ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ
ᑲᑎᒪᔾᔪᑎᒃᓴᑦ: ᑲᑎᒪᓂᕐᒧᑦ 001-2018
ᐱᖓᔪᐊᓐ, ᒫᔾᔨ 7th, 2018
ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑕ ᑲᑎᒪᕝᕕᐊᓂ, ᐃᖃᓗᐃᑦ, ᓄᓇᕗᑦ

	ᓇᒻᒪ	**ᑲᑎᒪᔾᔪᑎᒃᓴᖅ:**	**ᐃᔾᔪᔪᓂ**	**ᐅᖃᓪᓚᖕᓂᐊᖅᑐᖅ**	**ᐊᑯᓂᐅᓂᖓ**
9:00 AM to 9:05 AM	1	ᑲᑎᒪᒋᐊᓕᕐᓂᖅ/ᒪᑐᐃᖅᓯᓂᖅ ᑐᒃᓯᐊᕐᓂᒃᑯᑦ		ᐃᒃᓯᕙᐅᑕᐅᑲᐃᓐᓇᖅᑐᖅ	5 ᒥᓂᑦᔅ
9:05 AM to 9:10 AM	2	ᒪᑐᐃᖅᓯᓂᕐᒧᑦ ᐅᖃᐅᓯᒃᓴᑦ ᐊᒻᒪ ᑭᓇᒃᑰᓂᓐᖏᓐᓂᒃ ᓇᓗᓇᐃᖅᓯᓂᖅ		ᐃᒃᓯᕙᐅᑕᐅᑲᐃᓐᓇᖅᑐᖅ	5 ᒥᓂᑦᔅ
9:10 AM to 9:15 AM	3	ᑲᑎᒪᔾᔪᑎᒃᓴᑦ ᕿᒥᕐᕈᔭᐅᓂᖏᑦ ᐊᒻᒪᓗ ᐊᖏᕈᑕᐅᓂᖏᑦ	1	ᐃᒃᓯᕙᐅᑕᐅᑲᐃᓐᓇᖅᑐᖅ	5 ᒥᓂᑦᔅ
		ᐱᓕᕆᕝᕕᖓᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ - ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦ: ᐱᔾᔪᑎᑦ/ᐅᖃᐅᓯᒃᓴᑦ			
9:15 AM to 10:45 AM	*4*	ᐸᕙᓐ ᐸᐃᒍᐊᖅᑕᖅᑐᑦ ᓇᓄᐃᖏᓐᓂᑦ ᑲᑎᓐᖓᔪᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᑭᒡᓕᖏᑦ	2	ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᓐᓂ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ	*90 ᒥᓂᑦᔅ*
10:45 AM to 11:00 AM		***ᕿᑲᑲᐃᓐᓇᖅ***			15 ᒥᓂᑦᔅ
11:00 AM to 11:45 AM	5	Kane Basin -ᒍᐊᖅᑕᖅᑐᑦ ᓇᓄᐃᑦ ᑲᑎᓐᖓᔪᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᑭᒡᓕᖏᑦ	3	ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᓐᓂ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ	45 ᒥᓂᑦᔅ
11:45 AM to 1:15 PM		***ᐅᓪᓗᕈᒻᒥᑕᕐᓇᖅ***			ᐃᑲᕐᕋᖅ ᐊᑕᐅᓯᖅ ᐊᒻᒪᓗ 15 ᒥᓂᑦᔅ
		ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ: ᐱᔾᔪᑎᑦ/ᐅᖃᐅᓯᒃᓴᑦ			
1:15 PM to 1:30 PM	6	ᑕᕆᐅᖅ ᓴᐳᒻᒥᐅᓯᖅᑕᐅᓯᒪᓂᖏᓐᓄᑦ ᑐᓴᒐᒃᓴᑦ	4	ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ	15 ᒥᓂᑦᔅ
1:30 PM to 2:00 PM	7	ᓴᓇᔭᐅᓂᖏᑦ ᓄᓇᕗᒥ ᐃᒪᕐᒥᐅᑕᕐᓂᐊᕐᓂᕐᒧᑦ ᒪᓕᒐᕋᓛᖏᑦ	5	ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ	30 ᒥᓂᑦᔅ
2:00 PM to 2:30 PM	8	ᑲᑎᓐᖓᔪᑦ ᐱᔭᐅᔪᓐᓇᖅᑐᑦ ᐅᑭᐅᖅᑕᖅᑐᕐᒥ ᐊᒻᒪ ᑕᖅᓴᖅᑲᖅᑐᑦ ᑭᖑᒃᐸᒃᑕᐅᔪᓐᓇᖅᑐᑦ 2018/19-ᒥ ᑭᖑᒃᐸᖕᓂᐊᕐᓇᕐᒥ	6	ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ	30 ᒥᓂᑦᔅ
		ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ: ᐱᔾᔪᑎᖏᑦ/ᐅᖃᐅᓯᒃᓴᑦ			
		ᑎᑎᖅᑲᓂᒃ ᑐᓂᓯᓯᒪᕆᑦᑐᑦ	ᐊᑐᓐᖏ		

		ᑲᓇᑕᒥ ᒥᓐᒍᐃᕐᓯᕝᕙᓕᕆᔨᒃᑯᑦ: ᐱᔾᔪᑎᖏᑦ/ᐅᖃᐅᓯᒃᓴᑦ			
		ᑎᑎᕐᑲᓂᒃ ᑐᓂᓯᓯᒪᖏᑦᑐᑦ	ᐊᑐᓐᖏ		
		ᓄᓇᕗᑦ ᑐᓐᖓᕕᒃ ᑎᒥᖓᑦ: ᐱᔾᔪᑎᖏᑦ/ᐅᖃᐅᓯᒃᓴᑦ ᑐᓂᓯᓚᐅᖏᑦᑐᑦ ᑎᑎᕐᑲᓂᒃ			
		ᑎᑎᕐᑲᓂᒃ ᑐᓂᓯᓯᒪᖏᑦᑐᑦ	ᐊᑐᓐᖏ		
		ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ			
2:30 PM to 3:15 PM	9	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐅᓂᒃᑳᓕᐊᖓᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ	7	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔩᑦ	45 ᒥᓂᑦᓯ
3:15 PM to 3:30 PM		***ᕿᑲᑲᐃᓐᓇᖅ***			15 ᒥᓂᑦᓯ
		ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ			
3:30 PM to 4:00 PM	10	ᑲᖏᖅᑐᒑᐱᐅᑉ ᑐᒃᓯᕋᐅᑎᖓᑕ ᐅᓄᖅᓯᒋᐊᕐᑯᒡᓗᒋᑦ ᐃᖃᓗᒃᓴᓄᑦ ᑰᑕᖓᑦ ᐃᖃᓗᒃᓯᐅᕝᕕᖏᑦ ᑲᖏᖅᑐᐊᓗᖕᒥᑦ: ᐅᖃᐅᓯᑐᐃᓇᕐ		ᐆᒪᔪᓕᕆᔨ	30 ᒥᓂᑦᓯ
4:00 PM to 4:05 PM	11	ᓄᖅᑲᕐᓂᖅ		ᐃᒃᓯᕚᐅᑕᐅᑲᐃᓐᓇᖅᑐᖅ	5 ᒥᓂᑦᓯ

ᑐᓂᔭᐅᓯᒪᔪᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐃᒪᓇᐃᓘᖅᑕᐅᖁᓗᒋᑦ

ᑐᓴᐅᒪᔪᑎᒃᓴᖅ: ᐃᓱᒪᓕᐅᕈᑎᖅ: X

ᐋᖅᑭᒃᑕᐅᔭᕆᐊᓕᒃ: ᓇᓄᑦ ᑲᑎᓗᒋᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᓕᒫᑦ ᐊᑐᓕᖁᔨᔪᑎᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂ

ᐅᖃᐅᓯᖃᒃᓯᒪᓂᖏᑦ:

- ᑭᖑᓪᓕᖅᐹᖑᓚᐅᖅᑐᖅ ᐃᓘᓐᓇᓕᒫᖏᓐᓂᑦ ᐅᓄᕐᓂᖅᓯᐅᕆᓗᑎᒃ ᖃᐅᔨᓇᓱᐊᕐᓂᐅᓚᐅᖅᑐᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᑕ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ, ᐱᖃᖃᑎᖃᕈᑕᐅᓪᓗᓂ ᑲᓛᑦᓯᒥᐅᓂᑦ, ᐱᓕᕆᐊᖑᓚᐅᕐᒪᑕ ᐊᑯᓐᓂᖏᓐᓂᑦ 1994 ᐊᒻᒪᓗ 1997. 1997 ᐅᓄᕐᓂᖏᓐᓂᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑎᑦ ᐃᒪᓇᑎᒋᐅᓚᐅᖅᐳᑦ 2074 (95% ᐳᓴᓐᑎᑦ ᖁᓚᖏᓐᓂᕐᒧᑦ ᓄᖃᖓᑲᐃᓐᓇᕐᓂᖅ = 1,544-2,604) ᓇᓄᐃᑦ, ᐊᒻᒪᓗ ᐅᓄᕐᓂᖏᑦ ᐃᒪᓇᐅᓂᐊᓂᕋᒃᑕᐅᓚᐅᕐᒪᑕ ᐅᓄᕈᓐᓃᒃᑲᓐᓂᕐᓗᑎᒃ ᓇᓗᓇᕐᓂᖏᓄᑦ ᑲᓛᑦᓯᒥᐅᑦ ᓇᓐᓄᒐᓱᐊᖃᑦᑕᕐᓂᖏᑦ.
- ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᒃᓱᐊᓗᒃ ᓇᓗᓇᕐᓂᖏᑦ ᐅᓄᕐᓂᖏᑕ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᑦ ᑕᐃᒪᓐᓇᐃᑎᑕᐅᓪᓗᓂ ᐃᓚᖓᓄᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓂᖏᑕ ᓇᓗᓇᕐᓂᖏᓄᑦ ᐊᒻᒪᓗ ᐊᓯᕈᒃᐸᓪᓕᐊᓂᖓᓄᑦ ᐊᕙᑎᑦᑕ ᖃᓄᐃᓕᖓᓕᕐᓂᖓᑕ, ᐊᑐᖅᑕᐅᕙᓪᓕᐊᓗᓂ-ᐃᓚᓕᐅᑎᔭᐅᓗᓂ ᖃᑦᔫᓇᐅᕈᓇᐊᖅᑕᐅᓗᑎᒃ ᐊᖑᓇᓱᐊᖅᑕᐅᓂᖏᑕ ᐅᓄᕐᓂᖏᑦ ᐊᑐᓕᖅᑎᑕᐅᓚᐅᖅᓯᒪᖕᒪᑕ ᐊᑯᓐᓂᖏᓐᓂ 2010 ᐊᒻᒪᓗ 2014 ᑕᐃᑦᓯᒪᓂ ᓄᓇᕗᑦ ᐊᖑᓇᓱᐊᖅᐸᒃᑕᖏᑕ ᐅᓄᕐᓂᖏᑦ ᒥᑭᓪᓕᒋᐊᖅᑕᐅᓚᐅᖅᓯᒪᕗᑦ 105-ᖑᓚᐅᖅᑎᓗᒋᑦ ᓇᓄᐃᑦ ᐊᕐᕌᒍᓕᒫᒥ 2010-ᒥ, ᐃᒪᓇᐅᓕᖅᑐᑎᒃ 65 ᓇᓄᐃᑦ ᐊᕐᕌᒍᓕᒫᒥ 2014-ᒥ.
- ᑲᓇᑕ-ᖃᓚᑦᓰᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ ᓇᓄᕐᓂᑦ (JC) ᐃᓱᒪᓕᐅᕈᑎᖃᓚᐅᕐᒥᖕᒪᑕ ᓄᑖᒥᑦ ᖃᐅᔨᓇᓱᐊᖅᑐᖃᓇᐊᓕᐅᒐᓗᐊᖅ ᓄᑖᓂᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᖅᑖᖅᑭᑕᐅᒐᔭᕐᒪᑦ ᐊᒥᒐᖏᓐᓂᖏᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ; ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐱᓕᕆᖁᔭᐅᓕᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥ ᐃᖅᑲᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᑕᐃᑦᓯᒥᖓ ᖃᐅᔨᓇᓱᐊᖁᔭᐅᓪᓗᑎᒃ.
- ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐱᔭᕇᒃᓯᓚᐅᖅᐳᑦ ᐱᖓᓱᓂᑦ-ᐊᕐᕌᒍᓂᑦ ᖃᐅᔨᓇᓱᐊᖅᑕᒥᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᑦᓯᖃᖅᕕᒃᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ 2014-ᒥ ᐊᒻᒪᓗ ᑐᓂᓯᓚᐅᕆᓪᓗᑎᒃ ᐱᔭᕇᒃᓯᔪᑎᒥᖕᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒌᓄᖅᑕᐅᓪᓗᑎᒃ (JC) ᔪᓚᐃ 2016-ᒥ.
- ᒪᓕᒡᓗᒋᑦ ᐱᔭᕇᒃᓯᔪᑎᑦ ᐅᓂᒃᑳᓕᐊᖑᓯᒪᔪᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᑦᓯᖃᖅᕕᒃᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᓂᖏᑦ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐱᓕᕆᖁᔨᓕᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐱᔭᐅᑎᑦᑎᖁᓪᓗᒋᑦ ᐊᖑᓇᓱᐊᕈᓐᓇᕐᓂᕐᒧᑦ ᐊᑐᓕᖁᔨᔪᑎᓂᑦ ᐅᑯᑎᒎᓇ ᐊᖑᓇᓱᐊᑕᐅᓗᑎᒃ ᖁᒃᓴᓇᖅᑎᖅᒋᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑎᖅᒋᑕ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᔪᑎᖅᒋᑎᒍ.
- ᔪᓚᐃ 2017-ᒥ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᖅᑎᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑎᖅᒋᑦ ᐅᓂᒃᑳᓕᐊᖑᓯᒪᔪᑦ ᑐᓂᔭᐅᓚᐅᖅᐳᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᓐᓄᑦ (JC) ᐱᓕᕆᐊᖃᓚᐅᑐᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒍᑦ ᐱᓕᕆᐊᖑᔪᒪᔪᓂᑦ ᐱᔪᒪᔭᐅᓂᖃᖅᕕᐅᔪᓂᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ (JC).

- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ (GN) ᐊᐱᕐᓱᓚᐅᕐᒪᑕ ᐊᒃᑐᖅᑕᐅᓯᒪᔪᓂᑦ ᓄᓇᓕᖕᓂᑦ ᔭᓄᐊᕆ 2017-ᒥ ᐱᔭᕇᒃᓯᔪᑕᐅᔪᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᓂᕐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕐᕿᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖕᒋᓐᓂᑦ.

- ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᖃᐅᔨᓇᓱᐊᕐᓂᑦ ᖃᐅᔨᓇᓱᖕᓂᕐᒥᑦ ᐅᓂᒃᑳᓕᐊᕆᓯᒪᔪᑦ ᐅᖃᒃᓯᒪᓚᐅᕐᒪᑕ ᐅᓄᖅᓯᒃᑲᓐᓂᕐᓂᕋᐃᓐᓗᑎᒃ ᐅᓄᕐᓂᓪᓚᕆᖕᒋᓐᓂᑦ ᐅᓄᕐᓂᖕᒋᑕ ᒥᒃᓴᐅᓴᒃᑕᐅᓚᐅᑎᓗᒋᑦ ᓯᕗᓂᖓᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᕐᓂᖕᒋᓐᓂᑦ ᑕᒪᒃᑭᒃ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕐᕿᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖕᒋᑦ. ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᒫᓐᓇ ᓅᕙᔪᐊᓘᖕᒪᑦ ᐃᒪᓐᓇᐅᔪᓐᓇᖅᓯᓂᖕᒐ ᐅᓄᕐᓂᖕᒋᑦ ᐅᓄᖅᓯᒃᑲᓐᓂᕈᓐᓇᖅᓯᓪᓗᑎᒃ ᑕᒪᓐᓇᐅᑎᓪᓗᒍ ᖃᓄᐃᓐᓂᕐᓗᒍᑎᖃᖕᒋᑦᑎᐊᖅᑎᓗᒋᑦ ᐊᑎᖅᑕᖅᑖᖅᑲᑦᑕᕐᓂᖕᒋᑦ, ᑕᒪᐃᒃᑲᓗᐊᖅᑎᓗᒍ, ᐊᖑᑎᖕᒋᑕ ᐆᒪᑯᑖᖅᑲᑦᑕᕐᓂᖕᒋᑦ ᕿᓚᒻᒥᐅᓂᖅᓴᐅᖅᑲᑦᑕᖅᑐᔮᓕᕐᒪᑕ ᐊᒻᒪᓗ ᖃᑦᔫᓐᓇᐅᓂᖅᓴᐅᓕᖅᐸᓗᑎᒃ ᓯᑯᒥ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓯᓚᐅᑉ ᐊᓯᔾᔨᖅᐸᓪᓕᐊᓂᖓ ᐃᓱᒫᓗᓇᖃᓯᐅᑎᖕᒥᖕᒪᑦ.

- ᒫᓐᓇ ᐊᑐᖅᑕᐅᔪᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐅᓄᕐᓂᖕᒋᓐᓄᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑎᑦ, ᑕᐅᕙᖓᑦ 2011-2014–ᒥ ᖃᐅᔨᓇᓱᐊᕐᓂᖅ, ᐃᒪᓇᑎᒋᐅᒡᒪᑕ 2,826 95% ᐳᓴᓐᑎᑦ ᖁᓚᖕᒋᓐᓂᕐᒧᑦ ᓄᖃᖓᑲᐃᓐᓇᕐᓂᖅ = 2,059-3,593) ᓇᓄᐃᑦ, ᑕᐃᒃᑯᐊᓕ 1997-ᒥ ᒥᒃᓴᐅᓴᒃᑕᐅᓚᐅᖅᓯᒪᔪᑦ ᐃᒪᓇᑎᒋᐅᓚᐅᖅᓯᒪᖕᒪᑕ 2074 (95% ᐳᓴᓐᑎᑦ ᖁᓚᖕᒋᓐᓂᕐᒧᑦ ᓄᖃᖓᑲᐃᓐᓇᕐᓂᖅ = 1,544-2,604) ᓇᓄᐃᑦ. ᐅᓂᒃᑳᖅᓯᒪᔪᑦ ᐅᔾᔨᖅᑐᕐᑎᐊᖁᔨᖕᒪᑕ ᑖᒃᑯᐊ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᖃᐅᔨᓇᓱᐊᖅᑕᐅᔪᑦ ᓴᕿᔪᑎᖕᒋᑦ ᑕᐅᕙᓂ 1990-ᒥ ᓈᒪᑦᑎᐊᖅᑐᒥᑦ ᐊᔾᔨᐅᓂᖅᓯᐅᖅᑕᐅᔪᓐᓇᖕᒋᒪᑕ ᓄᑖᖑᓖᓄᑦ ᖃᐅᔨᓇᓱᐊᕈᑕᐅᓚᐅᖅᑐᓄᑦ ᓴᕿᑕᐅᓚᐅᖅᑐᓂᑦ, ᐱᓗᐊᙳᐊᖅᑐᒥᑦ ᐅᓄᖅᑐᐊᓗᓂᖅᑕ ᒥᒃᓴᐅᓴᒃᑕᐅᔪᑎᖕᒋᓐᓂᑦ. ᐊᓯᕈᒃᑎᓪᓗᒋᑦ ᐊᕙᑎᑦᑕ ᖃᓄᐃᓕᖓᓂᓇᓕᖅᑕᖅᒋᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓴᓚᒃᓴᓕᐅᕐᓂᕐᒧᑦ ᐱᓕᕆᔪᓯᐅᕙᒃᑐᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᓚᐅᖅᑐᑦ ᐊᑯᓐᓂᖓᓂ ᒪᕐᕉᒃ ᖃᐅᔨᓇᓱᐊᕐᓂᒃ ᐱᔪᓐᓇᖅᑎᑦᑎᖕᒋᒪᑎᒃ ᖃᐅᔨᔭᑕᐅᔪᓂᑦ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖕᒋᓐᓂᑦ ᒪᑯᓂᖓ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖕᒋᓐᓂᑦ. ᑕᒪᐃᒃᑲᓗᐊᖅᑎᓗᒍ, ᐃᓅᓇᓕᒫᖅ ᖃᐅᔨᓇᓱᐊᕐᓂᖅ ᓇᓗᓇᐃᖅᓯᖕᒪᑦ ᖃᑦᔫᓐᓇᐅᓂᖅᓴᐅᓕᕐᓂᖕᒋᓐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖕᒋᓐᓂᑦ ᑕᐃᑦᓱᒪᓂᐅᕋᑖᓚᐅᖅᑐᖅ ᐃᒪᓐᓇᐅᓂᐊᕐᓂᖃᒃᑕᐅᓚᐅᖅᓯᒪᑎᓪᓗᒋᑦ.

- ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᓄᑦ (BB) ᓇᓄᐃᑦ ᓇᓄᕋᖅᑭᑦ ᐊᐅᓪᓚᖅᑎᑕᐅᔭᕆᐊᖃᕈᓐᓃᓚᐅᖅᓯᒪᖕᒪᑕ ᓯᓚᑖᓄᑦ ᓄᓇᕗᑦ 2010-ᒥ ᐱᔾᔪᑕᐅᓪᓗᓂ ᑕᐅᖅᓯᑦᑕᐃᓕᓕᖅᑎᑕᐅᓂᖕᒋᓄᑦ. ᑕᐅᖅᓯᑦᑕᐃᓕᓂ ᑕᐃᒪᓐᓇᐃᓕᑎᑦᑎᑕᐅᓚᐅᖅᓯᒪᖕᒪᑦ ᐃᓱᒫᓗᑕᐅᓂᖕᒋᓄᑦ ᐊᑕᐅᑦᑎᒃᑯᖅᑎᓗᒋᑦ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᑲᑖᑦᓯᑦ ᐊᖑᓇᓱᐊᕐᓂᖕᒋᑦ ᑕᒫᖓᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖕᒋᓐᓂᑦ ᑲᔪᓯᔪᐊᓇᖃᓯᐅᑎᓯᒪᓚᐅᖅᓯᒪᖕᒪᑦ ᐊᒻᒪᓗ ᐅᓄᕐᓂᖕᒋᑦ ᐅᓄᖃᓐᓂᖅᐸᓪᓕᐊᓯᔭᐅᓚᐅᖅᓯᒪᖕᒪᑦ.

- ᐱᐅᓇᖅᓯᒪᖅᑭᑦᑐᑦ ᐋᓐᓂᖅᑕᐅᔪᑎᒋᓂᐅᓴᖅᑭᑦᑕᖅᑭᑦ ᖃᐅᔨᔭᐅᖅᒪᑕ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖕᒋᑦ ᐲᖅᑕᐅᓚᐅᖅᓯᒪᖕᒪᑕ ᑕᐃᒪᓐᓇᐃᓕᖅᑐᓂ ᑕᐅᖅᓯᑦᑕᐃᓕᓯᒪᓂ ᐲᖅᑕᐅᓚᐅᖅᓯᒪᖕᒪᑦ ᓇᓗᓇᐃᖅᓯᒪᓚᐅᖅᓯᒪᔪᑦ ᐊᖑᓇᓱᒃᑎᓄᑦ ᑕᐃᑦᓱᒪ ᐱᓇᓱᒃᑕᐅᕙᓚᐅᖅᑐᒥ ᐊᑐᖅᑕᐅᓇᓚᐅᕐᒪᑦ.

ᒫᓐᓇᐅᔪᖅ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ:

- ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐋᖅᑭᒃᓱᐃᓚᐅᕐᒪᑕ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓴᖅᓯᔪᑎᑦ ᐅᓂᒃᑳᓕᐅᕈᑎᖏᓐᓂᑦ ᐊᑐᕐᓗᑎᒃ ᒫᓐᓇᕋᑖᖅ ᐅᓄᕐᓂᕆᓚᐅᖅᑕᖏᓐᓂᑦ ᖃᐅᔨᒪᔪᑎᒃᓴᓂ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒥᒐᖏᒪᖔᑕ ᖃᐅᔨᓇᓱᐊᑕᐅᔪᑎᖏᓐᓂ. ᑕᐃᑯᓇᓂ ᐅᓂᒃᑳᓕᐊᕆᓯᒪᔪᓂ, ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᐊᑐᑐᐃᓐᓇᕆᐊᖃᓕᖅᑐᑦ ᐊᒻᒪᓗ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᕈᑎᔪᑦ ᖃᐅᔨᒪᓇᕈᑎᖃᒃᑎᑕᐅᓚᐅᕐᒪᑕ ᐱᓕᕆᒥᐊᕈᑕᐅᓪᓗᑎᒃ ᑐᒃᓯᕌᖑᓯᒪᓂᖏᓐᓄᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ (JC).
- ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓇᓱᐊᕈᑎᑦ ᐃᓚᒋᓕᐅᔨᓚᐅᕐᒪᑕ ᐊᔾᔨᒌᖏᑦᑐᓂᑦ ᐆᒪᔪᓯᖃᕐᓂᖏᑦ ᐱᓕᕆᐊᖑᒋᐊᕐᓗᑎᒃ ᐊᒻᒪᓗ ᐱᑕᖃᕆᐊᖃᓚᕆᒡᓂᖏᓐᓂᑦ ᖃᓄᑎᒋᐅᓂᖏᑦ ᑐᙵᕕᒋᓗᒋᑦ ᓇᓄᐃᑦ ᐆᒪᑎᓪᓗᒋᑦ ᖃᓄᐃᓘᖅᑕᑦᑕᕐᒪᖔᑕ, ᐊᒻᒪᓗ ᐊᒃᑐᐃᑐᐃᓇᕆᐊᖅᑲᖅᓂᖓ ᓯᕗᓂᒃᓴᒥ ᑕᕈᐅᑉ-ᓯᑯᖓᑕ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖓᓂ ᐅᓄᖅᓯᓂᖓᓂ ᐊᒻᒪᓗ ᖃᓄᐃᓕᕐᒪᖔᑕ ᑕᒪᑐᒨᓇ ᐃᒪᓐᓇᐅᓂᐊᕋᓯᕆᔭᐅᓪᓗᑎᒃ ᖃᓄᐃᓕᖔᓪᓚᐊᓂᖏᑦ ᑲᔪᓯᔪᓐᓇᕐᓂᖏᑎᒍ.
- ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐃᓱᒪᓕᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᒪᑯᓂᖓ ᖃᑦᓲᓐᓇᐅᓗᑎᒃ-ᐊᑯᓐᓈᖅᑭᔭᑦ ᖁᒃᓴᓇᕈᑎᖏᑕ ᐊᓂᒍᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ ᐊᐅᓚᑕᐅᔪᑎᖏᓐᓄᑦ ᐃᓕᕆᐊᖑᔪᒪᔪᑦ ᐋᖅᑭᐅᒪᑎᑕᐅᓗᑎᒃ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᖃᑦᓯᐅᑕᕐᓂᖏᓐᓄᑦ ᐋᖅᑭᐅᒪᑦᑎᐊᕐᓗᑎᒃ ᐅᓄᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐊᕙᑎᒦᑦᑐᑦ ᒪᑭᒪᑎᑕᐅᔪᓐᓇᕐᓗᑎᒃ.
- ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᓂᕈᐊᓚᐅᕐᒪᑕ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᐊᑐᑐᐃᓐᓇᕆᐊᓕᖕᓂᑦ ᐃᒪᓇᑎᒋ 70% ᐳᓴᓐᑎᑦ ᑲᔪᓯᑦᑎᐊᕈᓐᓇᑐᐃᓐᓇᕆᐊᖃᓯᑦᑐᑎᒃ ᐊᑐᕋᓇᕋᔭᖅᑐᑎᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᔪᒪᔪᓂᑦ. ᐱᑐᐊᙱᐊᖅᑐᒥᑦ, ᑕᐃᒃᑯᐊ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᑦ (BB) ᑐᑭᖃᕐᒪᑦ ᓇᓐᓄᒃᑕᐅᔭᐃᓐᓇᐅᖃᑦᑕᖅᐸᑕ ᐃᒪᓇᑎᒋᑦ 5.7% ᐳᓴᓐᑎᑦ, ᐅᕝᕙᓗᓂᑦ 160 ᓇᓄᐃᑦ ᐊᕐᕌᒍᑕᒪᑦ ᑲᓇᑕᒥ ᐊᒻᒪᓗ ᖃᓛᑦᓕᒥ, ᐊᕕᒡᓗᐊᖅᑕᐅᓗᑎᒃ ᑕᐃᒪᓪᓗᐊᖅᑲᑎᒋᓪᓗᑎᒃ, ᐃᓅᓇᓕᒫᖏᑦ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐅᓄᓕᓃᐅᓂᖏᑦ ᐃᒪᓇᑎᒋᐅᓗᑎᒃ 1:1. ᓂᕈᐊᖅᑕᐅᖏᓛᕐᓗᑎᒃ ᖁᒃᓴᓇᓗᐊᖏᓐᓂᕐᓴᖅᑕᐅᓗᑎᒃ ᐱᓕᕆᐊᖑᒐᔭᖅᐸᑕ 4.3% ᐳᓴᓐᑎᑦ ᐊᖑᓇᓱᖅᑕᐅᓂᖏᑦ ᐅᓄᕐᓂᖅᓯᖅᑐᑎᒃ ᑕᐃᒪᓐᓇᐅᓯᒍᓪᓗᓂᓗ ᐅᓄᕐᓂᖅᓴᓄᑦ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᓂᕈᐊᓚᐅᕐᒪᑕ ᐊᑯᓐᓈᖅᑭᔭᓂᑦ ᖁᒃᓴᓇᕈᑎᓂᖏᓐᓂᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᓗᑎᒃ ᖃᑦᓯᐅᓂᖏᓐᓄᑦ.
- ᐊᖑᓇᓱᐊᖅᑕᐅᓂᖏᓐᓄᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓴᖅᓯᓂᑦ ᐅᖃᐅᓯᖃᓚᐅᕐᒪᑕ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᐅᓕᖅᓯᒪᓂᖏᓐᓂ ᐆᒪᑯᑖᖃᑦᑕᕐᓂᖏᑕ ᐊᖑᑎᑦ ᐊᒻᒪᓗ ᓈᒪᒃᓴᖏᓐᓂᑦ ᓇᓪᓕᐊᖑᓂᖏᑕ ᐅᓄᓕᓃᐅᓂᖏᓐᓂᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᕐᒪᑕ ᐊᕐᓇᓪᓗᐃᑦ ᐃᒪᓇᓪᓗᐊᑎᒋᐸᓗᕐᒪᑕ ᖃᓂᒋᔮᓂᐸᓗᒡᓗᑎᒃ 70% ᐳᓴᓐᑎᑦ ᐃᒻᒥᓂᖅᓴᖅᑐᑦ ᓇᓄᐃᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB). ᐅᓂᒃᑳᓕᐅᕆᓯᒪᔪᖅ ᐅᖃᐅᓯᖃᓗᐊᙱᐊᕐᒪᑕ ᖃᑦᓲᓐᓇᕈᕐᓂᖏᓐᓂᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ ᓴᖅᑭᑉᐸᓪᓚᐊᑐᐃᓐᓇᕐᒪᑕ ᓄᖑᒍᑦᑕᐃᓕᒪᑎᑕᐅᖁᔭᐅᓪᓗᑎᒃ ᐃᓱᒫᓗᑕᐅᖅᒪᑕ. ᑕᒪᓐᓇᐅᑎᓪᓗᒍ ᐃᓱᒫᓗᑕᐅᔭᖅ, ᑲᑎᑕᐅᓗᑎᒃ ᐊᖑᓇᓯᒃᑕᐅᔪᑦ ᓇᓪᓕᐅᖑᒃᐸᑕ ᐅᓄᓕᓃᓂ 1 ᐅᖑᑦ ᐊᑐᓂ ᐊᕐᓇᓗᐃᑦ ᑕᒪᓐᓇᐅᖁᔭᐅᓚᐅᖅᐳᑦ.
- ᓂᕈᐊᖅᑕᐅᓂᖏᑦ ᐊᖑᓇᓱᒃᑕᐅᖁᔭᐅᔪᑦ ᖃᑦᓯᐅᐸᒐᔭᖅᐊᑕ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ (BB) ᓴᖅᑭᑎᑦᑎᕐᒪᑕ ᑕᐃᒃᑯᓂᖓᓪᓗᐊᖅ ᐅᓐᓄᕐᓂᖏᓐᓂᑦ ᐆᒪᔪᓯᕐᒥᑦ ᑕᒃᑯᐊ ᐅᓄᕐᓂᖏᑦ ᐊᒻᒪᓗ ᑕᐃᒪᓐᓇᐅᓂᕋᐃᔭᕐᑕᐅᓐᓇᕐᓇᑎᒃ ᐃᓅᓇᓕᒫᖓᓗ ᐊᓯᖏᑦ ᓄᓇᕗᑦ ᓇᓄᖏᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ. ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᓂ (BB) ᒫᓐᓇ ᓲᖑᔪᒥᑦ ᐅᓄᖅᓯᒃᑲᓐᓂᕋᔭᖅᓯᓐᓇᕐᒪᑕ ᐊᒻᒪᓗ ᖃᓄᐃᕐᒧᑎᖅᑲᖅᑭᑦᑐᓂᑦ ᐊᑎᖅᑕᖅᑖᖅᐸᒃᑐᑎᒃ. ᐊᑐᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᐊᐅᓚᑕᐅᕐᒪᑕ ᐱᓕᕆᐊᖑᕙᓪᓕᐊᓂᖏᓐᓄ ᒪᓕᒃᓗᒋᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᑐᕐᖁᔭᐅᔪᑦ ᒪᓕᑐᐃᓐᓇᐃᒪᑕ ᑕᐃᑯᓂᖓᓪᓗᐊᖅ ᐅᓄᕐᓂᖏᑕ ᐅᓄᖅᓯᕙᓪᓕᐊᓂᖏᓐᓂᑦ.
- ᐊᖑᓇᓯᒃᑕᐅᓂᖏᓐᓄᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓴᖅᓯᓂᖅ ᐱᔭᐅᑎᑦᑎᓯᕗᒪᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ ᐱᔭᐅᔭᓇᐊᓕᓐᓂᑦ ᖃᐅᔨᒪᓕᐊᖅᓯᔭᐊᑕᐅᖅᑕᐃᓐᓇᕐᓂᖏᓐᓂᑦ ᖃᓄᐃᓘᖃᑦᑕᕐᓂᖏᓐᓂᑦ ᐋᖅᑭᐅᒪᑎᑕᐅᖁᓗᒋᑦ ᐊᖑᓇᓯᒃᑕᐅᑦᑕᕐᓂᖏᑕ ᐅᓄᕐᓂᖏᑦ ᐃᓱᒪᓕᐅᕈᑎᒋᔭᐅᕙᒃᑐᓂ ᐊᐅᓚᑦᑎᓂᐅᑉ

ᐊᓯᕈᒃᑎᑕᐅᓂᖏᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᐊᕐᒪᑕ 15-ᐅᑭᐅᑦ ᐊᑯᓐᓂᖏᓐᓂ. ᐊᓯᕈᒃᑐᒥᓂᐅᒃᐸᑦ ᑕᓇᐅᑉ–ᓯᑯᖓ, ᐆᒪᑯᑖᖃᑦᑕᕐᓂᖏᑕ ᖃᓄᑎᒋᐅᓂᖏᑦ (ᐱᓗᐊᙳᐊᖅᑐᒥᑦ ᐊᖑᑎᑦ), ᐊᑎᖅᑕᖏᑕ ᖃᑦᓯᐅᖃᑦᑕᕐᓂᖏᑦ, ᐅᕝᕙᓗᓂᑦ ᓇᓐᓄᒃᑕᐅᔪᑦ ᓇᓪᓕᐊᖑᓂᖏᑕ ᐅᓄᓕᓈᒡᓂᖏᑦ, ᑕᐃᒪᓐᓇᐃᓕᖅᐸᑦ ᐊᖑᓇᓱᒃᑕᐅᖁᔭᐅᔪᑦ ᐅᓄᕐᓂᖏᑦ ᐋᖅᑭᒋᐊᖅᑕᐅᔭᕆᐊᖃᓕᕐᔭᕐᒪᑕ ᐅᓄᖏᓐᓂᖅᓴᐅᓕᖅᑎᑕᐅᓗᑎᒃ ᐱᔭᐅᔪᓐᓇᖅᓯᖁᓗᒋᑦ ᐋᖅᑭᒃᑕᐅᓯᒪᔪᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᓇᔪᒪᔪᑦ (ᑐᑭᓕᐅᑎᓗᒍ ᐋᖅᑭᐅᒪᑎᑕᐅᓗᑎᒃ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐊᒥᓲᓂᖏᑦ ᐋᖅᑭᒍᑎᑦᑎᐊᖅᓯᒪᔪᑦ ᐅᓄᕐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ ᐊᕙᑎᑦᑎᓐᓄᑦ ᒪᑭᒪᑎᑕᐅᔪᓐᓇᖅᑐᑦ).

ᐊᐱᖅᓱᕐᓃᑦ:

- ᓄᓇᓕᖕᓂᑦ ᐊᐱᖅᓱᖅᑐᖃᓚᐅᕐᒪᑦ ᐊᖑᓇᓱᐊᖅᑐᓕᕆᔾᔪᑎᒃᑯᑦ ᐃᖃᓇᐃᔭᕐᕕᖁᑎᖓᓂᑦ (HTO) ᑭᒡᒐᖅᑐᐃᔨᓂᑦ ᒥᑦᑎᒪᑕᓕᖕᒥᑦ, ᑲᖏᖅᖢᒑᐱᖕᒥᑦ, ᐊᒻᒪᓗ ᕿᑭᖅᑕᕐᔪᐊᒥᑦ ᑕᐅᕙᓂ ᔭᓄᐊᓇ 9-15, 2017-ᒥᑦ, ᐱᖃᓯᐅᑎᓗᒋᑦ ᐱᓕᕆᖃᑕᐅᔪᑦ ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓃᖑᖅᑐᑦ (NTI) ᐊᒻᒪᓗ ᕿᑭᒃᑕᐅᓂ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂᑦ. ᑕᐃᑦᓯᒪᓂ ᑲᑎᒪᑎᓪᓗᒋᑦ, ᐋᖅᑭᒃᑕᐅᔪᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔮᑦ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᖃᐅᔨᓇᓱᐊᑎᓂᑦ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᔪᑦ ᐅᓂᒃᑳᓕᐊᕆᓯᒪᔪᑦ ᐅᖃᖃᑎᒋᒍᑕᐅᓚᐅᕐᒪᑕ. ᓄᓇᓕᖕᓂᑦ ᐊᐱᖅᓱᖅᑐᖃᓂᐊᕐᒪᑦ ᐃᓚᒋᔭᐅᓗᑎᒃ ᐊᖑᓇᓱᐊᖅᑐᓕᕆᔾᔪᑎᒃᑯᑦ ᐃᖃᓇᐃᔭᕐᕕᖁᑎᖏᑕ (HTO) ᑭᒡᒐᖅᑐᐃᔨᖏᑦ ᑕᐃᑯᓇᖓᑦᑕᐃᓐᓈᕐᓗᑎᒃ ᓄᓇᓕᖕᓂᑦ ᐊᑐᖅᑎᓗᒍ ᓯᕗᓪᓕᖅᐹᖓ ᐱᓇᓱᐊᕈᓯᖅ ᕕᕗᐊᓇ 2018–ᒥ ᐅᖃᖃᑎᒋᒍᑕᐅᓗᑎᒃ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᐊᑐᓕᖁᔨᔪᑎᑦ
- ᐃᓘᓇᑎᒃ ᐱᓕᕆᖃᑕᐅᔪᑦ ᐊᖏᖃᑎᒋᓚᐅᕐᒪᑕ ᐃᑲᔪᖅᓯᐃᔪᒪᓪᓗᑎᒃ ᑐᓂᓯᕚᓕᔪᒪᓂᕐᒥᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᓂ (BB) ᑲᑎᓗᒋᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᓂᑦ (TAH) ᐊᑐᓕᖁᔨᔪᑎᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓅᖅᑕᐅᓗᑎᒃ (NWMB), ᓯᕗᓂᖓᒍᑦ ᐊᐱᖅᓱᓇᐊᑐᖃᓚᐅᖏᓐᓂᖓᓂ. ᐊᐱᖅᓱᕐᓂᕐᒧᑦ ᓇᐃᓈᖅᓯᔪᑎᑦ ᑐᓂᔭᐅᑦᕙᐅᑎᒋᔭᓇᐊᖃᕐᒪᑕ ᐅᑎᓚᐅᖅᑎᓪᓗᒋᑦ ᐊᐱᖅᓱᓇᐊᖅᑐᖅᓯᒪᓂᕐᒥᑦ ᐊᒻᒪᓗ ᐱᔭᕇᒃᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᑐᑭᓕᐅᖅᑕᐅᓂᖏᑦ.
- ᒪᓕᑐᐃᓐᓇᕐᓂᐊᕐᒪᑕ ᖃᓄᐃᓕᔪᑕᐅᓂᖏᓐᓂᑦ ᐊᐱᖅᓱᓇᐊᖅᑐᖅᓯᒪᓂᐅᑉ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ (GN) ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᐱᔭᐅᖁᔨᑐᐃᓐᓇᓇᐊᖃᕐᒪᑕ ᑐᓴᐅᒪᔪᑎᒃᓴᒃᑲᓐᓂᕐᓂᑦ ᐅᕝᕙᓗᓂᑦ ᐊᑐᓕᖁᔨᔪᑎᓂᑦ ᐱᓕᕆᑎᓪᓗᒋᑦ ᒫᔾ 2018–ᒥ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ (NWMB) ᑲᑎᒪᓂᕆᓯᖓ (RM001-2018).

ᐊᑐᓕᖁᔭᐅᔪᑦ:

1. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᐃᑲᔪᖅᓯᐃᖕᒪᑕ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐊᑐᓕᖁᔨᔪᑎᖏᓐᓂᑦ **ᐅᓄᖅᓯᒋᐊᖅᑕᐅᖁᓗᒋᑦ ᒫᓐᓇ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᑲᑎᓗᒋᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ (TAH) ᐃᒪᓇᑎᒋᐅᓚᐅᖅᑎᓗᒋᑦ 65 ᐃᒪᓇᐅᓕᖁᓗᑎᒃ 80 ᓇᓄᐃᑦ** ᐊᖀᒍᑕᒪᑦ.
2. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᐊᑐᓕᖁᔨᕗᑦ ᑖᒃᑯᐊ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᓇᓪᓕᐊᖑᓂᖏᑕ ᐅᓄᓕᓈᒡᓂᖏᑦ **ᐋᖅᑭᒃᑕᐅᓚᐅᕐᒪᑕ ᐃᒪᓇᐅᓕᖁᓗᑎᒃ 1:1 ᓇᓪᓕᐊᖑᓂᖏᑕ ᐅᓄᓕᓈᒡᓂᖏᓐᓂ** (1 ᐊᖑᑦ ᓇᓐᓄᒃᑕᐅᓗᓂ ᓇᓐᓄᒃᑕᐅᔪᖃᒃᑎᓗᒍ 1 ᐊᕐᓇᓪᓗᖕᒥᑦ ᓇᓐᓄᒃᑐᖃᕋᖓᑦ) ᐱᔾᔪᑎᒋᓗᒍ ᐃᓱᒫᓗᑕᐅᓂᖓ ᖃᑦᓲᓇᕈᒃᑎᑕᐅᑐᐃᓐᓇᓇᐊᖃᕐᓂᖏᓐᓂᑦ ᐃᓐᓇᐃᑦ ᐊᖑᑎᑦ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂ.
3. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᐊᑐᓕᖁᔨᓕᖅᐳᑦ ᐲᖅᑕᐅᖁᔨᓪᓗᑎᒃ ᐊᑕᐅᓯᕐᒥᑦ ᐊᖑᒻᒧᑦ ᐅᕝᕙᓗᓂᑦ ᐊᑕᐅᓯᕐᒥᑦ ᐊᕐᓇᓪᓗᖕᒧᑦ ᓂᕕᖓᑖᕐᒥᑦ ᐊᖀᒍᓂᐊᖅᑐᒥ ᓄᓇᓕᑦ ᓂᕕᖓᑦᑕᖁᑎᖏᑕ ᓴᓂᕝᕙᒃᑕᐅᓂᓐᓂ ᐊᑐᓂ ᐆᒪᔪᒧᑦ ᐅᖓᑖᓂᑦ ᓇᓐᓄᒃᑐᖅᓯᓂᖅᐸᑦ ᓈᒪᒃᑎᖅᑕᐅᖃᑦᑕᖏᓪᓚᕆᓪᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓄᑦ ᐱᓕᕆᔪᑎᑦ.
4. ᓴᓂᕝᕙᐃᓂᖅ ᐅᑯᓇᖓᑦ ᑲᑖᑦᑖᓂᑦ ᑲᓇᑕᒥᓪᓗ = 160 (80 ᐊᑐᓂ ᓄᓇᓕᑦᓯᐊᓂᑦ).

Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations

Final Report to the Canada-Greenland Joint Commission on Polar Bear
from the Scientific Working Group (SWG) of the Canada-Greenland Joint Commission on Polar Bear

Prepared by: Stephen Atkinson (SWG), Erik W. Born (SWG), Kristin L. Laidre (SWG), Nicholas J. Lunn (SWG), Øystein Wiig (SWG) and external experts Todd Arnold, Markus Dyck, Eric V. Regehr, Harry Stern, and Seth Stapleton

Edited by: Kristin L. Laidre and Nicholas J. Lunn

31 July 2016

Please cite as: SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear. 31 July 2016: x + 636 pp.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS

Over the previous 6 years during which this research program was developed, field work undertaken, data analyzed, and a final report written and delivered to the Canada-Greenland Joint Commission on Polar Bear, many individuals and organizations provided invaluable assistance. We hope that the following acknowledgements capture their collective contributions appropriately but recognize that we may have missed some; omissions do not reflect diminished involvement but rather that it was not possible to name everyone involved.

Financial, logistical, and in-kind support were provided by: Air Greenland, Applied Physics Laboratory (University of Washington), Avannaa Resources Inc. (Copenhagen), Niels Berthelsen (Upenavik), Bureau of Mineral and Petroleum (Nuuk), Canada Department of National Defense, Environment and Climate Change Canada, Environmental Protection Agency (Ministry of Environment and Food of Denmark) -DANCEA Programme, Government of Nunavut, Greenland Department of Fisheries, Hunting and Agriculture (DFHA), Greenland Institute of Natural Resources, Greenland Pharmaceutical Authority (Nuuk), the Hospital in Upernavik – Peqqik, KNAPK (Greenland Hunters and Fishermen Organization), Kullorsuaq School, Lasø ApS (Upernavik), Mitteqarfik airports in Qaarsut and Upernavik, the Mittimatalik Hunters and Trappers Organization, the National Aeronautical and Space Agency (NASA) Development and Testing of Potential Indicators for the National Climate Assessment program grant NNX13AN28G (PI: H. Stern), NASA Climate and Biological Response grant NNX11A063G (PI: K. Laidre), the Namautaq Hunters and Trappers Organization, the Nativak Hunters and Trappers Organization, Nuna Minerals Inc. (Nuuk), Nunavut Wildlife Management Board, Parks Canada Agency, Polar Continental Shelf Project, Polar Science Center (University of Washington), Qaanaap Kommunia, Qikiqtaaluk Corporation, Quantum Murray LP, Royal

Canadian Mounted Police, School of Aquatic and Fishery Sciences (University of Washington), Thule Air Force Base (Base Commander, US Air Force and Danish Liaison Officer, Danish Navy), Universal Helicopters, University of Oslo, Upernavik Seafood (Upernavik), Uummannaq Sundhedscenter – Peqqik, Vetlesen Foundation, and World Wildlife Fund (Nunavut General Monitoring Program).

The success of the aerial survey and genetic mark-recapture programs was largely due to the pilots and engineers who went above and beyond their call of duty. In particular, we thank Geir Akse, Jim Berry, Gerard Hartery, Sebastian Holst, John Innes, Steve Lodge, and Lorne Pike.

Mark Akeeagok, Dennis Andriashek, Apoosie Apak, Ilannguaq Egede, Carey Elverum, Paul Frame, Bruce J. Hainnu, Peter Hegelund, Jacob Jaypoody, Martika Jensen, Jaypeetee Kalliktee, Robert Kautak, Jopee Kigutak, Brian Koonoo, Jaloo Kooneloosie, George Koonoo, Juulut Kruse, Karl Kristian Kruse, Juilie Kuksiak, Robbie Kullualik, Jon Neely, James Noble Jr., Timothæus Petersen, Adrian Jimmy Pitseolak, polar bear hunters in Niaqornat, Kullorsuaq and Upernavik, Tyler Ross, Vicki Sahanatien, H.C. Schmidt, Guillaume Szor, and Mathias Therkelsen provided invaluable assistance and friendship in the field.

We thank the staff of Wildlife Genetics International (Nelson, BC, Canada) for their expertise in genotyping such a large number of samples and for on-going management of a polar bear genotype database. Staff at the Danish Center for Environment (Århus University) prepared and aged polar bear teeth.

We also thank Drs. Jon Aars (Norwegian Polar Institute), Andrew Derocher (University of Alberta), Jeff Laake (NOAA), and Elizabeth Peacock (USGS) for reviewing the survey options assessment report for Baffin Bay and Kane Basin. We are grateful to Dr. Gary White, Colorado State University, who kindly reviewed our mark-recapture modeling approach and interpretations of the analysis. Professor Patrick Heagerty, University of Washington, offered statistical consultation on resource selection models. Erica Escajeda conducted denning analyses as part of her Masters work at the University of Washington and Richard McGovern assisted with GIS mapping. We also thank professor Lutz Bachmann (University of Oslo) for reviewing the results of the population genetic analyses.

Permits and authorizations to undertake this work were issued by Canadian Border Services Agency (SERL-12-008, SERL-13-012, SERL-13-130), Danish Transport and Aviation Authority, Environment Canada National Wildlife Areas research permit (Nun-NWA-13-07), Greenland Aviation Authority, Greenland Department of Environment (Nuuk), Greenland Department of Fisheries, Hunting and Agriculture (DFHA), Greenland Department of Health, Greenland Home Rule Government, Government of Nunavut Wildlife Research Permits (2011-041, 2012-010, 2012-015, 2013-011, 2013-0292014-007), Nunavut Impact Review Board screening (13YN017 – exemption from environmental assessment screening), Parks Canada Research and Collection Permit (ANP-2011-8533), Qikiqtani Inuit Association (authorization Q13X009 – permission to access Inuit-owned lands).

The Animal and Use Care Committee at the University of Minnesota (1403-31404A), Greenland Department of Environment (Nuuk), and Greenland Department of Health reviewed and approved the methods used in this research.

CHAPTER 1

INTRODUCTION

We briefly present the history of the different polar bear management systems in Nunavut/Canada and Greenland to provide the background and context on the current assessment of the status of the polar bear subpopulations in Baffin Bay (BB) and Kane Basin (KB). We summarize the results of the previous assessment of the BB and KB subpopulations in the 1990s and the framework established in 2009 for the Canada-Greenland joint management of these shared subpopulations. We describe the process leading to the decisions on major objectives of a re-assessment and the subsequent development of a research plan for re-assessing the status of the polar bear subpopulations in Baffin and Kane Basin. Finally, we present the schedule of the completion of the study.

1.1. History of Polar Bear Management in Canada and Greenland related to Baffin Bay (BB) and Kane Basin (KB)

Wildlife legislation in Canada did not address polar bear harvesting until 1935, when a hunting season from 1 October through 31 May was imposed. In 1949, hunting was restricted to native people. Arbitrary quotas for polar bears were introduced in Canada in 1967 that were based largely on the fur records from several preceding years (Lee and Taylor 1994, Prestrud and Stirling 1994).

Through a delegation of authority from the federal government, ultimate responsibility for the management of polar bears in Nunavut lies with the Government of Nunavut, as represented by the Minister of Environment (Lunn et al. 2010). However, this responsibility is subject to the terms of the Nunavut Land Claims Agreement (NLCA) that established a system of ‘co-management’ for wildlife. Under the NLCA, the Minister’s decision-making authority for wildlife management is shared with the Nunavut Wildlife Management Board and is subject to strict requirements for consultation with Regional Wildlife Organizations and community-based Hunters and Trappers Organizations. The intent of this co-management system is to ensure that decisions are based on the best available science and Inuit Qaujimajatuqangit (Inuit traditional knowledge), and that these decisions consider not only conservation as a founding principle but also take into account the values, beliefs, views and needs of Inuit. The system is also designed to ensure that Inuit are involved in all aspects of wildlife management including research, monitoring, and harvest management.

Polar bears occur in relatively discrete subpopulations and are managed as such in Canada. The annual polar bear harvest in Nunavut is within the estimated sustainable yield of females and controlled through a male-biased, sex-selective quota system (Taylor et al. 1987, 2008b). Females accompanied by cubs, cubs, and bears in or constructing dens are protected by law (Lunn et al. 2010). Currently, the quota year in Canada runs from 1 July through 30 June of the following year.

In Greenland, regulations for the catch of polar bears in the entire country were enforced beginning 1 January 1975 (prior to 1975 regulations in NE Greenland had existed since the 1937; Born 1995 and references therein). The regulations prior to the introduction of quotas in Greenland in 2006 to reduce hunting effort and protect females with cubs and also involved a closed season in July-August (ibid). Furthermore, hunting of polar bears was restricted to Greenlandic citizens who had hunting as their main occupation. Quotas for the Greenlanders´

catch of polar bears were introduced in 2005 and took effect 1 January 2006 (Lønstrup 2006). The quota year in Greenland is between 1 January and 31 December.

The management history and harvest monitoring in Nunavut and Greenland are described in more detail in Chapter 8.

1.2. Previous Delineations and Assessments of the Baffin Bay (BB) and Kane Basin (KB) Subpopulations

Delineation of the Baffin Bay and Kane Basin subpopulations (Figure 1.1) was largely based on movements of collared bears and the recapture or harvest of tagged animals and has been well documented (PBSG 1998, SWG 2010). The BB subpopulation is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island (Nunavut, Canada) to the west (Taylor et al. 2001). A relatively distinct southern boundary at Cape Dyer, Baffin Island, and the entrance to Kangerlussuaq/Søndre Strømfjord, Greenland, is evident from the movements of collared or tagged bears (Stirling *et al.* 1980, Taylor et al. 2001).

A study of microsatellite genetic variation based on biological samples collected during the first half of the 1990s revealed significant genetic variation between polar bears in BB and neighboring Davis Strait (DS) and Lancaster Sound (LS), but not between BB and KB (Paetkau et al. 1999).

The BB, KB and DS subpopulations are shared between Greenland and Canada (Taylor et al. 2001). Population inventories involving physical mark-recapture, in combination with satellite telemetry, were conducted jointly by Nunavut/Canada and Greenland during 1991-1997 with the objective to estimate the size of the BB and KB subpopulations. These resulted in an estimate for BB of 2074 bears (95% CI: 1544-2604; Taylor et al. 2005, PBSG 2006) and for KB of 164 bears (95% CI: 94-234; PBSG 2006, Taylor et al. 2008).

Due to concerns with respect to the reported harvest occurring in BB, modelling was used to project how many bears there would be in 2004 based on the 1997 BB estimate and associated vital rates plus the reported annual catches in Nunavut and Greenland. The results suggested that BB would have experienced a decline to 1546 polar bears in 2004 (95% confidence interval: 690-2402; PBSG 2006). Although a similar modeling exercise was not done for KB, both subpopulations were thought to be declining as a result of overharvest (PBSG 2006, 2010, 2015).

1.3. Canada-Greenland Joint Commission (JC) on Polar Bear and the Scientific Working Group (SWG)

The Canada-Greenland Joint Commission on Polar Bear (JC) was established with the signing of a "*Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*" (MOU) on 30 October 2009 (Anon. 2009). Primary objectives of the MOU are to: "(1) to manage polar bear within the Kane Basin and Baffin Bay management units in order to ensure their conservation and sustainable management into the future, and, (2) establish an effective system of management which will include adhering to the principles of conservation".

The JC subsequently established a 5-member Scientific Working Group (SWG) to provide it with scientific advice and recommendations with respect to the conservation and management of the BB and KB polar bear subpopulations. Two of the SWG´s members shall represent Canadian and two Greenlandic research institutes/agencies, whereas the fifth member is appointed by the SWG co-chairs from a research institute that is independent of both Canada and Greenland. To assist the SWG in providing the best scientific advice, external experts can be invited to participate in its work.

1.4. Subpopulation Re-Assessment

After an initial meeting in Ottawa, Canada in January 2010, the JC tasked the SWG with using the best available scientific information to:

(1) Propose Total Allowable Harvest (TAH) levels for the Kane Basin and Baffin Bay subpopulations and provide the Joint Commission with a written report of its recommendations; and,

(2) Provide science advice to the Joint Commission for monitoring the effects of habitat changes on polar bears.

The SWG provided a report to the JC at the 2nd meeting of the JC in Ilulissat, Greenland, in May 2010 (SWG 2010). The SWG noted that 100% of simulations using population viability analysis and current harvest levels showed that both the Baffin Bay and Kane Basin subpopulations declined after 10 years (PBSG 2010). The SWG also noted that simulations to assess the estimated risk of subpopulation decline are typically run 10-15 years into the future from the point in time at which data were last collected to estimate abundance and vital rates. Furthermore, it was noted that there was uncertainty surrounding the magnitude of impacts of environmental change on polar bears and that demographic rates could have significantly changed since the last inventories of these subpopulations in the 1990s.

The SWG also noted that a common Canada-Greenland management goal for the BB and KB subpopulations had not yet been specified in detail. Accordingly, the SWG assumed that a recommended TAH for BB and KB should ensure that each subpopulation remained at the accepted science-based estimate (PBSG 2010) until new scientific inventories had been conducted and management goals established.

Acknowledging the fact that considerable uncertainty existed about the status of the BB and KB subpopulations in 2010 and that clearly defined management goals had not been identified, the SWG estimated the sustainable TAH from the BB subpopulation to be 90 bears/year and to be 1-2 bears/year from the KB subpopulation (SWG 2010).

In order to address the second question posed by the JC, the SWG summarized items that should be monitored in BB and KB together with the possible monitoring approaches. A list of general scientific areas where monitoring is required, the rationale for the monitoring, and the potential methods that can help gather data under each topic were presented. Furthermore, the SWG indicated, where possible, how monitoring can be conducted by scientists and local users concomitantly.

Based on consideration of the key parameters that should be monitored in order to determine the effects of climate change on the BB and KB polar bear subpopulations and the need to ensure that harvesting of the two subpopulations is sustainable in a changing environment, the SWG recommended that the following research needs were given priority:

(1) Subpopulation size

(2) Distribution and movements

(3) Physical condition of individual bears

(4) Vital parameters (survival and reproduction)

The SWG also noted that other data useful for determining the effects of habitat change in polar bears should be collected routinely. Collection of these data involves the active participation of users through the submission of information and samples from harvested polar bears. Sampling from the harvest will provide important information on age and sex structure of the harvest, body condition, reproduction, and levels of pollutants in polar bears.

However, given the age of the data on abundance and vital rates (PBSG 2010) combined with large-scale environmental changes in Baffin Bay during the last decades, the SWG **strongly recommended** that new estimates of subpopulation abundance, subpopulation delineation, and vital rates be given high priority (SWG 2010).

Pilot aerial surveys in Baffin Bay in 2009 and 2010

In 2009 (i.e., prior to establishment of the SWG), scientists from the Government of Nunavut Department of Environment (GNDE) and Greenland Institute of Natural Resources (GINR) initiated a collaboration to determine the feasibility of using aerial surveys for estimating the size of polar bear subpopulations in BB and KB. A pilot survey was conducted in the Sirmilik National Park on NE Baffin Island in August 2009 to determine if the line-transect aerial survey technique could be used during the ice-free or open-water season in Baffin Bay. However, given the results from the rugged terrain and mountainous landscape, it was concluded that this method was not feasible (Stapleton 2010).

Based on the experience in 2009, scientists from Nunavut and Greenland decided to (1) conduct a pilot aerial survey over sea ice during spring in Baffin Bay, and (2) deploy satellite-transmitters on polar bears in Baffin Bay in order to collect data on distribution and movement necessary for evaluation of aerial survey data.

During 27 May-4 June 2010, a pilot survey was flown over the fast and pack ice in Baffin Bay off SE Baffin Island. The survey was determined to be a success and provided data that demonstrated promise for the approach in BB. Based on the results of the 2010 spring pilot survey, a group of experts in polar bear ecology and population dynamics, population assessment methods and sea ice from the Greenland Institute of Natural Resources, Government of Nunavut, University of Washington, University of Minnesota, and U.S. National Marine Mammal Laboratory evaluated the data and concluded that a springtime aerial survey would be a feasible approach for estimating abundance of polar bears in BB and KB and should result in estimates with adequate precision for management purposes (Laake 2010, Stapleton et al. 2010, SWG 2011).

Survey options review

Following the advice from the SWG, the JC tasked the SWG to review and evaluate various methods for assessing the number of polar bears in BB and KB (JC 2010a). The JC further requested that the SWG´s report include evaluation of benefits and limitations of using each of the proposed methods (aerial surveys, genetic mark-recapture, and physical mark recapture) and indicated that the review should be based on a pilot aerial survey conducted in Baffin Bay in Spring 2010 (see preceding section). The SWG was also tasked with providing recommendations for one or more scenarios that represented the best way forward and, for each scenario, to identify major attributes, risks and management questions including level of funding (JC 2010a). Subsequently the JC would make recommendations on the most appropriate survey methodologies for assessing the BB and KB subpopulations (JC 2010a).

In January 2011, the SWG submitted to the JC a review of options for conducting new research including recommendations on appropriate methodologies (SWG 2011), which considered the pros and cons of using physical mark-recapture, genetic mark-recapture, or aerial surveys for subpopulation inventories in BB and KB. The general conclusion was that the physical MR method (1) is the most well established method available for estimating abundance of polar bears, (2) is the most widely accepted and recommended method by the greater scientific community, and (3) provides the maximum information needed for sound management advice on polar bears. In addition to an abundance estimate, it provides information that could be used to assess effects on bears of climate change and pollution.

The SWG concluded that the physical MR method was superior to aerial surveys and genetic MR because it yields the most detailed information and recommended that physical MR be used for estimating the abundance of polar bears in Baffin Bay. It was stated that aerial surveys and a multiple-year genetic MR may be considered as alternatives for assessing the number of polar bears in Baffin Bay. However, the SWG noted that there are disadvantages to both methods.

Decision of the JC

In March 2011, the JC discussed survey methods for Kane Basin and Baffin Bay with Drs. Erik Born and Stephen Atkinson who represented the SWG. Following discussion of the SWG's report, the JC concluded that the genetic mark-recapture method was preferable based on lack of support by Inuit in Nunavut for physical mark-recapture, and on concerns that variability in sea-ice conditions have the potential to create an unacceptable amount of risk in obtaining accurate subpopulation estimates via aerial surveys. Thereafter, the JC recommended to the signatories of the MOU that a 3-year biopsy darting research program be developed.

1.5. A Multi-Year Research Plan for Baffin Bay and Kane Basin

In March 2011, the JC requested the SWG prepare a research plan for the re-assessment of the Baffin Bay and Kane Basin polar bear subpopulations. It was determined that preparation of such a plan was the responsibility of those members of the SWG who would serve as lead investigators on behalf of Nunavut and Greenland. Consequently, a multi-year research plan (Atkinson et al. 2011) for re-assessment of the Baffin Bay and Kane Basin subpopulations was developed by Drs. Stephen Atkinson (Nunavut), Erik Born and Kristin Laidre (Greenland Institute of Natural Resources).

The plan outlined a multi-year research program to be carried out collaboratively by scientists from Canada (Nunavut) and Greenland together with external collaborators (local people and scientists) participating in various parts of the study. The plan presented tentative schedules and budgets. The main goals of the research program were (1) to determine the size of the Baffin Bay and Kane Basin subpopulations of polar bears, and (2) evaluate how polar bears in these areas are affected by the decrease in sea ice.

The proposed research program had three basic field components: (1) Biopsying polar bears along E Baffin Island, in NW Greenland and in the Kane Basin region, (2) deployment of satellite transmitters on male and female polar bears in NW Greenland and Kane Basin, and (3) hunter collection of tissue samples from the catch of polar bears (harvest recoveries) in BB and KB (and adjacent subpopulations). A 3-year study was proposed beginning in the fall of 2011 with the purpose to:

1) Estimate the abundance and sex (and approximate age) composition of polar bears in BB and KB;

2) Compare a new estimate of abundance with those derived from previous studies (1991-1997) in-order to gain insight into subpopulations trend;

3) Delineate the boundaries of the BB and KB subpopulations and reassess the validity of these areas as a demographic unit;

4) Estimate survival and reproductive parameters (to the extent possible) in-order to facilitate population viability analyses; and,

5) Evaluate polar bear distribution with respect to environmental variables, particularly ice conditions, topography and food availability/distribution.

Results generated by the proposed research program have the following potential applications:

1) The development of an updated status report for BB including recommendations on sustainable harvest levels; and,

2) The development of models to assess the effects of changes in habitat (in particular sea ice) on bear distribution.

Schedule

In BB, main field operations were conducted during spring and fall. Due to logistical constraints (remoteness of the survey area and consideration of light conditions in fall) field work was concentrated in spring in KB. The schedule of the proposed study was:

Biopsying for genetic mark-recapture assessment

1) Fall biopsying along eastern Baffin Island fall 2011, 2012 and 2013.

2) Fall biopsying in NW Greenland fall 2012 and 2013.

3) Spring biopsying in NW Greenland 2011, 2012 and 2013.

4) Spring biopsying in Kane Basin 2012, 2013, and 2014 (optional).

Deployment of satellite radios

1) Deployment during spring in NW Greenland 2011, 2012 and 2013 of satellite radios (satellite radios had also been deployed in these areas during spring 2009 and 2010).

2) Deployment of satellite radios during spring in Kane Basin 2012 and 2013.

Hunter collection of tissues from the polar bear catch (harvest recoveries)

1) Nunavut spring 2011 and onward.

2) Greenland spring 2012 and onward.

Data analyses

1) Genetic laboratory analyses, analyses of MR data, genetic data and satellite telemetry data 2012 and onward

Final reporting

1) September-October 2014

1.6. Process of BB and KB Assessment (2011-2016) and Delays

Program activities

Activities (field operations, laboratory analyses, data analyses, and reporting) in connection with the research program are summarized here. Details of the various activities are presented in the Materials and Methods sections of the various chapters of this report.

Field activities

Personnel from several research institutions participated in planning and conducting the field work including GINR, GNDE, University of Oslo, and University of Minnesota. In addition, local polar bear hunters in Nunavut and Greenland participated on several flights aiming to obtain biopsy samples from unrestrained bears and/or immobilizing bears to furnish them with satellite radios (NW Greenland and Kane Basin). In both Nunavut and Greenland, community consultations were conducted each year and information about the study was provided through meetings and via public media to local communities and the broader public audience.

The spring and fall biopsy program (2011-2013) along Baffin Island and in W and NW Greenland from the BB subpopulation was conducted as planned. From 2011 to 2013, 1,111 bears were biopsy darted along eastern Baffin Island (and genotyped) which was substantially more than anticipated. From 2009 to 2013, 143 bears were physically marked or biopsy darted (and genotyped) in W and NW Greenland. The spring biopsying program in Kane Basin was also successful although a third spring season was needed and completed during 2014. From 2012 to 2014, 129 bears were physically marked and genotyped or biopsy darted and genotyped in KB (Chapter 2 and 5).

Additional to the original research plan and concurrent with the biopsy sampling, a systematic aerial survey using sight-resight distance sampling protocols was conducted during spring 2014 to assess the number of polar bears in the Kane Basin subpopulation. Adding this extra component allowed for a comparison of estimates of subpopulation size via two different methods (i.e., genetic MR and aerial survey).

During 2011-2013 a total of 66 satellite radios (35 F, 31 M) were deployed in W and NW Greenland (in addition 35 satellite radios had been deployed there in 2009 and 2010; 20 F, 15 M). During 2012 and 2013 a total of 36 satellite radios (21 F, 15 M) were deployed in the Kane Basin region. Some individuals were recaptured during the study and furnished with new satellite radios. Hence, a total of 91 individual bears were tagged with satellite transmitters in BB and 34 individual bears in KB (Chapter 2). The satellite radios included small ear satellite tags developed by GINR for tracking adult male polar bears and sub-adults of both sexes (Born et al. 2010, Laidre et al. 2012).

A total of 234 hunter recoveries (tissue samples) were obtained from the catch of polar bears in Nunavut and Greenland (1993-2013). The hunter recovery program was instituted in Greenland for the first time in 2012. In addition, 635 biopsies from physical MR operations to assess BB and KB subpopulations in the 1990s (*cf.* Taylor et al. 2005, 2008) were included in the recent MR assessment analyses (Chapter 2 and 5).

In summary – All field operations were conducted as planned and were very successful. The number of biopsies obtained from the BB subpopulation was substantially higher than expected. All handling in NW Greenland and Kane Basin of individual polar bears in connection with deployment of satellite radios were made without any complications. The general pubic and local communities were informed about the operations and local polar bear hunters participated on several of the flights to obtain biopsies or immobilize polar bears.

Analyses

Wildlife Genetics International (Nelson, British Columbia, Canada) analyzed all genetic samples (9 nuclear markers).

Satellite telemetry data (habitat analyses) were analyzed under leadership of the Greenland Institute of Natural Resources and Dr. Kristin Laidre.

Analyses of ice metrics were conducted by Dr. Harry Stern at the Polar Science Center (University of Washington, USA) in collaboration with Laidre.

Analyses of the genetic MR recapture data were conducted at Department of Fisheries, Wildlife and Conservation Biology (University of Minnesota) under the leadership of postdoctoral research associate Dr. Seth Stapleton, and his supervisor, Professor Todd Arnold. Salary for Dr. Stapleton´s postdoctoral fellowship was provided by the Government of Nunavut, Environment Canada, and GINR. Close cooperation on MR analyses was conducted with outside expert Dr. Eric V. Regehr (US Fish and Wildlife Service).

Analyses of field observations of polar bear body condition were made by Nunavut under leadership of Dr. Stephen Atkinson.

Population genetic analyses to determine the demographic identities of Baffin Bay and Kane Basin subpopulation were made by Dr. Liselotte Wesley Andersen at Department of Bioscience (University of Århus, Denmark) based on the results of analyses of nine nuclear markers used in the genetic MR study.

All SWG members participated in various phases of analyses. External experts (Appendix A) also participated in the analyses and in three face-to-face progress meetings of the SWG held at the Polar Science Center, University of Washington.

Timeline of analyses and reporting

The SWG originally proposed an October 2014 deadline for submission of a final report to the JC (SWG 2011). However, due to the time required for completion of sample processing in the genetic laboratory this deadline could not be met. The delay was a consequence of the largely successful field work, which exceeded expectations in terms of the quality and number of biopsy samples. As the mark-recapture modeling could not begin until the laboratory analyses were completed and the final datasets compiled, a new deadline for a final report of 30 April 2015 was proposed by the SWG (SWG 2014). However, further unanticipated delays occurred related to availability of historical samples and a final comprehensive dataset was not available until mid-December 2015. This delay put the analyses approximately 6 months behind schedule (SWG letter to JC).

In October 2015, the SWG held a 2nd face-to-face meeting in Seattle, Washington. At the meeting, preliminary results from analyses on abundance and vital rates, population genetic structure, range use, distribution, seasonal movements, and trends in sea-ice habitat were presented and evaluated. Near-final results of the analyses of range use, distribution, seasonal movements, and trends in sea-ice habitat and genetic analyses were also presented. However, the presentation of preliminary results from the mark-recapture modeling revealed that a considerable amount of additional work was still required from collaborators at University of Minnesota (SWG 2015). This work included, among other things, more in-depth error checking and a more detailed exploration of data from the MR studies in the 1990s in order to assess bias and potentially detect trends in abundance. This required revising the basic structure of the population models, developing and running model simulations, and validating the final model results before a final report could be completed.

Following this meeting the SWG and collaborators held 13 teleconferences between November 2015 and April 2016 where progress in the MR modeling of abundance in BB and KB was discussed and evaluated. A third face-to-face meeting was held in February 2016 to evaluate the revised modeling results.

After detailed discussions of the results of the MR assessment of the BB and KB subpopulations the SWG identified items for further analyses by University of Minnesota before final results could be sent to external scientific review (SWG 2016). In May 2016, results of the mark-recapture assessments were sent to Dr. Gary White (Professor Emeritus, Colorado State University) for a courtesy, external peer-review. Dr. White is a world-expert in mark-recapture population analyses. Dr. White gave the analyses a positive review and approved the analytical methods used and their results.

In summary the analyses of sea ice, movement and habitat use were conducted according to the original time plan. However, unanticipated delays in getting a final genetic dataset and complications related to the modeling of the genetic data resulted in delays in preparing the final report to the JC.

1.7. Literature Cited

Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations,* 9 pp.

Atkinson, S., E. W. Born, and K. Laidre. 2011. A research plan for re-assessment of the Baffin Bay and Kane Basin polar bear sub-populations. Unpublished report submitted to the Canada-Greenland Joint Commission on Polar Bear, 28 pp.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E.W., C. Sonne, and R. Dietz 2010. Research on polar bears in Greenland 2005-2009, Pages 135-148 *in* M. E. Obbard, G. W. Thiemann, E. Peacock and T. DeBruyn, editors. Proceedings of 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2010a. Request for review of survey methods. Letter from the Canada-Greenland Joint Commission on Polar Bear to the Scientific Working Group to the Joint Commission on Polar Bear, 26 October 2010.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2010b. Report of the 2nd meeting of the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 3 pp plus appendices.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2011. Greenland Canada Nunavut JC - Monitoring recommendations. E-mail from Basile van Havre to the JC and SWG, 2 March 2011.

Laake, J. 2010. Unpublished report from J. Laake, National Marine Mammal Laboratory, Seattle, to project collaborators: Government of Nunavut, Greenland Institute of Natural Resources, Qikiqtarjuaq HTO, the University of Minnesota, and the University of Washington, 7 pp.

Laidre, K.L., E.W. Born, E. Gurarie, Ø. Wiig, R. Dietz, and H. Stern 2012. Females roam while males patrol: Divergence in breeding season movements of pack-ice polar bears (*Ursus maritimus*). Proceedings of the Royal Society B: Biological Science (London) doi.10.1098/rspb.2012.2371

Lee, J., and M. Taylor. 1994. Aspects of the polar bear harvest in the Northwest Territories, Canada. International Conference on Bear Research and Management 9:237–243.

Lønstrup, J. 2006. Polar bear management in Greenland. Pages 133–134 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Lunn, N. J., M. Branigan, L. Carpenter, J. Justus, D. Hedman, D. Larsen, S. Lefort, R. Maraj, M. E. Obbard, E. Peacock, and F. Pokiak. 2010. Polar bear management in Canada, 2005-2008. Pages 87–113 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1995. Summary of polar bear population status 1993. Pages 19–24 *in* Ø. Wiig, E. W. Born, and G. W. Garner, editors. Polar Bears: Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014. http://pbsg.npolar.no/en/status/status-table.html, Accessed 18 June 2016.

Peacock, E., A. E. Derocher, G. W. Thiemann, and I. Stirling. 2011. Conservation and management of Canada's polar bears (*Ursus maritimus*) in a changing Arctic. Canadian Journal of Zoology 89:371–385.

Prestrud, P., and I. Stirling. 1994. The International Polar Bear Agreement and the current status of polar bear conservation. Aquatic Mammals 20:113–124.

Stapleton, S. 2010. Polar bear monitoring and aerial survey assessment in Sirmilik National Park: Final Project Report. Unpublished report to Parks Canada, Mittimatilik Hunters and Trappers Organization, Government of Nunavut, University of Minnesota, and WWF-Canada, 11 pp.

Stapleton, S., T. Arnold, S. Atkinson, and D. Garshelis. 2010. Baffin Bay polar bear aerial survey: Pilot results and research alternatives. Unpublished report to project collaborators: Government of Nunavut, Greenland Institute of Natural Resources, Qikiqtarjuaq HTO, and the University of Minnesota, 23 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2010. Report of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 18 pp

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2014. Outcome of January 2014 Scientific Working Group Meeting, Letter to the Canada-Greenland Joint Commission on Polar Bear, 24 January 2014, 3 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2015. Report of Working Meeting of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 5-7 October 2015, Polar Science Center, University of Washington, Seattle, Washington, USA, 13 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Report of Working Meeting of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 15-19 February 2016, Polar Science Center, University of Washington, Seattle, Washington, USA, 45 pp.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987. Modelling the sustainable harvest of female polar bears. Journal of Wildlife Management 51:811–820.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

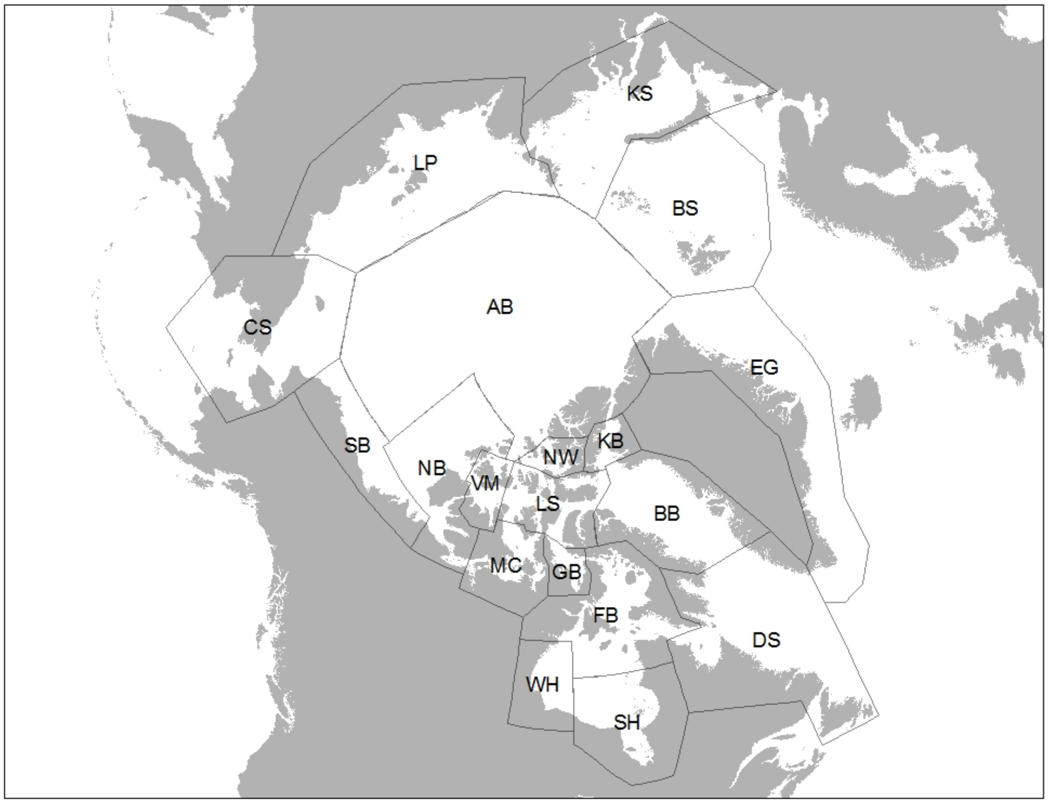


Figure 1.1. Map of the circumpolar Arctic showing the 19 subpopulations of polar bears, including Baffin Bay (BB) and Kane Basin (KB). Polar Bear subpopulations: Arctic Basin (AB), Baffin Bay, Barents Sea (BS), Chukchi Sea (CS), Davis Strait (DS), East Greenland (EG), Foxe Basin (FB), Gulf of Boothia (GB), Kane Basin, Kara Sea (KS), Lancaster Sound (LS), Laptev Sea, M’Clintock Channel (MC), Northern Beaufort Sea (NB), Norwegian Bay (NW), Southern Beaufort Sea (SB), Southern Hudson Bay (SB), Viscount Melville Sound (VM), and Western Hudson Bay (WH).

CHAPTER 2

SUBPOPULATION DELINEATION OF BAFFIN BAY AND KANE BASIN

KEY FINDINGS

Baffin Bay (BB)

- The 1990s and 2000s satellite telemetry data are comparable for analysis; approximately 92% of adult females collared in West Greenland in spring during the 2000s use the same area on Baffin Island where adult females were captured and collared in fall in the 1990s.
- There was a significant reduction in the size of the 2000s BB 95% bivariate normal kernel range (i.e. a measure of the area used by collared bears) in all months and seasons when compared to the range in the 1990s. The most marked reduction was a 60% decline in subpopulation range size summer.
- The overlap of the 1990s and 2000s BB ranges was < 50% in all months, reflecting both a contraction and shift of the BB subpopulation range in the 2000s. These shifts are related to the loss of annual sea ice and changes in breakup timing, contracting the range of the BB subpopulation and shifting the distribution of BB polar bears northward in all seasons. The BB subpopulation is still distributed within the current management boundaries.
- There were significant shifts north in the median subpopulation latitude in all seasons in BB across decades.
- Bears in the 2000s were significantly less likely to leave BB than in the 1990s ($p<0.001$), with reductions in the number of bears moving into Davis Strait and Lancaster Sound.
- Genetic analyses using eight polymorphic microsatellites in recent samples (2009-2014) had low genetic resolution. BB and KB polar bears could not be differentiated genetically. Also in accordance with previous genetic studies, BB-KB polar bears were found to be genetically different from polar bears in Lancaster Sound and Davis Strait.
- During the genetic mark-recapture studies in BB and KB (2011-2014) there were very low levels of recapture or harvest recovery of bears outside their subpopulation of origin.
- Satellite telemetry demonstrates that emigration from BB has been significantly reduced since the 1990s, largely due to reduced sea-ice extent in winter and absence of sea ice in summer. This suggests the BB subpopulation has become more discrete, with less exchange between it and other subpopulations.

Kane Basin (KB)

- KB mean 95% kernel ranges have generally expanded since the 1990s. The increase in range use in the 2000s occurs in all seasons, however is statistically significant only in summer (June-September), where ranges doubled between the 1990s and the 2000s. This range expansion is likely related to changes in sea ice, as KB is trending towards the characteristics of an annual ice ecoregion (like BB) where ice melts out almost completely each summer.
- There is still considerable seasonal overlap in KB subpopulation ranges for bears in the 1990s and 2000s (50-98% overlap over decades), suggesting that bears generally continue to use the same areas of KB.
- There were significant northward shifts in KB median latitude of polar bear locations in the 2000s in spring and summer, although these shifts were smaller than observed in BB. Variability in the range of latitudes has increased; bears in the 2000s use a broader range of latitudes. There has been no change during winter.
- These distribution patterns did not change with a sensitivity analysis in which bears captured in eastern KB were excluded from the 2000s data, to match the distribution of captures in the 1990s. This suggests that our key findings for KB were not influenced by the distribution of capture locations.
- **BB and KB**

Overall, our findings based on satellite telemetry, movement of marked bears and genetics suggest that the existing boundaries of the BB and KB subpopulations continue to be relevant for harvest management purposes and population monitoring.

2.1. Subpopulation Delineation and Status Background

Cluster analyses of movement data from satellite-collared bears (Taylor et al. 2001), genetic analyses (e.g., Paetkau et al. 1999), and recaptures and harvest recoveries of marked (tagged) bears (Taylor and Lee 1995, Taylor et al. 2001) are among the methods that have been used to evaluate and delineate the boundaries of the Baffin Bay (BB) and Kane Basin (KB) subpopulations. For BB, genetic analyses suggest a lack of genetic differentiation of BB from the adjacent KB subpopulation to the north, but a significant genetic difference from polar bears in the Davis Strait (DS) subpopulation (Paetkau et al. 1999). Analyses of satellite collar data and tag returns suggest that some interchange occurs among BB and adjacent subpopulations including DS, Lancaster Sound (LS) and KB (e.g., Taylor et al. 2001). However, for the purposes of management, BB is considered a distinct demographic unit, and its dynamics are largely driven by intrinsic rates of reproduction and mortality rather than exchange with neighboring subpopulations.

For KB, analyses of satellite collar data and tag returns suggest partial closure. However, the discreteness of this subpopulation from neighbouring units has been questioned, in part due to the lack of genetic differentiation from surrounding subpopulations and the potential for immigration from these much larger subpopulations to significantly influence demographic processes in a source-sink dynamic (Taylor et al. 2008). Particularly notable interchange occurs with BB and LS. The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a barrier between KB and BB – LS.

Sea ice in BB and KB has decreased markedly during the last 3 decades (Stirling and Parkinson 2006, Peacock et al. 2012, Laidre et al. 2015, Chapter 4 and 9), with earlier spring break up and later fall formation. The extent to which these trends in sea ice will affect the distribution and boundaries of these subpopulations remains uncertain and requires updated information. In particular, there is a need to re-evaluate these boundaries when undertaking studies to estimate abundance and vital rates to ensure sampling remains consistent with the distribution of the biological subpopulations.

The delineation and status of the BB subpopulation has been documented and updated by the IUCN/SSC Polar Bear Specialist Group (PBSG 1995, 1998, 2002, 2006, 2010, and http://pbsg.npolar.no/en/index.html) and annually by the Canadian Polar Bear Technical

Committee (unpublished). Based on the movements of adult females with satellite radio-collars and the recapture or harvest of tagged animals, the BB subpopulation of polar bears is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island, Nunavut, Canada to the west (Taylor et al. 2001) (Figure 1.1). A relatively distinct southern boundary at Cape Dyer, Baffin Island, and the entrance to Kangerlussuaq/ Søndre Strømfjord, Greenland, is evident from the movements of tagged bears (Stirling et al. 1980) and from adult female polar bears monitored by satellite telemetry (Taylor et al. 2001).

Analysis using microsatellites revealed significant genetic variation between polar bears in BB and neighboring DS, but not between polar bears in BB and neighboring KB (Paetkau et al. 1999). However, bears from BB-KB differed genetically from polar bears in the neighboring LS and DS subpopulations (Paetkau et al. 1999, Peacock et al. 2015, Malenfant et al. 2016). The original separation of the polar bears subpopulations into the two management units Kane Basin and Baffin Bay was based on studies of movement of polar bears with satellite collars in the 1990s and the fact that the North Water Polynya in northern Baffin Bay, to a certain extent, acts as a barrier to movement between BB and KB (PBSG 1998, Taylor et al. 2001). The samples included in the genetic study by Paetkau et al. (1999) were collected in the early 1990s. A study by Peacock et al. (2015) was based on more recent samples (i.e., BB: mainly 2006-2008; DS: 2005-2007; LS: mainly 2008), however the samples from KB were from the 1990s. Malenfant et al. (2016) conducted a re-analysis and relied upon subsets of the same data in Peacock et al. (2015). Hence, the KB-samples in Malenfant et al. (2016) were also not temporally congruent with samples from BB, LS and DS.

Based on the movements of KB adult females with satellite collars and the recapture or harvest of tagged animals, the boundaries of the KB subpopulation include the North Water

Polynya (to the south of KB), and Greenland and Ellesmere Island to the west, north, and east and the southern part of Kennedy Channel to the north (Taylor et al. 2001; Figure 1.1).

There have been no new scientific studies in BB or KB to update information on subpopulation delineation since 1997. We used new data from genetics, satellite telemetry and information on the movements of bears amongst BB, KB and surrounding subpopulations as detected by the recapture or harvest of marked individuals to assess the current BB boundaries. Our objective was to use these data to support other lines of evidence relating to the delineation of these subpopulations.

2.2. Methods

Genetics

In this study, a large recent sample of polar bear tissues was obtained from the subsistence catch and from scientific biopsies in BB, KB, LS and DS (2009-2014). This was obtained primarily for the genetic mark-recapture as reported in Chapters 5 and 10. Data from these analyses were therefore available for an updated examination of population genetics in BB and KB. The majority of the samples were collected between 2011 and 2014. Hence, in contrast to previous studies, the samples collected in connection with the genetic mark-recapture study were both more recent and also temporally congruent (Table 2.1).

The genetic analyses reported here used the same eight polymorphic microsatellite markers as in MR analyses to explore to what extent polar bears in KB, BB, LS and DS differed genetically (e.g., PBSG 2010). The decision to make these analyses was made *post hoc* and genetic analyses were not a part of the original study plan (see Chapter 1). Hence, these analyses were based on a platform of opportunity (i.e., the samples had been analyzed genetically and the data were available), fully recognizing that more nuclear markers would result in a higher resolution.

Genetic data from a total of 1,364 individual polar bears from the BB, KB, LS and DS subpopulations were available for the genetic analyses. A subset of 402 polar bears sampled during the winter-spring season (Nov-June) in 2012-2014 represented all four subpopulations under consideration. This subset consisted of the spring biopsy samples from Kane Basin, and winter-spring harvest samples from BB, LS and DS. We considered the samples to have been collected within a narrow time frame (i.e., “temporally congruent”) that would exclude major displacement of groups of polar bears among BB, KB, LS and DS during the sampling period (Table 2.1).

The population genetic analyses were conducted by Department of Bioscience (Aarhus University, Denmark) using standardized analytical tools and methods (ADEGENET package, Jombart 2008; ARLEQUIN Version 3.5.1, Excoffier and Lischer 2010; BA3-3.0.3, Wilson and Rannala 2003; DAPC, Jombart et al. 2010; FSTAT, Goudet 1995; GENECLASS2, Piry et al. 2004; GENELAND, Guillot et al. 2005, Guillot 2008; STRUCTURE, Prichard et al. 2000).

Satellite Telemetry Studies of Movements and Range Use

Polar bears were tagged in NW Greenland on the fast and pack ice between mid-March and mid-April 2009-2013 in Baffin Bay and mid-April to early May 2012-2013 in Kane Basin. Field operations were based out of coastal settlements in West Greenland or research stations on Ellesmere Island. Searches for bears in BB occurred out to a maximum distance of 150 km from the coast. Areas with consolidated glacier ice at the glacial terminus were also searched in both BB and KB. A total of 91 individual bears were tagged with satellite transmitters in BB and 34 individual bears in KB. Of these, 38 were AFs collared in BB and 20 collared AFs in KB (Table 2.2, Figures 2.1 and 2.2). In the 2000s, both sexes and subadults were tagged. These data were combined with a historical data set of captured and tracked from both areas in the 1990s. In BB, 1991-1995 43 collars were deployed on AFs, with the majority deployed during the ice free season in fall on Baffin Island (n=11 deployed in spring in NWG, 9 of which transmitted for sufficient time to be included in the analyses) (Table 2.3). In KB, 1992-1994, 12 collars were deployed on AFs on the west side of KB in the fjords and fast ice. Only bears captured within the BB or KB subpopulation boundaries as defined by PBSG (2010) were included in the comparative analysis.

Polar bears at all sites were darted and immobilized from an Ecureuil AS350 (BB) or Bell 206 LR (KB) helicopter and handled according to procedures described in Stirling et al. (1989). Standard body measurements (standard length and axillary girth) were taken and total body mass was estimated using the approach of Derocher and Wiig (2002). Field estimates of age and reproductive status were recorded.

Adult female polar bears in the 2000s in both areas were fitted with TAW-4610H satellite radio collars (Telonics, Mesa, Arizona, USA). Satellite collars provided information on geographic location, internal transmitter temperature, and activity. Collars were programmed to transmit during one six-hour period each day on 4-day intervals. In the 2000s, all adult male polar bears and subadults of both sexes were fitted with SPOT-5 S227 satellite radio transmitters (Wildlife Computers, Redmond, Washington, USA) attached to the ear similar to numbered plastic ear tags used in conventional studies (right ear). The SPOT-5 transmitters weighed 32 g and 60 g with attachment system (Born et al. 2010). Ear transmitters were duty cycled to extend battery life, with most tags transmitting on 4-day intervals and others on daily intervals. Satellite tags transmitted around noon local time each day and were programmed so that several locations were received per transmission day.

Data filtering and sub-sampling – Data on locations and transmitter status from all polar bears were collected via the Argos Location Service Plus system (Toulouse, France). Location qualities are assigned by ARGOS to each position, with location qualities of 0–3 estimated to have errors of 1.5 km or less and those categorized as 'A,' 'B,' or 'Z' have no predicted accuracy. Unrealistic and poor quality locations were removed using a speed and angle filter in R version 2.13.2 (R Development Core Team 2012) using the package 'argosfilter' (Freitas et al. 2008). Positions exceeding a maximum between location travel velocity (10 km/h based on previous movement studies of polar bears, Laidre et al. 2013) and angle (measured from the track between three successive locations; set to the default) were removed by the filtering algorithm. The resulting locations for each bear were next reduced to a single position per day to reduce autocorrelation bias, standardize temporal sampling, and address the effects of variable duty cycling among the tags. To obtain a daily position for each tag, the first, best quality location within the period of peak satellite passage was selected. Daily positions, after filtering and optimal daily position selection, only consisted of ARGOS qualities 1–3. Distances between successive daily positions were calculated as the great circle route and used to compute minimum daily displacements.

As a result of variable experimental objectives in both subpopulations and decades, different duty cycles were used for tags in an effort to extend battery life or gather information from specific time periods. The 1990s collars were programmed to transmit on varying and intermittent intervals, ranging from 1 to 6 days, while the 2000s collars were all on a 4-day cycle. We sub-sampled the 1990s data and created a strict 4-, 5- or 6- day interval time series for each individual to best match the 2000s data. This ensured that the impact of serial autocorrelation was consistent.

Captured polar bears were classified as independent adult male (AM), adult female (AF), subadult male (SM), subadult female (SF), or dependent cubs (cub of year COY, yearling YRL or 2-year old 2YR). We also classified the habitat type where the bear was first located and captured into three categories– pack ice (open or loose ice with leads and cracks), fast ice (sea ice attached to land with no open water), glacier ice (consolidated glacier ice at glacier fronts), as well as captures on land.

Data were divided into seasons: Spring (March – July, which included the peak of sea-ice coverage and initiation of sea-ice break-up), Summer (August – October, which included the end of break-up and the on-land period) and Winter (November – February, which included the freeze-up period and time when bears went back out on the sea ice). All denning periods were identified (maternity and shelter dens) (Escajeda 2016) and removed from RSF models. RSF models were only conducted on AFs to enable comparison with the 1990s. Bears with < 3 locations were removed from analyses as this was likely due to transmitter failure immediately after capture. Ages of polar bears were provisionally estimated in the field and later confirmed more accurately from counting of cementum growth layers of a pre-molar extracted during capture following methods in Calvert and Ramsay (1998). Adult females were defined as ≥ 5 years old and adult males as ≥ 6 years old. Age group status as determined in the field was verified based on tooth analyses.

We assigned each polar bear location to its respective subpopulation boundary (starting point or origin as well as the boundary where the bear was located at each time step) based on the boundaries recognized by the IUCN Polar Bear Specialist Group (PBSG 2010).

Basis for comparison of the 1990s and 2000s BB satellite telemetry data – Polar bears within the subpopulation boundaries of BB are treated as belonging to a single management unit. The bears range over the entire Baffin Bay with the majority of bears spending the summer on Baffin Island during the ice-free period. There were differences in capture locations between decades (Figure 2.1), and although these captures were all within the bounds of the BB subpopulation per PBSG (2010) they were captured in different areas and seasons. Thus we conducted analyses to ensure that the movement of the bears was comparable across periods. In the 1990s, n=43 adult females were collared between 1991 and 1995. Approximately 72% of these were captured on land on Baffin Island in fall. In the 2000s, all n=38 adult females were captured on the spring fast ice and pack ice between 2009 and 2013 in West Greenland.

We spatially bounded the 1990s fall capture region along the coast of Baffin Island and examined what fraction of bears collared in the 2000s (in spring) in West Greenland used the same area the following fall. We defined fall as any period between August and November. All West Greenland 2000s bears were considered “independent” because they were captured in spring. Some individuals that remained resident during all seasons on or close to glaciers in Melville Bay and bears with collars that failed to transmit for >2.5 months after spring capture (which occurred in mid-April) were excluded from the calculations.

Monthly and Seasonal Kernel Density Estimates – Using a fixed kernel density approach (Worton 1989), we estimated the geographic areas characterized by a high probability of use by satellite-radio tagged AF polar bears in BB and KB. Kernel density estimators provide a non-parametric probability of using a given point in space and are reliably used to define the utilization distribution, or home range, for marine and terrestrial wildlife (Kie et al. 2010).

We calculated Gaussian bivariate normal kernel density estimates for each subpopulation (BB, KB, and KB West), in each decade, and for each month (n=12) and season (n=3). Kernel Density Estimates (KDEs) were calculated using the "bkde2D" function in "KernSmooth" R package (Wand 1994, Wand et al. 1995). The sample size of tagged AF differed between the 1990s and 2000s (Table 2.2, Table 2.3). To account for any potential bias in the KDEs or the fraction of overlap between decades due to differing numbers of AFs we randomly sampled with replacement from the pool of AF bears in each the two decades (1990s and 2000s) so that the sample sizes of collared bears were equivalent during each time period (n=38 bears in BB in both decades, n=12 bears in KB in both decades). We sampled bears with replacement 1,000 times for each monthly and seasonal KDE and calculated the area of the 95% contour polygon (bounding 95% of the KDE surface volume). We produced a mean and bootstrapped standard error (SE) for monthly and seasonal home ranges, calculated the fraction of overlap for each time period, and statistically compared time periods. We used the ‘intersect’ tool in ArcGIS to identify overlapping home ranges between subpopulations. We also estimated the proportion of home range overlap between the 1990s and the 2000s (Fieberg and Kochanny 2005) based on the bootstrapped mean. The cell size was set to 6 km and bandwidth of 50 km (approximately 50% of the 4-day movement step of AFs in this study). Cell size determines the smoothness of the resulting prediction, but has minimal impact on kernel density estimation relative to bandwidth selection. The bandwidth controls the width of the estimated kernel thereby determining how much regional variation is emphasized.

Changes in Median Latitude – We also calculated median latitude and longitude values for the 1990s and 2000s using pooled data from all AFs by season. The north-south orientation of the BB and KB subpopulation ranges allowed for this comparison. We compared changes in median latitude and longitude across decades with t-tests at a significance level of $\alpha = 0.05$. In the 1990s KB, all bears were caught along eastern Ellesmere Island (i.e., western KB) whereas bears were captured in both western and eastern KB in the 2000s. We performed a sensitivity analyses to evaluate this sampling difference with a subset of the KB bears captured in the 2000s. This subset included only those bears captured along the coast of Ellesmere Island in western KB. We compared them to the sample from the 1990s to examine any bias in comparison across decades.

Movements across Subpopulation Boundaries – We considered each 4 to 6 day AF bear trajectory as a single sample and calculated the elapsed time spent in the subpopulation region of origin (defined as where the bear was captured and tagged). Specifically, we calculated the number of days until each polar bear left its subpopulation region of origin and plotted the time-until-departure for each subpopulation and decade. Bears that never left their region of origin still contributed follow-up time, but their observation time was censored at time of last transmission. Statistical methods for censored event times were used to construct "survival" curves (Kaplan-Meier) to characterize the distribution of exit times from BB or KB and test for differences among different subpopulations/decades (log-rank test of equality) with $\alpha = 0.05$. We considered two time scales for departure: (1) a departure from the region of origin to be any length of time (4 days minimum) and (2) a departure from the region of origin that was at least 30 days long.

For bears that were observed to leave their subpopulation region of origin we summarized which subpopulation they departed to and the month of departure. We contrasted departures between BB and KB and across decades. As there were two capture seasons in the

1990s (spring in Melville Bay and summer on Baffin Island) we also tested whether capture season impacted the time until departure from the region of origin.

Recaptures and Harvest Recoveries of Marked Bears

Bears included in this study were marked in springtime (April – May) or fall (August – October) during three periods; 1991-1997, 2005-2007 and 2009-2014. From 1991 to 1997, 881 and 141 bears were captured and physically marked with ear tags and lip tattoos as part of studies in BB and KB, respectively (Taylor et al. 2005, 2008). In preparation for genetic mark-recapture studies in BB and KB commencing 2011, tissue samples collected from these bears were subsequently genotyped in 2011 with two exceptions: 1) bears that were known to have been harvested between 1991 and 2011 and 2) bears whose known or estimated age would have been greater than 35 years in 2011. In total, this dataset consisted of 650 individuals marked in the 1990s that would have been ≤ 35 years old and had not been harvested by the time genetic sampling began in 2011. Samples for genotyping were available for 635 of the 650 individuals. Genotyping followed methods described elsewhere in this report (Chapter 5).

From 2005 to 2007, 1518 bears in DS were physically marked (and subsequently genotyped) as part of a mark-recapture study (Peacock et al. 2013). From 2011 to 2013, 1111 bears were biopsy darted along western BB (Canada) and genotyped. From 2009 to 2013, 143 bears were physically marked and genotyped or biopsy darted and genotyped in eastern BB (Greenland). From 2012 to 2014, 129 bears were physically marked and genotyped or biopsy darted and genotyped in KB.

Recapture or harvest recovery of physically or genetically marked individuals was detected by two means. Prior to 2011, when biopsy darting began, marked BB and KB individuals were identifiable by ear tags and lip tattoos. During this period, recaptures of marked individuals were recorded during physical capture sampling in BB, KB, and surrounding subpopulations. Harvest recoveries of marked bears were detected via hunter returns of ear tags and / or lip tattoos as part of the on-going harvest monitoring program across all subpopulations in Canada and Greenland (Peacock et al. 2012). From 2011 onwards, all marked individuals were genotyped and some were both physically marked and genotyped. Recapture or harvest recovery of marked individuals was detected by physical marks recorded during capture sampling and harvest monitoring or by matching the genotypes of marked bears to samples collected during capture sampling, biopsy darting or harvest monitoring.

Although recaptures and harvest recoveries of marked bears from BB and KB have been previously reported (Taylor and Lee 1995, Taylor et al. 2001, Peacock et al. 2012), we incorporated additional data to supplement and update these analyses. We focused on three areas. To facilitate interpretation of results from the recent genetic mark-recapture studies in BB and KB (Chapters 5 and 10) we examined recaptures and recoveries of individuals marked between 2011 and 2014 to test the assumption that bears marked during these studies remained within their original subpopulations over the sampling period. For bears marked in the 1990s in BB and KB, we examined the number and sex of individuals recovered in the harvest up to 2014 to assess the degree of movement amongst subpopulations over the long term and to test the hypothesis that these movements are sex biased. Finally, we examined 3 sequential, intensive mark-recapture sampling sessions in BB (1991-97 and 2011-13) and neighboring DS (2005-07), to assess movements across the BB-DS boundary.

We incorporated capture, recapture or recovery events for which the location of bears was recorded at time of observation using a GPS. Sex was determined by physical examination or genotyping (Chapter 5 and 11). For bears marked between 1991 and 2008, age was determined based on previous capture history, known age (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). For bears "marked" from 2009 to 2013, the age of most individuals (i.e., those remotely biopsied along eastern Baffin Island, in contrast to those immobilized and handled in NW Greenland and in Kane Basin) could not be determined since they were not physically handled and teeth were not available for aging. Instead, age class (cub-of-the-year, yearling, sub-adult [ages 2 – 4], and adult) was estimated from the air at a range of 3-7 meters above ground. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B). For all capture-recapture analyses, recaptures of an individual within the same season and year of capture were excluded. Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

2.3. Results

Overall Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all of Bylot Island (66.2°N to 73.8°N) in Nunavut/ Canada, as well as parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al. 2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south (Figure 1.1, Figure 2.1). Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A small number of bears remain on land in northwestern Greenland throughout the ice-retreat period (Born 1995, Born et al. 2011, this study).

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the influx of polar pack ice from Arctic Basin, and reaches a minimum in late summer.

Genetics

The multi-locus F_{ST} estimates were generally low, although statistically significant. The F_{ST} analysis suggested a separation into three groups (1) BB-KB, (2) LS and (3) DS (L.W. Andersen, Institute of Bioscience, Århus University, Denmark, personal communication). This subdivision is in accordance with Paetkau et al. (1999), Peacock et al. (2015) and Malenfant et al. (2016). The analyses based on the 8 markers showed that the genetic resolution was low. Bayesian clustering methods had difficulties identifying more closely related groups without using spatial information (i.e., site of sampling or harvest).

Using the spatial information (i.e., GPS positions of individual samples sites) implemented in GENELAND a group structure was indicated where adult females and males sampled during the winter-spring season were divided in an eastern and a western group corresponding to BB-KB and LS, and a northern and a southern group corresponding to BB-KB and DS (L.W. Andersen, Institute of Bioscience, Århus University, Denmark, personal communication) (Table 2.1).

General Movements from Telemetry

Baffin Bay – All but n=12 of the 43 bears collared in BB between 1991 and 1995 were captured on land on Baffin Island in fall. The remaining 12 were captured on the sea ice in spring (n=3 off Baffin Island and n=9 in Melville Bay, West Greenland). The 1990s collared bears transmitted through 1997 (Figure 2.3). Overall in the 1990s 72% were captured in fall on land inside the sampled area on BI (i.e., the area in which biopsies were collected for the genetic MR assessment), 6% were captured on the sea ice off the Baffin Island coast, and 21% were captured on the sea ice in Melville Bay, West Greenland. In the 2000s, all n=38 adult females (100%) were collared between 2009-2013 on the spring sea ice in West Greenland and transmitting through April 2015 (Figure 2.1).

Adult females were tracked between ~6 months and four years. Adult male tracking durations were shorter (~2 months) due to ear attachments. Telemetry data were truncated at 01

April 2015 so that analyses could be completed on time for reporting deadlines, therefore locations from collared bears beyond that date were not included here.

Adult females were captured and collared in West Greenland in all reproductive states (alone, as mating pairs, with COYs= cub of the year, yearlings, and 2-year old cubs) (Figure 2.2). General patterns of movements show broad coverage of the BB region during the tracking period in each year (Figures 2.4-2.11). There was a series of collar failures in 2012 that resulted from poor release mechanisms (Figure 2.7), therefore tracking data from bears captured in that year had shorter durations. There was some exchange between BB and KB. Most bears followed the general pattern of moving from West Greenland to the coast of Baffin Island in the fall. However a new pattern was observed that was not detected in the 1990s satellite telemetry data. Of the 38 adult females collared in BB in the 2000s, n=7 (18.4%) remained in the glacier ice of Melville Bay for the entire tracking durations. For all but one of these bears, where collar failure occurred early, this period was between 1 and 2 years (with some bears still transmitting from Melville Bay after April 2015). Bears that remained in Melville Bay were captured in all years when the area was sampled (2011-2013). No bears showed this behavior in the 1990s, and only one of the tracked bears made a single excursion onto the fast ice in Melville Bay (Taylor et al. 2001 figure 3, and Figure 2.3). Of note, this comparison may include some bias because only one bear was captured and given a satellite collar at glacier fronts in Melville Bay in the 1990s (Taylor et al. 2001 figure 1) even though the area was searched. Unfortunately the satellite collar on this bear only transmitted for one day. Additionally n=2 of 38 bears captured in the 2000s moved back and forth between Melville Bay and KB but never visited Baffin Island. Furthermore, another two individuals in the 2000s moved between Melville Bay and KB, but in subsequent years also moved to Baffin Island.

Kane Basin – In the 1990s, 12 adult females were captured and collared in KB along the coast of Ellesmere Island (Figure 2.11). In the 2000s, 20 adult females were captured and collared in spring along both Ellesmere Island and the western coast of Greenland (Figure 2.1). Bears captured in 2012 remained in KB subpopulation boundaries for the entire tracking period (through the period when collars were removed, April 2014) whereas with bears captured in 2013 there was more variability in movements, with individuals moving into Baffin Bay and Jones Sound. No bears collared in KB moved to Melville Bay though contact with two individuals was lost close to the West Greenland coast (Figures 2.12-2.14). One bear collared in KB in 2013 moved along the northern coast of Greenland in the Arctic Basin to Severnaya Zemlya and then to Franz Josef Land (Figure 2.14 inset). This individual was considered an outlier and excluded from further analyses. Adult males captured and satellite tagged with ear tags in KB (Figure 2.15) remained in KB during their tracking periods, which were less than 30 days.

Basis for Comparison of the 1990s and 2000s BB Satellite Telemetry Data

The analysis examined if polar bears collared on the sea ice in spring in the 2000s (in West Greenland) represented the same subpopulation sampled in fall on Baffin Island (Figure 2.1). This was conducted to ensure the comparison of movements and habitat use between the two telemetry sets was valid. Overall 92% of the bears collared in West Greenland in the 2000s entered the 1990s capture region on Baffin Island in fall, providing a solid basis for comparing the movements of polar bears captured in spring and in fall as defined by a polygon encompassing all the 1990s fall captures. Bears collared in West Greenland used nearly the entire Baffin Island coastline in fall and were spread over the whole capture region used in the

1990s, with the exception of the area around Cape Dyer. These bears thus also were representative of the bears biopsied for the genetic MR in the 2000s.

Subpopulation KDEs and Overlap of Ranges

Baffin Bay – In BB in the 1990s, 95% kernel ranges for polar bears were similar in winter, spring and summer, ranging from approximately 700,000 – 900,000 km^2. In the 2000s, seasonal ranges were significantly smaller in all seasons, ranging from 255,000 to 729,000 km^2. When the 1990s ranges were compared to the 2000s, there was a significant reduction in the size of 95% seasonal ranges in all seasons (reduction of 20% area in winter and 30% in spring), with the most marked reduction being a 60% decline in area of the summer range, reduced from a mean of 716,767 km^2 (SE 57,850) to a mean of 255,992 km^2 (SE 28,627), based on 1,000 bootstrap samples standardized for sample size (Table 2.4, Figure 2.16).

In all months except May, the home range sizes for the 2000s were significantly smaller than those in the 1990s (Figure 2.17). In some spring months 95% monthly ranges were reduced by 30% in late winter and spring (February, March and April) and in summer months by 50% (August and September), with a difference of about 325,000-375,000 km^2 (Table 2.4, Figure 2.18). Home range sizes were calculated with and without the resident Melville Bay bears to test for changes in results with inclusion of resident bears. There were minimal changes to the home range sizes and no differences in the significance of results with or without these residents.

Kane Basin – Seasonally, 95% ranges in KB in the 1990s fell between 89,000 and 203,000 km^2 whereas in the 2000s 95% ranges were between 152,000 and 192,000 km^2 (Table 2.4, Figures 2.19-2.21). In most months there was a reverse pattern to that in BB, where 95% range sizes in the 2000s were generally larger than those in the 1990s. However the pattern of increased range size was statistically significant only for June-September ($p<0.001$). Increases in summer range size were detected both when including all KB 2000s bears and those collared in west KB only.

Overlap of Ranges between Decades

The percent overlap quantifies the similarity in space between the home ranges across decades. In BB, the overlap of 1990s and 2000s home ranges was lowest in the summer months (July through October), ranging from 21-34% overlap. It was higher in spring and mid-winter, reaching a maximum of 61% in June, however largely was < 50% in all months. The low overlap values reflected the significant contraction of the range in most months in BB when comparing the two decades.

In KB there was a higher level of overlap in ranges between decades, ranging from about 50% in November and December to 98% in September. In general, KB bears used similar areas between decades, however in 2000s these areas were larger. When the 1990s KB bears were compared with the KB bears tagged in the western portion of the area (KB-West 2000s), the overlap was similar. In general, bears captured and tagged in KB west did not differ from the full sample of KB bears in the 2000s, though those in the western portion of KB were more likely to move into BB.

Changes in Median Latitude from Telemetry

We detected significant shifts north in the median subpopulation latitude in all seasons in BB. In winter, the shift was nearly 5 degrees north (median 68.8 in the 1990s, median 73.5 in the 2000s, $p<0.001$), while in the spring the shift north was about 3 degrees ($p<0.001$) and in summer < 1 degree (p<0.001) (Table 2.5, Figure 2.22). In KB there were also significant shifts north in median latitude in spring and summer but they were smaller, ~ 1 to 1.5 degrees (Table 2.5, Figure 2.23). There was no change in median latitude in KB during winter (p=0.07). We tested both all KB 2000s bears and those in KB-west and there were no differences in significance.

Movements across Subpopulation Boundaries from Telemetry

We examined the trajectories and departure from regions of origin BB (n=43 AFs in 1990s, n=38 AFs in 2000s) and KB (n=12 AFs in 1990s, n=20 AFs in 2000s) for bears tracked up to 700 days in duration. Of bears captured in BB in the 1990s when departure of any length (minimum 4 days due to duty cycling) was considered, there was movement to two subpopuations: Davis Strait (n=14 bears) and Lancaster Sound (n=12 bears). In the 2000s, bears moved to three subpopulations: Davis Strait (n=3), Lancaster Sound (n=3) and Kane Basin (n=5) (Table 2.6). Overall polar bears in the 2000s were significantly less likely to leave BB than in the 1990s (p<0.001) (Table 2.7), with large reductions in the number of bears moving into Davis Strait and Lancaster Sound as observed in the 1990s. For example at 100 days after capture, approximately 58% of bears were remaining in BB in the 1990s, whereas at the same time step in the 2000s, over 90% of the collared bears were still in the area. At 300 days after capture, approximately 60% of the bears had departed from BB in the 1990s where at the same time step in the 2000s about <10% had left. Capture season in BB in the 1990s (spring vs. summer) was not a factor in timing of departure from BB, there was no difference in time until departure for the two decades (p=0.562) for either length of departure.

In KB there were no significant differences in percentage of bears departing over the decades studied (p=0.339). In Kane Basin, 2 AF polar bears in the 1990s moved out of the area (one to Arctic Basin and one to Baffin Bay) (Table 2.6), and movement in the 2000s was slightly higher and included departure to Lancaster Sound, but was not significantly different (p=0.351).

The timing of departures over the annual cycle varied significantly in BB (Figure 2.24, Figure 2.25). In the 1990s, bears were significantly more likely to depart from BB to Davis Strait during the winter or early spring months when the area was ice covered (November and April). Bears that departed to Lancaster Sound left BB in late spring and summer (June-September) to move on to remnants of sea ice in the archipelago area (p=0.002, Fisher's exact test). This pattern was weakly present in the 2000s although sample sizes were very low.

The number of boundary crossings by individual bears is shown in Table 2.9. Of the bears that departed from BB (crossed the BB subpopulation border) in the 1990s when departures of all time steps were considered, 5 of 26 did not return to BB (19%) at any point during the tracking period. Of the bears that departed in the 2000s, 3 of 11 (27%) did not return to BB.

When the threshold for departure was longer (>30 days) patterns were similar. Bears were significantly more likely to depart from BB in the 1990s and departures primarily occurred to Davis Strait and Lancaster Sound (Tables 2.10 and 2.11). For AF departures based on the 30 day time step, fewer bears left the subpopulations of origin (Figure 2.25). There were similar differences in proportion of bears departing BB; significantly more bears departed in the 1990s vs. the 2000s (p=0.009). The timing of departures was similar but sample sizes were smaller (Table 2.12). Of the bears that departed from BB (crossed the BB border) in the 1990s when departures >30 days were considered, 56% of the 1990s BB bears did not depart from the BB boundaries, where as 79% of the BB bears in the 2000s did not depart from the BB boundaries (Table 2.13).

Recaptures and Recoveries during Genetic Mark-Recapture Studies, 2011-14

From 2011 to 2014, the harvest of polar bears in BB, KB and all surrounding subpopulations was monitored genetically by genotyping of tissue samples from harvested bears. Of the bears genetically marked in BB during the recent mark-recapture study, 2011-2013, 85 individuals were recovered in the harvest, as detected by genotyping (Table 2.14). Of these individuals, 84 (99%) were recovered in the BB harvest and the other bear was harvested in DS. For bears marked in KB from 2012-2014, no individuals were recovered in the harvest in KB or surrounding subpopulations. Also during this period of harvest monitoring, an additional 12 bears marked in the 1990s in BB and KB were recovered in the harvest. Eleven were marked in BB, 1 was marked in KB. All of these ‘old’ 1990s marks were recovered in BB.

Of the bears genetically marked in BB during 2011-2013, there were 207 recapture events during mark-recapture sampling in BB and KB from 2011 to 2014, including >1 recapture of some individuals (Table 2.15). Two hundred and six (> 99%) of these recapture events occurred in BB and one BB mark was recaptured in KB. For bears marked in KB during 2012-2014, there were 29 recapture events during mark-recapture sampling in BB and KB from 2011 to 2014. Twenty-eight (> 96%) of these recaptures occurred in KB and one KB mark was recaptured in BB.

During this recent period of mark-recapture sampling there were 66 recapture events of bears marked in either BB or KB during 1991-1997. Fifty-four of these events were of individuals marked in BB. Twelve were of KB individuals. For the 54 ‘old’ BB marks, all were recaptured in BB. Of the ‘old’ KB marks, 8 and 4 were recaptured in KB and BB, respectively.

For bears that were biopsied on more one occasion during the ice-free seasons in BB between 2011 and 2013 we examined the straight line distances between mark and recapture locations. Intervals between capture and recapture were obviously constrained to a ranged from 1-2 years. Median distances between mark-recapture varied between age and sex classes (Figure 2.26; Kruskal-Wallis, H = 18.62, $P = 0.001$). Bears initially marked as yearlings were recaptured at a location farther from their initial capture site than COY and adult females. Adult males also had larger mark-recapture distances than COY (Figures 2.26 and 2.27).

Long-term Harvest Recoveries of BB and KB Marks

Of the 881 bears marked in BB from 1991 to 1997, 181 individuals were recovered in the harvest in Canada and Greenland between 1991 and 2014 (Table 2.16). Eighty-three percent of recoveries occurred within BB. Recoveries of marked bears outside BB tended to be male-biased (3.29 males per female) relative to recoveries within BB (1.85 males per female), but this tendency was not statistically significant (Fisher’s Exact test, $P = 0.483$).

Of the 141 bears marked in KB from 1991 to 1997, 21 individuals were recovered in the harvest in Canada and Greenland between 1991 and 2014. Forty-eight percent of these recoveries were within KB. There was no significant sex bias in recoveries within versus outside KB (Fisher’s Exact test, $P = 0.656$).

Movement of Bears between BB and DS

One hundred and fifty-five bears marked in BB during 1991 – 1997 were recovered in the harvest up to 2005, including 11 recoveries in DS. From 2005 to 2007, a total of 2,128 bears comprising 1,518 unique individuals were captured in DS. There were 13 recapture events of bears previously marked in BB recaptured during this sampling in DS.

From 2009 to 2013, 1,253 unique individuals (1,623 capture events) were recorded in BB. Sixteen (1%) of these individuals were originally marked in DS. In sum, from a total of 2,771 bears marked in either BB or DS during 2005 – 2013, we detected 29 instances (ca. 1%) where marked bears moved from one subpopulation to the other.

The straight line distance of these 29 inter-subpopulation movements was independent of the capture-recapture interval which ranged from 4 to 15 years (Table 2.15, Figure 2.28). Seventy percent of these individuals were originally captured and marked within 100 km of the boundary between BB and DS (Table 2.16, Figure 2.29). In comparison to other bears marked in these subpopulations, individuals that made inter-subpopulation movements were found significantly closer to the boundary ($\chi^2 = 169.48$, d.f. = 11, $p < 0.001$). Despite extensive marking of bears throughout the range of both subpopulations the recorded inter-poplation movements were clustered near the boundary (Figure 2.1).

2.4. Discussion

Genetics

The results of the analyses of migration direction and detection of first generation migrants were subtle and influenced by the lack of differentiation between Baffin Bay and Kane Basin. However the overall migration direction appeared to flow from Lancaster Sound and

Davis Strait to Baffin Bay and Kane Basin. More markers would be needed to verify this (L.W. Andersen, Institute of Bioscience, Århus University, Denmark, personal communication).

Although the genetic resolution was low due to use of only 8 nuclear markers pre-selected for other purposes these analyses support findings in previous studies (Paetkau et al. 1999, Peaccok et al. 2015, Malenfant et al. 2016) that (1) Kane Basin and Baffin Bay polar bears are not genetically different, and that (2) polar bears from Kane Basin-Baffin Bay are genetically different from polar bears from Lancaster Sound and Davis Strait.

Movements and Telemetry

BB bears home ranges have become significantly smaller, by a third to a half between the 1990s and 2000s. There is <50% overlap between areas used by bears in the 90s and what they use in the 2000s. This is consistent even when resident bears that remained in Melville Bay glacier fronts were excluded (removing any possible bias in comparisons across decades). There is a significant shift in median latitude northward in the 2000s for the core subpopulation range when compared to the 1990s. Overall analyses also indicate that BB bears are significantly less likely to depart from the BB subpopulation boundaries in the 2000s than the 1990s.

During the fall, most bears in BB are distributed on land or on the remaining pack-ice along the coastline of Baffin Island and associated islands (Ferguson et al. 1998, 2001, Taylor et al. 2001). Information from polar bear hunters with extensive experience from the Melville Bay area (Born et al. 2011) and miscellaneous observations (e.g., Taylor et al. 2001) suggest that some bears may also be present along the NW Greenland coast during the open-water season, in particular in Melville Bay. However, in this study for the first time we documented resident bears in Melville Bay via satellite telemetry. These bears remained in NW Greenland year-round, in some cases >2 years within a series of glacier fronts. Satellite telemetry studies of bear movements in the 1990s failed to demonstrate this behavior (Taylor et al. 2001) however only one adult female bear was collared on the fast ice in Melville Bay (at a glacier front) and the satellite radio only transmitted for one day (E.W. Born, pers. comm.).

KB bears home ranges have become overall larger, in some cases significantly so and twice the size in summer. There is more overlap in home ranges for KB between the 90s and 00s, between about 50 and 98%, probably because the subpopulation area is smaller and because the ice loss has not been as extreme (or bears can use more of the southernmost habitat still). However, a likely explanation may be that the sea ice in the Kane Basin region has become less consolidated (e.g., Born et al. 2011) and now resembles that of the annual sea-ice ecoregion (rather than the archipelago region with year-round consolidated sea ice, cf Amstrup et al. 2008). This shift in habitat likely forces bears to have larger-scale movements and large home ranges, following well with that found by Ferguson et al. (2001). There were also significant shifts north in median latitude in KB in spring and summer from the 90s, though the variability has increased in the range of latitudes used and no change during winter.

Changes in Polar Bear Densities in Melville Bay, Northwest Greenland

Another result of the comparison between ranges and movements is the apparent increased use of Melville Bay, northwest Greenland by BB bears. The telemetry data collected in the 2000s indicate an increased use in both spring and summer (Figure 2.16) when compared to the 1990s (see also Figure 2.3). Of 43 polar bears that were tracked by use of satellite telemetry during the 1990s remarkably few made excursions from offshore BB pack ice onto landfast ice in NW Greenland (Figure 2.3, Taylor et al. 2001: Figure 3, p. 696; Born and Dietz 2009: Figure 18, p. 82). This is in contrast to heavy use of the area in the 2000s (Figure 2.9).

During the 1990s spring sampling 1992 and 1993, 55 hours of active searching was spent on the fast ice, glacier fronts, and active offshore pack ice between *ca*. 74° N and *ca*. 76° N in northwest Greenland, including offshore areas 100-150 km from the coast (i.e., ferry time excluded; Born unpublished data). In both years, there was little sign of bear activity on the fast ice in Melville Bay and along glacier fronts even though ~ 25% of the active search time was flown over fast ice and along glacier fronts. Only four of 36 bears were tagged in fast ice and glacier fronts the 1990s (Born et al. 1992, Rosing-Asvid 1993). In contrast, in 2011-2013, 85 active search hours were concentrated in the Melville Bay area north of the settlement Kullorsuaq. In each year there were signs of recent polar bear activity on the fast ice and along glacier fronts, including the consolidated pack ice in Melville Bay. Approximately 82% of captures occurred in fast ice or consolidated pack ice, and of those 25% were along glacier fronts. Differences between 1992-1993 and 2011-2013 in the allocation of sampling effort and number of polar bears captured reflect an increased density of polar bears using landfast ice and glacial fronts in Melville Bay in the 2010s. The apparent change in densities in spring time Melville Bay is worth noting and is consistent with information obtained from experienced polar bear hunters that there has been an increased occurrence of polar bears in nearshore areas in NW Greenland (Born et al. 2011).

Recoveries from Marked Bears

Use of tag recoveries or recaptures is a relatively coarse means of assessing subpopulation closure for the purpose of mark-recapture analyses, but it can facilitate the inclusion of data from large numbers of individuals, relative to satellite telemetry. The probability of detecting the movements of individuals between subpopulations depends on numerous factors, including the number of marks deployed, the intensity of sampling effort following marking, and the intensity of harvest. Additionally, the detection of movement amongst subpopulations does not provide a means of quantifying rates of permanent emigration or immigration. Nevertheless, data on recaptures and harvest recoveries of marked bears provide a supplemental line of evidence to support subpopulation delineations based on more detailed methods such as telemetry data analyses (Taylor et al. 2001) and genetics (Paetkau et al. 1999, Peacock et al. 2015).

Short-term Movements

During the recent genetic mark-recapture studies in BB and KB (2011-2014), we documented very low levels of recapture or harvest recovery of bears outside their subpopulation of origin. It should be noted that bears marked in the final year of these studies had a zero probability of recapture or recovery because harvest monitoring and biopsy darting were not extended beyond the last year of marking. However, bears marked in the first two years of these studies were available for recapture or recovery, subject to rates of natural mortality. The total number of bears marked in years 1 and 2 was equivalent to ~34% and ~25% of the estimated subpopulation size in BB and KB, respectively (Chapters 5 and 10). Despite marking a large proportion of the subpopulation, instances of emigration were ≤ 1% of the recaptures and recoveries of BB marks. Similarly in KB, documented cases of emigration comprised < 4% of recaptures. Amongst these findings, rates of harvest recovery provide a more complete picture of movement amongst subpopulations because harvest was monitored genetically in BB, KB and all surrounding subpopulations throughout the study period. In contrast, capture effort only took place in BB and KB during the study.

We acknowledge that this work comprised a relatively short window of time, especially since harvest monitoring and biopsy darting were not extended beyond the last year of marking. However, our findings suggest that the existing subpopulation boundaries continue to be relevant for harvest management purposes and subpopulation monitoring. Bears marked in BB or KB tended to remain within their respective subpopulations at least over the short term. These units can be surveyed by means of mark-recapture or aerial survey with a reasonable degree of confidence in the assumption of closure during short-term studies.

Long-Term Harvest Recoveries

Over the period 1979-2009, Peacock et al. (2012) found that amongst harvest recoveries of bears marked in BB, 82% were recovered in BB versus other subpopulations. Using a subset of the same data plus newer recoveries, we obtained a similar level of recovery (83%) within BB. These findings imply that bears exhibit a reasonably high degree of long-term fidelity to this geographically defined unit, which is consistent with estimates of site fidelity derived from mark-recapture analyses (Chapter 5, but note that the site fidelity parameter pertains to the study area and not necessarily the subpopulation). In contrast, less than half of the KB marks deployed in the 1990s have been recovered in the harvest in KB, although the vast majority of recoveries occurred during the 1990s and early 2000s. This suggests that fidelity to KB may be lower than observed in BB. However, we note that the harvest rate in KB is an order of magnitude smaller than surrounding subpopulations; mean annual harvests in KB and BB have been 9.3 and 163 bears, respectively, over the period 1992-2014 (Chapter 8 in this report). This difference in sampling effort (specifically, the small sample sizes in KB) may contribute to the observed differences in fidelity between the subpopulations.

Sex and Age Class

Sex and age differences in movements and distribution patterns are well documented for many mammal species (e.g., Mabry et al. 2013). Until recently, studies of polar bear movements and the delineation of subpopulations have relied primarily on data collected from satellite collared adult females (e.g., Taylor et al. 2001, Amstrup et al. 2004, Parks et al. 2006, Cherry et al. 2013). Even now, studies of the movements of sub-adults and adult male polar bears are limited by available technology. Satellite transmitters attached to sub-adult and adult males have only lasted 4-5 months, limiting research to studies of seasonal movements and habitat use (Amstrup et al. 2001, Laidre et al. 2013). The extent to which subpopulation boundaries as currently defined reflect the long-term distribution of sub-adults and male polar bears therefore remains uncertain. This is an important area requiring on-going investigation because of the implications for defining biological populations or harvest management units that can be accurately monitored by methods such as mark-recapture.

Using data on the distance between capture and recapture locations, Taylor et al. (2001) found a tendency for sub-adults to exhibit longer-range movements than adults in BB and KB. Similar data from our study (2011-2013) suggest that mark-recapture distances were greater for bears initially marked as yearlings, relative to adult females and COY. This may reflect a degree of dispersal amongst young bears away from their maternal (natal) range post-weaning. This is a common phenomenon among mammals and is often male biased (Greenwood 1980), but sample sizes for yearlings in our study were insufficient to test for sex effects in mark-recapture distances. However, using measures of genetic relatedness (kin structure analysis), Zeyl et al. (2009) demonstrated a slight male bias in effective dispersal of polar bears. Nevertheless, fidelity to natal range was relatively strong in that study, suggesting that regardless of sex, dispersal of offspring is unlikely to be a major factor in determining the broad scale (subpopulation) structure of polar bears. With harvest recoveries, the sex ratio of marked BB bears harvested outside BB tended to be male biased relative to recoveries within the subpopulation but not significantly so. Collectively, these findings provide some support for the notion that subpopulation boundaries, delineated using data on the movements of satellite collared adult females are relevant to polar bears of all sex and age classes as also suggested by Taylor et al. (2001).

Movement between Baffin Bay and Davis Strait

The boundary between the BB and DS subpopulations is not delineated by a landmass or area of open water that creates an obvious barrier to the movement of polar bears. During the winter and spring, this boundary is spanned by both land fast and pack ice that provides an easy platform for bears to move between these subpopulations (Stirling et al. 1980). Nevertheless, bears in these two subpopulations have consistently shown distinct differences in studies of genetics (Paetkau et al. 1999, Peacock et al. 2015), movements (Taylor et al. 2001) and diet (Thiemann et al. 2008), suggesting that there is a real boundary between them. This boundary is likely the result of ocean current patterns caused by a submarine ridge between SE Baffin Island and Central West Greenland and associated differences in patterns of sea-ice formation and break-up in Baffin Bay and Davis Strait, respectively (Taylor et al. 2001).

Since 1990, there have been 3 periods of intensive mark-recapture study in BB and DS. These data provided an opportunity to examine movements between DS and BB. Additionally, DS has likely undergone a substantial increase in abundance since the 1970s and presently has one of the highest densities of polar bears amongst subpopulations (Peacock et al. 2013). Furthermore it may be speculated that because of a higher rate of sea-ice loss in Davis Strait compared to Baffin Bay during the last decades (e.g., Laidre et al. 2015) an increased number of polar bears may have immigrated to BB from DS. We were therefore keen to examine the hypothesis that this apparent expansion happening concomitantly with sea-ice loss in Davis Strait had resulted in the movement of bears from DS to BB, thereby influencing our estimate of abundance for BB.

The intensity of marking effort (i.e., number of unique individuals marked) in these subpopulations was equivalent to 41% (BB 1991-97), 70% (DS 2005-07) and 44% (BB 2011-2013) of estimated abundance at the time of marking. Despite this extensive marking of bears throughout the seasonal ranges of both subpopulations we detected very few cases of movement between DS and BB. Approximately 1% of bears sampled in DS were recaptures of bears marked in BB and vice versa. Our results support the notion that the boundary between these two subpopulations remains relatively strong and does not support the hypothesis that subpopulation expansion and sea-ice loss in Davis Strait have resulted in a large-scale northward movement of DS bears into BB, at least during the period from 2005 to 2013.

Bears that were documented to have moved between BB and DS tended to be those originally captured close to the management unit boundary (< 100 km). This clustering of inter-subpopulation movements around the boundary does not itself demonstrate the significance of the boundary as a barrier to movements. Instead it may reflect the high degree of fidelity that

BB and DS bears show to their seasonal on-shore range. A similar degree of fidelity has been previously documented in BB (Taylor et al. 2001) and other subpopulations (Stirling et al. 2004). Taylor et al. (2001) examined distances between capture and recapture locations for bears marked in BB during the 1990s finding that 59% were recaptured within 100 km of their original capture location. Amongst all BB bears marked between 2011 and 2013, 50 % (n = 166) of recaptures were within 100 km of initial capture location (GN unpublished data) suggesting that the tendency of local fidelity of BB bears has remained unchanged since the 1990s. Strong inter-annual fidelity to terrestrial habitat is further supported by our finding that distances between capture and recapture locations for bears that moved between DS and BB were independent of capture intervals ranging from 4 to 15 years. Bears appear to exhibit fidelity over long periods.

In conclusion, using data from satellite telemetry, recapture and / or harvest recovery of marked bears and genetic we found no evidence to suggest a change in the delineation of the BB and KB subpopulations. The boundaries of these subpopulations appear to be relevant from the stand point of mark-recapture or other forms of periodic survey and for harvest management purposes.

2.5. Literature Cited

Amstrup, S. C., G. M. Durner, T. L. McDonald, D. M. Mulcahy, and G. W. Garner. 2001. Comparing movement patterns of satellite-tagged male and female polar bears. Canadian Journal of Zoology 79:2147–2158.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E.W., C. Sonne, and R. Dietz 2010. Research on polar bears in Greenland 2005-2009, Pages 135-148 *in* M. E. Obbard, G. W. Thiemann, E. Peacock and T. DeBruyn, editors. Proceedings of 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Derocher, A. E., and Ø. Wiig. 2002. Postnatal growth in body length and mass of polar bears (*Ursus maritimus*) at Svalbard. Journal of Zoology 256:343–349.

Escajeda, E. D. 2016. Identifying shifts in maternity den phenology and habitat characteristics of polar bears (*Ursus maritimus*) in Baffin Bay and Kane Basin. MSc Thesis, University of Washington, Seattle, Washington, USA.

Excoffier, L., and H. E. L. Lischer. 2010. Arlequin suite ver 3.5: a new series of programs to perform population genetics analyses under Linux and Windows. Molecular Ecology Resources 10:564–567.

Ferguson, S. H., M. K. Taylor, E. W. Born, and F. Messier. 1998. Fractals, sea-ice landscape, and spatial patterns of polar bears. Journal of Biogeography 25:1081–1092.

Ferguson, S. H., M. K. Taylor, E. W. Born, A. Rosing-Asvid, and F. Messier. 2001. Activity and movement patterns of polar bears inhabiting consolidated versus active pack ice. Arctic 54:49–54.

Fieberg, J., and C. O. Kochanny. 2005. Quantifying home-range overlap: the importance of the utilization distribution. Journal of Wildlife Management 69:1346–1359.

Freitas, C., K. M. Kovacs, C. Lydersen, and R. Ims. 2008. A novel method for quantifying habitat selection and predicting habitat use. Journal of Applied Ecology 45:1213–1220.

Goudet, J. 1995. FSTAT (Version 1.2): a computer program to calculate F statistics. Journal of Heredity 86:485–486.

Greenwood, P. J. 1980. Mating systems, philopatry and dispersal in birds and mammals. Animal Behaviour 28:1140–1162.

Guillot, G. 2008. Inference of structure in subdivided populations at low levels of genetic differentiation. The correlated allele frequencies model revisited. Bioinformatics 24:2222-2228.

Guillot, G., F. Mortier, and A. Estoup. 2005. Geneland: a computer package for landscape genetics. Molecular Ecology Notes 5:712–715.

Jombart, T. 2008. *adegenet*: a R package for the multivariate analysis of genetic markers. Bioinformatics 24:1403–1405.

Jombart, T., S. Devillard, and F. Balloux. 2010. Discriminant analysis of principal components: a new method for the analysis of genetically structured populations. BMC Genetics 11:94.

Kie, J. G., J. Matthiopoulos, J. Fieberg, R. A. Powell, F. Cagnacci, M. S. Mitchell, J.-M. Gaillard, and P. R. Moorcroft. 2010. The home-range concept: are traditional estimators still relevant with modern telemetry technology? Philosophical Transactions of the Royal Society of London, Series B, Biological Sciences 365:2221–31.

Laidre, K. L., E. W. Born, E. Gurarie, Ø. Wiig, R. Dietz, and H. Stern. 2013. Females roam while males patrol: divergence in breeding season movements of pack-ice polar bears (*Ursus maritimus*). Proceedings of the Royal Society B: Biological Sciences 280: 20122371. doi:10.1098/rspb.2012.2371.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Mabry, K. E., E. L. Shelley, K. E. Davis, D. T. Blumstein, and D. H. van Vuren. 2013. Social mating system and sex-biased dispersal in mammals and birds: a phylogenetic analysis. PLoS ONE 8(3): e57980. doi:10.1371/journal.pone.0057980.

Malenfant, R. M., C. S. Davis, C. I. Cullingham, and D. W. Coltman. 2016. Circumpolar genetic structure and recent gene flow of polar bears: A reanalysis. PLoS ONE 11(3): e0148967. doi:10.1371/journal.pone.0148967.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

Parks, E. K., A. E. Derocher, and N. J. Lunn. 2006. Seasonal and annual movement patterns of polar bears on the sea ice of Hudson Bay. Canadian Journal of Zoology 84:1281–1294.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1995. Summary of polar bear population status 1993. Pages 19–24 *in* Ø. Wiig, E. W. Born, and G. W. Garner, editors. Polar Bears: Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2002. Status of the polar bear. Pages 21–35 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T.

D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Peacock, E., S. A. Sonsthagen, M. E. Obbard, A. Boltunov, E. V. Regehr, N. Ovsyanikov, J. Aars, S. Atkinson, G. K. Sage, A. G. Hope, E. Zeyl, L. Bachmann, D. Ehrich, K. T. Scribner, S. C. Amstrup, S. Belikov, E. W. Born, A. E., Derocher, I. Stirling, M. K. Taylor, Ø. Wiig, D. Paetkau, and S. L. Talbot. 2015. Implications of the circumpolar genetic structure of polar bears for their conservation in a rapidly warming Arctic. PLoS ONE 10:e112021. doi:10.1371/journal.pone.0112021.

Piry, S., A. Alapetite, J. M. Cornuet, D. Paetkau, L. Baudouin, and A. Estoup. 2004. GENECLASS2: A software for genetic assignment and first-generation migrant detection. Journal of Heredity 95:536–539.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., N. J. Lunn, J. Iacozza, C. Elliott, and M. Obbard. 2004. Polar bear distribution and abundance on the southwestern Hudson Bay coast during open water season, in relation to population trends and annual ice patterns. Arctic 57:15–26.

Stirling, I., C. Spencer, and D. Andriashek. 1989. Immobilization of polar bears (*Ursus maritimus*) with Telazol® in the Canadian Arctic. Journal of Wildlife Diseases 25:159–168.

Taylor, M. K., and J. Lee. 1995. Distribution and abundance of Canadian polar bear populations: a management perspective. Arctic 48:147–154.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating

Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Thiemann, G. W., S. J. Iverson, and I. Stirling. 2008. Polar bear diets and arctic marine food webs: Insights from fatty acid analysis. Ecological Monographs 78:591–613.

Wand, M. P. 1994. Fast Computation of Multivariate Kernel Estimators. Journal of Computational and Graphical Statistics 3:433–445.

Wand, M. P., and M. C. Jones. 1995. Kernel Smoothing. Chapman and Hall, London.

Wilson, G. A., and B. Rannala. 2003. Bayesian inference of recent migration rates using multilocus genotypes. Genetics 163:1177–1191.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164–168.

Zeyl, E., J. Aars, D. Ehrich, L. Bachmann, and Ø. Wiig. 2009. The mating system of polar bears: a genetic approach. Canadian Journal of Zoology 87:1195–1209.

Table 2.1. Polar bear samples (n = 1,364; sampling period: 2009-2014) from Baffin Bay (BB), Kane Basin (KB), Lancaster Sound (LS) and Davis Strait (DS) that were included in the population genetic analyses. The total sample was subdivided according to season, age category and sex for the analyses. A subset of 402 samples collected during winter and spring (2012-2014) represented all four subpopulations.

	BB	**KB**	**LS**	**DS**	**N**
	Biopsies and Harvest	**Biopsies**	**Harvest**	**Harvest**	**Total**
Period of sampling	2009-2014	2012-2014	2011-2013	2012-2013	
Total Sample	1051	99	142	72	1364
Winter-spring (WS)[1]	140	99	114	49	402
Winter-spring-adults (WSA)[2]	109	78	84	37	308
Winter-spring-subadults (WSS)[2]	31	21	30	12	94
Winter-spring-adults-females (WSAF)[3]	54	54	15	11	134
Winter-spring-adult-males (WSAM)[3]	55	24	69	26	174

[1] Winter defined as: November-February and spring defined as March-June

[2] A=adults and S=subadults

[3] F=females and M=males

Table 2.2. Sample sizes (number of individuals) polar bears captured and tagged with collars or ear tags in the 1990s and 2000s in BB and KB. In total 134 bears were tagged in BB and 46 bears were tagged in KB over two decades.

		AF	**AM**	**SF**	**SM**	**2YR**	**TOTAL**
1990s	BB	43					43
	KB	12					12
2000s	BB	38	30	4	6	13	91
	KB	20	9		5		34

Table 2.3. Breakdown of adult females (AF) collared in the 1990s and 2000s. Total n=113 AF bears over both decades and subpopulations.

		AF alone	AF+AM	AF+COY	AF+YRL	AF+2YR	Sum
1990s	BB	9		19	13	2	43
	KB	3		5	3	1	12
2000s	BB	10	2	6	12	8	38
	KB	5	1	7	3	4	20

Table 2.4. 95% mean kernel range sizes (in sq km) estimated from a bootstrap method for each of 38 bears in BB and 12 bears in KB by decade, month and season. The bootstrap was used to generate kernel range sizes based on equal sample sizes between decades (see Methods). Data are reported with bootstrapped SE of the mean in parentheses. Fraction of overlap is the 95% kernel probability area from the 1990s overlapped by the same in the 2000s. KB_West is reported only for bears tagged in western KB for direct comparison to the 1990s (where no bears were tagged in East KB). Data here include all bears in BB, including the resident bears in Melville Bay, though we also investigated bootstrap range size values without resident bears and significance remained.

	BB				**KB_All Bears**				**KB_West**	
	1990s mean (SE)	**2000s**	**P value comparing 95% area between decades**	**Overlap**	**1990s**	**2000s**	**P value comparing 95% area between decades**	**Overlap**	**2000s**	**Overlap (with KB ALL 1990s)**
January	684,409	558,957		0.40	86,556	163,892		0.77	175,730	0.77
	(60,692)	(56,594)	<0.001		(6,890)	(27,619)	0.007		(23,649)	
February	707,387	513,732		0.39	105,788	171,441		0.81	183,981	0.81
	(55,079)	(40,662)	0.005		(5,426)	(34,036)	0.057		(23,205)	
March	852,935	580,767		0.45	136,942	205,921		0.70	232,299	0.73
	(50,240)	(36,287)	<0.001		(15,859)	(49,119)	0.180		(38,259)	
April	795,859	506,739		0.46	131,963	183,184		0.73	180,913	0.69
	(45,652)	(26,529)	<0.001		(15,331)	(38,786)	0.219		(32,718)	
May	564,658	473,825		0.58	130,730	122,598		0.68	115,925	0.69
	(37,090)	(35,679)	0.078		(19,002)	(15,355)	0.741		(14,792)	
June	521,410	430,766		0.61	68,696	124,227		0.88	103,783	0.85
	(32,633)	(31,829)	0.047		(11,854)	(13,578)	<0.001		(9,942)	

July	536,992	376,891		0.52	54,681	130,518		0.91	112,655	0.88
	(32,000)	(30,062)	<0.001		(7,986)	(12,414)	<0.001		(13,872)	
August	596,411	216,881		0.21	68,649	119,251		0.90	106,314	0.92
	(44,692)	(18,958)	<0.001		(12,391)	(13,419)	<0.001		(11,694)	
September	551,395	226,427		0.26	74,368	131,558		0.98	109,697	0.84
	(48,280)	(21,538)	<0.001		(13,130)	(15,871)	<0.001		(14,187)	
October	459,230	276,198		0.34	99,855	153,820		0.91	132,177	0.75
	(43,932)	(29,264)	<0.001		(15,807)	(25,424)	0.072		(23,717)	
November	594,280	474,604		0.58	156,120	172,068		0.54	166,048	0.53
	(33,388)	(25,867)	<0.001		(22,728)	(23,943)	0.631		(20,773)	
December	702,091	524,787		0.52	150,392	143,969		0.54	137,782	0.53
	(35,173)	(38,123)	<0.001		(20,855)	(20,739)	0.826		(19,024)	
Winter	906,657	729,022		0.65	203,858	192,619		0.56	210,364	0.64
	(55,609)	(44,240)	0.012		(37,301)	(34,357)	0.826		(26,680)	
Spring	837,036	585,659		0.57	137,563	177,495		0.80	189,301	0.82
	(58,976)	(33,379)	<0.001		(17,600)	(37,516)	0.337		(37,470)	
Summer	716,676	255,992		0.24	89,066	152,747		0.97	141,118	0.94
	(57,850)	(28,627)	<0.001		(14,251)	(21,784)	0.014		(20,697)	

Table 2.5. Box plot statistics for median latitude (and interquartile range) for each season, subpopulation, and decade of polar bears tracked by use of satellite telemetry in BB and KB during the 1990s and 2000s. See Table 2.3 for sample sizes. In the case of BB 2000s, the resident bears in Melville Bay glacial fronts were not included

	Q1	Median	Q2	t statistic	df	p value
Winter						
BB 90s	66.90	68.88	72.13	-17.34	1683.4	p<0.001
BB 00s	69.76	72.01	74.90			
KB 90s	77.28	78.62	79.17	-1.86	173.54	p=0.07
KB 00s all	77.03	78.96	79.41			
Spring						
BB 90s	67.99	70.87	73.51	-18.18	1615.99	p<0.001
BB 00s	70.88	72.90	74.48			
KB 90s	77.63	77.91	79.13	-4.06	617.76	p<0.001
KB 00s all	77.25	79.06	79.42			
Summer						
BB 90s	67.26	70.29	72.96	-4.20	1758.07	p<0.001
BB 00s	69.51	70.68	71.60			
KB 90s	77.77	77.89	78.50	-5.47	840.41	p<0.001
KB 00s all	77.27	78.89	79.43			

Table 2.6. Movements of all polar bears (AF + AM) captured and tracked in the 1990s and 2000s from BB and KB. Movement to another subpopulation is enumerated if the bear departed for any length of time (4 days or greater).

			Movement to other subpopulation during tracking period				
Time period	**Subpopulation of origin**	**n**	**Arctic Basin**	**Baffin Bay**	**Davis Strait**	**Kane Basin**	**Lancaster Sound**
1990s	BB	43	0	0	14	0	12
2000s	BB	91	0	0	3	5	3
1990s	KB	12	1	1	0	0	0
2000s	KB	34	2	2	0	0	3

Table 2.7. Summary of observed and expected departures from region of origin (capture site) to any other subpopulation region in the 1990s and 2000s for AF polar bears for departures on all time steps. Log rank rest of equality conducted on each subpopulation comparing decades.

Subpopulation Decade	**n**	**Observed departures**	**Expected departures**	**(O-E)^2/E**	**(O-E)^2/V**
BB 1990s	43	26	15.7	6.72	12.1
BB 2000s	38	11	21.3	4.96	12.1
KB 1990s	12	2	3.29	0.509	0.871
KB 2000s	20	6	4.71	0.356	0.871

BB: $\chi^2 = 12.1$, df = 1, p = 0.000515

KB: $\chi^2 = 0.9$, df = 1, p = 0.351

Table 2.8. Timing of departures from Baffin Bay in the 1990s and 2000s for radio-collared adult female polar bears for departures of all time steps. Data are reported as number of AF bears departing to another subpopulation for the 1990s (with number of AF bears in 2000s in parentheses).

Month of departure from BB	Davis Strait	Lancaster Sound	Kane Basin
1	1 (0)	1 (0)	0 (3)
2	0 (0)	0 (0)	0 (0)
3	0 (0)	0 (0)	0 (0)
4	1 (1)	0 (0)	0 (0)
5	0 (0)	1 (1)	0 (0)
6	0 (0)	2 (1)	0 (0)
7	1 (0)	2 (0)	0 (1)
8	0 (0)	1 (0)	0 (0)
9	0 (0)	3 (0)	0 (0)
10	0 (0)	2 (1)	0 (0)
11	7 (0)	0 (0)	0 (0)
12	4 (2)	0 (0)	0 (1)

Table 2.9. Number of subpopulation boundary crossings made by individual AF bears in each subpopulation and decade for departures of all time steps.

	Number of subpopulation boundary crossings by individual AFs												
	0	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
BB 90s	17	3	7	1	3	1	3	0	3	3	1	0	1
BB 00s	27	3	4	0	2	0	0	1	1	0	0	0	0
KB 90s	10	0	1	0	0	0	0	0	0	1	0	0	0
KB 00s	14	1	2	0	1	1	0	0	1	0	0	0	0

Table 2.10. Movements of AF polar bears captured and tracked in the 1990s and 2000s from BB and KB. Movement to another subpopulation is enumerated if the bear departed for of >30 days or more.

			Movement to other subpopulation during tracking period				
Time period	**Subpopulation of origin**	**n**	**Arctic Basin**	**Baffin Bay**	**Davis Strait**	**Kane Basin**	**Lancaster Sound**
1990s	BB	43	0	0	14	0	12
2000s	BB	38	0	0	3	5	3
1990s	KB	12	1	1	0	0	0
2000s	KB	20	2	1	0	0	3

Table 2.11. Summary of observed and expected departures from region of origin (capture site) to any other subpopulation region in the 1990s and 2000s for radio-collared adult female polar bears for departures of >30 days or more. Log rank rest of equality conduced on each subpopulation comparing decades.

Subpopulation Decade	N	Observed departures	Expected departures	(O-E)^2/E	(O-E)^2/V
BB 1990s	43	19	12.4	3.580	6.760
BB 2000s	38	8	14.6	3.020	6.760
KB 1990s	12	1	2.06	0.532	0.914
KB 2000s	29	4	2.96	0.367	0.914

BB: $\chi^2 = 6.8$, df = 1, p = 0.009

KB: $\chi^2 = 0.9$, df = 1, p = 0.339

Table 2.12. Timing of departures from Baffin Bay in the 1990s and 2000s for radio-collared AF polar bears for departures of >30 days or more. Data are reported as number of AF bears departing to another subpopulation for the 1990s (number of AF bears departing in 2000s in parentheses).

Month of departure from BB	Davis Strait	Lancaster Sound	Kane Basin
1	1 (0)	1 (0)	0 (3)
2	0 (0)	0 (0)	0 (0)
3	0 (0)	0 (0)	0 (0)
4	1 (1)	0 (0)	0 (0)
5	0 (0)	1 (1)	0 (0)
6	0 (0)	2 (1)	0 (0)
7	1 (0)	2 (0)	0 (1)
8	0 (0)	1 (0)	0 (0)
9	0 (0)	3 (0)	0 (0)
10	0 (0)	2 (1)	0 (0)
11	7 (0)	0 (0)	0 (0)
12	4 (2)	0 (0)	0 (1)

Table 2.13. Number of subpopulation boundary crossings made by individual radio-collared adult female bears in each subpopulation and decade for departures of >30 days or more. 0 denotes the number of bears that never cross a boundary, so n=24 bears in BB 90s never departed from BB. Percentages shown as percent of total tagged bears.

	Number of subpopulation boundary crossings by individual AFs							
	0	**1**	**2**	**3**	**4**	**5**	**6**	**7**
BB 90s	24 (56%)	8 (19%)	6 (14%)	1 (2%)	1 (2%)	1 (2%)	1 (2%)	1 (2%)
BB 00s	30 (79%)	3 (8%)	2 (5%)	0	1 (3%)	1(3%)	0	1(3%)
KB 90s	11 (92%)	0	0	0	0	0	0	1 (8%)
KB 00s	16 (80%)	1 (5%)	0	1 (5%)	1(5%)	0	1(5%)	0

Table 2.14. Number and subpopulation location of harvest recoveries of bears marked genetically between 2011 and 2014. Data presented as number of individuals.

Subpopulation	**Subpopulation Recovered**[1]						
Marked	**BB**	**KB**	**LS**	**FB**	**DS**	**NW**	**GB**
BB	84	0	0	0	1	0	0
KB	0	0	0	0	0	0	0

[1] BB, Baffin Bay; KB, Kane Basin; LS, Lancaster Sound; FB, Foxe Basin; DS, Davis Strait; NW, Norwegian Bay; GB, Gulf of Boothia

Table 2.15. Recaptures of bears marked genetically between 2011 and 2014 in BB and KB. Data presented as number of recapture events. Some individuals were recaptured more than once. Excludes multiple recaptures of same individual within a season. Also includes COY that were not initially sampled but later seen as yearlings with mother and sampled.

Subpopulation Marked[1]	**Subpopulation Recaptured**	
	BB	**KB**
BB	206	1
KB	1	28

[1] BB, Baffin Bay; KB, Kane Basin

Table 2.16. Recoveries of polar bears tagged in Baffin Bay (1990-1997) in the harvest in Canada and Greenland, 1990 to 2014.

	Recovered in Harvest (1990-2014)		
Sex of Bear	**In Baffin Bay**	**Outside Baffin Bay**	**Total**
Female	53	7	60
Male	98	23	121

Figure 2.1. Distribution of capture locations of polar bears with satellite collars in BB and KB during the 1990s and the 2000s, respectively. See Table 2.2 for sample sizes. Note in 1990s bears in BB were mainly captured on Baffin Island in fall (12 were captured on sea ice in spring; 3 along Baffin Island and 9 in NW Greenland) whereas during the 2000s all bears were captured and tagged with satellite transmitters on the sea ice in NW Greenland in spring.

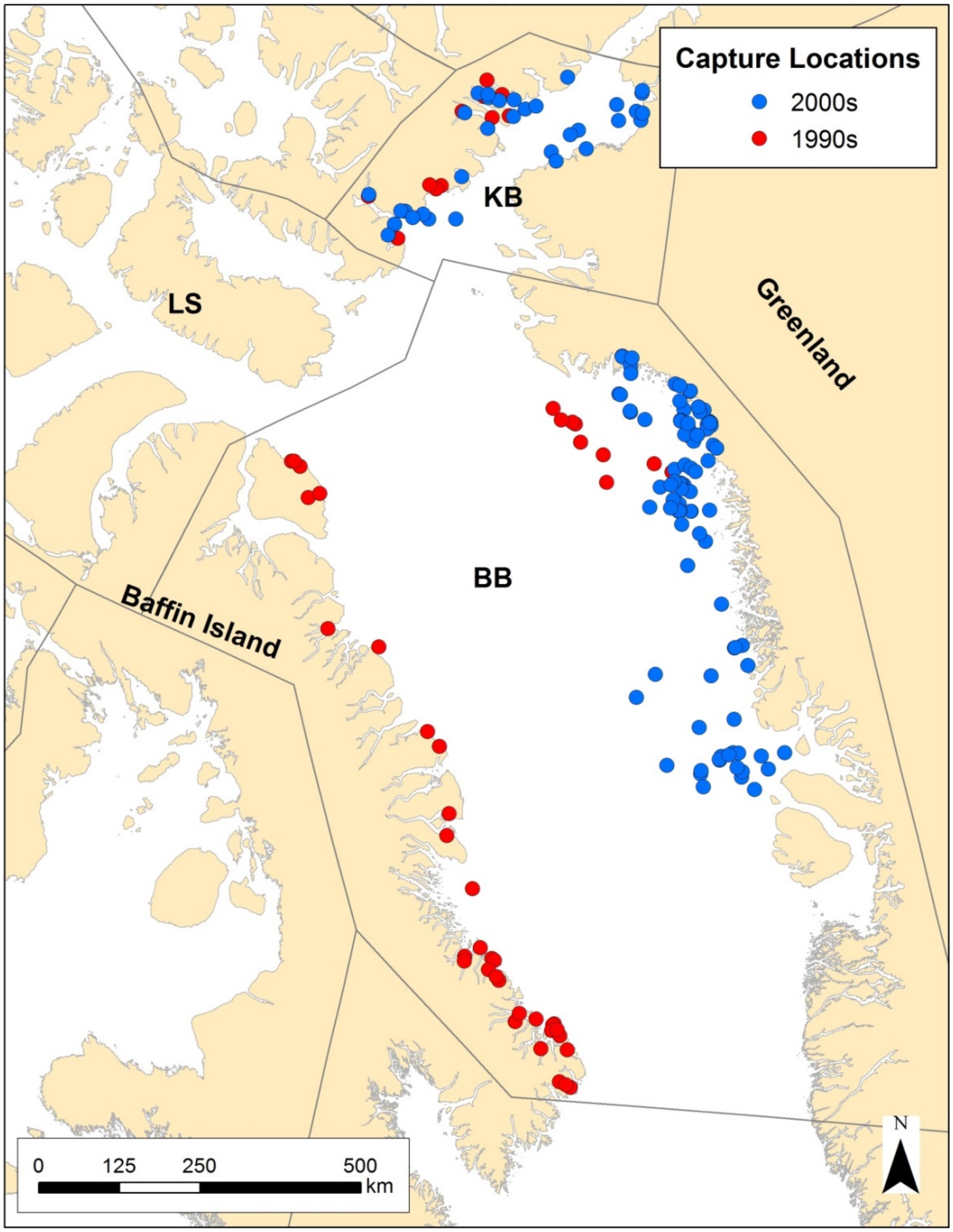


Figure 2.2. Distribution of ages and family groups of a total of 139 individual polar bears captured in spring in northwest Greenland, 2009-2013.

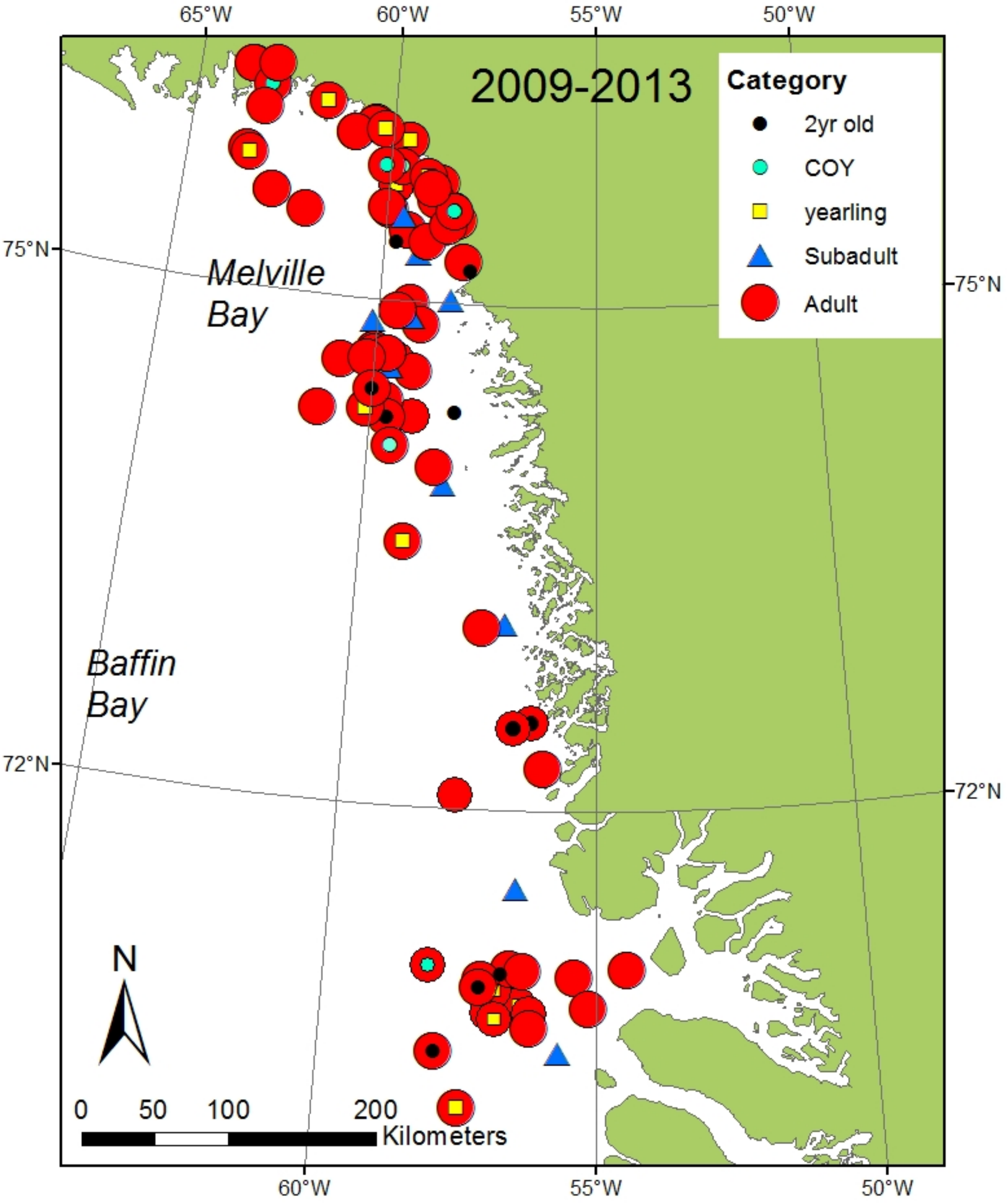


Figure 2.3. Tracklines from n=43 adult female polar bears satellite collared in the 1990s in BB.

Note the general absence of tracks on the fast ice in West and Northwest Greenland.

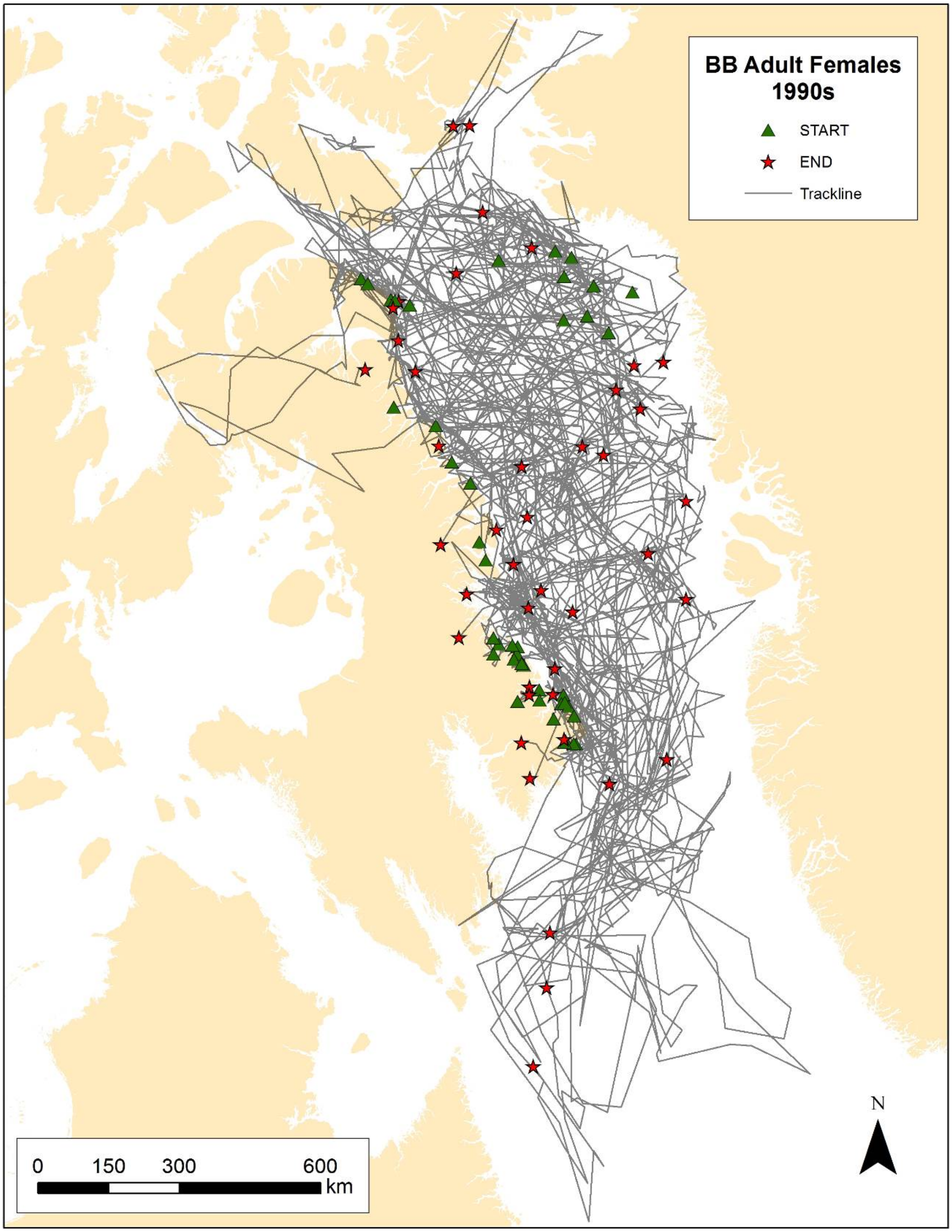


Figure 2.4. Tracklines from n=5 adult female polar bears satellite collared in 2009 in BB.

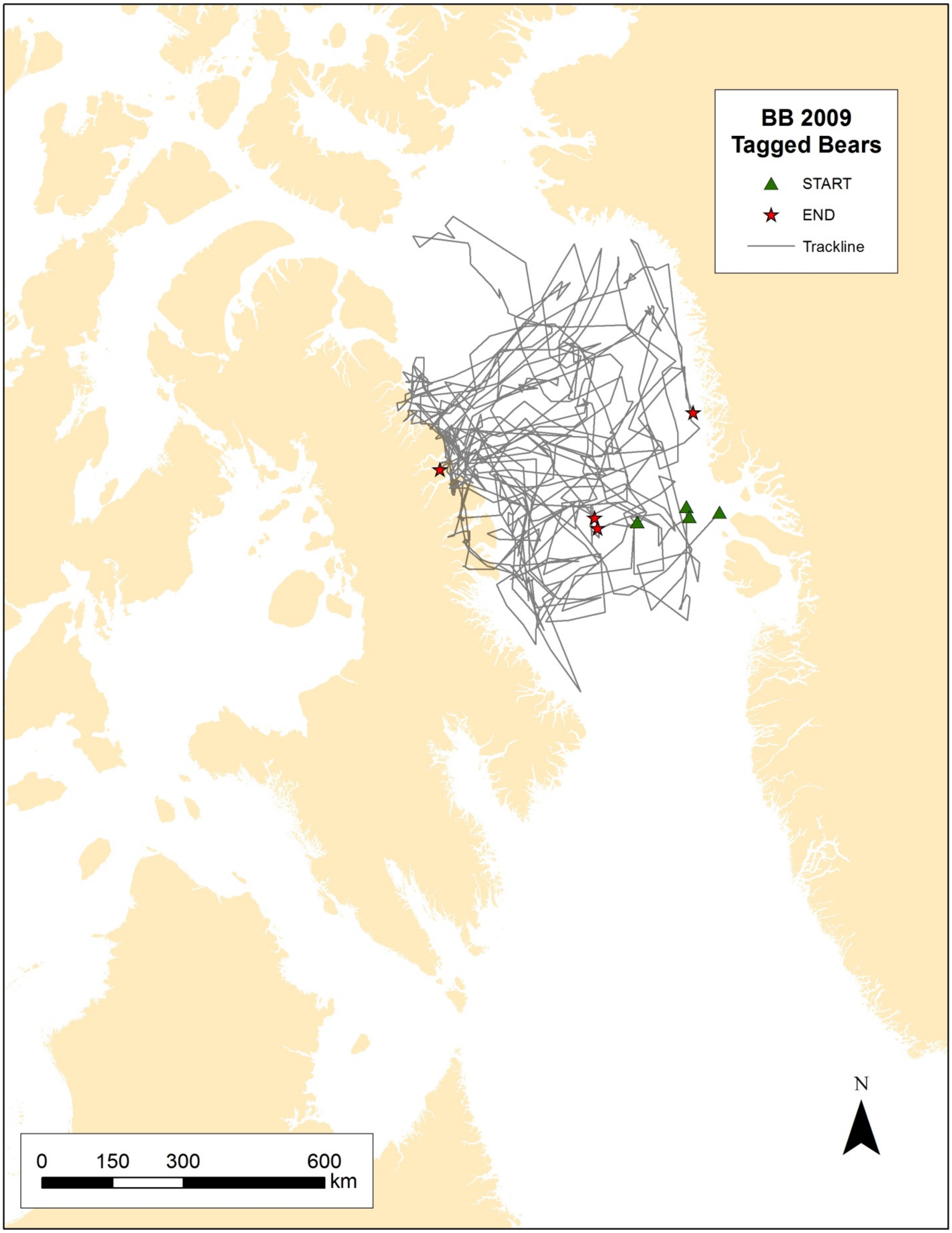


Figure 2.5. Tracklines from n=9 adult female polar bears satellite collared in 2010 in BB.

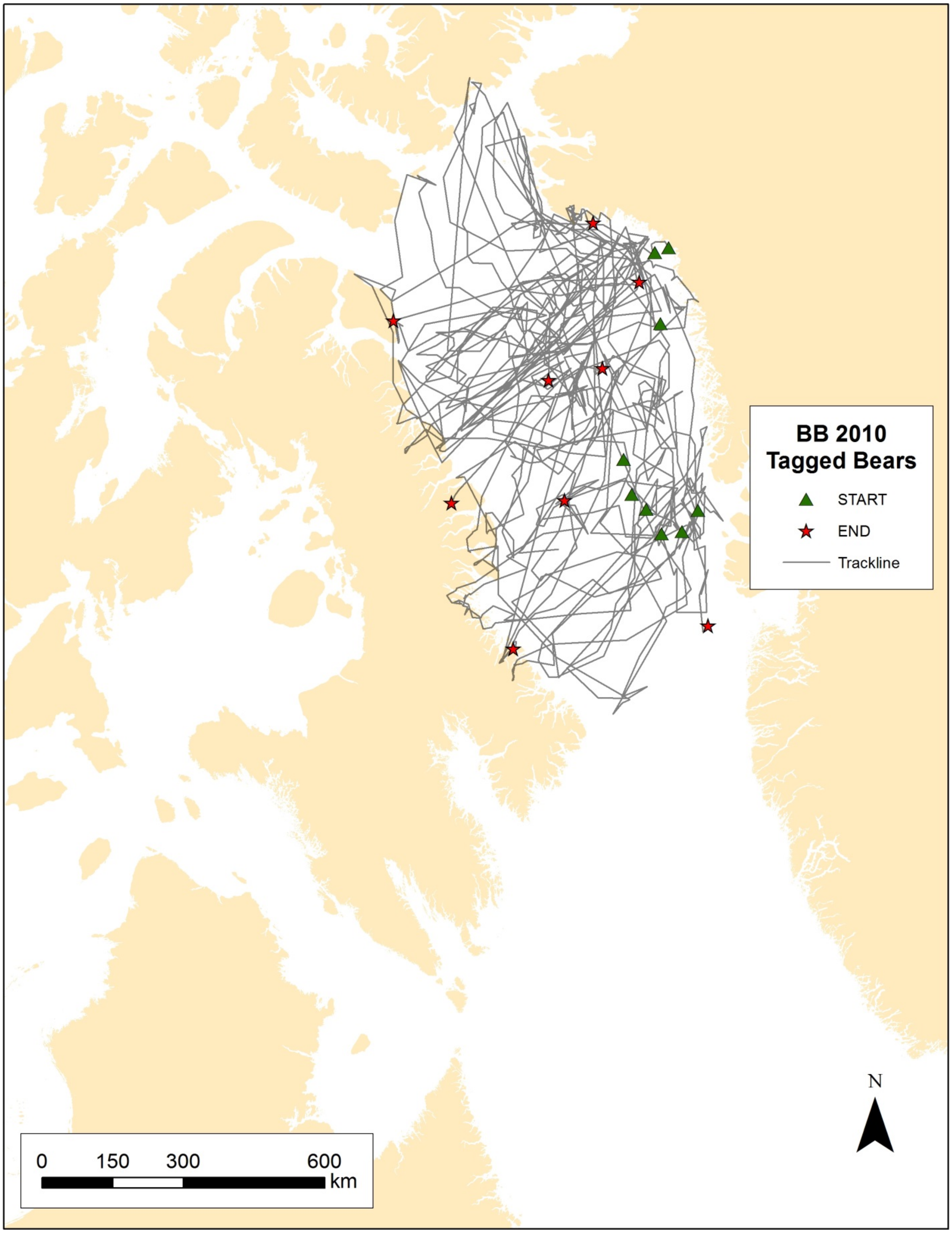


Figure 2.6. Tracklines from n=12 adult female polar bears satellite collared in 2011 in BB.

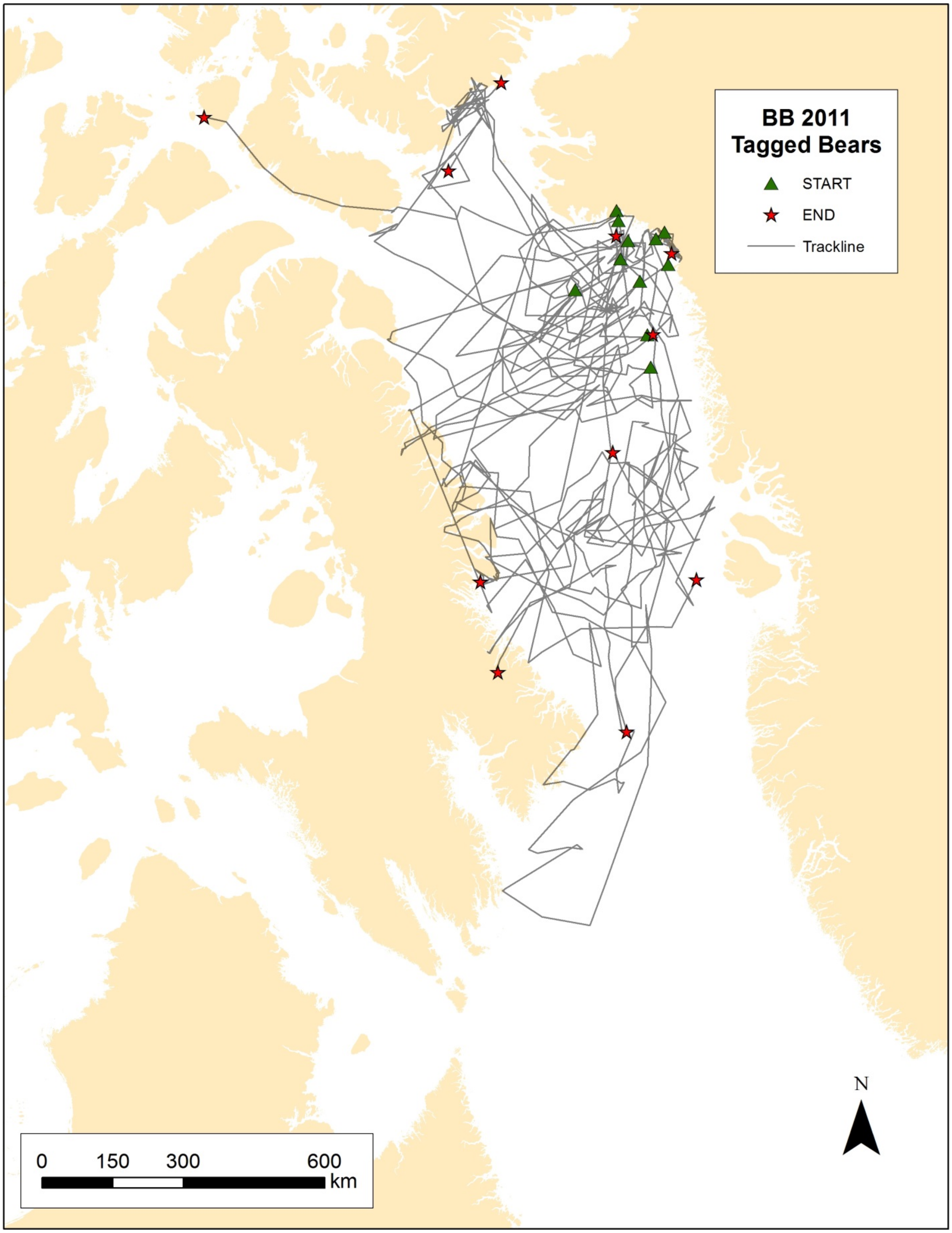


Figure 2.7. Tracklines from n=11 adult female polar bears satellite collared in 2012 in BB.

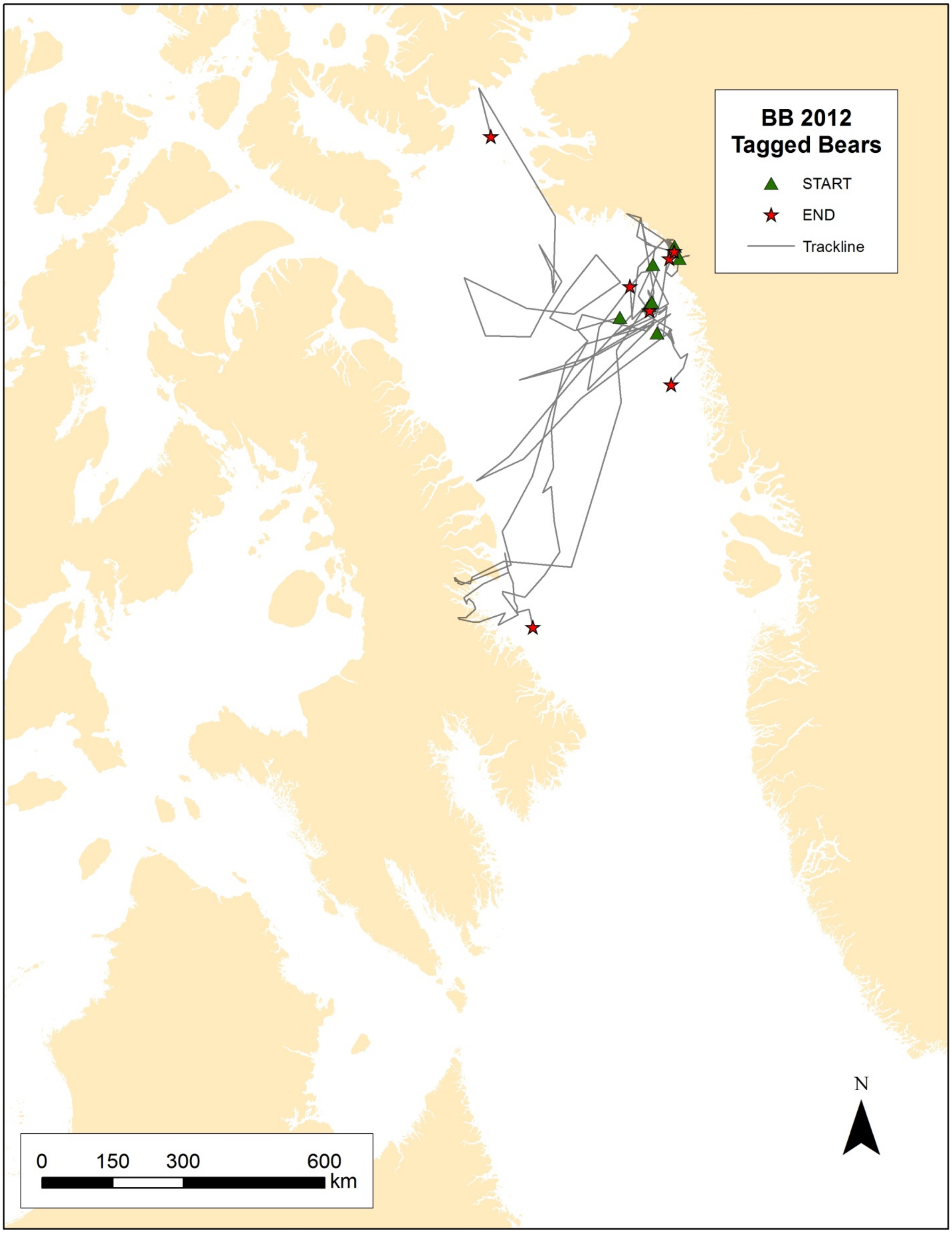


Figure 2.8. Tracklines from n=9 adult female polar bears satellite collared in 2013 in BB.

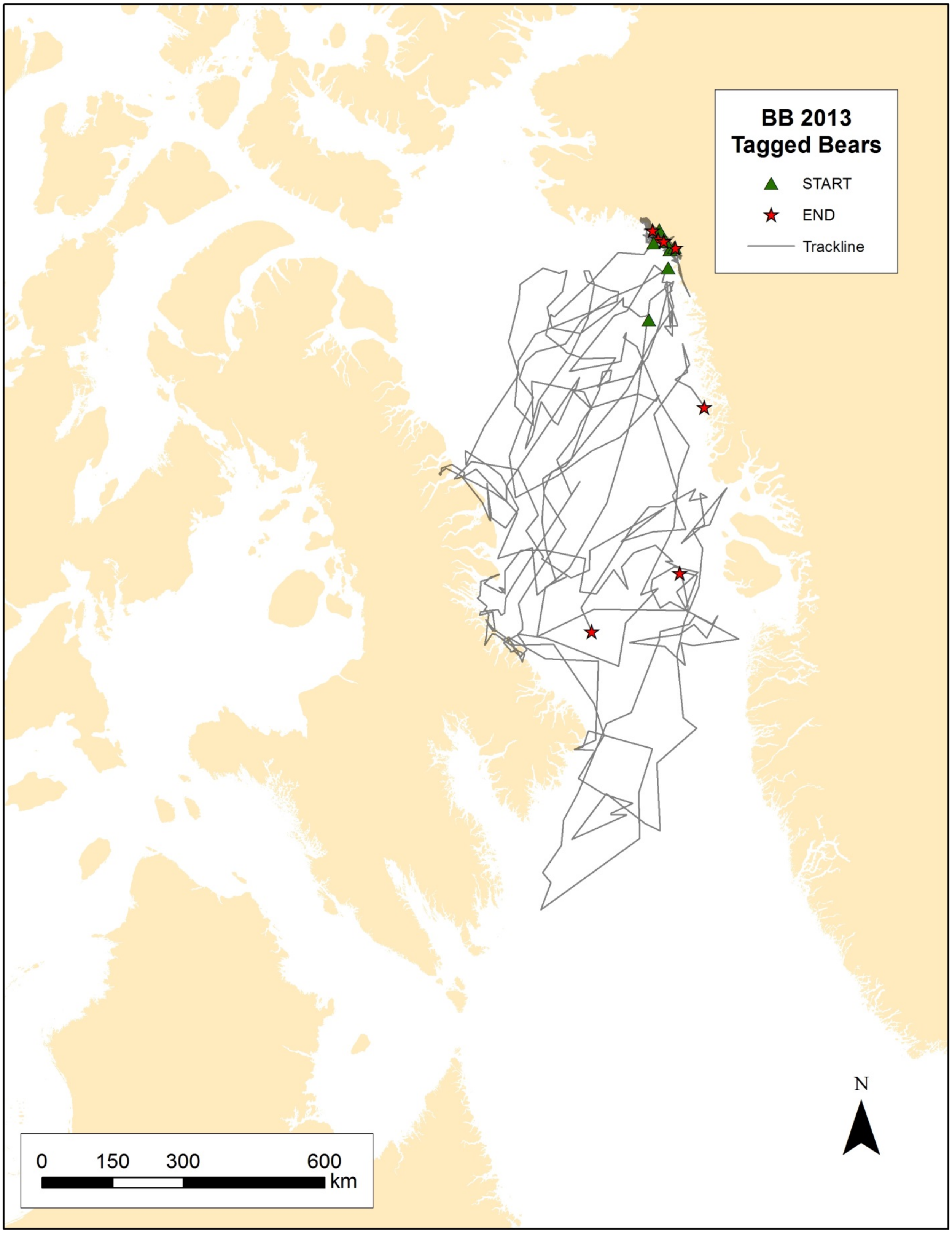


Figure 2.9. Tracklines from all adult female bears (n=38) collared between 2009 and 2013 shown together through April 2015, excluding bears where collars failed after a few days.

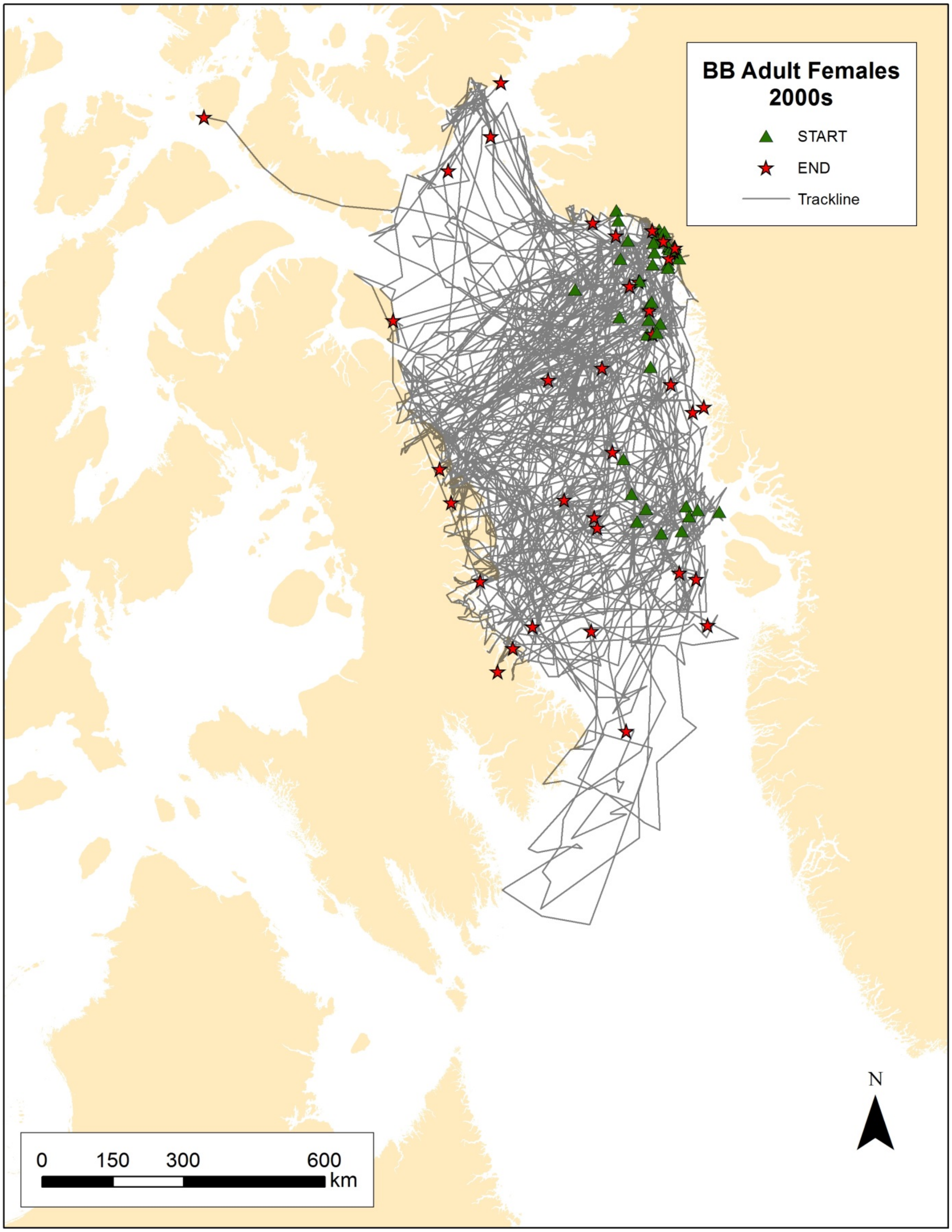


Figure 2.10. Tracklines from n=32 adult male bears tagged with ear transmitters between 2009-2013.

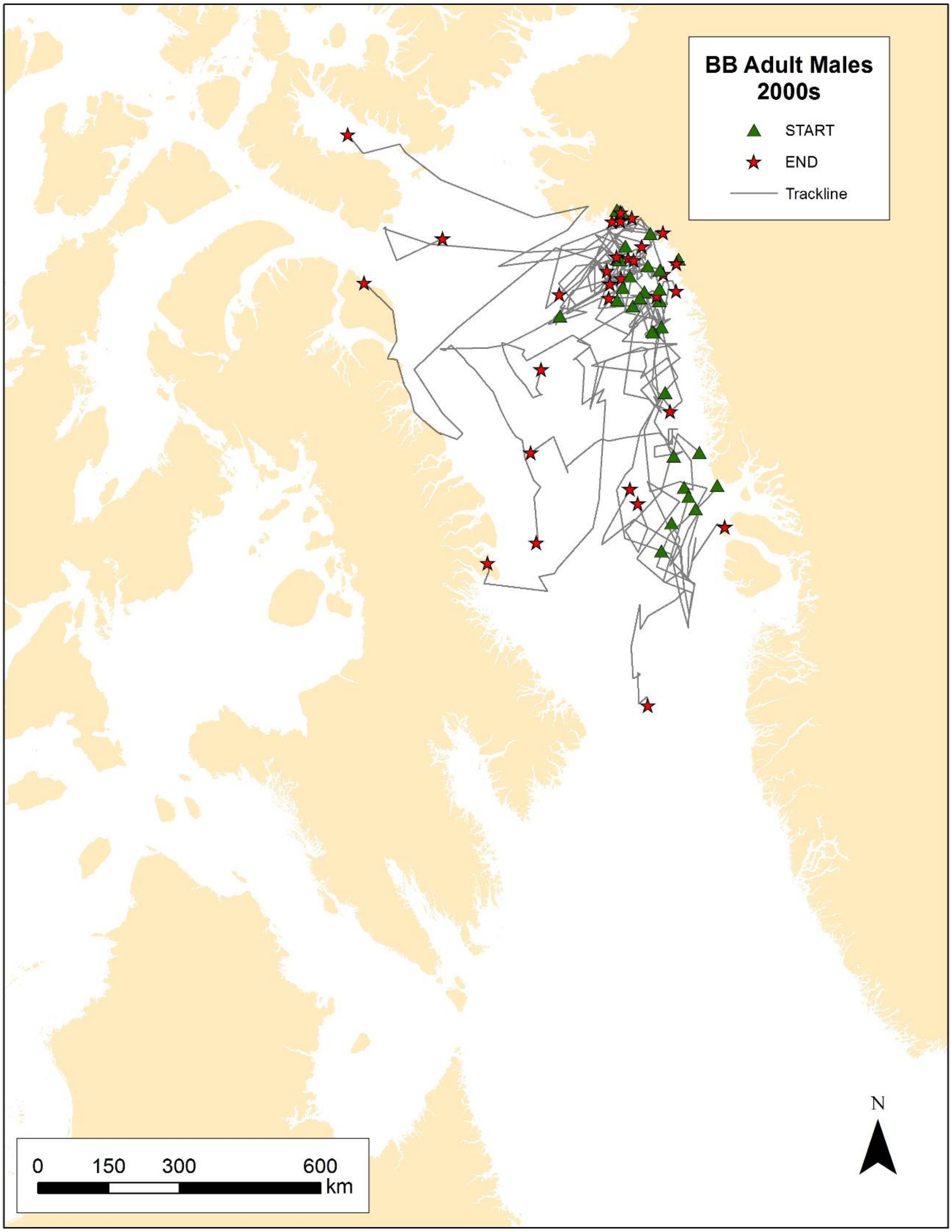


Figure 2.11. Tracklines from n=12 adult female polar bears satellite collared in the 1990s in KB.

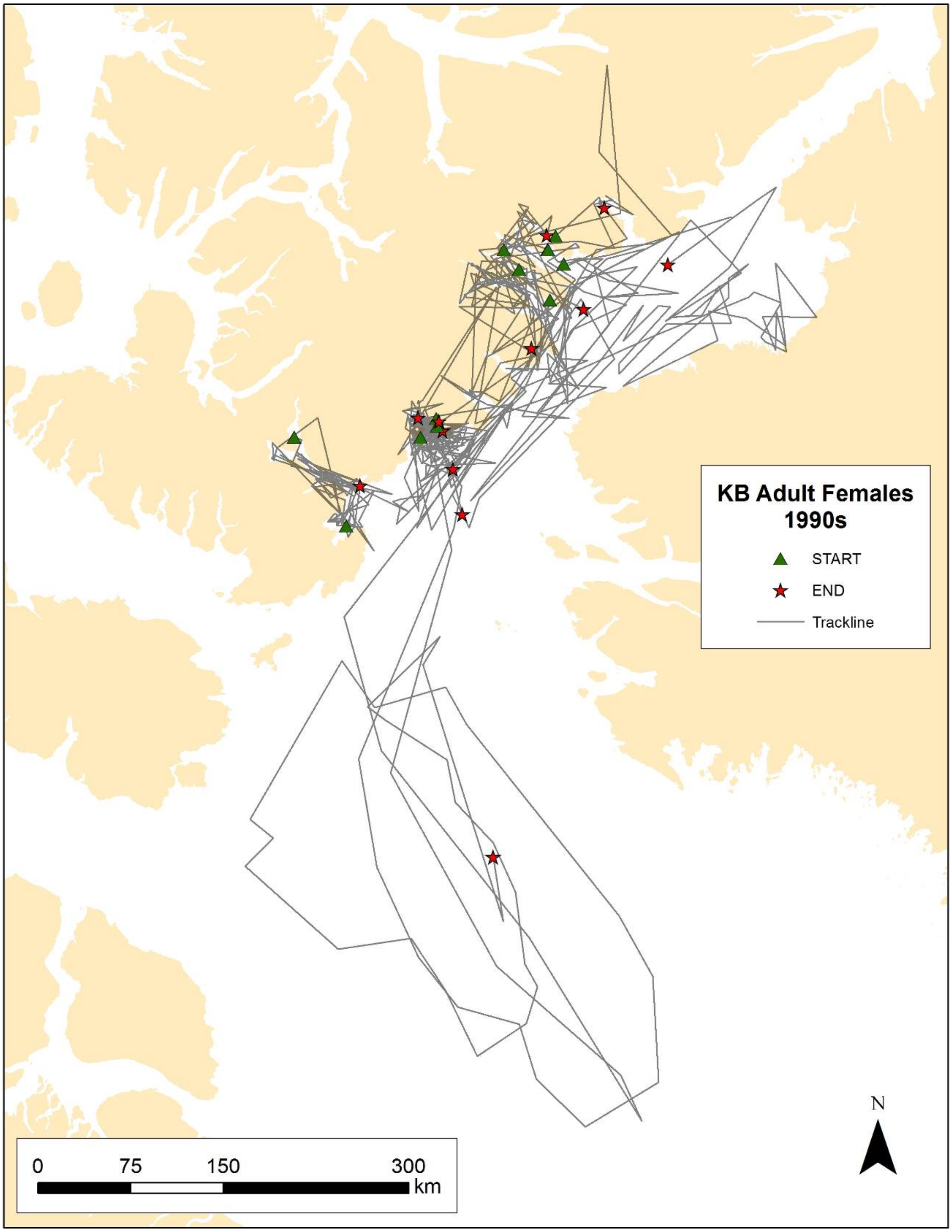


Figure 2.12. Tracklines from n=9 adult female polar bears satellite collared in 2012 in KB.

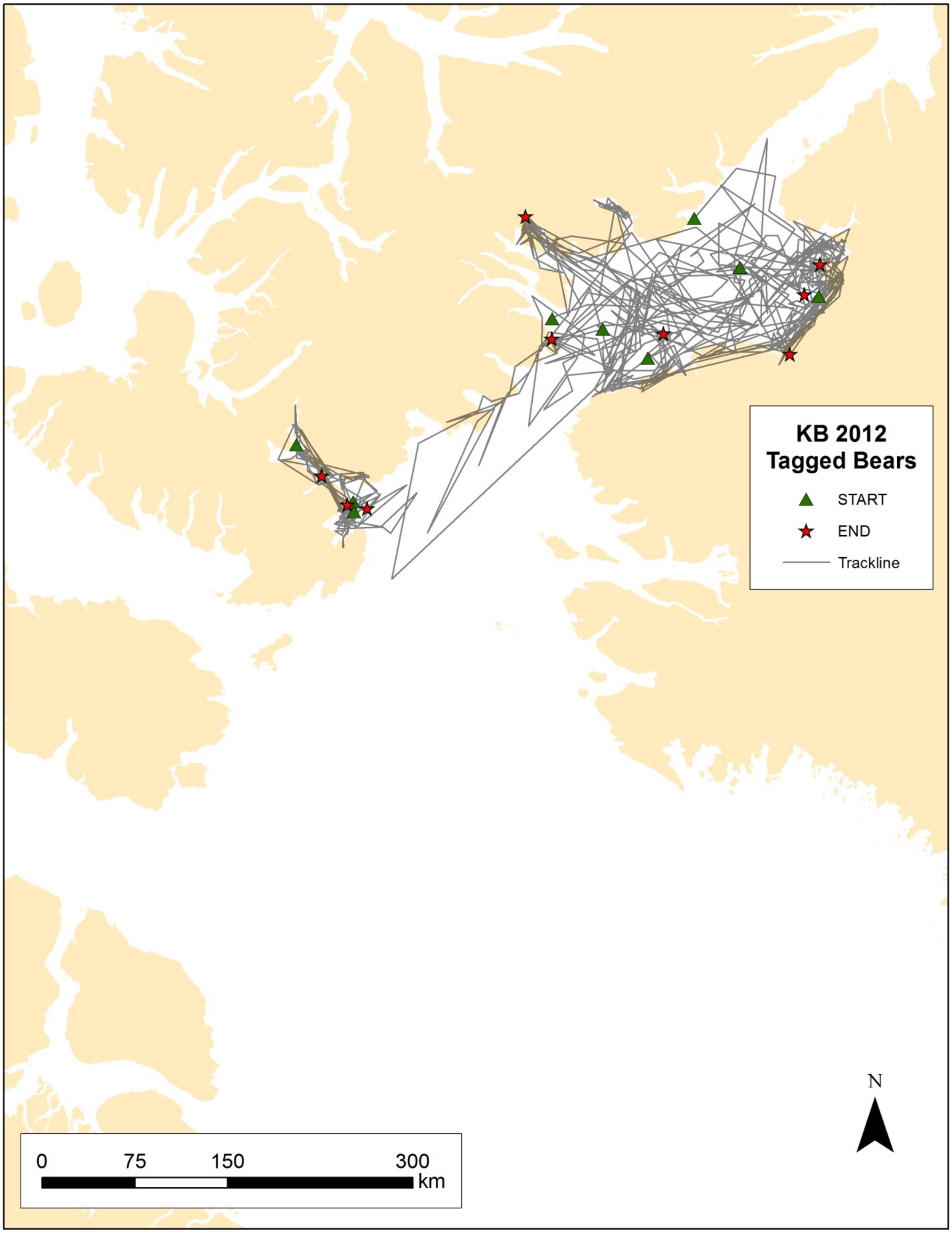


Figure 2.13. Tracklines from n=11 adult female polar bears satellite collared in 2013 in KB.

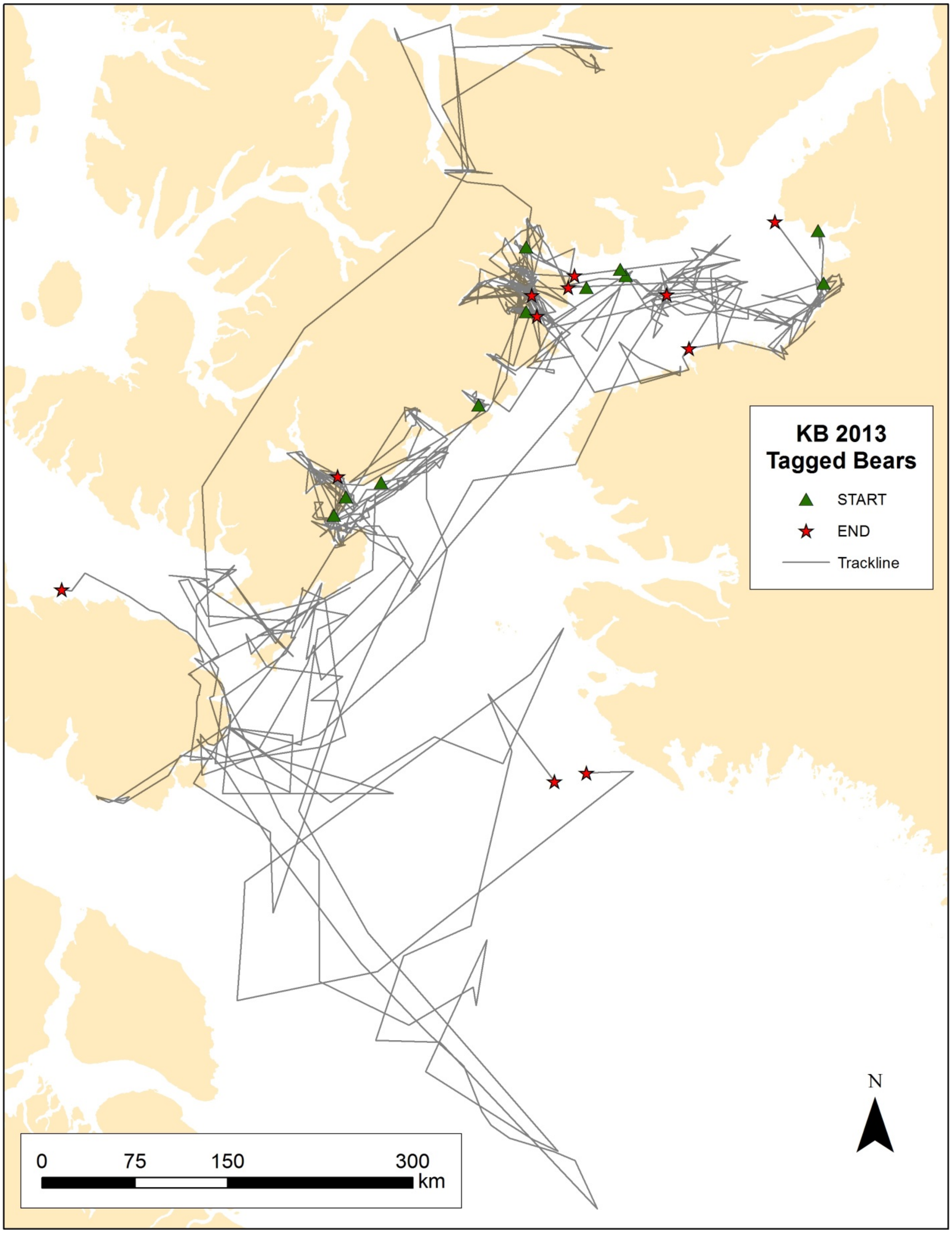


Figure 2.14. Tracklines from n=20 adult female polar bears satellite collared in 2012 and 2013 shown through April 2015 in KB. Inset shows n=1 bear that moved to Russia, excluded from analyses.

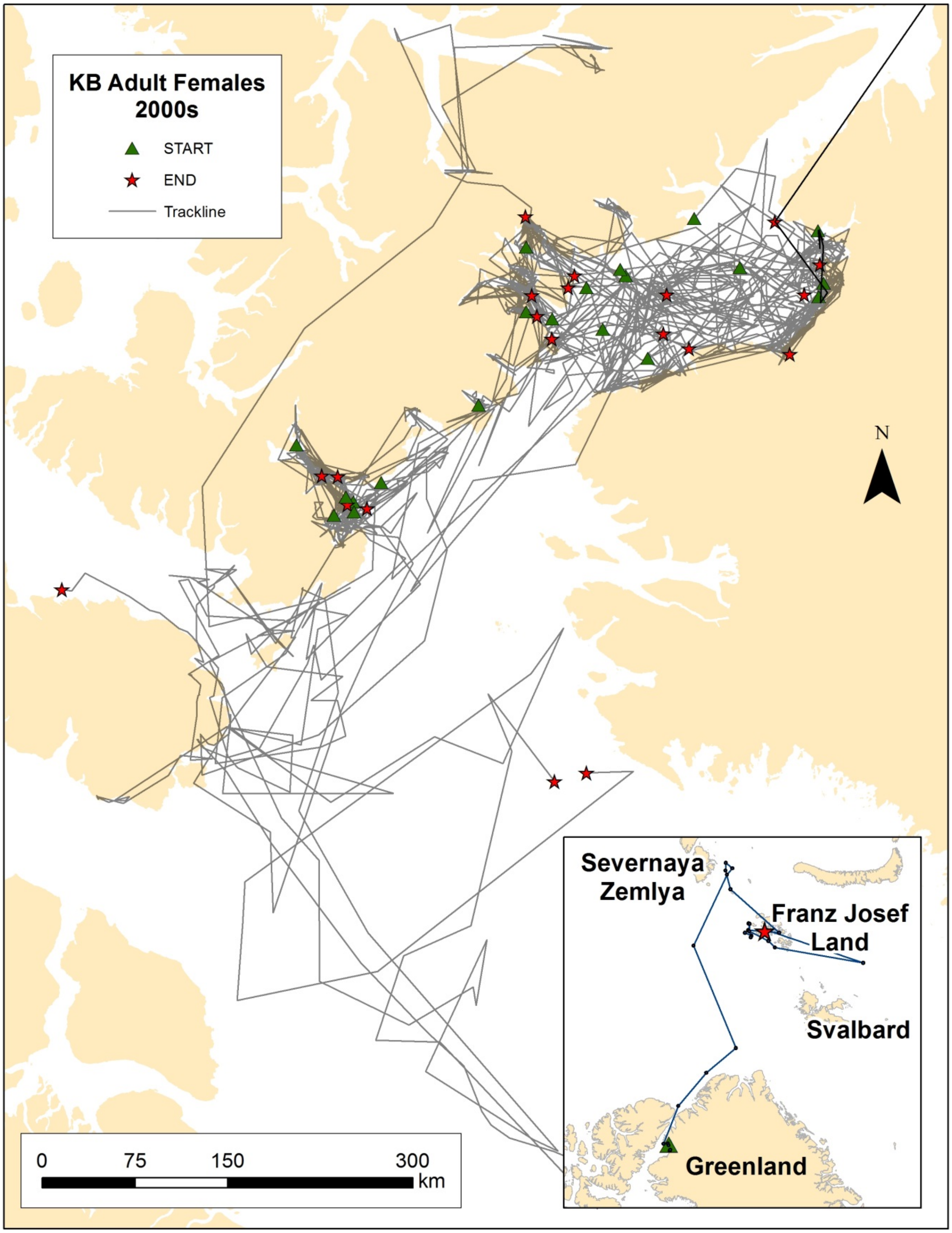


Figure 2.15. Tracklines from n=9 adult male polar bears satellite tagged with ear tags in 2012 and 2013 in KB.

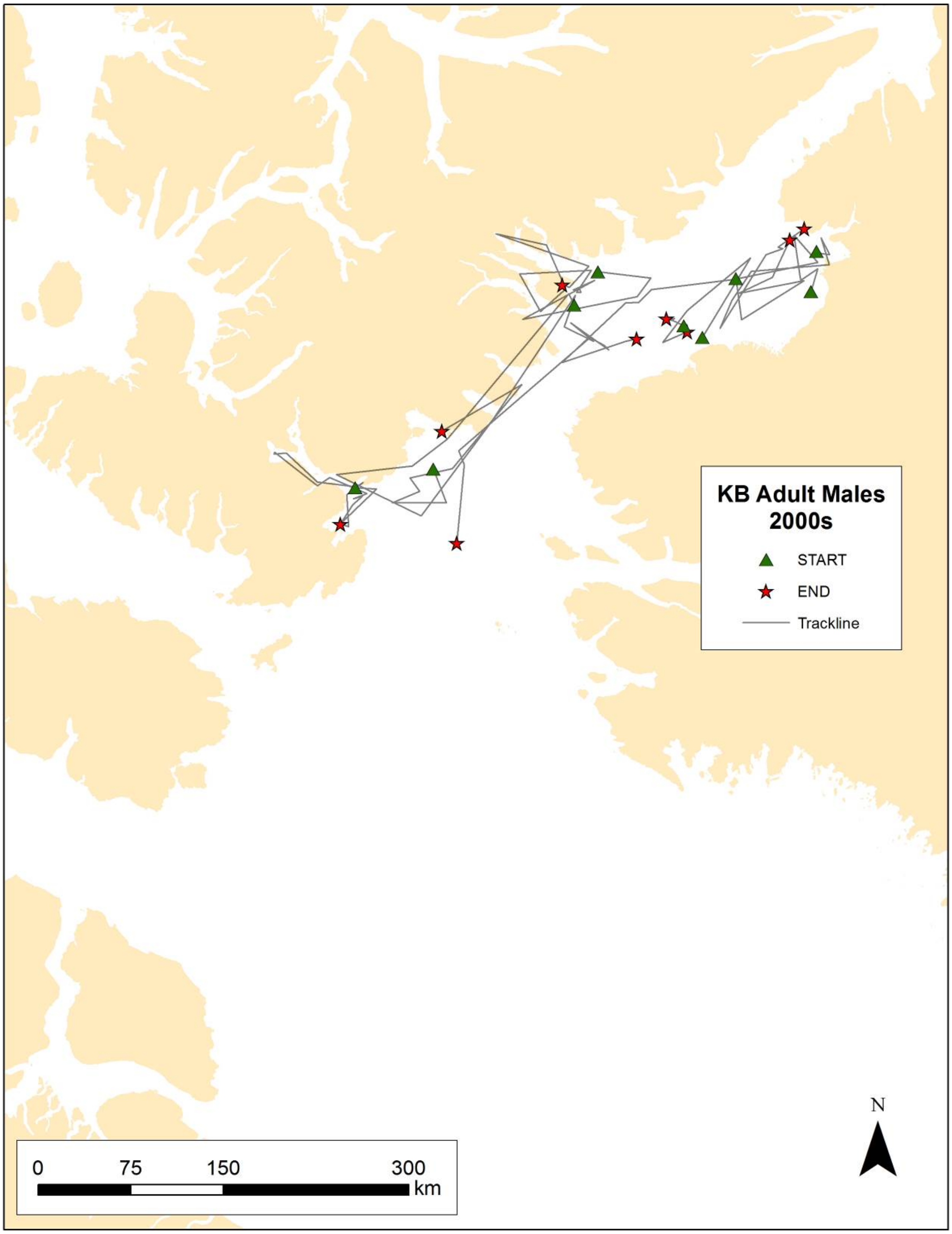


Figure 2.16. 95% kernel ranges for bears captured in 1990s and 2000s in BB by season (winter, spring and summer). See Table 2.4 for areas, overlap and tests for significance between decades.

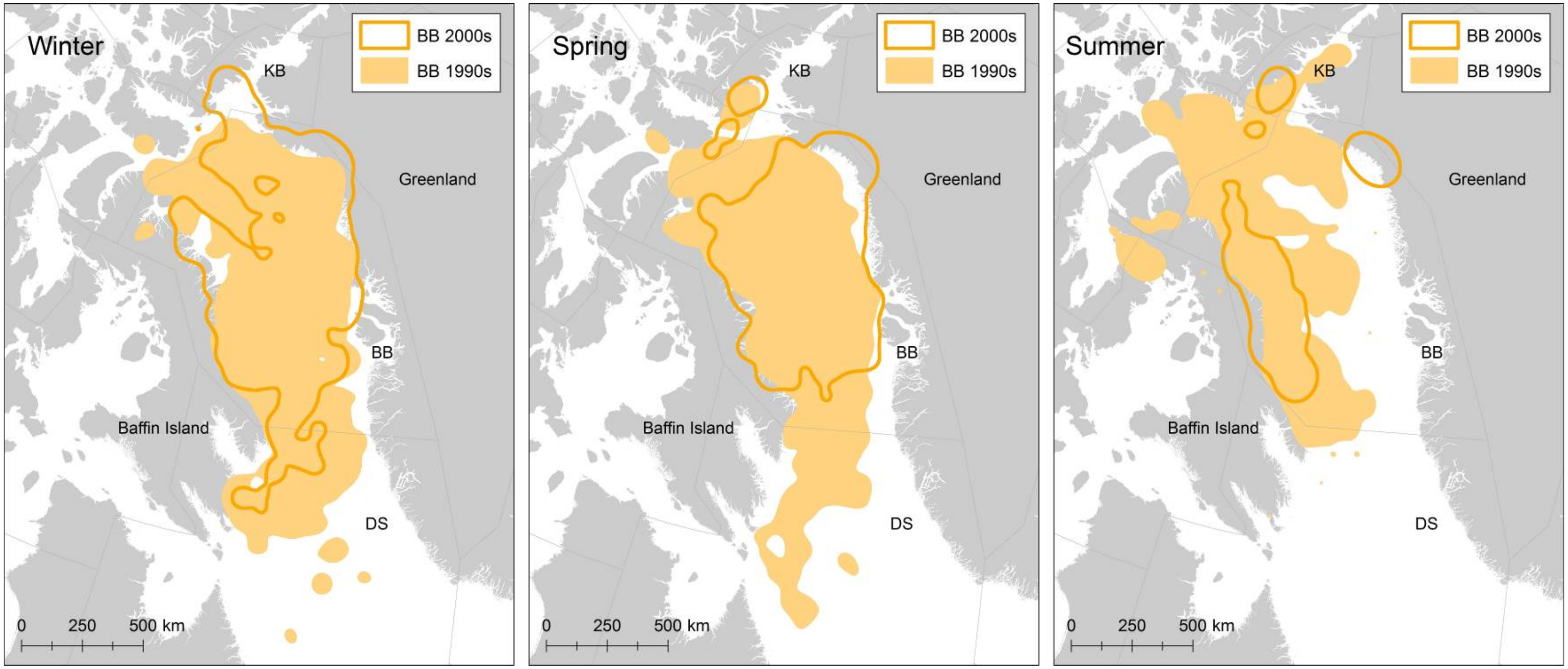


Figure 2.17. Matrix of home ranges shown by month in Baffin Bay for collared adult females in the 1990s and 2000s.

Figure 2.18. Home range sizes between decades for adult female polar bears in BB in the 1990s (red, n=43) and 2000s (blue, n=38). Line represents the mean values by month and shaded area +/- 2 SE. Graph excludes Melville Bay resident bears.

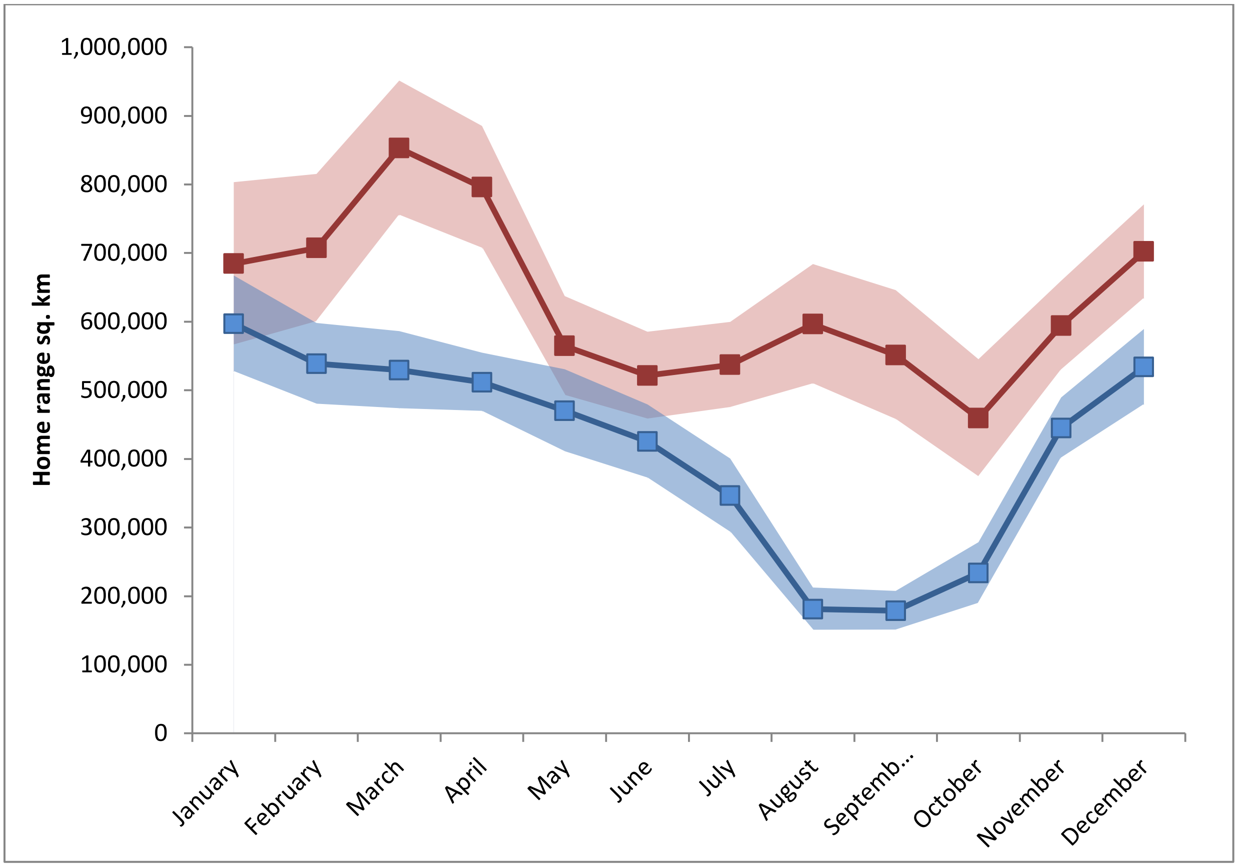


Figure 2.19. 95% kernel ranges for adult female bears captured in 1990s and 2000s in KB by season (winter, spring and summer).

See Table 2.4 for areas, overlap and test for significance between decades.

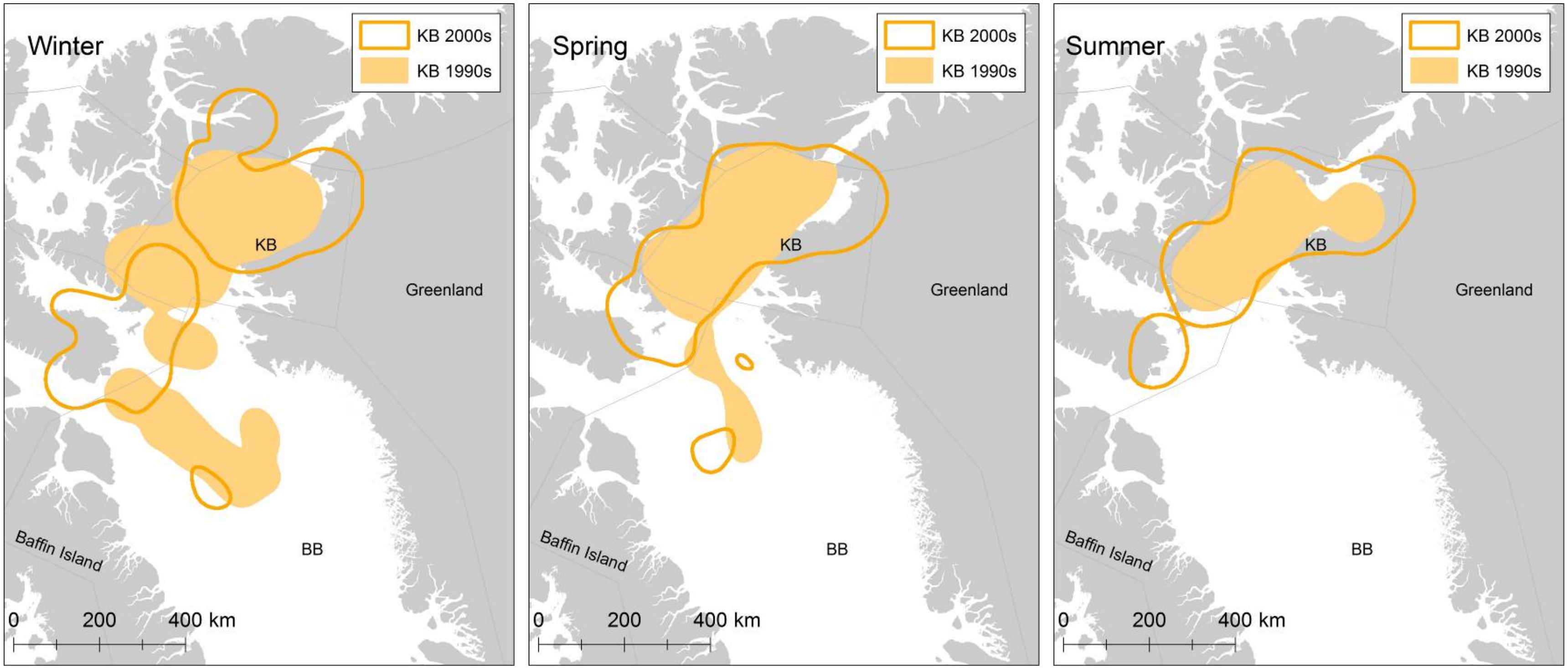


Figure 2.20. Matrix of home ranges shown by month for adult female polar bears in Kane Basin in the 1990s and 2000s.

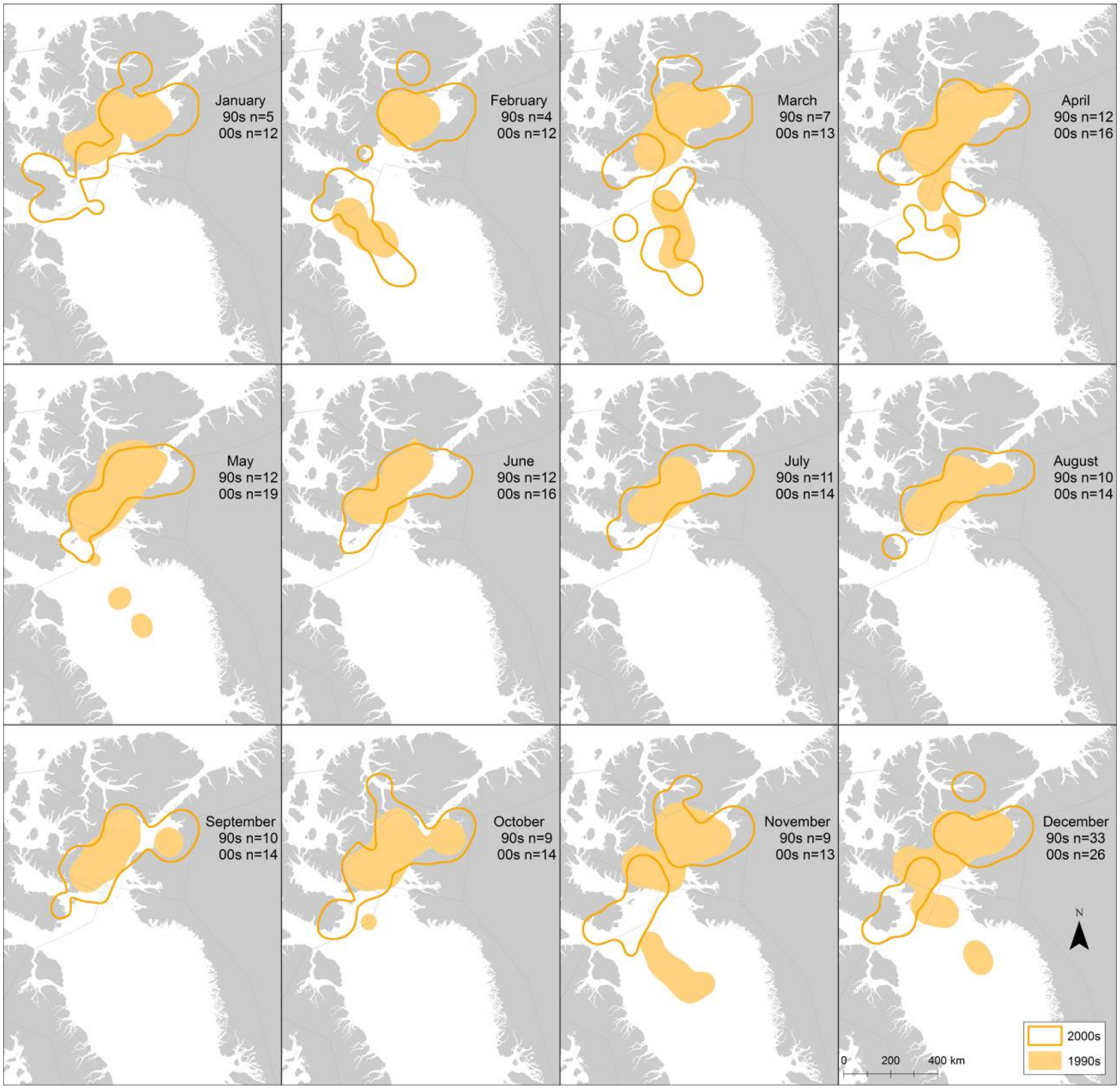


Figure 2.21. Home range sizes between decades for adult female polar bears in KB in the 1990s (red, n=12) and 2000s (blue, n=20). Line represents the mean values by month and shaded area +/- 2 SE.

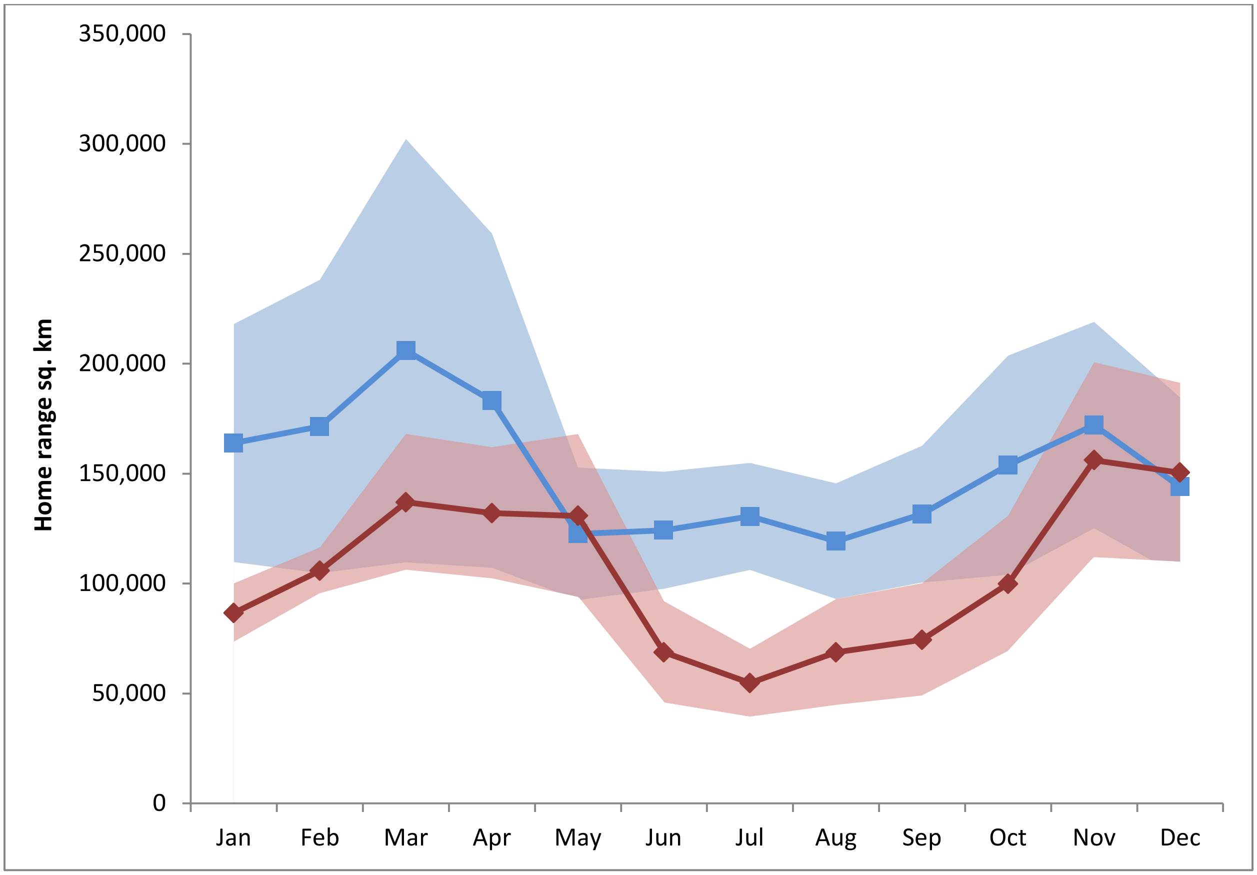


Figure 2.22. Box plots shown by season of median latitude for adult female polar bears in BB in the 1990s (n=43) and 2000s (n=38). Plot excludes the bears that are resident in Melville Bay though inclusion of these bears did not change the significance of the results.

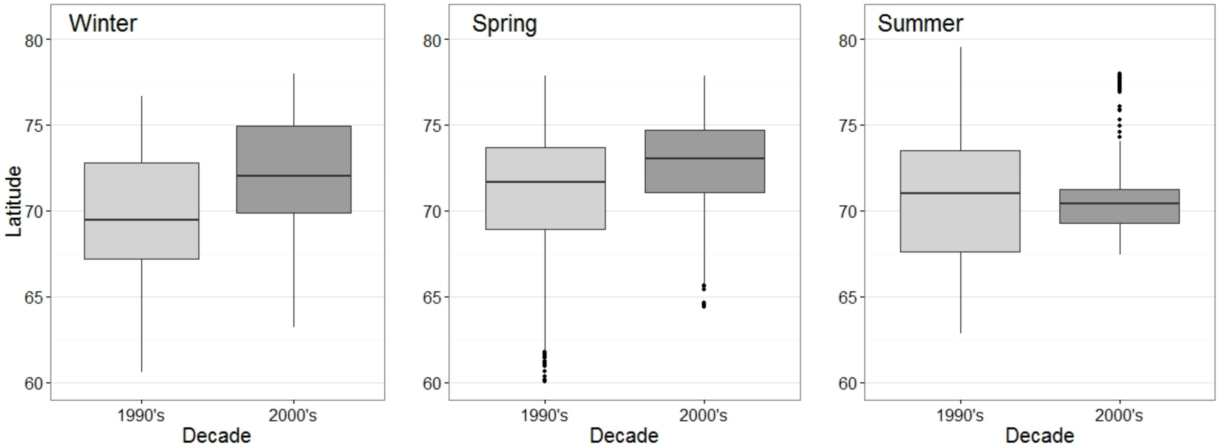


Figure 2.23. Box plots by season of median latitude for AF bears in KB in the 1990s (n=12) and 2000s (n=20).

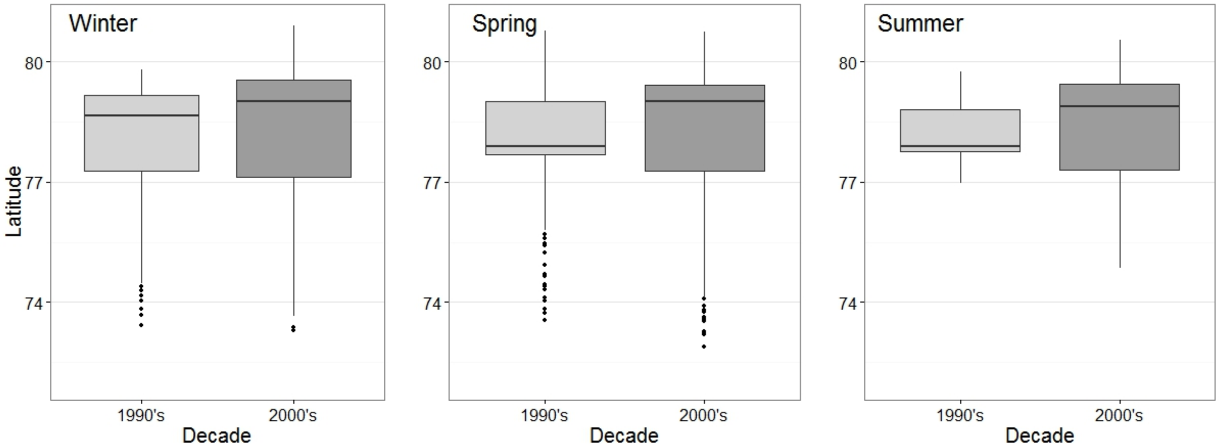


Figure 2.24. Plot of departure timing from region of origin for BB and KB bears in the 1990s and 2000s where departures of any length (min 4 days) were considered. See Table 2.3 for sample sizes.

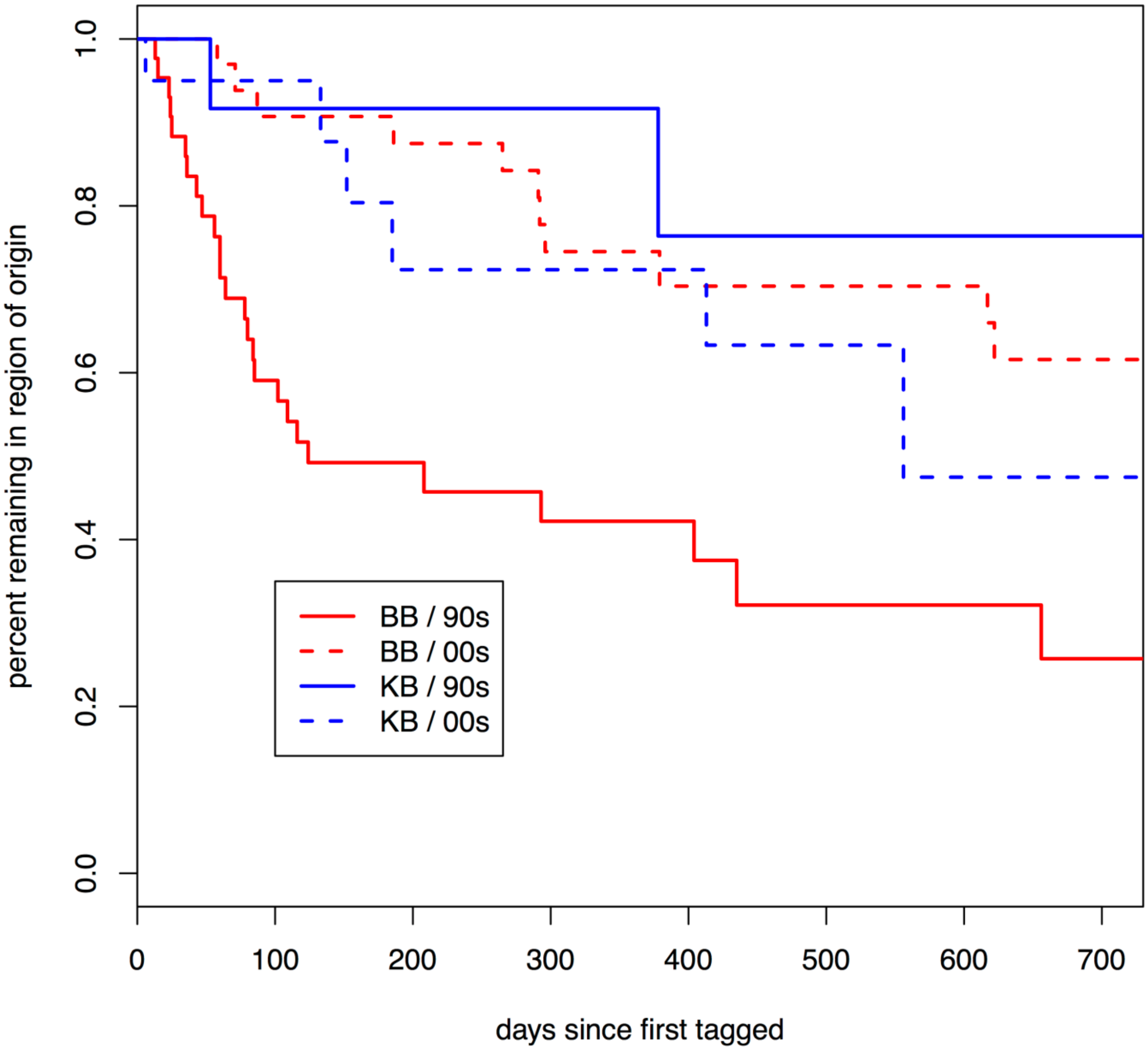


Figure 2.25. Plot of departure timing from region of origin for BB and KB bears in the 1990s and 2000s where only departures 30 days or greater were considered. See Table 2.3 for sample sizes.

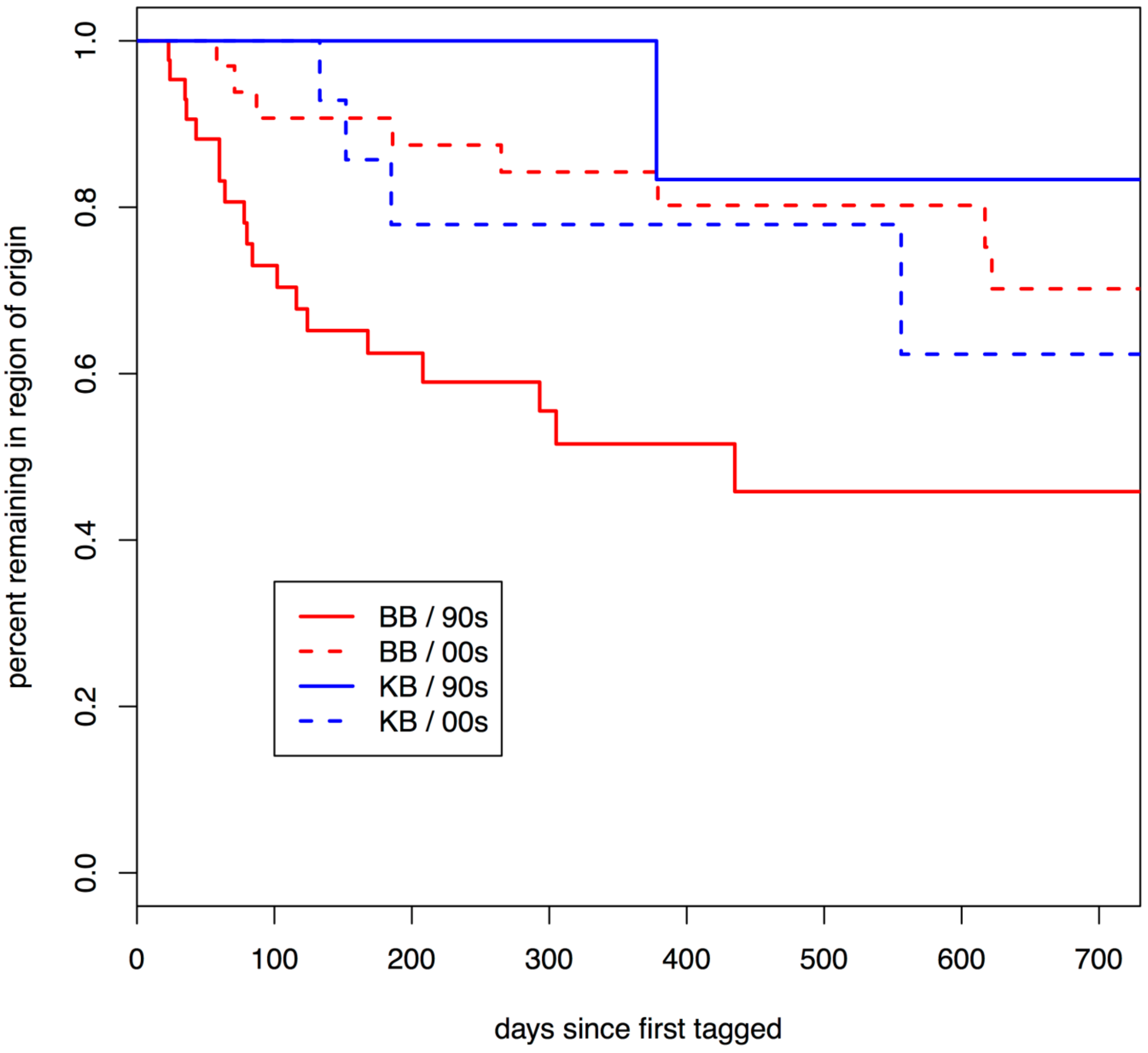


Figure 2.26. Distances (km) between mark and recapture locations of polar bears in Baffin Bay, 2011-2013. Adult females (AF), adult males (AM), cub-of-the-year (CO=COY), subadults (SA), yearlings (YR). Median distance is represented by the black line within each box. Box represents the interquartile range. Whiskers represent maximum and minimum values. Symbols denote significant differences between groups (Bonferonni correction for multiple comparisons, alpha = 0.05).

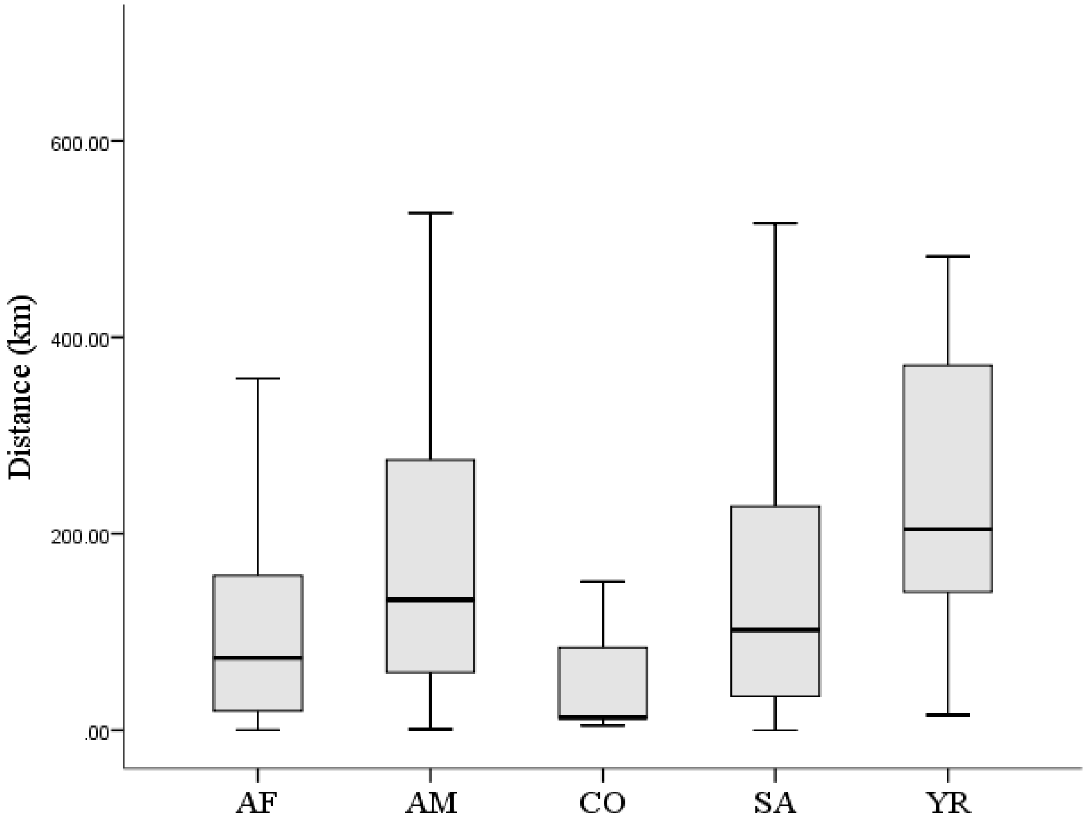


Figure 2.27. Relationship between capture-recapture interval and straight line displacement distance for 29 bears that moved between Baffin Bay and Davis Strait as detected by capture and recapture. Median distance is represented by the black line within each box. Box represents the interquartile range. Whiskers represent maximum and minimum values.

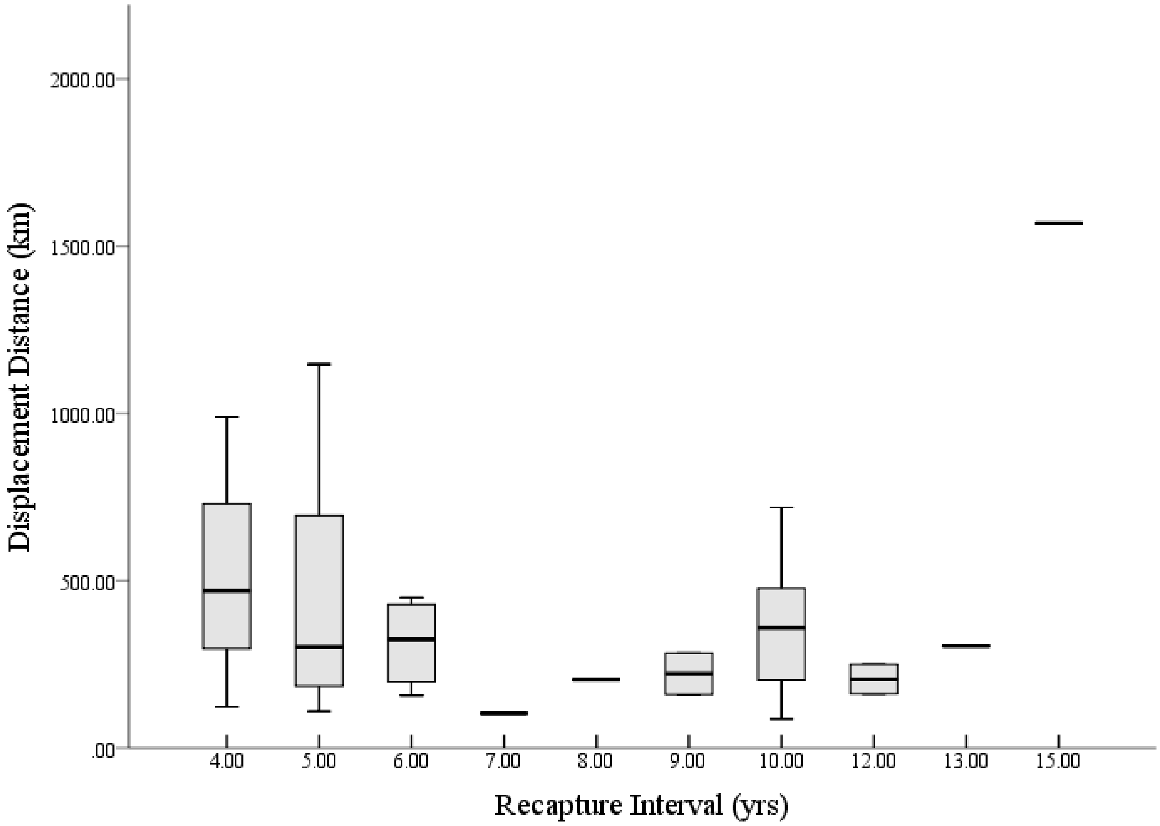


Figure 2.28. Frequency distribution of the distance between capture location and the boundary of the Baffin Bay (BB) and Davis Strait (DS) polar bear subpopulations for 29 individuals that made inter-subpopulation movements as detected by capture and recapture (grey bars), 1991-2013. Distances between capture locations and the BB-DS boundary for all bears (n = 2,771) marked in BB and DS are also shown (black bars). Bars represent proportion of captures occurring within each distance bin.

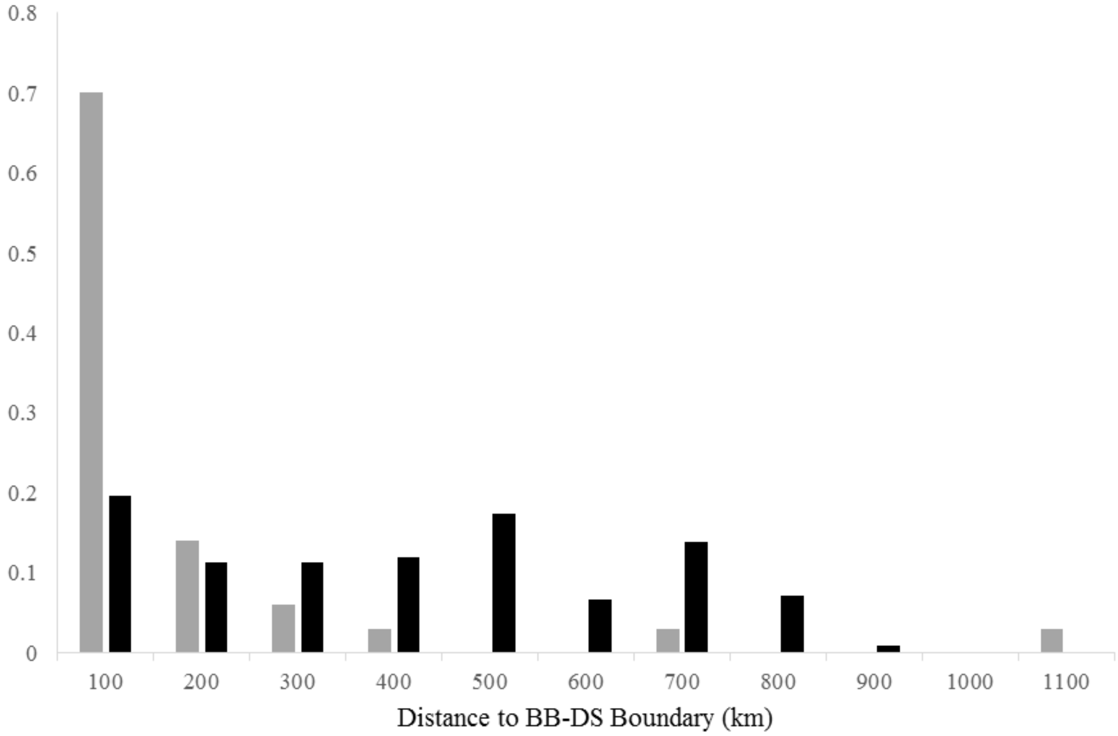
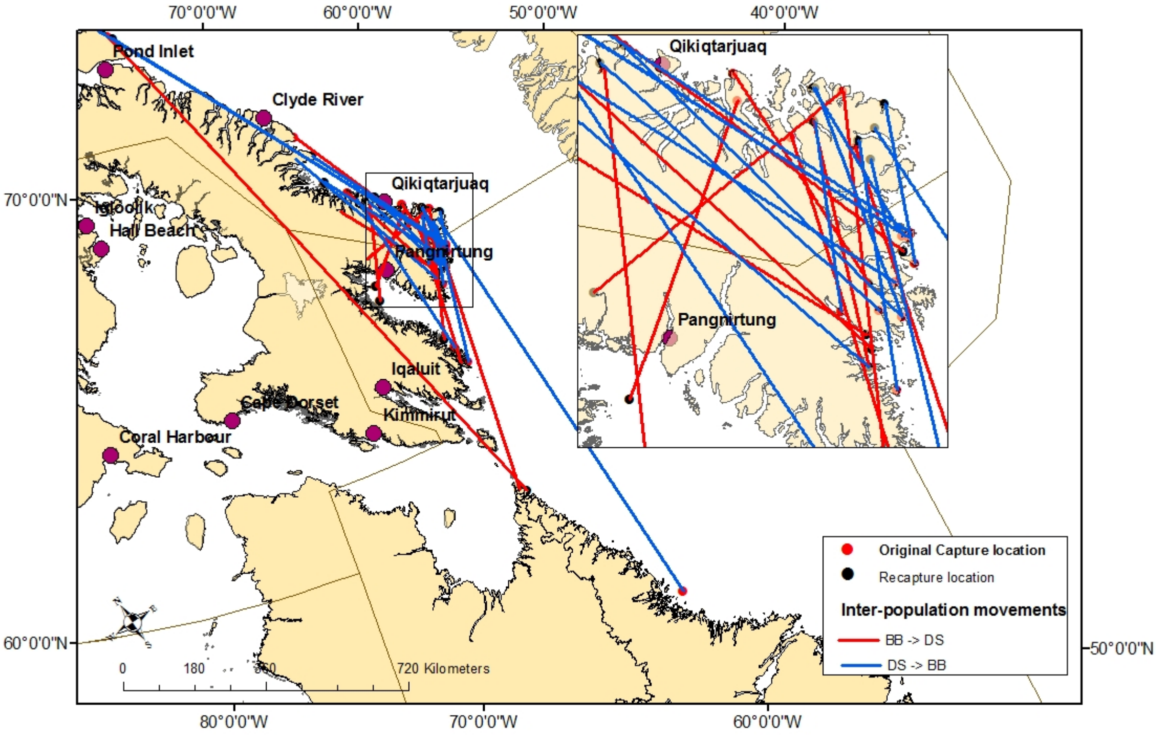


Figure 2.29. Capture and recapture locations of bears known to have made inter-subpopulation movements between mark-recapture sampling sessions in Baffin Bay (BB) (1990-97), Davis Strait (DS) (2005-07), and Baffin Bay (2009-2013).



CHAPTER 3

REASSESSING THE 1990S BAFFIN BAY DATA FOR BIAS AND COMPATIBILITY WITH THE 2010S DATA

KEY FINDINGS

- This chapter evaluates patterns in the 1990s physical MR data, including non-random and incomplete sampling, and the resulting potential for bias in estimates of demographic parameters.

- The 1990s MR sample size was small (average 229 total captures per sampling year), relative to the 2010s (average 470 total biopsies per sampling year), and the number of recaptures in the 1990s was low. There were few dead recoveries during the period between MR sampling studies (1998-2010), particularly in the latter years. Small sample sizes make it difficult to estimate demographic parameters and assess subpopulation trend, limiting both the strength of inference that can be drawn from the 1990s data and our ability to quantify and reduce bias in estimates of demographic parameters.

- The spatial distribution of polar bear physical captures and biopsy samples for the MR studies in the 1990s and 2010s was significantly different. In the 2010s, a larger fraction of bears were captured inland from the coastline, and inside fjords along Baffin Island.

- The difference in distribution of captures between sampling periods was not due to changes in habitat use. Analyses of satellite telemetry data from adult females, providing an unbiased assessment of land use between decades, showed no differences in distance inland or elevation for onshore bears between the 1990s and 2010s. Thus, the difference in capture distributions were a function of different sampling effort, with less effort expended away from coastlines and inside fjords in the 1990s.

- Consistent with the differences in sampling effort and temporary emigration between the 1990s and 2010s, there were significant differences in the composition of the MR samples (e.g., the proportion of bears within each age-sex class) between these two periods. Specifically, adult females were under-represented in the 1990s samples.

- The spatially-defined sampling area (km^2) in Nunavut encompassed the capture and biopsy locations in both decades and represented a minimum area sampled. The sampling area in the 1990s survey was less than ½ of that sampling in the 2010s. The 2010s sampling area encompassed most fjords along the coast and more inland habitat. To evaluate potential biases associated with the smaller sampling area of the 1990s, MR analyses and estimated parameters were compared from two datasets: (1) all 2010s MR data, and (2) a geographic subset of the 2010s MR data that was comparable to the sampling area in the 1990s (Chapter 5).

- In the 1990s there was likely a high degree of temporary emigration from the sampling area on the Baffin Island coast because bears used sea ice offshore in Baffin Bay or in the archipelago in summer. Significantly less sea ice was available in the 2010s and temporary emigration was lower. In the 1990s, ≤30% of radio-collared female bears were inside the sampling area during the MR sampling periods, compared to 70-80% in the 2010s. This suggests that a potentially significant proportion of bears were not available for capture each year during the 1990s, though sample sizes for analysis were small. Completely random temporary emigration from the sampling area should not result in biased demographic parameters. However, the degree of temporary emigration in the 1990s appeared variable and dependent on environmental conditions; and small samples sizes made it difficult to rule out significant bias.

- Additional sources of temporary emigration in the 1990s were non-random and linked to the reproductive cycle of females. Adult females in reproductive classes that were likely pregnant in fall moved farther inland on Baffin Island (e.g., to find suitable denning habitat), compared to non-pregnant females, which likely contributed to the under-sampling of adult females in some years in the 1990s because of the lack of inland sampling.

- There also were technical challenges with the 1990s MR data. Within the 1990s MR data there was uncertainty in identifying bears that were located with the aid of radio-telemetry vs. those located by standard search (i.e., random encounter). Original capture records could not be located and were inferred by comparing available information to the capture history files compiled for the 1990s BB demographic analysis. This uncertainty could result in bias, because knowing which bears were located by telemetry was important in the most-supported MR models for the 1990s data.

- Relative to the 2010s data, the 1990s data were characterized by relatively small sample sizes, incomplete geographic sampling, a likely higher degree of temporary emigration for bears that remained on sea ice during the summer, and potential non-random temporary emigration for adult females that moved farther inland to den. These issues led to an increased potential for bias in estimates of survival and abundance from the 1990s data. As a result, demographic parameters estimated from 1990s and 2010s BB data are not directly comparable and there is a limited ability to evaluate subpopulation trends.

3.1. Background

Accurate knowledge of demographic parameters (e.g., survival, abundance) is important for wildlife management decisions such as determining sustainable harvest levels and evaluating subpopulation viability. Mark-recapture (MR) studies are used to estimate demographic parameters because it is generally not feasible to monitor every individual in a subpopulation.

The results from MR studies can be biased by several factors, including heterogeneity in recapture probability (*p*) that is not accounted for through the choice of sampling design or modeling approach (Williams et al. 2002). The magnitude of bias is generally largest for abundance (Pollock et al. 1990) although estimates of survival probability can have meaningful bias as well (Devineau et al. 2006). Estimating accurate and unbiased demographic parameters for polar bears is particularly challenging. First, sample sizes are relatively small due to challenging environmental and logistical conditions, and the high cost of Arctic fieldwork. Second, polar bears are often distributed across large landscapes at low densities. Only a fraction of the study subpopulation may be accessible to researchers, and this fraction may change from year to year based on environmental conditions and logistical constraints. This limits sample sizes, leads to difficulty in delineating subpopulation boundaries, and means that the effective study subpopulation may be different than the biological population of interest. Third, the high mobility of polar bears and inter-annual variability of their sea-ice habitat can lead to nonrandom movements (i.e., temporary emigration) with respect to the sampling area. Fourth, female bears may be less-observable or unobservable for several months when pregnant or associated with maternal dens, leading to an ‘unobservable state’. Fifth, the three-year reproductive cycle of polar bears makes it difficult to estimate reproductive rates and their relationships with environmental conditions. Finally, relatively long-term datasets are required because of the long life span of polar bears and high inter-annual variability in the Arctic environment.

In recent years, methodological advances have led to an increased ability to detect, quantify, and mitigate bias in demographic parameters from MR studies arising from the challenges listed above. Advances include noninvasive genetic methods to increase sample size

(Lukacs and Burnham 2005); multiple sampling occasions per year under a “robust design” (Kendall et al. 1997); spatially-explicit models to account for heterogeneity in recapture probability as a function of site fidelity (Royle et al. 2014); models with “unobservable states” to account for temporary emigration (Schaub et al. 2004); and models that integrate data from multiple sources (Peñaloza et al. 2014). Some of these methods have been employed for polar bears, whereas others have not been used due to lack of familiarity or practical limits on the types of sampling that can be conducted.

The MR study of the Baffin Bay subpopulation 2011-2013 incorporated noninvasive genetic sampling and modelled live-recapture and dead-recovery data in the same analytical framework. Both of these approaches increased sample sizes and reduced susceptibility to some types of bias. Nonetheless, there remained major challenges to the application of MR models to the Baffin Bay data, and in this chapter we evaluate sampling and biological issues that have the potential to introduce bias in estimates of survival and abundance. Similar investigations of bias have become a standard part of MR studies for polar bears (e.g., Regehr et al. 2010), and are necessary to understand the strength of inference that can be drawn from MR studies. In this chapter we focus on reassessing the 1990s BB data because, compared to the 2010s data, the 1990s data had smaller sample sizes, reduced geographic coverage, and other uncertainties and limitations. This assessment directly informs our ability to compare results from the 1990s and 2010s data and evaluate trends in polar bear survival and abundance between sampling periods.

Distribution of Mark-Recapture Sampling on Baffin Island

Prior to the 2011-2013 survey of the Baffin Bay subpopulation, MR sampling occurred during several periods. Initial sampling was conducted during the 1970s (northern Baffin Island

and Bylot Island, near Lancaster Sound), early 1980s (east-central Baffin Island), and late 1980s to early 1990s (Canada and Greenland, as part of movement studies; Figure 5.2; Taylor et al. 2005). Early sampling efforts were generally restricted to spring-time and primarily occurred on landfast and nearshore pack ice. These studies documented that an unknown but likely large proportion of the subpopulation was on sea ice farther offshore during the spring and therefore unavailable for capture. We excluded these early data from present analyses (*cf*. Taylor et al. 2005, in which these early data were included) because the early sampling occurred in a different season (i.e., spring) and was spatially variable and restricted. Additionally, lack of tissue samples from early sampling precluded genetic identification for use in the present study.

In 1993 –1995 and 1997, more systematic sampling occurred during fall ice-free seasons (during September and October) on Baffin and Bylot islands (Figure 3.1). There was no fall sampling in 1996 due to logistical and resource constraints. These data formed the core of the study reported by Taylor et al. (2005) who estimated the number of polar bears in Baffin Bay at 2,074 (95% confidence interval: 1544-2604) in 1998. Taylor et al. (2001) indicated that a large majority of polar bears were onshore in summer retreat areas on Bylot and Baffin islands during the autumn. Taylor et al. (2005) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island. Consequently, Taylor et al. (2005) suggested that the autumn onshore sampling in 1993-1995 and 1997 provided improved coverage of the subpopulation and more reliable abundance estimates compared to those derived from the 1980s BB data, which Taylor et al. (2005) suggested were biased low.

In 2011-2013 we completed a second fall-time MR sampling study (August – October) on the coasts of Baffin Island (Figure 3.1). Data from West Greenland were also collected (see

Chapter 5, Figure 5.8). During this study, new data on movements and spatial distribution of bears were also obtained via satellite telemetry in BB and KB. This information was used to assess subpopulation boundaries (Chapter 2) and habitat use relative to the 1990s (Chapter 4), but also to improve MR study design (i.e., stratify the study site; Chapter 5) with the objectives of reducing heterogeneity in capture probabilities and more efficiently allocating survey effort. The 2011-2013 study (see Chapter 5) was largely modeled after Taylor et al. (2005) in that bears were targeted during the ice-free season, to obtain estimates of abundance and vital rates that might be comparable to Taylor et al. (2005) therefore useful for assessing trend.

Here we compare the spatial and temporal distribution of physical captures and biopsy sampling on Baffin Island for sampling 1993-1995 and 1997 vs. sampling during 2011-2013 (referred to as the “2010s”). The goal is to evaluate whether there were important differences in sampling, which could lead to different biases or different definitions of the effective study subpopulations (e.g., if a large group of bears was systematically missed in one study period, then the effective study subpopulation for that period would be smaller). Field records (e.g., Global Positioning System helicopter logs, navigation maps) delineating survey effort 1993-1995 and 1997 were unavailable. Therefore, we plotted sighting data from Taylor et al. (2005) in a Geographic Information System (GIS; ArcMap 10.2, ESRI, Redlands, California, USA) to examine the spatial distribution of captures compared to the 2010s. We also used historic and current radio telemetry data to identify whether potential differences in capture locations were influenced by changes in the onshore movements and habitat use of polar bears.

Methods – Maps of physical capture and biopsy sampling locations (hereafter collectively referred to as “captures”) on Baffin Island suggested that captures in the 1990s were more limited to coastal areas, whereas captures in the 2010s included bears located farther from the coast and deep inside fjords, including higher altitudes (Figure 3.1). We examined the hypothesis that the discrepancy in capture locations across periods reflects differences in sampling effort rather than a shift in the onshore distribution of bears. We calculated the distance to the nearest coastline and the distance to the smoothed outer Baffin Island coastline for each capture location in the 1990s and 2010s. The smoothed coastline followed the contour of the true physical coastline of Baffin Island, but was smoothed across fjords with a straight segment orthogonal to the fjord direction. We smoothed fjords only when the distance across the mouth of the fjord was $\leq$ 7 km using an Azimuthal Equidistant projection (WGS84 datum). We calculated the distance to both coastlines (original and smoothed) for all captures of independent bears (i.e., age 2 or older) that were located on mainland Baffin and Bylot islands (i.e., not on offshore islands) and were successfully genotyped.

We compared the distance-to-coast results to locations of radio-collared bears onshore during the 1990s and 2010s to evaluate whether differences in capture locations reflected differences in sampling effort or differences in the distribution of bears. Given that recent analyses of movement data suggest significant changes in sea-ice habitat use and onshore timing (Chapter 4), we considered the possibility that bears had also changed their behavior and habitat use while on land. First, we verified that the sample of 1990s bears collared in the fall on Baffin Island were comparable to the sample of 2010s bears collared in the spring in West Greenland, by assessing what fraction of spring-collared bears used the area on Baffin Island where bears were collared in the fall (see details in Chapter 2). Overall, 92% of the 2010s spring-captured bears used the fall collaring area. This suggests that, although radio-collaring occurred in different seasons and areas across the two time periods, the collared bears exhibited similar movement and habitat use patterns, and therefore provided comparable data for evaluating onshore habitat use across time periods.

Using satellite telemetry data, we calculated the distance inland from the smoothed coastline and Digital Elevation Model (DEM) elevation (m) for all locations of collared female bears during summer months (August-October). We used land covariates derived from the 22 m^2ASTER GDEM for all positions in Canada (http://www.jspacesystems.or.jp/ersdac/GDEM/E/4.html). We only used adult female bears on Baffin Island and calculations excluded resident bears that remained year-round on the Melville Bay glacier ice.

We also examined distance to the smoothed Baffin Island coastline for adult females as a function of reproductive status (captured alone, as mating pairs; or with COY, yearlings, 2-year old cubs) to evaluate whether this factor may have influenced temporary emigration with respect to the sampling area (particularly the nearer-shore sampling area in the 1990s). For this specific analysis (reproductive state examination) we only examined adult females in the year of collar deployment because their reproductive status was known at the time of capture in spring, thus could be assumed in fall. We excluded bears on sea ice during August-October.

Results – The mean distance of captures to the smoothed coastline was smaller in the 1990s ($\bar{x}$= 5.1 km, SD = 7.2, n = 438) compared to the 2010s ($\bar{x}$= 8.6, SD = 11.9, n = 766, Mann-Whitney U test: z = 3.4, P < 0.001). Detailed results are provided in Table 3.2. Furthermore, a greater proportion of independent bears were captured near the smoothed coastline during the 1990s than the 2010s (Figure 3.2). For example, 84% of captures occurred within 10 km of the smoothed coastline during 1993 – 1997, compared to 72% of captures during 2011 – 2013. Similarly, one independent bear was captured > 35 km from the smoothed coastline during the 1990s sampling, whereas 28 independent bears were sampled > 35 km from the smoothed coastline during the 2010s. The corresponding analysis using satellite telemetry found no significant differences in the distance of adult females from the smoothed Baffin Island coastline between the 1990s and 2010s; adult female bears on average in the 1990s were about a mean 17 km from the smoothed coast in August and September, where as in the 2000s they were about 13 km in those months, however standard errors were overlapping (Figure 3.4). Also, there were no differences in the mean monthly elevation used by adult females on Baffin Island between the 1990s and 2010s (Figures 3.4 and 3.5).

Satellite telemetry analyses further documented differences in the inland distance of adult females on Baffin Island as a function of reproductive status. Females that were most likely available to breed and become pregnant in spring (e.g., those captured alone, with 2 year old cubs, or as mating pairs in spring) were significantly farther inland in fall than adult females captured with COYs or yearling cubs (Table 3.2). This was especially pronounced for adult females captured in mating pairs (on average 27-35 km inland).

In contrast to analyses based on distance to the smoothed coastline, the distance of captures to the true coastline (not smoothed) was consistent between sampling periods (Figure 3.3), averaging 1.8 km (SD = 2.8) in the 1990s and 1.5 km (SD = 2.5) in the 2010s. This suggests that the difference in capture locations between the two sampling periods was largely due to less effort spent searching and capturing bears in the inland portions of fjords in the 1990s compared to the 2010s. For adult females, mean distances to the true coastline were 6.4 km (SD: 8.0) and 10.2 km (SD: 12.6) during the 1990s and 2010s, respectively (Figures 3.2 and 3.3).

Summary – The distribution of polar bear captures on Baffin Island differed significantly between sampling in the 1990s and 2010s. Specifically, the capture data indicate an under-representation of bears in fjords and inland regions during the 1990s (see also Chapter 5). Satellite telemetry location data, which were collected from independent bears over several years and were not influenced by which areas were searched in any given year, did not suggest a shift in the onshore distribution of polar bears. Given that no changes in adult female use of land habitats was detected (also see Chapter 4 terrestrial resource selection), the differences in capture distribution can be attributed to differences in sampling. During the 1990s, capture effort was concentrated on islands, along the outer coastline, and near the mouths of fjords (Figures 3.2 and 3.4). During the 2010s, these areas were searched as well as the inland portions of fjords. This is particularly prominent in central and northern Baffin Island, where no captures were recorded beyond the mouths of fjords during the 1990s. In contrast to the southern parts of Baffin Island the central and northern parts have a higher and more mountainous terrain. Finally, satellite telemetry data also indicate that adult females in different reproductive status show a non-random pattern of moving farther inland, likely in search of locations to construct maternal dens. These findings suggest a non-random probability of being a temporary emigrant as a function of the multi-year reproductive state. Taken together, these findings suggest that restricted geographic sampling in the 1990s likely led to higher probabilities of temporary emigration from the sampling area during that time period, compared to the 2010s. Furthermore, the probability of being a temporary emigrant appears non-random. Variable and non-random temporary emigration is known to introduce bias into estimates of survival and abundance under some conditions (Peñaloza et al. 2014).

Size of the Mark-Recapture Sampling Area on Baffin Island

Following from the previous section, we calculated the sizes of the effective MR sampling areas on Baffin Island in the 1990s and 2010s.

Methods – We delineated the sampling areas based on the spatial distribution of capture locations. We first used ArcGIS to create 99% kernel density contour around all capture locations in each time period. We then adjusted this contour on a point-by-point basis to ensure that the final estimated sampling area was within 1 km of the outermost capture locations. The sampling area did not extend offshore, except in a few cases in the 1990s where there were offshore points, in which case the boundary was kept within 1 km of those points. When capture locations occurred inside a fjord, it was assumed that sampling effort occurred everywhere from the mouth of that fjord to the capture location.

Results – The size of the MR sampling areas differed significantly between the 1990s and 2010s. The estimated sampling area was ~28,700 km^2 in the 1990s and ~60,200 km^2 in the 2010s. The 2010s sampling area included most fjords along the Baffin Island coast and reached farther inland than the 1990s (Figures 3.6 and 3.7). Furthermore, the 1990s sampled area was almost entirely contained within the 2010s sampling area (Figure 3.8). This made it possible to subsample the 2010s capture data, using the restricted 1990s sampling area, for the purpose of evaluating the influence of the size of sampling area on estimates of abundance from the two time periods (see Chapter 5).

Temporary Emigration Related to the Availability of Sea ice

Previous sections in this chapter documented a smaller onshore sampling area in the 1990s, which likely resulted in higher and potentially non-random temporary emigration from the sampling area in the 1990s. Here we evaluate temporary emigration related to the availability of sea ice, which declined between the two study periods in all months of the year, including the summer when sampling on Baffin Island occurred. We used satellite telemetry data to assess the fraction of adult females that were located in the sampling area vs. out of the sampling area (including on the sea ice) in the 1990s compared to the 2010s.

Methods – For each year of sampling in the 1990s and 2010s, we used the specific date range when sampling occurred (Table 3.3) to calculate the proportion of independent collared bears located inside the sampling area, as well as the proportion of locations from each individual bear that were inside the sampling area. First, we identified independent adult females that were wearing functional radio-collars during the sampling period. To ensure that location data were independent, we did not include locations from the same sampling period on which an adult female was captured and fitted with a radio-collar. For example, if a bear was captured and collared on October 1, 1993, locations from that individual through October 8, 1993 were not used (Table 3.3). However, locations from that individual in 1994 and 1995 were considered independent and included in analyses. If a bear was captured in spring of a given year, her location data were considered independent by fall of that year. We considered a bear to be located inside the sampling area if that bear had 1 (or more) telemetry location inside the sampling area.

We evaluated average sea-ice conditions in Baffin Bay during each sampling period for the 1990s and 2010s to determine whether bears that were located outside of the sampling area, were located on sea ice. For each sampling period, we mapped mean sea-ice concentration during the week that encompassed the mid-point of the sampling period, using the Passive Microwave data (SMMR/SSMI) sea-ice concentration dataset from the National Snow and Ice Data Center (see Chapter 4). We then superimposed independent bear locations on the sea-ice concentration map, and visually examined whether bears located outside of the sampled area were in an area with a substantial concentration of sea ice and therefore likely using the sea ice.

Results – Table 3.3 shows the date range of MR sampling in each year. There were a maximum of 13 independent adult female bears transmitting with satellite collars during the 1990s sampling periods. The number of individuals declined over the course of the 1990s study because most collars were deployed at the beginning of the study and some collars failed (Table 3.4). The largest number of transmitting independent bears occurred in 1993, and by 1997 there were none. There were also a maximum of 13 transmitting independent bears during a given sampling period in the 2010s, although sample sizes remained higher through the 2010s due to longer collar attachment periods (Table 3.4). We found large differences in the proportion of transmitting independent bears using the sampling areas between 1990s and 2010s. In the 1990s, 0-20% of females occurred within the sampling area during the MR sampling period (Table 3.4, Figure 3.9 - 3.11). In the 2010s, 67-80% of females occurred within the sampling area during the MR sampling period (Table 3.4, Figure 3.12 - 3.14).

Sea-ice availability in Baffin Bay declined between the 1990s and 2010s. In the 1990s, a substantial amount of sea ice was available in offshore central Baffin Bay; within the Canadian archipelago, including around Devon Island; and in Lancaster Sound and Kane Basin (Figures 3.15-3.21). In 1993, when the largest proportion of independent bears was offshore during the sample period (Figure 3.15), there was a persistent area of sea ice available in central Baffin Bay. In other years in the 1990s, some bears were located on the advancing sea ice forming in northern Baffin Bay (Figures 3.15-3.17). In contrast, in the 2010s all bears (excluding resident bears in Melville Bay) were distributed on land on Baffin Island or in Kane Basin (Figures 3.18-3.20) during the sampling periods. There were no bears on offshore ice in the 2010s, because sea ice had melted completely in central Baffin Bay by July (see Chapter 4). The differences in sea-ice conditions between the 1990s and 2010s can been seen clearly using juxtaposed sea-ice concentration maps (Figure 3.21).

In addition to relatively fewer adult females being present in the sampling area during the 1990s, most bears with >1 location in the sampling area did not spend the entire sampling period there, but rather were passing through (Table 3.5). In the 1990s, approximately 44% of locations received for bears that used the sampling area, were located inside the sampled area (see Chapter 1 for information on location filtering and subsampling). In the 2010s, approximately 94% of locations received for bears that used the sampling area, were located inside the sampled area. Although sample sizes were small and unevenly distributed across years, the higher probability of bears in the 1990s being located outside the sampling area appeared largely due to the presence of sea ice, whereas in the 2010s sea ice was absent and bears exhibited reduced summertime movement rates (see Chapter 4).

Summary – Temporary emigration from the sampling area during the autumn sampling period has the potential to introduce bias into estimates of demographic parameters from this study. Our analyses suggest that the proportion of adult females (and presumably other sex and age classes) in the sampling area was likely lower in the 1990s compared to the 2010s, for two reasons. First, some bears located inland in the 1990s were not available to capture teams because there was apparently limited inland search effort, and in particularly bears were not captured in the deep inland portions of fjords. Furthermore, the location of bears from the coast—and therefore the susceptibility of bears to capture—appeared related to reproductive status, in which case the probability of being a temporary emigrant may have been nonrandom. Second, a proportion of radio-collared polar bears used offshore ice in the 1990s, whereas sea ice was less available in the 2010s and therefore a substantially higher proportion of bears were likely inside the sampling area. Because of small sample sizes that varied across years, we were unable to calculate precise estimates of temporary emigration rates or to evaluate the magnitude and direction nonrandom patterns (e.g., Markovian dependence) in a statistically rigorous manner. Nonetheless, multiple lines of evidence indicate higher temporary emigration in the 1990s, compared to the 2010s. The most likely effect of temporary emigration is an unknown but potentially meaningful negative bias in estimates of survival and abundance (Schaub et al. 2004, Devineau et al. 2006, Peñaloza et al. 2014).

Additional sampling considerations

Small sample sizes lead to multiple challenges into MR studies, including high variance in estimated parameters, small-sample bias, susceptibility to bias due to violation of modeling assumptions (e.g., un-modeled heterogeneity in recapture probability), and limited options for quantifying or mitigating bias (Williams et al. 2002). Compared to the 2010s data, sample sizes in the 1990s were small and had a low proportion of recaptures (Table 3.1). For example, the entire dataset for adult females (F2+ age group) included only 5 animals recaptured by standard search in 1995, and 14 animals recaptured by standard search in 1997 (note that numbers in Table 3.1 are higher, because they include “likely” recaptures and re-sightings of bears located by radio telemetry; see below). Furthermore, there were relatively few dead recoveries during the interim period when no sampling occurred (1998-2010), particularly in the later years. For example, an average of 1.3 research-marked females per year were recovered in the harvest, from 1998-2010. Conceptually, it is apparent that the small number of live recaptures during 1990s live-encounter sampling, the gap years between 1990s and 2010s sampling, and the small number of dead recoveries during the gap years contain a limited amount of information and will lead to estimates of demographic parameters that have substantial uncertainty and low resolution (i.e., that few demographic parameters can be estimated, requiring the estimation of “average” parameters over years or groups of animals).

There were significant differences in the composition of the MR samples (i.e., the proportion of bears within each age-sex class, based on initial captures) between the 1990s and 2010s in Baffin Bay (Table 3.1). There were more adult and sub-adult male captures in the 1990s, whereas there were more sub-adult and adult female captures in the 2010s. The proportion of total female captures in the1990s was less than the 2010s (mean annual proportion of age 2+ female captures : total 2+ captures, 1990s: 0.42; 2010s: 0.53; Table 3.1). Given the spatial segregation of bears by sex and age-classes and reproductive states (see section Distribution of Mark-recapture Sampling on Baffin Island), the apparent under-representation of females in the 1990s samples likely reflects at least in part the coastal-focused sampling protocols during that period, rather than true differences in the composition of the subpopulation (although we cannot rule out progressive depletion of males through the 2010s due to high harvest).

Development of an Individual Covariate to Explain Inland Habitat Use

Given the apparent differences in sampling effort between the 1990s and 2010s, the spatial segregation of bears by sex and age class, and differences in the composition of capture samples, we hypothesized that proximity to the coastline may explain variation in recapture probabilities. We also wanted to explore whether proximity to the coastline for an individual bear was nonrandom across years (e.g., whether bears captured inland were more likely to be recaptured inland). We assigned capture locations to either coastal or inland categories, using a threshold of 2 km from true and smoothed coastlines, and compiled contingency tables for individuals captured in multiple sampling periods. For individuals captured three or more times, we used only an individual's first two capture events and included only those bears initially captured as independent animals, since the locations of cubs-of-the-year and yearlings were dependent on the location of their mothers.

Use of inland areas appeared nonrandom. Individual polar bears initially captured inland from the true coastline were more likely to be recaptured inland in subsequent years (all data: $\chi^2 = 10.4$, $P = 0.0012$; 1990s only: Fisher's exact test $P = 0.10$; 2010s only: Fisher's exact test $P = 0.02$). Similarly, bears initially captured inland of the smoothed coastline were more likely to be recaptured inland (all data $\chi^2 = 18.1$, $P < 0.0001$), a pattern which was driven largely by the 2010s (Fisher's exact test $P < 0.0001$; 2010s only: Fisher's exact test $P = 0.21$; 1990s only). As such, we incorporated a proximity to coastline covariate for modeling recapture probability in demographic analyses (see Chapter 5).

Challenges with Using the 1990s Radio Telemetry Data

Some aspects of the 1990s radio-telemetry data were uncertain or unavailable, presenting challenges to the use of these data in the current analysis. As part of a study examining subpopulation delineation and spatial ecology (Ferguson et al. 1997, Taylor et al. 2001), a sample of adult female polar bears was fitted with satellite radio-collars in Baffin Bay (from both Canada and Greenland) during the 1990s. Some of these bears ($n = 14$) were captured on Baffin and Bylot Islands during autumn 1993 – 1997. Taylor et al. (2005) report that collared bears and their dependent young were often relocated using VHF during the 1990s study period. The probability of locating and recapturing a bear with a collar is likely higher than the probability of recapturing a bear without a collar. Therefore, a radio telemetry covariate, describing whether a bear was wearing a functional radio-collar that could have allowed it to be located by telemetry, was important for explaining variation in recapture probabilities; and all of the most-supported models in the 1990s included a radio telemetry covariate (Taylor et al. 2005). Taylor et al. (2005:209) reported that "The probability of autumn recapture was lower for females and yearling cubs than for adult males and sub-adults, except for radio-collared females and their young" which indicates that radio-collared females were recaptured using radio-location data. Unfortunately, the data archives did not include complete information on which bears were wearing functional radio-collars and located using VHF. Furthermore, in some cases where records could be located, there were inconsistencies among databases and historical hard-copy files. This presented a challenge to MR modeling because the live-capture data in the 1990s were sparse, particularly for adult females, and we anticipated that the additional records for bears likely recaptured using VHF would be important for explaining patterns in survival and recapture probability (see Taylor et al. 2005). To address this issue, we manually reviewed capture histories and covariates compiled for the previous Baffin Bay analysis. We compared these historical files with our available records to identify events in which a bear was likely located via VHF (see also Chapter 5). Based on this, we added 7 recapture events of 5 age 2+ individuals previously in the dataset, and 6 capture events of 5 age 2+ individuals not previously included in the dataset. We believe that this protocol accurately incorporated most of the data for polar bears captured by VHF in the 1990s, although some uncertainty remains given that the original data were not available.

Ramifications of Issues with the 1990s Baffin Bay Data

It is difficult to estimate demographic parameters and detect trends in parameters, for long-lived animals using short time-series of live-encounter data, especially when recapture rates are low, environmental variation is high, and the entire study subpopulation is not exposed to sampling effort on each occasion (Williams et al. 2002). The analyses described above identify specific challenges with 1990s Baffin Bay MR data that arise from both sampling issues and environmental factors. These challenges may lead to bias in estimates of survival and abundance, and ambiguity in the definitions of parameters being estimated (e.g., whether a model is estimating apparent survival, which reflects emigration from the study subpopulation, or true survival).

Survival – A statistical assessment of trends in polar bear survival between the 1990s and 2010s is not possible due to the short duration of live-encounter sampling periods, the large gap between 1990s and 2010s live-encounter sampling, low recapture probabilities, low numbers of dead recoveries, changes in the sampling area between the 1990s and 2010s, and evidence for changes in polar bear movements with respect to the sampling area. This conclusion was supported by computer simulations (T. Arnold, University of Minnesota, unpubl data) in Program MARK to generate datasets that resembled the actual Baffin Bay data but included a known effect (e.g., large reduction in survival), and evaluating the power of MR model to detect such effects (T. Arnold, University of Minnesota, unpublished data). In the context of small and variable sample sizes, a primary challenge for estimating survival is the difficulty of delineating temporary vs. permanent emigration from the study area, and the effects of emigration on estimates of survival. MR modeling was performed using Burnham models, which assume that emigration from the study subpopulation is permanent. Burnham models directly estimate the probability of permanent emigration (*F*) based on patterns in live-encounter data in conjunction with harvest data collected from an area that is larger than the MR sampling area. Under the Burnham model, the survival parameter (S) is technically defined as true survival (i.e., does not include an emigration component). However, research-marked bears that are harvested outside the sampling area may be temporary rather than permanent emigrants (i.e., the bears could have returned to the sampling area in future years, if they had not been killed), and the short duration of the study, small sample sizes, and likely high interannual variability in the probability of being a temporary emigrant (e.g., as related to sea-ice availability) make it difficult to delineate temporary vs. permanent emigration. Simulations suggested that the Baffin Bay data were too sparse to fit Barker models, which relax the assumption that emigration is permanent, and are capable of estimating temporary emigration rates, including non-random temporary emigration. The consequence of using Burnham models either with *F* estimated or with *F* fixed = 1 (i.e., assuming no permanent emigration if *F* is estimated), is that variation across individuals and sampling occasions in the probability of being a temporary emigrant is not explicitly accounted for, and therefore exists as variation in recapture probabilities. Heterogeneity in recapture probabilities has the potential to introduce bias into estimates of S (Schaub et al. 2004). The directionality of bias is often negative and its magnitude tends to increase in the final years of a study (Devineau et al. 2006). Furthermore, non-random patterns in temporary emigration are known to cause bias in estimates of survival (Kendall et al. 1997), and the availability of adult females for capture in the 1990s was related to their multi-year reproductive cycle. Interpretation of trends in survival between the 1990s and 2010s is further complicated because radio-telemetry data suggest changes in fidelity to the MR sampling study area between the epochs, and because the geographic extent of the MR study area itself changed. We conclude that estimates of survival from the current MR analysis of Baffin Bay data must be interpreted with caution. Although estimates of survival provide the basis for discussion and ecological interpretation, they are unlikely to be directly comparable between the 1990s and 2010s, and will require further analysis (e.g., regarding different assumptions about movements between epochs) if used in matrix-type models for subpopulation projections.

Abundance – Estimating abundance is one of the more difficult challenges in wildlife management (Williams et al. 2002). Deriving accurate estimates of abundance and evaluating trends in abundance over time require an appropriate study design and, especially, consistent distribution of sampling effort in time and space. In the current study, the difference between the distributions of captures in the 1990s and 2010s suggest that the sampling area on Baffin Island expanded substantially from the 1990s to the 2010s. Specifically, sampling was spatially restricted to a portion of the subpopulation’s fall range during the 1990s, thus excluding bears with seasonal fidelity to inland areas. Furthermore, an unknown but potentially significant portion of the Baffin Bay subpopulation may not have been exposed to sampling in the 1990s due to the higher presence of sea ice, which some bears used throughout the year rather than coming onto land. We conclude that the abundance estimate in the 2010s, based on MR data from the entire sampling area, is not directly comparable to the previous 1990s abundance estimate. To investigate the extent to which differences in sampling affected abundance estimates from the 1990s and 2010s, we used the 1990s sampling area to create a subset of the 2010s data, and subsequently derived a 2010s abundance estimate based on this restricted subset of the data. We included only those 2011 – 2013 capture events that were located within the estimated 1990s sampling frame and completed supplemental demographic analyses (see Chapter 5). This analysis helped evaluate the potential biases associated with the more restricted area of onshore sampling on Baffin Island in the 1990s. However, it did not address the potential effects of polar bears using the sea ice in the 1990s. When there is temporary emigration from the sampling area, estimates of abundance from Burnham models represent the "superpopulation" (defined as all animals with a probability of moving through the sampling area, even if not every animal was actually in the sampling area on every sampling occasion). If temporary emigration from the sampling area is completely random, it will not introduce bias into estimates of abundance. However, nonrandom temporary emigration (e.g., if some individuals are often or always temporary emigrants) has a similar effect on estimates of demographic parameters from MR models as un-modeled heterogeneity in recapture probability, and generally introduces negative bias into estimates of abundance (Kendall et al. 1997).

MR model covariates – 1990s sampling bias may also impact the individual, geographic fidelity covariate (proximity to smoothed coastline). Analyses did not suggest a significant relationship between initial and subsequent capture locations in the 1990s, but this may be due to sampling (e.g., not enough effort was expended inland, to identify animals with fidelity to inland areas). The relationship is driven by the 2010s data. Also, the radio telemetry covariate may be biased in some unknown direction due to the uncertainty as to whether the subsequent capture of a collared bear was facilitated by the radio tracking. Sensitivity analyses outlined above may help better understand potential biases. Given the differences between the 1990s and 2010s, including epoch effects for the binary 'proximity to smoothed coastline' is important.

3.2. Literature Cited

Born, E.W. and R. Dietz. 2009. Polar bear *Ursus maritimus*. Pages 81-88 *in* D. Boertmann, A. Mosbech, D. Schiedek, and K. Johansen, editors. The Eastern Baffin Bay. A preliminary strategic environmental impact assessment of hydrocarbon activities in the KANUMAS

West area. NERI Technical Report No. 720, National Environmental Research Institute, Aarhus University, Denmark. http://www.dmu.dk/Pub/FR720.pdf.

Born, E. W., A. Rosing-Asvid, and D. Andriashek. 1992. Tagging of polar bears in NW Greenland, 2-19 May 1992. Report on field work. Unpublished report, Greenland Fisheries Research Institute, Copenhagen, Denmark, 6 pp. Available at Greenland Institute of Natural Resources, P.O. Box 570, 3900 Nuuk, Greenland.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Devineau, O., R. Choquet, and J-D. Lebreton. 2006. Planning capture-recapture studies: straightforward precision, bias, and power calculations. Wildlife Society Bulletin 34:1028–1035.

Ferguson, S. H., M. K. Taylor, E. W. Born, A. Rosing-Asvid, and F. Messier. 1999. Determinants of home range size in polar bears. Ecology Letters 2:311–318.

Ferguson, S. H., M. K. Taylor, E. W. Born, A. Rosing-Asvid, and F. Messier. 2001. Activity and movement patterns of polar bears inhabiting consolidated versus active pack ice. Arctic 54:49–54.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use by polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock's robust design. Ecology 78:563–578.

Lukacs, P. M., and K. P. Burnham. 2005. Review of capture–recapture methods applicable to noninvasive genetic sampling. Molecular Ecology 14:3909–3919.

Peñaloza, C. L., W. L. Kendall, and C. A. Langtimm. 2014. Reducing bias in survival under nonrandom temporary emigration. Ecological Applications 24:1155-1166.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:3–97.

Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117–127.

Rosing-Asvid, A. 1993. Tagging of polar bears in NW Greenland, May 1993. Report on field work. Unpublished report, Greenland Fisheries Research Institute, Copenhagen, Denmark, 8 pp. Available at Greenland Institute of Natural Resources, P.O. Box 570, 3900 Nuuk, Greenland.

Royle, J. A., R. B. Chandler, R. Sollmann, and B. Gardner. 2014. Spatial Capture-Recapture. First edition. Academic Press, Waltham, Massachusetts, USA.

Schaub, M., O. Gimenez, B. R. Schmidt, and R. Pradel. 2004. Estimating survival and temporary emigration in the multistate capture–recapture framework. Ecology 85:2107–2113.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA.

Table 3.1. Summary table of live captures and dead recoveries during the mark-recapture study of the Baffin Bay polar bear subpopulation in Nunavut, Canada, and Greenland, 1993 – 2010. Shaded cells indicate that data were not possible due to an absence of marking or recapture.

	Initial captures						**Live recaptures**				**Dead recoveries**					
	Females			*Males*			*Females*		*Males*		*Females*			*Males*		
	Coy	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Yrl**	**2+**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**
1993	14	8	53	12	8	61					0	0	1	0	0	0
1994	26	13	65	16	9	77	0	5	0	14	0	0	3	0	0	7
1995	15	11	62	19	11	85	4	11	4	23	0	2	6	1	0	8
1996												1	8		0	7
1997	22	10	60	19	13	113		20		31	0	0	6	0	1	9
1998												0	3		0	11
1999													3			9
2000													0			8
2001													2			8
2002													0			11
2003													0			7
2004													1			7
2005													2			3
2006													3			6
2007													1			2
2008													2			4

2009													2			0
2010													0			1
2011	2	23	163	1	20	148		5		5	0	0	4	0	0	20
2012	40	30	221	35	30	192	3	41	0	54	0	0	8	0	2	14
2013	28	15	121	16	15	90	4	48	5	55	0	1	8	1	0	20
Totals	**147**	**110**	**745**	**118**	**106**	**766**	**11**	**130**	**9**	**182**	**0**	**4**	**63**	**2**	**3**	**162**

Table 3.2. Metrics for adult females satellite collared in the 1990s (fall) and 2010s (spring) for the distance inland from the outer Baffin Island coast. Distance is reported in km.

		August			**September**			**October**		
Adult Female Accompanied by	**N**	**Mean distance inland**	**SD**	**Count of locations**	**Mean distance inland**	**SD**	**Count of locations**	**Mean distance inland**	**SD**	**Count of locations**
1990s										
2YR	1							5.6	4.4	3
AM	0									
COY	15	10.8	14.2	10	19.7	15.1	13	9.6	5.9	46
YRL	12	6.0	5.6	3	18.0	17.8	25	8.4	5.8	41
ALONE	5				8.8	9.4	6	13.4	11.8	8
2010s										
2YR	5	13.5	9.3	25	27.0	8.3	20	16.2	13.5	11
AM -in spring	2	27.1	10.2	9	32.6	12.0	5	35.1	9.3	4
COY	2	5.5	4.8	3	7.5	4.3	13	3.6	4.3	11
YRL	7	3.5	5.0	20	6.8	6.2	33	6.2	7.3	27
ALONE	6	11.9	10.8	25	16.3	7.6	22	14.4	7.8	18

Table 3.3. Time periods when the BB fall sampling period occurred in each decade. These dates were used to asses if independent bears were in or out of the sampled area.

Year of sampling	Start	End
1993	23 August	8 October
1994	7 September	19 October
1995	17 September	19 October
1996	n/a	n/a
1997	21 September	29 October
2011	4 September	14 October
2012	26 August	29 September
2013	20 August	11 October

Table 3.4. The overall fraction of independent collared adult female (AF) bears found in the sampling range by year.

Year of sampling	n independent collared bears	n independent AF bears in the sampled area (minimum of n=1 location during date range)	% independent AF bears in the sampled area for each decade
1993	13	3	23
1994	5	1	20
1995	1	0	0
1997	0		
2011	12	8	67
2012	13	11	85
2013	6	4	67

Table 3.5. Independent BB adult female bears with satellite collars transmitting during the MR sample periods. Bears listed are only those that used the sampled area on Baffin Island for each decade. The fraction of locations inside the sampled area is shown for each bear.

YEAR + capture season	ID (PTT + Year)	Start Date	End Date	n independent bears during this year	n independent bears in the sampled area	Fraction of total locations inside sampled area during the sampling dates	Proportion of locations
1993		**23-Aug**	**8-Oct**	**13**	**3**		
fall	199111062	27-Aug	16-Sep			1/4	0.25
spring	19922718	24-Aug	7-Oct			1/8	0.13
fall	19922700	25-Aug	25-Aug			1/1	1.00
1994		**7-Sep**	**19-Oct**	**5**	**1**		
spring	19922701	8-Sep	6-Oct			2/6	0.33
2011		**4-Sep**	**14-Oct**	**12**	**8**		
spring	201068010	6-Sep	8-Oct			8/9	0.89
spring	2011105814	24-Sep	10-Oct			4/4	1.00
spring	201074768	6-Sep	12-Oct			8/8	1.00
spring	2011105809	6-Oct	6-Oct			1/1	1.00
spring	200974767	6-Sep	12-Oct			10/10	1.00
spring	2011105817	4-Sep	14-Oct			10/10	1.00
spring	2011105816	4-Sep	14-Oct			5/5	1.00

spring	200968005	6-Sep	12-Oct			10/10	1.00
2012		**26-Aug**	**29-Sep**	**13**	**11**		
spring	201074774	29-Aug	26-Sep			6/7	0.86
spring	2012105829	29-Aug	26-Sep			5/7	0.71
spring	201068010	12-Sep	24-Sep			2/3	0.67
spring	2011105814	29-Aug	26-Sep			5/7	0.71
spring	201074768	27-Aug	28-Sep			7/8	0.88
spring	2011105808	29-Aug	26-Sep			7/8	0.88
spring	2011105809	6-Sep	6-Sep			1/1	1.00
spring	200974767	27-Aug	28-Sep			1/9	1.90
spring	200974771	29-Aug	26-Sep			8/8	1.00
spring	2011105813	29-Aug	22-Sep			2/6	0.33
spring	200968005	27-Aug	27-Aug			1/1	1.00
2013		**20-Aug**	**11-Oct**	**6**	**4**		
spring	2013105818	20-Aug	11-Oct			12/12	1.00
spring	2013128265	20-Aug	11-Oct			14/14	1.00

Figure 3.1. Locations of polar bears sampled in Baffin Bay during the 1990s (August – October, 1993 – 1995, 1997, red) and 2010s (August – October, 2011 – 2013, blue). Sampling in Greenland in the 2010s occurred near Melville Bay but is not shown. Note the absence of captures in fjords on Baffin Island during the 1990s in the inset.

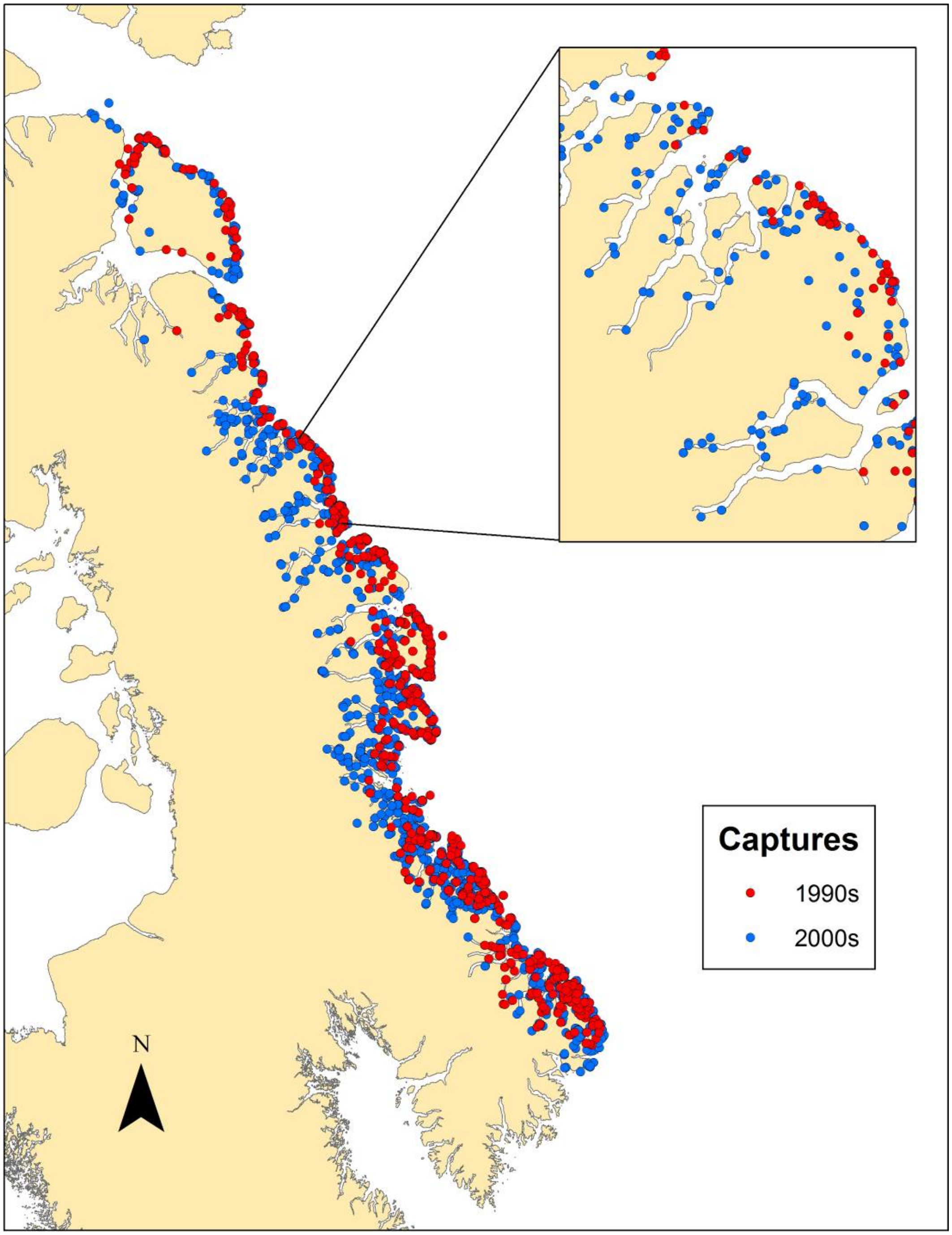


Figure 3.2. Distances independent bears were captured from the smoothed coastlines of Baffin and Bylot Islands during fall-time sampling in the Baffin Bay subpopulation, 1993 – 1997 and 2011 – 2013.

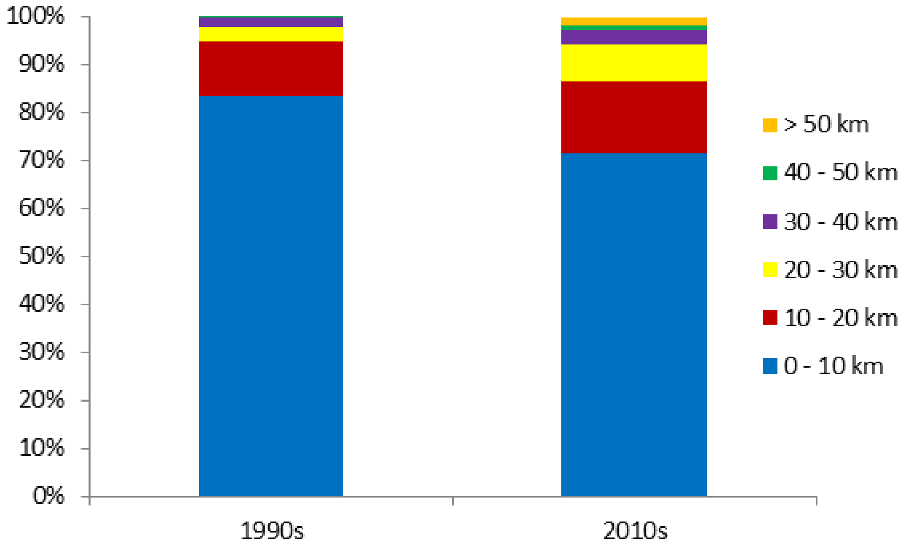


Figure 3.3. Distances independent bears were captured from the true coastlines of Baffin and Bylot Islands during fall-time sampling in the Baffin Bay subpopulation, 1993 – 1997 and 2011 – 2013.

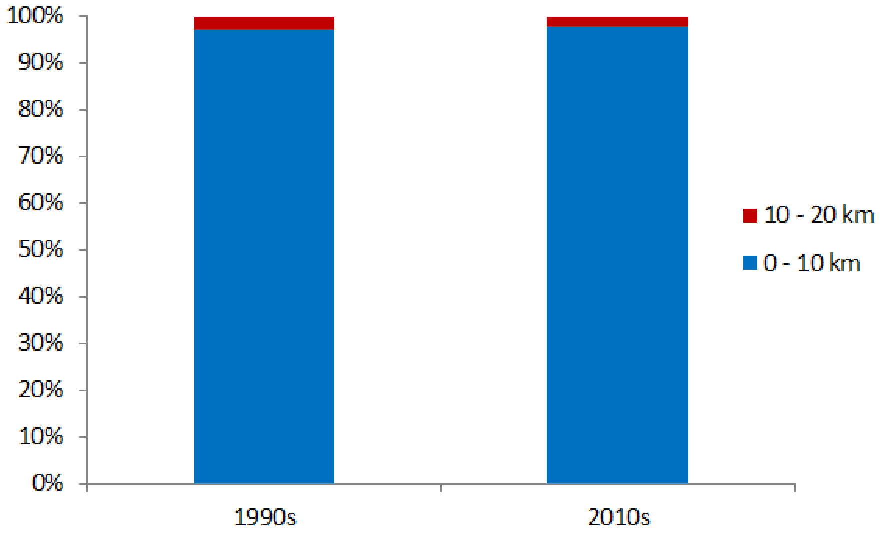


Figure 3.4. Distance to smoothed Baffin Island coastline shown in all summer months using satellite telemetry data from adult females in the 1990s (red) and 2010s (blue) located on Baffin Island. Shaded regions represent 2 SE from the mean. Numbers above represent numbers of telemetry locations for each month. There was no difference in distance inland (or distance to the outer Baffin Island coast) between decades.

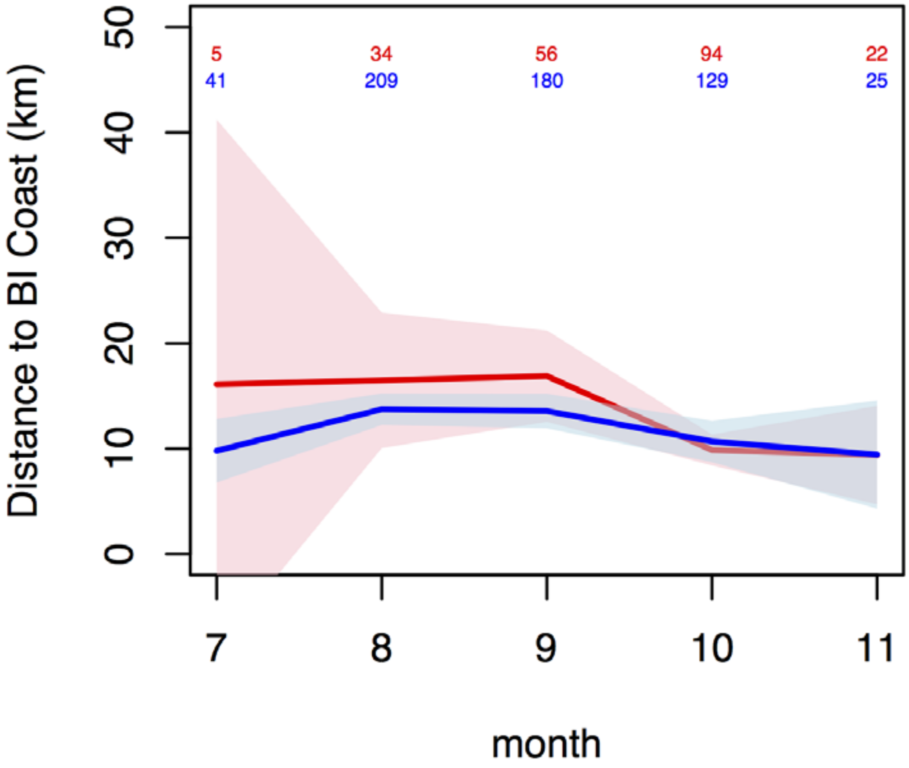


Figure 3.5. Elevation of adult female polar bears on Baffin Island shown in all summer months using satellite telemetry data from the 1990s (red) and 2010s (blue). Shaded regions represent 2 SE from the mean. Numbers above represent numbers of telemetry locations for each month. There was no difference in elevations used by polar bears across months between decades.

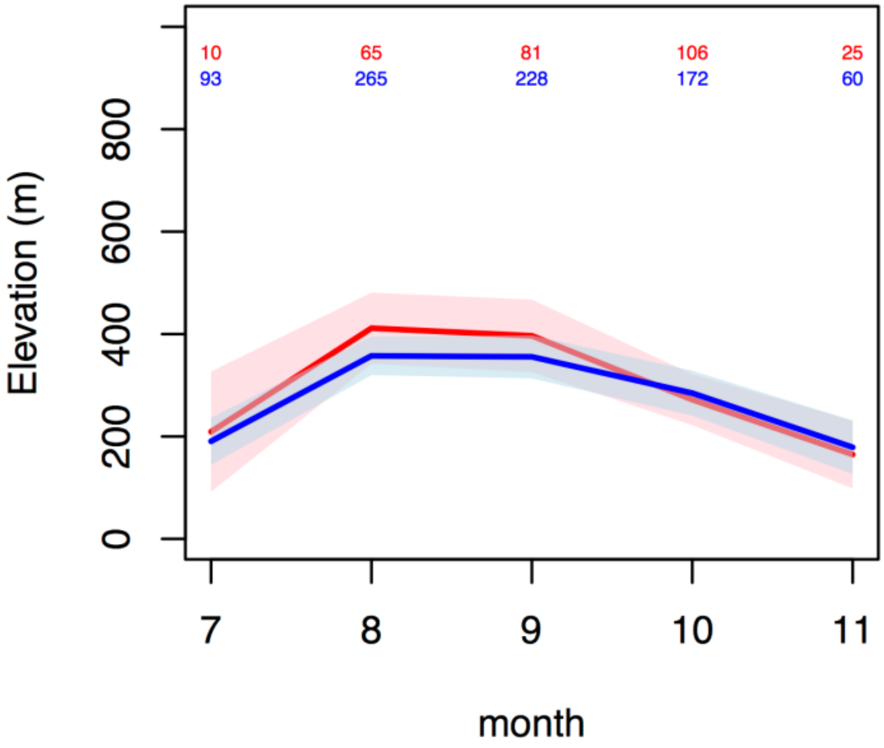


Figure 3.6. The delineation of the sampled area shown with a red outline for the 1990s with capture locations collected during the MR sampling.

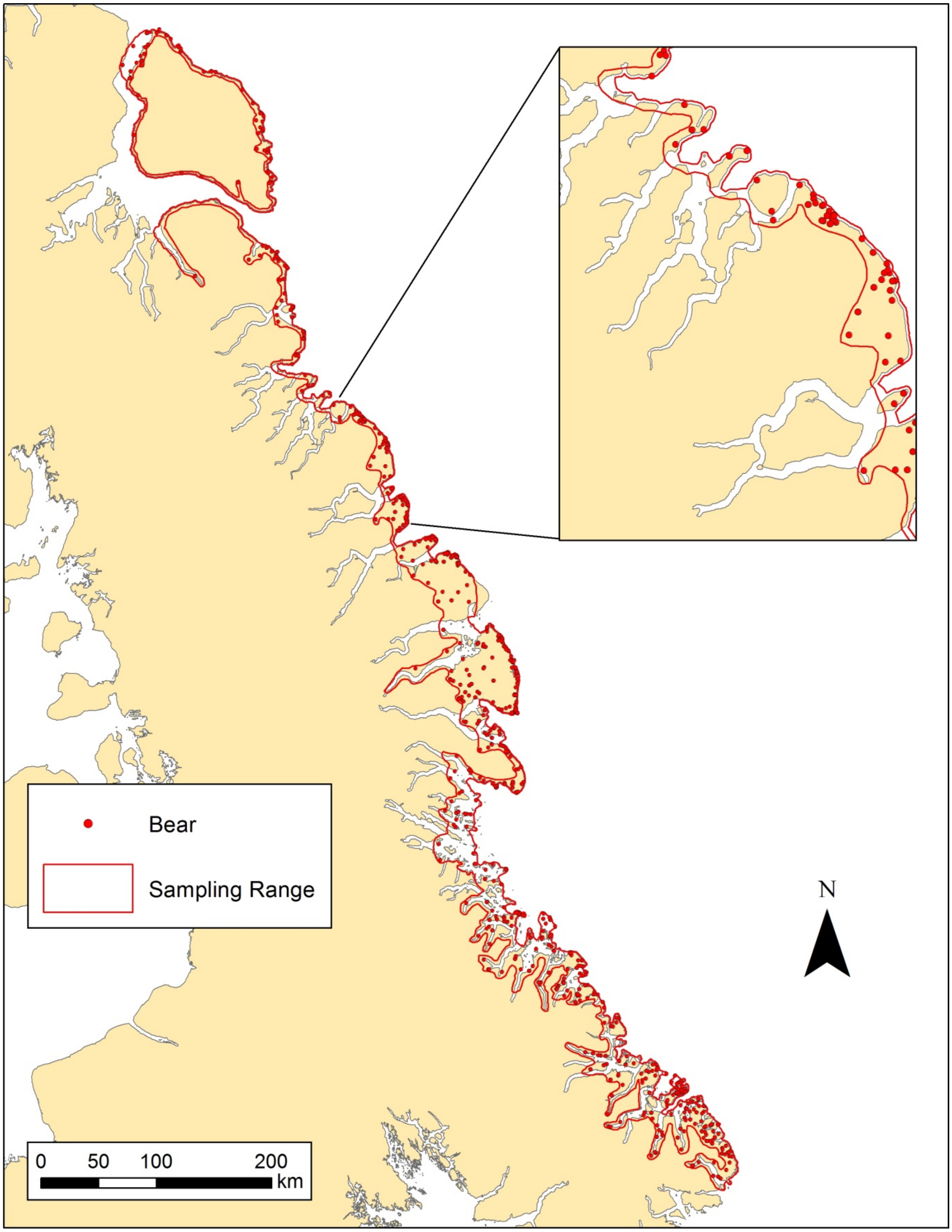


Figure 3.7. The delineation of the sampled area shown with a red outline for the 2010s with biopsy locations collected during the MR sampling (2011-2013).

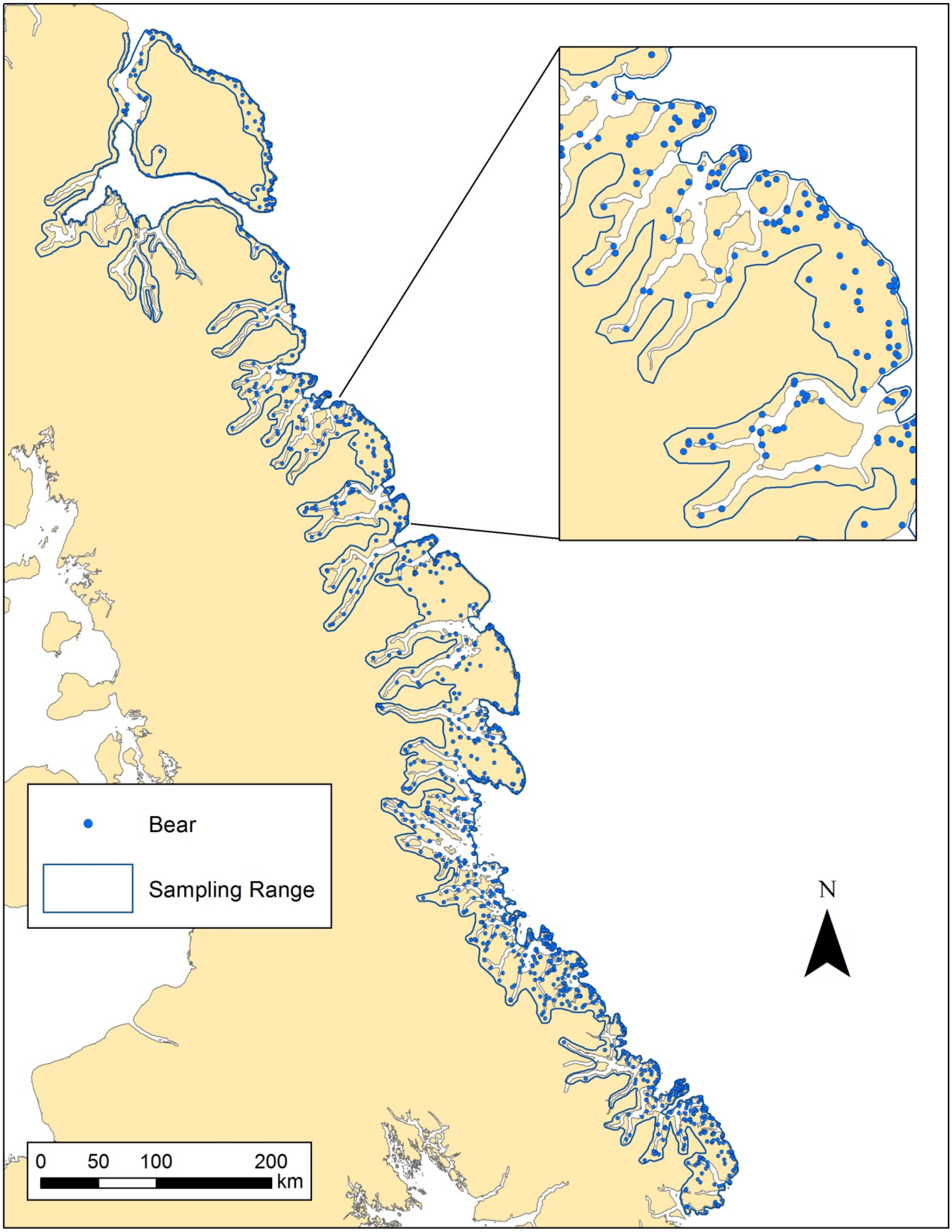


Figure 3.8. Geographic sampling ranges for the MR in the 1990s and 2010s.

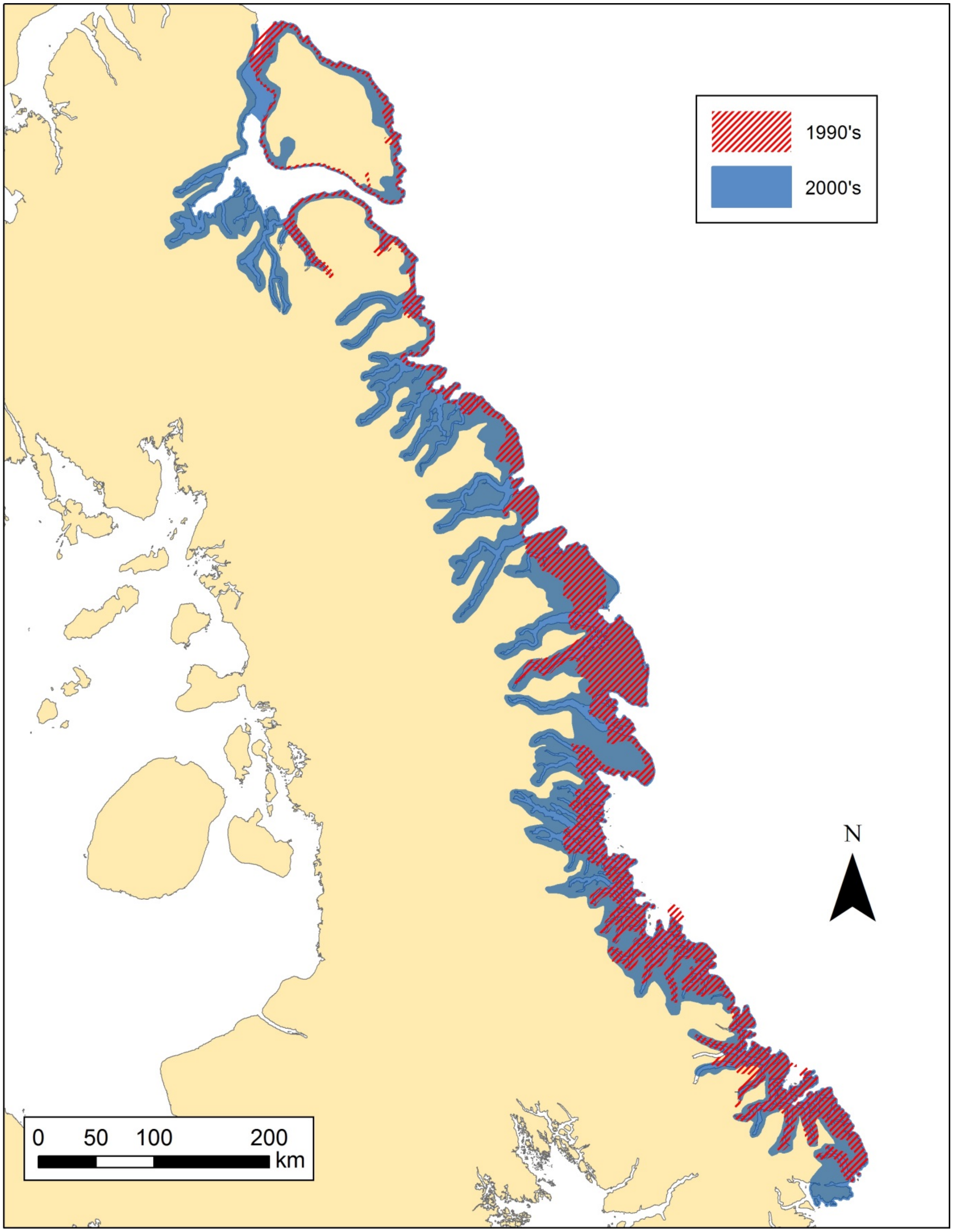


Figure 3.9. All telemetry locations from independent adult female bears with satellite collars transmitting during the 1993 sampling period dates (See Table 3.3). The 1990s sampled area for the MR study is shown in the red outline. Bears in central BB are on sea ice (Figure 3.15).

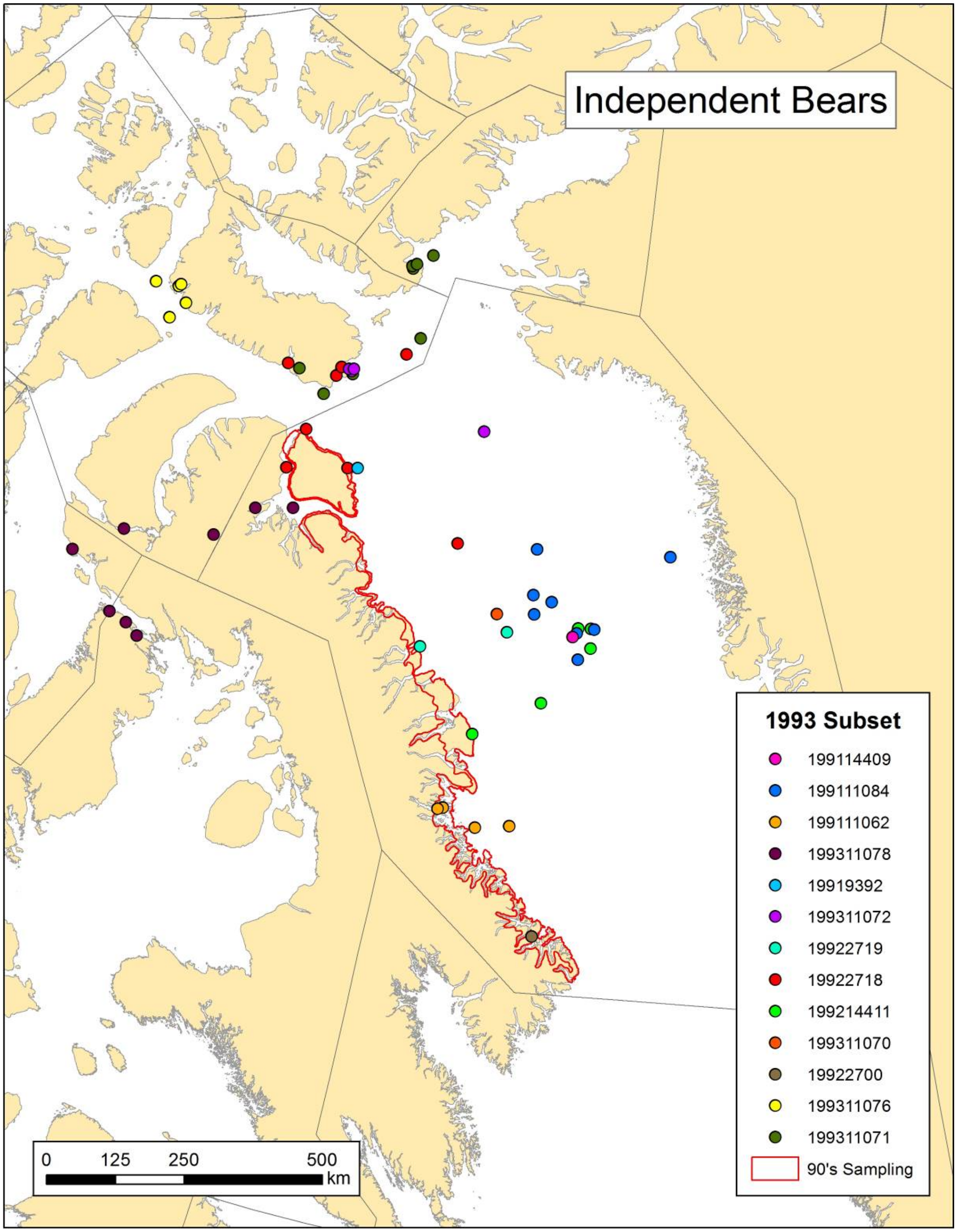


Figure 3.10. All telemetry locations from independent adult female bears with satellite collars transmitting during the 1994 sampling period dates (See Table 3.3). The 1990s sampled area for the MR study is shown in the red outline.

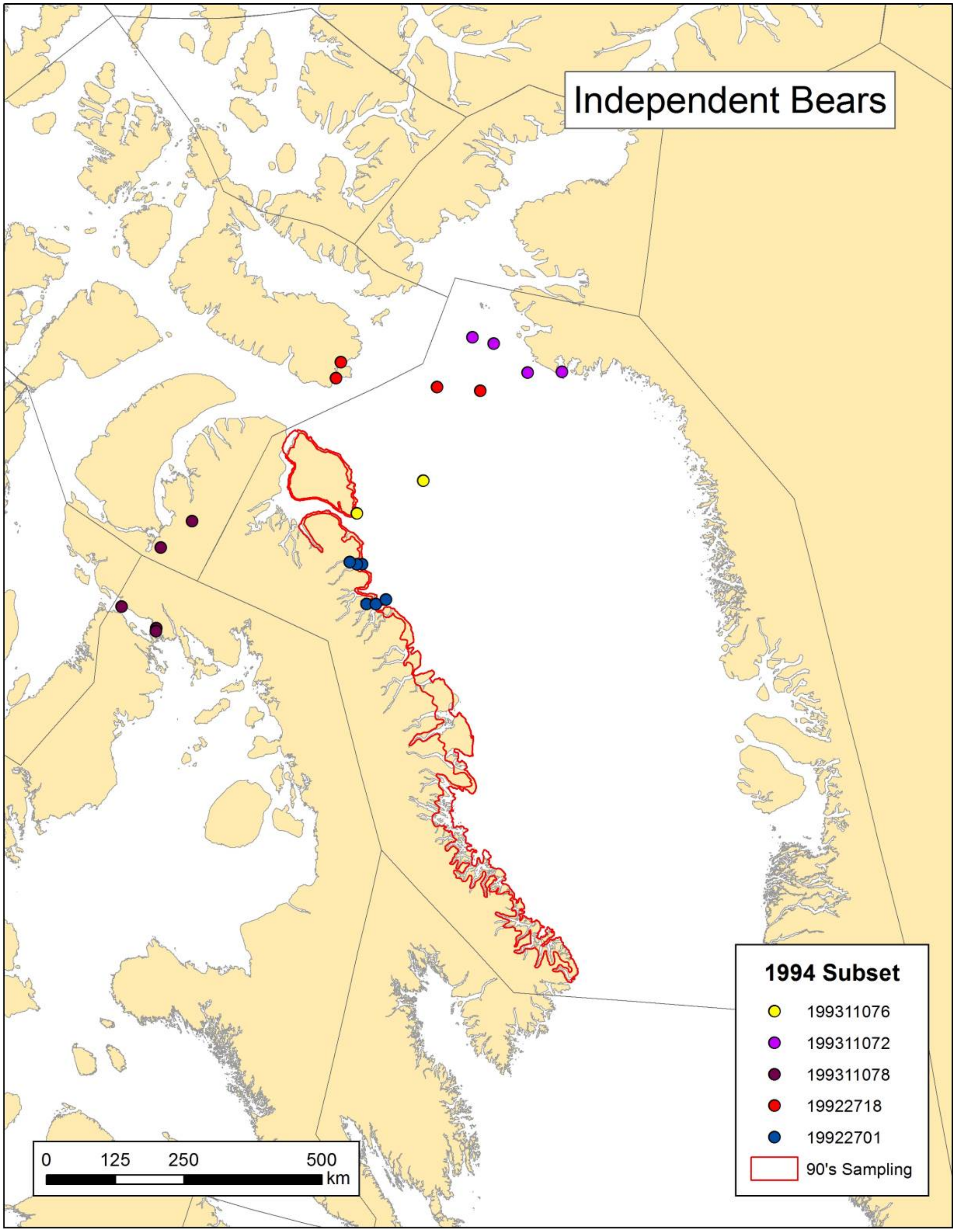


Figure 3.11. All telemetry locations from independent adult female bears with satellite collars transmitting during the 1995 sampling period dates (See Table 3.3). The 1990s sampled area for the MR study is shown in the red outline.

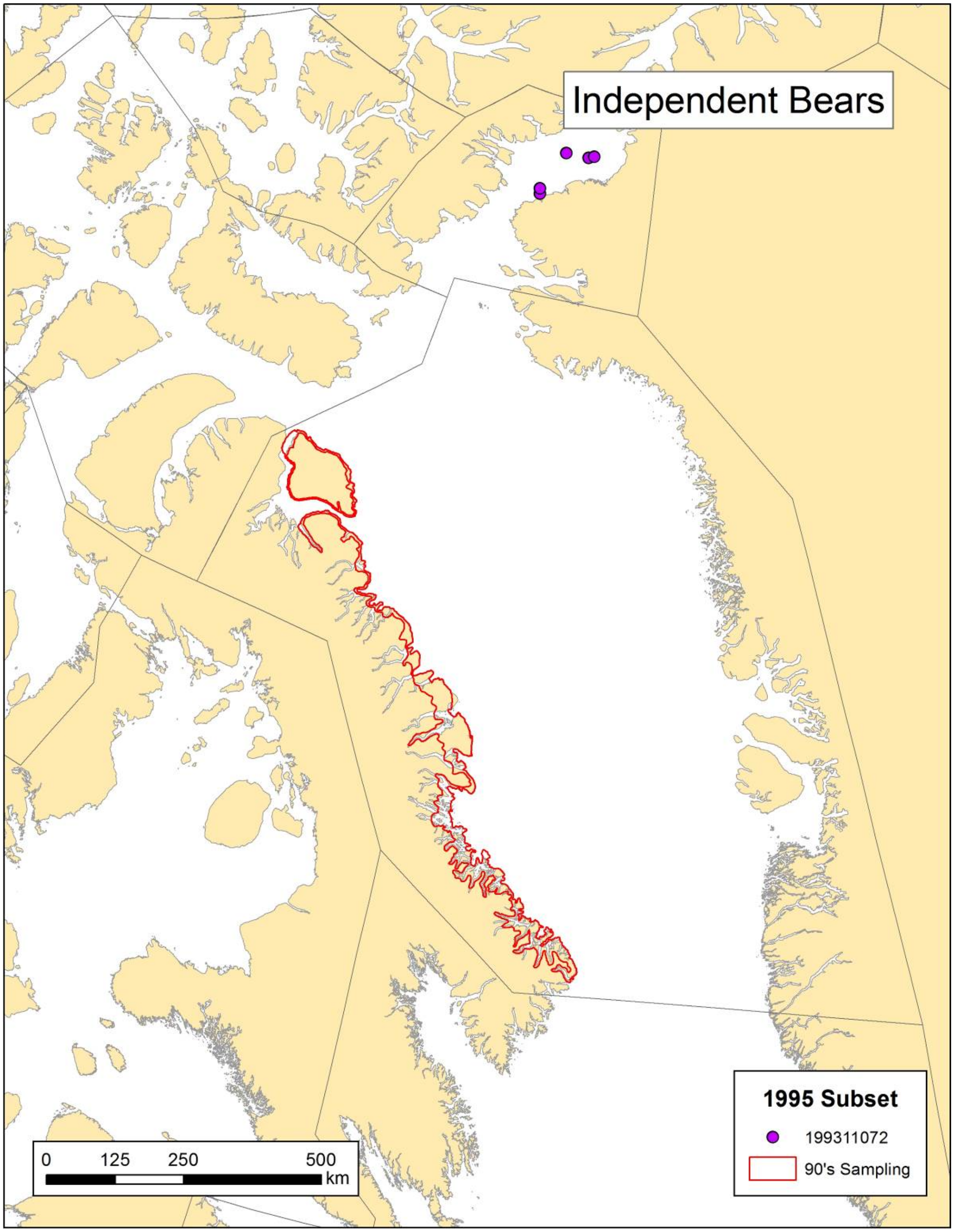


Figure 3.12. All telemetry locations from independent adult female bears with satellite collars transmitting during the 2011 sampling period dates (See Table 3.3). The 2010s sampled area for the MR study is shown in the blue outline.

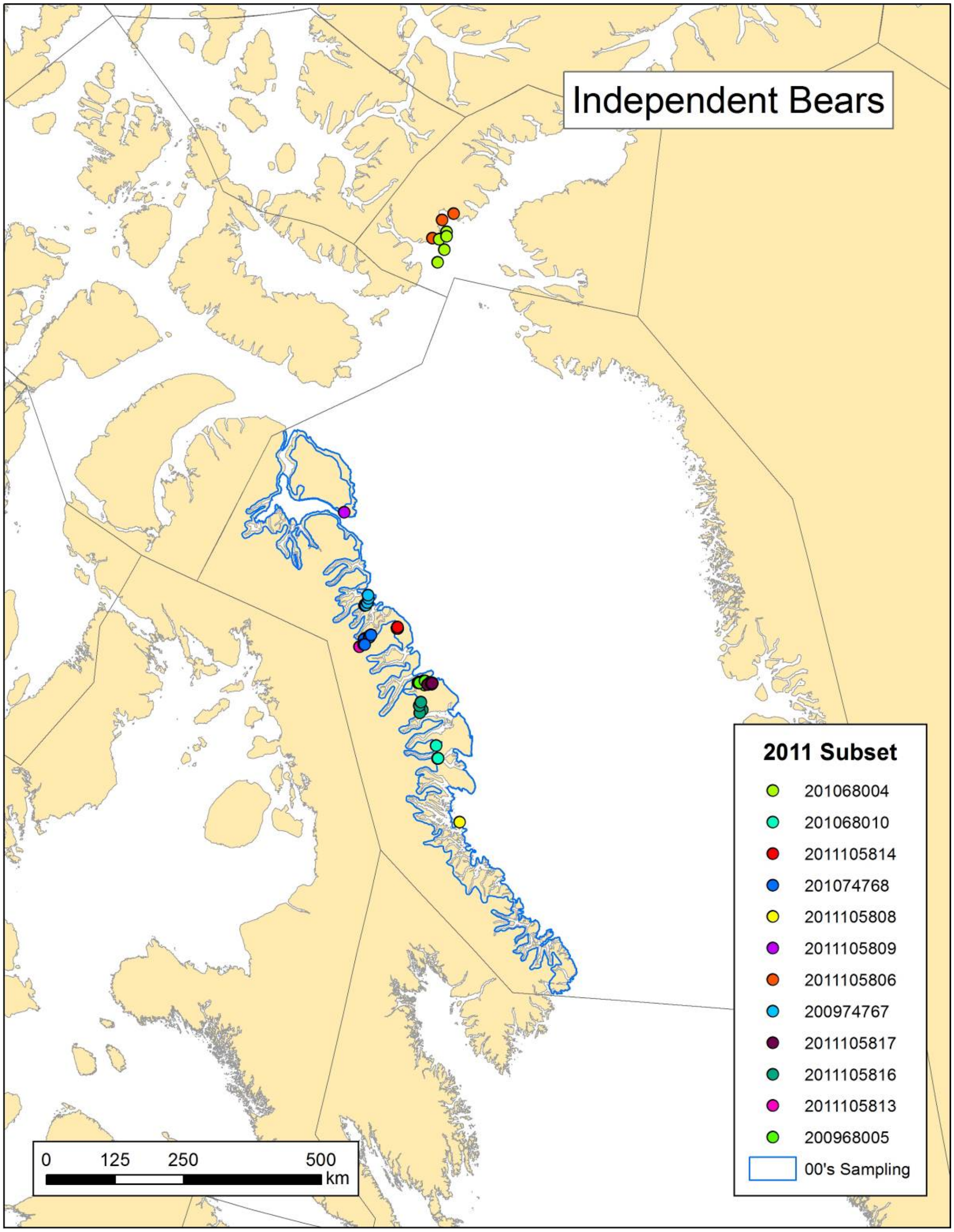


Figure 3.13. All telemetry locations from independent adult female bears with satellite collars transmitting during the 2012 sampling period dates (See Table 3.3). The 2010s sampled area for the MR study is shown in the blue outline.

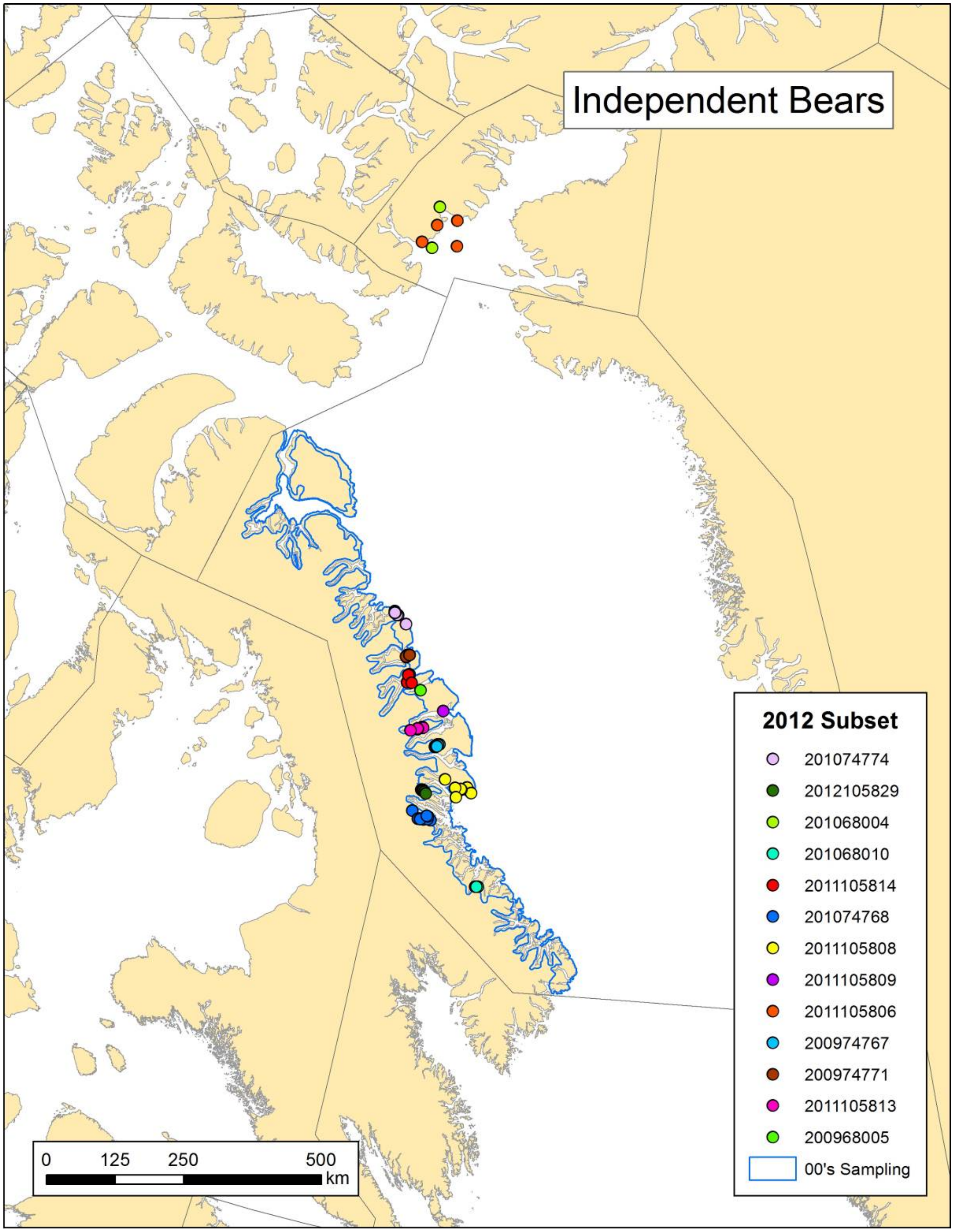


Figure 3.14. All telemetry locations from independent adult female bears with satellite collars transmitting during the 2013 sampling dates (See Table 3.3). The 2010s sampled area for the MR study is shown in the blue outline.

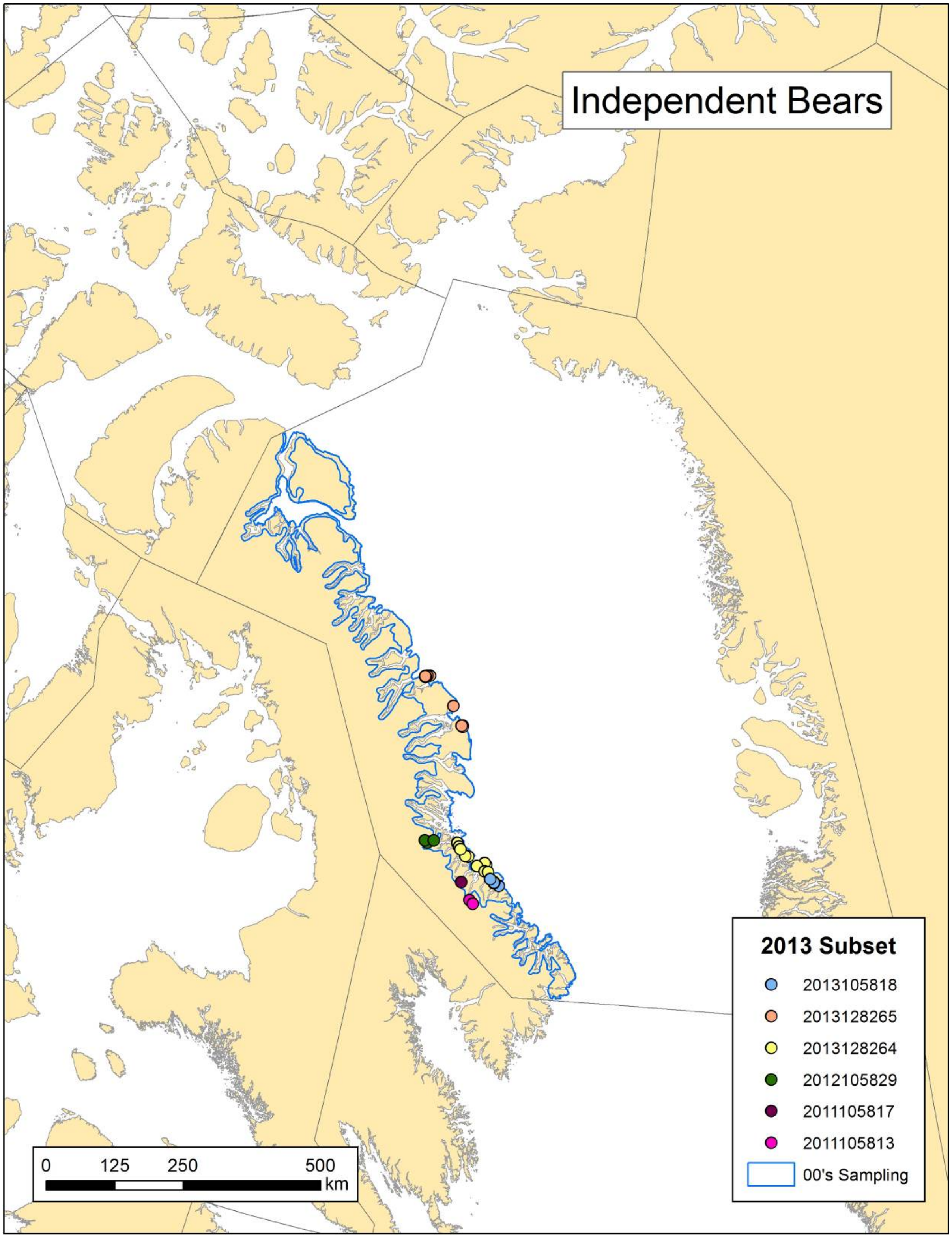


Figure 3.15. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 1993 (August week 4). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 1993 sampling period are shown.

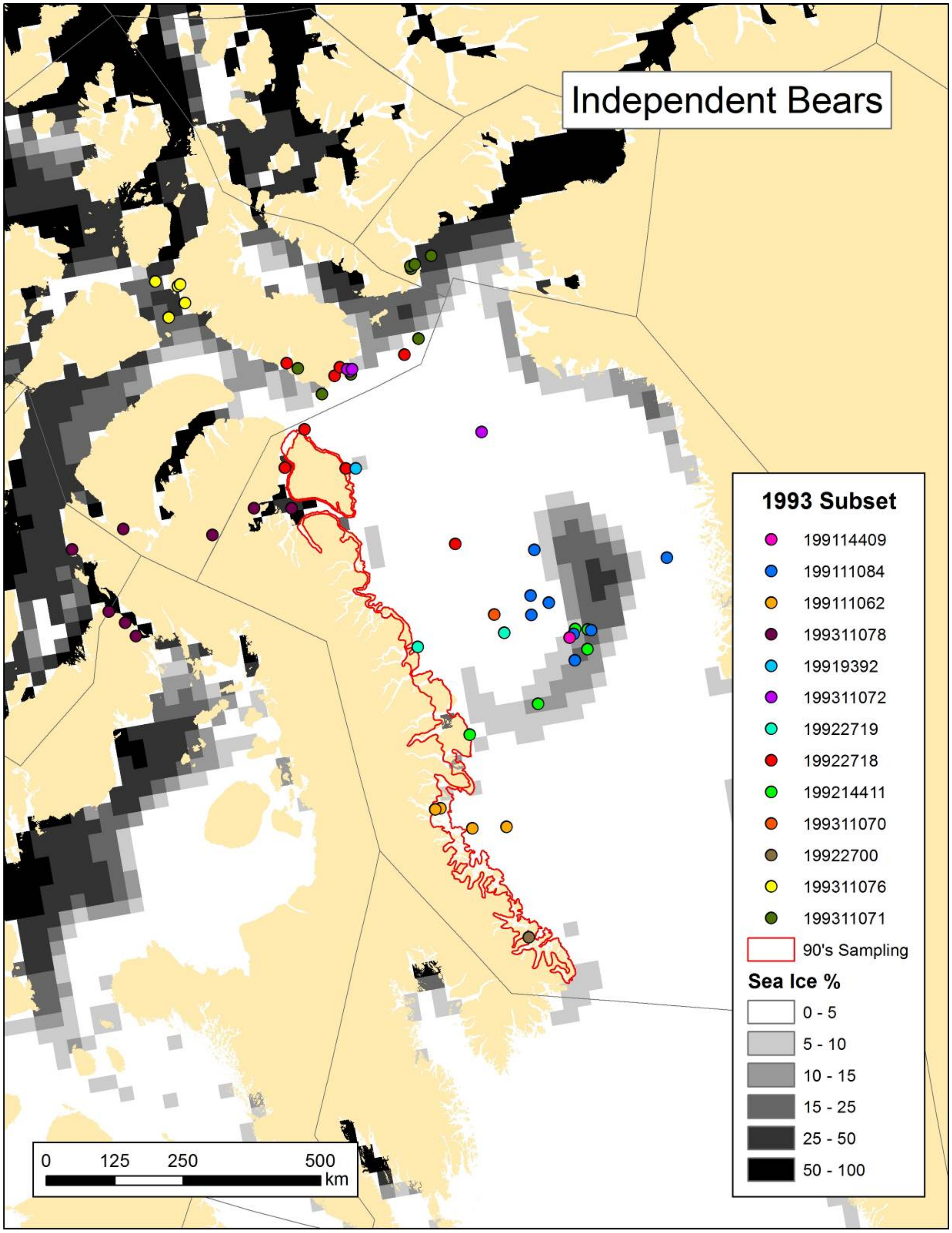


Figure 3.16. Distribution of weekly mean sea ice concentrations (SSMI) during the mid-point of the sampling period in 1994 (October week 1). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 1994 sampling period are shown.

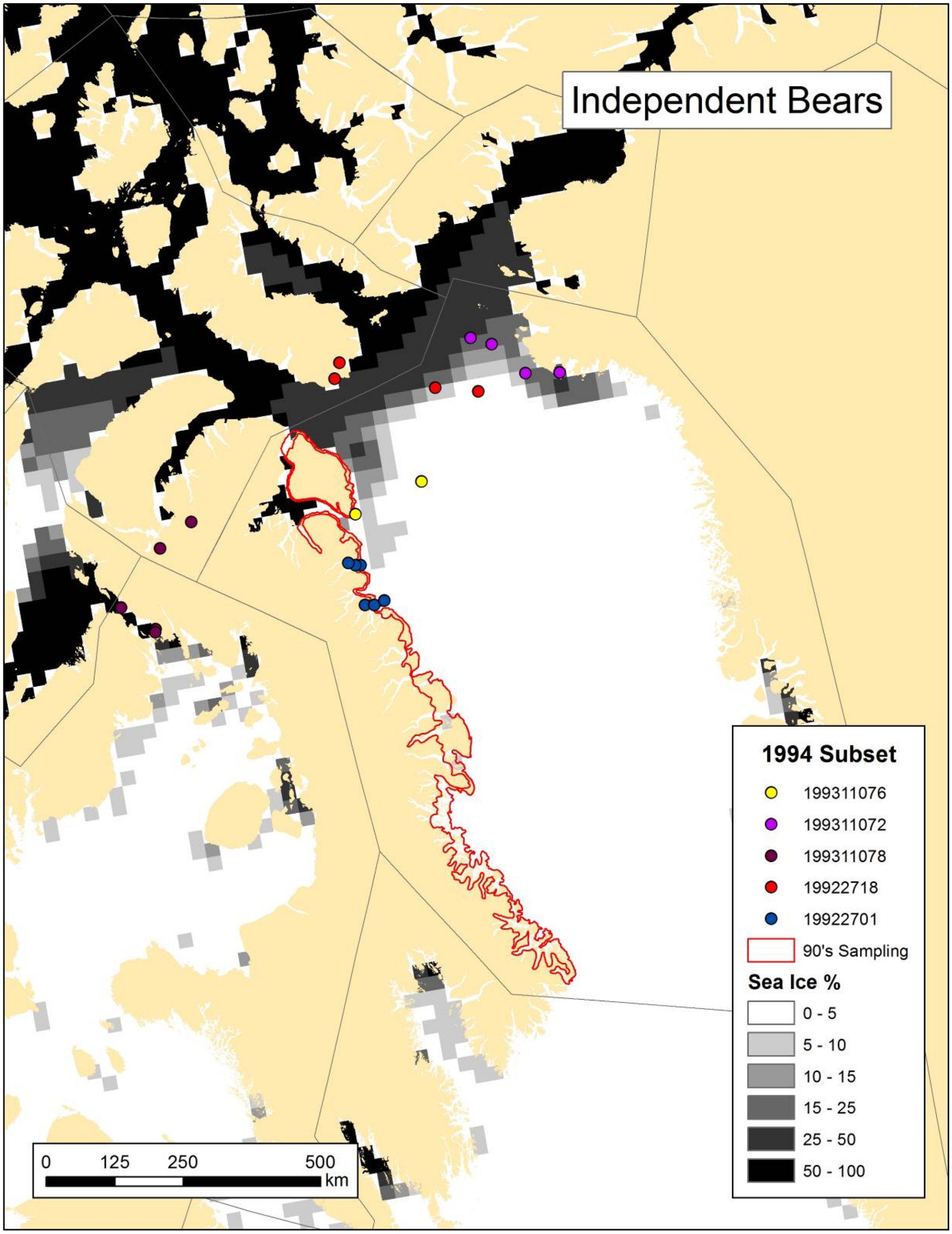


Figure 3.17. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 1995 (October week 2). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 1995 sampling period are shown.

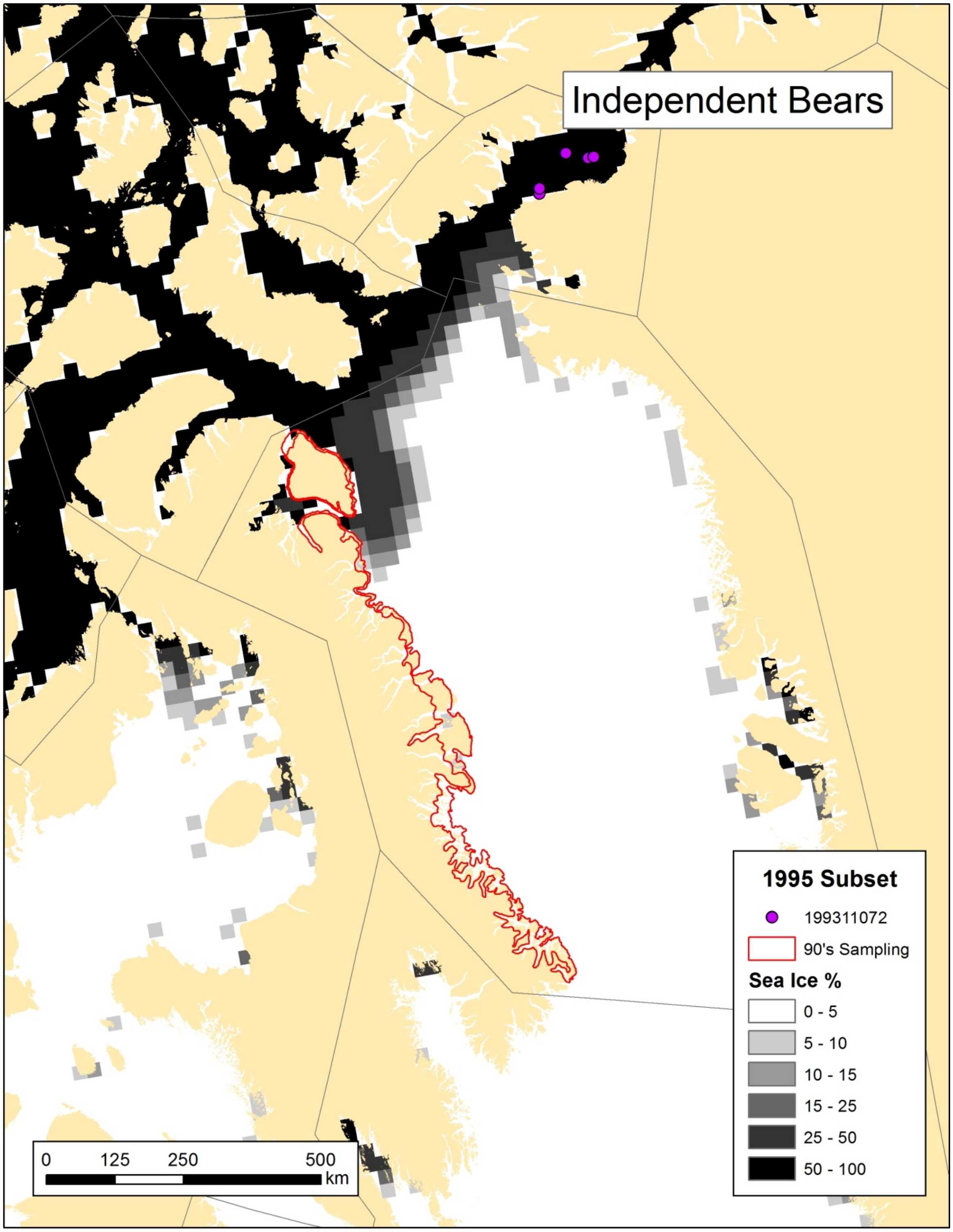


Figure 3.18. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 2011 (September week 3). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 2011 sampling period are shown.

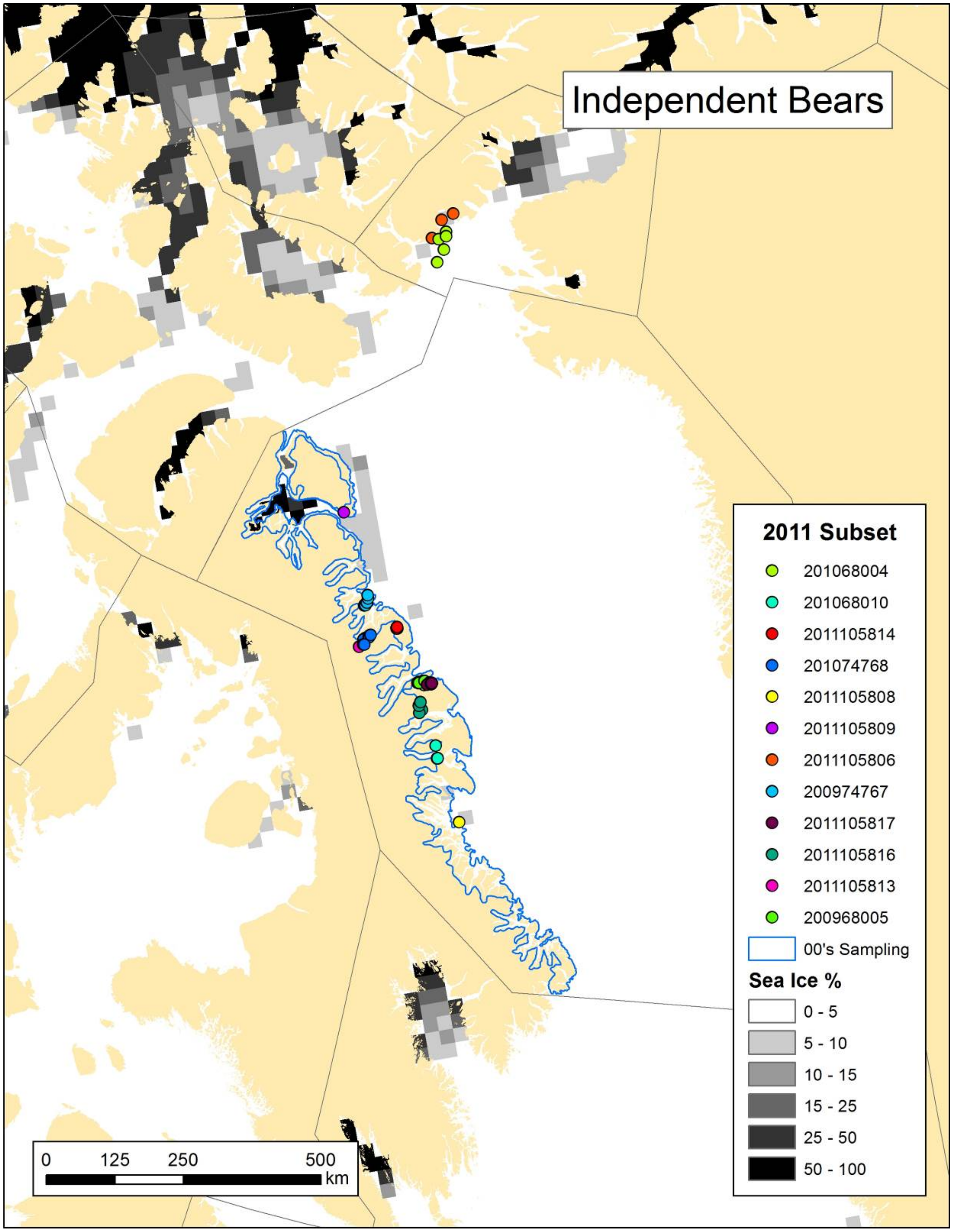


Figure 3.19. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 2012 (September week 2). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 2012 sampling period are shown.

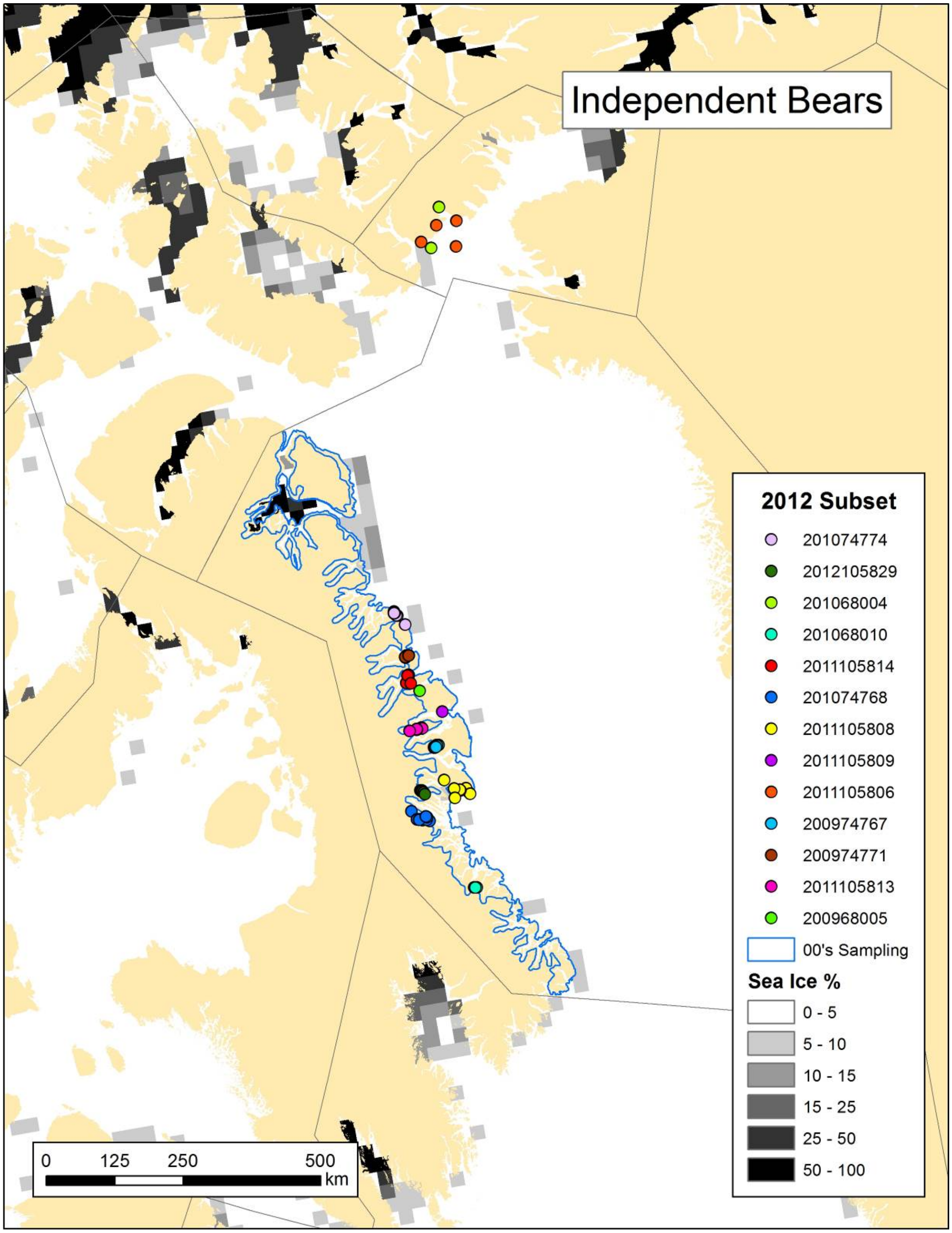


Figure 3.20. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 2013 (September week 3). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 2013 sampling period are shown.

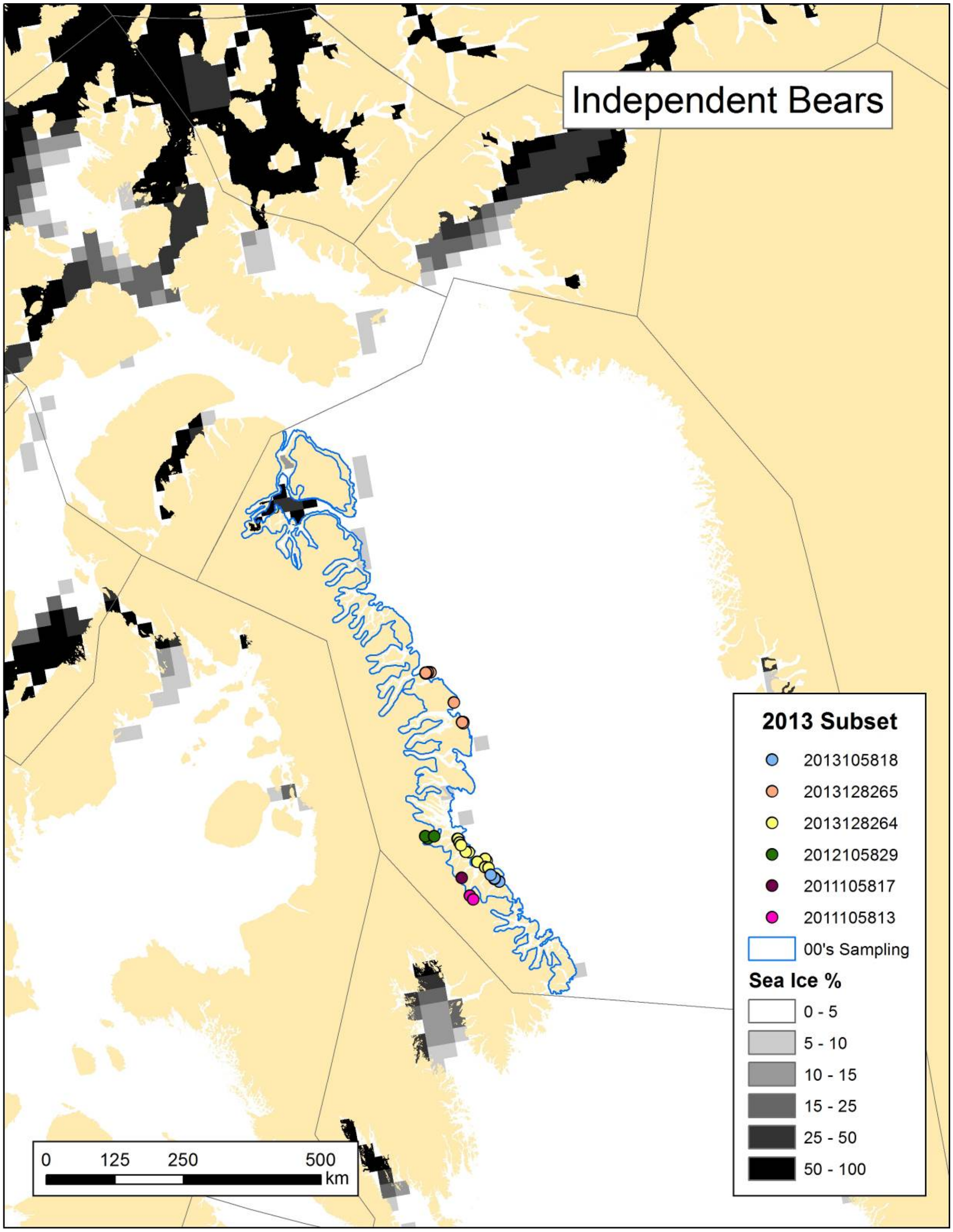
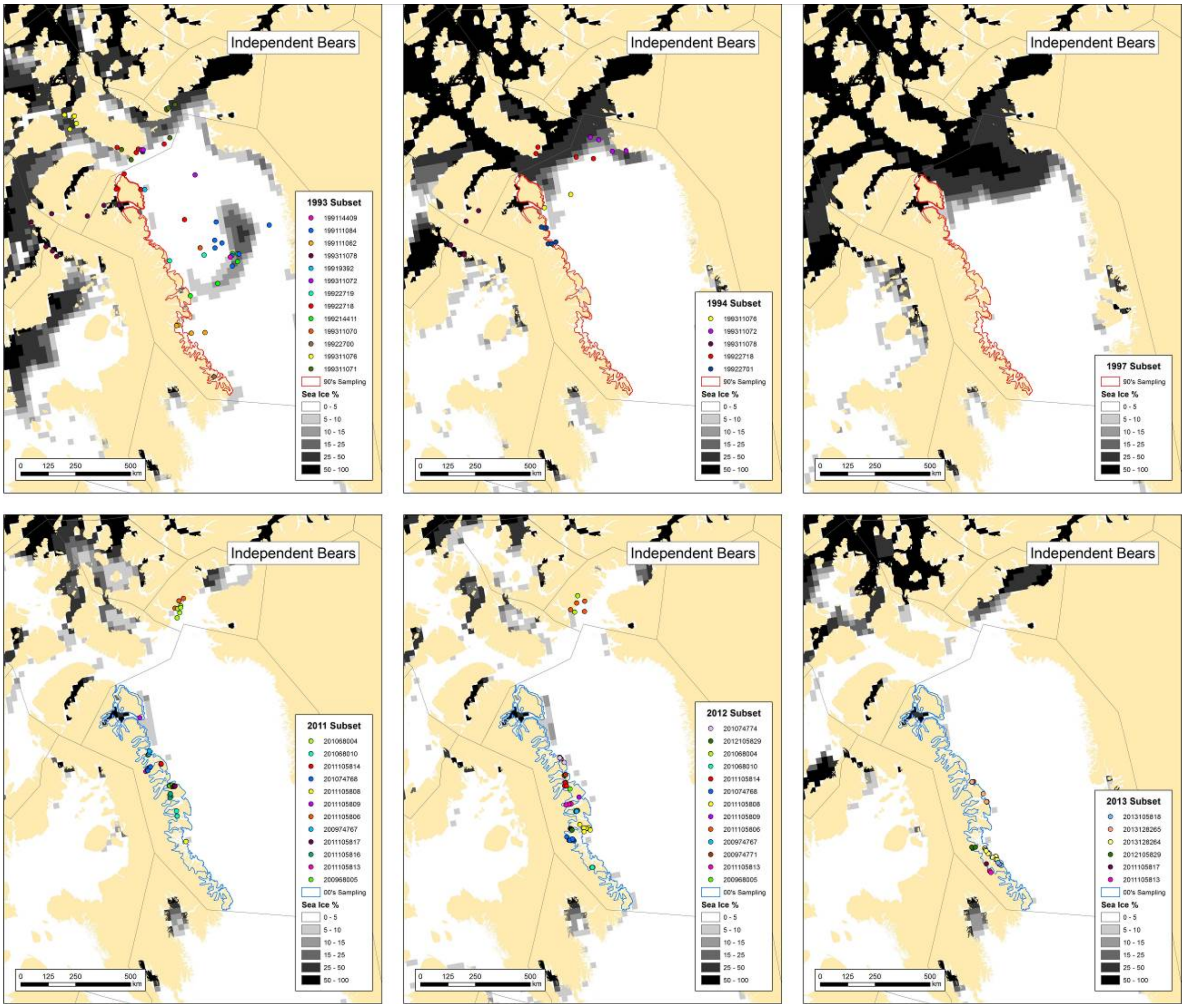


Figure 3.21. Distribution of sea-ice conditions (SSMI) during 1990s MR (top left to right 1993, 1994 and 1997) and 2010s MR (bottom left to right 2011, 2012, and 2013). Independent bears transmitting during the sampling are shown for reference. Note sampling occurred in 1997 but there were no independent collared bears for assessment of presence in the sampling area.



CHAPTER 4

HABITAT AND HABITAT USE IN BAFFIN BAY

KEY FINDINGS

- Sea-ice habitat in BB has significantly declined over the period of satellite observations, especially since the mid-1990s. The length of summer (number of days from sea-ice retreat in spring to sea-ice advance in fall) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 4 percent/decade. The general pattern of melt has not changed but occurs about 3-4 weeks earlier than in 1990s.

- Four-day movement rates of adult female polar bears have significantly declined during summer (August-October) in the 2000s due to disappearance of offshore and archipelago summertime sea ice. Bears are significantly less likely to move from land onto sea ice in summer in the 2000s.

- Bears use significantly lower sea-ice concentrations in winter and spring in the 2000s than the 1990s. Bears had stronger preferences to be closer to the 300 m depth contour (on shelf waters and near land) in the 2000s. Sea-ice concentration alone did not determine preferred habitat, adult females selected for lower sea-ice concentrations if it allowed them access to continental shelf waters (<300 m).

- Adult female bears were significantly closer to land in all months in the 2000s except at the end of breakup (June-July), when they remained on offshore sea ice as long as possible to maximize feeding.

- Potential long-distance swimming events were identified, defined as events in the summer during which bears traveled >100 km from offshore sea ice, to Baffin Island, through areas with <10% sea-ice concentration, and with a concurrent period of reduced or absent collar transmissions (i.e., because collars generally do not transmit when bears are in water). These events were observed in both decades, but the frequency increased in the 2000s, particularly in 2011.

- Bears spend significantly more time on land on Baffin Island; arrival dates on Baffin Island in summer were one month earlier in 2000s. The amount of time bears spend on land has increased by 20-30 days since the 1990s. Bears in the 2000s no longer arrive on Devon or Ellesmere Islands but only on Baffin Island (some also remain in Melville Bay Greenland).

- Entry dates into maternity dens were >1 month later in the 2000s. Exit dates from maternity dens did not change. Overall there was a significantly shorter maternity den duration in the 2000s.

- The first date of arrival on land by pregnant females was significantly earlier in the 2000s than the 1990s and bears spend more time onshore before entering maternity dens.
- Maternity dens in the 2000s occured at higher elevations and steeper slopes than maternity dens in the 1990s, likely due to reduced snow cover.

4.1. Introduction

Information on habitat can be used to facilitate subpopulation status assessment and harvest recommendations in the context of changes that are occurring in both available habitat and habitat use by polar bears. Habitat analyses can be used to evaluate polar bear distribution with respect to environmental variables, particularly ice conditions, topography and food availability or distribution (e.g., Ferguson et al. 1999. Durner et al. 2009, Wilson et al. 2014) and to inform subpopulation status relative to changes in environmental carrying capacity (Regehr et al. 2015). Using habitat information to identify key areas of use, or areas that will be critical in the future, is also important for conservation. Finally habitat studies can also be used to provide important context for interpreting both point estimates and trends in vital rates or MR results, which often have considerable bias and uncertainty. Large changes in sea-ice habitat for polar bears have occurred across all 19 subpopulations (Stern and Laidre, in review), particularly BB which occurs in the seasonal sea-ice ecoregion (Amstrup et al. 2008).

We assessed changes in sea-ice habitat for the BB subpopulation and used satellite telemetry data collected over two decades to assess changes in movement rates of bears, sea-ice habitat use, terrestrial habitat use, arrival and departure dates on/off land, and maternity denning. The results of this work provide important context for the MR results and add perspective on how environmental changes may explain observed results. They also provide scientific information which can potentially be used for comparison to observations from LEK studies, including observations of increased densities of polar bears on land or closer to shore (and communities).

4.2. Methods

Sea ice Analysis

Methods are described in detail in Stern and Laidre (in review), however, are briefly reviewed here. We used daily satellite data of sea ice for the period 1979-2014 from the National Snow and Ice Data Center in Boulder, CO (Cavalieri et al. 1996, updated yearly) to examine seasonal patterns of sea ice and how they are changing over time. These products are designed to provide a consistent time series of sea-ice concentration (the fraction, or percentage, of ocean area covered by sea ice) spanning the coverage of several passive microwave instruments. The gridded data have a cell size of 25 × 25 km. We used ETOPO1 for bathymetry, a 1 arc-minute global relief model of Earth's surface that integrates land topography and ocean bathymetry, built from numerous global and regional data sets (Amante and Eakins 2009). We averaged the ETOPO1 data over each 25-km grid cell to get the mean ocean depth for the cell, which we used to distinguish the continental shelf (less than 300 meters depth) from the deeper ocean. Baffin Bay consists of 1042 grid cells (656 × 103 km2); 28% have mean depth < 300 m, 72% have mean depth > 300 m. The shallow regions are located along the east coast of Baffin Island and the west coast of Greenland.

From the sea-ice concentration data, we calculated the daily area of sea ice for three sets of grid cells in Baffin Bay: (1) all ocean depths, (2) shallow depths only (< 300 m), and (3) shallow depths (< 300 m) along the coast of Baffin Island only. Figure 4.1 shows the seasonal cycle of sea ice area in Baffin Bay (all depths). We calculated the March average sea-ice area and the September average sea-ice area over all years (1979-2014), and defined a threshold (T) to be halfway between the two averages. We then found the date each spring when the sea-ice area dropped below T on its way to the summer minimum, and the date each fall when the sea-ice area rose above T on its way to the winter maximum. We call these the dates of sea-ice retreat and advance, respectively. These dates were calculated for each year (1979-2014) for the three regions (all depths, shallow depths only, and shallow depths along Baffin Island only).

In addition to the dates of sea-ice retreat and advance, we calculated the number of days from retreat to advance (also called length of summer), and the mean sea-ice concentration during June through October. Finally, we calculated the number of ice-covered days per year at every grid cell in Baffin Bay. This is the number of days per year that the sea-ice concentration exceeds 15%.

Sea-ice Habitat Use and Resource Selection Models

Polar bears were tagged in Northwest Greenland on the fast and pack ice between mid-March and mid-April 2009-2013 in Baffin Bay as described in Chapter 2. A total of 91 bears were tagged with satellite transmitters in BB. Of these, 38 were AFs collared in BB. These data were combined with a historical data set of AFs captured and tracked in the 1990s. In BB, 1991-1995 43 collars were deployed on AFs, with the majority deployed during the ice free season in fall on Baffin Island (n=11 deployed in spring in NWG, of these n=9 transmitted long enough to be included in the analyses). Only bears captured within the BB subpopulation boundaries were included in the comparative analysis, as defined by PBSG (2010).

Data filtering and sub-sampling – Methods on data filtering and sub-sampling are described in Chapter 2. Data were divided into seasons included Spring (March – July, which included the peak of sea-ice coverage and initiation of sea-ice break-up), Summer (August –

October, which included the end of break-up and the on-land period) and Winter (November – February, which included the freeze-up period and time when bears went back out on the sea ice). Adult females were defined as ≥ 5 years old and adult males as ≥ 6 years old. Age group status as determined in the field was verified based on tooth analyses (Table 4.1).

Movement rates – We examined daily (4- or 5-day) movement rates for AFs in each subpopulation by decade and month. For the RSF, we used the mean monthly movement rate + 2SD for the radius of potential habitat selection at each time step, following the approach used by Durner et al. (2009) and Laidre et al. (2015). We used monthly values that were specific to each decade. We used a spatial distance limit of 400 km (or roughly 12 days) as the maximum step length possible in the data. Any gaps longer than that were skipped.

Habitat covariates for RSF – In the comparative analysis between polar bears tracked in the 1990s and 2000s daily sea-ice concentration values were used from satellite passive microwave data (SSM/I) from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data (Cavalieri et al. 1996) available from the National Snow and Ice Data Center (NSIDC) in Boulder, Colorado, USA. Sea-ice concentrations were provided in a polar stereographic projection with a nominal grid cell size of 25 × 25 km. Temporal coverage was every other day from 26 October 1978 through 9 July 1987, and daily through 01 April 2015. Sea-ice habitat was defined around each polar bear location at two spatial scales: the sea-ice concentration pixel value where the bear was located and the mean sea-ice concentration within a region consisting of the 3 × 3 block of pixels centered at the bear location (nominal area 5,625 km^2) with the corners removed in order to approximate a circle. All denning periods were identified (maternity and shelter dens) (Escajeda 2016) and removed from resource selection (RSF) models. RSF models were only conducted on AFs to enable comparison with the 1990s. Bears with a low n (less than 3 locations) were removed from the analysis.

We also calculated the distance from each polar bear location to the sea-ice edge (defined with two concentration thresholds) and the distance from each polar bear location to the mainland Baffin Island coastline. The sea-ice edge covariate estimated the distance (in km) from the bear's location to the center of the nearest pixel with either 15% or 50% sea-ice concentration. We used the 15% sea-ice concentration as a delimiter between sea ice and open water. We used the 50% sea-ice concentration as a delimiter between suitable polar bear habitat and breakup conditions (see Stirling and Parkinson 2006). Distances were determined by great circle calculations based on latitude and longitude and therefore were not subject to pixel size.

We included variables about bathymetry in the models. We used IBCAO 3.0 (500 m^2) to estimate the depth value (m) at the bear location and to determine if the bear was in categorical depth categories (shelf: 0 - <300 m, intermediate: 300 - 1000 m, and basin: >1000). We also calculated the distance of the bear to the shelf break (where the shelf break was considered to be > 300 m). In 931 cases (out of 500,000) at the southern extent of the range offshore in Davis Strait, there were no values in the IBCAO grid south of 60°N. For these values we used the ETOPO-5 grid (1 km^2) to retrieve depth information. Finally we included a variable that quantified if bears moved from sea-ice in winter or spring to land.

Buffers were created around each polar bear location that were representative of available habitat bears could select on a 4-day (or occasionally 5 or 6-day) interval depending on satellite collar duty cycle (cycling of transmissions for battery longevity). The radius of the buffer was based on mean monthly movement rates for bears grouped into decades +2 SD (1990s and 2000s). Fifty random locations in each buffer were sampled for each time step and represented candidate locations not selected by the bear at each given movement step (i.e., pseudo-absence locations). This control data set was considered to represent local habitat availability. All pseudo-absence locations were linked to the same habitat variables listed above using ArcGIS (ArcGIS 10.1, ESRI, Redlands, CA, USA). A maximum time gap of 12 days or 400 km distance was selected between locations to minimize the size of the buffer.

RSF sea-ice models – Univariate habitat utilization was quantified and contrasted for each habitat covariate (e.g, only pixels where the bear was present) in the 1990s and 2000s. Multivariate conditional logistic RSF models were built for each season and decade. We selected variables for multivariate analyses a priori based on biological hypotheses. We did not include variables in the same model that were biologically redundant (continuous depth vs. distance to 300 m shelf) or highly correlated (e.g., distance to 15% and 50% sea-ice concentration). Models were fit to each decade and one combined model using both decades was fit to facilitate testing for differences in effects across decades through the use of a covariate by decade interaction. We used conditional logistic regression with matched location/pseudo-absence sets (CLOGIT function from SURVIVAL package) (R Development Core Team 2013; Therneau 2015) to model the strength of preference for habitat parameters in the 1990s and 2000s.

Terrestrial Habitat Use and Selection

Terrestrial RSF – For collared polar bears on land in summer, RSF models included land covariates derived from two Digital Elevation Models (DEMs): the ASTER GDEM for all positions in Canada (http://www.jspacesystems.or.jp/ersdac/GDEM/E/4.html, 22 m2) and the GIMP for all positions in West Greenland (http://bpcrc.osu.edu/gdg/data/gimpdem). The

ASTER DEM, which was used for the majority of telemetry locations, consisted of a mosaic of tiles from the Advanced Spaceborne Thermal Emission and Reflection Radiometer Global Digital Elevation Model (ASTER GDEM; Version 2), a product of Japan's Ministry of Economy, Trade, and Industry (METI) and the United States National Aeronautics and Space Administration (NASA). The ASTER GDEM had an overall horizontal resolution of ~17 m at the 95% confidence interval with a vertical resolution of 75 m. The DEM was produced at a 22.625 x 22.625 m resolution with elevations positioned in the WGS 1984 datum and projected in a North Pole Stereographic projection with a central meridian of -55°W.

At all bear locations we extracted the value of elevation (m), slope (calculated as % rise) and aspect (calculated as 0-360 degrees) at the bear location using the DEM at the highest resolution. We included a variable that quantified whether bears moved from land habitats to sea ice in summer. We followed a similar process for the sea ice RSF models above but only examined land covariates in the summer season (August-October), including interactions for the 1990s and 2000s.

Arrival and departure dates on land – We used location data from satellite collars to compare the timing of land use patterns by AF polar bears in the Baffin Bay between two time periods (1991-1997 and 2009-2015) when substantial summer sea-ice loss occurred. We excluded bears that remained in Melville Bay for the entire tracking period (1-2 years) as these bears, though close to shore, were using glacial ice throughout the summer and we could not determine if they were on land or in a glacial mélange. We quantified the date individual BB bears arrived on land in fall, the duration of time spent on land, the date bears were back on the sea ice in spring.

All radio-collar locations were determined by the Argos System with accuracy from < 250 m to > 1500 m (see http://www.argos-system.org/web/en/78-faq.php#faq-theme-55). Location data were filtered to remove implausible locations as described previously. Instances of dropped collars were identified based on activity and temperature sensors and data collected post-drop were removed.

We considered a bear to be on land if its Argos location was within 5 km of the high resolution coastline as identified by 0-pleth line of the IBCAO digital elevation model (Jakobsson et al. 2012). The 5 km buffer was used to encompass small barrier islands that may be used by polar bears in the summer but are not depicted as land in the GIS file and to account for low accuracy of some locations. Our 5 km buffer might have resulted in some offshore bears being classified as on land, but this was less likely to occur during the focal time periods of our analysis (fall and spring) because landfast ice was either disappearing or forming in the seasonal ice zone. We were most interested in the date bears arrived on land during the fall sea-ice recession and the date bears returned to the ice during spring ice formation. Thus any short-term visits to land were not included in the analysis and we focused on large seasonal patterns. Due to the nature of the sea ice cycle in BB, bears within 5 km of the coast during fall were likely to either have been in open water or on land.

Bears were required to enter the 5 km buffer and stay within 5 km or less of the coastline for at last 14 days before they were considered to be ‘on land’. The same criteria were used for bears departing from land in spring (>= 14 days on the ice). For all AF bears identified to be in maternity dens (Escajeda 2016), we excluded dates of return to the sea ice in spring, as the maternity denning period dictated the date of return, not the formation of sea ice.

For pairs of positions that were separated by 4 or 8 days, we linearly interpolated the date on or offshore. We excluded data when observed locations were separated by >8 days, except in the case of when bears were offshore in summer on <15% sea ice (>100-200 km from the nearest coastline) and next subsequent position was on land. In these cases, there were data gaps (12-30 days) in locations due to potential long distance swimming from central Baffin Bay to the shore; see next section.

Potential swimming events – We identified potential long distance swimming events from central BB during the break-up season. There are frequent drop-outs in transmission from collars (i.e., missing positions) such that the sampling interval was 8 days, or 12 days, or longer. The drop-outs occurred when the bear was in a region of extremely low sea-ice concentration, suggesting that the failure to transmit a position may be because the antennae is in the water, i.e., the bear was swimming (Pagano et al. 2012).

For every polar bear location, we extracted the sea-ice concentration at the SSM/I grid cell in which the polar bear was located providing a time series of sea-ice concentration following the bear's trajectory. We also calculated the distance from every polar bear position to the nearest coastline and developed plot distance-to-land vs. time, using symbols representing the sea-ice concentration at the bear position. We examined the trajectory of each bear and identified the timing and occurrence of gaps in collar transmissions. We selected bears where the transmissions ceased during the break-up season when the bear was offshore and examined both the bear's trajectory and the sea-ice conditions for potential swimming to land.

Maternity denning – Solitary females or females with two year-old cubs were considered candidates for denning the following winter after capture (Wiig 1998). All of the satellite collars in the 2000s provided temperature and motion data along with position coordinates.

Temperature was measured by a thermistor within the collar (Fischbach et al. 2007). Data collected on polar bear dens in the Baffin Bay and Kane Basin in the 1990s were originally published by Ferguson et al. (1997), and included 29 dens from Baffin Bay and 9 from Kane Basin. The 1990s data were obtained in the form of one position per day.

Although the reading from the collars is influenced by the animal's body temperature, the temperature reported by the collar is a general representation of the ambient temperature of the surrounding habitat (Harris et al. 1990). Temperature data were extracted from transmitters using the Telonics Data Converter software (Version 2.21; Telonics, Mesa, AZ). Position and temperature data for bears identified as denning candidates were examined from July to June of the following year. Only one best quality position and one temperature reading were used for each day. The designated position for each day was selected by choosing the first position with the best location quality score. The temperature reading for each day was selected by first removing any temperature points ≥ 40ºC or ≤ -40ºC which were considered outliers (Tchernova 2010), and calculating the average. Three variables signal that a female is in a den: high temperature readings compared to ambient air temperature (10 to 40ºC warmer), constant position on land, and decreased quality and frequency of transmissions (Amstrup and Garner 1994; Messier et al. 1994; Wiig 1998; Fischbach et al. 2007). Temperature data were available for all Baffin Bay bears.

Since transmissions were received on a 4-day duty cycle, the position data were particularly coarse as points found within the denning period did not closely center on a single den position, but rather consisted of a cluster of points within a small area. The data for all adult females were examined for sustained high temperatures (greater than 0°C), decreased transmission quality and frequency, and a stationary position on land during the denning period

(September through March). We created a subset of the position data for each denning candidate during this time period and mapped the points in ArcMap v.10.1 (Environmental Systems Research Institute, Inc. [ESRI], Redlands, CA 2012). Individual point shapefiles were created from the identified subset for the three location quality categories: LQ 1, LQ 2, and LQ 3. A buffer was then drawn around each point with radii equivalent to the maximum error estimate for each LQ score (points with LQ 1 had a buffer of 1500 m, LQ 2: 500 m, LQ 3: 250 m). The mean center of the intersection of these buffers then defined the den position. The method provided a probable location for the den site based on the error estimate of the satellite telemetry positions during the denning period and is independent of the number of positions as well as any spatial outliers. Note that not all of the dens were determined using this method, some bears had sparse location data within the denning period and thus the den positions had to be determined using variants of the buffer method.

Den Phenology Analyses – Length of denning was used to distinguish maternity dens from shelter dens. Shelter dens were typically occupied for a short period of time (> 14 days to < 4 months) whereas females will typically remain in maternity dens for > 5 months (Messier et al. 1994). Though shelter dens were included in the den habitat characteristics analyses, they were excluded from the phenology analyses. In addition to denning duration, den entry and exit dates were compared to the 1990s data (Ferguson et al. 2000). The exit date for each den was established as the median date between the female's last transmission from the den and the first movement outside the den, indicated by a significant drop in temperature and movement away from the den site. Most of the entry dates for the 2000s dens were determined by creating a 1 km buffer around each den site and selecting the median date between the last date outside of the buffer and the first date inside the buffer. The entry date was verified by comparing the autumn temperature data for each denning bear with temperature readings from a non-denning bear that same year to check for a difference of more than 10°C. The dates when the temperature readings diverged by ≥ 10°C were then compared to the entry dates determined by the position data. Both the den entry and exit dates were measured as day-of-year (DOY; Day #1 is 1 January), which we then used to calculate the denning duration in number of days (Messier et al. 1994; Wiig 1998; Ferguson et al. 2000).

We used two-sample Mann-Whitney U tests to test for differences between the entry/exit dates between the 1990s and 2000s datasets and duration spent in the dens without assuming normality. Additionally we tested for a correlation between den entry date and latitude using a Kendall's tau test. An alpha value of 0.05 was used for all statistical analyses.

First Date on Land (FDOL) by pregnant females – The date of entry onto land was determined for each bear entering a maternity den. Cherry et al. (2013) defined the first date on land (FDOL) as the date that the bear first came onto land without returning to the sea ice until freeze-up the following fall; we modified this for pregnant females to be the first date on land after which the bear did not return to the sea ice until she emerged in the spring. First dates on land for sheltering bears were also calculated, but not analyzed.

Den Habitat Characteristics Analyses – All den positions were imported into ArcMap and overlaid with the ASTER DEM of the study area. The elevation, aspect, and slope of each den site were extracted from the DEM while straight-line distance to the nearest shoreline was measured using a vector shapefile of Canada's coastline (US Defense Mapping Agency). Elevation was calculated as the elevation of the cell containing the den site and was measured in meters. Aspect is the compass direction (in degrees) the cell faces, while slope measures the rate of maximum change of elevation in degrees.

A principal component analysis (PCA) using a correlation matrix was performed on the habitat data of the maternity dens in order to determine which variables drive any dissimilarity among den sites. In the data matrix for the PCA, each den site was input with its elevation, aspect, slope, and distance to coast. A second matrix organized the den sites into two groups: sample period (1990s or 2000s), and latitude zone (south of 70^{o} N, central 70 ^{o} N to 75 ^{o} N, and north > 75^{o} N). Prior to the analyses, the environmental variables in each dataset were log10-transformed to control for skewed data (Kenkel 2006). After computing the PCA, a Monte Carlo randomized approach was used to test the significance of the eigenvalues ($\alpha = 0.05$). All analyses were performed using the statistical software R version 3.0.2 along with the "vegan" package (Oksanen et al. 2013) and the "Biostats" R package (McGarigal 2015).

In order to test for differences in environmental descriptors between the two groups, a permutational multivariate analysis of variance (perMANOVA) was performed on the maternity den matrix. For the distance matrices, Euclidean distances were calculated for each log10-transformed and column-standardized matrix of raw data. A permutation test was then used to evaluate the significance of the resulting pseudo-F statistic as compared to a null hypothesis of no difference between groups. To determine the source(s) of dissimilarity detected by the perMANOVA, we applied a supplementary test of multivariate homogeneity of group dispersion (DISPER; Anderson 2006). DISPER involves computation of the distance of each group member to the group's centroid and applies an ANOVA to the distances with a null hypothesis of no difference in variation among groups.

4.3. Results

Sea-ice Habitat

In Baffin Bay, sea ice is retreating earlier in spring by 7 days/decade and advancing later in fall by 5 days/decade (Figure 4.2). Trends in four of the sea-ice metrics (Table 4.2) show consistent loss of polar bear habitat. The length of summer (number of days from retreat to advance) is increasing by 12 days/decade, and the mean sea-ice concentration during June-October is decreasing by 4 percent/decade (Figure 4.3). All these trends are statistically significant (Table 4.2).

Trends in the dates of spring sea-ice retreat and fall sea-ice advance are stronger for the shallow depths (< 300 m) than for all depths collectively. When only the shallow depths on the Baffin Island shelf are considered, the trends are weaker than for all depths collectively, which suggests that the trends are stronger for the West Greenland shelf than for the entire Baffin Bay. This can be seen in the trend in the number of ice-covered days (Figure 4.4), which shows a greater loss of ice-covered days along the western coast of Greenland than in central Baffin Bay.

The pattern of spring sea-ice retreat in Baffin Bay begins with melting along the southwest coast of Greenland and progresses northward. At the same time, the North Water Polynya (located at the north end of the Baffin Bay region, in Smith Sound, and the south end of the Kane Basin region) begins to melt out. At some point during summer, these two open water areas connect as Melville Bay melts out, severing the continuous ice connection between Baffin Island and Greenland. The sea ice then continues to melt back toward the coast of Baffin Island. Occasionally a “sea-ice island” becomes the last remnant of ice in Baffin Bay, if the ice along the coast of Baffin Island melts out first. This general pattern of melt has not changed over time, but it is occurring earlier in the summer now (in the last decade) than in the 1980s and 1990s (Figure 4.5). The early part of the melt pattern, up the coast of Greenland and across Melville Bay, is trending even earlier than the melt on the western side of Baffin Bay. In October and November, sea ice advances southward through Baffin Bay, generally with the leading edge along the coast of Baffin Island. This pattern is occurring later in the fall now (in the last decade) than in the 1980s and 1990s (Figure 4.6).

In summary, the coast of Baffin Island is generally the last place in Baffin Bay to lose sea ice in summer, and the first place in Baffin Bay to regain sea ice in fall. The spring retreat of sea ice there is trending earlier by about 7 days/decade, or 24 days over the 3 ½ decades of this study. The fall advance of sea ice there is trending later by about 4 days/decade, or 14 days over the period of this study. During the months of June through October, the mean sea-ice concentration is trending downward by about 4% per decade, or about 14% over the period of this study. There is year-to-year variability in all the sea-ice metrics, but the trends are all statistically significant.

Movement rates – In Baffin Bay, mean monthly movement rates for adult females in the 1990s ranged from 5.5 km/day (in October) to 15.8 km/day (in December) (Figure 4.7). Rates for adult females in the 2000s ranged from 1.9 km/day (in September) to 13.8 km/day (in December) (Figure 4.8, Table 4.3). In general, rates were lowest in both decades during September and October, and highest in December. Adult females had significantly higher movement rates in the 1990s during May, August and September. In the summer months (Aug-Sept), movement rates in the 2000s were one-half to one-third of the rates in the 1990s ($p<0.001$) (Table 4.3). Overall there was also greater variability in movement rates in the 2000s than the 1990s, perhaps representing a broader range of strategies in changing habitats. The greatest variability was found in the early winter months, November and December, especially in the 2000s. This may represent changes in timing of freeze up and when bears access the sea ice.

RSF sea-ice models – We first examined univariate relationships for each covariate as a continuous function over the entire annual cycle (Figure 4.9 and 4.10). Adult female polar bears use significantly lower sea-ice concentrations in the 2000s than the 1990s in all seasons except May/June (Figure 4.9). Bears in the 2000s were also significant closer to land in all months except June and July. Overall there was a significantly smaller fraction of observations on the sea ice in the 2000s than the 1990s over the period that each bear was tracked (Figure 4.10).

In winter in the 1990s, the full multi-variate RSF model demonstrated a positive association between adult female polar bears and sea-ice concentration, meaning bears selected areas with higher sea-ice concentrations when available. The association in the 1990s with depth was negative, demonstrating preference for shallow shelf waters and avoidance of deep areas (Table 4.4). There was a strong negative association for movement on to land in winter. In the 2000s, preference for higher sea-ice concentrations was not as important as distance to shallow shelf waters. The distance to the 50% sea-ice concentration threshold was strongly negative, and adult females showed a strong and significant preference for regions <300 m in depth, similar to the 1990s. In both decades bears did not prefer to move from sea ice to land. Interactions between decades (changes in preference over time) showed that in winter adult female polar bears in the 2000s used lower sea-ice concentrations than bears did in the 1990s. In the 2000s there was also an increase in preference for being close to the 300 m depth contour (on shelf waters), which also serves as a proxy for distance to land.

In spring in the 1990s, the full multivariate model showed that adult females had a strong significant preference for higher sea-ice concentrations (Table 4.5), more so than in the winter months. There was also a negative association with increasing distance from 50% sea-ice concentration. This was similar to bears in the 2000s, where sea-ice concentration and distance to 50% sea-ice concentration were important model variables. When the two decades were compared using interactions, the preference for high ice concentrations was significantly stronger in the 2000s than the 1990s. Furthermore, being farther from 50% ice concentration was less preferred in the 2000s than the 1990s (it was not preferred in either decade but more so in the 2000s). There was no change in the association with land, in both decades bears strongly avoided going to land in spring.

Terrestrial Habitat Use and Selection

Terrestrial RSF – In summer both in the 1990s and 2000s, resource selection models suggested that adult female polar bears significantly preferred areas of lower elevatio and steeper slope (Figure 4.11, Table 4.6). There was no preference for aspect. Also in both decades adult females showed a negative preference for being further inland from the outer (smoothed coast) of Baffin Island. In the 1990s there was a significant preference to stay on land (bears tended not move on to ice or into water). In the 2000s, all of the preferences were similar to the 1990s. There were no changes in preference between decades for on-land habitat types (elevation, slope, aspect or distance inland from the outer coast) (Figure 4.12). The primary difference between decades was that bears in the 2000s had a stronger preference to stay on land, meaning they were less likely to move to sea ice in summer months (once on land) than in the 1990s.

Timing on land and duration on land – We obtained 78 arrival dates for AF bears between 1991 and 2013. In some cases one bear contributed more than one arrival date because the bear was tracked over multiple years. Of these, bears arrived on land on Baffin Island (n=71 dates), Devon Island (n=5 dates) or Ellesmere Island (n=2 dates, in Kane Basin). We also obtained 71 departure dates in fall, of these 66 were from Baffin Island, Devon Island (n=3 dates) and Ellesmere Island (n=2 dates). We excluded the dates from two individual bears on Ellesmere Island because the sea-ice formation and break-up in the Arctic Archipelago system is different than the seasonal ecoregion in BB, which is the region of focus.

The mean date of arrival on land in the 1990s was August 24 (SD 16 days, n=30), which was significantly later than the mean date of arrival on land in the 2000s (August 4, SD 11 days, n=46) ($p<0.001$). The mean date of departure from land in the 1990s was November 1 (SD 21 days, n=42) which was not statistically different than the mean date of departure from land in the 2000s (November 8, SD 9 days, n=27) ($p=0.06$) (Table 4.7).

The overall mean arrival latitude in the 1990s was 70.7^{o} N (SD 3.0, n=30), which was not statistically different than the overall mean latitude of arrival in the 2000s of 70.3^{o} N (SD 1.3, n=46). The overall mean departure latitude in the 1990s was 69.7^{o} N (SD 3.0, n=42) and was not statistically different than the overall mean latitude in the 2000s of 70.4^{o} N (SD 1.4, n=27). When latitude of arrival and departure was examined in 2 degree blocks, there was a clear difference between decades. In general the distribution of arrival and departure latitudes was more restricted in the 2000s than the 1990s (Figure 4.13). For example, bears in the 2000s had significantly earlier arrivals that were contracted into a smaller band of latitudes. In the 1990s, bears departed from a larger range of latitudes (75 to 67^{o} N), where as in the 2000s departures were condensed into a latitude band ranging from $73\text{-}68^{o}$ N. The departures showed the general pattern of sea-ice formation from north to sound (so more northerly latitudes were available earlier) (Figure 4.14).

Estimation of time on land requires both an arrival and a departure of the same bear in the same year (paired dates). In this case we had n=14 pairs of dates in the 1990s and n=26 pairs of dates in the 2000s. We used this to estimate the number of weeks spent on land and changes between decades. We did not include the two individuals on Ellesmere Island.

During the 1990s, the mean duration of time on land in summer was 62.3 days (SD 25, range 8-99 days). In the 2000s, the mean duration of time on land in summer was significantly longer (<0.001) by 33 days, with an average time on land of 94.8 days (SD 15.7, range 56-120). These values include bears that used any coastline within the seasonal sea-ice ecoregion (e.g., including Devon Island, where bears arrived in the 1990s). If only bears using Baffin Island were compared then the mean duration of time on land in summer in the 1990s increased to 74.5 days (SD 14, range 54-99 days, n=10 bears). This value was still significantly different when compared to the 2000s (<0.001), with bears in the 2000s spending on average 20 days longer on land (Table 4.8).

The dates of adult females arrival and departure from land, described above, had a remarkably close correlation with the sea-ice metrics in BB (Figure 4.2 and 4.15). In both decades the dates of fall sea-ice advance were correlated and highly similar to the dates of departure from land. In both decades the dates of spring sea-ice retreat were correlated to the dates of arrival on land, however there was a lag between the retreat and arrival dates while bears used sea ice in BB as long as possible before being forced to land (Figure 4.15).

Swimming to land – There were potential long-distance swimming events observed both decades however they were observed in larger numbers in the 2000s. In the 1990s there were at least two potential long-distance swimming events. In the 2000s, there were 15 potential events in July 2011, one in July 2012, and one in July 2013. Examples of data time series from these events are shown in Figures 4.16 and 4.17. The events in July 2011 were especially prominent because in that year sea-ice breakup occurred early when a large number of bears (n=15) were transmitting from offshore BB. In this case, at least 6 bears appeared to swim to Baffin Island, where large gaps in locations occurred over a period of 2-3 weeks and then bears first transmission after the gaps was onshore on Baffin Island.

Additionally, in five cases events were identified where satellite collars were functioning normally and transmissions from individual bears ceased when the bear was in <10% sea ice and several hundred kilometers from shore. These events appeared to be the start of what would have been swimming events, though the bears never reappeared on shore. (See Figure 4.18abc for examples).

Maternity denning – Maternity denning is described in detail in Escajeda (2016). Overall we found 21 dens between 2009 and 2015, including 16 maternity dens (Figure 4.19) and five shelter dens (Figure 4.20). In the 1990s, Ferguson et al. (1997) found 29 dens between 1991 and 1997 including 8 maternity dens and 21 shelter dens (Table 4.9). All but one of the dens were located on land (one 1990s shelter den was located on landfast ice inside a fjord near the shore of Baffin Island). In the 2000s, most maternity dens were found on Baffin Island except for one den on Coburg Island and one den in Melville Bay, Greenland. The maternity den in Greenland was included in the phenology analyses but not the habitat characteristics analyses because of glacial ice. The lowest latitude for the 1990s dens was 66.4° N, and 67.5° N for the 2000s dens.

Among the three bears that built maternity dens twice in our dataset only one exhibited fidelity to a maternity denning area. The bear built a den in 2012 that was 1.25 km away from her previous denning site in 2009 on a peninsula close to Eglinton Fjord, Baffin Island. The other females denned in areas far from previous den sites and showed no fidelity. Also, one female built two maternity dens in consecutive years (2011 and 2012).

The majority of maternity dens were found on a north-facing slope ($n = 21$) and were located within 21 km of the coast (Table 4.10). All of the maternity dens were located further inland than shelter dens, on average 8.5 km from the coast in comparison to 6 km among shelter dens. The maternity dens were also located at higher elevations ($\bar{x} = 524.2$ m) in comparison to shelter dens ($\bar{x} = 395.7$ m) ($p = 0.086$). One bear denned a little less than 35 km away from Qikiqtarjuaq, Baffin Island, Canada, however most bears denned far from human settlements ($\bar{x} = 143.1$ km).

Adult female polar bears in BB in the 2000s spent on average less time in their dens ($\bar{x} =$ 167.1 days, SD = 27.6 days) than in the 1990s ($\bar{x} = 194.1$ days, SD = 21 days; Table 4.11). There was a significant difference in den duration between the two time periods with a *p*-value of 0.017 (Figure 4.21). Timing of entry in the dens differed significantly among the two periods ($p = 0.018$), however no significant difference was found among exit dates ($p = 0.399$; Figure 4.22). The median entry date for dens in the 2000s dataset (3 October) was more than a full month later than the median date of entry for dens in the 1990s (28 August). Therefore, differences in entry dates accounted for the observed difference in duration among the two time periods. There was no significant correlation between latitude and den entry ($p = 0.383$) and exit dates ($p = 0.212$) for the Baffin Bay maternity dens, and the negative correlation of den duration with latitude was not significant ($p = 0.278$) (Table 4.11).

The dates of entry onto land in the 2000s significantly differed from the 1990s. The median first date on land among the Baffin Bay maternity denning bears was 7 August in the 2000s (SD = 9.1 days) compared to 25 August in the 1990s (SD = 19 days) (Figure 4.23). This follows well with the results examining first date on land for all collared females in BB, not just those that build maternity dens in fall.

The elevation and slope significantly differed between the 1990s and 2000s maternity dens ($p = 0.003$; Table 4.12). There were more dens at mid to high elevations and steeper slopes in the 2000s than in the 1990s (Figure 4.11). The average elevation and slope among the 2000s Baffin Bay maternity dens (elevation: $\bar{x} = 707$ m, SD = 285; slope: $\bar{x} = 23.1°$, SD = 7.4) was double that of the 1990s dens (elevation: $\bar{x} = 351$ m, SD = 194.5; slope: $\bar{x} = 11.9°$, SD = 6.4). Although most dens were found at southern-facing aspects in the 2000s and most were found at northern-facing aspects in the 1990s (Figure 4.24), there was no significant difference between the two time periods ($p = 0.392$). Females in BB maintained dens at similar distances to the coast between the two time periods.

The PCA ordination analysis on the Baffin Bay maternity dens matrix produced two principal components (PC) that together explained 65.69% of the variation (Table 4.13). The first component, PC 1, explained 37.31% of the variation and had strong loadings from elevation and slope (loadings > 0.6 or < -0.6 were considered significant; Table 4.13). PC 2 explained almost a third of the variation at 28.38% and was strongly loaded by aspect. We tested the statistical significance of the first four eigenvalues by applying a Monte Carlo randomization test and found both PC 1 ($p = 0.455$) and PC 2 ($p = 0.4$) to be insignificant. The perMANOVA analysis detected a significant difference between the habitat variables of the year groups ($p = 0.003$), but not the latitudinal zone group ($p = 0.775$).

Year groups were visualized in ordination space using a PCA biplot, with dispersion ellipses drawn around the year groups using the ordiellipse function from the “vegan” package (Figure 4.25). The ellipses are drawn around the standard deviations of the point scores, and the directions of their principal axes are defined by the weighted correlations (Oksanen et al. 2013). In the biplot, most of the 2000s maternity dens are positioned to the left of the plot whereas the

1990s dens are on the right. Since elevation and slope had the highest loadings for PC 1 and were both negative, the dens in the left half of the biplot have higher elevation and slope than those to the right. There was little overlap in the ellipses between time periods, which reinforces results showing a significant difference in habitat characteristics between the 1990s and the 2000s. The DISPER test on group dispersion did not indicate a significant difference in the variances among the year or zone groups. Therefore the observed difference between the year groups cannot be attributed to variance alone.

4.4. Discussion

Overall there have been large changes in BB habitat and BB polar bear habitat use since the 1990s. The sea-ice habitat has significantly declined over the period of satellite observations, especially since the mid-1990s. The length of summer (number of days from sea-ice retreat in spring to sea-ice advance in fall) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 4 percent/decade. The general pattern of melt has not changed but occurs about 3-4 weeks earlier than in 1990s. These patterns are consistent with that observed for the whole Arctic and has been reported in other polar bear subpopulations.

These changes have had impacts on the movements and habitat choice by polar bears in BB. Movement rates of adult females have declined significantly during summer in the 2000s largely due to disappearance of offshore and archipelago summertime sea ice. Bears are now concentrated on shore on Baffin Island in contrast to the 1990s where bears ranged more widely in summer and had access to sea ice. Some bears also spend the summer in NW Greenland at glacier fronts. This results in localized on-land movements and reduced movement rates in the

2000s. Bears are significantly less likely to move from land onto sea ice in summer in the 2000s than in the 2000s. Adult female bears are significantly closer to land in all months in the 2000s except at the end of breakup (June-July), when they stay on remnant offshore sea ice as long as possible to maximize feeding. This follows well with observations in BB of bears being closer to communities and in higher densities onshore (Dowsley 2005).

Adult female bears use significantly lower sea-ice concentrations in winter and spring in the 2000s than the 1990s. Bears have stronger preferences to be closer to the 300 m depth contour (on shelf waters) in the 2000s, which is also a proxy for land thus meaning bears are closer to the coast. Assigning this to a shift in preference is difficult given the concurrent changes in habitat (e.g., late sea-ice formation in fall influences how far offshore polar bears could potentially be in winter). Models indicated that sea-ice concentration alone does not determine preferred habitat, adult females select for lower sea-ice concentrations if it allows them access to continental shelf waters (<300 m).

Potential long-distance swimming events in BB were, defined as bears traveling rapidly from central Baffin Bay to Baffin Island in summer on sea ice <10% concentration with reduced or no collar transmissions. This was observed in both decades but was more frequent in the 2000s. This has been documented for other polar bear subpopulations were sea ice is increasing and springtime breakup occurs earlier increasing the frequency of long-distance swimming (Pagano et al. 2012).

Overall adult females in BB spend significantly more time on land. Arrival dates on Baffin Island in summer are one month earlier in 2000s. The amount of time bears spent on land has increased by 20-30 days since the 1990s. This follows well with studies in other areas that show similar pattern with sea-ice loss (Atwood et al. 2015, Rode et al. 2015). Adult females in

BB in the 2000s no longer arrive on Devon or Ellesmere Islands but only use Baffin Island. Of note some bears remain in Melville Bay in summer.

Maternity denning appears to have changed in association with environmental changes. Entry dates into maternity dens are >1 month later in the 2000s, although exit dates have not changed. Overall the period of maternity den duration is significantly shorter in BB in the 2000s. The first date of arrival on land by pregnant females is significantly earlier in the 2000s than the 1990s, following well with that of all adult females (see above). Finally habitat selection for den sites has changed, maternity dens in the 2000s now occur at higher elevations and steeper slopes than maternity dens in the 1990s. This may be due to changes in snow cover (reduced snow cover at lower latitudes) though more detailed habitat availability studies are needed.

4.5. Literature Cited

Amante, C., and B. W. Eakins. 2009. ETOPO1 1 Arc-Minute Global Relief Model: Procedures, Data Sources and Analysis. NOAA Technical Memorandum NESDIS NGDC-24, National Geophysical Data Center, NOAA. doi:10.7289/V5C8276M (http://www.ngdc.noaa.gov/mgg/global/global.html).

Amstrup, S. C., and C. L. Gardner. 1994. Polar bear maternity denning in the Beaufort Sea. Journal of Wildlife Management 58:1–10.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Anderson, M. J. 2006. Distance-based tests for homogeneity of multivariate dispersion. Biometrics 62:245–253.

Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015. Demographic composition and behavior of polar bears summering on shore in Alaska:

U.S. Geological Survey Administrative Report, 26 pp.

Cavalieri, D. J., C. L. Parkinson, P. Gloersen, and H. J. Zwally. 1996, updated yearly. Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS passive microwave data, Version 1. NASA National Snow and Ice Data Center Distributed Active Archive Center, Boulder, Colorado, USA, doi: http://dx.doi.org/10.5067/8GQ8LZQVL0VL.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Escajeda, E. D. 2016. Identifying shifts in maternity den phenology and habitat characteristics of polar bears (*Ursus maritimus*) in Baffin Bay and Kane Basin. MSc Thesis, University of Washington, Seattle, Washington, USA.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use by polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Ferguson, S. H., M. K. Taylor, A. Rosing-Asvid, E. W. Born, and F. Messier. 2000. Relationships between denning of polar bears and conditions of sea ice. Journal of Mammalogy 81:1118–1127.

Fischbach, A. S., S. C. Amstrup, and D. C. Douglas. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. Polar Biology 30:1395–1405.

Harris, R. B., S. G. Fancy, D. C. Douglas, G. W. Garner, S. C. Amstrup, T. R. McCabe, and L. F. Pank. 1990. Tracking wildlife by satellite: current systems and performance. U.S. Fish and Wildlife Service Technical Report 30, Washington, D.C., USA.

Jakobsson M., L. Mayer, B. Coakley, J. A. Dowdeswell, S. Forbes, B. Fridman, H. Hodnesdal, R. Noormets, R. Pedersen, M. Rebesco, H. W. Schenke, Y. Zarayskaya, D. Accettella, A. Armstrong, R. M. Anderson, P. Bienhoff, A. Camerlenghi, I. Church, M. Edwards, J. V. Gardner, J. K. Hall, B. Hell, O. Hestvik, Y. Kristoffersen, C. Marcussen, R. Mohammad, D. Mosher, S. V. Nghiem, M. T. Pedrosa, P. G. Travaglini, and P. Weatherall. 2012. The international bathymetric chart of the Arctic ocean (IBCAO) Version 3.0. Geophysical Research Letters 39 (12).

Kenkel, N. C. 2006. On selecting an appropriate multivariate analysis. Canadian Journal of Plant Science 86:663–676.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

McGarigal, K. 2015. BIOSTATS documentation. University of Massachusetts, Amherst, Massachusetts, USA.

Messier, F., M. K. Taylor, and M. A. Ramsay. 1994. Denning ecology of polar bears in the Canadian Arctic Archipelago. Journal of Mammalogy 75:420–430.

Oksanen, J., F. G. Blanchet, R. Kindt, P. Legendre, P. R. Michin, R. B. O'Hara, P. S. Simpson, P. Solymos, H. H. Stevens, and H. H. Wagner. 2013. Vegan: Community Ecology Package. R package version 2.0–9.

Pagano, A. M., G. M. Durner, S. C. Amstrup, K. S. Simac, and G. S. York. 2012. Long-distance swimming by polar bears (*Ursus maritimus*) of the southern Beaufort Sea during years of extensive open water. Canadian Journal of Zoology 90:663–676.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

R Development Core Team. 2013. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rode, K. D., R. R. Wilson, E. V. Reghr, M. St Martin, D. C. Douglas, and J. Olson. 2015.

Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere Discussion Papers, in review.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Tchernova, J. 2010. Denning characteristics and movement patterns of female polar bears with cubs in Svalbard during the first month after emergence: implication for detecting denning locations. MSc Thesis, University of Tromsø, Tromsø, Norway.

Therneau, T. 2015. A Package for Survival Analysis in S. version 2.38, http://CRAN.R-project.org/package=survival.

Wiig, Ø. 1998. Survival and reproductive rates for polar bears at Svalbard. Ursus 10:25–32.

Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. Ecosphere 5(10):136. doi:10.1890/ES14-00193.1

Table 4.1. Breakdown of adult females (AF) collared in the 1990s and 2000s. AF = adult female, AM = adult male, COY = Cub of the Year, YRL = Yearling, 2YR = 2 Year old cub.

		AF alone	AF+AM	AF+COY	AF+YRL	AF+2YR	Sum
1990s	BB	9		19	13	2	43
2000s	BB	10	2	6	12	8	38

Table 4.2. Trends in date of spring sea-ice retreat, fall sea-ice advance, fall – spring dates, and summer (June-Oct) sea-ice concentration in Baffin Bay (all depths, and shallow sub-regions). All trends are statistically significant at the 99% level according to a 2-sided F test.

Baffin Bay region	**Trend in date of spring ice retreat (days/decade)**	**Trend in date of fall ice advance (days/decade)**	**Trend in fall – spring (days/decade)**	**Trend in ice con. June-October (percent/decade)**
All depths	−7.3	+5.4	+12.7	−4.1
Depths < 300 m	−8.4	+9.7	+18.1	−3.3
Depths < 300 m Baffin I. shelf	−6.8	+3.9	+10.7	−4.7

Table 4.3. Mean monthly movement rates in BB for adult female in the 1990s and 2000s. One SE about the mean is given. We conducted a parametric test of significance between decades.

Month	Mean 1990s	SE 1990s	n 1990s	n steps	Mean 2000s	SE 2000s	n 2000s	n steps	t-test p value
1	10.29	0.88	33	195	10.07	1.38	26	225	0.8933
2	11.31	0.96	32	173	8.68	1.18	27	218	0.0898
3	11.83	1.16	32	203	9.47	1.13	27	261	0.1492
4	13.37	1.2	30	244	10.33	1	38	546	0.0558
5	14.05	0.99	31	256	9.68	0.94	37	563	0.0021
6	13.03	0.66	31	269	11.31	1.04	34	480	0.168
7	12.88	1.06	30	239	10.49	1	32	402	0.1057
8	10.52	1.17	27	193	4.77	0.63	30	337	<0.001
9	7.73	1.13	28	172	1.88	0.45	28	258	<0.001
10	5.49	0.82	38	222	4.32	1.36	27	244	0.4683
11	14.52	1.24	37	252	13.43	1.58	26	230	0.5903
12	15.82	1.19	34	213	13.79	1.76	25	233	0.3442

Table 4.4. Sea-ice resource selection function (RSF) model coefficients for the Winter season in BB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled for ease of interpretation. “Mean ice conc.10” is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. “Dist to 50%” is the distance to the 50% sea-ice concentration scaled by 100 km. “Dist to 300 m” is the distance to the 300 m depth contour scaled by units of 100 m. “Depth.100” is the absolute value of bathymetry scaled by 100 m. “Land” is the variable that describes the tendency of a bear to move from sea ice on to land.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
Mean ice conc.10	0.075	0.025	0.0025	-0.022	0.023	0.3452	0.0043
Dist to 50%.100	-0.100	0.044	0.0237	-0.107	0.052	0.0383	0.9197
Dist to 300 m.100	0.001	0.148	0.9968	-0.619	0.205	0.0026	0.0144
Depth.100	-0.032	0.010	0.0011	-0.019	0.012	0.1021	0.4266
Land	-1.617	0.295	0.0000	-2.12	0.267	0	0.2065

Table 4.5. Sea-ice resource selection function (RSF) model coefficients for the Spring season in BB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled for ease of interpretation. “Mean ice conc.10” is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. “Dist to 50%” is the distance to the 50% sea-ice concentration scaled by 100 km. “Dist to 300 m” is the distance to the 300 m depth contour scaled by units of 100 m. “Depth.100” is the absolute value of bathymetry scaled by 100 m. “Land” is the variable that describes the tendency of a bear to move from sea ice on to land.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
Mean ice conc.10	0.162	0.022	0	0.255	0.026	0	0.0066
Dist to 50%.100	-0.115	0.041	0.0047	-0.376	0.042	0	0
Dist to 300 m.100	-0.199	0.111	0.0727	-0.088	0.111	0.4265	0.4789
Depth.100	-0.014	0.007	0.0606	0.011	0.007	0.1111	0.014
Land	-1.738	0.291	0	-1.059	0.312	0.0007	0.1111

Table 4.6. Sea-ice resource selection function (RSF) model coefficients for the Summer season in BB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled for ease of interpretation. "Elev.100" is elevation scaled by units of 100 m. "Slope.10" is slope in degrees scaled by units of 10 degrees. "Aspect.10" is aspect scaled by units of 10 degrees. "BIdistCoast.10" is the distance to the smoothed Baffin Island coastline (measured from a point inland) scaled by 10 km. "Not Land" is the tendency of a bear to move from land on to sea ice.

	1990s coef	SE	p-value 1990s	2000s coef	SE	p-value 2000s	p-value delta
elev.100	-0.09	0.03	0.0033	-0.091	0.017	0	0.9627
slope.10	0.22	0.065	0.0007	0.202	0.036	0	0.8118
aspect.10	0.001	0.007	0.8484	-0.005	0.004	0.1933	0.4231
BIdistCoast.10	-0.456	0.069	0	-0.358	0.042	0	0.2243
NotLand	-1.182	0.234	0	-2.44	0.16	0	0

Table 4.7. Summary table of 78 arrival dates and 71 departure dates for individual radio-collared adult female bears arriving on Baffin Island or Devon Island (within the seasonal sea-ice ecoregion). We excluded two bears from the 1990s that arrived on Ellesmere (2 arrivals and 2 departures by two individuals = i.e., in the archipelago ecoregion cf Amstrup et al. 2008). There was a significantly ($p<0.001$) earlier arrival on land in the 2000s. No difference in departure date from land between 1990s and 2000s. Significance did not change with the inclusion of the Ellesmere bears.

Decade	**Arrival date on land**	**Departure date from land**
1990s	Aug 23 (SD 16 days) n=30	Nov 1 (SD 21 days) n=42
2000s	Aug 4 (SD 11 days) n=46	Nov 8 (SD 9 days) n=27

Table 4.8. Table of number of days (time on land) on Baffin Island (or Devon Island in the 1990s). Data composed from 56 paired arrival/departure dates from individual adult female bears. There was a significant (*p*<0.001) increase in time on land in the 2000s, between ~30 days longer between decades.

Decade	n	Min # days on land	Max # days on land	Mean # days on land	SD of mean
1990s	14	8	99	62.3	25.0
2000s	26	56	120	94.8	15.7

Table 4.9. Number of maternity and shelter dens in Baffin Bay in the 1990s and 2000s.

All Dens		Maternity Dens		Shelter Dens	
1990s	29	1990s	8	1990s	21
2000s	21	2000s	16	2000s	5
Total	50	Total	24	Total	26

Table 4.10. Summary table of the habitat characteristics for Baffin Bay maternity and shelter dens (two NW Greenland dens were not included). Elev. = elevation (meters), Asp. = aspect (degrees), CoastDist = distance to nearest coastline (kilometers).

	All Maternity Dens (*n* = 24)				**All Shelter Dens (*n* = 26)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	583.3	166.2	19.2	9.5	421.8	224.7	19.9	7.4
Min	101.0	0.0	2.4	0.4	0	–1	0	0.1
Max	1323.0	357.6	32.9	20.2	1116	357.2	46.7	54.5
Median	623.0	175.2	18.5	7.8	354	249.3	19.4	4.7
SD	306.1	102.4	8.8	6.3	320.5	124.8	12.6	10.7
	1990s Maternity Dens (*n* = 8)				**1990s Shelter Dens (*n* = 21)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	351.3	150.6	11.9	10.2	414.9	210.1	20.4	7.9
Min	131	18.4	2.4	2.2	0	–1	0	0.1
Max	623	357.6	21.4	20.2	1116	357.2	46.7	54.5
Median	279	99.1	12.1	7.6	354	247.6	20.1	4.9
SD	194.5	135.7	6.4	6.5	335.5	128.5	13.5	11.4
	2000s Maternity Dens (*n* = 15)				**2000s Shelter Dens (*n* = 4)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	707.0	174.4	23.1	9.2	458.3	301.7	17.1	4.9
Min	101.0	0.0	6.1	0.4	169.0	196.4	10.3	0.2
Max	1323.0	320.6	32.9	18.6	728.0	344.4	26.9	15.3
Median	693.0	182.2	23.0	8.3	468.0	332.9	15.5	2.0
SD	284.9	83.9	7.4	6.3	263.2	70.4	7.4	7.0

Table 4.11. Summary table of the phenology for Baffin Bay maternity dens including the Greenland maternity den. One maternity den from the 2000s data did not have an exit date and was excluded from the table. Entry and exit dates were quantified as day of year (DOY; Day #1 is 1 January).

	1990s **Maternity Dens (*n* = 8)**			**2000s** **Maternity Dens (*n* = 15)**		
	Entry DOY	Exit DOY	Duration (# days)	Entry DOY	Exit DOY	Duration (# days)
Mean	249.8	78.9	194.1	277.7	79.8	167.1
Min	230	73	163	237	60	121
Max	281	82	217	324	91	212
Median	240	79.5	201	276	80	164
SD	21.3	3.6	21	27.7	8.7	27.6

Table 4.12. Results of the two-sample Mann-Whitney U tests comparing habitat characteristics for Baffin Bay maternity and shelter dens (note that the den in Greenland was not included).

	Maternity Dens (*n* = 23)		**Shelter Dens (*n* = 25)**	
	W	***p*-value**	**W**	***p*-value**
Elevation	105	0.003	20	0.695
Slope	105	0.003	25	0.695
Aspect	74	0.392	17	0.262
Distance to Coast	55	0.776	24	0.369

Table 4.13. Summary table of results from the principal component analysis on the Baffin Bay maternity dens matrix (the Greenland maternity den was omitted). None of the principal components (PC) were significant, though PC 1 and 2 were able to capture over half of the variation in the data. Principal component loadings greater than 0.6 or less than –0.6 were considered significant (in bold).

Principal Component Analysis Summary					**Principal Component Loadings**		
	Eigenvalue	% Var.	Cum. % Var.	*p*-value		PC 1 (37.31%)	PC 2 (28.38%)
PC 1	1.49	37.31	37.31	0.465	Elevation	–0.659	0.375
PC 2	1.24	28.38	65.69	0.413	Aspect	–0.210	–0.770
PC 3	0.99	24.64	90.33	0.061	Slope	–0.604	–0.375
PC 4	0.39	9.67	100	0.908	CoastDist	–0.396	0.356

Figure 4.1. Sea-ice area in Baffin Bay (shown for all depths) for the years 1979-2014 (gray curves) using SSM-I passive microwave data. Two six-year averages are also shown (red and blue curves) that approximate the sampling dates for the MR. The threshold for defining the dates of sea-ice retreat and advance (middle horizontal dotted line) is halfway between the average March sea-ice area (upper dotted line) and the average September sea-ice area (lower dotted line).

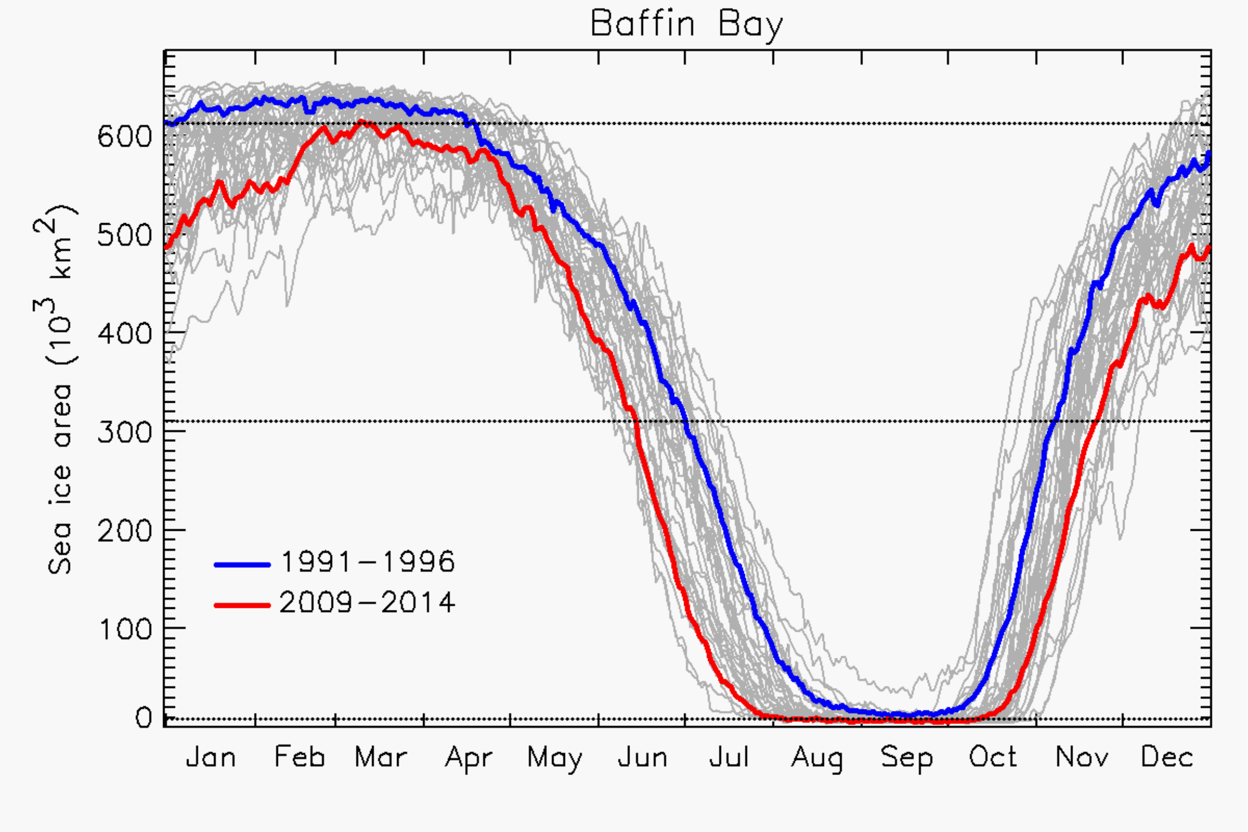
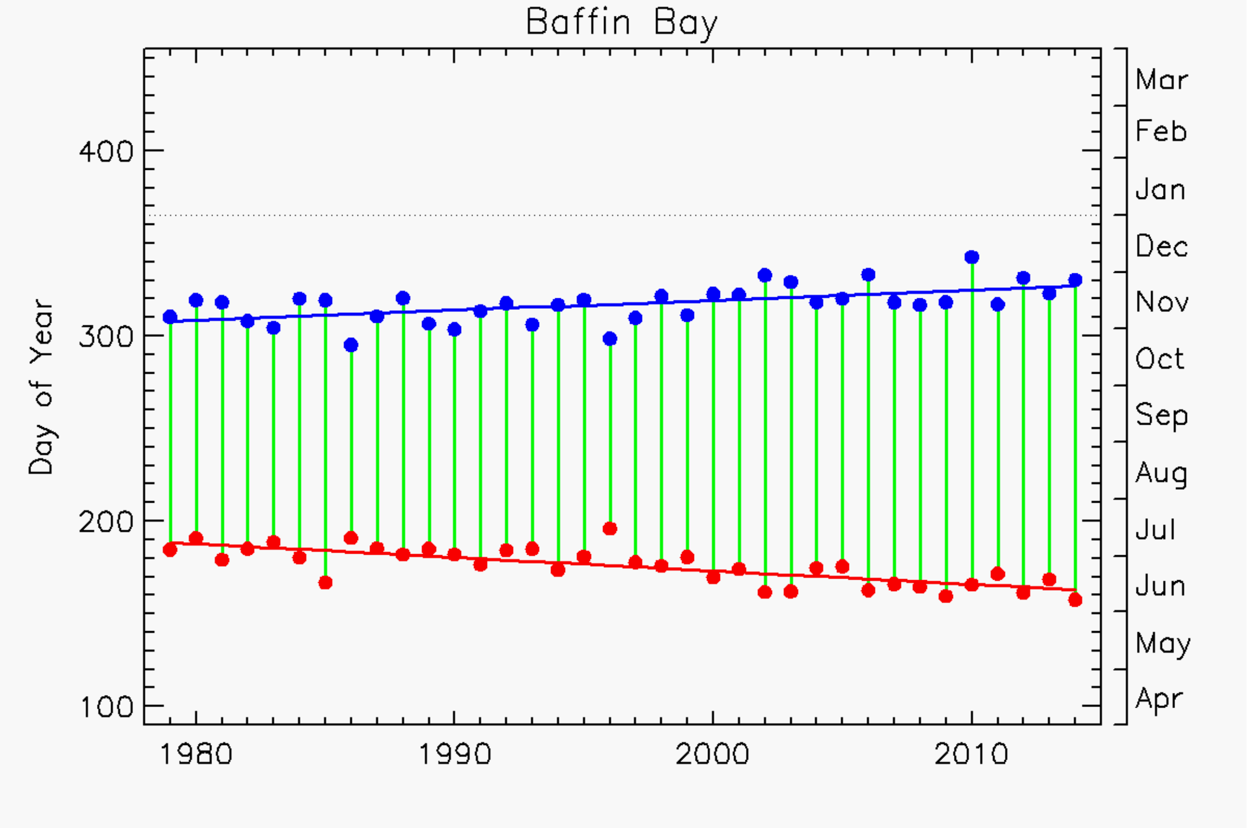


Figure 4.2. Day of spring sea-ice retreat (red circles), fall sea-ice advance (blue circles), and the interval between them (green lines), for Baffin Bay (all depths), 1979-2014. Least-squares fits to spring and fall dates are shown (red and blue lines). Trends are given in Table 4.1.



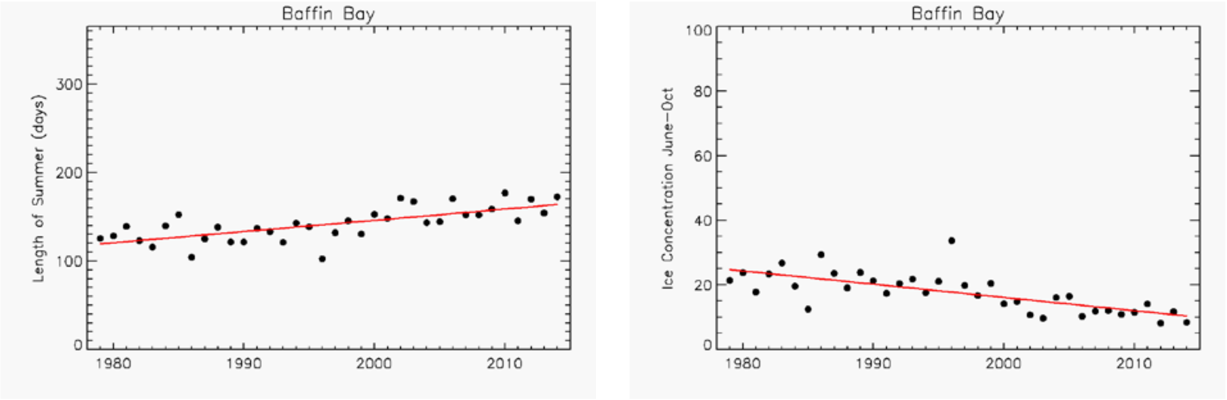


Figure 4.3. Length of summer (left) and mean sea-ice concentration during June-October (right) for Baffin Bay (all depths), 1979-2014. Length of summer is the interval from spring sea-ice retreat to fall sea-ice advance (see Figure 4.2, green lines). Least-squares fits are shown (red lines); trends are given in Table 4.1.

Figure 4.4. Trend in the number of ice-covered days, 1979-2014, color-coded for each grid cell, as indicated in the legend at left. Time series of the number of ice-covered days are shown for two specific grid cells, one in central Baffin Bay (upper right) and one close to the coast of Greenland (lower right). Least-squares fits (red lines) and numerical trends are indicated. An ice-covered day is one in which the sea-ice concentration exceeds 15%.

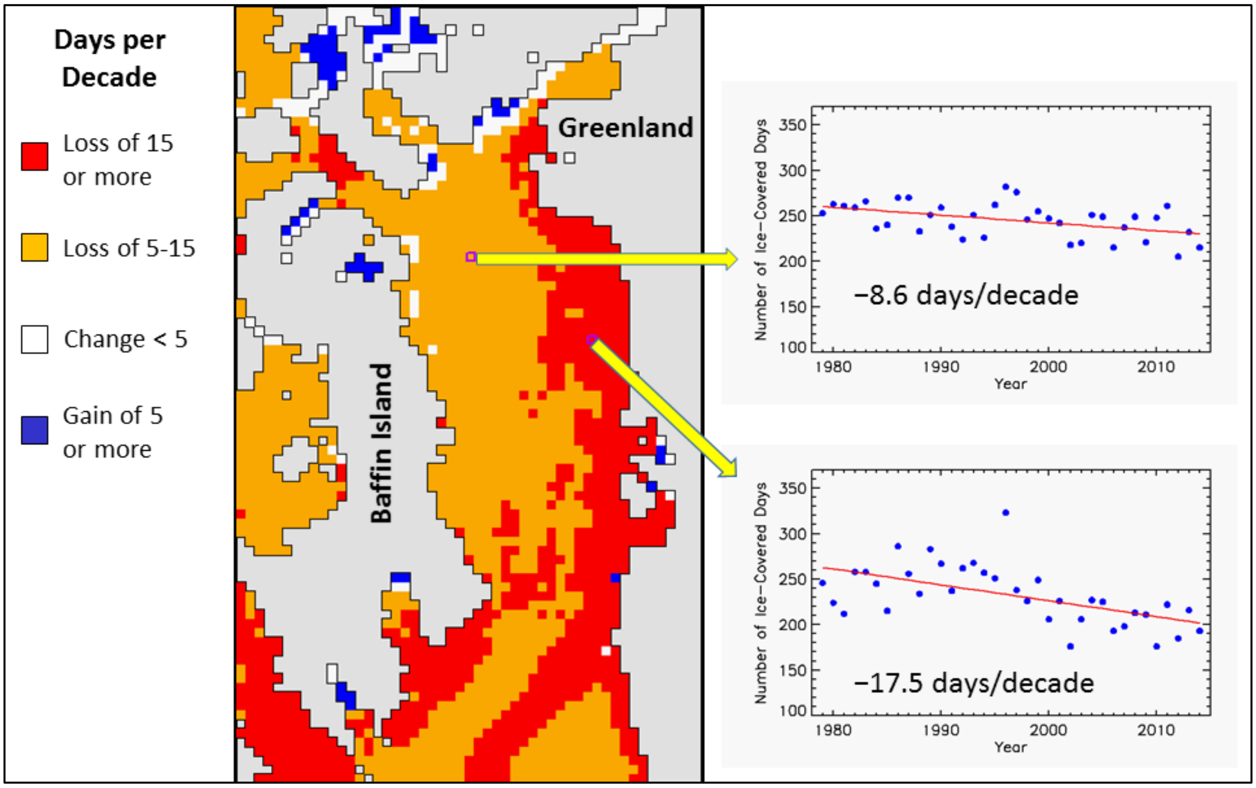


Figure 4.5. Sea-ice concentration in Baffin Bay on July 15 of every year from 1979 (upper left) to 2014 (lower right). Color coding: 15-50% (blue), 50-85% (green), 85-95% (yellow), 95-99% (orange), 99-100% (red). Black dots in Baffin Bay indicate shallow depths (< 300 m).

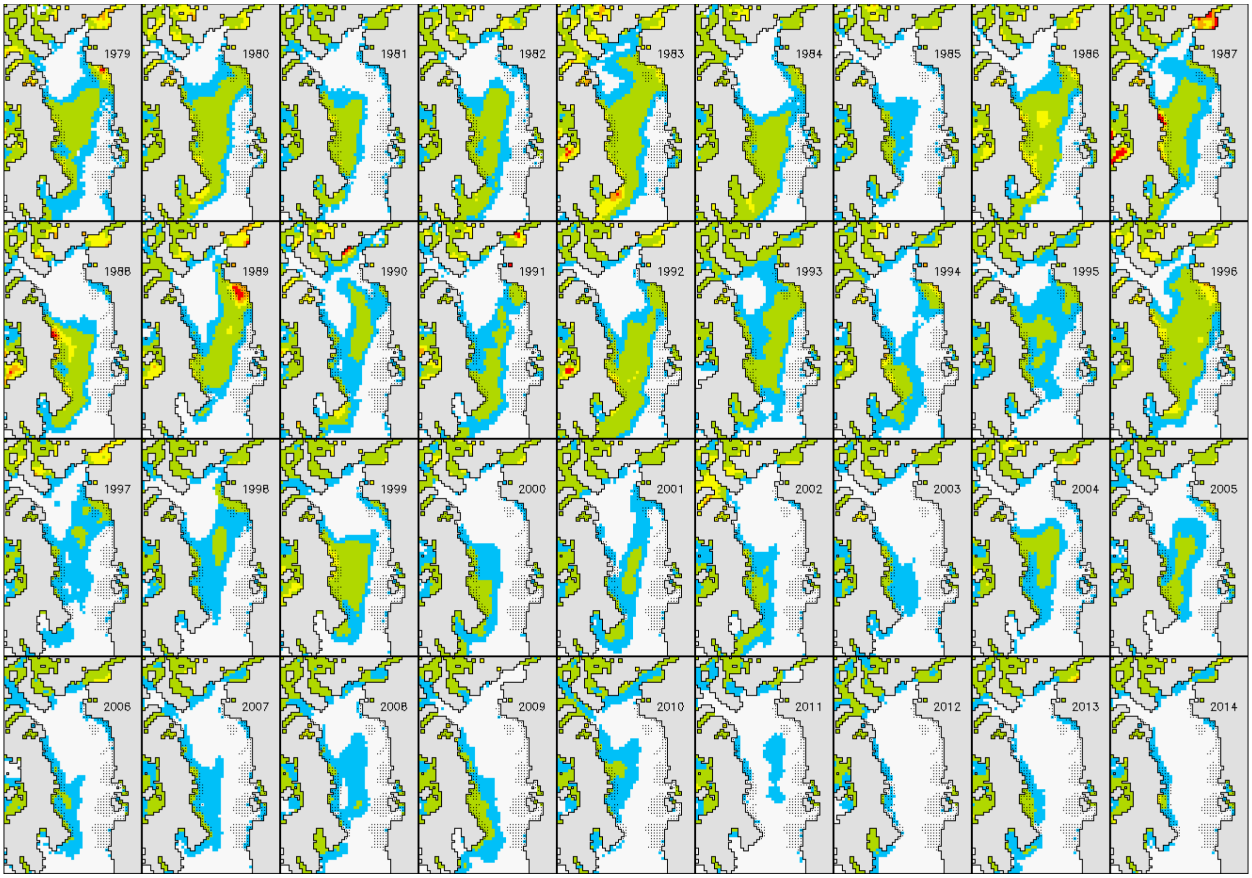


Figure 4.6. Location of the leading edge of sea ice (> 50% concentration) along the coast of Baffin Island on November 1 in the years 1991-1995 (top row) and 2009-2013 (bottom row). Color coding of sea-ice concentration is the same as in Figure 4.5. In the top row, black circles mark the leading edge of sea ice, and the horizontal dashed black line is the average position. In the bottom row, red circles mark the leading edge of sea ice, and the horizontal dashed red line is the average position. The dashed black line is the same as in the top row, showing that in the later period, sea ice has not advanced as far south by November 1 as in the early period.

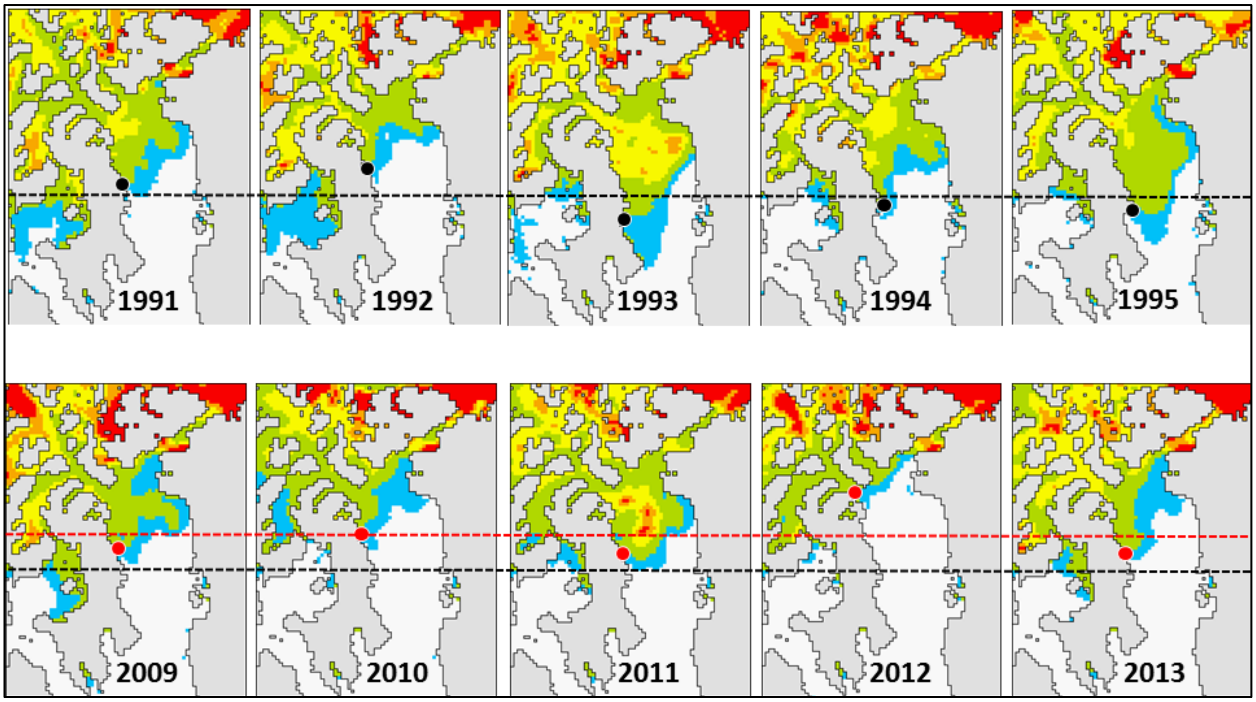


Figure 4.7. Movement rate of BB adult female bears (km/day) in the 1990s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

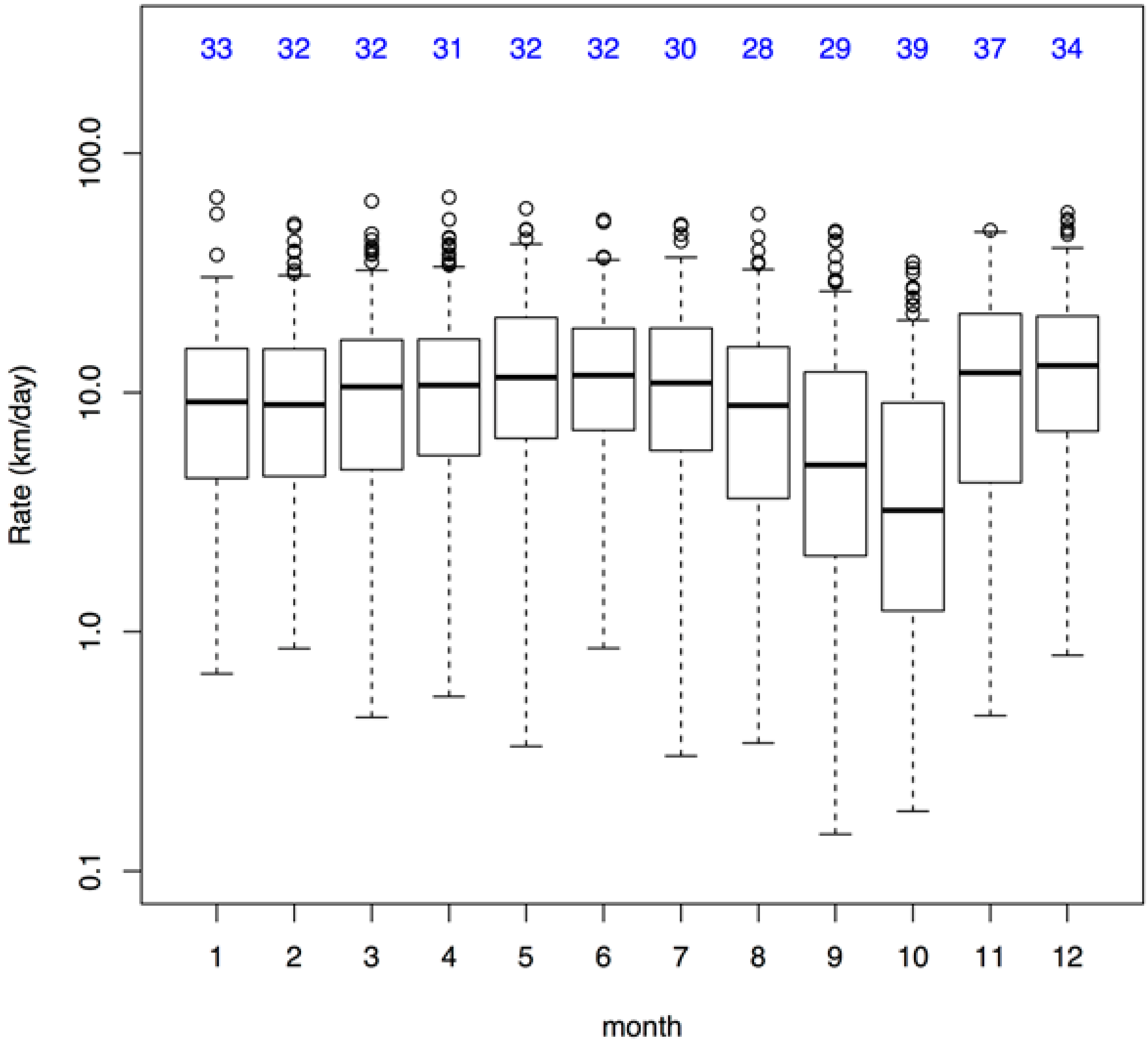


Figure 4.8. Movement rate of BB adult female bears (km/day) in the 2000s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

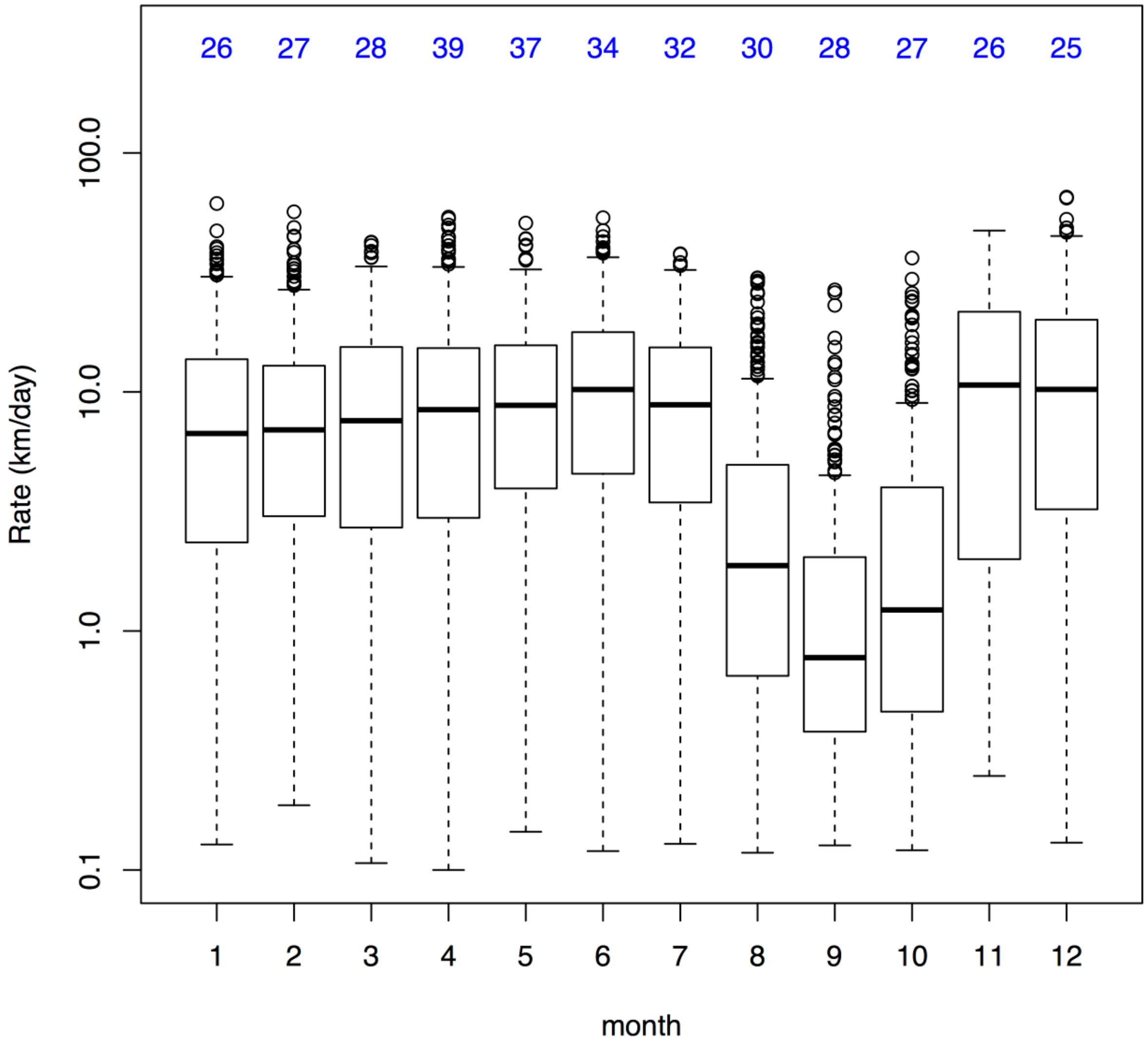


Fig 4.9. 1990s and 2000s adult female polar bear habitat use in BB for each of four sea-ice habitat variables: sea-ice concentration in small buffer, distance to 15% sea-ice concentration, distance to 50% sea-ice concentration, and distance to the nearest land. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer) used in the analysis. SSM/I sea-ice concentration is used in both decades. All positions are for bears on sea ice or water and resident bears in Melville Bay are excluded. Months 8 -10 also represent land use by bears. The small numbers in blue and red represent the number of movements for each month and decade.

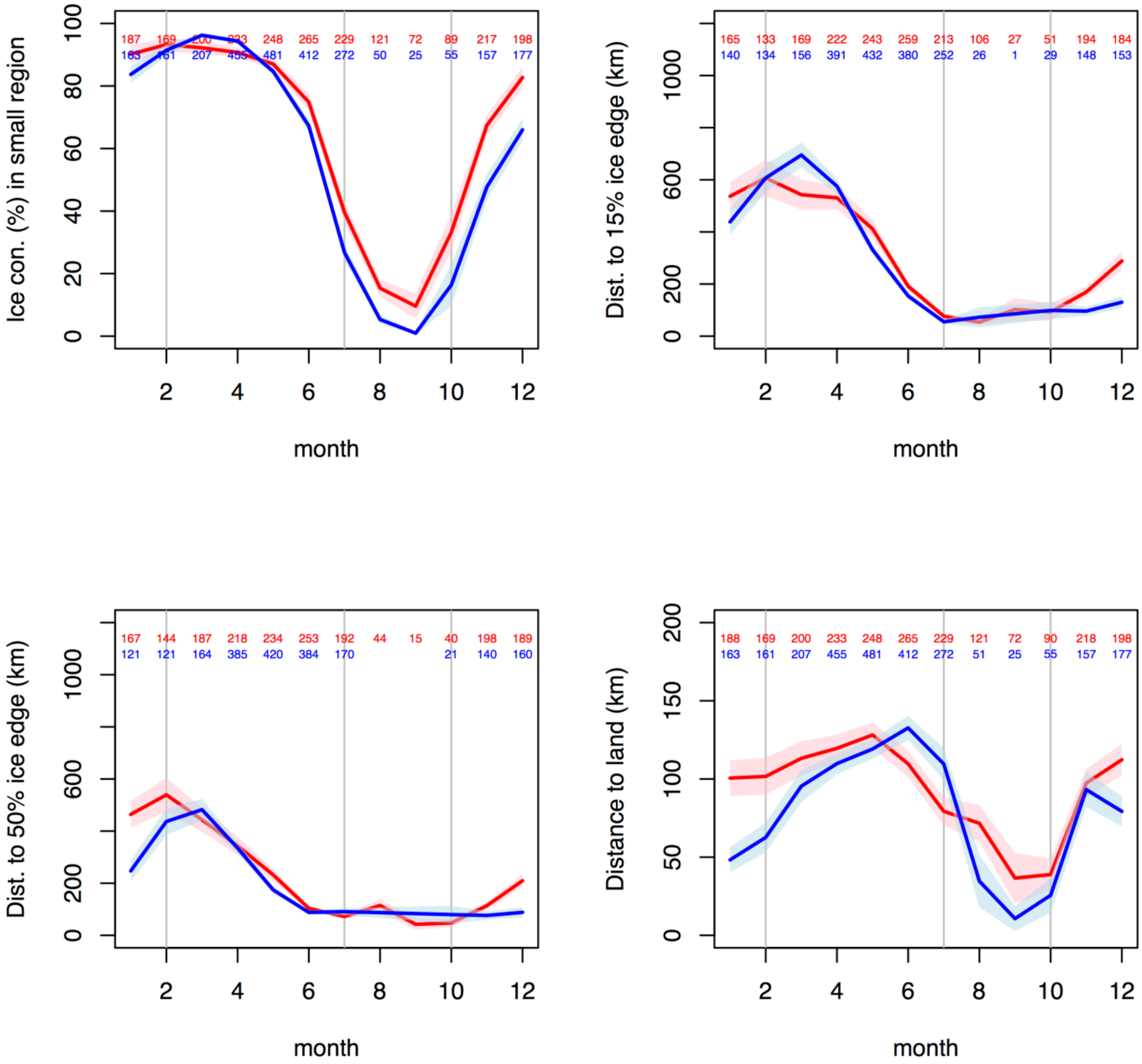


Fig 4.10. 1990s and 2000s adult female polar bear habitat use in BB for each of three sea-ice habitat variables: distance to 300 m depth contour, depth (bathymetry), and percentage of observations on the sea ice. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer). SSM/I sea-ice concentration is used in both decades. All positions are for bears on ice or water and resident bears in Melville Bay are excluded. Months 8 -10 also represent land use by bears. The small numbers in blue and red represent the number of movements for each month and decade.

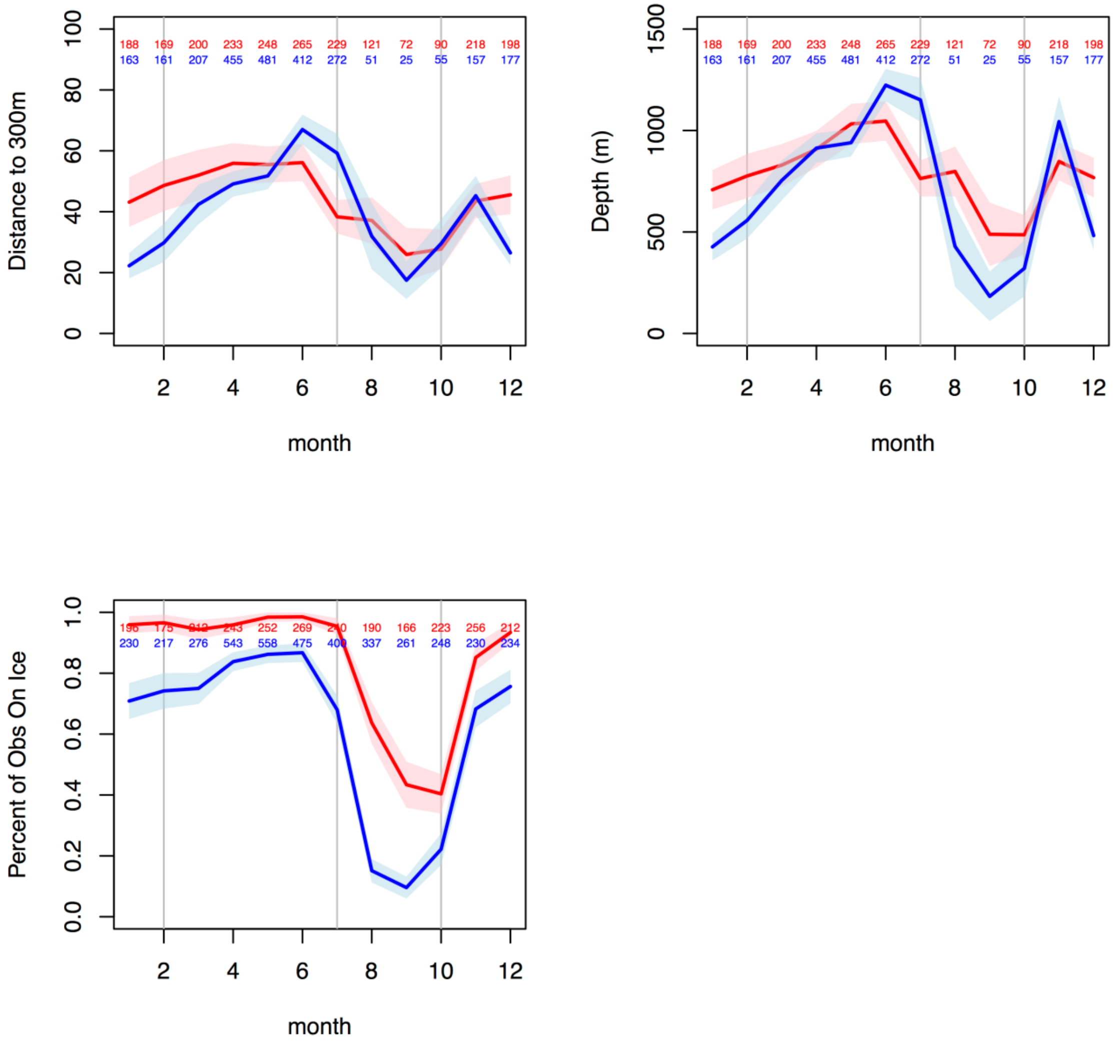


Figure 4.11. 1990s and 2000s adult female polar bear habitat use for each of four land habitat variables: Elevation, Slope, Aspect and Distance to the Baffin Island coast (from inland). Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. The small numbers in blue and red represent the number of movements for each month and decade.

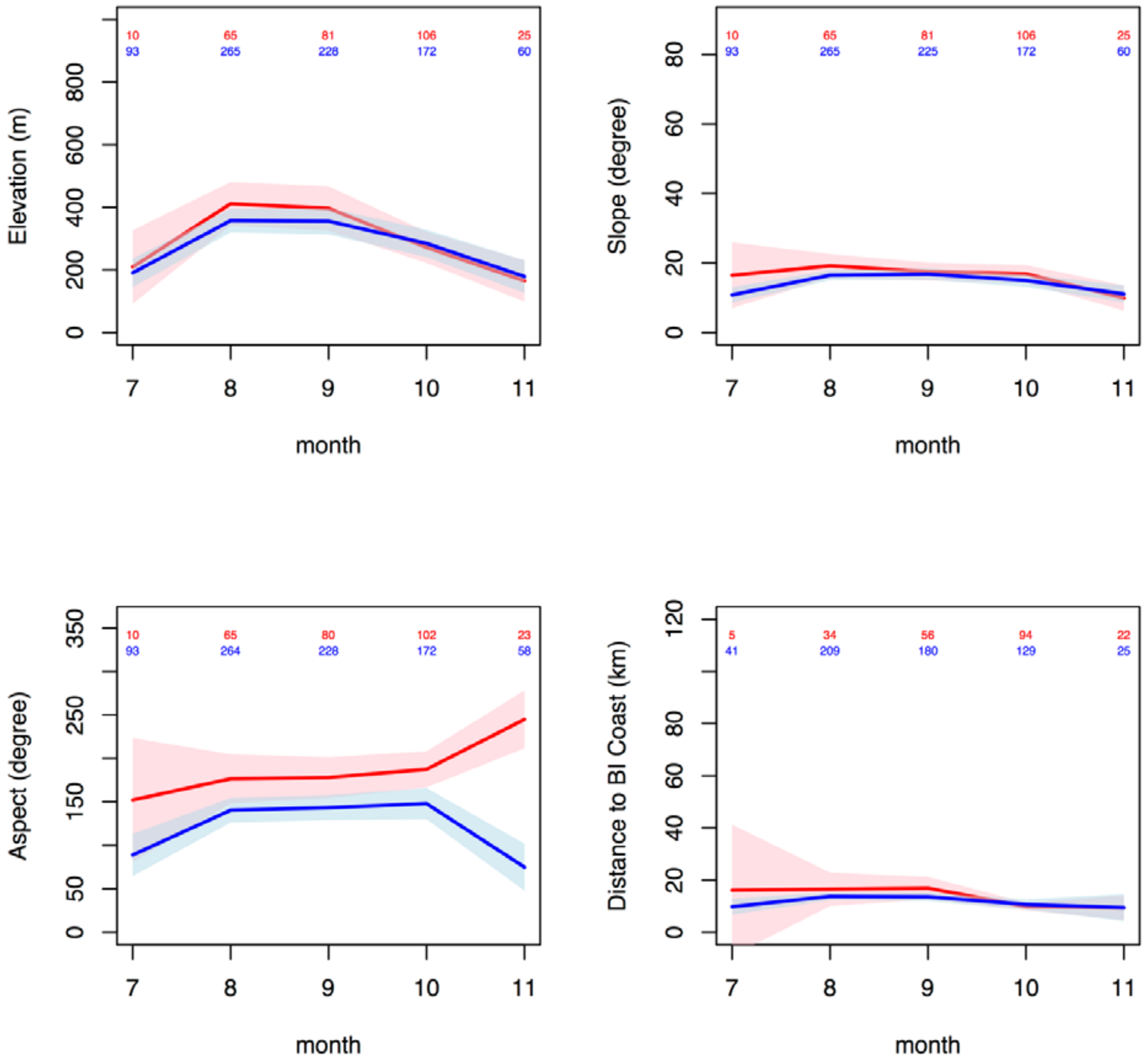


Figure 4.12. Map of ASTER Digital Elevation Model (DEM) data and IBCAO bathymetry shown for Baffin Island used in the RSF analyses. Partial tracks from a single adult female (68005) collared in 2009 in West Greenland are shown for reference.

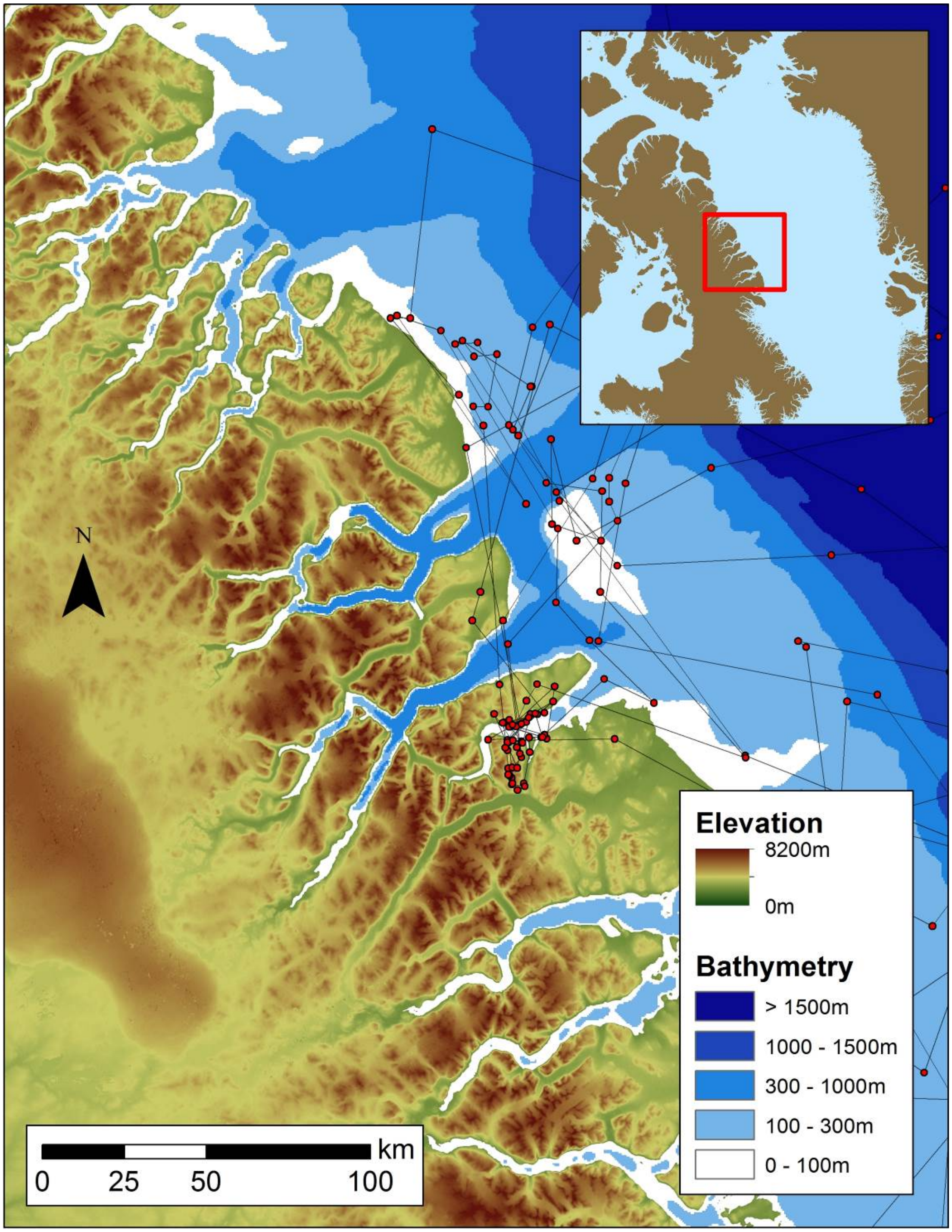


Figure 4.13. Arrival dates (on land) for adult female BB polar bears in the 1990s and 2000s. This does not include two BB bears arriving on Ellesmere Island in the 1990s. Also bears in glacial fronts in Melville Bay not included as they remained in coastal habitat year-round.

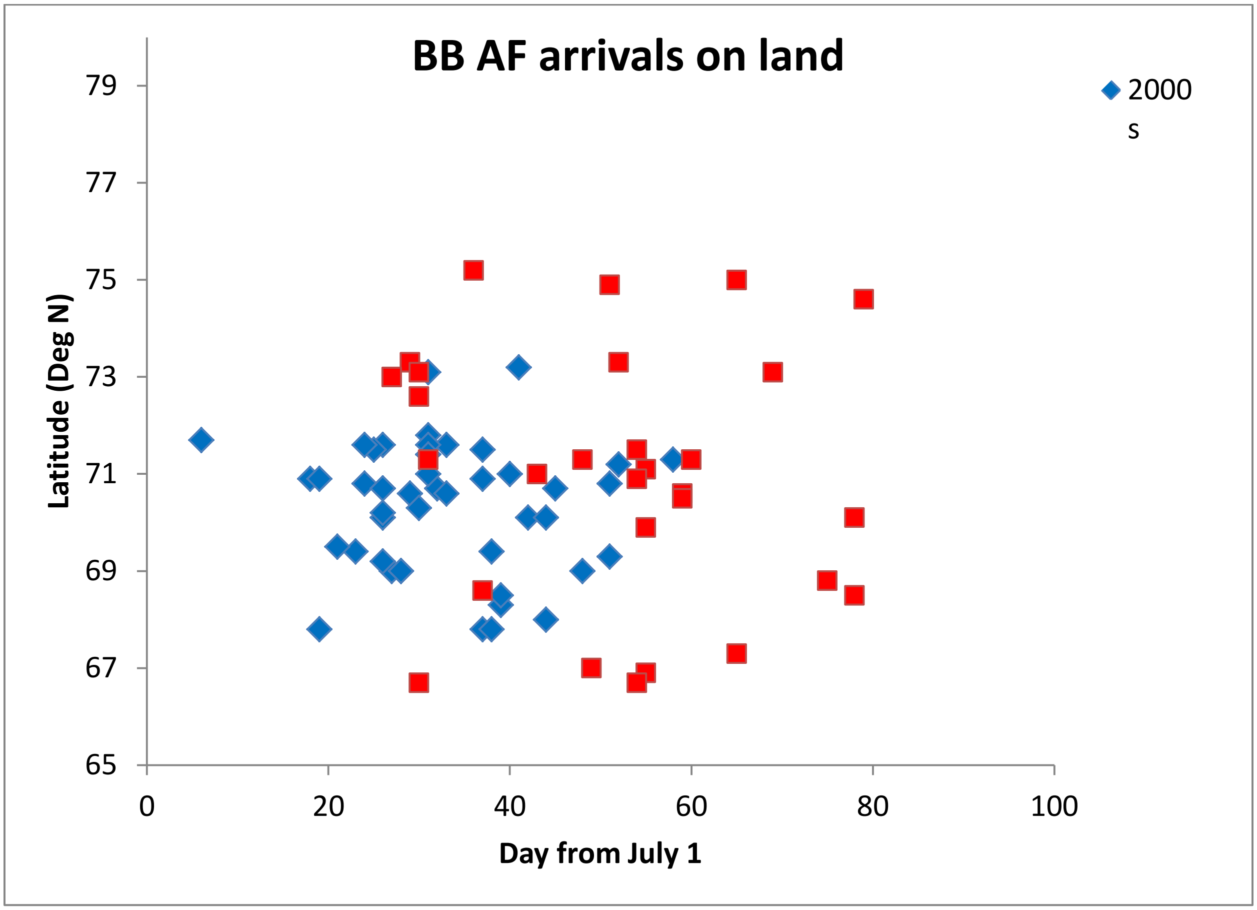


Fig 4.14. Departure dates in fall (off land) for adult female BB polar bears in the 1990s and 2000s.

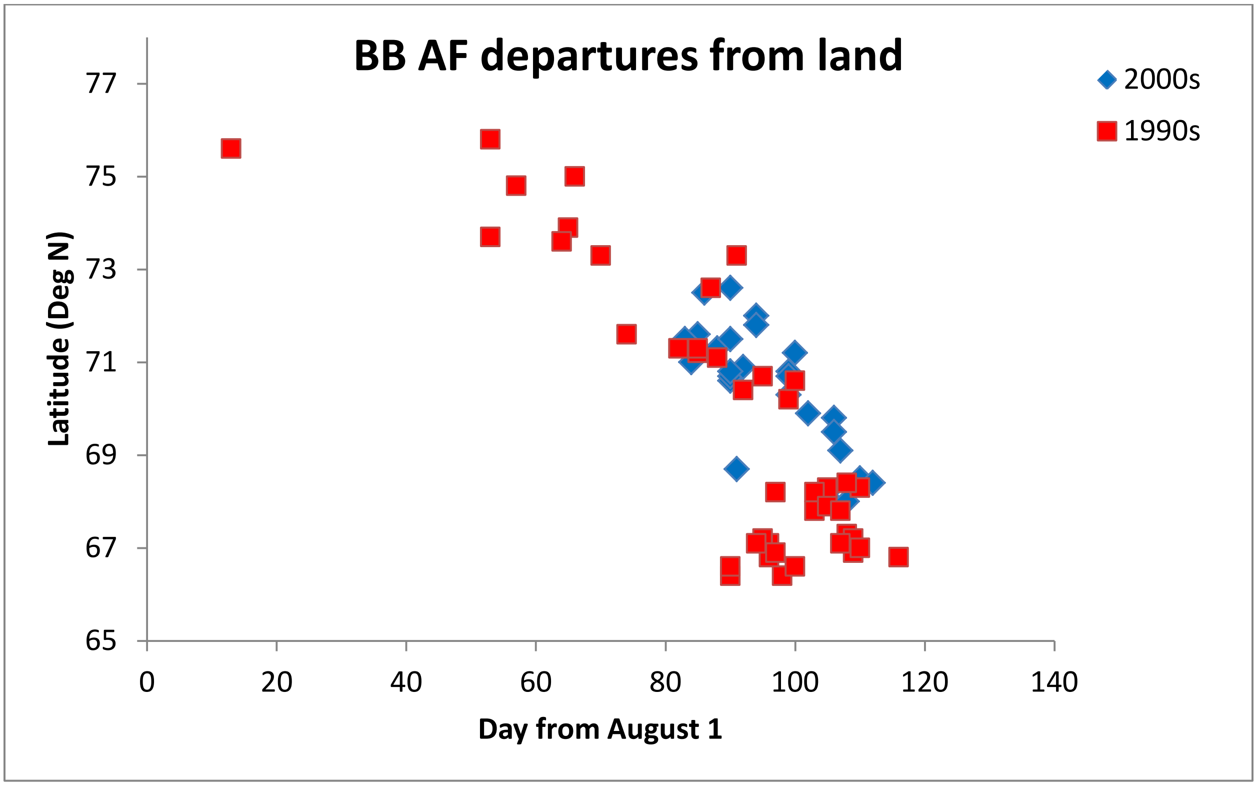


Figure 4.15. Correlations between the sea-ice retreat and advance metrics (see methods) and the arrival and departure dates on Baffin Island for adult females in both decades.

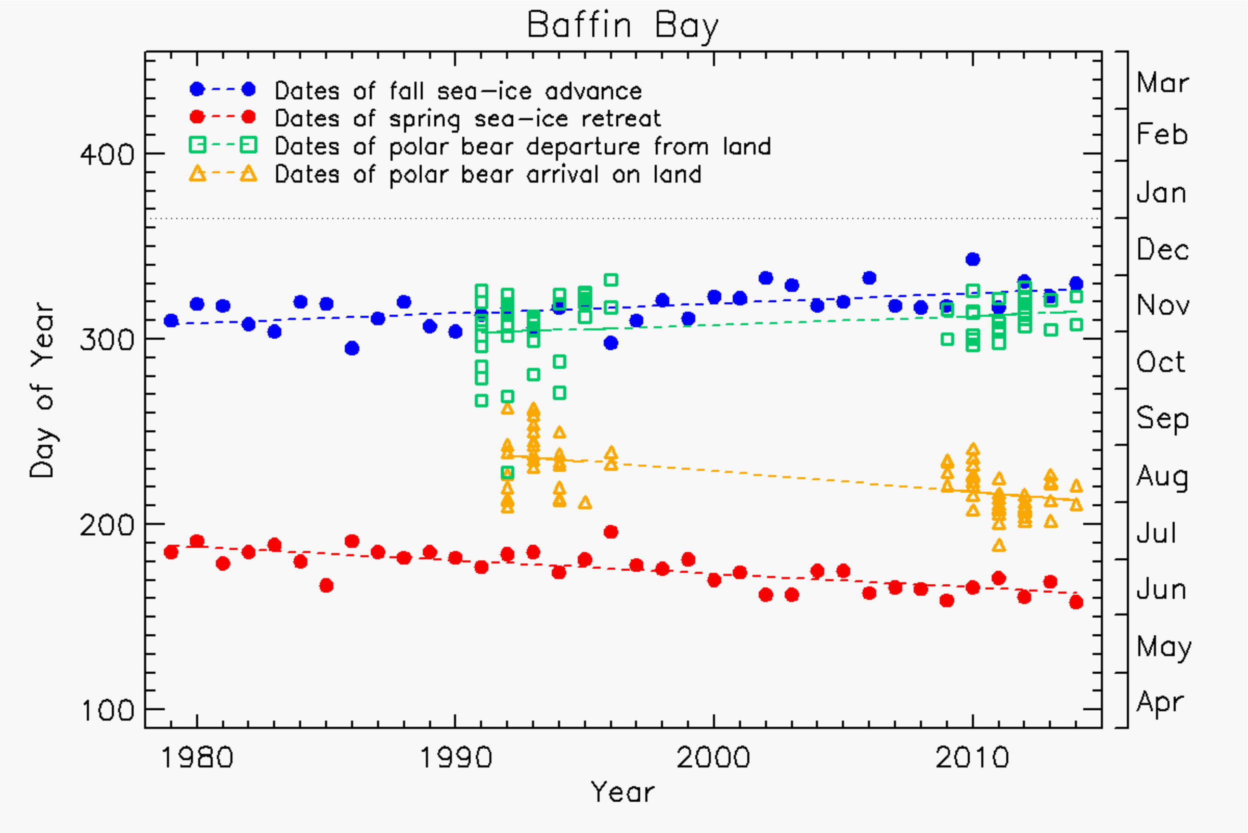
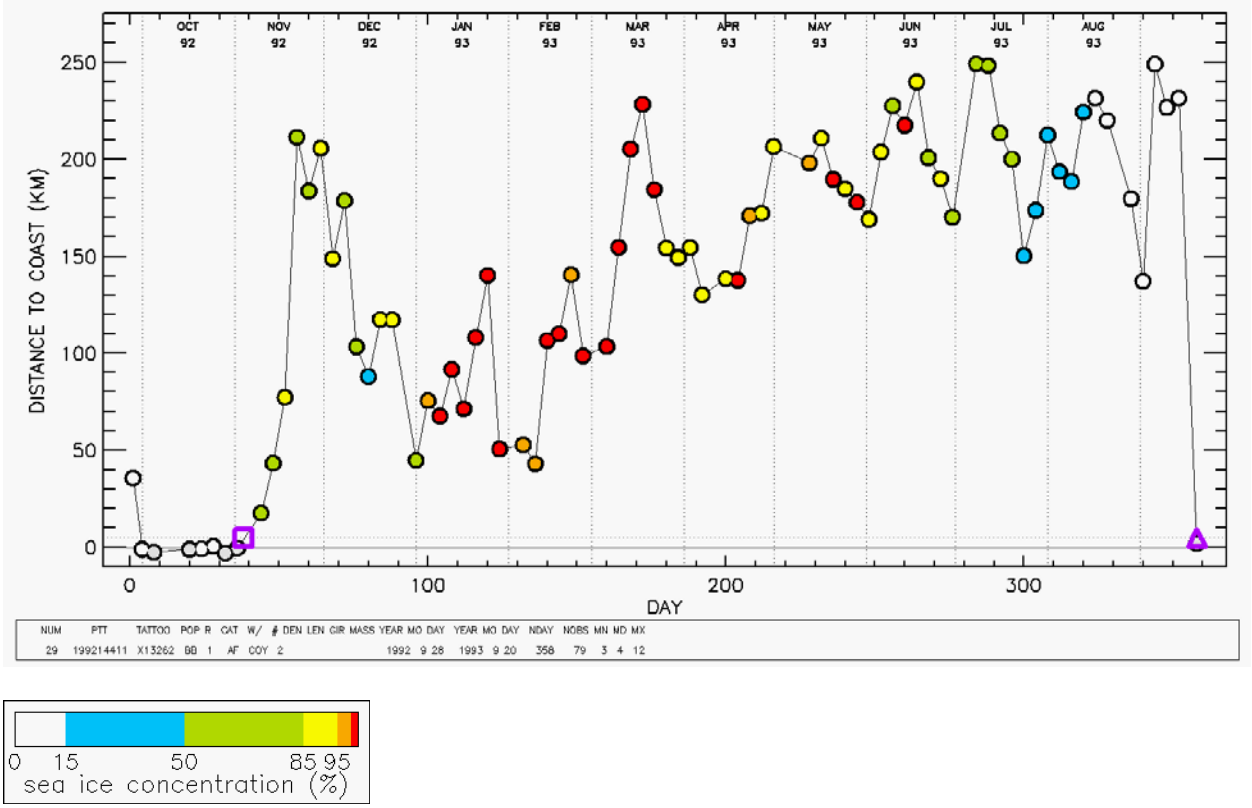


Figure 4.16. Distance to land vs. time plot for polar bear captured in 1992 (PTT 14411) showing a potential swimming event in September 1993. Purple squares denote the departure date from land on to sea ice and purple triangles denote the arrival date from sea ice/water on to land for that individual.



NUM	PTT	TATTOO	POP	R	CAT	W/	#	DEN	LEN	GIR	MASS	YEAR	MO	DAY	YEAR	MO	DAY	NDAY	NOBS	MN	MD	MX
29	199214411	X13262	BB	1	AF	COY	2					1992	9	28	1993	9	20	358	79	3	4	12

Figure 4.17. Distance to land vs. time for polar bear captured in 2011 (PTT 105808) showing an example of a swimming event in both July 2011 and July 2012. Purple squares denote the departure date from land on to sea ice and purple triangles denote the arrival date from sea ice/water on to land for that individual.

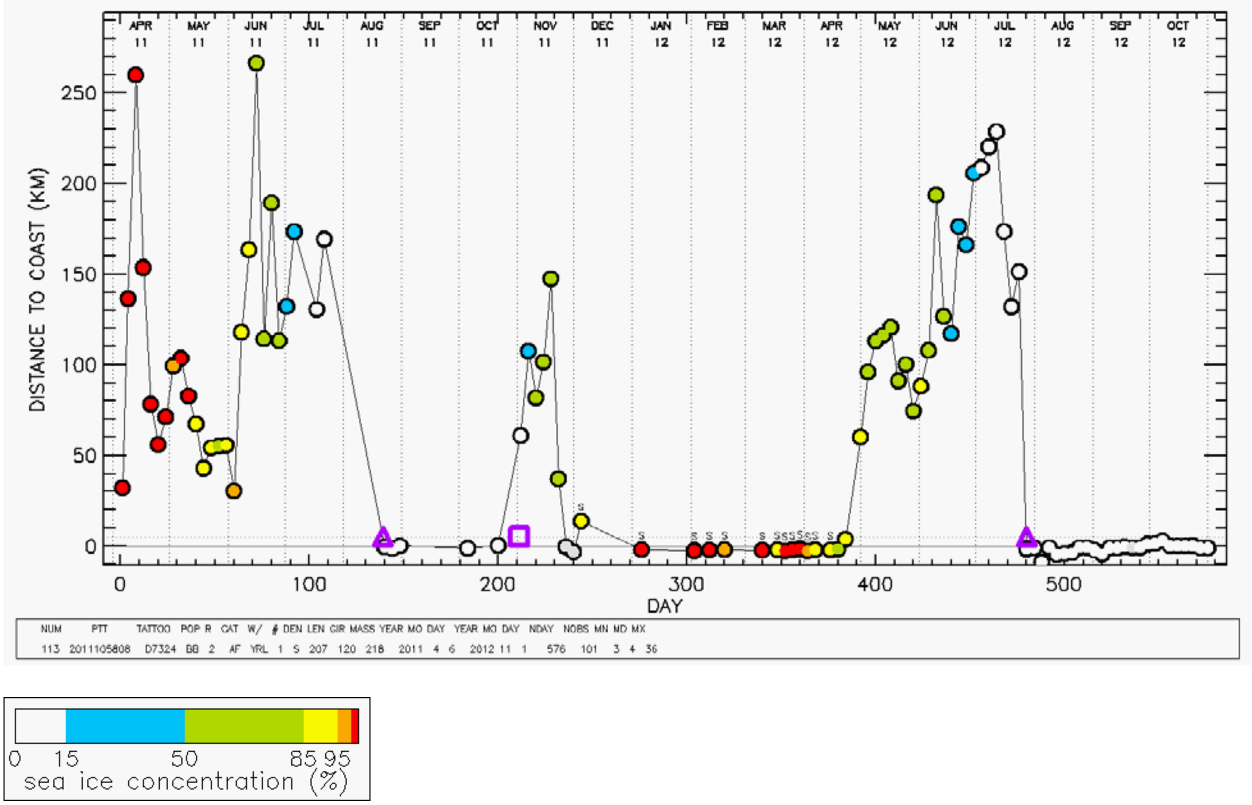
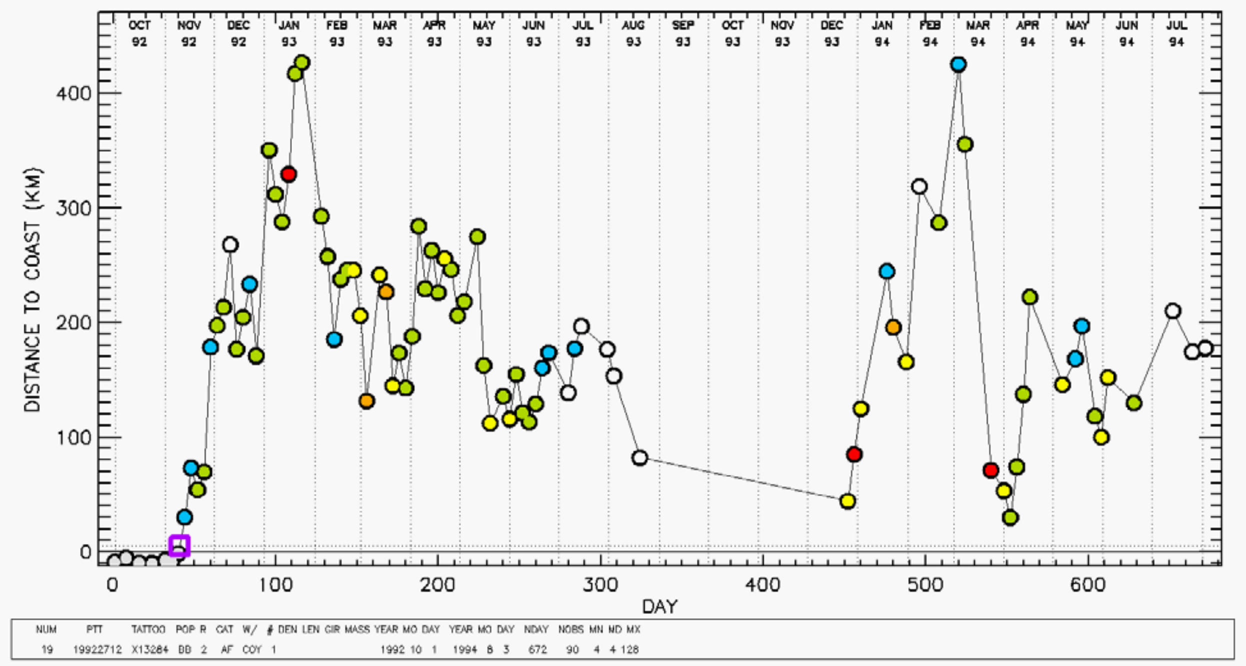


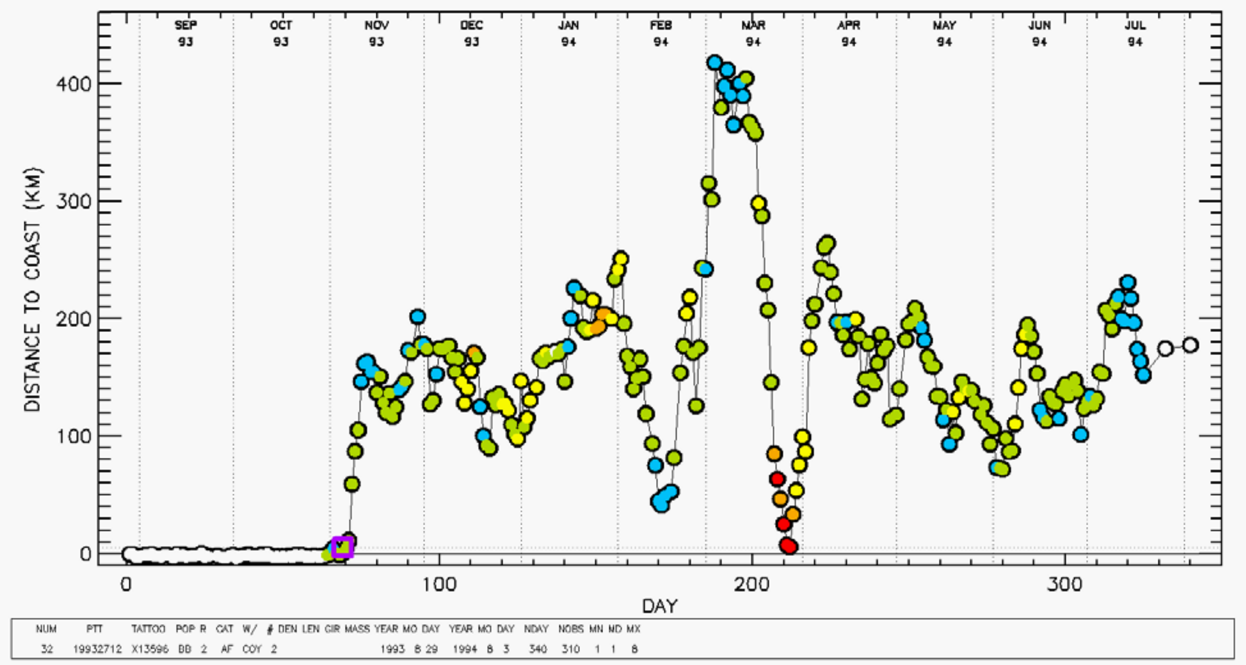
Figure 4.18. **a**) Track of adult female bear tagged in 1992 where the last three locations from the individual occur between July-August 1994 in open water 180 km from land; **b)** track of adult female bear collared in 1993 through August 1994, where the last two locations occur in open water 180 km from land; and, **c)** track of an adult female bear collared in April 2011 through July 2011. The last position is 80 km from land in <15% sea ice.

a)



NUM	PTT	TATTOO	POP	R	CAT	W/	#	DEN	LEN	GIR	MASS	YEAR	MO	DAY	YEAR	MO	DAY	NDAY	NOBS	MN	MD	MX
19	19922712	X13284	BB	2	AF	COY	1					1992	10	1	1994	8	3	672	90	4	4	128

b)



NUM	PTT	TATTOO	POP	R	CAT	W/	#	DEN	LEN	GIR	MASS	YEAR	MO	DAY	YEAR	MO	DAY	NDAY	NOBS	MN	MD	MX
32	19932712	X13596	BB	2	AF	COY	2					1993	8	29	1994	8	3	340	310	1	1	8

c)

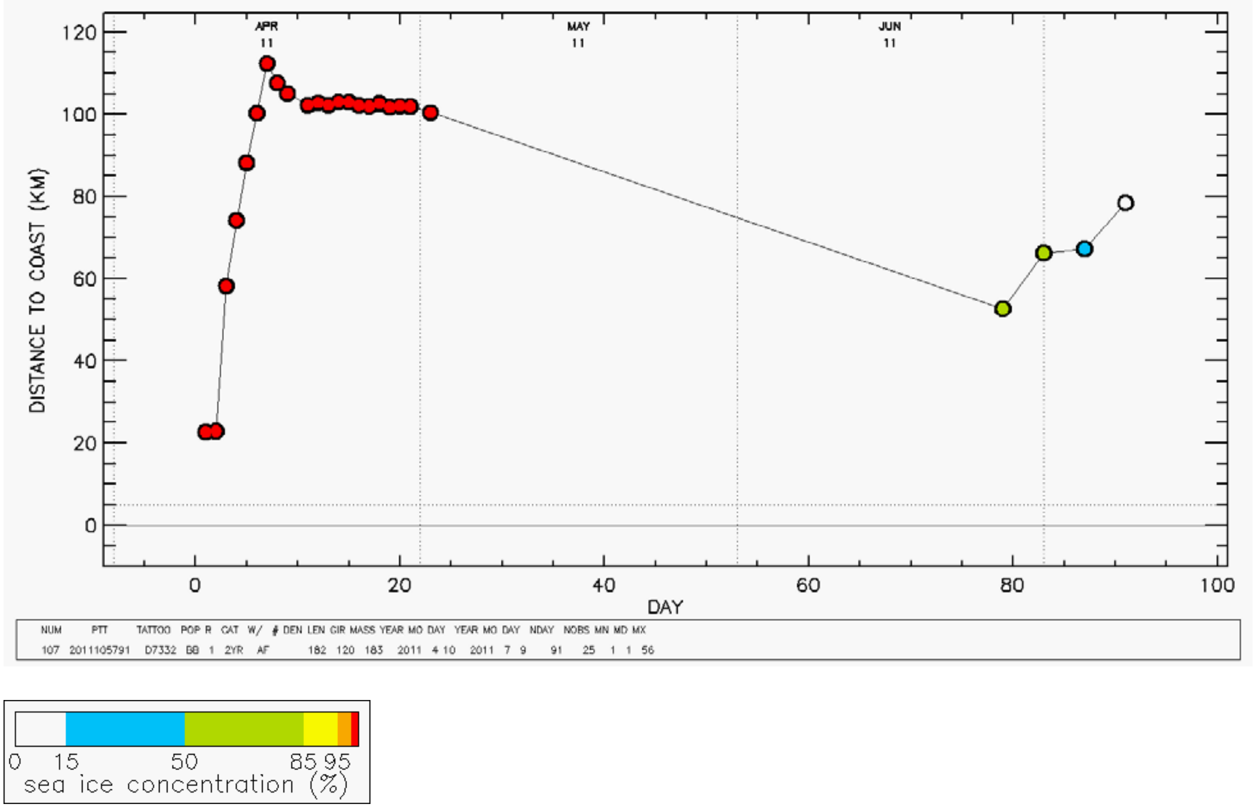

APR
11
MAY
11
JUN
11
120
100
80
60
40
20
0
DISTANCE TO COAST (KM)
0
20
40
60
80
100
DAY
NUM PTT TATTOO POP R CAT W/ # DEN LEN GIR MASS YEAR MO DAY YEAR MO DAY NDAY NOBS MN MD MX
107 2011105791 D7332 BB 1 2YR AF 182 120 183 2011 4 10 2011 7 9 91 25 1 1 56
0 15 50 85 95
sea ice concentration (%)

Figure 4.19. Maternity den locations in BB by decade.

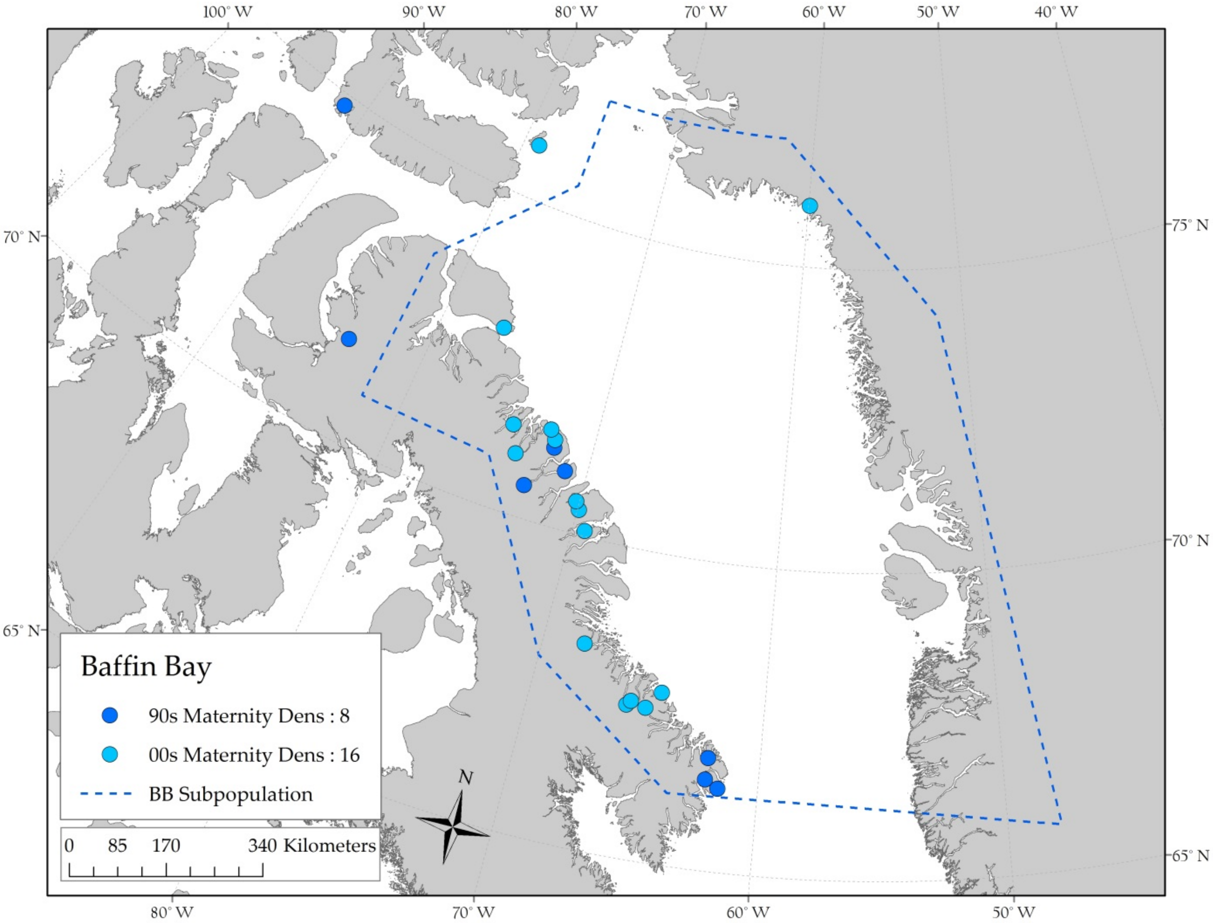


Figure 4.20. Shelter den locations in BB by decade.

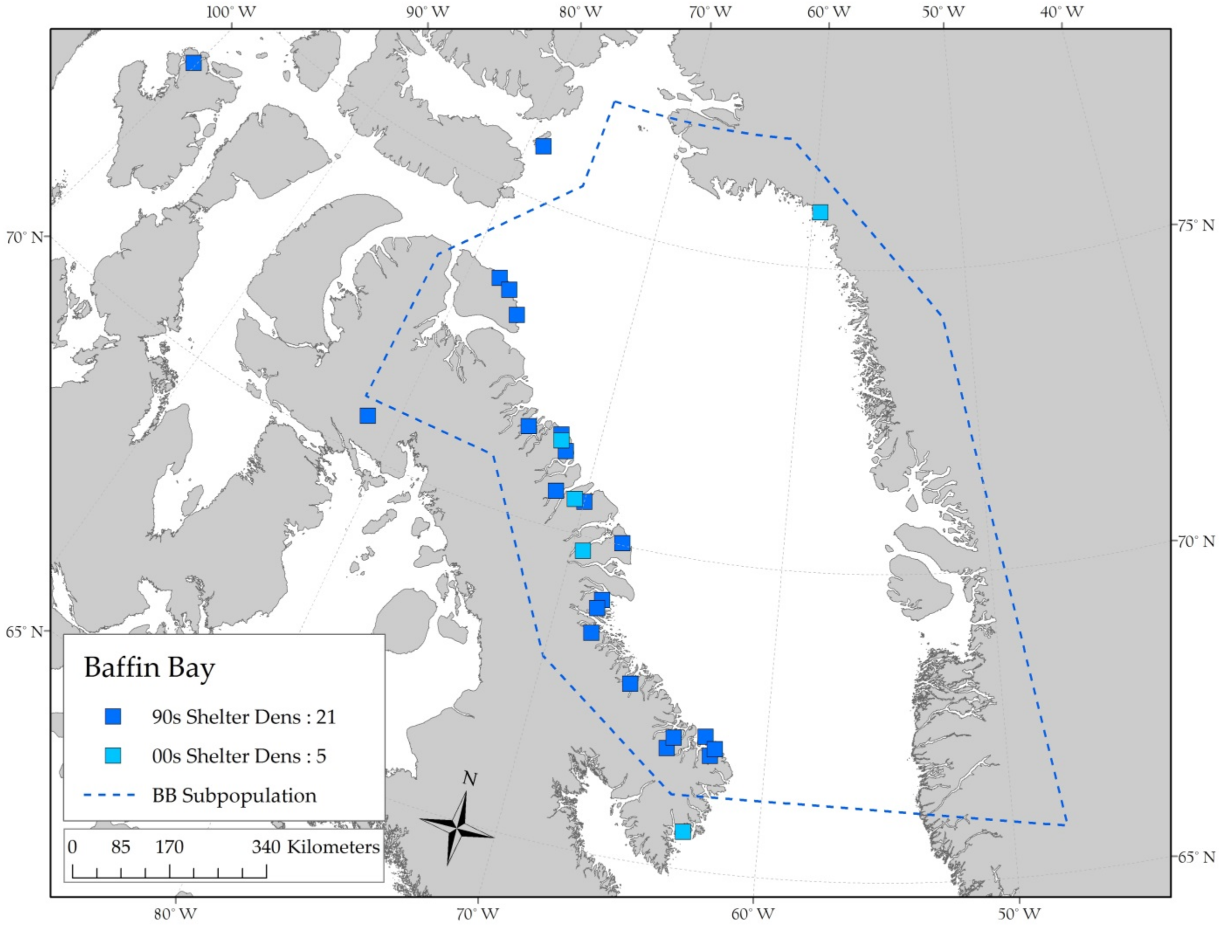


Figure 4.21. Boxplots comparing maternity den duration of Baffin Bay maternity dens (p = 0.017) (1990s: n = 8; 2000s: n = 16).

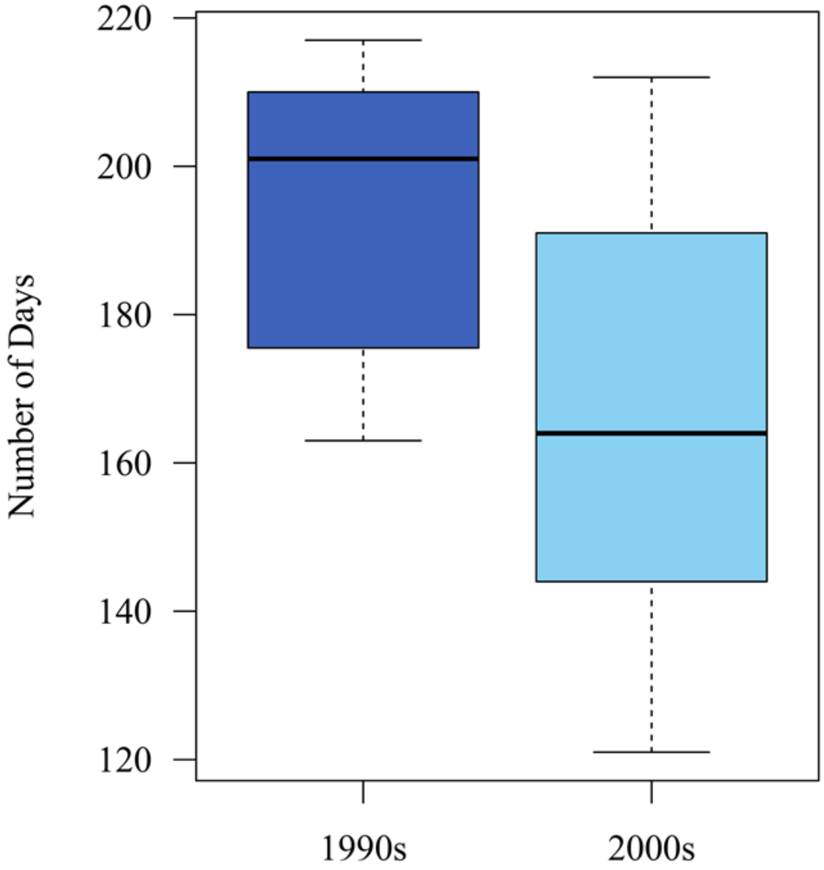


Figure 4.22. Boxplots comparing entry ($p = 0.018$) and exit dates ($p = 0.399$) of Baffin Bay maternity dens (1990s: $n = 8$; 2000s: $n = 16$)

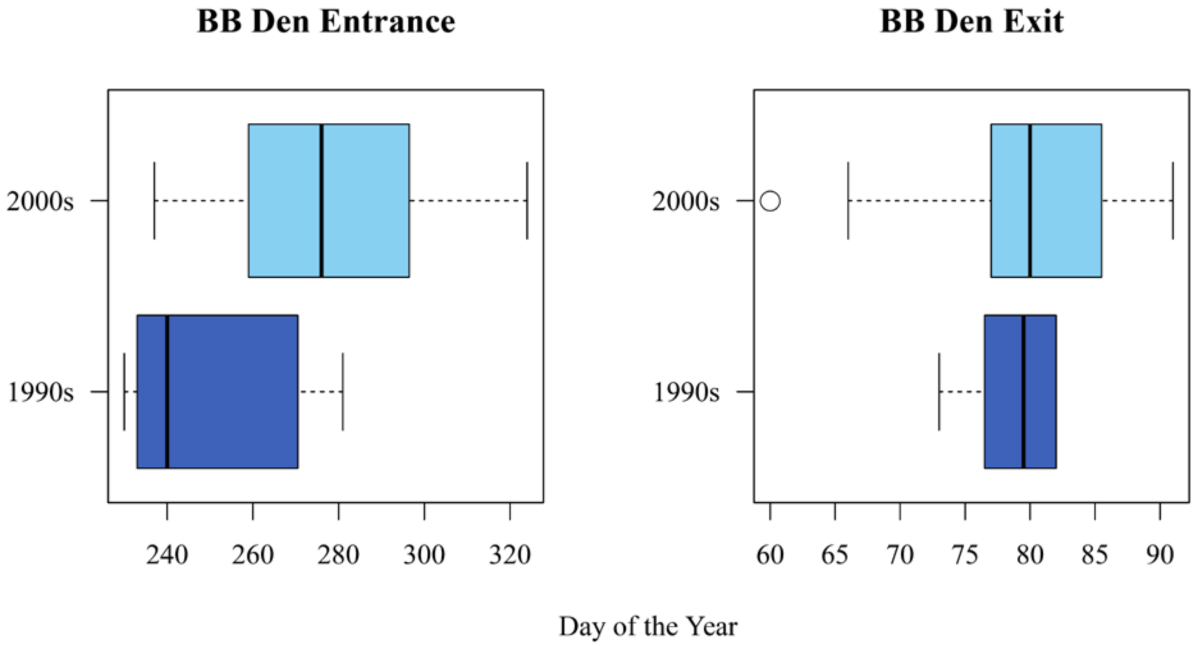


Figure 4.23. Boxplots comparing the first date on land (FDOL) of pregnant females in BB in the 1990s ($n = 8$) and 2000s ($n = 16$) (First FDOL used; $p = 0.002$).

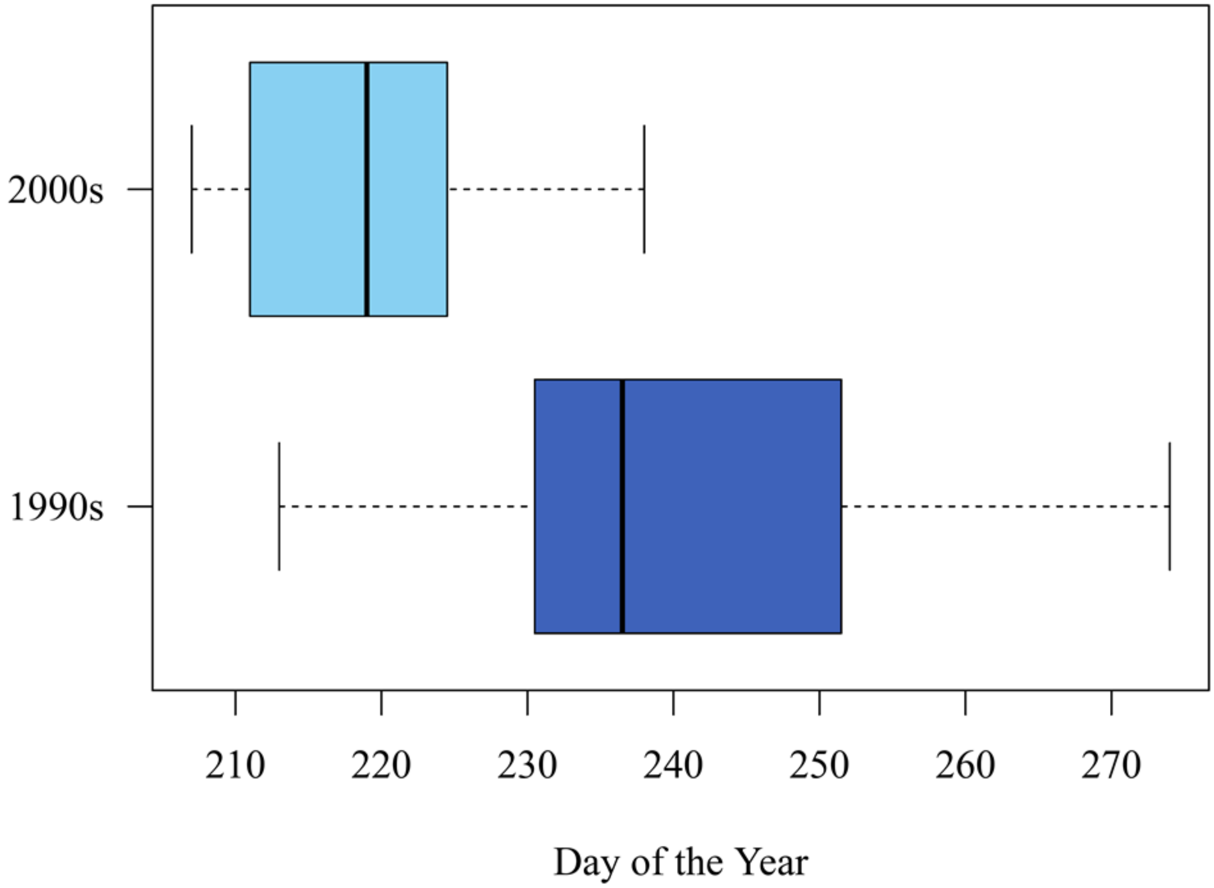


Figure 4.24. Plots comparing the aspect, slope, elevation, and distance to coast of the 1990s (*n* = 8) and 2000s (*n* = 15) maternity dens in Baffin Bay (the den on Greenland was omitted). The aspect plot consists of a compass face with lines marking the directions that dens faced. The lines are annotated with numbers noting how many dens were found at that aspect. Elevation and slope significantly differed between the two time periods (*p* = 0.003), whereas no significant difference was detected for aspect (*p* = 0.392) or distance to coast (*p* = 0.776).

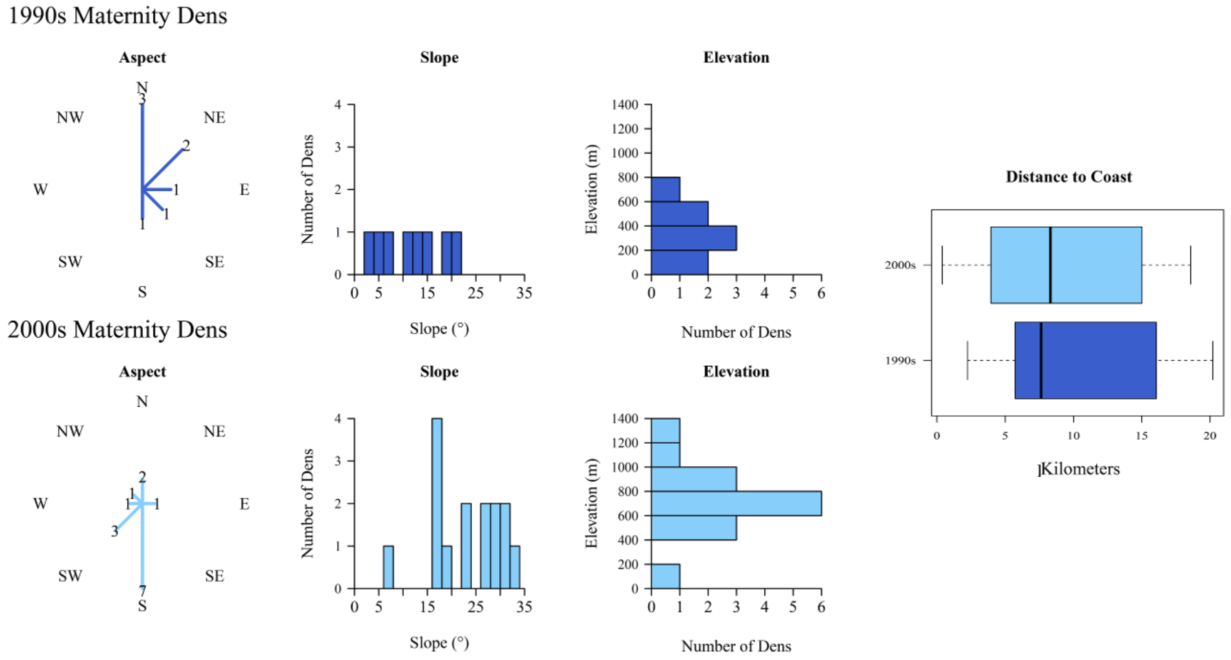
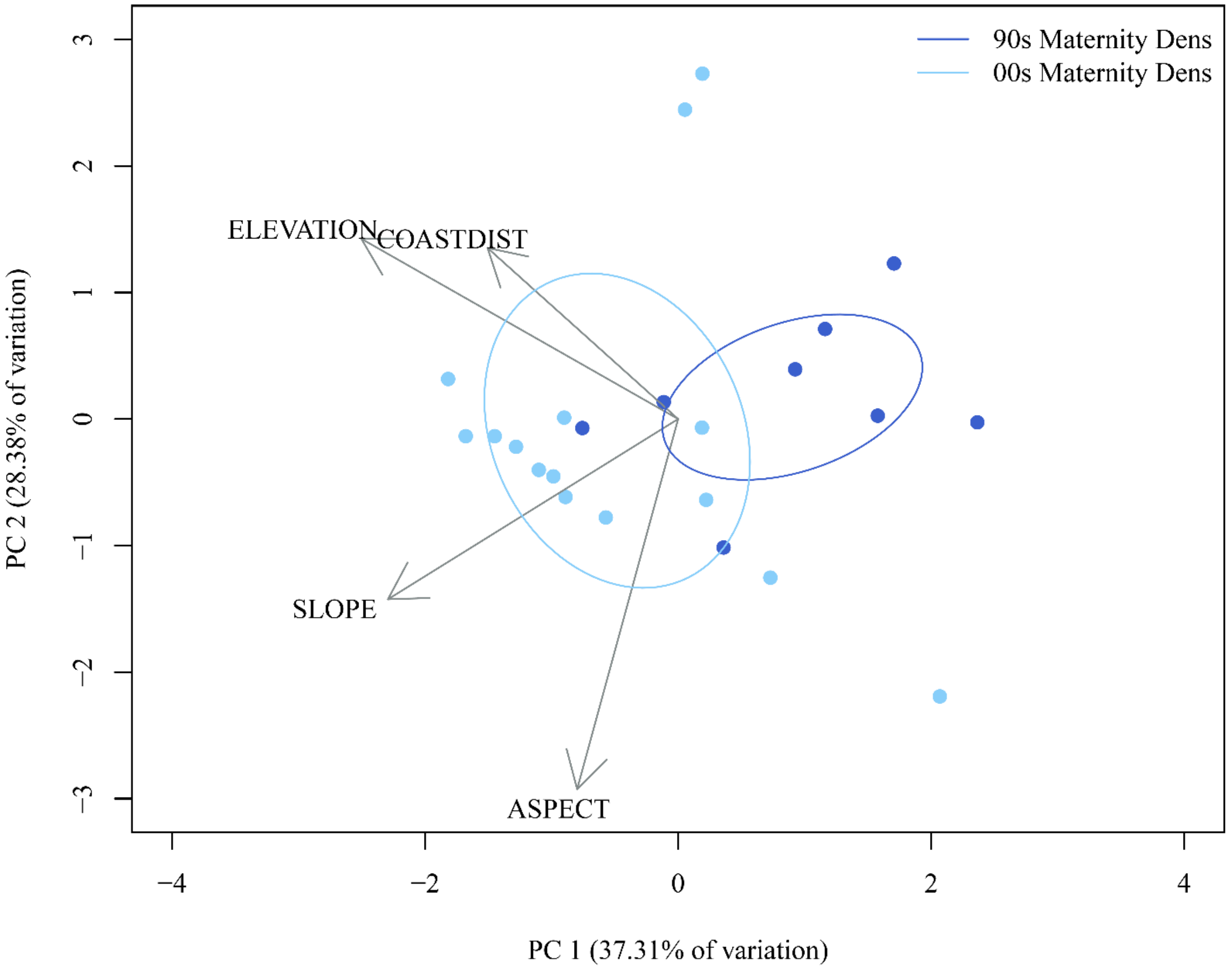


Figure 4.25. Biplot symbolizing the results for the principal component analysis (PCA) ordination of the Baffin Bay maternity dens and their habitat descriptors (elevation, slope, aspect, and distance to coast or 'coastdist'), with ordiellipses drawn around year groups (1990s and 2000s; confidence level = 0.95). The 1990s dens (*n* = 8) are symbolized by dark blue points and the light blue points are the 2000s dens (*n* = 15; the Greenland maternity den was omitted).



CHAPTER 5

GENETIC MARK-RECAPTURE STUDY OF POLAR BEARS IN BAFFIN BAY

KEY FINDINGS

- We used joint live-recapture and dead-recovery mark-recapture models to analyze data for the Baffin Bay (BB) polar bear subpopulation, with the goal of updating estimates of subpopulation size and survival. The dataset consisted of 914 physical captures 1993-1995 and 1997; 1,410 genetic samples obtained from biopsy darting 2011-2013; and 243 harvest returns of research-marked bears 1993-2013.

- The mean estimate of total abundance of the BB subpopulation in 2012-2013 was 2,826 (95% CI = 2,059-3,593) polar bears. The mean estimate of total abundance 1994-1997 was 2,173 (95% CI = 1,252-3,093) bears, similar to the estimate reported by Taylor et al. (2005). Estimates of abundance for the 1990s and 2010s are not directly comparable due to changes in sampling design and environmental conditions.

- The mean estimates of total (i.e., including harvest mortality) survival in 2011-2013 were 0.90 (SE = 0.05) for females age ≥ 2 years, and 0.78 (SE = 0.06) for males age 2 ≥ years. The time-constant estimate of total survival for a combined age class of cubs-of-the-year and yearlings, over the entire period 1993-2013, was 0.87 (SE = 0.06). Estimates of unharvested survival in 2011-2013 for females and males age ≥ 2 years were 0.91 (SE = 0.05) and 0.83 (SE = 0.06), respectively. Estimates of survival for both sexes may have included negative bias due to temporary emigration (see Chapter 3).

- We performed a comparative assessment of sampling design and environmental conditions in the 1990s and 2010s to help interpret parameter estimates, quantify potential bias, and understand trends. An evaluation of the spatial distribution of onshore captures, together with data on habitat use from satellite telemetry, suggested that more systematic live-recapture sampling, including inland areas and the backs of fjords, occurred during 2011 – 2013 compared to the 1990s. Furthermore, offshore sea ice was available to polar bears during the annual sampling periods in the 1990s, but largely unavailable in the 2010s.

- We created a geographic subset of the 2010s data based on the estimated sampling area from the 1990s to investigate the effects of sampling differences. Analyses suggested that geographically-restricted sampling such as occurred during the 1990s could result in approximately 10% negative bias in estimates of abundance. Furthermore, satellite telemetry data suggested that a potentially significant proportion of the BB subpopulation may have been located outside the sampling area or on the sea ice during mark-recapture sampling in the 1990s, although no sampling was conducted on the ice. This represents another potential source of negative bias in the 1990s abundance estimate although we were not able to quantify its magnitude.

- Differences in sampling design and environmental conditions between the 1990s and 2010s introduced variable levels of heterogeneity into recapture probabilities. This heterogeneity was due in large part to the temporary emigration of bears from the sampling area, which was more pronounced in the 1990s (i.e., more bears were located farther inland, where there was no sampling, or on the sea ice). Although moderate levels of random temporary emigration are not problematic for mark-recapture models, high or variable levels of temporary emigration combined with short live-encounter sampling windows, or non-random temporary emigration, are well-known sources of bias. Our approach of including harvest returns in the same analytical framework as live-capture data likely mitigated bias to some extent. However, the BB data were too sparse to fit MR models that explicitly estimated temporary emigration and thus minimized its effects on parameters.

- Considering statistical uncertainty in estimated parameters and evidence that the sampling design and environmental conditions likely resulted in an underestimate of abundance in the 1990s, it is not possible to conclude that the estimate of total abundance in the 2010s represents an increase in the size of the BB subpopulation. Although the 2010s abundance estimate represents the best-available information and is suitable for informing management, we cannot reliably determine the trend in subpopulation size over the 1993-2013 study period.

- The 2011 – 2013 estimate of total survival for independent females is likely too low to support a stable subpopulation, although subsequent demographic modeling (e.g., population viability analysis integrating survival, recruitment, and harvest) is necessary to estimate observed and potential (i.e., in the absence of human-caused removals) subpopulation growth rates. The low estimates of total survival for independent males may warrant concern and further investigation. However, the short time-series of live-recapture data in the 1990s and 2010s, statistical uncertainty, and potential negative bias due to temporary emigration (such bias is generally most pronounced toward the end of a study) limit inference about trends in survival or the current status of the BB subpopulation based on estimated survival rates.

5.1. Introduction

Sea-ice dynamics are rapidly changing across the circumpolar Arctic (Comiso et al. 2008, Stroeve et al. 2012), including the Baffin Bay region (Laidre et al. 2015), with a general trend toward reduced spatial extent and temporal availability of sea ice. These changes are expected to have negative impacts on sea ice dependent polar bears in the long-term (Atwood et al. 2015). In the near-term, the effects of sea-ice loss are expected to vary among subpopulations, with some of the earliest impacts anticipated in the seasonal ice subpopulations (Amstrup et al. 2008, Stirling and Derocher 2012, Rode et al. 2012, 2014, Bromaghin et al. 2015, Obbard et al. 2015, Lunn et al. 2016).

The Baffin Bay (BB) polar bear subpopulation, located between Canada and Greenland, forms part of the seasonal ice ecoregion (Amstrup et al. 2008) and is characterized by the formation and near-complete melting of sea ice each year. These sea-ice dynamics force most bears to spend the low-ice (or ice-free) summer on land. In this region, the interval between the spring sea-ice retreat and the fall sea-ice advance has increased by ~12 days per decade since 1979 (Chapter 4, Laidre et al. 2015), suggesting that polar bears are likely experiencing reduced sea-ice availability during important spring and fall foraging periods. Rode et al. (2012) reported declining body condition in BB and suggested that this finding may be due to loss of sea-ice habitat.

Abundance of the BB subpopulation was estimated as 2,074 (95% CI= 1544 - 2604) bears based on a physical mark-recapture study conducted 1993 – 1997 (Taylor et al. 2005). Taylor et al. (2005) reported relatively high survival for subadult and adult females (ages 1 – 4: 0.90, SE= 0.045; ages 5 – 20: 0.94, SE= 0.021) and estimated an unharvested population growth rate (λ) of 1.055 (SD: 0.01), suggesting strong potential for subpopulation growth relative to other demographic studies of polar bears (*cf.* Taylor et al. 2006, 2008, 2009). The combined Canada-Greenland harvest from BB was high during the early to mid-2000s ($\bar{x}$: 214 from 2001 – 2005, Chapter 8), and Peacock et al. (2012) reported lower survival rates in BB for the 2003 – 2009 period using harvest recoveries, although the estimated survival rates were characterized by high uncertainty and potential bias due to sparse data.

No new research has been conducted to update estimates of abundance since the 1993 – 1997 research, but projections using estimated abundance and vital rates from the 1990s (Taylor et al. 2005) and reported harvest suggested a declining subpopulation and a projected abundance of ~1,550 bears as of 2004 (PBSG 2010). In addition, the IUCN / Polar Bear Specialist Group (2015) assessed BB to be declining based on 100% of population viability analysis (PVA) simulations resulting in a decline in abundance after 10 years, largely attributable to the effects of harvest. In the absence of an updated demographic and ecological assessment, there has been considerable uncertainty about the current abundance and status of polar bears in BB. Given the large-scale environmental changes occurring in Baffin Bay and concerns regarding previous and current harvest levels, there was a need for new information on subpopulation status (Chapter 1).

Our objective was to obtain updated estimates of abundance and, to the extent possible, vital rates including survival for polar bears in the BB subpopulation. These estimates, combined with information on sea-ice dynamics, polar bear movements, reproductive output, body condition, and other ecological metrics, will be used to assess subpopulation status, develop management plans, and inform subsistence harvest levels. To address our objectives, we conducted a 3-year genetic mark-recapture study during 2011 – 2013. These data were analyzed together with data from physical mark-recapture research (1993 – 1995, 1997). Harvest recoveries were incorporated throughout the 21-year (1993 – 2013) study period, but no live recapture sampling occurred during the 13-year interval from 1998 – 2010.

To address concerns regarding the potential impacts of immobilization and handling on wildlife and better reflect values of northern Canadian communities, the Canada-Greenland Joint Commission on Polar Bears elected to use genetic, rather than physical, mark-recapture methods (Chapter 1). This work is part of a broad, inter-jurisdictional initiative to develop less-invasive methods (i.e., compared to physical capture) to study polar bear subpopulations. Whereas use of aerial surveys has become increasingly widespread for polar bears (e.g., Aars et al. 2009, Stapleton et al. 2014, 2016, Obbard et al. 2015), genetic mark-recapture has not been implemented at large spatial scales. However, genetic mark-recapture is an established technique that has been used in wildlife studies for decades (e.g., Palsbøll et al. 1997, Boersen et al. 2003, Boulanger et al. 2004), including small-scale studies of polar bears (Herreman and Peacock 2013). This study and concurrent research in the neighboring Kane Basin subpopulation (Chapter 10) represent the first subpopulation-scale applications of genetic mark-recapture for assessment of a polar bear subpopulation.

5.2. Materials and Methods

Study Area

The boundaries of the Baffin Bay polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all Bylot Island (66.2° N to 73.8° N) in Nunavut/ Canada and parts of West and Northwest Greenland (66.0° N to 77.0° N; Taylor et al. 2005; Figure 5.1). BB is bounded by Greenland to the east, Baffin and Bylot islands to the west, the North Water polynya to the north and Davis Strait to the south. Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N.

Baffin Bay is ice covered in winter but typically ice free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay. The last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot islands (Taylor et al. 2005), although an unquantified but probably small number remains on land in northwestern Greenland throughout the ice-retreat period (see Chapter 4). Sea ice in Baffin Bay has decreased markedly during the last few decades (Stirling and Parkinson 2006, Peacock et al. 2012, Laidre et al. 2015), with earlier spring break up and later fall formation. During the 1990s, some sea ice remained in offshore Baffin Bay during summer and was used by bears; this remnant ice was largely unavailable to polar bears in the 2010s (Chapters 3 and 4). The east coasts of Baffin and Bylot islands are characterized by high topographic relief and deep, steep-sided fjords, creating difficult conditions in which to sample bears.

Movement data of satellite-collared bears (Taylor et al. 2001), genetic analyses (e.g., Paetkau et al. 1999, Peacock et al. 2015), and recaptures and harvest recoveries of research-marked bears (Taylor and Lee 1995) have been used to evaluate subpopulation delineation between BB and other subpopulations in the Canadian and Greenlandic Arctic. Taylor et al. (2001) reported some sub-structuring of BB on a north-south gradient, and genetic analyses suggest a lack of genetic differentiation between BB from the adjacent Kane Basin subpopulation to the north, but a significant genetic difference between BB and the Davis Strait subpopulation (Paetkau et al. 1999, Peacock et al. 2015). These findings were corroborated by analyses of recent samples from BB, KB, DS and Lancaster Sound (see Chapter 2). Although some interchange occurs among BB and adjacent subpopulations including Davis Strait, Lancaster Sound and Kane Basin (e.g., Taylor et al. 2001, Chapter 3), the BB subpopulation is considered a distinct demographic unit for management purposes.

Mark-Recapture Sampling Design

Mark-recapture sampling of polar bears in BB has occurred over three periods. Early field sampling was conducted during the 1970s and 1980s (Figure 5.2) but restricted to spring-time captures on landfast ice (i.e., ice occurring nearshore; Taylor et al. 2005). Because this early sampling occurred in a different season (i.e., spring versus fall) and was spatially more variable and restricted than later sampling, we excluded early data (1970s and 1980s) from the present analyses. In addition, lack of tissue samples from early sampling precluded genetic identification, a primary method used in the current study. More systematic capture-based sampling occurred during fall ice-free seasons in 1993 –1995 and 1997, but there was no fall sampling in 1996 due to logistical and resource constraints (Taylor et al. 2005).

We completed a recent fall-time sampling session (August – October) from 2011 to 2013. This session differed from sampling in the 1990s in several important ways. First, sampling was conducted by biopsy darting to obtain tissue for genotyping individuals, rather than via physical capture and tagging. Second, new information obtained via satellite telemetry on the movements and spatial distribution of bears in Baffin Bay and Kane Basin, collected during the 1990s sampling session and during 2009 – 2010, was used to improve study design, with the objective of improving sampling coverage of the BB subpopulation and thereby reducing heterogeneity in recapture probabilities. In Baffin Bay, heterogeneity during fall sampling is likely to result primarily from the spatial distribution of bears in relation to sampling effort. Although bears are concentrated along the Baffin Island coast during the ice-free season, some individuals travel significant distances inland, move to higher elevations, or remain on offshore ice where access for sampling is difficult (Ferguson et al. 1997, 2000, Taylor et al. 2001, Chapter 4). Bears also may segregate by age and reproductive status. For example, adult females with cubs tend to select fjords, avoiding offshore islands and coastal regions where densities of adult males are higher (Ferguson et al. 1997; Chapter 4), and pregnant bears select inland and upland denning habitats where they are less available for capture (Chapter 4).

Although Taylor et al. (2005:205) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island, examination of the satellite telemetry data from adult female polar bears collared in the 1990s indicated an under-representation of bears in fjords and inland regions and offshore pack ice (see Chapter 3). This finding suggested that capture effort during the 1990s was concentrated on islands, along the coastline, and near the mouths of fjords (Figure 5.3). This pattern was particularly noticeable in central and northern Baffin Island, where no captures were recorded beyond the mouths of fjords during the entire study period. We conclude that sampling was spatially restricted to a portion of the subpopulation's fall range during the 1990s, thus potentially increasing heterogeneity in recapture probabilities, which can bias estimates of subpopulation size and demographic parameters.

Analysis of the 1990s telemetry data also showed that a potentially significant proportion of collared bears remained on offshore sea ice during the fall onshore sampling period (see Chapter 3). The proportion of collared bears present in the sampled area each year was estimated based on the total number of collars that were transmitting during the capture sampling period. To evaluate movements and fidelity with respect to the onshore sampling area in years $t + 1, 2, \ldots k$, we only used data from bears that were captured in the onshore sampling area and fitted with collars during year t (i.e., we excluded data from the year of capture, because bears were captured onshore and their locations following capture were not random). In addition to the mark-recapture sampling in the fall, some bears were captured and fitted with collars in the spring. Data from bears captured in spring of year t were incorporated into summaries of movement and fidelity for fall of year t.

During the 1990s, there were three years with telemetry data available during mark-recapture sampling (1993 – 1995). In those years, 0 – 23% of collared bears transmitting during the sampling period (n = 1 – 13 collared bears / year) were present in the sampling area. By contrast, during the 2010s, 67 – 85% of collared bears were present in the sampling area during the sampling period (n = 6 – 12 collared bears / year) (see Chapter 3). Although sample sizes are small and telemetry data are limited to adult females, this finding suggests that seasonal fidelity to the sampling area changed significantly between the two mark-recapture sampling periods. This follows well with the change observed in the sea-ice habitat during those periods (Chapter 4). Further investigation showed that a substantial portion of 1990s adult females outfitted with satellite collars were on the offshore pack ice of Baffin Island or on remnant ice around Lancaster Sound and Devon Island. In contrast, there was little sea-ice habitat available in summer during the 2010s, and bears were concentrated on Baffin Island and Northwest Greenland. Polar bears that used offshore sea ice during the 1990s were unavailable for capture, and to the extent that individual bears consistently used offshore sea ice throughout the 1990s sampling period, these bears would not have been enumerated in the subpopulation estimate. These issues were less problematic during the 2010s due to the expanded onshore sampling area and the lower availability of offshore sea ice.

For sampling on Baffin and Bylot islands during 2011 – 2013, we defined sampling strata to guide effort and improve survey coverage and efficiency. Stratification primarily was based on satellite telemetry data obtained from adult female polar bears collared during fall and spring along eastern Baffin Island (1993-1997) and in spring in W and NW Greenland in 2009 and 2010. We summarized location data by proximity to the coastline and used the proportion of locations in different inland zones (e.g., 0 – 5 km inland, 5 – 10 km inland) to inform stratification. We delineated a high-density stratum including the coastline and offshore islands, extending 5 km inland; a moderate-density stratum including inland regions 5 – 10 km from the coastline; and a low-density stratum extending up to 30 km inland (Figure 5.4). We attempted to allocate roughly 65%, 25%, and 10% of helicopter search effort in the high-, moderate-, and low-density strata, respectively, to efficiently sample the study area. We set *a priori* guidelines to systematically distribute inland search effort along the entirety of the islands.

It was not feasible to sample bears that may have remained on offshore ice floes in either decade. However, long-term trends in sea-ice conditions in Baffin Bay have resulted in significant reductions in offshore ice during the fall in the 2010s, relative to the 1990s (Laidre et al. 2015; see Chapter 4). Thus, the presence of bears on offshore ice during the recent sampling session was considerably reduced (*cf.* Chapter 3 Figures 3.18, 3.19, and 3.20). Additionally, using real-time data on sea ice (see Field Methods below) and the location of telemetry-instrumented bears, sampling during the 2011 – 2013 period was timed to coincide with the period when sea-ice cover was at a minimum and most collared bears were on land.

Various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that an unquantified but presumably small number of bears in the BB subpopulation spend the summer in the Qimmusseriarsuaq / Melville Bay area of Northwest Greenland, rather than moving with the retreating sea ice and summering on Baffin and Bylot islands (see also Chapter 2). To account for this portion of the subpopulation, we extended our sampling efforts to include this region during fall (Figure 5.4). Because satellite telemetry indicated that polar bears were not present during fall in the Melville Bay region in the 1990s (Taylor et al. 2001), NW Greenland was not sampled during fall during the 1990 physical mark-recapture study

(Taylor et al. 2005). In sum, study design for 2011 – 2013 was intended to (1) maximize sample size; (2) sample bears across the known seasonal range of the subpopulation; (3) efficiently allocate sampling effort based on expected densities across the study area; and (4) accommodate the spatial segregation of sex, age, and reproductive classes. These considerations are important to reducing potential bias in estimates of demographic parameters, particularly abundance, from mark-recapture studies (Pollock et al. 1990, Williams et al. 2002).

Field Methods

Capture, sampling, aging, and data collection protocols for bears marked during 1993 – 1995 and 1997 are described in Taylor et al. (2005). For genetic mark-recapture sampling from 2011 – 2013, field work was timed to coincide with minimum sea-ice cover in Baffin Bay based on Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery (https://worldview.earthdata.nasa.gov) and Canadian Ice Service maps (August – October in all years). The locations of collared bears were also reviewed prior to field work to confirm the timing and location of bears coming ashore. During field sampling, locations of collared bears were not reviewed and telemetry equipment was not used to help locate bears.

In Nunavut (Canada), sampling in 2011 – 2013 was conducted via remote biopsy darting (Pagano et al. 2014) using two helicopters (Bell 206 LongRangers). The helicopters began sampling at opposite ends of the study area; one proceeding north to south, and the other south to north until they overlapped. We sampled Baffin and Bylot islands from September 4 – October 14, 2011; August 26 – September 29, 2012; and August 20 – October 11, 2013. With the exception of 1993 (August 23 – October 8), sampling started and ended earlier than research in the 1990s. Approximately 300 hours of total helicopter flying time was allocated each year, including travel time to and from daily start points and refueling caches. To the extent possible, effort was allocated according to the *a priori* stratification and criteria noted previously. However, we modified sampling where necessary based on terrain and weather conditions that limited access to some areas. Flight paths during searches were recorded via GPS to facilitate *post-hoc* assessment of the distribution of search effort. We made a concerted effort to search inland and at high elevations. We searched most fjords along their entire lengths and a majority (>95%) of offshore islands.

As outlined above, a small number of polar bears summer in the Melville Bay area of NW Greenland (e.g., Born et al. 2011), and we also searched these areas to collect biopsies. The areas between 74° 34´ N and 76° 46´ N (i.e., Melville Bay *sensu lato*) were searched during 4-11 September 2012 and 7-17 September 2013 (a total of nearly 60 hours of active on-effort search) using an AS350 Ecureuil B3 helicopter. Coastlines, mountain sides, inland nunataks, glacier fronts and most offshore islands up to 40 km from the coast were searched. On each flight, three dedicated observers and the pilot searched for bears; and on several flights, a local polar bear hunter assisted in the search. Flight paths were recorded using GPS.

In 2011, we initially tested two types of biopsy dart to assess reliability and the quality of samples yielded (Figure 5.5). The Pneudart DNA dart (Pneudart Inc., Williamsport, USA) was highly reliable at yielding good quality tissue samples (95% success). In contrast, the Pneudart Biopsy dart was unreliable (<50% success), although it did provide good quality samples when successfully deployed. Consequently, we used DNA darts exclusively during the remainder of the field work. Biopsy darting was quick and minimally invasive; the time between spotting a bear and obtaining a sample was typically < 2 minutes. Since bears were not chemically immobilized, they could be safely darted in locations that would have been unsuitable for capture due to the risks of drowning or falling. The only exception was for bears sighted in the water. Most bears found in water were directed to land before darting since the darts do not float well. A small number of bears were darted while still in the water using Pneudart DNA Marker darts, which float for a short time in calm waters and may be retrieved.

Bears showed little or no reaction to the impact of a biopsy dart, and no visible mark was left in most cases. Immediately after darting, each bear was allowed to move away from the helicopter before the dart was retrieved. Darts were coated in fluorescent paint to aid retrieval. When working in deep snow, we also rolled a length (~ 20 cm) of flagging tape around the shaft of each dart. This tape unrolled during flight and helped in locating darts when they sank into the snow. Because biopsy darts leave no visible mark, there was potential for repeated sampling of the same individuals within a single field season. To minimize duplicate sampling, daily searches were limited to areas not previously searched. Where possible, we used natural barriers to polar bear movements for the purpose of delineating daily break points in search effort. When sampling members of a family group consisting of an adult female with cubs-of-the-year or yearlings, the 1 to 3 dependent young with each mother were distinguished based on differences in size or other features (e.g., marks on fur) to avoid sampling the same individual twice. When it was not possible to distinguish between cubs, we used DNA-Marker darts to distinguish among litter mates. This dart takes a tissue sample and leaves a temporary dye mark that can be used to distinguish sampled from non-sampled individuals.

A limitation of biopsy darting was the challenge of sampling cubs-of-the-year. Although many cubs were large enough to be sampled, doing so involved a risk of separating them from their mother. Unlike physical capture methods, in which the adult female is first immobilized and can be used as an ‘anchor point’ around which cubs are captured, members of a family group that are being biopsy darted may move in opposite directions. With the rugged and steep terrain along Baffin and Bylot islands and Melville Bay, cubs can quickly lose sight of their mother and are at risk of injury or separation. For these reasons, only about half of the cubs-of-the-year that we encountered during 2011 – 2013 were biopsy darted, although we recorded the sighting of all individual cubs for calculating proportions of females with cubs and mean litter sizes.

Following retrieval, darts were checked to ensure they contained a suitable tissue sample. Each sample was divided into two parts for storage and labelled with a unique biopsy number. Samples were initially stored cooled or frozen. Samples sent for DNA extraction were taped onto an absorbent card, placed into individual envelopes and later oven dried for submission, or stored in vials with DMSO. For each bear encountered, we recorded GPS coordinates and data on location, weather conditions, habitat, behavior, body condition (thin, average, and obese bears; see Stirling et al. 2008), group size, and estimated age-class and sex.

We remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) from the air at a range of 3 – 7 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males), body shape and proportions, presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under the tail in females). Field notes assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified for some bears from other encounter records in which the individual was captured and physically examined, or by using genetic identification to assign membership to a known family group (Appendix B). We were able to assess the accuracy of estimating age-class and sex of polar bears using this sample of known age-class bears (Appendix B).

Harvest recoveries

We assembled data from bears killed in the harvest or as problem bears during the 1993 - 2013 study period. Between 1993 and 2010, recoveries of research-marked bears in the harvest were detected by the return of ear tags and / or lip tattoos from hunters in Canada and Greenland. Between 2011 and 2013, recoveries of bears that were physically tagged or genetically marked (i.e., without physical tags) were detected by the return of ear tags or lip tattoos (and satellite radios in Greenland); or from genetic monitoring of harvest (i.e., genotyping of harvested bears). Although we expected a majority of bears marked in Baffin Bay to remain within BB, previous studies of tag recoveries in the harvest and satellite telemetry suggest that some bears emigrate from the subpopulation (Stirling et al. 1980, Taylor and Lee 1995, Taylor et al. 2001, Taylor et al. 2005). To account for marked individuals that were harvested outside BB (Burnham 1993), samples were collected from bears harvested in Baffin Bay as well as all surrounding subpopulations (Lancaster Sound, Kane Basin, Gulf of Boothia and Davis Strait, Foxe Basin; Burnham 1993). For each harvested bear, data including age, sex, date and location of kill were recorded. Canada's quota-based mandatory harvest reporting system was in place throughout the 1993 – 2013 research period. Greenland implemented a quota system and made improvements to the reporting system in the mid-2000s and, in 2012, instituted a mandatory harvest reporting system for collection of a tissue sample and premolar tooth for age determination (Appendix D).

Genetic Analysis

To identify bears physically marked (with ear tags and lip tattoos) during the 1990s that survived and were subsequently encountered during the genetic mark-recapture session (2011-13), we genotyped tissue samples from all bears sampled in the 1990s except: 1) bears that were recorded as harvested between 1993 and 2011, 2) bears whose known or estimated age would have been greater than 35 years in 2011, and 3) 33 bears that met the above 2 criteria, but lacked tissue samples. In total, this dataset consisted of 650 individuals marked in the 1990s that would have been ≤ 35 years old and had not been harvested by the time genetic sampling began in 2011; the 33 bears lacking tissue samples were assigned to unique attribute groups to acknowledge they were unavailable for genetic recapture during the 2011-2013 sampling period (see below).

Dried biopsy samples and harvest specimens (frozen or in ethanol) were sent to Wildlife Genetics International (Nelson, B.C., Canada) for analysis using protocols previously validated for bears (Kendall et al. 2009). DNA was extracted from ~ 3mm^2 pieces of tissue with QIAGEN DNeasy Blood and Tissue Kits (http://www.qiagen.com/). Most samples consisted of a plug of a skin and sub-cutaneous tissue, which provided ample material for DNA extraction and residual tissue for future analyses. In a small proportion of cases, the available sample consisted of a tuft of hair. DNA was extracted from hair samples using approximately 10 guard hair roots or 30 pieces of underfur. In a few cases, where a biopsy sample contained no visible tissue, DNA was successfully extracted by soaking the barbed needle from the biopsy dart in the lysis mix (QIAGEN buffer ATL + proteinase K).

To select markers for the analysis of individual identity, we used allele frequency data from 1,771 polar bears for which complete 20-locus genotypes existed before the genetic mark-recapture study began (Government of Nunavut, unpublished data). We ranked the 20 microsatellite markers in the dataset by expected heterozygosity. The 8 most variable markers that could be analyzed together in a single sequencer lane were selected for use. These surpassed the required standard for marker variability (HE = 0.80; Paetkau 2003). In addition to the 8 microsatellite markers, we analyzed sex on every sample, using a *ZFX*/*ZFY* marker. This 9th marker roughly halved the match probability (assuming a balanced sex ratio), even for close relatives, as well as providing replication of sex data for individuals that were sampled more than once.

The analysis of individual identity followed a 3-phase approach. Phase 1 was a first pass of all extracted samples using the 9 selected markers (*G10B*, *CXX20*, *G10H*, *G10P*, *145P07*, *MU50*, *MU59*, *G10X* and *ZFX*/*ZFY*). Samples that failed at > 6 of 9 markers on the first pass were set aside and did not proceed further in the analyses. Previous experience has shown that such samples are prone to errors and run out of DNA before generating a complete (phase 2) and reproducible (phase 3) genotype (D. Paetkau, pers. comm.).

The first pass was followed by a cleanup phase in which data points that were weak or difficult to read the first time were re-analyzed. During cleanup we used 5 µL of DNA per reaction instead of the 3 µL was used during first pass. At the conclusion of the cleanup phase, the remaining samples (99.5%) had high-confidence scores for all 9 markers. In cases where the genetic sex result contradicted the reported sex based on field assessment, genetic sex was checked using a second independent marker (*amelogenin*; http://www.ncbi.nlm.nih.gov/pubmed/7695123), thus confirming the results, and ruling out the possibility that a mutation at a particular marker was to blame. In all cases, results from the second marker confirmed that the field data was the source of error.

The third and final phase of analysis was error-checking, following the published protocol of reanalyzing the mismatching markers in highly similar pairs of genotypes (Paetkau 2003). This error-check included genotypes from the 4,657 polar bears in the database, plus published data from 473 individuals (Paetkau et al. 1999). The error-checking protocol functions on the principle that when ≥ 2 samples are genotyped from a given individual, and when 1 of those genotypes contains an error, the result is a pair of genotypes that match at all-but-1 marker (a ‘1MMpair’). Less commonly, 2MM-pairs are created when 2 errors have been made in the genotypes of the samples from a given individual.

An important distinction with this protocol is that it is designed to ensure accurate individual ID, and has been proven to do so with a high degree of efficiency (Kendall et al. 2009), but it is not intended or expected to correct errors when just 1 sample has been genotyped from a given individual. In addition to re-analyzing mismatching markers this protocol also involved the inclusion of additional markers for some samples. Finally, we searched the dataset for genotype matches that seemed unlikely based on our field data. In each case, 3 extra markers were added to the genotypes to lower the probability of chance matches between individuals. The extra loci confirmed all of these matches. Once the genotyping and error-checking was complete, we defined an individual for each unique 9-locus genotype.

In total, 1610 biopsy samples (99.2% of those submitted for analysis) were successfully extracted and genotyped. A further 868 samples (99.3% of those submitted) from bears harvested in BB and surrounding subpopulations 1993-2013 were genotyped successfully, with success defined as satisfying the lab’s visual and peak-height criteria for high-confidence scoring at each of the 9 markers.

In addition to the genotyping errors that were targeted during error-checking, DNA-based datasets are prone to a second source of error, when match probabilities are so high that some individuals have identical genotypes. Calculated match probabilities provide no practical insight into the risk of sampling individuals with matching genotypes, because the calculations are so dependent on the assumptions made about the degree of relatedness among the sampled individuals. We therefore used the direct, empirical approach of extrapolation from the observed mismatch curve (Figure 5.6). We expect to see roughly order-of-magnitude decreases in the number of pairs of individuals whose genotypes match at increasing numbers of markers (Paetkau 2003). In our dataset the slope of this curve was reasonably true to that rule of thumb. From this curve, it is estimated that we would have sampled ~ 0.3 0MM-pairs (individuals whose genotypes matched at 9 markers) in this multiyear dataset of 4,657 individuals; a very small risk of error in proportion to the size of the dataset. In addition to reducing the risk of sampling individuals with the same genotype, another benefit to having such a powerful marker system was realized during error-checking, where the amount of time required to reanalyze the mismatching markers underlying 1MM- and 2MM pairs was trivial in proportion to the scale of the project, because there were so few such pairs.

Statistical Analysis

We used the Burnham (1993) model in Program MARK (Cooch and White 2015) to analyze joint live-recapture and dead-recovery data from the 21-year (1993 – 2013) study period in BB. The Burnham model is an extension of the Cormack-Jolly-Seber live encounter model that facilitates the inclusion of dead-recovery data (i.e., combining the CJS model with the Brownie-Seber dead-recovery model) and estimates survival probability (S; the probability of surviving interval t to t+1), recapture probability (p; the probability of live-recapturing a marked animal), dead reporting probability (r; the probability that a bear is killed by humans and reported to authorities), and fidelity (F; the probability that a bear does not permanently emigrate from the sampling area, and is therefore available for capture in future years). The inclusion of dead recoveries allows for estimation of true survival (i.e., a biological survival rate that does not include permanent emigration), because whereas live-encounter models only measure the probability of remaining alive and within the live-recapture area, the inclusion of dead recoveries from throughout Greenland and Canada in the Burnham model allow for estimation of true survival independent of potential emigration. More importantly, inclusion of dead recovery data increased the amount of information available on the fates of individual bears in the BB subpopulation, likely decreased susceptibility to bias because the sampling mechanism for dead recoveries was different from live-recapture sampling, and allowed us to estimate survival during intervening years between live-recapture periods (i.e., 1998-2010; Peacock et al. 2012). Similarly, with respect to dead-recovery models, live recaptures provide large amounts of additional data, allowing for more precise estimation of survival than would be possible using dead recoveries only (Cooch and White 2015).

Estimates of demographic parameters from mark-recapture studies may be impacted by temporary emigration (e.g., Peñaloza et al. 2014). Completely random temporary emigration is generally reflected in the parameter p for long-term studies (i.e., the probability of recapture reflects both the probability that an individual is in the sampling area on a given occasion, and the probability that the individual will be recaptured conditional on being in the sampling area; Burnham 1993, Barker and White 2001). However, for shorter studies with high or variable rates of temporary emigration, especially when the probability of temporary emigrants remaining outside the study area is non-random (e.g., if the probability of being an emigrant at sampling occasion *t*, depends on emigration status at sampling occasion *t*-1) the parameter *S* may be susceptible to bias as well. Bias typically increases toward the end of the study, and is referred to as terminal bias, because bears that leave the study area during the final years have no opportunity to return and be resampled, and thus cannot be distinguished from individuals that died or emigrated permanently (Peñaloza et al. 2014). Such terminal bias can suggest spurious correlations with environmental covariates, particularly for short studies, if habitat quality declines toward the end of the study and there is concurrent terminal bias (Devineau et al. 2006). With the Burnham model, the inclusion of dead recovery data can mitigate these issues to some extent (Peacock et al. 2012), as can formal incorporation of telemetry data (Peñaloza et al. 2014). More complex mark-recapture approaches are available that can theoretically model temporary emigration with respect to the sampling area, further mitigating potential bias (e.g., the Barker model, Barker and White 2001; multistate models with unobservable states, Schaub et al. 2004). However, these models require large datasets and can be difficult to fit in practice (Converse et al. 2009). We conducted simulations in Program MARK (White and Burnham 1999) and determined that existing mark-recapture data from BB were inadequate for measuring temporary emigration (i.e., the *F* and *F'* parameters in the Barker model, or the *a"* and *a'* parameters in the Barker robust design). Simulations suggested that, using the Burnham model, adult survival (*S*) and recapture (*p*) probabilities were relatively unbiased in the presence of low to moderate levels of random temporary emigration, under which conditions the estimates of *p* reflected the product of recapture probability and presence on the study area (T. Arnold, unpubl. data). Simulations suggested that if temporary emigration was non-random or temporally variable, survival rates would be negatively biased (especially if dead recovery rates were low, as for adult females);

however, the product of capture probability and presence on the study area exhibited relatively small bias, the directionality of which tended to underestimate abundance.

Whereas live-capture sessions are assumed to be instantaneous, dead recoveries may occur year-round between the live-capture sampling periods. For the BB data, there was some temporal overlap of live recapture and dead recovery periods, but the assumption of non-overlap between live and dead recovery periods was generally met. We considered harvests prior to August 31 in year t as occurring after the live recapture sampling period in year $t – 1$, whereas harvests after September 1 were assumed to have occurred after live-recapture sampling in year t. This coding protocol resulted in no instances of bears being coded as harvested before being observed alive during the sampling period in year t.

We analyzed data and built models in Program MARK (White and Burnham 1999). We set up the analysis using a Barker modeling framework to provide flexibility if the data proved sufficient to model temporary emigration, but initially constrained parameters of the Barker model to correspond to the simpler Burnham model (i.e., we fixed the following parameters to 0: $F'(i)$ [the probability a bear not at risk of capture in i is at risk of capture in period i + 1], $R(i)$ [the probability that a bear surviving from occasion i to i + 1 is resighted alive between i and i + 1], and $R'(i)$ [the probability that a bear dies during i to i + 1 without being reported dead is re-sighted alive between i and i + 1 before its death; Barker 1997, 1999). This approach allowed for the possibility of altering model structures, in the event that we elected to explicitly model temporary emigration (F') or wanted to simulate the consequences of constraining this parameter to 0, rather than allowing random temporary emigration to be incorporated in the parameter p (Burnham 1993, Barker 1997, 1999). We included harvest data through 2013 and compiled individual capture histories with the live capture and dead encounter data.

Whereas researchers during the 1990s study period were able to estimate age by physical examination and by counting annular rings on a bear's extracted premolar (Calvert and Ramsay 1998), we did not physically capture bears during 2011 – 2013, and individual age classes were assessed from the air. As such, there was uncertainty in our assignment of bears to age classes (Appendix B). Hence, during mark-recapture modeling we elected to simplify age structure relative to previous work (*cf.* Taylor et al. 2005), resulting in the following age classes: cubs-of-the-year (*coy*), yearlings (*yrl*), and individuals ≥ 2 years old (*age 2+*). Dependent young (coy and yrl) were assumed to be aged without error because of clear differences in the body size of these two age classes.

We identified a limited number of candidate sub-model structures for the parameters S, p, r, and F in the Burnham model. Because we expected that survival would vary among age classes (e.g., Taylor et al. 2005), we incorporated age structure in all 10 candidate sub-models (Table 5.1). We hypothesized that survival of coy would differ from yrl and thus constructed a three age-class structure (coy, yrl, age 2+). Because many of the coy that were sighted during the 2011-2013 sampling period were not biopsy darted, we also examined a two age-class structure in which coy and yrl were pooled for estimation of S. We hypothesized that the sexes would differ in S for the age 2+ class, primarily due to sex-selective harvest (2:1 male-to-female harvest ratio), but not for coy and yrl since they are dependent on their mothers for survival. Given sparseness of data, we examined time-constant structures for S, and a structure allowing temporal variation in adult survival across three sampling epochs corresponding to the live recapture and dead recovery periods (i.e., 1993 – 1997, 1998 – 2010, and 2011 – 2013). We did not have a biological reason to suspect that temporal changes in survival aligned with these sampling epochs (e.g., that survival exhibited a step change between 1997 and 1998). This structure aligned with changes in sampling design and available information on changes in bear movements, and allowed assessment of whether estimates of *S* varied whether they were based on combined data (epochs 1 and 3) or dead recoveries only (epoch 2). We specified the time-constant and epoch-based structures for *S* by constraining the design matrix in Program MARK, while maintaining full temporal structure on adult survival within the parameter index matrices (PIMs). This approach facilitated modeling of environmental covariates (see below) and future use of random effect models or Markov Chain Monte Carlo (MCMC) approaches to explore annual variation in survival (Cooch and White 2015).

We created two annual covariates, *ice transition* and *ice area* (both standardized about the mean and standard deviation) to explore relationships between *S* and environmental conditions. First, we hypothesized that the duration of the summer sea-ice transition period over the continental shelf of BB (*ice transition*; i.e., the time between break-up and freeze-up; see Chapter 4 for description of derivation of sea-ice metrics) would have a negative relationship with survival for the age 2+ classes, such that increasing duration of the ice transition period would be correlated with decreasing survival (e.g., Regehr et al. 2007). Similarly, we hypothesized that the area of ice over the continental shelf of the BB subpopulation during late spring (*ice area;* mean area of sea ice during May and June) would have a positive relationship with survival of age 2+ bears. Duration of the ice transition period and ice area over the continental shelf increased and decreased, respectively, during the 21-year study period (*ice transition*: $\beta = 2.7$ (days), $t = 3.1$, $P = 0.005$; *ice area*: $\beta = -1{,}362$ (km^2), $t = -4.2$, $P < 0.001$: -0.70). Because we did not sample many coy during 2011 – 2013 and we could only estimate survival from 7 cohorts, data were insufficient to explore relationships between time-varying covariates and the survival of dependent bears.

Preliminary analyses suggested a positive relationship between age 2+ female survival and duration of the sea-ice transition period. However, we believe this finding was an artifact of study design. No live recapture sampling occurred in 1996, which coincided with the heaviest observed sea-ice conditions during the period 1993-2013 (mean$_{1993\text{-}2013}$: 190 days; 1996: 129 days; standardized effect size: -2.18). Furthermore, the following year 1997 was the last year of live-capture data and the proportion of adult females in the sample was low compared to previous years, which we hypothesized was due to the higher levels of temporary emigration in the 1990s associated with limited geographic sampling and the availability of offshore sea ice (see Chapter 3). The combination of extreme environmental conditions in 1996, lack of live-capture sampling in 1996 and 1998-2009, and auxiliary data suggesting high levels of temporary emigration and nonrandom sampling in the 1990s led us to the conclusion that the data were likely insufficient to evaluate year-to-year variation in survival, especially toward the end of 1990s live-capture sampling. We explored the relationships between environmental covariates and S in other years by setting the 1996 value of standardized covariates to 0 (i.e., the standardized mean), and found there was not a significant relationship between sea ice and female survival. Based on these considerations we excluded sea-ice metrics from further consideration for evaluating temporal variation in S, although we explored the robustness of these results using additional post hoc analyses (see Discussion).

We created 12 candidate structures to model recapture probability (Table 5.2). We modeled coy to have the same p as females, since they remain in family groups as yearlings and are recaptured with their mothers. However, we hypothesized that p of age 2+ males (including p of male yearlings, recaptured at age 2 after break-up of family groups) would differ due to spatial segregation of bears onshore by sex and age classes (Taylor et al. 2005), and we included this two-group structure for p (*family*, including 2+ females and dependent young; and age 2+ males) in all candidate sub-models. We evaluated two candidate structures for temporal variability in p: 1) given differences in sampling protocols, search effort and survey teams between the 1990s and 2000s, p was allowed to differ between the 1990s and 2010s live-capture sampling epochs (additive or interactive effects with family); and 2) a fully-time varying structure (additive with family) for p (i.e., allowing for year-to-year variability), given that both sampling effort and environmental conditions varied significantly among years. In all candidate structures, p was fixed to 0 for the years 1996 and 1998-2010. This was necessary because p represents the probability of live-recapturing a previously-marked bear, and no live-capture sampling occurred in these years.

We hypothesized that timing of sea-ice break-up over the continental shelf may influence the distribution of polar bears, and thus p, in fall. We evaluated the standardized spring sea-ice transition (50% threshold) date in some structures (*spring ice*, Table 5.2). Exploratory analyses suggested that proximity to the coastline also may explain variability in p. Specifically, contingency tables suggested that bears initially captured inland were more likely to be recaptured inland; and conversely that bears initially captured near the coast were more likely to be recaptured in coastal regions. We created a binary geographic covariate based on an individual's first capture location, using a threshold of 2 km from a smoothed coastline (*coastline*; i.e., the coastline excluding deep fjords, see Chapter 3). We considered two temporal structures for this covariate: 1) given the apparent differences in sampling between the 1990s and 2010s, we estimated separate effects by epoch; and 2) we included the covariate effect only for the 2010s, as exploratory analyses suggested this epoch showed the strongest relationship between p and the covariate. A small number of bears (n = 33) initially captured in the first sampling period that could be alive by 2011 (<35 years of age and not harvested) were not genotyped due to inadequate tissue samples or an absence of samples. Because all sampling was conducted via remote biopsy darting and subsequent genetic analysis during 2011 – 2013, these individuals had zero probability of live recapture in this period, so we assigned these bears to unique attribute groups to fix $p = 0$ during the last 3 years of the study. As part of a study evaluating spatial ecology and population delineation (Ferguson et al. 1997, Taylor et al. 2001), a sample of adult female bears was outfitted with satellite collars in BB during 1992 – 1997. Some of these bears ($n = 14$) were captured or radio-located on Baffin and Bylot islands during fall, 1993 – 1997. Because Taylor et al. (2005) state that the locations of collared bears were known at various times of the year and this information was used to assist in recaptures during 1994, 1995, and 1997, we constructed three binary radio covariates (rad94, rad95, rad97) to identify when individual females were likely wearing functional radiocollars and therefore may have been more vulnerable to capture. Not all collared bears were recaptured, however, so we did not fix $p = 1$ for these individuals, but instead used the covariates to allow for a potentially higher recapture probability if such an effect were supported by the data. We coded dependent offspring such that radio covariates matched their mothers.

Although earlier studies (e.g., Taylor et al. 2005, 2008) have assumed that all research-marked bears were reported in the harvest, current genetic analyses identified some marked bears that were harvested but not reported as marked, possibly due to marker loss (Government of Nunavut, unpublished data; Chapter 8). Hence, reporting probabilities (r) may be biased low, especially for bears that survived many years after initial marking during which their plastic ear tags could be lost and their lip tattoos could fade. Given restrictions on the harvest of females with dependent offspring and sex-restrictive quotas, we hypothesized that r would be lower for cubs, yearlings, and 2+ females; however, we did not fix $r = 0$ for cubs because at least 1 cub was harvested. We also hypothesized that improvements in the Greenlandic reporting system during the mid-2000s might yield increased reporting rates, so we considered models where r differed by early vs. later years (1993 – 2005 vs. 2006 – 2013). Since few cubs or yearlings were harvested, we incorporated this temporal structure as an additive effect for age 2+ individuals only.

We considered three candidate structures for site fidelity (F). Previous studies (e.g., Taylor et al. 2005) assumed perfect overlap of the areas sampled by live recapture and dead recovery sampling, with no permanent emigration from the study population. To reflect this assumption we evaluated a sub-model in which F was fixed to 1 for all sex and age-classes (F=1), without assessing a parameter penalty for QAIC_c. Using this approach, any permanent emigration that actually occurs for the BB subpopulation would be reflected in lower estimates of survival. Given that the subpopulation boundaries are only partially discrete, interchange is known to occur among subpopulations, and some harvest recoveries occurred outside the BB population boundaries (Figure 5.7), we also hypothesized that bears may permanently emigrate from the BB sampling area. We therefore evaluated a structure in which F was estimated as a constant across all age-sex classes (F.). Using this approach, survival estimates would not be biased by permanent emigration, but simulations indicated that under high levels of temporary emigration, rates of permanent emigration would be overestimated (i.e., temporary emigration would be misidentified as permanent emigration due to small sample sizes and short live-recapture sampling epochs relative to the life span of polar bears). We also hypothesized that adult males would exhibit lower site fidelity, so we considered a model structure in which independent males (age 2+) had a different fidelity rate than females and their dependent offspring (F_{family}).

Overdispersion, or extra-binomial variation, exists in mark-recapture data when the capture histories of individual animals are not independent (e.g., as is the case for family groups, in which the fate of the cubs depends on the adult female, or when emigration is non-random). Correcting for overdispersion is necessary to avoid underestimating the variance of parameters. To estimate overdispersion, we constructed our most highly parameterized model and used the median $\hat{c}$ approach as implemented in Program MARK (Cooch and White 2015). This method employs simulations to generate an estimate of c, the over-dispersion parameter. Results suggested that the BB data were modestly over-dispersed (i.e., $\hat{c} = 1.1$), as would be expected given the dependency between females and their cubs (Taylor et al. 2005), so we inflated $\hat{c}$ and based model selection and inference on QAIC$_c$ (Burnham 1998).

Given 10 sub-model structures for S, 12 for p, 1 for r, and 3 for F, there would be 360 potential model structures if all possible combinations of the sub-model structures were considered. We used a modified version of the plausible combinations approach outlined in Bromaghin et al. (2013) to identify supported sub-model structures. This process entailed holding constant the most generalized structure (excluding individual covariates) for three of the four sub-models while evaluating structures for the fourth sub-model. We considered sub-model structures with ΔQAIC$_c$ < 4 as representing plausible structures and constructed all possible combinations from these sub-model structures. We note that S was poorly estimated for coy (i.e., at implausibly high rates near 1, but not inestimable), a finding which we attributed to the scarcity of data for coy, particularly during the 2010s sampling period in which many coy were not marked. Hence, we estimated a pooled S rate for coy and yrlg in all subsequent models.

Because coy were approximately 9 months old at the time of marking, it is biologically plausible that their survival rates were similar to those of yearlings.

We computed model-averaged estimates (Burnham and Anderson 2002) of parameters (S, r and F) using a threshold of $\Delta\text{QAIC}_c < 4$. Because our estimates of survival reflected harvest mortality, we derived unharvested survival (S*; also referred to as "natural" survival) using the equation as $S^* = S + r \times (1 - S)$ (e.g, Taylor et al. 2005, 2008, Peacock et al. 2013) and estimated the variance of S* using the delta method (following Taylor et al. 2008). This derivation of unharvested survival is based on several assumptions. First, it assumes harvest of all marked bears is reported; under-reporting of the harvest, which has been documented (Government of Nunavut, unpublished data; Chapter 8), leads to negative bias in estimates of S*. This derivation also assumes that harvest mortality is completely additive. In other words, that no harvested bears would otherwise die during a given interval, and that the higher population density for an unharvested subpopulation would not lead to lower vital rates for all bears in future years. A violation of the assumption of additive mortality would result in positive bias in estimates of S*. A more appropriate equation for unharvested survival would be: $S^* = S/[1 - r \times (1 - S)]$, which assumes that harvested bears are subject to the same natural mortality rate as other bears. In this equation, the quantity in brackets represents the probability of surviving the hunting season and S represents the product of natural and hunting mortality (Anderson and Burnham 1976). We used the Taylor et al. (2005) derivation for unharvested survival to maintain consistency with earlier studies, noting that the resulting potential for bias is small given high unharvested survival rates and relatively low harvest mortality for polar bears.

For highly supported models, we obtained annual estimates of abundance for groups of individuals that share common estimates of p (e.g., certain age and sex-classes), using a generalized Horvitz-Thompson estimator for the yrl and age 2+ classes, in which $\widehat{N}_{i,t} = \frac{n_{i,t}}{\hat{p}_{i,t}}$, where $n_{i,t}$ is the number of bears captured in group *i* during year *t*, and $\hat{p}_{i,t}$ is the recapture probability for group *i* during year *t*. However, estimates of n and p did not accurately represent coy because we did not sample all coy during the 2011 – 2013 sampling period. Thus, we estimated coy abundance as the product of age 2+ females with coy litters (estimated via a Horvitz-Thompson estimator) and mean observed coy litter size. We summed estimates of abundance across groups to obtain total estimates of abundance (derived for each model) by year. Similar to earlier studies (e.g., Taylor et al. 2005, Peacock et al. 2013), we used the delta method (Seber 1982, Powell 2007) to estimate variances for annual abundance estimates [R (R Core Team 2015) package emdbook (Bolker 2016)]. Variance estimates incorporated parameter variances and covariances (as computed in Program MARK) as well as variances of mean coy litter sizes. We used model weights to model-average estimates of total abundance by year and their associated variances. We also calculated mean estimates of total abundance by sampling epoch and estimated variance using the delta method. Given the 13-year interval without live captures preceding 2011, estimates of abundance for 2011 were based on values of *p* estimated for the relatively small number of bears that were marked during the 1990s and survived until 2011. These estimates of *p* were characterized by high uncertainty and potential small-sample bias. Their use in the denominator of the Horvitz-Thompson estimator, in conjunction with the large sample of first-time captures in the numerator of the estimator, had the potential to produce spurious results. Therefore, we excluded the less-reliable estimate of abundance from 2011 when calculating mean total abundance for the 2010s sampling epoch.

Geographic subsetting to evaluate the effects of different sampling methods in the 1990s and 2010s

Because sampling during the 1990s was spatially restricted to a portion of the BB subpopulation's fall range and did not include bears located farther inland, particularly within deep fjords, or on the sea ice. In contrast, from 2011-2013 onshore sampling was more comprehensive and systematic. To explore the potential impact of differences in sampling on estimates of subpopulation size, we delineated the extent of the sampling area in the 1990s based on capture locations (see Chapter 3). We then created a subset of the 2010s live-capture data that only included captures that occurred within the more restricted sampling area of the 1990s, recompiled the individual capture histories, and repeated our mark-recapture analyses using the same procedures as outlined above. We expected that comparison of abundance estimates for the 2010s using full dataset (i.e., for the complete sampling area) vs. the restricted dataset (i.e., for the restricted sampling area), would help inform the potential bias in estimates of abundance from the 1990s based on an incomplete sampling frame. Conceptually, this assumed that $\widehat{N}_{2010s}^{subset\ data}/\widehat{N}_{2010s}^{full\ data} \approx \widehat{N}_{1990s}^{subset\ data}/\widehat{N}_{1990s}^{full\ data}$; where $\widehat{N}$ represents estimates of abundance, and $\widehat{N}_{1990s}^{full\ data}$represents the (unknown) estimate of abundance that would have been obtained in the 1990s if the complete sampling area had been covered. This assumption seemed plausible given that satellite telemetry data indicated that onshore habitat use of polar bears did not vary between the 1990s and 2010s. Although this investigation provides information on the effects of difference in onshore sampling between the two epochs, it did not provide any information on potential bias in the 1990s abundance estimate due to bears using offshore sea ice in the 1990s.

5.3. Results

The BB mark-recapture dataset consisted of 2,324 total captures of 1,992 individuals (i.e., there were 332 live recaptures), and 234 dead recoveries during the 1993 – 2013 study period (Table 5.3). Data were relatively sparse for live captures of age 2+ females during the 1990s, and dead recoveries of all bears during the early- to mid-2000s. During September 2012 and 2013, we sighted 30 polar bears (including 21 independent bears) in Greenland (*cf.* 1,043 total captures during 2012 and 2013; Table 5.3), suggesting a relatively low number of bears resided in Greenland during the late summer in those years (see Chapter 3).

Females comprised a greater proportion of live captures of age 2+ bears in the 2010s compared to the 1990s (mean annual proportion female during the 1990s: 0.42; 2010s: 0.54; Table 5.3). We hypothesize that under-representation of age 2+ females was a result of the greater use of inland habitats by denning females and lack of sampling in those habitats during the 1990s. Age 2+ males comprised nearly 70% of the reported harvest of marked bears over the 21-year study period (162 of 234; Table 5.3), with adult females exhibiting sparse recovery data, especially during the interim epoch (1998-2010) with no live encounter data.

The plausible combinations approach indicated that the following sub-model structures were supported by the data: one S structure (3 temporal epochs with an interactive effect with sex for the 2+ age class; Table 5.4); two p structures [including (1) fully time-varying p and (2) spring sea-ice transition date to explain variability in p; Table 5.5]; all three F structures (Table 5.6); and one r structure as candidate structures from which to construct the final set of models. Although within 4 ΔQAIC_c of the most highly supported p structures, we excluded p sub-models that incorporated the inland proximity to coastline covariate, as this was an uninformative parameter that was not supported by lower QAIC_c relative to hierarchically simpler models (Burnham and Anderson 1992, Arnold 2010, Peacock et al. 2012).

We constructed the final candidate model set using all combinations of the well-supported sub-model structures as identified above. The most-supported model included a fully time-varying p structure and estimated F as constant across all sex and age classes. Three additional model structures were within Δ4 QAIC$_c$ of the most-supported model, including a model with F estimated separately for independent males, and a model with F fixed to 1 (Table 5.7).

The time-constant, model-averaged estimate of survival for dependent bears was (S = 0.87, SE = 0.06; Table 5.8). Estimates of S for age 2+ females (1993 – 1997: 0.84, SE = 0.04; 1998 – 2010: 0.95, SE = 0.02; 2011 – 2013: 0.90, SE = 0.05) and males (1993 – 1997: 0.89, SE = 0.02; 1998 – 2010: 0.87, SE = 0.02; 2011 – 2013: 0.78, SE = 0.06) varied among epochs, although statistical uncertainty and potential bias made it difficult to evaluate whether this variation was meaningful (see Discussion). Reporting rates were nearly 2-fold higher for age 2+ males than 2+ females, reflecting male-biased harvest, but there was not a strong difference in estimates of r before and after 1995. Estimated natural survival for age 2+ males (0.83, SE = 0.06) was less than age 2+ females (0.91, 0.05) during 2011 – 2013 (coy: 0.88, SE = 0.06; yrl: 0.89, SE = 0.06). Bears exhibited strong fidelity to the study area (F = 0.96 for females and dependent young, and F = 0.97 for age 2+ males). This suggests that approximately 3 – 4% of the study population permanently emigrated from the sampling area each year, although we did not utilize Barker models that additionally measure the probability that some of these bears might have returned.

Mean estimates of total abundance for the BB subpopulation were 2,173 (95% CI = 1,252 – 3,093) for the 1994 – 1997 sampling epoch and 2,826 (95% CI = 2,059 – 3,593) for the years 2012 – 2013 (Table 5.9), although these estimates correspond to different sampling frames in the

1990s and 2010s. The mean estimate of abundance for the years 2012-2013 was approximately 10% lower for the geographic subset dataset (2,553 ± 433) compared to the full dataset (2,826 ± 391). In addition, the annual point estimates for 2012 – 2013 derived from the geographic subset were lower than estimates based on the full data (Table 5.12). These findings suggest that the restricted sampling frame introduced negative bias into estimates of abundance for the 1990s. The difference in the spatial distribution of captures between the 1990s and 2010s was consistent with our conclusion that the sampling frames differed substantially between epochs, particularly with respect to the inland distribution of bears in central and northern Baffin Island (Figures 5.1, 5.8, and 5.9). Very few bears were recorded beyond the mouths of fjords in these regions during the 1990s, whereas observations were numerous there during the 2011 – 2013 epoch. This finding was reinforced by telemetry data during the 2000s that indicated no large-scale shift in onshore distribution (relative to the coastline) between epochs (see also Chapters 3 and 4).

The geographic subset included 1,679 total individuals, as >300 bears from the 2000s were censored from this analysis based on their locations outside the estimated sampling frame of the 1990s. Model selection results were generally similar to the comprehensive data set, although the coastline covariate (for modeling p) was more highly supported in some structures (Table 5.10). Parameter estimates also were consistent with the comprehensive data set (Table 5.11).

5.4. Discussion

We used physical mark-recapture data collected 1993-1995 and 1997, genetic mark-recapture data collected 2011-2013, and dead recovery data from the 21-year period 1993-2013 to estimate demographic parameters for the BB subpopulation. Our mean estimate of total abundance for the years 2012-2013 was 2,826 (95% CI = 2,059 – 3,593). Our estimate of mean abundance for the period 1994-1997 (2,173; 95% CI = 1,252 – 3,093) was consistent with the previous estimate from Taylor et al. (2005) (2,074; SE = 266), despite minor differences in the data and analytical methods. Although our 2012-2013 estimate of abundance is ~30% higher than our 1990s estimate, differences in sampling protocols and changes in environmental conditions between epochs make interpretation about true changes in population size difficult. Notably, the distribution of capture locations was different between epochs, with a higher proportion of captures in inland areas and deep within fjords during the 2010s (Figures 5.3, 5.8 and 5.9, see also Chapter 3 Figure 3.1). We used satellite telemetry data to compare on-land distribution and summer habitat use between the two epochs. These analyses provided no evidence for changes in on-land distribution in the summer, suggesting that differences in the spatial distribution of captures resulted from a more restricted sampling frame in the 1990s. In the 1990s only one helicopter was used for sampling in western Baffin Bay in fall whereas the 2000-sampling involved the use of two helicopters operating at the same time. In addition, satellite telemetry data suggested that an unknown but potentially large number of bears were not present in the 1990s study area due to the presence of summer pack ice offshore from Baffin Island, whereas offshore ice was largely unavailable in the 2010s.

We investigated potential bias resulting from differences in sampling protocols between epochs. Although flight paths for the helicopters used to capture bears in the 1990s were unavailable, we estimated the spatial extent of the sampling area using capture locations, and then created a geographical subset of the 2010s data based on this restricted sampling frame. These results indicated that a lack of inland sampling in the 2010s would have resulted in approximately 10% negative bias in the mean estimate of total abundance for 2012-2013, which suggests that similar bias likely existed in the mean estimate of total abundance for the 1990s, due to restricted geographic sampling. The source of this bias is individual heterogeneity in p due to polar bear movements with respect to the sampling area (i.e., temporary emigration; Schaub et al. 2004); which in the extreme can result in some bears effectively having $p = 0$, and therefore being completely excluded (i.e., “missed”) from the study. The presence of offshore sea ice during the 1990s was another potential source of negative bias, and an issue that we were unable to resolve. These environmental conditions are problematic because bears using the sea ice were either temporarily or permanently absent from the mark-recapture sampling area during the 1990s. Although sample sizes of independent collared females were small, telemetry data suggested that >~75% of collared bears were outside of the sampled area or on remnant sea ice during the fall sampling period in some years during 1990s. As such, an unknown but potentially large proportion of the population was unavailable for capture in some years. By contrast, in the 2010s sea ice was not present in Baffin Bay in late summer and bears were more concentrated in the onshore sampling areas (i.e., on Baffin or Bylot islands, or West Greenland); 68-85% of collared bears were inside the sampling areas in all years (2011-2013). If the probability of being a temporary emigrant in the 1990s was sufficiently high, relative to the short duration of the study and small sample sizes; or if there was Markovian dependence in the probability of being a temporary emigrant (e.g., if a bear that was on remnant sea ice in year t tended to return to the ice in year $t + 1$), then abundance estimates from the 1990s may be subject to additional bias. The sign of this bias was likely negative (i.e., it is possible that a meaningful proportion of the subpopulation was effectively excluded from abundance estimates), although the component of bias due to potential Markovian dependence could be either positive or negative depending on the directionality of Markovian dependence (Schaub et al. 2004). Our assessment of temporary emigration in the 1990s is consistent with traditional ecological knowledge in the Baffin Bay region, which suggests that some bears spend the entire year on sea ice and do not come ashore (S. Atkinson, pers. obs.). The effects of temporary emigration on 1990s abundance estimates are difficult to quantify because of the short live-recapture sampling window (1993 – 1995; 1997) and low recapture probabilities.

Given the multiple potential sources of negative bias in the 1990s abundance estimate, and statistical uncertainty in estimated parameters, we cannot conclude that the size of the BB subpopulation increased between the 1990s and 2010s. The 2010s estimate of abundance constitutes the best-available information and is suitable for informing management, but we cannot reliably determine the trend in subpopulation size between the 1990s and 2010s.

It should, however, be mentioned that during TEK-studies in both Nunavut (Dowsley 2005, Dowsley and Taylor 2006) and West Greenland (Rosing-Asvid and Born 1990, Born et al. 2011) several interviewees noted an increased occurrence of BB-polar bears in coastal areas since sometime in the 1990s. Some interviewees were of the opinion that this reflected an increase in subpopulation (BB) size whereas others thought that it reflected a change in to the bears´ behavior, and the fact that they occur closer to land as a reaction to the reduction in the sea-ice cover (cf. Born et al. 2011:206-207 for a discussion of this). Born et al. (2011) concluded that it was not possible from their interview survey to determine the extent to which an increased occurrence of polar bears in the hunting areas represents an increase in the population or a change in distribution (or for that sake a combination of these factors). Chapter 4 documents changes in the on-ice behavior of BB bears, with bears located closer to the coastline (and closer to shallow depths) in all seasons.

The sampling issues outlined above also have the potential to affect estimates of survival probability (Peñaloza et al. 2014), although these issues are mitigated to some extent by the inclusion of dead recovery data (Peacock et al. 2012). Our estimates of survival for age 2+ females 1993 – 1997 are notably lower than estimates of similar age classes obtained by Taylor et al. (2005) and Peacock et al. (2012), but our estimates of coy and yearling survival are higher. Factors that may contribute to these differences include: different treatment of data (e.g., exclusion of spring captures, recognition of coarser age classes in the present study relative to Taylor et al. 2005 and Peacock et al. 2012); our exclusion of data collected during the 1970s and 1980s, which were included in Taylor et al. (2005) as initial captures but not fully modeled as individual capture histories; and the inclusion of additional information (e.g., harvest recoveries during the 2000s and live captures during the 2010s), given that some parameters were estimated using information that was shared across sampling epochs.

The mean estimate of total annual survival of age 2+ males was particularly low (0.78, SE = 0.06) during 2011 – 2013, compared to values from earlier periods in this study and values reported for other polar bear subpopulations (PBSG 2010), with the exception of the Southern Hudson Bay subpopulation for which low estimates of adult male survival were reported for the final years of the study (Obbard et al. 2007). Although r for 2+ males was higher than 2+ females due to the sex-selective harvest, estimates of unharvested male survival 2011-2013 (S^* = 0.83) were also significantly lower than estimates of S^* for females during this period (S^* = 0.91). Low survival of adult males in the 2010s may be a biological signal that reflects a disproportionate impact of environmental change on males, either through lower true survival or increased dispersal of young males to adjacent subpopulations due to density-dependent effects. However, interpretation of point estimates and potential trends in S is difficult due to the short time-frame (i.e., only 3 years) of live recaptures in the 2010s and confounding of parameters and potential bias in estimates of survival during terminal years (e.g., Peñaloza et al. 2014). We suggest that, when viewed together with information on habitat loss (Chapter 4) and nutritional condition (Chapter 7) for BB polar bears, the low estimates of S for 2+ males 2011-2013 may signal negative density-dependent population effects. However, we emphasize that additional years of live-recapture and dead-recovery data would be necessary to determine the degree to which low estimates of survival were influenced by temporary emigration and other factors.

Adjusting total survival with r to derive unharvested survival yielded estimates of female survival in the 2010s that appear too low to support stable or positive population growth in the absence of harvest. Regehr et al. (2015) suggest that a minimum unharvested adult female survival rate of ~0.93, referenced to a population density at maximum net productivity level, is necessary for long-term persistence. We note, however, that our estimates of dependent young survival were high (*cf*. Taylor et al. 2005), although these estimates applied to the entire 1993-2013 study period. Based on the life history of polar bears, survival of dependent young would be expected to decline (e.g., in response to negative environmental conditions) before the survival of adult females declined (e.g., Eberhardt 2002). In addition, due to the lack of precise numeric age information available from non-invasive genetic sampling in the 2010s, we estimated survival for a single age class of polar bears age ≥ 2 years. Under this approach subadult and senescent bears, which likely have lower survival rates (e.g., Regehr et al. 2007), were included with prime age adults. This likely had the effect of reducing the overall estimate of female survival. We also note that the lowest estimates of total and unharvested survival for age 2+ females occurred during 1994-1997, a period during which sea-ice habitats were more available compared to the 2010s. These considerations, in conjunction with the high and variable levels of temporary emigration from the sampling area in the 1990s, and our approach of aligning temporal epochs in the survival sub-model with changes in sampling design, could have led to negative bias in estimates of female survival.

We recommend further caution in interpretation of survival estimates. First, the three temporal epochs for which we estimated survival were based on – and are confounded with – sampling methodologies (i.e., whether estimates were derived from both live recapture and dead recovery data, or dead recovery data alone). In addition, the 2011 – 2013 epoch represents the minimum length of a time series from which it is possible to estimate survival using Cormack-Jolly-Seber models, such that a single anomalous year (from either a sampling or biological perspective) has a greater impact on the pooled estimate. Additionally, negative terminal bias in survival estimates is a well-known challenge with mark-recapture studies, especially under scenarios with pronounced temporary emigration (Peñaloza et al. 2014).

To provide additional insight into our findings, we conducted complementary modeling using Markov chain Monte Carlo (MCMC) methods in Program MARK. We re-fit several well-supported models and incorporated annual random effects for S, r, and in some cases p. Annual random effects can offer advantages over fixed effect approaches by representing temporal patterns in the data via a long-term mean and annual shrinkage estimates, such that annual estimates of a parameter only deviate from the mean to the extent that any difference is supported by the data (Link and Barker 2004, White et al. 2009). MCMC methods also enable delineation of sampling from process variation. However, the ability to obtain useful annual estimates from the Baffin Bay data was somewhat limited by small sample sizes and the unusual survey design (i.e., the short time series of live-encounters at the beginning and end of the study, separated by a longer period of dead-recovery only data in the middle of the study).

MCMC analyses suggested that the low estimates of mean survival for 2+ females during 1993 – 1997 and for 2+ males during 2011 – 2013 were due in part to relatively higher levels of human-caused mortality (i.e., as represented by estimates of r) for females and males in 1996 (also the year in which no live capture sampling occurred) and 2011, respectively. Importantly, these analyses also indicated that the low estimate of survival for 2+ males during 2011 – 2013 was strongly influenced by the cohort of newly-marked bears in 2011. The long period of time without live recapture sampling (1998 – 2010) meant that bears initially marked in the 1990s (i.e., older bears) were pooled with new captures during 2011 (including younger bears) for estimation of parameters. MCMC analyses suggest that 2011 was a particularly poor year for survival of age 2+ males, but this impact was only evident among newly marked bears and not among surviving bears first marked in the 1990s. This finding suggests possible individual or finer-scale age-based variation in survival, but sample sizes and study design (i.e., the 13-year interval with no live captures) were insufficient to fit models with individual random effects using either maximum likelihood or MCMC methods. In sum, MCMC analyses supported our interpretation that the BB data do not provide strong evidence for temporal changes in survival, with the exception of the two years noted above. We recommend that future work in Baffin Bay and elsewhere further explore models with annual random effects. In addition, incorporation of dead recovery data after 2013 will assist in estimating survival during the 2011-2013 sampling period (Peacock et al. 2012).

Based on supplementary analyses, conducted as part of this investigation, that suggest a relatively strong ability to distinguish subadults from adults using field assessments and genetic information (i.e., sex) obtained from biopsy darting (Appendix B), future modeling of data from genetic mark-recapture studies could consider a more detailed age structure. Recent advances in analytical methods (e.g., mark-recapture models with state uncertainty; Pradel 2009) could be used to model the relatively low occurrence of errors in estimation of field ages. Given the broader base of evidence that the BB subpopulation is responding to losses of sea-ice habitat (e.g., Rode et al. 2011, Peacock et al. 2012; Chapters 4, 6, 7), considering a more detailed age structure could help to assess whether survival rates for adults and subadults exhibited different temporal trends (e.g., per the expectation that subadult survival rates are among the first demographic parameters to respond to environmental changes; Regehr et al. 2007, Stirling and Derocher 2012).

To further assess bias and precision of estimated model parameters, we completed simulations in which Barker models, which are capable of explicitly modeling temporary emigration, were fitted to datasets that closely resembled the BB data, but included known levels and types of temporary emigration. Under moderate to high random temporary emigration (F=F'= 0.3 or 0.5), survival estimates were unbiased when estimated using Barker models, but moderately negatively biased when based on Burnham models (especially for females, which had lower recovery rates). CJS and Seber models both produced highly biased estimates of S when temporary emigration occurred. Markovian emigration may yield negative bias in S, although sparse telemetry data did not permit assessment of Markovian patterns in temporary emigration with the actual Baffin Bay data. Although we were unable to simulate data on abundance, Barker models provided unbiased estimates of p, whereas Burnham models provided unbiased estimates of the product of $p \times F$. These findings suggest that Barker models would provide reasonable estimates of the number of bears located within the sampling area on any given sampling occasion, whereas Burnham models would provide better estimates of the "super-population" (i.e., the larger group of bears with a non-negligible probability of using the sampling area, even if some of these bears were outside of the sampling area [i.e., were temporary emigrants] on any given sampling occasion). Given that the super-population corresponds more closely to the study population of interest in BB from a biological and management perspective, this supports our decision to use the Burnham model to estimate abundance.

Although there are uncertainties in the BB subpopulation related to the demographic analyses, additional sources of information, including sea-ice conditions, movement ecology, and reproductive metrics, are useful for informing current subpopulation status. These auxiliary data suggest a lengthening of the ice-free season (Chapter 4), a significant increase in the amount of time bears spend on land during the ice-free season (Chapter 4), a northward shift in their ranges (Chapter 2), decreased reproductive output (Chapter 6), and declining body condition (Rode et al. 2011; Chapter 7). These signs point to a subpopulation that could be exhibiting density-dependent effects associated with declining carrying capacity. The relatively low estimates of unharvested survival for the 2010s, especially for adult males, are consistent with this explanation, although as stated previously we cannot rule out the presence of negative bias in survival estimates.

The Baffin Bay study highlights potential challenges in interpreting long-term trends in abundance and survival. Although Taylor et al. (2005) and York et al. (2016) assert that the BB subpopulation was uniformly and comprehensively sampled during the 1990s, we documented evidence of changes in the sampling frames between epochs (i.e., incomplete spatial sampling during the 1990s, relative to the 2000s). These changes precluded an assessment of trends in abundance, and the 13-year interval between live capture sessions limited our ability to assess temporal trends in survival, and likely resulted in increased individual heterogeneity in survival (which we could not explicitly model) as newly-marked bears in the 2010s were pooled with older individuals first marked in the 1990s. We attempted to reconcile and understand the impacts of these issues through supplemental analyses, and future work could explore the usefulness of new analytical methods (e.g., multistate models with unobservable states; spatially-explicit models [Royle et al. 2013]) to mitigate potential bias. However, recent analyses of mark-recapture data for polar bears have identified the limitations of model-based methods to account for inconsistent sampling or violated modeling assumptions (e.g., Bromaghin et al. 2015). Therefore, we underscore the critical importance of complete and consistent sampling of the study area for robust and unbiased inference regarding population status. We also recommend evaluating the current inventory schedule and using *a priori* study design analyses to evaluate whether modifications (e.g., extending the live capture sampling periods, shortening the interval between successive capture periods, incorporation of more intensive “robust design” sampling [Converse et al. 2009]) may improve the ability to detect changes in abundance and associated vital rates. Finally we recommend considering other survey methods (i.e., aerial surveys; e.g., Aars et al. 2009) for assessing polar bear subpopulations.

5.5. Literature Cited

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea-ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Anderson, D. R. and K. P. Burnham. 1976. Population ecology of the mallard: VI. The effect of exploitation on survival. Resource Publication No. 128, U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Arnold, T. W. 2010 Uninformative parameters and model selection using Akaike's Information Criterion. Journal of Wildlife Management 74:1175–1178.

Atwood, T., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. E. Bromaghin. 2015. Evaluating and ranking threats to the long-term persistence of polar bears. U.S. Geological Survey Open-File Report 201-1254. U.S. Geological Survey, Reston, Virginia, USA.

Barker, R. J. 1997. Joint modeling of live-recapture, tag-resight, and tag-recovery data. Biometrics 53:666–677.

Barker, R. J. 1999. Joint analysis of mark-recapture, resighting and ring-recovery data with age-dependence and marking effect. Bird Study 46:S82–S91.

Barker, R. J., and G. C. White. 2001. Joint analysis of live and dead encounters of marked animals. Pages 361–367 *in* R. Field, R. J. Warren, H. Okarma, and P. R. Sievert, editors. Wildlife, Land, and People: Priorities for the 21st Century. Proceedings of the Second International Wildlife Management Congress, The Wildlife Society, Bethesda, Maryland, USA.

Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. Wildlife Society Bulletin 31:197–207.

Bolker, B. 2016. Ecological Models and Data in R. R package version 1.3.9.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Boulanger, J., S. Himmer, and C. Swan. 2004. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British

Columbia. Canadian Journal of Zoology 82:1267–1277.

Bromaghin, J. F., T. L. McDonald, and S. C. Amstrup. 2013. Plausible combinations: an improved method to evaluate the covariate structure of Cormack-Jolly-Seber mark-recapture models. Open Journal of Ecology 3:11–22.

Burnham, K. P. 1993. A theory for combined analysis of ring recovery and recapture data. Pages 199–213 *in* J. D. Lebreton and P. M. North, editors. Marked individuals in the study of bird populations. Birkhauser-Verlag, Basel, Switzerland.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.

Comiso, J. C., C. L. Parkinson, R. Gertsen, and L. Stock. 2008. Accelerated decline in the Arctic sea ice cover. Geophysical Research Letters 35:L01703.

Converse, S. J., W. L. Kendall, P. F. Doherty, and P. G. Ryan. 2009. Mutistate models for estimation of survival and reproduction in the grey-headed albatross (Thalassarche chrysostoma). The Auk 126:77-88.

Cooch, E.G., White, G.W., 2015. Program MARK: A Gentle Introduction. 14th ed. Colorado State University, Fort Collins, CO, USA.

Devineau, O., R. Choquet, and J. D. Lebreton. 2006. Planning capture-recapture studies: straightforward precision, bias, and power calculations. Wildlife Society Bulletin 34:1028-1035.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1: 43 pp.

Dowsley, M. and M.K. Taylor. 2006. Community consultations with Qikiqtarjuaq, Clyde River and Pond Inlet on management concerns for the Baffin Bay (BB) polar bear population: A summary of Inuit knowledge and community consultations. Nunavut Wildlife Research Group Final Report No. 2: 83 pp.

Eberhardt, L. L. 2002. A paradigm for population analysis of long-lived vertebrates. Ecology 83:2841-2854.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use of polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. Ecology 81:761–772.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. Journal of Wildlife Management 73:3–17.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lebreton, J.-D., J. D. Nichols, R. J. Barker, R. Pradel, and J. A. Spendlow. 2009. Modeling individual animal histories with multistate capture-recapture models.*in* H. Caswell, editor. Advances in Ecological Research Volume 41. Academic Press, New York.

Link, W. A., and R. J. Barker. 2004. Hierarchical mark-recapture models: a framework for inference about demographic processes. Animal Biodiversity and Conservation 27:441–449.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications doi:10.1890/15-1256.

Obbard, M. E., T. L. McDonald, E. J. Howe, E. V. Regehr, and E. S. Richardson. 2007. Polar bear population status in Southern Hudson Bay, Canada. Administrative report, U.S. Geological Survey, Reston, Virginia, USA.

Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. Molecular Ecology 12:1375–1387.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Palsbøll, P. J., J. Allen, M. Bérube, P. J. Clapham, T. P. Feddersen, P. S. Hammond, R. R. Hudson, H. Jørgensen, S. Katona, A. H. Larsen, F. Larsen, J. Lien, D. K. Mattila, J. Sigurjónsson, R. Sears, T. Smith, R. Sponer, P. Stevick, and N. Øien. 1997. Genetic tagging of humpback whales. Nature 388:767–769.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014. http://pbsg.npolar.no/en/status/status-table.html, Accessed 4 May 2016.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Peacock, E., S. A. Sonsthagen, M. E. Obbard, A. Boltunov, E. V. Regehr, N. Ovsyanikov, J. Aars, S. Atkinson, G. K. Sage, A. G. Hope, E. Zeyl, L. Bachmann, D. Ehrich, K. T. Scribner, S. C. Amstrup, S. Belikov, E. W. Born, A. E., Derocher, I. Stirling, M. K. Taylor, Ø. Wiig, D. Paetkau, and S. L. Talbot. 2015. Implications of the circumpolar genetic structure of polar bears for their conservation in a rapidly warming Arctic. PLoS ONE 10:e112021. doi:10.1371/journal.pone.0112021

Peñaloza, C. L., W. L. Kendall, and C. A. Langtimm. 2014. Reducing bias in survival under nonrandom temporary emigration. Ecological Applications 24:1155–1166.

Powell, L. 2007. Approximating variance of demographic parameters using the delta method: A reference for avian biologists. Condor 109:949–954.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Rosing-Asvid, A. and E.W. Born. 1990. Fangst af isbjørn (*Ursus maritimus*) i Avanersuaq og Upernavik kommuner: En interviewundersøgelse (The catch of polar bears, *Ursus maritimus*, in the municipalities of Avarnersuaq and Upernavik: An interview survey. Grønlands Hjemmestyres Miljø –og Naturforvaltning Teknisk Rapport Nr. 23: 63 pp. (In Danish with an English and Greenlandic Abstract).

Royle, J. A., R. B. Chandler, R. Sollmann, and B. Gardner. 2013. Spatial capture-recapture. Academic Press, Waltham, Massachusetts, USA.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Macmillan, New York, New York, USA.

Schaub, M., O. Gimenez, B. R. Schmidt, and R. Pradel. 2004. Estimating survival and temporary emigration in the multistate capture-recapture framework. Ecology **85**:2107-2113.

Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., G. W. Thiemann, and E. Richardson. 2008. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568-574.

Stroeve, J. C., M. C. Serreze, M. M. Holland, J. E. Kay, J. Malanik, and A. P. Barrett. 2012. The Arctic's rapidly shrinking sea ice cover: A research synthesis. Climatic Change 110:1005–1027.

Taylor, M. K., and J. Lee. 1995. Distribution and abundance of Canadian polar bear populations: a management perspective. Arctic 48:147–154.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M'Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667–1673.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. Marine Mammal Science 25:778–796.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46:S120–S139.

White, G. C., K. P. Burnham, and R. J. Barker. 2009. Evaluation of a Bayesian MCMC random effects inference methodology for capture-mark-recapture data. Pages 1119–1127 *in* D. L. Thomson, E. G. Cooch, and M. J. Conroy, editors. Modeling Demographic Processes in Marked Populations, Springer, New York, USA.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution 6:2897–2924.

Table 5.1. Survival sub-model structures evaluated in mark-recapture analysis of the Baffin Bay polar bear subpopulation data.

S sub-model	Age	Sex	Temporal	Environmental
1	2 class	Age 2+ only	Constant	None
2	2 class	Age 2+ only	3 epoch + sex	None
3	2 class	Age 2+ only	3 epoch × sex	None
4	2 class	Age 2+ only	Constant	Ice transition
5	2 class	Age 2+ only	Constant	Ice area
6	3 class	Age 2+ only	Constant	None
7	3 class	Age 2+ only	3 epoch + sex	None
8	3 class	Age 2+ only	3 epoch × sex	None
9	3 class	Age 2+ only	Constant	Ice transition
10	3 class	Age 2+ only	Constant	Ice area

Table 5.2. Recapture probability sub-model structures evaluated in mark-recapture analysis of the Baffin Bay polar bear subpopulation. All models included a Radio covariate for bears that were outfitted with a satellite collar that may have been used to locate individuals for recapture. Bears that were not genotyped were unavailable to be recaptured during the 2011 – 2013 sampling window, so p was fixed to 0 for non-genotyped bears.

p sub-model	Family	Temporal	Geographic	Ice
1	Yes	2 epoch + family	None	None
2	Yes	2 epoch + family	Coastline, 2 epoch	None
3	Yes	2 epoch + family	Coastline, 2010s	None
4	Yes	2 epoch + family	None	Spring
5	Yes	2 epoch + family	Coastline, 2 epoch	Spring
6	Yes	2 epoch + family	Coastline, 2010s	Spring
7	Yes	2 epoch × family	None	None
8	Yes	2 epoch × family	Coastline, 2 epoch	None
9	Yes	2 epoch × family	Coastline, 2010s	None
10	Yes	Annual + family	None	None
11	Yes	Annual + family	Coastline, 2 epoch	None
12	Yes	Annual + family	Coastline, 2010s	None

Table 5.3. Summary table of live captures and dead recoveries during the mark-recapture study of the Baffin Bay polar bear subpopulation in Nunavut, Canada, and Greenland, 1993 – 2013. Shaded cells indicate that data are not possible due to an absence of marking or recapture.

	Initial captures						**Live recaptures**				**Dead recoveries**					
	Females			*Males*			*Females*		*Males*		*Females*			*Males*		
Year	**Coy**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Yrl**	**2+**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**
1993	14	8	53	12	8	61					0	0	1	0	0	0
1994	26	13	65	16	9	77	0	5	0	14	0	0	3	0	0	7
1995	15	11	62	19	11	85	4	11	4	23	0	2	6	1	0	8
1996												1	8		0	7
1997	22	10	60	19	13	113		20		31	0	0	6	0	1	9
1998												0	3		0	11
1999													3			9
2000													0			8
2001													2			8
2002													0			11
2003													0			7
2004													1			7
2005													2			3
2006													3			6
2007													1			2
2008													2			4
2009													2			0
2010													0			1
2011	2	23	163	1	20	148		5		5	0	0	4	0	0	20
2012	40	30	221	35	30	192	3	41	0	54	0	0	8	0	2	14
2013	28	15	121	16	15	90	4	48	5	55	0	1	8	1	0	20
Totals	**147**	**110**	**745**	**118**	**106**	**766**	**11**	**130**	**9**	**182**	**0**	**4**	**63**	**2**	**3**	**162**

Table 5.4. Survival (*S*) sub-model selection results from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation. Epoch = periods defined by sampling method (1993 – 1997, 1998 – 2010, and 2011 – 2013). Preliminary analyses suggested that QAIC*c* scores of structures including sea-ice metrics were critically dependent on 1996, the year in which there was no live recapture sampling, which also happened to coincide with heavy sea ice. Structures with sea-ice covariates thus were eliminated from further consideration.

***S* sub-model structure**	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
coy yrl, 2+(sex × epoch)	22	0.00	0.978	3878.0
coy yrl, 2+(sex + epoch)	20	8.36	0.015	3890.4
coy yrl, 2+(sex)	18	9.83	0.007	3896.0

Table 5.5. Recapture probability (p) sub-model selection results from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Family = females / dependent bears and independent males (2 age / sex classes); ice = spring transition date; epoch = sampling period (1993 – 1995, 1997; 2011 – 2013); t = full time variation; and inland = proximity of individual's first capture location to smoothed coastline (2 km threshold; binary). All p structures incorporated the radio collar covariate representing bears that were outfitted with collars that may have been used to locate them.

***p* sub-model structure**	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
family + t	22	0	0.418	3878.0
family + t + coastline (2010s)	23	1.31	0.217	3877.3
family + t + coastline (epoch)	24	1.32	0.216	3875.2
family + epoch + ice	19	3.50	0.073	3887.6
family + epoch + ice + coastline (epoch)	21	4.78	0.038	3884.8
family + epoch + ice + coastline (2010s)	20	4.78	0.038	3886.8
family + epoch	18	15.49	0.0002	3901.6
family + epoch + coastline (2010s)	19	16.96	0.0001	3901.0
family + epoch + coastline (epoch)	20	17.08	0.0001	3899.1
family × epoch	19	17.31	0.0001	3901.4
family × epoch + coastline (epoch)	21	18.66	<0.0001	3898.7
family × epoch + coastline (2010s)	20	18.71	<0.0001	3900.8

Table 5.6. Fidelity (*F*) sub-model selection results from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation.

***F* sub-model structure**	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
Constant	21	0.00	0.57	3878.4
coy yrl 2+ F, 2+ M	22	1.62	0.25	3878.0
Fixed = 1	20	2.28	0.18	3882.7

Table 5.7. Model selection results (< 4 $\Delta QAIC_c$) from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation. For S, epoch = periods defined by sampling method (1993 – 1997, 1998 – 2010, and 2011 – 2013). For p, family = females / dependent bears and independent males (2 age / sex classes); ice = spring transition date; epoch = sampling period (1993 – 1995, 1997; 2011 – 2013); and t = full time variation. For r, time = 1992 – 2005 and 2006 – 2013. All p structures incorporated the radio collar covariate representing bears that were outfitted with collars that may have been used to locate them.

Model Structures							
S	*P*	*R*	*F*	**Parameters**	**ΔAICc**	**AICc Weights**	**QDeviance**
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Constant	21	0	0.52	3878.4
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	22	1.62	0.23	3878.0
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Fixed = 1	20	2.28	0.17	3882.7
coy yrl, 2+(sex × epoch)	family + epoch + ice	coy, yrl, 2+ (sex + time)	Constant	18	3.57	0.09	3888.1

Table 5.8. Model averaged (<Δ 4 $QAIC_c$) parameter estimates obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1993 – 2013.

Parameter	Class	Estimate (SE)
Survival (total)		
	Cubs of the year / yearlings	0.87 (0.06)
	2+ Females, 1993 – 1997	0.84 (0.04)
	2+ Females, 1998 – 2010	0.95 (0.02)
	2+ Females, 2011 – 2013	0.90 (0.05)
	2+ Males, 1993 – 1997	0.89 (0.02)
	2+ Males, 1998 – 2010	0.87 (0.02)
	2+ Males, 2011 – 2013	0.78 (0.06)
Reporting		
	Cubs of the year	0.06 (0.05)
	Yearlings	0.13 (0.07)
	2+ Females, 1993 – 2005	0.19 (0.05)
	2+ Females, 2006 - 2013	0.16 (0.05)
	2+ Males, 1993 – 2005	0.30 (0.03)
	2+ Males, 2006 – 2013	0.26 (0.06)
Fidelity		
	Cubs of the year, yearlings, and 2+ females	0.96 (0.03)
	2+ Males	0.97 (0.02)

Table 5.9. Model averaged (<Δ 4 $QAIC_c$) estimates of abundance ($\widehat{N}$ ± SE; [95% Confidence Interval]) obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1994 – 1997, 2011 – 2013. The 2011 estimate is believed to be biased based on a limited sample of surviving bears from the 1990s that were available for recapture (see Methods).

1994	1995	1997	2011	2012	2013
2280 ± 615 (1073-3486)	1999 ± 359 (1295-2703)	2239 ± 393 (1469-3009)	4202 ± 1762 (749-7656)	2595 ± 352 (1905-3286)	3056 ± 426 (2221-3893)

Table 5.10. Model selection results (< 4 $\Delta QAIC_c$) from analysis of geographic subset of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation. For S, epoch = periods defined by sampling method (1993 – 1997, 1998 – 2010, and 2011 – 2013). For p, family = females / dependent bears and independent males (2 age / sex classes); ice = spring transition date; epoch = sampling period (1993 – 1995, 1997; 2011 – 2013); inland = proximity of initial captur to smoothed coastline; and t = full time variation. For r, time = 1992 – 2005 and 2006 – 2013. All p structures incorporated the radio collar covariate for bears that were outfitted with collars that may have been used to locate them.

Model Structures							
S	*P*	*r*	*F*	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Constant	21	0.00	0.29	3361.5
coy yrl, 2+(sex × epoch)	family + epoch + ice + coastline (epoch)	coy, yrl, 2+ (sex + time)	Constant	20	0.80	0.19	3364.3
coy yrl, 2+(sex × epoch)	family + epoch + ice	coy, yrl, 2+ (sex + time)	Constant	18	1.09	0.17	3368.7
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	22	1.71	0.12	3361.2
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Fixed = 1	20	2.46	0.08	3366.0
coy yrl, 2+(sex × epoch)	family + epoch + ice + coastline (epoch)	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	21	2.49	0.08	3364.0
coy yrl, 2+(sex × epoch)	family + epoch + ice	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	19	2.76	0.07	3368.3

Table 5.11. Model averaged (<Δ 4 $QAIC_c$) parameter estimates obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1993 – 2013, using the geographic data subset.

Parameter	Class	Estimate (SE)
Survival (total)		
	Cubs of the year / yearlings	0.89 (0.06)
	2+ Females, 1990s	0.85 (0.04)
	2+ Females, Gap	0.95 (0.02)
	2+ Females, 2011 – 2013	0.91 (0.05)
	2+ Males, 1990s	0.89 (0.03)
	2+ Males, Gap	0.87 (0.02)
	2+ Males, 2011 – 2013	0.78 (0.06)
Reporting		
	Cubs of the year	0.08 (0.07)
	Yearlings	0.10 (0.07)
	2+ Females, 1993 – 2005	0.19 (0.05)
	2+ Females, 2006 - 2013	0.17 (0.06)
	2+ Males, 1993 – 2005	0.29 (0.03)
	2+ Males, 2006 – 2013	0.27 (0.06)
Fidelity		
	Cubs of the year, yearlings, and 2+ females	0.95 (0.03)
	2+ Males	0.95 (0.03)

Table 5.12. Model averaged (<Δ 4 $QAIC_c$) estimates of abundance ($\widehat{N}$ ± SE) obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1994 – 1997 and 2011 – 2013, using the geographic data subset.

1994	1995	1997	2011	2012	2013
2545 ± 597	2208 ± 382	2225 ± 418	2516 ± 1473	2447 ± 423	2659 ± 442

Figure 5.1. The BB subpopulation boundaries include portions of Nunavut, Canada, and West Greenland.

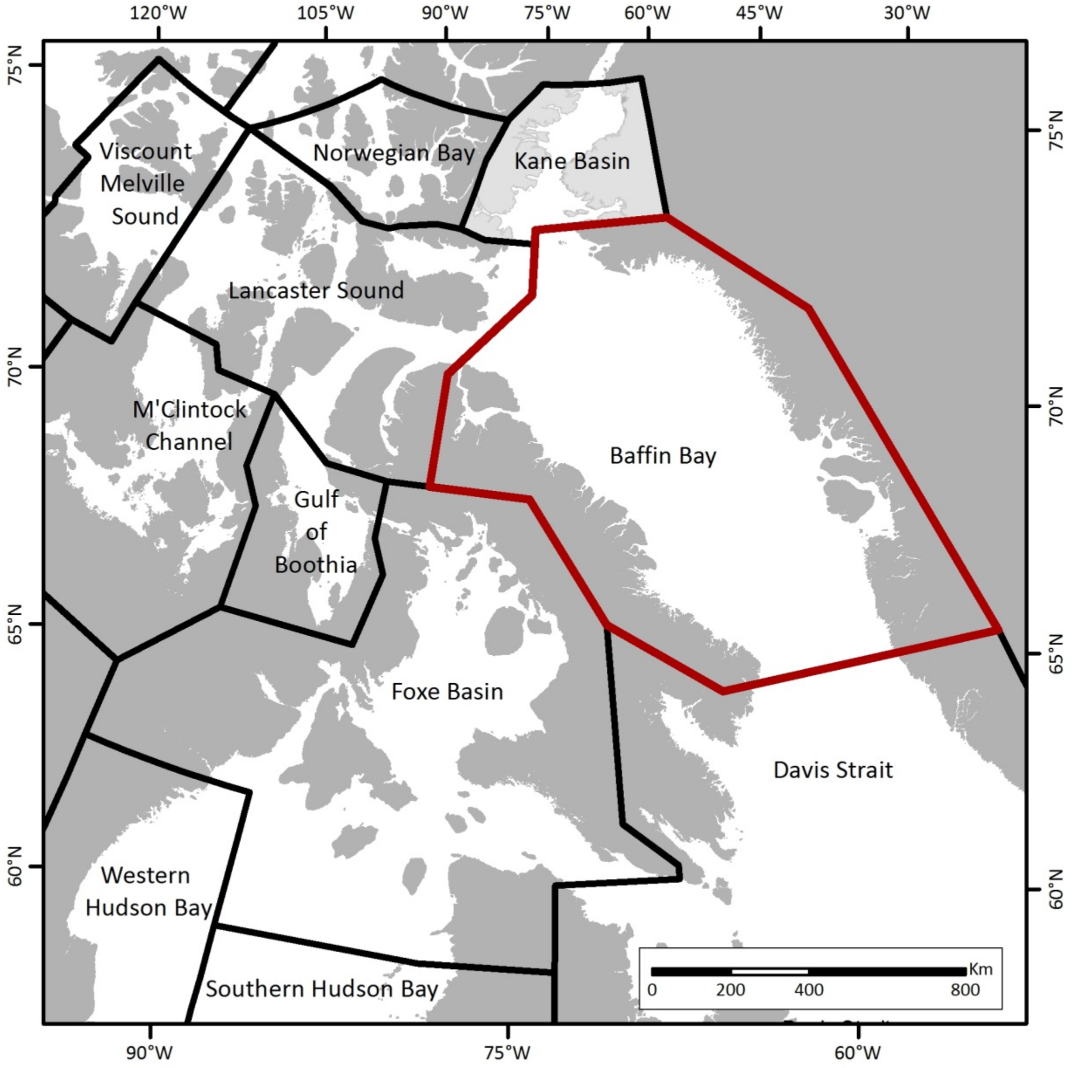


Figure 5.2. Locations of polar bears captured in Baffin Bay during the 1970s, 1980 - 1985, and 1989 – 1993.

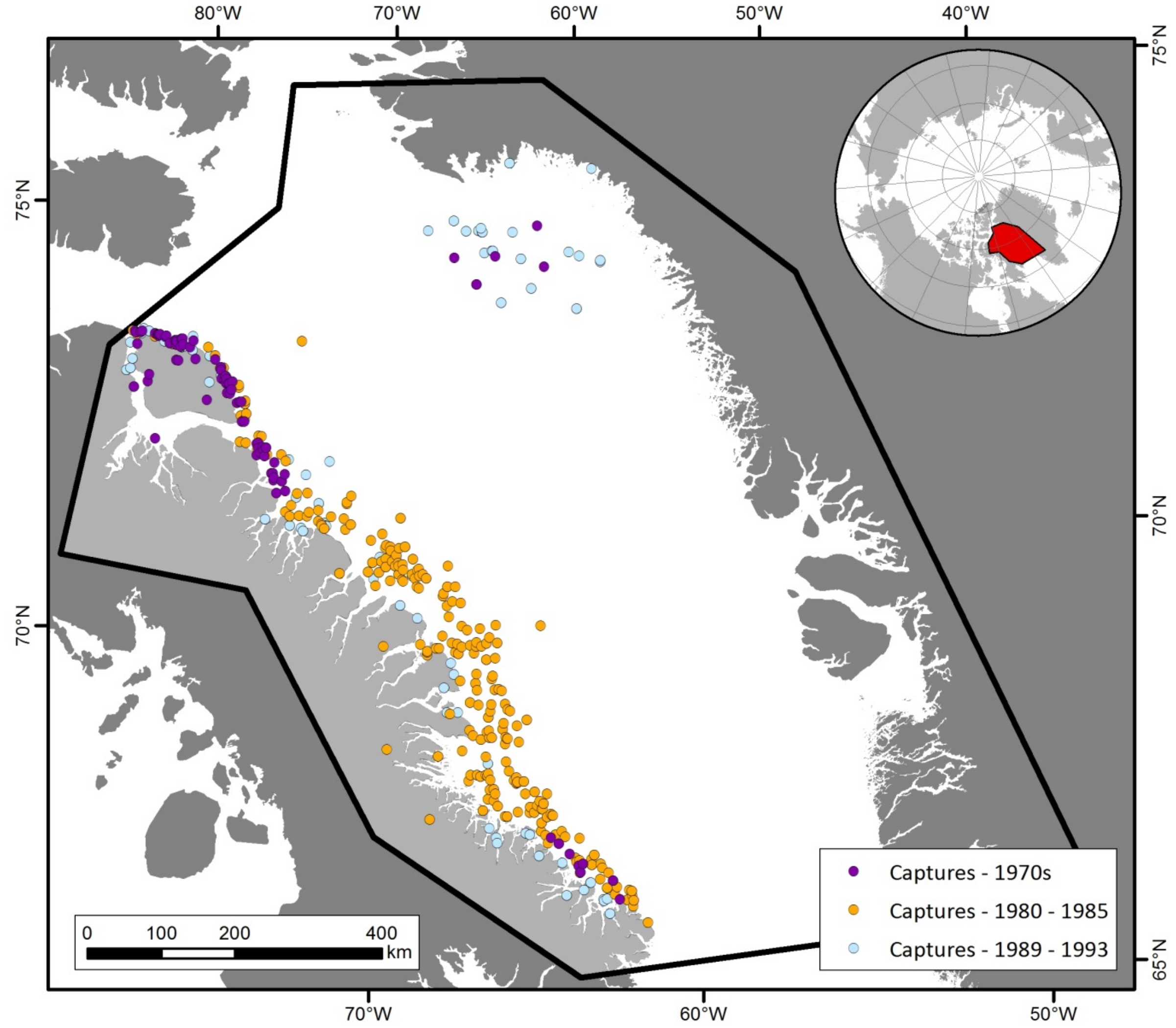


Figure 5.3. Locations of polar bears captured in Baffin Bay during August – October, 1993 – 1995 and 1997.

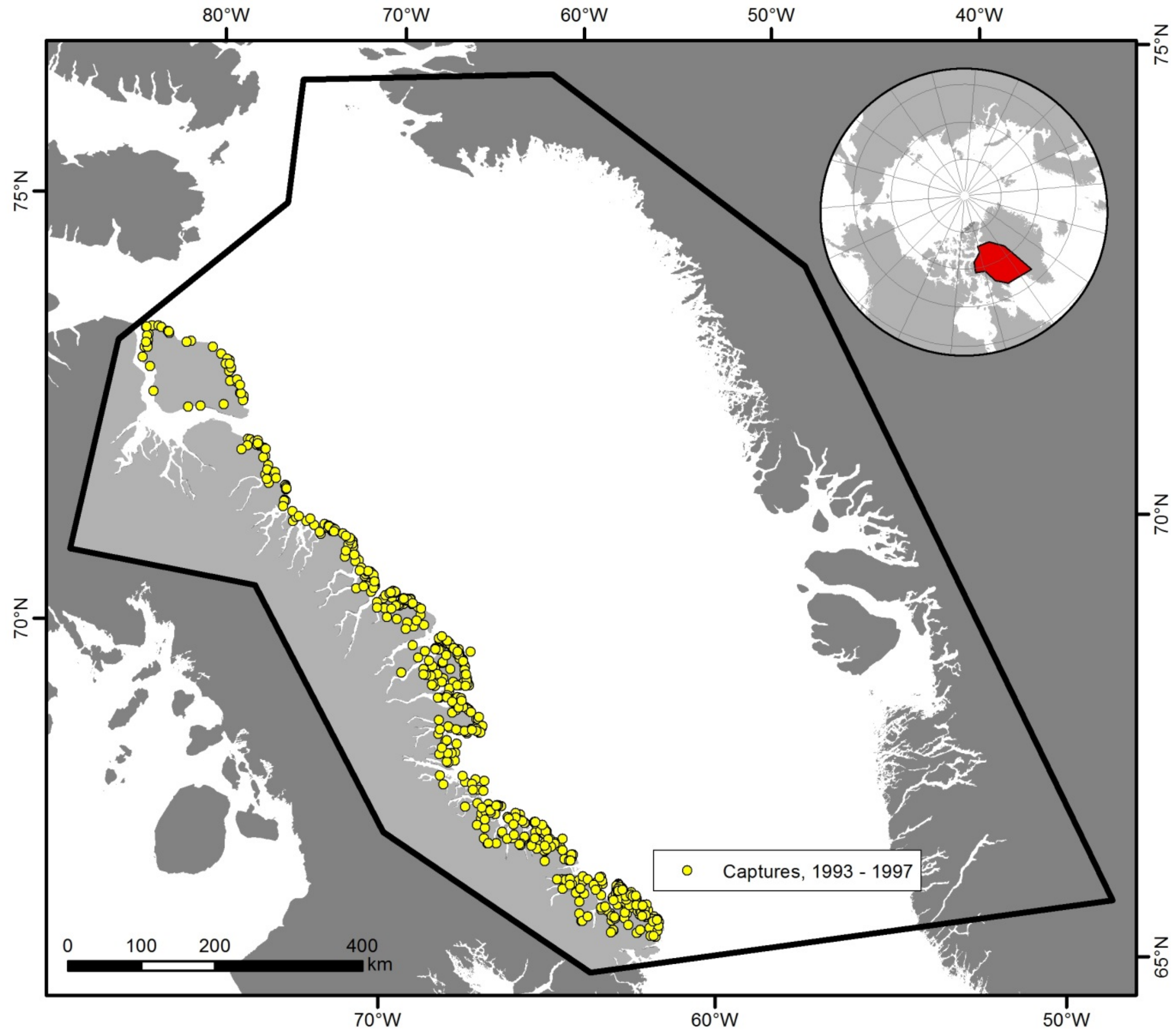


Figure 5.4. Sampling strata delineated on Baffin Island, Canada, for genetic mark-recapture study completed during 2011 – 2013. Fall sampling also was completed in the nearshore regions around Melville Bay, Greenland, denoted by the yellow star.

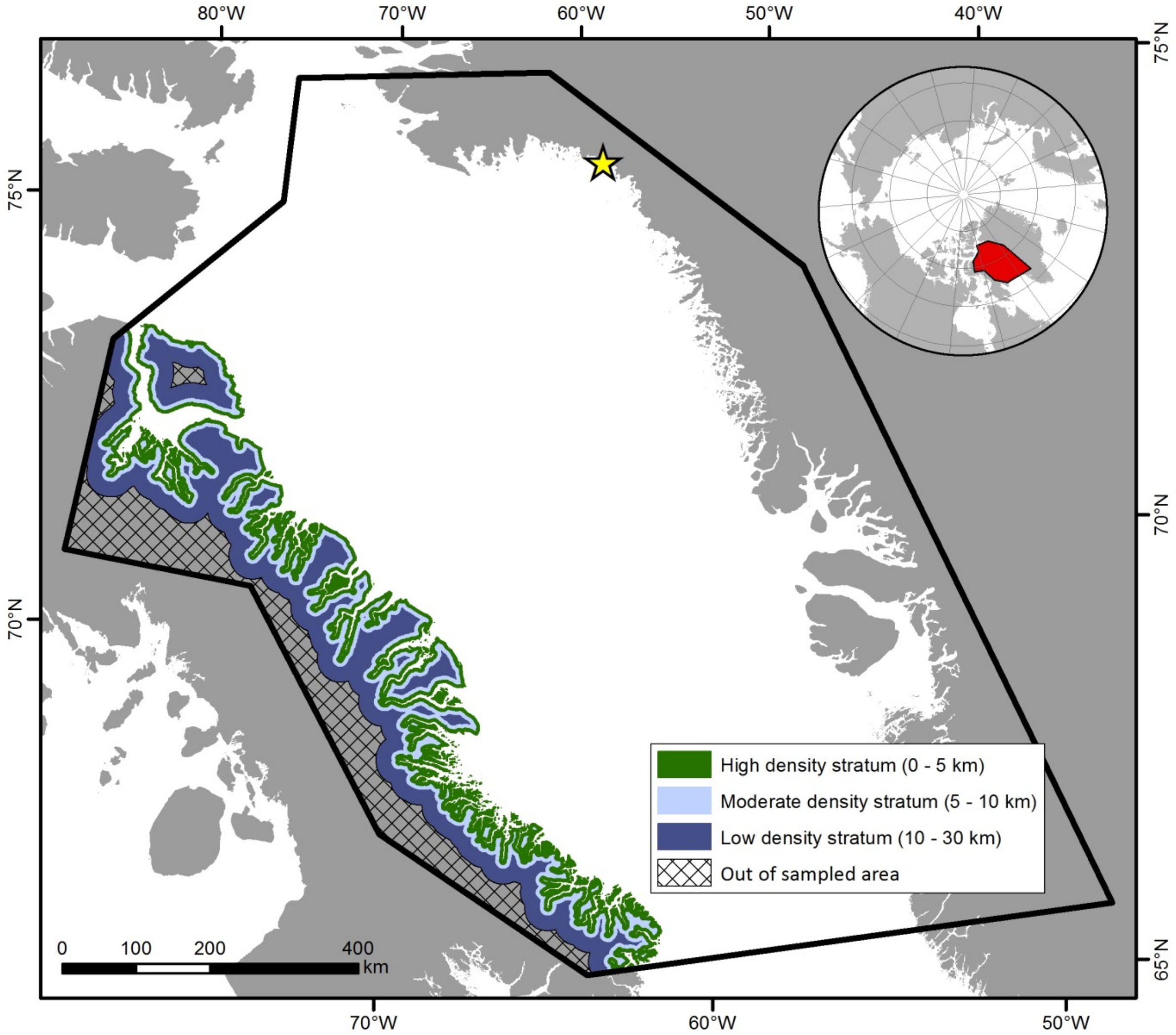


Figure 5.5. Left: Pneudart Inc. DNA (top) and Biopsy (bottom) darts used during the 2011 genetic-mark recapture in Baffin Bay. Right: A sample of skin and fat provide by a DNA dart.

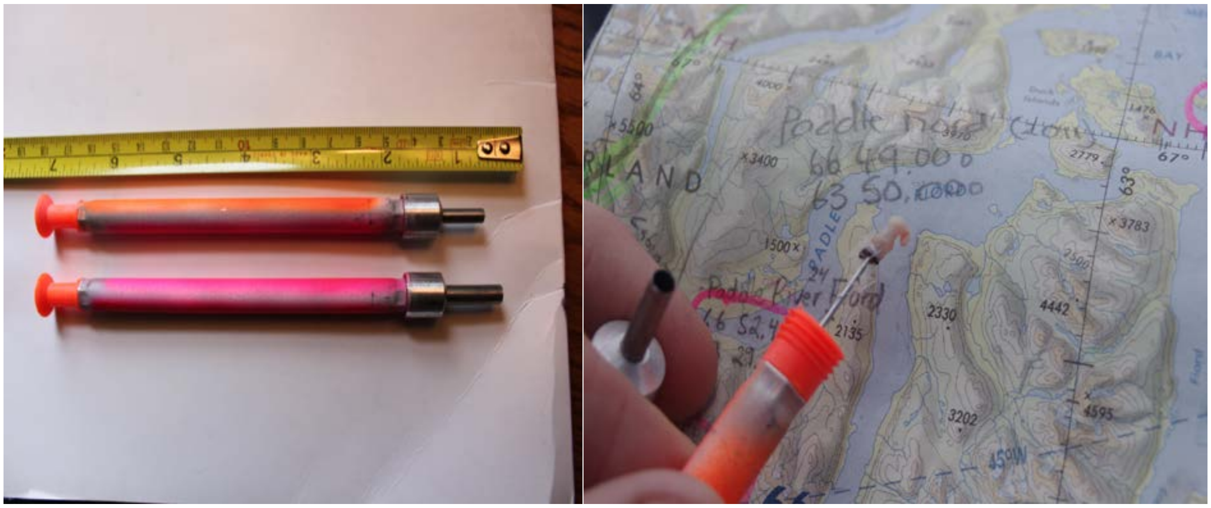


Figure 5.6. 9-locus mismatch distribution for 4,657 polar bears from Nunavut and the Greenland side of BB.

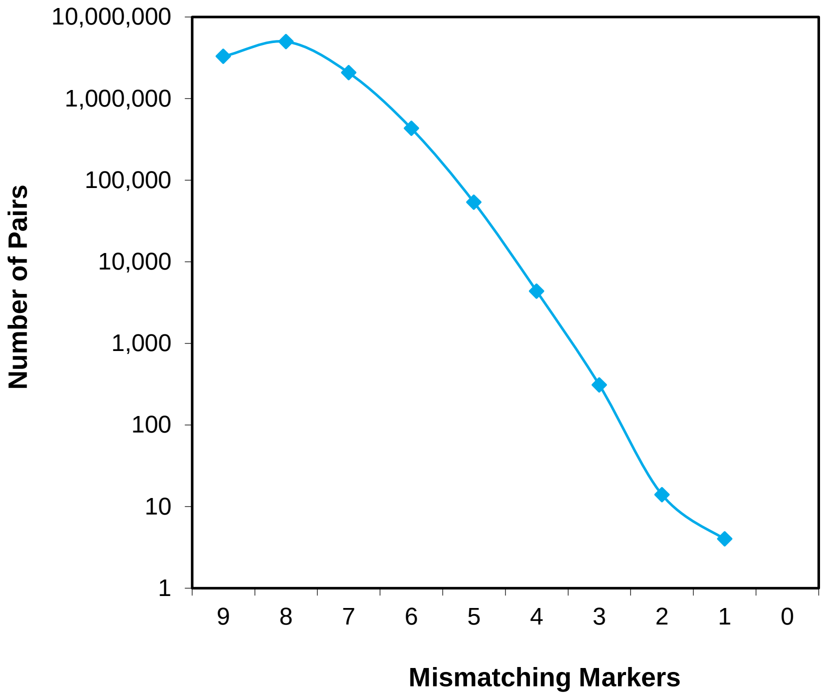


Figure 5.7. Locations of marked BB polar bears recovered in the harvest in BB and surrounding subpopulations during 1993 – 2013.

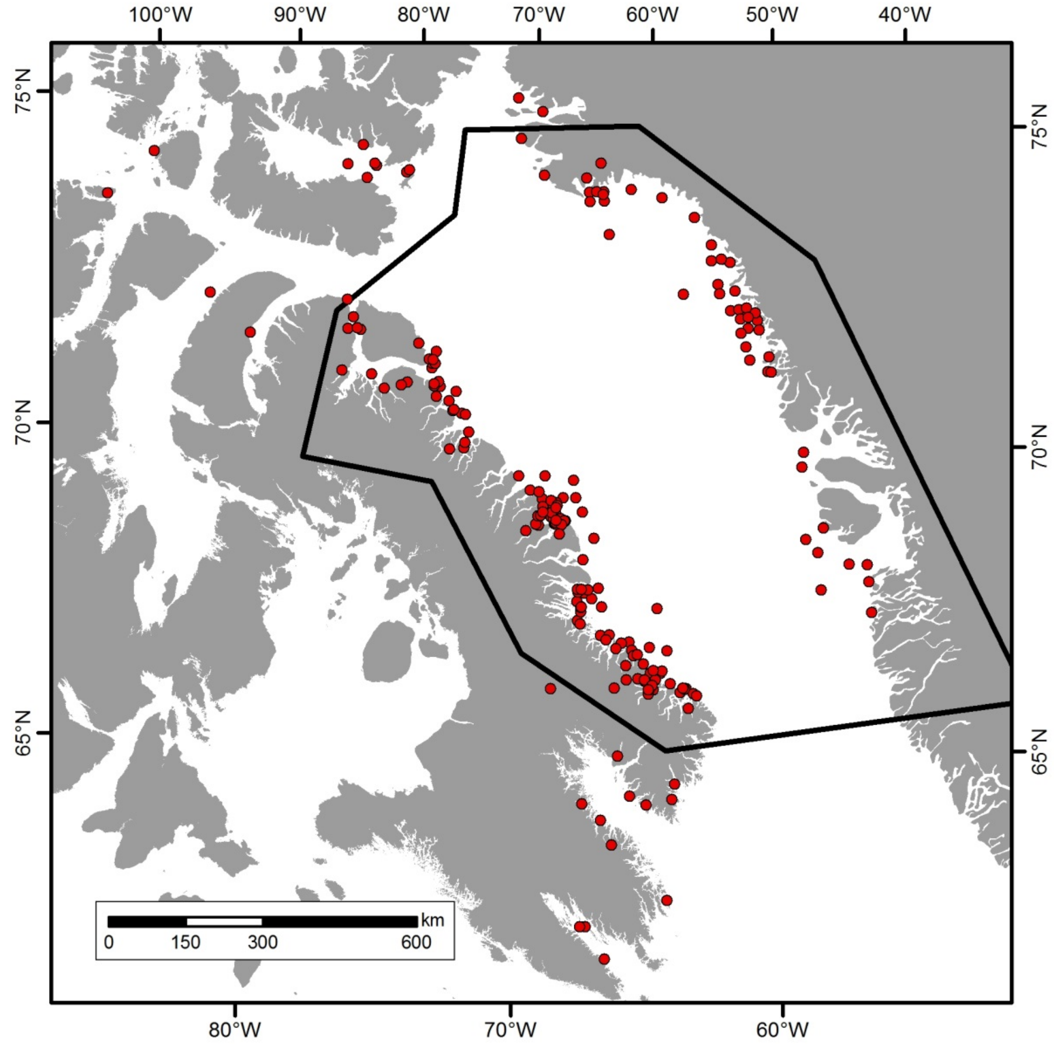


Figure 5.8. Locations of polar bears sampled in Baffin Bay during August – October, 2011 – 2013. Sampling in Greenland occurred near Melville Bay.

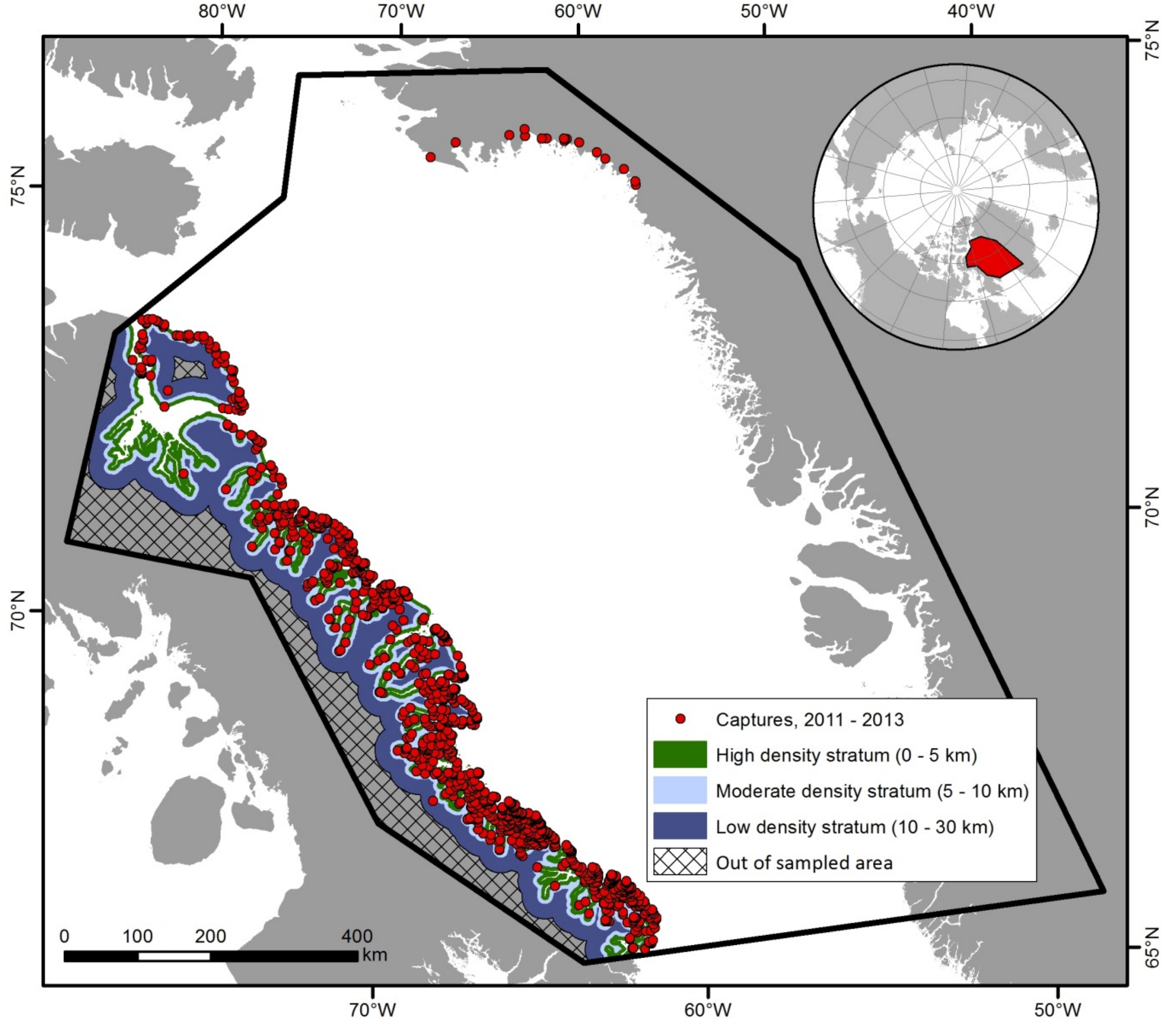
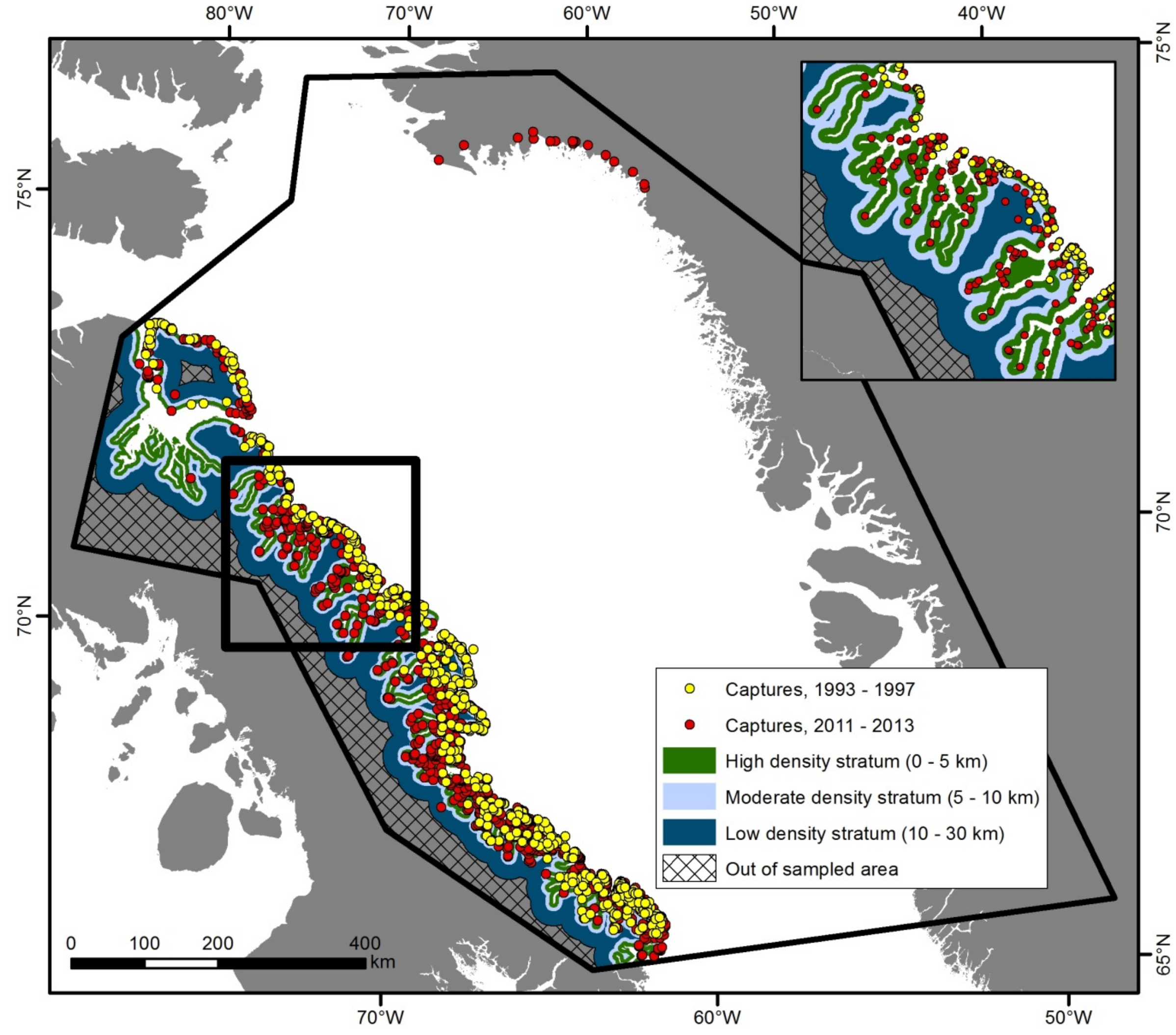


Figure 5.9. Locations of polar bears sampled in Baffin Bay during August – October, 1993 – 1995, 1997 and 2011 – 2013. The region bounded by the black square is enlarged in the inset. Note the absence of captures in fjords on Baffin Island and in northwestern Greenland during the 1990s.



CHAPTER 6

REPRODUCTIVE METRICS FOR MARK-RECAPTURE SAMPLED POLAR BEARS IN BAFFIN BAY

KEY FINDINGS

- We found little evidence of changes in litter size amongst COY or yearlings in BB. Annual variation in litter size was largely not significant and there were no trends over time or in association with spring transition date.
- We found evidence of a temporal trend in our index of COY recruitment (calculated as the number of COY per adult female in the MR sample) that was closely associated with variation in sea-ice conditions. From 1993-1995, 1997, 2011-2013, cub recruitment declined concurrent with a trend towards earlier spring sea-ice break-up. Similar declines in reproduction over time and in association with sea-ice conditions have been previously reported for polar bear subpopulations, but not in Baffin Bay.
- We found evidence of declines in body condition amongst bears in BB, following with previous studies that showed the same (Rode et al. 2012).
- Estimated annual recruitment (calculated as the number of yearlings per adult female in the MR sample) for BB during 1993-2013 ranged from 0.24 to 0.51, suggesting that BB continues to exhibit the level of reproduction requires for a viable population, according to the metric reported by Regehr et al. (2015), who suggest that variation in yearling recruitment may be a primary mechanism driving changes in population growth with values ranging from 0.1 to 0.3 necessary for population persistence.
- We found an association between COY recruitment in year t and yearling recruitment in year *t* + 1, as well as a strong association between COY litter size and yearling recruitment to following year. These findings suggest that a majority of mortality of polar bear cubs occurs during the first 8 months of life, such that recruitment of yearlings is heavily dependent on the number of COY that are born and survive up to the first ice-free period. Given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set.
- Although values in the 2000s tended to be lower than those observed in the 1990s, we did not find a significant temporal trend in annual yearling recruitment. However, given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set.
- Mean litter sizes in Baffin Bay and the proportions of COY and yearlings were similar to those observed in Foxe Basin and Southern Hudson Bay. In contrast, metrics for Baffin Bay were notably higher than those for estimated for Davis Strait and Western Hudson Bay. These comparisons suggest that BB remains a relatively productive subpopulation despite what appear to be recent declines in reproduction.

6.1. Introduction

For populations of large, long-lived mammals, changes in reproductive performance can be one of the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1981, 1987). In populations approaching carrying capacity, declines in reproductive performance are likely to occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring indices or metrics of reproduction may therefore provide a useful tool for detecting potential population trends that may warrant more in-depth study. This is especially true for populations in which cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring reproductive metrics may provide a form of surveillance that can be used to trigger more intensive study.

Reproductive metrics have been identified as an important component for monitoring polar bears across their circumpolar range (Vongraven et al. 2012). These metrics may be used to track long-term trends in the status of subpopulations, parameterize population viability models and support harvest risk assessments (Regehr et al. 2015). In particular, changes in reproduction are predicted to be amongst the first subpopulation-level effects of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Indeed, declines in reproduction have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (Regehr et al. 2007, Rode et al. 2010, Rode et al. 2014).

Changes in reproductive metrics can signal significant changes in subpopulation status of polar bears. However, observations of poor reproductive performance alone do not necessarily imply a decline in subpopulation status. Studies of several polar bear subpopulations have documented declines in reproduction associated with increases in abundance that may be the result of density dependence (Derocher 2005, Peacock et al. 2013). In other cases, variation in reproductive performance within or amongst subpopulations has been attributed to geographic or annual variation in biological productivity and prey availability (i.e., variable carrying capacity; Stirling and Øritsland 1995, Stirling and Lunn 1997, Stirling 2002, Rode et al. 2014). Information on reproduction must therefore be considered alongside other measures of subpopulation performance in-order to properly assess status.

The Baffin Bay (BB) subpopulation is part of the seasonal ice ecoregion as defined by Amstrup et al. (2008) in which sea ice melts almost entirely in the summer and bears are forced ashore for extended periods of time, during which they have no or reduced access to food. Baffin Bay has experienced a long-term reduction in sea-ice cover and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapter 3). As a result, bears are spending an increasing amount of time on land. Previous studies have documented a decline in body condition amongst BB bears in association with these trends in sea ice (Rode et al. 2012), and similar trends were found in the current study (see Chapter 7), however changes in reproductive metrics in BB have not been reported.

We summarized reproductive metrics for the Baffin Bay polar bear subpopulation using data collected during two periods of mark-recapture sampling from 1993 to 1997 (Taylor et al. 2005) and 2011 to 2013 (this report). Annual variation in reproduction was examined to assess trends over time and to evaluate the hypothesis that reproductive performance varied with sea-ice conditions. Because Baffin Bay has been infrequently monitored, we also sought to assess the utility of reproductive metrics as a surveillance tool for monitoring subpopulation status between periods of more in-depth demographic study, especially since some metrics can be obtained by methods that do not require physical capture (e.g., aerial surveys, harvest monitoring). Finally, we compared reproductive metrics for BB with other subpopulations to make inferences about the relative performance of this subpopulation. Results from these analyses provide context for understanding the status of BB polar bears.

6.2. Materials and Methods

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all Bylot Island (66.2°N to 73.8°N) in Nunavut/ Canada and parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al. 2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south. Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). Some bears remain on land in northwestern Greenland throughout the ice-retreat period.

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in BB, Canada. Sampling occurred along eastern Baffin and Bylot Islands during the ice-free season from late August to mid-October in 1993-95, 1997 and 2011-2013. Most bears in Baffin Bay move onto land on Baffin and Bylot Islands in late summer as the sea ice breaks up and remain on land until freeze-up in the late fall (Taylor et al. 2005). Sampling was extensive across this on-land study area during both periods (1990s and 2000s) of the study (Figure 6.1). The remote biopsy sampling in Greenland conducted in 2012 and 2013 were not included in this analyses.

Various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that a presumably relatively small proportion of the BB subpopulation summers in the Qimmusseriarsuaq / Melville Bay area of NW Greenland rather than moving with the retreating sea ice and summering on Baffin and Bylot Islands.

During the 1990s, bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2005). Data on the sex, estimated age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred by biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. We remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) from the air at a range of 10-20 feet above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males, e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B).

Reproductive Metrics

We calculated annual reproductive metrics that have been previously recommended (Vongraven et al. 2012) or used in studies of polar bears (e.g., Stirling et al. 1980, Derocher and Stirling 1995, Rode et al. 2010, Peacock et al. 2013, Regehr et al. 2015), including mean litter sizes (± SD) for cubs-of-the-year (COY) and yearlings. Because we did not have estimated ages for adult females sampled during 2011-2013, we calculated a pooled mean for each year rather than age-specific values. Recruitment indices were calculated as the total number of COYs or yearlings divided by the total number of adult females in the sample (Derocher and Stirling 1995, Regehr et al. 2015). Calculation of reproductive metrics considered only those COY and yearlings accompanying their mother at time of observation. During the 1990s, approximately 6% of yearlings were found to be independent of their mother during the ice-free period (GN unpublished data). These independent yearlings were not included in the calculation of yearling recruitment since we were not able to identify independent yearlings encountered during the 2000s sampling period with a known degree of accuracy.

Although Taylor et al. (2005) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island, recent examination of mark-recapture and telemetry data collected in the 1990s suggest sampling was spatially restricted to a portion of the BB subpopulation’s seasonal range and did not sample bears located farther inland or on the sea ice (GN unpublished data, Chapter 3). In contrast, sampling during 2011 – 2013 was more comprehensive and systematic on onshore areas, and the amount of un-searched sea ice during the sampling period was greatly reduced. To explore the potential impact of this difference in sampling between epochs on the calculation of reproductive metrics, we estimated and delineated the minimum extent of the sampling frame from the 1990s using capture locations in a GIS. We subset the 2000s data using this layer to create a dataset collected over the same sampling area in both time periods. We then recalculated reproductive metrics using this geographic subset. We expected that reproductive metrics calculated for the 2000s using the subset sampling area would reduce potential bias by adjusting for bears that may have been functionally missed by the limited geographic scope of sampling on land in the 1990s, given that satellite telemetry data indicated that onshore habitat use did not vary between epochs. However, we note that this geographic subset exercise would not correct for bears that may have been missed during the 1990s due to their location on the sea ice during the fall sampling period. Within season recaptures of individuals were excluded from both data sets for analyses.

Sea-ice Metrics

Based on findings from other studies (Stirling et al. 1999, Durner et al. 2004, 2006, 2009, Regehr et al. 2007, Rode et al. 2010, 2014), we hypothesized that timing of sea-ice break-up over the continental shelf (< 300m depth) of BB may influence the annual reproductive performance of polar bears as indexed from data collected during the ice-free period. We therefore calculated the annual sea-ice metric, spring transition date, as day of the year (1-365) when ice area over the continental shelf of BB reached 50% in the spring (see Chapter 2 for methodology). Whereas some researchers have used lower sea ice thresholds for studying polar bears (e.g., Cherry et al. 2013, Rode et al. 2014, Obbard et al. 2016), the exact threshold level is less important in Baffin Bay because the changes in sea-ice area during spring and fall occur quickly, such that relatively small differences in transition dates result from small changes in the threshold values (Laidre et al. 2015). For sea ice in Baffin Bay, spring transition date is also correlated with other sea-ice metrics including fall transition date (negative correlation) and the interval between spring and fall transitions (Laidre et al. 2015).

We examined annual variation, temporal trends, and relationships to sea ice for reproductive metrics. Similar to Rode et al. (2014), we did not include both time and sea ice in the same analyses since long-term trends in the spring transition date were well-established *a priori* (Laidre et al. 2015, this report) and these two parameters were correlated over the 7 years of sampling (Pearson coefficient = -0.846, $p = 0.017$). We also examined relationships between metrics for COYs and those for yearlings in the following year for periods when sampling occurred in successive years.

To examine temporal trends in reproductive metrics and associations with sea ice, we used weighted least squares regression (General Linear Model procedure), with number of litters sampled each year as the weighting variable. Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

6.3. Results

During mark-recapture sampling in 1993-1995, 1997, and 2011-2013, we sampled 251 family groups consisting of an adult female and 1-3 dependent COY (400 COY in total; Table 6.1). During this period we also sampled 152 family groups consisting of an adult female and 1-3 dependent yearlings (231 yearlings in total). The mean number of family groups sampled annually was 35.9 (range: 16-62) and 21.7 (range: 8-44) for COY and yearling families respectively. From these data we created a second subsampled dataset base on geographic location to exclude bears that were encountered outside the standardized sampling area as described previously. This geographic subset consisted of 191 COY family groups containing 1-3 dependent COY (300 COYs in total) and 105 yearling family groups containing 1-3 dependent yearlings (160 yearling in total; Table 6.2). The mean number of family groups sampled annually was 27.3 (range: 16-48) and 15.0 (range: 8-25) for COY and yearling families respectively. Annual reproductive metrics for these two datasets are presented in Tables 6.1 and 6.2.

Litter Size

There was annual variation in mean litter size amongst adult females with COYs but differences amongst years were non-significant for both the full (Kruskal-Wallis, $P = 0.153$) and geographic subset (Kruskal-Wallis, $P = 0.069$) data (Figure 6.2). For adult females with yearlings, differences in mean annual litter size were not significant for the full dataset (Kruskal-

Wallis, *P* = 0.051). For the subset data, differences in mean yearling litter size were significant (Kruskal-Wallis, *P* = 0.035) with both 1997 and 2013 having significantly lower litter sizes than 1994 and 1993 (Bonferonni correction for multiple comparisons, *P* < 0.050 in each case; Figure 6.3).

There were no statistically significant temporal trends in annual COY or yearling mean litter size from 1993 to 2013 (Table 6.3). Similarly, mean litter sizes were not associated with date of spring sea-ice transition.

Recruitment

COY recruitment showed a significant negative temporal trend from 1993 to 2013 for the geographic subset data, but not the full dataset (Table 6.3). COY recruitment also exhibited a positive association with date of spring sea-ice transition (Figure 6.4) for both the full and geographic subset data (i.e., later spring break-up was associated with higher COY recruitment). For yearlings, annual recruitment was not associated with either time or spring transition date (Figure 6.4).

There were 4 instances where sampling occurred over 2 successive years: 1993-94, 1994-95, 2011-2012, and 2012-2013. For these back-to-back years we compared reproductive metrics for COY (time t) to those of yearlings in the following year (time t+1). There was no relationship between annual mean litter size for COY in year *t* and yearling litter size in year *t + 1* (Table 6.4; Figure 6.5). COY recruitment was positively associated with yearling recruitment the following year for the geographic subset data but not the full dataset. COY litter size was closely associated with yearling recruitment in the following year for both datasets (Table 6.4; Figure 6.5), such that higher mean COY litter size in year *t* resulted in higher yearling recruitment in year *t + 1*. We also examined the relationship between yearling reproductive metrics and spring ice transition date the previous year finding no association for either the full ($F_{1,6} = 0.128$, $r^2 = 0.025$, $P = 0.735$) or subset data sets ($F_{1,6} = 0.095$, $r^2 = 0.019$, $P = 0.771$).

6.4. Discussion

Calculating annual reproductive metrics from mark-recapture field data is subject to several potential sources of error, including non-random sampling with respect to the overall study subpopulation. Similar to other seasonally ice-free subpopulations (e.g., Derocher and Stirling 1990), polar bears in Baffin Bay are known to exhibit a degree of spatial segregation by sex, age class and reproductive status with respect to the use of terrestrial habitat during the ice-free period (Ferguson et al. 1997; Chapter 4). For example, adult females with cubs tend to select fjords and avoid offshore islands and coastal regions where densities of adult males are higher (Ferguson et al. 1997). Pregnant bears select inland and upland denning habitats (Chapter 3). Mark-recapture sampling in the 1990s was more restricted in geographic extent relative to the 2010s (Chapter 3). The extent to which this difference in sampling between the two time periods introduced error and more importantly, systematic bias into our estimates of reproduction, cannot be fully evaluated. However, sampling bias is unlikely to account for the results of our analyses for several reasons. First, we attempted to account for differences in sampling by restricting some analyses to data collected within a standardized sampling area. This made little difference to the results. Results based on full and subset data were very similar. Second, sampling bias between the 1990s and 2000s would not account for the temporal trends in reproduction or associations with the timing of spring sea-ice transition unless sampling bias varied in proportion to these factors; something that is unlikely. Third, under-sampling of fjord habitats in the 1990s may have introduced bias in sampling of adult females with offspring. However, the most likely impact of this bias would have been underestimation of recruitment indices since fewer adult females with offspring would have been sampled relative to adult females overall. Fourth, sampling bias would not account for the association observed between reproductive metrics in successive years and the closeness of this association in some cases.

Another source of error in estimation of reproductive parameters in our study originated from the misclassification of bears that were observed from the air rather than handled during 2011-2013. However, when combined with genetic sexing, the accuracy of this method of classification is high even for lone adult females[1] (Appendix B). Furthermore, this source of error does not necessarily introduce systematic bias. Lone adult females could only have been misclassified as lone subadult females (and vice-versa), and there is no evidence to suggest inaccuracy in this area favours one age-class versus the other.

Finally, analyses of telemetry data collected from collared adult females suggest that a proportion of collar bears remained on the remnant sea ice in some years during the 1990s and were unavailable for sampling (Chapter 3). Although this could have introduced bias if certain classes of bears tended to remain on the ice while others moved to shore, we were unable to correct for this potential source of bias. However, the most likely effect would have been over-representation of lone (pregnant) adult females in our sampling data since this is the class of adult females that has an obligate need to come ashore to look for suitable denning habitat. This, in turn, would have led to underestimation of recruitment in the 1990s relative to the 2000s.

Litter Size

---

[1] Approximately 84% of lone adult females were correctly classified (GN unpublished data; see chapter 5, appendix 1.)

We found little evidence of changes in litter size amongst COY or yearlings in BB. Annual variation in litter size was for the most part not significant and there were no trends over time or in association with spring transition date. This finding is consistent with Molnar et al. (2011), who found that although litter size is predicted to vary in response to changes in maternal body condition and environmental conditions, it is a relatively insensitive reproductive metric. Large changes in maternal condition and environment are necessary to produce statistically significant differences in litter size. Litter size does, however, remain an important reproductive metric for monitoring polar bear subpopulations (Vongraven et al. 2012). Changes in litter size have been associated with temporal and geographic variation in ecosystem productivity (Stirling and Lunn 1997, Peacock et al. 2013), and long-term trends have been detected in association with changing subpopulation status (Derocher and Stirling 1995).

Trends in COY Recruitment and Association with Sea-ice Conditions

We found evidence of a temporal trend in our index of COY recruitment that was also closely associated with variation in sea-ice conditions. From 1993 to 2013, cub recruitment declined concurrent with a trend towards earlier spring sea-ice break-up. Similar declines in reproduction over time and in association with sea-ice conditions have been previously reported for polar bear subpopulations (Derocher and Stirling 1995, Derocher 2005, Rode et al. 2010, Peacock et al. 2013, Rode et al. 2014), but not in Baffin Bay. Earlier spring break-up (also associated with later fall freeze-up) presumably decreases feeding opportunities for polar bears, thereby resulting in poorer maternal body condition and reduced investment in reproduction. This, in turn, will be manifested as reduced natality rates and / or lower offspring survival. Our index of cub recruitment incorporates both of these parameters reflecting to an unknown degree a blend of decreased cub production and lowered cub survival over the first 8 months of life. The association between cub recruitment and spring transition date in our study suggests that lower cub survival from birth to the first ice-free season may be a primary mechanism driving lower reproduction in Baffin Bay. However, we have not demonstrated a causal relationship; other factors may play an important role in cub recruitment, particularly since recruitment was associated with both time (year) and spring transition date. These two parameters are correlated with one another and may also be associated with other parameters that we did not consider. As such, there is some uncertainty as to the extent to which declining reproduction in BB is mediated by sea-ice conditions.

Declining reproduction and body condition are amongst the first subpopulation level effects predicted occur in polar bears as a result of climate change (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Our evidence of a decline in reproduction in BB from 1993 to 2013 is accompanied by evidence of concurrent declines in body condition amongst bears in this subpopulation over the same period (Rode et al. 2012, Chapter 6). These changes may signal a reduction in the carrying capacity of BB. Although the point estimate of abundance from our recent genetic mark-recapture was higher than the 1990s estimate, the difference between estimates was not statistically significant (Chapter 5). Additionally, differences in these point estimates may be largely explained by differences in sampling design between the two time periods. Regardless of whether density effects are at play, if the observed association between sea ice and reproduction is real and the well documented trend in sea-ice continues, it is reasonable to expect that this subpopulation will experience significant changes in reproductive performance as a result of declining habitat (Laidre et al. 2015).

Yearling Recruitment

Recruitment calculated as the number of yearlings per adult female has been identified as an important reproductive metric to monitor in polar bear subpopulations, incorporating both natality and survival of COY (Vongraven et al. 2012). Regehr et al. (2015) suggest that variation in yearling recruitment may be a primary mechanism driving changes in subpopulation growth, with values ranging from 0.1 to 0.3 necessary for subpopulation persistence. Estimated annual recruitment values for Baffin Bay during 1993 to 2013 ranged from 0.24 to 0.51, suggesting that BB continues to function as a viable subpopulation, according to this metric. Interestingly, in contrast to previous studies (e.g., Rode et al. 2010, 2014), we did not find evidence that yearling recruitment was associated with sea-ice conditions. Instead, our results suggest that recruitment of yearlings is largely determined by reproductive metrics for COYs in the previous year. We found an association between COY recruitment in year t and yearling recruitment in year t + 1, as well as a strong association between COY litter size and yearling recruitment to following year. These findings suggest that a majority of mortality of polar bear cubs occurs during the first 8 months of life, such that recruitment of yearlings is heavily dependent on the number of COY that are born and survive up to the first ice-free period. Since COY recruitment itself appears to be associated with spring transition date, yearling recruitment up to the ice-free period may be influenced to a greater degree by ice conditions the previous year than by ice conditions in the current year. We did not find evidence of this lag effect but our sample size was very small.

Although values in the 2000s tended to be lower than those observed in the 1990s, we did not find a significant temporal trend in annual yearling recruitment. However, given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set. In this context, we suggest that monitoring annual litter size of COY may be a useful tool for tracking trends in recruitment in the absence of more intensive subpopulation studies. This metric can be readily acquired from aerial surveys without capture or biopsy of bears, without the need for extensive observer experience in identifying age-sex classes, and at relatively low cost. Our very limited data suggest that mean annual COY litter size is closely related to yearling recruitment the following year. However, we acknowledge that the robustness of this relationship has not been validated with a larger data set and under a range of environmental conditions.

Comparison with other Subpopulations

Indices of reproduction for BB were comparable to other polar bear subpopulations in the seasonal ice ecoregion (Amstrup et al. 2008) that have been recently studied by mark-recapture or aerial survey (Table 6.5). Mean litter sizes in Baffin Bay and the proportions of COY and yearlings were similar to those observed in Foxe Basin (Stapleton et al. 2016) and Southern Hudson Bay (Obbard et al. 2015); two subpopulations classified as stable (PBSG 2010). In contrast, metrics for Baffin Bay were notably higher than those estimated for Davis Strait (Peacock et al. 2013) and Western Hudson Bay (Lunn et al. 2014). These comparisons suggest that BB remains a relatively productive subpopulation despite apparent recent declines in reproduction.

6.5. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Cavalieri, D. J., C. L. Parkinson, P. Gloersen, and H. J. Zwally. 1996, updated yearly. Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS passive microwave data, Version 1. NASA National Snow and Ice Data Center Distributed Active Archive Center, Boulder, Colorado, USA, doi: http://dx.doi.org/10.5067/8GQ8LZQVL0VL.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. Population Ecology 47:267–275.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1990. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395–1403.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Durner, G. M., S. C. Amstrup, R. Nielson, and T. McDonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea.

Pages 107–120 *in* S. Huzurbazar, editor. Resource selection methods and applications. Omnipress, Madison, Wisconsin, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use of polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Fowler, C. W. 1981. Density dependence as related to life history strategy. Ecology 62:602–610.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. Peacock, M. K. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø. Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Stapleton, S., S. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., and N. J. Lunn. 1997. Environmental fluctuations in arctic marine ecosystems as reflected by variability in reproduction of polar bears and ringed seals. Pages 167–181 *in* S. J. Woodin, and M. Marquiss, editors. Ecology of arctic environments. Special Publication No. 13 of the British Ecological Society, Blackwell Science Ltd., Oxford, UK.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Table 6.1. Reproductive metrics for annual mark-recapture sampling data from Baffin Bay. Full dataset.

Year	**Mean Litter Size (n, SD)**		**Recruitment Index[2] (Offspring/adult female)**	
	COY	**Yearling [1]**	**COY**	**Yearlings [1]**
1993	1.63 (16, 0.50)	1.75 (8, 0.46)	0.79	0.42
1994	1.87 (23, 0.51)	1.82 (11, 0.38)	0.83	0.38
1995	1.52 (21, 0.51)	1.64 (14, 0.63)	0.71	0.51
1997	1.64 (25, 0.49)	1.25 (12, 0.45)	0.65	0.24
2011	1.57 (58, 0.50)	1.53 (34, 0.51)	0.68	0.39
2012	1.47 (62, 0.50)	1.55 (44, 0.54)	0.47	0.35
2013	1.65 (46, 0.49)	1.34 (29, 0.49)	0.60	0.31

[1] Dependent yearlings only. Capture data for 1993-97 suggests up to 6% of yearlings are independent of their mother in the fall

[2] Calculated per Derocher and Stirling (1995), Vongraven et al. (2012), Regehr et al. (2015)

Table 6.2. Reproductive metrics for annual mark-recapture sampling data from Baffin Bay. Data for 2011-13 were filtered to exclude individuals encountered outside the area where sampling was estimated to have occurred in the 1990s.

Year	Mean Litter Size (n, SD)		Recruitment Index[2] (Offspring/adult female)	
	COY	Yearling [1]	COY	Yearlings [1]
1993	1.63 (16, 0.50)	1.75 (8, 0.46)	0.79	0.42
1994	1.87 (23, 0.51)	1.82 (11, 0.38)	0.83	0.38
1995	1.52 (21, 0.51)	1.64 (14, 0.63)	0.71	0.51
1997	1.64 (25, 0.49)	1.25 (12, 0.45)	0.65	0.24
2011	1.48 (33, 0.51)	1.60 (20, 0.50)	0.61	0.40
2012	1.42 (48, 0.50)	1.48 (25, 0.51)	0.55	0.30
2013	1.64 (25, 0.49)	1.27 (15, 0.46)	0.55	0.26

[1] Dependent yearlings only. Capture data for 1993-97 suggests up to 6% of yearlings are independent of their mother in the fall

[2] Calculated per Derocher and Stirling (1995), Vongraven et al. (2012), Regehr et al. (2015)

Table 6.3. Results of least squares regressions for annual reproductive metrics derived from mark-recapture sampling of polar bears during the ice-free period (August-October) in Baffin Bay. Number of litters was used as a weighting variable. Significant relationships in bold.

Reproductive Metric	Litter Age	Explanatory Variable	Dataset	$F_{1,6}$	r^2	P
Litter Size	COY	Year	Full	1.64	0.25	0.256
Litter Size	COY	Year	Subset	3.37	0.40	0.126
Litter Size	COY	Spring Ice Transition	Full	3.42	0.41	0.124
Litter Size	COY	Spring Ice Transition	Subset	3.92	0.44	0.105
Litter Size	Yearling	Year	Full	1.22	0.20	0.319
Litter Size	Yearling	Year	Subset	1.16	0.19	0.331
Litter Size	Yearling	Spring Ice Transition	Full	1.64	0.25	0.256
Litter Size	Yearling	Spring Ice Transition	Subset	1.38	0.22	0.293
Recruitment	COY	Year	Full	5.34	0.52	0.069
Recruitment	COY	Year	Subset	22.43	0.82	**0.005**
Recruitment	COY	Spring Ice Transition	Full	53.90	0.92	**0.001**
Recruitment	COY	Spring Ice Transition	Subset	11.60	0.70	**0.019**
Recruitment	Yearling	Year	Full	0.72	0.13	0.434
Recruitment	Yearling	Year	Subset	1.37	0.21	0.295
Recruitment	Yearling	Spring Ice Transition	Full	0.64	0.11	0.460
Recruitment	Yearling	Spring Ice Transition	Subset	1.91	0.28	0.225

Table 6.4. Results of least squares regressions for annual reproductive metrics derived in year *t* and year *t+1* from mark-recapture sampling of polar bears during the ice-free period (August-October) in Baffin Bay, Canada. Number of litters in year *t+1* was used as a weighting variable.

Reproductive Metric (year t)	**Reproductive Metric (year t+1)**	**Dataset**	$F_{1,3}$	r^2	***P***
COY Litter Size	Yearling Litter Size	Full	0.80	0.29	0.465
COY Litter Size	Yearling Litter Size	Subset	0.92	0.29	0.431
COY Recruitment	Yearling Recruitment	Full	3.88	0.66	0.188
COY Recruitment	Yearling Recruitment	Subset	20.33	0.91	**0.046**
COY Litter Size	Yearling Recruitment	Full	474.43	0.99	**0.002**
COY Litter Size	Yearling Recruitment	Subset	1854.94	0.99	**0.001**

Table 6.5. Comparison of reproductive metrics for polar bear subpopulations in the seasonal ice ecoregion. Sampling occurred during ice-free periods.

Subpopulation	Mean Litter Size		Proportion of Total Observations[2]		Source
	COY	Yearling	COY	Yearlings	
Baffin Bay (1993-97)	1.67	1.60	0.16	0.09	Taylor et al. (2005)
Baffin Bay (2011-13)[1]	1.55	1.48	0.15	0.09	This study
Davis Strait (2005-07)	1.49	1.22	0.08	0.09	Peacock et al. (2013)
Foxe Basin (2009-2010)	1.55	1.48	0.13	0.10	Stapleton et al. (2016)
Southern Hudson Bay (2011)	1.56	1.49	0.16	0.12	Obbard et al. (2015)
Western Hudson Bay (2011)	1.43	1.22	0.07	0.03	Stapleton et al. (2014)

[1] Based on sampling across study area

[2] Some of these recent studies relied on aerial survey which is less accurate in identifying adult females (without genotyping to determine sex). For this reason we used published data on mean litter sizes and the proportion of COY and yearlings within these studies rather than calculating indices of recruitment used in the present study

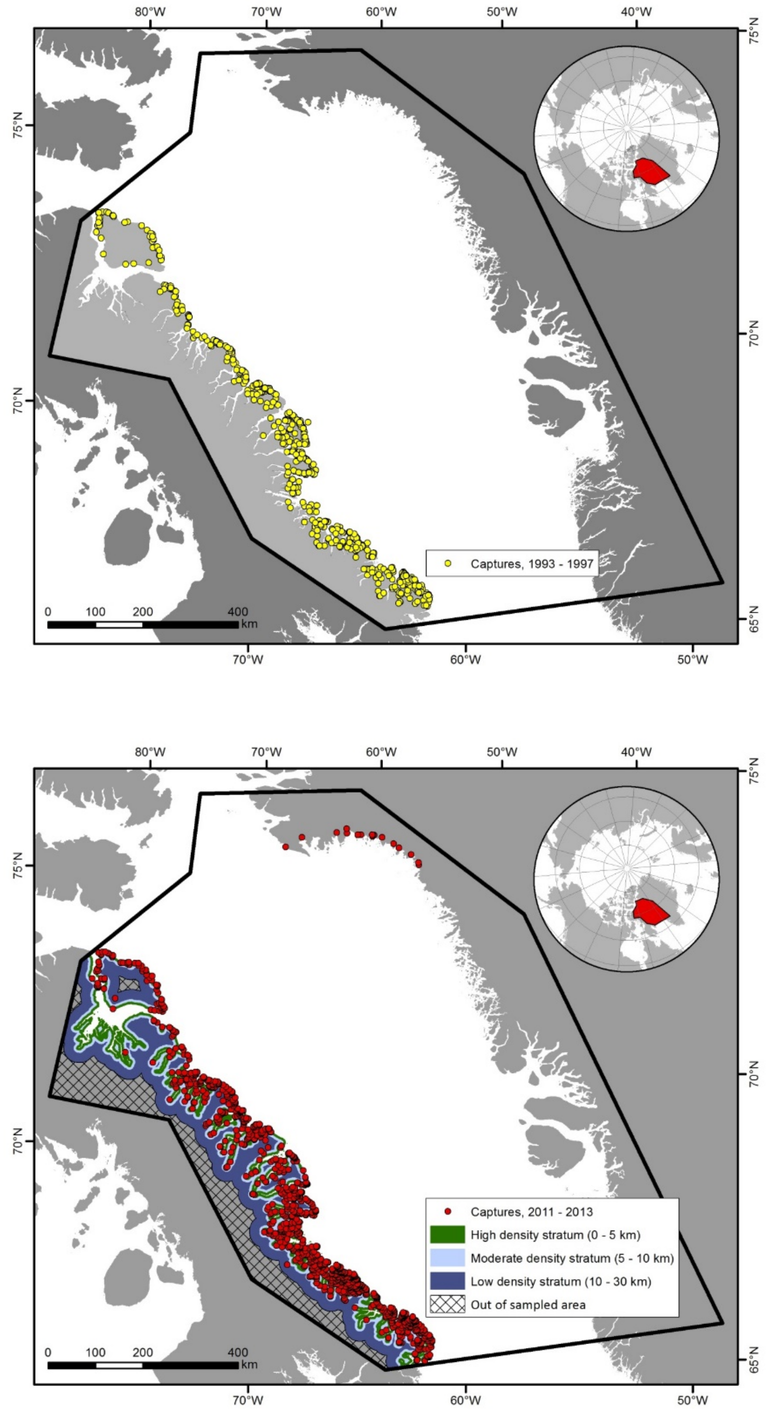


Figure 6.1. Spatial distribution of bears recorded during sampling in the Baffin Bay polar bear subpopulation, 1993 – 1995, 1997 (top), and 2011 – 2013 (bottom).

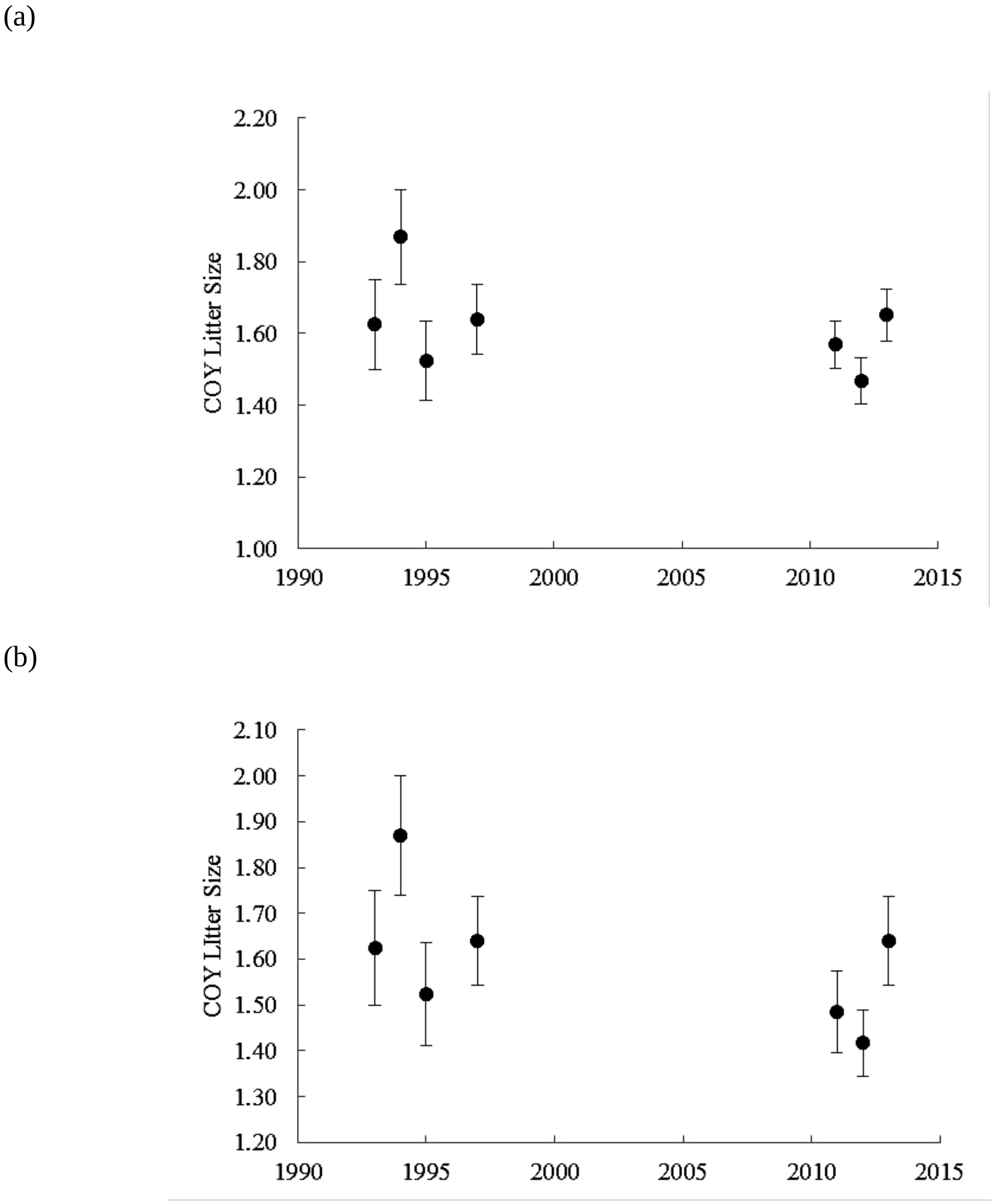
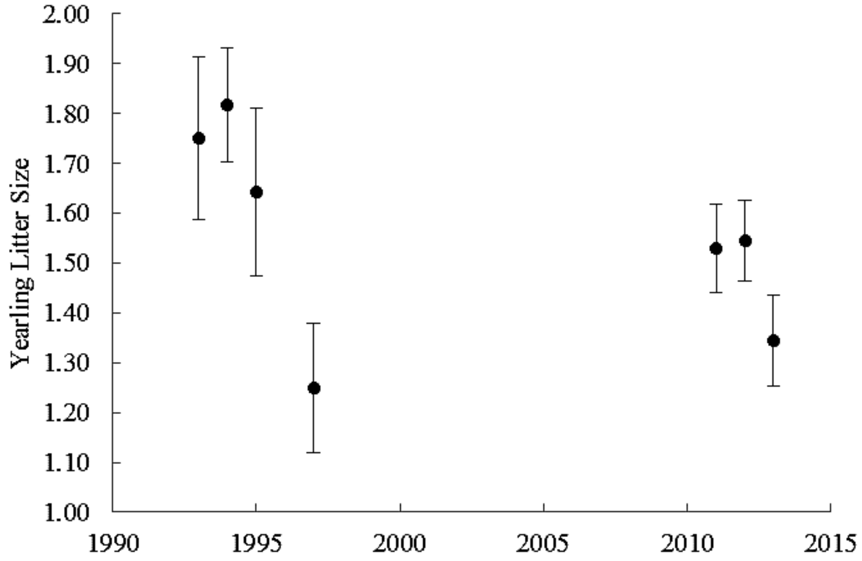


Figure 6.2. Mean litter sizes (+/- SE) for adult females with cubs-of the year (COY) during the ice-free period in Baffin Bay. Calculated using the (a) full and (b) geographically subset data (see text).

(a)



(b)

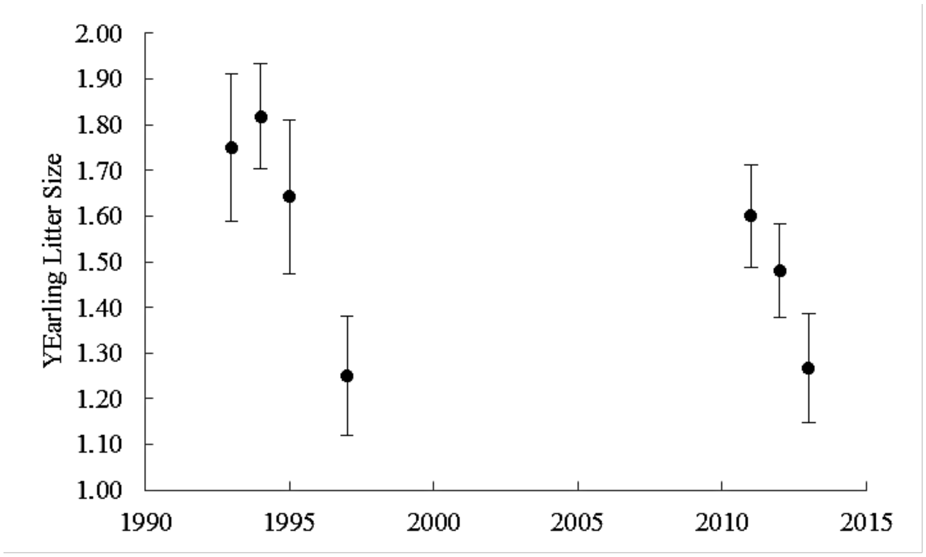


Figure 6.3. Mean litter sizes (+/- SE) for adult females with yearlings during the ice-free period in Baffin Bay. Calculated using the (a) full and (b) geographically subset data (see text).

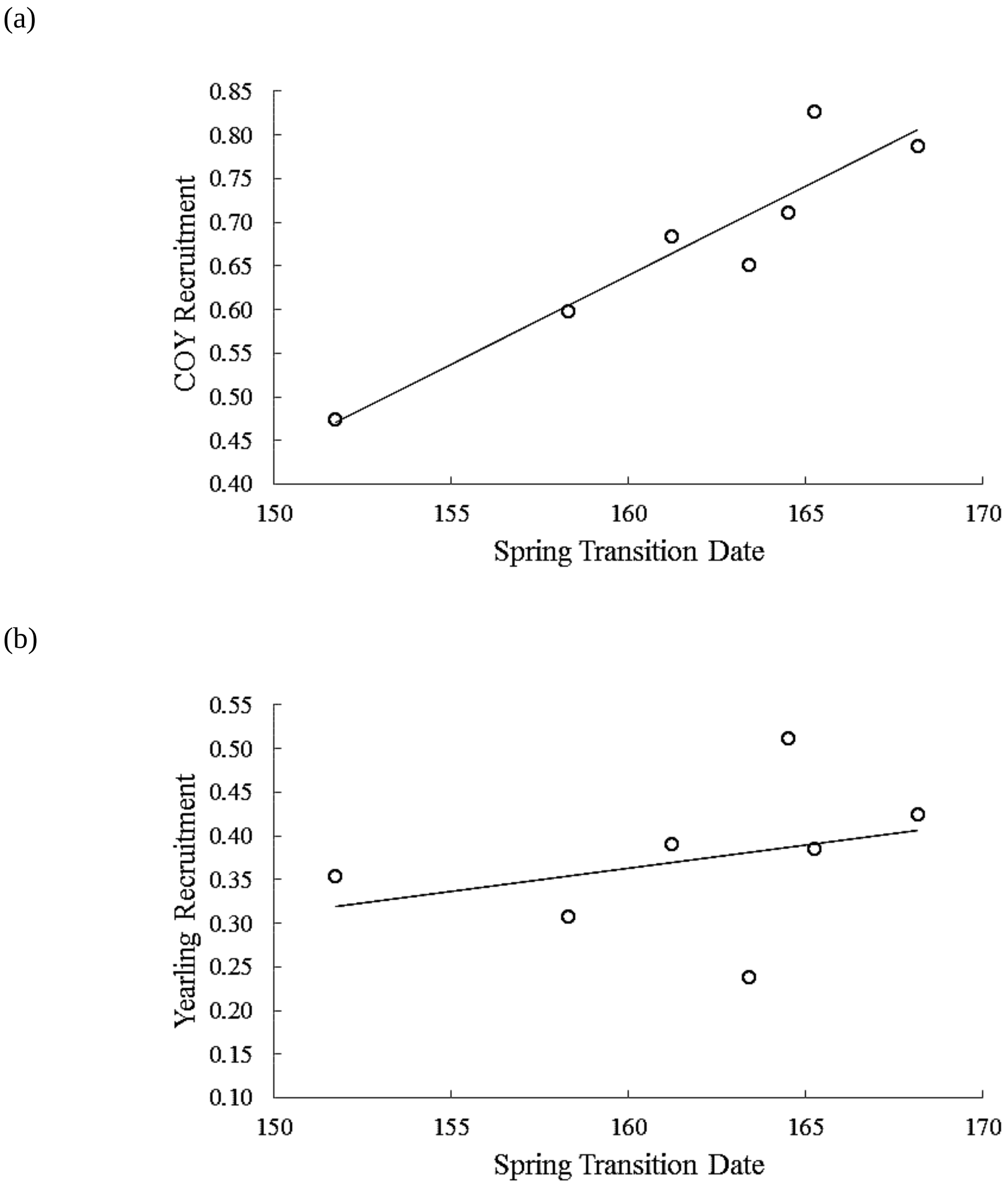


Figure 6.4. The relationship between spring transition date and annual recruitment index for (a) cub-of-the-year and (b) yearling polar bears during the ice free period (August-October), in Baffin Bay. Recruitment calculated as the number of COY or yearlings per adult female in the sample subpopulation using the full mark-recapture dataset.

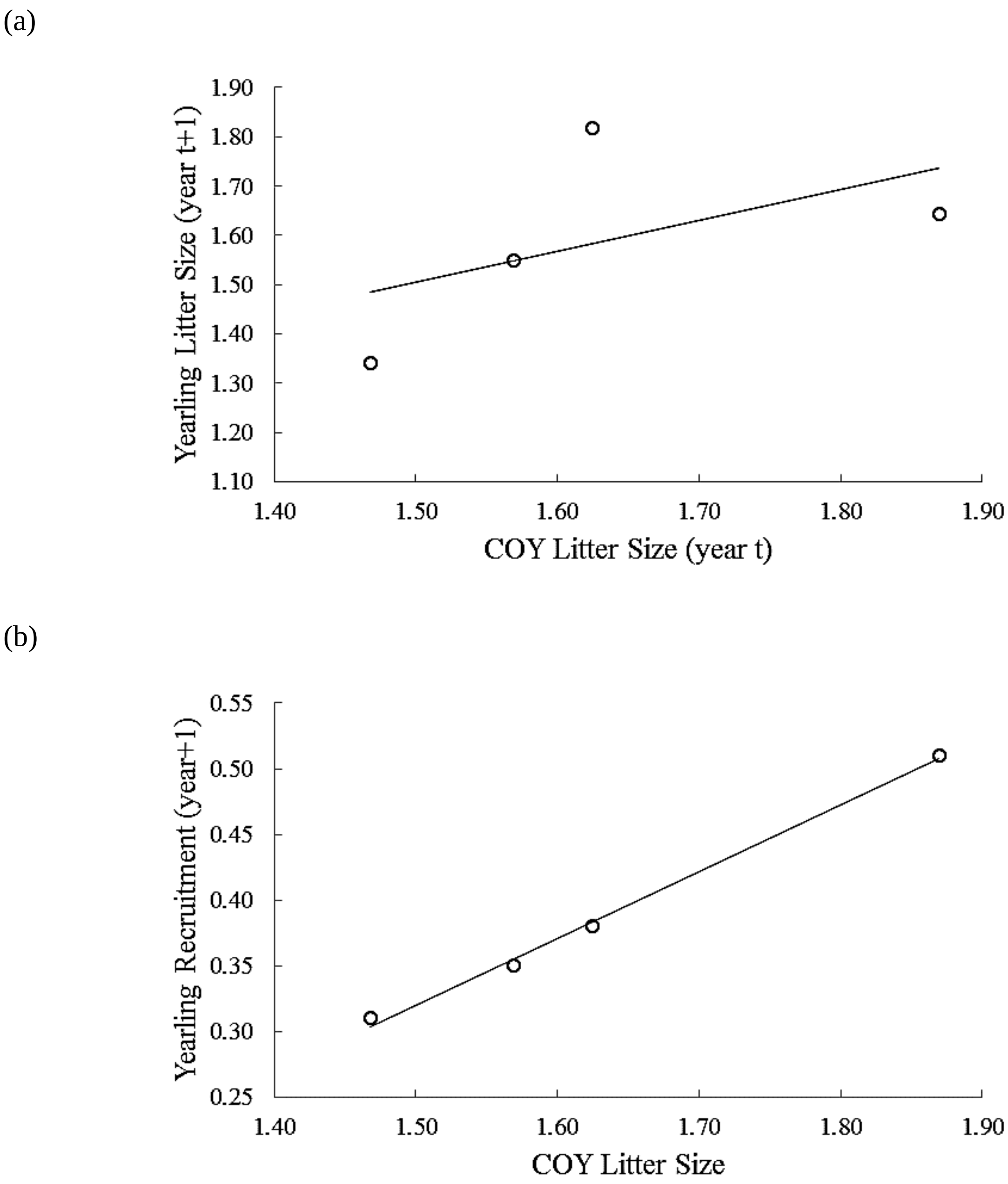


Figure 6.5. The relationship between mean annual litter size amongst cub-of-the-year litters and (a) yearling litter size and (b) yearling recruitment in the following year (year +1). Data are for polar bears sampled during the ice free period (August-October), in Baffin Bay. Metrics calculated using full mark-recapture dataset.

CHAPTER 7

BODY CONDITION OF BAFFIN BAY POLAR BEARS

KEY FINDINGS

- Fatness index (FI) scores were collected during two periods of mark-recapture sampling in BB, 1993-1995, 1997 and 2011-2013. We examined trends in this metric of body condition across both sampling periods in relation to sea-ice conditions.
- We found evidence of declines in body condition amongst bears in BB over the period 1993 to 2013. Body condition in BB polar bears declined in close association with the ice-free period and spring sea-ice transition dates. This is consistent with the hypothesis that reduced time on the sea ice is a primary mechanism driving this decline.
- Our results follow with previous studies that showed similar results through 2010 with different metrics derived from physical handing of bears (Rode et al. 2012).
- The springtime aerial survey was successfully implemented due to the small geographic These findings are consistent with available traditional knowledge suggesting that body condition of polar bears in BB was poorer in the early 2000s relative to the 1990s (Dowsley and Wenzel 2008, Born et al. 2011).
- We found evidence of recent foraging in approximately 9% of the BB bears observed during the on-land period. Marine mammals, in particular seals and walrus made up almost half of the identifiable food sources.

7.1. Background

For populations of large, long-lived mammals changes in body condition will be among the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1987, 1990, Zedrosser et al. 2006). In populations approaching K, declines in condition will occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring body condition may therefore provide a useful tool for the early detection of population trends that warrant more in-depth study. This is especially true for populations where cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring condition may provide a form of surveillance that can be used to trigger periods of more intensive study.

The annual life-cycle of polar bears (*Ursus maritimus*) is characterized by large seasonal changes in body condition (Watts and Hansen 1987, Ramsay et al. 1992, Ramsay and Stirling 1988, Atkinson and Ramsay 1995). Throughout most of their circumpolar range, bears are thought to gain condition during the spring and early summer when juvenile seals are abundant and relatively susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). This period of hyperphagia is followed by a scarcity of food in the late summer and fall when sea ice reaches a minimum throughout the Arctic. During this season, bears in some regions are forced onto land by the melting sea ice where access to seals and other marine mammal prey is greatly reduced (Stirling et al. 1977, Derocher and Stirling 1990a,b, Ramsay et al. 1991). In other regions, bears remain on off-shore pack-ice but likely also have reduced access to and/or less success in catching seals (Amstrup et al. 2000, Stirling 2002, Atwood et al. 2015, Rode et al. 2015). Some individuals utilize terrestrial foods sources during the summer and fall. However, the extent to which this occurs and the significance of terrestrial foods to energy budgets remains the subject of on-going debate and research (e.g., Derocher et al. 1993a, Hobson et al. 2009, Gormezano and Rockwell 2013, 2015, Rode et al. 2015, Tartu et al. 2016). Regardless, it is well documented that many bears especially those in the seasonal ice ecoregion like Baffin Bay (Amstrup et al. 2008) rely on nutrients and energy stored within adipose and other body tissues to meet a significant portion of their maintenance requirements for survival during this period. Consequently, body condition amongst most individuals declines progressively through the summer and fall until access to sea ice increases in the late fall and early winter (Atkinson and Ramsay 1995, Atkinson and Ramsay 1996, Derocher and Stirling 1995, Rode et al. 2011, Obbard et al. 2016). Additionally, adult females rely on body stores to support reproductive activities. Mothers with cubs continue to lactate during the summer and fall if their condition and/ or available food is sufficient (Derocher et al.1993b, Derocher and Stirling, 1996). Pregnant females enter dens in the fall where they rely exclusively on body stores to support gestation and early to mid-lactation over a period of 6-8 months (Atkinson and Ramsay 1995, Derocher and Stirling 1995). Given this dynamic cycle of feeding and fasting, body condition attained during the spring and early summer is expected to exert a significant influence on the survival, reproductive performance and thus status of polar bear subpopulations (Atkinson and Ramsay 1995, Derocher and Stirling 1995, 1996, Molnar et al. 2010, Molnar et al. 2011).

Tracking long-term trends in body condition has been identified as an important component of the monitoring scheme for polar bears across their circumpolar range (Vongraven et al. 2012, Patyk et al. 2015). In the absence of more intensive studies, simple body condition metrics may be useful indices for monitoring subpopulations and detecting responses to changing environmental conditions (Amstrup et al. 2006, Stirling et al. 2008a, Vongraven et al. 2012). Of particular concern, changes in body condition are predicted to be amongst the first subpopulation-level impacts of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Wiig et al. 2008, Stirling and Derocher 2012). Indeed, declines in condition have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016). Although these trends in body condition can signal significant changes, observations of declining condition alone do not necessarily imply a decline in subpopulation status. Studies of some polar bear subpopulations have documented declines in condition in association with increases in abundance that may be the result of density dependence (Stirling et al. 1999, Rode et al. 2011, Peacock et al. 2013).

Information on body condition must therefore be interpreted alongside other measures of subpopulation performance in-order to properly assess status.

A variety of quantitative and qualitative body condition indices have been used on polar bears including body weight estimated from girth (e.g., Derocher and Stirling 1995, Rode et al. 2011), body mass indices standardized for length (e.g., Stirling et al. 1999, Cattet et al. 2002, Obbard et al. 2016), skull width (Rode et al. 2010, 2011), percent body fat determined by isotopic dilution or bioelectrical impedance analysis (Atkinson and Ramsay 1995, 1996, McKinney et al. 2014), percent lipid content of adipose tissue biopsies (Stirling et al. 2008b, McKinney et al. 2014) and a visually assigned fatness index (Amstrup et al. 2006, Stirling et al. 2008a,b). Most of these condition indices require the handling of bears to collect measurements. However, the fatness index (FI) and potentially the lipid content of adipose tissue (Pagano et al. 2014, McKinney et al. 2014) may be obtained without handling thus making them suitable for use in subpopulations monitored by less invasive methods such as aerial survey or genetic mark-recapture.

The Baffin Bay (BB) subpopulation is part of the seasonal ice-free ecoregion as defined by Amstrup et al. (2008) in which sea ice melts almost entirely in the summer and bears are forced ashore for an extended period of time, during which they have no or reduced access to food. Baffin Bay has experienced a long term reduction in sea-ice cover and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015). As a result, bears are spending an increasing amount of time on land. Examining data on the morphometric measurements of BB polar bears (girth, length and skull width) for the period 1977 to 2010, Rode et al. (2011) detected a decline in body condition concurrent with declining sea-ice cover. However, geographically restricted sampling and uncertainty about trends in subpopulation density during the latter years of this study limited the interpretation of these findings and conclusions regarding subpopulation status (York et al. 2016).

Here we summarize information on the body condition of polar bears in BB using a different measure of condition; the fatness index (FI). FI scores were collected during two periods of mark-recapture sampling in Baffin Bay from 1993 to 1997 and 2011 to 2013. During the latter period of sampling bears were surveyed by genetic mark-recapture using biopsy darts. Because biopsy darted bears were not handled our collection of body condition data was limited to visually assigned FI scores only. The FI has been validated as a measure of condition in polar bears, being closely correlated with more quantitative condition indices (Stirling et al. 2008b, McKinney et al. 2014) and other biological factors (e.g., Henricksen et al. 2001, Amstrup et al. 2006). Our study examined body condition using a different index of condition collected over a different (albeit overlapping) temporal and spatial sampling frame to that of Rode et al. (2011). We examined trends in condition in relationship to sea ice. During part of this study, we also collected information on the foraging habits of BB polar bears to assess the range of food sources utilized by bears during the ice-free period. Our results provide supplementary information for interpreting the results of the recent genetic mark-recapture in BB and for understanding the present status of this subpopulation.

7.2. Materials and Methods

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and Bylot Island (66.2°N to 73.8°N) in Nunavut/Canada and parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al.

2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south (Figure 7.1). Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Historically sea ice also remained in Melville Bay, NW Greenland (Born 1995). Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A presumably small number of bears remain on land in northwestern Greenland throughout the ice-retreat period (Born 1995, this study).

Field Sampling

Data for the study were collected during two periods of systematic mark-recapture sampling on Baffin Island in Baffin Bay. Sampling occurred along eastern Baffin and Bylot Islands during the ice-free season from late August to mid-October in 1993-95, 1997 and 2011-2013. Most bears in Baffin Bay move onto land on Baffin Island and Bylot in late summer as the sea ice breaks up and remain on land until freeze-up in the late fall (Taylor et al. 2005). Sampling was spatially extensive across this on-land study area during all years (Figure 7.1) although there were some noted differences in sampling strategy between the two periods (1990s and 2000s) of the study (Chapter 3).

Using helicopters we searched for bears across the study area. During the 1990s, bears were sampled by physical capture on Baffin Island and examination using methods previously described (Taylor et al. 2005). Data on the sex, estimated age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred by remote biopsy darting (Pagano et al. 2014) and subsequent genetic analysis to determine genetic sex and identify individuals. We remotely estimated sex, age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) and reproductive status from the air at a range of 3-10 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B).

During both sampling periods, all encountered bears were assigned a FI score on a scale of 1-5 where 1 and 5 represent the leanest and most obese bears, respectively (Stirling et al. 2008a).

During the 1990s, this score was based on physical examination of captured bears. For bears in the 2000s, FI scores were assigned based on examination from the air at a distance of 3-7 m above ground. Additional information collected for all bears at the time of observation included the identity of the observer, date, and location (coordinates).

Additionally various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that small, albeit unknown, proportion of the BB subpopulation summers in the Qimmusseriarsuaq / Melville Bay area of NW Greenland rather than moving with the retreating sea ice and summering on Baffin and Bylot Islands. For comparative reasons estimates of body condition index from bears biopsy dated in NW Greenland are presented separately from that on Baffin Island. During 1-14 September 2012 and 2013 a total of 20 biopsies (2012: 6, 2013: 14) were sampled on land and along glacier fronts in the Melville Bay area using the same methods described above in Nunavut. During sampling the FI index was scored for each bear by three observers with extensive experience in judging body condition of polar bears both during examination from the air and during subsequent physical handling. The sex of all biopsied individuals was determined genetically post hoc.

Body Condition Scoring

Bears were initially scored according to the standard FI on a scale of 1 to 5 (Stirling et al. 2008b). This scoring system was subsequently simplified to a 3 point scale of poor (FI = 1 or 2), fair (FI = 3) and good (FI = 4 or 5) condition; hereafter termed the Body Condition Score (BCS). Similar modifications of the FI for polar bears have been employed in other studies to facilitate analyses (Stirling et al. 2008a) or have been recommended for use in general monitoring schemes for polar bears (Vongraven et al. 2012). In our case, this refinement was made in part due to the lower frequencies of bears scored as 1 and 5, but also to address concerns about potential bias. The assumption was made that a simplified scale would be subject to less bias resulting from different observers and / or distance from bear at time of scoring. Experienced observers should be able to discriminate a bear in poor, fair or good condition even at distances of up to 7 m.

Bears coming off the ice in summer are thought to be at or near their annual peak in body condition having recently gone through a period of hyperphagia when juvenile seals are relatively abundant and susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). For our analyses, therefore, we calculated the proportion of bears rated as being in good condition (BCS = 3) as a metric of annual body condition within each sex and age class. Repeated observations of the same individual (as identified by physical mark or genotype) within a given year were excluded from the analyses. Observations of the same individual in different years were included. Similar to Stirling et al. (2008a), we assumed that observations of the same individual in different years were statistically independent given the dynamic nature of body condition in polar bears (Watts and Hansen 1987, Atkinson and Ramsay 1995, 1996) and it’s response to annual variation in environmental conditions.

Sea-ice Metric

Based on findings from other studies (Stirling et al. 1999, Durner et al. 2004, 2006, 2009, Regehr et al. 2007, Rode et al. 2010, 2014), we hypothesized that timing of sea-ice break-up over the continental shelf (< 300m depth) of BB may influence the body condition of polar bears during the ice-free period. We therefore calculated the annual sea-ice metric, spring transition date, as day of the year (1-365) when ice area over the continental shelf of BB reached 50% in the spring (Chapter 4). We used *Sea-ice Concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data* (Cavalieri et al., 1996) available from the National Snow and Ice Data Center (NSIDC) in Boulder, Co (See Laidre et al. 2015 Appendix S1 for additional details). While some researchers have used lower ice cover thresholds for studying polar bear relationships to sea ice (e.g., Cherry et al. 2013, Rode et al. 2014, Obbard et al. 2016), Laidre et al. (2015) reported that spring transition dates in Baffin Bay were not sensitive to the choice of threshold because usually the decrease of sea-ice area in the spring and the increase of sea-ice area in the fall proceed relatively quickly. A small change in the threshold results in a small change in the transition dates (Laidre et al. 2015). For sea ice in Baffin Bay, spring transition date is also correlated with other sea-ice metrics including fall transition date (negative correlation) and the interval between spring and fall transitions (Laidre et al. 2015).

We examined annual variation in our body condition metric, trends over time, and relation to spring sea-ice transition date. Similar to Rode et al. (2014), we did not include both time and sea ice in the same analyses since long term trends in our sea-ice metric were well established *a priori* (Laidre et al. 2015, Chapter 4) and these two parameters were closely correlated over the 7 years of sampling (Pearson coefficient = -0.846, P = 0.017). Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

Foraging Observations

Observations of bears feeding or evidence that they had recently fed were collected during the second sampling session, 2011-13, only. In 2011, systematic observations of the feeding activities were not collected. In 2012 and 2013, all bears encountered were examined from the air for evidence of feeding activity. Evidence of feeding included the presence of kills and other obvious food items, fresh oil or blood staining around the mouth, pendulous/distended abdomens (full stomachs), the production of black tar-like feces during pursuit (normally seen in bears that have been eating marine mammals) and the production of feces containing visible berries.

7.3. Results

Body Condition Scores

In total, 2500 polar bears were assigned a BCS during mark recapture sampling. Six bears assigned a BCS were not assigned to a sex-age class. These were excluded from the further analyses. Amongst independent bears, samples sizes were largest for adult males (n = 783), adult females with offspring (423) and lone adult females (225). Our analyses focussed on these three groups of bears for the following reasons: (1) Sample sizes were relatively large and/or (2) the accuracy of classifying bears into sex and age class at the time of aerial observation and BCS assignment was relatively good (Appendix B)[1]. Amongst dependent offspring we examined BCS for cubs-of-the-year (COY) and yearlings both of which can be identified with good accuracy from the air.

During the period 1993-97, all observations were made by a single individual (Table 7.1). During the period 2011-13, observations were made by 3 individuals but a majority (79%) were made by a single individual. Of the 2496 bears in our study, four were observed by two observers in the same year. In each case the condition scores assigned by the observers were the same (adult male in poor condition, adult female in fair condition, adult female in poor condition,

---

[1] Accuracy for adult males, adult females with offspring, lone adult females was 95%, 100% and 74%, respectively. In comparison accuracy for sub-adult males and females was <40%.

subadult female in fair condition). Because the observers collected BCS data on bears in different years and / or different parts of the study area from each other, it was not possible to distinguish observer effects from other factors.

In general, body condition was better amongst adult males and lone adult females than other age classes (Table 7.2). For example, pooling data across years, 25 and 32 % of individuals were classified as being in good condition amongst adult males and lone adult females, respectively. In contrast, amongst adult females with offspring, subadult females and subadult males, the percentage of bears in good condition was 9, 4 and 4% respectively.

There was annual variation in body condition. For example, during the period 1993-97, when all observations were collected by a single individual, the distribution of adult male BCS varied significantly from year-to-year ($\chi^2 = 24.01$, d.f. = 6, $P < 0.001$) with more males in good condition in 1993 (58.8%) compared to other years, in particular 1997 (32.0%). Annual variation in condition was similar, although not significant, for lone adult females ($\chi^2 = 11.51$, d.f. =6, $P = 0.070$) and females with offspring ($\chi^2 = 9.94$, d.f. =6, $P = 0.132$) during this period. For the period, 2011-13, when a majority of observations were made by a single but different observer, condition was also found to vary significantly from year-to-year amongst adult males ($\chi^2 = 24.31$, d.f. =4, $P < 0.001$) but not lone females ($\chi^2 = 4.89$, d.f. =4, $P = 0.300$) or females with offspring ($\chi^2 = 7.71$, d.f. =4, $p < 0.100$). During this recent sampling period, 2011 tended to be a better year for body condition.

Our annual body condition metric was associated spring sea-ice transition date amongst some sex and age classes of bears (Table 7.3). The proportion of adult males assigned a BCS of 3 (good) in a given year was closely associated with the timing of spring sea-ice transition. A higher proportion of adult males were in good condition in years with a later spring transition date (Figure 7.2a). A similar association was evident for adult females with offspring (Figure 7.2b) but not lone adult females (Figure 7.2c). These associations between condition and sea ice were also evident for adult males (Exponential curve; $F_{1,3} = 101.27$, $r^2 = 0.98$, $P = 0.010$) and females with offspring (Exponential curve; $F_{1,3} = 72.12$, $r^2 = 0.97$, $P = 0.014$) when analyses were limited to the period 1993 to 1997; the 4 years when data were collected by a single observer.

For COY, the proportion of bears in good condition was unrelated to the timing of spring sea-ice transition (Table 7.3). In contrast, later spring transition was associated with a higher proportion of yearlings in good condition. Similar to adult males and females with offspring, this association was also evident when analysis was limited to the period 1993-97 when analyses were limited to the period 1993 to 1997; the 4 years when data were collected by a single observer (Linear regression; $F_{1,3} = 51.30$, $r^2 = 0.96$, $P = 0.019$).

Body condition showed a negative trend over time amongst some sex and age classes (Table 7.3). For adult males and adult females with offspring the proportion of bears in good condition during the ice-free period declined from 1993 to 2013. As similar trend, although not significant ($P = 0.065$), was evident for yearlings.

In Melville Bay, NW Greenland in 2012 and 2013 the adult bears were generally in good body condition. The samples collected from this area included 10 adult females, 5 adult males, 3 subadults (1 F, 2 M) and 2 female yearlings. Three adult females and 1 adult male scored FI = 4 and 7 adult females and 4 adult males scored F = 3. Three subadults (1 F, 2 M) and 2 female yearlings all scored F = 3. Hence, although sample size in Greenland was low, BB bears in this area were in good body condition despite an on-land period which is longer than BB polar bears that summer on Baffin Island.

Foraging During the Ice-Free period

One hundred and seven (9.3%) of the 1146 bears observed in 2012 and 2013 showed evidence of feeding. Prevalence of feeding was lowest amongst adult males (8%) and dependent offspring (7%), and highest amongst subadults (13%). Across sex and age classes, 50% of feeding observations were among adult males and subadults (Figure 7.3). The distribution of feeding observations amongst sex and age-classes did not differ significantly from the sex and age-class composition of all bears observed (feeding and not feeding); although there was a tendency for subadult bears to be over-represented amongst those observed feeding ($\chi^2 = 5.607$, d.f. = 4, $P = 0.23$). There was no seasonal trend in prevalence of feeding observations during the sampling period (Figure 7.4).

Bears were observed feeding on a range of food items including seals (species unknown), walrus (*Odobenus rosmarus*), narwhal (*Monodon monoceros*), Arctic char (*Salvelinus alpinus*), Greenland shark (*Somniosus microcephalus*) and berries (Table 7.4). Where food source was known, marine mammals comprised 47% of the observations of bears feeding. At two locations, congregations of bears were encountered along streams where char were observed to be running in large numbers. Fish carcasses found at these sites and the presence of bears standing in close quarters along these watercourses suggested bears were actively fishing. Seventeen seal kills were noted while searching for bears. Most were located along the shores of fjords rather than the main open coastline (Figure 7.5).

In Melville Bay polar bears were observed feeding on seals on patches of floes of fast ice and more or less consolidated bergy bits at glacier fronts. During September in both 2012 and 2013 numerous narwhal, ringed, bearded, harp and hooded seals were observed close to glacier fronts and land in Melville Bay (Born et al. 2012, 2013) indicating that suitable food for BB polar bears is abundant in this area during the open-water season.

7.4. Discussion

Body Condition of Baffin Bay Bears

Our results demonstrate that body condition amongst BB polar bears declined over the period 1993 to 2013. The close association between condition during the ice-free period and spring sea-ice transition date is consistent with the hypothesis that reduced time on the sea ice and presumably declining access to prey during the important spring to early summer feeding period is a primary mechanism driving this decline. However, with our qualitative body condition data and simplistic analysis we have not demonstrated cause and effect. Annual variation in body condition was associated with both time (year) and spring sea-ice transition date. These two parameters are correlated with one another and may also be associated with other parameters that we did not consider. The extent to which declining condition in BB is mediated by ice conditions therefore remains uncertain.

Using body condition metrics different from those used in the present study, Rode et al. (2011) detected a decline in the condition of BB polar bears between 1990 and 2010 concurrent with declining sea-ice cover. Our findings are consistent with this earlier study except that we did not find associations between body condition and sea-ice cover amongst all sex and age classes of bears. This may be due to limitations of the qualitative condition data used in the present analyses and sample size issues. The BCS is a qualitative and thus less precise measure of condition than the quantitative metrics used by Rode et al. (2011) that were derived from morphometric measurements acquired during physical capture and handling of bears.

Additionally, during the latter years of our study (2011-2013) the BCS for each bear was assessed based on examination from a distance rather than capture and physical examination. Inaccuracies in classifying bears by age class and sex from the air, combined with a lack of direct physical handling to assess condition may have introduced more error in assigning BCS. Given these limitations, BCS data are likely a less robust and less sensitive means of detecting changes in body condition over time or in response to ecological parameters (Vongraven et al. 2012, McKinney et al. 2014). Direct, quantitative measurement of body condition by morphometry or adipose tissue lipid content (McKinney et al. 2014) remains the most reliable and precise means of monitoring condition.

Nevertheless, our study extends the findings of Rode et al. (2011) in three notable ways. First, we find that trends in body condition and the association with sea-ice conditions have continued beyond 2010. Second, Rode et al. (2011) suggest that important trends in body condition that can affect reproduction might not be detectable from on the ground observations without capture and physical measurement of bears. Our findings suggest that long-term trends in body condition can be detected without handling of bears albeit with less sensitivity. As found in other studies (Amstrup et al. 2006, Stirling et al. 2008a), visually assigned body condition scores are a useful means of monitoring body condition and polar bear responses to environmental conditions. In the absence of physical capture programs and / or more intensive monitoring schemes, the collection of condition scores provides a simple and low cost means to track general trends in BB and likely other polar bear subpopulations. Finally, York et al. (2016) maintain that the evidence linking reduced body condition to sea-ice decline in Baffin Bay (Rode et al. 2011) is ambiguous because the body condition data used in the analyses were collected in varying parts of the subpopulation area over the period of study rather than range wide throughout. For example, during 1990s data on condition during the ice-free period were collected from bears across the subpopulation's seasonal range on eastern Baffin Island. In contrast, sampling in the 2000s was restricted to a relatively small southern portion of the range near the boundary with Davis Strait. Consequently, condition data for the 2000s may not have been representative of the subpopulation as a whole. However, sampling in our study was range wide during both time periods (1993-1997 and 2011-2013) as illustrated in Figure 7.1 suggesting that the findings of Rode et al. (2012) were representative of BB.

Several sources of bias were possible in our study. BCS data were collected by several observers in different years and different parts of the study area. Notably a single observer collected all data during the 1990s. Several different observers collected data during the 2000s and in different parts of the study area from one another. To reduce potential observer bias in assigning qualitative condition scores, we employed a simplified body condition scoring system that required observers to discriminate between bears in poor, fair and good condition. All observers in the study were experienced polar bear biologists who had previously handled hundreds or thousands of bears in varying condition and should have been capable of easily discriminating bears in good condition. However, since individual bears were not scored by more than one observer, teasing out observer effects is challenging because differences in scored condition may reflect real temporal or spatial differences in the bears sampled. Never-the-less, several lines of evidence suggest observer bias was likely not a significant factor in our study. First, a majority of observations were made by a single observer within each time period (1990s and 2000s). Differences between these two observers in scoring body condition are unlikely to explain the linear and non-linear trends in condition we observed or the close association between condition and sea ice. Additionally, the statistical significance of these associations was maintained when analyses were restricted to a single time period when all observations were collected by a single observer. Finally, as noted by Stirling et al. (2008a) although the FI from which our condition metric was derived is a qualitative index and thus subjective, it has been found to be "*repeatable between individual biologists when blind comparisons are done in the field over both short and long time periods*." In other studies, FI data collected by multiple observers have been found to correlate closely with quantitative indices of condition (e.g., Stirling et al. 2008b, McKinney et al. 2014). Therefore, while we cannot exclude the possibility of observer bias in our study, we suggest that this potential source of bias is unlikely to account for our results.

Body condition scores in the 1990s were collected from bears captured and physically examined. In contrast scores in the 2000s were from bears observed from the air without handling. The effect of close-up versus distance examination on the scoring of condition is unknown. McKinney et al. (2014) found that remotely assigned FI ratings did not correlate with the % lipid content of adipose tissue; another measure of condition. However, their sample sizes were small and limited to comparisons of bears of FI 3 and 4 only whereas bears in our study had FI ratings ranging from 1 to 5. Remotely scoring FI may be a less robust (precise) index of condition but is not necessarily inherently biased relative to physical examination. For many of the same reasons discussed previously concerning observer effects, we suggest that this potential source of bias is unlikely to account for our results. The use of a simplified scoring system (poor, fair, good) in our study should have helped to reduce errors in scoring for bears observed from the air. Trends in condition over time and the close association with sea-ice metrics cannot be explained by differences in examination distance.

Another source of error in our study associated with differences in sampling between the

1990s (physical capture) and 2000s (aerial observation) was in the classification of bears by sex and age-class based on aerial observation rather than handling. Classifying bears from the air is without doubt less accurate than physical examination. However, aerial classification is accurate in most instances (Appendix B), especially amongst adult males and adult females with offspring; the two classes exhibiting the strongest trends in condition in our study. Additionally, despite being less precise we have no evidence to suggest that aerial classification results in a bias in age and sex classifications amongst a group of bears. This source of measurement error thus seems unlikely to account for our results.

An assumption of our study was that bears sampled within our study area were representative of the BB subpopulation. Although sampling during both the 1990s and 2000s was extensive across the seasonal range of BB bears, the proportion of bears in the subpopulation exposed to sampling may have differed between these two periods (Chapter 3). In the 1990s, a high proportion of collared bears did not come ashore on Baffin Island during the sampling windows but instead remained on remnant offshore sea ice where they could not be sampled. This observation suggests that a significant portion of the subpopulation was not sampled in the 1990s. Whether this biased our estimates of body condition is unknown. However, we note that bears remaining out on the ice were likely still able to hunt seals to some extent and may therefore have been in better condition than those coming ashore. Consequently, any bias in our sampling would have resulted in underestimation of condition in the 1990s or in years when spring transition occurred later. This would therefore not account for the trends in condition we observed over time or in association with date of spring sea-ice transition.

Relative to the 2000s, sampling in the 1990s was also more concentrated near the coast with less inland sampling (Chapter 3). Similar to other seasonally ice-free subpopulations (e.g., Derocher and Stirling 1990) polar bears in Baffin Bay are known to exhibit a degree of spatial segregation by sex, age class and reproductive status with respect to the use of terrestrial habitat during the ice-free period (Ferguson et al. 1997, Chapter 4). Adult females with cubs tend to select fjords and avoid offshore islands and coastal regions where densities of adult males are higher. Pregnant bears select inland and upland denning habitats. While limited inland sampling in the 1990s may have resulted in under sampling of certain sex, age and reproductive classes we are unaware of any evidence to suggest that this would also have biased body condition data. However, to explore the potential impact of this difference in sampling between epochs on the body condition data, we estimated and delineated the minimum extent of the sampling frame from the 1990s using capture locations in a GIS. We subset the 2000s data using this layer to create a dataset collected over the same sampling area in both time periods and repeated our analyses of trends in body condition. The results were essentially the same (Appendix C), suggesting that this sampling difference between epochs did not influence our findings.

Bears in BB lose condition through the summer and fall while on land in BB (Rode et al. 2011). Differences amongst years in the timing of sampling could therefore have affected our annual body condition metric. Sampling occurred between late August and late October but varied somewhat in timing from year-to-year. We did not consider timing of sampling in our analysis. During preliminary exploration of the data we noted that the 3 years where the median date (Julian day) of sampling was earliest were the best (1993) and two worst (2012 and 2013) years for body condition amongst both adult males and adult females with offspring, as measured by our metric. Median date of sampling was also not associated with our condition metric for any of the sex and age classes of bears. Similarly, looking at the number of days between spring transition date and the date of sampling for each bear as an index of timing of sampling relative to seas-ice breakup we found similar results suggesting that timing of sampling did not account for the annual variation in body condition that we were observing at a broad scale with our somewhat crude measure of condition. However, we acknowledge that a more sophisticated analysis such as a polynomial logistic regression could incorporate sampling date as a covariate.

Declining body condition and reproduction are amongst the first subpopulation level effects predicted occur in polar bears as a result of climate change (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Our evidence of a decline in condition in BB from 1993 to 2013, along with similar findings from a previous study (Rode et al. 2011), is accompanied by evidence of a concurrent decline in reproduction in this subpopulation. These findings are also consistent with available traditional knowledge suggesting that body condition of polar bears in BB was poorer in the early 2000s relative to the 1990s (Dowsley and Wenzel 2008, Born et al. 2011). These changes may signal a reduction in the carrying capacity of BB.

The bears that were biopsied in Melville Bay in September 2012 and 2013 generally appeared to be in good body condition. However, during an interview survey among experienced polar bear hunters in NW Greenland ca. 24% of the 72 interviewees noted that polar bears had generally become thinner (Born et al. 2011).

Similar to recent observations in the Davis Strait subpopulation (Rode et al. 2011, Peacock et al. 2013) we cannot rule-out possible density effects on body condition and reproduction resulting from a declining sea-ice platform. Regardless of whether density effects are at play, if the observed association between sea ice and body condition is real and the well documented trend in sea ice continues it is reasonable to expect that this subpopulation will experience an on-going decline in condition as a result of declining habitat (Laidre et al. 2015). This in turn is predicted to lead to demographic consequences including reduced adult survival (Molnar et al. 2010, 2011).

Measures of body condition have been identified as one of the most important metrics needed to evaluate polar bear health (Patyk et al. 2015). Similar to previous studies (e.g., Amstrup et al. 2006, Stirling et al. 2008a,b, McKinney et al. 2014) we have demonstrated the utility of a simple qualitative metric for monitoring trends in body condition in polar bear subpopulations where more detailed quantitative measures of condition may not be available. In circumstances where demographic studies are conducted periodically rather than on an on-going basis or where the selected methods of survey do not involved capture and handling, collection of visually assigned body condition scores from harvested bears or from opportunistic observations of free ranging animals offers a useful means of surveillance. Such surveillance may be carried out by government agencies but there is also potential for implementation as part of a community-based ecosystem monitoring scheme. Changes in condition detected through this method of monitoring may serve as a trigger to initiate more intensive studies.

We acknowledge that results from analyses of FI scores, including those of the present study, must be interpreted cautiously given the many potential biases associated with this type of data. While many of these potential biases can be mitigated through study design and analyses, further work is needed to examine the robustness of these data before this method of monitoring is implemented more widely in government or community-based monitoring schemes.

Foraging During the Ice-Free period

Polar bears have been found to opportunistically exploit a wide variety of food sources while on land during the summer and fall (e.g., Derocher et al. 1993a, Brook and Richardson

2002, Dyck and Romberg 2007, Gormenzano and Rockwell 2013, Iverson et al. 2014, Rogers et al. 2015, Tartu et al. 2016). Most of these foods are terrestrial in origin including berries, bird eggs, birds, small mammals and occasionally large mammal prey such as caribou or reindeer. Although bears have been observed catching fish (Dyck and Romberg 2007) and seals in open water (Furnell and Oolooyuk 1980), and scavenging the bone piles of human hunted bowhead whales (Rogers et al. 2015), the available evidence suggests marine mammals generally represent a small portion of the diet during this period of minimum sea ice. In Western Hudson Bay, for example, Gormenzo et al. (2013) found evidence of marine mammal remains in less than 5% of polar bear fecal samples collected during the on-land period. In contrast, terrestrial foods such as vegetation and eggs made up the majority of material in these samples. Similar dietary habits have been documented in Southern Hudson Bay (Russell 1975)[2].

In Baffin Bay we found evidence of recent foraging in approximately 9% of the bears observed during the on-land period. The type of food consumed was known for approximately half of these individuals. Marine mammals, in particular seals and walrus made up almost half of the identifiable food sources. While these are opportunistic observations and therefore subject to numerous potential biases, these findings suggest that bears in Baffin Bay may make greater use of marine mammals during the ice-free period than bears in some other subpopulations. This may be the result of differences in habitat and / or the availability of marine mammal prey. In contrast to the lowlands of the Hudson and James Bays, the east coast of Baffin Island is characterized by rugged coastline with high mountains, long, deep fjords and glaciers some of which run directly into the marine environment. Of the seal kills documented during our study most were located along the shores of fjords rather than the main open coastline (Figure 7.5). As suggested by Derocher et al. (2004) fjords may offer preferred seal hunting habitat for polar

---

[2] Russell (1975) found seal remains in 9% of polar bear scats collected on-land during the summer.

bears for several reasons. Stable sea ice at tide water glaciers provides prime breeding habitat for ringed seals (Lydersen et al. 2014). Additionally, remnant sea ice that persists longer into the summer, the shedding of ice from glaciers, the early formation of new sea ice around freshwater outflows and the availability of Arctic char and other food sources near the mouths of rivers may make fjords good habitat for seals. These same features may also make fjords good polar bear habitat. Stable and persistent ice provides a platform to hunt from and the steep sides of fjords give polar bears easy access to the deeper waters in which seals may be swimming during the open water period.

In Melville Bay polar bears were observed in September near or at glacier fronts where numerous ringed seals were also observed (Born et al. 2012, 2013). Satellite telemetry has shown that some bears remain in this habitat throughout the year for several years (Chapter 2). Hence, clearly some bears are able to sustain year round on prey (likely mainly ringed seals) taken in this type of habitat.

We observed a number of bears that appeared to be feeding on anadromous Arctic char during their seasonal runs into creeks and lakes. Observations of polar bears feeding on Arctic char have been previously reported (Dyck and Romberg 2007, Dyck and Kebreab 2009) and traditional knowledge of certain Arctic char runs that are visited annually by polar bears is present amongst residents of Nunavut (S. Atkinson pers. comm.). In Baffin Bay, the significance of Arctic char to polar bear nutritional budgets is unknown. However, we note there are numerous char runs and the available biomass is potentially high. While this food source is only available during a short seasonal window in late summer, the timing and location of char runs is highly predictable. This makes it a reliable source of food for some bears (at least locally) during the ice-free period when other foods are scarce. In some grizzly bear populations, access to anadromous fish has been shown to directly affect body mass, litter size and population density (Hildebrand et al. 1999, 2004). Whether Arctic char hold similar significance for polar bears is unknown but is worthy of further investigation. Based on energetics modelling, Dyck and Kebreab (2009) speculated that polar bears with access to char could in theory maintain or gain body weight during the ice-free period. In contrast, Rode et al. (2010b) suggested that the use of char by polar bears was limited by the availability of suitable water bodies (creeks and rivers) in which bears could capture anadromous fish with an energetic efficiency high enough to permit maintenance or gains in weight. To date, however, there have been no direct empirical studies of the significance of Arctic char in the diets and energetics of polar bears. Arctic char have not been included in prey models for quantitative free fatty acid signature analysis (QFFASA) studies of polar bear diet (e.g., Thiemann et al. 2008, 2009, Galicia et al. 2015). Consequently, this prey's signature would not have been detected in dietary studies conducted to date. We suggest that QFFASA models of polar bear diets should be calibrated to include the signatures of Arctic char sampled from the same regions as the polar bears being studied.

Sea-ice conditions are changing in Baffin Bay (Laidre et al. 2015, Chapter 4). Studies suggest this is affecting the movements and distribution (Chapter 2), habitat use (Chapter 4), body condition (Rode et al. 2011, this study) and reproductive performance (Chapter 6) of polar bears in the region. Declining condition and reproduction is presumably mediated by reduced per capita food intake but precisely how availability of food for polar bears is changing is unknown in part due to lack of knowledge about trends in marine mammal populations (Laidre et al. 2015). Changes in prey abundance and vulnerability to predation are both potential mechanisms. Changes in prey diversity are also possible. One such change may be increased access to sub-Arctic seals such as harp seals (*Pagophilus groenlandicus*). Baffin Bay is part of the summer range of the western north Atlantic harp seal population. The near 2.5 fold increase in this seal population over the last 30 decades is one of the mechanisms postulated to have supported an increase in polar bear abundance in the neighbouring Davis Strait (DS) subpopulation (Peacock et al. 2013). Similarly, McKinney et al. (2013) attributed improving body condition (expressed as adipose tissue lipid content) amongst polar bears in East Greenland (EG) to increased access to sub-Arctic seals including harp seals. Unlike DS and EG, however, bears in BB do not have access to harp seal whelping areas and are therefore not able to access this prey species during its most vulnerable season. For bears in BB, hunting of harp seals is limited to late spring through to fall when predation success rates amongst polar bears hunting in low ice cover or open water are likely relatively low. Polar bears in Svalbard are known to prey on harp seals in the summer (Derocher et al. 2002). Bears in BB may have similar summer foraging opportunities. Indeed, using QFFASA, Galicia et al. (2015) found that adult male polar bears from BB had a higher proportion of harp seal in their diet relative to bears from other subpopulations. This suggests that polar bears in BB have been able to benefit to some extent from the availability of this species. However, the observation that body condition amongst BB bears has declined over the last 3 decades suggests that any shift in prey availability associated with harp seal population expansion has not offset the effects of declining sea-ice conditions on access to other food sources.

7.5. Literature Cited

Amstrup, S. C., G. M. Durner, I. Stirling, N. J. Lunn, and F. Messier, F. 2000. Movements and distribution of polar bears in the Beaufort Sea. Canadian Journal of Zoology 78:948–966.

Amstrup, S. C., I. Stirling, T. S. Smith, C. Perham, and G. W. Thiemann. 2006. Recent observations of intraspecific predation and cannibalism among polar bears in the southern

Beaufort Sea. Polar Biology 29:997–1002.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Atkinson, S. N., and M. A. Ramsay. 1995. The effects of prolonged fasting on the body composition and reproductive success of female polar bears (*Ursus maritimus*). Functional Ecology 9:559–567.

Atkinson, S. N., R. A. Nelson, and M. A. Ramsay. 1996. Changes in body composition of fasting polar bears (*Ursus maritimus*): the effect of relative fatness on protein conservation. Physiological Zoology 69:304–316.

Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015. Demographic composition and behavior of polar bears summering on shore in Alaska: U.S. Geological Survey Administrative Report, 26 pp.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears: Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Born, E.W., K. Laidre, and Ø. Wiig. 2012. Genetic mark-recapture assessment of the Baffin Bay and Kane Basin polar bear subpopulations: Interim report on field Work 2012. Unpublished report from the Greenland Institute of Natural Resources (Nuuk) to the Greenland Government´s Department of Fisheries, Hunting and Agriculture (Nuuk) and The Danish Environmental Protection Agency (Ministry of the Environment, Copenhagen), 7 November 2012. Available from Greenland Institute of Natural Resources, Nuuk, www.natur.gl

Born, E.W., K. Laidre, and Ø. Wiig. 2013. Genetic mark-recapture assessment of the Baffin Bay and Kane Basin polar bear subpopulations: Interim report on field work 2013.

Unpublished report from the Greenland Institute of Natural Resources (Nuuk) to the Greenland Government´s Department of Fisheries, Hunting and Agriculture (Nuuk) and The Danish Environmental Protection Agency (Ministry of the Environment, Copenhagen), 10 December 2013. Available from Greenland Institute of Natural Resources, Nuuk, www.natur.gl

Brook, R. K., and E. S. Richardson. 2002. Observations of polar bear predatory behavior toward caribou. Arctic 55:193–196.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Cattet, M. R. L., N. A. Caulkett, M. E. Obbard, and G. B. Stenhouse. 2002. A body condition index for ursids. Canadian Journal of Zoology 80:1156–1161.

Cavalieri, D. J., C. L. Parkinson, P. Gloersen, and H. J. Zwally. 1996, updated yearly. Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS passive microwave data, Version 1. NASA National Snow and Ice Data Center Distributed Active Archive Center, Boulder, Colorado, USA, doi: http://dx.doi.org/10.5067/8GQ8LZQVL0VL.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Derocher, A. E., D. Andriashek, and I. Stirling. 1993a. Terrestrial foraging by polar bears during the ice-free period in western Hudson Bay. Arctic 46:251–254.

Derocher, A. E., D. Andriashek, and J. P. Y. Arnould. 1993b. Aspects of milk composition and lactation in polar bears. Canadian Journal Zoology 71: 561–567.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1990a. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395–1403.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Derocher, A. E., and I. Stirling. 1996. Aspects of survival in juvenile polar bears. Canadian Journal of Zoology 74: 1246–1252.

Derocher, A.E., Ø. Wiig, and M. Andersen. 2002. Diet composition of polar bears in Svalbard and the western Barents Sea. Marine Mammal Science 25:448–452.

Dowsley, M., and G. W. Wenzel. 2008. ‘‘The time of the most polar bears’’: A co-management conflict in Nunavut. Arctic 61:177–189.

Durner, G. M., S. C. Amstrup, R. Nielson, and T. Macdonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea. Pages 107–120 *in* S. Huzurbazar, editor. Resource selection methods and applications. Omnipress, Madison, Wisconsin, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Dyck, M. G., and E. Kebreab. 2009. Estimating the energetic contribution of polar bear (*Ursus maritimus*) summer diets to the total energy budget. Journal of Mammalogy 90:585–593.

Dyck, M. G., and S. Romberg. 2007. Observations of a wild polar bear (*Ursus maritimus*) successfully fishing Arctic Charr (*Salvelinus alpinus*) and Fourhorn Sculpin (*Myoxocephalus quadricornis*). Polar Biology 30:1625–1628.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use of polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Fowler, C. W. 1990. Density dependence in northern fur seals (*Callorhinus ursinus*). Marine Mammal Science 6:171–196.

Furnell, D.J., and D. Oolooyuk. 1980. Polar bear predation on ringed seals in ice-free water. Canadian Field-Naturalist 94:88–89.

Galicia, M. P., G. W. Thiemann, M. G. Dyck, and S. H. Ferguson. 2015. Characterization of polar bear (*Ursus maritimus*) diets in the Canadian High Arctic. Polar Biology 38:1983–1992.

Gormezano, L. J., and R. F. Rockwell. 2013. Dietary composition and spatial patterns of polar bear foraging on land in western Hudson Bay. BioMed Central Ecology 13:51.

Gormezano, L.J., and R. F. Rockwell. 2015. The energetic value of land-based foods in Western Hudson Bay and their potential to alleviate energy deficits of starving adult male polar bears. PLoS ONE 10:e0128520.doi:10.1371/journal.pone.0128520.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Marine Mammal Science 7:123–1 35.

Henriksen, E. O., Ø. Wiig, J. U. Skaare, G. W. Gabrielsen, and A. E. Derocher. 2001. Monitoring PCBs in polar bears: lessons learned from Svalbard. Journal of Environmental Monitoring 3:493–498.

Hilderbrand, G. V., S. D. Farley, C. C. Schwartz, and C. T. Robbins. 2004. Importance of salmon to wildlife: Implications for integrated management. Ursus 15:1–9.

Hilderbrand, G. V, C. T. Robbins, M. E. Jacoby, T. A. Hanley, S. M. Arthur and C. Servheen. 1999. The importance of meat, particularly salmon, to body size, population productivity, and conservation of North American brown bears. Canadian Journal of Zoology 77:132–138.

Hobson, K. A., I. Stirling, and D. S. Andriashek. 2009. Isotopic homogeneity of breath CO2 from fasting and berry-eating polar bears: Implications for tracing reliance on terrestrial foods in a changing Arctic. Canadian Journal of Zoology 85:193–201.

Iverson, S. A., H. G. Gilchrist, P. A. Smith, A. J. Gaston, and M. R. Forbes. 2014. Longer ice-free seasons increase the risk of nest depredation by polar bears for colonial breeding birds in the Canadian Arctic. Proceedings of the Royal Society B 281:20133128.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lydersen, C., P. Assmy, S. Falk-Petersen, J. Kohler, K. M. Kovacs, M. Reigstad, H. Steen, H. Strøm, A. Sundfjord, Ø. Varpe, W. Walczowski, J. M. Weslawski, and M. Zajaczkowski.

2014. The importance of tidewater glaciers for marine mammals and seabirds in Svalbard, Norway. Journal of Marine Systems 129:452– 471.

McKinney, M. A., S. J. Iverson, A. T. Fisk, C. Sonne, F. F. Riget, R. J. Letcher, M. T. Arts, E. W. Born, A. Rosing-Asvid, and R. Dietz. 2013. Global change effects on the long-term feeding ecology and contaminant exposures of East Greenland polar bears. Global Change Biology 19:2360–2372.

McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

Molnár, P. K., A. E. Derocher, G. W. Thiemann, and M. A. Lewis. 2010. Predicting survival, reproduction and abundance of polar bears under climate change. Biological Conservation 143:1612–1622.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.

Ramsay, M. A., R. A. Nelson, and I. Stirling. 1991. Seasonal changes in the ratio of serum urea to creatinine in feeding and fasting polar bears. Canadian Journal of Zoology 69:298–302.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. Peacock, M. K. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø. Wiig. 2011. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Rode, K.D, J. Reist, E. Peacock, and I. Stirling. 2010b. Comments in response to “Estimating the energetic contribution of polar bear (*Ursus maritimus*) summer diets to the total energy budget”. Journal of Mammalogy 91:1517–1523.

Rode, K. D., R. R. Wilson, E. V. Regehr, M. St Martin, D. C. Douglas, and J. Olson. 2015. Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.

Rogers, M. C., E. Peacock, K. Simac, M. B. O’Dell, and J. M. Welker. 2015. Diet of female polar bears in the southern Beaufort Sea of Alaska: evidence for an emerging alternative foraging strategy in response to environmental change. Polar Biology 38:1035–1047.

Russell, R. H. 1975. The food habits of polar bears of James Bay and southwest Hudson Bay in summer and autumn. Arctic 28:117–129.

Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., E. Richardson, G. W. Thiemann, and A. E. Derocher. 2008a. Unusual predation attempts of polar bears on ringed seals in the southern Beaufort Sea: Possible significance of changing spring ice conditions. Arctic 61:14–22.

Stirling, I., G. W. Thiemann, and E. Richardson. 2008b. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568–574.

Tartu, S., S. Bourgeon, J. Aars, M. Andersen, D. Ehrich, G. W. Thiemann, J. Welker, and H. Routti. 2016. Geographical area and life history traits influence diet in an arctic marine predator. PLoS ONE 11:e0155980. doi:10.1371/journal.pone.0155980.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Thiemann, G. W., S. J. Iverson, and I. Stirling. 2008. Polar bear diets and arctic marine food webs: Insights from fatty acid analysis. Ecological Monographs 78:591–613.

Thiemann, G. W., S. J. Iverson, and I. Stirling. 2009. Using fatty acids to study marine mammal foraging: The evidence from an extensive and growing literature. Marine Mammal Science 25:243–249.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Watts, P. D., and S. E. Hansen. 1987. Cyclic starvation as a reproductive strategy in the polar bear. Symposia of the Zoological Society of London 57:305–318.

Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. Science Progress 91:151–173.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution doi:10.1002/ece3.2030.

Zedrosser, A., B. Dahle, and J. E. Swenson. 2006. Population density and food conditions determine adult female body size in brown bears. Journal of Mammalogy 87:510–518.

Table 7.1. Frequency of observations for body condition scores of polar bears in Baffin Bay. Proportion of within-year observations in parentheses.

	Observer				
Year	**1**	**2**	**3**	**4**	**Total**
1993	-	-	-	149 (1.0)	149
1994	-	-	-	220 (1.0)	220
1995	-	-	-	243 (1.0)	243
1997	-	-	-	285 (1.0)	285
2011	31 (0.06)	415 (0.87)	36 (0.07)	-	482
2012	-	529 (0.79)	142 (0.21)	-	671
2013	-	316 (0.70)	134 (0.30)	-	450
Total	31	1260	312	897	2500

Table 7.2. Frequency of body condition scores (BCS) assigned to polar bears on Baffin Island. BCS scores were derived from Fatness Index (FI) scores (1-5) assigned to polar bears during field observations (following Stirling et al. 2008b). FI scores of 1-2, 3 and 4-5 were assigned BCS of poor, fair and good respectively.

			BCS			
Sex-Age Class			**Poor**	**Fair**	**Good**	**Total**
Adult Male	Year	1993	7	13	30	51
		1994	14	30	27	71
		1995	21	28	31	80
		1997	13	72	40	125
		2011	16	88	35	139
		2012	52	123	21	196
		2013	23	86	12	121
	Total		133	440	185	783
Adult Female (Lone)	Year	1993	2	2	6	10
		1994	3	6	3	12
		1995	0	4	2	6
		1997	2	20	5	27
		2011	3	26	14	43
		2012	8	44	19	72
		2013	5	26	24	55
	Total		21	128	61	225
Adult Female (w/offspring)	Year	1993	4	13	8	25
		1994	15	19	6	40
		1995	15	22	5	42
		1997	11	24	3	38
		2011	7	75	9	91
		2012	18	91	3	112
		2013	8	63	4	75
	Total		76	307	38	423
Subadult Female	Year	1993	3	7	2	12
		1994	6	6	0	12
		1995	4	14	0	18
		1997	4	10	1	15
		2011	3	31	3	37
		2012	13	62	0	75

		2013	4	35	3	42
	Total		36	165	9	211
Subadult Male	Year	1993	1	7	1	9
		1994	4	11	1	16
		1995	14	13	2	27
		1997	8	9	0	17
		2011	4	19	1	24
		2012	8	45	1	54
		2013	8	27	1	36
	Total		40	131	7	183
COY	Year	1993	2	22	2	26
		1994	16	28	0	44
		1995	15	15	4	34
		1997	22	19	0	41
		2011	4	80	12	96
		2012	8	85	1	94
		2013	5	64	8	77
	Total		67	313	25	412
Yearling	Year	1993	1	8	6	15
		1994	3	16	6	25
		1995	7	18	5	30
		1997	3	18	1	22
		2011	0	43	5	48
		2012	5	51	1	57
		2013	5	35	2	42
	Total		24	190	25	239
2-Year-Olds	Year	1993	0	1	0	1
		1995	2	3	1	6
		2011	2	0	0	2
		2012	0	8	1	9
	Total		4	12	1	18
Total	Year	1993	18	73	51	149
		1994	59	116	37	220
		1995	71	117	45	243
		1997	60	172	48	285
		2011	38	362	76	480
		2012	100	510	41	669
		2013	55	336	53	448
	Total[1]		401	1686	351	2494

[1] Excludes 6 bears of unrecorded sex-age class

Table 7.3. Regression results for an annual body condition metric for polar bears on Baffin Island. The metric, proportion of bears in good condition, was derived from observed frequencies of Fatness Index (FI) scores ranging from 1 to 5 (Stirling et al. 2008). Bears of FI 4 or 5 were in good condition. Spring ice transition was the decimal day (1-365) when ice cover over the continental shelf of BB reached 50%. Regressions were performed in the Curve Estimation procedure of SPSS (Version 24.0).

Sex-Age Class	**Dependent Variable**	F_6	r^2	P	***Curve Type***
Adult Male	Spring Ice Transition	102.99	0.98	≤ **0.001**	Quadratic (2nd order)
Adult Male	Year	18.50	0.79	**0.008**	Linear
Adult Female (alone)	Spring Ice Transition	0.65	0.12	0.456	Linear
Adult Female (alone)	Year	0.03	0.01	0.863	Linear
Adult Female (with offspring)	Spring Ice Transition	53.29	0.91	**0.001**	Exponential
Adult Female (with offspring)	Year	7.31	0.59	**0.043**	Exponential
Yearling	Spring Ice Transition	21.57	0.81	**0.006**	Exponential
Yearling	Year	5.526	0.53	0.065	Exponential
COY	Spring Ice Transition	0.10	0.02	0.760	Linear
COY	Year	0.334	0.06	0.587	Linear

Table 7.4. Food sources used by bears on Baffin Island during Aug to Oct, 2012 and 2013.

Food Source	**Number of Bears Observed**
Berries	10
Arctic Char	14
Walrus	10
Seal	11
Narwhal	1
Greenland Shark	1
Unknown	60
Total	107

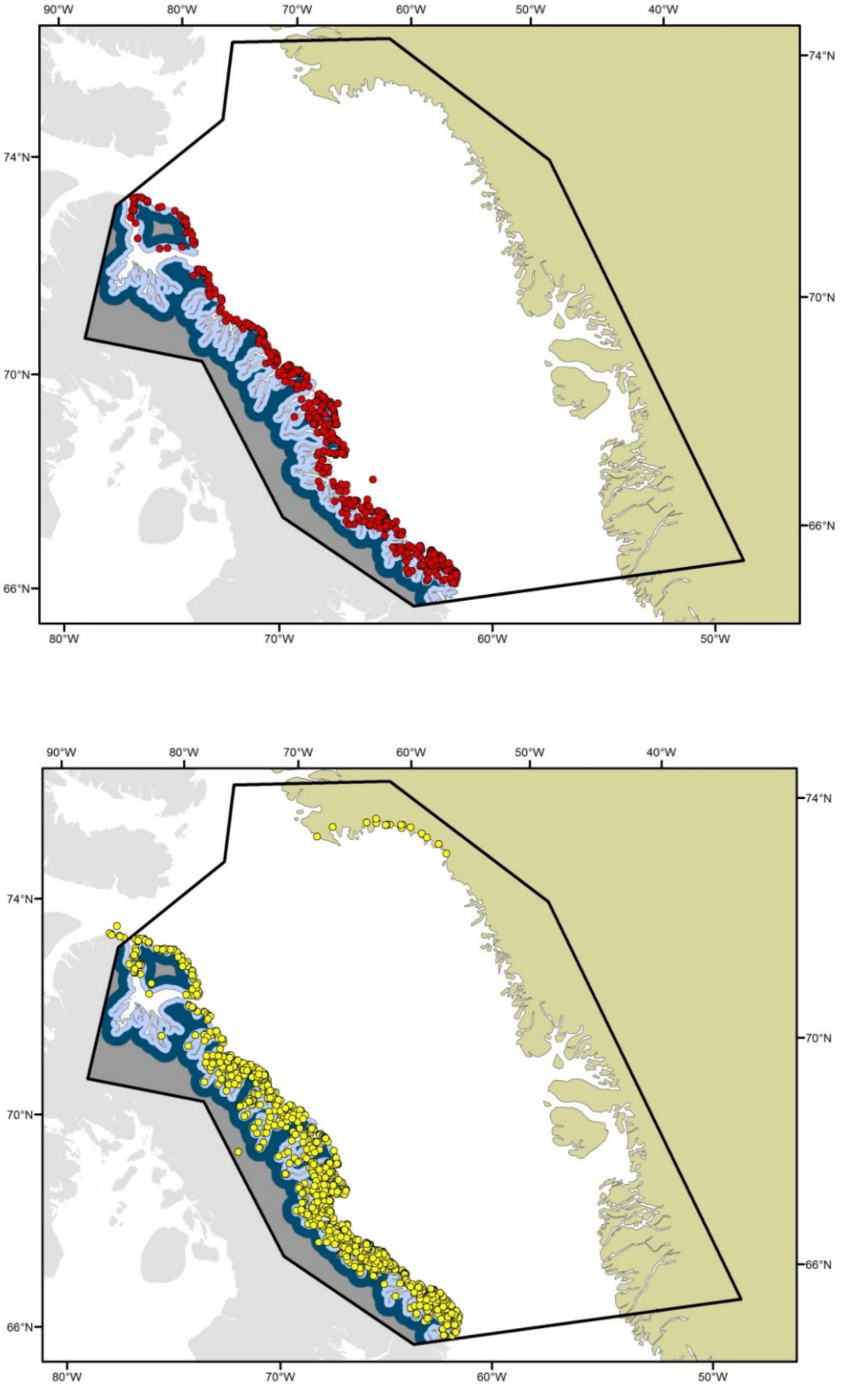
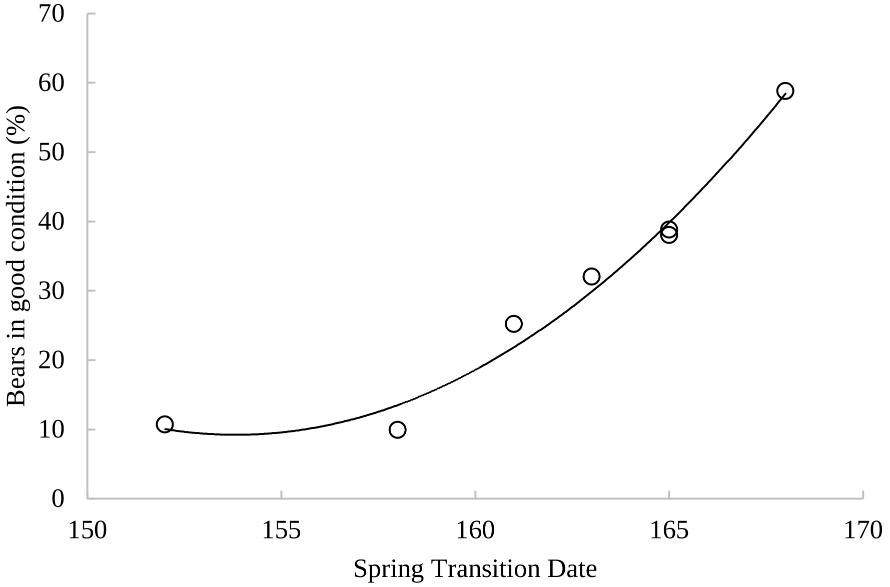


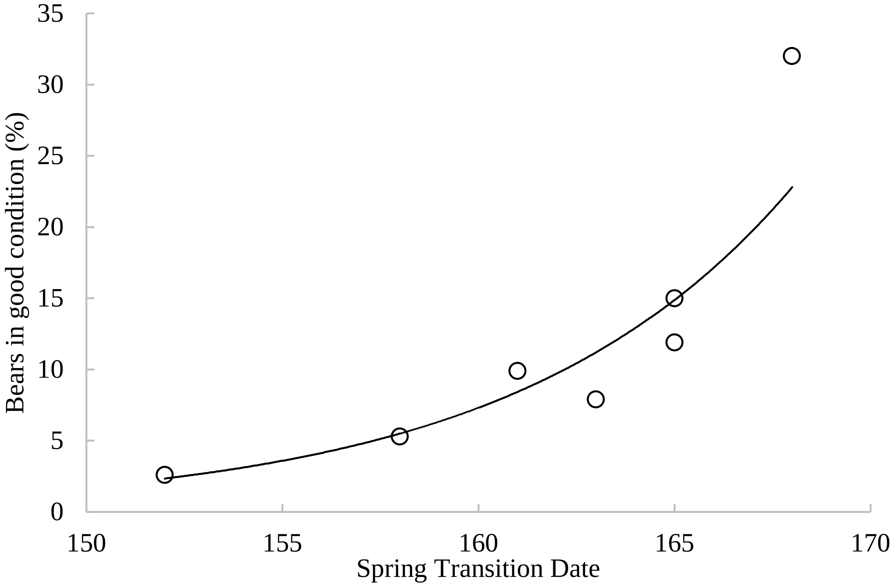
Figure 7.1. Spatial distribution of live captures recorded during sampling in the western parts of the range of the Baffin Bay polar bear subpopulation, 1993 – 1995, 1997 (top), and 2011 – 2013

(bottom). Red, 1990s. Yellow, 2010s. During both periods polar bears were also live captured in the eastern parts (i.e., the Melville Bay area) of the subpopulation´s range (data not shown).

(a)


70
60
50
40
30
20
10
0
Bears in good condition (%)
150
155
160
165
170
Spring Transition Date

(b)


35
30
25
20
15
10
5
0
Bears in good condition (%)
150
155
160
165
170
Spring Transition Date

(c)

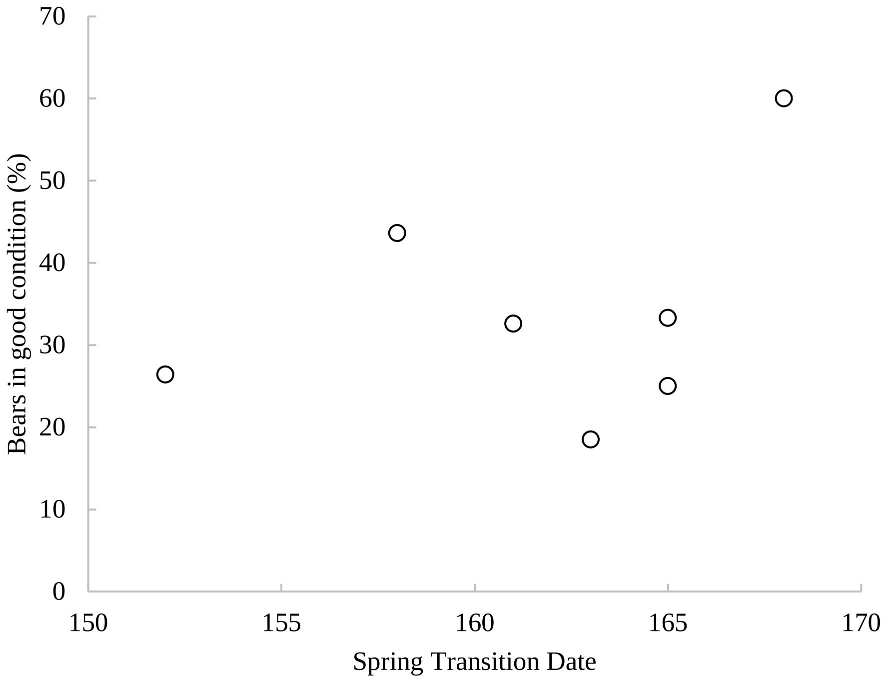


Figure 7.2. The association between the percent of bears in good body condition in western Baffin Bay and the timing of spring sea-ice transition date for (a) adult males, (b) adult females with dependent offspring and (c) lone adult females.

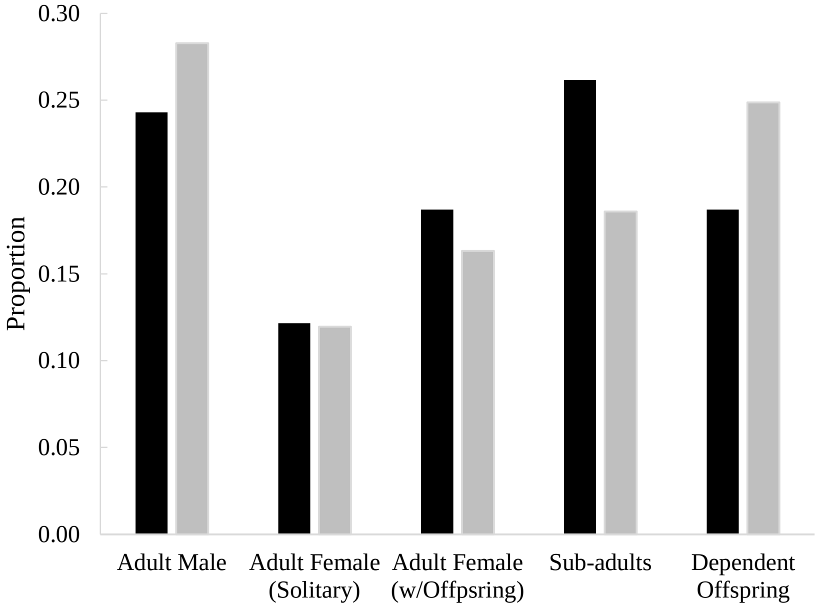


Figure 7.3. The sex and age class distribution of polar bears showing evidence of recent feeding (black) as compared to all the bears observed (grey) on Baffin Island in Baffin Bay during August-October, 2012 and 2013.

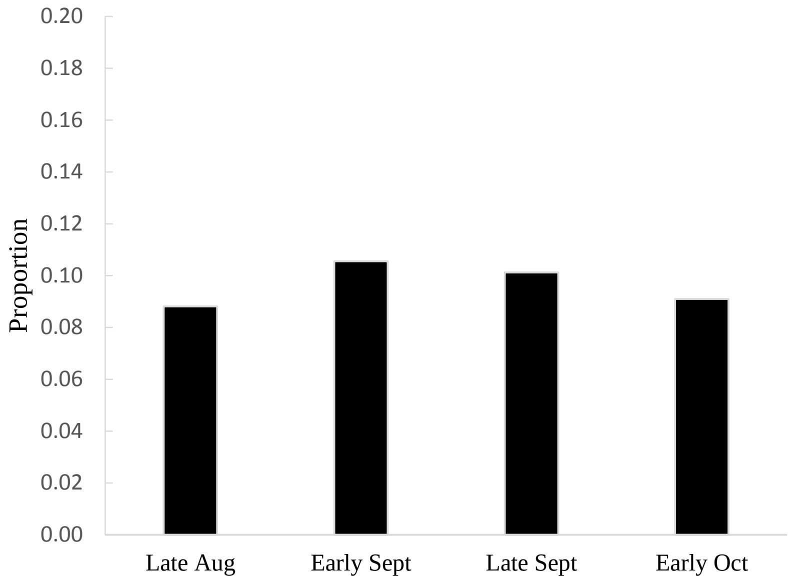


Figure 7.4. Prevalence of feeding evidence amongst bears on Baffin Island in Baffin Bay, 2012 and 2013. Data presented bimonthly.

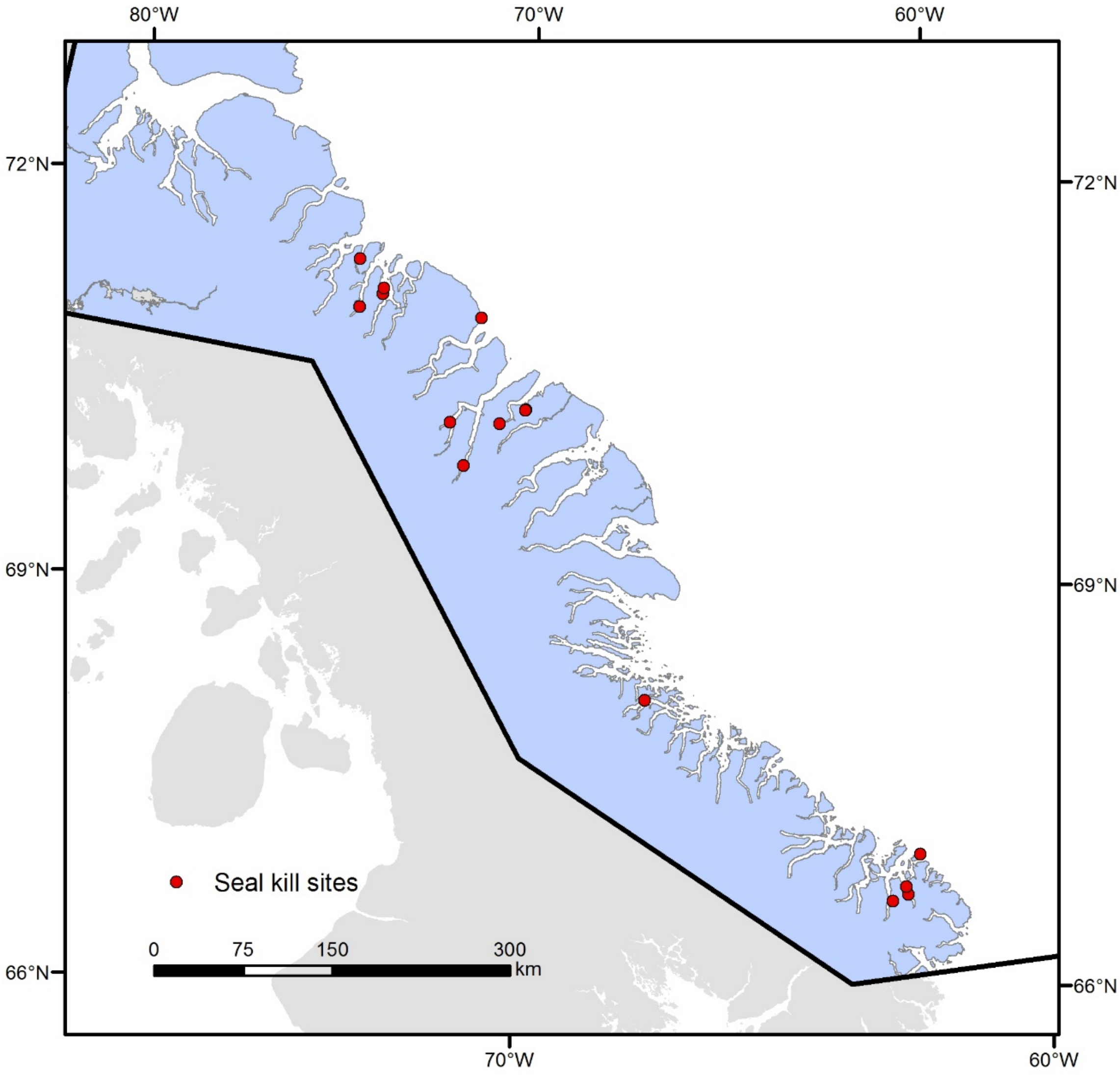


Figure 7.5. Distribution of seal kill sites observed during polar bear biopsy darting along eastern Baffin Island, Aug-Oct 2011-13.

CHAPTER 8

POLAR BEAR HARVESTING IN BAFFIN BAY AND KANE BASIN: A SUMMARY OF HISTORICAL HARVEST AND HARVEST REPORTING, 1993 TO 2014

KEY FINDINGS

- Both Canada (Nunavut) and Greenland harvest from the shared subpopulations of polar bears in Baffin Bay and Kane Basin.
- During 1993-2005 (i.e., before quotas were introduced in Greenland) the combined annual harvest averaged 165 polar bears (range: 120-268) from the Baffin Bay subpopulation and 12 polar bears (range: 6-26) from Kane Basin (for several of the years, harvest reported from Kane Basin was based on an estimate).
- During 2006-2014 the combined annual harvest averaged 161 (range: 138-176) from Baffin Bay and 6 (range: 3-9) polar bears from Kane Basin.
- Total harvest peaked between 2002 and 2005 coinciding with several events in harvest reporting and harvest management in both Canada and Greenland.
- In Baffin Bay the sex ratio of the combined harvest has remained around 2:1 (male: females) with an annual mean of 35% females amongst independent bears.
- In Kane Basin the sex composition of the combined harvest was 33% females overall for the period 1993-2014. The estimated composition of the harvest since the introduction of a quota in Greenland is 44% female but the factual basis for estimation of the sex ratio in the harvest is weak.
- In Greenland the vast majority of bears are harvested between January and June in Baffin Bay and Kane Basin whereas in Nunavut *ca.* 40% of the harvest in Baffin Bay is in the summer to fall (August – November) while bears are on or near shore. In Nunavut, all bears harvested from Kane Basin occurred in the spring.
- Sport hunting of polar bears is permitted in Canada but not Greenland. Sport hunting activity average 16% of annual harvest and peaked in 2008 coincident with several management actions. This type of hunting is highly selective for older, adult males.
- Defense-of-life-and-property kills (DLPs) of polar bears in Baffin Bay was highly selective for young (2-3 years old) individuals. We did not find evidence of a trend in the annual number of DLPs between 1993 and 2014. Most DLPs occur during the open water

period (August-November) when bears are on or near shore. The seasonal frequencies of DLPs in Canada (Nunavut) declined through the summer and fall.

- Genetic validation of the sex of individual bears as reported by the hunters showed that the gender of harvested bears was incorrectly reported in a significant number of cases. Inaccuracies in gender reporting were greatest in Greenland.
- Based on genetic validation, the presence of physical marks (i.e., ear tags or tattoos) on bears was under reported in the harvest. Detection of marked bears declined with increasing age of marks suggesting that mark loss (especially loss of ear tags) was a problem. This finding has implications for the use of harvest recoveries in MR recapture studies.
- Management and the history of management of polar bears in Canada and Greenland differs in some respects: (1) quotas were introduced in Canada in 1967 and in Greenland in 2006, (2) sport hunting is allowed in Canada but not permitted in Greenland, (3) Canadian management is designed to achieve a target harvest sex ration of 2:1 (males to females) whereas the harvest of independent polar bears in GL is non-selective, (4) the vast majority of polar bears in Greenland are taken from dog sleds whereas in Canada the vast majority are taken from snowmobiles.

8.1. Introduction

Climate induced loss of sea-ice habitat has been identified as the ultimate threat to the persistence of polar bears across their circumpolar range (PBSG 2010; Wiig et al. 2015). Other threats to the species, including pollution, industrial development, tourism and over-harvest are considered to be of varying importance amongst the different subpopulations. Three of the 19 recognized subpopulations of polar bears are currently designated as declining by the IUCN/SSC Polar Bear Specialist Group (http://pbsg.npolar.no/en/index.html). Of these three (Southern Beaufort Sea, Baffin Bay and Kane Basin), potential over-harvest has been identified as a concern in Baffin Bay (BB) and Kane Basin (KB).

Based on a mark-recapture study, Taylor et al. (2005) estimated the number of polar bears in BB at 2,074 (95% CI: 1544-2604) in 1998. A subsequent population viability analysis (PVA) using the 1998 abundance estimate, associated vital rates and available harvest data projected a decline to 1,564 bears (95% CI: 690-2402) in 2004 (PBSG 2010). Importantly, these simulations suggested that the combined harvest of bears from BB within Canada and Greenland was unsustainable. Furthermore, this PVA did not take into account on-going sea-ice habitat loss which was predicted to exacerbate the potential impacts of the harvest. In contrast to these scientific findings traditional ecological knowledge (TEK) of polar bears in BB does not suggest this subpopulation is presently declining in response to harvest and / or sea ice (Dowsley and Wenzel 2008, Born et al. 2011). York et al. (2016) have postulated that this disparity between science and TEK may be the result of inaccuracies in the scientific data for BB. In particular, potential under reporting of tagged bears in the harvest and / or over reporting of total harvest in Greenland (both under and over reporting of the harvest may result in bias in mark-recapture estimates of abundance and survival rates). However, the validity of these assertions regarding harvest reporting is unproven.

Based on a mark-recapture study, the size of the KB subpopulation was estimated to be 164 (95% CI: 94-234) polar bears for 1994 - 1997 (Taylor et al. 2008a). Taylor et al. (2008a) also suggested that Kane Basin might act as a sink (i.e., some bears may move from Baffin Bay into Kane Basin) because of unsustainable rates of harvest in KB (ibid.), and lack of genetic differentiation from Baffin Bay (Paetkau et al. 1999). Using the abundance and vital rates estimated from the 1994-97 capture study and catches reported from Nunavut and Greenland, 100% of the PVA simulations resulted in a decline in this subpopulation size after 10 years (PBSG 2010). Similar to BB, this simulated decline in subpopulation size as a result of harvest did not take into account changes in sea-ice habitat. York et al. (2016) postulated that the available TEK for KB, although limited, did not appear to contradict these scientific findings. However, neither of the sources cited by York et al. (2016) – i.e., COSEWIC (2008) and M.K. Taylor (pers. comm. 1986-2008) - bring any documentation for this statement. In contrast, a detailed study involving the interviewing of Greenland hunters with extensive experience hunting polar bears in the Kane Basin region documented the TEK perception that the occurrence of polar bear in KB has increased (Born et al. 2011).

In response to concerns regarding the projected declines in abundance, the impacts of harvest, on-going sea-ice habitat loss and the apparent disparity between scientific findings and TEK, Canada and Greenland jointly initiated new mark-recapture surveys of the BB and KB subpopulations from 2011-2014. Results from these surveys are presented elsewhere in this report (Chapters 5 and 10). Here we summarize available polar bear harvest information for BB and KB for the period 1993 to 2014; the period spanning both the recent and previous surveys in these subpopulations. Furthermore, a new system of reporting the catch of polar bears was introduced in Greenland in 1993. The objectives of this summary are to provide background information for discussion around future harvest levels and subpopulation status in light of results from the new surveys. We examine the level and composition of the harvest looking for trends over time or in relation to other factors. We also examine the harvest monitoring systems to identify issues that may affect the accuracy of harvest monitoring and the ability of jurisdictions to effectively manage harvest. Finally we make recommendations on improvements to the harvest reporting/monitoring systems.

Both BB and KB are jointly managed by Canada (Nunavut) and Greenland. Systems of harvest management and monitoring differ substantially between these two jurisdictions. Detailed descriptions of the respective systems and relevant history are presented in Appendix D. This information constitutes an important reference for the results presented herein.

8.2. Methods

Canadian Harvest Data

Data on polar bears harvested from BB and KB within Canada (Nunavut) were obtained from the Government of Nunavut's (GN) polar bear harvest database. This database contains records for all bears reported to have been harvested from these subpopulations. The methods of data collection are described elsewhere (Appendix D). From this database we extracted records for bears harvested from Jul 1, 1992 to Jun 30, 2014. The regulatory season for polar bear hunting in Canada (Nunavut) is from July 1 to June 30 the following year. This differs from Greenland where the regulatory season for polar bear hunting is from Jan 1 to Dec 31. Because adjustments in Total Allowable Harvest (quota) are made on a seasonal basis in response to recorded harvest levels in the preceding season, harvest figures were reported by season rather than by calendar year. For example, we used the notation 1992/93 when reporting harvest figures for the season beginning Jul 1, 1992 and ending Jun 30, 1993.

The extracted data included the sex, age class (adult, sub adult, 2-year-olds, yearling and cub-of-the-year [COY]), age (years), date, location and type of harvest for each individual. In reporting total harvest levels we considered all forms of human-caused mortality including illegal harvest and accidental kills but excluded bears killed for humane reasons (i.e., diseased or dying from natural causes). Other analyses were limited to the three main types of harvesting: Regular (subsistence) hunts, sport hunts and the harvest of bears in Defense-Life-and-Property (DLPs). We examined harvest by time (year or season), sex, type, monthly distribution and age. Although known for most individuals, records where sex, age class or date of harvest was unknown were excluded from the corresponding analyses where those parameters were used. Individual ages (years) based on ageing of harvested bears were only available up to June 2010 limiting our analyses of age of harvest to the period 1992/93 – 2009/10.

For DLPs we also examined the relationship with several annual metrics of sea ice to test the hypothesis that the number of human-bear conflicts, hence DLPs, increases in years when sea-ice breaks up earlier and/or forms later thereby forcing bears to remain on land for longer periods without access to their marine mammal prey (Stirling and Parkinson 2006, Stirling and Derocher 2012). As an index of the timing of spring sea-ice break-up, spring transition date was calculated as the date (Julian day) that spring sea-ice cover reached 50% over the continental shelf of BB. As an index of the timing of sea-ice formation in the fall, fall transition was calculated as the date (Julian day) that fall sea-ice cover reached 50% over the continental shelf of BB. The difference between spring and fall transition dates was used as an index for the length of the ice-free season. Addition details of these sea-ice metrics are provided in Chapter 4.

Greenlandic Harvest Data

Magnitude of the Annual Catch from KB and BB – For the period, 1993-2005 (i.e., prior to the introduction of quotas in 2006), data on the Greenland annual catch of polar bears from the KB and BB subpopulations were based on the “Piniarneq” reporting-system (for a description of this system see Appendix D). Information on catches reported via this system is compiled annually and published by the Greenland Government´s Department of Fisheries, Hunting and Agriculture (DFHA) in Nuuk. However, after the introduction of quotas when the polar bear harvest was more closely monitored detailed information on every kill was collected and compiled by the DFHA. Data for this report on the total annual kill of polar bears in 1993-2005 and in 2006-2014 were provided by the DFHA (*in litt.* August 2015)

Seasons of Catch – Information on date (day/month/year) of each polar bear catch was provided by the DFHA (*in litt.* 2014, 2015). This information was used to describe the seasonal distribution of the Greenland catch of polar bears from the KB and the BB subpopulations during 2006-2014.

Means of Transportation When Hunting Polar Bears – For each bear caught, hunters reported the type of vehicle used during the hunt (i.e., dog sled, skiff or boat; or a combination of these means of transportation). The terms “skiff” describes a boat up to 20 feet in length and usually powered with an outboard engine. Boats larger than 20 BRT/15 BT must not be used during polar bear hunts or for transportation to or from polar bear hunting grounds (Anon. 2005). These boats are small-type fishing vessel made of wood, fiber glass or steel with an inboard engine.

The information reported during 2006-2014 on vehicles used during 445 individual polar bear hunts from the KB and BB subpopulations was used to describe hunting methods both overall and regionally. Due to differences hunting traditions not at least related to stability and duration of the sea-ice cover and therefore distribution of polar bears there are regional variations in hunting methods from north to south in NW and W Greenland (*cf.* also Born et al. 2011). The NW and W coast of Greenland between Kane Basin and the town of Sisimiut was therefore subdivided into six areas: (1) Subarea 1 compasses the area to the north of Ullersuaq/Kap Alexander - 78° 10' N (i.e., Kane Basin), (2) subarea 2 consists of the area between Ullersuaq and Innaaganeq/Kap York (76° 30' N), (3) subarea 3 encompasses the areas between Innaanganeq/Kap York and the peninsula Nuussuaq/Kraulshavn at *ca.* 74° N (i.e., the Melville Bay area), (4) subarea 4 comprises the southern Upernavik area beween 74° N and the peninsula Sigguk at 71° 30' N, subarea 5 is the areas between Sigguk and the town of Aasiaat at 68° 45' N (i.e., the Uummannaq, Disko Island and Disko Bay area), and subarea 6 comprises the area between Aasiaat and Sisimiut at *ca.* 66° 55' N.

Sex and Age Composition of the Catch – Before the introduction of quotas information on the sex and age composition of the catch from KB and BB was obtained during interview surveys (Rosing-Asvid and Born 1990, Rosing-Asvid 2002, Born et al. 2011), and few biological samples (Rosing-Asvid 2002). Information on sex and age composition (i.e., independent males and females and dependent cubs) was extracted from these sources.

During an interview survey among experienced polar bear hunters in 2006 in NW Greenland (i.e., the Upernavik and Qaanaaq areas) detailed information was collected on age category (i.e., old, adult, young, and cubs) and sex of the individual bears that the interviewees had shot. Information on the composition of 588 catches (754 individual bears) going back to the early 1950s was presented in Born et al. (2011). Data on age and sex in the Greenlandic catch of polar bears during 1991-2005 (n = 354) were extracted from the interview survey database (Greenland Institute of Natural Resources, GINR, unpublished) and presented separately for the Kane Basin (KB) and Baffin Bay (BB) subpopulations.

Information on sex and age category (cub, young, adult, old) from individual polar bear kills was obtained from the DFHA based on the filled in forms provided by the licensed hunter for each polar bear killed (2006-2014). In case of catches reported by hunters living in the northernmost area (i.e., the Qaanaaq area) a polar bear may have been taken from either the KB or the BB subpopulation. In several cases coordinates of the location of kill was not given whereas the Greenlandic name of the site was noted by the hunter. In most cases it was possible to determine whether a bear was killed inside the KB management unit (i.e., north of the southern border of the KB management unit area on the Greenland side of the mid-sector line in Smith Sound; this border had been placed more or less arbitrarily at exactly 77° N; *cf*. Derocher et al. 1998, Taylor et al. 2001), or alternatively in the BB management unit. In cases where neither coordinates nor name of site of kill were stated, it was assumed that if the reporting hunter lived in the northernmost settlement Siorapaluk the bear had been taken in KB. The hunters in this settlement traditionally harvest bears from the KB management unit (Rosing-Asvid and Born 1990, Born et al. 2011). About 10.0% (5 of 49) of the kills (2006-2014) allocated to KB were based on information about hometown of the hunter reporting (or *ca.* 3.5% of a total of a total of 147 catches reported from the Qaanaaq area).

The file contained information on sex and age category of *ca.* 96% of the total catch reported for KB during 2006-2014, and *ca.* 92% of the catches reported for BB during the same period.

Detailed Age Structures – The age (and sex) structure of the Greenland catch of polar bears was examined using several sources of data. Samples (soft tissues and a vestigial tooth, i.e., 1st premolar) were collected from 55 BB polar bears caught by the hunters in 2012 and 2013. This sample represents *ca.* 40% of the reported catch in Greenland from the BB subpopulation in 2012 and 2013. The gender of each sample was determined genetically (Chapter 5). Individual ages were obtained from counting growth layer groups in premolar teeth (Calvert and Ramsay 1998).

In 2009-2013, a total of 131 individual BB-polar bears were live captured by GINR during spring (Mar 23-Apr 23) in W and NW Greenland (i.e., between 70° 14´ N and 76° 20´ N; i.e., between northern Disko Island and the settlement Savissivik) in connection with a study of movement (GINR unpublished data). These bears were tagged on fast ice and in the offshore pack ice in areas where the hunters usually take polar bears and therefore likely represent the sex and age composition of bears available on the polar bear hunting grounds. The gender of each sample was determined in the field and verified genetically (Chapter 5). Individual ages were obtained from counting growth layer groups in premolar teeth (Calvert and Ramsay 1998). The sex and age composition of this sample was included for comparison. Also for comparative purposes the age structure of a sample of 63 polar bears collected by the subsistence hunters in NW Greenland during 1988-1996 was inferred from figure 26 in Rosing-Asvid (2002:21).

Pooling of Canadian and Greenlandic Harvest Data

In contrast to Greenland where harvest is reported on a calendar year, harvest in Canada (Nunavut) is reported and quotas are adjusted on a hunting season basis, as described above. This presented some challenges in pooling data from the two jurisdictions. However, as documented later in this report, most harvesting in Greenland (99% in BB and 87% in KB) occurs between January 1 and June 30. We therefore pooled Greenland and Canadian data and report total harvest for BB and KB on the basis of Canadian harvest seasons. For example, harvest for 1998/99 includes bears harvested in Canada from July 1, 1998 to June 30, 1999 and in Greenland for January 1 to December 31, 1999.

Genetic Monitoring of Harvest

From 2011 to 2014, as part of genetic mark-recapture studies, the polar bear harvest in BB, KB and surrounding subpopulations was monitored to detect recoveries of genetically ‘marked’ bears. Tissue samples from harvested bears were submitted for genotyping and genetic sexing as described elsewhere in this report (e.g., Chapter 5). This provided data on genetic sex for comparison with the reported sex of each individual thereby allowing us to examine the accuracy of gender reporting. Additionally, several groups of bears that were physically marked with ear tags and lip tattoos during previous studies were detectable from this genetic monitoring of harvest. Archived tissues samples from all bears physically marked in BB and KB from 1990 to 1997 were genotyped and genetically sexed with the exception of bears whose ages would have been >35 years when genetic monitoring began in 2011 and a small number of individuals (n = 15) for which archived tissue samples were unavailable. Bears physically marked in BB and KB from 2010-2013 were also genotyped. Finally, bears marked during a recent physical mark-recapture study in the neighbouring Davis Strait subpopulation (Peacock et al. 2013) were genotyped. The harvest of one of these genetically and physically marked individuals could be detected by two means; the presence of ear tags or lip tattoos as reported by hunters or by matching the genotype of a harvested bear to that of a previously marked bear. This permitted a comparison of the efficiency of detection of marked bears by these two methods.

8.3. Results and Discussion

Canadian Harvest from BB and KB

Annual Harvest from Baffin Bay – In Canada (Nunavut), a total of 1,633 bears were harvested from the BB subpopulation over the period 1992/93 to 2013/14. Mean annual harvest was 74 bears per year (SD = 17.13, range 49-103, n = 22). Total Allowable Harvest (TAH) in Nunavut for the BB subpopulation over this period varied from 64 bears per year in the 1990s up to 105 starting in 2004/05 (Table 8.1). This variation reflects harvest management initiatives at the subpopulation level. An increase in TAH was implemented from 2004/05 to 2009/10 as a result of a new memorandum of understanding (MOU) between the Government of Nunavut and communities in BB. Starting in 2010/11, in response to concerns about the sustainability of the combined Canadian and Greenlandic harvest level, the TAH in Canada (Nunavut) was reduced by 10 bears per year over four years and had decreased to 65 by 2013/14. In most years, harvest from BB remained within the TAH. From 1996 onwards, exceedances of the allowable harvest resulted in adjustments of allowable harvest for the following year, in accordance with the flexible quota system (see Appendix D).

A majority (96%) of the harvest was comprised of independent bears (i.e., ≥ 2 years old) consistent with regulatory prohibitions on harvesting of family groups except in Defense-of-Life-and-Property (DLP). Also in accordance with the management system, the harvest was sex selective. Pooling data across years, the proportion of females in the harvest was 0.34 for independent bears (excludes COY and yearlings) and 0.35 for all bears which is consistent with the target sex ratio of 0.33 (i.e., 2 or more males per female). From 1996 onward, annual variation around this sex ratio (Figure 8.1: range 0.22 – 0.45) was regulated by the flexible quota system. When more than the recommended number of females were harvested, the TAH for the following season was reduced to compensate for the over harvest of females and deviation above the target sex ratio. Consequently, there was no temporal trend in harvest sex ratio.

Annual Harvest from Kane Basin – In Canada (Nunavut), harvest from KB has been minimal over the period 1992/93 to 2013/14 (Table 8.2). Total harvest during this period was 9 bears with a mean annual harvest of 0.4 bears per year (SD = 0.59, range = 0-2, n = 22). The sex ratio of the pooled harvest data (1993-2014) was 33% females (i.e., 2 males per female) consistent with the target sex ratio of the management system. The TAH in Nunavut of 5 bears per year for KB since 1996 has not been exceeded in any year. All bears were harvested in the spring (February to May).

The low level of hunting in KB is in part due to its remoteness and the logistical challenges of travelling in this subpopulation. The nearest Canadian community is Aujuittuq/Grise Fiord on southern Ellesmere Island with a population of <200 people. Access to

KB from Aujuittuq/Grise Fiord involves overland travel across southern Ellesmere Island. Costs for equipment and supplies make this journey less feasible relative to other hunting opportunities in the region (such as Jones Sounds in the Lancaster Sound subpopulation). In addition, spring-time travel into Kane Basin has been increasing in difficulty in recent years due to snow conditions (too little or early melts) on overland trails (M. Akeeagok, pers. comm.). Traditionally the Kane Basin region was a main polar bear hunting ground for the Inuit living in the Qaanaaq area in Northwest Greenland (Vibe 1968) and it is still an importing hunting area for them (Born et al. 2011).

Given the sparsity of the Canadian harvest from KB we do not report further on it in this section.

Timing of Harvest from Baffin Bay – Month of harvest was recorded for 1,594 of the 1,633 bears harvested in BB between 1992/93 and 2013/14. Pooling data across years, we examined the distribution of hunting activity across months. Approximately 40% of annual harvesting from Baffin Bay occurred between August and November when bears were on or close to shore. Harvesting activity peaked in October (23% of total).

The three main types of human-caused mortality (i.e., subsistence hunts, sport hunts and the killing of bears in defense, DLP) each varied differently in frequency throughout the year (Figure 8.2). Subsistence hunting activity (n = 1,107), termed “regular” hunting, peaked in October (29% of total hunts of this type) and continued through the fall, winter and spring until sea-ice break-up. A majority (91%) of sport hunting (n = 248) occurred on the sea ice in spring (March-May). DLPs kills (n = 185) peaked in August and declined steadily through the fall. Fifty percent of DLPs occurred in August and September. Notably, this peak in DLPs coincides with the time when bears move to shore from the melting sea ice rather than the timing of freeze-up (Taylor et al. 2005; Chapter 4). This finding is similar to Dyck (2006) but inconsistent with the hypothesis that human-bear conflicts are driven by poor body condition (Stirling and Parkinson 2006) since body condition will be at a seasonal high point in August, decreasing progressively through the fall (Rode et al. 2011). We speculate that this peak may result from the concentration of bears along the coast as they come off the ice (prior to dispersing inland) combined with the increased range and frequency of boat travel and use of camps by residents of communities along Baffin Island at this time of year. Hence higher rates of DLPs during the summer may be the product of human-bear encounter probability. However, this finding does not discount the hypothesis that bears in poor condition are more likely to come into conflict with people.

Number of Sport Hunts in Baffin Bay – Overall, sport hunting accounted for 16% of the harvest from BB in Canada from 1992/93 to 2013/14. Sport hunting activity increased through the 1990s and early 2000s before sharply declining in 2009 (Figure 8.3). This decline in sport hunting after 2008 coincided with two events. First, the 2008 listing of polar bears as "threatened" under the US Endangered Species Act and the subsequent ban on importation of polar bear hides into the US pursuant to the US Marine Mammal Protection Act (US Department of the Interior: Fish and Wildlife Service 2008). Second, Canada's issuance in 2010 of a negative non-detriment finding for Baffin Bay under the Convention on International Trade in Endangered Species (CITES) which triggered a ban on Canada's export of polar bear hides from this subpopulation. These two events seemingly reduced the pool of sport hunters interested in hunting opportunities in BB (Weber et al. 2015).

Number of Defense-of-Life-and-Property Kills (DLPs) in Baffin Bay – Bears killed as DLPs are the only type of polar bear harvest that is not limited by quota in Canada. All DLPs are deducted from the available quota but the total number of DLPs in a given year is not limited. The number of DLPs in BB varied annually with no apparent trend over the period 1992/93 to 2013/14 (Figure 8.4). DLP-related harvest averaged 8.6 bears per year or about 12% of annual harvest. There was no trend in the proportion of annual TAH allocated to account for DLP kills over the period 1993-2014 (Linear regression, $F_{1,\,21} = 0.19$, $P > 0.700$). Thus, problems bears do not appear to be using an increasing portion of the available quota in BB.

Restricting the data to independent bears(in this case, independent bears were defined as all individuals except for COYs and yearlings accompanying their mother) during the months of Aug to Nov, when most bears in BB were on land, did not reveal a trend in number of DLPs kills over time (Figure 8.4; Linear regression, $F_{1,\,21} = 1.404$, $P > 0.200$). The number of DLPs of independent bears during the Aug to Nov period was also unrelated to annual date of spring sea-ice transition (Linear regression, $F_{1,\,21} = 0.028$, $P > 0.800$), fall transition date (Linear regression, $F_{1,\,21} = 0.179$, $P > 0.700$) and the number of days between these two dates (used as proxy for length of the ice-free period: Linear regression, $F_{1,\,21} = 0.121$, $P > 0.700$). This suggests that earlier spring sea-ice break-up, later fall freeze-up and a lengthening ice free period was not associated with increased DLP kills in BB. This finding is inconsistent with the prediction that problem bear kills will increase as sea-ice habitat deteriorates and bears spend more time on land (Stirling and Parkinson 2006, Stirling and Derocher 2012) as has been observed in Western Hudson (Towns et al. 2009). However, we note that the relatively small samples size (n = 112) and shorter time series in our study relative to Towns et al. (2009) may have limited the findings. Moreover, unlike the situation in Western Hudson Bay where there is no subsistence hunting of polar bears in the province of Manitoba, some of the bears in BB that were taken by hunters and recorded as part of the subsistence harvest might otherwise have ended up becoming DLPs, especially those harvested as they approached communities or camps. The number of DLPs recorded amongst our data is thus likely to be an under and somewhat variable estimate of actual or potential DLPs occurring. Our results relating frequency of DLPs to sea-ice conditions should therefore be interpreted with caution.

Grouping data on DLPs into multiyear blocks (1993-95, 1996-00, 2001-05, 2006-10, 2011-13), there was no significant difference in the timing (mean Julian day) of DLPs of independent bears during Aug-Nov (ANOVA, $F = 0.846$, $P > 0.40$).

Sex Ratio of the Harvest in Baffin Bay – As reported above, the sex ratio of the harvest from BB for the period 1992/93 to 2013/14 was 0.34 amongst independent bears and 0.35 for amongst all bears. As expected there were significant differences in sex ratio amongst the different types of hunting ($\chi^2 = 16.03$, $P < 0.001$, df = 2) with sport hunting being selective for males (3:1 sex ratio) and DLPs being closer to a 1:1 ratio (Fig 8.5). Sport hunting contributes to the maintenance of a male-selective harvest; compensating for the less selective nature of DLPs.

Age Structure of Harvest in Baffin Bay – Several features of the harvest management system in Canada (Nunavut) that tend to select for or against bears in certain age ranges. While there is a regulatory limit (i.e., the TAH) on the total number of bears harvested each year and adjustments in this limit are made to compensate for the sex ratio of the harvest, there are no specific limitations on the age of bears harvested in BB; with one exception. A prohibition (under the Nunavut Wildlife Act) on the harvesting of family groups (defined as an adult female accompanied by COYs, yearlings or 2-year-olds), except in defense-of-life-and-property, protects most cubs-of-the-year (COY) and yearlings from hunting, as well as some 2-year-olds. This is reflected in the harvest data. Overall, between 1992/93 and 2013/14, *ca.* 20% (37/189) of

DLPs were adult females and their accompanying offspring. Comparable figures for regular hunting and sport hunts were *ca.* 0.4% (44/1104) and 0% (n = 260) respectively.

Sport hunting tends to select for larger, presumably older, adult bears. Looking at the available data on the known or tooth-derived age of harvested bears from BB, there were significant differences in the median ages of harvested female bears amongst the 3 main types of harvest: DLP, regular, sport (Fig 8.6: Kruskal Wallis, $H = 10.97$, $P = 0.004$). Females taken as sport hunts tended to be older than those harvested as DLPs. Similarly, the median age of male bears was significantly different amongst types of hunting (Fig 8.6: Kruskal Wallis, $H = 61.38$, $P < 0.001$). Sport hunting was highly selective for older males relative to both regular hunts and DLPs. DLPs selected for younger males.

Dyck (2006) found that a majority of DLPs occurring in polar bear subpopulations across Nunavut involved bears < 7 years old. Looking more closely at the age distribution of DLPs for Baffin Bay, we see that most bears (≈ 60%) coming into conflict with people are ≤ 3 years of age. Specifically, juveniles aged 2 and 3 years were over-represented amongst the DLPs relative to the ‘population’ age structure derived from mark-recapture sampling, especially amongst males (Figure 8.7; Males: $\chi^2 = 64.55$, df = 13, $P < 0.001$; Females $\chi^2 = 30.41$, df = 13, $P < 0.005$). Bears aged 2-3 years are at a stage, between weaning and the on-set of sexual maturity, when they may be particularly vulnerable to conflict with humans (Towns et al. 2009). Food availability for these newly independent juveniles may be relatively low since hunting skills are still developing and their relatively small body size limits the ability to compete with larger bears for food. The demands of continued growth during a life-stage of relative food scarcity may lead to poor body condition and reduced survival amongst this age class particularly as environmental conditions deteriorate (Regehr et al. 2007). Poor body condition may increase the tendency for juveniles to seek food from sites of human activity. Additionally, these individuals may be naïve with respect to risks of interacting with humans; having not yet acquired learned aversions to humans and adopted avoidance behaviors.

Annual variation in the ages of harvested bears was examined by sex and harvest type for the period 1992/93 and 2010; the period for which data were available. The median age of harvested bears did not exhibit trends over time for any of the different types of hunting (for example see Figure 8.8).

Greenlandic Harvest from BB and KB

Annual Harvest from BB – Sport hunting of polar bears is not permitted in Greenland (Anon. 2005). Hence, the only harvest types recorded by the management authorities are (1) regular subsistence harvest under quota, (2) bears killed in Defense-Life-and-Property (DLPs), and (3) illegal hunts.

The Greenlanders´ catch of polar bears from the BB subpopulation according to the Piniarneq catch recording system (see Appendix D) during 1993-2005 (i.e., the year before introduction of quotas) are shown in Figure 8.9. The trend in numbers reported per year during 1993-2005 in the Qaanaaq area (i.e., north of the Upernavik area) is not statistically significant ($R = 0.272$, $Z = 0.881$, $P = 0.378$, n=13). However, the catch in the Uummannaq-Sisimiut area (i.e., south of Upernavik) increased statistically significantly during the same period ($R = 0.594$, $Z = 2.163$, $P = 0.031$, n=13). The catch in these areas amounted to *ca.* 24% of the total catch reported by Greenlanders from BB during 1993-2005. Similarly, the catch reported from the Upernavik area (between *ca.* 74° 35ʾN and *ca.* 71° 30ʾN) increased significantly during the same period ($R = 0.794$, $Z = 3.426$, $P = 0.001$, n = 13). On average the reported catch of polar bears in the Upernavik area amounted to *ca.* 57% of the total catch reported in Greenland from BB during 1993-2005 (Figure 8.9). Hence, there are indications that the Greenland catch of polar bears from BB showed a real increase and especially after *ca.* 2000. An increase in availability of polar bears in the Upernavik area during the 1990s and 2000s was also indicated by traditional ecological knowledge, TEK (Rosing-Asvid and Born 1990, Born et al. 2011).

The Piniarneq-data have played an important role in the assessment of abundance and trends in abundance of the BB and the KB subpopulations. Since 1993, the Piniarneq-data on annual catch for the shared subpopulations (BB, KB and DS) as compiled and published by the DFHA have been provided annually to the Canadian Polar Bear Technical Committee and to the meetings of the IUCN/SSC Polar Bear Specialist Group. The data have been incorporated with the reported catch of polar bears in Nunavut from BB in modeling of subpopulation status (e.g., Aars et al. 2006, York et al. 2016). In a PVA (Population Viability Analysis) re-assessment of polar bear subpopulations including BB and KB based on historical data, York et al. (2016) simply assumed that the Greenland catch of polar bears reported in Piniarneq for BB was/are overestimated (“over-reporting”). However, these authors did not present any validation of the Piniarneq-data or new evidence in support of this assumption (Ibid.). As indicated in the previous other evidence suggests that the increase in the Greenlanders´ catch from BB during 1993-2005 was real. Although over-reporting, or under-reporting, to an unknown extent cannot be ruled out.

Annual Harvest from KB – During 1993-2005 (i.e., prior to introduction of quotas in Greenland in 2006) the Greenland annual catch of polar bears from the Kane Basin (KB) subpopulation was 11 (SD = 4.4 bears, range: 6-25/year; n,= 13 years) with no apparent trend.

However, it should be noted that during 1993-1999 the annual catches from KB were estimated at 10 each year during 1993-1999 (PBSG 2002, 2010) based on an interview survey in 1989 (Rosing-Asvid and Born 1990). The statistics on annual catches after 1999 were based on reportings in Piniarneq (2000-2005) and special reporting forms under the quota system (2006-2014).

After the introduction of quotas the Greenlanders´ catch of polar bears from the KB management unit (2006-2014) has averaged 6/year (SD = 1.7, range: 2-8/year, n = 9).

Defense-of-Life-and-Property Kills (DLPs) – No defense kills were reported for the Kane Basin subpopulation during 2006-2014. During the same period 7 defense kills were reported for the Baffin Bay subpopulation (2007: 2, 2011: 1, 2012: 4). The kills comprised 1 young male, 3 adult females, 1 young with sex not stated, and 2 with sex and age not stated. The months during which these incidences occurred were: January (n = 1), February (1), July (2), October (2) and December (1). Hence, since the introduction of quotas in 2006 when the recording of defense kills began, there have been no apparent annual or seasonal trends in defense kills from the KB and BB subpopulations.

In Greenland DLPs in one year are not subtracted from next year quota (DFHA, *in litt.* 2016).

Uncertainties in Catch Reporting – Prior to 2006, when quotas were introduced, there was significant uncertainty in the accuracy of the reported polar bear harvest (catch) from BB and KB in Greenland. Reporting occurred via the Piniarneq system. It has been suggested that the polar bear catch reported through the Piniarneq system may be both an under-estimate (i.e., some kills not reported) and an over-estimate (Born 1998, 2002, 2006, Jessen 2002, Rosing-Asvid 2002) of the actual catch. To obtain a hunting license for the coming year, a hunter must report his catch through the Piniarneq (in practice the catch of all species is reported in a booklet). Inevitably, a hunter may sometimes fail to report a catch of a particular species. Very often several hunters participate in a polar bear hunt (Rosing-Asvid 2002, Born et al. 2011). "Multiple reporting" (*i.e.*, one kill is reported by more than one hunter) leading to an over-estimate may thus occur when several hunters, who have participated in the same hunt and are proud that a bear was taken, each report the same kill via Piniarneq regardless of whether they shot the bear (Jessen 2002).

When evaluating the Piniarneq shortly after it was introduced, Kapel and Rosing-Asvid (1996) wrote that some hunters were not used to paperwork, and they may not see the point of keeping exact notes on the dates and numbers of animals taken. Whether this resulted in under-reporting, over-reporting, or just arbitrary reporting in order to have something to report when renewing the license, was not clear (Ibid.). In a study of the Greenland catch of ringed seals, Teilmann and Kapel (1998) identified examples of both under-reporting and over-reporting.

Generally, the numbers reported in Piniarneq are higher than those reported in the previous system of recording catches (i.e., The Hunters Lists of Game, *cf.* Teilmann and Kapel 1998). This apparent difference may be caused by several factors: (1) previous information was incomplete and the estimates of unreported catches too low, (2) the Piniarneq-system overestimates the catch due to "multiple"-reporting, (3) a real increase in the catch, or (4) a combination of all these factors. An example of sources of error in *Piniarneq* is the report in 2004 of 24 and 10 polar bears reported for Sisimiut and Maniitsoq, respectively (Born and Sonne 2006). Some of these (10 and 5) were reported by hunters with a "part-time" hunting license and were suspected to be of muskoxen (O. Heinrich, DFHA, *in litt.* 2005).

Rosing-Asvid (2002) compared information from various sources (trade in hide, information from sampling of biological tissues, and interview survey and Piniarneq) about the catch of polar bears in Greenland for the period 1993–1998. He found cases of under-reporting and of over-reporting. In the Piniarneq, simple errors like ringed seals reported as walruses or polar bears occur. However, validation of the information is not a standard procedure and some over-reporting is found in most of the species where the annual catch is low (Rosing-Asvid 2002). Another type of error may occur because the hunter does not have to report to Piniarneq where the polar bear was shot. The kill is assigned to the municipality in which the hunter lives and is therefore in some cases misplaced if the hunter has taken the bear in another area (ibid.).

According to Rosing-Asvid (2002) validating the Piniarneq-data was (is) not a standard procedure and some over-reporting is found in most of the species where the annual harvest is low. The number of polar bear kills reported in Central Greenland might be overestimated with this new reporting system, however, the trend toward more polar bears caught in West Greenland since the mid 1980’s is undoubtedly real. For the period 1970-87 the reported catches only averaged 2/year in Central West Greenland, which is less than reported through the media in the latest years or by forms that for some kills have been filled out at local offices since 1995. The interviewed hunters from Upernavik also reported a marked increase in the number of polar bears in the area since mid-1980s (Born et al. 2011).

As indicated the Piniarneq-data may in some cases represent under-reporting and in other cases “over-“ or “multiple”-reporting. As there has been no standard procedure in place in Greenland for validating to which extent (and/or in which direction) the Piniarneq-system is influenced by these potential errors when comes to polar bears one must be cautious when using and interpreting the data in Piniarneq, as pointed out by Born (2002).

Because of a good correlation between the number of ringed seal hides traded and the number of ringed seals reported via Piniarneq in the Upernavik municipality, Rosing-Asvid (2002) concluded that generally the Piniarneq system worked (works) well in this area (*Ibid*.). Hence, he indicated that the increase in reported catches of polar bears in the Upernavik area from the BB subpopulation was real. The average reported catch of polar bears in the Upernavik area during 1970-1987 (HLG) was 9 bears/year (range: 1-41/year) whereas during 1993-1998 it was 37 bears/year (range: 25-48/year) according to the Piniarneq (Ibid.).

Timing of Harvest in Baffin Bay – The Greenlanders catch of polar bears from the Baffin Bay subpopulation is almost exclusively concentrated in winter and spring (Jan-Jun). About 99% (n = 589) of the catches reported with month during 2006-2014 were taken during this period. The remaining *ca.* 1% (n = 5) is caught during Sep-Dec (Figure 8.10). This timing of the catches is in marked contrast to the situation in Nunavut where *ca.* 40% of the harvest is between Aug-Nov with a peak in October.

About 99% of the bears reported from BB were caught during Jan-Jun which is in contrast to pre-2006 when a relatively larger proportion was taken during fall and early winter. The fact that the catches have been more concentrated to the beginning of the year may be a result of the quota for BB (the quota year starts 1 January) being used up fast. According to the interviews conducted in NW Greenland in 2006 the availability of polar bears during spring in BB has increased (Born et al. 2011) which may explain that the quota is used up relatively soon and the catches therefore concentrated to late winter and spring.

Timing of Harvest in Kane Basin – Of 49 catches reported from the Kane Basin subpopulation during 2006-2014, 87.8% (n = 43) were taken during winter-spring (Jan-Jun) with a clear peak in April. Twenty-six (53.1%) of these catches were taken during April. The remainder (12.2%) of the catches were reported from summer and fall (Aug-Oct); Figure 8.11.

The tendency with a peak in hunting activity in Kane Basin in March-April and a less pronounced peak in fall was also seen prior to 2006 (Born et al. 2011:185). However, after 2006 the seasonal distribution of catches from the KB subpopulation has shown a more conspicuous peak in April. It should however, be noted that the sample size from KB was small.

Means of Transportation – In some cases information on means of transportation used during the polar bear hunt is lacking from the reports on individual catches. However, during 2006-2014 there was information on means of transportation used during 445 individual polar bear hunts from areas between Kane Basin and Sisimiut (i.e., from the KB and BB subpopulations, respectively). Overall, 63.6% of the bears had been caught during a hunt involving dog sled. During 35.3% of the hunts a skiff was used and only in 1.1% (n = 5) of the cases the bear was caught from a <20 BRT/15 BT boat.

Means of Transportation in Baffin Bay – There was regional variation in means of transportation used during polar bear hunts in Baffin Bay. In Subareas 3 and 4 (encompassing the Melville Bay and the Upernavik areas) where *ca.* 85% of the Greenlanders´ annual catch of bears from the BB subpopulation are taken, an average of 71.1% of the bears are caught during dog sled trips (the remainder are taken from skiffs) with a clear difference between the northern part and the southern part of the area (Table 8.3) reflecting differences in density of sea-ice cover and timing spring break-up. In Subareas 5 and 6 (Uummannaq, Disko Bay and areas south to Sisimiut) where only *ca.* 15% of the bears caught by Greenlanders´ from the BB subpopulation are taken, 7.8% of the bears reported during 2006-2014 were killed during dog sled hunts, 88.2% were taken from skiff and 3.9% from a small boat (Table 8.3). These areas have open water (or light pack ice conditions) during winter and/or early spring (e.g., Buch 2001). Dog sleds were used in 93.5% of the cases in the important polar bear hunting areas 1-3 and in 20.8% of the cases in areas 4-6 (i.e., the areas south of 74° N). This difference in means of transportation between the two overall areas was statistically significant ($\chi^2 = 246.283$, $P < 0.0001$, df = 1).

Only in one instance during 2006-2014 was it specified that a polar bear had been caught during a hunt involving the use of a snowmobile (using a snowmobile in connection with hunting polar bears is illegal in Greenland; Anon. 2005). This case involved the illegal kill of a male bear in the Sisimiut area in March 2011.

In none of the areas was there a statistically significant annual trend in fraction of bears taken from skiff or boat during 2006-2014 (linear regressions of weighted percentages of skiff+boat versus year; data not shown).

Means of Transportation in Kane Basin – Of 39 individual polar bear catches (2006-2014) from the Kane Basin subpopulation (i.e., Subarea 1 and 2; only 2 catches reported from Subarea 2), 76.9% were taken from dog sled, 15.4% from skiffs 7.7% from a boat. The catches from skiffs and boats were taken during May (n = 1) and June-October (n = 8); Table 8.3.

The data on means of transportation showed a marked north-south gradient in the use of sleds vs. skiffs related both to differences in hunting traditions and availability of dense fast or pack ice. In the southern areas where sea-ice conditions to a large extent are influenced by the inflow of relatively warm current from the south (e.g., Buch 2001) the majority of polar bears are taken from skiffs (and in a few cases from small-type fishing vessels) whereas in the areas north of *ca.* 74° N, where there is fast ice and dense pack ice, the majority of polar bears are caught by dog sled and this means of transportation is still an important element in the traditional way of living and hunting. The 2006-interview survey indicated that there has been an increase in the use of skiffs for hunting polar bears especially in the Upernavik area since the early 1990s (Born et al. 2011). This development was ascribed to the fact that the sea-ice conditions have become more unstable (for driving a dog sled) and there is an earlier spring ice break-up (ibid.). The same tendency was indicated in the 2006-2014 records of hunting methods (present study) but was not statistically significant.

Hence, in Greenland, the majority of polar bears that are taken from the KB and BB subpopulations are still taken during dedicated polar bear hunts where the dog sled (in the majority of cases) is used for transportation and tracking of polar bears. This maintains an old and traditional way of hunting polar bears.

Since 1968 snowmobiles have been used increasingly in the polar bear hunt. In Arctic Canada, polar bears are nowadays hunted almost exclusively with snowmobiles (except for guided sport hunts, which are required to use a dog team); Slavik (2013 and references therein).

Sex Ratio in Baffin Bay – The sex and age composition of the Greenlanders´ catch of polar bears from the BB and KB subpopulations is presented in Table 8.4 and 8.5, respectively.

In Baffin Bay the relative proportions of independent polar bear females and males in three different sets of data (1982-2005; Table 8.4) were not statistically different ($\chi^2 = 1.096$, $P = 0.578$, df = 2). Overall, independent females constituted *ca.* 28.3% of the total annual catch (note: including cubs which constituted *ca.* 20% of the catch) prior to 2006.

Independent female polar bears constituted *ca.* 32.5% of the total annual catch during 2006-2014. Overall, the ratio of independent F:M in the catch was *ca.* 1:2 both during 1982-2005 and 2006-2014.

In a sample of 55 bears (2012 and 2013) for which gender was determined genetically and tooth-derived ages were known independent females constituted 45.5% (Table 8.4). The relative proportions of independent females and males in this sample differed significantly at the 5% level from those in the larger sample (2006-2014) which was based on reports from the hunters ($\chi^2 = 3.972$, $P = 0.046$, df = 1).

Sex Ratio in Kane Basin – In Kane Basin the relative proportions of independent polar bear females and males in three different sets of data (1982-2005; Table 8.5) did not differ ($\chi^2 = 4.609$, $P = 0.099$, df = 2). Overall, independent females constituted *ca.* 25.4% of the total annual catch (note: including cubs which made up *ca.* 8% of the total catch) prior to the introduction of quotas in 2006. During 2006-2014 independent female polar bears constituted *ca.* 44.9% of the total annual catch (note: after 2005 it has been illegal to catch dependent cubs irrespective of their age). The relative proportions of independent females and males during 1982-2005 and 2006-2014, respectively, differed significantly ($\chi^2 = 5.130$, $P = 0.024$, df = 1); independent females constituting a higher proportion of the catch in KB after 2006. However, it must be kept in mind that the basis for data before 2005 is heterogeneous and sample size after 2006 is relatively small.

Age Structure – In a hunter collected sample of a total of 55 polar bears caught in BB Greenland during 2012 and 2013 individuals less than 10 years of age constituted 85.5% and ≥ 10 years olds were 14.5% of the catch. The oldest bears were two 17 year old females (Figure 8.12). In comparison, polar bears less than 10 years of age (i.e., 2-9 years of age) constituted 63.9% and 10+ olds 36.1%, respectively, in the sample of live captured bears (2009-2013) from BB. Hence, the proportion of polar bears ≥ 10 years of age was significantly higher than in the sample from the harvest ($\chi^2 = 8.026$, $P = 0.005$, df = 1). The oldest live captured polar bear was a 23 year old male (Figure 8.13).

Adult (i.e., sexually mature) females were defined as greater ≥ 4 years old and adult males as greater than or equal to 5 years old (Molnár et al. 2008). The relative proportion of sexually immature and sexually mature polar bears in the 2012-2013-sample did not differ significantly from a sample of 55 polar bears aged 2+ sampled in NW Greenland during 1988-1996 (Rosing-Asvid 2002: figure 26; $\chi^2 = 1.094$, $P = 0.296$, df = 1). COYs and 1 year olds were excluded from this comparison because only the sample from 1988-1996 contained these age groups. The oldest bear in the 1988-1996- sample was a 16 year old female.

Polar bear cubs usually follow the mother for two years and are weaned at 2.5 years of age although some are weaned already during their second spring (range: 1.3-2.3 years; Lønø 1970, Lentfer et al. 1980, Ramsay and Stirling 1986, Amstrup and Durner 1995). Hence for modeling purposes bears 3 years of age and older are considered “independent” or “adult (Taylor et al. 1987). Prior to introduction of quotas (2006) it was prohibited to catch COYs and yearlings in W and NW Greenland whereas after 2005 it became prohibited to take dependent cubs (i.e., cubs demonstrably belonging to a family group) irrespective of their age. Hence, we assume that the 2-year-olds reported in the Greenland catch in recent years (Figure 8.12) had left their mothers. Rosing-Asvid (2002: figure 26) presented an age composition of a sample (1988-1996) that comprises COYs, yearlings and 2-year-olds. Hence, due to differences in hunting regulations during the two periods (i.e., prior and after quotas were introduced) only 2-year-olds and older bears were considered in the comparison of age-structure in the catch. The recent sample from the catch and the sample from 1988-96 did not comprise any bears older than 17 years of age. In the sample from NW Greenland (1988-1996) Rosing-Asvid (2002) found the oldest male to be 14 years and the oldest female 16 years of age. A comparison of the two admittedly not large data sets indicates no change in age composition of the Greenlanders catch from BB over time.

The finding of relatively few polar bears ≥ 10 years of age and no individuals older than 17 years of age in the harvest is perhaps surprising and in contrast to the situation in the sample of live captured bears (2009-2013) from NW Greenland.

The sample from the harvest in Nunavut from Baffin Bay contained several bears older than 17 year of age (*cf.* Figure 8.7).

In contrast, in Central East Greenland where in a sample of 238 polar bears (1983-1996; Rosing-Asvid 2002: figure 8) *ca.* 9% were older than16 years. The oldest female was 26 years and the oldest male 30 years (ibid.).

Combined Canadian and Greenlandic Harvest from BB and KB

Baffin Bay – Data from Nunavut and Greenland were combined to examine overall harvest levels in BB for the period 1993-2014 (Table 8.6). Mean annual harvest was 163 bears (SD = 37.9, range 120-268, n = 22 years). For the period 1993-2005, prior to the introduction of a quota in Greenland, the mean annual total harvest was 165 bears (SD = 48.9, range = 120-268, n = 13 years). For period 2006-2014, after the introduction of a quota in Greenland, the mean annual harvest was 161 bears (SD = 13.6, range = 138-176, n = 9 years).

Total harvest in Baffin Bay peaked between 2002 and 2005 (Figure 8.14). This peak was the result of two events: (1) an increase in allowable harvest in Canada in 2004 (from 64 to 105 bears per year) as part of a new management agreement for the subpopulation and (2) a large increase in reported harvest in Greenland.

Total harvest declined after 2005 through to 2014 as a result of two management initiatives: (1) the introduction of a quota in Greenland in 2006 and a subsequent decrease in annual quotas and (2) implementation of a phased reduction (10 bears per year for 4 years) in Total Allowable Harvest in Canada.

The sex ratio of the reported harvest has remained around 2:1 (male: females) with an annual mean of 0.35 females amongst independent bears.

Kane Basin – Total estimated harvest in Kane Basin for the period 1993-2014 (Table 8.2) was 204 bears with a mean of 9.3 bears per year (SD = 4.63, n = 22, range = 3-26). Prior to the introduction of a quota in Greenland, mean estimated harvest in KB was 11.6 bears/year (1993-2005: SD = 4.61, n = 13, range 6-26). Following the introduction of a quota in Greenland, harvest decreased to a mean of 5.9 bears/year (2006-2014: SD = 1.62, n = 9, range = 3-9).

Thus since the introduction of a quota, the estimated harvest has halved in size and there has been a significant reduction in annual variation. However, the uncertainty of the number of polar bears taken from KB prior to 2006 must be mentioned. The polar bear hunters living in the Qaanaaq area in NW Greenland harvest polar bears both from the Kane Basin and from the Baffin Bay subpopulation (e.g., Born et al. 2011). The annual reports of total catch in the Qaanaaq area during 1993-1999 did not specify whether a bear had been taken from KB or from BB, respectively. Consequently it was assumed that 10 of the total number of polar bears reported from the Qaanaaq annually had been extracted from KB during this period. The remainder was assumed to have been taken from the BB subpopulation. However, the estimate of 10/year for KB represented the upper range of an estimate of 5-10/year which was based on an interview survey conducted in 1989 (Rosing-Asvid and Born 1990, 1995, PBSG 2010). During 2000-2005 the estimates of the fraction of bears reported from the Qaanaaq area that had been taken in KB were based on location of settlement reporting whereas during 2006-2014 the numbers are based on report of actual site of the kill.

Adding to the uncertainty of the exact number of polar bears that are taken by Greenland from the KB subpopulations is the simple fact that it cannot be determined with any certainty whether a bear taken in the central parts of the Qaanaaq area (i.e., close to the border at 77° N between the KB and BB management zone) belong the KB or the BB subpopulation. This uncertainty will of course have greater implications for the relatively small catch from KB than for BB.

Overall for the period 1993-2014, the estimated sex ratio of bears harvested in Kane Basin was 33% females. However, the sex ratio of the harvest since the introduction of a quota in Greenland has been approximately 44% female (based on pooled data for the period 2006-2014).

Accuracy of Harvest Reporting as Assessed from Genetic Studies of Sex and Individual Identity

Reporting of the Sex of Harvested Bears – During the recent genetic mark-recapture studies in BB and KB (2011-14) bears harvested in BB, KB and surrounding subpopulations were genotyped to establish genetic sex and individual identity in-order to detect recoveries of genetically marked (biopsied) individuals (Table 8.7). Rates of tissue sampling, reporting of sex and genotyping of bears harvested in BB and KB were less than 100%. Overall, 270 (75%) of the 359 bears that were reported as harvested in BB during the mark-recapture sampling period were tissue sampled and genotyped. For Kane Basin, 4 (40%) of the 10 harvested bears were genotyped. Sampling of harvested bears was lower in Greenland than Nunavut.

Using these genotyped bears we compared the gender of harvested bears as reported in harvest records in Canada and Greenland to the gender as determined by genetics. As part of the genetic analyses, conflicts between reported and genetic sex were investigated via additional genotyping to confirm genetic sex (see description of genetic methods in Chapter 5). Thus confidence in the genetic sex data is high. The results indicate there was significant inaccuracy in gender reporting with a bias towards under reporting of females. Pooling data for Canada and Greenland, 16% of genetic females in the harvest were reported as males (Table 8.8). In contrast, 4% of genetic males were reported as females. The bias was greatest in the Greenland harvest, where 39% of genetic females were reported as males and 12% of genetic males were reported as female (Table 8.9 and Figure 8.15a). In Nunavut, 5% of females were reported as males. Two percent of males were reported as females (Table 8.10 and Figure 8.15b). Overall, the sex composition of the genotyped harvest as reported in official harvest records was 37% females. The genetic composition of this harvest was 42% females. For the Greenland harvest, the sex composition of the reported harvest was 39% females. The genetic composition was 54% females. For the Nunavut harvest, the sex composition of the reported harvest was 36% females. The genetic composition was 37% females. Considering only independent bears (subadults & adults), for the Greenland harvest, the sex composition of the reported harvest was 40% females. The genetic composition was 54% females. For the Nunavut harvest, the sex composition of the reported harvest was 36% females. The genetic composition was 36% females.

Assuming these data are representative of the overall harvest, harvest in Greenland appears to be non-selective for sex. Harvest in Nunavut is approximately 2:1 males to females in accordance with target sex ratio of the flexible quota management system. In Nunavut verification of the sex of harvested bears is a regulatory requirement. Hunters are required to submit the baculum from harvested males. Where proof of sex is not provided sex is verified by genotyping. Our finding that gender reporting in the Canadian (Nunavut) data is accurate was thus expected.

Inaccuracies in reporting the sex of harvested bears is a management issue for these subpopulations. Determination of sustainable harvest levels in part depends on the sex ratio of the harvest (Taylor et al. 2008b; Regehr et al. 2015). The less selective the harvest, the lower the sustainable harvest. Incorrect reporting of gender for harvested bears may also be an issue for some of the past demographic analyses for BB and KB (e.g., PVA's) that have been used to establish quotas and subpopulation status; albeit to an unknown extent at present.

Reporting of Marked Bears in the Harvest – Using data for bears that were physically marked (tagged and tattooed) in either Baffin Bay (1990-1997) or Davis Strait (2005-2007), and subsequently also genotyped, we examined the accuracy of reporting of ear tagged and lip tattooed bears in the harvest relative to the detection of these marked bears via genotyping. Due to small samples sizes we pooled data on recoveries of physically marked bears in Nunavut and Greenland.

In the harvest data for Baffin Bay for the period 2011-2014, 9 recoveries of physically marked bears were detected by genotyping, 4 of which were not reported as tagged or tattooed in official harvest records. Expanding this dataset to the Davis Strait harvest records, resulted in detection of 42 physically marked bears, 12 of which were unreported as marked in harvest records. One of the unreported marks was a Greenland harvest record.

Approximately 29% of recoveries were not reported as being marked. These findings suggest that a significant portion of physically marked bears that are recovered in the harvest are undetected via the harvest reporting systems currently in place. Although the data are limited, detection probability appears to be a function of the age of a mark; the interval between application of tags and tattoos and subsequent harvest (Fig 8.16). Older marks are less likely to be reported suggesting that loss of tags or fading of tattoos may affect the ability of hunters to detect when they have harvested a marked bear. Indeed, in cases where marked bears were reported in the harvest, 62% were reported based on the presence of lip tattoos only. Examining data for bears physically marked in the Baffin Bay during 1990-97 that were recaptured during the recent Davis Strait inventory (2005-07) we found that 24 of 24 marked individuals whose recapture was detected by genotyping were also detected as marked by field workers. In most cases, notes on the field data sheets indicated that the recaptured bears had lost both ear-tags and were identified by means of their lip tattoo only. Mean capture interval (i.e., age of mark) was 11 years (range 8 to 15) amongst this sample of 24. This suggests that loss of ear-tags is the primary problem affecting detection of marked bears in the harvest. These findings also suggest that the problem of detecting marked individuals may be limited to the harvest data only.

The implications of this finding require careful consideration with respect to past and future mark-recapture studies. The assumption that all marked bears recovered in the harvest are reported, an assumption made in previous polar bear mark-recapture studies in Baffin Bay (Taylor et al. 2005, Peacock et al. 2012), Kane Basin (Taylor et al. 2008a), Davis Strait (Peacock et al. 2013) and elsewhere, appears to be invalid. Under-reporting of marked bears in the harvest may have introduced bias resulting in underestimation of natural survival rates in these studies. However, the extent (significance) of the bias is unknown at present. We recommend further investigation of this issue.

In our admittedly limited sample of harvest recoveries, detection of marks ≤ 5 years old was > 90%. Moving forward, this finding highlights the importance of maintaining a sample of recently (within 5 years) marked bears in the subpopulation when relying on detection of physical marks to estimate survival rates. Alternatively, we recommend genetic monitoring of the harvest in future studies where detection of 'old' marks is anticipated to play an important role. Further research into materials and designs for increasing the endurance of ear-tags may also be warranted.

8.4. Literature Cited

Aars, J., N. J. Lunn, and A. E. Derocher, editors. 2006. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Amstrup, S. C., and G. M. Durner. 1995. Survival rates of radio-collared female polar bears and their dependent young. Canadian Journal of Zoology 73:1312–1322.

Anon. 1954-87. Sammendrag af Grønlands Fangstlister. Ministeriet for Grønland [Summaries of "Hunters´ Lists of Game"]. Summarized during 1954-1984 by the Danish Ministry of Greenland, Copenhagen, and during 1985-1987 by the Ministry for Fisheries, Hunting and Agriculture, Nuuk. [In Danish].

Anon. 1973. *Agreement on the Conservation of Polar Bears*. Available at: http://pbsg.npolar.no/en/agreements/agreement1973.html.

Anon. 1976. Bekendtgørelse om ændring af bekendtgørelse om fredning af isbjørne i Grønland [Executive order of preservation of polar bears in Greenland]. Nalunaerutit-Grønlandsk Lovsamling, Serie A, Nr. I, MJ.G.j. nr. 1701-D8: 109. [In Danish].

Anon. 2005. Hjemmestyrets bekendtgørelse nr. 21 af 22. september 2005 om beskyttelse og fangst af isbjørne [Greenland Home Rule Executive order No. 21 of 22 September 2005 on protection and harvest of polar bears], 5 pp. [In Danish].

Born, E. W. 1998. Research on polar bears in Greenland, primo 1993 to primo 1997. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears:

Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W. 2002. Research on polar bears in Greenland 1997-2001. Pages 67–74 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., and C. Sonne 2006. Research on polar bears in Greenland, 2001-2005. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Buch, E. 2001. The ocean environment. Pages 111–122 *in* E. W. Born, and J. Böcher, editors. The Ecology of Greenland. Atuakkiorfik/Education Publishers, Nuuk, Greenland.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Derocher, A. E., G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. 1998. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Dowsley, M., and G. W. Wenzel. 2008. ‘‘The time of the most polar bears’’: A co-management conflict in Nunavut. Arctic 61:177–189.

Dyck, M. G. 2006. Characteristics of polar bears killed in defense of life and property in Nunavut, Canada, 1970– 2000. Ursus 17:52–62.

Jessen, A. 2002. A note on the management of polar bears in Greenland. Page 65 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Kapel, F. O., and A. Rosing-Asvid. 1996. Seal hunting statistics for Greenland 1993 and 1994, according to a new system of collecting information, compared to the previous Lists-of-Game. NAFO Scientific Council Studies 26:71–86.

Lentfer, J. W., R. J. Hensel, J. R. Gilbert, and F. E. Sorensen. 1980. Population characteristics of Alaskan polar bears. International Conference on Bear Research and Management 4:109–115.

Lønø, O. 1970. The polar bear (*Ursus maritimus*, Phipps) in the Svalbard area. Norsk Polarinstitutt Skrifter 149:1–115.

Molnár**,** P. K., A. E. Derocher, M. A. Lewis, and M. K. Taylor. 2008. Modelling the mating system of polar bears: a mechanistic approach to the Allee effect. Proceedings Royal Society B 275:217–226.

Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. 2010. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2002. Status of the polar bear. Pages 21–35 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Ramsay, M. A., and I. Stirling. 1986. On the mating system of polar bears. Canadian Journal of Zoology 64:2142–2151.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rosing-Asvid, A. 2002. The polar bear hunt in Greenland. Greenland Institute of Natural Resources Technical Report 45, 25 pp.

Rosing-Asvid, A., and E. W. Born. 1990. Fangst af isbjørn (*Ursus maritimus*) i Avanersuaq og Upernavik kommuner: En interviewundersøgelse [Catch of polar bears in the municipalities of Avanersuaq and Upernavik: an interview survey]. Grønlands Hjemmestyres Miljø- og Naturforvaltning Teknisk Rapport Nr. 23, 63 pp. [In Danish with English summary].

Rosing-Asvid, A., and E. W. Born. 1995. The catch of polar bears in northwestern Greenland. Page 188 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Slavik, D. V. W 2013. Knowing *nanuut*: Bankslanders' knowledge and indicators of polar bear population health. MSc thesis, Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta, Canada.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

Teilmann, J., and Kapel, F.O. 1998. Exploitation of ringed seals (*Phoca hispida*) in Greenland. NAMMCO Scientific Publications 1:130–151.

Towns, L., A. E. Derocher, I. Stirling, N. J. Lunn, and D. Hedman. 2009. Spatial and temporal patterns of problem polar bears in Churchill, Manitoba. Polar Biology 32:1529–1537.

Vibe, C. 1968. Thule-eskimoernes isbjørnejagter [Polar bear hunting of the Thule-Eskimos]. Tidsskriftet Grønland 6:175-180. [In Danish].

Vibe, C. 1985. Miscellaneous information relating to polar bears in Greenland. Pages 44–57 *in* Polar Bears: Proceedings of the 8th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland.

Weber, D. S., T. Mandler, M. Dyck, P. J. Van Coeverden de Groot, D. S. Lee, and D. A. Clark. 2015. Unexpected and undesired conservation outcomes of wildlife trade bans - An emerging problem for stakeholders? Global Ecology and Conservation 3:389–400.

Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution doi:10.1002/ece3.2030.

Table 8.1. Summary of Canadian polar bear harvest from the Baffin Bay subpopulation for the period 1992/93 to 2013/14.

Harvest Season[1]	Total Harvest	Total Allowable Harvest[2]	Proportion Female (Total Harvest)[3]	Proportion of Harvest Made up of Independent Bears[4]	Proportion Female (Independent Bears Only)[5]
1992/93	62	n/a	0.36	0.94	0.35
1993/94	60	n/a	0.37	0.88	0.34
1994/95	60	64	0.33	0.92	0.35
1995/96	55	64	0.33	0.95	0.35
1996/97	60	64	0.41	0.88	0.42
1997/98	69	64	0.38	0.96	0.38
1998/99	49	64	0.35	0.98	0.36
1999/00	58	64	0.41	0.95	0.40
2000/01	61	64	0.28	0.98	0.28
2001/02	64	64	0.30	1.00	0.30
2002/03	62	64	0.26	0.97	0.22
2003/04	69	64	0.28	0.99	0.25
2004/05	101	105	0.37	0.98	0.38
2005/06	94	105	0.32	0.98	0.28
2006/07	89	105	0.36	0.99	0.38
2007/08	101	105	0.28	0.97	0.26
2008/09	103	105	0.39	0.98	0.39
2009/10	86	105	0.41	1.00	0.41
2010/11	94	95	0.33	0.98	0.34
2011/12	90	85	0.40	0.96	0.37
2012/13	74	75	0.47	0.92	0.45
2013/14	72	65	0.31	0.97	0.29
Mean	74.23		0.35	0.96	0.34

[1] The hunting season in Canada runs from July 1st to June 30th

[2] Total Allowable Harvest (TAH) is the regulated limit for all human-caused mortalities. Formerly referred to as the quota

[3] Excludes bears of unknown sex (which constituted a mean of *ca.* 1% of harvest annually for the period 1989/90 to 2013/14)

[4] Excludes bears of unknown dependency (which constituted a mean of *ca.* 4% of harvest annually for the period 1989/90 to 2013/14)

[5] Excludes bears of unknown sex and dependency

Table 8.2. Canadian and Greenlandic polar bear harvest from the Kane Basin subpopulation for the period 1992/93 to 2013/14.

Harvest Season[1]	Total Harvest	Total Allowable Harvest[2]	Harvest (Canada)	Harvest (Greenland)[3]	Proportion Female (Independent bears only)
1992/93	10	-	0	10	0.25
1993/94	11	-	1	10	0.23
1994/95	10	-	0	10	0.25
1995/96	10	-	0	10	0.25
1996/97	11	-	1	10	0.23
1997/98	12	-	2	10	0.30
1998/99	11	-	1	10	0.23
1999/00	6	-	0	6	0.25
2000/01	11	-	1	10	0.32
2001/02	12	-	0	12	0.25
2002/03	12	-	0	12	0.25
2003/04	9	-	0	9	0.25
2004/05	26	-	1	25	0.28
2005/06	9	-	1	8	0.40
2006/07	6	15	0	6	0.46
2007/08	7	13	0	7	0.46
2008/09	5	11	0	5	0.46
2009/10	3	11	1	2	0.30
2010/11	6	11	0	6	0.46
2011/12	6	11	0	6	0.46
2012/13	6	11	0	6	0.46
2013/14	5	11	0	5	0.46
Mean	9.3	11.8	0.4	8.9	0.33

[1] Greenland harvest data for a given calendar year were included in the harvest season ending in that calendar year

[2] No quota in Greenland prior to 2006 thus total quota only presented for period 2006-2014

[3] Annual harvest in Greenland (1993-1999) was estimated from an interview survey conducted in 1989 (Rosing-Asvid and Born 1990). Information on annual catch was based on information in Piniarneq during 2000-2005 and thereafter on the special reporting under the quota system (see Appendix D)

Table 8.3. Means of transportation used during 445 polar bears catches in West and Northwest Greenland from Subareas 1 and 2 (i.e., the Kane Basin subpopulation) and Subareas 3-6 (the Baffin Bay subpopulation) during 2006-2014.

Subarea	Dog sled	Skiff	Boat	Total	% Dog sled	% skiff and boat
1+2[1]	30	6	3	39	76.9	23.1
3	215	8	0	223	96.4	3.6
4	34	98	0	132	25.8	74.2
5	4	25	0	29	13.8	86.2
6	0	20	2	22	0.0	100.0
Total	283	157	5	445	63.6	36.4

[1] Only 2 catches reported from Subarea 2

Table 8.4. Sex and age composition of the Greenland catch of polar bears from the BB subpopulation (1982-2014) based on various sources.

Period	% Females[1]	% Males[1]	Independent F:M ratio	% Cubs	F (n)	Cubs (n)	M (n)	Total (n)	Source	Source
1982-1989	29.9	57.7	1:1.9	12.4	41	17	79	137	Interviews 1989-90	Rosing-Asvid and Born 1990, Rosing-Asvid 2000
1988-1996	33.3	46.0	1:1.4	20.6	21	13	29	63	Samples	Rosing-Asvid[2] 2002: table 3
1991-2005	26.8	50.6	1:1.9	22.6	95	80	179	354	Interviews 2006	Born unpublished, and Born *et al.* 2011
1982-2005	28.3	51.8	1:1.9	19.9	157	110	287	554		
2006-2014	32.2	67.8	1:2.1	0.0	192	n.a.[3]	404	596	Special Reporting Forms	2006-13:DFHA[4] *in litt. (*2014), 2014:DFHA *in litt.* (2015)
2012-2013	45.5	54.5	1:1.2	0.0	25	n.a.	30	55	Samples	Samples collected by hunters, GINR unpublished
2006-2014	32.5	67.5	1:2.1	0.0	193	n.a.	401	594		

[1] Percentage of total annual catch. Adult and subadult females and adult and subadult males = individuals 3 years old and older (*cf.* Taylor et al. 1987)

[2] A comparison of figure 26 and table 3 in Rosing-Asvid (2002) shows that the percentages of independent bears given in his table 3 are based on 2+ years old bears

[3] Since 2006 it has been prohibited to kill dependent cubs irrespective of their age

[4] DFHA = Department of Fisheries, Hunting and Agriculture (Nuuk)

Table 8.5. Sex and age composition of the Greenland catch of polar bears from the KB subpopulation (1982-2014) based on various sources.

Period	% Females[1]	% Males[1]	Independent F:M ratio	% Cubs	F (n)	Cubs (n)	M (n)	Total (n)	Source	Source
1982-1989	29.2	63.1	1:2.2	7.7	19	5	41	65	Interviews 1989-90	Rosing-Asvid and Born 1990, Rosing-Asvid 2000
1988-1996	36.7	56.7	1:1.5	6.7	11	2	17	30	Samples	Rosing-Asvid 2002: table 3
1991-2005	17.9	73.1	1:4.1	9.0	14	7	57	78	Interviews 2006	Born unpublished, and Born et al. 2011
1982-2005	25.4	66.5	1:2.6	8.1	44	14	115	173		
2006-2014	44.9	55.1	1:1.2	0.0	22	n.a.[2]	27	49	Special Reporting Forms	2006-13:DFHA[3] *in litt. (*2014), 2014:DFHA *in litt.* (2015)

[1] Independent females as percentage of total annual catch

[2] Since 2006 it has been prohibited to kill dependent cubs irrespective of their age

[3] DFHA = Department of Fisheries, Hunting and Agriculture (Nuuk)

Table 8.6. The combined Canadian and Greenlandic polar bear annual harvest from the Baffin Bay subpopulation for the period 1992/93 to 2013/14. Annual average proportion of independent female polar bears is shown.

Harvest Season[1]	Total Harvest	Total Allowable Harvest[2]	Harvest (Canada)	Harvest (Greenland)[3]	Proportion Female (Independent bears only)
1992/93	134		62	72	0.35
1993/94	120		60	60	0.35
1994/95	124		60	64	0.35
1995/96	122		55	67	0.35
1996/97	139		60	79	0.38
1997/98	165		69	96	0.36
1998/99	146		49	97	0.36
1999/00	126		58	68	0.37
2000/01	158		61	97	0.33
2001/02	182		64	118	0.33
2002/03	268		62	206	0.32
2003/04	225		69	156	0.32
2004/05	236		101	135	0.36
2005/06	173		94	79	0.30
2006/07	165	178	89	76	0.35
2007/08	176	176	101	75	0.29
2008/09	174	173	103	71	0.36
2009/10	150	171	86	64	0.37
2010/11	165	160	94	71	0.33
2011/12	165	152	90	75	0.35
2012/13	137	142	74	63	0.39
2013/14	146	132	72	74	0.31
Mean	163	161	74.23	89.00	0.35

[1] Greenland harvest data for a given calendar year were included in the harvest season ending in that calendar year

[2] No quota in Greenland prior to 2006 thus total quota only presented for period 2006-2014

[3] Harvest in Greenland is estimated from reported harvest in west Greenland and the estimated portion of this harvest that occurs in Baffin Bay

Table 8.7. Genotyping of bears harvested in Canada and Greenland, 2011-2014. Data presented as the percentage of individuals in the reported harvest that were sampled and genotyped. Total number of individuals reported as harvested is presented in parenthesis. Data are organized by Nunavut hunting seasons which run from July1 to June 30.

	2011/12		2012/13		2013/14	
Subpopulation	Canada	Greenland	Canada	Greenland	Canada	Greenland
BB	75 (85)	64 (69)	90 (78)	42 (60)	100 (67)	n/a[1]
DS	74 (38)	-	92 (60)	-	-	-
LS	75 (92)	-	91 (92)	-	-	-
KB	0 (0)	50 (6)	0 (0)	25 (4)	-	-
FB	86 (107)	-	91 (109)	-	-	-
NW	0 (0)	-	33 (3)	-	-	-

---

[1] Greenland harvest during this season occurred after mark-recapture sampling ceased and is therefore not reported.

Table 8.8. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Data from Canada (Nunavut) and Greenland harvest.

	Genetic Sex	
Reported Sex	Male	Female
Male	156	19
Female	6	97

Table 8.9. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Greenland harvest only.

	Genetic Sex	
Reported Sex	Male	Female
Male	29	15
Female	4	24

Table 8.10. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Data from Canada (Nunavut) harvest only.

	Genetic Sex	
Reported Sex	Male	Female
Male	127	4
Female	2	73

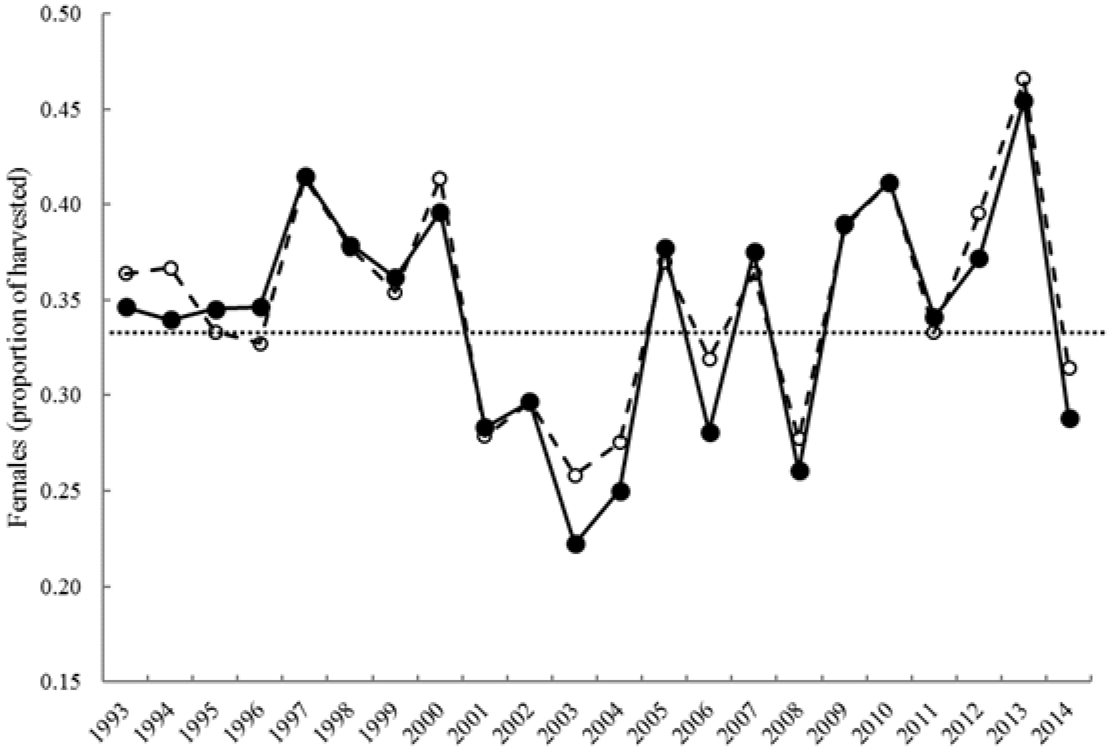


Figure 8.1. Harvest of female polar bear as a proportion of total Canadian harvest from Baffin Bay. All females (dashed line) and independent females only (solid line). Dotted line indicates target sex ratio (0.33).

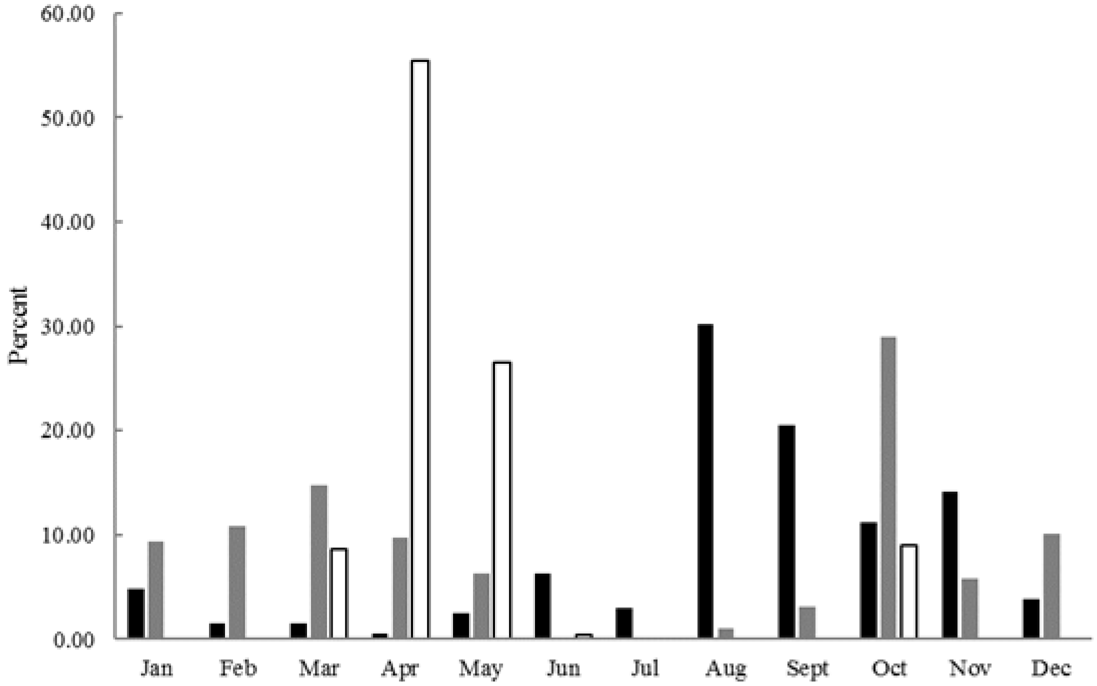


Figure 8.2. Monthly distribution of polar bear harvesting in Baffin Bay by type of harvest (1992/93-2013/14). Regular hunts (grey), defense-of-life-and-property kills (black) and sport hunts (white). Bars represent the percentage of hunting of a given type that occurred each month.

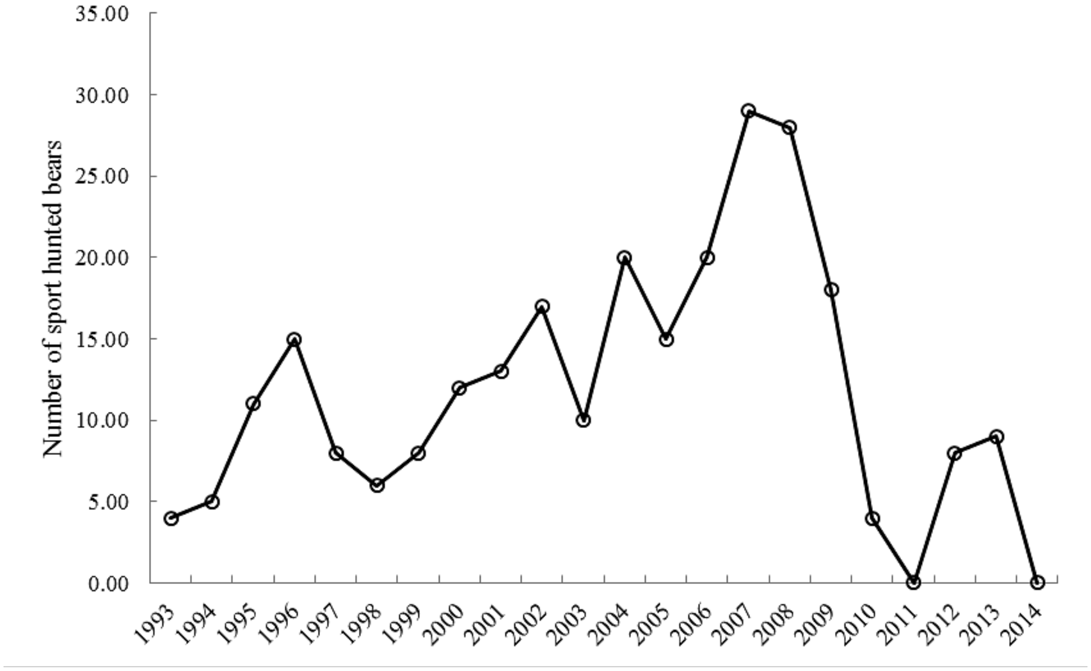


Figure 8.3. Numbers of polar bear taken by sport hunters in Baffin Bay, Canada.

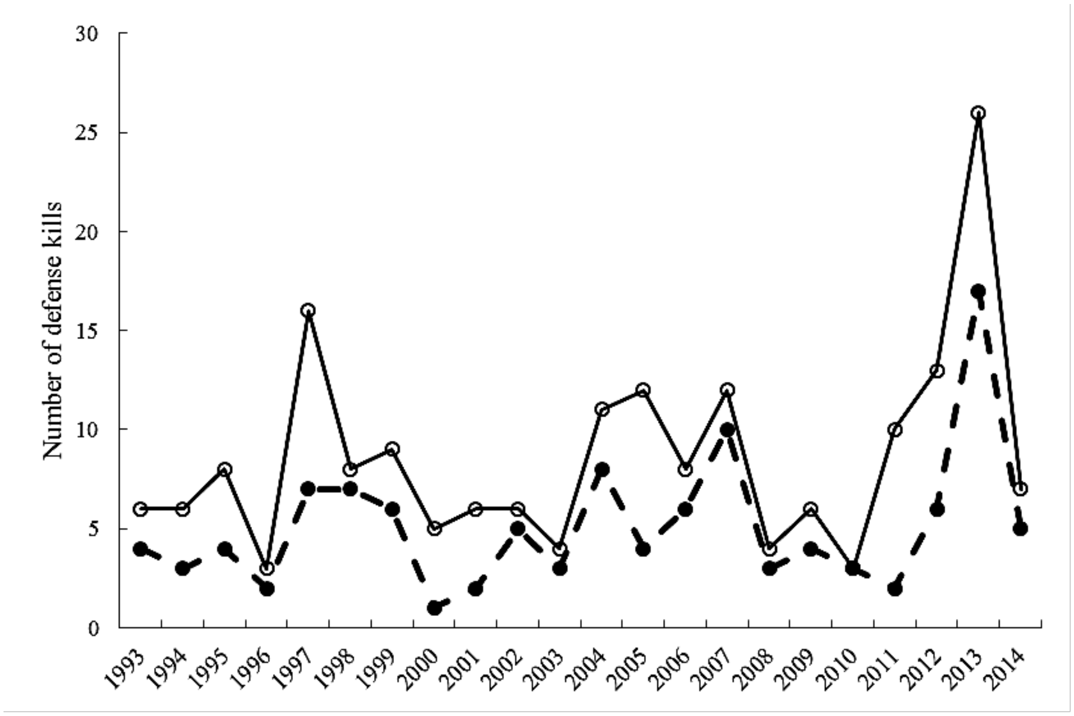


Figure 8.4. Number of polar bears harvested in defense-of-life-and-property (DLP) in Baffin Bay, Canada. Total DLPs per year (solid line) and DLPs of independent bears during Aug-November each year (dashed line).

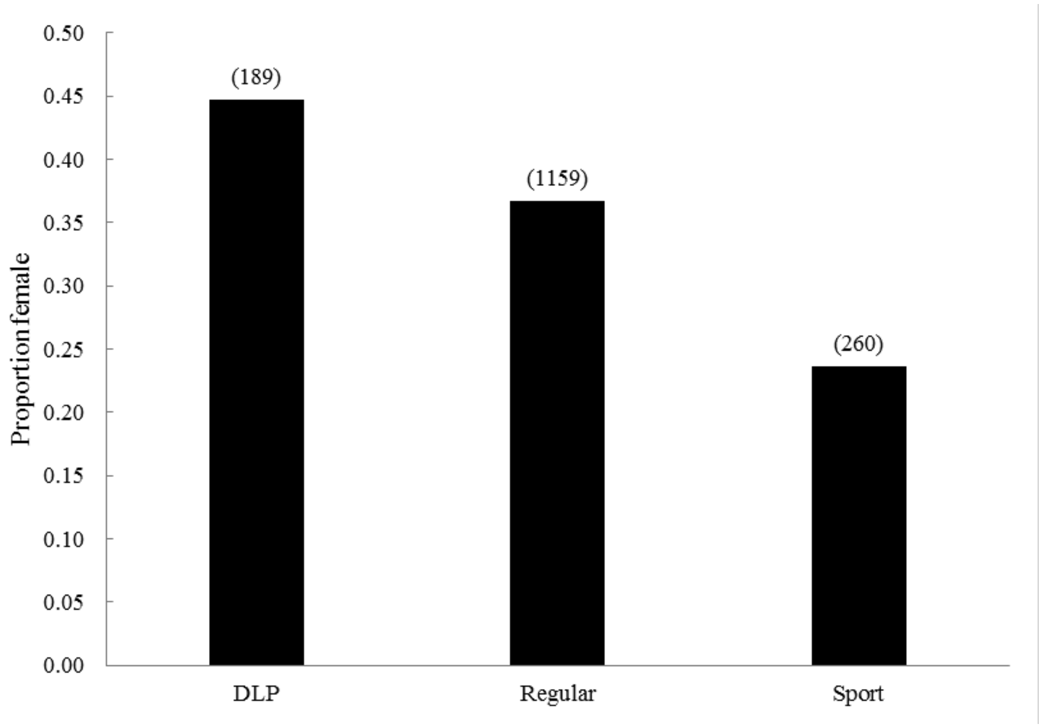


Figure 8.5. Sex composition of the three main types of polar bear harvesting in Baffin Bay, Canada, for the period 1992/93 to 2013/14. Bars represent the proportion of harvest that was female. Data are for independent bears only. Sample sizes in parentheses.

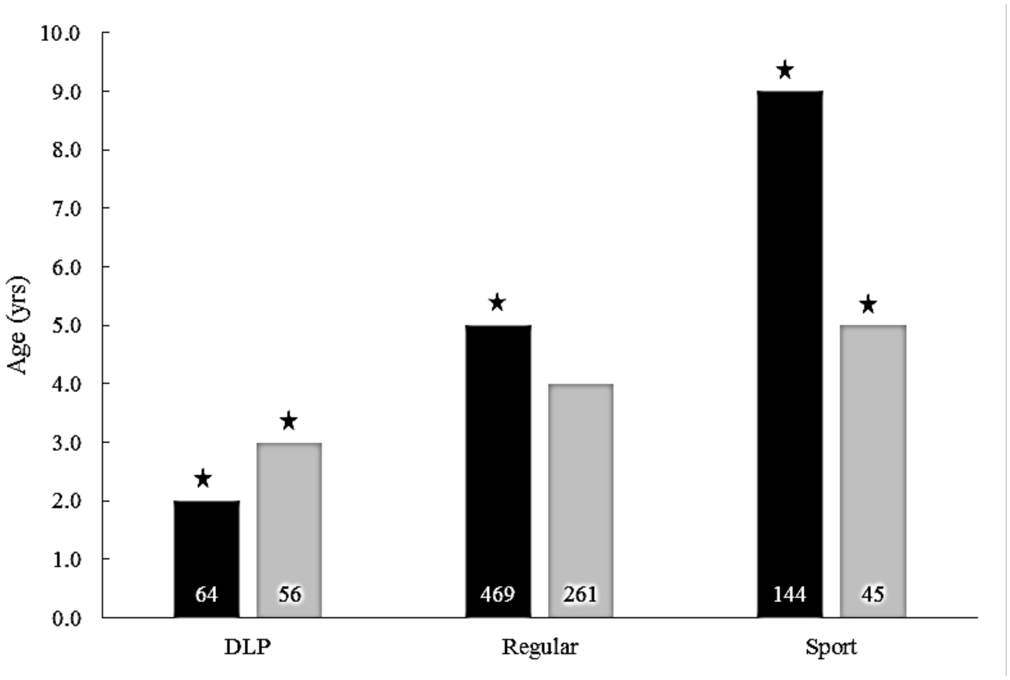
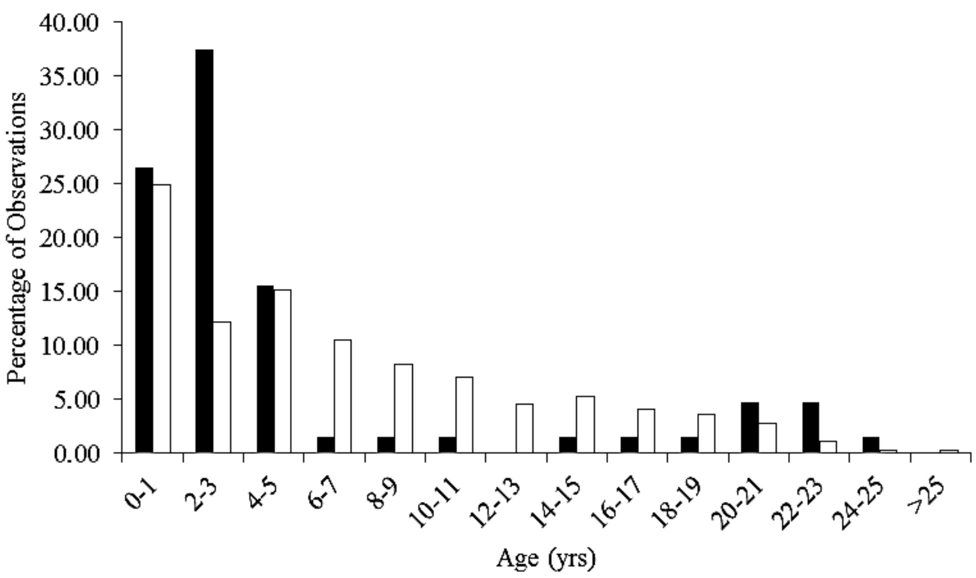


Figure 8.6. Median age of male (black) and female (grey) bears harvested for defense-of-life-and-property (DLP), regular (subsistence) and sport hunting purposes in Baffin Bay, Canada, 1993-2010. Within sexes significant differences denoted by * (Based on Mann-Whitney *U* test with Bonferroni correction for multiple comparisons, two-tailed, alpha = 0.05). Sample sizes within bars.

(a)



(b)

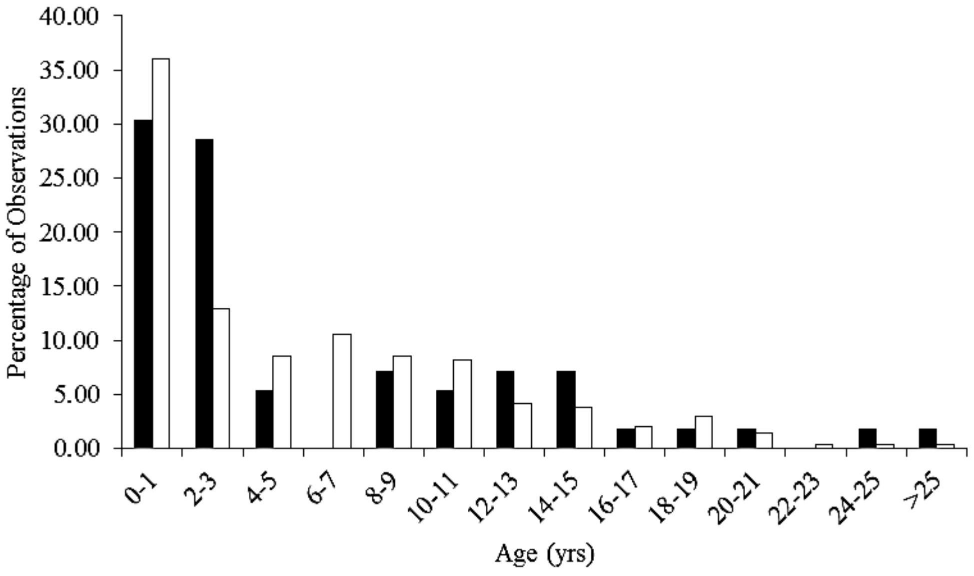


Figure 8.7. The age distribution of (a) 64 male and (b) 56 female bears harvested in defense-of-life-and-property (black) in Baffin Bay (Canada), 1993-2010, relative to the age distribution of 778 bears captured during mark-recapture sampling (white), 1993-1997 (GN unpublished data).

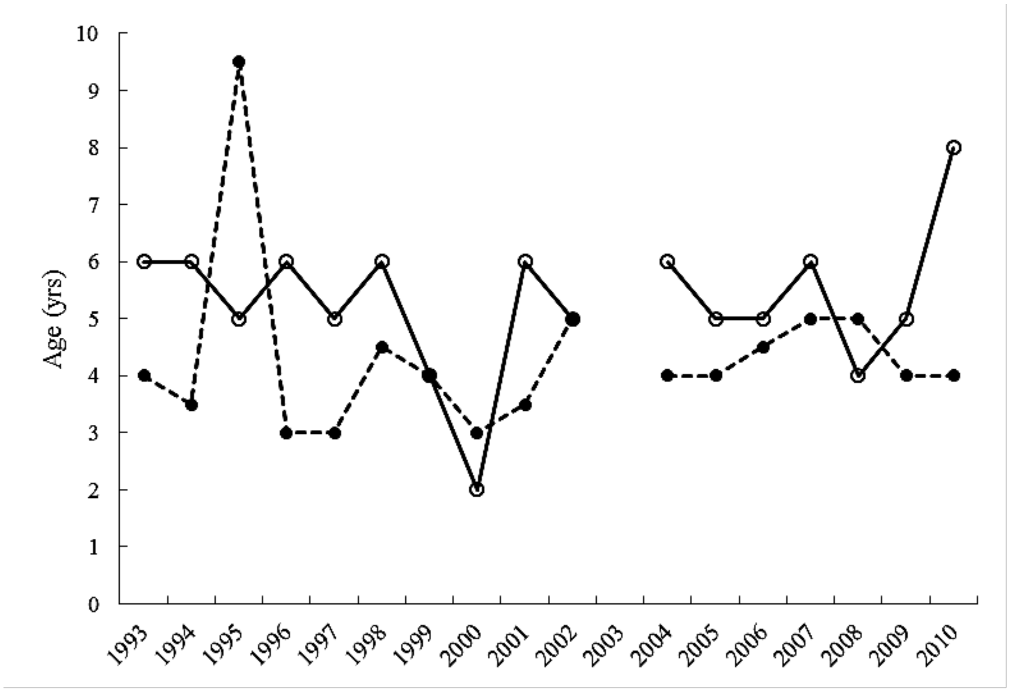


Figure 8.8. Median age of harvested male (solid line) and female (dashed line) polar bears in Baffin Bay, Canada. Regular hunts only.

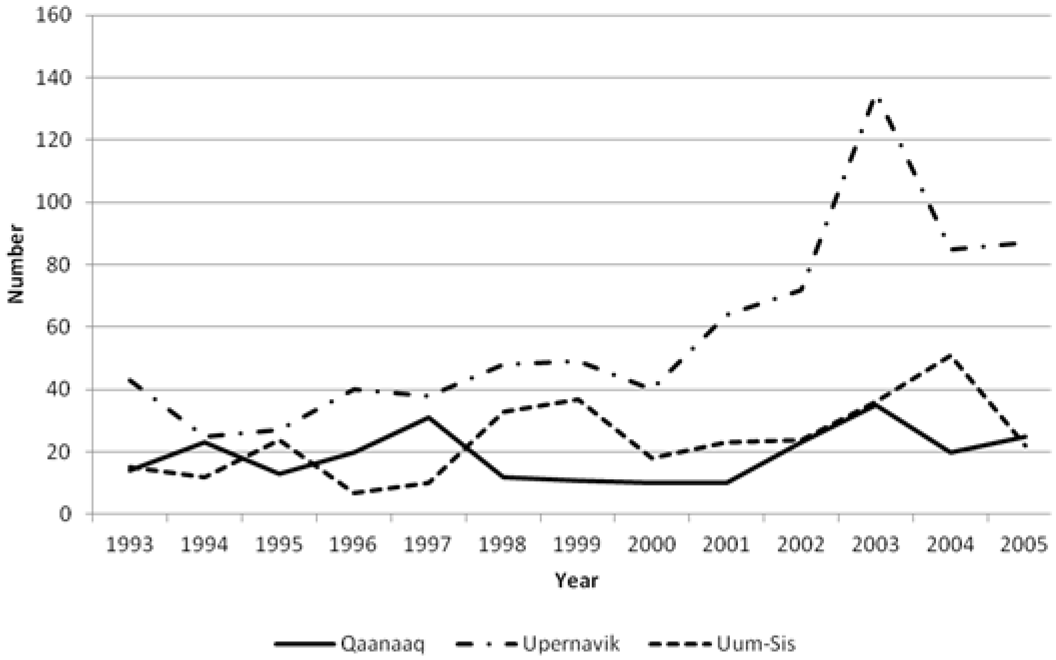


Figure 8.9. The Greenland catch of polar bears from the Baffin Bay subpopulation reported in Piniarneq (1993-2005). The catch is shown for three different regions: (1) The Qaanaaq region representing polar bears taken between 74° 35´ N and 76° 20´ N (i.e., the Melville Bay region sensu latu), (2) the Upernavik area between *ca.* 71° 30´ N and 74° 35´ N, and (3) the areas between Uummannaq and Sisimiut between 66° 55´ N and 71° 30´ N.

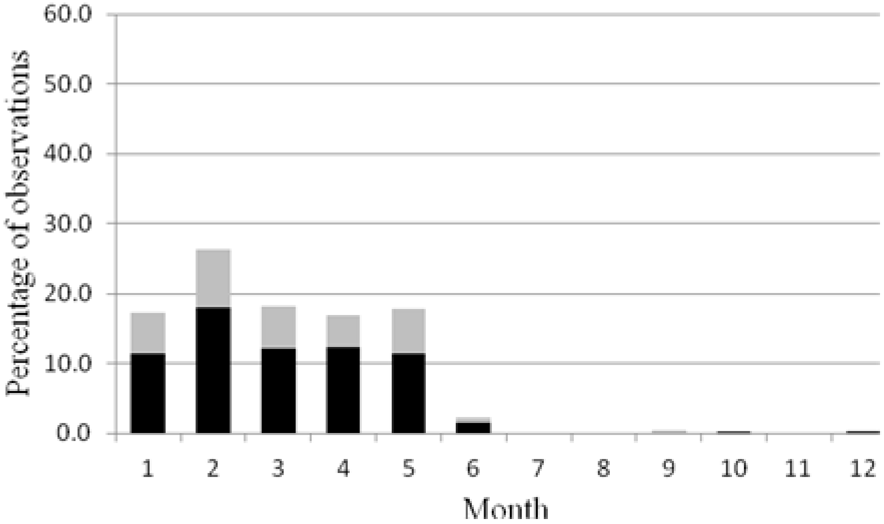


Figure 8.10. Seasonal distribution of catches of polar bears (n = 594; 587 legal and 7 illegal catches) taken in NW and W Greenland from the Baffin Bay subpopulation based on reports from the licensed hunters (2006-2014). Black = Males. Grey = Females. Source: DFHA (in litt 2014 and 2015).

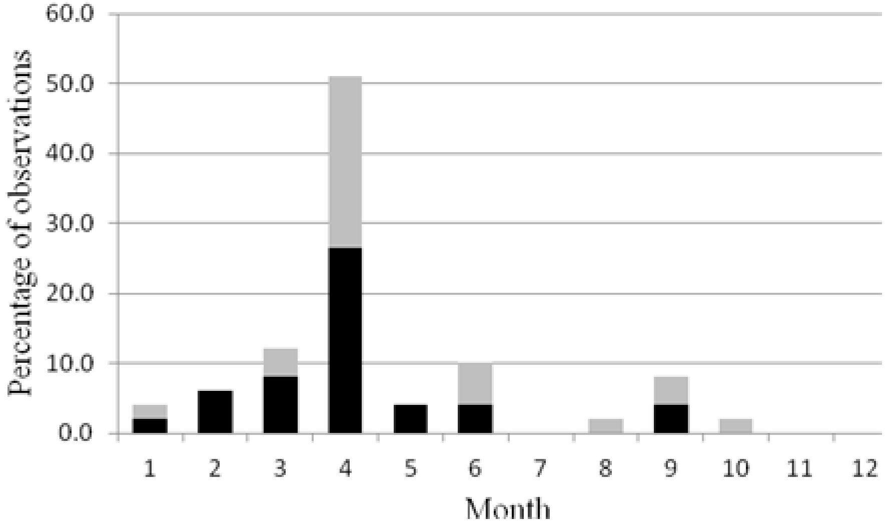


Figure 8.11. Seasonal distribution of catches of polar bears (n = 49; 48 legal and 1 illegal catches) taken in NW Greenland from the Kane Basin subpopulation based on reports from the licensed hunters (2006-2014). Black = Males. Grey = Females. Source: DFHA (*in litt.* 2014 and 2015).

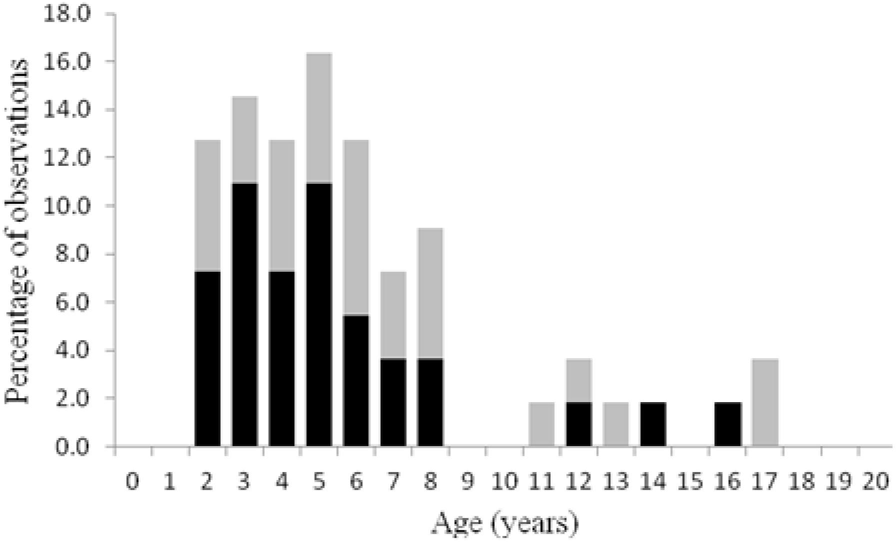


Figure 8.12. Age distribution of a total of 55 polar bears (Grey = Females: n = 25; Black = Males: n = 30) that were taken from the BB subpopulation by Greenland subsistence hunters in 2012 (n = 33) and 2013 (n = 22). Sex was determined genetically.

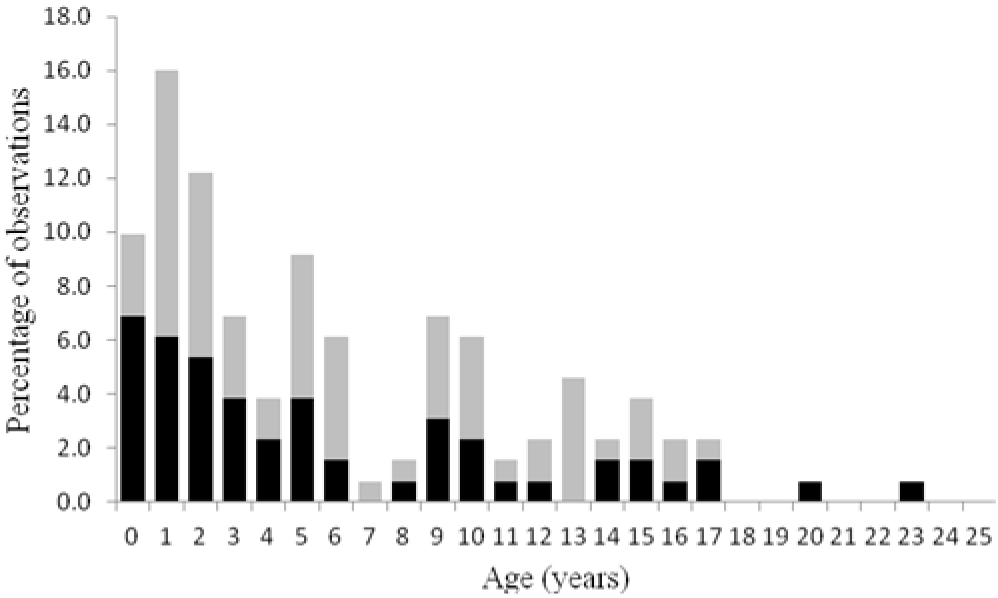


Figure 8.13. Age distribution of 131 polar bears (Grey = Females: n=73; Black = Males: n=58) that were live captured in NW Greenland north of *ca.* 70° N from the BB subpopulation during spring 2009-2013 in connection with the present study (GINR unpublished data).

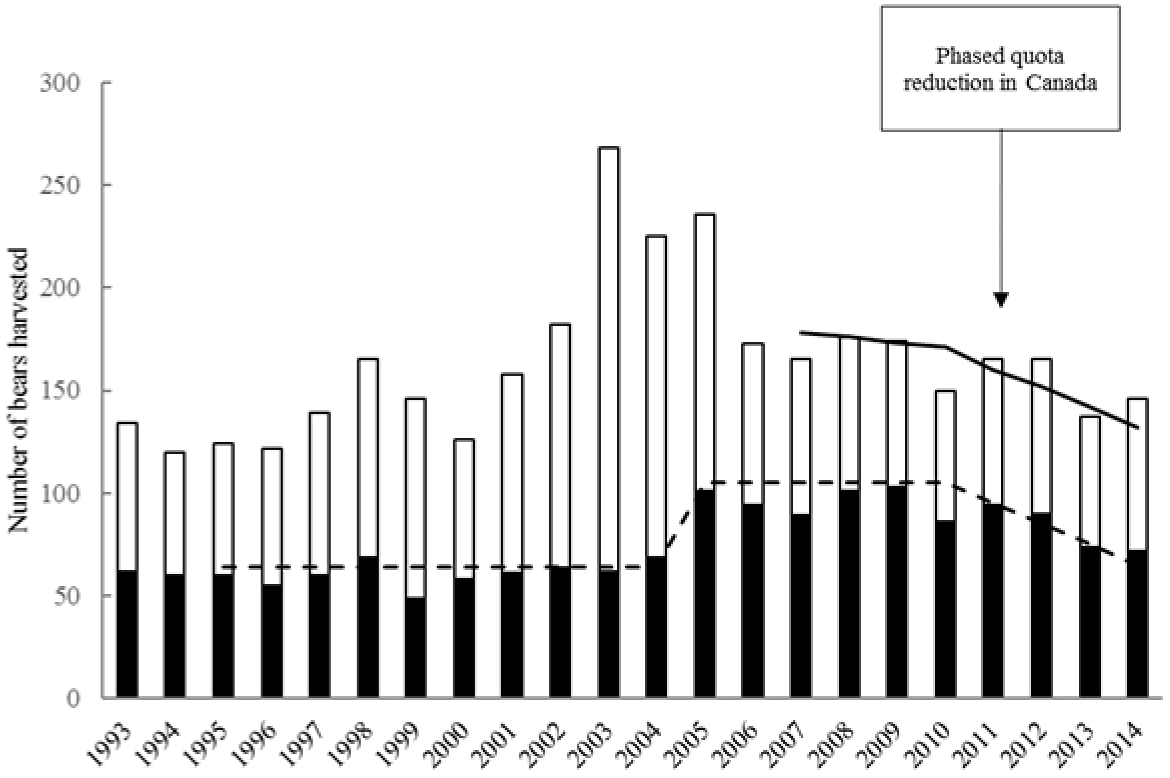
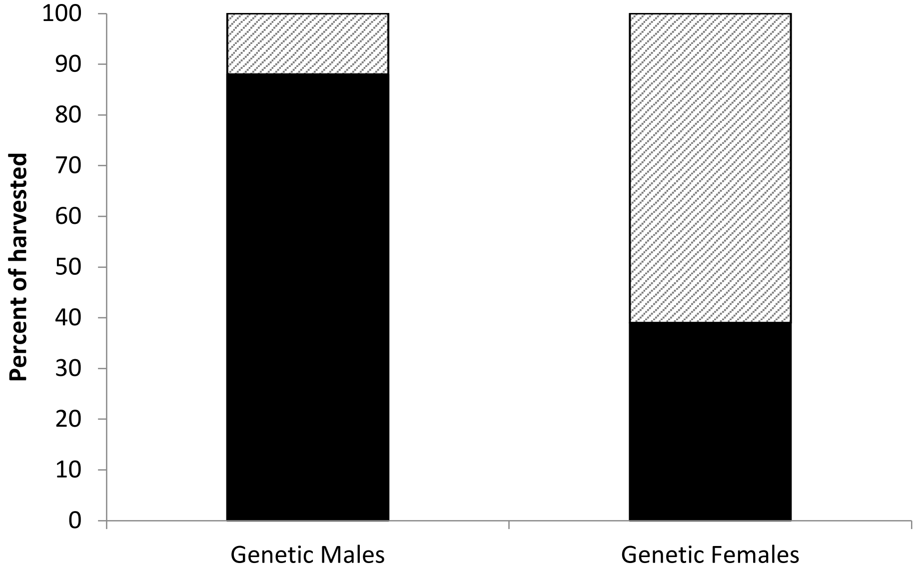


Figure 8.14. Combined Canadian (black) and Greenlandic (white) harvest of polar bears from the Baffin Bay subpopulation. Total allowable harvest in Canada (dashed line) and Canada-Greenland combined total permitted harvest (solid line) levels are also shown.

(a)



(b)

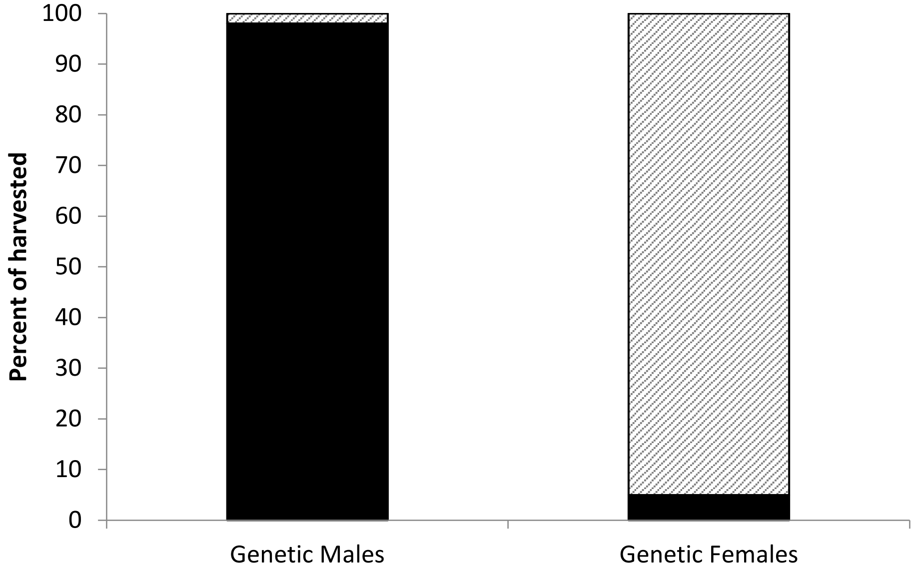


Figure 8.15. Accuracy of gender reporting for polar bears harvested in Baffin Bay and Kane Basin (2011-14) in (a) Greenland and (b) Nunavut. Bears reported as males and females are indicated in black and hatched, respectively

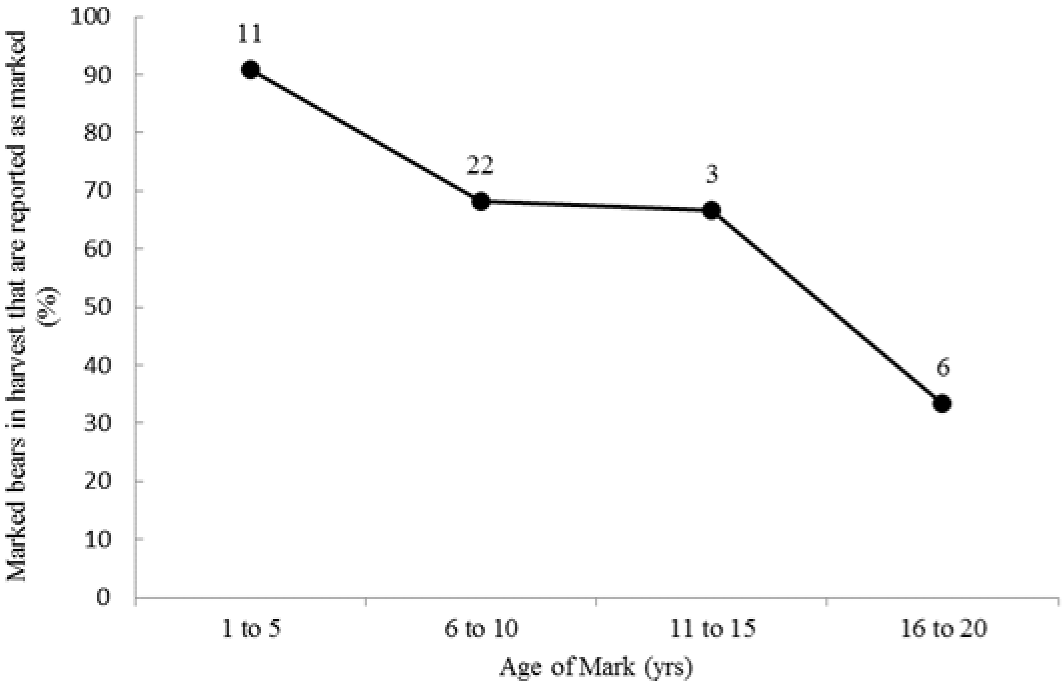


Figure 8.16. Accuracy of reporting of physically marked (tag and tattooed) bears in the harvest in Canada and Greenland. Data are for bears physically marked in Baffin Bay (1990-97) and Davis Strait (2005-07) that were recovered in the harvest 2011-14. Sample sizes above points.

CHAPTER 9

HABITAT AND HABITAT USE IN KANE BASIN

KEY FINDINGS

- The annual cycle of sea-ice habitat in KB has shifted from a largely year-round ice platform (>30% coverage in summer) to a cycle that resembles the seasonal ice ecoregion with complete melt-out in summer (<5% coverage).
- Sea ice is retreating earlier in Kane Basin spring by 7 days/decade, and advancing later in fall by 5-6 days/decade and length of summer (number of days from retreat to advance) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 5-6 percent/decade.
- The KB subpopulation has responded to changing sea-ice conditions with broad movement and habitat use patterns that are more similar to those of bears in seasonal sea-ice ecoregions (e.g., expanded seasonal home ranges, see Chapter 2 and use of lower sea-ice concentrations in summer and fall).
- Four-day movement rates in KB are lower than those in BB and have a less pronounced seasonal cycle. There are no significant differences in movement rates between the 1990s and 2000s except in October where rates were higher.
- Land use in KB during summer remains intermittent because some sea ice remains in fjords and coastal areas. No on-land arrival and departure dates could be determined from satellite telemetry.
- Three maternity dens were found in KB in the 2000s. All were located on Ellesmere Island. There was no significant difference in maternity denning duration, entry dates, or exit dates between the 1990s and 2000s.
- Overall, the movement (Chapter 2) and habitat results combined with reproductive metrics (Chapter 11) and body condition (Chapter 12), indicate that KB bears are experiencing more seasonal sea-ice ecoregion-like conditions, which since the 1990s may have increased overall biological productivity of the area.

9.1. Introduction

The Kane Basin (KB) subpopulation of polar bears occurs between the Canadian Arctic Archipelago and Northwest Greenland, referred to by some as the Arctic Archipelago ecoregion, historically characterized by year-round sea-ice habitat in islands in the Canadian High Arctic and in the Kane Basin region in Northwest Greenland (Hamilton et al. 2014). Ice in this ecoregion is largely multi-year, except in Kane Basin where much of the ice is annual though partially present year-round. In contrast to the Canadian Arctic Archipelago, the Kane Basin region is also characterized and influenced by influx of multi-year ice flowing to the area from the Arctic Ocean. In these areas, sea ice remains along coastal areas in summer providing temporally-stable hunting opportunities for polar bears during summer. This is in contrast to the seasonal ecoregion (Baffin Bay), where sea ice disappears almost completely in summer and bears are forced onshore (Amstrup et al. 2008, this study).

Earlier studies comparing movements and habitat selection of polar bears in the archipelago and seasonal ecoregions documented differences in the influence of sea-ice regimes on movements and habitat use. Ferguson et al. (2000) showed that bears in the archipelago regions (including KB) are strongly influenced by the temporal and spatial distribution of land fast ice around islands providing a consistent habitat across the annual cycle, though possibly lower density year-round prey base. This was in contrast to seasonal ecoregion bears that are strongly influenced by the availability of seasonal sea ice, resulting in a more productive but temporally limited the duration of feeding. Bears in these two ecoregions also demonstrate contrasting movement patterns: bears in the archipelago region (e.g., KB) tended to have smaller home ranges and greater irregularity in movement patterns because they utilize small coastal areas around the complex land masses (or in case of the Kane Basin region fast ice in fjords or fields of pack ice in Nares Strait-Kane Basin), whereas bears in the seasonal sea-ice region (e.g., BB) had large home range sizes (Ferguson et al. 1998) and more regular movement patterns associated with the seasonal growth and recession of sea ice.

Recent work examining Global Climate Model projections of sea-ice habitat for polar bears in the Canadian Arctic Archipelago suggest that conditions will shift towards a seasonal-sea-ice ecoregion before the end of the 21st century (Hamilton et al. 2014). This shift is expected to initially increase productivity in the archipelago system, as thicker ice is replaced by thinner annual ice increasing production blooms and prey platforms for ice seals. Such a change would be expected to be associated with changes in movement patterns and habitat use similar to that exhibited by bears in the seasonal ecoregion. To date, no studies have quantified such behavioral changes. In this chapter we document changes in sea-ice habitat of KB over the satellite record, and quantify change in habitat use using satellite telemetry data collected from collared adult females in the 1990s and 2000s.

9.2. Methods

We refer to “Kane Basin” as the region within the boundaries of the Kane Basin (KB) polar bear management unit (PBSG 2010; Figure 1.1.) that encompasses the northern part of Smith Sound, Nares Strait and Kane Basin and the southern part of Kennedy Channel and adjacent fjords on eastern Ellesmere Island and in NW Greenland.

KB polar bears were captured and tagged between mid-April and early May 2012-2013 (Figure 9.1). Field operations were based out of the Alexandra Fjord station on Ellesmere Island. A total of 34 bears were furnished with satellite-transmitters in KB. Twenty were adult females who received a satellite collar (Table 9.1) and 14 were adult males or subadults (given satellite radio ear tags). Data from adult females were combined with a historical data set from 12 adult females collared between 1992 and 1994 on the west side of KB in the fjords and fast ice (Taylor et al. 2001). Only bears captured within the KB subpopulation boundaries (PBSG 2010) were included in the analysis. The eastern side of the Kane Basin region was surveyed during 1994-1997 but no female polar bears for collar deployment were found in these areas (Taylor et al. 2001). Hence, radio collars were only deployed on the west side of KB in the 1990s (ibid.). Given the different distribution of collar deployments between decades we tested for differences in area use and mean latitude using only bears captured in West KB in the 2000s and found no differences (see Chapter 2). Therefore, all bears from the 1990s were compared to all bears in the 2000s in habitat models. Bears in all decades largely utilized KB, however in both decades bears moved from KB into BB, LS and in the 2000s out of KB into the Arctic Basin.

Sea ice

Data sources and methods for sea-ice analysis in KB are the same as those described for BB (see Chapter 4 and Stern and Laidre 2016). The entire KB region, as defined by PBSG (Figure 1.1), was used for the sea-ice habitat analyses. The area was roughly divided into two distinct parts. The northern part, or Kane Basin proper, is bounded on the south by Smith Sound, consisting of almost entirely of shallow (< 300 m) water. The southern part contains the northern part of the North Water Polynya, and is bounded on the south by roughly 77° N latitude where it adjoins northern Baffin Bay. The southern part of Kane Basin consists mostly of deep (> 300 m) water on the Ellesmere Island side and shallow water on the Greenland side.

Habitat Use Analyses

Methods for KB movement rates, habitat covariates, and multivariate RSF modeling are the same as those described in Chapter 4 for BB. In the multivariate terrestrial RSF we did not include the variable pertaining to the distance to the smoothed Baffin Island coastline. In the multivariate sea-ice RSF we excluded the covariates representing the distance to the 15 and 50% sea-ice concentration because in some months in KB the distance resulted in unrealistic potential movements of bears.

We attempted to quantify the dates of arrival and departure on land in KB as described for BB in Chapter 4. This analysis was confounded by the fact that KB bears have access to sea ice much of the summer. Thus KB bears used land intermittently and it was not possible to quantify a specific date where bears arrived on shore and did not leave. There were no potential swimming events identified in KB. Maternity denning analyses were conducted with the same methods as described in Chapter 4 for BB.

9.3 Results and Discussion

Sea-ice habitat

Kane Basin consisted of 81 SSMI sea ice grid cells (53 × 103 km^2); 68% had a mean depth < 300 m, 32% had a mean depth > 300 m. The seasonal cycle of the sea ice in KB has changed dramatically since the 1990s (Figure 9.2). In the 1990 sea ice did not disappear from KB and in summer months >50% of KB was ice covered. In the 2000s, there has been greater extent of sea-ice loss and KB reaches ~5% coverage in summer. The sea-ice loss has been most pronounced from May and through the late fall, and there are few differences in sea-ice coverage between January and April. Trends in the four sea-ice metrics (described in Chapter 4) are provided in Table 9.2. All trends are statistically significant and show a loss of sea-ice habitat. In Kane Basin as a whole, sea ice is retreating earlier in spring by 7 days/decade, and advancing later in fall by 5-6 days/decade (Figure 9.3). The length of summer (number of days from retreat to advance) is increasing by 12 days/decade, and the mean sea-ice concentration during June-

October is decreasing by 5-6 percent/decade (Figure 9.4ab). The trend in the date of spring sea-ice retreat is apparently stronger for the shallow depths of KB than for all depths. The trend in the date of fall sea-ice advance is the same for both depth categories. The downward trend in the June-October sea-ice concentration is apparently stronger for the shallow depths.

The year-to-year variability in sea-ice metrics for Kane Basin was larger than for Baffin Bay, i.e., the scatter about the trend lines was larger. There are several reasons for this variability: (1) the North Water Polynya is an area of dynamic sea-ice activity that affects Kane Basin; (2) there is typically an “ice arch” north of Kane Basin that determines whether ice lingers in the basin (arch intact) or is flushed out (arch collapses); and (3) Baffin Bay is much larger than Kane Basin and so is less affected by such relatively small-scale phenomena as (1) and (2).

The trend in the annual number of ice-covered days in Kane Basin is between −5 and −15 days/decade for most of the areas with shallow depths (Table 9.2). For the southern portion of Kane Basin, the trend is steeper than −15 days/decade on the Greenland side and there is almost no trend on the deeper Ellesmere Island side. Thus, the pattern of extreme sea-ice loss in Baffin Bay along the coast of Greenland (see Chapter 4) extends northward into the southeast portion of Kane Basin.

Spring sea-ice melt in the Kane Basin region begins in May in the North Water Polyna, which generally becomes ice-free by July. Kane Basin proper, to the north, generally holds some sea ice all summer. Figure 4.5 shows that on July 15, Kane Basin proper is almost always ice-covered, often with 50% or more sea-ice concentration. The year 2009 was exceptional, when all the ice in Kane Basin was swept out in May and June. In October, sea ice advances from north to south through Kane Basin, but the date of advance is generally trending later (Table 9.2 and Figure 9.2).

Movement rates – In Kane Basin, mean monthly movement rates for adult females were overall lower than in Baffin Bay. In KB in the 1990s, mean monthly movement rates ranged from a low of 3.4 km/day (in August) to a high of 9.4 km/day (in February) (Figure 9.5, Table 9.3). Rates for adult females in the 2000s were similar and ranged from 4.7 km/day (in Septembe) to 6.94 km/day (in November) (Figure 9.5, Table 9.3). Compared to Baffin Bay, there was a substantially less pronounced cycle to movement rates over the year in KB. There were no significant differences in movement rates between decades except in October in the 2000s where rates were higher than the 1990s (Table 9.3).

RSF sea-ice models – The sample sizes of adult females in KB in the 1990s and 2000s were smaller than in BB (12 and 20 bears, respectively). Collars deployed between 2012 and 2013 were removed in April 2014 thus tracking durations in the later period were also shorter. We examined univariate relationships for multiple habitat covariates over the annual cycle of sea ice (Figure 9.7 and 9.8). KB bears in the 1990 used similar sea-ice concentrations as bears in the 2000s between January and May (Figure 9.7). Starting in late spring (June) and continuing through December, KB bears in the 2000s used significantly lower sea-ice concentrations than in the 1990s. This was most pronounced in August-October. Distances from bears to the 15% or 50% sea-ice concentration thresholds varied widely across the annual cycle and were similar between decades, though in the 2000s bears were significantly closer to the 50% sea-ice edge in March and April. There were no large differences in bears’ distance to land in either decade, other than bears being closer to land from October-December in the 2000s.

The multivariate RSF model in winter demonstrated adult female polar bears in the 1990s had a strong preference for higher ice concentrations. This preference was not present in the

2000s. In both decades bears had a similar strength of preference for the distance to the 300 m depth contour and preferred shallower depths (more strongly and significantly in the 2000s) (Table 9.4). The multivariate RSF model for spring showed that bears also had a strong significant preference for higher ice concentrations in the 1990s (Table 9.5). The preference was reduced in the 2000s but there was no significant difference between decades. In spring in the 1990s, bears were farther from the shelf break (300 m contour), whereas in the 2000s they were closer to 300 m and this change was significant between decades. There was no preference for depth in either decade in spring. In both decades, there was a preference not to move to land, but this was significantly stronger in the 2000s.

RSF terrestrial models – Adult female use of land was intermittent in KB, thus land use models reflect use of land largely near the shoreline as bears moved on and off sea ice (Figure 9.9, Figure 9.10). The terrestrial models demonstrated that KB bears preferred lower elevations, a preference which has significantly increased in the 2000s. Bears tend to avoid steep slopes in both decades and were significantly less likely to move to sea ice once they were on land (Table 9.6).

Arrival and departure dates – KB is part of the Archipelago ecoregion, which in contrast to the seasonal ice ecoregion, historically does not melt out completely each year. Bears in KB exhibit fundamental differences in their habitat use because of the availability of sea ice between systems. In general, KB bears had access to sea ice for most of the summer, especially in the 1990s though this has been significantly reduced in the 2000s. Some bears utilized fjord ice for most of the summer and never arrived on land, while others spent intermittent time on land. Overall patterns of land use among individuals were not consistent and thus it was not possible to quantify on-land arrival and departure dates. No long-distance swimming events were observed, though one of the swimming events recorded for BB in July 2010 resulted in a BB collared bear arriving on Ellesmere Island after a long distance swim from offshore pack ice in Northern BB.

Kane Basin denning – Nine dens were found from 2012 to 2015 in KB: three maternity dens (Figure 9.11) and six shelter dens (Figure 9.12). In the 1990s data, Ferguson et al. (1997) also found nine dens, of which three were maternity dens and six were shelter dens (Table 9.7). All dens were on land with the exception of one 1990s shelter den that was located on landfast ice nine kilometers from the shore of Ellesmere Island. Most of the dens were located on Ellesmere Island except for three dens on Devon Island. None of the adult females from KB denned on Greenland. The minimum latitude for the 1990s dens was 77.94° N, and 77.04° N for the 2000s dens.

There was no significant difference in maternity denning duration ($p = 1$) (Table 9.8, Figure 9.13), entry dates ($P = 0.6$) and exit dates ($P = 1$) (Figure 9.14). Only four of the KB bears in the 2000s provided useable temperature data for inferring exit dates and no temperature data were available from the 1990s. There was no significant correlation between latitude and maternity den entry dates ($\tau = 0.138$, $P = 0.848$) or duration ($\tau = 0.2$, $P = 0.707$). The median first date on land for the n=3 pregnant females in the 1990s was 18 September (SD = 31 days) and in the 2000s was 23 August (SD = 20.8 days; Figure 9.15). The difference between the two time periods was not significant despite the median FDOLs being 27 days apart. The sample size was small and there was considerable variability. Habitat characteristics among maternity dens did not significantly differ between decades (Figure 9.16; Table 9.9, 9.10).

9.4. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T.

DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea-ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Ferguson, S. H., M. K. Taylor, E. W. Born, and F. Messier. 1998. Fractals, sea-ice landscape, and spatial patterns of polar bears. Journal of Biogeography 25:1081–1092.

Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. Ecology 81:761–772.

Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS ONE 9(11): e113746. doi:10.1371/journal.pone.0113746.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere Discussion Papers, in review.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Table 9.1. Breakdown of adult females (AF) collared in the Kane Basin region in the 1990s and 2000s. AF = adult female, AM = adult male, COY = Cub of the Year, YRL = Yearling, 2YR = 2 Year old cub.

		AF alone	AF+AM	AF+COY	AF+YRL	AF+2YR	Sum
1990s	KB	3	0	5	3	1	12
2000s	KB	5	1	7	3	4	20

Table 9.2. Trends in date of spring sea-ice retreat, fall sea-ice advance, fall – spring dates, and summer (June-Oct) sea-ice concentration in Kane Basin (all depths, and depths < 300 m). All trends are statistically significant at the 99% level according to a 2-sided F test, except the date of spring retreat (all depths), which is significant at the 95% level.

Baffin Bay region	**Trend in date of spring ice retreat (days/decade)**	**Trend in date of fall ice advance (days/decade)**	**Trend in fall – spring (days/decade)**	**Trend in ice con. June-October (percent/decade)**
All depths	−6.8	+5.6	+12.4	−5.4
Depths < 300 m	−9.7	+5.5	+15.2	−6.9

Table 9.3. Mean monthly movement rates in KB for radio-collared adult female polar bears (AFs) in the 1990s and 2000s. One SE about the mean is given. We conducted a parametric test of significance between decades; bold = significant at the 5% level.

Month	Mean 1990s	SE 1990s	N 1990s	N steps	Mean 2000s	SE 2000s	N 2000s	N steps	t-test P value
1	3.62	3.78	5	21	4.87	7.59	12	98	0.8691
2	9.39	9.74	4	16	5.9	6.99	11	93	0.4504
3	8.52	9.17	7	35	5.63	6.73	13	117	0.3802
4	5.79	7.28	13	80	5.66	6.57	13	138	0.5197
5	7.35	10.19	12	92	6.75	5.87	21	230	0.6035
6	5.31	7.2	12	85	5.22	4.47	18	193	0.6012
7	3.6	3.19	11	81	5.75	4.13	14	166	0.1011
8	3.41	3.45	10	62	4.84	4.66	14	132	0.5136
9	3.91	5.9	10	63	4.5	4.57	14	132	0.1466
10	4	3.07	9	51	6.42	7.1	14	141	**0.0394**
11	7.87	8.96	9	40	6.94	6.57	13	118	0.986
12	7.87	8.82	8	36	4.68	5.22	12	116	0.3835

Table 9.4. Sea-ice resource selection function (RSF) model coefficients for the Winter season in KB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled such by a certain number of units for more meaningful interpretation. "Mean ice conc.10" is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. "Dist to 50%" is the distance to the 50% sea-ice concentration scaled by 100 km. "Dist to 300 m" is the distance to the 300 m depth contour scaled by units of 100 m. "Depth.100" is the absolute value of bathymetry scaled by 100 m. "Land" is the variable that describes the tendency of a bear to move from sea ice on to land.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
Mean ice conc.10	0.479	0.564	0.3963	0.074	0.192	0.699	0.4976
Dist to 50%.100	-0.014	0.134	0.918	-0.032	0.115	0.7823	0.9181
Dist to 300 m.100	1.82	0.824	0.0272	1.738	0.826	0.0353	0.9436
Depth.100	-0.034	0.078	0.6579	-0.065	0.084	0.4387	0.7892
Land	3.377	4.895	0.4903	1.108	1.504	0.4615	0.6577

Table 9.5. Sea-ice resource selection function (RSF) model coefficients for the Spring season in KB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled such by a certain number of units for more meaningful interpretation. "Mean ice conc.10" is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. "Dist to 50%" is the distance to the 50% sea-ice concentration scaled by 100 km. "Dist to 300 m" is the distance to the 300 m depth contour scaled by units of 100 m. "Depth.100" is the absolute value of bathymetry scaled by 100 m. "Land" is the variable that describes the tendency of a bear to move from sea ice on to land. Bold = significant at the 5% level.

	1990s coef	**SE**	**P-value 1990s**	**2000s coef**	**SE**	**P-value 2000s**	**P-value delta**
Mean ice conc.10	0.255	0.062	**<0.001**	0.223	0.058	**0.0001**	0.7107
Dist to 50%.100	-0.007	0.051	0.8867	-0.118	0.071	0.0949	0.2031
Dist to 300 m.100	1.324	0.502	**0.0084**	-0.997	0.483	**0.0392**	<0.001
Depth.100	-0.159	0.043	**0.0002**	-0.102	0.051	**0.0451**	0.4
Land	-0.349	0.476	0.4639	-0.588	0.469	0.2107	0.7208

Table 9.6. Terrestrial resource selection function (RSF) model coefficients for the Summer season in KB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled such by a certain number of units for more meaningful interpretation. "Elev.100" is elevation scaled by units of 100 m. "Slope.10" is slope in degrees scaled by units of 10 degrees. "Aspect.10" is aspect scaled by units of 10 degrees. "Not Land" is the tendency of a bear to move from land on to sea ice. Note not all bears used land in summer and land-use was intermittent. Bold = significant at the 5% level.

	1990s coef	**SE**	**P-value 1990s**	**2000s coef**	**SE**	**P-value 2000s**	**P-value delta**
elev.100	-0.452	0.07	**<0.001**	-0.74	0.054	**<0.001**	0.001
slope.10	0.256	0.136	0.0594	0.316	0.07	**<0.001**	0.692
aspect.10	-0.019	0.013	0.1621	0.007	0.007	0.3166	0.088
NotLand	-0.443	0.317	0.1619	-1.186	0.166	**<0.001**	0.03

Table 9.7. Number of polar bear maternity and shelter dens in Kane Basin in the 1990s and 2000s.

All Dens		Maternity Dens		Shelter Dens	
1990s	9	1990s	3	1990s	6
2000s	9	2000s	3	2000s	6
Total	18	Total	6	Total	12

Table 9.8. Summary table of the phenology for Kane Basin polar bear maternity dens.

	1990s **Maternity Dens (*n* = 3)**			**2000s** **Maternity Dens (*n* = 3)**		
	Entry DOY	Exit DOY	Duration (# days)	Entry DOY	Exit DOY	Duration (# days)
Mean	279	78.3	164.3	274	77.7	168.7
Min	274	69	145	252	65	144
Max	289	89	180	301	88	184
Median	274	77	168	269	80	178
SD	8.7	10.1	17.8	24.9	11.7	21.6

Table 9.9. Summary table of the habitat characteristics for Kane Basin polar bear maternity and shelter dens. Elev. = elevation (meters), Asp. = aspect (degrees), CoastDist = distance to nearest coastline (kilometers).

	All Maternity Dens (*n* = 6)				**All Shelter Dens (*n* = 12)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	327	165.8	13.4	5.2	366.6	141.8	15.4	3.4
Min	9	28.3	3.7	0.7	6	5.5	1.8	0.1
Max	506	229	17.8	12.5	855	350	36.9	8
Median	408	186.8	15.9	3.4	318	168.3	12	2.8
SD	188.4	69.8	5.4	4.8	274.8	126.3	10.8	2.4
	1990s Maternity Dens (*n* = 3)				**1990s Shelter Dens (*n* = 6)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	207.7	201	11.9	2.1	386.7	134	12.9	2.6
Min	9	185.9	3.7	0.7	6	12	1.8	0.1
Max	422	229	16.1	4.3	855	349.7	36.9	5.5
Median	192	188.1	15.8	1.2	257	105.7	9.9	2.3
SD	206.9	24.3	7.1	1.9	378.5	137.4	12.4	2.1
	2000s Maternity Dens (*n* = 3)				**2000s Shelter Dens (*n* = 6)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	446.3	130.6	15	8.3	346.5	149.5	17.9	4.2
Min	394	28.3	10.7	2.6	149	5.5	7.2	1.6
Max	506	187.7	17.8	12.5	500	350	32.5	8
Median	439	175.9	16.5	9.7	355.5	168.3	16.4	4.1
SD	56.4	88.8	3.8	5.1	148.2	126.7	9.3	2.5

Table 9.10. Results of the two-sample Mann-Whitney U tests comparing habitat characteristics for Kane Basin polar bear maternity and shelter dens.

	Maternity Dens (*n* = 6)		**Shelter Dens (*n* = 12)**	
	W	***p*-value**	**W**	***p*-value**
Elevation	8	0.2	20	0.818
Slope	7	0.4	25	0.31
Aspect	1	0.2	17	0.937
Distance to Coast	8	0.2	24	0.394

Figure 9.1. Distribution of capture locations for polar bears in Kane Basin (KB) in each decade. See Table 9.1 for sample sizes in each year. Note in 1990s bears in KB were captured on the west side of KB whereas 2000s bears were captured on both the east and west side.

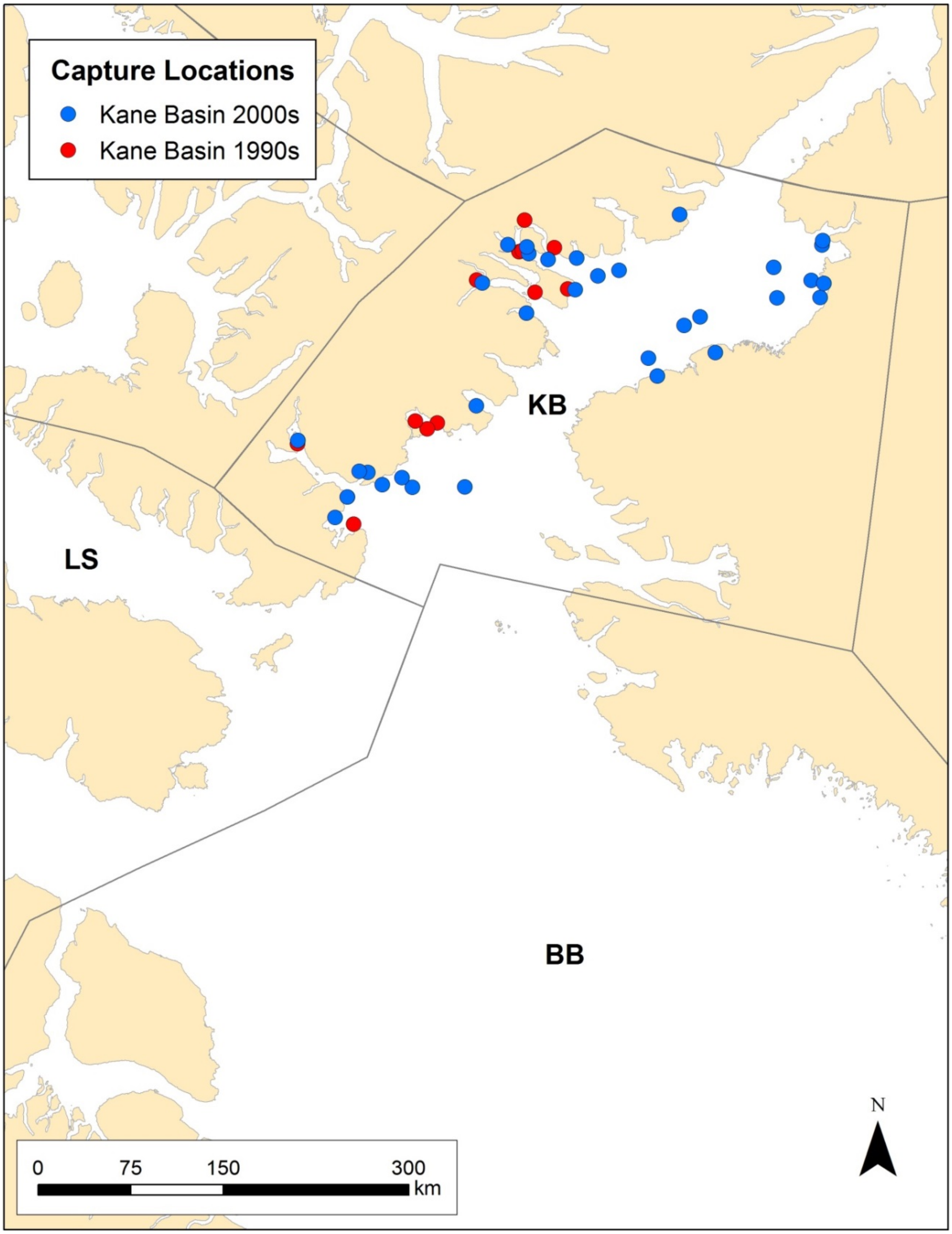


Figure 9.2 Sea-ice area in Kane Basin (all depths) for the years 1979-2014 (gray curves). Two six-year averages are also shown (colored curves). The threshold for defining the dates of sea-ice retreat and advance (middle horizontal dotted line) is halfway between the average March sea-ice area (upper dotted line) and the average September sea-ice area (lower dotted line).

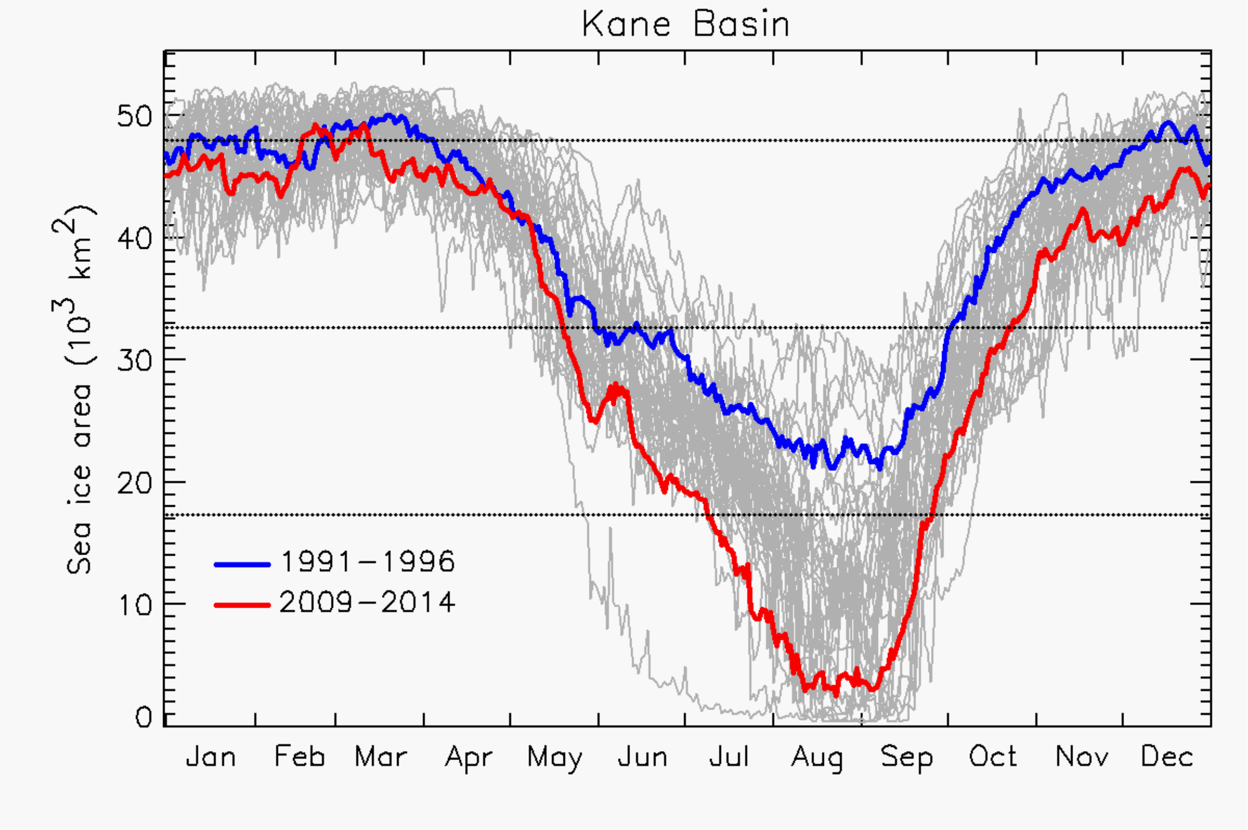


Figure 9.3 Day of spring sea-ice retreat (red circles), fall sea-ice advance (blue circles), and the interval between them (green lines), for Kane Basin (all depths), 1979-2014. Least-squares fits to spring and fall dates are shown (red and blue lines). Trends are given in Table 9.2.

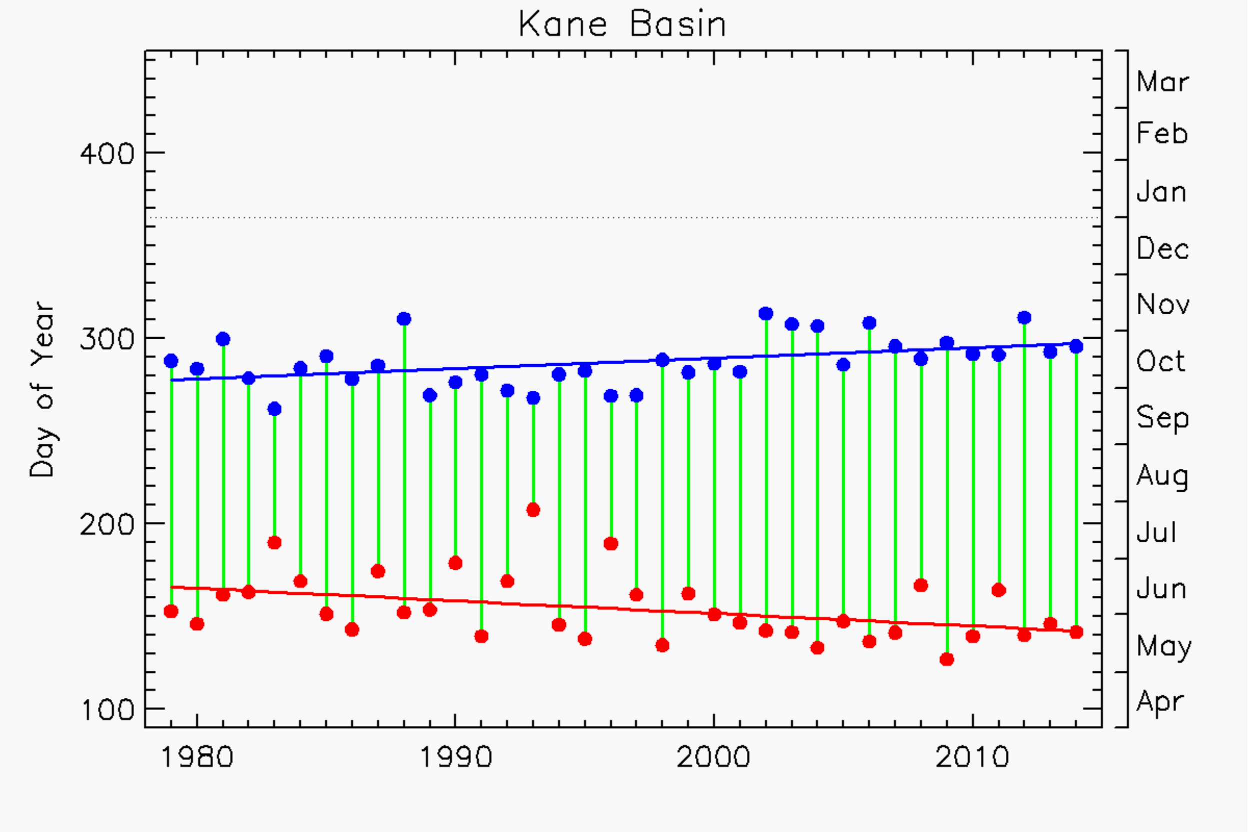
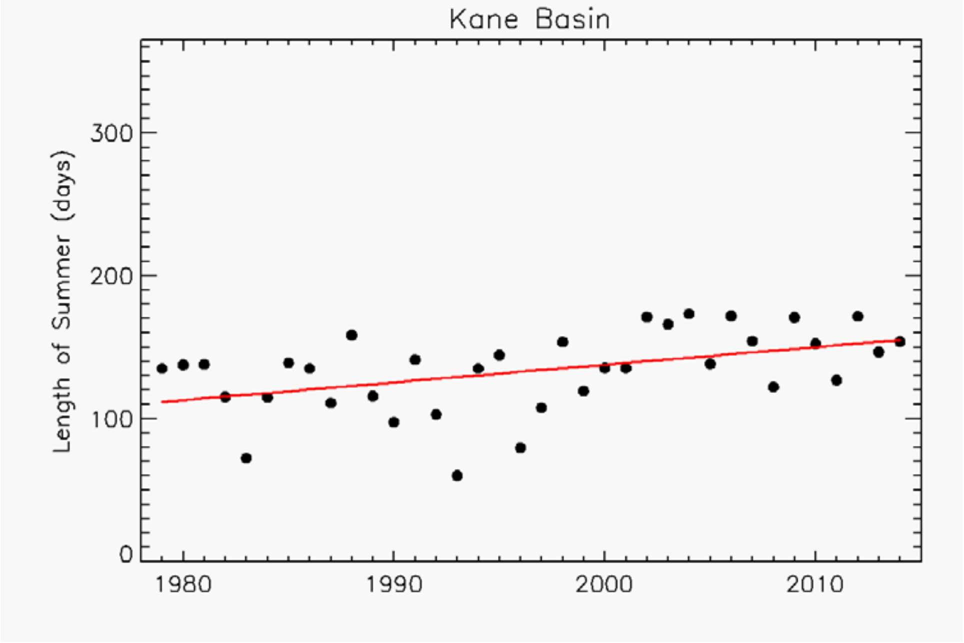


Figure 9.4. Length of summer (a) and mean sea-ice concentration during June-October (b) for Kane Basin (all depths), 1979-2014. Length of summer is the interval from spring sea-ice retreat to fall sea-ice advance (see Figure 9.2, green lines). Least-squares fits are shown (red lines); trends are given in Table 9.2.

(a)



(b)

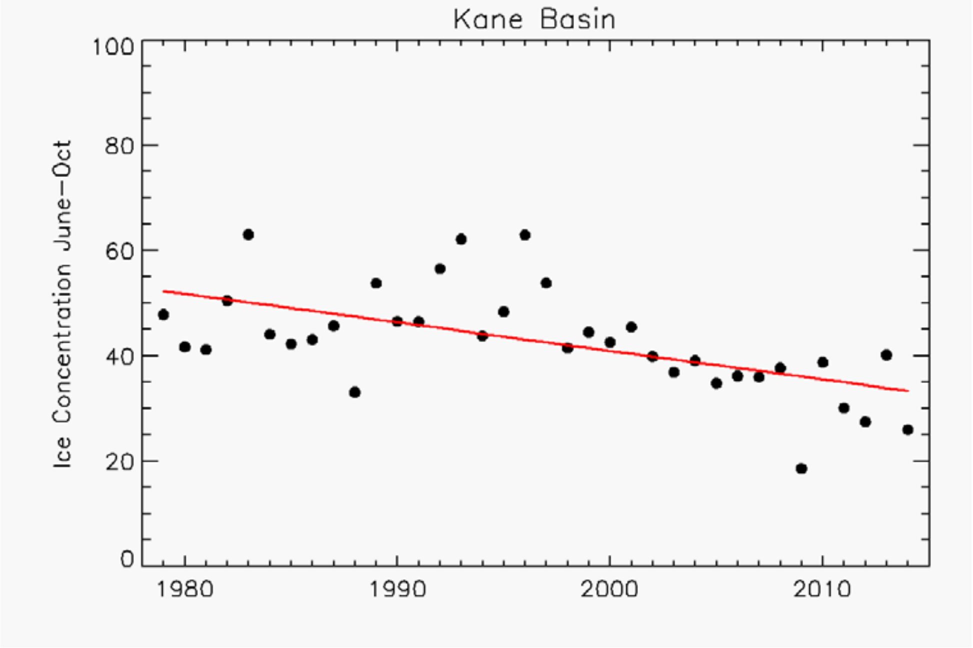


Figure 9.5. Movement rate of KB adult female bears (km/day) in the 1990s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

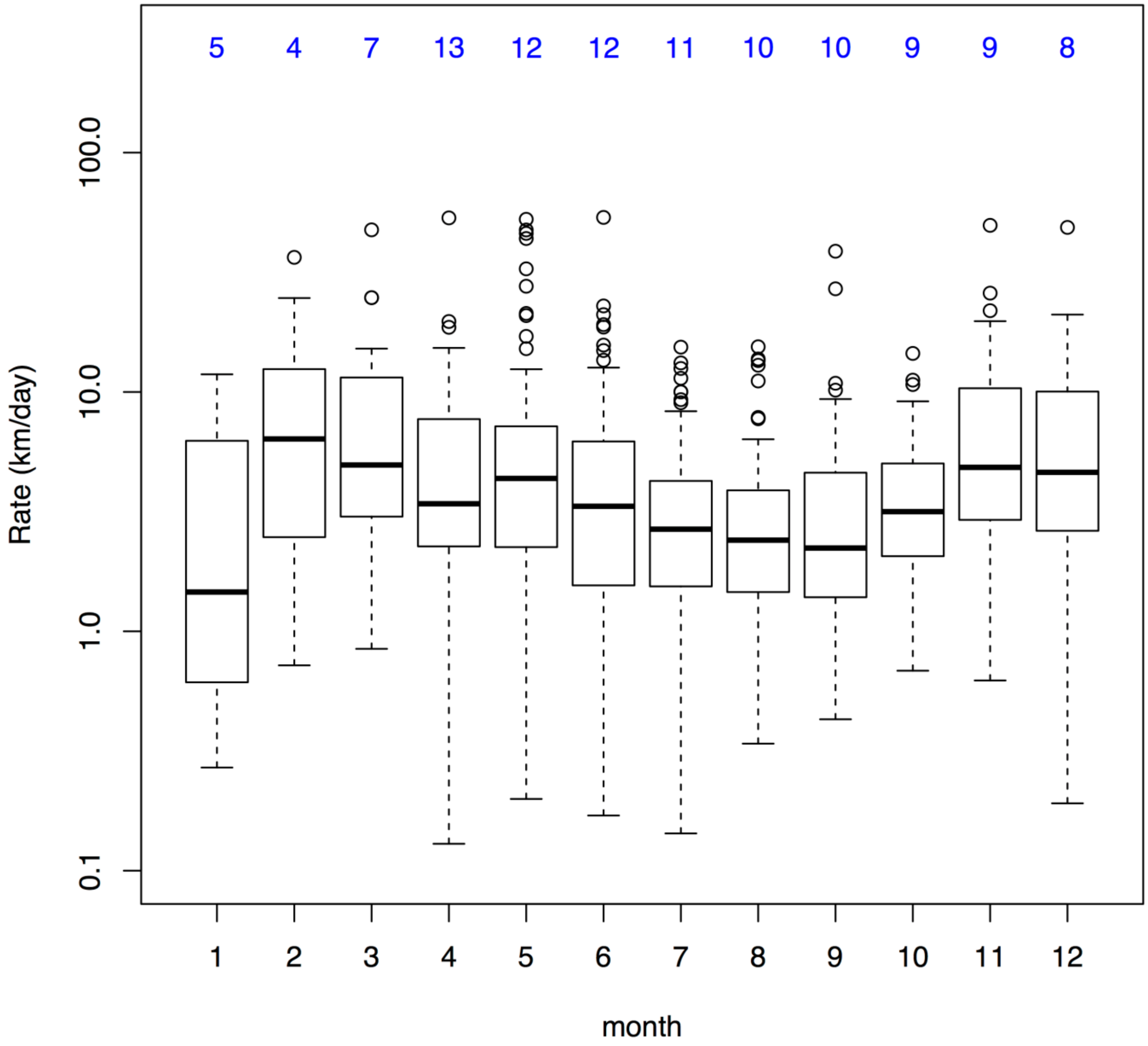


Figure 9.6. Movement rate of BB adult female bears (km/day) in the 2000s. Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

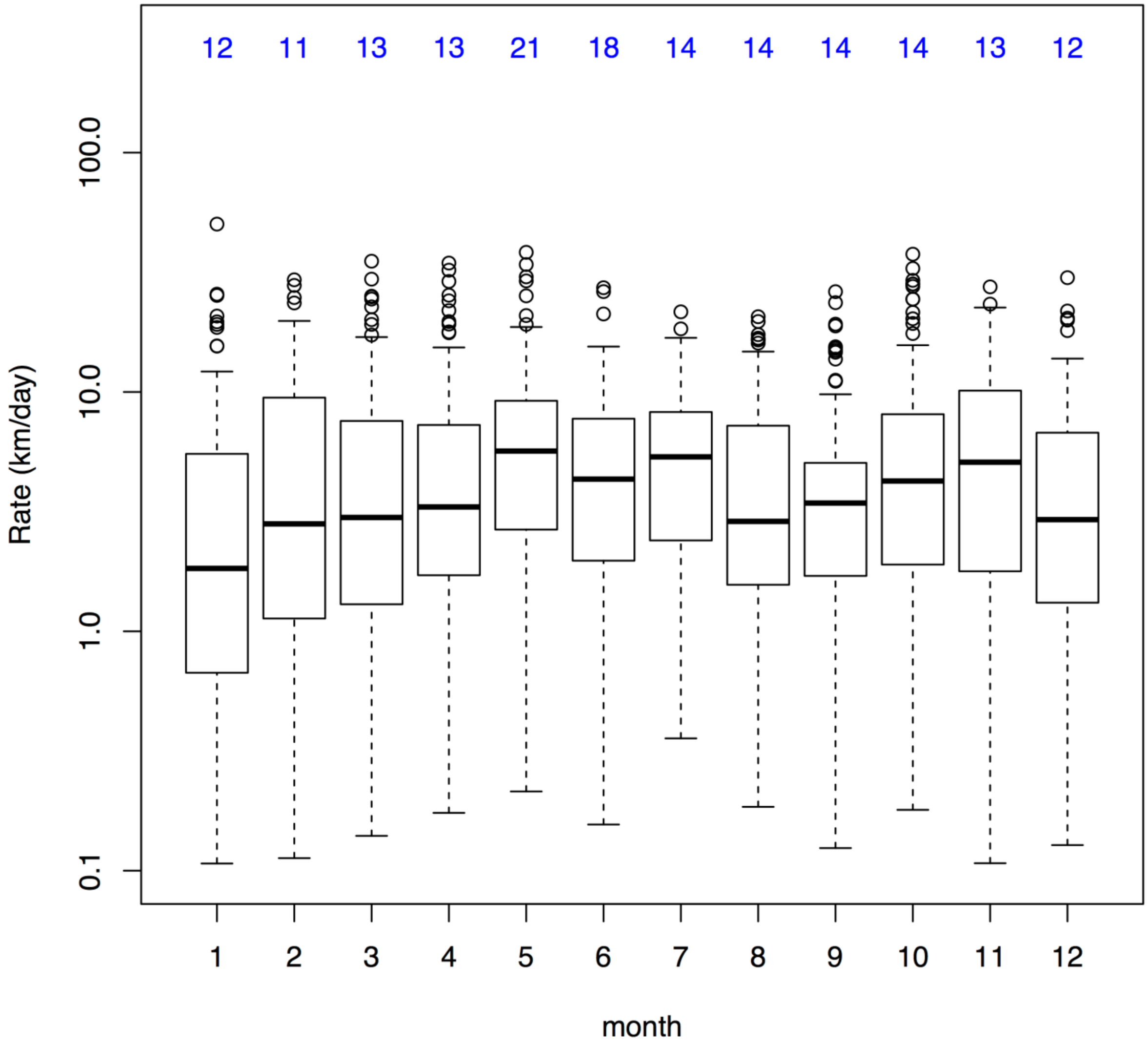


Figure 9.7. 1990s and 2000s adult female polar bear habitat use in KB for each of four sea-ice habitat variables: sea-ice concentration in small buffer, distance to 15% sea-ice concentration, distance to 50% sea-ice concentration, and distance to the nearest land. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer) used in the analysis. SSM/I sea-ice concentration is used in both decades. Months 8 -10 also represent land use by bears. The small numbers indicate the number of movements captured within each months.

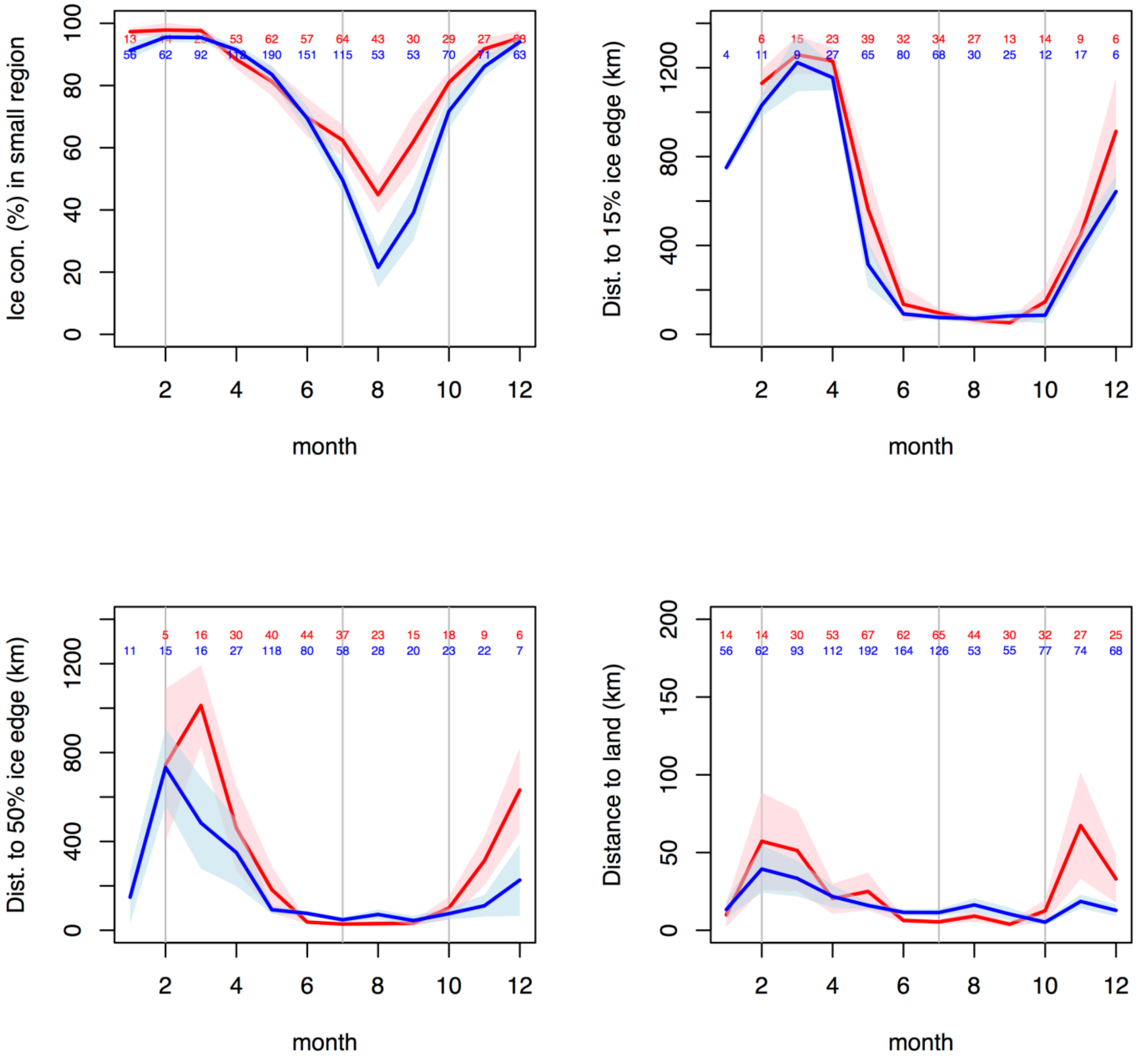


Figure 9.8. 1990s and 2000s adult female polar bear habitat use in KB for each of three sea-ice habitat variables: distance to 300 m depth contour, depth (bathymetry), and percentage of observations on the sea ice. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer). SSM/I sea-ice concentration is used in both decades. Months 8 -10 also represent land use by bears.

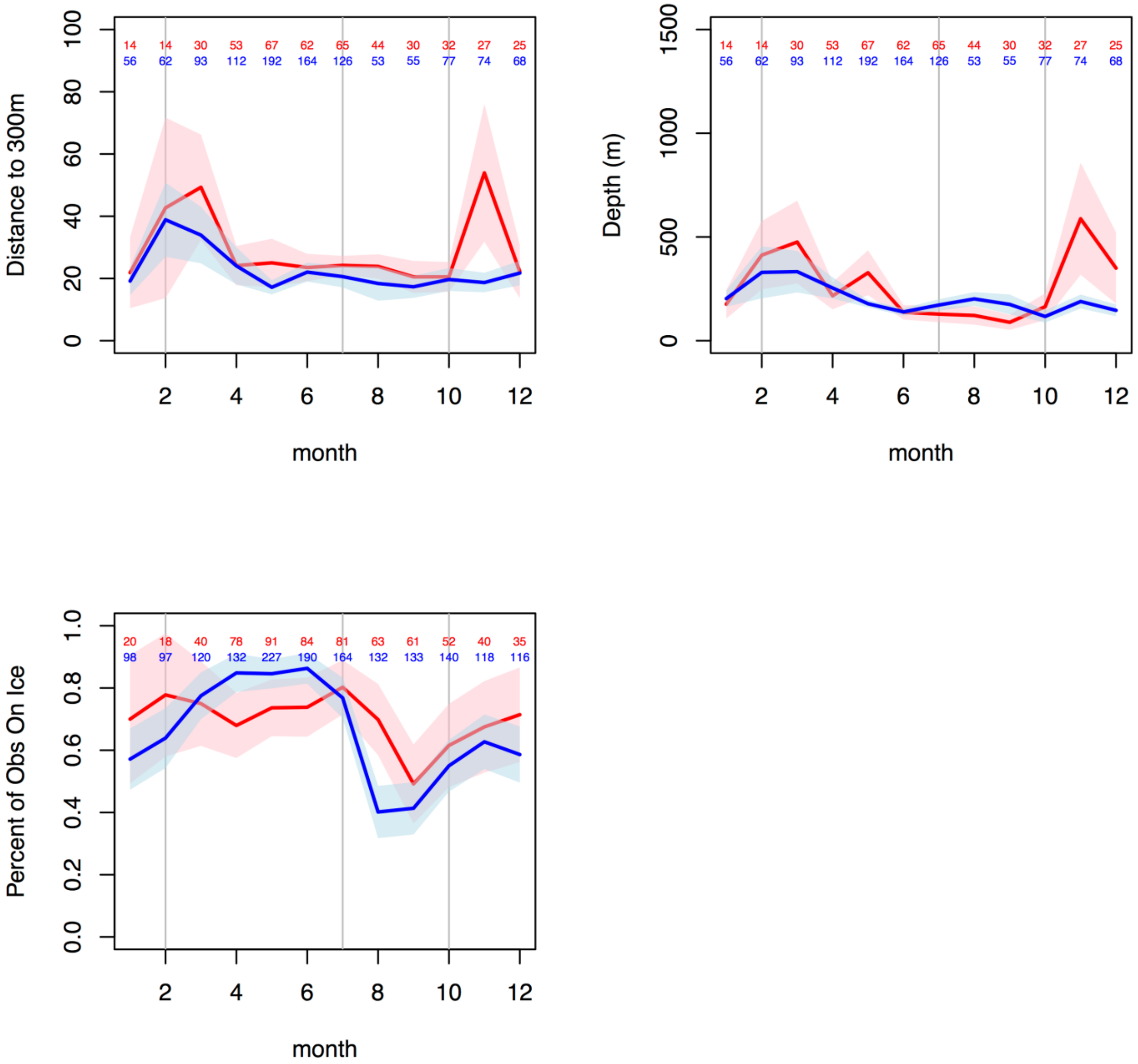


Figure 9.9. 1990s and 2000s adult female polar bear habitat use in KB for each of three terrestrial habitat variables: elevation, slope, and aspect. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Months 8 -10 also represent land use by bears.

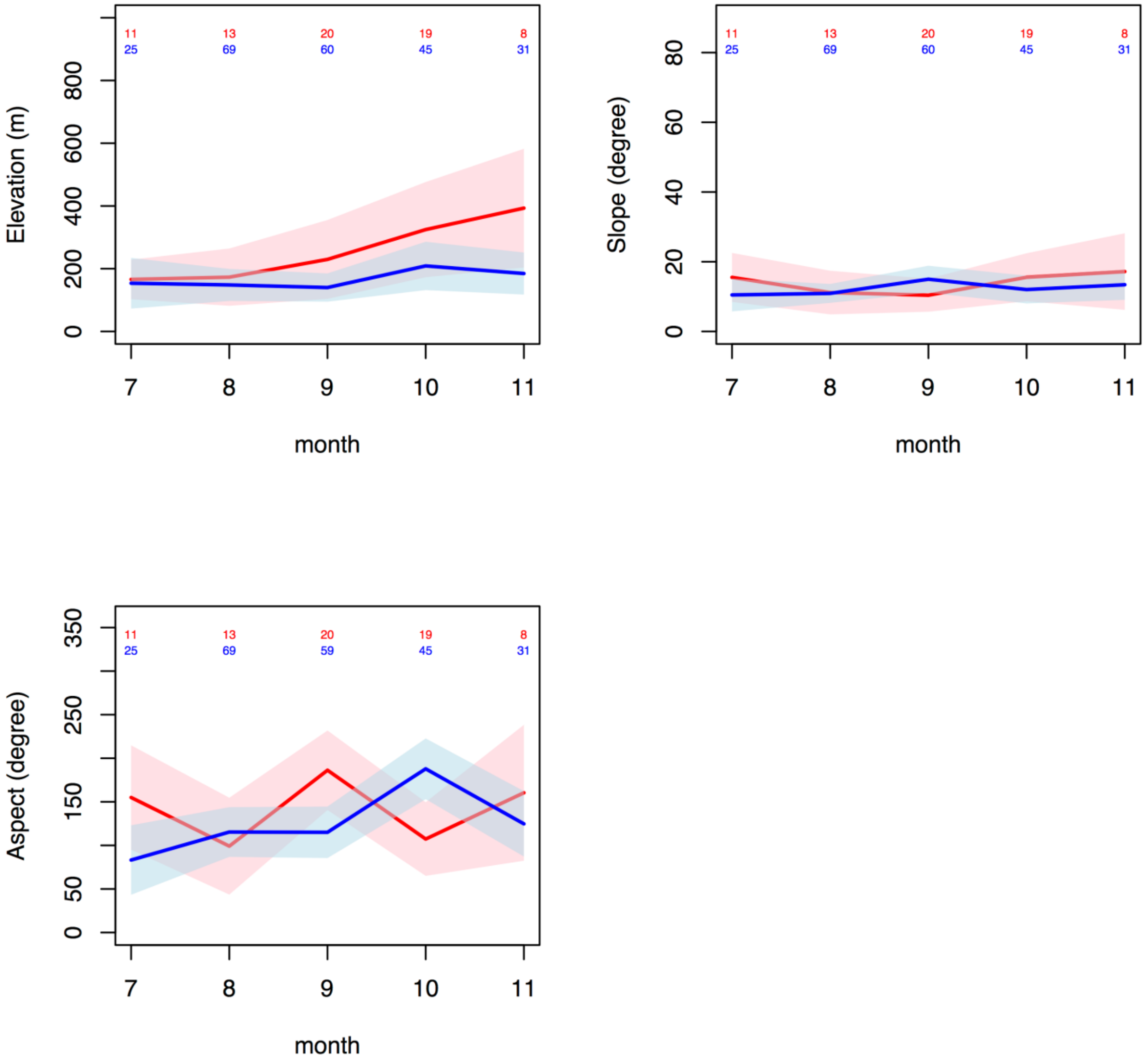


Figure 9.10. ASTER Digital Elevation Model (DEM) used in KB with trackline of a single bear tagged in the 2000s.

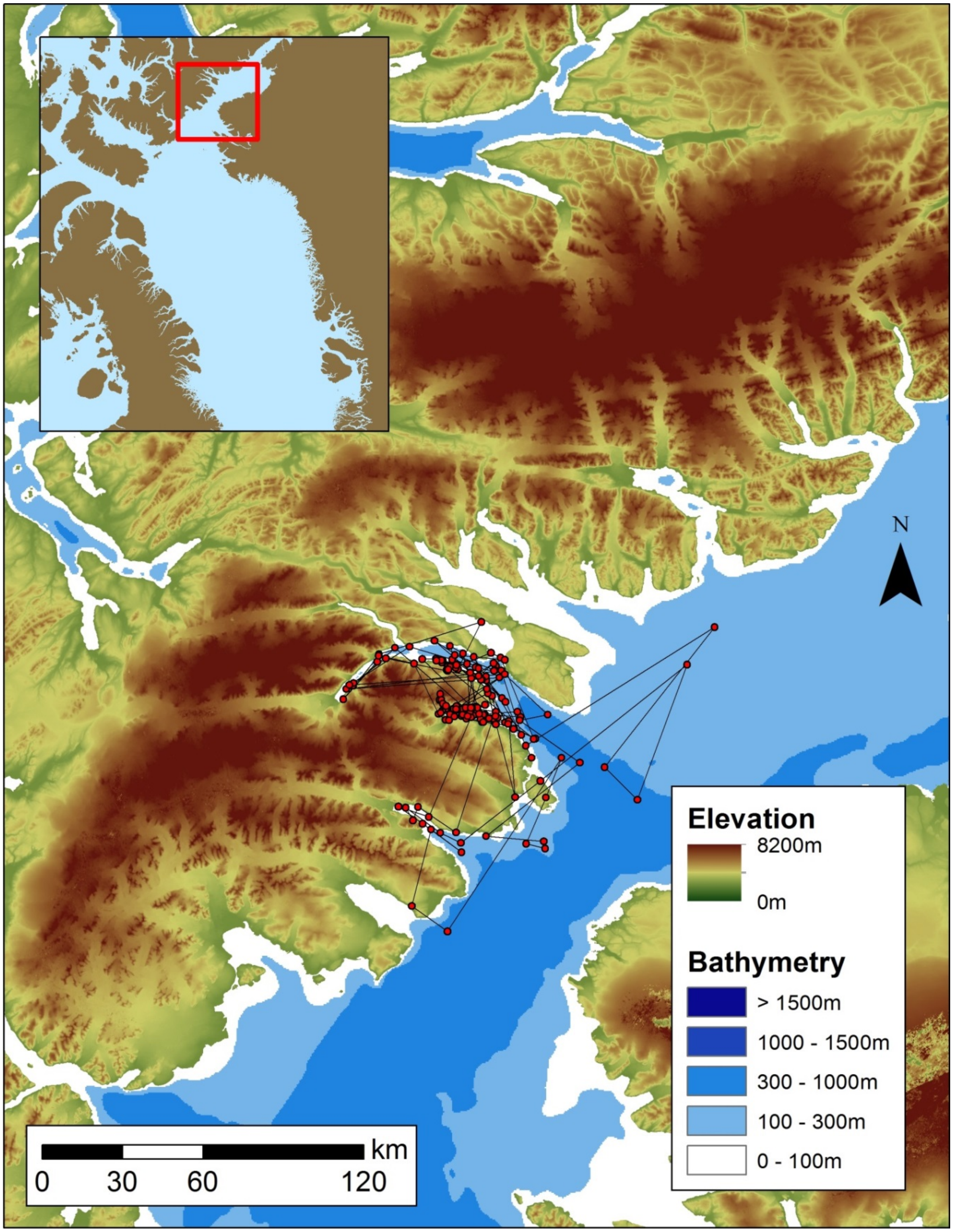


Figure 9.11. Distribution of KB polar bear maternity dens in the 1990s and 2000s.

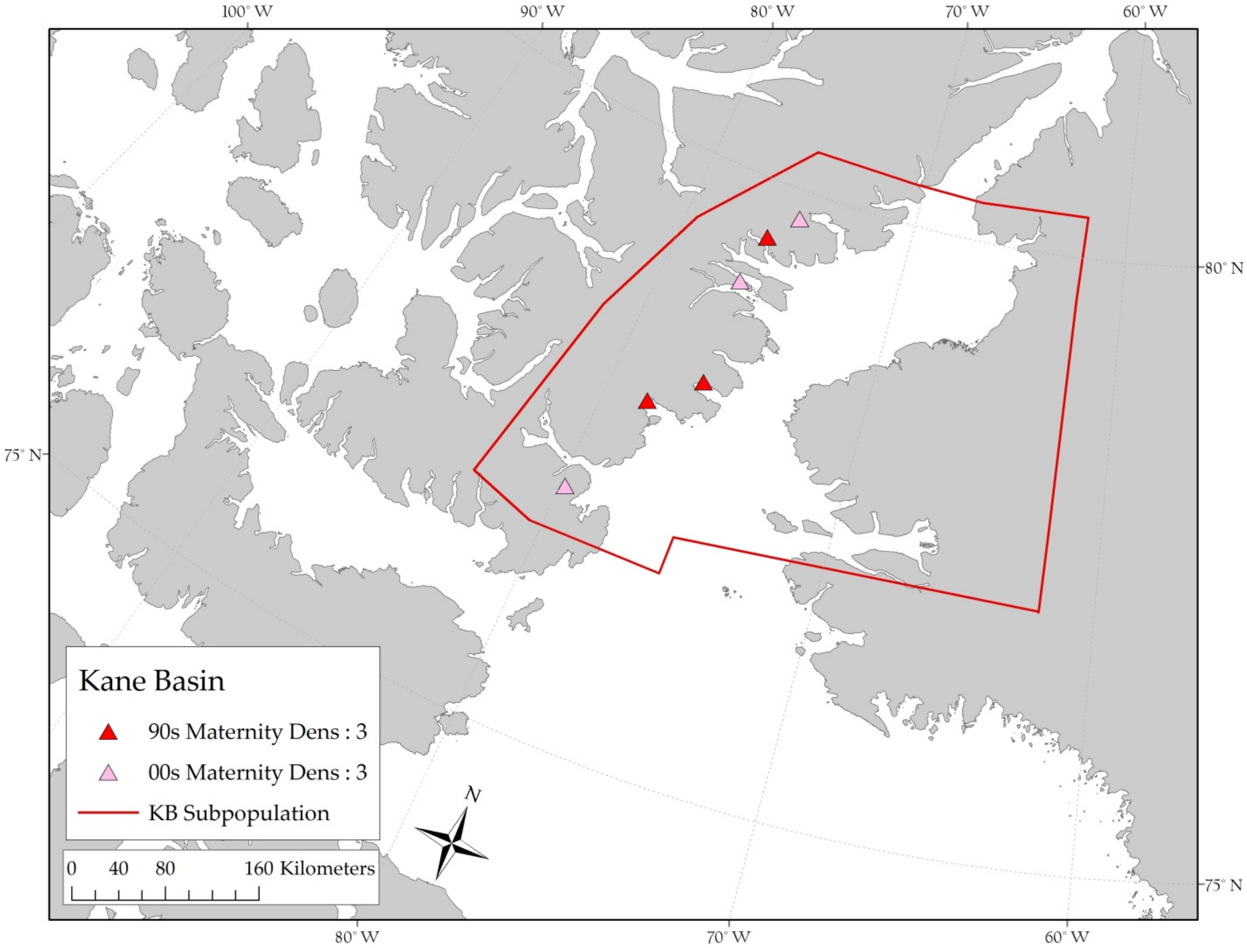


Figure 9.12. Distribution of KB polar bear shelter dens in the 1990s and 2000s.

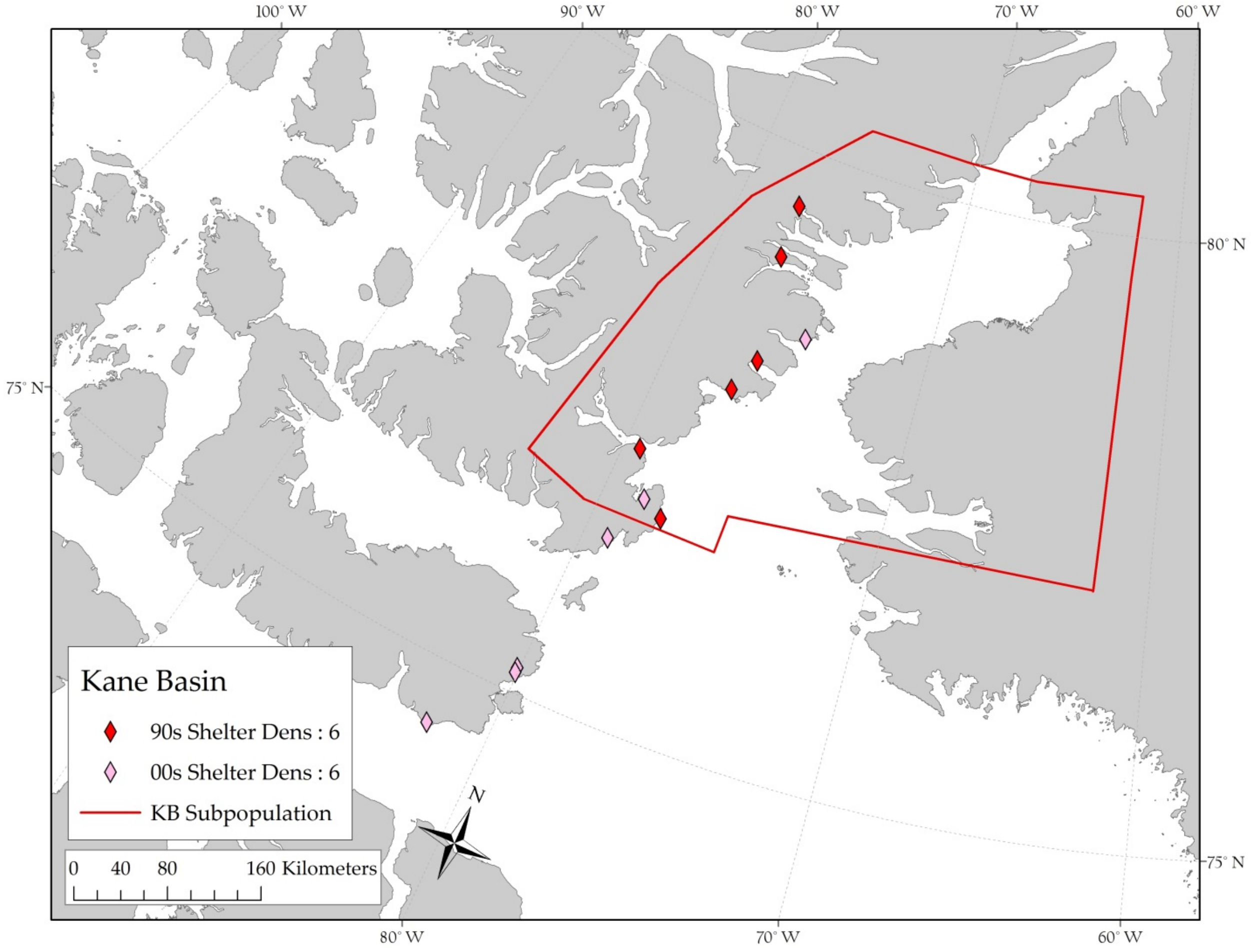


Figure 9.13. Boxplots comparing den duration of Kane Basin (KB) polar bear maternity dens (*P* = 1) (1990s: *n* = 3; 2000s: *n* = 3).

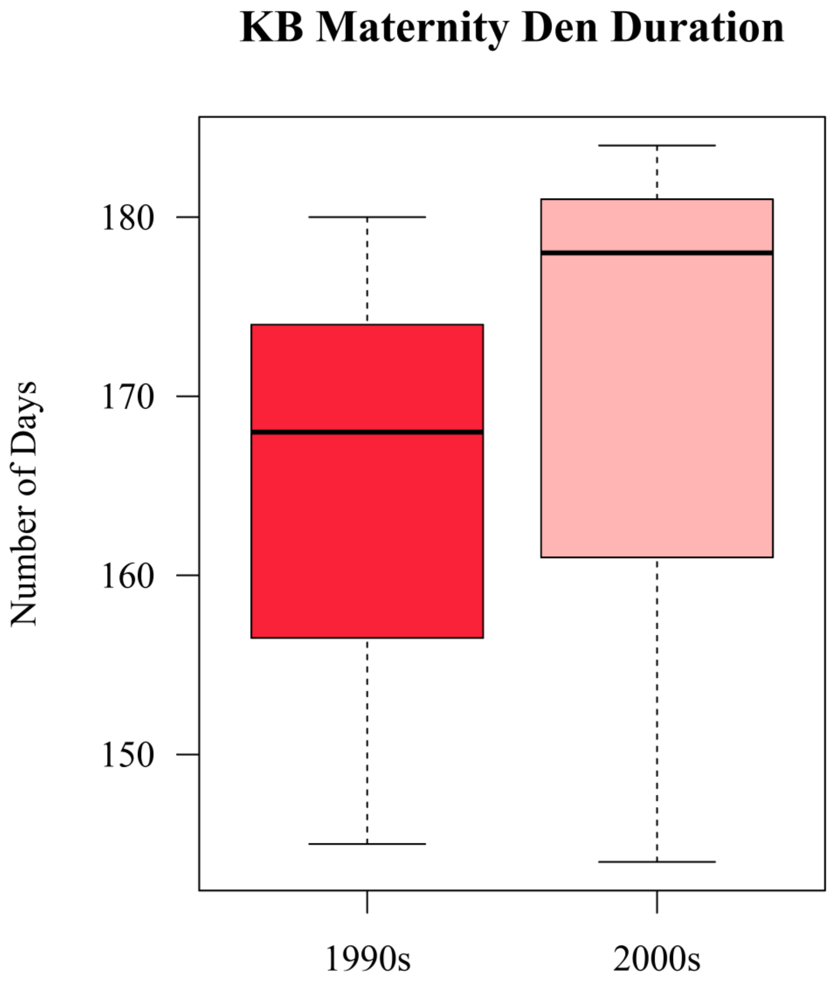


Figure 9.14. Boxplots comparing entry (*P* = 0.6) and exit dates (*P* = 1) of Kane Basin (KB) polar bear maternity dens (1990s: *n* = 3; 2000s: *n* = 3).

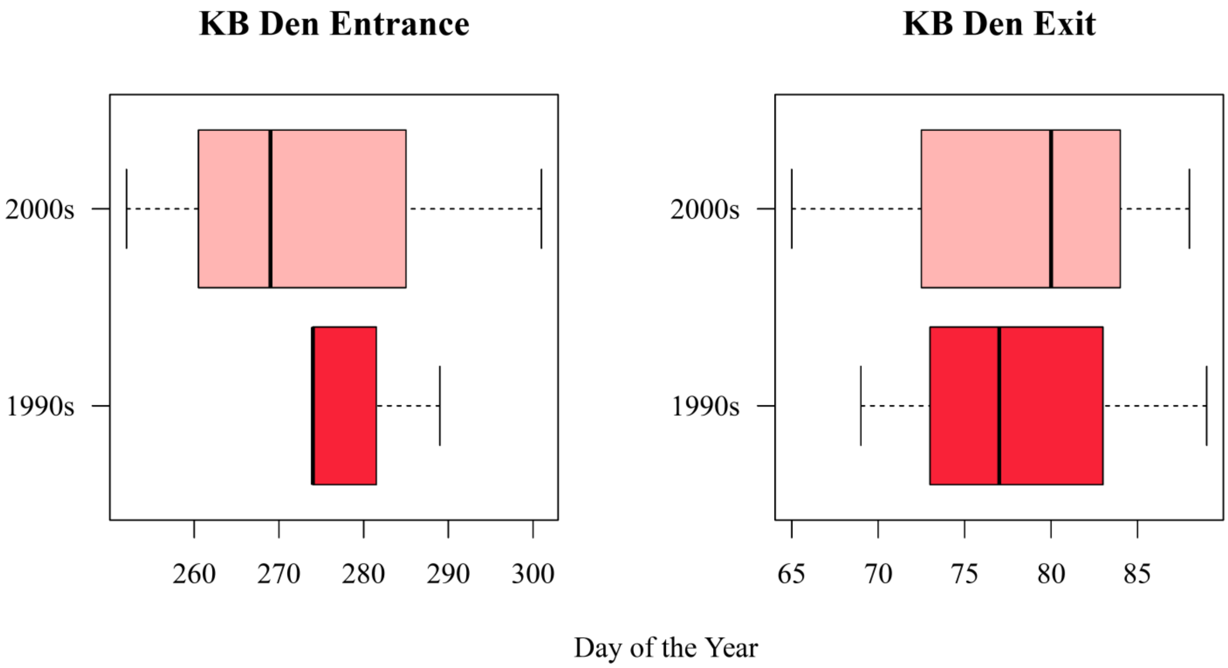


Figure 9.15. Boxplots comparing the first date on land (FDOL) of pregnant female polar bears from the 1990s (*n* = 3) and 2000s (*n* = 3) in Kane Basin (KB) (*P* = 1).

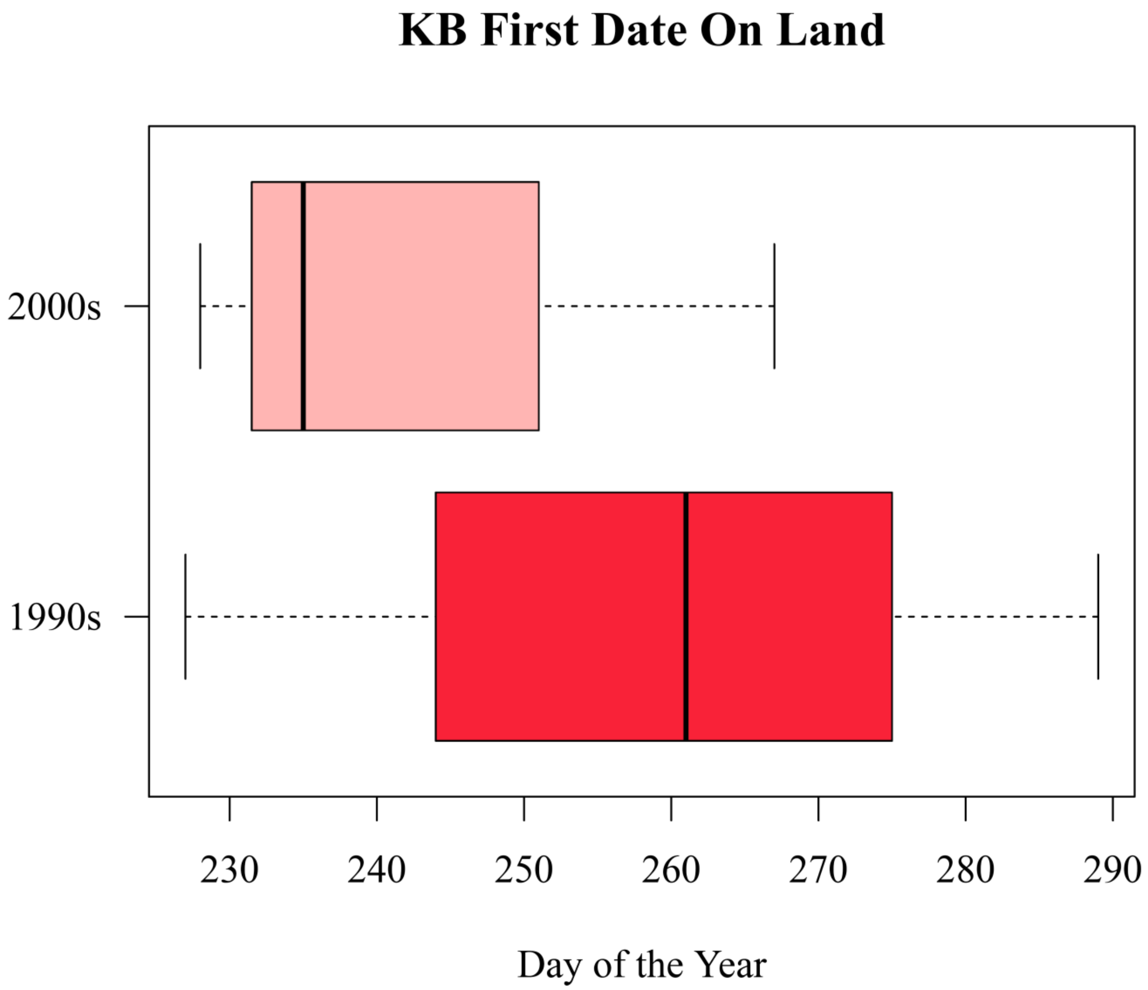
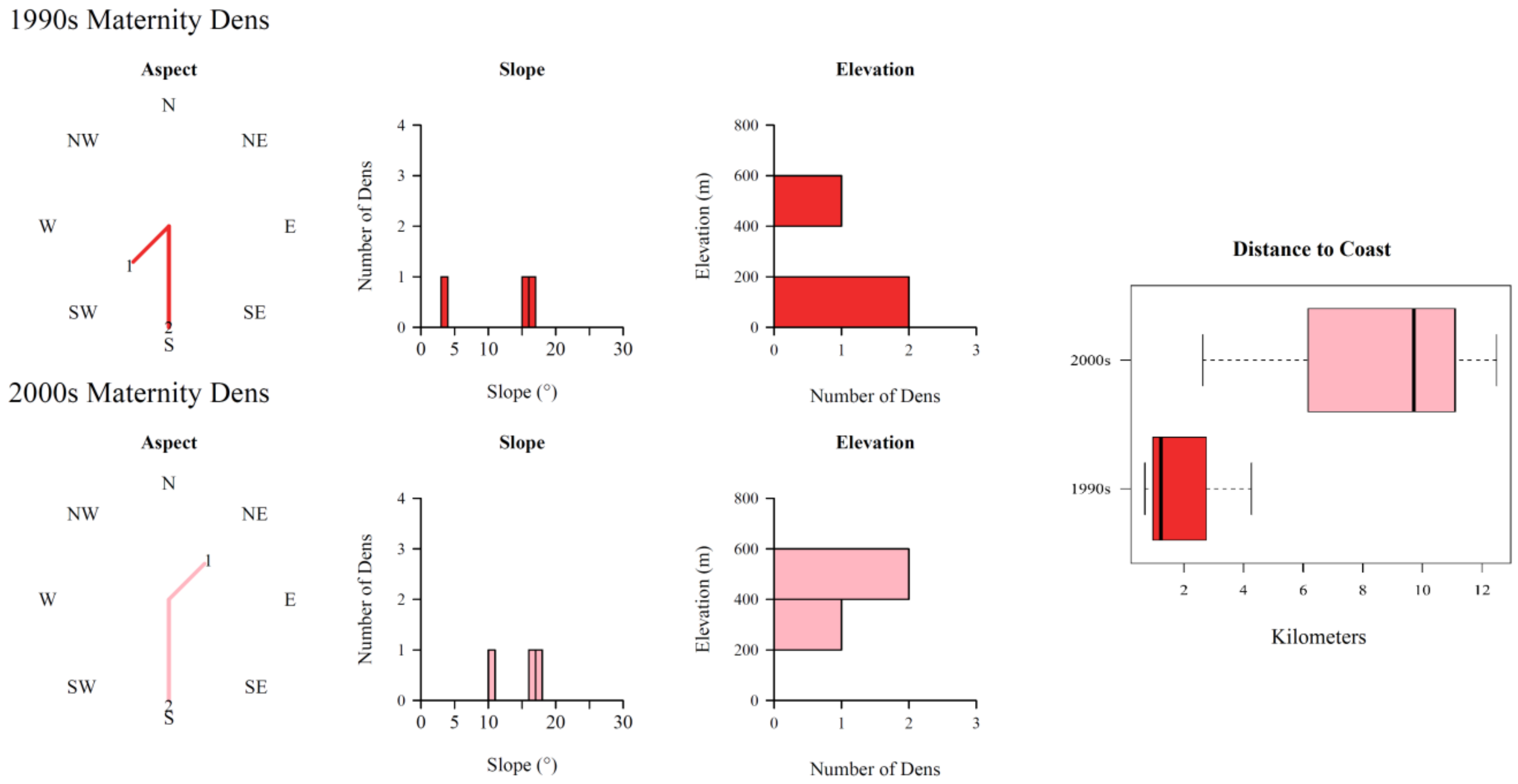


Figure 9.16. Plots comparing aspect, slope, elevation, and distance to coast for the 1990s (*n* = 3) and 2000s (*n* = 3) polar bear maternity dens in Kane Basin. The aspect plot consists of a compass face with lines marking the directions that dens faced. The lines are annotated with numbers noting how many dens were found at that aspect. None of the habitat variables significantly differed between the two time periods (elevation, aspect, distance to coast: $P = 0.2$; slope: $P = 0.4$).



CHAPTER 10

GENETIC MARK-RECAPTURE STUDY OF POLAR BEARS IN KANE BASIN

KEY FINDINGS

- We used joint live-recapture and dead-recovery mark-recapture models to analyze data for the Kane Basin (KB) polar bear subpopulation, with the goal of updating estimates of subpopulation size and survival. The dataset consisted of 277 initial live captures (1992-1997 = 150, 2012-2014 = 127), 89 live recaptures (1992-1997 = 53, 2012-2014 = 36), and 24 harvest returns of research-marked bears 1992-2014.

- Mark-recapture research conducted in the Kane Basin subpopulation yielded an estimate of abundance of 357 polar bears (95% CI: 221 – 493) for 2013 – 2014. An estimate derived during 1995 – 1997 yielded 224 bears (95% CI: 145 – 303). Based on physical MR, the size of the KB subpopulation was previously estimated to be 164 polar bears (95% CI: 94-234) for 1994-1997 (noting that this estimate applies to different years than our re-analysis; Taylor et al. 2008).

- We documented more bears in the eastern regions of the Kane Basin subpopulation during 2012 – 2014 than during the 1990s. Eastern Kane Basin was searched during the 1990s although with less effort than in the 2010s due to the low density of bears observed there. The difference in distribution between the 1990s and 2010s may reflect differences in spatial distribution of bears, possibly influenced by reduced hunting pressure by Greenland in eastern KB and thus an increased density of bears in KB, but also some differences in sampling protocols.

- The 2013 – 2014 estimate of abundance suggests 357 (221 – 493) bears currently use KB in springtime (i.e., the Kane Basin super-population; Kendall et al. 1997), and the current point estimate is higher than the historical estimate. Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of KB abundance for 2013-2014 and the estimate for 1995-1997 was approximately 133 bears (standard deviation of the difference ≈ 80 bears), with 95% of the sampling distribution suggesting that population change between the two time periods could have been positive. This suggests relatively strong evidence for a stable to increasing subpopulation, and is consistent with data on movements, condition and reproduction. We encourage some caution in interpretation of population growth due to potential expansion of the sampling frame and differences in sampling protocols between the 1990s versus the 2010s study periods.

- Current estimates of total survival for age 3+ females (0.95; SE: 0.04) and dependent bears were consistent with previous research. Estimates of unharvested survival for 3+

females appear sufficiently high for positive population growth. Updated estimates of total survival are lower for age 3+ males (0.87; SE: 0.06). Our longer-term data set and several other ecological, sampling, and technical considerations may contribute to this result.

- We documented a reduction in mortality associated with harvest, likely attributable to implementation of Greenland's harvest quota in 2006.
- Demographic modeling suggests Kane Basin bears exhibit relatively high fidelity to the springtime study area, with <5% of marked bears emigrating on an annual basis.

10.1. Introduction

Large-scale environmental changes are occurring across the circumpolar Arctic (Comiso et al. 2008, Stroeve et al. 2012, Laidre et al. 2015; see also Chapters 5 and 9), with general reductions in the temporal availability and spatial extent of sea ice. For sea ice obligate polar bears, which are among the most highly sensitive of marine mammals to the projected impacts of climate change (Laidre et al. 2008), long-term impacts are anticipated to be negative (Atwood et al. 2015). However, there likely will be significant temporal and spatial variability among subpopulations in the short-term (Stirling and Derocher 2012). Indeed, the effects of the changing Arctic environment on polar bears have been documented in some regions but are less clear or have not been realized elsewhere (e.g., Rode et al. 2012, 2014, Bromaghin et al. 2015, Obbard et al. 2015, Lunn et al. 2016).

The Kane Basin (KB) polar bear subpopulation, regarded as part of the Arctic archipelago region (Amstrup et al. 2008), covers a small region between Nunavut, Canada and NW Greenland. Abundance of KB was last estimated at ~164 (SE: 35) polar bears based on a physical mark-recapture study completed during 1992 – 1997 (Taylor et al. 2008). At the time of this estimate, the harvested population growth rate ($\lambda = 0.919$) indicated that the subpopulation was over-exploited. The unharvested growth rate also was low ($\lambda = 1.009$; Taylor et al. 2008), suggesting limited capacity for the KB subpopulation to increase even in the absence of human-caused removals. In response, Greenland implemented a quota in 2006 that significantly reduced the total harvest from Kane Basin (see Methods below). The small subpopulation size, low growth rates, and long-term exploitation led Taylor et al. (2008) to suggest that Kane Basin may act as a sink for neighboring subpopulations such as Baffin Bay.

The KB subpopulation is currently considered to be declining (PBSG 2015): 100% of population viability analysis (PVA) simulations (using data on abundance and vital rates from Taylor et al. 2008 and reported Canadian and Greenlandic catches) resulted in a decline in abundance within 10 years. However, no new research to update estimates of abundance or vital rates has occurred since the 1990s study. Given the outdated demographic information, the substantial changes in Arctic sea-ice habitats over the past several decades (e.g., Stroeve et al. 2012, Chapters 4 and 9), and the reduction in harvest in 2006, there was uncertainty as to the current status of polar bears in Kane Basin. As such, there was a need for new information to inform status and harvest management (Chapter 1).

Although bears in KB are not genetically different from those in Baffin Bay (Paetkau et al. 1999, Chapter 2), satellite telemetry and capture records indicate that they move among KB and neighboring subpopulations but exhibit strong fidelity to specific regions (Taylor et al. 2001, Chapters 2, 5, and 9). These data have formed the basis for population delineation, and polar bears in Kane Basin are considered a distinct demographic unit for management purposes.

Our objective was to estimate the current abundance and vital rates, including survival, of polar bears in the KB subpopulation. We sought to compare new estimates of abundance with those derived from earlier research (Taylor et al. 2008). These results, in conjunction with information on sea-ice dynamics, spatial ecology, reproductive output, survival, and other metrics, will be used to inform subpopulation status. The data used in this project spanned a 23-

year period (1992-2014): an initial 6-year physical capture and dead recovery sampling period (1992 – 1997) was followed by a 14-year period with dead recoveries only (1998 – 2011) and a recent (2012 – 2014) live capture (physical and genetic) and dead recovery session. Jurisdictions across the Arctic have increasingly invested in non-physical capture based monitoring methods, largely to address social considerations, particularly in Nunavut, regarding wildlife handling (*cf.* Chapter 1) and to facilitate more rapid monitoring. Prior to this study and research in the Baffin Bay subpopulation (Chapter 5), however, the focus of such alternative methods has been aerial surveys (e.g., Aars et al. 2009, Obbard et al. 2015, Stapleton et al. 2016).

10.2. Materials and Methods

Study Area

The KB subpopulation covers ~150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (boundaries evaluated in Taylor et al. 2001; Figure 10.1). However, the boundaries of the KB subpopulation encompass a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* Figure 10.3 and 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area) and the southern part of Kennedy Channel. It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the Baffin Bay (BB) and Lancaster Sound (LS) subpopulations, and to the west by Norwegian Bay (NW; PBSG 2010). The KB subpopulation is regarded as a part of the Arctic archipelago ecoregion (Amstrup et al. 2008); historically sea ice remained present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, this pattern has changed markedly in recent decades (Chapter 9, Figure 9.2). KB is partially connected to neighboring subpopulations; particularly notable though limited interchange occurs with BB and LS (Chapter 4). The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a partial barrier between KB and BB – LS. The KB subpopulation is subjected to subsistence harvest by Inuit living in Jones Sound (Canada) and the Qaanaaq area (NW Greenland; PBSG 2010, Born et al. 2011).

Field Sampling

Initial surveying was conducted in Kane Basin during springs (April – May), 1992 – 1997 (described in Taylor et al. 2008). Additional sampling was completed during fall 1994, but we excluded these data from the present analyses to reduce temporal heterogeneity (e.g., sampling cubs-of-the-year in spring versus fall yields substantial differences in estimates of survival). All sighted bears, including dependent offspring, were chemically immobilized (Stirling et al. 1989) and uniquely marked with plastic ear tags and permanent lip tattoos (Taylor et al. 2008). Ages of independent bears were determined by extracting vestigial premolars and counting annular rings (Calvert and Ramsay 1998), whereas cubs-of-the-year and yearling bears were considered of known age. A sample of adult females was outfitted with satellite collars as part of a separate study quantifying movements and spatial ecology (Taylor et al. 2001, Chapter 9).

Although Taylor et al. (2008) reported that they conducted a uniform search of the study site each year, records delineating their survey effort were unavailable. Subsequent examination of annual distributions of captures suggested incremental increases in the size of the study area, progressing northwards, with apparent expansions in the sampling frame between 1992 and 1993 (northward along eastern Ellesmere Island into the Nares Strait region). Between 1994 and 1995, survey efforts were expanded eastward into Kane Basin proper off the Humboldt Glacier in Northwest Greenland (E. Born, pers. obs.) which inferred from Taylor et al. (2001) was also the case in 1996 and 1997; Figure 10.2). No live-recapture sampling occurred during 1998 – 2011, but we obtained recoveries of harvested bears during this interval. Available information also suggests eastern Kane Basin was covered in the 1990s, though no captures were made there.

We surveyed KB during 25 April – 6 May, 2012; 27 April – 10 May, 2013; and 28 April – 19 May, 2014. Sampling windows were comparable to the 1990s, although surveying in 1992 and 1993 occurred earlier (mid-April) and for shorter windows of time. We sampled sea-ice habitats by helicopter (Bell 206 LongRanger) across the entirety of western and northern Kane Basin, including landfast ice in fjords and nearshore areas as well as offshore pack ice, but excluded more open water habitats of the North Water polynya. We also did not survey the sea ice in the fjords of the populated Qaanaaq area in NW Greenland (i.e., the eastern parts of the North Water polynya) because hunting pressure for marine mammals in these areas is generally high and consequently “resident” polar bears do not exist in the Qaanaaq area (E. Born, pers. obs.). Sampling was primarily completed via directed searching in 2012, with searches focused in areas believed to provide the most suitable polar bear habitat (“adaptive sampling”). In 2013, we completed directed searching and also flew ad hoc transects oriented approximately perpendicular to the coastline, particularly near Greenland, to ensure that effort was well-distributed across the landscape.

In 2012 and 2013, most bears were sampled via physical capture, including chemical immobilization and application of ear tags and lip tattoos as described above. We collected tissue samples from physically captured bears (for genotyping) and recorded additional information including sex, family status, field-estimated age class (cub-of-the-year, yearling, 2-year old, subadult, or adult) and standard morphometric measurements. We completed additional sampling in 2012 and 2013 via remote biopsy darting (Pagano et al. 2014) to collect genetic tissues for subsequent genotyping and analyses (e.g., Herreman and Peacock 2013). Cubs-of-the-year were too small in springtime to be biopsy darted and thus were not sampled when their mothers were biopsy darted (although COY were sampled during physical captures). For bears that were not physically immobilized, sex was confirmed upon genetic analyses (see below).

In connection with immobilization and handling, we deployed satellite transmitters on 36 polar bears in 2012 and 2013 (see Chapter 9; 2012: 6 satellite radio collars on adult females; 10 satellite ear-tags on adults and subadults of both sexes; 2013: 10 satellite radio collars on adult females; 10 satellite ear-tags on adults and subadults of both sexes). This work enabled us to evaluate distribution and habitat use during the genetic mark-recapture sampling and the aerial survey (Chapter 11) and to conduct a post hoc assessment of sampling representativeness during sampling.

We modified our sampling strategy during 2014. We stratified the study area into high- and low-density areas based on our observations of polar bears in 2012 and 2013 (i.e., presumed densities) and searched for bears from systematically spaced transects. This design enabled us to more efficiently allocate effort and reduced the potential for spatial heterogeneity in detection. Systematic sampling also facilitated the simultaneous completion of an aerial survey (Chapter

11) to derive an abundance estimate, based on different methodology, for comparison with the capture-based estimate.

Strata conformed to general landscape features and ice types: the high-density stratum included landfast ice within fjords as well as nearshore pack ice (within ~30 km of the nearest land mass); the low-density stratum included farther offshore pack ice (Figure 10.3). We delineated the landward extent of the study area using current GIS layers from Greenland and Nunavut. We used Moderate Resolution Imaging Spectroradiometer (MODIS; http://modis.gsfc.nasa.gov/) images with 1 km resolution to delineate the extent of available habitat by approximating the edge of the North Water polynya. Because the polynya's boundaries can change rapidly, we delineated the extent of the polynya adjacent to the section surveyed on a particular day using MODIS imagery from that day (when possible) or as to close to that date as possible (when imagery was unclear on that date due to atmospheric conditions). We examined the delineated study area in relation to weekly regional sea-ice charts produced by the Canadian Ice Service (https://www.ec.gc.ca/glaces-ice/) for confirmation. During sampling, we also collected GPS waypoints at the edge of the polynya to verify delineation.

Transects were systematically spaced at 6-km and 18-km intervals in the high- and low-density strata, respectively, based on anticipated encounter rates and available resources. We also sampled during ferry flights (e.g., between survey transects). Survey protocols in 2014 (detailed in Chapter 11) were designed to facilitate the simultaneous collection of data for mark-recapture and the aerial survey. All mark-recapture sampling in 2014 was conducted via remote biopsy darting.

Harvest Recoveries

We used harvest records to compile dead recovery data for polar bears captured in KB and subsequently harvested there or in neighboring subpopulations during 1992 – 2013 (Burnham 1993). Harvest was monitored by the return of tags or lip tattoos during 1992 – 2010 and by genotyping during 2011 – 2014. Data including date and location of recovery and sex and estimated age were recorded for harvested bears and individuals killed in defense of life and property. Reported harvest rate in KB was relatively high during the 1990s (range: 6 – 17 bears / year) but significantly decreased by the mid-2000s (2 – 8 bears / year; Chapter 8), likely due to factors including changes in sea-ice conditions limiting hunter access by use of dog sleds to northeastern KB (E. Born, pers. obs.) and the implementation of a Greenlandic quota system in 2006. Greenland's reporting system also improved with the implementation of the quota (Chapter 8). Previous studies assumed that harvests of all marked bears were reported when natural survival was calculated and, therefore, the reporting rate r was interpreted as the proportion of mortality due to harvest (e.g., Taylor et al. 2005, 2008). However, more recently, genetic data suggested under-reporting of marked bears in the harvest, with decreases in reporting correlated with increasing marker age (Chapter 8).

Genetic Analyses

DNA Extraction – Dried biopsy samples, new and archived tissue samples, and harvest specimens (frozen or in ethanol) were sent to Wildlife Genetics International (Nelson, B.C., Canada) for analysis using protocols previously validated for bears (Kendall et al. 2009). DNA was extracted from ~ 3mm^2 pieces of tissue with QIAGEN DNeasy Blood and Tissue Kits (http://www.qiagen.com/). Most biopsy darting samples consisted of a plug of a skin and sub-cutaneous tissue. This provided ample material for DNA extraction and residual tissue for future analyses. In a small proportion of cases, the available sample consisted of a tuft of hair. DNA was extracted from these hair samples using approximately 10 guard hair roots or 30 pieces of underfur. In a few cases, where a biopsy sample contained no visible tissue, DNA was successfully extracted by soaking the barbed needle from the biopsy dart in the lysis mix (QIAGEN buffer ATL + proteinase K).

Marker Selection – To select markers for the analysis of individual identity, we used allele frequency data from 1,771 polar bears for which complete 20-locus genotypes existed before the genetic mark-recapture began (Government of Nunavut unpublished data). We ranked the 20 microsatellite markers in the dataset by expected heterozygosity. The 8 most variable markers that could be analyzed together in a single sequencer lane were selected for use. These surpassed the required standard for marker variability (HE = 0.80; Paetkau 2003). In addition to the 8 microsatellite markers, we analyzed sex on every sample, using a *ZFX/ZFY* marker. This 9th marker roughly halved the match probability (assuming a balanced sex ratio), even for close relatives, as well as providing replication of sex data for individuals that were sampled more than once.

Genotyping – The analysis of individual identity followed a 3-phase approach. Phase 1 was a first pass of all extracted samples using the 9 selected markers (*G10B, CXX20, G10H, G10P, 145P07, MU50, MU59, G10X and ZFX/ZFY*). Samples that failed at > 6 of 9 markers on the first pass were set aside and did not proceed further in the analyses. Previous experience has shown that such samples are prone to errors and run out of DNA before generating a complete (phase 2) and reproducible (phase 3) genotype (D. Paetkau, pers. comm.).

The first pass was followed by a cleanup phase in which data points that were weak or difficult to read the first time were re-analyzed. During cleanup we used 5 µL of DNA per reaction instead of the 3 µL was used during first pass. At the conclusion of the cleanup phase, the remaining samples (99.5%) had high-confidence scores for all 9 markers. In cases where the genetic sex result contradicted the reported sex based on field assessment, genetic sex was checked using a second independent marker (*amelogenin*; http://www.ncbi.nlm.nih.gov/pubmed/7695123), thus confirming the results, and ruling out the possibility that a mutation at a particular marker was to blame. In all cases, results from the second marker confirmed that the field data was the source of error.

The third and final phase of analysis was error-checking, following the published protocol of reanalyzing the mismatching markers in highly similar pairs of genotypes (Paetkau 2003). This error-check included genotypes from the 4,657 polar bears in the database, plus published data from 473 individuals (Paetkau et al. 1999). The error-checking protocol functions on the principle that when ≥ 2 samples are genotyped from a given individual, and when 1 of those genotypes contains an error, the result is a pair of genotypes which match at all-but-1 marker (a '1MMpair'). Less commonly, 2MM-pairs are created when 2 errors have been made in the genotypes of the samples from a given individual.

An important distinction with this protocol is that it is designed to ensure accurate individual ID — and has been proven to do so with a high degree of efficiency (Kendall et al. 2009) — but it is not intended or expected to correct errors when just one sample has been genotyped from a given individual. In addition to re-analyzing mismatching markers, this protocol also involved the inclusion of additional markers for some samples. Finally, we also searched the dataset for genotype matches that seemed unlikely based on our field data. In each case, three extra markers were added to the genotypes to lower the probability of chance matches between individuals. The extra loci confirmed all of these matches. Once the genotyping and error-checking was complete, we defined an individual for each unique 9-locus genotype.

Marker Power – In addition to the genotyping errors that were targeted during error-checking, DNA-based datasets are prone to a second source of error, when match probabilities are so high that some individuals have identical genotypes. Calculated match probabilities provide no practical insight into the risk of sampling individuals with matching genotypes, because the calculations are so dependent on the assumptions made about the degree of relatedness among the sampled individuals. We therefore used the direct, empirical approach of extrapolation from the observed mismatch curve (Figure 10.4). We expect to see roughly order-of-magnitude decreases in the number of pairs of individuals whose genotypes match at increasing numbers of markers (Paetkau 2003). In our dataset the slope of this curve was reasonably true to that rule of thumb. From this curve, it is estimated that we would have sampled ~ 0.3 0MM-pairs (individuals whose genotypes matched at 9 markers) in this multiyear dataset of 4,657 individuals; a very small risk of error in proportion to the size of the dataset. In addition to reducing the risk of sampling individuals with the same genotype, another benefit to having such a powerful marker system was realized during error-checking, where the amount of time required to reanalyze the mismatching markers underlying 1MM- and 2MM-pairs was trivial in proportion to the scale of the project, because there were so few such pairs.

Statistical Analyses

We analyzed joint live-capture and dead-recovery data from the KB subpopulation with the Burnham (1993) mark-recapture model, which combines the Cormack-Jolly-Seber (CJS) live-recapture model with the Brownie-Seber dead-recovery model to estimate survival (S), recapture (p), reporting (r), and fidelity (F) probabilities. With the Burnham model, live recaptures are assumed to occur (relatively) instantaneously within the study area, whereas dead recoveries can occur year-round between live capture periods and may take place within or outside the live encounter study area. We assumed that harvests prior to April 15 occurred before the live encounter period in year *t*, (i.e., in year *t* - 1); post-April 15 harvests were considered to have occurred after the live encounter period (i.e., year *t*). This treatment of the data resulted in no instances in which a bear was recovered before being captured alive. We acknowledge that there was some temporal overlap of live recapture and dead recovery periods in KB, but for a long-lived species such as polar bears, the exact timing of harvest relative to the live capture sampling period is less important.

We analyzed data and constructed models in program MARK (White and Burnham 1999). We assembled capture histories from the live capture and dead recovery data and included harvest recoveries through 2013. Although ages were estimated with high resolution during the initial 1990s study period, there was uncertainty in field assessment of age during the 2012 – 2014 sampling frame, particularly with biopsy darting. Hence, we identified relatively coarse age classes (*cf.* Taylor et al. 2008, Peacock et al. 2013), including cubs-of-the-year (coy), yearlings (yrl), 2-year olds (2yr), and individuals age 3 and above (age 3+). Because KB is a small subpopulation, capture and recovery data were very sparse, and we identified a limited number of relatively simple candidate sub-model structures.

We hypothesized that survival would differ among age classes and included age structure in all candidate models; however, we constrained yearling survival equal to 2-year old survival due to sparse data. Because coy are fully dependent on their mothers for their survival, we assumed that survival would not vary between male and female coy. However, we expected that survival would differ between sexes for older age classes, largely due to the 2 : 1 male-to-female sex ratio in the harvest, so we examined structures in which (1) *S* differed between sexes for age 3+ bears only and (2) *S* differed between sexes for yrl / 2yr and age 3+ bears (additive effect of sex). Given the sparseness of the data, we did not examine year-to-year variability in *S*, or relationships between *S* and time-varying environmental covariates.

We examined seven sub-model structures for *p* (i.e., estimation is conditional on first capture). Estimates of *p* in the Burnham model reflect both the probability of an animal being located in the sampling area and thus available for recapture, and the probability of the animal being recaptured conditional on its presence in the sampling area (i.e., random temporary emigration is incorporated in *p*; Burnham 1993). We hypothesized that female bears and dependent offspring (ages 0 and 1) may have a different *p* than independent male bears and evaluated models with this sex and age-class structure (family; sub-model structure 1). In addition, we suspected that search effort and sampling protocols may have differed between the two sampling epochs (1992-1997 vs. 2012-2014), so we considered structures with a temporal epoch effect (epoch; 2). Although the data were scant, we hypothesized that inter-annual variability in weather and sea-ice conditions may have resulted in *p* that varied significantly among years, so we also considered a fully time varying structure (time; 3). We considered structures with additive effects between (4) family and epoch and (5) family and time, as well as a structure including (6) an interactive effect between family and epoch. We also evaluated a null *p* sub-model (i.e., constant *p*; 7).

Because some adult females in our 1990s sample were outfitted with satellite collars (n = 12) that may have assisted in locating them, we created a binary radio covariate indicating whether a bear was theoretically available for recapture with the assistance of radio telemetry. We applied the covariate for 2 years post-collaring during the 1990s sampling period, unless there was evidence that the collar was physically removed from the bear. We included the radio covariate in all structures and coded dependent offspring such that they had the same covariate structure as their mother. Satellite collars were not used to locate bears during the 2012 – 2014 period. Some individuals were not successfully genotyped ($n = 25$) because either tissue samples were not located among the archives or the samples were inadequate to facilitate genotyping. These individuals had a reduced p during 2012 and 2013 (when physical capture and biopsy darting both occurred), but no probability of detection during 2014 (when bears only were sampled via biopsy darting). To reflect this, we created a binary 'genotyped' covariate (0 = successfully genotyped; 1 = not genotyped) and included it in all model structures for 2012 and 2013; for non-genotyped individuals, we fixed $p = 0$ in 2014. We also fixed $p = 0$ during 1998 – 2011, when there was no live recapture sampling.

The reporting (r) parameter represents the probability that a dead bear is identified and reported to authorities. Here, r reflected the proportion of mortality that can be attributed to reported harvest (including bears killed to protect life or property). We hypothesized that r would vary among age classes (yrl / 2yr and age 3+) and by sex for age 3+ individuals (sub-model structure 1) due to harvest regulations, including sex-selective harvest (2 males : 1 female). Because recovery data were sparse (≤ 5 total recoveries per year; typically 0 – 2 recoveries per year), we did not consider models with annual variation in r, but we created an alternative structure which included an additive effect for time period (pre-2006; 2006 - 2013) for age 3+ individuals to reflect the changes in harvest and improvements in the Greenlandic reporting system over the past decade (structure 2). There were no records of cubs-of-the-year marked in KB harvested during the first year post marking, so we fixed r_{coy} to 0. Because only harvest data through 2013 were included in analyses, we fixed r to 0 for all age classes in 2014.

We hypothesized that polar bears may permanently emigrate from KB, based on the semi-discreteness of subpopulation boundaries (Taylor et al. 2001) and the spatial distribution of historical recapture and recovery data. Thus, we chose to estimate the F parameter, rather than assume that there was no permanent emigration and fix F to 1, as done in previous studies (e.g., Taylor et al. 2005, 2008, 2009). We considered structures in which (1) F was estimated as constant across all sex and age classes (constant) and (2) F was different for a combined class of coy, yrl / 2yr, and age 3+ females vs. age 3+ males (3+ males).

We constructed the most generalized model (excluding individual covariates) and used the median $\hat{c}$ method, as implemented in Program MARK, to estimate over-dispersion. Because results suggested the data were not significantly over-dispersed (i.e., $\hat{c}$ was approximately 1), we proceeded with model selection via AIC$_c$. Given the relatively small set of candidate sub-model structures, we constructed all possible combinations of candidate sub-models.

We evaluated models via AIC$_c$ and model-averaged parameters for models with ΔAIC$_c$ < 4 (Burnham and Anderson 2002), based on an initial sensitivity analysis. Our estimates of survival reflected harvest mortality, so we derived estimates of natural survival as $S + r * (1 - S)$ (following, e.g., Taylor et al. 2005, 2008, Peacock et al. 2013) and estimated variance via the delta method (following Taylor et al. 2008). This equation relies on several key assumptions. First, it assumes harvest of all marked bears is reported; under-reporting of the harvest, which has been documented (Government of Nunavut, unpublished data), would lead to negative bias in estimates of natural survival. However, this derivation of natural survival also assumes that harvest mortality is completely additive, i.e., no bears that are harvested would otherwise die during a given interval. In contrast to under-reporting of marked bears in the harvest, a violation of the assumption of additive mortality would result in positive bias in estimates of natural survival.

For highly supported models, we used a generalized Horvitz-Thompson estimator, $\widehat{N} = \frac{n}{\hat{p}}$, where n is the number captured in group i and $\hat{p}$ is the recapture probability for group i, to generate estimates of abundance by attribute group (e.g., family group status) for the yrl / 2yr and age 3+ classes. Because some coy were not marked during the 2012 – 2014 sampling period and estimates of n and p did not accurately reflect this age class, we incorporated coy by estimating the number of age 3+ females with coy litters via a Horvitz-Thompson estimator and multiplying by mean observed coy litter size. To obtain an overall estimate of abundance for KB by year, we summed individual estimates across groups. Following previous work (e.g., Taylor et al. 2005, 2008, Peacock et al. 2013), we estimated variances for total abundance estimates and incorporated variances and covariances (calculated in MARK) as well as variance of mean litter sizes via the delta method (Seber 1982, Powell 2007) using R (R Core Team 2015) package emdbook (Bolker 2016). We model-averaged estimates of total abundance using model weights for recapture probabilities and variances obtained with the delta method. We calculated mean overall estimates of abundance by sampling epoch and estimated variance using the delta method. We excluded 1993 – 1994 and 2012 from these mean estimates given the initial expansions of the sampling frame between 1992 and 1995 and the long interval without live recaptures preceding 2012, respectively (i.e., estimation of subpopulation size in 2012 was based on estimated recapture rates of bears marked during the 1990s applied to newly encountered bears in 2012).

10.3. Results

We recorded a total of 277 initial captures, 89 recaptures, and 24 dead recoveries over the course of the 23-year study period (Table 10.1). Markedly more bears were captured in the eastern regions of KB (i.e., off Humboldt Glacier in Northwest Greenland) during 2012 – 2014 than during the 1990s (Figures 10.2 and 10.5). Capture data were particularly sparse during the 1990s, although sampling in 1995 yielded significantly more captures than other years in the 1990s (Table 10.1). Similarly, very few bears were recovered via the harvest during the 2000s (Table 10.1). Notably, no males initially marked in KB during the 1990s were recaptured during 2012 – 2014, and only one male marked in the 1990s was reported in the harvest after 2002. Although no COY were sampled in 2014 (all sampling was conducted via biopsy darting), we observed a total of 23 COY with their mothers that year. In addition, 3 COY with their mothers were not biopsy darted in both 2012 and 2013. Mean observed COY litter size during 2012 – 2014 was 1.60 (SD: 0.5).

The most highly supported models included an additive effect of sex for the yrl / 2yr and 3+ age classes for S and a temporal effect (break at 2006) for r (Table 10.2). Although there was not clear support for specific structures for modeling p, complex (e.g., fully time-varying) structures for p were not supported in model selection, which was not surprising given the sparseness of the data. For model-averaging, we included 12 of 56 total models (cumulative model weight = 0.76).

Estimates of total survival of males were markedly lower than females for both the yrl / 2yr and 3+ age classes, although we note that the additive effect in S was shared across age classes and not estimated separately for yrl/2yr vs. 3+ bears (Table 10.3). This pattern was also evident in estimates of unharvested survival (yrl / 2yr females: 0.74, SE: 0.15; yrl / 2yr males: 0.54, SE: 0.17; age 3+ females, 2006 – 2013: 0.96, SE: 0.04 and 3+ males, 2006 – 2013: 0.88, 0.05). As hypothesized, recent (2006 - 2013) estimates of r were less than 1992-2005 values, although r did not significantly differ among age and sex classes (Table 10.3). Estimates of F suggest relatively strong fidelity to the springtime study area for females and dependent bears (F: 0.98, SE: 0.04) as well as age 3+ males (F: 0.96, SE: 0.07).

Annual estimates of abundance largely reflected the variability in sample sizes among years (e.g., 1995; Table 10.4, *cf*. Table 10.1). The estimated mean total abundance of the KB subpopulation during the 1995 – 1997 period was 224 (SE: 40; 95% CI: 145 – 303). The estimated mean total abundance for 2013 – 2014 was 357 (92; 221 – 493).

10.4. Discussion

We used a combination of physical and genetic mark-recapture techniques, including live recaptures and dead recoveries, to estimate demographic parameters of the Kane Basin polar bear subpopulation over a 23-year study period. Our estimate of abundance from the 1990s (224, 95% CI 145 – 303; averaged over 1995 – 1997) was consistent with previous analyses (164, averaged over 1994 - 1997; Taylor et al. 2008). Although the 2010s point estimate is ~36% greater than the 1990s estimate of Taylor et al. (2008), from the 1990s; this difference is largely attributable to our decision to derive a mean estimate of abundance from only 1995 – 1997. The sampling frame expanded during the 1990s, progressing northward and eastward in incremental steps such that, in the initial years, only a portion of KB was surveyed (Figure 10.2). Hence, we calculated mean abundance estimates by epoch only during periods when sampling was consistent and the sampling frames were generally comparable (1990s: 1995 – 1997). By contrast, Taylor et al.'s (2008) estimate was calculated as the mean estimated from 1994 – 1997; including 1994 in our estimate would reduce our point estimate from 224 to 198.

The 2013 – 2014 estimate of abundance suggests 357 (221 – 493) bears currently use KB in springtime (i.e., the Kane Basin super-population; Kendall et al. 1997), and the current point estimate is higher than the historical estimate. Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of KB abundance for 2013-2014 and the estimate for 1995-1997 was approximately 133 bears (standard deviation of the difference ≈ 80 bears), with 95% of the sampling distribution suggesting that population change between the two time periods could have been positive. This result suggests a stable to increasing subpopulation and is consistent with data on movements (Chapter 9), reproductive output (Chapter 12), and body condition (Chapter 13), suggesting that Kane Basin is currently a healthy subpopulation. However, we encourage caution in interpretation. We attempted to mitigate the impacts of apparent changes in sampling frames, particularly during the 1990s, by excluding 1993 and 1994 from our mean estimate of abundance during the 1990s. However, we were unable to address potential changes in survey effort between the 1995 – 1997 and 2012 – 2014 epochs.

Sampling occurred in the eastern regions of the KB subpopulation (i.e., near the Humboldt Glacier) during 1995 – 1997 and 2012 – 2014. When the eastern parts of Kane Basin (i.e., the areas east of the mid-sector line in the Nares Strait-Kane Basin area off the Humboldt Glacier) were surveyed in 1994 and 1995, only few signs of polar bear activity (i.e., tracks) were observed there and consequently only a few bears were tagged (0 in 1994 and 4 in 1995; E. Born, pers. obs., Taylor et al. 2001). Similarly, no bears were found and tagged there in 1996 and only 3 in 1997 (Taylor et al. 2001). The apparent very low densities of polar bears in eastern KB was assumed to reflect a long-term avoidance response because eastern KB has been hunted relatively intensively by hunters from the Qaanaaq region and in particular after it no longer became permitted for Greenland hunters to hunt polar bears in Canadian territory in the late

1960s. The presence of ringed seals in eastern KB was noted during the surveys in the 1990s and it was concluded that these areas were favourable habitat for polar bears (E. Born, pers. obs., Taylor et al. 2001).

However, it cannot be precluded that, this difference between the 1990s and the 2000s in the spatial distribution of bears, to a certain extent reflect some differences in sampling protocols, including increased survey intensity near the Humboldt Glacier and more uniform distribution of effort during 2012 – 2014. However, during both periods relative allocation of survey effort to a certain extent was decided based on assumptions of what was suitable polar bear habitat (i.e., areas with anticipated polar bear occurrence and/or areas with observed signs of polar bear habitat). In the 2000s large areas in central and southern KB with relatively open pack ice were not surveyed although satellite telemetry (Chapter 2) and aerial surveys (Heide-Jørgensen et al. 2013) indicate the presence of polar bears in this habitat albeit likely few. Whereas in the 1990s when the spring sea ice in KB was more consolidated (E. Born, pers. obs; Chapter 9) areas with little or no signs of polar bear activity in the eastern parts were surveyed less intensively.

It should be mentioned that sea-ice dynamics in Kane Basin also have changed since the 1990s (Chapter 9), limiting access of hunters from Greenland to the eastern parts of the region (Born et al. 2011) possibly resulting in an increased occurrence of polar bears in this area. Hence, we hypothesize that these differences in sea-ice dynamics and associated change in hunting pressure have led to the apparent shift in the distribution of bears toward the eastern parts of the Nares Strait-Kane Basin region.

We note that the Horvitz-Thompson estimator used to generate estimates of abundance in Kane Basin yielded biologically implausible rates of growth from 1994 to 1995 (119 – 318), 1995 to 1996 (318 – 189), and 2012 to 2013 (221 – 328; Table 10.4). Although the apparent changes in sampling frame (and thus the definition of the effective study population) from 1994 to 1995 may contribute to this finding in part, this result is primarily an artifact of the estimator itself. Horvitz-Thompson (H-T) estimators are calculated as $\widehat{N} = \frac{n}{p}$ for each group (e.g., age class and sex), and the total abundance estimate is derived by summing estimates across all groups. As such, H-T estimators are sensitive to sample size, particularly if recapture probabilities are estimated as temporal constants. Because data for this analysis were very sparse and models specifying inter-annual variation in estimates of recapture probability were not supported, our Horvitz-Thompson estimates of abundance are influenced by variation in annual sample size of captured bears (n). We attempted to address this issue by integrating annual random effects for estimating recapture probability with complementary Markov Chain Monte Carlo (MCMC) analyses, but this approach did not resolve the issue. Given this limitation, we do not recommend interpreting inter-annual variation in estimates of abundance, and suggest that the larger estimate of abundance in 2013-2014 may be partially attributed to larger sample sizes in those years, compared to 1995-1997.

Our estimates of survival for dependent bears and age 3+ females are consistent with previous work in Kane Basin (Taylor et al. 2008), and estimates of unharvested survival rates of independent females appear capable of supporting positive subpopulation growth (Regehr et al. 2015). Our estimates of age 3+ male survival (present analysis: age 3+ males, 0.87, SE 0.06) are lower than previous work in KB (Taylor et al. 2008: age 5+ males, 0.96, SE: 0.05) but consistent with estimates of adult male survival derived in some other studies (e.g., Stirling et al. 2011, Peacock et al. 2012). We further note that data on males were particularly scant in this study (Table 10.1): no male bears initially marked in the 1990s were subsequently recaptured in the

2000s, and very few individuals were reported in the harvest over the past 15 years. We are uncertain if the disparity in estimates of male survival between our study and Taylor et al. (2008) reflects differences in modeling approaches, such as the broader age class designations in the present study and consideration of different model structures (unlike Taylor et al. [2008], we did not consider a structure in which S was estimated as constant between the sexes for age 3+ individuals); our inclusion of longer-term data; a disproportionate impact of the changing environment on males; reduced fidelity of males to the study area that was not effectively captured by our model-based estimates of the fidelity (*F*) parameter given the paucity of recovery data; under-reporting of male bears in the harvest; or some combination thereof.

Although we calculated estimates of natural survival following previous studies (e.g., Taylor et al. 2005), we note that the formula used to derive these estimates [$S + r * (1 - S)$] makes simplifying assumptions that can introduce bias into estimates of unharvested *S* under some conditions, although the impacts of using this vs. an alternative equation is likely minimal for KB data due to high estimates of *S* and low estimates of *r*. The documented under-reporting of marked bears in the harvest leads to an underestimation of natural survival, but this may be offset, to some extent, by a likely violation of the assumption that harvest mortality is completely additive. For example, Taylor et al. (2008) estimated natural survival for both adult females and males to be 0.997, meaning that virtually all mortality of adult (age 5+) bears in Kane Basin during 1992 - 1998 resulted from harvest, and <1 in 300 adult bears would die annually in the absence of harvest. Although Taylor et al. (2008) did not report their estimates of *r*, back calculating from survival rates in their Table 3 yields unrealistically high estimates of *r* = 0.91 for adult females and *r* = 0.93 for adult males, so their estimates of natural and harvest mortality should be treated cautiously.

Changing sea-ice conditions, a reduction in accessibility of Kane Basin to hunters from Canada and (in the Humboldt Glacier region) Greenland, and the implementation of a quota in Greenland have contributed to a net reduction in harvest since the 1990s (Chapter 8). This decline is reflected in estimates of the reporting parameter, as *r* is estimated lower for the period from 2006 – 2014 than 1992 – 2005 (Table 10.3). We note, however, that under-reporting of harvest, which anecdotally appears to increase with greater marker age (Chapter 8), also may contribute to lower estimates of *r* during 2006 – 2013, especially given the 14-year interval without live recaptures.

Despite the sparseness of the data and the unusual study design (6-year and 3-year live capture sessions connected by a 14-year period with dead recoveries only), we were able to generate estimates of *F* that seem biologically realistic (age 3+ males: 0.96; females and dependent bears: 0.98): polar bears show strong seasonal fidelity to the region in which they were captured, but a small proportion of individuals permanently emigrate to other subpopulations. These estimates appear consistent with findings from satellite telemetry data and capture records (Taylor et al. 2001, Chapters 2 and 9).

Our study indicates that there has been no decline in the size of the KB subpopulation. These findings are in accordance with TEK (Born et al. 2011).

Using estimates of subpopulation size in KB and vital parameters from the 1990s, York et al. (2016) modeled a decline in KB with an estimated subpopulation size of zero in 2013. According to York et al. (2016:9,18) the projected decline is consistent with TEK. It is mentioned that KB has been subject to chronic long-term overharvest and would not persist if it did not receive immigrants from adjacent subpopulations (Ibid.). As basis for the TEK information York et al. (2016) cite COSEWIC (2008) and M. Taylor (pers. comm. 1986-2008) in the text, and COSEWIC (2008), CWS (2009) and PBTC (2014) in their table 2b but not a comprehensive TEK study in which NW Greenland polar bear hunters were interviewed (Born et al. 2011).

During this interview survey in Greenland experienced polar bear hunters who had been hunting in Kane Basin were of the opinion that polar bears in this region had expanded their range. Previously the hunters had to travel north to Washington Land (*ca.* 80° N) to find bears, whereas nowadays they only have to go as far as Inglefield Land (*ca.* 78° 30ʾ N) to hunt polar bears in the eastern Nares Strait-Kane Basin region. Their reason for this was because "the bears have come closer" (Born et al. 2011:75,79). It was mentioned that previously polar bears were scarce in the eastern Kane Basin area (i.e., in front of the Humboldt Glacier) but now had expanded their range from Ellesmere Island eastward to the Nares Strait-Kane Basin region (ibid:80). Some of the interviewees were of the opinion that this change represented an increase in the number of polar bears. It was also mentioned that the hunting pressure in Kane Basin had decreased because poor sea-ice conditions (i.e., lack of dense sea ice) had made travels with dog sleds north more difficult (Born et al. 2011). Hence, information from experienced polar bear hunters in NW Greenland indicates that the KB-polar bear subpopulation has expanded its distribution area and increased in size which is in accordance with our study.

This study and concurrent research in the neighboring Baffin Bay subpopulation (Chapter 5) represent the first attempts to implement genetic mark-recapture for polar bears at a subpopulation-wide scale. The ability to successfully genotype bears from archived tissue and samples obtained via remote biopsy darting, combined with the ability to analyze data in well-established mark-recapture models, suggest that this approach is a promising tool for future polar bear inventories. However, there are some limitations of the technique including a generalized age structure as bears are identified from the air. We opted for a conservative approach when designating age classes because there was inherent uncertainty in estimating age class remotely rather than aging via physical examination or with annular rings from an extracted tooth. However, our ability to accurately classify bears by age-class remotely (Chapter 5) suggests that future studies may be able to increase the resolution of age classification for obtaining estimates of survival (and other parameters), thus enhancing the utility of the technique.

10.5. Literature Cited

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Atwood, T., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. E. Bromaghin. 2015. Evaluating and ranking threats to the long-term persistence of polar bears. U.S. Geological Survey Open-File Report 201-1254. U.S. Geological Survey, Reston, Virginia, USA.

Bolker, B. 2016. Ecological Models and Data in R. R package version 1.3.9.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological

Applications 25:634–651.

Burnham, K. P. 1993. A theory for combined analysis of ring recovery and recapture data. Pages 199–213 *in* J. D. Lebreton and P. M. North, editors. Marked individuals in the study of bird populations. Birkhauser-Verlag, Basel, Switzerland.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Comiso, J. C., C. L. Parkinson, R. Gertsen and L. Stock. 2008. Accelerated decline in the Arctic sea ice cover. Geophysical Research Letters 35:L01703.

COSEWIC [Committee on the Status of Endangered Wildlife in Canada]. 2008. COSEWIC assessment and update status report on the polar bear Ursus maritimus in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, vii + 75 pp.

CWS [Canadian Wildlife Service]. 2009. Nunavut Consultation Report. Consultations on the Proposed Listing of the Polar Bear as Special Concern under the Species at Risk Act. Conducted February-April 2009, Iqaluit, Nunavut, 247 pp.

Heide-Jørgensen, M. P., L. M. Burt, R. G. Hansen, N. H. Nielsen, M. Rasmussen, S. Fossette, and H. Stern. 2013. The significance of the North Water Polynya to Arctic top predators. Ambio 42:596–610.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. Journal of Wildlife Management 73:3–17.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock’s robust design. Ecology 78:563–578.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications 18:S97–S125.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D.

Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. Molecular Ecology 12:1375–1387.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014: Status table. <http://pbsg.npolar.no/export/sites/pbsg/en/docs/status-table-2014.pdf>. Accessed 4 May 2016.

PBTC [Canadian Polar Bear Technical Committee]. 2014. 2014 Polar Bear Technical Committee Status Table, unpublished data.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Powell, L. 2007. Approximating variance of demographic parameters using the delta method: A reference for avian biologists. Condor 109:949–954.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Sahanatien, V., and A. E. Derocher. 2012. Monitoring sea ice habitat fragmentation for polar bear conservation. Animal Conservation 15:397–406.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Macmillan, New York, New York, USA.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stirling, I., C. Spencer, and D. Andriashek. 1989. Immobilization of polar bears (*Ursus maritimus*) with Telazol® in the Canadian Arctic. Journal of Wildlife Diseases 25:159–168.

Stirling, I., T. L. McDonald, E. S. Richardson, E. V. Regehr, and S. C. Amstrup. 2011. Polar bear population status in the northern Beaufort Sea, Canada, 1971-2006. Ecological Applications 21:859–876.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. Global Change Biology 18:2694–2706.

Stroeve, J. C., M. C. Serreze, M. M. Holland, J. E. Kay, J. Malanik, and A. P. Barrett. 2012. The Arctic's rapidly shrinking sea ice cover: A research synthesis. Climatic Change 110:1005–1027.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46:S120–S139.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution 6:2897-2924.

Table 10.1. Summary table of live captures and dead recoveries during the mark-recapture study of the Kane Basin polar bear subpopulation in Nunavut, Canada, and Greenland, 1992 – 2014. Shaded cells indicate that data are not possible due to an absence of marking or recapture.

	Initial captures						**Live recaptures**				**Dead recoveries**					
	Females			*Males*			*Females*		*Males*		*Females*			*Males*		
	Coy	**Yrl / 2yr**	**3+**	**Coy**	**Yrl / 2yr**	**3+**	**Yrl / 2yr**	**3+**	**Yrl / 2yr**	**3+**	**Coy**	**Yrl / 2yr**	**3+**	**Coy**	**Yrl / 2yr**	**3+**
1992	4	0	7	2	0	3					0	0	1	0	0	0
1993	1	3	6	3	1	8	0	2	0	0	0	0	0	0	0	0
1994	2	0	9	3	0	4	1	3	2	2	0	0	0	0	0	0
1995	12	3	21	5	2	13	0	7	0	6	0	0	1	0	0	1
1996	5	2	8	2	2	4	1	7	0	5	0	0	0	0	0	1
1997	0	4	4	3	1	3	1	8	0	8	0	1	1	0	0	2
1998												0	3		0	2
1999													2			2
2000													0			0
2001													1			0
2002													0			1
2003													2			0
2004													1			0
2005													0			0
2006													0			0
2007													0			0
2008													0			0
2009													0			0
2010													0			1
2011													0			0
2012	2	3	19	1	4	11		2		0	0	0	1	0	0	0
2013	6	4	20	2	2	19	0	9	1	3	0	0	0	0	0	0
2014	0	2	21	0	1	10	2	12	0	7	0	0	0	0	0	0
Totals	**32**	**21**	**115**	**21**	**13**	**75**	**5**	**50**	**3**	**31**	**0**	**1**	**13**	**0**	**0**	**10**

Table 10.2. Model selection results (< ΔAIC_c 4) from analysis of mark-recapture-recovery data from the Kane Basin polar bear subpopulation, 1992 – 2014. Coy = cubs of the year. Yrl = yearlings and 2-year olds. 3+ = bears aged 3 and older. For p, family = females / dependent bears and independent males (2 age / sex classes); and epoch = sampling period (1992 – 1997; 2012 – 2014). For r, time = 1992 – 2005 and 2006 – 2013.

Model Structures							
S	p	r	F	**Parameters**	**ΔAICc**	**AICc Weights**	**Deviance**
coy, yrl, 3+(add sex with yrl)	Family	yrl, 3+(sex + time)	Constant	13	0	0.23	723.71
coy, yrl, 3+(add sex with yrl)	Family	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	14	1.22	0.13	722.76
coy, yrl, 3+(add sex with yrl)	family + epoch	yrl, 3+(sex + time)	Constant	14	1.23	0.13	722.77
coy, yrl, 3+(add sex with yrl)	Constant	yrl, 3+(sex + time)	Constant	12	1.88	0.09	727.74
coy, yrl, 3+(sex)	Family	yrl, 3+(sex + time)	Constant	13	2.19	0.08	725.90
coy, yrl, 3+(add sex with yrl)	family + epoch	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	15	2.57	0.06	721.94
coy, yrl, 3+(add sex with yrl)	family * epoch	yrl, 3+(sex + time)	Constant	15	2.62	0.06	721.99
coy, yrl, 3+(sex)	Family	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	14	2.83	0.06	724.38
coy, yrl, 3+(sex)	Constant	yrl, 3+(sex + time)	Constant	12	3.06	0.05	728.92
coy, yrl, 3+(add sex with yrl)	Epoch	yrl, 3+(sex + time)	Constant	13	3.37	0.04	727.08
coy, yrl, 3+(sex)	family + epoch	yrl, 3+(sex + time)	Constant	14	3.45	0.04	724.99
coy, yrl, 3+(add sex with yrl)	Constant	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	13	3.90	0.03	727.61

Table 10.3. Model averaged (<Δ 4 AICc) parameter estimates for the Kane Basin polar bear subpopulation obtained from mark-recapture study, 1992 – 2014.

Parameter	Class	Estimate (SE)
Total Survival (*S*)		
	Cubs of the year	0.45 (0.15)
	Yearlings / 2-year old females	0.73 (0.13)
	Yearlings / 2-year old males	0.52 (0.17)
	3+ females	0.95 (0.04)
	3+ males	0.87 (0.06)
Reporting (*r*)		
	Yearlings / 2-year olds	0.04 (0.04)
	3+ females, 1992 – 2005	0.42 (0.26)
	3+ females, 2006 - 2013	0.09 (0.08)
	3+ males, 1992 – 2005	0.32 (0.12)
	3+ males, 2006 – 2013	0.06 (0.05)
Fidelity (*F*)		
	Cubs of the year, yearlings, 2-year olds, and 3+ females	0.98 (0.04)
	3+ males	0.96 (0.07)

Table 10.4. Model averaged (<Δ 4 AICc) estimates of abundance ($\hat{N}$ ± SE; [95% Confidence Interval]) of the Kane Basin polar bear subpopulation from mark-recapture study, 1992 – 2014.

1993	1994	1995	1996	1997	2012	2013	2014
120 ± 19 (83-156)	119 ± 21 (77-160)	318 ± 53 (214-429)	189 ± 36 (119-259)	164 ± 28 (110-218)	221 ± 41 (141-301)	328 ± 60 (211-445)	385 ± 78 (233-537)

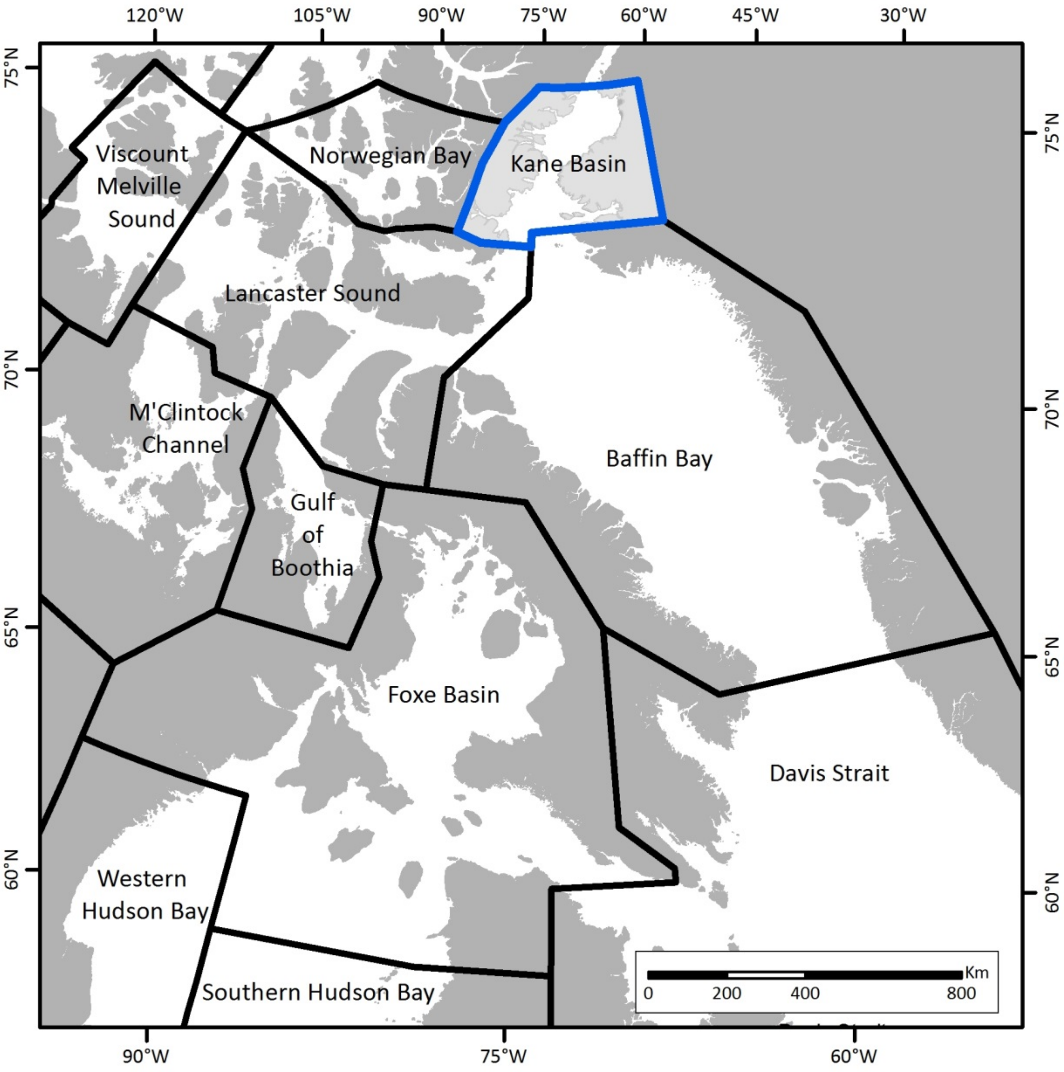


Figure 10.1. The Kane Basin polar bear subpopulation is located between Nunavut, Canada and Greenland and is regarded as belonging to the Arctic Archipelago region.

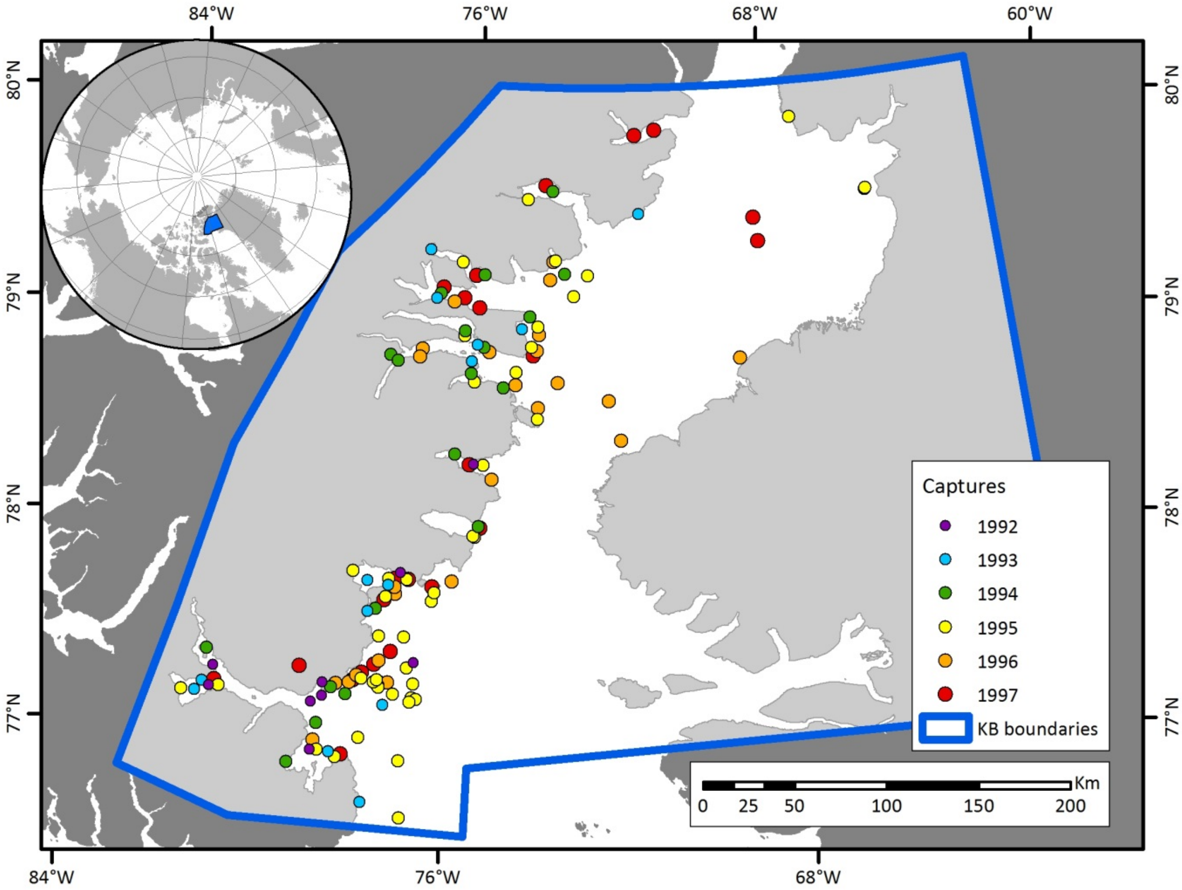


Figure 10.2. Locations of polar bears captured in the Kane Basin subpopulation during springtime, 1993 – 1995 and 1997. Kane Basin is highlighted in blue in the inset.

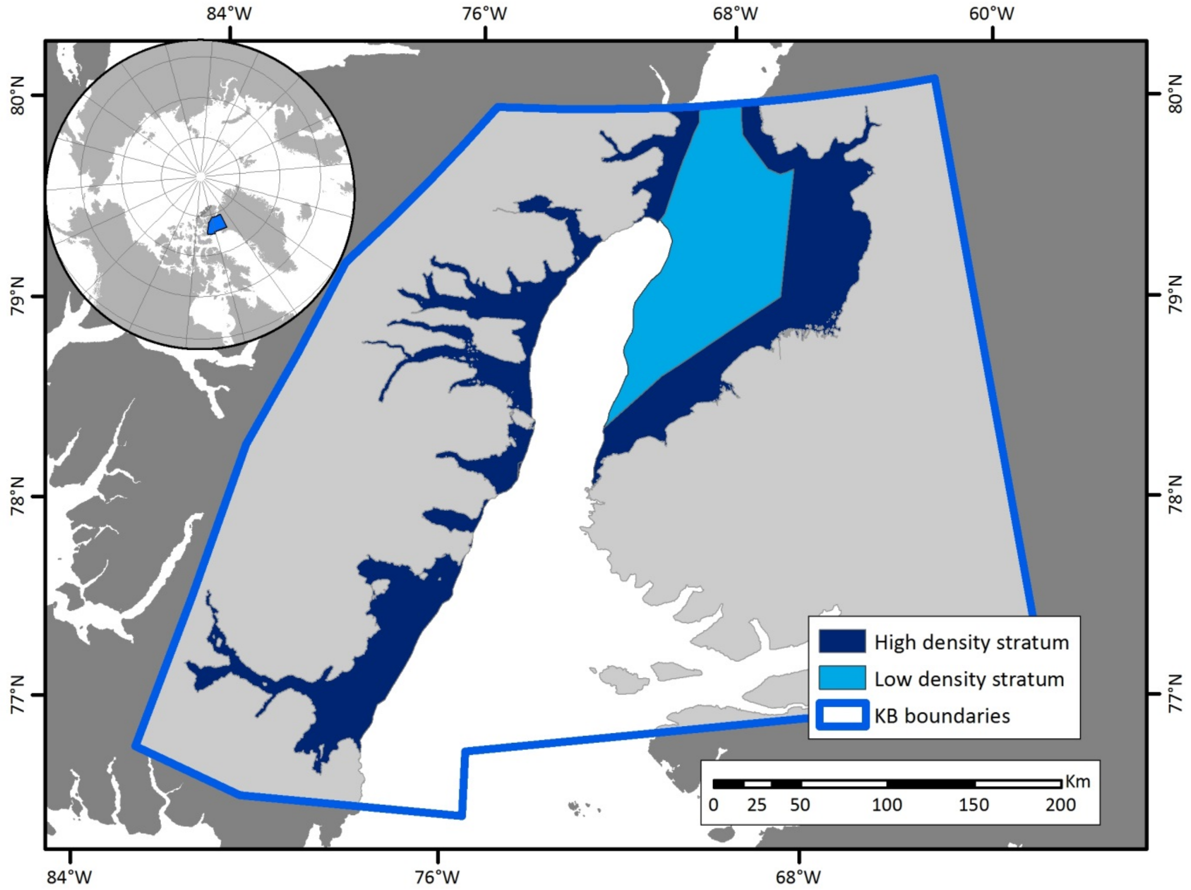


Figure 10.3. Sampling strata for genetic mark-recapture and aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

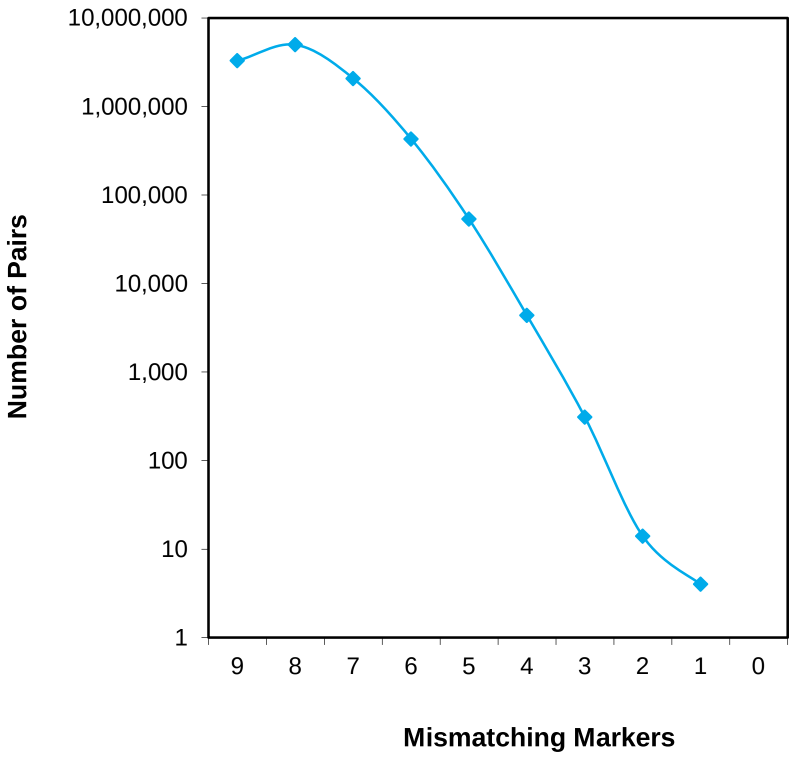


Figure 10.4. 9-locus mismatch distribution for 4,657 polar bears from Nunavut and the Greenland side of the Baffin Bay and Kane Basin polar bear subpopulations.

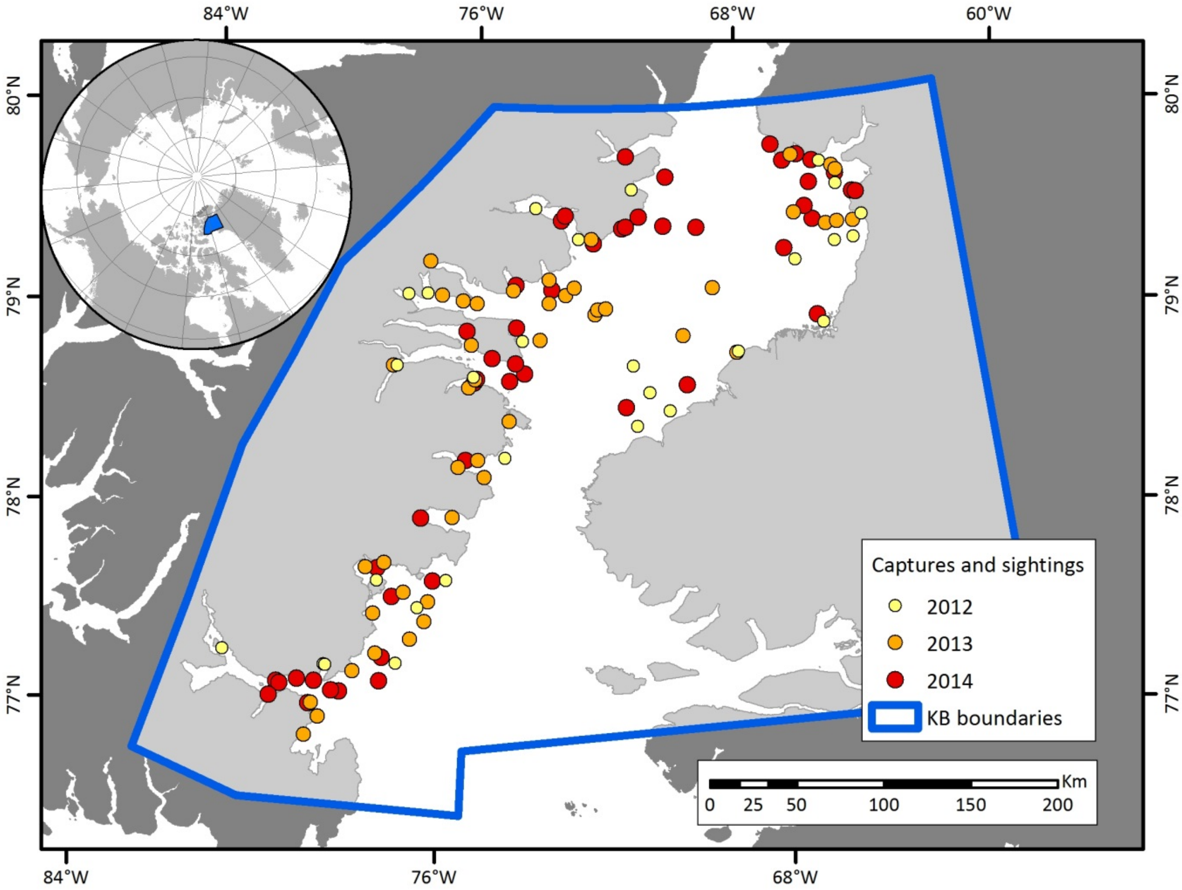


Figure 10.5. Locations of polar bears sighted in Kane Basin during research in April and May, 2012 – 2014. Kane Basin is highlighted in blue in the inset. The North Water polynya varied among years, but in general, included the south-central portion of the subpopulation in all years. We did not sample sea ice in southeastern Kane Basin due to logistical constraints presented by the polynya and anticipated low densities.

CHAPTER 11

AERIAL SURVEY OF POLAR BEARS IN KANE BASIN

KEY FINDINGS

- The estimate of abundance based on the springtime 2014 aerial survey in KB was 206 bears (95% lognormal CI: 83 - 510). However, due to insufficient coverage of offshore polar bear habitat this estimate is likely negatively biased.
- Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of total abundance 2013-2014 from the MR study (357 bears, 95% CI = 221 – 493) and the aerial survey estimate was approximately 151 bears (standard deviation of the difference ≈ 127 bears), with 88% of the sampling distribution suggesting that the difference was positive (i.e., that the MR estimate was at least one bear larger than the aerial survey estimate).
- Differences between MR and aerial survey point estimates in KB require caution when comparing results from different techniques conducted during springtime. Aerial surveys yield a snapshot estimate of abundance, whereas MR generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period. We suggest that the MR estimate is appropriate for use in management.
- As shown also in other areas of the Arctic aerial surveys provide a useful tool for inventorying polar bear subpopulations and the method has been used on even larger subpopulations than KB in remote areas (e.g. the Barents Sea).
- The springtime aerial survey was successfully implemented due to the small geographic area and a period of good weather, but precision could be improved by increasing survey effort to better estimate the detection function and by ensuring that the entire range of the subpopulation is covered.
- Aerial surveys of polar bears that also range in areas with offshore loose drift ice and open water, like Kane Basin, should be conducted from fixed-winged aircraft with a longer endurance than the single-engine helicopter used in the present study. This allows for offshore polar bear habitat to be monitored and will result in a more accurate estimate of abundance.

11.1. Introduction

Physical mark-recapture has formed the basis for demographic studies of polar bears throughout the North American Arctic (e.g., Taylor et al. 2005, 2008, Peacock et al. 2013).

Capture-based research has generated information on abundance, vital rates, and harvest management, and facilitated a variety of other studies, including assessments of body condition (e.g., Rode et al. 2012), movements, habitat use and spatial ecology (e.g., Durner et al. 2009, Cherry et al. 2013), and diet (e.g., Thiemann et al. 2008). Over the past decade, however, jurisdictions have invested substantial resources in the development and implementation of less invasive monitoring techniques, in part to better address social concerns regarding wildlife handling and immobilization. Genetic mark-recapture, one such alternative method, has been used to estimate the number of polar bears using whale carcasses in Alaska (Herreman and Peacock 2013) and, more recently, to estimate the abundance and associated vital rates for the Baffin Bay and Kane Basin subpopulations (see Chapters 5 and 10, respectively). Aerial surveys also have been widely implemented, including studies conducted over land in seasonally ice-free subpopulations (Stapleton et al. 2014, 2016, Obbard et al. 2015) and over land and sea ice in the Barents Sea (Aars et al. 2009). Aerial surveys yield less detailed information on sex, age, body condition, and vital rates than both physical and genetic mark-recapture methods, but they can enable more frequent monitoring, an important consideration in the face of a rapidly changing Arctic.

Our objective was to evaluate the feasibility of estimating abundance with an aerial survey flown over springtime sea ice in the Kane Basin (KB) subpopulation. We designed and implemented the aerial survey to be conducted alongside a concurrent mark-recapture study in the KB subpopulation during 2014. This protocol ensured consistency in the sampling frames and study periods. It also allowed us to derive independent estimates of abundance from the two techniques, enabling us to directly compare and assess the results of the 2 methods. This important step is necessary to properly integrate population estimates derived from different survey techniques. Although research elsewhere in has facilitated broad comparisons between mark-recapture and aerial survey methods (Western Hudson Bay – Stapleton et al. 2014, Lunn et al. 2016; Southern Hudson Bay – Obbard 2008, Obbard et al. 2015), the work in Kane Basin represents the first study in which an aerial survey was designed and implemented with a sampling frame identical to a simultaneous mark-recapture study. Aerial surveys yield snapshot estimates of abundance (i.e., the number of bears occupying the survey area during the study period; Buckland et al. 2001), whereas mark-recapture generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period (including individuals that are currently outside the survey area due to temporary emigration; Kendall et al. 1997). Because there is a lack of geographic closure among polar bear subpopulations such that they are only partially discrete (Taylor et al. 2001, Chapter 9), we hypothesized that our aerial survey-based estimate would be smaller than our mark-recapture-based estimate.

11.2. Materials and Methods

Study Area

The KB subpopulation covers ~150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (boundaries evaluated in Taylor et al. 2001; Figure 10.1). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq areas). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the Baffin

Bay (BB) and Lancaster Sound (LS) subpopulations, and to the west by Norwegian Bay (NW; PBSG 2010). The KB subpopulation is regarded as belonging to the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, sea-ice conditions have changed markedly in the Kane Basin region in recent decades (Born et al. 2011; Figure 9.2). KB is partially connected to neighboring subpopulations; particularly notable interchange occurs with BB and LS. The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a barrier between KB and BB – LS.

Field Sampling

Using a helicopter (Bell 206 LongRanger), we implemented a line-transect aerial survey over springtime sea ice in the KB subpopulation during 28 April – 12 May 2014. To efficiently allocate effort and ensure that the study area was sampled as comprehensively as possible, we stratified the subpopulation into high- and low-density areas based on observations of polar bears during 2012 and 2013 mark-recapture surveys (i.e., presumed densities; see Chapter 10). Strata conformed to general landscape features and ice types: the high-density stratum included landfast ice along the coastline and within fjords as well as nearshore pack ice within ~30 km of the nearest land mass (~18,870 km^2), whereas the low-density stratum included pack ice located farther offshore (~9,110 km^2; Figure 11.1). Since the survey was conducted after adult females had left dens, we assumed that no bears were located on land during the study period. We used

GIS layers from Greenland and Nunavut to delineate the landward extent (i.e., coastline) of the study area. We delineated the extent of available habitat by approximating the edge of the North Water polynya with Moderate Resolution Imaging Spectroradiometer (MODIS; http://modis.gsfc.nasa.gov/) images (1 km resolution). The polynya's boundaries can change rapidly, so we delineated the extent of the polynya adjacent to the section surveyed on a particular day using MODIS imagery from that day, or from the closest date possible when same-day imagery was unclear due to atmospheric conditions. We also examined the delineated study area in relation to weekly regional sea-ice charts produced by the Canadian Ice Service (https://www.ec.gc.ca/glaces-ice/). During sampling, we collected GPS waypoints at the edge of the polynya to verify delineation. We did not sample in the polynya due to safety considerations. Polar bears occur in the polynya area (Heide-Jørgensen et al. 2013) but ice conditions in spring 2014 suggested that it was not suitable springtime habitat for polar bears due to its thin, forming (i.e., new and grey) ice and expansive open water (*cf*. Sahanatien and Derocher 2012). We also did not survey the sea ice in the fjords (~3,245 km^2; Figure 11.1) of the populated Qaanaaq area in NW Greenland (i.e., in the eastern parts of the North Water polynya) because hunting pressure for marine mammals in these areas is generally high and consequently "resident" polar bears do not exist in the Qaanaaq area (Born et al. 2011, E. Born, pers. obs.).

Aerial transects were systematically spaced at 6-km and 18-km width intervals in the high- and low-density strata, respectively, based on anticipated encounter rates and available resources. We arranged transects in an east – west direction in open areas, but oriented them perpendicular to fjords (i.e., across the widths of fjords) to improve variance estimation (i.e., more numerous short transects) and reduce bias (i.e., sighting distances did not reflect potential density gradients, with highest densities along the sides of fjords; Figure 11.2).

During line-transect sampling, we surveyed at an altitude of ~120 m and groundspeed of ~150 km / hr. We sampled from most planned transects and included some ferry flights (during which we sampled) that were random with respect to the distribution of bears and presumed density gradients (i.e., highest densities near the polynya edge and along the sides of fjords) in analyses. Although many groups were observed during flights between consecutive transects, these typically occurred near the sides of fjords. As such, observations may have reflected a density gradient as well as the probability of detection and were thus inappropriate to include in distance sampling analyses (Stapleton et al. 2014).

We collected aerial survey data with mark-recapture distance sampling protocols (Laake and Borchers 2004, see also Stapleton et al. 2014, 2016). Two front (including the pilot) and two rear observers comprised the first and second capture periods, respectively, and teams of observers worked independently until both groups were afforded a full opportunity to observe a bear. After announcing a sighting, we flew off-transect to record the bear's initial location with a GPS, and we later estimated distance from transects in a GIS (Marques et al. 2006). During off-transect flights, we flew to within ~5 – 10 m of bears to obtain a tissue sample via biopsy darting for genetic analysis (see Chapter 10) and to estimate sex and age class of the bear. For each sighting, we recorded 3 covariates that potentially impacted detection probability: 1) habitat structure within a 30-m radius (smooth / low structure or moderate to high structure; i.e., smooth versus rough ice); 2) visibility (good or poor, due to fog, glare or precipitation); and 3) light conditions (i.e., cloud cover; clear: 0 – 25%; partly cloudy: 25 – 50%; mostly cloudy: 50 – 75%; or overcast: 75 – 100%).

Statistical Analyses

We analyzed line-transect data using distance sampling, which fits a function to observational data to describe how detection changes with increasing distance from the sampling transect (Buckland et al. 2001). We initially intended to use double observer (i.e., mark-recapture) distance sampling for analyses, but small sample sizes precluded this approach. We defined clusters as discrete groups of bears with non-independent detection probabilities (i.e., an adult female with 1 or more offspring or a breeding pair). We first examined a left-truncated data set (i.e., 75 m was subtracted from all observations; observations within 75 m were censored to account for blind spots directly beneath the helicopter; e.g., Borchers et al. 2006, Stapleton et al. 2014) to evaluate distance sampling's fundamental assumption of complete detection on the transect line (Buckland et al. 2001). Because these results indicated that the probability of detection by at least one observer was >96% at the adjusted transect line, we considered this assumption to be approximately met and proceeded with analyses including all observations (i.e., data were not left-truncated).

We completed analyses in the mark-recapture distance sampling (MRDS) engine of Program DISTANCE 6.2 (Thomas et al. 2010) and modeled the survey data as a single-observer study. We examined half-normal and hazard rate key functions and used multiple covariate distance sampling (Marques and Buckland 2003) to include a maximum of 1 covariate per model due to sample size constraints. We condensed light conditions into a binary covariate (0 – 25% cloud cover; >25% cloud cover) due to underrepresentation of some values. We considered each transect the sampling unit for variance estimation and used the Innes et al. (2002) method to estimate variance associated with global density and overall abundance.

11.3. Results

We surveyed 4,160 km of transects, including 3,850 km along 234 transects in the high-density stratum and 610 km along 14 transects in the low-density stratum. We observed 29 groups of polar bears (Figure 11.2), including 49 total bears (30 independent bears); cub-of-the-year and yearling litter sizes in this sample averaged 1.55 (SD: 0.5, n = 11) and 1.0 (SD: 0.0, n = 2), respectively. We right-truncated sightings data at 1,400 m to improve model fit and parsimony (Buckland et al. 2001), censoring one observation of an independent bear at >3,500 m, leaving 28 groups for estimating the detection function and abundance; 27 of these sightings occurred in the high-density stratum.

Sighting distance was not correlated with polar bear group size (r = -0.10, P = 0.61), so we used mean group size for abundance estimation. Histograms summarizing sightings distances indicated strong-support for a distance-based detection function (Figure 11.3), and all highly supported distance sampling models indicated adequate goodness-of-fit (chi-squared, Cramér-von Mises and Komolgorov-Smirnov tests: P >0.05). The most highly supported model (half-normal key function) suggested that light conditions (cloud cover) affected detection probability (Figure 11.4). However, the small number of observations (see Buckland et al. 2001: at least 60 – 80 sightings are recommended for estimating the detection function) resulted in uncertainty in density and abundance estimation, and a model with a hazard rate key function had nearly equivalent support and estimated much higher densities (Figure 11.3, Table 11.1). Thus, we elected to model-average (Burnham and Anderson 2002) the 2 most highly supported models and obtained a subpopulation-wide estimate of 206 bears (SE: 101; 95% lognormal CI: 83 – 510; CV: 49%) in 2014.

11.4. Discussion

The estimate of abundance based on the springtime 2014 aerial survey in KB was 206 bears (95% lognormal CI: 83 - 510). However, due to insufficient coverage of offshore polar bear habitat (vast areas of offshore habitat in the North Water Polynya was not surveyed) this estimate is likely negatively biased. The estimate of abundance obtained from the aerial survey was negatively biased by about 30% or more (see below).

Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of total abundance 2013-2014 from the MR study (357 bears, 95% CI = 221 – 493) and the aerial survey estimate was approximately 151 bears (standard deviation of the difference ≈ 127 bears), with 88% of the sampling distribution suggesting that the difference was positive (i.e., that the MR estimate was at least one bear larger than the aerial survey estimate).

Differences between MR and aerial survey point estimates in KB require caution when comparing results from different techniques conducted during springtime. Aerial surveys yield a snapshot estimate of abundance (i.e., the number of bears occupying the survey area during the study period), whereas MR generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period (including individuals that are currently outside the survey area due to temporary emigration) (Kendall et al. 1997). We suggest that the MR estimate is appropriate for use in management.

This finding reinforces that boundaries between subpopulations are not discrete, a result consistent with satellite telemetry (Taylor et al. 2001, Chapter 9) and capture and harvest records (e.g., Peacock et al. 2012, Chapter 10). Interchange among subpopulations is particularly prevalent during the springtime (Chapter 9), meaning that a large number of bears were likely exposed to sampling during the 3-year mark-recapture study period in the KB subpopulation.

These seasonal movement patterns contribute to the finding that the mark-recapture point estimate was ~85% greater than the aerial survey point estimate. We note that data used for the mark-recapture analysis were sparse and precluded implementing the Barker model (1997, 1999) to explicitly model temporary emigration. However, such an approach would enable a more direct comparison between methods by defining the mark-recapture estimate as pertaining to only those bears that were present in the study area and available for capture, rather than the entire super-population.

Although the aerial survey was not ideally designed (it relied up the use of a single-engine helicopter with limited range so that offshore habitat could not be surveyed), the KB subpopulation study provides the first opportunity to directly compare simultaneous mark-recapture and aerial survey studies. In Western Hudson Bay, estimates of abundance derived from mark-recapture and an aerial survey were similar (although the aerial survey snapshot estimate was somewhat greater than the mark-recapture super-population estimate), but differences in sampling frames limited inference (Stapleton et al. 2014, Lunn et al. 2016). Similarly, abundance estimates from an aerial survey and mark-recapture in Southern Hudson Bay were consistent, but several years elapsed between the inventories, and the mark-recapture estimate was adjusted upwards to reflect potential heterogeneity in capture probabilities and to account for un-sampled areas (Obbard 2008, Obbard et al. 2007, 2015). As jurisdictions incorporate alternative (non-capture based) methods for estimating abundance and monitoring populations, understanding the ability to compare results from different techniques will be critical to correctly interpreting status and trend (Stapleton et al. 2014). The differences in survey methods resulting in estimates of different “populations” (i.e., the MR estimate of the “super”-population versus the aerial survey´s real-time snapshot of abundance) suggest caution when comparing results and assessing trends from different techniques implemented during the springtime, when polar bear movements among subpopulations are greatest.

We acknowledge that our estimate of abundance derived from the aerial survey is likely biased low. First, we did not sample the southeastern portion of the KB subpopulation and the large area of the North Water polynya because of logistical and safety considerations in a helicopter and the presumed relatively low densities of bears in these regions (Heide-Jørgensen et al. 2013). Satellite telemetry data indicated no collared bears (*n* = 20 adult females) were present in the un-surveyed areas during the aerial survey sampling period (see also Chapter 9). However, the un-sampled regions covered extensive areas (sea ice near Qaanaaq: 3,245 km^2, and the North Water polynya: 27,214 km^2), such that even very low densities may significantly contribute to an overall estimate of abundance. This unsurveyed area in the North Water Polynya amounts to *ca.* 34% - 40% of the extension of the polynya (70,000-80,000 km^2; Born et al. 2004 and references therein). Extrapolating our model-averaged estimate of density from the low-density stratum (3.39 bears / 1,000 km^2) to the sea ice near Qaanaaq in southeastern KB yielded ~11 bears. For the North Water polynya, extrapolating a very low estimate of density (1.13 bears / 1,000 km^2, or roughly a third of the estimated density used for the sea ice near Qaanaaq) added 31 bears.

During May 2009 and 2010, Heide-Jørgensen et al. (2013) conducted an aerial survey over the North Water Polynya (NOW) between 76° N and 79° N (i.e., north to the southernmost part of the Nares Strait-Kane Basin region). Hence, they in effect covered a major part of the NOW with loose drift ice and open water which were not covered by us for safety reasons and because it was judged by us to be suboptimal or unsuitable polar bear habitat. Despite that their survey was a multi-species survey mainly targeting beluga (*Delpinapterus leucas*), narwhal

(*Monodon monoceros*), walrus (*Odobenus rosmarus*), and seals, they detected polar bears both in water and on ice. Heide-Jørgensen et al. (2013) derived an estimate of 60 polar bears (CV 0.96, range: 12-293 bears). Although their point estimate had a large uncertainty due to low sample size, it indicates that a substantial number of polar bears may occur "offshore" on loose drift ice in NOW (i.e., in habitat not covered during our 2014 survey). Given the inherent uncertainty in estimates of density for the unsampled regions, we hypothesize that negative bias arising from incomplete sampling of the Kane Basin polar bear subpopulation may have been roughly 10 – 30%.

Second, a fundamental assumption of distance sampling is perfect detection of target objects on the transect line (i.e., at distance 0; Buckland et al. 2001). Preliminary analyses with a left-truncated data set suggested that the probability of bears near the aircraft being sighted by at least one observer was >96%, so we considered this assumption to be approximately valid. Our data were too sparse to permit mark-recapture distance sampling analyses (Laake and Borchers 2004) to correct for less than perfect detection at distance 0, but our initial double-observer analyses suggest that any resultant negative bias was modest (<5%).

Population-wide aerial surveys of polar bears have been completed in the autumn over land in Foxe Basin, Western Hudson Bay, and Southern Hudson Bay (Stapleton et al. 2014, 2016, Obbard et al. 2015) and over both land and sea ice in the Barents Sea (Aars et al. 2009). Similarly, recent pilot aerial survey studies over springtime sea ice have been completed in the Baffin Bay and Southern Beaufort Sea subpopulations (Stapleton 2013). However, the aerial survey in the KB subpopulation represents the first attempt to complete a subpopulation-wide survey on springtime sea ice. Although most aerial surveys of polar bears have been conducted during the fall ice-free period, this study illustrates that, in small areas and under favorable weather conditions, aerial surveys can provide a useful inventory technique on springtime sea ice as well. Aerial surveys may be a particularly valuable tool for monitoring small, remote subpopulations that are not subject to significant harvest pressure and where acquiring detailed demographic information through mark-recapture may be too costly to justify.

We note, however, that larger geographic areas and periods of inclement weather may require more time to complete a comprehensive aerial survey, thereby necessitating more complex study designs to accommodate potential changes in bear densities and the study area itself, especially if sea-ice dynamics are changing during the survey window. Aerial surveys of larger areas like Baffin Bay will require the use of more than one fixed-winged aircraft with long endurance to ensure that the entire area (including remote offshore habitat) is covered within a relatively narrow time frame (e.g., SWG 2011, Nielson et al. 2013). Nevertheless, based on the pilot aerial survey along SE Baffin Island in spring 2010, a group of survey experts concluded that it would be feasible to assess polar bear populations with a larger range (i.e., Baffin Bay) using aerial surveys (Chapter 1).

We recognize that large offshore areas with loose drift ice in the Kane Basin subpopulation´s range could not be surveyed by us for safety reasons and because our helicopter had a relatively low range and endurance. Hence, future aerial surveys should consider using fixed-winged aircraft perhaps in combination with a helicopter (SWG 2011).

Our aerial survey estimate of abundance was based on a very small number of encounters (n = 28), resulting in some uncertainty in estimation of the detection function. Increasing the number of observations via greater sampling effort will likely improve precision; a minimum of 60 – 80 observations are recommended with distance sampling (Buckland et al. 2001), but even a marginal increase in sightings would improve estimation of the detection function. In addition, if other sites adopt on-ice surveys using similar study designs and survey platforms, joint analysis in which observations are pooled might yield more reliable estimates of the detection function, thereby improving precision of abundance estimates.

11.5. Literature Cited

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Barker, R. J. 1997. Joint modeling of live-recapture, tag-resight, and tag-recovery data. Biometrics 53:666–677.

Barker, R. J. 1999. Joint analysis of mark-recapture, resighting and ring-recovery data with age-dependence and marking effect. Bird Study 46:S82–S91.

Born, E. W., J. Teilmann, M. Acquarone, and F. Riget. 2004. Habitat use of ringed seal (*Phoca hispida*) in the North Water area (North Baffin Bay). Arctic 57:129–142.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Borchers, D. L., J. L. Laake, C. Southwell, and C. G. M. Paxton. 2006. Accommodating unmodeled heterogeneity in double-observer distance sampling surveys. Biometrics 62:372–378.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to Distance sampling: Estimating abundance of biological populations. Oxford University Press, Oxford, UK.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Heidi-Jørgensen, M. P., L. M. Burt, R. G. Hansen, N. H. Nielsen, M. Rasmussen, S. Fossette, and H. Stern. 2013. The significance of the North Water Polynya to Arctic top predators. Ambio 42:596–610.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Innes, S., M. P. Heide-Jørgensen, J. L. Laake, K. L. Laidre, H. J. Cleator, P. Richard, and R. E. A. Stewart. 2002. Surveys of belugas and narwhals in the Canadian High Arctic in 1996. NAMMCO Scientific Publication 4:169–190.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock's robust design. Ecology 78:563–578.

Laake, J. L., and D. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in* S. Buckland, D. Anderson, K. Burnham, J. Laake, D. Borchers, and L. Thomas, editors. Advanced Distance Sampling. Oxford University Press, Oxford, UK.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Marques, F. F. C., and S. Buckland. 2003. Incorporating covariates into standard line transect analyses. Biometrics 59:924–935.

Marques, T. A., M. Andersen, S. Christensen-Dalsgaard, S. Belikov, A. Boltunov, Ø. Wiig, S. T. Buckland, and J. Aars. 2006. The use of global positioning systems to record distances in a helicopter line-transect survey. Wildlife Society Bulletin 34:759–763.

Nielson, R. M., T. J. Evans, and M. B. Stahl. 2013. Investigating the potential use of aerial line transect surveys for estimating polar bear abundance in sea ice habitats: a case study for the Chukchi Sea. Marine Mammal Science 29:389–406.

Obbard, M. E. 2008. Southern Hudson Bay polar bear project 2003 – 2005: final report. Unpublished report, Wildlife Research and Development Section, Ontario Ministry of Natural Resources, Peterborough, Ontario, Canada.

Obbard, M. E., T. L. McDonald, E. J. Howe, E. V. Regehr, and E. S. Richardson. 2007. Polar bear population status in Southern Hudson Bay, Canada. Administrative report, U.S. Geological Survey, Reston, Virginia, USA.

Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., J. Laake, K.L. Laidre, E.W. Born, and S.N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Sahanatien, V., and A. E. Derocher. 2012. Monitoring sea ice habitat fragmentation for polar bear conservation. Animal Conservation 15:397–406.

Stapleton, S. 2013. Alternative methods for monitoring polar bears in the North American Arctic. PhD Dissertation, University of Minnesota, St. Paul, Minnesota, USA.

Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal

Science 32:181–201.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. B. Bishop, T. Marquest, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology 47:5–14.

Table 11.1. Results from distance sampling analyses of an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014. The most highly supported models ($\Delta AIC_c < 2$) are shown. In the column Model, the key function is followed by the covariate (Light = light conditions). p = detection probability. High- and low- density refer to stratum-specific estimates.

			Density (Bears / 1,000 km^2)			
Model	**ΔAIC_c**	**p**	**High-density**	**Low-density**	**Global**	**Abundance (SE)**
Half-Normal / Light	0.00	0.60 (0.09)	7.5 (2.0)	3.1 (3.1)	6.1 (1.7)	170 (49)
Hazard / None	0.13	0.43 (0.20)	11.1 (6.0)	3.7 (4.0)	8.7 (4.5)	243 (125)
Half-Normal / None	0.37	0.62 (0.09)	7.6 (1.9)	2.5 (2.5)	5.9 (1.6)	166 (44)

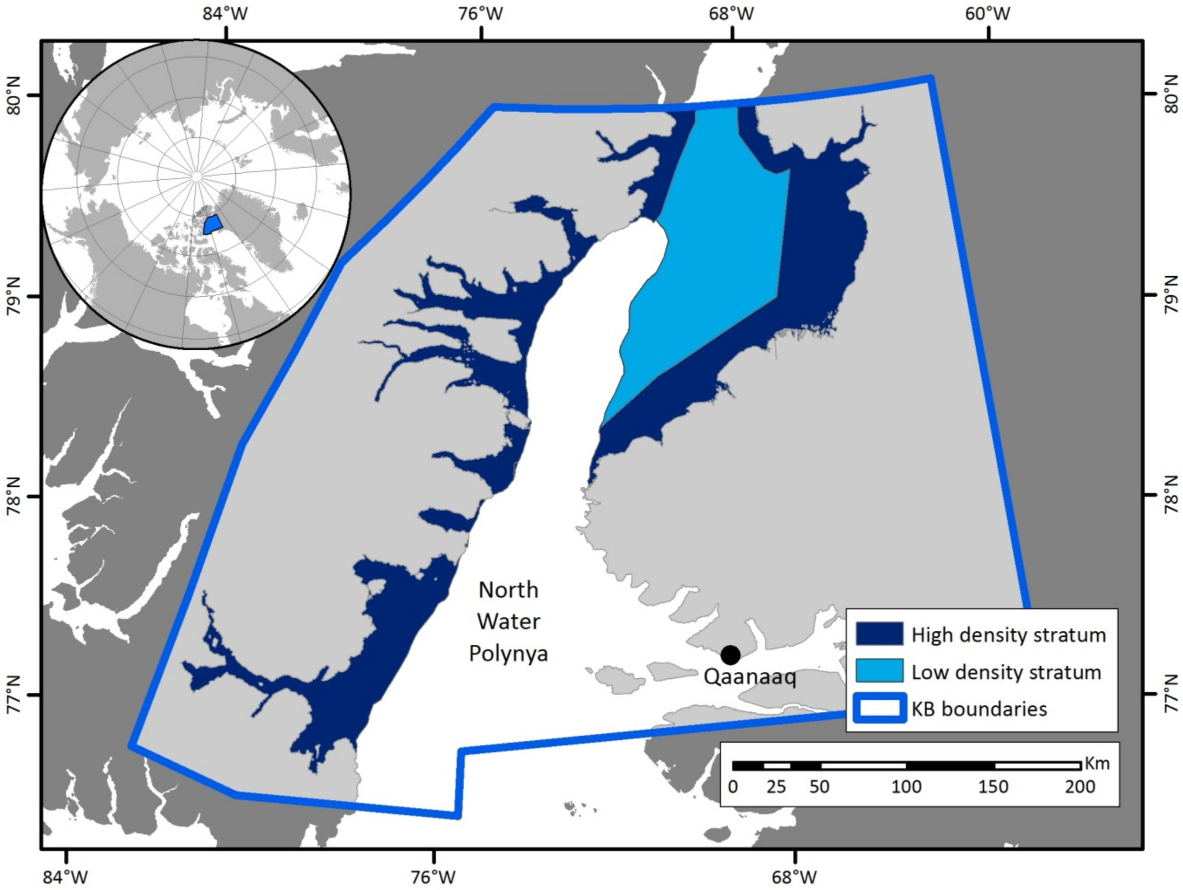


Figure 11.1. Sampling strata for genetic mark-recapture and aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

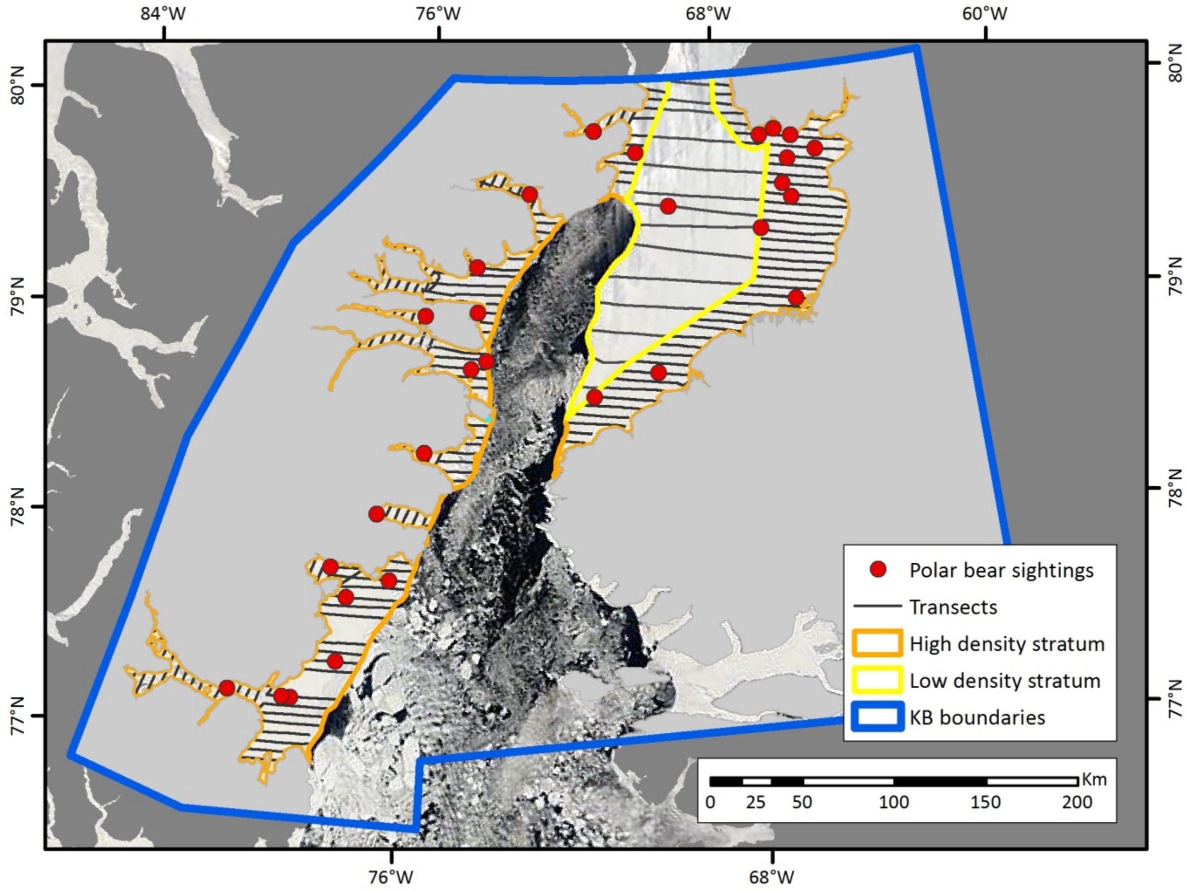


Figure 11.2. Transects surveyed and polar bear groups sighted during transect surveys of the Kane Basin subpopulation during April – May, 2014. Transects and sightings are overlaid on MODIS image (1 km resolution; available: http://modis.gsfc.nasa.gov/) collected on 5 May 2014. Sea ice in southeastern Kane Basin (i.e., to left of figure legend) was not sampled due to safety and logistical constraints presented by the North Water polynya and because we anticipated very low densities.

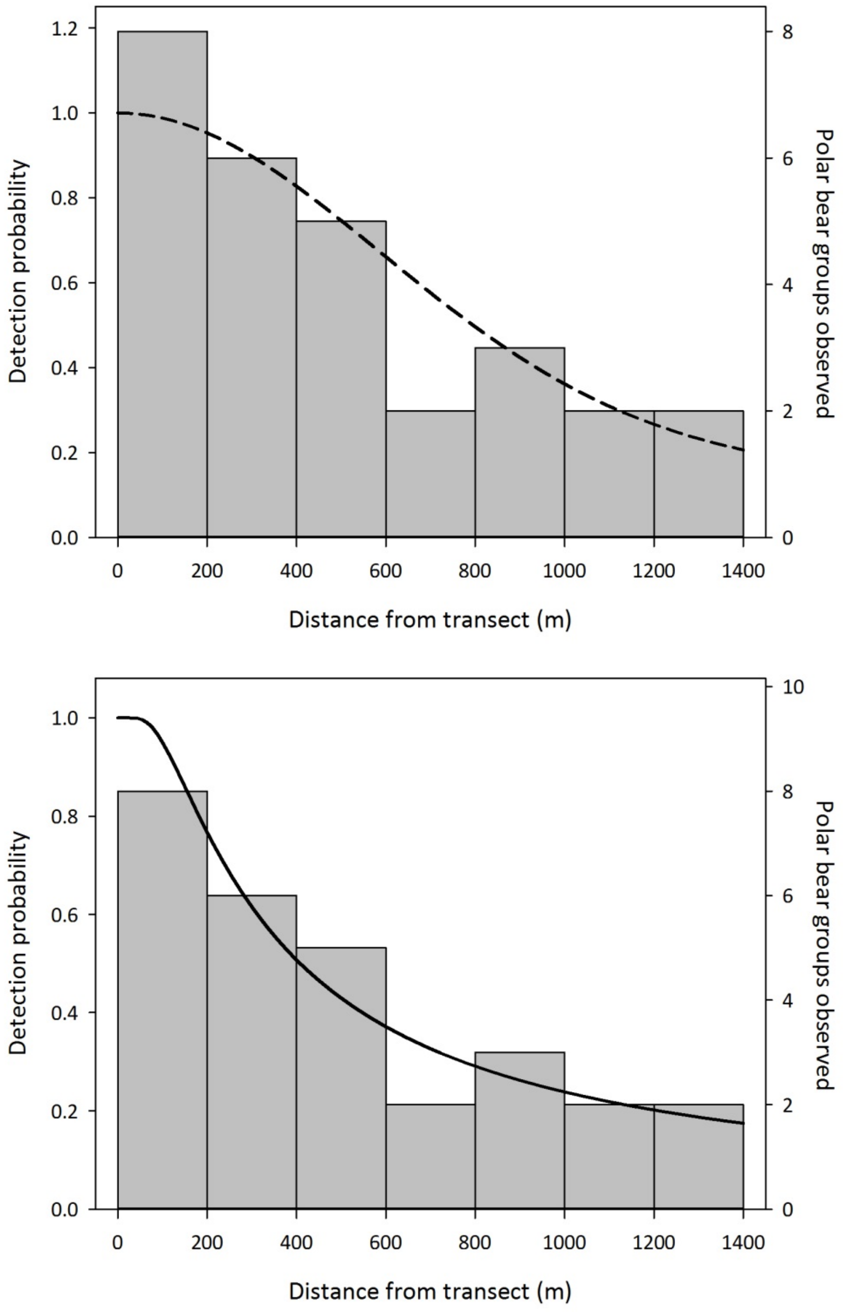


Figure 11.3. Histograms summarizing sighting distances and estimated detection functions from an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014. Top: Half-normal key function including a binary light conditions covariate. Bottom: Hazard rate key function with no adjustment terms or covariates.

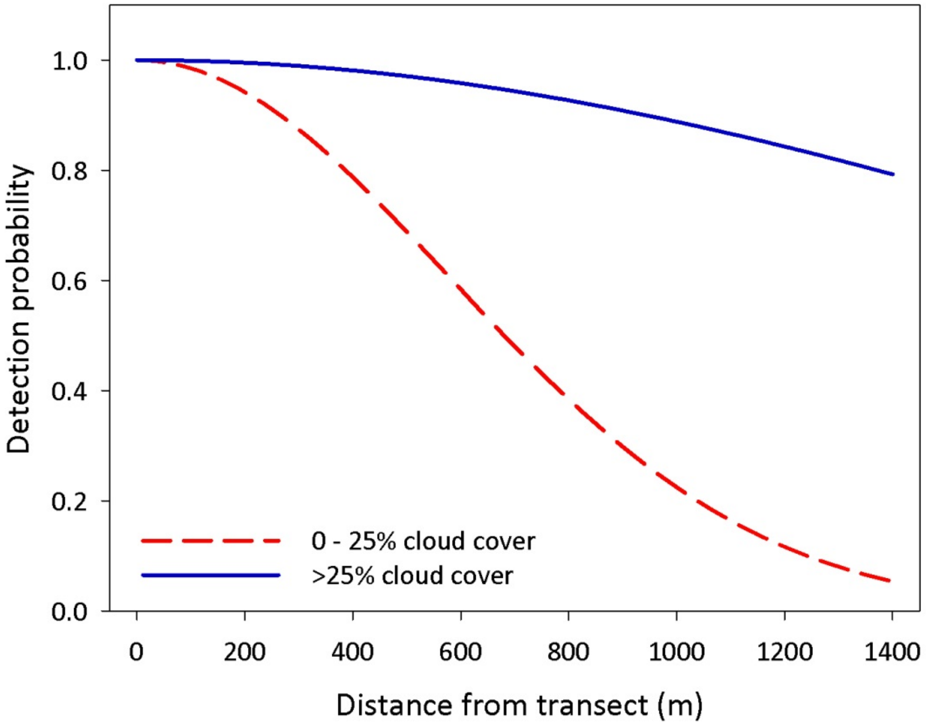


Figure 11.4. Distance sampling detection function (half-normal key function with binary light conditions covariate) estimated from data collected during an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

CHAPTER 12

REPRODUCTIVE METRICS FOR MARK-RECAPTURE SAMPLED POLAR BEARS IN KANE BASIN

KEY FINDINGS

- Data for the study were collected during two periods of MR sampling in KB. Sampling occurred on the sea ice in April and May. During the 1990s, bears were sampled by physical capture and examination using methods previously described. During the 2000s, sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals.

- Reproductive metrics for KB, including mean litters sizes for cubs-of-the-year (COY) and yearlings, and an index of recruitment (calculated as the number of yearlings per adult female in the MR sample), were comparable between the 1990s and 2010s sampling periods. Mean litter sizes in KB (for COY: 1.67 in the 1990s and 1.60 in the 2010s) were similar to those observed in other polar bear subpopulations in the archipelago ecoregion (range 1.65 - 1.71). We found no evidence of lower reproductive performance in KB, but sparse data limited our conclusions.

- During the years with the largest sample sizes (1995, 2013, and 2014), the total sample exceeded 50 bears. In these years, there was notable variation in the proportions of COY (15% - 30%). In contrast, proportions of yearlings and the recruitment index were relatively invariant among years.

12.1. Introduction

For populations of large, long-lived mammals, changes in reproductive performance can be one of the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1981, 1987). In populations approaching carrying capacity, declines in reproductive performance are likely to occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring indices or metrics of reproduction therefore may provide a useful tool for the early detection of potential population trends that may warrant more in-depth study. This is especially true for populations in which cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring reproductive metrics may provide a form of surveillance that can be used to trigger periods of more intensive study.

Reproductive metrics have been identified as an important component for monitoring polar bears across their circumpolar range (Vongraven et al. 2012). These metrics may be used to track long-term trends in the status of subpopulations, parameterize population viability models and support harvest risk assessments (Regehr et al. 2015). Of particular concern, changes in reproduction are predicted to be amongst the first subpopulation-level effects of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Molnár et al. 2011, Stirling and Derocher 2012). Indeed, declines in reproduction have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (Regehr et al. 2007, Rode et al. 2010, Rode et al. 2014).

Changes in reproductive metrics can signal significant changes in subpopulation status of polar bears. However, observations of poor reproductive performance alone do not necessarily imply a decline in subpopulation status. Studies of several polar bear subpopulations have documented declines in reproduction in association with increases in abundance that may be the result of density dependence (Derocher 2005, Peacock et al. 2013). In other cases, variation in reproductive performance within or amongst subpopulations has been attributed to geographic or annual variation in biological productivity and prey availability (i.e., fluctuating carrying capacity; Stirling and Øritsland 1995, Stirling and Lunn 1997, Stirling 2002, Rode et al. 2014). Information on reproduction therefore must be considered alongside other measures of subpopulation performance in-order to properly assess status.

The Kane Basin (KB) polar bear subpopulation is part of the archipelago ecoregion as defined by Amstrup et al. (2008), where sea ice does not melt entirely in the summer and some bears remain on the ice year-round. However, the sea ice situation in Kane Basin has changed markedly in recent decades (Chapter 9). Although currently designated as declining based on population viability modelling (PBSG 2010), a comparison of results from two mark recapture studies suggests the abundance of this small, low density subpopulation has not changed significantly over the two decades (Taylor et al. 2008a, Chapter 10). KB has experienced long term changes in sea-ice composition, and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapters 4 and 9). However, model projections predict that KB will be one of the last polar bear subpopulations to experience the negative consequences of climate change including reproductive failure (Amstrup et al. 2008, Hamilton et al. 2014). Here, we summarize reproductive metrics for KB using data collected during two periods of mark-recapture sampling from 1992 to 1997 (Taylor et al. 2008a) and 2012 to 2014 (Chapter 10). Our results provide additional context for interpreting the results of mark-recapture analyses and assessment of this subpopulation's present status.

12.2. Materials and Methods

Study Area

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008a). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit; *cf.* Figure 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq areas). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, the amount of sea ice during summer in Nares Strait-Kane Basin has dropped markedly in recent decades (e.g., Figure 9.2).

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in KB. Sampling occurred on the sea ice in April and May. During the 1990s, bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2008a). Data on the sex, age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. Sampling occurred during 25 April-6 May in 2012 and during 27 April and 10 May in 2013. With biopsy darting, we remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 –

4], and adult) from the air at a range of 3-7 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males; e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of known age-class individuals (Appendix B).

Reproductive Metrics

We calculated annual reproductive metrics that have been previously recommended (Vongraven et al. 2012) or used in studies of polar bears (e.g., Derocher and Stirling 1995, Rode et al. 2010, Peacock et al. 2013, Stapleton et al. 2014, Regehr et al. 2015). For cubs-of-the-year (COY) and yearlings, mean litter sizes were calculated from observed litter sizes. Because we did not have estimated ages for adult females sampled in 2012-2014 and because samples sizes were small in most years, we calculated a pooled mean for each year rather than age-specific values. Numbers of COY and yearling were expressed as a proportion of the total bears sampled each year. An index of recruitment was calculated as the total number yearlings divided by the total number of adult females in the sample (Derocher and Stirling 1995, Regehr et al. 2015).

Some individuals were sampled more than once in a given year. These recaptures were excluded from analyses. Captures of the same individual over multiple years were included.

We examined annual variation in reproductive metrics and compared metrics between the two epochs (1992-1997 and 2012-2014). Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

12.3. Results

During 1992-1997 and 2012-2014, we sampled 53 family groups consisting of an adult female and 1-2 dependent COY (87 COY in total). We also sampled 24 family groups consisting of an adult female and 1-3 dependent yearlings (32 yearlings in total). The mean number of family groups sampled annually was 5.9 (range: 2-15) and 2.7 (range: 0-5) for COY and yearling families respectively. Annual reproductive metrics are presented in Table 12.1.

Annual variation in observed litters sizes was not significant amongst COY (Kruskal-Wallis, $H = 4.86$, $P = 0.772$) or yearlings (Kruskal-Wallis, $H = 9.49$, $P = 0.219$). COY comprised between 9 and 38% of the bears sampled annually. Yearlings comprised between 0 and 16% of annual observations. Recruitment ranged from zero to 0.43. Sample sizes were too small to permit further analyses of annual reproductive metrics. Pooling data within epochs there were no differences in mean litter sizes between the 1990s and 2000s (Mann-Whitney U test, $U = 369$, $P = 0.700$ for COY; $U = 79.5$, $P = 0.671$ for yearlings). Proportions of COY and yearlings were also similar between epochs (Table 12.2).

During sampling in 2012-2014, 9 (12%) of 78 adult females encountered were of known age (marked during the 1990s) and ranged in age from 18 to 35 years. Five were between 18 and 20 years old, three of which were observed with litters. None of the 4 (5%) bears > 20 years old were observed with offspring. In comparison, during sampling in the 1990s, 77 adult females were sampled. Of these, 10 (13%) were 18 years old or greater (based on tooth aging) ranging in age from 18 to 21 years, and seven were accompanied by offspring. Two (2.6%) were greater than 20 years of age.

12.4. Discussion

Sample sizes were too small in most years to permit investigation of annual variation in reproductive metrics in KB. Observed variation likely was primarily the product of sample size rather than biological effect. During the years with the largest sample sizes (1995, 2013, and 2014), the total sample exceeded 50 bears, representing a sizeable portion of this small subpopulation. In these years, there was notable variation in the proportions of COY (15% - 30%; Table 12.1). In contrast, proportions of yearlings and the recruitment index were relatively invariant among years. We are unaware of any source of sampling bias that would account for this apparent variation in COY production and suggest that it may reflect pulsing or synchrony in reproduction. Anecdotally, dates of spring and fall sea-ice transition in the years prior to these 3 years were unremarkable in terms of variation; offering no explanation in terms of environmental conditions.

Our surveys were conducted in late April-early May. Den emergence date even at high latitutes (i.e., > 76° N) late March (Ferguson et al. 2000; Chapter 9, Figure 9.14). Hence, we are confident that our observations of adult females with COYs were representative.

During recent sampling, 2012-2014, we recaptured a small number of older, known age adult females that were originally marked in the 1990s. Amongst this sample, none of the individuals >20 years of age were accompanied by offspring. Although the sample size is small, this is consistent with the reproductive senescence reported in some other subpopulations Ramsay and Stirling 1988, Derocher and Stirling 1994).

Reproductive metrics for KB in both time periods were comparable, and mean litter sizes in KB were within the range of observed variation for other polar bear subpopulations in the archipelago ecoregion (Table 12.2). In summary, we found no evidence of lower reproductive performance in KB, but sparse data limited our conclusions.

12.5. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180. American Geophysical Union, Washington, DC, USA.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. Population Ecology 47:267–275.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1994. Age-specific reproductive performance of female polar bears (*Ursus maritimus*). Journal of Zoology 234:527–536.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Ferguson, S. H., M. K. Taylor, A. Rosing-Asvid, E. W. Born, and F. Messier. 2000. Relationships between denning of polar bears and conditions of sea ice. Journal of Mammalogy 81:1118–1127.

Fowler, C. W. 1981. Density dependence as related to life history strategy. Ecology 62:602–610.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS ONE 9(11): e113746. doi:10.1371/journal.pone.0113746.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., and N. J. Lunn. 1997. Environmental fluctuations in arctic marine ecosystems as reflected by variability in reproduction of polar bears and ringed seals. Pages 167–181 *in* S. J. Woodin, and M. Marquiss, editors. Ecology of arctic environments. Special Publication No. 13 of the British Ecological Society, Blackwell Science Ltd., Oxford, UK.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667-1673.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2008b. Mark-recapture and stochastic population models for polar bears of the high arctic. Arctic 61:143–152.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. Marine Mammal Science 25:778-796.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Table 12.1. Reproductive metrics derived from annual mark-recapture sampling data from Kane Basin. Captures and between season recaptures are included.

Year	Mean Litter Size (n, SD)		Proportion of Total Observations (n)		Recruitment Index [1] (Yearlings/adult female)
	COY	Yearling	COY	Yearlings	
1992	2.00 (3, 0.00)	0.00 (0, 0.00)	0.38 (16)	0.00 (16)	0.00
1993	2.00 (2, 0.00)	1.00 (3, 0.00)	0.17 (23)	0.13 (23)	0.43
1994	1.60 (5, 0.55)	2.00 (2, 0.00)	0.26 (31)	0.13 (31)	0.40
1995	1.70 (10, 0.48)	1.50 (2, 0.71)	0.25 (67)	0.04 (67)	0.14
1996	1.40 (5, 0.55)	1.00 (3, 0.00)	0.19 (36)	0.08 (36)	0.23
1997	1.50 (2, 0.71)	2.00 (2, 1.41)	0.09 (32)	0.13 (32)	0.40
2012	1.50 (4, 0.58)	1.40 (5, 0.55)	0.14 (44)	0.16 (44)	0.37
2013	1.57 (7, 0.53)	1.33 (3, 0.58)	0.15 (71)	0.06 (71)	0.14
2014	1.65 (14, 0.50)	1.00 (4, 0.00)	0.30 (84)	0.05 (84)	0.13

[1] – *Sensu* Regehr et al. (2015)

Table 12.2. Comparison of reproductive metrics for some polar bear subpopulations in the Arctic archipelago ecoregion (Amstrup et al. 2008). Sampling occurred during ice-free periods.

Subpopulation	Mean Litter Size		Proportion of Total Observations		Source
	COY	Yearling	COY	Yearlings	
Kane Basin (1992-97)	1.67	1.42	0.22	0.08	Taylor et al. (2008a)
Kane Basin (2012-14)	1.60	1.25	0.21	0.08	This study
GB	1.65	-	-	-	Taylor et al. (2009)
LS	1.69	-	-	-	Taylor et al. (2008b)
MC	1.68	-	-	-	Taylor et al. (2006)
NW	1.71	-	-	-	Taylor et al. (2008b)

CHAPTER 13

OBSERVATIONS ON THE BODY CONDITION AND FORAGING HABITS OF POLAR BEARS IN KANE BASIN DURING THE SPRING

KEY FINDINGS

- Body condition was better amongst KB subadults and adult females with yearlings during the 2010s relative to the 1990s, though sample sizes were limited. In contrast, condition amongst adult males, adult females with COYs, and lone adult females was similar between these time periods.
- Improved condition in the 2010s may reflect natural variation or a response to long-term changes in the sea-ice regimen in Kane Basin, largely turning into a system resembling a seasonal sea-ice ecoregion.
- Seals and polar bears were similarly distributed in Kane Basin.
- Relatively high densities of both seals and bears in northeastern Kane Basin, near the Humbolt Glacier, indicates that this region has high productivity and is important habitat for polar bears in the subpopulation.
- A high proportion of KB bears were found to have extensive hair loss and skin ulcerations on their feet. The cause of these lesions is unknown to science. Traditional knowledge suggests this phenomenon is the result of abrasive injuries sustained by walking and digging in hard, icy, coarse snow cover on the spring sea ice combined with increased rates of movement during the peak mating and feeding periods.

13.1. Introduction

For populations of large, long-lived mammals changes in body condition will be among the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1987, 1990, Zedrosser et al. 2006). In populations approaching K, declines in condition will occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring body condition may therefore provide a useful tool for the early detection of population trends that warrant more in-depth study. This is especially true for populations where cost or logistical constraints limit the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring condition may provide a form of surveillance that can be used to trigger periods of more intensive study.

The annual life-cycle of polar bears is characterized by large seasonal changes in body condition (Watts & Hansen, 1987, Ramsay et al., 1992, Ramsay and Stirling 1988, Atkinson and Ramsay 1995). Throughout most of their circumpolar range, bears are thought to gain condition during the spring and early summer when juvenile seals are abundant and relatively susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). This period of hyperphagia is followed by a scarcity of food in the late summer and fall when sea ice reaches a minimum throughout the Arctic. During this season, bears in some regions are forced onto land by the melting sea ice where access to seals and other marine mammal prey is greatly reduced (Stirling et al. 1977, Derocher and Stirling 1990, Ramsay et al. 1991). In other regions, bears remain on off-shore pack-ice but likely also have reduced access to and/or less success in catching seals (Amstrup et al. 2000, Stirling 2002, Atwood et al. 2015a, Rode et al. 2015).

Given this dynamic cycle of feeding and fasting, body condition attained during the spring and early summer is expected to exert a significant influence on the survival, reproductive performance and thus status of polar bear subpopulations (Atkinson and Ramsay 1995, Derocher and Stirling 1995, 1996, Molnár et al. 2010, Molnár et al. 2011). Tracking long-term trends in body condition has thus been identified as an important component of the monitoring scheme for polar bears across their circumpolar range (Vongraven et al. 2012, Patyk et al. 2015). In the absence of more intensive studies, simple body condition metrics may be useful indices for monitoring subpopulations and detecting responses to changing environmental conditions

(Amstrup et al. 2006, Stirling et al. 2008a, Vongraven et al. 2012). Of particular concern, changes in body condition are predicted to be amongst the first subpopulation-level impacts of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Wiig et al 2008, Stirling and Derocher 2012). Indeed, declines in condition have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016).

A variety of quantitative and qualitative body condition indices have been used on polar bears including body weight estimated from girth (e.g., Derocher and Stirling 1995, Rode et al. 2011), body mass indices standardized for length (e.g., Stirling et al 1999, Cattet et al. 2002, Obbard et al. 2016), skull width (Rode et al. 2010, 2011), percent body fat determined by isotopic dilution or bioelectrical impedance analysis (Atkinson and Ramsay 1995, 1996, McKinney et al. 2014), percent lipid content of adipose tissue biopsies (Stirling et al. 2008b, McKinney et al. 2014) and a visually assigned fatness index (Amstrup et al. 2006, Stirling et al. 2008a, b). Most of these condition indices require the handling of bears to collect measurements. However, the fatness index (FI) and potentially the lipid content of adipose tissue (Pagano et al. 2014, McKinney et al. 2014) may be obtained without handling thus making them suitable for use in subpopulations monitored by less invasive methods such as aerial survey or genetic mark-recapture.

The Kane Basin (KB) polar bear subpopulation is part of the archipelago ecoregion as defined by Amstrup et al. (2008), where sea ice does not melt entirely in the summer and some bears remain on the ice year-round. Although currently designated as declining based on population viability modelling (PBSG 2010), a comparison of results from two mark recapture studies suggests the abundance of this small, low density subpopulation has not changed significantly over the two decades (Taylor et al. 2008a, Chapter 10). KB has experienced long term changes in sea-ice composition, and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapter 4). However, model projections predict that KB will be one of the last polar bear subpopulations to experience the negative consequences of climate change including reproductive failure (Amstrup et al. 2008, Hamilton et al. 2014).

Here we summarize information on the body condition of polar bears in KB collected during two periods of mark-recapture sampling from 1992 to 1997 and 2012 to 2014. Using the Fatness Index (FI: Stirling et al. 2008b) as a qualitative metric we examine differences in condition between the two time periods. Our results provide supplementary information for interpreting the results of the genetic mark-recapture (Chapter 10) and other recent studies in KB (Chapters 2 and 9), and for understanding the present status of this subpopulation. We also report on incidental observations of prominent skin lesions that were found on some KB bears during the latter period of the study. Finally, we report incidental observations of spring time foraging by bears and the distribution of seals in KB.

13.2. Materials and Methods

Study Area

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* Figure 13.1). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, sea-ice conditions have changed markedly in the Kane Basin region in recent decades (Born et al. 2011; Figure 9.2).

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in KB. In both periods sampling occurred on the sea ice in April and May using a helicopter flying at 300-500 feet above sea-level to search for bears across the study area. During the 1990s (1992-97), bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2008a). Data on the sex, age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2010s (2012-14), sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. With biopsy darting, we remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 –

4], and adult) from the air at a range of 3-7 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males; e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of known age-class individuals (Appendix B).

Body Condition Scoring

Because most of the bears sampled during the latter period of sampling (2012-2014) were biopsy darted rather than captured and handled, our ability to compare body condition between time periods was limited to visually assigned Fatness Index (FI) scores only. The FI has been validated as a measure of condition in polar bears, being closely correlated with more quantitative condition indices (Stirling et al. 2008b, McKinney et al. 2014) and other biological factors (e.g., Henricksen et al. 2001, Amstrup et al. 2006). During both sampling periods, all encountered bears were assigned a FI score on a scale of 1-5 where 1 and 5 represent the leanest and most obese bears, respectively (Stirling et al. 2008b). During the 1990s, this score was based on physical examination of captured bears. For bears in the 2010s, FI scores for most

(67%) individuals were assigned based on examination from the air at a distance of 3-7 m above ground. The remaining portion was assigned FI scores based on physical examination after capture.

All bears were initially scored in the field according to the standard FI on a scale of 1 to 5 (Stirling et al. 2008b). This scoring system was subsequently simplified to a binary Body Condition Score (BCS) where individuals in poor (FI = 1, 2) and fair-good (FI = 3, 4 or 5) condition were assigned scores of 1 and 2 respectively. Similar modifications of the FI for polar bears have been employed in other studies to facilitate analyses (Stirling et al. 2008a) or have been recommended for use in general monitoring schemes for polar bears (Vongraven et al. 2012). In our case, this refinement was necessary due to the small samples sizes overall in our study and the low frequencies of bears at the extremes of the 5 point FI scale (i.e., very few or no bears with FI scores of 1 or 5). This simplified scoring system was also a potential means to reduce bias in assigning condition scores. The assumption made was that a simplified scale would be subject to less bias resulting from different observers and / or distance from bear at time of scoring. Experienced observers should be able to discriminate a bear in poor condition even at distances of up to 7 m. All observers in our study had extensive experience studying polar bears including capture, handling and body condition scoring.

For analyses, we pooled BCS data collected in different years into two periods (epochs); the 1990s (1992-97) and the 2010 (2012-2014). Again this was necessary due to low samples sizes. Repeated observations of the same individual (as identified by physical mark or genotype) within a given year were excluded from the analyses. Observations of the same individual in different years were included. Similar to Stirling et al. (2008a), we assumed that observations of the same individual in different years were statistically independent given the dynamic nature of body condition in polar bears (Watts and Hansen 1987, Atkinson and Ramsay 1995, 1996) and it's response to annual variation in environmental conditions.

For different sex, age and reproductive classes of polar bears we compared BCS between the two epochs using contingency tables analyses (Cross Tabs procedure in SPSS Version 24.0, IBM Corp. 2016). We also considered the potential effect of the timing of sampling on the BCS of bears. Along with the binary categorical variable (Epoch: 1990s and 2010s), Julian Day of sampling was used as an independent variable in a logistic regression (Binary Logistic procedure in SPSS Version 24.0, IBM Corp. 2016) to examined variation in BCS. Both variables were entered into the regression model. We did not specify an interaction between Julian Day and Epoch. All tests were two-tailed with alpha at 0.05.

Other Observations

During sampling from 2012-2014, we also made several other types of observations either systematically or opportunistically. For each bear encountered we noted any evidence of recent feeding. The presence of a seal kill or bears with full pendulous stomachs constituted evidence of feeding. While searching for bears in 2013, we noted the locations of live seals using a GPS. Each group comprising 1 or more individuals was recorded as a single observation. Finally, during capture and physical examination of bears in 2011 and 2012 we noted the presence or absence of some prominent skin lesions that had not previously been described in the literature.

13.3. Results

Body Condition Scores

Body Condition Scores (BCS) were assigned to 129 and 135 subadult and adult polar bears encountered during sampling in KB in the 1990s and 2010s, respectively. BCS for adult males were similar between the two epochs (Table 13.1). In contrast, across all reproductive classes, adult females in the 2010s tended to be in better condition than those in the 1990s; although this was statistically significant for adult females with yearlings only. Similarly, subadults in the 2010s were in better condition at time of encounter.

Although sampling occurred in April and early May during both epochs, timing of sampling differed (Mann-Whitney U = 1,557.00, P = 0.002). Median Julian day of sampling was slightly earlier during the 1990s (121.45) relative to the 2010s (124.68) across all sex-age classes. Within sex-age classes, these slight differences in timing of sampling were maintained (e.g., adult females with yearlings, Mann-Whitney U = 104.50, P = 0.060; sub adults, Mann-Whitney U 275.0, P = 0.02). Incorporating Julian day of sampling into a logistic regression did not explain variation in body condition amongst most classes of bears with the exception of adult females with cubs-of-the-year (COY) where bears sampled later tended to be in better condition (Appendix E). Amongst adult females with yearlings, Julian day was not a predictor of BCS and condition was better in the 2010s than in the 1990s. For subadults there was no effect of either timing of sampling or epoch on the probability of a bear being in poor versus fair-good condition.

Other Observations

Feeding – During sampling in 2012-2014, 14% of bears encountered showed evidence of recent feeding (excluding dependent offspring). Prevalence of feeding observations was highest amongst adult females with offspring and lowest in adult males and subadults (Table 13.2).

Seals – In 2013, 94 groups consisting of one or live ringed seals, *Phoca hispida*, were observed while searching for bears in KB. Notable concentrations of seals were encountered in north east Kane Basin in front of the Humbolt glacier and inside fiords along eastern Ellesmere Island (Figure 13.1).

Skin Lesions –In 2012 and 2013, 40% of the bears that were captured and physically examined were found to have unusual skin lesions. These were characterized as locally extensive alopecia (hair loss) over the feet, in most cases (75%) affecting all four feet (Figure 13.2). In addition, some of the affected individuals had multi-focal ulcerations on the plantar/palmar heel and digital foot pads and on the dorsal aspects of all 4 feet (Figure 13.3). Discharge from these lesions was purulent and sanguinous. Granulation tissue forming in some of these ulcers indicated they were chronic in nature. Even under anesthesia, some bears exhibited notable discomfort when these ulcers were gently palpated during examination, often reacting by moving the foot or lifting their head. Finally, two individuals (an adult male and a yearling) were found to have mild generalized alopecia over the dorsal neck, thorax and abdomen.

The prevalence of foot lesions was highest amongst adult males (75%) and lowest amongst cubs-of-the-years (0%) (Table 13.3). The prevalence of bleeding ulcerations on the feet, an indication of the severity and/or chronicity of the condition, was highest amongst adult males with 75% exhibiting some degree of ulceration. Also of note were two bears captured in 2012 without lesions that were recaptured in 2013 with lesions.

13.4. Discussion

Body Condition

With the limited data in this study we found evidence of differences in the spring time body condition of KB polar bears between the 1990s and 2010s. Condition amongst adult females with yearlings and subadults was better in 2010s. In contrast, condition amongst adult males, adult females with COY, and lone adult females was similar between these time periods. These findings may be attributable to several factors including bias in the data, natural variation in condition and long-term trends in environmental conditions.

Several sources of bias were possible in our study associated with use of a qualitative body condition score rather than a quantitative metric. BCS data were collected by several observers. In the 1990s, most data were collected by a single observer. In the 2010s all data were collected by a single but different observer. Differences in the assignment of condition scores by these two observers could therefore generate the apparent differences in condition between time periods. Since individual bears were not scored by more than one observer, teasing out potential observer effects is challenging. While we cannot exclude the possibility of observer bias in our study, several lines of evidence suggest that this potential bias is unlikely to account for our results. First, to reduce observer bias we employed (*post-hoc*) a simplified body condition scoring system that required observers to discriminate between bears in poor versus fair-good condition. All observers in the study were experienced polar bear biologists who had previously handled hundreds or thousands of bears in varying condition and should have been capable of accurately discriminating such bears. Second, as noted by Stirling et al. (2008a) although the FI from which our condition metric was derived is a qualitative index and thus subjective, it has been found to be "*repeatable between individual biologists when blind comparisons are done in the field over both short and long time periods*." In other studies, FI data collected by multiple observers have been found to correlate closely with quantitative indices of condition (e.g., Stirling et al. 2008b, McKinney et al. 2014). Finally, we found differences in condition between time periods amongst adult females with yearling and subadult only. If these differences in condition reflect observer bias we would expect this to be evident in all classes of bears.

Body condition scores in the 1990s were collected from bears captured and physically examined. In contrast scores in the 2000s were from bears either captured (33%) or observed from the air without handling (67%). The effect of close-up versus distance examination on the scoring of condition is unknown. McKinney et al. (2014) found that remotely assigned FI ratings did not correlate with the % lipid content of adipose tissue; another measure of condition. However, their sample sizes were small and limited to comparisons of bears of FI 3 and 4 only whereas bears in our study had FI ratings ranging from 1 to 5. Remotely scoring FI may be a less robust (precise) index of condition but is not necessarily inherently biased relative to physical examination. For many of the same reasons discussed previously concerning observer effects, we suggest that this potential source of bias is unlikely to account for our results. The use of a simplified scoring system (poor versus fair-good) in our study should have helped to reduce errors in scoring for bears observed from the air.

Another source of error in our study associated with differences in sampling between the 1990s (physical capture) and 2010s (physical capture or aerial observation) was in the classification of bears by sex and age-class based on aerial observation rather than handling. Classifying bears from the air is without doubt less accurate than physical examination. However, aerial classification is accurate in most instances (Chapter 5, Appendix B), especially amongst adult males and adult females with offspring. Misclassification was therefore unlikely to explain differences in condition of adult females with yearlings. Additionally, despite being less precise we have no evidence to suggest that aerial classification results in a bias in age and sex classifications amongst a group of bears. This source of measurement error thus seems unlikely to account for our results.

Body condition amongst bears likely improves progressively during the spring and early summer as the availability of seals increases (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). Sampling in the 1990s tended to occur earlier in the spring than in the 2010s. This difference in the timing of sampling could therefore partially account for the better condition we observed amongst some classes of bears in the 2010s. However, we note that differences in timing although significant were relatively small (i.e., 3-4 day difference in median day of sampling) so the effect on condition data may be minor. Additionally, timing of sampling was not a significant predictor of body condition amongst adult females with yearling or subadults; the two classes of bear in which differences in condition scores were detected. Finally, if timing of sampling were a significant factor we would have expected similar bias in other classes. Interestingly, we found that condition amongst adult females with COY was a function of Julian day of sampling suggesting consistent with the hypothesis that females emerging from maternal dens begin to steadily recover lost body condition in the spring.

Several ecological explanations could explain our findings. Body condition amongst polar bears fluctuates on temporal and spatial scales in response to annual variation in environmental conditions regardless of any underlying long-term trends (Kingsley 1979, Stirling 2002). Our findings may simply reflect this normal variation in condition whereby sampling in the 2010s occurred at a higher point in condition than in the 1990s. However, this would not account for the fact that improved condition was only detected amongst adult females with yearlings and subadults; since presumably all classes of bears can capitalize on improved environmental conditions. Failure to find differences in condition amongst other age classes may be due to the limited samples sizes, lack of precision in condition scoring and / or bias in our study. We note that condition tended to be better in 2010s amongst all classes of adult females; although only statistically significant for those with yearlings.

As an alternative explanation, differences in body condition between the 1990s and 2010s may reflect long-term changes in environmental conditions. At the southern extent of the polar bears' range, declining condition has been associated with reduced sea-ice cover resulting from climate change (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016). However, High Arctic polar bears such as those in KB are predicted to be amongst the last members of the species negatively impacted by climate change (Derocher et al. 2004). An initial impact of climate change in KB has been an observed reduction in the extent of multi-year ice and replacement with thinner annual ice (Hamilton et al. 2014, Chapter 4). Such changes in ice regimen are predicted to have a positive effect on polar bears via increased primary productivity and access to prey (Derocher et al. 2004). Our finding of improved body condition amongst adult females and subadults is consistent with this prediction and may be a sign of improved environmental conditions (albeit temporarily). In some subpopulations where effects of climate change have been reported, body condition has been negatively affected to a greater and / or more easily detectable degree amongst the adult female and subadult classes (Obbard et al. 2006, Rode et al. 2010). This suggests that sensitivity to deteriorating environmental conditions varies by sex, age, and reproductive status; presumably as a result of differing nutritional and energetic requirements and / or rates of food intake. Conversely, it is reasonable to assume that the age classes most readily impacted by negative changes in the environment will be the first to respond positively as conditions improved. This hypothesis is consistent with our finding that condition improved significantly amongst adult females and subadults but not adult males.

Given the limitations of our data set as discussed above, we are unable to confidently resolve between the differing explanations for our finding of improved body condition between the 1990s and 2010s. We therefore urge caution in interpreting these results. Never-the-less, it is reasonable to conclude that there has been no decline in condition in KB.

Feeding Observations

During the 2010s, the proportion of individuals showing signs of having recently fed was lowest amongst adult males. This is consistent with the hypothesis that adult males forgo foraging opportunities during the spring mating period while pursuing estrous females (Cherry et al. 2009, Stirling et al. 2015). The relatively low feeding rates we observed amongst subadults is consistent with the hypothesis that juvenile bears are less successful in hunting. We did not have data on feeding rates during the 1990s sampling period for comparison. Consequently, we are unable to assess potential changes in hunting success that could account for the improved condition observed amongst bears in the 2010s.

Seal Distribution

The distribution of seal observations made during mark-recapture sampling of polar bears was uncorrected for search effort. However, we note that the relative densities of seals along our search tracks was similar to the distribution of the polar bears we encountered (Chapters 10 and 11). Not surprisingly, where we found relatively high numbers of seals, we also found relatively high numbers of bears. The high densities of ringed seals found in north eastern KB at the front of the Humbolt Glacier is consistent with other studies that have found sea ice in front of tide-water glaciers to be prime breeding habitat for seals (Lydersen et al. 2014). This area of Kane Basin appears to have high productivity and is likely an important feeding area for polar bears. This is consistent with traditional ecological knowledge indicating that the area was a preferred hunting location for Greenlandic Inuit (Born et al. 2011). According to PBSG (1998), Taylor et al. (2001), and Born (pers. obs. 1994 and 1995) both the Greenland and Canadian portions of Kane Basin were mostly mixed annual and multi-year sea ice that appeared to be favourable habitat for polar bears in the 1990s. Ringed seals were common in both eastern and western KB (ibid.). The relatively few polar bears encountered in this region during the 1990s was thus likely the result of hunting pressure rather than habitat suitability (Taylor et al. 2001).

Skin Lesions

We documented hair loss (alopecia) and ulcerations on the feet of polar bears in Kane Basin during capture sampling in 2012 and 2013. These types of lesions have not been reported previously in the literature. Atwood et al. (2015b) documented an alopecia syndrome of unknown etiology amongst polar bears from the Southern Beaufort Sea (SB). However, those lesions were largely confined to the head, neck and shoulders rather than the feet. Additionally, they were most prevalent amongst subadult bears whereas those in KB were most common amongst adult males. Overall prevalence of lesions in KB bears (40% in 2012 and 2013) was higher than peak prevalence observed in the SB (28%). In particular, we note that seventy-five percent of adult males sampled in KB in 2012 and 2013 were affected. Lesions on adult males were also more severe than on other age classes as indicated by the presence of skin ulcerations some of which were apparently very painful. Atwood et al. (2015b) found that bears with alopecia were in poorer body condition. What impact, if any, the lesions found on KB bears may have had on condition is unknown since samples sizes were too small to support analyses.

Whether the lesions reported in SB and KB share a common etiology (cause) is unknown. Discussions with Inuit hunters from Grise Fiord (near Kane Basin) suggest this phenomenon of hair loss on the feet is well known to occur in spring time (M. Akeeagok, J. Kiguktak, D. Akeeagok pers. comm.). It is believed that increased rates of movement in spring, when bears are mating and hunting activity is high, result in abrasions to the feet. Dog teams in the Canadian Arctic are well known get similar lesions in the spring when travelling over coarse, icing snow formed by melting and refreezing as temperatures fluctuate throughout the day (D. Iqqaqrialu pers. comm.). The condition may become so severe that some dogs become lame and unable to pull sleds. Snow cover on the sea-ice in KB in 2012 and 2013 was noticeably icy and granular in composition (Figure 13.4) with a hard ice-covered crust on top. It is therefore plausible that the lesions we observed were the result of snow conditions. The finding that prevalence was highest in adult males may be due to their greater weight and the likelihood that they break the ice crust while walking, combined with potentially increased time allocated to travelling in the spring while searching for estrous females[1]. Lesions similar to those observed in 2012 and 2013 were not seen on polar bears captured in KB between 1992 and 1997 (M. K. Taylor and E. W. Born, pers. comm.). Whether this is a new phenomenon in KB brought about by changing snow conditions and progressively warmer spring temperatures or an incidental observation in the years we were sampling bears is unknown.

13.5. Literature Cited

Amstrup, S. C., G. M. Durner, I. Stirling, N. J. Lunn, and F. Messier, F. 2000. Movements and

---

[1] Versus time spent stalking prey or resting

distribution of polar bears in the Beaufort Sea. Canadian Journal of Zoology 78:948–966.

Amstrup, S. C., I. Stirling, T. S. Smith, C. Perham, and G. W. Thiemann. 2006. Recent observations of intraspecific predation and cannibalism among polar bears in the southern Beaufort Sea. Polar Biology 29:997–1002.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Atkinson, S. N., and M. A. Ramsay. 1995. The effects of prolonged fasting on the body composition and reproductive success of female polar bears (*Ursus maritimus*). Functional Ecology 9:559–567.

Atkinson, S. N., R. A. Nelson, and M. A. Ramsay. 1996. Changes in body composition of fasting polar bears (*Ursus maritimus*): the effect of relative fatness on protein conservation. Physiological Zoology 69:304–316.

Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015a. Demographic composition and behavior of polar bears summering on shore in Alaska: U.S. Geological Survey Administrative Report, 26 pp.

Atwood, T., E. Peacock, K. Burek-Huntington, V. Shearn-Bochsler, B. Bodenstein, K. Beckmen, and G. Durner. 2015. Prevalence and spatio-temporal variation of an alopecia syndrome in polar bears (*Ursus maritimus*) of the southern Beaufort Sea. Journal of Wildlife Diseases 51:48-59.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Cattet, M. R. L., N. A. Caulkett, M. E. Obbard, and G. B. Stenhouse. 2002. A body condition index for ursids. Canadian Journal of Zoology 80:1156–1161.

Cherry, S. G., A. E. Derocher, I. Stirling, and E. S. Richardson. 2009. Fasting physiology of polar bears in relation to environmental change and breeding behavior in the Beaufort Sea. Polar Biology 32:383–391.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1990. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395–1403.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Derocher, A. E., and I. Stirling. 1996. Aspects of survival in juvenile polar bears. Canadian Journal of Zoology 74: 1246–1252.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Fowler, C. W. 1990. Density dependence in northern fur seals (*Callorhinus ursinus*). Marine Mammal Science 6:171–196.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Marine Mammal Science 7:123–1 35.

Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS ONE 9(11): e113746. doi:10.1371/journal.pone.0113746.

Henriksen, E. O., Ø. Wiig, J. U. Skaare, G. W. Gabrielsen, and A. E. Derocher. 2001. Monitoring PCBs in polar bears: lessons learned from Svalbard. Journal of Environmental Monitoring 3:493–498.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lydersen, C., P. Assmy, S. Falk-Petersen, J. Kohler, K. M. Kovacs, M. Reigstad, H. Steen, H. Strøm, A. Sundfjord, Ø. Varpe, W. Walczowski, J. M. Weslawski, and M. Zajaczkowski. 2014. The importance of tidewater glaciers for marine mammals and seabirds in

Svalbard, Norway. Journal of Marine Systems 129:452– 471.

McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

Molnár, P. K., A. E. Derocher, G. W. Thiemann, and M. A. Lewis. 2010. Predicting survival, reproduction and abundance of polar bears under climate change. Biological Conservation 143:1612–1622.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., M. R. L. Cattet, T. Moody, L. R. Walton, D. Potter, J. Inglis, and C. Chenier. 2006. Temporal trends in the body condition of Southern Hudson Bay polar bears. Climate Change Research Information Note 3, Ontario Ministry of Natural Resources, Applied Research and Development Branch, Sault Ste. Marie, Ontario, Canada.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Patyk, K. A., C. Duncan, P. Nol, C. Sonne, K. Laidre, M. Obbard, Ø. Wiig, J. Aars, E. Regehr, L. L. Gustafson, and T. Atwood. 2015. Establishing a definition of polar bear (*Ursus maritimus*) health: A guide to research and management activities. Science of the Total Environment 514: 371-378.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.

Ramsay, M. A., R. A. Nelson, and I. Stirling. 1991. Seasonal changes in the ratio of serum urea to creatinine in feeding and fasting polar bears. Canadian Journal of Zoology 69:298–302.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Rode, K. D., R. R. Wilson, E. V. Regehr, M. St Martin, D. C. Douglas, and J. Olson. 2015. Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.

Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.

Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., E. Richardson, G. W. Thiemann, and A. E. Derocher. 2008a. Unusual predation attempts of polar bears on ringed seals in the southern Beaufort Sea: Possible significance of changing spring ice conditions. Arctic 61:14–22.

Stirling, I., G. W. Thiemann, and E. Richardson. 2008b. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568-574.

Stirling, I., C. Spencer, and D. Andriashek. 2016. Behavior and activity budgets of wild breeding polar bears (*Ursus maritimus*). Marine Mammal Science 32:13–37.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Watts, P. D., and S. E. Hansen. 1987. Cyclic starvation as a reproductive strategy in the polar bear. Symposia of the Zoological Society of London 57:305–318.

Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. Science Progress 91:151–173.

Zedrosser, A., B. Dahle, and J. E. Swenson. 2006. Population density and food conditions determine adult female body size in brown bears. Journal of Mammalogy 87:510–518.

Table 13.1. Body condition scores assigned to polar bears in Kane Basin during sampling in the 1990s (1992-1997) and 2010s (2012-2014). Within each epoch and sex-age class, the proportion of individuals in the two BCS categories is presented in parenthesis.

Sex-Age Class	Epoch	Body Condition Score Poor	Body Condition Score Fair-Good	Test Result[1]
Adult Male	1990s	5 (0.10)	44 (0.90)	*P* = 1.000
	2010s	5 (0.11)	40 (0.89)	
Adult Female (Lone)	1990s	9 (0.28)	23 (0.72)	χ^2 = 2.100
	2010s	5 (0.14)	31 (0.86)	*P* = 0.147
Adult Female (w/COY[2])	1990s	11 (0.42)	15 (0.58)	χ^2 = 1.922
	2010s	6 (0.24)	19 (0.76)	*P* = 0.166
Adult Female (w/yearling)	1990s	8 (0.67)	4 (0.33)	*P* = 0.015
	2010s	2 (0.15)	11 (0.85)	
Subadults	1990s	4 (0.29)	10 (0.71)	*P* = 0.037
	2010s	0 (0.00)	16 (1.00)	

[1] Unless otherwise indicated all tests results report values of *P* for Fisher's exact test (two-tailed)
[2] Cub-of-the-year (COY)

Table 13.2. Observations of recent feeding amongst polar bears encountered in Kane Basin, 2012-2014. Evidence of feeding includes presence of seal kills and bears encountered with full, pendulous stomachs.

Sex-age class	**Proportion Feeding (n)**
Adult Male	0.07 (46)
Adult Female (Lone)	0.17 (36)
Adult Female (with offspring)	0.26 (43)
Subadult	0.07 (16)

Table 13.3. Frequency of hair-loss (alopecia) and skin ulcerations on the feet of polar bears captured in Kane Basin, Nunavut, 2012 and 2013.

Sex-Age Class	**No. of Individuals Examined**	**Proportion with Alopecia**	**Proportion with Ulcerations**
Adult Male	12	0.75	0.75
Adult Female (Lone)	9	0.22	0.00
Adult Female (with COY)	7	0.29	0.00
Adult Female (with Yearling)	6	0.50	0.17
Adult Female (with 2-year-old)	1	0.00	0.00
Subadult	4	0.50	0.25
2-year-old	1	0.00	0.00
Yearling	8	0.63	0.25
Cub-of-the-year (COY)	11	0.00	0.00
Total	**59**	**0.40**	**0.22**

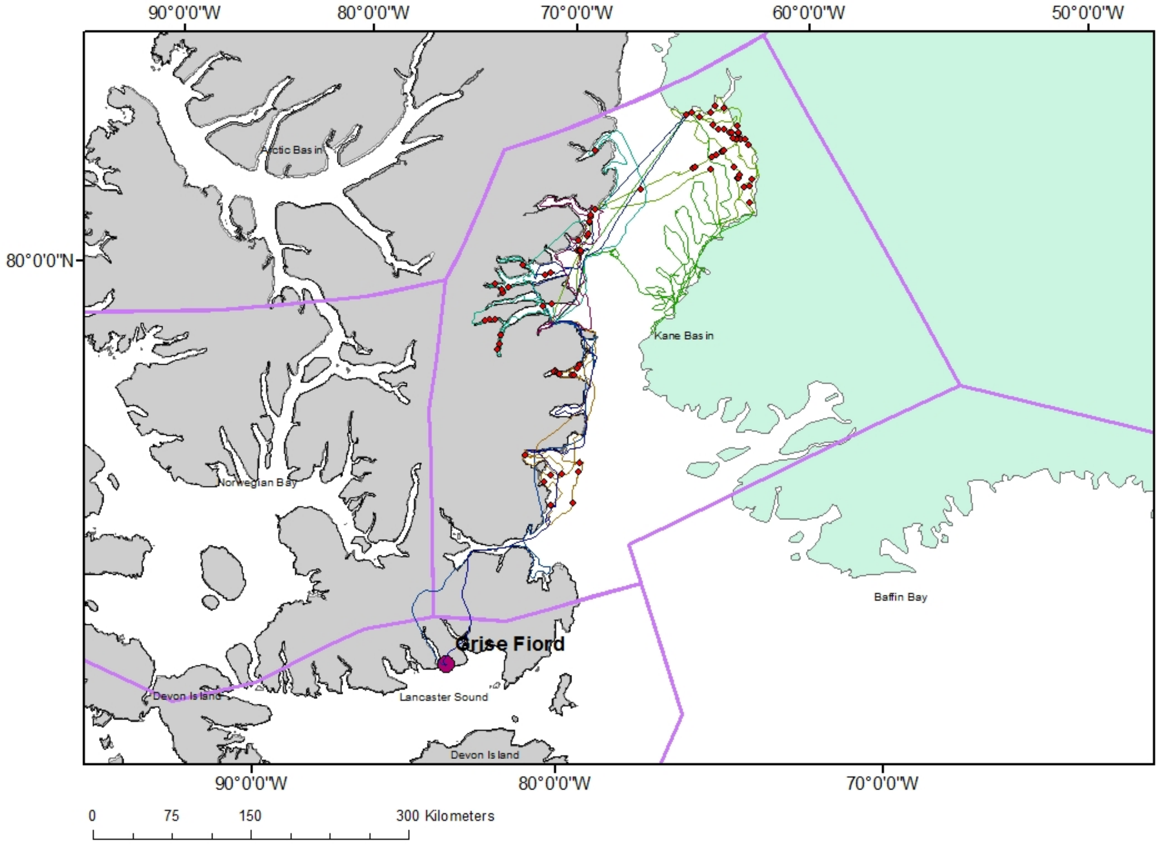


Figure 13.1. Distribution of seal observations during polar bear mark-recapture sampling in Kane Basin, 2013. Flight tracks are shown.

Figure 13.2. Examples of alopecia over the feet of polar bears handled in Kane Basin, April/May 2012 and 2013.

Figure 13.3. Examples of ulcerative lesions on the feet of polar bears handled in Kane Basin, April/May 2012 and 2013.

Figure 13.4. An example of the granular snow found in many parts of Kane basin during polar bear sampling in April/May 2012 and 2013.

CHAPTER 14

GENERAL DISCUSSION AND ADVICE TO THE JOINT COMMISSION

14.1. Conclusions

In 2010, the Canada-Greenland Joint Commission on Polar Bear (JC) tasked the Scientific Working Group with using the best available scientific information to (1) propose Total Allowable Harvest (TAH) levels for the Baffin Bay and Kane Basin subpopulations and provide the JC with a written report of its recommendations, and (2) provide science advice to the JC for monitoring the effects of habitat changes on polar bears. Given the age of the mark-recapture data on which abundance and vital rates for BB and KB were estimated combined with large-scale environmental changes in Baffin Bay during the last decades and suspected large-scale environmental changes in Kane Basin in recent time, the SWG strongly recommended that new estimates of subpopulation abundance, population delineation, and vital rates be given high priority.

Based on the decisions of the JC that physical MR should not be used in this study a multi-year programs began in 2011 (BB) and 2012 (KB) to re-assess the size of both subpopulations using genetic mark-recapture (MR) techniques that involved biopsy sampling from both live and harvested polar bears. In addition, satellite transmitters were deployed on male and female polar bears in NW Greenland during 2009-2013 to study polar bear movement and habitat choice and to gather data for planning and interpretation of the genetic MR study in Baffin Bay. With the same purpose satellite transmitters were deployed on male and female polar bears in both the Canadian (Nunavut) and Greenland parts of Kane Basin in 2012 and

2013. In addition, a helicopter-based aerial survey was flown in Kane Basin in spring 2014 concomitantly with the MR operation to evaluate the feasibility of estimating KB abundance with an aerial survey flown over springtime sea ice.

A total of 2,690 genetic samples were collected from live and harvested polar bears in BB and KB, 125 satellite transmitters were deployed, and 4,160 linear km of transects flown during the aerial survey in KB as part of the overall research program. In addition, many hours were subsequently spent processing samples; analyzing genetic, aerial survey, and sea-ice data; in discussion interpreting the results; and, writing this report.

The study has resulted in new estimates of abundance of polar bears in the Baffin Bay and Kane Basin subpopulations and provided significant and comprehensive information about polar bear ecology and sea-ice dynamics in Baffin Bay and Kane Basin. The results of this program are described in detail in Chapters 2 through 13. They are also summarized concisely in the Executive Summary document.

14.2. Lessons from Genetic Mark-Recapture

The SWG recommended that physical mark-recapture be used for assessing the size of the Baffin Bay and Kane Basin subpopulations. However, as described in Chapter 1 the Canada-Greenland Joint Commission decided to use genetic mark-recapture for assessment.

The Baffin Bay and Kane Basin studies represent the first time that genetic mark-recapture has been implemented at population-wide scales for estimating polar bear abundance and demographic rates. As such, these studies provide valuable information about the utility of genetic mark-recapture as a monitoring tool for assessing polar bear subpopulations ranging over large areas like Baffin Bay with dynamic sea-ice conditions, and how the technique may be most successfully implemented in the future.

Sampling via biopsy darting was highly successful in yielding tissue samples suitable for genotyping in both Baffin Bay and Kane Basin (Chapters 5 and 10) essential if genetic mark-recapture is to be implemented in other subpopulations.

Moreover, genetic analyses identified that reporting of marked bears in the harvest is incomplete; specifically, decreases in reporting are related to greater marker age, suggesting that the loss of physical markers (loss of ear tags and fading of lip tattoos) over time makes it difficult for hunters to correctly identify marked bears. As such, we encourage the use of genetics for identifying marked bears in the harvest in the future; for those subpopulations not inventoried via genetic mark-recapture, this will require genotyping archived samples as well (see also [3] below).

Collecting samples for genotyping via biopsy darting is generally fast, efficient, and less invasive than physical mark-recapture, since bears are not immobilized (Chapter 5). Because biopsying for genetic MR estimation is less time consuming than handling individual bears during physical MR operations genetic MR has the potential of resulting in more “marks” and “recaptures” which theoretically improve precision of estimates of abundance in MR. In Baffin Bay success in sampling a large number of biopsies in the huge coastal distribution areas within a relatively short time was obtained by using three helicopters (2 along eastern Baffin Island and 1 in NW Greenland) during the same time in fall. Using three helicopters during fall biopsying for several years inevitably increased the costs of the surveys.

In physical mark-recapture information on a recapture is obtained from direct physical inspection of the presence of numbered ear tags and/or a number in the lips. Data on marking and recapture is therefore readily at hand for analyses. In contrast, during genetic mark-recapture the information on whether a biopsied bear was a "recapture" or not is not obtained until after genetic analyses are conducted in a qualified laboratory. This adds a delay in the analyses. In our case the commitment of the contracted laboratory, one that is recognized world-wide as an expert and of high quality, to undertake other genetic analyses caused a serious delay in processing the polar bear samples. This resulted in a delay in data analyses for the abundance estimation.

However, there were trade-offs with the genetic mark-recapture method that resulted in lack of information that would have been available with a physical capture protocol. Physical mark-recapture provides a wider range of information including estimates of rates of birth and death, detailed age-structure and body condition information, and a suite of physical samples can further inform individual and population status. Because bears had to be identified from the air using genetic MR, there is uncertainty in the age structure especially for younger bears. Overall, physical mark-recapture permits a more comprehensive assessment of population status, as previously recommended by SWG (2010).

Furthermore, even if physical mark-recapture is not used for assessment, some physical capture is necessary for studies at this scale to provide data on movements of bears and habitat use. In this study, 139 bears were captured in West Greenland, and of these 38 adult females with collars informed both the mark-recapture assessment itself (e.g., temporary emigration analyses, range sizes) as well as provided key information on changes in sea-ice habitat use that set the mark-recapture results into context.

Research in the Kane Basin subpopulation during 2014 illustrated that aerial survey and genetic mark-recapture methods can be implemented simultaneously (Chapters 10 and 11) to generate more comprehensive demographic information and to ensure efficient and representative allocation of sampling effort. In the present study we illustrate that combining the two methods simultaneously is feasible for subpopulations which are surveyed on sea ice during spring and which have a relatively small geographical distribution like the Kane Basin subpopulation. However, using a helicopter like we did with a relatively short range for both biopsying and aerial surveys at the same time may be suboptimal. Using only a single helicopter limits the ability to expand survey effort to offshore polar bear habitat with loose drift ice and open water as demonstrated in our study where a substantial portion of the KB subpopulation's range could not be surveyed (also for safety reasons) leading to an abundance estimate which was negatively biased to an unknown extent.

With genetic mark-recapture, the ability to leverage historical data to improve estimates of survival is limited by the availability of archived samples for genotyping. For both Baffin Bay and Kane Basin, tissues samples suitable for genotyping were available for most – but not all – bears initially marked during research in the 1990s. This lack of tissue samples for a small proportion of the sampled population necessitated identifying those individuals which could still be alive (based on harvest records and age at time of first capture; Chapters 5 and 10) and fixing their recapture probabilities during the 2010s sampling to zero. Although this solution complicated analyses, it enabled us to incorporate historical capture data. Moreover, the large sampling interval between the past and present BB and KB studies did not favor recoveries of old "marks" that also could have assisted in improving some vital rates. If successive population studies are envisaged within a 10-15 year time frame, then a single-year biopsy sampling session should be implemented 5-7 years after the study was completed in order to increase or maintain marks in the population that can be recovered through either harvest or the subsequent new study.

During physical mark-recapture individual age is obtained from every single bear that is handled and marked. During immobilization a vestigial tooth is extracted and individual age is obtained from reading growth-layers in the cementum. This allow for implementing age-structured models for estimating abundance and vital parameters. The fact that individual ages are not obtained from bears that are biopsied during genetic mark-recapture represents a limitation on *post hoc* analyses as indicated in Chapter 5. Hence, we implemented only coarse age structures for demographic analyses, pooling individuals ≥2 years and ≥3 years for the Baffin Bay and Kane Basin studies, respectively (Chapters 5 and 10). Thus, survival could not be estimated for 2 year olds and subadults separately from adults. Ancillary data suggest that experienced biologists may be able to discriminate among finer age classes (i.e., 2 – 4 year old subadults versus adults ≥5 years) with a high degree of accuracy for adults and a lesser degree for subadults (Appendix B), particularly since the sex of individuals is confirmed via genetics (Appendix B). As such, genetic mark-recapture may have the potential to yield estimates of survival and reproductive output that are relevant to management and comparable to previous research however more work needs to be done (e.g., Taylor et al. 2005, 2008a, Peacock et al. 2013). However, it must be kept in mind that field-estimation of individual age during genetic mark-recapture relies heavily upon the individual researcher’s experience in assigning polar bears to more specific age classes.

We completed a detailed review of historical records to assess previous research in Baffin Bay and Kane Basin. These reviews proved critical to our interpretation of results, as apparent changes in the sampling frames between epochs (i.e., incomplete spatial sampling during the

1990s, relative to the 2010s) limited our ability to assess trends in abundance. However, we were not able to locate original and detailed data files from the physical mark-recapture study and telemetry study conducted in Baffin Bay and Kane Basin in the 1990s. This finding underscores the need to archive data securely and for complete and consistent sampling of the study areas. We recommend that any polar bear study archives data securely at institutes that were central in the collaborative studies to allow future studies to thoroughly assess historical inventories to ensure proper interpretation of results. For the same reason, the Canada/Nunavut/Greenland institutes that conducted the recent study in Baffin Bay and Kane Basin have signed a contract where collaborating institutes have access to and must securely archive all data stemming from the joint study.

We defined and stratified the study areas for recent inventories based on recent telemetry data (and historical capture records). For this purpose, data obtained from satellite transmitters deployed in West and Northwest Greenland in 2009 and 2010 for other purposes proved to be an important tool. This process improved our allocation of effort and ensured that results reflected the entire subpopulations as best as possible. Satellite telemetry data (both historical from the 1990s and recent from 2009-2014) were also used to interpret *post-hoc* to what extent the biopsy samplings in the 1990s and 2000s were for the entire subpopulations in BB and KB. Hence, information on movement and area occupancy obtained from satellite telemetry is an essential tool in mark-recapture studies of polar bears.

Finally, the telemetry data identified that 18% of the adult females collared in West Greenland remained in Melville Bay for a least one summer season and in some cases for over a year. There would be value in future work investigating what fraction of the BB subpopulation uses Melville Bay year-round. This could make use of the existing genetic marks from this study, with the addition of one or two sampling seasons in fall in Melville Bay only. Densities appeared low based on observations during data collection in 2012 and 2013 however there would be value in quantifying this using more detailed sampling of the area.

In conclusion, the present study has shown that a genetic mark-recapture assessment of a polar bear subpopulation at a large scale is possible. However, such a study would likely involve an absolute minimum of three years of sampling and, subsequently, additional time for laboratory analysis of samples, analysis of data, and report writing. Thus, these timelines must be considered in advance if timely information on abundance for management is paramount. In this study, our reporting of abundance of polar bears in Baffin Bay and Kane Basin is given 5+ years after initiation of the study in 2011.

14.3. Recommendations on Sustainable Harvest

Historically, the management of polar bears assumed that their sea-ice habitat was relatively stable and, once subpopulation size was known, conservation of polar bears could be achieved through harvest management (SWG 2011). The sustainable harvest of a subpopulation would, therefore, be largely dependent on an estimation of abundance, demographic rates (e.g., birth and death vital rates), and the magnitude, and sex and age composition of the harvest. Since the 1980s, management and conservation of polar bears, particularly in Canada, has been informed by predictive modelling, which has provided guidance to managers on the setting of sustainable harvest levels that have been based on a better understanding of the risk associated with different harvest scenarios (Taylor et al. 1987b, 2002, 2005). The primary predictive model used, RISKMAN (Taylor et al. 2001), focused on harvest management and did not account for any future change in survival or birth rates during the period over which the simulations were run. Furthermore it did not include changes in carrying capacity (K).

Based on demographic rates derived from pooled subpopulation data collected across the Canadian High Arctic in the 1970s and 1980s, modeling indicated that under optimal conditions the sustainable yield of adult females is typically <1.6% of the total population (Taylor et al. 1987a). Hence, level of total sustainable catch (C) was inferred from this relationship (PBSG 1998): C = N * 0.015/sex ratio in harvest; where N is subpopulation size, 0.015 is the sustainable rate of harvest of adult (independent) females from the population, and the denominator is the ratio of adult females in the harvest. Hence, the ratio of adult female bears in the harvest was important for the sustainable total yield. Using this relationship and a general 2:1 sex ratio of males to females in the harvest, the historical standard for the harvest rate of polar bear subpopulations has been 4.5%, which was based on a 2:1 sex ratio of males to females and, initially, on generalized demographic rates derived from pooled subpopulation data collected across the Canadian High Arctic in the 1970s and 1980s (Taylor et al. 1987a, b). More recently, as more studies were undertaken and additional analytical methods developed, it became clear that demographic rates were subpopulation specific (e.g., Amstrup et al. 2001, Taylor et al. 2002, 2005, PBSG 2006, Taylor et al. 2006, Regehr et al. 2007, PBSG 2010). In their recent development of a matrix-based demographic model for polar bears, Regehr et al. (2015) noted that harvest management based on this standard rate and the 2:1 male-biased sex ratio is reasonable under many biological and management conditions; although in some cases, lower or higher rates may be more appropriate.

The ultimate threat to polar bears throughout their range is the reduction in sea-ice habitat expanse, duration, and quality as a consequence of climate change (Derocher et al. 2004, Laidre et al. 2008, 2015a, Wiig et al. 2015). At their 2009 Meeting in Tromsø, Norway, the Polar Bear Range States agreed that the impacts of climate change constitute the most important threat to polar bear conservation (Polar Bear Range States 2015). Hence, the effective and sound management of polar bears can no longer rely solely on estimates of abundance but must also incorporate impacts of a changing environment (e.g., loss of ice and reduction in carrying capacity). In addition, as other threats (i.e., pollution, resource exploration and development, tourism) become better understood, management of polar bear subpopulations will need to be modified, particularly if reproduction or survival rates are negatively affected (Vongraven and Peacock 2011, Vongraven et al. 2012). Thus, understanding the risks associated with a range of harvest management options is important for polar bear conservation.

One of the stated purposes and objectives of the 2009 Canada-Nunavut-Greenland Memorandum of Understanding is to manage polar bears within the KB and BB management units in order to ensure their conservation and sustainable management into the future (Anon. 2009). The SWG was subsequently tasked with proposing Total Allowable Harvest (TAH) levels for both the Baffin Bay and Kane Basin subpopulations (Chapter 1) but was not provided with any specific guidance on management goals.

To facilitate the ability of the SWG to provide the Joint Commission (JC) with useful recommendations on TAH, the SWG requested that the JC provide:

1) A statement of management objectives for each subpopulation,
2) Information on the expected frequency and intensity of future monitoring, and
3) A statement of risk tolerance with respect to the effects of human caused removals.

As a consequence of not receiving the necessary information from the JC on which to base recommendations on TAH despite repeated requests, the SWG is currently unable to provide these recommendations. Specific objectives for the management and conservation of BB and KB polar bears are critical as these will largely influence recommendations on TAH, which in turn will ensure that decision makers have all the necessary information available to them. Risk tolerance and management objectives are not decisions to be made by scientific staff but rather by those with management authority for the resource and, ideally, made after consideration of not only subsistence harvest but also other sources of human-caused removals (e.g., human-bear interactions) and after consultation with all stakeholders.

Once the JC provides the requested information, the SWG strongly recommends that subsequent advice on TAH be based on population simulations using models that have the ability to incorporate changing sea-ice conditions (e.g., reductions in carrying capacity) and demographic data quality as part of the overall process to assess risk under different harvest management scenarios. For example, Regehr et al. (2015) developed a state-dependent management framework that linked the demographic model to simulated population assessments, which can be used to estimate the maximum sustainable rate of human-caused removals. It can also be used to calculate a recommended sustainable harvest rate, which Regehr et al. (2015) note is generally lower than the maximum sustainable rate because it is dependent on management objectives, the precision and frequency of population data, and risk tolerance.

14.4. Recommendations on Monitoring Habitat Change on Polar Bears

There have been numerous reports regarding the effects of climate change and in particular the loss of sea ice, on polar bears (Derocher et al. 1994, Laidre et al. 2008, Wiig et al. 2008). Changes in distribution of polar bears in several populations including in BB have been summarized by Stirling and Parkinson (2006). Furthermore, information on local observations of the effects of climate change on polar bears in BB made by hunters and elders have been presented by Dowsley (2005), Dowsley and Taylor (2006), and Dowsley and Wenzel (2008) for eastern Baffin Island and by Born et al. (2008a, b, 2011) based on interviews with experienced polar bear hunters NW Greenland. These sources indicate that polar bears in BB (and likely also KB) currently are affected by large-scale environmental changes.

Monitoring habitat change will improve our understanding of the relationship between BB and KB polar bears and the environment. It provides insights into how factors such as sea ice and prey abundance and availability affect polar bear distribution and vital rates. The results of the habitat assessment work conducted in Baffin Bay and Kane Basin since 2009, largely based on satellite telemetry data from collared adult female polar bears, have provided a wealth of new information on habitat and habitat change, which directly inform the status of the BB and KB subpopulations (Chapters 2, 4, and 9). The movement information has also assisted in interpreting the BB and KB abundance and demographic data, which were associated with some biases that could be addressed through the use of the telemetry data (Chapter 3).

Vongraven et al (2012) developed a circumpolar monitoring framework for polar bears, which was focused on the sustained long-term monitoring necessary to understand ongoing effects of climate warming and other population-level stressors in order to inform management and policy responses to changing worldwide polar bear status and trends. The current scientific understanding of polar bears and their reliance on sea-ice habitats is the result of long-term monitoring that has been conducted in only a few subpopulations. There is variability in the response of each subpopulation to loss of sea ice, as manifested in this study through the differences in responses in KB and BB despite roughly the same rates of sea-ice loss (Chapters 4 and 9). Therefore it is critical that scientific studies be conducted within the subpopulation(s) of interest to gain an in depth understanding of the complex ecological ramifications of climate change in that area (e.g., Rode et al. 2014), rather than making assumptions based on studies in other areas that may have different responses.

The BB and KB satellite telemetry studies allowed for comparison and quantification of range use across decades, seasons and months, changes in overlap of the population range over time, shifts in median latitude of bears across seasons, changes in immigration and emigration across subpopulation boundaries, changes in movement rates, shifts in sea ice and terrestrial habitat use and habitat selection, and changes in maternity denning timing, and changes in maternity denning areas and habitat. Though not included in the report, time series from captures in BB and KB provide information on causal links between factors that determine health, nutritional ecology and population-level processes. Analyses have been initiated (using samples from recent BB captures) and are expected to provide new information on feeding and nutritional ecology.

The SWG concludes that future physical capture and satellite tagging studies in BB and KB (collaring and tracking adult female bears for periods of years) will be critical to extending the current time series and informing managers of the impacts of sea-ice habitat loss. Given the large physical changes documented in this study, and clear responses of polar bears in both areas, continuation of the time series of satellite telemetry data will improve our understanding of the impacts of future biotic and abiotic changes on the two subpopulations. The satellite telemetry studies should be conducted on intervals of 10 years or less, with samples sizes roughly equivalent to those collected in the 1991-1997 and 2009-2013 (approximately 40 adult females tracked over a period of several years). Lower sample sizes will make assessments more difficult due to individual variability and lack of model convergence (as seen for KB).

By use of satellite telemetry, the present study documented that a group of adult female polar bears occur year round at glacier fronts in Melville Bay in NW Greenland (Chapter 2). To protect important polar bear habitat the Melville Bay Nature Reserve was established in 1980. All access and hunting within the central zone of the nature reserve is prohibited (Appendix D). The (re-)establishment of a local group of polar bears with affinity to Melville Bay appears to be relatively recent and is likely an effect of the protection places on this important polar bear habitat. Satellite telemetry in the 2000s has also shown females to be denning in the Melville Bay Nature Reserve. When the areas were surveyed during spring 1992 and 1993 very little signs of polar bear activity were observed in the nature reserve and of the 1990s satellite telemetry indicated that adult females did not use the Melville Bay (Chapter 2).

Hence, recent data indicate that the Melville Bay Nature Reserve represents an example of how important polar bear habitat can be protected resulting in polar bears reestablishing groups "locally" in prime habitat. Hence, the nature reserve exemplifies a means of protecting polar bears in the future. However, the broader effect of the nature reserve on polar bears should be followed. This can be done by regularly conducting a genetic mark-recapture estimation of trends in numbers in the local group of bears (baseline genetic data now exist from the present study) and by tracking individuals by use of satellite telemetry at 5-10 years intervals.

Continued assessment of changes in habitat in BB and KB via satellite-based observation of sea ice (passive microwave data, MODIS, or Radar images) provide useful context for changes in the physical environment. Arctic sea ice is the most critical habitat for the survival of polar bear subpopulations as distribution and timing of ice relative to critical phases of polar bear life history have been linked to subpopulation status and trend (Stirling et al. 1999, Hunter et al. 2010, Regehr et al. 2010). The SWG recommends continued monitoring of sea-ice habitat change through these studies (described herein as well as Stern and Laidre 2016). Furthermore, contrasting changes in BB and KB with other polar bear subpopulations provides an important baseline for comparison.

Continued development and refinement of habitat models will be necessary in the future to identify habitat selection changes and better predict critical habitat in BB and KB. Standardized methods of developing habitat models (resource selection functions, RSFs) for polar bears have been developed for several subpopulations (Mauritzen et al. 2003, Ferguson et al. 2000, Durner et al. 2004, 2006, Wilson et al. 2014, Laidre et al. 2015b, Chapters 4 and 9) and within a large part of polar bear range (Durner et al. 2009). RSFs are developed from satellite radio telemetry data of adult female bears and readily available sea-ice data in geographic information system (GIS) format. Habitat models are powerful tools for predicting the occurrence of terrestrial den habitat (Howlin et al. 2002, Richardson et al. 2005). Knowledge of the distribution of maternal den habitat has significant management potential to protect polar bears in dens. Trends in sea-ice den habitat may be estimated by monitoring sea-ice conditions as changes in the composition of sea ice has been linked to changes in den distribution (Fischbach et al. 2007).

Habitat availability and change have been linked to polar bear demography and/or condition in some subpopulations (Regehr et al. 2007, Bromaghin et al. 2015, Lunn et al. 2016). Quantitative links between habitat and demographic parameters are complex and need to be refined and specific to the subpopulation of interest. Continued habitat monitoring will improve the understanding between the links to demography and productivity for both BB and KB.

Stable isotope (Bentzen et al. 2007), fatty acid analysis (Iverson et al. 2006), and lipid content in adipose tissues (McKinney et al. 2014) conducted from blood, fat and hair collected during captures can provide information on the polar bear prey base and help to identify shifts in food webs and body condition in BB and KB. This information can be used in concert with information on movements and habitat use from the telemetry to better inform how bears use the ecosystem. While some of this information can be collected from harvest sampling, physical capture of polar bears provides opportunities that are not available from harvest samples (e.g., collection of samples from the same individuals over time).

Continued monitoring of TEK and LEK in BB and KB will also be critical for providing information on how changes in sea ice are impacting the polar bear hunt (e.g., hunting practices), the overall harvest, and the condition of bears harvested. Local perspectives on changes both to the physical environment and the population are important inputs to managers. Repeated studies with a robust interview study design and data collection process (e.g., Born et al. 2011) are needed.

Finally continued subsistence harvest monitoring is needed in both Canada and Greenland, providing critical information on numbers, sex ratios and ages of bears taken in both areas. In this study, this information provided important content into changes in harvest patterns and composition of the harvest (Chapter 8). Genetic validation of the sex of individual bears (as reported by the hunters) showed that the gender was incorrectly reported in a significant number of cases. Improvement in gender reporting is needed; inaccuracies in gender reporting were greatest in Greenland.

14.5. Literature Cited

Amstrup, S. C., T. L. McDonald, and I. Stirling. 2001. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. Journal of Agricultural, Biological, and Environmental Statistics 6:221–234.

Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*, 9 pp.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2008a. Isbjørne i Nordvestgrønland En interviewundersøgelse om fangst og klima, 2008. Pinngortitaleriffik – Grønlands Naturinstitut Teknisk Rapport Nr. 70, 112 pp. (In Danish and Greenlandic with English abstract).

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2008b. Nannut Kalaallit Nunaata avannaata kitaani – Piniarneq silallu pissusaa pillugit apersuilluni misissuineq, 2006. – Pinngortitaleriffik – Nalunaarusiaq Teknikkimut tunngasoq No. 70, 116 pp. (In Greenlandic with English summary).

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Bentzen, T. W., E. H. Follmann, S. C. Amstrup, G. S. York, M. J. Wooller, and T. M. O'Hara. 2007. Variation in winter diet of southern Beaufort Sea polar bears inferred from stable isotope analysis. Canadian Journal of Zoology 85:596–608.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634–651.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2007. Advanced Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, Oxford, UK.

Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. Population Ecology 47:267–275.

Derocher, A. E., and I. Stirling. 1994. Age-specific reproductive performance of female polar bears (*Ursus maritimus*). Journal of Zoology 234:527–536.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1, Government of Nunavut, 43 pp.

Dowsley, M., and M. K. Taylor. 2006. Community consultations with Qikiqtarjuaq, Clyde River and Pond Inlet on management concerns for the Baffin Bay (BB) polar bear population: A summary of Inuit knowledge and community consultations. Nunavut Wildlife Research Group Final Report No. 2, Government of Nunavut, 83 pp.

Dowsley, M., and G. W. Wenzel. 2008. ‘‘The time of the most polar bears’’: A co-management conflict in Nunavut. Arctic 61:177–189.

Durner, G. M., S. C. Amstrup, R. Nielson, and T. McDonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea. Pages 107–120 *in* S. Huzurbazar, editor. Resource selection methods and applications. Omnipress, Madison, Wisconsin, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. Ecology 81:761–772.

Fischbach, A. S., S. C. Amstrup, and D. C. Douglas. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. Polar Biology 30:1395–1405.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Marine Mammal Science 7:123–1 35.

Hansen, K. W. 2010. Polar bear management in Greenland, 2005-2008. Pages 133-134 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Howlin, S., M. Stishov, L. L. McDonald, and S. L. Schliebe. 2002. Den site selection by polar bears on Wrangel Island. Pages 127–138 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Hunter, C. M, H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. Ecology 91:2883–2897.

Iverson, S. J., I. Stirling, and S. L. C. Lang. 2006. Spatial and temporal variation in the diets of polar bears across the Canadian Arctic: indicators of changes in prey populations and environment. Pages 98–117 *in* I. L. Boyd, S. Wanless, and C. J. Camphuysen, editors. Top Predators in Marine Ecosystems: Their Role in Monitoring and Management. Cambridge University Press, Cambridge, UK.

Laidre, K. L., E. W. Born, P. Heagerty, Ø. Wiig, H. Stern, R. Dietz, J. Aars, and M. Andersen. 2015b. Shifts in female polar bear (*Ursus maritimus*) habitat use in East Greenland. Polar Biology 38:879–893.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015a. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications 18:S97–S125.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Mauritzen, M., S. E. Belikov, A. N. Boltunov, A. E. Derocher, E. Hansen, R. A. Ims, Ø. Wiig, and N. Yoccoz. 2003. Functional responses in polar bear habitat selection. Oikos 100:112–124.

McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117–127.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Richardson, E., I. Stirling, and D. S. Hik. 2005. Polar bear (*Ursus maritimus*) maternity denning habitat in western Hudson Bay: a bottom-up approach to resource selection functions. Canadian Journal of Zoology 83:860–870.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere Discussion Papers, in review.

Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

Taylor, M., F. Bunnell, D. DeMaster, R. Schweinsburg, and J. Smith. 1987b. ANURSUS: a population analysis system for polar bears (*Ursus maritimus*). International Conference on Bear Research and Management 7:117-125.

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987a. Modelling the sustainable harvest of female polar bears. Journal of Wildlife Management 51:811–820.

Taylor, M. K., J. Laake, H. D. Cluff, M. Ramsay, and F. Messier. 2002. Managing the risk from hunting for the Viscount Melville Sound polar bear population. Ursus 13:185–202.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667-1673.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2008b. Mark-recapture and stochastic population models for polar bears of the high arctic. Arctic 61:143–152.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. Marine Mammal Science 25:778-796.

Taylor, M., M. Obbard, B. Pond, M. Kuc, and D. Abraham. 2001. RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations. Government of Nunavut, Iqaluit, Nunavut, Canada.

Vongraven, D., and E. Peacock. 2011. Development of a pan-Arctic monitoring plan for polar bears: background paper. Circumpolar Biodiversity Monitoring Programme, CAFF Monitoring Series Report No. 1, CAFF International Secretariat, Akureyri, Iceland.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. Science Progress 91:151–173.

Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA.

Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. Ecosphere 5(10):136. doi:10.1890/ES14-00193.1

APPENDICES

APPENDIX A. List of Scientific Working Group members and external experts involved in the re-assessment of the Baffin Bay and Kane Basin polar bear subpopulations.

Scientific Working Group Members	External Experts
Stephen N. Atkinson Government of Nunavut Canada	Todd Arnold University of Minnesota USA
Erik W. Born Greenland Institute of Natural Resources Greenland	Markus Dyck Government of Nunavut Canada
Kristin L. Laidre University of Washington and Greenland Institute of Natural Resources Greenland	Eric V. Regehr US Fish and Wildlife Service USA
Nicholas J. Lunn Environment and Climate Change Canada Canada	Seth Stapleton University of Minnesota USA
Øystein Wiig University of Oslo Norway	Harry Stern University of Washington USA

APPENDIX B. The accuracy of estimating polar bear age-class and sex from helicopter-based, aerial observations: Implications for the use of non-invasive survey methods in monitoring subpopulations.

INTRODUCTION

Although some subpopulations of polar bears (*Ursus maritimus*) have been studied and monitored for more than 4 decades (e.g., Stirling et al. 1977, Lunn et al. 2016), there is considerable variation in the depth and scope of knowledge across their circumpolar range (Vongraven et al. 2012). Large gaps in basic information exist. Growing concerns about the impacts of climate change, increasing industrial development, harvest and contaminants in the Arctic have prompted range state governments, researchers, environmental organizations and local communities to call for an enhanced and coordinated circumpolar monitoring effort (Vongraven et al. 2012, Range States 2015).

Polar bears typically have been monitored by means of physical mark-recapture. This method has yielded detailed demographic data, allowing researchers to assess the status of subpopulations and closely examine the impacts of climate change and other threats. Hence, physical mark recapture provides tissue samples and samples of e.g., blood and milk which have been used in a wealth of studies of health and pollution in polar bear populations (e.g., Regehr et al. 2007, Hunter et al. 2010, Sonne 2010, Peacock et al. 2013, Bromaghin et al. 2015, Lunn et al. 2016). However, gaps in knowledge and the demand for rapid dissemination of up-to-date information have generated interest in the use of alternative methods for monitoring polar bears. Aerial surveys have proven to be an effective and expedient way of assessing abundance in polar bear subpopulations even in subpopulations with a large range (Aars et al. 2009, Stapleton et al.

2014, 2016). Genetic mark-recapture (Herreman and Peacock 2013, Pagano et al. 2014, Chapters 5 and 10) and remote sensing (Stapleton et al. 2014) may also offer viable alternatives in situations where logistical and financial limitations preclude the use of physical mark-recapture or where concerns about the impacts of handling bears outweigh the benefits (Vongraven et al. 2012).

Aerial surveys are widely used for monitoring wildlife populations. Unlike physical mark-recapture studies where individuals are captured, marked and released, aerial surveys do not provide detailed demographic data such as estimates of birth rates and survival that can be used to project population growth. Nevertheless, aerial surveys, which only rely on one season are an effective means of yielding a snapshot of estimates of abundance/status. Inferences about population trends can be derived from repeated aerial surveys. In addition to estimating abundance, aerial surveys can also provide information on spatial distribution, sex and age composition, body condition and reproductive performance that can be used to facilitate population status assessment (e.g., Stapleton et al. 2014). In contrast to genetic and physical mark-recapture estimation, which relies on several years of sampling, aerial surveys can provide an estimate of abundance from only one season of study.

Genetic mark-recapture has been increasingly used for wildlife population monitoring (Palsbøll et al. 1997, Boersen et al. 2003, Boulanger et al. 2004, Kendall et al. 2009). Like aerial surveys, genetic mark-recapture does not require the capture and physical handling of individuals. Tissue samples are collected for genotyping and identification of individuals by methods such as biopsy darting or hair-snagging. With protocols such as biopsy darting, individuals can be observed from a distance, facilitating collection of additional information on sex, age class and body condition. However, the reliability of this ancillary information depends on the ability to correctly classify individuals by sex and age class.

Using data collected during two recent genetic mark recapture studies of polar bears in the Baffin Bay (BB) and Kane Basin (KB) subpopulations, we assessed the accuracy of classifying polar bears into sex and age classes from the air without physical handling. From a sample of bears of known sex and age class, we examined variation in accuracy of classification dependent on the method of survey (aerial vs genetic mark-recapture) and amongst sex-age classes. We discuss the implications of the results for expanding the utility of aerial survey and genetic mark-recapture as less invasive methods for monitoring species status.

METHODS

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all Bylot Island (66.2° N to 73.8° N) in Nunavut/ Canada and parts of West and Northwest Greenland (66.0° N to 77.0° N; Taylor et al. 2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south. Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A small number of bears remain on land in northwestern Greenland throughout the ice-retreat period.

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit; *cf.* Figure 13.1). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, in recent decades, sea ice conditions in KB have changed markedly (Chapter 9)

<u>Genetic Mark-Recapture Study</u>

Genetic mark-recapture studies were conducted in BB and KB between 2011-2013 and 2012-2014 respectively (Chapters 5 and 10 in this report). Sampling of bears in BB occurred from late August to mid- October along the east coast of Baffin Island and around Bylot Island, Canada. During this period, bears were on land in a variety of habitats ranging from flat coastal plains and beaches to steep rocky slopes and glaciers. Bears were observed against different backgrounds including sand, rocks, low lying vegetation, snow and water. Sampling in KB occurred in April and May while bears were on the sea ice. Bears were observed on a range of sea-ice types including flat, shorefast ice, consolidated pack-ice and unconsolidated pack-ice.

Using a helicopter (Bell 206 LongRanger), we searched for and biopsy darted polar bears using methods described previously (Chapters 5 and 10). Upon encounter, the sex and age class (cub-of-the-year [COY], yearling, subadult [ages 2 – 4], and adult) of each bear was estimated from the air at a range of 3 – 7 m above ground. The individual identity and sex of each bear was later confirmed via genetic analysis (Chapter 5 and 11). In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males; e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under the tail of females). Fields notes, and in some cases photographs, also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. All observations were made by experienced polar bear biologists who had previously participated in physical mark-recapture studies.

At the time of encounter, observers had no prior knowledge of the sex or age class of bears. However, amongst the individuals encountered, a proportion were of known sex and age class based on one or more lines of evidence (Table B1), including a number of bears whose genotypes matched those of bears handled during physical mark-recapture studies in BB (Taylor et al. 2005), KB (Taylor et al. 2008, Chapter 10 in this report) and Davis Strait (Peacock et al. 2013). We assessed the accuracy of remote classification using this sample of ‘known’ bears and examined two scenarios. The first scenario simulated the outcome of an aerial survey in which the sex of bears cannot be confirmed via genotyping. Sex and age classification under this scenario therefore relies solely on field observation. The second scenario simulated a genetic mark-recapture, whereby inaccuracies in field sexing of bears can be corrected following genotyping and field notes made at the time of observation can be used to make *post-hoc* adjustments to age class once genetic sex is known[1].

We restricted our analyses to bears that were sub adults or adults at time of encounter due to small sample sizes for COYs and yearlings and because these dependent offspring can be easily identified when part of a family group (> 96% and 91% accurate for COY and yearlings respectively; GN unpublished data from Davis Strait). Our analysis was a simple comparison of the estimated and known frequencies of bears in each sex and age class under these two scenarios.

RESULTS

During genetic mark-recapture studies in BB and KB, 2011-2014, there were 309 encounters with individuals classified from the air as adult females based on the presence of accompanying offspring (either COY or yearlings), including 29 instances in which the sex and age class of the adult female was also known from capture and physical examination (n = 2) and tooth aging (n = 27) on a prior or future occasion. Twenty-five adult females were subject to aerial classification during fall when they were accompanied by COY (n = 12) or yearlings (n = 4) and 4 were classified during spring (3 with COY, 1 with yearlings). All adult females with dependent offspring were correctly classified from the air.

In addition to adult females with accompanying offspring, we recorded 128 unencumbered bears of known sex and age class (Table B2). Aerial classification of these bears without subsequent genotyping and reclassification based on genetic sex (i.e., the aerial survey scenario) resulted in an overall accuracy of 73%. For lone adults, 95% males and 74% of females were

---

[1] An example of a field note used for post-hoc adjustment of age class would be: "*If not female is a sub adult male*".

correctly classified. Inaccuracies were greatest amongst subadult bears. Although 70% of subadults were correctly classified, only 23% were correctly classified as sub adults of a particular sex.

Aerial classification combined with subsequent reclassification based on genetic sex, field notes and photographs (i.e., the genetic mark-recapture scenario) resulted in an overall accuracy of 91% amongst the 128 known age, independent bears. Again, accuracy varied by sex and age class (Figure B1); accuracy was highest for adult males (97%) and lowest for sub adult females (79%) (Table B3).

DISCUSSION

One of the criteria used to classify adult females was the presence of dependent offspring (COY or yearling) at the time of aerial observation or during a prior encounter. Use of this criterion was based on the assumption that accuracy in identifying females with offspring of this age was at, or near 100%. Although the sample size was relatively small, our results support this assumption and the validity of this age classification criterion. All of the adult females with offspring whose age could also be confirmed by tooth aging or physical examination were correctly classified from the air. However, we did not have any adult females accompanied by 2-year-olds in our sample of known-aged bears so we were unable to test the accuracy of classifying adult females based on the presence of 2-year-old offspring nor were we able to test accuracy in classifying 2-year-olds themselves.

Our results suggest that experienced observers can estimate the sex and age class of bears from the air with high accuracy for most sex and age classes, particularly when aerial observations are combined with genetic sexing, field notes and photographs. These findings are consistent with results from a similar study using a larger dataset of known-age bears (n = 445 based on tooth aging) from Davis Strait in which aerial observers correctly classified 97%, 88%, 80% and 80% of adult males, adult females, subadult males and subadult females respectively (GN unpublished data).

Not surprisingly, the greatest inaccuracies occur in classifying subadult bears. While the ability to classify an individual as a subadult is reasonably good from the air, the ability to determine the sex of subadult bears based on aerial observation alone is poor. Another area of potential inaccuracy that we were unable to test was the identification of independent yearlings. In some polar bear subpopulations, a proportion of yearlings are found alone during the summer or fall; presumably having been weaned (Derocher and Stirling 1995, 2012, Stirling et al. 1999). In Baffin Bay during the 1990s, approximately 6% of yearlings encountered during mark-recapture sampling were independent (GN unpublished data). The extent to which weaning of yearlings is occurring in BB at present is unknown. Of 16 bears recaptured as yearlings during genetic mark-recapture sampling from 2011 to 2013, all were still with their mother, but this small sample size limits inferences. However, in Western Hudson Bay, the proportion of yearlings that are independent during the fall has declined dramatically from > 81% prior to 1980 to almost zero at present (Stirling and Derocher 2012). This decline in early weaning of offspring has occurred in association with changing sea-ice conditions leading to the suggestion that early weaning is associated with favorable environmental conditions. Given trends in sea ice in BB (Laidre et al. 2015), a reduction in the proportions of independent yearlings may also be occurring. The number of independent yearlings encountered during our genetic mark-recapture was likely negligible.

Based on the accuracy of classification documented in this study, we conclude that the sex and age class data derived from aerial surveys or genetic mark-recapture studies can provide reliable data to support monitoring and assessment of population status. From aerial observations, adult males, adult females, COY, and yearlings can be identified with high accuracy. Subadult age classes also can be accurately determined, but classification by sex is poor. Combined with genetic sexing, field notes and photographs, accuracy is improved for all classes of bears. While we cannot derive specific ages for bears from aerial observations, such data do support monitoring the basic age structure of subpopulations. In addition, with genetic mark-recapture, there is an opportunity to model survival of specific age classes, albeit with a degree of uncertainty. For example, remote classification of sex and age classes does not permit modeling senescent age classes, nor can we model the transition from subadult to adult age classes with certainty. Finally, given the accuracy in identifying adult females and their dependent COY and yearling offspring, reproductive indices such as litters size and recruitment (yearlings per adult female) can be reliably obtained.

In our genetic mark-recapture studies in Baffin Bay and Kane Basin approximately one-third and two-thirds of sampling, respectively, occurred for bears of known age class based on our criteria (Table B1). The higher proportion of known age bears in Kane Basin was the result of physical captures completed to deploy satellite telemetry instruments. For the two thirds and one third of bears of ‘unknown’ age that were age classed based on aerial observations, genetic sex, field notes and photographs we can be confident in the accuracy of those classifications. For mark-recapture analyses, we adopted a coarser age class structure than was assessed in the present study due to concerns about the ability to remotely classify bears (Chapters 5 and 10). Our findings suggest that the accuracy of remote classification is sufficient to justify the use of finer scale age-class structures in the future.

LITERATURE CITED

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. Wildlife Society Bulletin 31:197–207.

Boulanger, J., S. Himmer, and C. Swan. 2004. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British Columbia. Canadian Journal of Zoology 82:1267–1277.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634–651.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Derocher, A. E., and I. Sitrling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Hunter, C. M, H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. Ecology 91:2883–2897.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. Journal of Wildlife Management 73:3–17.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016.

Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Palsbøll, P. J., J. Allen, M. Bérube, P.J. Clapham, T. P. Feddersen, P. S. Hammond, R. R. Hudson, H. Jørgensen, S. Katona, A. H. Larsen, F. Larsen, J. Lien, D. K. Mattila, J. Sigurjónsson, R. Sears, T. Smith, R. Sponer, P. Stevick, and N. Øien. 1997. Genetic tagging of humpback whales. Nature 388:767–769.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Sonne, C. 2010. Health effects from long-range transported contaminants in Arctic top predators: An integrated review based on studies of polar bears and relevant model species. Environment International 36:461–491.

Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014a. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stapleton, S., M. A. LaRue, N. Lecomte, S. N. Atkinson, D. Garshelis, C. Porter, and T. Atwood. 2014b. Polar bears from space: Assessing satellite imagery as a tool to monitor *Ursus maritimus*. PLoS ONE doi:10.1371/journal.pone.0101513.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

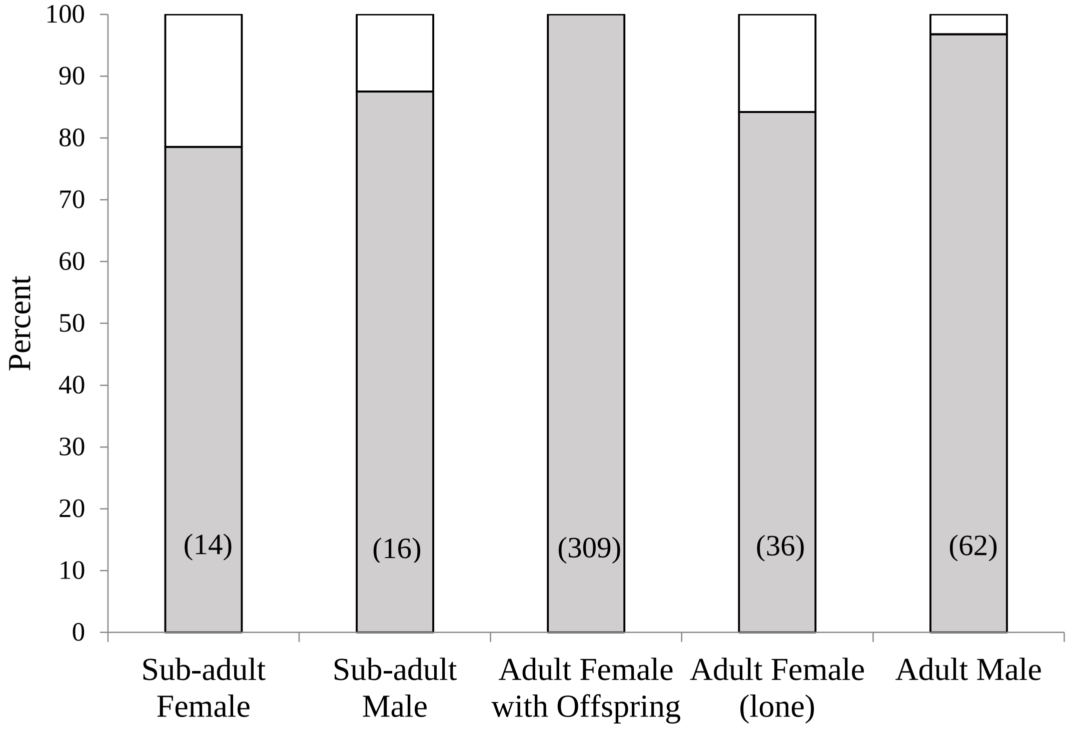


Figure B1. Accuracy of estimating the age class of bears during genetic mark-recapture studies in Baffin Bay and Kane Basin (2011-2014). Data are for bears of known age-class and sex. Sex is based on genotyping and age class is based on one or more of the criteria listed in Table A1. Data are presented as percentages correctly (grey) and incorrectly (white) classified with sample sizes in parentheses.

Table B1. Lines of evidence (criteria) used to determine the 'known' sex and age class of polar bears.

Sex-Age Class	Evidence Used to Determine Class
Subadults	• Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior or future occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be subadult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].
Adult Females with dependent offspring (COY or yearling)	• Accompanied by dependent offspring at time of encounter or during previous encounter and / or capture. • Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior or future occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].
Lone Adult Females	• Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior occasion or future occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to an adult female previously accompanied by dependent offspring. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].
Adult Males	• Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].

[1] Age estimated by counting annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998).

Table B2. Comparison of sex and age classes as estimated from the air versus known sex and age for bears observed in Baffin Bay and Kane Basin, Canada, 2011-2014. Estimated sex and age based on aerial observation only (aerial survey scenario). Data are frequencies. Percentage of bears correctly classified are presented in parentheses.

			Known				
			Sub adult (<5yrs)		**Adult**		
			Female	**Male**	**Female (with offspring)**	**Female (Lone)**	**Male**
Estimated	**Sub adult (<5yrs)**	**Female**	3 (21.4)				
		Male	3	4 (25.0)			1
		Unknown/Not recorded	5	6			
	Sub adult or adult	**Female**				1	
		Male				2	1
		Unknown/Not recorded				1	
	Adult	**Female (with offspring)**			309 (100)		
		Female (Lone)	3	4		28 (73.7)	1
		Male		2		4	59 (95.2)
		Total Individuals	14	16	309	36	62

Table B3. Comparison of sex and age classes as estimated from the air versus known sex and age for bears observed in Baffin Bay and Kane Basin, Canada, 2011-2014. Estimated sex and age based on aerial observation and post-hoc correction for genetic sex (genetic mark-recapture scenario). Data are frequencies. Percentage of bears correctly classified are presented in parentheses.

			Known				
			Sub adult (<5yrs)		**Adult**		
			Female	Male	Female (with offspring)	Female (Lone)	Male
Estimated	**Sub adult (<5yrs)**	Female	11 (78.6)				
		Male		14 (87.5)			1
		Unknown/Not recorded					
	Sub adult or adult	Female				4	
		Male					1
		Unknown/Not recorded					
	Adult	Female (with offspring)			309 (100)		
		Female (Lone)	3			32 (88.9)	
		Male		2			60 (96.8)
		Total Individuals	14	16	309	36	62

APPENDIX C. Regression results for an annual body condition metric for polar bears in Baffin Bay (BB). The metric, proportion of bears in good condition, was derived from observed frequencies of Fatness Index (FI) scores ranging from 1 to 5 (Stirling et al. 2008). Bears of FI 4 or 5 were in good condition. Spring ice transition was the decimal day (1-365) when ice cover over the continental shelf of BB reached 50%. Data for all years were collected within a standardized sampling area (see chapter 3). Regressions were performed in the Curve Estimation procedure of SPSS (Version 24.0).

Sex-Age Class	**Dependent Variable**	F_6	r^2	***P***	***Curve Type***
Adult Male	Spring Ice Transition	59.89	0.97	**0.001**	Quadratic (2nd order)
Adult Male	Year	18.90	0.79	**0.007**	Linear
Adult Female (alone)	Spring Ice Transition	0.25	0.05	0.635	Linear
Adult Female (alone)	Year	2.09	0.29	0.208	Linear
Adult Female (with offspring)	Spring Ice Transition	51.77	0.91	**0.001**	Exponential
Adult Female (with offspring)	Year	13.24	0.73	**0.015**	Exponential
Yearling	Spring Ice Transition	9.75	0.83	**0.029**	Quadratic (2nd order)
Yearling	Year	4.71	0.49	0.082	Linear

LITERATURE CITED

Stirling, I., G. W. Thiemann, and E. Richardson. 2008. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568–574.

APPENDIX D. Description of the polar bear harvest management and monitoring systems in Canada and Greenland.

HARVEST MANAGEMENT AND MONITORING IN CANADA

Within Canada, harvesting of polar bears has been managed on a quota system since 1967 (Lee and Taylor 1994). This system has undergone several revisions over time both in terms of the size of quotas and the methods of management. Since 1996, the quota system for BB, KB and other neighboring subpopulations managed by the Government of the Northwest Territories (and now the Government of Nunavut) has had several notable features. The term quota has been replaced by the term Total Allowable Harvest (TAH) to reflect language in the Nunavut Land Claims Agreement (NLCA). However, TAH and quota are often used interchangeably in reports. The TAH is managed on a flexible system that allows the annual level of harvest to vary somewhat from year-to-year to some degree. Each community hunting from a subpopulation is allocated a portion of the TAH. Exceedance of that allocation in a given year is compensated for by a reduction in the community's allowable harvest the following year. Exceedances occur when the total number of bears harvested is greater than the available limit or when too many females are harvested. This sex selective harvest management system is based on a target sex ratio of 2 or more males for every female harvested which allows a higher sustainable harvest than a 1:1 sex ratio (Taylor et al. 2008b). All human-caused mortalities are counted against the available TAH. Adult females with dependent offspring (cubs-of-the-year, yearling or two-year-olds) and those in or constructing dens are protected from hunting. The hunting season runs from July 1st to June 30th the following year. Most hunting is for subsistence purposes by Inuit. However, a portion of each community's TAH may be allocated to guided sport hunting by non-Inuit, at the discretion of Inuit (Tyrell 2009; Wenzel 2008, 2011). Methods of hunting are regulated under the *Nunavut Wildlife Act*. With the exception of sport hunters who must travel by dog-team only, hunting may be facilitated by use of various forms of transportation including snow machine, All Terrain Vehicle or boat and with firearms of authorized calibers.

In Canada (Nunavut), the reporting of all harvested bears is mandatory. Hunters are required to provide evidence of the sex in the form of a baculum from harvested males. In the absence of a baculum, sex may be determined by DNA (Prior to 2005, proof of sex could also be established by sworn affidavit). When proof of sex is unavailable a harvested bear is counted as a female for quota management purposes. In addition to proof of sex, hunters are required to submit a set of standard specimens from each harvested bear including the lower jaw (or skull) for extraction of a tooth for aging, lip tattoos and ear tags if present. Payment is provided by the Government of Nunavut (GN) for these specimens.

Following harvest of a bear, hunters submit the required specimens and other information to local GN Conservation Officers. For each bear, details are recorded including location of harvest, date, hide length, estimated age, sex, type of hunt (e.g., regular subsistence, sport hunt, defense-of-life-and-property), ear tag number (and tags) if present and lip tattoo number (and tattoo) if present. The hide from each harvested bear is then marked with a uniquely numbered tag (hide seal) that is permanently affixed. This hide seal is required for export of hides from Nunavut and sale.

The information collected on each harvested bear is recorded on Hunter Kill Return (HKR) forms completed by Conservation Officers. HKR forms are submitted to the GN's polar bear management program where they were checked for omissions and errors before entry into the GN’s polar bear harvest database. The sex of harvested bears is again verified by submission of a baculum or by DNA submitted for analysis to Wildlife Genetics International (Nelson, BC, Canada). The age of harvested bears is determined from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998).

HARVEST MANAGEMENT AND MONITORING IN GREENLAND

Harvest management

Following the signing of the *Agreement on Conservation of Polar Bears* in 1973 (Anon. 1973), regulations for the harvest and the protection of polar bears for all areas of Greenland were introduced and were enforced by 1 January 1975 (Anon. 1976, Vibe 1985). Since then, several amendments have been made to the regulations (*cf*. Born 1995). The latest amendment was made in 2005 (Anon. 2005).

When Greenland Home Rule was established in 1979, Greenland took over the legal responsibility for management of its renewable resources, including polar bears. In October 2005, a new Executive Order (Anon. 2005) came into force. Some important protective measures in this executive order are (Anon. 2005, Lønstrup 2006, Hansen 2010):

- year round protection of all cubs (regardless of age) and females accompanied by cubs. The executive order also introduces a prohibition of the export of polar bear cubs;
- protection of all polar bears from 1 July to 31 August; in the local authority districts of Ittoqqortoormiit og Ammassalik from 1 August to 30 September;
- prohibition to disturb or dig out polar bears in dens;
- introduction of quotas from 1 January 2006 and the possibility that part of the quota may be used for trophy hunting. There has never been and currently is no trophy

hunting of polar bears in Greenland. Special provisions on trophy hunting will be laid down in a separate executive order;

- only Greenland residents who hunt as a full-time occupation are allowed to hunt polar bears;
- it is mandatory to report to the Greenland management authorities all catches including struck-and-lost polar bears;
- aircraft, helicopters, motorized vehicles, including snow scooters and boats larger than 20 GRT/15GT are not allowed in the hunt or for transportation to and from the hunting grounds;
- poison, traps, foot snares or self-shooting guns are not allowed;
- rim-fire rifles, shot guns or semi- or fully automatic weapons are not allowed. Polar bears may only be hunted using a rifle with a minimum caliber of 30.06 (7.62 mm);
- all meat, skin and other useable parts of the bear must be brought back (or cached in the field for later use); and,
- no parts of the polar bear must be sold until the catch has been officially registered and the license has received an official stamp.

The Melville Bay Nature Reserve offers protection of polar bears in the Baffin Bay subpopulation. This reserve (10 500 km^2) was established in1980 to protect important polar bear habitat. All hunting within the central (coastal) zone I of nature reserve is prohibited (Vibe 1985, Anon. 1989).

The Greenland Home Rule Act. No. 12 of 29 October 1999 provides the legal framework for wildlife management. Various laws on environmental protection and animal welfare also apply to the management of polar bears (Polar Bear Range States 2015).

Responsibility for the management of polar bears resides with the Department of Fisheries, Hunting and Agriculture, DFHA (Aalisarnermut, Piniarnermut Nunalerinermullu Naalakkersuisoqarfik, APNN) of the Greenland Government.

Quotas for the take of polar bears in Greenland were introduced in 2005 taking effect 1 January 2006 (Lønstrup 2006). The Minister of Fisheries, Hunting and Agriculture sets an annual polar bear quota. The minister drafts a preliminary regional allocation of the quota based on the latest scientific advice and harvest results for the preceeding harvest season, and then sends the draft to the Hunters´ National Association, the municipalities, the Ministry of Nature and Environment and the Greenland Institute of Natural Resources for consultation for a period of not less than five weeks. Based on the resulting consultation, the Ministry of Fisheries, Hunting and Agriculture prepares a final presentation of the total annual quota to the Minister (Polar Bear Range States 2015). Licenses to hunt polar bears are issues by the municipalities, within annual quotas set by DFHA and the National Government.

The Greenland Fisheries License Control Authority is tasked with enforcing the regulations set by the government and the municipalities (Polar Bear Range States 2015).

When polar bear studies conducted by the Greenland Fisheries Research Institute (predecessor of Greenland Institute of Natural Resources, Nuuk) were initiated in 1991, Greenland de facto took over the responsibility providing scientific data for the management of its polar bear subpopulations as outlined in the 1973 *Agreement on Conservation of Polar Bears*.

During the fall of 2000, the Greenland Home Rule Government signed a Memorandum of Understanding (MOU) with the Government of Nunavut (Canada). An appendix to this MOU contains a prioritized list of items, including that there should be cooperation between both regarding shared polar bear subpopulations (Lønstrup 2006).

Harvest Monitoring

Home Rule was established in Greenland 1979. Since then the Department of Fisheries, Hunting and Agriculture (DFHA, Nuuk) has been responsible for organizing the collection of catch statistics in Greenland and for summarizing and publishing the data. Until 1987 information about the number of polar bears taken in Greenland was available through the Hunters´ Lists of Game (HLG) where hunters reported their catch of various wildlife including polar bears voluntarily. The HLG (Anon. 1954-83 and unpublished 1984-87) was based upon the principle that an appointed person from each settlement kept count of the catch of various hunting animals by all the hunters in his settlement and reported the numbers to the authorities (Rosing-Asvid 2002). When such reports for some reason were missing, the central authority added an estimate to account for unreported catch based upon "other information" (i.e., for example notices of catches in newspapers or records of trade of skin etc.). The HLG-summaries of the catch, including estimates of unreported catch, were published annually by the Ministry for Greenland in Copenhagen (until 1983). After the Greenland Home Rule Government took over the HLG-system, unpublished summaries of the catch in 1985, 1986 and 1987 became available from the Department of Fisheries, Hunting and Agriculture, DFHA (Nuuk). However, the reliability of the reporting in the HLG deteriorated since about the mid-1970s (Born 1995, Rosing 1998) and this way of monitoring the catch in Greenland stopped in 1987.

Some information (HLG and trade) was available during the decade prior to the introduction of a new system of reporting catches in 1993 (see the following). However, none of these sources gave the total picture and the size of the annual catch of polar bears from the BB and KB subpopulations in the 1980s and the information of annual catches was largely based on estimates (Born 1995). Born (1995) and Rosing-Asvid (2002) estimated that during 1980-1992 a total of between 25(30) and 70(80) polar bears were taken each year by Greenlanders from the BB and KB subpopulations (the estimates of the total catch was not separated to BB and KB).

On January 1993 a new system of reporting catches – the ”Piniarneq” (Greenlandic word for “catch”) – was introduced in Greenland on 1 January 1996. The “Piniarneq” relies upon each hunter voluntarily reporting his annual catch (between 1 October and 30 September) of various species including polar bears. The Piniarneq system is linked to the issuing of hunting licenses, of which two categories exist: one for full-time hunters and another for part-time hunters. Hunters in both categories have to pay a small fee for renewal of the license, at which time they are obliged to report their catches during the previous 12 months. Only full-time hunters can get a license to hunt polar bears. The information on catches is compiled by the Department of Fishery, Hunting and Agriculture (Nuuk) which publishes summaries of the annual catch by area. In the summaries, the catch is reported by municipality, meaning that in ”Piniarneq” there are no records of the exact site of kill. There is also no information on sex and age of the caught bears or whether the bear(s) was (were) killed during a hunt involving more hunters than the one reporting the catch(es).

In recognition of the fact that a potential problem of reporting catches of polar bears via Piniarneq might be that more than one hunter reports the catch of a bear because several hunters participated in the hunt leading to multiple- reporting of a kill (see Discussion) it became mandatory from 1994 to report every kill of a polar bear on specific forms (“Special Reporting Forms”; “Særmeldingsskemaer”). On these forms the hunter´s name, civil registration number, settlement/town, place and date of the kill, sex and approximate age (young, adult, old) of the bear ha to be given. To be able to reduce the problem of potential multi-reporting of a single kill the hunter who finished off the bear was supposed to fill in the form and also give the name etc.

of the other participants in the catch. During the years this system worked with variable success and not all bears that were caught were reported (Born 1998).

Quotas on polar hunting were first introduced in Greenland in 2006. After the introduction of quotas taking effect 1 January 2006 the Ministry of Fisheries, Hunting and Agriculture has improved the hunting statistics by developing a new database and a double reporting system. This means that a hunter must be issued a license before the hunt and immediately following the hunt the hunter must report the catch to the local authority using a standardized form. This standardized form includes information on the name of the hunter(s), place of residence, date, license number, location of kill, and the sex and age category, and whether the bear was marked. As an additional control, all hunters must report their annual harvest of all species (including polar bears) in Piniarneq (Hansen 2010).

In connection with studies of movement and subpopulation assessment more than 1500 polar bears have been physically marked in the Kane Basin and Baffin Bay subpopulations since the 1970s (Born 1995, Taylor et al. 2005, 2008, Peacock et al. 2012). Before the quotas the hunters received a token payment for returning marks (and transmitters) to the authorities with information on the bear (date, site, sex etc. etc). By the introduction of quotas it became mandatory to report whether a bear was marked or not and return tags (and transmitters) (Anon. 2005).

Since the 1980s biological samples from the polar bear catch (various tissues, sexual organs, teeth for ageing etc.) have been collected during various specific programs in connection with studies of pollution and the demography of the catch (e.g., Rosing-Asvid 2002, Sonne et al. 2012). However, these programs which relied upon the hunters collecting the samples with an economical compensation served specific purposes and in case of monitoring the catch have been intermittent (Rosing-Asvid 2002).

In order to continuously monitor the Greenland catch of polar bears demographically and provide information on sex and age composition of the catch it became mandatory in 2012 for the polar bear hunters to deliver a tissue sample (for genetic analyses) and a small vestigial tooth (for age determination) from each bear killed. The samples shall be sent to the Greenland Institute of Natural Resources in Nuuk which is responsible for arranging the sampling program practically and for processing the samples. Each sample is accompanied by a filled form where with details about the catch (date, site, name of hunter, sex of the bear and its approximate age etc. etc.). The hunters are required to also send the same information to the Greenland management authorities (i.e., the Department of Fisheries Hunting and Agriculture in Nuuk).

LITERATURE CITED

Anon. 1954-87. Sammendrag af Grønlands Fangstlister. Ministeriet for Grønland [Summaries of “Hunters´ Lists of Game”]. Summarized during 1954-1984 by the Danish Ministry of Greenland, Copenhagen, and during 1985-1987 by the Ministry for Fisheries, Hunting and Agriculture, Nuuk. [In Danish].

Anon. 1973. *Agreement on the Conservation of Polar Bears*. Available at: http://pbsg.npolar.no/en/agreements/agreement1973.html.

Anon. 1976. Bekendtgørelse om ændring af bekendtgørelse om fredning af isbjørne i Grønland [Executive order of preservation of polar bears in Greenland]. Nalunaerutit-Grønlandsk Lovsamling, Serie A, Nr. I, MJ.G.j. nr. 1701-D8: 109. [In Danish].

Anon. 1989. Hjemmestyrets bekendtgørelse nr. 21 af 17. maj 1989 om naturreservatet i Melville Bugt [Greenland Home Rule Executive order No. 21 on the Melville Bay Nature Reserve]. http://www.lovgivning.gl/gh.gl love/dk/1989/bkg/bkg_nr_21-1989_dk/bkg_nr_21-19, 05-01-2010. [In Danish].

Anon. 2005. Hjemmestyrets bekendtgørelse nr. 21 af 22. september 2005 om beskyttelse og fangst af isbjørne [Greenland Home Rule Executive order No. 21 of 22 September 2005 on protection and harvest of polar bears], 5 pp. [In Danish].

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W. 1998. Research on polar bears in Greenland, primo 1993 to primo 1997. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Hansen, K. W. 2010. Polar bear management in Greenland, 2005-2008. Pages 133-134 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Lee, J., and M. Taylor. 1994. Aspects of the polar bear harvest in the Northwest Territories, Canada. International Conference on Bear Research and Management 9:237–243.

Lønstrup, J. 2006. Polar bear management in Greenland. Pages 133–134 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

Rosing, B. 1998. Issues pertaining to polar bear management in Greenland. Pages 93-95 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Rosing-Asvid, A. 2002. The polar bear hunt in Greenland. Greenland Institute of Natural Resources Technical Report 45, 25 pp.

Sonne, C., R. J. Letcher, T. Ø. Bechshøft, F. F. Rigét, D. C. G. Muir, P. S. Leifsson, E. W. Born, L. Hyldstrup, N. Basu, M. Kirkegaard, and R. Dietz. 2012. Two decades of biomonitoring polar bear health in Greenland: A review. Acta Veterinaria Scandinavica 54 (Supplement 1):S15 doi:10.1186/1751-0147-54-S1-S15.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

Tyrrell, M. 2009. Guiding, opportunity, identity: the multiple roles of the Arviat polar bear conservation hunt. Pages 25–38 *in* M. M. R. Freeman, and L. Foote, editors. Inuit, Polar Bears and Sustainable Use: Local, National and International Perspectives. Canadian Circumpolar Institute Press, Edmonton, Alberta, Canada.

Vibe, C. 1985. Miscellaneous information relating to polar bears in Greenland. Pages 44–57 *in* Polar Bears: Proceedings of the 8th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland.

Wenzel, G. 2008. Sometimes hunting can seem like business: Polar bear sport hunting in Nunavut. Canadian Circumpolar Institute Press, Edmonton, Alberta, Canada.

Wenzel, G. W. 2011. Polar bear management, sport hunting and Inuit subsistence at Clyde River, Nunavut. Marine Policy 35:457–465.

APPENDIX E. Results of binary logistic regressions for body condition scores (poor, fair-good) for polar bears in Kane Basin.

Sex-Age Class	**Independent Variables**	***B***	**Wald Statistic**	***P***
Adult male	Julian Day	-0.095	1.617	0.204
	Epoch (1990s-2010s)	0.230	0.082	0.774
Adult female (lone)	Julian Day	0.104	2.413	0.120
	Epoch (1990s-2010s)	0.775	1.152	0.283
Adult female (with COY)	Julian Day	0.251	9.210	**0.002**
	Epoch (1990s-2010s)	0.327	0.194	0.660
Adult female (with yearling)	Julian Day	0.082	0.617	0.432
	Epoch (1990s-2010s)	2.064	4.081	**0.043**
Subadult	Julian Day	0.111	0.741	0.389
	Epoch (1990s-2010s)	20.056	0.000	0.998

ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᖅ ᐊᖑᓇᓱᒃᑎᓕᕆᔨᒃᑯᓐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᑲᑎᔪᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖁᔭᐅᔪᓂᒃ

ᕕᕗᐊᕆ 3-5, 2018

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ, ᓄᓇᕗᑦ ᒐᕙᒪᖓ, ᐃᖃᓗᐃᑦ, ᓄᓇᕗᑦ

ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓇᐃᓈᕐᓯᒪᔪᑦ

ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓄᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᑭᒡᒐᖅᑐᐃᔨᖏᑦ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᓚᐅᕐᒪᑕ ᐊᖑᓇᓱᒃᑎᓕᕆᔨᒃᑯᓐᓂᑦ (HTOs) ᕕᕗᐊᕆ 3-ᒥ 5-ᒧ, 2018 ᐊᐃᑉᐸᕆᓕᕐᖢᓂᐅᒃ ᐅᖃᓪᓚᑲᑕᒃᓂᖏᓐᓂᒃ. ᐱᒋᐊᓖᓴᕈᑎᓂᒃ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᖅᑕᖃᓚᐅᕐᑐᖅ ᔭᓄᐊᕆ 9-ᒥ 15-ᒧ, 2017 ᐱᑕᖃᖅᑎᑦᑎᒐᓱᒃᖢᑎᒃ ᐊᐅᓚᑦᑎᓂᒃᑯᑦ ᐱᓕᕆᖅᑎᒌᒃᓂᕐᒥᑦ ᐱᔾᔪᑎᖃᓗᐊᖅᖢᑎᒃ ᑭᖑᓕᖅᐹᓂᒃ ᖃᐅᔨᓴᖅᑐᓕᕆᓂᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᓂᒃ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᒃ (BB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ, ᐊᒻᒪᓗ ᓄᐊᑦᑎᓇᓱᒃᖢᑎᒃ ᑕᒪᑐᒧᖓ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᑦ (TK) ᑲᑎᑦᑎᓇᓱᒃᖢᑎᒃ. ᐊᐃᑉᐸᖓᓂ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᖅ ᖃᐅᔨᓇᓱᒍᑎᐅᓚᐅᕐᑐᖅ ᐆᒥᖓ ᑲᑎᖢᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᔪᓂᒃ ᐅᑯᓇᙵ ᑲᓇᑕ-ᑲᓛᓪᓖᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᕿᒥᕐᕈᐊᕐᓂᖅᑕᖃᓚᐅᕐᑎᓪᓗᒍ ᓇᓐᓄᒃᑕᐅᔪᓄᑦ ᖁᐊᕐᓵᕐᓇᕈᑎᐅᓇᔭᖅᑐᓂ ᖃᐅᔨᓴᕈᑎᓂᒃ ᐱᐊᓂᒃᑕᐅᓂᑯ ᐅᑯᓇᙵ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᓖᑦ ᐱᓕᕆᖅᑎᒌᒃᑐᓂᑦ (SWG). ᐊᖑᓇᓱᒃᑐᓕᕆᔨᑐᐊᑦ ᓄᓇᓕᒃᓂ ᐊᖑᓇᓱᒃᐸᒃᑐᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᓇᓄᕐᓂ ᖃᐅᔨᓇᓱᒡᕕᐅᓚᐅᕐᑐᑦ.

ᑐᓴᖅᑕᐅᕙᓪᓕᐊᔪᑦ ᐊᒻᒪ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏ (TK) ᓄᐊᑕᐅᕙᓪᓕᐊᔪᑦ ᓯᕗᓪᓕᕐᒥ ᖃᐅᔨᓇᓱᖃᑦᑕᖅᐸᓪᓕᐊᓕᖅᑎᓪᓗᒋ ᐃᓱᒪᒋᔭᐅᓚᐅᕐᑐᑦ ᑲᓇᑕ-ᑲᓛᓪᓖᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᐅᑯᓂᖓ ᑲᑎᖢᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐋᕿᒃᓱᐃᕙᓪᓕᐊᑎᓪᓗᒋ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᓇᓄᕐᓂ. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑐᓂᓯᓇᔭᖅᑐᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᓂᒃ ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔩᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ (NWMB) ᐃᓱᒪᓕᐊᕆᔭᐅᓂᐊᕐᒪᑦ.

ᐅᓇ ᐅᓂᒃᑳᖅ ᓈᐃᓈᕐᓯᒪᔪᒃᑯᑦ ᐅᖃᐅᓯᐅᕙᓪᓕᐊᔪᓂᒃ ᑎᑎᕋᐅᓯᖃᕋᓱᒃᑐᖅ ᐃᓚᐅᓂᑯᓂᒃ ᐊᐃᑉᐸᖓᓂᒃ ᖃᐅᔨᓇᓱᒃᓂᕐᒥᑦ ᑲᑎᒪᑎᑦᑎᓂᒃᑯᑦ.

ᐅᖃᐅᓯᖃᕆᐊᓕᕈᑎᖓ

ᐅᓇ ᐅᓂᒃᑳᖅ ᓴᕿᑎᑦᑎᓯᒪᔪᖅ ᐊᑭᑎᓕᕆᔾᒃᑯᑦ ᐱᔪᓐᓇᕐᓂᕆᑦᑎᒍᑦ ᑕᒻᒪᕐᓯᒪᙱᑦᑐᓂᒃ ᑐᓴᖅᐸᓪᓕᐊᓯᒪᔭᕆᓐᓂ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᒃᑯᑦ ᑲᑎᒪᑎᓪᓗᑎᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔾᒃᑯᓐᓂᑦ ᒥᑦᑎᒪᑕᓕᒃᒥᑦ, ᑲᖏᖅᑐᒫᐱᒃᒥᑦ ᐊᒻᒪ ᕿᑭᖅᑕᖅᔪᐊᖅᒥᑦ.

ᐃᓱᒪᕆᔭᐅᓯᒪᔪᑦ ᐅᑭᓂ ᐊᑭᑎᓕᕆᔾᒃᑯᓐᓂᖓᕐᓯᒪᙱᑦᑐᑦ, ᐅᑯᓘᓐᓃᑦ ᓄᓇᕗᑦ ᒐᕙᒪᖓᓄᑦ.

ᕿᓂᕐᕕᒃ

1.0 ᐅᓂᒃᑳᓄᑦ ᐱᔾᔪᑎᐅᔪᑦ ᐊᒻᒪ ᐋᕿᒃᓯᒪᓂᖅ

ᐅᓇ ᐅᓂᒃᑳᖅ ᐅᖃᐅᓯᒃᓴᓂ ᑲᑎᕐᓱᐃᒍᑎᐅᔪᖅ ᐊᒻᒪ ᓈᐃᓈᕐᓯᒪᔪᒃᑯᑦ ᐅᖃᐅᓯᕐᓂᒃ, ᐊᐱᕐᑯᑎᓂᒃ, ᐃᓱᒫᓘᑎᓂᒃ ᐊᒻᒪ ᐃᓱᒪᒋᔭᐅᕐᑯᔭᐅᔪᓂᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓄᑦ ᑭᐅᓯᒪᓂᕐᒧᑦ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᒋᔭᐅᔪᑦ ᓴᓐᓂᕈᑎᐅ ᐃᒪᖓᓂ (BB) ᓇᓄᕐᓂᒃ. ᐱᒋᐊᕈᑎᐅᓂᑯᑦ ᖃᐅᔨᓇᓱᒃᐸᓪᓕᐊᓂᕐᒥ ᐱᓕᕆᓂᕐᒥᑦ ᓄᓇᓕᒃᓂᑦ ᓄᐊᑦᑎᓇᓱᒃᓗᑎᒃ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᒃ (TK) ᓯᕗᓂᐊᒍᑦ ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᔪᓂᑦ ᐆᒧᖓ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓄᑦ (BB). ᐅᑯᐊ ᓄᓇᓖᑦ ᐅᖃᓪᓚᖃᑎᒋᔭᐅᖃᑦᑕᓚᐅᕐᑐᑦ ᕕᕗᐊᕆ 3-ᒥ 5-ᒧ, 2018:

- ᒥᑦᑎᒪᑕᓕᒃ, ᕕᕗᐊᕆ 3, 2018
- ᑲᖏᖅᑐᒑᐱᒃ, ᕕᕗᐊᕆ 4, 2018
- ᕿᑭᖅᑕᕐᔪᐊᖅ, ᕕᕗᐊᕆ 5, 2018

ᐱᒋᐊᕈᑎᐅᓂᑯᓂᒃ ᖃᐅᔨᓇᓱᒃᐸᓪᓕᐊᓚᐅᕐᑎᓪᓗᒋᑦ, ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᑦ ᐅᖃᐅᓯᐅᓚᐅᕐᑐᑦ ᐊᒻᒪ ᐃᓱᒪᓕᐊᕆᔾᔪᑎᐅᓪᓗᓂ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᒪᑎᓪᓗᒋ ᐅᒃᑑᐱᕆ 2017-ᒥᑦ. ᐊᐃᑉᐸᖓᓂ ᖃᐅᔨᓇᓱᒃᐸᓪᓕᐊᓂᕐᒥ ᐱᓕᕆᐊᖃᓚᐅᕐᑐᑦ ᑖᒃᑯᓇᓂᑦᑕᐃᓐᓇᖅ ᓄᓇᓕᒃᓂᑦ ᐅᖃᐅᓯᖃᖅᓗᑎᒃ ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᓂᒃ ᐃᑲᔪᕈᑎᐅᓂᐊᖅᑐᓂᒃ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᒃ ᑐᓂᓯᑎᓪᓗᒋᑦ ᓄᓇᕗᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ ᐃᓱᒪᓕᐊᖏᓐᓂᒃ.

ᑭᒡᒐᖅᑐᐃᔪᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᑦ, ᓄᓇᕗᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓂᑦ, ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓂᑦ (NTI), ᐊᒻᒪᓗ ᕿᑭᖅᑖᓗᒃᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂᑦ (QWB) ᐅᐸᒍᑎᓚᐅᕐᑐᑦ ᑕᒪᐃᓐᓄᑦ ᖃᐅᔨᓇᓱᒃᐸᓪᓕᐊᓂᕐᒧᑦ.

2.0 ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᕐᒧᑦ ᐱᔾᔪᑎᒋᔭᐅᔪᑦ

ᓯᕗᓪᓕᕐᒥ ᐅᖃᓪᓚᖃᑎᖃᕐᓂᒃᑯᑦ ᐅᖃᐅᓯᖃᓚᐅᕐᑐᑦ ᓄᑖᖑᓖᓂᒃ ᑐᓴᒐᒃᓴᓂᒃ ᐆᒧᖓ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᐅᓂᒃᑲᐅᓯᐅᓂᖏᑦᑎᒍᑦ ᖃᐅᔨᓴᖅᑐᓕᕆᔨᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᖏᓐᓂᒃ ᐋᕿᒃᑕᐅᓯᒪᔪᖅ ᐅᑯᓇᙵᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᓖᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᓂᑦ (SWG). ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᓖᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᑦ (SWG) ᐱᒋᐊᖅᑕᐅᓚᐅᕐᓯᒪᔪᑦ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᓄᐊᑦᑎᓂᐊᕐᓗᑎᒃ ᖃᐅᔨᓴᕐᓂᒃᑯᑦ ᓄᐊᑕᐅᕙᓪᓕᐊᔪᓂᒃ ᐊᒻᒪ ᓄᑖᖑᕆᐊᖅᑕᐅᓗᓂ ᖃᓄᖅ ᐱᓕᕆᖃᑎᒌᒃᑐᓂᕐᒥᒃ ᓇᓄᕐᓂ ᐅᑯᑎᒎᓇᖅ ᑲᓇᑕ/ᓄᓇᕗᑦ ᐊᒻᒪ ᑲᓛᓪᓛᑦ ᓄᓇᖓᓂ. ᓯᕗᓪᓕᕐᒥᑦ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᓚᐅᖅᑎᓪᓗᒋᑦ ᐊᒻᒪᓗ ᐱᐊᓂᒃᓯᑎᓪᓗᒋᑦ ᓇᓄᒃᑕᐅᔪᑦ ᖁᐊᕐᓵᕐᓇᖅᑐᒦᓐᓇᓴᕆᐊᒃᓴᖏᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᕐᒥᑦ ᐅᑯᓄᖓ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᓖᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᓂᑦ (SWG), ᐅᑯᐊ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᒪᓚᐅᕐᑐᑦ ᐆᒃᑑᐱᕆ 2017-ᒥ ᐃᓱᒪᓕᐅᕆᒐᓱᒃᓗᑎᒃ ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ (TAH). ᐱᔾᔪᑎᒋᔭᐅᔪᑦᑕ ᐊᐃᑉᐸᖓ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᕐᒧᑦ ᓄᓇᓕᒃᓂ ᐅᑯᐊ ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐃᓱᒪᓕᐊᕆᓯᒪᔭᖏᓐᓂᒃ ᐅᑯᓇᙵ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᐆᒧᖓ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᓇᓄᕐᓂ.

2.1 ᑲᑎᒪᓂᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ

ᑲᑎᒪᖃᑦᑕᓚᐅᕐᑐᑦ ᐅᓐᓄᒃᑯᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᓪᓗᒃᑯᑦ ᐊᒻᒪ ᐃᑲᕐᕋᓂ 3-ᓄ 4-ᓄ ᑲᑎᒪᕙᒃᓗᑎᒃ ᒪᓕᒃᓗᑎᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᖃᓄᐃᑦᑕᕆᐊᖃᕐᓂᖏᓐᓄᑦ. ᑲᑎᒪᓃᑦ ᐋᕿᒃᑕᐅᓯᒪᓚᐅᕐᑐᑦ ᐊᒻᒪ ᑐᑭᒧᐊᖅᑕᐅᓪᓗᓂ ᐱᓕᕆᓕᒪᓂᕐᓴᒥᒃ ᐊᐅᓚᑦᑎᔨᒥ ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᐱᓕᕆᐊᓂᒃ, ᔦᐃᓴᓐ ᐊᓕᖅᑐᖅᑐᖅ, ᐊᒻᒪ ᐱᓕᕆᓕᒪᓂᕐᓴᖅ ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᐅᖃᐅᔾᔭᐃᔭ, ᑭᐅᓇᓐ ᓯᒥᑦ. ᐊᑐᓂ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᐅᔪᑦ ᐱᒋᐊᕈᑎᖃᓚᐅᕐᐳᑦ ᓴᕿᑎᑦᑎᓂᕐᒥᒃ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓂᒃ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᔪᓂᒃ (ᓇᓗᓇᐃᒃᑯᑎ 1). ᑲᑎᒪᖅᑕᐅᔪᑦ ᑐᙵᓱᒃᑎᑕᐅᓚᐅᕐᑐᑦ ᐊᐱᕐᓯᕈᓐᓇᕐᓗᑎᒃ, ᐃᓱᒫᓘᑎᖏᓐᓂᒃ ᐅᖃᐅᓯᖃᕐᓗᑎᒃ, ᐅᕝᕙᓘᓐᓃᑦ ᐊᓯᐊᒍᖕᒎᖅ ᐊᑐᖅᑕᐅᔪᓐᓇᕐᓴᖕᒎᖅᑐᓂᒃ ᑲᑎᒪᑎᓪᓗᒋ. ᓴᕿᑎᑦᑎᑕᐅᖅᑎᓪᓗᒋᑦ, ᐊᐱᕐᑯᑎᑦ/ᐅᖃᐅᓯᒃᓴᐃᓪᓗ ᑲᔪᓯᑕᐅᕐᑐᖅ ᑭᓯᐊᓂ ᐊᐱᕐᑯᑎᒃᓴᖃᕈᓐᓃᕐᒪᑕ.

3.0 ᓇᐃᓈᕐᓯᒪᔪᑦ ᓄᓇᓕᒃᑎᒍᑦ

ᖃᓄᐃᓕᐅᕈᑎᐅᔪᒪᔪᑦ ᑲᑎᒪᑲᑕᒃᓂᕐᒥ ᓇᓗᓇᐃᖅᑕᐅᑎᐊᓚᐅᕐᑐᑦ ᐊᕝᒍᓇᓱᒃᑐᓕᕆᔾᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᓯᕗᓂᐊᒍᑦ ᐊᒻᒪᓗ ᑲᑎᒪᓂᖅ ᐱᒋᐊᓕᖋᖔ. ᐊᒥᓲᓚᐅᕐᑐᑦ ᐊᔾᔨᒌᒃᑐᑦ ᐊᐱᕐᑯᑎᐅᔪᑦ, ᐃᓱᒫᓘᑎᐅᔪᑦ ᐊᒻᒪ ᐃᓱᒪᒋᔭᐅᓯᒪᔪᑦ ᐊᕝᒍᓇᓱᒃᑐᓕᕆᔾᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂᑦ ᑕᒪᐃᓐᓂ ᓄᓇᓕᒃᓂ ᐅᖃᓪᓚᖃᑎᒋᔭᐅᑲᑕᒃᑐᓂᑦ ᑲᑎᒪᓂᒃᑯ. ᐅᓂᒃᑳ ᐊᑕᖏᕐᓯᒪᑦᑎᐊᖅᑐ ᐊᐱᕐᑯᑎᓂᒃ ᐊᒻᒪ ᐅᖃᐅᓯᒃᓴᓂ ᑕᒪᐃᓐᓂ ᓄᓇᓕᖏᓂ ᑕᑯᔭᐅᔪᓐᓇᖅᑐᑦ ᓇᓗᓇᐃᒃᑯᑎ 4-ᒥᑦ. ᐅᑯᐊ ᐊᐅᓯᐃᑦᑐᕐᒥᐅᑦ ᓄᓇᓕᑐᐊᕝᒍᓂᕐᕆᓐᓄᑦ ᐊᕝᒍᓇᓯᖅᑳᖅᑐᓂᒃ ᐱᕐᓕᕋᕐᕕᔅᔪᐊᑉ ᑕᕆᐅᖕᒍᓂ (KB) ᐃᓚᖏᓐᓂᒃ ᐃᓱᒪᒋᔭᐅᔪᓂ ᐊᓯᖃᖃᑦᑕᓚᐅᕐᑐᑦ ᐅᑯᓇᙵᑦ ᐱᖓᓱᓂᒃᑕᐅᖅ ᓄᓇᓕᒃᓂᑦ.

3.1 ᒥᑎᒪᑕᓕᒃ

<u>**ᐅᓪᓗᖓ:**</u> ᔭᓄᐊᕆ 3, 2018

<u>**ᑮᓇᖅᑐᐃᔨᐅᔪᑦ:**</u>

- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒫᓂᕐᔅᖅ ᐊᐅᓚᑦᑎᔨ ᐊᐅᓚᑕᐅᔪᓄᑦ: ᔭᐃᔅᓐ ᐊᓕᖃᖅᑐᖅ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒫᓂᕐᔅᖅ ᓂᕐᔪᑎᓄᑦ ᐅᖃᐅᔾᔪᐃᔨ: ᑭᐅᕆᓐ ᓯᒥᑦ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᖁᑭᖅᑖᓗ ᐅᐊᖕᓇᖕᒐᓄᑦ ᐊᐅᓚᑦᑎᔨ: ᓯᑳᑦ ᔫᓐᓴᓐ
- ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ, ᐊᕙᑎᓕᕆᔨ ᓯᐅᕈᓪ ᕗᕇ
- ᓄᓇᕗᑦᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑕᓂᔅ ᓇᑎᓗ
- ᓄᓇᕗᑦᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑲᐃᐅᓪ ᐅᕆᑦᓯ
- ᖁᑭᖅᑖᓗᒃᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ, ᑲᑎᒪᔨ: ᐋᑕᒥ ᓄᓇ
- ᒥᑎᒪᑕᓕᒃ ᐊᕝᒍᓇᓱᒃᑐᓕᕆᔾᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ
    - ᔭᐃᑯᓛᓯ ᕿᓕᑎ
    - ᑲᐃᓚ ᓴᕝᒍᔭ
    - ᓃᓇ ᑲᐅᑕᖅ
    - ᐱᓕ ᒥᑯᓵᖅ
    - ᐸᓄᐃᓕ ᐃᓄᑯᓗ
    - ᑕᐃᓇ ᐃᓄᑯᓗ
    - ᐃᓚᐃᔭ ᐸᓂᒃᐸᑯᑦᑐᒃ

<u>**ᐅᖃᐅᓯᒃᓴᑦ ᐊᒻᒪ ᐊᐱᕐᑯᑎᒃᓴᑦ:**</u>

ᐊᕝᒍᓇᓱᒃᑐᓕᕆᔾᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐃᓱᒫᓘᑎᖏᓐᓂᒃ ᓴᖁᑎᑦᑎᓚᐅᕐᑐᑦ ᓇᓄᐃᑦ ᐅᓄᕐᓯᕚᓪᓕᕐᓂᖏᓐᓂᒃ ᑲᒪᒋᔭᕆᐊᖅᑲᑦᑕᖅᑕᖏᓐ ᓄᓇᓕᒃᓂ ᖃᓂᒋᔭᕐᒐᓂᒡᓗ. ᓄᓇᓕᒃᓂᑦ ᓇᓄᒐᓱᖅᑲᑦᑕᖅᑐᑦ ᐱᕕᖃᓗᐊᖅᐸᙱᓐᒪᑕ ᐊᕝᒍᓇᓯᕆᐊᒃᓴᖅ ᐱᔾᔪᑎᕆᓪᓗᒍ ᓇᓄᐃᑦ ᐱᔭᐅᓂᕐᓴᐅᖅᑲᑦᑕᓗᐊᕐᒪᑕ ᐸᒃᕕᒋᔭᐅᓕᖅᑲᕐᒥᒃ. ᓇᓄᐃᑦ ᓄᓇᓕᒃᓄᐊᖅᑲᑦᑕᖅᑐᑦ ᐊᒻᒪ ᖃᓂᒋᔭᕐᒐᓄᑦ ᓵᓪᓗᐊᓘᖅᑲᑦᑕᖅᑐᑦ ᐊᒻᒪᓗ ᓂᕆᔭᒃᓴᐅᙵᕆᕗᑎᒃ ᐃᓖᓐᓂᒃᑯᑦ. ᑕᒪᐅᕐᓘᕐᒐᔪᒥᑦ ᐃᓱᒫᓘᑎᒥ ᓴᖁᑎᓚᐅᕐᒥᔪᑦ ᐊᕝᒍᓇᓱᒃᑐᓕᖁᔾᒃᑯ ᐃᒪᓐᓇ ᐃᓱᒪᓪᓗᑎᒃ ᓴᐳᔾᔨᓯᒪᓂᕐᒥᒃ ᐃᓅᓯᕐᒥᑦ ᐊᒻᒪ ᐱᕐᑯᑎᓂᒃ ᑐᕐᑯᑕᐅᔪᑦ (DLPKs) ᓂᐱᙵᑕᖅᑲᓇᐊᖅᑲᙱᓐᓂᙱᓐᓂᒃ. ᑲᐃᓚ ᓴᕝᒍᔭ ᐊᐱᕆᔪᖅ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᖃᐅᔨᓴᕆᐊᒃᓴᖏᓐᓂᒃ ᓇᓄᐃᑦ ᐱᓇᓱᐊᖅᐸᒃᑕᖏᓐᓂᒃ ᐆᒪᔪᓂᒃ. ᓄᓇᓕᒃᓃᑦᑐ ᐅᔾᔨᕆᓕᕐᒪᑕ ᓇᑏᑦ ᐊᒥᓲᖑᓃᖅᐸᓪᓕᐊᓂᖏᓐᓂᒃ. ᐃᓚᐃᔭ ᐸᓂᒃᐸᑯᑦᑐᒃ ᑐᑭᓯᒋᐊᖅᑎᑦᑎᔪᖅ ᓇᓄᐃᑦ ᓱᕋᐃᕙᓪᓕᐊᓂᖏᓐᓂᒃ ᐱᕐᑯᑎᒋᔭᐅᔪᓂᒃ ᐊᒻᒪ ᖁᖓᓂᕆᔭᐅᔪᓂᒃ. ᓂᕿᓄᑦ ᖁᖕᓃᑦ ᐱᒻᒪᕆᐊᓘᒃᒪᑕ ᐃᓄᒃᓄᑦ ᓂᕿᒃᓴᐅᓂᖏᓐᓄᑦ ᓄᓇᓕᒃᓂᑦ

ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᑦ ᑲᑎᒪᔩᖏ ᐊᖏᓚᐅᖅᑐᑦ ᑎᑎᖅᑲᓂᒃ ᐃᑲᔪᕈᑎᒃᓴᓂᒃ ᓇᒃᓯᐅᔾᔨᓂᐊᕐᒧᑎᒃ ᐅᑯᓄᖓ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᑦ ᑲᑎᒪᔩᖏᓐᓄᑦ ᕕᕗᐊᕆ 23 ᑐᖓᓂ ᑖᓐᓇ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᒃᑯᑦ ᑲᑎᒪᔩᖏᑦᑕ ᐊᕙᑎᓕᕆᔨᖓ ᐃᓚᐅᑎᑦᑎᔪᓐᓇᕐᓂᐊᕐᒪᑦ ᒫᔾ 2018-ᒥ ᑲᑎᒪᔾᔪᑎᒃᓴᓄᑦ. ᑲᑎᒪᔩ ᓇᓗᓇᐃᓚᐅᕐᒥᔪᑦ ᑎᑎᖅᑲᓂ ᐊᑐᕋᔭᕐᓂᖏᓐᓂᒃ ᐃᓱᒫᓘᑎᖏᓐᓄᑦ ᐱᔾᔪᑎᖃᖅᑐᒥᒃ ᓴᐳᔾᔨᓯᒪᓂᕐᒥᑦ ᐃᓅᓯᕐᒥᒃ ᐊᒻᒪ ᐱᖁᑎᒋᔭᐅᔪᓄᑦ ᑐᖁᑕᐅᖃᑦᑕᖅᑐᓂᒃ. ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᑦ ᐃᓱᒪᒃᒪᑕ ᐊᕕᒃᑐᖅᓯᒪᓂᒃᑯᑦ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᑦ ᖃᐅᔨᓴᕈᓐᓇᓪᓗᐊᕐᓂᖏᓐᓂᒃ ᓂᕕᙵᑕᐃᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᕐᓂᖏᓐᓂᒃ ᐊᑕᐅᔫᖅᑕᐅᓇᓂᒃ ᐊᕐᕌᒍᒥᑦ ᑲᑎᒪᕕᒡᔪᐊᕐᓇᒥ. ᐅᓄᕐᓯᕙᓪᓕᐊᒍᑎᒃ ᓈᒻᒪᙱᓕᐅᕈᑎᐅᕙᒃᑐᑦ ᓇᓄᐃᑦ ᑕᐃᒪᓕ ᐅᓄᕈᓐᓃᖅᑎᑦᑎᕚᓪᓕᕈᓐᓇᖅᑐᑦ ᓇᓐᓄᒐᓱᑐᐃᓐᓇᖅᑐᓂ ᐊᑭᓖᓂᒃᑯᑦ ᑕᒪᒃᑯᐊ ᓂᕕᙵᑕᐃᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᕐᓂᐊᕐᒪᑕ ᑐᖁᑕᐅᑐᐃᓐᓇᖅᑐᓄᑦ ᓇᓄᕐᓄᑦ ᐸᕝᕕᒋᔭᐅᓂᖏᓐᓄᑦ.

3.2 ᑲᖏᖅᑐᒑᐱᒃᒥ ᑲᑎᒪᑎᑦᑎᓂᕐᒧᑦ ᓇᐃᓈᖅᓯᒪᔪᑦ ᐅᓂᒃᑳᑦ

ᑲᑎᒪᓂᐅᑉ ᐅᓪᓗᖏᑦ: ᕕᕗᐊᕆ 4, 2018

ᑭᒡᒐᖅᑐᐃᔨᐅᔪᑦ:

- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒪᓂᖅᓴᖅ ᐊᐅᓚᑦᑎᔨ ᐊᐅᓚᑕᐅᔪᓄᑦ: ᔭᐃᓴᓐ ᐊᓕᖅᑲᖅᑐᖅᑐᖅ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒪᓂᖅᓴᖅ ᓂᕐᔪᑎᓄᑦ ᐅᖃᐅᔾᔪᐃᔨ: ᑭᐅᓇᓐ ᓯᒥᑦ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᕿᑭᖅᑖᓗ ᐅᐊᖕᓇᖕᒐᓄᑦ ᐊᐅᓚᑦᑎᔨ: ᓯᑳᑦ ᔮᓐᓴᓐ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᑦ, ᐆᒪᔪᓕᕆᔨ II: ᐳᕉᔅ ᔨᐅᓇ (ᐱᔭᐃ) Hᐊᐃᓐᓄ
- ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ, ᐊᕙᑎᓕᕆᔨ ᓯᐅᕈᓪ ᕗᓃ
- ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑕᓂᔅ ᓇᑎᓗ
- ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑲᐃᐅᑉ ᐅᓇᕐᔾ
- ᕿᑭᖅᑖᓗᒃᒥᑦ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ, ᑲᑎᒪᔨ: ᐋᑕᒥ ᓄᓇ
- ᓇᒻᒪᐅᑕᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ
  - ᒨᓯ ᐸᓖᑦᑐᖅ
  - ᒍᐊᑕᓐ ᑲᐅᑕᖅ
  - ᓇᐃᓴᓇ ᕿᓪᓚᖅ
  - ᔭᐊᕙ ᐃᑐᐊᖓᑦ
  - ᔭᐃᓯ ᑎᒍᓪᓚᒐᖅ
  - ᓕᕙᐃ ᐸᓖᑦᑐᖅ

ᐅᖃᐅᓯᒃᓴᑦ ᐊᒻᒪ ᐊᐱᖁᑎᒃᓴᑦ:

ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᓐᓄᑦ ᐃᓚᒋᔭᐅᔪᑦ ᐃᓱᒫᓘᑎᖃᓚᐅᖅᑐᑦ ᓱᓕ ᐊᑐᖅᓯᒪᓂᑯᓄᑦ ᐊᑭᓖᒥᐊᖃᖅᓴᖅᐸᑦ ᑖᓐᓇ ᑲᑎᖑᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᓄᑖᖅ ᐋᖅᑭᒃᑕᐅᓚᐅᖅᑎᓪᓗᒍ. ᑐᓴᖅᑎᑕᐅᓚᐅᖅᑐ ᖃᓄᓕᒫᖅ ᐊᑭᓕᕆᐊᖃᖅᑕᖏᑦ ᔨᑭᙳᓛᕐᓂᖏᓐᓂᒃ ᑖᓐᓇ ᓄᑖᖅ ᑲᑎᖑᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐊᑐᓕᖅᑎᑕᐅᒃᐸᑦ. ᐊᖑᓇᓱᒃᑐᓕᕆᔩᒃᑯᑦ ᓈᒻᒪᒃᓴᙱᑦᓚᕆᒃᑐᑦ ᐊᓯᙳᕈᒎᔭᖃᑦᑕᖅᑐᒥᑦ ᓇᓐᓄᒐᒃᓴᓕᕆᓂᕐᒥ. ᐃᓱᒪᒃᒪᑕ ᑕᒪᓐᓇ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓄᑦ ᐃᑲᔫᑎᐅᓂᐊᕐᓗᓂ ᑭᓯᐊᓂᓕ ᓄᓇᓕᒃᓄᑦ ᐃᑲᔫᑕᐅᙱᓪᓗᓂ. ᑐᑭᓯᓯᒪᓪᓚᕆᙱᑦᑐᑦ ᖃᓄᖅ ᐱᔪᓐᓇᕋᔭᕆᐊᒃᓴᖏᓐᓂᒃ ᐊᑭᓕᐅᑎᒋᔭᕆᐊᖃᖅᑕᓂ ᑲᑎᑉᐸᓪᓕᐊᔪᑦ ᐊᒻᒪᓗ ᐱᔪᓐᓇᕐᓂᖃᓂᒃ ᐊᓯᐅᔨᓂᕐᒥᒃ ᐊᒥᓱᓗᐊᓂᒃ ᓇᓐᓄᒃᓯᒪᓂᕐᒧᑦ ᐋᓐᓂᕈᑎᐅᒃᒪᑦ ᓄᓇᓕᒃᓄᑦ. ᐅᖃᖅᑐᖃᓚᐅᖅᑐᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯ ᑎᑎᕋᕈᓐᓇᕐᓂᖏᓐᓂᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔩᑦᔪᐊᑦ ᑲᑎᒪᔩᖏᓐᓄᑦ ᐱᔪᓐᓇᖅᓯᔪᔪᒪᓂᕐᒥᑦ ᐊᔪᙱᔾᔪᑎᐅᔪᓐᓇᖅᑐᓂᒃ ᐊᖅᑮᒎᓕᖅᑐᓂᒃ.

ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᒻᒪ ᐱᐅᒃᓴᖏᓪᓚᕆᒃᑐᑦ ᓂᕕᖓᑕᓂ ᐊᑐᖅᑲᑦᑕᕆᐊᖅᑲᖅᓂᖏᓐᓂᒃ ᐸᕝᕕᓇᖅᑐᓂ ᓇᓄᕐᓂ ᑐᖁᑕᐅᔪᓄ ᐅᕙᖔᖅᑐᒥ ᑲᑎᑦᑐᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH). ᐃᓱᒪᒃᒪᑕ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓄᑦ ᐃᖅᑲᓇᐃᔭᖅᑎ ᓇᓄᕐᒥ ᑐᖂᑦᑎᔭᕆᐊᖃᖅᐸᑦ, ᓄᓇᓖᑦ ᑖᔅᓱᒧᖓ ᓂᕕᖓᑦᑕᖅᑲᓐᓂᕈᑎᖃᕆᐊᖃᖏᓐᓂᖏᓐᓂᒃ. ᓂᕈᐊᕐᓂᖅᓴᐅᒃᒪᑕᓕ ᐊᓯᐊᒎᖓᔪᒃᑯᑦ ᓂᕕᖓᑕᖅᑲᑕᖅᑲᖅᓗᓂ ᓇᓄᕐᓄᑦ ᐸᕝᕕᓴᕆᑕᖅᑲᑕᖅᑐᓄᑦ. ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓄᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᐅᔪᑦ ᐅᖃᓚᐅᖅᑐᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᓱᒪᒋᔭᖏᓐᓂᒃ ᐅᖃᐅᓯᖃᕆᐊᖃᖅᑭᓂᖏᓐᓂᒃ ᐆᒥᖓ ᐊᔾᔨᒌᖃᑦᑕᖏᓗᐊᖅᑐᖅ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓕᕆᓂᕐᒥᒃ ᐊᒻᒪᓗ ᐸᕝᕙᓇᖅᑐᓂᒃ ᓇᓄᕐᓂ ᑐᖂᑦᑎᖅᑲᑦᑕᕐᓂᕐᒥᒃ ᑎᑎᖅᑲᑎᒍᑦ ᓄᓇᕗᒥ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᒻᒪᓗ ᐃᓱᒫᓗᑎᖏᓐᓂᒃ ᐅᖃᐅᓯᖅᑲᖅᓗᑎᒃ ᐅᐸᒍᑎᓯᒪᓂᒃᑯᑦ ᑭᒃᑯᓕᒫᓂᒃ ᑲᑎᒪᑎᑦᑎᓕᕈᑎᒃ ᐋᖅᑭᒃᓱᖅᑕᐅᕙᓪᓕᐊᔪᓂᒃ ᓇᓄᕐᓂ ᐊᐅᓚᑦᑎᓂᖅᑯᑦ ᐸᖅᓇᐅᑎᓂᒃ. ᐱᔪᒪᔪᖅᑲᓚᐅᖅᑐᖅ ᐃᓚᖓᓐᓂᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖓᓂ ᑲᑎᖅᑲᑎᕐᕈᒪᓂᖅᒥᑦ ᐅᐸᖅᑲᑎᒌᒃᓯᒪᓂᒃᑯᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓂᑦ ᐅᖅᑲᐅᓯᖅᑲᖅᓗᑎᒃ ᐋᖅᑭᒃᓱᖅᑕᐅᕙᓪᓕᐊᓂᐅᔪᖅ ᓇᓄᕐᓂ ᐊᐅᓚᑦᑎᓂᖅᑯᑦ ᐸᖅᓇᐅᑎᒃᓴᓂᒃ.

ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᖏᖅᑐᒑᐱᒃᒥ ᑲᑎᖅᑲᑎᖅᑯᔨᔪᑦ ᑖᒃᑯᓂᖓ ᐱᖓᓱᓂᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓂᒃ ᐊᖑᓇᓱᒃᐸᒃᑐᓂᒃ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᑕ ᓇᓄᖏᓐᓂᒃ ᐅᖅᑲᐅᓯᖅᑲᖅᓗᑎᒃ ᐃᓱᒫᓗᑎᑦᓯᖏᓐᓂᑦ ᐊᔾᔨᒌᖅᑲᑦᑕᖏᓗᐊᖅᑐᖅ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓕᕆᓂᕐᒥᒃ ᐊᒻᒪᓗ ᐃᓱᒫᓗᑎᐅᕙᒃᑐᓂᒃ ᓇᓄᕐᓂᒃ ᐸᕝᕕᑦᓴᐅᓕᖅᑐᑦ ᑐᖁᑕᐅᖅᑲᑦᑕᕐᓂᖏᓐᓂᒃ. ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖏ ᐃᓱᒪᓕᐅᕆᓚᐅᖅᑐᑦ ᐱᐅᓇᔭᕐᓂᖓᓂᒃ ᓯᕗᓂᐊᒍᑦ ᑭᒃᑯᓕᒫᓂᒃ ᑲᑎᒪᑎᑦᑎᓚᐅᖅᑎᓐᓇᓇᒥᑦ ᐱᒋᐊᕈᑎᓄᑦ ᓇᓄᕐᓂ ᐊᐅᓚᑦᑎᓂᖅᑯᑦ ᐸᖅᓇᐅᑎᓂᒃ. ᐊᒥᓱᑦ ᑲᑎᒪᔨᐅᔪᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓄᑦ ᓇᑐᓇᐃᓚᐅᖅᑐᑦ ᐃᑲᔪᖅᑐᐃᓇᔭᕐᓂᖏᓐᓂᒃ ᐅᓄᖅᓯᒋᐊᖅᑎᑦᑎᓂᖅᒥᒃ ᑲᑎᑦᑐᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ (TAH). ᐊᐱᕆᔭᐅᓚᐅᖅᑐ ᑎᑎᖅᑲᓂᒃ ᓇᒃᓯᐅᔾᔨᖅᑯᔭᐅᓪᓗᑎᒃ ᖃᕋᓴᐅᔭᒃᑯᓪᓗᓐᓃᑦ ᑎᑎᕋᖅᓗᑎ ᕕᕗᐊᕆ 23, 2018 ᓯᕗᓂᐊᒍᑦ ᐃᓚᒋᔭᐅᑎᑕᐅᔪᓐᓇᖅᓂᐊᖅᒪᑦ ᑭᖑᓪᓕᕐᒥ ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᑲᑎᒪᓕᖅᐸᑕ.

3.3 ᕿᑭᖅᑕᕐᔪᐊᕐᒥᑦ ᑲᑎᒪᑎᑦᑎᓂᕐᒧᑦ ᓇᐃᓈᖅᓯᒪᔪᑦ ᐅᓂᒃᑳᑦ

ᐅᓪᓗᖓ: ᕕᕗᐊᕆ 5, 2018

ᑭᒡᒐᖅᑐᐃᔨᐅᔪᑦ:

- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒪᓂᖅᓴᖅ ᐊᐅᓚᑦᑎᔨ ᐊᐅᓚᑕᐅᔪᓄᑦ: ᔭᐃᓴᓐ ᐊᓕᖅᑲᑐᖅᑐᖅ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒪᓂᖅᓴᖅ ᓂᖅᔪᑎᓄᑦ ᐅᖅᑲᐅᔾᔪᐃᔨ: ᑭᐅᓇ ᓯᒥᑦ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᕿᑭᖅᑖᓗ ᐅᐊᖕᓇᖓᓄᑦ ᐊᐅᓚᑦᑎᔨ: ᓯᑳᑦ ᔫᓐᓴᓐ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᑦ, ᐆᒪᔪᓕᕆᔨ II: ᔩᓴᕝ ᒧᐃ
- ᓄᓇᕗᑦ ᑐᖓᕕᒃᑯᑦ, ᐊᕙᑎᓕᕆᔨ ᓯᐅᕈᓪ ᕗᓖ
- ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑕᓂᔅ ᓇᑎᓗ
- ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑲᐃᐅᓪ ᐅᓇᕐᔪ
- ᕿᑭᖅᑖᓗᒃᒥᑦ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ, ᑲᑎᒪᔨ: ᐋᑕᒥ ᓄᓇ
- ᓇᑦᑎᕙᒃ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ
  - ᔭᐃᐱᑎ ᓄᕿᕈᐊᒃ
  - ᐋᓚᓴ ᑰᐸᓚᒃ
  - ᔭᐃᓕ ᑯᑯᐊᒃ
  - ᔭᐊᓯ ᐊᐅᑦᓚᑭᐊᒃ
  - ᓴᓯᐱ ᐊᐅᑦᓚᑭᐊᒃ
  - ᔭᕿᐃᐊ ᓄᕿᖓᒃ

ᐅᖃᐅᓯᒃᓴᑦ ᐊᒻᒪ ᐊᐱᖁᑎᒃᓴᑦ:

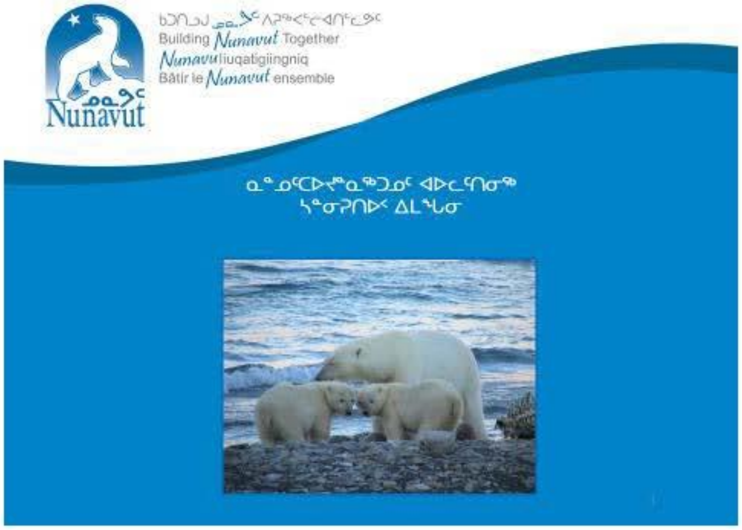
ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏ ᐃᓚᐅᔪᑦ ᐃᑲᔪᕐᓯᒪᓚᐅᕐᑐᑦ ᐃᓱᒪᓕᐊᕆᓯᒪᓂᕐᒥ ᐅᓂᕐᓯᕚᓪᓕᖅᑎᑦᑎᓂᕐᒥ ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ (TAH) ᐊᒻᒪ ᑎᑎᕋᕈᒪᓪᓗᑎᒃ ᐃᑲᔪᖅᑐᐃᒍᑎᒃᓴᒥᑦ ᐅᑯᓂᖓ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂᒃ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᖃᓚᐅᕐᒥᔪᑦ ᑭᒡᒐᖅᑐᐃᓂᖏᓐᓂᒃ ᕿᑭᖅᑕᖅᔪᐊᕐᒥᐅᓂᒃ ᐊᒻᒪᓗ ᐃᓱᒪᓕᐊᕆᔭᐅᔪᑦ ᓄᓇᓕᒃᓄᑦ ᑐᓴᖅᑕᐅᑎᒋᐊᖃᖅᓗᒋᑦ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᐱᕆᓚᐅᕐᑐᑦ ᑮᓇᐅᔭᓂᒃ ᐱᕕᒃᓴᖅᑕᖃᕐᒪᖔᑦ ᐊᖑᓇᓱᒃᑎᓄᑦ ᓇᓄᐃᑦ ᐊᒥᖏᓐᓂᒃ ᐱᓯᒪᔪᓂᒃ ᓂᐅᕐᕈᑕᐅᔪᓐᓇᖏᓐᓂᖏᓐᓄᑦ ᓲᖃᐃᒻᒪ ᐱᔭᐅᓚᐅᕐᓯᒪᒃᒪᑕ 2010-ᒧ 2013-ᒧ ᐊᑯᓐᓂᖓᓂ. ᐃᓱᒫᓘᑎᒥ ᐅᖃᖅᑐᖃᓚᐅᕐᑐᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖓᓂ ᐱᔾᔪᑎᖃᖅᑐᒥᑦ ᐊᖑᓴᓪᓗᐃᓪᓗ ᐊᕐᓇᓪᓗᐃᓪᓗ ᖃᓄᐃᓕᖓᓂᖏᓐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᑕ ᓇᓄᖏᓐᓂᒃ. ᐃᓱᒫᓘᑎᖃᕐᓂᖃᓚᐅᕐᒪᑕ ᐊᖑᓴᓪᓗᐃᑦ ᒥᒃᓵᓄᑦ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᑦᑎᒍ ᐃᓱᒪᓪᓗᑎᒃ ᐊᓯᒃᓴᖏᓐᓂᒃ ᐱᑕᖃᓗᐊᕈᓐᓃᖃᔭᕆᐊᒃᓴᖏᑦ ᐊᖑᓇᓱᓗᐊᖃᑦᑕᕈᑎᒃ ᐊᒥᓱᓗᐊᓂᒃ ᐊᖑᓴᓪᓗᓐᓂᒃ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏ ᖃᐅᔨᓇᓱᓚᐅᕐᒥᔪᑦ ᖃᑯᒍ ᑭᖑᓪᓕᕐᒥ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᖃᐅᔨᓵᓕᕐᒪᖔᑕ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᒃ (BB) ᐊᒻᒪᓗ ᖃᐅᔨᓴᖃᑦᑕᕐᓂᖏ ᓇᐃᓐᓂᕐᓴᐅᕙᒃᓗᑎᒃ. ᐃᓚᖏᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᖃᐅᓯᖃᓚᐅᕐᒥᔪᑦ ᐃᓱᒫᓘᑎᖏᓐᓂᑦ ᒪᑯᐊ ᖃᓄᐃᑕᐅᖃᑦᑕᕐᒪᖔᑕ ᓇᓄᐃᑦ ᑐᖁᑕᐅᖃᑦᑕᖅᑐᑦ ᐆᒪᔪᕐᓂᐊᖅᑎᒃᑯᓐᓂᑦ.

ᐃᓚᖏᑦ ᓄᓇᓖᑦ ᐊᒻᒪ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏ ᐅᔾᔨᕆᓯᒪᓕᕐᒪᑕ ᓴᓪᓘᓂᕐᓴᐅᓕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᓄᓇᓕᒃᓄᐊᖅᑐᑦ ᐅᕙᓘᓐᓃᑦ ᓄᓇᓖᑦ ᖃᓂᒋᔮᓄᐊᖃᑦᑕᖅᑐᑦ. ᐊᒥᓱᑦ ᐃᓱᒪᒃᒪᑕ ᓴᓪᓗᐊᓗᐃᑦ ᓇᓄᐃᑦ ᐃᓐᓇᐅᓂᕐᓴᐅᔭᕆᐊᒃᓴᖏᓐᓂᒃ ᐊᖑᓇᓱᒍᓐᓇᐃᓪᓕᔪᑦ ᐊᒻᒪ ᐃᓐᓇᐅᓂᖏᓐᓄᑐᐃᓐᓇᖅ.

ᐅᖃᐅᓯᐅᓚᐅᑎᓪᓗᓂ ᐊᓯᖏᓐᓂᒃ ᓄᓇᓕᒃᓂᑦ, ᕿᑭᖅᑕᖅᔪᐊᕐᒥᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐃᓱᒪᔪᑦ ᐊᑑᑎᖃᕐᓴᕐᓂᕐᒪᓂᒃ ᑖᒃᑯᓄᖓ ᐱᖓᓱᓄᑦ ᓄᓇᓕᒃᓄᑦ ᐊᖑᓇᓱᒃᐸᒃᑐᓂᒃ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᒃ (BB) ᓇᓄᕐᓂᒃ (ᒥᑦᑎᒪᑕᓕᒃ, ᑲᖏᖅᑐᒑᐱᒃ, ᐊᒻᒪ ᕿᑭᖅᑕᖅᔪᐊᖅ) ᑲᑎᖃᑎᒌᒃᓗᑎᒃ ᐅᖃᐅᓯᖃᕐᓂᐊᕐᒪᑕ ᐃᓱᒫᓘᑎᒋᔭᖏᓐᓂᒃ. ᐃᖅᑲᓇᐃᔭᖅᑎᐅᔪᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖓᓄ ᐅᖃᓚᐅᕐᑐᑦ ᐃᓱᒫᓘᑎᖏᓐᓂᒃ ᑎᑎᕋᐅᓯᖃᕆᓐᓇᕐᓂᖏᓐᓂᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ ᐅᕙᓘᓐᓃᑦ ᐅᐸᒍᑎᓯᒪᓗᑎᒃ ᑭᒃᑯᓕᒫᓂᒃ ᑲᑎᒪᑎᑦᑎᓕᖅᐸᑕ ᐋᕿᒃᓱᖅᑕᐅᕙᓪᓕᐊᔪᒥᑦ ᓇᓄᕐᓄᑦ ᐊᐅᓚᑦᑎᔾᔪᑎᒃᓴᓄᑦ ᐸᕐᓇᐅᑎᓂᒃ (ᐅᓪᓗᖅ ᐊᒻᒪ ᓇᓅᓐᓂᐊᕐᓂᖓ ᐅᖃᐅᓯᐅᓛᖅᑐᖅ).

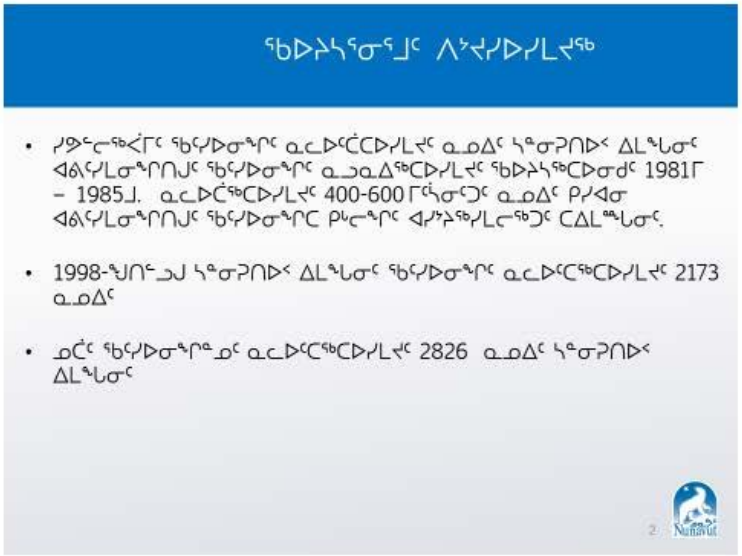
4.0 ᓇᐃᓈᖅᓯᔪᑎᑦ

ᑕᒪᐃᓐᓂ ᐱᖓᓱᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᓂᕈᐊᕐᓯᒪᔪᐃᓐᓇᐅᓚᐅᕐᑐᑦ ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᒥ ᐅᓄᕐᓯᒋᐊᖅᑎᑦᑎᓂᕐᒥ ᑲᑎᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐊᒻᒪᓗ ᐊᓯᖑᖅᑎᑦᑎᓂᕐᒥ ᓇᓕᐊᓐᓂ ᐊᖑᑎᓂᒃ ᐊᕐᓇᓂᒡᓗ ᐊᔾᔨᒌᖏᑎᑦᑎᓂᕐᒥ ᐋᕿᒃᑕᐅᓗᓂ 1:1 (ᐊᑕᐅᓯᖅ ᐊᖑᑎ ᐊᑕᐅᓯᕐᒥ ᐊᕐᓇᒧᑦ). ᐊᑐᓂ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᓇᓗᐃᓚᐅᕐᑐᑦ ᓄᓇᖏᓐᓂᑦ ᐅᖃᓪᓚᖃᑎᖃᕐᓂᐊᕐᓂᕋᖅᑐᑎᒃ ᐊᒻᒪᓗ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓐᓄᑦ ᐅᐸᒍᑎᔪᓐᓇᓚᐅᖏᑦᑐᓄᑦ ᑲᑎᒪᓂᕐᓄᑦ, ᑐᓂᓯᓚᐅᖏᓐᓂᖏᓐᓂᒃ ᑎᑎᖅᑲᓂᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᓄᑦ ᐃᑲᔪᖅᑐᐃᒐᓱᒃᓂᕐᒧᑦ. ᐊᑐᓂ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᖃᓚᐅᕐᑐᑦ ᑕᑯᔪᒪᓂᖃᖅᓗᑎᒃ ᐊᓯᖑᕐᓯᒪᔪᒥᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᓇᓄᐃᑦ ᓂᕙᓚᑕᖏᓐᓂᒃ. ᐊᔾᔨᒌᓂᒃᑕᐅᖅ ᐃᓱᒫᓘᑎᖃᓚᐅᕐᑐᑦ ᓄᓇᓖᑦ ᐆᒥᖓ ᐊᓯᔾᔨᓗᐊᖃᑦᑕᖅᑐᒥᒃ ᓇᓄᒍᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᔾᔪᓯᕐᒥᑦ ᐊᒻᒪᓗ ᖁᐊᔅᓵᕐᓇᕐᓂᕐᒧᑦ ᑐᖁᑕᐅᔪᑦ ᓂᕙᓚᑕᖏᓐᓂ ᐊᑐᕈᑎᐅᕙᒃᓂᖏᓐᓂᒃ.



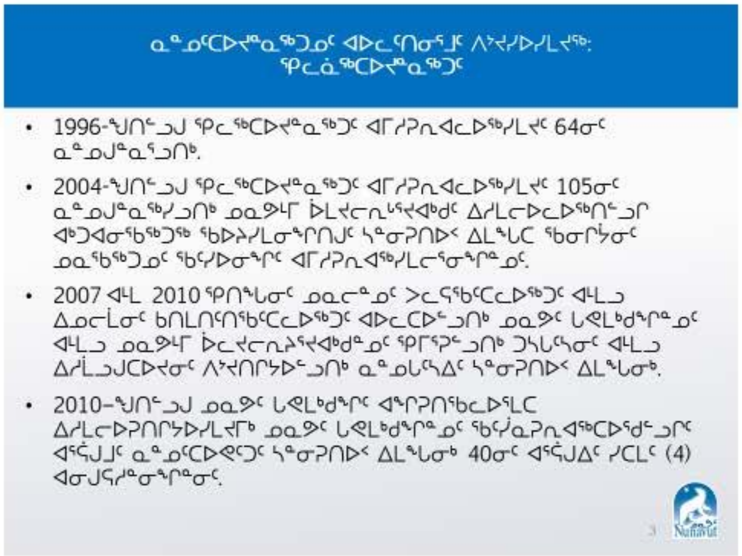
ᑲᑐᑎᓗᒍ ᓄᓇᕗᑦ ᐱᕈᖅᐸᓪᓕᐊᑎᓪᓗᒍ
Building *Nunavut* Together
*Nunavu*liuqatigiingniq
Bâtir le *Nunavut* ensemble

ᓇᓐᓄᕐᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐊᐅᓚᕿᑎᓂᖅ
ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ



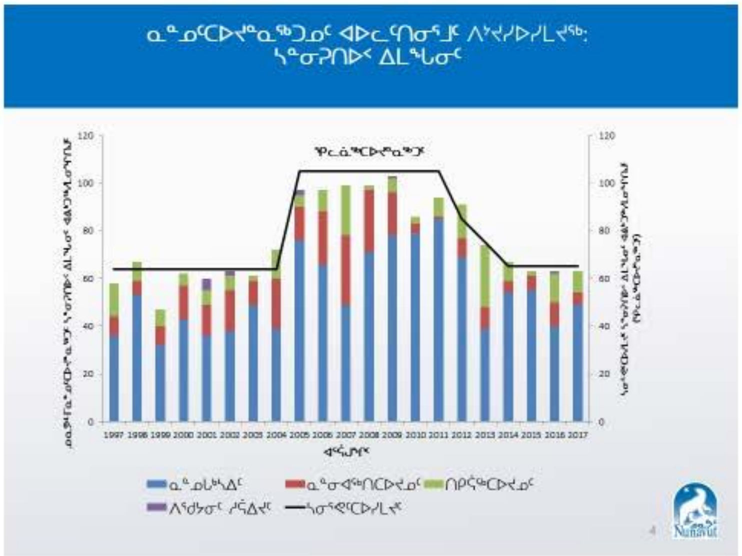
ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐱᔾᔪᓯᐅᓯᒪᔪᖅ

- ᓯᕗᓪᓕᖅᐹᒥᑦ ᖃᑦᓯᐅᓂᖏᑦ ᓇᓚᐅᑦᑖᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ ᐊᕙᑦᓯᒪᓂᖏᑎᒍᑦ ᖃᑦᓯᐅᓂᖏᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖓᓂᑦ 1981ᒥ – 1985ᒧ. ᓇᓚᐅᑖᖅᑕᐅᓯᒪᔪᑦ 400-600 ᒥᑦᓵᓂᑦᑐᑦ ᓇᓄᐃᑦ ᑭᓯᐊᓂ ᐊᕙᑦᓯᒪᓂᖏᑎᒍᑦ ᖃᑦᓯᐅᓂᖏᑕ ᑭᒡᓕᖏᑦ ᐊᓯᔾᔨᖅᓯᒪᓕᖅᑐᑦ ᑕᐃᒪᙵᓂᑦ.
- 1998-ᖑᑎᓪᓗᒍ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ ᖃᑦᓯᐅᓂᖏᑦ ᓇᓚᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ 2173 ᓇᓄᐃᑦ
- ᓄᑖᑦ ᖃᑦᓯᐅᓂᖏᓐᓄᑦ ᓇᓚᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ 2826 ᓇᓄᐃᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ

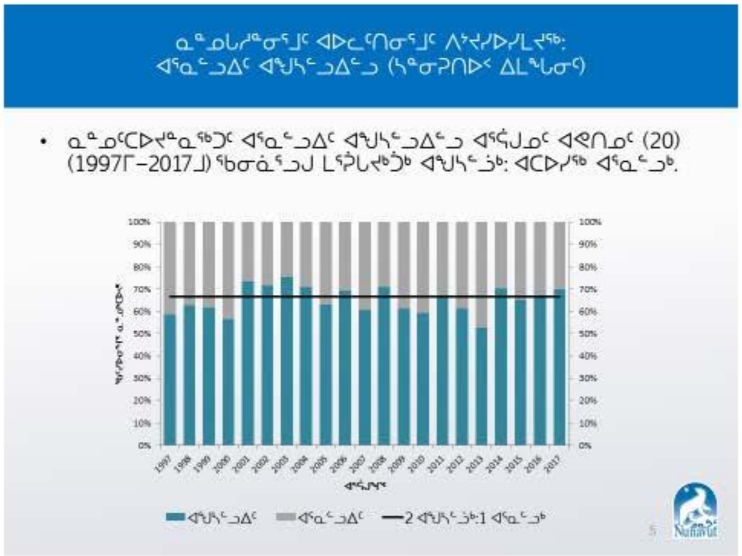


ᓇᓄᕐᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᔾᔪᑎᐅᓯᒪᔪᖅ: ᕿᓚᓈᖅᑕᐅᔪᓐᓇᖅᑐᑦ

- 1996-ᖑᑎᓪᓗᒍ ᕿᓚᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᒥᓱᕈᓐᓇᐊᓚᐅᖅᓯᒪᔪᑦ 64ᓂᑦ ᓇᓄᒧᓐᓇᕐᓗᑎᒃ.
- 2004-ᖑᑎᓪᓗᒍ ᕿᓚᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᒥᓱᕈᓐᓇᐊᓚᐅᖅᓯᒪᔪᑦ 105ᓂᑦ ᓇᓄᒧᓐᓇᖅᓯᓗᑎᒃ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᐃᓱᒪᓕᐅᓚᐅᖅᑎᓪᓗᒋ ᐊᒃᑐᐊᓂᖅᑲᖅᑐᖅ ᖃᐅᔨᓯᒪᓂᖏᑎᒍᑦ ᓴᓇᕈᑎᐅᑉ ᐃᒪᖓᑕ ᖃᓂᒋᔮᓂᑦ ᓄᓇᖅᑲᖅᑐᓄᑦ ᖃᑦᓯᐅᓂᖏᑦ ᐊᒥᓱᕈᓐᓇᐊᖅᓯᒪᓕᕐᓂᖏᓐᓄᑦ.
- 2007 ᐊᒻᒪ 2010 ᕿᑎᖓᓂᑦ ᓄᓇᓕᓐᓄᑦ ᐳᓚᕋᖅᑕᑦᑕᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ ᐃᓄᓕᒫᓂᑦ ᑲᑎᒪᑎᑦᑎᖅᑕᑦᑕᓚᐅᖅᑐᑦ ᐊᐅᓚᑕᐅᓪᓗᑎᒃ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᓐᓄᑦ ᐊᒻᒪᓗ ᓄᓇᕗᒻᒥ ᐆᓚᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ ᕿᒥᕐᕈᓪᓗᑎᒃ ᑐᓴᒐᔅᓴᓂᑦ ᐊᒻᒪᓗ ᐃᓱᒫᓗᒍᑕᐅᔪᓂᑦ ᐱᔾᔪᑎᒋᔭᐅᓪᓗᑎᒃ ᓇᓄᒐᔅᓴᐃᑦ ᓴᓇᕈᑎᐅᑉ ᐃᒪᖓᓂᒃ.
- 2010–ᖑᑎᓪᓗᒍ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᑦ ᐊᖏᕈᑎᖃᓚᐅᕐᒪᑕ ᐃᓱᒪᓕᐅᕈᑎᒋᔭᐅᓯᒪᔪᒥᒃ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᖏᓐᓄᑦ ᖃᔾᔮᓇᕈᓐᓇᐊᖅᑕᐅᖁᓪᓗᒋᑦ ᐊᕐᕌᒍᒧᑦ ᓇᓄᕐᑕᐅᕙᑦᑐᑦ ᓴᓇᕈᑎᐅᑉ ᐃᒪᖓᓂᒃ 40ᓂᑦ ᐊᕐᕌᒍᐃᑦ ᓯᑕᒪᑦ (4) ᐊᓂᒍᕐᓯᓐᓂᖏᓐᓂᑦ.

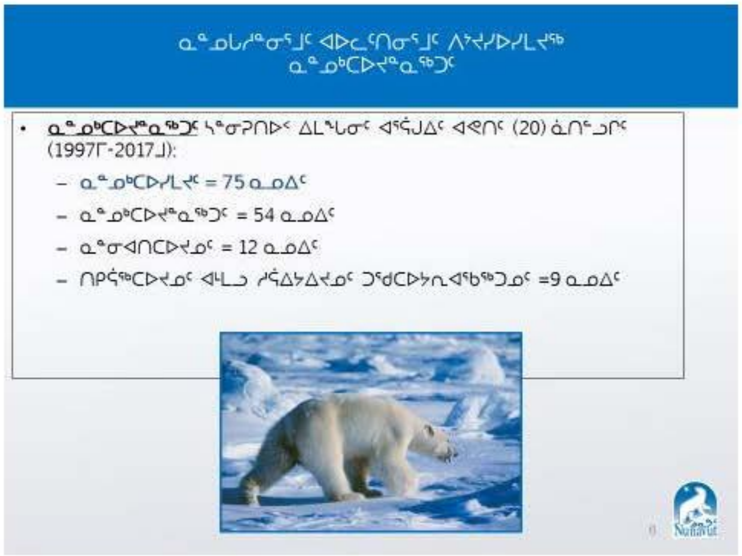


ᓇᓄᕐᑕᐅᔪᓐᓇᖅᑐᓄᑦ ᐊᐅᓚᕐᑎᓂᕐᒧᑦ ᐱᔾᔪᑎᐅᓯᒪᔪᖅ: ᓴᓇᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ



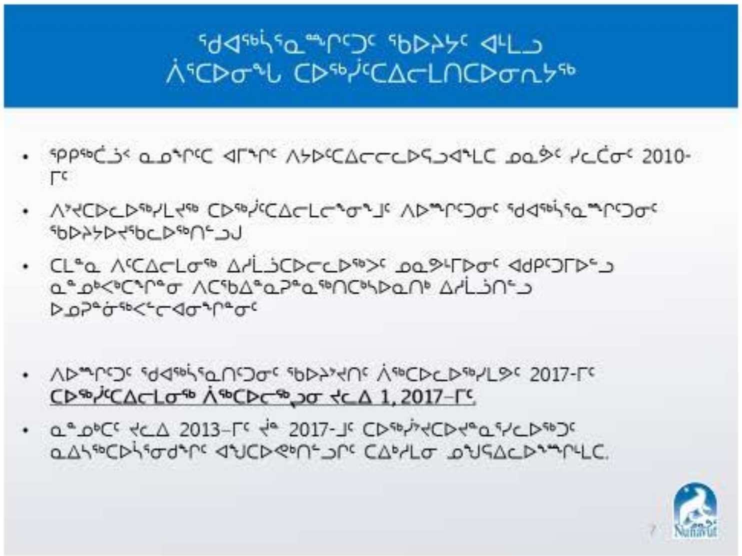
ᓇᓄᓕᓲᓐᓂᕐᒧᑦ ᐊᐅᓚᕿᑎᓂᕐᒧᑦ ᐱᓕᕆᐅᓯᒪᔪᖅ: ᐊᕐᓇᓕᒫᓗᐃᑦ ᐊᖑᑦᓯᓕᒫᓗᐃᑦ (ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᕐ)

- ᓇᓄᕐᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᕐᓇᓕᒫᓗᐃᑦ ᐊᖑᑦᓯᓕᒫᓗᐃᓪᓗ ᐊᕐᕌᒍᓄᑦ ᐊᕿᑎᓄᑦ (20) (1997ᒥ−2017ᒧ) ᖃᓂᖕᓇᕐᓗᒍ ᒪᕐᕉᓕᖅᑯᖅ ᐊᖑᑦᓯᓕᒫᖅ: ᐊᑕᐅᓯᖅ ᐊᕐᓇᓕᒫᖅ.



ᓇᓄᓕᓲᓐᓂᕐᒧᑦ ᐊᐅᓚᕿᑎᓂᕐᒧᑦ ᐱᓕᕆᐅᓯᒪᔪᖅ ᓇᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ

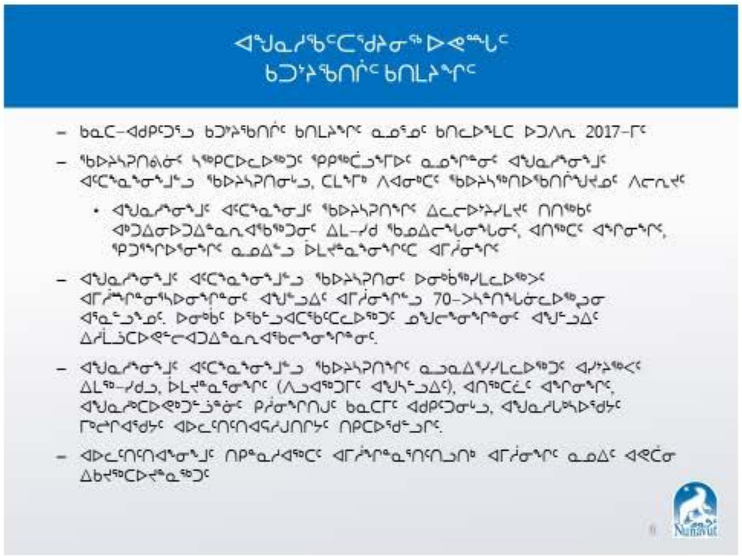
- <u>ᓇᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ</u> ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦ ᐊᕐᕌᒍᐃᑦ ᐊᕿᑦ (20) ᓈᑎᓪᓗᒋᑦ (1997ᒥ-2017ᒧ):
  - ᓇᓄᒃᑕᐅᓯᒪᔪᑦ = 75 ᓇᓄᐃᑦ
  - ᓇᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ = 54 ᓇᓄᐃᑦ
  - ᓇᓂᐊᑎᑕᐅᔪᓄᑦ = 12 ᓇᓄᐃᑦ
  - ᑎᑭᕝᕙᒃᑕᐅᔪᓄᑦ ᐊᒻᒪᓗ ᓯᕗᐃᔾᔭᐃᔪᓄᑦ ᑐᕐᖁᑕᐅᔾᔪᑎᓇᐊᖅᑲᖅᑐᓄᑦ =9 ᓇᓄᐃᑦ



ᖁᐊᖅᑳᖅᓇᙱᑦᑐᑦ ᖃᐅᔨᓴᑦ ᐊᒻᒪᓗ ᐱᑦᑕᐅᓂᖓ ᑕᐅᖅᓯᑦᑕᐃᓕᒪᑎᑕᐅᓂᕆᓴᖅ

- ᕿᑭᖅᑖᓗᑉ ᓇᓄᕐᓯᑦᑕ ᐊᒥᖅᓯᑦ ᐱᔭᐅᑦᑕᐃᓕᓕᓚᐅᕋᓱᐊᕐᒪᑕ ᓄᓇᕘᑦ ᓯᓚᑖᓂᑦ 2010-ᒥᑦ
- ᐱᔾᔪᑕᐅᓚᐅᖅᓯᒪᔪᖅ ᑕᐅᖅᓲᑦᑕᐃᓕᒪᓕᕐᓂᕐᒧᑦ ᐱᐅᙱᑦᑐᓂᑦ ᖁᐊᖅᑳᖅᓇᙱᑦᑐᓂᑦ ᖃᐅᔨᓴᐅᔪᖅᑲᓚᐅᖅᑎᓪᓗᒍ
- ᑕᒪᓐᓇ ᐱᑦᑕᐃᓕᒪᓂᖅ ᐃᓱᒫᓘᑕᐅᓕᓚᐅᖅᐳᑦ ᓄᓇᕗᒻᒥᐅᓂᑦ ᐊᑯᑭᑦᑐᒥᐅᓪᓗ ᓇᓐᓄᒃᐸᒃᑕᖅᑎᓐᓂ ᐱᑕᖃᐃᓐᓇᕈᓐᓇᖅᑎᑦᓯᐅᓇᑎᒃ ᐃᓱᒫᓘᑎᓪᓗ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᐊᓂᖏᓐᓂᑦ

- ᐱᐅᙱᑦᑐᑦ ᖁᐊᖅᑳᖅᓇᑎᑦᑐᓂᑦ ᖃᐅᔨᔾᔪᑎᑦ ᐲᖅᑕᐅᓚᐅᖅᓯᒪᕗᑦ 2017-ᒥᑦ <u>ᑕᐅᖅᓲᑦᑕᐃᓕᒪᓂᖅ ᐲᖅᑕᐅᓕᖅᑐᓂ ᔪᓚᐃ 1, 2017-ᒥᑦ</u>
- ᓇᓐᓄᒃᑕᑦ ᔪᓚᐃ 2013-ᒥᑦ ᔫᓐ 2017-ᒧᑦ ᑕᐅᖅᓲᔾᔪᑕᐅᔪᓐᓇᖅᓯᓚᐅᖅᑐᑦ ᓇᐃᓴᖅᑕᐅᓵᖅᓂᑯᖏᑦ ᐊᖑᑕᐅᕙᒃᑎᓪᓗᒋᑦ ᑕᐃᒃᓯᒪᓂ ᓄᖑᕋᐃᓚᐅᖅᓯᒪᓕᕐᒪᑕ.



ᐊᖑᓇᓱᒃᑲᑦᑕᖁᔨᓂᖅ ᐅᕙᙵᑦ ᑲᑐᔾᔨᖃᑎᒌᑦ ᑲᑎᒪᔨᖏᑦ

- ᑲᓇᑕ–ᐊᑯᑭᑦᑐᕐᓗ ᑲᑐᔾᔨᖃᑎᒌᑦ ᑲᑎᒪᔨᖏᑦ ᓇᓄᕐᓄᑦ ᑲᑎᓚᐅᖅᒪᑕ ᐅᑐᐱᕆ 2017–ᒥᑦ
- ᖃᐅᔨᓴᖅᑎᕕᓃᑦ ᓴᖅᑭᑕᐅᓚᐅᖅᑐᑦ ᕿᑭᖅᑖᓗᖕᒥᐅᑦ ᓇᓄᖅᑎᓐᓂᑦ ᐊᖑᓇᓱᒃᑎᖏᓐᓄᑦ ᐊᑦᑕᕐᓇᖅᑐᖏᓐᓗ ᖃᐅᔨᓴᖅᑎᓂᓪᓗ, ᑕᒪᕐᒥᒃ ᐱᐊᓂᒃᑕᑦ ᖃᐅᔨᓴᖅᑎᐅᖃᑎᒌᖑᔪᓄᑦ ᐱᓕᕆᔪᑦ
    - ᐊᖑᓇᓱᒃᑎᖏᓐᓄᑦ ᐊᑦᑕᕐᓇᖅᑐᓄᑦ ᖃᐅᔨᓴᖅᑎᖏᑦ ᐃᓚᓕᐅᔾᔨᓯᒪᔪᑦ ᑎᑎᖅᑲᑦ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᓇᐊᖅᑐᓂᑦ ᐃᒪ–ᓯᑯ ᖃᓄᐃᓕᖓᓂᖓᓂᑦ, ᐊᑎᖅᑕᑦ ᐊᖏᓂᖏᑦ, ᕿᑐᙳᕆᐅᖅᓯᓂᖏᑦ ᓇᓄᐃᓪᓗ ᐆᒪᔪᖃᕐᓂᖏᑦᑕ ᐊᒥᓱᓂᖏᑦ
- ᐊᖑᓇᓱᒃᑎᖏᓐᓄᑦ ᐊᑦᑕᕐᓇᖅᑐᖏᓐᓗ ᖃᐅᔨᓴᖅᑎᓂᑦ ᐅᓂᒃᑳᖅᓯᒪᓚᐅᖅᐳᑦ ᐊᒥᓲᙳᕆᐊᓂᖅᓴᐅᓂᖏᓐᓂᒃ ᐊᖑᓪᓗᐃᑦ ᐊᒥᓱᓂᖏᓐᓗ 70–ᐳᓴᓐᑎᖓᓂᓚᐅᖅᑐᓂ ᐊᕐᓇᓪᓗᖕᓄᑦ. ᐅᓂᒃᑳᑦ ᐅᖃᓪᓗᐊᑕᖅᑕᑦᑕᐅᔪᑦ ᓄᖑᓕᖅᑎᖅᑎᓐᓂᒃ ᐊᖑᓪᓗᐃᑦ ᐃᓱᒫᓘᑕᐅᕙᓪᓕᐊᑐᐃᓐᓇᓇᐊᖅᑕᓕᖅᑎᖅᑎᓐᓂᒃ.
- ᐊᖑᓇᓱᒃᑎᖏᓐᓄᑦ ᐊᑦᑕᕐᓇᖅᑐᖏᓐᓗ ᖃᐅᔨᓴᖅᑎᖏᑦ ᓇᓗᓇᐃᖅᓯᒪᓚᐅᖅᐳᑦ ᐊᓯᔾᔨᖅᐸᑦ ᐃᒪᖅ–ᓯᑯᓗ, ᐆᒪᔪᖃᕐᓂᖏᑦ (ᐱᓗᐊᖅᑐᒥᑦ ᐊᖑᓴᓪᓗᐃᑦ), ᐊᑎᖅᑕᓖᑦ ᐊᖏᓂᖏᑦ, ᐊᖑᓇᓱᒃᑕᐅᕙᒃᑐᓐᓇᖅᓂᖏᑦ ᑭᓯᓂᖏᑎᒍᑦ ᑲᓇᑕᒥᑦ ᐊᑯᑭᑦᑐᓂᓪᓗ, ᐊᖑᓇᓱᒪᓂᖅᐸᖅᑐᑦ ᒥᖅᑯᓕᐊᖅᑐᑦ ᐊᐅᓚᑦᑎᑦᑎᐊᕋᓱᒍᑎᖏᑦ ᑎᑭᑕᐅᖅᑯᓗᑎᑦ.
- ᐊᐅᓚᑦᑎᑦᑎᐊᕐᓂᖏᓐᓄᑦ ᑎᑭᓐᓇᓯᐊᖅᑕᑦ ᐊᒥᓲᖓᕐᓇᖅᑎᑦᑎᓗᑎᒃ ᐊᒥᓱᓂᖏᑦ ᓇᓄᐃᑦ ᐊᕙᑖᓂ ᐃᑲᔪᖅᑕᐅᔭᓐᓇᖅᑐᑦ

ᐊᖑᓇᓱᖅᑲᑦᑕᖅᑯᓯᓂᖅ ᐅᕙᖕᒐᑦ
ᑲᑐᔾᔨᖅᑲᑎᒌᑦ ᑲᑎᒪᔨᖏᑦ

- ᑖᒃᑯᐊ ᑲᑐᔾᔨᖅᑲᑎᒌᑦ ᑲᑎᒪᔨᖏᑦ **ᐊᒥᓱᑭᓇᐊᖅᑯᔨᕗᑦ ᒫᓐᓇ ᖁᑭᖅᑖᓗᖕᒥᑦ ᐊᖑᑕᐅᔪᓐᓇᖅᑐᓂᑦ 65-ᒥᑦ 80–ᒧᑦ ᓇᓄᖅᓂᑦ** ᐊᖅᑰᑕᒫᑦ (ᐊᑯᑭᑦᑐᑦ = 80-ᓂᑦ ᓇᓄᖅᓂᑦ)
- ᖁᑭᖅᑖᓗᖕᒥᑦ ᓇᓐᓄᓴᓯᒃᑕᐅᔪᓐᓇᖅᑯᔨᔪᑦ ᑭᓲᓂᖕᒥᑦ 1:1–ᖑᓕᖅᓗᓂ (ᐊᑕᐅᓯᖅ ᐊᖑᓴᓪᓗᖅᑕᖅ ᐊᑕᐅᓯᖕᒧᑦ ᐊᖅᓇᓪᓗᒃᑲᖅᑕᒧᑦ) ᐱᔾᔪᑎᒋᓪᓗᒍ ᐃᓱᒫᓘᑕᐅᓂᖓ ᓄᖑᑉᐸᓪᓕᐊᑐᐃᓐᓇᓇᐊᖅᑲᓂᖏᑦ ᐊᖑᓴᓪᓗᐃᑦ.
- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᐊᓯᔾᔨᖅᑕᖅᑐᓂᑦ ᐱᔪᓐᓇᒐᒃᓴᓕᕆᔾᔪᑎᖅᑲᖅᑯᔨᕗᑦ ᐊᑐᖅᓂᐊᖅᓗᓂᔾᔪᒃ ᐋᖅᑭᒃᓯᐃᓂᖕᒥᓐᓂᑦ ᐲᖅᓯᓗᑎᒃ ᐊᖑᓴᓪᓗ ᐊᑕᐅᓯ ᐊᖅᓇᓪᓗᒃᑲᖅᓘᓐᓃᑦ ᓂᕈᐊᖅᑕᐅᒃᑯᑦᑕᑦ ᐆᒪᔪᓕᒫᓄ ᐱᔭᔭᓄᑦ ᓄᓇᓕᖕᓂᑦ ᓂᕈᐊᖅᑕᐅᒃᑯᑎᒃᑯᑦ ᑎᒍᓯᖅᑲᑦᑕᖅᓂᖅᓂᖅᓂᑦ ᐊᖑᒐᒃᓴᓇᔭᓂᑦ.

ᖁᔭᓐᓇᒦᒃ
THANK YOU
QUANAQUTIN
MERCI

ᐊᖑᓇᓱᒃᑕᐅᔪᓂᒃ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᐊᒻᒪ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ ᓇᓄᕐᓂᒃ ᐊᕕᒃᑐᖅᓯᒪᔪᓄᑦ ᑲᑎᙵᐅᖅᑐᓄᑦ: ᓇᐃᓈᕈᑎᖏᑦ ᓇᓂᓯᐅᔪᓂ

ᐱᕙᒌᔭᖅᑕᐅᓯᒪᔪᑦ ᑲᑎᒪᓂᐊᕐᓂᖏᓐᓄᑦ ᑲᓇᑕᒥ–ᑲᓛᕐᓖ ᓄᓈᑦ ᑲᑐᔾᔨᖃᑎᒌᒃᑐᓄᑦ ᑲᑎᒪᔨᓄᑦ ᓇᓄᕐᓄᑦ

ᐋᑐᐋ, ᑲᓇᑕᒥ

ᐅᑐᐱᕆ 25–27, 2017

ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᑲᑐᔾᔭᐅᓪᓗᑎᒃ ᐊᐅᓚᑕᐅᖕᒪᑕ ᑲᓇᑕᒥᑦ (ᓄᓇᕗᑦ) ᐊᒻᒪᓗ ᖃᓛᖕᓃᑦ. *ᑲᓇᑕᒥ–ᑲᓛᕐᓖ ᓄᓈᑦ ᑲᑐᔾᔨᖃᑎᒌᒃᑐᓄᑦ ᑲᑎᒪᔨᓄᑦ ᓇᓄᕐᓄᑦ* (JC) ᐱᓕᕆᐊᕆᔭᕐᓂᖅᓯᑎᑦᑎᖕᒪᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᒪᖀᖕᓂ ᐱᓕᕆᔪᓐᓇᕐᓂᖅᑲᕐᕕᐅᔫᖕᓂ. ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᕐᑯᔨᔪᑦ (JC) ᐅᖃᐅᔭᐅᒋᐊᓱᖑᒪᑕ ᐅᑯᓇᖓᑦ *ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ* (SWG) ᐃᓚᖃᕐᓗᑎᒃ ᐃᓚᒋᔭᐅᔪᓂᑦ ᑕᒪᒃᑭᖏᓐᓂᑦ ᓄᓇᓕᕐᔪᒥᐅᑕᓂᑦ. ᐱᒋᐊᕐᓗᓂ 2011–2014, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐱᓕᕆᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᓗᑎᒃ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ. ᐱᔾᔪᑕᐅᔪᖅ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒧᑦ ᑐᑭᓯᓇᓱᐊᕐᓗᑎᒃ ᐊᒃᑐᐃᓂᖓᓂᑦ ᐊᓯᔾᔨᕐᓂᖓᑕ ᑕᕆᐅᑉ–ᓯᑯᖓᑕ ᖃᓄᐃᑦᑑᓂᕆᕙᒃᑕᖓᓂᑦ ᐊᒻᒪᓗ ᐃᓱᒪᓕᐅᕈᑕᐅᔪᒪᓪᓗᓂ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ [TAH]; ᑲᑎᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐃᓄᖕᓂᑦ ᐊᑐᓂ ᐊᕐᕌᒍᒥ ᓄᖑᓗᐊᖅᑎᑕᐅᖏᓪᓗᑎᒃ ᐊᖑᓇᓱᒃᑕᐅᑎᓪᓗᒋᑦ). ᓄᑖᑦ ᖃᐅᔨᓇᓱᐊᕐᓃᑦ ᐱᔭᐅᔭᕆᐊᖃᓕᓚᐅᕐᒪᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑐᓴᐅᒪᔪᑎᒃᓴᑦ ᐅᑯᓇᖔᖅᑐᑦ ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᓗᑎᒃ ᑕᐅᕙᓂ 1990–ᖏᓐᓂ ᐱᑐᖃᐅᓗᐊᓕᓚᐅᕐᒪᑦ. ᑕᒪᒃᑮᔪᑦ ᓴᕿᑕᐅᓯᒪᔪᑦ ᐱᒋᐊᕐᓗᑎᒃ 2011–2014 ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᑦ ᓴᕿᑕᐅᓯᒪᖕᒪᑕ ᐊᓯᐊᓃᓪᓗᑎᒃ ᐅᓂᒃᑳᓕᐊᕆᓯᒪᔪᓂ.

ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᕐᑯᔨᔪᑦ (JC) ᑐᒃᓯᕋᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᐊᑐᕈᒪᓪᓗᑎᒃ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ 2011–2014-ᒥᖔᖅᑐᓂᑦ ᖃᐅᔨᓇᓱᐊᕈᑎᓂᑦ ᐅᖃᐅᔨᒥᐊᕈᑕᐅᖁᓪᓗᒋᑦ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᔨᓂᑦ. ᐱᓗᐊᙳᐊᖅᑐᒥᑦ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᕐᑯᔨᔪᑦ (JC) ᖃᐅᔨᔪᒪᓚᐅᕐᒪᑕ ᖃᓄᑎᒋᐅᒪᖔᑕ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᓈᒪᒃᑲᔭᕐᓗᑎᒃ ᐱᖓᓱᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᓄᑦ ᑎᑭᑕᐅᔪᒪᔪᓄᑦ: (1) ᐋᖅᑭᐅᒪᑎᑦᑎᓗᑎᒃ ᓄᖑᑉᐸᓪᓕᐊᕐᑯᓇᒋᑦ ᐅᓄᕐᓂᒃᑲᓂᖏᑕ ᐊᖏᔫᓂᑦ; (2) ᐋᖅᑭᐅᒪᑎᓪᓗᒋᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐅᖏᔫᓂᖏᑦ ᐋᖅᑭᑦᑎᐊᖅᓯᒪᕐᑯᓪᓗᒋᑦ ᐅᓄᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐊᕙᑎᑦᑎᓐᓄᑦ ᒪᑭᒪᐃᓐᓇᖅᑎᑕᐅᔪᓐᓇᕐᓂᖏᑦ; ᐊᒻᒪᓗ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᑕᒪᐅᖓᑐᐊᖅ, **(3) ᐅᓄᖏᓐᓂᖅᓴᐅᓕᖅᑎᑕᐅᓗᑎᒃ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐊᖏᔫᓂᖓ ᐃᒪᓐᓇᑎᒋᓪᓗᐊᐸᓗᒃ 30% ᐳᓴᓐᑎᑦ 10–15 ᐊᕐᕌᒍᓂ.** ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᕐᑯᔨᔪᑦ (JC) ᐃᓕᓴᖅᓯᓚᐅᕐᒪᑕ ᑕᒪᒃᑯᐊ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ

ᖃᐅᔨᓴᓂᑦ ᐃᓚᖃᕐᒪᑕ ᓇᓗᓈᕿᔪᓂᑦ, ᐊᔪᕐᓇᕐᓂᐊᕐᒪᑦ ᑖᒃᑯᓄᖓ ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓇᓚᐅᑦᑎᔪᒥᑦ ᑮᐅᑦᑎᐊᕈᑎᖃᕆᐊᒃᓴᖅ. ᑕᐃᒪᐃᒪᑦ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑦ (JC) ᑐᒃᓯᕋᓕᖅᐸᑦ, ᐊᑐᓂ ᐅᓄᕐᓂᒃᑲᓐᓂᐅᔪᑦ, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓈᓴᐅᓯᕆᖁᓪᓗᒋᑦ ᖃᓄᑎᒋᐅᒐᔭᕐᓂᖏᓐᓂᑦ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᖁᕝᕙᓯᒃᑐᒪᕆᐅᔪᓐᓇᕐᓗᑎᒃ (90% ᐳᓴᓐᑎᓂᑦ) ᐱᔪᓐᓇᕋᔭᕐᓂᖏᑦ ᐊᑐᓂ ᐊᐅᓚᑦᑎᓂᐅᔪᓄᑦ ᑎᑭᑕᐅᔪᒪᔪᓂᑦ (ᑕᒪᓐᓇ ᑐᑭᖃᒃᐳᖅ ᐊᒃᐸᓯᒃᑐᒪᕆᐅᒐᔭᕐᖁᓪᓗᒋᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᖁᓇᒋᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ), ᐱᖃᓯᐅᑎᓗᒋᑦ ᖃᓄᑎᒋᐅᒐᔭᕐᓂᖏᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᐊᑯᓐᓈᕿᓗᑎᒃ (70% ᐳᓴᓐᑎᖏᓂᑦ) ᐱᔪᓐᓇᕋᔭᕐᓗᑎᒃ ᐊᑐᓂ ᐊᐅᓚᑦᑎᓂᐅᔪᓄᑦ ᑎᑭᑕᐅᔪᒪᔪᓂᑦ (ᑕᒪᓐᓇᓗ ᑐᑭᖃᕐᓗᓂ ᐊᑯᓐᓈᕿᔪᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᖁᓇᒋᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓅᑦᑐᑦ).

ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᑭᐅᔪᑎᖃᓚᐅᕐᒪᑕ ᐊᐱᖅᑯᑎᒋᔭᐅᔪᓄᑦ ᐊᑐᕐᓗᑎᒃ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ ᐅᑯᓇᖖᖅᑐᓂᑦ 2011-2014 ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᓂᑦ ᐃᓚᒋᔭᐅᓗᑎᒃ ᐅᓄᕐᓂᖏᓐᓄᑦ ᐋᖅᑭᒃᓯᐃᔪᑎᓂᑦ ᓇᓄᕐᓂᑦ. ᐅᓄᕐᓂᖏᓐᓂᑦ ᐋᖅᑭᒃᓯᐃᔪᑎᑦ ᐱᖃᓯᐅᔾᔨᕙᒻᒪᑕ ᑕᕆᐅᑉ–ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓄᑦ. ᖃᐅᔨᓴᖅᓯᔪᑎᒋᔭᖏᓐᓂ, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᓱᒪᒃᓴᓯᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᐅᓄᖅᑐᓂᑦ ᖃᓄᐃᓕᔪᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᐊᒃᑐᐃᓂᖏᓐᓂᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH), ᐱᖃᓯᐅᑎᓗᒋᑦ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐊᖑᑎᑦ ᐊᕐᓇᓪᓗᐃᓪᓗ ᐊᒻᒪᓗ ᐅᑭᐅᖃᕐᓂᖏᑦ ᓇᓐᓄᒃᑕᐅᕙᒃᑐᑦ ᓇᓄᐃᑦ, ᐊᒻᒪᓗ ᓯᕗᓂᒃᓴᒥ ᐱᓕᕆᓂᐅᔭᖅᑐᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐊᒻᒪᓗ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᕆᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᕐᒧᑦ (TEK) ᖃᐅᔨᓇᓱᐊᕐᓃᑦ. ᓯᕗᓂᒃᓴᒥ ᐱᓕᕆᐊᖑᒐᔭᖅᑐᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᑦ ᐱᒻᒪᕆᐊᓘᒻᒪᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ, ᓄᑖᓂᒃ ᖃᐅᔨᓇᓱᐊᕐᓂᖅᑕᖃᖅᑲᑦᑕᖅᐸᑦ ᕿᓚᒻᒥᐅᓗᑎᒃ, ᐊᒃᐸᓯᒡᓂᖅᓴᐅᒐᔭᕐᒪᑕ ᐅᓄᕈᓐᓃᖅᑎᑕᐅᓗᐊᕋᔭᕐᓂᖏᑕ ᐅᖓᑎᓗᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᖃᑦᑕᖃᑦᑕᖅᐸᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᖃᐅᓇᓯᐊᕈᑎᑦ ᐱᔭᐅᑎᑦᑎᒐᔭᕐᒪᑕ ᓄᑖᓂᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ ᐊᐅᓚᑦᑎᔨᓄᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᕋᔭᖅᑐᓂᒃ ᐋᖅᑭᒋᐊᖅᓯᔪᑕᐅᔪᓐᓇᕐᓗᑎᒃ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH).

ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓ

2011–2013 ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᓃᑦ ᒥᒃᓴᐅᓴᒃᓯᓚᐅᕐᒪᑕ ᐃᒪᓐᓇᐸᓘᓇᓱᒋᔭᐅᓪᓗᑎᒃ 2800 ᓇᓄᐃᑦ (95% ᓈᓴᐅᑏᑦ ᐃᓗᐊᓂ [CI]=2,059–3,593) ᑕᒫᓂ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᕆᔭᐅᔪᓂᑦ. ᑕᕆᐅ ᓯᑯᖓ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᒥᑭᓪᓕᒃᑲᓐᓂᖅᓯᒪᖕᒪᑦ ᐊᓂᒍᖅᑐᓂ ᖃᑦᓯᑲᓪᓚᐃᑦ ᖁᔨᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ. ᑕᒪᓐᓇ ᒥᑭᓪᓕᒋᐊᖅᑎᑦᑎᓯᒪᖕᒪᑦ ᓇᔪᖅᑕᐅᓯᒥᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᓯᒥᑦ, ᐊᒻᒪᓗ ᐱᒋᐊᓕᖅᑎᑦᑎᓯᒪᑐᐃᓐᓇᕆᐊᖃᕐᒪᑦ ᓇᓄᕐᓂᑦ ᓴᓗᒡᓂᖅᓴᐅᓕᖅᑎᑦᑎᓗᓂ ᐊᒻᒪᓗ ᐊᑎᖅᑕᖃᖏᓐᓂᖅᓴᐅᓕᖅᑎᑦᑎᓗᓂ. ᓇᔪᖅᑕᐅᕙᒃᑐᖅ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᒪᑭᒪᑎᑦᑎᓂᐊᖅᑐᒃᓴᐅᓕᕐᒪᑦ ᖃᑦᓰᓐᓇᐅᓂᖅᓴᓂᑦ ᓇᓄᕐᓂᑦ ᓯᕗᓂᒃᓴᒥ ᑕᕆᐅᑉ-ᓯᑯᖓ ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓗᓂ ᑲᔪᓯᑐᐃᓐᓇᕋᔭᖅᐸᑦ. ᑕᐃᒪᐃᒪᑦ, ᑕᕆᐅᑉ-ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᐱᖃᓯᐅᔾᔭᐅᓚᐅᖅᐳᖅ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᖃᒃᑎᑕᐅᓗᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂ ᐅᑯᑎᒎᓇ ᐅᓄᕐᓂᖏᓐᓄᑦ ᐋᖅᑭᒃᓱᐃᔪᑎᖓᓂ.

ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᓂᑦ SWG ᖃᐅᔨᔪᑕᐅᓚᐅᕐᒪᑦ ᐊᔪᕐᓇᕋᔭᕐᓂᖓᓂᑦ ᐊᑐᕋᓯᒋᐊᖓ ᑎᑭᑕᐅᔪᒪᔪᖅ (1) ᐋᖅᑭᐅᒪᑎᑦᑎᓂᖅ ᓄᖑᑉᐸᓪᓕᐊᖏᓪᓗᑎᒃ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᓯᕗᓂᒃᓴᒥ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᑉ-ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᖃᑦᓰᓐᓇᐅᓂᖅᓴᐅᓕᖅᑎᑦᑎᒐᔭᕐᒪᑦ ᓇᓄᕐᓂᑦ ᖃᓄᑎᒋᐅᒐᔭᕐᓂᕋᓗᐊᖅᐸᑕ ᐊᖑᓇᓱᐊᖅᑕᐅᓂᖏᑦ ᐊᒻᒪᓗ ᐊᑐᓚᓇᖑᐊᖅᑎᑦᑎᓂᕆᔭᐅᓚᐅᖅᑐᖅ ᐃᒪᓐᓇᐅᓂᕋᐃᖕᒪᑦ ᐊᑐᖅᑕᐅᒐᔭᕐᓂᖅᐸᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ (3) 30% ᐳᓴᓐᑎᖏᓐᓂᑦ ᐅᓄᖏᓐᓂᖅᓴᐅᓕᖅᑎᑦᑎᓂᖅ ᐱᔭᒃᓴᐅᒐᔭᖏᒻᒪᑦ ᑕᐃᒪᓐᓇ ᐅᖃᐅᑎᒋᔭᐅᖕᒪᑦ ᐅᓄᕐᓂᖏᑕ ᖁᒃᓴᓇᕐᓂᖏᓐᓂ ᐱᔭᐅᐊᓐᓇᖅᑎᑕᐅᒐᔭᕐᓂᖏᓐᓂᑦ. ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᓱᒪᓕᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᖁᕝᕙᓯᒃᑐᒪᓇᐅᓂᖓᓂᑦ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓄᐊᖃᕐᓂᖓᑕ ᑎᑭᑕᐅᔪᒪᔪᑦ (2) ᐊᑐᕐᓗᒍ ᐅᓪᓗᒥᐅᓕᖅᑐᖅ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᑎᑭᐅᒪᑎᓪᓗᒋᑦ 120–ᓄᑦ ᓇᓄᐃᑦ ᐊᑐᓂ ᐊᕐᕌᖑᒥ (ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑎᑭᐅᒪᓗᑎᑦ 4.3% ᐳᓴᓐᑎᓄᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ), ᐊᒻᒪᓗ ᐊᑯᓐᓈᕿᔪᖅ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓄᐊᖅᑲᖅᒪᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ (2) ᐅᓪᓗᒥᐅᓕᖅᑐᖅ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᑎᑭᐅᒪᓗᑎᒃ 160–ᓄᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᑐᓂ ᐊᕐᕌᖑᒥ (ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐅᓄᕐᓂᖃᕐᓗᑎᒃ 5.7% ᐳᓴᓐᑎᓂᒃ ᐅᓄᕐᓂᖏᓐᓂᑦ). ᑕᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᒪᓕᑐᐃᓐᓇᕐᒪᑕ ᖃᑦᓯᑲᓪᓚᖕᓂᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᓂᑦ. ᓯᕗᓪᓕᖅᐹᖅ, ᐃᓱᒪᒋᔭᐅᓚᐅᕐᒪᑦ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐊᖑᑦ ᐊᕐᓇᓪᓗᐃᓪᓘᓐᓂᑦ ᓇᓐᓄᒃᑕᐅᖃᑦᑕᕐᓂᕆᓚᐅᖅᑕᖏᑦ ᑕᐃᒪᓐᓇᓴᐃᓐᓈᕐᓂᐊᖅᐸᑕ ᐊᑐᖅᑕᐅᖃᑦᑕᓚᐅᖅᑐᓂ ᐊᓂᒍᖅᑐᓂ ᖁᓪᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ. ᑕᐃᒪᓕ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᖃᓛᑦᓚᑦ ᓇᓐᓄᒃᐸᒃᑕᖏᑦ ᑲᑎᑕᐅᑎᓪᓗᒋᑦ, ᐃᒪᓐᓇᐅᒐᔪᓚᐅᖅᐸᖅ ᓇᓐᓄᒃᑕᐅᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ 1998 ᐊᒻᒪᓗ 2013 ᒪᑯᐊᖑᓚᐅᕐᒪᑕ 1.25 ᐊᖑᑎᑦ ᐱᔭᐅᑎᓪᓗᒍ 1 ᐊᕐᓇᓪᓗᒃ. ᑕᒪᓐᓇᓕ ᐊᖑᑕᐅᒃᐸᑕ ᐊᕐᓇᓪᓘᒃᐸᑕ ᓇᓐᓄᒃᑕᐅᓯᑦ ᐊᓯᔾᔨᖅᐸᑕ, ᐊᓯᐃᓪᓛᒃᑕᐅᖅ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐊᓯᔾᔨᓯᔭᖅᑐᒃᓴᐅᖕᒥᔪᑦ. ᐱᖃᓯᐅᔾᔭᐅᔪᖅ, ᑕᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᐃᒪᓐᓇᐅᓯᓇᐊᖅᑲᓕᖅᑎᑦᑎᖕᒪᑦ ᓄᓛᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᒃᑯᑦ ᐊᒻᒪᓗ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᑕᓕᕆᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᑦ (TEK) ᖃᐅᔨᓇᓱᐊᕈᑎᑦ ᐱᓕᕆᐊᖑᖃᑦᑕᕐᓂᐊᕐᓗᑎᒃ ᕿᓚᒻᒥᐅᓂᖅᓴᐅᖃᑦᑕᕐᓗᑎᒃ ᓯᕗᓂᒃᓴᒥ, ᓄᓛᓂᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ ᐊᑐᐃᓐᓇᐅᔪᖃᕐᓂᖅᓴᐅᖃᑦᑕᖁᓪᓗᒍ ᓈᓴᐅᓯᓇᐊᔪᑕᐅᖁᓪᓗᒍ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᓈᑐᐊᕋᖕᒪᑕ 15 ᐊᕐᕌᒍᑦ. ᑕᐃᒪᓕ ᓄᓛᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᒃᑯᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᐃᑦ ᐊᑐᐃᓐᓇᐅᖏᑉᐸᑕ ᓈᓴᐅᓯᓇᐊᔪᑕᐅᖁᓪᓗᒋᑦ

ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) 15 ᐊᕐᕌᒍᓂ, ᕐᑯᕝᕙᓯᒡᓂᖓ ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᐅᕐᑲᐅᑕᐅᕐᑲᐅᔪᖅ ᑕᑉᐸᒃᑯᓇᓂ ᐊᑐᖅᑕᐅᑎᑦᑎᔪᓐᓇᕐᔭᖏᒪᑕ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔪᒥᑦ ᐅᕐᑲᐅᑕᐅᔪᒥᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᕐᑯᔨᔪᑎᓂᑦ (JC) ᐊᒻᒪᓗ ᕐᑯᕝᕙᓯᒡᓂᖅᓵᓗᑐᐃᓐᓇᕆᐊᕐᑲᒃᓗᓂ ᐅᓗᕆᐊᓇᖅᑐᒦᓕᖅᑎᑦᑎᒐᔭᕐᓂᖓᑦ ᐅᖓᑎᑐᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᕐᑲᑦᑕᕐᓂᕆᒐᔭᖅᑕᖏᑕ ᐊᑐᕐᓂᐊᑐᐃᓐᓇᖅᑐᓂ ᕐᑲᑦᓯᑲᑦᓚᖕᓂ ᕐᑯᓕᐅᔭᖅᑐᑦ ᐊᕐᕌᒍᓂ. ᐱᖓᔪᖓᓐᓂ, ᑕᒪᒃᑯᐊ ᓴᕐᑭᑕᐅᔪᑦ ᐃᒪᓐᓇᐅᓕᖅᑎᑦᑎᖕᒪᑕ ᐊᒃᐸᓯᒪᑕ ᒥᒃᓴᐅᓴᒃᓯᔪᑕᐅᔪᑦ ᐆᒪᑯᑖᕐᑲᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᑐᖅᑎᓪᓗᒋᑦ 2011-2013 ᓯᓂᓚᐅᖏᒪᑕ ᓴᖅᑭᑕᐅᔪᑦ. ᐊᑐᓚᓇᒃᑐᒥ, ᑕᒪᒃᑯᐊ ᐊᒃᐸᓯᒃᑐᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑕᐅᔪᑦ ᐃᓚᖓᒍᑦ ᓯᓚᓂᕐᑲᑐᐃᓐᓇᕆᐊᓚᐅᒐᓗᐊᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᕐᑲᕐᓂᖓᓄᑦ ᑕᕆᐅᑉ–ᓯᑯᕐᑲᕈᓐᓃᖅᐸᓪᓕᐊᓂᐅᑉ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᓇᓄᕐᓂᑦ.

ᕐᑲᐅᔩᓇᓯᐊᖅᑎᑦ ᐃᕐᑲᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᕐᑲᑎᒋᑦ (SWG) ᓇᑐᓇᐃᖅᓯᓚᐅᕐᒪᑦ ᕐᑲᑦᓯᑲᑦᓚᖕᓂᑦ ᐃᓯᒫᑐᑕᐅᔪᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᕆᔭᐅᔪᓂᑦ. ᓯᕗᓪᓕᖅ, ᒫᓇ ᕐᑲᑦᓯᓐᓇᐅᓂᖅᓴᐅᓕᕐᒪᑕ ᐊᖑᑎᑦ ᓇᓄᐃᑦ ᐊᕐᓇᓪᓗᖕᓂᓪᓚ ᐅᓄᕐᓂᒃᑲᓐᓂᕆᔭᐅᔪᓂ, ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐆᒪᑯᑖᕐᑲᑦᑕᕐᓂᖏᑦ ᐊᒃᐸᓯᒡᓂᖅᓴᐅᓪᓗᑎᒃ ᐊᕐᓇᓪᓗᐃᑦ ᐆᒪᑯᑖᕐᑲᑦᑕᕐᓂᖏᓐᓂᑦ. ᐊᖑᑎᑦ ᓇᓄᐃᑦ ᑲᔪᓯᑐᑎᒃ ᕐᑲᑦᓵᓇᕈᒃᐸᓪᓕᐊᑐᐃᓐᓇᕐᑲᐸᑕ ᐃᒪᓐᓇᐅᓕᖅᑎᑦᑎᔪᓐᓇᕐᒪᑦ ᕐᑲᑦᓯᓐᓇᐅᓂᖅᓴᐅᓕᕐᑐᑎᒃ ᐊᑎᖅᑕᐃᑦ ᐃᕐᓂᐊᖑᔪᑦ, ᐊᒻᒪᓗ ᕐᑲᑦᓯᓐᓇᐅᓂᖅᓴᐅᓕᕋᔭᕐᒪᑕ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐅᓄᕐᓂᖏᑦ. ᑐᑭᓯᓂᖅ ᕐᑲᓄᐃᓕᕙᓪᓕᐊᓂᖏᓐᓂᑦ ᐊᖑᑎᑦ ᓇᓄᐃᑦ ᐱᒻᒪᕆᐊᒎᖕᒪᑦ ᐊᑐᕋᔭᕐᓂᖓᓄᑦ ᓯᕗᓂᒃᓴᒥ ᕐᑲᐅᔩᓇᓯᐊᕐᓂᕐᒧᑦ. ᐱᕐᑲᑖᓂ, ᐊᐅᓚᑦᑎᔩᑦ ᓂᕆᐅᒋᐊᕐᑲᓕᖅᑐᑦ ᑕᒪᓐᓇ ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᑕᒫᓂ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓇᓂ ᕐᑲᑦᓵᓇᕈᒃᐸᓪᓕᐊᓂᐊᕐᒪᑕ ᐊᑐᕐᓂᐊᖅᑐᓂ ᕐᑲᑦᓯᑲᑦᓚᖕᓂ ᕐᑯᓕᐅᔭᖅᑐᑦ ᐊᕐᕌᒍᓂ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᑉ–ᓯᑯᕐᑲᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᕐᑲᑦᓵᓇᕈᕐᑎᑦᑎᑲᓐᓂᕐᓂᐊᕐᒪᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ ᓇᓄᐃᑦ ᑕᒫᓂ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᓇᔪᖅᑕᐅᕙᒃᑐᒥ ᒪᑭᒪᔪᑕᐅᔪᓐᓇᕐᓂᖓᓂ. ᐱᖓᔪᖓᓐᓂ, ᐃᒪᓐᓇᔪᓐᓇᕐᒥᔪᖅ ᐅᓄᕐᓂᕆᔭᐅᔪᓄᑦ ᐱᓕᕆᔪᑕᐅᕙᒃᑐᖅ ᐊᑐᖅᑕᐅᓯᖅ ᕐᑲᐅᔩᓇᓯᐊᖅᑎᑦ ᐃᕐᑲᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᕐᑲᑎᒋᓂᑦ (SWG) ᑐᖓᐅᑦᑎᑐᑎᒃ–ᒥᒃᓴᐅᓴᒃᓯᔪᒥᓂᐅᑐᐃᓐᓇᕆᐊᕐᑲᕐᒪᑕ ᐱᐅᓈᖏᑦᑐᓂᑦ ᐊᒃᑐᐃᓂᕐᑲᕐᓯᒋᑐᒋᑦ ᑕᕆᐅᑉ–ᓯᑯᕐᑲᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᓇᓄᐃᑦ. ᑕᒪᓐᓇ ᑕᐃᒪᓐᓇᐃᑉᐸᑦ, ᐊᓯᐃᓪᓚᒃ ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᕐᑯᕝᕙᓯᒡᓂᖏᑦ ᐅᕐᑲᐅᑕᐅᕐᑲᐅᔪᑦ ᑕᑉᐸᒃᑯᓇᓂ ᕐᑯᑖᓂ ᐃᒪᓐᓇᐃᓕᑎᑦᑎᔪᓐᓇᕐᒪᑕ ᐅᖓᑎᑐᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᑎᑦᑎᕐᑲᑦᑕᕐᑐᑎᒃ. ᑕᐃᒪᓇᐅᒐᓗᖅᑎᓪᓗᒍ ᑕᕆᐅᑉ–ᓯᑯᕐᑲᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᐅᓗᕆᐊᓇᕈᑎᕐᑲᕐᓂᐊᕐᒪᑦ ᐅᓄᕐᓂᖅᓴᓄᑦ ᓇᓄᕐᓄᑦ ᐊᑐᕐᓂᐊᖅᑐᒧᑦ–ᑕᐅᕗᖓᑐᕐᑲᒃ, ᐊᓯᖏᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᒫᓐᓇ ᐊᑐᖅᐸᑦᑕᐊᓕᕐᒪᑕ ᑕᕆᐅᑉ–ᓯᑯᕐᑲᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᐊᔾᔨᒋᖏᑕᖏᑎᒍ ᕐᑲᓄᐃᓕᕙᓪᓕᐊᑦᑐᑎᒃ, ᐊᒻᒪᓗ ᐊᔪᕐᓇᐸᑐᖕᒪᑦ ᓇᓚᐅᑐᒋᑦ ᕐᑲᐅᔩᒪᕙᒦᕋᓯᒋᐊᖏᑕ ᓯᕗᓂᒃᓴᒥ ᐊᒃᑐᐃᓂᕆᒐᔭᖅᑕᖓ ᑕᒪᑐᒪ ᑕᕆᐅᑉ–ᓯᑯᕐᑲᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓᑕ. ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᑕᒪᒃᑯᐊ ᐃᓯᒫᑐᑕᐅᔪᑦ, ᕐᑲᐅᔩᓇᓯᐊᖅᑎᑦ ᐃᕐᑲᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᕐᑲᑎᒋᑦ (SWG) ᐊᑐᓕᕐᑯᔨᓕᕐᑯᔨᓕᒪᐅᕐᒪᑕ ᐱᓂᕐᓗᒃᑕᐅᔭᐃᖅᓯᓗᑎᒃ ᐱᓕᕆᕐᑯᔨᑦᑐᑎᒃ ᓇᓐᓄᕐᑲᑦᑕᕐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ ᐃᑲᔪᕐᑯᑦᑐᒍ ᐊᒃᓲᖏᓐᓂᖅᓴᐅᓕᖅᑎᑦᑎᔪᑕᐅᓗᓂ ᐅᖓᑎᑐᐊᖓᓄᑦ–ᐊᖑᑦᑕᐅᓂᖏᓐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᕆᔭᐅᔪᓂ.

ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓ

2012–2014 ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᓃᑦ ᒥᒃᓵᐅᓴᒃᓯᓚᐅᕐᒪᑕ ᐃᒪᓐᓇᐸᔫᓇᓱᒋᔭᐅᓪᓗᑎᒃ 360 ᓇᓄᐃᑦ (95% ᓈᓴᐅᑏᑦ ᐃᓗᐊᓂ [CI]=221–493) ᑕᒫᓂ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ KB ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂ. ᑕᐃᒪᐃᑲᓗᐊᖅᑎᓪᓗᒍ ᑕᓇᐅᑉ ᓯᑯᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ ᒥᑭᓂᖅᓴᐅᓕᕋᓗᐊᖅᑎᓗᒍ ᑕᐃᒪᖓᓂᑦ ᐊᓂᒍᖅᑐᓂ ᖃᑦᓯᑲᓪᓚᖕᓂ ᖁᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ, ᑕᒪᓐᓇ ᓱᖅᑭᑎᑦᑎᖏᒪᑦ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᕐᒥᑦ ᒪᑯᓂᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ ᓇᓄᖏᓐᓂᑦ. ᐋᓚᒃ, ᑕᓇᐅᑉ–ᓯᑯᖓᑕ ᖃᓄᐃᑦᑑᓕᕐᓂᖓ ᐃᒪᓐᓇᐃᓕᑎᑦᑎᑐᐃᓐᓇᓇᐊᖃᕐᒪᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓃᑦᑐᓂᑦ ᖃᑦᔫᓐᓇᕈᒃᐸᓪᓕᐊᑎᑦᑎᖏᓪᓗᓂ ᓇᓄᕐᓂᑦ. ᑕᐃᒪᐃᒪᑦ, ᑕᓇᐅᑉ–ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᐃᓚᒋᓕᐅᑎᔭᐅᓚᐅᖏᓚᖅ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᖓᓄᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ ᓇᓄᐃᑦ ᐱᓕᓇᔪᑕᐅᖃᑦᑕᖅᑐᓂ.

ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᓱᒪᓕᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᐃᒪᓐᓇᐅᓂᖓᓂᑦ ᐊᕙᕐᓇᕐᒪᑦ ᓇᓗᓇᐃᖅᓯᔭᓇᐊᒃᓴᖅ ᐅᓄᕐᓂᖏᓐᓂᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᐃᒪᓐᓇᐅᔪᓐᓇᕋᔭᕐᓂᖅᐸᑕ ᓂᕆᐅᓇᑦᑎᐊᕐᓗᑎᒃ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓇᐊᖃᕐᓂᖏᑦ ᑎᑭᑕᐅᔪᒪᔪᑦ (1) ᐊᒻᒪᓗ (2). ᑖᓐᓇ ᐱᔾᔪᑎᖃᖏᑦᑐᖅ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐱᕚᓪᓕᐊᑎᐊᖏᓐᓂᖏᓐᓂᑦ. ᐃᒪᓐᓇᖔᕐᓕ, ᐱᔾᔪᑎᖃᒃᐳᖅ ᓇᓗᓈᕿᔪᓂᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᑕᖃᓚᐅᕐᒪᑦ ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᖃᐅᔨᓇᓱᐊᕈᑎᖏᓐᓂ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᖃᑦᔫᓐᓇᐅᖓᑕ ᐊᒻᒪᓗ ᐊᕝᕙᐸᓗᒃᓗᑎᒃ ᖃᐅᔨᓇᓯᒋᐊᖏᑕ. ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᖃᐅᔨᓚᐅᕐᒪᑕ ᐊᑯᓐᓈᕿᖓᒪᑦ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓇᐊᖃᕐᓂᖓ ᑎᑭᑕᐅᔪᒪᔫᑉ (1) ᐅᓗᒥᐅᓕᖅᑐᖅ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᑎᑭᐅᒪᓗᑎᒃ 10 ᓇᓄᕐᓄᑦ ᐊᕐᕌᒍᑕᒪᑦ (ᐊᖑᑦᓯᑕᖃᑦᑕᕐᓂᖕᑦ ᑎᑭᐅᒪᓗᑎᒃ 2.8% ᐳᓴᓐᑎᓄᑦ). ᑖᓐᓇ ᖃᐅᔨᔭᐅᔪᖅ ᑐᙵᕕᖃᒃᑎᑕᐅᖓᑦ ᑕᒪᒃᑭᖏᓐᓂᑦ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᒪᔪᑎᑦ ᑕᐅᕗᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᓇᓄᖏᓐᓄᑦ, ᐊᒻᒪᓗ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᓇᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᖅ (TEK) ᐃᒪᓐᓇᐅᓂᕋᐃᖓᑕ ᑕᐃᒃᑯᐊ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᐅᓄᖅᓯᒃᑲᓐᓂᖅᓯᒪᕐᑯᔨᓕᕐᒪᑕ ᑕᐃᒪᖕᖓᓂᑦ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ 1990-ᒥᑦ. ᐊᔾᔨᐸᓗᒋᓪᓗᓂᒋᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᑕ ᓴᓐᓂᕈᑎᐅᑉ ᑕᓇᐅᖓᑕ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ, ᑕᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᒪᓕᑐᐃᓐᓇᕐᒪᑕ ᖃᑦᓯᑲᓪᓚᖕᓂᑦ ᖃᓄᐃᓕᖓᓂᓇᔭᖏᑕ. ᓯᕗᓪᓕᖅᐹᖅ, ᐃᓱᒪᒋᔭᐅᓚᐅᕐᒪᑦ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐊᖑᑦ ᐊᕐᓇᓪᓗᐃᓪᓘᓐᓂᑦ ᓇᓐᓄᒃᑕᐅᖃᑦᑕᕐᓂᓇᓚᐅᖅᑕᖏᑦ ᑕᐃᒪᓐᓇᕆᓴᐃᓐᓈᕐᓂᐊᖅᐸᑕ ᐊᑐᖅᑕᐅᖃᑦᑕᓚᐅᖅᑐᓂ ᐊᓂᒍᖅᑐᓂ ᖁᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ. ᑕᐃᒪᓕ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᖃᓖᓪᓕᒥᐅᑦ ᐊᖑᓇᓱᐊᖅᐸᒃᑕᖏᑦ ᑲᑎᑕᐅᔭᕐᕚᖓᑕ, ᐃᒪᓐᓇᑎᒋᐅᕙᔾᔮᑦ 0.94 ᐊᖑᑎᑦ ᐊᑕᐅᓯᕐᒥᑦ 1 ᐊᕐᓇᓪᓗᓕᖓᖅᑐᖅᑲᕝᖓᑦ. ᐱᖃᑖᓂ, ᑕᒃᑯᐊ ᓴᕿᑦᑐᑦ ᐃᒪᓐᓇᐅᔭᓇᐊᖃᒃᑎᑦᑎᖓᒪᑕ ᓄᑖᓂᒃ ᖃᐅᔨᓇᓱᐊᕐᓂᒃᑯᑦ ᐊᒻᒪᓗ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᓇᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᖅ (TEK) ᖃᐅᔨᓇᓱᐊᖅᑕᐅᖃᑦᑕᕐᓂᐊᕐᒪᑕ ᐊᑯᓚᐃᑐᒥᑦ ᓯᕗᓂᒃᓴᒥ, ᓄᑖᑦ ᖃᐅᔨᒪᔪᑎᒃᓴᐃᑦ ᐊᑐᐃᓐᓇᐅᖃᑦᑕᕐᓂᐊᕐᒪᑕ ᓈᓴᐅᓯᓇᔪᑕᐅᒃᑲᓐᓂᖃᑦᑕᕐᓂᐊᕐᒪᑕ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᓈᒍᐊᖒᖓᑕ 15 ᐊᕐᕌᒍᑦ. ᑕᐃᒪᓕ ᓄᑖᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒧᑦ ᖃᐅᔨᒪᔪᑎᒃᓴᓂᑦ ᐊᑐᐃᓐᓇᐅᔪᕐᑲᖏᐸᑉ ᓈᓴᐅᓯᓇᔪᑕᐅᒃᑲᓐᓂᖃᑦᑕᕐᓂᐊᖅᑐᓂᑦ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ

ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) 15-ᓂ ᐊᕐᕌᒍᓂ, ᐅᓄᕐᓂᖕᒋᑦ ᑖᒃᑯᐊ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᓴᕿᑎᑕᐅᔪᑦ ᑕᑉᐸᒃᑯᓇᓂ ᐊᒃᓲᓂᖅᓴᖅ ᐱᕕᒡᔪᐊᕈᑕᐅᔪᓐᓇᖅᑐᓂᒃ ᐅᖓᑎᓗᐊᖓᓄᑦ-ᓇᓄᒃᑕᐅᖃᑦᑕᓗᐊᖅᐸᑕ ᐊᑐᖅᑎᓪᓗᒋᑦ ᐊᑐᕐᓂᐊᖅᑐᑦ ᖃᑦᓯᑲᓪᓚᐃᑦ ᖁᓕᐅᔪᖅᑐᑦ ᐊᕐᕌᒍᐃᑦ.

ᑕᐅᕙᓂᑦᑎᐊᖅ ᐱᓕᕆᓂᐅᓂᐊᖅᑐᒥ, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓇᓗᓇᐃᖅᓯᓚᐅᕐᒪᑕ ᐅᓄᖕᒋᓐᓂᖅᓴᓂᒃ ᐃᓱᒫᓗᑕᐅᔪᓂᑦ ᑕᐃᒃᑯᓄᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᐅᖃᑕᐅᔪᓂᑦ ᑕᒪᒃᑯᓇᖓᓪᓕ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᑕ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ. ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ ᐱᑕᖃᕐᒪᑦ ᖃᑦᓯᑲᓪᓚᖏᓐᓂᑦ ᐱᒻᒪᕆᐊᓗᖏᓐᓂᑦ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᔭᕆᐊᓕᖏᓐᓂᑦ. ᐊᔾᔨᓪᓗᐊᐸᓗᒋᓗᓂᒋᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᑕ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖓᑕ, ᒫᓇ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᐅᖓᑕ ᐊᖑᑏᑦ ᐊᕐᓇᓪᓗᖕᓂᑦ ᑕᒫᓂ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖓᓐᓂ, ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐆᒪᑯᑖᒍᓐᓇᖕᒋᓐᓂᖅᓴᐅᖕᒪᑕ ᐊᕐᓇᓪᓗᖕᓂᑦ ᐆᒪᑯᑖᓪᓂᕆᔫᖕᒋᓐᓂᑦ. ᐱᖃᑖᓂ, ᑕᕆᐅᑉ-ᓯᑯᖓ ᓄᖑᓴᕋᐃᓕᕙᓪᓕᐊᒐᓗᐊᖅᑎᓪᓗᒍ ᐱᐅᓈᖅᑐᓂᒃ ᐊᒃᑐᐃᓂᖃᕐᒪᑦ ᑖᒃᑯᓂᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓃᑦᑐᓂᑦ (KB) ᓇᓄᕐᓂᑦ ᒫᓐᓇᕐᑖᖑᓚᐅᖅᑐᖅ ᖁᓕᐅᔪᖅᑐᑦ ᐊᕐᕌᖑᓂ, ᑕᒪᒃᑯᐊ ᐊᒃᑐᐃᓂᖓᑦ ᓂᕆᐅᒋᔭᐅᑦ ᐱᐅᓈᕈᓐᓂᕐᓂᐊᕋᓱᕆᔭᐅᓪᓗᑎᒃ ᐊᑯᓂ ᑕᐅᕗᖓᑐᖃᒃ ᑕᕆᐅᑉ-ᓯᑯᖓ ᓄᖑᓴᕋᐃᓕᐊᑐᐃᓐᓇᖅᑎᓪᓗᒍ. ᐱᖓᓱᖓᓐᓂ, ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖓᑦ ᑲᔪᓯᑐᐃᓐᓇᕐᓂᐊᕐᒪᑕ ᐊᔪᕐᓇᐸᓗᐃᓐᓇᕐᓗᑎᒃ ᖃᐅᔨᓇᓱᐊᕆᐊᖓᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᓄᖕᒋᑦᑐᑯᓘᖕᒪᑕ ᐊᒻᒪᓗ ᐅᐸᒐᕐᓂᖕᒋᑦᑐᒦᑦᓗᑎᒃ ᐊᐅᓚᑦᑎᓂᒧᑦ ᐱᓕᕆᕕᐅᖃᑦᑕᓯᒥ. ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᒪᓐᓇᐃᓗᖁᔨᓕᓚᐅᕐᒪᑦ ᑲᑐᔾᔭᐅᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑎᑦ (JC) ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᖃᕐᓗᒋᑦ ᐱᓕᕆᒋᐊᖃᓐᓂᕈᑎᒃᓴᐃᑦ, ᐱᓕᕆᐊᖑᓴᕋᐃᓐᓂᖅᓴᐅᕙᓪᓗᑎᒃ ᖃᐅᔨᒪᒋᐊᖅᓯᓂᕐᒧᑦ ᐸᕐᓇᒍᑎᑦ ᑕᐃᑯᓄᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖓᓄᑦ, ᓱᕐᓗ ᒪᑯᓇᓂ ᓈᒪᒃᓯᑐᐊᕋᖓᑦ ᖃᖓᑕᓱᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᕙᓪᓗᑎᒃ ᐊᑐᖅᓯᐅᑎᓗᒋᒃ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᕆᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᑦ (TEK), ᑕᒪᓐᓇᓗ ᐃᑲᔪᕈᓐᓇᑐᐃᓐᓇᕆᐊᖃᕐᒪᑦ ᐊᐅᓚᑦᑎᔨᓂᑦ ᖃᐅᔨᔪᑕᐅᔪᓐᓇᕐᓗᓂ ᐅᖓᑎᓗᐊᖓᓄᑦ-ᓇᓄᒃᑕᐅᖃᑦᑕᓕᕐᓂᖅᐸᑕ ᐊᒻᒪᓗ ᐊᓯᖏᓐᓄᑦ ᐊᑲᐅᙱᓕᐅᕆᑎᓄᑦ ᓯᕗᓂᖓᒍᑦ ᐊᒃᓱᑐᐊᕌᓗᒃ ᐱᓂᕐᓗᒃᓯᓚᐅᖕᒋᓐᓂᖕᒋᓐᓂᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖕᒋᓐᓂᑦ.

Harvest Assessment for the Baffin Bay and Kane Basin Polar Bear Subpopulations

Final Report to the Canada-Greenland Joint Commission on Polar Bear

Eric V. Regehr[1], Stephen Atkinson[2,*], Erik W. Born[3,*], Kristin L. Laidre[1,2,*], Nicholas J. Lunn[4,*], and Øystein Wiig[5,*]

[1] Polar Science Center, Applied Physics Laboratory, University of Washington, Seattle, Washington 98105, USA

[2] Box 19, Group 7, RR#2, Dugald, Manitoba, R0E 0K0, Canada

[3] Greenland Institute of Natural Resources, 3900 Nuuk, Greenland

[4] Environment and Climate Change Canada, University of Alberta, Edmonton, Alberta, T6G 2E9 Canada

[5] Natural History Museum, University of Oslo, NO-0318 Oslo, Norway

[*] Scientific Working Group of the Canada-Greenland Joint Commission on Polar Bear

31 July 2017

Please cite as: Regehr, E.V., S. Atkinson, E.W. Born, K.L. Laidre, N.J. Lunn, and Ø. Wiig. 2017. Harvest Assessment for the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear, 31 July 2017: iii + 107 pp.

Table of Contents

Executive Summary

Background

The Baffin Bay (BB) and Kane Basin (KB) polar bear subpopulations are jointly managed by Canada (Nunavut) and Greenland. The Canada-Greenland Joint Commission on Polar Bear (JC) facilitates coordination between the two jurisdictions. In 2011, due to concerns about potentially unsustainable harvest and the demographic effects of observed, long-term changes in sea-ice habitat, the JC tasked its Scientific Working Group (SWG) with reassessing the status of the BB and KB subpopulations. Both subpopulation were surveyed in the 1990s. However, by 2011 these earlier surveys were considered too old to provide a reliable basis for assessment or future harvest management. Consequently, the SWG undertook a research program using genetic capture-recapture (BB and KB), radiotelemetry (BB and KB), and aerial surveys (KB) from 2011–2014 to obtain updated information on subpopulation size, status, delineation, habitat quality, and habitat use (SWG 2016).

Results from these recent studies suggest that, over the last few decades, the BB subpopulation has undergone significant changes in range, movements, habitat use, body condition, and reproductive performance concurrent with a decline in sea-ice extent, duration, and quality. Baffin Bay is a relatively abundant subpopulation with an estimated 2,826 bears in 2011–2013 (95% CI = 2,059–3,593). However, due to differences in capture-recapture (CR) sampling designs, results from the surveys in the 1990s and 2010’s cannot be directly compared to assess trends in the size of the subpopulation. For KB, recent studies suggest that the subpopulation is transitioning from a multiyear sea-ice system towards a sea-ice regimen characteristic of the seasonal ice ecoregion, where sea ice melts almost entirely during the summer. The KB subpopulation has responded to changing sea ice by expanding its range since the 1990s, especially during summer. Larger and more variable home ranges, and the use of lower sea-ice concentrations in summer and fall, have also been observed (SWG 2016). The current abundance of the KB subpopulation was estimated at 357 bears in 2012–2014 (95% CI = 221–493), with the available evidence suggesting this subpopulation has been stable or increasing since the 1990s.

Harvest assessment approach

Following a review of research findings for BB and KB (SWG 2016), the JC provided the SWG with three potential alternatives for subpopulation Management Objectives: (1) maintain a relatively stable subpopulation size; (2) maintain a subpopulation size that achieves maximum sustainable yield, with respect to a potentially changing environmental carrying capacity; and (3) reduce subpopulation size by approximately 30% in 10–15 years. Management Objectives 1 and 2 were considered potential objectives for both BB and KB, while an exploration of Management Objective 3 was associated with concerns about human-bear conflicts, and was requested for BB only. The JC requested that the SWG use the best-available information to provide advice on harvest management strategies, including levels of Total Allowable Harvest, under which these objectives could be achieved. The JC provided two levels of risk tolerance ("low" and "medium") for not meeting each objective.

In this report, we use the ecological and demographic data from SWG (2016) to evaluate a suite of potential harvest strategies for the BB and KB subpopulations. We evaluated all strategies against the specific management objectives and risk tolerances provided by the JC, and for each strategy we also recorded other metrics of biological or management interest (e.g., the probability of severely depleting adult male bears). We interpreted the JC's request for advice on "low" and "medium" risk tolerance to mean a 90% and 70% chance of successfully meeting a management objective, respectively (alternatively, a 10% and 30% chance of failing to meet a management objective).

We performed a quantitative risk assessment using a demographic model based on the life history of polar bears, which can include the effects of environmental change (Regehr et al. 2017). The potential effects of future changes in sea-ice conditions on subpopulation size and status, and the resulting implications for harvest management, were incorporated in the analyses by using projected trends in carrying capacity (*K*, the capacity of the environment to support a given number of polar bears). In addition to this direct environmental effect, the model included a mechanistic submodel of Allee effects in the mating system, which limited reproduction under conditions of low subpopulation density or imbalanced numbers of adult females and males. We considered several different scenarios of the vital rates (e.g., rates of reproduction and survival) for each subpopulation due to uncertainty and potential bias in some demographic parameters from CR studies for both BB and KB.

For each combination of management objective and vital rates scenario, we evaluated multiple harvest strategies. Harvest strategies were defined in terms of the key elements that can be identified and adaptively managed by authorizing agencies, including harvest rate and harvest level (measured in number of independent bears [i.e., not including cubs-of-the-year or yearlings] removed annually), the sex and age composition of the harvest, the management interval, and the quality (i.e., level of statistical precision) of available survey data. We evaluated 10, 15, and 20-year management intervals, defined as the number of years between successive changes to the harvest based on new data from subpopulation surveys. The management interval often used as an objective in Canada is 15 years. For BB and KB, the interval between the two most recent surveys was 18 years (1993 to 2011), although some management adjustments occurred during this period. Thus, a management interval of 15–20 years approximates current practices for these subpopulations. A management interval of 10 years was used to illustrate the effects of more frequent subpopulation surveys and management changes.

Strategies using three harvest sex ratios (*SR*) were examined; *SR* = 1 (i.e., a 1:1 male-to-female ratio) reflecting conditions where harvest is not selective for either males or females; *SR* = 2 reflecting the target ratio for sex-selective harvest currently implemented in Canada, and reflecting the reported sex ratio of the combined Canada-Greenland harvest 1998–2013; and, for each subpopulation, a '*status quo*' sex ratio of the combined Canada-Greenland harvest 1998–2013, based on results from recent genetic sampling (2011–2013) indicating that sex was incorrectly reported for a substantial number of harvested bears.

All population projections assumed a state-dependent (i.e., dependent on current conditions) management approach, under which harvest levels did not remain constant in the future, but rather were updated according to the management interval. This means that the harvest strategies are tied directly to the timeline for reassessing subpopulation abundance and vital rates.

Baffin Bay

For the BB subpopulation, projections included a proxy for changes in *K* estimated from the number of ice-covered days per year in the BB region, which decline by approximately 5.5% per decade when projected forward in time. Use of a projected, declining trend in *K* is consistent with evidence for range contractions, and changes in nutritional condition and reproductive rates of BB polar bears associated with sea-ice loss (SWG 2016). In the demographic model, we assumed that the current rates of survival and reproduction for BB polar bears reflect a subpopulation that is near its maximum net productivity level (MNPL, the subpopulation size that results in the greatest net annual increment in numbers resulting from reproduction minus losses due to natural mortality). In other words, we assumed that subpopulation size has been held below *K* due to harvest, and that BB bears are not currently experiencing strong density-dependent suppression of survival or reproduction. Our projections did not include potential density-independent effects of sea-ice loss, which could reduce subpopulation resilience and capacity to support harvest in the future. If such changes occur rapidly compared to the schedule for future subpopulation surveys and harvest changes (as determined by the management interval), the risk of negative population outcomes would be higher than estimated from our projections.

We evaluated three scenarios of the vital rates for the BB subpopulation. Each scenario used the same rates of reproduction (litter production rate and cub-of-the-year litter size) as calculated from the recent genetic CR data (2011–2013) but differed in the rates of survival. Scenario 1 used estimates of unharvested survival (S^*) calculated from CR data for the period 2011–2013. Scenario 2 used estimates of S^* calculated from CR data for the period 1998–2010. Scenario 3 used estimates of S^* representing the “average” rates seen amongst polar bear subpopulations; this provided a benchmark for comparison with other subpopulations. Of the three scenarios, we considered Scenario 2 a more likely representation of the status of the BB subpopulation, because it used data specific to BB and could reproduce plausible trends in subpopulation abundance and sex ratio that were consistent with the available scientific information and Traditional Ecological Knowledge. Additionally, unlike Scenario 1, estimates of S^* for the period 1998–2010 were less susceptible to terminal bias (i.e., bias at the end of a time series of estimates, a common problem in CR studies).

Scenario 2 resulted in an unharvested asymptotic population growth rate $\lambda = 1.08$ (SE = 0.02) annually (i.e., 8% per year). For harvest strategies with $SR = 1.25$ (i.e., a 1.25:1 male-to-female sex ratio in the harvest, the estimated *status quo* for BB based on genetic data), an age composition based on historic harvest data, and a 15-year management interval, present-day harvest rates of up to 4.3% and 5.7% were consistent with Management Objective 2 under “low” and “medium” risk tolerances, respectively. We focused on Management Objective 2 for the BB subpopulation because this objective is more relevant to sustainable harvest when *K* is declining. Applying these harvest rates to the current subpopulation size of 2,826 would result in present-day harvest levels of up to approximately 120 and 160 bears per year, depending on risk tolerance. Under this harvest strategy, the present-day harvest level would be maintained for a 15-year period, at which point a new subpopulation survey should have been completed and the harvest should have been re-calculated. Over the next 35 years (approximately three polar bear generations), the harvest level would be expected to decline due to declining *K* with sea-ice loss, and possibly due to other demographic effects.

The harvest strategies listed above (i.e., present-day harvest rates of up to 4.3% and 5.7%) should be interpreted with caution for several reasons. First, strategies at the upper end of this range were associated with up to a 12% probability of severely depleting adult male bears, and up to a 4% probability of extirpation (i.e., reduction to a very small and non-viable subpopulation size) after 35 years. Second, recent subpopulation studies provided evidence for ecological effects of sea-ice loss that could, now or in the future, translate into negative demographic effects above and beyond the declining trend in *K* and Allee effects included in our model. Although there were several reasons to place less confidence in the relatively low estimates of *S** from 2011–2013 (Scenario 1), it is possible that these lower estimates reflected, to some extent, a reduced capacity for subpopulation growth due to sea-ice loss. If that is the case, the risks of negative population outcomes could be much higher than estimated under Scenario 2. Our analyses did not make purposefully-conservative assumptions, and therefore could have understated the future effects of sea-ice loss. Such risks could be reduced through a precautionary approach to harvest management.

Simulations for Management Objective 1 (maintaining a relatively stable subpopulation size) demonstrated that this objective likely cannot be achieved in the mid- to long-term due to projected declines in *K*, which would reduce subpopulation size regardless of harvest level. Simulations also suggested that Management Objective 3 for the BB subpopulation (reduction of 30% in 10–15 years) is probably not feasible at the level of risk tolerance stated by the JC. The largest-possible subpopulation reduction that remained within risk tolerance (with some caveats—see main text) was approximately 25% over 15 years. This required a 1:1 male-to-female sex ratio in the harvest, a 5-year management interval, and improved precision in the vital rates estimated from future subpopulation surveys. Under this harvest strategy, a present-day harvest rate of 8.7% (approximately 245 bears per year) would be applied for a 5-year period, at which point a new subpopulation survey would have been completed and the harvest would be re-calculated. Over a 15-year period, the harvest level would need to decline rapidly as subpopulation size declined. Results suggested that attempting a managed subpopulation reduction without a near-optimal, state-dependent approach—for example, applying a fixed-level harvest of 245 bears per year without new subpopulation surveys—would be associated with high probabilities of severe male depletion and extirpation after 15 years.

Kane Basin

For the KB subpopulation, projections included interannual variation in K but no declining trend, reflecting evidence that decreasing sea ice in the multiyear-ice region of KB may have positive ecological effects in the near term (e.g., increased marine productivity as the system transitions to annual sea-ice dynamics; SWG 2016). Similar to BB, we assumed that the estimated rates of survival and reproduction for KB polar bears reflect a subpopulation that is currently functioning near MNPL.

We evaluated two scenarios of the vital rates that differed in terms of estimated survival rates for young bears. Scenario 1 used time-constant estimates of S^* calculated from CR data for the period 1992–2014; and Scenario 2 used similar estimates, but with mean values of S^* for bears less than or equal to 2 years of age modified, to reproduce the estimated increase in subpopulation abundance from 224 bears in the 1990s to 357 bears in the 2010s (SWG 2016). We considered Scenario 2 a more likely representation of the status of the KB subpopulation, because under Scenario 1 the unmodified estimates of S^* for bears age 2 years or less were low (range across age classes of 0.45–0.73) compared to other subpopulations of similar productivity, and had high statistical uncertainty due to sparse data (e.g., less than 4 cubs-of-the-year were sampled per year 2012–2014). Additionally, use of survival rates under Scenario 1 suggested a poor demographic status for the KB subpopulation, yielding an unharvested population growth rate of 1% per year ($\lambda = 1.01$ [SE = 0.04]). This low rate is inconsistent with multiple lines of evidence for the KB subpopulation, including estimated increases in subpopulation size since the 1990s (SWG 2016), the likely positive trends in marine productivity in the region, recent information on nutritional condition and reproduction, and available Traditional Ecological Knowledge.

Scenario 2 for the KB subpopulation resulted in an unharvested population growth rate of 5% per year (λ =1.05 [SE = 0.06]). For harvest strategies with a SR = 0.94 (the estimated *status quo*, based on genetic data), an age composition based on historic harvest data, and a 15-year management interval, present-day harvest rates up to 1.7% and 1.1% were consistent with Management Objectives 1 and 2, respectively, at the "medium" level of risk tolerance. At the "low" level of risk tolerance, Management Objectives 1 and 2 could not be met in the absence of harvest, due to variability in subpopulation trajectories resulting from high uncertainty in the vital rates. Applying harvest rates of 1.1% to 1.7% to the current subpopulation size of 357 would result in a present-day harvest levels of up to approximately 4–6 bears per year. Under this harvest strategy, the present-day harvest level would be maintained for a 15-year period, at which point a new subpopulation survey would be completed and the harvest would be re-calculated. Over the next 35 years, the harvest level would be expected to remain stable or increase due to stable or potentially increasing K. Harvest strategies at the upper end of this range were associated with up to a 17% probability of severely depleting adult male bears, and up to a 3% increased probability of extirpation compared to projections with no harvest, after 35 years.

Under Scenario 2, harvest rates that met management objectives for the KB subpopulation were lower than the observed harvest rate for the period 1998–2014, during which subpopulation size likely increased (SWG 2016). This inconsistency was due primarily to high statistical uncertainty in estimates of S^* for bears age 2 years or less, a consequence of small sample sizes and relatively short study periods of research in KB. If alternative assumptions were made for uncertainty in estimates of S^* for bears age 2 years or less (e.g., if the precision of these estimates was increased to match the precision of survival estimates for older bears), present-day harvest rates up to 2.2% to 2.8% (8–10 bears per year) were consistent with Management Objective 1 at a "medium" risk tolerance, when following a state-dependent approach with a 15-year management interval.

Considering all available ecological and demographic data for the KB subpopulation, present-day harvest rates up to approximately 2.8% (10 bears per year) seem unlikely to cause negative population outcomes under a state-dependent approach with effective monitoring. It is possible that the logistical challenges of studying the KB subpopulation may lead to continued difficulty in obtaining accurate and precise estimates of vital rates, despite increased survey efforts. We therefore suggest developing a suite of ecological and demographic indicators to monitor subpopulation status, including accurate information on the level and composition of the harvest, marine productivity, habitat availability, reproductive rates, and estimates or indices of subpopulation size (via aerial survey or CR).

Monitoring requirements and further research

The results in this report are intended to help inform and guide subsequent decisions of the JC with respect to determining appropriate levels of harvest for these two shared subpopulations of polar bears. Both BB and KB are experiencing long-term trends in the extent, duration, and quality of sea-ice habitat. Our analyses identify harvest strategies that are designed to maintain subpopulation size near MNPL with respect to a changing *K*, and to limit negative effects of harvest on the probability of subpopulation persistence. All of the harvest strategies presented in this report require the existence of a coupled research-management system under which both the sustainable harvest rate and the harvest level are adjusted periodically, based on new scientific information from subpopulation surveys and other sources. For both BB and KB, our analyses demonstrate that shorter management intervals and more precise data can substantially reduce the risk of negative population outcomes associated with a given harvest strategy. A state-dependent management approach is an effective means of reducing the risk of overexploitation while maintaining opportunities for use. This is especially important if sea-ice loss is currently having ecological effects on polar bears that may signal negative demographic effects in the future (BB); or if a less-conservative harvest strategy is selected when the currently available estimates of vital rates have high uncertainty and appear inconsistent with other lines of evidence (KB). Harvest strategies that appear sustainable over the next three polar bear generations under a state-dependent approach could lead to subpopulation depletion or extirpation under a fixed-level approach that removes the same number of bears annually without reassessment.

In addition to regular, periodic surveys to estimate subpopulation size and vital rates, we recommend more frequent but less intensive monitoring of sea-ice habitat, movement and habitat use, nutritional condition, and reproductive indices based on research and harvest data, and Traditional Ecological Knowledge. Continuous genetic monitoring of the harvest to detect recoveries of genetically marked animals, and improving the accuracy of harvest reporting, are also needed. Systematic analysis of all harvest data, especially for the BB subpopulation, could provide complimentary estimates of harvest rate and other demographic parameters. During the intervening years between scheduled subpopulation surveys, these monitoring programs may provide a mechanism to detect sudden shifts in environmental conditions or subpopulation status that might necessitate a change in harvest strategy (e.g., a shortening of the management interval). Also, these monitoring programs will provide information essential for designing and implementing periodic subpopulation assessments. We suggest that future subpopulation assessment protocols consider the use of integrated population models, which can analyze data from multiple sources (e.g., different types of research, harvest, and Traditional Ecological Knowledge) in a unified framework, potentially leading to improved assessments of overall subpopulation status.

For the BB and KB subpopulations, the harvest sex ratios in recent decades, based on genetic sex determination, indicate weaker selection for males compared to the reported sex ratio and the management goal of a 2:1 male-to-female ratio. In most of our simulations, a harvest strategy with $SR = 2$ (instead of the lower *status quo* values of SR) did not result in higher harvest rates that met management objectives. This is because, for both subpopulations, estimates of S^* were lower for males than females, and females currently comprise approximately 70% of independent bears. These factors, if combined with a strongly male-selective harvest, often led to the severe depletion of adult males in our simulations, which had negative effects on reproduction and increased the probability of extirpation due to Allee effects in the mating system. These findings do not provide evidence against the conservation value of sex-selective harvest for polar bears. Rather, they indicate that depletion of males may be an emerging conservation concern for the BB and KB subpopulations. Given the current regulation of harvest in Canada (Nunavut) based on a sex 2:1 male-to-female sex ratio, it is important to further investigate this issue and particularly the lower estimates of survival for male bears. Our finding of skewed sex ratios in both subpopulations, despite an overall harvest that may not be strongly selective, suggests that these lower survival rates have a biological basis. Concurrent monitoring of the sex ratio in the harvest, the sex and age composition of the subpopulation, and the litter production rate, are necessary to determine the extent to which reduction of male bears could negatively affect the productivity of the BB and KB subpopulations.

Literature Cited in Executive Summary

Regehr, E. V., R. R. Wilson, K. D. Rode, M. C. Runge, and H. L. Stern. 2017. Harvesting wildlife affected by climate change: a modeling and management approach for polar bears. Journal of Applied Ecology doi:10.1111/1365-2664.12864.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear. 31 July 2016: x + 636 pp.

Introduction

Background on the Joint Commission, and the Baffin Bay and Kane Basin subpopulation reassessments

The Canada-Greenland Joint Commission on Polar Bear (JC) was established with the signing of a "*Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*" on 30 October 2009 (Anon. 2009). The primary objectives of this Memorandum of Understanding are to: "(1) to manage polar bear within the Kane Basin and Baffin Bay management units in order to ensure their conservation and sustainable management into the future, and, (2) establish an effective system of management which will include adhering to the principles of conservation". The JC subsequently established a Scientific Working Group (SWG) to provide scientific advice and recommendations with respect to the conservation and management of the Baffin Bay (BB) and Kane Basin (KB) polar bear subpopulations. In 2010, the JC tasked the SWG with using the best-available scientific information to:

(1) Propose Total Allowable Harvest (TAH) levels for the Kane Basin and Baffin Bay subpopulations.

(2) Provide science advice to the Joint Commission for monitoring the effects of habitat changes on polar bears.

The SWG reviewed the available scientific information and reported (SWG 2010) that for both subpopulations the most recent status updates indicated that 100% of population viability analysis (PVA) simulations, using current harvest levels, resulted in subpopulation declines after 10 years (PBSG 2010). The SWG also noted that PVA simulations are typically run 10–15 years beyond the point in time that abundance and vital rates were estimated. Given that the most recent estimates of demographic parameters for the BB and KB subpopulations were from the mid- to late 1990s (Taylor et al. 2005, 2008a; PBSG 2010), the SWG concluded that the available information was outdated and it was unknown whether demographic parameters had changed over time. These factors, in combination with the large-scale environmental changes in BB during recent decades, led the SWG to recommend that a high priority be given to developing new estimates of subpopulation abundance, subpopulation delineation, and vital rates (SWG 2010).

As a result, the JC tasked the SWG with evaluating various methods for assessing the number of polar bears in BB and KB (JC 2010). The SWG considered the pros and cons of physical CR, genetic CR, and aerial surveys; and concluded that physical CR was the preferred method to obtain robust ecological and demographic data (SWG 2011). However, due to the lack of support for physical CR among Inuit in Nunavut, and concerns that variability in sea-ice conditions can make it difficult to obtain accurate abundance estimates from aerial surveys, the JC recommended development of a 3-year research program based on genetic CR methods using biopsy darting. Following this recommendation, field research programs were conducted 2011–2014 as part of comprehensive reassessments of the BB and KB polar bear subpopulations (SWG 2016).

Estimation of sustainable harvest for polar bears

Historically, polar bear management was based on the assumption that sea-ice habitat was relatively stable over the long term and that, once subpopulation size (*N*) had been estimated, conservation could be achieved through harvest management (SWG 2011). Sustainable harvest, therefore, would depend on estimates of abundance and vital rates (e.g., probabilities of survival and reproduction), the harvest level, and the sex and age composition of the harvest. Early modeling suggested that sex-selective harvest at a rate of 4.5% of total population size, was sustainable for polar bears under optimal conditions (Taylor et al. 1987a, b). Recently, Regehr et al. (2015, 2017) also found that a 4.5% harvest rate, with a 2:1 male-to-female sex ratio in the harvest, was reasonable under many biological and management conditions, provided that population surveys were conducted periodically and harvest levels were adjusted when necessary. Regehr et al. (2017) also noted that sustainable harvest rates could be lower or higher than 4.5% under some conditions. This finding is consistent with recent studies that have provided a better understanding of how vital rates vary across subpopulations and change over time (e.g., Amstrup et al. 2001; Taylor et al. 2002, 2005, 2006; PBSG 2006, 2010; Regehr et al. 2007, 2010).

In recent decades, management and conservation of polar bears, particularly in Canada, have been informed by predictive modeling that incorporates subpopulation-specific vital rates.

The development of RISKMAN (Taylor et al. 2001), a stochastic population model, made it possible to perform detailed harvest assessments for multiple subpopulations, providing managers with a better understanding of the risk associated with different harvest strategies (Taylor et al. 2002, 2005, 2006, 2008a). However, RISKMAN was primarily intended to inform near-term management under stable conditions. It did not include a detailed model of density dependence, allow for future changes in environmental conditions or demographic parameters, or provide a way to directly assess how the frequency and intensity of subpopulation surveys can affect the risk of different management actions.

At present, the primary threat to polar bears throughout their range is the reduction in sea-ice habitat area, duration, and quality as a consequence of climate change (Derocher et al. 2004; Laidre et al. 2008, 2015; Wiig et al. 2015; Atwood et al. 2016). In 2009, the Polar Bear Range States (2015) agreed that the impacts of climate change constitute the most important threat to polar bear conservation and recommended that best management practices should “Consider the cumulative effects of climate change and human activities on polar bear subpopulations when making management decisions using tools such as predictive modeling”. Hence, in many situations sound harvest management will no longer rely solely on an estimate of abundance and a fixed annual harvest rate (e.g., 4.5%), or on predictive modeling that assumes stable conditions. The current demographic status of the world’s 19 polar bear subpopulations is variable (PBSG 2017) due to ecological variation, different rates of habitat change, and the influence of anthropogenic stressors (Vongraven and Peacock 2011; Atwood et al. 2016). Furthermore, although national and international agreements provide common standards for polar bear conservation, near-term management goals can vary across jurisdictions as a function of environmental, social, and other factors (Polar Bear Range States 2015). In light of this variability and the primary threat of habitat loss due to climate change, there is a need for improved risk assessment tools that can incorporate a broad range of environmental and direct human-caused factors to address specific management goals.

Harvest assessment for the Baffin Bay and Kane Basin subpopulations

Recent reassessments of the BB and KB polar bear subpopulations provided the ecological and demographic data necessary for harvest risk assessments (SWG 2016). However, the SWG was unable to provide harvest options as part of its final report because the JC had not provided specific guidance on (i) management objectives for each subpopulation, (ii) the expected frequency and intensity of future monitoring, and (iii) risk tolerance with respect to the effects of human-caused removals. Subsequently, the JC requested that the SWG propose TAH levels for both the BB and KB subpopulations based on the abundance estimates in SWG (2016), historical harvest levels, an expected 7–15 year frequency between subpopulation surveys, and that TAH be evaluated relative to the following management objectives (JC 2016):

1a) Maintaining a stable subpopulation at the current subpopulation estimate, with a low tolerance for the risk of declines below 90% of this level.

1b) Maintaining a stable subpopulation at the current subpopulation estimate, with a medium tolerance for the risk of declines below 90% of this level.

2a) A TAH that would ensure a maximum sustainable yield, with a low level of risk tolerance for the subpopulation declining below this level.

2b) A TAH that would ensure a maximum sustainable yield, with a medium level of risk tolerance for the subpopulation declining below this level.

Lacking further guidance, the SWG interpreted “low” and “medium” tolerance for the risk of a subpopulation decline below a specified level, to mean requiring a 90% or 70% probability of maintaining a subpopulation size above the specified level, respectively.

In addition to the management objectives above, the JC subsequently requested that the SWG explore possible methods to achieve a managed reduction of the BB subpopulation, from the current estimate of 2,826 animals (SWG 2016) to approximately 2,000 animals, over a 10–15 year period (JC 2017). Possible reasons for considering a managed reduction could be to reduce human-bear conflicts in circumstances where densities of polar bears have increased or nutritionally-stressed bears are increasingly coming in close proximity to humans, which may occur as sea-ice loss continues (Wilder et al. 2017).

In this report, we used recent estimates of abundance and vital rates (SWG 2016) in a matrix-based demographic model (adapted from Regehr et al. 2015, 2017) to evaluate TAH for the BB and KB polar bear subpopulations, relative to management objectives provided by the JC.

The demographic model is based on the life history of polar bears, and provides several advantages compared to other predictive modeling tools, including (i) an ability to incorporate the effects of a changing habitat (e.g., through a variable or declining *K*), (ii) a species-specific model of density dependence, which is important when evaluating the combined effects of habitat change and human-caused removals; (iii) an integrated model of Allee effects in the mating system, based on the work of Molnár et al. (2008, 2014); and (iv) a direct link between research and management actions, so that harvest strategies can be evaluated in light of the frequency and intensity of future subpopulation surveys.

For each subpopulation, we used data from SWG (2016) to develop alternative scenarios of the vital rates, which either represented plausible conditions for the current status of the subpopulation or provided a useful benchmark (e.g., for comparison with a subpopulation of "average" productivity). We considered multiple scenarios because estimates of demographic parameters from CR studies for polar bears can include uncertainty that is not reflected in the statistical distributions of the parameters, as well as multiple types of bias (Regehr et al. 2009; Chapter 3 in SWG 2016). To obtain robust results from predictive modeling it is important that demographic parameters be evaluated for biological realism and within the framework of other available information (e.g., Skalski et al. 2012).

Our analyses focused on harvest management strategies that were defined in terms of the harvest rate (percentage of the total subpopulation size removed per year), harvest level (measured in number of independent bears removed per year), sex and age composition of removed animals, management interval (number of years between successive subpopulation surveys and management changes), and the precision of demographic parameters estimated from subpopulation surveys. All harvest strategies were evaluated using a state-dependent (i.e., dependent on current conditions; Lyons et al. 2008) management approach, under which both the harvest rate and harvest level were updated periodically according to the management interval. State-dependent management has many features in common with the "adaptive management" approach recommended by the Range States (Polar Bear Range States 2015). It also has many advantages over other management approaches (e.g., a fixed-level harvest), both in terms of mitigating harvest risks and making it possible to maximize long-term yield (Regehr et al. 2017). Because our analyses incorporated multiple types of variability (e.g., statistical uncertainty and environmental variation), results are presented in a probabilistic manner that is consistent with management objectives provided by the JC.

The final results of this analysis are a series of potential harvest strategies for the BB and KB subpopulations. It is intended that these results help inform and guide subsequent decisions of the JC with respect to its determination of appropriate levels of harvest for these two shared subpopulations of polar bears.

Methods

Demographic and management model

We performed population projections using the matrix-based demographic model described in Regehr et al. (2015, 2017). The demographic model is based on the polar bear life cycle (Figure 1), with six female stages representing age and reproductive status, and four male stages representing age (Hunter et al. 2010; Regehr et al. 2010). Transitions between stages are defined by vital rates relative to a post-breeding census from the autumn or spring of year *t* to the autumn or spring of year *t* + 1, for the BB and KB subpopulations, respectively. Projections were referenced to independent bears (i.e., bears age ≥ 2 years that are not members of a family group) because cubs-of-the-year (C0) and yearlings (C1) are not included as individuals in the life cycle, but rather are used to define the reproductive status of an adult female (adult females with C0, stage 5; adult females with C1, stage 6). Projections were referenced to individual subpopulations and were not designed to consider immigration, emigration, or metapopulation dynamics. Unless otherwise noted, details of the projection model and its application follow from Regehr et al. (2015, 2017). A list of abbreviations, acronyms, symbols, and definitions is provided at the end of this report.

Density dependence

Including density dependence is necessary to evaluate the combined effects of habitat change and human-caused removals (Guthery and Shaw 2013). We constructed density-dependent curves of the vital rates (sample curves shown in Figure BB1) using methods and shape parameters described in Appendix S2 of Regehr et al. (2017). We assumed that available vital rates for the BB and KB subpopulations (section Vital rates) corresponded to an estimated subpopulation size at maximum net productivity level (MNPL; section Abbreviations, Acronyms etc.) under asymptotic population dynamics.

The matrix model was used to project hypothetical polar bear subpopulations forward over annual time steps starting at $t = 1$ (section Population projections). At each time step, density was determined as the sum of metabolic energetic equivalent (*mee*) values in the subpopulation, divided by carrying capacity (K; section Carrying capacity and environmental variation) expressed as energetic equivalents (Regehr et al. 2017). Under this approach, larger bears (e.g., adult males) occupied more energetic space and therefore had a greater density effect than smaller bears (e.g., subadult females). Regehr et al. (2017) found that individual variation in energetic requirements can influence population productivity because a given environment can generally support more females than males. We used *mee* values from Regehr et al. (2017) that were calculated from data on body mass and diet for the Chukchi Sea and Southern Beaufort Sea subpopulations (Table S2 in Regehr et al. 2017) because equivalent data were not available for the BB and KB subpopulations.

The demographic model incorporated a mechanistic submodel for Allee effects in the mating system, following the recommendation of Regehr et al. (2017). Molnár et al. (2008, 2014) proposed that, under some conditions, reproductive rates for polar bears may decline due to limitations in mate finding. Such declines can occur if adult males are depleted relative to adult females, which is possible under sex-selective harvest (McLoughlin et al. 2005; Taylor et al. 2008b); or if polar bear densities are low during the breeding season. Because Allee effects in the BB and KB subpopulations have not been studied directly, the submodel for Allee effects was based on equation 3 from Molnár et al. (2014) with input parameters for a "generic population". For both the BB and KB subpopulations, we calibrated the Allee submodel by calculating the degree of mating season aggregation that would result in a litter production rate equal to the estimated value from recent subpopulation studies (section Vital rates). This ensured that reproductive rates at $t = 1$ were equivalent to observed values. In subsequent years of projections ($t = 2, 3, \ldots$), the estimate of litter production rate from the Allee submodel was standardized by dividing by its value at $t = 1$. The resulting value was constrained to the interval [0,1] and used to modify the value of litter production rate (β_4) obtained from the density-dependent curves of the vital rates. Under this approach, a subpopulation that did not experience male depletion or significant reductions in abundance, did not experience declines in reproduction due to Allee effects.

For the BB subpopulation, we used an on-ice area of 656,000 km^2 (Stern and Laidre 2016) to calculate the densities of female and male bears available to breed at each time step (stages 4 and 10, respectively), which are inputs to the Allee submodel (Molnár et al. 2014). A mating season aggregation parameter of 0.43 resulted in a litter production rate of 0.93, as estimated from 2011–2013 field data (section Results). For the KB subpopulation, we used an on-ice area of 53,000 km^2 (Stern and Laidre 2016). We set the mating season aggregation parameter to 1.0 because the Allee submodel could produce a litter production rate of 0.71, as estimated from 2012–2014 field data (section Results), without a reduction in the effective subpopulation area.

Carrying capacity and environmental variation

Modeling wildlife populations under climate change required consideration of the effects of variability and trends in the environment (Boyce et al. 2006). We derived a proxy metric to represent potential changes in K using satellite data of sea-ice extent. We calculated separate metrics for the BB and KB subpopulations, based on the number of ice-covered days per year within the management boundary for each subpopulation (Chapters 4 and 9 in SWG 2016). We used the number of ice-covered days because it integrates spatial and temporal variation in sea-ice availability in a manner that is biologically relevant to polar bears (Stern and Laidre 2016). For each subpopulation, we fit a linear model to the time series of ice-covered days from 1979–2014. We then used the fitted model to project correlated values of ice-covered days forward in time, using methods of Gelman and Hill (2007) to simulate uncertainty in the slope coefficient and residual standard errors. Finally, we standardized the metric by dividing the projected values of ice-covered days at year $t = 1, 2, \ldots k$, by the fitted value at year $t = 1$. This resulted in a dimensionless metric (κ) representing proportional changes in K. During population projections, carrying capacity at year t, calculated as $K(t) = K(t = 1) \times \kappa(t)$, operated on vital rates through the density-dependent relationships.

The number of ice-covered days in the BB management area declined over the period 1979–2014 (slope = -1.22 days/year, SE = 0.23, $P < 0.001$), from fitted values of 245 days in 1979 to 203 days in 2014. This is a decline of approximately 5.5% per decade. During harvest assessment analyses for the BB subpopulation, projected values of κ were based on the estimated slope coefficient due to evidence for the effects of sea-ice loss on subpopulation ecology (SWG 2016). Therefore all population projections for BB included a proxy for *K* that varied from year-to-year, and declined by approximately 23% over three polar bear generations (section Population projections).

The number of ice-covered days in the KB management area declined over the period 1979–2014 (slope = -1.24 days/year, SE = 0.41, $P < 0.01$), from fitted values of 253 days in 1979 to 210 days in 2014. This is a decline of approximately 5.3% per decade. During harvest assessment analyses for the KB subpopulation, projected values of κ were based on a slope coefficient of 0, due to evidence for potential increases in productivity of the KB subpopulation associated with a transition from a multi-year ice region to seasonal ice conditions (SWG 2016). Therefore all population projections for KB included a proxy for *K* that varied from year-to-year, but remained stable over three polar bear generations (section Population projections). In other words, unlike for the BB subpopulation, projections for KB did not reflect the potential effects of long-term, decreasing trends in sea-ice cover.

In addition to density-dependent variation in the vital rates resulting from variation in *K*, we subjectively included additional density-independent variation as 25% of total uncertainty (i.e., temporal variation plus sampling uncertainty) in estimated vital rates, following the example of Taylor et al. (2002). Density-independent variation was implemented using the correlation matrix from Regehr et al. (2010), because that analysis estimated vital rates with a multistate CR model that was based on a life cycle graph similar to Figure 1.

Harvest and simulated population assessments

During population projections, harvest was implemented annually at a calculated level. Throughout our analyses the harvest level refers to the number of independent bears removed from a subpopulation by humans (i.e., the combination of subsistence harvest, sport hunting, removals of problem bears, defense kills, etc.). The calculated harvest level was updated every several years, according to the management interval (section Abbreviations, Acronyms etc.). To account for selectivity in human-caused removals and individual variation in the reproductive value of polar bears, harvest was implemented using stage-specific harvest vulnerability vectors. For females and males separately, we estimated harvest vulnerability by comparing the age structure of the harvest to the estimated age structure of the subpopulation (section Population initialization). For the BB subpopulation, age structure of the harvest in Canada was estimated using data from 805 bears for which age had been determined from counts of cementum annuli (Christensen-Dalsgaard et al. 2010) or from individual capture histories, during the period 1998–2013. Because age determination for polar bears is referenced to the spring, whereas stage transitions in the life cycle graph were referenced to autumn for BB, we subtracted 1 from the known age of bears harvested January through August. For example, a 4-year-old male bear in the spring of calendar year *t* remained a member of stage 8 (3 years) until it transitioned to stage 9 (4 years) in the autumn of year *t*. Age structure of the Greenland harvest was determined from 212 bears taken during the period 2012–2015. We assumed this sample was representative of the Greenlandic harvest because cementum ages were not available for other years. Overall harvest vulnerability vectors were derived by averaging the Canadian and Greenlandic vectors, weighted by the total reported harvest in each country 1998–2014 (Table 8.6 in SWG 2016). The resulting harvest vulnerability vectors for females (stages 1–6) and males (stages 7–10) were [0.93, 1.17, 1.10, 1.00, 0.00, 0.00] and [1.25, 1.34, 1.01, 1.00], respectively. During population projections for KB, we used the same harvest vulnerability vectors as were estimated for BB, because harvest data for the KB subpopulation were sparse and the two subpopulations are subject to similar harvest management regimes (Chapter 8 in SWG 2016).

At the beginning of each population projection (i.e., $t = 1$), the harvest level was calculated using the mean values of the vital rates and *N* as estimated from recent subpopulation surveys (section State-dependent management approach). This ensured that starting harvest levels reflected current data for the BB and KB subpopulations. At the beginning of each subsequent management interval, the harvest level was calculated using estimates of vital rates and *N* derived from simulated population assessments. Conceptually, the simulated population assessments represent new subpopulation surveys, performed in the future, to obtain updated data that can be used for management. The simulated population assessments included sampling uncertainty, for which the level and correlation structure were based on recent CR studies (SWG 2016). In other words, each successive simulated population assessment produced demographic parameters of similar precision to the most recent genetic CR studies for the BB and KB subpopulations (SWG 2016). Exceptions were made for some simulations, which included a modified level of precision in simulated population assessments (section Population projections).

This allowed us to evaluate the effects on harvest management of future survey methods that provide more precise or less precise estimates of *N* and the vital rates (section Simulations).

State-dependent management approach

We used a state-dependent management approach to calculate harvest level (Regehr et al. 2017) as a function of *N* and the intrinsic population growth rate (*r*, which depends on the vital rates) as follows:

$$H^{female}(t) = F_O \times \tilde{r}_{MNPL}(t) \times 0.5 \times \widetilde{N}(t) \qquad \text{[eqn 1]}$$

and

$$H^{male}(t) = H^{female}(t) \times SR \qquad \text{[eqn 2]}$$

where	H^{female}	is the number of females that can be removed annually;
	F_O	is a factor that directly adjusts the harvest rate to reflect management objectives and the risk tolerance of managers with respect to harvest;
	$\tilde{r}_{MNPL}$	is an estimate of the intrinsic population growth rate from subpopulation studies, referenced to population density at MNPL and selected as the 50th percentile of its sampling distribution;
	0.5	is a factor to calculate female removals assuming an equal sex ratio in the subpopulation, which serves to protect against excessive female removals when the male segment of a subpopulation is depleted;
	$\widetilde{N}$	is an estimate of *N* from subpopulation studies and selected as the 50th percentile of its sampling distribution;
	H^{male}	is the number of males that can be removed annually; and
	SR	is a factor that specifies the male-to-female ratio in removals.

To implement this state-dependent approach, managers must choose input values of the parameters F_O and *SR*. The parameter F_O directly influences the harvest rate: higher values lead to a higher harvest, which can eventually increase the risk of negative population outcomes (e.g., depletion). The parameter SR determines the sex ratio of the harvest.

In our analyses, values of F_O and SR remain constant for the duration of population projections, so that each harvest strategy had a consistent definition. In practice these parameters could be adjusted over time in response to changing biological or management conditions. In contrast, in our analyses the biological parameters in equations 1 and 2 (i.e., the true values of r_{MNPL} and N, as well as their estimated values $\tilde{r}_{MNPL}$ and $\widetilde{N}$) varied during population projections (e.g., N declined over time due to declining K). The notation for time (t) in equations 1 and 2 indicates that the estimated parameters $\tilde{r}_{MNPL}$ and $\widetilde{N}$ are updated periodically, as determined by the management interval. Equations 1 and 2 are written in terms of harvest level for convenience; the harvest rate (in this instance, referenced to the number of independent bears in the subpopulation) for females is the right side of equation 1 before multiplying by $\widetilde{N}$.

Management Objectives

We evaluated harvest relative to three management objectives provided by the JC (section Introduction; Table BB1).

Management Objective 1 was to maintain N above 90% of its starting value. Although we report results for this objective for both subpopulations, it was of limited value for BB because our analyses included projected trends in K that made it unlikely to meet Management Objective 1 even with no harvest (section Carrying capacity and environmental variation).

For Management Objective 2, we interpreted the language "…ensure a maximum sustainable yield" (JC 2016) as a desire to maintain a subpopulation size above MNPL relative to a potentially changing K (Management Objective 2 in Table BB1). To evaluate this objective we used a single value of MNPL corresponding to a subpopulation density (N/K) = 0.70, which is similar to the mean estimate of density at MNPL across a wide range of vital rates (Regehr et al. 2017). Using a single value of MNPL across all population projections, had the benefit of providing a consistent point of reference for management decisions.

Management Objective 3 was specific to BB, and reflected the goal of a managed reduction in total subpopulation size to 2,000 bears in 10–15 years. This corresponds to a reduction of approximately 30% relative to the mean estimate of 2,826 for the period 2012–2013

(SWG 2016). In Table BB1, we express this objective as a desire to maintain a subpopulation size above 70% of its starting value (Table BB1), noting that Joint Commission (2017) indicated that Management Objective 3 should also consider "…not achieving an abundance below the level that would produce maximum sustainable yield" as well as "…a potentially changing environmental carrying capacity". We interpret this as meaning that, to achieve Management Objective 3, the conditions for both Management Objectives 2 and 3 must be met.

In Table BB1 there are two versions of each Management Objective, reflecting "low" and "medium" levels of risk tolerance for not meeting the objective (section Introduction). We interpreted Management Objectives 1 and 2 as mid- to long-term objectives, and therefore evaluated them at the final time step $t = 36$ years, corresponding to approximately three polar bear generations in the future (section Population projections). Management Objective 3 was a short-term objective and was evaluated at $t = 15$. When reporting which harvest strategies met the management objectives, we included an additional condition requiring that the increased probability of a subpopulation being extirpated ($P_{extirpation}$) due to harvest, compared to an identical projection without harvest, not to exceed 0.05. In other words, this condition sought to ensure that harvest alone would not result in more than a 1-in-20 chance of extirpation. Applying this condition and Management Objective 2 together, over a sufficiently long time period, is consistent with the definition of "sustainable harvest" suggested by Regehr et al. (2107). Harvest strategies that met Management Objectives 1 and 2 generally were not associated with a high probability of extirpation, which meant that the condition on $P_{extirpation}$ had only a minor influence on the results. The exception was for some harvest strategies associated with Management Objective 3 (section Results).

Vital rates

We parameterized the matrix-based projection model using estimates of vital rates for the BB and KB subpopulations from recent genetic CR studies (SWG 2016). The published vital rates were adapted to the matrix-based projection model using methods described in Appendix S1 of Regehr et al. (2017).

For both the BB and KB subpopulations, litter production rate (equivalent to the parameter β_4 in Figure 1, for bears age ≥ 5 years) was not reported in SWG (2016) due to uncertainty in the age of bears that were observed from the air but not physically captured, and therefore did not provide a tooth for subsequent age determination. Also, there was uncertainty in the sex of some animals for which genetic samples were not obtained. We used simulation methods to estimate reproductive parameters for each subpopulation in a manner that accounted for this uncertainty. First, we created 10,000 bootstrap datasets by resampling, with replacement, from the 2010s field data. In each bootstrap dataset, animals with known reproductive status (i.e., adult females with dependent young), or of known age (from counts of cementum annuli) and known sex (from genetic analysis of tissue samples), were deterministically assigned to a life cycle stage. For animals of known sex but unknown age, a value for numeric age was sampled from a multinomial distribution created for bears of the same field-estimated age class. The multinomial distribution used probabilities calculated from Table B3 in SWG (2016), which compares sex and age classes as estimated from the air, with known sex and age for bears observed in BB and KB during the period 2011–2014. For animals of unknown sex, a similar procedure was used that considered both sex and age, with probabilities calculated from Table B2 in SWG (2016). For each bootstrap dataset, we estimated mean litter production rate as the number of females with C0 (stage 5) in year $t + 1$ divided by the product of adult female survival and the number of females available to breed (stage 4) in year t, taking into account annual sample sizes (Taylor et al. 1987b). The number of bears in each stage was calculated directly from the field data because CR modeling did not identify differences in recapture probabilities among female bears (Chapters 5 and 10 in SWG 2016). Standard error in the reproductive parameters was estimated as the standard deviation of point estimates from the 10,000 bootstrap datasets.

Baffin Bay

During recent subpopulation studies for BB, there was uncertainty and concern about bias in estimates of survival probability, particularly during the period 2011–2013 (Chapter 5 in SWG 2016). Therefore, we considered three alternative scenarios for the vital rates of BB polar bears, which represented a potential range of conditions (i.e., from low to high) for the current status of the subpopulation. We performed population projections using the vital rates for all three scenarios (section Simulations) and attempt to provide guidance about which results are most applicable to management.

Reproductive parameters

All three scenarios of the vital rates for the BB subpopulation used reproductive parameters estimated from CR studies 2011–2013, based on field data described in Chapter 6 of SWG (2016), and using the methods described above. To calculate litter production rate we used $S = 0.95$, the estimate of total survival for females ≥ 2 years during the period 1998–2010 (Table 5.8 in SWG 2016). The resulting litter production rate for bears age ≥ 5 years (β_4) was 0.93, which suggests relatively high mating success and cub production. Other reproductive parameters were similar to values presented in Chapter 6 of SWG (2016), with minor differences in the point estimates and variances due to the simulation methods used here (Table BB1). The parameter β_4 is the most important breeding parameter in the matrix-based projection model (Hunter et al 2007). Due to the lack of age data for most observations in BB during the period 2011–2013, we were unable to estimate litter production rate for 4-year-old bears (β_3) with an acceptable degree of accuracy, and therefore used the value 0.10 for this parameter from Taylor et al. 2005. We set the value for β_5 to 0, because females that are observed with C0 in the autumn of year t are not able to subsequently lose their cubs, re-breed, and be observed with a new litter of C0 in year $t + 1$.

Scenarios for survival

We evaluated three scenarios for survival rates of BB polar bears, representing alternative hypotheses for the current status of the subpopulation. Scenario 1 used estimates of natural (i.e., unharvested) survival (S^*) for the period 2011–2013 (page 261 in SWG 2016). Scenario 1 represents the hypothesis that estimates of S^* for 2011–2013 are accurate, in contrast to the higher estimates for BB polar bears during the period 1998–2010 and the higher mean estimates for most other subpopulations (Appendix S1 in Regehr et al. 2017). This scenario suggests a relatively poor status for the BB subpopulation (section Results).

For Scenario 2, we calculated S^* from estimates of total survival (i.e., including harvest) and harvest reporting probabilities for the period 1998–2010 (Table 5.8 in SWG 2016), using the equations for natural survival on page 257 of SWG (2016). Scenario 2 represents the hypothesis that the 1998–2010 estimates of S^* for BB are accurate, whereas the 2011–2013 estimates were negatively biased. SWG (2016) proposed that such bias may occur due to heterogeneity in recapture probability, non-random patterns of temporary emigration, of other factors. Terminal bias (i.e., bias at the end of a time series) in survival estimates commonly occurs in CR studies of long-lived, mobile animals (Peñaloza et al. 2014). Similar to our rationale in using Scenario 2, other studies have elected not to use terminal estimates S^* in population viability analyses due to concerns about bias (e.g., Langtimm 2009).

Scenario 3 used hypothetical survival rates representing an "average" polar bear subpopulation. To derive mean estimates of S^* for Scenario 3, we started with the estimates of S^* from Scenario 1 for female bears, and set estimates of S^* for male bears equal to 99% of these values. This reflects observations from other case studies that natural survival is generally similar for females and males (Table S1 in Regehr et al. 2017). Next, we increased all estimates of S^* in equal increments, until they produced an intrinsic population growth rate (r) of 0.05 under deterministic and asymptotic population dynamics. The value $r = 0.05$ is the mean estimate across case studies for polar bears as reviewed in Appendix S2 of Regehr et al. (2017). For Scenario 3, we used an amount of sampling uncertainty equivalent to data precision level 3 in Regehr et al. (2017). This represents the 50th percentile of estimated sampling uncertainty in case studies for polar bears, and is therefore typical of recent studies for the species (Appendix S4 in Regehr et al. 2017). Scenario 3 provides a benchmark for comparison with scenarios 1 and 2; it does not represent a data-based hypothesis for the current status of the BB subpopulation.

Estimates of S^* corresponding to the three scenarios of the vital rates are presented in Table BB3. For each scenario, we used the matrix-based projection model to calculate basic demographic parameters under asymptotic population dynamics (Table BB4). These parameters provide a general sense of the capacity for subpopulation growth under each scenario, but do not fully describe how the subpopulations behaved in the demographic model, because the model includes multiple types of stochasticity and can produce transient dynamics.

Kane Basin

During recent subpopulation studies estimates of S^* were derived from CR and harvest data collected during the period 1992–2014 (Chapter 10 in SWG 2016). Due to sparse data, SWG (2016) did not consider estimation models that allowed for temporal variation in survival. This presents a challenge for harvest assessment because all available estimates of S^* represent average values over a 23-year period, and do not reflect potential changes in survival in recent years. Furthermore, exploratory population reconstruction using the time-constant estimates of S^* and observed harvest patterns, resulted in a declining subpopulation trend over the period 1998–2014 (section Population initialization). This is inconsistent with estimated increases in abundance of the KB subpopulation from CR modeling, from 224 (SE = 40) for the period 1995–1997, to 357 (SE = 92) for the period 2013–2014 (Chapter 10 in SWG 2016). SWG (2016) provide several caveats for the estimated increase in abundance, including potentially inconsistent sampling frames between the 1990s and 2010s. However, other lines of evidence from subpopulation ecology and Traditional Ecological Knowledge also suggest that the KB subpopulation is currently healthy and has been stable or increasing in recent years (SWG 2016). We considered two alternative scenarios of the vital rates for the KB subpopulation to reflect uncertainty in current subpopulation status.

Reproductive parameters

The two scenarios of the vital rates for KB used reproductive parameters estimated from CR studies 2012–2014, based on field data described in Chapter 21 of SWG (2016) and using the methods described above. To calculate litter production rate we used $S = 0.95$, the estimate of total survival for females ≥ 3 years during the period 1992–2014 (Table 10.3 in SWG 2016). The resulting litter production rate for bears age ≥ 5 years (β_4) was 0.71 (Table KB1). Because we were unable to estimate litter production rate for 4-year-old bears (β_3) using data from genetic sampling conducted 2012–2014, we set this parameter to 0 based on the finding by Taylor et al. (2008) that 4-year-old bears did not reproduce. We set the value for β_5 to 0.10 based on Regehr et al. (2010), which is the only study to directly estimate the probability that a female observed with C0 in the spring of year t, conditional on losing her cubs, will re-breed and produce a new litter of C0 in the spring of year $t + 1$. This likely had a minor effect on results, due to the relative unimportance of β_5 to population growth (Hunter et al. 2007).

Scenarios for survival

We evaluated two scenarios for survival rates of KB polar bears. Scenario 1 used un-modified estimates of S^* as reported on page 496 of SWG (2016). Scenario 1 represents the hypothesis that time-constant estimates of S^* are accurate and represent the current status of the

KB subpopulation, despite being too low to reproduce the estimated increase in abundance over the period 1998–2014 (section Population initialization). Scenario 1 suggests a relatively poor status for the KB subpopulation (section Results).

For Scenario 2, we modified values of S* for some sex and age classes until the vital rates were sufficiently high to reproduce the estimated increase in abundance over the period 1998–2014. Specifically, for female and male polar bears age ≤ 2 years, we created 10 equal-increment values of *S** from a minimum corresponding to the point estimate for that sex and age class, to a maximum corresponding to the estimate of *S** for bears age ≥ 3 years of the same sex. This approach retained the lower values of unharvested survival for males compared to females, a pattern that was apparent for both the BB and KB subpopulations (SWG 2016). We chose to modify values of *S** for bears ≤ 2 years, rather than for adults, because CR and dead-recovery sample sizes were small for younger bears (Table 10.1 in SWG 2016), which resulted in high sampling uncertainty and increased potential for bias (Pollock et al. 1990). We performed population reconstructions for each set of equal-increment values of *S**, to determine the magnitude of increases in *S** necessary to achieve a 50% probability of reproducing the estimated increase in abundance of the KB subpopulation (section Population initialization). Thus, Scenario 2 represents the hypothesis that estimates of *N* for the KB subpopulation are accurate and provide a valid basis for inference about the subpopulation’s capacity to grow and support harvest. Uncertainty in estimates of *S** for Scenario 2 was calculated from the relative standard deviations for Scenario 1, which meant that even though estimates of *S** for bears ≤ 2 years were increased under Scenario 2, the corresponding level of data precision was not improved.

Estimates of *S** corresponding to the two scenarios of the vital rates are presented in Table KB2. Basic demographic parameters, calculated under asymptotic population dynamics, are presented in Table KB3.

Population projections

We performed population projections to evaluate the dynamics of the BB and KB polar bear subpopulations and to investigate the effects of different harvest strategies. For a given projection, the main biological inputs were: a starting value of *N* (expressed as a number of independent bears); mean values from a scenario of the vital rates, referenced to MNPL;

estimates of sampling variation, process variation, and the correlation structure of the vital rates; a starting stage distribution; a starting subpopulation density, expressed as the ratio N/K; stage-specific *mee* values; and a stochastic projection of the dimensionless metric κ, representing future variation in K. The main management inputs were: a value of F_O for use in Equation 1 to calculate harvest rate; a value of *SR* for use in Equation 2 to calculate harvest level; stage-specific harvest vulnerability vectors for females and males; a value for the management interval; and *rsd.mod*, the modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty, which was applied to future subpopulation assessments (section Abbreviations, Acronyms, etc.).

Populations were projected 35 years into the future (i.e., from $t = 1, 2, \ldots 36$), which is equivalent to approximately three polar bear generations (Regehr et al. 2016). At each time step $t = 2, 3, \ldots k$, the following operations were performed. First, subpopulations were projected forward 1 year using a stage-structured matrix model: $\boldsymbol{n}(\mathrm{t}+1) = \boldsymbol{A}(\mathrm{t}) \times \boldsymbol{n}(\mathrm{t})$, where $\boldsymbol{n}(\mathrm{t})$ is a stage distribution vector representing the number of animals in each life cycle stage at time step t, and $\boldsymbol{A}(\mathrm{t})$ is a 10×10 projection matrix (Caswell 2001). Entries in $\boldsymbol{A}(t)$ were defined in terms of vital rates in the life cycle graph (Figure 1). Demographic stochasticity was not included, because it is considered relatively unimportant at subpopulation sizes typical of polar bears (White 2000). Second, harvest was allocated among stages using a multinomial distribution with the probability for each stage calculated as the product of its proportional stage distribution and harvest vulnerability vector. For some projections, selective harvest led to the depletion of bears in one or more stages. If the specified harvest level exceeded the number of bears in a stage, the excess harvest was applied to adult bears of the same sex (i.e., stages 4 or 10). If the specified harvest exceeded the total number of one sex, the excess harvest was applied to adult bears of the other sex. Third, subpopulation density was calculated by summing *mee* values across animals in the subpopulation, then dividing by the total *mee* values available at carrying capacity. The survival and reproductive rates corresponding to this density were determined from the density-dependent curves, with modifications applied to the parameter β_4 based on the Allee submodel. Fourth, these vital rates were subject to density-independent stochastic variation. Finally, the resulting vital rates were used to construct a projection matrix for the next time step $\boldsymbol{A}(t+1)$.

During population projections, we defined persistence as maintaining a subpopulation size greater than a pre-determined quasi-extinction threshold. We used a threshold of 100 independent bears for the BB subpopulation, which is similar to values that have been used for brown bears (Wielgus 2002). We used a threshold of 25 independent bears for KB, because it is a smaller subpopulation and 25 bears was likely high enough to avoid negative small-population dynamics resulting from demographic stochasticity (Morris and Doak 2002). We note that Regehr et al. (2017) used higher quasi-extinction thresholds, calculated as 15% of starting *N*. We did not follow that approach because our analyses incorporated an Allee submodel, which provided a mechanistic description of small-population dynamics that have been suggested as important for polar bears (Molnár et al. 2014). During projections, subpopulations that crossed below the quasi-extinction threshold were considered extirpated and could not recover.

Population initialization

For both BB and KB we performed exploratory population reconstruction, which consisted of retrospective projections that used historic biological and management conditions. This helped to evaluate the vital rates, establish reasonable initial conditions for the main population projections (e.g., a subpopulation composition that was consistent with the history of sex-selective harvest), and reduce transient dynamics in early years of projections (Caswell 2001).

Baffin Bay

We performed population reconstruction for the period 1998–2010 using vital rates from Scenario 2. We did not include 2011–2013 in the reconstruction due to concerns about bias in the survival estimates for those years (SWG 2016). For the population reconstruction, the mean value of starting *N* was set to 1,968 independent bears. This was calculated from the estimated total subpopulation size (i.e., including C0 and C1) of 2,826 for the period 2011–2013 (Chapter 5 in SWG 2016). We used this starting value because of potential bias in the 1993–1997 estimate of *N* due to limited geographic sampling in the 1990s (Chapters 3 and 6 in SWG 2016). Methods to convert from total subpopulation size, to the number of independent bears, are described below. Starting N/K (i.e., at $t = 1$) was set to 0.67, which corresponded to MNPL as estimated from the density-dependent curves of the vital rates for Scenario 2. Harvest was implemented at a fixed level of 162 independent bears per year, which was the mean reported harvest for the BB subpopulation during the period 1998–2010 (Table 8.4 in SWG 2016).

We used a male-to-female sex ratio in the harvest (*SR*) of 1.25 for population reconstruction. This value was derived by averaging Canadian and Greenlandic harvest sex ratios, weighted by the total reported harvest in each country. The Canadian sex ratio was calculated directly from hunter-reported sex as it agreed with the genetically-determined sex, which indicated that 0.34 of harvested bears were female (Chapter 8 in SWG 2016). The Greenlandic sex ratio was estimated separately for this analysis, due to apparent discrepancies between hunter-reported sex and genetically-determined sex (Chapter 8 in SWG 2016). Specifically, we used the genetic sex data from tissue samples collected from 77 polar bears harvested in Greenland during the period 2011–2013, to estimate that 0.53 of the reported harvest was female. Lacking genetic sex data for other years, we assumed that this proportion was representative of the Greenlandic harvest 1998–2010. Harvest was implemented using the stage-specific harvest vulnerability vectors for the BB subpopulation. For population reconstruction, we used a deterministic proxy metric for *K* that was based directly on the observed time series of ice-covered days 1998–2010. Other specifications for population reconstruction were the same as for the full suite of projections (section Simulations).

The reconstructed subpopulation, using vital rates from Scenario 2 and a fixed-level harvest of 162 bears per year, exhibited a relatively stable trajectory during the period 1998–2010. The observed population growth rate was 1.01 (SE = 0.10) per year as calculated using the methods of Humbert et al. (2009). The ending proportion of females in the subpopulation was 0.69. For comparison, the mean proportion of females in the BB subpopulation during the period 2011–2013 was 0.66, as calculated using sex- and age-specific estimates of abundance for bears age ≥ 2 years from the most-supported CR model (Chapter 5 in SWG 2016). The similarity between the proportions 0.69 and 0.66, and the ability of population reconstruction to produce plausible population dynamics (i.e., a stable subpopulation under the observed harvest), provide a degree of confidence in the vital rates of Scenario 2.

To obtain a starting stage distribution for the full suite of population projections, we adjusted the final stage distribution from the population reconstruction until the proportion of females was 0.66, keeping the within-sex stage distributions constant. This produced the 10-stage distribution vector [0.07, 0.05, 0.05, 0.17, 0.18, 0.14, 0.07, 0.05, 0.05, 0.17]. When combined with estimates of C0 and C1 litter size for Scenario 2, this produced a ratio of independent bears to total bears of 0.70. Therefore, all projections for the BB subpopulation started with a mean value of $N_{t=1} = 2{,}826 \times 0.70 \approx 1{,}968$ independent bears, where 2,826 is the estimated total subpopulation size for 2011–2013 from SWG (2016). To reduce transient dynamics, all projections started at a subpopulation density $N/K = 0.81$, the median estimated density at the final year of population reconstruction.

Kane Basin

We performed population reconstruction for the period 1998–2014, to evaluate the vital rates from Scenario 1 and to identify values of S^* for Scenario 2 (section Vital rates). For the population reconstruction, the mean value of starting N was set to 153 independent bears. This was calculated from the estimated total subpopulation size of 224 for the period 1995–1997 (Chapter 10 in SWG 2016). Methods to convert from total subpopulation size, to the number of independent bears, are described below. Starting N/K was set to 0.73, which corresponded to MNPL as estimated from the density-dependent curves of the vital rates for Scenario 1. Harvest was implemented at a fixed level of 8 independent bears per year, which is the mean harvest reported for the KB subpopulation during the period 1998–2013 (Tables 8.2 and 8.5 in SWG 2016).

We used $SR = 0.94$ for population reconstruction. This value was derived by averaging Canadian and Greenlandic harvest sex ratios, weighted by the total reported harvest in each country. The Canadian sex ratio was calculated directly from hunter-reported sex, which indicated that 0.33 of harvested bears were female (Chapter 8 in SWG 2016). The Greenlandic sex ratio was based on genetically-determined sex for bears harvested from the KB subpopulation during the period 2011–2014, which indicated that 0.53 of harvested bears were female (Chapter 8 in SWG 2016). Lacking genetic sex data for other years, we assumed that this proportion was representative of the Greenlandic harvest 1998–2014. Harvest was implemented using the stage-specific harvest vulnerability vectors that were calculated for the BB subpopulation.

For KB population reconstruction, we used a deterministic proxy metric for K with interannual variation based on the observed time series of ice-covered days 1998–2014, but with an increasing trend of approximately 3% per year. We subjectively included this trend in *K*, rather than using observed values of the sea-ice metric, because an increasing trend would be necessary to allow *N* to increase from 224 bears in 1995–1997 to 357 bears in 2012–2014 (i.e., $N_{t=17} / N_{t=1} \approx 1.6$; Chapter 10 in SWG 2016), conditional on vital rates that were sufficiently high to produce such an increase. Other specifications for population reconstruction were the same as for the full suite of projections (section Simulations).

An initial KB population reconstruction used vital rates from Scenario 1 and a fixed-level harvest of 8 bears per year. This resulted in a declining trajectory during the period 1998–2014, with an observed population growth rate of 0.98 (SE = 0.40) per year. The large variance in the observed growth rate was due primarily to high sampling uncertainty in vital rates for the KB subpopulation (Chapter 10 in SWG 2016). The fact that the growth rate was negative, despite an increasing proxy for *K*, suggests that a subpopulation with vital rates similar to Scenario 1 would be unlikely to support a harvest of 8 bears per year, even in the absence of density-dependent regulation.

Subsequent KB population reconstructions used the same conditions as described above, but with the estimates of $S*$ from Scenario 1 modified to include incremental increases in survival for bears age ≤ 2 years (section Vital rates). We found that an average proportional increase in $S*$ of 38% (i.e., $S^*_{modified} = 1.38 \times S^*_{original}$) produced an increasing trajectory during the period 1998–2014, with an observed population growth rate of 1.03 (SE = 0.33). This corresponded to a median increase in abundance of $N_{t=17} / N_{t=1} = 1.59$ (SE = 0.71), which is similar to the estimated increase in abundance for the KB subpopulation from 1995–1997 to 2012–2014. Therefore, Scenario 2 of the vital rates for the KB subpopulation included these modified estimates of $S*$ for bears ≤ 2 years (section Results).

The reconstructed subpopulation, using vital rates from Scenario 2 and a fixed-level harvest of 8 bears per year, produced an ending proportion of females in the subpopulation of 0.70. For comparison, the mean proportion of females in the KB subpopulation during the period 2012–2014 was 0.71, as calculated from sex- and age-specific estimates of abundance for bears ≥ 2 years from the most-supported CR model (Chapter 10 in SWG 2016). To obtain a starting stage distribution for the full suite of projections, we adjusted the final stage distribution from the population reconstruction until the proportion of females was 0.71, keeping the within-sex stage distributions constant. This produced the 10-stage distribution vector [0.06, 0.06, 0.05, 0.24, 0.16, 0.14, 0.05, 0.04, 0.03, 0.17]. When combined with estimates of C0 and C1 litter size for Scenario 2, this leads to a ratio of independent bears to total bears of 0.68. Therefore, all projections for the KB subpopulation started with a mean value of $N_{t=1} = 357 \times 0.68 \approx 244$ independent bears, where 357 is the estimated total subpopulation size for 2012–2014 from SWG (2016). To reduce transient dynamics, all projections started at a subpopulation density $N/K = 0.69$, the median estimated density at the final year of population reconstruction.

Simulations

We define a "simulation" as multiple replicates of a population projection, where each replicate has the same mean biological inputs and the same management inputs (section Population projections). For each simulation, we used a parametric bootstrap procedure that generated 250 correlated random samples of the input vital rates and starting value of N, for the purpose of representing sampling variation in the vital rates (White 2000). We subjectively included sampling variation as 75% of total uncertainty (i.e., temporal variation plus sampling uncertainty) following the example of Taylor et al. (2002). Samples of the vital rates were generated using either a multivariate beta distribution or a stretched beta distribution (Morris and Doak 2002), as described in Appendix S3 of Regehr et al. (2017). When vital rates were near the boundary conditions [0,1] and variances were large, shape parameters for the beta distribution occasionally could not be determined. When this occurred, we generated a sample for the vital rate in question using a truncated normal distribution. The correlation structure for sampling variation in the vital rates was informed by the most-supported CR model for the BB subpopulation (Table 5.7 in SWG 2016). Specifically, we used a correlation coefficient of 1 within the following sets of parameters, and a correlation coefficient of 0 between the sets: $[\sigma_1, \sigma_2, \sigma_3, \sigma_4, \sigma_5, \sigma_6]$, $[\sigma_7, \sigma_8, \sigma_9, \sigma_{10}]$, $[\sigma_{L0}, \sigma_{L1}]$, $[\beta_3, \beta_4, \beta_5]$, and $[N]$. This correlation structure was also used for simulated population assessments, based on the assumption that future estimation methods would be broadly similar to SWG (2016).

For each random sample of the vital rates and starting N, we ran 50 projections, each with a different stochastic projection of κ (section Carrying capacity and environmental variation). Also, the projections included stochastic, density-independent variation at each time step. Therefore, for each simulation the resulting 250 × 50 = 12,500 replicate projections reflected both sampling and environmental variation. Although 12,500 is a relatively small number of replicates for population viability analysis (White 2000), this number was computationally feasible (section Software) and gave reproducible results at the levels of precision we report.

For each subpopulation, we performed simulations designed to evaluate a range of biological conditions and harvest strategies relevant to polar bear management (see below). For each simulation, we report the probability of meeting management objectives, calculated as the number of replicates that met the corresponding population condition (Table BB1) divided by the total number of replicates. At specific time steps, we also report $P_{extirpation}$, defined as the proportion of replicates for which N declined below the quasi-extinction threshold at any time step prior to time step $t = k$; and the probability of male depletion ($P_{male.dep}$), defined as the proportion of replicates for which the number of adult males (stage 10) was below 50% of the quasi-extinction threshold at $t = k$. The metric $P_{male.dep}$ is relevant because lower values of $S*$ for males compared to females, combined with sex-selective harvest, led to severe depletion of adult male bears under some conditions. Finally, we report the median change in subpopulation size ($N_{t=k}$ / $N_{t=1}$), the median subpopulation density ($N_{t=k}$ / $K_{t=k}$), and the mean realized harvest level ($H_{t=k}$). These values were calculated over all replicates, including those that led to extirpation. The metric H is relevant because some simulations included declining N (to declining K or to high harvest) and declining r (due to Allee effects caused by male depletion), which led to declining values of H over time under state-dependent approach.

Baffin Bay

We performed a primary set of simulations for the BB subpopulation to evaluate sustainable harvest for the three scenarios of the vital rates. All primary simulations used a management interval of 15 years and *rsd.mod* = 1. For each scenario, we performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Five, 11, and 8 values of the management factor F_O for Scenarios 1, 2, and 3, respectively. These values of F_O corresponded to starting harvest levels that differed by 20 bears per year, and encompassed the estimates of maximum sustainable yield for each scenario based on asymptotic dynamics and non-selective harvest (section Results).

2. Three values of sex ratio in the harvest, corresponding to *SR* = 1.0, 1.25, and 2.0. The value of *SR* = 1.25 represents the current status quo for the BB subpopulation under the assumptions made during population reconstruction (section Population initialization). We included *SR* = 2 because it is a common management objective for polar bears (Taylor et al. 2008b). We included *SR* = 1 to evaluate non-sex selective harvest, which might be a strategy for managed population reduction.

We performed a secondary set of simulations using the vital rates for Scenario 2 only, which we considered the most likely representation of the current status of the BB subpopulation. The objectives were to evaluate the effects of changes in the management interval and the precision of data obtained from future subpopulation assessments. All secondary simulations used *SR* = 1.25. We performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Nine values of the management factor F_O, corresponding to starting harvest levels that differed by 10 bears per year, and encompassed the range of harvest that met management objectives for Scenario 2 during primary simulations.
2. Three values for the management interval corresponding to 10, 15, and 20 years.
3. Three levels of precision in subpopulation data, corresponding to *rsd.mod* = 0.5, 1.0, and 1.5.

We also performed several *post hoc* simulations that were focused on the issue of managed population reduction.

Kane Basin

We performed a primary set of simulations for the KB subpopulation to evaluate sustainable harvest for the two scenarios of the vital rates. All primary simulations used a management interval of 15 years and *rsd.mod* = 1. For each scenario, we performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Five and 9 values of the management factor F_O for Scenarios 1 and 2, respectively. These values of F_O corresponded to starting harvest levels that differed by 2 bears per year, and encompassed the estimates of maximum sustainable yield for each scenario based on asymptotic dynamics and non-selective harvest (section Results).

2. Two values of sex ratio in the harvest, corresponding to *SR* = 0.94 and 2.0. The value of *SR* = 0.94 represents the current *status quo* for the KB subpopulation under the assumptions made during population reconstruction (section Population initialization).

We performed a secondary set of simulations using the vital rates for Scenario 2 only, to evaluate the effects of changes in the management interval and the precision of data obtained from future subpopulation assessments. Scenario 2 was more useful for this investigation because, unlike Scenario 1, it led to non-zero harvest levels that met management objectives. All secondary simulations used *SR* = 0.94. We performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Nine values of the management factor F_O, corresponding to starting harvest levels that differed by 1 bear per year, and encompassed the range of harvest that met management objectives for Scenario 2 during the primary simulations.
2. Three values for the management interval corresponding to 10, 15, and 20 years.
3. Three levels of precision in subpopulation data, corresponding to *rsd.mod* = 0.5, 1.0, and 1.5.

Finally, we performed several *post hoc* simulations focused on the ramifications of high uncertainty in estimates of S^* for bears age ≤ 2 years.

Software

Computations were performed in the R computing language (version R 3.4.0; The R Project for Statistical Computing; http://www.r-project.org). Simulations were run the Amazon Elastic Compute Cloud (http://aws.amazon.com/ec2/) using an Amazon Machine Image for RStudio Server (RStudio 2016) developed by L. Aslett (http://www.louisaslett.com/RStudio_AMI/). Each simulation took approximately 60 minutes using a Memory Optimized r4.xlarge computing instance.

Results

Baffin Bay

Primary simulations

We performed a primary set of simulations to evaluate population dynamics and sustainable harvest, for three scenarios of the vital rates, over a period of three polar bear generations during which K declined. Projections used a 15-year management interval, a baseline level of data precision (i.e., *rsd.mod* = 1.0) based on recent subpopulation surveys (SWG 2016), and a mean starting subpopulation size $N_{t=1}$ = 1,968 (SE = 236) independent bears with a stage distribution and subpopulation density determined from population reconstruction (section Population initialization). Tables BB5–BB7 present the highest harvest strategies that met management objectives for each scenario of the vital rates (see below). The harvest rate (h) in these tables is presented as the percentage of total subpopulation size (i.e., subpopulation size including C0s and C1s) that is removed each year as independent bears, because this definition of h is commonly used in polar bear management. Strategies with lower values of F_O than appear in Tables BB5–BB7, but otherwise similar inputs (e.g., the same harvest sex ratio), also met management objectives. Some of the harvest strategies in Tables BB5–BB7 could result in the depletion of adult male bears; probabilities of extirpation, compared to projections with no harvest, that approach the upper limit of 0.05; or declines in the calculated harvest level over time. Detailed results from the primary simulations are presented in Appendix S1.

Scenario 1

Scenario 1 of the vital rates resulted in an asymptotic intrinsic growth rate at MNPL (r_{MNPL}) of 0.03 (Table BB4), suggesting a limited capacity for growth and low resilience relative to other polar bear subpopulations (Regehr et al. 2017). Statistical uncertainty in estimates of S^* for the period 2011–2013 (Table BB3) contributed to high uncertainty in the estimate of r_{MNPL}, with approximately 26% of its sampling distribution below 0 (i.e., corresponding to a negative intrinsic growth rate). Management Objective 1 was not achievable even in the absence of harvest, due to the combined effects of low r, declining K, and high uncertainty in the vital rates (Table BB5). Management Objective 2 could be met using F_O = 0 to 0.41, depending on the value of *SR* and risk tolerance. This corresponds to a starting harvest rate $h_{t=1}$ = 0 to 0.7%, and a starting harvest level $H_{t=1}$ = 0 to 20 bears per year. Upper limits on F_O that met Management Objective 3 were 0 to 1.22, depending on *SR* and risk tolerance. However, due in part to high uncertainty in the vital rates, harvest strategies at the upper end of this range simultaneously increased $P_{extirpation}$ toward the upper condition of 0.05 at t = 15, while being unlikely to reduce the median subpopulation size by 30% (Table S.BB1). A subpopulation similar to Scenario 1 would have little capacity to support harvest, and would risk a 2 to 3% chance of extirpation at t = 36 in the absence of harvest (Table S.BB1).

Scenario 2

Scenario 2 resulted in r_{MNPL} = 0.08 (Table BB4), suggesting a strong capacity for growth and relatively high resilience. Due to declining *K*, Management Objective 1 was either not achievable or only achievable with no harvest (Table BB6). Management Objective 2 could be met using F_O = 0.43 to 1.03, depending on *SR* and risk tolerance. This corresponds to a starting harvest rate $h_{t=1}$ = 3.6 to 5.7%, and a starting harvest level $H_{t=1}$ = 100 to 160 bears per year. At the upper end of this range, a harvest strategy using F_O = 0.92 and the *status quo* value of *SR* = 1.25 corresponds to $h_{t=1}$ = 5.7% and $H_{t=1}$ = 160 bears per year, which is similar to harvest of the BB subpopulation in recent decades (Chapter 8 in SWG 2016). A sample of replicates from population projections can help visualize the effects of this harvest strategy on subpopulation trajectories. Figure BB3 shows that the median subpopulation size declines in parallel with (but not faster than) declining *K*, which is a consequence of a state-dependent management approach that meets Management Objective 2. The color-coding in Figure BB3 identifies the potential for male depletion or subpopulation extirpation in later years (Table S.BB2). For replicates that experienced male depletion, reproductive rates declined due to Allee effects in the mating system. This reduced the subpopulation's capacity for growth and resulted in lower calculated harvest levels under the state-dependent approach. Figure BB4 illustrates these effects, for the same harvest strategy that was shown in Figure BB3. Other harvest strategies that were more selective for males (i.e., *SR* = 2), including some strategies that met Management Objective 2 (Table BB6), had higher probabilities of causing male depletion (e.g., up to 0.25 at t = 36; Table S.BB2). Evaluation of Management Objective 3 is presented with results from the secondary simulations (see below).

Scenario 3

Scenario 3 was based on hypothetical vital rates that resulted in $r_{MNPL} = 0.05$ (Table BB4), representing a subpopulation with average capacity for growth and resilience. This scenario provided a benchmark for comparison with the data-based Scenarios 1 and 2. Similar to Scenario 2, Management Objective 1 was either not achievable or only achievable with no harvest, due to declining K (Table BB7). Management Objective 2 could be met using $F_O = 0.53$ to 0.89, depending on SR and risk tolerance. This corresponds to a starting harvest rate $h_{t=1} = 2.1$ to 3.6%, and a starting harvest level $H_{t=1} = 60$ to 100 bears per year. Harvest strategies under Scenario 3 that used $SR = 2$ were less likely to cause male depletion than similar strategies under Scenario 2 (Table S.BB3), because under Scenario 3 values of S^* were similar for adult females and adult males. Upper limits on F_O that met Management Objective 3 were 0.66 to 1.40, depending on SR and risk tolerance. In some cases, the condition requiring that $P_{extirpation} < 0.05$ at $t = 15$, compared to a similar projection without harvest, was the limiting factor for Management Objective 3. In other words, a harvest strategy could achieve an acceptable level of risk with respect to the population condition $N_{t=15} > (0.7 \times N_{t=1})$ but also result in $P_{extirpation} > 0.05$. Harvest strategies that met Management Objective 3, in terms of both $N_{t=15} > (0.7 \times N_{t=1})$ and the condition on $P_{extirpation}$, were unlikely to reduce the subpopulation size by 30% (Table S.BB3). These findings suggest that a managed population reduction of 30% is unlikely to be achieved, within the guidelines for risk tolerance provided by the JC, when using a 15-year management interval for a subpopulation with vital rates and a level of data precision similar to Scenario 3.

Secondary simulations

We performed a secondary set of simulations for Scenario 2 to evaluate the effects of management interval and data precision. Table BB8 shows the highest harvest strategies that met Management Objective 2b as a function of these factors, illustrating the potential impact of different management conditions. For example, $h_{t=1}$ is 54% higher for a 10-year management interval and $rsd.mod = 0.5$ (which corresponds to a relative standard deviation of 0.01 in σ_4 due to sampling uncertainty), compared to a 20-year management interval and $rsd.mod = 1.5$ (which corresponds to a relative standard deviation of 0.03 in σ_4). Table S.BB4 provides detailed results for the simulations summarized in Table BB8. In Table S.BB4, results for the expected value of $H_{t=36}$ require additional explanation. It appears counterintuitive that $H_{t=36}$ is higher for a 20-year management interval than for a 15-year management interval. This occurs because the harvest strategies in Table S.BB4 are fairly aggressive, and can result in moderate degrees of male depletion and reduced capacity for growth at $t = 36$. Using a 15-yr management interval, these negative effects result in a reduced harvest level at the second subpopulation assessment, which occurs at $t = 32$. In contrast, using a 20-year management interval, the second subpopulation assessment does not occur until $t = 42$, which is beyond the duration of projections. The ramifications are that using a 20-year management interval (i) leads to higher probabilities of extirpation at $t = 36$, and (ii) would be expected to result in large reductions to the calculated harvest level at $t = 42$.

We used results from the secondary set of simulations to evaluate Management Objective 3, because the primary simulations suggested that achieving a subpopulation reduction, within the specified risk tolerance, would require a short management interval and improved data precision. In the secondary simulations, the harvest strategy with the highest harvest and best management conditions was $F_O = 1.15$, a 10-year management interval, and *rsd.mod* = 0.5. This corresponded to $h_{t=1} = 7.1\%$ and $H_{t=1} = 200$. For this strategy, the median value of $N_{t=15} / N_{t=1}$ was 0.86 at $t = 15$ (i.e., a median reduction of 14% in starting subpopulation size; Table S.BB4). Although this strategy met Management Objective 3b as stated in Table BB1, it did not reduce the subpopulation by 30%. Also, it led to a 0.38 probability that subpopulation size was below MNPL at $t = 15$, which exceeded the risk tolerance for the condition on maximum sustainable yield that was associated with Management Objective 3 (section Management Objectives).

Post hoc simulations

We performed two *post hoc* simulations to inform future considerations for managed population reduction. First, we identified a state-dependent harvest strategy (BB_S1) that came as close as possible to achieving a subpopulation reduction of 30% in 15 years, while meeting the population condition for Management Objective 3b as stated in Table BB1, but without the additional condition related to maximum sustainable yield. Harvest strategy BB_S1 used $F_O = 1.58$, $SR = 1.0$, a 5-year management interval, and *rsd.mod* = 0.5. Use of $SR = 1$ promoted subpopulation reduction by removing more females, compared to a sex-selective harvest. This strategy corresponded to $h_{t=1}$ = 8.7% and $H_{t=1}$ = 245. At t = 15, the median value of $N_{t=15}$ / $N_{t=1}$ was 0.75 (i.e., a median reduction of 25% in starting subpopulation size) and Management Objective 3b was met (Table S.BB5). The mean harvest level declined at each management interval, concurrent with declining subpopulation size (e.g., $H_{t=15}$ = 180 bears per year; Table S.BB5). Conceptually, BB_S1 represents a near-optimal harvest strategy that would require nearly continuous surveys and rapid management response.

For comparison with BB_S1, we evaluated a second harvest strategy (BB_S2) that used the same starting harvest level, but did not follow a state-dependent approach and used the *status quo* value *SR* = 1.25. Strategy BB_S2 applied a fixed-level harvest of 250 bear per year, for a period of 15 years, without new subpopulation assessments or changes to management during this period. At t = 15, the median value of $N_{t=15}$ / $N_{t=1}$ was 0.55, indicating that the subpopulation was depleted beyond the desired 30% reduction. Also, strategy BB_S2 did not meet Management Objective 3 and resulted in a 0.30 probability of male depletion and a 0.23 probability of subpopulation extirpation at t = 15 (Table S.BB5).

Kane Basin

Primary simulations

We performed a primary set of simulations to evaluate population dynamics and sustainable harvest, for two scenarios of the vital rates, over a period of three polar bear generations during which *K* remained stable. Projections used a 15-year management interval, the baseline level of data precision (i.e., *rsd.mod* = 1.0), and a mean starting subpopulation size $N_{t=1}$ = 244 (SE = 41) independent bears with a stage distribution and subpopulation density determined from population reconstruction (section Population initialization). Tables KB4–KB6 present the highest harvest strategies that met management objectives for each scenario of the vital rates. Strategies with lower values of F_O, but otherwise similar inputs, also met management objectives. Some of the harvest strategies in Tables KB4–KB6 could result in the depletion of adult male bears or increased probabilities of extirpation, compared to projections with no harvest, that approached the upper limit of 0.05. Detailed results for the primary simulations are presented in Appendix S1.

Scenario 1

Scenario 1 of the vital rates resulted in $r_{MNPL} = 0.01$ (Table KB3), suggesting a very limited capacity for growth and low resilience. Statistical uncertainty in estimates of S^* for the period 2012–2014 (Table KB2), especially for bears ≤ 2 years, contributed to high uncertainty in the estimate of r_{MNPL}, with approximately 29% of its sampling distribution below 0 (i.e., corresponding to a negative intrinsic growth rate). Due to low values of r and high uncertainty in the vital rates, most management objectives could not be met even with no harvest (Table KB4). This is illustrated by Figure KB1, which shows a sample of replicates from population projections with $F_O = 0$ (i.e., no harvest). Although the median N increases gradually over time, the subpopulation trajectories are highly variable. The color-coding in Figure KB1 indicates that male depletion is possible due to lower estimates of S^* for males compared to females (e.g., $P_{male.dep} = 0.10$ at $t = 15$; Table S.KB1). A subpopulation with vital rates similar to Scenario 1 would have little guarantee of supporting harvest, and would face a 2 to 4% chance of extirpation at $t = 36$ in the absence of harvest (Table S.KB1). The ramifications of high uncertainty in vital rates for the KB subpopulation were evaluated in the secondary and *post hoc* simulations (see below).

Scenario 2

Scenario 2 resulted in $r_{MNPL} = 0.05$ (Table KB3), suggesting a medium capacity for growth and resilience. Management Objectives 1a and 2a (i.e., the version of the objectives with low risk tolerance for not achieving the population condition; Table BB1) could not be met with no harvest, due to variability in subpopulation trajectories arising from uncertainty in the vital rates (Table S.KB2). Management Objective 1b could be met using $F_O = 0.31$ to 0.48, depending on the value of SR (Table KB5). This corresponds to a starting harvest rate $h_{t=1} = 1.7\%$ and a starting harvest level $H_{t=1} = 6$ bears per year, which is similar to recent harvest of the KB subpopulation (SWG 2016). Management Objective 2b could be met using $F_O = 0.21$ to 0.31, depending on the value of SR. This corresponds to a starting harvest rate $h_{t=1} = 1.1\%$ and a starting harvest level $H_{t=1} = 4$ bears per year. The harvest strategy that meets Management Objective 2 is lower than the strategy that meets Management Objective 1, because projections for the KB subpopulation included a stable trend in K (i.e., $K_{t=36} \approx K_{t=1}$) Therefore, at $t = 36$, Management Objective 1 requires a subpopulation size that is greater than $0.9 \times N_{t=1} = 0.90 \times (0.69 \times K_{t=1}) \approx 0.62 \times K_{t=1}$; whereas Management Objective 2 requires a subpopulation size that is greater than $0.70 \times K_{t=36} \approx 0.70 \times K_{t=1}$ (i.e., a lower subpopulation size). Harvest strategies that met Management Objectives 1 and 2 were associated with increases in median N of up to 21% at $t = 36$ (Table S.KB2). This indicates that uncertainty in the vital rates, rather than the mean values of the rates, was a limiting factor in meeting management objectives. At $t = 36$, the harvest strategies in Table KB5 were associated with probabilities of causing male depletion of up to 0.27, due in part to lower S^* of males; and increased probabilities of extirpation, compared to projections with no harvest, of up to 0.03 (Table S.KB2).

Secondary simulations

We performed secondary simulations for Scenario 2 to evaluate the effects of management interval and data precision. Table KB6 shows the highest harvest strategies that met Management Objective 1b as a function of these factors, indicating the potential impact of different management conditions on harvest strategies for the KB subpopulation. For example, the highest harvest strategy under improved management conditions (i.e., a 10-year management interval and $rsd.mod = 0.5$) corresponded to $h_{t=1} = 2.2\%$ and $H_{t=1} = 8$. This harvest rate is 57% higher than the rate for a 20-year management interval and $rsd.mod = 1.5$.

In contrast to the BB subpopulation, the highest starting harvest level for the KB subpopulation, under improved management conditions, was lower than the expected value of maximum sustainable yield based on asymptotic population dynamics (i.e., 13 bears per year; Table KB3). This finding is largely due to high uncertainty in estimates of S^* for bears age ≤ 2 years (Table KB3). The reason is that improved data precision (i.e., $rsd.mod = 0.5$) only reduced sampling variation for simulated population assessments that occur in the future. Therefore, all subpopulation trajectories were highly variable during the first management interval (i.e., for the first 10, 15, or 20 years) due to high uncertainty in the baseline vital rates for Scenario 2. Because we only evaluated strategies with time-constant values of F_O, the range of harvest strategies that met management objectives was constrained by high uncertainty in the currently-available data for the KB subpopulation. A consequence of this effect is that, for some harvest strategies, the calculated harvest level increased after the first management interval, and remained 1-2 bears higher than the starting value at $t = 15$ and 36 (Table S.KB3).

Post hoc simulations

We performed two *post hoc* simulations to explore the ramifications of high uncertainty in estimates of S^* for bears age ≤ 2 years. First, we reduced sampling variation in these estimates by 75%, which resulted in levels of uncertainty similar to bears age ≥ 3 years (e.g., the reduced relative standard deviation due to sampling uncertainty was 0.04 for the vital rate σ_1, which is equivalent to the un-modified value for σ_4). Conceptually, this permitted exploration of how a higher level of confidence in estimates of S^* for bears age ≤ 2 years might affect sustainable harvest. After reducing the uncertainty in estimates of S^*, we performed simulations under conditions similar to current harvest practices for the KB subpopulation (i.e., $SR = 0.94$, a 15-year management interval, and $rsd.mod = 1.0$) and identified the highest strategy that met Management Objective 1b. The resulting strategy (KB_S1) corresponded to $F_O = 0.80$, $h_{t=1} = 2.8\%$, and $H_{t=1} = 10$ bears per year (Table S.KB4). Interpretation of these results requires caution, because there was not an analytical basis for reducing uncertainty in estimates of S^* for bears age ≤ 2 years. Nonetheless, this *post hoc* simulation can provide guidance on what a sustainable harvest strategy might be, if it was assumed with an increased degree of confidence that survival rates of bears age ≤ 2 years have been sufficiently high to produce the estimated increase in abundance for the KB subpopulation.

Second, we explored uncertainty in estimates of S^* for bears age ≤ 2 years using a different approach. For all other simulations in this report, total uncertainty in the vital rates was partitioned as 75% sampling variation and 25% process variation, following the example of Taylor et al. (2002). Because time-constant estimates of S^* for KB bears were referenced to a 23-year period from 1992–2014, this partitioning may not be accurate for younger animals, which often exhibit higher interannual variation in survival compared to adults (Eberhardt 2002). We were not able to perform an analysis of variance components in S^* (e.g., Cooch and White 2016) due to sparse data. Therefore, for bears age ≤ 2 years, we subjectively repartitioned total uncertainty as 25% sampling variation and 75% process variation. After making this change, we performed simulations under conditions similar to current harvest practices for the KB subpopulation (i.e., $SR = 0.94$, a 15-year management interval, and $rsd.mod = 1.0$) and identified the highest harvest strategy that met Management Objective 1b. The resulting strategy (KB_S2) corresponded to $F_O = 0.64$, $h_{t=1} = 2.2\%$, and $H_{t=1} = 8$ bears per year (Table S.KB4).

Discussion

We used subpopulation data from SWG (2016) in a demographic model adapted from Regehr et al. (2017) to evaluate a suite of potential harvest strategies for the BB and KB polar bear subpopulations, relative to management objectives and risk tolerances provided by the JC.

Demographic and management model

Additional details of the demographic model are presented in Regehr et al. (2015, 2017), including caveats and topics for future work. In this report we integrated the demographic model with a mechanistic submodel of Allee effects in the mating system (Molnár et al. 2008, 2014). In simulations for both the BB and KB subpopulations, lower estimates of S^* for males than females, combined with a sex-selective harvest, produced stage distributions that were skewed toward females. If the demographic model did not incorporate Allee effects, such subpopulations could exhibit unrealistically high values of r (i.e., in excess of the theoretical r_{max} under asymptotic population dynamics) because most adults were female and litter production rates could potentially remain high even in the near-absence of adult males. This effect could be compounded by the fact that the model tracked subpopulation density in terms of metabolic energetic equivalents, which allowed a given resource base to support a larger number of females compared to males (section Density dependence). We suggest that deriving parameters of the Allee submodel for the BB and KB subpopulations, and validating model-based predictions against field data, are areas for future work.

The demographic model differed from RISKMAN (Taylor et al. 2001) in several ways, including its ability to model temporal changes in vital rates and K. Also, we used a correlation structure for sampling variation that was informed by the CR models used to estimate vital rates (Chapter 6 in SWG 2016), whereas RISKMAN assumes sampling errors are independent (Taylor et al. 2006). This can influence PVA results, with correlated vital rates generally resulting in more variable subpopulation trajectories and higher risks of negative outcomes. To illustrate, we used the demographic model to calculate an unharvested, asymptotic population growth rate (λ)

using vital rates for the BB subpopulation as reported in Taylor et al. (2005). Our estimate of $\lambda = 1.053$ (SE = 0.022) was similar to the geometric mean estimate of unharvested $\lambda = 1.055$ (SE = 0.011) in Taylor et al. (2005), but our estimated standard error was twice as large. We suggest that our approach is more consistent with recommended practices in population viability analysis (Morris and Doak 2002).

Application of the demographic model required several key assumptions. First, to create density-dependent curves of the vital rates it was necessary to specify a subpopulation density (N/K) at which the vital rates were estimated. In practice, it is not possible to directly estimate K for wildlife populations (e.g., Gerrodette and Demaster 1990). Therefore, we inferred that N/K corresponded to a subpopulation size in the vicinity of MNPL, based on evidence that harvest in recent decades had been near maximum sustainable yield (Regehr et al. 2017). If actual N/K corresponded to a subpopulation size below MNPL, our estimates of intrinsic growth rate (r) could be positively biased. That is, the value of r corresponding to the vital rates would be closer to r_{max}, whereas we assumed it was equivalent to r_{MNPL}. If actual values of N/K corresponded to a subpopulation size significantly above MNPL, bias would be in the opposite direction. Second, we initialized population projections at time step $t = 1$ at a subpopulation size close to MNPL. This assumed that the BB and KB subpopulations are currently not experiencing strong density-dependent suppression of demographic parameters, which could be inaccurate for BB given evidence of range contraction (Chapter 4 in SWG 2016) and links between sea-ice and nutritional condition and reproductive rates (Chapters 6 and 7 in SWG 2016). Although we placed low confidence in the low estimates of S^* for the BB subpopulation from 2011–2013 (Scenario 1; see below), it is possible these estimates partially reflected negative effects of sea-ice loss. In combination, the short duration of recent subpopulation assessments in BB and KB, statistical uncertainty and potential bias in demographic parameters, and interannual variation, precluded direct estimation of subpopulation density. Our modeling approach did not make purposefully conservative assumptions about current density effects, especially for the BB subpopulation, and therefore could have understated the current and future effects of sea-ice loss. We recommend that future predictive modeling include sensitivity analyses with respect to key assumptions (e.g., Zabel et al. 2006).

Polar bears are distributed throughout the circumpolar Arctic in 19 subpopulations (PBSG 2010). Their life history is dependent on sea ice (Laidre and Regehr 2017), which is used as a platform from which to hunt their primary prey, ringed seals (*Pusa hispida*) and bearded seals (*Erignathus barbatus*). Earlier sea-ice breakup and reductions in optimal ice habitat have been linked to reductions in polar bear body condition, survival, reproduction, and abundance in some subpopulations (Stirling et al. 1999; Regehr et al. 2007, 2010; Rode et al. 2012; Bromaghin et al. 2015; Lunn et al. 2016; Obbard et al. 2016). Additional studies have documented use of less optimal sea ice habitat in several polar bear subpopulations (e.g. Durner et al. 2009, Wilson et al. 2014, Laidre et al. 2015, McCall et al. 2015). Our population projections for the BB and KB subpopulations included environmental variation primarily through the proxy metric for *K*, which was calculated from remote-sensing data for sea ice. Sea-ice metrics from other case studies for polar bears are reviewed by Stern and Laidre (2016). We did not consider rapid, non-linear declines in *K* or potentially catastrophic ecological or demographic effects due to climate change (e.g., Derocher et al. 2013), although the modeling framework could readily be adapted to include such effects. If subpopulations experience negative density-dependent effects that are larger or more abrupt than represented by the proxy metric for *K*, or negative density-independent effects that occur rapidly with respect to the management interval (i.e., so that multiple years elapse before such effects are detected), the harvest strategies identified in this report might cease to meet management objectives, resulting in increased risk of negative outcomes. Following a state-dependent management approach with a relatively short management interval (e.g., 10–15 years) can mitigate such risks, because reductions in *N* and the vital rates, whatever their cause, could be detected in future subpopulation surveys, and harvest strategies adjusted accordingly. Population dynamics and harvest strategies for declining populations are reviewed in detail in USFWS (2016).

Management objectives

We evaluated Management Objectives 1 and 2 at $t = 36$, corresponding to three polar bear generations (Regehr et al. 2016) in the future, a common time reference for population projections (e.g., IUCN 2017). We also report results at $t = 15$ years to provide insight into near-term population dynamics and identify potential metrics for monitoring (e.g., the proportion of females, see below).

Management Objective 1, which desired to achieve $N \geq 90\%$ of its current value, is more relevant to harvest assessments when habitat is stable or increasing. Under conditions of declining *K*, it is not possible to meet Management Objective 1 over the long term, even in the absence of harvest. For example, population projections for BB only met Management Objective 1b under cessation of harvest, which resulted in transient subpopulation increases as *N* approached *K*, followed by declines as *N*/*K* reached 1 (Tables BB6 and S.BB2).

Management Objective 2, which desired to keep $N \geq$ MNPL with respect to a changing *K*, is more relevant to harvest assessments when habitat is declining. The goal is to maintain a constant ratio of *N*/*K*, such that subpopulation size and carrying capacity decline in parallel. If *N* remains far enough below *K* due to harvest (e.g., at MNPL), density effects are alleviated and there is a harvestable surplus. Under a harvest strategy that fulfills these conditions, long-term declines in *N* are driven primarily by declines in *K*. Regehr et al. (2017) proposed that such strategies are possible for polar bears, as long as habitat loss affects subpopulations primarily through density-dependent mechanisms (e.g., increased crowing and competition for limited resources), or if a state-dependent management approach is followed and the management interval is short enough to respond to density-independent reductions in *r* (e.g., reduced reproductive success because bears have insufficient time to hunt seals on the sea ice, regardless of density). Figure BB3 shows sample replicates from population projections that illustrate this concept, except toward the end of the projections when the probability of severe male depletion increases and causes reproductive failure (see below).

Management Objective 3 desired to achieve, but not exceed, a 30% reduction in *N* in 10–15 years, while maintaining subpopulation size above the level necessary to achieve maximum sustainable yield (i.e., above MNPL). Simultaneously meeting these two population conditions is likely not possible. That is because MNPL for polar bears occurs at approximately $N/K = 0.70$ (Regehr et al. 2017). Unless a subpopulation started at $N/K = 1$, a 30% reduction in *N* would necessarily result in a density $N/K < 0.70$ (i.e., below the subpopulation size that would produce maximum sustainable yield).

Harvest and subpopulation sex ratio

For both subpopulations, we performed projections with multiple values of sex ratio in the harvest (*SR*). Male-biased harvest is a common wildlife management and conservation tool (e.g., Mysterud 2011). For polar bears, seeking to harvest at $SR = 2$ (i.e., a 2:1 male-to-female ratio) is intended to protect adult females (Taylor et al. 2008b), which have the highest reproductive value (Hunter et al. 2007). For the BB and KB subpopulations, harvest data from recent decades suggested that *SR* = 1.25 and 0.94, respectively. These estimates were based on hunter-reported sex in Canada, which genetic testing suggests is highly accurate; and genetic sex determination of harvest samples from Greenland in the 2010s, which was assumed to represent the long-term sex ratio in the Greenlandic harvest due to apparent inaccuracies in hunter-reported sex (Chapter 8 in SWG 2016). For most of our simulations, harvest strategies that used *SR* = 2, instead of the lower *status quo* values of *SR*, did not result in higher harvest rates that met management objectives. This should not be interpreted as evidence against the conservation value of sex-selective harvest. Rather, it is a consequence of lower estimates of *S** for males than females in both subpopulations which, in conjunction with a sex-selective harvest, often led to the depletion of males. This had negative effects on reproduction via Allee effects in the mating system, translating into lower realized values of *r* and lower sustainable harvest. Taylor et al. (2008b) suggested that a 2:1 male-to-female harvest designed to achieve maximum sustainable yield, is unlikely to reduce the abundance or mean age of male bears to the point of reducing mating success. However, that analysis used equal survival rates for males and females, and the harvest rate was calculated relative to a different interpretation of maximum sustainable yield than is used here. Accurate monitoring of the sex ratio in the harvest, as well as the sex composition of the subpopulation and the litter production rate, are necessary to determine the extent to which reduction of male bears might affect the productivity of the BB and KB subpopulations. We suggest that it is important to investigate the analytical and biological reasons for lower estimates of *S** for male bears. Our finding that females comprise approximately 70% of independent bears in both subpopulations, despite harvest that may not be strongly selective for males, suggests that there is a biological basis for this finding, which could signal an emerging conservation concern and have demographic consequences not considered in our analyses.

We estimated stage-specific harvest vulnerability vectors for females and male using age data from the BB subpopulation (section Population initialization). Results suggested a slight preferential selection for juvenile bears of both sexes, compared to their representation in the subpopulation. For example, male two-year-olds (stage 7) were 25% more likely to be harvested compared to what would be expected based on their relative abundance. Strong selection against adult female bears with dependent young (stages 5 and 6) was consistent with the protection of family groups in BB (Chapter 8 in SWG 2016). All population projections assumed that harvest vulnerability will remain constant in the future. If this is not the case (e.g., if selection becomes stronger for adults and weaker for subadults), the harvest strategies that meet management objectives might change.

Baffin Bay

We suggest that demographic modeling results for both subpopulations should be interpreted within the context of other available information. Sea-ice habitat in the BB region significantly declined between the previous subpopulation assessment in the 1990s and the recent reassessment in the 2010s (Chapter 4 in SWG 2106). The length of summer (i.e., the number of days from sea-ice retreat in spring to sea-ice advance in fall) increased by 12 days/decade since 1979. The mean sea-ice concentration during June–October decreased by 4% per decade. The general pattern of melt occurs about 3–4 weeks earlier in the 2010s than in 1990s. In general, BB has incurred large changes in the sea-ice regime experienced by polar bears and this has resulted in habitat loss (Stern and Laidre 2016; SWG 2016), which has translated to biological changes in the subpopulation. BB bears used significantly lower sea-ice concentrations in winter and spring in the 2010s than the 1990s (SWG 2016). Adult female bears are significantly closer to land in all months than in the 1990s, except at the end of breakup (June–July) when they stay on offshore sea ice as long as possible, likely to maximize feeding. Arrival dates on Baffin Island in summer are one month earlier in the 2010s than in the 1990s, and therefore the amount of time bears spent on land has increased by 20–30 days since the 1990s. There is a significantly shorter maternity den duration in the 2010s and maternity dens occur at higher elevations and steeper slopes than maternity dens in the 1990s, likely due to reduced snow cover (Escajeda et al. 2017).

Body condition declined in BB between 1993 and 2013, and declines were in close association with the duration of the ice-free period and spring sea ice transition dates. Reproductive metrics indicate that, from 1993 to 2013, an annual index of C0 recruitment declined concurrent with a trend towards earlier spring sea-ice break-up (SWG 2016). There has also been a significant reduction in the size of the 2010s BB 95% kernel range (i.e., a measure of the area used by bears fitted with radiocollars) in all months and seasons compared to the range in the 1990s. The most marked reduction is a 60% decline in subpopulation range size in summer. With respect to movements across subpopulation boundaries, BB bears in the 2010s were significantly less likely to leave BB than in the 1990s. In particular, there was a reduction in the number of collared bears moving into Davis Strait and Lancaster Sound, apparently due to reduced winter sea-ice coverage. This suggests the BB subpopulation has become more discrete, with less exchange between it and other subpopulations.

For the BB subpopulation, litter production rate for females age ≥5 years (β_4) was estimated to be 0.93 (SE = 0.08) from field data collected 2011–2013. Taking into account statistical uncertainty, this value is similar to the values of 0.88 for bears age 5 years, and 1.00 for bears age ≥ 6 years, reported for BB by Taylor et al. (2005); and higher than the mean value of 0.80 for bears age ≥ 6 years across 11 other subpopulations (range = 0.44 to 0.98; Table S1 in Regehr et al. 2017). Relatively high litter production is consistent with our modeling assumption that, despite evidence for ecological change, the BB subpopulation is currently not experiencing strong density-dependent limitation in demographic parameters. We estimated β_4 directly from the sample of observed bears (section Methods), which was assumed to reflect the subpopulation because CR modeling did not identify differences in recapture probabilities (Chapter 5 in SWG 2016). However, during the autumn single adult females were more likely to be inland or at high elevations (SWG 2016), which could have led to heterogeneity in recapture probabilities that was not detected in the modeling process. If this was the case, single adult females could have been under-represented in the observation sample, which could lead to positive bias in estimates of β_4 because the number of single adult females appears in the denominator of the equation for litter production rate.

The three scenarios of the vital rates for BB corresponded to significant differences in subpopulation status (Table BB4) and therefore in harvest strategies. We placed less confidence in Scenario 1, because estimates of S^* for the period 2011–2103 were based primarily on three years of sampling, and bias in survival during the terminal years of a CR study is common when there is un-modeled heterogeneity in recapture probabilities or non-random temporary emigration from the sampling area (Peñaloza et al. 2014). Estimates of S^* for Scenario 2 were referenced to 1998–2010, a longer period that was bracketed by intensive sampling in the 1990s and 2010s, and throughout which research marks were returned in the harvest. Furthermore, population reconstruction suggested that a subpopulation with the vital rates from Scenario 2 could exhibit a stable trajectory over the period 1998–2010, when subject to the observed harvest of approximately 162 bears per year and observed variation in sea-ice conditions. We started the population reconstruction in 1998 at an assumed abundance of 2,826 (i.e., the estimate for the period 2011–2013), because SWG (2016) indicated that lower estimates of *N* from the 1990s included an unknown level of negative bias, and that trends in the size of the BB subpopulation could not be reliably determined. Our finding that the vital rates of Scenario 2 were capable of maintaining a stable subpopulation does not constitute evidence that this occurred. We did not complete a comprehensive suite of population reconstructions, and other combinations of factors (e.g., higher starting *N* and lower vital rates) might reproduce equally plausible histories. Nonetheless, the vital rates of Scenario 2 appear consistent with available information on the history of the BB subpopulation, to the extent this can be determined given uncertainties and potential biases in the data. Also, population reconstruction from 1998–2010 led to an ending proportion of females in the subpopulation that was similar to the value estimated from sex- and age-specific abundance estimates for the period 2011–2013.

For the reasons discussed above, we considered Scenario 2 to be the more likely representation of the current status of the BB subpopulation. Harvest strategies in Table BB6 that met Management Objective 2 are likely to satisfy the definition of sustainable harvest proposed by Regehr et al. (2017), which requires maintaining a subpopulation size above MNPL with respect to a changing *K*, and limiting the negative effects of harvest on persistence. For harvest strategies using the *status quo* value of *SR* = 1.25 and a 15-year management interval, the upper limits on present-day harvest rate ($h_{t=1}$) were 4.3 and 5.7% for "low" and "medium" risk tolerances, as stated by the JC. This corresponds to present-day harvest levels of up to 120 and 160 bears per year, respectively, which would be applied for a period of 15 years and then updated. This range encompasses current TAH of 132 for the BB subpopulation (SWG 2016). The sustainability of these harvest strategies is conditional on the input data and assumptions of our modeling approach, including (1) that Scenario 2, the most optimistic scenario of the vital rates, is an accurate representation of the current and future status of the BB subpopulation; and (2) adherence to a state-dependent management approach over the next 35 years, with a 15-year management interval and future subpopulation assessments that provide a level of precision similar to the 1998–2010 estimates of S^* (SWG 2016). The harvest strategy corresponding to "low" risk tolerance (i.e., $h_{t=1}$ = 4.3%) is associated with lower probabilities of male depletion and extirpation in later years of the projection. For the BB subpopulation, nearly all harvest strategies can be expected to require reductions in the harvest level over time, due primarily to declining *K*, but also potentially due to harvest if there are inaccuracies in the input data or our modeling approach. It is also possible that the harvest rate will decline over time due to Allee effects in the mating system or to density-independent reductions in *r* that were not considered in our analyses.

We included Scenario 3 of the vital rates, to provide a means of comparison with an "average" polar bear subpopulation. Table BB7 indicates that, using the *status quo* $SR = 1.25$ and a 15-year management interval, harvest strategies with $h_{t=1} = 3.6\%$ could meet Management Objective 2 at the "medium" level of risk tolerance. This corresponds to a present-day harvest level of up to 100 bears per year. The upper limit on $h_{t=1}$ was also 3.6% for a harvest strategy with $SR = 2$, which is lower than the historic standard 4.5% harvest rate when using a 2:1 male-to-female sex ratio, for subpopulations experiencing positive environmental conditions (Taylor et al. 1987a). This difference is partially due to our inclusion of a declining trend in *K* for the BB subpopulation. It also suggests that our demographic modeling approach, when used in conjunction with Management Objective 2 and a "medium" risk tolerance as stated by the JC, may be slightly more conservative than previous predictive modeling for polar bears.

For the BB subpopulation, the challenges of meeting Management Objective 3, as it was stated by the JC, were presented above. To inform future discussion of subpopulation reduction, we identified a harvest strategy that resulted in a 25% reduction in starting subpopulation size over 15 years, while remaining with the stated risk tolerance for not exceeding a 30% reduction (Management strategy BB_S1 in Table S.BB5). This strategy required $SR = 1$, a 5-year management interval, and improved precision in the vital rates estimated from future subpopulation surveys. The starting harvest rate was 8.7%, corresponding to a present-day harvest level of 245 bears per year. It is unlikely that the near-optimal management conditions required by this strategy are feasible in practice, suggesting that either the management objectives or risk tolerances associated with a managed subpopulation reduction require reconsideration. Another practical challenge of managed reduction is that harvest must be rapidly reduced from very high levels in early years, to much lower levels once the target subpopulation size has been achieved. The risks of not reducing harvest in this manner were demonstrated by harvest strategy BB_S2, which maintained a fixed-level harvest of 245 bears per year for 15 years, without new subpopulation assessments or adjustments to the harvest. That strategy resulted high probabilities of extirpation (Table S.BB5), emphasizing the critical importance of monitoring and responsive management under aggressive harvest regimens.

Kane Basin

The annual cycle of sea-ice habitat in KB has shifted from a largely year-round ice platform (>30% coverage in summer) to a cycle that resembles the seasonal ice ecoregion (Amstrup et al. 2008) with complete melt-out in summer (<5% coverage; SWG 2016). The KB subpopulation has responded to changing sea-ice conditions with broad movement and habitat use patterns that are more similar to those of bears in seasonal sea-ice ecoregions (e.g., expanded seasonal home ranges). Apparent improvement in body condition in the 2010s, and no evidence of changes in reproductive performance in KB between the 1990s and 2010s, may reflect natural variation or a response to long-term changes in sea-ice dynamics in KB (SWG 2016). These observed changes reflect general differences in habitat use of bears occupying the archipelago vs. seasonal ice ecoregions: bears inhabiting seasonal ice regions have larger and more variable home ranges as they temporally track sea ice, whereas bears in archipelago regions have smaller home ranges with less variation.

For the KB subpopulation, litter production rate for females age ≥5 years (β_4) was estimated to be 0.71 (SE = 0.16) from field data collected 2012–2014. Considering statistical uncertainty, this suggests reproductive success similar to, or slightly lower than, other subpopulations (Table S1 in Regehr et al. 2017). The two scenarios of the vital rates for KB corresponded to significant differences in subpopulation status (Table KB3) and therefore in harvest strategies. Scenario 1 was characterized by low capacity for growth and high uncertainty ($\lambda = 0.01$ [SE = 0.04]), due largely to low and uncertain estimates of S^* for bears age ≤ 2 years (Table KB2). We placed less confidence in Scenario 1 because survival estimates of young bears were based on very small sample sizes during the period 2012–2014 (e.g., < 4 C0 marked per year). Furthermore, population reconstruction suggested that a subpopulation with vital rates from Scenario 1 would exhibit a declining trend from 1998–2014, when subject to the observed harvest of approximately 8 bears per year and an increasing trend in K. This is inconsistent with the estimated increase in abundance from 224 in the 1990s to 357 in the 2010s (Chapter 10 in SWG 2016), and with other evidence for productivity of the KB subpopulation (Chapters 9, 12, and 13 in SWG 2016).

Scenario 2 for KB included modified values of S^* for bears age ≤ 2 years, as necessary to reproduce the estimated subpopulation trend between the 1990s and 2010s, keeping other vital rates (e.g., adult survival) identical to Scenario 1. Vital rates in Scenario 2 retained the same amount of sampling variation as Scenario 1. Therefore, trajectories during population reconstruction were highly variable, corresponding to an 80% probability that the ending subpopulation size was at least one bear larger than the starting subpopulation size (i.e., that $N_{2014} > N_{1998}$). This stochastic representation of the history of the KB subpopulation was slightly more pessimistic (i.e., more likely to correspond to a declining subpopulation) compared to findings in SWG (2016), which suggested that the probability of a positive subpopulation change between the 1990s and 2010s was 95%. For Scenario 2, the modified estimates of S^* for bears age ≤ 2 years (Table KB2) were lower than the corresponding estimates for BB (noting that survival has a different time reference for the two subpopulation based on spring vs. autumn sampling), and within the range of juvenile survival estimates for other subpopulations with spring sampling (Table S1 in Regehr et al. 2017). The estimate of $\lambda = 1.05$ for Scenario 2 was equivalent to the mean estimate of unharvested population growth rate for other subpopulations (Regehr et al. 2017). Considering other lines of evidence for increasing productivity in the KB region, we suggest that Scenario 2 is a plausible representation of the current demographic status of this subpopulation.

For Scenario 2 of the vital rates and using the *status quo* value of $SR = 0.94$ and a 15-year management interval, the highest harvest strategy that met Management Objective 1 at "medium" risk tolerance corresponded to $h_{t=1} = 1.7\%$ and $H_{t=1} = 6$ bears per year. This finding is conditional on the input data and assumptions of the modeling approach, including (1) that Scenario 2 is an accurate representation of the KB subpopulation; (2) that K will remain stable for the next 35 years; and (3) adherence to a state-dependent management approach over the next 35 years, with a 15-year management interval and future subpopulation assessments that provide a level of precision similar to the estimates of S^* in SWG (2016). Sustainable harvest strategies for KB exhibited slight increases in mean harvest level over time (Table S.KB2), due in part to stability in K. If increasing biological productivity in the KB region leads to increasing trends over time in K and the intrinsic population growth rate, sustainable harvest levels would be expected to increase as well.

The relatively low estimate of sustainable harvest for KB was largely due to high uncertainty in vital rates, particularly estimates of S^* for bears age ≤ 2 years. This is evidenced by comparing a harvest level of up to 6 bears per year, calculated from the stochastic model projections (see above), with the estimated maximum sustainable yield of 13 bears per year based on asymptotic population dynamics (Table KB3). To investigate further, we performed two *post hoc* simulations with alternative assumptions for sampling uncertainty. The first simulation, which reduced uncertainty in estimates of S^* for bears ≤ 2 years to match the level of uncertainty in other survival estimates, resulted in a sustainable harvest of up to 2.8% (10 bears per year) under Management Objective 1 with “medium” risk tolerance, when using a 15-year management interval. The second simulation, which reallocated total uncertainty between sampling and process variation, resulted in a sustainable harvest of up to 2.2% (8 bears per year) under the same management conditions. These simulations were relevant because it is difficult to obtain precise and accurate estimates of vital rates for small and remote subpopulations such as KB. Without such estimates, the options available to managers include (1) inferring subpopulation status and sustainable harvest based on data other than the estimated vital rates (e.g., by modifying some estimates of S^* based on other information, similar to Scenario 2 and the *post hoc* simulations); or (2) adopting a conservative harvest strategy (e.g., Taylor et al. 2002).

Considering all available ecological and demographic data for the KB subpopulation, we suggest that present-day harvest rates up to approximately 2.8% (10 bears per year) are unlikely to cause negative population outcomes, if coupled with effective monitoring under a state-dependent approach. Use of a 10-year management interval would reduce the risks of harvest associated with high uncertainty in the currently available vital rates. If the challenges of studying the KB subpopulation lead to continued difficulty in obtaining accurate and precise estimates of vital rates, despite increased survey efforts, supplementary monitoring that is more frequent but less intensive may be valuable. We suggest developing a suite of ecological and demographic indicators to monitor subpopulation status, including accurate information on the level and composition of the harvest, marine productivity, habitat use and availability, reproductive rates, and estimates or indices of subpopulation size (via aerial survey or CR).

Monitoring

All of the harvest strategies considered in this report require the existence of a coupled research-management system under which both the sustainable harvest rate and the harvest level are adjusted periodically, based on new scientific information from subpopulation surveys and other sources. For both subpopulations, shorter management intervals and more precise estimates of *N* and vital rates, can substantially reduce the risk of negative population outcomes associated with a given harvest strategy. Results from the secondary simulations can help managers balance trade-offs between monitoring frequency and intensity (and therefore cost), the sustainable harvest rate, and harvest risks (Tables BB8 and KB6).

In our simulations, the management interval corresponded to the exact number of years between changes to the harvest level. For example, during population projections a 15-year management interval meant that new simulated population assessments were completed, and changes to the harvest level implemented, every 15 years. In practice, time lags in the coupled research-management system will likely result in departures from this simplified representation. For example, even if on-the-ground changes to TAH are implemented every 15 years, each change might be based on data from subpopulation surveys that were completed 2–3 years earlier. Application of the findings in this report should consider major differences, if they exist, between the definition of the management interval in practice and the definition used here.

Periodically obtaining new estimates of *N* and the vital rates (which determine *r*) is a central feature of a state-dependent management approach (Regehr et al. 2017). These parameters can be difficult and expensive to collect for wildlife populations (e.g., Williams et al. 2002), although both field methods (e.g., genetic CR; SWG 2016) and analytical approaches (e.g., Bayesian implementation of multistate models; Lunn et al. 2016) continue to evolve. Our analyses highlight the challenges of using estimated demographic parameters in harvest assessments for polar bears, even when the parameters were obtained from well-designed CR studies (SWG 2016). We sought to address these challenges primarily through consideration of multiple scenarios of the vital rates, which were developed based on (i) the estimated vital rates, referenced to different time periods; (ii) insights into the magnitude and directionality of potential bias (e.g., Schaub et al. 2004; Peñaloza et al. 2014); (iii) population reconstruction; and (iv) comparison with other case studies for polar bears. Although the magnitude of bias is generally lower in estimates of survival from CR studies compared to estimates of abundance

(Williams et al. 2002), the ramifications of relatively small bias in survival (e.g., 1–2%) can be profound when survival rates are used for population projections (e.g., Regehr et al. 2009). Skalski et al. (2012) recommend that biological realism should serve as a fundamental check for estimated demographic parameters and trends, and that auxiliary information should be used whenever possible to validate the results from predictive modeling. Peacock et al. (2011) recommend that management decisions for polar bears also include assessments of changes in body condition, habitat, population, and genetic delineation, and simultaneous surveys on Traditional Ecological Knowledge and human dimensions. Most of these subjects were successfully addressed during recent studies in BB and KB (SWG 2016). In this report, analyses relied heavily on estimates of subpopulation abundance and vital rates from SWG (2016), with limited interpretation of these estimates for the purpose of developing alternative scenarios and *post hoc* simulations.

Detailed recommendations on how to improve future estimates of N and r for the BB and KB subpopulations are beyond the scope of this report. Analyses to optimize the design of CR studies are being conducted under implementation of the Circumpolar Action Plan for polar bears (Polar Bear Range States 2015). For BB and KB, we suggest considering CR studies with a modified sampling scheme (e.g., sampling every other year for a period of six years, rather than sampling annually for three years), which could be combined with ongoing, less-intensive monitoring (see below). Also, we suggest that future study plans consider collecting and analyzing multiple types of data under the framework of an integrated population model (e.g., Frederiksen et al. 2014). Integrated population models can offer benefits for precision, accuracy, and the number of relevant parameters that can be estimated. For example, Regehr et al. (In preparation) concurrently analyzed radiotelemetry and CR data, which allowed direct estimation of temporary emigration and likely reduced bias in estimates of survival. Integrated population models can lead to increased consistency among demographic parameters (e.g., such that estimates of survival from studies of individually-marked animals, are consistent with estimates of subpopulation trend from aerial surveys). Finally, these models could allow integration of multiple types of research data with information from the harvest, local observations, and Traditional Ecological Knowledge. This has the potential to provide an improved assessment of overall subpopulation status that represents multiple perspectives.

Together with the conclusions and recommendations in SWG (2016), our analyses highlight important ecological and demographic indices for monitoring the BB and KB subpopulations. Accurate knowledge of the number, and sex and age composition, of human-caused removals is critical to understanding the effects of harvest on a subpopulation. Incomplete harvest reporting can lead to subpopulation depletion and other undesired outcomes, including negative bias in estimates of S^* that result in pessimistic population projections and lower estimates of sustainable harvest. We recommend that all harvested bears for the BB and KB subpopulations be genetically monitored, to detect recoveries of animals that were genetically marked during research. When used in CR models, accurate recovery data provide important information on survival and can reduce bias compared to studies with live recaptures only (Kendall et al. 2013). Furthermore, a systematic analysis of the sex and age composition of all harvest data (i.e., not only from research-marked bears) could provide complementary estimates of harvest rate and other demographic parameters (Skalski et al. 2005). Although such analyses can be limited by low statistical power and untestable assumptions, these problems are reduced if sample sizes are large (e.g., for the BB subpopulation) and recent data are available from a comprehensive subpopulation assessment. We suggest that systematic analysis of harvest data can be a useful monitoring tool between subpopulation surveys.

Nutritional condition and reproductive rates should be monitored as key indicators of subpopulation productivity (e.g., Vongraven et al. 2012). Changes in these parameters may precede or occur at the same time as other demographic changes (e.g., declining survival). Estimated relationships between time series of reproductive rates (or any other vital rate) and environmental conditions can be used in population projections to evaluate the future effects of habitat loss (e.g., Hunter et al. 2010). When such relationships are available, this represents an empirical alternative to our approach of projecting future trends in K that operated on vital rates through the density-dependent functions.

For a subpopulation that is harvested near maximum sustainable yield, and therefore in theory should function at a density well below carrying capacity (e.g., $N/K = 0.70$), declining nutritional condition and reproductive rates may signal negative density-independent effects. Density-independent limitation can result in lower values of r_{MNPL}, thus reducing the sustainable harvest rate that is calculated from equations 1 and 2. Under strong density-independent limitation, continued harvest without adjustment could lead to predominately additive mortality, with the potential to accelerate subpopulation declines compared to what would be expected under sea-ice loss only (USFWS 2016). Given that there is uncertainty about the extent to which sea-ice loss affects polar bears via density-dependent vs. density-independent mechanisms (e.g., Rode et al. 2012; Lunn et al. 2016), a high priority should be placed on monitoring spatial and temporal changes in habitat availability (e.g., as estimated from remote-sensing data of sea ice; Stern and Laidre 2016) along with indices of nutritional status and reproduction. SWG (2016) presented multiple reproductive indices for the BB and KB subpopulations, and we suggest that the number of yearlings per adult female may be particularly useful because it integrates litter production rate and juvenile survival (Rode et al. 2014; Regehr et al. 2015). Furthermore, concurrent monitoring of reproduction and the proportion of females in the subpopulation, is important to detect potential declines in subpopulation productivity due low male survival, skewed sex ratios in the subpopulation, and potential Allee effects in the mating system. Our analyses highlight these issues as potential conservation concerns for both the BB and KB subpopulations.

Acknowledgements

This report was prepared with contract support for EVR from the Greenland Institute of Natural Resources (Nuuk), through support by a grant from the Danish Cooperation For Environment In The Arctic (DANCEA). H. Stern (University of Washington) provided sea-ice data. M. Runge (U.S. Geological Survey) provided advice on technical details of the demographic model. A complete list of acknowledgements for the research conducted in BB and KB can be found in SWG (2016).

Abbreviations, Acronyms, Symbols, and Definitions

Adult – A polar bear age ≥ 5 years.

Allee effect – In this report, Allee effects refer to changes in reproductive rates due to density effects in the mating system. Declining reproductive rates can occur if adult males are depleted relative to adult females, or if overall subpopulation density is low during the spring on-ice breeding season.

Capture-recapture (**CR) –** A type of research study in which animals are individually marked, often through biopsy darting to obtain a genetic sample, or chemical immobilization to apply a physical mark (e.g., ear tags and lip tattoos). Over multiple years, data on individually marked animals can be used to estimate abundance and vital rates.

Carrying capacity (***K***) **–** The maximum number of individuals in a subpopulation that can be supported by the environment. This limit reflects the availability of food, habitat, and other resources. In this report, K is measured in the number of independent bears. Within the demographic model K is converted to metabolic energetic equivalents for the purpose of tracking subpopulation density over time.

Coefficient of variation (CV) – Ratio of the standard deviation to the mean of a statistical distribution of values. The CV reflects the level of uncertainty in an estimate, compared to the value of the estimate.

Confidence interval (CI) – A range of values that describes the uncertainty surrounding an estimate. Estimates of abundance and vital rates are often accompanied by a 95% CI.

Cub-of-the-year (**C0) –** A polar bear cub less than one year of age. In the polar bear life cycle it is assumed that C0 are born on 01 January of each year.

Density dependence – Demographic processes that change the birth or death rates as subpopulation density (i.e., the number of individuals per unit of habitat) changes.

Dependent young – A polar bear age ≤ 2 years that is accompanied by its mother.

Extirpation – The functional extinction of a subpopulation, which occurs in the population projections when a subpopulation size falls below the quasi-extinction threshold. In our analyses, extirpation is an irreversible condition that cannot be recovered from once the quasi-extinction threshold is crossed.

Harvest – In this report, harvest refers to all types of human-caused removals (i.e., subsistence harvest, sport hunting, removal of problem bears, defense kills, etc.).

Harvest level (*H*) – The number of independent bears removed each year through harvest.

Harvest rate (*h*) – Percentage of the total subpopulation size (i.e., the number of all bears, including dependent young) that is removed as independent bears each year through harvest.

Harvest strategy – A particular set of management and research conditions that define how harvest is conducted within the context of a state-dependent management approach. A harvest strategy is specified by inputs to equations 1 and 2, which determine the level and sex ratio of the harvest; as well as by the management interval and the level of precision in subpopulation data.

Independent bears – Polar bears age ≥ 2 years that are not with their mothers. Includes all polar bears in a subpopulation except for yearlings, cubs-of-the-year, and dependent two-year olds.

Intrinsic population growth rate (*r*) – The intrinsic population growth rate in the absence of human-caused removals. The maximum intrinsic growth rate ($\boldsymbol{r_{max}}$) occurs at a low density relative to carrying capacity. The intrinsic growth rate at a density referenced to maximum net productivity level is denoted $\boldsymbol{r_{MNPL}}$. Both r_{max} and r_{MNPL} are unharvested, potential growth rates that provide measures of the resilience of a subpopulation.

$\boldsymbol{\kappa}$ – A dimensionless metric representing proportional changes in carrying capacity (K), calculated from the number of ice-covered days per year. During projections, carrying capacity at year t, calculated as $K(t) = K(t = 1) * \kappa\ (t)$, operated on vital rates through the density-dependent relationships.

Litter production rate ($\boldsymbol{\beta_4}$) – The proportion of adult females that are available to breed in year t, which produce a litter of cubs-of-the-year in year t+1.

Management interval (*mgmt.interval*) – Duration (in years) of the interval between successive changes to the harvest level based on new data from completed subpopulation surveys. For example, under a 10-year management interval, a harvest level would calculated in year $t = 1$ and then applied each year $t = 1, 2, ... 10$. During the later years of this period, a subpopulation survey would be completed to provide updated estimates of abundance and the vital rates. A new harvest level would be calculated using these data equations 1 and 2, and the new harvest level would be applied in each year $t = 11, 12, … 20$.

Management Objective – An overall goal for management of a subpopulation, as stated by the responsible management agencies. In this report, management objectives are presented as a desired population condition (e.g., maintaining a relatively stable subpopulation size) along with a risk tolerance for not meeting the population condition.

Maximum net productivity level (MNPL) – The subpopulation size that results in the greatest net annual increment in subpopulation numbers resulting from reproduction minus losses due to natural mortality. The value of MNPL depends on how density dependence operates in a subpopulation. Regehr et al. (2017) suggested that for polar bears MNPL occurs at approximately 70% of the maximum number of animals the environment can support on average (i.e., MNPL ≈ 0.70 × *K*).

Metabolic energetic equivalent value (*mee*) – The energetic requirements of an individual bear, expressed relative to the energetic requirements of an average adult female. Larger bears (e.g., adult males) have higher *mee* values that smaller bears (e.g., subadult females), and therefore occupy more "energetic space" and make a greater individual contribution to density effects.

$P_{extirpation}$ **–** The probability of extirpation for a subpopulation.

$P_{male.dep}$ **–** The probability of severe male depletion, defined as the number of adult males in a subpopulation (stage 10 in Figure 1) falling below 50% of the quasi-extinction threshold.

$P_{Objective}$ **–** The probability of meeting the population condition corresponding to a Management Objective as defined in Table BB1.

Population growth rate (λ) **–** The rate of change of subpopulation size, measured in numbers of individuals per unit time.

Population projection – A simulated process in which the matrix-based model is used to project the size and composition of a subpopulation forward over a certain number of annual time steps. Each projection was defined by a specific set of biological and management conditions.

Population reconstruction – In this report, population reconstruction refers to retrospective population projections that used historic biological and management conditions. Population reconstruction was used to explore the past performance of the BB and KB subpopulations.

Quasi-extinction threshold – The size below which a subpopulation is considered to be extirpated. Population viability analyses often use quasi-extinction thresholds that are larger than one animal, because at very low numbers there can be negative small-population dynamics that reduce viability and accelerate extirpation. In this report, the quasi-extinction thresholds were 100 and 25 independent bears for the BB and KB subpopulations, respectively.

Risk tolerance – The attitude toward risk of the responsible management agencies. In this report, risk tolerance is expressed as the required probability of meeting the population condition associated with a Management Objective (e.g., the required probability, as stated by managers, of maintaining subpopulation size above a desired level).

rsd.mod **–** A modifier on the baseline relative standard deviation (SD) of the vital rates due to sampling uncertainty. Using different values of *rsd.mod* in population projections, allowed evaluation of the effects of different levels of precision in the data obtained from future subpopulation surveys. For example, a projection with *rsd.mod* = 0.5 meant that simulated population assessments would produce estimates of the vital rates and subpopulation size with approximately 50% less sampling variation, compared to the actual amount of sampling variation for the corresponding scenario of the vital rates.

Scenario of the vital rates – A specific set of vital rates assumed to represent the current status of a subpopulation. In this report, multiple scenarios of the vital rates were considered because of uncertainty and potential bias in estimates of certain demographic parameters from CR studies for both the BB and KB subpopulations (SWG 2016).

Stage – Stages in the life cycle graph representing bears of different sex, age, and reproductive status (Figure 1).

Standard deviation (SD) – A statistical measure that quantifies the amount of variation of a set of numbers around the mean (i.e., average) value. A low standard deviation means that most numbers are very close to the mean.

Standard error (SE) – A statistical measure that quantifies the amount of variation associated with an estimated parameter. The standard error is the standard deviation of a parameter's sampling distribution (i.e., its probability distribution, as estimated from a random sample of data).

Sex ratio (S*R*) – A factor that specifies the male-to-female ratio in the harvest. For example, *SR* = 2 is equivalent to a 2:1 male-to-female sex ratio.

Simulated population assessments – Simulated subpopulation surveys that were performed during population projections, on a schedule according to the management interval. The simulated population assessments provided updated estimates of subpopulation size and vital rates, which were used in equations 1 and 2 to calculate an updated harvest level, which was applied for the subsequent management interval.

State-dependent management – An approach under which management actions are based on the current state (status) of the subpopulation. In this report, state-dependent management refers to a coupled research-management system under which both the harvest rate and the harvest level are adjusted periodically, based on new scientific information from subpopulation surveys.

Subadult – Independent polar bear aged 2–4 years

Subpopulation – One of the 19 polar bear subpopulations recognized by the International Union for the Conservation of Nature (in the present case the Baffin Bay and Kane Basin subpopulations).

Subpopulation size (*N*) – The number of bears in a subpopulation.

t – Annual time step in a subpopulation projection. Quantities labeled with a subscript *t* are referenced to a specific time step. For example, $H_{t=1}$ is a harvest level at year 15.

Vital rates – Demographic parameters such as reproductive rates and survival rates, which define transitions in the life cycle graph (Figure 1) and determine the composition and growth of a subpopulation.

Yearling (C1) – A polar bear cub between one and two years of age.

Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. *in* E. T. DeWeaver, C. M. Bitz, and L.-B. Tremblay, editors. Arctic Sea Ice Decline: Observations, Projections, Mechanisms, and Implications. American Geophysical Union, Washington, D.C., USA.

Amstrup, S. C., T. L. McDonald, and I. Stirling. 2001. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. Journal of Agricultural, Biological, and Environmental Statistics 6:221–234.

Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*, 9 pp.

Atwood, T. C., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. F. Bromaghin. 2016. Forecasting the relative influence of environmental and anthropogenic stressors on polar bears. Ecosphere 7:1-22.

Boyce, M. S., C. V. Haridas, C. T. Lee, and N. S. Demography. 2006. Demography in an increasingly variable world. Trends in Ecology & Evolution 21:141–148.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634-651.

Caswell, H. 2001. Matrix Population Models, 2nd edition. Sinauer Associates Inc., Sunderland, Massachusetts, USA.

Christensen-Dalsgaard, S. N., J. Aars, M. Andersen, C. Lockyer, and N. G. Yoccoz. 2010. Accuracy and precision in estimation of age of Norwegian Arctic polar bears (*Ursus maritimus*) using dental cementum layers from known-age individuals. Polar Biology 33:589–597.

Cooch, E. G., and G. C. White, editors. 2016. Program MARK: A Gentle Introduction. 16th Edition. Available from: http://www.phidot.org/software/mark/docs/book/.

Derocher, A. E., J. Aars, S. C. Amstrup, A. Cutting, N. J. Lunn, P. K. Molnar, M. E. Obbard, I. Stirling, G. W. Thiemann, D. Vongraven, O. Wiig, and G. York. 2013. Rapid ecosystem change and polar bear conservation. Conservation Letters 6:368-375.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, O. Wiig, E. DeWeaver, M. C. Serreze, S. Belikov, M. Holland, J. A. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25-58.

Eberhardt, L. L. 2002. A paradigm for population analysis of long-lived vertebrates. Ecology 83:2841-2854.

Escajeda, E., K. L. Laidre, E. W. Born, Ø. Wiig, S. Atkinson, M. Dyck, S. H. Ferguson, N. J. Lunn. 2017. Identifying shifts in maternity den phenology and habitat characteristics of polar bears (*Ursus maritimus*) in Baffin Bay and Kane Basin. Polar Biology DOI: 10.1007/s00300-017-2172-6

Frederiksen, M., J.-D. Lebreton, R. Pradel, R. Choquet, and O. Gimenez. 2014. Identifying links between vital rates and environment: a toolbox for the applied ecologist. Journal of Applied Ecology 51:71-81.

Gelman, A., and J. Hill. 2007. Data analysis using regression and multilevel/hierarchical models. Cambridge University Press, New York, New York, USA.

Gerrodette, T., and D. P. Demaster. 1990. Quantitative determination of Optimum Sustainable Population level. Marine Mammal Science 6:1-16.

Guthery, F. S., and J. H. Shaw. 2013. Density dependence: applications in wildlife management. Journal of Wildlife Management 77:33–38.

Humbert, J.-Y., L. S. Mills, J. S. Horne, and B. Dennis. 2009. A better way to estimate population trends. Oikos 118:1940–1946.

Hunter, C. M., H. Caswell, M. C. Runge, S. C. Amstrup, E. V. Regehr, and I. Stirling. 2007. Polar bears in the southern Beaufort Sea II – Demography and population growth in relation to sea ice conditions. U.S. Geological Survey Administrative Report, U.S. Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Hunter, C. M., H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. Ecology 91:2883–2897.

IUCN Standards and Petitions Subcommittee. 2017. Guidelines for Using the IUCN Red List Categories and Criteria. Version 13. Prepared by the Standards and Petitions Subcommittee. Available from http://iucnredlist.org./documents/RedListGuidelines.pdf

JC [Canada-Greenland Joint Commission on Polar Bear]. 2010. Request for review of survey methods. Letter from the Canada-Greenland Joint Commission on Polar Bear to the Scientific Working Group to the Joint Commission on Polar Bear, 26 October 2010.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2016. Response to the SWG from the Joint Commission re: Harvest risk analysis. Email correspondence from C. Ladanowski, 7 October 2016.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2017. Response to the SWG from the Joint Commission re: Harvest risk analysis. Email correspondence from C. Ladanowski, 20 February 2017.

Kendall, W. L., R. J. Barker, G. C. White, M. S. Lindberg, C. A. Langtimm, and C. L. Penaloza. 2013. Combining dead recovery, auxiliary observations and robust design data to estimate demographic parameters from marked individuals. Methods in Ecology and Evolution 4:828-835.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications 18:S97–S125.

Laidre, K. L., and E. V. Regehr. 2017. Arctic marine mammals and sea ice. Pages 516-533 *in* D. N. Thomas, editor. Sea Ice, 3rd Edition. Wiley-Blackwell, West Sussex, UK.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Langtimm, C. A. 2009. Non-random temporary emigration and the robust design: conditions for bias at the end of a time series. Pages 745–761 *in* D. L. Thomson, E.G. Cooch, and M. J. Conroy, editors. Modeling Demographic Processes in Marked Populations. Environmental and Ecological Statistics, Volume 3. Springer Verlag, New York, New York, USA.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range: impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302-1320.

Lyons, J. E., M. C. Runge, H. P. Laskowski, and W. L. Kendall. 2008. Monitoring in the Context of Structured Decision-Making and Adaptive Management. Journal of Wildlife Management 72:1683-1692.

McCall, A. G., A. E. Derocher, and N. J. Lunn. 2015. Home range distribution of polar bears in western Hudson Bay. Polar Biology 38:343-355.

McLoughlin, P. D., M. K. Taylor, and F. Messier. 2005. Conservation risks of male-selective harvest for mammals with low reproductive potential. Journal of Wildlife Management 69:1592–1600.

Molnár, P. K., A. E. Derocher, M. A. Lewis, and M. K. Taylor. 2008. Modeling the mating system of polar bears: a mechanistic approach to the Allee effect. Proceedings of the Royal Society B: Biological Science 275:217–226.

Molnár, P. K., M. A. Lewis, and A. E. Derocher. 2014. Estimating allee dynamics before they can be observed: Polar bears as a case study. PLoS ONE 9:e85410. doi:10.1371/journal.pone.0085410.

Morris, W. F., and D. F. Doak. 2002. Quantitative Conservation Biology: Theory and Practice of Population Viability Analysis. Sinauer Associates Inc., Sunderland, Massachusetts, USA.

Mysterud, A. 2011. Selective harvesting of large mammals: how often does it result in directional selection? Journal of Applied Ecology 48:827-834.

Obbard, M. E., M. R. I. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15-32.

Peacock, E., A. E. Derocher, G. W. Thiemann, and I. Stirling. 2011. Conservation and management of Canada's polar bears (*Ursus maritimus*) in a changing Arctic. Canadian Journal of Zoology 89:371-385.

Peñaloza, C. L., W. L. Kendall, and C. A. Langtimm. 2014. Reducing bias in survival under nonrandom temporary emigration. Ecological Applications 24:1155-1166.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 Agreement on the Conservation of Polar Bears. Available from http://naalakkersuisut.gl/~/media/Nanoq/Files/Attached%20Files/Fiskeri_Fangst_Landbrug/Polarbear%202015/CAP/CAP%20Book.pdf

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:1-97.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T.

D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2017. Summary of polar bear population status per 2017. Published 30 March 2017. Available from http://pbsg.npolar.no/en/status/status-table.html

Regehr, E. V., M. Ben-David, S. C. Amstrup, G. M. Durner, and J. S. Horne. 2009. PhD Dissertation. Chapter 4. Quantifying bias in capture-recapture studies for mobile species: a case study with polar bears. University of Wyoming, Laramie, Wyoming USA.

Regehr, E.V., N. Hostetter, R. Wilson, K. Rode, A. Von Duyke, G. Pendleton, and S. Converse. (In preparation). Estimating demographic parameters for mobile species with sparse data: a Bayesian capture-recapture model for polar bears in the Chukchi Sea.

Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117-127.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., K. L. Laidre, H. R. Akçakaya, S. C. Amstrup, T. C. Atwood, N. J. Lunn, M. Obbard, H. Stern, G. W. Thiemann, and Ø. Wiig 2016. Conservation status of polar bears (*Ursus maritimus*) in relation to projected sea-ice declines. Biology Letters 12:20160556, doi:10.1098/rsbl.2016.0556.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Regehr, E. V., R. R. Wilson, K. D. Rode, M. C. Runge, and H. L. Stern. 2017. Harvesting wildlife affected by climate change: a modeling and management approach for polar bears. Journal of Applied Ecology doi:10.1111/1365-2664.12864.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768-782.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and O. Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3-18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76-88.

RStudio. 2016. RStudio: Integrated Development for R. RStudio, Inc., Boston, Massachusetts, USA. Available from http://www.rstudio.com/

Schaub, M., O. Gimenez, B. R. Schmidt, and R. Pradel. 2004. Estimating survival and temporary emigration in the multistate capture-recapture framework. Ecology 85:2107-2113.

Skalski, J. R., M. V. Clawson, and J. J. Millspaugh. 2012. Model evaluation in statistical population reconstruction. Wildlife Biology 18:225-234.

Skalski, J. R., K. E. Ryding, and J. J. Millspaugh. 2005. Wildlife Demography: Analysis of Sex, Age, and Count Data. Elsevier Academic Press, San Diego, California, USA. 636 pp.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere 10:2027–2041.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2010. Report of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 18 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear. 31 July 2016: x + 636 pp.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294-306.

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987a. Modelling the sustainable harvest of female polar bears. Journal of Wildlife Management 51:811–820.

Taylor, M., F. Bunnell, D. DeMaster, R. Schweinsburg, and J. Smith. 1987b. ANURSUS: a population analysis system for polar bears (*Ursus maritimus*). International Conference on Bear Research and Management 7:117-125.

Taylor, M. K., J. Laake, H. D. Cluff, M. Ramsay, and F. Messier. 2002. Managing the risk from hunting for the Viscount Melville Sound polar bear population. Ursus 13:185–202.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667-1673.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

Taylor, M.K., Obbard, M., Pond, B., Kuc, M., and Abraham, D. 2001. RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations. Iqaluit: Government of Nunavut.

Taylor, M., M. Obbard, B. Pond, M. Kuc, and D. Abraham. 2006. A guide to using RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations, Version 1.9.003. The Queen’s Printer for Ontario. 58 pp.

USFWS [United States Fish and Wildlife Service]. 2016. Polar Bear (*Ursus maritimus*) Conservation Management Plan, Final. U.S. Fish and Wildlife, Region 7, Anchorage, Alaska, USA, 104 pp.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Vongraven, D., and E. Peacock. 2011. Development of a pan-Arctic monitoring plan for polar bears: background paper. Circumpolar Biodiversity Monitoring Programme, CAFF Monitoring Series Report No. 1, CAFF International Secretariat, Akureyri, Iceland.

White, G. C. 2000. Population viability analysis: data requirements and essential analyses. Pages 288-331 *in* L. Boitani, and T. K. Fuller, editors. Research Techniques in Animal Ecology: Controversies and Consequences. Columbia University Press, New York, New York, USA.

Wielgus, R. B. 2002. Minimum viable population and reserve sizes for naturally regulated grizzly bears in British Columbia. Biological Conservation 106:381-388.

Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

Wilder, J. M., D. Vongraven, T. Atwood, B. Hansen, A. Jessen, A. Kochnev, G. Yowrk, R. Vallender, D. Hedman, and M. Gibbons. 2017. Polar bear attaches on humans: implications of a changing climate. Wildlife Society Bulletin: DOI: 10.1002/wsb.783.

Williams, B. K., M. J. Conroy, and J. D. Nichols, editors. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA. 817 pp.

Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. Ecosphere 5:1-24.

Zabel, R. W., M. D. Scheuerell, M. M. Mcclure, and J. G. Williams. 2006. The Interplay Between Climate Variability and Density Dependence in the Population Viability of Chinook Salmon. Conservation Biology 20:190-200.

Tables and Figures

Table BB1. Potential Management Objectives for the Baffin Bay (1-3) and Kane Basin (1-2) polar bear subpopulations. Population size (*N*) and carrying capacity (*K*) are measured in the number of independent bears and referenced to an annual time step (*t*) during population projections.

Management Objective	Population condition	Required probability of meeting objective
1a	$N_{t=36} > (0.9 \times N_{t=1})$	≥ 0.90
1b	$N_{t=36} > (0.9 \times N_{t=1})$	≥ 0.70
2a	$N_{t=36} > (0.7 \times K_{t=36})$	≥ 0.90
2b	$N_{t=36} > (0.7 \times K_{t=36})$	≥ 0.70
3a	$N_{t=15} > (0.7 \times N_{t=1})$	≥ 0.90
3b	$N_{t=15} > (0.7 \times N_{t=1})$	≥ 0.30

Table BB2. Reproductive parameters for the Baffin Bay polar bear subpopulation estimated from field data collected 2011-2013. Dependent young are cubs-of-the-year (C0) and yearlings (C1). Adult females are ≥ 5 years. The mean and standard error (SE) we calculated using simulations methods described in the main text.

Parameter	Mean	SE
Litter production rate for adult females (β_4)	0.93	0.08
C0 per adult female	0.58	0.04
Proportion of adult females with C0	0.38	0.02
C0 litter size	1.55	0.04
C1 per adult female	0.35	0.03
Proportion of adult females with C1	0.24	0.02
C1 litter size	1.47	0.05

Table BB3. Estimates (mean and standard error [SE]) of unharvested survival (*S**) for three scenarios of the vital rates for the Baffin Bay polar bear subpopulation. The scenarios are described in the main text.

†The life cycle graph (Figure 1) does not include separate stages for cubs-of-the-year (C0) and yearlings (C1), but survival rates for these age classes contribute to transition probabilities between reproductive stages for adult females.

			Scenario 1		Scenario 2		Scenario 3	
Sex	Age class	Stage	Mean	SE	Mean	SE	Mean	SE
female	C0	†	0.88	0.06	0.88	0.06	0.88	0.06
female	C1	†	0.89	0.06	0.89	0.06	0.89	0.06
female	2-4 year	1-3	0.91	0.05	0.96	0.02	0.93	0.05
female	≥5 year	4-6	0.91	0.05	0.96	0.02	0.93	0.05
male	C0	†	0.88	0.06	0.88	0.06	0.88	0.06
male	C1	†	0.89	0.06	0.89	0.06	0.89	0.06
male	2-4 year	7-9	0.83	0.06	0.91	0.02	0.92	0.06
male	≥5 year	10	0.83	0.06	0.91	0.02	0.92	0.06

Table BB4. Demographic parameters (mean and standard error [SE]) for the Baffin Bay polar bear subpopulation, corresponding to the three scenarios of the vital rates, based on asymptotic population dynamics. The parameters are: unharvested population growth rate (λ); subpopulation density (i.e., N/K) corresponding to maximum net productivity level (MNPL); intrinsic population growth rate at MNPL (r_{MNPL}); intrinsic population growth rate at low population density (r_{max}); and maximum sustainable yield (MSY) measured in numbers of independent bears under non-selective harvest.

	Scenario 1		**Scenario 2**		**Scenario 3**	
Sex	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
λ	1.03	0.05	1.08	0.04	1.05	0.03
MNPL	0.72	0.04	0.67	0.03	0.70	0.03
r_{MNPL}	0.03	0.06	0.08	0.03	0.05	0.03
r_{max}	0.03	0.06	0.10	0.04	0.06	0.04
MSY	49	67	156	50	100	59

Table BB5. Summary of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 1 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., t = 1) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 1.0**			***SR* = 1.25**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
2a	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
2b	0.41	20	0.7%	0.36	20	0.7%	0.27	20	0.7%
3a	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
3b	1.22	60	2.1%	1.08	60	2.1%	1.08	80	2.8%

Table BB6. Summary of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., t = 1) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 1.0**			***SR* = 1.25**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
2a	0.78	120	4.3%	0.69	120	4.3%	0.43	100	3.6%
2b	1.03	160	5.7%	0.92	160	5.7%	0.60	140	5.0%
3a	1.03	160	5.7%	0.92	160	5.7%	0.60	140	5.0%
3b	1.16	180	6.4%	1.15	200	7.1%	0.78	180	6.4%

Table BB7. Summary of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 3 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 1.0**			***SR* = 1.25**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
2a	0.60	60	2.1%	0.53	60	2.1%	0.53	80	2.8%
2b	0.80	80	2.8%	0.89	100	3.6%	0.66	100	3.6%
3a	1.00	100	3.6%	0.71	80	2.8%	0.66	100	3.6%
3b	1.40	140	5.0%	1.24	140	5.0%	0.93	140	5.0%

Table BB8. Summary of secondary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Rows are different levels of *rsd.mod*, a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty. Columns are different management intervals. Values in the cells represent the upper limits that meet Management Objective 2b (Table BB1); where F_O is a factor to calculate the harvest rate, and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with SR = 1.25.

	Management interval (years)		
rsd.mod	**10**	**15**	**20**
	(Results reported as values of F_O)		
0.5	1.15	0.98	0.86
1.0	1.03	0.92	0.80
1.5	0.92	0.80	0.75
	(Results reported as values of $h_{t=1}$)		
0.5	7.1%	6.0%	5.3%
1.0	6.4%	5.7%	5.0%
1.5	5.7%	5.0%	4.6%

Table KB1. Reproductive parameters for the Kane Basin polar bear subpopulation estimated from field data collected 2012-2014. Dependent young are cubs-of-the-year (C0) and yearlings (C1). Adult females are ≥ 5 years. The mean and standard error (SE) we calculated using simulations methods described in the main text.

Parameter	**Mean**	**SE**
Litter production rate for adult females (β_4)	0.71	0.16
C0 per adult female	0.55	0.10
Proportion of adult females with C0	0.34	0.06
C0 litter size	1.64	0.10
C1 per adult female	0.22	0.06
Proportion of adult females with C1	0.17	0.04
C1 litter size	1.23	0.12

Table KB2. Estimates (mean and standard error [SE]) of unharvested survival (*S**) for two scenarios of the vital rates for the Kane Basin polar bear subpopulation. The scenarios are described in the main text.

†The life cycle graph (Figure 1) does not include separate stages for cubs-of-the-year (C0) and yearlings (C1), but survival rates for these age classes contribute to transition probabilities between reproductive stages for adult females.

			Scenario 1		Scenario 2	
Sex	**Age class**	**Stage**	**Mean**	**SE**	**Mean**	**SE**
female	C0	†	0.45	0.15	0.74	0.25
female	C1	†	0.74	0.15	0.87	0.15
female	2 year	1	0.74	0.15	0.87	0.15
female	3 year	2	0.97	0.04	0.97	0.04
female	4 year	3	0.97	0.04	0.97	0.04
female	≥5 year	4-6	0.97	0.04	0.97	0.04
male	C0	†	0.45	0.15	0.70	0.23
male	C1	†	0.54	0.17	0.74	0.23
male	2 year	7	0.54	0.17	0.74	0.23
male	3 year	8	0.90	0.06	0.90	0.06
male	4 year	9	0.90	0.06	0.90	0.06
male	≥5 year	10	0.90	0.06	0.90	0.06

Table KB3. Demographic parameters (mean and standard error [SE]) for the Kane Basin polar bear subpopulation, corresponding to the two scenarios of the vital rates, based on asymptotic population dynamics. The parameters are: unharvested population growth rate (λ); subpopulation density (i.e., N/K) corresponding to maximum net productivity level (MNPL); intrinsic population growth rate at MNPL (r_{MNPL}); intrinsic population growth rate at low population density (r_{max}); intrinsic population growth rate at low population density in the absence of human-caused removals (r_{max}); and maximum sustainable yield (MSY) measured in numbers of independent bears under non-selective harvest.

	Scenario 1		**Scenario 2**	
Sex	**Mean**	**SE**	**Mean**	**SE**
λ	1.01	0.04	1.05	0.06
MNPL	0.73	0.05	0.69	0.04
r_{MNPL}	0.01	0.04	0.05	0.06
r_{max}	0.02	0.05	0.07	0.06
MSY	3	6	13	13

Table KB4. Summary of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 1 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 0.94**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.00	0	0.0%	*NA*	*NA*	*NA*
2a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
2b	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*

Table KB5. Summary of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., t = 1) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 0.94**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.48	6	1.7%	0.31	6	1.7%
2a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
2b	0.32	4	1.1%	0.21	4	1.1%

Table KB6. Summary of secondary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Rows are different levels of *rsd.mod*, a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty. Columns are different management intervals. Values in the cells represent the upper limits that meet Management Objective 1b (Table BB1); where F_O is a factor to calculate the harvest rate, and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with SR = 0.94.

	Management interval (years)		
rsd.mod	**10**	**15**	**20**
	(Results reported as values of F_O)		
0.5	0.64	0.64	0.56
1.0	0.56	0.48	0.48
1.5	0.56	0.40	0.40
	(Results reported as values of $h_{t=1}$)		
0.5	2.2%	2.2%	2.0%
1.0	2.0%	1.7%	1.7%
1.5	2.0%	1.4%	1.4%

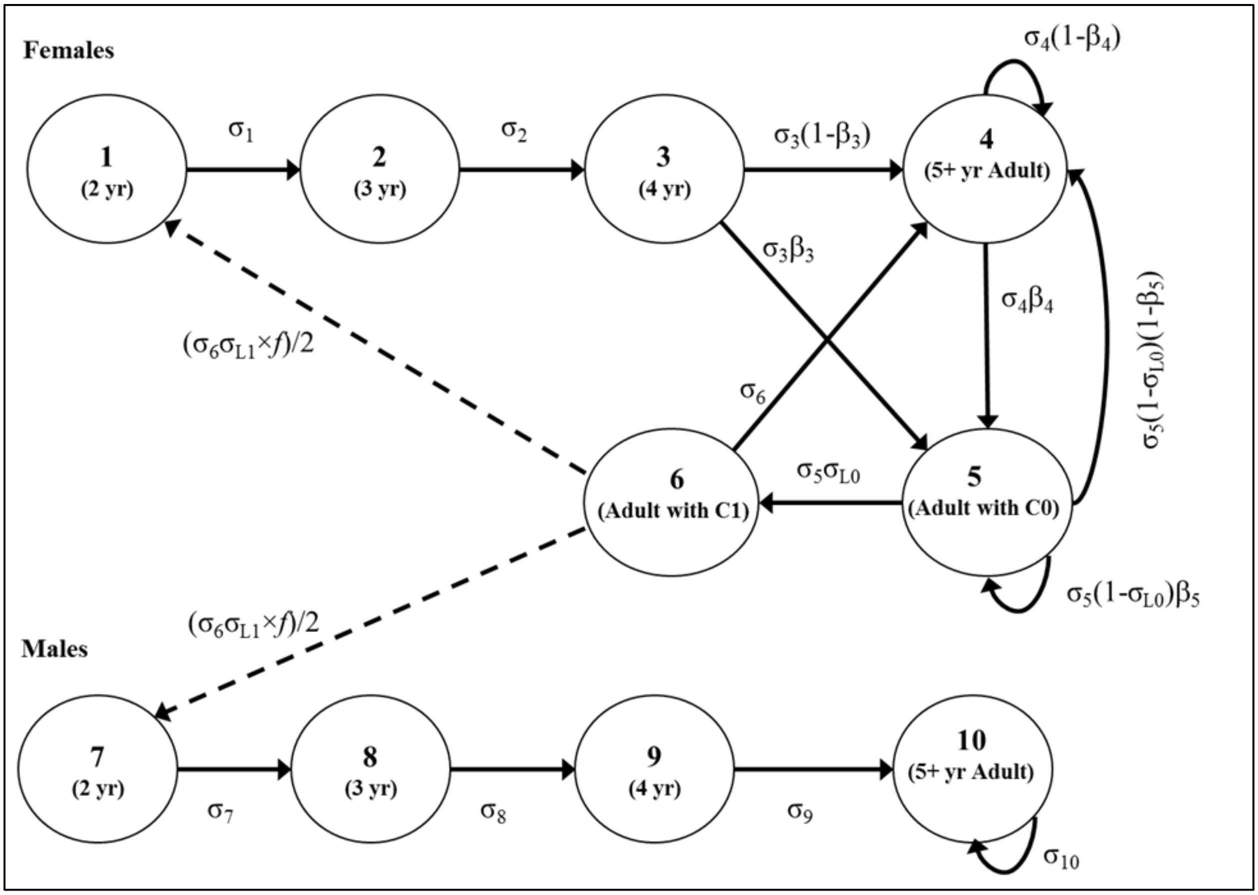


Figure BB1. The polar bear life cycle graph underlying the matrix-based projection model, reproduced from Figure 1 in Regehr et al. (2017). Stages 1–6 are females and stages 7–10 are males; σ_i is the annual probability of survival of an individual in stage *i*, σ_{L0} and σ_{L1} are the probabilities of at least one member of a cub-of-the-year (C0) or yearling (C1) litter surviving, *f* is the expected size of C1 litters that survive to 2 years, and β_i is the probability, conditional on survival, of an individual in stage *i* breeding, thereby producing a C0 litter with at least one member surviving. Solid lines are stage transitions and dashed lines are reproductive contributions.

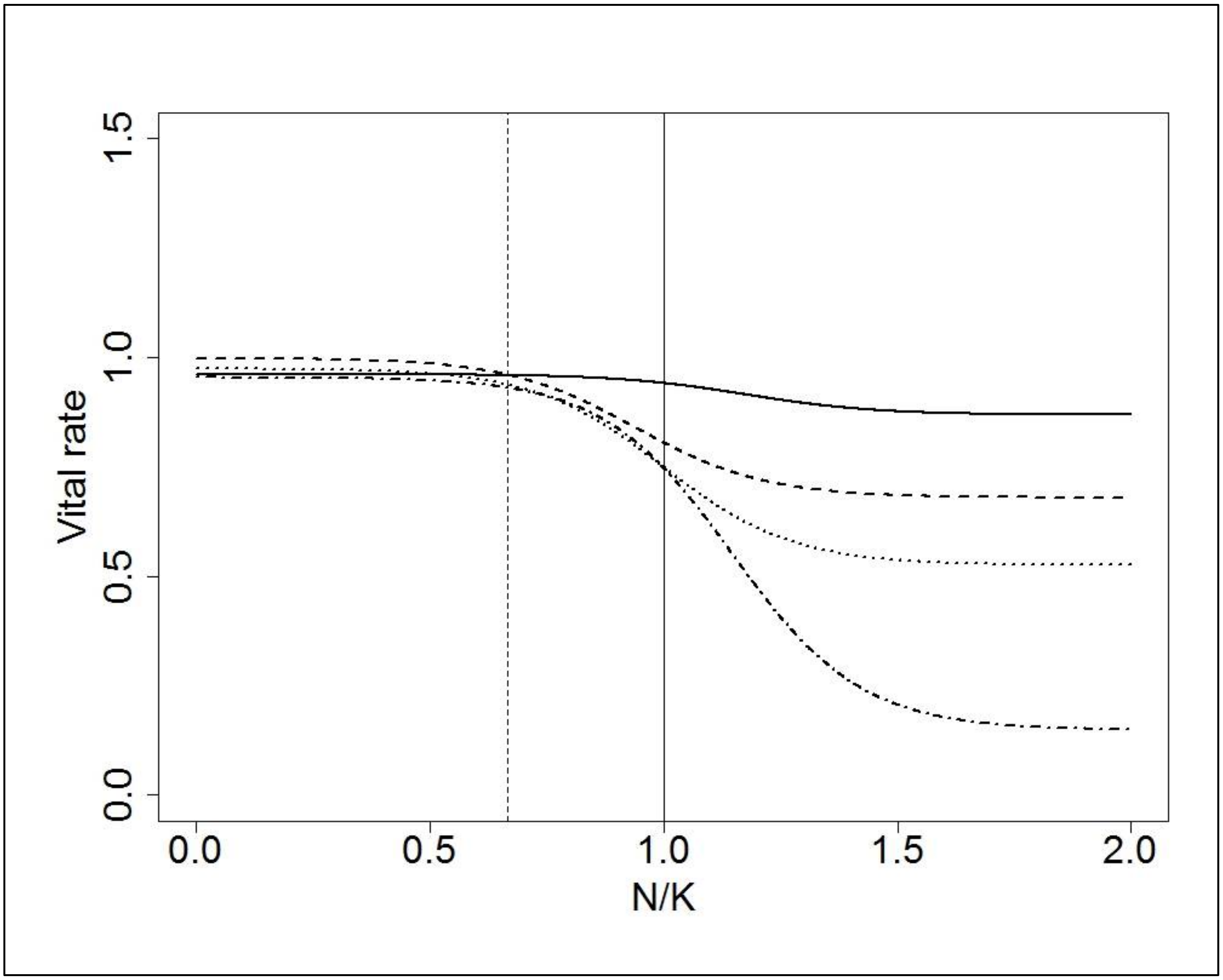


Figure BB2. Sample density-dependent curves of the vital rates for the Baffin Bay polar bear subpopulation for Scenario 2. Vital rates shown are survival probability for stage 4 (σ_4, solid line), survival probability for stage 1 (σ_1, dashed line), survival probability for cub-of-the-year litters (σ_{L0}, dotted line), and breeding probability for stage 4 (β_4, dash-dot line). Density on the x-axis is expressed as the ratio of population size (*N*) to carrying capacity (*K*). The solid vertical line corresponds to *N*/*K* = 1 at carrying capacity. The vital rates at this density would result in a stable subpopulation (i.e., intrinsic population growth rate [*r*] = 0) assuming asymptotic dynamics. The dashed vertical line corresponds to maximum net productivity level (MNPL). The vital rates at a subpopulation size equivalent to MNPL are the mean parameter values for Scenario 2 (Tables BB2 and BB3).

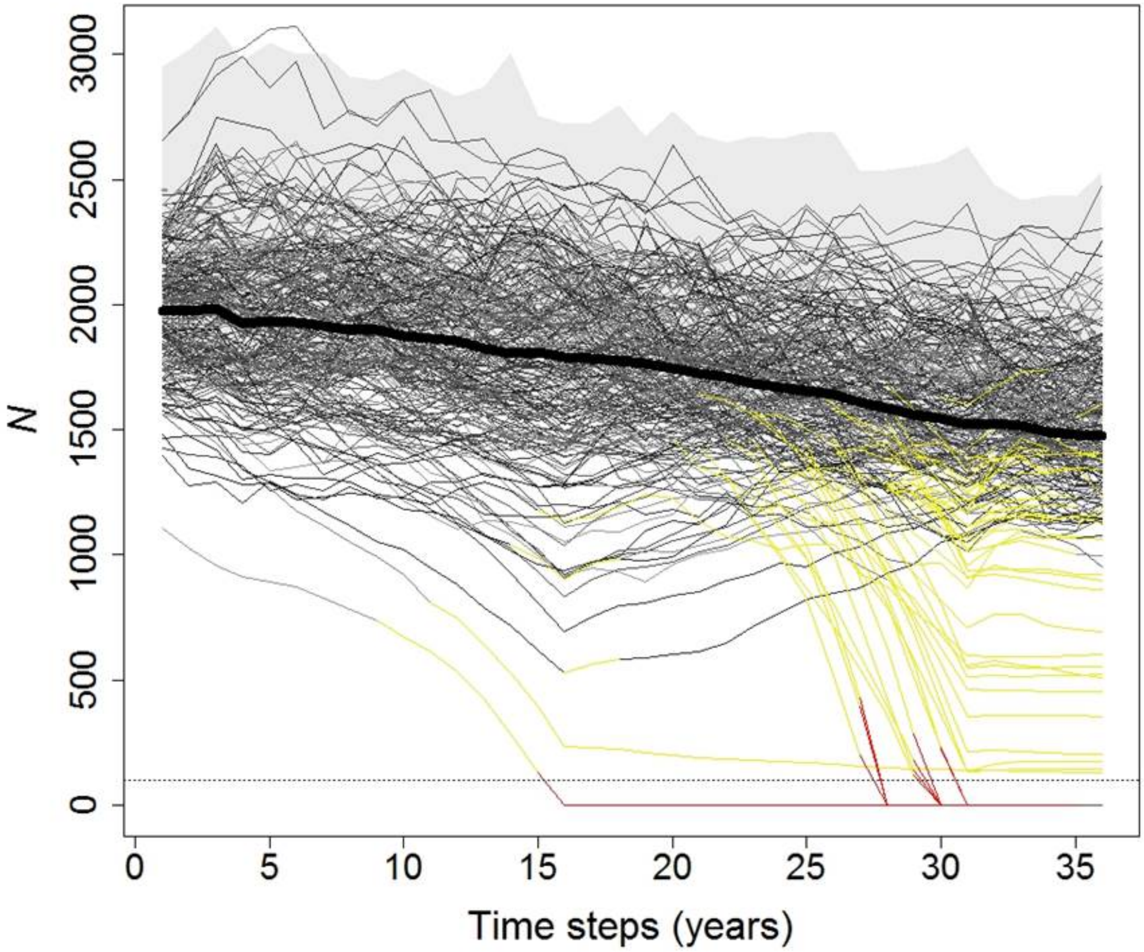


Figure BB3. Sample replicates (black lines) from population projections for the Baffin Bay subpopulation, using vital rates from Scenario 2. The grey shaded area in the background represents the upper 95% confidence interval for carrying capacity, measured in number of bears, which declined at a rate of approximately 5.5% per decade. The y-axis *N* is subpopulation size referenced to independent bears, and the heavy black line is median subpopulation size. Replicates are shaded yellow and red for time steps at which they experienced male depletion or extirpation, respectively. Projections are for a harvest strategy with F_0 = 0.92, *SR* = 1.25, a 15-year management interval, and *rsd.mod* = 1.0 (management inputs are defined in the main text). This harvest strategy equates to a starting (i.e., *t* = 1) harvest level of 160 bears per year.

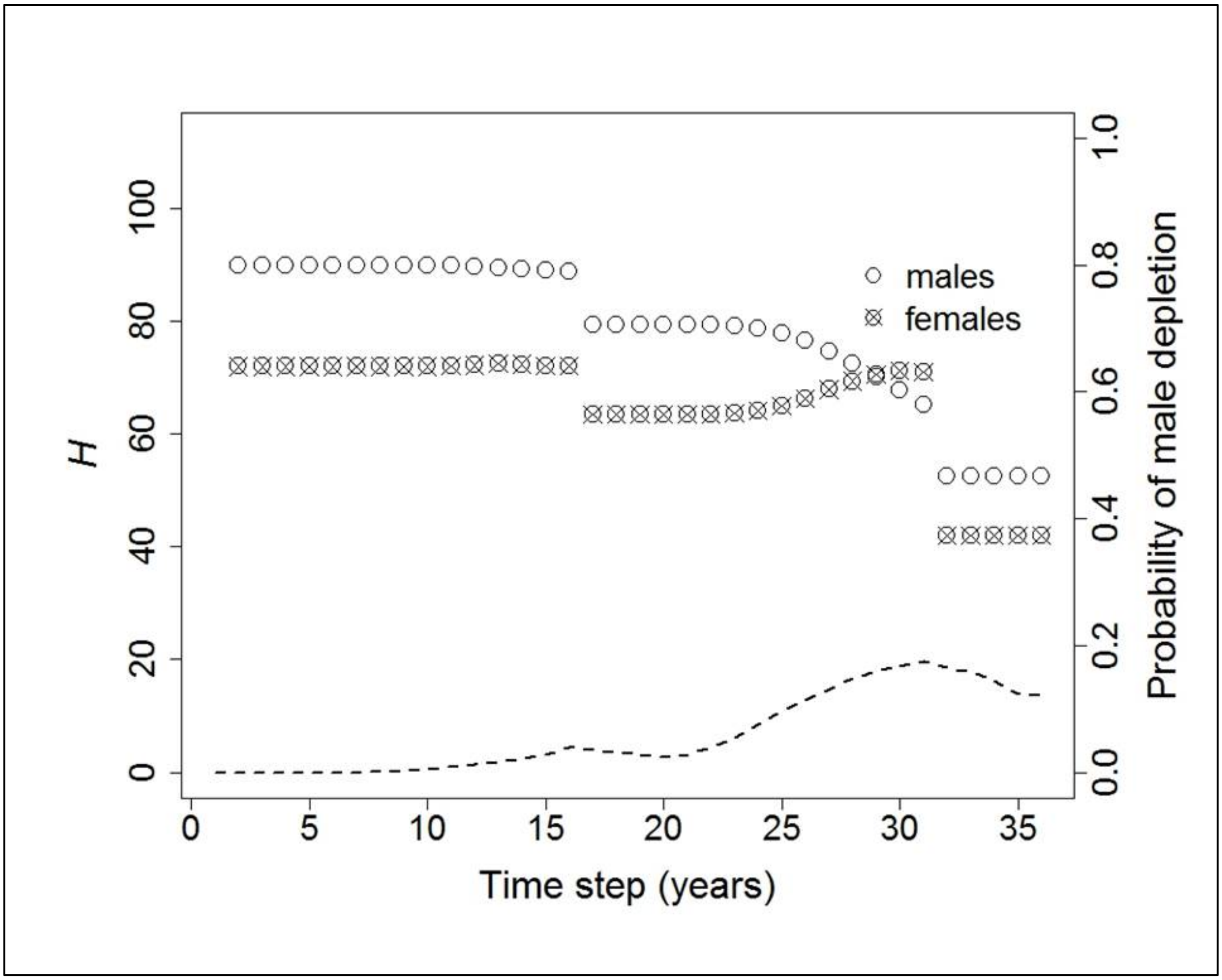


Figure BB4. Example results from population projections for the Baffin Bay polar bear subpopulation, using vital rates from Scenario 2. The left y-axis *H* is the harvest level, with the circles and cross-circles representing the average number of independent male and female polar bear removed per year, respectively, under a state-dependent management approach. The right y-axis is the probability of severe male depletion, values of which are plotted as the dashed line. Projections are for a harvest strategy with F_0 = 0.92, *SR* = 1.25, a 15-year management interval, and *rsd.mod* = 1.0 (i.e., the same harvest strategy as Figure BB3; management inputs are defined in the main text). This harvest strategy equates to a starting (i.e., *t* = 1) harvest level of 160 bears per year.

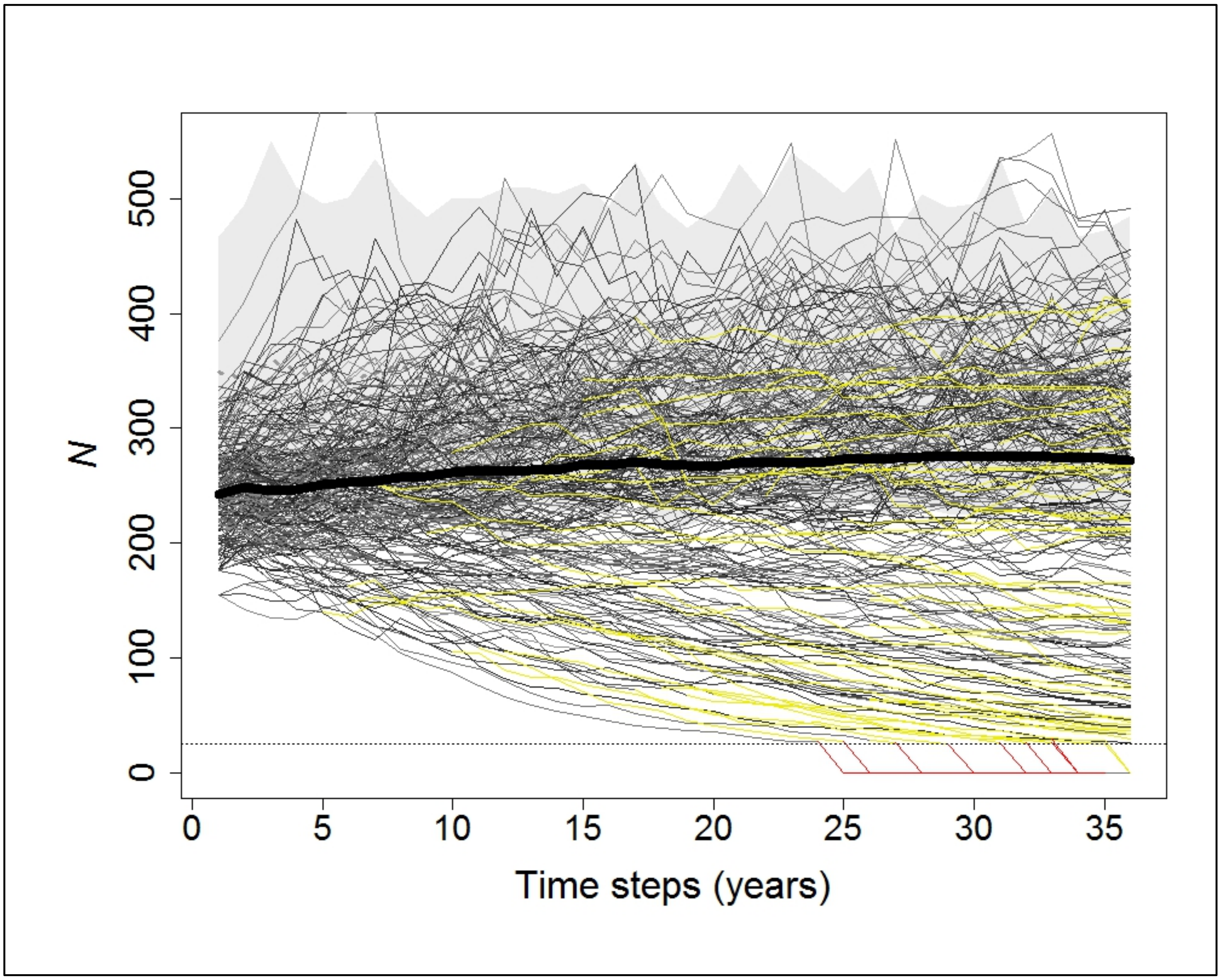


Figure KB1. Sample replicates (black lines) from population projections for the Kane Basin subpopulation, using vital rates from Scenario 1. The grey shaded area in the background represents the upper 95% confidence interval for carrying capacity, measured in number of bears, which declined at a rate of approximately 5.5% per decade. The y-axis *N* is subpopulation size referenced to independent bears, and the heavy black line is median subpopulation size. Replicates are shaded yellow and red for time steps at which they experienced severe male depletion or extirpation, respectively. Projections are for a subpopulation with no harvest.

Appendix S1.

Table S.BB1. Detailed results of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 1 of the vital rates. Each column corresponds to one simulation, with results reported at time steps *t* = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

	***t* = 1**					***t* = 1**					***t* = 1**				
F_O	0.00	0.41	0.81	1.22	1.63	0.00	0.36	0.72	1.08	1.45	0.00	0.27	0.54	0.81	1.08
$h_{t=1}$	0.0%	0.7%	1.4%	2.1%	2.8%	0.0%	0.7%	1.4%	2.1%	2.8%	0.0%	0.7%	1.4%	2.1%	2.8%
$H_{t=1}$	0	20	40	60	80	0	20	40	60	80	0	20	40	60	80
SR	1.00	1.00	1.00	1.00	1.00	1.25	1.25	1.25	1.25	1.25	2.00	2.00	2.00	2.00	2.00
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	***t* = 15**					***t* = 15**					***t* = 15**				
N_t/N_1	1.05	1.05	0.95	0.90	0.89	1.08	1.03	0.99	0.93	0.87	1.07	1.04	0.98	0.97	0.90
N_t/K_t	0.94	0.91	0.85	0.80	0.78	0.95	0.91	0.87	0.82	0.76	0.94	0.93	0.87	0.86	0.81
H_t	0	20	40	59	77	0	20	40	58	77	0	20	39	59	78
$P_{extirpation}$	0.00	0.00	0.01	0.03	0.04	0.00	0.00	0.00	0.02	0.04	0.00	0.00	0.01	0.01	0.02
$P_{male.dep}$	0.00	0.00	0.01	0.04	0.05	0.00	0.00	0.00	0.04	0.07	0.00	0.01	0.01	0.06	0.11
$P_{Objective1}$	0.78	0.73	0.66	0.58	0.53	0.81	0.78	0.69	0.60	0.51	0.81	0.79	0.69	0.68	0.57
$P_{Objective2}$	0.83	0.79	0.72	0.67	0.63	0.86	0.82	0.77	0.69	0.60	0.87	0.82	0.76	0.76	0.67
$P_{Objective3}$	0.86	0.83	0.77	0.71	0.69	0.89	0.86	0.82	0.74	0.66	0.91	0.85	0.80	0.81	0.73

Table S.BB1. Continued.

			***t* = 36**					***t* = 36**					***t* = 36**		
N_t/N_1	0.88	0.83	0.65	0.42	0.37	0.90	0.82	0.69	0.45	0.30	0.89	0.83	0.63	0.39	0.25
N_t/K_t	0.93	0.85	0.68	0.45	0.38	0.94	0.87	0.72	0.47	0.32	0.94	0.88	0.66	0.41	0.27
H_t	0	27	39	41	46	0	28	39	40	38	0	27	31	35	33
$P_{extirpation}$	0.03	0.05	0.07	0.21	0.22	0.02	0.03	0.06	0.20	0.26	0.03	0.04	0.06	0.25	0.33
$P_{male.dep}$	0.01	0.04	0.11	0.18	0.20	0.02	0.03	0.12	0.20	0.22	0.01	0.05	0.23	0.26	0.25
$P_{Objective1}$	0.53	0.30	0.13	0.07	0.05	0.55	0.34	0.15	0.08	0.05	0.55	0.37	0.15	0.09	0.07
$P_{Objective2}$	0.78	0.72	0.49	0.31	0.26	0.81	0.77	0.53	0.33	0.24	0.81	0.77	0.48	0.33	0.26
$P_{Objective3}$	0.78	0.71	0.47	0.28	0.23	0.80	0.76	0.50	0.29	0.21	0.80	0.76	0.45	0.29	0.22

Table S.BB2. Detailed results of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

(a) male-to-female sex ratio in harvest (*SR*) = 1.0

						t = 1					
F_O	0.000	0.516	0.646	0.775	0.904	1.033	1.162	1.291	1.420	1.549	1.679
$h_{t=1}$	0.0%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%	7.1%	7.8%	8.5%	9.2%
$H_{t=1}$	0	80	100	120	140	160	180	200	220	240	260
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
						t = 15					
N_t/N_1	1.14	1.03	1.01	0.97	0.95	0.87	0.85	0.77	0.73	0.57	0.47
N_t/K_t	1.00	0.91	0.89	0.86	0.83	0.78	0.75	0.68	0.64	0.51	0.41
H_t	0	82	102	122	142	162	180	198	220	220	225
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.04	0.03	0.16	0.21
$P_{male.dep}$	0.00	0.00	0.00	0.00	0.00	0.01	0.03	0.10	0.12	0.21	0.26
$P_{Objective1}$	1.00	0.98	0.92	0.83	0.68	0.45	0.31	0.17	0.09	0.04	0.03
$P_{Objective2}$	1.00	1.00	0.99	0.95	0.91	0.79	0.66	0.45	0.34	0.19	0.14
$P_{Objective3}$	1.00	1.00	1.00	0.97	0.96	0.90	0.80	0.62	0.55	0.36	0.27
						t = 36					
N_t/N_1	0.96	0.86	0.84	0.80	0.78	0.72	0.69	0.62	0.57	0.39	0.20
N_t/K_t	1.00	0.90	0.88	0.85	0.81	0.77	0.72	0.65	0.59	0.41	0.21
H_t	0	72	86	97	104	105	107	105	108	94	90
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.01	0.03	0.07	0.14	0.18	0.35	0.43
$P_{male.dep}$	0.00	0.00	0.00	0.00	0.02	0.04	0.05	0.08	0.11	0.13	0.13
$P_{Objective1}$	0.75	0.32	0.22	0.13	0.08	0.05	0.03	0.02	0.02	0.01	0.00
$P_{Objective2}$	1.00	1.00	0.99	0.95	0.84	0.71	0.55	0.40	0.30	0.18	0.11
$P_{Objective3}$	1.00	0.99	0.97	0.91	0.80	0.66	0.50	0.33	0.24	0.14	0.08

Table S.BB2. Continued

(b) male-to-female ratio in the harvest (*SR*) = 1.25

t = 1											
F_O	0.000	0.459	0.574	0.689	0.804	0.918	1.033	1.148	1.263	1.378	1.492
$h_{t=1}$	0.0%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%	7.1%	7.8%	8.5%	9.2%
$H_{t=1}$	0	80	100	120	140	160	180	200	220	240	260
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
t = 15											
N_t/N_1	1.12	1.06	1.00	0.96	0.96	0.92	0.88	0.80	0.75	0.59	0.41
N_t/K_t	1.01	0.92	0.89	0.87	0.85	0.82	0.78	0.71	0.66	0.53	0.37
H_t	0	81	101	122	142	161	181	198	211	212	205
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.03	0.07	0.18	0.28
$P_{male.dep}$	0.00	0.00	0.00	0.00	0.01	0.03	0.05	0.14	0.23	0.34	0.39
$P_{Objective1}$	1.00	0.98	0.93	0.86	0.75	0.59	0.43	0.26	0.16	0.07	0.03
$P_{Objective2}$	1.00	0.99	0.98	0.96	0.93	0.85	0.73	0.55	0.42	0.25	0.12
$P_{Objective3}$	1.00	1.00	0.99	0.98	0.97	0.92	0.85	0.72	0.58	0.39	0.24
t = 36											
N_t/N_1	0.94	0.88	0.85	0.81	0.78	0.75	0.71	0.65	0.56	0.37	0.14
N_t/K_t	1.00	0.91	0.89	0.87	0.82	0.79	0.74	0.68	0.58	0.39	0.15
H_t	0	71	81	89	93	94	98	100	99	96	93
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.02	0.04	0.11	0.16	0.25	0.36	0.45
$P_{male.dep}$	0.00	0.00	0.01	0.02	0.08	0.12	0.14	0.20	0.22	0.21	0.23
$P_{Objective1}$	0.76	0.39	0.28	0.21	0.13	0.09	0.06	0.04	0.03	0.02	0.01
$P_{Objective2}$	1.00	1.00	0.99	0.95	0.84	0.74	0.64	0.51	0.37	0.25	0.17
$P_{Objective3}$	1.00	0.99	0.97	0.93	0.81	0.69	0.57	0.42	0.29	0.18	0.12

Table S.BB2. Continued

(c) male-to-female ratio in the harvest (*SR*) = 2.00

						***t* = 1**					
F_O	0.000	0.344	0.430	0.516	0.603	0.689	0.775	0.861	0.947	1.033	1.119
$h_{t=1}$	0.0%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%	7.1%	7.8%	8.5%	9.2%
$H_{t=1}$	0	80	100	120	140	160	180	200	220	240	260
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
						***t* = 15**					
N_t/N_1	1.12	1.04	1.02	1.01	0.99	0.94	0.85	0.78	0.60	0.41	0.15
N_t/K_t	1.01	0.94	0.91	0.89	0.87	0.83	0.76	0.69	0.54	0.36	0.13
H_t	0	81	102	122	142	161	181	195	198	197	181
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.08	0.16	0.26	0.42
$P_{male.dep}$	0.00	0.00	0.00	0.01	0.03	0.17	0.32	0.46	0.57	0.60	0.51
$P_{Objective1}$	1.00	1.00	0.96	0.92	0.81	0.63	0.39	0.24	0.09	0.06	0.02
$P_{Objective2}$	1.00	1.00	0.99	0.98	0.94	0.83	0.69	0.51	0.31	0.20	0.11
$P_{Objective3}$	1.00	1.00	1.00	1.00	0.97	0.89	0.79	0.61	0.41	0.27	0.15
						***t* = 36**					
N_t/N_1	0.94	0.88	0.85	0.83	0.79	0.73	0.68	0.61	0.48	0.22	0.00
N_t/K_t	1.00	0.93	0.90	0.87	0.82	0.77	0.72	0.64	0.51	0.23	0.00
H_t	0	64	66	68	67	73	87	94	101	92	77
$P_{extirpation}$	0.00	0.00	0.00	0.01	0.05	0.09	0.14	0.19	0.27	0.40	0.55
$P_{male.dep}$	0.00	0.02	0.08	0.17	0.25	0.29	0.34	0.34	0.33	0.34	0.27
$P_{Objective1}$	0.76	0.47	0.37	0.28	0.20	0.14	0.10	0.06	0.04	0.02	0.01
$P_{Objective2}$	1.00	0.99	0.93	0.85	0.74	0.66	0.60	0.47	0.36	0.21	0.11
$P_{Objective3}$	1.00	0.98	0.92	0.83	0.72	0.62	0.53	0.39	0.28	0.15	0.08

Table S.BB3. Detailed results of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 3 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

(a) male-to-female ratio in the harvest (*SR*) = 1.0

				***t* = 1**				
F_O	0.000	0.598	0.797	0.996	1.196	1.395	1.594	1.793
$h_{t=1}$	0.0%	2.1%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%
$H_{t=1}$	0	60	80	100	120	140	160	180
mgmt.interval	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
				***t* = 15**				
N_t/N_1	1.14	1.05	1.00	0.98	0.94	0.87	0.82	0.73
N_t/K_t	1.02	0.93	0.90	0.87	0.83	0.78	0.72	0.66
H_t	0	60	80	100	119	136	153	172
$P_{extirpation}$	0.00	0.00	0.00	0.01	0.01	0.04	0.06	0.06
$P_{male.dep}$	0.00	0.00	0.00	0.01	0.01	0.01	0.05	0.06
$P_{Objective1}$	0.97	0.94	0.88	0.80	0.68	0.48	0.35	0.21
$P_{Objective2}$	0.99	0.97	0.94	0.89	0.84	0.73	0.60	0.48
$P_{Objective3}$	1.00	0.98	0.97	0.92	0.89	0.81	0.70	0.59
				***t* = 36**				
N_t/N_1	0.94	0.84	0.79	0.73	0.68	0.64	0.61	0.57
N_t/K_t	0.99	0.88	0.83	0.77	0.72	0.68	0.64	0.60
H_t	0	51	66	76	79	80	82	80
$P_{extirpation}$	0.00	0.00	0.00	0.02	0.04	0.07	0.12	0.15
$P_{male.dep}$	0.00	0.00	0.00	0.01	0.01	0.02	0.02	0.02
$P_{Objective1}$	0.72	0.26	0.17	0.11	0.07	0.06	0.05	0.03
$P_{Objective2}$	0.97	0.94	0.86	0.68	0.54	0.46	0.41	0.36
$P_{Objective3}$	0.97	0.93	0.83	0.64	0.49	0.41	0.36	0.31

Table S.BB3. Continued

(b) male-to-female ratio in the harvest (*SR*) = 1.25

				***t* = 1**				
F_O	0.000	0.531	0.709	0.886	1.063	1.240	1.417	1.594
$h_{t=1}$	0.0%	2.1%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%
$H_{t=1}$	0	60	80	100	120	140	160	180
mgmt.interval	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
				***t* = 15**				
N_t/N_1	1.15	1.05	1.02	0.99	0.94	0.87	0.83	0.79
N_t/K_t	1.01	0.93	0.91	0.87	0.83	0.78	0.74	0.70
H_t	0	60	80	99	119	136	156	167
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.02	0.03	0.05	0.09
$P_{male.dep}$	0.00	0.01	0.00	0.00	0.03	0.04	0.07	0.06
$P_{Objective1}$	0.96	0.92	0.88	0.80	0.69	0.52	0.42	0.33
$P_{Objective2}$	0.97	0.94	0.92	0.86	0.82	0.71	0.64	0.56
$P_{Objective3}$	0.98	0.95	0.94	0.89	0.86	0.79	0.73	0.66
				***t* = 36**				
N_t/N_1	0.95	0.85	0.82	0.77	0.73	0.69	0.66	0.67
N_t/K_t	0.99	0.89	0.86	0.80	0.76	0.73	0.70	0.70
H_t	0	52	63	74	79	82	86	89
$P_{extirpation}$	0.00	0.01	0.00	0.01	0.06	0.08	0.10	0.13
$P_{male.dep}$	0.00	0.01	0.01	0.02	0.02	0.03	0.03	0.04
$P_{Objective1}$	0.70	0.33	0.23	0.17	0.12	0.09	0.07	0.06
$P_{Objective2}$	0.96	0.93	0.88	0.75	0.66	0.58	0.52	0.51
$P_{Objective3}$	0.96	0.92	0.86	0.72	0.62	0.53	0.47	0.45

Table S.BB3. Continued

(c) male-to-female ratio in the harvest (*SR*) = 2.00

				t = 1				
F_O	0.000	0.399	0.531	0.664	0.797	0.930	1.063	1.196
$h_{t=1}$	0.0%	2.1%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%
$H_{t=1}$	0	60	80	100	120	140	160	180
mgmt.interval	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
				t = 15				
N_t/N_1	1.14	1.07	1.05	1.02	0.98	0.94	0.91	0.84
N_t/K_t	1.0	0.9	0.9	0.9	0.9	0.8	0.8	0.7
H_t	0	60	80	99	118	136	153	171
$P_{extirpation}$	0.00	0.00	0.00	0.01	0.02	0.03	0.06	0.06
$P_{male.dep}$	0.00	0.00	0.00	0.03	0.04	0.07	0.12	0.17
$P_{Objective1}$	0.97	0.94	0.91	0.84	0.80	0.67	0.59	0.42
$P_{Objective2}$	0.99	0.96	0.95	0.89	0.86	0.80	0.72	0.60
$P_{Objective3}$	0.99	0.97	0.97	0.91	0.89	0.85	0.77	0.69
				t = 36				
N_t/N_1	0.93	0.89	0.86	0.82	0.77	0.74	0.72	0.71
N_t/K_t	0.98	0.93	0.91	0.86	0.81	0.79	0.76	0.74
H_t	0	50	65	68	73	75	79	80
$P_{extirpation}$	0.00	0.00	0.01	0.03	0.07	0.09	0.14	0.14
$P_{male.dep}$	0.00	0.00	0.02	0.05	0.09	0.13	0.13	0.14
$P_{Objective1}$	0.70	0.47	0.38	0.29	0.22	0.18	0.16	0.14
$P_{Objective2}$	0.96	0.96	0.94	0.82	0.74	0.68	0.62	0.59
$P_{Objective3}$	0.96	0.95	0.93	0.80	0.71	0.64	0.58	0.54

Table S.BB4. Detailed results of secondary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. Results are presented for the highest strategies that meet Management Objective 2b, for each unique combination of *mgmt.interval* and *rsd.mod*. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

					***t* = 1**				
F_O	1.148	0.976	0.861	1.033	0.918	0.804	0.918	0.804	0.746
$h_{t=1}$	7.1%	6.0%	5.3%	6.4%	5.7%	5.0%	5.7%	5.0%	4.6%
$H_{t=1}$	200	170	150	180	160	140	160	140	130
SR	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25
mgmt.interval	10	15	20	10	15	20	10	15	20
rsd.mod	0.50	0.50	0.50	1.00	1.00	1.00	1.50	1.50	1.50
					***t* = 15**				
N_t/N_1	0.86	0.89	0.93	0.88	0.90	0.96	0.92	0.96	0.96
N_t/K_t	0.76	0.78	0.82	0.78	0.81	0.85	0.81	0.84	0.85
H_t	159	173	153	163	161	142	154	142	132
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
$P_{male.dep}$	0.03	0.02	0.01	0.01	0.03	0.00	0.01	0.01	0.00
$P_{Objective1}$	0.25	0.47	0.62	0.41	0.55	0.77	0.55	0.75	0.79
$P_{Objective2}$	0.62	0.77	0.87	0.75	0.80	0.93	0.85	0.92	0.93
$P_{Objective3}$	0.89	0.88	0.95	0.92	0.89	0.98	0.96	0.96	0.97
					***t* = 36**				
N_t/N_1	0.74	0.77	0.78	0.74	0.75	0.76	0.76	0.77	0.77
N_t/K_t	0.77	0.80	0.81	0.77	0.79	0.79	0.79	0.80	0.81
H_t	126	97	122	115	94	130	107	90	113
$P_{extirpation}$	0.00	0.03	0.01	0.01	0.04	0.02	0.01	0.05	0.03
$P_{male.dep}$	0.10	0.11	0.12	0.14	0.10	0.18	0.10	0.10	0.13
$P_{Objective1}$	0.02	0.06	0.12	0.07	0.09	0.14	0.12	0.14	0.18
$P_{Objective2}$	0.76	0.80	0.83	0.71	0.76	0.77	0.73	0.76	0.78
$P_{Objective3}$	0.68	0.76	0.76	0.65	0.71	0.70	0.68	0.71	0.72

Table S.BB5. Detailed results of *post hoc* simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time step t = 15. Management strategies BB_S1 and BB_S2 are described in the main text. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; N is population size referenced to independent bears; and K is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1).

	Management strategy	
	BB_S1	**BB_S2**
	***t* = 1**	
F_O	1.58	-
$h_{t=1}$	8.7%	8.7%
$H_{t=1}$	245	245
SR	1.00	1.25
mgmt.interval	5	-
rsd.mod	0.50	-
	***t* = 15**	
N_t/N_1	0.75	0.55
N_t/K_t	0.67	0.49
H_t	180	212
$P_{extirpation}$	0.00	0.23
$P_{male.dep}$	0.01	0.30
$P_{Objective1}$	0.05	0.07
$P_{Objective2}$	0.22	0.24
$P_{Objective3}$	0.73	0.36

Table S.KB1. Detailed results of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 1 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

			***t* = 1**					***t* = 1**		
F_O	0.000	0.645	1.289	1.934	2.579	0.000	0.417	0.834	1.251	1.668
$h_{t=1}$	0.0%	0.6%	1.1%	1.7%	2.2%	0.0%	0.6%	1.1%	1.7%	2.2%
$H_{t=1}$	0	2	4	6	8	0	2	4	6	8
SR	0.94	0.94	0.94	0.94	0.94	2.00	2.00	2.00	2.00	2.00
mgmt.interval	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
			***t* = 15**					***t* = 15**		
N_t/N_1	1.10	1.02	0.90	0.79	0.79	1.09	1.02	0.98	0.83	0.75
N_t/K_t	0.78	0.72	0.64	0.56	0.56	0.76	0.70	0.69	0.58	0.53
H_t	0	2	4	6	7	0	2	4	6	7
$P_{extirpation}$	0.00	0.01	0.01	0.02	0.06	0.01	0.00	0.01	0.01	0.07
$P_{male.dep}$	0.10	0.16	0.26	0.34	0.37	0.10	0.15	0.34	0.39	0.46
$P_{Objective1}$	0.74	0.64	0.52	0.46	0.43	0.71	0.61	0.63	0.48	0.42
$P_{Objective2}$	0.59	0.46	0.37	0.33	0.30	0.56	0.45	0.45	0.34	0.28
			***t* = 36**					***t* = 36**		
N_t/N_1	1.12	0.82	0.44	0.28	0.22	1.12	0.79	0.54	0.29	0.24
N_t/K_t	0.80	0.59	0.31	0.20	0.16	0.79	0.55	0.38	0.21	0.17
H_t	0	3	4	4	4	0	3	3	3	3
$P_{extirpation}$	0.02	0.09	0.19	0.28	0.38	0.04	0.08	0.17	0.31	0.36
$P_{male.dep}$	0.20	0.36	0.45	0.47	0.43	0.27	0.45	0.59	0.50	0.51
$P_{Objective1}$	0.70	0.48	0.21	0.13	0.10	0.68	0.43	0.25	0.14	0.10
$P_{Objective2}$	0.64	0.33	0.14	0.08	0.07	0.61	0.31	0.16	0.09	0.06

Table S.KB2. Detailed results of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; N is population size referenced to independent bears; and K is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

	***t* = 1**									***t* = 1**								
F_O	0.000	0.159	0.319	0.478	0.638	0.797	0.956	1.116	1.275	0.000	0.103	0.206	0.309	0.412	0.516	0.619	0.722	0.825
$h_{t=1}$	0.0%	0.6%	1.1%	1.7%	2.2%	2.8%	3.4%	3.9%	4.5%	0.0%	0.6%	1.1%	1.7%	2.2%	2.8%	3.4%	3.9%	4.5%
$H_{t=1}$	0	2	4	6	8	10	12	14	16	0	2	4	6	8	10	12	14	16
SR	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	***t* = 15**									***t* = 15**								
N_t/N_1	1.28	1.31	1.26	1.28	1.19	1.16	1.10	1.06	0.99	1.34	1.28	1.26	1.23	1.20	1.16	1.13	1.08	1.01
N_t/K_t	0.92	0.91	0.88	0.88	0.83	0.81	0.78	0.75	0.70	0.92	0.91	0.89	0.86	0.84	0.82	0.78	0.76	0.70
H_t	0	2	4	6	8	10	11	13	13	0	2	4	6	8	9	11	13	13
$P_{extirpation}$	0.00	0.00	0.02	0.02	0.04	0.03	0.04	0.09	0.18	0.00	0.00	0.00	0.02	0.03	0.04	0.06	0.09	0.18
$P_{male.dep}$	0.06	0.10	0.13	0.12	0.16	0.20	0.24	0.26	0.18	0.05	0.08	0.14	0.15	0.23	0.28	0.31	0.32	0.36
$P_{Objective1}$	0.85	0.85	0.78	0.78	0.73	0.74	0.69	0.63	0.57	0.86	0.85	0.78	0.78	0.72	0.73	0.70	0.63	0.57
$P_{Objective2}$	0.77	0.77	0.72	0.69	0.64	0.64	0.58	0.53	0.48	0.79	0.78	0.71	0.70	0.64	0.65	0.61	0.54	0.49
	***t* = 36**									***t* = 36**								
N_t/N_1	1.26	1.29	1.20	1.19	1.07	0.97	0.84	0.72	0.59	1.33	1.26	1.21	1.14	1.03	0.91	0.78	0.54	0.48
N_t/K_t	0.92	0.90	0.85	0.83	0.75	0.68	0.60	0.52	0.42	0.93	0.90	0.87	0.81	0.73	0.65	0.55	0.39	0.34
H_t	0	3	5	7	8	9	9	9	10	0	3	4	6	6	7	7	6	7
$P_{extirpation}$	0.03	0.03	0.05	0.06	0.07	0.08	0.14	0.22	0.29	0.03	0.03	0.05	0.06	0.10	0.12	0.17	0.26	0.32
$P_{male.dep}$	0.11	0.16	0.18	0.17	0.22	0.29	0.32	0.33	0.25	0.13	0.15	0.22	0.27	0.36	0.43	0.47	0.47	0.44
$P_{Objective1}$	0.81	0.83	0.75	0.73	0.66	0.60	0.50	0.42	0.37	0.84	0.83	0.75	0.72	0.61	0.56	0.46	0.36	0.33
$P_{Objective2}$	0.78	0.79	0.70	0.66	0.57	0.48	0.39	0.31	0.28	0.80	0.78	0.70	0.65	0.53	0.46	0.36	0.28	0.25

Table S.KB3. Detailed results of secondary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. Results are presented for the highest strategies that meet Management Objective 1b, for each unique combination of *mgmt.interval* and *rsd.mod*. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

					t = 1				
F_O	0.638	0.638	0.558	0.558	0.478	0.478	0.558	0.398	0.398
$h_{t=1}$	2.2%	2.2%	2.0%	2.0%	1.7%	1.7%	2.0%	1.4%	1.4%
$H_{t=1}$	8	8	7	7	6	6	7	5	5
SR	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94
mgmt.interval	10	15	20	10	15	20	10	15	20
rsd.mod	0.50	0.50	0.50	1.00	1.00	1.00	1.50	1.50	1.50
					t = 15				
N_t/N_1	1.12	1.19	1.22	1.15	1.25	1.24	1.17	1.19	1.24
N_t/K_t	0.80	0.82	0.86	0.82	0.89	0.88	0.84	0.84	0.87
H_t	10	8	7	9	6	6	8	5	5
$P_{extirpation}$	0.01	0.02	0.03	0.00	0.02	0.01	0.01	0.01	0.02
$P_{male.dep}$	0.17	0.15	0.11	0.13	0.11	0.14	0.10	0.09	0.09
$P_{Objective1}$	0.74	0.76	0.77	0.79	0.82	0.80	0.79	0.77	0.79
$P_{Objective2}$	0.61	0.66	0.69	0.69	0.74	0.72	0.69	0.67	0.70
					t = 36				
N_t/N_1	1.08	1.10	1.11	1.10	1.17	1.15	1.12	1.15	1.18
N_t/K_t	0.78	0.77	0.79	0.80	0.84	0.82	0.81	0.82	0.84
H_t	9	9	9	8	7	7	7	5	6
$P_{extirpation}$	0.04	0.06	0.07	0.05	0.05	0.07	0.05	0.05	0.08
$P_{male.dep}$	0.23	0.21	0.19	0.21	0.18	0.21	0.16	0.14	0.15
$P_{Objective1}$	0.70	0.72	0.72	0.75	0.79	0.75	0.74	0.72	0.74
$P_{Objective2}$	0.59	0.61	0.64	0.66	0.71	0.67	0.66	0.66	0.68

Table S.KB4. Detailed results of *post hoc* simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time step t = 15 and t = 36. Harvest strategies KB_S1 and KB_S2 and the simulation conditions on which they are based, are described in the main text. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

	Management strategy	
	KB_S1	**KB_S2**
	***t* = 1**	
F_O	0.80	0.60
$h_{t=1}$	2.8%	2.2%
$H_{t=1}$	10	8
SR	0.94	0.94
mgmt.interval	15	15
rsd.mod	1.00	1.00
	***t* = 15**	
N_t/N_1	1.20	1.19
N_t/K_t	0.85	0.83
H_t	10	8
$P_{extirpation}$	0.02	0.00
$P_{male.dep}$	0.07	0.11
$P_{Objective1}$	0.82	0.79
$P_{Objective2}$	0.69	0.65
	***t* = 36**	
N_t/N_1	1.09	1.08
N_t/K_t	0.80	0.77
H_t	10	8
$P_{extirpation}$	0.04	0.04
$P_{male.dep}$	0.21	0.18
$P_{Objective1}$	0.71	0.70
$P_{Objective2}$	0.56	0.55

ᑐᓂᔭᐅᓯᒪᔪᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐃᒪᓇᐃᓘᖅᑕᐅᕐᑯᓗᒋᑦ

ᑐᓴᐅᒪᔪᑎᒃᓴᑦ: ᐃᓱᒪᓕᐅᕈᑎᑦ: X

ᐋᖅᑭᒃᑕᐅᔭᕆᐊᓕᒃ: ᓇᓄᑦ ᑲᑎᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᓄᑦ ᐊᑐᓕᕐᑯᐃᔪᑏᑦ ᑕᐅᕗᖓ ᑲᐃᓐ ᐃᑭᕋᓴᖓᓂ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓂ

ᐅᖃᐅᓯᖃᒃᓯᒪᓂᖏᑦ:

- ᑭᖑᓪᓕᖅᑳᖑᓚᐅᖅᓯᒪᔪᖅ ᑕᒪᐃᓐᓂᓕᒫᖅ ᐅᓄᕐᓂᖏᓐᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᕐᓕᕋᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ, ᐱᖃᖃᑎᒋᒍᑕᐅᔪᓂ ᖃᓛᖦᓂᑦ, ᐱᓕᕆᐊᖑᓚᐅᖅᓯᒪᖓᑕ ᐊᑯᓐᓂᖏᓐᓂ 1992 ᐊᒻᒪᓗ 1997. 1997 ᐅᓄᕐᓂᖏᓐᓂᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑎᑦ ᐃᒪᓇᑎᒋᐅᓚᐅᖅᓯᒪᖓᑕ 164-ᖑᓗᑎᒃ ᓇᓄᐃᑦ, ᐊᒻᒪᓗ ᐅᓄᕐᓂᖏᑦ ᖃᑦᔫᓇᕈᒃᐸᓪᓕᐊᓂᐊᕐᓂᕋᒃᑕᐅᓚᐅᖅᓯᒪᓪᓗᑎᒃ ᐅᓄᓗᐊᕌᓗᒃᑎᓐᓇᒋᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᓗᐊᕐᓇᑎᒡᓗ ᑕᐃᑉᓯᒪᓂ. 2006-ᒥ ᑲᓛᖦᓯᒥᐅᑦ ᐊᑐᓕᖅᑎᑦᑎᓚᐅᕐᒪᑕ ᓇᓐᓄᒐᓱᐊᒐᒃᓴᓂᑦ ᐱᕐᓕᕋᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ ᐅᓄᖄᓃᓇᐊᖅᑎᑦᑎᔪᑕᐅᓗᑎᒃ ᖃᓚᖦᖢᑦ ᐱᔪᓐᓇᖅᑕᕐᕆᓐᓂᑦ ᑕᐃᒪᓐᓇᐃᓕᔪᑕᐅᓪᓗᑎᒃ ᑕᐃᒫᑦᑕᐃᓐᓇᓕᕐᕆᓐᓂᕐᕆᓄᑦ ᓇᓐᓄᒃᑦᑕᕐᓂᕐᕆᑦ.
- ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᓇᓗᓈᕿᓗᐊᕐᓂᕐᕆᑦ ᐅᓄᕐᓂᕐᕆᑕ ᖃᓄᐃᓕᖓᓂᕐᕆᑦ ᑕᐃᒪᓐᓇᐃᓕᖅᑎᑕᐅᓯᒪᓪᓗᓂ ᐃᓚᒐᒎᑦ ᐅᓄᕐᕆᓗᐊᕐᓂᕐᕆᓐᓄᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᓂᕐᕆᑕ, ᓇᓐᓄᒐᓱᐊᕐᓂᕐᒍᑦ ᓇᓗᓈᕿᓃᑦ ᐊᒻᒪᓗ ᐊᓯᑭᓕᖅᑎᓪᓗᒍ ᐊᕙᑎᑦᑕ ᖃᓄᐃᓕᖓᓂᖓ, ᑲᓇᑕ–ᖃᓛᖦᓗᑦ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ ᓇᓄᕐᓂᑦ (JC) ᐃᒪᓐᓇᐃᓘᕐᑯᔨᓕᓚᐅᕐᒪᑕ ᓄᑖᒥᑦ ᖃᐅᔨᓇᓱᐊᖅᑐᖃᕐᑐᓂ ᐱᓕᕆᐊᖑᓗᓂ ᖃᐅᔨᔭᐅᑎᐊᕐᑯᓪᓗᒋᑦ ᐅᓪᓗᒥᒧᑦ–ᐊᖑᒪᑎᓕᕐᓗᑎᒃ ᖃᓄᐃᓕᖓᓂᕐᕆᓇ ᑖᒃᑯᐊ ᐅᓄᕐᓂᒃᑲᓐᓂᕐᕆᑕ. ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐱᓕᕆᕐᑯᔨᓕᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔮᓛᑦ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᑕᐃᒃᑯᓂᖓ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᓯᒪᔪᓂᑦ.
- ᓄᑖᖅ ᐃᓇᕐᕆᖅᓯᓇᓱᐊᓱᕐᕈᕐᕆᑦᑐᑦ ᐊᔾᔨᒌᒍᓂᓛᑦ ᓇᓗᓇᐃᑯᑎᖅᑕᐅᓯᒪᔪᑦ-ᐱᔭᐅᒃᑲᓐᓂᖅᓯᒪᔪᑦ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᑎᓪᓗᒋᑦ ᐱᓕᕆᐊᖑᓚᐅᕐᒪᑕ ᐊᑯᓐᓂᕐᕆᓐᓂ 2012-2014. ᓄᑖᑦ ᐅᓄᖅᑐᐊᕌᓂᕐᕆᓐᓂᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑎᑦ ᐱᕐᓕᕋᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ (KB) ᓇᓄᕐᕆᓐᓄᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᕐᕆᓐᓂ ᐃᒪᓇᑎᒋᐅᖓᑕ 357 ᓇᓄᐃᑦ (95% ᐳᓴᓐᑎᑦ ᖁᓚᕐᕆᓐᓂᕐᒧᑦ ᓄᖃᖓᑲᐃᓐᓇᕐᓂᖅ: 221-493), ᑕᐃᒃᑯᐊᓕ ᓱᓕ 1997-ᒥ ᒥᒃᓴᐅᓴᒃᑕᐅᔪᑦ ᐃᒪᓇᑎᒋᐅᓚᐅᕐᒪᑕ 224 ᓇᓄᐃᑦ.
- ᒪᓕᒡᓗᒋᑦ ᑭᖑᓪᓕᖅᐹᑦ ᐅᓂᒃᑳᓕᐅᕐᕈᓯᒪᔪᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕋᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᓂᕐᕆᑦ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᐱᓕᕆᕐᑯᔨᓕᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᖅᑕᐅᕐᑯᔨᓪᓗᑎᒃ ᐃᖃᓇᐃᔮᓛᑦ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᐊᕐᕈᓇᓱᒃᑕᐅᔪᓐᓇᕐᑯᓪᓗᒋᑦ ᐊᑐᓕᕐᑯᐃᔪᑎᓂᑦ ᐊᕐᕈᓇᓱᐊᖅᑕᐅᓂᕐᕆᑕ ᕐᑯᒃᓴᓇᕈᑎᕐᕆᓐᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᑦ ᖃᐅᔨᓇᓱᒃᑕᐅᔪᑎᕐᕆᑦ.
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ (GN) ᐊᐱᕐᓱᓚᐅᕐᒪᑕ ᐊᒃᑐᖅᑕᐅᓯᒪᔪᓂᑦ ᓄᓇᓕᕐᓂᑦ ᔪᓄᐊᕆ 2017-ᒥ ᐱᔭᕇᒃᓯᔪᑕᐅᔪᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔮᓛᑦ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᓂᕐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂᑦᑐᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕋᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᕐᕆᓐᓂᑦ.

ᒫᓐᓇᐅᔪᖅ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ:

- ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᑲᑎᓚᐅᕐᒪᑕ ᐅᑐᐱᕆ 2017-ᒥ ᐃᓱᒪᓕᐅᕈᑎᖃᕈᒪᓪᓗᑎᒃ ᑲᑎᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᓂᑦ (TAH) ᐊᑐᓕᖁᔨᔪᑎᑦ ᑐᙵᕕᖃᖅᑐᑎᒃ ᓴᕿᑕᐅᔪᓂᑦ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᑦ ᐱᔭᕇᒃᑕᐅᓂᖏᓐᓂᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒌᓂᑦ (SWG).
- ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓇᓱᐊᓂᖅ ᐊᑐᓕᖁᔨᓚᐅᕐᒪᑦ ᐲᖅᑕᐅᖁᔨᓪᓗᓂ ᑎᑭᐅᒪᔪᓂᑦ 10-ᓄᑦ (2.8% ᐳᓴᓐᑎᓄᑦ) ᓇᓄᖏᓐᓂᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ ᑕᐃᒪᓐᓇᐃᓕᑎᑕᐅᓗᓂ ᓇᓗᓈᕿᓗᐊᕐᓂᖏᓐᓄᑦ ᐱᑕᖃᕆᐊᖃᓪᓚᕆᖕᓂᖏᑕ ᐊᒻᒪᓗ ᐃᓘᓇᓕᒫᖏᑦ ᒥᑭᔪᕈᓗᐃᑦ ᖃᐅᔨᓴᒐᒃᓴᑦ ᐊᖏᓂᖏᑦ ᑕᐃᑦᓱᒪᓂ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᖓᑕ.
- ᐱᔪᒪᔭᐅᓗᐊᕐᓂᖏᓐᓄᑦ ᐊᒻᒪᓗ ᑐᙵᕕᒋᓗᒋᑦ ᓱᓕᓂᖏᑦ ᐅᓄᕐᓂᖏᓐᓄᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑎᑦ ᐅᓄᕐᓂᖅᓴᐅᖓᑕ ᑕᐃᑦᓱᒪᓂᐅᕙᑖᓚᐅᖅᑐᒥᑦ ᐅᓄᕐᓂᕆᔭᐅᓚᐅᖅᑐᓂᑦ, ᖃᓛᑦᓯᑦ ᐊᑐᓕᖁᔨᓚᐅᕐᒪᑕ ᐊᖑᓇᓱᒍᓐᓇᕈᒪᓪᓗᑎᒃ 15-ᓂᑦ ᓇᓄᕐᓂᑦ (4.5% ᐳᓴᓐᑎᑦ).
- ᖃᓚᑦᓯᒥᐅᑦ ᑐᒃᓯᕋᓚᐅᕐᒪᑕ ᓴᓂᕐᕙᒃᑕᐅᖁᔨᓪᓗᑎᒃ 10–ᓂᑦ ᓇᓄᕐᓂᑦ ᖃᓚᑦᓯᒥᐅᓄᑦ ᐊᒻᒪᓗ 5–ᓂᑦ ᓄᓇᕗᒻᒧᑦ ᓄᓇᓕᖓᓄᑦ ᐊᐅᓯᐃᑦᑑᑉ ᓇᓐᓄᕋᓯᓗᐊᓯᖑᖏᑦᑐᓄᑦ ᑖᒃᑯᓂᖓᑦ ᓇᓄᕐᓂᑦ ᐅᓄᕐᓂᕆᔭᐅᔪᓃᖓᖅᑐᓂᑦ. ᖃᓛᑦᓯᒥᐅᑦ ᑭᖑᓂᖓᒍᑦ ᖃᑦᔫᓐᓇᕈᓂᐊᖅᓯᓚᐅᕐᒪᑕ ᓇᓐᓄᕋᒃᓴᒥᖕᓂᑦ ᐃᒪᓐᓇᑎᑭᐅᓕᖅᑎᑕᐅᓗᑎᒃ 9 ᓇᓄᐃᑦ
- ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ 4.5% ᐳᓴᓐᑎᑦ ᐊᖏᖅᑕᐅᓐᓂᕋᓗᐊᖅᐸᑕ, ᐃᒪᓐᓇᐅᒐᔭᖅᑑᔭᖏᒻᒪᑕ ᑲᑎᑕᐅᒐᓗᐊᖅᐊᑕ ᐲᖅᑕᐅᓗᑎᓪᓗ ᐊᕐᕌᒍᑕᒪᑦ ᐅᖓᑖᓄᕋᔭᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐊᑐᓕᖁᔭᖏᓐᓂ 10 ᓇᓄᐃᑦ (2.8% ᐳᓴᓐᑎᑦ)
- ᑖᒃᑯᐊ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᓈᒪᒃᓴᓕᓚᐅᖏᒪᑕ ᐊᑐᖅᑕᐅᖁᔭᐅᔪᓂᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᓂᑦ ᑕᐅᕙᖓᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂᑦ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑭᒡᒐᖅᑐᐃᔪᖃᓚᐅᖏᒻᒪᑦ ᐊᐅᓯᐃᑦᑐᕐᒦᖓᖅᑐᒥᑦ ᑕᐃᑲᓂ ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (JC) ᑲᑎᒪᓂᖓᓐᓂ.
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ (GN) ᐊᒻᒪᓗ ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ (NTI) ᐅᖄᓚᖃᑎᖃᓚᐅᕐᒪᑕ ᐊᐅᓯᐃᑦᑑᑉ ᐊᖑᓇᓱᐊᖅᑐᓕᕆᔨᒃᑯᖏᓐᓂᑦ (HTO) ᔭᓄᐊᕆ 9, 2017-ᒥ ᐅᖃᖃᑎᒋᒍᑎᖃᕐᓗᑎᒃ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐊᑐᓕᖁᔨᔪᑎᓂᑦ ᐊᒻᒪᓗ ᑲᓛᑦᓯᑦ ᐊᖑᓇᓱᐊᖅᐸᒃᑕᖏᓄᑦ ᑐᒃᓯᕋᐅᑎᓂᑦ. ᐊᖑᓇᓱᐊᖅᑐᓕᕆᔨᒃᑯᑦ (HTO) ᓇᓗᓇᐃᖅᓯᓚᐅᕐᒪᑕ ᐊᓯᕈᒃᑕᐅᖁᔨᖏᓐᓂᕐᒥᖕᓂᑦ 5-ᓂᑦ ᓂᕕᖓᑖᖁᑎᒥᖕᓂᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᐊᒻᒪᓗ ᐃᓱᒫᓗᑕᐅᓇᑎᒃ ᖃᓛᑦᓯᑦ ᐊᖑᓇᓱᒍᓐᓇᖅᑕᖏᓐᓂᑦ 9-ᓂᑦ.
- ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᐅᐸᒍᑎᓯᒪᓗᑎᒃ ᐊᐱᖅᓱᕐᓂᐊᕐᒪᑕ ᐊᐅᓯᐃᑦᑑᑉ ᐊᖑᓇᓱᐊᖅᑐᓕᕆᔨᒃᑯᖏᓐᓂᑦ ᐱᒋᐊᓕᕋᑖᕐᓂᖓᓂ ᕕᕝᕗᐊᕆ 2018.
- ᐊᐱᖅᓱᕐᓂᕐᒧᑦ ᓇᐃᓈᕈᑎᑦ ᐅᓂᒃᑳᖅᓯᒪᔪᑦ ᐱᔭᐅᑎᑕᐅᓂᐊᖅᐳᑦ ᑲᑎᒪᔨᓄᖅᑕᐅᓗᑎᒃ ᐱᔭᐅᑎᑕᐅᔪᓐᓇᖅᓯᑐᐊᖅᐸᑕ ᑭᖑᓂᖓᒍᑦ ᓄᓇᓕᖕᓂᑦ ᐊᐱᖅᓱᕆᐊᖅᑐᖅᓯᒪᓚᐅᖅᑎᓗᒋᑦ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᔪᓐᓇᖁᓗᒋᑦ ᒫᔨᒥ ᑲᑎᒪᓂᕆᓂᐊᖅᑕᖏᓄᑦ.

ᐊᑐᓕᖁᔭᐅᔪᑦ:

1. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᐃᒪᓐᓇᐅᖁᔨᓕᖅᐳᑦ **ᐊᓯᕈᒃᑐᖅᑲᖏᓪᓗᓂ** ᒫᓐᓇ ᑲᐃᓐ ᐃᑭᕋᓴᖓᓂ ᐅᓄᕐᓂᖏᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ (TAH) 5 ᓇᓄᐃᑦ.

ᓇᐅᓇᐃᔭᖅᓯᒪᓂᖏᑦ

ᐅᑯᐊ ᑎᑎᕋᖅᓯᒪᔪᑦ ᓇᐅᓇᐃᔭᐃᔪᑦ ᐅᓂᒃᑳᓂ ᑕᐃᔭᐅᔪᖅ *“ᖃᐅᔨᓴᒃᑲᓂᕐᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᕐ ᐃᒪᖓ ᐊᒻᒪ Kane Basin ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ: ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᐱᓕᕆᖃᑎᒌᓐᓂᕐᒧᑦ ᓇᓄᕐᓂᑦ”* ᑐᓂᔭᐅᔪᑦ ᑲᒪᔨᓄᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᓇᓄᕐᓂᑦ 31 ᔪᓚᐃ 2016. ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᒪᔭᐅᓂᐊᖅᑐᑦ “SWG 2016” (ᑕᑯᒃᑭᑦ ᐅᖃᖅᑕᐅᓯᒪᔪᑦ ᐊᑖᓂ). ᓇᐅᓴᐃᔭᖅᓯᒪᓂᖏᑦ ᓇᐅᓇᐃᔭᐃᔪᑦ ᐃᓗᓕᖏᓐᓂ 14 ᓴᑉᑕᓂ ᑕᕝᕙᓂ ᐅᓂᒃᑳᓂᑦ, ᐅᖃᐅᓯᐅᔪᑦ ᖃᐅᔾᔭᐅᓪᓗᐊᑕᖅᓯᒪᔪᑦ, ᐊᒻᒪ ᐊᑐᓕᕐᑯᔭᓕᐅᖅᑐᑎ ᓯᕗᓂᖕᓴᒥ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) Kane Basin (KB) ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ.

ᐅᓂᒃᑳᓂᑦ ᐅᖃᖅᑕᐅᓯᒪᔪᑦ: SWG [ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᖃᑎᒌᑦᑐᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᑦ ᓇᓄᕐᓂᑦ]. 2016. ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ: ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᑦ ᑲᓇᑕᒥ−ᐊᑯᑭᑦᑐᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᑦ ᓇᓄᕐᓂᑦ. 31 ᔪᓚᐃ 2016: x + 636 pp.

ᐅᓂᒃᑳᑦ ᑎᑎᕋᖅᑕᐅᔪᑦ S. N. ᐋᑦᑭᓐᓴᓐ (SWG), E. W. ᐳᐊᕐᓐ (SWG), K. L. ᓚᐃᑐᖃᐃ (SWG), N. J. ᓚᓐ (SWG) ᐊᒻᒪ Ø. ᕗᐃᒡ (SWG) ᐊᒻᒪ ᓯᒪᑖᓃᖕᓘᖅᑐᑦ ᐊᔪᖏᓐᓂᓖᑦ T. ᐋᓄᕐᑦ, M. ᑎᒃ, E. V. ᐱᕆᐅ, H. ᓯᑑᓐ, ᐊᒻᒪ S. ᓯᑕᐃᐳᑕᓐ ᐊᒻᒪ ᓇᑲᑎᖅᑕᐅᓪᓗᓂ K. L. ᓚᐃᑐᕋᐃ ᐊᒻᒪ N. J. ᓚᓐ.

ᓵᑉᑕ 1: ᐱᒋᐊᕐᓂᖓ

ᑲᓇᑕᒥ−ᐊᑯᑭᑦᑐᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᑦ ᓇᓄᕐᓂᑦ (JC) ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᖅ ᐊᑎᓕᐅᖅᑕᐅᓪᓗᓂ “ᐊᖏᖃᑎᒌᒍᒻᒥᒃ ᑐᑭᓯᐅᒪᖃᑎᒌᓐᓂᕐᒧᑦ ᑲᓇᑕᐅᑉ ᒐᕙᒪᒃᑯᖏᓐᓂ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ, ᐊᑯᑭᑦᑐᑦ ᒐᕙᒪᒃᑯᖏᑦ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓇᓄᕐᓂᑦ” (MOU) 30 ᐅᑦᑑᐱᕆ 2009 (Anon. 2009). ᑐᕌᒐᓪᓗᐊᑕᕆᔭᖏᑦ MOU ᐱᖁᓯᐅᔾᔨᓗᑎ (1) ᐊᐅᓚᑦᑎᓗᑎ ᓇᓄᕐᓂᑦ ᐃᓗᐊᓂ Kane Basin ᐊᒻᒪ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᐅᓚᑕᐅᔪ)ᓂᑦ ᐱᐅᖅᓯᐊᕐᓂᕐᒧᑦ ᐊᒻᒪ ᓄᖑᓴᐃᖏᓐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓯᕗᓂᖕᓴᒧᑦ ᐊᒻᒪᓗ (2) ᓴᖅᑭᑎᑦᑎᓗᑎ ᐊᐅᓚᓂᖃᑦᑎᐊᖅᑐᒥᑦ ᐊᐅᓚᑕᐅᓂᖏᓐᓄᑦ ᐱᖁᓯᐅᔾᔨᓂᐊᖅᑐᑦ ᒪᓕᓪᓗᑎ ᐊᑐᖅᑕᐅᕆᐊᓕᓐᓂᑦ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ. JC ᓴᖅᑭᑎᑦᑎᓚᐅᖅᑐᑦ ᑕᓪᒪᓂᑦ ᐃᓚᒋᔭᐅᔪᓂᑦ ᖃᐅᔨᓴᖅᑎᓂᑦ ᐱᓕᕆᖃᑎᒌᑦᑐᓂᑦ (SWG) ᐅᖃᐅᔾᔨᒋᐊᕈᓐᓇᕐᓗᑎ ᖃᐅᔨᓴᖅᑎᓃᖕᓘᖅᑐᓂᑦ ᐊᒻᒪ ᐊᑐᓕᕐᑯᔭᓕᐅᕐᓗᑎ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐊᒻᒪ Kane Basin (KB) ᓇᓄᖏᓐᓂ. JC ᐱᓕᕆᑎᑦᑎᓚᐅᖅᑐᑦ SWG−ᑯᓐᓂᑦ ᐊᑐᖅᑐᑎ ᐱᐅᓛᖑᔪᓂᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ:
(1) ᐱᕐᑯᔨᓗᑎ ᖃᑎᑦᑐᒋᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᕐᑯᕝᕙᓯᓐᓂᖏᑦ KB−ᒧᑦ ᐊᒻᒪ BB ᓇᓄᖏᓐᓄᑦ ᑐᓂᓯᓗᒋᑦ JC ᐅᓂᒃᑳᓂᑦ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐊᑐᓕᕐᑯᔭᖏᓐᓄᑦ; ᐊᒻᒪ (2) ᖃᐅᔨᓴᖅᑎᓃᖕᓘᖅᑐᓂ ᐅᖃᐅᔾᔨᒋᐊᕐᓗᑎ JC−ᒧᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᑦᑐᖅᑕᐅᔪᑦ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᓇᓄᕐᓄᑦ.

ᒪᐃᖑᑎᓪᓗᒍ 2010 SWG ᐅᓂᒃᑳᓕᐅᓚᐅᖅᑐᑦ JC−ᑯᓐᓄᑦ (SWG 2010) ᓇᐅᓇᐃᔭᐃᔪᓂᑦ ᒦᓐᓇᐅᔪᒥᑦ ᐊᑐᐃᓐᓇᐅᔪᑦ ᑐᑭᓯᒋᐊᕈᑎᓯᐊᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓄᑦ. SWG ᓄᖅᑲᐅᓯᓚᐅᖅᑐᑦ ᒥᓯᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᒻᒪ ᕐᑯᕝᕙᓯᓐᓂᕆᐊᖃᖅᖏᑦ BB−ᒧᑦ ᐊᒻᒪ KB−ᒧᑦ ᐅᓪᓗᒥᒨᖓᓚᐅᖏᑦᑐᑦ. ᐃᓚᒋᐊᕐᓗᒍ, ᐊᒥᓲᓂᖏᓐᓄᑦ ᓈᒻᒪᓐᓂᓇᔪᓐᓇᖅᑕᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᖅ (PVA) ᑖᒃᑯᐊ ᐊᑐᖅᑐᒋᑦ ᐊᒻᒪ ᒦᓐᓇᐅᔪᒥᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᑦ ᕐᑯᕝᕙᓯᓐᓂᑦ 100%−ᒥ ᕐᑯᕝᕙᓇᐊᓚᐅᖅᑐᑦ ᐱᓪᓚᓇᖑᐊᖅᑎᑕᐅᔪᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᑕᒪᒃᑭᓐᓄ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᑦᑏᓐᓇᕈᓇᐊᕋᔭᖅᑐᑎᑦ ᕐᑯᓕᑦ ᐊᕐᕋᒍᐃᑦ ᖃ ᖏᖅᓯᒪᓕᖅᑲᑕ (PBSG 2010). SWG ᖃᐅᔾᑲᐃᓚᐅᖅᑐᑦ JC−ᑯᓐᓂᑦ ᐱᓪᓚᓇᖑᐊᖅᑎᑕᐅᔪᑦ ᐊᐅᓚᓂᖃᑦᑕᔪᓃᑦ ᐅᖓᑖᓅᖏᑦᑐᑦ 10−15 ᐊᕐᕋᒍᐃᑦ ᐅᖓᑖᓅᑦ ᑭᖑᓪᓕᖅᐹᒥ ᑐᑭᓯᒋᐊᕈᑏᑦ ᐱᔭᐅᑎᓪᓗᒋᑦ, ᑐᑭᖃᖅᑎᑦᑎᔪᖅ PVA−ᖑᔪᑦ ᐅᓪᓗᒥᒨᖓᓚᐅᖏᑦᑐᑦ ᑕᐃᒪᐃᓐᓂᖓᓄᓪᓗ ᑐᓐᖓᕕᒋᔭᔅᓴᐅᓇᑎ. ᐃᓚᒋᐊᕐᓗᒍ, SWG

ᓇᐅᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᑦᑐᖅᑕᐅᓂᐅᔪᓐᓇᖅᑐᑦ ᐊᕙᑎᒧᑦ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ, ᐊᒻᒪ ᓄᓇᒥ ᕐᑯᕝᕙᓯᓐᓂᖏᑦ ᐊᖏᔪᑲᓪᓚᒻᒥᑦ ᐊᓯᔾᔨᕈᓐᓇᖅᓯ,ᔪᖅ ᑭᖑᓪᓕᖅᐹᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᑖᒃᑯᓂᖓ ᐊᒥᓲᓂᖏᓐᓂᑦ 1990−ᖏᓐᓂ. ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᓪᓗᒥᒨᖓᖏᑦᑐᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᒻᒪ ᕐᑯᕝᕙᓯᓐᓂᓇᔭᓇᐊᖅᑕᖏᑦ (PBSG 2010) ᐊᒻᒪ ᐊᖏᔪᒥᑦ ᐊᕙᑎᖓᓂᑦ ᐊᓯᔾᔨᖅᑐᑦ ᖃᐅᔨᔭᐅᔪᑦᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᕐᑯᓕᓂᑦ ᐊᕐᕋᒎᓕᖅᑐᓕᑦ, SWG ᐊᑐᓕᕐᑯᔮᓚᐅᖅᑐᑦ ᓄᑖᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ

ᐊᒥᓲᓂᖏᑦ, ᐊᒥᓲᓂᖏᓐᓄᑦ ᓇᓗᓇᐃᔭᐃᓂᖅ, ᐊᒻᒪ ᖁᕝᕙᓯᖕᓂᓇᒋᐊᖃᖅᑕᖏᑦᐱᔭᐅᓗᑎ BB−ᒧᑦ ᐊᒻᒪ KB (SWG 2010).

ᐊᑐᐃᓐᓇᕈᖅᐸᓕᐊᓪᓗᓂ ᐱᒋᐊᖅᑎᑦᑎᓂᕐᒥᑦ ᐊᑕᖐᔪᒥᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᓈᓴᐃᓂᕐᒥᑦ BB−ᒥᑦ ᐊᒻᒪ KB, ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᖃᐅᔨᓴᕈᑎᖃᓚᐅᖅᑐᑦ ᓯᑯᒃᑯᑦ ᓂᒋᖓᑕ ᑲᓇᓐᓇᖓᓂ ᕿᑭᖅᑖᓗᒻᒥ ᐅᐱᕐᖔᖓᓂ 2010 ᕿᒥᕐᕈᓪᓗᑎ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᑦ. ᑕᕝᕙᓐᖓᑦ ᖃᐅᔨᓴᓇᐊᓐᖓᐅᑎᐅᔪᒥᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᖅ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᐃᓚᒋᔭᖓ ᐊᒥᓱᒐᓚᓐᓂᑦ ᓈᒻᒪᑦᑐᓂ ᓇᓖᕌᕋᔅᓴᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ BB−ᒥ ᐊᒻᒪ KB. ᐅᓂᒃᑳᓕᐅᖅᑐᖃᕋᑖᓕᐅᖅᑐᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐊᔾᔨᒌᖏᑦᑐᓂᑦ ᓇᓖᕌᕋᔅᓴᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐊᑐᓕᖁᔭᐅᔪᑦ ᓈᒻᒪᑦᑐᓂᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᓄᑦ (MR), ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓃᑦ, ᐊᒻᒪ ᐊᐅᖏᑎᒍᑦ M-R (SWG 2011). SWG ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᖅ ᖃᓄᐃᓕᖓᓂᖓ MR ᐱᐅᓛᖑᓚᐅᖅᑐᖅ ᐊᑐᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᒻᒪ ᖁᕝᕙᓯᖕᓂᓇᒋᐊᖃᖅᑕᖏᓐᓄᑦ BB−ᒥ. ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓗᓴᐃᖅᑕᐅᓚᐅᖅᑐᖅ ᐱᐅᓛᖑᓪᓗᓂ ᐊᑐᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᕕᒋᓗᒍ KB. SWG ᐃᓕᓴᖅᓯᓚᐅᖅᑐᑦ ᐱᔪᓐᓇᕈᑎᒃᑲᓐᓂᖏᓐᓂᑦ ᐊᒻᒪ ᐱᔪᓐᓇᖏᔾᔪᑎᖏᓐᓂᑦ ᑕᒪᒃᑭᓐᓄᑦ ᐊᑐᖅᑕᐅᔪᓄᑦ. ᐅᖃᖃᑎᒌᔾᔪᑎᒋᓚᐅᖅᑐᒍ SWG ᐅᓂᒃᑳᖓ, JC ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᐅᖏᑎᒍᑦ MR ᐊᑐᖅᑕᐅᖔᕈᒪᔪᖅ ᑕᒪᒃᑭᓐᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐃᑲᔪᖅᓯᑕᐅᑦᑎᐊᖏᓐᓂᖏᓐᓄᑦ ᐃᓄᓐᓄᑦ ᓄᓇᕘᒻᒥ ᑎᒥᑎᒍ MR ᐊᑐᖅᑕᐅᓂᖓ, ᐊᒻᒪ ᐱᔅᑎᕆᓪᓗᒍ ᓯᑯᖓᑕ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐊᑦᑕᓇᖅᑑᑎᑦᑎᑐᐃᓐᓇᓇᑎᐊᖅᑕᑦ ᓯᓚᖏᑦᑐᓂ ᐱᓗᓂ ᐊᒥᓲᓂᖏᓐᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᒃᑯᑦ.

ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, JC ᐊᑐᓕᖁᔨᓚᐅᖅᑐᑦ ᐊᑎᓕᐅᖅᓯᒪᔪᓄᑦ MOU−ᒥ ᐊᖅᖀᒍᓄᑦ ᐱᖓᓱᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᐊᒥᑦ ᐊᐅᖏᑎᒍᑦ MR BB−ᒥ ᐊᒻᒪ KBᒥ ᓴᖅᑭᖅᑎᑕᐅᓗᓂ.

ᐊᖅᖀᒍᒪᓚᖓᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᐊᖅ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᖅ SWG−ᑯᓐᓄᑦ ᐃᓗᓕᖅᑕᐅᓚᐅᖅᑐᖅ ᐱᖓᓱᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ: (1) ᐊᐅᖏᓐᓂ ᖃᐅᔨᓴᖅᑕᑦᑕᕐᓂᖅ ᓇᓄᕐᓂᑦ ᐱᓗᑎ ᖃᐅᔨᓴᕋᕋᔅᓴᓂᑦ ᑲᓇᓐᓇᖓᓂ ᕿᑭᖅᑖᓗᒃ, ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐊᒻᒪ KB-ᒥᑦ; (2) ᐋᖅᑭᑦᑕᐅᓂᖏᑦ ᖁᓐᒧᐊᑦᑎᑕᐅᓯᒪᔫᒃᑯᑦ ᖃᐅᔨᓴᐅᑏᑦ ᐊᖑᑎᓄᑦ ᐊᒻᒪ ᐊᕐᓇᓄᑦ ᓇᓄᕐᓂᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐊᒻᒪ KB-ᒥᑦ ᐊᒻᒪᓗ (3) ᐊᖑᓇᓱᑦᑎᓂᑦ ᓄᐊᑕᐅᓂᖏᑦ ᖃᐅᔨᓴᕋᕋᐃᑦ ᐊᑐᓂ (ᓇᓄᑦᑕᐅᔪᓂᑦ) BB-ᒥ, KB ᐊᒻᒪ ᐊᑦᑐᐊᓂᓃᑦ. ᐱᖓᓱᓄᑦ ᐊᖅᖀᒍᓄᑦ ᖃᐅᔨᓴᕈᒪᓂᖃᓚᐅᖅᑐᑦ ᐅᑭᐊᖅᓯᓴᑖᖅᑎᓪᓗᒍ 2011 ᐅᑯᓂᖓ ᑐᕌᒐᖃᖅᑐᑎᑦ: 1) ᒥᔅᓴᐅᓴᑦᑕᐅᓗᑎ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᓪᓗ (ᐊᒻᒪ ᐊᖅᖀᒍᖏᑦ) ᓇᓄᖏᑦ BB−ᒥ ᐊᒻᒪ KBᒥ; 2) ᐊᔾᔨᒌᖑᐊᑎᓪᓗᒋᑦ ᓄᑖᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ (1991−1997), ᖃᐅᔨᓴᓗᒋᑦ ᐊᒥᓲᓂᖏᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ; 3) ᓇᓗᓇᐃᖅᓗᒋᑦ ᑭᓪᓕᖏᑦ BB−ᒧ ᐊᒻᒪ KB−ᒧᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᒻᒪ ᖃᐅᔨᓴᒃᑲᓂᕐᓗᒋᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓂᖏᓐᓄᑦ ᑖᒃᑯᐊ; 4) ᒥᔅᓴᐅᓴᓗᒋᑦ ᐆᒪᓂᓇᕙᑦᑕᖏᑦ ᐊᒻᒪ ᕿᑐᕐᖏᐅᕐᓂᖏᑦ (ᐊᑕᖏᕈᓐᓇᖅᑐᓕᒫᓂᑦ) ᐊᑐᖅᑕᐅᓗᑎ ᐊᒥᓲᓂᖏᓐᓄᑦ ᓈᒻᒪᓐᓂᖏᓐᓄ ᖃᐃᔭᐃᑐᑎᓗ; ᐊᒻᒪ 5) ᖃᐅᔨᓴᕐᓗᒋᑦ ᓇᓄᐃᑦ ᓇᒧᖓᔾᔭᑦᑕᕐᓂᖏᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐊᕙᑎᖓᓄᑦ ᐊᔾᔨᒌᖏᑦᑐᒪᓂᖓᓄᑦ, ᐱᓗᐊᖅᑐᒥ ᓯᑯᖓ ᖃᓄᐃᓐᓂᖓ, ᓄᓇᖓ ᐊᒻᒪ ᓇᒧᖓᕐᕕᑦᑕᖏᑦ ᐊᒻᒪ ᐊᑐᐃᓐᓇᐅᓂᖏᑦ ᓂᕿᖏᑦ. ᒫᓐᓇᐅᔪᒥᑦ ᐅᓂᒃᑳᑦ (SWG 2016) ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᑐᖅᑕᓕᒫᖑᔪᓂᑦ, ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᐊᑐᖅᑕᐅᔪᑦ, ᖃᓄᐃᓐᓂᖏᑦ ᐊᒻᒪ ᐅᖃᐅᓯᐅᓂᖏᑦ ᑖᔅᓱᒥᖓ ᐊᖅᖀᒍᒪᓚᖓᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᐊᖅ.

ᓵᑉᑕ 2: ᐊᒥᓲᓂᖏᑦ ᓇᓗᓇᐃᓴᖅᑕᐅᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin

ᓄᑖᓂᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᖃᐅᔨᓴᕐᓂᖅᑕᖃᓯᒪᖏᑦᑐᖅ BB–ᒥ ᐅᕝᕙᓘᓐᓃᑦ KBᒥ ᐅᓪᓗᒥᒧᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᓇᓗᓇᐃᓴᖅᑕᐅᓂᖏᓐᓄᑦ 1997–ᒥᑦ. ᓵᑉᑕ 2 ᐊᑐᖅᑐᖅ ᒫᓐᓇᕋᑖᖑᔪᒥ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᐅᖏᑎᒍᑦ, ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ, ᐊᒻᒪ ᐱᓴᐅᒃᑲᓂᖅᑐᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓇᓐᓄᑦᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᓂᕐᒨᖓᔪᓂᑦ BB–ᒥ ᐊᒻᒪ KBᒥ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᖅᑐᑎ ᐊᒥᓲᓂᖏᓐᓄᑦ ᑭᓪᓕᖏᑦ.

ᓵᑉᑕ 2 ᓯᕗᓪᓕᕐᒥ ᓇᓗᓇᐃᓴᐃᔪᖅ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᔾᔨᕆᓐᖑᐊᖅᑎᑕᐅᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂ ᐊᒻᒪ 2000–ᖏᓐᓂ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᖏᑦ ᓈᒻᒪᒃᑲᓗᐊᕐᒪᖗᑕ. ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᕙᓚᐅᖅᑐᑦ ᖁᖓᓯᕈᓕᖅᑕᐅᓪᓗᑎ BB ᓇᓄᖏᓐᓂᑦ ᐊᒻᒪ ᐊᔾᔨᕇᖏᑦᑐᓂ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᑖᒃᑯᓂᖓ ᒪᕐᕉᔪᓐᓂᒃ (1990–ᖏᓐᓂ ᐅᑭᐊᕐᓵᒃᑯᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐊᒻᒪ ᐅᐱᕐᖖᒫᒃᑯᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ; 2000–ᖏᓐᓂ ᐅᐱᕐᖖᒫᒃᑯᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᓯᑯᖓᓂ) ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᓯᕗᓪᓕᖅᐹᖑᓚᐅᖅᑐᖅ ᖃᐅᔨᓴᕐᓗᒋᑦ ᓇᓄᐃᑦ ᐱᓴᐅᖁᓚᐅᖅᑐᑦ ᐅᐱᕐᖖᒫᒃᑯᑦ ᑕᒪᔾᓱᒥᖓᔅᓴᐃᓐᓇᖅ ᓇᔪᖅᓯᒪᖗᑕ ᐱᓴᐅᓚᐅᖅᑐᓂᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ. ᖃᓄᐃᓐᓂᕆᓴᖏᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ 1990–ᖏᓐᓂ ᐊᒻᒪ 2000–ᖏᓐᓂ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᓈᒻᒪᓐᓂᖃᖅᑐᑎᑦ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐊᔾᔨᕆᓐᖑᐊᑎᑦᑐᕆᑦ ᐊᒥᓲᓂᖏᑦ ᓇᓗᓇᐃᓴᖅᑦᓯᒪᓂᖏᑦ ᐊᕐᕋᒎᒐᓚᐅᓕᖅᑐᓂᑦ' ᖃᓂᒋᓴᐸᓗᖏᓐᓂ 92% ᐃᓐᓇᐅᔪᓂᑦ ᐊᕐᓇᓂᑦ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐅᐱᕐᖖᒍᑦ ᑖᔅᓯᒥᖓᔅᓴᐃᓐᓇᖅ ᓇᔪᖅᓯᔪᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐃᓐᓇᖏᑦ ᐊᕐᓇᐃᑦ ᐱᓴᐅᔪᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ 1990–ᖏᓐᓂ.

ᖃᐅᔨᓴᓂᖃᓕᓚᐅᕐᒥᔪᒍᑦ ᐊᑐᖅᑐᑕ ᖃᓄᐃᓕᖓᓂᐅᓯᒪᖃᑦᑕᖅᑐᑐᖃᕐᓂᑦ ᐊᒻᒪ ᒫᓐᓇᕋᑖᖑᔪᒥ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᖃᐅᔨᓴᖅᑐᑎᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᖁᓕᓂᒃ ᐊᕐᕌᒍᓯᒪᔪᓂᑦ ᒪᕐᕉᓐᓂ ᐊᑐᖅᑕᓕᒫᖑᔪᓂᑦ (ᓲᕐᓗ, ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᖏᓂᖏᑦ) ᐅᖓᓯᓐᓂᖓ ᖁᓕᓇᖃᑦᑕᐅᑎᓂᖏᑦ, ᐊᓯᔾᔨᕐᓂᖏᑦ ᓇᒦᓐᓂᖏᑦ ᐊᕐᕌᒍᓕᓚᓐᓄᑦ, ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐅᑎᖅᑕᕐᕕᖏᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᑎᖅᑕᖅᑐᓂ ᑭᓪᓕᖏᑎᒍᑦ. ᖃᐅᔨᓴᕈᑎᒦᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᖏᔪᒥ ᒥᑭᓪᓕᒋᐊᖅᓯᒪᔪᑦ 2000–ᖏᓐᓂᑦ BB 95% ᐅᖓᓯᓐᓂᖓ (ᓲᕐᓗ, ᐊᑐᖅᑕᐅᔪᑦ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓄᑦ ᓇᓄᕐᓄᑦ) ᑕᖅᑭᓕᒫᖏᓐᓂᑦ ᐊᒻᒪ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐊᔾᔨᒋᑎᓐᖑᐊᖅᑐᒍ 1990–ᖏᓐᓂᑦ. ᖃᑦᑏᓐᓇᕈᓇᐊᖅᓯᒪᓛᖑᔪᑦ 60% ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᖏᓂᖏᑦ ᐊᐅᓴᒃᑯᑦ. ᖁᓕᓇᖃᑦᑕᐅᑎᓂᐅᔪᑦ 1990ᖏᓐᓂ ᐊᒻᒪ 2000–ᖏᓐᓂᑦ BB ᐅᖓᓯᓐᓂᓇᓚᐅᖅᑕᖏᑦ < 50% ᑕᖅᑭᓕᒫᓂᑦ, ᑕᑯᔅᓴᐅᑎᑦᑎᔪᖅ ᑕᒪᒃᑭᓐᓂ ᐊᖏᓪᓕᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᓴᖑᖃᑦᑕᕐᓂᖏᑦ BB–ᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ 2000–ᖏᓐᓂᑦ. ᑖᒃᑯᐊ ᓴᖑᓂᐅᔪᑦ ᐊᑦᑐᐊᓂᓖᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂᑦ ᓯᑯᐃᖅᑕᑦᑕᕐᓂᖓᓄᑦ, ᐊᓯᔾᔩᔪᖅ ᐅᖓᓯᓐᓂᖏᓐᓄᑦ Bbu ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᓴᖑᑎᑦᑎᓪᓗᑎ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ BB–ᒥ ᓇᓄᖏᑦ ᐅᐊᓐᓇᖅᐸᓯᐊᓄ ᐊᕐᕌᒍ ᐃᓗᐊᓂ. ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᑭᓪᓕᖏᑦ, BB ᓇᓄᖏᑦ 2000–ᖏᓐᓂᑦ ᖃᑦᑏᓐᓇᐅᓂᖅᓴᐅᓚᐅᖅᑐᑦ BB–ᒥᖑᓐᓃᕐᓂᖏᓐᓄᑦ ᓴᓂᐊᓂ 1990–ᖏᓐᓂ. ᐱᓗᐊᖅᑐᒥ, ᖃᑦᑏᓐᓇᕈᓇᐊᓚᐅᖅᑐᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ ᕿᑭᖅᑖᓘᑉ ᐃᒪᖓᓂ ᐊᒻᒪ ᑕᓪᓗᕈᑎᐅᑉ ᑕᓇᐅᖓᓂ BB–ᒥ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᓯᑯᖃᖏᓐᓂᖓᓄᑦ, ᑖᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅ BB–ᒥ ᐊᒥᓲᓂᖏᑦ ᑲᑎᓐᖓᓴᐅᒍᓐᓃᖅᑐᑦ, ᖃᓄᐃᓘᖃᑦᑕᐅᑎᖏᓂᖅᓴᐅᓪᓗᑎ ᐊᒻᒪ ᐊᓯᖏᓐᓂ ᓇᓄᕐᓂᑦ.

KB–ᒥᑦ, ᖁᕝᕙᓯᓐᓂᓇᓚᔪᑦᑕᖓ 95% ᐅᖓᓯᓐᓂᖏᑦ ᐊᖏᓪᓕᒋᐊᖅᓯᒪᔪᑦ 1990–ᖏᓐᓂ. ᐊᖏᓪᓕᒋᐊᒃᓂᖓ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᑐᖅᑕᐅᔪᑦ 2000–ᖏᓐᓂ ᐊᕐᕌᒎᓕᒫᖑᕙᑦᑐᒥᑦ, ᑭᓯᐊᓂᓕ ᐊᒥᓲᓂᖅᓴᐅᔪᑦ ᐊᐅᓴᒃᑯᑐᐊᖅ (ᔫᓂ-ᓯᑎᐱᕆ), ᓇᒧᖓᕐᓂᕆᕙᑦᑕᖏᑦ ᒪᕈᐊᑎᖅᓗᐊᖅᓯᒪᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂ ᐊᒻᒪ 2000–ᖏᓐᓂᑦ ᓇᒧᖓᓂᕆᕙᑦᑕᖏᑦ ᐊᖏᓪᓕᒋᐊᕈᑎᒋᓴᖓ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᓯᑯᖓ, KB ᐊᕐᕌᒍᑕᒫᑦ ᓯᑯᖃᑦᑕᖏᓐᓂᖅᓴᐅᕙᓪᓕᐊᓕᖅᑐᖅ (BB-ᑎᑐᑦ) ᓯᑯᓕᒫᐸᓗᖓ ᐊᐅᖃᑦᑕᖅᑦᑐᓂ ᐊᐅᓴᑕᒫᑦ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᐅᔪᑦ ᐊᓯᔾᔨᖅᑐᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᓚᔪᑦᑐᒃ ᐊ–ᔨᕆᖏᓐᓂᖏᓐᓂᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᓄᑦ ᓇᓄᕐᓄᑦ ᕿᑭᖅᑕᓂᑦ ᓴᓂᐊᓂ

ᓯᑯᐃᖅᑕᑦᑕᕐᓂᖓᓄᑦ: ᓇᓄᐃᑦ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ ᓯᑯᒥ ᐊᖏᓂᖅᓴᒥᑦ ᐊᒻᒪ ᐊᔾᔨᕆᖏᓂᖅᓴᓂᑦ ᐊᖏᕋᖃᖅᑐᑦ ᓯᑯᒃᑰᑲᐃᓐᓇᐸᓪᒪᑕ ᓇᓄᐃᑦ ᕿᑭᖅᑕᒐᓚᓐᓃᑦᑐᑦ ᒥᑭᓂᖅᓴᓂᑦ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ ᐊᓯᔾᔨᒃᓂᖃᖏᓂᖅᓴᐅᓪᓗᑎ.

ᓲᓕ ᐊᖏᔪᒥᑦ ᖄᖓᒨᕈᑎᖃᖅᑐᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ KB ᓇᓄᖏᓐᓂ 1990–ᖏᓐᓂ ᐊᒻᒪ 2000–ᖏᓐᓂ (50-98%

ᖁᓕᕇᖃᑦᑕᐅᑎᔪᑦ ᐊᕐᕌᒍᒐᓚᓐᓂᑦ), ᓇᓗᓇᐃᖅᓯᔪᑦ ᓇᓄᐃᑦ ᑕᒪᔅᓯᒥᙵᔅᓴᐃᖅ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ KB−ᒥ. ᐊᖏᔪᑲᓪᓚᒻᒥᑦ ᐅᐊᓐᓇᙵᓅᓚᐅᖅᑐᑦ KB ᓇᓄᖏᑦ ᓇᒦᓐᓂᖏᑦ 2000−ᖏᓐᓂᑦ ᐅᐱᕐᙶᒃᑯᑦ ᐊᒻᒪ ᐊᐅᔾᒃᑯᑦ (ᒥᑭᓂᖕᓴᐅᔪᑦ BB−ᒥ) ᑭᓯᐊᓂ ᐊᔾᔨᒌᖏᒻᓐᓂᖏᑦ ᖁᕝᕙᕆᐊᖅᓯᒪᔪᑦ. ᐊᓯᔾᔨᕐᓂᖅᑳᖏᑦᑐᖅ ᐅᑭᐅᖑᑎᓪᓗᒍ. ᑖᒃᑯᐊ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᐊᓯᔾᔨᓚᐅᖏᑦᑐᑦ ᐊᑦᑐᖅᑕᐅᓴᕋᐃᓐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᓇᓄᓇᐃᑦ ᐱᔭᐅᑎᓪᓗᒋᑦ ᑲᓇᑖᓐᓇᙵᓂ KB ᐱᖃᓯᐅᔾᔭᐅᓚᐅᖏᑦᑐᑦ 2000−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, ᐊᔾᔨᑭᓕᖅᑎᓪᓗᒍ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᐱᔭᐅᔪᑦ 1990−ᖏᓐᓂ. ᑕᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅ ᖃᐅᔨᔭᓪᓗᐊᑕᐅᔪᑦ KB−ᒧᑦ ᐊᔭᐅᖅᑕᐅᓚᐅᖏᑦᑐᑦ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐱᔭᐅᔪᑦ ᓇᒦᓐᓂᖏᑦ..

ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᐃᓂᖃᓚᐅᖅᑐᑦ ᐊᑐᖅᑐᑎ 8 polymorphic microsatellites ᐃᓚᒋᓪᓗᓂᐅᒃ ᐊᖏᔪᓂᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᐅᖏᑎᒍᑦ ᑐᑭᓯᒋᐊᕈᑎᑦ ᓄᐊᑕᐅᔪᑦ MR ᖃᐅᔨᓴᐃᓂᐅᔪᓄᑦ (ᓵᑉᑕ 5 ᐊᒻᒪ 10). ᐊᑕᖏᖅᑐᒥᑦ, ᐊᑦᑎᑦᑑᓚᐅᖅᑐᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᒥᓲᖏᓐᓂᖏᓐᓄᑦ ᒥᑭᔪᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᑦ. BB ᐊᒻᒪ KB ᓇᓄᖏᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᔭᐅᔪᓐᓇᓚᐅᖏᑦᑐᑦ. ᐊᒻᒪᓗᑦᑕᐅᖅ ᒪᓕᑦᑐᒍ ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᕐᓂᐅᓯᒪᔪᓂᑦ, BB ᖃᒻᒪ KB ᓇᓄᖏᑦᐊᐅᖏᑎᒍᐳᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᖅᑐᑦ ᓇᓄᖏᓐᓂ ᓴᓂᕈᑎᐅᑉ ᐃᒪᙵᓂ ᐊᒻᒪ ᕿᑭᖅᑖᓗᑉ ᑕᕆᐅᙵᓂ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᔭᐃᓂᕐᒥᑦ BB−ᒧᑦ ᐊᒻᒪ KB−ᒧ ᐊᓯᔾᔨᖅᓯᒪᖏᑦᑐᑦ.

ᐱᔭᐅᒃᑲᓂᖅᑐᑦ ᐊᒻᒪ ᐊᙳᓇᓯᑦᑕᐅᔪᓂᑦ ᑐᑭᓯᒋᐊᖃᕈᑏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐱᙵᓱᓂᑦ ᐊᕐᕌᒍᓂ MR ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ BB−ᒥᑦ ᐊᒻᒪ KB ᐊᑦᑎᑦᑑᓚᐅᖅᑐᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᑦ ᖁᑎᕐᓂᖏᓪᓘᓐᓂᑦ ᓇᓄᐃᑦ ᓯᓚᑖᓂ ᐊᒥᓲᓂᖏᓐᓂᑦ ᓇᑭᙶᕐᕕᑎᑎᐊᓐᖓᖅᓯᒪᔭᖏᓐᓂᑦ, ᑲᑎᑦᑐᒋᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᑦ ᐊᕐᕌᒍᖏᓐᓂ 1 ᐊᒻᒪ 2 ᑖᒃᑯᓂᖓ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ (2011−2012 BB−ᒥᑦ ᐊᒻᒪ 2012-2013 KB−ᒥᑦ) ᐊᔾᔨᐸᓗᒋᓚᐅᖅᑕᖏᑦ ~34% ᐊᒻᒪ ~25% ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᐊᖏᓂᖏᑦ BB ᐊᒻᒪ KB, ᑐᖏᓕᕇᑦᑐᑎ. ᑖᓐᓇ ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ, ᐅᑎᖅᑕᖃᑦᑕᖅᕕᒋᔭᖏᑦ ≤ 1% ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᐊᒻᒪ ᐅᑎᖅᑐᓂᑦ BB ᓇᓄᖏᓐᓄᑦ. ᐊᔾᔨᐸᓗᙵᓗ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ, ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐅᑎᖅᑕᒃᑕᕐᓂᖏᑦ < 4% ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᐊᒻᒪ ᐅᑎᖅᑐᓂᑦ. ᑖᓐᓇ ᒪᓕᓐᓂᖅᒥᔪᖅ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᑐᓐᖓᓂᖅᑐᑎᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐅᑎᖅᑕᖃᑦᓴᑦᑕᕐᓂᖏᑦ BB ᐊᖏᔪᑲᓪᓚᒻᒥᑦ ᖃᑦᑏᓐᓇᕈᓇᐊᖅᓯᒪᔪᑦ 1990−ᖏᓂ, ᐱᔾᔪᑎᖃᓗᐊᖅᑐᖅ ᓯᑯᖃᓗᐊᕈᓐᓅᕐᓂᖓᓄᑦ ᐅᑭᐅᒃᑯᑦ ᐊᒻᒪ ᓯᑯᑕᖃᖏᓐᓂᖓᓄᑦ ᐊᐅᔾᒃᑯᑦ.

ᐊᑕᖏᖅᑐᒍ, ᖃᐅᔨᔭᐅᔪᑦ ᑐᓐᖓᓂᖅᑐᑎᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ, ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ, ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᒫᓐᓇᐅᔪᒥ ᑭᒡᓕᕆᔭᖏᑦ BB−ᒧ ᐊᒻᒪ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ ᐊᑐᐊᓂᖅᖔᔭᖅᑐᑦ ᓈᓄᑦᑕᐅᕚᑦᑐᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔾᔪᑕᐅᔪᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ.

ᓵᑉᑕ 3: ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖅ 1990−ᖏᓐᓂ ᓴᓂᕈᑎᐅᑉ ᐃᒪᙵᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᐃᓚᐃᓐᓇᐅᓂᙵᓄ ᐊᒻᒪ ᓈᒪᓐᓂᙵᓄᑦ 2000−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ

ᓵᑉᑕ 3 ᖃᐅᔨᓴᐃᔪᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᒻᒪ ᑎᒥᙵᓃᙶᖅᑐᓂᑦ ᐱᔾᔪᑕᐅᔪᓂᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᑕᒪᙵᑐᐃᓇᐅᖏᑦᑐᖅ ᐊᒻᒪ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐱᔭᕇᖅᑕᐅᓯᒪᖏᑦᑐᓂᑦ, ᓴᖅᑭᑦᑎᔪᓐᓇᑐᐃᓐᓇᕆᐊᓖᑦ ᐃᓚᐃᓐᓇᙵᓂ ᒥᔅᓴᐅᓴᓐᓂᕐᒥᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐊᒻᒪ ᐊᒥᓲᓂᖏᓐᓄᑦ MR ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ BB ᓇᓄᖏᓐᓂᑦ.

ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ BB−ᒥᑦ 2011-2013−ᒥ ᐱᖃᓯᐅᔾᔨᓚᐅᖅᑐᖅ ᐸᕙᓴᐃᖏᑦᑐᓂ ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᒐᔅᓴᓂᑦ ᐊᒻᒪ ᐊᑐᖅᑐᑎ ᐆᒪᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᖁᖓᔪᓂᑦ ᖃᐅᔨᓴᒐᔅᓴᓂᑦ ᐊᔾᔨᖓᑎᑐᑦ ᓴᓇᓯᒪᓂᖓᓂᑦ. ᑖᒃᑯᐊ ᑕᒪᒃᑭ ᖁᕝᕙᕆᐊᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᒐᔅᓴᓂᖏᑦ ᐊᖏᓂᖏᓐᓂᑦ ᐊᒻᒪ ᐊᑦᑎᓪᓕᑎᑦᑎᓪᓗᓂ ᖃᓄᐃᑦᑐᓂᖏᑦ ᐃᓚᐃᓇᖏᓐᓃᑎᑦᑎᒍᓐᓇᖅᑐᑦ.
ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ, ᐊᔅᓯᕈᕈᑕᐅᔪᑦ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ MR ᐊᑐᖅᑕᐅᔪᓐᓇᑐᓄᑦ BB ᑐᑭᓯᒋᐊᕈᑎᖏᓐᓄᑦ BB−ᒧᑦ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᒫᓐᓇᐅᔪᒥᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᑲᑎᖓᔪᓂᑦ 1993-2013 ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ. 1990−ᖏᓐᓂ ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖃᓚᐅᖅᑐᑦ BB ᑐᑭᓯᒋᐊᕈᑎᖏᓐᓂᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ, ᐊᔾᔨᒋᑎᖑᐊᖅᑐᒍ 2010−ᖏᓐᓂ ᑐᑭᓯᒋᐊᕈᑏᑦ, 1990−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᒥᑭᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᒐᔅᓴᐃᑦ ᐊᖏᓂᖏᑦ, ᓄᓇᖓᓂ ᒥᑭᓂᖅᓴᓂ ᖃᐅᔨᓴᖅᑐᑎ, ᐊᒻᒪ ᐊᕙᑎᖓᓂ ᐊᔾᔨᒋᖏᑕᖓᓂ, ᑲᑎᖓᑎᓪᓗᒋᐱᑦ ᓴᖅᑭᑎᑦᑎᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᖅᓯᓕᕆᖏᑦᑐᓂᑦ ᐊᒻᒪ ᑭᓪᓕᖃᖅᑎᑦᑎᑐᐃᓐᓇᕆᐊᖃᕐᓂᖓᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, **ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᑖᒃᑯᐊ ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᕐᒧᑦ** (ᓵᑉᑕ 3) ᑕᕝᕙᖓᑦᑕᐅᑎᒋ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᔾᔨᒌᖑᐊᖅᑎᒐᔅᓴᐅᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ BB−ᒥ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᐃᖅᓱᒋᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᓄᓇᐃᑦ ᐆᒪᓂᖏᑦ ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ.

1990−ᖏᓐᓂ MR ᖃᐅᔨᓴᒐᔅᓴᐃᑦ ᐊᖏᓂᖏᑦ (229 ᑲᑎᑦᑐᒋᑦ ᐱᔭᐅᔪᑦ ᖃᐅᔨᓴᕐᕕᒻᒧᑦ ᐊᕐᕌᒍᒧᑦ), ᐊᔾᔨᐸᓗᖓ 2010−ᖏᓐᓂ (470 ᑲᑎᑦᑐᒋᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᑐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ ᐊᕐᕌᒍᒥ), ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᑦ 1990−ᖏᓐᓂ ᐊᑦᑎᑦᑑᓚᐅᖅᑐᑦ. ᐆᒪᔪᓂᑦ MR ᖃᐅᔨᓴᕐᓂᖅᑐᐃᓐᓇᓚᐅᖅᑐᑦ ᐃᓚᖏᓐᓂ ᐊᕐᕌᒍᓂᑦ ᐊᑐᓂ ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ: 1993-1996 ᐊᒻᒪ 1997 ᓯ−ᔭᖓᓂ ᕿᑭᖅᑕᓗᒃ ᐊᒻᒪ ᐅᐊᖕᓇᖓᑕ ᐱᖓᓇᖓ ᐊᑯᑭᑦᑐᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ, ᐊᒥᓱᓚᐅᓚᐅᖅᑐᑦ ᑐᖁᖓᓚᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ MR ᖃᐅᔨᓴᓂᕐᒥᑦ (1998-2010), ᐱᓗᐊᖅᑐᒥ ᐊᕐᕌᒍᐃᑦ ᑭᖑᓂᐊᒍᑦ. ᒥᑭᔪᓂᑦ ᖃᐅᔨᓴᒐᔅᓴᓂᑦ ᐊᔪᕐᓇᑎᑦᑎᓚᐅᖅᑐᖅ ᒥᔅᓴᐅᓴᑦᑐᒥᑦ ᓄᓇᒥ ᐊᑐᖅᑕᕐᓂ ᐊᒻᒪ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᑭᓪᓕᖃᖅᑎᑦᑎᔪᖅ ᑕᒪᒃᑭᓐᓄ ᓴᖏᓂᖓᓄᑦ ᐱᔭᐅᔭᓐᓇᖅᑐᓂᑦ 1990−ᖏᓐᓂ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒻᒪ ᓇᑐᓇᐃᑭᓐᓇᓂᑦᑎᓐᓄᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐊᑦᑎᓪᓕᓕᐊᖅᑐᒍ ᐃᓚᐃᓇᕐᓂ ᒥᔅᓴᐅᓴᓐᓂᖅ.

ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᓄᓇᐃᑦ ᐱᔭᐅᔪᑦ ᐊᒻᒪ ᖃᐅᔨᓴᒐᔅᓴᐃᑦ MR ᖃᐅᔨᓴᕐᓂᐅᔪᓄᑦ 1990−ᖏᓐᓂ ᐊᒻᒪ 2010−ᖏᓐᓂ ᐊᔾᔨᒌᖏᓐᓂᖃᓪᓗᐊᑲᓪᓚᓚᐅᖅᑐᑦ. 2010−ᖏᓐᓂᑦ, ᐊᒥᓲᓂᖅᓴᐃᑦ ᓄᓇᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓄᓇᖅᐸᒻᒥ ᓯᔾᔭᖏᓐᓂ ᐊᒻᒪ ᑲᖏᖅᑐᓂᑦ ᕿᑭᖅᑖᓗᒥ. ᑖᓐᓇ ᐊᔾᔨᒌᖏᓐᓂᖓ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐱᔭᐅᔪᓂᑦ ᐊᑯᓐᓂᖏᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᐊᓯᔾᔨᕐᓂᖏᓐᓃᖏᑦᑐᖅ ᓇᖁᑦᑕᖅᑕᑦᑕᖏᓐᓂᑦ. ᖃᐅᔨᓴᐃᓂᖅ ᖁᓪᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᐃᓐᓇᓂ ᐊᕐᓇᓂᑦ, ᖃᐅᔨᓴᐃᒍᓐᓇᑎᑦᑎᔪᖅ ᓄᓇᒥ ᐊᑐᖅᑕᐅᕙᑦᑐᓂᑦ ᐊᕐᕌᒍᒪᓚᓐᓂ ᐃᓛᒃᑯᖅᑐᖅ ᓴᓂᐊᓂ ᐱᔭᐅᔪᓂᑦ ᓇᒦᓐᓂᖏᓐᓂᑦ, ᑕᑯᔅᓴᐅᑎᑦᑎᓚᐅᖏᑦᑐᑦ ᐅᖓᓯᓐᓂᖏᓐᓂᑦ ᓄᓇᒥ ᐊᒻᒪ ᖁᕝᕙᓯᓐᓂᖓᓂᑦ ᓇᓄᕐᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990s ᐊᒻᒪ 2010s. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐱᔭᐅᔪᓂᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᐅᓚᓂᖃᓚᐅᖅᑐᑦ ᐊᔾᔨᒋᖏᑕᖓᓂ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ, ᓄᓇᖅᐸᓯᓪᒦᖏᓐᓂᖅᓴᐅᓪᓗᑎ ᐊᒻᒪ ᑲᖏᖅᑐᓂ 1990−ᖏᓐᓂ.

ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᒐᔅᓴᐃᑦ ᓄᓇᕗᒻᒥ 1990−ᖏᓐᓂ ᑐᖖᓂᓚᐅᖅᑐᑦ ½ ᖃᐅᔨᓴᕐᕕᐅᔪᓂᑦ 2010−ᖏᓐᓂ. 2010−ᖏᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᑲᖏᖅᑑᑕᐅᓂᖅᓴᐃᑦ ᓯᔾᔭᖓᓂ ᐊᒻᒪ ᓄᓇᖅᐸᒥᐅᓂᖅᓴᓂ ᓇᔪᖅᑕᐅᔪᓂᑦ, ᖃᐅᔨᓴᖅᑐᒋᑦ ᐃᓚᖏᓈᕈᓐᓇᖅᑐᑦ ᐊᑦᑐᐊᓂᓰᑦ ᒥᑭᓂᖅᓴᐃᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ 1990−ᖏᓐᓂ, MR ᖃᐅᔨᓴᓂᖅ ᐊᒻᒪ ᒥᓯᓴᖅᑕᐅᔪᑦ ᐊᔾᔨᒌᖑᐊᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᒪᕐᕉᓐᓄ ᑐᑭᓯᒋᐊᕈᑎᓄᑦ (1) ᐊᑕᐅᓯᖅᑐᒥᑦ 2010−ᖏᓐᓂ MR ᑐᑭᓯᒋᐊᕈᑏᑦ, ᐊᒻᒪ (2)ᓄᓇᖏᑎᒍᓪᓗ 2010−ᖏᓐᓂ MR ᑐᑭᓯᒋᐊᕈᑏᑦ ᐊᔾᔨᒌᖑᐊᖅᑎᒐᔅᓴᐅᔪᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ 1990−ᖏᓐᓂ (ᓵᑉᑕ 5).

ᐊᑦᑐᐊᓂᓪᓕᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐊᕐᕌᒍᓚᐅᔪᓂᑦ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᒥ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ. 1990−ᖏᓐᓂ ᐊᒥᓲᑲᐃᓇᐸᓪᓚᐃᓚᐅᖅᑐᑦ ᐅᑎᖅᑕᖅᑕᖅᑐᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ ᕿᑭᖅᑖᓗᒥ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓄᓇᐃᑦ ᓯᑯᒥ ᐊᑐᓚᐅᖅᑐᑦ ᐃᒪᐃᓐᓇᒦᑦᑐᒥ ᐅᕝᕙᓴᓐᓃᑦ ᓯᑯᓂᑦ ᖃᐅᔨᓴᖅᑕᓚᓚᓐᓂ

ᐊᐅᔾᒃᑯᑦ. ᑕᒪᓐᓇ ᑕᑯᔅᓴᐅᑎᑕᐅᓚᐅᖅᑐᖅ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑕᒪᑭᒃ ᖃᐅᑕᒫᑦ ᐃᑳᕝᕕᐅᔪᑦ ᓯᑯᕋᐃᑦ ᑐᑭᓯᑎᐊᕈᑎᖏᑦ. ᓯᑯᖓ ᐊᖏᔪᒥ ᒥᑭᓪᓕᑎᐊᓚᐅᖅᑐᖅ 2010−ᖏᓐᓂᑦ, ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᓯᑯᖃᕈᓐᓃᖅᑐᓂ ᐋᓯᐅᓯᒥ ᐊᒻᒪ ᐊᖅᑯᑎᐅᕙᑦᑐᑦ ᐊᑦᑎᓂᖅᓴᕐᓗᐊᑲᓪᓚᐅᓚᐅᖅᑐᖅ. ᐊᑐᓂ ᐊᕐᕌᒍᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2013, ᐊᒥᓱᓂᖏᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ ᐊᑕᐅᓯᒡᒍᑏᑦ ᐃᓐᓇᖏ ᐊᒃᓇᐃᑎ ᖃᐅᔨᓴᕐᕕᒦᑦᑐᑦ ᐱᖃᓯᐅᔾᔭᐅᓚᐅᖅᑐᑦ. 1990−ᖏᓐᓂ, ᖃᓂᒋᔭᐸᓗᖏᓐᓂ <30% ᖁᖓᓯᕈᖅᓯᒪᔪᑦ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑰᖓᔪᓂᑦ ᐊᕐᓇᓂᑦ ᓇᓄᕐᓂᑦ ᐃᓗᐊᓃᑦᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ MR−ᒥ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ. ᑖᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅᐊᒥᓲᓂᖅᓴᕆᔭᖏᑦ ᓇᓄᐃᑦ ᐊᑐᐃᓐᓇᐅᓚᐅᖏᑦᑐᑦ ᐱᔭᐅᓗᑎ ᐃᓚᖏᓐᓂᑦ ᐊᕐᕌᒍᖏᓐᓂᑦ 1990−ᖏᓐᓂᑦ. ᑕᑯᓐᓇᖅᑐᒍ 2010−ᖏᓐᓂᑦ, ᐊᑯᓐᓂᖏᓐᓂ 70-80% ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᑦ ᕐᓇᖏᑦ ᐊᕐᓇᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᕕᒻᒥᓚᐅᖅᑐᑦ.

ᕿᑎᕋᖅᑐᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᑖᒃᑲᑐᐃᓐᓇᖅ ᐅᑎᖅᑕᕝᕕᐅᑲᐃᓐᓇᖅᑐᒥᑦ ᖃᐅᔨᓴᕝᕕᐅᔪᒥᑦ ᐃᓚᐃᓐᓇᖏᓐᓃᖅᓯᒪᒋᐊᖃᖏᑦᑐᑦ ᓄᓇᖓᓄᑦ ᑭᓪᓕᖏᓐᓄᑦ. ᑭᓯᐊᓂᓕ, ᐊᖏᓂᖓ ᐊᖅᑯᑎᐅᑲᐃᓐᓇᖅᑐᖅ 1990−ᖏᓐᓂ ᐊᔾᔨᒌᖏᑦᑑᔭᖅᑐᑦ ᐊᒻᒪ ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᐊᕙᑎᖓᓄ ᖃᓄᐃᓐᓂᖓᓄᑦ; ᐊᒻᒪ ᒥᑭᔫᑎᑦ ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐊᔪᕐᓇᑎᑦᑎᓚᐅᖅᑐᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᓗᑎ. ᐊᖅᑯᑎᒋᔭᐅᒃᑲᓐᓂᑲᐃᓐᓇᖅᑐᑦ 1990−ᖏᓐᓂ ᑖᒃᑲᑐᐃᓐᓈᖅᓯᒪᓚᐅᖏᑦᑐᑦ ᐊᒻᒪ ᐊᑦᑐᐊᓂᖃᖅᑐᑎᑦ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᓇᔾᔮᓚᐅᖅᑐᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ ᓄᓇᖅᐸᒌᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᕿᑭᖅᑖᓗᒻᒥ (ᓲᕐᓗ, ᓇᔪᖅᓴᓯᐅᖅᑐᑎᑦ ᑎᓯᒋᓂᐊᖅᑕᖏᓐᓂᑦ), ᓴᓂᐊᓂ ᓇᔾᔨᖏᑦᑐᓂᑦ, ᖃᐅᔨᓴᖅᑕᐅᑎᐊᖃᑦᑕᖏᓐᓂᖏᓐᓄᑦ ᐊᕐᓇᐃᑦ ᐃᓐᓇᐃᑦ ᓄᓇᖅᐸᒌᖃᑦᑕᖅᑐᑦ ᐃᓚᖏᓐᓂᑦ ᐊᕐᕌᒍᖏᓐᓂᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᖃᑦᑕᖏᓐᓂᖏᓐᓄᑦ ᓄᓇᒦᑦᑐᓂᑦ.

ᒪᓕᓐᓂᖃᖅᑐᖅ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐅᑎᖅᑕᕝᕕᐅᑲᐃᓐᓇᖅᑐᓂᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010-ᖏᓐᓂᑦ, ᐊᖏᔪᒥᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖅᑐᑦ ᐃᓗᓕᖏᓐᓂᑦ MR ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ (ᓲᕐᓗ, ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᐊᕐᓇᐅᓂᖏᓐᓄᑦ ᐊᖑᑕᐅᓂᖏᓐᓄᑦ) ᑖᒃᑯᓂᖓ ᒪᕐᕉᔪᓐᓂᒃ. ᐱᓗᐊᖅᑐᒥ, ᐃᓐᓇᖏᑦ ᐊᕐᓇᐃᑦ ᐱᑕᖃᖏᓐᓂᖅᓴᐅᔫᖅᑐᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ.

ᐊᔅᓱᕈᕈᑎᑕᖃᓚᐅᕐᒥᔪᖅ 1990−ᖏᓐᓂ MR ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓇᓗᓇᕈᑎᑕᖃᓚᐅᖅᑐᖅ ᓇᓗᓇᐃᖅᑕᐅᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᓇᒦᓐᓂᖏᓐᓄᑦ ᖁᖓᓯᒍᖏᑎᒍᑦ ᓴᓂᐊᓂ ᖃᐅᔨᔭᐅᔪᓂᑦ ᕿᓂᕐᓂᑐᐃᓐᓇᒃᑯᑦ (ᓲᕐᓗ, ᑕᑯᔭᐅᑳᓪᓚᑦᑐᑦ) ᐅᐱᓐᓇᕋᓂ ᐱᔭᐅᓯᒪᔪᓂᑦ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᑖᔅᓯᒥᖓ ᓇᓗᓇᐃᖅᓯᓂᖃᓚᐅᖏᑦᑐᖅ. ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᐃᑲᔪᖅᑕᐅᓪᓗᑎ ᐱᔭᐅᔪᑦ ᑕᐃᒪᐃᓐᓂᖓᓄ ᐊᔾᔨᒋᓐᖑᐊᖅᑐᒍ ᐊᑐᐃᓐᓇᐅᔪᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐱᓗᒋᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᓄᐊᑕᐅᓯᒪᔪᑦ 1990−ᖏᓐᓂ ᖃᐅᔨᓴᕆᐊᖓᐅᑕᐅᔪᓂᑦs BB−ᒧ ᖃᐅᔨᓴᐃᓂᖅ ᑕᐃᒪ ᐃᓚᖏᓪᓗ. (2005). ᑖᓐᓇ ᓇᓗᓇᕈᑎᐅᔪᖅ ᐃᓚᐃᓐᓇᖏᓐᓃᖅᑎᑦᑎᔾᔪᓐᓇᖅᑐᖅ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᖃᐅᔨᒪᓪᓗᒋᑦ ᓇᓪᓕᐊ ᓇᓄᐃᑦ ᓇᒦᓚᐅᕐᒪᖔᑕ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᐱᒻᒪᕆᐅᓚᐅᖅᑐᖅ ᐊᒥᓲᓂᖅᓴᓄᑦ ᐃᑲᔪᓱᖅᑕᖃᐅᔪᑦ MR ᐊᑐᖅᑕᐅᔪᓂᑦ 1990−ᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ.

ᐊᑕᖏᖅᑐᒋᑦ, ᐊᔾᔨᐸᓗᖏᑦ 2010−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᒦᓐᓂᑦ, 1990−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒦᓐᓂᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᒥᑭᔫᑕᐅᓂᖅᓴᓂᑦ ᖃᐅᔨᓴᕋᔅᓴᓄᑦ, ᐱᔭᓐᖏᖅᓯᒪᖏᑦᑐᑦ ᓄᓇᖓᑎᒍᑦ ᖃᐅᔨᓴᕐᓂᖅ, ᐊᒥᓱᕈᖅᑎᑦᑎᑲᐃᓐᓇᖅᑐᖅ ᐊᖅᑯᑎᒋᔭᐅᑲᐃᓐᓇᖅᑐᓂᑦ ᓇᓄᕐᓄᑦ ᓯᑯᒦᖏᓐᓇᓚᐅᖅᑐᓄᑦ ᐊᐅᔾᒃᑯᑦ, ᐊᒻᒪ ᑖᒃᑲᑐᐃᓐᓇᖅ ᐃᖄᖅᑐᑦ ᐃᓐᓇᕐᓄᑦ ᐊᕐᓇᓄᑦ ᓄᓇᒨᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᑎᓯᖏᓐᓄᑦ. ᑖᒃᑯᐊ ᐃᓚᐃᓐᓇᖏᓐᓂᑎᑦᑎᑐᐃᓐᓇᕆᐊᖃᕐᓇᖅᑐᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᐆᒪᓂᕐᒧᑦ ᐊᒻᒪ ᐊᒥᓲᓂᖏᓐᓄᑦ 1990−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᓄᓇᖓᑎᒍᑦ ᑭᓪᓕᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010-ᖏᓐᓂᑦ BB ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᑕᕝᕙᖓᑦᑕᐅᑎᒋ ᐊᔾᔨᒋᓐᖑᐊᖅᑎᒐᔅᓴᐅᖏᑦᑐᑦ, ᑭᓪᓕᖅᑎᑦᑎᔪᖅ ᖃᐅᔨᓴᐃᔪᓐᓇᓂᕐᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖃᑦᑕᖅᑐᓂᑦ.

ᓵᐸᑕ 4: ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᓇᔪᖅᑕᐅᔪᑦ ᐊᑐᖅᑕᐅᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᓵᐸᑕ 4 ᖃᐅᔨᓴᖅᑐᖅ ᐊᓯᔾᔨᖅᑐᓂᑦ BB–ᒥ ᓯᑯᒥ ᓇᔪᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂᑦ BB–ᒥᑦ ᓇᓄᕐᓄᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2010-ᖏᓐᓂᑦ (2009-2015). ᖃᐅᔨᓴᕈᑎᒦᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᑕ–ᕗᖓᑦᑕᐅᑎᒋ ᑐᔾᓯᕚᖑᔪᓂᑦ JC–ᒧᑦ (ᖃᐅᔨᓴᕐᓂᖅ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᕐᓂᖏᓐᓄᑦ ᓯᑯᖏᓐᓂᑦ) ᐊᒻᒪᓗᑦᑕᐅᖅ ᐱᒻᒪᕆᐅᔪᖅ MR–ᒧ ᖃᐅᔨᓴᓂᕐᒧᑦ, ᓲᕐᓗ ᖃᓄᖅ ᐊᕙᑎ ᐊᓯᔾᔨᕐᓂᖓ ᓇᓗᓇᐃᖅᓯᔪᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ. ᖃᐅᔨᓴᕈᑎᒦᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᖃᐅᔨᓴᖅᑎᓃᙳᖅᑐᓂᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᔾᔩᓐᓂᖑᐊᖅᑎᑕᐅᓗᑎ ᑕᑯᔭᐅᔪᓂᑦ ᓄᓇᓕᒻᒥ ᖃᐅᔨᒪᔭᐅᔪᓂᑦ (LEK) ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᖃᐅᔨᔭᐅᔪᑦ ᐊᒥᓱᕈᓇᐊᕐᓂᖏᑦ ᓇᓄᐃᑦ ᓄᓇᒥ ᓯᕗᔭᖅᐸᓯᐅᓂᖅᓴᒥᓪᓘᓐᓃᑦ ᐊᒻᒪ ᓄᓇᓕᓐᓂᑦ.

ᓯᑯᖓᓂ ᓇᔪᖅᑕᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ BB–ᒧᑦ ᐱᒋᐊᖅᓯᒪᓪᓗᓂ 1979–ᒥᑦ, ᐱᒋᐊᕐᓂᖃᓕᖅᑎᓪᓗᒍ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑎᑎᕋᖅᑕᐅᓂᖏᑦ, ᐊᒻᒪ ᑕᔪᓯᓪᓗᓂ 2015–ᒧ ᐊᑐᖅᑐᑎ ᖃᐅᔨᓴᐅᑎᓂᑦ (ᖃᐅᑕᒫᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᖅ ᓯᑯᖓᓂ). ᓯᑯᖓᓂ ᓇᔪᖅᑕᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ BB–ᒥᑦ ᐊᖏᔪᒥᑦ ᒥᑭᓪᓕᒋᐊᖅᓯᒪᔪᖅ ᑕᕝᕙᓂ ᐊᑯᓂᐅᓂᖓᓂ, ᐱᓗᐊᖅᑐᒥ ᐱᒋᐊᖅᑐᓂ ᕿᑎᖏᓐᓂ 1990–ᖏᓐᓂᑦ. ᐊᑯᓂᐅᓂᖓ ᐊᐅᔭᖓ (ᓲᕐᓗ, ᐅᓪᓗᐃᑦ ᐊᒥᓲᓂᖏᑦ ᓯᑯᐃᖃᑦᑕᕐᓂᖓ ᐅᐱᕐᖓᓵᒃᑯᑦ ᐅᑭᐊᒃᓵᒧᑦ) ᐊᒥᓱᕈᓇᐊᖅᓯᒪᔪᑦ ᐅᓪᓗᐃᑦ/ᐊᕐᖒᒍᐳᐃᑦ ᖁᓖᑦ 1979–ᒥᓂᑦ. ᓯᕐᖁᖃᕐᓂᓇᓚᔪᓚᐅᖅᑕᖓ ᔫᓐᓂ–ᐅᑦᑑᐱᓇᒧ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᔪᖅ 4% ᖁᓕᓂᑕᒫᑦ ᐊᕐᖒᒍᓂᑦ. ᐊᐅᓐᓂᓇᓚᔪᖃᑦᑕᖅᑕᖓ ᐊᓯᔾᔨᖅᓯᒪᖏᑦᑐᖅ ᑭᓯᐊᓂ ᕿᓚᒥᐅᓂᖅᓴᐅᕙᓕᖅᑐᖅ 3-4 ᐱᓇᓱᐊᕈᓯᕐᓂᑦ 2000–ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990–ᖏᓐᓂᑦ. BB ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᖅᓯᒪᔪᖅ ᓯᑯᖓᓂ ᐊᑐᖅᑕᐅᕙᑦᑐᑦ ᓇᓄᕐᓄᑦ ᐊᒻᒪ ᑕᒪᓐᓇ ᓇᔪᖃᑦᑕᖅᑕᖏᓐᓂᑦ ᓴᓚᐃᑎᑦᑎᔪᖅ.

ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᐊᒥᓱᒐᓚᐃᑦ ᐊᓯᔾᔨᖅᑐᑦ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᓇᔪᖃᑦᑕᖅᑕᖏᑦ BB ᓇᓄᖏᑦ. ᓇᓄᖓᖃᑦᑕᕐᓂᖏᑦ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᓇᓄᐃᑦ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᓪᓗᐊᑲᓪᓚᑦᑐᑦ ᐃᒪᐃᓐᓇᐅᑎᓪᓗᒍ (ᐋᒡᒌᓯ–ᐅᑦᑑᐱᓇ) 2000–ᖏᓐᓂᑦ ᐃᐃᓐᓇᕐᒥ ᕿᑭᖅᑕᖏᓐᓂᓪᓗ ᓯᑯᖃᕈᓐᓂᕐᓂᖓᓄᑦ. ᓇᓄᐃᑦ ᐃᒫᓅᕋᔪᖏᑦᑐᑦ ᓄᓇᒦᓐᓇᓛᖅᑐᑎᑦ 2000–ᖏᓐᓂᑦ ᐃᒪᖓᓂ ᓯᑯᖃᑦᑎᐊᖏᓐᓂᖓᓄᑦ ᐃᒪᐃᓐᓇᐅᖃᑦᑕᑎᓪᓗᒍ.

BB ᓇᓄᐃᑦ ᓯᑯᓂᑦ ᐊᑐᖏᓂᖅᓴᐅᕙᓚᐅᖅᑐᑦ ᓯᑯᖃᖏᓂᖅᓴᐅᓂᖓᓄᑦ ᐅᑭᐅᒃᑯᑦ ᐊᒻᒪ ᐅᐱᕐᖔᒃᑯᑦ 2000–ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990–ᖏᓐᓂᑦ. ᓇᓄᐃᑦ ᖃᓂᓴᕈᒪᓂᖅᓴᐅᓚᐅᖅᑐᑦ 300 m ᐃᑎᓂᓕᒻᒥᑦ (ᓯᑯᒥ ᐃᒪᖓᓂ ᐊᒻᒪ ᓄᓇᖅᐸᓯᒻᒥ) 2000–ᖏᓐᓂᑦ. ᓯᑯᖃᕐᓂᑐᐊᓇᖏᑕᖓ ᓇᓗᓇᐃᖅᓯᔫᓚᐅᖏᑦᑐᑦ ᓇᔪᖅᑕᐅᔪᓂᑦ; ᐊᕐᓇᕐᓗᐃᑦ ᓂᕈᐊᖅᑕᐅᔪᑦ ᓯᑯᖃᖏᓂᖅᓴᓄᑦ ᓯᑯᒨᕈᓐᓇᑎᑦᑎᓚᐅᖅᑲᑕ ᐃᒪᖏᓐᓄᑦ (<300 m). ᐊᕐᓇᕐᓗᐃᑦ ᓇᓄᐃᑦ ᓄᓇᒧᑦ ᖃᓂᒃᓴᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᑕᖅᑭᓕᒫᓂᑦ 2000–ᖏᓐᓂᑦ ᑭᓯᐊᓂ ᓯᑯᐃᖅᑎᓪᓗᐳ (ᔩᓂ–ᔪᓚᐃ), ᐃᒪᐃᓐᓇᒥ ᓯᑯᒦᓚᐅᖅᑐᑦ ᐊᑯᓂᐅᒍᓐᓇᕐᓂᓕᒫᖏᓐᓂᑦ, ᓂᕿᒃᓴᓯᐅᖅᑐᑦ.

ᐊᑯᓂᐅᔪᒥᑦ ᐳᐃᒥᑐᐃᓐᓇᓇᐊᖃᕐᓂᖏᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ, ᓇᓗᓇᐃᖅᑕᐅᔪᑦ ᐊᐅᔮᒃᑯᑦ ᖃᓄᐃᓘᖅᑕᐅᔪᑦ ᓇᓄᐃᑦ ᐃᖏᕋᓂᖃᖅᑐᑎᑦ >100 km ᐃᒪᐃᓐᓇᕐᒥᑦ ᓯᑯᒥᑦ, ᕿᑭᖅᑖᓗᒻᒧᑦ, <10%ᔾᒥ ᓯᑯᖃᕐᓂᖓᒍᑦ, ᐊᑕᐅᑦᑎᑯᓪᓗ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᔪᑦ ᐱᑕᖃᖏᑐᓪᓘᓐᓃᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ (ᓲᕐᓗ, ᖁᖓᓯᕈᐃᑦ ᖃᐅᔨᔭᔅᓴᐅᖏᒻᒪᑕ ᐃᒫᓃᑦᑎᓪᓗᒋᑦ). ᑖᒃᑯᐊ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᑕᒪᒃᑭᓐᓂ ᖁᓂᒡᒍᑎᓂᑦ ᐊᕐᖒᒍᓂᑦ, ᑭᓯᐊᓂ ᖁᕝᕙᓇᐊᓚᐅᖅᑐᑦ 2000–ᖏᓐᓂᑦ, ᐱᓗᐊᖅᑐᒥ 2011–ᒥᑦ.

ᐊᓯᔾᔨᖅᑐᑦ ᓄᓇᖓᓄᑦ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ ᕿᑭᖅᑖᓗᒻᒥ ᖃᐅᔨᔭᐅᖅᑐᑦ ᐊᒻᒪ ᓯᑯᐃᔅᓴᐃᓐᓂᖓᓄ ᐊᒻᒪ ᓯᒥᒍᓇᓵᕐᓂᖓᓄᑦ. 2000–ᖏᓐᓂᑦ, ᓇᓄᐃᑦ ᐊᑯᓂᐅᓂᖅᓴᖅ ᓄᓇᒦᓚᐅᖅᑐᑦ

ᕿᑭᖅᑖᓗᒻᒥ, ᐊᒻᒪ ᑎᑭᕝᕕᖏᑦ ᐅᓪᓗᖏᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐊᐅᔭᒃᑯᑦ ᑕᖅᑭᓪᓗᐊᒥ ᑎᑭᓴᕋᐃᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ. ᓇᓄᐃᑦ ᓄᓇᒦᓐᓂᕆᔭᖏᑦ ᖁᕝᕙᕆᐊᓚᐅᖅᑐᖅ 20-30 ᐅᓪᓗᓂᑦ 1990−ᖏᓐᓂᐅᕙᔪᒻᒪᓂᑦ. ᓇᓅᕙᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ ᑕᓪᓗᕈᒻᒨᖅᐸᒍᓐᓃᖅᑐᑦ ᐊᐅᓯᐃᑦᑐᒧᓪᓗᓐᓃᑦ ᕿᑭᖅᑖᓗᐊᓂ ᑭᓯᐊᓂ ᕿᑭᖅᑖᓗᒻᒥᑐᐊᖅ (ᐊᒥᓲᖏᑦᑐᑦ Melville Bay ᐊᑯᑭᑦᑐᑦ, ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᓇᔪᖅᓯᔪᑦ ᖃᐅᔨᓴᐅᓚᐅᖅᑐᑦ ᓯᕗᓪᓕᖅᐹᒥ ᐊᑐᖅᑐᑎᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔫᒃᑯᑦ). ᖃᓂᒌᑦᑐᑎᑦ ᖃᓄᐃᓘᖃᑦᑕᐅᑎᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ ᐅᓪᓗᖏᓐᓂᑦ ᑎᑭᑦᑎᓪᓗᒋᑦ ᕿᑭᖅᑖᓗᒻᒧ ᐊᒻᒪ ᓯᑯᐃᖅᑎᓪᓗᒍ, ᐊᒻᒪᓗᑦᑕᐅᖅ ᐅᓪᓗᖏᓐᓂᑦ ᕿᑭᖅᑖᓗᒻᒦᒍᓐᓃᖅᑐᑎᑦ ᓯᑯᒨᖅᑐᑎ ᐊᒻᒪ ᐅᑭᐊᖅᓴᑯᑦ ᓯᑯᓂᖓ.

16 ᑎᓯᐅᔪᑦ ᖃᐅᔨᓴᐅᓚᐅᖅᑐᑦ ᐊᑐᖅᑐᑎ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ 2009 ᐊᒻᒪ 2015 (15 ᕿᑭᖅᑖᓗᒻᒥ ᐊᒻᒪ 1 Melville Bay, ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ). ᐊᓯᔾᔨᖅᑐᓂᑦ ᖃᐅᔨᓴᐅᓚᐅᖅᑐᑦ ᑎᓯᒥᖃᕐᓂᖏᑦ BB−ᒥᑦ. ᑎᓯᒥᑦ ᖃᐅᔨᓚᐅᖅᑐᑦ Melville Bay−ᒥ, ᑕᐃᒪᐃᓚᐅᖏᑦᑐᖅ 1990−ᖏᓐᓂᑦ, ᐃᑲᔪᖅᑐᐃᔪᖅ ᐊᓯᖏᓐᓂᑦ ᖃᐅᔨᓴᐅᓯᒪᔪᓂᑦ ᐊᑐᖅᑕᐅᓂᖅᓴᐅᓂᖓᓄᑦ Melville Bay ᐊᕐᕌᒎᓕᓚᐅᓯᒪᓕᖅᑐᓂᑦ. ᐃᓚᒋᐊᕐᓗᒍ, ᐅᓪᓗᖏᑦ ᑎᓯᒨᖃᑦᑕᕐᓂᖏᑦ >1 ᑕᖅᑭᒥᑦ ᑭᖑᕙᓯᓐᓂᖃᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ. ᐊᓂᕕᒋᔭᖏᑦ ᐅᓪᓗᖏᑦ ᑎᓯᖏᓐᓂᑦ ᐊᓯᔾᔨᓚᐅᖏᑦᑐᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᕿᓚᒥᐅᓂᖅᓴᐅᖃᑦᑕᖅᓯᒪᔪᑦ BB−ᒥ 2000−ᖏᓐᓂ; ᓯᕗᓪᓕᖅᐹᖓ ᐅᓪᓗᖓᓂ ᓇᔾᔨᔪᒥ ᓇᓄᕐᒥᑦ ᕿᓚᒥᐅᓂᖅᓴᐅᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᓇᓄᐃᑦ ᐊᑯᓂᐅᓂᖅᓴᖅ ᓄᓇᒦᓚᐅᖅᑐᑦ ᑎᓯᖏᓐᓅᓚᐅᖅᑎᓐᓇᒋᑦ. ᑎᓯᕆᔭᐅᔪᑦ 2000−ᖏᓐᓂᑦ ᖁᕝᕙᓯᓐᓂᖅᓵᓚᐅᖅᑐᑦ ᐊᒻᒪ ᓯᕕᖓᓂᖅᓴᓂᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ, ᐊᐳᑎᖅᑦᑎᐊᖏᓐᓂᖓᓅᔪᔅᓴᐅᔪᖅ.

ᓄᖃᐅᓯᕐᓗᒍ, ᐊᖏᔪᓂᑦ ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖅᑐᑦ BB−ᒥᑦ ᐊᕐᕌᒎᓕᖅᑐᓂᑦ 25−ᓂ ᐊᑦᑐᐊᓂᖃᖅᑐᑦ ᓯᑯᖃᕈᓐᓂᕐᓂᖓᓄᑦ.

ᓵᑉᑕ 5: ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᓂᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓄᖏᓐᓂᑦ ᓵᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᓵᑉᑕ 5 ᓇᓗᓇᐃᔭᐃᔪᖅ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ ᐱᖓᓱᓄᑦ ᐊᕐᕌᒍᓄᑦ MR ᖃᐅᔨᓴᓂᕐᒥᑦ BB ᓇᓄᖏᓐᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ 2011-2013. ᑲᑎᖓᔪᓂᑦ ᐆᒪᔪᓂᑦ −ᐱᔭᐅᔪᑦ ᐊᒻᒪ ᑐᖁᖓᔪᓂᑦ NR ᐊᑐᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᑐᕆᑦ ᖃᐅᓴᕈᑎᒥᓃᑦ ᑐᕌᖓᖅᑐᑎᑦ ᐅᓪᓗᒥ ᒨᖓᓕᖅᑎᑕᐅᓗᑎ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐆᒪᓂᖏᑦ ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ. ᖃᐅᔨᓴᕈᑎᒥᓂᖅᓵᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ 914 ᐱᔭᐅᔪᑦ ᐊᕐᕌᒍᖏᓐᓂᑦ 1993-1995 ᐊᒻᒪᓗ 1997 ᐊᒻᒪ 1,410 ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ ᐅᖓᓯᑦᑐᒥᑦ ᕿᑭᐅᑦᑕᐅᔭᓄᑦ 2011- 2013. ᓇᓐᓄᑦᑕᐅᔪᓂᑦ ᐱᔭᐅᔪᑦ (n=243) ᑲᓇᑕᒥ ᐊᒻᒪ ᐊᑯᑭᑦᑐᑦ ᐃᓚᓕᐅᑎᓯᒪᔪᑦ 21−ᓂ ᐊᕐᕌᒍᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ r (1993-2013) ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ, ᑭᓯᐊᓂ ᐱᔭᐅᒃᑲᖃᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᖃᓚᐅᖏᑦᑐᖅ 13−ᓂ ᐊᕐᕌᒍᓯᒪᔪᓂᑦ 1998 – 2010.

ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒥᑦ ᐊᒥᓲᓂᖏᑦ BB ᓇᓄᖏᑦ 2012-2013−ᒥ ᐅᑯᐊᖑᓚᐅᖅᑐᑦ 2,826 (95% CI = 2,059-3,593)ᓇᓄᐃᑦ. ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒥᑦ ᐊᒥᓲᓂᖏᑦ 1994-1997 ᐅᑯᐊᖑᓚᐅᖅᑐᑦ 2,173 (95% CI = 1,252-3,093) ᓇᓄᐃᑦ, ᐊᔾᔨᐸᓗᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᐅᓂᒃᑲᓂᑦ ᑕᐃᓗ ᐃᓚᖏᓪᓗ. (2005). ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᐊᔾᔨᒌᖑᐊᖅᑎᒐᔅᓴᐅᖏᑦᑐᑦ ᐊᓯᔾᔨᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᓴᓇᓯᒪᓂᖓ ᐊᒻᒪ ᐊᕙᑎᖓᓄᑦ ᖃᓄᐃᓐᓂᖓᓄᑦ.

ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᒃ ᑲᑎᑦᑐᒥᑦ (ᓲᕐᓗ, ᐱᖃᓯᐅᑎᓪᓗᑎ (ᓇᓐᓄᑦᑕᐅᕙᑦᑐᑦ) ᐆᒪᓂᖏᓐᓄᑦ 2011-2013 ᐅᑯᐊᖑᓚᐅᖅᑐᑦ 0.90 (SE = 0.05) ᐊᕐᓇᓄᑦ ᐊᕐᕌᒍᓖᑦ ≥ 2, ᐊᒻᒪ 0.78 (SE = 0.06) ᐊᖑᑎᓄᑦ ᐊᕐᕌᒍᓕᓐᓄᑦ 2 ≥. ᐊᑯᓂᐅᓂᖓ ᒥᔅᐅᓴᑦᑕᐅᔪᖅ ᑲᑎᑦᑐᒥᑦ ᐆᒪᓂᖏᑦ ᑲᑎᖓᔪᓂᑦ ᐊᕐᕌᒍᓕᓐᓂᑦ ᐊᑎᖃᑕᖏᓐᓄᑦ ᐊᕐᕌᒍᒥ ᐊᒻᒪ ᐊᑎᖅᑖᖏᑦ, ᖃᐅᔨᓴᕐᓂᓕᒫᖑᔪᓂᑦ 1993-2013, ᐃᒪᐃᓕᖓᓚᐅᖅᑐ 0.87 (SE = 0.06). ᒥᔅᐅᓴᑦᑕᐅᔪᑦ ᓇᓐᓄᑦᑕᐅᓯᒪᖏᑦᑐᑦ

ᐆᒪᖃᑦᑕᕐᓂᖏᑦ 2011-2013 ᐊᕐᓇᓄᑦ ᐊᒻᒪ ᐊᖑᑎᓄᑦ ᐊᕿᒍᓖᑦ ≥ 2 ᐅᑯᐊᖑᒪᐅᖅᑐᑦ 0.91 (SE = 0.05) ᐊᒻᒪ 0.83 (SE = 0.06), ᑐᖏᓕᕆᑦᑐᑎᑦ. ᒥᔅᐅᓴᑦᑕᐅᔪᑦ ᐆᒪᓂᖏᓐᓄᑦ ᑕᒪᒃᑭᓐᓄᑦ ᐊᖑᑎᓄᑦ ᐊᕐᓇᓄᑦ ᐱᖃᓯᐅᔾᔨᖃᕈᓐᓇᖅᑐᑦ ᐱᐅᖏᑦᑐᓂ ᐃᓚᒋᔭᐅᑎᑦᑎᖏᓐᓂᕐᒥᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᖅᑯᑕᐅᑲᐃᓐᓇᖅᑐᒥ (ᑕᑯᒃ ᓵᑕ 3).

ᐊᔾᔨᒌᓐᖑᐊᑎᑦᑎᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᓂᖏᓐᓂᑦ ᐊᒻᒪ ᐊᑮᑎᒧᑦ ᖃᓄᐃᓐᓂᕆᔭᖓ 1990−ᖏᓐᓂ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᐃᑲᔪᖅᑐᒋᑦ ᓇᑐᓇᐃᖅᑐᒋᑦ ᑭᓪᓕᒋᔭᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᑦ, ᓇᑐᓇᐃᑉᑐᒋᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᔪᓐᓇᖅᑐᑦ, ᐊᒻᒪ ᑐᑭᓯᕗᒋᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᐊᒥᓲᓂᖏᑦ. ᖃᐅᔨᓴᐃᓂᖅ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᓄᓇᖅᐸᓯᒻᒥ ᐱᔭᐅᔪᑦ, ᑕᒪᒃᑭ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᐊᑐᖅᑐᑎ ᕿᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ, ᓇᑐᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᓂᖅᓴᐅᔪᑦ ᐊᒻᒪ ᓄᓇᖓᓂ ᐊᖏᓂᖅᓴᒥ ᐆᒪᔪᓂᑦ ᐱᔭᐅᕗᓂᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ, ᐱᖃᓯᐅᑎᓪᓄᑐᑎ ᓄᓇᖅᐸᒻᒥ ᐊᒻᒪ ᐃᓱᖏᓐᓂᑦ ᑲᖏᖅᑐᓂᑦ, ᐊᕿᒍᖏᓐᓂᑦ 2011 – 2013 ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ, ᐃᓚᒋᐊᕐᓗᒍ, ᐃᒪᐃᓐᓴᕐᒥ ᓯᑯᑕᖃᓚᐅᖅᑐᖅ ᓇᓄᕐᓄᖅ ᐊᕿᒍᑕᒫᑦ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ 1990−ᖏᓐᓂᑦ, ᑭᓯᐊᓂ ᐱᑕᖃᑦᑎᐊᓚᐅᖏᑦᑐᖅ 2010−ᖏᓐᓂᑦ.

ᐋᖅᑭᔅᓯᓚᐅᖅᑐᒍᑦ ᓄᓇᖓᓄ ᐋᖅᑭᔅᓯᒪᔪᓂ 2010−ᖏᓐᓂ ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᖅᑐᑎ ᐊᑦᑐᐃᓂᖃᕐᓂᖏ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐊᔾᔨᒌᖏᓐᖏᓂᖏᑦ. ᖃᐅᔨᓴᕈᑎᒦᑦ ᓇᑐᓇᐃᖅᓯᔪᑦ ᓄᓇᖓᑎᒍᑦ ᑭᓪᓕᖃᖅᑎᑕᐅᔪᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᓰᕐᓗ ᑕᐃᒪᐃᓚᐅᖅᑐᖅ 1990−ᖏᓐᓂᑦ ᓴᖅᑭᑎᑦᑎᔪᓐᓇᖅᑐᖅ ᖃᓂᒋᔭᐸᓗᖏᓐᓂᑦ 10% ᐱᐅᖏᑐᓂ ᐱᖃᓯᐅᔾᔨᖏᑦᑐᓂᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ . ᐃᓚᒋᐊᕐᓗᒍ, ᕿᒻᒧᐊᑦᑎᑕᐅᓯᒪᔫᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᓇᑐᓇᐃᖅᓯᔪᑦ ᐊᖏᓂᓪᓗᐊᑲᓪᓚᒋᔪᓐᓇᖅᑕᖓ BB−ᒥ ᓇᓄᖏᑦ ᓯᓚᑖᓃᖑᖅᓯᒪᑐᐃᓐᓇᕆᐊᓖᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ ᐅᕙᓘᓐᓃᑦ ᓯᕐᑯᒥ MR−ᒥ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ 1990−ᖏᓐᓂᑦ, ᑭᓯᐊᓂ ᓯᑯᒥ ᖃᐅᔨᓴᕐᓂᖃᓚᐅᖏᑦᑐᑦ, ᑖᓐᓇ ᓇᑐᓇᐃᖅᓯᔪᖅ ᓇᑭᓐᖖᕕᒋᔪᓐᓇᖅᑕᖏ ᐱᖃᓯᐅᔾᔨᑎᑦᑎᖏᑦᑐᓂ 1990−ᖏᓐᓂᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ ᑭᓯᐊᓂ ᓇᑐᓇᐃᕈᓐᓇᓚᐅᖏᑦᑕᕗᑦ ᐊᖏᓂᖓ.

ᐊᔾᔨᒌᖏᓐᖏᓂᖏᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᔪᑦ ᐊᒻᒪ ᐊᑮᑎᖓᓄᑦ ᖃᓄᐃᓐᓂᖏᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᓴᖅᑭᑎᑦᑎᓚᐅᖅᑐᑦ ᐊᔾᔨᒌᖏᑦᑐᓂ ᕿᑭᕚᓯᓐᓂᖏᑦ ᐱᔭᐅᒃᑲᓂᑐᐃᓐᓇᕆᐊᖃᖅᑐᓂᑦ. ᑖᓐᓇ ᐊᔾᔨᒌᖏᑐᓂ ᐆᒪᔪᓂᑦ ᐱᔾᔪᑎᒋᓚᐅᖅᑕᖓ ᐅᑎᖅᑕᕐᕕᒋᑲᐃᓐᓇᖅᑕᖏᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ, ᓇᑐᓇᐃᖅᑕᐅᓚᐅᖅᑐᖅ 1990−ᖏᓐᓂᑦ (, ᓇᓄᐃᑦ ᐊᒥᓲᓂᖅᓴᐃᑦ ᓄᓇᒦᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ, ᖃᐅᔨᓴᕐᕕᐅᓂᖃᖏᑦᑐᒥ, ᐅᕙᓘᓐᓃᑦ ᓯᑯᖓᓂ). ᕿᑎᕋᖅᑐᑦ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᑦ ᑕᒪᐅᓇᑐᐃᓐᓇᖅ ᐊᑲᐅᖏᓕᐅᕈᑕᐅᖏᑦᑐᑦ MR ᐊᑐᖅᑕᒧᑦ, ᕿᕙᓯᑦᑐᑦ ᐅᕙᓘᓐᓃᑦ ᐊᔾᔨᒌᖏᖃᑦᑕᖅᑐᑦ ᕿᕙᓯᓐᓂᖏᑦ ᐃᑳᕐᕕᐅᑲᐃᓐᓇᖅᑐᓂᑦ ᑲᑎᖓᓪᓗᑎ ᐊᖅᑰᓴᖅᑕᐅᔪᓂ ᖃᐅᔨᓴᐃᓂᖅ, ᐅᕙᓘᓐᓃᑦ ᑕᒪᐅᓇᑐᐃᓐᓇᐅᖏᑦᑐᖅ ᐃᑳᕐᕕᐅᑲᐃᓐᓇᖅᑐᑦ, ᖃᐅᔨᒪᔭᐅᑦᑎᐊᖅᑐᑦ ᐱᖃᓯᐅᔾᔨᑎᑦᑎᖏᓐᓂᕐᒥᑦ. ᐊᑐᖅᑕᕗᑦ ᐱᖃᓯᐅᑎᓪᓗᑎ ᓇᓐᓄᑦᑕᒥᓂᕐᓂᑦ ᐊᔾᔨᖓᑎᑐᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᓐᖓ ᐆᒪᔪᓂᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᓯᐊᒻᒪᖅᓯᒋᐊᖅᑎᑦᑎᔭᖅ ᐱᖃᓯᐅᔾᔨᓂᖏᓂᕐᒥᑦ ᐃᓚᖏᓐᓄᑦ. ᑭᓯᐊᓂᓕ BB ᖃᐅᔨᓴᕈᑎᓃᑦ ᒥᑭᓗᐊᖅᑐᑦ MR−ᒨᕐᓗᑎ ᒥᔅᐅᓴᑦᑕᐅᓚᐅᖅᑐᑦ ᐊᑐᖅᑕᐅᑲᐃᓐᓇᑦᑐᒥ ᐊᒻᒪ ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᐊᑦᑐᐃᓂᖃᓗᐊᖏᑦᑐᖅ ᑭᓪᓕᖏᓐᓄᑦ.

ᐃᓱᒪᒋᓪᓗᒋᑦ ᑭᓪᓕᓯᓂᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᑐᓇᕈᑎᐅᔪᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ ᑭᓪᓕᖏᓐᓂᑦ ᐊᒻᒪ ᖃᐅᔨᔭᐅᓯᒪᔪᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᓂᖓ ᐊᒻᒪ ᐊᑮᑎᒧᑦ ᖃᓄᐃᓐᓂᖏᑦ ᑕᐃᒪᐃᓐᓂᖅᑰᖅᑐᖅ ᒥᔅᐅᓴᑦᑕᐅᓂᖏᓐᓄᑦ ᐊᑦᑎᑦᑐᒦᕐᓯᒪᔪᑦ ᐊᒥᓲᓂᖏᑦ 1990−ᖏᓐᓂᑦ, ᓄᖅᑲᐅᓯᕋᔅᓴᐅᖏᑦᑐᖅ ᑖᒃᑯᐊ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᓲᓂᖏᑦ 2010−ᖏᓐᓂᑦ ᑭᒻᒐᖅᑐᐅᓗᑎ ᕿᕙᓇᐊᖅᑐᓂᑦ ᐊᖏᓂᖓᓄ BB ᓇᓄᖏᓐᓄᑦ. 2010−ᖏᓐᓂ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑭᒻᒐᖅᑐᐃᒐᓗᐊᖅᑎᓪᓗᒋᑦ ᐱᐅᓛᓂ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒻᒪ ᓈᒻᒪᑦᑐᑎ ᐊᐅᓚᑕᐅᓂᖏᓐᓄᑦ, ᑐᓐᖓᕕᒋᓗᒋᑦ ᓇᑐᓇᐃᖅᓯᒡᒍᑎᒋᔪᓐᓇᖏᑕᕙᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᕿᒌᓯᒪᔪᓂᑦ 1993−2013 ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ.

2011 – 2013 ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐃᒻᒥᑰᖅᑐᓄᑦ ᐊᒃᓇᓄᑦ ᐊᑦᑎᓗᐊᖅᑰᖅᑐᖅ ᐃᑲᔪᖅᑐᐃᓗᓂ ᒪᑭᑕᑦᑎᐊᖅᑐᓂ ᐊᒥᕉᓂᖏᑦ, ᑭᓯᐊᓂ ᓄᓇᖓᓯᒍᑦ ᐊᑐᖑᐊᕐᓂᖅ (ᓱᕐᓗ, ᐊᒥᕉᓂᖏᑦ ᓈᒻᒪᓐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᖃᓯᐅᔾᔭᐅᓗᑎ ᐆᒪᓂᖏᑦ, ᐱᔭᐅᔪᑦ ᐊᒻᒪ ᓇᓐᓄᑦᑕᐅᔪᑦ) ᐱᑕᖃᓇᐊᖃᖅᑐᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᖃᐅᔮᔭᐅᔪᓂᑦ ᐊᒻᒪ ᐊᒥᕉᓂᓇᔪᓐᓇᖅᑕᖏᑦ (ᓱᕐᓗ, ᐃᓄᓐᓅᖏᑦᑐᖅ ᐋᖅᑕᐅᑎᓪᓗᒋᑦ) ᓇᓄᐃᑦ ᐊᒥᓱᖅᐸᓪᓕᐊᓂᖏᑦ. ᐊᑦᑎᑦᑐᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐃᒻᒥᑰᖅᑐᓄᑦ ᐊᖑᑎᓄᑦ ᐃᓱᒫᓘᑎᖃᕈᓐᓇᖅᑐᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᕐᓗᓂ. ᑭᓯᐊᓂᓕ, ᕿᓚᒥᐅᔪᒧᑦ ᐆᒪᔪᓂᑦ ᐱᔭᐅᔪᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2010–ᖏᓐᓂᑦ, ᑭᓪᓕᓯᓂᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓗᓇᕈᑎᖅᑐᑎᑦ, ᐊᒻᒪ ᐱᐅᖏᑦᑐᒥᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᔪᓐᓇᕐᓂᖓ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᑐᖅᑕᐅᑲᐃᓐᓇᕐᓂᕙᑦᑕᖓ (ᓵᕐᓗ ᐱᖃᑕᐅᖏᑦᑎᓂᐅᔪᑦ ᐅᖃᐅᓯᐅᓚᔪᑦᑐᑦ ᐃᓱᐊᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ) ᑭᓪᓕᖅᑎᑦᑎᔪᖅ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒥᕉᓂᖏᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐅᕝᕙᓘᓐᓃᑦ ᒫᓐᓇᐅᔪᒥᑦ ᖃᓄᐃᓕᖓᓂᕆᔭᖏᑦ BB ᓇᓄᖏᑦ ᑐᓐᖓᓂᖅᑐᑎᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᐆᒪᓂᖏᓐᓄᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ.

ᓵᑉᑕ 6: ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᓂᑦ–ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᐊᑐᖅᑐᑎᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓴᒪᕐᖀᓐᓂᒃ ᖃᐅᔨᓴᕐᓂᖅᕕᐅᓯᒪᔫᓐᓂᒃ MR–ᒥ ᖃᐅᔨᓴᓂᕐᒥᑦ BB–ᒥᑦ (1993-1995, 1997 ᐊᒻᒪᓗ 2011- 2013) ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔨᒌᖏᓂᖏᑦ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖏᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓪᓗᑎ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᐊᒥᕉᓂᖏᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓴᖅᑐᒋᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᕿᑐᓐᖏᖃᑦᑕᕐᓂᖏᑦ ᐊᔾᔨᒌᖏᓂᖃᓚᐅᖅᑐᑦ ᓯᑯᖓᓄᑦ ᖃᓄᐃᓐᓂᕆᔭᖓᓄᑦ. BB–ᒥ, ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐊᒥᕉᓂᕆᖃᑦᑕᖅᑕᖏᑦ ᐊᑎᖅᑖᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᕐᕌᒍᒧᑦ (COY) ᐅᕝᕙᓘᓐᓃᑦ ᐊᑎᖅᑕᐃᑦ. ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᒥᕉᓂᖏᑦ ᐊᖏᔫᓗᐊᓚᐅᖏᑦᑐᖅ ᐊᒻᒪ ᐊᓯᔾᔨᕐᓂᖃᓚᐅᖏᑦᑐᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᐱᕐᖓᔅᓵᖅᐸᓪᓕᐊᓂᖓᓄᑦ.

ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᒥᑦ ᐊᓯᔾᔨᑲᐃᓐᓇᖅᑕᖅᑐᓂᑦ ᐅᕙᒍᑦ ᑎᑎᕋᖅᓯᒪᔭᖏᓐᓂ COY ᐊᕐᕌᒍᑕᒫᑦ (ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ COY ᐊᒥᕉᓂᖏᑦ ᐊᑕᐅᓯᖅ ᐊᕐᓇᓪᓗᑦ MR–ᒥ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ) ᐊᑦᑐᐊᓂᖃᓚᐅᖅᑐᑦ ᐊᔾᔨᒌᖏᖃᑦᑕᕐᓂᖓᓄᑦ ᓯᑯᖓ ᖃᓄᐃᓐᓂᖓ. 1993–ᒥ 2013–ᒧᑦ. ᐊᔾᔭᐸᓗᖏᑦ ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑦ ᐊᑕᐅᑦᑎᒃᑯᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐅᐱᕐᖔᒃᑯᑦ ᓯᑯᐃᔅᓴᕋᐃᓐᓂᖓᓄᑦ. ᐊᔾᔭᐸᓗᖏᑦ ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑦ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᐊᑦᑐᐊᓂᖃᖅᑐᓂ ᓯᑯᖓᓄ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓄᑦ ᐅᓂᒃᑲᓕᐊᖑᕋᑖᓚᐅᖅᑐᑦ ᐊᓯᖏᓐᓄᑦ ᓇᓄᕐᓄᑦ.

ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ (ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᕉᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᑕᐅᓯᕐᒧᑦ ᐊᕐᓇᓪᓗᒧᑦ MR–ᒥ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ) BB-ᒧᑦ ᖃᐅᔨᓴᕐᓂᖅᑎᓪᓗᒋᑦ 1993-2013 ᐊᔾᔨᒌᖏᓂᖃᓚᐅᖅᑐᑦ 0.24 ᑎᑭᑦᑐᒍ 0.51. ᑖᒃᑯᐊ ᖃᐅᔮᔭᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ BB ᑕᑯᔅᓴᐅᑎᑦᑎᕙᓪᓕᐊᖏᓐᓇᖅᑐᖅ ᖁᕝᕙᓯᓐᓂᖏᑦ ᕿᑐᓐᖏᐅᕐᓂᓇᖃᑦᑕᓇᐊᖃᖅᑕᖏᑦ ᓈᒻᒪᑦᑐᓄᑦ ᐊᒥᕉᓂᖏᓐᓄᑦ, ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓃᖑᖅᑐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐅᓂᒃᑲᓕᐊᖑᔪᑦ Regehr et al. (2015), ᐅᖃᐅᔾᔨᓚᐅᖅᑐᑦ ᐊᔾᔨᒌᓐᓂᖏᑦ ᐊᑎᖅᑕᖏᓐᓂᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᓯᔾᔨᕈᑎᓪᓗᐊᑕᓇᔭᖓ ᐊᒥᕉᓂᖏᓐᓄᑦ ᐱᕈᖅᐸᓪᓕᐊᓂᖏᓐᓄᑦ ᐅᖓᓯᓐᓂᖃᖅᑐᓂᑦᖕ 0.1 ᑎᑭᑦᑐᒍ 0.3 ᐊᒥᕉᓂᖏᓐᓄᑦ ᐊᒥᓱᖅᐸᓪᓕᐊᓗᑎ.

ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᐊᑦᑐᐊᓂᖃᖅᑐᓂᑦ ᐊᑯᓐᓂᖏᓐᓂ COY ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᕐᕌᒍᖓᓂ t ᐊᒻᒪ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ ᐊᕐᕌᒍᒥ year *t* + 1, ᐊᒻᒪᓗᑦᑕᐅᖅ ᓴᖏᔪᒥᑦ ᐊᑦᑐᐊᓂᖃᖅᑐᓂ COY ᐊᑎᖅᑕᐃᑦ ᐊᒥᕉᓂᖏᑦ ᐊᒻᒪ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐊᕐᕌᒎᓂᐊᖅᑐᒧᑦ. ᑖᒃᑯᐊ ᖃᐅᔮᔭᐅᔪᑦ ᐊᒥᕉᓂᖅᓴᓇᔭᖏᑦ ᑐᕐᑯᖃᑦᑕᖅᑐᑦ ᓇᓄᐃᑦ ᐊᑎᖅᑕᐅᑎᓪᓗᒋᑦ ᓯᕗᓪᓕᖅᐹᖏᓐᓂᑦ 8–ᓂ ᑕᖅᑭᓂᑦ, ᑕᐃᒪᓐᓇ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᑐᓐᖓᓂᖅᑐᑦᐊᒥᕉᓂᖏᑦ COY ᐃᕐᓂᐊᖑᓚᐅᖅᑐᑦ ᐊᒻᒪ ᐆᒪᓂᖅᑐᑎᑦ ᑎᑭᑦᑐᒍ ᓯᕗᓪᓕᖅᐹᖑᓗ ᓯᑯᖃᕐᓃᓯᑎᓪᓗᒍ.

ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᕐᓂᖃᖅᑐᓂᑦ ᐊᒥᕉᓂᖏᓐᓂᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᑎᖅᑕᓂᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ. ᑭᓯᐊᓂᓕ, ᐊᑦᑐᐊᓂᖃᕐᓂᖓᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ COY ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒦᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᑦ ᐊᒻᒪ ᓯᑯᖓ, ᐊᒻᒪ ᐊᑯᓂᐅᔪᒧᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᓯᑯᖓᓂ BB–ᒥᑦ, ᑲᖐᓯᑦᑐᒍᑦ ᐊᑦᑎᓪᓕᕆᐊᕐᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ ᖃᐅᔨᓇᕋᔭᖅᑐᖅ ᐊᖏᓂᖅᓴᓄᑦ ᐊᒻᒪ ᐊᑯᓂᐅᓂᖅᓴᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᒧᑦ.

ᐊᑎᖅᑕᐃᑦ ᐊᒥᓲᓂᕆᖃᑦᑕᖅᑕᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ ᐃᒪᒋᔭᖏᓐᓂᑦ COY ᐊᒻᒪ ᐊᑎᖅᑕᐃᑦ ᐊᕝᕚᐸᓗᕆᓚᐅᖅᑕᖏᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᓂᑦ Foxe Basin– ᐊᒻᒪ, ᓂᕆᖓᓂ ᑕᓱᐅᔭᕐᔪᐊᖅ . ᑕᑯᓐᓇᖅᑐᒍ, ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᕿᑦᑎᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᓴᓂᐊᓂ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓄᑦ ᕿᑭᖅᑖᓘᑉ ᐃᑭᑫᓴᖓᓄᑦ ᐊᒻᒪ ᐱᖓᓐᓇᖓᓂ ᑕᓱᐅᔭᕐᔪᐊᖅ. ᑖᒃᑯᐊ ᐊᓪᕚᕿᓐᖑᐊᑎᑦᑐᕆᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ BB ᐊᓲᕝᕚᖅᓯᒪᓗᐊᖏᑦᑐᖅ ᕿᑐᓐᖏᐅᕝᕙᐅᖃᑦᑕᕐᓂᖓ ᑕᒫᓂ ᖃᑦᑏᓐᓇᕈᓇᐊᖅᓯᒪᔫᔭᕋᑖᓚᐅᕋᓗᐊᖅᑎᓪᓗᕆᑦ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ.

ᓈᑉᑕ 7: ᓇᓄᐃᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᐅᖅᓱᖃᕐᓂᖓᓄᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ (FI) ᐊᒥᓱᒐᓚᐃᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᔭᐅᔪᓂᑦ ᒪᕿᓐᓂ MR ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ BB–ᒥᑦ, 1993-1995, 1997 ᐊᒻᒪᓗ 2011- 2013. ᖃᐅᔨᓴᓚᐅᖅᑕᕗᑦ ᐊᓲᕝᕚᖃᑦᑕᕐᓂᖏᑦ ᖃᐅᔨᓴᕈᑎᒥᕐᓂᑦ ᑎᒥᖏᓐᓄᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᑕᒪᒃᑭᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔫᓐᓂᒃ ᐊᑦᒃᑐᐊᓂᖃᖅᑐᑎᑦ ᓯᑯᖓᓄᑦ ᖃᓄᐃᓐᓂᓇᔭᖓᓄᑦ.

ᖃᐅᔾᓚᐅᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᐊᑦᑎᓪᓕᒋᐊᕐᓂᖏᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᓇᓄᕐᓂᑦ BB–ᒥ ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2013. ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᓇᓄᕐᓂᑦ BB–ᒥ ᐊᑦᑎᓪᓕᒋᐊᓚᐅᖅᑐᑦ ᐊᑦᑐᐊᓂᖃᖅᑐᑎᑦ ᐃᒪᐅᖏᓐᓇᕐᓂᖓᓄᑦ ᐊᒻᒪ ᐅᐱᕐᖔᒃᑯᑦ ᓯᑯᐃᕝᕕᒋᕚᑦᑕᖓᓄᑦ. ᑖᓇ ᒪᓕᓐᓂᖃᖅᑐᖅ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᓇᐊᖃᖅᑐᓂᑦ ᐊᑯᓂᐅᓂᖓᓂᑦ ᓯᑯᒥᒍᓐᓃᕐᓂᓇᓚᐅᖅᑕᖏᑦ (ᐊᒻᒪᓗᖃᐃ ᐊᑦᑎᓪᓕᒋᐊᖅᑐᑎᑦ ᓂᓇᖃᑦᑕᖅᑕᖏᑦ ᐅᐱᕐᖔᒃᑯᑦ ᐊᐅᔭᕈᕋᑖᖅᑎᓪᓗᒍ ᐊᐅᓚᔾᔪᑎᓪᓗᐊᑕᓇᔭᖓ ᐊᑦᑎᓪᓕᒋᐊᕐᓂᖓᓄᑦ.

ᖃᐅᔨᓴᕈᑎᒥᓂᖅ>ᑦ ᖃᐅᔨᓴᕐᓂᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᑦᑎᓪᓗᓪᓕᐊᓪᓗᑎ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ BB–ᒥ ᓇᓄᖏᓐᓂ 2010–ᒥ ᐊᑐᑐᑎ ᐊᕝᕚᕿᖏᑦᑐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᕐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᐅᓚᑕᐅᔪᓂᑦ ᓇᓄᕐᓂᑦ (Rode et al. 2012). ᖃᐅᔨᓴᖅᑐᑎ ᐅᓂᒃᑲᓕᐊᓂᑦ ᓈᑉᑕ 7 ᐊᑕᖑᔪᒦᓚᐅᖅᑐᖅ ᑕᒪᒃᑭᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᓯᒪᔫᓐᓂᒃ (1993-1997 ᐊᒻᒪ 2011-2013) ᓇᓗᓇᐃᖅᓯᔪᑦ ᖃᐅᔨᔭᐅᔪᑦ Rode et al. (2012) ᑭᒡᒐᖅᑐᐃᔪᑦ BB–ᒥᑦ ᐊᖏᓂᖓ ᒥᑭᓂᖅᓴᐅᓚᐅᕋᓗᐊᖅᑎᓪᓗᒍ ᑭᓪᓕᖅᑎᑕᐅᓂᖓ.

ᑖᒃᑯᐊ ᖃᐅᔨᔭᐅᔪᑦ ᒪᓕᓐᓂᖃᕐᒥᔪᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ BB–ᒥ ᐱᐅᖏᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐱᒋᐊᕐᓂᖏᓐᓂ 200–ᖏᓐᓂᑦ ᐊᕝᕚᑲᐸᓗᑦᑐᓂ 1990–ᖏᓐᓂᑦ (Dowsley ᐊᒻᒪ Wenzel 2008, Born et al. 2011)

ᖃᐅᔾᓚᐅᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᓄᓇᒥ ᓂᓇᓇᓱᐊᖅᑐᓂᑦ ᖃᓂᒋᔭᐸᓗᖏᓐᓂ 9% BB ᓇᓄᖏᑦ ᖃᐅᔨᔭᐅᔪᑦ ᓄᓇᒦᑦᑎᓪᓗᒋᑦ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ. ᐃᒪᕐᒥᐅᑕᐃᑦ ᐆᒪᔪᐃᑦ, ᐱᓗᐊᖅᑐᒥ ᓇᑦᑏᑦ ᐊᒻᒪ ᐊᐃᕖᑦ, ᓇᐸᓪᓗᐊᐸᓗᕆᓚᐅᖅᑕᖏᑦᓇᓗᓇᐃᕋᔅᓴᐅᔪᓂᑦ ᓂᕆᖃᑦᑕᖅᑕᖏᓐᓂᑦ.

ᓈᑉᑕ 8: ᓇᓐᓄᑦᑕᐅᔪᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin: ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᓯᒪᔪᑦ ᐊᒻᒪ ᓇᓐᓄᑦᑕᐅᔪᓂᑦ ᑐᓴᖅᑎᑦᑎᓂᖅ, 1993 ᑎᑭᑦᑐᒍ 2014.

ᓈᑉᑕ 8 ᓇᓗᓇᐃᔭᐃᔪᖅ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᑐᓂᑦ BB-ᒥ ᐊᒻᒪ KB–ᒥᑦ. ᖃᐅᔨᓴᓚᐅᖅᑕᕗᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐊᒻᒪ ᐋᖅᑭᓯᒪᓂᖏᑦ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᑐᓂᑦ ᖁᓕᓂᑦ ᐊᕐᕌᒍᖃᑦᑕᖅᓯᒪᔪᓂᑦ ᐊᓲᕝᕚᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒥᓲᓂᖏᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᑦᑐᐊᓂᖃᕐᓂᖏᓐᓄᑦ ᐊᓯᖏᓐᓄᑦ ᐱᔾᔪᑕᐅᔪᓄᑦ. ᖃᐅᔨᓴᓚᐅᕐᒥᔪᒍᑦᑕᐅᖅ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓄᑦ ᐊᑐᖅᑕᐅᕙᑦᑐᑦ ᓖᓐᓇᐅᔪᒥ; ᓇᓗᓇᐃᖅᑐᒋᑦ ᐊᑦᑐᐃᓂᖃᕈᓐᓇᖅᑐᓂᑦ ᑕᒻᒪᖅᓯᒪᖏᓐᓂᖏᓐᓄᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᒻᒪ ᐱᔫᓇᕐᓂᖏᑦ ᓚᕙᒪᖃᕐᕕᐅᔪᑦ ᐊᐅᓚᑦᑎᑎᐊᕐᓗᑎ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓂᑦ. ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᐅᖃᐅᒥᓯᐅᔫᓇᖅᑐᓂᑦ ᓯᕗᓂᕐᓴᒥ ᓇᓐᓄᑦᑕᐅᔫᓇᖅᑐᓂᑦ ᖁᕝᕙᓯᓐᓂᖏᓐᓂᑦ ᐊᒻᒪ ᓇᓗᓇᐃᖅᑕᑲᐅᓂᖓ ᐊᒥᓲᓂᖏᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ. ᐊᑐᓕᖁᔭᐅᔪᑦ ᐱᐅᓯᑎᕆᐊᖅᑕᐅᓗᑎᑦ ᓇᓐᓄᑦᑲᑕᐅᕙᑦᑐᓂᑦ ᑐᓴᖅᑎᑦᑎᓂᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐱᑕᖃᖅᑎᑕᐅᓗᑎᑦ.

ᑕᒪᒃᑭ ᑲᓇᑕ (ᓄᓇᕗᑦ) ᐊᒻᒪ ᐊᑯᑦᑐᓂᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᑦ BB−ᒥᑦ ᐊᒻᒪ KB−ᒥᑦ ᓇᓄᖏᓐᓂᑦ. 1993-2005−ᒧᑦ (ᓲᕐᓗ, ᑰᑕᖃᖅᑎᑕᐅᓚᐅᖅᑎᓐᓇᒋᑦ ᐊᑯᑭᑦᑐᑦ) ᑲᑎᓐᖓᓪᓗᒋ (ᓲᕐᓗ, ᑕᒪᒃᑭᓐᓂ ᓄᓇᓕᕐᔪᐊᓂᑦ) ᐊᕐᕌᒍᑕᓖᑦ ᓇᓄᑦᑕᐅᓚᔪᓚᐅᖅᑐᑦ 165 (ᐊᒥᓲᓂᖏᑦ: 120-268) 12 (ᐊᒥᓲᓂᖏᓐᓂᑦ: 6-26) ᓇᓄᐃᑦ BB-ᒥᑦ ᐊᒻᒪ KB−ᒥ ᓇᓄᖏᓐᓄᑦ, ᑐᖏᓕᕇᑦᑐᑎ. ᑲᑎᑦᑐᒋᑦ ᓇᓄᑦᑕᐅᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ 2002 ᐊᒻᒪ 5005 ᒪᓕᓐᓂᖃᖅᑐᑦ ᐊᒥᓱᓚᓐᓂᑦ ᖃᓄᐃᓘᖅᑐᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ ᑐᓴᖅᑎᑦᑎᓂᕐᒥᑦ ᐊᒻᒪ ᓇᓄᑦᑕᐅᔪᓐᓇᖅᑐᓂ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᑕᒪᒃᑭᓐᓂ ᑲᓇᑕᒥ ᐊᑯᑭᑦᑐᓂᓗ.

BB−ᒥ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ ᑲᑎᓐᖓ)ᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ ᖃᓂᒋᔭᖕᖓᓃᖏᓐᓇᖅᓯᒪᔪᖅ 2:1 (ᐊᖑᑎᑦ: ᐊᕐᓇᐃᑦ) ᐊᕐᕌᒍᑕᓖᖑᓚᔪᑦᑐᓂ 35% ᐊᕐᓇᕐᓗᐃᑦ ᐃᒻᒥᒎᖅᑐᑦ ᓇᓄᐃᑦ. KB−ᒥ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ ᑲᑎᓐᖓᔪᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ 33%−ᖑᓚᐅᖅᑐᖅ ᐊᕐᓇᐃᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ 1993-2014. ᒥᒃᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ KB−ᒥᑦ ᓇᓄᑦᑕᐅᓯᒪᔪᑦ ᑰᑕᖃᓕᖅᑎᑕᐅᔪᒻᒪᑕᓂᑦ ᐊᑯᑭᑦᑐᑦ 44%−ᖑᓯᒪᔪᑦ ᐊᕐᓇᐃᑦ ᑭᓯᐊᓂ ᐱᓪᓚᕇᒋᔭᖓ ᒥᒃᓴᐅᓴᑦᑕᐅᓂᖓᓄᑦ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ KB−ᒥ ᓇᓄᑦᑕᑲᐅᔪᓂᑦ ᓴᖑᑦᑐᖅ.

ᐊᑯᑭᑦᑐᓂᑦ ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᓇᓄᑦᑕᑲᐅᕙᑦᑐᑦ ᐊᑯᓐᓂᖏᓐᓂ ᔫᓐᓄᐊᕆ ᐊᒻᒪ ᔫᓂ BB−ᒥ ᐊᒻᒪ KB−ᒥᑦ, ᓄᓇᕗᒻᒥᓕ ca. 40% ᓇᓄᑦᑕᐅᔪᑦ BB−ᒥᑦ ᐊᐅᔭᒃᑯᑦ ᐅᑭᐊᒃᓵᒧᑦ (ᐋᒡᒌᓯ − ᓄᕖᐱᕆ) ᓇᓄᐃᑦ ᓯᔾᔭᖅᐸᓯᒻᒦᑎᓪᓗᒋᑦ. ᓄᓇᕗᒻᒥ, ᓇᓄᑦᑕᐅᖃᑦᑕᖅᑐᑦ KB−ᒥᑦ ᐅᐱᕐᖔᒃᑯᑦ.

ᓇᓄᓚᓱᑐᐃᓐᓇᕐᓂᖅ ᐱᔪᓐᓇᖅᑎᑕᐅᔪᖅ ᑲᓇᑕᒥ ᑭᓯᐊᓂ ᐊᑯᑭᑦᑐᓂᐅᖏᑦᑐᖅ. ᓇᓄᓚᓱᑐᐃᓐᓇᕐᓂᖅ ᑭᓪᓚᖅᑐᐃᔪᖅ 16% ᐊᕐᕌᒍᑕᓖᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐊᒻᒪ ᖁᕝᕙᓯᓐᓂᖃᓲᖑᓚᐅᖅᑐᑎ 2008−ᒥ ᒪᓕᓐᓂᖃᖅᑐᑦ ᐊᒥᓱᓚᓐᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᔪᓂᑦ. ᐊᖑᓇᓱᑐᐃᓐᓇᕐᓂᐅᔪᑦ ᓂᕈᐊᖃᑦᑕᓲᑦ ᓇᓄᑐᖃᐅᓂᖅᓴᓂᑦ, ᐃᓐᓇᕐᓂᑦ ᐊᖑᑎᓂᑦ.

ᐃᒻᒥᓂ ᐱᖁᑎᓂᓪᓗ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᑐᖁᑕᐅᔪᑦ (DLPs) ᓇᓄᐃᑦ BB−ᒥᑦ ᐊᒥᓲᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᒪᒃᑯᓂᖅᓴᐃᑦ (2-3 ᐊᕐᕌᒍᓖᑦ). ᖃᐅᔨᓂᒥᖃᓚᐅᖏᑦᑐᒍᑦ ᐊᓯᑉᔨᖃᑦᑕᖅᑐᓂᑦ ᐊᕐᕌᒍᑕᓖᑦ ᐊᒥᒨᓂᖏᑦ DLPs ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2014. ᐊᒥᓲᓂᖅᓴᐃᑦ DLPs ᐃᒪᐃᓪᓇᕖᖅᐸᓚᐅᖅᑐᑦ (ᐋᒡᒌᓯ−ᓄᕖᐱᕆ) ᓇᓄᐃᑦ ᓯᔾᔮᑦᑎᓪᓗᒋᑦ ᓯᔾᔭᖅᐸᓯᒻᒥᓘᓐᓃᑦ. ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓴᖅᑭᕐᓂᕆᕙᑦᑕᖏᑦ DLPs ᑲᓇᑕᒥ (ᓄᓇᕗᑦ) ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᔪᑦ ᐊᐅᔭᒃᑯᑦ ᐅᑭᐊᒃᓵᑯᓪᓗ.

ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᖅᑕᐅᓃᑦ ᐊᖑᑕᐅᓂᖏᓐᓄᑦ ᐊᕐᓇᐅᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ, ᐅᓂᒃᑳᓕᐊᖑᔪᑦ ᐊᖑᓇᓱᑦᑎᓄᑦ, ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᖑᑕᐅᓂᖓ ᐊᕐᓇᐅᓂᖓ ᓇᓗᓇᐃᖅᑕᐅᑲᒻᒪᔅᓯᒪᓚᐅᖅᑐᑦ ᐊᒥᓱᓚᐅᔪᓂᑦ ᐊᒻᒪ ᑖᒻᒪᕐᓂᐅᔪᑦ ᐊᒥᓲᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐊᑯᑭᑦᑐᓂᑦ. ᑐᓐᖓᓂᖃᖅᑐᑎ ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᖅᑕᐅᓂᖏᓐᓄᑦ, ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᑕᑯᔅᓴᐅᓂᖏᑎᒍᑦ (ᓲᕐᓗ ᓯᐅᑎᖏᑎᒍᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ ᐊᒥᐊᖅᑕᐅᓯᒪᔪᓪᓘᓐᓃᑦ) ᓇᓄᕐᓂᑦ ᖃᑦᐲᓐᓇᐅᓗᐊᖅᑐᑦ ᑐᓴᖅᑎᑦᑎᔪᑕᐅᓯᒪᔪᑦ. ᖃᐅᔨᓴᐅᓂᖏᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᔪᑦ ᐱᑐᖃᕈᖅᐸᓕᐊᓪᓗᑎ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᔭᒐᐃᓪᓗᑎ, ᐱᓗᐊᖅᑐᒥ ᔭᒐᑦᑐᑎ ᓯᐅᑎᖏᑎᒍᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ, ᐊᑲᐅᖏᓕᐅᕈᑕᐅᓚᐅᖅᑐᖅ. ᑖᓐᓇ ᖃᐅᔨᓴᐅᓂᐅᔪᖅ ᐊᑦᑐᕈᑎᖃᖅᑐᖅ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐅᑎᖅᑎᑕᐅᓂᖏᓐᓄᑦ MR−ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ.

ᐊᐅᓚᑦᑎᓂᐅᔪᖅ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᐅᖃᑦᑕᖅᓯᒪᔪᑦ ᓇᓄᕐᓂᑦ ᑲᓇᑕᒥ ᐊᑯᑭᑦᑐᓂᓗ ᐊᔾᔨᒌᖏᑦᑐᑦ ᐃᓚᖓᑎᒍᑦ: (1) ᓇᓄᑦᑕᐅᔪᓐᓇᖅᑐᑦ ᑭᓪᓕᖏᑦ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᑦ 1967−ᒥᑦ ᐊᒻᒪ ᐊᑯᑭᑦᑐᓂᑦ 2006−ᒥᑦ, (2) ᓇᓄᓚᓱᐊᖅᑐᖃᑐᐃᓐᓇᕈᓐᓇᖅᑐᖅ ᑲᓇᑕᒥ ᑭᓯᐊᓂ ᐊᑯᑭᑦᑐᓂᐅᖏᑦᑐᖅ, (3) ᑲᓇᑕᒥ ᐊᐅᓚᑕᐅᓂᖏᑦ ᓴᓇᓯᒪᔪᑦ ᓇᓄᑦᑕᐅᔪᓐᓇᕐᓗᑎ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ 2:1 (ᐊᖑᑎᓂᑦ ᐊᕐᓇᓄᑦ) ᓇᓄᑦᑕᐅᔪᑦ ᐊᑯᑭᑦᑐᓂᑦ ᓂᕈᐊᕐᓂᖅᓯᒪᖏᑦᑐᑦ, ᐊᒻᒪ (4) ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᐊᑯᑭᑦᑐᓂᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᑦ ᕿᒧᒃᓯᒃᑯᑦ ᑲᓇᑕᒥᓕ ᖃᒧᑕᐅᔭᒃᑯᑦ.

ᓵᑉᑕ 9: ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖏᑦ Kane Basin−ᒥᑦ

ᐊᕐᕌᒍᑕᒫᑦ ᓯᑯᖓ ᓇᔪᖅᑕᐅᖃᑦᑕᕐᓂᖓ KB−ᒥᑦ ᐊᓯᔾᔨᖅᓯᒪᔪᖅ ᐊᕐᕌᒍᓕᒫᒥ ᓯᑰᖏᓐᓇᖅᑐᒥ (>30% ᐊᐅᔭᒃᑯᑦ) ᐊᔾᔨᐸᓗᒋᓕᖅᑕᖓᓄᑦ ᐊᕐᕌᒍᑕᒫᑦ ᓯᑯᖃᑦᑕᕐᓂᖓᓄᑦ ᐊᐅᑦᑐᐃᓇᐅᓪᓗᓂ ᐊᐅᔭᒃᑯᑦ (<5% ᐱᔭᐅᓯᒪᔪᑦ). ᓯᑯᐃᖅᓴᖁᐅᓐᓂᖅᓴᐅᓕᖅᑐᖅ ᐅᐱᕐᖔᒃᑯᑦ 7−ᓂ ᐅᓪᓗᓂ/ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ, ᐊᒻᒪ ᓯᑯᓇᓵᕐᓂᓴᐅᓪᓗᓂ ᐅᑭᐊᔅᓵᒃᑯᑦ 5-6 ᐅᓪᓗᓂ/ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ, ᐊᒻᒪ ᐊᑯᓂᐅᓂᖓ ᐊᐅᔭᖓ (ᓲᕐᓗ, ᐅᓪᓗᐃᑦ ᐊᒥᓲᓂᖏᑦ ᓯᑯᐃᕐᓂᖓᓄᑦ ᐊᒻᒪ ᓯᑯᓂᖓᓄᑦ) ᖁᕝᕙᕆᐊᖅᓯᒪᔪᑦ 12 ᐅᓪᓗᓂ/ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ. ᓯᑯᖃᑦᑕᕐᓂᕆᒐᔪᑦᑕᖓ ᔫᓂᒥ ᐅᑦᑑᐱᕆᒧᖅ ᐊᑎᓪᓕᕚᓪᓕᐊᔪᖅ 5-6% ᖁᓕᐅᓂᖅᑕᑕᒫᑦ ᐊᕐᕌᒍᓂᑦ.

KB ᓇᓄᖏᑦ ᖃᓄᐃᓕᓯᒪᔪᑦ ᐊᓯᔾᔨᐸᓪᓕᐊᓂᖓᓄᑦ ᓯᑯᖓ ᖃᓄᐃᓐᓂᕆᔭᖓ ᐅᖓᓯᓐᓂᖅᓴᒧᑦ ᓇᒧᖓᖅᐸᓕᖅᑐᑎᑦ ᐊᒻᒪ ᓇᔪᖅᐸᑦᑕᖏᑦ ᐊᑐᖃᑦᑕᖅᑕᖏᓐᓄᑦ ᐊᔾᔨᖃᐸᓗᑦᑐᑦ ᓇᓄᖏᓐᓄᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓯᑯᒦᑉᐸᑦᑐᑦ (ᒎᕐᓗ, ᐊᖏᓪᓕᒋᐊᖅᓯᒪᔪᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓇᔪᖅᐸᑦᑕᖏᑦ, ᑕᑯᒍᑦ ᓵᑉᑕ 2 ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖓ ᓯᑯᖃᖏᓐᓂᖅᓴᐅᓂᖓ ᐊᐅᔭᒃᑯᑦ ᐊᒻᒪ ᐅᑭᐊᔅᓵᒃᑯᑦ). ᐅᓪᓗᓂᑦ ᑎᓴᒪᓂᑦ ᐃᖏᕋᓂᖃᖅᑐᑦ KB−ᒥᑦ ᐊᑦᑎᓐᓂᖅᓴᐅᔪᑦ ᓴᓂᐊᓂ BBᒥᑦ ᐊᒻᒪ ᖃᑦᑏᓐᓇᐅᓂᖅᓴᐅᓯᒪᔪᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐊᐅᓚᓂᖏᓐᓂᑦ. ᐊᖏᔪᓂᑦ ᐊᔾᔩᖏᓐᓂᖃᓚᐅᖏᑦᑐᑦ ᑎᓴᒪᓄᑦ ᐅᓪᓗᑦ ᐃᖏᕋᓂᖏᓐᓄᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ KB−ᒥᑦ ᐊᑯᓐᓂᖓᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᕿᑎᐊᓂ ᐅᑦᑑᐱᕆᒥ ᖁᕝᕙᓯᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ r.

ᓄᓇ ᐊᑐᖅᑕᐅᓂᖓ KB−ᒥᒃ ᐊᐅᔭᒃᑯᑦ ᑕᐃᒫᔅᓴᐃᓈᖏᑦᑐᖅ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐃᓚᖏᑦ ᖃᖏᖅᑐᐃᑦ ᓯᑰᖏᓐᓇᕐᓂᖏᓐᓄᒃ ᐊᒻᒪ ᓯᔾᔭᖓ. ᓄᓇᓘᕐᓂᖏᑦ ᓄᓇᒦᒍᓐᓃᕐᓂᖏᓪᓗ ᐅᔾ;ᖏᑦ ᓇᓗᓇᐃᖅᑕᐅᔪᓐᓇᓚᐅᖏᑦᑐᑦ ᖃᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᐅᒻᒧᑦ.

ᐱᖓᓱᑦ ᑎᓲᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ KB−ᒥᑦ 2000−ᖏᓐᓂᑦ. ᐊᐅᓯᐃᑦᑑᑉ ᕿᕿᖅᑖᓗᐊᓃᑦᑐᐃᓐᓇᐅᓚᐅᖅᑐᑦ. ᐊᖏᔪᒥᑦ ᐊᔾᔩᖏᓐᓂᖃᓚᐅᖏᑦᑐᑦ ᑎᓯᓕᐅᕐᓂᖏᓐᓄᑦ, ᐃᓯᕐᕕᖏᑦ ᐅᓪᓗᖏᑦ, ᐊᓂᕕᖏᓪᓘᓐᓃᑦ ᐅᓪᓗᖏᑦ ᐊᑯᓐᓂᖓᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ.

ᐊᑕᖏᖅᑐᒍ, ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ (ᓵᑉᑕ 2) ᐊᒻᒪ ᓇᔪᖅᑕᐅᔪᑦ ᖃᓄᐃᓐᓂᖏᑦ ᑲᑎᖓᖃᓯᐅᑎᓪᓗᑎ ᕿᑐᓐᖏᐅᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᓃᑦ (ᓵᑉᑕ 11) ᐊᒻᒪ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ (ᓵᑉᑕ 12)ᒪ ᓇᓗᓇᐃᖅᓯᔪᑦ ᑖᒃᑯᐊ KB−ᒥ ᓇᓄᐃᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓯᑯᓯᐅᖏᓂᖅᓴᐅᕙᓕᖅᑐᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ, 1990−ᖏᓪᓂᓂᑦ ᖁᕝᕙᕆᐊᖅᓯᒪᔪᓐᓇᖅᑐᖅ ᐊᑕᖏᖅᑐᒍ ᕿᑐᓐᖏᐅᕐᕕᒋᕙᑦᑕᖏᑦ.

ᓵᑉᑕ 10: ᐊᐅᖏᑎᒎᖅᑐᓂ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓄᖏᓐᓂᑦ Kane Basin−ᒥ

ᓵᑉᑕ 10 ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᐊᐅᖏᑎᒎᖅᑐᓂ MR ᖃᐅᔨᓴᕐᓂᐅᔪᒥ KB−ᒥ, ᖃᐅᔨᓴᖅᑐᑦ ᐃᓚᒋᓪᓗᓂᐅᒃ ᐊᑯᓂᐅᔪᒧᑦ (23-ᐊᕐᕌᒍᐃᑦ) ᑕᑯᔭᐅᓂᖏᑎᒍᑦ ᐊᐅᖏᑎᒍᓪᓗ MR ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ. ᑖᓐᓇ ᐱᖃᓯᐅᔾᔨᓚᐅᖅᑐᖅ ᐱᒋᐊᕐᓂᖏᓐᓂ ᐊᕐᕕᓂᓕᓐᓂᑦ ᐊᕐᕌᒍᓂᖅ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᑐᖁᖓᔪᓂᑦ ᐅᑎᖅᑎᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ (1992 – 1997) ᑭᖑᓂᐊᒍᑦ 14-ᓄᑦ ᐊᕐᕌᒍᓄᑦ ᑐᖁᖓᔪᓄᑦ ᐅᑎᖅᑐᑐᐊᑦ (1998 – 2011) ᒫᓐᓇᕋᑖᖑᓯᒪᔪᕐᓗ (2012 – 2014) ᐆᒪᔪᓂᑦ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᑐᖁᖓᔪᓂᑦ ᐅᑎᖅᑎᑕᐅᓂᖏᓐᓄᑦ. ᑐᕌᒐᓚᐅᖅᑕᕗᑦ ᒥᔅᓴᐅᓴᕐᓗᒋᑦ ᒫᓐᓇᐅᔪᒥ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐱᒻᒪᕆᐅᔪᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ ᐊᒻᒪ ᐊᔾᔨᒋᑎᓐᖑᐊᕐᓗᒋᑦ ᐅᓪᓗᒥ ᒨᖓᔪᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑖᒃᑯᓂᖓ ᓴᖅᑭᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᓯᒪᔪᓂᑦ (Taylor et al. 2008).

ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ KB−ᒥ 357 ᓇᓄᐃᑦ (95% CI: 221 – 493) 2013 – 2014−ᒧᑦ. ᐋᖅᑭᑕᐅᒃᑲᓂᖅᑐᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ 224 ᓇᓄᐃᑦ (95% CI: 145 – 303) ᐊᕐᕌᒍᓄᑦ 1995 – 1997. ᑐᙵᓂᖃᖅᑐᑎ MR, ᐊᖏᓂᖓ KB

ᐊᒥᓲᓂᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᓯᒪᓚᐅᖅᑐᑦ 164 ᓇᓄᐃᑦ (95% CI: 94-234) ᐊᕐᕌᒍᓄᑦ 1994-1997 (ᓇᓄᓇᐃᖅᑐᒍ ᑖᓐᓇ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᖅ ᐊᑐᖅᑕᖅ ᐊᔾᔨᒌᖏᑦᑐᓄᑦ ᐊᕐᕌᒍᓄᑦ ᓴᓂᐊᓂ ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᕆᔭᑦᑎᓐᓂᑦ; Taylor et al. 2008).

ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᑎᑎᕋᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᓇᓐᓇᖓᓂ KB ᐊᕐᕌᒍᖏᓐᓂ 2012 – 2014 ᓴᓂᐊᓂ 1994-1997. ᑲᓇᓐᓇᖓ Kane Basin ᕿᓂᕐᕙᐅᓚᐅᖅᑐᖅ 1990–ᖏᓐᓂᑦ ᕿᓂᖏᓂᖅᓴᐅᓪᓗᑎ 2010–ᖏᓐᓂᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓇᓄᖃᑦᑎᐊᖏᓐᓂᖓᓄ ᖃᐅᔨᔭᐅᔪᓂᑦ ᑕᐃᑲᓂ. ᓇᑦᑎᖅᑕᖃᕐᓂᖓ ᑲᓇᓐᓇᖓᓂ Kane Basin ᐅᐱᕐᙳᐹᒃᑯᑦ 1990–ᖏᓐᓂᑦ ᓇᓄᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᑕᒪᓐᓇ ᓇᓄᕐᓄᑦ ᓇᔪᕋᔅᓴᑦᑎᐅᕙᐅᓪᓗᓂ (Taylor et al. 2001). ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᓇᒨᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂ ᐊᒻᒪ 2010–ᖏᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᓐᓇᖅᑐᖅ ᐊᔾᔨᒌᖏᓐᓂᖓᓂ ᓇᒨᖓᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ, ᐊᖑᓇᓱᑦᑕᐅᖏᓐᓂᖓᔫᓄᔅᓴᐅᑐᐃᓐᓇᕆᐊᓕ ᐊᑯᑭᑦᑐᒧᑦ ᑲᓇᓐᓇᖓᓂ KB ᐊᒻᒪ ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᐊᒥᓱᕈᕆᐊᒃᑲᓂᖅᓯᒪᔪᑦ ᓇᒧᓄᐃᑦ KB–ᒥᑦ, ᑭᓯᐊᓂᑦᑕᐅᖅ ᐃᓚᖏᓐᓂᑦ ᐊᔾᔨᒌᖏᓂᖃᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᓂᑦ.

2013 – 2014 ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᓇᓄᓇᐃᖅᓯᔪᑦ 357 (221 – 493) ᓇᓄᐃᑦ ᓛᓐᓇᐅᔪᒥᑦ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ KB–ᒥᑦ ᐅᐱᕐᙳᐹᒃᑯᑦ (ᓅᕐᓗ, Kane Basin ᓇᓄᓕᓛᖏᑦ; Kendall et al. 1997), ᐊᒻᒪ ᓛᓐᓇᐅᔪᒥᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᖁᑦᑎᓂᖅᓴᐅᔪᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᖃᑦᑕᖅᓯᒪᔪᑐᖃᕐᓂᑦ. ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᑕᓖᖓᑐᐃᓐᓇᖅ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᓇᒨᖓᖃᑦᑕᕐᓂᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ, ᐊᔾᔨᒌᖏᓂᓪᓗᐊᑕᓇᕙᑦᑕᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ KB–ᒧᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ 2013-2014 ᐊᒻᒪ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1995-1997 ᖃᓂᒋᔭᐸᓗᖏᓐᓃᓚᐅᖅᑐᑦ 133 ᓇᓄᐃᑦ (ᐊᑐᖅᑕᐅᒐᔪᑦᑐᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ≈ 80 ᓇᓄᐃᑦ), 95% ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᓇᓄᓇᐃᖅᓯᔪᑦ ᐊᒥᓲᑑᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖅᑐᑦ ᒪᕐᕈᓐᓂ ᖃᐅᔨᓴᕐᕙᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ ᐱᐅᔫᔪᓐᓇᓚᐅᖅᑐᖅ. ᑖᓐᓇ ᓇᓄᓇᐃᖅᓯᔪᖅ ᓴᖏᔪᓂᒥᑦ ᓇᓄᓇᐃᖅᓯᒍᑎᖃᖅᑐᓂ ᐊᒥᓱᕈᖅᐸᓪᓕᐊᓂᖏᓐᓂᑦ, ᐊᒻᒪ ᒪᓕᓐᓂᖃᖅᑐᖅ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓇᒨᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᖃᓄᐃᓐᓂᖓ ᐊᒻᒪ ᕿᑐᙵᕆᐅᖃᑦᑕᕐᓂᖏᑦ. ᐅᔾᔨᖅᓱᕐᑯᔨᔪᒧᑦ ᓇᓄᓇᐃᖏᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᒥᓱᕈᒃᑲᓂᕐᓂᖏᓐᓄᑦ ᐊᖏᓪᓕᒋᐊᖅᑕᐅᔪ)ᓐᓇᕐᓂᖓᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐊᑯᓂᐅᓂᖓ ᐊᒻᒪ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᑦ 1990–ᖏᓐᓂᑦ ᓴᓂᐊᓂ 2010–ᖏᓐᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ.

ᓛᓐᓇᐅᔪᒥᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᖃᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐊᕐᕌᒍᓕᓐᓄᑦ 3+ ᐊᕐᓇᐃᑦ (0.95; SE: 0.04) ᐊᒻᒪ ᐱᓯᒪᔭᐅᔪᑦ ᓇᓄᐃᑦ ᒪᓕᓐᓂᖅᑲᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᐅᑌᖅᓯᒪᔪᒥᑦ, ᑭᓯᐊᓂ ᓄᑖᖅ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐊᑦᑎᓂᖅᓴᐅᓪᓗᑎ ᐊᕐᕌᒍᓕᓐᓄᑦ 3+ ᐊᖑᑎᑦ (0.87; SE: 0.06). ᐊᑯᓂᐅᔪᒧᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᖅᐳᑦ, ᐱᔭᐅᔪᖃᓚᐅᖏᑦᑐᖅ ᐊᐅᖏᑎᒍᐳᓪᓘᓐᓃᑦ ᖃᐅᔨᓴᖅᑐᑎᑦ ᐊᒻᒪ ᖃᑦᑏᓐᓇᓂᑦ ᐅᑎᓚᐅᖅᑐᑦ 1998-2011, ᐊᒻᒪ ᐊᒥᓱᓚᐃᑦ ᐊᒥᖏᑦ ᐊᕙᑎᖓᓄ, ᖃᐅᔨᓴᕐᓂᖅ, ᐊᒻᒪ ᐃᖅᑲᓴᐃᔫᖑᔪᓄᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᖃᓄᐃᓐᓂᖅᑎᑦᑎᔪᓐᓇᖅᑐᑦ.

ᑎᑎᕋᖅᓯᓚᐅᖅᑐᒍᑦ ᖃᑦᑏᓐᓇᕈᓇᐊᖅᑐᓂᑦ ᑐᑯᖃᑦᑕᖅᑐᓂᑦ ᐊᑦᑐᐊᓂᓖᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓄᑦ, ᐱᔾᔪᑎᖃᑐᐃᓐᓇᕆᐊᓕ ᐊᑐᓕᖅᑎᑕᐅᓂᖓᓄᑦ ᐊᑯᑭᑦᑐᓂᑦ ᓇᓐᓄᑦᑕᐅᔫᓇᖅᑐᑦ 2006–ᒥ. ᓄᓇᖏᑎᒍᑦ ᐊᑐᖅᑕᐅᓃᑦ ᓇᓄᓇᐃᖅᓯᔪᑦ KB–ᒥ ᓇᓄᐃᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᖁᕝᕙᓯᑦᑐᒥ ᑐᓐᖓᕕᒋᔭᔅᓴᐅᔪᓂᑦ ᐅᐱᕐᙳᐹᒃᑯᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ, <4% ᓇᓄᓇᐃᒃᑯᓯᖅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ ᓅᑦᑐᑎ. ᑖᓐᓇ ᒪᓕᑦᑐᖅ ᓇᒨᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓵᑉᑕ 2–ᒦᑦᑐᑦ.

ᓵᑉᑕ 11: ᖃᖓᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓄᖏᓐᓂ Kane Basin

ᖃᖓᑕᔫᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖃᓚᐅᖅᑐᑦ (ᓵᑉᑕ 10). ᐊᑐᖅᑕᐅᔪᖅ ᒪᓕᓐᓂᖃᓚᐅᖅᑯᑐᖅ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ ᐊᑯᓂᐅᓂᖓᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᕐᕕᐅᔪᓂᑦ KB–ᒥᑦ ᐊᒻᒪ ᐅᕙᑦᑎᓐᓂ ᓇᒻᒥᓂᖅ ᒥᔅᓴᐅᒍᓐᓇᖅᑎᑎᓪᓗᑎ ᐊᒥᓲᓂᖏᓐᓂᑦ ᒪᕐᕉᓐᓂᒃ ᐊᑐᖅᑕᐅᔫᓐᓂᒃ, ᐅᕙᑦᑎᓂᑦ ᐊᔾᔨᖑᐊᖅᑎᑦᑎᔫᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, ᐃᓱᒪᒌᓐᓇᖅᑐᒍᑐᑭᓯᒋᔭᖓ “ᖃᐅᔨᓴᕐᓂᖅ ᐊᒥᓲᓂᖏᓐᓂᑦ” ᐊᔾᔨᒋᒍᓐᓇᖅᑐᑦ ᒪᕐᕉᓐᓂᒃ ᐊᑐᖅᑕᐅᔫᓐᓂᒃ.

ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᐅᐱᕐᖔᒃᑯᑦ 2014–ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ KB–ᒥᑦ 206–ᖑᓚᐅᖅᑐᑦ ᓇᓄᐃᑦ (95% ᑎᑎᕋᖅᑕᐅᔪᑦ CI: 83 - 510). ᑭᓯᐊᓂᓕ, ᖃᐅᔨᓴᖅᑕᐅᑎᐊᖏᓐᓂᖓᓄᑦ ᐃᒪᐃᒻᕼᕐᒥ ᓇᓄᕐᓄᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᓂᑦ ᑖᓐᓇ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᖅ ᐱᐅᖏᑦᑐᒥᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᔪᖅ. ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᑕᒫᖓᑐᐃᓐᓇᖅ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᐊᑐᖅᑕᐅᒐᔪᑦᑐᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ, ᐊᔾᔨᒌᖏᓐᓂᕆᒐᔪᑦᑕᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᑲᑎᑦᑐᒥᑦ ᐊᒥᓲᓂᖏᑦ 2013-2014 MR ᖃᐊᐅᔨᓴᕐᓂᐅᔪᒥᑦ (357 ᓇᓄᐃᑦ, 95% CI = 221 – 493) ᐊᒻᒪ ᖃᖓᑕᓲᒃᑯᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᖃᓂᒋᔭᖏᓐᓃᓚᐅᖅᑐᑦ 151 ᓇᓄᐃᑦ ᐊᔾᔨᒌᖏᓐᓂᕆᒐᔪᑦᑕᖏᑦ ≈ 127 ᓇᓄᐃᑦ), 88%–ᒥ ᖃᐅᔨᓴᖃᑦᑕᓂᕐᒥᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐱᐅᔫᓚᐅᖅᑐᑦ (ᓲᕐᓗ, MR ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᐊᑕᐅᓯᕐᒥᑦ ᓇᓄᕐᒥᑦ ᐊᖏᓂᖅᓴᐅᓚᐅᖅᑐᖅ ᓴᓂᐊᓂ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ).

ᐊᔾᔨᒌᖏᓐᓂᖏᑦ MR ᐊᒻᒪ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ KB–ᒥ ᖃᐅᔨᒪᔭᐅᑦᑎᐊᕆᐊᓖᑦ ᐊᔾᔩᒻᖑᐊᖅᑎᑕᐅᓗᑎ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ ᐊᔾᔨᒋᖕᒋᑕᖏᓐᓂᑦ ᐊᑐᖅᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐅᐱᕐᖔᒃᑯᑦ. ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᑕᑯᑎᑦᑎᓚᐅᖅᑐᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ (ᓲᕐᓗ, ᐊᒥᓲᓂᖏᑦ ᓇᓄᐃᑦ ᓇᔪᖅᓯᔪᑦ ᖃᐅᔨᓴᕝᕕᐅᔪᒥᑦ). NR ᐊᑕᖐᑎᑦᑎᓪᓗᒍ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᖅ ᓇᓄᓕᒫᓂᑦ ᐱᔭᐅᖏᑐᖅᑕᖃᕈᓐᓇᖏᑦᑐᑎᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᒧᑦ). ᐅᖃᐅᔾᔨᒋᐊᖅᑐᒍᑦ MR ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᓈᒻᒪᑦᑐᑎ ᐊᑐᖅᑕᐅᓗᑎ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᑕᑯᔅᓴᐅᑎᑕᐅᒻᒥᔪᖅ ᐊᓯᖏᓐᓂᑦ ᐅᑭᐅᑕᖅᑐᒥ (ᓲᕐᓗ Barents Sea; Aars et al. 2009) ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐊᑑᑎᖃᖅᑐᒥ ᓴᓇᕐᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᑎᑎᕋᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᒥᓲᓂᖏᑦ. ᐊᑐᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᖅ ᑲᔪᓯᑦᑎᐊᕐᓂᖃᖅᓯᒪᔪᑦᐊᖏᓂᖅᓴᐅᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᓴᓂᐊᓂ KB. ᐅᐱᕐᖔᒃᑯᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ KB–ᒥ ᐊᑐᓕᖅᑎᑕᐅᑦᑎᐊᓚᐅᖅᑐᖅ ᒥᑭᔫᓂᖓᓄᑦ ᖃᐅᔨᓴᕝᕕᐅᔪᑉ ᐊᒻᒪ ᓯᓚᑦᑎᐊᕙᐅᖃᑦᑕᕐᓂᖓᓄᑦ, ᑭᓯᐊᓂ ᐱᐅᓯᑎᒋᐊᖅᑕᐅᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᐊᖏᓪᓕᒋᐊᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᒥᔅᓴᐅᓴᑦᑎᐊᓂᖅᓴᖓᓂᐊᕐᓗᒋᑦ ᖃᐅᔨᓴᐃᔭᑦᑕᐅᔪᑦ ᐊᒻᒪ ᐊᑕᖏᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᖅᑯᐅᔪᖅ ᐊᒥᓲᓂᖏᓐᓄᑦ.

ᓵᑉᑕ 12: ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦᒃ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅ)ᔪᓄᑦ–ᐱᔭᐅᒃᑲᓂᖅᑐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ Kane Basin–ᒥ

ᓵᑉᑕ 12 ᓇᓗᓇᐃᔭᐃᔪᖅ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ KB–ᒧᑦ ᐊᑐᖅᑐᑎ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ MR ᖃᐅᔨᓴᕐᓂᐅᔫᓐᓂᒃ ᐱᒋᐊᖅᑐᓂ 1992 ᑎᑭᑦᑐᒍ 1997 (Taylor et al. 2008) ᐊᒻᒪ 2012 ᑎᑭᑦᑐᒍ 2014 (ᓵᑉᑕ 10). ᖃᐅᔨᓴᕈᓯᒥᓃᑦ ᓇᓗᓇᐃᖅᓯᑎᑦᑎᑲᓐᓂᖅᑐᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ MR ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᒻᒪ ᖃᐅᔨᓴᓂᕐᒥᑦ ᒫᓐᓇᐅᔪᒥᑦ ᖃᓄᐃᓕᖓᓂᖓ KB–ᒧ ᐊᒥᓲᓂᖏᑦ.

ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ KB–ᒧᑦ, ᐱᖁᓯᐅᑎᓪᓗᑎ ᐊᒥᓲᓂᕆᒐᔪᑦᑕᖏᑦ ᐊᑎᖅᑕᖏᑦ ᐊᕐᕌᒍᒧᑦ (COY) ᐊᒻᒪ ᐊᑎᖅᑕᓖᑦ, ᐊᒻᒪ ᑎᑎᕋᖅᓯᒪᔪᑦ ᐱᖁᓯᐅᔾᔨᓂᕐᒧᑦ (ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᑎᖅᑕᐃᑦ ᐊᑐᓂ ᐊᑕᐅᓯᕐᒧᑦ ᐊᕐᓇᓪᓗᒧᑦ MR–ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ), ᐊᔾᔨᒌᖑᐊᖅᑎᓪᓚᓴᐅᓚᐅᖅᑐᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2010–ᖏᓐᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ. ᐊᒥᓲᓂᕆᒐᔪᑦᑕᖏᑦ ᐊᑎᖅᑕᐃᑦ KB–ᒥ (COY–ᒧᑦ: 1.67 1990–ᖏᓐᓂ ᐊᒻᒪ 1.60 2010–ᖏᓐᓂᑦ) ᐊᔾᔨᐸᓗᒋᓚᐅᖅᑕᖏᑦ ᖃᐅᔨᔭᐅᔪᓄᑦ ᐊᓯᖏᓐᓄᑦ ᓇᓄᕐᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᕿᑭᖅᑕᓗᓐᓂᑦ (ᐊᑯᓐᓂᖏᓐᓂ 1.65 - 1.71). ᐊᕿᒍᑕᒫᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᒥᑭᓗᐊᖃᑦᑕᓚᐅᖅᑐᑦ ᐃᑲᔪᖅᑐᐃᓗᑎ ᐊᔾᔨᒌᖑᐊᖅᑎᓪᓚᓴᐅᓗᒋᑦ ᐊᕐ ᕿᒎᓯᒪᔪᓂᑦ. ᑭᓯᐊᓂᓕ, ᐊᕿᒍᓄᑦ ᐊᖏᓖᖑᓯᒪᔪᑦ ᖃᐅᔨᓴᕝᕕᐅᔪᑦ ᐊᖏᓂᖏᑦ (1995, 2013, ᐊᒻᒪ 2014), ᑲᑎᑦᑐᒥᑦ ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐅᖓᑕᐅᔾᔨᓚᐅᖅᑐᑦ 50–ᓂᑦ ᖃᐅᔨᔭᔅᓴᐅᓚᐅᖅᑐᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐃᓚᖏᓐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᓇᓄᕐᓂᑦ ᐊᑎᖅᑕᐅᓚᐅᖅᑐᑦ ᐊᕿᒍᒧᑦ (ᐊᑯᓐᓂᖏᓐᓂ 15% - 30%) ᓇᓗᓇᐃᖅᓯᔪᖅ ᐊᖏᔪᒥᑦ ᐊᕿᒍᑕᒫᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐊᑎᖅᑕᑖᖃᑦᑕᕐᓂᖏᑦ. ᑕᑯᓐᓇᖅᑐᒍᐃᓚᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᒻᒪ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖏᑦᑐᑦ ᐊᕿᒎᔪᓂᑦ

ᓇᓗᓇᐃᖅᑐᒍ, ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᓇᓗᓇᐃᕈᑎᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ KBᒧᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ, ᑐᑭᓯᒋᐊᕈᑎᔅᓴᖅᑕᖃᑦᑎᐊᖏᓐᓂᖓᓄᑦ ᓄᖅᑲᐅᓯᕐᓂᖃᖏᑦᑐᖅ.

ᓵᑉᑕ 13: ᖃᐅᔨᓴᐅᔪᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᓄᓇᒥ ᓂᕆᓇᓱᓐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ Kane Basin−ᒥ ᐅᐱᕐᖔᒃᑯᑦ

ᐊᑐᖅᑐᒍ ᐅᖅᓱᖃᕐᓂᖓᓄᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ (ᑕᑯᓗᒍᑦᑕᐅᖅ ᓵᑉᑕ 7) ᐊᔾᔨᒌᓐᖑᐊᖅᑎᓚᐅᖅᑕᕗᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ ᓇᓄᐃᑦ KB−ᒥᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ. ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐱᐅᓂᖅᓴᐅᓚᐅᖅᑐᑦ KB-ᒥ ᐃᓐᓇᕐᓂᑦ ᐊᕐᓇᓂᑦ ᐊᑎᖅᑕᓕᓐᓂᑦ 2010−ᖏᓐᓂᑦ ᐊᔾᔨᒋᐸᔭᑦᑕᖏᑦ 1990−ᖏᓐᓂᑦ. ᑭᓯᐊᓂ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓂᖏᑦ ᑭᓪᓕᖅᑎᑕᐅᓚᐅᖅᑐᑦ. ᑕᑯᓐᓇᖅᑐᒍ, ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐃᓐᓇᖏᓐᓂᑦ ᓇᓄᕐᓂᑦ ᐊᖑᑎᓂᑦ, ᐃᓐᓇᓂᑦ ᐊᕐᓇᓂᑦ Coy−ᖃᖅᑐᑦ, ᐊᒻᒪ ᐃᓄᑑᔪᑦ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᐊᔾᔨᒌᐲᐸᕋᓗᓚᐅᖅᑐᑦ ᑕᕝᕙᓂ ᖃᐅᔨᓴᕐᓂᐅᔫᓐᓂᒃ.

ᖃᐅᔨᓴᔅᓴᐅᔪᑦ ᐱᐅᓯᕚᓪᓕᖅᑐᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ 2010−ᖏᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᖅ ᐃᒻᒥᒃᑯᑦ ᐊᔾᔨᒌᖏᓐᓂᕆᖃᑦᑕᖅᑕᖏᑦ ᐅᕝᕙᓘᓐᓂᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐊᑯᓂᐅᔪᒧᑦ ᐊᓯᔾᔨᖅᑐᒧᑦᓯᑯᖓᓂ KB−ᒥ, ᐊᔾᔨᖃᐸᓗᓕᖅᑐᖅ ᐊᕐᕋᒍ ᐃᓗᐊᓂ ᓯᑯᖃᕈᓐᓃᖃᑦᑕᕐᓂᖓ. ᓇᑦᑏᑦ ᐊᒻᒪ ᓇᓄᐃᑦ ᐊᔾᔨᒌᐸᓗᓐᓂᑦ ᓇᒧᖓᖃᑦᑕᓕᕐᓗᑎᒃ Kane Basin. ᐊᒥᓱᓕᒫᓪᓗᐊᑕᖅᑐᑦ ᑕᒪᒃᑭᓐᓂ ᓇᑦᑏᑦ ᐊᒻᒪ ᓇᓄᐃᑦ ᐅᐊᓐᓴᖓᑕ ᑲᓇᓐᓇᖓᓂ Kane Basin, ᖃᓂᒋᔭᖓᓂ Humbolt Glacier (ᖃᓇᓐᓇᖓᓂ KB), ᓇᓗᓇᐃᖅᓯᔪᖅ ᑕᒪᓐᓇ ᖁᕝᕙᓯᑦᑐᒥᑦ ᐆᒪᔪᓕᐅᑦᐸᑦᑐᑦ ᐊᒻᒪ ᐱᒻᒪᕆᐅᓗᓂ ᓇᓄᕐᓄᑦ ᓇᔪᖅᑕᐅᓗᓂ ᓇᓄᕐᓄᑦ.

ᐊᒥᓲᔪᑦ ᐃᒪᖏᑦ KBᐟᒥ ᓇᓄᐃᑦ ᖃᐅᔨᓴᐅᓚᐅᖅᑐᑦ ᒥᖅᑯᐃᔭᖃᑦᑕᖅᓯᒪᒻᒪᕆᑦᑐᑎ ᐊᒻᒪ ᐅᕙᓂᖏᑦ ᐲᔭᕐᓂᖏᑦ ᐃᓯᒪᕐᑕ ᐊᑖᒍᑦ. ᑖᒃᑯᓄᖓ ᐱᔾᔪᑕᐅᔪᑦ ᖃᐅᔨᒪᔭᐅᖏᑦᑐᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ. ᐃᓄᓐᓄᑦ ᖃᐅᔨᒪᔭᐅᔪᓄᑦ ᐅᖃᐅᓯᕐᔪᐊᖅ ᑕᒪᓐᓇ ᑕᐃᒪᐃᓐᓂᕆᔭᖓ ᐊᕐᖏᖃᑦᑕᖅᓯᒪᔪᖅ ᐱᓯᑦᑐᑎ ᐊᒻᒪ ᐊᓪᓚᑦᑐᑎ ᑎᓯᔪᒥᑦ, ᓂᓚᒥᑦ, ᒪᓂᑦᑐᒥᑦ ᐊᐳᒥᑦ ᐅᐱᕐᖔᒃᑯᑦ ᓯᑯᒃᑯᑦ ᐃᒪᒥᖅᑲᓯᐅᑎᔭᖓ ᓅᖅᑲᑕᓂᖅᓯᐅᔪᓐᓂᖅᓴᓄᑦ ᓄᓕᐊᖅᑎᓐᓄᑦ ᐊᒻᒪ ᓂᕆᓇᓱᑦᑎᓐᓄᑦ.

ᓵᑉᑕ 14: ᐅᖃᐅᓯᔅᓴᐃᑦ ᐊᒻᒪ ᐅᖃᐅᔾᔨᒋᐊᕈᑎᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᓂᑦ

ᖃᐅᔨᓴᐃᓂᖅ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ

BB ᐊᒻᒪ KB−ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᓯᕗᓪᓕᖅᐹᒥ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᓂ ᐊᑐᓕᖅᑎᑕᐅᓯᒪᔪᖅ ᐊᒥᓲᓂᓕᒫᖏᓐᓄᑦ ᓴᔅᓴᐅᓴᑦᑕᐅᓗᑎ ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᓇᒦᓐᓂᖏᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ. ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᖁᑭᐅᑦᑕᐅᔭᓄ ᑲᔪᓯᑦᑎᐊᕐᓂᖃᓚᐅᖅᑐᖅ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᓈᒻᒪᑦᑐᓂ ᓇᓗᓇᐃᖅᑕᑲᐅᓂᖏᓐᓄᑦ ᑕᒪᒃᑭᖏᓐᓄ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓ ᐊᒻᒪ Kane Basin. ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐱᖃᑦᑕᕐᓂᖅ ᓱᑲᑦᑑᔪᖅ, ᐊᐅᓚᓂᖃᑦᑎᐊᖅᑐᓂ, ᐊᒻᒪ ᐸᕝᕙᓴᐃᖏᓐᓂᖅᓴᐅᓗᓂ ᓴᓂᐊᓂ ᐱᔭᐅᔪᓂᑦ, ᓇᓄᐃᑦ ᐊᐅᓚᒍᓐᓇᐃᓕᑎᑕᐅᕙᖏᒪᑕ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᐊᑑᑎᖃᖅᑎᑦᑎᔾᔪᓇᖅᑐᖅ, ᐸᕝᕙᓴᐃᖏᓐᓂᖅᓴᐅᓗᓂ ᓴᓇᕿᑎ.

ᑭᓯᐊᓂᓕ ᐊᓯᖏᑦ ᖃᓄᐃᓕᓂᖃᖔᖅᑐᑦ ᑖᔅᓯᒪᖓ ᐊᑐᖅᑕᐅᔪᒧᑦ ᐃᓚᖏᓐᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᑐᐃᓐᓇᐅᑎᑦᑎᖏᑦᑐᑦ ᓇᓗᓇᐃᖅᓯᑎᐊᕐᓂᖅᓴᐅᐸᐅᖅᑐᓂᑦ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ (ᑭᒍᑎᖏᑦ ᐊᕐᕌᒍᖏᑦ ᓇᓄᕐᓂᑦ ᓇᓗᓇᐃᖅᓯᑎᐊᑲᓐᓂᐊᖅᑎᑦᑎᔪᖅ ᐊᖅᑮᒍᑎᖃᑎᖅᒌᑦᑐᓂᑦ,ᑐᑭᓯᒋᐊᕈᑎᔅᓴᖅᑕᖃᑦᑎᐊᓂᖅᓴᐅᓗᓂ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ, ᐊᖏᔪᓂᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᓄᐊᑦᑎᓚᐅᖏᑦᑐᑦ ᐊᐅᑎᒍᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐱᖃᑦᑕᑎᓪᓗᒋᑦ, ᖃᑦᑏᓐᓇᐅᓂᖅᓴᐃᑦ ᖁᖓᓯᕈᑦᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᒃᑯᓂᑦ). ᐃᓚᒋᐊᕐᓗᒍ, ᐃᓚᖏᑦ ᐆᒪᔪᑦ ᐱᔭᐅᔪᑦ ᐱᒻᒪᕿᐅᓚᐅᖅᑐᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᑦᑎᑎᓪᓗᐊᑕᖅᑐᑎᑦ ᑖᔅᓯᒪᖓ ᖃᐅᔨᓴᓂᕐᒧᑦ. ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᑐᑭᓯᒋᐊᕈᑎ ᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ ᓇᓄᕐᓂᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐊᑐᖅᑕᐅᓚᐅᖅᑐᑦ

ᓇᓄᓇᐃᓯᐃᑎᓪᓗᒋᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, ᑕᒪᒃᑮᓐᓂ ᐃᓱᒪᒋᔭᐅᔪᑦ ᒪᒍᖅᑕᕐᕆᓐᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᑭᓱᐊᓂ ᖃᐅᔨᒃᑲᐃᓪᓗᓂ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ.

ᐊᕐᕆᔪᑦ ᐱᓕᕆᐊᑦ ᐊᔾᔨᙱᑎᑐᑦ BB−ᒥ ᐊᒻᒪ KB−ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐊᑭᑐᔪᑦ ᐊᒻᒪ ᐃᖅᑲᓇᐃᓯᐅᑎᓄᑦ ᐊᑦᓯᕈᓐᓇᖅᑐᓂ ᐊᒻᒪ ᑭᓪᓕᖅᖢᑎᑕᐅᓪᓗᑎ ᓴᖅᑭᖃᑦᑕᕐᓂᕐᕆᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒐᕙᒪᖃᕐᕕᐅᔪᓂᑦ ᐱᓕᕆᐊᕆᔫᓐᓇᖅᑕᕐᕆᑦ. ᐊᒥᓲᓂᕐᕆᓐᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᕐᕆᓐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᖃᑦᑕᖅᖃᑕ ᑕᑯᓐᓇᖅᑕᐅᓗᑎ 10-15 ᐊᕐᕌᒍᓂᑕᒫᑦ BB-ᒥ ᐊᒻᒪ KB−ᒥᑦ, ᑕᐃᒪ ᐊᑐᓕᕐᑯᔭᐅᕗᖅ ᒪᑭᒪᑎᓪᓗᒋᑉ ᐊᒻᒪ ᐊᒥᓱᕈᖅᑎᒋᐊᕐᓗᒋᑦ ᓇᓄᓇᐃᒃᑯᑕᐃᑦ ᐊᒥᓲᓂᕐᕆᓐᓂ ᐊᑕᐅᓯᐊᕐᓗᑎ ᐊᕐᕌᒍᒥ ᖃᐅᔨᓴᖃᒃᑕᕐᓗᑎ 5-7 ᐊᕐᕌᒍᓂᑕᒫᑦ. ᑕᒪᓐᓇ ᓇᓄᓇᐃᖅᓯᓂᐊᖅᑐᖅ ᓇᓄᓇᐃᒃᑯᑕ ᐊᐃᕐᕆᑎᒨᖅᑐᓂ ᓇᓄᓇᐃᒃᑯᑖᓪᓚᑦ ᓇᓄᐃᑦ ᐊᒥᓲᓂᕐᕆᓐᓂᑦ ᐅᑎᖅᑎᑕᐅᓗᑎᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓄᑖᒥ ᐊᒥᓲᓂᕐᕆᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᐅᓚᓂᖃᑦᑎᐊᓂᖅᓴᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐃᓚᐅᑎᑦᑎᕐᕆᓐᓂᖅᓴᐅᕐᕆᑦᑐᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᐱᓪᓗᓈᐅᔪᓂᑦ ᖁᕝᕙᓯᓐᓂᓇᕆᐊᖃᖅᑕᓴᕐᕆᑦ.

ᑭᓱᐊᓂᓕ, ᖃᐅᔨᒪᔭᐅᔭᓇᐊᖃᕐᒥᔪᖅ ᓇᓄᓇᐃᒃᑯᓯᖅᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐱᖕᖓᓱᓄᑦ ᐊᕐᕌᒍᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᔭᓇᐊᖏᓪᓚᑦ ᑐᓐᖓᕕᖃᖅᑐᓂᑦ ᒥᒃᓴᐅᓴᑦᑕᐅᓂᕐᕆᓐᓄᑦ ᓈᒻᒪᒋᔭᐅᔪᒥᑦ ᓇᓄᓴᐃᔭᖅᓯᒪᓂᕐᕆᓐᓄᑦ ᖃᐅᔨᓴᓂᒧᑦ. ᑕᑯᒃᓴᐅᑎᑕᐅᓯᒪᔪᑦ ᒫᓐᓇᐅᔪᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᓂ *post hoc* ᐊᐅᕐᕆᑎᒨᖅᑐᓂ ᖃᐅᔨᓴᐃᓂᖅ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᖃᓄᐃᓐᓂᓇᔭᕐᕆᑦ ᐊᑐᐃᓐᓇᐅᔫᓐᓇᖅᑐᑦ ᑕᓪᓕᒪᓂ ᐊᕐᕌᒍᓂ ᐱᒋᐊᖅᑎᑕᐅᓯᒪᓕᖅᑎᓪᓗᒍ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ. ᐃᓚᒋᐊᕐᓗᒍ, ᐊᐃᕐᕆᑎᒨᖅᑐᓂ ᓇᓄᓇᐃᒃᑯᓯᖅᓯᓂᖅ (ᐊᔾᔨᐸᓗᙱ ᐆᒪᔪᓂ ᓇᓄᓇᐃᒃᑯᓯᑦᓯᒥᖅ) ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᕐᕆᔪᒥᑦ ᓇᒧᙱᖃᑦᑕᖅᑐᑎᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᓯᔾᔨᖃᑦᑕᖅᑐᓂᑦ ᐊᕙᑎᙱᓂ (ᓲᕐᓗ, ᐊᓯᔾᔨᖃᑦᑕᖅᑐᑦ ᓯᑯᙱ) ᑕᓐᓂᕈᑎᐅᑉ ᐃᒪᙱᑎᑐᑦ ᐊᔭᐅᖅᑕᐅᔫᓐᓇᖅᑐᖅ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᕐᕆᓐᓄᑦ ᐱᔫᓐᓇᕐᓗᑎ ᑭᒡᒐᖅᑐᐃᓗᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ (ᓲᕐᓗ, ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᖅ, ᐃᓚᕐᕆᓐᓄᑦ ᐊᒥᓲᓂᕐᕆᓐᓄᑦ ᐊᑐᐃᓐᓇᐅᕐᕆᑦᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᓗᑎ ᐃᒪᐃᓐᓇᓲᑦᑐᑦ ᐅᙱᓯᑦᑐᒥ ᓯᑯᒥ).

ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᕐᓗᑎ ᓇᓄᐃᑦ ᐊᒥᓲᓂᕐᕆᑦ ᐊᕐᕆᔪᒥᑦ ᐃᓱᒪᒋᔭᐅᔫᓐᓇᕐᒥᔪᖅ ᖃᙱᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᖃᓯᐅᔾᔨᓗᓂ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔫᒃᑯᑦ ᒥᒃᓴᐅᓴᑦᑕᐅᓂᕐᕆᓐᓄᑦ, ᓇᓄᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᐊᐆᑎᖃᖅᑐᑎᑦ ᐊᓯᕐᕆᓐᓄᑦ (Aars et al. 2009). ᐅᐱᕐᙰᒃᑯᑦ ᓯᓚᑦᑎᐊᕙᐅᖃᑦᑕᕈᓂᖃᙱᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖃᕈᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᖅᑕᐅ)ᔫᓐᓇᖅᑐᖅ ᐊᕐᕌᒍᒥ ᐊᑕᐅᓯᕐᒥ. ᑭᓱᐊᓂᓕ, ᑕᑯᓐᓇᖅᑐᒍ ᓇᓄᓇᐃᒃᑯᓯᖅᑕᐅᔪᓄᑦ ᖃᙱᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᓯᓚᑲᒻᒪᐅᖃᑦᑐᐃᓐᓇᖅᐸᓕᑦ ᐊᑕᐅᓯᕐᒥ ᐊᕐᕌᒍᒥ.

SWG ᐱᓕᕆᐊᖅᑖᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᐱᔪᒪᓂᖃᖅᑐᑦ ᑲᑎᑦᑐᒋᑦ ᓇᓄᑦᑕᐅᔫᓐᓇᖅᑐᓂᑦ (TAH) ᖁᕝᕙᓯᓐᓂᕐᕆᑦ ᑕᒪᒃᑮᓄ ᓴᓂᕈᑎᐅᑉ ᐃᒪᙱᓄ ᐊᒻᒪ Kane Basin ᑭᓱᐊᓂ ᐱᑕᖃᖅᑎᑕᐅᓚᐅᕐᕆᑦᑐᑦ ᖃᓄᐃᑦᑐᑐᐃᓐᓇᕐᒧᑦ ᓇᓄHᓇᐃᖅᓯᒪᔪᒧᑦ ᑐᑭᒧᐊᕈᑎᒥᑦ ᐊᐅᓚᑕᐅᓂᕐᕆᓐᓄᑦ ᑐᕌᖅᑕᐅᔪᓄᑦ ᑖᒃᑯᓄᙵ ᐊᒥᓲᓂᕐᕆᓐᓄᑦ JC−ᑯᓐᓄᑦ, ᐊᒥᓱᐊᖅᑎᒐᓚᑦᑐᑎᑦ ᑐᔅᓯᕋᖃᑦᑕᑦᖅᓯᒪᒐᓗᐊᖅᑐᑎ ᐊᕐᕌᒍᒐᒪᐅᔪᓂᑦ. SWG ᑐᔅᓯᕋᓚᐅᖅᑐᑦ ᑖᒃᑯᐊ JC ᐱᑕᖃᖅᑎᑦᑎᓗᑎ:

1) ᐅᓂᒃᑳᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᕐᒍᓂᐊᖅᑐᑦ ᐊᑐᓂ ᐊᒥᓲᓂᕐᕆᓐᓄᑦ,

2) ᑐᑭᓯᒋᐊᕈᑏᑦ ᓂᕿᐅᒋᔭᐅᔪᑦ ᓴᖅᑭᖃᑦᑕᓂᐊᕐᓂᖓ ᐊᒻᒪ ᐊᕐᕆᓂᖓ ᓯᕗᓂᔅᓴᒥ ᖃᐅᔨᓴᓂᕐᒧᑦ, ᐊᒻᒪ

3) ᐅᓂᒃᑳᑦ ᐊᑦᑕᓇᕈᑎᖃᕈᓐᓴᕐᓂᖓᓄ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐊᑦᑐᖅᑕᐅᔪᓄᑦ ᐃᓄᓐᓄᑦ ᐹᖅᑕᐅᔪᓄᑦ.

ᐱᔭᐅᕐᕆᓐᓂᕐᕆᓐᓄᑦ ᐱᔭᐅᔭᓇᐊᖏᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᑐᓐᖓᕕᒋᓂᐊᖅᑕᕐᕆᑦ ᐊᑐᓕᕐᑯᔭᐅᔪᑦ TAH JC−ᑯᓐᓂᑦ, SWG−ᑯᑦ ᒫᓐᓇᐅᔪᒥᑦ ᑖᒃᑯᓂᖓ ᐊᑐᓕᕐᑯᔭᓕᐅᕈᓐᓇᕐᕆᑦᑐᑦ. ᓇᓄᓇᐃᖅᓯᒪᔪᑦ ᐱᓕᕆᐊᕐᒍᕆᐊᖏᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᒻᒪ ᐱᐅᖅᓱᐊᓂᕐᒧᑦ BB−ᒧᑦ ᐊᒻᒪ KB−ᒧᑦ ᓇᓄᕐᕆᓐᓄᑦ ᐱᓪᓗᓈᐅᔪᖅ ᑖᒃᑯᐊ ᐊᔭᐅᕈᑎᖃᓪᓗᐊᓂᐊᖅᑐᑦ ᐊᑐᓕᕐᑯᔭᐅᔪᓂᑦ TAH, ᐃᓱᒪᓕᐅᖃᑦᑕᖅᑐᓂᑦ ᐊᑐᐃᓐᓇᖃᓂᐊᖅᑐᑦ ᑐᑭᓯᒋᐊᕈᑎᐅᕆᐊᖃᖅᑐᓂᑦ.

ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᕌᒐᐃᑦ ᐃᓱᒪᓕᐅᕈᑎᐅᖏᑦᑐᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᐃᖅᑲᓴᐃᔭᖅᑎᓄᑦ ᑭᓯᐅᓂᓕ ᑕᐃᒃᑯᓄᖓ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔪᓐᓇᐅᑎᓕᓐᓄᑦ ᓇᓄᕐᓄᑦ ᐊᒻᒪ, ᐃᓱᒪᒋᔭᔅᓴᐅᔪᖅ, ᐃᓱᒪᓕᐅᕈᑕᐅᔪᑦᒃ ᐃᓱᒪᒋ.ᖃᒪᐅᖅᑐᑎᑦ ᓂᕿᔅᓴᓄᓃᖏᑦᑐᖅ ᓇᓄᑦᑕᐅᔪᑦ ᑭᓯᐊᓂ ᐊᓯᖏᑦ ᐃᓄᓐᓄᑦ ᐲᖅᑕᐅᔪᑦ ᓃᕐᓗ, ᐃᓄᐃᑦ−ᓇᓄᐃᑦ ᖃᓄᐃᓘᖃᑦᑕᐅᑎᓂᖏᑦ) ᐊᒻᒪ ᐅᖃᖃᑎᖃᒪᐅᖅᑎᓪᓗᒋᑦ ᑎᒍᒥᐊᖅᑎᐅᖃᑕᐅᔪᓂᑦ.

JC ᑐᓂᓯᑐᐊᕈᓂ ᑐᔅᓯᕌᖑᔪᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ, SWG ᐊᑐᓕᖁᔨᔪᑦ ᐅᖃᐅᔾᔨᒋᐊᕈᑎᐅᕙᑖᓚᐅᖅᑐᖅ TAH−ᒥ ᑐᓐᖓᓂᖃᕐᓗᓂ ᐊᒥᓲᓂᕆᓐᖑᐊᖅᑎᑕᐅᔪᓄᑦ ᐊᑐᖅᑐᓗᑎ ᐱᖃᓯᐅᔾᔨᔪᓐᓇᖅᑐᓂᑦ ᐊᓯᔾ"ᒍᓐᓇᓂᕐᒧᑦ ᐊᓯᓅᖅᓯᔪᓐᓇᓂᕐᒧᑦ (ᓃᕐᓗ, ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ) ᐊᒻᒪ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᐱᐅᓂᖏᑦ ᐃᒪᒋᓪᓗᓂᐅᒃ ᐊᑕᖏᖅᑐᒥ ᐱᕚᓪᓕᐊᓂᖓᓂᑦ ᑲᐅᔨᓴᐃᓪᓗᑎ ᐊᑦᑕᓇᖅᑐᒦᒍᓐᓇᕐᓂᖓᓂ ᐊᔾᔨᒌᖏᑦᑐᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ ᐊᐅᓚᑕᐅᔪᓂᑦ ᖃᓄᐃᓐᓂᐅᔪᓐᓚᖅᑐᓂᑦ. SWG ᐊᑐᓕᖁᔨᔪᑦ ᐊᑐᖅᑕᐅᓗᓂ ᓇᓗᓇᐃᖅᓯᒪᔪᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᑐᐊᒐᖅ, ᓴᖅᑭᖅᑎᑕᐅᓯᒪᔪᖅ Regehr et al. (2015), ᑎᒥᒧᑦ−ᑐᓐᖓᓂᖃᖅᑐᖅ ᐊᐅᓚᑕᐅᓂᖓᓄᑦ ᐋᖅᑭᔅᓯᒪᓂᖓ ᑲᓯᕐᓂᖃᖅᑎᑦᑎᔪᖅ ᐊᑐᖅᑕᐅᔪᒧᑦ ᐊᒥᓲᓂᕆᓐᖑᐊᖅᑕᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᖅ ᐊᒻᒪ ᐊᑐᖅᑕᐅᔪᓐᓇᕐᒥᔪᖅ ᒥᔅᓴᐅᓴᑦᑕᐅᓗᑎ ᐊᒥᓲᓕᕆᔪᓐᓇᖅᑕᖏᑦ ᒪᑭᒪᑎᑦᑎᓂᕐᒧᑦ ᖁ−ᕗᓯᓐᓂᖏᑦ ᐃᓄᓐᓄᑦ ᐲᖅᑕᐅᔪᑦ, ᐊᑐᖅᑲᑕᐅᔪᓐᓇᕐᒥᔪᖅᑕᐅᖅ ᐋᑦᑭᔅᓯᓗᑎ ᐊᑐᓕᖁᔮᐅᔪᓂ ᒪᑭᒪᑎᑕᔅᓴᐅᔪᓂᑦ ᓇᓄᑦᑕᐅᖃᑦᑕᖅᑐᓂᑦ ᖁᕝᕗᓯᓐᓂᖏᓐᓄᑦ, Regehr et al. (2015) ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᖅ ᐊᑦᑎᓂᖅᓴᐅᓚᔪᑦᑐᖅ ᑭᓪᓕᒋᔪᓐᓇᖅᑕᖓ ᒪᑭᒪᑎᑦᑎᓂᕐᒧᑦ ᖁᕝᕗᓯᓐᓂᖏᑦ ᑐᓐᖓᓂᖃᕐᓂᖓᓄᑦ ᐊᐅᓚᑕᐅᓂᖏᓐᓄᑦ ᐱᑎᕆᐊᓄᑦ, ᓇᓗᓇᐃᖅᓯᒪᓕᕆᓐᓂᖓ ᐊᒻᒪ ᖃᖓᒃᑰᖃᑦᑕᕐᓂᖏᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᐊᒻᒪ ᐊᑦᑕᓇᕈᓇᖏᓐᓂᕐᒧᑦ.

ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᑦ

ᑕᒪᒃᑭ ᖃᐅᔨᓴᖅᑎᓄᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ (ᐅᕝᕙ ᐊᒻᒪᓗᑦᑕᐅᖅ ᓯᕗᓂᐊᒍᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ) ᐊᒻᒪ ᐃᓄᓐᓄᑦ ᖃᐅᔨᒪᔭᐅᔪᑦ (Dowsley et al. 2005, Dowsley and Wenzel 2008 and Born et al. 2011) ᑎᑎᕋᖅᓯᓯᒪᔪᑦ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ (ᓃᕐᓗ, ᓯᑯᖃᕈᓐᓃᕐᓂᖓ) ᓇᓄᕐᓄᑦ BB−ᒥᑦ (KB−ᒥᐅᔪᔅᓴᐅᒻᒥᔪᖅᑕᐅᖅ). ᓇᔪᖅᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᒻᒪᕆᐅᓂᐊᖅᑐᖅ ᑲᔪᓯᑎᓪᓗᒍ ᖃᓄᐃᓐᓂᕆᐊᓯᒋᔭᐅᔪᑦ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᐅᑭᐅᑕᖅᑐᒥ. ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓇᔪᖅᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin−ᒥ 2009−ᒥᓂᑦ, ᑐᓐᖓᓂᖃᓗᐊᖅᑐᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᕐᓇᓐᓄᓂᑦ ᓇᓄᕐᓂᑦ, ᐱᐅᔪᓂᑦ ᓄᑖᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᖅ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂ ᓇᔪᖅᑕᐅᔪᓂᑦ. ᑐᑭᒧᐊᖅᑐᖅ ᖃᓄᐃᓕᖓᓂᕆᔭᖓᓂ BB−ᒥ ᐊᒻᒪ KB−ᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ

SWG ᓇᓗᓇᐃᖅᓯᔪᑦ ᓯᕗᓂᔅᓴᒥ ᐆᒪᔪᓂᑦ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᖅ BB−ᒥ ᐊᒻᒪ KB−ᒥᑦ (ᖁᖓᓯᕈᖅᓯᓂᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᑦ ᓇᒦᓐᖏᓐᓄᑦ ᐊᕐᓇᓪᓗᐃᑦ ᐊᕐᕌᒍᓗᒪᓐᓄᑦ) ᐱᒻᒪᕿᐅᓂᐊᖅᑐᖅ ᐅᖓᕐᕗᓇᐊᖅᑕᐅᓗᑎᑦ ᐊᑯᓂᐅᓂᕆᔭᖏᑦ ᐊᒻᒪ ᖃᐅᔨᒃᑲᐃᔪᖅ ᐊᐅᓚᑦᑎᔨᓂᑦ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᕐᕌᒍᓗᒪᓚᐅᓂᐊᖅᑐᓂᑦ. ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᖅᓯᒪᓂᖓᓄᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᑕᕝᕙᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ, ᐊᒻᒪ ᓇᓗᓇᖏᑦᑐᑦ ᖃᓄᐃᓕᓂᕆᔭᖏᑦ ᓇᓄᐃᑦ ᑕᒪᒃᑭᓐᓂ, ᑲᔪᓯᑎᑕᐅᓂᖓ ᖁᒻᒧᐊᑦᖅᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᑐᑭᓯᑎᑦᑎᑎᐊᓂᖅᓴᐅᓂᐊᖅᑐᖅ ᐊᑦᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᓯᕗᓂᔅᓴᒥ ᐊᒻᒪ ᐆᒪᑎᓐᓇᕆᓐᓂᖏᓐᓂᖏᓐᓂᖅᑐᑦ ᐊᓯᔾᔨᖅᑐᑦ. ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᖃᐅᔨᓴᕐᓂᖃᓇᐊᖅᑲᖅᑐᑦ ᖁᓕᓂᑕᒫᑦ ᐊᕐᕌᒍᓂᑦ ᑐᖕᓕᓂᓪᓗᓐᓃᑦ, ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓂᖏᑦ ᐊᔾᔮᐸᓗᒋᔭᖏᑦ ᓄᐊᑕᐅᔪᓂᑦ 1991-1997 ᐊᒻᒪ 2009-2013 ᖃᐅᔨᓴᕐᓂᐅᔪᑦ (ᖃᓂᒋᔭᐸᓗᖏᓐᓂ 40 ᐊᕐᓇᓪᓗᐃᑦ ᖃᐅᔨᔭᐅᔪᑦ ᐊᕐᕌᒍᓗᒪᐅᔪᒪᓐᓄᑦ). ᐊᑎᓂᖅᓴᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓂᖏᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᔾᑲᓇᑎᑦᑎᑲᓐᓂᓂᐊᖅᑐᖅ ᐃᒻᒥᒎᖅᑐᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᐊᑐᐊᒪᔅᓴᖅᑕᖅᑎᐊᖏᓐᓂᖕᒪᓄᑦ RSFs (ᑕᑯᔭᐅᓯᒪᔪᑦ KB−ᒥᑦ).

ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓯᕗᓪᓕᖅᐹᒥ ᑎᑎᕋᖅᑕᓚᐅᖅᑐᑦ ᑖᒃᑯᐊ 18% ᐊᕐᓇᓪᓗᓂᑦ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓂᑦ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ Melville Bay−ᒦᖏᓐᓇᓚᐅᖅᑐᑦ

ᐊᑕᐅᓯᕐᒥ ᐊᐅᔭᒥᑦ ᐊᒻᒪ ᐃᓚᖏᑎᒍᑦ ᐊᕐᕿᒍᒧᑦ. ᑕᒪᓐᓇ ᒪᓕᓐᓂᖅᑐᖅ ᖃᐅᔨᔭᐅᔪᓂᑦ ᐊᖑᓇᓱᑦᑎᓪᓚᕆᓐᓄᑦ Melville Bay−ᒥ, ᖃᐅᔨᓯᒪᔪᑦ ᓇᓄᖅᑕᖃᓕᖅᐸᓪᓕᐊᓂᖓᓄᑦ Melville Bay, ᐊᒻᒪᓗᑦᑕᐅᖅ ᑎᑎᕋᖅᑕᐅᔪᑦ ᐅᑯᓄᖓ Taylor et al. (2005) ᐊᒻᒪ Born et al. (2011). ᐊᑐᓕᖁᔭᐅᒃᑲᓂᖅᑐᑦ ᓯᕗᓂᔅᓴᒥ ᐃᖅᑲᓇᐃᔮᖑᓂᐊᖅᑐᓂᑦ ᖃᐅᔨᓴᖅᓱᑎ ᖃᑦᑎᑦ ᐃᒪᖏᑦ BB−ᓇᓄᖏᑦ ᐊᑐᖃᑦᑕᖅᖃᑦ Melville Bay ᐊᕐᕿᒍᓕᒫᒥᑦ ᐊᒻᒪ ᒪᓕᓪᓗᒋᑦ ᑖᒃᑯᐊ ᑲᑎᙵᔪᑦ ᓇᓄᐃᑦ ᐊᕐᕿᒍᓕᒫᒥ Melville Bay−ᒦᑉᐸᑦᑐᑦ ᐆᒪᔪᖃᕐᕕᒻᒥᑦ. ᑖᓐᓇ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᖅ ᒫᓐᓇᐅᔪᒥ ᐊᐅᖏᑎᔫᖅᓯᒪᔪᓂᑦ ᓇᓗᓇᐃᒃᑯᑕᓂ ᑖᔅᓱᒧᙵ ᖃᐅᔨᓴᓂᕐᒥᑦ, ᐃᓚᒍᑕᖃᓪᓗᑎ ᐊᑕᐅᓯᕐ ᒪᕐᕿᓂᓪᓘᓐᓃᑦ ᐊᐅᖏᑎᔫᖅᑐᓂᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ Melville Bay−ᒥ ᐊᑕᐅᑎᒃᑯᑦ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔅᓴᓂᕐᒥᑦ.

SWG ᐊᑐᓕᖁᔨᓚᐅᕐᒥᔪᑦ ᑲᔪᓯᑎᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᓂᖅ ᐊᓯᔾᔨᖅᑐᓂ ᓇᔪᖅᑕᐅᔪᓂᑦ BB−ᒥᑦ ᐊᒻᒪ KB−ᒥᑦ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓗᑎ ᓯᑯᖓ (ᐊᖅᑯᑎᖏᓐᓄ ᑐᑭᓯᒋᐊᕈᑏᑦ, MODIS, ᐅᕝᕙᓘᓐᓃᑦ Radar ᐊᔾᔨᙳᐊᖏᑦ, ᐱᔭᕆᐊᑎᑕᐅᓂᖏᑦ ᓇᔪᖅᑕᐅᔪᓄᑦ ᐊᑐᖅᑕᐅᔪᑦ (ᒪᓕᑦᑐᒍ Durner et al. 2009, Laidre et al. 2015 ᐊᒻᒪ ᓅᑉᑕᖏᑦ 4 ᐊᒻᒪ 9), ᑲᔪᓯᑎᑕᐅᔪᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐊᑦᑐᐊᓂᖅᑐᑦ ᓇᔪᑦᖅᑕᐅᔪᓄᑦ ᐊᓯᔾᔨᖅᑐᓄᑦ ᓇᓗᓇᐃᖅᓯᒪᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ, ᐊᒻᒪ ᑲᔪᓯᑎᑕᐅᖏᓐᓇᕐᓗᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ TEK ᐊᒻᒪ LEK ᑲᒪᒃᑭᓐᓂ ᑲᓇᑕᒥ ᐊᒻᒪ ᑲᓇᑕᒥ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᓂᐊᖅᑐᑦ ᖃᓄᖅ ᐊᓯᔾᔨᒻᑐᓂᑦ ᓯᑯᓂᑦ ᐊᑦᑐᐃᓂᖃᕐᒪᖔᑕ ᓇᓄᕐᓂᑦ ᐊᖑᓇᓱᑦᑕᐅᕙᑦᑐᓂᑦ (ᓲᕐᓗ, ᓇᓐᓄᓚᓯᑐᐃᓐᓇᕐᓂᖅ), ᐊᑕᖏᖅᑐᒍ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᑐᑦ, ᐊᒻᒪ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᓇᓐᓄᑦᑕᐅᔪᑦ.

ᐃᓱᒪᒋᔭᐅᔪᑦ

ᔮᓐᓄᐊᕆᒥ 2010, JC ᐱᓕᕆᐊᖃᖅᑎᑦᑎᓚᐅᖅᑐᑦ SWG−ᑯᓐᓂᑦ ᐊᑐᖅᑐᑎᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐃᒪᐃᓐᓂᐊᖅᑐᑎ: (1) ᓴᖅᑭᑎᑦᑎᓗᑎ ᑲᑎᑦᑐᒋᑦ ᓈᓄᑦᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᖁᕝᕙᓯᓐᓂᖏᑦ Kane Basin−ᒧᑦ ᐊᒻᒪ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓄᑦ ᓇᓄᖏᓐᓄᑦ ᐊᒻᒪ JC ᑐᓂᓯᔭᐅᓗᑎ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐅᓂᒃᑳᓂᑦ ᐊᑐᓕᖁᔭᖏᓐᓂᑦ, (2) ᖃᐅᔨᓴᖅᑎᓃᖔᖅᑐᓂ ᐅᖃᐅᔾᔨᒋᐊᕐᓂᖅ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖓᓄ ᐊᑦᑐᖅᑕᐅᓂᖏᑦ ᓇᔪᖅᑕᖏᑕ ᐊᓯᔾᔨᕐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ.

ᕿᒥᕐᕈᓚᐅᖅᑐᒋᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ, SWG ᓄᖅᑲᐅᓯᓚᐅᖅᑐᑦ (SWG 2011) ᒥᔅᐅᓴᑦᑕᐅᕋᑖᖅᓯᒪᔪᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓄᑦ (BB) ᐊᒻᒪ Kane Basin (KB) ᐊᒻᒪ ᐱᒪᕆᐅᔪᑦ ᑭᓪᓕᕆᔭᖏᑦ ᓴᖅᑭᖅᑐᑦ ᑲᑐᑎᔭᐅᔪᓂᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ (1993-1997) BB−ᒥ ᐊᒻᒪ KB−ᒥ ᐅᓪᓗᒥ ᒨᖓᓚᐅᖏᑦᑐᑦ ᐊᒻᒪ ᓇᓗᓇᕈᑎᑕᖅᑐᑎ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᓯᔾᔨᓪᓗᐊᑲᓪᓚᔾᔨᒪᓂᖓᓄᑦ ᐅᓂᒃᑲᓕᐊᖑᔪᓂᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐊᒻᒪ ᓯᑯᖓ ᖃᓄᐃᓕᖓᓂᖓ ᑖᒃᑯᐊ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐱᔭᕇᖅᑕᐅᓯᒪᓕᖅᑎᓪᓗᒋᑦ.
ᐃᓚᒋᐊᕐᓗᒍᓕ, ᑐᑭᓯᒋᐊᕈᑏᑦ ᖃᓄᐃᓕᖓᓂᕆᔭᖏᓐᓂᑦ ᑖᒃᑯᓂᖓ ᐱᔭᐅᔪᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᓴᓂᐊᓂ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ (TEK) ᒪᓕᓐᓂᖅᑑᔭᓚᐅᖏᑦᑐᖅ. ᑖᓐᓇ ᐋᖅᑭᓂᐊᕐᓗᒍ ᓇᓗᓇᕈᑎᐅᔪᖅ, SWG ᐱᒋᐊᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᓄᑖᒥᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ 2011−ᒥ ᐃᒪᐃᓐᓂᐊᖅᑐᑎ (1) ᖃᐅᔨᓴᕐᓗᒋᑦ ᐊᒥᓲᓂᖏᑦ ᓇᓄᐃᑦ BB-ᒥᑦ ᐊᒻᒪ KB−ᒥᑦ, ᐊᒻᒪ (2) ᑲᑎᖅᓯᐃᓗᑎ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᓇᓄᕐᓄᑦ ᓯᑯᖅᑲᕈᓃᕐᓂᖓᓄᑦ ᑖᒃᑯᓄᖓ ᐊᕙᑦᑐᖅᓯᒪᔪᓄᑦ.

ᐊᑕᖏᖅᑐᒥᑦ ᖃᓄᐃᓕᓂᐅᔪᑦ ᐊᓯᔾᔨᖅᑐᓄᑦ ᓇᔪᖅᑕᐅᔪᓄᑦ

ᑕᒪᒃᑭᓐᓂ ᐊᒥᓲᓂᖏᓐᓂ ᑕᑯᔅᓴᐅᑎᑦᑎᓯᒪᔪᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᐊᓯᔾᔨᖅᑐᑦ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᓂᕆᔭᖏᑦ ᐊᖏᔪᒧᑦ, ᐊᑯᓂᐅᔪᒧᑦ ᓯᑯᖅᑲᕈᓐᓃᑲᓐᓂᕐᓂᖓᓄᑦ BB−ᒥᑦ ᐊᒻᒪ KB−ᒥᑦ ᑭᖑᓪᓕᖅᐹᓂᓯᒪᔭᖏᓐᓂᑦ 25 ᐊᕐᕿᒍᓂᑦ. BB ᐊᒥᓲᓂᖏᑦ ᑭᓪᓕᖅᑲᓕᖅᓯᒪᔪᖅ ᐅᖓᓯᓐᓂᖓ ᐊᒻᒪ ᓴᙳᓯᒪᓪᓗᑎ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᐅᐊᓐᓇᒧᑦ ᐊᕐᕿᒍᓕᒫᒥ ᐃᓗᐊᓂ ᐊᔾᔨᒋᑎᙳᐊᖅᑐᒍ ᖃᓄᐃᓕᖓᓂᕆᔭᖓᓂ 1990−ᖏᓐᓂ. ᑖᒃᑯᐊ ᐊᒥᓲᓂᖏᑦ ᐃᓛᒃᑰᓕᒃᑲᓂᖅᓯᒪᔪᑦ, ᖃᓄᐃᓘᖅᑲᑦᑕᐅᑎᖏᓐᓂᖅᓴᐅᓗᑎ ᓴᓂᐊᓃᑦᑐᓂᑦ (KB, ᑕᓗᕈᑎᐅᑉ ᐃᒪᖓ ᐊᒻᒪ

ᕿᑭᖅᑖᓘᑉ ᐃᑭᕋᓵᖓ). ᐅᐱᕐᙳᒃᑯᑦ ᓯᑯᐃᖅᓴᕋᐃᖃᑦᑕᕐᓂᖓᓄᑦ, ᐊᐅᑦᑐᐃᓐᓇᐅᖃᑦᑕᑎᓪᓗᒋᑦ ᐊᕐᕋᒍᑕᒫᑦ ᓯᑰᔦᑦ ᐊᒻᒪ ᐅᑭᐊᒃᓵᒃᑯᑦ ᓯᑯᓈᕐᓂᖓᓄᑦ, BB ᓇᓄᖏᑦ ᐊᑯᓂᐅᓂᖅᓴᐅᕙᓪᓕᐊᓕᖅᑐᑦ ᓄᓇᒦᓐᓂᖁᕙᑦᑕᖏᑦ ᐃᒫᓅᕈᓐᓇᖏᑎᑦᑎᔪᑦ ᐃᒪᕐᒥᐅᑕᓄᑦ ᐆᒪᔪᓄᑦ ᓂᖃᑦᑕᖅᑕᖏᓐᓄᑦ.

1990-ᖏᓐᓂᑦ, ᓯᑯᖓᓄᑦ ᐊᑐᖅᑕᐅᒐᔪᑦᑐᖅ KB−ᒧᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᖅᑐᖅᓯᒪᔪᓂᑦ ᓯᑯᓂᑦ ᐊᕐᕋᒍᑕᒫᒃ ᐊᒻᒪ ᐊᕐᕋᒍᒐᓚᐅᓐᓄᑦ ᐊᐅᔭᐅᓚᐅᖅᑐᒥᑦ. ᑕᒪᓐᓇ KB ᓇᓄᖏᓐᓂᑦ ᐊᕐᕋᒍᓕᒫᒥᑦ ᓯᑯᒦᕈᓐᓇᑎᑦᑎᔪᖅ ᓇᔪᖃᑦᑕᖅᑕᖓᓄ ᓯᑯᐃᖃᑦᑕᕐᓂᖓᓅᖏᙳᖅᑐᖅ ᓵᕐᓗ BB−ᑎᑐᑦ. 1990−ᖏᓐᓂᐅᕙᔪᒻᒪᓂᑦ, ᑭᓯᐊᓂᓕ, ᖃᓄᐃᓐᓂᕆᔭᖏᑦ KB−ᒥᑦ ᑐᕌᖅᐸᓪᓕᐊᓯᒪᔪᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓄᑦ ᓯᑯᖏᑦ BB−ᒥ ᐊᐅᑦᑐᐃᓐᓇᐅᖃᑦᑕᑎᓪᓗᒋᑦ ᐊᐅᔭᑕᒫᑦ (i.e., "ᓯᑯᐃᖃᑦᑕᕐᓂᖓ ᓯᑯᖃᑦᑕᕐᓂᖓᓄᓪᓗ" ᖃᓄᐃᓐᓂᕆᔭᖏᑦ). ᖃᐅᔨᓴᕈᑎᒥᓂᖅᑎᓐᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ KB ᐊᒥᕐᓂᖏᑦ ᖃᓄᐃᓕᓂᖃᖅᓯᒪᔪᑦ ᑖᓯᒪᖓ ᐊᓯᔾᔨᕐᓂᐅᔪᒧᑦ ᓯᑯᖃᑦᑕᕐᓂᖓᓄ ᓯᑯᐃᖃᑦᑕᕐᓂᖓᓄᑦ ᐊᖏᓪᓕᒋᐊᖅᑐᑎᑦ ᐅᖓᓯᓐᓂᖏᑦ 1990−ᖏᓐᓂᐅᕙᔪᒻᒪᓂᑦ ᐱᓗᐊᖅᑐᒥ ᐊᐅᔮᒃᑯᑦ. ᐊᖏᓂᖅᓴᐃᑦ ᐊᒻᒪ ᐊᔾᔨᑮᖏᒍᓐᓇᓂᖅᓴᐅᔪᑦ ᐅᖓᓯᓐᓂᖏᑦ ᓇᔪᖅᑕᖏᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖓ ᓯᑯᖃᑦᑎᐊᖏᓐᓂᖓ ᐊᐅᔮᒃᑯᑦ ᐅᑭᐊᒃᓵᑯᓪᓗ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ. ᑖᒃᑯᐊ ᐊᓯᔾᔨᕐᓂᐅᔪᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᐃᖏᕋᓂᖏᑦ ᖃᓄᐃᓘᕐᓂᖏᑦ ᐊᑐᖅᑕᐅᕙᑦᑐᑦ ᓇᓄᕐᓄᑦ ᓇᔪᖅᓯᔪᓂᑦ "ᐊᕙᑎᖓ ᐊᕚᑦᑐᖅᓯᒪᔪᒥ" ᐊᕐᕋᒍᑕᒫᑦ ᓯᑯᖓᓄᑦ.

ᐊᓯᔾᔨᖅᑐᑦ ᐊᕐᕋᒍ ᐃᓗᐊᓂ ᐅᑎᖅᑕᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᖃᐅᔨᓴᕐᓂᖅ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ

BB−ᒥᑦ ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᔭᕐᓴᐅᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐊᕐᕋᒍ ᐃᓗᐊᓂ ᐅᑎᖅᑕᕐᓂᖏᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ BB ᓇᓄᖏᑦ ᖃᓄᐃᓕᓂᖃᖅᑐᑦ ᐊᒥᓱᓪᓚᓐᓂᑦ ᒥᑭᓕᒋᐊᕐᓂᖏᓐᓄᑦ ᓈᒻᒪᑦᑐᑦ ᓇᔪᖅᑕᐅᔪᑦ 1990−ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ. ᑕᒪᓐᓇ ᓂᖃᐅᕆᔭᐅᔪᖅ ᑲᔪᓯᓗᓂ ᑲᔪᓰᓐᓇᕐᓂᖓᓄᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ.

ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᖃᓄᐃᓘᖃᑦᑕᕐᓂᖏᑦ, ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᐃᓴᐃᑦ ᐊᕐᓇᐃᑦ ᐊᑐᖃᑦᑕᓕᖅᑐᑦ ᓯᑯᑭᓐᓂᖅᓴᐅᔪᒥ ᓯᑯᒥ ᓇᔪᖃᑦᑕᖅᑕᖏᓐᓂ ᓄᓇᒧᑦ ᖃᓐᓂᓴᐅᓗᑎ. ᓇᓄᐃᑦ ᓄᓇᒨᖃᑦᑕᓕᖅᑐᑦ ᕿᑭᖅᑖᓗᒻᒧᑦ ᓯᕗᕙᓯᓐᓂᓴᐅᓗᑎ ᑕᖅᑭᓗᐊᒥᑦ ᐊᒻᒪ ᐊᑯᓂᐅᓂᖅᓴ ᓇᔪᖅᓯᕙᓪᓕᖅᑐᑦ (20-30 ᐅᓗᓂᑦ) ᓄᓇᒥᑦ ᐅᑎᓚᐅᖅᑎᓇᒋᑦ ᓯᑯᕋᑖᒧᑦ ᓯᑯᓈᖅᓯᒪᔪᒥᑦ. ᐊᒥᕐᓂᖏᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐅᖓᓯᑦᑐᒧᑦ ᐳᐃᒥᓂᐅᔪᓂᑦ ᐊᕐᓇᓂᑦ ᐊᒥᓱᕈᖃᐊᖅᓯᒪᔪᑦ ᓇᓗᓇᐃᖅᓯᔪ ᓇᓄᐃᑦ ᖃᓄᐃᓕᓂᖅᑐᑦ ᓯᑯᖃᑦᑎᐊᖏᓐᓂᖓᓄᑦ ᐱᓯᒪᐃᓐᓇᓂᐊᖅᑐᒋᑦ ᐊᐅᔭᒃᑯᑦ ᓇᔪᖃᑦᑕᖅᑕᖏᑦ ᕿᑭᖅᑖᓗᒻᒥ. ᓇᔾᔨᔦᑦ ᓇᓄᐃᑦ ᑎᓯᖏᓐᓅᖅᓴᕋᐃᓐᓂᖅᓴᐅᓕᖅᑐᑦ ᐊᑕᐅᓯᖅᒥᑦ ᑕᖅᑭᕐᒥᑦ ᑭᓯᐊᓂ ᐊᓂᕕᕆᔭᖏᑦ ᐅᓪᓗᖏᑦ ᐊᓯᔾᔨᖅᓯᒪᖏᑦᑐᑦ. ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᕿᓚᒥᐅᓂᖅᓴᐅᔪᒥ ᑎᓯᖓᓃᓐᓂᖏᓐᓄᑦ ᖃᐅᔨᒪᔭᐅᖏᑦᑐᑦ. ᑭᓯᐊᓂᓕ, ᖃᐅᔨᔭᐅᔪᑦ ᑎᓯᖏᑦ ᖁᕙᓯᓐᓂᖅᓱᒦᖃᑦᑕᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ ᐊᐳᑎᖃᑦᑎᐊᖏᓐᓂᖓᓄ ᐃᓅᕈᖏᑦ ᑲᓇᓐᓇᖓᓂ ᕿᑭᖅᑖᓗᒃ ᐱᔾᔪᑎᒋᔭᐅᔭᓐᓇᖅᑐᑦ.

ᖃᐅᔨᓚᐅᕐᒥᔪᒍᑦ ᐊᑦᑎᓕᕆᐊᖅᑐᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᑎᒥᖏᓐᓂᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ BB ᓇᓄᖏᑦ. ᑎᑎᕋᖅᑕᐅᖃᓯᐅᑎᔭᐅᓂᖏᑦ ᐊᑎᖅᑕᐃᒃ ᐊᕐᕋᒍᒧᑦ ᖃᑦᑎᓐᓇᕈᕆᐊᖅᓯᒪᔪᑦ ᒪᓕᓐᓂᖃᖅᑐᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐅᐱᕐᙳᒃᑯᑦ ᓯᑯᐃᖅᓴᕋᐃᖃᑦᑕᕐᓂᖓᓄᑦ 1990−ᖏᓐᓂᓂᑦ. ᑭᓕᖃᖅᑐᓂᑦ ᒥᔅᓴᐅᓴᐅᒧᓐᓇᕋᓗᐊᖅᑎᓗᑕ ᐆᒪᓂᖏᓐᓄᑦ, ᐊᑦᑎᓚᐅᖅᑐᑦ ᐃᒻᒥᒍᖅᑐᑦ ᐊᖑᑎᑦ 2000ᙵᖏᓐᓂᑦ ᐊᔾᔨᒋᑎᙳᐊᖅᑐᒍ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓄᑦ ᑐᑯᔅᓴᐅᑎᑦᑎᔾᔪᓐᓇᖅᑐ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᐊᕙᑎᖓ ᖃᓄᐃᓐᓂᕆᔭᖓ. ᑭᖑᓪᓕᖅᐹᒥ, ᐊᑦᑎᓕᕆᐊᖅᑐᑦ ᑎᒥᖏᓐᓄᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᓇᓄᕐᓂᑦ BBᒥᑦ ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2013 ᑕᐃᒪᐃᔅᓯᒪᔪᑦ ᐊᑦᑐᐊᓂᖅᑐᓂ ᐊᑯᓂᐅᓂᖓᓄ ᓯᑯᖅᑎᓐᓇᒍ ᐊᒻᒪ ᐅᐱᕐᙳᒃᑯᑦ ᓯᑯᐃᖃᑦᑕᕐᓂ ᐅᓪᓗᖏᑦ.

ᑕᑯᓐᓇᖅᑐᒍ BB–ᒧᑦ, ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᕿᑐᓐᖏᕐᑯᑲᑦᑕᕐᓂᖏᓐᓄᑦ KB–ᒧᑦ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2000–ᖏᓐᓂᑦ (ᑭᓯᐊᓂ ᓯᐊᒻᒪᖅᓯᔪᓂᑦ ᑐᑭᓯᐊᕈᑎᑦ ᑮᓕᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᐅᕙᖑᑦ ᓄᖃᐅᓯᖓᓄᑦ), ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᔭᔅᓴᐅᔪᓂᑦ ᐱᐅᓯᕆᐊᖅᑐᓂᑦ ᑎᒥᖏᓐᓂ ᖃᓄᐃᓐᓂᓇᔭᖏᓐᓂ 2000–ᖏᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᓐᓇᖅᑐᖅ ᓴᖅᑭᖃᑦᑕᖅᑐᓂᑦ ᑭᓯᐊᓂ ᐃᓚᒋᔪᓐᓇᖅᑕᖓ ᖃᓄᐃᓕᓂᖓ ᐊᑯᓂᐅᔪᒧᑦ ᐊᓯᔾᔨᖅᑐᓄᑦ ᓯᑯᖓᓄᑦ Kane Basin, ᓴᓂᐊᓃᑦᑐᑦ ᐆᒪᔪᓕᐅᖃᑦᑕᕐᓂᖏᑦ ᐊᕙᑎᖓᓂ ᕐᑯᕝ>ᕆᐊᖅᓯᒪᔪᖅ.

ᐱᑕᖃᕐᓂᖏᑦ

ᐊᐅᖏᑎᒎᖅᑐᓂ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᖅᕝᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᐅᓂᖅ BB– ᐊᒥᓲᓂᖏᓐᓄᑦ ᓴᖅᑭᑎᑦᑎᓯᒪᔪᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᒐᔪᑦᑐᓂᑦ (2012-2013) ᑲᑎᑦᑐᒥᑦ 2,826 ᓇᓄᐃᑦ (95% CI = 2,059-3,593). ᐱᔾᔪᑎᖃᖅᑐᑎ ᐊᒥᓲᒐᓚᓐᓂᑦ ᑮᓕᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ. ᐅᖃᐅᓯᐅᔪᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᓪᓗᑎ ᐅᕙᓂ ᐅᓂᒃᑳᓂᑦ, ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2000–ᖏᓐᓂᑦ ᑕᕝᕚᖓᑦᑕᐅᑎᒥ ᐊᔾᔩᙳᐊᖅᑎᒐᔅᓴᐅᖏᑦᑐᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᓇᓗᓇᐃᖅᓯᓂᖃᕈᓐᓇᖏᑦᑐᖅ ᐊᓯᔾᔨᖃᑦᑕᖅᑐᓂᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ BB ᓇᓄᖏᓐᓄᑦ. ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᑕᑯᔅᓴᐅᑎᑦᑎᓯᒪᔪᖅ ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᐊᑖᒍ ᐊᒥᓲᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᖏᓂᖅᓴᐅᔪᖅ ᓴᓂᐊᓂ ᓇᓗᓇᐃᖅᑕᐅᔪᒥ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ (1,546 ᓇᓄᐃᑦ cf. PBSG 2015); ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᐃᓚᒋᓕᓚᐅᖅᑕᖏᑦ PBSG’s (2015) ᓇᓗᓇᐃᖅᑕᐅᓂᖓ BB ᐊᑐᑦᑎᓪᓕᕚᓪᓕᐊᓂᖏᑦ ᐊᒥᓲᓂᖏᑦ. TEK ᑐᑭᓯᐊᕈᑏᑦ ᓇᓗᓇᐃᖅᓯᓯᒪᖏᑦᑐᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᑦᑎᓪᓕᒋᐊᖅᑐᓂᑦ BB ᓇᓄᖏᓐᓄᑦ (ᓲᕐᓗ, Dowsley and Taylor 2006, Born et al. 2011).

ᓇᓗᓇᕈᑎᑕᖃᕋᓗᐊᖅᑎᓪᓗᒍ ᖃᐅᔨᔭᐅᔪᓂᑦ BB–ᒥᑦ, ᐊᒥᓲᒐᒪᐃᑦ ᖃᐅᔨᔭᐅᔪᑦ ᐊᑦᑐᐊᓂᓗᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᐃᖏᕋᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᓇᔪᖅᑕᐅᔪᒥ ᐊᑐᐃᓐᓇᐅᓂᖓᓄᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ, ᖃᓄᐃᑑᕐᓂᖓ, ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ, ᐊᒻᒪ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᑖᒃᑯᐊ BB ᓇᓄᖏᑦ ᒫᓐᓇᐅᔪᒥᑦ ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᕐᓂᑰᖅᑐᑦ ᓄᖑᑉᐸᓕᐊᓂᖏᓐᓄᑦ ᓇᔪᖅᑲᑦᑕᖅᑕᖅ. ᒫᓇᐅᔪᒥᑦ ᑕᐃᒪᐃᓕᖓᑐᐃᓐᓇᖅᑲᑕ ᓯᑯᖓ ᓈᒻᒪᑦᑐᒥ ᒥᔅᓴᐅᓴᔅᓴᐅᔪᑦ ᐊᓯᔾᔨᑲᓂᓂᐊᖅᑐᑎᑦ ᑕᕝᕙᓂ ᐊᒥᓲᓂᖏᓐᓂᑦ ᐱᖃᓯᐅᑎᓪᓗᑎ, ᑕᐃᒪᓂᖃᑲᓇᐊᖃᖅᑐᓂ, ᐊᑦᑎᓪᓕᒋᐊᖅᑐᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐱᒻᒪᕆᐅᔪᑦ ᕐᑯᕝᕙᓯᓐᓂᓇᕆᐊᖃᖅᑕᖏᑦ, ᑖᓐᓇ ᖃᐅᔨᒪᕆᐊᖃᖅᑎᑦᑎᔪᖅ ᑕᒪᒃᑭᓐᓂ ᓯᕗᓂᖅᓴᒥ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ.

ᐊᑐᖅᑐᒥᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ–ᐱᔭᐅᒃᑲᓂᖅᑐᑦ KB–ᒥᑦ, ᓄᑖᖑᔪᖅ (2013-2014) ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᑖᒃᑯᓄᖓ ᐊᒥᓱᓂᖏᓐᓄᑦ 357 ᓇᓄᐃᑦ (95% CI: 221 – 493). ᐋᖅᑭᑕᐅᒃᑲᓂᖅᓯᒪᔪᖅ 1990 ᐱᔭ.ᐅᒃᑲᓂᖅᑐᒥᑦ ᑐᑭᓯᐊᕈᑎᓂᑦ KB–ᒥᑦ ᒥᔅᓴᐅᓴᑎᑦᑎᓯᒪᔪᑦ 224 ᓇᓄᐃᑦ (95% CI: 145 – 303) ᐊᑯᓐᓂᖏᓐᓄ 1995 – 1997. ᐊᔾᔩᙳᐊᖅᑎᑕᐅᔪᑦ ᑖᒃᑯᓂᖓ ᒪᕐᕉᓐᓂ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓐᓂᒃ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐱᑕᖃᕐᓂᖓᓄ ᒪᑭᑕᑦᑎᐊᖅᑐᒥ ᐊᒥᓱᕈᖅᐸᓪᓕᐊᔪᓂᓪᓗᓐᓃᑦ, ᒪᓕᓐᓂᖃᖅᑐᖅ ᑐᑭᓯᐊᕈᑎᓂᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᒪᓕᓐᓂᖃᖅᑐᑎ TEK (Born et al. 2011). ᑖᒃᑯᐊ ᐊᓯᔾᔨᕐᓂᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᒥᓲᓂᖏᑦ ᑖᒃᑯᐊ ᓈᒻᒪᑎᐊᖅᑐᑦ ᐊᒻᒪ ᑕᐃᒪᐃᓕᖓᐃᓐᓇᕈᔪᑦᑐᑎ. ᑖᒃᑯᐊ ᖃᐅᔨᔭᐅᓂᐅᔪᑦ ᐊᔾᔩᖏᑕᖏᑦ ᐊᑦᑎᓂᖅᓴᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐱᔭᐅᔪᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ (Taylor et al. 2008); ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᐃᓚᒋᓕᓚᐅᖅᑕᖏᑦ PBSG’s (2010) ᓇᓗᓇᐃᖅᑕᐅᓂᖓ KB ᐊᑐᑦᑎᓪᓕᕚᓪᓕᐊᓂᖏᑦ ᐊᒥᓲᓂᖏᑦ.

ᑖᒃᑯᐊ ᐊᓯᔾᔨᖅᑐᑦ KB–ᒥᑦ ᖃᐅᔨᑎᑦᑎᒍᓐᓇᖅᑐᑦ ᐱᐅᔪᒥ ᖃᓄᐃᓐᓂᐅᔪᒥᑦ ᐊᑦᑐᖅᑕᐅᒋᐊᓐᖓᐅᑎᓄᑦ ᓯᓚᐅᑉ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᕐᑯᑎᑦᑐᒥ ᐅᑭᐅᑕᖅᑐᒥ ᐃᓱᒪᒋᔭᐅᓯᒪᓚᐅᖅᑐᑦ ᓇᓄᕐᓄᑦ. ᑭᓯᐊᓂᓕ, ᓇᓗᓇᐃᖅᓯᔪᒧᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᓯᑯᒥ ᑲᔪᓯᑉᐸᑕ ᑖᒃᑯᐊ ᐱᐅᔪ)ᓂᑦ ᐊᑦᑐᐃᓂᖃᕐᓂᐊᖅᑐᒋᔭᐅᔪᑦ ᐋᖅᑕᐅᓗᓂ ᑭᖑᔅᕝᕙᖅᑕᐅᓗᓂᓗᓐᓃᑦ ᐱᐅᖏᑦᑐᓄᑦ ᐊᑦᑐᖅᑕᐅᔪᓄᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᐊᑐᕐᑯᓯᔪᒧᑦ ᖃᐅᔨᓴᐃᓇᕐᓗᒥᑦ ᓯᑯᓪᓗ ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᖃᐅᔨᓴᑲᐃᓇᖃᑦᑕᕐᓂ ᖃᐅᔨᒪᔭᐅᑎᓄᑦ ᐊᓯᔾᔨᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᑖᒃᑯᓂᖓ ᑲᑦᑏᓐᓇᐅᔪᓂᑦ ᐊᓯᒥᓲᓂᖏᓐᓄᑦ.

ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐃᓕᓴᕆᔭᐅᔪᑦ

Aars, J., T.A. Marques, S.T. Buckland, M.Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear population size. Marine Mammal Science 25:35-52.

Anon. 2009. Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations, 9 pp.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1, Government of Nunavut, 43 pp.

Dowsley, M., and G. W. Wenzel. 2008. ''The time of the most polar bears'': A co-management conflict in Nunavut. Arctic 61:177–189.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st- century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Laidre, K. L., E. W. Born, P. Heagerty, Ø. Wiig, H. Stern, R. Dietz, J. Aars, and M. Andersen. 2015. Shifts in female polar bear (*Ursus maritimus*) habitat use in East Greenland. Polar Biology 38:879–893.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320

Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Population status. Status table. http://pbsg.npolar.no/en/status/status-table.html

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2010. Report of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 18 pp

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

Taylor, M.K., S. Akeeagok, D. Andriashek, W. Barbour, E.W. Born, W. Calvert, D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, F. Messier 2001. Delineation of Canadian and Greenland Polar Bear (*Ursus maritimus)* populations by cluster analysis of movements Canadian Journal of Zoology 79: 690-709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing- Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations

Final Report to the Canada-Greenland Joint Commission on Polar Bear
from the Scientific Working Group (SWG) of the Canada-Greenland Joint Commission on Polar Bear

Prepared by: Stephen Atkinson (SWG), Erik W. Born (SWG), Kristin L. Laidre (SWG), Nicholas J. Lunn (SWG), Øystein Wiig (SWG) and external experts Todd Arnold, Markus Dyck, Eric V. Regehr, Harry Stern, and Seth Stapleton

Edited by: Kristin L. Laidre and Nicholas J. Lunn

31 July 2016

Please cite as: SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear. 31 July 2016: x + 636 pp.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS

Over the previous 6 years during which this research program was developed, field work undertaken, data analyzed, and a final report written and delivered to the Canada-Greenland Joint Commission on Polar Bear, many individuals and organizations provided invaluable assistance. We hope that the following acknowledgements capture their collective contributions appropriately but recognize that we may have missed some; omissions do not reflect diminished involvement but rather that it was not possible to name everyone involved.

Financial, logistical, and in-kind support were provided by: Air Greenland, Applied Physics Laboratory (University of Washington), Avannaa Resources Inc. (Copenhagen), Niels Berthelsen (Upenavik), Bureau of Mineral and Petroleum (Nuuk), Canada Department of National Defense, Environment and Climate Change Canada, Environmental Protection Agency (Ministry of Environment and Food of Denmark) -DANCEA Programme, Government of Nunavut, Greenland Department of Fisheries, Hunting and Agriculture (DFHA), Greenland Institute of Natural Resources, Greenland Pharmaceutical Authority (Nuuk), the Hospital in Upernavik – Peqqik, KNAPK (Greenland Hunters and Fishermen Organization), Kullorsuaq School, Lasø ApS (Upernavik), Mitteqarfik airports in Qaarsut and Upernavik, the Mittimatalik Hunters and Trappers Organization, the National Aeronautical and Space Agency (NASA) Development and Testing of Potential Indicators for the National Climate Assessment program grant NNX13AN28G (PI: H. Stern), NASA Climate and Biological Response grant NNX11A063G (PI: K. Laidre), the Namautaq Hunters and Trappers Organization, the Nativak Hunters and Trappers Organization, Nuna Minerals Inc. (Nuuk), Nunavut Wildlife Management Board, Parks Canada Agency, Polar Continental Shelf Project, Polar Science Center (University of Washington), Qaanaap Kommunia, Qikiqtaaluk Corporation, Quantum Murray LP, Royal

Canadian Mounted Police, School of Aquatic and Fishery Sciences (University of Washington), Thule Air Force Base (Base Commander, US Air Force and Danish Liaison Officer, Danish Navy), Universal Helicopters, University of Oslo, Upernavik Seafood (Upernavik), Uummannaq Sundhedscenter – Peqqik, Vetlesen Foundation, and World Wildlife Fund (Nunavut General Monitoring Program).

The success of the aerial survey and genetic mark-recapture programs was largely due to the pilots and engineers who went above and beyond their call of duty. In particular, we thank Geir Akse, Jim Berry, Gerard Hartery, Sebastian Holst, John Innes, Steve Lodge, and Lorne Pike.

Mark Akeeagok, Dennis Andriashek, Apoosie Apak, Ilannguaq Egede, Carey Elverum, Paul Frame, Bruce J. Hainnu, Peter Hegelund, Jacob Jaypoody, Martika Jensen, Jaypeetee Kalliktee, Robert Kautak, Jopee Kigutak, Brian Koonoo, Jaloo Kooneloosie, George Koonoo, Juulut Kruse, Karl Kristian Kruse, Juilie Kuksiak, Robbie Kullualik, Jon Neely, James Noble Jr., Timothæus Petersen, Adrian Jimmy Pitseolak, polar bear hunters in Niaqornat, Kullorsuaq and Upernavik, Tyler Ross, Vicki Sahanatien, H.C. Schmidt, Guillaume Szor, and Mathias Therkelsen provided invaluable assistance and friendship in the field.

We thank the staff of Wildlife Genetics International (Nelson, BC, Canada) for their expertise in genotyping such a large number of samples and for on-going management of a polar bear genotype database. Staff at the Danish Center for Environment (Århus University) prepared and aged polar bear teeth.

We also thank Drs. Jon Aars (Norwegian Polar Institute), Andrew Derocher (University of Alberta), Jeff Laake (NOAA), and Elizabeth Peacock (USGS) for reviewing the survey options assessment report for Baffin Bay and Kane Basin. We are grateful to Dr. Gary White, Colorado State University, who kindly reviewed our mark-recapture modeling approach and interpretations of the analysis. Professor Patrick Heagerty, University of Washington, offered statistical consultation on resource selection models. Erica Escajeda conducted denning analyses as part of her Masters work at the University of Washington and Richard McGovern assisted with GIS mapping. We also thank professor Lutz Bachmann (University of Oslo) for reviewing the results of the population genetic analyses.

Permits and authorizations to undertake this work were issued by Canadian Border Services Agency (SERL-12-008, SERL-13-012, SERL-13-130), Danish Transport and Aviation Authority, Environment Canada National Wildlife Areas research permit (Nun-NWA-13-07), Greenland Aviation Authority, Greenland Department of Environment (Nuuk), Greenland Department of Fisheries, Hunting and Agriculture (DFHA), Greenland Department of Health, Greenland Home Rule Government, Government of Nunavut Wildlife Research Permits (2011-041, 2012-010, 2012-015, 2013-011, 2013-0292014-007), Nunavut Impact Review Board screening (13YN017 – exemption from environmental assessment screening), Parks Canada Research and Collection Permit (ANP-2011-8533), Qikiqtani Inuit Association (authorization Q13X009 – permission to access Inuit-owned lands).

The Animal and Use Care Committee at the University of Minnesota (1403-31404A), Greenland Department of Environment (Nuuk), and Greenland Department of Health reviewed and approved the methods used in this research.

CHAPTER 1

INTRODUCTION

We briefly present the history of the different polar bear management systems in Nunavut/Canada and Greenland to provide the background and context on the current assessment of the status of the polar bear subpopulations in Baffin Bay (BB) and Kane Basin (KB). We summarize the results of the previous assessment of the BB and KB subpopulations in the 1990s and the framework established in 2009 for the Canada-Greenland joint management of these shared subpopulations. We describe the process leading to the decisions on major objectives of a re-assessment and the subsequent development of a research plan for re-assessing the status of the polar bear subpopulations in Baffin and Kane Basin. Finally, we present the schedule of the completion of the study.

1.1. History of Polar Bear Management in Canada and Greenland related to Baffin Bay (BB) and Kane Basin (KB)

Wildlife legislation in Canada did not address polar bear harvesting until 1935, when a hunting season from 1 October through 31 May was imposed. In 1949, hunting was restricted to native people. Arbitrary quotas for polar bears were introduced in Canada in 1967 that were based largely on the fur records from several preceding years (Lee and Taylor 1994, Prestrud and Stirling 1994).

Through a delegation of authority from the federal government, ultimate responsibility for the management of polar bears in Nunavut lies with the Government of Nunavut, as represented by the Minister of Environment (Lunn et al. 2010). However, this responsibility is subject to the terms of the Nunavut Land Claims Agreement (NLCA) that established a system of ‘co-management’ for wildlife. Under the NLCA, the Minister’s decision-making authority for wildlife management is shared with the Nunavut Wildlife Management Board and is subject to strict requirements for consultation with Regional Wildlife Organizations and community-based Hunters and Trappers Organizations. The intent of this co-management system is to ensure that decisions are based on the best available science and Inuit Qaujimajatuqangit (Inuit traditional knowledge), and that these decisions consider not only conservation as a founding principle but also take into account the values, beliefs, views and needs of Inuit. The system is also designed to ensure that Inuit are involved in all aspects of wildlife management including research, monitoring, and harvest management.

Polar bears occur in relatively discrete subpopulations and are managed as such in Canada. The annual polar bear harvest in Nunavut is within the estimated sustainable yield of females and controlled through a male-biased, sex-selective quota system (Taylor et al. 1987, 2008b). Females accompanied by cubs, cubs, and bears in or constructing dens are protected by law (Lunn et al. 2010). Currently, the quota year in Canada runs from 1 July through 30 June of the following year.

In Greenland, regulations for the catch of polar bears in the entire country were enforced beginning 1 January 1975 (prior to 1975 regulations in NE Greenland had existed since the 1937; Born 1995 and references therein). The regulations prior to the introduction of quotas in Greenland in 2006 to reduce hunting effort and protect females with cubs and also involved a closed season in July-August (ibid). Furthermore, hunting of polar bears was restricted to Greenlandic citizens who had hunting as their main occupation. Quotas for the Greenlanders´

catch of polar bears were introduced in 2005 and took effect 1 January 2006 (Lønstrup 2006). The quota year in Greenland is between 1 January and 31 December.

The management history and harvest monitoring in Nunavut and Greenland are described in more detail in Chapter 8.

1.2. Previous Delineations and Assessments of the Baffin Bay (BB) and Kane Basin (KB) Subpopulations

Delineation of the Baffin Bay and Kane Basin subpopulations (Figure 1.1) was largely based on movements of collared bears and the recapture or harvest of tagged animals and has been well documented (PBSG 1998, SWG 2010). The BB subpopulation is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island (Nunavut, Canada) to the west (Taylor et al. 2001). A relatively distinct southern boundary at Cape Dyer, Baffin Island, and the entrance to Kangerlussuaq/Søndre Strømfjord, Greenland, is evident from the movements of collared or tagged bears (Stirling *et al.* 1980, Taylor et al. 2001).

A study of microsatellite genetic variation based on biological samples collected during the first half of the 1990s revealed significant genetic variation between polar bears in BB and neighboring Davis Strait (DS) and Lancaster Sound (LS), but not between BB and KB (Paetkau et al. 1999).

The BB, KB and DS subpopulations are shared between Greenland and Canada (Taylor et al. 2001). Population inventories involving physical mark-recapture, in combination with satellite telemetry, were conducted jointly by Nunavut/Canada and Greenland during 1991-1997 with the objective to estimate the size of the BB and KB subpopulations. These resulted in an estimate for BB of 2074 bears (95% CI: 1544-2604; Taylor et al. 2005, PBSG 2006) and for KB of 164 bears (95% CI: 94-234; PBSG 2006, Taylor et al. 2008).

Due to concerns with respect to the reported harvest occurring in BB, modelling was used to project how many bears there would be in 2004 based on the 1997 BB estimate and associated vital rates plus the reported annual catches in Nunavut and Greenland. The results suggested that BB would have experienced a decline to 1546 polar bears in 2004 (95% confidence interval: 690-2402; PBSG 2006). Although a similar modeling exercise was not done for KB, both subpopulations were thought to be declining as a result of overharvest (PBSG 2006, 2010, 2015).

1.3. Canada-Greenland Joint Commission (JC) on Polar Bear and the Scientific Working Group (SWG)

The Canada-Greenland Joint Commission on Polar Bear (JC) was established with the signing of a "*Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*" (MOU) on 30 October 2009 (Anon. 2009). Primary objectives of the MOU are to: "(1) to manage polar bear within the Kane Basin and Baffin Bay management units in order to ensure their conservation and sustainable management into the future, and, (2) establish an effective system of management which will include adhering to the principles of conservation".

The JC subsequently established a 5-member Scientific Working Group (SWG) to provide it with scientific advice and recommendations with respect to the conservation and management of the BB and KB polar bear subpopulations. Two of the SWG´s members shall represent Canadian and two Greenlandic research institutes/agencies, whereas the fifth member is appointed by the SWG co-chairs from a research institute that is independent of both Canada and Greenland. To assist the SWG in providing the best scientific advice, external experts can be invited to participate in its work.

1.4. Subpopulation Re-Assessment

After an initial meeting in Ottawa, Canada in January 2010, the JC tasked the SWG with using the best available scientific information to:

(1) Propose Total Allowable Harvest (TAH) levels for the Kane Basin and Baffin Bay subpopulations and provide the Joint Commission with a written report of its recommendations; and,

(2) Provide science advice to the Joint Commission for monitoring the effects of habitat changes on polar bears.

The SWG provided a report to the JC at the 2nd meeting of the JC in Ilulissat, Greenland, in May 2010 (SWG 2010). The SWG noted that 100% of simulations using population viability analysis and current harvest levels showed that both the Baffin Bay and Kane Basin subpopulations declined after 10 years (PBSG 2010). The SWG also noted that simulations to assess the estimated risk of subpopulation decline are typically run 10-15 years into the future from the point in time at which data were last collected to estimate abundance and vital rates. Furthermore, it was noted that there was uncertainty surrounding the magnitude of impacts of environmental change on polar bears and that demographic rates could have significantly changed since the last inventories of these subpopulations in the 1990s.

The SWG also noted that a common Canada-Greenland management goal for the BB and KB subpopulations had not yet been specified in detail. Accordingly, the SWG assumed that a recommended TAH for BB and KB should ensure that each subpopulation remained at the accepted science-based estimate (PBSG 2010) until new scientific inventories had been conducted and management goals established.

Acknowledging the fact that considerable uncertainty existed about the status of the BB and KB subpopulations in 2010 and that clearly defined management goals had not been identified, the SWG estimated the sustainable TAH from the BB subpopulation to be 90 bears/year and to be 1-2 bears/year from the KB subpopulation (SWG 2010).

In order to address the second question posed by the JC, the SWG summarized items that should be monitored in BB and KB together with the possible monitoring approaches. A list of general scientific areas where monitoring is required, the rationale for the monitoring, and the potential methods that can help gather data under each topic were presented. Furthermore, the SWG indicated, where possible, how monitoring can be conducted by scientists and local users concomitantly.

Based on consideration of the key parameters that should be monitored in order to determine the effects of climate change on the BB and KB polar bear subpopulations and the need to ensure that harvesting of the two subpopulations is sustainable in a changing environment, the SWG recommended that the following research needs were given priority:

(1) Subpopulation size

(2) Distribution and movements

(3) Physical condition of individual bears

(4) Vital parameters (survival and reproduction)

The SWG also noted that other data useful for determining the effects of habitat change in polar bears should be collected routinely. Collection of these data involves the active participation of users through the submission of information and samples from harvested polar bears. Sampling from the harvest will provide important information on age and sex structure of the harvest, body condition, reproduction, and levels of pollutants in polar bears.

However, given the age of the data on abundance and vital rates (PBSG 2010) combined with large-scale environmental changes in Baffin Bay during the last decades, the SWG **strongly recommended** that new estimates of subpopulation abundance, subpopulation delineation, and vital rates be given high priority (SWG 2010).

Pilot aerial surveys in Baffin Bay in 2009 and 2010

In 2009 (i.e., prior to establishment of the SWG), scientists from the Government of Nunavut Department of Environment (GNDE) and Greenland Institute of Natural Resources (GINR) initiated a collaboration to determine the feasibility of using aerial surveys for estimating the size of polar bear subpopulations in BB and KB. A pilot survey was conducted in the Sirmilik National Park on NE Baffin Island in August 2009 to determine if the line-transect aerial survey technique could be used during the ice-free or open-water season in Baffin Bay. However, given the results from the rugged terrain and mountainous landscape, it was concluded that this method was not feasible (Stapleton 2010).

Based on the experience in 2009, scientists from Nunavut and Greenland decided to (1) conduct a pilot aerial survey over sea ice during spring in Baffin Bay, and (2) deploy satellite-transmitters on polar bears in Baffin Bay in order to collect data on distribution and movement necessary for evaluation of aerial survey data.

During 27 May-4 June 2010, a pilot survey was flown over the fast and pack ice in Baffin Bay off SE Baffin Island. The survey was determined to be a success and provided data that demonstrated promise for the approach in BB. Based on the results of the 2010 spring pilot survey, a group of experts in polar bear ecology and population dynamics, population assessment methods and sea ice from the Greenland Institute of Natural Resources, Government of Nunavut, University of Washington, University of Minnesota, and U.S. National Marine Mammal Laboratory evaluated the data and concluded that a springtime aerial survey would be a feasible approach for estimating abundance of polar bears in BB and KB and should result in estimates with adequate precision for management purposes (Laake 2010, Stapleton et al. 2010, SWG 2011).

Survey options review

Following the advice from the SWG, the JC tasked the SWG to review and evaluate various methods for assessing the number of polar bears in BB and KB (JC 2010a). The JC further requested that the SWG´s report include evaluation of benefits and limitations of using each of the proposed methods (aerial surveys, genetic mark-recapture, and physical mark recapture) and indicated that the review should be based on a pilot aerial survey conducted in Baffin Bay in Spring 2010 (see preceding section). The SWG was also tasked with providing recommendations for one or more scenarios that represented the best way forward and, for each scenario, to identify major attributes, risks and management questions including level of funding (JC 2010a). Subsequently the JC would make recommendations on the most appropriate survey methodologies for assessing the BB and KB subpopulations (JC 2010a).

In January 2011, the SWG submitted to the JC a review of options for conducting new research including recommendations on appropriate methodologies (SWG 2011), which considered the pros and cons of using physical mark-recapture, genetic mark-recapture, or aerial surveys for subpopulation inventories in BB and KB. The general conclusion was that the physical MR method (1) is the most well established method available for estimating abundance of polar bears, (2) is the most widely accepted and recommended method by the greater scientific community, and (3) provides the maximum information needed for sound management advice on polar bears. In addition to an abundance estimate, it provides information that could be used to assess effects on bears of climate change and pollution.

The SWG concluded that the physical MR method was superior to aerial surveys and genetic MR because it yields the most detailed information and recommended that physical MR be used for estimating the abundance of polar bears in Baffin Bay. It was stated that aerial surveys and a multiple-year genetic MR may be considered as alternatives for assessing the number of polar bears in Baffin Bay. However, the SWG noted that there are disadvantages to both methods.

Decision of the JC

In March 2011, the JC discussed survey methods for Kane Basin and Baffin Bay with Drs. Erik Born and Stephen Atkinson who represented the SWG. Following discussion of the SWG's report, the JC concluded that the genetic mark-recapture method was preferable based on lack of support by Inuit in Nunavut for physical mark-recapture, and on concerns that variability in sea-ice conditions have the potential to create an unacceptable amount of risk in obtaining accurate subpopulation estimates via aerial surveys. Thereafter, the JC recommended to the signatories of the MOU that a 3-year biopsy darting research program be developed.

1.5. A Multi-Year Research Plan for Baffin Bay and Kane Basin

In March 2011, the JC requested the SWG prepare a research plan for the re-assessment of the Baffin Bay and Kane Basin polar bear subpopulations. It was determined that preparation of such a plan was the responsibility of those members of the SWG who would serve as lead investigators on behalf of Nunavut and Greenland. Consequently, a multi-year research plan (Atkinson et al. 2011) for re-assessment of the Baffin Bay and Kane Basin subpopulations was developed by Drs. Stephen Atkinson (Nunavut), Erik Born and Kristin Laidre (Greenland Institute of Natural Resources).

The plan outlined a multi-year research program to be carried out collaboratively by scientists from Canada (Nunavut) and Greenland together with external collaborators (local people and scientists) participating in various parts of the study. The plan presented tentative schedules and budgets. The main goals of the research program were (1) to determine the size of the Baffin Bay and Kane Basin subpopulations of polar bears, and (2) evaluate how polar bears in these areas are affected by the decrease in sea ice.

The proposed research program had three basic field components: (1) Biopsying polar bears along E Baffin Island, in NW Greenland and in the Kane Basin region, (2) deployment of satellite transmitters on male and female polar bears in NW Greenland and Kane Basin, and (3) hunter collection of tissue samples from the catch of polar bears (harvest recoveries) in BB and KB (and adjacent subpopulations). A 3-year study was proposed beginning in the fall of 2011 with the purpose to:

1) Estimate the abundance and sex (and approximate age) composition of polar bears in BB and KB;

2) Compare a new estimate of abundance with those derived from previous studies (1991-1997) in-order to gain insight into subpopulations trend;

3) Delineate the boundaries of the BB and KB subpopulations and reassess the validity of these areas as a demographic unit;

4) Estimate survival and reproductive parameters (to the extent possible) in-order to facilitate population viability analyses; and,

5) Evaluate polar bear distribution with respect to environmental variables, particularly ice conditions, topography and food availability/distribution.

Results generated by the proposed research program have the following potential applications:

1) The development of an updated status report for BB including recommendations on sustainable harvest levels; and,

2) The development of models to assess the effects of changes in habitat (in particular sea ice) on bear distribution.

Schedule

In BB, main field operations were conducted during spring and fall. Due to logistical constraints (remoteness of the survey area and consideration of light conditions in fall) field work was concentrated in spring in KB. The schedule of the proposed study was:

Biopsying for genetic mark-recapture assessment

1) Fall biopsying along eastern Baffin Island fall 2011, 2012 and 2013.

2) Fall biopsying in NW Greenland fall 2012 and 2013.

3) Spring biopsying in NW Greenland 2011, 2012 and 2013.

4) Spring biopsying in Kane Basin 2012, 2013, and 2014 (optional).

Deployment of satellite radios

1) Deployment during spring in NW Greenland 2011, 2012 and 2013 of satellite radios (satellite radios had also been deployed in these areas during spring 2009 and 2010).

2) Deployment of satellite radios during spring in Kane Basin 2012 and 2013.

Hunter collection of tissues from the polar bear catch (harvest recoveries)

1) Nunavut spring 2011 and onward.

2) Greenland spring 2012 and onward.

Data analyses

1) Genetic laboratory analyses, analyses of MR data, genetic data and satellite telemetry data 2012 and onward

Final reporting

1) September-October 2014

1.6. Process of BB and KB Assessment (2011-2016) and Delays

Program activities

Activities (field operations, laboratory analyses, data analyses, and reporting) in connection with the research program are summarized here. Details of the various activities are presented in the Materials and Methods sections of the various chapters of this report.

Field activities

Personnel from several research institutions participated in planning and conducting the field work including GINR, GNDE, University of Oslo, and University of Minnesota. In addition, local polar bear hunters in Nunavut and Greenland participated on several flights aiming to obtain biopsy samples from unrestrained bears and/or immobilizing bears to furnish them with satellite radios (NW Greenland and Kane Basin). In both Nunavut and Greenland, community consultations were conducted each year and information about the study was provided through meetings and via public media to local communities and the broader public audience.

The spring and fall biopsy program (2011-2013) along Baffin Island and in W and NW Greenland from the BB subpopulation was conducted as planned. From 2011 to 2013, 1,111 bears were biopsy darted along eastern Baffin Island (and genotyped) which was substantially more than anticipated. From 2009 to 2013, 143 bears were physically marked or biopsy darted (and genotyped) in W and NW Greenland. The spring biopsying program in Kane Basin was also successful although a third spring season was needed and completed during 2014. From 2012 to 2014, 129 bears were physically marked and genotyped or biopsy darted and genotyped in KB (Chapter 2 and 5).

Additional to the original research plan and concurrent with the biopsy sampling, a systematic aerial survey using sight-resight distance sampling protocols was conducted during spring 2014 to assess the number of polar bears in the Kane Basin subpopulation. Adding this extra component allowed for a comparison of estimates of subpopulation size via two different methods (i.e., genetic MR and aerial survey).

During 2011-2013 a total of 66 satellite radios (35 F, 31 M) were deployed in W and NW Greenland (in addition 35 satellite radios had been deployed there in 2009 and 2010; 20 F, 15 M). During 2012 and 2013 a total of 36 satellite radios (21 F, 15 M) were deployed in the Kane Basin region. Some individuals were recaptured during the study and furnished with new satellite radios. Hence, a total of 91 individual bears were tagged with satellite transmitters in BB and 34 individual bears in KB (Chapter 2). The satellite radios included small ear satellite tags developed by GINR for tracking adult male polar bears and sub-adults of both sexes (Born et al. 2010, Laidre et al. 2012).

A total of 234 hunter recoveries (tissue samples) were obtained from the catch of polar bears in Nunavut and Greenland (1993-2013). The hunter recovery program was instituted in Greenland for the first time in 2012. In addition, 635 biopsies from physical MR operations to assess BB and KB subpopulations in the 1990s (*cf.* Taylor et al. 2005, 2008) were included in the recent MR assessment analyses (Chapter 2 and 5).

In summary – All field operations were conducted as planned and were very successful. The number of biopsies obtained from the BB subpopulation was substantially higher than expected. All handling in NW Greenland and Kane Basin of individual polar bears in connection with deployment of satellite radios were made without any complications. The general pubic and local communities were informed about the operations and local polar bear hunters participated on several of the flights to obtain biopsies or immobilize polar bears.

Analyses

Wildlife Genetics International (Nelson, British Columbia, Canada) analyzed all genetic samples (9 nuclear markers).

Satellite telemetry data (habitat analyses) were analyzed under leadership of the Greenland Institute of Natural Resources and Dr. Kristin Laidre.

Analyses of ice metrics were conducted by Dr. Harry Stern at the Polar Science Center (University of Washington, USA) in collaboration with Laidre.

Analyses of the genetic MR recapture data were conducted at Department of Fisheries, Wildlife and Conservation Biology (University of Minnesota) under the leadership of postdoctoral research associate Dr. Seth Stapleton, and his supervisor, Professor Todd Arnold. Salary for Dr. Stapleton´s postdoctoral fellowship was provided by the Government of Nunavut, Environment Canada, and GINR. Close cooperation on MR analyses was conducted with outside expert Dr. Eric V. Regehr (US Fish and Wildlife Service).

Analyses of field observations of polar bear body condition were made by Nunavut under leadership of Dr. Stephen Atkinson.

Population genetic analyses to determine the demographic identities of Baffin Bay and Kane Basin subpopulation were made by Dr. Liselotte Wesley Andersen at Department of Bioscience (University of Århus, Denmark) based on the results of analyses of nine nuclear markers used in the genetic MR study.

All SWG members participated in various phases of analyses. External experts (Appendix A) also participated in the analyses and in three face-to-face progress meetings of the SWG held at the Polar Science Center, University of Washington.

Timeline of analyses and reporting

The SWG originally proposed an October 2014 deadline for submission of a final report to the JC (SWG 2011). However, due to the time required for completion of sample processing in the genetic laboratory this deadline could not be met. The delay was a consequence of the largely successful field work, which exceeded expectations in terms of the quality and number of biopsy samples. As the mark-recapture modeling could not begin until the laboratory analyses were completed and the final datasets compiled, a new deadline for a final report of 30 April 2015 was proposed by the SWG (SWG 2014). However, further unanticipated delays occurred related to availability of historical samples and a final comprehensive dataset was not available until mid-December 2015. This delay put the analyses approximately 6 months behind schedule (SWG letter to JC).

In October 2015, the SWG held a 2nd face-to-face meeting in Seattle, Washington. At the meeting, preliminary results from analyses on abundance and vital rates, population genetic structure, range use, distribution, seasonal movements, and trends in sea-ice habitat were presented and evaluated. Near-final results of the analyses of range use, distribution, seasonal movements, and trends in sea-ice habitat and genetic analyses were also presented. However, the presentation of preliminary results from the mark-recapture modeling revealed that a considerable amount of additional work was still required from collaborators at University of Minnesota (SWG 2015). This work included, among other things, more in-depth error checking and a more detailed exploration of data from the MR studies in the 1990s in order to assess bias and potentially detect trends in abundance. This required revising the basic structure of the population models, developing and running model simulations, and validating the final model results before a final report could be completed.

Following this meeting the SWG and collaborators held 13 teleconferences between November 2015 and April 2016 where progress in the MR modeling of abundance in BB and KB was discussed and evaluated. A third face-to-face meeting was held in February 2016 to evaluate the revised modeling results.

After detailed discussions of the results of the MR assessment of the BB and KB subpopulations the SWG identified items for further analyses by University of Minnesota before final results could be sent to external scientific review (SWG 2016). In May 2016, results of the mark-recapture assessments were sent to Dr. Gary White (Professor Emeritus, Colorado State University) for a courtesy, external peer-review. Dr. White is a world-expert in mark-recapture population analyses. Dr. White gave the analyses a positive review and approved the analytical methods used and their results.

In summary the analyses of sea ice, movement and habitat use were conducted according to the original time plan. However, unanticipated delays in getting a final genetic dataset and complications related to the modeling of the genetic data resulted in delays in preparing the final report to the JC.

1.7. Literature Cited

Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations,* 9 pp.

Atkinson, S., E. W. Born, and K. Laidre. 2011. A research plan for re-assessment of the Baffin Bay and Kane Basin polar bear sub-populations. Unpublished report submitted to the Canada-Greenland Joint Commission on Polar Bear, 28 pp.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E.W., C. Sonne, and R. Dietz 2010. Research on polar bears in Greenland 2005-2009, Pages 135-148 *in* M. E. Obbard, G. W. Thiemann, E. Peacock and T. DeBruyn, editors. Proceedings of 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2010a. Request for review of survey methods. Letter from the Canada-Greenland Joint Commission on Polar Bear to the Scientific Working Group to the Joint Commission on Polar Bear, 26 October 2010.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2010b. Report of the 2nd meeting of the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 3 pp plus appendices.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2011. Greenland Canada Nunavut JC - Monitoring recommendations. E-mail from Basile van Havre to the JC and SWG, 2 March 2011.

Laake, J. 2010. Unpublished report from J. Laake, National Marine Mammal Laboratory, Seattle, to project collaborators: Government of Nunavut, Greenland Institute of Natural Resources, Qikiqtarjuaq HTO, the University of Minnesota, and the University of Washington, 7 pp.

Laidre, K.L., E.W. Born, E. Gurarie, Ø. Wiig, R. Dietz, and H. Stern 2012. Females roam while males patrol: Divergence in breeding season movements of pack-ice polar bears (*Ursus maritimus*). Proceedings of the Royal Society B: Biological Science (London) doi.10.1098/rspb.2012.2371

Lee, J., and M. Taylor. 1994. Aspects of the polar bear harvest in the Northwest Territories, Canada. International Conference on Bear Research and Management 9:237–243.

Lønstrup, J. 2006. Polar bear management in Greenland. Pages 133–134 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Lunn, N. J., M. Branigan, L. Carpenter, J. Justus, D. Hedman, D. Larsen, S. Lefort, R. Maraj, M. E. Obbard, E. Peacock, and F. Pokiak. 2010. Polar bear management in Canada, 2005-2008. Pages 87–113 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1995. Summary of polar bear population status 1993. Pages 19–24 *in* Ø. Wiig, E. W. Born, and G. W. Garner, editors. Polar Bears: Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014. http://pbsg.npolar.no/en/status/status-table.html, Accessed 18 June 2016.

Peacock, E., A. E. Derocher, G. W. Thiemann, and I. Stirling. 2011. Conservation and management of Canada’s polar bears (*Ursus maritimus*) in a changing Arctic. Canadian Journal of Zoology 89:371–385.

Prestrud, P., and I. Stirling. 1994. The International Polar Bear Agreement and the current status of polar bear conservation. Aquatic Mammals 20:113–124.

Stapleton, S. 2010. Polar bear monitoring and aerial survey assessment in Sirmilik National Park: Final Project Report. Unpublished report to Parks Canada, Mittimatilik Hunters and Trappers Organization, Government of Nunavut, University of Minnesota, and WWF-Canada, 11 pp.

Stapleton, S., T. Arnold, S. Atkinson, and D. Garshelis. 2010. Baffin Bay polar bear aerial survey: Pilot results and research alternatives. Unpublished report to project collaborators: Government of Nunavut, Greenland Institute of Natural Resources, Qikiqtarjuaq HTO, and the University of Minnesota, 23 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2010. Report of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 18 pp

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2014. Outcome of January 2014 Scientific Working Group Meeting, Letter to the Canada-Greenland Joint Commission on Polar Bear, 24 January 2014, 3 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2015. Report of Working Meeting of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 5-7 October 2015, Polar Science Center, University of Washington, Seattle, Washington, USA, 13 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Report of Working Meeting of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 15-19 February 2016, Polar Science Center, University of Washington, Seattle, Washington, USA, 45 pp.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987. Modelling the sustainable harvest of female polar bears. Journal of Wildlife Management 51:811–820.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

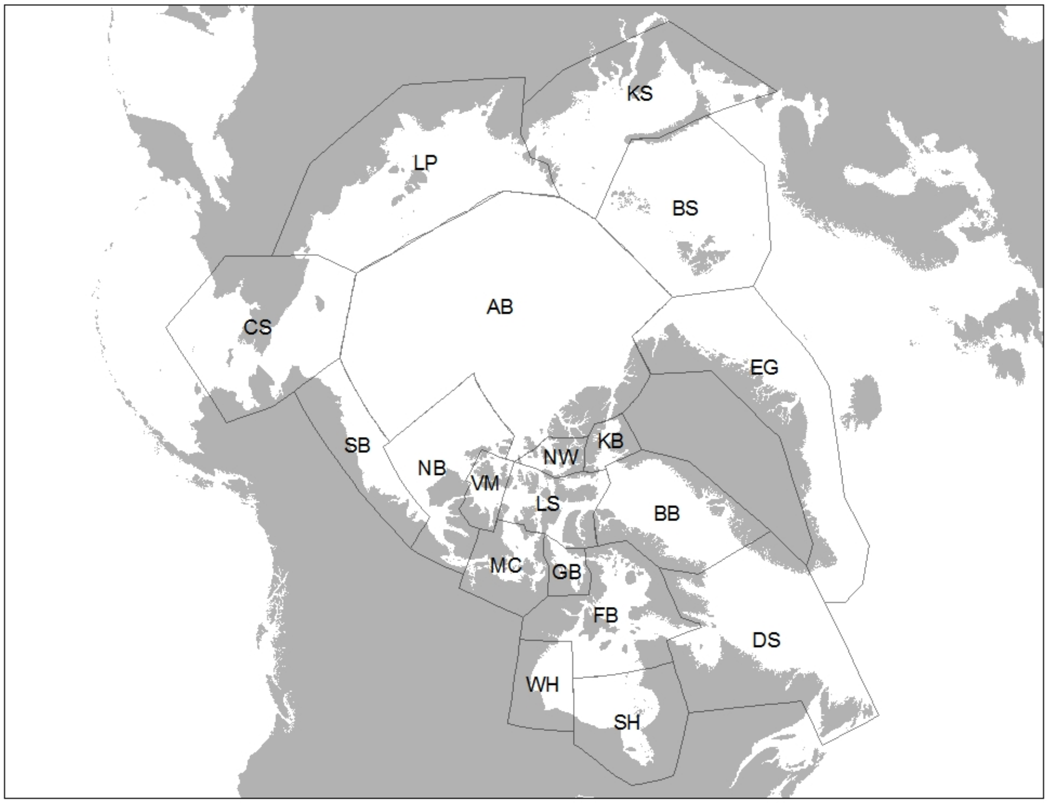


Figure 1.1. Map of the circumpolar Arctic showing the 19 subpopulations of polar bears, including Baffin Bay (BB) and Kane Basin (KB). Polar Bear subpopulations: Arctic Basin (AB), Baffin Bay, Barents Sea (BS), Chukchi Sea (CS), Davis Strait (DS), East Greenland (EG), Foxe Basin (FB), Gulf of Boothia (GB), Kane Basin, Kara Sea (KS), Lancaster Sound (LS), Laptev Sea, M'Clintock Channel (MC), Northern Beaufort Sea (NB), Norwegian Bay (NW), Southern Beaufort Sea (SB), Southern Hudson Bay (SB), Viscount Melville Sound (VM), and Western Hudson Bay (WH).

CHAPTER 2

SUBPOPULATION DELINEATION OF BAFFIN BAY AND KANE BASIN

KEY FINDINGS

Baffin Bay (BB)

- The 1990s and 2000s satellite telemetry data are comparable for analysis; approximately 92% of adult females collared in West Greenland in spring during the 2000s use the same area on Baffin Island where adult females were captured and collared in fall in the 1990s.
- There was a significant reduction in the size of the 2000s BB 95% bivariate normal kernel range (i.e. a measure of the area used by collared bears) in all months and seasons when compared to the range in the 1990s. The most marked reduction was a 60% decline in subpopulation range size summer.
- The overlap of the 1990s and 2000s BB ranges was < 50% in all months, reflecting both a contraction and shift of the BB subpopulation range in the 2000s. These shifts are related to the loss of annual sea ice and changes in breakup timing, contracting the range of the BB subpopulation and shifting the distribution of BB polar bears northward in all seasons. The BB subpopulation is still distributed within the current management boundaries.
- There were significant shifts north in the median subpopulation latitude in all seasons in BB across decades.
- Bears in the 2000s were significantly less likely to leave BB than in the 1990s ($p<0.001$), with reductions in the number of bears moving into Davis Strait and Lancaster Sound.
- Genetic analyses using eight polymorphic microsatellites in recent samples (2009-2014) had low genetic resolution. BB and KB polar bears could not be differentiated genetically. Also in accordance with previous genetic studies, BB-KB polar bears were found to be genetically different from polar bears in Lancaster Sound and Davis Strait.
- During the genetic mark-recapture studies in BB and KB (2011-2014) there were very low levels of recapture or harvest recovery of bears outside their subpopulation of origin.
- Satellite telemetry demonstrates that emigration from BB has been significantly reduced since the 1990s, largely due to reduced sea-ice extent in winter and absence of sea ice in summer. This suggests the BB subpopulation has become more discrete, with less exchange between it and other subpopulations.

Kane Basin (KB)

- KB mean 95% kernel ranges have generally expanded since the 1990s. The increase in range use in the 2000s occurs in all seasons, however is statistically significant only in summer (June-September), where ranges doubled between the 1990s and the 2000s. This range expansion is likely related to changes in sea ice, as KB is trending towards the characteristics of an annual ice ecoregion (like BB) where ice melts out almost completely each summer.
- There is still considerable seasonal overlap in KB subpopulation ranges for bears in the 1990s and 2000s (50-98% overlap over decades), suggesting that bears generally continue to use the same areas of KB.
- There were significant northward shifts in KB median latitude of polar bear locations in the 2000s in spring and summer, although these shifts were smaller than observed in BB. Variability in the range of latitudes has increased; bears in the 2000s use a broader range of latitudes. There has been no change during winter.
- These distribution patterns did not change with a sensitivity analysis in which bears captured in eastern KB were excluded from the 2000s data, to match the distribution of captures in the 1990s. This suggests that our key findings for KB were not influenced by the distribution of capture locations.
- **BB and KB**

Overall, our findings based on satellite telemetry, movement of marked bears and genetics suggest that the existing boundaries of the BB and KB subpopulations continue to be relevant for harvest management purposes and population monitoring.

2.1. Subpopulation Delineation and Status Background

Cluster analyses of movement data from satellite-collared bears (Taylor et al. 2001), genetic analyses (e.g., Paetkau et al. 1999), and recaptures and harvest recoveries of marked (tagged) bears (Taylor and Lee 1995, Taylor et al. 2001) are among the methods that have been used to evaluate and delineate the boundaries of the Baffin Bay (BB) and Kane Basin (KB) subpopulations. For BB, genetic analyses suggest a lack of genetic differentiation of BB from the adjacent KB subpopulation to the north, but a significant genetic difference from polar bears in the Davis Strait (DS) subpopulation (Paetkau et al. 1999). Analyses of satellite collar data and tag returns suggest that some interchange occurs among BB and adjacent subpopulations including DS, Lancaster Sound (LS) and KB (e.g., Taylor et al. 2001). However, for the purposes of management, BB is considered a distinct demographic unit, and its dynamics are largely driven by intrinsic rates of reproduction and mortality rather than exchange with neighboring subpopulations.

For KB, analyses of satellite collar data and tag returns suggest partial closure. However, the discreteness of this subpopulation from neighbouring units has been questioned, in part due to the lack of genetic differentiation from surrounding subpopulations and the potential for immigration from these much larger subpopulations to significantly influence demographic processes in a source-sink dynamic (Taylor et al. 2008). Particularly notable interchange occurs with BB and LS. The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a barrier between KB and BB – LS.

Sea ice in BB and KB has decreased markedly during the last 3 decades (Stirling and Parkinson 2006, Peacock et al. 2012, Laidre et al. 2015, Chapter 4 and 9), with earlier spring break up and later fall formation. The extent to which these trends in sea ice will affect the distribution and boundaries of these subpopulations remains uncertain and requires updated information. In particular, there is a need to re-evaluate these boundaries when undertaking studies to estimate abundance and vital rates to ensure sampling remains consistent with the distribution of the biological subpopulations.

The delineation and status of the BB subpopulation has been documented and updated by the IUCN/SSC Polar Bear Specialist Group (PBSG 1995, 1998, 2002, 2006, 2010, and http://pbsg.npolar.no/en/index.html) and annually by the Canadian Polar Bear Technical

Committee (unpublished). Based on the movements of adult females with satellite radio-collars and the recapture or harvest of tagged animals, the BB subpopulation of polar bears is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island, Nunavut, Canada to the west (Taylor et al. 2001) (Figure 1.1). A relatively distinct southern boundary at Cape Dyer, Baffin Island, and the entrance to Kangerlussuaq/ Søndre Strømfjord, Greenland, is evident from the movements of tagged bears (Stirling et al. 1980) and from adult female polar bears monitored by satellite telemetry (Taylor et al. 2001).

Analysis using microsatellites revealed significant genetic variation between polar bears in BB and neighboring DS, but not between polar bears in BB and neighboring KB (Paetkau et al. 1999). However, bears from BB-KB differed genetically from polar bears in the neighboring LS and DS subpopulations (Paetkau et al. 1999, Peacock et al. 2015, Malenfant et al. 2016). The original separation of the polar bears subpopulations into the two management units Kane Basin and Baffin Bay was based on studies of movement of polar bears with satellite collars in the 1990s and the fact that the North Water Polynya in northern Baffin Bay, to a certain extent, acts as a barrier to movement between BB and KB (PBSG 1998, Taylor et al. 2001). The samples included in the genetic study by Paetkau et al. (1999) were collected in the early 1990s. A study by Peacock et al. (2015) was based on more recent samples (i.e., BB: mainly 2006-2008; DS: 2005-2007; LS: mainly 2008), however the samples from KB were from the 1990s. Malenfant et al. (2016) conducted a re-analysis and relied upon subsets of the same data in Peacock et al. (2015). Hence, the KB-samples in Malenfant et al. (2016) were also not temporally congruent with samples from BB, LS and DS.

Based on the movements of KB adult females with satellite collars and the recapture or harvest of tagged animals, the boundaries of the KB subpopulation include the North Water

Polynya (to the south of KB), and Greenland and Ellesmere Island to the west, north, and east and the southern part of Kennedy Channel to the north (Taylor et al. 2001; Figure 1.1).

There have been no new scientific studies in BB or KB to update information on subpopulation delineation since 1997. We used new data from genetics, satellite telemetry and information on the movements of bears amongst BB, KB and surrounding subpopulations as detected by the recapture or harvest of marked individuals to assess the current BB boundaries. Our objective was to use these data to support other lines of evidence relating to the delineation of these subpopulations.

2.2. Methods

Genetics

In this study, a large recent sample of polar bear tissues was obtained from the subsistence catch and from scientific biopsies in BB, KB, LS and DS (2009-2014). This was obtained primarily for the genetic mark-recapture as reported in Chapters 5 and 10. Data from these analyses were therefore available for an updated examination of population genetics in BB and KB. The majority of the samples were collected between 2011 and 2014. Hence, in contrast to previous studies, the samples collected in connection with the genetic mark-recapture study were both more recent and also temporally congruent (Table 2.1).

The genetic analyses reported here used the same eight polymorphic microsatellite markers as in MR analyses to explore to what extent polar bears in KB, BB, LS and DS differed genetically (e.g., PBSG 2010). The decision to make these analyses was made *post hoc* and genetic analyses were not a part of the original study plan (see Chapter 1). Hence, these analyses were based on a platform of opportunity (i.e., the samples had been analyzed genetically and the data were available), fully recognizing that more nuclear markers would result in a higher resolution.

Genetic data from a total of 1,364 individual polar bears from the BB, KB, LS and DS subpopulations were available for the genetic analyses. A subset of 402 polar bears sampled during the winter-spring season (Nov-June) in 2012-2014 represented all four subpopulations under consideration. This subset consisted of the spring biopsy samples from Kane Basin, and winter-spring harvest samples from BB, LS and DS. We considered the samples to have been collected within a narrow time frame (i.e., “temporally congruent”) that would exclude major displacement of groups of polar bears among BB, KB, LS and DS during the sampling period (Table 2.1).

The population genetic analyses were conducted by Department of Bioscience (Aarhus University, Denmark) using standardized analytical tools and methods (ADEGENET package, Jombart 2008; ARLEQUIN Version 3.5.1, Excoffier and Lischer 2010; BA3-3.0.3, Wilson and Rannala 2003; DAPC, Jombart et al. 2010; FSTAT, Goudet 1995; GENECLASS2, Piry et al. 2004; GENELAND, Guillot et al. 2005, Guillot 2008; STRUCTURE, Prichard et al. 2000).

Satellite Telemetry Studies of Movements and Range Use

Polar bears were tagged in NW Greenland on the fast and pack ice between mid-March and mid-April 2009-2013 in Baffin Bay and mid-April to early May 2012-2013 in Kane Basin. Field operations were based out of coastal settlements in West Greenland or research stations on Ellesmere Island. Searches for bears in BB occurred out to a maximum distance of 150 km from the coast. Areas with consolidated glacier ice at the glacial terminus were also searched in both BB and KB. A total of 91 individual bears were tagged with satellite transmitters in BB and 34 individual bears in KB. Of these, 38 were AFs collared in BB and 20 collared AFs in KB (Table 2.2, Figures 2.1 and 2.2). In the 2000s, both sexes and subadults were tagged. These data were combined with a historical data set of captured and tracked from both areas in the 1990s. In BB, 1991-1995 43 collars were deployed on AFs, with the majority deployed during the ice free season in fall on Baffin Island (n=11 deployed in spring in NWG, 9 of which transmitted for sufficient time to be included in the analyses) (Table 2.3). In KB, 1992-1994, 12 collars were deployed on AFs on the west side of KB in the fjords and fast ice. Only bears captured within the BB or KB subpopulation boundaries as defined by PBSG (2010) were included in the comparative analysis.

Polar bears at all sites were darted and immobilized from an Ecureuil AS350 (BB) or Bell 206 LR (KB) helicopter and handled according to procedures described in Stirling et al. (1989). Standard body measurements (standard length and axillary girth) were taken and total body mass was estimated using the approach of Derocher and Wiig (2002). Field estimates of age and reproductive status were recorded.

Adult female polar bears in the 2000s in both areas were fitted with TAW-4610H satellite radio collars (Telonics, Mesa, Arizona, USA). Satellite collars provided information on geographic location, internal transmitter temperature, and activity. Collars were programmed to transmit during one six-hour period each day on 4-day intervals. In the 2000s, all adult male polar bears and subadults of both sexes were fitted with SPOT-5 S227 satellite radio transmitters (Wildlife Computers, Redmond, Washington, USA) attached to the ear similar to numbered plastic ear tags used in conventional studies (right ear). The SPOT-5 transmitters weighed 32 g and 60 g with attachment system (Born et al. 2010). Ear transmitters were duty cycled to extend battery life, with most tags transmitting on 4-day intervals and others on daily intervals. Satellite tags transmitted around noon local time each day and were programmed so that several locations were received per transmission day.

Data filtering and sub-sampling – Data on locations and transmitter status from all polar bears were collected via the Argos Location Service Plus system (Toulouse, France). Location qualities are assigned by ARGOS to each position, with location qualities of 0–3 estimated to have errors of 1.5 km or less and those categorized as 'A,' 'B,' or 'Z' have no predicted accuracy. Unrealistic and poor quality locations were removed using a speed and angle filter in R version 2.13.2 (R Development Core Team 2012) using the package 'argosfilter' (Freitas et al. 2008). Positions exceeding a maximum between location travel velocity (10 km/h based on previous movement studies of polar bears, Laidre et al. 2013) and angle (measured from the track between three successive locations; set to the default) were removed by the filtering algorithm. The resulting locations for each bear were next reduced to a single position per day to reduce autocorrelation bias, standardize temporal sampling, and address the effects of variable duty cycling among the tags. To obtain a daily position for each tag, the first, best quality location within the period of peak satellite passage was selected. Daily positions, after filtering and optimal daily position selection, only consisted of ARGOS qualities 1–3. Distances between successive daily positions were calculated as the great circle route and used to compute minimum daily displacements.

As a result of variable experimental objectives in both subpopulations and decades, different duty cycles were used for tags in an effort to extend battery life or gather information from specific time periods. The 1990s collars were programmed to transmit on varying and intermittent intervals, ranging from 1 to 6 days, while the 2000s collars were all on a 4-day cycle. We sub-sampled the 1990s data and created a strict 4-, 5- or 6- day interval time series for each individual to best match the 2000s data. This ensured that the impact of serial autocorrelation was consistent.

Captured polar bears were classified as independent adult male (AM), adult female (AF), subadult male (SM), subadult female (SF), or dependent cubs (cub of year COY, yearling YRL or 2-year old 2YR). We also classified the habitat type where the bear was first located and captured into three categories– pack ice (open or loose ice with leads and cracks), fast ice (sea ice attached to land with no open water), glacier ice (consolidated glacier ice at glacier fronts), as well as captures on land.

Data were divided into seasons: Spring (March – July, which included the peak of sea-ice coverage and initiation of sea-ice break-up), Summer (August – October, which included the end of break-up and the on-land period) and Winter (November – February, which included the freeze-up period and time when bears went back out on the sea ice). All denning periods were identified (maternity and shelter dens) (Escajeda 2016) and removed from RSF models. RSF models were only conducted on AFs to enable comparison with the 1990s. Bears with < 3 locations were removed from analyses as this was likely due to transmitter failure immediately after capture. Ages of polar bears were provisionally estimated in the field and later confirmed more accurately from counting of cementum growth layers of a pre-molar extracted during capture following methods in Calvert and Ramsay (1998). Adult females were defined as ≥ 5 years old and adult males as ≥ 6 years old. Age group status as determined in the field was verified based on tooth analyses.

We assigned each polar bear location to its respective subpopulation boundary (starting point or origin as well as the boundary where the bear was located at each time step) based on the boundaries recognized by the IUCN Polar Bear Specialist Group (PBSG 2010).

Basis for comparison of the 1990s and 2000s BB satellite telemetry data – Polar bears within the subpopulation boundaries of BB are treated as belonging to a single management unit. The bears range over the entire Baffin Bay with the majority of bears spending the summer on Baffin Island during the ice-free period. There were differences in capture locations between decades (Figure 2.1), and although these captures were all within the bounds of the BB subpopulation per PBSG (2010) they were captured in different areas and seasons. Thus we conducted analyses to ensure that the movement of the bears was comparable across periods. In the 1990s, n=43 adult females were collared between 1991 and 1995. Approximately 72% of these were captured on land on Baffin Island in fall. In the 2000s, all n=38 adult females were captured on the spring fast ice and pack ice between 2009 and 2013 in West Greenland.

We spatially bounded the 1990s fall capture region along the coast of Baffin Island and examined what fraction of bears collared in the 2000s (in spring) in West Greenland used the same area the following fall. We defined fall as any period between August and November. All West Greenland 2000s bears were considered “independent” because they were captured in spring. Some individuals that remained resident during all seasons on or close to glaciers in Melville Bay and bears with collars that failed to transmit for >2.5 months after spring capture (which occurred in mid-April) were excluded from the calculations.

Monthly and Seasonal Kernel Density Estimates – Using a fixed kernel density approach (Worton 1989), we estimated the geographic areas characterized by a high probability of use by satellite-radio tagged AF polar bears in BB and KB. Kernel density estimators provide a non-parametric probability of using a given point in space and are reliably used to define the utilization distribution, or home range, for marine and terrestrial wildlife (Kie et al. 2010).

We calculated Gaussian bivariate normal kernel density estimates for each subpopulation (BB, KB, and KB West), in each decade, and for each month (n=12) and season (n=3). Kernel Density Estimates (KDEs) were calculated using the "bkde2D" function in "KernSmooth" R package (Wand 1994, Wand et al. 1995). The sample size of tagged AF differed between the 1990s and 2000s (Table 2.2, Table 2.3). To account for any potential bias in the KDEs or the fraction of overlap between decades due to differing numbers of AFs we randomly sampled with replacement from the pool of AF bears in each the two decades (1990s and 2000s) so that the sample sizes of collared bears were equivalent during each time period (n=38 bears in BB in both decades, n=12 bears in KB in both decades). We sampled bears with replacement 1,000 times for each monthly and seasonal KDE and calculated the area of the 95% contour polygon (bounding 95% of the KDE surface volume). We produced a mean and bootstrapped standard error (SE) for monthly and seasonal home ranges, calculated the fraction of overlap for each time period, and statistically compared time periods. We used the ‘intersect’ tool in ArcGIS to identify overlapping home ranges between subpopulations. We also estimated the proportion of home range overlap between the 1990s and the 2000s (Fieberg and Kochanny 2005) based on the bootstrapped mean. The cell size was set to 6 km and bandwidth of 50 km (approximately 50% of the 4-day movement step of AFs in this study). Cell size determines the smoothness of the resulting prediction, but has minimal impact on kernel density estimation relative to bandwidth selection. The bandwidth controls the width of the estimated kernel thereby determining how much regional variation is emphasized.

Changes in Median Latitude – We also calculated median latitude and longitude values for the 1990s and 2000s using pooled data from all AFs by season. The north-south orientation of the BB and KB subpopulation ranges allowed for this comparison. We compared changes in median latitude and longitude across decades with t-tests at a significance level of α = 0.05. In the 1990s KB, all bears were caught along eastern Ellesmere Island (i.e., western KB) whereas bears were captured in both western and eastern KB in the 2000s. We performed a sensitivity analyses to evaluate this sampling difference with a subset of the KB bears captured in the 2000s. This subset included only those bears captured along the coast of Ellesmere Island in western KB. We compared them to the sample from the 1990s to examine any bias in comparison across decades.

Movements across Subpopulation Boundaries – We considered each 4 to 6 day AF bear trajectory as a single sample and calculated the elapsed time spent in the subpopulation region of origin (defined as where the bear was captured and tagged). Specifically, we calculated the number of days until each polar bear left its subpopulation region of origin and plotted the time-until-departure for each subpopulation and decade. Bears that never left their region of origin still contributed follow-up time, but their observation time was censored at time of last transmission. Statistical methods for censored event times were used to construct "survival" curves (Kaplan-Meier) to characterize the distribution of exit times from BB or KB and test for differences among different subpopulations/decades (log-rank test of equality) with α =0.05. We considered two time scales for departure: (1) a departure from the region of origin to be any length of time (4 days minimum) and (2) a departure from the region of origin that was at least 30 days long.

For bears that were observed to leave their subpopulation region of origin we summarized which subpopulation they departed to and the month of departure. We contrasted departures between BB and KB and across decades. As there were two capture seasons in the

1990s (spring in Melville Bay and summer on Baffin Island) we also tested whether capture season impacted the time until departure from the region of origin.

Recaptures and Harvest Recoveries of Marked Bears

Bears included in this study were marked in springtime (April – May) or fall (August – October) during three periods; 1991-1997, 2005-2007 and 2009-2014. From 1991 to 1997, 881 and 141 bears were captured and physically marked with ear tags and lip tattoos as part of studies in BB and KB, respectively (Taylor et al. 2005, 2008). In preparation for genetic mark-recapture studies in BB and KB commencing 2011, tissue samples collected from these bears were subsequently genotyped in 2011 with two exceptions: 1) bears that were known to have been harvested between 1991 and 2011 and 2) bears whose known or estimated age would have been greater than 35 years in 2011. In total, this dataset consisted of 650 individuals marked in the 1990s that would have been ≤ 35 years old and had not been harvested by the time genetic sampling began in 2011. Samples for genotyping were available for 635 of the 650 individuals. Genotyping followed methods described elsewhere in this report (Chapter 5).

From 2005 to 2007, 1518 bears in DS were physically marked (and subsequently genotyped) as part of a mark-recapture study (Peacock et al. 2013). From 2011 to 2013, 1111 bears were biopsy darted along western BB (Canada) and genotyped. From 2009 to 2013, 143 bears were physically marked and genotyped or biopsy darted and genotyped in eastern BB (Greenland). From 2012 to 2014, 129 bears were physically marked and genotyped or biopsy darted and genotyped in KB.

Recapture or harvest recovery of physically or genetically marked individuals was detected by two means. Prior to 2011, when biopsy darting began, marked BB and KB individuals were identifiable by ear tags and lip tattoos. During this period, recaptures of marked individuals were recorded during physical capture sampling in BB, KB, and surrounding subpopulations. Harvest recoveries of marked bears were detected via hunter returns of ear tags and / or lip tattoos as part of the on-going harvest monitoring program across all subpopulations in Canada and Greenland (Peacock et al. 2012). From 2011 onwards, all marked individuals were genotyped and some were both physically marked and genotyped. Recapture or harvest recovery of marked individuals was detected by physical marks recorded during capture sampling and harvest monitoring or by matching the genotypes of marked bears to samples collected during capture sampling, biopsy darting or harvest monitoring.

Although recaptures and harvest recoveries of marked bears from BB and KB have been previously reported (Taylor and Lee 1995, Taylor et al. 2001, Peacock et al. 2012), we incorporated additional data to supplement and update these analyses. We focused on three areas. To facilitate interpretation of results from the recent genetic mark-recapture studies in BB and KB (Chapters 5 and 10) we examined recaptures and recoveries of individuals marked between 2011 and 2014 to test the assumption that bears marked during these studies remained within their original subpopulations over the sampling period. For bears marked in the 1990s in BB and KB, we examined the number and sex of individuals recovered in the harvest up to 2014 to assess the degree of movement amongst subpopulations over the long term and to test the hypothesis that these movements are sex biased. Finally, we examined 3 sequential, intensive mark-recapture sampling sessions in BB (1991-97 and 2011-13) and neighboring DS (2005-07), to assess movements across the BB-DS boundary.

We incorporated capture, recapture or recovery events for which the location of bears was recorded at time of observation using a GPS. Sex was determined by physical examination or genotyping (Chapter 5 and 11). For bears marked between 1991 and 2008, age was determined based on previous capture history, known age (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). For bears “marked” from 2009 to 2013, the age of most individuals (i.e., those remotely biopsied along eastern Baffin Island, in contrast to those immobilized and handled in NW Greenland and in Kane Basin) could not be determined since they were not physically handled and teeth were not available for aging. Instead, age class (cub-of-the-year, yearling, sub-adult [ages 2 – 4], and adult) was estimated from the air at a range of 3-7 meters above ground. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B). For all capture-recapture analyses, recaptures of an individual within the same season and year of capture were excluded. Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

2.3. Results

Overall Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all of Bylot Island (66.2°N to 73.8°N) in Nunavut/ Canada, as well as parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al. 2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south (Figure 1.1, Figure 2.1). Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A small number of bears remain on land in northwestern Greenland throughout the ice-retreat period (Born 1995, Born et al. 2011, this study).

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the influx of polar pack ice from Arctic Basin, and reaches a minimum in late summer.

Genetics

The multi-locus F_{ST} estimates were generally low, although statistically significant. The F_{ST} analysis suggested a separation into three groups (1) BB-KB, (2) LS and (3) DS (L.W. Andersen, Institute of Bioscience, Århus University, Denmark, personal communication). This subdivision is in accordance with Paetkau et al. (1999), Peacock et al. (2015) and Malenfant et al. (2016). The analyses based on the 8 markers showed that the genetic resolution was low. Bayesian clustering methods had difficulties identifying more closely related groups without using spatial information (i.e., site of sampling or harvest).

Using the spatial information (i.e., GPS positions of individual samples sites) implemented in GENELAND a group structure was indicated where adult females and males sampled during the winter-spring season were divided in an eastern and a western group corresponding to BB-KB and LS, and a northern and a southern group corresponding to BB-KB and DS (L.W. Andersen, Institute of Bioscience, Århus University, Denmark, personal communication) (Table 2.1).

General Movements from Telemetry

Baffin Bay – All but n=12 of the 43 bears collared in BB between 1991 and 1995 were captured on land on Baffin Island in fall. The remaining 12 were captured on the sea ice in spring (n=3 off Baffin Island and n=9 in Melville Bay, West Greenland). The 1990s collared bears transmitted through 1997 (Figure 2.3). Overall in the 1990s 72% were captured in fall on land inside the sampled area on BI (i.e., the area in which biopsies were collected for the genetic MR assessment), 6% were captured on the sea ice off the Baffin Island coast, and 21% were captured on the sea ice in Melville Bay, West Greenland. In the 2000s, all n=38 adult females (100%) were collared between 2009-2013 on the spring sea ice in West Greenland and transmitting through April 2015 (Figure 2.1).

Adult females were tracked between ~6 months and four years. Adult male tracking durations were shorter (~2 months) due to ear attachments. Telemetry data were truncated at 01

April 2015 so that analyses could be completed on time for reporting deadlines, therefore locations from collared bears beyond that date were not included here.

Adult females were captured and collared in West Greenland in all reproductive states (alone, as mating pairs, with COYs= cub of the year, yearlings, and 2-year old cubs) (Figure 2.2). General patterns of movements show broad coverage of the BB region during the tracking period in each year (Figures 2.4-2.11). There was a series of collar failures in 2012 that resulted from poor release mechanisms (Figure 2.7), therefore tracking data from bears captured in that year had shorter durations. There was some exchange between BB and KB. Most bears followed the general pattern of moving from West Greenland to the coast of Baffin Island in the fall. However a new pattern was observed that was not detected in the 1990s satellite telemetry data. Of the 38 adult females collared in BB in the 2000s, n=7 (18.4%) remained in the glacier ice of Melville Bay for the entire tracking durations. For all but one of these bears, where collar failure occurred early, this period was between 1 and 2 years (with some bears still transmitting from Melville Bay after April 2015). Bears that remained in Melville Bay were captured in all years when the area was sampled (2011-2013). No bears showed this behavior in the 1990s, and only one of the tracked bears made a single excursion onto the fast ice in Melville Bay (Taylor et al. 2001 figure 3, and Figure 2.3). Of note, this comparison may include some bias because only one bear was captured and given a satellite collar at glacier fronts in Melville Bay in the 1990s (Taylor et al. 2001 figure 1) even though the area was searched. Unfortunately the satellite collar on this bear only transmitted for one day. Additionally n=2 of 38 bears captured in the 2000s moved back and forth between Melville Bay and KB but never visited Baffin Island. Furthermore, another two individuals in the 2000s moved between Melville Bay and KB, but in subsequent years also moved to Baffin Island.

Kane Basin – In the 1990s, 12 adult females were captured and collared in KB along the coast of Ellesmere Island (Figure 2.11). In the 2000s, 20 adult females were captured and collared in spring along both Ellesmere Island and the western coast of Greenland (Figure 2.1). Bears captured in 2012 remained in KB subpopulation boundaries for the entire tracking period (through the period when collars were removed, April 2014) whereas with bears captured in 2013 there was more variability in movements, with individuals moving into Baffin Bay and Jones Sound. No bears collared in KB moved to Melville Bay though contact with two individuals was lost close to the West Greenland coast (Figures 2.12-2.14). One bear collared in KB in 2013 moved along the northern coast of Greenland in the Arctic Basin to Severnaya Zemlya and then to Franz Josef Land (Figure 2.14 inset). This individual was considered an outlier and excluded from further analyses. Adult males captured and satellite tagged with ear tags in KB (Figure 2.15) remained in KB during their tracking periods, which were less than 30 days.

Basis for Comparison of the 1990s and 2000s BB Satellite Telemetry Data

The analysis examined if polar bears collared on the sea ice in spring in the 2000s (in West Greenland) represented the same subpopulation sampled in fall on Baffin Island (Figure 2.1). This was conducted to ensure the comparison of movements and habitat use between the two telemetry sets was valid. Overall 92% of the bears collared in West Greenland in the 2000s entered the 1990s capture region on Baffin Island in fall, providing a solid basis for comparing the movements of polar bears captured in spring and in fall as defined by a polygon encompassing all the 1990s fall captures. Bears collared in West Greenland used nearly the entire Baffin Island coastline in fall and were spread over the whole capture region used in the

1990s, with the exception of the area around Cape Dyer. These bears thus also were representative of the bears biopsied for the genetic MR in the 2000s.

Subpopulation KDEs and Overlap of Ranges

Baffin Bay – In BB in the 1990s, 95% kernel ranges for polar bears were similar in winter, spring and summer, ranging from approximately 700,000 – 900,000 km^2. In the 2000s, seasonal ranges were significantly smaller in all seasons, ranging from 255,000 to 729,000 km^2. When the 1990s ranges were compared to the 2000s, there was a significant reduction in the size of 95% seasonal ranges in all seasons (reduction of 20% area in winter and 30% in spring), with the most marked reduction being a 60% decline in area of the summer range, reduced from a mean of 716,767 km^2 (SE 57,850) to a mean of 255,992 km^2 (SE 28,627), based on 1,000 bootstrap samples standardized for sample size (Table 2.4, Figure 2.16).

In all months except May, the home range sizes for the 2000s were significantly smaller than those in the 1990s (Figure 2.17). In some spring months 95% monthly ranges were reduced by 30% in late winter and spring (February, March and April) and in summer months by 50% (August and September), with a difference of about 325,000-375,000 km^2 (Table 2.4, Figure 2.18). Home range sizes were calculated with and without the resident Melville Bay bears to test for changes in results with inclusion of resident bears. There were minimal changes to the home range sizes and no differences in the significance of results with or without these residents.

Kane Basin – Seasonally, 95% ranges in KB in the 1990s fell between 89,000 and 203,000 km^2 whereas in the 2000s 95% ranges were between 152,000 and 192,000 km^2 (Table 2.4, Figures 2.19-2.21). In most months there was a reverse pattern to that in BB, where 95% range sizes in the 2000s were generally larger than those in the 1990s. However the pattern of increased range size was statistically significant only for June-September (p<0.001). Increases in summer range size were detected both when including all KB 2000s bears and those collared in west KB only.

Overlap of Ranges between Decades

The percent overlap quantifies the similarity in space between the home ranges across decades. In BB, the overlap of 1990s and 2000s home ranges was lowest in the summer months (July through October), ranging from 21-34% overlap. It was higher in spring and mid-winter, reaching a maximum of 61% in June, however largely was < 50% in all months. The low overlap values reflected the significant contraction of the range in most months in BB when comparing the two decades.

In KB there was a higher level of overlap in ranges between decades, ranging from about 50% in November and December to 98% in September. In general, KB bears used similar areas between decades, however in 2000s these areas were larger. When the 1990s KB bears were compared with the KB bears tagged in the western portion of the area (KB-West 2000s), the overlap was similar. In general, bears captured and tagged in KB west did not differ from the full sample of KB bears in the 2000s, though those in the western portion of KB were more likely to move into BB.

Changes in Median Latitude from Telemetry

We detected significant shifts north in the median subpopulation latitude in all seasons in BB. In winter, the shift was nearly 5 degrees north (median 68.8 in the 1990s, median 73.5 in the 2000s, p<0.001), while in the spring the shift north was about 3 degrees (p<0.001) and in summer < 1 degree (p<0.001) (Table 2.5, Figure 2.22). In KB there were also significant shifts north in median latitude in spring and summer but they were smaller, ~ 1 to 1.5 degrees (Table 2.5, Figure 2.23). There was no change in median latitude in KB during winter (p=0.07). We tested both all KB 2000s bears and those in KB-west and there were no differences in significance.

Movements across Subpopulation Boundaries from Telemetry

We examined the trajectories and departure from regions of origin BB (n=43 AFs in 1990s, n=38 AFs in 2000s) and KB (n=12 AFs in 1990s, n=20 AFs in 2000s) for bears tracked up to 700 days in duration. Of bears captured in BB in the 1990s when departure of any length (minimum 4 days due to duty cycling) was considered, there was movement to two subpopuations: Davis Strait (n=14 bears) and Lancaster Sound (n=12 bears). In the 2000s, bears moved to three subpopulations: Davis Strait (n=3), Lancaster Sound (n=3) and Kane Basin (n=5) (Table 2.6). Overall polar bears in the 2000s were significantly less likely to leave BB than in the 1990s (p<0.001) (Table 2.7), with large reductions in the number of bears moving into Davis Strait and Lancaster Sound as observed in the 1990s. For example at 100 days after capture, approximately 58% of bears were remaining in BB in the 1990s, whereas at the same time step in the 2000s, over 90% of the collared bears were still in the area. At 300 days after capture, approximately 60% of the bears had departed from BB in the 1990s where at the same time step in the 2000s about <10% had left. Capture season in BB in the 1990s (spring vs. summer) was not a factor in timing of departure from BB, there was no difference in time until departure for the two decades (p=0.562) for either length of departure.

In KB there were no significant differences in percentage of bears departing over the decades studied (p=0.339). In Kane Basin, 2 AF polar bears in the 1990s moved out of the area (one to Arctic Basin and one to Baffin Bay) (Table 2.6), and movement in the 2000s was slightly higher and included departure to Lancaster Sound, but was not significantly different (p=0.351).

The timing of departures over the annual cycle varied significantly in BB (Figure 2.24, Figure 2.25). In the 1990s, bears were significantly more likely to depart from BB to Davis Strait during the winter or early spring months when the area was ice covered (November and April). Bears that departed to Lancaster Sound left BB in late spring and summer (June-September) to move on to remnants of sea ice in the archipelago area (p=0.002, Fisher's exact test). This pattern was weakly present in the 2000s although sample sizes were very low.

The number of boundary crossings by individual bears is shown in Table 2.9. Of the bears that departed from BB (crossed the BB subpopulation border) in the 1990s when departures of all time steps were considered, 5 of 26 did not return to BB (19%) at any point during the tracking period. Of the bears that departed in the 2000s, 3 of 11 (27%) did not return to BB.

When the threshold for departure was longer (>30 days) patterns were similar. Bears were significantly more likely to depart from BB in the 1990s and departures primarily occurred to Davis Strait and Lancaster Sound (Tables 2.10 and 2.11). For AF departures based on the 30 day time step, fewer bears left the subpopulations of origin (Figure 2.25). There were similar differences in proportion of bears departing BB; significantly more bears departed in the 1990s vs. the 2000s (p=0.009). The timing of departures was similar but sample sizes were smaller (Table 2.12). Of the bears that departed from BB (crossed the BB border) in the 1990s when departures >30 days were considered, 56% of the 1990s BB bears did not depart from the BB boundaries, where as 79% of the BB bears in the 2000s did not depart from the BB boundaries (Table 2.13).

Recaptures and Recoveries during Genetic Mark-Recapture Studies, 2011-14

From 2011 to 2014, the harvest of polar bears in BB, KB and all surrounding subpopulations was monitored genetically by genotyping of tissue samples from harvested bears. Of the bears genetically marked in BB during the recent mark-recapture study, 2011-2013, 85 individuals were recovered in the harvest, as detected by genotyping (Table 2.14). Of these individuals, 84 (99%) were recovered in the BB harvest and the other bear was harvested in DS. For bears marked in KB from 2012-2014, no individuals were recovered in the harvest in KB or surrounding subpopulations. Also during this period of harvest monitoring, an additional 12 bears marked in the 1990s in BB and KB were recovered in the harvest. Eleven were marked in BB, 1 was marked in KB. All of these ‘old’ 1990s marks were recovered in BB.

Of the bears genetically marked in BB during 2011-2013, there were 207 recapture events during mark-recapture sampling in BB and KB from 2011 to 2014, including >1 recapture of some individuals (Table 2.15). Two hundred and six (> 99%) of these recapture events occurred in BB and one BB mark was recaptured in KB. For bears marked in KB during 2012-2014, there were 29 recapture events during mark-recapture sampling in BB and KB from 2011 to 2014. Twenty-eight (> 96%) of these recaptures occurred in KB and one KB mark was recaptured in BB.

During this recent period of mark-recapture sampling there were 66 recapture events of bears marked in either BB or KB during 1991-1997. Fifty-four of these events were of individuals marked in BB. Twelve were of KB individuals. For the 54 ‘old’ BB marks, all were recaptured in BB. Of the ‘old’ KB marks, 8 and 4 were recaptured in KB and BB, respectively.

For bears that were biopsied on more one occasion during the ice-free seasons in BB between 2011 and 2013 we examined the straight line distances between mark and recapture locations. Intervals between capture and recapture were obviously constrained to a ranged from 1-2 years. Median distances between mark-recapture varied between age and sex classes (Figure 2.26; Kruskal-Wallis, H = 18.62, $P = 0.001$). Bears initially marked as yearlings were recaptured at a location farther from their initial capture site than COY and adult females. Adult males also had larger mark-recapture distances than COY (Figures 2.26 and 2.27).

Long-term Harvest Recoveries of BB and KB Marks

Of the 881 bears marked in BB from 1991 to 1997, 181 individuals were recovered in the harvest in Canada and Greenland between 1991 and 2014 (Table 2.16). Eighty-three percent of recoveries occurred within BB. Recoveries of marked bears outside BB tended to be male-biased (3.29 males per female) relative to recoveries within BB (1.85 males per female), but this tendency was not statistically significant (Fisher’s Exact test, $P = 0.483$).

Of the 141 bears marked in KB from 1991 to 1997, 21 individuals were recovered in the harvest in Canada and Greenland between 1991 and 2014. Forty-eight percent of these recoveries were within KB. There was no significant sex bias in recoveries within versus outside KB (Fisher’s Exact test, $P = 0.656$).

Movement of Bears between BB and DS

One hundred and fifty-five bears marked in BB during 1991 – 1997 were recovered in the harvest up to 2005, including 11 recoveries in DS. From 2005 to 2007, a total of 2,128 bears comprising 1,518 unique individuals were captured in DS. There were 13 recapture events of bears previously marked in BB recaptured during this sampling in DS.

From 2009 to 2013, 1,253 unique individuals (1,623 capture events) were recorded in BB. Sixteen (1%) of these individuals were originally marked in DS. In sum, from a total of 2,771 bears marked in either BB or DS during 2005 – 2013, we detected 29 instances (ca. 1%) where marked bears moved from one subpopulation to the other.

The straight line distance of these 29 inter-subpopulation movements was independent of the capture-recapture interval which ranged from 4 to 15 years (Table 2.15, Figure 2.28). Seventy percent of these individuals were originally captured and marked within 100 km of the boundary between BB and DS (Table 2.16, Figure 2.29). In comparison to other bears marked in these subpopulations, individuals that made inter-subpopulation movements were found significantly closer to the boundary ($\chi^2 = 169.48$, d.f. = 11, $p < 0.001$). Despite extensive marking of bears throughout the range of both subpopulations the recorded inter-poplation movements were clustered near the boundary (Figure 2.1).

2.4. Discussion

Genetics

The results of the analyses of migration direction and detection of first generation migrants were subtle and influenced by the lack of differentiation between Baffin Bay and Kane Basin. However the overall migration direction appeared to flow from Lancaster Sound and

Davis Strait to Baffin Bay and Kane Basin. More markers would be needed to verify this (L.W. Andersen, Institute of Bioscience, Århus University, Denmark, personal communication).

Although the genetic resolution was low due to use of only 8 nuclear markers pre-selected for other purposes these analyses support findings in previous studies (Paetkau et al. 1999, Peaccok et al. 2015, Malenfant et al. 2016) that (1) Kane Basin and Baffin Bay polar bears are not genetically different, and that (2) polar bears from Kane Basin-Baffin Bay are genetically different from polar bears from Lancaster Sound and Davis Strait.

Movements and Telemetry

BB bears home ranges have become significantly smaller, by a third to a half between the 1990s and 2000s. There is <50% overlap between areas used by bears in the 90s and what they use in the 2000s. This is consistent even when resident bears that remained in Melville Bay glacier fronts were excluded (removing any possible bias in comparisons across decades). There is a significant shift in median latitude northward in the 2000s for the core subpopulation range when compared to the 1990s. Overall analyses also indicate that BB bears are significantly less likely to depart from the BB subpopulation boundaries in the 2000s than the 1990s.

During the fall, most bears in BB are distributed on land or on the remaining pack-ice along the coastline of Baffin Island and associated islands (Ferguson et al. 1998, 2001, Taylor et al. 2001). Information from polar bear hunters with extensive experience from the Melville Bay area (Born et al. 2011) and miscellaneous observations (e.g., Taylor et al. 2001) suggest that some bears may also be present along the NW Greenland coast during the open-water season, in particular in Melville Bay. However, in this study for the first time we documented resident bears in Melville Bay via satellite telemetry. These bears remained in NW Greenland year-round, in some cases >2 years within a series of glacier fronts. Satellite telemetry studies of bear movements in the 1990s failed to demonstrate this behavior (Taylor et al. 2001) however only one adult female bear was collared on the fast ice in Melville Bay (at a glacier front) and the satellite radio only transmitted for one day (E.W. Born, pers. comm.).

KB bears home ranges have become overall larger, in some cases significantly so and twice the size in summer. There is more overlap in home ranges for KB between the 90s and 00s, between about 50 and 98%, probably because the subpopulation area is smaller and because the ice loss has not been as extreme (or bears can use more of the southernmost habitat still). However, a likely explanation may be that the sea ice in the Kane Basin region has become less consolidated (e.g., Born et al. 2011) and now resembles that of the annual sea-ice ecoregion (rather than the archipelago region with year-round consolidated sea ice, cf Amstrup et al. 2008). This shift in habitat likely forces bears to have larger-scale movements and large home ranges, following well with that found by Ferguson et al. (2001). There were also significant shifts north in median latitude in KB in spring and summer from the 90s, though the variability has increased in the range of latitudes used and no change during winter.

Changes in Polar Bear Densities in Melville Bay, Northwest Greenland

Another result of the comparison between ranges and movements is the apparent increased use of Melville Bay, northwest Greenland by BB bears. The telemetry data collected in the 2000s indicate an increased use in both spring and summer (Figure 2.16) when compared to the 1990s (see also Figure 2.3). Of 43 polar bears that were tracked by use of satellite telemetry during the 1990s remarkably few made excursions from offshore BB pack ice onto landfast ice in NW Greenland (Figure 2.3, Taylor et al. 2001: Figure 3, p. 696; Born and Dietz 2009: Figure 18, p. 82). This is in contrast to heavy use of the area in the 2000s (Figure 2.9).

During the 1990s spring sampling 1992 and 1993, 55 hours of active searching was spent on the fast ice, glacier fronts, and active offshore pack ice between *ca*. 74° N and *ca*. 76° N in northwest Greenland, including offshore areas 100-150 km from the coast (i.e., ferry time excluded; Born unpublished data). In both years, there was little sign of bear activity on the fast ice in Melville Bay and along glacier fronts even though ~ 25% of the active search time was flown over fast ice and along glacier fronts. Only four of 36 bears were tagged in fast ice and glacier fronts the 1990s (Born et al. 1992, Rosing-Asvid 1993). In contrast, in 2011-2013, 85 active search hours were concentrated in the Melville Bay area north of the settlement Kullorsuaq. In each year there were signs of recent polar bear activity on the fast ice and along glacier fronts, including the consolidated pack ice in Melville Bay. Approximately 82% of captures occurred in fast ice or consolidated pack ice, and of those 25% were along glacier fronts. Differences between 1992-1993 and 2011-2013 in the allocation of sampling effort and number of polar bears captured reflect an increased density of polar bears using landfast ice and glacial fronts in Melville Bay in the 2010s. The apparent change in densities in spring time Melville Bay is worth noting and is consistent with information obtained from experienced polar bear hunters that there has been an increased occurrence of polar bears in nearshore areas in NW Greenland (Born et al. 2011).

Recoveries from Marked Bears

Use of tag recoveries or recaptures is a relatively coarse means of assessing subpopulation closure for the purpose of mark-recapture analyses, but it can facilitate the inclusion of data from large numbers of individuals, relative to satellite telemetry. The probability of detecting the movements of individuals between subpopulations depends on numerous factors, including the number of marks deployed, the intensity of sampling effort following marking, and the intensity of harvest. Additionally, the detection of movement amongst subpopulations does not provide a means of quantifying rates of permanent emigration or immigration. Nevertheless, data on recaptures and harvest recoveries of marked bears provide a supplemental line of evidence to support subpopulation delineations based on more detailed methods such as telemetry data analyses (Taylor et al. 2001) and genetics (Paetkau et al. 1999, Peacock et al. 2015).

Short-term Movements

During the recent genetic mark-recapture studies in BB and KB (2011-2014), we documented very low levels of recapture or harvest recovery of bears outside their subpopulation of origin. It should be noted that bears marked in the final year of these studies had a zero probability of recapture or recovery because harvest monitoring and biopsy darting were not extended beyond the last year of marking. However, bears marked in the first two years of these studies were available for recapture or recovery, subject to rates of natural mortality. The total number of bears marked in years 1 and 2 was equivalent to ~34% and ~25% of the estimated subpopulation size in BB and KB, respectively (Chapters 5 and 10). Despite marking a large proportion of the subpopulation, instances of emigration were ≤ 1% of the recaptures and recoveries of BB marks. Similarly in KB, documented cases of emigration comprised < 4% of recaptures. Amongst these findings, rates of harvest recovery provide a more complete picture of movement amongst subpopulations because harvest was monitored genetically in BB, KB and all surrounding subpopulations throughout the study period. In contrast, capture effort only took place in BB and KB during the study.

We acknowledge that this work comprised a relatively short window of time, especially since harvest monitoring and biopsy darting were not extended beyond the last year of marking. However, our findings suggest that the existing subpopulation boundaries continue to be relevant for harvest management purposes and subpopulation monitoring. Bears marked in BB or KB tended to remain within their respective subpopulations at least over the short term. These units can be surveyed by means of mark-recapture or aerial survey with a reasonable degree of confidence in the assumption of closure during short-term studies.

Long-Term Harvest Recoveries

Over the period 1979-2009, Peacock et al. (2012) found that amongst harvest recoveries of bears marked in BB, 82% were recovered in BB versus other subpopulations. Using a subset of the same data plus newer recoveries, we obtained a similar level of recovery (83%) within BB. These findings imply that bears exhibit a reasonably high degree of long-term fidelity to this geographically defined unit, which is consistent with estimates of site fidelity derived from mark-recapture analyses (Chapter 5, but note that the site fidelity parameter pertains to the study area and not necessarily the subpopulation). In contrast, less than half of the KB marks deployed in the 1990s have been recovered in the harvest in KB, although the vast majority of recoveries occurred during the 1990s and early 2000s. This suggests that fidelity to KB may be lower than observed in BB. However, we note that the harvest rate in KB is an order of magnitude smaller than surrounding subpopulations; mean annual harvests in KB and BB have been 9.3 and 163 bears, respectively, over the period 1992-2014 (Chapter 8 in this report). This difference in sampling effort (specifically, the small sample sizes in KB) may contribute to the observed differences in fidelity between the subpopulations.

Sex and Age Class

Sex and age differences in movements and distribution patterns are well documented for many mammal species (e.g., Mabry et al. 2013). Until recently, studies of polar bear movements and the delineation of subpopulations have relied primarily on data collected from satellite collared adult females (e.g., Taylor et al. 2001, Amstrup et al. 2004, Parks et al. 2006, Cherry et al. 2013). Even now, studies of the movements of sub-adults and adult male polar bears are limited by available technology. Satellite transmitters attached to sub-adult and adult males have only lasted 4-5 months, limiting research to studies of seasonal movements and habitat use (Amstrup et al. 2001, Laidre et al. 2013). The extent to which subpopulation boundaries as currently defined reflect the long-term distribution of sub-adults and male polar bears therefore remains uncertain. This is an important area requiring on-going investigation because of the implications for defining biological populations or harvest management units that can be accurately monitored by methods such as mark-recapture.

Using data on the distance between capture and recapture locations, Taylor et al. (2001) found a tendency for sub-adults to exhibit longer-range movements than adults in BB and KB. Similar data from our study (2011-2013) suggest that mark-recapture distances were greater for bears initially marked as yearlings, relative to adult females and COY. This may reflect a degree of dispersal amongst young bears away from their maternal (natal) range post-weaning. This is a common phenomenon among mammals and is often male biased (Greenwood 1980), but sample sizes for yearlings in our study were insufficient to test for sex effects in mark-recapture distances. However, using measures of genetic relatedness (kin structure analysis), Zeyl et al. (2009) demonstrated a slight male bias in effective dispersal of polar bears. Nevertheless, fidelity to natal range was relatively strong in that study, suggesting that regardless of sex, dispersal of offspring is unlikely to be a major factor in determining the broad scale (subpopulation) structure of polar bears. With harvest recoveries, the sex ratio of marked BB bears harvested outside BB tended to be male biased relative to recoveries within the subpopulation but not significantly so. Collectively, these findings provide some support for the notion that subpopulation boundaries, delineated using data on the movements of satellite collared adult females are relevant to polar bears of all sex and age classes as also suggested by Taylor et al. (2001).

Movement between Baffin Bay and Davis Strait

The boundary between the BB and DS subpopulations is not delineated by a landmass or area of open water that creates an obvious barrier to the movement of polar bears. During the winter and spring, this boundary is spanned by both land fast and pack ice that provides an easy platform for bears to move between these subpopulations (Stirling et al. 1980). Nevertheless, bears in these two subpopulations have consistently shown distinct differences in studies of genetics (Paetkau et al. 1999, Peacock et al. 2015), movements (Taylor et al. 2001) and diet (Thiemann et al. 2008), suggesting that there is a real boundary between them. This boundary is likely the result of ocean current patterns caused by a submarine ridge between SE Baffin Island and Central West Greenland and associated differences in patterns of sea-ice formation and break-up in Baffin Bay and Davis Strait, respectively (Taylor et al. 2001).

Since 1990, there have been 3 periods of intensive mark-recapture study in BB and DS. These data provided an opportunity to examine movements between DS and BB. Additionally, DS has likely undergone a substantial increase in abundance since the 1970s and presently has one of the highest densities of polar bears amongst subpopulations (Peacock et al. 2013). Furthermore it may be speculated that because of a higher rate of sea-ice loss in Davis Strait compared to Baffin Bay during the last decades (e.g., Laidre et al. 2015) an increased number of polar bears may have immigrated to BB from DS. We were therefore keen to examine the hypothesis that this apparent expansion happening concomitantly with sea-ice loss in Davis Strait had resulted in the movement of bears from DS to BB, thereby influencing our estimate of abundance for BB.

The intensity of marking effort (i.e., number of unique individuals marked) in these subpopulations was equivalent to 41% (BB 1991-97), 70% (DS 2005-07) and 44% (BB 2011-2013) of estimated abundance at the time of marking. Despite this extensive marking of bears throughout the seasonal ranges of both subpopulations we detected very few cases of movement between DS and BB. Approximately 1% of bears sampled in DS were recaptures of bears marked in BB and vice versa. Our results support the notion that the boundary between these two subpopulations remains relatively strong and does not support the hypothesis that subpopulation expansion and sea-ice loss in Davis Strait have resulted in a large-scale northward movement of DS bears into BB, at least during the period from 2005 to 2013.

Bears that were documented to have moved between BB and DS tended to be those originally captured close to the management unit boundary (< 100 km). This clustering of inter-subpopulation movements around the boundary does not itself demonstrate the significance of the boundary as a barrier to movements. Instead it may reflect the high degree of fidelity that

BB and DS bears show to their seasonal on-shore range. A similar degree of fidelity has been previously documented in BB (Taylor et al. 2001) and other subpopulations (Stirling et al. 2004). Taylor et al. (2001) examined distances between capture and recapture locations for bears marked in BB during the 1990s finding that 59% were recaptured within 100 km of their original capture location. Amongst all BB bears marked between 2011 and 2013, 50 % (n = 166) of recaptures were within 100 km of initial capture location (GN unpublished data) suggesting that the tendency of local fidelity of BB bears has remained unchanged since the 1990s. Strong inter-annual fidelity to terrestrial habitat is further supported by our finding that distances between capture and recapture locations for bears that moved between DS and BB were independent of capture intervals ranging from 4 to 15 years. Bears appear to exhibit fidelity over long periods.

In conclusion, using data from satellite telemetry, recapture and / or harvest recovery of marked bears and genetic we found no evidence to suggest a change in the delineation of the BB and KB subpopulations. The boundaries of these subpopulations appear to be relevant from the stand point of mark-recapture or other forms of periodic survey and for harvest management purposes.

2.5. Literature Cited

Amstrup, S. C., G. M. Durner, T. L. McDonald, D. M. Mulcahy, and G. W. Garner. 2001. Comparing movement patterns of satellite-tagged male and female polar bears. Canadian Journal of Zoology 79:2147–2158.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E.W., C. Sonne, and R. Dietz 2010. Research on polar bears in Greenland 2005-2009, Pages 135-148 *in* M. E. Obbard, G. W. Thiemann, E. Peacock and T. DeBruyn, editors. Proceedings of 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Derocher, A. E., and Ø. Wiig. 2002. Postnatal growth in body length and mass of polar bears (*Ursus maritimus*) at Svalbard. Journal of Zoology 256:343–349.

Escajeda, E. D. 2016. Identifying shifts in maternity den phenology and habitat characteristics of polar bears (*Ursus maritimus*) in Baffin Bay and Kane Basin. MSc Thesis, University of Washington, Seattle, Washington, USA.

Excoffier, L., and H. E. L. Lischer. 2010. Arlequin suite ver 3.5: a new series of programs to perform population genetics analyses under Linux and Windows. Molecular Ecology Resources 10:564–567.

Ferguson, S. H., M. K. Taylor, E. W. Born, and F. Messier. 1998. Fractals, sea-ice landscape, and spatial patterns of polar bears. Journal of Biogeography 25:1081–1092.

Ferguson, S. H., M. K. Taylor, E. W. Born, A. Rosing-Asvid, and F. Messier. 2001. Activity and movement patterns of polar bears inhabiting consolidated versus active pack ice. Arctic 54:49–54.

Fieberg, J., and C. O. Kochanny. 2005. Quantifying home-range overlap: the importance of the utilization distribution. Journal of Wildlife Management 69:1346–1359.

Freitas, C., K. M. Kovacs, C. Lydersen, and R. Ims. 2008. A novel method for quantifying habitat selection and predicting habitat use. Journal of Applied Ecology 45:1213–1220.

Goudet, J. 1995. FSTAT (Version 1.2): a computer program to calculate F statistics. Journal of Heredity 86:485–486.

Greenwood, P. J. 1980. Mating systems, philopatry and dispersal in birds and mammals. Animal Behaviour 28:1140–1162.

Guillot, G. 2008. Inference of structure in subdivided populations at low levels of genetic differentiation. The correlated allele frequencies model revisited. Bioinformatics 24:2222-2228.

Guillot, G., F. Mortier, and A. Estoup. 2005. Geneland: a computer package for landscape genetics. Molecular Ecology Notes 5:712–715.

Jombart, T. 2008. *adegenet*: a R package for the multivariate analysis of genetic markers. Bioinformatics 24:1403–1405.

Jombart, T., S. Devillard, and F. Balloux. 2010. Discriminant analysis of principal components: a new method for the analysis of genetically structured populations. BMC Genetics 11:94.

Kie, J. G., J. Matthiopoulos, J. Fieberg, R. A. Powell, F. Cagnacci, M. S. Mitchell, J.-M. Gaillard, and P. R. Moorcroft. 2010. The home-range concept: are traditional estimators still relevant with modern telemetry technology? Philosophical Transactions of the Royal Society of London, Series B, Biological Sciences 365:2221–31.

Laidre, K. L., E. W. Born, E. Gurarie, Ø. Wiig, R. Dietz, and H. Stern. 2013. Females roam while males patrol: divergence in breeding season movements of pack-ice polar bears (*Ursus maritimus*). Proceedings of the Royal Society B: Biological Sciences 280: 20122371. doi:10.1098/rspb.2012.2371.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Mabry, K. E., E. L. Shelley, K. E. Davis, D. T. Blumstein, and D. H. van Vuren. 2013. Social mating system and sex-biased dispersal in mammals and birds: a phylogenetic analysis. PLoS ONE 8(3): e57980. doi:10.1371/journal.pone.0057980.

Malenfant, R. M., C. S. Davis, C. I. Cullingham, and D. W. Coltman. 2016. Circumpolar genetic structure and recent gene flow of polar bears: A reanalysis. PLoS ONE 11(3): e0148967. doi:10.1371/journal.pone.0148967.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

Parks, E. K., A. E. Derocher, and N. J. Lunn. 2006. Seasonal and annual movement patterns of polar bears on the sea ice of Hudson Bay. Canadian Journal of Zoology 84:1281–1294.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1995. Summary of polar bear population status 1993. Pages 19–24 *in* Ø. Wiig, E. W. Born, and G. W. Garner, editors. Polar Bears: Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2002. Status of the polar bear. Pages 21–35 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T.

D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Peacock, E., S. A. Sonsthagen, M. E. Obbard, A. Boltunov, E. V. Regehr, N. Ovsyanikov, J. Aars, S. Atkinson, G. K. Sage, A. G. Hope, E. Zeyl, L. Bachmann, D. Ehrich, K. T. Scribner, S. C. Amstrup, S. Belikov, E. W. Born, A. E., Derocher, I. Stirling, M. K. Taylor, Ø. Wiig, D. Paetkau, and S. L. Talbot. 2015. Implications of the circumpolar genetic structure of polar bears for their conservation in a rapidly warming Arctic. PLoS ONE 10:e112021. doi:10.1371/journal.pone.0112021.

Piry, S., A. Alapetite, J. M. Cornuet, D. Paetkau, L. Baudouin, and A. Estoup. 2004. GENECLASS2: A software for genetic assignment and first-generation migrant detection. Journal of Heredity 95:536–539.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., N. J. Lunn, J. Iacozza, C. Elliott, and M. Obbard. 2004. Polar bear distribution and abundance on the southwestern Hudson Bay coast during open water season, in relation to population trends and annual ice patterns. Arctic 57:15–26.

Stirling, I., C. Spencer, and D. Andriashek. 1989. Immobilization of polar bears (*Ursus maritimus*) with Telazol® in the Canadian Arctic. Journal of Wildlife Diseases 25:159–168.

Taylor, M. K., and J. Lee. 1995. Distribution and abundance of Canadian polar bear populations: a management perspective. Arctic 48:147–154.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating

Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Thiemann, G. W., S. J. Iverson, and I. Stirling. 2008. Polar bear diets and arctic marine food webs: Insights from fatty acid analysis. Ecological Monographs 78:591–613.

Wand, M. P. 1994. Fast Computation of Multivariate Kernel Estimators. Journal of Computational and Graphical Statistics 3:433–445.

Wand, M. P., and M. C. Jones. 1995. Kernel Smoothing. Chapman and Hall, London.

Wilson, G. A., and B. Rannala. 2003. Bayesian inference of recent migration rates using multilocus genotypes. Genetics 163:1177–1191.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164–168.

Zeyl, E., J. Aars, D. Ehrich, L. Bachmann, and Ø. Wiig. 2009. The mating system of polar bears: a genetic approach. Canadian Journal of Zoology 87:1195–1209.

Table 2.1. Polar bear samples (n = 1,364; sampling period: 2009-2014) from Baffin Bay (BB), Kane Basin (KB), Lancaster Sound (LS) and Davis Strait (DS) that were included in the population genetic analyses. The total sample was subdivided according to season, age category and sex for the analyses. A subset of 402 samples collected during winter and spring (2012-2014) represented all four subpopulations.

	BB	**KB**	**LS**	**DS**	**N**
	Biopsies and Harvest	**Biopsies**	**Harvest**	**Harvest**	**Total**
Period of sampling	2009-2014	2012-2014	2011-2013	2012-2013	
Total Sample	1051	99	142	72	1364
Winter-spring (WS)[1]	140	99	114	49	402
Winter-spring-adults (WSA)[2]	109	78	84	37	308
Winter-spring-subadults (WSS) [2]	31	21	30	12	94
Winter-spring-adults-females (WSAF)[3]	54	54	15	11	134
Winter-spring-adult-males (WSAM) [3]	55	24	69	26	174

[1] Winter defined as: November-February and spring defined as March-June

[2] A=adults and S=subadults

[3] F=females and M=males

Table 2.2. Sample sizes (number of individuals) polar bears captured and tagged with collars or ear tags in the 1990s and 2000s in BB and KB. In total 134 bears were tagged in BB and 46 bears were tagged in KB over two decades.

		AF	AM	SF	SM	2YR	TOTAL
1990s	BB	43					43
	KB	12					12
2000s	BB	38	30	4	6	13	91
	KB	20	9		5		34

Table 2.3. Breakdown of adult females (AF) collared in the 1990s and 2000s. Total n=113 AF bears over both decades and subpopulations.

		AF alone	AF+AM	AF+COY	AF+YRL	AF+2YR	Sum
1990s	BB	9		19	13	2	43
	KB	3		5	3	1	12
2000s	BB	10	2	6	12	8	38
	KB	5	1	7	3	4	20

Table 2.4. 95% mean kernel range sizes (in sq km) estimated from a bootstrap method for each of 38 bears in BB and 12 bears in KB by decade, month and season. The bootstrap was used to generate kernel range sizes based on equal sample sizes between decades (see Methods). Data are reported with bootstrapped SE of the mean in parentheses. Fraction of overlap is the 95% kernel probability area from the 1990s overlapped by the same in the 2000s. KB_West is reported only for bears tagged in western KB for direct comparison to the 1990s (where no bears were tagged in East KB). Data here include all bears in BB, including the resident bears in Melville Bay, though we also investigated bootstrap range size values without resident bears and significance remained.

	BB				**KB_All Bears**				**KB_West**	
	1990s mean (SE)	**2000s**	**P value comparing 95% area between decades**	**Overlap**	**1990s**	**2000s**	**P value comparing 95% area between decades**	**Overlap**	**2000s**	**Overlap (with KB ALL 1990s)**
January	684,409	558,957		0.40	86,556	163,892		0.77	175,730	0.77
	(60,692)	(56,594)	<0.001		(6,890)	(27,619)	0.007		(23,649)	
February	707,387	513,732		0.39	105,788	171,441		0.81	183,981	0.81
	(55,079)	(40,662)	0.005		(5,426)	(34,036)	0.057		(23,205)	
March	852,935	580,767		0.45	136,942	205,921		0.70	232,299	0.73
	(50,240)	(36,287)	<0.001		(15,859)	(49,119)	0.180		(38,259)	
April	795,859	506,739		0.46	131,963	183,184		0.73	180,913	0.69
	(45,652)	(26,529)	<0.001		(15,331)	(38,786)	0.219		(32,718)	
May	564,658	473,825		0.58	130,730	122,598		0.68	115,925	0.69
	(37,090)	(35,679)	0.078		(19,002)	(15,355)	0.741		(14,792)	
June	521,410	430,766		0.61	68,696	124,227		0.88	103,783	0.85
	(32,633)	(31,829)	0.047		(11,854)	(13,578)	<0.001		(9,942)	

July	536,992	376,891		0.52	54,681	130,518		0.91	112,655	0.88
	(32,000)	(30,062)	<0.001		(7,986)	(12,414)	<0.001		(13,872)	
August	596,411	216,881		0.21	68,649	119,251		0.90	106,314	0.92
	(44,692)	(18,958)	<0.001		(12,391)	(13,419)	<0.001		(11,694)	
September	551,395	226,427		0.26	74,368	131,558		0.98	109,697	0.84
	(48,280)	(21,538)	<0.001		(13,130)	(15,871)	<0.001		(14,187)	
October	459,230	276,198		0.34	99,855	153,820		0.91	132,177	0.75
	(43,932)	(29,264)	<0.001		(15,807)	(25,424)	0.072		(23,717)	
November	594,280	474,604		0.58	156,120	172,068		0.54	166,048	0.53
	(33,388)	(25,867)	<0.001		(22,728)	(23,943)	0.631		(20,773)	
December	702,091	524,787		0.52	150,392	143,969		0.54	137,782	0.53
	(35,173)	(38,123)	<0.001		(20,855)	(20,739)	0.826		(19,024)	
Winter	906,657	729,022		0.65	203,858	192,619		0.56	210,364	0.64
	(55,609)	(44,240)	0.012		(37,301)	(34,357)	0.826		(26,680)	
Spring	837,036	585,659		0.57	137,563	177,495		0.80	189,301	0.82
	(58,976)	(33,379)	<0.001		(17,600)	(37,516)	0.337		(37,470)	
Summer	716,676	255,992		0.24	89,066	152,747		0.97	141,118	0.94
	(57,850)	(28,627)	<0.001		(14,251)	(21,784)	0.014		(20,697)	

Table 2.5. Box plot statistics for median latitude (and interquartile range) for each season, subpopulation, and decade of polar bears tracked by use of satellite telemetry in BB and KB during the 1990s and 2000s. See Table 2.3 for sample sizes. In the case of BB 2000s, the resident bears in Melville Bay glacial fronts were not included

	Q1	Median	Q2	t statistic	df	p value
Winter						
BB 90s	66.90	68.88	72.13	-17.34	1683.4	p<0.001
BB 00s	69.76	72.01	74.90			
KB 90s	77.28	78.62	79.17	-1.86	173.54	p=0.07
KB 00s all	77.03	78.96	79.41			
Spring						
BB 90s	67.99	70.87	73.51	-18.18	1615.99	p<0.001
BB 00s	70.88	72.90	74.48			
KB 90s	77.63	77.91	79.13	-4.06	617.76	p<0.001
KB 00s all	77.25	79.06	79.42			
Summer						
BB 90s	67.26	70.29	72.96	-4.20	1758.07	p<0.001
BB 00s	69.51	70.68	71.60			
KB 90s	77.77	77.89	78.50	-5.47	840.41	p<0.001
KB 00s all	77.27	78.89	79.43			

Table 2.6. Movements of all polar bears (AF + AM) captured and tracked in the 1990s and 2000s from BB and KB. Movement to another subpopulation is enumerated if the bear departed for any length of time (4 days or greater).

			Movement to other subpopulation during tracking period				
Time period	**Subpopulation of origin**	**n**	**Arctic Basin**	**Baffin Bay**	**Davis Strait**	**Kane Basin**	**Lancaster Sound**
1990s	BB	43	0	0	14	0	12
2000s	BB	91	0	0	3	5	3
1990s	KB	12	1	1	0	0	0
2000s	KB	34	2	2	0	0	3

Table 2.7. Summary of observed and expected departures from region of origin (capture site) to any other subpopulation region in the 1990s and 2000s for AF polar bears for departures on all time steps. Log rank rest of equality conducted on each subpopulation comparing decades.

Subpopulation Decade	**n**	**Observed departures**	**Expected departures**	**(O-E)^2/E**	**(O-E)^2/V**
BB 1990s	43	26	15.7	6.72	12.1
BB 2000s	38	11	21.3	4.96	12.1
KB 1990s	12	2	3.29	0.509	0.871
KB 2000s	20	6	4.71	0.356	0.871

BB: $\chi^2 = 12.1$, df = 1, p = 0.000515

KB: $\chi^2 = 0.9$, df = 1, p = 0.351

Table 2.8. Timing of departures from Baffin Bay in the 1990s and 2000s for radio-collared adult female polar bears for departures of all time steps. Data are reported as number of AF bears departing to another subpopulation for the 1990s (with number of AF bears in 2000s in parentheses).

Month of departure from BB	**Davis Strait**	**Lancaster Sound**	**Kane Basin**
1	1 (0)	1 (0)	0 (3)
2	0 (0)	0 (0)	0 (0)
3	0 (0)	0 (0)	0 (0)
4	1 (1)	0 (0)	0 (0)
5	0 (0)	1 (1)	0 (0)
6	0 (0)	2 (1)	0 (0)
7	1 (0)	2 (0)	0 (1)
8	0 (0)	1 (0)	0 (0)
9	0 (0)	3 (0)	0 (0)
10	0 (0)	2 (1)	0 (0)
11	7 (0)	0 (0)	0 (0)
12	4 (2)	0 (0)	0 (1)

Table 2.9. Number of subpopulation boundary crossings made by individual AF bears in each subpopulation and decade for departures of all time steps.

	Number of subpopulation boundary crossings by individual AFs												
	0	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
BB 90s	17	3	7	1	3	1	3	0	3	3	1	0	1
BB 00s	27	3	4	0	2	0	0	1	1	0	0	0	0
KB 90s	10	0	1	0	0	0	0	0	0	1	0	0	0
KB 00s	14	1	2	0	1	1	0	0	1	0	0	0	0

Table 2.10. Movements of AF polar bears captured and tracked in the 1990s and 2000s from BB and KB. Movement to another subpopulation is enumerated if the bear departed for of >30 days or more.

			Movement to other subpopulation during tracking period				
Time period	**Subpopulation of origin**	**n**	**Arctic Basin**	**Baffin Bay**	**Davis Strait**	**Kane Basin**	**Lancaster Sound**
1990s	BB	43	0	0	14	0	12
2000s	BB	38	0	0	3	5	3
1990s	KB	12	1	1	0	0	0
2000s	KB	20	2	1	0	0	3

Table 2.11. Summary of observed and expected departures from region of origin (capture site) to any other subpopulation region in the 1990s and 2000s for radio-collared adult female polar bears for departures of >30 days or more. Log rank rest of equality conduced on each subpopulation comparing decades.

Subpopulation Decade	N	Observed departures	Expected departures	(O-E)^2/E	(O-E)^2/V
BB 1990s	43	19	12.4	3.580	6.760
BB 2000s	38	8	14.6	3.020	6.760
KB 1990s	12	1	2.06	0.532	0.914
KB 2000s	29	4	2.96	0.367	0.914

BB: $\chi^2 = 6.8$, df = 1, p = 0.009

KB: $\chi^2 = 0.9$, df = 1, p = 0.339

Table 2.12. Timing of departures from Baffin Bay in the 1990s and 2000s for radio-collared AF polar bears for departures of >30 days or more. Data are reported as number of AF bears departing to another subpopulation for the 1990s (number of AF bears departing in 2000s in parentheses).

Month of departure from BB	Davis Strait	Lancaster Sound	Kane Basin
1	1 (0)	1 (0)	0 (3)
2	0 (0)	0 (0)	0 (0)
3	0 (0)	0 (0)	0 (0)
4	1 (1)	0 (0)	0 (0)
5	0 (0)	1 (1)	0 (0)
6	0 (0)	2 (1)	0 (0)
7	1 (0)	2 (0)	0 (1)
8	0 (0)	1 (0)	0 (0)
9	0 (0)	3 (0)	0 (0)
10	0 (0)	2 (1)	0 (0)
11	7 (0)	0 (0)	0 (0)
12	4 (2)	0 (0)	0 (1)

Table 2.13. Number of subpopulation boundary crossings made by individual radio-collared adult female bears in each subpopulation and decade for departures of >30 days or more. 0 denotes the number of bears that never cross a boundary, so n=24 bears in BB 90s never departed from BB. Percentages shown as percent of total tagged bears.

	Number of subpopulation boundary crossings by individual AFs							
	0	**1**	**2**	**3**	**4**	**5**	**6**	**7**
BB 90s	24 (56%)	8 (19%)	6 (14%)	1 (2%)	1 (2%)	1 (2%)	1 (2%)	1 (2%)
BB 00s	30 (79%)	3 (8%)	2 (5%)	0	1 (3%)	1(3%)	0	1(3%)
KB 90s	11 (92%)	0	0	0	0	0	0	1 (8%)
KB 00s	16 (80%)	1 (5%)	0	1 (5%)	1(5%)	0	1(5%)	0

Table 2.14. Number and subpopulation location of harvest recoveries of bears marked genetically between 2011 and 2014. Data presented as number of individuals.

Subpopulation	Subpopulation Recovered[1]						
Marked	BB	KB	LS	FB	DS	NW	GB
BB	84	0	0	0	1	0	0
KB	0	0	0	0	0	0	0

[1] BB, Baffin Bay; KB, Kane Basin; LS, Lancaster Sound; FB, Foxe Basin; DS, Davis Strait; NW, Norwegian Bay; GB, Gulf of Boothia

Table 2.15. Recaptures of bears marked genetically between 2011 and 2014 in BB and KB. Data presented as number of recapture events. Some individuals were recaptured more than once. Excludes multiple recaptures of same individual within a season. Also includes COY that were not initially sampled but later seen as yearlings with mother and sampled.

Subpopulation Marked[1]	**Subpopulation Recaptured**	
	BB	**KB**
BB	206	1
KB	1	28

[1] BB, Baffin Bay; KB, Kane Basin

Table 2.16. Recoveries of polar bears tagged in Baffin Bay (1990-1997) in the harvest in Canada and Greenland, 1990 to 2014.

	Recovered in Harvest (1990-2014)		
Sex of Bear	**In Baffin Bay**	**Outside Baffin Bay**	**Total**
Female	53	7	60
Male	98	23	121

Figure 2.1. Distribution of capture locations of polar bears with satellite collars in BB and KB during the 1990s and the 2000s, respectively. See Table 2.2 for sample sizes. Note in 1990s bears in BB were mainly captured on Baffin Island in fall (12 were captured on sea ice in spring; 3 along Baffin Island and 9 in NW Greenland) whereas during the 2000s all bears were captured and tagged with satellite transmitters on the sea ice in NW Greenland in spring.

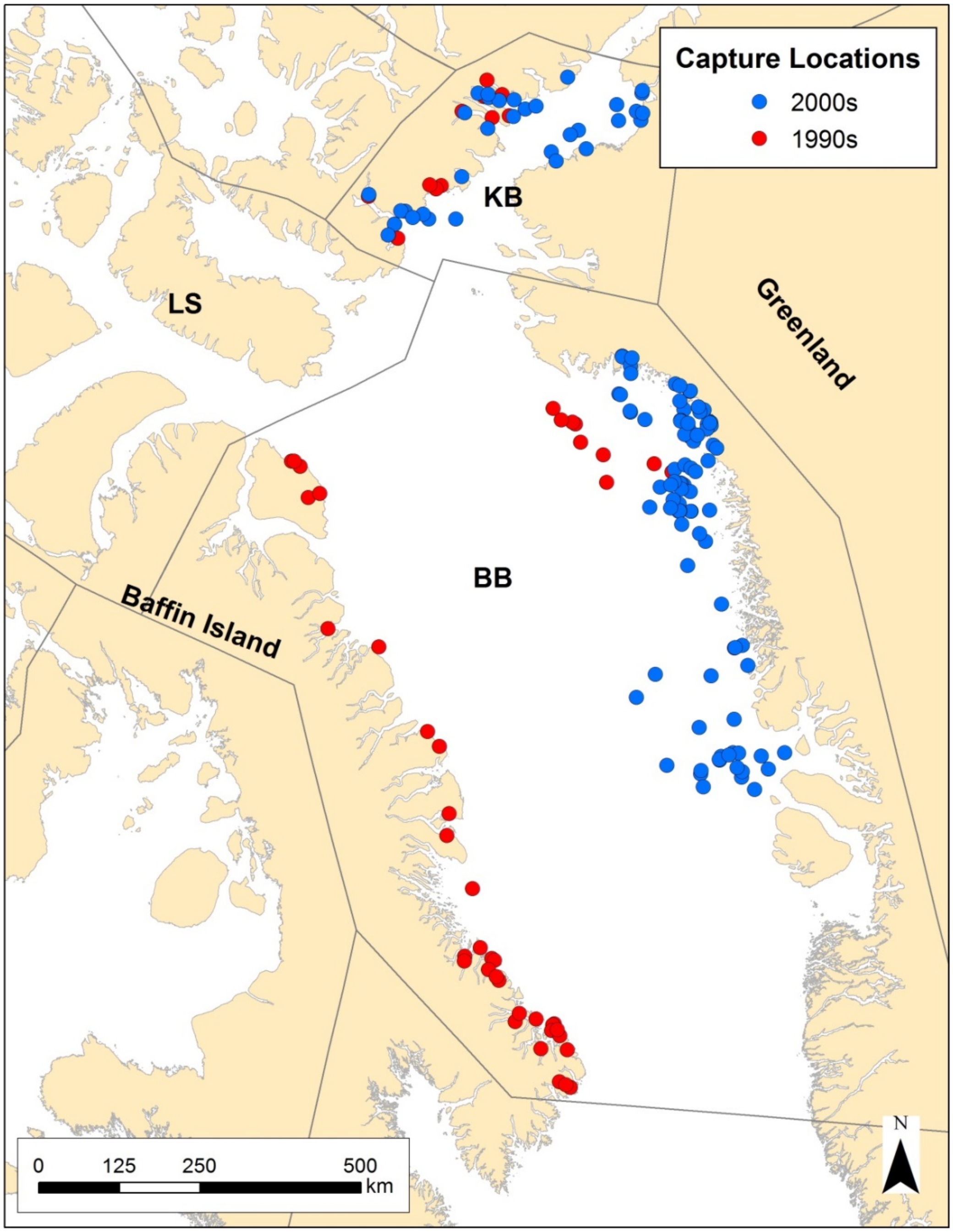


Figure 2.2. Distribution of ages and family groups of a total of 139 individual polar bears captured in spring in northwest Greenland, 2009-2013.

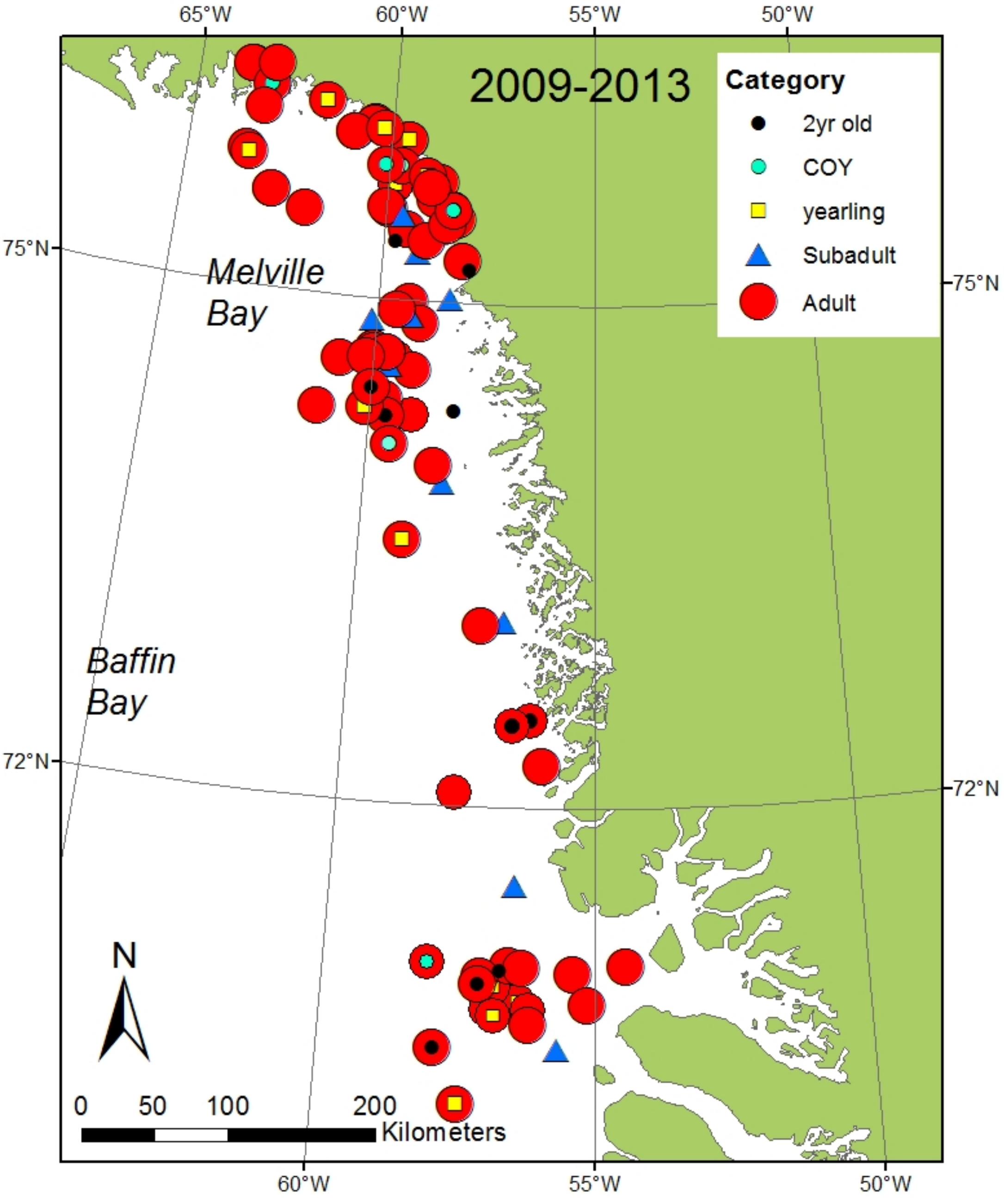


Figure 2.3. Tracklines from n=43 adult female polar bears satellite collared in the 1990s in BB.

Note the general absence of tracks on the fast ice in West and Northwest Greenland.

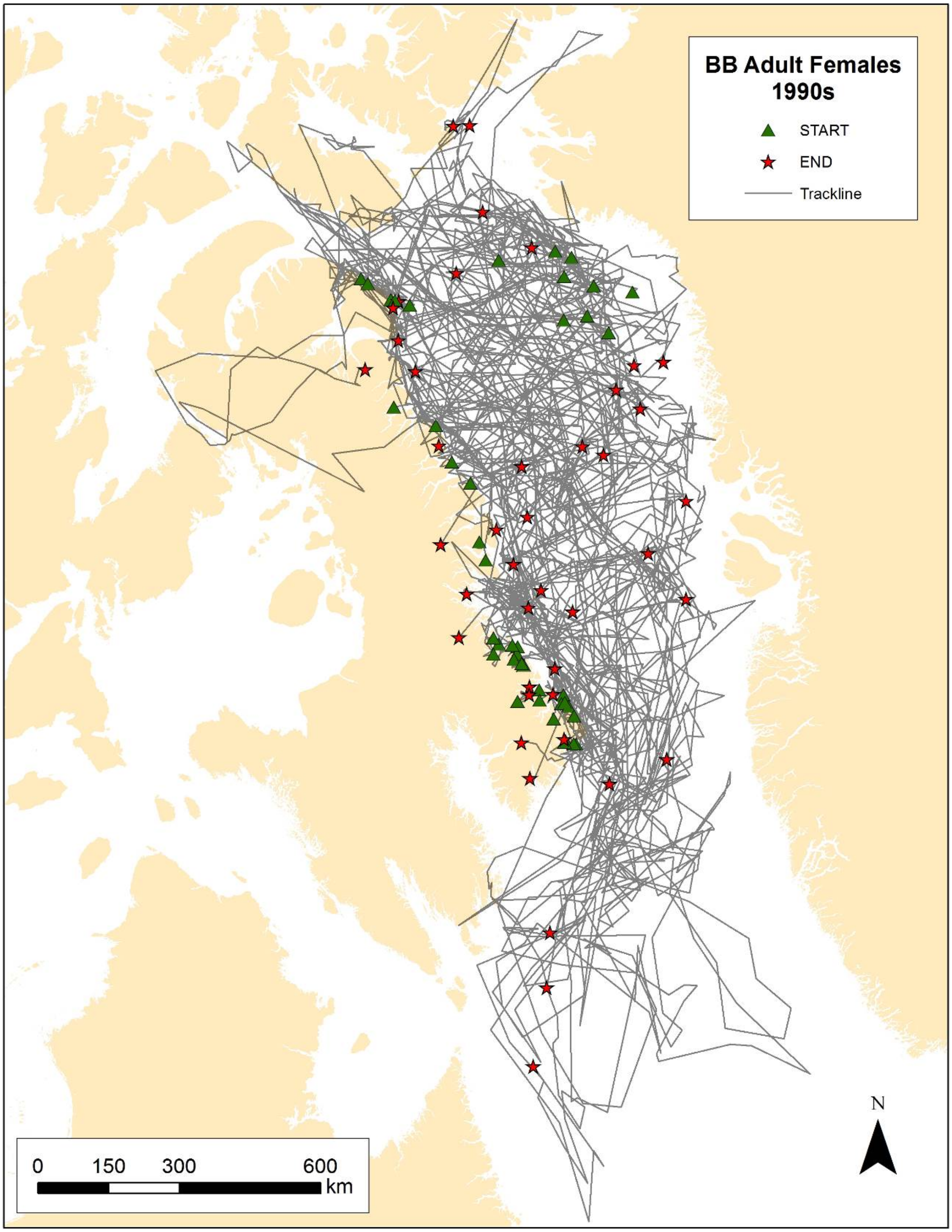


Figure 2.4. Tracklines from n=5 adult female polar bears satellite collared in 2009 in BB.

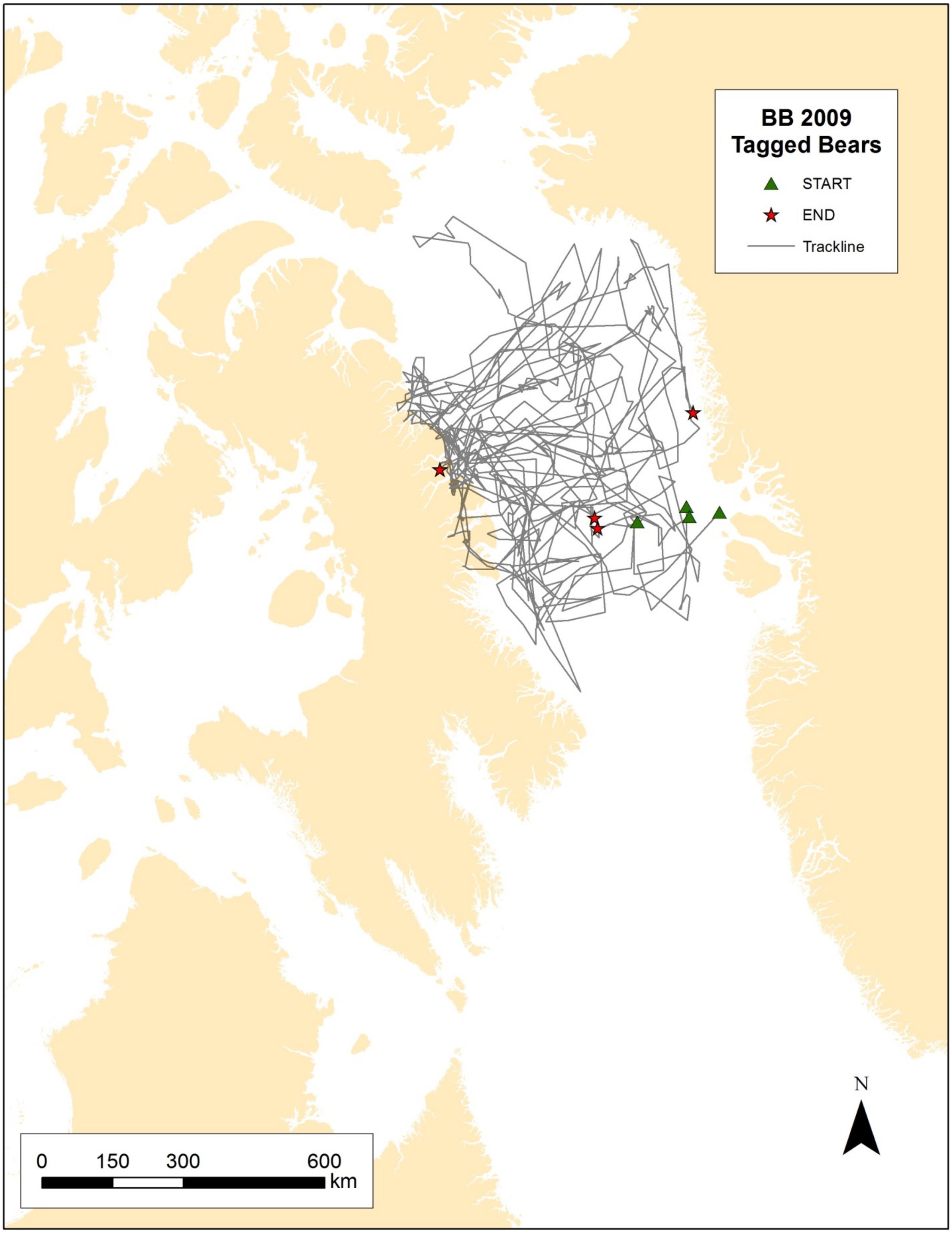


Figure 2.5. Tracklines from n=9 adult female polar bears satellite collared in 2010 in BB.

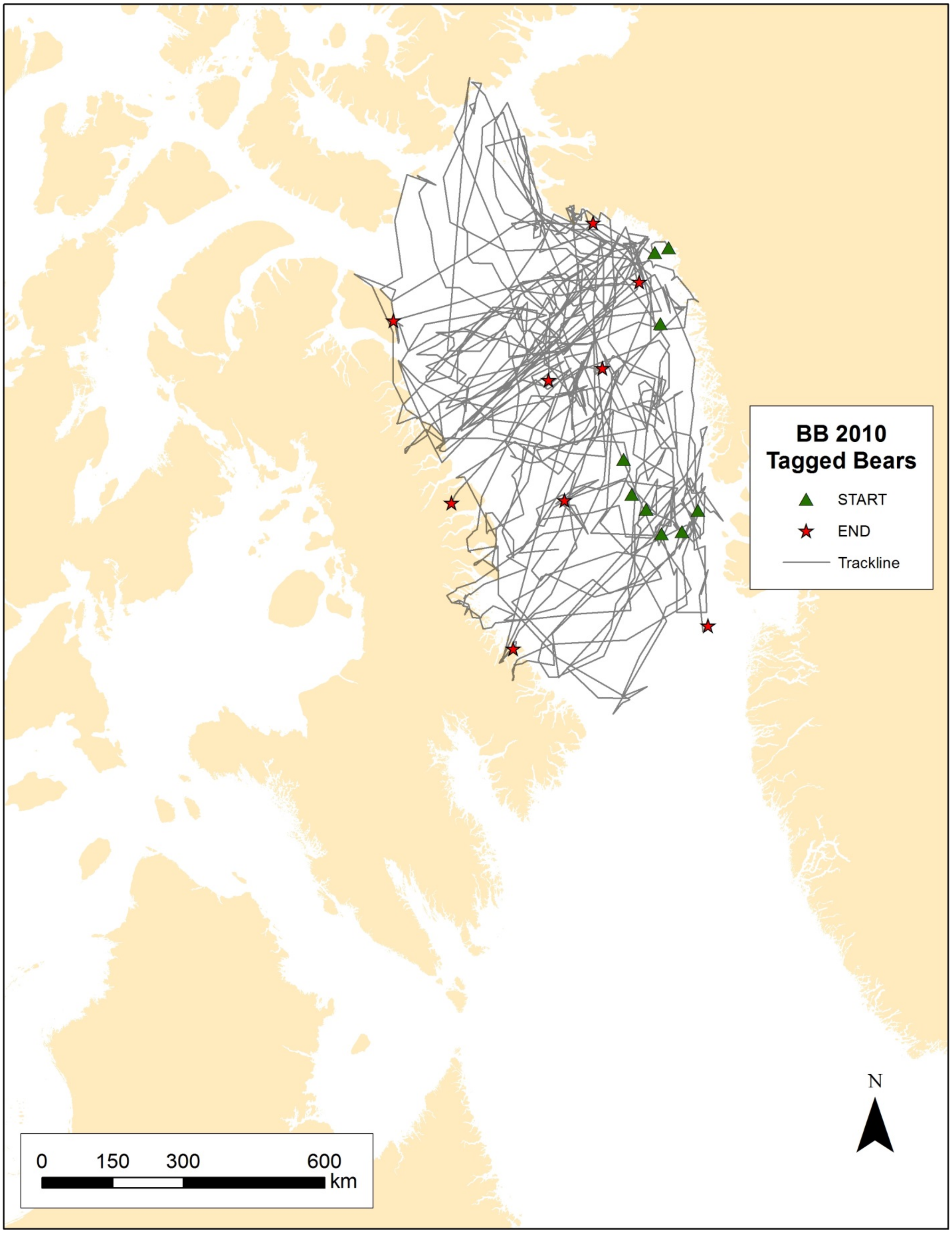


Figure 2.6. Tracklines from n=12 adult female polar bears satellite collared in 2011 in BB.

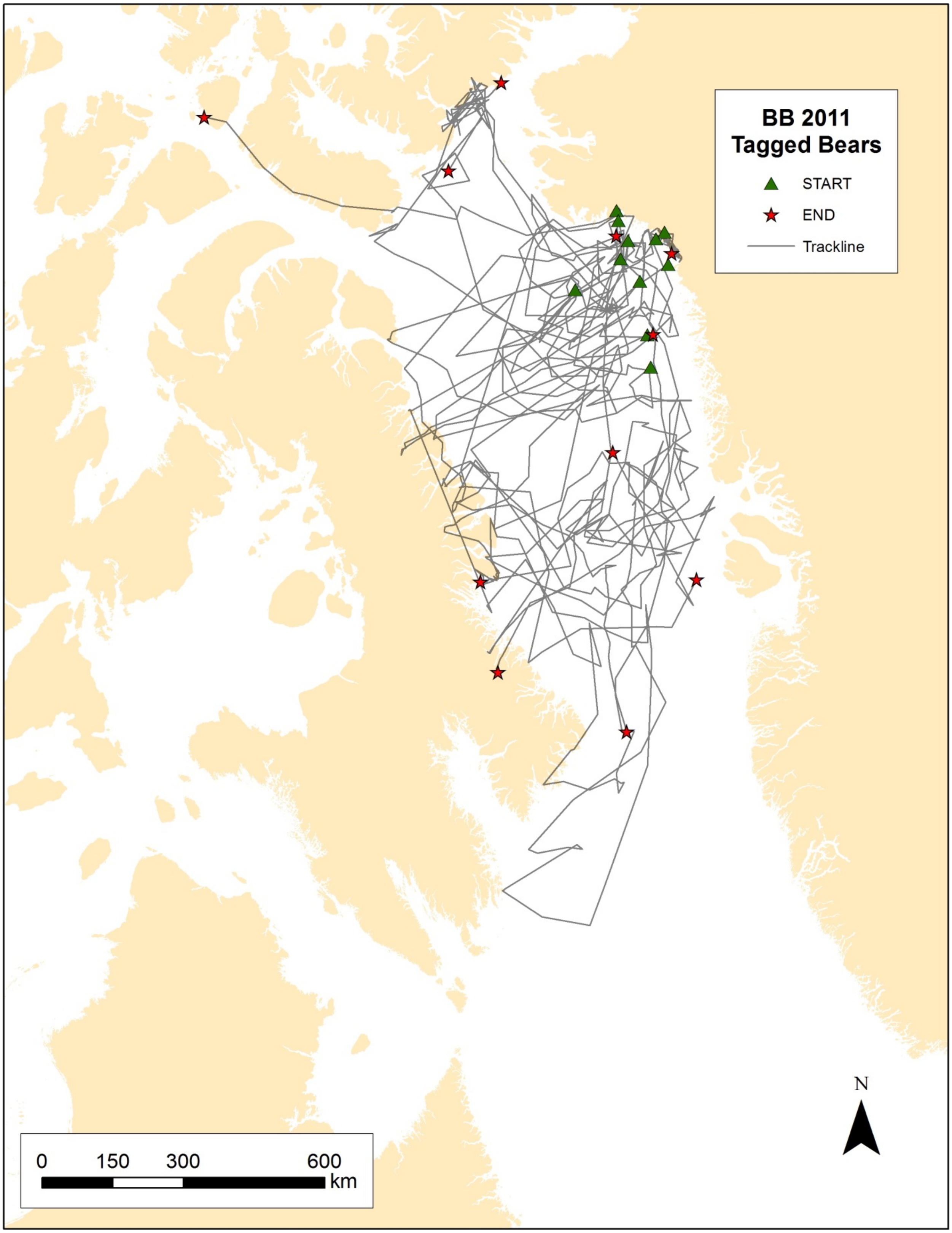


Figure 2.7. Tracklines from n=11 adult female polar bears satellite collared in 2012 in BB.

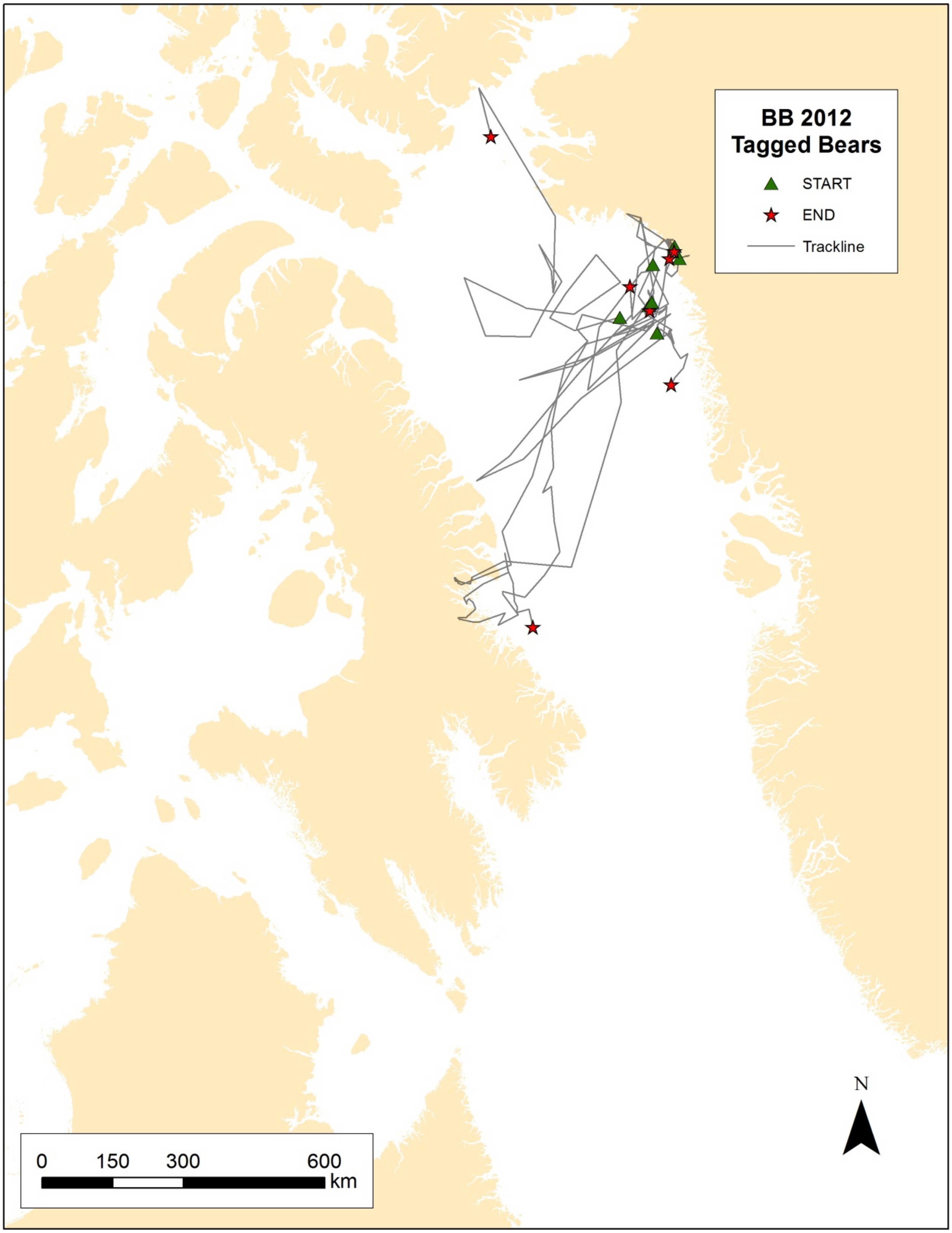


Figure 2.8. Tracklines from n=9 adult female polar bears satellite collared in 2013 in BB.

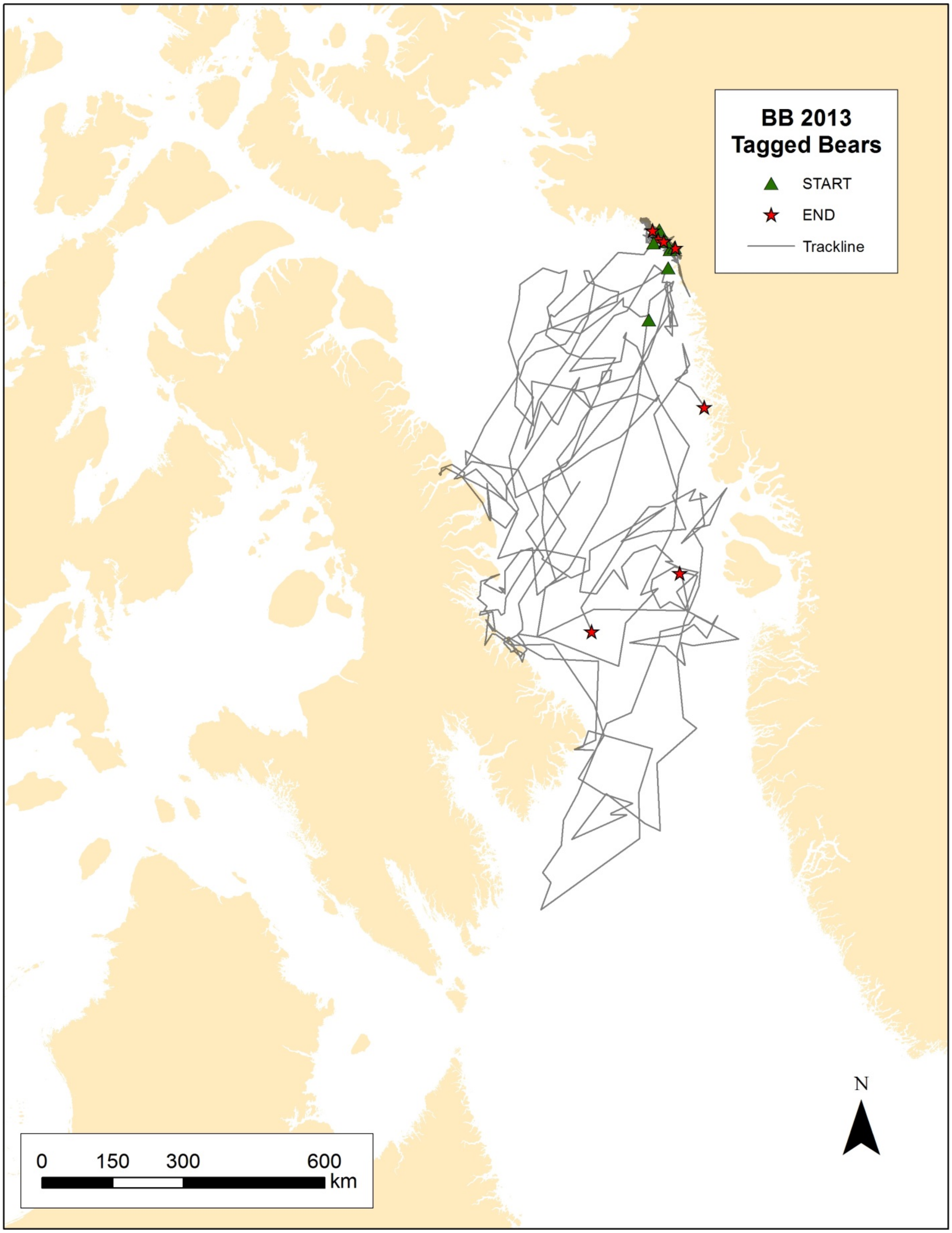


Figure 2.9. Tracklines from all adult female bears (n=38) collared between 2009 and 2013 shown together through April 2015, excluding bears where collars failed after a few days.

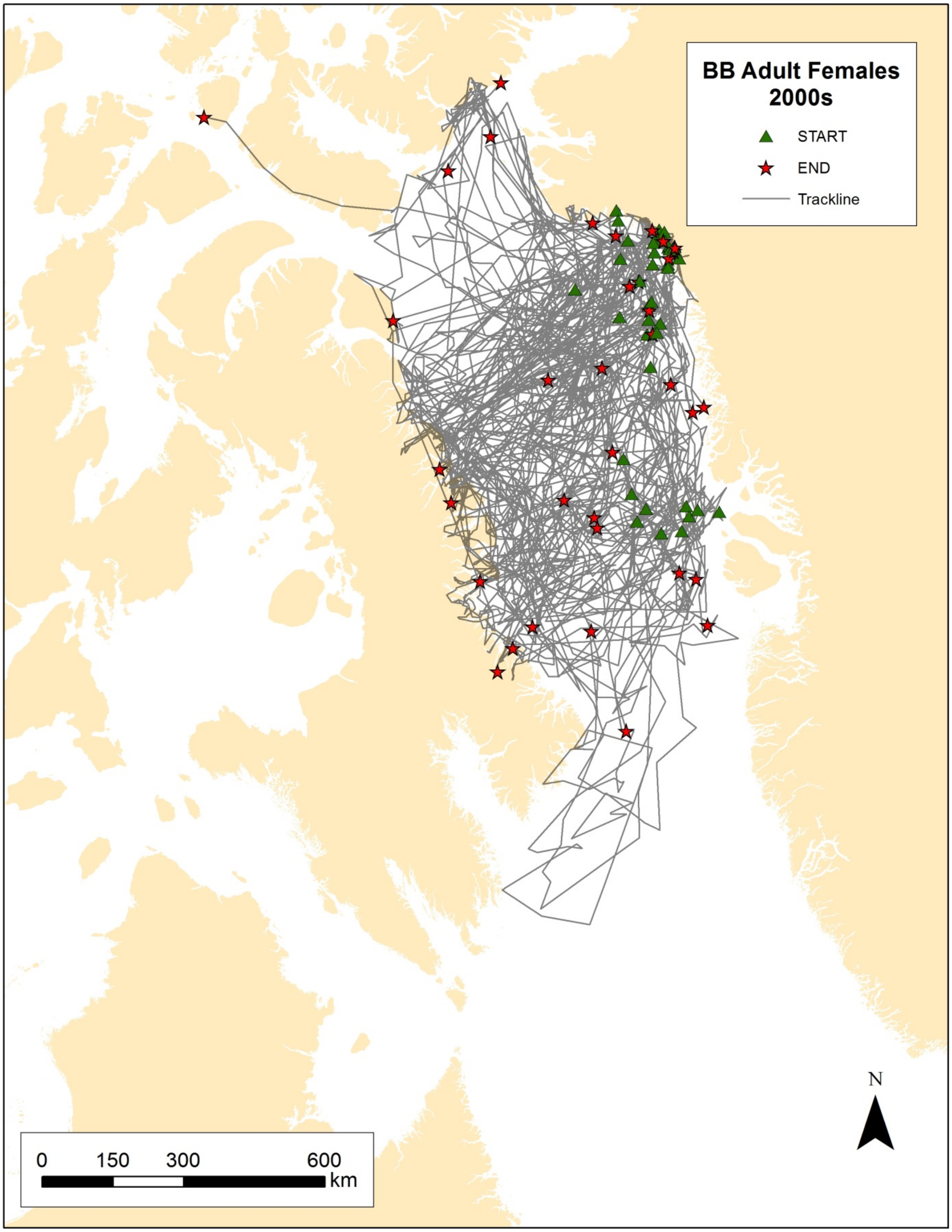


Figure 2.10. Tracklines from n=32 adult male bears tagged with ear transmitters between 2009-2013.

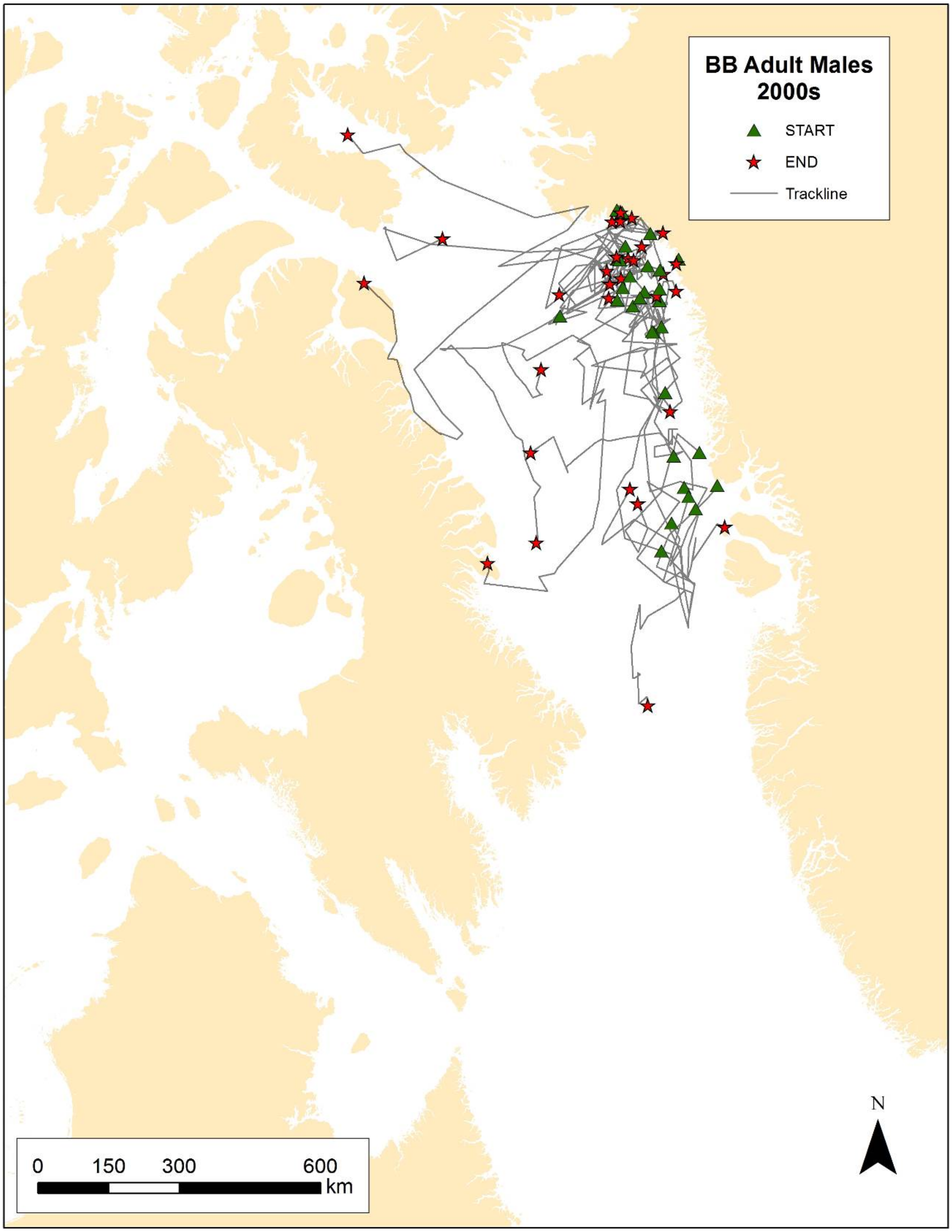


Figure 2.11. Tracklines from n=12 adult female polar bears satellite collared in the 1990s in KB.

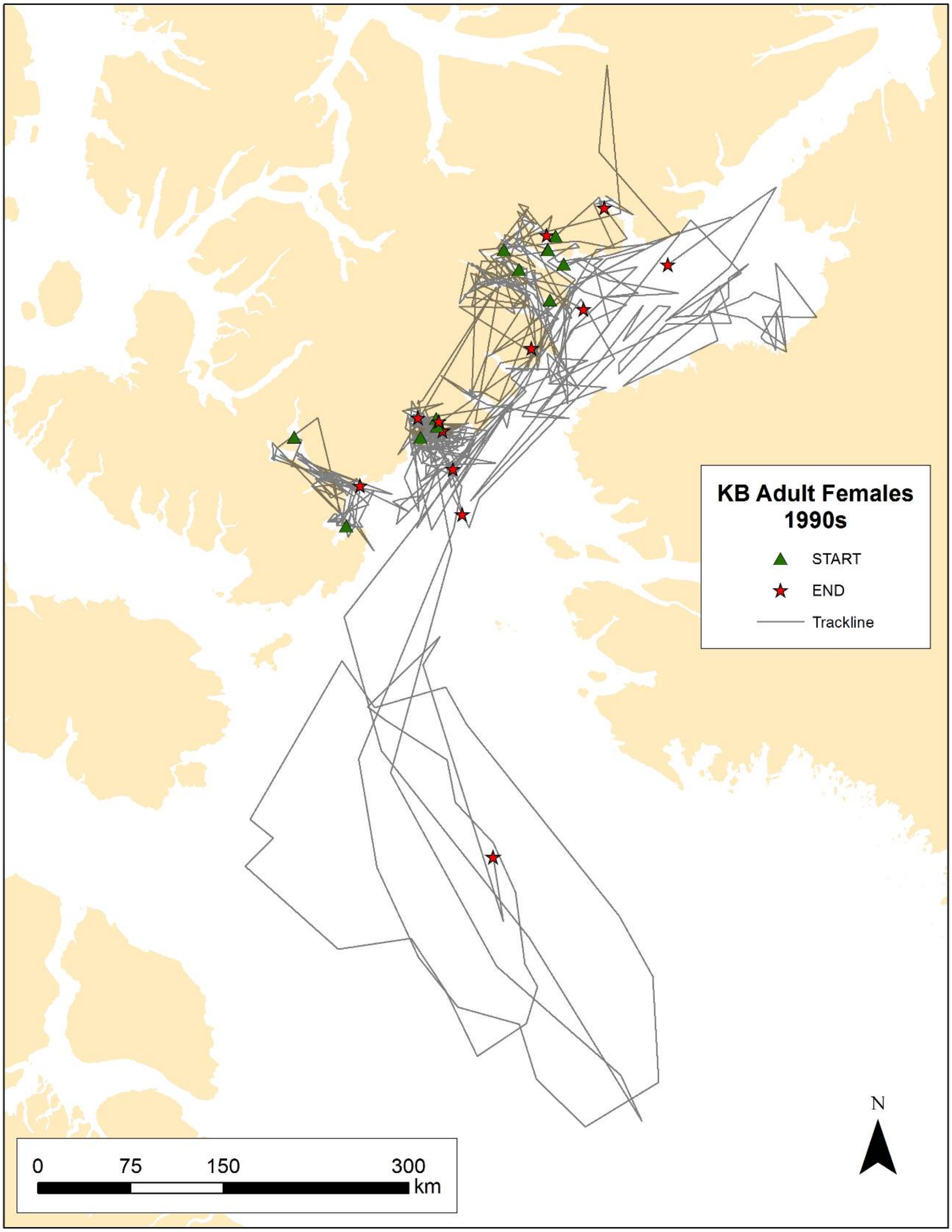


Figure 2.12. Tracklines from n=9 adult female polar bears satellite collared in 2012 in KB.

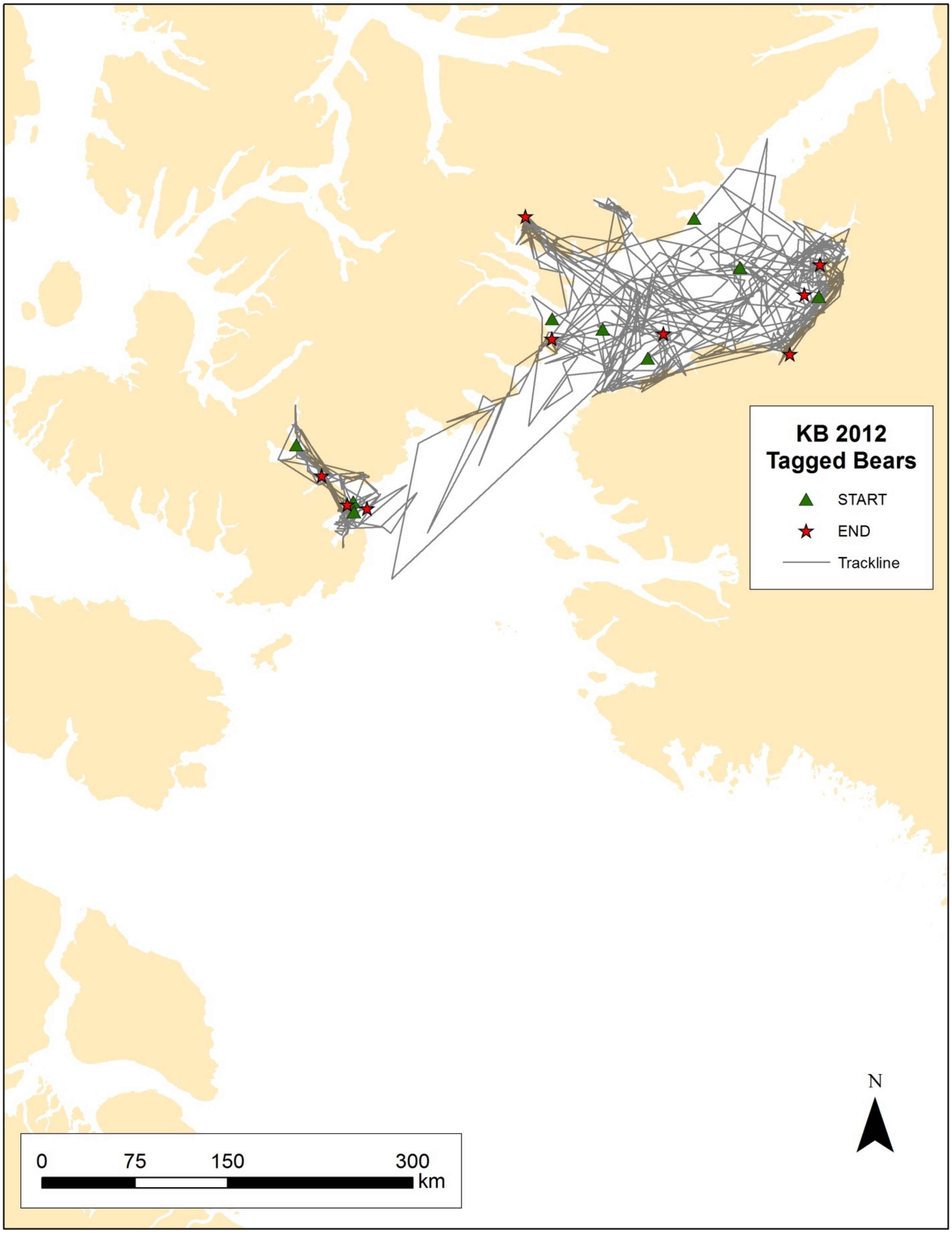


Figure 2.13. Tracklines from n=11 adult female polar bears satellite collared in 2013 in KB.

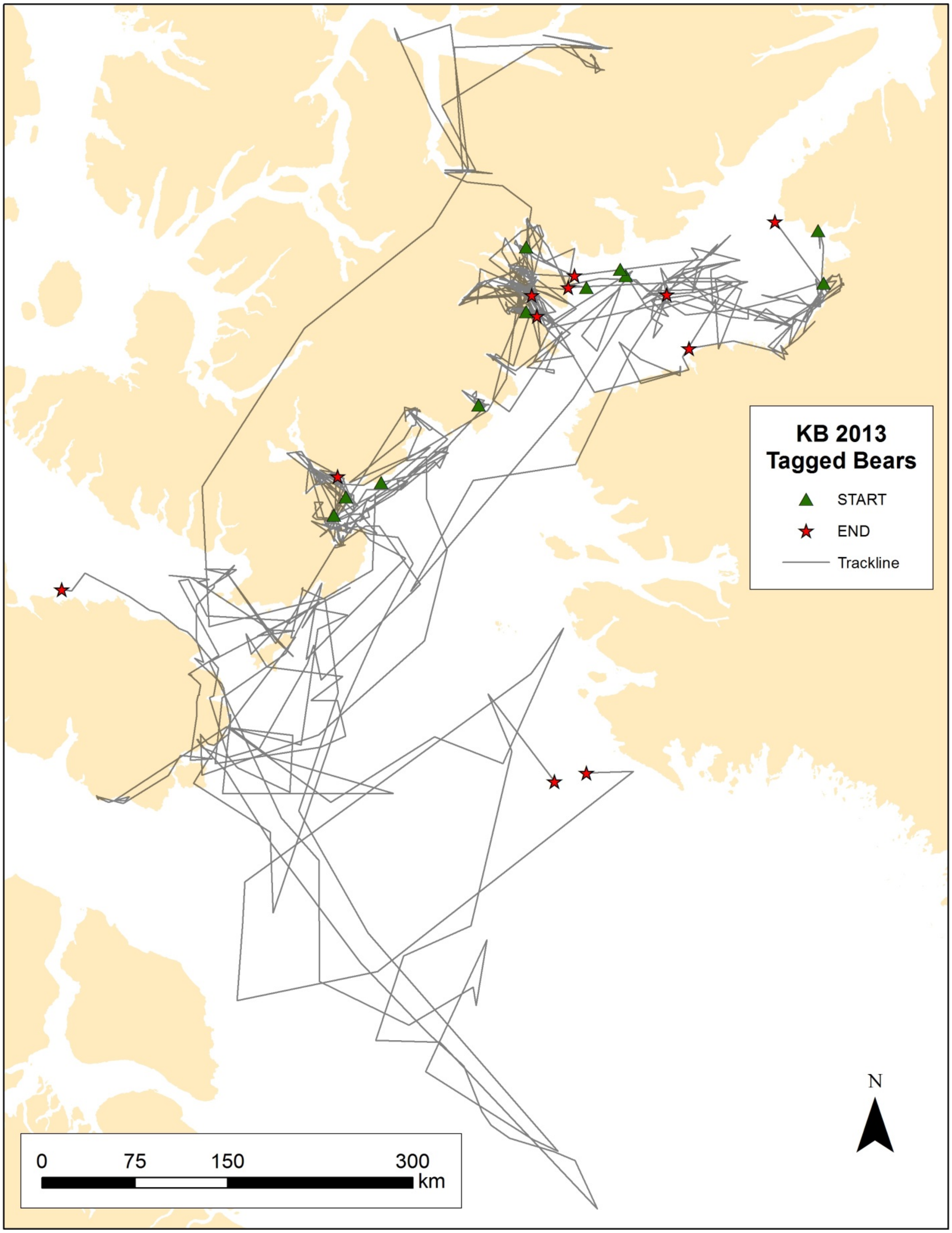


Figure 2.14. Tracklines from n=20 adult female polar bears satellite collared in 2012 and 2013 shown through April 2015 in KB. Inset shows n=1 bear that moved to Russia, excluded from analyses.

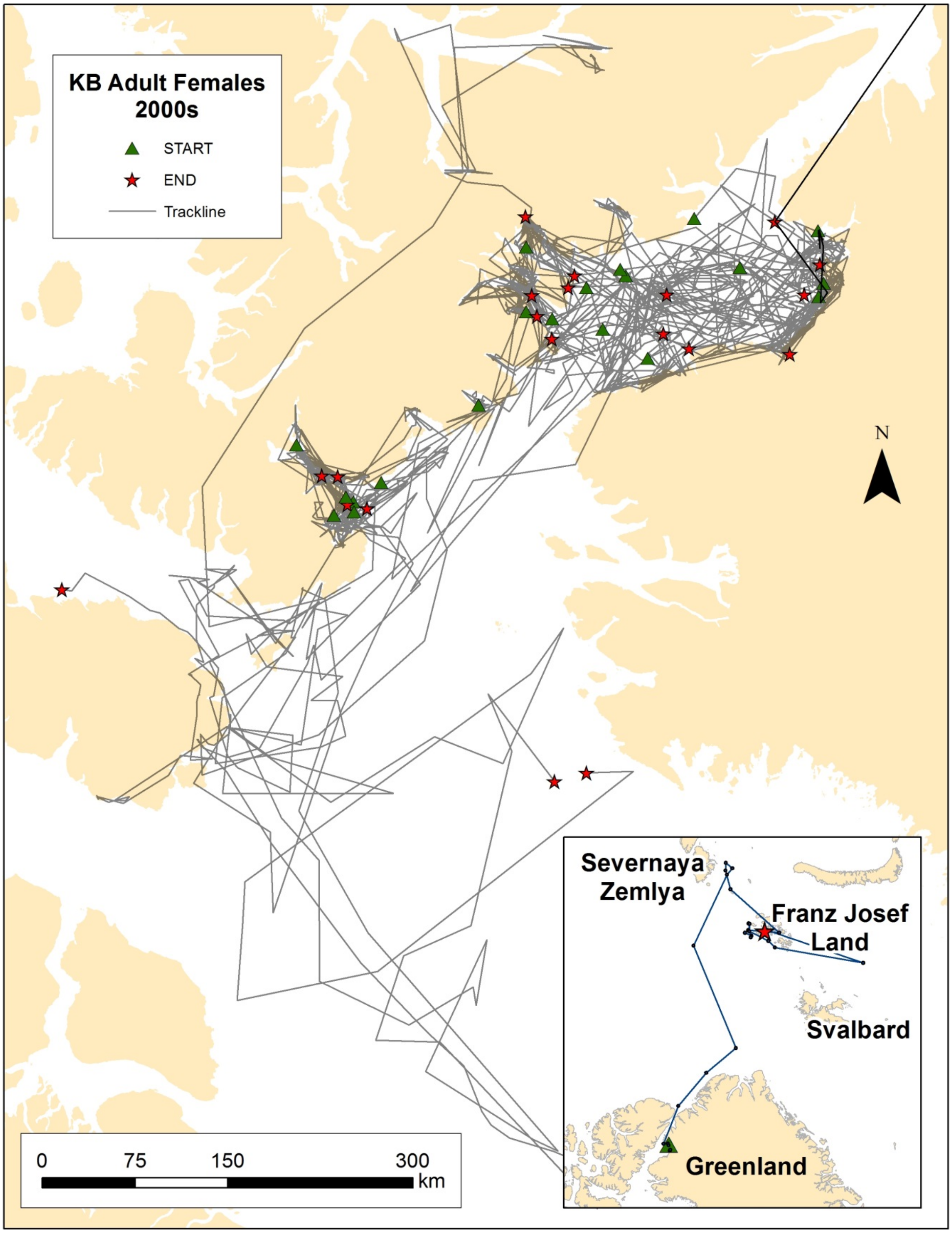


Figure 2.15. Tracklines from n=9 adult male polar bears satellite tagged with ear tags in 2012 and 2013 in KB.

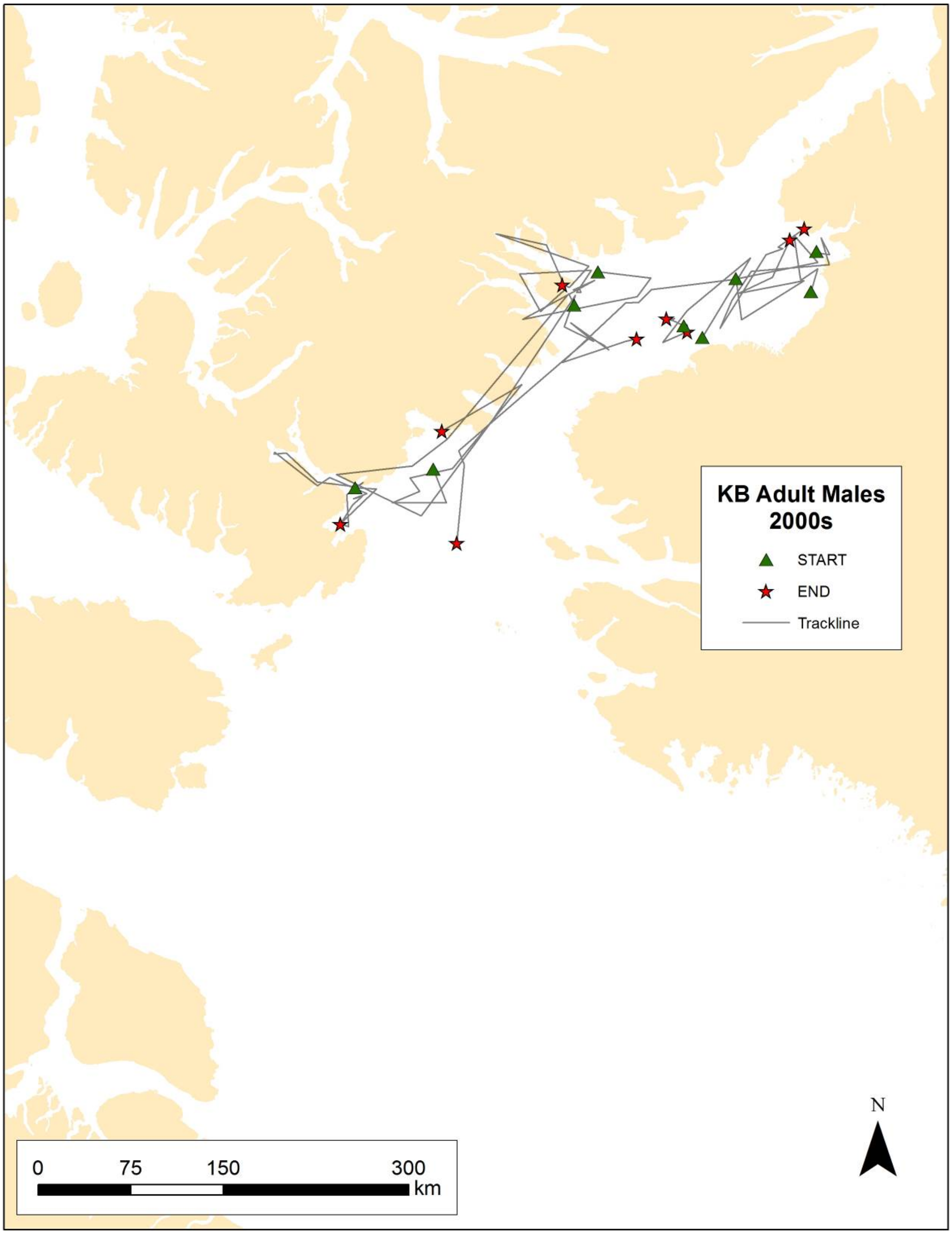


Figure 2.16. 95% kernel ranges for bears captured in 1990s and 2000s in BB by season (winter, spring and summer). See Table 2.4 for areas, overlap and tests for significance between decades.

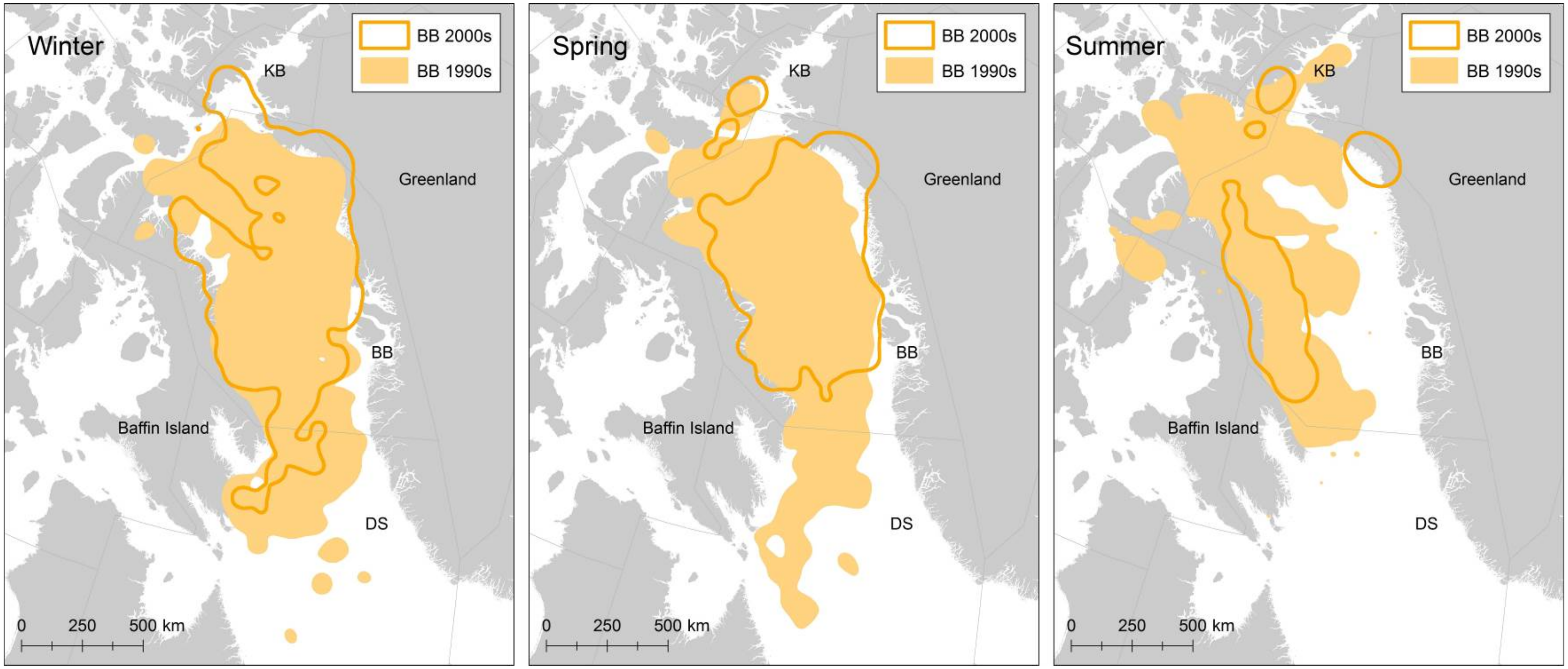


Figure 2.17. Matrix of home ranges shown by month in Baffin Bay for collared adult females in the 1990s and 2000s.

Figure 2.18. Home range sizes between decades for adult female polar bears in BB in the 1990s (red, n=43) and 2000s (blue, n=38). Line represents the mean values by month and shaded area +/- 2 SE. Graph excludes Melville Bay resident bears.

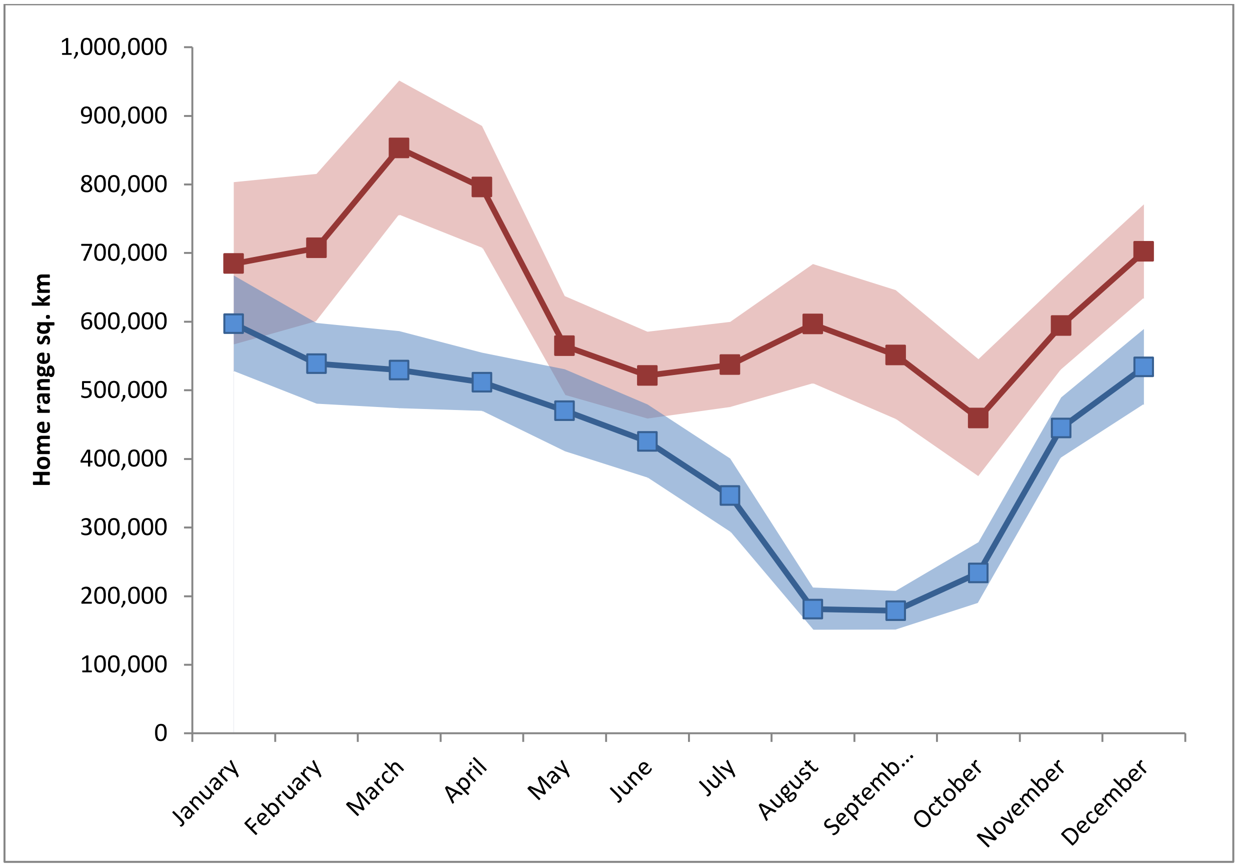


Figure 2.19. 95% kernel ranges for adult female bears captured in 1990s and 2000s in KB by season (winter, spring and summer).

See Table 2.4 for areas, overlap and test for significance between decades.

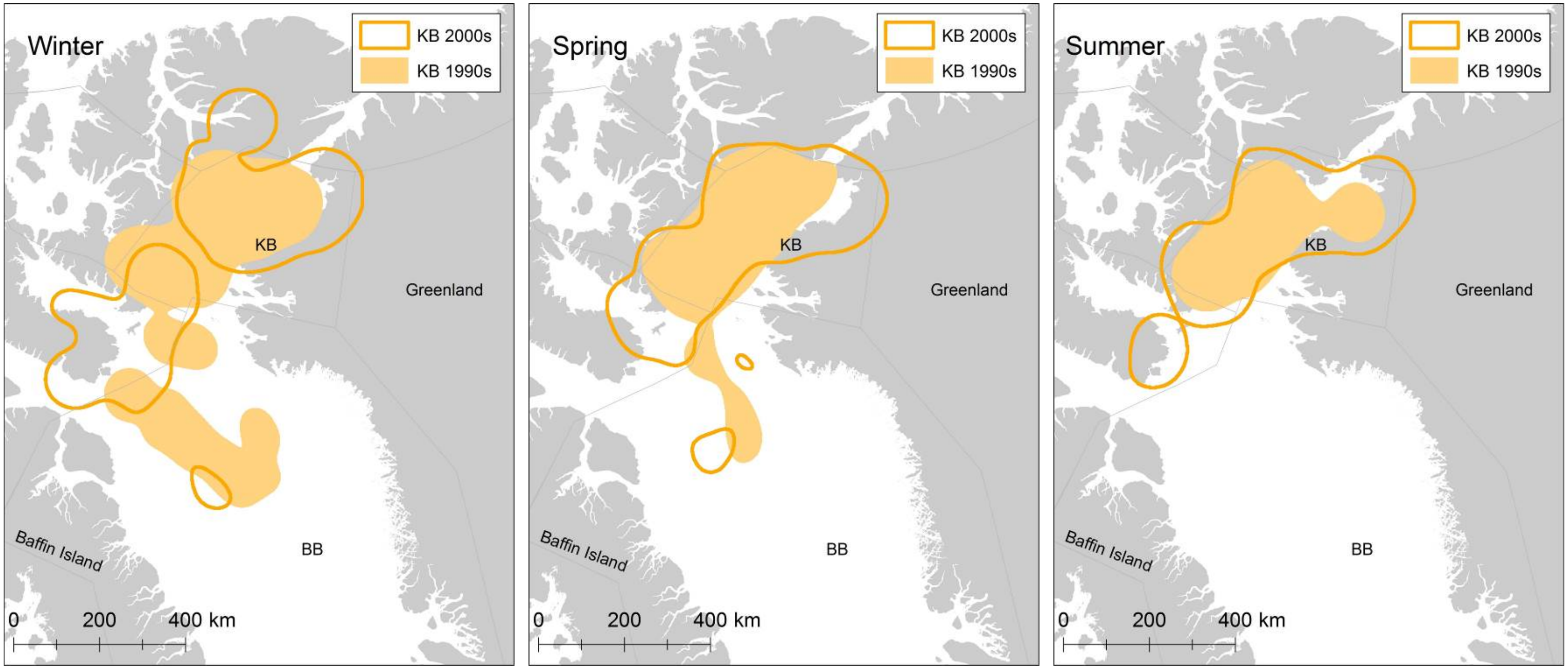


Figure 2.20. Matrix of home ranges shown by month for adult female polar bears in Kane Basin in the 1990s and 2000s.

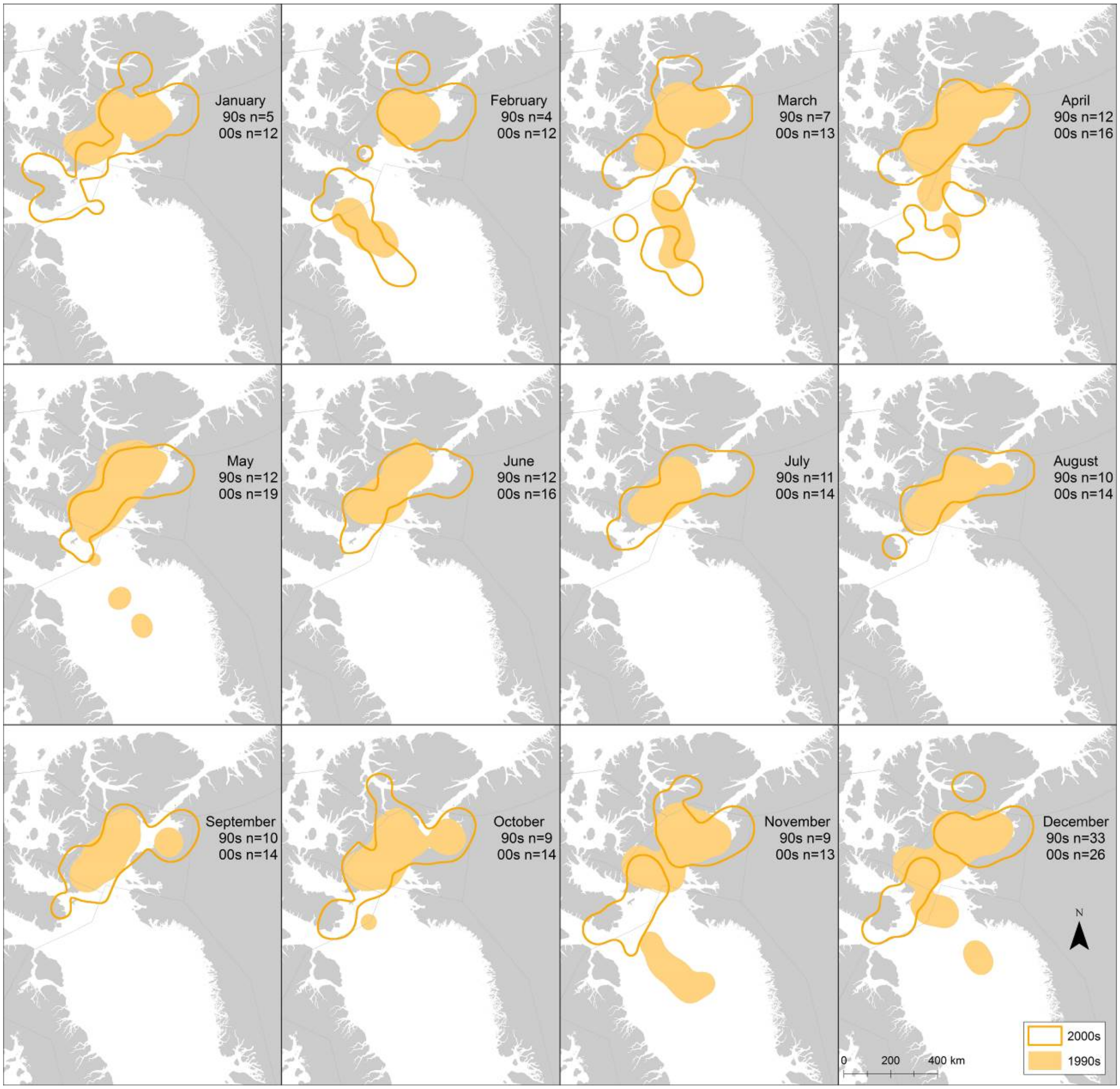


Figure 2.21. Home range sizes between decades for adult female polar bears in KB in the 1990s (red, n=12) and 2000s (blue, n=20). Line represents the mean values by month and shaded area +/- 2 SE.

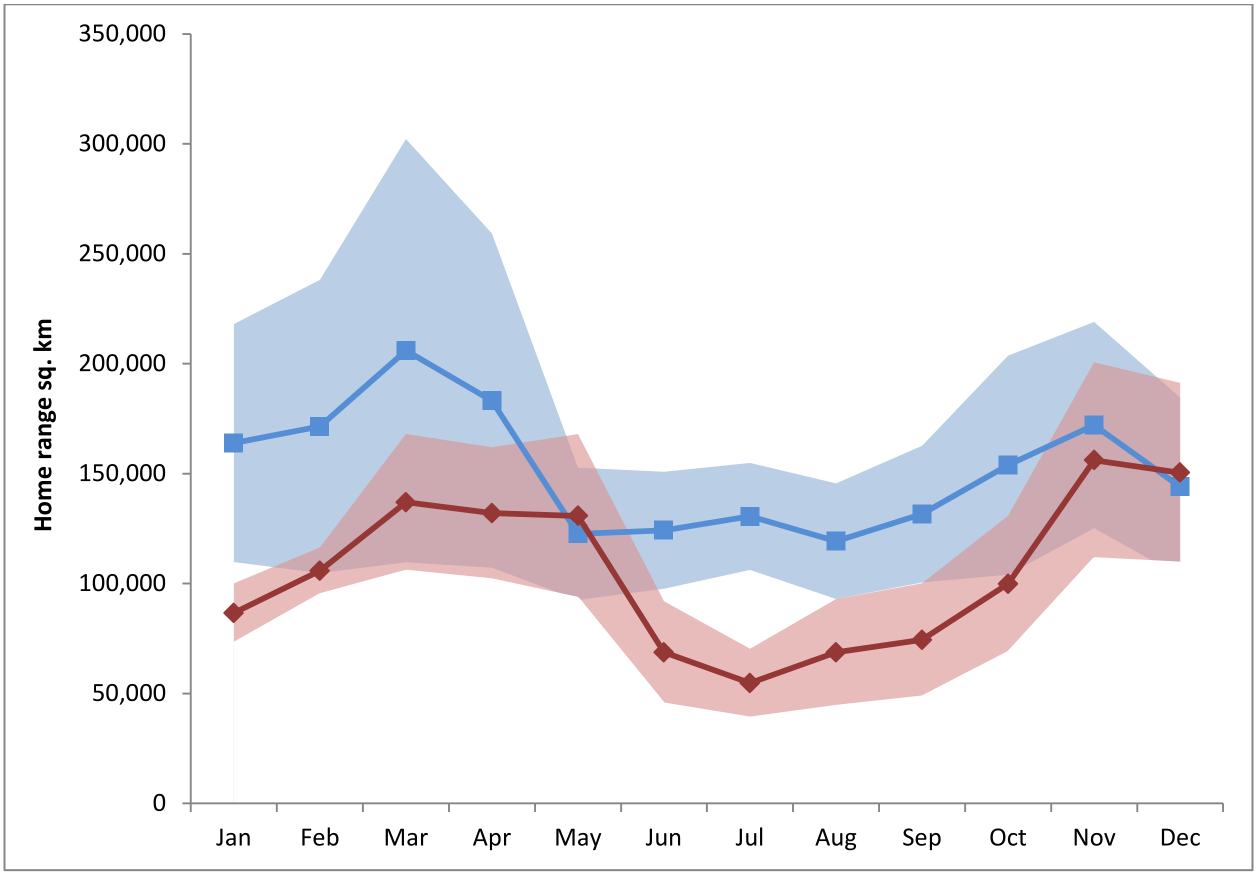


Figure 2.22. Box plots shown by season of median latitude for adult female polar bears in BB in the 1990s (n=43) and 2000s (n=38). Plot excludes the bears that are resident in Melville Bay though inclusion of these bears did not change the significance of the results.

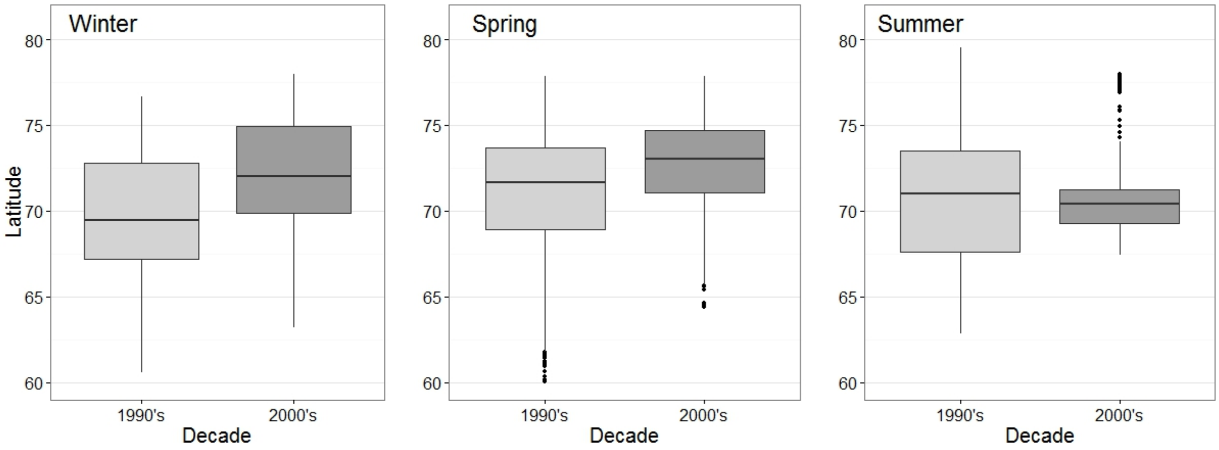


Figure 2.23. Box plots by season of median latitude for AF bears in KB in the 1990s (n=12) and 2000s (n=20).

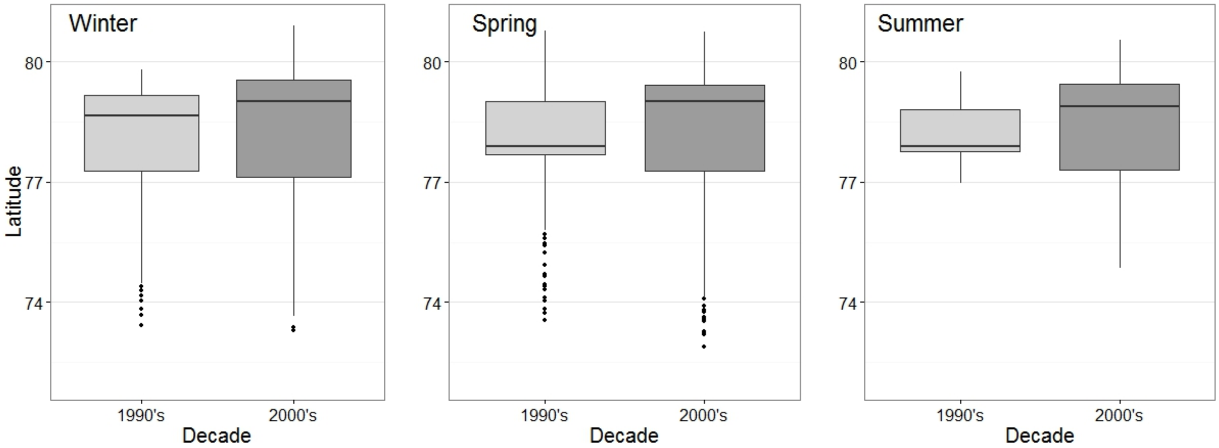


Figure 2.24. Plot of departure timing from region of origin for BB and KB bears in the 1990s and 2000s where departures of any length (min 4 days) were considered. See Table 2.3 for sample sizes.

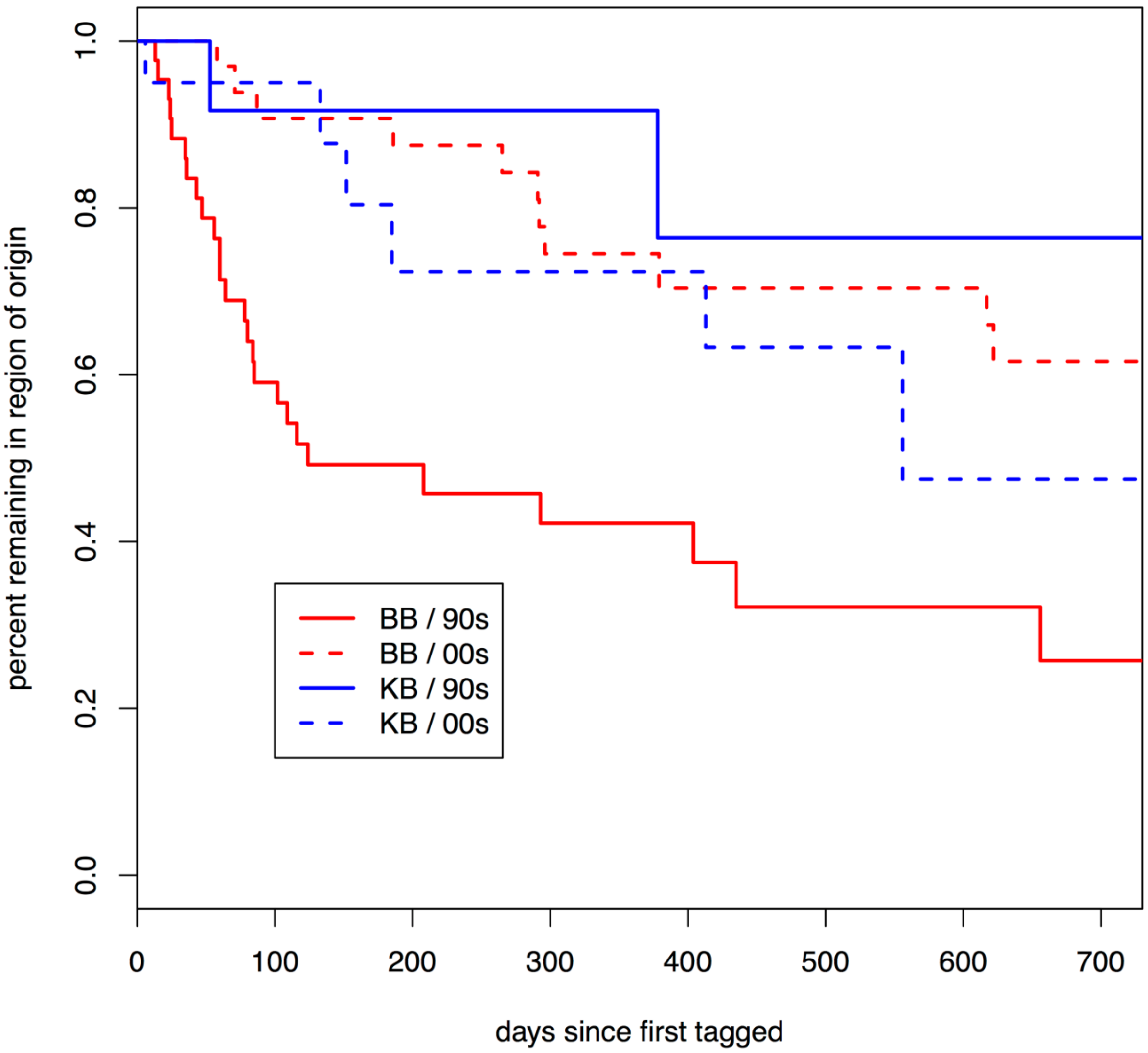


Figure 2.25. Plot of departure timing from region of origin for BB and KB bears in the 1990s and 2000s where only departures 30 days or greater were considered. See Table 2.3 for sample sizes.

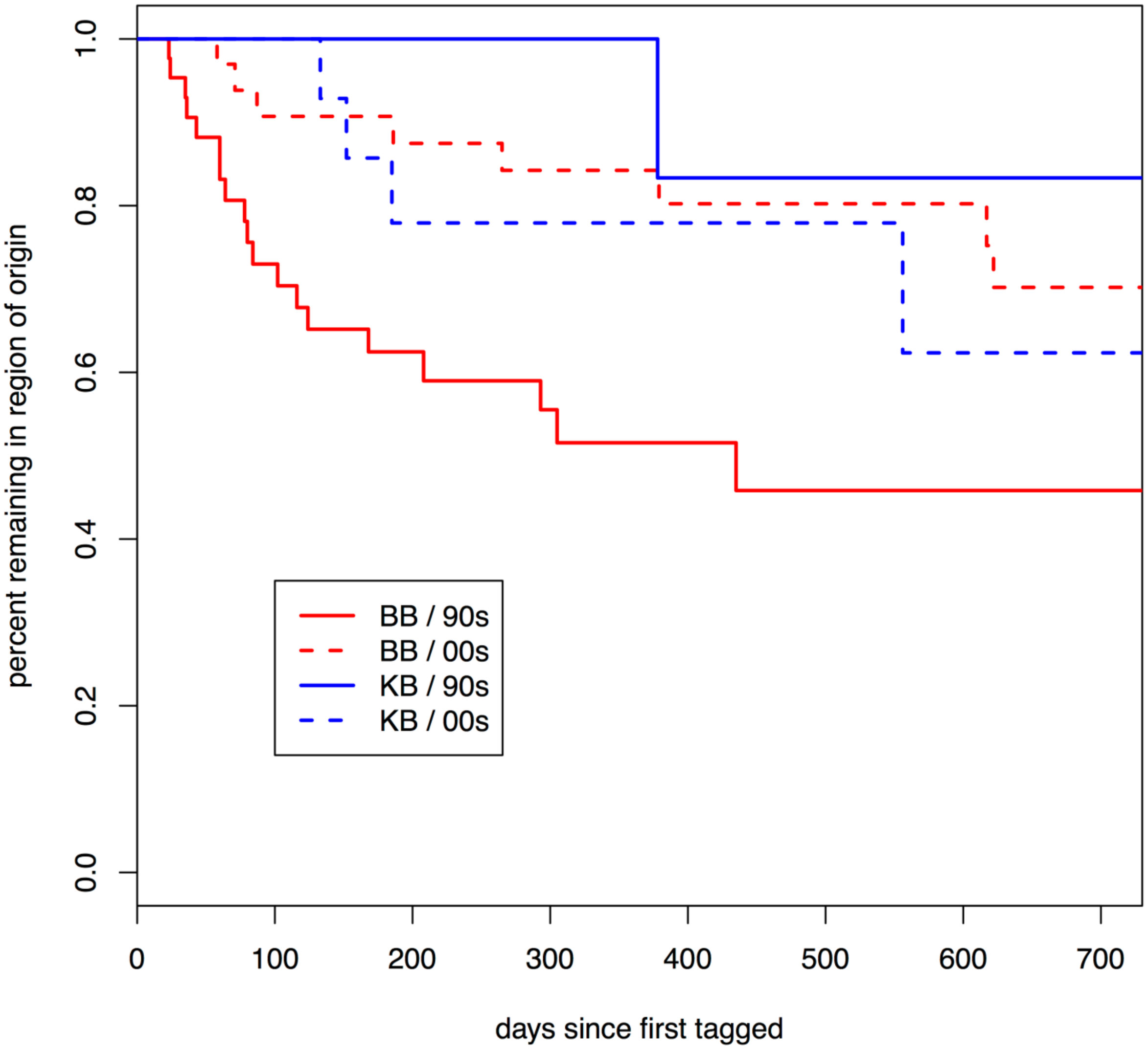


Figure 2.26. Distances (km) between mark and recapture locations of polar bears in Baffin Bay, 2011-2013. Adult females (AF), adult males (AM), cub-of-the-year (CO=COY), subadults (SA), yearlings (YR). Median distance is represented by the black line within each box. Box represents the interquartile range. Whiskers represent maximum and minimum values. Symbols denote significant differences between groups (Bonferonni correction for multiple comparisons, alpha = 0.05).

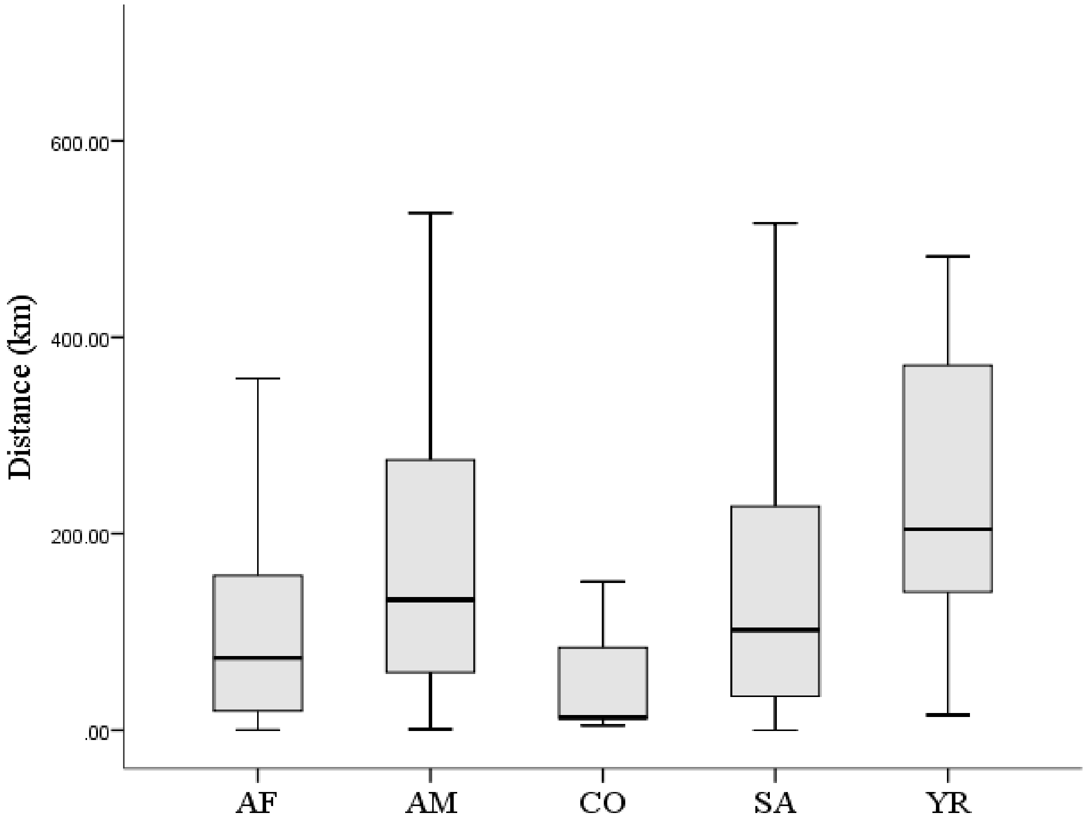


Figure 2.27. Relationship between capture-recapture interval and straight line displacement distance for 29 bears that moved between Baffin Bay and Davis Strait as detected by capture and recapture. Median distance is represented by the black line within each box. Box represents the interquartile range. Whiskers represent maximum and minimum values.

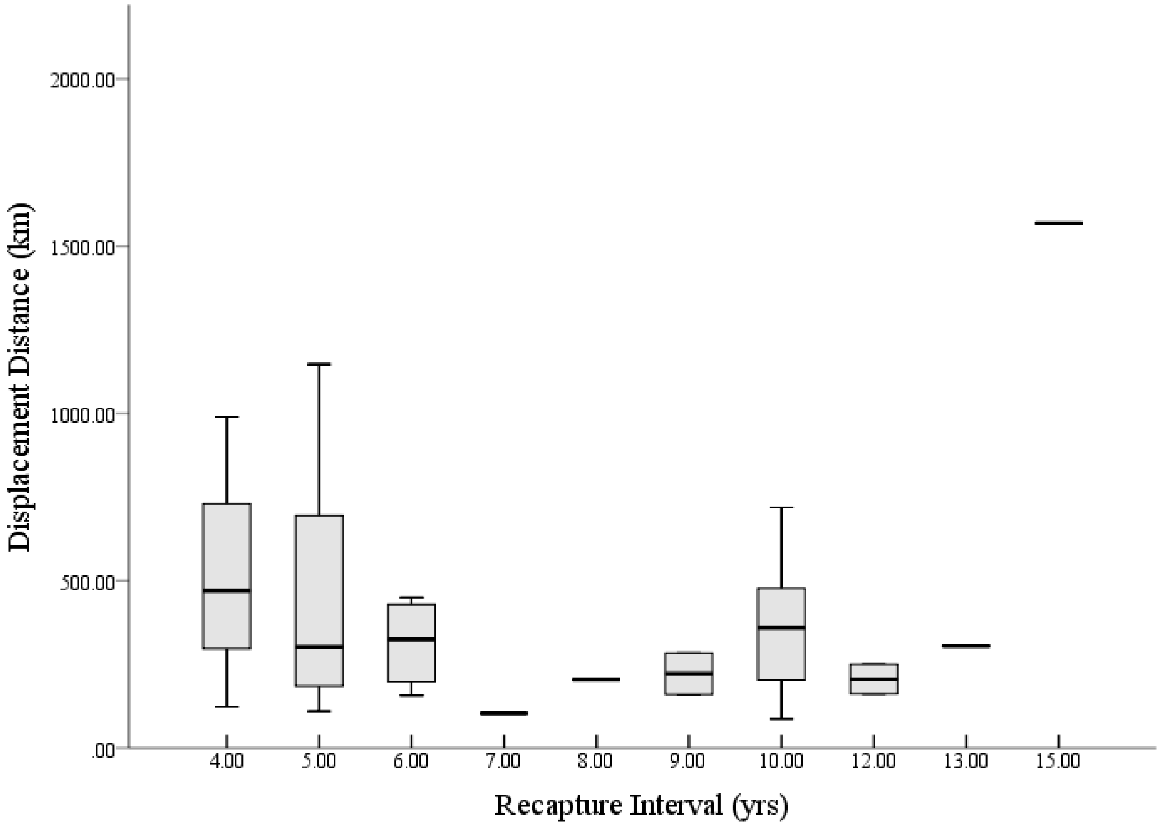


Figure 2.28. Frequency distribution of the distance between capture location and the boundary of the Baffin Bay (BB) and Davis Strait (DS) polar bear subpopulations for 29 individuals that made inter-subpopulation movements as detected by capture and recapture (grey bars), 1991-2013. Distances between capture locations and the BB-DS boundary for all bears (n = 2,771) marked in BB and DS are also shown (black bars). Bars represent proportion of captures occurring within each distance bin.

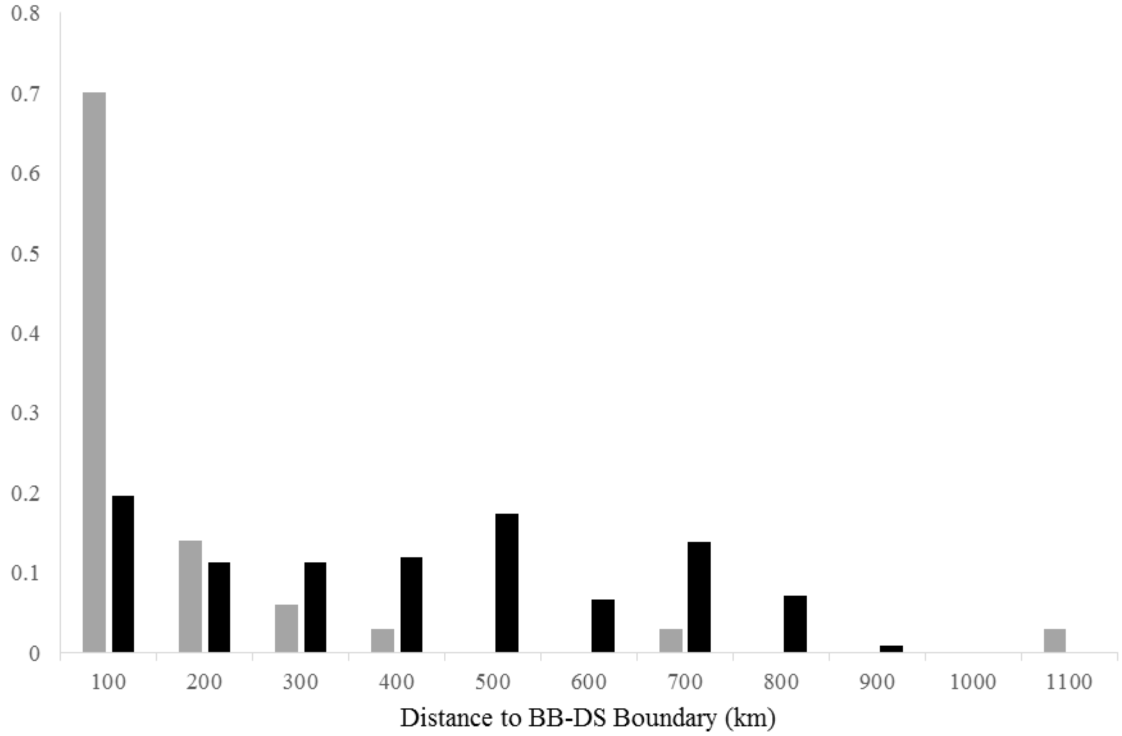
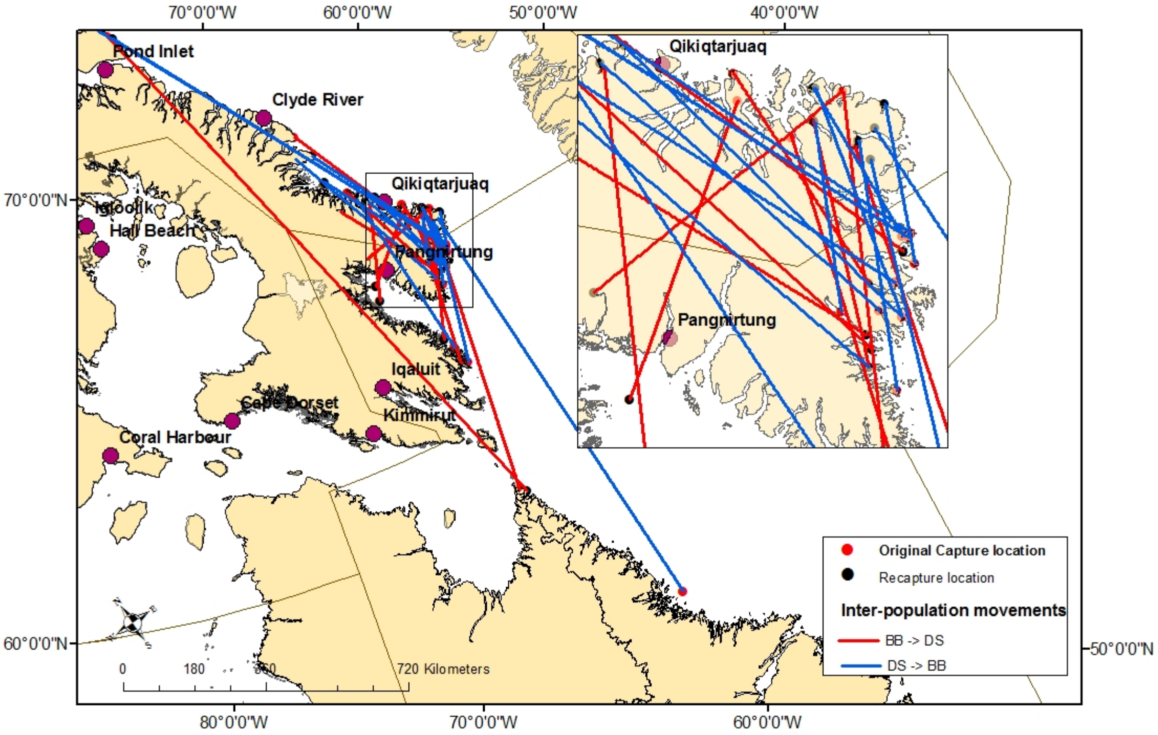


Figure 2.29. Capture and recapture locations of bears known to have made inter-subpopulation movements between mark-recapture sampling sessions in Baffin Bay (BB) (1990-97), Davis Strait (DS) (2005-07), and Baffin Bay (2009-2013).



CHAPTER 3

REASSESSING THE 1990S BAFFIN BAY DATA FOR BIAS AND COMPATIBILITY WITH THE 2010S DATA

KEY FINDINGS

- This chapter evaluates patterns in the 1990s physical MR data, including non-random and incomplete sampling, and the resulting potential for bias in estimates of demographic parameters.

- The 1990s MR sample size was small (average 229 total captures per sampling year), relative to the 2010s (average 470 total biopsies per sampling year), and the number of recaptures in the 1990s was low. There were few dead recoveries during the period between MR sampling studies (1998-2010), particularly in the latter years. Small sample sizes make it difficult to estimate demographic parameters and assess subpopulation trend, limiting both the strength of inference that can be drawn from the 1990s data and our ability to quantify and reduce bias in estimates of demographic parameters.

- The spatial distribution of polar bear physical captures and biopsy samples for the MR studies in the 1990s and 2010s was significantly different. In the 2010s, a larger fraction of bears were captured inland from the coastline, and inside fjords along Baffin Island.

- The difference in distribution of captures between sampling periods was not due to changes in habitat use. Analyses of satellite telemetry data from adult females, providing an unbiased assessment of land use between decades, showed no differences in distance inland or elevation for onshore bears between the 1990s and 2010s. Thus, the difference in capture distributions were a function of different sampling effort, with less effort expended away from coastlines and inside fjords in the 1990s.

- Consistent with the differences in sampling effort and temporary emigration between the 1990s and 2010s, there were significant differences in the composition of the MR samples (e.g., the proportion of bears within each age-sex class) between these two periods. Specifically, adult females were under-represented in the 1990s samples.

- The spatially-defined sampling area (km^2) in Nunavut encompassed the capture and biopsy locations in both decades and represented a minimum area sampled. The sampling area in the 1990s survey was less than ½ of that sampling in the 2010s. The 2010s sampling area encompassed most fjords along the coast and more inland habitat. To evaluate potential biases associated with the smaller sampling area of the 1990s, MR analyses and estimated parameters were compared from two datasets: (1) all 2010s MR data, and (2) a geographic subset of the 2010s MR data that was comparable to the sampling area in the 1990s (Chapter 5).

- In the 1990s there was likely a high degree of temporary emigration from the sampling area on the Baffin Island coast because bears used sea ice offshore in Baffin Bay or in the archipelago in summer. Significantly less sea ice was available in the 2010s and temporary emigration was lower. In the 1990s, $\leq$30% of radio-collared female bears were inside the sampling area during the MR sampling periods, compared to 70-80% in the 2010s. This suggests that a potentially significant proportion of bears were not available for capture each year during the 1990s, though sample sizes for analysis were small. Completely random temporary emigration from the sampling area should not result in biased demographic parameters. However, the degree of temporary emigration in the 1990s appeared variable and dependent on environmental conditions; and small samples sizes made it difficult to rule out significant bias.

- Additional sources of temporary emigration in the 1990s were non-random and linked to the reproductive cycle of females. Adult females in reproductive classes that were likely pregnant in fall moved farther inland on Baffin Island (e.g., to find suitable denning habitat), compared to non-pregnant females, which likely contributed to the under-sampling of adult females in some years in the 1990s because of the lack of inland sampling.

- There also were technical challenges with the 1990s MR data. Within the 1990s MR data there was uncertainty in identifying bears that were located with the aid of radio-telemetry vs. those located by standard search (i.e., random encounter). Original capture records could not be located and were inferred by comparing available information to the capture history files compiled for the 1990s BB demographic analysis. This uncertainty could result in bias, because knowing which bears were located by telemetry was important in the most-supported MR models for the 1990s data.

- Relative to the 2010s data, the 1990s data were characterized by relatively small sample sizes, incomplete geographic sampling, a likely higher degree of temporary emigration for bears that remained on sea ice during the summer, and potential non-random temporary emigration for adult females that moved farther inland to den. These issues led to an increased potential for bias in estimates of survival and abundance from the 1990s data. As a result, demographic parameters estimated from 1990s and 2010s BB data are not directly comparable and there is a limited ability to evaluate subpopulation trends.

3.1. Background

Accurate knowledge of demographic parameters (e.g., survival, abundance) is important for wildlife management decisions such as determining sustainable harvest levels and evaluating subpopulation viability. Mark-recapture (MR) studies are used to estimate demographic parameters because it is generally not feasible to monitor every individual in a subpopulation.

The results from MR studies can be biased by several factors, including heterogeneity in recapture probability (*p*) that is not accounted for through the choice of sampling design or modeling approach (Williams et al. 2002). The magnitude of bias is generally largest for abundance (Pollock et al. 1990) although estimates of survival probability can have meaningful bias as well (Devineau et al. 2006). Estimating accurate and unbiased demographic parameters for polar bears is particularly challenging. First, sample sizes are relatively small due to challenging environmental and logistical conditions, and the high cost of Arctic fieldwork. Second, polar bears are often distributed across large landscapes at low densities. Only a fraction of the study subpopulation may be accessible to researchers, and this fraction may change from year to year based on environmental conditions and logistical constraints. This limits sample sizes, leads to difficulty in delineating subpopulation boundaries, and means that the effective study subpopulation may be different than the biological population of interest. Third, the high mobility of polar bears and inter-annual variability of their sea-ice habitat can lead to nonrandom movements (i.e., temporary emigration) with respect to the sampling area. Fourth, female bears may be less-observable or unobservable for several months when pregnant or associated with maternal dens, leading to an ‘unobservable state’. Fifth, the three-year reproductive cycle of polar bears makes it difficult to estimate reproductive rates and their relationships with environmental conditions. Finally, relatively long-term datasets are required because of the long life span of polar bears and high inter-annual variability in the Arctic environment.

In recent years, methodological advances have led to an increased ability to detect, quantify, and mitigate bias in demographic parameters from MR studies arising from the challenges listed above. Advances include noninvasive genetic methods to increase sample size

(Lukacs and Burnham 2005); multiple sampling occasions per year under a “robust design” (Kendall et al. 1997); spatially-explicit models to account for heterogeneity in recapture probability as a function of site fidelity (Royle et al. 2014); models with “unobservable states” to account for temporary emigration (Schaub et al. 2004); and models that integrate data from multiple sources (Peñaloza et al. 2014). Some of these methods have been employed for polar bears, whereas others have not been used due to lack of familiarity or practical limits on the types of sampling that can be conducted.

The MR study of the Baffin Bay subpopulation 2011-2013 incorporated noninvasive genetic sampling and modelled live-recapture and dead-recovery data in the same analytical framework. Both of these approaches increased sample sizes and reduced susceptibility to some types of bias. Nonetheless, there remained major challenges to the application of MR models to the Baffin Bay data, and in this chapter we evaluate sampling and biological issues that have the potential to introduce bias in estimates of survival and abundance. Similar investigations of bias have become a standard part of MR studies for polar bears (e.g., Regehr et al. 2010), and are necessary to understand the strength of inference that can be drawn from MR studies. In this chapter we focus on reassessing the 1990s BB data because, compared to the 2010s data, the 1990s data had smaller sample sizes, reduced geographic coverage, and other uncertainties and limitations. This assessment directly informs our ability to compare results from the 1990s and 2010s data and evaluate trends in polar bear survival and abundance between sampling periods.

Distribution of Mark-Recapture Sampling on Baffin Island

Prior to the 2011-2013 survey of the Baffin Bay subpopulation, MR sampling occurred during several periods. Initial sampling was conducted during the 1970s (northern Baffin Island

and Bylot Island, near Lancaster Sound), early 1980s (east-central Baffin Island), and late 1980s to early 1990s (Canada and Greenland, as part of movement studies; Figure 5.2; Taylor et al. 2005). Early sampling efforts were generally restricted to spring-time and primarily occurred on landfast and nearshore pack ice. These studies documented that an unknown but likely large proportion of the subpopulation was on sea ice farther offshore during the spring and therefore unavailable for capture. We excluded these early data from present analyses (*cf*. Taylor et al. 2005, in which these early data were included) because the early sampling occurred in a different season (i.e., spring) and was spatially variable and restricted. Additionally, lack of tissue samples from early sampling precluded genetic identification for use in the present study.

In 1993 –1995 and 1997, more systematic sampling occurred during fall ice-free seasons (during September and October) on Baffin and Bylot islands (Figure 3.1). There was no fall sampling in 1996 due to logistical and resource constraints. These data formed the core of the study reported by Taylor et al. (2005) who estimated the number of polar bears in Baffin Bay at 2,074 (95% confidence interval: 1544-2604) in 1998. Taylor et al. (2001) indicated that a large majority of polar bears were onshore in summer retreat areas on Bylot and Baffin islands during the autumn. Taylor et al. (2005) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island. Consequently, Taylor et al. (2005) suggested that the autumn onshore sampling in 1993-1995 and 1997 provided improved coverage of the subpopulation and more reliable abundance estimates compared to those derived from the 1980s BB data, which Taylor et al. (2005) suggested were biased low.

In 2011-2013 we completed a second fall-time MR sampling study (August – October) on the coasts of Baffin Island (Figure 3.1). Data from West Greenland were also collected (see

Chapter 5, Figure 5.8). During this study, new data on movements and spatial distribution of bears were also obtained via satellite telemetry in BB and KB. This information was used to assess subpopulation boundaries (Chapter 2) and habitat use relative to the 1990s (Chapter 4), but also to improve MR study design (i.e., stratify the study site; Chapter 5) with the objectives of reducing heterogeneity in capture probabilities and more efficiently allocating survey effort. The 2011-2013 study (see Chapter 5) was largely modeled after Taylor et al. (2005) in that bears were targeted during the ice-free season, to obtain estimates of abundance and vital rates that might be comparable to Taylor et al. (2005) therefore useful for assessing trend.

Here we compare the spatial and temporal distribution of physical captures and biopsy sampling on Baffin Island for sampling 1993-1995 and 1997 vs. sampling during 2011-2013 (referred to as the “2010s”). The goal is to evaluate whether there were important differences in sampling, which could lead to different biases or different definitions of the effective study subpopulations (e.g., if a large group of bears was systematically missed in one study period, then the effective study subpopulation for that period would be smaller). Field records (e.g., Global Positioning System helicopter logs, navigation maps) delineating survey effort 1993-1995 and 1997 were unavailable. Therefore, we plotted sighting data from Taylor et al. (2005) in a Geographic Information System (GIS; ArcMap 10.2, ESRI, Redlands, California, USA) to examine the spatial distribution of captures compared to the 2010s. We also used historic and current radio telemetry data to identify whether potential differences in capture locations were influenced by changes in the onshore movements and habitat use of polar bears.

Methods – Maps of physical capture and biopsy sampling locations (hereafter collectively referred to as “captures”) on Baffin Island suggested that captures in the 1990s were more limited to coastal areas, whereas captures in the 2010s included bears located farther from the coast and deep inside fjords, including higher altitudes (Figure 3.1). We examined the hypothesis that the discrepancy in capture locations across periods reflects differences in sampling effort rather than a shift in the onshore distribution of bears. We calculated the distance to the nearest coastline and the distance to the smoothed outer Baffin Island coastline for each capture location in the 1990s and 2010s. The smoothed coastline followed the contour of the true physical coastline of Baffin Island, but was smoothed across fjords with a straight segment orthogonal to the fjord direction. We smoothed fjords only when the distance across the mouth of the fjord was ≤ 7 km using an Azimuthal Equidistant projection (WGS84 datum). We calculated the distance to both coastlines (original and smoothed) for all captures of independent bears (i.e., age 2 or older) that were located on mainland Baffin and Bylot islands (i.e., not on offshore islands) and were successfully genotyped.

We compared the distance-to-coast results to locations of radio-collared bears onshore during the 1990s and 2010s to evaluate whether differences in capture locations reflected differences in sampling effort or differences in the distribution of bears. Given that recent analyses of movement data suggest significant changes in sea-ice habitat use and onshore timing (Chapter 4), we considered the possibility that bears had also changed their behavior and habitat use while on land. First, we verified that the sample of 1990s bears collared in the fall on Baffin Island were comparable to the sample of 2010s bears collared in the spring in West Greenland, by assessing what fraction of spring-collared bears used the area on Baffin Island where bears were collared in the fall (see details in Chapter 2). Overall, 92% of the 2010s spring-captured bears used the fall collaring area. This suggests that, although radio-collaring occurred in different seasons and areas across the two time periods, the collared bears exhibited similar movement and habitat use patterns, and therefore provided comparable data for evaluating onshore habitat use across time periods.

Using satellite telemetry data, we calculated the distance inland from the smoothed coastline and Digital Elevation Model (DEM) elevation (m) for all locations of collared female bears during summer months (August-October). We used land covariates derived from the 22 m^2ASTER GDEM for all positions in Canada (http://www.jspacesystems.or.jp/ersdac/GDEM/E/4.html). We only used adult female bears on Baffin Island and calculations excluded resident bears that remained year-round on the Melville Bay glacier ice.

We also examined distance to the smoothed Baffin Island coastline for adult females as a function of reproductive status (captured alone, as mating pairs; or with COY, yearlings, 2-year old cubs) to evaluate whether this factor may have influenced temporary emigration with respect to the sampling area (particularly the nearer-shore sampling area in the 1990s). For this specific analysis (reproductive state examination) we only examined adult females in the year of collar deployment because their reproductive status was known at the time of capture in spring, thus could be assumed in fall. We excluded bears on sea ice during August-October.

Results – The mean distance of captures to the smoothed coastline was smaller in the 1990s ($\bar{x}$= 5.1 km, SD = 7.2, n = 438) compared to the 2010s ($\bar{x}$= 8.6, SD = 11.9, n = 766, Mann-Whitney U test: z = 3.4, P < 0.001). Detailed results are provided in Table 3.2. Furthermore, a greater proportion of independent bears were captured near the smoothed coastline during the 1990s than the 2010s (Figure 3.2). For example, 84% of captures occurred within 10 km of the smoothed coastline during 1993 – 1997, compared to 72% of captures during 2011 – 2013. Similarly, one independent bear was captured > 35 km from the smoothed coastline during the 1990s sampling, whereas 28 independent bears were sampled > 35 km from the smoothed coastline during the 2010s. The corresponding analysis using satellite telemetry found no significant differences in the distance of adult females from the smoothed Baffin Island coastline between the 1990s and 2010s; adult female bears on average in the 1990s were about a mean 17 km from the smoothed coast in August and September, where as in the 2000s they were about 13 km in those months, however standard errors were overlapping (Figure 3.4). Also, there were no differences in the mean monthly elevation used by adult females on Baffin Island between the 1990s and 2010s (Figures 3.4 and 3.5).

Satellite telemetry analyses further documented differences in the inland distance of adult females on Baffin Island as a function of reproductive status. Females that were most likely available to breed and become pregnant in spring (e.g., those captured alone, with 2 year old cubs, or as mating pairs in spring) were significantly farther inland in fall than adult females captured with COYs or yearling cubs (Table 3.2). This was especially pronounced for adult females captured in mating pairs (on average 27-35 km inland).

In contrast to analyses based on distance to the smoothed coastline, the distance of captures to the true coastline (not smoothed) was consistent between sampling periods (Figure 3.3), averaging 1.8 km (SD = 2.8) in the 1990s and 1.5 km (SD = 2.5) in the 2010s. This suggests that the difference in capture locations between the two sampling periods was largely due to less effort spent searching and capturing bears in the inland portions of fjords in the 1990s compared to the 2010s. For adult females, mean distances to the true coastline were 6.4 km (SD: 8.0) and 10.2 km (SD: 12.6) during the 1990s and 2010s, respectively (Figures 3.2 and 3.3).

Summary – The distribution of polar bear captures on Baffin Island differed significantly between sampling in the 1990s and 2010s. Specifically, the capture data indicate an under-representation of bears in fjords and inland regions during the 1990s (see also Chapter 5). Satellite telemetry location data, which were collected from independent bears over several years and were not influenced by which areas were searched in any given year, did not suggest a shift in the onshore distribution of polar bears. Given that no changes in adult female use of land habitats was detected (also see Chapter 4 terrestrial resource selection), the differences in capture distribution can be attributed to differences in sampling. During the 1990s, capture effort was concentrated on islands, along the outer coastline, and near the mouths of fjords (Figures 3.2 and 3.4). During the 2010s, these areas were searched as well as the inland portions of fjords. This is particularly prominent in central and northern Baffin Island, where no captures were recorded beyond the mouths of fjords during the 1990s. In contrast to the southern parts of Baffin Island the central and northern parts have a higher and more mountainous terrain. Finally, satellite telemetry data also indicate that adult females in different reproductive status show a non-random pattern of moving farther inland, likely in search of locations to construct maternal dens. These findings suggest a non-random probability of being a temporary emigrant as a function of the multi-year reproductive state. Taken together, these findings suggest that restricted geographic sampling in the 1990s likely led to higher probabilities of temporary emigration from the sampling area during that time period, compared to the 2010s. Furthermore, the probability of being a temporary emigrant appears non-random. Variable and non-random temporary emigration is known to introduce bias into estimates of survival and abundance under some conditions (Peñaloza et al. 2014).

Size of the Mark-Recapture Sampling Area on Baffin Island

Following from the previous section, we calculated the sizes of the effective MR sampling areas on Baffin Island in the 1990s and 2010s.

Methods – We delineated the sampling areas based on the spatial distribution of capture locations. We first used ArcGIS to create 99% kernel density contour around all capture locations in each time period. We then adjusted this contour on a point-by-point basis to ensure that the final estimated sampling area was within 1 km of the outermost capture locations. The sampling area did not extend offshore, except in a few cases in the 1990s where there were offshore points, in which case the boundary was kept within 1 km of those points. When capture locations occurred inside a fjord, it was assumed that sampling effort occurred everywhere from the mouth of that fjord to the capture location.

Results – The size of the MR sampling areas differed significantly between the 1990s and 2010s. The estimated sampling area was ~28,700 km^2 in the 1990s and ~60,200 km^2 in the 2010s. The 2010s sampling area included most fjords along the Baffin Island coast and reached farther inland than the 1990s (Figures 3.6 and 3.7). Furthermore, the 1990s sampled area was almost entirely contained within the 2010s sampling area (Figure 3.8). This made it possible to subsample the 2010s capture data, using the restricted 1990s sampling area, for the purpose of evaluating the influence of the size of sampling area on estimates of abundance from the two time periods (see Chapter 5).

Temporary Emigration Related to the Availability of Sea ice

Previous sections in this chapter documented a smaller onshore sampling area in the 1990s, which likely resulted in higher and potentially non-random temporary emigration from the sampling area in the 1990s. Here we evaluate temporary emigration related to the availability of sea ice, which declined between the two study periods in all months of the year, including the summer when sampling on Baffin Island occurred. We used satellite telemetry data to assess the fraction of adult females that were located in the sampling area vs. out of the sampling area (including on the sea ice) in the 1990s compared to the 2010s.

Methods – For each year of sampling in the 1990s and 2010s, we used the specific date range when sampling occurred (Table 3.3) to calculate the proportion of independent collared bears located inside the sampling area, as well as the proportion of locations from each individual bear that were inside the sampling area. First, we identified independent adult females that were wearing functional radio-collars during the sampling period. To ensure that location data were independent, we did not include locations from the same sampling period on which an adult female was captured and fitted with a radio-collar. For example, if a bear was captured and collared on October 1, 1993, locations from that individual through October 8, 1993 were not used (Table 3.3). However, locations from that individual in 1994 and 1995 were considered independent and included in analyses. If a bear was captured in spring of a given year, her location data were considered independent by fall of that year. We considered a bear to be located inside the sampling area if that bear had 1 (or more) telemetry location inside the sampling area.

We evaluated average sea-ice conditions in Baffin Bay during each sampling period for the 1990s and 2010s to determine whether bears that were located outside of the sampling area, were located on sea ice. For each sampling period, we mapped mean sea-ice concentration during the week that encompassed the mid-point of the sampling period, using the Passive Microwave data (SMMR/SSMI) sea-ice concentration dataset from the National Snow and Ice Data Center (see Chapter 4). We then superimposed independent bear locations on the sea-ice concentration map, and visually examined whether bears located outside of the sampled area were in an area with a substantial concentration of sea ice and therefore likely using the sea ice.

Results – Table 3.3 shows the date range of MR sampling in each year. There were a maximum of 13 independent adult female bears transmitting with satellite collars during the 1990s sampling periods. The number of individuals declined over the course of the 1990s study because most collars were deployed at the beginning of the study and some collars failed (Table 3.4). The largest number of transmitting independent bears occurred in 1993, and by 1997 there were none. There were also a maximum of 13 transmitting independent bears during a given sampling period in the 2010s, although sample sizes remained higher through the 2010s due to longer collar attachment periods (Table 3.4). We found large differences in the proportion of transmitting independent bears using the sampling areas between 1990s and 2010s. In the 1990s, 0-20% of females occurred within the sampling area during the MR sampling period (Table 3.4, Figure 3.9 - 3.11). In the 2010s, 67-80% of females occurred within the sampling area during the MR sampling period (Table 3.4, Figure 3.12 - 3.14).

Sea-ice availability in Baffin Bay declined between the 1990s and 2010s. In the 1990s, a substantial amount of sea ice was available in offshore central Baffin Bay; within the Canadian archipelago, including around Devon Island; and in Lancaster Sound and Kane Basin (Figures 3.15-3.21). In 1993, when the largest proportion of independent bears was offshore during the sample period (Figure 3.15), there was a persistent area of sea ice available in central Baffin Bay. In other years in the 1990s, some bears were located on the advancing sea ice forming in northern Baffin Bay (Figures 3.15-3.17). In contrast, in the 2010s all bears (excluding resident bears in Melville Bay) were distributed on land on Baffin Island or in Kane Basin (Figures 3.18-3.20) during the sampling periods. There were no bears on offshore ice in the 2010s, because sea ice had melted completely in central Baffin Bay by July (see Chapter 4). The differences in sea-ice conditions between the 1990s and 2010s can been seen clearly using juxtaposed sea-ice concentration maps (Figure 3.21).

In addition to relatively fewer adult females being present in the sampling area during the 1990s, most bears with >1 location in the sampling area did not spend the entire sampling period there, but rather were passing through (Table 3.5). In the 1990s, approximately 44% of locations received for bears that used the sampling area, were located inside the sampled area (see Chapter 1 for information on location filtering and subsampling). In the 2010s, approximately 94% of locations received for bears that used the sampling area, were located inside the sampled area. Although sample sizes were small and unevenly distributed across years, the higher probability of bears in the 1990s being located outside the sampling area appeared largely due to the presence of sea ice, whereas in the 2010s sea ice was absent and bears exhibited reduced summertime movement rates (see Chapter 4).

Summary – Temporary emigration from the sampling area during the autumn sampling period has the potential to introduce bias into estimates of demographic parameters from this study. Our analyses suggest that the proportion of adult females (and presumably other sex and age classes) in the sampling area was likely lower in the 1990s compared to the 2010s, for two reasons. First, some bears located inland in the 1990s were not available to capture teams because there was apparently limited inland search effort, and in particularly bears were not captured in the deep inland portions of fjords. Furthermore, the location of bears from the coast—and therefore the susceptibility of bears to capture—appeared related to reproductive status, in which case the probability of being a temporary emigrant may have been nonrandom. Second, a proportion of radio-collared polar bears used offshore ice in the 1990s, whereas sea ice was less available in the 2010s and therefore a substantially higher proportion of bears were likely inside the sampling area. Because of small sample sizes that varied across years, we were unable to calculate precise estimates of temporary emigration rates or to evaluate the magnitude and direction nonrandom patterns (e.g., Markovian dependence) in a statistically rigorous manner. Nonetheless, multiple lines of evidence indicate higher temporary emigration in the 1990s, compared to the 2010s. The most likely effect of temporary emigration is an unknown but potentially meaningful negative bias in estimates of survival and abundance (Schaub et al. 2004, Devineau et al. 2006, Peñaloza et al. 2014).

Additional sampling considerations

Small sample sizes lead to multiple challenges into MR studies, including high variance in estimated parameters, small-sample bias, susceptibility to bias due to violation of modeling assumptions (e.g., un-modeled heterogeneity in recapture probability), and limited options for quantifying or mitigating bias (Williams et al. 2002). Compared to the 2010s data, sample sizes in the 1990s were small and had a low proportion of recaptures (Table 3.1). For example, the entire dataset for adult females (F2+ age group) included only 5 animals recaptured by standard search in 1995, and 14 animals recaptured by standard search in 1997 (note that numbers in Table 3.1 are higher, because they include "likely" recaptures and re-sightings of bears located by radio telemetry; see below). Furthermore, there were relatively few dead recoveries during the interim period when no sampling occurred (1998-2010), particularly in the later years. For example, an average of 1.3 research-marked females per year were recovered in the harvest, from 1998-2010. Conceptually, it is apparent that the small number of live recaptures during 1990s live-encounter sampling, the gap years between 1990s and 2010s sampling, and the small number of dead recoveries during the gap years contain a limited amount of information and will lead to estimates of demographic parameters that have substantial uncertainty and low resolution (i.e., that few demographic parameters can be estimated, requiring the estimation of "average" parameters over years or groups of animals).

There were significant differences in the composition of the MR samples (i.e., the proportion of bears within each age-sex class, based on initial captures) between the 1990s and 2010s in Baffin Bay (Table 3.1). There were more adult and sub-adult male captures in the 1990s, whereas there were more sub-adult and adult female captures in the 2010s. The proportion of total female captures in the1990s was less than the 2010s (mean annual proportion of age 2+ female captures : total 2+ captures, 1990s: 0.42; 2010s: 0.53; Table 3.1). Given the spatial segregation of bears by sex and age-classes and reproductive states (see section Distribution of Mark-recapture Sampling on Baffin Island), the apparent under-representation of females in the 1990s samples likely reflects at least in part the coastal-focused sampling protocols during that period, rather than true differences in the composition of the subpopulation (although we cannot rule out progressive depletion of males through the 2010s due to high harvest).

Development of an Individual Covariate to Explain Inland Habitat Use

Given the apparent differences in sampling effort between the 1990s and 2010s, the spatial segregation of bears by sex and age class, and differences in the composition of capture samples, we hypothesized that proximity to the coastline may explain variation in recapture probabilities. We also wanted to explore whether proximity to the coastline for an individual bear was nonrandom across years (e.g., whether bears captured inland were more likely to be recaptured inland). We assigned capture locations to either coastal or inland categories, using a threshold of 2 km from true and smoothed coastlines, and compiled contingency tables for individuals captured in multiple sampling periods. For individuals captured three or more times, we used only an individual's first two capture events and included only those bears initially captured as independent animals, since the locations of cubs-of-the-year and yearlings were dependent on the location of their mothers.

Use of inland areas appeared nonrandom. Individual polar bears initially captured inland from the true coastline were more likely to be recaptured inland in subsequent years (all data: $\chi^2 = 10.4$, $P = 0.0012$; 1990s only: Fisher's exact test $P = 0.10$; 2010s only: Fisher's exact test $P = 0.02$). Similarly, bears initially captured inland of the smoothed coastline were more likely to be recaptured inland (all data $\chi^2 = 18.1$, $P < 0.0001$), a pattern which was driven largely by the 2010s (Fisher's exact test $P < 0.0001$; 2010s only: Fisher's exact test $P = 0.21$; 1990s only). As such, we incorporated a proximity to coastline covariate for modeling recapture probability in demographic analyses (see Chapter 5).

Challenges with Using the 1990s Radio Telemetry Data

Some aspects of the 1990s radio-telemetry data were uncertain or unavailable, presenting challenges to the use of these data in the current analysis. As part of a study examining subpopulation delineation and spatial ecology (Ferguson et al. 1997, Taylor et al. 2001), a sample of adult female polar bears was fitted with satellite radio-collars in Baffin Bay (from both Canada and Greenland) during the 1990s. Some of these bears ($n = 14$) were captured on Baffin and Bylot Islands during autumn 1993 – 1997. Taylor et al. (2005) report that collared bears and their dependent young were often relocated using VHF during the 1990s study period. The probability of locating and recapturing a bear with a collar is likely higher than the probability of recapturing a bear without a collar. Therefore, a radio telemetry covariate, describing whether a bear was wearing a functional radio-collar that could have allowed it to be located by telemetry, was important for explaining variation in recapture probabilities; and all of the most-supported models in the 1990s included a radio telemetry covariate (Taylor et al. 2005). Taylor et al. (2005:209) reported that “The probability of autumn recapture was lower for females and yearling cubs than for adult males and sub-adults, except for radio-collared females and their young” which indicates that radio-collared females were recaptured using radio-location data. Unfortunately, the data archives did not include complete information on which bears were wearing functional radio-collars and located using VHF. Furthermore, in some cases where records could be located, there were inconsistencies among databases and historical hard-copy files. This presented a challenge to MR modeling because the live-capture data in the 1990s were sparse, particularly for adult females, and we anticipated that the additional records for bears likely recaptured using VHF would be important for explaining patterns in survival and recapture probability (see Taylor et al. 2005). To address this issue, we manually reviewed capture histories and covariates compiled for the previous Baffin Bay analysis. We compared these historical files with our available records to identify events in which a bear was likely located via VHF (see also Chapter 5). Based on this, we added 7 recapture events of 5 age 2+ individuals previously in the dataset, and 6 capture events of 5 age 2+ individuals not previously included in the dataset. We believe that this protocol accurately incorporated most of the data for polar bears captured by VHF in the 1990s, although some uncertainty remains given that the original data were not available.

Ramifications of Issues with the 1990s Baffin Bay Data

It is difficult to estimate demographic parameters and detect trends in parameters, for long-lived animals using short time-series of live-encounter data, especially when recapture rates are low, environmental variation is high, and the entire study subpopulation is not exposed to sampling effort on each occasion (Williams et al. 2002). The analyses described above identify specific challenges with 1990s Baffin Bay MR data that arise from both sampling issues and environmental factors. These challenges may lead to bias in estimates of survival and abundance, and ambiguity in the definitions of parameters being estimated (e.g., whether a model is estimating apparent survival, which reflects emigration from the study subpopulation, or true survival).

Survival – A statistical assessment of trends in polar bear survival between the 1990s and 2010s is not possible due to the short duration of live-encounter sampling periods, the large gap between 1990s and 2010s live-encounter sampling, low recapture probabilities, low numbers of dead recoveries, changes in the sampling area between the 1990s and 2010s, and evidence for changes in polar bear movements with respect to the sampling area. This conclusion was supported by computer simulations (T. Arnold, University of Minnesota, unpubl data) in Program MARK to generate datasets that resembled the actual Baffin Bay data but included a known effect (e.g., large reduction in survival), and evaluating the power of MR model to detect such effects (T. Arnold, University of Minnesota, unpublished data). In the context of small and variable sample sizes, a primary challenge for estimating survival is the difficulty of delineating temporary vs. permanent emigration from the study area, and the effects of emigration on estimates of survival. MR modeling was performed using Burnham models, which assume that emigration from the study subpopulation is permanent. Burnham models directly estimate the probability of permanent emigration (*F*) based on patterns in live-encounter data in conjunction with harvest data collected from an area that is larger than the MR sampling area. Under the Burnham model, the survival parameter (S) is technically defined as true survival (i.e., does not include an emigration component). However, research-marked bears that are harvested outside the sampling area may be temporary rather than permanent emigrants (i.e., the bears could have returned to the sampling area in future years, if they had not been killed), and the short duration of the study, small sample sizes, and likely high interannual variability in the probability of being a temporary emigrant (e.g., as related to sea-ice availability) make it difficult to delineate temporary vs. permanent emigration. Simulations suggested that the Baffin Bay data were too sparse to fit Barker models, which relax the assumption that emigration is permanent, and are capable of estimating temporary emigration rates, including non-random temporary emigration. The consequence of using Burnham models either with *F* estimated or with *F* fixed = 1 (i.e., assuming no permanent emigration if *F* is estimated), is that variation across individuals and sampling occasions in the probability of being a temporary emigrant is not explicitly accounted for, and therefore exists as variation in recapture probabilities. Heterogeneity in recapture probabilities has the potential to introduce bias into estimates of S (Schaub et al. 2004). The directionality of bias is often negative and its magnitude tends to increase in the final years of a study (Devineau et al. 2006). Furthermore, non-random patterns in temporary emigration are known to cause bias in estimates of survival (Kendall et al. 1997), and the availability of adult females for capture in the 1990s was related to their multi-year reproductive cycle. Interpretation of trends in survival between the 1990s and 2010s is further complicated because radio-telemetry data suggest changes in fidelity to the MR sampling study area between the epochs, and because the geographic extent of the MR study area itself changed. We conclude that estimates of survival from the current MR analysis of Baffin Bay data must be interpreted with caution. Although estimates of survival provide the basis for discussion and ecological interpretation, they are unlikely to be directly comparable between the 1990s and 2010s, and will require further analysis (e.g., regarding different assumptions about movements between epochs) if used in matrix-type models for subpopulation projections.

Abundance – Estimating abundance is one of the more difficult challenges in wildlife management (Williams et al. 2002). Deriving accurate estimates of abundance and evaluating trends in abundance over time require an appropriate study design and, especially, consistent distribution of sampling effort in time and space. In the current study, the difference between the distributions of captures in the 1990s and 2010s suggest that the sampling area on Baffin Island expanded substantially from the 1990s to the 2010s. Specifically, sampling was spatially restricted to a portion of the subpopulation's fall range during the 1990s, thus excluding bears with seasonal fidelity to inland areas. Furthermore, an unknown but potentially significant portion of the Baffin Bay subpopulation may not have been exposed to sampling in the 1990s due to the higher presence of sea ice, which some bears used throughout the year rather than coming onto land. We conclude that the abundance estimate in the 2010s, based on MR data from the entire sampling area, is not directly comparable to the previous 1990s abundance estimate. To investigate the extent to which differences in sampling affected abundance estimates from the 1990s and 2010s, we used the 1990s sampling area to create a subset of the 2010s data, and subsequently derived a 2010s abundance estimate based on this restricted subset of the data. We included only those 2011 – 2013 capture events that were located within the estimated 1990s sampling frame and completed supplemental demographic analyses (see Chapter 5). This analysis helped evaluate the potential biases associated with the more restricted area of onshore sampling on Baffin Island in the 1990s. However, it did not address the potential effects of polar bears using the sea ice in the 1990s. When there is temporary emigration from the sampling area, estimates of abundance from Burnham models represent the "superpopulation" (defined as all animals with a probability of moving through the sampling area, even if not every animal was actually in the sampling area on every sampling occasion). If temporary emigration from the sampling area is completely random, it will not introduce bias into estimates of abundance. However, nonrandom temporary emigration (e.g., if some individuals are often or always temporary emigrants) has a similar effect on estimates of demographic parameters from MR models as un-modeled heterogeneity in recapture probability, and generally introduces negative bias into estimates of abundance (Kendall et al. 1997).

MR model covariates – 1990s sampling bias may also impact the individual, geographic fidelity covariate (proximity to smoothed coastline). Analyses did not suggest a significant relationship between initial and subsequent capture locations in the 1990s, but this may be due to sampling (e.g., not enough effort was expended inland, to identify animals with fidelity to inland areas). The relationship is driven by the 2010s data. Also, the radio telemetry covariate may be biased in some unknown direction due to the uncertainty as to whether the subsequent capture of a collared bear was facilitated by the radio tracking. Sensitivity analyses outlined above may help better understand potential biases. Given the differences between the 1990s and 2010s, including epoch effects for the binary 'proximity to smoothed coastline' is important.

3.2. Literature Cited

Born, E.W. and R. Dietz. 2009. Polar bear *Ursus maritimus*. Pages 81-88 *in* D. Boertmann, A. Mosbech, D. Schiedek, and K. Johansen, editors. The Eastern Baffin Bay. A preliminary strategic environmental impact assessment of hydrocarbon activities in the KANUMAS

West area. NERI Technical Report No. 720, National Environmental Research Institute, Aarhus University, Denmark. http://www.dmu.dk/Pub/FR720.pdf.

Born, E. W., A. Rosing-Asvid, and D. Andriashek. 1992. Tagging of polar bears in NW Greenland, 2-19 May 1992. Report on field work. Unpublished report, Greenland Fisheries Research Institute, Copenhagen, Denmark, 6 pp. Available at Greenland Institute of Natural Resources, P.O. Box 570, 3900 Nuuk, Greenland.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Devineau, O., R. Choquet, and J-D. Lebreton. 2006. Planning capture-recapture studies: straightforward precision, bias, and power calculations. Wildlife Society Bulletin 34:1028–1035.

Ferguson, S. H., M. K. Taylor, E. W. Born, A. Rosing-Asvid, and F. Messier. 1999. Determinants of home range size in polar bears. Ecology Letters 2:311–318.

Ferguson, S. H., M. K. Taylor, E. W. Born, A. Rosing-Asvid, and F. Messier. 2001. Activity and movement patterns of polar bears inhabiting consolidated versus active pack ice. Arctic 54:49–54.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use by polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock’s robust design. Ecology 78:563–578.

Lukacs, P. M., and K. P. Burnham. 2005. Review of capture–recapture methods applicable to noninvasive genetic sampling. Molecular Ecology 14:3909–3919.

Peñaloza, C. L., W. L. Kendall, and C. A. Langtimm. 2014. Reducing bias in survival under nonrandom temporary emigration. Ecological Applications 24:1155-1166.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:3–97.

Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117–127.

Rosing-Asvid, A. 1993. Tagging of polar bears in NW Greenland, May 1993. Report on field work. Unpublished report, Greenland Fisheries Research Institute, Copenhagen, Denmark, 8 pp. Available at Greenland Institute of Natural Resources, P.O. Box 570, 3900 Nuuk, Greenland.

Royle, J. A., R. B. Chandler, R. Sollmann, and B. Gardner. 2014. Spatial Capture-Recapture. First edition. Academic Press, Waltham, Massachusetts, USA.

Schaub, M., O. Gimenez, B. R. Schmidt, and R. Pradel. 2004. Estimating survival and temporary emigration in the multistate capture–recapture framework. Ecology 85:2107–2113.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA.

Table 3.1. Summary table of live captures and dead recoveries during the mark-recapture study of the Baffin Bay polar bear subpopulation in Nunavut, Canada, and Greenland, 1993 – 2010. Shaded cells indicate that data were not possible due to an absence of marking or recapture.

	Initial captures						**Live recaptures**				**Dead recoveries**					
	Females			*Males*			*Females*		*Males*		*Females*			*Males*		
	Coy	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Yrl**	**2+**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**
1993	14	8	53	12	8	61					0	0	1	0	0	0
1994	26	13	65	16	9	77	0	5	0	14	0	0	3	0	0	7
1995	15	11	62	19	11	85	4	11	4	23	0	2	6	1	0	8
1996												1	8		0	7
1997	22	10	60	19	13	113		20		31	0	0	6	0	1	9
1998												0	3		0	11
1999													3			9
2000													0			8
2001													2			8
2002													0			11
2003													0			7
2004													1			7
2005													2			3
2006													3			6
2007													1			2
2008													2			4

2009													2			0
2010													0			1
2011	2	23	163	1	20	148		5		5	0	0	4	0	0	20
2012	40	30	221	35	30	192	3	41	0	54	0	0	8	0	2	14
2013	28	15	121	16	15	90	4	48	5	55	0	1	8	1	0	20
Totals	**147**	**110**	**745**	**118**	**106**	**766**	**11**	**130**	**9**	**182**	**0**	**4**	**63**	**2**	**3**	**162**

Table 3.2. Metrics for adult females satellite collared in the 1990s (fall) and 2010s (spring) for the distance inland from the outer Baffin Island coast. Distance is reported in km.

		August			**September**			**October**		
Adult Female Accompanied by	**N**	**Mean distance inland**	**SD**	**Count of locations**	**Mean distance inland**	**SD**	**Count of locations**	**Mean distance inland**	**SD**	**Count of locations**
1990s										
2YR	1							5.6	4.4	3
AM	0									
COY	15	10.8	14.2	10	19.7	15.1	13	9.6	5.9	46
YRL	12	6.0	5.6	3	18.0	17.8	25	8.4	5.8	41
ALONE	5				8.8	9.4	6	13.4	11.8	8
2010s										
2YR	5	13.5	9.3	25	27.0	8.3	20	16.2	13.5	11
AM -in spring	2	27.1	10.2	9	32.6	12.0	5	35.1	9.3	4
COY	2	5.5	4.8	3	7.5	4.3	13	3.6	4.3	11
YRL	7	3.5	5.0	20	6.8	6.2	33	6.2	7.3	27
ALONE	6	11.9	10.8	25	16.3	7.6	22	14.4	7.8	18

Table 3.3. Time periods when the BB fall sampling period occurred in each decade. These dates were used to asses if independent bears were in or out of the sampled area.

Year of sampling	Start	End
1993	23 August	8 October
1994	7 September	19 October
1995	17 September	19 October
1996	n/a	n/a
1997	21 September	29 October
2011	4 September	14 October
2012	26 August	29 September
2013	20 August	11 October

Table 3.4. The overall fraction of independent collared adult female (AF) bears found in the sampling range by year.

Year of sampling	n independent collared bears	n independent AF bears in the sampled area (minimum of n=1 location during date range)	% independent AF bears in the sampled area for each decade
1993	13	3	23
1994	5	1	20
1995	1	0	0
1997	0		
2011	12	8	67
2012	13	11	85
2013	6	4	67

Table 3.5. Independent BB adult female bears with satellite collars transmitting during the MR sample periods. Bears listed are only those that used the sampled area on Baffin Island for each decade. The fraction of locations inside the sampled area is shown for each bear.

YEAR + capture season	**ID (PTT + Year)**	**Start Date**	**End Date**	**n independent bears during this year**	**n independent bears in the sampled area**	**Fraction of total locations inside sampled area during the sampling dates**	**Proportion of locations**
1993		**23-Aug**	**8-Oct**	**13**	**3**		
fall	199111062	27-Aug	16-Sep			1/4	0.25
spring	19922718	24-Aug	7-Oct			1/8	0.13
fall	19922700	25-Aug	25-Aug			1/1	1.00
1994		**7-Sep**	**19-Oct**	**5**	**1**		
spring	19922701	8-Sep	6-Oct			2/6	0.33
2011		**4-Sep**	**14-Oct**	**12**	**8**		
spring	201068010	6-Sep	8-Oct			8/9	0.89
spring	2011105814	24-Sep	10-Oct			4/4	1.00
spring	201074768	6-Sep	12-Oct			8/8	1.00
spring	2011105809	6-Oct	6-Oct			1/1	1.00
spring	200974767	6-Sep	12-Oct			10/10	1.00
spring	2011105817	4-Sep	14-Oct			10/10	1.00
spring	2011105816	4-Sep	14-Oct			5/5	1.00

spring	200968005	6-Sep	12-Oct			10/10	1.00
2012		**26-Aug**	**29-Sep**	**13**	**11**		
spring	201074774	29-Aug	26-Sep			6/7	0.86
spring	2012105829	29-Aug	26-Sep			5/7	0.71
spring	201068010	12-Sep	24-Sep			2/3	0.67
spring	2011105814	29-Aug	26-Sep			5/7	0.71
spring	201074768	27-Aug	28-Sep			7/8	0.88
spring	2011105808	29-Aug	26-Sep			7/8	0.88
spring	2011105809	6-Sep	6-Sep			1/1	1.00
spring	200974767	27-Aug	28-Sep			1/9	1.90
spring	200974771	29-Aug	26-Sep			8/8	1.00
spring	2011105813	29-Aug	22-Sep			2/6	0.33
spring	200968005	27-Aug	27-Aug			1/1	1.00
2013		**20-Aug**	**11-Oct**	**6**	**4**		
spring	2013105818	20-Aug	11-Oct			12/12	1.00
spring	2013128265	20-Aug	11-Oct			14/14	1.00

Figure 3.1. Locations of polar bears sampled in Baffin Bay during the 1990s (August – October, 1993 – 1995, 1997, red) and 2010s (August – October, 2011 – 2013, blue). Sampling in Greenland in the 2010s occurred near Melville Bay but is not shown. Note the absence of captures in fjords on Baffin Island during the 1990s in the inset.

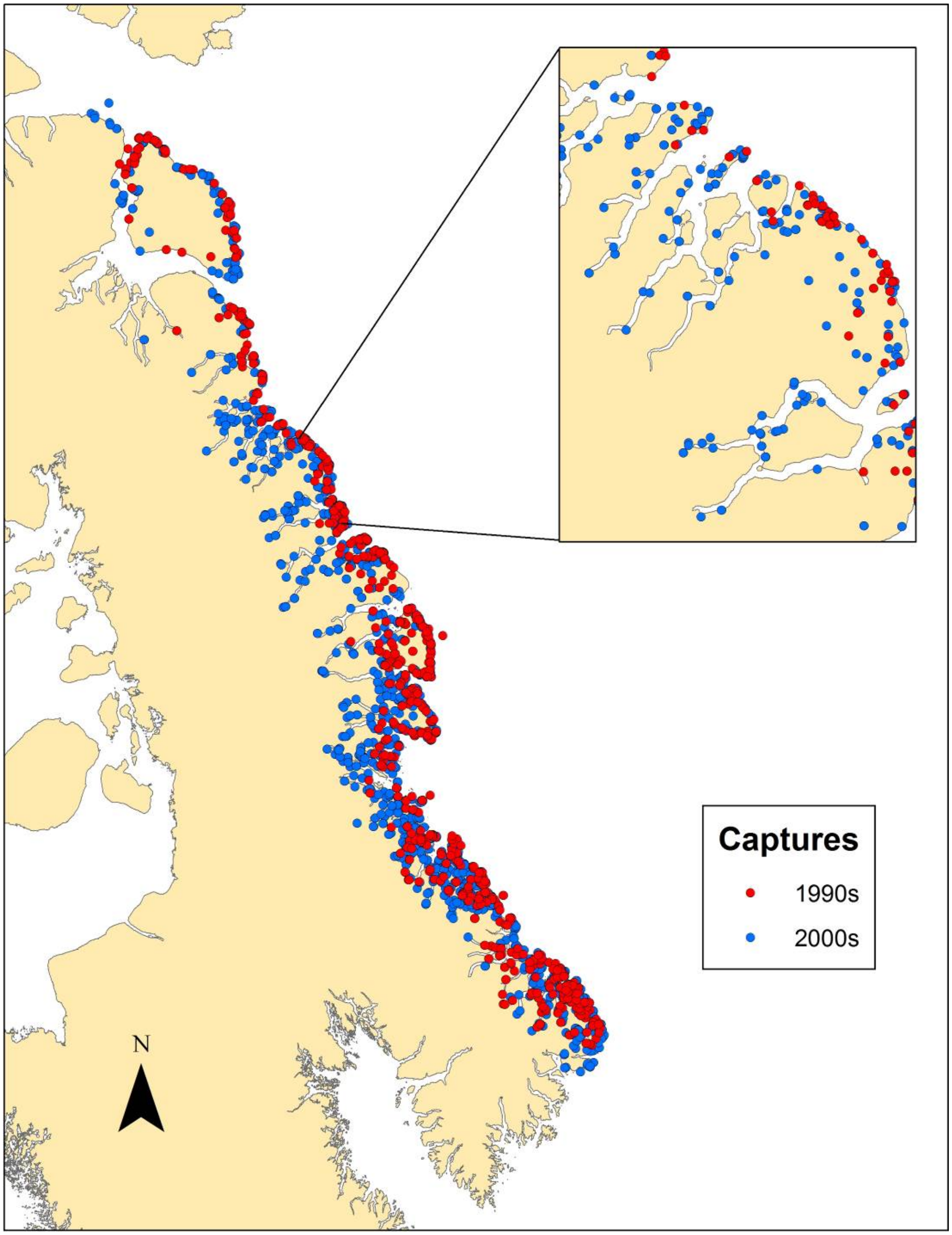


Figure 3.2. Distances independent bears were captured from the smoothed coastlines of Baffin and Bylot Islands during fall-time sampling in the Baffin Bay subpopulation, 1993 – 1997 and 2011 – 2013.

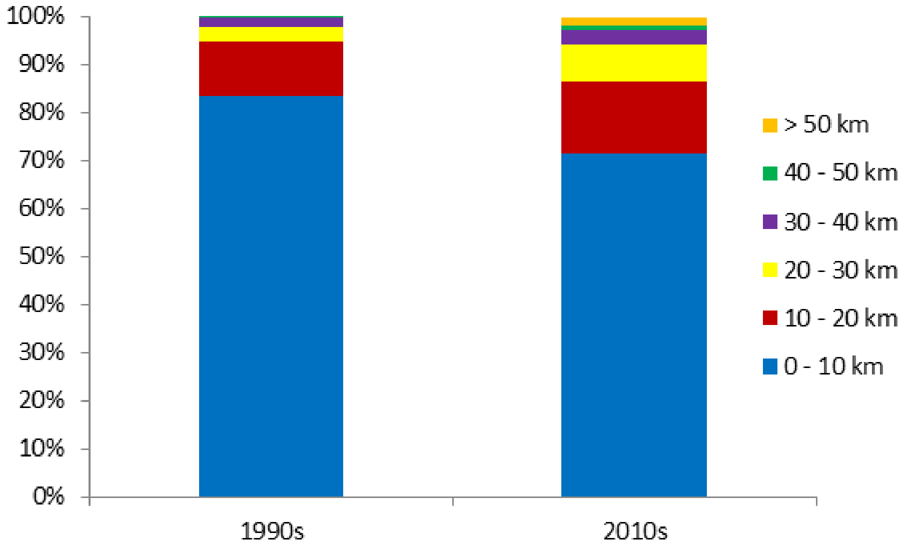


Figure 3.3. Distances independent bears were captured from the true coastlines of Baffin and Bylot Islands during fall-time sampling in the Baffin Bay subpopulation, 1993 – 1997 and 2011 – 2013.

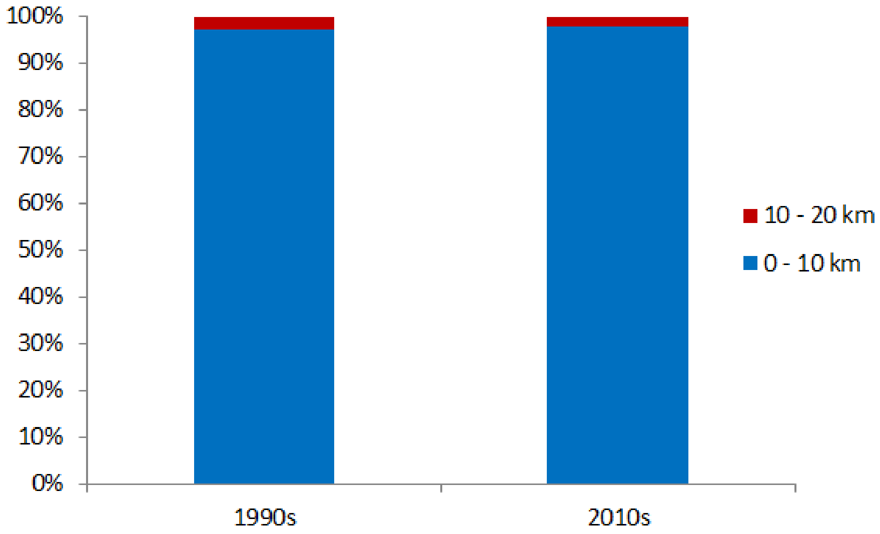


Figure 3.4. Distance to smoothed Baffin Island coastline shown in all summer months using satellite telemetry data from adult females in the 1990s (red) and 2010s (blue) located on Baffin Island. Shaded regions represent 2 SE from the mean. Numbers above represent numbers of telemetry locations for each month. There was no difference in distance inland (or distance to the outer Baffin Island coast) between decades.

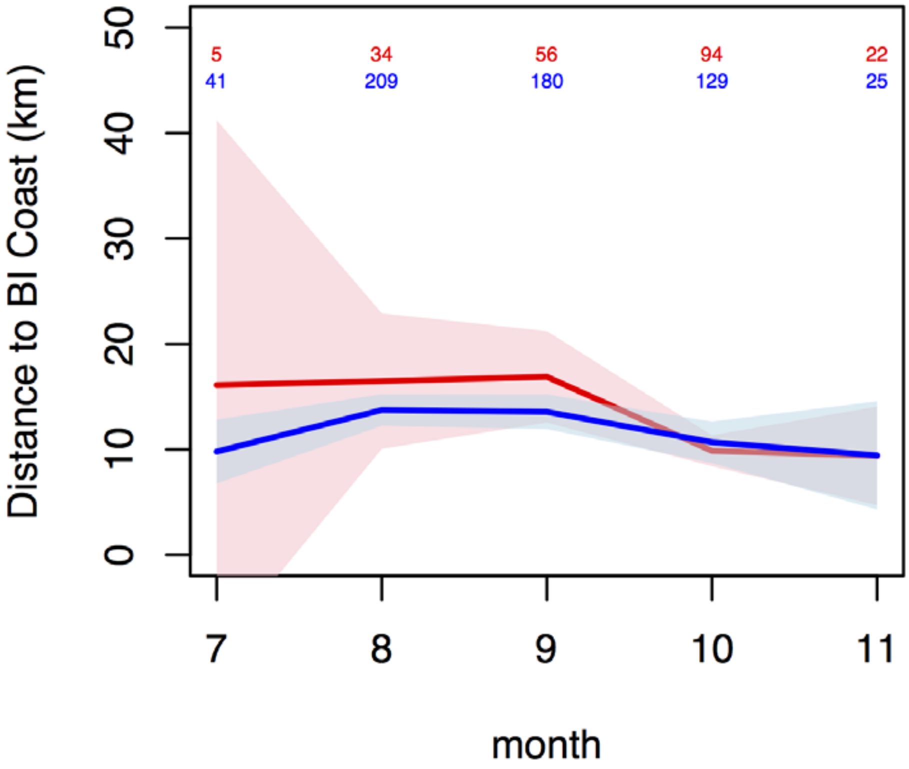


Figure 3.5. Elevation of adult female polar bears on Baffin Island shown in all summer months using satellite telemetry data from the 1990s (red) and 2010s (blue). Shaded regions represent 2 SE from the mean. Numbers above represent numbers of telemetry locations for each month. There was no difference in elevations used by polar bears across months between decades.

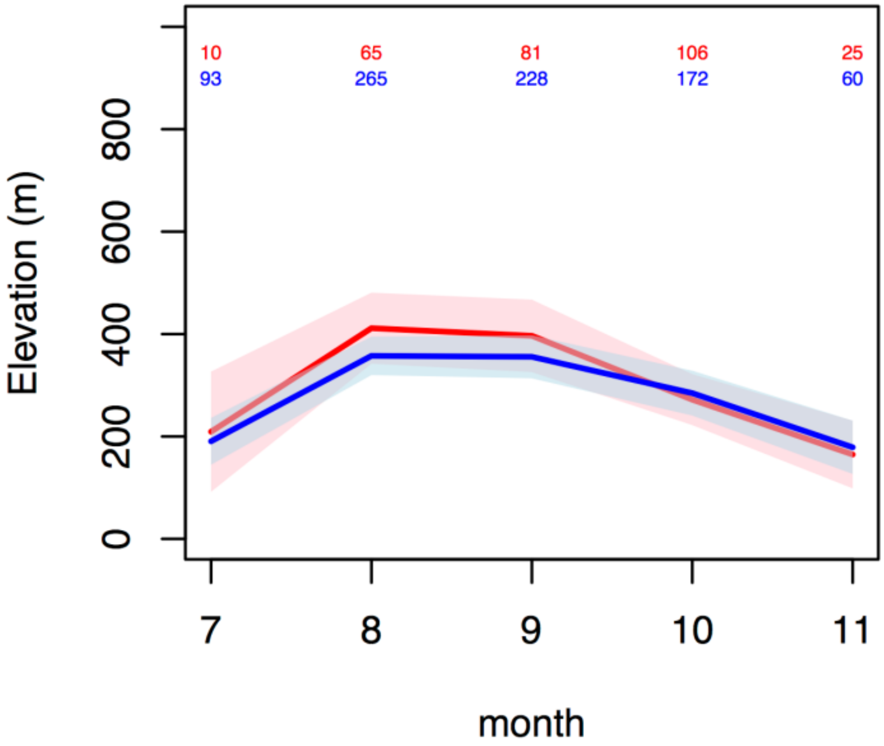


Figure 3.6. The delineation of the sampled area shown with a red outline for the 1990s with capture locations collected during the MR sampling.

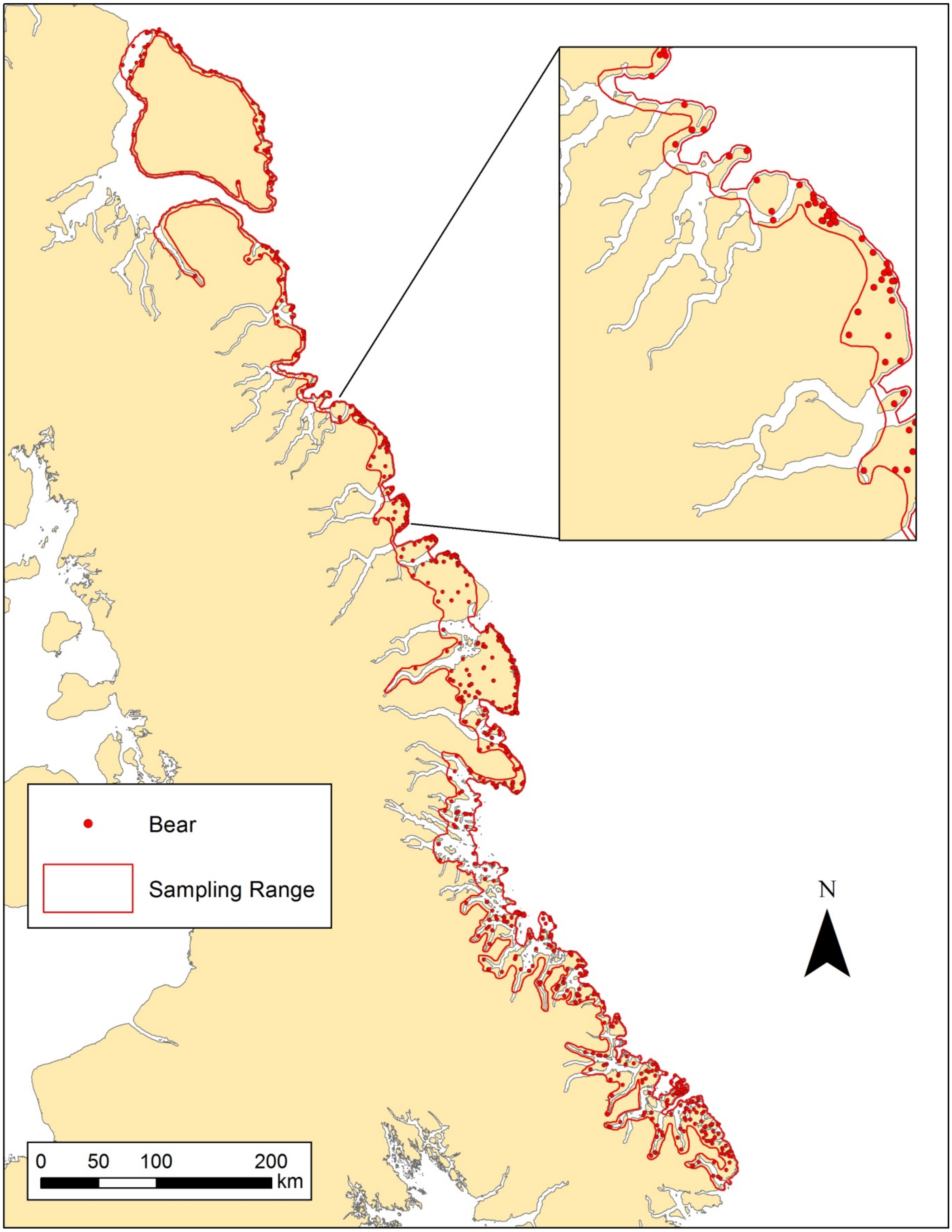


Figure 3.7. The delineation of the sampled area shown with a red outline for the 2010s with biopsy locations collected during the MR sampling (2011-2013).

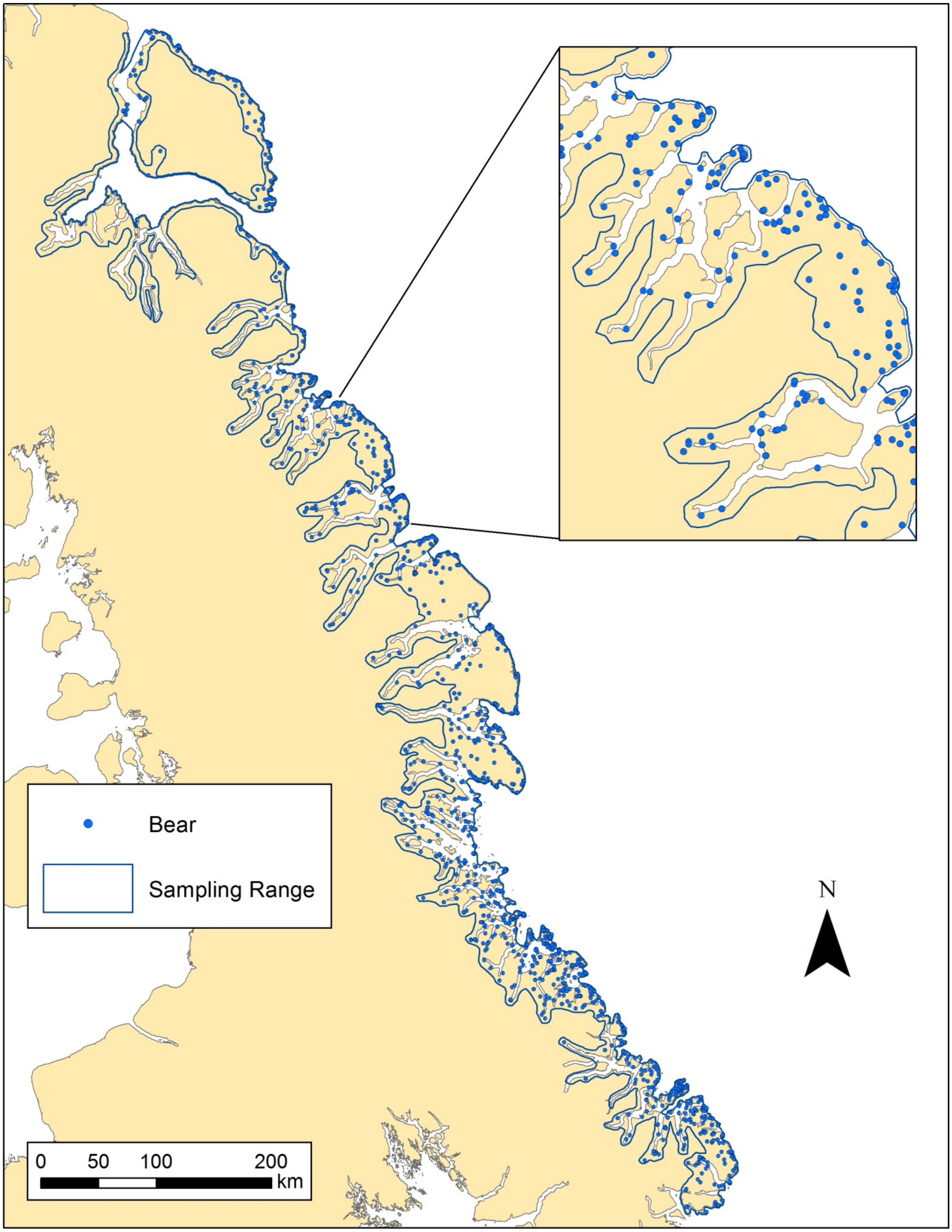


Figure 3.8. Geographic sampling ranges for the MR in the 1990s and 2010s.

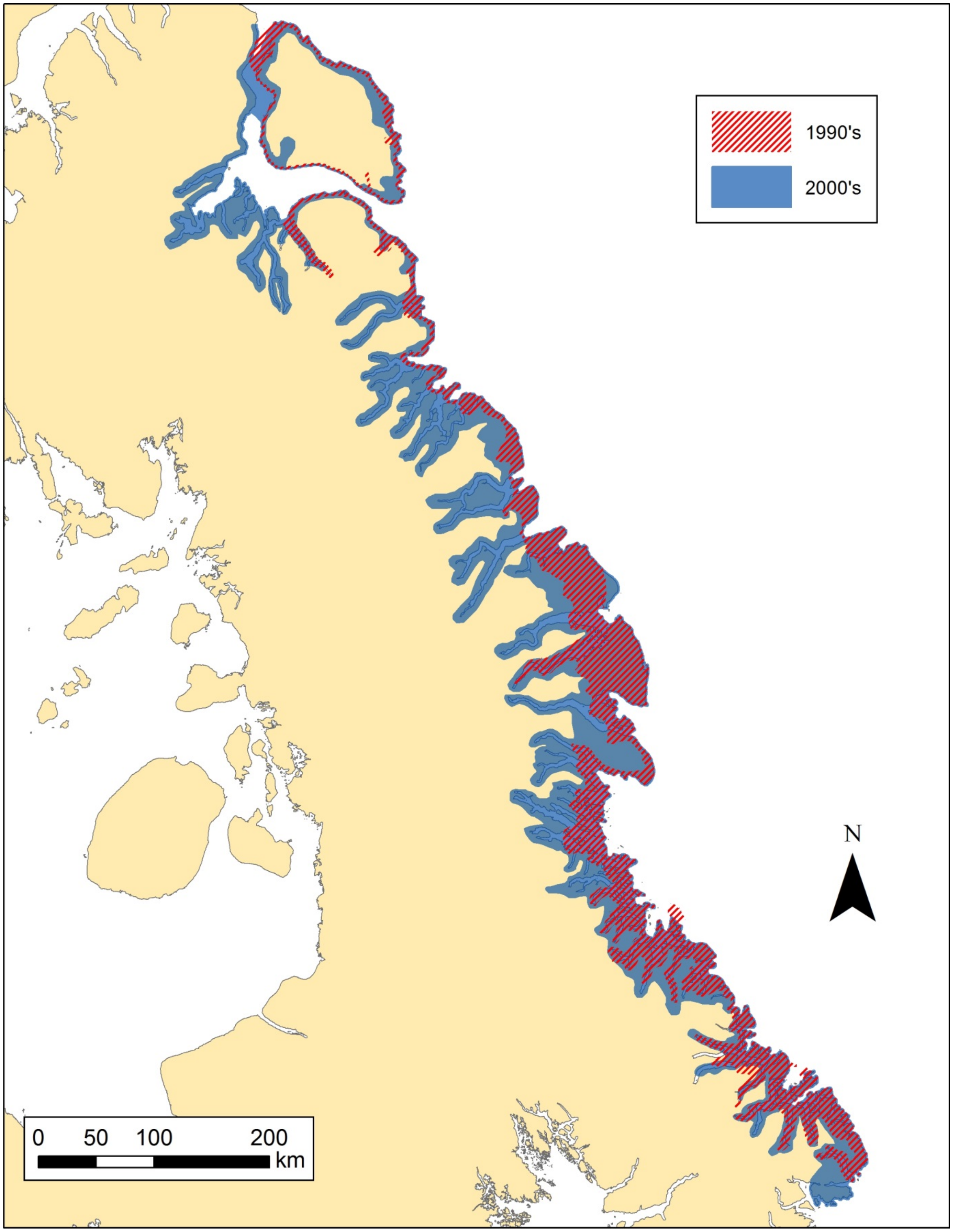


Figure 3.9. All telemetry locations from independent adult female bears with satellite collars transmitting during the 1993 sampling period dates (See Table 3.3). The 1990s sampled area for the MR study is shown in the red outline. Bears in central BB are on sea ice (Figure 3.15).

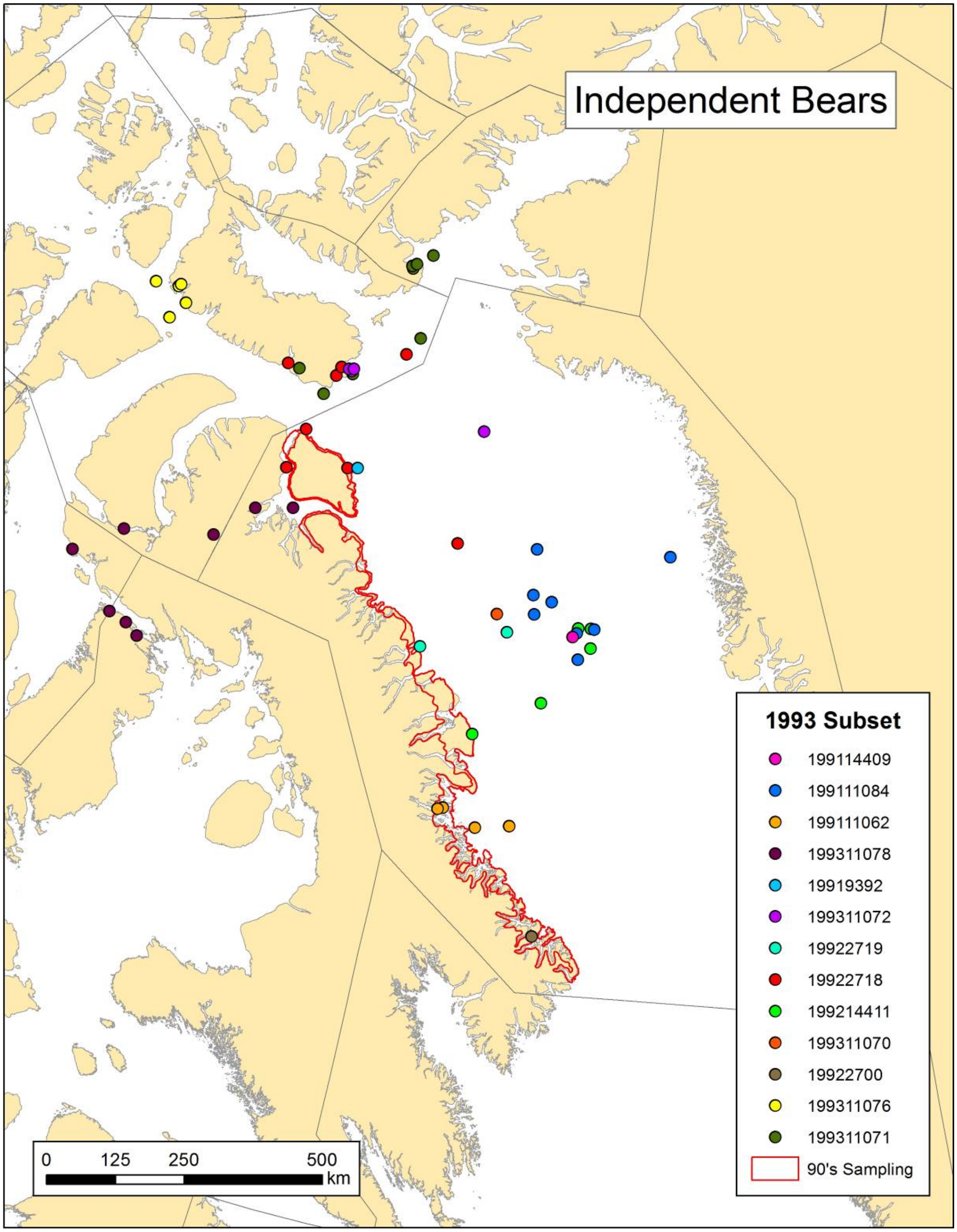


Figure 3.10. All telemetry locations from independent adult female bears with satellite collars transmitting during the 1994 sampling period dates (See Table 3.3). The 1990s sampled area for the MR study is shown in the red outline.

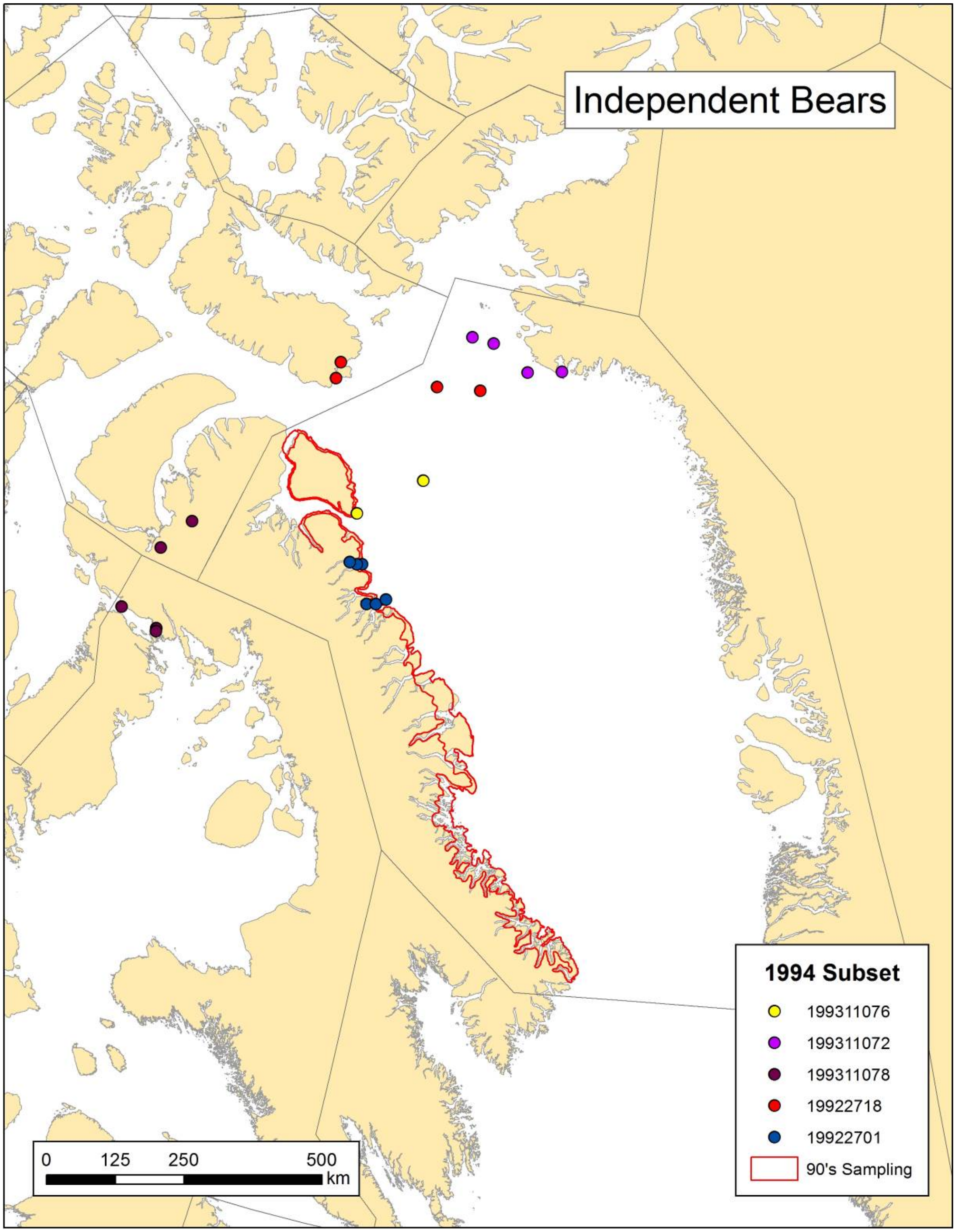


Figure 3.11. All telemetry locations from independent adult female bears with satellite collars transmitting during the 1995 sampling period dates (See Table 3.3). The 1990s sampled area for the MR study is shown in the red outline.

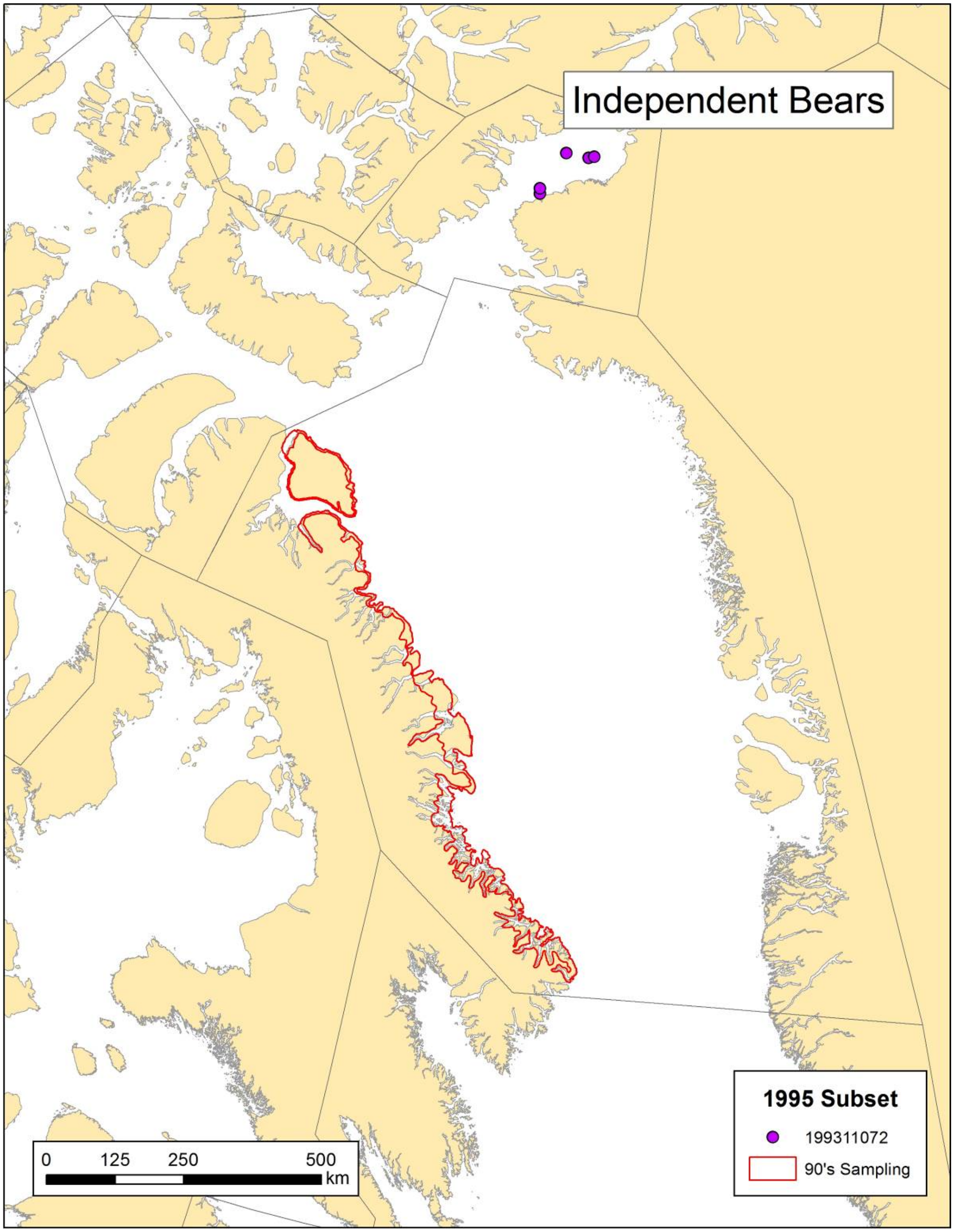


Figure 3.12. All telemetry locations from independent adult female bears with satellite collars transmitting during the 2011 sampling period dates (See Table 3.3). The 2010s sampled area for the MR study is shown in the blue outline.

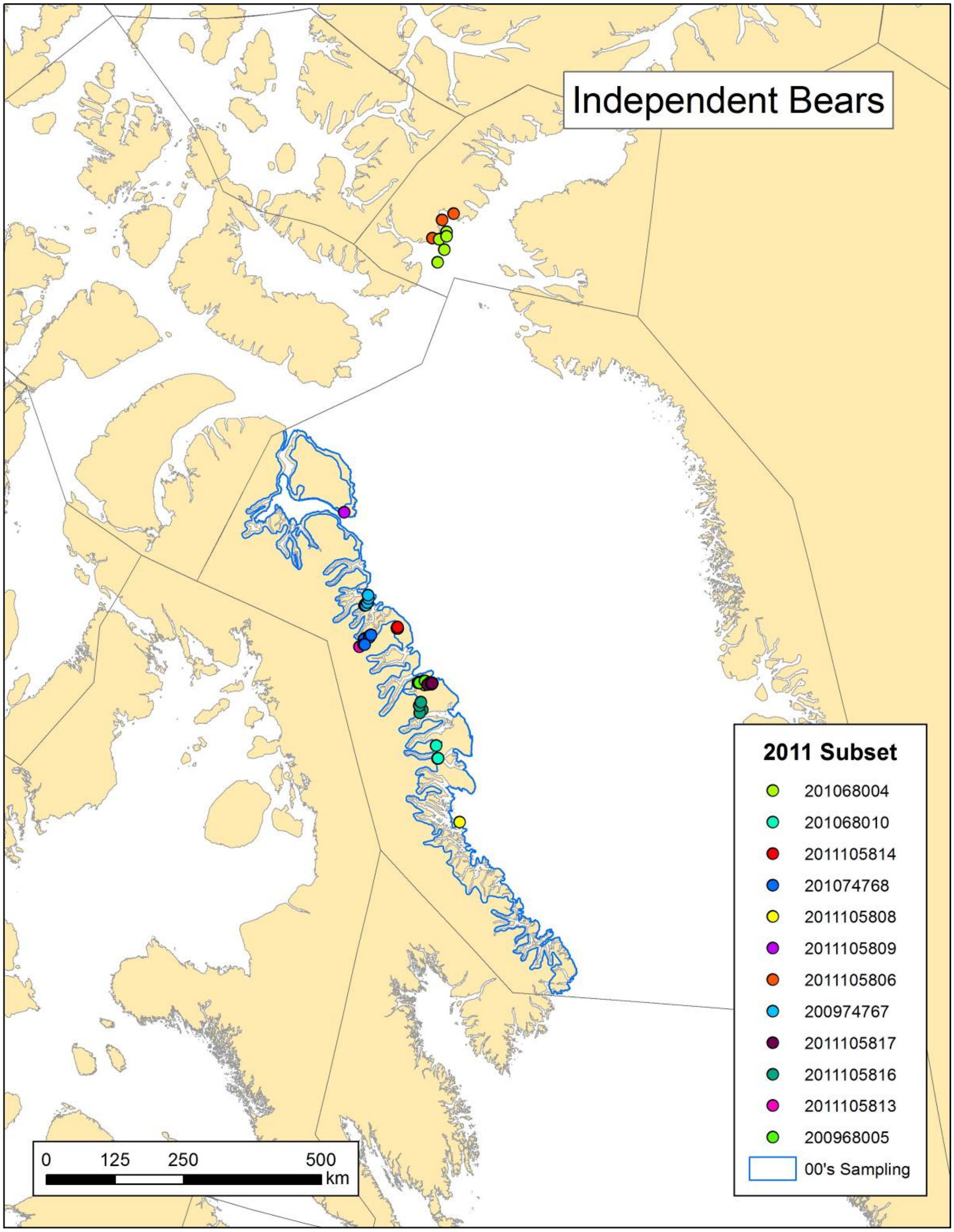


Figure 3.13. All telemetry locations from independent adult female bears with satellite collars transmitting during the 2012 sampling period dates (See Table 3.3). The 2010s sampled area for the MR study is shown in the blue outline.

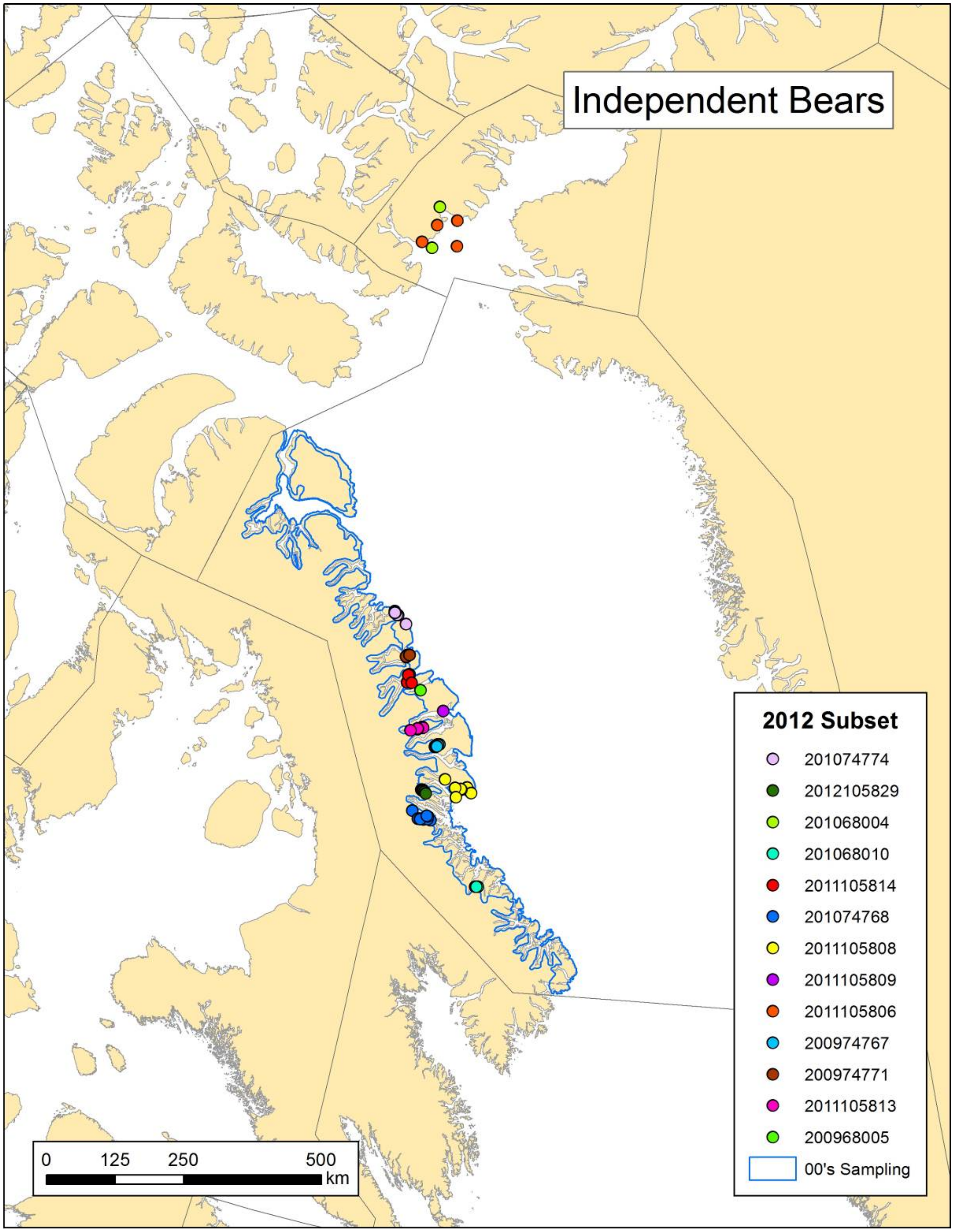


Figure 3.14. All telemetry locations from independent adult female bears with satellite collars transmitting during the 2013 sampling dates (See Table 3.3). The 2010s sampled area for the MR study is shown in the blue outline.

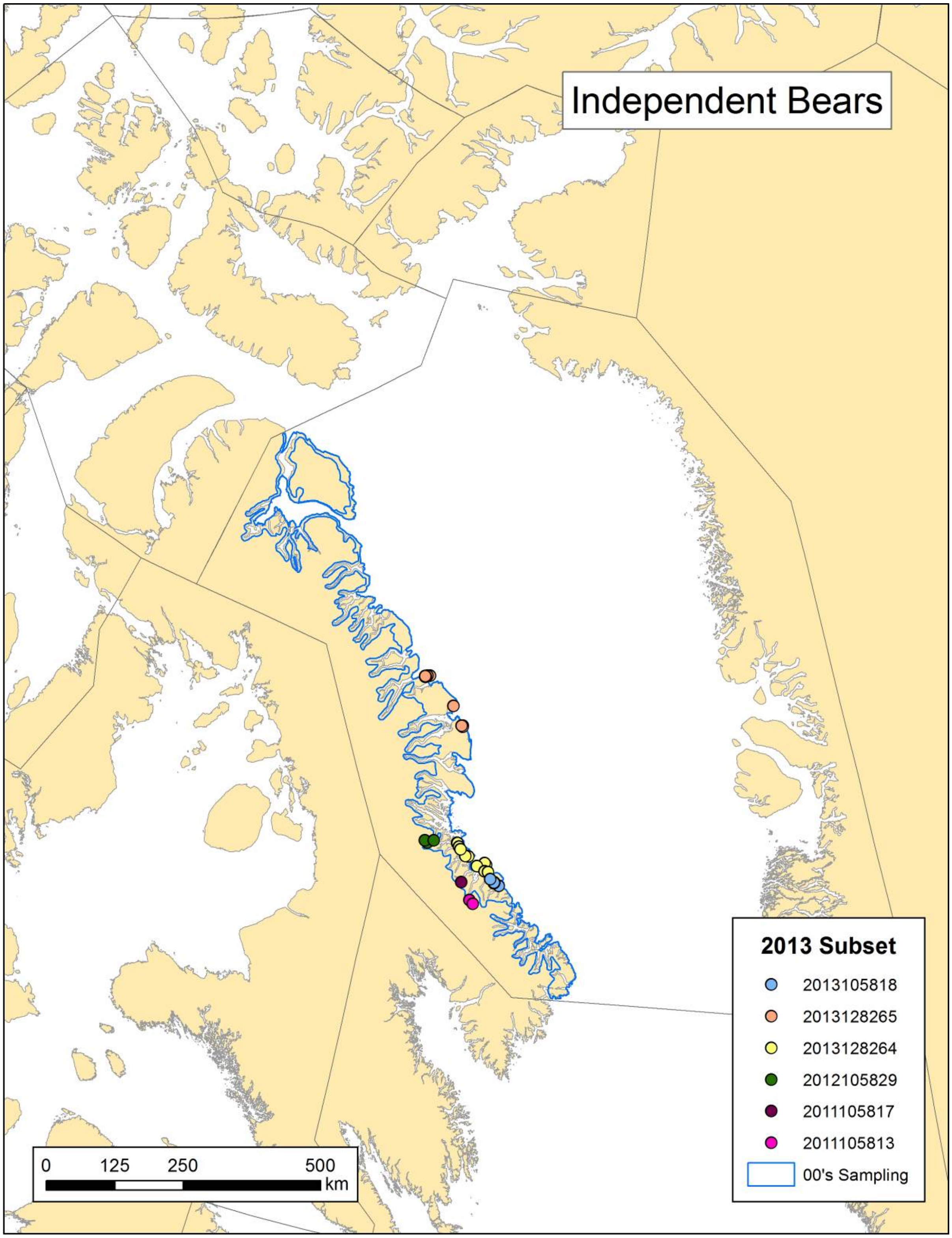


Figure 3.15. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 1993 (August week 4). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 1993 sampling period are shown.

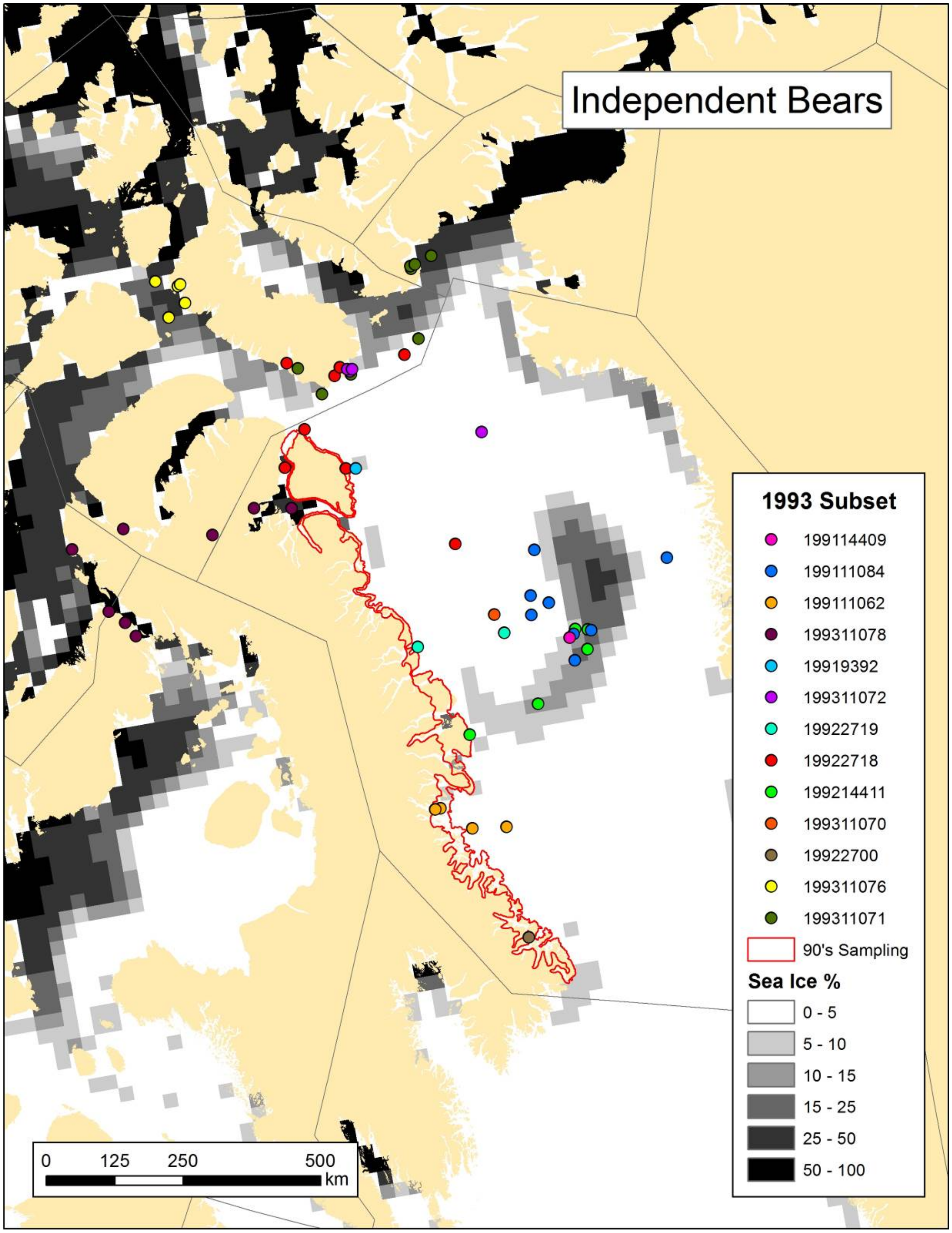


Figure 3.16. Distribution of weekly mean sea ice concentrations (SSMI) during the mid-point of the sampling period in 1994 (October week 1). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 1994 sampling period are shown.

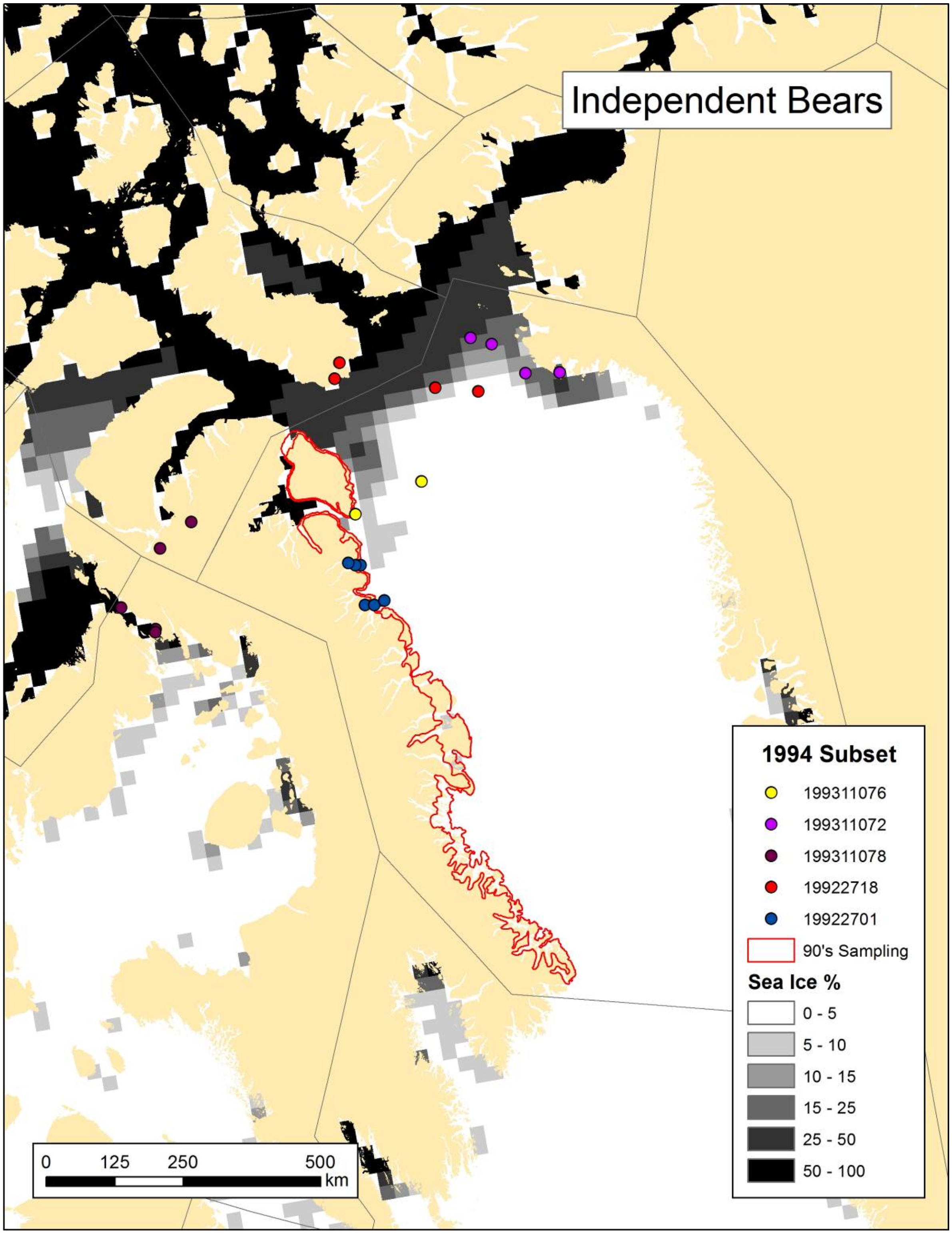


Figure 3.17. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 1995 (October week 2). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 1995 sampling period are shown.

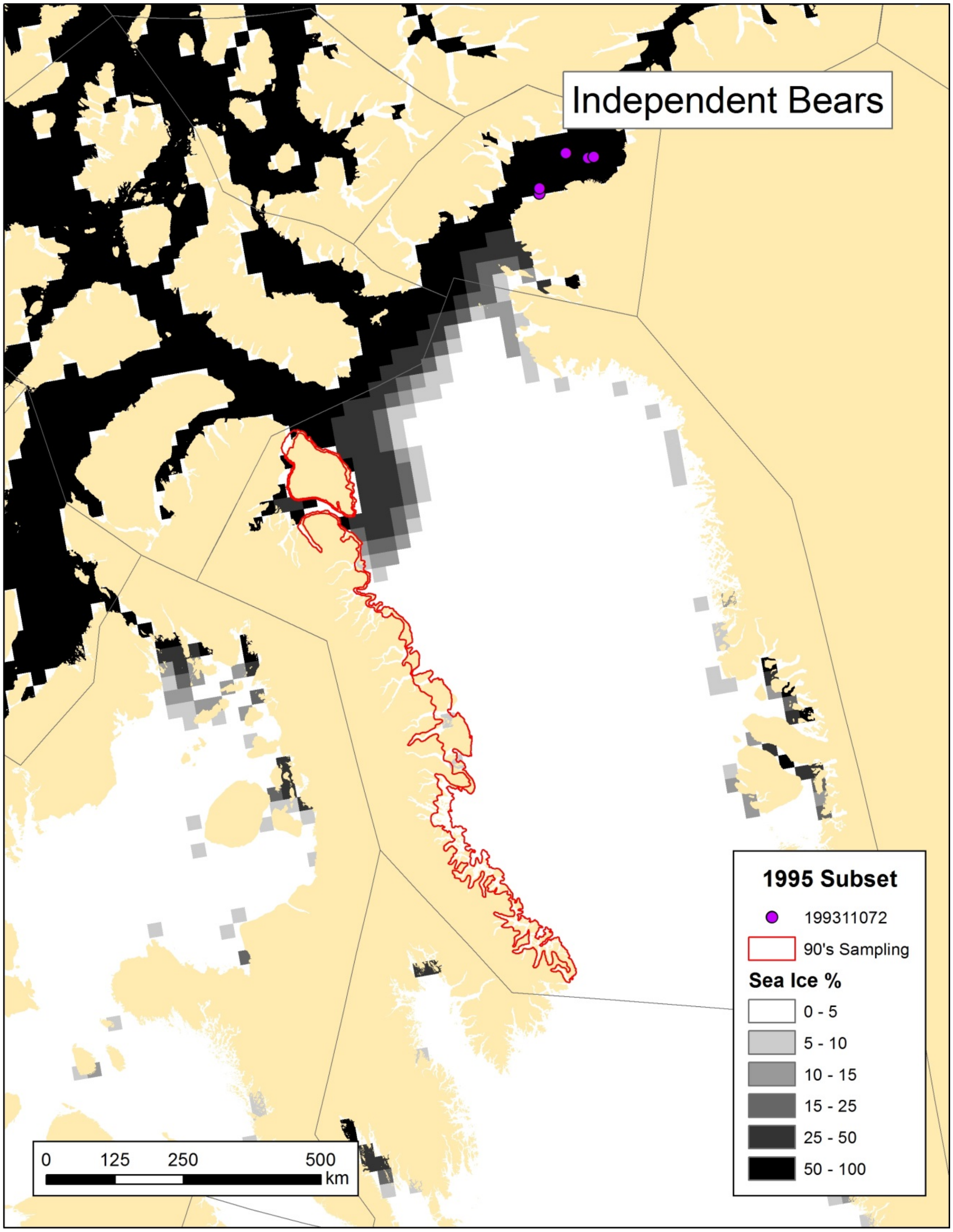


Figure 3.18. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 2011 (September week 3). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 2011 sampling period are shown.

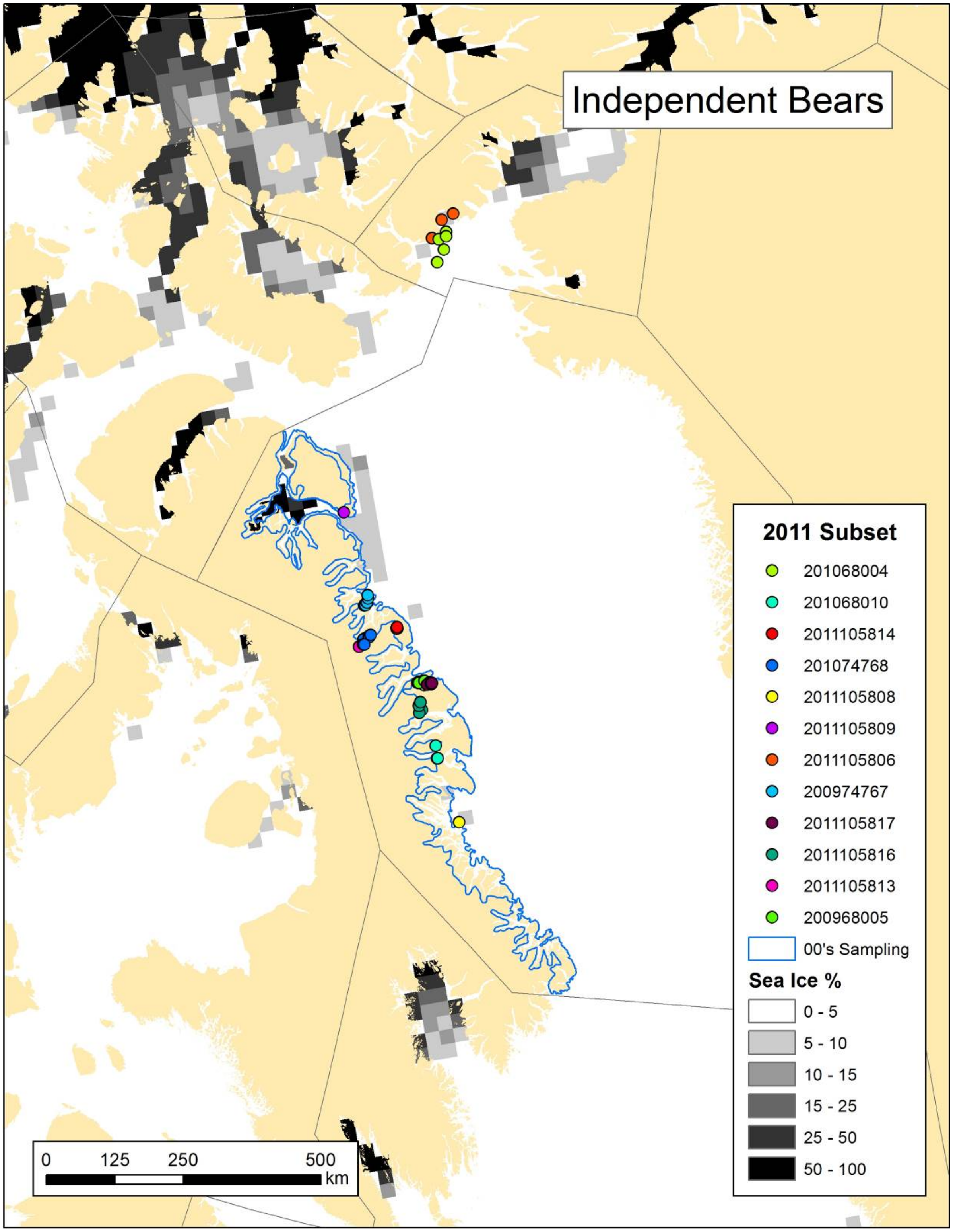


Figure 3.19. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 2012 (September week 2). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 2012 sampling period are shown.

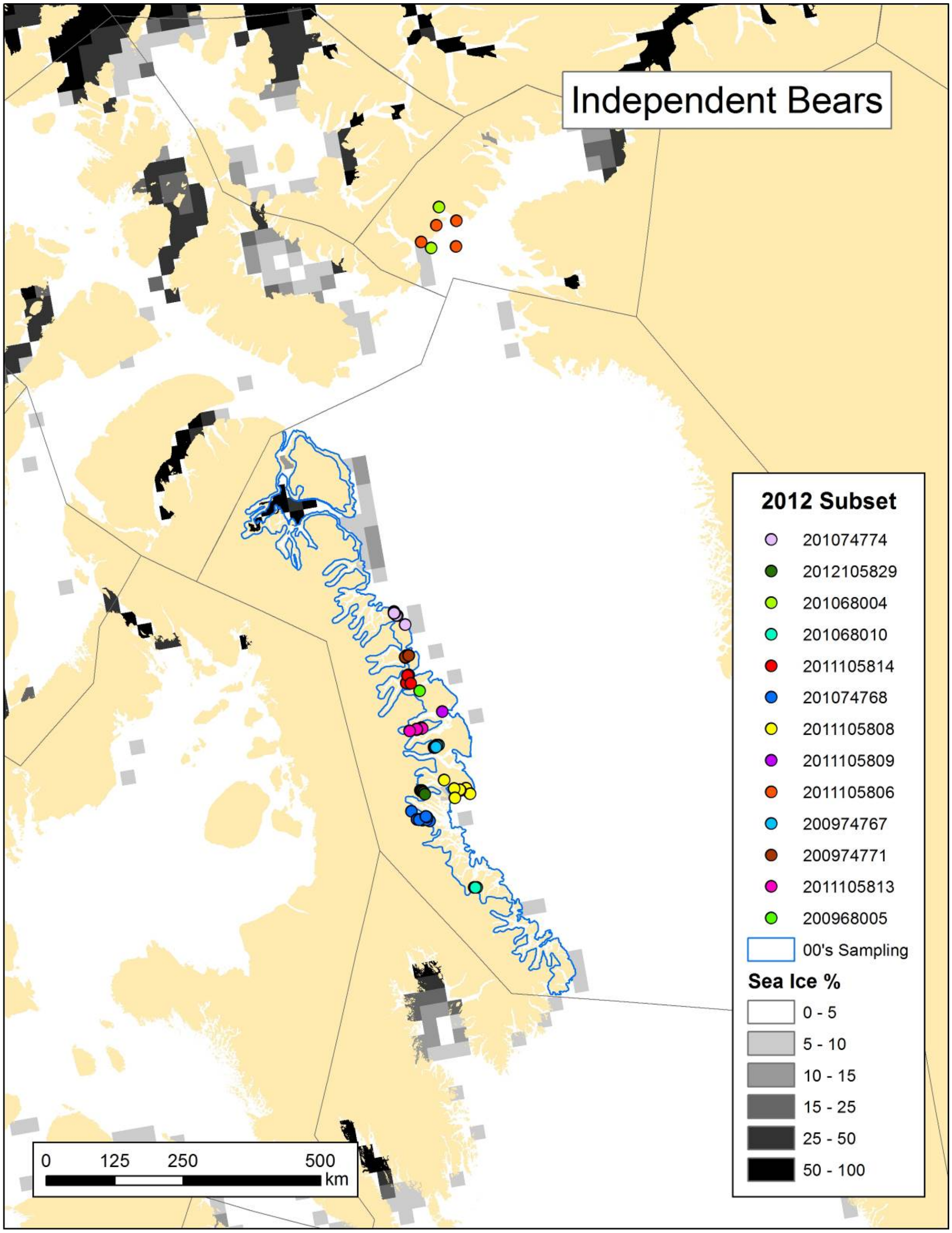


Figure 3.20. Distribution of weekly mean sea-ice concentrations (SSMI) during the mid-point of the sampling period in 2013 (September week 3). Sea ice is shown in 25 km^2 pixels. Locations of independent AF bears during the 2013 sampling period are shown.

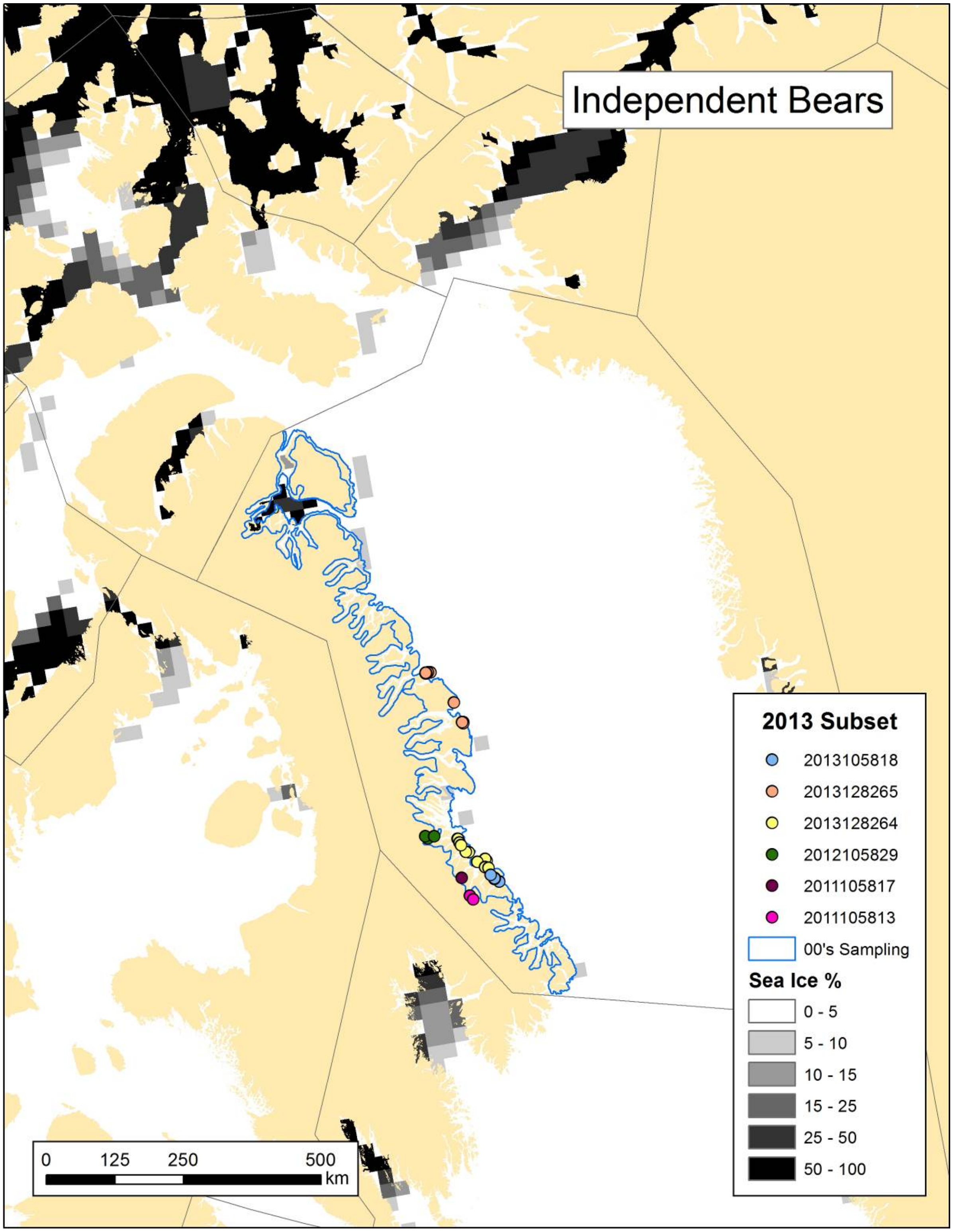
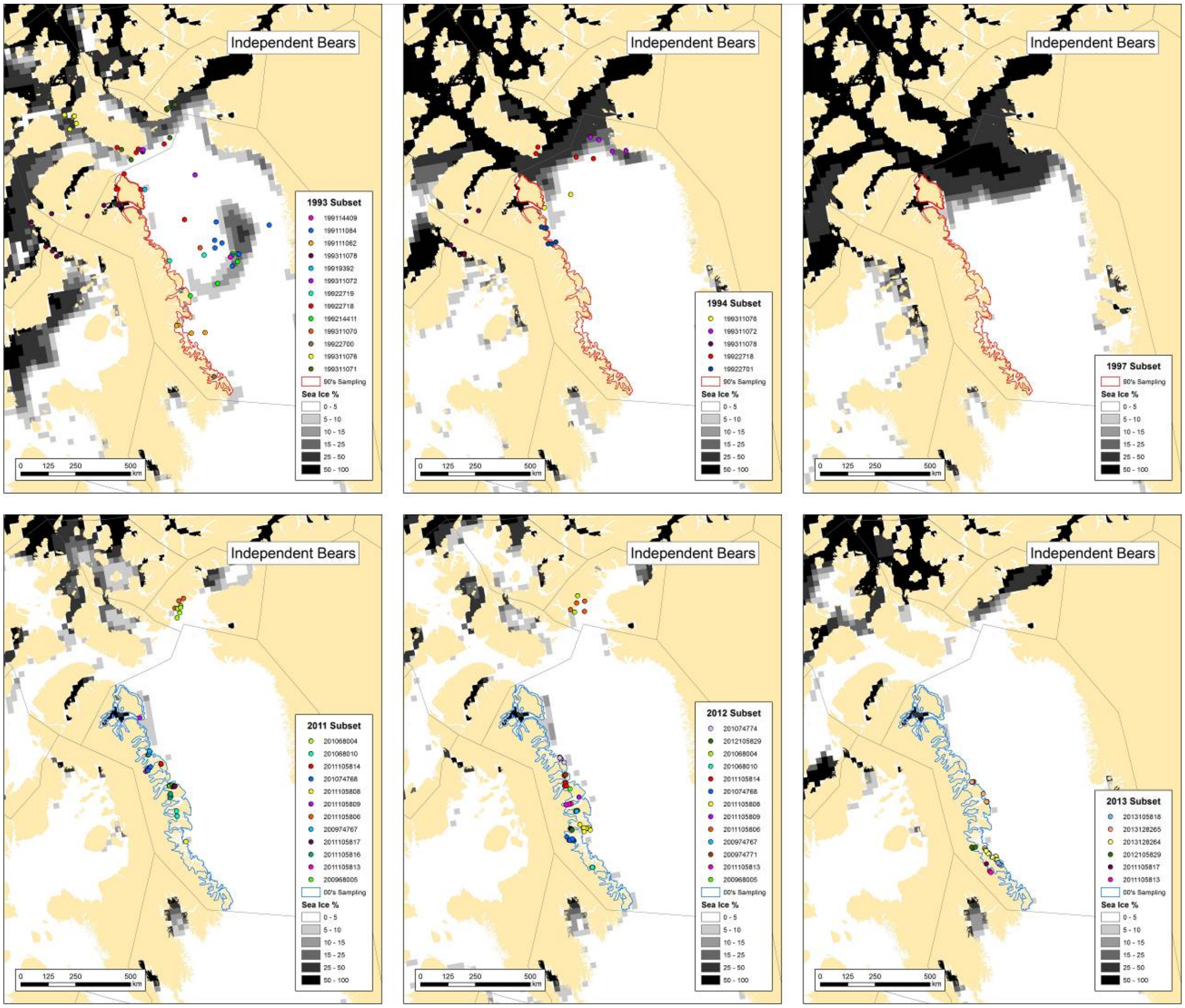


Figure 3.21. Distribution of sea-ice conditions (SSMI) during 1990s MR (top left to right 1993, 1994 and 1997) and 2010s MR (bottom left to right 2011, 2012, and 2013). Independent bears transmitting during the sampling are shown for reference. Note sampling occurred in 1997 but there were no independent collared bears for assessment of presence in the sampling area.



CHAPTER 4

HABITAT AND HABITAT USE IN BAFFIN BAY

KEY FINDINGS

- Sea-ice habitat in BB has significantly declined over the period of satellite observations, especially since the mid-1990s. The length of summer (number of days from sea-ice retreat in spring to sea-ice advance in fall) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 4 percent/decade. The general pattern of melt has not changed but occurs about 3-4 weeks earlier than in 1990s.

- Four-day movement rates of adult female polar bears have significantly declined during summer (August-October) in the 2000s due to disappearance of offshore and archipelago summertime sea ice. Bears are significantly less likely to move from land onto sea ice in summer in the 2000s.

- Bears use significantly lower sea-ice concentrations in winter and spring in the 2000s than the 1990s. Bears had stronger preferences to be closer to the 300 m depth contour (on shelf waters and near land) in the 2000s. Sea-ice concentration alone did not determine preferred habitat, adult females selected for lower sea-ice concentrations if it allowed them access to continental shelf waters (<300 m).

- Adult female bears were significantly closer to land in all months in the 2000s except at the end of breakup (June-July), when they remained on offshore sea ice as long as possible to maximize feeding.

- Potential long-distance swimming events were identified, defined as events in the summer during which bears traveled >100 km from offshore sea ice, to Baffin Island, through areas with <10% sea-ice concentration, and with a concurrent period of reduced or absent collar transmissions (i.e., because collars generally do not transmit when bears are in water). These events were observed in both decades, but the frequency increased in the 2000s, particularly in 2011.

- Bears spend significantly more time on land on Baffin Island; arrival dates on Baffin Island in summer were one month earlier in 2000s. The amount of time bears spend on land has increased by 20-30 days since the 1990s. Bears in the 2000s no longer arrive on Devon or Ellesmere Islands but only on Baffin Island (some also remain in Melville Bay Greenland).

- Entry dates into maternity dens were >1 month later in the 2000s. Exit dates from maternity dens did not change. Overall there was a significantly shorter maternity den duration in the 2000s.

- The first date of arrival on land by pregnant females was significantly earlier in the 2000s than the 1990s and bears spend more time onshore before entering maternity dens.
- Maternity dens in the 2000s occured at higher elevations and steeper slopes than maternity dens in the 1990s, likely due to reduced snow cover.

4.1. Introduction

Information on habitat can be used to facilitate subpopulation status assessment and harvest recommendations in the context of changes that are occurring in both available habitat and habitat use by polar bears. Habitat analyses can be used to evaluate polar bear distribution with respect to environmental variables, particularly ice conditions, topography and food availability or distribution (e.g., Ferguson et al. 1999. Durner et al. 2009, Wilson et al. 2014) and to inform subpopulation status relative to changes in environmental carrying capacity (Regehr et al. 2015). Using habitat information to identify key areas of use, or areas that will be critical in the future, is also important for conservation. Finally habitat studies can also be used to provide important context for interpreting both point estimates and trends in vital rates or MR results, which often have considerable bias and uncertainty. Large changes in sea-ice habitat for polar bears have occurred across all 19 subpopulations (Stern and Laidre, in review), particularly BB which occurs in the seasonal sea-ice ecoregion (Amstrup et al. 2008).

We assessed changes in sea-ice habitat for the BB subpopulation and used satellite telemetry data collected over two decades to assess changes in movement rates of bears, sea-ice habitat use, terrestrial habitat use, arrival and departure dates on/off land, and maternity denning. The results of this work provide important context for the MR results and add perspective on how environmental changes may explain observed results. They also provide scientific information which can potentially be used for comparison to observations from LEK studies, including observations of increased densities of polar bears on land or closer to shore (and communities).

4.2. Methods

Sea ice Analysis

Methods are described in detail in Stern and Laidre (in review), however, are briefly reviewed here. We used daily satellite data of sea ice for the period 1979-2014 from the National Snow and Ice Data Center in Boulder, CO (Cavalieri et al. 1996, updated yearly) to examine seasonal patterns of sea ice and how they are changing over time. These products are designed to provide a consistent time series of sea-ice concentration (the fraction, or percentage, of ocean area covered by sea ice) spanning the coverage of several passive microwave instruments. The gridded data have a cell size of 25 × 25 km. We used ETOPO1 for bathymetry, a 1 arc-minute global relief model of Earth's surface that integrates land topography and ocean bathymetry, built from numerous global and regional data sets (Amante and Eakins 2009). We averaged the ETOPO1 data over each 25-km grid cell to get the mean ocean depth for the cell, which we used to distinguish the continental shelf (less than 300 meters depth) from the deeper ocean. Baffin Bay consists of 1042 grid cells (656 × 103 km2); 28% have mean depth < 300 m, 72% have mean depth > 300 m. The shallow regions are located along the east coast of Baffin Island and the west coast of Greenland.

From the sea-ice concentration data, we calculated the daily area of sea ice for three sets of grid cells in Baffin Bay: (1) all ocean depths, (2) shallow depths only (< 300 m), and (3) shallow depths (< 300 m) along the coast of Baffin Island only. Figure 4.1 shows the seasonal cycle of sea ice area in Baffin Bay (all depths). We calculated the March average sea-ice area and the September average sea-ice area over all years (1979-2014), and defined a threshold (T) to be halfway between the two averages. We then found the date each spring when the sea-ice area dropped below T on its way to the summer minimum, and the date each fall when the sea-ice area rose above T on its way to the winter maximum. We call these the dates of sea-ice retreat and advance, respectively. These dates were calculated for each year (1979-2014) for the three regions (all depths, shallow depths only, and shallow depths along Baffin Island only).

In addition to the dates of sea-ice retreat and advance, we calculated the number of days from retreat to advance (also called length of summer), and the mean sea-ice concentration during June through October. Finally, we calculated the number of ice-covered days per year at every grid cell in Baffin Bay. This is the number of days per year that the sea-ice concentration exceeds 15%.

Sea-ice Habitat Use and Resource Selection Models

Polar bears were tagged in Northwest Greenland on the fast and pack ice between mid-March and mid-April 2009-2013 in Baffin Bay as described in Chapter 2. A total of 91 bears were tagged with satellite transmitters in BB. Of these, 38 were AFs collared in BB. These data were combined with a historical data set of AFs captured and tracked in the 1990s. In BB, 1991-1995 43 collars were deployed on AFs, with the majority deployed during the ice free season in fall on Baffin Island (n=11 deployed in spring in NWG, of these n=9 transmitted long enough to be included in the analyses). Only bears captured within the BB subpopulation boundaries were included in the comparative analysis, as defined by PBSG (2010).

Data filtering and sub-sampling – Methods on data filtering and sub-sampling are described in Chapter 2. Data were divided into seasons included Spring (March – July, which included the peak of sea-ice coverage and initiation of sea-ice break-up), Summer (August –

October, which included the end of break-up and the on-land period) and Winter (November – February, which included the freeze-up period and time when bears went back out on the sea ice). Adult females were defined as ≥ 5 years old and adult males as ≥ 6 years old. Age group status as determined in the field was verified based on tooth analyses (Table 4.1).

Movement rates – We examined daily (4- or 5-day) movement rates for AFs in each subpopulation by decade and month. For the RSF, we used the mean monthly movement rate + 2SD for the radius of potential habitat selection at each time step, following the approach used by Durner et al. (2009) and Laidre et al. (2015). We used monthly values that were specific to each decade. We used a spatial distance limit of 400 km (or roughly 12 days) as the maximum step length possible in the data. Any gaps longer than that were skipped.

Habitat covariates for RSF – In the comparative analysis between polar bears tracked in the 1990s and 2000s daily sea-ice concentration values were used from satellite passive microwave data (SSM/I) from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data (Cavalieri et al. 1996) available from the National Snow and Ice Data Center (NSIDC) in Boulder, Colorado, USA. Sea-ice concentrations were provided in a polar stereographic projection with a nominal grid cell size of 25 × 25 km. Temporal coverage was every other day from 26 October 1978 through 9 July 1987, and daily through 01 April 2015. Sea-ice habitat was defined around each polar bear location at two spatial scales: the sea-ice concentration pixel value where the bear was located and the mean sea-ice concentration within a region consisting of the 3 × 3 block of pixels centered at the bear location (nominal area 5,625 km^2) with the corners removed in order to approximate a circle. All denning periods were identified (maternity and shelter dens) (Escajeda 2016) and removed from resource selection (RSF) models. RSF models were only conducted on AFs to enable comparison with the 1990s. Bears with a low n (less than 3 locations) were removed from the analysis.

We also calculated the distance from each polar bear location to the sea-ice edge (defined with two concentration thresholds) and the distance from each polar bear location to the mainland Baffin Island coastline. The sea-ice edge covariate estimated the distance (in km) from the bear's location to the center of the nearest pixel with either 15% or 50% sea-ice concentration. We used the 15% sea-ice concentration as a delimiter between sea ice and open water. We used the 50% sea-ice concentration as a delimiter between suitable polar bear habitat and breakup conditions (see Stirling and Parkinson 2006). Distances were determined by great circle calculations based on latitude and longitude and therefore were not subject to pixel size.

We included variables about bathymetry in the models. We used IBCAO 3.0 (500 m^2) to estimate the depth value (m) at the bear location and to determine if the bear was in categorical depth categories (shelf: 0 - <300 m, intermediate: 300 - 1000 m, and basin: >1000). We also calculated the distance of the bear to the shelf break (where the shelf break was considered to be > 300 m). In 931 cases (out of 500,000) at the southern extent of the range offshore in Davis Strait, there were no values in the IBCAO grid south of 60^{o}N. For these values we used the ETOPO-5 grid (1 km^2) to retrieve depth information. Finally we included a variable that quantified if bears moved from sea-ice in winter or spring to land.

Buffers were created around each polar bear location that were representative of available habitat bears could select on a 4-day (or occasionally 5 or 6-day) interval depending on satellite collar duty cycle (cycling of transmissions for battery longevity). The radius of the buffer was based on mean monthly movement rates for bears grouped into decades +2 SD (1990s and 2000s). Fifty random locations in each buffer were sampled for each time step and represented candidate locations not selected by the bear at each given movement step (i.e., pseudo-absence locations). This control data set was considered to represent local habitat availability. All pseudo-absence locations were linked to the same habitat variables listed above using ArcGIS (ArcGIS 10.1, ESRI, Redlands, CA, USA). A maximum time gap of 12 days or 400 km distance was selected between locations to minimize the size of the buffer.

RSF sea-ice models – Univariate habitat utilization was quantified and contrasted for each habitat covariate (e.g, only pixels where the bear was present) in the 1990s and 2000s. Multivariate conditional logistic RSF models were built for each season and decade. We selected variables for multivariate analyses a priori based on biological hypotheses. We did not include variables in the same model that were biologically redundant (continuous depth vs. distance to 300 m shelf) or highly correlated (e.g., distance to 15% and 50% sea-ice concentration). Models were fit to each decade and one combined model using both decades was fit to facilitate testing for differences in effects across decades through the use of a covariate by decade interaction. We used conditional logistic regression with matched location/pseudo-absence sets (CLOGIT function from SURVIVAL package) (R Development Core Team 2013; Therneau 2015) to model the strength of preference for habitat parameters in the 1990s and 2000s.

Terrestrial Habitat Use and Selection

Terrestrial RSF – For collared polar bears on land in summer, RSF models included land covariates derived from two Digital Elevation Models (DEMs): the ASTER GDEM for all positions in Canada (http://www.jspacesystems.or.jp/ersdac/GDEM/E/4.html, 22 m2) and the GIMP for all positions in West Greenland (http://bpcrc.osu.edu/gdg/data/gimpdem). The

ASTER DEM, which was used for the majority of telemetry locations, consisted of a mosaic of tiles from the Advanced Spaceborne Thermal Emission and Reflection Radiometer Global Digital Elevation Model (ASTER GDEM; Version 2), a product of Japan's Ministry of Economy, Trade, and Industry (METI) and the United States National Aeronautics and Space Administration (NASA). The ASTER GDEM had an overall horizontal resolution of ~17 m at the 95% confidence interval with a vertical resolution of 75 m. The DEM was produced at a 22.625 x 22.625 m resolution with elevations positioned in the WGS 1984 datum and projected in a North Pole Stereographic projection with a central meridian of -55°W.

At all bear locations we extracted the value of elevation (m), slope (calculated as % rise) and aspect (calculated as 0-360 degrees) at the bear location using the DEM at the highest resolution. We included a variable that quantified whether bears moved from land habitats to sea ice in summer. We followed a similar process for the sea ice RSF models above but only examined land covariates in the summer season (August-October), including interactions for the 1990s and 2000s.

Arrival and departure dates on land – We used location data from satellite collars to compare the timing of land use patterns by AF polar bears in the Baffin Bay between two time periods (1991-1997 and 2009-2015) when substantial summer sea-ice loss occurred. We excluded bears that remained in Melville Bay for the entire tracking period (1-2 years) as these bears, though close to shore, were using glacial ice throughout the summer and we could not determine if they were on land or in a glacial mélange. We quantified the date individual BB bears arrived on land in fall, the duration of time spent on land, the date bears were back on the sea ice in spring.

All radio-collar locations were determined by the Argos System with accuracy from < 250 m to > 1500 m (see http://www.argos-system.org/web/en/78-faq.php#faq-theme-55). Location data were filtered to remove implausible locations as described previously. Instances of dropped collars were identified based on activity and temperature sensors and data collected post-drop were removed.

We considered a bear to be on land if its Argos location was within 5 km of the high resolution coastline as identified by 0-pleth line of the IBCAO digital elevation model (Jakobsson et al. 2012). The 5 km buffer was used to encompass small barrier islands that may be used by polar bears in the summer but are not depicted as land in the GIS file and to account for low accuracy of some locations. Our 5 km buffer might have resulted in some offshore bears being classified as on land, but this was less likely to occur during the focal time periods of our analysis (fall and spring) because landfast ice was either disappearing or forming in the seasonal ice zone. We were most interested in the date bears arrived on land during the fall sea-ice recession and the date bears returned to the ice during spring ice formation. Thus any short-term visits to land were not included in the analysis and we focused on large seasonal patterns. Due to the nature of the sea ice cycle in BB, bears within 5 km of the coast during fall were likely to either have been in open water or on land.

Bears were required to enter the 5 km buffer and stay within 5 km or less of the coastline for at last 14 days before they were considered to be ‘on land’. The same criteria were used for bears departing from land in spring (>= 14 days on the ice). For all AF bears identified to be in maternity dens (Escajeda 2016), we excluded dates of return to the sea ice in spring, as the maternity denning period dictated the date of return, not the formation of sea ice.

For pairs of positions that were separated by 4 or 8 days, we linearly interpolated the date on or offshore. We excluded data when observed locations were separated by >8 days, except in the case of when bears were offshore in summer on <15% sea ice (>100-200 km from the nearest coastline) and next subsequent position was on land. In these cases, there were data gaps (12-30 days) in locations due to potential long distance swimming from central Baffin Bay to the shore; see next section.

Potential swimming events – We identified potential long distance swimming events from central BB during the break-up season. There are frequent drop-outs in transmission from collars (i.e., missing positions) such that the sampling interval was 8 days, or 12 days, or longer. The drop-outs occurred when the bear was in a region of extremely low sea-ice concentration, suggesting that the failure to transmit a position may be because the antennae is in the water, i.e., the bear was swimming (Pagano et al. 2012).

For every polar bear location, we extracted the sea-ice concentration at the SSM/I grid cell in which the polar bear was located providing a time series of sea-ice concentration following the bear’s trajectory. We also calculated the distance from every polar bear position to the nearest coastline and developed plot distance-to-land vs. time, using symbols representing the sea-ice concentration at the bear position. We examined the trajectory of each bear and identified the timing and occurrence of gaps in collar transmissions. We selected bears where the transmissions ceased during the break-up season when the bear was offshore and examined both the bear’s trajectory and the sea-ice conditions for potential swimming to land.

Maternity denning – Solitary females or females with two year-old cubs were considered candidates for denning the following winter after capture (Wiig 1998). All of the satellite collars in the 2000s provided temperature and motion data along with position coordinates.

Temperature was measured by a thermistor within the collar (Fischbach et al. 2007). Data collected on polar bear dens in the Baffin Bay and Kane Basin in the 1990s were originally published by Ferguson et al. (1997), and included 29 dens from Baffin Bay and 9 from Kane Basin. The 1990s data were obtained in the form of one position per day.

Although the reading from the collars is influenced by the animal's body temperature, the temperature reported by the collar is a general representation of the ambient temperature of the surrounding habitat (Harris et al. 1990). Temperature data were extracted from transmitters using the Telonics Data Converter software (Version 2.21; Telonics, Mesa, AZ). Position and temperature data for bears identified as denning candidates were examined from July to June of the following year. Only one best quality position and one temperature reading were used for each day. The designated position for each day was selected by choosing the first position with the best location quality score. The temperature reading for each day was selected by first removing any temperature points ≥ 40ºC or ≤ -40ºC which were considered outliers (Tchernova 2010), and calculating the average. Three variables signal that a female is in a den: high temperature readings compared to ambient air temperature (10 to 40^{o}C warmer), constant position on land, and decreased quality and frequency of transmissions (Amstrup and Garner 1994; Messier et al. 1994; Wiig 1998; Fischbach et al. 2007). Temperature data were available for all Baffin Bay bears.

Since transmissions were received on a 4-day duty cycle, the position data were particularly coarse as points found within the denning period did not closely center on a single den position, but rather consisted of a cluster of points within a small area. The data for all adult females were examined for sustained high temperatures (greater than 0°C), decreased transmission quality and frequency, and a stationary position on land during the denning period

(September through March). We created a subset of the position data for each denning candidate during this time period and mapped the points in ArcMap v.10.1 (Environmental Systems Research Institute, Inc. [ESRI], Redlands, CA 2012). Individual point shapefiles were created from the identified subset for the three location quality categories: LQ 1, LQ 2, and LQ 3. A buffer was then drawn around each point with radii equivalent to the maximum error estimate for each LQ score (points with LQ 1 had a buffer of 1500 m, LQ 2: 500 m, LQ 3: 250 m). The mean center of the intersection of these buffers then defined the den position. The method provided a probable location for the den site based on the error estimate of the satellite telemetry positions during the denning period and is independent of the number of positions as well as any spatial outliers. Note that not all of the dens were determined using this method, some bears had sparse location data within the denning period and thus the den positions had to be determined using variants of the buffer method.

Den Phenology Analyses – Length of denning was used to distinguish maternity dens from shelter dens. Shelter dens were typically occupied for a short period of time (> 14 days to < 4 months) whereas females will typically remain in maternity dens for > 5 months (Messier et al. 1994). Though shelter dens were included in the den habitat characteristics analyses, they were excluded from the phenology analyses. In addition to denning duration, den entry and exit dates were compared to the 1990s data (Ferguson et al. 2000). The exit date for each den was established as the median date between the female’s last transmission from the den and the first movement outside the den, indicated by a significant drop in temperature and movement away from the den site. Most of the entry dates for the 2000s dens were determined by creating a 1 km buffer around each den site and selecting the median date between the last date outside of the buffer and the first date inside the buffer. The entry date was verified by comparing the autumn temperature data for each denning bear with temperature readings from a non-denning bear that same year to check for a difference of more than 10°C. The dates when the temperature readings diverged by ≥ 10°C were then compared to the entry dates determined by the position data. Both the den entry and exit dates were measured as day-of-year (DOY; Day #1 is 1 January), which we then used to calculate the denning duration in number of days (Messier et al. 1994; Wiig 1998; Ferguson et al. 2000).

We used two-sample Mann-Whitney U tests to test for differences between the entry/exit dates between the 1990s and 2000s datasets and duration spent in the dens without assuming normality. Additionally we tested for a correlation between den entry date and latitude using a Kendall's tau test. An alpha value of 0.05 was used for all statistical analyses.

First Date on Land (FDOL) by pregnant females – The date of entry onto land was determined for each bear entering a maternity den. Cherry et al. (2013) defined the first date on land (FDOL) as the date that the bear first came onto land without returning to the sea ice until freeze-up the following fall; we modified this for pregnant females to be the first date on land after which the bear did not return to the sea ice until she emerged in the spring. First dates on land for sheltering bears were also calculated, but not analyzed.

Den Habitat Characteristics Analyses – All den positions were imported into ArcMap and overlaid with the ASTER DEM of the study area. The elevation, aspect, and slope of each den site were extracted from the DEM while straight-line distance to the nearest shoreline was measured using a vector shapefile of Canada's coastline (US Defense Mapping Agency). Elevation was calculated as the elevation of the cell containing the den site and was measured in meters. Aspect is the compass direction (in degrees) the cell faces, while slope measures the rate of maximum change of elevation in degrees.

A principal component analysis (PCA) using a correlation matrix was performed on the habitat data of the maternity dens in order to determine which variables drive any dissimilarity among den sites. In the data matrix for the PCA, each den site was input with its elevation, aspect, slope, and distance to coast. A second matrix organized the den sites into two groups: sample period (1990s or 2000s), and latitude zone (south of 70^{o} N, central 70^{o} N to 75^{o} N, and north > 75^{o} N). Prior to the analyses, the environmental variables in each dataset were log10-transformed to control for skewed data (Kenkel 2006). After computing the PCA, a Monte Carlo randomized approach was used to test the significance of the eigenvalues ($\alpha = 0.05$). All analyses were performed using the statistical software R version 3.0.2 along with the “vegan” package (Oksanen et al. 2013) and the “Biostats” R package (McGarigal 2015).

In order to test for differences in environmental descriptors between the two groups, a permutational multivariate analysis of variance (perMANOVA) was performed on the maternity den matrix. For the distance matrices, Euclidean distances were calculated for each log10-transformed and column-standardized matrix of raw data. A permutation test was then used to evaluate the significance of the resulting pseudo-F statistic as compared to a null hypothesis of no difference between groups. To determine the source(s) of dissimilarity detected by the perMANOVA, we applied a supplementary test of multivariate homogeneity of group dispersion (DISPER; Anderson 2006). DISPER involves computation of the distance of each group member to the group’s centroid and applies an ANOVA to the distances with a null hypothesis of no difference in variation among groups.

4.3. Results

<u>Sea-ice Habitat</u>

In Baffin Bay, sea ice is retreating earlier in spring by 7 days/decade and advancing later in fall by 5 days/decade (Figure 4.2). Trends in four of the sea-ice metrics (Table 4.2) show consistent loss of polar bear habitat. The length of summer (number of days from retreat to advance) is increasing by 12 days/decade, and the mean sea-ice concentration during June-October is decreasing by 4 percent/decade (Figure 4.3). All these trends are statistically significant (Table 4.2).

Trends in the dates of spring sea-ice retreat and fall sea-ice advance are stronger for the shallow depths (< 300 m) than for all depths collectively. When only the shallow depths on the Baffin Island shelf are considered, the trends are weaker than for all depths collectively, which suggests that the trends are stronger for the West Greenland shelf than for the entire Baffin Bay. This can be seen in the trend in the number of ice-covered days (Figure 4.4), which shows a greater loss of ice-covered days along the western coast of Greenland than in central Baffin Bay.

The pattern of spring sea-ice retreat in Baffin Bay begins with melting along the southwest coast of Greenland and progresses northward. At the same time, the North Water Polynya (located at the north end of the Baffin Bay region, in Smith Sound, and the south end of the Kane Basin region) begins to melt out. At some point during summer, these two open water areas connect as Melville Bay melts out, severing the continuous ice connection between Baffin Island and Greenland. The sea ice then continues to melt back toward the coast of Baffin Island. Occasionally a “sea-ice island” becomes the last remnant of ice in Baffin Bay, if the ice along the coast of Baffin Island melts out first. This general pattern of melt has not changed over time, but it is occurring earlier in the summer now (in the last decade) than in the 1980s and 1990s (Figure 4.5). The early part of the melt pattern, up the coast of Greenland and across Melville Bay, is trending even earlier than the melt on the western side of Baffin Bay. In October and November, sea ice advances southward through Baffin Bay, generally with the leading edge along the coast of Baffin Island. This pattern is occurring later in the fall now (in the last decade) than in the 1980s and 1990s (Figure 4.6).

In summary, the coast of Baffin Island is generally the last place in Baffin Bay to lose sea ice in summer, and the first place in Baffin Bay to regain sea ice in fall. The spring retreat of sea ice there is trending earlier by about 7 days/decade, or 24 days over the 3 ½ decades of this study. The fall advance of sea ice there is trending later by about 4 days/decade, or 14 days over the period of this study. During the months of June through October, the mean sea-ice concentration is trending downward by about 4% per decade, or about 14% over the period of this study. There is year-to-year variability in all the sea-ice metrics, but the trends are all statistically significant.

Movement rates – In Baffin Bay, mean monthly movement rates for adult females in the 1990s ranged from 5.5 km/day (in October) to 15.8 km/day (in December) (Figure 4.7). Rates for adult females in the 2000s ranged from 1.9 km/day (in September) to 13.8 km/day (in December) (Figure 4.8, Table 4.3). In general, rates were lowest in both decades during September and October, and highest in December. Adult females had significantly higher movement rates in the 1990s during May, August and September. In the summer months (Aug-Sept), movement rates in the 2000s were one-half to one-third of the rates in the 1990s ($p<0.001$) (Table 4.3). Overall there was also greater variability in movement rates in the 2000s than the 1990s, perhaps representing a broader range of strategies in changing habitats. The greatest variability was found in the early winter months, November and December, especially in the 2000s. This may represent changes in timing of freeze up and when bears access the sea ice.

RSF sea-ice models – We first examined univariate relationships for each covariate as a continuous function over the entire annual cycle (Figure 4.9 and 4.10). Adult female polar bears use significantly lower sea-ice concentrations in the 2000s than the 1990s in all seasons except May/June (Figure 4.9). Bears in the 2000s were also significant closer to land in all months except June and July. Overall there was a significantly smaller fraction of observations on the sea ice in the 2000s than the 1990s over the period that each bear was tracked (Figure 4.10).

In winter in the 1990s, the full multi-variate RSF model demonstrated a positive association between adult female polar bears and sea-ice concentration, meaning bears selected areas with higher sea-ice concentrations when available. The association in the 1990s with depth was negative, demonstrating preference for shallow shelf waters and avoidance of deep areas (Table 4.4). There was a strong negative association for movement on to land in winter. In the 2000s, preference for higher sea-ice concentrations was not as important as distance to shallow shelf waters. The distance to the 50% sea-ice concentration threshold was strongly negative, and adult females showed a strong and significant preference for regions <300 m in depth, similar to the 1990s. In both decades bears did not prefer to move from sea ice to land. Interactions between decades (changes in preference over time) showed that in winter adult female polar bears in the 2000s used lower sea-ice concentrations than bears did in the 1990s. In the 2000s there was also an increase in preference for being close to the 300 m depth contour (on shelf waters), which also serves as a proxy for distance to land.

In spring in the 1990s, the full multivariate model showed that adult females had a strong significant preference for higher sea-ice concentrations (Table 4.5), more so than in the winter months. There was also a negative association with increasing distance from 50% sea-ice concentration. This was similar to bears in the 2000s, where sea-ice concentration and distance to 50% sea-ice concentration were important model variables. When the two decades were compared using interactions, the preference for high ice concentrations was significantly stronger in the 2000s than the 1990s. Furthermore, being farther from 50% ice concentration was less preferred in the 2000s than the 1990s (it was not preferred in either decade but more so in the 2000s). There was no change in the association with land, in both decades bears strongly avoided going to land in spring.

Terrestrial Habitat Use and Selection

Terrestrial RSF – In summer both in the 1990s and 2000s, resource selection models suggested that adult female polar bears significantly preferred areas of lower elevatio and steeper slope (Figure 4.11, Table 4.6). There was no preference for aspect. Also in both decades adult females showed a negative preference for being further inland from the outer (smoothed coast) of Baffin Island. In the 1990s there was a significant preference to stay on land (bears tended not move on to ice or into water). In the 2000s, all of the preferences were similar to the 1990s. There were no changes in preference between decades for on-land habitat types (elevation, slope, aspect or distance inland from the outer coast) (Figure 4.12). The primary difference between decades was that bears in the 2000s had a stronger preference to stay on land, meaning they were less likely to move to sea ice in summer months (once on land) than in the 1990s.

Timing on land and duration on land – We obtained 78 arrival dates for AF bears between 1991 and 2013. In some cases one bear contributed more than one arrival date because the bear was tracked over multiple years. Of these, bears arrived on land on Baffin Island (n=71 dates), Devon Island (n=5 dates) or Ellesmere Island (n=2 dates, in Kane Basin). We also obtained 71 departure dates in fall, of these 66 were from Baffin Island, Devon Island (n=3 dates) and Ellesmere Island (n=2 dates). We excluded the dates from two individual bears on Ellesmere Island because the sea-ice formation and break-up in the Arctic Archipelago system is different than the seasonal ecoregion in BB, which is the region of focus.

The mean date of arrival on land in the 1990s was August 24 (SD 16 days, n=30), which was significantly later than the mean date of arrival on land in the 2000s (August 4, SD 11 days, n=46) ($p<0.001$). The mean date of departure from land in the 1990s was November 1 (SD 21 days, n=42) which was not statistically different than the mean date of departure from land in the 2000s (November 8, SD 9 days, n=27) ($p=0.06$) (Table 4.7).

The overall mean arrival latitude in the 1990s was 70.7^{o} N (SD 3.0, n=30), which was not statistically different than the overall mean latitude of arrival in the 2000s of 70.3^{o} N (SD 1.3, n=46). The overall mean departure latitude in the 1990s was 69.7^{o} N (SD 3.0, n=42) and was not statistically different than the overall mean latitude in the 2000s of 70.4^{o} N (SD 1.4, n=27). When latitude of arrival and departure was examined in 2 degree blocks, there was a clear difference between decades. In general the distribution of arrival and departure latitudes was more restricted in the 2000s than the 1990s (Figure 4.13). For example, bears in the 2000s had significantly earlier arrivals that were contracted into a smaller band of latitudes. In the 1990s, bears departed from a larger range of latitudes (75 to 67^{o} N), where as in the 2000s departures were condensed into a latitude band ranging from 73-68^{o} N. The departures showed the general pattern of sea-ice formation from north to sound (so more northerly latitudes were available earlier) (Figure 4.14).

Estimation of time on land requires both an arrival and a departure of the same bear in the same year (paired dates). In this case we had n=14 pairs of dates in the 1990s and n=26 pairs of dates in the 2000s. We used this to estimate the number of weeks spent on land and changes between decades. We did not include the two individuals on Ellesmere Island.

During the 1990s, the mean duration of time on land in summer was 62.3 days (SD 25, range 8-99 days). In the 2000s, the mean duration of time on land in summer was significantly longer (<0.001) by 33 days, with an average time on land of 94.8 days (SD 15.7, range 56-120). These values include bears that used any coastline within the seasonal sea-ice ecoregion (e.g., including Devon Island, where bears arrived in the 1990s). If only bears using Baffin Island were compared then the mean duration of time on land in summer in the 1990s increased to 74.5 days (SD 14, range 54-99 days, n=10 bears). This value was still significantly different when compared to the 2000s (<0.001), with bears in the 2000s spending on average 20 days longer on land (Table 4.8).

The dates of adult females arrival and departure from land, described above, had a remarkably close correlation with the sea-ice metrics in BB (Figure 4.2 and 4.15). In both decades the dates of fall sea-ice advance were correlated and highly similar to the dates of departure from land. In both decades the dates of spring sea-ice retreat were correlated to the dates of arrival on land, however there was a lag between the retreat and arrival dates while bears used sea ice in BB as long as possible before being forced to land (Figure 4.15).

Swimming to land – There were potential long-distance swimming events observed both decades however they were observed in larger numbers in the 2000s. In the 1990s there were at least two potential long-distance swimming events. In the 2000s, there were 15 potential events in July 2011, one in July 2012, and one in July 2013. Examples of data time series from these events are shown in Figures 4.16 and 4.17. The events in July 2011 were especially prominent because in that year sea-ice breakup occurred early when a large number of bears (n=15) were transmitting from offshore BB. In this case, at least 6 bears appeared to swim to Baffin Island, where large gaps in locations occurred over a period of 2-3 weeks and then bears first transmission after the gaps was onshore on Baffin Island.

Additionally, in five cases events were identified where satellite collars were functioning normally and transmissions from individual bears ceased when the bear was in <10% sea ice and several hundred kilometers from shore. These events appeared to be the start of what would have been swimming events, though the bears never reappeared on shore. (See Figure 4.18abc for examples).

Maternity denning – Maternity denning is described in detail in Escajeda (2016). Overall we found 21 dens between 2009 and 2015, including 16 maternity dens (Figure 4.19) and five shelter dens (Figure 4.20). In the 1990s, Ferguson et al. (1997) found 29 dens between 1991 and 1997 including 8 maternity dens and 21 shelter dens (Table 4.9). All but one of the dens were located on land (one 1990s shelter den was located on landfast ice inside a fjord near the shore of Baffin Island). In the 2000s, most maternity dens were found on Baffin Island except for one den on Coburg Island and one den in Melville Bay, Greenland. The maternity den in Greenland was included in the phenology analyses but not the habitat characteristics analyses because of glacial ice. The lowest latitude for the 1990s dens was 66.4° N, and 67.5° N for the 2000s dens.

Among the three bears that built maternity dens twice in our dataset only one exhibited fidelity to a maternity denning area. The bear built a den in 2012 that was 1.25 km away from her previous denning site in 2009 on a peninsula close to Eglinton Fjord, Baffin Island. The other females denned in areas far from previous den sites and showed no fidelity. Also, one female built two maternity dens in consecutive years (2011 and 2012).

The majority of maternity dens were found on a north-facing slope ($n = 21$) and were located within 21 km of the coast (Table 4.10). All of the maternity dens were located further inland than shelter dens, on average 8.5 km from the coast in comparison to 6 km among shelter dens. The maternity dens were also located at higher elevations ($\bar{x} = 524.2$ m) in comparison to shelter dens ($\bar{x} = 395.7$ m) ($p = 0.086$). One bear denned a little less than 35 km away from Qikiqtarjuaq, Baffin Island, Canada, however most bears denned far from human settlements ($\bar{x} = 143.1$ km).

Adult female polar bears in BB in the 2000s spent on average less time in their dens ($\bar{x} = 167.1$ days, SD = 27.6 days) than in the 1990s ($\bar{x} = 194.1$ days, SD = 21 days; Table 4.11). There was a significant difference in den duration between the two time periods with a p-value of 0.017 (Figure 4.21). Timing of entry in the dens differed significantly among the two periods ($p = 0.018$), however no significant difference was found among exit dates ($p = 0.399$; Figure 4.22). The median entry date for dens in the 2000s dataset (3 October) was more than a full month later than the median date of entry for dens in the 1990s (28 August). Therefore, differences in entry dates accounted for the observed difference in duration among the two time periods. There was no significant correlation between latitude and den entry ($p = 0.383$) and exit dates ($p = 0.212$) for the Baffin Bay maternity dens, and the negative correlation of den duration with latitude was not significant ($p = 0.278$) (Table 4.11).

The dates of entry onto land in the 2000s significantly differed from the 1990s. The median first date on land among the Baffin Bay maternity denning bears was 7 August in the 2000s (SD = 9.1 days) compared to 25 August in the 1990s (SD = 19 days) (Figure 4.23). This follows well with the results examining first date on land for all collared females in BB, not just those that build maternity dens in fall.

The elevation and slope significantly differed between the 1990s and 2000s maternity dens ($p = 0.003$; Table 4.12). There were more dens at mid to high elevations and steeper slopes in the 2000s than in the 1990s (Figure 4.11). The average elevation and slope among the 2000s Baffin Bay maternity dens (elevation: $\bar{x} = 707$ m, SD = 285; slope: $\bar{x} = 23.1°$, SD = 7.4) was double that of the 1990s dens (elevation: $\bar{x} = 351$ m, SD = 194.5; slope: $\bar{x} = 11.9°$, SD = 6.4). Although most dens were found at southern-facing aspects in the 2000s and most were found at northern-facing aspects in the 1990s (Figure 4.24), there was no significant difference between the two time periods ($p = 0.392$). Females in BB maintained dens at similar distances to the coast between the two time periods.

The PCA ordination analysis on the Baffin Bay maternity dens matrix produced two principal components (PC) that together explained 65.69% of the variation (Table 4.13). The first component, PC 1, explained 37.31% of the variation and had strong loadings from elevation and slope (loadings > 0.6 or < -0.6 were considered significant; Table 4.13). PC 2 explained almost a third of the variation at 28.38% and was strongly loaded by aspect. We tested the statistical significance of the first four eigenvalues by applying a Monte Carlo randomization test and found both PC 1 ($p = 0.455$) and PC 2 ($p = 0.4$) to be insignificant. The perMANOVA analysis detected a significant difference between the habitat variables of the year groups ($p = 0.003$), but not the latitudinal zone group ($p = 0.775$).

Year groups were visualized in ordination space using a PCA biplot, with dispersion ellipses drawn around the year groups using the ordiellipse function from the “vegan” package (Figure 4.25). The ellipses are drawn around the standard deviations of the point scores, and the directions of their principal axes are defined by the weighted correlations (Oksanen et al. 2013). In the biplot, most of the 2000s maternity dens are positioned to the left of the plot whereas the

1990s dens are on the right. Since elevation and slope had the highest loadings for PC 1 and were both negative, the dens in the left half of the biplot have higher elevation and slope than those to the right. There was little overlap in the ellipses between time periods, which reinforces results showing a significant difference in habitat characteristics between the 1990s and the 2000s. The DISPER test on group dispersion did not indicate a significant difference in the variances among the year or zone groups. Therefore the observed difference between the year groups cannot be attributed to variance alone.

4.4. Discussion

Overall there have been large changes in BB habitat and BB polar bear habitat use since the 1990s. The sea-ice habitat has significantly declined over the period of satellite observations, especially since the mid-1990s. The length of summer (number of days from sea-ice retreat in spring to sea-ice advance in fall) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 4 percent/decade. The general pattern of melt has not changed but occurs about 3-4 weeks earlier than in 1990s. These patterns are consistent with that observed for the whole Arctic and has been reported in other polar bear subpopulations.

These changes have had impacts on the movements and habitat choice by polar bears in BB. Movement rates of adult females have declined significantly during summer in the 2000s largely due to disappearance of offshore and archipelago summertime sea ice. Bears are now concentrated on shore on Baffin Island in contrast to the 1990s where bears ranged more widely in summer and had access to sea ice. Some bears also spend the summer in NW Greenland at glacier fronts. This results in localized on-land movements and reduced movement rates in the

2000s. Bears are significantly less likely to move from land onto sea ice in summer in the 2000s than in the 2000s. Adult female bears are significantly closer to land in all months in the 2000s except at the end of breakup (June-July), when they stay on remnant offshore sea ice as long as possible to maximize feeding. This follows well with observations in BB of bears being closer to communities and in higher densities onshore (Dowsley 2005).

Adult female bears use significantly lower sea-ice concentrations in winter and spring in the 2000s than the 1990s. Bears have stronger preferences to be closer to the 300 m depth contour (on shelf waters) in the 2000s, which is also a proxy for land thus meaning bears are closer to the coast. Assigning this to a shift in preference is difficult given the concurrent changes in habitat (e.g., late sea-ice formation in fall influences how far offshore polar bears could potentially be in winter). Models indicated that sea-ice concentration alone does not determine preferred habitat, adult females select for lower sea-ice concentrations if it allows them access to continental shelf waters (<300 m).

Potential long-distance swimming events in BB were, defined as bears traveling rapidly from central Baffin Bay to Baffin Island in summer on sea ice <10% concentration with reduced or no collar transmissions. This was observed in both decades but was more frequent in the 2000s. This has been documented for other polar bear subpopulations were sea ice is increasing and springtime breakup occurs earlier increasing the frequency of long-distance swimming (Pagano et al. 2012).

Overall adult females in BB spend significantly more time on land. Arrival dates on Baffin Island in summer are one month earlier in 2000s. The amount of time bears spent on land has increased by 20-30 days since the 1990s. This follows well with studies in other areas that show similar pattern with sea-ice loss (Atwood et al. 2015, Rode et al. 2015). Adult females in

BB in the 2000s no longer arrive on Devon or Ellesmere Islands but only use Baffin Island. Of note some bears remain in Melville Bay in summer.

Maternity denning appears to have changed in association with environmental changes. Entry dates into maternity dens are >1 month later in the 2000s, although exit dates have not changed. Overall the period of maternity den duration is significantly shorter in BB in the 2000s. The first date of arrival on land by pregnant females is significantly earlier in the 2000s than the 1990s, following well with that of all adult females (see above). Finally habitat selection for den sites has changed, maternity dens in the 2000s now occur at higher elevations and steeper slopes than maternity dens in the 1990s. This may be due to changes in snow cover (reduced snow cover at lower latitudes) though more detailed habitat availability studies are needed.

4.5. Literature Cited

Amante, C., and B. W. Eakins. 2009. ETOPO1 1 Arc-Minute Global Relief Model: Procedures, Data Sources and Analysis. NOAA Technical Memorandum NESDIS NGDC-24, National Geophysical Data Center, NOAA. doi:10.7289/V5C8276M (http://www.ngdc.noaa.gov/mgg/global/global.html).

Amstrup, S. C., and C. L. Gardner. 1994. Polar bear maternity denning in the Beaufort Sea. Journal of Wildlife Management 58:1–10.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Anderson, M. J. 2006. Distance-based tests for homogeneity of multivariate dispersion. Biometrics 62:245–253.

Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015. Demographic composition and behavior of polar bears summering on shore in Alaska:

U.S. Geological Survey Administrative Report, 26 pp.

Cavalieri, D. J., C. L. Parkinson, P. Gloersen, and H. J. Zwally. 1996, updated yearly. Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS passive microwave data, Version 1. NASA National Snow and Ice Data Center Distributed Active Archive Center, Boulder, Colorado, USA, doi: http://dx.doi.org/10.5067/8GQ8LZQVL0VL.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Escajeda, E. D. 2016. Identifying shifts in maternity den phenology and habitat characteristics of polar bears (*Ursus maritimus*) in Baffin Bay and Kane Basin. MSc Thesis, University of Washington, Seattle, Washington, USA.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use by polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Ferguson, S. H., M. K. Taylor, A. Rosing-Asvid, E. W. Born, and F. Messier. 2000. Relationships between denning of polar bears and conditions of sea ice. Journal of Mammalogy 81:1118–1127.

Fischbach, A. S., S. C. Amstrup, and D. C. Douglas. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. Polar Biology 30:1395–1405.

Harris, R. B., S. G. Fancy, D. C. Douglas, G. W. Garner, S. C. Amstrup, T. R. McCabe, and L. F. Pank. 1990. Tracking wildlife by satellite: current systems and performance. U.S. Fish and Wildlife Service Technical Report 30, Washington, D.C., USA.

Jakobsson M., L. Mayer, B. Coakley, J. A. Dowdeswell, S. Forbes, B. Fridman, H. Hodnesdal, R. Noormets, R. Pedersen, M. Rebesco, H. W. Schenke, Y. Zarayskaya, D. Accettella, A. Armstrong, R. M. Anderson, P. Bienhoff, A. Camerlenghi, I. Church, M. Edwards, J. V. Gardner, J. K. Hall, B. Hell, O. Hestvik, Y. Kristoffersen, C. Marcussen, R. Mohammad, D. Mosher, S. V. Nghiem, M. T. Pedrosa, P. G. Travaglini, and P. Weatherall. 2012. The international bathymetric chart of the Arctic ocean (IBCAO) Version 3.0. Geophysical Research Letters 39 (12).

Kenkel, N. C. 2006. On selecting an appropriate multivariate analysis. Canadian Journal of Plant Science 86:663–676.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

McGarigal, K. 2015. BIOSTATS documentation. University of Massachusetts, Amherst, Massachusetts, USA.

Messier, F., M. K. Taylor, and M. A. Ramsay. 1994. Denning ecology of polar bears in the Canadian Arctic Archipelago. Journal of Mammalogy 75:420–430.

Oksanen, J., F. G. Blanchet, R. Kindt, P. Legendre, P. R. Michin, R. B. O'Hara, P. S. Simpson, P. Solymos, H. H. Stevens, and H. H. Wagner. 2013. Vegan: Community Ecology Package. R package version 2.0–9.

Pagano, A. M., G. M. Durner, S. C. Amstrup, K. S. Simac, and G. S. York. 2012. Long-distance swimming by polar bears (*Ursus maritimus*) of the southern Beaufort Sea during years of extensive open water. Canadian Journal of Zoology 90:663–676.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

R Development Core Team. 2013. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rode, K. D., R. R. Wilson, E. V. Reghr, M. St Martin, D. C. Douglas, and J. Olson. 2015.

Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere Discussion Papers, in review.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Tchernova, J. 2010. Denning characteristics and movement patterns of female polar bears with cubs in Svalbard during the first month after emergence: implication for detecting denning locations. MSc Thesis, University of Tromsø, Tromsø, Norway.

Therneau, T. 2015. A Package for Survival Analysis in S. version 2.38, http://CRAN.R-project.org/package=survival.

Wiig, Ø. 1998. Survival and reproductive rates for polar bears at Svalbard. Ursus 10:25–32.

Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. Ecosphere 5(10):136. doi:10.1890/ES14-00193.1

Table 4.1. Breakdown of adult females (AF) collared in the 1990s and 2000s. AF = adult female, AM = adult male, COY = Cub of the Year, YRL = Yearling, 2YR = 2 Year old cub.

		AF alone	AF+AM	AF+COY	AF+YRL	AF+2YR	Sum
1990s	BB	9		19	13	2	43
2000s	BB	10	2	6	12	8	38

Table 4.2. Trends in date of spring sea-ice retreat, fall sea-ice advance, fall – spring dates, and summer (June-Oct) sea-ice concentration in Baffin Bay (all depths, and shallow sub-regions). All trends are statistically significant at the 99% level according to a 2-sided F test.

Baffin Bay region	**Trend in date of spring ice retreat (days/decade)**	**Trend in date of fall ice advance (days/decade)**	**Trend in fall – spring (days/decade)**	**Trend in ice con. June-October (percent/decade)**
All depths	−7.3	+5.4	+12.7	−4.1
Depths < 300 m	−8.4	+9.7	+18.1	−3.3
Depths < 300 m Baffin I. shelf	−6.8	+3.9	+10.7	−4.7

Table 4.3. Mean monthly movement rates in BB for adult female in the 1990s and 2000s. One SE about the mean is given. We conducted a parametric test of significance between decades.

Month	Mean 1990s	SE 1990s	n 1990s	n steps	Mean 2000s	SE 2000s	n 2000s	n steps	t-test p value
1	10.29	0.88	33	195	10.07	1.38	26	225	0.8933
2	11.31	0.96	32	173	8.68	1.18	27	218	0.0898
3	11.83	1.16	32	203	9.47	1.13	27	261	0.1492
4	13.37	1.2	30	244	10.33	1	38	546	0.0558
5	14.05	0.99	31	256	9.68	0.94	37	563	0.0021
6	13.03	0.66	31	269	11.31	1.04	34	480	0.168
7	12.88	1.06	30	239	10.49	1	32	402	0.1057
8	10.52	1.17	27	193	4.77	0.63	30	337	<0.001
9	7.73	1.13	28	172	1.88	0.45	28	258	<0.001
10	5.49	0.82	38	222	4.32	1.36	27	244	0.4683
11	14.52	1.24	37	252	13.43	1.58	26	230	0.5903
12	15.82	1.19	34	213	13.79	1.76	25	233	0.3442

Table 4.4. Sea-ice resource selection function (RSF) model coefficients for the Winter season in BB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled for ease of interpretation. “Mean ice conc.10” is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. “Dist to 50%” is the distance to the 50% sea-ice concentration scaled by 100 km. “Dist to 300 m” is the distance to the 300 m depth contour scaled by units of 100 m. “Depth.100” is the absolute value of bathymetry scaled by 100 m. “Land” is the variable that describes the tendency of a bear to move from sea ice on to land.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
Mean ice conc.10	0.075	0.025	0.0025	-0.022	0.023	0.3452	0.0043
Dist to 50%.100	-0.100	0.044	0.0237	-0.107	0.052	0.0383	0.9197
Dist to 300 m.100	0.001	0.148	0.9968	-0.619	0.205	0.0026	0.0144
Depth.100	-0.032	0.010	0.0011	-0.019	0.012	0.1021	0.4266
Land	-1.617	0.295	0.0000	-2.12	0.267	0	0.2065

Table 4.5. Sea-ice resource selection function (RSF) model coefficients for the Spring season in BB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled for ease of interpretation. “Mean ice conc.10” is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. “Dist to 50%” is the distance to the 50% sea-ice concentration scaled by 100 km. “Dist to 300 m” is the distance to the 300 m depth contour scaled by units of 100 m. “Depth.100” is the absolute value of bathymetry scaled by 100 m. “Land” is the variable that describes the tendency of a bear to move from sea ice on to land.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
Mean ice conc.10	0.162	0.022	0	0.255	0.026	0	0.0066
Dist to 50%.100	-0.115	0.041	0.0047	-0.376	0.042	0	0
Dist to 300 m.100	-0.199	0.111	0.0727	-0.088	0.111	0.4265	0.4789
Depth.100	-0.014	0.007	0.0606	0.011	0.007	0.1111	0.014
Land	-1.738	0.291	0	-1.059	0.312	0.0007	0.1111

Table 4.6. Sea-ice resource selection function (RSF) model coefficients for the Summer season in BB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled for ease of interpretation. "Elev.100" is elevation scaled by units of 100 m. "Slope.10" is slope in degrees scaled by units of 10 degrees. "Aspect.10" is aspect scaled by units of 10 degrees. "BIdistCoast.10" is the distance to the smoothed Baffin Island coastline (measured from a point inland) scaled by 10 km. "Not Land" is the tendency of a bear to move from land on to sea ice.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
elev.100	-0.09	0.03	0.0033	-0.091	0.017	0	0.9627
slope.10	0.22	0.065	0.0007	0.202	0.036	0	0.8118
aspect.10	0.001	0.007	0.8484	-0.005	0.004	0.1933	0.4231
BIdistCoast.10	-0.456	0.069	0	-0.358	0.042	0	0.2243
NotLand	-1.182	0.234	0	-2.44	0.16	0	0

Table 4.7. Summary table of 78 arrival dates and 71 departure dates for individual radio-collared adult female bears arriving on Baffin Island or Devon Island (within the seasonal sea-ice ecoregion). We excluded two bears from the 1990s that arrived on Ellesmere (2 arrivals and 2 departures by two individuals = i.e., in the archipelago ecoregion cf Amstrup et al. 2008). There was a significantly (*p*<0.001) earlier arrival on land in the 2000s. No difference in departure date from land between 1990s and 2000s. Significance did not change with the inclusion of the Ellesmere bears.

Decade	**Arrival date on land**	**Departure date from land**
1990s	Aug 23 (SD 16 days) n=30	Nov 1 (SD 21 days) n=42
2000s	Aug 4 (SD 11 days) n=46	Nov 8 (SD 9 days) n=27

Table 4.8. Table of number of days (time on land) on Baffin Island (or Devon Island in the 1990s). Data composed from 56 paired arrival/departure dates from individual adult female bears. There was a significant (*p*<0.001) increase in time on land in the 2000s, between ~30 days longer between decades.

Decade	n	Min # days on land	Max # days on land	Mean # days on land	SD of mean
1990s	14	8	99	62.3	25.0
2000s	26	56	120	94.8	15.7

Table 4.9. Number of maternity and shelter dens in Baffin Bay in the 1990s and 2000s.

All Dens		Maternity Dens		Shelter Dens	
1990s	29	1990s	8	1990s	21
2000s	21	2000s	16	2000s	5
Total	50	Total	24	Total	26

Table 4.10. Summary table of the habitat characteristics for Baffin Bay maternity and shelter dens (two NW Greenland dens were not included). Elev. = elevation (meters), Asp. = aspect (degrees), CoastDist = distance to nearest coastline (kilometers).

	All Maternity Dens (*n* = 24)				**All Shelter Dens (*n* = 26)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	583.3	166.2	19.2	9.5	421.8	224.7	19.9	7.4
Min	101.0	0.0	2.4	0.4	0	–1	0	0.1
Max	1323.0	357.6	32.9	20.2	1116	357.2	46.7	54.5
Median	623.0	175.2	18.5	7.8	354	249.3	19.4	4.7
SD	306.1	102.4	8.8	6.3	320.5	124.8	12.6	10.7

	1990s Maternity Dens (*n* = 8)				**1990s Shelter Dens (*n* = 21)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	351.3	150.6	11.9	10.2	414.9	210.1	20.4	7.9
Min	131	18.4	2.4	2.2	0	–1	0	0.1
Max	623	357.6	21.4	20.2	1116	357.2	46.7	54.5
Median	279	99.1	12.1	7.6	354	247.6	20.1	4.9
SD	194.5	135.7	6.4	6.5	335.5	128.5	13.5	11.4

	2000s Maternity Dens (*n* = 15)				**2000s Shelter Dens (*n* = 4)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	707.0	174.4	23.1	9.2	458.3	301.7	17.1	4.9
Min	101.0	0.0	6.1	0.4	169.0	196.4	10.3	0.2
Max	1323.0	320.6	32.9	18.6	728.0	344.4	26.9	15.3
Median	693.0	182.2	23.0	8.3	468.0	332.9	15.5	2.0
SD	284.9	83.9	7.4	6.3	263.2	70.4	7.4	7.0

Table 4.11. Summary table of the phenology for Baffin Bay maternity dens including the Greenland maternity den. One maternity den from the 2000s data did not have an exit date and was excluded from the table. Entry and exit dates were quantified as day of year (DOY; Day #1 is 1 January).

	1990s Maternity Dens (*n* = 8)			**2000s Maternity Dens (*n* = 15)**		
	Entry DOY	Exit DOY	Duration (# days)	Entry DOY	Exit DOY	Duration (# days)
Mean	249.8	78.9	194.1	277.7	79.8	167.1
Min	230	73	163	237	60	121
Max	281	82	217	324	91	212
Median	240	79.5	201	276	80	164
SD	21.3	3.6	21	27.7	8.7	27.6

Table 4.12. Results of the two-sample Mann-Whitney U tests comparing habitat characteristics for Baffin Bay maternity and shelter dens (note that the den in Greenland was not included).

	Maternity Dens (*n* = 23)		**Shelter Dens (*n* = 25)**	
	W	***p*-value**	**W**	***p*-value**
Elevation	105	0.003	20	0.695
Slope	105	0.003	25	0.695
Aspect	74	0.392	17	0.262
Distance to Coast	55	0.776	24	0.369

Table 4.13. Summary table of results from the principal component analysis on the Baffin Bay maternity dens matrix (the Greenland maternity den was omitted). None of the principal components (PC) were significant, though PC 1 and 2 were able to capture over half of the variation in the data. Principal component loadings greater than 0.6 or less than –0.6 were considered significant (in bold).

Principal Component Analysis Summary					**Principal Component Loadings**		
	Eigenvalue	% Var.	Cum. % Var.	*p*-value		PC 1 (37.31%)	PC 2 (28.38%)
PC 1	1.49	37.31	37.31	0.465	Elevation	–0.659	0.375
PC 2	1.24	28.38	65.69	0.413	Aspect	–0.210	–0.770
PC 3	0.99	24.64	90.33	0.061	Slope	–0.604	–0.375
PC 4	0.39	9.67	100	0.908	CoastDist	–0.396	0.356

Figure 4.1. Sea-ice area in Baffin Bay (shown for all depths) for the years 1979-2014 (gray curves) using SSM-I passive microwave data. Two six-year averages are also shown (red and blue curves) that approximate the sampling dates for the MR. The threshold for defining the dates of sea-ice retreat and advance (middle horizontal dotted line) is halfway between the average March sea-ice area (upper dotted line) and the average September sea-ice area (lower dotted line).

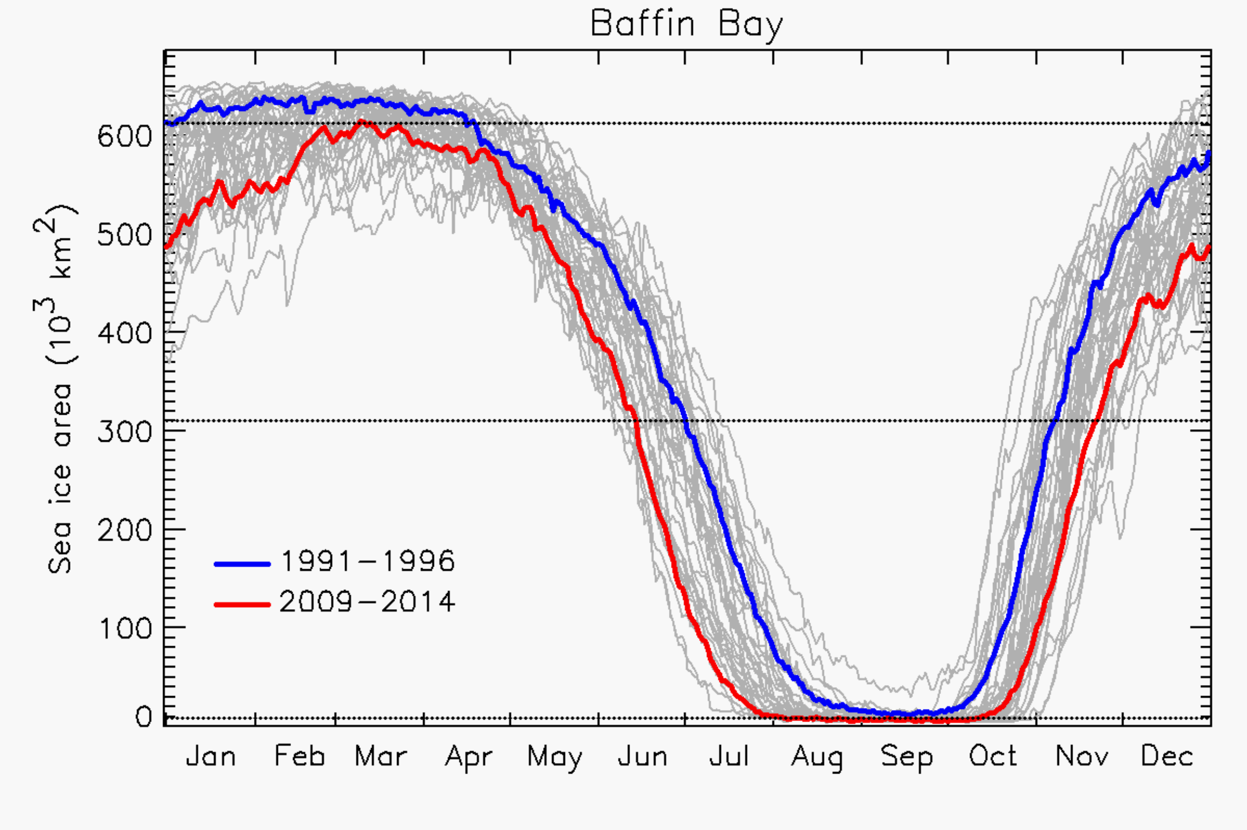
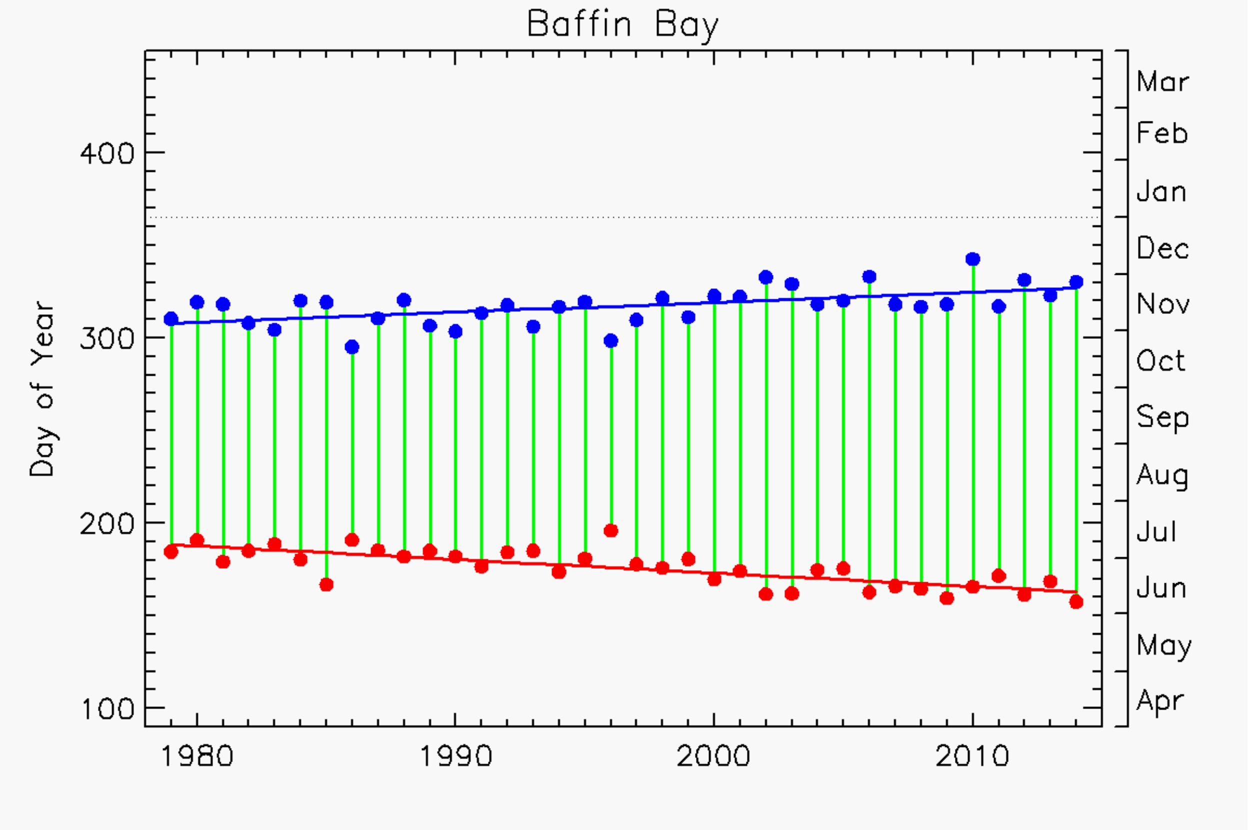


Figure 4.2. Day of spring sea-ice retreat (red circles), fall sea-ice advance (blue circles), and the interval between them (green lines), for Baffin Bay (all depths), 1979-2014. Least-squares fits to spring and fall dates are shown (red and blue lines). Trends are given in Table 4.1.



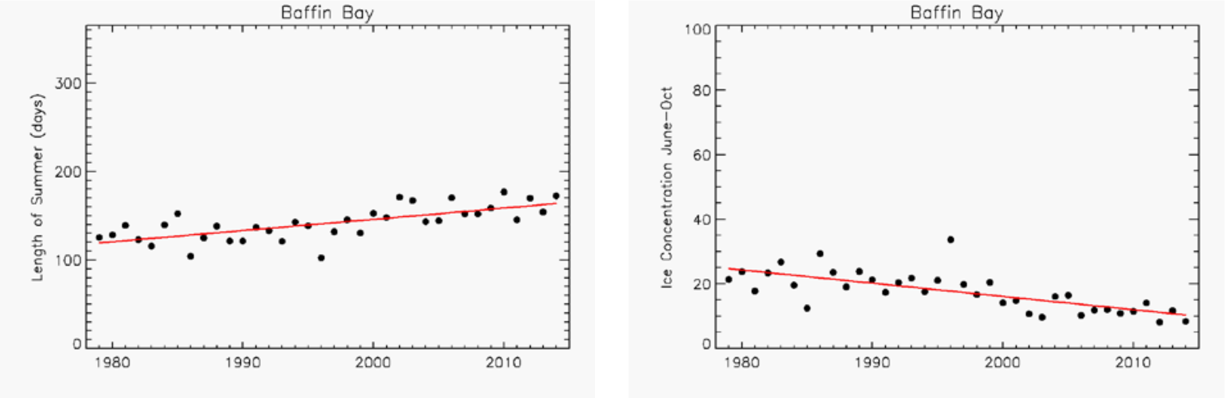


Figure 4.3. Length of summer (left) and mean sea-ice concentration during June-October (right) for Baffin Bay (all depths), 1979-2014. Length of summer is the interval from spring sea-ice retreat to fall sea-ice advance (see Figure 4.2, green lines). Least-squares fits are shown (red lines); trends are given in Table 4.1.

Figure 4.4. Trend in the number of ice-covered days, 1979-2014, color-coded for each grid cell, as indicated in the legend at left. Time series of the number of ice-covered days are shown for two specific grid cells, one in central Baffin Bay (upper right) and one close to the coast of Greenland (lower right). Least-squares fits (red lines) and numerical trends are indicated. An ice-covered day is one in which the sea-ice concentration exceeds 15%.

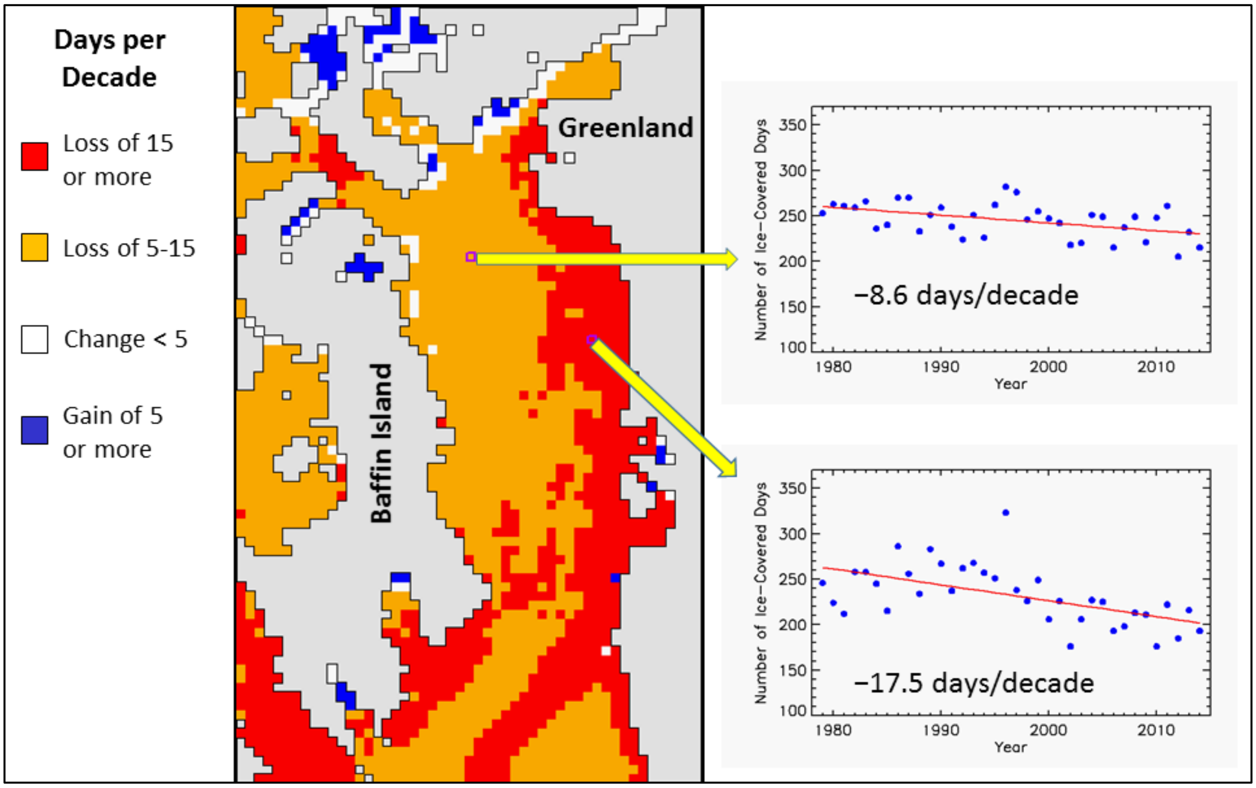


Figure 4.5. Sea-ice concentration in Baffin Bay on July 15 of every year from 1979 (upper left) to 2014 (lower right). Color coding: 15-50% (blue), 50-85% (green), 85-95% (yellow), 95-99% (orange), 99-100% (red). Black dots in Baffin Bay indicate shallow depths (< 300 m).

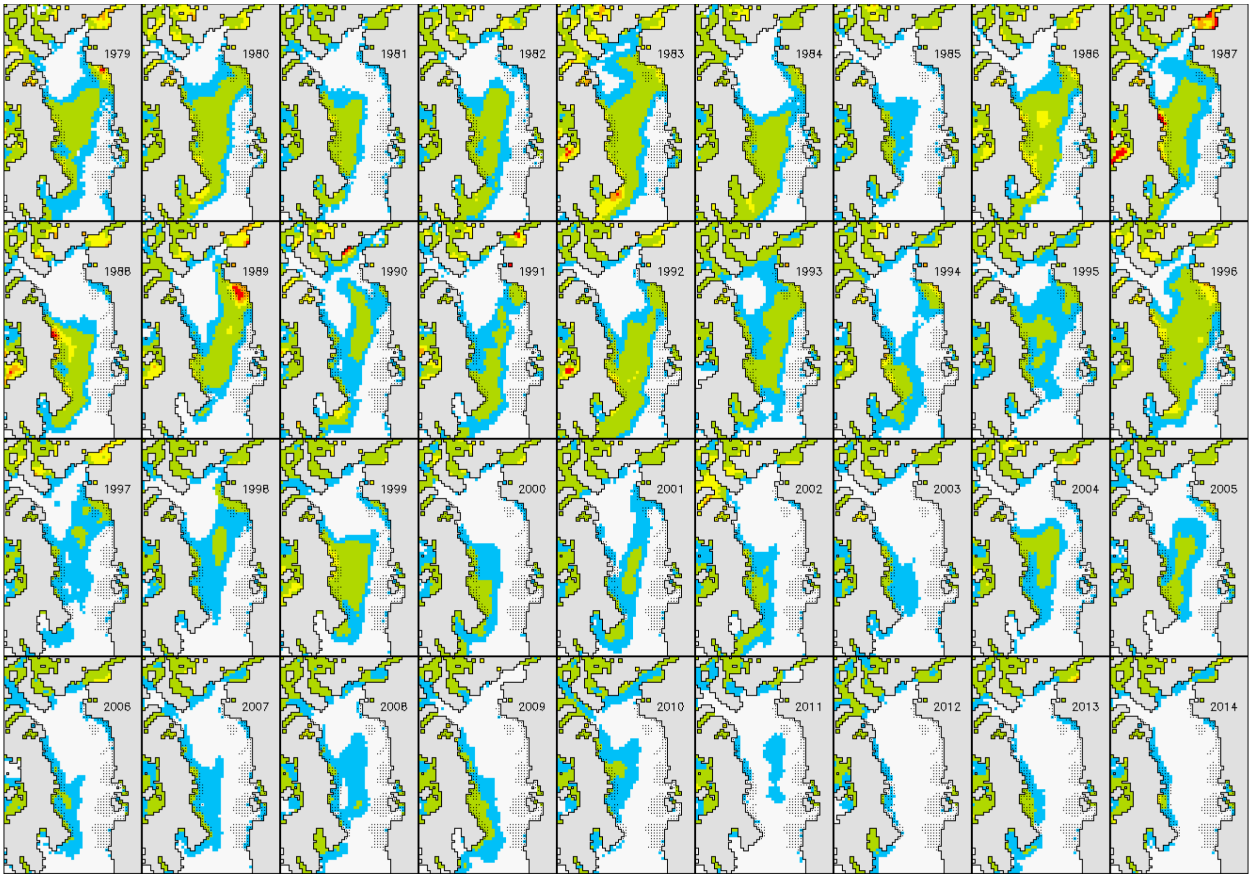


Figure 4.6. Location of the leading edge of sea ice (> 50% concentration) along the coast of Baffin Island on November 1 in the years 1991-1995 (top row) and 2009-2013 (bottom row). Color coding of sea-ice concentration is the same as in Figure 4.5. In the top row, black circles mark the leading edge of sea ice, and the horizontal dashed black line is the average position. In the bottom row, red circles mark the leading edge of sea ice, and the horizontal dashed red line is the average position. The dashed black line is the same as in the top row, showing that in the later period, sea ice has not advanced as far south by November 1 as in the early period.

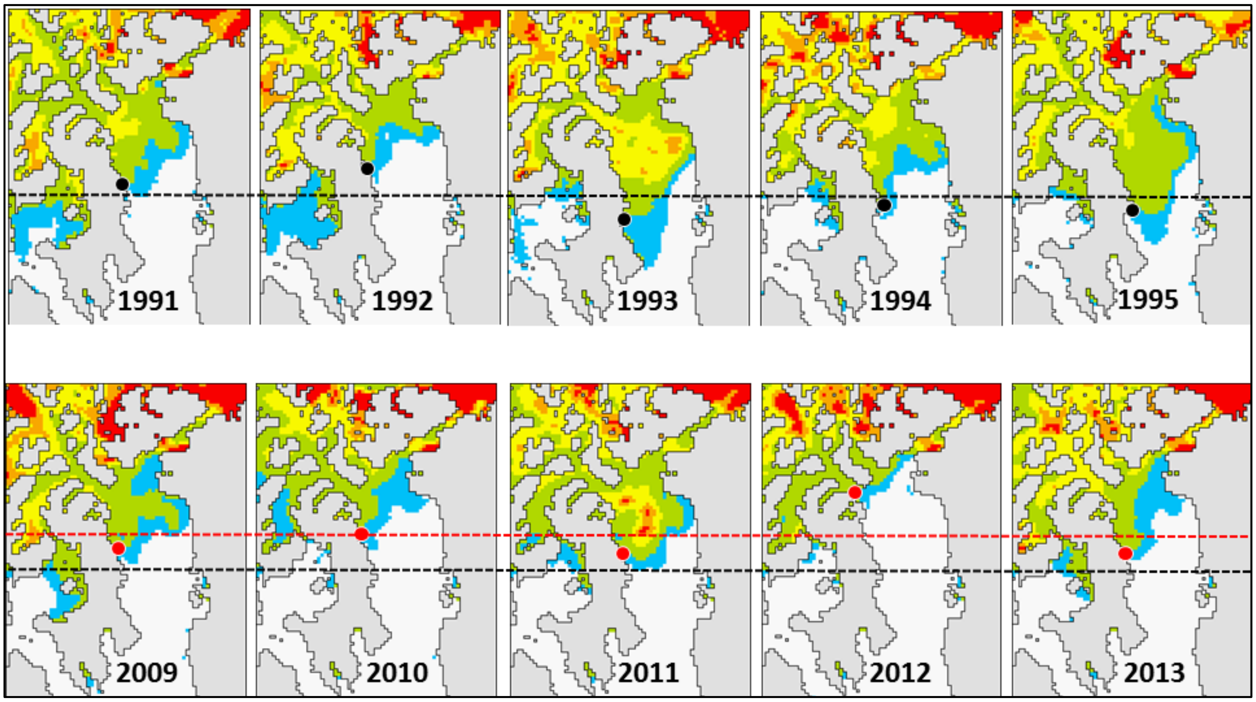


Figure 4.7. Movement rate of BB adult female bears (km/day) in the 1990s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

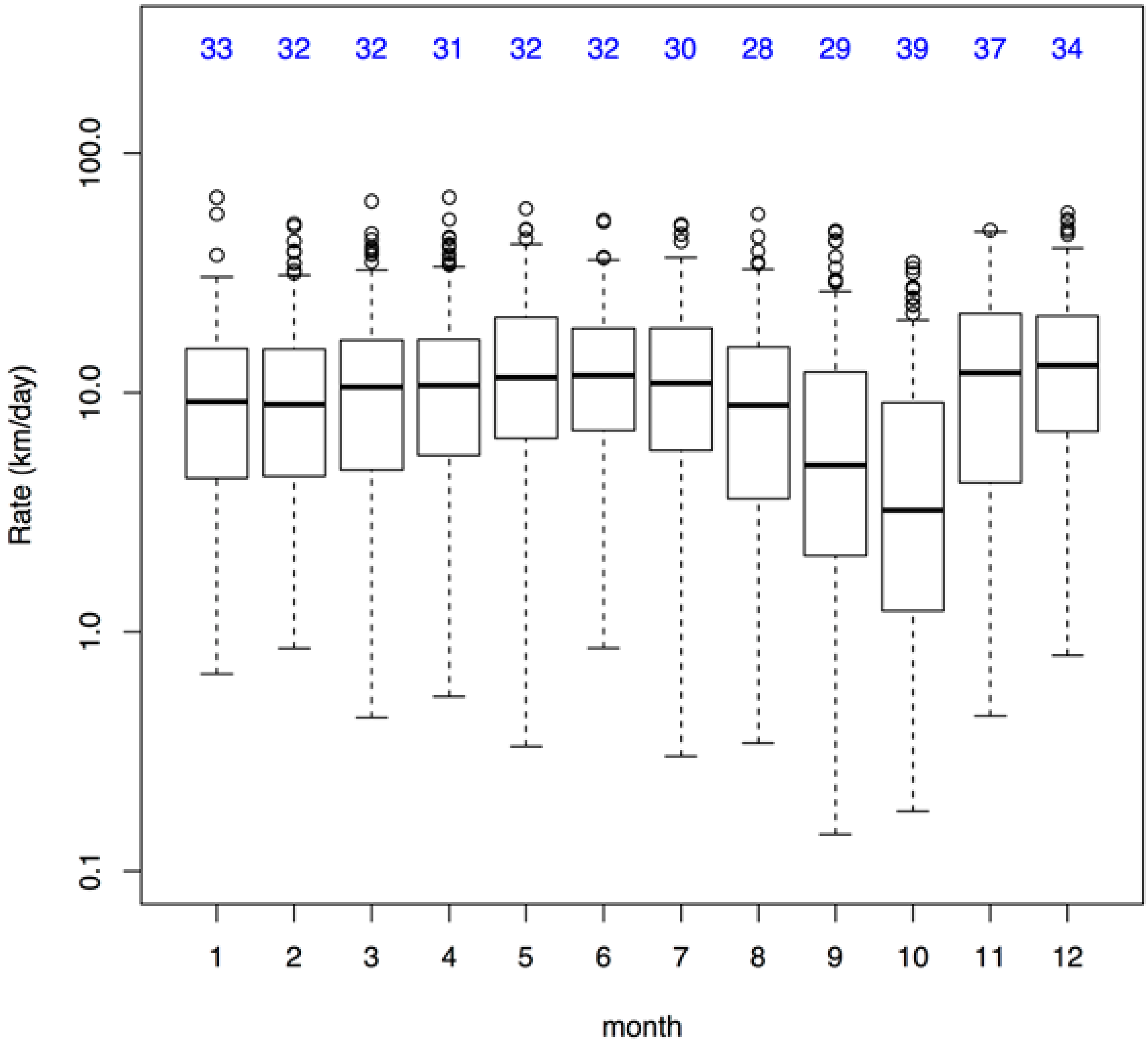


Figure 4.8. Movement rate of BB adult female bears (km/day) in the 2000s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

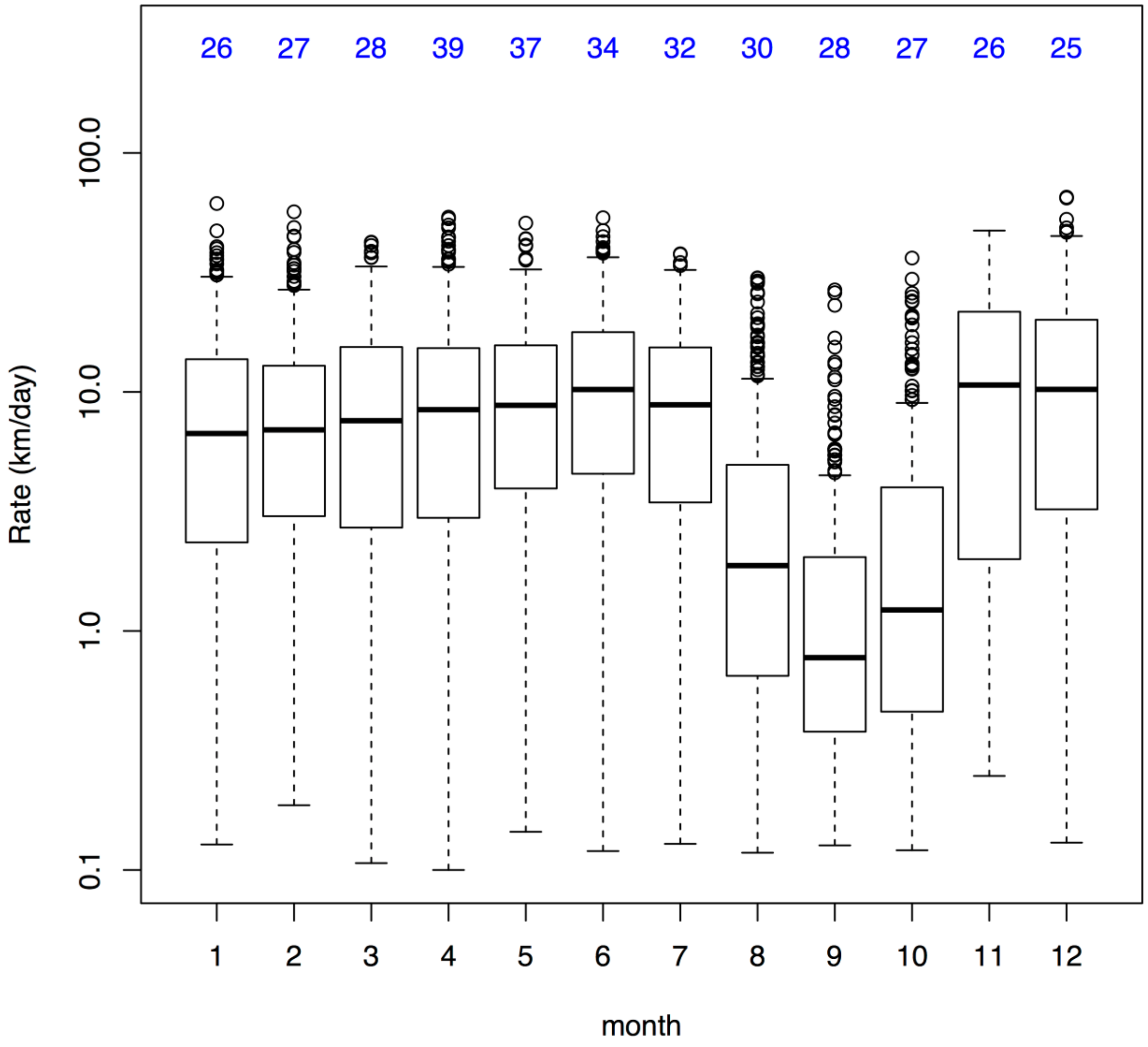


Fig 4.9. 1990s and 2000s adult female polar bear habitat use in BB for each of four sea-ice habitat variables: sea-ice concentration in small buffer, distance to 15% sea-ice concentration, distance to 50% sea-ice concentration, and distance to the nearest land. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer) used in the analysis. SSM/I sea-ice concentration is used in both decades. All positions are for bears on sea ice or water and resident bears in Melville Bay are excluded. Months 8 -10 also represent land use by bears. The small numbers in blue and red represent the number of movements for each month and decade.

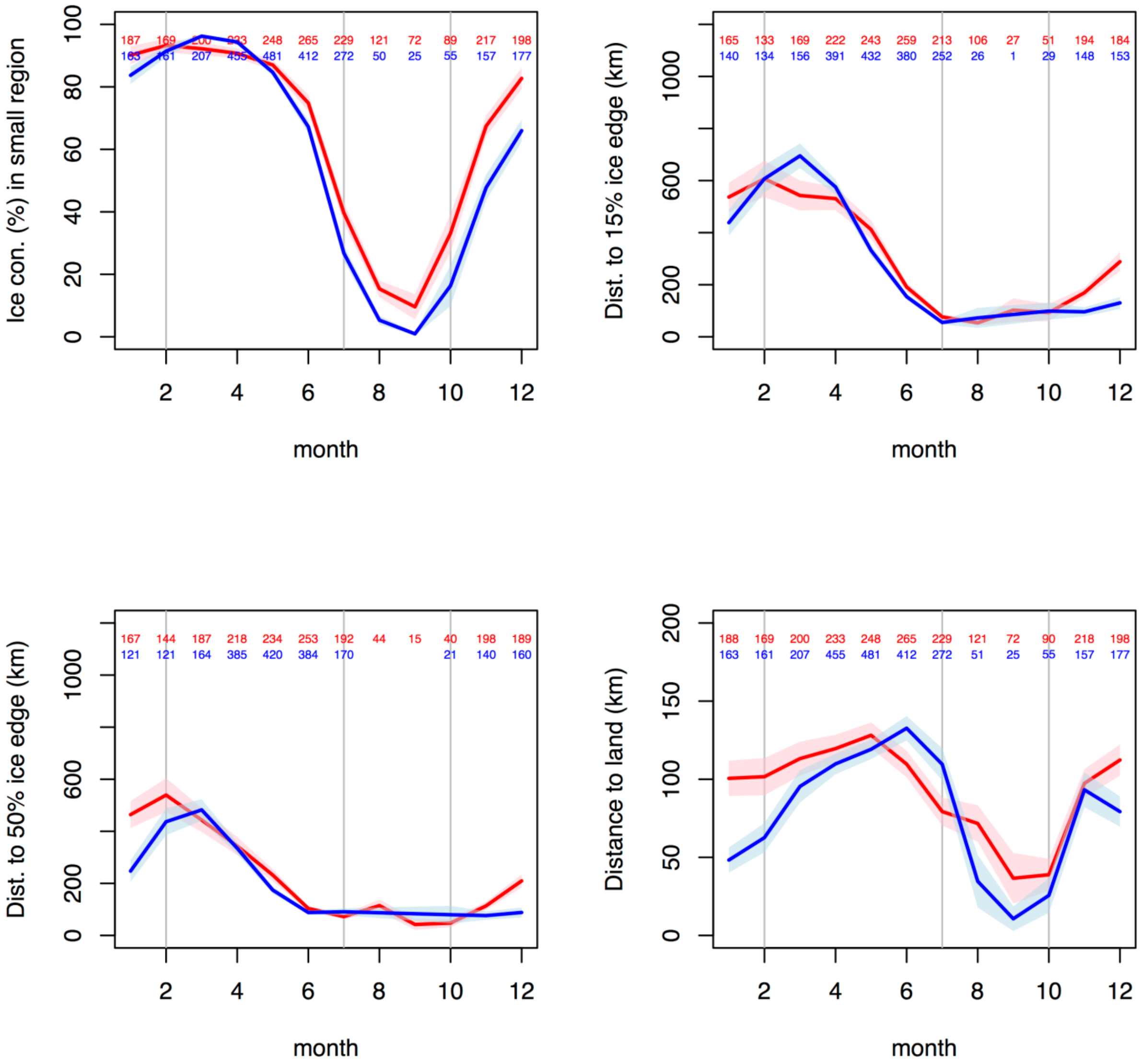


Fig 4.10. 1990s and 2000s adult female polar bear habitat use in BB for each of three sea-ice habitat variables: distance to 300 m depth contour, depth (bathymetry), and percentage of observations on the sea ice. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer). SSM/I sea-ice concentration is used in both decades. All positions are for bears on ice or water and resident bears in Melville Bay are excluded. Months 8 -10 also represent land use by bears. The small numbers in blue and red represent the number of movements for each month and decade.

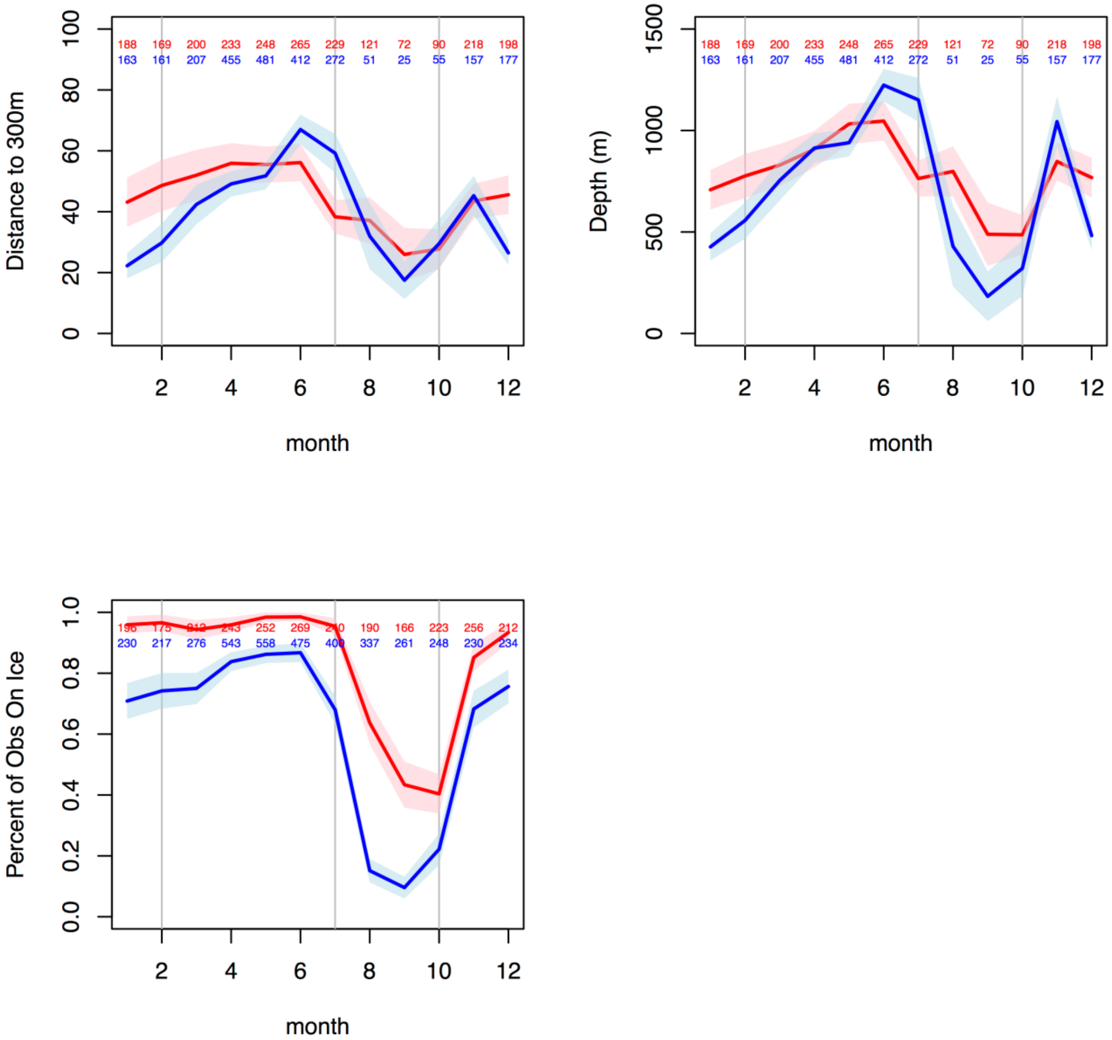


Figure 4.11. 1990s and 2000s adult female polar bear habitat use for each of four land habitat variables: Elevation, Slope, Aspect and Distance to the Baffin Island coast (from inland). Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. The small numbers in blue and red represent the number of movements for each month and decade.

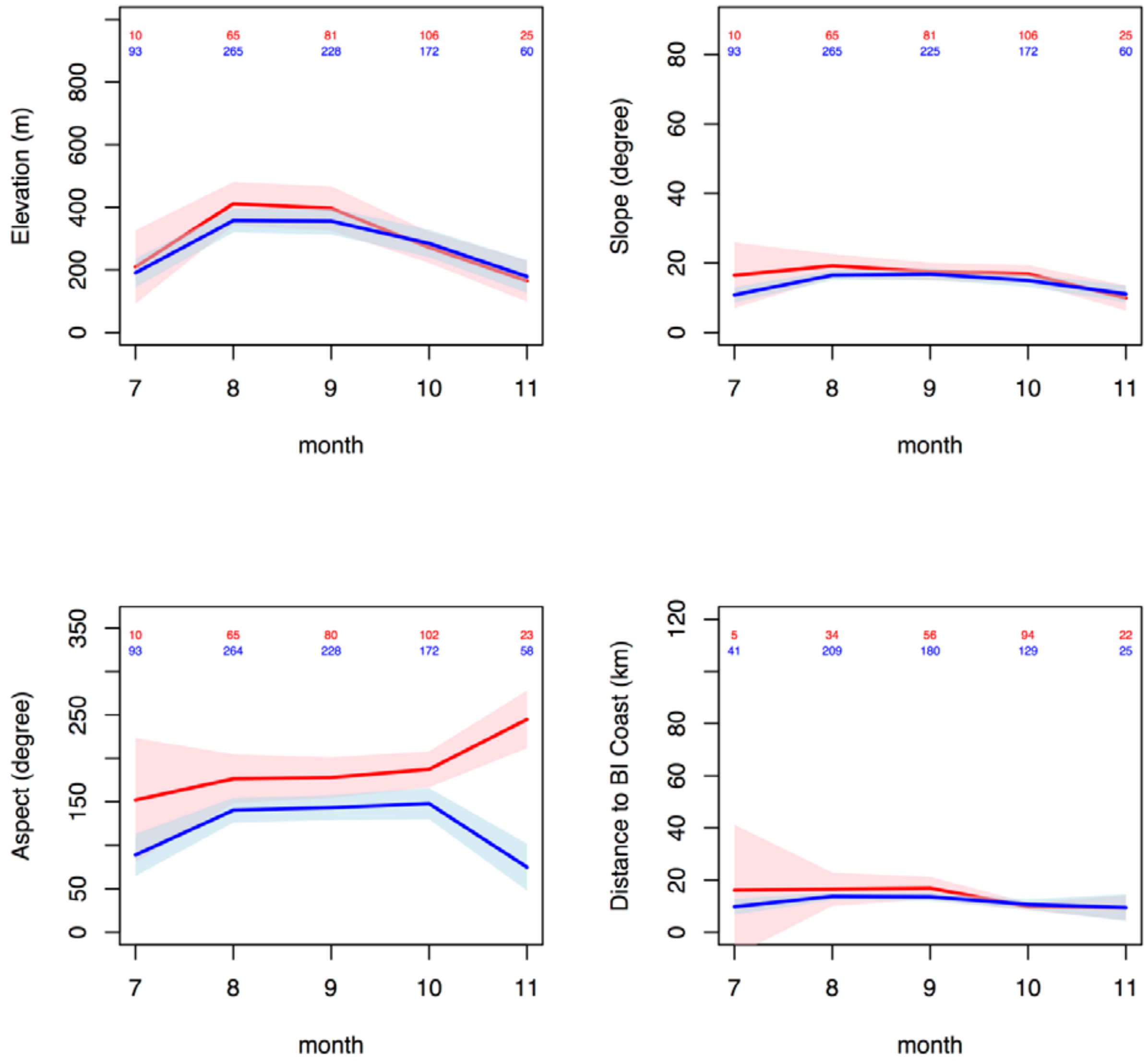


Figure 4.12. Map of ASTER Digital Elevation Model (DEM) data and IBCAO bathymetry shown for Baffin Island used in the RSF analyses. Partial tracks from a single adult female (68005) collared in 2009 in West Greenland are shown for reference.

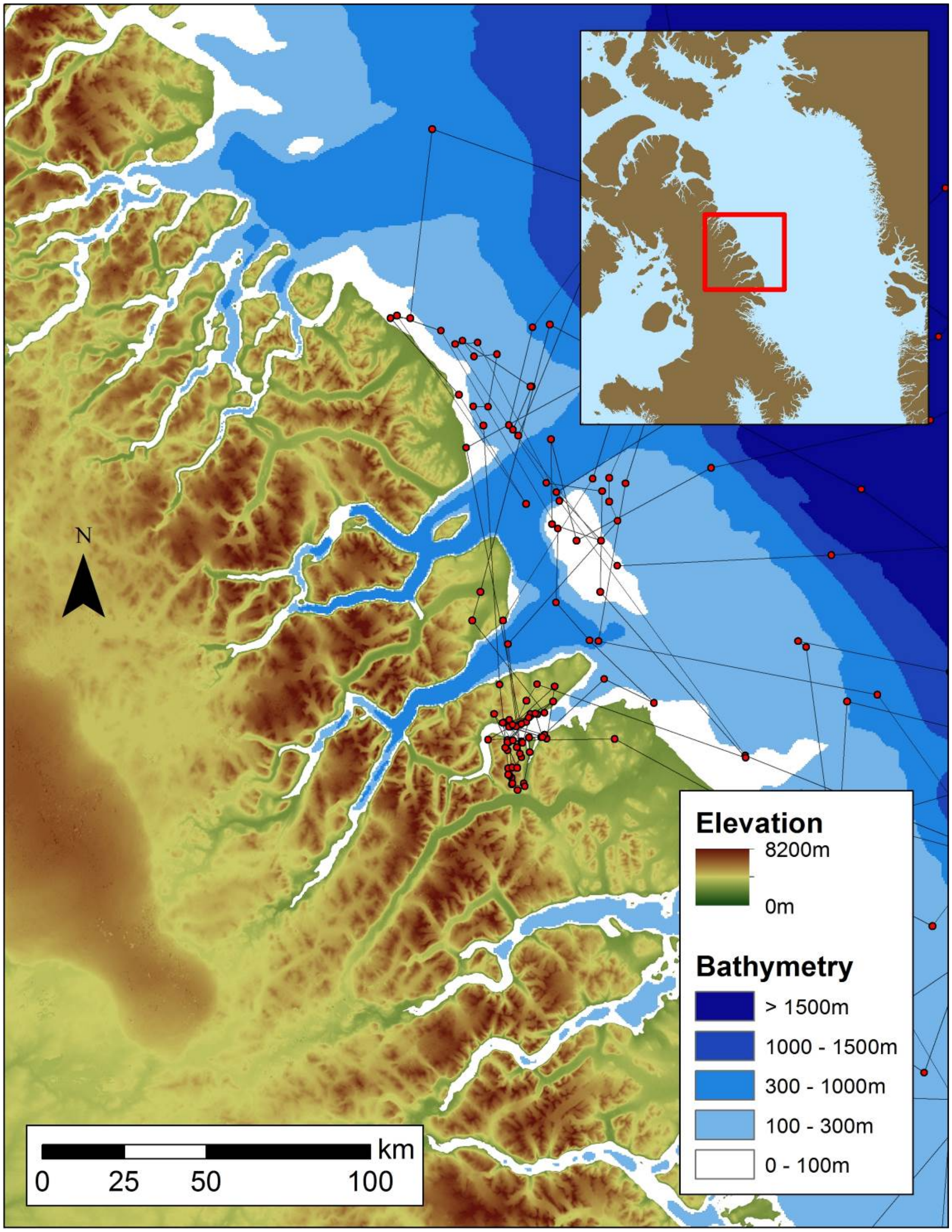


Figure 4.13. Arrival dates (on land) for adult female BB polar bears in the 1990s and 2000s. This does not include two BB bears arriving on Ellesmere Island in the 1990s. Also bears in glacial fronts in Melville Bay not included as they remained in coastal habitat year-round.

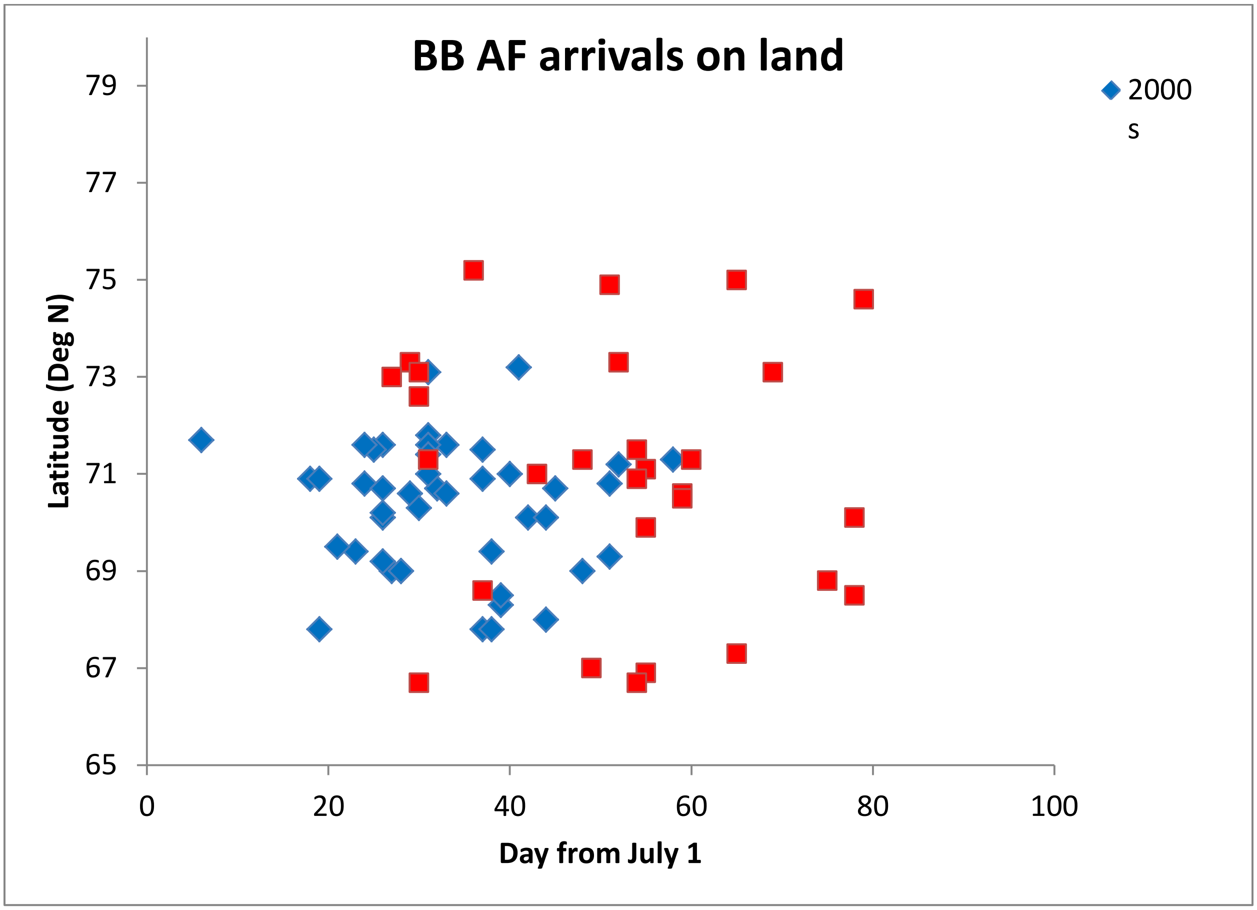


Fig 4.14. Departure dates in fall (off land) for adult female BB polar bears in the 1990s and 2000s.

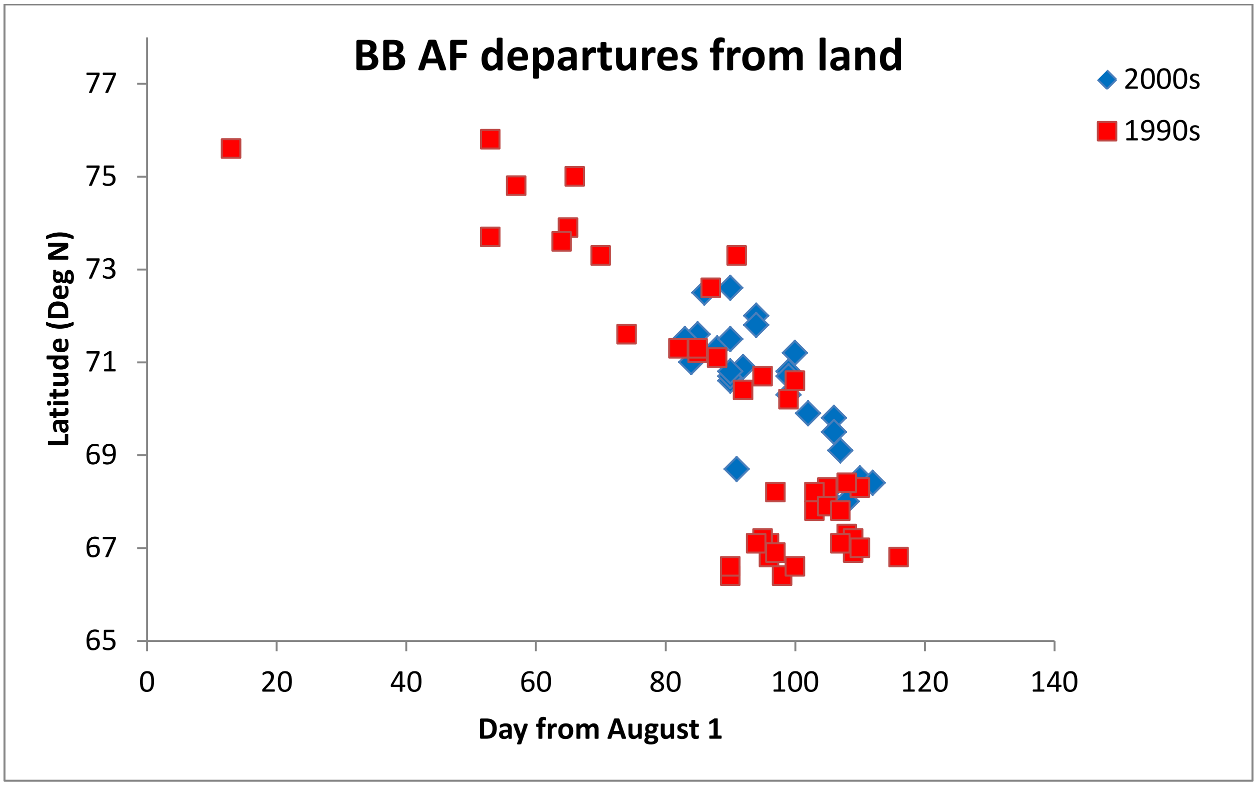


Figure 4.15. Correlations between the sea-ice retreat and advance metrics (see methods) and the arrival and departure dates on Baffin Island for adult females in both decades.

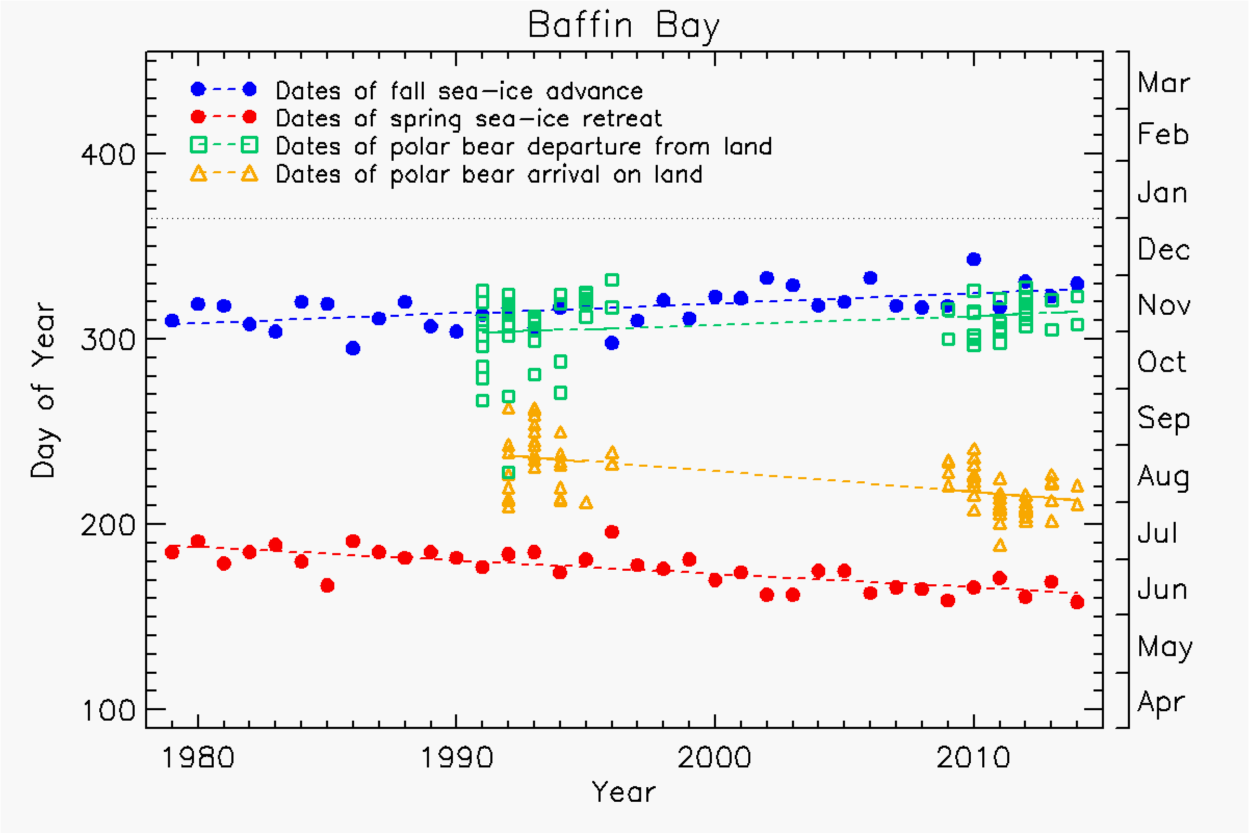
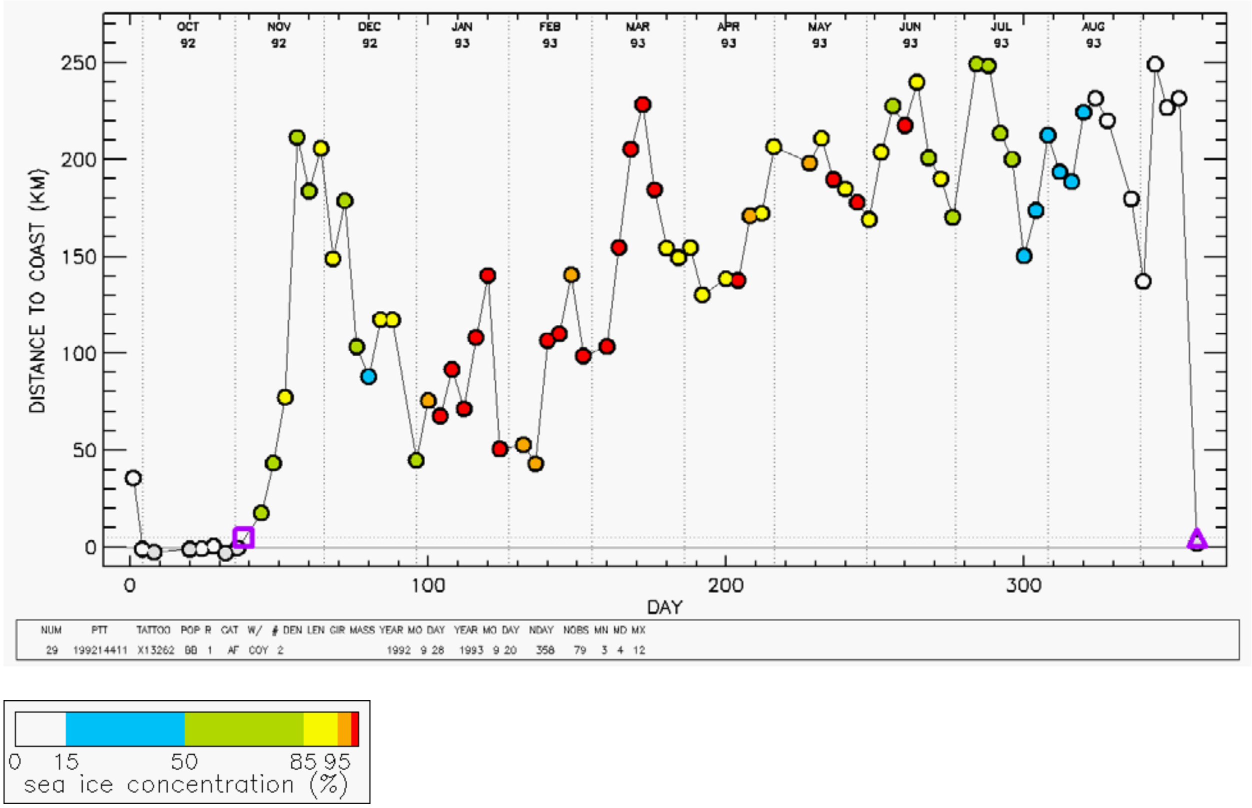
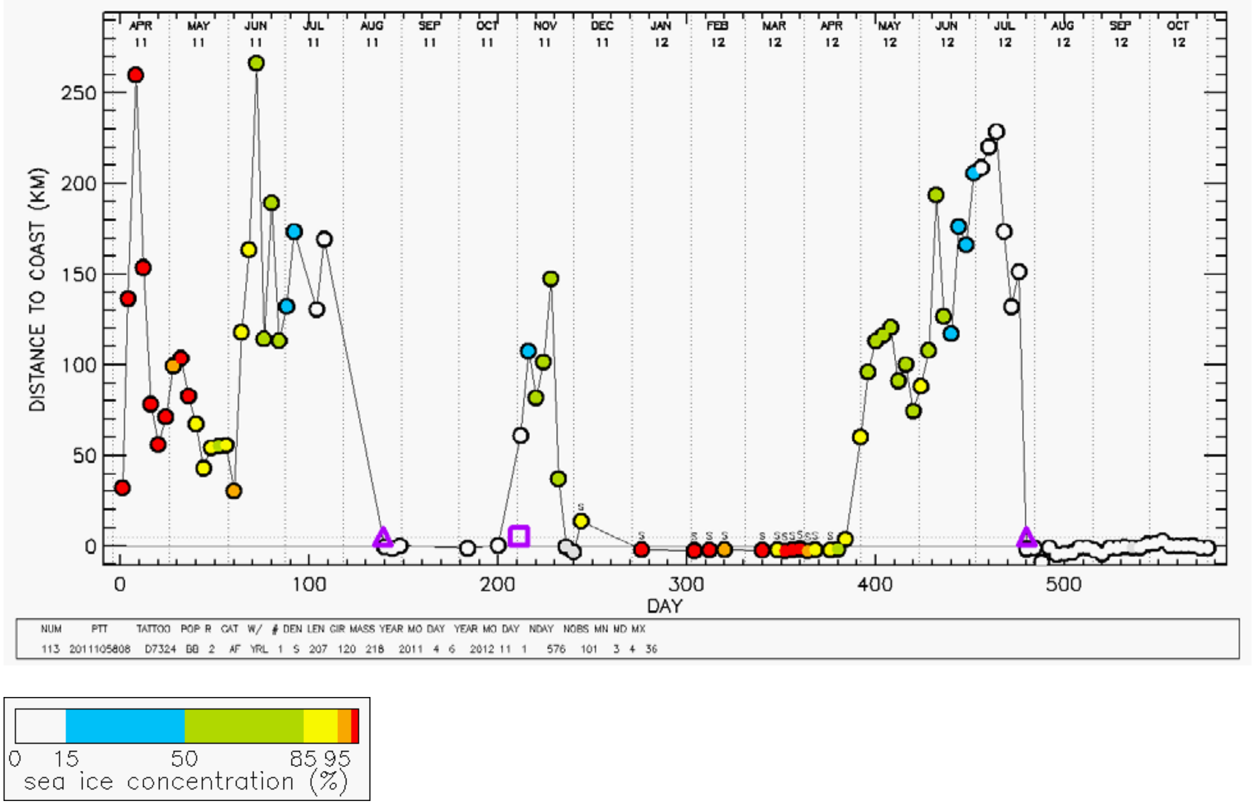


Figure 4.16. Distance to land vs. time plot for polar bear captured in 1992 (PTT 14411) showing a potential swimming event in September 1993. Purple squares denote the departure date from land on to sea ice and purple triangles denote the arrival date from sea ice/water on to land for that individual.



NUM	PTT	TATTOO	POP	R	CAT	W/	#	DEN	LEN	GIR	MASS	YEAR	MO	DAY	YEAR	MO	DAY	NDAY	NOBS	MN	MD	MX
29	199214411	X13262	BB	1	AF	COY	2					1992	9	28	1993	9	20	358	79	3	4	12

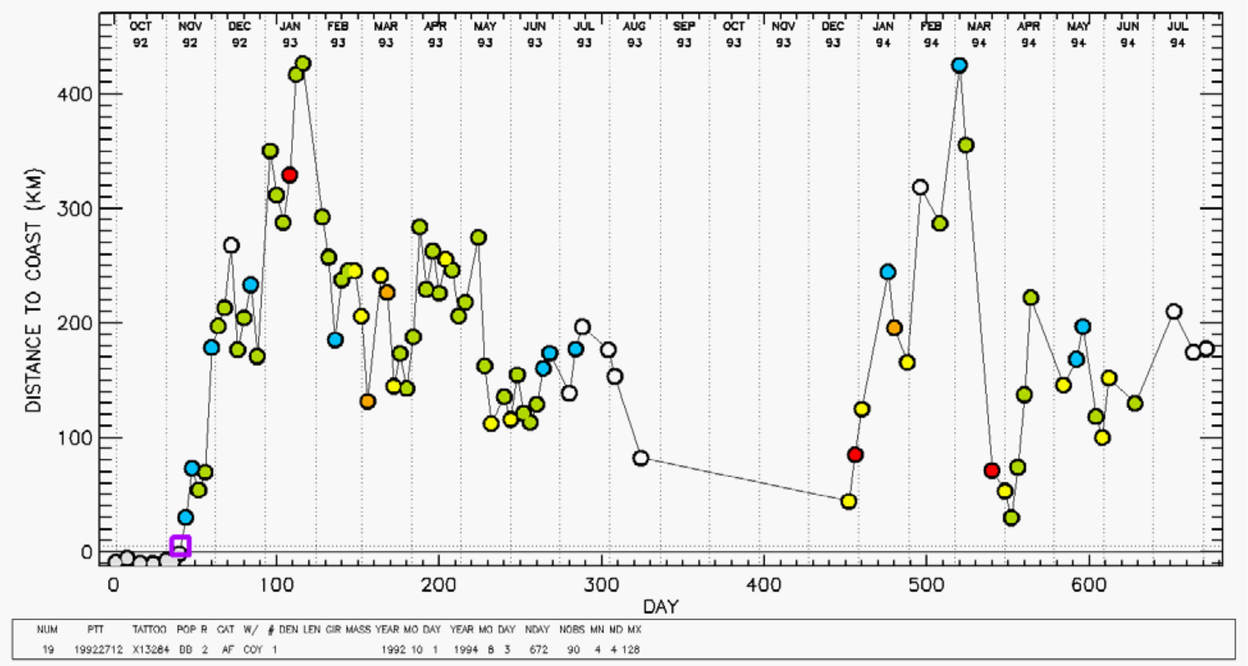
Figure 4.17. Distance to land vs. time for polar bear captured in 2011 (PTT 105808) showing an example of a swimming event in both July 2011 and July 2012. Purple squares denote the departure date from land on to sea ice and purple triangles denote the arrival date from sea ice/water on to land for that individual.



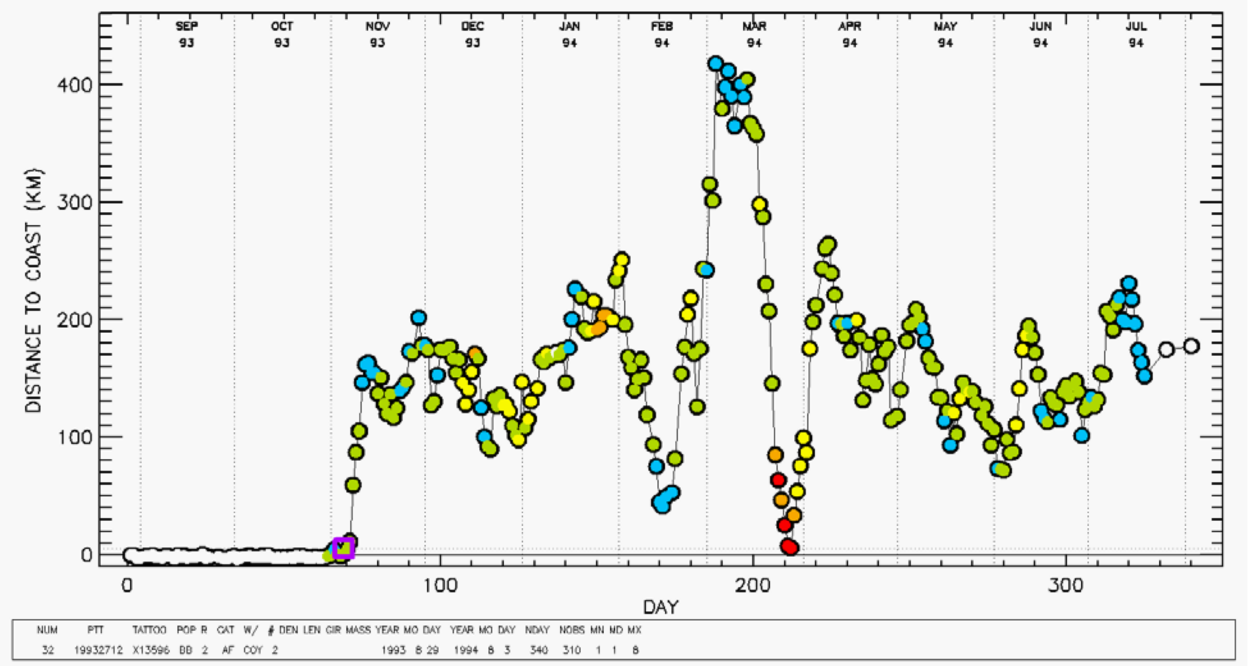
NUM	PTT	TATTOO	POP	R	CAT	W/	#	DEN	LEN	GIR	MASS	YEAR	MO	DAY	YEAR	MO	DAY	NDAY	NOBS	MN	MD	MX
113	2011105808	D7324	BB	2	AF	YRL	1	S	207	120	218	2011	4	6	2012	11	1	576	101	3	4	36

Figure 4.18. a) Track of adult female bear tagged in 1992 where the last three locations from the individual occur between July-August 1994 in open water 180 km from land; **b)** track of adult female bear collared in 1993 through August 1994, where the last two locations occur in open water 180 km from land; and, **c)** track of an adult female bear collared in April 2011 through July 2011. The last position is 80 km from land in <15% sea ice.

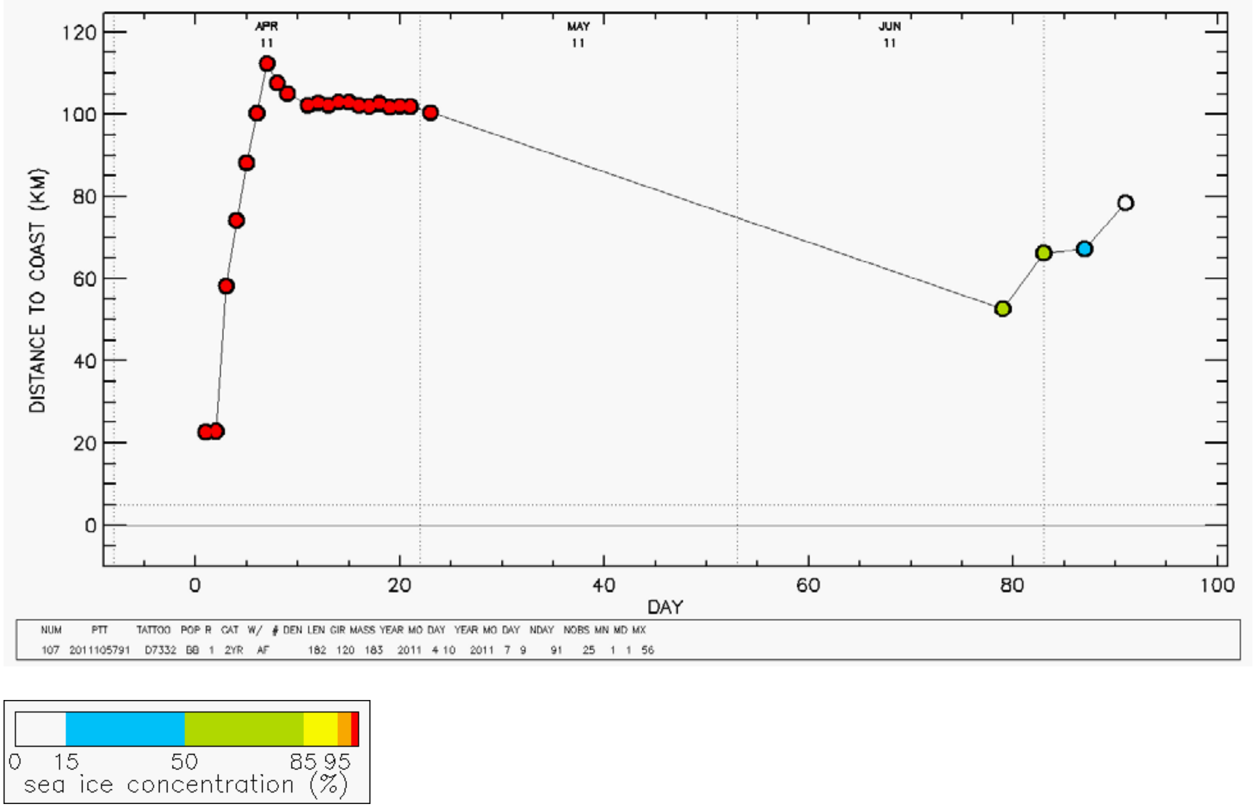
a)



b)



c)



NUM	PTT	TATTOO	POP	R	CAT	W/	#	DEN	LEN	GIR	MASS	YEAR	MO	DAY	YEAR	MO	DAY	NDAY	NOBS	MN	MD	MX
107	2011105791	D7332	BB	1	2YR	AF			182	120	183	2011	4	10	2011	7	9	91	25	1	1	56

Figure 4.19. Maternity den locations in BB by decade.

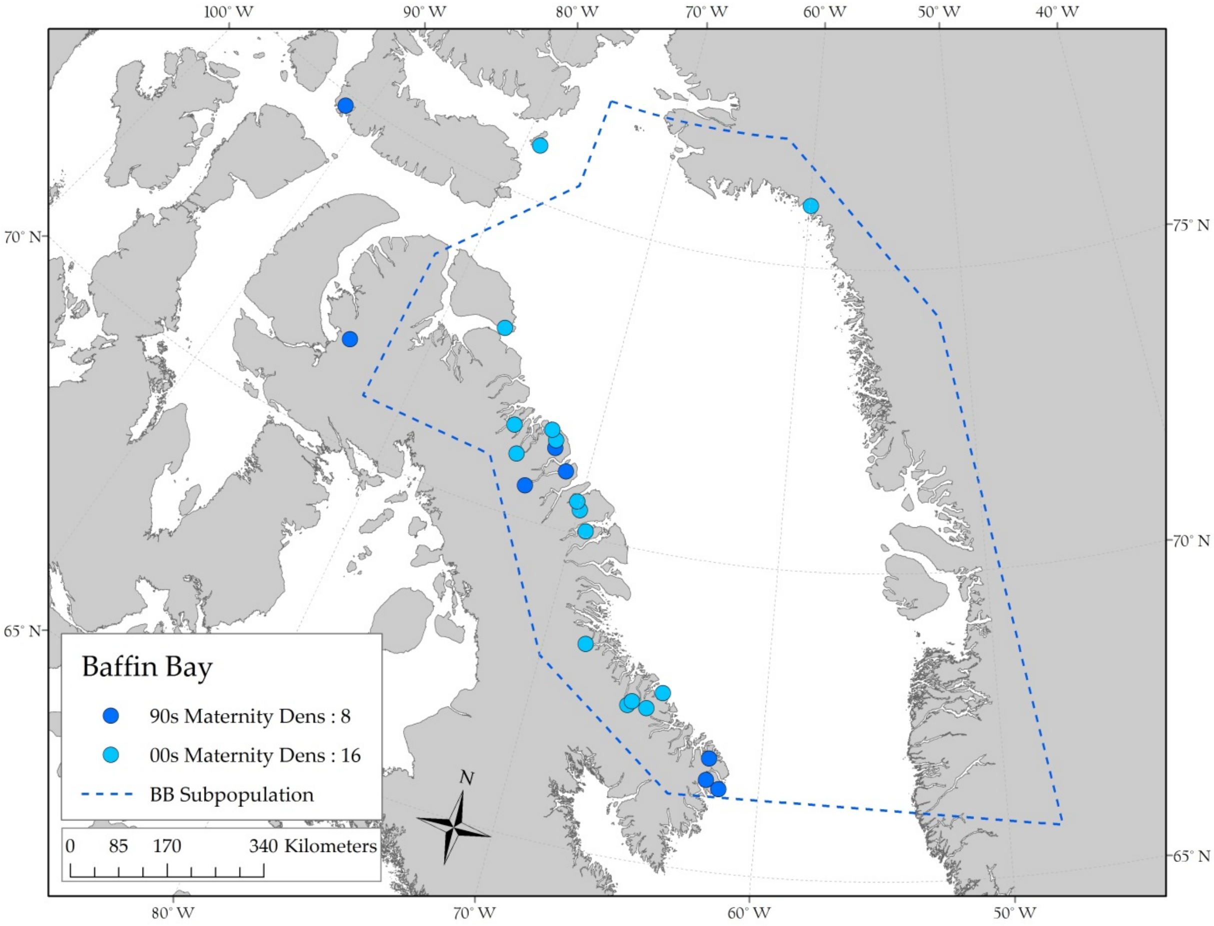


Figure 4.20. Shelter den locations in BB by decade.

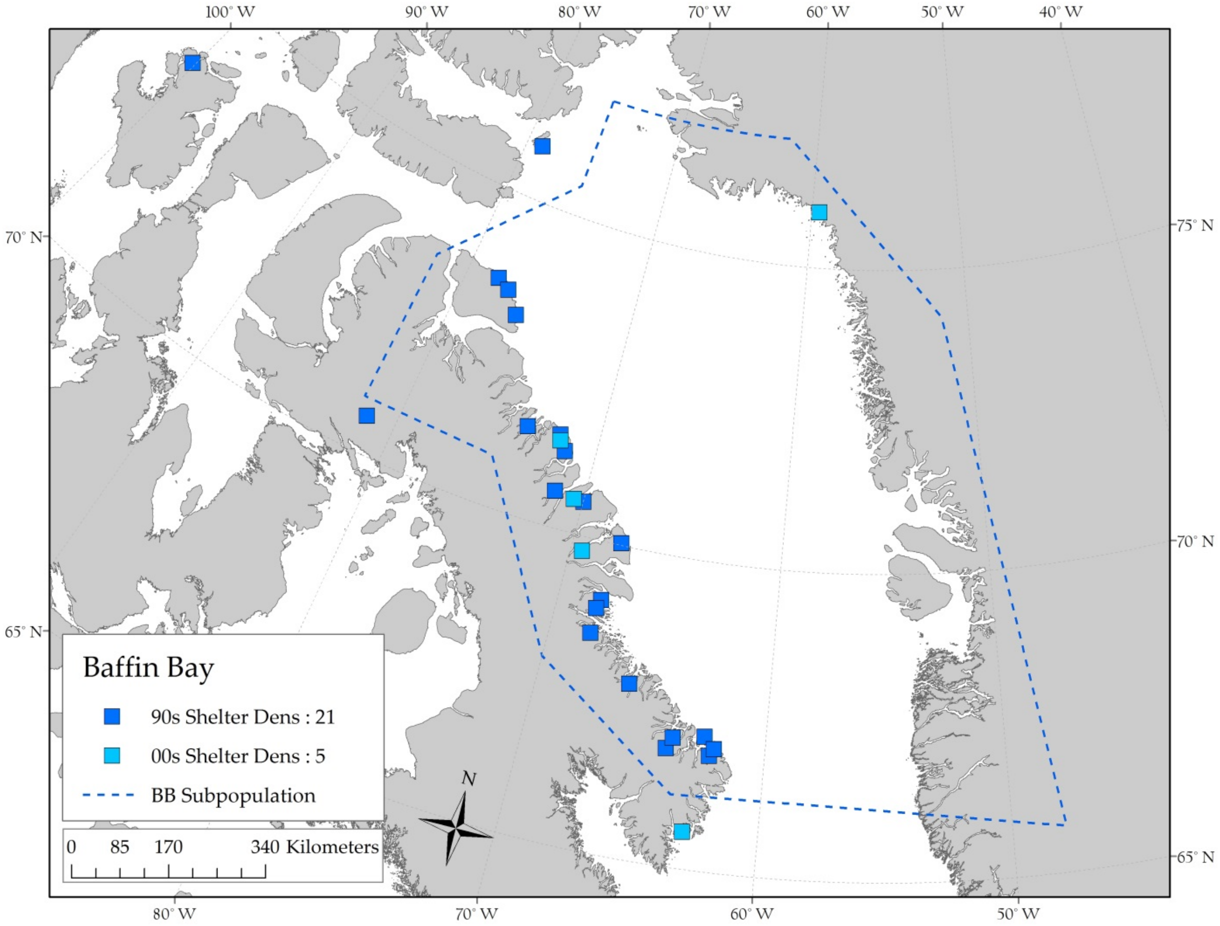


Figure 4.21. Boxplots comparing maternity den duration of Baffin Bay maternity dens (*p* = 0.017) (1990s: *n* = 8; 2000s: *n* = 16).

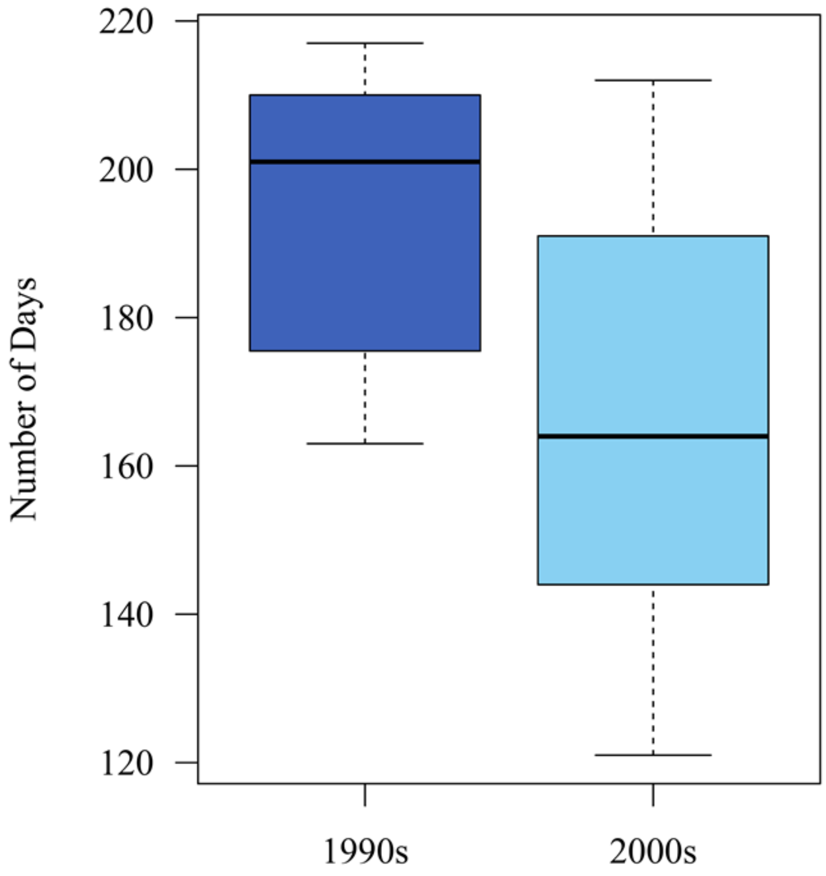


Figure 4.22. Boxplots comparing entry (p = 0.018) and exit dates (p = 0.399) of Baffin Bay maternity dens (1990s: n = 8; 2000s: n = 16)

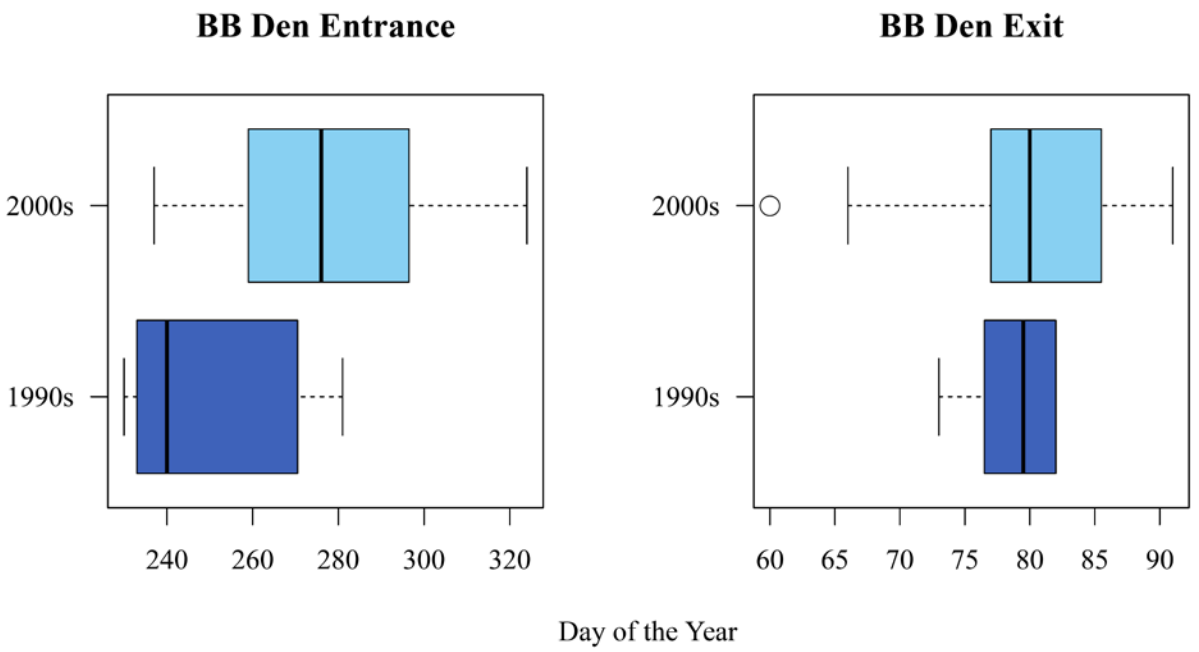


Figure 4.23. Boxplots comparing the first date on land (FDOL) of pregnant females in BB in the 1990s (*n* = 8) and 2000s (*n* = 16) (First FDOL used; *p* = 0.002).

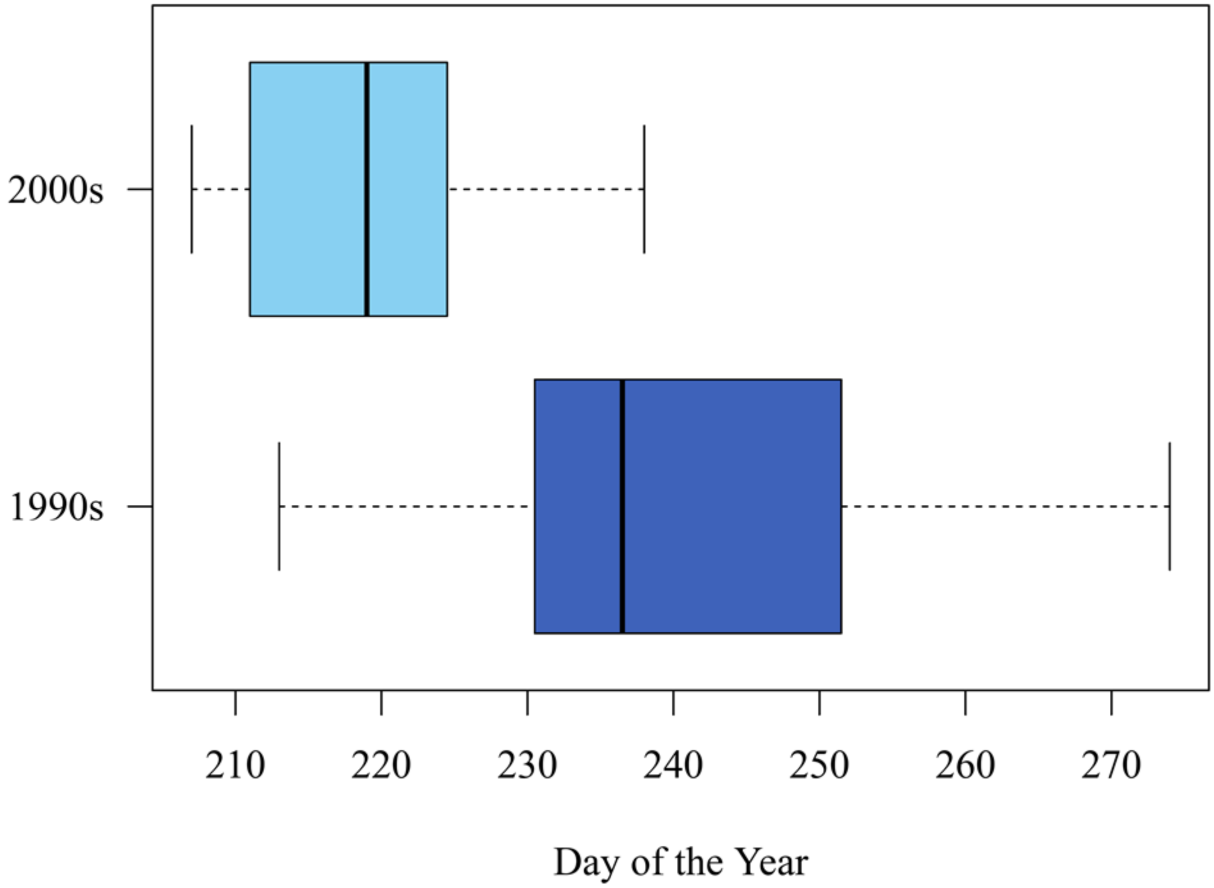


Figure 4.24. Plots comparing the aspect, slope, elevation, and distance to coast of the 1990s (*n* = 8) and 2000s (*n* = 15) maternity dens in Baffin Bay (the den on Greenland was omitted). The aspect plot consists of a compass face with lines marking the directions that dens faced. The lines are annotated with numbers noting how many dens were found at that aspect. Elevation and slope significantly differed between the two time periods (*p* = 0.003), whereas no significant difference was detected for aspect (*p* = 0.392) or distance to coast (*p* = 0.776).

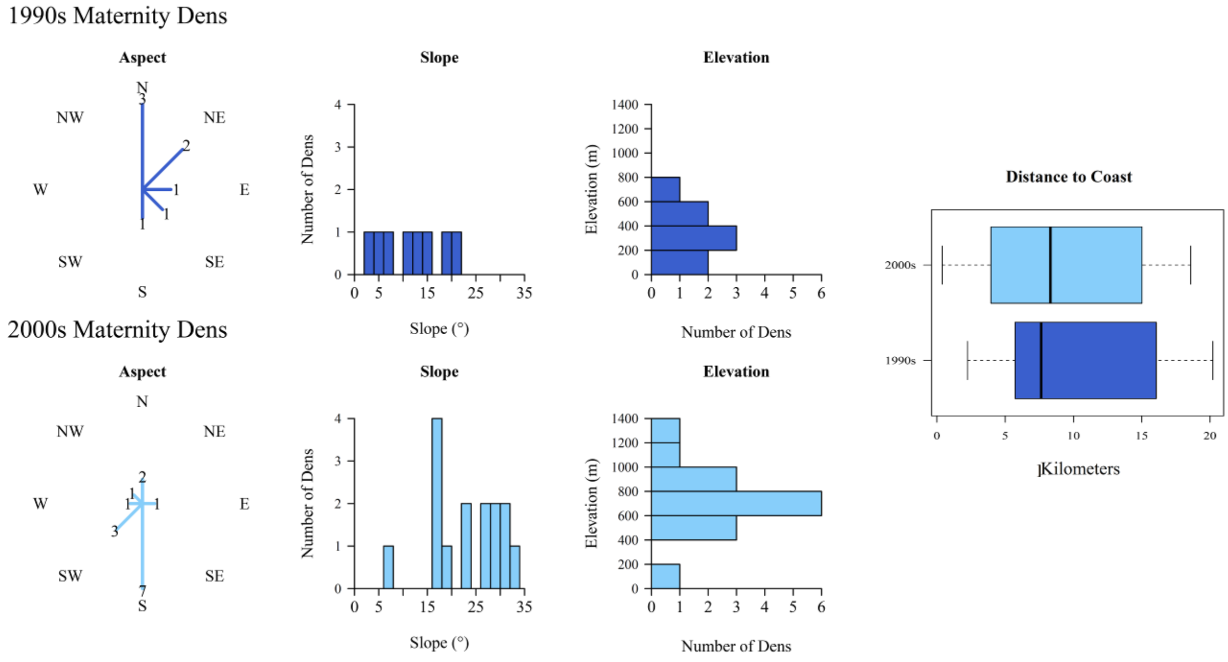
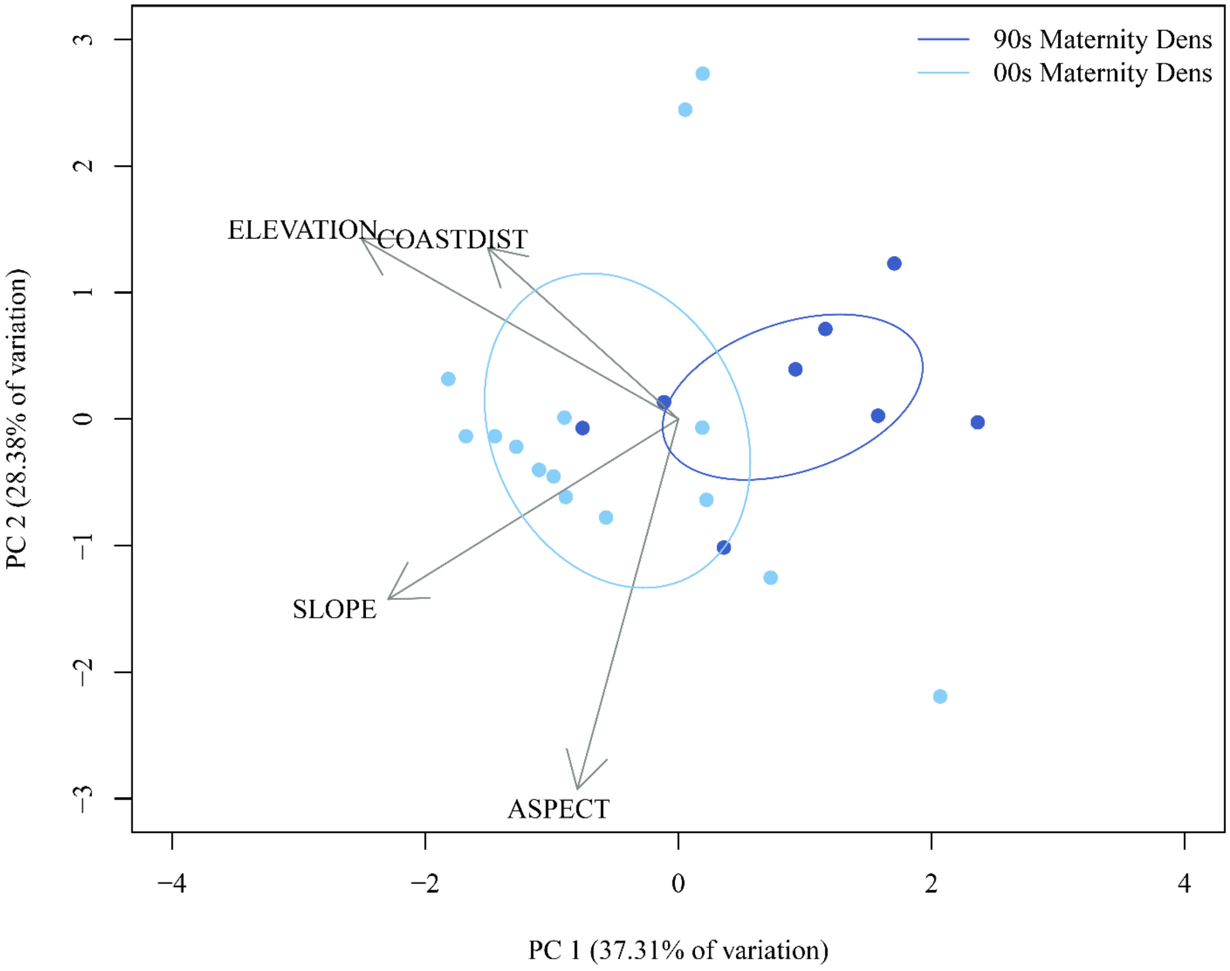


Figure 4.25. Biplot symbolizing the results for the principal component analysis (PCA) ordination of the Baffin Bay maternity dens and their habitat descriptors (elevation, slope, aspect, and distance to coast or 'coastdist'), with ordiellipses drawn around year groups (1990s and 2000s; confidence level = 0.95). The 1990s dens (*n* = 8) are symbolized by dark blue points and the light blue points are the 2000s dens (*n* = 15; the Greenland maternity den was omitted).



CHAPTER 5

GENETIC MARK-RECAPTURE STUDY OF POLAR BEARS IN BAFFIN BAY

KEY FINDINGS

- We used joint live-recapture and dead-recovery mark-recapture models to analyze data for the Baffin Bay (BB) polar bear subpopulation, with the goal of updating estimates of subpopulation size and survival. The dataset consisted of 914 physical captures 1993-1995 and 1997; 1,410 genetic samples obtained from biopsy darting 2011-2013; and 243 harvest returns of research-marked bears 1993-2013.

- The mean estimate of total abundance of the BB subpopulation in 2012-2013 was 2,826 (95% CI = 2,059-3,593) polar bears. The mean estimate of total abundance 1994-1997 was 2,173 (95% CI = 1,252-3,093) bears, similar to the estimate reported by Taylor et al. (2005). Estimates of abundance for the 1990s and 2010s are not directly comparable due to changes in sampling design and environmental conditions.

- The mean estimates of total (i.e., including harvest mortality) survival in 2011-2013 were 0.90 (SE = 0.05) for females age ≥ 2 years, and 0.78 (SE = 0.06) for males age 2 ≥ years. The time-constant estimate of total survival for a combined age class of cubs-of-the-year and yearlings, over the entire period 1993-2013, was 0.87 (SE = 0.06). Estimates of unharvested survival in 2011-2013 for females and males age ≥ 2 years were 0.91 (SE = 0.05) and 0.83 (SE = 0.06), respectively. Estimates of survival for both sexes may have included negative bias due to temporary emigration (see Chapter 3).

- We performed a comparative assessment of sampling design and environmental conditions in the 1990s and 2010s to help interpret parameter estimates, quantify potential bias, and understand trends. An evaluation of the spatial distribution of onshore captures, together with data on habitat use from satellite telemetry, suggested that more systematic live-recapture sampling, including inland areas and the backs of fjords, occurred during 2011 – 2013 compared to the 1990s. Furthermore, offshore sea ice was available to polar bears during the annual sampling periods in the 1990s, but largely unavailable in the 2010s.

- We created a geographic subset of the 2010s data based on the estimated sampling area from the 1990s to investigate the effects of sampling differences. Analyses suggested that geographically-restricted sampling such as occurred during the 1990s could result in approximately 10% negative bias in estimates of abundance. Furthermore, satellite telemetry data suggested that a potentially significant proportion of the BB subpopulation may have been located outside the sampling area or on the sea ice during mark-recapture sampling in the 1990s, although no sampling was conducted on the ice. This represents another potential source of negative bias in the 1990s abundance estimate although we were not able to quantify its magnitude.

- Differences in sampling design and environmental conditions between the 1990s and 2010s introduced variable levels of heterogeneity into recapture probabilities. This heterogeneity was due in large part to the temporary emigration of bears from the sampling area, which was more pronounced in the 1990s (i.e., more bears were located farther inland, where there was no sampling, or on the sea ice). Although moderate levels of random temporary emigration are not problematic for mark-recapture models, high or variable levels of temporary emigration combined with short live-encounter sampling windows, or non-random temporary emigration, are well-known sources of bias. Our approach of including harvest returns in the same analytical framework as live-capture data likely mitigated bias to some extent. However, the BB data were too sparse to fit MR models that explicitly estimated temporary emigration and thus minimized its effects on parameters.

- Considering statistical uncertainty in estimated parameters and evidence that the sampling design and environmental conditions likely resulted in an underestimate of abundance in the 1990s, it is not possible to conclude that the estimate of total abundance in the 2010s represents an increase in the size of the BB subpopulation. Although the 2010s abundance estimate represents the best-available information and is suitable for informing management, we cannot reliably determine the trend in subpopulation size over the 1993-2013 study period.

- The 2011 – 2013 estimate of total survival for independent females is likely too low to support a stable subpopulation, although subsequent demographic modeling (e.g., population viability analysis integrating survival, recruitment, and harvest) is necessary to estimate observed and potential (i.e., in the absence of human-caused removals) subpopulation growth rates. The low estimates of total survival for independent males may warrant concern and further investigation. However, the short time-series of live-recapture data in the 1990s and 2010s, statistical uncertainty, and potential negative bias due to temporary emigration (such bias is generally most pronounced toward the end of a study) limit inference about trends in survival or the current status of the BB subpopulation based on estimated survival rates.

5.1. Introduction

Sea-ice dynamics are rapidly changing across the circumpolar Arctic (Comiso et al. 2008, Stroeve et al. 2012), including the Baffin Bay region (Laidre et al. 2015), with a general trend toward reduced spatial extent and temporal availability of sea ice. These changes are expected to have negative impacts on sea ice dependent polar bears in the long-term (Atwood et al. 2015). In the near-term, the effects of sea-ice loss are expected to vary among subpopulations, with some of the earliest impacts anticipated in the seasonal ice subpopulations (Amstrup et al. 2008, Stirling and Derocher 2012, Rode et al. 2012, 2014, Bromaghin et al. 2015, Obbard et al. 2015, Lunn et al. 2016).

The Baffin Bay (BB) polar bear subpopulation, located between Canada and Greenland, forms part of the seasonal ice ecoregion (Amstrup et al. 2008) and is characterized by the formation and near-complete melting of sea ice each year. These sea-ice dynamics force most bears to spend the low-ice (or ice-free) summer on land. In this region, the interval between the spring sea-ice retreat and the fall sea-ice advance has increased by ~12 days per decade since 1979 (Chapter 4, Laidre et al. 2015), suggesting that polar bears are likely experiencing reduced sea-ice availability during important spring and fall foraging periods. Rode et al. (2012) reported declining body condition in BB and suggested that this finding may be due to loss of sea-ice habitat.

Abundance of the BB subpopulation was estimated as 2,074 (95% CI= 1544 - 2604) bears based on a physical mark-recapture study conducted 1993 – 1997 (Taylor et al. 2005). Taylor et al. (2005) reported relatively high survival for subadult and adult females (ages 1 – 4: 0.90, SE= 0.045; ages 5 – 20: 0.94, SE= 0.021) and estimated an unharvested population growth rate (λ) of 1.055 (SD: 0.01), suggesting strong potential for subpopulation growth relative to other demographic studies of polar bears (*cf.* Taylor et al. 2006, 2008, 2009). The combined Canada-Greenland harvest from BB was high during the early to mid-2000s ($\bar{x}$: 214 from 2001 – 2005, Chapter 8), and Peacock et al. (2012) reported lower survival rates in BB for the 2003 – 2009 period using harvest recoveries, although the estimated survival rates were characterized by high uncertainty and potential bias due to sparse data.

No new research has been conducted to update estimates of abundance since the 1993 – 1997 research, but projections using estimated abundance and vital rates from the 1990s (Taylor et al. 2005) and reported harvest suggested a declining subpopulation and a projected abundance of ~1,550 bears as of 2004 (PBSG 2010). In addition, the IUCN / Polar Bear Specialist Group (2015) assessed BB to be declining based on 100% of population viability analysis (PVA) simulations resulting in a decline in abundance after 10 years, largely attributable to the effects of harvest. In the absence of an updated demographic and ecological assessment, there has been considerable uncertainty about the current abundance and status of polar bears in BB. Given the large-scale environmental changes occurring in Baffin Bay and concerns regarding previous and current harvest levels, there was a need for new information on subpopulation status (Chapter 1).

Our objective was to obtain updated estimates of abundance and, to the extent possible, vital rates including survival for polar bears in the BB subpopulation. These estimates, combined with information on sea-ice dynamics, polar bear movements, reproductive output, body condition, and other ecological metrics, will be used to assess subpopulation status, develop management plans, and inform subsistence harvest levels. To address our objectives, we conducted a 3-year genetic mark-recapture study during 2011 – 2013. These data were analyzed together with data from physical mark-recapture research (1993 – 1995, 1997). Harvest recoveries were incorporated throughout the 21-year (1993 – 2013) study period, but no live recapture sampling occurred during the 13-year interval from 1998 – 2010.

To address concerns regarding the potential impacts of immobilization and handling on wildlife and better reflect values of northern Canadian communities, the Canada-Greenland Joint Commission on Polar Bears elected to use genetic, rather than physical, mark-recapture methods (Chapter 1). This work is part of a broad, inter-jurisdictional initiative to develop less-invasive methods (i.e., compared to physical capture) to study polar bear subpopulations. Whereas use of aerial surveys has become increasingly widespread for polar bears (e.g., Aars et al. 2009, Stapleton et al. 2014, 2016, Obbard et al. 2015), genetic mark-recapture has not been implemented at large spatial scales. However, genetic mark-recapture is an established technique that has been used in wildlife studies for decades (e.g., Palsbøll et al. 1997, Boersen et al. 2003, Boulanger et al. 2004), including small-scale studies of polar bears (Herreman and Peacock 2013). This study and concurrent research in the neighboring Kane Basin subpopulation (Chapter 10) represent the first subpopulation-scale applications of genetic mark-recapture for assessment of a polar bear subpopulation.

5.2. Materials and Methods

Study Area

The boundaries of the Baffin Bay polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all Bylot Island (66.2° N to 73.8° N) in Nunavut/ Canada and parts of West and Northwest Greenland (66.0° N to 77.0° N; Taylor et al. 2005; Figure 5.1). BB is bounded by Greenland to the east, Baffin and Bylot islands to the west, the North Water polynya to the north and Davis Strait to the south. Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N.

Baffin Bay is ice covered in winter but typically ice free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay. The last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot islands (Taylor et al. 2005), although an unquantified but probably small number remains on land in northwestern Greenland throughout the ice-retreat period (see Chapter 4). Sea ice in Baffin Bay has decreased markedly during the last few decades (Stirling and Parkinson 2006, Peacock et al. 2012, Laidre et al. 2015), with earlier spring break up and later fall formation. During the 1990s, some sea ice remained in offshore Baffin Bay during summer and was used by bears; this remnant ice was largely unavailable to polar bears in the 2010s (Chapters 3 and 4). The east coasts of Baffin and Bylot islands are characterized by high topographic relief and deep, steep-sided fjords, creating difficult conditions in which to sample bears.

Movement data of satellite-collared bears (Taylor et al. 2001), genetic analyses (e.g., Paetkau et al. 1999, Peacock et al. 2015), and recaptures and harvest recoveries of research-marked bears (Taylor and Lee 1995) have been used to evaluate subpopulation delineation between BB and other subpopulations in the Canadian and Greenlandic Arctic. Taylor et al. (2001) reported some sub-structuring of BB on a north-south gradient, and genetic analyses suggest a lack of genetic differentiation between BB from the adjacent Kane Basin subpopulation to the north, but a significant genetic difference between BB and the Davis Strait subpopulation (Paetkau et al. 1999, Peacock et al. 2015). These findings were corroborated by analyses of recent samples from BB, KB, DS and Lancaster Sound (see Chapter 2). Although some interchange occurs among BB and adjacent subpopulations including Davis Strait, Lancaster Sound and Kane Basin (e.g., Taylor et al. 2001, Chapter 3), the BB subpopulation is considered a distinct demographic unit for management purposes.

Mark-Recapture Sampling Design

Mark-recapture sampling of polar bears in BB has occurred over three periods. Early field sampling was conducted during the 1970s and 1980s (Figure 5.2) but restricted to spring-time captures on landfast ice (i.e., ice occurring nearshore; Taylor et al. 2005). Because this early sampling occurred in a different season (i.e., spring versus fall) and was spatially more variable and restricted than later sampling, we excluded early data (1970s and 1980s) from the present analyses. In addition, lack of tissue samples from early sampling precluded genetic identification, a primary method used in the current study. More systematic capture-based sampling occurred during fall ice-free seasons in 1993 –1995 and 1997, but there was no fall sampling in 1996 due to logistical and resource constraints (Taylor et al. 2005).

We completed a recent fall-time sampling session (August – October) from 2011 to 2013. This session differed from sampling in the 1990s in several important ways. First, sampling was conducted by biopsy darting to obtain tissue for genotyping individuals, rather than via physical capture and tagging. Second, new information obtained via satellite telemetry on the movements and spatial distribution of bears in Baffin Bay and Kane Basin, collected during the 1990s sampling session and during 2009 – 2010, was used to improve study design, with the objective of improving sampling coverage of the BB subpopulation and thereby reducing heterogeneity in recapture probabilities. In Baffin Bay, heterogeneity during fall sampling is likely to result primarily from the spatial distribution of bears in relation to sampling effort. Although bears are concentrated along the Baffin Island coast during the ice-free season, some individuals travel significant distances inland, move to higher elevations, or remain on offshore ice where access for sampling is difficult (Ferguson et al. 1997, 2000, Taylor et al. 2001, Chapter 4). Bears also may segregate by age and reproductive status. For example, adult females with cubs tend to select fjords, avoiding offshore islands and coastal regions where densities of adult males are higher (Ferguson et al. 1997; Chapter 4), and pregnant bears select inland and upland denning habitats where they are less available for capture (Chapter 4).

Although Taylor et al. (2005:205) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island, examination of the satellite telemetry data from adult female polar bears collared in the 1990s indicated an under-representation of bears in fjords and inland regions and offshore pack ice (see Chapter 3). This finding suggested that capture effort during the 1990s was concentrated on islands, along the coastline, and near the mouths of fjords (Figure 5.3). This pattern was particularly noticeable in central and northern Baffin Island, where no captures were recorded beyond the mouths of fjords during the entire study period. We conclude that sampling was spatially restricted to a portion of the subpopulation's fall range during the 1990s, thus potentially increasing heterogeneity in recapture probabilities, which can bias estimates of subpopulation size and demographic parameters.

Analysis of the 1990s telemetry data also showed that a potentially significant proportion of collared bears remained on offshore sea ice during the fall onshore sampling period (see Chapter 3). The proportion of collared bears present in the sampled area each year was estimated based on the total number of collars that were transmitting during the capture sampling period. To evaluate movements and fidelity with respect to the onshore sampling area in years $t + 1, 2, \ldots k$, we only used data from bears that were captured in the onshore sampling area and fitted with collars during year t (i.e., we excluded data from the year of capture, because bears were captured onshore and their locations following capture were not random). In addition to the mark-recapture sampling in the fall, some bears were captured and fitted with collars in the spring. Data from bears captured in spring of year t were incorporated into summaries of movement and fidelity for fall of year t.

During the 1990s, there were three years with telemetry data available during mark-recapture sampling (1993 – 1995). In those years, 0 – 23% of collared bears transmitting during the sampling period (n = 1 – 13 collared bears / year) were present in the sampling area. By contrast, during the 2010s, 67 – 85% of collared bears were present in the sampling area during the sampling period (n = 6 – 12 collared bears / year) (see Chapter 3). Although sample sizes are small and telemetry data are limited to adult females, this finding suggests that seasonal fidelity to the sampling area changed significantly between the two mark-recapture sampling periods. This follows well with the change observed in the sea-ice habitat during those periods (Chapter 4). Further investigation showed that a substantial portion of 1990s adult females outfitted with satellite collars were on the offshore pack ice of Baffin Island or on remnant ice around Lancaster Sound and Devon Island. In contrast, there was little sea-ice habitat available in summer during the 2010s, and bears were concentrated on Baffin Island and Northwest Greenland. Polar bears that used offshore sea ice during the 1990s were unavailable for capture, and to the extent that individual bears consistently used offshore sea ice throughout the 1990s sampling period, these bears would not have been enumerated in the subpopulation estimate. These issues were less problematic during the 2010s due to the expanded onshore sampling area and the lower availability of offshore sea ice.

For sampling on Baffin and Bylot islands during 2011 – 2013, we defined sampling strata to guide effort and improve survey coverage and efficiency. Stratification primarily was based on satellite telemetry data obtained from adult female polar bears collared during fall and spring along eastern Baffin Island (1993-1997) and in spring in W and NW Greenland in 2009 and 2010. We summarized location data by proximity to the coastline and used the proportion of locations in different inland zones (e.g., 0 – 5 km inland, 5 – 10 km inland) to inform stratification. We delineated a high-density stratum including the coastline and offshore islands, extending 5 km inland; a moderate-density stratum including inland regions 5 – 10 km from the coastline; and a low-density stratum extending up to 30 km inland (Figure 5.4). We attempted to allocate roughly 65%, 25%, and 10% of helicopter search effort in the high-, moderate-, and low-density strata, respectively, to efficiently sample the study area. We set *a priori* guidelines to systematically distribute inland search effort along the entirety of the islands.

It was not feasible to sample bears that may have remained on offshore ice floes in either decade. However, long-term trends in sea-ice conditions in Baffin Bay have resulted in significant reductions in offshore ice during the fall in the 2010s, relative to the 1990s (Laidre et al. 2015; see Chapter 4). Thus, the presence of bears on offshore ice during the recent sampling session was considerably reduced (*cf.* Chapter 3 Figures 3.18, 3.19, and 3.20). Additionally, using real-time data on sea ice (see Field Methods below) and the location of telemetry-instrumented bears, sampling during the 2011 – 2013 period was timed to coincide with the period when sea-ice cover was at a minimum and most collared bears were on land.

Various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that an unquantified but presumably small number of bears in the BB subpopulation spend the summer in the Qimmusseriarsuaq / Melville Bay area of Northwest Greenland, rather than moving with the retreating sea ice and summering on Baffin and Bylot islands (see also Chapter 2). To account for this portion of the subpopulation, we extended our sampling efforts to include this region during fall (Figure 5.4). Because satellite telemetry indicated that polar bears were not present during fall in the Melville Bay region in the 1990s (Taylor et al. 2001), NW Greenland was not sampled during fall during the 1990 physical mark-recapture study

(Taylor et al. 2005). In sum, study design for 2011 – 2013 was intended to (1) maximize sample size; (2) sample bears across the known seasonal range of the subpopulation; (3) efficiently allocate sampling effort based on expected densities across the study area; and (4) accommodate the spatial segregation of sex, age, and reproductive classes. These considerations are important to reducing potential bias in estimates of demographic parameters, particularly abundance, from mark-recapture studies (Pollock et al. 1990, Williams et al. 2002).

Field Methods

Capture, sampling, aging, and data collection protocols for bears marked during 1993 – 1995 and 1997 are described in Taylor et al. (2005). For genetic mark-recapture sampling from 2011 – 2013, field work was timed to coincide with minimum sea-ice cover in Baffin Bay based on Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery (https://worldview.earthdata.nasa.gov) and Canadian Ice Service maps (August – October in all years). The locations of collared bears were also reviewed prior to field work to confirm the timing and location of bears coming ashore. During field sampling, locations of collared bears were not reviewed and telemetry equipment was not used to help locate bears.

In Nunavut (Canada), sampling in 2011 – 2013 was conducted via remote biopsy darting (Pagano et al. 2014) using two helicopters (Bell 206 LongRangers). The helicopters began sampling at opposite ends of the study area; one proceeding north to south, and the other south to north until they overlapped. We sampled Baffin and Bylot islands from September 4 – October 14, 2011; August 26 – September 29, 2012; and August 20 – October 11, 2013. With the exception of 1993 (August 23 – October 8), sampling started and ended earlier than research in the 1990s. Approximately 300 hours of total helicopter flying time was allocated each year, including travel time to and from daily start points and refueling caches. To the extent possible, effort was allocated according to the *a priori* stratification and criteria noted previously. However, we modified sampling where necessary based on terrain and weather conditions that limited access to some areas. Flight paths during searches were recorded via GPS to facilitate *post-hoc* assessment of the distribution of search effort. We made a concerted effort to search inland and at high elevations. We searched most fjords along their entire lengths and a majority (>95%) of offshore islands.

As outlined above, a small number of polar bears summer in the Melville Bay area of NW Greenland (e.g., Born et al. 2011), and we also searched these areas to collect biopsies. The areas between 74° 34´ N and 76° 46´ N (i.e., Melville Bay *sensu lato*) were searched during 4-11 September 2012 and 7-17 September 2013 (a total of nearly 60 hours of active on-effort search) using an AS350 Ecureuil B3 helicopter. Coastlines, mountain sides, inland nunataks, glacier fronts and most offshore islands up to 40 km from the coast were searched. On each flight, three dedicated observers and the pilot searched for bears; and on several flights, a local polar bear hunter assisted in the search. Flight paths were recorded using GPS.

In 2011, we initially tested two types of biopsy dart to assess reliability and the quality of samples yielded (Figure 5.5). The Pneudart DNA dart (Pneudart Inc., Williamsport, USA) was highly reliable at yielding good quality tissue samples (95% success). In contrast, the Pneudart Biopsy dart was unreliable (<50% success), although it did provide good quality samples when successfully deployed. Consequently, we used DNA darts exclusively during the remainder of the field work. Biopsy darting was quick and minimally invasive; the time between spotting a bear and obtaining a sample was typically < 2 minutes. Since bears were not chemically immobilized, they could be safely darted in locations that would have been unsuitable for capture due to the risks of drowning or falling. The only exception was for bears sighted in the water. Most bears found in water were directed to land before darting since the darts do not float well. A small number of bears were darted while still in the water using Pneudart DNA Marker darts, which float for a short time in calm waters and may be retrieved.

Bears showed little or no reaction to the impact of a biopsy dart, and no visible mark was left in most cases. Immediately after darting, each bear was allowed to move away from the helicopter before the dart was retrieved. Darts were coated in fluorescent paint to aid retrieval. When working in deep snow, we also rolled a length (~ 20 cm) of flagging tape around the shaft of each dart. This tape unrolled during flight and helped in locating darts when they sank into the snow. Because biopsy darts leave no visible mark, there was potential for repeated sampling of the same individuals within a single field season. To minimize duplicate sampling, daily searches were limited to areas not previously searched. Where possible, we used natural barriers to polar bear movements for the purpose of delineating daily break points in search effort. When sampling members of a family group consisting of an adult female with cubs-of-the-year or yearlings, the 1 to 3 dependent young with each mother were distinguished based on differences in size or other features (e.g., marks on fur) to avoid sampling the same individual twice. When it was not possible to distinguish between cubs, we used DNA-Marker darts to distinguish among litter mates. This dart takes a tissue sample and leaves a temporary dye mark that can be used to distinguish sampled from non-sampled individuals.

A limitation of biopsy darting was the challenge of sampling cubs-of-the-year. Although many cubs were large enough to be sampled, doing so involved a risk of separating them from their mother. Unlike physical capture methods, in which the adult female is first immobilized and can be used as an ‘anchor point’ around which cubs are captured, members of a family group that are being biopsy darted may move in opposite directions. With the rugged and steep terrain along Baffin and Bylot islands and Melville Bay, cubs can quickly lose sight of their mother and are at risk of injury or separation. For these reasons, only about half of the cubs-of-the-year that we encountered during 2011 – 2013 were biopsy darted, although we recorded the sighting of all individual cubs for calculating proportions of females with cubs and mean litter sizes.

Following retrieval, darts were checked to ensure they contained a suitable tissue sample. Each sample was divided into two parts for storage and labelled with a unique biopsy number. Samples were initially stored cooled or frozen. Samples sent for DNA extraction were taped onto an absorbent card, placed into individual envelopes and later oven dried for submission, or stored in vials with DMSO. For each bear encountered, we recorded GPS coordinates and data on location, weather conditions, habitat, behavior, body condition (thin, average, and obese bears; see Stirling et al. 2008), group size, and estimated age-class and sex.

We remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) from the air at a range of 3 – 7 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males), body shape and proportions, presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under the tail in females). Field notes assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified for some bears from other encounter records in which the individual was captured and physically examined, or by using genetic identification to assign membership to a known family group (Appendix B). We were able to assess the accuracy of estimating age-class and sex of polar bears using this sample of known age-class bears (Appendix B).

Harvest recoveries

We assembled data from bears killed in the harvest or as problem bears during the 1993 - 2013 study period. Between 1993 and 2010, recoveries of research-marked bears in the harvest were detected by the return of ear tags and / or lip tattoos from hunters in Canada and Greenland. Between 2011 and 2013, recoveries of bears that were physically tagged or genetically marked (i.e., without physical tags) were detected by the return of ear tags or lip tattoos (and satellite radios in Greenland); or from genetic monitoring of harvest (i.e., genotyping of harvested bears). Although we expected a majority of bears marked in Baffin Bay to remain within BB, previous studies of tag recoveries in the harvest and satellite telemetry suggest that some bears emigrate from the subpopulation (Stirling et al. 1980, Taylor and Lee 1995, Taylor et al. 2001, Taylor et al. 2005). To account for marked individuals that were harvested outside BB (Burnham 1993), samples were collected from bears harvested in Baffin Bay as well as all surrounding subpopulations (Lancaster Sound, Kane Basin, Gulf of Boothia and Davis Strait, Foxe Basin; Burnham 1993). For each harvested bear, data including age, sex, date and location of kill were recorded. Canada's quota-based mandatory harvest reporting system was in place throughout the 1993 – 2013 research period. Greenland implemented a quota system and made improvements to the reporting system in the mid-2000s and, in 2012, instituted a mandatory harvest reporting system for collection of a tissue sample and premolar tooth for age determination (Appendix D).

Genetic Analysis

To identify bears physically marked (with ear tags and lip tattoos) during the 1990s that survived and were subsequently encountered during the genetic mark-recapture session (2011-13), we genotyped tissue samples from all bears sampled in the 1990s except: 1) bears that were recorded as harvested between 1993 and 2011, 2) bears whose known or estimated age would have been greater than 35 years in 2011, and 3) 33 bears that met the above 2 criteria, but lacked tissue samples. In total, this dataset consisted of 650 individuals marked in the 1990s that would have been ≤ 35 years old and had not been harvested by the time genetic sampling began in 2011; the 33 bears lacking tissue samples were assigned to unique attribute groups to acknowledge they were unavailable for genetic recapture during the 2011-2013 sampling period (see below).

Dried biopsy samples and harvest specimens (frozen or in ethanol) were sent to Wildlife Genetics International (Nelson, B.C., Canada) for analysis using protocols previously validated for bears (Kendall et al. 2009). DNA was extracted from ~ 3mm^2 pieces of tissue with QIAGEN DNeasy Blood and Tissue Kits (http://www.qiagen.com/). Most samples consisted of a plug of a skin and sub-cutaneous tissue, which provided ample material for DNA extraction and residual tissue for future analyses. In a small proportion of cases, the available sample consisted of a tuft of hair. DNA was extracted from hair samples using approximately 10 guard hair roots or 30 pieces of underfur. In a few cases, where a biopsy sample contained no visible tissue, DNA was successfully extracted by soaking the barbed needle from the biopsy dart in the lysis mix (QIAGEN buffer ATL + proteinase K).

To select markers for the analysis of individual identity, we used allele frequency data from 1,771 polar bears for which complete 20-locus genotypes existed before the genetic mark-recapture study began (Government of Nunavut, unpublished data). We ranked the 20 microsatellite markers in the dataset by expected heterozygosity. The 8 most variable markers that could be analyzed together in a single sequencer lane were selected for use. These surpassed the required standard for marker variability (HE = 0.80; Paetkau 2003). In addition to the 8 microsatellite markers, we analyzed sex on every sample, using a *ZFX*/*ZFY* marker. This 9th marker roughly halved the match probability (assuming a balanced sex ratio), even for close relatives, as well as providing replication of sex data for individuals that were sampled more than once.

The analysis of individual identity followed a 3-phase approach. Phase 1 was a first pass of all extracted samples using the 9 selected markers (*G10B*, *CXX20*, *G10H*, *G10P*, *145P07*, *MU50*, *MU59*, *G10X* and *ZFX*/*ZFY*). Samples that failed at > 6 of 9 markers on the first pass were set aside and did not proceed further in the analyses. Previous experience has shown that such samples are prone to errors and run out of DNA before generating a complete (phase 2) and reproducible (phase 3) genotype (D. Paetkau, pers. comm.).

The first pass was followed by a cleanup phase in which data points that were weak or difficult to read the first time were re-analyzed. During cleanup we used 5 µL of DNA per reaction instead of the 3 µL was used during first pass. At the conclusion of the cleanup phase, the remaining samples (99.5%) had high-confidence scores for all 9 markers. In cases where the genetic sex result contradicted the reported sex based on field assessment, genetic sex was checked using a second independent marker (*amelogenin*; http://www.ncbi.nlm.nih.gov/pubmed/7695123), thus confirming the results, and ruling out the possibility that a mutation at a particular marker was to blame. In all cases, results from the second marker confirmed that the field data was the source of error.

The third and final phase of analysis was error-checking, following the published protocol of reanalyzing the mismatching markers in highly similar pairs of genotypes (Paetkau 2003). This error-check included genotypes from the 4,657 polar bears in the database, plus published data from 473 individuals (Paetkau et al. 1999). The error-checking protocol functions on the principle that when ≥ 2 samples are genotyped from a given individual, and when 1 of those genotypes contains an error, the result is a pair of genotypes that match at all-but-1 marker (a '1MMpair'). Less commonly, 2MM-pairs are created when 2 errors have been made in the genotypes of the samples from a given individual.

An important distinction with this protocol is that it is designed to ensure accurate individual ID, and has been proven to do so with a high degree of efficiency (Kendall et al. 2009), but it is not intended or expected to correct errors when just 1 sample has been genotyped from a given individual. In addition to re-analyzing mismatching markers this protocol also involved the inclusion of additional markers for some samples. Finally, we searched the dataset for genotype matches that seemed unlikely based on our field data. In each case, 3 extra markers were added to the genotypes to lower the probability of chance matches between individuals. The extra loci confirmed all of these matches. Once the genotyping and error-checking was complete, we defined an individual for each unique 9-locus genotype.

In total, 1610 biopsy samples (99.2% of those submitted for analysis) were successfully extracted and genotyped. A further 868 samples (99.3% of those submitted) from bears harvested in BB and surrounding subpopulations 1993-2013 were genotyped successfully, with success defined as satisfying the lab's visual and peak-height criteria for high-confidence scoring at each of the 9 markers.

In addition to the genotyping errors that were targeted during error-checking, DNA-based datasets are prone to a second source of error, when match probabilities are so high that some individuals have identical genotypes. Calculated match probabilities provide no practical insight into the risk of sampling individuals with matching genotypes, because the calculations are so dependent on the assumptions made about the degree of relatedness among the sampled individuals. We therefore used the direct, empirical approach of extrapolation from the observed mismatch curve (Figure 5.6). We expect to see roughly order-of-magnitude decreases in the number of pairs of individuals whose genotypes match at increasing numbers of markers (Paetkau 2003). In our dataset the slope of this curve was reasonably true to that rule of thumb. From this curve, it is estimated that we would have sampled ~ 0.3 0MM-pairs (individuals whose genotypes matched at 9 markers) in this multiyear dataset of 4,657 individuals; a very small risk of error in proportion to the size of the dataset. In addition to reducing the risk of sampling individuals with the same genotype, another benefit to having such a powerful marker system was realized during error-checking, where the amount of time required to reanalyze the mismatching markers underlying 1MM- and 2MM pairs was trivial in proportion to the scale of the project, because there were so few such pairs.

Statistical Analysis

We used the Burnham (1993) model in Program MARK (Cooch and White 2015) to analyze joint live-recapture and dead-recovery data from the 21-year (1993 – 2013) study period in BB. The Burnham model is an extension of the Cormack-Jolly-Seber live encounter model that facilitates the inclusion of dead-recovery data (i.e., combining the CJS model with the Brownie-Seber dead-recovery model) and estimates survival probability (S; the probability of surviving interval t to t+1), recapture probability (p; the probability of live-recapturing a marked animal), dead reporting probability (r; the probability that a bear is killed by humans and reported to authorities), and fidelity (F; the probability that a bear does not permanently emigrate from the sampling area, and is therefore available for capture in future years). The inclusion of dead recoveries allows for estimation of true survival (i.e., a biological survival rate that does not include permanent emigration), because whereas live-encounter models only measure the probability of remaining alive and within the live-recapture area, the inclusion of dead recoveries from throughout Greenland and Canada in the Burnham model allow for estimation of true survival independent of potential emigration. More importantly, inclusion of dead recovery data increased the amount of information available on the fates of individual bears in the BB subpopulation, likely decreased susceptibility to bias because the sampling mechanism for dead recoveries was different from live-recapture sampling, and allowed us to estimate survival during intervening years between live-recapture periods (i.e., 1998-2010; Peacock et al. 2012). Similarly, with respect to dead-recovery models, live recaptures provide large amounts of additional data, allowing for more precise estimation of survival than would be possible using dead recoveries only (Cooch and White 2015).

Estimates of demographic parameters from mark-recapture studies may be impacted by temporary emigration (e.g., Peñaloza et al. 2014). Completely random temporary emigration is generally reflected in the parameter p for long-term studies (i.e., the probability of recapture reflects both the probability that an individual is in the sampling area on a given occasion, and the probability that the individual will be recaptured conditional on being in the sampling area; Burnham 1993, Barker and White 2001). However, for shorter studies with high or variable rates of temporary emigration, especially when the probability of temporary emigrants remaining outside the study area is non-random (e.g., if the probability of being an emigrant at sampling occasion *t*, depends on emigration status at sampling occasion *t*-1) the parameter *S* may be susceptible to bias as well. Bias typically increases toward the end of the study, and is referred to as terminal bias, because bears that leave the study area during the final years have no opportunity to return and be resampled, and thus cannot be distinguished from individuals that died or emigrated permanently (Peñaloza et al. 2014). Such terminal bias can suggest spurious correlations with environmental covariates, particularly for short studies, if habitat quality declines toward the end of the study and there is concurrent terminal bias (Devineau et al. 2006). With the Burnham model, the inclusion of dead recovery data can mitigate these issues to some extent (Peacock et al. 2012), as can formal incorporation of telemetry data (Peñaloza et al. 2014). More complex mark-recapture approaches are available that can theoretically model temporary emigration with respect to the sampling area, further mitigating potential bias (e.g., the Barker model, Barker and White 2001; multistate models with unobservable states, Schaub et al. 2004). However, these models require large datasets and can be difficult to fit in practice (Converse et al. 2009). We conducted simulations in Program MARK (White and Burnham 1999) and determined that existing mark-recapture data from BB were inadequate for measuring temporary emigration (i.e., the *F* and *F'* parameters in the Barker model, or the *a"* and *a'* parameters in the Barker robust design). Simulations suggested that, using the Burnham model, adult survival (*S*) and recapture (*p*) probabilities were relatively unbiased in the presence of low to moderate levels of random temporary emigration, under which conditions the estimates of *p* reflected the product of recapture probability and presence on the study area (T. Arnold, unpubl. data). Simulations suggested that if temporary emigration was non-random or temporally variable, survival rates would be negatively biased (especially if dead recovery rates were low, as for adult females);

however, the product of capture probability and presence on the study area exhibited relatively small bias, the directionality of which tended to underestimate abundance.

Whereas live-capture sessions are assumed to be instantaneous, dead recoveries may occur year-round between the live-capture sampling periods. For the BB data, there was some temporal overlap of live recapture and dead recovery periods, but the assumption of non-overlap between live and dead recovery periods was generally met. We considered harvests prior to August 31 in year *t* as occurring after the live recapture sampling period in year *t – 1*, whereas harvests after September 1 were assumed to have occurred after live-recapture sampling in year *t*. This coding protocol resulted in no instances of bears being coded as harvested before being observed alive during the sampling period in year *t*.

We analyzed data and built models in Program MARK (White and Burnham 1999). We set up the analysis using a Barker modeling framework to provide flexibility if the data proved sufficient to model temporary emigration, but initially constrained parameters of the Barker model to correspond to the simpler Burnham model (i.e., we fixed the following parameters to 0: $F'(i)$ [the probability a bear not at risk of capture in *i* is at risk of capture in period *i* + 1], $R(i)$ [the probability that a bear surviving from occasion i to *i* + 1 is resighted alive between i and *i* + 1], and $R'(i)$ [the probability that a bear dies during i to *i* + 1 without being reported dead is re-sighted alive between i and *i* + 1 before its death; Barker 1997, 1999). This approach allowed for the possibility of altering model structures, in the event that we elected to explicitly model temporary emigration (F') or wanted to simulate the consequences of constraining this parameter to 0, rather than allowing random temporary emigration to be incorporated in the parameter p (Burnham 1993, Barker 1997, 1999). We included harvest data through 2013 and compiled individual capture histories with the live capture and dead encounter data.

Whereas researchers during the 1990s study period were able to estimate age by physical examination and by counting annular rings on a bear's extracted premolar (Calvert and Ramsay 1998), we did not physically capture bears during 2011 – 2013, and individual age classes were assessed from the air. As such, there was uncertainty in our assignment of bears to age classes (Appendix B). Hence, during mark-recapture modeling we elected to simplify age structure relative to previous work (*cf.* Taylor et al. 2005), resulting in the following age classes: cubs-of-the-year (*coy*), yearlings (*yrl*), and individuals ≥ 2 years old (*age 2+*). Dependent young (coy and yrl) were assumed to be aged without error because of clear differences in the body size of these two age classes.

We identified a limited number of candidate sub-model structures for the parameters S, p, r, and F in the Burnham model. Because we expected that survival would vary among age classes (e.g., Taylor et al. 2005), we incorporated age structure in all 10 candidate sub-models (Table 5.1). We hypothesized that survival of coy would differ from yrl and thus constructed a three age-class structure (coy, yrl, age 2+). Because many of the coy that were sighted during the 2011-2013 sampling period were not biopsy darted, we also examined a two age-class structure in which coy and yrl were pooled for estimation of S. We hypothesized that the sexes would differ in S for the age 2+ class, primarily due to sex-selective harvest (2:1 male-to-female harvest ratio), but not for coy and yrl since they are dependent on their mothers for survival. Given sparseness of data, we examined time-constant structures for *S*, and a structure allowing temporal variation in adult survival across three sampling epochs corresponding to the live recapture and dead recovery periods (i.e., 1993 – 1997, 1998 – 2010, and 2011 – 2013). We did not have a biological reason to suspect that temporal changes in survival aligned with these sampling epochs (e.g., that survival exhibited a step change between 1997 and 1998). This structure aligned with changes in sampling design and available information on changes in bear movements, and allowed assessment of whether estimates of *S* varied whether they were based on combined data (epochs 1 and 3) or dead recoveries only (epoch 2). We specified the time-constant and epoch-based structures for *S* by constraining the design matrix in Program MARK, while maintaining full temporal structure on adult survival within the parameter index matrices (PIMs). This approach facilitated modeling of environmental covariates (see below) and future use of random effect models or Markov Chain Monte Carlo (MCMC) approaches to explore annual variation in survival (Cooch and White 2015).

We created two annual covariates, *ice transition* and *ice area* (both standardized about the mean and standard deviation) to explore relationships between *S* and environmental conditions. First, we hypothesized that the duration of the summer sea-ice transition period over the continental shelf of BB (*ice transition*; i.e., the time between break-up and freeze-up; see Chapter 4 for description of derivation of sea-ice metrics) would have a negative relationship with survival for the age 2+ classes, such that increasing duration of the ice transition period would be correlated with decreasing survival (e.g., Regehr et al. 2007). Similarly, we hypothesized that the area of ice over the continental shelf of the BB subpopulation during late spring (*ice area;* mean area of sea ice during May and June) would have a positive relationship with survival of age 2+ bears. Duration of the ice transition period and ice area over the continental shelf increased and decreased, respectively, during the 21-year study period (*ice transition*: $\beta = 2.7$ (days), $t = 3.1$, $P = 0.005$; *ice area*: $\beta = -1{,}362$ (km^2), $t = -4.2$, $P < 0.001$: -0.70). Because we did not sample many coy during 2011 – 2013 and we could only estimate survival from 7 cohorts, data were insufficient to explore relationships between time-varying covariates and the survival of dependent bears.

Preliminary analyses suggested a positive relationship between age 2+ female survival and duration of the sea-ice transition period. However, we believe this finding was an artifact of study design. No live recapture sampling occurred in 1996, which coincided with the heaviest observed sea-ice conditions during the period 1993-2013 (mean$_{1993\text{-}2013}$: 190 days; 1996: 129 days; standardized effect size: -2.18). Furthermore, the following year 1997 was the last year of live-capture data and the proportion of adult females in the sample was low compared to previous years, which we hypothesized was due to the higher levels of temporary emigration in the 1990s associated with limited geographic sampling and the availability of offshore sea ice (see Chapter 3). The combination of extreme environmental conditions in 1996, lack of live-capture sampling in 1996 and 1998-2009, and auxiliary data suggesting high levels of temporary emigration and nonrandom sampling in the 1990s led us to the conclusion that the data were likely insufficient to evaluate year-to-year variation in survival, especially toward the end of 1990s live-capture sampling. We explored the relationships between environmental covariates and S in other years by setting the 1996 value of standardized covariates to 0 (i.e., the standardized mean), and found there was not a significant relationship between sea ice and female survival. Based on these considerations we excluded sea-ice metrics from further consideration for evaluating temporal variation in S, although we explored the robustness of these results using additional post hoc analyses (see Discussion).

We created 12 candidate structures to model recapture probability (Table 5.2). We modeled coy to have the same p as females, since they remain in family groups as yearlings and are recaptured with their mothers. However, we hypothesized that p of age 2+ males (including p of male yearlings, recaptured at age 2 after break-up of family groups) would differ due to spatial segregation of bears onshore by sex and age classes (Taylor et al. 2005), and we included this two-group structure for p (*family*, including 2+ females and dependent young; and age 2+ males) in all candidate sub-models. We evaluated two candidate structures for temporal variability in p: 1) given differences in sampling protocols, search effort and survey teams between the 1990s and 2000s, p was allowed to differ between the 1990s and 2010s live-capture sampling epochs (additive or interactive effects with family); and 2) a fully-time varying structure (additive with family) for p (i.e., allowing for year-to-year variability), given that both sampling effort and environmental conditions varied significantly among years. In all candidate structures, p was fixed to 0 for the years 1996 and 1998-2010. This was necessary because p represents the probability of live-recapturing a previously-marked bear, and no live-capture sampling occurred in these years.

We hypothesized that timing of sea-ice break-up over the continental shelf may influence the distribution of polar bears, and thus p, in fall. We evaluated the standardized spring sea-ice transition (50% threshold) date in some structures (*spring ice*, Table 5.2). Exploratory analyses suggested that proximity to the coastline also may explain variability in p. Specifically, contingency tables suggested that bears initially captured inland were more likely to be recaptured inland; and conversely that bears initially captured near the coast were more likely to be recaptured in coastal regions. We created a binary geographic covariate based on an individual's first capture location, using a threshold of 2 km from a smoothed coastline (*coastline*; i.e., the coastline excluding deep fjords, see Chapter 3). We considered two temporal structures for this covariate: 1) given the apparent differences in sampling between the 1990s and 2010s, we estimated separate effects by epoch; and 2) we included the covariate effect only for the 2010s, as exploratory analyses suggested this epoch showed the strongest relationship between p and the covariate. A small number of bears (n = 33) initially captured in the first sampling period that could be alive by 2011 (<35 years of age and not harvested) were not genotyped due to inadequate tissue samples or an absence of samples. Because all sampling was conducted via remote biopsy darting and subsequent genetic analysis during 2011 – 2013, these individuals had zero probability of live recapture in this period, so we assigned these bears to unique attribute groups to fix $p = 0$ during the last 3 years of the study. As part of a study evaluating spatial ecology and population delineation (Ferguson et al. 1997, Taylor et al. 2001), a sample of adult female bears was outfitted with satellite collars in BB during 1992 – 1997. Some of these bears ($n = 14$) were captured or radio-located on Baffin and Bylot islands during fall, 1993 – 1997. Because Taylor et al. (2005) state that the locations of collared bears were known at various times of the year and this information was used to assist in recaptures during 1994, 1995, and 1997, we constructed three binary radio covariates (rad94, rad95, rad97) to identify when individual females were likely wearing functional radiocollars and therefore may have been more vulnerable to capture. Not all collared bears were recaptured, however, so we did not fix $p = 1$ for these individuals, but instead used the covariates to allow for a potentially higher recapture probability if such an effect were supported by the data. We coded dependent offspring such that radio covariates matched their mothers.

Although earlier studies (e.g., Taylor et al. 2005, 2008) have assumed that all research-marked bears were reported in the harvest, current genetic analyses identified some marked bears that were harvested but not reported as marked, possibly due to marker loss (Government of Nunavut, unpublished data; Chapter 8). Hence, reporting probabilities (r) may be biased low, especially for bears that survived many years after initial marking during which their plastic ear tags could be lost and their lip tattoos could fade. Given restrictions on the harvest of females with dependent offspring and sex-restrictive quotas, we hypothesized that r would be lower for cubs, yearlings, and 2+ females; however, we did not fix $r = 0$ for cubs because at least 1 cub was harvested. We also hypothesized that improvements in the Greenlandic reporting system during the mid-2000s might yield increased reporting rates, so we considered models where r differed by early vs. later years (1993 – 2005 vs. 2006 – 2013). Since few cubs or yearlings were harvested, we incorporated this temporal structure as an additive effect for age 2+ individuals only.

We considered three candidate structures for site fidelity (F). Previous studies (e.g., Taylor et al. 2005) assumed perfect overlap of the areas sampled by live recapture and dead recovery sampling, with no permanent emigration from the study population. To reflect this assumption we evaluated a sub-model in which F was fixed to 1 for all sex and age-classes (F=1), without assessing a parameter penalty for QAIC_c. Using this approach, any permanent emigration that actually occurs for the BB subpopulation would be reflected in lower estimates of survival. Given that the subpopulation boundaries are only partially discrete, interchange is known to occur among subpopulations, and some harvest recoveries occurred outside the BB population boundaries (Figure 5.7), we also hypothesized that bears may permanently emigrate from the BB sampling area. We therefore evaluated a structure in which F was estimated as a constant across all age-sex classes (F.). Using this approach, survival estimates would not be biased by permanent emigration, but simulations indicated that under high levels of temporary emigration, rates of permanent emigration would be overestimated (i.e., temporary emigration would be misidentified as permanent emigration due to small sample sizes and short live-recapture sampling epochs relative to the life span of polar bears). We also hypothesized that adult males would exhibit lower site fidelity, so we considered a model structure in which independent males (age 2+) had a different fidelity rate than females and their dependent offspring (F_{family}).

Overdispersion, or extra-binomial variation, exists in mark-recapture data when the capture histories of individual animals are not independent (e.g., as is the case for family groups, in which the fate of the cubs depends on the adult female, or when emigration is non-random). Correcting for overdispersion is necessary to avoid underestimating the variance of parameters. To estimate overdispersion, we constructed our most highly parameterized model and used the median $\hat{c}$ approach as implemented in Program MARK (Cooch and White 2015). This method employs simulations to generate an estimate of c, the over-dispersion parameter. Results suggested that the BB data were modestly over-dispersed (i.e., $\hat{c} = 1.1$), as would be expected given the dependency between females and their cubs (Taylor et al. 2005), so we inflated $\hat{c}$ and based model selection and inference on QAIC$_c$ (Burnham 1998).

Given 10 sub-model structures for S, 12 for p, 1 for r, and 3 for F, there would be 360 potential model structures if all possible combinations of the sub-model structures were considered. We used a modified version of the plausible combinations approach outlined in Bromaghin et al. (2013) to identify supported sub-model structures. This process entailed holding constant the most generalized structure (excluding individual covariates) for three of the four sub-models while evaluating structures for the fourth sub-model. We considered sub-model structures with ΔQAIC$_c$ < 4 as representing plausible structures and constructed all possible combinations from these sub-model structures. We note that S was poorly estimated for coy (i.e., at implausibly high rates near 1, but not inestimable), a finding which we attributed to the scarcity of data for coy, particularly during the 2010s sampling period in which many coy were not marked. Hence, we estimated a pooled S rate for coy and yrlg in all subsequent models.

Because coy were approximately 9 months old at the time of marking, it is biologically plausible that their survival rates were similar to those of yearlings.

We computed model-averaged estimates (Burnham and Anderson 2002) of parameters (S, r and F) using a threshold of $\Delta \text{QAIC}_c < 4$. Because our estimates of survival reflected harvest mortality, we derived unharvested survival (S*; also referred to as “natural” survival) using the equation as $S^* = S + r \times (1 - S)$ (e.g, Taylor et al. 2005, 2008, Peacock et al. 2013) and estimated the variance of S* using the delta method (following Taylor et al. 2008). This derivation of unharvested survival is based on several assumptions. First, it assumes harvest of all marked bears is reported; under-reporting of the harvest, which has been documented (Government of Nunavut, unpublished data; Chapter 8), leads to negative bias in estimates of S*. This derivation also assumes that harvest mortality is completely additive. In other words, that no harvested bears would otherwise die during a given interval, and that the higher population density for an unharvested subpopulation would not lead to lower vital rates for all bears in future years. A violation of the assumption of additive mortality would result in positive bias in estimates of S*. A more appropriate equation for unharvested survival would be: $S^* = S/[1 - r \times (1 - S)]$, which assumes that harvested bears are subject to the same natural mortality rate as other bears. In this equation, the quantity in brackets represents the probability of surviving the hunting season and S represents the product of natural and hunting mortality (Anderson and Burnham 1976). We used the Taylor et al. (2005) derivation for unharvested survival to maintain consistency with earlier studies, noting that the resulting potential for bias is small given high unharvested survival rates and relatively low harvest mortality for polar bears.

For highly supported models, we obtained annual estimates of abundance for groups of individuals that share common estimates of p (e.g., certain age and sex-classes), using a generalized Horvitz-Thompson estimator for the yrl and age 2+ classes, in which $\widehat{N}_{i,t} = \frac{n_{i,t}}{\hat{p}_{i,t}}$, where $n_{i,t}$ is the number of bears captured in group *i* during year *t*, and $\hat{p}_{i,t}$ is the recapture probability for group *i* during year *t*. However, estimates of n and p did not accurately represent coy because we did not sample all coy during the 2011 – 2013 sampling period. Thus, we estimated coy abundance as the product of age 2+ females with coy litters (estimated via a Horvitz-Thompson estimator) and mean observed coy litter size. We summed estimates of abundance across groups to obtain total estimates of abundance (derived for each model) by year. Similar to earlier studies (e.g., Taylor et al. 2005, Peacock et al. 2013), we used the delta method (Seber 1982, Powell 2007) to estimate variances for annual abundance estimates [R (R Core Team 2015) package emdbook (Bolker 2016)]. Variance estimates incorporated parameter variances and covariances (as computed in Program MARK) as well as variances of mean coy litter sizes. We used model weights to model-average estimates of total abundance by year and their associated variances. We also calculated mean estimates of total abundance by sampling epoch and estimated variance using the delta method. Given the 13-year interval without live captures preceding 2011, estimates of abundance for 2011 were based on values of *p* estimated for the relatively small number of bears that were marked during the 1990s and survived until 2011. These estimates of *p* were characterized by high uncertainty and potential small-sample bias. Their use in the denominator of the Horvitz-Thompson estimator, in conjunction with the large sample of first-time captures in the numerator of the estimator, had the potential to produce spurious results. Therefore, we excluded the less-reliable estimate of abundance from 2011 when calculating mean total abundance for the 2010s sampling epoch.

Geographic subsetting to evaluate the effects of different sampling methods in the 1990s and 2010s

Because sampling during the 1990s was spatially restricted to a portion of the BB subpopulation's fall range and did not include bears located farther inland, particularly within deep fjords, or on the sea ice. In contrast, from 2011-2013 onshore sampling was more comprehensive and systematic. To explore the potential impact of differences in sampling on estimates of subpopulation size, we delineated the extent of the sampling area in the 1990s based on capture locations (see Chapter 3). We then created a subset of the 2010s live-capture data that only included captures that occurred within the more restricted sampling area of the 1990s, recompiled the individual capture histories, and repeated our mark-recapture analyses using the same procedures as outlined above. We expected that comparison of abundance estimates for the 2010s using full dataset (i.e., for the complete sampling area) vs. the restricted dataset (i.e., for the restricted sampling area), would help inform the potential bias in estimates of abundance from the 1990s based on an incomplete sampling frame. Conceptually, this assumed that $\widehat{N}_{2010s}^{subset\ data}/\widehat{N}_{2010s}^{full\ data} \approx \widehat{N}_{1990s}^{subset\ data}/\widehat{N}_{1990s}^{full\ data}$; where $\widehat{N}$ represents estimates of abundance, and $\widehat{N}_{1990s}^{full\ data}$ represents the (unknown) estimate of abundance that would have been obtained in the 1990s if the complete sampling area had been covered. This assumption seemed plausible given that satellite telemetry data indicated that onshore habitat use of polar bears did not vary between the 1990s and 2010s. Although this investigation provides information on the effects of difference in onshore sampling between the two epochs, it did not provide any information on potential bias in the 1990s abundance estimate due to bears using offshore sea ice in the 1990s.

5.3. Results

The BB mark-recapture dataset consisted of 2,324 total captures of 1,992 individuals (i.e., there were 332 live recaptures), and 234 dead recoveries during the 1993 – 2013 study period (Table 5.3). Data were relatively sparse for live captures of age 2+ females during the 1990s, and dead recoveries of all bears during the early- to mid-2000s. During September 2012 and 2013, we sighted 30 polar bears (including 21 independent bears) in Greenland (*cf.* 1,043 total captures during 2012 and 2013; Table 5.3), suggesting a relatively low number of bears resided in Greenland during the late summer in those years (see Chapter 3).

Females comprised a greater proportion of live captures of age 2+ bears in the 2010s compared to the 1990s (mean annual proportion female during the 1990s: 0.42; 2010s: 0.54; Table 5.3). We hypothesize that under-representation of age 2+ females was a result of the greater use of inland habitats by denning females and lack of sampling in those habitats during the 1990s. Age 2+ males comprised nearly 70% of the reported harvest of marked bears over the 21-year study period (162 of 234; Table 5.3), with adult females exhibiting sparse recovery data, especially during the interim epoch (1998-2010) with no live encounter data.

The plausible combinations approach indicated that the following sub-model structures were supported by the data: one S structure (3 temporal epochs with an interactive effect with sex for the 2+ age class; Table 5.4); two p structures [including (1) fully time-varying p and (2) spring sea-ice transition date to explain variability in p; Table 5.5]; all three F structures (Table 5.6); and one r structure as candidate structures from which to construct the final set of models. Although within 4 ΔQAIC_c of the most highly supported p structures, we excluded p sub-models that incorporated the inland proximity to coastline covariate, as this was an uninformative parameter that was not supported by lower QAIC_c relative to hierarchically simpler models (Burnham and Anderson 1992, Arnold 2010, Peacock et al. 2012).

We constructed the final candidate model set using all combinations of the well-supported sub-model structures as identified above. The most-supported model included a fully time-varying p structure and estimated F as constant across all sex and age classes. Three additional model structures were within Δ4 QAIC_c of the most-supported model, including a model with F estimated separately for independent males, and a model with F fixed to 1 (Table 5.7).

The time-constant, model-averaged estimate of survival for dependent bears was (S = 0.87, SE = 0.06; Table 5.8). Estimates of S for age 2+ females (1993 – 1997: 0.84, SE = 0.04; 1998 – 2010: 0.95, SE = 0.02; 2011 – 2013: 0.90, SE = 0.05) and males (1993 – 1997: 0.89, SE = 0.02; 1998 – 2010: 0.87, SE = 0.02; 2011 – 2013: 0.78, SE = 0.06) varied among epochs, although statistical uncertainty and potential bias made it difficult to evaluate whether this variation was meaningful (see Discussion). Reporting rates were nearly 2-fold higher for age 2+ males than 2+ females, reflecting male-biased harvest, but there was not a strong difference in estimates of r before and after 1995. Estimated natural survival for age 2+ males (0.83, SE = 0.06) was less than age 2+ females (0.91, 0.05) during 2011 – 2013 (coy: 0.88, SE = 0.06; yrl: 0.89, SE = 0.06). Bears exhibited strong fidelity to the study area (F = 0.96 for females and dependent young, and F = 0.97 for age 2+ males). This suggests that approximately 3 – 4% of the study population permanently emigrated from the sampling area each year, although we did not utilize Barker models that additionally measure the probability that some of these bears might have returned.

Mean estimates of total abundance for the BB subpopulation were 2,173 (95% CI = 1,252 – 3,093) for the 1994 – 1997 sampling epoch and 2,826 (95% CI = 2,059 – 3,593) for the years 2012 – 2013 (Table 5.9), although these estimates correspond to different sampling frames in the

1990s and 2010s. The mean estimate of abundance for the years 2012-2013 was approximately 10% lower for the geographic subset dataset (2,553 ± 433) compared to the full dataset (2,826 ± 391). In addition, the annual point estimates for 2012 – 2013 derived from the geographic subset were lower than estimates based on the full data (Table 5.12). These findings suggest that the restricted sampling frame introduced negative bias into estimates of abundance for the 1990s. The difference in the spatial distribution of captures between the 1990s and 2010s was consistent with our conclusion that the sampling frames differed substantially between epochs, particularly with respect to the inland distribution of bears in central and northern Baffin Island (Figures 5.1, 5.8, and 5.9). Very few bears were recorded beyond the mouths of fjords in these regions during the 1990s, whereas observations were numerous there during the 2011 – 2013 epoch. This finding was reinforced by telemetry data during the 2000s that indicated no large-scale shift in onshore distribution (relative to the coastline) between epochs (see also Chapters 3 and 4).

The geographic subset included 1,679 total individuals, as >300 bears from the 2000s were censored from this analysis based on their locations outside the estimated sampling frame of the 1990s. Model selection results were generally similar to the comprehensive data set, although the coastline covariate (for modeling p) was more highly supported in some structures (Table 5.10). Parameter estimates also were consistent with the comprehensive data set (Table 5.11).

5.4. Discussion

We used physical mark-recapture data collected 1993-1995 and 1997, genetic mark-recapture data collected 2011-2013, and dead recovery data from the 21-year period 1993-2013 to estimate demographic parameters for the BB subpopulation. Our mean estimate of total abundance for the years 2012-2013 was 2,826 (95% CI = 2,059 – 3,593). Our estimate of mean abundance for the period 1994-1997 (2,173; 95% CI = 1,252 – 3,093) was consistent with the previous estimate from Taylor et al. (2005) (2,074; SE = 266), despite minor differences in the data and analytical methods. Although our 2012-2013 estimate of abundance is ~30% higher than our 1990s estimate, differences in sampling protocols and changes in environmental conditions between epochs make interpretation about true changes in population size difficult. Notably, the distribution of capture locations was different between epochs, with a higher proportion of captures in inland areas and deep within fjords during the 2010s (Figures 5.3, 5.8 and 5.9, see also Chapter 3 Figure 3.1). We used satellite telemetry data to compare on-land distribution and summer habitat use between the two epochs. These analyses provided no evidence for changes in on-land distribution in the summer, suggesting that differences in the spatial distribution of captures resulted from a more restricted sampling frame in the 1990s. In the 1990s only one helicopter was used for sampling in western Baffin Bay in fall whereas the 2000-sampling involved the use of two helicopters operating at the same time. In addition, satellite telemetry data suggested that an unknown but potentially large number of bears were not present in the 1990s study area due to the presence of summer pack ice offshore from Baffin Island, whereas offshore ice was largely unavailable in the 2010s.

We investigated potential bias resulting from differences in sampling protocols between epochs. Although flight paths for the helicopters used to capture bears in the 1990s were unavailable, we estimated the spatial extent of the sampling area using capture locations, and then created a geographical subset of the 2010s data based on this restricted sampling frame. These results indicated that a lack of inland sampling in the 2010s would have resulted in approximately 10% negative bias in the mean estimate of total abundance for 2012-2013, which suggests that similar bias likely existed in the mean estimate of total abundance for the 1990s, due to restricted geographic sampling. The source of this bias is individual heterogeneity in p due to polar bear movements with respect to the sampling area (i.e., temporary emigration; Schaub et al. 2004); which in the extreme can result in some bears effectively having $p = 0$, and therefore being completely excluded (i.e., “missed”) from the study. The presence of offshore sea ice during the 1990s was another potential source of negative bias, and an issue that we were unable to resolve. These environmental conditions are problematic because bears using the sea ice were either temporarily or permanently absent from the mark-recapture sampling area during the 1990s. Although sample sizes of independent collared females were small, telemetry data suggested that >~75% of collared bears were outside of the sampled area or on remnant sea ice during the fall sampling period in some years during 1990s. As such, an unknown but potentially large proportion of the population was unavailable for capture in some years. By contrast, in the 2010s sea ice was not present in Baffin Bay in late summer and bears were more concentrated in the onshore sampling areas (i.e., on Baffin or Bylot islands, or West Greenland); 68-85% of collared bears were inside the sampling areas in all years (2011-2013). If the probability of being a temporary emigrant in the 1990s was sufficiently high, relative to the short duration of the study and small sample sizes; or if there was Markovian dependence in the probability of being a temporary emigrant (e.g., if a bear that was on remnant sea ice in year t tended to return to the ice in year $t + 1$), then abundance estimates from the 1990s may be subject to additional bias. The sign of this bias was likely negative (i.e., it is possible that a meaningful proportion of the subpopulation was effectively excluded from abundance estimates), although the component of bias due to potential Markovian dependence could be either positive or negative depending on the directionality of Markovian dependence (Schaub et al. 2004). Our assessment of temporary emigration in the 1990s is consistent with traditional ecological knowledge in the Baffin Bay region, which suggests that some bears spend the entire year on sea ice and do not come ashore (S. Atkinson, pers. obs.). The effects of temporary emigration on 1990s abundance estimates are difficult to quantify because of the short live-recapture sampling window (1993 – 1995; 1997) and low recapture probabilities.

Given the multiple potential sources of negative bias in the 1990s abundance estimate, and statistical uncertainty in estimated parameters, we cannot conclude that the size of the BB subpopulation increased between the 1990s and 2010s. The 2010s estimate of abundance constitutes the best-available information and is suitable for informing management, but we cannot reliably determine the trend in subpopulation size between the 1990s and 2010s.

It should, however, be mentioned that during TEK-studies in both Nunavut (Dowsley 2005, Dowsley and Taylor 2006) and West Greenland (Rosing-Asvid and Born 1990, Born et al. 2011) several interviewees noted an increased occurrence of BB-polar bears in coastal areas since sometime in the 1990s. Some interviewees were of the opinion that this reflected an increase in subpopulation (BB) size whereas others thought that it reflected a change in to the bears´ behavior, and the fact that they occur closer to land as a reaction to the reduction in the sea-ice cover (cf. Born et al. 2011:206-207 for a discussion of this). Born et al. (2011) concluded that it was not possible from their interview survey to determine the extent to which an increased occurrence of polar bears in the hunting areas represents an increase in the population or a change in distribution (or for that sake a combination of these factors). Chapter 4 documents changes in the on-ice behavior of BB bears, with bears located closer to the coastline (and closer to shallow depths) in all seasons.

The sampling issues outlined above also have the potential to affect estimates of survival probability (Peñaloza et al. 2014), although these issues are mitigated to some extent by the inclusion of dead recovery data (Peacock et al. 2012). Our estimates of survival for age 2+ females 1993 – 1997 are notably lower than estimates of similar age classes obtained by Taylor et al. (2005) and Peacock et al. (2012), but our estimates of coy and yearling survival are higher. Factors that may contribute to these differences include: different treatment of data (e.g., exclusion of spring captures, recognition of coarser age classes in the present study relative to Taylor et al. 2005 and Peacock et al. 2012); our exclusion of data collected during the 1970s and 1980s, which were included in Taylor et al. (2005) as initial captures but not fully modeled as individual capture histories; and the inclusion of additional information (e.g., harvest recoveries during the 2000s and live captures during the 2010s), given that some parameters were estimated using information that was shared across sampling epochs.

The mean estimate of total annual survival of age 2+ males was particularly low (0.78, SE = 0.06) during 2011 – 2013, compared to values from earlier periods in this study and values reported for other polar bear subpopulations (PBSG 2010), with the exception of the Southern Hudson Bay subpopulation for which low estimates of adult male survival were reported for the final years of the study (Obbard et al. 2007). Although r for 2+ males was higher than 2+ females due to the sex-selective harvest, estimates of unharvested male survival 2011-2013 (S^* = 0.83) were also significantly lower than estimates of S^* for females during this period (S^* = 0.91). Low survival of adult males in the 2010s may be a biological signal that reflects a disproportionate impact of environmental change on males, either through lower true survival or increased dispersal of young males to adjacent subpopulations due to density-dependent effects. However, interpretation of point estimates and potential trends in S is difficult due to the short time-frame (i.e., only 3 years) of live recaptures in the 2010s and confounding of parameters and potential bias in estimates of survival during terminal years (e.g., Peñaloza et al. 2014). We suggest that, when viewed together with information on habitat loss (Chapter 4) and nutritional condition (Chapter 7) for BB polar bears, the low estimates of S for 2+ males 2011-2013 may signal negative density-dependent population effects. However, we emphasize that additional years of live-recapture and dead-recovery data would be necessary to determine the degree to which low estimates of survival were influenced by temporary emigration and other factors.

Adjusting total survival with r to derive unharvested survival yielded estimates of female survival in the 2010s that appear too low to support stable or positive population growth in the absence of harvest. Regehr et al. (2015) suggest that a minimum unharvested adult female survival rate of ~0.93, referenced to a population density at maximum net productivity level, is necessary for long-term persistence. We note, however, that our estimates of dependent young survival were high (*cf*. Taylor et al. 2005), although these estimates applied to the entire 1993-2013 study period. Based on the life history of polar bears, survival of dependent young would be expected to decline (e.g., in response to negative environmental conditions) before the survival of adult females declined (e.g., Eberhardt 2002). In addition, due to the lack of precise numeric age information available from non-invasive genetic sampling in the 2010s, we estimated survival for a single age class of polar bears age ≥ 2 years. Under this approach subadult and senescent bears, which likely have lower survival rates (e.g., Regehr et al. 2007), were included with prime age adults. This likely had the effect of reducing the overall estimate of female survival. We also note that the lowest estimates of total and unharvested survival for age 2+ females occurred during 1994-1997, a period during which sea-ice habitats were more available compared to the 2010s. These considerations, in conjunction with the high and variable levels of temporary emigration from the sampling area in the 1990s, and our approach of aligning temporal epochs in the survival sub-model with changes in sampling design, could have led to negative bias in estimates of female survival.

We recommend further caution in interpretation of survival estimates. First, the three temporal epochs for which we estimated survival were based on – and are confounded with – sampling methodologies (i.e., whether estimates were derived from both live recapture and dead recovery data, or dead recovery data alone). In addition, the 2011 – 2013 epoch represents the minimum length of a time series from which it is possible to estimate survival using Cormack-Jolly-Seber models, such that a single anomalous year (from either a sampling or biological perspective) has a greater impact on the pooled estimate. Additionally, negative terminal bias in survival estimates is a well-known challenge with mark-recapture studies, especially under scenarios with pronounced temporary emigration (Peñaloza et al. 2014).

To provide additional insight into our findings, we conducted complementary modeling using Markov chain Monte Carlo (MCMC) methods in Program MARK. We re-fit several well-supported models and incorporated annual random effects for S, r, and in some cases p. Annual random effects can offer advantages over fixed effect approaches by representing temporal patterns in the data via a long-term mean and annual shrinkage estimates, such that annual estimates of a parameter only deviate from the mean to the extent that any difference is supported by the data (Link and Barker 2004, White et al. 2009). MCMC methods also enable delineation of sampling from process variation. However, the ability to obtain useful annual estimates from the Baffin Bay data was somewhat limited by small sample sizes and the unusual survey design (i.e., the short time series of live-encounters at the beginning and end of the study, separated by a longer period of dead-recovery only data in the middle of the study).

MCMC analyses suggested that the low estimates of mean survival for 2+ females during 1993 – 1997 and for 2+ males during 2011 – 2013 were due in part to relatively higher levels of human-caused mortality (i.e., as represented by estimates of r) for females and males in 1996 (also the year in which no live capture sampling occurred) and 2011, respectively. Importantly, these analyses also indicated that the low estimate of survival for 2+ males during 2011 – 2013 was strongly influenced by the cohort of newly-marked bears in 2011. The long period of time without live recapture sampling (1998 – 2010) meant that bears initially marked in the 1990s (i.e., older bears) were pooled with new captures during 2011 (including younger bears) for estimation of parameters. MCMC analyses suggest that 2011 was a particularly poor year for survival of age 2+ males, but this impact was only evident among newly marked bears and not among surviving bears first marked in the 1990s. This finding suggests possible individual or finer-scale age-based variation in survival, but sample sizes and study design (i.e., the 13-year interval with no live captures) were insufficient to fit models with individual random effects using either maximum likelihood or MCMC methods. In sum, MCMC analyses supported our interpretation that the BB data do not provide strong evidence for temporal changes in survival, with the exception of the two years noted above. We recommend that future work in Baffin Bay and elsewhere further explore models with annual random effects. In addition, incorporation of dead recovery data after 2013 will assist in estimating survival during the 2011-2013 sampling period (Peacock et al. 2012).

Based on supplementary analyses, conducted as part of this investigation, that suggest a relatively strong ability to distinguish subadults from adults using field assessments and genetic information (i.e., sex) obtained from biopsy darting (Appendix B), future modeling of data from genetic mark-recapture studies could consider a more detailed age structure. Recent advances in analytical methods (e.g., mark-recapture models with state uncertainty; Pradel 2009) could be used to model the relatively low occurrence of errors in estimation of field ages. Given the broader base of evidence that the BB subpopulation is responding to losses of sea-ice habitat (e.g., Rode et al. 2011, Peacock et al. 2012; Chapters 4, 6, 7), considering a more detailed age structure could help to assess whether survival rates for adults and subadults exhibited different temporal trends (e.g., per the expectation that subadult survival rates are among the first demographic parameters to respond to environmental changes; Regehr et al. 2007, Stirling and Derocher 2012).

To further assess bias and precision of estimated model parameters, we completed simulations in which Barker models, which are capable of explicitly modeling temporary emigration, were fitted to datasets that closely resembled the BB data, but included known levels and types of temporary emigration. Under moderate to high random temporary emigration (F=F'= 0.3 or 0.5), survival estimates were unbiased when estimated using Barker models, but moderately negatively biased when based on Burnham models (especially for females, which had lower recovery rates). CJS and Seber models both produced highly biased estimates of S when temporary emigration occurred. Markovian emigration may yield negative bias in S, although sparse telemetry data did not permit assessment of Markovian patterns in temporary emigration with the actual Baffin Bay data. Although we were unable to simulate data on abundance, Barker models provided unbiased estimates of p, whereas Burnham models provided unbiased estimates of the product of $p \times F$. These findings suggest that Barker models would provide reasonable estimates of the number of bears located within the sampling area on any given sampling occasion, whereas Burnham models would provide better estimates of the "super-population" (i.e., the larger group of bears with a non-negligible probability of using the sampling area, even if some of these bears were outside of the sampling area [i.e., were temporary emigrants] on any given sampling occasion). Given that the super-population corresponds more closely to the study population of interest in BB from a biological and management perspective, this supports our decision to use the Burnham model to estimate abundance.

Although there are uncertainties in the BB subpopulation related to the demographic analyses, additional sources of information, including sea-ice conditions, movement ecology, and reproductive metrics, are useful for informing current subpopulation status. These auxiliary data suggest a lengthening of the ice-free season (Chapter 4), a significant increase in the amount of time bears spend on land during the ice-free season (Chapter 4), a northward shift in their ranges (Chapter 2), decreased reproductive output (Chapter 6), and declining body condition (Rode et al. 2011; Chapter 7). These signs point to a subpopulation that could be exhibiting density-dependent effects associated with declining carrying capacity. The relatively low estimates of unharvested survival for the 2010s, especially for adult males, are consistent with this explanation, although as stated previously we cannot rule out the presence of negative bias in survival estimates.

The Baffin Bay study highlights potential challenges in interpreting long-term trends in abundance and survival. Although Taylor et al. (2005) and York et al. (2016) assert that the BB subpopulation was uniformly and comprehensively sampled during the 1990s, we documented evidence of changes in the sampling frames between epochs (i.e., incomplete spatial sampling during the 1990s, relative to the 2000s). These changes precluded an assessment of trends in abundance, and the 13-year interval between live capture sessions limited our ability to assess temporal trends in survival, and likely resulted in increased individual heterogeneity in survival (which we could not explicitly model) as newly-marked bears in the 2010s were pooled with older individuals first marked in the 1990s. We attempted to reconcile and understand the impacts of these issues through supplemental analyses, and future work could explore the usefulness of new analytical methods (e.g., multistate models with unobservable states; spatially-explicit models [Royle et al. 2013]) to mitigate potential bias. However, recent analyses of mark-recapture data for polar bears have identified the limitations of model-based methods to account for inconsistent sampling or violated modeling assumptions (e.g., Bromaghin et al. 2015). Therefore, we underscore the critical importance of complete and consistent sampling of the study area for robust and unbiased inference regarding population status. We also recommend evaluating the current inventory schedule and using *a priori* study design analyses to evaluate whether modifications (e.g., extending the live capture sampling periods, shortening the interval between successive capture periods, incorporation of more intensive "robust design" sampling [Converse et al. 2009]) may improve the ability to detect changes in abundance and associated vital rates. Finally we recommend considering other survey methods (i.e., aerial surveys; e.g., Aars et al. 2009) for assessing polar bear subpopulations.

5.5. Literature Cited

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea-ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Anderson, D. R. and K. P. Burnham. 1976. Population ecology of the mallard: VI. The effect of exploitation on survival. Resource Publication No. 128, U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Arnold, T. W. 2010 Uninformative parameters and model selection using Akaike's Information Criterion. Journal of Wildlife Management 74:1175–1178.

Atwood, T., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. E. Bromaghin. 2015. Evaluating and ranking threats to the long-term persistence of polar bears. U.S. Geological Survey Open-File Report 201-1254. U.S. Geological Survey, Reston, Virginia, USA.

Barker, R. J. 1997. Joint modeling of live-recapture, tag-resight, and tag-recovery data. Biometrics 53:666–677.

Barker, R. J. 1999. Joint analysis of mark-recapture, resighting and ring-recovery data with age-dependence and marking effect. Bird Study 46:S82–S91.

Barker, R. J., and G. C. White. 2001. Joint analysis of live and dead encounters of marked animals. Pages 361–367 *in* R. Field, R. J. Warren, H. Okarma, and P. R. Sievert, editors. Wildlife, Land, and People: Priorities for the 21st Century. Proceedings of the Second International Wildlife Management Congress, The Wildlife Society, Bethesda, Maryland, USA.

Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. Wildlife Society Bulletin 31:197–207.

Bolker, B. 2016. Ecological Models and Data in R. R package version 1.3.9.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Boulanger, J., S. Himmer, and C. Swan. 2004. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British

Columbia. Canadian Journal of Zoology 82:1267–1277.

Bromaghin, J. F., T. L. McDonald, and S. C. Amstrup. 2013. Plausible combinations: an improved method to evaluate the covariate structure of Cormack-Jolly-Seber mark-recapture models. Open Journal of Ecology 3:11–22.

Burnham, K. P. 1993. A theory for combined analysis of ring recovery and recapture data. Pages 199–213 *in* J. D. Lebreton and P. M. North, editors. Marked individuals in the study of bird populations. Birkhauser-Verlag, Basel, Switzerland.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.

Comiso, J. C., C. L. Parkinson, R. Gertsen, and L. Stock. 2008. Accelerated decline in the Arctic sea ice cover. Geophysical Research Letters 35:L01703.

Converse, S. J., W. L. Kendall, P. F. Doherty, and P. G. Ryan. 2009. Mutistate models for estimation of survival and reproduction in the grey-headed albatross (Thalassarche chrysostoma). The Auk 126:77-88.

Cooch, E.G., White, G.W., 2015. Program MARK: A Gentle Introduction. 14th ed. Colorado State University, Fort Collins, CO, USA.

Devineau, O., R. Choquet, and J. D. Lebreton. 2006. Planning capture-recapture studies: straightforward precision, bias, and power calculations. Wildlife Society Bulletin 34:1028-1035.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1: 43 pp.

Dowsley, M. and M.K. Taylor. 2006. Community consultations with Qikiqtarjuaq, Clyde River and Pond Inlet on management concerns for the Baffin Bay (BB) polar bear population: A summary of Inuit knowledge and community consultations. Nunavut Wildlife Research Group Final Report No. 2: 83 pp.

Eberhardt, L. L. 2002. A paradigm for population analysis of long-lived vertebrates. Ecology 83:2841-2854.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use of polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. Ecology 81:761–772.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. Journal of Wildlife Management 73:3–17.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lebreton, J.-D., J. D. Nichols, R. J. Barker, R. Pradel, and J. A. Spendlow. 2009. Modeling individual animal histories with multistate capture-recapture models.*in* H. Caswell, editor. Advances in Ecological Research Volume 41. Academic Press, New York.

Link, W. A., and R. J. Barker. 2004. Hierarchical mark-recapture models: a framework for inference about demographic processes. Animal Biodiversity and Conservation 27:441–449.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications doi:10.1890/15-1256.

Obbard, M. E., T. L. McDonald, E. J. Howe, E. V. Regehr, and E. S. Richardson. 2007. Polar bear population status in Southern Hudson Bay, Canada. Administrative report, U.S. Geological Survey, Reston, Virginia, USA.

Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. Molecular Ecology 12:1375–1387.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Palsbøll, P. J., J. Allen, M. Bérube, P. J. Clapham, T. P. Feddersen, P. S. Hammond, R. R. Hudson, H. Jørgensen, S. Katona, A. H. Larsen, F. Larsen, J. Lien, D. K. Mattila, J. Sigurjónsson, R. Sears, T. Smith, R. Sponer, P. Stevick, and N. Øien. 1997. Genetic tagging of humpback whales. Nature 388:767–769.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014. http://pbsg.npolar.no/en/status/status-table.html, Accessed 4 May 2016.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Peacock, E., S. A. Sonsthagen, M. E. Obbard, A. Boltunov, E. V. Regehr, N. Ovsyanikov, J. Aars, S. Atkinson, G. K. Sage, A. G. Hope, E. Zeyl, L. Bachmann, D. Ehrich, K. T. Scribner, S. C. Amstrup, S. Belikov, E. W. Born, A. E., Derocher, I. Stirling, M. K. Taylor, Ø. Wiig, D. Paetkau, and S. L. Talbot. 2015. Implications of the circumpolar genetic structure of polar bears for their conservation in a rapidly warming Arctic. PLoS ONE 10:e112021. doi:10.1371/journal.pone.0112021

Peñaloza, C. L., W. L. Kendall, and C. A. Langtimm. 2014. Reducing bias in survival under nonrandom temporary emigration. Ecological Applications 24:1155–1166.

Powell, L. 2007. Approximating variance of demographic parameters using the delta method: A reference for avian biologists. Condor 109:949–954.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Rosing-Asvid, A. and E.W. Born. 1990. Fangst af isbjørn (*Ursus maritimus*) i Avanersuaq og Upernavik kommuner: En interviewundersøgelse (The catch of polar bears, *Ursus maritimus*, in the municipalities of Avarnersuaq and Upernavik: An interview survey. Grønlands Hjemmestyres Miljø –og Naturforvaltning Teknisk Rapport Nr. 23: 63 pp. (In Danish with an English and Greenlandic Abstract).

Royle, J. A., R. B. Chandler, R. Sollmann, and B. Gardner. 2013. Spatial capture-recapture. Academic Press, Waltham, Massachusetts, USA.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Macmillan, New York, New York, USA.

Schaub, M., O. Gimenez, B. R. Schmidt, and R. Pradel. 2004. Estimating survival and temporary emigration in the multistate capture-recapture framework. Ecology **85**:2107-2113.

Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., G. W. Thiemann, and E. Richardson. 2008. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568-574.

Stroeve, J. C., M. C. Serreze, M. M. Holland, J. E. Kay, J. Malanik, and A. P. Barrett. 2012. The Arctic’s rapidly shrinking sea ice cover: A research synthesis. Climatic Change 110:1005–1027.

Taylor, M. K., and J. Lee. 1995. Distribution and abundance of Canadian polar bear populations: a management perspective. Arctic 48:147–154.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667–1673.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. Marine Mammal Science 25:778–796.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46:S120–S139.

White, G. C., K. P. Burnham, and R. J. Barker. 2009. Evaluation of a Bayesian MCMC random effects inference methodology for capture-mark-recapture data. Pages 1119–1127 *in* D. L. Thomson, E. G. Cooch, and M. J. Conroy, editors. Modeling Demographic Processes in Marked Populations, Springer, New York, USA.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution 6:2897–2924.

Table 5.1. Survival sub-model structures evaluated in mark-recapture analysis of the Baffin Bay polar bear subpopulation data.

S sub-model	Age	Sex	Temporal	Environmental
1	2 class	Age 2+ only	Constant	None
2	2 class	Age 2+ only	3 epoch + sex	None
3	2 class	Age 2+ only	3 epoch × sex	None
4	2 class	Age 2+ only	Constant	Ice transition
5	2 class	Age 2+ only	Constant	Ice area
6	3 class	Age 2+ only	Constant	None
7	3 class	Age 2+ only	3 epoch + sex	None
8	3 class	Age 2+ only	3 epoch × sex	None
9	3 class	Age 2+ only	Constant	Ice transition
10	3 class	Age 2+ only	Constant	Ice area

Table 5.2. Recapture probability sub-model structures evaluated in mark-recapture analysis of the Baffin Bay polar bear subpopulation. All models included a Radio covariate for bears that were outfitted with a satellite collar that may have been used to locate individuals for recapture. Bears that were not genotyped were unavailable to be recaptured during the 2011 – 2013 sampling window, so p was fixed to 0 for non-genotyped bears.

p sub-model	Family	Temporal	Geographic	Ice
1	Yes	2 epoch + family	None	None
2	Yes	2 epoch + family	Coastline, 2 epoch	None
3	Yes	2 epoch + family	Coastline, 2010s	None
4	Yes	2 epoch + family	None	Spring
5	Yes	2 epoch + family	Coastline, 2 epoch	Spring
6	Yes	2 epoch + family	Coastline, 2010s	Spring
7	Yes	2 epoch × family	None	None
8	Yes	2 epoch × family	Coastline, 2 epoch	None
9	Yes	2 epoch × family	Coastline, 2010s	None
10	Yes	Annual + family	None	None
11	Yes	Annual + family	Coastline, 2 epoch	None
12	Yes	Annual + family	Coastline, 2010s	None

Table 5.3. Summary table of live captures and dead recoveries during the mark-recapture study of the Baffin Bay polar bear subpopulation in Nunavut, Canada, and Greenland, 1993 – 2013. Shaded cells indicate that data are not possible due to an absence of marking or recapture.

	Initial captures						**Live recaptures**				**Dead recoveries**					
	Females			*Males*			*Females*		*Males*		*Females*			*Males*		
Year	**Coy**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Yrl**	**2+**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**	**Coy**	**Yrl**	**2+**
1993	14	8	53	12	8	61					0	0	1	0	0	0
1994	26	13	65	16	9	77	0	5	0	14	0	0	3	0	0	7
1995	15	11	62	19	11	85	4	11	4	23	0	2	6	1	0	8
1996												1	8		0	7
1997	22	10	60	19	13	113		20		31	0	0	6	0	1	9
1998												0	3		0	11
1999													3			9
2000													0			8
2001													2			8
2002													0			11
2003													0			7
2004													1			7
2005													2			3
2006													3			6
2007													1			2
2008													2			4
2009													2			0
2010													0			1
2011	2	23	163	1	20	148		5		5	0	0	4	0	0	20
2012	40	30	221	35	30	192	3	41	0	54	0	0	8	0	2	14
2013	28	15	121	16	15	90	4	48	5	55	0	1	8	1	0	20
Totals	**147**	**110**	**745**	**118**	**106**	**766**	**11**	**130**	**9**	**182**	**0**	**4**	**63**	**2**	**3**	**162**

Table 5.4. Survival (*S*) sub-model selection results from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation. Epoch = periods defined by sampling method (1993 – 1997, 1998 – 2010, and 2011 – 2013). Preliminary analyses suggested that QAIC*c* scores of structures including sea-ice metrics were critically dependent on 1996, the year in which there was no live recapture sampling, which also happened to coincide with heavy sea ice. Structures with sea-ice covariates thus were eliminated from further consideration.

***S* sub-model structure**	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
coy yrl, 2+(sex × epoch)	22	0.00	0.978	3878.0
coy yrl, 2+(sex + epoch)	20	8.36	0.015	3890.4
coy yrl, 2+(sex)	18	9.83	0.007	3896.0

Table 5.5. Recapture probability (*p*) sub-model selection results from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Family = females / dependent bears and independent males (2 age / sex classes); ice = spring transition date; epoch = sampling period (1993 – 1995, 1997; 2011 – 2013); t = full time variation; and inland = proximity of individual's first capture location to smoothed coastline (2 km threshold; binary). All *p* structures incorporated the radio collar covariate representing bears that were outfitted with collars that may have been used to locate them.

***p* sub-model structure**	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
family + t	22	0	0.418	3878.0
family + t + coastline (2010s)	23	1.31	0.217	3877.3
family + t + coastline (epoch)	24	1.32	0.216	3875.2
family + epoch + ice	19	3.50	0.073	3887.6
family + epoch + ice + coastline (epoch)	21	4.78	0.038	3884.8
family + epoch + ice + coastline (2010s)	20	4.78	0.038	3886.8
family + epoch	18	15.49	0.0002	3901.6
family + epoch + coastline (2010s)	19	16.96	0.0001	3901.0
family + epoch + coastline (epoch)	20	17.08	0.0001	3899.1
family × epoch	19	17.31	0.0001	3901.4
family × epoch + coastline (epoch)	21	18.66	<0.0001	3898.7
family × epoch + coastline (2010s)	20	18.71	<0.0001	3900.8

Table 5.6. Fidelity (*F*) sub-model selection results from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation.

***F* sub-model structure**	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
Constant	21	0.00	0.57	3878.4
coy yrl 2+ F, 2+ M	22	1.62	0.25	3878.0
Fixed = 1	20	2.28	0.18	3882.7

Table 5.7. Model selection results (< 4 $\Delta QAIC_c$) from analysis of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation. For S, epoch = periods defined by sampling method (1993 – 1997, 1998 – 2010, and 2011 – 2013). For p, family = females / dependent bears and independent males (2 age / sex classes); ice = spring transition date; epoch = sampling period (1993 – 1995, 1997; 2011 – 2013); and t = full time variation. For r, time = 1992 – 2005 and 2006 – 2013. All p structures incorporated the radio collar covariate representing bears that were outfitted with collars that may have been used to locate them.

Model Structures							
S	*P*	*R*	*F*	**Parameters**	**ΔAICc**	**AICc Weights**	**QDeviance**
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Constant	21	0	0.52	3878.4
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	22	1.62	0.23	3878.0
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Fixed = 1	20	2.28	0.17	3882.7
coy yrl, 2+(sex × epoch)	family + epoch + ice	coy, yrl, 2+ (sex + time)	Constant	18	3.57	0.09	3888.1

Table 5.8. Model averaged (<Δ 4 $QAIC_c$) parameter estimates obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1993 – 2013.

Parameter	Class	Estimate (SE)
Survival (total)		
	Cubs of the year / yearlings	0.87 (0.06)
	2+ Females, 1993 – 1997	0.84 (0.04)
	2+ Females, 1998 – 2010	0.95 (0.02)
	2+ Females, 2011 – 2013	0.90 (0.05)
	2+ Males, 1993 – 1997	0.89 (0.02)
	2+ Males, 1998 – 2010	0.87 (0.02)
	2+ Males, 2011 – 2013	0.78 (0.06)
Reporting		
	Cubs of the year	0.06 (0.05)
	Yearlings	0.13 (0.07)
	2+ Females, 1993 – 2005	0.19 (0.05)
	2+ Females, 2006 - 2013	0.16 (0.05)
	2+ Males, 1993 – 2005	0.30 (0.03)
	2+ Males, 2006 – 2013	0.26 (0.06)
Fidelity		
	Cubs of the year, yearlings, and 2+ females	0.96 (0.03)
	2+ Males	0.97 (0.02)

Table 5.9. Model averaged (<Δ 4 $QAIC_c$) estimates of abundance ($\hat{N}$ ± SE; [95% Confidence Interval]) obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1994 – 1997, 2011 – 2013. The 2011 estimate is believed to be biased based on a limited sample of surviving bears from the 1990s that were available for recapture (see Methods).

1994	1995	1997	2011	2012	2013
2280 ± 615 (1073-3486)	1999 ± 359 (1295-2703)	2239 ± 393 (1469-3009)	4202 ± 1762 (749-7656)	2595 ± 352 (1905-3286)	3056 ± 426 (2221-3893)

Table 5.10. Model selection results (< 4 $\Delta QAIC_c$) from analysis of geographic subset of mark-recapture-recovery data from the Baffin Bay polar bear subpopulation, 1993 – 2013. Coy = cubs of the year. Yrl = yearlings. 2+ = bears aged 2 and older. Age classes separated by a comma were estimated independently; classes not separated by a comma were pooled for estimation. For S, epoch = periods defined by sampling method (1993 – 1997, 1998 – 2010, and 2011 – 2013). For p, family = females / dependent bears and independent males (2 age / sex classes); ice = spring transition date; epoch = sampling period (1993 – 1995, 1997; 2011 – 2013); inland = proximity of initial captur to smoothed coastline; and t = full time variation. For r, time = 1992 – 2005 and 2006 – 2013. All p structures incorporated the radio collar covariate for bears that were outfitted with collars that may have been used to locate them.

Model Structures							
S	*P*	*r*	*F*	**Parameters**	**ΔQAICc**	**QAICc Weights**	**QDeviance**
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Constant	21	0.00	0.29	3361.5
coy yrl, 2+(sex × epoch)	family + epoch + ice + coastline (epoch)	coy, yrl, 2+ (sex + time)	Constant	20	0.80	0.19	3364.3
coy yrl, 2+(sex × epoch)	family + epoch + ice	coy, yrl, 2+ (sex + time)	Constant	18	1.09	0.17	3368.7
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	22	1.71	0.12	3361.2
coy yrl, 2+(sex × epoch)	family + t	coy, yrl, 2+ (sex + time)	Fixed = 1	20	2.46	0.08	3366.0
coy yrl, 2+(sex × epoch)	family + epoch + ice + coastline (epoch)	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	21	2.49	0.08	3364.0
coy yrl, 2+(sex × epoch)	family + epoch + ice	coy, yrl, 2+ (sex + time)	coy yrl 3+ F, 3+ M	19	2.76	0.07	3368.3

Table 5.11. Model averaged ($<\Delta$ 4 QAIC$_c$) parameter estimates obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1993 – 2013, using the geographic data subset.

Parameter	Class	Estimate (SE)
Survival (total)		
	Cubs of the year / yearlings	0.89 (0.06)
	2+ Females, 1990s	0.85 (0.04)
	2+ Females, Gap	0.95 (0.02)
	2+ Females, 2011 – 2013	0.91 (0.05)
	2+ Males, 1990s	0.89 (0.03)
	2+ Males, Gap	0.87 (0.02)
	2+ Males, 2011 – 2013	0.78 (0.06)
Reporting		
	Cubs of the year	0.08 (0.07)
	Yearlings	0.10 (0.07)
	2+ Females, 1993 – 2005	0.19 (0.05)
	2+ Females, 2006 - 2013	0.17 (0.06)
	2+ Males, 1993 – 2005	0.29 (0.03)
	2+ Males, 2006 – 2013	0.27 (0.06)
Fidelity		
	Cubs of the year, yearlings, and 2+ females	0.95 (0.03)
	2+ Males	0.95 (0.03)

Table 5.12. Model averaged ($<\Delta$ 4 QAIC$_c$) estimates of abundance ($\widehat{N}$ ± SE) obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1994 – 1997 and 2011 – 2013, using the geographic data subset.

1994	1995	1997	2011	2012	2013
2545 ± 597	2208 ± 382	2225 ± 418	2516 ± 1473	2447 ± 423	2659 ± 442

Figure 5.1. The BB subpopulation boundaries include portions of Nunavut, Canada, and West Greenland.

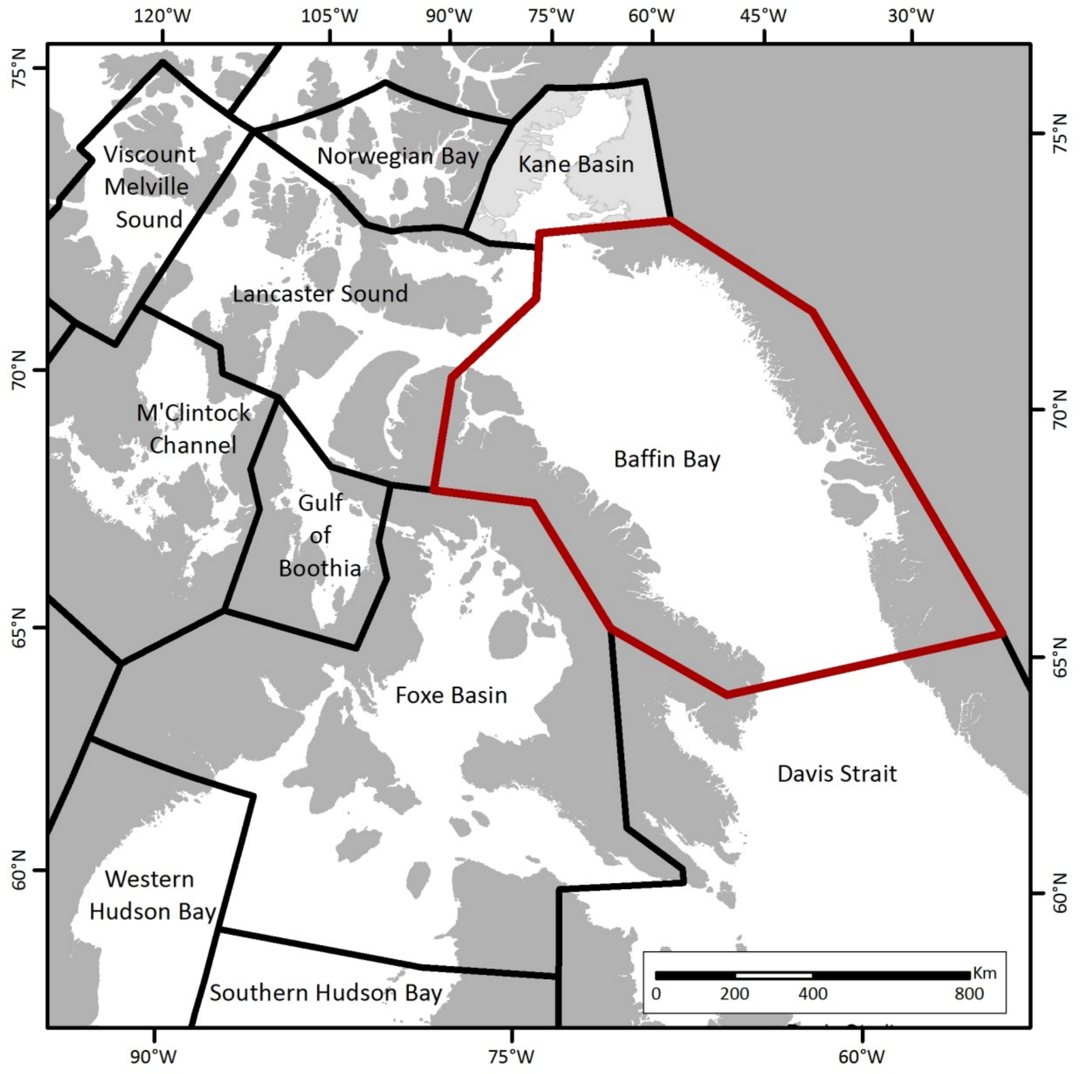


Figure 5.2. Locations of polar bears captured in Baffin Bay during the 1970s, 1980 - 1985, and 1989 – 1993.

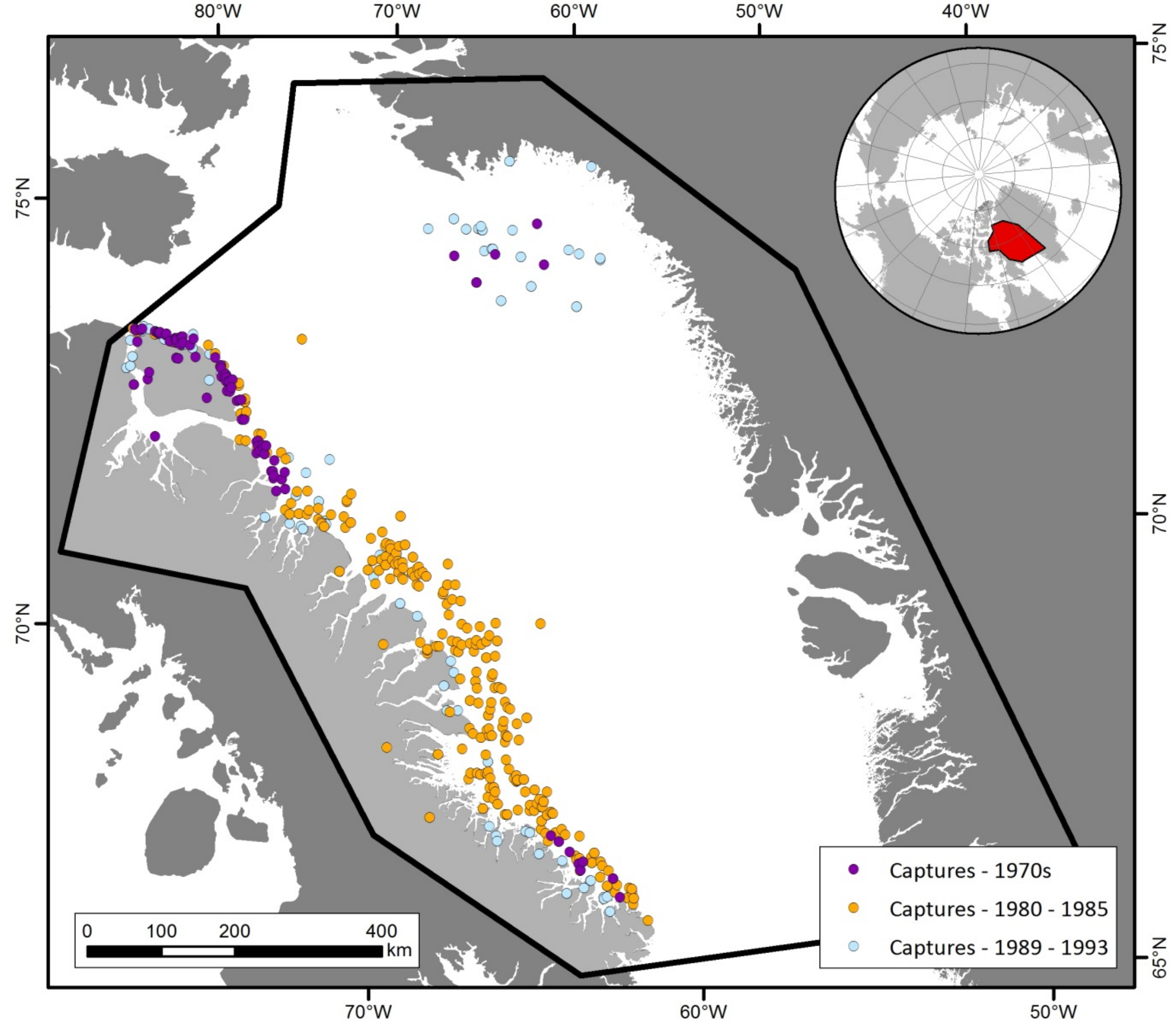


Figure 5.3. Locations of polar bears captured in Baffin Bay during August – October, 1993 – 1995 and 1997.

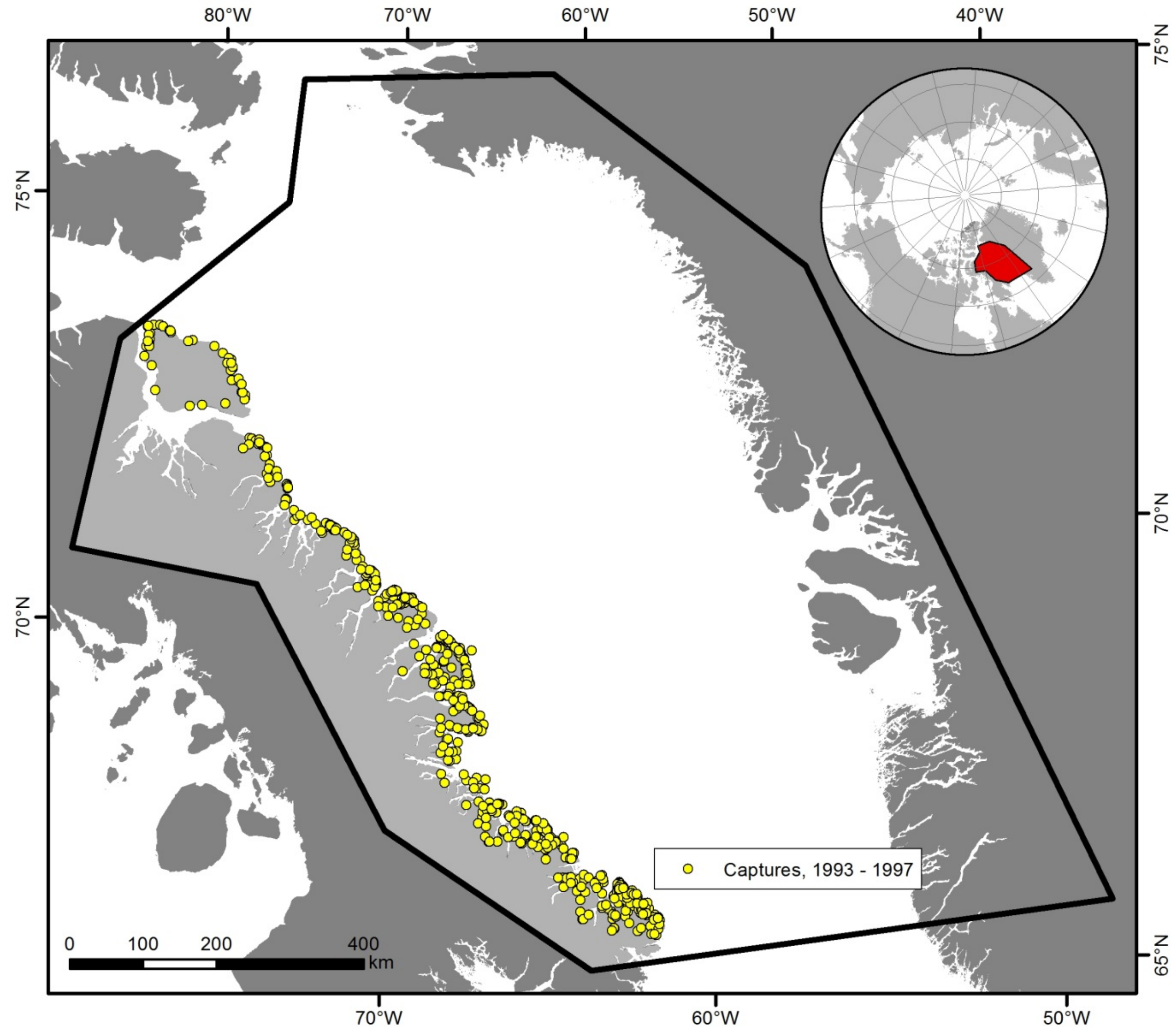


Figure 5.4. Sampling strata delineated on Baffin Island, Canada, for genetic mark-recapture study completed during 2011 – 2013. Fall sampling also was completed in the nearshore regions around Melville Bay, Greenland, denoted by the yellow star.

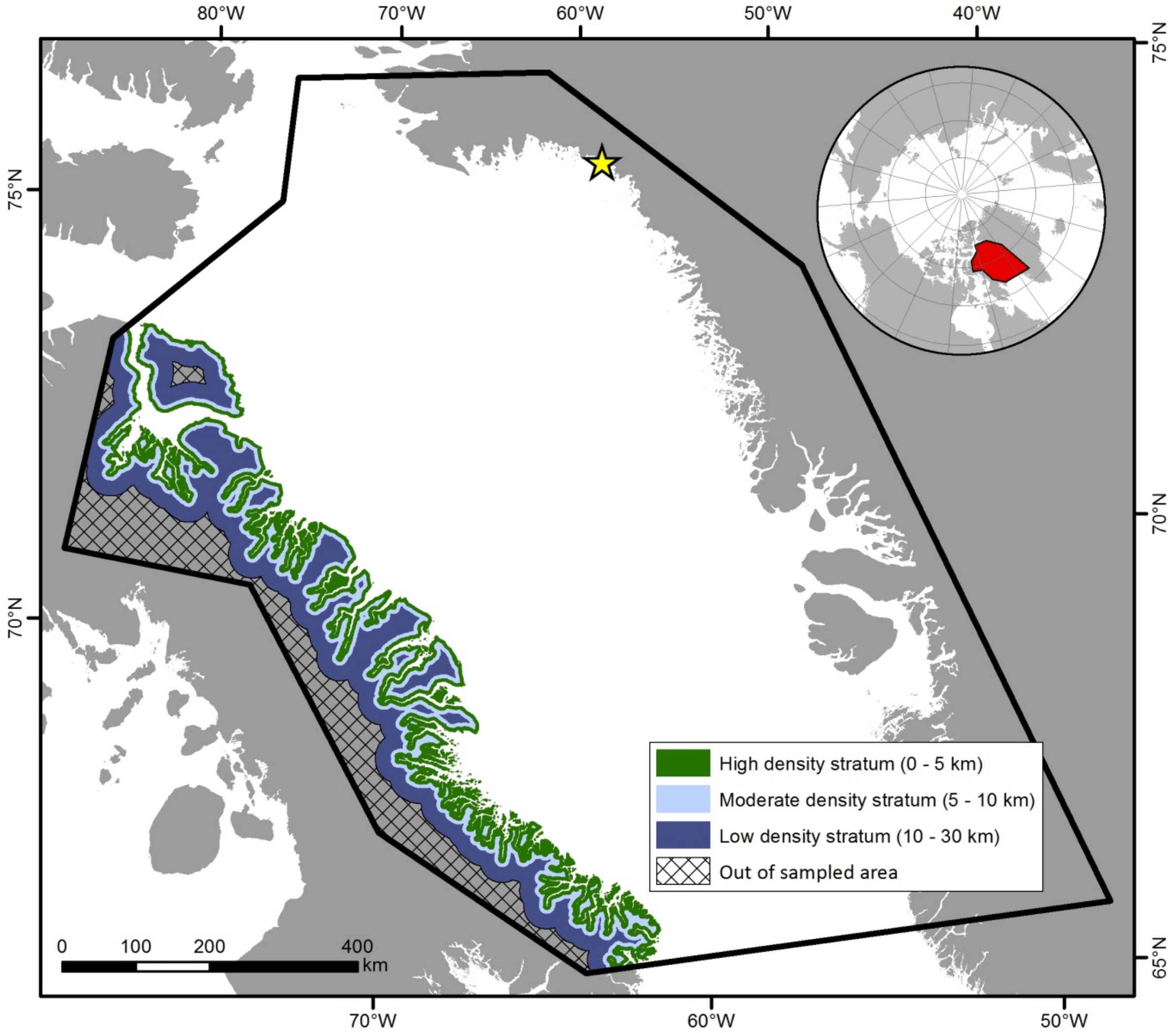


Figure 5.5. Left: Pneudart Inc. DNA (top) and Biopsy (bottom) darts used during the 2011 genetic-mark recapture in Baffin Bay. Right: A sample of skin and fat provide by a DNA dart.

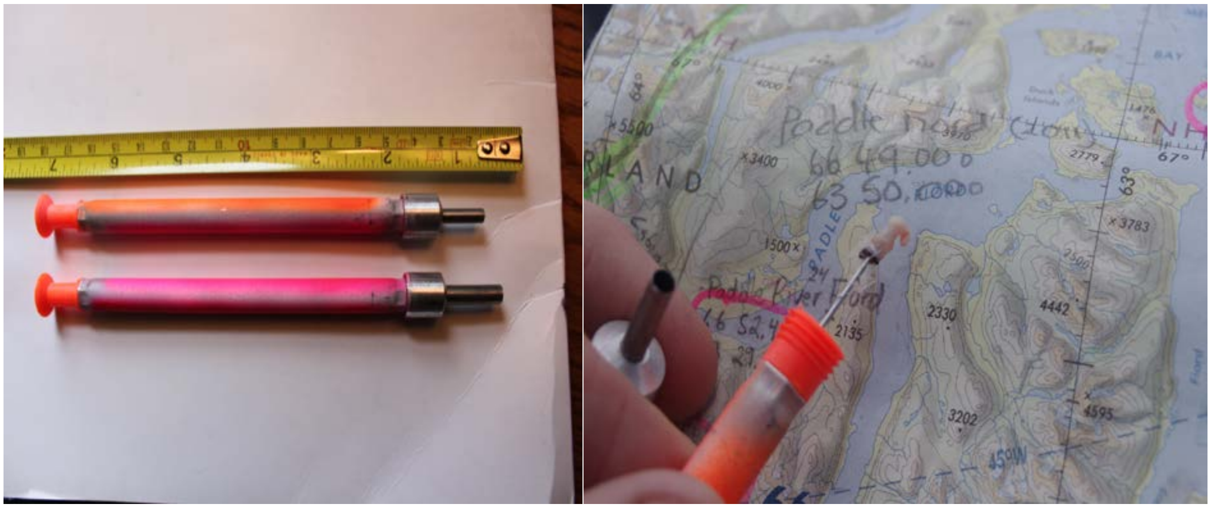


Figure 5.6. 9-locus mismatch distribution for 4,657 polar bears from Nunavut and the Greenland side of BB.

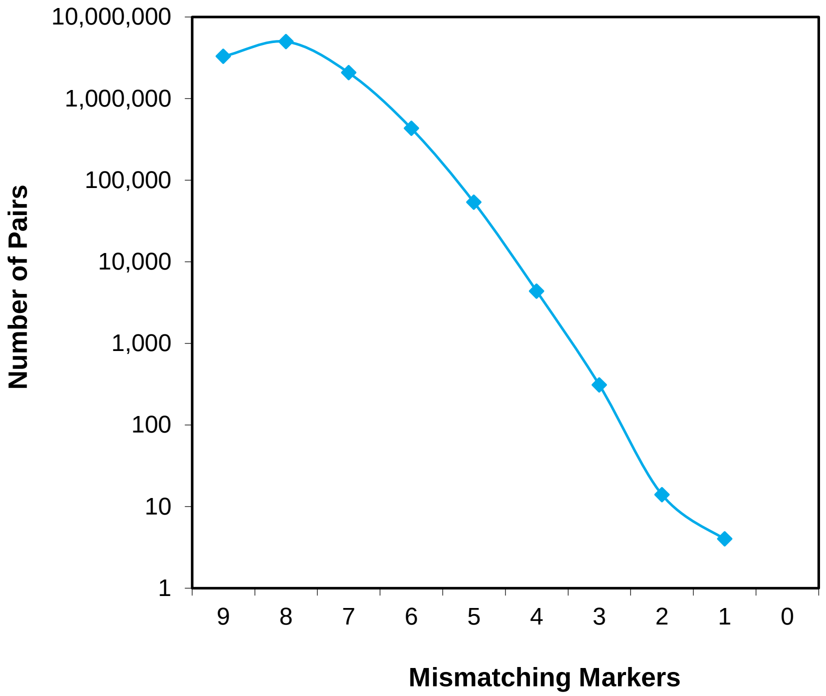


Figure 5.7. Locations of marked BB polar bears recovered in the harvest in BB and surrounding subpopulations during 1993 – 2013.

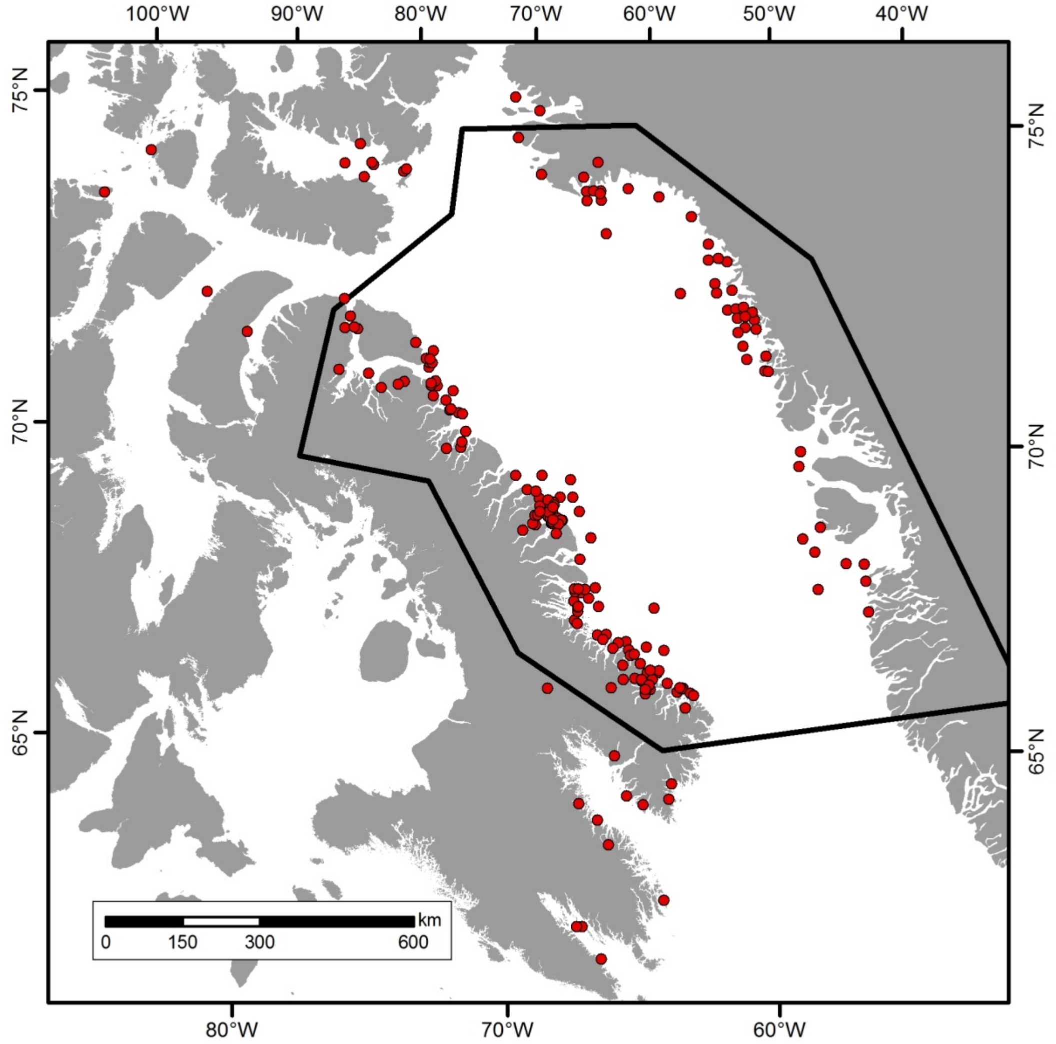


Figure 5.8. Locations of polar bears sampled in Baffin Bay during August – October, 2011 – 2013. Sampling in Greenland occurred near Melville Bay.

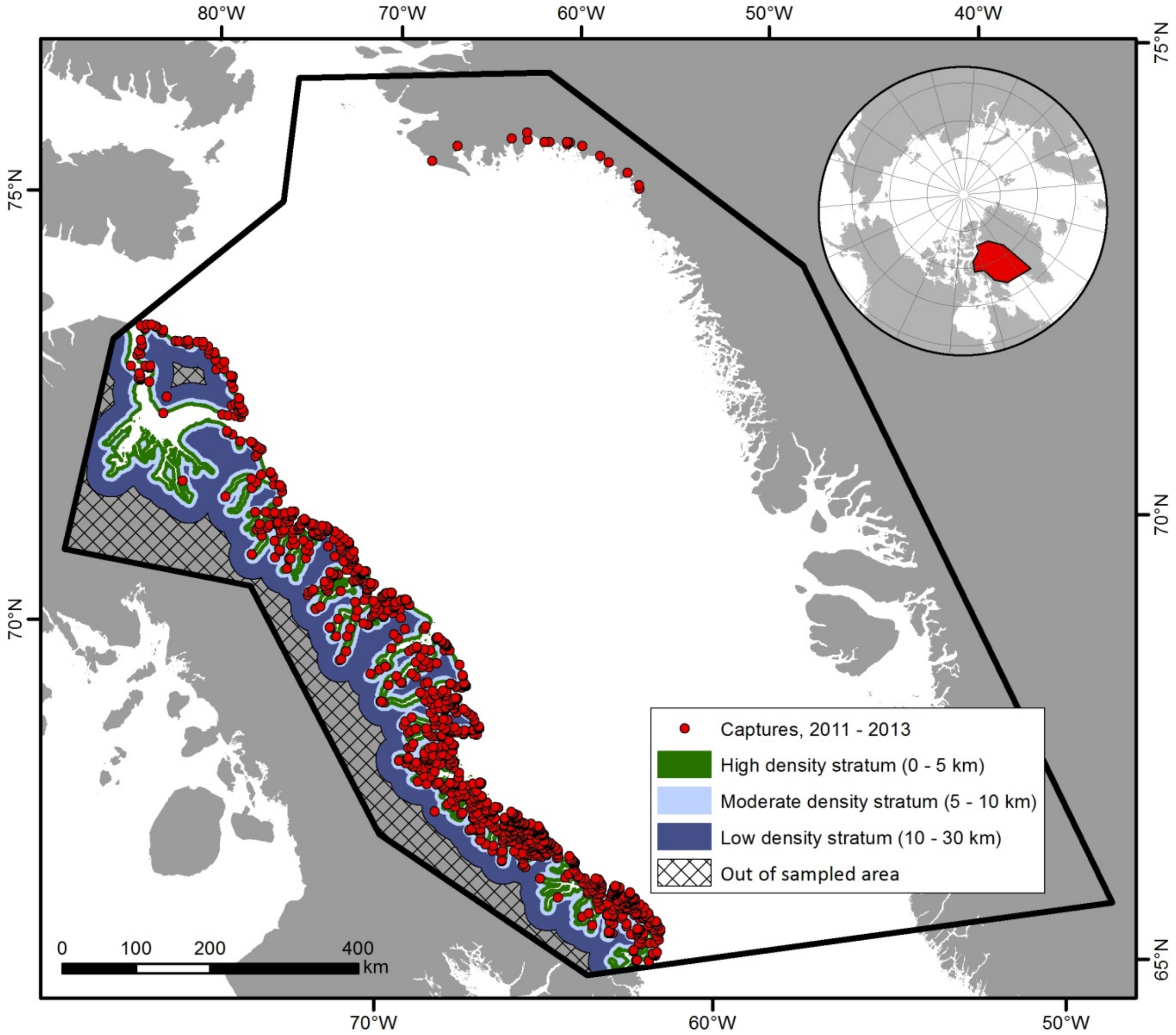
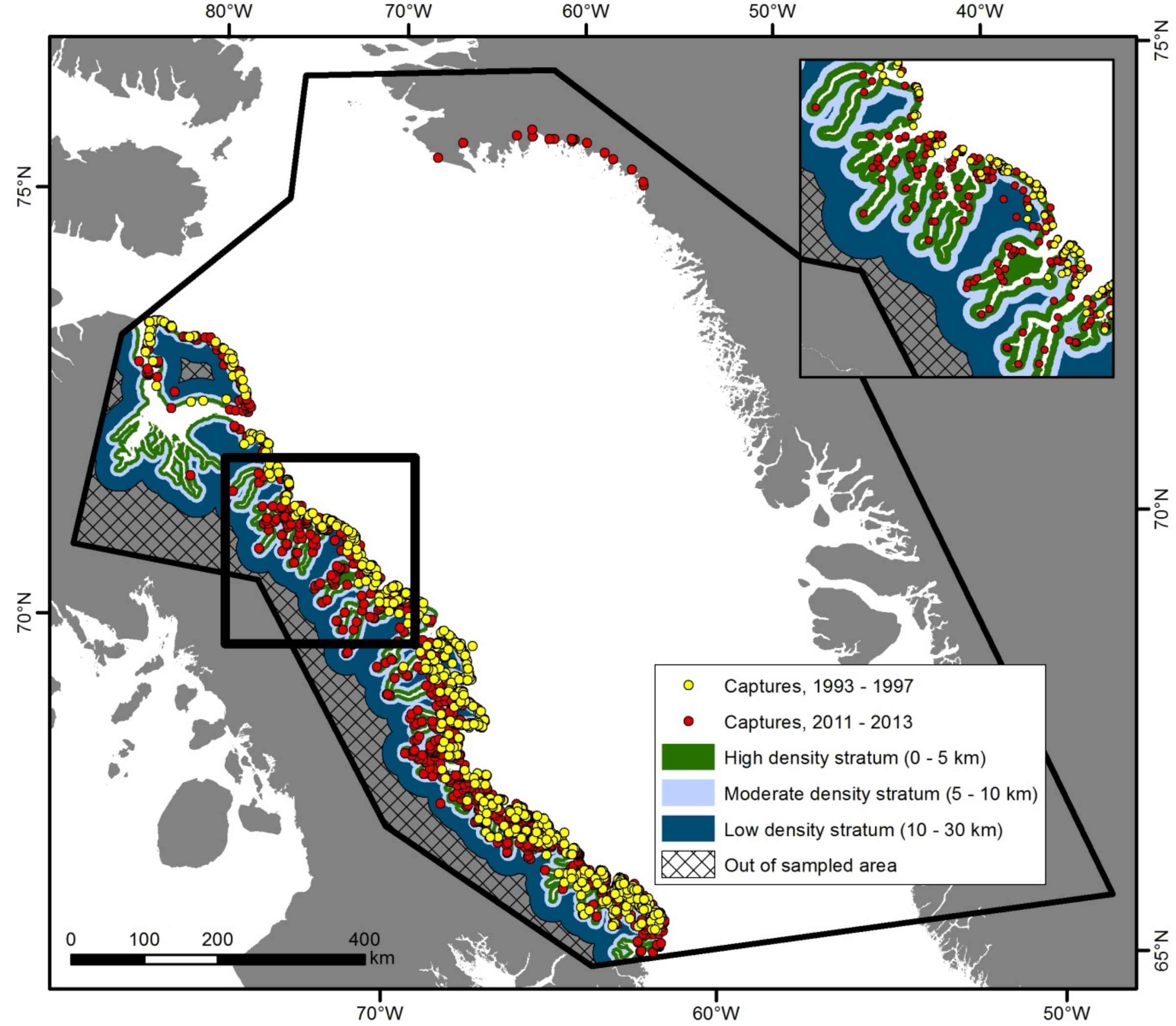


Figure 5.9. Locations of polar bears sampled in Baffin Bay during August – October, 1993 – 1995, 1997 and 2011 – 2013. The region bounded by the black square is enlarged in the inset. Note the absence of captures in fjords on Baffin Island and in northwestern Greenland during the 1990s.



CHAPTER 6

REPRODUCTIVE METRICS FOR MARK-RECAPTURE SAMPLED POLAR BEARS IN BAFFIN BAY

KEY FINDINGS

- We found little evidence of changes in litter size amongst COY or yearlings in BB. Annual variation in litter size was largely not significant and there were no trends over time or in association with spring transition date.
- We found evidence of a temporal trend in our index of COY recruitment (calculated as the number of COY per adult female in the MR sample) that was closely associated with variation in sea-ice conditions. From 1993-1995, 1997, 2011-2013, cub recruitment declined concurrent with a trend towards earlier spring sea-ice break-up. Similar declines in reproduction over time and in association with sea-ice conditions have been previously reported for polar bear subpopulations, but not in Baffin Bay.
- We found evidence of declines in body condition amongst bears in BB, following with previous studies that showed the same (Rode et al. 2012).
- Estimated annual recruitment (calculated as the number of yearlings per adult female in the MR sample) for BB during 1993-2013 ranged from 0.24 to 0.51, suggesting that BB continues to exhibit the level of reproduction requires for a viable population, according to the metric reported by Regehr et al. (2015), who suggest that variation in yearling recruitment may be a primary mechanism driving changes in population growth with values ranging from 0.1 to 0.3 necessary for population persistence.
- We found an association between COY recruitment in year t and yearling recruitment in year $t + 1$, as well as a strong association between COY litter size and yearling recruitment to following year. These findings suggest that a majority of mortality of polar bear cubs occurs during the first 8 months of life, such that recruitment of yearlings is heavily dependent on the number of COY that are born and survive up to the first ice-free period. Given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set.
- Although values in the 2000s tended to be lower than those observed in the 1990s, we did not find a significant temporal trend in annual yearling recruitment. However, given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set.
- Mean litter sizes in Baffin Bay and the proportions of COY and yearlings were similar to those observed in Foxe Basin and Southern Hudson Bay. In contrast, metrics for Baffin Bay were notably higher than those for estimated for Davis Strait and Western Hudson Bay. These comparisons suggest that BB remains a relatively productive subpopulation despite what appear to be recent declines in reproduction.

6.1. Introduction

For populations of large, long-lived mammals, changes in reproductive performance can be one of the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1981, 1987). In populations approaching carrying capacity, declines in reproductive performance are likely to occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring indices or metrics of reproduction may therefore provide a useful tool for detecting potential population trends that may warrant more in-depth study. This is especially true for populations in which cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring reproductive metrics may provide a form of surveillance that can be used to trigger more intensive study.

Reproductive metrics have been identified as an important component for monitoring polar bears across their circumpolar range (Vongraven et al. 2012). These metrics may be used to track long-term trends in the status of subpopulations, parameterize population viability models and support harvest risk assessments (Regehr et al. 2015). In particular, changes in reproduction are predicted to be amongst the first subpopulation-level effects of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Indeed, declines in reproduction have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (Regehr et al. 2007, Rode et al. 2010, Rode et al. 2014).

Changes in reproductive metrics can signal significant changes in subpopulation status of polar bears. However, observations of poor reproductive performance alone do not necessarily imply a decline in subpopulation status. Studies of several polar bear subpopulations have documented declines in reproduction associated with increases in abundance that may be the result of density dependence (Derocher 2005, Peacock et al. 2013). In other cases, variation in reproductive performance within or amongst subpopulations has been attributed to geographic or annual variation in biological productivity and prey availability (i.e., variable carrying capacity; Stirling and Øritsland 1995, Stirling and Lunn 1997, Stirling 2002, Rode et al. 2014). Information on reproduction must therefore be considered alongside other measures of subpopulation performance in-order to properly assess status.

The Baffin Bay (BB) subpopulation is part of the seasonal ice ecoregion as defined by Amstrup et al. (2008) in which sea ice melts almost entirely in the summer and bears are forced ashore for extended periods of time, during which they have no or reduced access to food. Baffin Bay has experienced a long-term reduction in sea-ice cover and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapter 3). As a result, bears are spending an increasing amount of time on land. Previous studies have documented a decline in body condition amongst BB bears in association with these trends in sea ice (Rode et al. 2012), and similar trends were found in the current study (see Chapter 7), however changes in reproductive metrics in BB have not been reported.

We summarized reproductive metrics for the Baffin Bay polar bear subpopulation using data collected during two periods of mark-recapture sampling from 1993 to 1997 (Taylor et al. 2005) and 2011 to 2013 (this report). Annual variation in reproduction was examined to assess trends over time and to evaluate the hypothesis that reproductive performance varied with sea-ice conditions. Because Baffin Bay has been infrequently monitored, we also sought to assess the utility of reproductive metrics as a surveillance tool for monitoring subpopulation status between periods of more in-depth demographic study, especially since some metrics can be obtained by methods that do not require physical capture (e.g., aerial surveys, harvest monitoring). Finally, we compared reproductive metrics for BB with other subpopulations to make inferences about the relative performance of this subpopulation. Results from these analyses provide context for understanding the status of BB polar bears.

6.2. Materials and Methods

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all Bylot Island (66.2°N to 73.8°N) in Nunavut/ Canada and parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al. 2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south. Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). Some bears remain on land in northwestern Greenland throughout the ice-retreat period.

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in BB, Canada. Sampling occurred along eastern Baffin and Bylot Islands during the ice-free season from late August to mid-October in 1993-95, 1997 and 2011-2013. Most bears in Baffin Bay move onto land on Baffin and Bylot Islands in late summer as the sea ice breaks up and remain on land until freeze-up in the late fall (Taylor et al. 2005). Sampling was extensive across this on-land study area during both periods (1990s and 2000s) of the study (Figure 6.1). The remote biopsy sampling in Greenland conducted in 2012 and 2013 were not included in this analyses.

Various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that a presumably relatively small proportion of the BB subpopulation summers in the Qimmusseriarsuaq / Melville Bay area of NW Greenland rather than moving with the retreating sea ice and summering on Baffin and Bylot Islands.

During the 1990s, bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2005). Data on the sex, estimated age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred by biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. We remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) from the air at a range of 10-20 feet above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males, e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B).

Reproductive Metrics

We calculated annual reproductive metrics that have been previously recommended (Vongraven et al. 2012) or used in studies of polar bears (e.g., Stirling et al. 1980, Derocher and Stirling 1995, Rode et al. 2010, Peacock et al. 2013, Regehr et al. 2015), including mean litter sizes (± SD) for cubs-of-the-year (COY) and yearlings. Because we did not have estimated ages for adult females sampled during 2011-2013, we calculated a pooled mean for each year rather than age-specific values. Recruitment indices were calculated as the total number of COYs or yearlings divided by the total number of adult females in the sample (Derocher and Stirling 1995, Regehr et al. 2015). Calculation of reproductive metrics considered only those COY and yearlings accompanying their mother at time of observation. During the 1990s, approximately 6% of yearlings were found to be independent of their mother during the ice-free period (GN unpublished data). These independent yearlings were not included in the calculation of yearling recruitment since we were not able to identify independent yearlings encountered during the 2000s sampling period with a known degree of accuracy.

Although Taylor et al. (2005) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island, recent examination of mark-recapture and telemetry data collected in the 1990s suggest sampling was spatially restricted to a portion of the BB subpopulation’s seasonal range and did not sample bears located farther inland or on the sea ice (GN unpublished data, Chapter 3). In contrast, sampling during 2011 – 2013 was more comprehensive and systematic on onshore areas, and the amount of un-searched sea ice during the sampling period was greatly reduced. To explore the potential impact of this difference in sampling between epochs on the calculation of reproductive metrics, we estimated and delineated the minimum extent of the sampling frame from the 1990s using capture locations in a GIS. We subset the 2000s data using this layer to create a dataset collected over the same sampling area in both time periods. We then recalculated reproductive metrics using this geographic subset. We expected that reproductive metrics calculated for the 2000s using the subset sampling area would reduce potential bias by adjusting for bears that may have been functionally missed by the limited geographic scope of sampling on land in the 1990s, given that satellite telemetry data indicated that onshore habitat use did not vary between epochs. However, we note that this geographic subset exercise would not correct for bears that may have been missed during the 1990s due to their location on the sea ice during the fall sampling period. Within season recaptures of individuals were excluded from both data sets for analyses.

Sea-ice Metrics

Based on findings from other studies (Stirling et al. 1999, Durner et al. 2004, 2006, 2009, Regehr et al. 2007, Rode et al. 2010, 2014), we hypothesized that timing of sea-ice break-up over the continental shelf (< 300m depth) of BB may influence the annual reproductive performance of polar bears as indexed from data collected during the ice-free period. We therefore calculated the annual sea-ice metric, spring transition date, as day of the year (1-365) when ice area over the continental shelf of BB reached 50% in the spring (see Chapter 2 for methodology). Whereas some researchers have used lower sea ice thresholds for studying polar bears (e.g., Cherry et al. 2013, Rode et al. 2014, Obbard et al. 2016), the exact threshold level is less important in Baffin Bay because the changes in sea-ice area during spring and fall occur quickly, such that relatively small differences in transition dates result from small changes in the threshold values (Laidre et al. 2015). For sea ice in Baffin Bay, spring transition date is also correlated with other sea-ice metrics including fall transition date (negative correlation) and the interval between spring and fall transitions (Laidre et al. 2015).

We examined annual variation, temporal trends, and relationships to sea ice for reproductive metrics. Similar to Rode et al. (2014), we did not include both time and sea ice in the same analyses since long-term trends in the spring transition date were well-established *a priori* (Laidre et al. 2015, this report) and these two parameters were correlated over the 7 years of sampling (Pearson coefficient = -0.846, $p = 0.017$). We also examined relationships between metrics for COYs and those for yearlings in the following year for periods when sampling occurred in successive years.

To examine temporal trends in reproductive metrics and associations with sea ice, we used weighted least squares regression (General Linear Model procedure), with number of litters sampled each year as the weighting variable. Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

6.3. Results

During mark-recapture sampling in 1993-1995, 1997, and 2011-2013, we sampled 251 family groups consisting of an adult female and 1-3 dependent COY (400 COY in total; Table 6.1). During this period we also sampled 152 family groups consisting of an adult female and 1-3 dependent yearlings (231 yearlings in total). The mean number of family groups sampled annually was 35.9 (range: 16-62) and 21.7 (range: 8-44) for COY and yearling families respectively. From these data we created a second subsampled dataset base on geographic location to exclude bears that were encountered outside the standardized sampling area as described previously. This geographic subset consisted of 191 COY family groups containing 1-3 dependent COY (300 COYs in total) and 105 yearling family groups containing 1-3 dependent yearlings (160 yearling in total; Table 6.2). The mean number of family groups sampled annually was 27.3 (range: 16-48) and 15.0 (range: 8-25) for COY and yearling families respectively. Annual reproductive metrics for these two datasets are presented in Tables 6.1 and 6.2.

Litter Size

There was annual variation in mean litter size amongst adult females with COYs but differences amongst years were non-significant for both the full (Kruskal-Wallis, $P = 0.153$) and geographic subset (Kruskal-Wallis, $P = 0.069$) data (Figure 6.2). For adult females with yearlings, differences in mean annual litter size were not significant for the full dataset (Kruskal-

Wallis, $P = 0.051$). For the subset data, differences in mean yearling litter size were significant (Kruskal-Wallis, $P = 0.035$) with both 1997 and 2013 having significantly lower litter sizes than 1994 and 1993 (Bonferonni correction for multiple comparisons, $P < 0.050$ in each case; Figure 6.3).

There were no statistically significant temporal trends in annual COY or yearling mean litter size from 1993 to 2013 (Table 6.3). Similarly, mean litter sizes were not associated with date of spring sea-ice transition.

Recruitment

COY recruitment showed a significant negative temporal trend from 1993 to 2013 for the geographic subset data, but not the full dataset (Table 6.3). COY recruitment also exhibited a positive association with date of spring sea-ice transition (Figure 6.4) for both the full and geographic subset data (i.e., later spring break-up was associated with higher COY recruitment). For yearlings, annual recruitment was not associated with either time or spring transition date (Figure 6.4).

There were 4 instances where sampling occurred over 2 successive years: 1993-94, 1994-95, 2011-2012, and 2012-2013. For these back-to-back years we compared reproductive metrics for COY (time t) to those of yearlings in the following year (time t+1). There was no relationship between annual mean litter size for COY in year t and yearling litter size in year $t + 1$ (Table 6.4; Figure 6.5). COY recruitment was positively associated with yearling recruitment the following year for the geographic subset data but not the full dataset. COY litter size was closely associated with yearling recruitment in the following year for both datasets (Table 6.4; Figure 6.5), such that higher mean COY litter size in year t resulted in higher yearling recruitment in year $t + 1$. We also examined the relationship between yearling reproductive metrics and spring ice transition date the previous year finding no association for either the full ($F_{1,6} = 0.128$, $r^2 = 0.025$, $P = 0.735$) or subset data sets ($F_{1,6} = 0.095$, $r^2 = 0.019$, $P = 0.771$).

6.4. Discussion

Calculating annual reproductive metrics from mark-recapture field data is subject to several potential sources of error, including non-random sampling with respect to the overall study subpopulation. Similar to other seasonally ice-free subpopulations (e.g., Derocher and Stirling 1990), polar bears in Baffin Bay are known to exhibit a degree of spatial segregation by sex, age class and reproductive status with respect to the use of terrestrial habitat during the ice-free period (Ferguson et al. 1997; Chapter 4). For example, adult females with cubs tend to select fjords and avoid offshore islands and coastal regions where densities of adult males are higher (Ferguson et al. 1997). Pregnant bears select inland and upland denning habitats (Chapter 3). Mark-recapture sampling in the 1990s was more restricted in geographic extent relative to the 2010s (Chapter 3). The extent to which this difference in sampling between the two time periods introduced error and more importantly, systematic bias into our estimates of reproduction, cannot be fully evaluated. However, sampling bias is unlikely to account for the results of our analyses for several reasons. First, we attempted to account for differences in sampling by restricting some analyses to data collected within a standardized sampling area. This made little difference to the results. Results based on full and subset data were very similar. Second, sampling bias between the 1990s and 2000s would not account for the temporal trends in reproduction or associations with the timing of spring sea-ice transition unless sampling bias varied in proportion to these factors; something that is unlikely. Third, under-sampling of fjord habitats in the 1990s may have introduced bias in sampling of adult females with offspring. However, the most likely impact of this bias would have been underestimation of recruitment indices since fewer adult females with offspring would have been sampled relative to adult females overall. Fourth, sampling bias would not account for the association observed between reproductive metrics in successive years and the closeness of this association in some cases.

Another source of error in estimation of reproductive parameters in our study originated from the misclassification of bears that were observed from the air rather than handled during 2011-2013. However, when combined with genetic sexing, the accuracy of this method of classification is high even for lone adult females[1] (Appendix B). Furthermore, this source of error does not necessarily introduce systematic bias. Lone adult females could only have been misclassified as lone subadult females (and vice-versa), and there is no evidence to suggest inaccuracy in this area favours one age-class versus the other.

Finally, analyses of telemetry data collected from collared adult females suggest that a proportion of collar bears remained on the remnant sea ice in some years during the 1990s and were unavailable for sampling (Chapter 3). Although this could have introduced bias if certain classes of bears tended to remain on the ice while others moved to shore, we were unable to correct for this potential source of bias. However, the most likely effect would have been over-representation of lone (pregnant) adult females in our sampling data since this is the class of adult females that has an obligate need to come ashore to look for suitable denning habitat. This, in turn, would have led to underestimation of recruitment in the 1990s relative to the 2000s.

Litter Size

---

[1] Approximately 84% of lone adult females were correctly classified (GN unpublished data; see chapter 5, appendix 1.)

We found little evidence of changes in litter size amongst COY or yearlings in BB. Annual variation in litter size was for the most part not significant and there were no trends over time or in association with spring transition date. This finding is consistent with Molnar et al. (2011), who found that although litter size is predicted to vary in response to changes in maternal body condition and environmental conditions, it is a relatively insensitive reproductive metric. Large changes in maternal condition and environment are necessary to produce statistically significant differences in litter size. Litter size does, however, remain an important reproductive metric for monitoring polar bear subpopulations (Vongraven et al. 2012). Changes in litter size have been associated with temporal and geographic variation in ecosystem productivity (Stirling and Lunn 1997, Peacock et al. 2013), and long-term trends have been detected in association with changing subpopulation status (Derocher and Stirling 1995).

Trends in COY Recruitment and Association with Sea-ice Conditions

We found evidence of a temporal trend in our index of COY recruitment that was also closely associated with variation in sea-ice conditions. From 1993 to 2013, cub recruitment declined concurrent with a trend towards earlier spring sea-ice break-up. Similar declines in reproduction over time and in association with sea-ice conditions have been previously reported for polar bear subpopulations (Derocher and Stirling 1995, Derocher 2005, Rode et al. 2010, Peacock et al. 2013, Rode et al. 2014), but not in Baffin Bay. Earlier spring break-up (also associated with later fall freeze-up) presumably decreases feeding opportunities for polar bears, thereby resulting in poorer maternal body condition and reduced investment in reproduction. This, in turn, will be manifested as reduced natality rates and / or lower offspring survival. Our index of cub recruitment incorporates both of these parameters reflecting to an unknown degree a blend of decreased cub production and lowered cub survival over the first 8 months of life. The association between cub recruitment and spring transition date in our study suggests that lower cub survival from birth to the first ice-free season may be a primary mechanism driving lower reproduction in Baffin Bay. However, we have not demonstrated a causal relationship; other factors may play an important role in cub recruitment, particularly since recruitment was associated with both time (year) and spring transition date. These two parameters are correlated with one another and may also be associated with other parameters that we did not consider. As such, there is some uncertainty as to the extent to which declining reproduction in BB is mediated by sea-ice conditions.

Declining reproduction and body condition are amongst the first subpopulation level effects predicted occur in polar bears as a result of climate change (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Our evidence of a decline in reproduction in BB from 1993 to 2013 is accompanied by evidence of concurrent declines in body condition amongst bears in this subpopulation over the same period (Rode et al. 2012, Chapter 6). These changes may signal a reduction in the carrying capacity of BB. Although the point estimate of abundance from our recent genetic mark-recapture was higher than the 1990s estimate, the difference between estimates was not statistically significant (Chapter 5). Additionally, differences in these point estimates may be largely explained by differences in sampling design between the two time periods. Regardless of whether density effects are at play, if the observed association between sea ice and reproduction is real and the well documented trend in sea-ice continues, it is reasonable to expect that this subpopulation will experience significant changes in reproductive performance as a result of declining habitat (Laidre et al. 2015).

Yearling Recruitment

Recruitment calculated as the number of yearlings per adult female has been identified as an important reproductive metric to monitor in polar bear subpopulations, incorporating both natality and survival of COY (Vongraven et al. 2012). Regehr et al. (2015) suggest that variation in yearling recruitment may be a primary mechanism driving changes in subpopulation growth, with values ranging from 0.1 to 0.3 necessary for subpopulation persistence. Estimated annual recruitment values for Baffin Bay during 1993 to 2013 ranged from 0.24 to 0.51, suggesting that BB continues to function as a viable subpopulation, according to this metric. Interestingly, in contrast to previous studies (e.g., Rode et al. 2010, 2014), we did not find evidence that yearling recruitment was associated with sea-ice conditions. Instead, our results suggest that recruitment of yearlings is largely determined by reproductive metrics for COYs in the previous year. We found an association between COY recruitment in year t and yearling recruitment in year t + 1, as well as a strong association between COY litter size and yearling recruitment to following year. These findings suggest that a majority of mortality of polar bear cubs occurs during the first 8 months of life, such that recruitment of yearlings is heavily dependent on the number of COY that are born and survive up to the first ice-free period. Since COY recruitment itself appears to be associated with spring transition date, yearling recruitment up to the ice-free period may be influenced to a greater degree by ice conditions the previous year than by ice conditions in the current year. We did not find evidence of this lag effect but our sample size was very small.

Although values in the 2000s tended to be lower than those observed in the 1990s, we did not find a significant temporal trend in annual yearling recruitment. However, given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set. In this context, we suggest that monitoring annual litter size of COY may be a useful tool for tracking trends in recruitment in the absence of more intensive subpopulation studies. This metric can be readily acquired from aerial surveys without capture or biopsy of bears, without the need for extensive observer experience in identifying age-sex classes, and at relatively low cost. Our very limited data suggest that mean annual COY litter size is closely related to yearling recruitment the following year. However, we acknowledge that the robustness of this relationship has not been validated with a larger data set and under a range of environmental conditions.

Comparison with other Subpopulations

Indices of reproduction for BB were comparable to other polar bear subpopulations in the seasonal ice ecoregion (Amstrup et al. 2008) that have been recently studied by mark-recapture or aerial survey (Table 6.5). Mean litter sizes in Baffin Bay and the proportions of COY and yearlings were similar to those observed in Foxe Basin (Stapleton et al. 2016) and Southern Hudson Bay (Obbard et al. 2015); two subpopulations classified as stable (PBSG 2010). In contrast, metrics for Baffin Bay were notably higher than those estimated for Davis Strait (Peacock et al. 2013) and Western Hudson Bay (Lunn et al. 2014). These comparisons suggest that BB remains a relatively productive subpopulation despite apparent recent declines in reproduction.

6.5. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Cavalieri, D. J., C. L. Parkinson, P. Gloersen, and H. J. Zwally. 1996, updated yearly. Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS passive microwave data, Version 1. NASA National Snow and Ice Data Center Distributed Active Archive Center, Boulder, Colorado, USA, doi: http://dx.doi.org/10.5067/8GQ8LZQVL0VL.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. Population Ecology 47:267–275.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1990. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395–1403.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Durner, G. M., S. C. Amstrup, R. Nielson, and T. McDonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea.

Pages 107–120 *in* S. Huzurbazar, editor. Resource selection methods and applications. Omnipress, Madison, Wisconsin, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use of polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Fowler, C. W. 1981. Density dependence as related to life history strategy. Ecology 62:602–610.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. Peacock, M. K. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø. Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Stapleton, S., S. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., and N. J. Lunn. 1997. Environmental fluctuations in arctic marine ecosystems as reflected by variability in reproduction of polar bears and ringed seals. Pages 167–181 *in* S. J. Woodin, and M. Marquiss, editors. Ecology of arctic environments. Special Publication No. 13 of the British Ecological Society, Blackwell Science Ltd., Oxford, UK.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Table 6.1. Reproductive metrics for annual mark-recapture sampling data from Baffin Bay. Full dataset.

Year	**Mean Litter Size (n, SD)**		**Recruitment Index[2] (Offspring/adult female)**	
	COY	**Yearling [1]**	**COY**	**Yearlings [1]**
1993	1.63 (16, 0.50)	1.75 (8, 0.46)	0.79	0.42
1994	1.87 (23, 0.51)	1.82 (11, 0.38)	0.83	0.38
1995	1.52 (21, 0.51)	1.64 (14, 0.63)	0.71	0.51
1997	1.64 (25, 0.49)	1.25 (12, 0.45)	0.65	0.24
2011	1.57 (58, 0.50)	1.53 (34, 0.51)	0.68	0.39
2012	1.47 (62, 0.50)	1.55 (44, 0.54)	0.47	0.35
2013	1.65 (46, 0.49)	1.34 (29, 0.49)	0.60	0.31

[1] Dependent yearlings only. Capture data for 1993-97 suggests up to 6% of yearlings are independent of their mother in the fall

[2] Calculated per Derocher and Stirling (1995), Vongraven et al. (2012), Regehr et al. (2015)

Table 6.2. Reproductive metrics for annual mark-recapture sampling data from Baffin Bay. Data for 2011-13 were filtered to exclude individuals encountered outside the area where sampling was estimated to have occurred in the 1990s.

Year	Mean Litter Size (n, SD)		Recruitment Index[2] (Offspring/adult female)	
	COY	Yearling [1]	COY	Yearlings [1]
1993	1.63 (16, 0.50)	1.75 (8, 0.46)	0.79	0.42
1994	1.87 (23, 0.51)	1.82 (11, 0.38)	0.83	0.38
1995	1.52 (21, 0.51)	1.64 (14, 0.63)	0.71	0.51
1997	1.64 (25, 0.49)	1.25 (12, 0.45)	0.65	0.24
2011	1.48 (33, 0.51)	1.60 (20, 0.50)	0.61	0.40
2012	1.42 (48, 0.50)	1.48 (25, 0.51)	0.55	0.30
2013	1.64 (25, 0.49)	1.27 (15, 0.46)	0.55	0.26

[1] Dependent yearlings only. Capture data for 1993-97 suggests up to 6% of yearlings are independent of their mother in the fall

[2] Calculated per Derocher and Stirling (1995), Vongraven et al. (2012), Regehr et al. (2015)

Table 6.3. Results of least squares regressions for annual reproductive metrics derived from mark-recapture sampling of polar bears during the ice-free period (August-October) in Baffin Bay. Number of litters was used as a weighting variable. Significant relationships in bold.

Reproductive Metric	Litter Age	Explanatory Variable	Dataset	$F_{1,6}$	r^2	P
Litter Size	COY	Year	Full	1.64	0.25	0.256
Litter Size	COY	Year	Subset	3.37	0.40	0.126
Litter Size	COY	Spring Ice Transition	Full	3.42	0.41	0.124
Litter Size	COY	Spring Ice Transition	Subset	3.92	0.44	0.105
Litter Size	Yearling	Year	Full	1.22	0.20	0.319
Litter Size	Yearling	Year	Subset	1.16	0.19	0.331
Litter Size	Yearling	Spring Ice Transition	Full	1.64	0.25	0.256
Litter Size	Yearling	Spring Ice Transition	Subset	1.38	0.22	0.293
Recruitment	COY	Year	Full	5.34	0.52	0.069
Recruitment	COY	Year	Subset	22.43	0.82	**0.005**
Recruitment	COY	Spring Ice Transition	Full	53.90	0.92	**0.001**
Recruitment	COY	Spring Ice Transition	Subset	11.60	0.70	**0.019**
Recruitment	Yearling	Year	Full	0.72	0.13	0.434
Recruitment	Yearling	Year	Subset	1.37	0.21	0.295
Recruitment	Yearling	Spring Ice Transition	Full	0.64	0.11	0.460
Recruitment	Yearling	Spring Ice Transition	Subset	1.91	0.28	0.225

Table 6.4. Results of least squares regressions for annual reproductive metrics derived in year *t* and year *t+1* from mark-recapture sampling of polar bears during the ice-free period (August-October) in Baffin Bay, Canada. Number of litters in year *t+1* was used as a weighting variable.

Reproductive Metric (year t)	**Reproductive Metric (year t+1)**	**Dataset**	$F_{1,3}$	r^2	P
COY Litter Size	Yearling Litter Size	Full	0.80	0.29	0.465
COY Litter Size	Yearling Litter Size	Subset	0.92	0.29	0.431
COY Recruitment	Yearling Recruitment	Full	3.88	0.66	0.188
COY Recruitment	Yearling Recruitment	Subset	20.33	0.91	**0.046**
COY Litter Size	Yearling Recruitment	Full	474.43	0.99	**0.002**
COY Litter Size	Yearling Recruitment	Subset	1854.94	0.99	**0.001**

Table 6.5. Comparison of reproductive metrics for polar bear subpopulations in the seasonal ice ecoregion. Sampling occurred during ice-free periods.

Subpopulation	Mean Litter Size		Proportion of Total Observations[2]		Source
	COY	Yearling	COY	Yearlings	
Baffin Bay (1993-97)	1.67	1.60	0.16	0.09	Taylor et al. (2005)
Baffin Bay (2011-13)[1]	1.55	1.48	0.15	0.09	This study
Davis Strait (2005-07)	1.49	1.22	0.08	0.09	Peacock et al. (2013)
Foxe Basin (2009-2010)	1.55	1.48	0.13	0.10	Stapleton et al. (2016)
Southern Hudson Bay (2011)	1.56	1.49	0.16	0.12	Obbard et al. (2015)
Western Hudson Bay (2011)	1.43	1.22	0.07	0.03	Stapleton et al. (2014)

[1] Based on sampling across study area

[2] Some of these recent studies relied on aerial survey which is less accurate in identifying adult females (without genotyping to determine sex). For this reason we used published data on mean litter sizes and the proportion of COY and yearlings within these studies rather than calculating indices of recruitment used in the present study

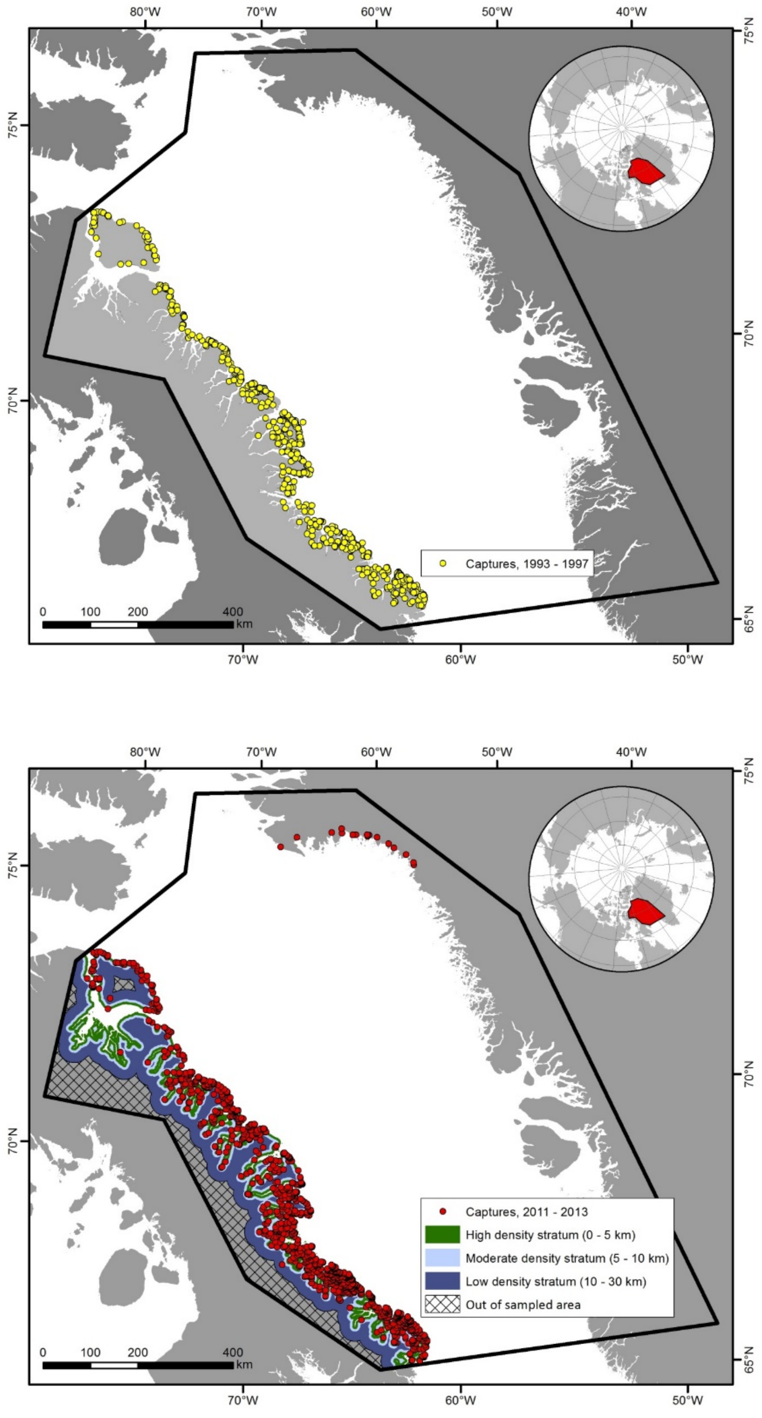
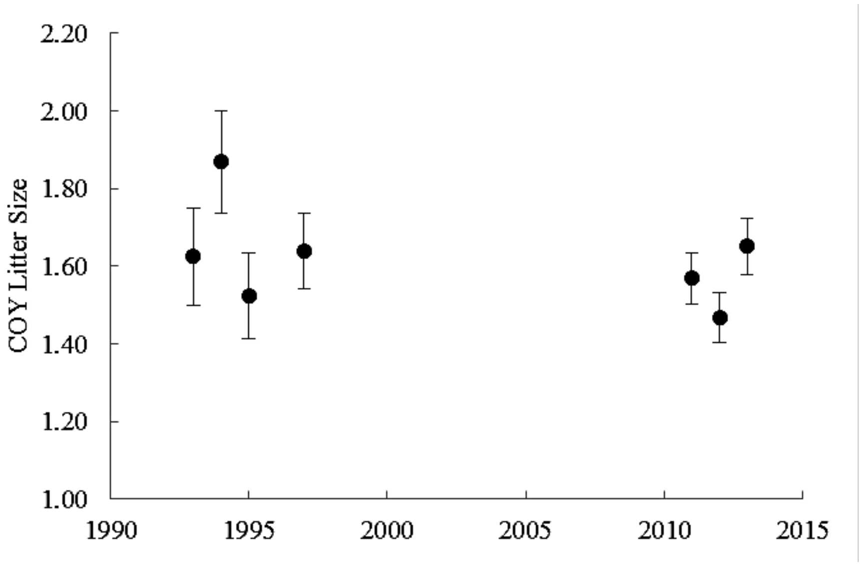


Figure 6.1. Spatial distribution of bears recorded during sampling in the Baffin Bay polar bear subpopulation, 1993 – 1995, 1997 (top), and 2011 – 2013 (bottom).

(a)



(b)

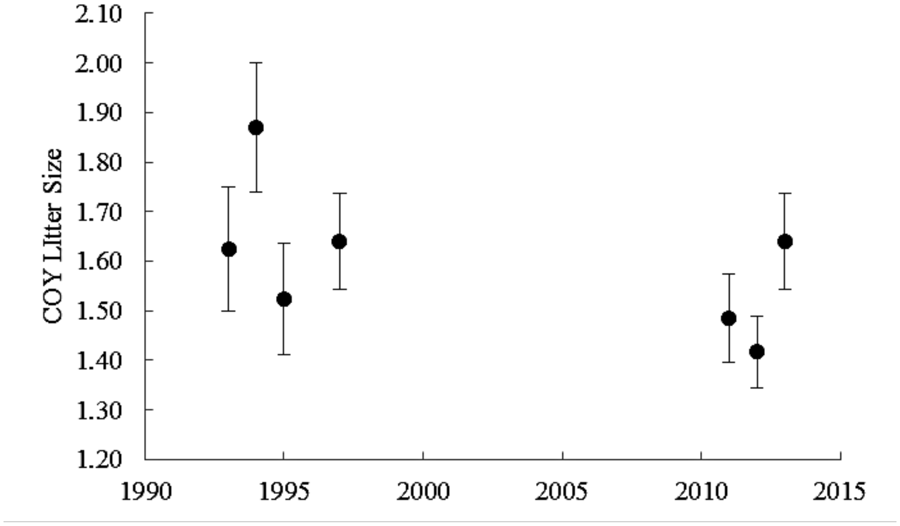
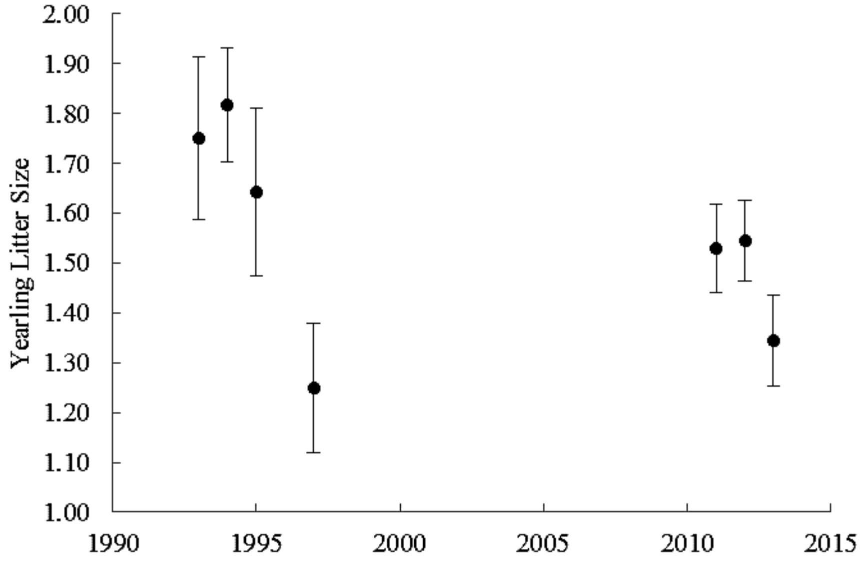


Figure 6.2. Mean litter sizes (+/- SE) for adult females with cubs-of the year (COY) during the ice-free period in Baffin Bay. Calculated using the (a) full and (b) geographically subset data (see text).

(a)



(b)

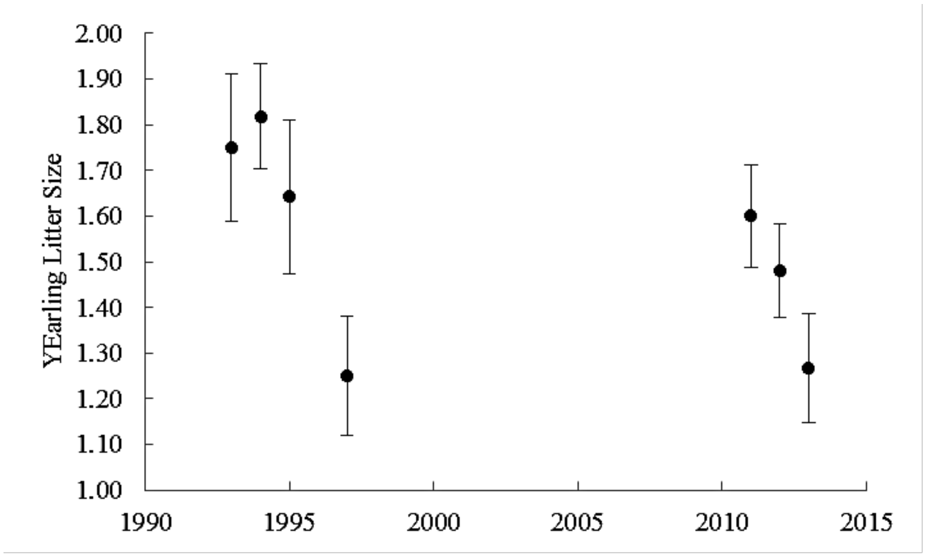
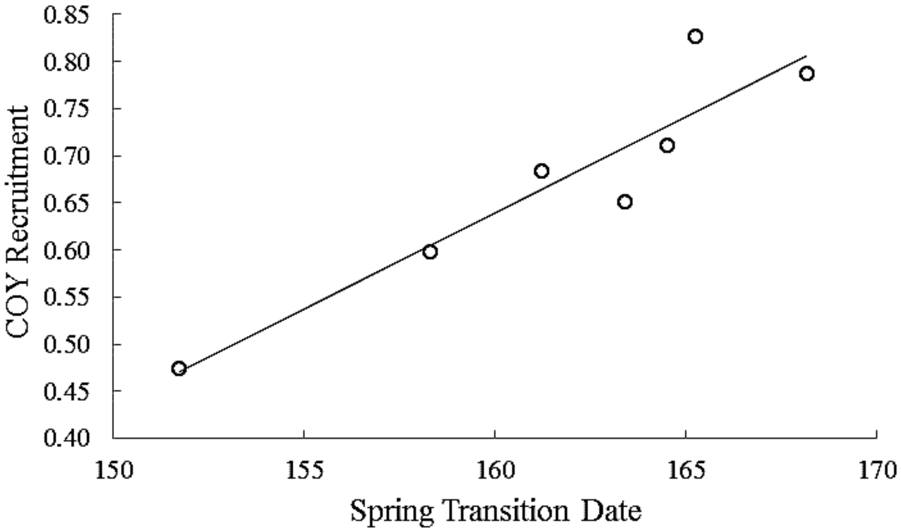


Figure 6.3. Mean litter sizes (+/- SE) for adult females with yearlings during the ice-free period in Baffin Bay. Calculated using the (a) full and (b) geographically subset data (see text).

(a)



(b)

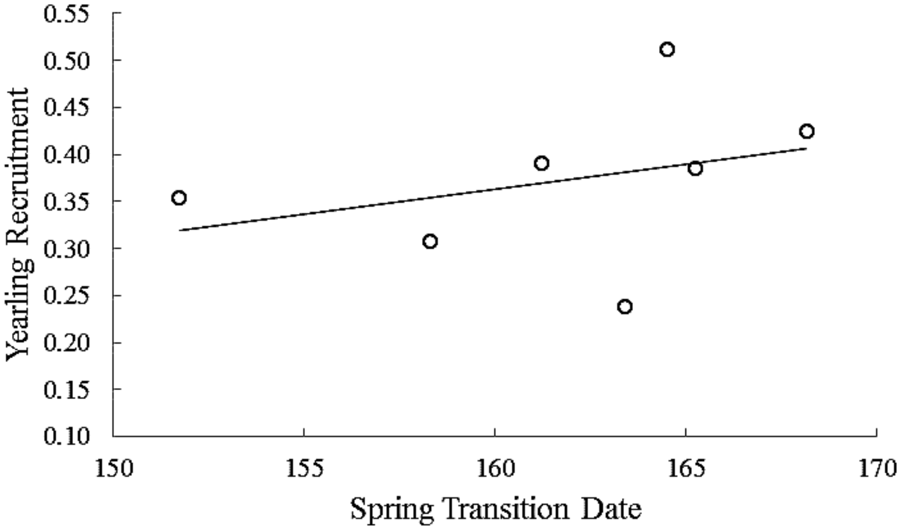


Figure 6.4. The relationship between spring transition date and annual recruitment index for (a) cub-of-the-year and (b) yearling polar bears during the ice free period (August-October), in Baffin Bay. Recruitment calculated as the number of COY or yearlings per adult female in the sample subpopulation using the full mark-recapture dataset.

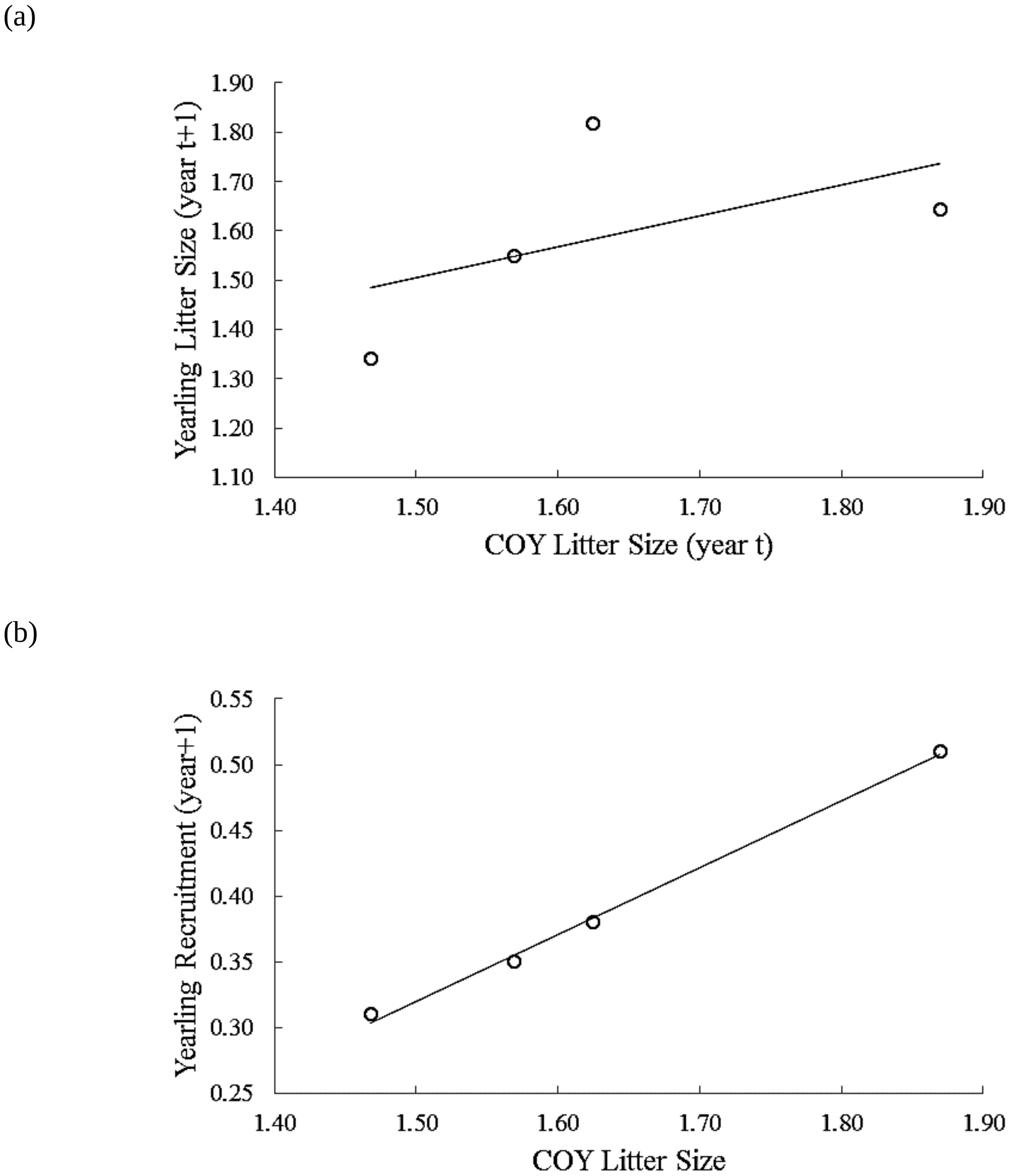


Figure 6.5. The relationship between mean annual litter size amongst cub-of-the-year litters and (a) yearling litter size and (b) yearling recruitment in the following year (year +1). Data are for polar bears sampled during the ice free period (August-October), in Baffin Bay. Metrics calculated using full mark-recapture dataset.

CHAPTER 7

BODY CONDITION OF BAFFIN BAY POLAR BEARS

KEY FINDINGS

- Fatness index (FI) scores were collected during two periods of mark-recapture sampling in BB, 1993-1995, 1997 and 2011-2013. We examined trends in this metric of body condition across both sampling periods in relation to sea-ice conditions.
- We found evidence of declines in body condition amongst bears in BB over the period 1993 to 2013. Body condition in BB polar bears declined in close association with the ice-free period and spring sea-ice transition dates. This is consistent with the hypothesis that reduced time on the sea ice is a primary mechanism driving this decline.
- Our results follow with previous studies that showed similar results through 2010 with different metrics derived from physical handing of bears (Rode et al. 2012).
- The springtime aerial survey was successfully implemented due to the small geographic These findings are consistent with available traditional knowledge suggesting that body condition of polar bears in BB was poorer in the early 2000s relative to the 1990s (Dowsley and Wenzel 2008, Born et al. 2011).
- We found evidence of recent foraging in approximately 9% of the BB bears observed during the on-land period. Marine mammals, in particular seals and walrus made up almost half of the identifiable food sources.

7.1. Background

For populations of large, long-lived mammals changes in body condition will be among the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1987, 1990, Zedrosser et al. 2006). In populations approaching K, declines in condition will occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring body condition may therefore provide a useful tool for the early detection of population trends that warrant more in-depth study. This is especially true for populations where cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring condition may provide a form of surveillance that can be used to trigger periods of more intensive study.

The annual life-cycle of polar bears (*Ursus maritimus*) is characterized by large seasonal changes in body condition (Watts and Hansen 1987, Ramsay et al. 1992, Ramsay and Stirling 1988, Atkinson and Ramsay 1995). Throughout most of their circumpolar range, bears are thought to gain condition during the spring and early summer when juvenile seals are abundant and relatively susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). This period of hyperphagia is followed by a scarcity of food in the late summer and fall when sea ice reaches a minimum throughout the Arctic. During this season, bears in some regions are forced onto land by the melting sea ice where access to seals and other marine mammal prey is greatly reduced (Stirling et al. 1977, Derocher and Stirling 1990a,b, Ramsay et al. 1991). In other regions, bears remain on off-shore pack-ice but likely also have reduced access to and/or less success in catching seals (Amstrup et al. 2000, Stirling 2002, Atwood et al. 2015, Rode et al. 2015). Some individuals utilize terrestrial foods sources during the summer and fall. However, the extent to which this occurs and the significance of terrestrial foods to energy budgets remains the subject of on-going debate and research (e.g., Derocher et al. 1993a, Hobson et al. 2009, Gormezano and Rockwell 2013, 2015, Rode et al. 2015, Tartu et al. 2016). Regardless, it is well documented that many bears especially those in the seasonal ice ecoregion like Baffin Bay (Amstrup et al. 2008) rely on nutrients and energy stored within adipose and other body tissues to meet a significant portion of their maintenance requirements for survival during this period. Consequently, body condition amongst most individuals declines progressively through the summer and fall until access to sea ice increases in the late fall and early winter (Atkinson and Ramsay 1995, Atkinson and Ramsay 1996, Derocher and Stirling 1995, Rode et al. 2011, Obbard et al. 2016). Additionally, adult females rely on body stores to support reproductive activities. Mothers with cubs continue to lactate during the summer and fall if their condition and/ or available food is sufficient (Derocher et al.1993b, Derocher and Stirling, 1996). Pregnant females enter dens in the fall where they rely exclusively on body stores to support gestation and early to mid-lactation over a period of 6-8 months (Atkinson and Ramsay 1995, Derocher and Stirling 1995). Given this dynamic cycle of feeding and fasting, body condition attained during the spring and early summer is expected to exert a significant influence on the survival, reproductive performance and thus status of polar bear subpopulations (Atkinson and Ramsay 1995, Derocher and Stirling 1995, 1996, Molnar et al. 2010, Molnar et al. 2011).

Tracking long-term trends in body condition has been identified as an important component of the monitoring scheme for polar bears across their circumpolar range (Vongraven et al. 2012, Patyk et al. 2015). In the absence of more intensive studies, simple body condition metrics may be useful indices for monitoring subpopulations and detecting responses to changing environmental conditions (Amstrup et al. 2006, Stirling et al. 2008a, Vongraven et al. 2012). Of particular concern, changes in body condition are predicted to be amongst the first subpopulation-level impacts of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Wiig et al. 2008, Stirling and Derocher 2012). Indeed, declines in condition have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016). Although these trends in body condition can signal significant changes, observations of declining condition alone do not necessarily imply a decline in subpopulation status. Studies of some polar bear subpopulations have documented declines in condition in association with increases in abundance that may be the result of density dependence (Stirling et al. 1999, Rode et al. 2011, Peacock et al. 2013).

Information on body condition must therefore be interpreted alongside other measures of subpopulation performance in-order to properly assess status.

A variety of quantitative and qualitative body condition indices have been used on polar bears including body weight estimated from girth (e.g., Derocher and Stirling 1995, Rode et al. 2011), body mass indices standardized for length (e.g., Stirling et al. 1999, Cattet et al. 2002, Obbard et al. 2016), skull width (Rode et al. 2010, 2011), percent body fat determined by isotopic dilution or bioelectrical impedance analysis (Atkinson and Ramsay 1995, 1996, McKinney et al. 2014), percent lipid content of adipose tissue biopsies (Stirling et al. 2008b, McKinney et al. 2014) and a visually assigned fatness index (Amstrup et al. 2006, Stirling et al. 2008a,b). Most of these condition indices require the handling of bears to collect measurements. However, the fatness index (FI) and potentially the lipid content of adipose tissue (Pagano et al. 2014, McKinney et al. 2014) may be obtained without handling thus making them suitable for use in subpopulations monitored by less invasive methods such as aerial survey or genetic mark-recapture.

The Baffin Bay (BB) subpopulation is part of the seasonal ice-free ecoregion as defined by Amstrup et al. (2008) in which sea ice melts almost entirely in the summer and bears are forced ashore for an extended period of time, during which they have no or reduced access to food. Baffin Bay has experienced a long term reduction in sea-ice cover and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015). As a result, bears are spending an increasing amount of time on land. Examining data on the morphometric measurements of BB polar bears (girth, length and skull width) for the period 1977 to 2010, Rode et al. (2011) detected a decline in body condition concurrent with declining sea-ice cover. However, geographically restricted sampling and uncertainty about trends in subpopulation density during the latter years of this study limited the interpretation of these findings and conclusions regarding subpopulation status (York et al. 2016).

Here we summarize information on the body condition of polar bears in BB using a different measure of condition; the fatness index (FI). FI scores were collected during two periods of mark-recapture sampling in Baffin Bay from 1993 to 1997 and 2011 to 2013. During the latter period of sampling bears were surveyed by genetic mark-recapture using biopsy darts. Because biopsy darted bears were not handled our collection of body condition data was limited to visually assigned FI scores only. The FI has been validated as a measure of condition in polar bears, being closely correlated with more quantitative condition indices (Stirling et al. 2008b, McKinney et al. 2014) and other biological factors (e.g., Henricksen et al. 2001, Amstrup et al. 2006). Our study examined body condition using a different index of condition collected over a different (albeit overlapping) temporal and spatial sampling frame to that of Rode et al. (2011). We examined trends in condition in relationship to sea ice. During part of this study, we also collected information on the foraging habits of BB polar bears to assess the range of food sources utilized by bears during the ice-free period. Our results provide supplementary information for interpreting the results of the recent genetic mark-recapture in BB and for understanding the present status of this subpopulation.

7.2. Materials and Methods

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and Bylot Island (66.2°N to 73.8°N) in Nunavut/Canada and parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al.

2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south (Figure 7.1). Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Historically sea ice also remained in Melville Bay, NW Greenland (Born 1995). Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A presumably small number of bears remain on land in northwestern Greenland throughout the ice-retreat period (Born 1995, this study).

Field Sampling

Data for the study were collected during two periods of systematic mark-recapture sampling on Baffin Island in Baffin Bay. Sampling occurred along eastern Baffin and Bylot Islands during the ice-free season from late August to mid-October in 1993-95, 1997 and 2011-2013. Most bears in Baffin Bay move onto land on Baffin Island and Bylot in late summer as the sea ice breaks up and remain on land until freeze-up in the late fall (Taylor et al. 2005). Sampling was spatially extensive across this on-land study area during all years (Figure 7.1) although there were some noted differences in sampling strategy between the two periods (1990s and 2000s) of the study (Chapter 3).

Using helicopters we searched for bears across the study area. During the 1990s, bears were sampled by physical capture on Baffin Island and examination using methods previously described (Taylor et al. 2005). Data on the sex, estimated age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred by remote biopsy darting (Pagano et al. 2014) and subsequent genetic analysis to determine genetic sex and identify individuals. We remotely estimated sex, age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) and reproductive status from the air at a range of 3-10 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B).

During both sampling periods, all encountered bears were assigned a FI score on a scale of 1-5 where 1 and 5 represent the leanest and most obese bears, respectively (Stirling et al. 2008a).

During the 1990s, this score was based on physical examination of captured bears. For bears in the 2000s, FI scores were assigned based on examination from the air at a distance of 3-7 m above ground. Additional information collected for all bears at the time of observation included the identity of the observer, date, and location (coordinates).

Additionally various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that small, albeit unknown, proportion of the BB subpopulation summers in the Qimmusseriarsuaq / Melville Bay area of NW Greenland rather than moving with the retreating sea ice and summering on Baffin and Bylot Islands. For comparative reasons estimates of body condition index from bears biopsy dated in NW Greenland are presented separately from that on Baffin Island. During 1-14 September 2012 and 2013 a total of 20 biopsies (2012: 6, 2013: 14) were sampled on land and along glacier fronts in the Melville Bay area using the same methods described above in Nunavut. During sampling the FI index was scored for each bear by three observers with extensive experience in judging body condition of polar bears both during examination from the air and during subsequent physical handling. The sex of all biopsied individuals was determined genetically post hoc.

Body Condition Scoring

Bears were initially scored according to the standard FI on a scale of 1 to 5 (Stirling et al. 2008b). This scoring system was subsequently simplified to a 3 point scale of poor (FI = 1 or 2), fair (FI = 3) and good (FI = 4 or 5) condition; hereafter termed the Body Condition Score (BCS). Similar modifications of the FI for polar bears have been employed in other studies to facilitate analyses (Stirling et al. 2008a) or have been recommended for use in general monitoring schemes for polar bears (Vongraven et al. 2012). In our case, this refinement was made in part due to the lower frequencies of bears scored as 1 and 5, but also to address concerns about potential bias. The assumption was made that a simplified scale would be subject to less bias resulting from different observers and / or distance from bear at time of scoring. Experienced observers should be able to discriminate a bear in poor, fair or good condition even at distances of up to 7 m.

Bears coming off the ice in summer are thought to be at or near their annual peak in body condition having recently gone through a period of hyperphagia when juvenile seals are relatively abundant and susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). For our analyses, therefore, we calculated the proportion of bears rated as being in good condition (BCS = 3) as a metric of annual body condition within each sex and age class. Repeated observations of the same individual (as identified by physical mark or genotype) within a given year were excluded from the analyses. Observations of the same individual in different years were included. Similar to Stirling et al. (2008a), we assumed that observations of the same individual in different years were statistically independent given the dynamic nature of body condition in polar bears (Watts and Hansen 1987, Atkinson and Ramsay 1995, 1996) and it's response to annual variation in environmental conditions.

Sea-ice Metric

Based on findings from other studies (Stirling et al. 1999, Durner et al. 2004, 2006, 2009, Regehr et al. 2007, Rode et al. 2010, 2014), we hypothesized that timing of sea-ice break-up over the continental shelf (< 300m depth) of BB may influence the body condition of polar bears during the ice-free period. We therefore calculated the annual sea-ice metric, spring transition date, as day of the year (1-365) when ice area over the continental shelf of BB reached 50% in the spring (Chapter 4). We used *Sea-ice Concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data* (Cavalieri et al., 1996) available from the National Snow and Ice Data Center (NSIDC) in Boulder, Co (See Laidre et al. 2015 Appendix S1 for additional details). While some researchers have used lower ice cover thresholds for studying polar bear relationships to sea ice (e.g., Cherry et al. 2013, Rode et al. 2014, Obbard et al. 2016), Laidre et al. (2015) reported that spring transition dates in Baffin Bay were not sensitive to the choice of threshold because usually the decrease of sea-ice area in the spring and the increase of sea-ice area in the fall proceed relatively quickly. A small change in the threshold results in a small change in the transition dates (Laidre et al. 2015). For sea ice in Baffin Bay, spring transition date is also correlated with other sea-ice metrics including fall transition date (negative correlation) and the interval between spring and fall transitions (Laidre et al. 2015).

We examined annual variation in our body condition metric, trends over time, and relation to spring sea-ice transition date. Similar to Rode et al. (2014), we did not include both time and sea ice in the same analyses since long term trends in our sea-ice metric were well established *a priori* (Laidre et al. 2015, Chapter 4) and these two parameters were closely correlated over the 7 years of sampling (Pearson coefficient = -0.846, $P = 0.017$). Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

Foraging Observations

Observations of bears feeding or evidence that they had recently fed were collected during the second sampling session, 2011-13, only. In 2011, systematic observations of the feeding activities were not collected. In 2012 and 2013, all bears encountered were examined from the air for evidence of feeding activity. Evidence of feeding included the presence of kills and other obvious food items, fresh oil or blood staining around the mouth, pendulous/distended abdomens (full stomachs), the production of black tar-like feces during pursuit (normally seen in bears that have been eating marine mammals) and the production of feces containing visible berries.

7.3. Results

Body Condition Scores

In total, 2500 polar bears were assigned a BCS during mark recapture sampling. Six bears assigned a BCS were not assigned to a sex-age class. These were excluded from the further analyses. Amongst independent bears, samples sizes were largest for adult males (n = 783), adult females with offspring (423) and lone adult females (225). Our analyses focussed on these three groups of bears for the following reasons: (1) Sample sizes were relatively large and/or (2) the accuracy of classifying bears into sex and age class at the time of aerial observation and BCS assignment was relatively good (Appendix B)[1]. Amongst dependent offspring we examined BCS for cubs-of-the-year (COY) and yearlings both of which can be identified with good accuracy from the air.

During the period 1993-97, all observations were made by a single individual (Table 7.1). During the period 2011-13, observations were made by 3 individuals but a majority (79%) were made by a single individual. Of the 2496 bears in our study, four were observed by two observers in the same year. In each case the condition scores assigned by the observers were the same (adult male in poor condition, adult female in fair condition, adult female in poor condition,

---

[1] Accuracy for adult males, adult females with offspring, lone adult females was 95%, 100% and 74%, respectively. In comparison accuracy for sub-adult males and females was <40%.

subadult female in fair condition). Because the observers collected BCS data on bears in different years and / or different parts of the study area from each other, it was not possible to distinguish observer effects from other factors.

In general, body condition was better amongst adult males and lone adult females than other age classes (Table 7.2). For example, pooling data across years, 25 and 32 % of individuals were classified as being in good condition amongst adult males and lone adult females, respectively. In contrast, amongst adult females with offspring, subadult females and subadult males, the percentage of bears in good condition was 9, 4 and 4% respectively.

There was annual variation in body condition. For example, during the period 1993-97, when all observations were collected by a single individual, the distribution of adult male BCS varied significantly from year-to-year ($\chi^2 = 24.01$, d.f. = 6, $P < 0.001$) with more males in good condition in 1993 (58.8%) compared to other years, in particular 1997 (32.0%). Annual variation in condition was similar, although not significant, for lone adult females ($\chi^2 = 11.51$, d.f. =6, $P = 0.070$) and females with offspring ($\chi^2 = 9.94$, d.f. =6, $P = 0.132$) during this period. For the period, 2011-13, when a majority of observations were made by a single but different observer, condition was also found to vary significantly from year-to-year amongst adult males ($\chi^2 = 24.31$, d.f. =4, $P < 0.001$) but not lone females ($\chi^2 = 4.89$, d.f. =4, $P = 0.300$) or females with offspring ($\chi^2 = 7.71$, d.f. =4, $p < 0.100$). During this recent sampling period, 2011 tended to be a better year for body condition.

Our annual body condition metric was associated spring sea-ice transition date amongst some sex and age classes of bears (Table 7.3). The proportion of adult males assigned a BCS of 3 (good) in a given year was closely associated with the timing of spring sea-ice transition. A higher proportion of adult males were in good condition in years with a later spring transition date (Figure 7.2a). A similar association was evident for adult females with offspring (Figure 7.2b) but not lone adult females (Figure 7.2c). These associations between condition and sea ice were also evident for adult males (Exponential curve; $F_{1, 3} = 101.27$, $r^2 = 0.98$, $P = 0.010$) and females with offspring (Exponential curve; $F_{1, 3} = 72.12$, $r^2 = 0.97$, $P = 0.014$) when analyses were limited to the period 1993 to 1997; the 4 years when data were collected by a single observer.

For COY, the proportion of bears in good condition was unrelated to the timing of spring sea-ice transition (Table 7.3). In contrast, later spring transition was associated with a higher proportion of yearlings in good condition. Similar to adult males and females with offspring, this association was also evident when analysis was limited to the period 1993-97 when analyses were limited to the period 1993 to 1997; the 4 years when data were collected by a single observer (Linear regression; $F_{1, 3} = 51.30$, $r^2 = 0.96$, $P = 0.019$).

Body condition showed a negative trend over time amongst some sex and age classes (Table 7.3). For adult males and adult females with offspring the proportion of bears in good condition during the ice-free period declined from 1993 to 2013. As similar trend, although not significant ($P = 0.065$), was evident for yearlings.

In Melville Bay, NW Greenland in 2012 and 2013 the adult bears were generally in good body condition. The samples collected from this area included 10 adult females, 5 adult males, 3 subadults (1 F, 2 M) and 2 female yearlings. Three adult females and 1 adult male scored FI = 4 and 7 adult females and 4 adult males scored F = 3. Three subadults (1 F, 2 M) and 2 female yearlings all scored F = 3. Hence, although sample size in Greenland was low, BB bears in this area were in good body condition despite an on-land period which is longer than BB polar bears that summer on Baffin Island.

Foraging During the Ice-Free period

One hundred and seven (9.3%) of the 1146 bears observed in 2012 and 2013 showed evidence of feeding. Prevalence of feeding was lowest amongst adult males (8%) and dependent offspring (7%), and highest amongst subadults (13%). Across sex and age classes, 50% of feeding observations were among adult males and subadults (Figure 7.3). The distribution of feeding observations amongst sex and age-classes did not differ significantly from the sex and age-class composition of all bears observed (feeding and not feeding); although there was a tendency for subadult bears to be over-represented amongst those observed feeding ($\chi^2 = 5.607$, d.f. = 4, $P = 0.23$). There was no seasonal trend in prevalence of feeding observations during the sampling period (Figure 7.4).

Bears were observed feeding on a range of food items including seals (species unknown), walrus (*Odobenus rosmarus*), narwhal (*Monodon monoceros*), Arctic char (*Salvelinus alpinus*), Greenland shark (*Somniosus microcephalus*) and berries (Table 7.4). Where food source was known, marine mammals comprised 47% of the observations of bears feeding. At two locations, congregations of bears were encountered along streams where char were observed to be running in large numbers. Fish carcasses found at these sites and the presence of bears standing in close quarters along these watercourses suggested bears were actively fishing. Seventeen seal kills were noted while searching for bears. Most were located along the shores of fjords rather than the main open coastline (Figure 7.5).

In Melville Bay polar bears were observed feeding on seals on patches of floes of fast ice and more or less consolidated bergy bits at glacier fronts. During September in both 2012 and 2013 numerous narwhal, ringed, bearded, harp and hooded seals were observed close to glacier fronts and land in Melville Bay (Born et al. 2012, 2013) indicating that suitable food for BB polar bears is abundant in this area during the open-water season.

7.4. Discussion

Body Condition of Baffin Bay Bears

Our results demonstrate that body condition amongst BB polar bears declined over the period 1993 to 2013. The close association between condition during the ice-free period and spring sea-ice transition date is consistent with the hypothesis that reduced time on the sea ice and presumably declining access to prey during the important spring to early summer feeding period is a primary mechanism driving this decline. However, with our qualitative body condition data and simplistic analysis we have not demonstrated cause and effect. Annual variation in body condition was associated with both time (year) and spring sea-ice transition date. These two parameters are correlated with one another and may also be associated with other parameters that we did not consider. The extent to which declining condition in BB is mediated by ice conditions therefore remains uncertain.

Using body condition metrics different from those used in the present study, Rode et al. (2011) detected a decline in the condition of BB polar bears between 1990 and 2010 concurrent with declining sea-ice cover. Our findings are consistent with this earlier study except that we did not find associations between body condition and sea-ice cover amongst all sex and age classes of bears. This may be due to limitations of the qualitative condition data used in the present analyses and sample size issues. The BCS is a qualitative and thus less precise measure of condition than the quantitative metrics used by Rode et al. (2011) that were derived from morphometric measurements acquired during physical capture and handling of bears.

Additionally, during the latter years of our study (2011-2013) the BCS for each bear was assessed based on examination from a distance rather than capture and physical examination. Inaccuracies in classifying bears by age class and sex from the air, combined with a lack of direct physical handling to assess condition may have introduced more error in assigning BCS. Given these limitations, BCS data are likely a less robust and less sensitive means of detecting changes in body condition over time or in response to ecological parameters (Vongraven et al. 2012, McKinney et al. 2014). Direct, quantitative measurement of body condition by morphometry or adipose tissue lipid content (McKinney et al. 2014) remains the most reliable and precise means of monitoring condition.

Nevertheless, our study extends the findings of Rode et al. (2011) in three notable ways. First, we find that trends in body condition and the association with sea-ice conditions have continued beyond 2010. Second, Rode et al. (2011) suggest that important trends in body condition that can affect reproduction might not be detectable from on the ground observations without capture and physical measurement of bears. Our findings suggest that long-term trends in body condition can be detected without handling of bears albeit with less sensitivity. As found in other studies (Amstrup et al. 2006, Stirling et al. 2008a), visually assigned body condition scores are a useful means of monitoring body condition and polar bear responses to environmental conditions. In the absence of physical capture programs and / or more intensive monitoring schemes, the collection of condition scores provides a simple and low cost means to track general trends in BB and likely other polar bear subpopulations. Finally, York et al. (2016) maintain that the evidence linking reduced body condition to sea-ice decline in Baffin Bay (Rode et al. 2011) is ambiguous because the body condition data used in the analyses were collected in varying parts of the subpopulation area over the period of study rather than range wide throughout. For example, during 1990s data on condition during the ice-free period were collected from bears across the subpopulation’s seasonal range on eastern Baffin Island. In contrast, sampling in the 2000s was restricted to a relatively small southern portion of the range near the boundary with Davis Strait. Consequently, condition data for the 2000s may not have been representative of the subpopulation as a whole. However, sampling in our study was range wide during both time periods (1993-1997 and 2011-2013) as illustrated in Figure 7.1 suggesting that the findings of Rode et al. (2012) were representative of BB.

Several sources of bias were possible in our study. BCS data were collected by several observers in different years and different parts of the study area. Notably a single observer collected all data during the 1990s. Several different observers collected data during the 2000s and in different parts of the study area from one another. To reduce potential observer bias in assigning qualitative condition scores, we employed a simplified body condition scoring system that required observers to discriminate between bears in poor, fair and good condition. All observers in the study were experienced polar bear biologists who had previously handled hundreds or thousands of bears in varying condition and should have been capable of easily discriminating bears in good condition. However, since individual bears were not scored by more than one observer, teasing out observer effects is challenging because differences in scored condition may reflect real temporal or spatial differences in the bears sampled. Never-the-less, several lines of evidence suggest observer bias was likely not a significant factor in our study. First, a majority of observations were made by a single observer within each time period (1990s and 2000s). Differences between these two observers in scoring body condition are unlikely to explain the linear and non-linear trends in condition we observed or the close association between condition and sea ice. Additionally, the statistical significance of these associations was maintained when analyses were restricted to a single time period when all observations were collected by a single observer. Finally, as noted by Stirling et al. (2008a) although the FI from which our condition metric was derived is a qualitative index and thus subjective, it has been found to be "*repeatable between individual biologists when blind comparisons are done in the field over both short and long time periods*." In other studies, FI data collected by multiple observers have been found to correlate closely with quantitative indices of condition (e.g., Stirling et al. 2008b, McKinney et al. 2014). Therefore, while we cannot exclude the possibility of observer bias in our study, we suggest that this potential source of bias is unlikely to account for our results.

Body condition scores in the 1990s were collected from bears captured and physically examined. In contrast scores in the 2000s were from bears observed from the air without handling. The effect of close-up versus distance examination on the scoring of condition is unknown. McKinney et al. (2014) found that remotely assigned FI ratings did not correlate with the % lipid content of adipose tissue; another measure of condition. However, their sample sizes were small and limited to comparisons of bears of FI 3 and 4 only whereas bears in our study had FI ratings ranging from 1 to 5. Remotely scoring FI may be a less robust (precise) index of condition but is not necessarily inherently biased relative to physical examination. For many of the same reasons discussed previously concerning observer effects, we suggest that this potential source of bias is unlikely to account for our results. The use of a simplified scoring system (poor, fair, good) in our study should have helped to reduce errors in scoring for bears observed from the air. Trends in condition over time and the close association with sea-ice metrics cannot be explained by differences in examination distance.

Another source of error in our study associated with differences in sampling between the

1990s (physical capture) and 2000s (aerial observation) was in the classification of bears by sex and age-class based on aerial observation rather than handling. Classifying bears from the air is without doubt less accurate than physical examination. However, aerial classification is accurate in most instances (Appendix B), especially amongst adult males and adult females with offspring; the two classes exhibiting the strongest trends in condition in our study. Additionally, despite being less precise we have no evidence to suggest that aerial classification results in a bias in age and sex classifications amongst a group of bears. This source of measurement error thus seems unlikely to account for our results.

An assumption of our study was that bears sampled within our study area were representative of the BB subpopulation. Although sampling during both the 1990s and 2000s was extensive across the seasonal range of BB bears, the proportion of bears in the subpopulation exposed to sampling may have differed between these two periods (Chapter 3). In the 1990s, a high proportion of collared bears did not come ashore on Baffin Island during the sampling windows but instead remained on remnant offshore sea ice where they could not be sampled. This observation suggests that a significant portion of the subpopulation was not sampled in the 1990s. Whether this biased our estimates of body condition is unknown. However, we note that bears remaining out on the ice were likely still able to hunt seals to some extent and may therefore have been in better condition than those coming ashore. Consequently, any bias in our sampling would have resulted in underestimation of condition in the 1990s or in years when spring transition occurred later. This would therefore not account for the trends in condition we observed over time or in association with date of spring sea-ice transition.

Relative to the 2000s, sampling in the 1990s was also more concentrated near the coast with less inland sampling (Chapter 3). Similar to other seasonally ice-free subpopulations (e.g., Derocher and Stirling 1990) polar bears in Baffin Bay are known to exhibit a degree of spatial segregation by sex, age class and reproductive status with respect to the use of terrestrial habitat during the ice-free period (Ferguson et al. 1997, Chapter 4). Adult females with cubs tend to select fjords and avoid offshore islands and coastal regions where densities of adult males are higher. Pregnant bears select inland and upland denning habitats. While limited inland sampling in the 1990s may have resulted in under sampling of certain sex, age and reproductive classes we are unaware of any evidence to suggest that this would also have biased body condition data. However, to explore the potential impact of this difference in sampling between epochs on the body condition data, we estimated and delineated the minimum extent of the sampling frame from the 1990s using capture locations in a GIS. We subset the 2000s data using this layer to create a dataset collected over the same sampling area in both time periods and repeated our analyses of trends in body condition. The results were essentially the same (Appendix C), suggesting that this sampling difference between epochs did not influence our findings.

Bears in BB lose condition through the summer and fall while on land in BB (Rode et al. 2011). Differences amongst years in the timing of sampling could therefore have affected our annual body condition metric. Sampling occurred between late August and late October but varied somewhat in timing from year-to-year. We did not consider timing of sampling in our analysis. During preliminary exploration of the data we noted that the 3 years where the median date (Julian day) of sampling was earliest were the best (1993) and two worst (2012 and 2013) years for body condition amongst both adult males and adult females with offspring, as measured by our metric. Median date of sampling was also not associated with our condition metric for any of the sex and age classes of bears. Similarly, looking at the number of days between spring transition date and the date of sampling for each bear as an index of timing of sampling relative to seas-ice breakup we found similar results suggesting that timing of sampling did not account for the annual variation in body condition that we were observing at a broad scale with our somewhat crude measure of condition. However, we acknowledge that a more sophisticated analysis such as a polynomial logistic regression could incorporate sampling date as a covariate.

Declining body condition and reproduction are amongst the first subpopulation level effects predicted occur in polar bears as a result of climate change (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Our evidence of a decline in condition in BB from 1993 to 2013, along with similar findings from a previous study (Rode et al. 2011), is accompanied by evidence of a concurrent decline in reproduction in this subpopulation. These findings are also consistent with available traditional knowledge suggesting that body condition of polar bears in BB was poorer in the early 2000s relative to the 1990s (Dowsley and Wenzel 2008, Born et al. 2011). These changes may signal a reduction in the carrying capacity of BB.

The bears that were biopsied in Melville Bay in September 2012 and 2013 generally appeared to be in good body condition. However, during an interview survey among experienced polar bear hunters in NW Greenland ca. 24% of the 72 interviewees noted that polar bears had generally become thinner (Born et al. 2011).

Similar to recent observations in the Davis Strait subpopulation (Rode et al. 2011, Peacock et al. 2013) we cannot rule-out possible density effects on body condition and reproduction resulting from a declining sea-ice platform. Regardless of whether density effects are at play, if the observed association between sea ice and body condition is real and the well documented trend in sea ice continues it is reasonable to expect that this subpopulation will experience an on-going decline in condition as a result of declining habitat (Laidre et al. 2015). This in turn is predicted to lead to demographic consequences including reduced adult survival (Molnar et al. 2010, 2011).

Measures of body condition have been identified as one of the most important metrics needed to evaluate polar bear health (Patyk et al. 2015). Similar to previous studies (e.g., Amstrup et al. 2006, Stirling et al. 2008a,b, McKinney et al. 2014) we have demonstrated the utility of a simple qualitative metric for monitoring trends in body condition in polar bear subpopulations where more detailed quantitative measures of condition may not be available. In circumstances where demographic studies are conducted periodically rather than on an on-going basis or where the selected methods of survey do not involved capture and handling, collection of visually assigned body condition scores from harvested bears or from opportunistic observations of free ranging animals offers a useful means of surveillance. Such surveillance may be carried out by government agencies but there is also potential for implementation as part of a community-based ecosystem monitoring scheme. Changes in condition detected through this method of monitoring may serve as a trigger to initiate more intensive studies.

We acknowledge that results from analyses of FI scores, including those of the present study, must be interpreted cautiously given the many potential biases associated with this type of data. While many of these potential biases can be mitigated through study design and analyses, further work is needed to examine the robustness of these data before this method of monitoring is implemented more widely in government or community-based monitoring schemes.

<u>Foraging During the Ice-Free period</u>

Polar bears have been found to opportunistically exploit a wide variety of food sources while on land during the summer and fall (e.g., Derocher et al. 1993a, Brook and Richardson

2002, Dyck and Romberg 2007, Gormenzano and Rockwell 2013, Iverson et al. 2014, Rogers et al. 2015, Tartu et al. 2016). Most of these foods are terrestrial in origin including berries, bird eggs, birds, small mammals and occasionally large mammal prey such as caribou or reindeer. Although bears have been observed catching fish (Dyck and Romberg 2007) and seals in open water (Furnell and Oolooyuk 1980), and scavenging the bone piles of human hunted bowhead whales (Rogers et al. 2015), the available evidence suggests marine mammals generally represent a small portion of the diet during this period of minimum sea ice. In Western Hudson Bay, for example, Gormenzo et al. (2013) found evidence of marine mammal remains in less than 5% of polar bear fecal samples collected during the on-land period. In contrast, terrestrial foods such as vegetation and eggs made up the majority of material in these samples. Similar dietary habits have been documented in Southern Hudson Bay (Russell 1975)[2].

In Baffin Bay we found evidence of recent foraging in approximately 9% of the bears observed during the on-land period. The type of food consumed was known for approximately half of these individuals. Marine mammals, in particular seals and walrus made up almost half of the identifiable food sources. While these are opportunistic observations and therefore subject to numerous potential biases, these findings suggest that bears in Baffin Bay may make greater use of marine mammals during the ice-free period than bears in some other subpopulations. This may be the result of differences in habitat and / or the availability of marine mammal prey. In contrast to the lowlands of the Hudson and James Bays, the east coast of Baffin Island is characterized by rugged coastline with high mountains, long, deep fjords and glaciers some of which run directly into the marine environment. Of the seal kills documented during our study most were located along the shores of fjords rather than the main open coastline (Figure 7.5). As suggested by Derocher et al. (2004) fjords may offer preferred seal hunting habitat for polar

---

[2] Russell (1975) found seal remains in 9% of polar bear scats collected on-land during the summer.

bears for several reasons. Stable sea ice at tide water glaciers provides prime breeding habitat for ringed seals (Lydersen et al. 2014). Additionally, remnant sea ice that persists longer into the summer, the shedding of ice from glaciers, the early formation of new sea ice around freshwater outflows and the availability of Arctic char and other food sources near the mouths of rivers may make fjords good habitat for seals. These same features may also make fjords good polar bear habitat. Stable and persistent ice provides a platform to hunt from and the steep sides of fjords give polar bears easy access to the deeper waters in which seals may be swimming during the open water period.

In Melville Bay polar bears were observed in September near or at glacier fronts where numerous ringed seals were also observed (Born et al. 2012, 2013). Satellite telemetry has shown that some bears remain in this habitat throughout the year for several years (Chapter 2). Hence, clearly some bears are able to sustain year round on prey (likely mainly ringed seals) taken in this type of habitat.

We observed a number of bears that appeared to be feeding on anadromous Arctic char during their seasonal runs into creeks and lakes. Observations of polar bears feeding on Arctic char have been previously reported (Dyck and Romberg 2007, Dyck and Kebreab 2009) and traditional knowledge of certain Arctic char runs that are visited annually by polar bears is present amongst residents of Nunavut (S. Atkinson pers. comm.). In Baffin Bay, the significance of Arctic char to polar bear nutritional budgets is unknown. However, we note there are numerous char runs and the available biomass is potentially high. While this food source is only available during a short seasonal window in late summer, the timing and location of char runs is highly predictable. This makes it a reliable source of food for some bears (at least locally) during the ice-free period when other foods are scarce. In some grizzly bear populations, access to anadromous fish has been shown to directly affect body mass, litter size and population density (Hildebrand et al. 1999, 2004). Whether Arctic char hold similar significance for polar bears is unknown but is worthy of further investigation. Based on energetics modelling, Dyck and Kebreab (2009) speculated that polar bears with access to char could in theory maintain or gain body weight during the ice-free period. In contrast, Rode et al. (2010b) suggested that the use of char by polar bears was limited by the availability of suitable water bodies (creeks and rivers) in which bears could capture anadromous fish with an energetic efficiency high enough to permit maintenance or gains in weight. To date, however, there have been no direct empirical studies of the significance of Arctic char in the diets and energetics of polar bears. Arctic char have not been included in prey models for quantitative free fatty acid signature analysis (QFFASA) studies of polar bear diet (e.g., Thiemann et al. 2008, 2009, Galicia et al. 2015). Consequently, this prey's signature would not have been detected in dietary studies conducted to date. We suggest that QFFASA models of polar bear diets should be calibrated to include the signatures of Arctic char sampled from the same regions as the polar bears being studied.

Sea-ice conditions are changing in Baffin Bay (Laidre et al. 2015, Chapter 4). Studies suggest this is affecting the movements and distribution (Chapter 2), habitat use (Chapter 4), body condition (Rode et al. 2011, this study) and reproductive performance (Chapter 6) of polar bears in the region. Declining condition and reproduction is presumably mediated by reduced per capita food intake but precisely how availability of food for polar bears is changing is unknown in part due to lack of knowledge about trends in marine mammal populations (Laidre et al. 2015). Changes in prey abundance and vulnerability to predation are both potential mechanisms. Changes in prey diversity are also possible. One such change may be increased access to sub-Arctic seals such as harp seals (*Pagophilus groenlandicus*). Baffin Bay is part of the summer range of the western north Atlantic harp seal population. The near 2.5 fold increase in this seal population over the last 30 decades is one of the mechanisms postulated to have supported an increase in polar bear abundance in the neighbouring Davis Strait (DS) subpopulation (Peacock et al. 2013). Similarly, McKinney et al. (2013) attributed improving body condition (expressed as adipose tissue lipid content) amongst polar bears in East Greenland (EG) to increased access to sub-Arctic seals including harp seals. Unlike DS and EG, however, bears in BB do not have access to harp seal whelping areas and are therefore not able to access this prey species during its most vulnerable season. For bears in BB, hunting of harp seals is limited to late spring through to fall when predation success rates amongst polar bears hunting in low ice cover or open water are likely relatively low. Polar bears in Svalbard are known to prey on harp seals in the summer (Derocher et al. 2002). Bears in BB may have similar summer foraging opportunities. Indeed, using QFFASA, Galicia et al. (2015) found that adult male polar bears from BB had a higher proportion of harp seal in their diet relative to bears from other subpopulations. This suggests that polar bears in BB have been able to benefit to some extent from the availability of this species. However, the observation that body condition amongst BB bears has declined over the last 3 decades suggests that any shift in prey availability associated with harp seal population expansion has not offset the effects of declining sea-ice conditions on access to other food sources.

7.5. Literature Cited

Amstrup, S. C., G. M. Durner, I. Stirling, N. J. Lunn, and F. Messier, F. 2000. Movements and distribution of polar bears in the Beaufort Sea. Canadian Journal of Zoology 78:948–966.

Amstrup, S. C., I. Stirling, T. S. Smith, C. Perham, and G. W. Thiemann. 2006. Recent observations of intraspecific predation and cannibalism among polar bears in the southern

Beaufort Sea. Polar Biology 29:997–1002.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Atkinson, S. N., and M. A. Ramsay. 1995. The effects of prolonged fasting on the body composition and reproductive success of female polar bears (*Ursus maritimus*). Functional Ecology 9:559–567.

Atkinson, S. N., R. A. Nelson, and M. A. Ramsay. 1996. Changes in body composition of fasting polar bears (*Ursus maritimus*): the effect of relative fatness on protein conservation. Physiological Zoology 69:304–316.

Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015. Demographic composition and behavior of polar bears summering on shore in Alaska: U.S. Geological Survey Administrative Report, 26 pp.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears: Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Born, E.W., K. Laidre, and Ø. Wiig. 2012. Genetic mark-recapture assessment of the Baffin Bay and Kane Basin polar bear subpopulations: Interim report on field Work 2012. Unpublished report from the Greenland Institute of Natural Resources (Nuuk) to the Greenland Government´s Department of Fisheries, Hunting and Agriculture (Nuuk) and The Danish Environmental Protection Agency (Ministry of the Environment, Copenhagen), 7 November 2012. Available from Greenland Institute of Natural Resources, Nuuk, www.natur.gl

Born, E.W., K. Laidre, and Ø. Wiig. 2013. Genetic mark-recapture assessment of the Baffin Bay and Kane Basin polar bear subpopulations: Interim report on field work 2013.

Unpublished report from the Greenland Institute of Natural Resources (Nuuk) to the Greenland Government´s Department of Fisheries, Hunting and Agriculture (Nuuk) and The Danish Environmental Protection Agency (Ministry of the Environment, Copenhagen), 10 December 2013. Available from Greenland Institute of Natural Resources, Nuuk, www.natur.gl

Brook, R. K., and E. S. Richardson. 2002. Observations of polar bear predatory behavior toward caribou. Arctic 55:193–196.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Cattet, M. R. L., N. A. Caulkett, M. E. Obbard, and G. B. Stenhouse. 2002. A body condition index for ursids. Canadian Journal of Zoology 80:1156–1161.

Cavalieri, D. J., C. L. Parkinson, P. Gloersen, and H. J. Zwally. 1996, updated yearly. Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS passive microwave data, Version 1. NASA National Snow and Ice Data Center Distributed Active Archive Center, Boulder, Colorado, USA, doi: http://dx.doi.org/10.5067/8GQ8LZQVL0VL.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Derocher, A. E., D. Andriashek, and I. Stirling. 1993a. Terrestrial foraging by polar bears during the ice-free period in western Hudson Bay. Arctic 46:251–254.

Derocher, A. E., D. Andriashek, and J. P. Y. Arnould. 1993b. Aspects of milk composition and lactation in polar bears. Canadian Journal Zoology 71: 561–567.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1990a. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395–1403.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Derocher, A. E., and I. Stirling. 1996. Aspects of survival in juvenile polar bears. Canadian Journal of Zoology 74: 1246–1252.

Derocher, A.E., Ø. Wiig, and M. Andersen. 2002. Diet composition of polar bears in Svalbard and the western Barents Sea. Marine Mammal Science 25:448–452.

Dowsley, M., and G. W. Wenzel. 2008. ‘‘The time of the most polar bears’’: A co-management conflict in Nunavut. Arctic 61:177–189.

Durner, G. M., S. C. Amstrup, R. Nielson, and T. Macdonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea. Pages 107–120 *in* S. Huzurbazar, editor. Resource selection methods and applications. Omnipress, Madison, Wisconsin, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Dyck, M. G., and E. Kebreab. 2009. Estimating the energetic contribution of polar bear (*Ursus maritimus*) summer diets to the total energy budget. Journal of Mammalogy 90:585–593.

Dyck, M. G., and S. Romberg. 2007. Observations of a wild polar bear (*Ursus maritimus*) successfully fishing Arctic Charr (*Salvelinus alpinus*) and Fourhorn Sculpin (*Myoxocephalus quadricornis*). Polar Biology 30:1625–1628.

Ferguson, S. H., M. K. Taylor, and F. Messier. 1997. Space use of polar bears in and around Auyuittuq National Park, Northwest Territories, during the ice-free period. Canadian Journal of Zoology 75:1585–1594.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Fowler, C. W. 1990. Density dependence in northern fur seals (*Callorhinus ursinus*). Marine Mammal Science 6:171–196.

Furnell, D.J., and D. Oolooyuk. 1980. Polar bear predation on ringed seals in ice-free water. Canadian Field-Naturalist 94:88–89.

Galicia, M. P., G. W. Thiemann, M. G. Dyck, and S. H. Ferguson. 2015. Characterization of polar bear (*Ursus maritimus*) diets in the Canadian High Arctic. Polar Biology 38:1983–1992.

Gormezano, L. J., and R. F. Rockwell. 2013. Dietary composition and spatial patterns of polar bear foraging on land in western Hudson Bay. BioMed Central Ecology 13:51.

Gormezano, L.J., and R. F. Rockwell. 2015. The energetic value of land-based foods in Western Hudson Bay and their potential to alleviate energy deficits of starving adult male polar bears. PLoS ONE 10:e0128520.doi:10.1371/journal.pone.0128520.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Marine Mammal Science 7:123–1 35.

Henriksen, E. O., Ø. Wiig, J. U. Skaare, G. W. Gabrielsen, and A. E. Derocher. 2001. Monitoring PCBs in polar bears: lessons learned from Svalbard. Journal of Environmental Monitoring 3:493–498.

Hilderbrand, G. V., S. D. Farley, C. C. Schwartz, and C. T. Robbins. 2004. Importance of salmon to wildlife: Implications for integrated management. Ursus 15:1–9.

Hilderbrand, G. V, C. T. Robbins, M. E. Jacoby, T. A. Hanley, S. M. Arthur and C. Servheen. 1999. The importance of meat, particularly salmon, to body size, population productivity, and conservation of North American brown bears. Canadian Journal of Zoology 77:132–138.

Hobson, K. A., I. Stirling, and D. S. Andriashek. 2009. Isotopic homogeneity of breath CO2 from fasting and berry-eating polar bears: Implications for tracing reliance on terrestrial foods in a changing Arctic. Canadian Journal of Zoology 85:193–201.

Iverson, S. A., H. G. Gilchrist, P. A. Smith, A. J. Gaston, and M. R. Forbes. 2014. Longer ice-free seasons increase the risk of nest depredation by polar bears for colonial breeding birds in the Canadian Arctic. Proceedings of the Royal Society B 281:20133128.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lydersen, C., P. Assmy, S. Falk-Petersen, J. Kohler, K. M. Kovacs, M. Reigstad, H. Steen, H. Strøm, A. Sundfjord, Ø. Varpe, W. Walczowski, J. M. Weslawski, and M. Zajaczkowski.

2014. The importance of tidewater glaciers for marine mammals and seabirds in Svalbard, Norway. Journal of Marine Systems 129:452– 471.

McKinney, M. A., S. J. Iverson, A. T. Fisk, C. Sonne, F. F. Riget, R. J. Letcher, M. T. Arts, E. W. Born, A. Rosing-Asvid, and R. Dietz. 2013. Global change effects on the long-term feeding ecology and contaminant exposures of East Greenland polar bears. Global Change Biology 19:2360–2372.

McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

Molnár, P. K., A. E. Derocher, G. W. Thiemann, and M. A. Lewis. 2010. Predicting survival, reproduction and abundance of polar bears under climate change. Biological Conservation 143:1612–1622.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.

Ramsay, M. A., R. A. Nelson, and I. Stirling. 1991. Seasonal changes in the ratio of serum urea to creatinine in feeding and fasting polar bears. Canadian Journal of Zoology 69:298–302.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. Peacock, M. K. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø. Wiig. 2011. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Rode, K.D, J. Reist, E. Peacock, and I. Stirling. 2010b. Comments in response to “Estimating the energetic contribution of polar bear (*Ursus maritimus*) summer diets to the total energy budget”. Journal of Mammalogy 91:1517–1523.

Rode, K. D., R. R. Wilson, E. V. Regehr, M. St Martin, D. C. Douglas, and J. Olson. 2015. Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.

Rogers, M. C., E. Peacock, K. Simac, M. B. O’Dell, and J. M. Welker. 2015. Diet of female polar bears in the southern Beaufort Sea of Alaska: evidence for an emerging alternative foraging strategy in response to environmental change. Polar Biology 38:1035–1047.

Russell, R. H. 1975. The food habits of polar bears of James Bay and southwest Hudson Bay in summer and autumn. Arctic 28:117–129.

Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., E. Richardson, G. W. Thiemann, and A. E. Derocher. 2008a. Unusual predation attempts of polar bears on ringed seals in the southern Beaufort Sea: Possible significance of changing spring ice conditions. Arctic 61:14–22.

Stirling, I., G. W. Thiemann, and E. Richardson. 2008b. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568–574.

Tartu, S., S. Bourgeon, J. Aars, M. Andersen, D. Ehrich, G. W. Thiemann, J. Welker, and H. Routti. 2016. Geographical area and life history traits influence diet in an arctic marine predator. PLoS ONE 11:e0155980. doi:10.1371/journal.pone.0155980.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Thiemann, G. W., S. J. Iverson, and I. Stirling. 2008. Polar bear diets and arctic marine food webs: Insights from fatty acid analysis. Ecological Monographs 78:591–613.

Thiemann, G. W., S. J. Iverson, and I. Stirling. 2009. Using fatty acids to study marine mammal foraging: The evidence from an extensive and growing literature. Marine Mammal Science 25:243–249.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Watts, P. D., and S. E. Hansen. 1987. Cyclic starvation as a reproductive strategy in the polar bear. Symposia of the Zoological Society of London 57:305–318.

Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. Science Progress 91:151–173.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution doi:10.1002/ece3.2030.

Zedrosser, A., B. Dahle, and J. E. Swenson. 2006. Population density and food conditions determine adult female body size in brown bears. Journal of Mammalogy 87:510–518.

Table 7.1. Frequency of observations for body condition scores of polar bears in Baffin Bay. Proportion of within-year observations in parentheses.

	Observer				
Year	**1**	**2**	**3**	**4**	**Total**
1993	-	-	-	149 (1.0)	149
1994	-	-	-	220 (1.0)	220
1995	-	-	-	243 (1.0)	243
1997	-	-	-	285 (1.0)	285
2011	31 (0.06)	415 (0.87)	36 (0.07)	-	482
2012	-	529 (0.79)	142 (0.21)	-	671
2013	-	316 (0.70)	134 (0.30)	-	450
Total	31	1260	312	897	2500

Table 7.2. Frequency of body condition scores (BCS) assigned to polar bears on Baffin Island. BCS scores were derived from Fatness Index (FI) scores (1-5) assigned to polar bears during field observations (following Stirling et al. 2008b). FI scores of 1-2, 3 and 4-5 were assigned BCS of poor, fair and good respectively.

			BCS			
Sex-Age Class			**Poor**	**Fair**	**Good**	**Total**
Adult Male	Year	1993	7	13	30	51
		1994	14	30	27	71
		1995	21	28	31	80
		1997	13	72	40	125
		2011	16	88	35	139
		2012	52	123	21	196
		2013	23	86	12	121
	Total		133	440	185	783
Adult Female (Lone)	Year	1993	2	2	6	10
		1994	3	6	3	12
		1995	0	4	2	6
		1997	2	20	5	27
		2011	3	26	14	43
		2012	8	44	19	72
		2013	5	26	24	55
	Total		21	128	61	225
Adult Female (w/offspring)	Year	1993	4	13	8	25
		1994	15	19	6	40
		1995	15	22	5	42
		1997	11	24	3	38
		2011	7	75	9	91
		2012	18	91	3	112
		2013	8	63	4	75
	Total		76	307	38	423
Subadult Female	Year	1993	3	7	2	12
		1994	6	6	0	12
		1995	4	14	0	18
		1997	4	10	1	15
		2011	3	31	3	37
		2012	13	62	0	75

		2013	4	35	3	42
	Total		36	165	9	211
Subadult Male	Year	1993	1	7	1	9
		1994	4	11	1	16
		1995	14	13	2	27
		1997	8	9	0	17
		2011	4	19	1	24
		2012	8	45	1	54
		2013	8	27	1	36
	Total		40	131	7	183
COY	Year	1993	2	22	2	26
		1994	16	28	0	44
		1995	15	15	4	34
		1997	22	19	0	41
		2011	4	80	12	96
		2012	8	85	1	94
		2013	5	64	8	77
	Total		67	313	25	412
Yearling	Year	1993	1	8	6	15
		1994	3	16	6	25
		1995	7	18	5	30
		1997	3	18	1	22
		2011	0	43	5	48
		2012	5	51	1	57
		2013	5	35	2	42
	Total		24	190	25	239
2-Year-Olds	Year	1993	0	1	0	1
		1995	2	3	1	6
		2011	2	0	0	2
		2012	0	8	1	9
	Total		4	12	1	18
Total	Year	1993	18	73	51	149
		1994	59	116	37	220
		1995	71	117	45	243
		1997	60	172	48	285
		2011	38	362	76	480
		2012	100	510	41	669
		2013	55	336	53	448
	Total[1]		401	1686	351	2494

[1] Excludes 6 bears of unrecorded sex-age class

Table 7.3. Regression results for an annual body condition metric for polar bears on Baffin Island. The metric, proportion of bears in good condition, was derived from observed frequencies of Fatness Index (FI) scores ranging from 1 to 5 (Stirling et al. 2008). Bears of FI 4 or 5 were in good condition. Spring ice transition was the decimal day (1-365) when ice cover over the continental shelf of BB reached 50%. Regressions were performed in the Curve Estimation procedure of SPSS (Version 24.0).

Sex-Age Class	**Dependent Variable**	F_6	r^2	***P***	***Curve Type***
Adult Male	Spring Ice Transition	102.99	0.98	≤ **0.001**	Quadratic (2nd order)
Adult Male	Year	18.50	0.79	**0.008**	Linear
Adult Female (alone)	Spring Ice Transition	0.65	0.12	0.456	Linear
Adult Female (alone)	Year	0.03	0.01	0.863	Linear
Adult Female (with offspring)	Spring Ice Transition	53.29	0.91	**0.001**	Exponential
Adult Female (with offspring)	Year	7.31	0.59	**0.043**	Exponential
Yearling	Spring Ice Transition	21.57	0.81	**0.006**	Exponential
Yearling	Year	5.526	0.53	0.065	Exponential
COY	Spring Ice Transition	0.10	0.02	0.760	Linear
COY	Year	0.334	0.06	0.587	Linear

Table 7.4. Food sources used by bears on Baffin Island during Aug to Oct, 2012 and 2013.

Food Source	**Number of Bears Observed**
Berries	10
Arctic Char	14
Walrus	10
Seal	11
Narwhal	1
Greenland Shark	1
Unknown	60
Total	107

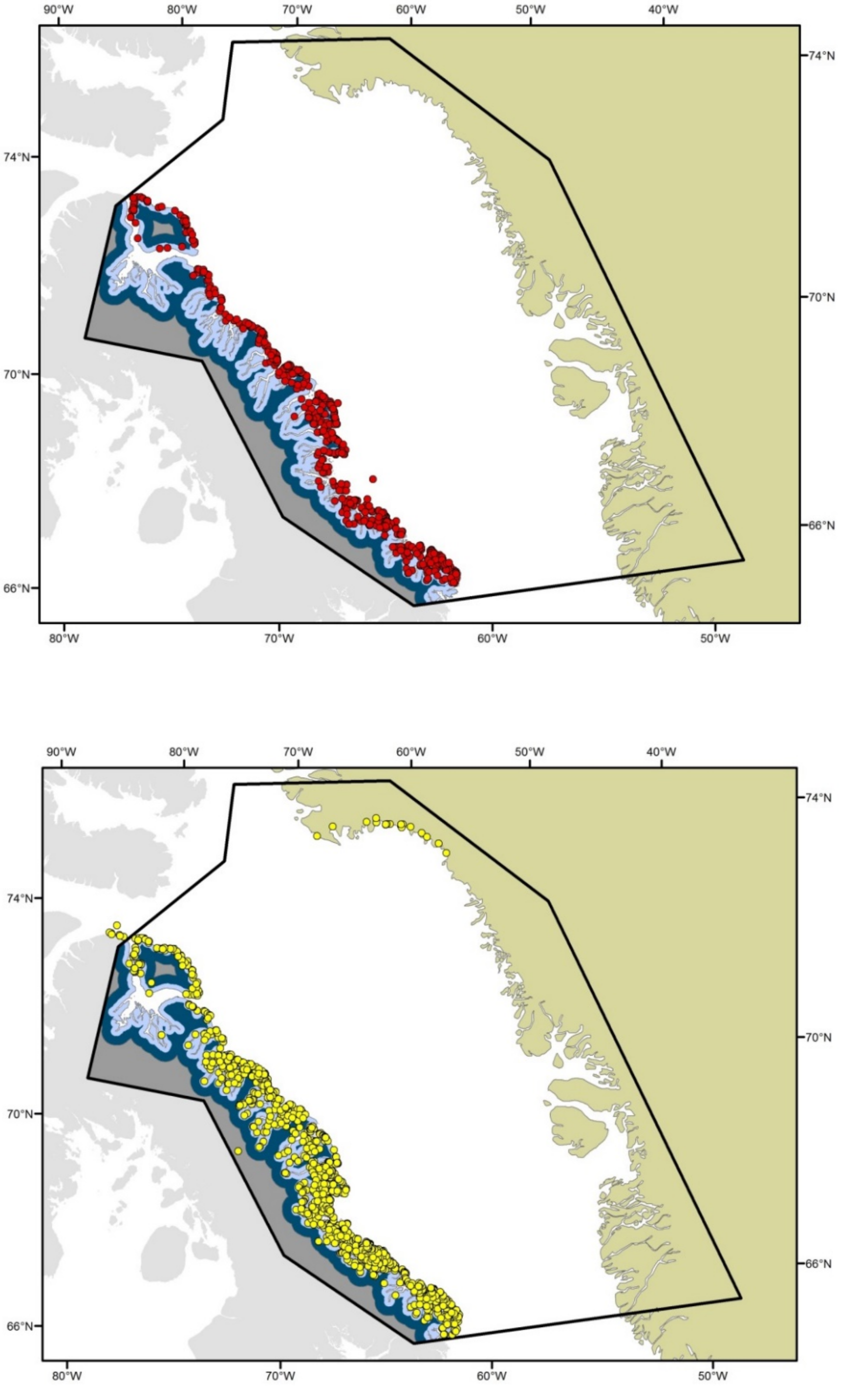
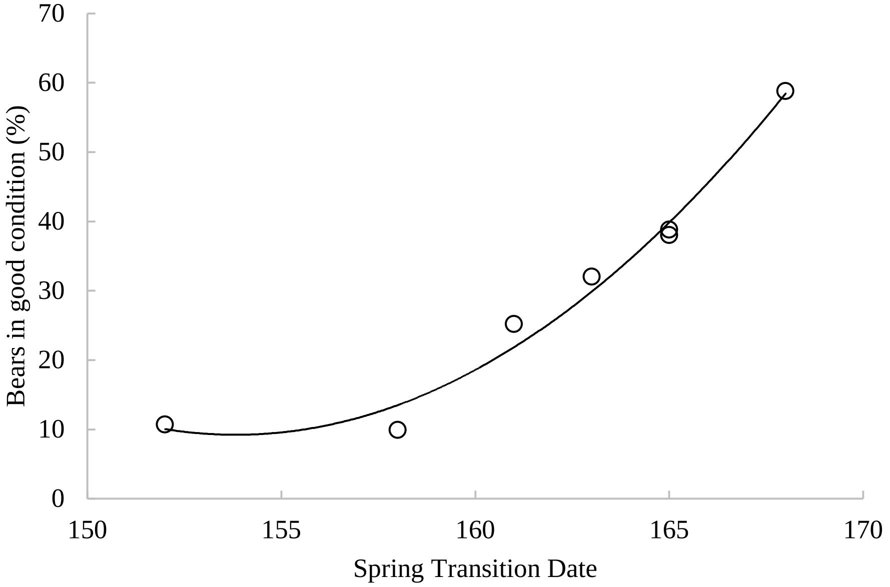


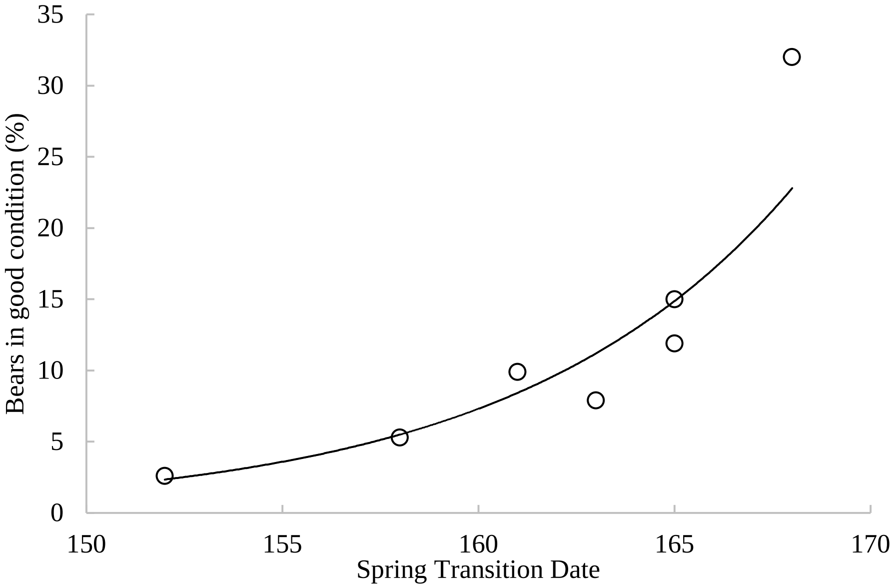
Figure 7.1. Spatial distribution of live captures recorded during sampling in the western parts of the range of the Baffin Bay polar bear subpopulation, 1993 – 1995, 1997 (top), and 2011 – 2013

(bottom). Red, 1990s. Yellow, 2010s. During both periods polar bears were also live captured in the eastern parts (i.e., the Melville Bay area) of the subpopulation´s range (data not shown).

(a)


Bears in good condition (%)
0
10
20
30
40
50
60
70
150
155
160
165
170
Spring Transition Date

(b)


Bears in good condition (%)
0
5
10
15
20
25
30
35
150
155
160
165
170
Spring Transition Date

(c)

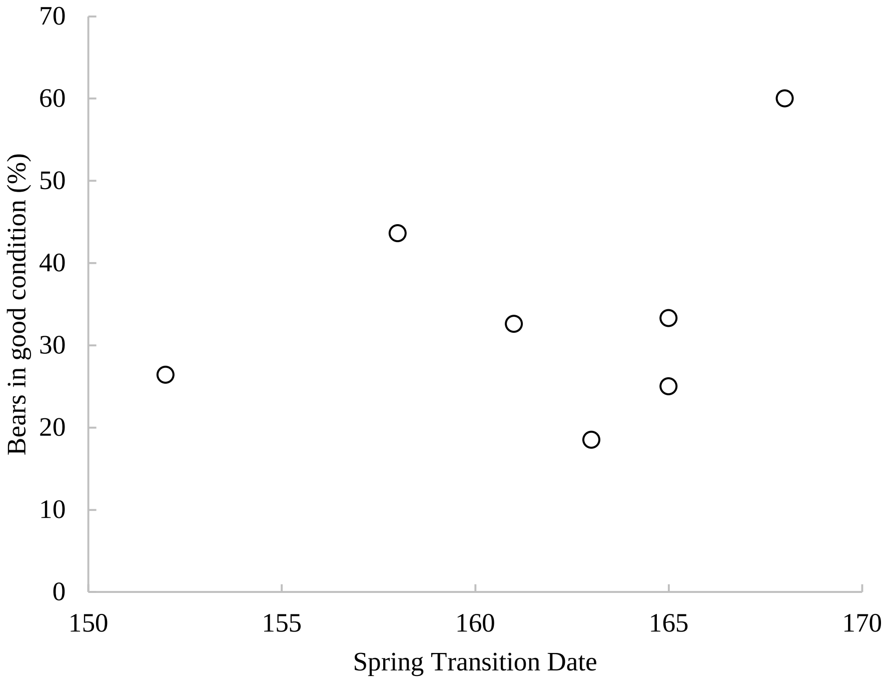


Figure 7.2. The association between the percent of bears in good body condition in western Baffin Bay and the timing of spring sea-ice transition date for (a) adult males, (b) adult females with dependent offspring and (c) lone adult females.

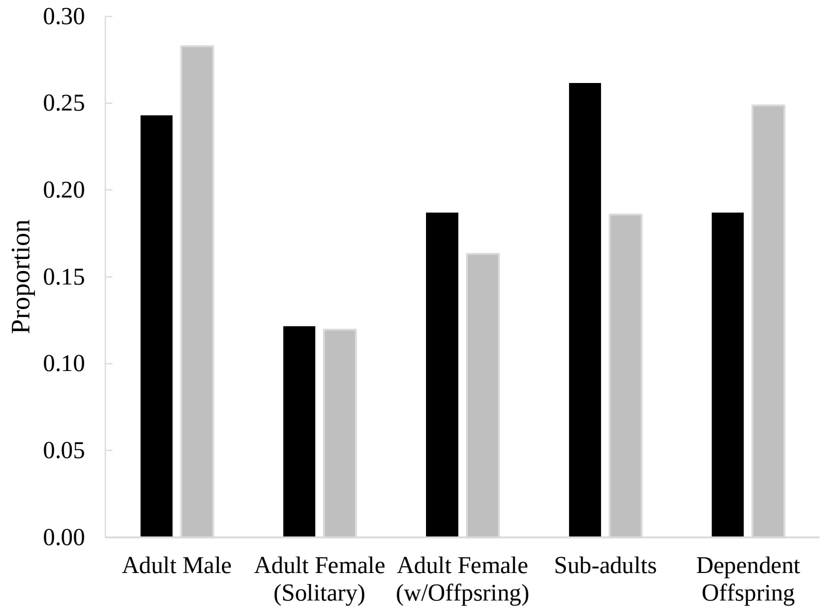


Figure 7.3. The sex and age class distribution of polar bears showing evidence of recent feeding (black) as compared to all the bears observed (grey) on Baffin Island in Baffin Bay during August-October, 2012 and 2013.

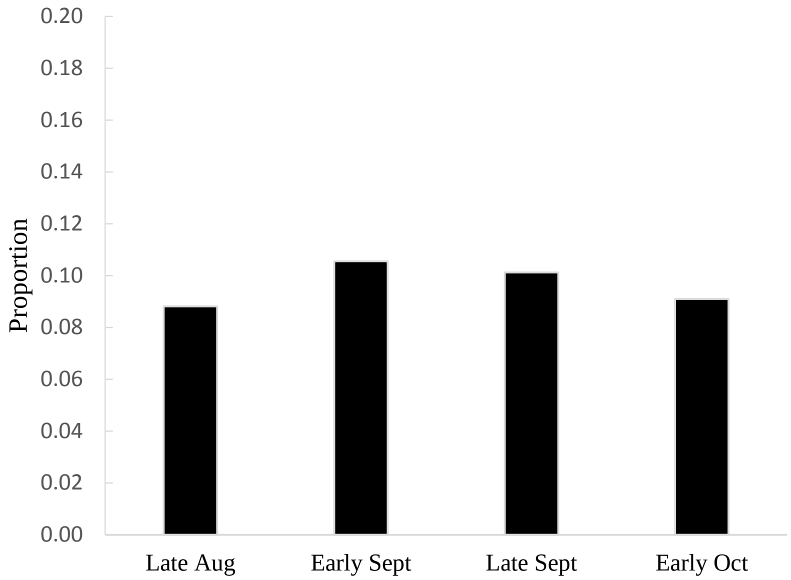


Figure 7.4. Prevalence of feeding evidence amongst bears on Baffin Island in Baffin Bay, 2012 and 2013. Data presented bimonthly.

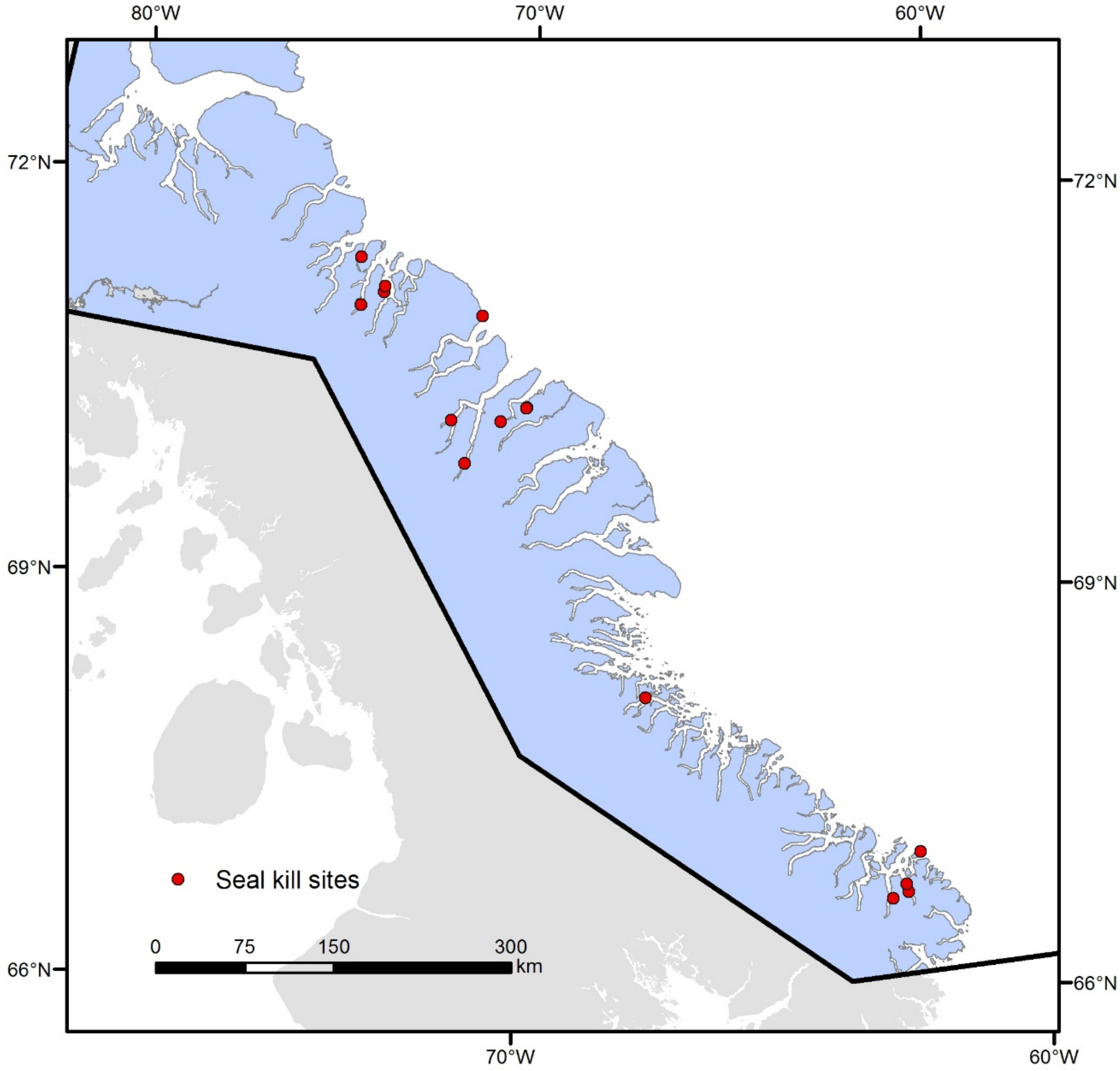


Figure 7.5. Distribution of seal kill sites observed during polar bear biopsy darting along eastern Baffin Island, Aug-Oct 2011-13.

CHAPTER 8

POLAR BEAR HARVESTING IN BAFFIN BAY AND KANE BASIN: A SUMMARY OF HISTORICAL HARVEST AND HARVEST REPORTING, 1993 TO 2014

KEY FINDINGS

- Both Canada (Nunavut) and Greenland harvest from the shared subpopulations of polar bears in Baffin Bay and Kane Basin.
- During 1993-2005 (i.e., before quotas were introduced in Greenland) the combined annual harvest averaged 165 polar bears (range: 120-268) from the Baffin Bay subpopulation and 12 polar bears (range: 6-26) from Kane Basin (for several of the years, harvest reported from Kane Basin was based on an estimate).
- During 2006-2014 the combined annual harvest averaged 161 (range: 138-176) from Baffin Bay and 6 (range: 3-9) polar bears from Kane Basin.
- Total harvest peaked between 2002 and 2005 coinciding with several events in harvest reporting and harvest management in both Canada and Greenland.
- In Baffin Bay the sex ratio of the combined harvest has remained around 2:1 (male: females) with an annual mean of 35% females amongst independent bears.
- In Kane Basin the sex composition of the combined harvest was 33% females overall for the period 1993-2014. The estimated composition of the harvest since the introduction of a quota in Greenland is 44% female but the factual basis for estimation of the sex ratio in the harvest is weak.
- In Greenland the vast majority of bears are harvested between January and June in Baffin Bay and Kane Basin whereas in Nunavut *ca.* 40% of the harvest in Baffin Bay is in the summer to fall (August – November) while bears are on or near shore. In Nunavut, all bears harvested from Kane Basin occurred in the spring.
- Sport hunting of polar bears is permitted in Canada but not Greenland. Sport hunting activity average 16% of annual harvest and peaked in 2008 coincident with several management actions. This type of hunting is highly selective for older, adult males.
- Defense-of-life-and-property kills (DLPs) of polar bears in Baffin Bay was highly selective for young (2-3 years old) individuals. We did not find evidence of a trend in the annual number of DLPs between 1993 and 2014. Most DLPs occur during the open water

period (August-November) when bears are on or near shore. The seasonal frequencies of DLPs in Canada (Nunavut) declined through the summer and fall.

- Genetic validation of the sex of individual bears as reported by the hunters showed that the gender of harvested bears was incorrectly reported in a significant number of cases. Inaccuracies in gender reporting were greatest in Greenland.
- Based on genetic validation, the presence of physical marks (i.e., ear tags or tattoos) on bears was under reported in the harvest. Detection of marked bears declined with increasing age of marks suggesting that mark loss (especially loss of ear tags) was a problem. This finding has implications for the use of harvest recoveries in MR recapture studies.
- Management and the history of management of polar bears in Canada and Greenland differs in some respects: (1) quotas were introduced in Canada in 1967 and in Greenland in 2006, (2) sport hunting is allowed in Canada but not permitted in Greenland, (3) Canadian management is designed to achieve a target harvest sex ration of 2:1 (males to females) whereas the harvest of independent polar bears in GL is non-selective, (4) the vast majority of polar bears in Greenland are taken from dog sleds whereas in Canada the vast majority are taken from snowmobiles.

8.1. Introduction

Climate induced loss of sea-ice habitat has been identified as the ultimate threat to the persistence of polar bears across their circumpolar range (PBSG 2010; Wiig et al. 2015). Other threats to the species, including pollution, industrial development, tourism and over-harvest are considered to be of varying importance amongst the different subpopulations. Three of the 19 recognized subpopulations of polar bears are currently designated as declining by the IUCN/SSC Polar Bear Specialist Group (http://pbsg.npolar.no/en/index.html). Of these three (Southern Beaufort Sea, Baffin Bay and Kane Basin), potential over-harvest has been identified as a concern in Baffin Bay (BB) and Kane Basin (KB).

Based on a mark-recapture study, Taylor et al. (2005) estimated the number of polar bears in BB at 2,074 (95% CI: 1544-2604) in 1998. A subsequent population viability analysis (PVA) using the 1998 abundance estimate, associated vital rates and available harvest data projected a decline to 1,564 bears (95% CI: 690-2402) in 2004 (PBSG 2010). Importantly, these simulations suggested that the combined harvest of bears from BB within Canada and Greenland was unsustainable. Furthermore, this PVA did not take into account on-going sea-ice habitat loss which was predicted to exacerbate the potential impacts of the harvest. In contrast to these scientific findings traditional ecological knowledge (TEK) of polar bears in BB does not suggest this subpopulation is presently declining in response to harvest and / or sea ice (Dowsley and Wenzel 2008, Born et al. 2011). York et al. (2016) have postulated that this disparity between science and TEK may be the result of inaccuracies in the scientific data for BB. In particular, potential under reporting of tagged bears in the harvest and / or over reporting of total harvest in Greenland (both under and over reporting of the harvest may result in bias in mark-recapture estimates of abundance and survival rates). However, the validity of these assertions regarding harvest reporting is unproven.

Based on a mark-recapture study, the size of the KB subpopulation was estimated to be 164 (95% CI: 94-234) polar bears for 1994 - 1997 (Taylor et al. 2008a). Taylor et al. (2008a) also suggested that Kane Basin might act as a sink (i.e., some bears may move from Baffin Bay into Kane Basin) because of unsustainable rates of harvest in KB (ibid.), and lack of genetic differentiation from Baffin Bay (Paetkau et al. 1999). Using the abundance and vital rates estimated from the 1994-97 capture study and catches reported from Nunavut and Greenland, 100% of the PVA simulations resulted in a decline in this subpopulation size after 10 years (PBSG 2010). Similar to BB, this simulated decline in subpopulation size as a result of harvest did not take into account changes in sea-ice habitat. York et al. (2016) postulated that the available TEK for KB, although limited, did not appear to contradict these scientific findings. However, neither of the sources cited by York et al. (2016) – i.e., COSEWIC (2008) and M.K. Taylor (pers. comm. 1986-2008) - bring any documentation for this statement. In contrast, a detailed study involving the interviewing of Greenland hunters with extensive experience hunting polar bears in the Kane Basin region documented the TEK perception that the occurrence of polar bear in KB has increased (Born et al. 2011).

In response to concerns regarding the projected declines in abundance, the impacts of harvest, on-going sea-ice habitat loss and the apparent disparity between scientific findings and TEK, Canada and Greenland jointly initiated new mark-recapture surveys of the BB and KB subpopulations from 2011-2014. Results from these surveys are presented elsewhere in this report (Chapters 5 and 10). Here we summarize available polar bear harvest information for BB and KB for the period 1993 to 2014; the period spanning both the recent and previous surveys in these subpopulations. Furthermore, a new system of reporting the catch of polar bears was introduced in Greenland in 1993. The objectives of this summary are to provide background information for discussion around future harvest levels and subpopulation status in light of results from the new surveys. We examine the level and composition of the harvest looking for trends over time or in relation to other factors. We also examine the harvest monitoring systems to identify issues that may affect the accuracy of harvest monitoring and the ability of jurisdictions to effectively manage harvest. Finally we make recommendations on improvements to the harvest reporting/monitoring systems.

Both BB and KB are jointly managed by Canada (Nunavut) and Greenland. Systems of harvest management and monitoring differ substantially between these two jurisdictions. Detailed descriptions of the respective systems and relevant history are presented in Appendix D. This information constitutes an important reference for the results presented herein.

8.2. Methods

Canadian Harvest Data

Data on polar bears harvested from BB and KB within Canada (Nunavut) were obtained from the Government of Nunavut's (GN) polar bear harvest database. This database contains records for all bears reported to have been harvested from these subpopulations. The methods of data collection are described elsewhere (Appendix D). From this database we extracted records for bears harvested from Jul 1, 1992 to Jun 30, 2014. The regulatory season for polar bear hunting in Canada (Nunavut) is from July 1 to June 30 the following year. This differs from Greenland where the regulatory season for polar bear hunting is from Jan 1 to Dec 31. Because adjustments in Total Allowable Harvest (quota) are made on a seasonal basis in response to recorded harvest levels in the preceding season, harvest figures were reported by season rather than by calendar year. For example, we used the notation 1992/93 when reporting harvest figures for the season beginning Jul 1, 1992 and ending Jun 30, 1993.

The extracted data included the sex, age class (adult, sub adult, 2-year-olds, yearling and cub-of-the-year [COY]), age (years), date, location and type of harvest for each individual. In reporting total harvest levels we considered all forms of human-caused mortality including illegal harvest and accidental kills but excluded bears killed for humane reasons (i.e., diseased or dying from natural causes). Other analyses were limited to the three main types of harvesting: Regular (subsistence) hunts, sport hunts and the harvest of bears in Defense-Life-and-Property (DLPs). We examined harvest by time (year or season), sex, type, monthly distribution and age. Although known for most individuals, records where sex, age class or date of harvest was unknown were excluded from the corresponding analyses where those parameters were used. Individual ages (years) based on ageing of harvested bears were only available up to June 2010 limiting our analyses of age of harvest to the period 1992/93 – 2009/10.

For DLPs we also examined the relationship with several annual metrics of sea ice to test the hypothesis that the number of human-bear conflicts, hence DLPs, increases in years when sea-ice breaks up earlier and/or forms later thereby forcing bears to remain on land for longer periods without access to their marine mammal prey (Stirling and Parkinson 2006, Stirling and Derocher 2012). As an index of the timing of spring sea-ice break-up, spring transition date was calculated as the date (Julian day) that spring sea-ice cover reached 50% over the continental shelf of BB. As an index of the timing of sea-ice formation in the fall, fall transition was calculated as the date (Julian day) that fall sea-ice cover reached 50% over the continental shelf of BB. The difference between spring and fall transition dates was used as an index for the length of the ice-free season. Addition details of these sea-ice metrics are provided in Chapter 4.

Greenlandic Harvest Data

Magnitude of the Annual Catch from KB and BB – For the period, 1993-2005 (i.e., prior to the introduction of quotas in 2006), data on the Greenland annual catch of polar bears from the KB and BB subpopulations were based on the “Piniarneq” reporting-system (for a description of this system see Appendix D). Information on catches reported via this system is compiled annually and published by the Greenland Government´s Department of Fisheries, Hunting and Agriculture (DFHA) in Nuuk. However, after the introduction of quotas when the polar bear harvest was more closely monitored detailed information on every kill was collected and compiled by the DFHA. Data for this report on the total annual kill of polar bears in 1993-2005 and in 2006-2014 were provided by the DFHA (*in litt.* August 2015)

Seasons of Catch – Information on date (day/month/year) of each polar bear catch was provided by the DFHA (*in litt.* 2014, 2015). This information was used to describe the seasonal distribution of the Greenland catch of polar bears from the KB and the BB subpopulations during 2006-2014.

Means of Transportation When Hunting Polar Bears – For each bear caught, hunters reported the type of vehicle used during the hunt (i.e., dog sled, skiff or boat; or a combination of these means of transportation). The terms “skiff” describes a boat up to 20 feet in length and usually powered with an outboard engine. Boats larger than 20 BRT/15 BT must not be used during polar bear hunts or for transportation to or from polar bear hunting grounds (Anon. 2005). These boats are small-type fishing vessel made of wood, fiber glass or steel with an inboard engine.

The information reported during 2006-2014 on vehicles used during 445 individual polar bear hunts from the KB and BB subpopulations was used to describe hunting methods both overall and regionally. Due to differences hunting traditions not at least related to stability and duration of the sea-ice cover and therefore distribution of polar bears there are regional variations in hunting methods from north to south in NW and W Greenland (*cf.* also Born et al. 2011). The NW and W coast of Greenland between Kane Basin and the town of Sisimiut was therefore subdivided into six areas: (1) Subarea 1 compasses the area to the north of Ullersuaq/Kap Alexander - 78° 10' N (i.e., Kane Basin), (2) subarea 2 consists of the area between Ullersuaq and Innaaganeq/Kap York (76° 30' N), (3) subarea 3 encompasses the areas between Innaanganeq/Kap York and the peninsula Nuussuaq/Kraulshavn at *ca.* 74° N (i.e., the Melville Bay area), (4) subarea 4 comprises the southern Upernavik area beween 74° N and the peninsula Sigguk at 71° 30' N, subarea 5 is the areas between Sigguk and the town of Aasiaat at 68° 45' N (i.e., the Uummannaq, Disko Island and Disko Bay area), and subarea 6 comprises the area between Aasiaat and Sisimiut at *ca.* 66° 55' N.

Sex and Age Composition of the Catch – Before the introduction of quotas information on the sex and age composition of the catch from KB and BB was obtained during interview surveys (Rosing-Asvid and Born 1990, Rosing-Asvid 2002, Born et al. 2011), and few biological samples (Rosing-Asvid 2002). Information on sex and age composition (i.e., independent males and females and dependent cubs) was extracted from these sources.

During an interview survey among experienced polar bear hunters in 2006 in NW Greenland (i.e., the Upernavik and Qaanaaq areas) detailed information was collected on age category (i.e., old, adult, young, and cubs) and sex of the individual bears that the interviewees had shot. Information on the composition of 588 catches (754 individual bears) going back to the early 1950s was presented in Born et al. (2011). Data on age and sex in the Greenlandic catch of polar bears during 1991-2005 (n = 354) were extracted from the interview survey database (Greenland Institute of Natural Resources, GINR, unpublished) and presented separately for the Kane Basin (KB) and Baffin Bay (BB) subpopulations.

Information on sex and age category (cub, young, adult, old) from individual polar bear kills was obtained from the DFHA based on the filled in forms provided by the licensed hunter for each polar bear killed (2006-2014). In case of catches reported by hunters living in the northernmost area (i.e., the Qaanaaq area) a polar bear may have been taken from either the KB or the BB subpopulation. In several cases coordinates of the location of kill was not given whereas the Greenlandic name of the site was noted by the hunter. In most cases it was possible to determine whether a bear was killed inside the KB management unit (i.e., north of the southern border of the KB management unit area on the Greenland side of the mid-sector line in Smith Sound; this border had been placed more or less arbitrarily at exactly 77° N; *cf*. Derocher et al. 1998, Taylor et al. 2001), or alternatively in the BB management unit. In cases where neither coordinates nor name of site of kill were stated, it was assumed that if the reporting hunter lived in the northernmost settlement Siorapaluk the bear had been taken in KB. The hunters in this settlement traditionally harvest bears from the KB management unit (Rosing-Asvid and Born 1990, Born et al. 2011). About 10.0% (5 of 49) of the kills (2006-2014) allocated to KB were based on information about hometown of the hunter reporting (or *ca.* 3.5% of a total of a total of 147 catches reported from the Qaanaaq area).

The file contained information on sex and age category of *ca.* 96% of the total catch reported for KB during 2006-2014, and *ca.* 92% of the catches reported for BB during the same period.

Detailed Age Structures – The age (and sex) structure of the Greenland catch of polar bears was examined using several sources of data. Samples (soft tissues and a vestigial tooth, i.e., 1st premolar) were collected from 55 BB polar bears caught by the hunters in 2012 and 2013. This sample represents *ca.* 40% of the reported catch in Greenland from the BB subpopulation in 2012 and 2013. The gender of each sample was determined genetically (Chapter 5). Individual ages were obtained from counting growth layer groups in premolar teeth (Calvert and Ramsay 1998).

In 2009-2013, a total of 131 individual BB-polar bears were live captured by GINR during spring (Mar 23-Apr 23) in W and NW Greenland (i.e., between 70° 14´ N and 76° 20´ N; i.e., between northern Disko Island and the settlement Savissivik) in connection with a study of movement (GINR unpublished data). These bears were tagged on fast ice and in the offshore pack ice in areas where the hunters usually take polar bears and therefore likely represent the sex and age composition of bears available on the polar bear hunting grounds. The gender of each sample was determined in the field and verified genetically (Chapter 5). Individual ages were obtained from counting growth layer groups in premolar teeth (Calvert and Ramsay 1998). The sex and age composition of this sample was included for comparison. Also for comparative purposes the age structure of a sample of 63 polar bears collected by the subsistence hunters in NW Greenland during 1988-1996 was inferred from figure 26 in Rosing-Asvid (2002:21).

Pooling of Canadian and Greenlandic Harvest Data

In contrast to Greenland where harvest is reported on a calendar year, harvest in Canada (Nunavut) is reported and quotas are adjusted on a hunting season basis, as described above. This presented some challenges in pooling data from the two jurisdictions. However, as documented later in this report, most harvesting in Greenland (99% in BB and 87% in KB) occurs between January 1 and June 30. We therefore pooled Greenland and Canadian data and report total harvest for BB and KB on the basis of Canadian harvest seasons. For example, harvest for 1998/99 includes bears harvested in Canada from July 1, 1998 to June 30, 1999 and in Greenland for January 1 to December 31, 1999.

Genetic Monitoring of Harvest

From 2011 to 2014, as part of genetic mark-recapture studies, the polar bear harvest in BB, KB and surrounding subpopulations was monitored to detect recoveries of genetically ‘marked’ bears. Tissue samples from harvested bears were submitted for genotyping and genetic sexing as described elsewhere in this report (e.g., Chapter 5). This provided data on genetic sex for comparison with the reported sex of each individual thereby allowing us to examine the accuracy of gender reporting. Additionally, several groups of bears that were physically marked with ear tags and lip tattoos during previous studies were detectable from this genetic monitoring of harvest. Archived tissues samples from all bears physically marked in BB and KB from 1990 to 1997 were genotyped and genetically sexed with the exception of bears whose ages would have been >35 years when genetic monitoring began in 2011 and a small number of individuals (n = 15) for which archived tissue samples were unavailable. Bears physically marked in BB and KB from 2010-2013 were also genotyped. Finally, bears marked during a recent physical mark-recapture study in the neighbouring Davis Strait subpopulation (Peacock et al. 2013) were genotyped. The harvest of one of these genetically and physically marked individuals could be detected by two means; the presence of ear tags or lip tattoos as reported by hunters or by matching the genotype of a harvested bear to that of a previously marked bear. This permitted a comparison of the efficiency of detection of marked bears by these two methods.

8.3. Results and Discussion

Canadian Harvest from BB and KB

Annual Harvest from Baffin Bay – In Canada (Nunavut), a total of 1,633 bears were harvested from the BB subpopulation over the period 1992/93 to 2013/14. Mean annual harvest was 74 bears per year (SD = 17.13, range 49-103, n = 22). Total Allowable Harvest (TAH) in Nunavut for the BB subpopulation over this period varied from 64 bears per year in the 1990s up to 105 starting in 2004/05 (Table 8.1). This variation reflects harvest management initiatives at the subpopulation level. An increase in TAH was implemented from 2004/05 to 2009/10 as a result of a new memorandum of understanding (MOU) between the Government of Nunavut and communities in BB. Starting in 2010/11, in response to concerns about the sustainability of the combined Canadian and Greenlandic harvest level, the TAH in Canada (Nunavut) was reduced by 10 bears per year over four years and had decreased to 65 by 2013/14. In most years, harvest from BB remained within the TAH. From 1996 onwards, exceedances of the allowable harvest resulted in adjustments of allowable harvest for the following year, in accordance with the flexible quota system (see Appendix D).

A majority (96%) of the harvest was comprised of independent bears (i.e., ≥ 2 years old) consistent with regulatory prohibitions on harvesting of family groups except in Defense-of-Life-and-Property (DLP). Also in accordance with the management system, the harvest was sex selective. Pooling data across years, the proportion of females in the harvest was 0.34 for independent bears (excludes COY and yearlings) and 0.35 for all bears which is consistent with the target sex ratio of 0.33 (i.e., 2 or more males per female). From 1996 onward, annual variation around this sex ratio (Figure 8.1: range 0.22 – 0.45) was regulated by the flexible quota system. When more than the recommended number of females were harvested, the TAH for the following season was reduced to compensate for the over harvest of females and deviation above the target sex ratio. Consequently, there was no temporal trend in harvest sex ratio.

Annual Harvest from Kane Basin – In Canada (Nunavut), harvest from KB has been minimal over the period 1992/93 to 2013/14 (Table 8.2). Total harvest during this period was 9 bears with a mean annual harvest of 0.4 bears per year (SD = 0.59, range = 0-2, n = 22). The sex ratio of the pooled harvest data (1993-2014) was 33% females (i.e., 2 males per female) consistent with the target sex ratio of the management system. The TAH in Nunavut of 5 bears per year for KB since 1996 has not been exceeded in any year. All bears were harvested in the spring (February to May).

The low level of hunting in KB is in part due to its remoteness and the logistical challenges of travelling in this subpopulation. The nearest Canadian community is Aujuittuq/Grise Fiord on southern Ellesmere Island with a population of <200 people. Access to

KB from Aujuittuq/Grise Fiord involves overland travel across southern Ellesmere Island. Costs for equipment and supplies make this journey less feasible relative to other hunting opportunities in the region (such as Jones Sounds in the Lancaster Sound subpopulation). In addition, spring-time travel into Kane Basin has been increasing in difficulty in recent years due to snow conditions (too little or early melts) on overland trails (M. Akeeagok, pers. comm.). Traditionally the Kane Basin region was a main polar bear hunting ground for the Inuit living in the Qaanaaq area in Northwest Greenland (Vibe 1968) and it is still an importing hunting area for them (Born et al. 2011).

Given the sparsity of the Canadian harvest from KB we do not report further on it in this section.

Timing of Harvest from Baffin Bay – Month of harvest was recorded for 1,594 of the 1,633 bears harvested in BB between 1992/93 and 2013/14. Pooling data across years, we examined the distribution of hunting activity across months. Approximately 40% of annual harvesting from Baffin Bay occurred between August and November when bears were on or close to shore. Harvesting activity peaked in October (23% of total).

The three main types of human-caused mortality (i.e., subsistence hunts, sport hunts and the killing of bears in defense, DLP) each varied differently in frequency throughout the year (Figure 8.2). Subsistence hunting activity (n = 1,107), termed “regular” hunting, peaked in October (29% of total hunts of this type) and continued through the fall, winter and spring until sea-ice break-up. A majority (91%) of sport hunting (n = 248) occurred on the sea ice in spring (March-May). DLPs kills (n = 185) peaked in August and declined steadily through the fall. Fifty percent of DLPs occurred in August and September. Notably, this peak in DLPs coincides with the time when bears move to shore from the melting sea ice rather than the timing of freeze-up (Taylor et al. 2005; Chapter 4). This finding is similar to Dyck (2006) but inconsistent with the hypothesis that human-bear conflicts are driven by poor body condition (Stirling and Parkinson 2006) since body condition will be at a seasonal high point in August, decreasing progressively through the fall (Rode et al. 2011). We speculate that this peak may result from the concentration of bears along the coast as they come off the ice (prior to dispersing inland) combined with the increased range and frequency of boat travel and use of camps by residents of communities along Baffin Island at this time of year. Hence higher rates of DLPs during the summer may be the product of human-bear encounter probability. However, this finding does not discount the hypothesis that bears in poor condition are more likely to come into conflict with people.

Number of Sport Hunts in Baffin Bay – Overall, sport hunting accounted for 16% of the harvest from BB in Canada from 1992/93 to 2013/14. Sport hunting activity increased through the 1990s and early 2000s before sharply declining in 2009 (Figure 8.3). This decline in sport hunting after 2008 coincided with two events. First, the 2008 listing of polar bears as “threatened” under the US Endangered Species Act and the subsequent ban on importation of polar bear hides into the US pursuant to the US Marine Mammal Protection Act (US Department of the Interior: Fish and Wildlife Service 2008). Second, Canada’s issuance in 2010 of a negative non-detriment finding for Baffin Bay under the Convention on International Trade in Endangered Species (CITES) which triggered a ban on Canada’s export of polar bear hides from this subpopulation. These two events seemingly reduced the pool of sport hunters interested in hunting opportunities in BB (Weber et al. 2015).

Number of Defense-of-Life-and-Property Kills (DLPs) in Baffin Bay – Bears killed as DLPs are the only type of polar bear harvest that is not limited by quota in Canada. All DLPs are deducted from the available quota but the total number of DLPs in a given year is not limited. The number of DLPs in BB varied annually with no apparent trend over the period 1992/93 to 2013/14 (Figure 8.4). DLP-related harvest averaged 8.6 bears per year or about 12% of annual harvest. There was no trend in the proportion of annual TAH allocated to account for DLP kills over the period 1993-2014 (Linear regression, $F_{1,\,21} = 0.19$, $P > 0.700$). Thus, problems bears do not appear to be using an increasing portion of the available quota in BB.

Restricting the data to independent bears(in this case, independent bears were defined as all individuals except for COYs and yearlings accompanying their mother) during the months of Aug to Nov, when most bears in BB were on land, did not reveal a trend in number of DLPs kills over time (Figure 8.4; Linear regression, $F_{1,\,21} = 1.404$, $P > 0.200$). The number of DLPs of independent bears during the Aug to Nov period was also unrelated to annual date of spring sea-ice transition (Linear regression, $F_{1,\,21} = 0.028$, $P > 0.800$), fall transition date (Linear regression, $F_{1,\,21} = 0.179$, $P > 0.700$) and the number of days between these two dates (used as proxy for length of the ice-free period: Linear regression, $F_{1,\,21} = 0.121$, $P > 0.700$). This suggests that earlier spring sea-ice break-up, later fall freeze-up and a lengthening ice free period was not associated with increased DLP kills in BB. This finding is inconsistent with the prediction that problem bear kills will increase as sea-ice habitat deteriorates and bears spend more time on land (Stirling and Parkinson 2006, Stirling and Derocher 2012) as has been observed in Western Hudson (Towns et al. 2009). However, we note that the relatively small samples size (n = 112) and shorter time series in our study relative to Towns et al. (2009) may have limited the findings. Moreover, unlike the situation in Western Hudson Bay where there is no subsistence hunting of polar bears in the province of Manitoba, some of the bears in BB that were taken by hunters and recorded as part of the subsistence harvest might otherwise have ended up becoming DLPs, especially those harvested as they approached communities or camps. The number of DLPs recorded amongst our data is thus likely to be an under and somewhat variable estimate of actual or potential DLPs occurring. Our results relating frequency of DLPs to sea-ice conditions should therefore be interpreted with caution.

Grouping data on DLPs into multiyear blocks (1993-95, 1996-00, 2001-05, 2006-10, 2011-13), there was no significant difference in the timing (mean Julian day) of DLPs of independent bears during Aug-Nov (ANOVA, $F = 0.846$, $P > 0.40$).

Sex Ratio of the Harvest in Baffin Bay – As reported above, the sex ratio of the harvest from BB for the period 1992/93 to 2013/14 was 0.34 amongst independent bears and 0.35 for amongst all bears. As expected there were significant differences in sex ratio amongst the different types of hunting ($\chi^2 = 16.03$, $P < 0.001$, df = 2) with sport hunting being selective for males (3:1 sex ratio) and DLPs being closer to a 1:1 ratio (Fig 8.5). Sport hunting contributes to the maintenance of a male-selective harvest; compensating for the less selective nature of DLPs.

Age Structure of Harvest in Baffin Bay – Several features of the harvest management system in Canada (Nunavut) that tend to select for or against bears in certain age ranges. While there is a regulatory limit (i.e., the TAH) on the total number of bears harvested each year and adjustments in this limit are made to compensate for the sex ratio of the harvest, there are no specific limitations on the age of bears harvested in BB; with one exception. A prohibition (under the Nunavut Wildlife Act) on the harvesting of family groups (defined as an adult female accompanied by COYs, yearlings or 2-year-olds), except in defense-of-life-and-property, protects most cubs-of-the-year (COY) and yearlings from hunting, as well as some 2-year-olds. This is reflected in the harvest data. Overall, between 1992/93 and 2013/14, *ca.* 20% (37/189) of

DLPs were adult females and their accompanying offspring. Comparable figures for regular hunting and sport hunts were *ca.* 0.4% (44/1104) and 0% (n = 260) respectively.

Sport hunting tends to select for larger, presumably older, adult bears. Looking at the available data on the known or tooth-derived age of harvested bears from BB, there were significant differences in the median ages of harvested female bears amongst the 3 main types of harvest: DLP, regular, sport (Fig 8.6: Kruskal Wallis, $H = 10.97$, $P = 0.004$). Females taken as sport hunts tended to be older than those harvested as DLPs. Similarly, the median age of male bears was significantly different amongst types of hunting (Fig 8.6: Kruskal Wallis, $H = 61.38$, $P < 0.001$). Sport hunting was highly selective for older males relative to both regular hunts and DLPs. DLPs selected for younger males.

Dyck (2006) found that a majority of DLPs occurring in polar bear subpopulations across Nunavut involved bears < 7 years old. Looking more closely at the age distribution of DLPs for Baffin Bay, we see that most bears (≈ 60%) coming into conflict with people are ≤ 3 years of age. Specifically, juveniles aged 2 and 3 years were over-represented amongst the DLPs relative to the ‘population’ age structure derived from mark-recapture sampling, especially amongst males (Figure 8.7; Males: $\chi^2 = 64.55$, df = 13, $P < 0.001$; Females $\chi^2 = 30.41$, df = 13, $P < 0.005$). Bears aged 2-3 years are at a stage, between weaning and the on-set of sexual maturity, when they may be particularly vulnerable to conflict with humans (Towns et al. 2009). Food availability for these newly independent juveniles may be relatively low since hunting skills are still developing and their relatively small body size limits the ability to compete with larger bears for food. The demands of continued growth during a life-stage of relative food scarcity may lead to poor body condition and reduced survival amongst this age class particularly as environmental conditions deteriorate (Regehr et al. 2007). Poor body condition may increase the tendency for juveniles to seek food from sites of human activity. Additionally, these individuals may be naïve with respect to risks of interacting with humans; having not yet acquired learned aversions to humans and adopted avoidance behaviors.

Annual variation in the ages of harvested bears was examined by sex and harvest type for the period 1992/93 and 2010; the period for which data were available. The median age of harvested bears did not exhibit trends over time for any of the different types of hunting (for example see Figure 8.8).

Greenlandic Harvest from BB and KB

Annual Harvest from BB – Sport hunting of polar bears is not permitted in Greenland (Anon. 2005). Hence, the only harvest types recorded by the management authorities are (1) regular subsistence harvest under quota, (2) bears killed in Defense-Life-and-Property (DLPs), and (3) illegal hunts.

The Greenlanders´ catch of polar bears from the BB subpopulation according to the Piniarneq catch recording system (see Appendix D) during 1993-2005 (i.e., the year before introduction of quotas) are shown in Figure 8.9. The trend in numbers reported per year during 1993-2005 in the Qaanaaq area (i.e., north of the Upernavik area) is not statistically significant (R = 0.272, Z = 0.881, P = 0.378, n=13). However, the catch in the Uummannaq-Sisimiut area (i.e., south of Upernavik) increased statistically significantly during the same period (R = 0.594, Z = 2.163, P = 0.031, n=13). The catch in these areas amounted to *ca.* 24% of the total catch reported by Greenlanders from BB during 1993-2005. Similarly, the catch reported from the Upernavik area (between *ca.* 74° 35’N and *ca.* 71° 30’N) increased significantly during the same period (R = 0.794, Z = 3.426, P = 0.001, n = 13). On average the reported catch of polar bears in the Upernavik area amounted to *ca.* 57% of the total catch reported in Greenland from BB during 1993-2005 (Figure 8.9). Hence, there are indications that the Greenland catch of polar bears from BB showed a real increase and especially after *ca.* 2000. An increase in availability of polar bears in the Upernavik area during the 1990s and 2000s was also indicated by traditional ecological knowledge, TEK (Rosing-Asvid and Born 1990, Born et al. 2011).

The Piniarneq-data have played an important role in the assessment of abundance and trends in abundance of the BB and the KB subpopulations. Since 1993, the Piniarneq-data on annual catch for the shared subpopulations (BB, KB and DS) as compiled and published by the DFHA have been provided annually to the Canadian Polar Bear Technical Committee and to the meetings of the IUCN/SSC Polar Bear Specialist Group. The data have been incorporated with the reported catch of polar bears in Nunavut from BB in modeling of subpopulation status (e.g., Aars et al. 2006, York et al. 2016). In a PVA (Population Viability Analysis) re-assessment of polar bear subpopulations including BB and KB based on historical data, York et al. (2016) simply assumed that the Greenland catch of polar bears reported in Piniarneq for BB was/are overestimated (“over-reporting”). However, these authors did not present any validation of the Piniarneq-data or new evidence in support of this assumption (Ibid.). As indicated in the previous other evidence suggests that the increase in the Greenlanders´ catch from BB during 1993-2005 was real. Although over-reporting, or under-reporting, to an unknown extent cannot be ruled out.

Annual Harvest from KB – During 1993-2005 (i.e., prior to introduction of quotas in Greenland in 2006) the Greenland annual catch of polar bears from the Kane Basin (KB) subpopulation was 11 (SD = 4.4 bears, range: 6-25/year; n,= 13 years) with no apparent trend.

However, it should be noted that during 1993-1999 the annual catches from KB were estimated at 10 each year during 1993-1999 (PBSG 2002, 2010) based on an interview survey in 1989 (Rosing-Asvid and Born 1990). The statistics on annual catches after 1999 were based on reportings in Piniarneq (2000-2005) and special reporting forms under the quota system (2006-2014).

After the introduction of quotas the Greenlanders´ catch of polar bears from the KB management unit (2006-2014) has averaged 6/year (SD = 1.7, range: 2-8/year, n = 9).

Defense-of-Life-and-Property Kills (DLPs) – No defense kills were reported for the Kane Basin subpopulation during 2006-2014. During the same period 7 defense kills were reported for the Baffin Bay subpopulation (2007: 2, 2011: 1, 2012: 4). The kills comprised 1 young male, 3 adult females, 1 young with sex not stated, and 2 with sex and age not stated. The months during which these incidences occurred were: January (n = 1), February (1), July (2), October (2) and December (1). Hence, since the introduction of quotas in 2006 when the recording of defense kills began, there have been no apparent annual or seasonal trends in defense kills from the KB and BB subpopulations.

In Greenland DLPs in one year are not subtracted from next year quota (DFHA, *in litt.* 2016).

Uncertainties in Catch Reporting – Prior to 2006, when quotas were introduced, there was significant uncertainty in the accuracy of the reported polar bear harvest (catch) from BB and KB in Greenland. Reporting occurred via the Piniarneq system. It has been suggested that the polar bear catch reported through the Piniarneq system may be both an under-estimate (i.e., some kills not reported) and an over-estimate (Born 1998, 2002, 2006, Jessen 2002, Rosing-Asvid 2002) of the actual catch. To obtain a hunting license for the coming year, a hunter must report his catch through the Piniarneq (in practice the catch of all species is reported in a booklet). Inevitably, a hunter may sometimes fail to report a catch of a particular species. Very often several hunters participate in a polar bear hunt (Rosing-Asvid 2002, Born et al. 2011). “Multiple reporting” (*i.e.*, one kill is reported by more than one hunter) leading to an over-estimate may thus occur when several hunters, who have participated in the same hunt and are proud that a bear was taken, each report the same kill via Piniarneq regardless of whether they shot the bear (Jessen 2002).

When evaluating the Piniarneq shortly after it was introduced, Kapel and Rosing-Asvid (1996) wrote that some hunters were not used to paperwork, and they may not see the point of keeping exact notes on the dates and numbers of animals taken. Whether this resulted in under-reporting, over-reporting, or just arbitrary reporting in order to have something to report when renewing the license, was not clear (Ibid.). In a study of the Greenland catch of ringed seals, Teilmann and Kapel (1998) identified examples of both under-reporting and over-reporting.

Generally, the numbers reported in Piniarneq are higher than those reported in the previous system of recording catches (i.e., The Hunters Lists of Game, *cf.* Teilmann and Kapel 1998). This apparent difference may be caused by several factors: (1) previous information was incomplete and the estimates of unreported catches too low, (2) the Piniarneq-system overestimates the catch due to “multiple”-reporting, (3) a real increase in the catch, or (4) a combination of all these factors. An example of sources of error in *Piniarneq* is the report in 2004 of 24 and 10 polar bears reported for Sisimiut and Maniitsoq, respectively (Born and Sonne 2006). Some of these (10 and 5) were reported by hunters with a “part-time” hunting license and were suspected to be of muskoxen (O. Heinrich, DFHA, *in litt.* 2005).

Rosing-Asvid (2002) compared information from various sources (trade in hide, information from sampling of biological tissues, and interview survey and Piniarneq) about the catch of polar bears in Greenland for the period 1993–1998. He found cases of under-reporting and of over-reporting. In the Piniarneq, simple errors like ringed seals reported as walruses or polar bears occur. However, validation of the information is not a standard procedure and some over-reporting is found in most of the species where the annual catch is low (Rosing-Asvid 2002). Another type of error may occur because the hunter does not have to report to Piniarneq where the polar bear was shot. The kill is assigned to the municipality in which the hunter lives and is therefore in some cases misplaced if the hunter has taken the bear in another area (ibid.).

According to Rosing-Asvid (2002) validating the Piniarneq-data was (is) not a standard procedure and some over-reporting is found in most of the species where the annual harvest is low. The number of polar bear kills reported in Central Greenland might be overestimated with this new reporting system, however, the trend toward more polar bears caught in West Greenland since the mid 1980's is undoubtedly real. For the period 1970-87 the reported catches only averaged 2/year in Central West Greenland, which is less than reported through the media in the latest years or by forms that for some kills have been filled out at local offices since 1995. The interviewed hunters from Upernavik also reported a marked increase in the number of polar bears in the area since mid-1980s (Born et al. 2011).

As indicated the Piniarneq-data may in some cases represent under-reporting and in other cases "over-" or "multiple"-reporting. As there has been no standard procedure in place in Greenland for validating to which extent (and/or in which direction) the Piniarneq-system is influenced by these potential errors when comes to polar bears one must be cautious when using and interpreting the data in Piniarneq, as pointed out by Born (2002).

Because of a good correlation between the number of ringed seal hides traded and the number of ringed seals reported via Piniarneq in the Upernavik municipality, Rosing-Asvid (2002) concluded that generally the Piniarneq system worked (works) well in this area (*Ibid*.). Hence, he indicated that the increase in reported catches of polar bears in the Upernavik area from the BB subpopulation was real. The average reported catch of polar bears in the Upernavik area during 1970-1987 (HLG) was 9 bears/year (range: 1-41/year) whereas during 1993-1998 it was 37 bears/year (range: 25-48/year) according to the Piniarneq (Ibid.).

Timing of Harvest in Baffin Bay – The Greenlanders catch of polar bears from the Baffin Bay subpopulation is almost exclusively concentrated in winter and spring (Jan-Jun). About 99% (n = 589) of the catches reported with month during 2006-2014 were taken during this period. The remaining *ca.* 1% (n = 5) is caught during Sep-Dec (Figure 8.10). This timing of the catches is in marked contrast to the situation in Nunavut where *ca.* 40% of the harvest is between Aug-Nov with a peak in October.

About 99% of the bears reported from BB were caught during Jan-Jun which is in contrast to pre-2006 when a relatively larger proportion was taken during fall and early winter. The fact that the catches have been more concentrated to the beginning of the year may be a result of the quota for BB (the quota year starts 1 January) being used up fast. According to the interviews conducted in NW Greenland in 2006 the availability of polar bears during spring in BB has increased (Born et al. 2011) which may explain that the quota is used up relatively soon and the catches therefore concentrated to late winter and spring.

Timing of Harvest in Kane Basin – Of 49 catches reported from the Kane Basin subpopulation during 2006-2014, 87.8% (n = 43) were taken during winter-spring (Jan-Jun) with a clear peak in April. Twenty-six (53.1%) of these catches were taken during April. The remainder (12.2%) of the catches were reported from summer and fall (Aug-Oct); Figure 8.11.

The tendency with a peak in hunting activity in Kane Basin in March-April and a less pronounced peak in fall was also seen prior to 2006 (Born et al. 2011:185). However, after 2006 the seasonal distribution of catches from the KB subpopulation has shown a more conspicuous peak in April. It should however, be noted that the sample size from KB was small.

Means of Transportation – In some cases information on means of transportation used during the polar bear hunt is lacking from the reports on individual catches. However, during 2006-2014 there was information on means of transportation used during 445 individual polar bear hunts from areas between Kane Basin and Sisimiut (i.e., from the KB and BB subpopulations, respectively). Overall, 63.6% of the bears had been caught during a hunt involving dog sled. During 35.3% of the hunts a skiff was used and only in 1.1% (n = 5) of the cases the bear was caught from a <20 BRT/15 BT boat.

Means of Transportation in Baffin Bay – There was regional variation in means of transportation used during polar bear hunts in Baffin Bay. In Subareas 3 and 4 (encompassing the Melville Bay and the Upernavik areas) where *ca.* 85% of the Greenlanders´ annual catch of bears from the BB subpopulation are taken, an average of 71.1% of the bears are caught during dog sled trips (the remainder are taken from skiffs) with a clear difference between the northern part and the southern part of the area (Table 8.3) reflecting differences in density of sea-ice cover and timing spring break-up. In Subareas 5 and 6 (Uummannaq, Disko Bay and areas south to Sisimiut) where only *ca.* 15% of the bears caught by Greenlanders´ from the BB subpopulation are taken, 7.8% of the bears reported during 2006-2014 were killed during dog sled hunts, 88.2% were taken from skiff and 3.9% from a small boat (Table 8.3). These areas have open water (or light pack ice conditions) during winter and/or early spring (e.g., Buch 2001). Dog sleds were used in 93.5% of the cases in the important polar bear hunting areas 1-3 and in 20.8% of the cases in areas 4-6 (i.e., the areas south of 74° N). This difference in means of transportation between the two overall areas was statistically significant ($\chi^2 = 246.283$, $P < 0.0001$, df = 1).

Only in one instance during 2006-2014 was it specified that a polar bear had been caught during a hunt involving the use of a snowmobile (using a snowmobile in connection with hunting polar bears is illegal in Greenland; Anon. 2005). This case involved the illegal kill of a male bear in the Sisimiut area in March 2011.

In none of the areas was there a statistically significant annual trend in fraction of bears taken from skiff or boat during 2006-2014 (linear regressions of weighted percentages of skiff+boat versus year; data not shown).

Means of Transportation in Kane Basin – Of 39 individual polar bear catches (2006-2014) from the Kane Basin subpopulation (i.e., Subarea 1 and 2; only 2 catches reported from Subarea 2), 76.9% were taken from dog sled, 15.4% from skiffs 7.7% from a boat. The catches from skiffs and boats were taken during May (n = 1) and June-October (n = 8); Table 8.3.

The data on means of transportation showed a marked north-south gradient in the use of sleds vs. skiffs related both to differences in hunting traditions and availability of dense fast or pack ice. In the southern areas where sea-ice conditions to a large extent are influenced by the inflow of relatively warm current from the south (e.g., Buch 2001) the majority of polar bears are taken from skiffs (and in a few cases from small-type fishing vessels) whereas in the areas north of *ca*. 74° N, where there is fast ice and dense pack ice, the majority of polar bears are caught by dog sled and this means of transportation is still an important element in the traditional way of living and hunting. The 2006-interview survey indicated that there has been an increase in the use of skiffs for hunting polar bears especially in the Upernavik area since the early 1990s (Born et al. 2011). This development was ascribed to the fact that the sea-ice conditions have become more unstable (for driving a dog sled) and there is an earlier spring ice break-up (ibid.). The same tendency was indicated in the 2006-2014 records of hunting methods (present study) but was not statistically significant.

Hence, in Greenland, the majority of polar bears that are taken from the KB and BB subpopulations are still taken during dedicated polar bear hunts where the dog sled (in the majority of cases) is used for transportation and tracking of polar bears. This maintains an old and traditional way of hunting polar bears.

Since 1968 snowmobiles have been used increasingly in the polar bear hunt. In Arctic Canada, polar bears are nowadays hunted almost exclusively with snowmobiles (except for guided sport hunts, which are required to use a dog team); Slavik (2013 and references therein).

Sex Ratio in Baffin Bay – The sex and age composition of the Greenlanders´ catch of polar bears from the BB and KB subpopulations is presented in Table 8.4 and 8.5, respectively.

In Baffin Bay the relative proportions of independent polar bear females and males in three different sets of data (1982-2005; Table 8.4) were not statistically different ($\chi^2 = 1.096$, $P = 0.578$, df = 2). Overall, independent females constituted *ca.* 28.3% of the total annual catch (note: including cubs which constituted *ca.* 20% of the catch) prior to 2006.

Independent female polar bears constituted *ca.* 32.5% of the total annual catch during 2006-2014. Overall, the ratio of independent F:M in the catch was *ca.* 1:2 both during 1982-2005 and 2006-2014.

In a sample of 55 bears (2012 and 2013) for which gender was determined genetically and tooth-derived ages were known independent females constituted 45.5% (Table 8.4). The relative proportions of independent females and males in this sample differed significantly at the 5% level from those in the larger sample (2006-2014) which was based on reports from the hunters ($\chi^2 = 3.972$, $P = 0.046$, df = 1).

Sex Ratio in Kane Basin – In Kane Basin the relative proportions of independent polar bear females and males in three different sets of data (1982-2005; Table 8.5) did not differ ($\chi^2 = 4.609$, $P = 0.099$, df = 2). Overall, independent females constituted *ca.* 25.4% of the total annual catch (note: including cubs which made up *ca.* 8% of the total catch) prior to the introduction of quotas in 2006. During 2006-2014 independent female polar bears constituted *ca.* 44.9% of the total annual catch (note: after 2005 it has been illegal to catch dependent cubs irrespective of their age). The relative proportions of independent females and males during 1982-2005 and 2006-2014, respectively, differed significantly ($\chi^2 = 5.130$, $P = 0.024$, df = 1); independent females constituting a higher proportion of the catch in KB after 2006. However, it must be kept in mind that the basis for data before 2005 is heterogeneous and sample size after 2006 is relatively small.

Age Structure – In a hunter collected sample of a total of 55 polar bears caught in BB Greenland during 2012 and 2013 individuals less than 10 years of age constituted 85.5% and ≥ 10 years olds were 14.5% of the catch. The oldest bears were two 17 year old females (Figure 8.12). In comparison, polar bears less than 10 years of age (i.e., 2-9 years of age) constituted 63.9% and 10+ olds 36.1%, respectively, in the sample of live captured bears (2009-2013) from BB. Hence, the proportion of polar bears ≥ 10 years of age was significantly higher than in the sample from the harvest ($\chi^2 = 8.026$, $P = 0.005$, df = 1). The oldest live captured polar bear was a 23 year old male (Figure 8.13).

Adult (i.e., sexually mature) females were defined as greater ≥ 4 years old and adult males as greater than or equal to 5 years old (Molnár et al. 2008). The relative proportion of sexually immature and sexually mature polar bears in the 2012-2013-sample did not differ significantly from a sample of 55 polar bears aged 2+ sampled in NW Greenland during 1988-1996 (Rosing-Asvid 2002: figure 26; $\chi^2 = 1.094$, $P = 0.296$, df = 1). COYs and 1 year olds were excluded from this comparison because only the sample from 1988-1996 contained these age groups. The oldest bear in the 1988-1996- sample was a 16 year old female.

Polar bear cubs usually follow the mother for two years and are weaned at 2.5 years of age although some are weaned already during their second spring (range: 1.3-2.3 years; Lønø 1970, Lentfer et al. 1980, Ramsay and Stirling 1986, Amstrup and Durner 1995). Hence for modeling purposes bears 3 years of age and older are considered “independent” or “adult (Taylor et al. 1987). Prior to introduction of quotas (2006) it was prohibited to catch COYs and yearlings in W and NW Greenland whereas after 2005 it became prohibited to take dependent cubs (i.e., cubs demonstrably belonging to a family group) irrespective of their age. Hence, we assume that the 2-year-olds reported in the Greenland catch in recent years (Figure 8.12) had left their mothers. Rosing-Asvid (2002: figure 26) presented an age composition of a sample (1988-1996) that comprises COYs, yearlings and 2-year-olds. Hence, due to differences in hunting regulations during the two periods (i.e., prior and after quotas were introduced) only 2-year-olds and older bears were considered in the comparison of age-structure in the catch. The recent sample from the catch and the sample from 1988-96 did not comprise any bears older than 17 years of age. In the sample from NW Greenland (1988-1996) Rosing-Asvid (2002) found the oldest male to be 14 years and the oldest female 16 years of age. A comparison of the two admittedly not large data sets indicates no change in age composition of the Greenlanders catch from BB over time.

The finding of relatively few polar bears ≥ 10 years of age and no individuals older than 17 years of age in the harvest is perhaps surprising and in contrast to the situation in the sample of live captured bears (2009-2013) from NW Greenland.

The sample from the harvest in Nunavut from Baffin Bay contained several bears older than 17 year of age (*cf.* Figure 8.7).

In contrast, in Central East Greenland where in a sample of 238 polar bears (1983-1996; Rosing-Asvid 2002: figure 8) *ca.* 9% were older than16 years. The oldest female was 26 years and the oldest male 30 years (ibid.).

Combined Canadian and Greenlandic Harvest from BB and KB

Baffin Bay – Data from Nunavut and Greenland were combined to examine overall harvest levels in BB for the period 1993-2014 (Table 8.6). Mean annual harvest was 163 bears (SD = 37.9, range 120-268, n = 22 years). For the period 1993-2005, prior to the introduction of a quota in Greenland, the mean annual total harvest was 165 bears (SD = 48.9, range = 120-268, n = 13 years). For period 2006-2014, after the introduction of a quota in Greenland, the mean annual harvest was 161 bears (SD = 13.6, range = 138-176, n = 9 years).

Total harvest in Baffin Bay peaked between 2002 and 2005 (Figure 8.14). This peak was the result of two events: (1) an increase in allowable harvest in Canada in 2004 (from 64 to 105 bears per year) as part of a new management agreement for the subpopulation and (2) a large increase in reported harvest in Greenland.

Total harvest declined after 2005 through to 2014 as a result of two management initiatives: (1) the introduction of a quota in Greenland in 2006 and a subsequent decrease in annual quotas and (2) implementation of a phased reduction (10 bears per year for 4 years) in Total Allowable Harvest in Canada.

The sex ratio of the reported harvest has remained around 2:1 (male: females) with an annual mean of 0.35 females amongst independent bears.

Kane Basin – Total estimated harvest in Kane Basin for the period 1993-2014 (Table 8.2) was 204 bears with a mean of 9.3 bears per year (SD = 4.63, n = 22, range = 3-26). Prior to the introduction of a quota in Greenland, mean estimated harvest in KB was 11.6 bears/year (1993-2005: SD = 4.61, n = 13, range 6-26). Following the introduction of a quota in Greenland, harvest decreased to a mean of 5.9 bears/year (2006-2014: SD = 1.62, n = 9, range = 3-9).

Thus since the introduction of a quota, the estimated harvest has halved in size and there has been a significant reduction in annual variation. However, the uncertainty of the number of polar bears taken from KB prior to 2006 must be mentioned. The polar bear hunters living in the Qaanaaq area in NW Greenland harvest polar bears both from the Kane Basin and from the Baffin Bay subpopulation (e.g., Born et al. 2011). The annual reports of total catch in the Qaanaaq area during 1993-1999 did not specify whether a bear had been taken from KB or from BB, respectively. Consequently it was assumed that 10 of the total number of polar bears reported from the Qaanaaq annually had been extracted from KB during this period. The remainder was assumed to have been taken from the BB subpopulation. However, the estimate of 10/year for KB represented the upper range of an estimate of 5-10/year which was based on an interview survey conducted in 1989 (Rosing-Asvid and Born 1990, 1995, PBSG 2010). During 2000-2005 the estimates of the fraction of bears reported from the Qaanaaq area that had been taken in KB were based on location of settlement reporting whereas during 2006-2014 the numbers are based on report of actual site of the kill.

Adding to the uncertainty of the exact number of polar bears that are taken by Greenland from the KB subpopulations is the simple fact that it cannot be determined with any certainty whether a bear taken in the central parts of the Qaanaaq area (i.e., close to the border at 77° N between the KB and BB management zone) belong the KB or the BB subpopulation. This uncertainty will of course have greater implications for the relatively small catch from KB than for BB.

Overall for the period 1993-2014, the estimated sex ratio of bears harvested in Kane Basin was 33% females. However, the sex ratio of the harvest since the introduction of a quota in Greenland has been approximately 44% female (based on pooled data for the period 2006-2014).

Accuracy of Harvest Reporting as Assessed from Genetic Studies of Sex and Individual Identity

Reporting of the Sex of Harvested Bears – During the recent genetic mark-recapture studies in BB and KB (2011-14) bears harvested in BB, KB and surrounding subpopulations were genotyped to establish genetic sex and individual identity in-order to detect recoveries of genetically marked (biopsied) individuals (Table 8.7). Rates of tissue sampling, reporting of sex and genotyping of bears harvested in BB and KB were less than 100%. Overall, 270 (75%) of the 359 bears that were reported as harvested in BB during the mark-recapture sampling period were tissue sampled and genotyped. For Kane Basin, 4 (40%) of the 10 harvested bears were genotyped. Sampling of harvested bears was lower in Greenland than Nunavut.

Using these genotyped bears we compared the gender of harvested bears as reported in harvest records in Canada and Greenland to the gender as determined by genetics. As part of the genetic analyses, conflicts between reported and genetic sex were investigated via additional genotyping to confirm genetic sex (see description of genetic methods in Chapter 5). Thus confidence in the genetic sex data is high. The results indicate there was significant inaccuracy in gender reporting with a bias towards under reporting of females. Pooling data for Canada and Greenland, 16% of genetic females in the harvest were reported as males (Table 8.8). In contrast, 4% of genetic males were reported as females. The bias was greatest in the Greenland harvest, where 39% of genetic females were reported as males and 12% of genetic males were reported as female (Table 8.9 and Figure 8.15a). In Nunavut, 5% of females were reported as males. Two percent of males were reported as females (Table 8.10 and Figure 8.15b). Overall, the sex composition of the genotyped harvest as reported in official harvest records was 37% females. The genetic composition of this harvest was 42% females. For the Greenland harvest, the sex composition of the reported harvest was 39% females. The genetic composition was 54% females. For the Nunavut harvest, the sex composition of the reported harvest was 36% females. The genetic composition was 37% females. Considering only independent bears (subadults & adults), for the Greenland harvest, the sex composition of the reported harvest was 40% females. The genetic composition was 54% females. For the Nunavut harvest, the sex composition of the reported harvest was 36% females. The genetic composition was 36% females.

Assuming these data are representative of the overall harvest, harvest in Greenland appears to be non-selective for sex. Harvest in Nunavut is approximately 2:1 males to females in accordance with target sex ratio of the flexible quota management system. In Nunavut verification of the sex of harvested bears is a regulatory requirement. Hunters are required to submit the baculum from harvested males. Where proof of sex is not provided sex is verified by genotyping. Our finding that gender reporting in the Canadian (Nunavut) data is accurate was thus expected.

Inaccuracies in reporting the sex of harvested bears is a management issue for these subpopulations. Determination of sustainable harvest levels in part depends on the sex ratio of the harvest (Taylor et al. 2008b; Regehr et al. 2015). The less selective the harvest, the lower the sustainable harvest. Incorrect reporting of gender for harvested bears may also be an issue for some of the past demographic analyses for BB and KB (e.g., PVA's) that have been used to establish quotas and subpopulation status; albeit to an unknown extent at present.

Reporting of Marked Bears in the Harvest – Using data for bears that were physically marked (tagged and tattooed) in either Baffin Bay (1990-1997) or Davis Strait (2005-2007), and subsequently also genotyped, we examined the accuracy of reporting of ear tagged and lip tattooed bears in the harvest relative to the detection of these marked bears via genotyping. Due to small samples sizes we pooled data on recoveries of physically marked bears in Nunavut and Greenland.

In the harvest data for Baffin Bay for the period 2011-2014, 9 recoveries of physically marked bears were detected by genotyping, 4 of which were not reported as tagged or tattooed in official harvest records. Expanding this dataset to the Davis Strait harvest records, resulted in detection of 42 physically marked bears, 12 of which were unreported as marked in harvest records. One of the unreported marks was a Greenland harvest record.

Approximately 29% of recoveries were not reported as being marked. These findings suggest that a significant portion of physically marked bears that are recovered in the harvest are undetected via the harvest reporting systems currently in place. Although the data are limited, detection probability appears to be a function of the age of a mark; the interval between application of tags and tattoos and subsequent harvest (Fig 8.16). Older marks are less likely to be reported suggesting that loss of tags or fading of tattoos may affect the ability of hunters to detect when they have harvested a marked bear. Indeed, in cases where marked bears were reported in the harvest, 62% were reported based on the presence of lip tattoos only. Examining data for bears physically marked in the Baffin Bay during 1990-97 that were recaptured during the recent Davis Strait inventory (2005-07) we found that 24 of 24 marked individuals whose recapture was detected by genotyping were also detected as marked by field workers. In most cases, notes on the field data sheets indicated that the recaptured bears had lost both ear-tags and were identified by means of their lip tattoo only. Mean capture interval (i.e., age of mark) was 11 years (range 8 to 15) amongst this sample of 24. This suggests that loss of ear-tags is the primary problem affecting detection of marked bears in the harvest. These findings also suggest that the problem of detecting marked individuals may be limited to the harvest data only.

The implications of this finding require careful consideration with respect to past and future mark-recapture studies. The assumption that all marked bears recovered in the harvest are reported, an assumption made in previous polar bear mark-recapture studies in Baffin Bay (Taylor et al. 2005, Peacock et al. 2012), Kane Basin (Taylor et al. 2008a), Davis Strait (Peacock et al. 2013) and elsewhere, appears to be invalid. Under-reporting of marked bears in the harvest may have introduced bias resulting in underestimation of natural survival rates in these studies. However, the extent (significance) of the bias is unknown at present. We recommend further investigation of this issue.

In our admittedly limited sample of harvest recoveries, detection of marks ≤ 5 years old was > 90%. Moving forward, this finding highlights the importance of maintaining a sample of recently (within 5 years) marked bears in the subpopulation when relying on detection of physical marks to estimate survival rates. Alternatively, we recommend genetic monitoring of the harvest in future studies where detection of ‘old’ marks is anticipated to play an important role. Further research into materials and designs for increasing the endurance of ear-tags may also be warranted.

8.4. Literature Cited

Aars, J., N. J. Lunn, and A. E. Derocher, editors. 2006. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Amstrup, S. C., and G. M. Durner. 1995. Survival rates of radio-collared female polar bears and their dependent young. Canadian Journal of Zoology 73:1312–1322.

Anon. 1954-87. Sammendrag af Grønlands Fangstlister. Ministeriet for Grønland [Summaries of “Hunters´ Lists of Game”]. Summarized during 1954-1984 by the Danish Ministry of Greenland, Copenhagen, and during 1985-1987 by the Ministry for Fisheries, Hunting and Agriculture, Nuuk. [In Danish].

Anon. 1973. *Agreement on the Conservation of Polar Bears*. Available at: http://pbsg.npolar.no/en/agreements/agreement1973.html.

Anon. 1976. Bekendtgørelse om ændring af bekendtgørelse om fredning af isbjørne i Grønland [Executive order of preservation of polar bears in Greenland]. Nalunaerutit-Grønlandsk Lovsamling, Serie A, Nr. I, MJ.G.j. nr. 1701-D8: 109. [In Danish].

Anon. 2005. Hjemmestyrets bekendtgørelse nr. 21 af 22. september 2005 om beskyttelse og fangst af isbjørne [Greenland Home Rule Executive order No. 21 of 22 September 2005 on protection and harvest of polar bears], 5 pp. [In Danish].

Born, E. W. 1998. Research on polar bears in Greenland, primo 1993 to primo 1997. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears:

Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W. 2002. Research on polar bears in Greenland 1997-2001. Pages 67–74 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., and C. Sonne 2006. Research on polar bears in Greenland, 2001-2005. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Buch, E. 2001. The ocean environment. Pages 111–122 *in* E. W. Born, and J. Böcher, editors. The Ecology of Greenland. Atuakkiorfik/Education Publishers, Nuuk, Greenland.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Derocher, A. E., G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. 1998. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Dowsley, M., and G. W. Wenzel. 2008. ‘‘The time of the most polar bears’’: A co-management conflict in Nunavut. Arctic 61:177–189.

Dyck, M. G. 2006. Characteristics of polar bears killed in defense of life and property in Nunavut, Canada, 1970– 2000. Ursus 17:52–62.

Jessen, A. 2002. A note on the management of polar bears in Greenland. Page 65 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Kapel, F. O., and A. Rosing-Asvid. 1996. Seal hunting statistics for Greenland 1993 and 1994, according to a new system of collecting information, compared to the previous Lists-of-Game. NAFO Scientific Council Studies 26:71–86.

Lentfer, J. W., R. J. Hensel, J. R. Gilbert, and F. E. Sorensen. 1980. Population characteristics of Alaskan polar bears. International Conference on Bear Research and Management 4:109–115.

Lønø, O. 1970. The polar bear (*Ursus maritimus*, Phipps) in the Svalbard area. Norsk Polarinstitutt Skrifter 149:1–115.

Molnár**,** P. K., A. E. Derocher, M. A. Lewis, and M. K. Taylor. 2008. Modelling the mating system of polar bears: a mechanistic approach to the Allee effect. Proceedings Royal Society B 275:217–226.

Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. 2010. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2002. Status of the polar bear. Pages 21–35 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Ramsay, M. A., and I. Stirling. 1986. On the mating system of polar bears. Canadian Journal of Zoology 64:2142–2151.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rosing-Asvid, A. 2002. The polar bear hunt in Greenland. Greenland Institute of Natural Resources Technical Report 45, 25 pp.

Rosing-Asvid, A., and E. W. Born. 1990. Fangst af isbjørn (*Ursus maritimus*) i Avanersuaq og Upernavik kommuner: En interviewundersøgelse [Catch of polar bears in the municipalities of Avanersuaq and Upernavik: an interview survey]. Grønlands Hjemmestyres Miljø- og Naturforvaltning Teknisk Rapport Nr. 23, 63 pp. [In Danish with English summary].

Rosing-Asvid, A., and E. W. Born. 1995. The catch of polar bears in northwestern Greenland. Page 188 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Slavik, D. V. W 2013. Knowing *nanuut*: Bankslanders' knowledge and indicators of polar bear population health. MSc thesis, Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta, Canada.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

Teilmann, J., and Kapel, F.O. 1998. Exploitation of ringed seals (*Phoca hispida*) in Greenland. NAMMCO Scientific Publications 1:130–151.

Towns, L., A. E. Derocher, I. Stirling, N. J. Lunn, and D. Hedman. 2009. Spatial and temporal patterns of problem polar bears in Churchill, Manitoba. Polar Biology 32:1529–1537.

Vibe, C. 1968. Thule-eskimoernes isbjørnejagter [Polar bear hunting of the Thule-Eskimos]. Tidsskriftet Grønland 6:175-180. [In Danish].

Vibe, C. 1985. Miscellaneous information relating to polar bears in Greenland. Pages 44–57 *in* Polar Bears: Proceedings of the 8th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland.

Weber, D. S., T. Mandler, M. Dyck, P. J. Van Coeverden de Groot, D. S. Lee, and D. A. Clark. 2015. Unexpected and undesired conservation outcomes of wildlife trade bans - An emerging problem for stakeholders? Global Ecology and Conservation 3:389–400.

Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution doi:10.1002/ece3.2030.

Table 8.1. Summary of Canadian polar bear harvest from the Baffin Bay subpopulation for the period 1992/93 to 2013/14.

Harvest Season[1]	Total Harvest	Total Allowable Harvest[2]	Proportion Female (Total Harvest)[3]	Proportion of Harvest Made up of Independent Bears[4]	Proportion Female (Independent Bears Only)[5]
1992/93	62	n/a	0.36	0.94	0.35
1993/94	60	n/a	0.37	0.88	0.34
1994/95	60	64	0.33	0.92	0.35
1995/96	55	64	0.33	0.95	0.35
1996/97	60	64	0.41	0.88	0.42
1997/98	69	64	0.38	0.96	0.38
1998/99	49	64	0.35	0.98	0.36
1999/00	58	64	0.41	0.95	0.40
2000/01	61	64	0.28	0.98	0.28
2001/02	64	64	0.30	1.00	0.30
2002/03	62	64	0.26	0.97	0.22
2003/04	69	64	0.28	0.99	0.25
2004/05	101	105	0.37	0.98	0.38
2005/06	94	105	0.32	0.98	0.28
2006/07	89	105	0.36	0.99	0.38
2007/08	101	105	0.28	0.97	0.26
2008/09	103	105	0.39	0.98	0.39
2009/10	86	105	0.41	1.00	0.41
2010/11	94	95	0.33	0.98	0.34
2011/12	90	85	0.40	0.96	0.37
2012/13	74	75	0.47	0.92	0.45
2013/14	72	65	0.31	0.97	0.29
Mean	74.23		0.35	0.96	0.34

[1] The hunting season in Canada runs from July 1st to June 30th

[2] Total Allowable Harvest (TAH) is the regulated limit for all human-caused mortalities. Formerly referred to as the quota

[3] Excludes bears of unknown sex (which constituted a mean of *ca.* 1% of harvest annually for the period 1989/90 to 2013/14)

[4] Excludes bears of unknown dependency (which constituted a mean of *ca.* 4% of harvest annually for the period 1989/90 to 2013/14)

[5] Excludes bears of unknown sex and dependency

Table 8.2. Canadian and Greenlandic polar bear harvest from the Kane Basin subpopulation for the period 1992/93 to 2013/14.

Harvest Season[1]	Total Harvest	Total Allowable Harvest[2]	Harvest (Canada)	Harvest (Greenland)[3]	Proportion Female (Independent bears only)
1992/93	10	-	0	10	0.25
1993/94	11	-	1	10	0.23
1994/95	10	-	0	10	0.25
1995/96	10	-	0	10	0.25
1996/97	11	-	1	10	0.23
1997/98	12	-	2	10	0.30
1998/99	11	-	1	10	0.23
1999/00	6	-	0	6	0.25
2000/01	11	-	1	10	0.32
2001/02	12	-	0	12	0.25
2002/03	12	-	0	12	0.25
2003/04	9	-	0	9	0.25
2004/05	26	-	1	25	0.28
2005/06	9	-	1	8	0.40
2006/07	6	15	0	6	0.46
2007/08	7	13	0	7	0.46
2008/09	5	11	0	5	0.46
2009/10	3	11	1	2	0.30
2010/11	6	11	0	6	0.46
2011/12	6	11	0	6	0.46
2012/13	6	11	0	6	0.46
2013/14	5	11	0	5	0.46
Mean	9.3	11.8	0.4	8.9	0.33

[1] Greenland harvest data for a given calendar year were included in the harvest season ending in that calendar year

[2] No quota in Greenland prior to 2006 thus total quota only presented for period 2006-2014

[3] Annual harvest in Greenland (1993-1999) was estimated from an interview survey conducted in 1989 (Rosing-Asvid and Born 1990). Information on annual catch was based on information in Piniarneq during 2000-2005 and thereafter on the special reporting under the quota system (see Appendix D)

Table 8.3. Means of transportation used during 445 polar bears catches in West and Northwest Greenland from Subareas 1 and 2 (i.e., the Kane Basin subpopulation) and Subareas 3-6 (the Baffin Bay subpopulation) during 2006-2014.

Subarea	Dog sled	Skiff	Boat	Total	% Dog sled	% skiff and boat
1+2[1]	30	6	3	39	76.9	23.1
3	215	8	0	223	96.4	3.6
4	34	98	0	132	25.8	74.2
5	4	25	0	29	13.8	86.2
6	0	20	2	22	0.0	100.0
Total	283	157	5	445	63.6	36.4

[1] Only 2 catches reported from Subarea 2

Table 8.4. Sex and age composition of the Greenland catch of polar bears from the BB subpopulation (1982-2014) based on various sources.

Period	% Females[1]	% Males[1]	Independent F:M ratio	% Cubs	F (n)	Cubs (n)	M (n)	Total (n)	Source	Source
1982-1989	29.9	57.7	1:1.9	12.4	41	17	79	137	Interviews 1989-90	Rosing-Asvid and Born 1990, Rosing-Asvid 2000
1988-1996	33.3	46.0	1:1.4	20.6	21	13	29	63	Samples	Rosing-Asvid[2] 2002: table 3
1991-2005	26.8	50.6	1:1.9	22.6	95	80	179	354	Interviews 2006	Born unpublished, and Born *et al.* 2011
1982-2005	28.3	51.8	1:1.9	19.9	157	110	287	554		
2006-2014	32.2	67.8	1:2.1	0.0	192	n.a.[3]	404	596	Special Reporting Forms	2006-13:DFHA[4] *in litt. (*2014), 2014:DFHA *in litt.* (2015)
2012-2013	45.5	54.5	1:1.2	0.0	25	n.a.	30	55	Samples	Samples collected by hunters, GINR unpublished
2006-2014	32.5	67.5	1:2.1	0.0	193	n.a.	401	594		

[1] Percentage of total annual catch. Adult and subadult females and adult and subadult males = individuals 3 years old and older (*cf.* Taylor et al. 1987)

[2] A comparison of figure 26 and table 3 in Rosing-Asvid (2002) shows that the percentages of independent bears given in his table 3 are based on 2+ years old bears

[3] Since 2006 it has been prohibited to kill dependent cubs irrespective of their age

[4] DFHA = Department of Fisheries, Hunting and Agriculture (Nuuk)

Table 8.5. Sex and age composition of the Greenland catch of polar bears from the KB subpopulation (1982-2014) based on various sources.

Period	% Females[1]	% Males[1]	Independent F:M ratio	% Cubs	F (n)	Cubs (n)	M (n)	Total (n)	Source	Source
1982-1989	29.2	63.1	1:2.2	7.7	19	5	41	65	Interviews 1989-90	Rosing-Asvid and Born 1990, Rosing-Asvid 2000
1988-1996	36.7	56.7	1:1.5	6.7	11	2	17	30	Samples	Rosing-Asvid 2002: table 3
1991-2005	17.9	73.1	1:4.1	9.0	14	7	57	78	Interviews 2006	Born unpublished, and Born et al. 2011
1982-2005	25.4	66.5	1:2.6	8.1	44	14	115	173		
2006-2014	44.9	55.1	1:1.2	0.0	22	n.a.[2]	27	49	Special Reporting Forms	2006-13:DFHA[3] *in litt. (*2014), 2014:DFHA *in litt.* (2015)

[1] Independent females as percentage of total annual catch
[2] Since 2006 it has been prohibited to kill dependent cubs irrespective of their age
[3] DFHA = Department of Fisheries, Hunting and Agriculture (Nuuk)

Table 8.6. The combined Canadian and Greenlandic polar bear annual harvest from the Baffin Bay subpopulation for the period 1992/93 to 2013/14. Annual average proportion of independent female polar bears is shown.

Harvest Season[1]	Total Harvest	Total Allowable Harvest[2]	Harvest (Canada)	Harvest (Greenland)[3]	Proportion Female (Independent bears only)
1992/93	134		62	72	0.35
1993/94	120		60	60	0.35
1994/95	124		60	64	0.35
1995/96	122		55	67	0.35
1996/97	139		60	79	0.38
1997/98	165		69	96	0.36
1998/99	146		49	97	0.36
1999/00	126		58	68	0.37
2000/01	158		61	97	0.33
2001/02	182		64	118	0.33
2002/03	268		62	206	0.32
2003/04	225		69	156	0.32
2004/05	236		101	135	0.36
2005/06	173		94	79	0.30
2006/07	165	178	89	76	0.35
2007/08	176	176	101	75	0.29
2008/09	174	173	103	71	0.36
2009/10	150	171	86	64	0.37
2010/11	165	160	94	71	0.33
2011/12	165	152	90	75	0.35
2012/13	137	142	74	63	0.39
2013/14	146	132	72	74	0.31
Mean	163	161	74.23	89.00	0.35

[1] Greenland harvest data for a given calendar year were included in the harvest season ending in that calendar year

[2] No quota in Greenland prior to 2006 thus total quota only presented for period 2006-2014

[3] Harvest in Greenland is estimated from reported harvest in west Greenland and the estimated portion of this harvest that occurs in Baffin Bay

Table 8.7. Genotyping of bears harvested in Canada and Greenland, 2011-2014. Data presented as the percentage of individuals in the reported harvest that were sampled and genotyped. Total number of individuals reported as harvested is presented in parenthesis. Data are organized by Nunavut hunting seasons which run from July1 to June 30.

	2011/12		2012/13		2013/14	
Subpopulation	Canada	Greenland	Canada	Greenland	Canada	Greenland
BB	75 (85)	64 (69)	90 (78)	42 (60)	100 (67)	n/a[1]
DS	74 (38)	-	92 (60)	-	-	-
LS	75 (92)	-	91 (92)	-	-	-
KB	0 (0)	50 (6)	0 (0)	25 (4)	-	-
FB	86 (107)	-	91 (109)	-	-	-
NW	0 (0)	-	33 (3)	-	-	-

---

[1] Greenland harvest during this season occurred after mark-recapture sampling ceased and is therefore not reported.

Table 8.8. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Data from Canada (Nunavut) and Greenland harvest.

	Genetic Sex	
Reported Sex	Male	Female
Male	156	19
Female	6	97

Table 8.9. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Greenland harvest only.

	Genetic Sex	
Reported Sex	Male	Female
Male	29	15
Female	4	24

Table 8.10. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Data from Canada (Nunavut) harvest only.

	Genetic Sex	
Reported Sex	Male	Female
Male	127	4
Female	2	73

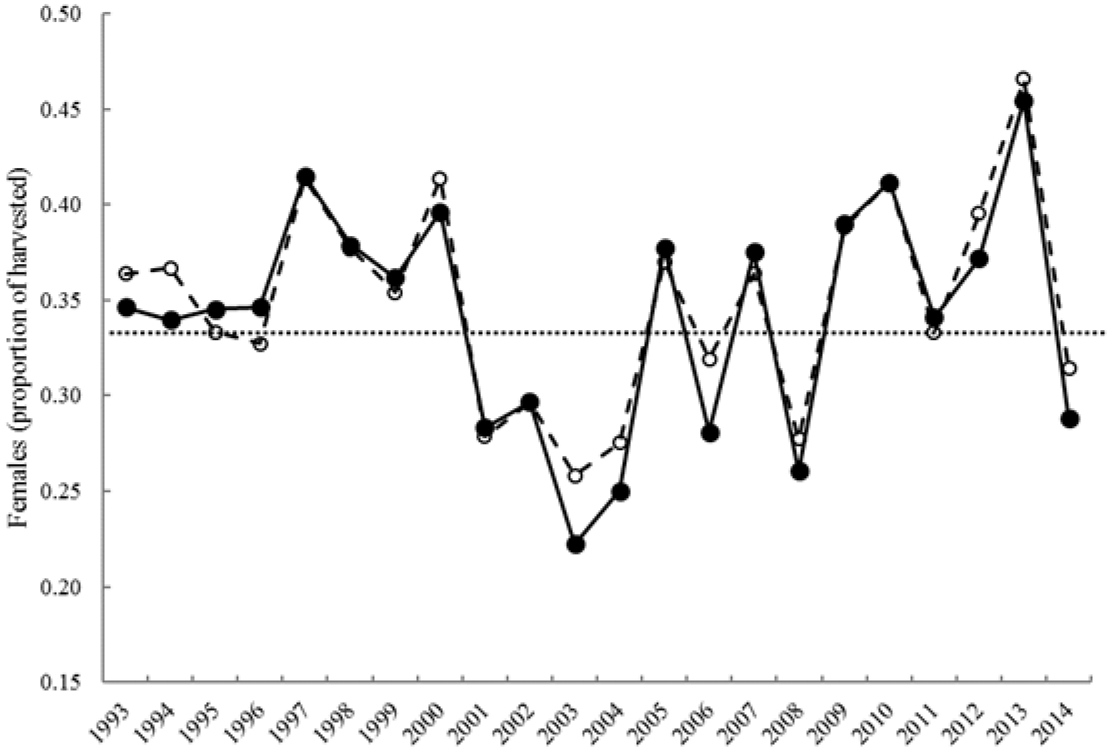


Figure 8.1. Harvest of female polar bear as a proportion of total Canadian harvest from Baffin Bay. All females (dashed line) and independent females only (solid line). Dotted line indicates target sex ratio (0.33).

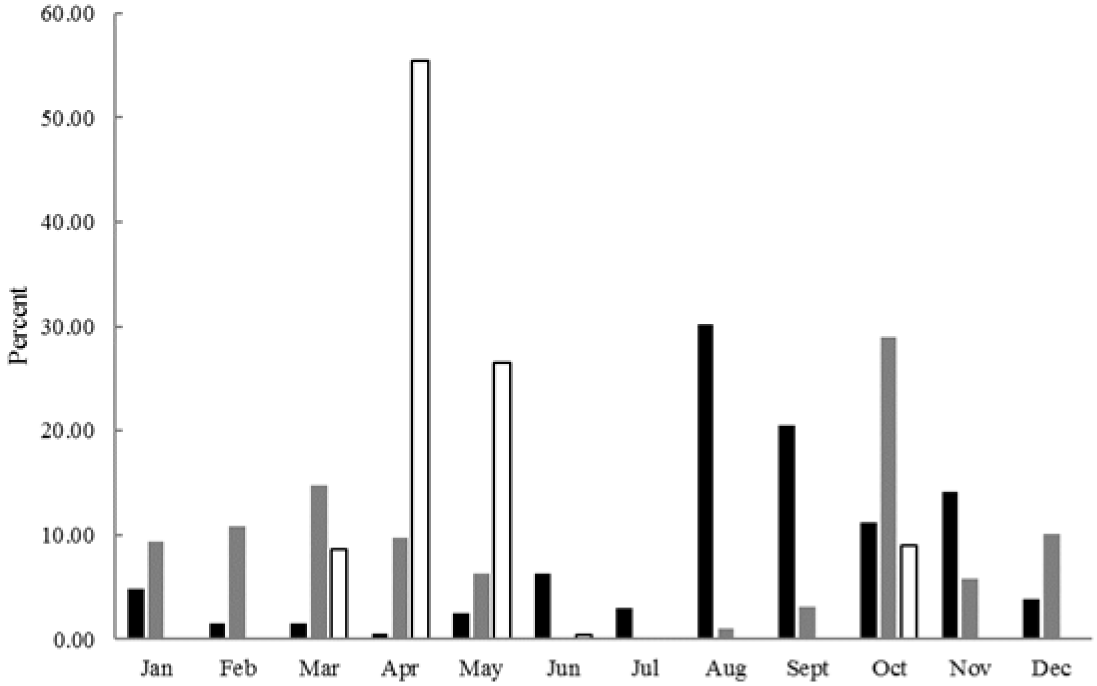


Figure 8.2. Monthly distribution of polar bear harvesting in Baffin Bay by type of harvest (1992/93-2013/14). Regular hunts (grey), defense-of-life-and-property kills (black) and sport hunts (white). Bars represent the percentage of hunting of a given type that occurred each month.

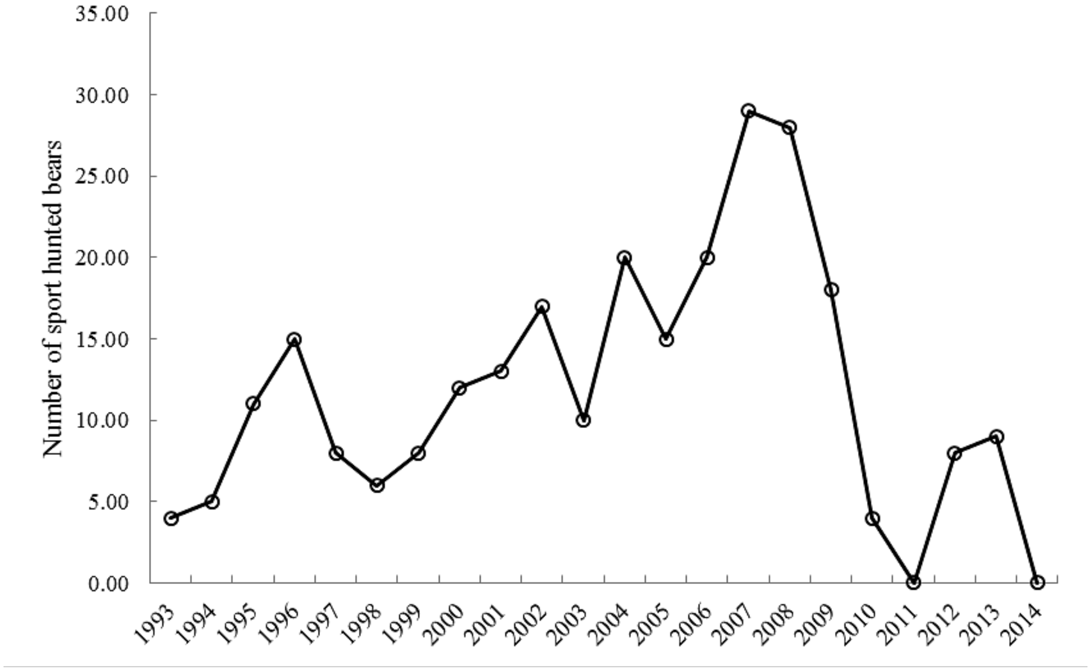


Figure 8.3. Numbers of polar bear taken by sport hunters in Baffin Bay, Canada.

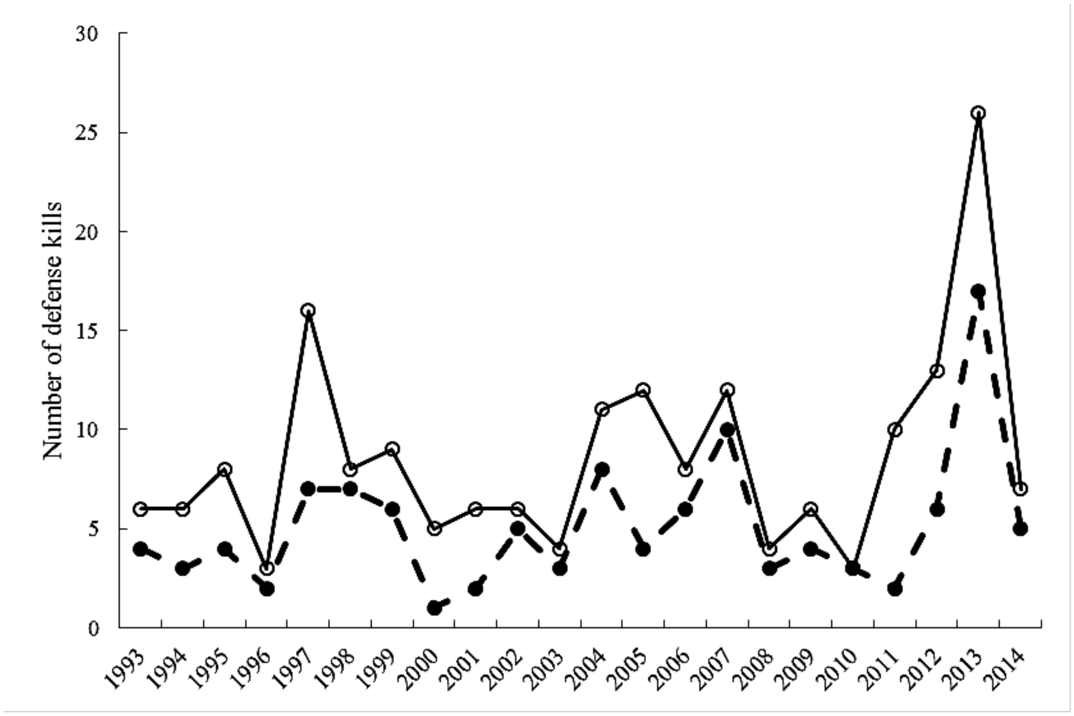


Figure 8.4. Number of polar bears harvested in defense-of-life-and-property (DLP) in Baffin Bay, Canada. Total DLPs per year (solid line) and DLPs of independent bears during Aug-November each year (dashed line).

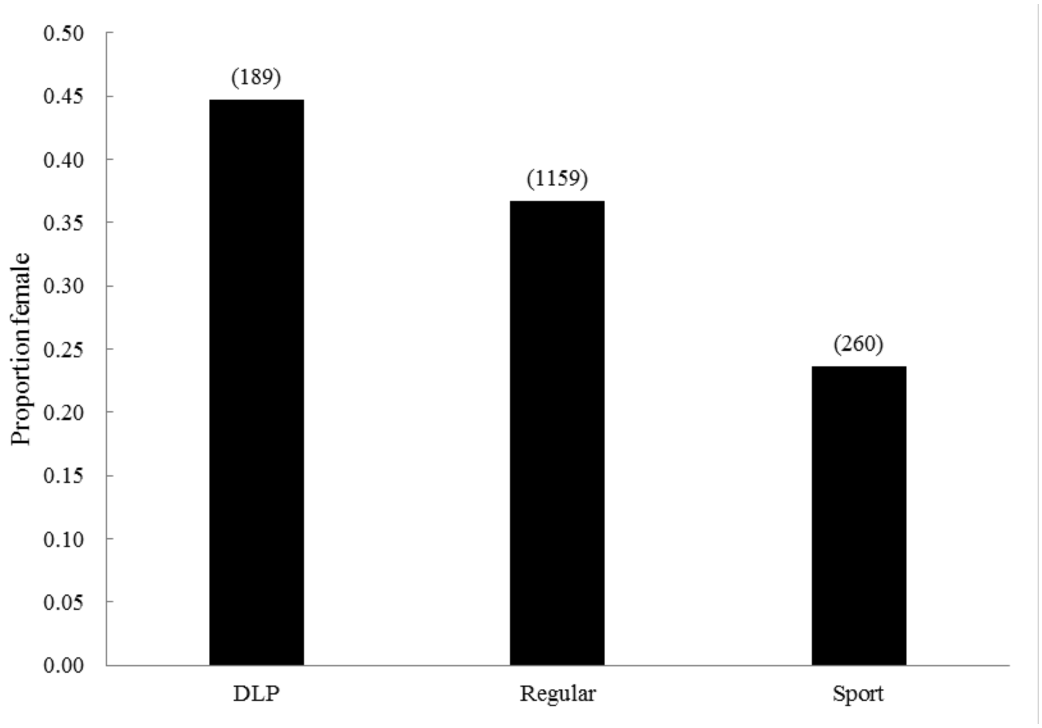


Figure 8.5. Sex composition of the three main types of polar bear harvesting in Baffin Bay, Canada, for the period 1992/93 to 2013/14. Bars represent the proportion of harvest that was female. Data are for independent bears only. Sample sizes in parentheses.

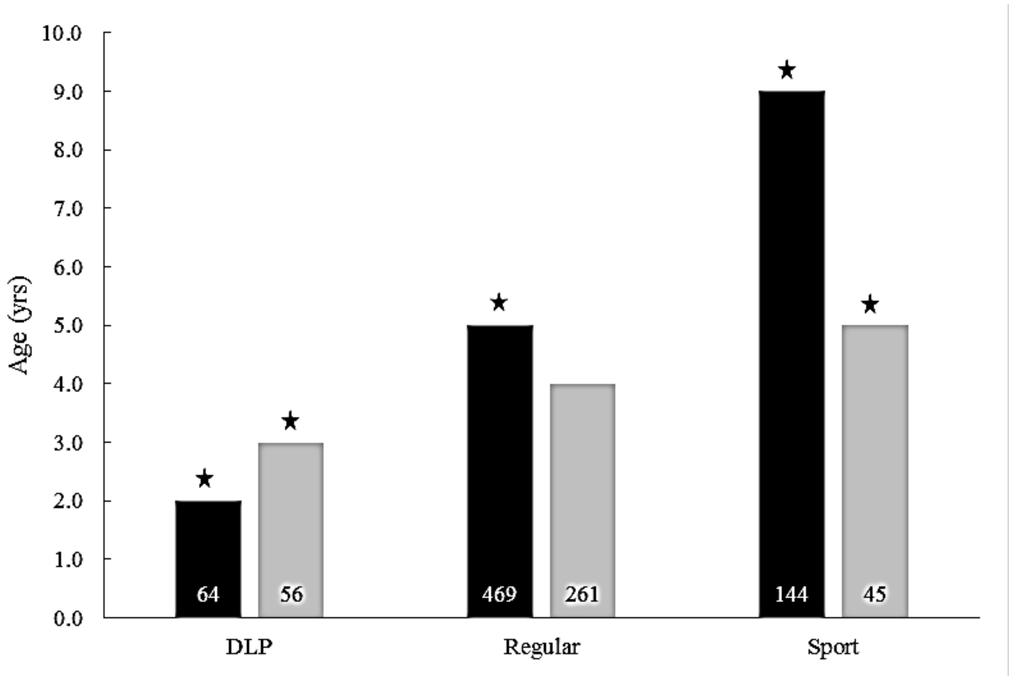


Figure 8.6. Median age of male (black) and female (grey) bears harvested for defense-of-life-and-property (DLP), regular (subsistence) and sport hunting purposes in Baffin Bay, Canada, 1993-2010. Within sexes significant differences denoted by * (Based on Mann-Whitney *U* test with Bonferroni correction for multiple comparisons, two-tailed, alpha = 0.05). Sample sizes within bars.

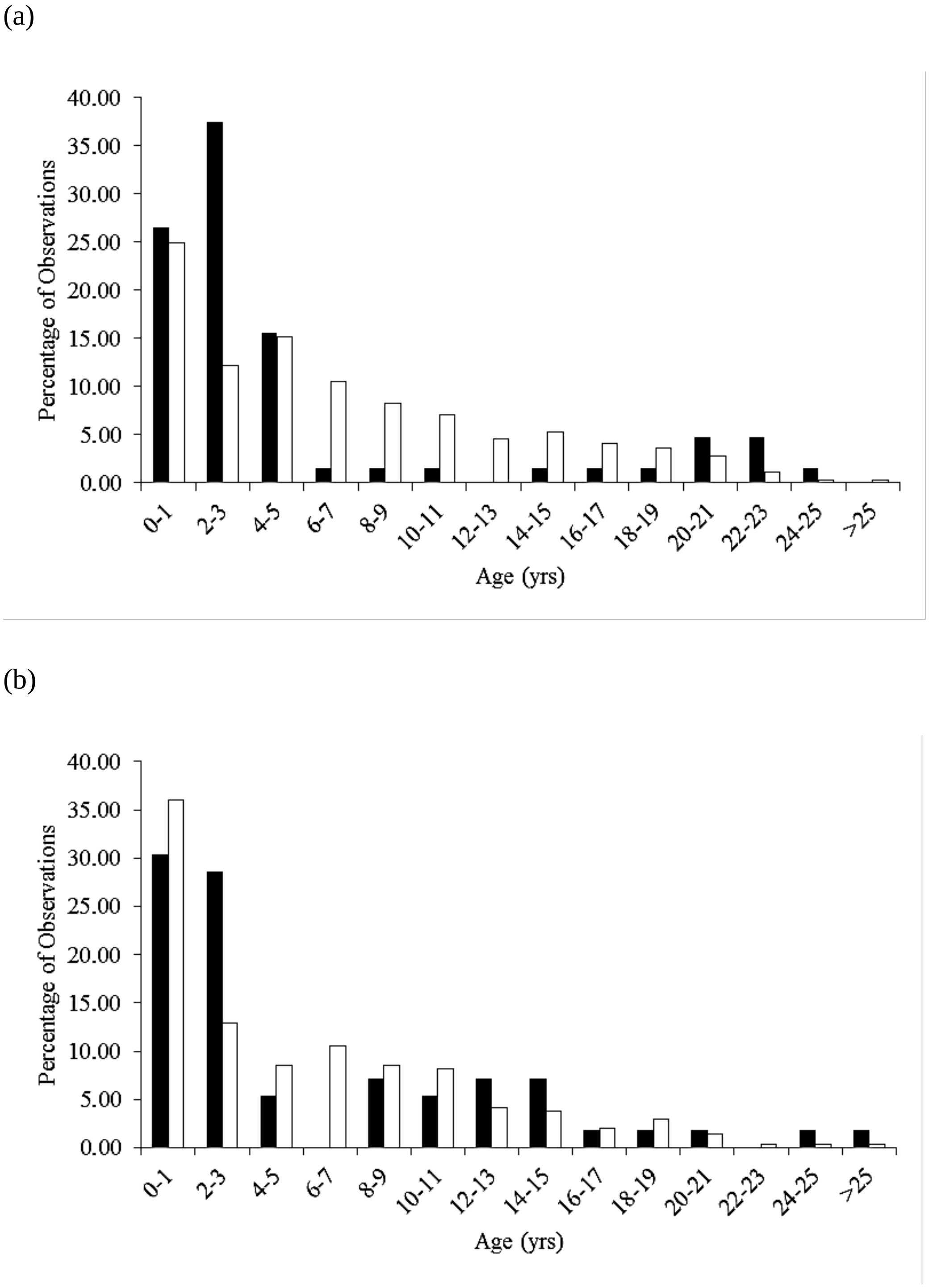


Figure 8.7. The age distribution of (a) 64 male and (b) 56 female bears harvested in defense-of-life-and-property (black) in Baffin Bay (Canada), 1993-2010, relative to the age distribution of 778 bears captured during mark-recapture sampling (white), 1993-1997 (GN unpublished data).

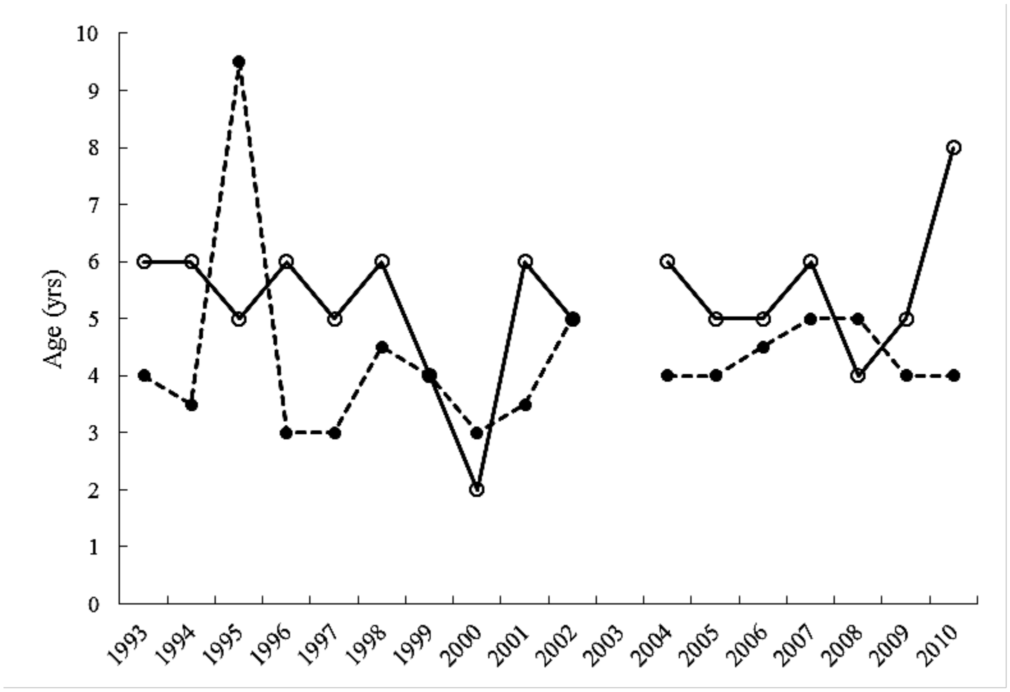


Figure 8.8. Median age of harvested male (solid line) and female (dashed line) polar bears in Baffin Bay, Canada. Regular hunts only.

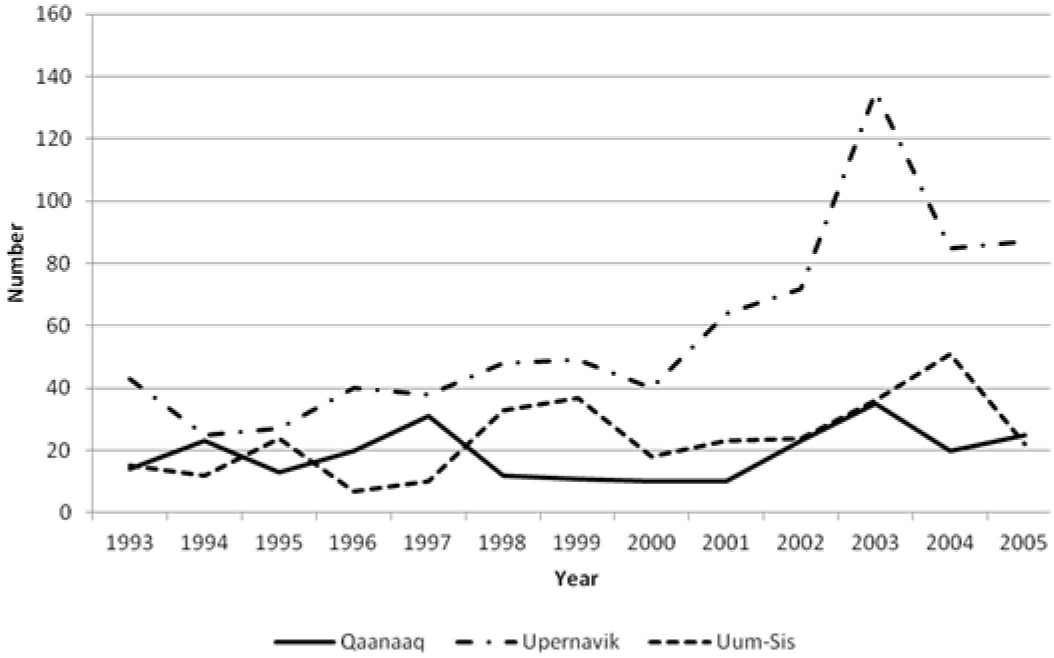


Figure 8.9. The Greenland catch of polar bears from the Baffin Bay subpopulation reported in Piniarneq (1993-2005). The catch is shown for three different regions: (1) The Qaanaaq region representing polar bears taken between 74° 35´ N and 76° 20´ N (i.e., the Melville Bay region sensu latu), (2) the Upernavik area between *ca.* 71° 30´ N and 74° 35´ N, and (3) the areas between Uummannaq and Sisimiut between 66° 55´ N and 71° 30´ N.

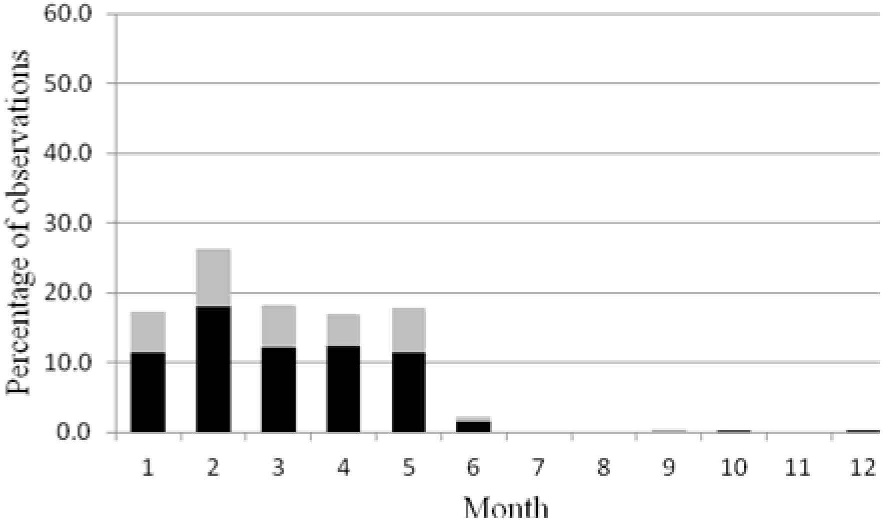


Figure 8.10. Seasonal distribution of catches of polar bears (n = 594; 587 legal and 7 illegal catches) taken in NW and W Greenland from the Baffin Bay subpopulation based on reports from the licensed hunters (2006-2014). Black = Males. Grey = Females. Source: DFHA (in litt 2014 and 2015).

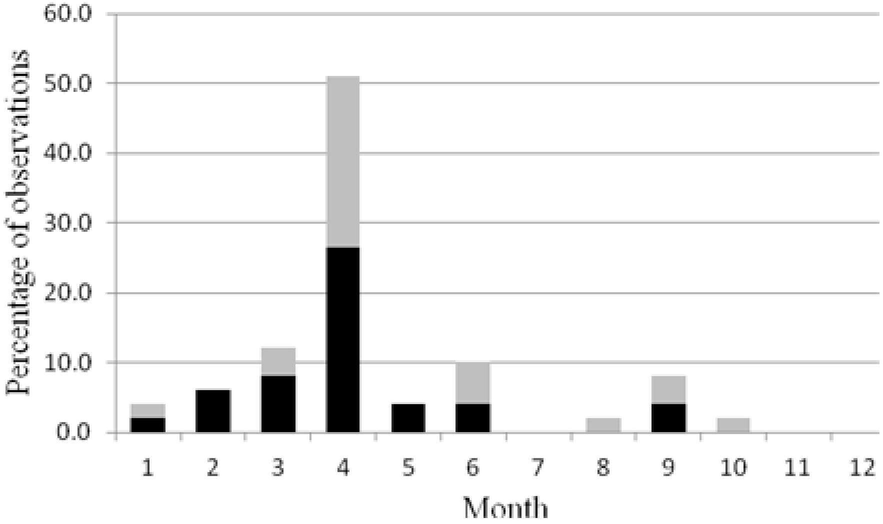


Figure 8.11. Seasonal distribution of catches of polar bears (n = 49; 48 legal and 1 illegal catches) taken in NW Greenland from the Kane Basin subpopulation based on reports from the licensed hunters (2006-2014). Black = Males. Grey = Females. Source: DFHA (*in litt.* 2014 and 2015).

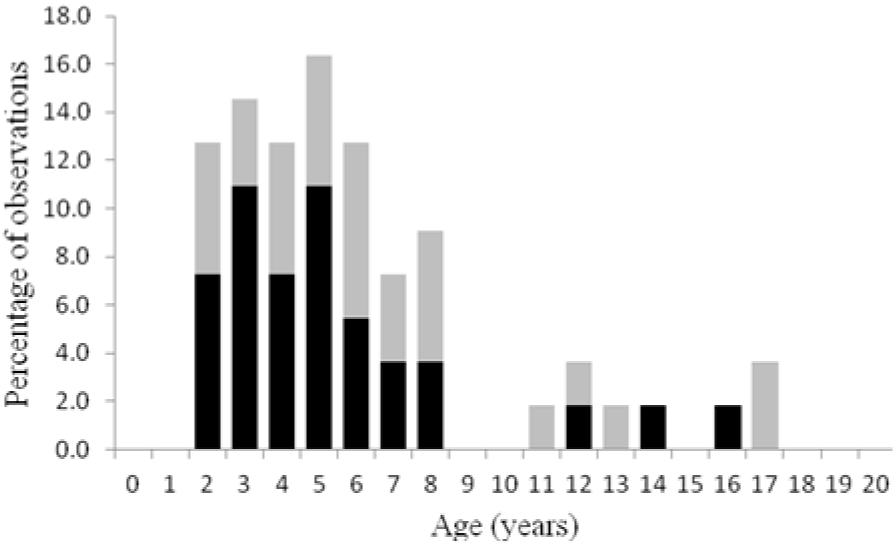


Figure 8.12. Age distribution of a total of 55 polar bears (Grey = Females: n = 25; Black = Males: n = 30) that were taken from the BB subpopulation by Greenland subsistence hunters in 2012 (n = 33) and 2013 (n = 22). Sex was determined genetically.

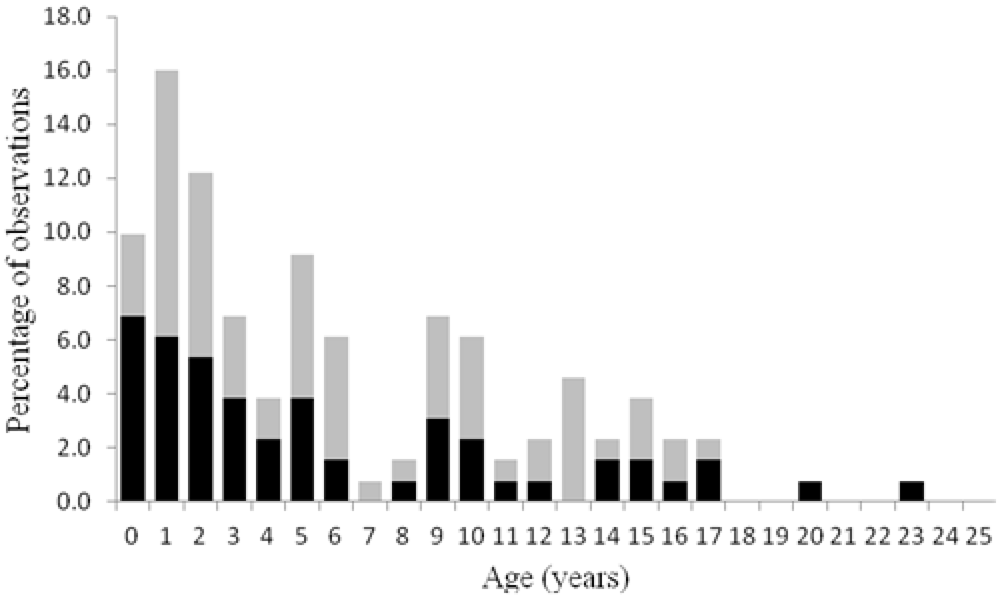


Figure 8.13. Age distribution of 131 polar bears (Grey = Females: n=73; Black = Males: n=58) that were live captured in NW Greenland north of *ca.* 70° N from the BB subpopulation during spring 2009-2013 in connection with the present study (GINR unpublished data).

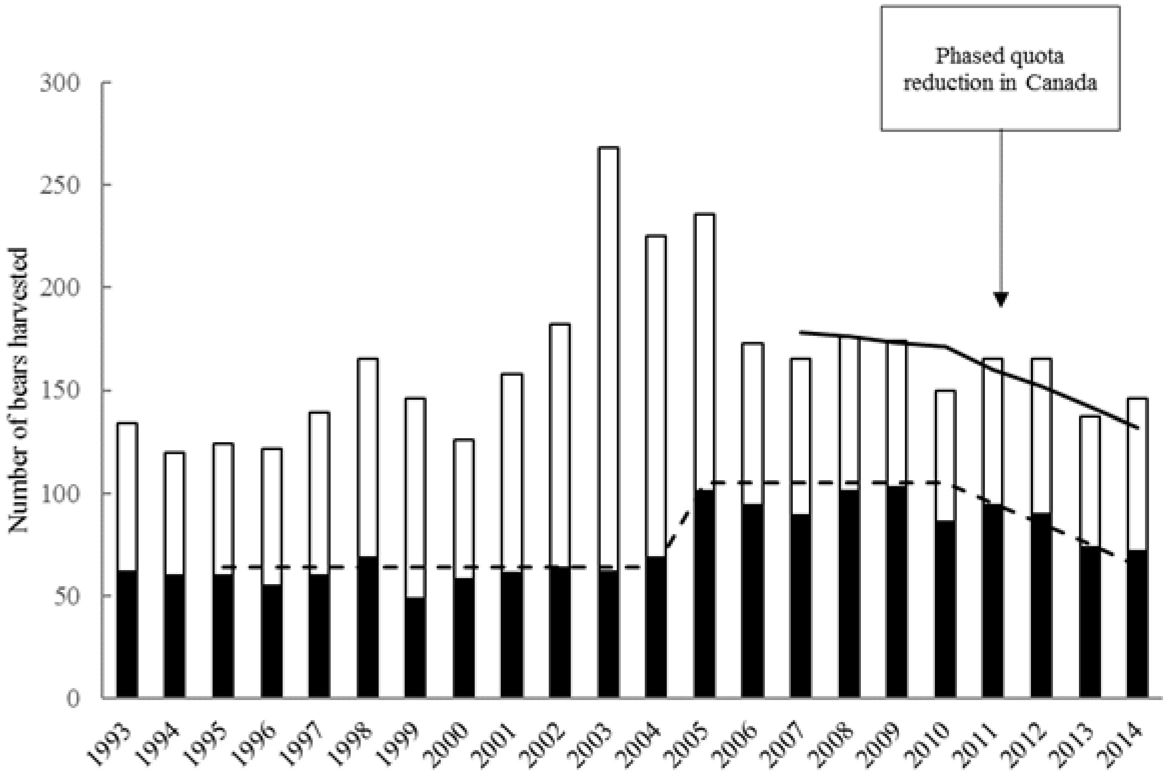


Figure 8.14. Combined Canadian (black) and Greenlandic (white) harvest of polar bears from the Baffin Bay subpopulation. Total allowable harvest in Canada (dashed line) and Canada-Greenland combined total permitted harvest (solid line) levels are also shown.

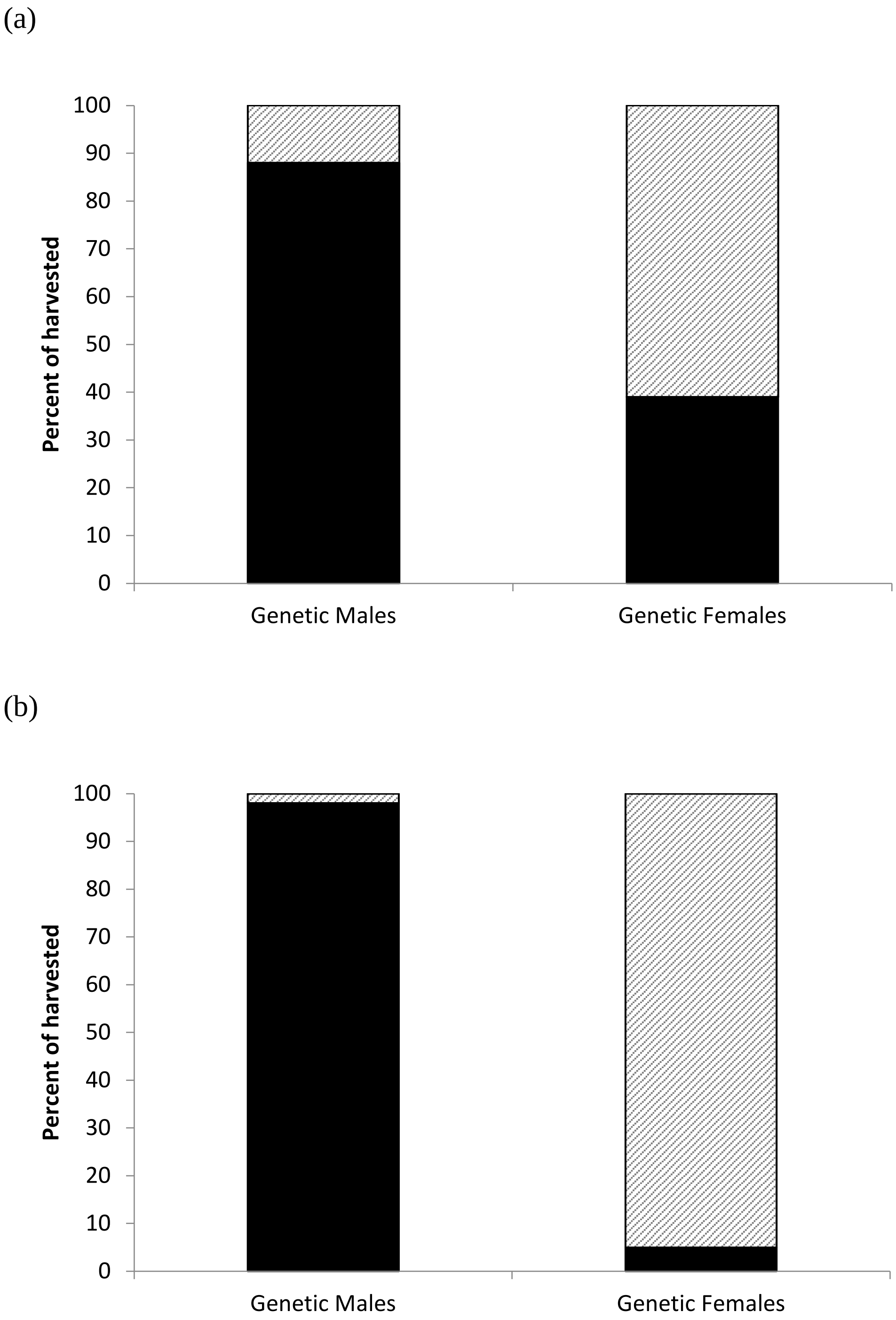


Figure 8.15. Accuracy of gender reporting for polar bears harvested in Baffin Bay and Kane Basin (2011-14) in (a) Greenland and (b) Nunavut. Bears reported as males and females are indicated in black and hatched, respectively

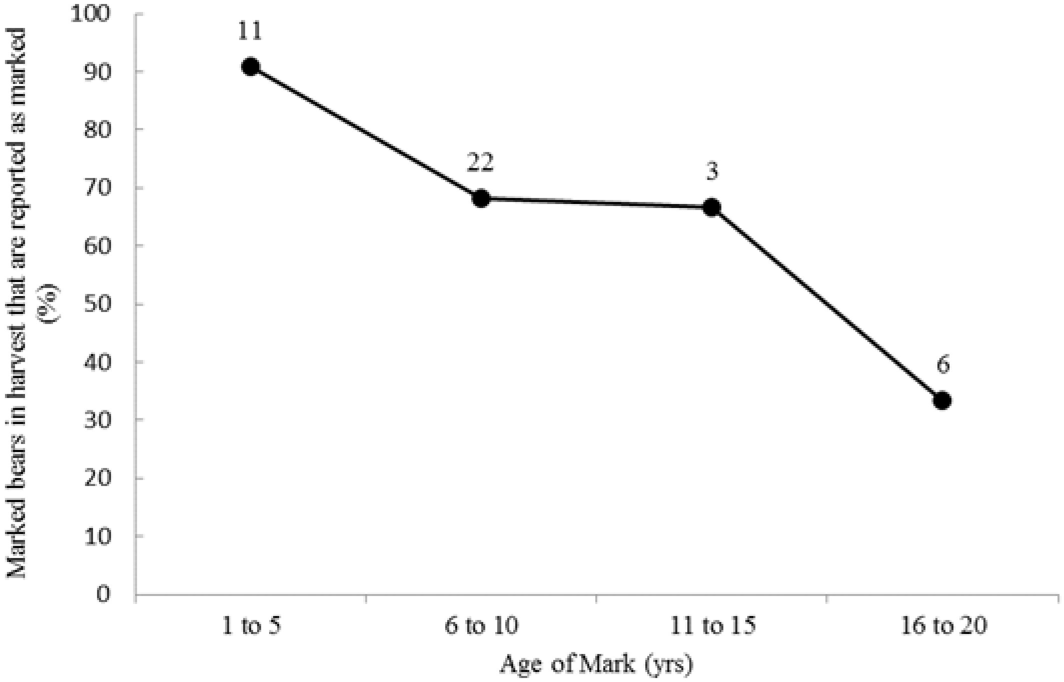


Figure 8.16. Accuracy of reporting of physically marked (tag and tattooed) bears in the harvest in Canada and Greenland. Data are for bears physically marked in Baffin Bay (1990-97) and Davis Strait (2005-07) that were recovered in the harvest 2011-14. Sample sizes above points.

CHAPTER 9

HABITAT AND HABITAT USE IN KANE BASIN

KEY FINDINGS

- The annual cycle of sea-ice habitat in KB has shifted from a largely year-round ice platform (>30% coverage in summer) to a cycle that resembles the seasonal ice ecoregion with complete melt-out in summer (<5% coverage).
- Sea ice is retreating earlier in Kane Basin spring by 7 days/decade, and advancing later in fall by 5-6 days/decade and length of summer (number of days from retreat to advance) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 5-6 percent/decade.
- The KB subpopulation has responded to changing sea-ice conditions with broad movement and habitat use patterns that are more similar to those of bears in seasonal sea-ice ecoregions (e.g., expanded seasonal home ranges, see Chapter 2 and use of lower sea-ice concentrations in summer and fall).
- Four-day movement rates in KB are lower than those in BB and have a less pronounced seasonal cycle. There are no significant differences in movement rates between the 1990s and 2000s except in October where rates were higher.
- Land use in KB during summer remains intermittent because some sea ice remains in fjords and coastal areas. No on-land arrival and departure dates could be determined from satellite telemetry.
- Three maternity dens were found in KB in the 2000s. All were located on Ellesmere Island. There was no significant difference in maternity denning duration, entry dates, or exit dates between the 1990s and 2000s.
- Overall, the movement (Chapter 2) and habitat results combined with reproductive metrics (Chapter 11) and body condition (Chapter 12), indicate that KB bears are experiencing more seasonal sea-ice ecoregion-like conditions, which since the 1990s may have increased overall biological productivity of the area.

9.1. Introduction

The Kane Basin (KB) subpopulation of polar bears occurs between the Canadian Arctic Archipelago and Northwest Greenland, referred to by some as the Arctic Archipelago ecoregion, historically characterized by year-round sea-ice habitat in islands in the Canadian High Arctic and in the Kane Basin region in Northwest Greenland (Hamilton et al. 2014). Ice in this ecoregion is largely multi-year, except in Kane Basin where much of the ice is annual though partially present year-round. In contrast to the Canadian Arctic Archipelago, the Kane Basin region is also characterized and influenced by influx of multi-year ice flowing to the area from the Arctic Ocean. In these areas, sea ice remains along coastal areas in summer providing temporally-stable hunting opportunities for polar bears during summer. This is in contrast to the seasonal ecoregion (Baffin Bay), where sea ice disappears almost completely in summer and bears are forced onshore (Amstrup et al. 2008, this study).

Earlier studies comparing movements and habitat selection of polar bears in the archipelago and seasonal ecoregions documented differences in the influence of sea-ice regimes on movements and habitat use. Ferguson et al. (2000) showed that bears in the archipelago regions (including KB) are strongly influenced by the temporal and spatial distribution of land fast ice around islands providing a consistent habitat across the annual cycle, though possibly lower density year-round prey base. This was in contrast to seasonal ecoregion bears that are strongly influenced by the availability of seasonal sea ice, resulting in a more productive but temporally limited the duration of feeding. Bears in these two ecoregions also demonstrate contrasting movement patterns: bears in the archipelago region (e.g., KB) tended to have smaller home ranges and greater irregularity in movement patterns because they utilize small coastal areas around the complex land masses (or in case of the Kane Basin region fast ice in fjords or fields of pack ice in Nares Strait-Kane Basin), whereas bears in the seasonal sea-ice region (e.g., BB) had large home range sizes (Ferguson et al. 1998) and more regular movement patterns associated with the seasonal growth and recession of sea ice.

Recent work examining Global Climate Model projections of sea-ice habitat for polar bears in the Canadian Arctic Archipelago suggest that conditions will shift towards a seasonal-sea-ice ecoregion before the end of the 21st century (Hamilton et al. 2014). This shift is expected to initially increase productivity in the archipelago system, as thicker ice is replaced by thinner annual ice increasing production blooms and prey platforms for ice seals. Such a change would be expected to be associated with changes in movement patterns and habitat use similar to that exhibited by bears in the seasonal ecoregion. To date, no studies have quantified such behavioral changes. In this chapter we document changes in sea-ice habitat of KB over the satellite record, and quantify change in habitat use using satellite telemetry data collected from collared adult females in the 1990s and 2000s.

9.2. Methods

We refer to “Kane Basin” as the region within the boundaries of the Kane Basin (KB) polar bear management unit (PBSG 2010; Figure 1.1.) that encompasses the northern part of Smith Sound, Nares Strait and Kane Basin and the southern part of Kennedy Channel and adjacent fjords on eastern Ellesmere Island and in NW Greenland.

KB polar bears were captured and tagged between mid-April and early May 2012-2013 (Figure 9.1). Field operations were based out of the Alexandra Fjord station on Ellesmere Island. A total of 34 bears were furnished with satellite-transmitters in KB. Twenty were adult females who received a satellite collar (Table 9.1) and 14 were adult males or subadults (given satellite radio ear tags). Data from adult females were combined with a historical data set from 12 adult females collared between 1992 and 1994 on the west side of KB in the fjords and fast ice (Taylor et al. 2001). Only bears captured within the KB subpopulation boundaries (PBSG 2010) were included in the analysis. The eastern side of the Kane Basin region was surveyed during 1994-1997 but no female polar bears for collar deployment were found in these areas (Taylor et al. 2001). Hence, radio collars were only deployed on the west side of KB in the 1990s (ibid.). Given the different distribution of collar deployments between decades we tested for differences in area use and mean latitude using only bears captured in West KB in the 2000s and found no differences (see Chapter 2). Therefore, all bears from the 1990s were compared to all bears in the 2000s in habitat models. Bears in all decades largely utilized KB, however in both decades bears moved from KB into BB, LS and in the 2000s out of KB into the Arctic Basin.

Sea ice

Data sources and methods for sea-ice analysis in KB are the same as those described for BB (see Chapter 4 and Stern and Laidre 2016). The entire KB region, as defined by PBSG (Figure 1.1), was used for the sea-ice habitat analyses. The area was roughly divided into two distinct parts. The northern part, or Kane Basin proper, is bounded on the south by Smith Sound, consisting of almost entirely of shallow (< 300 m) water. The southern part contains the northern part of the North Water Polynya, and is bounded on the south by roughly 77° N latitude where it adjoins northern Baffin Bay. The southern part of Kane Basin consists mostly of deep (> 300 m) water on the Ellesmere Island side and shallow water on the Greenland side.

Habitat Use Analyses

Methods for KB movement rates, habitat covariates, and multivariate RSF modeling are the same as those described in Chapter 4 for BB. In the multivariate terrestrial RSF we did not include the variable pertaining to the distance to the smoothed Baffin Island coastline. In the multivariate sea-ice RSF we excluded the covariates representing the distance to the 15 and 50% sea-ice concentration because in some months in KB the distance resulted in unrealistic potential movements of bears.

We attempted to quantify the dates of arrival and departure on land in KB as described for BB in Chapter 4. This analysis was confounded by the fact that KB bears have access to sea ice much of the summer. Thus KB bears used land intermittently and it was not possible to quantify a specific date where bears arrived on shore and did not leave. There were no potential swimming events identified in KB. Maternity denning analyses were conducted with the same methods as described in Chapter 4 for BB.

9.3 Results and Discussion

Sea-ice habitat

Kane Basin consisted of 81 SSMI sea ice grid cells (53 × 103 km^2); 68% had a mean depth < 300 m, 32% had a mean depth > 300 m. The seasonal cycle of the sea ice in KB has changed dramatically since the 1990s (Figure 9.2). In the 1990 sea ice did not disappear from KB and in summer months >50% of KB was ice covered. In the 2000s, there has been greater extent of sea-ice loss and KB reaches ~5% coverage in summer. The sea-ice loss has been most pronounced from May and through the late fall, and there are few differences in sea-ice coverage between January and April. Trends in the four sea-ice metrics (described in Chapter 4) are provided in Table 9.2. All trends are statistically significant and show a loss of sea-ice habitat. In Kane Basin as a whole, sea ice is retreating earlier in spring by 7 days/decade, and advancing later in fall by 5-6 days/decade (Figure 9.3). The length of summer (number of days from retreat to advance) is increasing by 12 days/decade, and the mean sea-ice concentration during June-

October is decreasing by 5-6 percent/decade (Figure 9.4ab). The trend in the date of spring sea-ice retreat is apparently stronger for the shallow depths of KB than for all depths. The trend in the date of fall sea-ice advance is the same for both depth categories. The downward trend in the June-October sea-ice concentration is apparently stronger for the shallow depths.

The year-to-year variability in sea-ice metrics for Kane Basin was larger than for Baffin Bay, i.e., the scatter about the trend lines was larger. There are several reasons for this variability: (1) the North Water Polynya is an area of dynamic sea-ice activity that affects Kane Basin; (2) there is typically an “ice arch” north of Kane Basin that determines whether ice lingers in the basin (arch intact) or is flushed out (arch collapses); and (3) Baffin Bay is much larger than Kane Basin and so is less affected by such relatively small-scale phenomena as (1) and (2).

The trend in the annual number of ice-covered days in Kane Basin is between −5 and −15 days/decade for most of the areas with shallow depths (Table 9.2). For the southern portion of Kane Basin, the trend is steeper than −15 days/decade on the Greenland side and there is almost no trend on the deeper Ellesmere Island side. Thus, the pattern of extreme sea-ice loss in Baffin Bay along the coast of Greenland (see Chapter 4) extends northward into the southeast portion of Kane Basin.

Spring sea-ice melt in the Kane Basin region begins in May in the North Water Polyna, which generally becomes ice-free by July. Kane Basin proper, to the north, generally holds some sea ice all summer. Figure 4.5 shows that on July 15, Kane Basin proper is almost always ice-covered, often with 50% or more sea-ice concentration. The year 2009 was exceptional, when all the ice in Kane Basin was swept out in May and June. In October, sea ice advances from north to south through Kane Basin, but the date of advance is generally trending later (Table 9.2 and Figure 9.2).

Movement rates – In Kane Basin, mean monthly movement rates for adult females were overall lower than in Baffin Bay. In KB in the 1990s, mean monthly movement rates ranged from a low of 3.4 km/day (in August) to a high of 9.4 km/day (in February) (Figure 9.5, Table 9.3). Rates for adult females in the 2000s were similar and ranged from 4.7 km/day (in Septembe) to 6.94 km/day (in November) (Figure 9.5, Table 9.3). Compared to Baffin Bay, there was a substantially less pronounced cycle to movement rates over the year in KB. There were no significant differences in movement rates between decades except in October in the 2000s where rates were higher than the 1990s (Table 9.3).

RSF sea-ice models – The sample sizes of adult females in KB in the 1990s and 2000s were smaller than in BB (12 and 20 bears, respectively). Collars deployed between 2012 and 2013 were removed in April 2014 thus tracking durations in the later period were also shorter. We examined univariate relationships for multiple habitat covariates over the annual cycle of sea ice (Figure 9.7 and 9.8). KB bears in the 1990 used similar sea-ice concentrations as bears in the 2000s between January and May (Figure 9.7). Starting in late spring (June) and continuing through December, KB bears in the 2000s used significantly lower sea-ice concentrations than in the 1990s. This was most pronounced in August-October. Distances from bears to the 15% or 50% sea-ice concentration thresholds varied widely across the annual cycle and were similar between decades, though in the 2000s bears were significantly closer to the 50% sea-ice edge in March and April. There were no large differences in bears' distance to land in either decade, other than bears being closer to land from October-December in the 2000s.

The multivariate RSF model in winter demonstrated adult female polar bears in the 1990s had a strong preference for higher ice concentrations. This preference was not present in the

2000s. In both decades bears had a similar strength of preference for the distance to the 300 m depth contour and preferred shallower depths (more strongly and significantly in the 2000s) (Table 9.4). The multivariate RSF model for spring showed that bears also had a strong significant preference for higher ice concentrations in the 1990s (Table 9.5). The preference was reduced in the 2000s but there was no significant difference between decades. In spring in the 1990s, bears were farther from the shelf break (300 m contour), whereas in the 2000s they were closer to 300 m and this change was significant between decades. There was no preference for depth in either decade in spring. In both decades, there was a preference not to move to land, but this was significantly stronger in the 2000s.

RSF terrestrial models – Adult female use of land was intermittent in KB, thus land use models reflect use of land largely near the shoreline as bears moved on and off sea ice (Figure 9.9, Figure 9.10). The terrestrial models demonstrated that KB bears preferred lower elevations, a preference which has significantly increased in the 2000s. Bears tend to avoid steep slopes in both decades and were significantly less likely to move to sea ice once they were on land (Table 9.6).

Arrival and departure dates – KB is part of the Archipelago ecoregion, which in contrast to the seasonal ice ecoregion, historically does not melt out completely each year. Bears in KB exhibit fundamental differences in their habitat use because of the availability of sea ice between systems. In general, KB bears had access to sea ice for most of the summer, especially in the 1990s though this has been significantly reduced in the 2000s. Some bears utilized fjord ice for most of the summer and never arrived on land, while others spent intermittent time on land. Overall patterns of land use among individuals were not consistent and thus it was not possible to quantify on-land arrival and departure dates. No long-distance swimming events were observed, though one of the swimming events recorded for BB in July 2010 resulted in a BB collared bear arriving on Ellesmere Island after a long distance swim from offshore pack ice in Northern BB.

Kane Basin denning – Nine dens were found from 2012 to 2015 in KB: three maternity dens (Figure 9.11) and six shelter dens (Figure 9.12). In the 1990s data, Ferguson et al. (1997) also found nine dens, of which three were maternity dens and six were shelter dens (Table 9.7). All dens were on land with the exception of one 1990s shelter den that was located on landfast ice nine kilometers from the shore of Ellesmere Island. Most of the dens were located on Ellesmere Island except for three dens on Devon Island. None of the adult females from KB denned on Greenland. The minimum latitude for the 1990s dens was 77.94° N, and 77.04° N for the 2000s dens.

There was no significant difference in maternity denning duration ($p = 1$) (Table 9.8, Figure 9.13), entry dates ($P = 0.6$) and exit dates ($P = 1$) (Figure 9.14). Only four of the KB bears in the 2000s provided useable temperature data for inferring exit dates and no temperature data were available from the 1990s. There was no significant correlation between latitude and maternity den entry dates ($\tau = 0.138$, $P = 0.848$) or duration ($\tau = 0.2$, $P = 0.707$). The median first date on land for the n=3 pregnant females in the 1990s was 18 September (SD = 31 days) and in the 2000s was 23 August (SD = 20.8 days; Figure 9.15). The difference between the two time periods was not significant despite the median FDOLs being 27 days apart. The sample size was small and there was considerable variability. Habitat characteristics among maternity dens did not significantly differ between decades (Figure 9.16; Table 9.9, 9.10).

9.4. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T.

DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea-ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Ferguson, S. H., M. K. Taylor, E. W. Born, and F. Messier. 1998. Fractals, sea-ice landscape, and spatial patterns of polar bears. Journal of Biogeography 25:1081–1092.

Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. Ecology 81:761–772.

Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS ONE 9(11): e113746. doi:10.1371/journal.pone.0113746.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere Discussion Papers, in review.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Table 9.1. Breakdown of adult females (AF) collared in the Kane Basin region in the 1990s and 2000s. AF = adult female, AM = adult male, COY = Cub of the Year, YRL = Yearling, 2YR = 2 Year old cub.

		AF alone	**AF+AM**	**AF+COY**	**AF+YRL**	**AF+2YR**	**Sum**
1990s	KB	3	0	5	3	1	12
2000s	KB	5	1	7	3	4	20

Table 9.2. Trends in date of spring sea-ice retreat, fall sea-ice advance, fall – spring dates, and summer (June-Oct) sea-ice concentration in Kane Basin (all depths, and depths < 300 m). All trends are statistically significant at the 99% level according to a 2-sided F test, except the date of spring retreat (all depths), which is significant at the 95% level.

Baffin Bay region	Trend in date of spring ice retreat (days/decade)	Trend in date of fall ice advance (days/decade)	Trend in fall – spring (days/decade)	Trend in ice con. June-October (percent/decade)
All depths	−6.8	+5.6	+12.4	−5.4
Depths < 300 m	−9.7	+5.5	+15.2	−6.9

Table 9.3. Mean monthly movement rates in KB for radio-collared adult female polar bears (AFs) in the 1990s and 2000s. One SE about the mean is given. We conducted a parametric test of significance between decades; bold = significant at the 5% level.

Month	Mean 1990s	SE 1990s	N 1990s	N steps	Mean 2000s	SE 2000s	N 2000s	N steps	t-test P value
1	3.62	3.78	5	21	4.87	7.59	12	98	0.8691
2	9.39	9.74	4	16	5.9	6.99	11	93	0.4504
3	8.52	9.17	7	35	5.63	6.73	13	117	0.3802
4	5.79	7.28	13	80	5.66	6.57	13	138	0.5197
5	7.35	10.19	12	92	6.75	5.87	21	230	0.6035
6	5.31	7.2	12	85	5.22	4.47	18	193	0.6012
7	3.6	3.19	11	81	5.75	4.13	14	166	0.1011
8	3.41	3.45	10	62	4.84	4.66	14	132	0.5136
9	3.91	5.9	10	63	4.5	4.57	14	132	0.1466
10	4	3.07	9	51	6.42	7.1	14	141	**0.0394**
11	7.87	8.96	9	40	6.94	6.57	13	118	0.986
12	7.87	8.82	8	36	4.68	5.22	12	116	0.3835

Table 9.4. Sea-ice resource selection function (RSF) model coefficients for the Winter season in KB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled such by a certain number of units for more meaningful interpretation. "Mean ice conc.10" is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. "Dist to 50%" is the distance to the 50% sea-ice concentration scaled by 100 km. "Dist to 300 m" is the distance to the 300 m depth contour scaled by units of 100 m. "Depth.100" is the absolute value of bathymetry scaled by 100 m. "Land" is the variable that describes the tendency of a bear to move from sea ice on to land.

	1990s coef	**SE**	**p-value 1990s**	**2000s coef**	**SE**	**p-value 2000s**	**p-value delta**
Mean ice conc.10	0.479	0.564	0.3963	0.074	0.192	0.699	0.4976
Dist to 50%.100	-0.014	0.134	0.918	-0.032	0.115	0.7823	0.9181
Dist to 300 m.100	1.82	0.824	0.0272	1.738	0.826	0.0353	0.9436
Depth.100	-0.034	0.078	0.6579	-0.065	0.084	0.4387	0.7892
Land	3.377	4.895	0.4903	1.108	1.504	0.4615	0.6577

Table 9.5. Sea-ice resource selection function (RSF) model coefficients for the Spring season in KB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled such by a certain number of units for more meaningful interpretation. “Mean ice conc.10” is the mean sea-ice concentration around the bear in a circular radius scaled by 10%. “Dist to 50%” is the distance to the 50% sea-ice concentration scaled by 100 km. “Dist to 300 m” is the distance to the 300 m depth contour scaled by units of 100 m. “Depth.100” is the absolute value of bathymetry scaled by 100 m. “Land” is the variable that describes the tendency of a bear to move from sea ice on to land. Bold = significant at the 5% level.

	1990s coef	**SE**	**P-value 1990s**	**2000s coef**	**SE**	**P-value 2000s**	**P-value delta**
Mean ice conc.10	0.255	0.062	**<0.001**	0.223	0.058	**0.0001**	0.7107
Dist to 50%.100	-0.007	0.051	0.8867	-0.118	0.071	0.0949	0.2031
Dist to 300 m.100	1.324	0.502	**0.0084**	-0.997	0.483	**0.0392**	<0.001
Depth.100	-0.159	0.043	**0.0002**	-0.102	0.051	**0.0451**	0.4
Land	-0.349	0.476	0.4639	-0.588	0.469	0.2107	0.7208

Table 9.6. Terrestrial resource selection function (RSF) model coefficients for the Summer season in KB using CLOGIT. The P-value delta is for the interaction between the 1990s to the 2000s for each covariate within the multivariate model. Coefficients are scaled such by a certain number of units for more meaningful interpretation. "Elev.100" is elevation scaled by units of 100 m. "Slope.10" is slope in degrees scaled by units of 10 degrees. "Aspect.10" is aspect scaled by units of 10 degrees. "Not Land" is the tendency of a bear to move from land on to sea ice. Note not all bears used land in summer and land-use was intermittent. Bold = significant at the 5% level.

	1990s coef	**SE**	**P-value 1990s**	**2000s coef**	**SE**	**P-value 2000s**	**P-value delta**
elev.100	-0.452	0.07	**<0.001**	-0.74	0.054	**<0.001**	0.001
slope.10	0.256	0.136	0.0594	0.316	0.07	**<0.001**	0.692
aspect.10	-0.019	0.013	0.1621	0.007	0.007	0.3166	0.088
NotLand	-0.443	0.317	0.1619	-1.186	0.166	**<0.001**	0.03

Table 9.7. Number of polar bear maternity and shelter dens in Kane Basin in the 1990s and 2000s.

All Dens		Maternity Dens		Shelter Dens	
1990s	9	1990s	3	1990s	6
2000s	9	2000s	3	2000s	6
Total	18	Total	6	Total	12

Table 9.8. Summary table of the phenology for Kane Basin polar bear maternity dens.

	1990s **Maternity Dens (*n* = 3)**			**2000s** **Maternity Dens (*n* = 3)**		
	Entry DOY	Exit DOY	Duration (# days)	Entry DOY	Exit DOY	Duration (# days)
Mean	279	78.3	164.3	274	77.7	168.7
Min	274	69	145	252	65	144
Max	289	89	180	301	88	184
Median	274	77	168	269	80	178
SD	8.7	10.1	17.8	24.9	11.7	21.6

Table 9.9. Summary table of the habitat characteristics for Kane Basin polar bear maternity and shelter dens. Elev. = elevation (meters), Asp. = aspect (degrees), CoastDist = distance to nearest coastline (kilometers).

	All Maternity Dens (*n* = 6)				**All Shelter Dens (*n* = 12)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	327	165.8	13.4	5.2	366.6	141.8	15.4	3.4
Min	9	28.3	3.7	0.7	6	5.5	1.8	0.1
Max	506	229	17.8	12.5	855	350	36.9	8
Median	408	186.8	15.9	3.4	318	168.3	12	2.8
SD	188.4	69.8	5.4	4.8	274.8	126.3	10.8	2.4
	1990s Maternity Dens (*n* = 3)				**1990s Shelter Dens (*n* = 6)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	207.7	201	11.9	2.1	386.7	134	12.9	2.6
Min	9	185.9	3.7	0.7	6	12	1.8	0.1
Max	422	229	16.1	4.3	855	349.7	36.9	5.5
Median	192	188.1	15.8	1.2	257	105.7	9.9	2.3
SD	206.9	24.3	7.1	1.9	378.5	137.4	12.4	2.1
	2000s Maternity Dens (*n* = 3)				**2000s Shelter Dens (*n* = 6)**			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	446.3	130.6	15	8.3	346.5	149.5	17.9	4.2
Min	394	28.3	10.7	2.6	149	5.5	7.2	1.6
Max	506	187.7	17.8	12.5	500	350	32.5	8
Median	439	175.9	16.5	9.7	355.5	168.3	16.4	4.1
SD	56.4	88.8	3.8	5.1	148.2	126.7	9.3	2.5

Table 9.10. Results of the two-sample Mann-Whitney U tests comparing habitat characteristics for Kane Basin polar bear maternity and shelter dens.

	Maternity Dens (*n* = 6)		**Shelter Dens (*n* = 12)**	
	W	***p*-value**	**W**	***p*-value**
Elevation	8	0.2	20	0.818
Slope	7	0.4	25	0.31
Aspect	1	0.2	17	0.937
Distance to Coast	8	0.2	24	0.394

Figure 9.1. Distribution of capture locations for polar bears in Kane Basin (KB) in each decade. See Table 9.1 for sample sizes in each year. Note in 1990s bears in KB were captured on the west side of KB whereas 2000s bears were captured on both the east and west side.

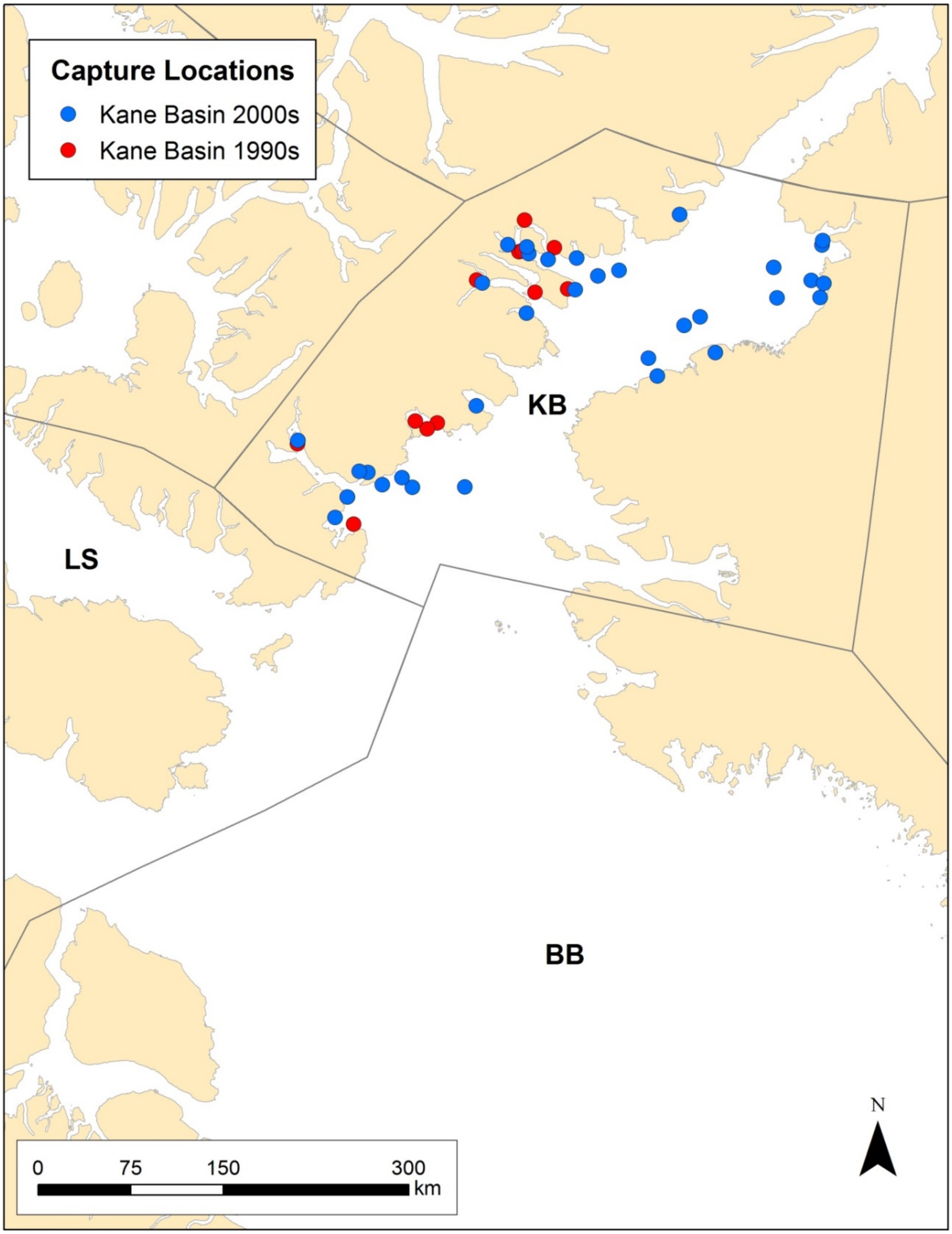


Figure 9.2 Sea-ice area in Kane Basin (all depths) for the years 1979-2014 (gray curves). Two six-year averages are also shown (colored curves). The threshold for defining the dates of sea-ice retreat and advance (middle horizontal dotted line) is halfway between the average March sea-ice area (upper dotted line) and the average September sea-ice area (lower dotted line).

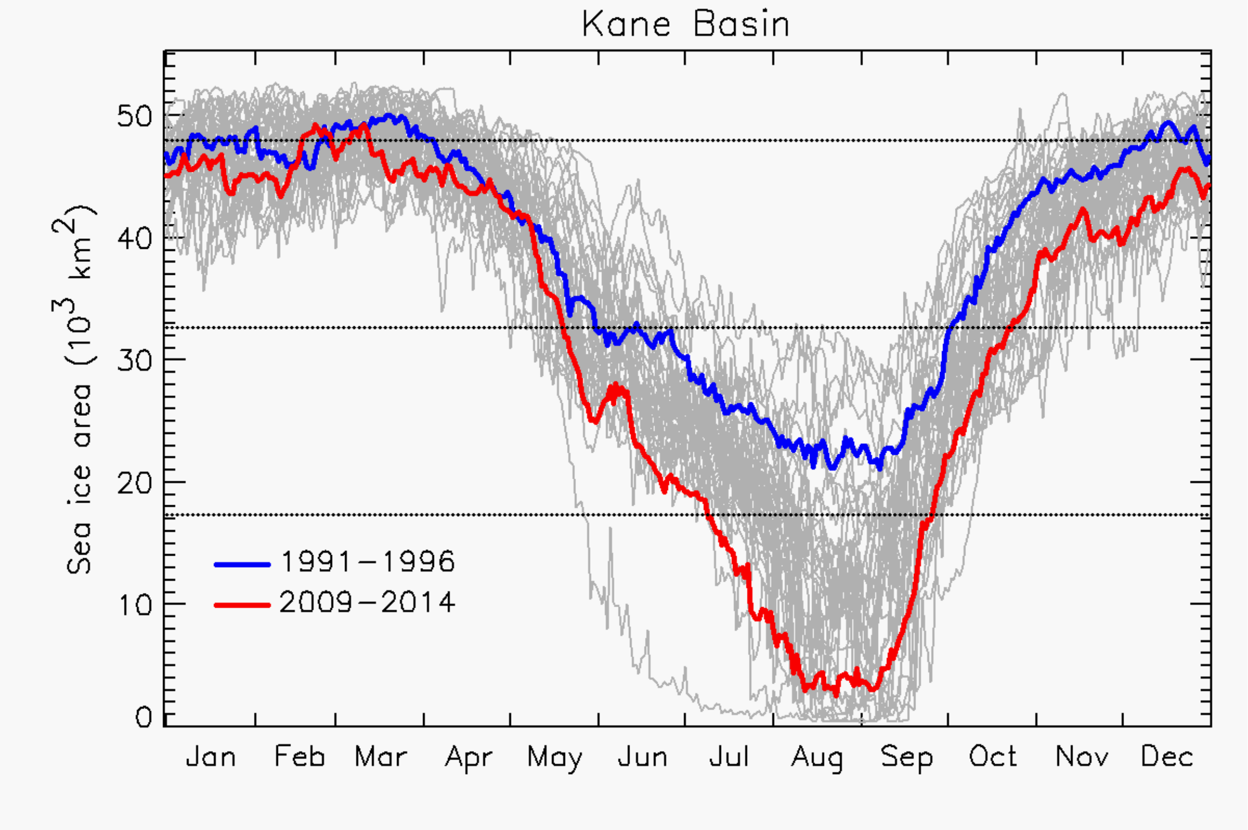


Figure 9.3 Day of spring sea-ice retreat (red circles), fall sea-ice advance (blue circles), and the interval between them (green lines), for Kane Basin (all depths), 1979-2014. Least-squares fits to spring and fall dates are shown (red and blue lines). Trends are given in Table 9.2.

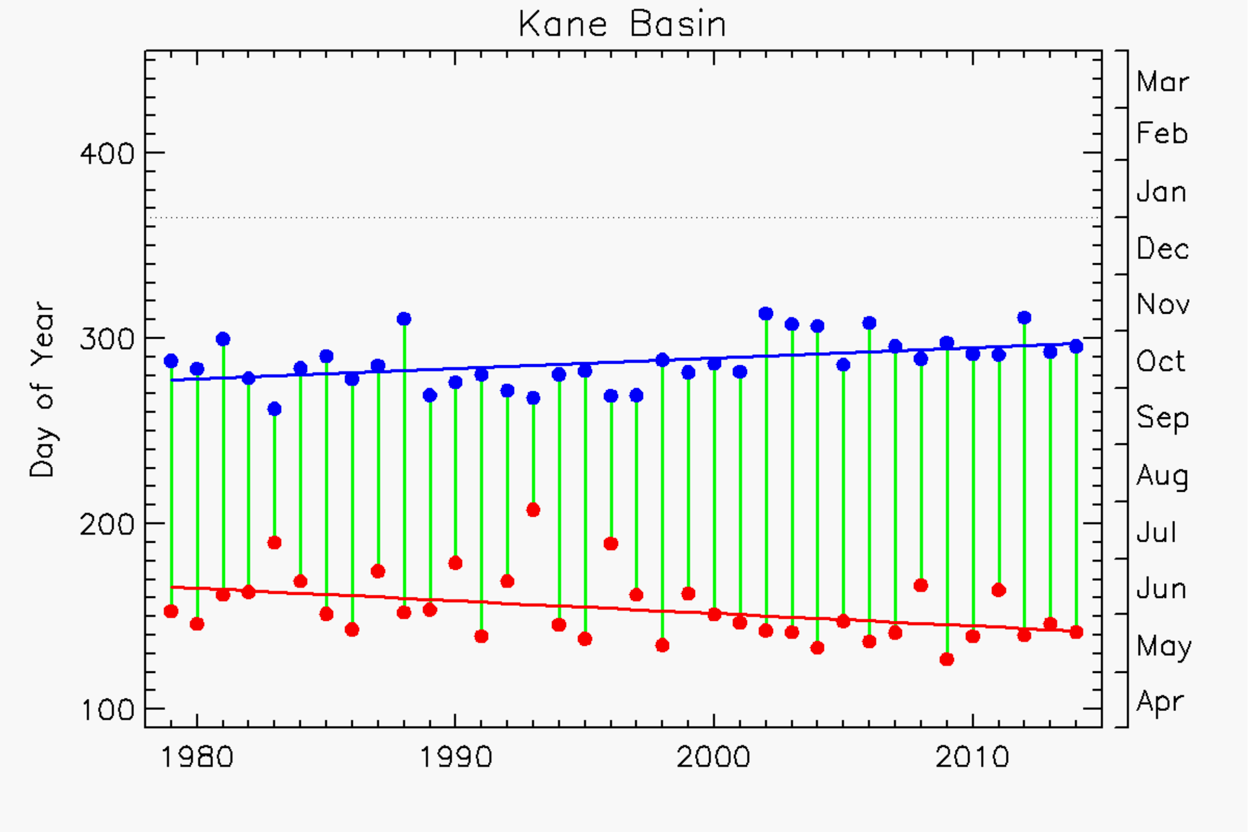
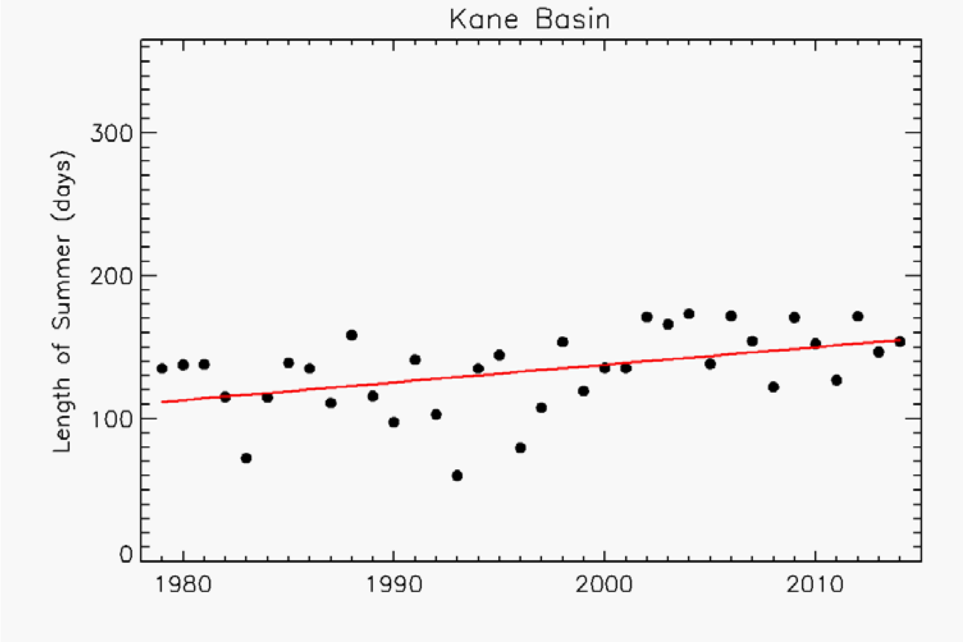


Figure 9.4. Length of summer (a) and mean sea-ice concentration during June-October (b) for Kane Basin (all depths), 1979-2014. Length of summer is the interval from spring sea-ice retreat to fall sea-ice advance (see Figure 9.2, green lines). Least-squares fits are shown (red lines); trends are given in Table 9.2.

(a)



(b)

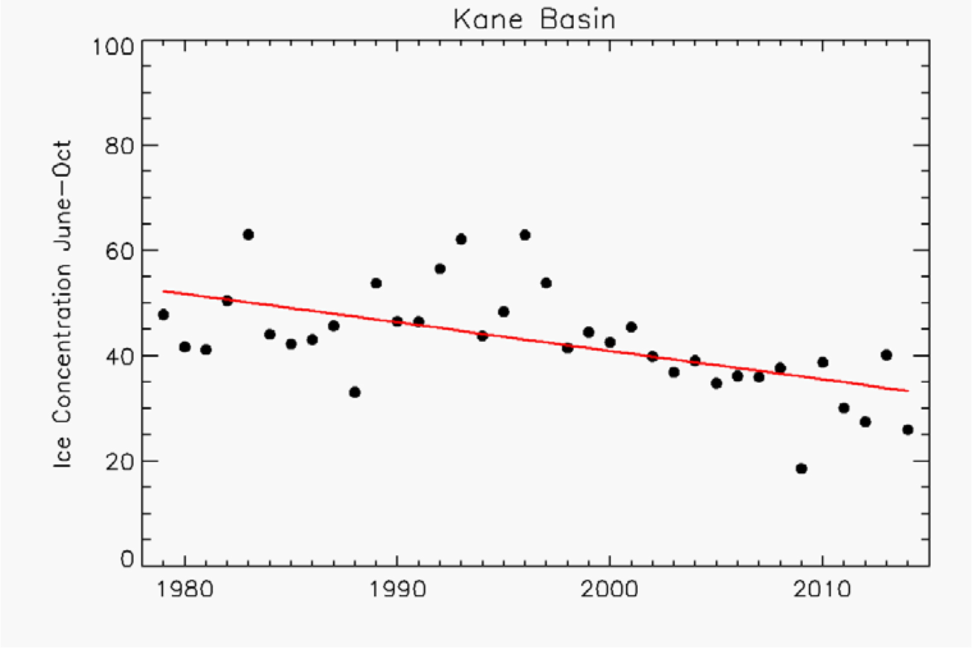


Figure 9.5. Movement rate of KB adult female bears (km/day) in the 1990s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

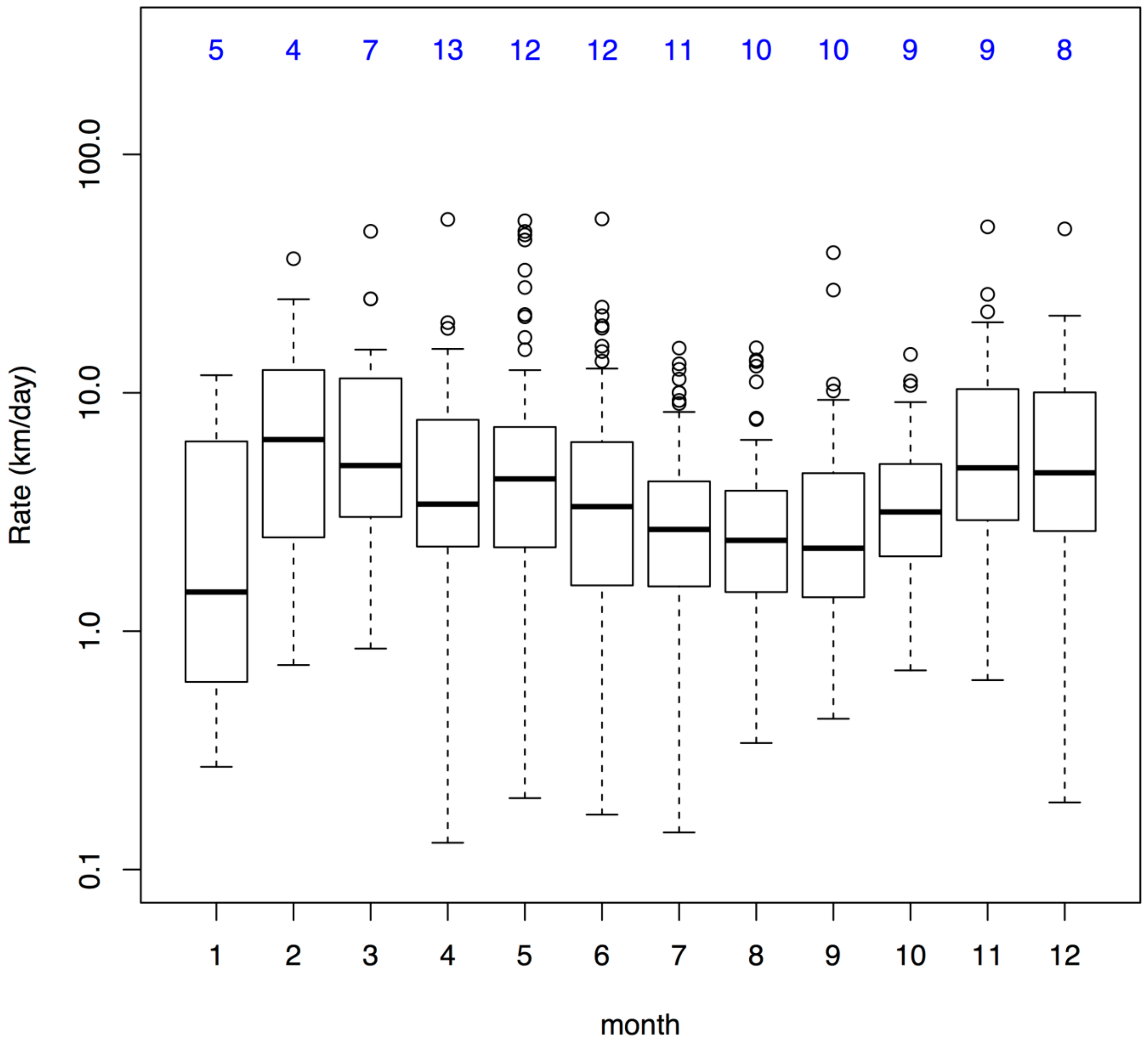


Figure 9.6. Movement rate of BB adult female bears (km/day) in the 2000s. Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

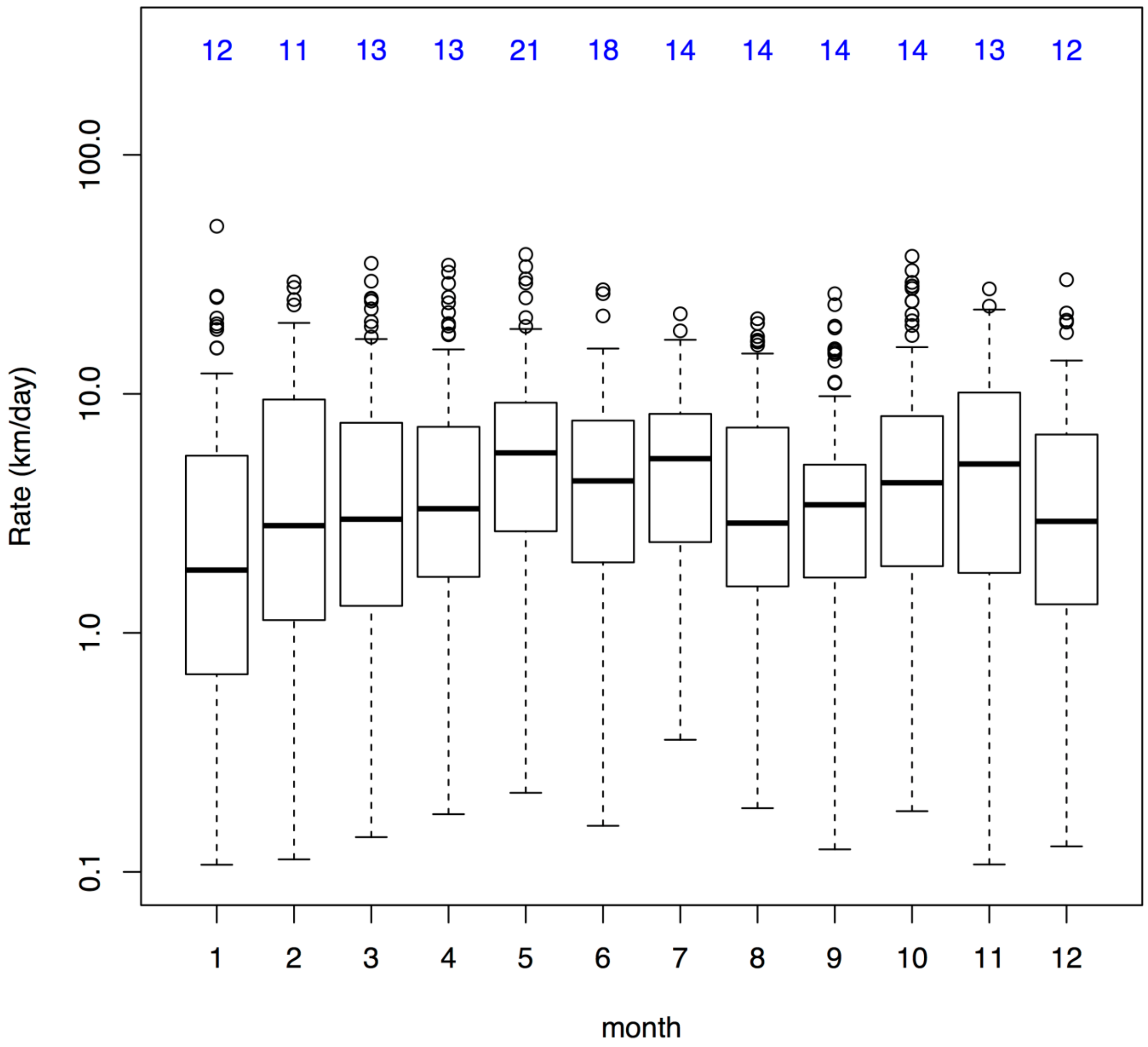


Figure 9.7. 1990s and 2000s adult female polar bear habitat use in KB for each of four sea-ice habitat variables: sea-ice concentration in small buffer, distance to 15% sea-ice concentration, distance to 50% sea-ice concentration, and distance to the nearest land. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer) used in the analysis. SSM/I sea-ice concentration is used in both decades. Months 8 -10 also represent land use by bears. The small numbers indicate the number of movements captured within each months.

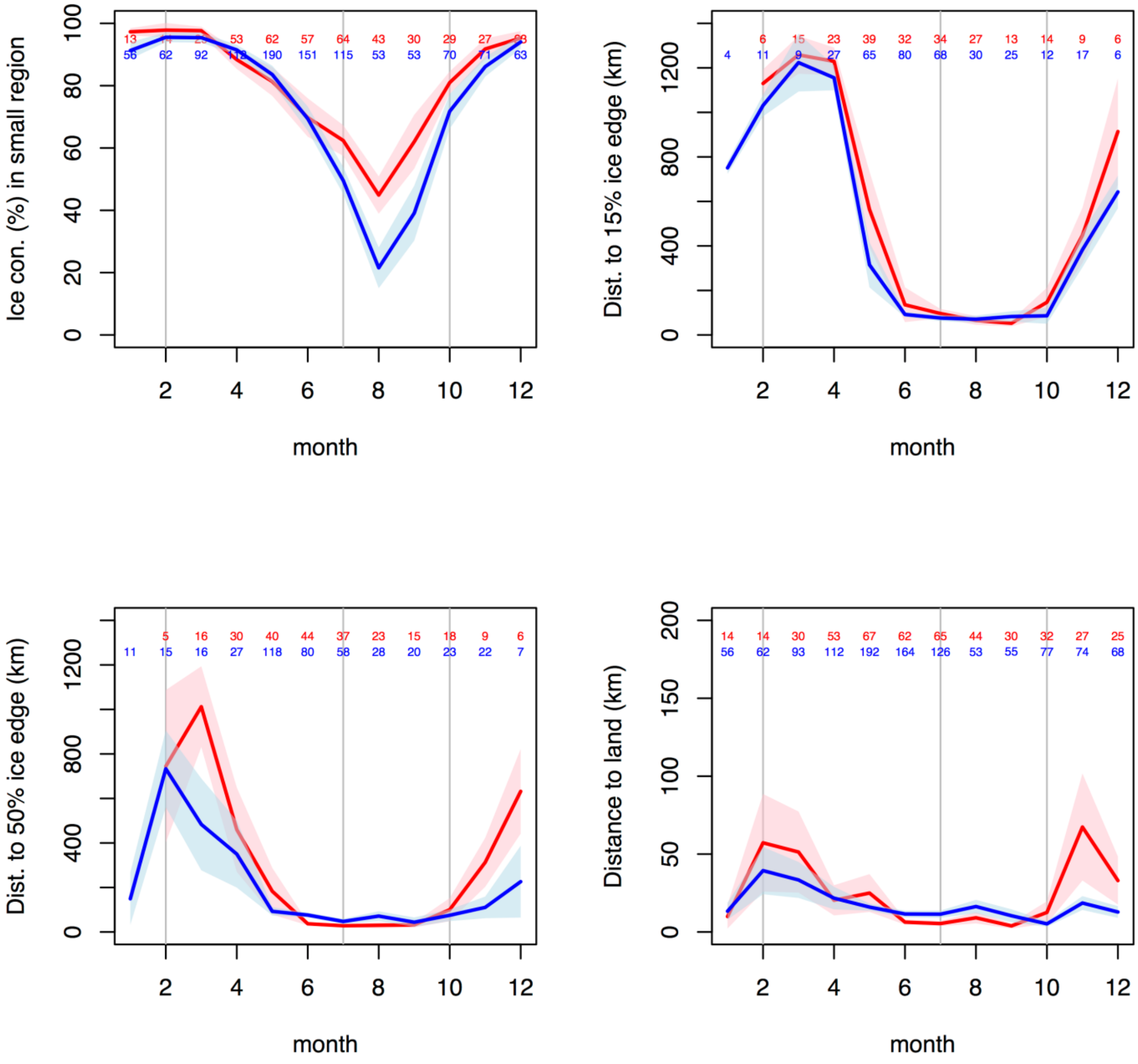


Figure 9.8. 1990s and 2000s adult female polar bear habitat use in KB for each of three sea-ice habitat variables: distance to 300 m depth contour, depth (bathymetry), and percentage of observations on the sea ice. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer). SSM/I sea-ice concentration is used in both decades. Months 8 -10 also represent land use by bears.

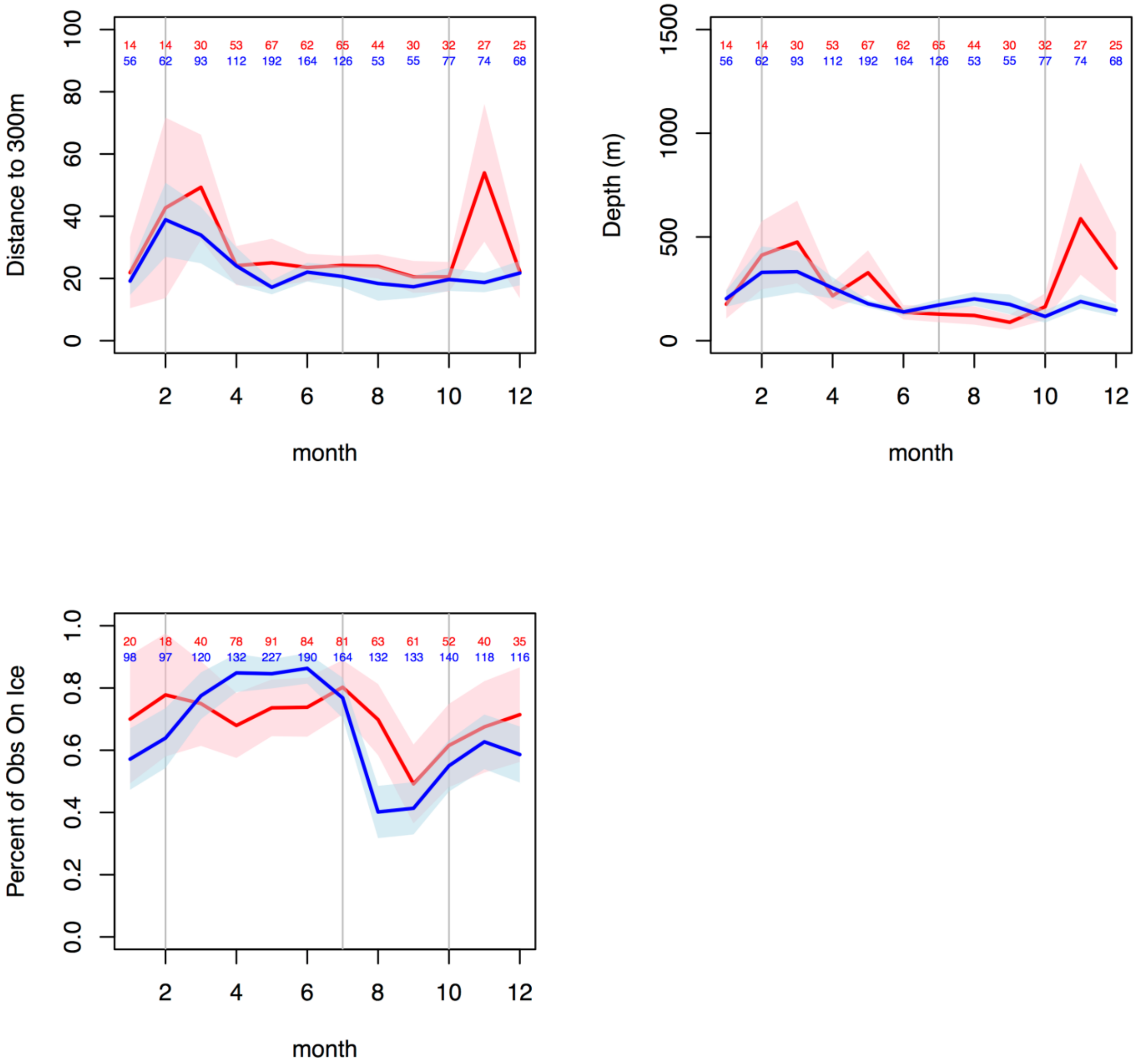


Figure 9.9. 1990s and 2000s adult female polar bear habitat use in KB for each of three terrestrial habitat variables: elevation, slope, and aspect. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Months 8 -10 also represent land use by bears.

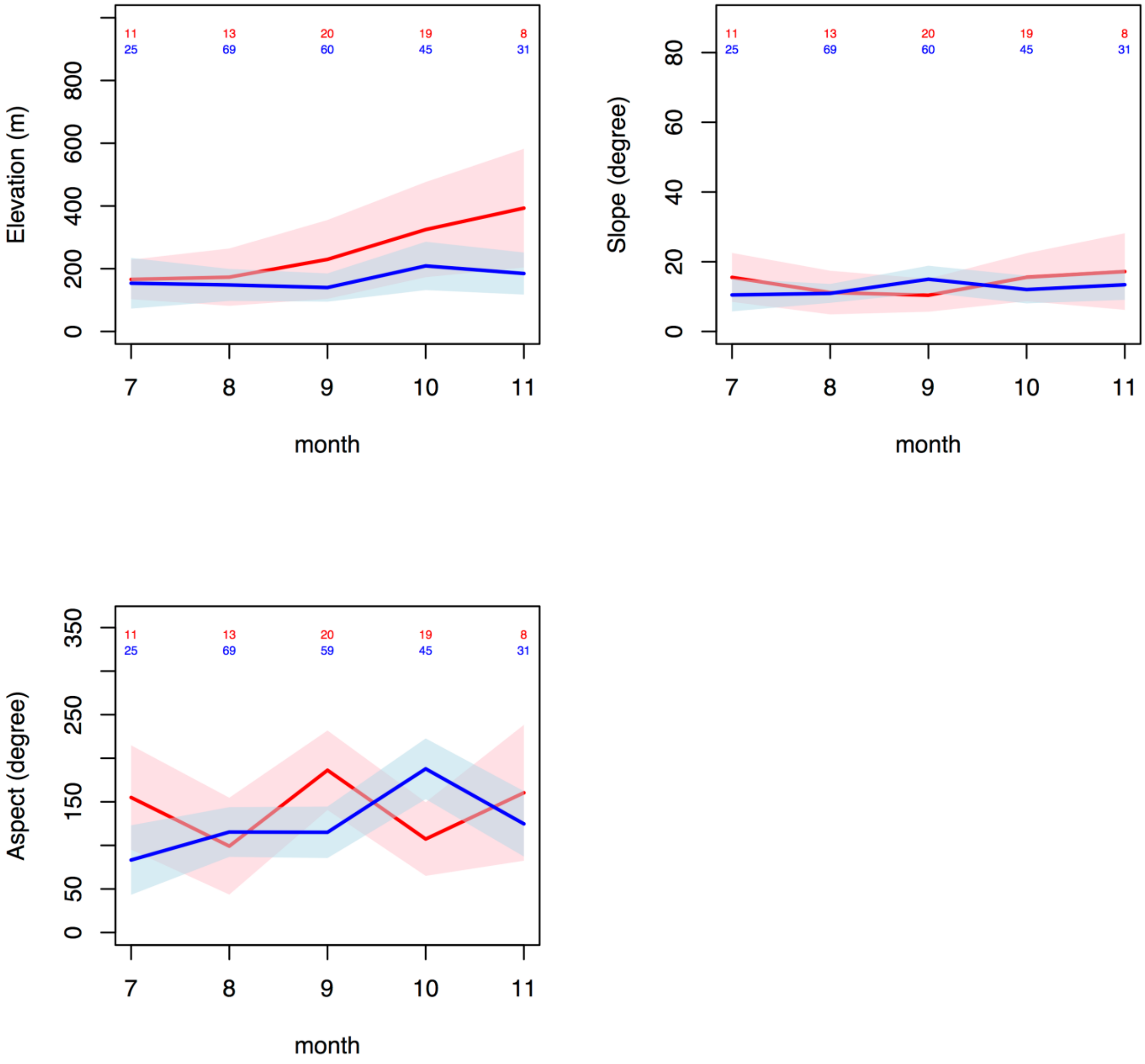


Figure 9.10. ASTER Digital Elevation Model (DEM) used in KB with trackline of a single bear tagged in the 2000s.

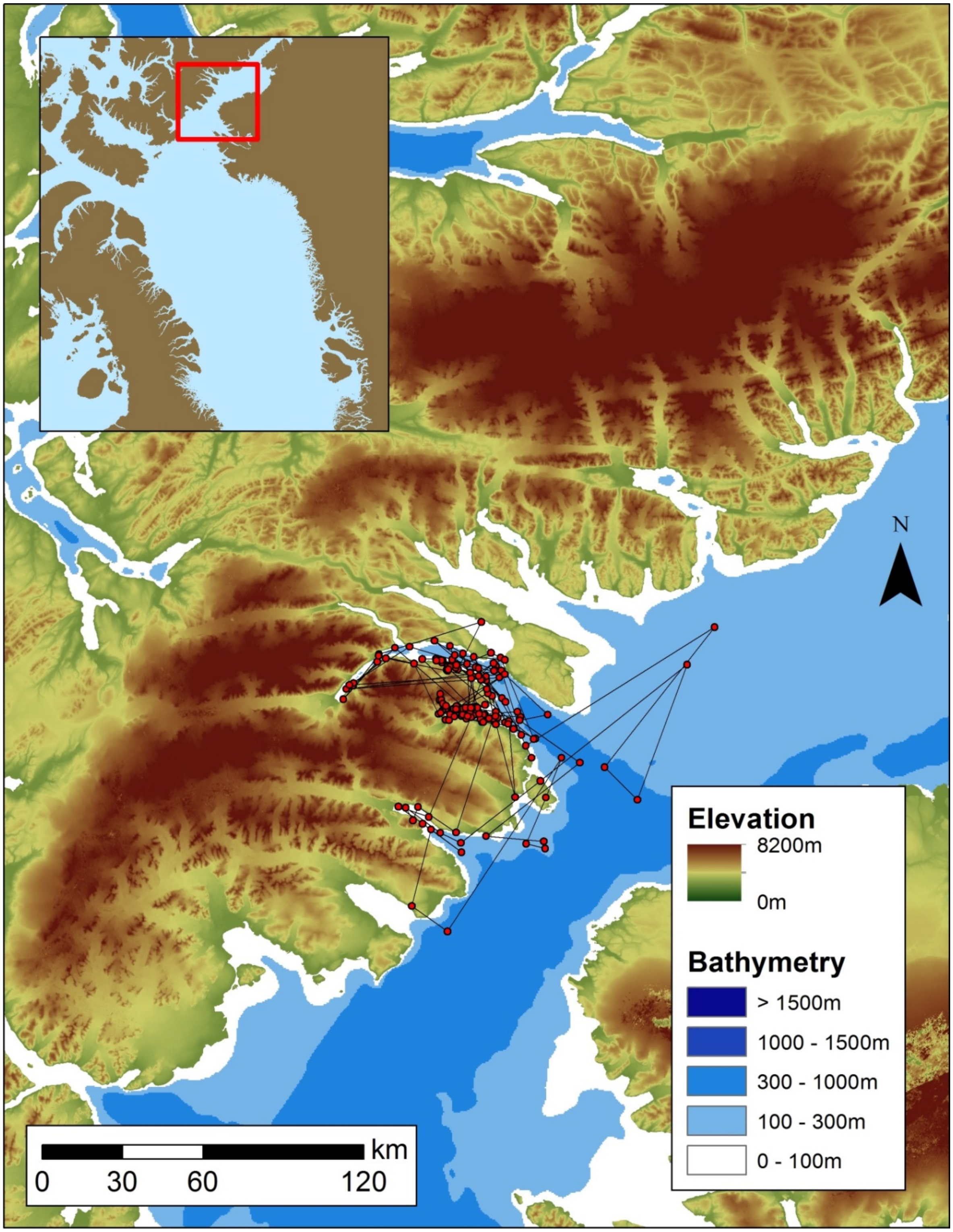


Figure 9.11. Distribution of KB polar bear maternity dens in the 1990s and 2000s.

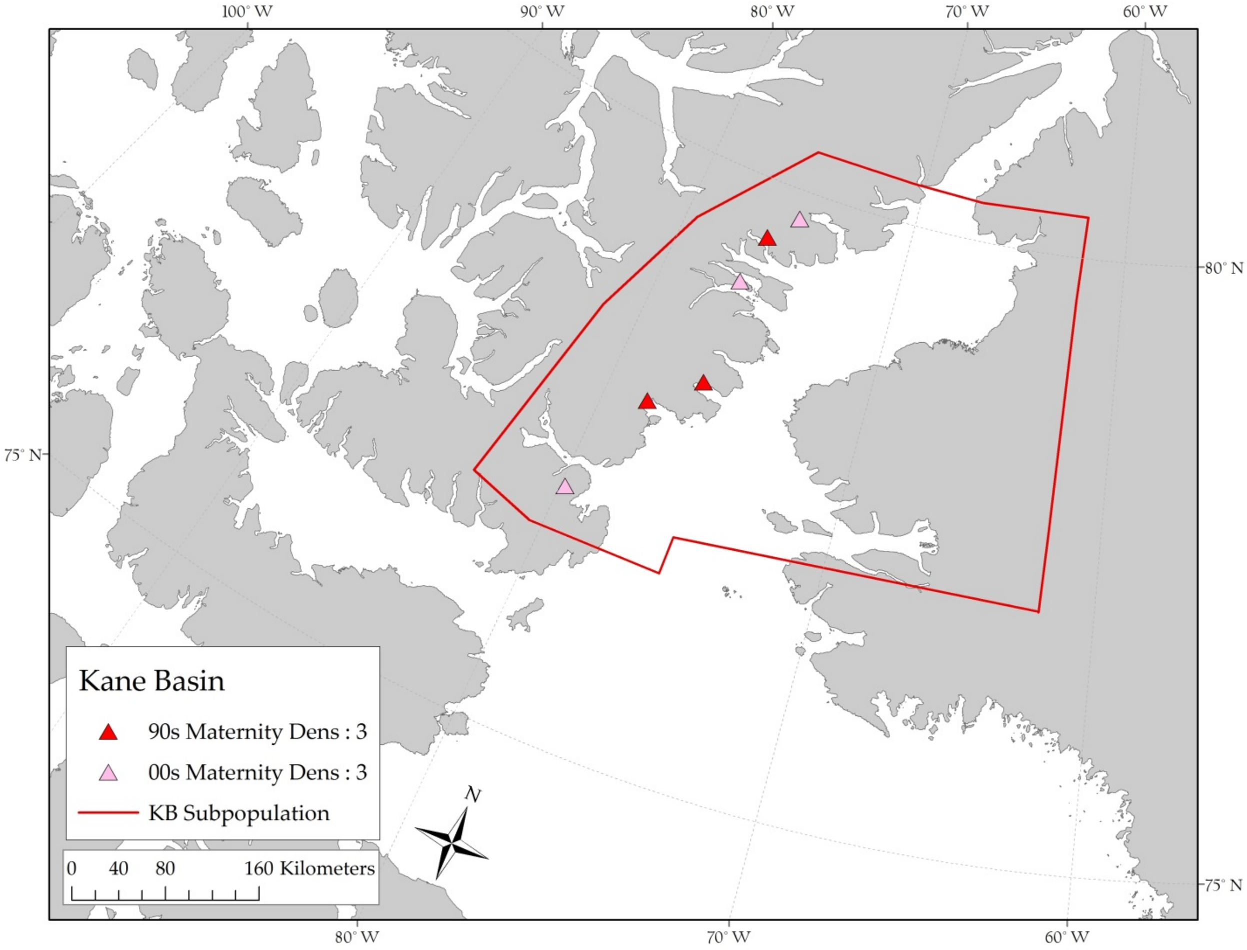


Figure 9.12. Distribution of KB polar bear shelter dens in the 1990s and 2000s.

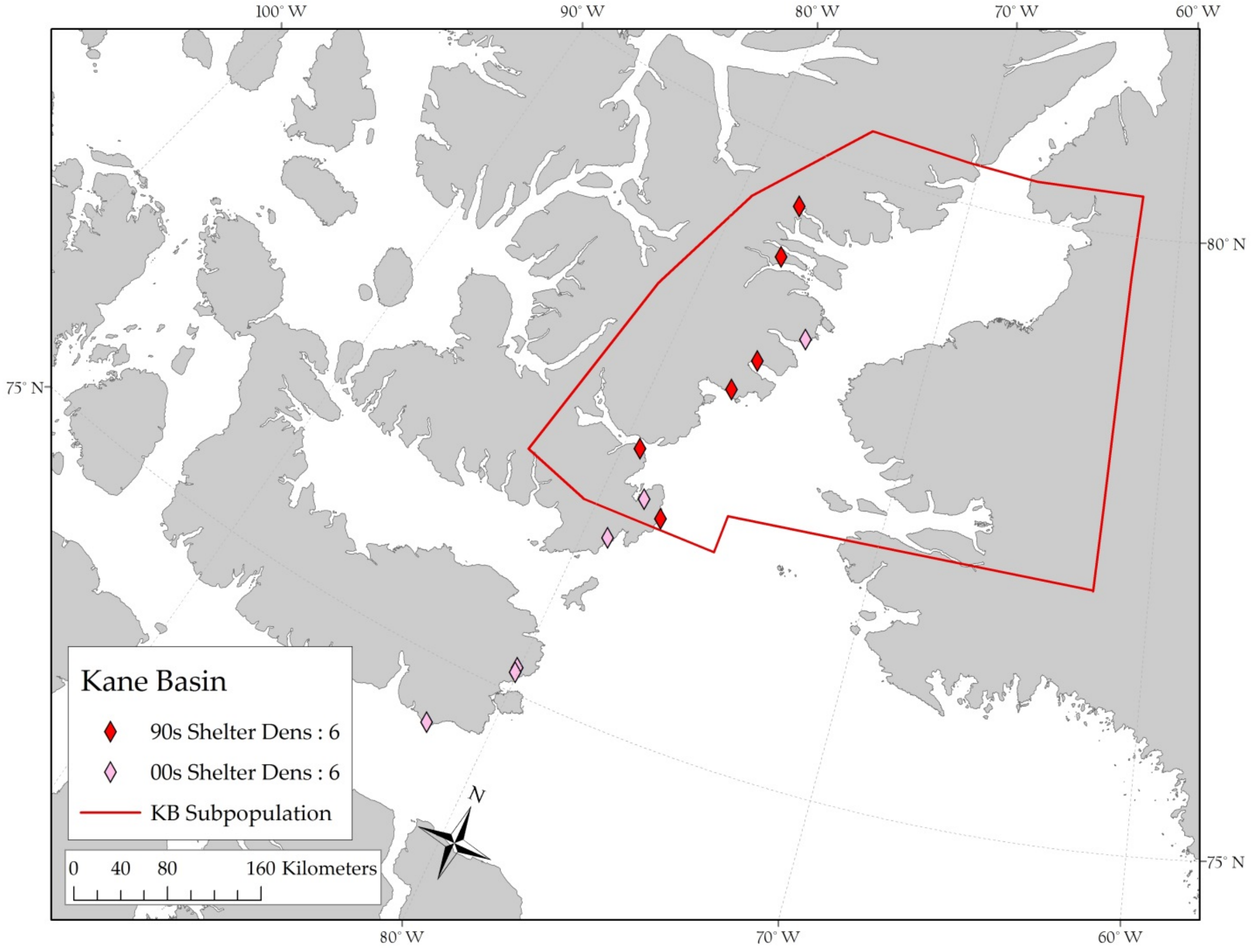


Figure 9.13. Boxplots comparing den duration of Kane Basin (KB) polar bear maternity dens (*P* = 1) (1990s: *n* = 3; 2000s: *n* = 3).

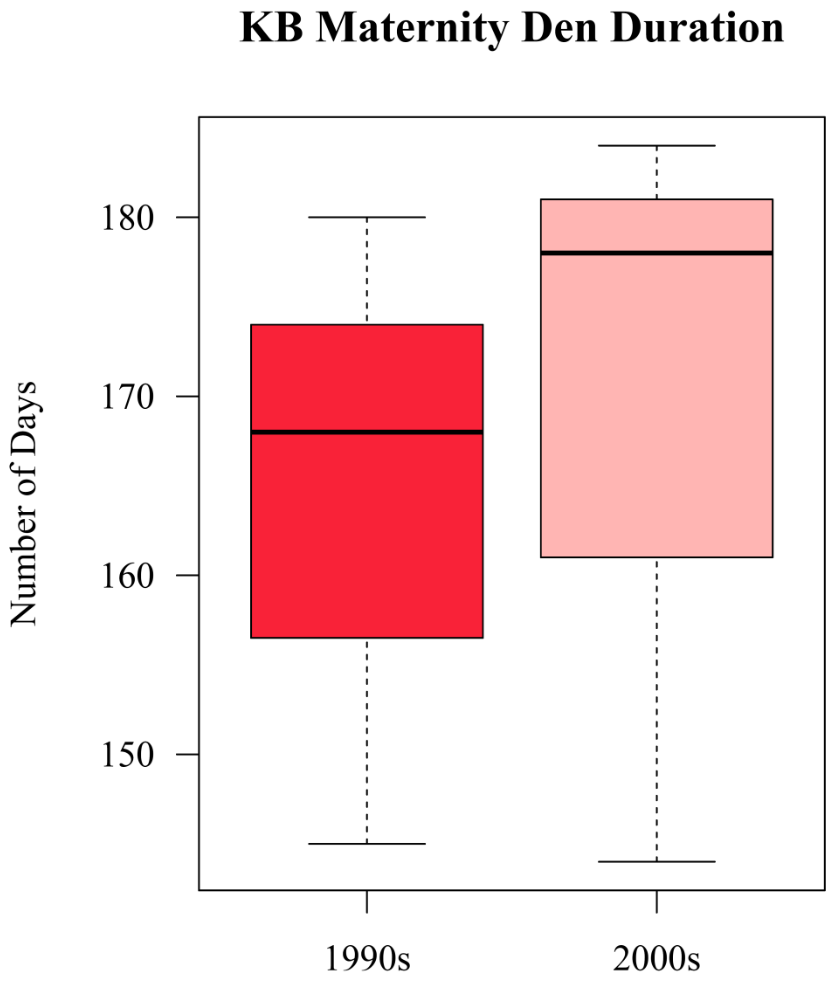


Figure 9.14. Boxplots comparing entry (*P* = 0.6) and exit dates (*P* = 1) of Kane Basin (KB) polar bear maternity dens (1990s: *n* = 3; 2000s: *n* = 3).

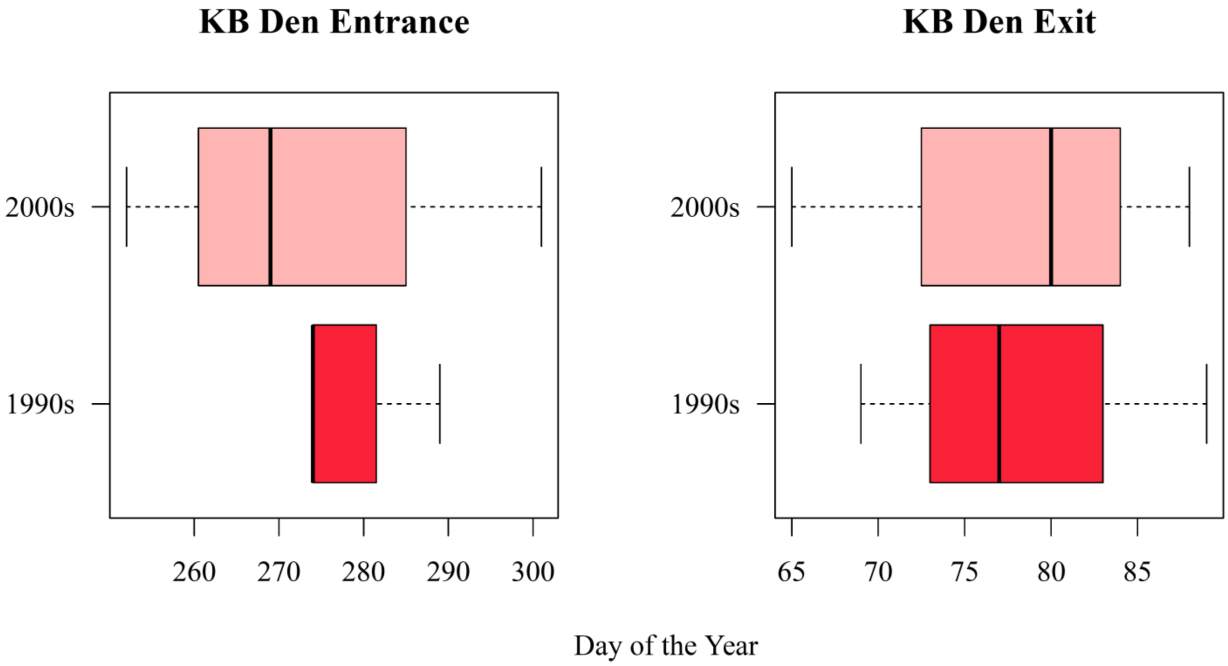


Figure 9.15. Boxplots comparing the first date on land (FDOL) of pregnant female polar bears from the 1990s (*n* = 3) and 2000s (*n* = 3) in Kane Basin (KB) (*P* = 1).

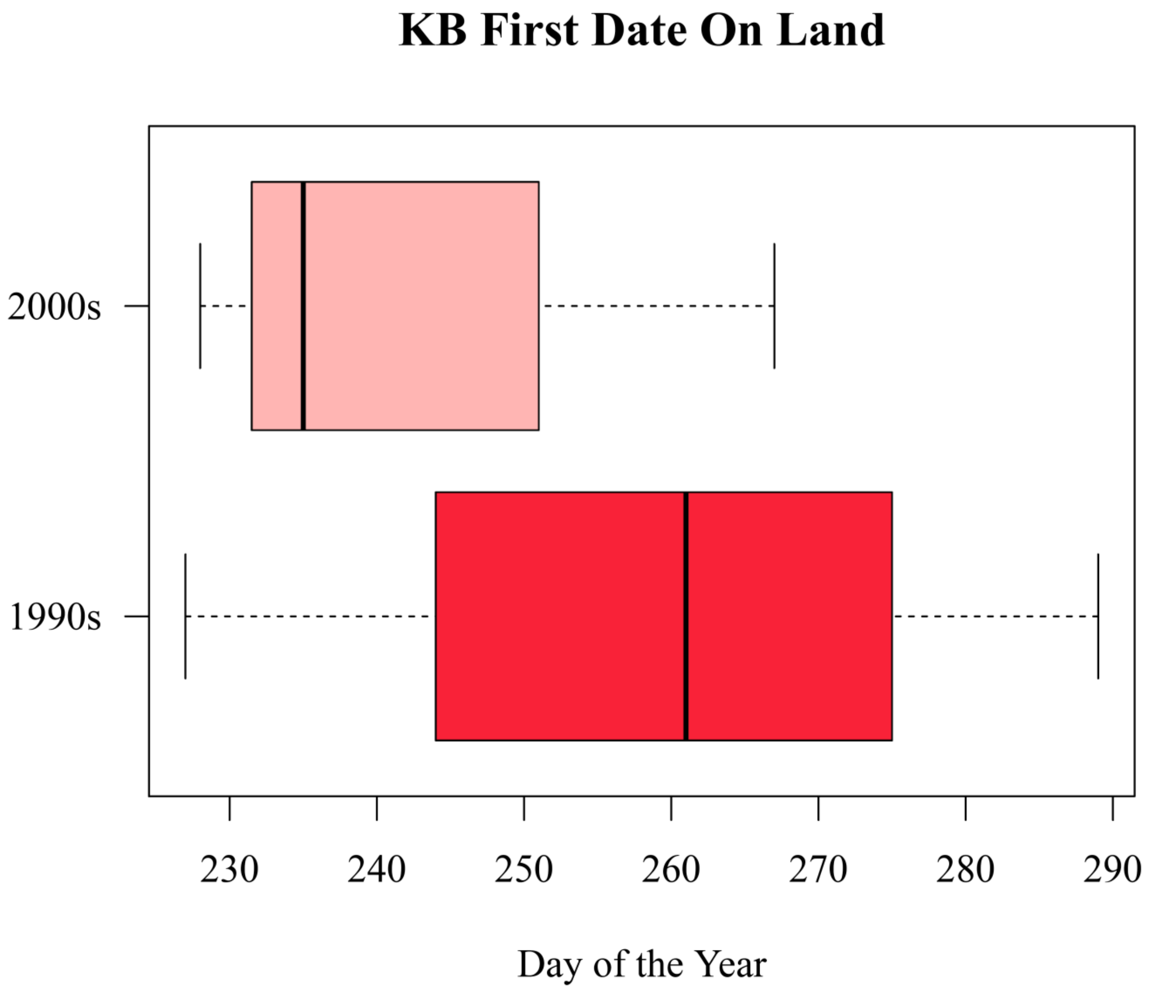
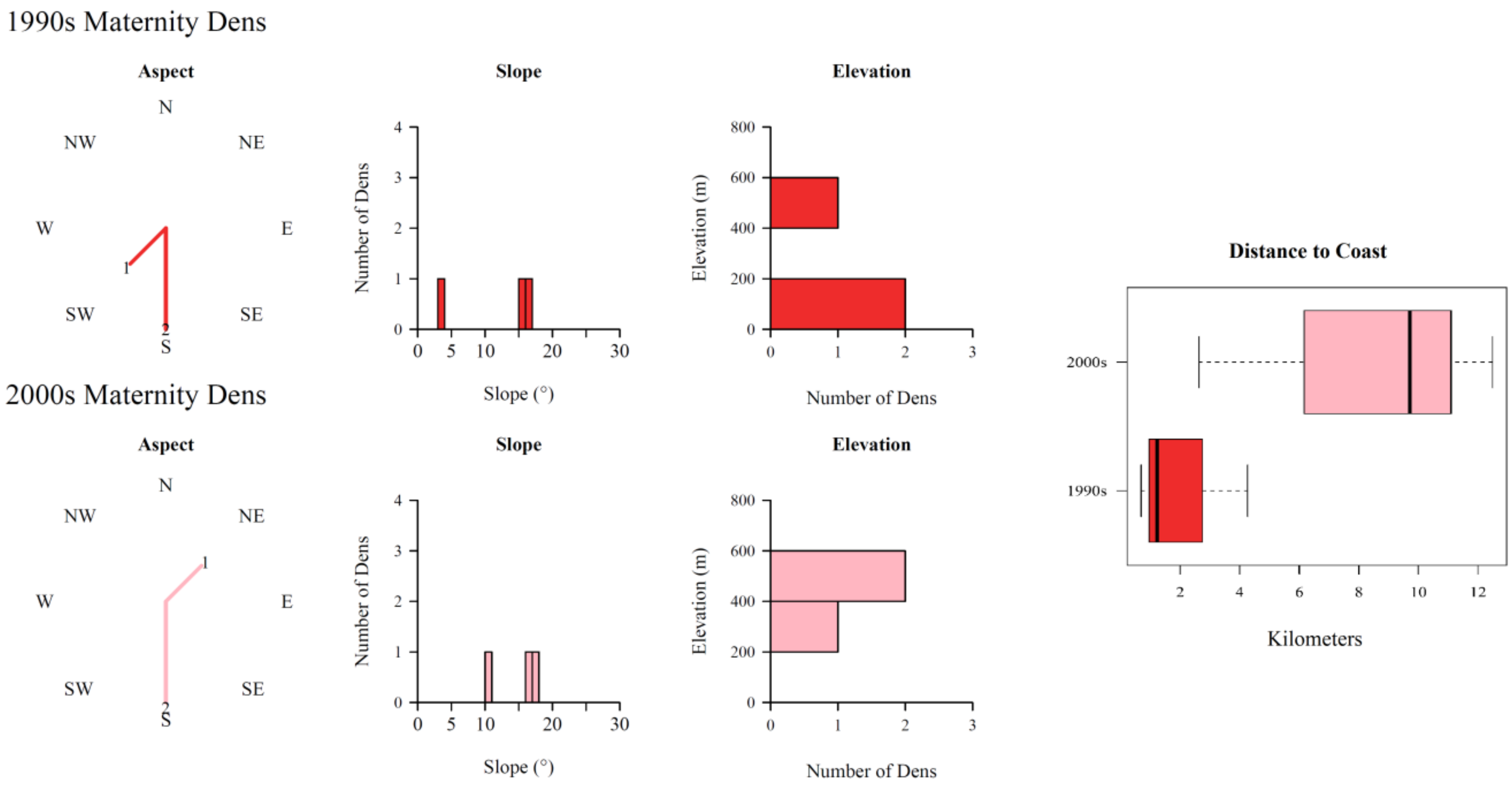


Figure 9.16. Plots comparing aspect, slope, elevation, and distance to coast for the 1990s (*n* = 3) and 2000s (*n* = 3) polar bear maternity dens in Kane Basin. The aspect plot consists of a compass face with lines marking the directions that dens faced. The lines are annotated with numbers noting how many dens were found at that aspect. None of the habitat variables significantly differed between the two time periods (elevation, aspect, distance to coast: *P* = 0.2; slope: *P* = 0.4).



CHAPTER 10

GENETIC MARK-RECAPTURE STUDY OF POLAR BEARS IN KANE BASIN

KEY FINDINGS

- We used joint live-recapture and dead-recovery mark-recapture models to analyze data for the Kane Basin (KB) polar bear subpopulation, with the goal of updating estimates of subpopulation size and survival. The dataset consisted of 277 initial live captures (1992-1997 = 150, 2012-2014 = 127), 89 live recaptures (1992-1997 = 53, 2012-2014 = 36), and 24 harvest returns of research-marked bears 1992-2014.

- Mark-recapture research conducted in the Kane Basin subpopulation yielded an estimate of abundance of 357 polar bears (95% CI: 221 – 493) for 2013 – 2014. An estimate derived during 1995 – 1997 yielded 224 bears (95% CI: 145 – 303). Based on physical MR, the size of the KB subpopulation was previously estimated to be 164 polar bears (95% CI: 94-234) for 1994-1997 (noting that this estimate applies to different years than our re-analysis; Taylor et al. 2008).

- We documented more bears in the eastern regions of the Kane Basin subpopulation during 2012 – 2014 than during the 1990s. Eastern Kane Basin was searched during the 1990s although with less effort than in the 2010s due to the low density of bears observed there. The difference in distribution between the 1990s and 2010s may reflect differences in spatial distribution of bears, possibly influenced by reduced hunting pressure by Greenland in eastern KB and thus an increased density of bears in KB, but also some differences in sampling protocols.

- The 2013 – 2014 estimate of abundance suggests 357 (221 – 493) bears currently use KB in springtime (i.e., the Kane Basin super-population; Kendall et al. 1997), and the current point estimate is higher than the historical estimate. Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of KB abundance for 2013-2014 and the estimate for 1995-1997 was approximately 133 bears (standard deviation of the difference ≈ 80 bears), with 95% of the sampling distribution suggesting that population change between the two time periods could have been positive. This suggests relatively strong evidence for a stable to increasing subpopulation, and is consistent with data on movements, condition and reproduction. We encourage some caution in interpretation of population growth due to potential expansion of the sampling frame and differences in sampling protocols between the 1990s versus the 2010s study periods.

- Current estimates of total survival for age 3+ females (0.95; SE: 0.04) and dependent bears were consistent with previous research. Estimates of unharvested survival for 3+

females appear sufficiently high for positive population growth. Updated estimates of total survival are lower for age 3+ males (0.87; SE: 0.06). Our longer-term data set and several other ecological, sampling, and technical considerations may contribute to this result.

- We documented a reduction in mortality associated with harvest, likely attributable to implementation of Greenland's harvest quota in 2006.
- Demographic modeling suggests Kane Basin bears exhibit relatively high fidelity to the springtime study area, with <5% of marked bears emigrating on an annual basis.

10.1. Introduction

Large-scale environmental changes are occurring across the circumpolar Arctic (Comiso et al. 2008, Stroeve et al. 2012, Laidre et al. 2015; see also Chapters 5 and 9), with general reductions in the temporal availability and spatial extent of sea ice. For sea ice obligate polar bears, which are among the most highly sensitive of marine mammals to the projected impacts of climate change (Laidre et al. 2008), long-term impacts are anticipated to be negative (Atwood et al. 2015). However, there likely will be significant temporal and spatial variability among subpopulations in the short-term (Stirling and Derocher 2012). Indeed, the effects of the changing Arctic environment on polar bears have been documented in some regions but are less clear or have not been realized elsewhere (e.g., Rode et al. 2012, 2014, Bromaghin et al. 2015, Obbard et al. 2015, Lunn et al. 2016).

The Kane Basin (KB) polar bear subpopulation, regarded as part of the Arctic archipelago region (Amstrup et al. 2008), covers a small region between Nunavut, Canada and NW Greenland. Abundance of KB was last estimated at ~164 (SE: 35) polar bears based on a physical mark-recapture study completed during 1992 – 1997 (Taylor et al. 2008). At the time of this estimate, the harvested population growth rate ($\lambda = 0.919$) indicated that the subpopulation was over-exploited. The unharvested growth rate also was low ($\lambda = 1.009$; Taylor et al. 2008), suggesting limited capacity for the KB subpopulation to increase even in the absence of human-caused removals. In response, Greenland implemented a quota in 2006 that significantly reduced the total harvest from Kane Basin (see Methods below). The small subpopulation size, low growth rates, and long-term exploitation led Taylor et al. (2008) to suggest that Kane Basin may act as a sink for neighboring subpopulations such as Baffin Bay.

The KB subpopulation is currently considered to be declining (PBSG 2015): 100% of population viability analysis (PVA) simulations (using data on abundance and vital rates from Taylor et al. 2008 and reported Canadian and Greenlandic catches) resulted in a decline in abundance within 10 years. However, no new research to update estimates of abundance or vital rates has occurred since the 1990s study. Given the outdated demographic information, the substantial changes in Arctic sea-ice habitats over the past several decades (e.g., Stroeve et al. 2012, Chapters 4 and 9), and the reduction in harvest in 2006, there was uncertainty as to the current status of polar bears in Kane Basin. As such, there was a need for new information to inform status and harvest management (Chapter 1).

Although bears in KB are not genetically different from those in Baffin Bay (Paetkau et al. 1999, Chapter 2), satellite telemetry and capture records indicate that they move among KB and neighboring subpopulations but exhibit strong fidelity to specific regions (Taylor et al. 2001, Chapters 2, 5, and 9). These data have formed the basis for population delineation, and polar bears in Kane Basin are considered a distinct demographic unit for management purposes.

Our objective was to estimate the current abundance and vital rates, including survival, of polar bears in the KB subpopulation. We sought to compare new estimates of abundance with those derived from earlier research (Taylor et al. 2008). These results, in conjunction with information on sea-ice dynamics, spatial ecology, reproductive output, survival, and other metrics, will be used to inform subpopulation status. The data used in this project spanned a 23-

year period (1992-2014): an initial 6-year physical capture and dead recovery sampling period (1992 – 1997) was followed by a 14-year period with dead recoveries only (1998 – 2011) and a recent (2012 – 2014) live capture (physical and genetic) and dead recovery session. Jurisdictions across the Arctic have increasingly invested in non-physical capture based monitoring methods, largely to address social considerations, particularly in Nunavut, regarding wildlife handling (*cf*. Chapter 1) and to facilitate more rapid monitoring. Prior to this study and research in the Baffin Bay subpopulation (Chapter 5), however, the focus of such alternative methods has been aerial surveys (e.g., Aars et al. 2009, Obbard et al. 2015, Stapleton et al. 2016).

10.2. Materials and Methods

Study Area

The KB subpopulation covers ~150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (boundaries evaluated in Taylor et al. 2001; Figure 10.1). However, the boundaries of the KB subpopulation encompass a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* Figure 10.3 and 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area) and the southern part of Kennedy Channel. It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the Baffin Bay (BB) and Lancaster Sound (LS) subpopulations, and to the west by Norwegian Bay (NW; PBSG 2010). The KB subpopulation is regarded as a part of the Arctic archipelago ecoregion (Amstrup et al. 2008); historically sea ice remained present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, this pattern has changed markedly in recent decades (Chapter 9, Figure 9.2). KB is partially connected to neighboring subpopulations; particularly notable though limited interchange occurs with BB and LS (Chapter 4). The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a partial barrier between KB and BB – LS. The KB subpopulation is subjected to subsistence harvest by Inuit living in Jones Sound (Canada) and the Qaanaaq area (NW Greenland; PBSG 2010, Born et al. 2011).

Field Sampling

Initial surveying was conducted in Kane Basin during springs (April – May), 1992 – 1997 (described in Taylor et al. 2008). Additional sampling was completed during fall 1994, but we excluded these data from the present analyses to reduce temporal heterogeneity (e.g., sampling cubs-of-the-year in spring versus fall yields substantial differences in estimates of survival). All sighted bears, including dependent offspring, were chemically immobilized (Stirling et al. 1989) and uniquely marked with plastic ear tags and permanent lip tattoos (Taylor et al. 2008). Ages of independent bears were determined by extracting vestigial premolars and counting annular rings (Calvert and Ramsay 1998), whereas cubs-of-the-year and yearling bears were considered of known age. A sample of adult females was outfitted with satellite collars as part of a separate study quantifying movements and spatial ecology (Taylor et al. 2001, Chapter 9).

Although Taylor et al. (2008) reported that they conducted a uniform search of the study site each year, records delineating their survey effort were unavailable. Subsequent examination of annual distributions of captures suggested incremental increases in the size of the study area, progressing northwards, with apparent expansions in the sampling frame between 1992 and 1993 (northward along eastern Ellesmere Island into the Nares Strait region). Between 1994 and 1995, survey efforts were expanded eastward into Kane Basin proper off the Humboldt Glacier in Northwest Greenland (E. Born, pers. obs.) which inferred from Taylor et al. (2001) was also the case in 1996 and 1997; Figure 10.2). No live-recapture sampling occurred during 1998 – 2011, but we obtained recoveries of harvested bears during this interval. Available information also suggests eastern Kane Basin was covered in the 1990s, though no captures were made there.

We surveyed KB during 25 April – 6 May, 2012; 27 April – 10 May, 2013; and 28 April – 19 May, 2014. Sampling windows were comparable to the 1990s, although surveying in 1992 and 1993 occurred earlier (mid-April) and for shorter windows of time. We sampled sea-ice habitats by helicopter (Bell 206 LongRanger) across the entirety of western and northern Kane Basin, including landfast ice in fjords and nearshore areas as well as offshore pack ice, but excluded more open water habitats of the North Water polynya. We also did not survey the sea ice in the fjords of the populated Qaanaaq area in NW Greenland (i.e., the eastern parts of the North Water polynya) because hunting pressure for marine mammals in these areas is generally high and consequently “resident” polar bears do not exist in the Qaanaaq area (E. Born, pers. obs.). Sampling was primarily completed via directed searching in 2012, with searches focused in areas believed to provide the most suitable polar bear habitat (“adaptive sampling”). In 2013, we completed directed searching and also flew ad hoc transects oriented approximately perpendicular to the coastline, particularly near Greenland, to ensure that effort was well-distributed across the landscape.

In 2012 and 2013, most bears were sampled via physical capture, including chemical immobilization and application of ear tags and lip tattoos as described above. We collected tissue samples from physically captured bears (for genotyping) and recorded additional information including sex, family status, field-estimated age class (cub-of-the-year, yearling, 2-year old, subadult, or adult) and standard morphometric measurements. We completed additional sampling in 2012 and 2013 via remote biopsy darting (Pagano et al. 2014) to collect genetic tissues for subsequent genotyping and analyses (e.g., Herreman and Peacock 2013). Cubs-of-the-year were too small in springtime to be biopsy darted and thus were not sampled when their mothers were biopsy darted (although COY were sampled during physical captures). For bears that were not physically immobilized, sex was confirmed upon genetic analyses (see below).

In connection with immobilization and handling, we deployed satellite transmitters on 36 polar bears in 2012 and 2013 (see Chapter 9; 2012: 6 satellite radio collars on adult females; 10 satellite ear-tags on adults and subadults of both sexes; 2013: 10 satellite radio collars on adult females; 10 satellite ear-tags on adults and subadults of both sexes). This work enabled us to evaluate distribution and habitat use during the genetic mark-recapture sampling and the aerial survey (Chapter 11) and to conduct a post hoc assessment of sampling representativeness during sampling.

We modified our sampling strategy during 2014. We stratified the study area into high- and low-density areas based on our observations of polar bears in 2012 and 2013 (i.e., presumed densities) and searched for bears from systematically spaced transects. This design enabled us to more efficiently allocate effort and reduced the potential for spatial heterogeneity in detection. Systematic sampling also facilitated the simultaneous completion of an aerial survey (Chapter

11) to derive an abundance estimate, based on different methodology, for comparison with the capture-based estimate.

Strata conformed to general landscape features and ice types: the high-density stratum included landfast ice within fjords as well as nearshore pack ice (within ~30 km of the nearest land mass); the low-density stratum included farther offshore pack ice (Figure 10.3). We delineated the landward extent of the study area using current GIS layers from Greenland and Nunavut. We used Moderate Resolution Imaging Spectroradiometer (MODIS; http://modis.gsfc.nasa.gov/) images with 1 km resolution to delineate the extent of available habitat by approximating the edge of the North Water polynya. Because the polynya's boundaries can change rapidly, we delineated the extent of the polynya adjacent to the section surveyed on a particular day using MODIS imagery from that day (when possible) or as to close to that date as possible (when imagery was unclear on that date due to atmospheric conditions). We examined the delineated study area in relation to weekly regional sea-ice charts produced by the Canadian Ice Service (https://www.ec.gc.ca/glaces-ice/) for confirmation. During sampling, we also collected GPS waypoints at the edge of the polynya to verify delineation.

Transects were systematically spaced at 6-km and 18-km intervals in the high- and low-density strata, respectively, based on anticipated encounter rates and available resources. We also sampled during ferry flights (e.g., between survey transects). Survey protocols in 2014 (detailed in Chapter 11) were designed to facilitate the simultaneous collection of data for mark-recapture and the aerial survey. All mark-recapture sampling in 2014 was conducted via remote biopsy darting.

Harvest Recoveries

We used harvest records to compile dead recovery data for polar bears captured in KB and subsequently harvested there or in neighboring subpopulations during 1992 – 2013 (Burnham 1993). Harvest was monitored by the return of tags or lip tattoos during 1992 – 2010 and by genotyping during 2011 – 2014. Data including date and location of recovery and sex and estimated age were recorded for harvested bears and individuals killed in defense of life and property. Reported harvest rate in KB was relatively high during the 1990s (range: 6 – 17 bears / year) but significantly decreased by the mid-2000s (2 – 8 bears / year; Chapter 8), likely due to factors including changes in sea-ice conditions limiting hunter access by use of dog sleds to northeastern KB (E. Born, pers. obs.) and the implementation of a Greenlandic quota system in 2006. Greenland's reporting system also improved with the implementation of the quota (Chapter 8). Previous studies assumed that harvests of all marked bears were reported when natural survival was calculated and, therefore, the reporting rate r was interpreted as the proportion of mortality due to harvest (e.g., Taylor et al. 2005, 2008). However, more recently, genetic data suggested under-reporting of marked bears in the harvest, with decreases in reporting correlated with increasing marker age (Chapter 8).

Genetic Analyses

DNA Extraction – Dried biopsy samples, new and archived tissue samples, and harvest specimens (frozen or in ethanol) were sent to Wildlife Genetics International (Nelson, B.C., Canada) for analysis using protocols previously validated for bears (Kendall et al. 2009). DNA was extracted from ~ $3mm^2$ pieces of tissue with QIAGEN DNeasy Blood and Tissue Kits (http://www.qiagen.com/). Most biopsy darting samples consisted of a plug of a skin and sub-cutaneous tissue. This provided ample material for DNA extraction and residual tissue for future analyses. In a small proportion of cases, the available sample consisted of a tuft of hair. DNA was extracted from these hair samples using approximately 10 guard hair roots or 30 pieces of underfur. In a few cases, where a biopsy sample contained no visible tissue, DNA was successfully extracted by soaking the barbed needle from the biopsy dart in the lysis mix (QIAGEN buffer ATL + proteinase K).

Marker Selection – To select markers for the analysis of individual identity, we used allele frequency data from 1,771 polar bears for which complete 20-locus genotypes existed before the genetic mark-recapture began (Government of Nunavut unpublished data). We ranked the 20 microsatellite markers in the dataset by expected heterozygosity. The 8 most variable markers that could be analyzed together in a single sequencer lane were selected for use. These surpassed the required standard for marker variability (HE = 0.80; Paetkau 2003). In addition to the 8 microsatellite markers, we analyzed sex on every sample, using a *ZFX/ZFY* marker. This 9th marker roughly halved the match probability (assuming a balanced sex ratio), even for close relatives, as well as providing replication of sex data for individuals that were sampled more than once.

Genotyping – The analysis of individual identity followed a 3-phase approach. Phase 1 was a first pass of all extracted samples using the 9 selected markers (*G10B, CXX20, G10H, G10P, 145P07, MU50, MU59, G10X and ZFX/ZFY*). Samples that failed at > 6 of 9 markers on the first pass were set aside and did not proceed further in the analyses. Previous experience has shown that such samples are prone to errors and run out of DNA before generating a complete (phase 2) and reproducible (phase 3) genotype (D. Paetkau, pers. comm.).

The first pass was followed by a cleanup phase in which data points that were weak or difficult to read the first time were re-analyzed. During cleanup we used 5 µL of DNA per reaction instead of the 3 µL was used during first pass. At the conclusion of the cleanup phase, the remaining samples (99.5%) had high-confidence scores for all 9 markers. In cases where the genetic sex result contradicted the reported sex based on field assessment, genetic sex was checked using a second independent marker (*amelogenin*; http://www.ncbi.nlm.nih.gov/pubmed/7695123), thus confirming the results, and ruling out the possibility that a mutation at a particular marker was to blame. In all cases, results from the second marker confirmed that the field data was the source of error.

The third and final phase of analysis was error-checking, following the published protocol of reanalyzing the mismatching markers in highly similar pairs of genotypes (Paetkau 2003). This error-check included genotypes from the 4,657 polar bears in the database, plus published data from 473 individuals (Paetkau et al. 1999). The error-checking protocol functions on the principle that when ≥ 2 samples are genotyped from a given individual, and when 1 of those genotypes contains an error, the result is a pair of genotypes which match at all-but-1 marker (a '1MMpair'). Less commonly, 2MM-pairs are created when 2 errors have been made in the genotypes of the samples from a given individual.

An important distinction with this protocol is that it is designed to ensure accurate individual ID — and has been proven to do so with a high degree of efficiency (Kendall et al. 2009) — but it is not intended or expected to correct errors when just one sample has been genotyped from a given individual. In addition to re-analyzing mismatching markers, this protocol also involved the inclusion of additional markers for some samples. Finally, we also searched the dataset for genotype matches that seemed unlikely based on our field data. In each case, three extra markers were added to the genotypes to lower the probability of chance matches between individuals. The extra loci confirmed all of these matches. Once the genotyping and error-checking was complete, we defined an individual for each unique 9-locus genotype.

Marker Power – In addition to the genotyping errors that were targeted during error-checking, DNA-based datasets are prone to a second source of error, when match probabilities are so high that some individuals have identical genotypes. Calculated match probabilities provide no practical insight into the risk of sampling individuals with matching genotypes, because the calculations are so dependent on the assumptions made about the degree of relatedness among the sampled individuals. We therefore used the direct, empirical approach of extrapolation from the observed mismatch curve (Figure 10.4). We expect to see roughly order-of-magnitude decreases in the number of pairs of individuals whose genotypes match at increasing numbers of markers (Paetkau 2003). In our dataset the slope of this curve was reasonably true to that rule of thumb. From this curve, it is estimated that we would have sampled ~ 0.3 0MM-pairs (individuals whose genotypes matched at 9 markers) in this multiyear dataset of 4,657 individuals; a very small risk of error in proportion to the size of the dataset. In addition to reducing the risk of sampling individuals with the same genotype, another benefit to having such a powerful marker system was realized during error-checking, where the amount of time required to reanalyze the mismatching markers underlying 1MM- and 2MM-pairs was trivial in proportion to the scale of the project, because there were so few such pairs.

Statistical Analyses

We analyzed joint live-capture and dead-recovery data from the KB subpopulation with the Burnham (1993) mark-recapture model, which combines the Cormack-Jolly-Seber (CJS) live-recapture model with the Brownie-Seber dead-recovery model to estimate survival (S), recapture (p), reporting (r), and fidelity (F) probabilities. With the Burnham model, live recaptures are assumed to occur (relatively) instantaneously within the study area, whereas dead recoveries can occur year-round between live capture periods and may take place within or outside the live encounter study area. We assumed that harvests prior to April 15 occurred before the live encounter period in year *t*, (i.e., in year *t* - 1); post-April 15 harvests were considered to have occurred after the live encounter period (i.e., year *t*). This treatment of the data resulted in no instances in which a bear was recovered before being captured alive. We acknowledge that there was some temporal overlap of live recapture and dead recovery periods in KB, but for a long-lived species such as polar bears, the exact timing of harvest relative to the live capture sampling period is less important.

We analyzed data and constructed models in program MARK (White and Burnham 1999). We assembled capture histories from the live capture and dead recovery data and included harvest recoveries through 2013. Although ages were estimated with high resolution during the initial 1990s study period, there was uncertainty in field assessment of age during the 2012 – 2014 sampling frame, particularly with biopsy darting. Hence, we identified relatively coarse age classes (*cf.* Taylor et al. 2008, Peacock et al. 2013), including cubs-of-the-year (coy), yearlings (yrl), 2-year olds (2yr), and individuals age 3 and above (age 3+). Because KB is a small subpopulation, capture and recovery data were very sparse, and we identified a limited number of relatively simple candidate sub-model structures.

We hypothesized that survival would differ among age classes and included age structure in all candidate models; however, we constrained yearling survival equal to 2-year old survival due to sparse data. Because coy are fully dependent on their mothers for their survival, we assumed that survival would not vary between male and female coy. However, we expected that survival would differ between sexes for older age classes, largely due to the 2 : 1 male-to-female sex ratio in the harvest, so we examined structures in which (1) *S* differed between sexes for age 3+ bears only and (2) *S* differed between sexes for yrl / 2yr and age 3+ bears (additive effect of sex). Given the sparseness of the data, we did not examine year-to-year variability in *S*, or relationships between *S* and time-varying environmental covariates.

We examined seven sub-model structures for *p* (i.e., estimation is conditional on first capture). Estimates of *p* in the Burnham model reflect both the probability of an animal being located in the sampling area and thus available for recapture, and the probability of the animal being recaptured conditional on its presence in the sampling area (i.e., random temporary emigration is incorporated in *p*; Burnham 1993). We hypothesized that female bears and dependent offspring (ages 0 and 1) may have a different *p* than independent male bears and evaluated models with this sex and age-class structure (family; sub-model structure 1). In addition, we suspected that search effort and sampling protocols may have differed between the two sampling epochs (1992-1997 vs. 2012-2014), so we considered structures with a temporal epoch effect (epoch; 2). Although the data were scant, we hypothesized that inter-annual variability in weather and sea-ice conditions may have resulted in *p* that varied significantly among years, so we also considered a fully time varying structure (time; 3). We considered structures with additive effects between (4) family and epoch and (5) family and time, as well as a structure including (6) an interactive effect between family and epoch. We also evaluated a null *p* sub-model (i.e., constant *p*; 7).

Because some adult females in our 1990s sample were outfitted with satellite collars (*n* = 12) that may have assisted in locating them, we created a binary radio covariate indicating whether a bear was theoretically available for recapture with the assistance of radio telemetry. We applied the covariate for 2 years post-collaring during the 1990s sampling period, unless there was evidence that the collar was physically removed from the bear. We included the radio covariate in all structures and coded dependent offspring such that they had the same covariate structure as their mother. Satellite collars were not used to locate bears during the 2012 – 2014 period. Some individuals were not successfully genotyped ($n = 25$) because either tissue samples were not located among the archives or the samples were inadequate to facilitate genotyping. These individuals had a reduced p during 2012 and 2013 (when physical capture and biopsy darting both occurred), but no probability of detection during 2014 (when bears only were sampled via biopsy darting). To reflect this, we created a binary ‘genotyped’ covariate (0 = successfully genotyped; 1 = not genotyped) and included it in all model structures for 2012 and 2013; for non-genotyped individuals, we fixed $p = 0$ in 2014. We also fixed $p = 0$ during 1998 – 2011, when there was no live recapture sampling.

The reporting (r) parameter represents the probability that a dead bear is identified and reported to authorities. Here, r reflected the proportion of mortality that can be attributed to reported harvest (including bears killed to protect life or property). We hypothesized that r would vary among age classes (yrl / 2yr and age 3+) and by sex for age 3+ individuals (sub-model structure 1) due to harvest regulations, including sex-selective harvest (2 males : 1 female). Because recovery data were sparse (≤ 5 total recoveries per year; typically 0 – 2 recoveries per year), we did not consider models with annual variation in r, but we created an alternative structure which included an additive effect for time period (pre-2006; 2006 - 2013) for age 3+ individuals to reflect the changes in harvest and improvements in the Greenlandic reporting system over the past decade (structure 2). There were no records of cubs-of-the-year marked in KB harvested during the first year post marking, so we fixed r_{coy} to 0. Because only harvest data through 2013 were included in analyses, we fixed r to 0 for all age classes in 2014.

We hypothesized that polar bears may permanently emigrate from KB, based on the semi-discreteness of subpopulation boundaries (Taylor et al. 2001) and the spatial distribution of historical recapture and recovery data. Thus, we chose to estimate the F parameter, rather than assume that there was no permanent emigration and fix F to 1, as done in previous studies (e.g., Taylor et al. 2005, 2008, 2009). We considered structures in which (1) F was estimated as constant across all sex and age classes (constant) and (2) F was different for a combined class of coy, yrl / 2yr, and age 3+ females vs. age 3+ males (3+ males).

We constructed the most generalized model (excluding individual covariates) and used the median $\hat{c}$ method, as implemented in Program MARK, to estimate over-dispersion. Because results suggested the data were not significantly over-dispersed (i.e., $\hat{c}$ was approximately 1), we proceeded with model selection via AIC_c. Given the relatively small set of candidate sub-model structures, we constructed all possible combinations of candidate sub-models.

We evaluated models via AIC_c and model-averaged parameters for models with $\Delta\text{AIC}_c < 4$ (Burnham and Anderson 2002), based on an initial sensitivity analysis. Our estimates of survival reflected harvest mortality, so we derived estimates of natural survival as $S + r * (1 - S)$ (following, e.g., Taylor et al. 2005, 2008, Peacock et al. 2013) and estimated variance via the delta method (following Taylor et al. 2008). This equation relies on several key assumptions. First, it assumes harvest of all marked bears is reported; under-reporting of the harvest, which has been documented (Government of Nunavut, unpublished data), would lead to negative bias in estimates of natural survival. However, this derivation of natural survival also assumes that harvest mortality is completely additive, i.e., no bears that are harvested would otherwise die during a given interval. In contrast to under-reporting of marked bears in the harvest, a violation of the assumption of additive mortality would result in positive bias in estimates of natural survival.

For highly supported models, we used a generalized Horvitz-Thompson estimator, $\hat{N} = \frac{n}{\hat{p}}$, where *n* is the number captured in group *i* and $\hat{p}$ is the recapture probability for group *i*, to generate estimates of abundance by attribute group (e.g., family group status) for the yrl / 2yr and age 3+ classes. Because some coy were not marked during the 2012 – 2014 sampling period and estimates of *n* and *p* did not accurately reflect this age class, we incorporated coy by estimating the number of age 3+ females with coy litters via a Horvitz-Thompson estimator and multiplying by mean observed coy litter size. To obtain an overall estimate of abundance for KB by year, we summed individual estimates across groups. Following previous work (e.g., Taylor et al. 2005, 2008, Peacock et al. 2013), we estimated variances for total abundance estimates and incorporated variances and covariances (calculated in MARK) as well as variance of mean litter sizes via the delta method (Seber 1982, Powell 2007) using R (R Core Team 2015) package emdbook (Bolker 2016). We model-averaged estimates of total abundance using model weights for recapture probabilities and variances obtained with the delta method. We calculated mean overall estimates of abundance by sampling epoch and estimated variance using the delta method. We excluded 1993 – 1994 and 2012 from these mean estimates given the initial expansions of the sampling frame between 1992 and 1995 and the long interval without live recaptures preceding 2012, respectively (i.e., estimation of subpopulation size in 2012 was based on estimated recapture rates of bears marked during the 1990s applied to newly encountered bears in 2012).

10.3. Results

We recorded a total of 277 initial captures, 89 recaptures, and 24 dead recoveries over the course of the 23-year study period (Table 10.1). Markedly more bears were captured in the eastern regions of KB (i.e., off Humboldt Glacier in Northwest Greenland) during 2012 – 2014 than during the 1990s (Figures 10.2 and 10.5). Capture data were particularly sparse during the 1990s, although sampling in 1995 yielded significantly more captures than other years in the 1990s (Table 10.1). Similarly, very few bears were recovered via the harvest during the 2000s (Table 10.1). Notably, no males initially marked in KB during the 1990s were recaptured during 2012 – 2014, and only one male marked in the 1990s was reported in the harvest after 2002. Although no COY were sampled in 2014 (all sampling was conducted via biopsy darting), we observed a total of 23 COY with their mothers that year. In addition, 3 COY with their mothers were not biopsy darted in both 2012 and 2013. Mean observed COY litter size during 2012 – 2014 was 1.60 (SD: 0.5).

The most highly supported models included an additive effect of sex for the yrl / 2yr and 3+ age classes for S and a temporal effect (break at 2006) for r (Table 10.2). Although there was not clear support for specific structures for modeling p, complex (e.g., fully time-varying) structures for p were not supported in model selection, which was not surprising given the sparseness of the data. For model-averaging, we included 12 of 56 total models (cumulative model weight = 0.76).

Estimates of total survival of males were markedly lower than females for both the yrl / 2yr and 3+ age classes, although we note that the additive effect in S was shared across age classes and not estimated separately for yrl/2yr vs. 3+ bears (Table 10.3). This pattern was also evident in estimates of unharvested survival (yrl / 2yr females: 0.74, SE: 0.15; yrl / 2yr males: 0.54, SE: 0.17; age 3+ females, 2006 – 2013: 0.96, SE: 0.04 and 3+ males, 2006 – 2013: 0.88, 0.05). As hypothesized, recent (2006 - 2013) estimates of r were less than 1992-2005 values, although r did not significantly differ among age and sex classes (Table 10.3). Estimates of F suggest relatively strong fidelity to the springtime study area for females and dependent bears (F: 0.98, SE: 0.04) as well as age 3+ males (F: 0.96, SE: 0.07).

Annual estimates of abundance largely reflected the variability in sample sizes among years (e.g., 1995; Table 10.4, *cf*. Table 10.1). The estimated mean total abundance of the KB subpopulation during the 1995 – 1997 period was 224 (SE: 40; 95% CI: 145 – 303). The estimated mean total abundance for 2013 – 2014 was 357 (92; 221 – 493).

10.4. Discussion

We used a combination of physical and genetic mark-recapture techniques, including live recaptures and dead recoveries, to estimate demographic parameters of the Kane Basin polar bear subpopulation over a 23-year study period. Our estimate of abundance from the 1990s (224, 95% CI 145 – 303; averaged over 1995 – 1997) was consistent with previous analyses (164, averaged over 1994 - 1997; Taylor et al. 2008). Although the 2010s point estimate is ~36% greater than the 1990s estimate of Taylor et al. (2008), from the 1990s; this difference is largely attributable to our decision to derive a mean estimate of abundance from only 1995 – 1997. The sampling frame expanded during the 1990s, progressing northward and eastward in incremental steps such that, in the initial years, only a portion of KB was surveyed (Figure 10.2). Hence, we calculated mean abundance estimates by epoch only during periods when sampling was consistent and the sampling frames were generally comparable (1990s: 1995 – 1997). By contrast, Taylor et al.'s (2008) estimate was calculated as the mean estimated from 1994 – 1997; including 1994 in our estimate would reduce our point estimate from 224 to 198.

The 2013 – 2014 estimate of abundance suggests 357 (221 – 493) bears currently use KB in springtime (i.e., the Kane Basin super-population; Kendall et al. 1997), and the current point estimate is higher than the historical estimate. Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of KB abundance for 2013-2014 and the estimate for 1995-1997 was approximately 133 bears (standard deviation of the difference ≈ 80 bears), with 95% of the sampling distribution suggesting that population change between the two time periods could have been positive. This result suggests a stable to increasing subpopulation and is consistent with data on movements (Chapter 9), reproductive output (Chapter 12), and body condition (Chapter 13), suggesting that Kane Basin is currently a healthy subpopulation. However, we encourage caution in interpretation. We attempted to mitigate the impacts of apparent changes in sampling frames, particularly during the 1990s, by excluding 1993 and 1994 from our mean estimate of abundance during the 1990s. However, we were unable to address potential changes in survey effort between the 1995 – 1997 and 2012 – 2014 epochs.

Sampling occurred in the eastern regions of the KB subpopulation (i.e., near the Humboldt Glacier) during 1995 – 1997 and 2012 – 2014. When the eastern parts of Kane Basin (i.e., the areas east of the mid-sector line in the Nares Strait-Kane Basin area off the Humboldt Glacier) were surveyed in 1994 and 1995, only few signs of polar bear activity (i.e., tracks) were observed there and consequently only a few bears were tagged (0 in 1994 and 4 in 1995; E. Born, pers. obs., Taylor et al. 2001). Similarly, no bears were found and tagged there in 1996 and only 3 in 1997 (Taylor et al. 2001). The apparent very low densities of polar bears in eastern KB was assumed to reflect a long-term avoidance response because eastern KB has been hunted relatively intensively by hunters from the Qaanaaq region and in particular after it no longer became permitted for Greenland hunters to hunt polar bears in Canadian territory in the late

1960s. The presence of ringed seals in eastern KB was noted during the surveys in the 1990s and it was concluded that these areas were favourable habitat for polar bears (E. Born, pers. obs., Taylor et al. 2001).

However, it cannot be precluded that, this difference between the 1990s and the 2000s in the spatial distribution of bears, to a certain extent reflect some differences in sampling protocols, including increased survey intensity near the Humboldt Glacier and more uniform distribution of effort during 2012 – 2014. However, during both periods relative allocation of survey effort to a certain extent was decided based on assumptions of what was suitable polar bear habitat (i.e., areas with anticipated polar bear occurrence and/or areas with observed signs of polar bear habitat). In the 2000s large areas in central and southern KB with relatively open pack ice were not surveyed although satellite telemetry (Chapter 2) and aerial surveys (Heide-Jørgensen et al. 2013) indicate the presence of polar bears in this habitat albeit likely few. Whereas in the 1990s when the spring sea ice in KB was more consolidated (E. Born, pers. obs; Chapter 9) areas with little or no signs of polar bear activity in the eastern parts were surveyed less intensively.

It should be mentioned that sea-ice dynamics in Kane Basin also have changed since the 1990s (Chapter 9), limiting access of hunters from Greenland to the eastern parts of the region (Born et al. 2011) possibly resulting in an increased occurrence of polar bears in this area. Hence, we hypothesize that these differences in sea-ice dynamics and associated change in hunting pressure have led to the apparent shift in the distribution of bears toward the eastern parts of the Nares Strait-Kane Basin region.

We note that the Horvitz-Thompson estimator used to generate estimates of abundance in Kane Basin yielded biologically implausible rates of growth from 1994 to 1995 (119 – 318), 1995 to 1996 (318 – 189), and 2012 to 2013 (221 – 328; Table 10.4). Although the apparent changes in sampling frame (and thus the definition of the effective study population) from 1994 to 1995 may contribute to this finding in part, this result is primarily an artifact of the estimator itself. Horvitz-Thompson (H-T) estimators are calculated as $\widehat{N} = \frac{n}{p}$ for each group (e.g., age class and sex), and the total abundance estimate is derived by summing estimates across all groups. As such, H-T estimators are sensitive to sample size, particularly if recapture probabilities are estimated as temporal constants. Because data for this analysis were very sparse and models specifying inter-annual variation in estimates of recapture probability were not supported, our Horvitz-Thompson estimates of abundance are influenced by variation in annual sample size of captured bears (n). We attempted to address this issue by integrating annual random effects for estimating recapture probability with complementary Markov Chain Monte Carlo (MCMC) analyses, but this approach did not resolve the issue. Given this limitation, we do not recommend interpreting inter-annual variation in estimates of abundance, and suggest that the larger estimate of abundance in 2013-2014 may be partially attributed to larger sample sizes in those years, compared to 1995-1997.

Our estimates of survival for dependent bears and age 3+ females are consistent with previous work in Kane Basin (Taylor et al. 2008), and estimates of unharvested survival rates of independent females appear capable of supporting positive subpopulation growth (Regehr et al. 2015). Our estimates of age 3+ male survival (present analysis: age 3+ males, 0.87, SE 0.06) are lower than previous work in KB (Taylor et al. 2008: age 5+ males, 0.96, SE: 0.05) but consistent with estimates of adult male survival derived in some other studies (e.g., Stirling et al. 2011, Peacock et al. 2012). We further note that data on males were particularly scant in this study (Table 10.1): no male bears initially marked in the 1990s were subsequently recaptured in the

2000s, and very few individuals were reported in the harvest over the past 15 years. We are uncertain if the disparity in estimates of male survival between our study and Taylor et al. (2008) reflects differences in modeling approaches, such as the broader age class designations in the present study and consideration of different model structures (unlike Taylor et al. [2008], we did not consider a structure in which S was estimated as constant between the sexes for age 3+ individuals); our inclusion of longer-term data; a disproportionate impact of the changing environment on males; reduced fidelity of males to the study area that was not effectively captured by our model-based estimates of the fidelity (*F*) parameter given the paucity of recovery data; under-reporting of male bears in the harvest; or some combination thereof.

Although we calculated estimates of natural survival following previous studies (e.g., Taylor et al. 2005), we note that the formula used to derive these estimates [$S + r * (1 - S)$] makes simplifying assumptions that can introduce bias into estimates of unharvested *S* under some conditions, although the impacts of using this vs. an alternative equation is likely minimal for KB data due to high estimates of *S* and low estimates of *r*. The documented under-reporting of marked bears in the harvest leads to an underestimation of natural survival, but this may be offset, to some extent, by a likely violation of the assumption that harvest mortality is completely additive. For example, Taylor et al. (2008) estimated natural survival for both adult females and males to be 0.997, meaning that virtually all mortality of adult (age 5+) bears in Kane Basin during 1992 - 1998 resulted from harvest, and <1 in 300 adult bears would die annually in the absence of harvest. Although Taylor et al. (2008) did not report their estimates of *r*, back calculating from survival rates in their Table 3 yields unrealistically high estimates of *r* = 0.91 for adult females and *r* = 0.93 for adult males, so their estimates of natural and harvest mortality should be treated cautiously.

Changing sea-ice conditions, a reduction in accessibility of Kane Basin to hunters from Canada and (in the Humboldt Glacier region) Greenland, and the implementation of a quota in Greenland have contributed to a net reduction in harvest since the 1990s (Chapter 8). This decline is reflected in estimates of the reporting parameter, as r is estimated lower for the period from 2006 – 2014 than 1992 – 2005 (Table 10.3). We note, however, that under-reporting of harvest, which anecdotally appears to increase with greater marker age (Chapter 8), also may contribute to lower estimates of r during 2006 – 2013, especially given the 14-year interval without live recaptures.

Despite the sparseness of the data and the unusual study design (6-year and 3-year live capture sessions connected by a 14-year period with dead recoveries only), we were able to generate estimates of F that seem biologically realistic (age 3+ males: 0.96; females and dependent bears: 0.98): polar bears show strong seasonal fidelity to the region in which they were captured, but a small proportion of individuals permanently emigrate to other subpopulations. These estimates appear consistent with findings from satellite telemetry data and capture records (Taylor et al. 2001, Chapters 2 and 9).

Our study indicates that there has been no decline in the size of the KB subpopulation. These findings are in accordance with TEK (Born et al. 2011).

Using estimates of subpopulation size in KB and vital parameters from the 1990s, York et al. (2016) modeled a decline in KB with an estimated subpopulation size of zero in 2013. According to York et al. (2016:9,18) the projected decline is consistent with TEK. It is mentioned that KB has been subject to chronic long-term overharvest and would not persist if it did not receive immigrants from adjacent subpopulations (Ibid.). As basis for the TEK information York et al. (2016) cite COSEWIC (2008) and M. Taylor (pers. comm. 1986-2008) in the text, and COSEWIC (2008), CWS (2009) and PBTC (2014) in their table 2b but not a comprehensive TEK study in which NW Greenland polar bear hunters were interviewed (Born et al. 2011).

During this interview survey in Greenland experienced polar bear hunters who had been hunting in Kane Basin were of the opinion that polar bears in this region had expanded their range. Previously the hunters had to travel north to Washington Land (*ca.* 80° N) to find bears, whereas nowadays they only have to go as far as Inglefield Land (*ca.* 78° 30' N) to hunt polar bears in the eastern Nares Strait-Kane Basin region. Their reason for this was because "the bears have come closer" (Born et al. 2011:75,79). It was mentioned that previously polar bears were scarce in the eastern Kane Basin area (i.e., in front of the Humboldt Glacier) but now had expanded their range from Ellesmere Island eastward to the Nares Strait-Kane Basin region (ibid:80). Some of the interviewees were of the opinion that this change represented an increase in the number of polar bears. It was also mentioned that the hunting pressure in Kane Basin had decreased because poor sea-ice conditions (i.e., lack of dense sea ice) had made travels with dog sleds north more difficult (Born et al. 2011). Hence, information from experienced polar bear hunters in NW Greenland indicates that the KB-polar bear subpopulation has expanded its distribution area and increased in size which is in accordance with our study.

This study and concurrent research in the neighboring Baffin Bay subpopulation (Chapter 5) represent the first attempts to implement genetic mark-recapture for polar bears at a subpopulation-wide scale. The ability to successfully genotype bears from archived tissue and samples obtained via remote biopsy darting, combined with the ability to analyze data in well-established mark-recapture models, suggest that this approach is a promising tool for future polar bear inventories. However, there are some limitations of the technique including a generalized age structure as bears are identified from the air. We opted for a conservative approach when designating age classes because there was inherent uncertainty in estimating age class remotely rather than aging via physical examination or with annular rings from an extracted tooth. However, our ability to accurately classify bears by age-class remotely (Chapter 5) suggests that future studies may be able to increase the resolution of age classification for obtaining estimates of survival (and other parameters), thus enhancing the utility of the technique.

10.5. Literature Cited

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Atwood, T., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. E. Bromaghin. 2015. Evaluating and ranking threats to the long-term persistence of polar bears. U.S. Geological Survey Open-File Report 201-1254. U.S. Geological Survey, Reston, Virginia, USA.

Bolker, B. 2016. Ecological Models and Data in R. R package version 1.3.9.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological

Applications 25:634–651.

Burnham, K. P. 1993. A theory for combined analysis of ring recovery and recapture data. Pages 199–213 *in* J. D. Lebreton and P. M. North, editors. Marked individuals in the study of bird populations. Birkhauser-Verlag, Basel, Switzerland.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Comiso, J. C., C. L. Parkinson, R. Gertsen and L. Stock. 2008. Accelerated decline in the Arctic sea ice cover. Geophysical Research Letters 35:L01703.

COSEWIC [Committee on the Status of Endangered Wildlife in Canada]. 2008. COSEWIC assessment and update status report on the polar bear Ursus maritimus in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, vii + 75 pp.

CWS [Canadian Wildlife Service]. 2009. Nunavut Consultation Report. Consultations on the Proposed Listing of the Polar Bear as Special Concern under the Species at Risk Act. Conducted February-April 2009, Iqaluit, Nunavut, 247 pp.

Heide-Jørgensen, M. P., L. M. Burt, R. G. Hansen, N. H. Nielsen, M. Rasmussen, S. Fossette, and H. Stern. 2013. The significance of the North Water Polynya to Arctic top predators. Ambio 42:596–610.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. Journal of Wildlife Management 73:3–17.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock’s robust design. Ecology 78:563–578.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications 18:S97–S125.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D.

Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. Molecular Ecology 12:1375–1387.

Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571–1584.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014: Status table. <http://pbsg.npolar.no/export/sites/pbsg/en/docs/status-table-2014.pdf>. Accessed 4 May 2016.

PBTC [Canadian Polar Bear Technical Committee]. 2014. 2014 Polar Bear Technical Committee Status Table, unpublished data.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Powell, L. 2007. Approximating variance of demographic parameters using the delta method: A reference for avian biologists. Condor 109:949–954.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Sahanatien, V., and A. E. Derocher. 2012. Monitoring sea ice habitat fragmentation for polar bear conservation. Animal Conservation 15:397–406.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Macmillan, New York, New York, USA.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stirling, I., C. Spencer, and D. Andriashek. 1989. Immobilization of polar bears (*Ursus maritimus*) with Telazol® in the Canadian Arctic. Journal of Wildlife Diseases 25:159–168.

Stirling, I., T. L. McDonald, E. S. Richardson, E. V. Regehr, and S. C. Amstrup. 2011. Polar bear population status in the northern Beaufort Sea, Canada, 1971-2006. Ecological Applications 21:859–876.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. Global Change Biology 18:2694–2706.

Stroeve, J. C., M. C. Serreze, M. M. Holland, J. E. Kay, J. Malanik, and A. P. Barrett. 2012. The Arctic's rapidly shrinking sea ice cover: A research synthesis. Climatic Change 110:1005–1027.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46:S120–S139.

York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. Ecology and Evolution 6:2897-2924.

Table 10.1. Summary table of live captures and dead recoveries during the mark-recapture study of the Kane Basin polar bear subpopulation in Nunavut, Canada, and Greenland, 1992 – 2014. Shaded cells indicate that data are not possible due to an absence of marking or recapture.

	Initial captures						**Live recaptures**				**Dead recoveries**					
	Females			*Males*			*Females*		*Males*		*Females*			*Males*		
	Coy	**Yrl / 2yr**	**3+**	**Coy**	**Yrl / 2yr**	**3+**	**Yrl / 2yr**	**3+**	**Yrl / 2yr**	**3+**	**Coy**	**Yrl / 2yr**	**3+**	**Coy**	**Yrl / 2yr**	**3+**
1992	4	0	7	2	0	3					0	0	1	0	0	0
1993	1	3	6	3	1	8	0	2	0	0	0	0	0	0	0	0
1994	2	0	9	3	0	4	1	3	2	2	0	0	0	0	0	0
1995	12	3	21	5	2	13	0	7	0	6	0	0	1	0	0	1
1996	5	2	8	2	2	4	1	7	0	5	0	0	0	0	0	1
1997	0	4	4	3	1	3	1	8	0	8	0	1	1	0	0	2
1998												0	3		0	2
1999													2			2
2000													0			0
2001													1			0
2002													0			1
2003													2			0
2004													1			0
2005													0			0
2006													0			0
2007													0			0
2008													0			0
2009													0			0
2010													0			1
2011													0			0
2012	2	3	19	1	4	11		2		0	0	0	1	0	0	0
2013	6	4	20	2	2	19	0	9	1	3	0	0	0	0	0	0
2014	0	2	21	0	1	10	2	12	0	7	0	0	0	0	0	0
Totals	**32**	**21**	**115**	**21**	**13**	**75**	**5**	**50**	**3**	**31**	**0**	**1**	**13**	**0**	**0**	**10**

Table 10.2. Model selection results (< ΔAIC_c 4) from analysis of mark-recapture-recovery data from the Kane Basin polar bear subpopulation, 1992 – 2014. Coy = cubs of the year. Yrl = yearlings and 2-year olds. 3+ = bears aged 3 and older. For p, family = females / dependent bears and independent males (2 age / sex classes); and epoch = sampling period (1992 – 1997; 2012 – 2014). For r, time = 1992 – 2005 and 2006 – 2013.

Model Structures							
S	p	r	F	**Parameters**	**ΔAICc**	**AICc Weights**	**Deviance**
coy, yrl, 3+(add sex with yrl)	Family	yrl, 3+(sex + time)	Constant	13	0	0.23	723.71
coy, yrl, 3+(add sex with yrl)	Family	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	14	1.22	0.13	722.76
coy, yrl, 3+(add sex with yrl)	family + epoch	yrl, 3+(sex + time)	Constant	14	1.23	0.13	722.77
coy, yrl, 3+(add sex with yrl)	Constant	yrl, 3+(sex + time)	Constant	12	1.88	0.09	727.74
coy, yrl, 3+(sex)	Family	yrl, 3+(sex + time)	Constant	13	2.19	0.08	725.90
coy, yrl, 3+(add sex with yrl)	family + epoch	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	15	2.57	0.06	721.94
coy, yrl, 3+(add sex with yrl)	family * epoch	yrl, 3+(sex + time)	Constant	15	2.62	0.06	721.99
coy, yrl, 3+(sex)	Family	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	14	2.83	0.06	724.38
coy, yrl, 3+(sex)	Constant	yrl, 3+(sex + time)	Constant	12	3.06	0.05	728.92
coy, yrl, 3+(add sex with yrl)	Epoch	yrl, 3+(sex + time)	Constant	13	3.37	0.04	727.08
coy, yrl, 3+(sex)	family + epoch	yrl, 3+(sex + time)	Constant	14	3.45	0.04	724.99
coy, yrl, 3+(add sex with yrl)	Constant	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	13	3.90	0.03	727.61

Table 10.3. Model averaged (<Δ 4 AICc) parameter estimates for the Kane Basin polar bear subpopulation obtained from mark-recapture study, 1992 – 2014.

Parameter	Class	Estimate (SE)
Total Survival (*S*)		
	Cubs of the year	0.45 (0.15)
	Yearlings / 2-year old females	0.73 (0.13)
	Yearlings / 2-year old males	0.52 (0.17)
	3+ females	0.95 (0.04)
	3+ males	0.87 (0.06)
Reporting (*r*)		
	Yearlings / 2-year olds	0.04 (0.04)
	3+ females, 1992 – 2005	0.42 (0.26)
	3+ females, 2006 - 2013	0.09 (0.08)
	3+ males, 1992 – 2005	0.32 (0.12)
	3+ males, 2006 – 2013	0.06 (0.05)
Fidelity (*F*)		
	Cubs of the year, yearlings, 2-year olds, and 3+ females	0.98 (0.04)
	3+ males	0.96 (0.07)

Table 10.4. Model averaged (<Δ 4 AICc) estimates of abundance ($\widehat{N}$ ± SE; [95% Confidence Interval]) of the Kane Basin polar bear subpopulation from mark-recapture study, 1992 – 2014.

1993	1994	1995	1996	1997	2012	2013	2014
120 ± 19 (83-156)	119 ± 21 (77-160)	318 ± 53 (214-429)	189 ± 36 (119-259)	164 ± 28 (110-218)	221 ± 41 (141-301)	328 ± 60 (211-445)	385 ± 78 (233-537)

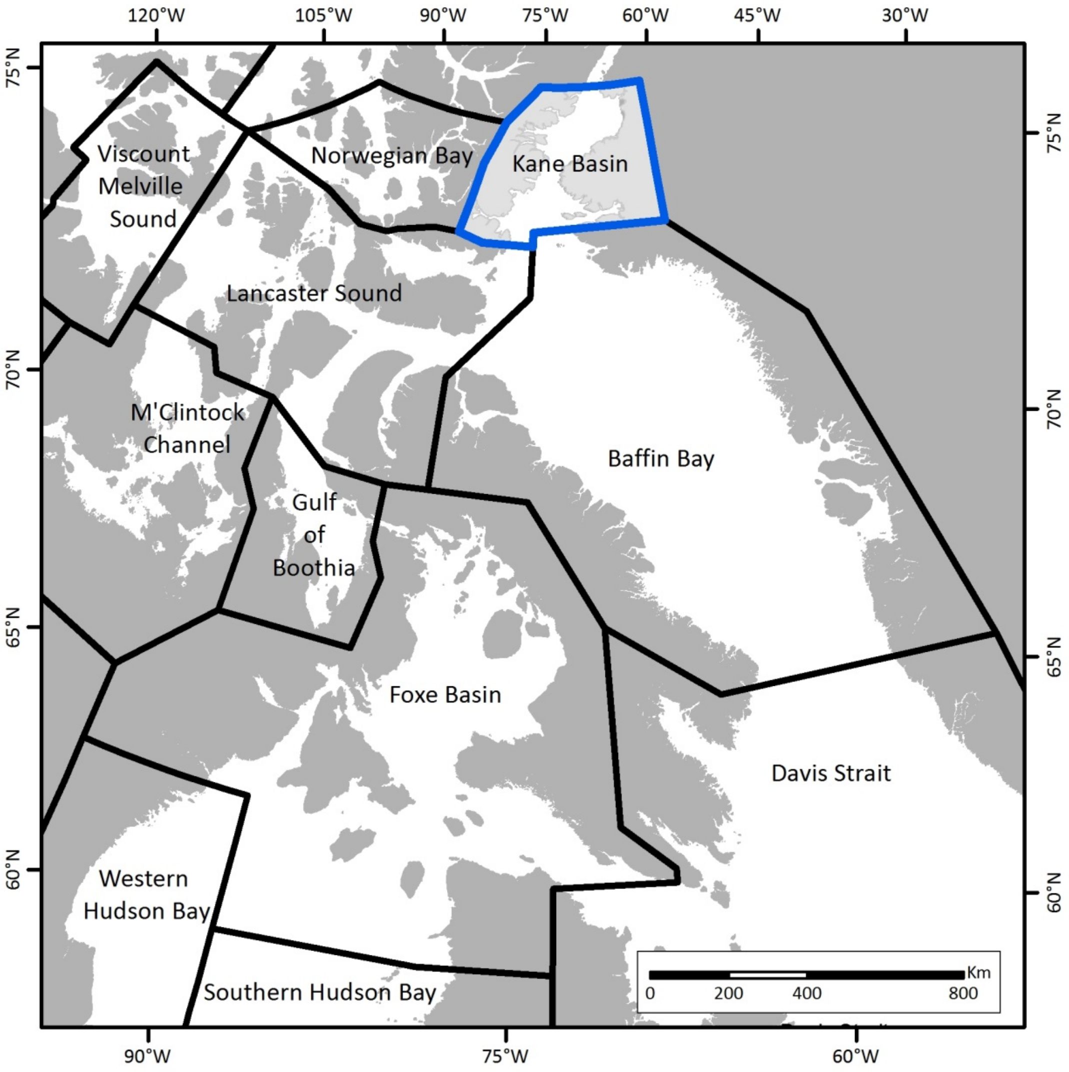


Figure 10.1. The Kane Basin polar bear subpopulation is located between Nunavut, Canada and Greenland and is regarded as belonging to the Arctic Archipelago region.

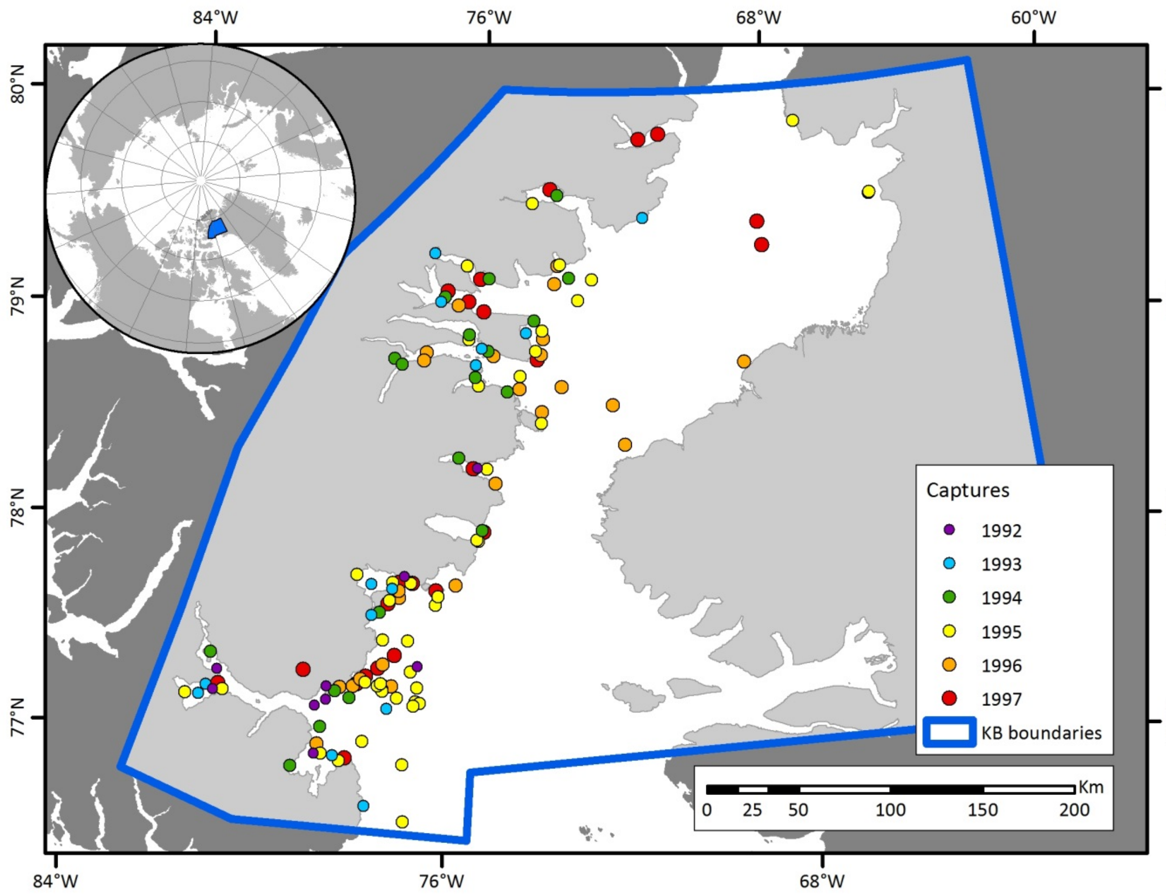


Figure 10.2. Locations of polar bears captured in the Kane Basin subpopulation during springtime, 1993 – 1995 and 1997. Kane Basin is highlighted in blue in the inset.

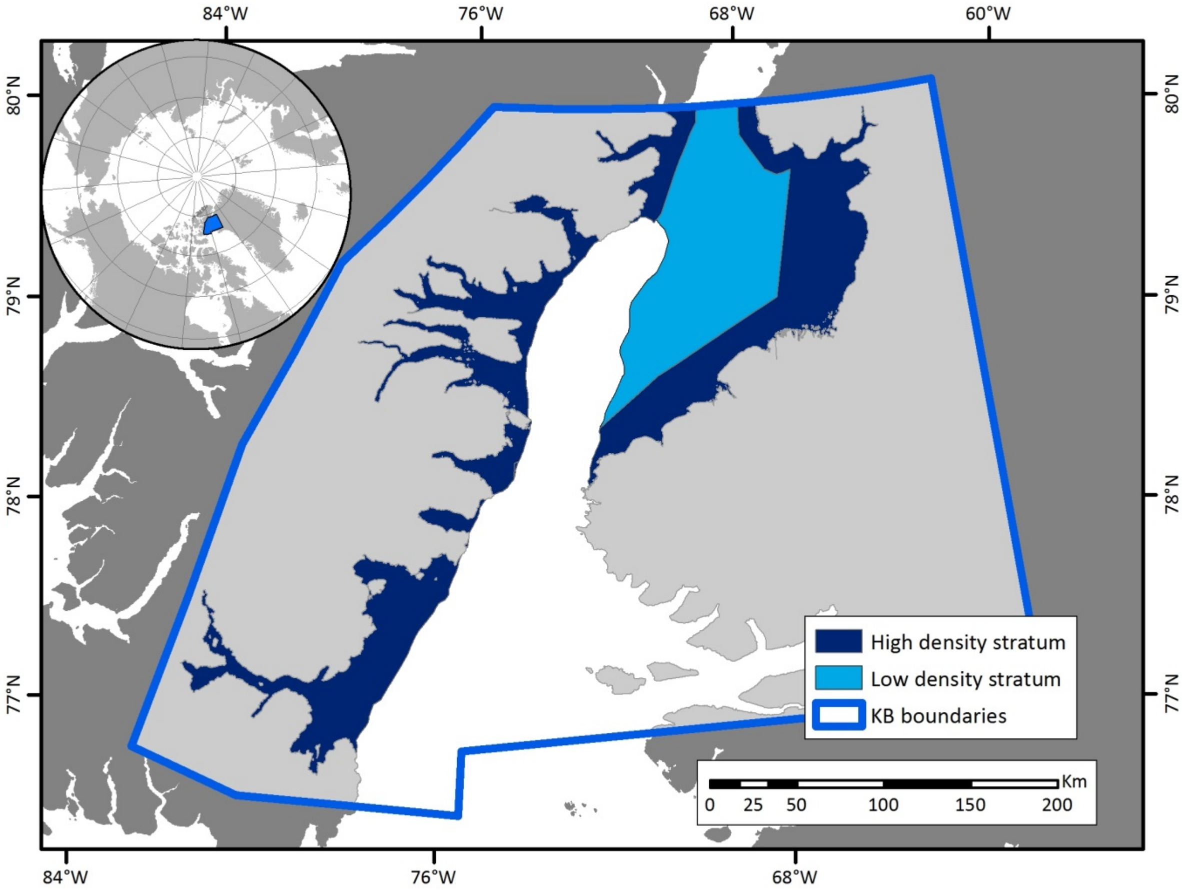


Figure 10.3. Sampling strata for genetic mark-recapture and aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

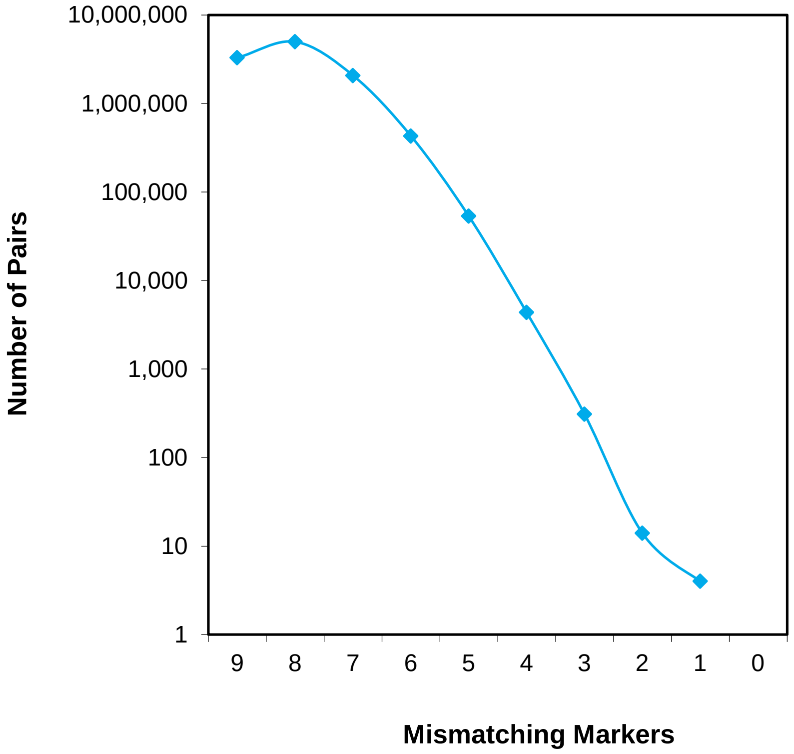


Figure 10.4. 9-locus mismatch distribution for 4,657 polar bears from Nunavut and the Greenland side of the Baffin Bay and Kane Basin polar bear subpopulations.

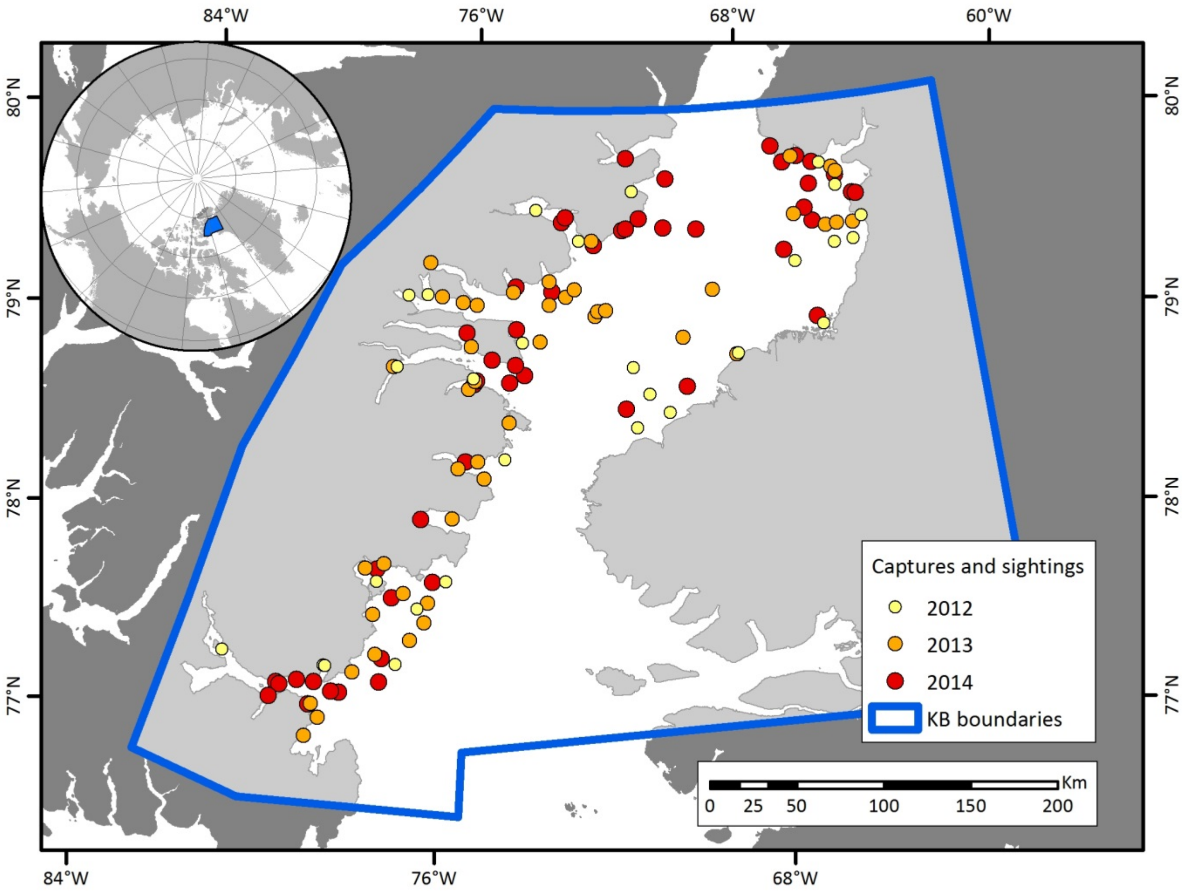


Figure 10.5. Locations of polar bears sighted in Kane Basin during research in April and May, 2012 – 2014. Kane Basin is highlighted in blue in the inset. The North Water polynya varied among years, but in general, included the south-central portion of the subpopulation in all years. We did not sample sea ice in southeastern Kane Basin due to logistical constraints presented by the polynya and anticipated low densities.

CHAPTER 11

AERIAL SURVEY OF POLAR BEARS IN KANE BASIN

KEY FINDINGS

- The estimate of abundance based on the springtime 2014 aerial survey in KB was 206 bears (95% lognormal CI: 83 - 510). However, due to insufficient coverage of offshore polar bear habitat this estimate is likely negatively biased.
- Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of total abundance 2013-2014 from the MR study (357 bears, 95% CI = 221 – 493) and the aerial survey estimate was approximately 151 bears (standard deviation of the difference ≈ 127 bears), with 88% of the sampling distribution suggesting that the difference was positive (i.e., that the MR estimate was at least one bear larger than the aerial survey estimate).
- Differences between MR and aerial survey point estimates in KB require caution when comparing results from different techniques conducted during springtime. Aerial surveys yield a snapshot estimate of abundance, whereas MR generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period. We suggest that the MR estimate is appropriate for use in management.
- As shown also in other areas of the Arctic aerial surveys provide a useful tool for inventorying polar bear subpopulations and the method has been used on even larger subpopulations than KB in remote areas (e.g. the Barents Sea).
- The springtime aerial survey was successfully implemented due to the small geographic area and a period of good weather, but precision could be improved by increasing survey effort to better estimate the detection function and by ensuring that the entire range of the subpopulation is covered.
- Aerial surveys of polar bears that also range in areas with offshore loose drift ice and open water, like Kane Basin, should be conducted from fixed-winged aircraft with a longer endurance than the single-engine helicopter used in the present study. This allows for offshore polar bear habitat to be monitored and will result in a more accurate estimate of abundance.

11.1. Introduction

Physical mark-recapture has formed the basis for demographic studies of polar bears throughout the North American Arctic (e.g., Taylor et al. 2005, 2008, Peacock et al. 2013).

Capture-based research has generated information on abundance, vital rates, and harvest management, and facilitated a variety of other studies, including assessments of body condition (e.g., Rode et al. 2012), movements, habitat use and spatial ecology (e.g., Durner et al. 2009, Cherry et al. 2013), and diet (e.g., Thiemann et al. 2008). Over the past decade, however, jurisdictions have invested substantial resources in the development and implementation of less invasive monitoring techniques, in part to better address social concerns regarding wildlife handling and immobilization. Genetic mark-recapture, one such alternative method, has been used to estimate the number of polar bears using whale carcasses in Alaska (Herreman and Peacock 2013) and, more recently, to estimate the abundance and associated vital rates for the Baffin Bay and Kane Basin subpopulations (see Chapters 5 and 10, respectively). Aerial surveys also have been widely implemented, including studies conducted over land in seasonally ice-free subpopulations (Stapleton et al. 2014, 2016, Obbard et al. 2015) and over land and sea ice in the Barents Sea (Aars et al. 2009). Aerial surveys yield less detailed information on sex, age, body condition, and vital rates than both physical and genetic mark-recapture methods, but they can enable more frequent monitoring, an important consideration in the face of a rapidly changing Arctic.

Our objective was to evaluate the feasibility of estimating abundance with an aerial survey flown over springtime sea ice in the Kane Basin (KB) subpopulation. We designed and implemented the aerial survey to be conducted alongside a concurrent mark-recapture study in the KB subpopulation during 2014. This protocol ensured consistency in the sampling frames and study periods. It also allowed us to derive independent estimates of abundance from the two techniques, enabling us to directly compare and assess the results of the 2 methods. This important step is necessary to properly integrate population estimates derived from different survey techniques. Although research elsewhere in has facilitated broad comparisons between mark-recapture and aerial survey methods (Western Hudson Bay – Stapleton et al. 2014, Lunn et al. 2016; Southern Hudson Bay – Obbard 2008, Obbard et al. 2015), the work in Kane Basin represents the first study in which an aerial survey was designed and implemented with a sampling frame identical to a simultaneous mark-recapture study. Aerial surveys yield snapshot estimates of abundance (i.e., the number of bears occupying the survey area during the study period; Buckland et al. 2001), whereas mark-recapture generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period (including individuals that are currently outside the survey area due to temporary emigration; Kendall et al. 1997). Because there is a lack of geographic closure among polar bear subpopulations such that they are only partially discrete (Taylor et al. 2001, Chapter 9), we hypothesized that our aerial survey-based estimate would be smaller than our mark-recapture-based estimate.

11.2. Materials and Methods

Study Area

The KB subpopulation covers ~150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (boundaries evaluated in Taylor et al. 2001; Figure 10.1). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq areas). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the Baffin

Bay (BB) and Lancaster Sound (LS) subpopulations, and to the west by Norwegian Bay (NW; PBSG 2010). The KB subpopulation is regarded as belonging to the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, sea-ice conditions have changed markedly in the Kane Basin region in recent decades (Born et al. 2011; Figure 9.2). KB is partially connected to neighboring subpopulations; particularly notable interchange occurs with BB and LS. The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a barrier between KB and BB – LS.

Field Sampling

Using a helicopter (Bell 206 LongRanger), we implemented a line-transect aerial survey over springtime sea ice in the KB subpopulation during 28 April – 12 May 2014. To efficiently allocate effort and ensure that the study area was sampled as comprehensively as possible, we stratified the subpopulation into high- and low-density areas based on observations of polar bears during 2012 and 2013 mark-recapture surveys (i.e., presumed densities; see Chapter 10). Strata conformed to general landscape features and ice types: the high-density stratum included landfast ice along the coastline and within fjords as well as nearshore pack ice within ~30 km of the nearest land mass (~18,870 km^2), whereas the low-density stratum included pack ice located farther offshore (~9,110 km^2; Figure 11.1). Since the survey was conducted after adult females had left dens, we assumed that no bears were located on land during the study period. We used

GIS layers from Greenland and Nunavut to delineate the landward extent (i.e., coastline) of the study area. We delineated the extent of available habitat by approximating the edge of the North Water polynya with Moderate Resolution Imaging Spectroradiometer (MODIS; http://modis.gsfc.nasa.gov/) images (1 km resolution). The polynya's boundaries can change rapidly, so we delineated the extent of the polynya adjacent to the section surveyed on a particular day using MODIS imagery from that day, or from the closest date possible when same-day imagery was unclear due to atmospheric conditions. We also examined the delineated study area in relation to weekly regional sea-ice charts produced by the Canadian Ice Service (https://www.ec.gc.ca/glaces-ice/). During sampling, we collected GPS waypoints at the edge of the polynya to verify delineation. We did not sample in the polynya due to safety considerations. Polar bears occur in the polynya area (Heide-Jørgensen et al. 2013) but ice conditions in spring 2014 suggested that it was not suitable springtime habitat for polar bears due to its thin, forming (i.e., new and grey) ice and expansive open water (*cf*. Sahanatien and Derocher 2012). We also did not survey the sea ice in the fjords (~3,245 km^2; Figure 11.1) of the populated Qaanaaq area in NW Greenland (i.e., in the eastern parts of the North Water polynya) because hunting pressure for marine mammals in these areas is generally high and consequently "resident" polar bears do not exist in the Qaanaaq area (Born et al. 2011, E. Born, pers. obs.).

Aerial transects were systematically spaced at 6-km and 18-km width intervals in the high- and low-density strata, respectively, based on anticipated encounter rates and available resources. We arranged transects in an east – west direction in open areas, but oriented them perpendicular to fjords (i.e., across the widths of fjords) to improve variance estimation (i.e., more numerous short transects) and reduce bias (i.e., sighting distances did not reflect potential density gradients, with highest densities along the sides of fjords; Figure 11.2).

During line-transect sampling, we surveyed at an altitude of ~120 m and groundspeed of ~150 km / hr. We sampled from most planned transects and included some ferry flights (during which we sampled) that were random with respect to the distribution of bears and presumed density gradients (i.e., highest densities near the polynya edge and along the sides of fjords) in analyses. Although many groups were observed during flights between consecutive transects, these typically occurred near the sides of fjords. As such, observations may have reflected a density gradient as well as the probability of detection and were thus inappropriate to include in distance sampling analyses (Stapleton et al. 2014).

We collected aerial survey data with mark-recapture distance sampling protocols (Laake and Borchers 2004, see also Stapleton et al. 2014, 2016). Two front (including the pilot) and two rear observers comprised the first and second capture periods, respectively, and teams of observers worked independently until both groups were afforded a full opportunity to observe a bear. After announcing a sighting, we flew off-transect to record the bear's initial location with a GPS, and we later estimated distance from transects in a GIS (Marques et al. 2006). During off-transect flights, we flew to within ~5 – 10 m of bears to obtain a tissue sample via biopsy darting for genetic analysis (see Chapter 10) and to estimate sex and age class of the bear. For each sighting, we recorded 3 covariates that potentially impacted detection probability: 1) habitat structure within a 30-m radius (smooth / low structure or moderate to high structure; i.e., smooth versus rough ice); 2) visibility (good or poor, due to fog, glare or precipitation); and 3) light conditions (i.e., cloud cover; clear: 0 – 25%; partly cloudy: 25 – 50%; mostly cloudy: 50 – 75%; or overcast: 75 – 100%).

Statistical Analyses

We analyzed line-transect data using distance sampling, which fits a function to observational data to describe how detection changes with increasing distance from the sampling transect (Buckland et al. 2001). We initially intended to use double observer (i.e., mark-recapture) distance sampling for analyses, but small sample sizes precluded this approach. We defined clusters as discrete groups of bears with non-independent detection probabilities (i.e., an adult female with 1 or more offspring or a breeding pair). We first examined a left-truncated data set (i.e., 75 m was subtracted from all observations; observations within 75 m were censored to account for blind spots directly beneath the helicopter; e.g., Borchers et al. 2006, Stapleton et al. 2014) to evaluate distance sampling's fundamental assumption of complete detection on the transect line (Buckland et al. 2001). Because these results indicated that the probability of detection by at least one observer was >96% at the adjusted transect line, we considered this assumption to be approximately met and proceeded with analyses including all observations (i.e., data were not left-truncated).

We completed analyses in the mark-recapture distance sampling (MRDS) engine of Program DISTANCE 6.2 (Thomas et al. 2010) and modeled the survey data as a single-observer study. We examined half-normal and hazard rate key functions and used multiple covariate distance sampling (Marques and Buckland 2003) to include a maximum of 1 covariate per model due to sample size constraints. We condensed light conditions into a binary covariate (0 – 25% cloud cover; >25% cloud cover) due to underrepresentation of some values. We considered each transect the sampling unit for variance estimation and used the Innes et al. (2002) method to estimate variance associated with global density and overall abundance.

11.3. Results

We surveyed 4,160 km of transects, including 3,850 km along 234 transects in the high-density stratum and 610 km along 14 transects in the low-density stratum. We observed 29 groups of polar bears (Figure 11.2), including 49 total bears (30 independent bears); cub-of-the-year and yearling litter sizes in this sample averaged 1.55 (SD: 0.5, $n = 11$) and 1.0 (SD: 0.0, $n = 2$), respectively. We right-truncated sightings data at 1,400 m to improve model fit and parsimony (Buckland et al. 2001), censoring one observation of an independent bear at >3,500 m, leaving 28 groups for estimating the detection function and abundance; 27 of these sightings occurred in the high-density stratum.

Sighting distance was not correlated with polar bear group size ($r = -0.10$, $P = 0.61$), so we used mean group size for abundance estimation. Histograms summarizing sightings distances indicated strong-support for a distance-based detection function (Figure 11.3), and all highly supported distance sampling models indicated adequate goodness-of-fit (chi-squared, Cramér-von Mises and Komolgorov-Smirnov tests: $P > 0.05$). The most highly supported model (half-normal key function) suggested that light conditions (cloud cover) affected detection probability (Figure 11.4). However, the small number of observations (see Buckland et al. 2001: at least 60 – 80 sightings are recommended for estimating the detection function) resulted in uncertainty in density and abundance estimation, and a model with a hazard rate key function had nearly equivalent support and estimated much higher densities (Figure 11.3, Table 11.1). Thus, we elected to model-average (Burnham and Anderson 2002) the 2 most highly supported models and obtained a subpopulation-wide estimate of 206 bears (SE: 101; 95% lognormal CI: 83 – 510; CV: 49%) in 2014.

11.4. Discussion

The estimate of abundance based on the springtime 2014 aerial survey in KB was 206 bears (95% lognormal CI: 83 - 510). However, due to insufficient coverage of offshore polar bear habitat (vast areas of offshore habitat in the North Water Polynya was not surveyed) this estimate is likely negatively biased. The estimate of abundance obtained from the aerial survey was negatively biased by about 30% or more (see below).

Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of total abundance 2013-2014 from the MR study (357 bears, 95% CI = 221 – 493) and the aerial survey estimate was approximately 151 bears (standard deviation of the difference ≈ 127 bears), with 88% of the sampling distribution suggesting that the difference was positive (i.e., that the MR estimate was at least one bear larger than the aerial survey estimate).

Differences between MR and aerial survey point estimates in KB require caution when comparing results from different techniques conducted during springtime. Aerial surveys yield a snapshot estimate of abundance (i.e., the number of bears occupying the survey area during the study period), whereas MR generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period (including individuals that are currently outside the survey area due to temporary emigration) (Kendall et al. 1997). We suggest that the MR estimate is appropriate for use in management.

This finding reinforces that boundaries between subpopulations are not discrete, a result consistent with satellite telemetry (Taylor et al. 2001, Chapter 9) and capture and harvest records (e.g., Peacock et al. 2012, Chapter 10). Interchange among subpopulations is particularly prevalent during the springtime (Chapter 9), meaning that a large number of bears were likely exposed to sampling during the 3-year mark-recapture study period in the KB subpopulation.

These seasonal movement patterns contribute to the finding that the mark-recapture point estimate was ~85% greater than the aerial survey point estimate. We note that data used for the mark-recapture analysis were sparse and precluded implementing the Barker model (1997, 1999) to explicitly model temporary emigration. However, such an approach would enable a more direct comparison between methods by defining the mark-recapture estimate as pertaining to only those bears that were present in the study area and available for capture, rather than the entire super-population.

Although the aerial survey was not ideally designed (it relied up the use of a single-engine helicopter with limited range so that offshore habitat could not be surveyed), the KB subpopulation study provides the first opportunity to directly compare simultaneous mark-recapture and aerial survey studies. In Western Hudson Bay, estimates of abundance derived from mark-recapture and an aerial survey were similar (although the aerial survey snapshot estimate was somewhat greater than the mark-recapture super-population estimate), but differences in sampling frames limited inference (Stapleton et al. 2014, Lunn et al. 2016). Similarly, abundance estimates from an aerial survey and mark-recapture in Southern Hudson Bay were consistent, but several years elapsed between the inventories, and the mark-recapture estimate was adjusted upwards to reflect potential heterogeneity in capture probabilities and to account for un-sampled areas (Obbard 2008, Obbard et al. 2007, 2015). As jurisdictions incorporate alternative (non-capture based) methods for estimating abundance and monitoring populations, understanding the ability to compare results from different techniques will be critical to correctly interpreting status and trend (Stapleton et al. 2014). The differences in survey methods resulting in estimates of different “populations” (i.e., the MR estimate of the “super”-population versus the aerial survey´s real-time snapshot of abundance) suggest caution when comparing results and assessing trends from different techniques implemented during the springtime, when polar bear movements among subpopulations are greatest.

We acknowledge that our estimate of abundance derived from the aerial survey is likely biased low. First, we did not sample the southeastern portion of the KB subpopulation and the large area of the North Water polynya because of logistical and safety considerations in a helicopter and the presumed relatively low densities of bears in these regions (Heide-Jørgensen et al. 2013). Satellite telemetry data indicated no collared bears (n = 20 adult females) were present in the un-surveyed areas during the aerial survey sampling period (see also Chapter 9). However, the un-sampled regions covered extensive areas (sea ice near Qaanaaq: 3,245 km^2, and the North Water polynya: 27,214 km^2), such that even very low densities may significantly contribute to an overall estimate of abundance. This unsurveyed area in the North Water Polynya amounts to *ca.* 34% - 40% of the extension of the polynya (70,000-80,000 km^2; Born et al. 2004 and references therein). Extrapolating our model-averaged estimate of density from the low-density stratum (3.39 bears / 1,000 km^2) to the sea ice near Qaanaaq in southeastern KB yielded ~11 bears. For the North Water polynya, extrapolating a very low estimate of density (1.13 bears / 1,000 km^2, or roughly a third of the estimated density used for the sea ice near Qaanaaq) added 31 bears.

During May 2009 and 2010, Heide-Jørgensen et al. (2013) conducted an aerial survey over the North Water Polynya (NOW) between 76° N and 79° N (i.e., north to the southernmost part of the Nares Strait-Kane Basin region). Hence, they in effect covered a major part of the NOW with loose drift ice and open water which were not covered by us for safety reasons and because it was judged by us to be suboptimal or unsuitable polar bear habitat. Despite that their survey was a multi-species survey mainly targeting beluga (*Delpinapterus leucas*), narwhal

(*Monodon monoceros*), walrus (*Odobenus rosmarus*), and seals, they detected polar bears both in water and on ice. Heide-Jørgensen et al. (2013) derived an estimate of 60 polar bears (CV 0.96, range: 12-293 bears). Although their point estimate had a large uncertainty due to low sample size, it indicates that a substantial number of polar bears may occur "offshore" on loose drift ice in NOW (i.e., in habitat not covered during our 2014 survey). Given the inherent uncertainty in estimates of density for the unsampled regions, we hypothesize that negative bias arising from incomplete sampling of the Kane Basin polar bear subpopulation may have been roughly 10 – 30%.

Second, a fundamental assumption of distance sampling is perfect detection of target objects on the transect line (i.e., at distance 0; Buckland et al. 2001). Preliminary analyses with a left-truncated data set suggested that the probability of bears near the aircraft being sighted by at least one observer was >96%, so we considered this assumption to be approximately valid. Our data were too sparse to permit mark-recapture distance sampling analyses (Laake and Borchers 2004) to correct for less than perfect detection at distance 0, but our initial double-observer analyses suggest that any resultant negative bias was modest (<5%).

Population-wide aerial surveys of polar bears have been completed in the autumn over land in Foxe Basin, Western Hudson Bay, and Southern Hudson Bay (Stapleton et al. 2014, 2016, Obbard et al. 2015) and over both land and sea ice in the Barents Sea (Aars et al. 2009). Similarly, recent pilot aerial survey studies over springtime sea ice have been completed in the Baffin Bay and Southern Beaufort Sea subpopulations (Stapleton 2013). However, the aerial survey in the KB subpopulation represents the first attempt to complete a subpopulation-wide survey on springtime sea ice. Although most aerial surveys of polar bears have been conducted during the fall ice-free period, this study illustrates that, in small areas and under favorable weather conditions, aerial surveys can provide a useful inventory technique on springtime sea ice as well. Aerial surveys may be a particularly valuable tool for monitoring small, remote subpopulations that are not subject to significant harvest pressure and where acquiring detailed demographic information through mark-recapture may be too costly to justify.

We note, however, that larger geographic areas and periods of inclement weather may require more time to complete a comprehensive aerial survey, thereby necessitating more complex study designs to accommodate potential changes in bear densities and the study area itself, especially if sea-ice dynamics are changing during the survey window. Aerial surveys of larger areas like Baffin Bay will require the use of more than one fixed-winged aircraft with long endurance to ensure that the entire area (including remote offshore habitat) is covered within a relatively narrow time frame (e.g., SWG 2011, Nielson et al. 2013). Nevertheless, based on the pilot aerial survey along SE Baffin Island in spring 2010, a group of survey experts concluded that it would be feasible to assess polar bear populations with a larger range (i.e., Baffin Bay) using aerial surveys (Chapter 1).

We recognize that large offshore areas with loose drift ice in the Kane Basin subpopulation´s range could not be surveyed by us for safety reasons and because our helicopter had a relatively low range and endurance. Hence, future aerial surveys should consider using fixed-winged aircraft perhaps in combination with a helicopter (SWG 2011).

Our aerial survey estimate of abundance was based on a very small number of encounters (n = 28), resulting in some uncertainty in estimation of the detection function. Increasing the number of observations via greater sampling effort will likely improve precision; a minimum of 60 – 80 observations are recommended with distance sampling (Buckland et al. 2001), but even a marginal increase in sightings would improve estimation of the detection function. In addition, if other sites adopt on-ice surveys using similar study designs and survey platforms, joint analysis in which observations are pooled might yield more reliable estimates of the detection function, thereby improving precision of abundance estimates.

11.5. Literature Cited

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Barker, R. J. 1997. Joint modeling of live-recapture, tag-resight, and tag-recovery data. Biometrics 53:666–677.

Barker, R. J. 1999. Joint analysis of mark-recapture, resighting and ring-recovery data with age-dependence and marking effect. Bird Study 46:S82–S91.

Born, E. W., J. Teilmann, M. Acquarone, and F. Riget. 2004. Habitat use of ringed seal (*Phoca hispida*) in the North Water area (North Baffin Bay). Arctic 57:129–142.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Borchers, D. L., J. L. Laake, C. Southwell, and C. G. M. Paxton. 2006. Accommodating unmodeled heterogeneity in double-observer distance sampling surveys. Biometrics 62:372–378.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to Distance sampling: Estimating abundance of biological populations. Oxford University Press, Oxford, UK.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912–921.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Heidi-Jørgensen, M. P., L. M. Burt, R. G. Hansen, N. H. Nielsen, M. Rasmussen, S. Fossette, and H. Stern. 2013. The significance of the North Water Polynya to Arctic top predators. Ambio 42:596–610.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Innes, S., M. P. Heide-Jørgensen, J. L. Laake, K. L. Laidre, H. J. Cleator, P. Richard, and R. E. A. Stewart. 2002. Surveys of belugas and narwhals in the Canadian High Arctic in 1996. NAMMCO Scientific Publication 4:169–190.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock’s robust design. Ecology 78:563–578.

Laake, J. L., and D. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 *in* S. Buckland, D. Anderson, K. Burnham, J. Laake, D. Borchers, and L. Thomas, editors. Advanced Distance Sampling. Oxford University Press, Oxford, UK.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Marques, F. F. C., and S. Buckland. 2003. Incorporating covariates into standard line transect analyses. Biometrics 59:924–935.

Marques, T. A., M. Andersen, S. Christensen-Dalsgaard, S. Belikov, A. Boltunov, Ø. Wiig, S. T. Buckland, and J. Aars. 2006. The use of global positioning systems to record distances in a helicopter line-transect survey. Wildlife Society Bulletin 34:759–763.

Nielson, R. M., T. J. Evans, and M. B. Stahl. 2013. Investigating the potential use of aerial line transect surveys for estimating polar bear abundance in sea ice habitats: a case study for the Chukchi Sea. Marine Mammal Science 29:389–406.

Obbard, M. E. 2008. Southern Hudson Bay polar bear project 2003 – 2005: final report. Unpublished report, Wildlife Research and Development Section, Ontario Ministry of Natural Resources, Peterborough, Ontario, Canada.

Obbard, M. E., T. L. McDonald, E. J. Howe, E. V. Regehr, and E. S. Richardson. 2007. Polar bear population status in Southern Hudson Bay, Canada. Administrative report, U.S. Geological Survey, Reston, Virginia, USA.

Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., J. Laake, K.L. Laidre, E.W. Born, and S.N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Sahanatien, V., and A. E. Derocher. 2012. Monitoring sea ice habitat fragmentation for polar bear conservation. Animal Conservation 15:397–406.

Stapleton, S. 2013. Alternative methods for monitoring polar bears in the North American Arctic. PhD Dissertation, University of Minnesota, St. Paul, Minnesota, USA.

Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal

Science 32:181–201.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. B. Bishop, T. Marquest, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology 47:5–14.

Table 11.1. Results from distance sampling analyses of an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014. The most highly supported models ($\Delta \text{AIC}_c < 2$) are shown. In the column Model, the key function is followed by the covariate (Light = light conditions). p = detection probability. High- and low- density refer to stratum-specific estimates.

			Density (Bears / 1,000 km^2)			
Model	**ΔAIC_c**	**p**	**High-density**	**Low-density**	**Global**	**Abundance (SE)**
Half-Normal / Light	0.00	0.60 (0.09)	7.5 (2.0)	3.1 (3.1)	6.1 (1.7)	170 (49)
Hazard / None	0.13	0.43 (0.20)	11.1 (6.0)	3.7 (4.0)	8.7 (4.5)	243 (125)
Half-Normal / None	0.37	0.62 (0.09)	7.6 (1.9)	2.5 (2.5)	5.9 (1.6)	166 (44)

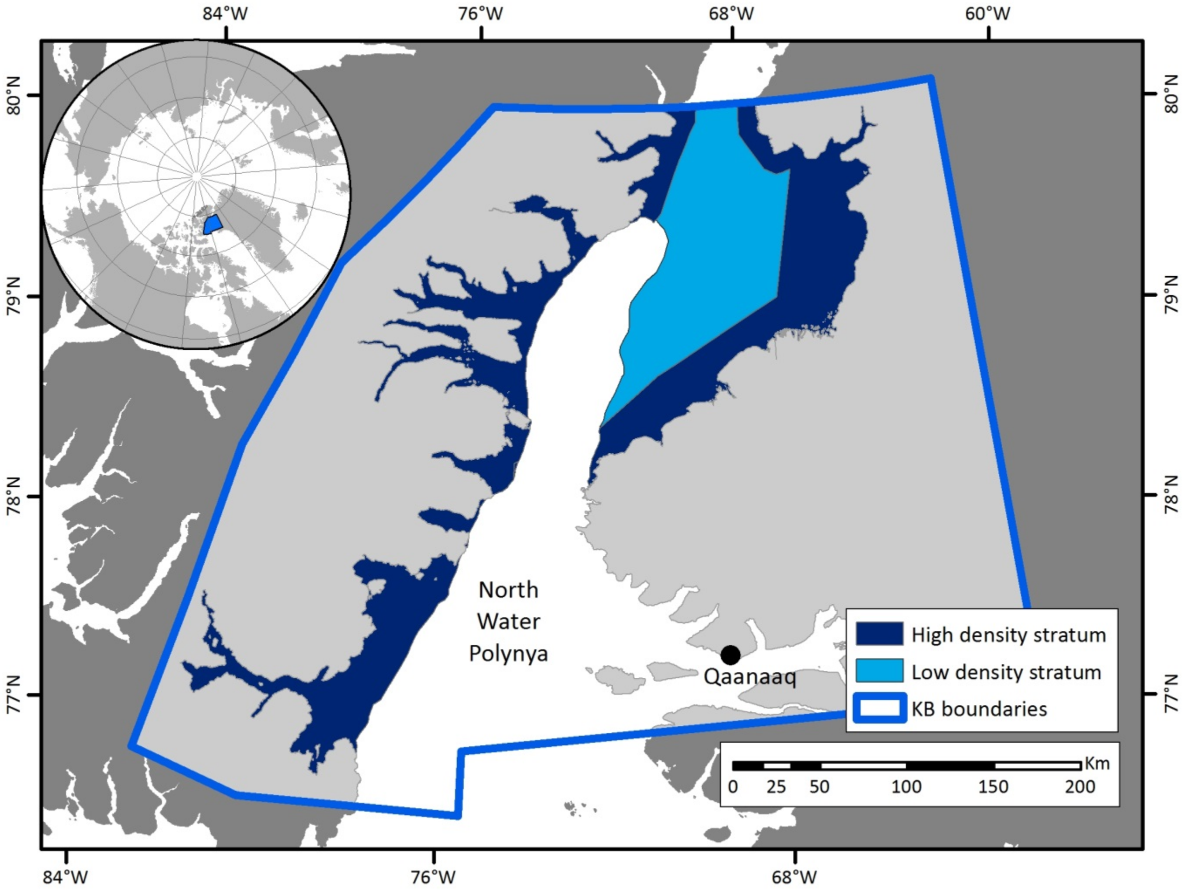


Figure 11.1. Sampling strata for genetic mark-recapture and aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

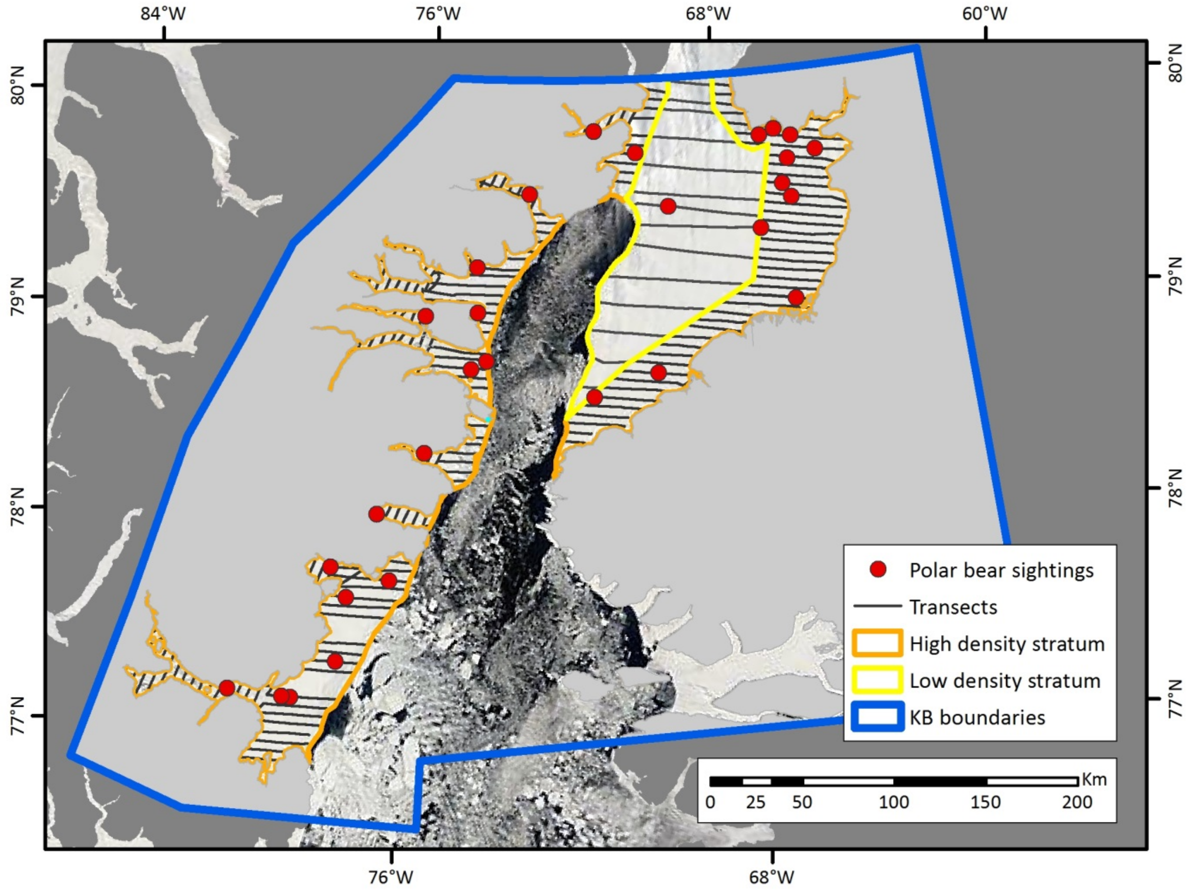


Figure 11.2. Transects surveyed and polar bear groups sighted during transect surveys of the Kane Basin subpopulation during April – May, 2014. Transects and sightings are overlaid on MODIS image (1 km resolution; available: http://modis.gsfc.nasa.gov/) collected on 5 May 2014. Sea ice in southeastern Kane Basin (i.e., to left of figure legend) was not sampled due to safety and logistical constraints presented by the North Water polynya and because we anticipated very low densities.

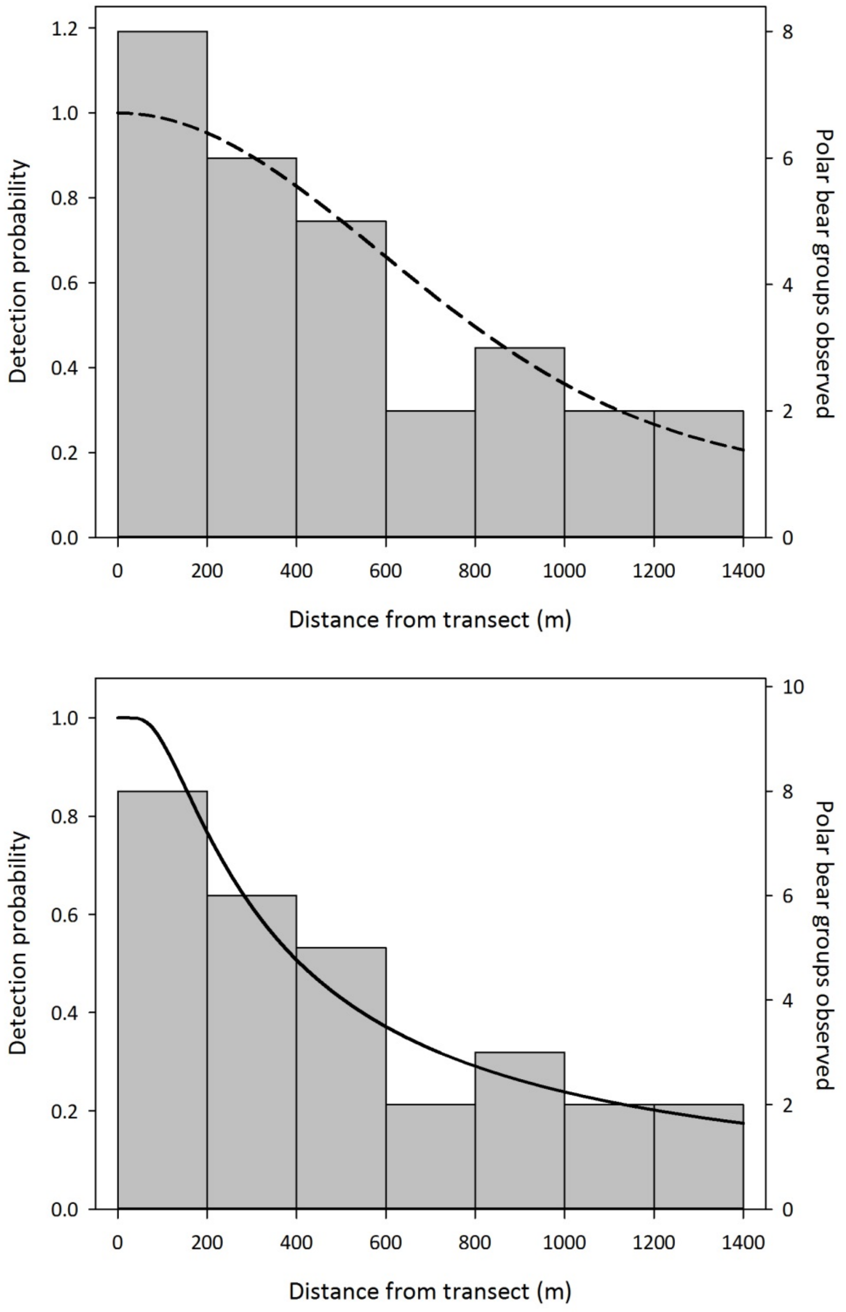


Figure 11.3. Histograms summarizing sighting distances and estimated detection functions from an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014. Top: Half-normal key function including a binary light conditions covariate. Bottom: Hazard rate key function with no adjustment terms or covariates.

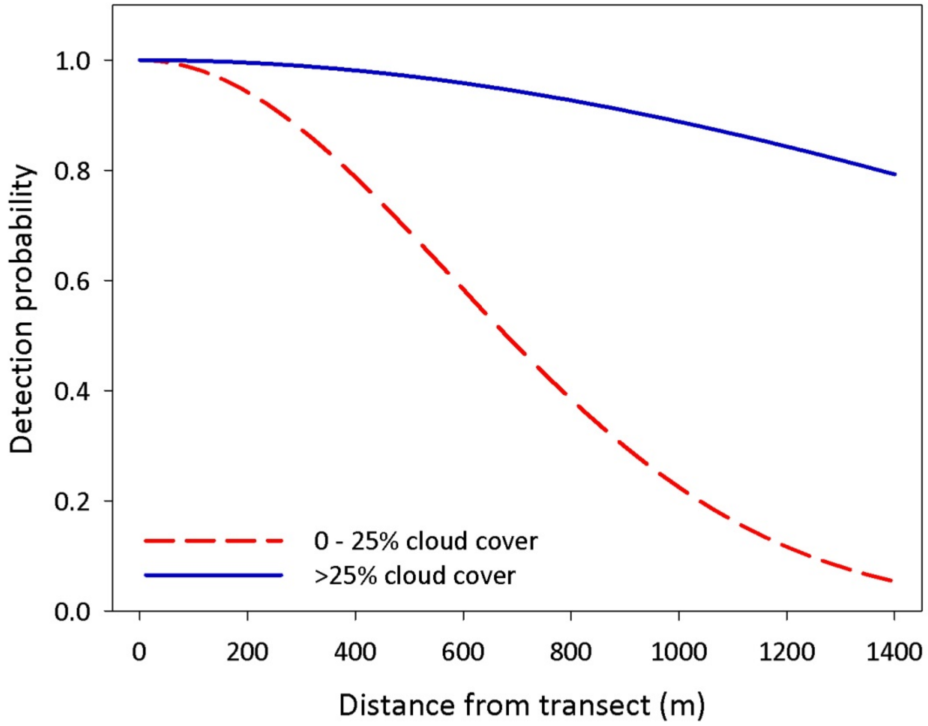


Figure 11.4. Distance sampling detection function (half-normal key function with binary light conditions covariate) estimated from data collected during an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

CHAPTER 12

REPRODUCTIVE METRICS FOR MARK-RECAPTURE SAMPLED POLAR BEARS IN KANE BASIN

KEY FINDINGS

- Data for the study were collected during two periods of MR sampling in KB. Sampling occurred on the sea ice in April and May. During the 1990s, bears were sampled by physical capture and examination using methods previously described. During the 2000s, sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals.
- Reproductive metrics for KB, including mean litters sizes for cubs-of-the-year (COY) and yearlings, and an index of recruitment (calculated as the number of yearlings per adult female in the MR sample), were comparable between the 1990s and 2010s sampling periods. Mean litter sizes in KB (for COY: 1.67 in the 1990s and 1.60 in the 2010s) were similar to those observed in other polar bear subpopulations in the archipelago ecoregion (range 1.65 - 1.71). We found no evidence of lower reproductive performance in KB, but sparse data limited our conclusions.
- During the years with the largest sample sizes (1995, 2013, and 2014), the total sample exceeded 50 bears. In these years, there was notable variation in the proportions of COY (15% - 30%). In contrast, proportions of yearlings and the recruitment index were relatively invariant among years.

12.1. Introduction

For populations of large, long-lived mammals, changes in reproductive performance can be one of the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1981, 1987). In populations approaching carrying capacity, declines in reproductive performance are likely to occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring indices or metrics of reproduction therefore may provide a useful tool for the early detection of potential population trends that may warrant more in-depth study. This is especially true for populations in which cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring reproductive metrics may provide a form of surveillance that can be used to trigger periods of more intensive study.

Reproductive metrics have been identified as an important component for monitoring polar bears across their circumpolar range (Vongraven et al. 2012). These metrics may be used to track long-term trends in the status of subpopulations, parameterize population viability models and support harvest risk assessments (Regehr et al. 2015). Of particular concern, changes in reproduction are predicted to be amongst the first subpopulation-level effects of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Molnár et al. 2011, Stirling and Derocher 2012). Indeed, declines in reproduction have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (Regehr et al. 2007, Rode et al. 2010, Rode et al. 2014).

Changes in reproductive metrics can signal significant changes in subpopulation status of polar bears. However, observations of poor reproductive performance alone do not necessarily imply a decline in subpopulation status. Studies of several polar bear subpopulations have documented declines in reproduction in association with increases in abundance that may be the result of density dependence (Derocher 2005, Peacock et al. 2013). In other cases, variation in reproductive performance within or amongst subpopulations has been attributed to geographic or annual variation in biological productivity and prey availability (i.e., fluctuating carrying capacity; Stirling and Øritsland 1995, Stirling and Lunn 1997, Stirling 2002, Rode et al. 2014). Information on reproduction therefore must be considered alongside other measures of subpopulation performance in-order to properly assess status.

The Kane Basin (KB) polar bear subpopulation is part of the archipelago ecoregion as defined by Amstrup et al. (2008), where sea ice does not melt entirely in the summer and some bears remain on the ice year-round. However, the sea ice situation in Kane Basin has changed markedly in recent decades (Chapter 9). Although currently designated as declining based on population viability modelling (PBSG 2010), a comparison of results from two mark recapture studies suggests the abundance of this small, low density subpopulation has not changed significantly over the two decades (Taylor et al. 2008a, Chapter 10). KB has experienced long term changes in sea-ice composition, and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapters 4 and 9). However, model projections predict that KB will be one of the last polar bear subpopulations to experience the negative consequences of climate change including reproductive failure (Amstrup et al. 2008, Hamilton et al. 2014). Here, we summarize reproductive metrics for KB using data collected during two periods of mark-recapture sampling from 1992 to 1997 (Taylor et al. 2008a) and 2012 to 2014 (Chapter 10). Our results provide additional context for interpreting the results of mark-recapture analyses and assessment of this subpopulation's present status.

12.2. Materials and Methods

Study Area

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008a). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit; *cf.* Figure 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq areas). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, the amount of sea ice during summer in Nares Strait-Kane Basin has dropped markedly in recent decades (e.g., Figure 9.2).

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in KB. Sampling occurred on the sea ice in April and May. During the 1990s, bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2008a). Data on the sex, age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. Sampling occurred during 25 April-6 May in 2012 and during 27 April and 10 May in 2013. With biopsy darting, we remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 –

4], and adult) from the air at a range of 3-7 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males; e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of known age-class individuals (Appendix B).

Reproductive Metrics

We calculated annual reproductive metrics that have been previously recommended (Vongraven et al. 2012) or used in studies of polar bears (e.g., Derocher and Stirling 1995, Rode et al. 2010, Peacock et al. 2013, Stapleton et al. 2014, Regehr et al. 2015). For cubs-of-the-year (COY) and yearlings, mean litter sizes were calculated from observed litter sizes. Because we did not have estimated ages for adult females sampled in 2012-2014 and because samples sizes were small in most years, we calculated a pooled mean for each year rather than age-specific values. Numbers of COY and yearling were expressed as a proportion of the total bears sampled each year. An index of recruitment was calculated as the total number yearlings divided by the total number of adult females in the sample (Derocher and Stirling 1995, Regehr et al. 2015).

Some individuals were sampled more than once in a given year. These recaptures were excluded from analyses. Captures of the same individual over multiple years were included.

We examined annual variation in reproductive metrics and compared metrics between the two epochs (1992-1997 and 2012-2014). Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

12.3. Results

During 1992-1997 and 2012-2014, we sampled 53 family groups consisting of an adult female and 1-2 dependent COY (87 COY in total). We also sampled 24 family groups consisting of an adult female and 1-3 dependent yearlings (32 yearlings in total). The mean number of family groups sampled annually was 5.9 (range: 2-15) and 2.7 (range: 0-5) for COY and yearling families respectively. Annual reproductive metrics are presented in Table 12.1.

Annual variation in observed litters sizes was not significant amongst COY (Kruskal-Wallis, $H = 4.86$, $P = 0.772$) or yearlings (Kruskal-Wallis, $H = 9.49$, $P = 0.219$). COY comprised between 9 and 38% of the bears sampled annually. Yearlings comprised between 0 and 16% of annual observations. Recruitment ranged from zero to 0.43. Sample sizes were too small to permit further analyses of annual reproductive metrics. Pooling data within epochs there were no differences in mean litter sizes between the 1990s and 2000s (Mann-Whitney U test, $U = 369$, $P = 0.700$ for COY; $U = 79.5$, $P = 0.671$ for yearlings). Proportions of COY and yearlings were also similar between epochs (Table 12.2).

During sampling in 2012-2014, 9 (12%) of 78 adult females encountered were of known age (marked during the 1990s) and ranged in age from 18 to 35 years. Five were between 18 and 20 years old, three of which were observed with litters. None of the 4 (5%) bears > 20 years old were observed with offspring. In comparison, during sampling in the 1990s, 77 adult females were sampled. Of these, 10 (13%) were 18 years old or greater (based on tooth aging) ranging in age from 18 to 21 years, and seven were accompanied by offspring. Two (2.6%) were greater than 20 years of age.

12.4. Discussion

Sample sizes were too small in most years to permit investigation of annual variation in reproductive metrics in KB. Observed variation likely was primarily the product of sample size rather than biological effect. During the years with the largest sample sizes (1995, 2013, and 2014), the total sample exceeded 50 bears, representing a sizeable portion of this small subpopulation. In these years, there was notable variation in the proportions of COY (15% - 30%; Table 12.1). In contrast, proportions of yearlings and the recruitment index were relatively invariant among years. We are unaware of any source of sampling bias that would account for this apparent variation in COY production and suggest that it may reflect pulsing or synchrony in reproduction. Anecdotally, dates of spring and fall sea-ice transition in the years prior to these 3 years were unremarkable in terms of variation; offering no explanation in terms of environmental conditions.

Our surveys were conducted in late April-early May. Den emergence date even at high latitutes (i.e., > 76° N) late March (Ferguson et al. 2000; Chapter 9, Figure 9.14). Hence, we are confident that our observations of adult females with COYs were representative.

During recent sampling, 2012-2014, we recaptured a small number of older, known age adult females that were originally marked in the 1990s. Amongst this sample, none of the individuals >20 years of age were accompanied by offspring. Although the sample size is small, this is consistent with the reproductive senescence reported in some other subpopulations Ramsay and Stirling 1988, Derocher and Stirling 1994).

Reproductive metrics for KB in both time periods were comparable, and mean litter sizes in KB were within the range of observed variation for other polar bear subpopulations in the archipelago ecoregion (Table 12.2). In summary, we found no evidence of lower reproductive performance in KB, but sparse data limited our conclusions.

12.5. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180. American Geophysical Union, Washington, DC, USA.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. Population Ecology 47:267–275.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1994. Age-specific reproductive performance of female polar bears (*Ursus maritimus*). Journal of Zoology 234:527–536.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Ferguson, S. H., M. K. Taylor, A. Rosing-Asvid, E. W. Born, and F. Messier. 2000. Relationships between denning of polar bears and conditions of sea ice. Journal of Mammalogy 81:1118–1127.

Fowler, C. W. 1981. Density dependence as related to life history strategy. Ecology 62:602–610.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS ONE 9(11): e113746. doi:10.1371/journal.pone.0113746.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., and N. J. Lunn. 1997. Environmental fluctuations in arctic marine ecosystems as reflected by variability in reproduction of polar bears and ringed seals. Pages 167–181 *in* S. J. Woodin, and M. Marquiss, editors. Ecology of arctic environments. Special Publication No. 13 of the British Ecological Society, Blackwell Science Ltd., Oxford, UK.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M'Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667-1673.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2008b. Mark-recapture and stochastic population models for polar bears of the high arctic. Arctic 61:143–152.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. Marine Mammal Science 25:778-796.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Table 12.1. Reproductive metrics derived from annual mark-recapture sampling data from Kane Basin. Captures and between season recaptures are included.

Year	**Mean Litter Size (n, SD)**		**Proportion of Total Observations (n)**		**Recruitment Index** [1] **(Yearlings/adult female)**
	COY	**Yearling**	**COY**	**Yearlings**	
1992	2.00 (3, 0.00)	0.00 (0, 0.00)	0.38 (16)	0.00 (16)	0.00
1993	2.00 (2, 0.00)	1.00 (3, 0.00)	0.17 (23)	0.13 (23)	0.43
1994	1.60 (5, 0.55)	2.00 (2, 0.00)	0.26 (31)	0.13 (31)	0.40
1995	1.70 (10, 0.48)	1.50 (2, 0.71)	0.25 (67)	0.04 (67)	0.14
1996	1.40 (5, 0.55)	1.00 (3, 0.00)	0.19 (36)	0.08 (36)	0.23
1997	1.50 (2, 0.71)	2.00 (2, 1.41)	0.09 (32)	0.13 (32)	0.40
2012	1.50 (4, 0.58)	1.40 (5, 0.55)	0.14 (44)	0.16 (44)	0.37
2013	1.57 (7, 0.53)	1.33 (3, 0.58)	0.15 (71)	0.06 (71)	0.14
2014	1.65 (14, 0.50)	1.00 (4, 0.00)	0.30 (84)	0.05 (84)	0.13

[1] – *Sensu* Regehr et al. (2015)

Table 12.2. Comparison of reproductive metrics for some polar bear subpopulations in the Arctic archipelago ecoregion (Amstrup et al. 2008). Sampling occurred during ice-free periods.

Subpopulation	Mean Litter Size		Proportion of Total Observations		Source
	COY	Yearling	COY	Yearlings	
Kane Basin (1992-97)	1.67	1.42	0.22	0.08	Taylor et al. (2008a)
Kane Basin (2012-14)	1.60	1.25	0.21	0.08	This study
GB	1.65	-	-	-	Taylor et al. (2009)
LS	1.69	-	-	-	Taylor et al. (2008b)
MC	1.68	-	-	-	Taylor et al. (2006)
NW	1.71	-	-	-	Taylor et al. (2008b)

CHAPTER 13

OBSERVATIONS ON THE BODY CONDITION AND FORAGING HABITS OF POLAR BEARS IN KANE BASIN DURING THE SPRING

KEY FINDINGS

- Body condition was better amongst KB subadults and adult females with yearlings during the 2010s relative to the 1990s, though sample sizes were limited. In contrast, condition amongst adult males, adult females with COYs, and lone adult females was similar between these time periods.
- Improved condition in the 2010s may reflect natural variation or a response to long-term changes in the sea-ice regimen in Kane Basin, largely turning into a system resembling a seasonal sea-ice ecoregion.
- Seals and polar bears were similarly distributed in Kane Basin.
- Relatively high densities of both seals and bears in northeastern Kane Basin, near the Humbolt Glacier, indicates that this region has high productivity and is important habitat for polar bears in the subpopulation.
- A high proportion of KB bears were found to have extensive hair loss and skin ulcerations on their feet. The cause of these lesions is unknown to science. Traditional knowledge suggests this phenomenon is the result of abrasive injuries sustained by walking and digging in hard, icy, coarse snow cover on the spring sea ice combined with increased rates of movement during the peak mating and feeding periods.

13.1. Introduction

For populations of large, long-lived mammals changes in body condition will be among the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1987, 1990, Zedrosser et al. 2006). In populations approaching K, declines in condition will occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring body condition may therefore provide a useful tool for the early detection of population trends that warrant more in-depth study. This is especially true for populations where cost or logistical constraints limit the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring condition may provide a form of surveillance that can be used to trigger periods of more intensive study.

The annual life-cycle of polar bears is characterized by large seasonal changes in body condition (Watts & Hansen, 1987, Ramsay et al., 1992, Ramsay and Stirling 1988, Atkinson and Ramsay 1995). Throughout most of their circumpolar range, bears are thought to gain condition during the spring and early summer when juvenile seals are abundant and relatively susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). This period of hyperphagia is followed by a scarcity of food in the late summer and fall when sea ice reaches a minimum throughout the Arctic. During this season, bears in some regions are forced onto land by the melting sea ice where access to seals and other marine mammal prey is greatly reduced (Stirling et al. 1977, Derocher and Stirling 1990, Ramsay et al. 1991). In other regions, bears remain on off-shore pack-ice but likely also have reduced access to and/or less success in catching seals (Amstrup et al. 2000, Stirling 2002, Atwood et al. 2015a, Rode et al. 2015).

Given this dynamic cycle of feeding and fasting, body condition attained during the spring and early summer is expected to exert a significant influence on the survival, reproductive performance and thus status of polar bear subpopulations (Atkinson and Ramsay 1995, Derocher and Stirling 1995, 1996, Molnár et al. 2010, Molnár et al. 2011). Tracking long-term trends in body condition has thus been identified as an important component of the monitoring scheme for polar bears across their circumpolar range (Vongraven et al. 2012, Patyk et al. 2015). In the absence of more intensive studies, simple body condition metrics may be useful indices for monitoring subpopulations and detecting responses to changing environmental conditions

(Amstrup et al. 2006, Stirling et al. 2008a, Vongraven et al. 2012). Of particular concern, changes in body condition are predicted to be amongst the first subpopulation-level impacts of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Wiig et al 2008, Stirling and Derocher 2012). Indeed, declines in condition have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016).

A variety of quantitative and qualitative body condition indices have been used on polar bears including body weight estimated from girth (e.g., Derocher and Stirling 1995, Rode et al. 2011), body mass indices standardized for length (e.g., Stirling et al 1999, Cattet et al. 2002, Obbard et al. 2016), skull width (Rode et al. 2010, 2011), percent body fat determined by isotopic dilution or bioelectrical impedance analysis (Atkinson and Ramsay 1995, 1996, McKinney et al. 2014), percent lipid content of adipose tissue biopsies (Stirling et al. 2008b, McKinney et al. 2014) and a visually assigned fatness index (Amstrup et al. 2006, Stirling et al. 2008a, b). Most of these condition indices require the handling of bears to collect measurements. However, the fatness index (FI) and potentially the lipid content of adipose tissue (Pagano et al. 2014, McKinney et al. 2014) may be obtained without handling thus making them suitable for use in subpopulations monitored by less invasive methods such as aerial survey or genetic mark-recapture.

The Kane Basin (KB) polar bear subpopulation is part of the archipelago ecoregion as defined by Amstrup et al. (2008), where sea ice does not melt entirely in the summer and some bears remain on the ice year-round. Although currently designated as declining based on population viability modelling (PBSG 2010), a comparison of results from two mark recapture studies suggests the abundance of this small, low density subpopulation has not changed significantly over the two decades (Taylor et al. 2008a, Chapter 10). KB has experienced long term changes in sea-ice composition, and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapter 4). However, model projections predict that KB will be one of the last polar bear subpopulations to experience the negative consequences of climate change including reproductive failure (Amstrup et al. 2008, Hamilton et al. 2014).

Here we summarize information on the body condition of polar bears in KB collected during two periods of mark-recapture sampling from 1992 to 1997 and 2012 to 2014. Using the Fatness Index (FI: Stirling et al. 2008b) as a qualitative metric we examine differences in condition between the two time periods. Our results provide supplementary information for interpreting the results of the genetic mark-recapture (Chapter 10) and other recent studies in KB (Chapters 2 and 9), and for understanding the present status of this subpopulation. We also report on incidental observations of prominent skin lesions that were found on some KB bears during the latter period of the study. Finally, we report incidental observations of spring time foraging by bears and the distribution of seals in KB.

13.2. Materials and Methods

Study Area

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* Figure 13.1). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, sea-ice conditions have changed markedly in the Kane Basin region in recent decades (Born et al. 2011; Figure 9.2).

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in KB. In both periods sampling occurred on the sea ice in April and May using a helicopter flying at 300-500 feet above sea-level to search for bears across the study area. During the 1990s (1992-97), bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2008a). Data on the sex, age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2010s (2012-14), sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. With biopsy darting, we remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 –

4], and adult) from the air at a range of 3-7 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males; e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of known age-class individuals (Appendix B).

Body Condition Scoring

Because most of the bears sampled during the latter period of sampling (2012-2014) were biopsy darted rather than captured and handled, our ability to compare body condition between time periods was limited to visually assigned Fatness Index (FI) scores only. The FI has been validated as a measure of condition in polar bears, being closely correlated with more quantitative condition indices (Stirling et al. 2008b, McKinney et al. 2014) and other biological factors (e.g., Henricksen et al. 2001, Amstrup et al. 2006). During both sampling periods, all encountered bears were assigned a FI score on a scale of 1-5 where 1 and 5 represent the leanest and most obese bears, respectively (Stirling et al. 2008b). During the 1990s, this score was based on physical examination of captured bears. For bears in the 2010s, FI scores for most

(67%) individuals were assigned based on examination from the air at a distance of 3-7 m above ground. The remaining portion was assigned FI scores based on physical examination after capture.

All bears were initially scored in the field according to the standard FI on a scale of 1 to 5 (Stirling et al. 2008b). This scoring system was subsequently simplified to a binary Body Condition Score (BCS) where individuals in poor (FI = 1, 2) and fair-good (FI = 3, 4 or 5) condition were assigned scores of 1 and 2 respectively. Similar modifications of the FI for polar bears have been employed in other studies to facilitate analyses (Stirling et al. 2008a) or have been recommended for use in general monitoring schemes for polar bears (Vongraven et al. 2012). In our case, this refinement was necessary due to the small samples sizes overall in our study and the low frequencies of bears at the extremes of the 5 point FI scale (i.e., very few or no bears with FI scores of 1 or 5). This simplified scoring system was also a potential means to reduce bias in assigning condition scores. The assumption made was that a simplified scale would be subject to less bias resulting from different observers and / or distance from bear at time of scoring. Experienced observers should be able to discriminate a bear in poor condition even at distances of up to 7 m. All observers in our study had extensive experience studying polar bears including capture, handling and body condition scoring.

For analyses, we pooled BCS data collected in different years into two periods (epochs); the 1990s (1992-97) and the 2010 (2012-2014). Again this was necessary due to low samples sizes. Repeated observations of the same individual (as identified by physical mark or genotype) within a given year were excluded from the analyses. Observations of the same individual in different years were included. Similar to Stirling et al. (2008a), we assumed that observations of the same individual in different years were statistically independent given the dynamic nature of body condition in polar bears (Watts and Hansen 1987, Atkinson and Ramsay 1995, 1996) and it's response to annual variation in environmental conditions.

For different sex, age and reproductive classes of polar bears we compared BCS between the two epochs using contingency tables analyses (Cross Tabs procedure in SPSS Version 24.0, IBM Corp. 2016). We also considered the potential effect of the timing of sampling on the BCS of bears. Along with the binary categorical variable (Epoch: 1990s and 2010s), Julian Day of sampling was used as an independent variable in a logistic regression (Binary Logistic procedure in SPSS Version 24.0, IBM Corp. 2016) to examined variation in BCS. Both variables were entered into the regression model. We did not specify an interaction between Julian Day and Epoch. All tests were two-tailed with alpha at 0.05.

Other Observations

During sampling from 2012-2014, we also made several other types of observations either systematically or opportunistically. For each bear encountered we noted any evidence of recent feeding. The presence of a seal kill or bears with full pendulous stomachs constituted evidence of feeding. While searching for bears in 2013, we noted the locations of live seals using a GPS. Each group comprising 1 or more individuals was recorded as a single observation. Finally, during capture and physical examination of bears in 2011 and 2012 we noted the presence or absence of some prominent skin lesions that had not previously been described in the literature.

13.3. Results

Body Condition Scores

Body Condition Scores (BCS) were assigned to 129 and 135 subadult and adult polar bears encountered during sampling in KB in the 1990s and 2010s, respectively. BCS for adult males were similar between the two epochs (Table 13.1). In contrast, across all reproductive classes, adult females in the 2010s tended to be in better condition than those in the 1990s; although this was statistically significant for adult females with yearlings only. Similarly, subadults in the 2010s were in better condition at time of encounter.

Although sampling occurred in April and early May during both epochs, timing of sampling differed (Mann-Whitney U = 1,557.00, P = 0.002). Median Julian day of sampling was slightly earlier during the 1990s (121.45) relative to the 2010s (124.68) across all sex-age classes. Within sex-age classes, these slight differences in timing of sampling were maintained (e.g., adult females with yearlings, Mann-Whitney U = 104.50, P = 0.060; sub adults, Mann-Whitney U 275.0, P = 0.02). Incorporating Julian day of sampling into a logistic regression did not explain variation in body condition amongst most classes of bears with the exception of adult females with cubs-of-the-year (COY) where bears sampled later tended to be in better condition (Appendix E). Amongst adult females with yearlings, Julian day was not a predictor of BCS and condition was better in the 2010s than in the 1990s. For subadults there was no effect of either timing of sampling or epoch on the probability of a bear being in poor versus fair-good condition.

Other Observations

Feeding – During sampling in 2012-2014, 14% of bears encountered showed evidence of recent feeding (excluding dependent offspring). Prevalence of feeding observations was highest amongst adult females with offspring and lowest in adult males and subadults (Table 13.2).

Seals – In 2013, 94 groups consisting of one or live ringed seals, *Phoca hispida*, were observed while searching for bears in KB. Notable concentrations of seals were encountered in north east Kane Basin in front of the Humbolt glacier and inside fiords along eastern Ellesmere Island (Figure 13.1).

Skin Lesions –In 2012 and 2013, 40% of the bears that were captured and physically examined were found to have unusual skin lesions. These were characterized as locally extensive alopecia (hair loss) over the feet, in most cases (75%) affecting all four feet (Figure 13.2). In addition, some of the affected individuals had multi-focal ulcerations on the plantar/palmar heel and digital foot pads and on the dorsal aspects of all 4 feet (Figure 13.3). Discharge from these lesions was purulent and sanguinous. Granulation tissue forming in some of these ulcers indicated they were chronic in nature. Even under anesthesia, some bears exhibited notable discomfort when these ulcers were gently palpated during examination, often reacting by moving the foot or lifting their head. Finally, two individuals (an adult male and a yearling) were found to have mild generalized alopecia over the dorsal neck, thorax and abdomen.

The prevalence of foot lesions was highest amongst adult males (75%) and lowest amongst cubs-of-the-years (0%) (Table 13.3). The prevalence of bleeding ulcerations on the feet, an indication of the severity and/or chronicity of the condition, was highest amongst adult males with 75% exhibiting some degree of ulceration. Also of note were two bears captured in 2012 without lesions that were recaptured in 2013 with lesions.

13.4. Discussion

Body Condition

With the limited data in this study we found evidence of differences in the spring time body condition of KB polar bears between the 1990s and 2010s. Condition amongst adult females with yearlings and subadults was better in 2010s. In contrast, condition amongst adult males, adult females with COY, and lone adult females was similar between these time periods. These findings may be attributable to several factors including bias in the data, natural variation in condition and long-term trends in environmental conditions.

Several sources of bias were possible in our study associated with use of a qualitative body condition score rather than a quantitative metric. BCS data were collected by several observers. In the 1990s, most data were collected by a single observer. In the 2010s all data were collected by a single but different observer. Differences in the assignment of condition scores by these two observers could therefore generate the apparent differences in condition between time periods. Since individual bears were not scored by more than one observer, teasing out potential observer effects is challenging. While we cannot exclude the possibility of observer bias in our study, several lines of evidence suggest that this potential bias is unlikely to account for our results. First, to reduce observer bias we employed (*post-hoc*) a simplified body condition scoring system that required observers to discriminate between bears in poor versus fair-good condition. All observers in the study were experienced polar bear biologists who had previously handled hundreds or thousands of bears in varying condition and should have been capable of accurately discriminating such bears. Second, as noted by Stirling et al. (2008a) although the FI from which our condition metric was derived is a qualitative index and thus subjective, it has been found to be "*repeatable between individual biologists when blind comparisons are done in the field over both short and long time periods*." In other studies, FI data collected by multiple observers have been found to correlate closely with quantitative indices of condition (e.g., Stirling et al. 2008b, McKinney et al. 2014). Finally, we found differences in condition between time periods amongst adult females with yearling and subadult only. If these differences in condition reflect observer bias we would expect this to be evident in all classes of bears.

Body condition scores in the 1990s were collected from bears captured and physically examined. In contrast scores in the 2000s were from bears either captured (33%) or observed from the air without handling (67%). The effect of close-up versus distance examination on the scoring of condition is unknown. McKinney et al. (2014) found that remotely assigned FI ratings did not correlate with the % lipid content of adipose tissue; another measure of condition. However, their sample sizes were small and limited to comparisons of bears of FI 3 and 4 only whereas bears in our study had FI ratings ranging from 1 to 5. Remotely scoring FI may be a less robust (precise) index of condition but is not necessarily inherently biased relative to physical examination. For many of the same reasons discussed previously concerning observer effects, we suggest that this potential source of bias is unlikely to account for our results. The use of a simplified scoring system (poor versus fair-good) in our study should have helped to reduce errors in scoring for bears observed from the air.

Another source of error in our study associated with differences in sampling between the 1990s (physical capture) and 2010s (physical capture or aerial observation) was in the classification of bears by sex and age-class based on aerial observation rather than handling. Classifying bears from the air is without doubt less accurate than physical examination. However, aerial classification is accurate in most instances (Chapter 5, Appendix B), especially amongst adult males and adult females with offspring. Misclassification was therefore unlikely to explain differences in condition of adult females with yearlings. Additionally, despite being less precise we have no evidence to suggest that aerial classification results in a bias in age and sex classifications amongst a group of bears. This source of measurement error thus seems unlikely to account for our results.

Body condition amongst bears likely improves progressively during the spring and early summer as the availability of seals increases (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). Sampling in the 1990s tended to occur earlier in the spring than in the 2010s. This difference in the timing of sampling could therefore partially account for the better condition we observed amongst some classes of bears in the 2010s. However, we note that differences in timing although significant were relatively small (i.e., 3-4 day difference in median day of sampling) so the effect on condition data may be minor. Additionally, timing of sampling was not a significant predictor of body condition amongst adult females with yearling or subadults; the two classes of bear in which differences in condition scores were detected. Finally, if timing of sampling were a significant factor we would have expected similar bias in other classes. Interestingly, we found that condition amongst adult females with COY was a function of Julian day of sampling suggesting consistent with the hypothesis that females emerging from maternal dens begin to steadily recover lost body condition in the spring.

Several ecological explanations could explain our findings. Body condition amongst polar bears fluctuates on temporal and spatial scales in response to annual variation in environmental conditions regardless of any underlying long-term trends (Kingsley 1979, Stirling 2002). Our findings may simply reflect this normal variation in condition whereby sampling in the 2010s occurred at a higher point in condition than in the 1990s. However, this would not account for the fact that improved condition was only detected amongst adult females with yearlings and subadults; since presumably all classes of bears can capitalize on improved environmental conditions. Failure to find differences in condition amongst other age classes may be due to the limited samples sizes, lack of precision in condition scoring and / or bias in our study. We note that condition tended to be better in 2010s amongst all classes of adult females; although only statistically significant for those with yearlings.

As an alternative explanation, differences in body condition between the 1990s and 2010s may reflect long-term changes in environmental conditions. At the southern extent of the polar bears' range, declining condition has been associated with reduced sea-ice cover resulting from climate change (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016). However, High Arctic polar bears such as those in KB are predicted to be amongst the last members of the species negatively impacted by climate change (Derocher et al. 2004). An initial impact of climate change in KB has been an observed reduction in the extent of multi-year ice and replacement with thinner annual ice (Hamilton et al. 2014, Chapter 4). Such changes in ice regimen are predicted to have a positive effect on polar bears via increased primary productivity and access to prey (Derocher et al. 2004). Our finding of improved body condition amongst adult females and subadults is consistent with this prediction and may be a sign of improved environmental conditions (albeit temporarily). In some subpopulations where effects of climate change have been reported, body condition has been negatively affected to a greater and / or more easily detectable degree amongst the adult female and subadult classes (Obbard et al. 2006, Rode et al. 2010). This suggests that sensitivity to deteriorating environmental conditions varies by sex, age, and reproductive status; presumably as a result of differing nutritional and energetic requirements and / or rates of food intake. Conversely, it is reasonable to assume that the age classes most readily impacted by negative changes in the environment will be the first to respond positively as conditions improved. This hypothesis is consistent with our finding that condition improved significantly amongst adult females and subadults but not adult males.

Given the limitations of our data set as discussed above, we are unable to confidently resolve between the differing explanations for our finding of improved body condition between the 1990s and 2010s. We therefore urge caution in interpreting these results. Never-the-less, it is reasonable to conclude that there has been no decline in condition in KB.

Feeding Observations

During the 2010s, the proportion of individuals showing signs of having recently fed was lowest amongst adult males. This is consistent with the hypothesis that adult males forgo foraging opportunities during the spring mating period while pursuing estrous females (Cherry et al. 2009, Stirling et al. 2015). The relatively low feeding rates we observed amongst subadults is consistent with the hypothesis that juvenile bears are less successful in hunting. We did not have data on feeding rates during the 1990s sampling period for comparison. Consequently, we are unable to assess potential changes in hunting success that could account for the improved condition observed amongst bears in the 2010s.

Seal Distribution

The distribution of seal observations made during mark-recapture sampling of polar bears was uncorrected for search effort. However, we note that the relative densities of seals along our search tracks was similar to the distribution of the polar bears we encountered (Chapters 10 and 11). Not surprisingly, where we found relatively high numbers of seals, we also found relatively high numbers of bears. The high densities of ringed seals found in north eastern KB at the front of the Humbolt Glacier is consistent with other studies that have found sea ice in front of tide-water glaciers to be prime breeding habitat for seals (Lydersen et al. 2014). This area of Kane Basin appears to have high productivity and is likely an important feeding area for polar bears. This is consistent with traditional ecological knowledge indicating that the area was a preferred hunting location for Greenlandic Inuit (Born et al. 2011). According to PBSG (1998), Taylor et al. (2001), and Born (pers. obs. 1994 and 1995) both the Greenland and Canadian portions of Kane Basin were mostly mixed annual and multi-year sea ice that appeared to be favourable habitat for polar bears in the 1990s. Ringed seals were common in both eastern and western KB (ibid.). The relatively few polar bears encountered in this region during the 1990s was thus likely the result of hunting pressure rather than habitat suitability (Taylor et al. 2001).

Skin Lesions

We documented hair loss (alopecia) and ulcerations on the feet of polar bears in Kane Basin during capture sampling in 2012 and 2013. These types of lesions have not been reported previously in the literature. Atwood et al. (2015b) documented an alopecia syndrome of unknown etiology amongst polar bears from the Southern Beaufort Sea (SB). However, those lesions were largely confined to the head, neck and shoulders rather than the feet. Additionally, they were most prevalent amongst subadult bears whereas those in KB were most common amongst adult males. Overall prevalence of lesions in KB bears (40% in 2012 and 2013) was higher than peak prevalence observed in the SB (28%). In particular, we note that seventy-five percent of adult males sampled in KB in 2012 and 2013 were affected. Lesions on adult males were also more severe than on other age classes as indicated by the presence of skin ulcerations some of which were apparently very painful. Atwood et al. (2015b) found that bears with alopecia were in poorer body condition. What impact, if any, the lesions found on KB bears may have had on condition is unknown since samples sizes were too small to support analyses.

Whether the lesions reported in SB and KB share a common etiology (cause) is unknown. Discussions with Inuit hunters from Grise Fiord (near Kane Basin) suggest this phenomenon of hair loss on the feet is well known to occur in spring time (M. Akeeagok, J. Kiguktak, D. Akeeagok pers. comm.). It is believed that increased rates of movement in spring, when bears are mating and hunting activity is high, result in abrasions to the feet. Dog teams in the Canadian Arctic are well known get similar lesions in the spring when travelling over coarse, icing snow formed by melting and refreezing as temperatures fluctuate throughout the day (D. Iqqaqrialu pers. comm.). The condition may become so severe that some dogs become lame and unable to pull sleds. Snow cover on the sea-ice in KB in 2012 and 2013 was noticeably icy and granular in composition (Figure 13.4) with a hard ice-covered crust on top. It is therefore plausible that the lesions we observed were the result of snow conditions. The finding that prevalence was highest in adult males may be due to their greater weight and the likelihood that they break the ice crust while walking, combined with potentially increased time allocated to travelling in the spring while searching for estrous females[1]. Lesions similar to those observed in 2012 and 2013 were not seen on polar bears captured in KB between 1992 and 1997 (M. K. Taylor and E. W. Born, pers. comm.). Whether this is a new phenomenon in KB brought about by changing snow conditions and progressively warmer spring temperatures or an incidental observation in the years we were sampling bears is unknown.

13.5. Literature Cited

Amstrup, S. C., G. M. Durner, I. Stirling, N. J. Lunn, and F. Messier, F. 2000. Movements and

---

[1] Versus time spent stalking prey or resting

distribution of polar bears in the Beaufort Sea. Canadian Journal of Zoology 78:948–966.

Amstrup, S. C., I. Stirling, T. S. Smith, C. Perham, and G. W. Thiemann. 2006. Recent observations of intraspecific predation and cannibalism among polar bears in the southern Beaufort Sea. Polar Biology 29:997–1002.

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.

Atkinson, S. N., and M. A. Ramsay. 1995. The effects of prolonged fasting on the body composition and reproductive success of female polar bears (*Ursus maritimus*). Functional Ecology 9:559–567.

Atkinson, S. N., R. A. Nelson, and M. A. Ramsay. 1996. Changes in body composition of fasting polar bears (*Ursus maritimus*): the effect of relative fatness on protein conservation. Physiological Zoology 69:304–316.

Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015a. Demographic composition and behavior of polar bears summering on shore in Alaska: U.S. Geological Survey Administrative Report, 26 pp.

Atwood, T., E. Peacock, K. Burek-Huntington, V. Shearn-Bochsler, B. Bodenstein, K. Beckmen, and G. Durner. 2015. Prevalence and spatio-temporal variation of an alopecia syndrome in polar bears (*Ursus maritimus*) of the southern Beaufort Sea. Journal of Wildlife Diseases 51:48-59.

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Cattet, M. R. L., N. A. Caulkett, M. E. Obbard, and G. B. Stenhouse. 2002. A body condition index for ursids. Canadian Journal of Zoology 80:1156–1161.

Cherry, S. G., A. E. Derocher, I. Stirling, and E. S. Richardson. 2009. Fasting physiology of polar bears in relation to environmental change and breeding behavior in the Beaufort Sea. Polar Biology 32:383–391.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Derocher, A. E., and I. Stirling. 1990. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395–1403.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Derocher, A. E., and I. Stirling. 1996. Aspects of survival in juvenile polar bears. Canadian Journal of Zoology 74: 1246–1252.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. Current Mammalogy 1:401–441.

Fowler, C. W. 1990. Density dependence in northern fur seals (*Callorhinus ursinus*). Marine Mammal Science 6:171–196.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Marine Mammal Science 7:123–1 35.

Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS ONE 9(11): e113746. doi:10.1371/journal.pone.0113746.

Henriksen, E. O., Ø. Wiig, J. U. Skaare, G. W. Gabrielsen, and A. E. Derocher. 2001. Monitoring PCBs in polar bears: lessons learned from Svalbard. Journal of Environmental Monitoring 3:493–498.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lydersen, C., P. Assmy, S. Falk-Petersen, J. Kohler, K. M. Kovacs, M. Reigstad, H. Steen, H. Strøm, A. Sundfjord, Ø. Varpe, W. Walczowski, J. M. Weslawski, and M. Zajaczkowski. 2014. The importance of tidewater glaciers for marine mammals and seabirds in

Svalbard, Norway. Journal of Marine Systems 129:452– 471.

McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

Molnár, P. K., A. E. Derocher, G. W. Thiemann, and M. A. Lewis. 2010. Predicting survival, reproduction and abundance of polar bears under climate change. Biological Conservation 143:1612–1622.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., M. R. L. Cattet, T. Moody, L. R. Walton, D. Potter, J. Inglis, and C. Chenier. 2006. Temporal trends in the body condition of Southern Hudson Bay polar bears. Climate Change Research Information Note 3, Ontario Ministry of Natural Resources, Applied Research and Development Branch, Sault Ste. Marie, Ontario, Canada.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Patyk, K. A., C. Duncan, P. Nol, C. Sonne, K. Laidre, M. Obbard, Ø. Wiig, J. Aars, E. Regehr, L. L. Gustafson, and T. Atwood. 2015. Establishing a definition of polar bear (*Ursus maritimus*) health: A guide to research and management activities. Science of the Total Environment 514: 371-378.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.

Ramsay, M. A., R. A. Nelson, and I. Stirling. 1991. Seasonal changes in the ratio of serum urea to creatinine in feeding and fasting polar bears. Canadian Journal of Zoology 69:298–302.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Rode, K. D., R. R. Wilson, E. V. Regehr, M. St Martin, D. C. Douglas, and J. Olson. 2015. Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.

Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.

Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.

Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.

Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

Stirling, I., E. Richardson, G. W. Thiemann, and A. E. Derocher. 2008a. Unusual predation attempts of polar bears on ringed seals in the southern Beaufort Sea: Possible significance of changing spring ice conditions. Arctic 61:14–22.

Stirling, I., G. W. Thiemann, and E. Richardson. 2008b. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568-574.

Stirling, I., C. Spencer, and D. Andriashek. 2016. Behavior and activity budgets of wild breeding polar bears (*Ursus maritimus*). Marine Mammal Science 32:13–37.

Taylor, M. K., S. Akeeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690–709.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Watts, P. D., and S. E. Hansen. 1987. Cyclic starvation as a reproductive strategy in the polar bear. Symposia of the Zoological Society of London 57:305–318.

Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. Science Progress 91:151–173.

Zedrosser, A., B. Dahle, and J. E. Swenson. 2006. Population density and food conditions determine adult female body size in brown bears. Journal of Mammalogy 87:510–518.

Table 13.1. Body condition scores assigned to polar bears in Kane Basin during sampling in the 1990s (1992-1997) and 2010s (2012-2014). Within each epoch and sex-age class, the proportion of individuals in the two BCS categories is presented in parenthesis.

		Body Condition Score		
Sex-Age Class	**Epoch**	**Poor**	**Fair-Good**	**Test Result[1]**
Adult Male	1990s	5 (0.10)	44 (0.90)	*P* = 1.000
	2010s	5 (0.11)	40 (0.89)	
Adult Female (Lone)	1990s	9 (0.28)	23 (0.72)	χ^2 = 2.100
	2010s	5 (0.14)	31 (0.86)	*P* = 0.147
Adult Female (w/COY[2])	1990s	11 (0.42)	15 (0.58)	χ^2 = 1.922
	2010s	6 (0.24)	19 (0.76)	*P* = 0.166
Adult Female (w/yearling)	1990s	8 (0.67)	4 (0.33)	*P* = 0.015
	2010s	2 (0.15)	11 (0.85)	
Subadults	1990s	4 (0.29)	10 (0.71)	*P* = 0.037
	2010s	0 (0.00)	16 (1.00)	

[1] Unless otherwise indicated all tests results report values of *P* for Fisher's exact test (two-tailed)

[2] Cub-of-the-year (COY)

Table 13.2. Observations of recent feeding amongst polar bears encountered in Kane Basin, 2012-2014. Evidence of feeding includes presence of seal kills and bears encountered with full, pendulous stomachs.

Sex-age class	**Proportion Feeding (n)**
Adult Male	0.07 (46)
Adult Female (Lone)	0.17 (36)
Adult Female (with offspring)	0.26 (43)
Subadult	0.07 (16)

Table 13.3. Frequency of hair-loss (alopecia) and skin ulcerations on the feet of polar bears captured in Kane Basin, Nunavut, 2012 and 2013.

Sex-Age Class	No. of Individuals Examined	Proportion with Alopecia	Proportion with Ulcerations
Adult Male	12	0.75	0.75
Adult Female (Lone)	9	0.22	0.00
Adult Female (with COY)	7	0.29	0.00
Adult Female (with Yearling)	6	0.50	0.17
Adult Female (with 2-year-old)	1	0.00	0.00
Subadult	4	0.50	0.25
2-year-old	1	0.00	0.00
Yearling	8	0.63	0.25
Cub-of-the-year (COY)	11	0.00	0.00
Total	**59**	**0.40**	**0.22**

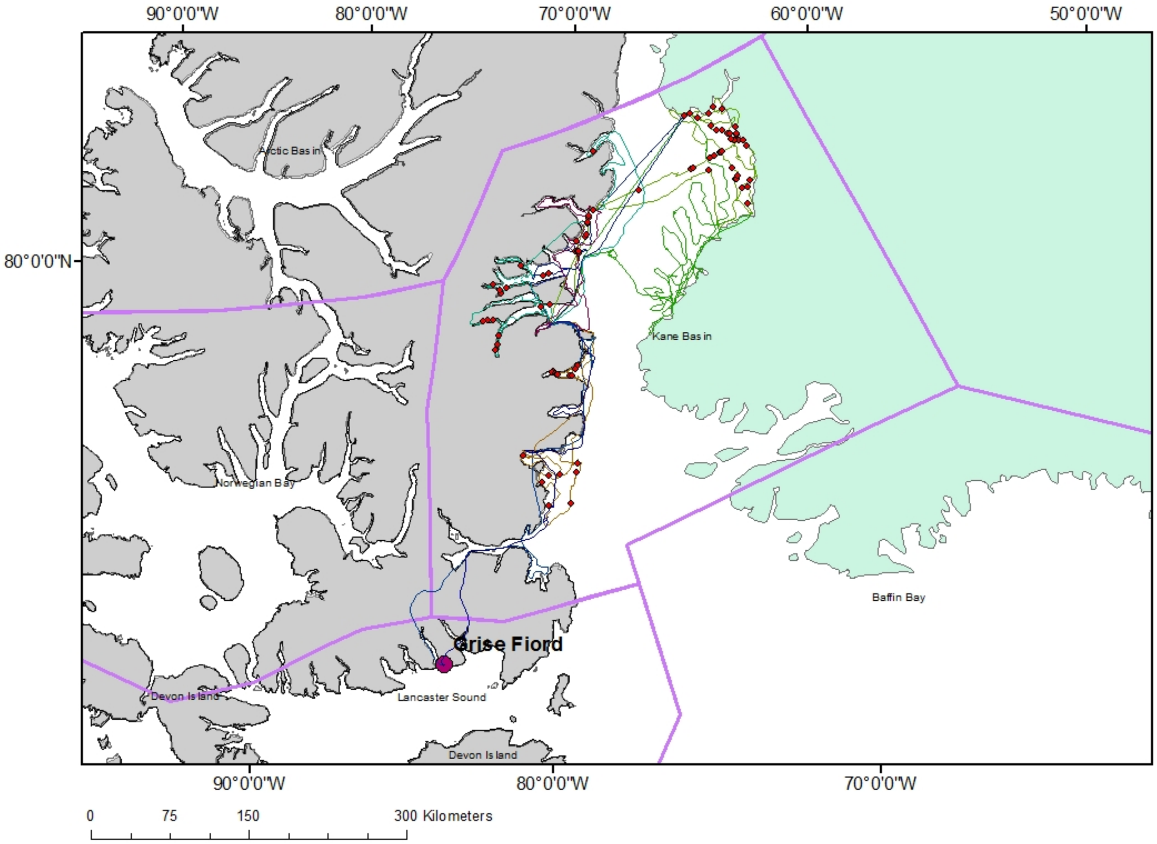


Figure 13.1. Distribution of seal observations during polar bear mark-recapture sampling in Kane Basin, 2013. Flight tracks are shown.

Figure 13.2. Examples of alopecia over the feet of polar bears handled in Kane Basin, April/May 2012 and 2013.

Figure 13.3. Examples of ulcerative lesions on the feet of polar bears handled in Kane Basin, April/May 2012 and 2013.

Figure 13.4. An example of the granular snow found in many parts of Kane basin during polar bear sampling in April/May 2012 and 2013.

CHAPTER 14

GENERAL DISCUSSION AND ADVICE TO THE JOINT COMMISSION

14.1. Conclusions

In 2010, the Canada-Greenland Joint Commission on Polar Bear (JC) tasked the Scientific Working Group with using the best available scientific information to (1) propose Total Allowable Harvest (TAH) levels for the Baffin Bay and Kane Basin subpopulations and provide the JC with a written report of its recommendations, and (2) provide science advice to the JC for monitoring the effects of habitat changes on polar bears. Given the age of the mark-recapture data on which abundance and vital rates for BB and KB were estimated combined with large-scale environmental changes in Baffin Bay during the last decades and suspected large-scale environmental changes in Kane Basin in recent time, the SWG strongly recommended that new estimates of subpopulation abundance, population delineation, and vital rates be given high priority.

Based on the decisions of the JC that physical MR should not be used in this study a multi-year programs began in 2011 (BB) and 2012 (KB) to re-assess the size of both subpopulations using genetic mark-recapture (MR) techniques that involved biopsy sampling from both live and harvested polar bears. In addition, satellite transmitters were deployed on male and female polar bears in NW Greenland during 2009-2013 to study polar bear movement and habitat choice and to gather data for planning and interpretation of the genetic MR study in Baffin Bay. With the same purpose satellite transmitters were deployed on male and female polar bears in both the Canadian (Nunavut) and Greenland parts of Kane Basin in 2012 and

2013. In addition, a helicopter-based aerial survey was flown in Kane Basin in spring 2014 concomitantly with the MR operation to evaluate the feasibility of estimating KB abundance with an aerial survey flown over springtime sea ice.

A total of 2,690 genetic samples were collected from live and harvested polar bears in BB and KB, 125 satellite transmitters were deployed, and 4,160 linear km of transects flown during the aerial survey in KB as part of the overall research program. In addition, many hours were subsequently spent processing samples; analyzing genetic, aerial survey, and sea-ice data; in discussion interpreting the results; and, writing this report.

The study has resulted in new estimates of abundance of polar bears in the Baffin Bay and Kane Basin subpopulations and provided significant and comprehensive information about polar bear ecology and sea-ice dynamics in Baffin Bay and Kane Basin. The results of this program are described in detail in Chapters 2 through 13. They are also summarized concisely in the Executive Summary document.

14.2. Lessons from Genetic Mark-Recapture

The SWG recommended that physical mark-recapture be used for assessing the size of the Baffin Bay and Kane Basin subpopulations. However, as described in Chapter 1 the Canada-Greenland Joint Commission decided to use genetic mark-recapture for assessment.

The Baffin Bay and Kane Basin studies represent the first time that genetic mark-recapture has been implemented at population-wide scales for estimating polar bear abundance and demographic rates. As such, these studies provide valuable information about the utility of genetic mark-recapture as a monitoring tool for assessing polar bear subpopulations ranging over large areas like Baffin Bay with dynamic sea-ice conditions, and how the technique may be most successfully implemented in the future.

Sampling via biopsy darting was highly successful in yielding tissue samples suitable for genotyping in both Baffin Bay and Kane Basin (Chapters 5 and 10) essential if genetic mark-recapture is to be implemented in other subpopulations.

Moreover, genetic analyses identified that reporting of marked bears in the harvest is incomplete; specifically, decreases in reporting are related to greater marker age, suggesting that the loss of physical markers (loss of ear tags and fading of lip tattoos) over time makes it difficult for hunters to correctly identify marked bears. As such, we encourage the use of genetics for identifying marked bears in the harvest in the future; for those subpopulations not inventoried via genetic mark-recapture, this will require genotyping archived samples as well (see also [3] below).

Collecting samples for genotyping via biopsy darting is generally fast, efficient, and less invasive than physical mark-recapture, since bears are not immobilized (Chapter 5). Because biopsying for genetic MR estimation is less time consuming than handling individual bears during physical MR operations genetic MR has the potential of resulting in more “marks” and “recaptures” which theoretically improve precision of estimates of abundance in MR. In Baffin Bay success in sampling a large number of biopsies in the huge coastal distribution areas within a relatively short time was obtained by using three helicopters (2 along eastern Baffin Island and 1 in NW Greenland) during the same time in fall. Using three helicopters during fall biopsying for several years inevitably increased the costs of the surveys.

In physical mark-recapture information on a recapture is obtained from direct physical inspection of the presence of numbered ear tags and/or a number in the lips. Data on marking and recapture is therefore readily at hand for analyses. In contrast, during genetic mark-recapture the information on whether a biopsied bear was a "recapture" or not is not obtained until after genetic analyses are conducted in a qualified laboratory. This adds a delay in the analyses. In our case the commitment of the contracted laboratory, one that is recognized world-wide as an expert and of high quality, to undertake other genetic analyses caused a serious delay in processing the polar bear samples. This resulted in a delay in data analyses for the abundance estimation.

However, there were trade-offs with the genetic mark-recapture method that resulted in lack of information that would have been available with a physical capture protocol. Physical mark-recapture provides a wider range of information including estimates of rates of birth and death, detailed age-structure and body condition information, and a suite of physical samples can further inform individual and population status. Because bears had to be identified from the air using genetic MR, there is uncertainty in the age structure especially for younger bears. Overall, physical mark-recapture permits a more comprehensive assessment of population status, as previously recommended by SWG (2010).

Furthermore, even if physical mark-recapture is not used for assessment, some physical capture is necessary for studies at this scale to provide data on movements of bears and habitat use. In this study, 139 bears were captured in West Greenland, and of these 38 adult females with collars informed both the mark-recapture assessment itself (e.g., temporary emigration analyses, range sizes) as well as provided key information on changes in sea-ice habitat use that set the mark-recapture results into context.

Research in the Kane Basin subpopulation during 2014 illustrated that aerial survey and genetic mark-recapture methods can be implemented simultaneously (Chapters 10 and 11) to generate more comprehensive demographic information and to ensure efficient and representative allocation of sampling effort. In the present study we illustrate that combining the two methods simultaneously is feasible for subpopulations which are surveyed on sea ice during spring and which have a relatively small geographical distribution like the Kane Basin subpopulation. However, using a helicopter like we did with a relatively short range for both biopsying and aerial surveys at the same time may be suboptimal. Using only a single helicopter limits the ability to expand survey effort to offshore polar bear habitat with loose drift ice and open water as demonstrated in our study where a substantial portion of the KB subpopulation's range could not be surveyed (also for safety reasons) leading to an abundance estimate which was negatively biased to an unknown extent.

With genetic mark-recapture, the ability to leverage historical data to improve estimates of survival is limited by the availability of archived samples for genotyping. For both Baffin Bay and Kane Basin, tissues samples suitable for genotyping were available for most – but not all – bears initially marked during research in the 1990s. This lack of tissue samples for a small proportion of the sampled population necessitated identifying those individuals which could still be alive (based on harvest records and age at time of first capture; Chapters 5 and 10) and fixing their recapture probabilities during the 2010s sampling to zero. Although this solution complicated analyses, it enabled us to incorporate historical capture data. Moreover, the large sampling interval between the past and present BB and KB studies did not favor recoveries of old "marks" that also could have assisted in improving some vital rates. If successive population studies are envisaged within a 10-15 year time frame, then a single-year biopsy sampling session should be implemented 5-7 years after the study was completed in order to increase or maintain marks in the population that can be recovered through either harvest or the subsequent new study.

During physical mark-recapture individual age is obtained from every single bear that is handled and marked. During immobilization a vestigial tooth is extracted and individual age is obtained from reading growth-layers in the cementum. This allow for implementing age-structured models for estimating abundance and vital parameters. The fact that individual ages are not obtained from bears that are biopsied during genetic mark-recapture represents a limitation on *post hoc* analyses as indicated in Chapter 5. Hence, we implemented only coarse age structures for demographic analyses, pooling individuals ≥2 years and ≥3 years for the Baffin Bay and Kane Basin studies, respectively (Chapters 5 and 10). Thus, survival could not be estimated for 2 year olds and subadults separately from adults. Ancillary data suggest that experienced biologists may be able to discriminate among finer age classes (i.e., 2 – 4 year old subadults versus adults ≥5 years) with a high degree of accuracy for adults and a lesser degree for subadults (Appendix B), particularly since the sex of individuals is confirmed via genetics (Appendix B). As such, genetic mark-recapture may have the potential to yield estimates of survival and reproductive output that are relevant to management and comparable to previous research however more work needs to be done (e.g., Taylor et al. 2005, 2008a, Peacock et al. 2013). However, it must be kept in mind that field-estimation of individual age during genetic mark-recapture relies heavily upon the individual researcher’s experience in assigning polar bears to more specific age classes.

We completed a detailed review of historical records to assess previous research in Baffin Bay and Kane Basin. These reviews proved critical to our interpretation of results, as apparent changes in the sampling frames between epochs (i.e., incomplete spatial sampling during the

1990s, relative to the 2010s) limited our ability to assess trends in abundance. However, we were not able to locate original and detailed data files from the physical mark-recapture study and telemetry study conducted in Baffin Bay and Kane Basin in the 1990s. This finding underscores the need to archive data securely and for complete and consistent sampling of the study areas. We recommend that any polar bear study archives data securely at institutes that were central in the collaborative studies to allow future studies to thoroughly assess historical inventories to ensure proper interpretation of results. For the same reason, the Canada/Nunavut/Greenland institutes that conducted the recent study in Baffin Bay and Kane Basin have signed a contract where collaborating institutes have access to and must securely archive all data stemming from the joint study.

We defined and stratified the study areas for recent inventories based on recent telemetry data (and historical capture records). For this purpose, data obtained from satellite transmitters deployed in West and Northwest Greenland in 2009 and 2010 for other purposes proved to be an important tool. This process improved our allocation of effort and ensured that results reflected the entire subpopulations as best as possible. Satellite telemetry data (both historical from the 1990s and recent from 2009-2014) were also used to interpret *post-hoc* to what extent the biopsy samplings in the 1990s and 2000s were for the entire subpopulations in BB and KB. Hence, information on movement and area occupancy obtained from satellite telemetry is an essential tool in mark-recapture studies of polar bears.

Finally, the telemetry data identified that 18% of the adult females collared in West Greenland remained in Melville Bay for a least one summer season and in some cases for over a year. There would be value in future work investigating what fraction of the BB subpopulation uses Melville Bay year-round. This could make use of the existing genetic marks from this study, with the addition of one or two sampling seasons in fall in Melville Bay only. Densities appeared low based on observations during data collection in 2012 and 2013 however there would be value in quantifying this using more detailed sampling of the area.

In conclusion, the present study has shown that a genetic mark-recapture assessment of a polar bear subpopulation at a large scale is possible. However, such a study would likely involve an absolute minimum of three years of sampling and, subsequently, additional time for laboratory analysis of samples, analysis of data, and report writing. Thus, these timelines must be considered in advance if timely information on abundance for management is paramount. In this study, our reporting of abundance of polar bears in Baffin Bay and Kane Basin is given 5+ years after initiation of the study in 2011.

14.3. Recommendations on Sustainable Harvest

Historically, the management of polar bears assumed that their sea-ice habitat was relatively stable and, once subpopulation size was known, conservation of polar bears could be achieved through harvest management (SWG 2011). The sustainable harvest of a subpopulation would, therefore, be largely dependent on an estimation of abundance, demographic rates (e.g., birth and death vital rates), and the magnitude, and sex and age composition of the harvest. Since the 1980s, management and conservation of polar bears, particularly in Canada, has been informed by predictive modelling, which has provided guidance to managers on the setting of sustainable harvest levels that have been based on a better understanding of the risk associated with different harvest scenarios (Taylor et al. 1987b, 2002, 2005). The primary predictive model used, RISKMAN (Taylor et al. 2001), focused on harvest management and did not account for any future change in survival or birth rates during the period over which the simulations were run. Furthermore it did not include changes in carrying capacity (K).

Based on demographic rates derived from pooled subpopulation data collected across the Canadian High Arctic in the 1970s and 1980s, modeling indicated that under optimal conditions the sustainable yield of adult females is typically <1.6% of the total population (Taylor et al. 1987a). Hence, level of total sustainable catch (C) was inferred from this relationship (PBSG 1998): $C = N * 0.015/\text{sex ratio in harvest}$; where N is subpopulation size, 0.015 is the sustainable rate of harvest of adult (independent) females from the population, and the denominator is the ratio of adult females in the harvest. Hence, the ratio of adult female bears in the harvest was important for the sustainable total yield. Using this relationship and a general 2:1 sex ratio of males to females in the harvest, the historical standard for the harvest rate of polar bear subpopulations has been 4.5%, which was based on a 2:1 sex ratio of males to females and, initially, on generalized demographic rates derived from pooled subpopulation data collected across the Canadian High Arctic in the 1970s and 1980s (Taylor et al. 1987a, b). More recently, as more studies were undertaken and additional analytical methods developed, it became clear that demographic rates were subpopulation specific (e.g., Amstrup et al. 2001, Taylor et al. 2002, 2005, PBSG 2006, Taylor et al. 2006, Regehr et al. 2007, PBSG 2010). In their recent development of a matrix-based demographic model for polar bears, Regehr et al. (2015) noted that harvest management based on this standard rate and the 2:1 male-biased sex ratio is reasonable under many biological and management conditions; although in some cases, lower or higher rates may be more appropriate.

The ultimate threat to polar bears throughout their range is the reduction in sea-ice habitat expanse, duration, and quality as a consequence of climate change (Derocher et al. 2004, Laidre et al. 2008, 2015a, Wiig et al. 2015). At their 2009 Meeting in Tromsø, Norway, the Polar Bear Range States agreed that the impacts of climate change constitute the most important threat to polar bear conservation (Polar Bear Range States 2015). Hence, the effective and sound management of polar bears can no longer rely solely on estimates of abundance but must also incorporate impacts of a changing environment (e.g., loss of ice and reduction in carrying capacity). In addition, as other threats (i.e., pollution, resource exploration and development, tourism) become better understood, management of polar bear subpopulations will need to be modified, particularly if reproduction or survival rates are negatively affected (Vongraven and Peacock 2011, Vongraven et al. 2012). Thus, understanding the risks associated with a range of harvest management options is important for polar bear conservation.

One of the stated purposes and objectives of the 2009 Canada-Nunavut-Greenland Memorandum of Understanding is to manage polar bears within the KB and BB management units in order to ensure their conservation and sustainable management into the future (Anon. 2009). The SWG was subsequently tasked with proposing Total Allowable Harvest (TAH) levels for both the Baffin Bay and Kane Basin subpopulations (Chapter 1) but was not provided with any specific guidance on management goals.

To facilitate the ability of the SWG to provide the Joint Commission (JC) with useful recommendations on TAH, the SWG requested that the JC provide:

1) A statement of management objectives for each subpopulation,
2) Information on the expected frequency and intensity of future monitoring, and
3) A statement of risk tolerance with respect to the effects of human caused removals.

As a consequence of not receiving the necessary information from the JC on which to base recommendations on TAH despite repeated requests, the SWG is currently unable to provide these recommendations. Specific objectives for the management and conservation of BB and KB polar bears are critical as these will largely influence recommendations on TAH, which in turn will ensure that decision makers have all the necessary information available to them. Risk tolerance and management objectives are not decisions to be made by scientific staff but rather by those with management authority for the resource and, ideally, made after consideration of not only subsistence harvest but also other sources of human-caused removals (e.g., human-bear interactions) and after consultation with all stakeholders.

Once the JC provides the requested information, the SWG strongly recommends that subsequent advice on TAH be based on population simulations using models that have the ability to incorporate changing sea-ice conditions (e.g., reductions in carrying capacity) and demographic data quality as part of the overall process to assess risk under different harvest management scenarios. For example, Regehr et al. (2015) developed a state-dependent management framework that linked the demographic model to simulated population assessments, which can be used to estimate the maximum sustainable rate of human-caused removals. It can also be used to calculate a recommended sustainable harvest rate, which Regehr et al. (2015) note is generally lower than the maximum sustainable rate because it is dependent on management objectives, the precision and frequency of population data, and risk tolerance.

14.4. Recommendations on Monitoring Habitat Change on Polar Bears

There have been numerous reports regarding the effects of climate change and in particular the loss of sea ice, on polar bears (Derocher et al. 1994, Laidre et al. 2008, Wiig et al. 2008). Changes in distribution of polar bears in several populations including in BB have been summarized by Stirling and Parkinson (2006). Furthermore, information on local observations of the effects of climate change on polar bears in BB made by hunters and elders have been presented by Dowsley (2005), Dowsley and Taylor (2006), and Dowsley and Wenzel (2008) for eastern Baffin Island and by Born et al. (2008a, b, 2011) based on interviews with experienced polar bear hunters NW Greenland. These sources indicate that polar bears in BB (and likely also KB) currently are affected by large-scale environmental changes.

Monitoring habitat change will improve our understanding of the relationship between BB and KB polar bears and the environment. It provides insights into how factors such as sea ice and prey abundance and availability affect polar bear distribution and vital rates. The results of the habitat assessment work conducted in Baffin Bay and Kane Basin since 2009, largely based on satellite telemetry data from collared adult female polar bears, have provided a wealth of new information on habitat and habitat change, which directly inform the status of the BB and KB subpopulations (Chapters 2, 4, and 9). The movement information has also assisted in interpreting the BB and KB abundance and demographic data, which were associated with some biases that could be addressed through the use of the telemetry data (Chapter 3).

Vongraven et al (2012) developed a circumpolar monitoring framework for polar bears, which was focused on the sustained long-term monitoring necessary to understand ongoing effects of climate warming and other population-level stressors in order to inform management and policy responses to changing worldwide polar bear status and trends. The current scientific understanding of polar bears and their reliance on sea-ice habitats is the result of long-term monitoring that has been conducted in only a few subpopulations. There is variability in the response of each subpopulation to loss of sea ice, as manifested in this study through the differences in responses in KB and BB despite roughly the same rates of sea-ice loss (Chapters 4 and 9). Therefore it is critical that scientific studies be conducted within the subpopulation(s) of interest to gain an in depth understanding of the complex ecological ramifications of climate change in that area (e.g., Rode et al. 2014), rather than making assumptions based on studies in other areas that may have different responses.

The BB and KB satellite telemetry studies allowed for comparison and quantification of range use across decades, seasons and months, changes in overlap of the population range over time, shifts in median latitude of bears across seasons, changes in immigration and emigration across subpopulation boundaries, changes in movement rates, shifts in sea ice and terrestrial habitat use and habitat selection, and changes in maternity denning timing, and changes in maternity denning areas and habitat. Though not included in the report, time series from captures in BB and KB provide information on causal links between factors that determine health, nutritional ecology and population-level processes. Analyses have been initiated (using samples from recent BB captures) and are expected to provide new information on feeding and nutritional ecology.

The SWG concludes that future physical capture and satellite tagging studies in BB and KB (collaring and tracking adult female bears for periods of years) will be critical to extending the current time series and informing managers of the impacts of sea-ice habitat loss. Given the large physical changes documented in this study, and clear responses of polar bears in both areas, continuation of the time series of satellite telemetry data will improve our understanding of the impacts of future biotic and abiotic changes on the two subpopulations. The satellite telemetry studies should be conducted on intervals of 10 years or less, with samples sizes roughly equivalent to those collected in the 1991-1997 and 2009-2013 (approximately 40 adult females tracked over a period of several years). Lower sample sizes will make assessments more difficult due to individual variability and lack of model convergence (as seen for KB).

By use of satellite telemetry, the present study documented that a group of adult female polar bears occur year round at glacier fronts in Melville Bay in NW Greenland (Chapter 2). To protect important polar bear habitat the Melville Bay Nature Reserve was established in 1980. All access and hunting within the central zone of the nature reserve is prohibited (Appendix D). The (re-)establishment of a local group of polar bears with affinity to Melville Bay appears to be relatively recent and is likely an effect of the protection places on this important polar bear habitat. Satellite telemetry in the 2000s has also shown females to be denning in the Melville Bay Nature Reserve. When the areas were surveyed during spring 1992 and 1993 very little signs of polar bear activity were observed in the nature reserve and of the 1990s satellite telemetry indicated that adult females did not use the Melville Bay (Chapter 2).

Hence, recent data indicate that the Melville Bay Nature Reserve represents an example of how important polar bear habitat can be protected resulting in polar bears reestablishing groups "locally" in prime habitat. Hence, the nature reserve exemplifies a means of protecting polar bears in the future. However, the broader effect of the nature reserve on polar bears should be followed. This can be done by regularly conducting a genetic mark-recapture estimation of trends in numbers in the local group of bears (baseline genetic data now exist from the present study) and by tracking individuals by use of satellite telemetry at 5-10 years intervals.

Continued assessment of changes in habitat in BB and KB via satellite-based observation of sea ice (passive microwave data, MODIS, or Radar images) provide useful context for changes in the physical environment. Arctic sea ice is the most critical habitat for the survival of polar bear subpopulations as distribution and timing of ice relative to critical phases of polar bear life history have been linked to subpopulation status and trend (Stirling et al. 1999, Hunter et al. 2010, Regehr et al. 2010). The SWG recommends continued monitoring of sea-ice habitat change through these studies (described herein as well as Stern and Laidre 2016). Furthermore, contrasting changes in BB and KB with other polar bear subpopulations provides an important baseline for comparison.

Continued development and refinement of habitat models will be necessary in the future to identify habitat selection changes and better predict critical habitat in BB and KB. Standardized methods of developing habitat models (resource selection functions, RSFs) for polar bears have been developed for several subpopulations (Mauritzen et al. 2003, Ferguson et al. 2000, Durner et al. 2004, 2006, Wilson et al. 2014, Laidre et al. 2015b, Chapters 4 and 9) and within a large part of polar bear range (Durner et al. 2009). RSFs are developed from satellite radio telemetry data of adult female bears and readily available sea-ice data in geographic information system (GIS) format. Habitat models are powerful tools for predicting the occurrence of terrestrial den habitat (Howlin et al. 2002, Richardson et al. 2005). Knowledge of the distribution of maternal den habitat has significant management potential to protect polar bears in dens. Trends in sea-ice den habitat may be estimated by monitoring sea-ice conditions as changes in the composition of sea ice has been linked to changes in den distribution (Fischbach et al. 2007).

Habitat availability and change have been linked to polar bear demography and/or condition in some subpopulations (Regehr et al. 2007, Bromaghin et al. 2015, Lunn et al. 2016). Quantitative links between habitat and demographic parameters are complex and need to be refined and specific to the subpopulation of interest. Continued habitat monitoring will improve the understanding between the links to demography and productivity for both BB and KB.

Stable isotope (Bentzen et al. 2007), fatty acid analysis (Iverson et al. 2006), and lipid content in adipose tissues (McKinney et al. 2014) conducted from blood, fat and hair collected during captures can provide information on the polar bear prey base and help to identify shifts in food webs and body condition in BB and KB. This information can be used in concert with information on movements and habitat use from the telemetry to better inform how bears use the ecosystem. While some of this information can be collected from harvest sampling, physical capture of polar bears provides opportunities that are not available from harvest samples (e.g., collection of samples from the same individuals over time).

Continued monitoring of TEK and LEK in BB and KB will also be critical for providing information on how changes in sea ice are impacting the polar bear hunt (e.g., hunting practices), the overall harvest, and the condition of bears harvested. Local perspectives on changes both to the physical environment and the population are important inputs to managers. Repeated studies with a robust interview study design and data collection process (e.g., Born et al. 2011) are needed.

Finally continued subsistence harvest monitoring is needed in both Canada and Greenland, providing critical information on numbers, sex ratios and ages of bears taken in both areas. In this study, this information provided important content into changes in harvest patterns and composition of the harvest (Chapter 8). Genetic validation of the sex of individual bears (as reported by the hunters) showed that the gender was incorrectly reported in a significant number of cases. Improvement in gender reporting is needed; inaccuracies in gender reporting were greatest in Greenland.

14.5. Literature Cited

Amstrup, S. C., T. L. McDonald, and I. Stirling. 2001. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. Journal of Agricultural, Biological, and Environmental Statistics 6:221–234.

Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*, 9 pp.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2008a. Isbjørne i Nordvestgrønland En interviewundersøgelse om fangst og klima, 2008. Pinngortitaleriffik – Grønlands Naturinstitut Teknisk Rapport Nr. 70, 112 pp. (In Danish and Greenlandic with English abstract).

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2008b. Nannut Kalaallit Nunaata avannaata kitaani – Piniarneq silallu pissusaa pillugit apersuilluni misissuineq, 2006. – Pinngortitaleriffik – Nalunaarusiaq Teknikkimut tunngasoq No. 70, 116 pp. (In Greenlandic with English summary).

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Bentzen, T. W., E. H. Follmann, S. C. Amstrup, G. S. York, M. J. Wooller, and T. M. O’Hara. 2007. Variation in winter diet of southern Beaufort Sea polar bears inferred from stable isotope analysis. Canadian Journal of Zoology 85:596–608.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634–651.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2007. Advanced Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, Oxford, UK.

Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. Population Ecology 47:267–275.

Derocher, A. E., and I. Stirling. 1994. Age-specific reproductive performance of female polar bears (*Ursus maritimus*). Journal of Zoology 234:527–536.

Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1, Government of Nunavut, 43 pp.

Dowsley, M., and M. K. Taylor. 2006. Community consultations with Qikiqtarjuaq, Clyde River and Pond Inlet on management concerns for the Baffin Bay (BB) polar bear population: A summary of Inuit knowledge and community consultations. Nunavut Wildlife Research Group Final Report No. 2, Government of Nunavut, 83 pp.

Dowsley, M., and G. W. Wenzel. 2008. ‘‘The time of the most polar bears’’: A co-management conflict in Nunavut. Arctic 61:177–189.

Durner, G. M., S. C. Amstrup, R. Nielson, and T. McDonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea. Pages 107–120 *in* S. Huzurbazar, editor. Resource selection methods and applications. Omnipress, Madison, Wisconsin, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. Ecology 81:761–772.

Fischbach, A. S., S. C. Amstrup, and D. C. Douglas. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. Polar Biology 30:1395–1405.

Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. Marine Mammal Science 7:123–1 35.

Hansen, K. W. 2010. Polar bear management in Greenland, 2005-2008. Pages 133-134 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Howlin, S., M. Stishov, L. L. McDonald, and S. L. Schliebe. 2002. Den site selection by polar bears on Wrangel Island. Pages 127–138 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Hunter, C. M, H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. Ecology 91:2883–2897.

Iverson, S. J., I. Stirling, and S. L. C. Lang. 2006. Spatial and temporal variation in the diets of polar bears across the Canadian Arctic: indicators of changes in prey populations and environment. Pages 98–117 *in* I. L. Boyd, S. Wanless, and C. J. Camphuysen, editors. Top Predators in Marine Ecosystems: Their Role in Monitoring and Management. Cambridge University Press, Cambridge, UK.

Laidre, K. L., E. W. Born, P. Heagerty, Ø. Wiig, H. Stern, R. Dietz, J. Aars, and M. Andersen. 2015b. Shifts in female polar bear (*Ursus maritimus*) habitat use in East Greenland. Polar Biology 38:879–893.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015a. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications 18:S97–S125.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Mauritzen, M., S. E. Belikov, A. N. Boltunov, A. E. Derocher, E. Hansen, R. A. Ims, Ø. Wiig, and N. Yoccoz. 2003. Functional responses in polar bear habitat selection. Oikos 100:112–124.

McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. Nature Communications 2:1–8.

Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.

Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117–127.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Richardson, E., I. Stirling, and D. S. Hik. 2005. Polar bear (*Ursus maritimus*) maternity denning habitat in western Hudson Bay: a bottom-up approach to resource selection functions. Canadian Journal of Zoology 83:860–870.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.

Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.

Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere Discussion Papers, in review.

Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. Canadian Journal of Fisheries and Aquatic Sciences 52:2594–2612.

Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261–275.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

Taylor, M., F. Bunnell, D. DeMaster, R. Schweinsburg, and J. Smith. 1987b. ANURSUS: a population analysis system for polar bears (*Ursus maritimus*). International Conference on Bear Research and Management 7:117-125.

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987a. Modelling the sustainable harvest of female polar bears. Journal of Wildlife Management 51:811–820.

Taylor, M. K., J. Laake, H. D. Cluff, M. Ramsay, and F. Messier. 2002. Managing the risk from hunting for the Viscount Melville Sound polar bear population. Ursus 13:185–202.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667-1673.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2008b. Mark-recapture and stochastic population models for polar bears of the high arctic. Arctic 61:143–152.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. Marine Mammal Science 25:778-796.

Taylor, M., M. Obbard, B. Pond, M. Kuc, and D. Abraham. 2001. RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations. Government of Nunavut, Iqaluit, Nunavut, Canada.

Vongraven, D., and E. Peacock. 2011. Development of a pan-Arctic monitoring plan for polar bears: background paper. Circumpolar Biodiversity Monitoring Programme, CAFF Monitoring Series Report No. 1, CAFF International Secretariat, Akureyri, Iceland.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. Science Progress 91:151–173.

Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA.

Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. Ecosphere 5(10):136. doi:10.1890/ES14-00193.1

APPENDICES

APPENDIX A. List of Scientific Working Group members and external experts involved in the re-assessment of the Baffin Bay and Kane Basin polar bear subpopulations.

Scientific Working Group Members	External Experts
Stephen N. Atkinson Government of Nunavut Canada	Todd Arnold University of Minnesota USA
Erik W. Born Greenland Institute of Natural Resources Greenland	Markus Dyck Government of Nunavut Canada
Kristin L. Laidre University of Washington and Greenland Institute of Natural Resources Greenland	Eric V. Regehr US Fish and Wildlife Service USA
Nicholas J. Lunn Environment and Climate Change Canada Canada	Seth Stapleton University of Minnesota USA
Øystein Wiig University of Oslo Norway	Harry Stern University of Washington USA

APPENDIX B. The accuracy of estimating polar bear age-class and sex from helicopter-based, aerial observations: Implications for the use of non-invasive survey methods in monitoring subpopulations.

INTRODUCTION

Although some subpopulations of polar bears (*Ursus maritimus*) have been studied and monitored for more than 4 decades (e.g., Stirling et al. 1977, Lunn et al. 2016), there is considerable variation in the depth and scope of knowledge across their circumpolar range (Vongraven et al. 2012). Large gaps in basic information exist. Growing concerns about the impacts of climate change, increasing industrial development, harvest and contaminants in the Arctic have prompted range state governments, researchers, environmental organizations and local communities to call for an enhanced and coordinated circumpolar monitoring effort (Vongraven et al. 2012, Range States 2015).

Polar bears typically have been monitored by means of physical mark-recapture. This method has yielded detailed demographic data, allowing researchers to assess the status of subpopulations and closely examine the impacts of climate change and other threats. Hence, physical mark recapture provides tissue samples and samples of e.g., blood and milk which have been used in a wealth of studies of health and pollution in polar bear populations (e.g., Regehr et al. 2007, Hunter et al. 2010, Sonne 2010, Peacock et al. 2013, Bromaghin et al. 2015, Lunn et al. 2016). However, gaps in knowledge and the demand for rapid dissemination of up-to-date information have generated interest in the use of alternative methods for monitoring polar bears. Aerial surveys have proven to be an effective and expedient way of assessing abundance in polar bear subpopulations even in subpopulations with a large range (Aars et al. 2009, Stapleton et al.

2014, 2016). Genetic mark-recapture (Herreman and Peacock 2013, Pagano et al. 2014, Chapters 5 and 10) and remote sensing (Stapleton et al. 2014) may also offer viable alternatives in situations where logistical and financial limitations preclude the use of physical mark-recapture or where concerns about the impacts of handling bears outweigh the benefits (Vongraven et al. 2012).

Aerial surveys are widely used for monitoring wildlife populations. Unlike physical mark-recapture studies where individuals are captured, marked and released, aerial surveys do not provide detailed demographic data such as estimates of birth rates and survival that can be used to project population growth. Nevertheless, aerial surveys, which only rely on one season are an effective means of yielding a snapshot of estimates of abundance/status. Inferences about population trends can be derived from repeated aerial surveys. In addition to estimating abundance, aerial surveys can also provide information on spatial distribution, sex and age composition, body condition and reproductive performance that can be used to facilitate population status assessment (e.g., Stapleton et al. 2014). In contrast to genetic and physical mark-recapture estimation, which relies on several years of sampling, aerial surveys can provide an estimate of abundance from only one season of study.

Genetic mark-recapture has been increasingly used for wildlife population monitoring (Palsbøll et al. 1997, Boersen et al. 2003, Boulanger et al. 2004, Kendall et al. 2009). Like aerial surveys, genetic mark-recapture does not require the capture and physical handling of individuals. Tissue samples are collected for genotyping and identification of individuals by methods such as biopsy darting or hair-snagging. With protocols such as biopsy darting, individuals can be observed from a distance, facilitating collection of additional information on sex, age class and body condition. However, the reliability of this ancillary information depends on the ability to correctly classify individuals by sex and age class.

Using data collected during two recent genetic mark recapture studies of polar bears in the Baffin Bay (BB) and Kane Basin (KB) subpopulations, we assessed the accuracy of classifying polar bears into sex and age classes from the air without physical handling. From a sample of bears of known sex and age class, we examined variation in accuracy of classification dependent on the method of survey (aerial vs genetic mark-recapture) and amongst sex-age classes. We discuss the implications of the results for expanding the utility of aerial survey and genetic mark-recapture as less invasive methods for monitoring species status.

METHODS

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km^2 in Baffin Bay, covering portions of Baffin Island and all Bylot Island (66.2° N to 73.8° N) in Nunavut/ Canada and parts of West and Northwest Greenland (66.0° N to 77.0° N; Taylor et al. 2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south. Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A small number of bears remain on land in northwestern Greenland throughout the ice-retreat period.

The KB polar bear subpopulation covers roughly 150,000 km^2 and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit; *cf.* Figure 13.1). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, in recent decades, sea ice conditions in KB have changed markedly (Chapter 9)

Genetic Mark-Recapture Study

Genetic mark-recapture studies were conducted in BB and KB between 2011-2013 and 2012-2014 respectively (Chapters 5 and 10 in this report). Sampling of bears in BB occurred from late August to mid- October along the east coast of Baffin Island and around Bylot Island, Canada. During this period, bears were on land in a variety of habitats ranging from flat coastal plains and beaches to steep rocky slopes and glaciers. Bears were observed against different backgrounds including sand, rocks, low lying vegetation, snow and water. Sampling in KB occurred in April and May while bears were on the sea ice. Bears were observed on a range of sea-ice types including flat, shorefast ice, consolidated pack-ice and unconsolidated pack-ice.

Using a helicopter (Bell 206 LongRanger), we searched for and biopsy darted polar bears using methods described previously (Chapters 5 and 10). Upon encounter, the sex and age class (cub-of-the-year [COY], yearling, subadult [ages 2 – 4], and adult) of each bear was estimated from the air at a range of 3 – 7 m above ground. The individual identity and sex of each bear was later confirmed via genetic analysis (Chapter 5 and 11). In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males; e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under the tail of females). Fields notes, and in some cases photographs, also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. All observations were made by experienced polar bear biologists who had previously participated in physical mark-recapture studies.

At the time of encounter, observers had no prior knowledge of the sex or age class of bears. However, amongst the individuals encountered, a proportion were of known sex and age class based on one or more lines of evidence (Table B1), including a number of bears whose genotypes matched those of bears handled during physical mark-recapture studies in BB (Taylor et al. 2005), KB (Taylor et al. 2008, Chapter 10 in this report) and Davis Strait (Peacock et al. 2013). We assessed the accuracy of remote classification using this sample of ‘known’ bears and examined two scenarios. The first scenario simulated the outcome of an aerial survey in which the sex of bears cannot be confirmed via genotyping. Sex and age classification under this scenario therefore relies solely on field observation. The second scenario simulated a genetic mark-recapture, whereby inaccuracies in field sexing of bears can be corrected following genotyping and field notes made at the time of observation can be used to make *post-hoc* adjustments to age class once genetic sex is known[1].

We restricted our analyses to bears that were sub adults or adults at time of encounter due to small sample sizes for COYs and yearlings and because these dependent offspring can be easily identified when part of a family group (> 96% and 91% accurate for COY and yearlings respectively; GN unpublished data from Davis Strait). Our analysis was a simple comparison of the estimated and known frequencies of bears in each sex and age class under these two scenarios.

RESULTS

During genetic mark-recapture studies in BB and KB, 2011-2014, there were 309 encounters with individuals classified from the air as adult females based on the presence of accompanying offspring (either COY or yearlings), including 29 instances in which the sex and age class of the adult female was also known from capture and physical examination (n = 2) and tooth aging (n = 27) on a prior or future occasion. Twenty-five adult females were subject to aerial classification during fall when they were accompanied by COY (n = 12) or yearlings (n = 4) and 4 were classified during spring (3 with COY, 1 with yearlings). All adult females with dependent offspring were correctly classified from the air.

In addition to adult females with accompanying offspring, we recorded 128 unencumbered bears of known sex and age class (Table B2). Aerial classification of these bears without subsequent genotyping and reclassification based on genetic sex (i.e., the aerial survey scenario) resulted in an overall accuracy of 73%. For lone adults, 95% males and 74% of females were

---

[1] An example of a field note used for post-hoc adjustment of age class would be: "*If not female is a sub adult male*".

correctly classified. Inaccuracies were greatest amongst subadult bears. Although 70% of subadults were correctly classified, only 23% were correctly classified as sub adults of a particular sex.

Aerial classification combined with subsequent reclassification based on genetic sex, field notes and photographs (i.e., the genetic mark-recapture scenario) resulted in an overall accuracy of 91% amongst the 128 known age, independent bears. Again, accuracy varied by sex and age class (Figure B1); accuracy was highest for adult males (97%) and lowest for sub adult females (79%) (Table B3).

DISCUSSION

One of the criteria used to classify adult females was the presence of dependent offspring (COY or yearling) at the time of aerial observation or during a prior encounter. Use of this criterion was based on the assumption that accuracy in identifying females with offspring of this age was at, or near 100%. Although the sample size was relatively small, our results support this assumption and the validity of this age classification criterion. All of the adult females with offspring whose age could also be confirmed by tooth aging or physical examination were correctly classified from the air. However, we did not have any adult females accompanied by 2-year-olds in our sample of known-aged bears so we were unable to test the accuracy of classifying adult females based on the presence of 2-year-old offspring nor were we able to test accuracy in classifying 2-year-olds themselves.

Our results suggest that experienced observers can estimate the sex and age class of bears from the air with high accuracy for most sex and age classes, particularly when aerial observations are combined with genetic sexing, field notes and photographs. These findings are consistent with results from a similar study using a larger dataset of known-age bears (n = 445 based on tooth aging) from Davis Strait in which aerial observers correctly classified 97%, 88%, 80% and 80% of adult males, adult females, subadult males and subadult females respectively (GN unpublished data).

Not surprisingly, the greatest inaccuracies occur in classifying subadult bears. While the ability to classify an individual as a subadult is reasonably good from the air, the ability to determine the sex of subadult bears based on aerial observation alone is poor. Another area of potential inaccuracy that we were unable to test was the identification of independent yearlings. In some polar bear subpopulations, a proportion of yearlings are found alone during the summer or fall; presumably having been weaned (Derocher and Stirling 1995, 2012, Stirling et al. 1999). In Baffin Bay during the 1990s, approximately 6% of yearlings encountered during mark-recapture sampling were independent (GN unpublished data). The extent to which weaning of yearlings is occurring in BB at present is unknown. Of 16 bears recaptured as yearlings during genetic mark-recapture sampling from 2011 to 2013, all were still with their mother, but this small sample size limits inferences. However, in Western Hudson Bay, the proportion of yearlings that are independent during the fall has declined dramatically from > 81% prior to 1980 to almost zero at present (Stirling and Derocher 2012). This decline in early weaning of offspring has occurred in association with changing sea-ice conditions leading to the suggestion that early weaning is associated with favorable environmental conditions. Given trends in sea ice in BB (Laidre et al. 2015), a reduction in the proportions of independent yearlings may also be occurring. The number of independent yearlings encountered during our genetic mark-recapture was likely negligible.

Based on the accuracy of classification documented in this study, we conclude that the sex and age class data derived from aerial surveys or genetic mark-recapture studies can provide reliable data to support monitoring and assessment of population status. From aerial observations, adult males, adult females, COY, and yearlings can be identified with high accuracy. Subadult age classes also can be accurately determined, but classification by sex is poor. Combined with genetic sexing, field notes and photographs, accuracy is improved for all classes of bears. While we cannot derive specific ages for bears from aerial observations, such data do support monitoring the basic age structure of subpopulations. In addition, with genetic mark-recapture, there is an opportunity to model survival of specific age classes, albeit with a degree of uncertainty. For example, remote classification of sex and age classes does not permit modeling senescent age classes, nor can we model the transition from subadult to adult age classes with certainty. Finally, given the accuracy in identifying adult females and their dependent COY and yearling offspring, reproductive indices such as litters size and recruitment (yearlings per adult female) can be reliably obtained.

In our genetic mark-recapture studies in Baffin Bay and Kane Basin approximately one-third and two-thirds of sampling, respectively, occurred for bears of known age class based on our criteria (Table B1). The higher proportion of known age bears in Kane Basin was the result of physical captures completed to deploy satellite telemetry instruments. For the two thirds and one third of bears of ‘unknown’ age that were age classed based on aerial observations, genetic sex, field notes and photographs we can be confident in the accuracy of those classifications. For mark-recapture analyses, we adopted a coarser age class structure than was assessed in the present study due to concerns about the ability to remotely classify bears (Chapters 5 and 10). Our findings suggest that the accuracy of remote classification is sufficient to justify the use of finer scale age-class structures in the future.

LITERATURE CITED

Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. Marine Mammal Science 25:35–52.

Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. Wildlife Society Bulletin 31:197–207.

Boulanger, J., S. Himmer, and C. Swan. 2004. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British Columbia. Canadian Journal of Zoology 82:1267–1277.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634–651.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Derocher, A. E., and I. Sitrling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657–1665.

Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. Ursus 24:148–163.

Hunter, C. M, H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. Ecology 91:2883–2897.

Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. Journal of Wildlife Management 73:3–17.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016.

Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320.

Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. Marine Mammal Science 30:169–183.

Palsbøll, P. J., J. Allen, M. Bérube, P.J. Clapham, T. P. Feddersen, P. S. Hammond, R. R. Hudson, H. Jørgensen, S. Katona, A. H. Larsen, F. Larsen, J. Lien, D. K. Mattila, J. Sigurjónsson, R. Sears, T. Smith, R. Sponer, P. Stevick, and N. Øien. 1997. Genetic tagging of humpback whales. Nature 388:767–769.

Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Sonne, C. 2010. Health effects from long-range transported contaminants in Arctic top predators: An integrated review based on studies of polar bears and relevant model species. Environment International 36:461–491.

Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014a. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38–47.

Stapleton, S., M. A. LaRue, N. Lecomte, S. N. Atkinson, D. Garshelis, C. Porter, and T. Atwood. 2014b. Polar bears from space: Assessing satellite imagery as a tool to monitor *Ursus maritimus*. PLoS ONE doi:10.1371/journal.pone.0101513.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. Global Change Biology 18:2694–2706.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294–306.

Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

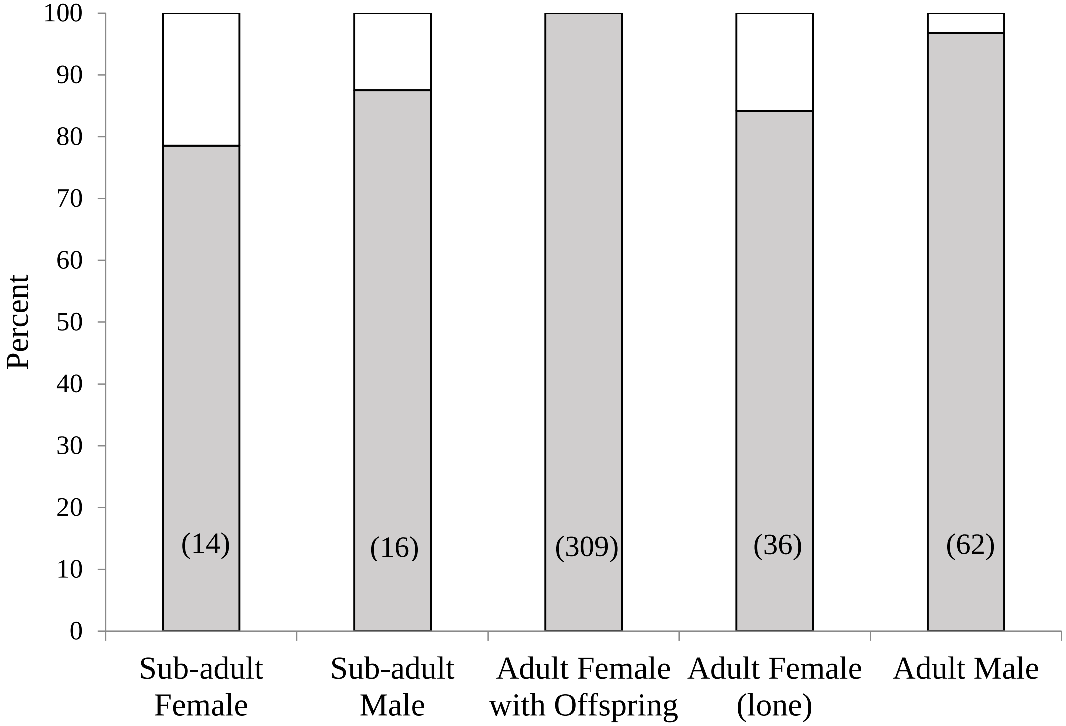


Figure B1. Accuracy of estimating the age class of bears during genetic mark-recapture studies in Baffin Bay and Kane Basin (2011-2014). Data are for bears of known age-class and sex. Sex is based on genotyping and age class is based on one or more of the criteria listed in Table A1. Data are presented as percentages correctly (grey) and incorrectly (white) classified with sample sizes in parentheses.

Table B1. Lines of evidence (criteria) used to determine the ‘known’ sex and age class of polar bears.

Sex-Age Class	Evidence Used to Determine Class
Subadults	• Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior or future occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be subadult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].
Adult Females with dependent offspring (COY or yearling)	• Accompanied by dependent offspring at time of encounter or during previous encounter and / or capture. • Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior or future occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].
Lone Adult Females	• Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior occasion or future occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to an adult female previously accompanied by dependent offspring. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].
Adult Males	• Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by tooth[1]. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth[1].

[1] Age estimated by counting annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998).

Table B2. Comparison of sex and age classes as estimated from the air versus known sex and age for bears observed in Baffin Bay and Kane Basin, Canada, 2011-2014. Estimated sex and age based on aerial observation only (aerial survey scenario). Data are frequencies. Percentage of bears correctly classified are presented in parentheses.

			Known				
			Sub adult (<5yrs)		**Adult**		
			Female	**Male**	**Female (with offspring)**	**Female (Lone)**	**Male**
Estimated	**Sub adult (<5yrs)**	**Female**	3 (21.4)				
		Male	3	4 (25.0)			1
		Unknown/Not recorded	5	6			
	Sub adult or adult	**Female**				1	
		Male				2	1
		Unknown/Not recorded				1	
	Adult	**Female (with offspring)**			309 (100)		
		Female (Lone)	3	4		28 (73.7)	1
		Male		2		4	59 (95.2)
		Total Individuals	14	16	309	36	62

Table B3. Comparison of sex and age classes as estimated from the air versus known sex and age for bears observed in Baffin Bay and Kane Basin, Canada, 2011-2014. Estimated sex and age based on aerial observation and post-hoc correction for genetic sex (genetic mark-recapture scenario). Data are frequencies. Percentage of bears correctly classified are presented in parentheses.

			Known				
			Sub adult (<5yrs)		**Adult**		
			Female	Male	Female (with offspring)	Female (Lone)	Male
Estimated	**Sub adult (<5yrs)**	Female	11 (78.6)				
		Male		14 (87.5)			1
		Unknown/Not recorded					
	Sub adult or adult	Female				4	
		Male					1
		Unknown/Not recorded					
	Adult	Female (with offspring)			309 (100)		
		Female (Lone)	3			32 (88.9)	
		Male		2			60 (96.8)
		Total Individuals	14	16	309	36	62

APPENDIX C. Regression results for an annual body condition metric for polar bears in Baffin Bay (BB). The metric, proportion of bears in good condition, was derived from observed frequencies of Fatness Index (FI) scores ranging from 1 to 5 (Stirling et al. 2008). Bears of FI 4 or 5 were in good condition. Spring ice transition was the decimal day (1-365) when ice cover over the continental shelf of BB reached 50%. Data for all years were collected within a standardized sampling area (see chapter 3). Regressions were performed in the Curve Estimation procedure of SPSS (Version 24.0).

Sex-Age Class	**Dependent Variable**	F_6	r^2	***P***	***Curve Type***
Adult Male	Spring Ice Transition	59.89	0.97	**0.001**	Quadratic (2nd order)
Adult Male	Year	18.90	0.79	**0.007**	Linear
Adult Female (alone)	Spring Ice Transition	0.25	0.05	0.635	Linear
Adult Female (alone)	Year	2.09	0.29	0.208	Linear
Adult Female (with offspring)	Spring Ice Transition	51.77	0.91	**0.001**	Exponential
Adult Female (with offspring)	Year	13.24	0.73	**0.015**	Exponential
Yearling	Spring Ice Transition	9.75	0.83	**0.029**	Quadratic (2nd order)
Yearling	Year	4.71	0.49	0.082	Linear

LITERATURE CITED

Stirling, I., G. W. Thiemann, and E. Richardson. 2008. Quantitative support for a subjective fatness index for immobilized polar bears. Journal of Wildlife Management 72:568–574.

APPENDIX D. Description of the polar bear harvest management and monitoring systems in Canada and Greenland.

HARVEST MANAGEMENT AND MONITORING IN CANADA

Within Canada, harvesting of polar bears has been managed on a quota system since 1967 (Lee and Taylor 1994). This system has undergone several revisions over time both in terms of the size of quotas and the methods of management. Since 1996, the quota system for BB, KB and other neighboring subpopulations managed by the Government of the Northwest Territories (and now the Government of Nunavut) has had several notable features. The term quota has been replaced by the term Total Allowable Harvest (TAH) to reflect language in the Nunavut Land Claims Agreement (NLCA). However, TAH and quota are often used interchangeably in reports. The TAH is managed on a flexible system that allows the annual level of harvest to vary somewhat from year-to-year to some degree. Each community hunting from a subpopulation is allocated a portion of the TAH. Exceedance of that allocation in a given year is compensated for by a reduction in the community's allowable harvest the following year. Exceedances occur when the total number of bears harvested is greater than the available limit or when too many females are harvested. This sex selective harvest management system is based on a target sex ratio of 2 or more males for every female harvested which allows a higher sustainable harvest than a 1:1 sex ratio (Taylor et al. 2008b). All human-caused mortalities are counted against the available TAH. Adult females with dependent offspring (cubs-of-the-year, yearling or two-year-olds) and those in or constructing dens are protected from hunting. The hunting season runs from July 1st to June 30th the following year. Most hunting is for subsistence purposes by Inuit. However, a portion of each community's TAH may be allocated to guided sport hunting by non-Inuit, at the discretion of Inuit (Tyrell 2009; Wenzel 2008, 2011). Methods of hunting are regulated under the *Nunavut Wildlife Act*. With the exception of sport hunters who must travel by dog-team only, hunting may be facilitated by use of various forms of transportation including snow machine, All Terrain Vehicle or boat and with firearms of authorized calibers.

In Canada (Nunavut), the reporting of all harvested bears is mandatory. Hunters are required to provide evidence of the sex in the form of a baculum from harvested males. In the absence of a baculum, sex may be determined by DNA (Prior to 2005, proof of sex could also be established by sworn affidavit). When proof of sex is unavailable a harvested bear is counted as a female for quota management purposes. In addition to proof of sex, hunters are required to submit a set of standard specimens from each harvested bear including the lower jaw (or skull) for extraction of a tooth for aging, lip tattoos and ear tags if present. Payment is provided by the Government of Nunavut (GN) for these specimens.

Following harvest of a bear, hunters submit the required specimens and other information to local GN Conservation Officers. For each bear, details are recorded including location of harvest, date, hide length, estimated age, sex, type of hunt (e.g., regular subsistence, sport hunt, defense-of-life-and-property), ear tag number (and tags) if present and lip tattoo number (and tattoo) if present. The hide from each harvested bear is then marked with a uniquely numbered tag (hide seal) that is permanently affixed. This hide seal is required for export of hides from Nunavut and sale.

The information collected on each harvested bear is recorded on Hunter Kill Return (HKR) forms completed by Conservation Officers. HKR forms are submitted to the GN’s polar bear management program where they were checked for omissions and errors before entry into the GN's polar bear harvest database. The sex of harvested bears is again verified by submission of a baculum or by DNA submitted for analysis to Wildlife Genetics International (Nelson, BC, Canada). The age of harvested bears is determined from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998).

HARVEST MANAGEMENT AND MONITORING IN GREENLAND

Harvest management

Following the signing of the *Agreement on Conservation of Polar Bears* in 1973 (Anon. 1973), regulations for the harvest and the protection of polar bears for all areas of Greenland were introduced and were enforced by 1 January 1975 (Anon. 1976, Vibe 1985). Since then, several amendments have been made to the regulations (*cf*. Born 1995). The latest amendment was made in 2005 (Anon. 2005).

When Greenland Home Rule was established in 1979, Greenland took over the legal responsibility for management of its renewable resources, including polar bears. In October 2005, a new Executive Order (Anon. 2005) came into force. Some important protective measures in this executive order are (Anon. 2005, Lønstrup 2006, Hansen 2010):

- year round protection of all cubs (regardless of age) and females accompanied by cubs. The executive order also introduces a prohibition of the export of polar bear cubs;
- protection of all polar bears from 1 July to 31 August; in the local authority districts of Ittoqqortoormiit og Ammassalik from 1 August to 30 September;
- prohibition to disturb or dig out polar bears in dens;
- introduction of quotas from 1 January 2006 and the possibility that part of the quota may be used for trophy hunting. There has never been and currently is no trophy

hunting of polar bears in Greenland. Special provisions on trophy hunting will be laid down in a separate executive order;

- only Greenland residents who hunt as a full-time occupation are allowed to hunt polar bears;
- it is mandatory to report to the Greenland management authorities all catches including struck-and-lost polar bears;
- aircraft, helicopters, motorized vehicles, including snow scooters and boats larger than 20 GRT/15GT are not allowed in the hunt or for transportation to and from the hunting grounds;
- poison, traps, foot snares or self-shooting guns are not allowed;
- rim-fire rifles, shot guns or semi- or fully automatic weapons are not allowed. Polar bears may only be hunted using a rifle with a minimum caliber of 30.06 (7.62 mm);
- all meat, skin and other useable parts of the bear must be brought back (or cached in the field for later use); and,
- no parts of the polar bear must be sold until the catch has been officially registered and the license has received an official stamp.

The Melville Bay Nature Reserve offers protection of polar bears in the Baffin Bay subpopulation. This reserve (10 500 km^2) was established in1980 to protect important polar bear habitat. All hunting within the central (coastal) zone I of nature reserve is prohibited (Vibe 1985, Anon. 1989).

The Greenland Home Rule Act. No. 12 of 29 October 1999 provides the legal framework for wildlife management. Various laws on environmental protection and animal welfare also apply to the management of polar bears (Polar Bear Range States 2015).

Responsibility for the management of polar bears resides with the Department of Fisheries, Hunting and Agriculture, DFHA (Aalisarnermut, Piniarnermut Nunalerinermullu Naalakkersuisoqarfik, APNN) of the Greenland Government.

Quotas for the take of polar bears in Greenland were introduced in 2005 taking effect 1 January 2006 (Lønstrup 2006). The Minister of Fisheries, Hunting and Agriculture sets an annual polar bear quota. The minister drafts a preliminary regional allocation of the quota based on the latest scientific advice and harvest results for the preceeding harvest season, and then sends the draft to the Hunters´ National Association, the municipalities, the Ministry of Nature and Environment and the Greenland Institute of Natural Resources for consultation for a period of not less than five weeks. Based on the resulting consultation, the Ministry of Fisheries, Hunting and Agriculture prepares a final presentation of the total annual quota to the Minister (Polar Bear Range States 2015). Licenses to hunt polar bears are issues by the municipalities, within annual quotas set by DFHA and the National Government.

The Greenland Fisheries License Control Authority is tasked with enforcing the regulations set by the government and the municipalities (Polar Bear Range States 2015).

When polar bear studies conducted by the Greenland Fisheries Research Institute (predecessor of Greenland Institute of Natural Resources, Nuuk) were initiated in 1991, Greenland de facto took over the responsibility providing scientific data for the management of its polar bear subpopulations as outlined in the 1973 *Agreement on Conservation of Polar Bears*.

During the fall of 2000, the Greenland Home Rule Government signed a Memorandum of Understanding (MOU) with the Government of Nunavut (Canada). An appendix to this MOU contains a prioritized list of items, including that there should be cooperation between both regarding shared polar bear subpopulations (Lønstrup 2006).

Harvest Monitoring

Home Rule was established in Greenland 1979. Since then the Department of Fisheries, Hunting and Agriculture (DFHA, Nuuk) has been responsible for organizing the collection of catch statistics in Greenland and for summarizing and publishing the data. Until 1987 information about the number of polar bears taken in Greenland was available through the Hunters´ Lists of Game (HLG) where hunters reported their catch of various wildlife including polar bears voluntarily. The HLG (Anon. 1954-83 and unpublished 1984-87) was based upon the principle that an appointed person from each settlement kept count of the catch of various hunting animals by all the hunters in his settlement and reported the numbers to the authorities (Rosing-Asvid 2002). When such reports for some reason were missing, the central authority added an estimate to account for unreported catch based upon “other information” (i.e., for example notices of catches in newspapers or records of trade of skin etc.). The HLG-summaries of the catch, including estimates of unreported catch, were published annually by the Ministry for Greenland in Copenhagen (until 1983). After the Greenland Home Rule Government took over the HLG-system, unpublished summaries of the catch in 1985, 1986 and 1987 became available from the Department of Fisheries, Hunting and Agriculture, DFHA (Nuuk). However, the reliability of the reporting in the HLG deteriorated since about the mid-1970s (Born 1995, Rosing 1998) and this way of monitoring the catch in Greenland stopped in 1987.

Some information (HLG and trade) was available during the decade prior to the introduction of a new system of reporting catches in 1993 (see the following). However, none of these sources gave the total picture and the size of the annual catch of polar bears from the BB and KB subpopulations in the 1980s and the information of annual catches was largely based on estimates (Born 1995). Born (1995) and Rosing-Asvid (2002) estimated that during 1980-1992 a total of between 25(30) and 70(80) polar bears were taken each year by Greenlanders from the BB and KB subpopulations (the estimates of the total catch was not separated to BB and KB).

On January 1993 a new system of reporting catches – the ”Piniarneq” (Greenlandic word for “catch”) – was introduced in Greenland on 1 January 1996. The “Piniarneq” relies upon each hunter voluntarily reporting his annual catch (between 1 October and 30 September) of various species including polar bears. The Piniarneq system is linked to the issuing of hunting licenses, of which two categories exist: one for full-time hunters and another for part-time hunters. Hunters in both categories have to pay a small fee for renewal of the license, at which time they are obliged to report their catches during the previous 12 months. Only full-time hunters can get a license to hunt polar bears. The information on catches is compiled by the Department of Fishery, Hunting and Agriculture (Nuuk) which publishes summaries of the annual catch by area. In the summaries, the catch is reported by municipality, meaning that in ”Piniarneq” there are no records of the exact site of kill. There is also no information on sex and age of the caught bears or whether the bear(s) was (were) killed during a hunt involving more hunters than the one reporting the catch(es).

In recognition of the fact that a potential problem of reporting catches of polar bears via Piniarneq might be that more than one hunter reports the catch of a bear because several hunters participated in the hunt leading to multiple- reporting of a kill (see Discussion) it became mandatory from 1994 to report every kill of a polar bear on specific forms (“Special Reporting Forms”; “Særmeldingsskemaer”). On these forms the hunter´s name, civil registration number, settlement/town, place and date of the kill, sex and approximate age (young, adult, old) of the bear ha to be given. To be able to reduce the problem of potential multi-reporting of a single kill the hunter who finished off the bear was supposed to fill in the form and also give the name etc.

of the other participants in the catch. During the years this system worked with variable success and not all bears that were caught were reported (Born 1998).

Quotas on polar hunting were first introduced in Greenland in 2006. After the introduction of quotas taking effect 1 January 2006 the Ministry of Fisheries, Hunting and Agriculture has improved the hunting statistics by developing a new database and a double reporting system. This means that a hunter must be issued a license before the hunt and immediately following the hunt the hunter must report the catch to the local authority using a standardized form. This standardized form includes information on the name of the hunter(s), place of residence, date, license number, location of kill, and the sex and age category, and whether the bear was marked. As an additional control, all hunters must report their annual harvest of all species (including polar bears) in Piniarneq (Hansen 2010).

In connection with studies of movement and subpopulation assessment more than 1500 polar bears have been physically marked in the Kane Basin and Baffin Bay subpopulations since the 1970s (Born 1995, Taylor et al. 2005, 2008, Peacock et al. 2012). Before the quotas the hunters received a token payment for returning marks (and transmitters) to the authorities with information on the bear (date, site, sex etc. etc). By the introduction of quotas it became mandatory to report whether a bear was marked or not and return tags (and transmitters) (Anon. 2005).

Since the 1980s biological samples from the polar bear catch (various tissues, sexual organs, teeth for ageing etc.) have been collected during various specific programs in connection with studies of pollution and the demography of the catch (e.g., Rosing-Asvid 2002, Sonne et al. 2012). However, these programs which relied upon the hunters collecting the samples with an economical compensation served specific purposes and in case of monitoring the catch have been intermittent (Rosing-Asvid 2002).

In order to continuously monitor the Greenland catch of polar bears demographically and provide information on sex and age composition of the catch it became mandatory in 2012 for the polar bear hunters to deliver a tissue sample (for genetic analyses) and a small vestigial tooth (for age determination) from each bear killed. The samples shall be sent to the Greenland Institute of Natural Resources in Nuuk which is responsible for arranging the sampling program practically and for processing the samples. Each sample is accompanied by a filled form where with details about the catch (date, site, name of hunter, sex of the bear and its approximate age etc. etc.). The hunters are required to also send the same information to the Greenland management authorities (i.e., the Department of Fisheries Hunting and Agriculture in Nuuk).

LITERATURE CITED

Anon. 1954-87. Sammendrag af Grønlands Fangstlister. Ministeriet for Grønland [Summaries of "Hunters´ Lists of Game"]. Summarized during 1954-1984 by the Danish Ministry of Greenland, Copenhagen, and during 1985-1987 by the Ministry for Fisheries, Hunting and Agriculture, Nuuk. [In Danish].

Anon. 1973. *Agreement on the Conservation of Polar Bears*. Available at: http://pbsg.npolar.no/en/agreements/agreement1973.html.

Anon. 1976. Bekendtgørelse om ændring af bekendtgørelse om fredning af isbjørne i Grønland [Executive order of preservation of polar bears in Greenland]. Nalunaerutit-Grønlandsk Lovsamling, Serie A, Nr. I, MJ.G.j. nr. 1701-D8: 109. [In Danish].

Anon. 1989. Hjemmestyrets bekendtgørelse nr. 21 af 17. maj 1989 om naturreservatet i Melville Bugt [Greenland Home Rule Executive order No. 21 on the Melville Bay Nature Reserve]. http://www.lovgivning.gl/gh.gl love/dk/1989/bkg/bkg_nr_21-1989_dk/bkg_nr_21-19, 05-01-2010. [In Danish].

Anon. 2005. Hjemmestyrets bekendtgørelse nr. 21 af 22. september 2005 om beskyttelse og fangst af isbjørne [Greenland Home Rule Executive order No. 21 of 22 September 2005 on protection and harvest of polar bears], 5 pp. [In Danish].

Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 *in* Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Born, E. W. 1998. Research on polar bears in Greenland, primo 1993 to primo 1997. Pages 23–44 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 10:449–453.

Hansen, K. W. 2010. Polar bear management in Greenland, 2005-2008. Pages 133-134 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Lee, J., and M. Taylor. 1994. Aspects of the polar bear harvest in the Northwest Territories, Canada. International Conference on Bear Research and Management 9:237–243.

Lønstrup, J. 2006. Polar bear management in Greenland. Pages 133–134 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

Rosing, B. 1998. Issues pertaining to polar bear management in Greenland. Pages 93-95 *in* A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Rosing-Asvid, A. 2002. The polar bear hunt in Greenland. Greenland Institute of Natural Resources Technical Report 45, 25 pp.

Sonne, C., R. J. Letcher, T. Ø. Bechshøft, F. F. Rigét, D. C. G. Muir, P. S. Leifsson, E. W. Born, L. Hyldstrup, N. Basu, M. Kirkegaard, and R. Dietz. 2012. Two decades of biomonitoring polar bear health in Greenland: A review. Acta Veterinaria Scandinavica 54 (Supplement 1):S15 doi:10.1186/1751-0147-54-S1-S15.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

Tyrrell, M. 2009. Guiding, opportunity, identity: the multiple roles of the Arviat polar bear conservation hunt. Pages 25–38 *in* M. M. R. Freeman, and L. Foote, editors. Inuit, Polar Bears and Sustainable Use: Local, National and International Perspectives. Canadian Circumpolar Institute Press, Edmonton, Alberta, Canada.

Vibe, C. 1985. Miscellaneous information relating to polar bears in Greenland. Pages 44–57 *in* Polar Bears: Proceedings of the 8th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland.

Wenzel, G. 2008. Sometimes hunting can seem like business: Polar bear sport hunting in Nunavut. Canadian Circumpolar Institute Press, Edmonton, Alberta, Canada.

Wenzel, G. W. 2011. Polar bear management, sport hunting and Inuit subsistence at Clyde River, Nunavut. Marine Policy 35:457–465.

APPENDIX E. Results of binary logistic regressions for body condition scores (poor, fair-good) for polar bears in Kane Basin.

Sex-Age Class	**Independent Variables**	***B***	**Wald Statistic**	***P***
Adult male	Julian Day	-0.095	1.617	0.204
	Epoch (1990s-2010s)	0.230	0.082	0.774
Adult female (lone)	Julian Day	0.104	2.413	0.120
	Epoch (1990s-2010s)	0.775	1.152	0.283
Adult female (with COY)	Julian Day	0.251	9.210	**0.002**
	Epoch (1990s-2010s)	0.327	0.194	0.660
Adult female (with yearling)	Julian Day	0.082	0.617	0.432
	Epoch (1990s-2010s)	2.064	4.081	**0.043**
Subadult	Julian Day	0.111	0.741	0.389
	Epoch (1990s-2010s)	20.056	0.000	0.998

ᐅᖅᑲᓪᓚᖅᑲᑎᖅᑲᑲᑕᒃᓂᖅ ᐊᐅᓱᐃᑦᑑᑉ ᐊᖑᓇᓱᒃᑎᓕᕆᔨᒃᑯᖕᒋᓐᓂᑦ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ ᑲᑎᒍᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑯᔨᐅᔪᓂᒃ

ᕕᕗᐊᕆ 2, 2018

ᐊᕙᑎᓕᕆᔨᒃᑯᑦ, ᓄᓇᕗᑦ ᒐᕙᒪᖓ, ᐃᖃᓗᐃᑦ, ᓄᓇᕗᑦ

ᐊᐅᓚᙱᓂᕐᒧᑦ ᓇᐃᓈᕐᓯᒪᔪᑦ

ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᓐᓄᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᑦ (DOE) ᑮᓇᖑᖅᑐᐃᔨᖏᑦ ᐅᕐᑲᓪᓚᕐᑲᑎᕐᑲᑲᑕᓚᐅᕐᒪᑕ ᐊᐅᓯᐃᑦᑑᑉ ᐊᖑᓇᓱᒃᑎᓕᕆᔨᒃᑯᖏᓐᓂᑦ (HTO) ᕕᕗᐊᕆ 2-ᒥ, 2018 ᐊᐃᑉᐸᕆᓕᕐᓗᓂᐅᒃ ᐅᕐᑲᓪᓚᑲᑕᒃᓂᖏᓐᓂᒃ. ᐱᒋᐊᓕᓵᕈᑎᓂᒃ ᐅᕐᑲᓪᓚᕐᑲᑎᕐᑲᑲᑕᒃᓂᖅᑕᕐᑲᓚᐅᕐᑐᖅ ᔭᓄᐊᕆ 9-ᒥ, 2017 ᐱᑕᕐᑲᖅᑎᑦᑎᓇᓱᒃᓗᑎᒃ ᐊᐅᓚᙱᓂᒃᑯᑦ ᐱᓕᕆᕐᑲᑎᒌᒃᓂᕐᒥᑦ ᐱᔾᔪᑎᕐᑲᓗᐊᖅᓗᑎᒃ ᑮᖑᓕᖅᐹᓂᒃ ᕐᑲᐅᔨᓴᖅᑐᓕᕆᓂᒃᑯᑦ ᕐᑲᐅᔨᓴᕈᑎᓂᒃ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ, ᐊᒻᒪᓗ ᓄᐊᑦᑎᓇᓱᒃᓗᑎᒃ ᑕᒪᑐᒎᖑ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᕐᑲᐅᔨᒪᔭᑐᕐᑲᖏᓐᓂᑦ (TK) ᑲᑎᑦᑎᓇᓱᒃᓗᑎᒃ. ᐊᐃᑉᐸᖓᓂ ᐅᕐᑲᓪᓚᕐᑲᑎᕐᑲᑲᑕᒃᓂᖅ ᕐᑲᐅᔨᓇᓱᑌᑎᐅᓚᐅᕐᑐᖅ ᐆᒥᖓ ᑲᑎᖓᑎᒃ ᓇᓄᒃᑕᐅᔪᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᔪᓂᒃ ᐅᑯᓇᙵ ᑲᓇᑕ-ᑲᓙᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᒼᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᕿᒥᕐᕈᐊᕐᓂᖅᑕᕐᑲᓚᐅᕐᑎᓪᓗᒍ ᓇᓄᒃᑕᐅᔪᓄᑦ ᕐᑯᐊᕐᓵᕐᓇᕈᑎᐅᓇᔭᖅᑐᓂ ᕐᑲᐅᔨᓴᕈᑎᓂᒃ ᐱᐊᓂᒃᑕᐅᓂᑯ ᐅᑯᓇᙵ ᕐᑲᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᓖᑦ ᐱᓕᕆᕐᑲᑎᒌᒃᑐᓂᑦ (SWG).

ᑐᓴᖅᑕᐅᕙᓪᓕᐊᔪᑦ ᐊᒻᒪ ᐃᓄᐃᑦ ᕐᑲᐅᔨᒪᔭᑐᕐᑲᖏ (TK) ᓄᐊᑕᐅᕙᓪᓕᐊᔪᑦ ᓱᕗᓪᓕᕐᒥ ᕐᑲᐅᔨᓇᓱᕐᑲᑦᑕᖅᐸᓪᓕᐊᓕᖅᑎᓪᓗᒋ ᐃᓱᒪᒋᔭᐅᓚᐅᕐᑐᑦ ᑲᓇᑕ-ᑲᓙᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᒼᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᐅᑯᓂᖓ ᑲᑎᖓᑎᒃ ᓇᓄᒃᑕᐅᔪᓇᖅᑐᑦ (TAH) ᐋᕿᒃᓯᐃᕙᓪᓕᐊᑎᓪᓗᒋ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ ᓇᓄᖏᓐᓂ. ᐊᕙᑎᓕᕆᔨᒃᑯᑦ ᑐᓂᓯᓇᔭᖅᑐᑦ ᑲᑎᖓᑎᒃ ᓇᓄᒃᑕᐅᔪᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᓂᒃ ᓄᓇᕗᑦᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ (NWMB) ᐃᓱᒪᓕᐊᕆᔭᐅᓂᐊᕐᒪᑦ.

ᐅᓇ ᐅᓂᒃᑳᖅ ᓈᐃᓈᕐᓯᒪᔫᒃᑯᑦ ᐅᕐᑲᐅᓯᐅᕙᓪᓕᐊᔪᓂᒃ ᑎᑎᕋᐅᓯᕐᑲᕋᓱᒃᑐᖅ ᐃᓚᐅᓂᑯᓂᒃ ᐊᐃᑉᐸᖓᓂᒃ ᕐᑲᐅᔨᓇᓱᒃᓂᕐᒥᑦ ᑲᑎᒪᑎᑦᑎᓂᒃᑯᑦ.

ᐅᕐᑲᐅᓯᕐᑲᕆᐊᓕᕈᑎᖓ

ᐅᓇ ᐅᓂᒃᑳᖅ ᓴᕿᑎᑦᑎᓯᒪᔪᖅ ᐊᑭᑎᓕᕆᔾᑯᑦ ᐱᔪᓐᓇᕐᓂᖏᑦᑎᒍᑦ ᑕᒻᒪᕐᓯᒪᙱᑦᑐᓂᒃ ᑐᓴᖅᐸᓪᓕᐊᓯᒪᔭᖏᓐᓂ ᐅᕐᑲᓪᓚᕐᑲᑎᕐᑲᑲᑕᒃᓂᒃᑯᑦ ᑲᑎᒪᕐᑲᑎᕐᑲᖅᑎᓪᓗᒋᑦ ᐊᐅᓯᐃᑦᑑᑉ ᐊᖑᓇᓱᒃᑐᓕᕆᔾᑯᖏᓐᓂᑦ.

ᐃᓱᒪᒋᔭᐅᓯᒪᔪᑦ ᐅᑭᓂ ᐊᑭᑎᓕᕆᔾᑯᓐᓂᖔᕐᓯᒪᙱᑦᑐᑦ, ᐅᑦᑭᓘᓐᓃᑦ ᓄᓇᕗᑦ ᒐᑭᒪᖓᓄᑦ.

ᕿᓂᕐᕕᒃ

1.0 ᐅᓂᒃᑳᓄᑦ ᐱᔾᔪᑎᐅᔪᑦ ᐊᒻᒪ ᐋᕿᒃᓯᒪᓂᖅ

ᐅᓇ ᐅᓂᒃᑳᖅ ᐅᖃᐅᓯᒃᓴᓂ ᑲᑎᕐᓯᐃᒍᑎᐅᔪᖅ ᐊᒻᒪ ᓈᐃᓈᕐᓯᒪᔪᒃᑯᑦ ᐅᖃᐅᓯᕐᓂᒃ, ᐊᐱᖁᑎᓂᒃ, ᐃᓱᒫᓘᑎᓂᒃ ᐊᒻᒪ ᐃᓱᒪᒋᔭᐅᖁᔭᐅᔪᓂᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᓄᑦ ᑭᐅᓯᒪᓂᕐᒧᑦ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᙵᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᒋᔭᐅᔪᑦ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ (KB) ᓇᓄᖏᓐᓂᒃ. ᐱᒋᐊᕈᑎᐅᓂᑯᑦ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᖕᓂᕐᒥᑦ ᐊᐅᓯᐃᑦᑐᕐᒥᑦ ᐱᓕᕆᓚᐅᖅᑐᑦ ᐃᓱᒪᒋᔭᐅᔪᓂᒃ ᐊᒻᒪ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᒃ (TK) ᓄᐊᓐᓇᓱᓪᓗᒋᑦ ᓯᕗᓂᐊᒍᑦ ᑲᑎᙵᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᔪᓂᑦ ᐆᒧᖓ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ (KB) ᓇᓄᖏᓐᓄᑦ. ᐊᐅᓯᐃᑦᑐᕐᒥᑦ ᐊᓯᐃᓛᒃ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᖅᑕᖃᓚᐅᕐᒥᔪᖅ ᕕᕗᐊᕆ 2–ᒥᑦ, 2018.

ᐱᒋᐊᕈᑎᐅᓂᑯᓂᒃ ᐅᖃᓪᓚᖃᑎᖃᑲᓚᐅᖅᑎᓪᓗᒋᑦ, ᑲᑎᙵᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᑦ ᐅᖃᐅᓯᐅᓚᐅᕐᑐᑦ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᒪᑎᓪᓗᒋ ᐅᑐᐱᕆ 2017-ᒥᑦ. ᐊᐃᑉᐸᖓᓂ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᖅ ᐱᓕᕆᐊᖃᓚᐅᕐᑐᑦ ᐅᖃᐅᓯᖃᖅᙵᑎᒃ ᐃᓱᒪᓕᐊᕆᔭᐅᓯᒪᔪᓂᒃ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᒃ ᑐᓂᓯᑎᓪᓗᒋᑦ ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯ ᑲᑎᒪᔨᖏᓐᓄᑦ (NWMB) ᐃᓱᒪᓕᐅᕆᔭᖏᓐᓂᒃ.

ᑭᖓᖅᑐᐃᔪᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᓐᓂᑦ (DOE), ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ (NWMB), ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᓐᓂᑦ (NTI), ᐊᒻᒪᓗ ᕿᑭᖅᑖᓗᒃᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂᑦ (QWB) ᐅᐸᒍᑎᓚᐅᕐᑐᑦ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᐅᔪᖅ.

2.0 ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᕐᒧᑦ ᐱᔾᔪᑎᒋᔭᐅᔪᑦ

ᓯᕗᓪᓕᕐᒥ ᐅᖃᓪᓚᖃᑎᖃᕐᓂᒃᑯᑦ ᐅᖃᐅᓯᖃᓚᐅᕐᑐᑦ ᓄᑖᙳᓛᓂᒃ ᑐᓴᒐᒃᓴᓂᒃ ᐆᒧᖓ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᐅᓂᒃᑲᐅᓯᐅᓂᖏᑦᑎᒍᑦ ᖃᐅᔨᓴᖅᑐᓕᕆᔨᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᖏᓐᓂᒃ ᐋᕿᒃᑕᐅᓯᒪᔪᖅ ᐅᑯᓇᙵᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᓂᑦ (SWG). ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᑦ (SWG) ᐱᒋᐊᖅᑕᐅᓚᐅᕐᓯᒪᔪᑦ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᓄᐊᑦᑎᓂᐊᕐᓗᑎᒃ ᖃᐅᔨᓴᕐᓂᒃᑯᑦ ᓄᐊᑕᐅᕙᓪᓕᐊᔪᓂᒃ ᐊᒻᒪ ᓄᑖᙳᕆᐊᖅᑕᐅᓗᓂ ᖃᓄᖅ ᐱᓕᕆᖃᑎᒌᒃᓂᕐᒥᒃ ᓇᓄᕐᓂ ᐅᑯᑎᒎᓇᖅ ᑲᓇᑕ/ᓄᓇᕗᑦ ᐊᒻᒪ ᑲᓛᓪᓛᑦ ᓄᓇᖓᓂᑦ. ᓯᕗᓪᓕᕐᒥᑦ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᓚᐅᖅᑎᓪᓗᒋᑦ ᐊᒻᒪᓗ ᐱᐊᓂᒃᓯᑎᓪᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᑦ ᖁᐊᕐᓵᕐᓇᖅᑐᒦᓐᓇᓴᕐᑎᐊᖅᓴᖏᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᕐᒥᑦ ᐅᑯᓄᖓ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔭᓖᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᓂᑦ (SWG), ᐅᑯᐊ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᒪᓚᐅᕐᑐᑦ ᐆᒃᑑᐱᕆ 2017-ᒥ ᐃᓱᒪᓕᐅᕆᓇᓱᒃᓗᑎᒃ ᑲᑎᙵᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ (TAH). ᐊᐃᑉᐸᖓᑕ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᖕᓂᐅᑉ ᐱᔾᔪᑎᖓ ᐊᐅᓯᐃᑦᑐᕐᒥᐅᑦ ᑐᓴᕐᕕᒋᒃᑲᓐᓂᕐᑐᒋᑦ ᑲᑎᙵᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐃᓱᒪᓕᐅᕆᔭᐅᓯᒪᔪᑦ ᑲᓇᑕ-ᑲᓛᓪᓛᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC)–ᑯᓐᓂᑦ ᐊᒻᒪᓗ ᐃᓱᒪᓕᐅᕆᔭᐅᔪᑦ, ᐱᑭᐊᓚᓱᔅᓱᐊᑉ ᓇᓄᖁᑎᖏᓐᓂ, ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ (NWMB) ᑐᓂᔭᐅᓂᐊᖅᑐᖅ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᓐᓄᑦ (DOE) ᐃᓱᒪᓕᐅᕆᔾᔪᑎᒃᓴᐅᔪᓄᑦ.

2.1 ᑲᑎᒪᓂᐅᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ

ᐅᓐᓄᒃᑯᑦ ᑲᑎᒪᓚᐅᖅᑐᑦ ᐊᒻᒪᓗ 5:30–ᒥᑦ 6:00–ᒧᑦ ᑲᑎᒪᓪᓗᑎᒡᓗ. ᑲᑎᒪᓂᖅ ᐊᐅᓚᑕᐅᓚᐅᖅᑐᖅ ᐊᒻᒪᓗ ᓯᕗᓕᖅᑎᒋᔭᐅᓪᓗᓂ ᖁᓪᓕᐅᔪᖅ ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᖃᐅᔨᒪᔨᑕᖅ, ᑭᐅᓇᓐ ᓯᒥᑦ, ᐊᒻᒪᓗ ᖁᓪᓕᐅᔪᖅ ᐊᐅᓚᑦᑎᔨ ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᐊᐅᓚᑕᐅᔪᓄᑦ, ᔭᐃᓴᓐ ᐊᓕᖃᑐᖅᑐᖅ. ᐊᑐᓂ ᐅᖃᓪᓚᖃᑎᖃᑲᑕᒃᓂᐅᔪᑦ ᐱᒋᐊᕈᑎᖃᓚᐅᕐᐳᑦ ᓴᕿᑎᑦᑎᓂᕐᒥᒃ ᖃᓄᐃᓕᖓᓂᖏᓐᓂᒃ ᐱᑭᐊᓚᓱᔅᓱᐊᖅᒥᑦ (KB) ᓇᓄᐃᑦ ᐊᒻᒪᓗ ᑲᑎᙵᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓱᒪᓕᐊᕆᔭᐅᔪᓂᒃ (ᓇᓗᓇᐃᒃᑯᑎ 1). ᑲᑎᒪᖅᑕᐅᔪᑦ ᑐᙵᓱᒃᑎᑕᐅᓚᐅᕐᑐᑦ ᐊᐱᕐᓯᕈᓐᓇᕐᓗᑎᒃ, ᐃᓱᒫᓘᑎᖏᓐᓂᒃ ᐅᖃᐅᓯᖃᕐᓗᑎᒃ, ᐅᕝᕙᓘᓐᓃᑦ ᐊᓯᐊᒍᖕᓛᖅ ᐊᑐᖅᑕᐅᔪᓐᓇᕐᓴᖕᓛᖅᑐᓂᒃ ᑲᑎᒪᑎᓪᓗᒋ. ᓴᕿᑎᑦᑎᓚᐅᖅᑎᓪᓗᒋᑦ, ᐊᐱᖁᑎᑦ/ᐅᖃᐅᓯᒃᓴᐃᑦᓗ ᑲᔪᓯᓚᐅᕐᑐᖅ ᑭᓯᐊᓂ ᐊᐱᖁᑎᒃᓴᖃᕈᓐᓃᕐᒪᑕ.

3.0 ᐅᖃᐅᓯᑦ ᓇᐃᕐᑐᑦ ᐊᐅᔪᐃᕐᑐᕐᒥ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᖏᓐᓂ ᐊᖃᓪᓚᖃᑎᖃᑲᑕᖕᓂᖅ

ᐅᓪᓗᖓ: ᔭᓄᐊᕆ 2, 2018

ᑭᒡᒐᖅᑐᐃᔨᐅᔪᑦ:

- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒪᓂᕐᓴᖅ ᐊᐅᓚᑦᑎᔨ ᐊᐅᓚᑕᐅᔪᓄᑦ: ᔭᐃᓴᓐ ᐊᓕᕐᑲᑐᖅᑐᖅ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᖏᑦ, ᐱᓕᕆᓕᒪᓂᕐᓴᖅ ᓂᕐᔪᑎᓄᑦ ᐅᖃᐅᔾᔪᐃᔨ: ᑭᐅᕆᓐ ᓯᒥᑦ
- ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᔭᑎᓕᕆᔨᖏᑦ, ᕿᑭᖅᑖᓗ ᐅᐊᖕᓇᖓᓄᑦ ᐊᐅᓚᑦᑎᔨ: ᓯᑳᑦ ᔮᓐᓴᓐ
- ᓄᓇᕗᑦ ᑐᙵᕕᒃᑯᑦ, ᐊᕙᑎᓕᕆᔨ ᓯᐅᕈᓪ ᕗᕆ
- ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑕᓂᔅ ᓇᑎᓗ
- ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᑦ, ᐊᕙᑎᓕᕆᔨ: ᑲᐃᐅᓪ ᐅᕆᕐᔪ
- ᕿᑭᖅᑖᓗᒃᒥᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ, ᑲᑎᒪᔨ: ᐋᑕᒥ ᓄᓇ
- ᐃᕕᖅ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ
    - ᔭᕗᕇ ᖃᐅᓐᓇᖅ
    - ᔮᕐᔫ ᑭᒍᑦᑕᖅ
    - ᓕᓴ ᓂᖏᐅᖅ
    - ᒪᐅᑎ ᑯᓗᒍᑕᖅ
    - ᐃᖑᑖᔅ ᑯᓂᔅᑕᖕᓴᖕ
    - ᐊᒪᓐ ᐊᕿᐊᕈᖅ

ᐅᖃᐅᓯᒃᓴᑦ ᐊᒻᒪ ᐊᐱᖅᑯᑎᒃᓴᑦ:

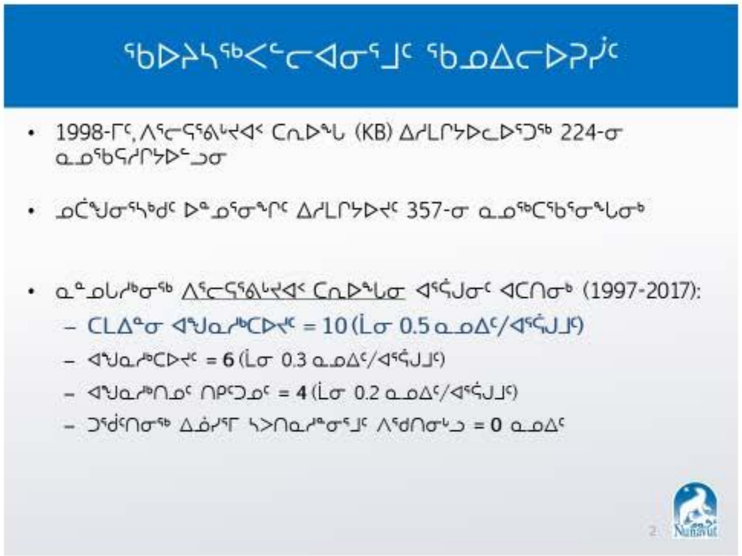
ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᖃᓚᐅᖅᑐᑦ ᐊᖑᓇᓱᒍᓐᓇᐅᑎᓂᑦ ᑕᓪᓕᒪᓂᑦ ᐱᓯᒪᔭᕆᓐᓂ ᐊᓯᐅᔨᒧᒪᕆᑦᑐ ᐱᑭᐊᓚᓱᔅᓯᐊᖅᒧᑦ ᑐᓂᔭᐅᓯᒪᔪᓄᑦ ᓇᓄᕐᑯᑎᖏᓐᓂ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᑐᑭᓯᔪᑦ ᓄᓇᓕᖓ ᐊᖑᓇᓱᒍᓐᓇᐅᑎᓂᒃ ᓄᖑᑦᑎᖅᑲᑦᑕᖏᓐᓂᖓ ᐊᖀᒍᑕᒫᖅ ᐊᒻᒪᓗ ᐊᖀᒍᐃᑦ ᐃᓚᖏᓐᓂ ᐊᖑᓇᓲᑎᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᖏᑦᑐᑦ. ᓄᓇᓕᒃ ᐅᔾᔨᕈᓱᒃᑐᖅ ᓯᕗᓂᒃᓴᕐᑎᓐᓂ ᐊᖅᑯᑎᖅᑲᕐᓂᖅᓴᐅᓛᕐᓂᕐᓂᐊᕐᓂᖓ ᐱᑭᐊᓚᓱᔅᓯᐊᖅᒥᑦ ᓇᓄᕐᑯᑎᖏᓐᓄ, ᐊᖑᓇᓲᑎᓂᒃ ᐊᑐᕈᓐᓇᕐᓂᕐᓴᐅᓕᕐᓗᑎᒃ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐅᖃᓚᐅᖅᑐᑦ ᐃᒃᐱᒍᓱᒃᑐᑦ ᐊᖑᓇᓲᑎᓂᑦ ᐅᑎᖅᑎᑦᑎᓂᖅ ᐊᔪᕐᓇᖅᑐᐊᓘᓪᓚᕐᓂᕐᓗᒍ ᓯᕗᓂᒃᓴᕐᑎᓐᓂ ᒫᓐᓇ ᐅᑎᖅᑎᑕᐅᒃᐸᑕ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᖏᖅᓯᒪᕗᑦ ᒫᓐᓇᐅᔪᖅ ᑕᑦᑕᒪᐃᑦ ᐊᖑᓇᓲᑏᑦ ᐱᓯᒪᔭᐅᑉᐸᑕ ᐃᓯᒪᑕᐅᓇᔭᐅᔪᒥᑦ 4.5% ᐊᖑᓇᓱᒍᓐᓇᕐᓂᕐᒥᑦ ᑲᓛᑦ ᑎᓕᐅᕆᔭᐅᓂᖓ, ᑕᒪᓐᓇ ᐊᖑᓇᓱᒍᓐᓇᕐᓂᖅ 2.8%–ᒧᑦ ᖃᓐᓂᖅᓴᐅᓂᐊᖅᑐᖅ ᑲᓇᑕ-ᑲᓛᓪᓖᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC)–ᑯᓐᓂᑦ ᐃᓯᒪᑕᐅᓇᔭᕐᓗᓂ.

4.0 ᓇᐃᓈᖅᓯᔪᑎᑦ

ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᐊᖏᖅᓯᒪᔪᑦ ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ ᐊᕙᑎᓕᕆᔨᒃᑯᖏᑦ (DOE)ᐃᑲᔪᖅᓱᕐᓗᒋᑦ ᑲᑎᓪᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐃᓯᒪᑕᐅᓇᔭᐅᔪᓂᑦ ᒫᓐᓇᐅᔪᖅ ᑲᑎᓪᓗᑎᒃ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᐊᓯᔾᔨᖅᑕᐅᖏᓪᓗᓂ ᑕᒪᒃᑯᐊ ᑕᑦᑕᒪᐃᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐱᑭᐊᓚᓱᔅᓯᐊᑉ ᓇᓄᕐᑯᑎᖏᓐᓂ. ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑉ ᐊᖏᖅᓯᒪᔪᑦ ᐃᑲᔪᖅᓯᐃᒐᑐᐊᖅᓗᑎᒃ ᑎᓕᐅᕆᔭᐅᔪᒥᑦ 4.5% ᓇᓐᓄᒍᓐᓇᕐᓂᖅ, ᑲᓛᓪᓖᑦ ᐱᔪᒪᓯᒪᔭᖏᓐᓂ, ᓇᓐᓄᒃᑕᐅᔪᑦ 2.8%-ᒥᑦ ᖃᓐᓂᖅᓴᐅᓂᐊᖅᑐ, ᑲᓇᑕ-ᑲᓛᓪᓖᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᑲᒥᓴᑯᓐᓂᑦ ᓇᓄᕐᓄᑦ (JC)–ᑯᓐᓂᑦ ᐃᓯᒪᑕᐅᓇᔭᕐᓗᓂ. ᐊᖑᓇᓱᒃᑐᓗᕆᔨᒃᑯᑦ ᐊᖏᖅᓯᒪᔪᑦ ᑎᑎᕋᕈᒪᓪᓗᑎᒃ ᖃᕋᓴᐅᔭᒃᑯᓪᓘᓐᓃᑦ ᐃᑲᔪᖅᓯᐃᒐᑐᑎᒃ ᐃᓯᒪᑕᐅᓇᔭᐅᔪ ᐊᒻᒪᓗ ᓄᓇᕗᕐᒥ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃ ᑲᑎᒪᔨᖏᓐᓄᑦ (NWMB) ᑐᓂᓗᒋᑦ.

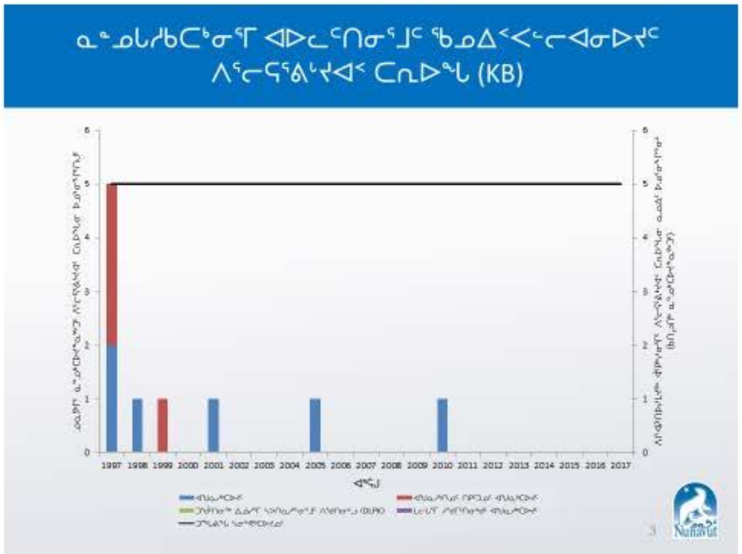
Building *Nunavut* Together
*Nunavut*liuqatigiingniq
Bâtir le *Nunavut* ensemble

ᓇᓄᒐᓯᖃᑦᑕᕐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ ᐅᕙᓂ ᐱᕐᓚᕐᕕᔾᔭᐊᑉ ᑕᓇᐅᖓ

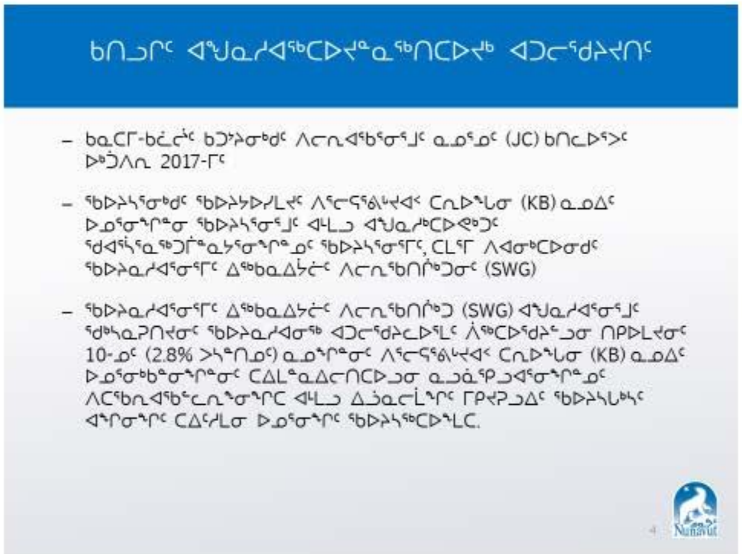


ᖃᐅᔨᓴᖅᐸᓪᓚᐊᓂᕐᒧᑦ ᖃᓄᐃᓕᐅᕈᓰᑦ

- 1998-ᒥᑦ, ᐱᕐᓚᕐᕕᔾᔭᐊᑉ ᑕᓇᐅᖓ (KB) ᐃᓯᒪᔨᐅᓚᐅᕐᑐᖅ 224-ᓂ ᓇᓄᖅᑲᓯᒋᔭᐅᓪᓗᓂ
- ᓄᑖᖑᓂᖅᓴᒃᑯᑦ ᐅᓐᓄᕐᓂᖓᒋᑦ ᐃᓯᒪᔨᐅᔪᑦ 357-ᓂ ᓇᓄᖅᑕᖃᕐᓂᖓᓂᒃ
- ᓇᓄᒐᓱᒃᓂᖅ ᐱᕐᓚᕐᕕᔾᔭᐊᑉ ᑕᓇᐅᖓᓂ ᐊᕐᕋᒍᓂᑦ ᐊᑕᑎᓂᒃ (1997-2017):
  - ᑕᒪᐃᓐᓂ ᐊᖑᓇᓱᒃᑕᐅᔪᑦ = 10 (ᒫᓂ 0.5 ᓇᓄᐃᑦ/ᐊᕐᕌᒍᒧᑦ)
  - ᐊᖑᓇᓱᒃᑕᐅᔪᑦ = **6** (ᒫᓂ 0.3 ᓇᓄᐃᑦ/ᐊᕐᕌᒍᒧᑦ)
  - ᐊᖑᓇᓱᒃᑎᓄᑦ ᑎᑭᑦᑐᓄᑦ = **4** (ᒫᓂ 0.2 ᓇᓄᐃᑦ/ᐊᕐᕌᒍᒧᑦ)
  - ᑐᕐᖁᑦᑎᓂᖅ ᐃᓅᓯᕐᒥ ᓴᐳᑎᓇᓱᓐᓂᕐᒧᑦ ᐱᕐᖁᑎᓂᒡᓗ = **0** ᓇᓄᐃᑦ

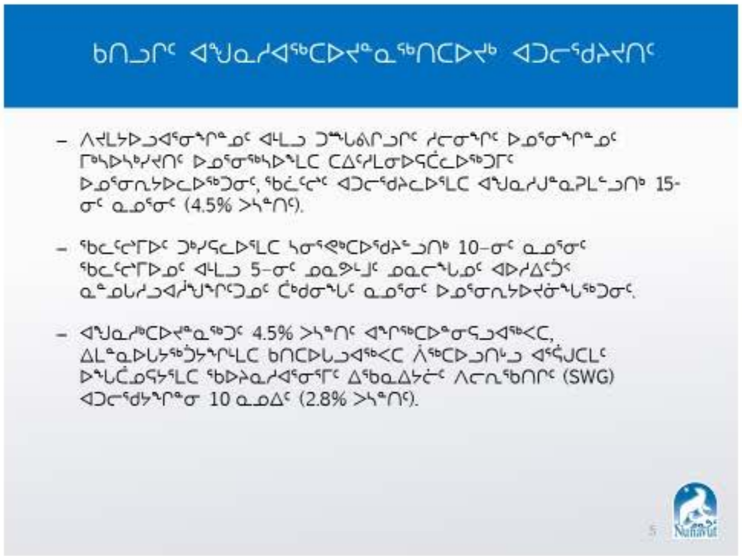


ᓇᓐᓄᒐᓯᑲᑕᒃᓂᕐᒥ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᖃᓄᐃᑉᐸᓪᓕᐊᓂᐅᔪᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᓐᖓ (KB)



ᑲᑎᒍᒥᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔭᓐᓇᖅᑎᑕᐅᔪᒃ ᐊᑐᓕᕐᑯᔨᔪᑎᑦ

- ᑲᓇᑕᒥ-ᑲᓛᓖᑦ ᑲᑐᔾᔨᓂᒃᑯᑦ ᐱᓕᕆᐊᖃᕐᓂᕐᒧᑦ ᓇᓄᕐᓄᑦ (JC) ᑲᑎᓚᐅᕐᐳᑦ ᐅᒃᑑᐱᕆ 2017-ᒥᑦ
- ᖃᐅᔨᓴᕐᓂᒃᑯᑦ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᓐᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᖅᓯᓐᓂ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᒻᒪᓗ ᐊᖑᓇᓱᒃᑕᐅᕙᒃᑐᑦ ᖁᐊᕐᓵᕐᓇᖅᑐᒦᓐᓇᓴᕐᓂᖅᑎᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᕐᒥᑦ, ᑕᒪᕐᒥ ᐱᐊᓂᒃᑕᐅᓂᒃᑯᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᑦ ᐱᓕᕆᖃᑎᒌᒃᑐᓂᑦ (SWG)
- ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖅᑲᓇᐃᔮᑦ ᐱᓕᕆᖃᑎᒌᒃᑐ (SWG) ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᖁᒃᓴᓇᕈᑎᔪᓂᑦ ᖃᐅᔨᓇᓱᐊᓂᖅ ᐊᑐᓕᕐᑯᔨᓚᐅᕐᒪᑦ ᐋᖅᑕᐅᕐᑯᔨᓪᓗᓂ ᑎᑭᐅᒪᔪᓂᑦ 10-ᓄᑦ (2.8% ᐳᓴᓐᑎᓄᑦ) ᓇᓄᖕᒋᓐᓂᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᓐᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖕᒋᓐᓂᑦ ᑕᐃᒪᓐᓇᐃᓕᑎᑕᐅᓗᓂ ᓇᓗᓈᕐᑭᓗᐊᕐᓂᖕᒋᓐᓄᑦ ᐱᑕᖃᓇᐊᖅᑲᓪᓚᓇᖕᓂᖕᒋᑕ ᐊᒻᒪᓗ ᐃᓅᓇᓕᒫᖕᒋᑦ ᒥᑭᔪᕈᐃᑦ ᖃᐅᔨᓴᒐᒃᓴᑦ ᐊᖏᓂᖕᒋᑦ ᑕᐃᑉᓯᒪᓂ ᐅᓄᕐᓂᖕᒋᑦ ᖃᐅᔨᓴᖅᑕᐅᖕᒪᑕ.



ᑲᑎᑐᒥᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑎᑕᐅᔪᒃ ᐊᑐᓕᖁᔨᔪᑎᑦ

- ᐱᔪᒪᔭᐅᓗᐊᕐᓂᖓᓐᓄᑦ ᐊᒻᒪᓗ ᑐᖕᖓᕕᒋᓗᒋᑦ ᓱᓕᓂᖓᑦ ᐅᓄᕐᓂᖓᓐᓄᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑎᑦ ᐅᓄᕐᓂᖅᓴᐅᖓᒪᑕ ᑕᐃᑦᓯᒪᓂᐅᕋᑖᓚᐅᖅᑐᒥᑦ ᐅᓄᕐᓂᕆᔭᐅᓚᐅᖅᑐᓂᑦ, ᖃᑖᑦᓕᓪᑦ ᐊᑐᓕᖁᔨᓚᐅᕐᒪᑕ ᐊᖑᓇᓱᒍᓐᓇᕈᒪᓪᓗᑎᒃ 15-ᓂᑦ ᓇᓄᕐᓂᑦ (4.5% ᐳᓴᓐᑎᑦ).
- ᖃᓪᓕᑦᓕᕆᒥᐅᑦ ᑐᒃᓯᕋᓚᐅᖅᒪᑕ ᓴᓂᖏᒃᑕᐅᖁᔨᓪᓗᑎᒃ 10–ᓂᑦ ᓇᓄᕐᓂᑦ ᖃᓪᓕᑦᓕᕆᒥᐅᓄᑦ ᐊᒻᒪᓗ 5–ᓂᑦ ᓄᓇᕗᒻᒧᑦ ᓄᓇᓕᖕᒐᓄᑦ ᐊᐅᓯᐃᑦᑑᑦ ᓇᓐᓄᒐᓱᓗᐊᓱᖕᖑᕆᑦᑐᓄᑦ ᑖᒃᑯᓂᖓᑦ ᓇᓄᕐᓂᑦ ᐅᓄᕐᓂᕆᔭᐅᔪᓃᖕᒐᖅᑐᓂᑦ.
- ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ 4.5% ᐳᓴᓐᑎᑦ ᐊᖏᖅᑕᐅᓂᖏᓗᐊᖅᐸᑕ, ᐃᒪᓐᓇᐅᒐᔭᖅᑑᔭᕆᕙᑕ ᑲᑎᑕᐅᒐᓗᐊᖅᐸᑕ ᐋᖅᑕᐅᓗᑎᒡᓗ ᐊᕐᕌᒍᑕᒪᑦ ᐅᖓᑖᓄᕋᔭᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐃᖃᓇᐃᔭᖏᑦ ᐱᓕᕆᖃᑎᒌᑦ (SWG) ᐊᑐᓕᖁᔭᕆᓐᓂ 10 ᓇᓄᐃᑦ (2.8% ᐳᓴᓐᑎᑦ).

ᖁᔭᓐᓇᒦᒃ
THANK YOU
QUANAQUTIN
MERCI

ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ

ᐅᑯᐊ ᑎᑎᕋᖅᓯᒪᔪᑦ ᓇᓗᓇᐃᔭᐃᔪᑦ ᐅᓂᒃᑳᓂ ᑕᐃᔭᐅᔪᖅ *“ᖃᐅᔨᓴᒃᑲᓂᕐᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᕐ ᐃᒪᖓ ᐊᒻᒪ Kane Basin ᓇᓄᐃᑦ ᐊᒥᒌᓂᖏᓐᓂᑦ: ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᐱᓕᕆᖃᑎᒌᓐᓂᕐᒧᑦ ᓇᓄᕐᓂᑦ”* ᑐᓂᔭᐅᔪᑦ ᑲᒪᔨᓄᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᓇᓄᕐᓂᑦ 31 ᔪᓚᐃ 2016. ᑎᑎᕋᖅᓯᒪᔪᑦ ᖃᐅᔨᒪᔭᐅᓂᐊᖅᑐᑦ “SWG 2016” (ᑕᑯᒃᑭᑦ ᐅᖃᖅᑕᐅᓯᒪᔪᑦ ᐊᑖᓂ). ᓇᓗᓴᐃᔭᖅᓯᒪᓂᖏᑦ ᓇᓗᓇᐃᔭᐃᔪᑦ ᐃᓗᓕᖏᓐᓂ 14 ᓴᑉᑕᓂ ᑕᕝᕙᓂ ᐅᓂᒃᑳᓂᑦ, ᐅᖃᐅᓯᐅᔪᑦ ᖃᐅᔨᔭᐅᓪᓗᐊᑕᖅᓯᒪᔪᑦ, ᐊᒻᒪ ᐊᑐᓕᕐᑯᔭᓕᐅᖅᑐᑎ ᓯᕗᓂᔅᓴᒥ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) Kane Basin (KB) ᓇᓄᐃᑦ ᐊᒥᒌᓂᖏᓐᓄᑦ.

ᐅᓂᒃᑳᓂᑦ ᐅᖃᖅᑕᐅᓯᒪᔪᑦ: SWG [ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᖃᑎᒌᑦᑐᑦ ᑲᓇᑕ−ᐊᑯᑭᑦᑐᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᑦ ᓇᓄᕐᓂᑦ]. 2016. ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin ᓇᓄᐃᑦ ᐊᒥᒌᓂᖏᓐᓂᑦ: ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᑦ ᑲᓇᑕᒥ−ᐊᑯᑭᑦᑐᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᑦ ᓇᓄᕐᓂᑦ. 31 ᔪᓚᐃ 2016: x + 636 pp.

ᐅᓂᒃᑳᑦ ᑎᑎᕋᖅᑕᐅᔪᑦ S. N. ᐋᑦᑭᓐᓴᓐ (SWG), E. W. ᐳᐊᕐᓐ (SWG), K. L. ᓚᐃᑐᖃᐃ (SWG), N. J. ᓚᓐ (SWG) ᐊᒻᒪ Ø. ᕗᐃᒃ (SWG) ᐊᒻᒪ ᓯᒪᑖᓃᖔᖅᑐᑦ ᐊᔪᖏᓐᓂᓖᑦ T. ᐋᓄᕐᑦ, M. ᑎᒃ, E. V. ᐱᕆᐅ, H. ᓯᑑᓐ, ᐊᒻᒪ S. ᓯᑕᐃᐳᑕᓐ ᐊᒻᒪ ᓇᑲᑎᖅᑕᐅᓪᓗᓂ K. L. ᓚᐃᑐᕋᐃ ᐊᒻᒪ N. J. ᓚᓐ.

ᓵᑉᑕ 1: ᐱᒋᐊᕐᓂᖓ

ᑲᓇᑕᒥ−ᐊᑯᑭᑦᑐᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᑦ ᓇᓄᕐᓂᑦ (JC) ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᖅ ᐊᑎᓕᐅᖅᑕᐅᓪᓗᓂ “ᐊᖏᖃᑎᒌᒍᓯᒥᒃ ᑐᑭᓯᐅᒪᖃᑎᒌᓐᓂᕐᒧᑦ ᑲᓇᑕᐅᑉ ᒐᕙᒪᒃᑯᖏᓐᓂ, ᓄᓇᕗᑦ ᒐᕙᒪᒃᑯᑦ, ᐊᑯᑭᑦᑐᑦ ᒐᕙᒪᒃᑯᖏᑦ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓇᓄᕐᓂᑦ” (MOU) 30 ᐅᑦᑑᐱᕆ 2009 (Anon. 2009). ᑐᕌᓗᐊᑕᕆᔭᖏᑦ MOU ᐱᖃᓯᐅᔾᔨᓗᑎ (1) ᐊᐅᓚᑦᑎᓗᑎ ᓇᓄᕐᓂᑦ ᐃᓗᐊᓂ Kane Basin ᐊᒻᒪ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᐅᓚᑕᐅᔪ)ᓂᑦ ᐱᐅᖅᓯᐊᕐᓂᒧᑦ ᐊᒻᒪ ᓄᖑᓴᐃᖏᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓯᕗᓂᔅᓴᒧᑦ ᐊᒻᒪᓗ (2) ᓴᖅᑭᑎᑦᑎᓗᑎ ᐊᐅᓚᓂᖃᑦᑎᐊᖅᑐᒥᑦ ᐊᐅᓚᑕᐅᓂᖏᓐᓄᑦ ᐱᖃᓯᐅᔾᔨᓂᐊᖅᑐᑦ ᒪᓕᒡᓗᑎ ᐊᑐᖅᑕᐅᒋᐊᓕᓐᓂᑦ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ. JC ᓴᖅᑭᑦᑎᓚᐅᖅᑐᑦ ᑕᒡᒪᓂᑦ ᐃᓚᒋᔭᐅᔪᓂᑦ ᖃᐅᔨᓴᖅᑎᓂᑦ ᐱᓕᕆᖃᑎᒌᑦᑐᓂᑦ (SWG) ᐅᖃᐅᔾᔨᒋᐊᕈᓐᓇᕐᓗᑎ ᖃᐅᔨᓴᖅᑎᓅᖔᖅᑐᓂᑦ ᐊᒻᒪ ᐊᑐᓕᕐᑯᔭᓕᐅᕐᓗᑎ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ (BB) ᐊᒻᒪ Kane Basin (KB) ᓇᓄᖏᓐᓂ. JC ᐱᓕᕆᑎᑦᑎᓚᐅᖅᑐᑦ SWG−ᑯᓐᓂᑦ ᐊᑐᖅᑐᑎ ᐱᐅᕌᖑᔪᓂᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ:
(1) ᐱᕐᑯᔾᔨᓗᑎ ᖃᑎᑦᑐᒥᑦ ᐆᒪᔪᖅᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᕐᑯᕝᕙᓯᓐᓂᖏᑦ KB−ᒧᑦ ᐊᒻᒪ BB ᓇᓄᖏᓐᓄᑦ ᑐᓂᓯᓗᒋᑦ JC ᐅᓂᒃᑳᓂᑦ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐊᑐᓕᕐᑯᔭᖏᓐᓄᑦ; ᐊᒻᒪ (2) ᖃᐅᔨᓴᖅᑎᓅᖔᖅᑐᓂ ᐅᖃᐅᔾᔨᒋᐊᕐᓗᑎ JC−ᒧᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᑦᑐᖅᑕᐅᔪᑦ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᓇᓄᕐᓄᑦ.

ᒪᐃᖑᑎᓪᓗᒍ 2010 SWG ᐅᓂᒃᑳᓕᐅᓚᐅᖅᑐᑦ JC−ᑯᓐᓄᑦ (SWG 2010) ᓇᓗᓇᐃᔭᐃᔪᓂᑦ ᒫᓐᓇᐅᔪᒥᑦ ᐊᑐᐃᓐᓇᐅᔪᑦ ᑐᑭᓯᒋᐊᕈᑎᔅᓴᐃᑦ ᐊᒥᒌᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓄᑦ. SWG ᓄᖅᑲᐅᓯᓚᐅᖅᑐᑦ ᒥᔅᑐᓴᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᒌᓂᖏᓐᓄᑦ ᐊᒻᒪ ᕐᑯᕝᕙᓯᓐᓂᓇᕆᐊᖃᖅᖏᑦ BB−ᒧᑦ ᐊᒻᒪ KB−ᒧᑦ ᐅᓪᓗᒥᒨᖓᓚᐅᖏᑦᑐᑦ. ᐃᓚᒋᐊᕐᓗᒍ, ᐊᒥᒌᓂᖏᓐᓄᑦ ᓈᒻᒪᓐᓂᓇᔪᓐᓇᖅᑕᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᖅ (PVA) ᑖᒃᑯᐊ ᐊᑐᖅᑐᒥᑦ ᐊᒻᒪ ᒫᓐᓇᐅᔪᒥᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᑦ ᕐᑯᕝᕙᓯᓐᓂᑦ 100%−ᒥ ᕐᑯᕝᕙᓇᐊᓚᐅᖅᑐᑦ ᐱᓪᓚᓇᖑᐊᖅᑎᑕᐅᔪᓄᑦ ᒥᔅᑐᓴᑦᑕᐅᓯᒪᔪᑦ ᑕᒪᒃᑭᓐᓄ ᐊᒥᒌᓂᖏᓐᓄᑦ ᖃᑦᑏᓐᓇᕈᓇᐊᕋᔭᖅᑐᑎᑦ ᕐᑯᓕᑦ ᐊᕐᕌᒍᐃᑦ ᖄᖏᖅᓯᒪᓕᖅᑲᑕ (PBSG 2010). SWG ᖃᐅᔨᒃᑲᐃᓚᐅᖅᑐᑦ JC−ᑯᓐᓂᑦ ᐱᓪᓚᓇᖑᐊᖅᑎᑕᐅᔪᑦ ᐊᐅᓚᓂᖃᕋᔪᒌᑦ ᐅᖓᑖᓅᖏᑦᑐᑦ 10−15 ᐊᕐᕌᒍᐃᑦ ᐅᖓᑖᓅᑦ ᑭᖑᓪᓕᖅᐹᒥ ᑐᑭᓯᒋᐊᕈᑏᑦ ᐱᔭᐅᑎᓪᓗᒋᑦ, ᑐᑭᖃᖅᑎᑦᑎᔪᖅ PVA−ᖑᔪᑦ ᐅᓪᓗᒥᒨᖓᓚᐅᖏᑦᑐᑦ ᑕᐃᒪᐃᓐᓂᖓᓄᓪᓗ ᑐᓐᖓᕕᒋᔭᔅᓴᐅᓇᑎ. ᐃᓚᒋᐊᕐᓗᒍ, SWG ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᑦᑐᖅᑕᐅᓂᐅᔪᓐᓇᖅᑐᑦ ᐊᕙᑎᒧᑦ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ, ᐊᒻᒪ ᓄᓇᒥ ᕐᑯᕝᕙᓯᓐᓂᖏᑦ ᐊᖏᔪᑲᓪᓚᒻᒥᑦ ᐊᓯᔾᔨᕈᓐᓇᖅᓯ,ᔪᖅ ᑭᖑᓪᓕᖅᐹᒥ ᓈᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ ᑖᒃᑯᓂᖓ ᐊᒥᒌᓂᖏᓐᓂᑦ 1990−ᖏᓐᓂ. ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᓪᓗᒥᒨᖓᖏᑦᑐᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᐊᒥᒌᓂᖏᓐᓄᑦ ᐊᒻᒪ ᕐᑯᕝᕙᓯᓐᓂᕆᔭᕆᐊᖃᖅᑕᖏᑦ (PBSG 2010) ᐊᒻᒪ ᐊᖏᔪᒥᑦ ᐊᕙᑎᖓᓂᑦ ᐊᓯᔾᔨᖅᑐᑦ ᖃᐅᔨᔭᐅᔪᑦᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᕐᑯᓕᓂᑦ ᐊᕐᕌᒎᓕᖅᑐᓕᑦ, SWG ᐊᑐᓕᕐᑯᔨᓚᐅᖅᑐᑦ ᓄᑖᑦ ᒥᔅᑐᓴᑦᑕᐅᓯᒪᔪᑦ

ᐊᒥᓲᓂᖏᑦ, ᐊᒥᓲᓂᖏᓐᓄᑦ ᓇᓗᓇᐃᔭᐃᓂᖅ, ᐊᒻᒪ ᖁᕝᕙᓯᓐᓂᕆᐊᖃᖅᑕᖏᑦᐱᔭᐅᓗᑎ BB−ᒧᑦ ᐊᒻᒪ KB (SWG 2010).

ᐊᑐᐃᓐᓇᕈᖅᐸᓕᐊᓪᓗᓂ ᐱᕆᐊᖅᑎᑦᑎᓂᕐᒥᑦ ᐊᑕᕐᕖᔪᒥᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᓈᓴᐃᓂᕐᒥᑦ BB−ᒥᑦ ᐊᒻᒪ KB, ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᖃᐅᔨᓴᕈᑎᖃᓚᐅᖅᑐᑦ ᓯᑯᒃᑯᑦ ᓂᕆᖓᑕ ᑲᓇᓐᓇᖓᓂ ᕿᑭᖅᑖᓗᒻᒥ ᐅᐱᕐᖓᖓᓂ 2010 ᕿᒥᕐᕈᓪᓗᑎ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᑦ. ᑕᕝᕙᖓᑦ ᖃᐅᔨᓴᕆᐊᖓᐅᑎᐅᔪᒥᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᖅ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᐃᓚᒋᔭᖓ ᐊᒥᓱᓚᖓᓂᑦ ᓈᒻᒪᑦᑐᓂ ᓇᓖᕋᔅᓴᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ BB−ᒥ ᐊᒻᒪ KB. ᐅᓂᒃᑳᓕᐅᖅᑐᖃᕋᑖᓚᐅᖅᑐᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐊᔾᔨᒌᖏᑦᑐᓂᑦ ᓇᓖᕋᔅᓴᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐊᑐᓕᖁᔭᐅᔪᑦ ᓈᒻᒪᑦᑐᓂᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᓄᑦ (MR), ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓃᑦ, ᐊᒻᒪ ᐊᐅᖏᑎᒍᑦ M-R (SWG 2011). SWG ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᖅ ᖃᓄᐃᓕᖓᓂᖓ MR ᐱᐅᓛᖑᓚᐅᖅᑐᖅ ᐊᑐᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᒻᒪ ᖁᕝᕙᓯᓐᓂᕆᐊᖃᖅᑕᖏᓐᓄᑦ BB−ᒥ. ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓗᓴᐃᖅᑕᐅᓚᐅᖅᑐᖅ ᐱᐅᓛᖑᓪᓗᓂ ᐊᑐᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᕕᒋᓗᒍ KB. SWG ᐃᓕᓴᖅᓯᓚᐅᖅᑐᑦ ᐱᔪᓐᓇᕈᑎᒃᑲᓐᓂᖏᓐᓂᑦ ᐊᒻᒪ ᐱᔪᓐᓇᖏᔾᔪᑎᖏᓐᓂᑦ ᑕᒪᒃᑭᓐᓄᑦ ᐊᑐᖅᑕᐅᔪᓄᑦ. ᐅᖃᖃᑎᒋᔾᔪᑎᒋᓚᐅᖅᑐᒍ SWG ᐅᓂᒃᑳᖓ, JC ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᐅᖏᑎᒍᑦ MR ᐊᑐᖅᑕᐅᖠᕈᒪᔪᖅ ᑕᒪᒃᑭᓐᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐃᑲᔪᖅᓱᑕᐅᑎᐊᖏᓐᓂᖏᓐᓄᑦ ᐃᓄᓐᓄᑦ ᓄᓇᕗᒻᒥ ᑎᒥᑎᒍ MR ᐊᑐᖅᑕᐅᓂᖓ, ᐊᒻᒪ ᐱᑎᕆᓪᓗᒍ ᓯᑯᖓ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᐊᑦᑕᓇᖅᑐᑏᑎᑦᑎᑐᐃᓐᓇᓇᐊᖃᖅᑐᑦ ᓱᓕᖏᑦᑐᓂ ᐱᓗᓂ ᐊᒥᓲᓂᖏᓐᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᒃᑯᑦ.

ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, JC ᐊᑐᓕᖁᔮᓚᐅᖅᑐᑦ ᐊᑎᓕᐅᖅᓯᒪᔪᓄᑦ MOU−ᒥ ᐊᕐᕌᒍᓄᑦ ᐱᖓᓱᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᐊᒥᑦ ᐊᐅᖏᑎᒍᑦ MR BB−ᒥ ᐊᒻᒪ KBᒥ ᓴᖅᑭᖅᑎᑕᐅᓗᓂ.

ᐊᕐᕌᒍᒐᓚᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᐊᖅ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᖅ SWG−ᑯᓐᓄᑦ ᐃᓗᓕᖃᓚᐅᖅᑐᖅ ᐱᖓᓱᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ: (1) ᐊᐅᖏᓪᓂ ᖃᐅᔨᓴᖃᑦᑕᕐᓂᖅ ᓇᓄᕐᓂᑦ ᐱᓗᑎ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᑲᓇᓐᓇᖓᓂ ᕿᑭᖅᑖᓗᒃ, ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐊᒻᒪ KB-ᒥᑦ; (2) ᐋᖅᑭᑦᑕᐅᓂᖏᑦ ᖁᓐᒧᐊᑦᑎᑕᐅᓯᒪᔫᒃᑯᑦ ᖃᐅᔨᓴᐅᑏᑦ ᐊᖑᑎᓄᑦ ᐊᒻᒪ ᐊᕐᓇᓄᑦ ᓇᓄᕐᓂᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐊᒻᒪ KB-ᒥᑦ ᐊᒻᒪᓗ (3) ᐊᖑᓇᓯᑦᑎᓂᑦ ᓄᐊᑕᐅᓂᖏᑦ ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐊᑐᓂ (ᓇᓐᓄᑦᑕᐅᔪᓂᑦ) BB-ᒥ, KB ᐊᒻᒪ ᐊᑦᑐᐊᓂᓖᑦ. ᐱᖓᓱᓄᑦ ᐊᕐᕌᒍᓄᑦ ᖃᐅᔨᓴᕈᒪᓂᖃᓚᐅᖅᑐᑦ ᐅᑭᐊᖕᓴᕈᕋᑖᖅᑎᓪᓗᒍ 2011 ᐅᑯᓂᖓ ᑐᕋᒪᖃᖅᑐᑎᑦ: 1) ᒥᔅᓴᐅᓴᑦᑕᐅᓗᑎ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᓪᓗ (ᐊᒻᒪ ᐊᕐᕌᒍᖏᑦ) ᓇᓄᖏᑦ BB−ᒥ ᐊᒻᒪ KBᒥ; 2) ᐊᔾᔨᒌᖑᐊᑎᓪᓗᒋᑦ ᓄᑖᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ (1991−1997), ᖃᐅᔨᓗᒋᑦ ᐊᒥᓲᓂᖏᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ; 3) ᓇᓗᓇᐃᕐᓗᒋᑦ ᑭᓪᓕᖏᑦ BB−ᒧ ᐊᒻᒪ KB−ᒧᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᒻᒪ ᖃᐅᔨᓴᒃᑲᓂᕐᓗᒋᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓂᖏᓐᓄᑦ ᑖᒃᑯᐊ; 4) ᒥᔅᓴᐅᓴᓪᓗᒋᑦ ᐆᒪᓂᓇᕙᑦᑕᖏᑦ ᐊᒻᒪ ᕿᑐᕐᖏᐅᕐᓂᖏᑦ (ᐊᑕᖏᕈᓐᓇᖅᑐᓕᒫᓂᑦ) ᐊᑐᖅᑕᐅᓗᑎ ᐊᒥᓲᓂᖏᓐᓄᑦ ᓈᒻᒪᓐᓂᖏᓐᓄ ᖃᐃᔨᓴᐃᓗᑎᓗ; ᐊᒻᒪ 5) ᖃᐅᔨᓴᕐᓗᒋᑦ ᓇᓄᐃᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐊᕙᑎᖓᓄᑦ ᐊᔾᔨᒌᖏᖃᑦᑕᕐᓂᖓᓄᑦ, ᐱᓗᐊᖅᑐᒥ ᓯᑯᖓ ᖃᓄᐃᓐᓂᖓ, ᓄᓇᖓ ᐊᒻᒪ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᐊᑐᐃᓐᓇᐅᓂᖏᑦ ᓂᕿᖏᑦ. ᒫᓐᓇᐅᔪᒥᑦ ᐅᓂᒃᑳᑦ (SWG 2016) ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᑐᖅᑕᓕᒫᖑᔪᓂᑦ, ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᐊᑐᖅᑕᐅᔪᑦ, ᖃᓄᐃᓐᓂᖏᑦ ᐊᒻᒪ ᐅᖃᐅᓯᐅᓂᖏᑦ ᑖᔅᓯᒪᖓ ᐊᕐᕌᒍᒐᓚᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐱᓕᕆᐊᖅ.

ᓭᑉᑕ 2: ᐊᒥᓲᓂᖏᑦ ᓇᓗᓇᐃᕝᕙᒃᑕᐅᓂᖏᑦ ᓴᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin

ᓄᑖᓂᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᖃᐅᔨᓴᕐᓂᖅᑕᖃᓯᒪᖏᑦᑐᖅ BB−ᒥ ᐅᕝᕙᓘᓐᓃᑦ KBᒥ ᐅᓪᓗᒥᒧᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᓇᓗᓇᐃᕝᕙᒃᑕᐅᓂᖏᓐᓄᑦ 1997−ᒥᑦ. ᓭᑉᑕ 2 ᐊᑐᖅᑐᖅ ᒫᓇᕋᑖᖑᔪᒥ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᐅᖏᑎᒍᑦ, ᕿᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ, ᐊᒻᒪ ᐱᕝᕚᒃᑲᓂᖅᑐᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓈᓄᑦᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᓂᕐᒨᖓᔪᓂᑦ BB−ᒥ ᐊᒻᒪ KBᒥ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᖅᑐᑎ ᐊᒥᓲᓂᖏᓐᓄᑦ ᑭᓪᓕᖏᑦ.

ᓭᑉᑕ 2 ᓯᕗᓪᓕᕐᒥ ᓇᓗᓇᐃᕝᐃᔪᖅ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᔾᔨᒌᓐᖑᐊᖅᑎᑕᐅᔪᑦ ᐊᒡᓂᖏᓐᓂ 1990−ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂ ᕿᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᖏᑦ ᓈᒻᒪᒃᑲᓗᐊᕐᒪᖔᑕ. ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᕙᓚᐅᖅᑐᑦ ᕿᖓᓯᕈᓕᖅᑕᐅᓪᓗᑎ BB ᓇᓄᖏᓐᓂᑦ ᐊᒻᒪ ᐊᔾᔨᒌᖏᑦᑐᓂ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᑖᒃᑯᓂᖓ ᒪᕐᕉᔫᓂᒃ (1990−ᖏᓐᓂ ᐅᑭᐊᔅᓵᒃᑯᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐊᒻᒪ ᐅᐱᕐᖔᒃᑯᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ; 2000−ᖏᓐᓂ ᐅᐱᕐᖔᒃᑯᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᓯᑯᖓᓂ) ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᓯᕗᓪᓕᖅᐸᖑᓚᐅᖅᑐᖅ ᖃᐅᔨᓴᕐᓗᒋᑦ ᓇᓄᐃᑦ ᐱᕙᐅᖁᓚᐅᖅᑐᑦ ᐅᐱᕐᖔᒃᑯᑦ ᑕᒪᔅᓯᒥᖓᔅᓴᐃᓐᓇᖅ ᓇᔪᖅᓯᒪᖔᑕ ᐱᕙᐅᓚᐅᖅᑐᓂᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ. ᖃᓄᐃᓐᓂᕆᕙᖏᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ 1990−ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂ ᕿᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕌᑦ ᓈᒻᒪᓂᖅᑐᑎᑦ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐊᔾᔨᒌᓐᖑᐊᑎᑦᑐᒋᑦ ᐊᒥᓲᓂᖏᑦ ᓇᓗᓇᐃᕝᑦᓯᒪᓂᖏᑦ ᐊᕿᒃᒍᓕᓚᐅᓕᖅᑐᓂᑦ' ᖃᓂᒋᔭᐸᓗᖏᓐᓂ 92% ᐃᓐᓇᐅᔪᓂᑦ ᐊᕐᓇᓂᑦ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐅᐱᕐᖔᑯᑦ ᑖᔅᓯᒥᖓᔅᓴᐃᓐᓇᖅ ᓇᔪᖅᓯᔪᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐃᓐᓇᖏᑦ ᐊᕐᓇᐃᑦ ᐱᕙᐅᔪᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ 1990−ᖏᓐᓂ.

ᖃᐅᔨᓴᓂᖃᓕᓚᐅᕐᒥᔪᒍᑦ ᐊᑐᖅᑐᑕ ᖃᓄᐃᓕᖓᓂᐅᓯᒪᖃᑦᑕᖅᑐᑐᖃᕐᓂᑦ ᐊᒻᒪ ᒫᓇᕋᑖᖑᔪᒥ ᕿᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᖃᐅᔨᓴᖅᑐᒋᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᕿᓕᓂᒃ ᐊᕐᕋᒍᓯᒪᔪᓂᑦ ᒪᕐᕉᓐᓂ ᐊᑐᖅᑕᓕᒫᖑᔪᓂᑦ (ᓲᕐᓗ, ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᖏᓂᖏᑦ) ᐅᖓᓯᓐᓂᖓ ᕿᓕᖑᖃᑦᑕᐅᑎᓂᖏᑦ, ᐊᓯᔾᔨᕐᓂᖏᑦ ᓇᒦᓐᓂᖏᑦ ᐊᕐᕋᒍᓚᓐᓄᑦ, ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐅᑎᖅᑕᕝᕕᖏᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᑎᖅᑕᖅᑐᓂ ᑭᓪᓕᖏᑎᒍᑦ. ᖃᐅᔨᓴᕈᑎᒦᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᖏᔪᒥ ᒥᑭᓪᓕᒋᐊᖅᓯᒪᔪᑦ 2000−ᖏᓐᓂᑦ BB 95% ᐅᖓᓯᓐᓂᖓ (ᓲᕐᓗ, ᐊᑐᖅᑕᐅᔪᑦ ᕿᖓᓯᕈᖅᑕᐅᓯᒪᔪᓄᑦ ᓇᓄᕐᓄᑦ) ᑕᖅᑭᓕᒫᖏᓐᓂᑦ ᐊᒻᒪ ᐊᕐᕋᒍ ᐃᓗᐊᓂ ᐊᔾᔨᒋᓐᖑᐊᖅᑐᒍ 1990−ᖏᓐᓂᑦ. ᖃᑦᑏᓐᓇᕈᓇᐊᖅᓯᒪᓛᖑᔪᑦ 60% ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᖏᓂᖏᑦ ᐊᐅᔭᒃᑯᑦ. ᕿᓕᓃᖃᑦᑕᐅᑎᓂᐅᔪᑦ 1990ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ BB ᐅᖓᓯᓐᓂᓇᓚᐅᖅᑕᖏᑦ < 50% ᑕᖅᑭᓕᒫᓂᑦ, ᑕᑯᔅᓴᐅᑎᑦᑎᔫᖅ ᑕᒪᒃᑭᓐᓂ ᐊᖏᓪᓕᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᓴᖑᖃᑦᑕᕐᓂᖏᑦ BB−ᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ 2000−ᖏᓐᓂᑦ. ᑖᒃᑯᐊ ᓴᖑᓂᐅᔪᑦ ᐊᑦᑐᐊᓂᓐᓂᑦ ᓯᖁᖃᕈᓐᓃᕐᓂᖓᓄᑦ ᐊᕿᒍᑕᓖᑦ ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂᑦ ᓯᑯᐃᖃᑦᑕᕐᓂᖓᓄᑦ, ᐊᓯᔾᔩᔪᖅ ᐅᖓᓯᓐᓂᖏᓐᓄᑦ Bbu ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᓴᖑᑎᑦᑎᓪᓗᑎ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ BB−ᒥ ᓇᓄᖏᑦ ᐅᐊᓐᓇᖅᐸᓯᐊᓄ ᐊᕿᒍ ᐃᓗᐊᓂ. ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᑭᓪᓕᖏᑦ, BB ᓇᓄᖏᑦ 2000−ᖏᓐᓂᑦ ᖃᑦᑏᓐᓇᐅᓂᖅᓴᐅᓚᐅᖅᑐᑦ BB−ᒥᖑᓐᓃᕐᓂᖏᓐᓄᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂ. ᐱᓗᐊᖅᑐᒥ, ᖃᑦᑏᓐᓇᕈᓇᐊᓚᐅᖅᑐᑦ ᕿᖓᓯᕈᖅᓯᒪᔪᑦ ᕿᑭᖅᑖᓗᑉ ᐃᒪᖓᓂ ᐊᒻᒪ ᑕᓪᓗᕈᑎᐅᑉ ᑕᓇᐅᖓᓂ BB−ᒥ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᓯᑯᖃᖏᓐᓂᖓᓄᑦ, ᑖᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅ BB−ᒥ ᐊᒥᓲᓂᖏᑦ ᑲᑎᓐᖓᔭᐅᒍᓐᓃᖅᑐᑦ, ᖃᓄᐃᓘᖃᑦᑕᐅᑎᖏᓂᖅᓴᐅᓪᓗᑎ ᐊᒻᒪ ᐊᓯᖏᓐᓂ ᓇᓄᕐᓂᑦ.

KB−ᒥᑦ, ᕿᕝᕙᓯᓐᓂᕆᓚᔪᑦᑕᖓ 95% ᐅᖓᓯᓐᓂᖏᑦ ᐊᖏᓪᓕᒋᐊᖅᓯᒪᔪᑦ 1990−ᖏᓐᓂ. ᐊᖏᓪᓕᒋᐊᒃᓂᖓ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᑐᖅᑕᐅᔪᑦ 2000−ᖏᓐᓂ ᐊᕿᒍᓕᒫᖑᕙᑦᑐᒥᑦ, ᑭᓯᐊᓂᓕ ᐊᒥᓲᓂᖅᓴᐅᔪᑦ ᐊᐅᔭᒃᑯᑐᐊᖅ (ᔫᓂ-ᓯᑎᐱᕆ), ᓇᒧᖓᕐᓂᕆᕙᑦᑕᖏᑦ ᒪᕈᐊᑎᖅᓗᐊᖅᓯᒪᔪᑦ ᐊᒡᓂᖏᓐᓂ 1990−ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ ᓇᒧᖓᓂᕆᕙᑦᑕᖏᑦ ᐊᖏᓪᓕᒋᐊᕈᑎᒋᔭᖓ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᓯᑯᖓ, KB ᐊᕿᒍᑕᓖᑦ ᓯᑯᖃᑦᑕᖏᓐᓂᖅᓴᐅᕙᓪᓕᐊᓕᖅᑐᖅ (BB-ᑎᑐᑦ) ᓯᑯᓕᒫᐸᓗᖓ ᐊᐅᖃᑦᑕᖅᑦᑐᓂ ᐊᐅᔭᑕᒫᑦ. ᑖᒃᑯᐊ ᖃᐅᔨᔭᐅᔪᑦ ᐊᓯᔾᔨᖅᑐᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᓚᔪᑦᑐᒃ ᐊ−ᔾᔨᒌᖏᓐᓂᖏᓐᓂᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᓄᑦ ᓇᓄᕐᓄᑦ ᕿᑭᖅᑕᓂᑦ ᓴᓂᐊᓂ

ᓯᑯᐃᖅᑕᕐᓂᖓᓄᑦ: ᓇᓄᐃᑦ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ ᓯᑯᒥ ᐊᖏᓂᖅᓴᒥᑦ ᐊᒻᒪ ᐊᔾᔨᒌᖏᓂᖅᓴᓂᑦ ᐊᖏᕋᖃᖅᑐᑦ ᓯᑯᒃᑯᑲᐃᓐᓇᐸᓪᓕᑕ ᓇᓄᐃᑦ ᕿᑭᖅᑕᓚᓐᓃᑦᑐᑦ ᒥᑭᓂᖅᓴᓂᑦ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ ᐊᓯᔾᔮᓂᖃᖏᓂᖅᓴᐅᓪᓗᑎ.

ᓱᓕ ᐊᖏᔪᒥᑦ ᖃᖓᒎᕈᑎᖃᖅᑐᑦ ᐊᕿᒍ ᐃᓗᐊᓂ KB ᓇᓄᖏᓐᓂ 1990−ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂ (50-98%

ᖁᓕᓈᖃᑦᑕᐅᑎᔪᑦ ᐊᕐᕌᒍᒐᓚᓐᓂᑦ), ᓇᓗᓇᐃᖅᓯᔪᑦ ᓇᓄᐃᑦ ᑕᒪᔅᓯᒥᙵᔅᓴᐃᖅ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ KB−ᒥ. ᐊᖏᔪᒃᑲᓪᓚᒻᒥᑦ ᐅᐊᓐᓇᙵᓅᓚᐅᖅᑐᑦ KB ᓇᓄᖏᑦ ᓇᒦᓐᓂᖏᑦ 2000−ᖏᓐᓂᑦ ᐅᐱᕐᙲᒃᑯᑦ ᐊᒻᒪ ᐊᐅᔾᔭᒃᑯᑦ (ᒥᑭᓂᖅᓴᐅᔪᑦ BB−ᒥ) ᑭᓯᐊᓂ ᐊᔾᔨᒌᖏᓪᓂᖏᑦ ᖁᕝᕙᓇᐊᖅᓯᒪᔪᑦ. ᐊᓯᔾᔨᕐᓂᖅᑲᖏᑦᑐᖅ ᐅᑭᐅᖑᑎᓪᓗᒍ. ᑖᒃᑯᐊ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᐊᓯᔾᔨᓚᐅᖏᑦᑐᑦ ᐊᑦᑐᖅᑕᐅᓴᕋᐃᓐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᓇᓄᓇᐃᑦ ᐱᔭᐅᑎᓪᓗᒋᑦ ᑲᓇᑕᓐᓇᙵᓂ KB ᐱᖃᓯᐅᔾᔭᐅᓚᐅᖏᑦᑐᑦ 2000−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, ᐊᔾᔨᒋᓕᖅᑎᓪᓗᒍ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᐱᔭᐅᔪᑦ 1990−ᖏᓐᓂ. ᑕᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅ ᖃᐅᔨᔭᓪᓗᐊᑕᐅᔪᑦ KB−ᒧᑦ ᐊᔭᐅᖅᑕᐅᓚᐅᖏᑦᑐᑦ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐱᔭᐅᔪᑦ ᓇᒦᓐᓂᖏᑦ..

ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᐃᓂᖃᓚᐅᖅᑐᑦ ᐊᑐᖅᑐᑎ 8 polymorphic microsatellites ᐃᓚᒋᓪᓗᓂᐅᒃ ᐊᖏᔪᓂᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᐅᖏᑎᒍᑦ ᑐᑭᓯᒋᐊᕈᑎᑦ ᓄᐊᑕᐅᔪᑦ MR ᖃᐅᔨᓴᐃᓂᐅᔪᓄᑦ (ᓵᑉᑕ 5 ᐊᒻᒪ 10). ᐊᑕᖏᖅᑐᒥᑦ, ᐊᑦᑎᑦᑑᓚᐅᖅᑐᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᒥᓱᖏᓐᓂᖏᓐᓄᑦ ᒥᑭᔪᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᑦ. BB ᐊᒻᒪ KB ᓇᓄᖏᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᔭᐅᔪᓐᓇᓚᐅᖏᑦᑐᑦ. ᐊᒻᒪᓗᑦᑕᐅᖅ ᒪᓕᑦᑐᒍ ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᕐᓂᐅᓯᒪᔪᓂᑦ, BB ᖃᒻᒪ KB ᓇᓄᖏᑦᐊᐅᖏᑎᒍᐳᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᖅᑐᑦ ᓇᓄᖏᓐᓂ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ ᕿᑭᖅᑖᓗᑉ ᑕᕆᐅᖓᓂ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓅᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᔭᐃᓂᕐᒥᑦ BB−ᒧᑦ ᐊᒻᒪ KB−ᒧ ᐊᓯᔾᔨᖅᓯᒪᖏᑦᑐᑦ.

ᐱᔭᐅᒃᑲᓂᖅᑐᑦ ᐊᒻᒪ ᐊᖑᓇᓯᑦᑕᐅᔪᓂᑦ ᑐᑭᓯᒋᐊᖃᕈᑏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐱᙵᓱᓂᑦ ᐊᕐᕌᒍᓂ MR ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ BB−ᒥᑦ ᐊᒻᒪ KB ᐊᑦᑎᑦᑑᓚᐅᖅᑐᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᑦ ᖁᑎᕐᓂᖏᓪᓘᓐᓂᑦ ᓇᓄᐃᑦ ᓯᓚᑖᓂ ᐊᒥᓲᓂᖏᓐᓂᑦ ᓇᑭᓐᖕᒎᕐᕕᑎᑎᐊᓐᖓᖅᓯᒪᔭᖏᓐᓂᑦ, ᑲᑎᑦᑐᒋᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᑦ ᐊᕐᕌᒍᖏᓐᓂ 1 ᐊᒻᒪ 2 ᑖᒃᑯᓂᖓ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ (2011−2012 BB−ᒥᑦ ᐊᒻᒪ 2012-2013 KB−ᒥᑦ) ᐊᔾᔨᐸᓗᒋᓚᐅᖅᑕᖏᑦ ~34% ᐊᒻᒪ ~25% ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᐊᒥᓱᓂᖏᓐᓂᑦ ᐊᖏᓂᖏᑦ BB ᐊᒻᒪ KB, ᑐᖏᓕᓇᑦᑐᑎ. ᑖᓐᓇ ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ, ᐅᑎᖅᑕᖃᑦᑕᖅᕕᒋᔭᖏᑦ ≤ 1% ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᐊᒻᒪ ᐅᑎᖅᑐᓂᑦ BB ᓇᓄᖏᓐᓄᑦ. ᐊᔾᔨᐸᓗᙵᓗ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ, ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐅᑎᖅᑕᒃᑕᕐᓂᖏᑦ < 4% ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᐊᒻᒪ ᐅᑎᖅᑐᓂᑦ. ᑖᓐᓇ ᒪᓕᓐᓂᖅᑲᕐᒥᔪᖅ ᓇᒧᙵᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᑐᓐᖓᓂᖅᑲᖅᑐᑎᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐅᑎᖅᑕᖃᑦᓴᑦᑕᕐᓂᖏᑦ BB ᐊᖏᔪᒃᑲᓪᓚᒻᒥᑦ ᖃᑦᑑᓐᓇᕈᓇᐊᖅᓯᒪᔪᑦ 1990−ᖏᓂ, ᐱᔾᔪᑎᖃᓗᐊᖅᑐᖅ ᓯᑯᖃᓗᐊᕈᓐᓃᕐᓂᖓᓄᑦ ᐅᑭᐅᒃᑯᑦ ᐊᒻᒪ ᓯᑯᑕᖃᖏᓐᓂᖓᓄᑦ ᐊᐅᔭᒃᑯᑦ.

ᐊᑕᖏᖅᑐᒍ, ᖃᐅᔨᔭᐅᔪᑦ ᑐᓐᖓᓂᖅᑲᖅᑐᑎᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ, ᓇᒧᙵᖃᑦᑕᕐᓂᖏᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ, ᐊᒻᒪ ᐊᒥᓱᓂᖏᑦ ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᒫᓇᐅᔪᒥ ᑭᒡᓕᒋᔭᖏᑦ BB−ᒧ ᐊᒻᒪ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ ᐊᑦᑐᐊᓂᖅᑑᔭᖅᑐᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔾᔪᑕᐅᔪᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ.

ᓵᑉᑕ 3: ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖅ 1990−ᖏᓐᓂ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᐃᓚᐃᓐᓇᐅᓂᖓᓄ ᐊᒻᒪ ᓈᒻᒪᓐᓂᖓᓄᑦ 2000−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ

ᓵᑉᑕ 3 ᖃᐅᔨᓴᐃᔪᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᒻᒪ ᑎᒥᖓᓃᖕᒎᖅᑐᓂᑦ ᐱᔾᔪᑕᐅᔪᓂᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᑕᒪᖓᑐᐃᓇᐅᖏᑦᑐᖅ ᐊᒻᒪ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐱᔭᓈᖅᑕᐅᓯᒪᖏᑦᑐᓂᑦ, ᓴᖅᑭᑎᑦᑎᔪᓐᓇᑐᐃᓐᓇᓇᐊᓖᑦ ᐃᓚᐃᓐᓇᙵᓂ ᒥᔅᓴᐅᓴᓐᓂᕐᒥᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐊᒻᒪ ᐊᒥᓱᓂᖏᓐᓄᑦ MR ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ BB ᓇᓄᖏᓐᓂᑦ.

ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ BB–ᒥᑦ 2011-2013–ᒥ ᐱᖃᓯᐅᔾᔨᓚᐅᖅᑐᖅ ᐸᕙᓴᐃᖏᑦᑐᓂ ᐊᐅᖏᑎᒍᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᒻᒪ ᐊᑐᖅᑐᑎ ᐆᒪᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᖁᖓᔪᓂᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᔾᔨᖓᑎᑐᑦ ᓴᓇᓯᒪᓂᖓᓂᑦ. ᑖᒃᑯᐊ ᑕᒪᒃᑭ ᖁᕝᕙᕆᐊᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᖏᑦ ᐊᖏᓂᖏᓐᓂᑦ ᐊᒻᒪ ᐊᑦᑎᓪᓕᑎᑦᑎᓪᓗᓂ ᖃᓄᐃᑦᑐᓂᖏᑦ ᐃᓚᐃᓇᖏᓐᓃᑎᑦᑎᒍᓐᓇᖅᑐᑦ.

ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ, ᐊᔅᓱᕈᕈᑕᐅᔪᑦ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ MR ᐊᑐᖅᑕᐅᔪᓐᓇᑐᓄᑦ BB ᑐᑭᓯᒋᐊᕈᑎᖏᓐᓄᑦ BB–ᒧᑦ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᒫᓐᓇᐅᔪᒥᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᑲᑎᙵᔪᓂᑦ 1993-2013 ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ. 1990–ᖏᓐᓂ ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖃᓚᐅᖅᑐᑦ BB ᑐᑭᓯᒋᐊᕈᑎᖏᓐᓂᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ, ᐊᔾᔨᒋᑎᖑᐊᖅᑐᒍ 2010–ᖏᓐᓂ ᑐᑭᓯᒋᐊᕈᑏᑦ, 1990–ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᒥᑭᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐊᖏᓂᖏᑦ, ᓄᓇᖓᓂ ᒥᑭᓂᖅᓴᓂ ᖃᐅᔨᓴᖅᑐᑎ, ᐊᒻᒪ ᐊᕙᑎᖓᓂ ᐊᔾᔨᒋᖏᑕᖓᓂ, ᑲᑎᙵᑎᓪᓗᒋᐱᑦ ᓴᖅᑭᑎᑦᑎᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᖅᓯᓪᓚᕆᖏᑦᑐᓂᑦ ᐊᒻᒪ ᑭᓪᓕᖃᖅᑎᑦᑎᑐᐃᓐᓇᕆᐊᖃᕐᓂᖓᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, **ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᑖᒃᑯᐊ ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᕐᒧᑦ** (ᓵᑉᑕ 3) ᑕᕝᕙᖓᑦᑕᐅᑎᒋ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᔾᔨᒌᙳᐊᖅᑎᒐᔅᓴᐅᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2010–ᖏᓐᓂᑦ BB–ᒥ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᐃᕐᓗᒋᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐆᒪᓂᖏᑦ ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ.

1990–ᖏᓐᓂ MR ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐊᖏᓂᖏᑦ (229 ᑲᑎᑦᑐᒋᑦ ᐱᔭᐅᔪᑦ ᖃᐅᔨᓴᕐᕕᒻᒧᑦ ᐊᕐᕌᒍᒧᑦ), ᐊᔾᔨᐸᓗᖓ 2010–ᖏᓐᓂ (470 ᑲᑎᑦᑐᒋᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᑐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ ᐊᕐᕌᒍᒥ), ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᑦ 1990–ᖏᓐᓂ ᐊᑦᑎᑦᑑᓚᐅᖅᑐᑦ. ᐆᒪᔪᓂᑦ MR ᖃᐅᔨᓴᕐᓂᖅᑐᐃᓐᓇᓚᐅᖅᑐᑦ ᐃᓚᖏᓐᓂ ᐊᕐᕌᒍᓂᑦ ᐊᑐᓂ ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ: 1993-1996 ᐊᒻᒪ 1997 ᓯ–ᔭᖓᓂ ᕿᑭᖅᑖᓗᒃ ᐊᒻᒪ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ, ᐊᒥᓱᓚᐅᓚᐅᖅᑐᑦ ᑐᖁᖓᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ MR ᖃᐅᔨᓴᓂᕐᒥᑦ (1998-2010), ᐱᓗᐊᖅᑐᒥ ᐊᕐᕌᒍᐃᑦ ᑭᖑᓂᐊᒍᑦ. ᒥᑭᔪᓂᑦ ᖃᐅᔨᓴᕋᔅᓴᓂᑦ ᐊᔪᕐᓇᑎᑦᑎᓚᐅᖅᑐᖅ ᒥᔅᓴᐅᓴᑦᑐᒋᑦ ᓄᓇᒥ ᐊᑐᖅᑕᖏᓐᓂ ᐊᒻᒪ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᑭᓐᓂᖅᑎᑦᑎᔪᖅ ᑕᒪᒃᑭᓐᓄ ᓴᖏᓂᖓᓄᑦ ᐱᔭᐅᔪᓐᓇᖅᑐᓂᒃ 1990–ᖏᓐᓂ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᒻᒪ ᓇᓗᓇᐃᕈᓐᓇᓂᑦᑎᓐᓄᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐊᑦᑎᓪᓕᕆᐊᖅᑐᒍ ᐃᓚᐃᓇᖏᓐᓂ ᒥᔅᓴᐅᓴᓐᓂᖅ.

ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐱᔭᐅᔪᑦ ᐊᒻᒪ ᖃᐅᔨᓴᕋᔅᓴᐃᑦ MR ᖃᐅᔨᓴᕐᓂᐅᔪᓄᑦ 1990–ᖏᓐᓂ ᐊᒻᒪ 2010–ᖏᓐᓂ ᐊᔾᔨᒌᖏᓐᓂᖃᕐᓗᐊᑲᓪᓚᓚᐅᖅᑐᑦ. 2010–ᖏᓐᓂᑦ, ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓄᓇᖅᐸᒻᒥ ᓯᔾᔭᖏᓐᓂ ᐊᒻᒪ ᑲᖏᖅᑐᓂᑦ ᕿᑭᖅᑖᓗᒻᒥ. ᑖᓐᓇ ᐊᔾᔨᒌᖏᓐᓂᖓ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐱᔭᐅᔪᓂᑦ ᐊᑯᓐᓂᖏᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᐊᓯᔾᔨᕐᓂᖏᓐᓃᖏᑦᑐᖅ ᓇᔪᖃᑦᑕᖅᑕᖏᓐᓂᑦ. ᖃᐅᔨᓴᐃᓂᖅ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᕙᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᐃᓐᓇᓂ ᐊᕐᓇᓂᑦ, ᖃᐅᔨᓴᐃᒍᓐᓇᑎᑦᑎᔪᖅ ᓄᓇᒥ ᐊᑐᖅᑕᐅᕙᑦᑐᓂᑦ ᐊᕐᕌᒍᒐᓚᓐᓄᑦ ᐃᓛᒃᑰᖅᑐᖅ ᓴᓂᐊᓂ ᐱᔭᐅᔪᓂᑦ ᓇᒦᓐᓂᖏᓐᓂᑦ, ᑕᑯᔅᓴᐅᑎᑦᑎᓚᐅᖏᑦᑐᑦ ᐅᖓᓯᓐᓂᖏᓐᓂᑦ ᓄᓇᒥ ᐊᒻᒪ ᖁᕝᕙᓯᓐᓂᖓᓂᑦ ᓇᓄᕐᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990s ᐊᒻᒪ 2010s. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐱᔭᐅᔪᓂᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐊᐅᓚᓂᖃᓚᐅᖅᑐᑦ ᐊᔾᔨᒋᖏᑕᖓᓂ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ, ᓄᓇᖅᐸᓯᒻᒦᖏᓂᖅᓴᐅᓪᓗᑎ ᐊᒻᒪ ᑲᖏᖅᑐᓂᑦ 1990–ᖏᓐᓂ.

ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᓄᓇᕗᒻᒥ 1990–ᖏᓐᓂ ᑐᖔᓃᓚᐅᖅᑐᑦ ½ ᖃᐅᔨᓴᕐᕕᐅᔪᓂᑦ 2010–ᖏᓐᓂ. 2010–ᖏᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᑲᖏᖅᑑᑕᐅᓂᖅᓴᐃᑦ ᓯᔾᔭᖓᓂ ᐊᒻᒪ ᓄᓇᖅᐸᒥᐅᓂᖅᓴᓂ ᓇᔪᖅᑕᐅᔪᓂᑦ, ᖃᐅᔨᓴᕐᓗᒋᑦ ᐃᓚᖏᓈᕈᓐᓇᖅᑐᑦ ᐊᑦᑐᐊᓂᓰᑦ ᒥᑭᓂᖅᓴᓂᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ 1990–ᖏᓐᓂ, MR ᖃᐅᔨᓴᓂᖅ ᐊᒻᒪ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᐊᔾᔨᒌᙳᐊᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᒪᕐᕉᓐᓄ ᑐᑭᓯᒋᐊᕈᑎᓄᑦ (1) ᐊᑕᖏᖅᑐᒋᑦ 2010–ᖏᓐᓂ MR ᑐᑭᓯᒋᐊᕈᑏᑦ, ᐊᒻᒪ (2)ᓄᓇᖏᑎᒍᓪᓗ 2010–ᖏᓐᓂ MR ᑐᑭᓯᒋᐊᕈᑏᑦ ᐊᔾᔨᒌᙳᐊᖅᑎᒐᔅᓴᐅᔪᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ 1990–ᖏᓐᓂ (ᓵᑉᑕ 5).

ᐊᑦᑐᐊᓂᓖᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐊᕐᕌᒍᓚᐅᔪᓂᑦ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓇᖅᑐᒥ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ. 1990–ᖏᓐᓂ ᐊᒥᓲᑲᐃᓐᓇᐸᓪᓚᐃᓚᐅᖅᑐᑦ ᐅᑎᖅᑕᖃᑦᑕᖅᑐᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓇᓄᐃᑦ ᓯᑯᒥ ᐊᑐᓚᐅᖅᑐᑦ ᐃᒪᐃᓐᓇᒦᑦᑐᒥ ᐅᕝᕙᓘᓐᓃᑦ ᓯᑯᓂᑦ ᕿᑭᖅᑕᓚᓚᓐᓂᑦ

ᐊᐅᔭᒃᑯᑦ. ᑕᒪᓐᓇ ᑕᑯᔅᓴᐅᑎᑕᐅᓚᐅᖅᑐᖅ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑕᒪᑭᒃ ᖃᐅᑕᒫᑦ ᐃᑳᕐᕕᐅᔪᑦ ᓯᑯᕋᐃᑦ ᑐᑭᓯᑎᐊᕈᑎᖏᑦ. ᓯᑯᖓ ᐊᖏᔪᒥ ᒥᑭᓪᓕᕆᐊᓚᐅᖅᑐᖅ 2010−ᖏᓐᓂᑦ, ᑖᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᓯᑯᖃᕈᓐᓃᖅᑐᓂ ᐋᓪᓖᓯᒥ ᐊᒻᒪ ᐊᖅᑯᑎᐅᕙᑦᑐᑦ ᐊᑦᑎᓂᖅᓴᕐᓗᐊᑲᓪᓚᐅᓚᐅᖅᑐᖅ. ᐊᑐᓂ ᐊᕐᕌᒍᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᐊᑯᓐᓂᖓᓂ 1993 ᐊᒻᒪ 2013, ᐊᒥᓱᓂᖏᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ ᐊᑕᐅᓯᒻᒧᑰᑦ ᐃᓐᓇᖏ ᐊᒃᓇᐃᑎ ᖃᐅᔨᓴᕐᕕᒦᑦᑐᑦ ᐱᖃᓯᐅᔾᔭᐅᓚᐅᖅᑐᑦ. 1990−ᖏᓐᓂ, ᖃᓂᒋᔭᐸᓗᖏᓐᓂ <30% ᖁᖓᓯᕈᖅᓯᒪᔪᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑰᖓᔪᓂᑦ ᐊᕐᓇᓂᑦ ᓇᓄᕐᓂᑦ ᐃᓗᐊᓃᑦᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ MR−ᒥ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ. ᑖᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅᐊᒥᒌᓂᖅᓴᕆᔭᖏᑦ ᓇᓄᐃᑦ ᐊᑐᐃᓐᓇᐅᓚᐅᖏᑦᑐᑦ ᐱᔭᐅᓗᑎ ᐃᓚᖏᓐᓂᑦ ᐊᕐᕌᒍᖏᓐᓂᑦ 1990−ᖏᓐᓂᑦ. ᑕᑯᓐᓇᖅᑐᒍ 2010−ᖏᓐᓂᑦ, ᐊᑯᓐᓂᖓᓂ 70-80% ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᑦ ᕐᓇᖏᑦ ᐊᕐᓇᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᕙᒻᒥᓚᐅᖅᑐᑦ.

ᕿᑎᖓᖅᑐᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᑕᒫᙵᑐᐃᓐᓇᖅ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᒥᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ ᐃᓚᐃᓐᓇᖏᓐᓃᖅᓯᒪᓕᕆᐊᖃᖏᑦᑐᑦ ᔪᓇᙵᓄᑦ ᑭᓪᓕᖏᓐᓄᑦ. ᑭᓯᐊᓂᓕ, ᐊᖏᓂᖓ ᐊᖅᑯᑎᐅᑲᐃᓐᓇᖅᑐᖅ 1990−ᖏᓐᓂ ᐊᔾᔨᒌᖏᑦᑑᔭᖅᑐᑦ ᐊᒻᒪ ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᐊᕙᑎᖓᓄ ᖃᓄᐃᓐᓂᖓᓄᑦ; ᐊᒻᒪ ᒥᑭᔩᑎᑦ ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐊᔪᕐᓇᑎᑦᑎᓚᐅᖅᑐᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᓗᑎ. ᐊᖁᑎᒋᔭᐅᒃᑲᓐᓂᑲᐃᓐᓇᖅᑐᑦ 1990−ᖏᓐᓂ ᑕᒫᙵᑐᐃᓈᖅᓯᒪᓚᐅᖏᑦᑐᑦ ᐊᒻᒪ ᐊᑦᑐᐊᓂᖃᖅᑐᑎᑦ ᕿᑐᙵᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᓇᔾᔪᓚᐅᖅᑐᑦ ᐅᑭᐊᔅᓵᒃᑯᑦ ᔪᓇᖅᐸᒨᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᕿᑭᖅᑖᓗᒻᒥ (ᓲᕐᓗ, ᓇᔪᖅᓴᓯᐅᖅᑐᑎᑦ ᑎᓯᒋᓂᐊᖅᑕᖏᓐᓂᑦ), ᓴᓂᐊᓂ ᓇᔾᔨᖏᑦᑐᓂᑦ, ᖃᐅᔨᓴᖅᑕᐅᑎᐊᖃᑦᑕᖏᓐᓂᖏᓐᓄᑦ ᐊᕐᓇᐃᑦ ᐃᓐᓇᐃᑦ ᔪᓇᖅᐸᒨᖃᑦᑕᖅᑐᑦ ᐃᓚᖏᓐᓂᑦ ᐊᕐᕌᒍᖏᓐᓂᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᖃᑦᑕᖏᓐᓂᖏᓐᓄᑦ ᔪᓇᒦᑦᑐᓂᑦ.

ᒪᓕᓐᓂᖃᖅᑐᖅ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᓂᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010-ᖏᓐᓂᑦ, ᐊᖏᔪᒥᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖅᑐᑦ ᐃᓗᓕᖏᓐᓂᑦ MR ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ (ᓲᕐᓗ, ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᐊᕐᓇᐅᓂᖏᓐᓄᑦ ᐊᖑᑕᐅᓂᖏᓐᓄᑦ) ᑖᒃᑯᓂᖓ ᒪᕐᕉᔪᓐᓂᒃ. ᐱᓗᐊᖅᑐᒥ, ᐃᓐᓇᖏᑦ ᐊᕐᓇᐃᑦ ᐱᑕᖃᖏᓐᓂᖅᓴᐅᔫᖅᑐᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ.

ᐊᔅᓱᕈᕈᑎᑕᖃᓚᐅᕐᒥᔪᖅ 1990−ᖏᓐᓂ MR ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓇᓗᓇᕈᑎᑕᖃᓚᐅᖅᑐᖅ ᓇᓗᓇᐃᖅᑕᐅᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᓇᒦᓐᓂᖏᓐᓄᑦ ᖁᖓᓯᒍᖏᑎᒍᑦ ᓴᓂᐊᓂ ᖃᐅᔨᔭᐅᔪᓂᑦ ᕿᓂᕐᓂᑐᐃᓐᓇᒃᑯᑦ (ᓲᕐᓗ, ᑕᑯᔭᐅᑳᓪᓚᑦᑐᑦ) ᐅᐱᓐᓇᕋᓂ ᐱᔭᐅᓯᒪᔪᓂᑦ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᑖᔅᓯᒥᙵ ᓇᓗᓇᐃᖅᓯᓂᖃᓚᐅᖅᑐᖅ. ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᐃᑲᔪᖅᑕᐅᓪᓗᑎ ᐱᔭᐅᔪᑦ ᑕᐃᒪᐃᓐᓂᖓᓄ ᐊᔅᓯᕆᑎᖑᐊᖅᑐᒍ ᐊᑐᐃᓐᓇᐅᔪᓄᑦ ᑐᑭᓯᑎᐊᕈᑎᓂᑦ ᐱᓗᓯᑦ ᑎᑎᖅᑕᐅᓯᒪᔪᑦ ᔪᐊᑕᐅᓯᒪᔪᑦ 1990−ᖏᓐᓂ ᖃᐅᔨᓴᖅᑕᐅᓂᐊᖅᖢᐅᑕᐅᔪᓂᑦs BB−ᒧ ᖃᐅᔨᓴᐃᓂᖅ ᑕᐃᒪ ᐃᓚᖏᓪᓗ. (2005). ᑖᓐᓇ ᓇᓗᓇᕈᑎᐅᔪᖅ ᐃᓚᐃᓐᓇᖏᓐᓃᖅᑎᑦᑎᔪᓐᓇᖅᑐᖅ, ᐱᔪᑎᒋᓪᓗᒍ ᖃᐅᔨᒪᓪᓗᒋᑦ ᓇᓪᓕᐊ ᓇᓄᐃᑦ ᓇᒦᓚᐅᕐᒪᖔᑕ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᐱᓪᒪᕆᐅᓚᐅᖅᑐᖅ ᐊᒥᒌᓂᖅᓴᐅᓂ ᐃᑲᔪᕐᓂᖅᑕᑲᐅᔪᑦ MR ᐊᑐᖅᑕᐅᔪᓂᑦ 1990−ᖏᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ.

ᐊᑕᖏᖅᑐᒋᑦ, ᐊᔾᔨᐸᓗᖏᑦ 2010−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ, 1990−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᒥᑭᔩᑕᐅᓂᖅᓴᓂᑦ ᖃᐅᔨᓴᖅᓴᓄᑦ, ᐱᔭᕇᖅᓯᒪᖏᑦᑐᑦ ᔪᓇᖓᑎᒍᑦ ᖃᐅᔨᓴᕐᓂᖅ, ᐊᒥᓱᕈᖅᑎᑦᑎᑲᐃᓐᓇᖅᑐᖅ ᐊᖁᑎᒋᔭᐅᑲᐃᓐᓇᖅᑐᓂᑦ ᓇᓄᕐᓄᑦ ᓯᑯᒦᖏᓐᓇᓚᐅᖅᑐᓄᑦ ᐊᐅᔭᒃᑯᑦ, ᐊᒻᒪ ᑕᒫᙵᑐᐃᓐᓇᖅ ᐃᖃᖅᑐᑦ ᐃᓐᓇᕐᓄᑦ ᐊᕐᓇᓄᑦ ᔪᓇᒨᕐᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᑎᓯᖏᓐᓄᑦ. ᑖᒃᑯᐊ ᐃᓚᐃᓐᓇᖏᓐᓂᑎᑦᑐᐃᓐᓇᕆᐊᖃᕐᓇᖅᑐᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᐆᒪᓂᕐᒧᑦ ᐊᒻᒪ ᐊᒥᒌᓐᖏᓄᑦ 1990−ᖏᓐᓂ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᔪᓇᖓᑎᒍᑦ ᑭᓪᓕᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010-ᖏᓐᓂᑦ BB ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᑕᕝᕙᖓᑦᑕᐅᑎᑭ ᐊᔾᔨᖑᐊᖅᑎᒋᔅᓴᐅᖏᑦᑐᑦ, ᑭᓪᓕᖅᑎᑦᑎᔪᖅ ᖃᐅᔨᓴᐃᔪᓇᓂᒥᑦ ᐊᒥᒌᓐᖏᓄᑦ ᐊᓯᔾᔨᖅᑕᖅᑐᓂᑦ.

ᓵᑉᑕ 4: ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᓇᔪᖅᑕᐅᔪᑦ ᐊᑐᖅᑕᐅᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᓵᑉᑕ 4 ᖃᐅᔨᓴᖅᑐᖅ ᐊᓯᔾᔨᖅᑐᓂᑦ BB–ᒥ ᓯᑯᒥ ᓇᔪᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂᑦ BB–ᒥᑦ ᓇᓄᕐᓄᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2010-ᖏᓐᓂᑦ (2009-2015). ᖃᐅᔨᓴᕈᑎᒦᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᑕ–ᕗᖓᑦᑕᐅᑎᒋ ᑐᔾᔨᕌᖑᔪᓂᑦ JC–ᒧᑦ (ᖃᐅᔨᓴᕐᓂᖅ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᕐᓂᖏᓐᓄᑦ ᓯᑯᖏᓐᓂᑦ) ᐊᒻᒪᓗᑦᑕᐅᖅ ᐱᒻᒪᕆᐅᔪᖅ MR–ᒧ ᖃᐅᔨᓴᓂᕐᒧᑦ, ᓲᕐᓗ ᖃᓄᖅ ᐊᕙᑎ ᐊᓯᔾᔨᕐᓂᖓ ᓇᓗᓇᐃᖅᓯᔪᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ. ᖃᐅᔨᓴᕈᑎᒦᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᖃᐅᔨᓴᖅᑎᓃᖕᒫᖅᑐᓂᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᔾᔨᒌᓐᓂᖑᐊᖅᑎᑕᐅᓗᑎ ᑕᑯᔭᐅᔪᓂᑦ ᓄᓇᓕᒻᒥ ᖃᐅᔨᒪᔭᐅᔪᓂᑦ (LEK) ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᖃᐅᔨᔭᐅᔪᑦ ᐊᒥᓲᕈᓇᐊᕐᓂᖏᑦ ᓇᓄᐃᑦ ᓄᓇᒥ ᓯᕐᔭᖅᐸᓯᐅᓂᖅᓴᒥᓪᓘᓐᓃᑦ ᐊᒻᒪ ᓄᓇᓕᓐᓂᑦ.

ᓯᑯᖓᓂ ᓇᔪᖅᑕᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ BB–ᒧᑦ ᐱᒋᐊᖅᓯᒪᓪᓗᓂ 1979–ᒥᑦ, ᐱᒋᐊᕐᓂᖃᓕᖅᑎᓪᓗᒍ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑎᑎᕋᖅᑕᐅᓂᖏᑦ, ᐊᒻᒪ ᑕᔪᓯᓪᓗᓂ 2015–ᒧ ᐊᑐᖅᑐᑎ ᖃᐅᔨᓴᐅᑎᓂᑦ (ᖃᐅᑕᒫᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᖅ ᓯᑯᖓᓂ). ᓯᑯᖓᓂ ᓇᔪᖅᑕᐅᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ BB–ᒥᑦ ᐊᖏᔪᒥᑦ ᒥᑭᓪᓕᒋᐊᖅᓯᒪᔪᖅ ᑕᕝᕙᓂ ᐊᑯᓂᐅᓂᖓᓂ, ᐱᓗᐊᖅᑐᒥ ᐱᒋᐊᖅᑐᓂ ᕿᑎᖏᓐᓂ 1990–ᖏᓐᓂᑦ. ᐊᑯᓂᐅᓂᖓ ᐊᐅᔭᖓ (ᓲᕐᓗ, ᐅᓪᓗᐃᑦ ᐊᒥᓲᓂᖏᑦ ᓯᑯᐃᖃᑦᑕᕐᓂᖓ ᐅᐱᕐᖓᓵᒃᑯᑦ ᐅᑭᐊᔅᓵᒧᑦ) ᐊᒥᓲᕈᓇᐊᖅᓯᒪᔪᑦ ᐅᓪᓗᐃᑦ/ᐊᕐᕌᒍᐳᐃᑦ ᖁᓖᑦ 1979–ᒥᓂᑦ. ᓯᖁᖃᕐᓂᓇᒐᔪᓚᐅᖅᑕᖓ ᔫᓐᓂ–ᐅᑦᑑᐱᕆᒧ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᔪᖅ 4% ᖁᓕᓂᑕᒫᑦ ᐊᕐᕌᒍᓂᑦ. ᐊᐅᓐᓂᓇᒐᔪᖅᑕᖅᑕᖓ ᐊᓯᔾᔨᖅᓯᒪᖏᑦᑐᖅ ᑭᓯᐊᓂ ᕿᓚᒥᐅᓂᖅᓴᐅᕙᓕᖅᑐᖅ 3-4 ᐱᓇᓱᐊᕈᓯᕐᓂᑦ 2000–ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990–ᖏᓐᓂᑦ. BB ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᖅᓯᒪᔪᖅ ᓯᑯᖓᓂ ᐊᑐᖅᑕᐅᕙᑦᑐᑦ ᓇᓄᕐᓄᑦ ᐊᒻᒪ ᑕᒪᓐᓇ ᓇᔪᖃᑦᑕᖅᑕᖏᓐᓂᑦ ᔭᒐᐃᑎᑦᑎᔪᖅ.

ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᐊᒥᓱᒐᓚᐃᑦ ᐊᓯᔾᔨᖅᑐᑦ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᓇᔪᖃᑦᑕᖅᑕᖏᑦ BB ᓇᓄᖏᑦ. ᓇᓄᖓᖃᑦᑕᕐᓂᖏᑦ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᓇᓄᐃᑦ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᓪᓗᐊᑲᓪᓚᑦᑐᑦ ᐃᒪᐃᓐᓇᐅᑎᓪᓗᒍ (ᐊᒡᒌᓯ–ᐅᑦᑑᐱᕆ) 2000–ᖏᓐᓂᑦ ᐃᐃᓐᓇᕐᒥ ᕿᑭᖅᑕᖏᓐᓂᓪᓗ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ. ᓇᓄᐃᑦ ᐃᒫᓃᕐᓯᔪᖏᑦᑐᑦ ᓄᓇᒦᖓᖅᑐᑎᑦ 2000–ᖏᓐᓂᑦ ᐃᒪᖓᓂ ᓯᑯᖃᑦᑎᐊᖏᓐᓇᕐᓂᖓᓂ ᐃᒪᐃᓐᓇᐅᑉᑕᑦᑎᓪᓗᒍ.

BB ᓇᓄᐃᑦ ᓯᑯᓂᑦ ᐊᑐᖏᓐᓂᖅᓴᐅᕙᓚᐅᖅᑐᑦ ᓯᑯᖃᖏᓐᓂᖅᓴᐅᓂᖓᓄᑦ ᐅᑭᐅᒃᑯᑦ ᐊᒻᒪ ᐅᐱᕐᖔᒃᑯᑦ 2000–ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990–ᖏᓐᓂᑦ. ᓇᓄᐃᑦ ᖃᓂᓴᕈᒪᓂᖅᓴᐅᓚᐅᖅᑐᑦ 300 m ᐃᑎᓂᓕᒻᒥᑦ (ᓯᑯᒥ ᐃᒪᖓᓂ ᐊᒻᒪ ᓄᓇᖅᐸᓯᒻᒥ) 2000–ᖏᓐᓂᑦ. ᓯᑯᖃᕐᓂᑐᐊᓇᖏᑕᖓ ᓇᓗᓇᐃᖅᓯᔪᑑᓚᐅᖏᑦᑐᑦ ᓇᔪᖅᑕᐅᔪᓂᑦ; ᐊᕐᓇᕐᓗᐃᑦ ᓂᕈᐊᖅᑕᐅᔪᑦ ᓯᑯᖃᖏᓐᓂᖅᓴᓄᑦ ᓯᑯᒨᑭᓐᓇᑎᑦᑎᓚᐅᖅᑲᑕ ᐃᒪᖏᓐᓄᑦ (<300 m). ᐊᕐᓇᕐᓗᐃᑦ ᓇᓄᐃᑦ ᓄᓇᒧᑦ ᖃᓂᔅᓴᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᑕᖅᑭᓖᓂᑦ 2000–ᖏᓐᓂᑦ ᑭᓯᐊᓂ ᓯᑯᐃᖅᑎᓪᓗᐳ (ᔫᓂ–ᔪᓚᐃ), ᐃᒪᐃᓐᓇᒥ ᓯᑯᒦᓚᐅᖅᑐᑦ ᐊᑯᓂᐅᒍᓐᓇᕐᓂᓕᒫᖏᓐᓂᑦ, ᓂᕿᔅᓴᓯᐅᖅᑐᑎᑦ.

ᐊᑯᓂᐅᔪᒥᑦ ᐳᐃᒥᑐᐃᓐᓇᓇᐊᖃᕐᓂᖏᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ, ᓇᓗᓇᐃᖅᑕᐅᔪᑦ ᐊᐅᔭᒃᑯᑦ ᖃᓄᐃᑑᖅᑕᐅᔪᑦ ᓇᓄᐃᑦ ᐃᖏᕋᓂᖃᖅᑐᑎᑦ >100 km ᐃᒪᐃᓐᓇᕐᒥᑦ ᓯᑯᒥᑦ, ᕿᑭᖅᑖᓗᒻᒧᑦ, <10%ᐟᒥ ᓯᑯᖃᕐᓂᖓᒍᑦ, ᐊᑕᐅᑦᑎᑯᓪᓗ ᐊᑦᑎᓪᓕᒋᐊᖅᓯᒪᔪᑦ ᐱᑕᖃᖏᑐᓪᓘᓐᓃᑦ ᖁᖓᓯᕈᖅᓯᒪᔪᑦ (ᓲᕐᓗ, ᖁᖓᓯᕈᐃᑦ ᖃᐅᔨᔭᔅᓴᐅᖅᐸᓪᓕᐊᑕ ᐃᒫᓃᑦᑎᓪᓗᒋᑦ). ᑖᒃᑯᐊ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᑕᒪᒃᑭᓐᓂ ᖁᓂᒡᒍᑎᓂᑦ ᐊᕐᕌᒍᓂᑦ, ᑭᓯᐊᓂ ᖁᕝᕙᓇᐊᓚᐅᖅᑐᑦ 2000–ᖏᓐᓂᑦ, ᐱᓗᐊᖅᑐᒥ 2011–ᒥᑦ.

ᐊᓯᔾᔨᖅᑐᑦ ᓄᓇᖓᓄᑦ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ ᕿᑭᖅᑖᓗᒻᒥ ᖃᐅᔨᔭᐅᖅᑐᑦ ᐊᒻᒪ ᓯᑯᐃᔅᓴᐃᓐᓂᖓᓗ ᐊᒻᒪ ᓯᒥᑯᓇᓵᕐᓂᖓᓄᑦ. 2000–ᖏᓐᓂᑦ, ᓇᓄᐃᑦ ᐊᑯᓂᐅᓂᖅᓴᖅ ᓄᓇᒦᓚᐅᖅᑐᑦ

ᕿᑭᖅᑖᓗᒻᒥ, ᐊᒻᒪ ᑎᑭᕝᕕᖏᑦ ᐅᓪᓗᖏᑦ ᕿᑭᖅᑖᓗᒻᒥ ᐊᐅᔭᒃᑯᑦ ᑕᖅᑭᓪᓗᐊᒥ ᑎᑭᓴᕋᐃᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ. ᓇᓄᐃᑦ ᓄᓇᒦᓐᓂᕆᔭᖏᑦ ᖁᕝᕙᕆᐊᓚᐅᖅᑐᖅ 20-30 ᐅᓪᓗᓂᑦ 1990−ᖏᓐᓂᐅᕙᓪᓕᒪᓂᑦ. ᓇᓅᕙᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ ᑕᓪᓗᕈᒻᒨᖅᐸᒍᓐᓃᖅᑐᑦ ᐊᐅᓯᐃᑦᑐᒧᓪᓘᓐᓃᑦ ᕿᑭᖅᑖᓗᐊᓂ ᑭᓯᐊᓂ ᕿᑭᖅᑖᓗᒻᒥᑐᐊᖅ (ᐊᒥᓲᖏᑦᑐᑦ Melville Bay ᐊᑯᑭᑦᑐᑦ, ᐃᓚᖏᑦ ᓇᓄᐃᑦ ᓇᔪᖅᓯᔪᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᓯᕗᓪᓕᖅᐹᒥ ᐊᑐᖅᑐᑎᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ). ᖃᓂᒌᑦᑐᑎᑦ ᖃᓄᐃᓘᖃᖅᑕᐅᑎᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ ᐅᓪᓗᖏᓐᓂᑦ ᑎᑭᑦᑎᓪᓗᒋᑦ ᕿᑭᖅᑖᓗᒻᒧ ᐊᒻᒪ ᓯᑯᐃᖅᑎᓪᓗᒍ, ᐊᒻᒪᓗᑦᑕᐅᖅ ᐅᓪᓗᖏᓐᓂᑦ ᕿᑭᖅᑖᓗᒻᒦᒍᓐᓃᖅᑐᑎᑦ ᓯᑯᒨᖅᑐᑎ ᐊᒻᒪ ᐅᑭᐊᖕᓴᑯᑦ ᓯᑯᓂᖓ.

16 ᑎᓯᐅᔪᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᐊᑐᖅᑐᑎ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ 2009 ᐊᒻᒪ 2015 (15 ᕿᑭᖅᑖᓗᒻᒥ ᐊᒻᒪ 1 Melville Bay, ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ). ᐊᓯᔾᔨᖅᑐᓂᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᑎᓯᒥᖅᑲᕐᓂᖏᑦ BB−ᒥᑦ. ᑎᓯᒥᑦ ᖃᐅᔨᓚᐅᖅᑐᑦ Melville Bay−ᒥ, ᑕᐃᒪᐃᓚᐅᖏᑦᑐᖅ 1990−ᖏᓐᓂᑦ, ᐃᑲᔪᖅᑐᐃᔪᖅ ᐊᓯᖏᓐᓂᑦ ᖃᐅᔨᔭᐅᓯᒪᔪᓂᑦ ᐊᑐᖅᑕᐅᓂᖅᓴᐅᓂᖓᓄᑦ Melville Bay ᐊᕐᕌᒍᓚᐅᓯᒪᓕᖅᑐᓂᑦ. ᐃᓚᒋᐊᕐᓗᒍ, ᐅᓪᓗᖏᑦ ᑎᓯᒨᖃᑦᑕᕐᓂᖏᑦ >1 ᑕᖅᑭᒥᑦ ᑭᖑᕙᓯᓐᓂᖅᑲᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ. ᐊᓂᕕᒋᔭᖏᑦ ᐅᓪᓗᖏᑦ ᑎᓯᖏᓐᓂᑦ ᐊᓯᔾᔨᓚᐅᖏᑦᑐᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᕿᓚᒥᐅᓂᖅᓴᐅᖃᑦᑕᖅᓯᒪᔪᑦ BB−ᒥ 2000−ᖏᓐᓂ; ᓯᕗᓪᓕᖅᐹᖓ ᐅᓪᓗᖓᓂ ᓇᔾᔨᔪᒥ ᓇᓄᕐᒥᑦ ᕿᓚᒥᐅᓂᖅᓴᐅᓚᐅᖅᑐᑦ 2000−ᖏᓐᓂᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᓇᓄᐃᑦ ᐊᑯᓂᐅᓂᖅᓴᖅ ᓄᓇᒦᓚᐅᖅᑐᑦ ᑎᓯᖏᓐᓅᓚᐅᖅᑎᓐᓇᒋᑦ. ᑎᓯᒋᔭᐅᔪᑦ 2000−ᖏᓐᓂᑦ ᖁᕝᕙᓯᓐᓂᖅᓵᓚᐅᖅᑐᑦ ᐊᒻᒪ ᓯᕕᖓᓂᖅᓴᓂᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ, ᐊᐳᑎᖃᑦᑎᐊᖏᓐᓂᖓᓅᔪᓐᓴᐅᔪᖅ.

ᓄᖅᑲᐅᓯᕐᓗᒍ, ᐊᖏᔪᓂᑦ ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖅᑐᑦ BB−ᒥᑦ ᐊᕐᕌᒍᓂᖅᑐᓂᑦ 25−ᓂ ᐊᑦᑐᐊᓂᖅᑐᑦ ᓯᑯᖃᕈᓐᓂᕐᓂᖓᓄᑦ.

ᓛᑉᑕ 5: ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᓂᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓄᖏᓐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᓛᑉᑕ 5 ᓇᓗᓇᐃᔭᐃᔪᖅ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ ᐱᖓᓱᓄᑦ ᐊᕐᕌᒍᓄᑦ MR ᖃᐅᔨᓴᓂᕐᒥᑦ BB ᓇᓄᖏᓐᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ 2011-2013. ᑲᑎᖓᔪᓂᑦ ᐆᒪᔪᓂᑦ −ᐱᔭᐅᔪᑦ ᐊᒻᒪ ᑐᖁᖓᔪᓂᑦ NR ᐊᑐᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᑕᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᑐᒋᑦ ᖃᐅᓴᕈᑎᒥᓃᑦ ᑐᕌᒐᖅᑐᑎᑦ ᐅᓪᓗᒥᒨᖕᓕᖅᑎᑕᐅᓗᑎ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐆᒪᓂᖏᑦ ᐊᒻᒪ ᐊᒥᓲᓂᖏᑦ. ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓃᓚᐅᖅᑐᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ 914 ᐱᔭᐅᔪᑦ ᐊᕐᕌᒍᖏᓐᓂᑦ 1993-1995 ᐊᒻᒪᓗ 1997 ᐊᒻᒪ 1,410 ᖃᐅᔨᓴᕋᔅᓴᐃᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ ᐅᖓᓯᑦᑐᒥᑦ ᖁᑭᐅᑦᑕᐅᔭᓄᑦ 2011- 2013. ᓇᓐᓄᑦᑕᐅᔪᑦ ᐱᔭᐅᔪᑦ (n=243) ᑲᓇᑕᒥ ᐊᒻᒪ ᐊᑯᑭᑦᑐᑦ ᐃᓚᓕᐅᑎᓯᒪᔪᑦ 21−ᓂ ᐊᕐᕌᒍᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ r (1993-2013) ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ, ᑭᓯᐊᓂ ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᖅᑲᓚᐅᖏᑦᑐᖅ 13−ᓂ ᐊᕐᕌᒍᓯᒪᔪᓂᑦ 1998 – 2010.

ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒋᑦ ᐊᒥᓲᓂᖏᑦ BB ᓇᓄᖏᑦ 2012-2013−ᒥ ᐅᑯᐊᖑᓚᐅᖅᑐᑦ 2,826 (95% CI = 2,059-3,593)ᓇᓄᐃᑦ. ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒋᑦ ᐊᒥᓲᓂᖏᑦ 1994-1997 ᐅᑯᐊᖑᓚᐅᖅᑐᑦ 2,173 (95% CI = 1,252-3,093) ᓇᓄᐃᑦ, ᐊᔾᔨᐸᓗᖏᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᐅᓂᒃᑳᓂᑦ ᑕᐃᓗ ᐃᓚᖏᓪᓗ. (2005). ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᐊᔾᔨᒌᓐᖑᐊᖅᑎᒐᔅᓴᐅᖏᑦᑐᑦ ᐊᓯᔾᔨᖅᑲᖏᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᓴᓇᓯᒪᓂᖓ ᐊᒻᒪ ᐊᕙᑎᖓᓄᑦ ᖃᓄᐃᓐᓂᖓᓄᑦ.

ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᒃ ᑲᑎᑦᑐᒋᑦ (ᓵᕐᓗ, ᐱᖅᑯᓯᐅᑎᓪᓗᑎ (ᓇᓐᓄᑦᑕᐅᕙᑦᑐᑦ) ᐆᒪᓂᖏᓐᓄᑦ 2011-2013 ᐅᑯᐊᖑᓚᐅᖅᑐᑦ 0.90 (SE = 0.05) ᐊᕐᓇᓄᑦ ᐊᕐᕌᒍᓖᑦ ≥ 2, ᐊᒻᒪ 0.78 (SE = 0.06) ᐊᖑᑎᓄᑦ ᐊᕐᕌᒍᓕᓐᓄᑦ 2 ≥. ᐊᑯᓂᐅᓂᖓ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᖅ ᑲᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᑦ ᑲᑎᖓᔪᓂᑦ ᐊᕐᕌᒍᓕᓐᓂᑦ ᐊᑎᖅᑲᑕᖏᓐᓄᑦ ᐊᕐᕌᒍᒥ ᐊᒻᒪ ᐊᑎᖅᑕᓖᖏᑦ, ᖃᐅᔨᓴᕐᓂᓕᒫᖑᔪᓂᑦ 1993-2013, ᐃᒪᐃᓕᖕᓚᐅᖅᑐ 0.87 (SE = 0.06). ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᓇᓐᓄᑦᑕᐅᓯᒪᖏᑦᑐᑦ

ᐆᒪᖃᑦᑕᕐᓂᖏᑦ 2011-2013 ᐊᕐᓇᓄᑦ ᐊᒻᒪ ᐊᖑᑎᓄᑦ ᐊᕐᕌᒍᓖᑦ ≥ 2 ᐅᑯᐊᖑᒪᐅᖅᑐᑦ 0.91 (SE = 0.05) ᐊᒻᒪ 0.83 (SE = 0.06), ᑐᖏᓕᕇᑦᑐᑎᑦ. ᒥᔅᐅᓴᑦᑕᐅᔪᑦ ᐆᒪᓂᖏᓐᓄᑦ ᑕᒪᒃᑭᓐᓄᑦ ᐊᖑᑎᓄᑦ ᐊᕐᓇᓄᑦ ᐱᖃᓯᐅᔾᔨᖃᕈᓐᓇᖅᑐᑦ ᐱᐅᖏᑦᑐᓂ ᐃᓚᒋᔭᐅᑎᑦᑎᖏᓐᓂᕐᒥᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᖅᑯᑕᐅᑲᐃᓐᓇᖅᑐᒥ (ᑕᑯᒍᒃ ᓈᑉᑕ 3).

ᐊᔾᔨᒌᓐᖑᐊᑎᑦᑎᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᓂᖏᓐᓂᑦ ᐊᒻᒪ ᐊᕙᑎᒧᑦ ᖃᓄᐃᓐᓂᕆᔭᖓ 1990−ᖏᓐᓂ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᐃᑲᔪᖅᑐᒋᑦ ᓇᓗᓇᐃᖅᑐᒋᑦ ᑭᓪᓕᒋᔭᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᑦ, ᓇᓗᓇᐃᑉᓗᒋᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᔪᓐᓇᖅᑐᑦ, ᐊᒻᒪ ᑐᑭᓯᓗᒋᑦ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᐊᒥᓲᓂᖏᑦ. ᖃᐅᔨᓴᐃᓂᖅ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᓄᓇᖅᐸᓯᒻᒥ ᐱᔭᐅᔪᑦ, ᑕᒪᒃᑭ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᐊᑐᖅᑐᑎ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ, ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᓂᖅᓴᐅᔪᑦ ᐊᒻᒪ ᓄᓇᖓᓂ ᐊᖏᓂᖅᓴᒥ ᐆᒪᔪᓂᑦ ᐱᔭᐅᕗᓂᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ, ᐱᖃᓯᐅᑎᓪᓄᓗᑎ ᓄᓇᖅᐸᒻᒥ ᐊᒻᒪ ᐃᓱᖏᓐᓂᑦ ᑲᖏᖅᑐᓂᑦ, ᐊᕐᕌᒍᖏᓐᓂᑦ 2011 – 2013 ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ, ᐃᓚᒋᐊᕐᓗᒍ, ᐃᒪᐃᓐᓴᕐᒥ ᓯᑯᑕᖃᓚᐅᖅᑐᖅ ᓇᓄᕐᓄᖅ ᐊᕐᕌᒍᑕᒫᑦ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ 1990−ᖏᓐᓂᑦ, ᑭᓯᐊᓂ ᐱᑕᖃᑦᑎᐊᓚᐅᖏᑦᑐᖅ 2010−ᖏᓐᓂᑦ.

ᐋᖅᑭᔅᓯᓚᐅᖅᑐᒍᑦ ᓄᓇᖓᓄ ᐋᖅᑭᔅᓯᒪᔪᓂ 2010−ᖏᓐᓂ ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᖅᑐᑎ ᐊᑦᑐᐃᓂᖃᕐᓂᖏ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ. ᖃᐅᔨᓴᕈᑎᒦᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᓄᓇᖓᑎᒍᑦ ᑭᓪᓕᖅᑎᑕᐅᔪᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᒫᕐᓗ ᑕᐃᒪᐃᓚᐅᖅᑐᖅ 1990−ᖏᓐᓂᑦ ᓴᖅᑭᑎᑦᑎᔪᓐᓇᖅᑐᖅ ᖃᓂᒋᔭᐸᓗᖏᓐᓂᑦ 10% ᐱᐅᖏᑦᑐᓂ ᐱᖃᓯᐅᔾᔨᖏᑦᑐᓂᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ . ᐃᓚᒋᐊᕐᓗᒍ, ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᖏᓂᓪᓗᐊᑲᓪᓕᒋᔪᓐᓇᖅᑕᖅ BB−ᒥ ᓇᓄᖏᑦ ᓯᓚᑖᓃᖔᖅᓯᒪᑐᐃᓐᓇᕆᐊᓖᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓯᕐᑯᒥ MR−ᒥ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ 1990−ᖏᓐᓂᑦ, ᑭᓯᐊᓂ ᓯᑯᒥ ᖃᐅᔨᓴᕐᓂᖃᓚᐅᖏᑦᑐᑦ, ᑖᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅ ᓇᑭᖔᕕᒋᔪᓐᓇᖅᑕᖏ ᐱᖃᓯᐅᔾᔨᑎᑦᑎᖏᑦᑐᓂ 1990−ᖏᓐᓂᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ ᑭᓯᐊᓂ ᓇᓗᓇᐃᖅᓴᓇᓚᐅᖏᑦᑕᕗᑦ ᐊᖏᓂᖓ.

ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᔪᑦ ᐊᒻᒪ ᐊᕙᑎᖓᓄᑦ ᖃᓄᐃᓐᓂᖏᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᓴᖅᑭᑎᑦᑎᓚᐅᖅᑐᑦ ᐊᔾᔨᒌᖏᑦᑐᓂ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐱᔭᐅᒃᑲᓂᑐᐃᓐᓇᓄᐊᖅᑕᖅᑐᓂᑦ. ᑖᓐᓇ ᐊᔾᔨᒌᖏᑦᑐᓂ ᐆᒪᔪᓂᑦ ᐱᔾᔪᑎᒋᓚᐅᖅᑕᖓ ᐅᑎᖅᑕᕝᕕᑲᐃᓐᓇᖅᑕᖅᑐᖅ ᓇᓄᐃᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥ, ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᖅ 1990−ᖏᓐᓂᑦ (, ᓇᓄᐃᑦ ᐊᒥᓲᓂᖅᓴᐃᑦ ᓄᓇᒦᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ, ᖃᐅᔨᓴᕐᕕᐅᓂᖅᓴᐅᖏᑦᑐᒥ, ᐅᕝᕙᓘᓐᓃᑦ ᓯᑯᖓᓂ). ᕿᑎᕋᖅᑐᑦ ᐅᑎᖅᑕᕝᕕᐅᑲᐃᓐᓇᖅᑐᑦ ᑕᒪᐅᓇᑐᐃᓐᓇᖅ ᐊᑲᐅᖏᓕᐅᕈᑕᐅᖏᑦᑐᑦ MR ᐊᑐᖅᑕᒧᑦ, ᖁᕝᕙᓯᑦᑐᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᔾᔨᒌᖏᖃᑦᑕᖅᑐᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐃᑳᕐᕕᐅᑲᐃᓐᓇᖅᑐᓂ ᑲᑎᖓᓗᑎ ᐊᖅᑯᓯᖅᑕᐅᔪᓂ ᖃᐅᔨᓴᐃᓂᖅ, ᐅᕝᕙᓘᓐᓃᑦ ᑕᒪᐅᓇᑐᐃᓐᓇᐅᖏᑦᑐᖅ ᐃᑳᕐᕕᐅᑲᐃᓐᓇᖅᑐᑦ, ᖃᐅᔨᒪᔭᐅᑦᑎᐊᖅᑐᑦ ᐱᖃᓯᐅᔾᔨᑎᑦᑎᖏᓐᓂᕐᒥᑦ. ᐊᑐᖅᑕᕗᑦ ᐱᖃᓯᐅᑎᓪᓗᑎ ᓇᓄᑦᑕᒥᓂᕐᓂᑦ ᐊᔾᔭᖑᑎᑐᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᓂᖅ ᐆᒪᔪᓂᑦ ᑐᑭᓯᒋᐊᖅᑎᑕᐅᓚᖓᔪᑦ ᓯᐊᒻᒪᖅᓯᒪᓕᐊᕐᑎᑦᑎᔪᑦ ᐱᖃᓯᐅᔾᔨᓂᖏᓐᓂᑦ ᐃᓚᖏᓐᓄᑦ. ᑭᓯᐊᓂᓕ BB ᖃᐅᔨᓴᕈᑎᓃᑦ ᒥᑭᓗᐊᖅᑐᑦ MR−ᒨᕐᓗᑎ ᒥᔅᐅᓴᑦᑕᐅᓚᐅᖅᑐᑦ ᐊᑐᖅᑕᐅᑲᐃᓐᓇᖅᑐᒥ ᐊᒻᒪ ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᐊᑦᑐᐃᓂᖃᓗᐊᖏᑦᑐᖅ ᑭᓪᓕᖏᓐᓄᑦ.

ᐃᓱᒪᒋᓪᓗᒋᑦ ᑭᓪᓕᓯᓂᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓗᓇᕈᑎᐅᔪᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂ ᑭᓪᓕᖏᓐᓂᑦ ᐊᒻᒪ ᖃᐅᔨᔭᐅᓯᒪᔪᓂ ᖃᐅᔨᓴᓂᕐᒧᑦ ᓴᓇᓯᒪᓂᖅ ᐊᒻᒪ ᐊᕙᑎᒧᑦ ᖃᓄᐃᓐᓂᖏᑦ ᑕᐃᒪᐃᓐᓂᖅᓴᖅᑯᑦ ᒥᔅᐅᓴᑦᑕᐅᓂᖏᓐᓄᑦ ᐊᑦᑎᑦᑐᒦᓯᒪᔪᑦ ᐊᒥᓲᓂᖏᑦ 1990−ᖏᓐᓂᑦ, ᓄᖅᑲᐅᓯᕋᔅᓴᐅᖏᑦᑐᖅ ᑖᒃᑯᐊ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᓲᓂᖏᑦ 2010−ᖏᓐᓂᑦ ᑭᒡᒐᖅᑐᐃᔾᔪᑎ ᖁᕝᕙᓯᑦᑐᓂᑦ ᐊᖏᓂᖓᓄ BB ᓇᓄᖏᓐᓄᑦ. 2010−ᖏᓐᓂ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑭᒡᒐᖅᑐᐃᓪᓗᐊᑕᕋᓕᐅᖅᑐᑦ ᐱᐅᓛᓂ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᑐᑭᓯᒋᐊᑭᓯᑎᓂᑦ ᐊᒻᒪ ᓈᒻᒪᑦᑐᑎ ᐊᐅᓚᑦᑎᓂᖅᓴᐅᓂᖏᓐᓄᑦ, ᑐᓐᖓᓕᐅᕈᑎᒋᓪᓗᒋᑦ ᓇᓗᓇᐃᖅᓯᓂᐅᔪᖅ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᕐᕌᒎᓯᒪᔪᓂᑦ 1993−2013 ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ.

2011 – 2013 ᒥᓵᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᕆᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐃᒻᒥᑰᖅᑐᓄᑦ ᐊᒃᓇᓄᑦ ᐊᑦᑎᓗᐊᖅᑰᖅᑐᖅ ᐃᑲᔪᖅᑐᐃᓗᓂ ᒪᑭᑕᑦᑎᐊᖅᑐᓂ ᐊᒥᕉᓂᖏᑦ, ᑭᓯᐊᓂ ᓄᓇᙱᓯᒍᑦ ᐊᑐᖑᐊᕐᓂᖅ (ᓱᕐᓗ, ᐊᒥᕉᓂᖏᑦ ᓈᒻᒪᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᖃᓯᐅᔾᔭᐅᓗᑎ ᐆᒪᓂᖏᑦ, ᐱᔭᐅᔪᑦ ᐊᒻᒪ ᓇᓄᑦᑕᐅᔪᑦ) ᐱᑕᖃᓐᐊᖅᖢᑦ ᒥᓵᐅᓴᑦᑕᐅᔪᓂᑦ ᖃᐅᔮᔭᐅᔪᓂᑦ ᐊᒻᒪ ᐊᒥᕉᓂᓇᔪᓐᓇᖅᑕᖏᑦ (ᓱᕐᓗ, ᐃᓄᓅᖏᑦᑐᖅ ᐋᖅᑕᐅᑎᓪᓗᒋᑦ) ᓇᓄᐃᑦ ᐊᒥᓯᕈᖅᐸᓪᓕᐊᓂᖏᑦ. ᐊᑦᑎᑦᑐᑦ ᒥᓵᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᕆᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐃᒻᒥᑰᖅᑐᓄᑦ ᐊᖑᑎᓄᑦ ᐃᓱᒫᓘᑎᖃᕈᓐᓇᖅᑐᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᕐᓗᓂ. ᑭᓯᐊᓂᓕ, ᕿᓚᒥᐅᔪᒧᑦ ᐆᒪᔪᓂᑦ ᐱᔭᐅᔪᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓅᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ, ᑭᓪᓕᓯᓂᐊᖅᑕᐅᓯᒪᔪᑦ ᓇᓗᓇᕈᑎᖃᖅᑐᑎᑦ, ᐊᒻᒪ ᐱᐅᖏᑦᑐᒥᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᔪᓐᓇᕐᓂᖓ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᑐᖅᑕᐅᑲᐃᓐᓇᕐᓂᕚᑦᑕᖓ (ᓵᕐᓗ ᐱᖃᑕᐅᖏᑎᑦᑎᓂᐅᔪᑦ ᐅᖃᐅᓯᐅᒐᔪᑦᑐᑦ ᐃᓱᐊᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ) ᑭᓪᓕᖃᖅᑎᑦᑎᔪᖅ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒥᕉᓂᖏᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐅᕝᕙᓘᓐᓃᑦ ᒫᓇᐅᔪᒥᑦ ᖃᓄᐃᓕᖓᓂᕆᔭᖏᑦ BB ᓇᓄᖏᑦ ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᒥᓵᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᐆᒪᓂᖏᓐᓄᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ.

ᒌᑉᑕ 6: ᕿᑐᙵᕆᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᓂᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ ᓴᓂᕈᑎᐅᑉ ᐃᒪᙱᓂ

ᐊᑐᖅᑐᑎᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓴᒪᕿᖀᓂᒃ ᖃᐅᔨᓴᕐᓂᖅᕹᐅᓯᒪᔫᓂᒃ MR−ᒥ ᖃᐅᔨᓴᓂᕐᒥᑦ BB−ᒥᑦ (1993-1995, 1997 ᐊᒻᒪᓗ 2011- 2013) ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᕿᑐᙵᕆᐅᖃᑦᑕᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᕐᒥᔪᑦ ᖃᐅᔨᓴᖅᑕᐅᓪᓗᑎ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᐊᒥᕉᓂᖏᑦ ᐊᒻᒪᓗ ᖃᐅᔨᓴᖅᑐᕆᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᕿᑐᙵᖃᑦᑕᕐᓂᖏᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖅᑐᑦ ᓯᑯᙱᓄᑦ ᖃᓄᐃᓐᓂᓕᔾᔪᙱᓄᑦ. BB−ᒥ, ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᐊᔾᔨᔾᔨᖅᑐᓂᑦ ᐊᒥᕉᓂᓇᖃᑦᑕᖅᑕᖏᑦ ᐊᑎᖅᑕᑏᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᕐᕋᒍᒧᑦ (COY) ᐅᕝᕙᓘᓐᓃᑦ ᐊᑎᖅᑕᐃᑦ. ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᒥᕉᓂᖏᑦ ᐊᖏᔫᓗᐊᓚᐅᖏᑦᑐᖅ ᐊᒻᒪ ᐊᔾᔨᔾᔨᕐᓂᖃᓚᐅᖏᑦᑐᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐅᐱᕐᙵᓵᕈᖅᐸᓪᓕᐊᓂᙱᓄᑦ.

ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᒥᑦ ᐊᓯᔾᔨᑲᐃᓇᖃᑦᑕᖅᑐᓂᑦ ᐅᕙᒍᑦ ᑎᑎᕋᖅᓯᒪᔭᖏᓐᓂᑦ COY ᐊᕐᕌᒍᑕᒫᑦ (ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ COY ᐊᒥᕉᓂᖏᑦ ᐊᑕᐅᓯᖅ ᐊᕐᓇᓪᓗᑦ MR−ᒥ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ) ᐊᑦᑐᐊᓂᖃᓚᐅᖅᑐᑦ ᐊᔾᔨᒌᖏᖃᑦᑕᕐᓂᙱᓄᑦ ᓯᑯᙱ ᖃᓄᐃᓐᓂᙱ. 1993−ᒥ 2013−ᒧᑦ. ᐊᔾᔨᐸᓗᖏᑦ ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑦ ᐊᑕᐅᑦᑎᒃᑯᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐅᐱᕐᙶᒃᑯᑦ ᓯᑯᐃᓵᕋᐃᓐᓂᙱᓄᑦ. ᐊᔾᔨᐸᓗᖏᑦ ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑦ ᕿᑐᙵᕆᐅᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᐊᑦᑐᐊᓂᖅᑐᓂ ᓯᑯᙱᓄ ᖃᓄᐃᓐᓂᓕᔾᔪᖏᓐᓄᑦ ᐅᓂᒃᑲᓕᐊᖑᕋᑖᓚᐅᖅᑐᑦ ᐊᓯᖏᓐᓄᑦ ᓇᓄᕐᓄᑦ.

ᒥᓵᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ (ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᕉᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᑕᐅᓯᕐᒧᑦ ᐊᕐᓇᓪᓗᒧᑦ MR−ᒥ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ) BB-ᒧᑦ ᖃᐅᔨᓴᕐᓂᖅᑎᓪᓗᒋᑦ 1993-2013 ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖅᑐᑦ 0.24 ᑎᑭᑦᑐᒍ 0.51. ᑖᒃᑯᐊ ᖃᐅᔮᔭᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ BB ᑕᑯᓵᐅᑎᑦᑎᕙᓪᓕᐊᖏᓇᖅᑐᖅ ᖁᕝᕙᓯᓐᓂᖏᑦ ᕿᑐᙵᕆᐅᕐᓂᖃᑦᑕᓇᐊᖅᖢᑕᖏᑦ ᓈᒻᒪᑦᑐᓄᑦ ᐊᒥᕉᓂᖏᓐᓄᑦ, ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓅᙶᖅᑐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐅᓂᒃᑲᓕᐊᖑᔪᑦ Regehr et al. (2015), ᐅᖃᐅᔾᔨᓚᐅᖅᑐᑦ ᐊᔾᔨᒋᓐᓂᖏᑦ ᐊᑎᖅᑕᖏᓐᓂᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᓯᔾᔨᕈᑎᓪᓗᐊᑕᓇᔭᙱ ᐊᒥᕉᓂᖏᓐᓄᑦ ᐱᕈᖅᐸᓪᓕᐊᓂᖏᓐᓄᑦ ᐅᖓᓯᓐᓂᖅᖢᑐᓂᑦᑫ 0.1 ᑎᑭᑦᑐᒍ 0.3 ᐊᒥᕉᓂᖏᓐᓄᑦ ᐊᒥᓯᕈᖅᐸᓪᓕᐊᓗᑎ.

ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᐊᑦᑐᐊᓂᖅᖢᑐᓂᑦ ᐊᑯᓐᓂᖏᓐᓂ COY ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᕐᕌᒍᙱᓂ t ᐊᒻᒪ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ ᐊᕐᕌᒍᒥ year *t* + 1, ᐊᒻᒪᓗᑦᑕᐅᖅ ᓴᖏᔪᒥᑦ ᐊᑦᑐᐊᓂᖅᖢᑐᓂ COY ᐊᑎᖅᑕᐃᑦ ᐊᒥᕉᓂᖏᑦ ᐊᒻᒪ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐊᕐᕌᒎᓂᐊᖅᑐᒧᑦ. ᑖᒃᑯᐊ ᖃᐅᔮᔭᐅᔪᑦ ᐊᒥᕉᓂᖅᓴᓇᔭᖏᑦ ᑐᕐᑯᖃᑦᑕᖅᑐᑦ ᓇᓄᐃᑦ ᐊᑎᖅᑕᐅᑎᓪᓗᒋᑦ ᓯᕗᓪᓕᖅᐹᖏᓐᓂᑦ 8−ᓂ ᑕᖅᑭᓂᑦ, ᑕᐃᒪᓐᓇ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᑐᓐᖓᓂᖃᖅᑐᑦᐊᒥᕉᓂᖏᑦ COY ᐃᕐᓂᐊᖑᓚᐅᖅᑐᑦ ᐊᒻᒪ ᐆᒪᓂᖃᖅᑐᑎᑦ ᑎᑭᑦᑐᒍ ᓯᕗᓪᓕᖅᐹᙱᓄ ᓯᑯᖃᕈᓐᓃᓯᑎᓪᓗᒍ.

ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᐊᖏᔪᒥᑦ ᐊᔾᔨᕐᓂᖃᖅᑐᓂᑦ ᐊᒥᕉᓂᖏᓐᓂᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᑎᖅᑕᓂᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ. ᑭᓯᐊᓂᓕ, ᐊᑦᑐᐊᓂᖃᕐᓂᙱᓄᑦ ᐊᑯᓐᓂᖏᓐᓂ COY ᕿᑐᙵᕆᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑏᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᑦ ᐊᒻᒪ ᓯᑯᙱ, ᐊᒻᒪ ᐊᑯᓂᐅᔪᒧᑦ ᐊᔾᔨᕐᓂᖏᑦ ᓯᑯᙱᓂ BB−ᒥᑦ, ᑲᙳᓯᑦᑐᒍᑦ ᐊᑦᑎᓪᓕᕆᐊᕐᓂᖏᑦ ᐊᑎᖅᑕᐃᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᓂᖏᑦ ᖃᐅᔨᓇᕋᔭᖅᑐᖅ ᐊᖏᓂᖅᓴᓄᑦ ᐊᒻᒪ ᐊᑯᓂᐅᓂᖅᓴᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᒧᑦ.

ᐊᑎᖅᑕᐃᑦ ᐊᒥᓲᓂᕆᖃᑦᑕᖅᑕᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ ᐃᒪᕆᔭᖏᓐᓂᑦ COY ᐊᒻᒪ ᐊᑎᖅᑕᐃᑦ ᐊᔾᔨᐸᓗᕆᓚᐅᖅᑕᖏᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᓂᑦ Foxe Basin– ᐊᒻᒪ, ᓂᕆᖓᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ . ᑕᒫᓐᓇᖅᑐᒍ, ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᕿᑦᑎᓂᖅᓵᐅᓚᐅᖅᑐᑦ ᓴᓂᐊᓂ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓄᑦ ᕿᑭᖅᑖᓗᑉ ᐃᑭᕋᓴᖓᓄᑦ ᐊᒻᒪ ᐱᖓᓐᓇᖓᓂ ᑕᓯᐅᔭᕐᔪᐊᖅ. ᑖᒃᑯᐊ ᐊᓪᔨᕖᓐᖑᐊᑎᑦᑐᕆᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ BB ᐊᓯᔾᔨᖅᓯᒪᓗᐊᖏᑦᑐᖅ ᕿᑐᓐᖏᕐᐅᕝᕙᐅᖃᑦᑕᕐᓂᖓ ᑕᒫᓂ ᖃᑦᑏᓐᓇᕈᓇᐊᖅᓯᒪᔫᔭᕋᑖᓚᐅᕋᓗᐊᖅᑎᓪᓗᒋᑦ ᕿᑐᓐᖏᕐᐅᖃᑦᑕᕐᓂᖏᑦ.

ᓵᑉᑕ 7: ᓇᓄᐃᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ

ᐅᖅᓱᖃᕐᓂᖓᓄᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ (FI) ᐊᒥᓱᒐᓚᐃᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ ᓇᓄᕐᓂᑦ ᖃᐅᔨᔭᐅᔪᓂᑦ ᒪᕐᕉᓐᓂ MR ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ BB–ᒥᑦ, 1993-1995, 1997 ᐊᒻᒪᓗ 2011- 2013. ᖃᐅᔨᓴᓚᐅᖅᑕᕗᑦ ᐊᔾᔨᖃᑦᑕᕐᓂᖏᑦ ᖃᐅᔨᓴᕈᑎᒥᕐᓂᑦ ᑎᒥᖏᓐᓄᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᑕᒪᒃᑭᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᔫᓐᓂᒃ ᐊᑦᒃᑐᐊᓂᖃᖅᑐᑎᑦ ᓱᑯᖓᓄᑦ ᖃᓄᐃᓐᓂᓇᔭᖓᓄᑦ.

ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᐊᑦᑎᓪᓕᕆᐊᕐᓂᖏᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᓇᓄᕐᓂᑦ BB–ᒥ ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2013. ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ ᓇᓄᕐᓂᑦ BB–ᒥ ᐊᑦᑎᓪᓕᕆᐊᓚᐅᖅᑐᑦ ᐊᑦᑐᐊᓂᖃᖅᑐᑎᑦ ᐃᒪᐅᖏᓐᓇᕐᓂᖓᓄᑦ ᐊᒻᒪ ᐅᐱᕐᖔᒃᑯᑦ ᓱᑯᐃᕐᕝᕆᕙᑦᑕᖓᓄᑦ. ᑖᓇ ᒪᓕᓐᓂᖃᖅᑐᖅ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᓇᐊᖃᖅᑐᓂᑦ ᐊᑯᓂᐅᓂᖓᓂᑦ ᓱᑯᒦᒍᓐᓃᕐᓂᓇᓚᐅᖅᑕᖏᑦ (ᐊᒻᒪᓗᖃᐃ ᐊᑦᑎᓪᓕᕆᐊᖅᑐᑎᑦ ᓂᓇᖃᑦᑕᖅᑕᖏᑦ ᐅᐱᕐᖔᒃᑯᑦ ᐊᐅᔭᕈᖁᑖᖅᑎᓪᓗᒍ ᐊᐅᓚᔾᒨᑎᓪᓗᐊᑕᓇᔭᖓ ᐊᑦᑎᓪᓕᕆᐊᕐᓂᖓᓄᑦ.

ᖃᐅᔨᓴᕈᑎᒥᓂᖅᐳᑦ ᖃᐅᔨᓴᕐᓂᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᐊᑦᑎᓪᓕᕝᓪᓕᐊᓪᓗᑎ ᖃᓄᐃᓐᓂᓇᔭᖏᑦ BB–ᒥ ᓇᓄᖏᓐᓂ 2010–ᒥ ᐊᑐᑐᑎ ᐊᔾᔨᕖᖏᑦᑐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᕐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᐅᓚᑕᐅᔪᓂᑦ ᓇᓄᕐᓂᑦ (Rode et al. 2012). ᖃᐅᔨᓴᖅᑐᑎ ᐅᓂᒃᑳᓕᐊᓂᑦ ᓵᑉᑕ 7 ᐊᑕᖐᔫᒥᓚᐅᖅᑐᖅ ᑕᒪᒃᑮᓐᓂ ᖃᐅᔨᓴᕐᓂᐅᓯᒪᔫᓐᓂᒃ (1993-1997 ᐊᒻᒪ 2011-2013) ᓇᓗᓇᐃᖅᓯᔪᑦ ᖃᐅᔨᔭᐅᔪᑦ Rode et al. (2012) ᑭᒡᒐᖅᑐᐃᔪᑦ BB–ᒥᑦ ᐊᖏᓂᖓ ᒥᑭᓂᖅᓴᐅᓚᐅᕋᓗᐊᖅᑎᓪᓗᒍ ᑭᓪᓕᖃᖅᑎᑕᐅᓂᖓ.

ᑖᒃᑯᐊ ᖃᐅᔨᔭᐅᔪᑦ ᒪᓕᓐᓂᖃᕐᒥᔪᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᓐᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ BB–ᒥ ᐱᐅᖏᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐱᕆᐊᕐᓂᖏᓐᓂ 200–ᖏᓐᓂᑦ ᐊᔾᔨᖃᐸᓗᑦᑐᓂ 1990–ᖏᓐᓂᑦ (Dowsley ᐊᒻᒪ Wenzel 2008, Born et al. 2011)

ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᔪᓂᑦ ᓄᓇᒥ ᓂᓇᓇᓯᐊᖅᑐᓂᑦ ᖃᓂᒋᔭᐸᓗᖏᓐᓂ 9% BB ᓇᓄᖏᑦ ᖃᐅᔨᔭᐅᔪᑦ ᓄᓇᒦᑦᑎᓪᓗᒋᑦ ᖃᐅᔨᓴᕐᓂᖃᖅᑎᓪᓗᒋᑦ. ᐃᒪᕐᒥᐅᑕᐃᑦ ᐆᒪᔪᐃᑦ, ᐱᓗᐊᖅᑐᒥ ᓇᑦᑏᑦ ᐊᒻᒪ ᐊᐃᕖᑦ, ᓇᑉᐸᓪᓗᐊᐸᓗᕆᓚᐅᖅᑕᖏᑦᓇᓗᓇᐃᖃᔅᓴᐅᔪᓂᑦ ᓂᕆᖃᑦᑕᖅᑕᖏᓐᓂᑦ.

ᓵᑉᑕ 8: ᓇᓐᓄᑦᑕᐅᔪᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin: ᓇᓗᓇᐃᔭᖅᓯᒪᓂᖏᑦ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᓯᒪᔪᑦ ᐊᒻᒪ ᓇᓐᓄᑦᑕᐅᔪᓂᑦ ᑐᓴᖅᑎᑦᑎᓂᖅ, 1993 ᑎᑭᑦᑐᒍ 2014.

ᓵᑉᑕ 8 ᓇᓗᓇᐃᔭᐃᔪᖅ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᑐᓂᑦ BB-ᒥ ᐊᒻᒪ KB–ᒥᑦ. ᖃᐅᔨᓴᓚᐅᖅᑕᕗᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐊᒻᒪ ᐋᖅᑭᔅᓯᒪᓂᖏᑦ ᓇᓐᓄᑦᑕᐅᖃᑦᑕᖅᑐᓂᑦ ᖁᓕᓂᑦ ᐊᕐᕌᒍᖃᑦᑕᖅᓯᒪᔪᓂᑦ ᐊᔾᔨᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᒥᓲᓂᖏᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐊᑦᑐᐊᓂᖃᕐᓂᖏᓐᓄᑦ ᐊᓯᖏᓐᓄᑦ ᐱᔾᔪᑕᐅᔪᓄᑦ. ᖃᐅᔨᓴᓚᐅᕐᒥᔪᒍᑦᑕᐅᖅ ᓇᓐᓄᑦᑕᐅᕝᕗᑦᑐᓄᑦ ᐊᑐᖅᑕᐅᕝᕗᑦᑐᑦ ᒫᓐᓇᐅᔪᒥ; ᓇᓗᓇᐃᖅᑐᕆᑦ ᐊᑦᑐᐃᓂᖃᕈᓐᓇᖅᑐᓂᑦ ᑕᒻᒪᖅᓯᒪᖏᓐᓂᖏᓐᓄᑦ ᓇᓐᓄᑦᑕᐅᕝᕗᑦᑐᓂᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᒻᒪ ᐱᔪᓐᓇᕐᓂᖏᑦ ᒐᕙᒪᖃᕐᕕᐅᔪᑦ ᐊᐅᓚᑦᑎᑎᐊᕐᓗᑎ ᓇᓐᓄᑦᑕᐅᕝᕗᑦᑐᓂᑦ. ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᑐᑭᓯᕆᐊᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᐅᖃᐅᒥᓯᐅᔪᓐᓇᖅᑐᓂᑦ ᓯᕗᓂᒃᓴᒥ ᓇᓐᓄᑦᑕᐅᔫᓐᓇᖅᑐᓂᑦ ᖁᕝᕙᓯᓐᓂᖏᓐᓂᑦ ᐊᒻᒪ ᓇᓗᓇᐃᖅᑕᑲᐅᓂᖓ ᐊᒥᓲᓂᖏᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ. ᐊᑐᓕᖁᔭᐅᔪᑦ ᐱᐅᓯᑎᕆᐊᖅᑕᐅᓗᑎᑦ ᓇᓐᓄᑦᑲᑕᐅᕝᕗᑦᑐᓂᑦ ᑐᓴᖅᑎᑦᑎᓂᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐱᑕᖅᑎᑕᐅᓗᑎᑦ.

ᑕᒪᒃᑭ ᑲᓇᑕ (ᓄᓇᕗᑦ) ᐊᒻᒪ ᐊᑯᑦᑐᓂᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᑦ BB–ᒥᑦ ᐊᒻᒪ KB–ᒥᑦ ᓇᓄᖏᓐᓂᑦ. 1993-2005–ᒧᑦ (ᓵᕐᓗ, ᑰᑕᖃᖅᑎᑕᐅᓚᐅᖅᑎᓐᓇᒋᑦ ᐊᑯᑭᑦᑐᑦ) ᑲᑎᓐᖓᓪᓗᒋ (ᓵᕐᓗ, ᑕᒪᒃᑭᓐᓂ ᓄᓇᓕᕐᔪᐊᓂᑦ) ᐊᕐᕋᒍᑕᒫᑦ ᓇᓄᑦᑕᐅᒐᔪᓚᐅᖅᑐᑦ 165 (ᐊᒥᓲᓂᖏᑦ: 120-268) 12 (ᐊᒥᓲᓂᖏᓐᓂᑦ: 6-26) ᓇᓄᐃᑦ BB-ᒥᑦ ᐊᒻᒪ KB–ᒥ ᓇᓄᖏᓐᓄᑦ, ᑐᖏᓕᕇᑦᑐᑎ. ᑲᑎᑦᑐᒋᑦ ᓇᓄᑦᑕᐅᔪᑦ ᐊᑯᓐᓂᖏᓐᓂ 2002 ᐊᒻᒪ 5005 ᒪᓕᓐᓂᖃᖅᑐᑦ ᐊᒥᓱᓚᓐᓂᑦ ᖃᓄᐃᓘᖅᑐᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ ᑐᓴᖅᑎᑦᑎᓂᕐᒥᑦ ᐊᒻᒪ ᓇᓄᑦᑕᐅᔪᓐᓇᖅᑐᓂ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᑕᒪᒃᑭᓐᓂ ᑲᓇᑕᒥ ᐊᑯᑭᑦᑐᓂᓗ.

BB–ᒥ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ ᑲᑎᓐᖓ)ᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ ᖃᓂᒋᔭᖓᓃᖏᓐᓇᖅᓯᒪᔪᖅ 2:1 (ᐊᖑᑎᑦ: ᐊᕐᓇᐃᑦ) ᐊᕐᕋᒍᑕᒫᖑᒐᔪᑦᑐᓂ 35% ᐊᕐᓇᕐᓗᐃᑦ ᐃᒻᒥᑰᖅᑐᑦ ᓇᓄᐃᑦ. KB–ᒥ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ ᑲᑎᓐᖓᔪᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ 33%–ᖑᓚᐅᖅᑐᖅ ᐊᕐᓇᐃᑦ ᖃᐅᔾᔭᕐᓂᐅᔪᒥᑦ 1993-2014. ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ KB–ᒥᑦ ᓇᓄᑦᑕᐅᓯᒪᔪᑦ ᑰᑕᖃᓕᖅᑎᑕᐅᔪᒻᒪᑕᓂᑦ ᐊᑯᑭᑦᑐᑦ 44%–ᖑᓯᒪᔪᑦ ᐊᕐᓇᐃᑦ ᑭᓯᐊᓂ ᐱᓪᓚᕆᒋᔭᖓ ᒥᔅᓴᐅᓴᑦᑕᐅᓂᖓᓄᑦ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ KB–ᒥ ᓇᓄᑦᑕᑲᐅᔪᓂᑦ ᓴᖑᑦᑐᖅ.

ᐊᑯᑭᑦᑐᓂᑦ ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᓇᓄᑦᑕᑲᐅᕙᑦᑐᑦ ᐊᑯᓐᓂᖏᓐᓂ ᔫᓐᓄᐊᕆ ᐊᒻᒪ ᔩᓂ BB–ᒥ ᐊᒻᒪ KB–ᒥᑦ, ᓄᓇᕗᒻᒥᓕ ca. 40% ᓇᓄᑦᑕᐅᔪᑦ BB–ᒥᑦ ᐊᐅᔭᒃᑯᑦ ᐅᑭᐊᔅᓵᒧᑦ (ᐋᒡᒌᓯ – ᓅᕖᐱᕆ) ᓇᓄᐃᑦ ᓯᔅᔭᖅᐸᓯᒻᒦᑎᓪᓗᒋᑦ. ᓄᓇᕗᒻᒥ, ᓇᓄᑦᑕᐅᖃᑦᑕᖅᑐᑦ KB–ᒥᑦ ᐅᐱᕐᖔᒃᑯᑦ.

ᓇᓄᓚᓯᑐᐃᓐᓇᕐᓂᖅ ᐱᔪᓐᓇᖅᑎᑕᐅᔪᖅ ᑲᓇᑕᒥ ᑭᓯᐊᓂ ᐊᑯᑭᑦᑐᓂᐅᖏᑦᑐᖅ. ᓇᓄᓚᓯᑐᐃᓐᓇᕐᓂᖅ ᑭᒡᓕᖅᑐᐃᔪᖅ 16% ᐊᕐᕋᒍᑕᒫᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐊᒻᒪ ᖁᕝᕙᓯᓐᓂᖃᑏᖑᓚᐅᖅᑐᑎ 2008–ᒥ ᒪᓕᓐᓂᖃᖅᑐᑦ ᐊᒥᓱᓚᓐᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᔪᓂᑦ. ᐊᖑᓇᓱᑐᐃᓐᓇᕐᓂᐅᔪᑦ ᓂᕈᐊᖃᑦᑕᓲᑦ ᓇᓄᑐᖃᐅᓂᖅᓴᓂᑦ, ᐃᓐᓇᕐᓂᑦ ᐊᖑᑎᓂᑦ.

ᐃᒻᒥᓂ ᐱᕐᑯᑎᓂᓪᓗ ᓴᐳᒻᒥᑦᑎᓂᕐᒧᑦ ᑐᕐᑯᑕᐅᔪᑦ (DLPs) ᓇᓄᐃᑦ BB–ᒥᑦ ᐊᒥᓲᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᒪᒃᑯᓂᖅᓴᐃᑦ (2-3 ᐊᕐᕋᒍᓖᑦ). ᖃᐅᔨᓂᒥᖃᓚᐅᖅᑦᑐᒍᑦ ᐊᓯᔾᔨᖃᑦᑕᖅᑐᓂᑦ ᐊᕐᕋᒍᑕᒫᑦ ᐊᒥᒨᓂᖏᑦ DLPs ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2014. ᐊᒥᓲᓂᖅᓴᐃᑦ DLPs ᐃᒪᐃᓪᓇᒦᖅᐸᓚᐅᖅᑐᑦ (ᐋᒡᒌᓯ–ᓅᕖᐱᕆ) ᓇᓄᐃᑦ ᓯᔅᔭᒦᑦᑎᓪᓗᒋᑦ ᓯᔅᔭᖅᐸᓯᒻᒥᒎᓐᓃᑦ. ᐊᕐᕋᒍ ᐃᓗᐊᓂ ᓴᖅᑭᕐᓂᕆᕙᑦᑕᖏᑦ DLPs ᑲᓇᑕᒥ (ᓄᓇᕗᑦ) ᐊᑦᑎᓕᒋᐊᖅᓯᒪᔪᑦ ᐊᐅᔭᒃᑯᑦ ᐅᑭᐊᔅᓵᑯᓪᓗ.

ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᖅᑕᐅᓃᑦ ᐊᖑᑕᐅᓂᖏᓐᓄᑦ ᐊᕐᓇᐅᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ, ᐅᓂᒃᑳᓕᐊᖑᔪᑦ ᐊᖑᓇᓱᑦᑎᓄᑦ, ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᐊᖑᑕᐅᓂᖓ ᐊᕐᓇᐅᓂᖓ ᓇᓗᓇᐃᖅᑕᐅᑲᒻᒪᔅᓯᒪᓚᐅᖅᑐᑦ ᐊᒥᓱᓚᐅᔪᓂᑦ ᐊᒻᒪ ᑕᒻᒪᕐᓂᐅᔪᑦ ᐊᒥᓲᓂᖅᓴᐅᓚᐅᖅᑐᑦ ᐊᑯᑭᑦᑐᓂᑦ. ᑐᓐᖓᓂᖃᖅᑐᑎ ᐊᐅᖏᑎᒍᑦ ᓇᓗᓇᐃᖅᑕᐅᓂᖏᓐᓄᑦ, ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᑕᑯᔅᓴᐅᓂᖏᑎᒍᑦ (ᓵᕐᓗ ᓯᐅᑎᖏᑎᒍᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ ᐊᒥᐊᖅᑕᐅᓯᒪᔪᓪᓗᓐᓃᑦ) ᓇᓄᕐᓂᑦ ᖃᑦᑏᓐᓇᐅᓗᐊᖅᑐᑦ ᑐᓴᖅᑎᑦᑎᔪᑕᐅᓯᒪᔪᑦ. ᖃᐅᔾᔭᐅᓂᖏᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᐊᑦᑎᓕᒋᐊᖅᓯᒪᔪᑦ ᐱᑐᖃᕈᖅᐸᓕᐊᓗᑎ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᔭᒐᐃᓪᓗᑎ, ᐱᓗᐊᖅᑐᒥ ᔭᒐᑦᑐᑎ ᓯᐅᑎᖏᑎᒍᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ, ᐊᑲᐅᖏᓕᐅᕈᑕᐅᓚᐅᖅᑐᖅ. ᑖᓐᓇ ᖃᐅᔾᔭᐅᓂᐅᔪᖅ ᐊᑦᑐᕈᑎᖃᖅᑐᖅ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐅᑎᖅᑎᑕᐅᓂᖏᓐᓄᑦ MR–ᒥ ᖃᐅᔾᔭᕐᓂᐅᔪᓂᑦ.

ᐊᐅᓚᑦᑎᓂᐅᔪᖅ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᐅᖃᑦᑕᖅᓯᒪᔪᑦ ᓇᓄᕐᓂᑦ ᑲᓇᑕᒥ ᐊᑯᑭᑦᑐᓂᓗ ᐊᔾᔨᒌᖏᑦᑐᑦ ᐃᓚᖓᑎᒍᑦ: (1) ᓇᓄᑦᑕᐅᔪᓐᓇᖅᑐᑦ ᑭᓕᖏᑦ ᓴᖅᑭᖅᑎᑕᐅᓚᐅᖅᑐᑦ 1967–ᒥᑦ ᐊᒻᒪ ᐊᑯᑭᑦᑐᓂᑦ 2006–ᒥᑦ,
(2) ᓇᓄᓚᓯᐊᖅᑐᖃᑐᐃᓐᓇᕈᓐᓇᖅᑐᖅ ᑲᓇᑕᒥ ᑭᓯᐊᓂ ᐊᑯᑭᑦᑐᓂᐅᖏᑦᑐᖅ, (3) ᑲᓇᑕᒥ ᐊᐅᓚᑕᐅᓂᖏᑦ ᓴᓇᓯᒪᔪᑦ ᓇᓄᑦᑕᐅᔪᓐᓇᕐᓗᑎ ᐊᖑᑕᐅᓂᖏᑦ ᐊᕐᓇᐅᓂᖏᑦ 2:1 (ᐊᖑᑎᓂᑦ ᐊᕐᓇᓄᑦ) ᓇᓄᑦᑕᐅᔪᑦ ᐊᑯᑭᑦᑐᓂᑦ ᓂᕈᐊᕐᓂᖃᖅᓯᒪᖏᑦᑐᑦ, ᐊᒻᒪ (4) ᐊᒥᓲᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᐊᑯᑭᑦᑐᓂᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᑦ ᕿᒧᒃᓯᒃᑯᑦ ᑲᓇᑕᒥᓕ ᖃᒧᑦᑕᐅᔭᒃᑯᑦ.

ᓵᑉᑕ 9: ᓇᔪᖅᑕᐅᔪᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖏᑦ Kane Basin−ᒥᑦ

ᐊᕐᕌᒍᑕᒫᑦ ᓯᑯᖓ ᓇᔪᖅᑕᐅᖃᑦᑕᕐᓂᖓ KB−ᒥᑦ ᐊᓯᔾᔨᖅᓯᒪᔪᖅ ᐊᕐᕌᒍᓕᒫᒥ ᓯᑰᖏᓐᓇᖅᑐᒥ (>30% ᐊᐅᔭᒃᑯᑦ) ᐊᔾᔨᐸᓗᒋᓕᖅᑕᖓᓄᑦ ᐊᕐᕌᒍᑕᒫᑦ ᓯᑯᖃᑦᑕᕐᓂᖓᓄᑦ ᐊᐅᑦᑐᐃᓇᐅᓪᓗᓂ ᐊᐅᔭᒃᑯᑦ (<5% ᐱᔭᐅᓯᒪᔪᑦ). ᓯᑯᐃᖅᓴᕋᐅᓐᓂᖅᓴᐅᓕᖅᑐᖅ ᐅᐱᕐᖔᒃᑯᑦ 7−ᓂ ᐅᓪᓗᓂ/ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ, ᐊᒻᒪ ᓯᑯᓇᓵᕐᓂᓴᐅᓪᓗᓂ ᐅᑭᐊᔅᓵᒃᑯᑦ 5-6 ᐅᓪᓗᓂ/ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ, ᐊᒻᒪ ᐊᑯᓂᐅᓂᖓ ᐊᐅᔭᖓ (ᓲᕐᓗ, ᐅᓪᓗᐃᑦ ᐊᒥᓲᓂᖏᑦ ᓯᑯᐃᕐᓂᖓᓄᑦ ᐊᒻᒪ ᓯᑯᓂᖓᓄᑦ) ᖁᕝᕙᕆᐊᖅᓯᒪᔪᑦ 12 ᐅᓪᓗᓂ/ᖁᓕᓂᑦ ᐊᕐᕌᒍᓂᑦ. ᓯᑯᖃᑦᑕᕐᓂᕆᓚᔪᑦᑕᖓ ᔫᓂᒥ ᐅᑦᑑᐱᕆᒧᖅ ᐊᑎᓪᓕᕙᓪᓕᐊᔪᖅ 5-6% ᖁᓕᐅᓂᖅᑕᑕᒫᑦ ᐊᕐᕌᒍᓂᑦ.

KB ᓇᓄᖏᑦ ᖃᓄᐃᓕᓯᒪᔪᑦ ᐊᓯᔾᔨᐸᓪᓕᐊᓂᖓᓄᑦ ᓯᑯᖓ ᖃᓄᐃᓐᓂᕆᔭᖓ ᐅᖓᓯᓐᓂᖅᓴᒧᑦ ᓇᒧᖓᖅᐸᓕᖅᑐᑎᑦ ᐊᒻᒪ ᓇᔪᖅᐸᑦᑕᖏᑦ ᐊᑐᖃᑦᑕᖅᑕᖏᓐᓄᑦ ᐊᔾᔨᖃᐸᓗᑦᑐᑦ ᓇᓄᖏᓐᓄᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓯᑯᒦᑉᐸᑦᑐᑦ (ᒨᕐᓗ, ᐊᖏᓪᓕᒋᐊᖅᓯᒪᔪᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓇᔪᖅᐸᑦᑕᖏᑦ, ᑕᑯᒍᑦ ᓵᑉᑕ 2 ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖓ ᓯᑯᖃᖏᓐᓂᖅᓴᐅᓂᖓ ᐊᐅᔭᒃᑯᑦ ᐊᒻᒪ ᐅᑭᐊᔅᓵᒃᑯᑦ). ᐅᓪᓗᓂᑦ ᑎᓴᒪᓂᑦ ᐃᖏᕋᓂᖃᖅᑐᑦ KB−ᒥᑦ ᐊᑦᑎᓂᖅᓴᐅᔪᑦ ᓴᓂᐊᓂ BBᒥᑦ ᐊᒻᒪ ᖃᑦᑏᓐᓇᐅᓂᖅᓴᐅᓯᒪᔪᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐊᐅᓚᓂᖏᓐᓂᑦ. ᐊᖏᔪᓂᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖏᑦᑐᑦ ᑎᓴᒪᓄᑦ ᐅᓪᓗᑦ ᐃᖏᕋᓂᖏᓐᓄᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ KB−ᒥᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᕿᑎᐊᓂ ᐅᑦᑑᐱᕆᒥ ᖁᕝᕙᓯᓐᓂᖅᓴᐅᓚᐅᖅᑐᑦ r.

ᓄᓇ ᐊᑐᖅᑕᐅᓂᖓ KB−ᒥᒃ ᐊᐅᔭᒃᑯᑦ ᑕᐃᒫᔅᓴᐃᓈᖏᑦᑐᖅ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐃᓚᖏᑦ ᖃᖏᖅᑐᐃᑦ ᓯᑰᖏᓐᓇᕐᓂᖏᓐᓄᒃ ᐊᒻᒪ ᓯᔾᔭᖓ. ᓄᓇᒨᕐᓂᖏᑦ ᓄᓇᒦᒍᓐᓃᕐᓂᖏᓪᓗ ᐅᔾ;ᖏᑦ ᓇᓗᓇᐃᖅᑕᐅᔪᓐᓇᓚᐅᖏᑦᑐᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᐅᒻᒧᑦ.

ᐱᖓᓱᑦ ᑎᓲᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ KB−ᒥᑦ 2000−ᖏᓐᓂᑦ. ᐊᐅᓯᐃᑦᑑᑉ ᕿᑭᖅᑖᓗᐊᓃᑦᑐᐃᓐᓇᐅᓚᐅᖅᑐᑦ. ᐊᖏᔪᒥᑦ ᐊᔾᔨᒌᖏᓐᓂᖃᓚᐅᖏᑦᑐᑦ ᑎᓯᓕᐅᕐᓂᖏᓐᓄᑦ, ᐃᓯᕐᕕᖏᑦ ᐅᓪᓗᖏᑦ, ᐊᓂᕕᖏᓪᓘᓐᓃᑦ ᐅᓪᓗᖏᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ.

ᐊᑕᖏᖅᑐᒍ, ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ (ᓵᑉᑕ 2) ᐊᒻᒪ ᓇᔪᖅᑕᐅᔪᑦ ᖃᓄᐃᓐᓂᖏᑦ ᑲᑎᖓᖃᓯᐅᑎᓪᓗᑎ ᕿᑐᖏᐅᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᓃᑦ (ᓵᑉᑕ 11) ᐊᒻᒪ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ (ᓵᑉᑕ 12)ᒪ ᓇᓗᓇᐃᖅᓯᔪᑦ ᑖᒃᑯᐊ KB−ᒥ ᓇᓄᐃᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓯᑯᓯᐅᖏᓐᓂᖅᓴᐅᕙᓕᖅᑐᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ, 1990−ᖏᓪᓂᓂᑦ ᖁᕝᕙᕆᐊᖅᓯᒪᔪᓐᓇᖅᑐᖅ ᐊᑕᖏᖅᑐᒍ ᕿᑐᖏᐅᕐᕕᒋᕙᑦᑕᖏᑦ.

ᓵᑉᑕ 10: ᐊᐅᖏᑎᒎᖅᑐᓂ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓄᖏᓐᓂᑦ Kane Basin−ᒥ

ᓵᑉᑕ 10 ᓇᓗᓇᐃᔭᖅᓯᒪᔪᑦ ᐊᐅᖏᑎᒎᖅᑐᓂ MR ᖃᐅᔨᓴᕐᓂᐅᔪᒥ KB−ᒥ, ᖃᐅᔨᓴᖅᑐᑦ ᐃᓚᒋᓪᓗᓂᐅᒃ ᐊᑯᓂᐅᔪᒧᑦ (23-ᐊᕐᕌᒍᐃᑦ) ᑕᑯᔭᐅᓂᖏᑎᒍᑦ ᐊᐅᖏᑎᒍᓪᓗ MR ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ. ᑖᓐᓇ ᐱᖃᓯᐅᔾᔨᓚᐅᖅᑐᖅ ᐱᒋᐊᕐᓂᖏᓐᓂ ᐊᕐᕕᓂᓕᓐᓂᑦ ᐊᕐᕌᒍᓂᖅ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᑐᖁᓪᓚᔪᓂᑦ ᐅᑎᖅᑎᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ (1992 – 1997) ᑭᖑᓂᐊᒍᑦ 14-ᓄᑦ ᐊᕐᕌᒍᓄᑦ ᑐᖁᓪᓚᔪᓄᑦ ᐅᑎᖅᑐᑐᐊᑦ (1998 – 2011) ᒫᓐᓇᕋᑖᖑᓯᒪᔪᕐᓗ (2012 – 2014) ᐆᒪᔪᓂᑦ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᑐᖁᓪᓚᔪᓂᑦ ᐅᑎᖅᑎᑕᐅᓂᖏᓐᓄᑦ. ᑐᕌᒐᓚᓇᐅᖅᑕᐅᕗᑦ ᒥᔅᐅᔅᑕᐅᓯᒪᔪᑦ ᒫᓐᓇᐅᔪᒥ ᐊᒥᓱᓂᖏᑦ ᐊᒻᒪ ᐱᒻᒪᕆᐅᔪᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ ᐊᒻᒪ ᐊᔾᔨᒋᑎᙳᐊᕐᓗᒋᑦ ᐅᓪᓗᒥᒨᖓᔪᑦ ᒥᔅᐅᔅᑕᐅᓯᒪᔪᑦ ᑖᒃᑯᓂᖓ ᓴᖅᑭᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᓯᒪᔪᓂᑦ (Taylor et al. 2008).

ᒥᔅᐅᔅᑕᐅᔪᑦ ᐊᒥᓱᓂᖏᓐᓄᑦ KB−ᒥ 357 ᓇᓄᐃᑦ (95% CI: 221 – 493) 2013 – 2014−ᒧᑦ. ᐋᖅᑭᑕᐅᒃᑲᓂᖅᑐᑦ 1990−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᒥᔅᐅᔅᑕᐅᓯᒪᔪᑦ 224 ᓇᓄᐃᑦ (95% CI: 145 – 303) ᐊᕐᕌᒍᓄᑦ 1995 – 1997. ᑐᙵᓂᖅᑐᑎ MR, ᐊᖏᓂᖓ KB

ᐊᒥᓃᓂᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᓯᒪᓚᐅᖅᑐᑦ 164 ᓇᓄᐃᑦ (95% CI: 94-234) ᐊᕐᕌᒍᓄᑦ 1994-1997 (ᓇᓗᓇᐃᖅᑐᒍ ᑖᓐᓇ ᒥᔅᐅᓴᑦᑕᐅᔪᖅ ᐊᑐᖅᑕᖅ ᐊᔾᔨᒋᖏᑦᑐᓄᑦ ᐊᕐᕌᒍᓄᑦ ᓴᓂᐊᓂ ᖃᐅᔨᓴᐃᒃᑲᓂᕐᓂᖑᔭᑦᑎᓐᓂᑦ; Taylor et al. 2008).

ᐊᒥᓃᓂᖅᓴᐃᑦ ᓇᓄᐃᑦ ᑎᑎᕋᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᓇᓐᓇᖓᓂ KB ᐊᕐᕌᒍᖏᓐᓂ 2012 – 2014 ᓴᓂᐊᓂ 1994-1997. ᑲᓇᓐᓇᖓ Kane Basin ᕿᓂᕐᕕᐅᓚᐅᖅᑐᖅ 1990–ᖏᓐᓂᑦ ᕿᓂᖏᓂᖅᓴᐅᓪᓗᑎ 2010–ᖏᓐᓂᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᓇᓄᖃᑦᑎᐊᖏᓐᓂᖕᒪᓄ ᖃᐅᔨᔭᐅᔪᓂᑦ ᑕᐃᑲᓂ. ᓇᑦᑎᖅᑕᖃᕐᓂᖕᒪ ᑲᓇᓐᓇᖓᓂ Kane Basin ᐅᐱᕐᖔᒃᑯᑦ 1990–ᖏᓐᓂᑦ ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᑦ ᑕᒪᓐᓇ ᓇᓄᕐᓄᑦ ᓇᔪᕋᔅᓴᑦᑎᐅᕙᐅᓪᓗᓂ (Taylor et al. 2001). ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᓇᒧᖕᒪᖃᑦᑕᕐᓂᖏᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂ ᐊᒻᒪ 2010–ᖏᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔫᓇᖅᑐᖅ ᐊᔾᔨᒌᖏᓐᓂᖕᒪᓂ ᓇᒧᖕᒪᖃᑦᑕᕐᓂᖏᑦ ᓇᓄᐃᑦ, ᐊᖑᓇᓱᑦᑕᐅᖏᓐᓂᖕᒪᔫᓄᔅᓴᐅᑐᐃᓐᓇᕆᐊᓕ ᐊᑯᑭᑦᑐᒧᑦ ᑲᓇᓐᓇᖓᓂ KB ᐊᒻᒪ ᑕᐃᒪᐃᓐᓂᖕᒪᓄᑦ ᐊᒥᓱᕈᕆᐊᒃᑲᓂᖅᓯᒪᔪᑦ ᓇᒧᓄᐃᑦ KB–ᒥᑦ, ᑭᓯᐊᓂᑦᑕᐅᖅ ᐃᓚᖏᓐᓂᑦ ᐊᔾᔨᒌᖏᓂᖃᖅᑐᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᓂᑦ.

2013 – 2014 ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ 357 (221 – 493) ᓇᓄᐃᑦ ᒫᓐᓇᐅᔪᒥᑦ ᓇᔪᖅᓯᖃᑦᑕᖅᑐᑦ KB–ᒥᑦ ᐅᐱᕐᖔᒃᑯᑦ (ᓃᕐᓗ, Kane Basin ᓇᓄᓕᒫᖏᑦ; Kendall et al. 1997), ᐊᒻᒪ ᒫᓐᓇᐅᔪᒥᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᖁᑦᑎᓂᖅᓴᐅᔪᑦ ᒥᔅᐅᓴᑦᑕᐅᖃᑦᑕᖅᓯᒪᔪᑐᖃᕐᓂᑦ. ᑐᓐᖓᓂᖃᖅᑐᑎᑦ ᑕᒫᖕᒐᑐᐃᓐᓇᖅ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᓇᒧᖕᒪᖃᑦᑕᕐᓂᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᑦ, ᐊᔾᔨᒌᖏᓂᓪᓗᐊᑕᕙᑦᑕᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ KB–ᒧᑦ ᐊᒥᓃᓂᖏᓐᓄᑦ 2013-2014 ᐊᒻᒪ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1995-1997 ᖃᓂᒋᔭᐸᓗᖏᓐᓃᓚᐅᖅᑐᑦ 133 ᓇᓄᐃᑦ (ᐊᑐᖅᑕᐅᕙᔪᑦᑐᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ≈ 80 ᓇᓄᐃᑦ), 95% ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᒥᓃᑑᓂᖏᓐᓄᑦ ᐊᓯᔾᔨᖅᑐᑦ ᒪᕐᕉᓐᓂ ᖃᐅᔨᓴᕐᕕᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ ᐱᐅᔫᔪᓐᓇᓚᐅᖅᑐᖅ. ᑖᓐᓇ ᓇᓗᓇᐃᖅᓯᔪᖅ ᓴᖏᔪᓂᒥᑦ ᓇᓗᓇᐃᖅᓯᒍᑎᖃᖅᑐᓂ ᐊᒥᓱᖅᐸᓪᓕᐊᓂᖏᓐᓂᑦ, ᐊᒻᒪ ᒪᓕᓐᓂᖃᖅᑐᖅ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ ᓇᒧᖕᒪᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᖃᓄᐃᓐᓂᖓ ᐊᒻᒪ ᕿᑐᓐᖏᕈᖃᑦᑕᕐᓂᖏᑦ. ᐅᔾᔨᖅᓱᕐᑯᔾᔪᒍᑦ ᓇᓗᓇᐃᖏᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᒥᓱᕈᒃᑲᓂᕐᓂᖏᓐᓄᑦ ᐊᖏᓪᓕᒋᐊᖅᑕᐅᔪ)ᓐᓇᕐᓂᖕᒪᓄᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᑯᓂᐅᓂᖕᒪ ᐊᒻᒪ ᐊᔾᔨᒌᖏᓐᓂᖏᑦ ᖃᐅᔨᓴᕐᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᔪᑦ 1990–ᖏᓐᓂᑦ ᓴᓂᐊᓂ 2010–ᖏᓐᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ.

ᒫᓐᓇᐅᔪᒥᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᖃᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐊᕐᕌᒍᓕᓐᓄᑦ 3+ ᐊᕐᓇᐃᑦ (0.95; SE: 0.04) ᐊᒻᒪ ᐱᓯᒪᔭᐅᔪᑦ ᓇᓄᐃᑦ ᒪᓕᓐᓂᖃᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕐᓂᐅᕋᑖᖅᓯᒪᔪᒥᑦ, ᑭᓯᐊᓂ ᓄᑖᖅ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᑦᑐᒋᑦ ᐆᒪᓂᖏᓐᓄᑦ ᐊᑦᑎᓂᖅᓴᐅᓪᓗᑎ ᐊᕐᕌᒍᓕᓐᓄᑦ 3+ ᐊᖑᑎᑦ (0.87; SE: 0.06). ᐊᑯᓂᐅᔪᒧᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᖅᐸᑦ, ᐱᔭᐅᔪᖃᓚᐅᖏᑦᑐᖅ ᐊᐅᖏᑎᒍᐸᓪᓘᓐᓃᑦ ᖃᐅᔨᓴᖅᑐᑎᑦ ᐊᒻᒪ ᖃᑦᑏᓐᓇᓂᑦ ᐅᑎᓚᐅᖅᑐᑦ 1998-2011, ᐊᒻᒪ ᐊᒥᓱᓚᐃᑦ ᐊᒥᖏᑦ ᐊᕙᑎᖕᒪᓄ, ᖃᐅᔨᓴᕐᓂᖅ, ᐊᒻᒪ ᐃᖅᑲᓴᐃᔩᖑᔪᓄᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᖃᓄᐃᓐᓂᖅᑎᑦᑎᔫᓇᖅᑐᑦ.

ᑎᑎᕋᖅᓯᓚᐅᖅᑐᒍᑦ ᖃᑦᑏᓐᓇᕈᓇᐊᖅᑐᓂᑦ ᑐᑯᖃᑦᑕᖅᑐᓂᑦ ᐊᑦᑐᐊᓂᓖᑦ ᓈᓄᑦᑕᐅᕙᑦᑐᓄᑦ, ᐱᔾᔪᑎᖃᑐᐃᓐᓇᕆᐊᓕ ᐊᑐᓕᖅᑎᑦᑕᐅᓂᖕᒪᓄᑦ ᐊᑯᑭᑦᑐᓂᑦ ᓈᓄᑦᑕᐅᔫᓇᖅᑐᑦ 2006–ᒥ. ᓄᓇᖏᑎᒍᑦ ᐊᑐᖅᑕᐅᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ KB–ᒥ ᓇᓄᐃᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᖁᕝᕙᓯᑦᑐᒥ ᑐᓐᖓᕕᒋᔭᔅᓴᐅᔪᓂᑦ ᐅᐱᕐᖔᒃᑯᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ, <4% ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᓂᑦ ᓇᓄᕐᓂᑦ ᓅᑦᑐᑎ. ᑖᓐᓇ ᒪᓕᑦᑐᖅ ᓇᒧᖕᒪᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓵᑕ 2–ᒦᑦᑐᑦ.

ᓵᑕ 11: ᖃᖓᑕᓰᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓇᓄᖏᓐᓂ Kane Basin

ᖃᖓᑕᓰᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖃᓚᐅᖅᑐᑦ (ᓵᑕ 10). ᐊᑐᖅᑕᐅᔪᖅ ᒪᓕᓐᓂᖃᓚᐅᖅᑯᑐᖅ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ ᐊᑯᓂᐅᓂᖓᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᕐᕕᐅᔪᓂᑦ KB–ᒥᑦ ᐊᒻᒪ ᐅᕙᑦᑎᓐᓂ ᓇᒻᒥᓂᖅ ᒥᔅᐅᓴᒍᓐᓇᖅᑎᑎᓪᓗᑎ ᐊᒥᓃᓂᖏᓐᓂᑦ ᒪᕐᕉᓐᓂᒃ ᐊᑐᖅᑕᐅᔫᓐᓂᒃ, ᐅᕙᑦᑎᓂᑦ ᐊᔾᔨᒌᖑᐊᖅᑎᑦᑎᔫᓇᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, ᐃᓱᒪᒌᓐᓇᖅᑐᒍᑐᑭᓯᒋᔭᖓ "ᖃᐅᔨᓴᕐᓂᖅ ᐊᒥᓃᓂᖏᓐᓂᑦ" ᐊᔾᔨᒌᖏᒍᓐᓇᖅᑐᑦ ᒪᕐᕉᓐᓂᒃ ᐊᑐᖅᑕᐅᔫᓐᓂᒃ.

ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᑦ ᑐᓐᖓᓂᖅᑰᑎᑦ ᐅᐱᕐᖒᒃᑯᑦ 2014−ᒥ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ KB−ᒥᑦ 206−ᖑᓚᐅᖅᑐᑦ ᓇᓄᐃᑦ (95% ᑎᑎᕋᖅᑕᐅᔪᑦ CI: 83 - 510). ᑭᓯᐊᓂᓕ, ᖃᐅᔨᓴᖅᑕᐅᑎᐊᖏᓐᓂᖓᓄᑦ ᐃᒪᐃᒻHᕐᒥ ᓇᓄᕐᓄᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᓂᑦ ᑖᓐᓇ ᒥᔅᐅᓴᑦᑕᐅᔪᖅ ᐱᐅᖏᑦᑐᒥᑦ ᐱᖃᑕᐅᖏᑎᑦᑎᔪᖅ. ᑐᓐᖓᓂᖅᑰᑎᑦ ᑕᒫᖓᑐᐃᓐᓇᖅ ᐊᑐᖅᑕᐅᔪᓂᑦ ᐃᓱᒪᒋᔭᐅᔪᑦ ᐊᑐᖅᑕᐅᒐᔪᑦᑐᑦ ᐊᒥᕉᓂᖏᓐᓄᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᑦ, ᐊᔾᔨᒋᖏᓐᓂᕆᒐᔪᑦᑕᖏᑦ ᒥᔅᐅᓴᑦᑕᐅᓯᒪᔪᓂᑦ ᑲᑎᑦᑐᒋᑦ ᐊᒥᕉᓂᖏᑦ 2013-2014 MR ᖃᐊᐅᔨᓴᕐᓂᐅᔪᒥᑦ (357 ᓇᓄᐃᑦ, 95% CI = 221 – 493) ᐊᒻᒪ ᖃᖓᑕᓲᒃᑯᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᑦ ᖃᓂᒋᔭᖏᓐᓃᓚᐅᖅᑐᑦ 151 ᓇᓄᐃᑦ ᐊᔾᔨᒋᖏᓐᓂᕆᒐᔪᑦᑕᖏᑦ ≈ 127 ᓇᓄᐃᑦ), 88%−ᒥ ᖃᐅᔨᓴᒃᑕᓂᕐᒥᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᔾᔨᒋᖏᓐᓂᖏᑦ ᐱᐅᔫᓚᐅᖅᑐᑦ (ᓲᕐᓗ, MR ᒥᔅᐅᓴᑦᑕᐅᔪᑦ ᐊᑕᐅᓯᕐᒥᑦ ᓇᓄᕐᒥᑦ ᐊᖏᓂᖅᓴᐅᓚᐅᖅᑐᖅ ᓴᓂᐊᓂ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ).

ᐊᔾᔨᒋᖏᓐᓂᖏᑦ MR ᐊᒻᒪ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᑦ KB−ᒥ ᖃᐅᔨᒪᔭᐅᑦᑎᐊᕆᐊᓖᑦ ᐊᔾᔨᒻᖑᐊᖅᑎᑕᐅᓗᑎ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓄᑦ ᐊᔾᔨᖏᑕᖏᓐᓂᑦ ᐊᑐᖅᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᓂᕐᒧᑦ ᐅᐱᕐᖒᒃᑯᑦ. ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᑕᑯᑎᑦᑎᓚᐅᖅᑐᑦ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ (ᓲᕐᓗ, ᐊᒥᕉᓂᖏᑦ ᓇᓄᐃᑦ ᓇᔪᖅᓯᔪᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᒥᑦ). NR ᐊᑕᖐᑎᑦᑎᓪᓗᒍ ᒥᔅᐅᓴᑦᑕᐅᔪᓂᑦ ᑕᑯᔅᐅᑎᑦᑎᔪᖅ ᓇᓄᓕᒫᓂᑦ ᐱᔭᐅᖏᑐᖅᑕᖃᕈᓐᓇᖏᑦᑐᑎᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᒧᑦ). ᐅᖃᐅᔾᔨᒋᐊᖅᑐᒍᑦ MR ᒥᔅᐅᓴᑦᑕᐅᔪᑦ ᓈᒻᒪᑦᑐᑎ ᐊᑐᖅᑕᐅᓗᑎ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ.

ᑕᑯᔅᐅᑎᑕᐅᒻᒥᔪᖅ ᐊᓯᖏᓐᓂᑦ ᐅᑭᐅᑕᖅᑐᒥ (ᓲᕐᓗ Barents Sea; Aars et al. 2009) ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐊᑑᑎᖃᖅᑐᒥ ᓴᓇᕐᕈᑎᖃᖅᑎᑦᑎᔪᑦ ᑎᑎᕋᖅᑕᐅᓂᖏᓐᓄᑦ ᐊᒥᕉᓂᖏᑦ. ᐊᑐᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᖅ ᑲᔪᓯᑦᑎᐊᕐᓂᖃᖅᓯᒪᔪᑦᐊᖏᓂᖅᓴᐅᔪᓂᑦ ᐊᒥᕉᓂᖏᓐᓂᑦ ᓴᓂᐊᓂ KB. ᐅᐱᕐᖒᒃᑯᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ KB−ᒥ ᐊᑐᓕᖅᑎᑕᐅᑦᑎᐊᓚᐅᖅᑐᖅ ᒥᑭᔫᓂᖓᓄᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᖅ ᐊᒻᒪ ᓯᓚᑦᑎᐊᕙᐅᒃᑲᕐᓂᖓᓄᑦ, ᑭᓯᐊᓂ ᐱᐅᓯᑎᕆᐊᖅᑕᐅᑲᓐᓂᕈᓐᓇᖅᑐᑦ ᐊᖏᓪᓕᕆᐊᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᒥᔅᐅᓴᑦᑎᐊᓂᖅᓴᓇᓂᐊᕐᓗᒋᑦ ᖃᐅᔾᔪᑕᐅᔪᑦ ᐊᒻᒪ ᐊᑕᖏᖅᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᑯᐅᔪᖅ ᐊᒥᕉᓂᖏᓐᓄᑦ.

ᓵᑉᑕ 12: ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦᒃ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅ)ᔪᓄᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᑦ ᓇᓄᐃᑦ Kane Basin−ᒥ

ᓵᑉᑕ 12 ᓇᓗᓇᐃᔭᐃᔪᖅ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ KB−ᒧᑦ ᐊᑐᖅᑐᑎ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ MR ᖃᐅᔨᓴᕐᓂᐅᔫᓐᓂᒃ ᐱᒋᐊᖅᑐᓂ 1992 ᑎᑭᑦᑐᒍ 1997 (Taylor et al. 2008) ᐊᒻᒪ 2012 ᑎᑭᑦᑐᒍ 2014 (ᓵᑉᑕ 10). ᖃᐅᔨᓴᕈᓯᒥᓃᑦ ᓇᓗᓇᐃᖅᓯᑎᑦᑎᑲᓐᓂᖅᑐᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ MR ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᒻᒪ ᖃᐅᔨᓴᓂᕐᒥᑦ ᒫᓐᓇᐅᔪᒥᑦ ᖃᓄᐃᓕᖓᓂᖓ KB−ᒧ ᐊᒥᕉᓂᖏᑦ.

ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ KB−ᒧᑦ, ᐱᖃᓯᐅᑎᓪᓗᑎ ᐊᒥᕉᓂᕆᒐᔪᑦᑕᖏᑦ ᐊᑎᖅᑕᖏᑦ ᐊᕐᕌᒍᒧᑦ (COY) ᐊᒻᒪ ᐊᑎᖅᑕᓛᕐ, ᐊᒻᒪ ᑎᑎᕋᖅᓯᒪᔪᑦ ᐱᖃᓯᐅᔾᔨᓂᕐᒧᑦ (ᐋᖅᑭᑦᑕᐅᓯᒪᔪᑦ ᐊᒥᕉᓂᖏᓐᓄᑦ ᐊᑎᖅᑕᐃᑦ ᐊᑐᓂ ᐊᑕᐅᓯᕐᒧᑦ ᐊᕐᓇᓪᓗᒧᑦ MR−ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᒥᑦ), ᐊᔾᔨᒌᓐᖑᐊᖅᑎᓪᓚᔅᐅᓚᐅᖅᑐᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2010−ᖏᓐᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ. ᐊᒥᕉᓂᕆᒐᔪᑦᑕᖏᑦ ᐊᑎᖅᑕᐃᑦ KB−ᒥ (COY−ᒧᑦ: 1.67 1990−ᖏᓐᓂ ᐊᒻᒪ 1.60 2010−ᖏᓐᓂᑦ) ᐊᔾᔨᐸᓗᒋᓚᐅᖅᑕᖏᑦ ᖃᐅᔨᔭᐅᔪᓄᑦ ᐊᓯᖏᓐᓄᑦ ᓇᓄᕐᓄᑦ ᐊᒥᕉᓂᖏᓐᓄᑦ ᕿᑭᖅᑕᒐᓚᓐᓂᑦ (ᐊᑯᓐᓂᖏᓐᓂ 1.65 - 1.71). ᐊᕐᕌᒍᑕᒫᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᒥᑭᓗᐊᖃᑦᑕᓚᐅᖅᑐᑦ ᐃᑲᔪᖅᑐᐃᓗᑎ ᐊᔾᔨᒌᓐᖑᐊᖅᑎᓪᓚᔅᐅᓗᒋᑦ ᐊᕐ ᕌᒎᓯᒪᔪᓂᑦ. ᑭᓯᐊᓂᓕ, ᐊᕐᕌᒍᓄᑦ ᐊᖏᒌᖑᓯᒪᔪᑦ ᖃᐅᔨᓴᕐᕕᐅᔪᑦ ᐊᖏᓂᖏᑦ (1995, 2013, ᐊᒻᒪ 2014), ᑲᑎᑦᑐᒋᑦ ᓇᓄᐃᑦ ᐊᒥᕉᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐅᖓᑕᐅᔾᔨᓚᐅᖅᑐᑦ 50−ᓂᑦ ᖃᐅᔨᔭᔅᓴᐅᓚᐅᖅᑐᑦ ᐊᔾᔨᒋᖏᓐᓂᖏᑦ ᐃᓚᖏᓐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᓇᓄᕐᓂᑦ ᐊᑎᖅᑕᐅᓚᐅᖅᑐᑦ ᐊᕐᕌᒍᒧᑦ (ᐊᑯᓐᓂᖏᓐᓂ 15% - 30%) ᓇᓗᓇᐃᖅᓯᔪᖅ ᐊᖏᔪᒥᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᔾᔨᒋᖏᓐᓂᖏᑦ ᐊᑎᖅᑕᑖᖃᑦᑕᕐᓂᖏᑦ. ᑕᑯᓐᓇᖅᑐᒧᐃᓚᖏᑦ ᐊᑎᖅᑕᐃᑦ ᐊᒻᒪ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐊᔾᔨᒋᖏᓐᓂᖃᓚᐅᖏᑦᑐᑦ ᐊᕐᕌᒎᔪᓂᑦ

ᓇᓗᓇᐃᖅᑐᒍ, ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᓇᓗᓇᐃᕈᑎᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᕿᑐᙱᐅᖃᑦᑕᕐᓂᖏᓐᓄᑦ KBᒧᑦ ᐊᑯᓐᓂᖏᓐᓂ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2000–ᖏᓐᓂᑦ, ᑐᑭᓯᒋᐊᕈᑎᔅᓴᖅᑕᖃᑦᑎᐊᖏᓐᓂᖓᓄᑦ ᓄᖅᑲᐅᓯᕐᓂᖃᖏᑦᑐᖅ.

ᓵᑉᑕ 13: ᖃᐅᔨᓴᐅᔪᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᓄᓇᒥ ᓂᕆᓇᓲᓐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ Kane Basin–ᒥ ᐅᐱᕐᖔᒃᑯᑦ

ᐊᑐᖅᑐᒍ ᐅᖅᓱᖃᕐᓂᖏᓐᓄᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ (ᑕᑯᓗᒍᑦᑕᐅᖅ ᓵᑉᑕ 7) ᐊᔾᔨᒌᙳᐊᖅᑎᓚᐅᖅᑕᕗᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ ᓇᓄᐃᑦ KB–ᒥᑦ 1990–ᖏᓐᓂᑦ ᐊᒻᒪ 2010–ᖏᓐᓂᑦ. ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐱᐅᓂᖅᓴᐅᓚᐅᖅᑐᑦ KB-ᒥ ᐃᓐᓇᕐᓂᑦ ᐊᕐᓇᓂᑦ ᐊᑎᖅᑕᓕᓐᓂᑦ 2010–ᖏᓐᓂᑦ ᐊᔾᔨᒋᐸᔪᑦᑕᖏᑦ 1990–ᖏᓐᓂᑦ. ᑭᓯᐊᓂ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓂᖏᑦ ᑭᓪᓕᖃᖅᑎᑕᐅᓚᐅᖅᑐᑦ. ᑕᑯᓐᓇᖅᑐᒍ, ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐃᓐᓇᖏᓐᓂᑦ ᓇᓄᕐᓂᑦ ᐊᖑᑎᓂᑦ, ᐃᓐᓇᓂᑦ ᐊᕐᓇᓂᑦ Coy–ᖃᖅᑐᑦ, ᐊᒻᒪ ᐃᓄᑑᔪᑦ ᐃᓐᓇᐃᑦ ᐊᕐᓇᐃᑦ ᐊᔾᔨᒌᐲᐸᕋᓗᓚᐅᖅᑐᑦ ᑕᕝᕙᓂ ᖃᐅᔨᓴᕐᓂᐅᔫᓐᓂᒃ.

ᖃᐅᔨᓴᔅᓴᐅᔪᑦ ᐱᐅᓯᕚᓪᓕᖅᑐᑦ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ 2010–ᖏᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᖅ ᐃᒻᒥᒃᑯᑦ ᐊᔾᔨᒌᖏᓐᓂᕆᖃᑦᑕᖅᑕᖏᑦ ᐅᕝᕙᓘᓐᓂᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐊᑯᓂᐅᔪᒧᑦ ᐊᓯᔾᔨᖅᑐᒧᑦᓯᑯᖓᓂ KB–ᒥ, ᐊᔾᔨᖃᐸᓗᓕᖅᑐᖅ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᓯᑯᖃᕈᓐᓃᖃᑦᑕᕐᓂᖓ. ᓇᑦᑏᑦ ᐊᒻᒪ ᓇᓄᐃᑦ ᐊᔾᔨᒌᐸᓗᓐᓂᑦ ᓇᒧᖓᖃᑦᑕᓚᐅᖅᑐᑦ Kane Basin. ᐊᒥᓱᒻᒪᕆᐅᓚᐅᖅᑐᑦ ᑕᒪᒃᑭᓐᓂ ᓇᑦᑏᑦ ᐊᒻᒪ ᓇᓄᐃᑦ ᐅᐊᓐᓇᖓᑕ ᑲᓇᓐᓇᖓᓂ Kane Basin, ᖃᓂᒋᔭᖓᓂ Humbolt Glacier (ᖃᓇᓐᓇᖓᓂ KB), ᓇᓗᓇᐃᖅᓯᔪᖅ ᑕᒪᓐᓇ ᖁᕝᕙᓯᑦᑐᒥᑦ ᐆᒪᔪᓕᐅᑦᐸᑦᑐᑦ ᐊᒻᒪ ᐱᒻᒪᕆᐅᓪᓗᓂ ᓇᓄᕐᓄᑦ ᓇᔪᖅᑕᐅᓪᓗᓂ ᓇᔪᖅᑕᐅᓗᓂ ᓇᓄᕐᓄᑦ.

ᐊᒥᓲᔪᑦ ᐃᒪᖏᑦ KBᑉᒥ ᓇᓄᐃᑦ ᖃᐅᔨᓴᐅᓚᐅᑦᖅᑐᑦ ᒥᖅᑯᐃᓴᖃᑦᑕᖅᓯᒪᒻᒪᕆᑦᑐᑎ ᐊᒻᒪ ᐅᕕᓂᖏᑦ ᐋᓴᕐᓂᖏᑦ ᐃᓯᒐᖏᑕ ᐊᑖᒍᑦ. ᑖᒃᑯᓄᖓ ᐱᔾᔪᑕᐅᔪᑦ ᖃᐅᔨᒪᔭᐅᖏᑦᑐᖅ ᖃᐅᔨᓴᖅᑎᓄᑦ. ᐃᓄᓐᓄᑦ ᖃᐅᔨᒪᔭᐅᔪᓂᑦ ᐅᖃᐅᔾᔨᔪᓂᑦ ᑕᒪᓐᓇ ᑕᐃᒪᐃᓐᓂᕆᔭᖓ ᐊᕐᖏᖃᑦᑕᖅᑐᓯᖅ ᐱᓯᑦᑐᑎ ᐊᒻᒪ ᐊᖕᒪᑦᑐᑎᑦ ᑎᓯᔪᒥᑦ, ᓂᓚᒥᑦ, ᒪᓃᑦᑐᒥᑦ ᐊᐳᒻᒥᑦ ᐅᐱᕐᖔᒃᑯᑦ ᓯᑯᒃᑯᑦ ᐃᒪᒋᖅᑲᓯᐅᑎᔭᖓ ᓅᖃᑦᑕᓂᖅᓴᐅᓂᖏᓐᓄᑦ ᓄᓕᐊᖅᑎᓪᓗᒋᑦ ᐊᒻᒪ ᓂᕆᓇᓯᑦᑎᓪᓗᒋᑦ.

ᓵᑉᑕ 14: ᐅᖃᐅᓯᔅᓴᐃᑦ ᐊᒻᒪ ᐅᖃᐅᔾᔨᒋᐊᕈᑎᑦ ᐊᐅᓚᑦᑎᖃᑎᒌᑦᑐᓂᑦ

ᖃᐅᔨᓴᐃᓂᖅ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖓ

BB ᐊᒻᒪ KB–ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᓯᕗᓪᓕᖅᐹᒥ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᑦ–ᐱᔭᐅᒃᑲᓂᖅᑐᓂ ᐊᑐᓕᖅᑎᑕᐅᓯᒪᔪᖅ ᐊᒥᓲᓂᓕᒫᖏᓐᓄᑦ ᓴᓴᐅᓴᑦᑕᐅᓗᑎ ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᓇᒦᓂᖏᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ. ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᖁᑭᐅᑦᑕᐅᔪᓄ ᑲᔪᓯᑦᑎᐊᕐᓂᖃᓚᐅᖅᑐᖅ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᖃᐅᔨᓴᕋᓱᒋᓂ ᓈᒻᒪᑦᑐᓂ ᓇᓗᓇᐃᖅᑕᕐᓂᖏᓐᓄᑦ ᑕᒪᒃᑭᖏᓐᓄ ᓴᓂᕈᑎᐅᑉ ᐃᒪᖓ ᐊᒻᒪ Kane Basin. ᖃᐅᔨᓴᖅᑎᓯᓂᑦ ᐱᖃᑦᑕᕐᓂᖅ ᓯᑲᑦᑑᔪᖅ, ᐊᐅᓚᓂᖃᑎᖃᖅᑐᓂ, ᐊᒻᒪ ᐸᕝᕕᓵᖏᓐᓂᓴᐅᓕᕐᓗᓂ ᓴᓂᐊᓂ ᐱᔭᐅᔪᓂᑦ, ᓇᓄᐃᑦ ᐊᐅᓚᒍᓐᓇᐃᓕᑎᑕᐅᕙᖕᒥᒪᑕ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᐊᑑᑎᖃᖅᑎᑦᑎᔪᓇᖅᑐᖅ, ᐸᕝᕕᓵᖏᓐᓂᓴᐅᓗᓂ ᓴᓇᕿᑎ.

ᑭᓯᐊᓂᓕ ᐊᓯᖏᑦ ᖃᓄᐃᓕᓂᖅᑲᓘᖅᑐᑦ ᑖᔅᓱᒧᖓ ᐊᑐᖅᑕᐅᔪᒧᑦ ᐃᓚᖏᑕ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᑐᐃᓐᓇᐅᑎᑦᑎᖏᑦᑐᑦ ᓇᓗᓇᐃᖅᓯᑎᐊᕐᓂᖅᓴᐅᐸᐅᖅᑐᓂᑦ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ (ᑭᒍᑎᖏᑦ ᐊᕐᕌᒍᖏᑦ ᓇᓄᕐᓂᑦ ᓇᓗᓇᐃᖅᓯᑦᑎᐊᒃᑲᓐᓂᐊᖅᑎᑦᑎᔪᖅ ᐊᕐᕌᒍᓂᑦ ᓴᖅᑭᑎᑦᑎᔾᔪᑎᖏᓐᓂᑦ,ᑐᑭᓯᒋᐊᕈᑎᔅᓴᖅᑕᖃᑦᑎᐊᕐᓂᖓᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ ᖃᓄᐃᓕᖓᓂᕆᔭᖏᑦ, ᐊᖏᔪᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᖅᑐᑦ ᓄᐊᑦᑎᓪᓗᒋᑦ). ᐃᓚᒋᐊᕐᓗᒍ, ᐃᓚᖏᑦ ᐆᒪᔪᑦ ᐱᔭᐅᔪᑦ ᐱᒻᒪᕆᐅᓚᐅᖅᑐᑦ ᑐᑭᓯᒋᐊᕈᑎᖅᑎᑎᓪᓗᐊᑕᖅᑐᑎᑦ ᑖᔅᓱᒧᖓ ᖃᐅᔨᓴᓂᕐᒧᑦ. ᓇᒧᖓᑦᑕᕐᓂᖏᑦ ᑐᑭᓯᒋᐊᕈᑎ ᑦ ᖁᓪᓗᐊᑦᑎᑕᐅᓯᒪᔪᕐᒧᑦ ᐱᔭᐅᓚᐅᖅᑐᑦ ᓇᓄᕐᓂᑦ ᐅᐊᓐᓇᖓᓂ ᐱᖓᓐᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ ᐊᑐᖅᑕᐅᓚᐅᖅᑐᑦ

ᓇᓗᓇᐃᔭᐃᑎᓪᓗᒋᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ, ᑕᒪᒃᑭᓐᓂ ᐃᓱᒪᒋᔭᐅᔪᑦ ᒪᒧᖅᑕᖏᓐᓂᑦ ᐊᓯᔾᔨᖑᓂᑦ ᑭᓯᐊᓂ ᖃᐅᔨᒃᑲᐃᓪᓗᓂ ᖃᐅᔨᓴᐃᓂᕐᒧᑦ.

ᐊᖏᔪᑦ ᐱᓕᕆᐊᑦ ᐊᔾᔨᖓᑎᑐᑦ BB−ᒥ ᐊᒻᒪ KB−ᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐊᑭᑐᔪᑦ ᐊᒻᒪ ᐃᖅᑲᓇᐃᔭᐅᑎᓄᑦ ᐊᔅᓱᕈᓐᓇᖅᑐᓂ ᐊᒻᒪ ᑭᓪᓕᖅᑎᑕᐅᓪᓗᑎ ᓴᖅᑭᖅᑕᑦᑕᕐᓂᖏᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᒐᕙᒪᒃᑯᕐᕕᐅᔪᓂᑦ ᐱᓕᕆᐊᕆᔭᓐᓇᖅᑕᖏᑦ. ᐊᒥᓲᓂᖏᓐᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᖃᑦᑕᖅᑲᑕ ᑕᑯᓐᓇᖅᑕᐅᓗᑎ 10-15 ᐊᕐᕌᒍᓂᑕᒫᑦ BB-ᒥ ᐊᒻᒪ KB−ᒥᑦ, ᑕᐃᒪ ᐊᑐᓕᕐᑯᔭᐅᕗᖅ ᒪᑭᒪᑎᓪᓗᒋᑦᒃ ᐊᒻᒪ ᐊᒥᓱᕈᖅᑎᒋᐊᕐᓗᒋᑦ ᓇᓗᓇᐃᒃᑯᑕᐃᑦ ᐊᒥᓲᓂᖏᓐᓂ ᐊᑕᐅᓯᐊᕐᓗᑎ ᐊᕐᕌᒍᒥ ᖃᐅᔨᓴᖃᒃᑕᕐᓗᑎ 5-7 ᐊᕐᕌᒍᓂᑕᒫᑦ. ᑕᒪᓐᓇ ᓇᓗᓇᐃᖅᓯᓂᐊᖅᑐᖅ ᓇᓗᓇᐃᒃᑯᑕ ᐊᐃᖏᑎᑰᖅᑐᓂ ᓇᓗᓇᐃᒃᑯᑕᓐᓃᑦ ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ ᐅᑎᖅᑎᑕᐅᓗᑎᑦ ᓇᓐᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓄᑖᒥ ᐊᒥᓲᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᐅᓚᓂᖅᑦᑎᐊᓂᖅᓴᒥᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐃᓚᐅᑎᑦᑎᖏᓐᓂᖅᓴᐅᖏᑦᑐᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓂᑦ ᐱᒻᒪᓇᐅᔪᓂᑦ ᖁᕝᕙᓯᓐᓂᓇᕆᐊᖅᑕᓴᖏᑦ.

ᑭᓯᐊᓂᓕ, ᖃᐅᔨᒪᔭᐅᔭᕆᐊᖃᕐᒥᔪᖅ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐱᖓᓱᓄᑦ ᐊᕐᕌᒍᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᔭᕆᐊᓖᑦ ᑐᓐᖓᕙᖃᖅᑐᓂᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓂᖏᓐᓄᑦ ᓈᒻᒪᒋᔭᐅᔪᒥᑦ ᓇᓗᓴᐃᔭᖅᓯᒪᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᓂᒧᑦ. ᑕᑯᔅᓴᐅᑎᑕᐅᓯᒪᔪᑦ ᒫᓇᐅᔪᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᓂ *post hoc* ᐊᐅᖏᑎᒎᖅᑐᓂ ᖃᐅᔨᓴᐃᓂᖅ ᖃᐅᔨᓴᕐᓴᓂᑦ ᖃᓄᐃᓐᓂᓇᔾᔨᑦ ᐊᑐᐃᓐᓇᐅᔪᓐᓇᖅᑐᑦ ᑕᓪᓕᒪᓂ ᐊᕐᕌᒍᓂ ᐱᒋᐊᖅᑎᑕᐅᓯᒪᓕᖅᑎᓪᓗᒍ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ. ᐃᓚᒋᐊᕐᓗᒍ, ᐊᐃᖏᑎᒎᖅᑐᓂ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᖅ (ᐊᔾᔨᐸᓗᖓ ᐆᒪᔪᓂ ᓇᓗᓇᐃᒃᑯᓯᑦᓯᒥᖅ) ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᔪᒥᑦ ᓇᒧᖓᖃᑦᑕᖅᑐᑎᑦ ᐊᕐᕌᒍᑕᒫᑦ ᐊᓯᔾᔨᖃᑦᑕᖅᑐᓂᑦ ᐊᕙᑎᖓᓂ (ᓲᕐᓗ, ᐊᓯᔾᔨᖃᑦᑕᖅᑐᑦ ᓯᑯᖓ) ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᑎᑐᑦ ᐊᔭᐅᖅᑕᐅᔭᓐᓇᖅᑐᖅ ᐊᓯᔾᔨᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐱᔭᓐᓇᕐᓗᑎ ᑭᒡᒐᖅᑐᐃᓗᒥ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ (ᓲᕐᓗ, ᐅᑎᖅᑕᕐᕕᐅᑲᐃᓐᓇᖅᑐᖅ, ᐃᓚᖏᓐᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᑐᐃᓐᓇᐅᖏᑦᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᓗᑎ ᐃᒪᐃᓐᓇᓲᑦᑐᑦ ᐅᖓᓯᑦᑐᒥ ᓯᑯᒥ).

ᖃᐅᔨᓴᖅᑕᐅᑲᓐᓂᕐᓗᑎ ᓇᓄᐃᑦ ᐊᒥᓲᓂᖏᑦ ᐊᖏᔪᒥᑦ ᐃᓱᒪᒋᔭᐅᔪᓐᓇᕐᒥᔪᖅ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᖃᓯᐅᔾᔨᓗᓂ ᖁᓪᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᓂᖏᓐᓄᑦ, ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᐊᑑᑎᖃᖅᑐᑎᑦ ᐊᓯᖏᓐᓄᑦ (Aars et al. 2009). ᐅᐱᕐᖔᒃᑯᑦ ᓯᓚᑦᑎᐊᕙᐅᖃᑦᑕᕈᓂᖃᖓᑕᕉᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᖃᕈᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᖅᑕᐅ)ᔪᓐᓇᖅᑐᖅ ᐊᕐᕌᒍᒥ ᐊᑕᐅᓯᕐᒥ. ᑭᓯᐊᓂᓕ, ᑕᑯᓐᓇᖅᑐᒍ ᓇᓗᓇᐃᒃᑯᓯᖅᑕᐅᔪᓄᑦ ᖃᖓᑕᓲᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ ᓯᓚᑲᒻᒪᐅᖃᑕᑐᐃᓐᓇᖃᐊᓕᑦ ᐊᑕᐅᓯᕐᒥ ᐊᕐᕌᒍᒥ.

SWG ᐱᓕᕆᐊᖅᑖᖅᑎᑕᐅᓚᐅᖅᑐᑦ ᐱᔪᒪᓂᖃᖅᑐᑦ ᑲᑎᑦᑐᒋᑦ ᓇᓐᓄᑦᑕᐅᔪᓐᓇᖅᑐᓂᑦ (TAH) ᖁᕝᕙᓯᓐᓂᖏᑦ ᑕᒪᒃᑭᓐᓄ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓄ ᐊᒻᒪ Kane Basin ᑭᓯᐊᓂ ᐱᑕᖅᑎᑕᐅᓚᐅᖏᑦᑐᑦ ᖃᓄᐃᑦᑐᑐᐃᓐᓇᕐᒧᑦ ᓇᓗHᓇᐃᖅᓯᒪᔪᒧᑦ ᑐᑭᒧᐊᕈᑎᒥᑦ ᐊᐅᓚᑕᐅᓂᖏᓐᓄᑦ ᑐᕌᖅᑕᐅᔪᓄᑦ ᑖᒃᑯᓄᖓ ᐊᒥᓲᓂᖏᓐᓄᑦ JC−ᑯᓐᓄᑦ, ᐊᒥᓱᐊᖅᑎᒐᓚᑦᑐᑎᑦ ᑐᔾᔨᖃᑦᑕᖅᓯᒪᒐᓗᐊᖅᑐᑎ ᐊᕐᕌᒍᒐᓚᐅᔪᓂᑦ. SWG ᑐᔾᔨᖅᓚᐅᖅᑐᑦ ᑖᒃᑯᐊ JC ᐱᑕᖅᑎᑦᑎᓗᑎ:

1) ᐅᓂᒃᑳᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᓕᕆᐊᖑᓂᐊᖅᑐᑦ ᐊᑐᓂ ᐊᒥᓲᓂᖏᓐᓄᑦ,

2) ᑐᑭᓯᒋᐊᕈᑏᑦ ᓂᕿᐅᒋᔭᐅᔪᑦ ᓴᖅᑭᖅᑕᑦᑕᓂᐊᕐᓂᖓ ᐊᒻᒪ ᐊᖏᓂᖓ ᓯᕗᓂᔅᓴᒥ ᖃᐅᔨᓴᓂᕐᒧᑦ, ᐊᒻᒪ

3) ᐅᓂᒃᑳᑦ ᐊᑦᑕᓇᕈᑎᖃᕈᓐᓇᕐᓂᖓᓄ ᐱᔾᔪᑎᖃᖅᑐᓂᑦ ᐊᑦᑐᖅᑕᐅᔪᓄᑦ ᐃᓄᓐᓄᑦ ᐲᖅᑕᐅᔪᓄᑦ.

ᐱᔭᐅᖏᓐᓂᖏᓐᓄᑦ ᐱᔭᐅᔭᕆᐊᓖᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᑐᓐᖓᕙᒋᓂᐊᖅᑕᖏᑦ ᐊᑐᓕᖅᑯᔭᐅᔪᑦ TAH JC−ᑯᓐᓂᑦ, SWG−ᑯᑦ ᒫᓇᐅᔪᒥᑦ ᑖᒃᑯᓂᖓ ᐊᑐᓕᖅᑯᔭᓕᐅᕈᓐᓇᖅᑕᖏᑦ. ᓇᓗᓇᐃᖅᓯᒪᔪᑦ ᐱᓕᕆᐊᖑᕆᐊᓖᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᒻᒪ ᐱᐅᖅᓯᐊᓂᕐᒧᑦ BB−ᒧᑦ ᐊᒻᒪ KB−ᒧᑦ ᓇᓄᖏᓐᓄᑦ ᐱᒻᒪᕆᐅᔪᖅ ᑕᒃᑯᐊ ᐊᔭᐅᕈᑎᒃᓗᐊᓂᐊᖅᑐᑦ ᐊᑐᓕᖅᑯᔭᐅᔪᓂᑦ TAH, ᐃᓱᒪᓕᐅᖃᑦᑕᖅᑐᓂᑦ ᐊᑐᐃᓐᓇᖃᓂᐊᖅᑐᑦ ᑐᑭᓯᒋᐊᕈᑎᐅᓂᐊᖅᑐᓂᑦ.

ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑐᕌᒐᐃᑦ ᐃᓱᒪᓕᐅᕈᑎᐅᖏᑦᑐᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᐃᖅᑲᓴᐃᔭᖅᑎᓄᑦ ᑭᓯᐅᓂᓕ ᑕᐃᒃᑯᓇᖓ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐱᔪᓐᓇᐅᑎᓕᓐᓄᑦ ᓇᓄᕐᓄᑦ ᐊᒻᒪ, ᐃᓱᒪᒋᔭᔅᓴᐅᔪᖅ, ᐃᓱᒪᓕᐅᕈᑕᐅᔪᑦᒃ ᐃᓱᒪᒋ.ᖃᒪᐅᖅᑐᑎᑦ ᓂᕿᔅᓴᓄᓲᖏᑦᑐᖅ ᓇᓄᑦᑕᐅᔪᑦ ᑭᓯᐊᓂ ᐊᓯᖏᑦ ᐃᓄᓐᓄᑦ ᐲᖅᑕᐅᔪᑦ ᓲᕐᓗ, ᐃᓄᐃᑦ−ᓇᓄᐃᑦ ᖃᓄᐃᓘᖃᑦᑕᐅᑎᓂᖏᑦ) ᐊᒻᒪ ᐅᖃᖃᑎᖃᒪᐅᖅᑎᓪᓗᒋᑦ ᑎᒍᒥᐊᖅᑎᐅᖃᑕᐅᔪᓂᑦ.

JC ᑐᓂᓯᑐᐊᕈᓂ ᑐᔅᓯᕌᖑᔪᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ, SWG ᐊᑐᕆᖅᑯᔨᔪᑦ ᐅᖃᐅᔾᔨᕆᐊᕈᑎᐅᕋᑖᓚᐅᖅᑐᖅ TAH−ᒥ ᑐᓐᖓᓂᖃᕐᓗᓂ ᐊᒥᓲᓂᕆᓐᖑᐊᖅᑎᑕᐅᔪᓄᑦ ᐊᑐᖅᑐᓗᑎ ᐱᖃᓯᐅᔾᔨᔪᓐᓇᖅᑐᓂᑦ ᐊᓯᔾ"ᒍᓐᓇᓂᕐᒧᑦ ᐊᓯᓅᖅᓯᔪᓐᓇᓂᕐᒧᑦ (ᓲᕐᓗ, ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ) ᐊᒻᒪ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᐱᐅᓂᖏᑦ ᐃᒪᒋᓪᓗᓂᐅᒃ ᐊᑕᖏᖅᑐᒥ ᐱᕙᓪᓕᐊᓂᖓᓂᑦ ᑲᐅᔨᓴᐃᓪᓗᑎ ᐊᑦᑕᓇᖅᑐᒦᒍᓐᓇᕐᓂᖓᓂ ᐊᔾᔨᕖᖏᑦᑐᓂᑦ ᓇᓄᑦᑕᐅᔪᓂᑦ ᐊᐅᓚᑕᐅᔪᓂᑦ ᖃᓄᐃᓐᓂᐅᔪᓐᓚᖅᑐᓂᑦ. SWG ᐊᑐᕆᖅᑯᔨᔪᑦ ᐊᑐᖅᑕᐅᓗᓂ ᓇᓗᓇᐃᖅᓯᒪᔪᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐊᑐᐊᒐᖅ, ᓴᖅᑭᖅᑎᑕᐅᓯᒪᔪᖅ Regehr et al. (2015), ᑎᒥᒧᑦ−ᑐᓐᖓᓂᖃᖅᑐᖅ ᐊᐅᓚᑕᐅᓂᖓᓄᑦ ᐋᖅᑭᓯᒪᓂᖓ ᑲᓯᕐᓂᖅᑎᑦᑎᔪᖅ ᐊᑐᖅᑕᐅᔪᒧᑦ ᐊᒥᓲᓂᕆᓐᖑᐊᖅᑕᖏᓐᓄᑦ ᖃᐅᔨᓴᐃᓂᖅ ᐊᒻᒪ ᐊᑐᖅᑕᐅᔪᓐᓇᕐᒥᔪᖅ ᒥᔅᓴᐅᓴᑦᑕᐅᓗᑎ ᐊᒥᓲᓕᕆᔪᓐᓇᖅᑕᖏᑦ ᒪᑭᒪᑎᑦᑎᓂᕐᒧᑦ ᖁ−ᕙᓯᓐᓂᖏᑦ ᐃᓄᓐᓄᑦ ᐲᖅᑕᐅᔪᑦ, ᐊᑐᖅᑲᑕᐅᔪᓐᓇᕐᒥᔪᖅᑕᐅᖅ ᐋᑦᑭᔅᓯᓗᑎ ᐊᑐᕆᖅᑯᔭᐅᔪᓂ ᒪᑭᒪᑎᑕᔅᓴᐅᔪᓂᑦ ᓇᓄᑦᑕᐅᖃᑦᑕᖅᑐᓂᑦ ᖁᕚᓯᓐᓂᖏᓐᓄᑦ, Regehr et al. (2015) ᓇᓗᓇᐃᖅᓯᓚᐅᖅᑐᖅ ᐊᑦᑎᓂᖅᓴᐅᒐᔪᑦᑐᖅ ᑭᓪᓕᒋᔪᓐᓇᖅᑕᖓᓂ ᒪᑭᒪᑎᑦᑎᓂᕐᒧᑦ ᖁᕚᓯᓐᓂᖏᑦ ᑐᓐᖓᓂᖃᕐᓂᖏᓐᓄᑦ ᐊᐅᓚᑕᐅᓂᖏᓐᓄᑦ ᐱᑎᓇᐊᓄᑦ, ᓇᓗᓇᐃᖅᓯᒪᓚᕆᓐᓂᖓ ᐊᒻᒪ ᖃᖓᒃᑰᖃᑦᑕᕐᓂᖏᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ ᐊᒻᒪ ᐊᑦᑕᓇᕈᓇᖏᓐᓂᕐᒧᑦ.

ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᑦ

ᑕᒪᒃᑭ ᖃᐅᔨᓴᖅᑎᓄᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ (ᐅᕙᓂ ᐊᒻᒪᓗᑦᑕᐅᖅ ᓯᕗᓂᐊᒍᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ) ᐊᒻᒪ ᐃᓄᓐᓄᑦ ᖃᐅᔨᒪᔭᐅᔪᑦ (Dowsley   et al. 2005, Dowsley and Wenzel 2008 and Born et al. 2011) ᑎᑎᕋᖅᓯᓯᒪᔪᑦ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᓇᔪᖅᑕᐅᕙᑦᑐᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ (ᓲᕐᓗ, ᓯᑯᖃᕈᓐᓃᕐᓂᖓ) ᓇᓄᕐᓄᑦ BB−ᒥᑦ (KB−ᒥᐅᔪᔅᓴᐅᒻᒥᔪᖅᑕᐅᖅ). ᓇᔪᖅᑕᐅᔪᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᐱᒻᒪᕆᐅᓂᐊᖅᑐᖅ ᑲᔪᓯᑎᓪᓗᒍ ᖃᓄᐃᓐᓂᕆᐊᓱᒋᔭᐅᔪᑦ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᐅᑭᐅᑕᖅᑐᒥ. ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓇᔪᖅᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᒻᒪ Kane Basin−ᒥ 2009−ᒥᓂᑦ, ᑐᓐᖓᓂᖃᓗᐊᖅᑐᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᕐᓇᓪᓗᓂᑦ ᓇᓄᕐᓂᑦ, ᐱᐅᔪᓂᑦ ᓄᑖᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᔪᖅ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᐊᓯᔾᔨᖅᑐᓂ ᓇᔪᖅᑕᐅᔪᓂᑦ. ᑐᑭᒧᐊᖅᑐᖅ ᖃᓄᐃᓕᖓᓂᕆᔭᖓᓂ BB−ᒥ ᐊᒻᒪ KB−ᒥᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ

SWG ᓇᓗᓇᐃᖅᓯᔪᑦ ᓯᕗᓂᔅᓴᒥ ᐆᒪᔪᓂᑦ ᐱᔭᐅᔪᓂᑦ ᐊᒻᒪ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᖅ BB−ᒥ ᐊᒻᒪ KB−ᒥᑦ (ᖁᖓᓯᕈᖅᓯᓂᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᑦ ᓇᒦᓐᖏᓐᓄᑦ ᐊᕐᓇᓪᓗᐃᑦ ᐊᕐᕌᒍᒐᓚᓐᓄᑦ) ᐱᒻᒪᕆᐅᓂᐊᖅᑐᖅ ᐅᖓᕐᕙᕆᐊᖅᑕᐅᓗᑎᑦ ᐊᑯᓂᐅᓂᕆᔭᖏᑦ ᐊᒻᒪ ᖃᐅᔨᒃᑲᐃᔪᖅ ᐊᐅᓚᑦᑎᔨᓂᑦ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓ ᓇᔪᖅᑕᐅᔪᓂᑦ ᐊᕐᕌᒍᒐᓚᐅᓂᐊᖅᑐᓂᑦ. ᐱᔾᔪᑎᒋᓪᓗᒍ ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᖅᓯᒪᓂᖓᓄᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᑕᕝᕙᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᒥ, ᐊᒻᒪ ᓇᓗᓇᖏᑦᑐᑦ ᖃᓄᐃᓕᓂᕆᔭᖏᑦ ᓇᓄᐃᑦ ᑕᒪᒃᑭᓐᓂ, ᑲᔪᓯᑎᑕᐅᓂᖓ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᑐᑭᓯᑎᑦᑎᑎᐊᓂᖅᓴᐅᓂᐊᖅᑐᖅ ᐊᑦᑐᖅᑕᐅᔪᓐᓇᖅᑐᑦ ᓯᕗᓂᔅᓴᒥ ᐊᒻᒪ ᐆᒪᑎᓐᓇᒥᓐᓂᖏᓐᓂᖒᖅᑐᑦ ᐊᓯᔾᔨᖅᑐᑦ. ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᖅ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᖃᐅᔨᓴᕐᓂᖃᕆᐊᖃᖅᑐᑦ ᖁᓕᓂᑕᒦᑦ ᐊᕐᕌᒍᓂᑦ ᑐᓖᓂᓪᓘᓐᓃᑦ, ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓂᖏᑦ ᐊᔾᔨᐸᓗᒋᔭᖏᑦ ᓄᐊᑕᐅᔪᓂᑦ 1991-1997 ᐊᒻᒪ 2009-2013 ᖃᐅᔨᓴᕐᓂᐅᔪᓂᑦ (ᖃᓂᒋᔮᐸᓗᖓᓂ 40 ᐊᕐᓇᓪᓗᐃᑦ ᖃᐅᔨᔭᐅᔪᑦ ᐊᔾᔨᒍᒐᓚᐅᔪᓂᑦ). ᐊᑦᑎᓂᖅᓴᐃᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓂᖏᑦ ᖃᐅᔨᓴᐃᓂᕐᒥᑦ ᐊᔪᒃᓇᑎᑦᑎᑲᓐᓂᓂᐊᖅᑐᖅ ᐃᒻᒥᑰᖅᑐᑦ ᐊᔾᔨᒌᖏᓐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᐊᑐᐊᒐᔅᓴᖃᑦᑎᐊᕐᓂᖏᓐᓄᑦ RSFs (ᑕᑯᔭᐅᓯᒪᔪᑦ KB−ᒥᑦ).

ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᖅ ᖃᐅᔨᓴᕈᑎᒥᓃᑦ ᓯᕗᓪᓕᖅᐹᒥ ᑎᑎᕋᖅᑕᓚᐅᖅᑐᑦ ᑖᒃᑯᐊ 18% ᐊᕐᓇᓪᓗᓂᑦ ᖁᖓᓯᕈᖅᑕᐅᓯᒪᔪᓂᑦ ᐱᖓᓇᖓᓂ ᐊᑯᑭᑦᑐᑦ Melville Bay−ᒦᖏᓐᓇᓚᐅᖅᑐᑦ

ᐊᑕᐅᓯᕐᒥ ᐊᐅᔭᒥᑦ ᐊᒻᒪ ᐃᓚᖏᑎᒍᑦ ᐊᕐᕌᒍᒧᑦ. ᑕᒪᓐᓇ ᒪᓕᓐᓂᖃᖅᑐᖅ ᖃᐅᔨᔭᐅᔪᓂᑦ ᐊᖑᓇᓱᑦᑎᓪᓚᕆᓐᓄᑦ Melville Bay–ᒥ, ᖃᐅᔨᓯᒪᔪᑦ ᓇᓄᖅᑕᖃᓕᖅᐸᓪᓕᐊᓂᖓᓄᑦ Melville Bay, ᐊᒻᒪᓗᑦᑕᐅᖅ ᑎᑎᕋᖅᑕᐅᔪᑦ ᐅᑯᓄᖓ Taylor et al. (2005) ᐊᒻᒪ Born et al. (2011). ᐊᑐᓕᖁᔭᐅᒃᑲᓂᖅᑐᑦ ᓯᕗᓂᔅᓴᒥ ᐃᖅᑲᓇᐃᔫᖑᓂᐊᖅᑐᓂᑦ ᖃᐅᔨᓴᕐᓗᑎ ᖃᑦᑎᑦ ᐃᒪᖏᑦ BB–ᓇᓄᖏᑦ ᐊᑐᖃᑦᑕᖅᖃᑦ Melville Bay ᐊᕐᕌᒍᓕᒫᒥᑦ ᐊᒻᒪ ᒪᓕᓪᓗᒋᑦ ᑖᒃᑯᐊ ᑲᑎᙵᔪᑦ ᓇᓄᐃᑦ ᐊᕐᕌᒍᓕᒫᒥ Melville Bay–ᒦᑉᐸᑦᑐᑦ ᐆᒪᔪᖃᕐᕕᒻᒥᑦ. ᑖᓐᓇ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᖅ ᒫᓐᓇᐅᔪᒥ ᐊᐅᖏᑎᒎᖅᓯᒪᔪᓂᑦ ᓇᓗᓇᐃᒃᑯᑕᓂ ᑖᔅᓱᒧᖓ ᖃᐅᔨᓴᓂᕐᒥᑦ, ᐃᓚᒍᑕᖃᕐᓗᑎ ᐊᑕᐅᓯᕐ ᒪᕐᕉᓂᓪᓘᓐᓃᑦ ᐊᐅᖏᑎᒎᖅᑐᓂᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ Melville Bay–ᒥ ᐊᑕᐅᑎᒃᑯᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔅᓴᓂᕐᒥᑦ.

SWG ᐊᑐᓕᖁᔨᓚᐅᕐᒥᔪᑦ ᑲᔪᓯᑎᑕᐅᓗᓂ ᖃᐅᔨᓴᕐᓂᖅ ᐊᓯᔾᔨᖅᑐᓂ ᓇᔪᖅᑕᐅᔪᓂᑦ BB–ᒥᑦ ᐊᒻᒪ KB–ᒥᑦ ᖁᒻᒧᐊᑦᑎᑕᐅᓯᒪᔪᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓗᑎ ᓯᑯᖓ (ᐊᖅᑯᑎᖏᓐᓄ ᑐᑭᓯᒋᐊᕈᑏᑦ, MODIS, ᐅᕝᕙᓘᓐᓃᑦ Radar ᐊᔾᔨᙳᐊᖏᑦ, ᐱᔭᓪᓕᐊᑎᑕᐅᓂᖏᑦ ᓇᔪᖅᑕᐅᔪᓄᑦ ᐊᑐᖅᑕᐅᔪᑦ (ᒪᓕᑦᑐᒍ Durner et al. 2009, Laidre et al. 2015 ᐊᒻᒪ ᓵᑉᑕᖏᑦ 4 ᐊᒻᒪ 9), ᑲᔪᓯᑎᑕᐅᔪᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐊᑦᑐᐊᓂᖃᖅᑐᑦ ᓇᔪᑦᖅᑕᐅᔪᓄᑦ ᐊᓯᔾᔨᖅᑐᓄᑦ ᓇᓗᓇᐃᖅᓯᒪᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ, ᐊᒻᒪ ᑲᔪᓯᑎᑕᐅᖏᓐᓇᕐᓗᓂ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ TEK ᐊᒻᒪ LEK ᑲᒪᒃᑭᓐᓂ ᑲᓇᑕᒥ ᐊᒻᒪ ᑲᓇᑕᒥ ᑐᑭᓯᒋᐊᕈᑎᖃᖅᑎᑦᑎᓂᐊᖅᑐᑦ ᖃᓄᖅ ᐊᓯᔾᔨᒻᑐᓂᑦ ᓯᑯᓂᑦ ᐊᑦᑐᐃᓂᖃᕐᒪᖔᑕ ᓇᓄᕐᓂᑦ ᐊᖑᓇᓱᑦᑕᐅᕙᑦᑐᓂᑦ (ᓅᕐᓗ, ᓇᓄᓕᕆᔨᑐᐃᓐᓇᕐᓂᖅ), ᐊᑕᖏᖅᑐᒍ ᓇᓄᑦᑕᐅᖃᑦᑕᖅᑐᑦ, ᐊᒻᒪ ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᓇᓄᑦᑕᐅᔪᑦ.

ᐃᓱᒪᕆᔭᐅᔪᑦ

ᔮᓐᓄᐊᕆᒥ 2010, JC ᐱᓕᕆᐊᖃᖅᑎᑦᑎᓚᐅᖅᑐᑦ SWG–ᑯᓐᓂᑦ ᐊᑐᖅᑐᑎᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᖃᐅᔨᓴᖅᑎᓄᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐃᒪᐃᓐᓂᐊᖅᑐᑎ: (1) ᓴᖅᑭᑎᑦᑎᓗᑎ ᑲᑎᑦᑐᒋᑦ ᓇᓄᑦᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᖁᕝᕙᓯᓐᓂᖏᑦ Kane Basin–ᒧᑦ ᐊᒻᒪ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓄᑦ ᓇᓄᖏᓐᓄᑦ ᐊᒻᒪ JC ᑐᓂᓯᔭᐅᓗᑎ ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐅᓂᒃᑳᓂᑦ ᐊᑐᓕᖁᔭᖏᓐᓂᑦ, (2) ᖃᐅᔨᓴᖅᑎᓐᓂᖑᖅᑐᓂ ᐅᖃᐅᓯᕆᐊᕐᓂᖅ ᐊᐅᓚᑦᑎᖃᑎᒋᑦᑐᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖓᓄ ᐊᑦᑐᖅᑕᐅᓂᖏᑦ ᓇᔪᖅᑕᖏᑦᑕ ᐊᓯᔾᔨᕐᓂᖏᓐᓄᑦ ᓇᓄᐃᑦ.

ᕿᒥᕐᕈᓚᐅᖅᑐᒋᑦ ᑐᑭᓯᒋᐊᕈᑏᑦ, SWG ᓄᖅᑲᐅᓯᓚᐅᖅᑐᑦ (SWG 2011) ᒥᔅᐅᓴᑦᑕᐅᕋᑖᖅᓯᒪᔪᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓄᑦ (BB) ᐊᒻᒪ Kane Basin (KB) ᐊᒻᒪ ᐱᒪᕆᐅᔪᑦ ᑭᓪᓕᕆᔭᖏᑦ ᓴᖅᑭᖅᑐᑦ ᑲᑐᑎᔭᐅᔪᓂᑦ ᑲᓇᑕ–ᐊᑯᑭᑦᑐᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ (1993-1997) BB–ᒥ ᐊᒻᒪ KB–ᒥ ᐅᓪᓗᒥ ᒨᖓᓚᐅᖏᑦᑐᑦ ᐊᒻᒪ ᓇᓗᓇᕈᑎᑕᖅᑐᑎ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᓯᔾᔨᓪᓗᐊᑲᓪᓚᔅᓯᒪᓂᖓᓄᑦ ᐅᓂᒃᑳᓕᐊᖑᔪᓂᑦ ᓇᓄᑦᑕᐅᕙᑦᑐᓂᑦ ᖁᕝᕙᓯᓐᓂᖏᑦ ᐊᒻᒪ ᓯᑯᖓ ᖃᓄᐃᓕᖓᓂᖓ ᑖᒃᑯᐊ ᖃᐅᔨᓴᕐᓂᐅᔪᑦ ᐱᔭᕇᖅᑕᐅᓯᒪᓕᖅᑎᓪᓗᒋᑦ. ᐃᓚᒋᐊᕐᓗᒍᓕ, ᑐᑭᓯᒋᐊᕈᑏᑦ ᖃᓄᐃᓕᖓᓂᕆᔭᖏᓐᓂᑦ ᑖᒃᑯᓂᖓ ᐱᔭᐅᔪᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᓴᓂᐊᓂ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔭᑐᖃᖏᑦ (TEK) ᒪᓕᓐᓂᖃᖅᑑᔮᓚᐅᖏᑦᑐᖅ. ᑖᓐᓇ ᐋᖅᑭᓂᐊᕐᓗᒍ ᓇᓗᓇᕈᑎᐅᔪᖅ, SWG ᐱᒋᐊᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᓄᑖᒥᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ 2011–ᒥ ᐃᒪᐃᓐᓂᐊᖅᑐᑎ (1) ᖃᐅᔨᓴᕐᓗᒋᑦ ᐊᒥᓲᓂᖏᑦ ᓇᓄᐃᑦ BB-ᒥᑦ ᐊᒻᒪ KB–ᒥᑦ, ᐊᒻᒪ (2) ᑲᑎᖅᓱᐃᓗᑎ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᓇᓄᕐᓄᑦ ᓯᑯᖃᕈᓐᓃᕐᓂᖓᓄᑦ ᑖᒃᑯᓄᖓ ᐊᕙᑦᑐᖅᓯᒪᔪᓄᑦ.

ᐊᑕᖏᖅᑐᒥᑦ ᖃᓄᐃᓕᓂᐅᔪᑦ ᐊᓯᔾᔨᖅᑐᓄᑦ ᓇᔪᖅᑕᐅᔪᓄᑦ

ᑕᒪᒃᑭᓐᓂ ᐊᒥᓲᓂᖏᓐᓂ ᑕᑯᔅᓴᐅᑎᑦᑎᓯᒪᔪᑦ ᓇᓗᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ ᐊᓯᔾᔨᖅᑐᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᖃᓄᐃᓕᓂᕆᔭᖏᑦ ᐊᖏᔪᒧᑦ, ᐊᑯᓂᐅᔪᒧᑦ ᓯᑯᖃᕈᓐᓃᑲᓐᓂᕐᓂᕐᓂᖓᓄᑦ BB–ᒥᑦ ᐊᒻᒪ KB–ᒥᑦ ᑭᖑᓪᓕᖅᐹᕆᓯᒪᔭᖏᓐᓂ 25 ᐊᕐᕌᒍᓂᑦ. BB ᐊᒥᓲᓂᖏᑦ ᑭᓪᓕᖃᓕᖅᓯᒪᔪᖅ ᐅᖓᓯᓐᓂᖓ ᐊᒻᒪ ᓴᖑᓯᒪᓪᓗᑎ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᑦ ᐅᐊᓐᓇᒧᑦ ᐊᕐᕌᒍᓕᒫᒥ ᐃᓗᐊᓂ ᐊᔾᔨᑭᑎᙳᐊᖅᑐᒍ ᖃᓄᐃᓕᖓᓂᕆᔭᖓᓂ 1990–ᖏᓐᓂᑦ. ᑖᒃᑯᐊ ᐊᒥᓲᓂᖏᑦ ᐃᓵᒃᑰᓕᒃᑲᓂᖅᓯᒪᔪᑦ, ᖃᓄᐃᓘᑲᑦᑕᐅᑎᖏᓐᓂᖅᓴᐅᓗᑎ ᓴᓂᐊᓃᑦᑐᓂᑦ (KB, ᑕᓪᓗᕈᑎᐅᑉ ᐃᒪᖓ ᐊᒻᒪ

ᕿᑭᖅᑖᓘᑉ ᐃᑭᕋᓴᖓ). ᐅᐱᕐᖔᒃᑯᑦ ᓯᑯᐃᖅᓴᕋᐃᖃᑦᑕᕐᓂᖓᓄᑦ, ᐊᐅᑦᑐᐃᓐᓇᐅᖃᑦᑕᑎᓪᓗᒋᑦ ᐊᕐᕌᒍᑕᒫᑦ ᓯᑰᔪᑦ ᐊᒻᒪ ᐅᑭᐊᔅᓵᒃᑯᑦ ᓯᑯᓇᓵᕐᓂᖓᓄᑦ, BB ᓇᓄᕐᒥᑦ ᐊᑯᓂᐅᓂᖅᓴᐅᕙᓪᓕᐊᓕᖅᑐᑦ ᓄᓇᒦᓐᓂᕆᕙᑦᑕᕐᒥᑦ ᐃᒫᓅᕈᓐᓇᕐᒥᑎᑦᑎᔪᑦ ᐃᒪᕐᒥᐅᑕᓄᑦ ᐆᒪᔪᓄᑦ ᓂᕆᒃᑦᑕᖅᑕᕐᒥᓄᑦ.

1990-ᖏᓐᓂᑦ, ᓯᑯᖓᓄᑦ ᐊᑐᖅᑕᐅᒐᔪᑦᑐᖅ KB−ᒧᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᑲᑎᖅᑐᖅᓯᒪᔪᓂᑦ ᓯᑯᓂᑦ ᐊᕐᕌᒍᑕᒫᒃ ᐊᔾᒪ ᐊᕐᕌᒍᒐᓚᓐᓄᑦ ᐊᐅᔭᐅᓚᐅᖅᑐᒥᑦ. ᑕᒪᓐᓇ KB ᓇᓄᖏᓐᓂᑦ ᐊᕐᕌᒍᓕᒫᒥᑦ ᓯᑯᒨᕈᓐᓇᑎᑦᑎᔪᖅ ᓇᔪᖅᑲᑦᑕᖅᑕᖅᓚᓄ ᓯᑯᐃᖃᑦᑕᕐᓂᖓᓅᖏᖔᖅᑐᖅ ᓃᕐᓗ BB−ᑎᑐᑦ. 1990−ᖏᓐᓂᐅᕙᔪᓪᒪᓂᑦ, ᑭᓯᐊᓂᓕ, ᖃᓄᐃᓐᓂᕆᔭᖏᑦ KB−ᒥᑦ ᑐᕌᖅᐸᓪᓕᐊᓯᒪᔪᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓄᑦ ᓯᑯᖏᑦ BB−ᒥ ᐊᐅᑦᑐᐃᓐᓇᐅᖃᑦᑕᑎᓪᓗᒋᑦ ᐊᐅᔭᑕᒫᑦ (i.e., “ᓯᑯᐃᖃᑦᑕᕐᓂᖓ ᓯᑯᖃᑦᑕᕐᓂᖓᓄᓗ” ᖃᓄᐃᓐᓂᕆᔭᖏᑦ). ᖃᐅᔨᓴᖅᑎᒥᓂᖅᑎᓐᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ KB ᐊᒥᓱᓂᖏᑦ ᖃᓄᐃᓕᓂᖃᖅᓯᒪᔪᑦ ᑖᔅᓱᒧᖓ ᐊᓯᔾᔨᕐᓂᐅᔪᒧᑦ ᓯᑯᖃᑦᑕᕐᓂᖓᓄ ᓯᑯᐃᖃᑦᑕᕐᓂᖓᓄᑦ ᐊᖏᓕᓕᐊᖅᑐᑎᑦ ᐅᓪᓗᓯᐅᓂᖏᑦ 1990−ᖏᓐᓂᐅᕙᔪᓪᒪᓂᑦ ᐱᓗᐊᖅᑐᒥ ᐊᐅᔭᒃᑯᑦ. ᐊᖏᓂᖅᓴᐃᑦ ᐊᒻᒪ ᐊᔾᔨᒌᖏᑐᓐᓇᓂᖅᓴᐅᔪᑦ ᐅᓪᓗᓯᓂᖏᑦ ᓇᔪᖅᑕᖏᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖓ ᓯᑯᖅᑎᐊᖏᓐᓂᖓ ᐊᐅᔭᒃᑯᓗ ᐅᑭᐊᔅᓵᒃᑯᓗ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ. ᑖᒃᑯᐊ ᐊᓯᔾᔨᕐᓂᐅᔪᑦ ᓇᒻᒪᖅᑕᕐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᐃᖏᕈᓐᓇᐅᔪᑦ ᖃᓄᐃᓲᓐᓂᖏᓐᓄᑦ ᐊᑐᖅᑕᐅᕙᑦᑐᑦ ᓇᓄᓐᓄᑦ ᓇᔪᖅᓯᒪᔪᓂ “ᐊᕙᑎᖓ ᐊᕕᑦᑐᖅᓯᒪᔪᒥ” ᐊᕐᕌᒍᑕᒫᑦ ᓯᑯᖓᓄᑦ.

ᐊᓯᔾᔨᖅᑐᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐅᑎᖅᑕᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᖃᐅᔨᓴᕐᓂᖅ ᖃᐅᔨᓴᖅᑎᒥᓂᕐᓂᑦ

BB−ᒥᑦ ᖃᐅᔨᓴᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᔭᔅᓴᐅᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐊᕐᕌᒍ ᐃᓗᐊᓂ ᐅᑎᖅᑕᕐᓂᖏᑦ ᖃᐅᔨᓴᓂᕐᒥᑦ ᖃᐅᔨᓴᖅᑎᒥᓂᕐᓂᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ BB ᓇᓄᕐᒥᑦ ᖃᓄᐃᓕᓂᖅᑕᖅ ᐊᒥᓯᓕᓐᓂᑦ ᒥᑭᓕᕆᐊᖅᓯᓂᖅᓯᓐᓂᑦ ᓈᒻᒪᑦᑐᑦ ᓇᔪᖅᑕᐅᔪᑦ 1990−ᖏᓐᓂ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ. ᑕᒪᓐᓇ ᓂᕆᐅᒋᔭᐅᔪᖅ ᑲᔪᓯᑐᓂ ᑲᔪᓯᓐᓇᕐᓂᖓᓄᑦ ᓯᑯᖃᕈᓐᓇᓂᕐᓂᖓᓄᑦ.

ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᖃᓄᐃᓲᖅᑕᖅᓯᓂᖏᑦ, ᖃᐅᔨᓴᓚᐅᖅᑐᒍᑦ ᐃᔅᓴᐃᑦ ᐊᔅᓇᐃᑦ ᐊᑐᖅᑕᖅᑕᓕᖅᑐᑦ ᓯᑯᑭᓐᓂᖅᓴᐅᔪᒥ ᓯᑯᒥ ᓇᔪᖅᑲᑦᑕᖅᑕᖓ ᓄᓇᒧᑦ ᖃᓂᒋᔭᐅᓗᑎ. ᓇᓄᐃᑦ ᓄᓇᒨᖅᑕᖅᑐᑦ ᕿᑭᖅᑖᓗᒻᒥᑦ ᓯᕗᕙᕐᓂᕐᓴᐅᓗᑎᑦ ᑕᖅᑭᓗᐊᕐᒥᑦ ᐊᒻᒪ ᐊᑯᓂᐅᓂᖅᓴ ᓇᔪᖅᓯᕙᓕᖅᑐᑦ (20-30 ᐅᓪᓗᓂᑦ) ᓄᓇᒥᑦ ᐅᑎᓚᐅᖅᑎᓇᒥᑦ ᓯᑯᕋᑖᒧᑦ ᓯᑯᓇᔅᓴᖅᓯᒪᔪᒥᑦ. ᐊᒥᓲᓂᕐᒥᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐅᓪᓗᓯᐅᑦᑐᒍᑦ >ᐃᒥᓐᓂᐅᔪᓂᑦ ᐊᖅᓇᓂᑦ ᐊᒥᓱᕆᐊᖅᓯᒪᔪᑦ ᓇᓄᐃᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ ᐱᓯᒪᐃᓐᓇᕆᐊᖅᑐᑦ ᐊᑐᖅᑐᒥᑦ ᓯᑯᑭᓐᓂᖅᓴᐅᔪᒥᑦ. ᓇᐃᔪᑦ ᓇᓄᐃᑦ ᑎᓯᖏᓐᓂᖅᓯᓯᒪᔪᑦ ᐊᓂᕈᓐᓇᖅᑐᑦ ᓯᑯᒥᑦ ᐅᓪᓗᖏᑦ ᐊᓯᔾᔨᖅᓯᒪᓕᖅᐳᑦ. ᖃᓄᐃᓐᓂᖏᑦ ᕿᓚᒥᐅᓂᖅᓴᐅᔪᒥ ᑎᓯᖓᓂᖏᓐᓂᑦ ᖃᐅᔨᒪᔭᐅᔪᑦ. ᑭᓯᐊᓂᓕ, ᖃᐅᔨᓴᐅᔪᑦ ᑎᓯᖏᑦ ᖁᕝᕙᓯᓐᓂᖅᓯᒪᓕᖅᑕᐅᓕᖅᑐᑦ 2000−ᖏᑦ ᓴᓂᐊᓂ 1990−ᖏᓐᓂᑦ ᐊᐳᑎᖃᑦᑕᕐᓂᖓ ᐃᓐᓈᕆᖏᑦ ᑲᓇᓐᓇᖓᓂ ᕿᑭᖅᑖᓗᒃ ᐱᔪᓐᓇᐅᔪᓐᓇᖅᑐᑦ.

ᖃᐅᔨᓴᐅᕕᔪᑦ ᐊᑦᑎᓕᕆᐊᖅᑐᓂ ᖃᐅᔨᓴᖅᑎᒥᓂᕐᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ ᐊᒻᒪ ᑎᒥᖏᓐᓂᑦ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂᑦ BB ᓇᓄᖏᑦ. ᑎᑎᕋᖅᑕᐅᖃᑎᒋᔭᐅᔪᓂᖏᑦ ᐊᑎᖅᑕᐃᒃ ᐊᕐᕌᒍᒧᑦ ᖃᑎᖓᓇᕐᓂᐊᖅᓯᒪᔪᑦ ᒪᓕᖓᓂᖅᑐᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᐅᐱᕐᙵᒃᑯᑦ ᓯᑯᐃᖅᓴᕋᐃᖅᑕᕐᓂᖓᓄᑦ 1990−ᖏᓐᓂᓂᑦ. ᑭᓕᖅᑐᓂᑦ ᒥᔅᓴᐅᓴᒧᓇᕋᓗᐊᖅᑎᓗᑕ ᐆᒪᓂᖏᓐᓂᑦ, ᐊᑎᓚᐅᖅᑐᑦ ᐃᓪᒥᑰᖅᑐᑦ ᐊᖑᑎᑦ 2000ᖑᖏᓐᓂᑦ ᐊᔾᔨᒋᐊᖑᐊᖅᑐᒍ ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᓄᑦ ᑐᑭᓴᐅᑎᑦᑎᔾᔪᓇᖅᑐ ᐊᑦᑐᖅᑕᐅᔪᓂᑦ ᐊᓯᔾᔨᕐᓂᖏᓐᓄᑦ ᐊᕙᑎᖓ ᖃᓄᐃᓐᓂᕆᔭᖓ. ᑭᖑᓪᓕᖅᐹᒥ, ᐊᑦᑎᓕᕆᐊᖅᑐᑦ ᑎᒥᖏᓐᓄᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᓇᓄᕐᓂᑦ BBᒥᑦ ᐊᑯᓐᓂᖏᓐᓂ 1993 ᐊᒻᒪ 2013 ᑕᐃᒪᐃᓯᒪᔪᑦ ᐊᑦᑐᐊᓂᖅᑐᓂ ᐊᑯᓂᐅᓂᖓᓄ ᓯᑯᖅᑎᓇᒍ ᐊᒻᒪ ᐅᐱᕐᙵᒃᑯᑦ ᓯᑯᐃᖅᑕᕐᓂ ᐅᓪᓗᖏᑦ.

ᑕᑯᓐᓇᖅᑐᒍ BB−ᒧᑦ, ᖃᐅᔨᓚᐅᖏᑦᑐᒍᑦ ᓇᓄᓇᐃᖅᓯᔪᓂᑦ ᐊᓯᔾᔨᖅᑐᓂᑦ ᕿᑐᓐᖏᖁᑲᑦᑕᕐᓂᖏᓐᓄᑦ KB−ᒧᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ (ᑭᓯᐊᓂ ᓯᐊᒻᒪᖅᓯᔪᓂᑦ ᑐᑭᓯᒋᐊᕈᑎᑦ ᑭᓪᓕᖅᑎᑦᑎᓚᐅᖅᑐᑦ ᐅᕙᙳᑦ ᓄᖅᑲᐅᓯᖓᓄᑦ), ᖃᐅᔨᓚᐅᖅᑐᒍᑦ ᖃᐅᔨᓴᖅᓴᐅᔪᓂᑦ ᐱᐅᓯᒋᐊᖅᑐᓂᑦ ᑎᒥᖏᓐᓂ ᖃᓄᐃᓐᓂᕆᔭᖏᓐᓂ 2000−ᖏᓐᓂᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᓐᓇᖅᑐᖅ ᓴᖅᑭᖃᑦᑕᖅᑐᓂᑦ ᑭᓯᐊᓂ ᐃᓚᒋᔪᓐᓇᖅᑕᖓ ᖃᓄᐃᓕᓂᖓ ᐊᑯᓂᐅᔪᒧᑦ ᐊᓯᔾᔨᖅᑐᓄᑦ ᓯᑯᖓᓄᑦ Kane Basin, ᓴᓂᐊᓅᑦᑐᑦ ᐆᒪᔪᓕᐅᖃᑦᑕᕐᓂᖏᑦ ᐊᕙᑎᖓᓂ ᖁᕝ>ᕆᐊᖅᓯᒪᔪᖅ.

ᐱᑕᖃᕐᓂᖏᑦ

ᐊᐅᖏᑎᒨᖅᑐᓂ ᓇᓄᓇᐃᒃᑯᓯᖅᓯᓂᖅᕝᐱᔭᐅᒃᑲᓂᖅᑐᓂᑦ ᖃᐅᔨᓴᐅᓂᖅ BB− ᐊᒥᓲᓂᖏᓐᓄᑦ ᓴᖅᑭᑎᑦᑎᓯᒪᔪᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᒐᔪᑦᑐᓂᑦ (2012-2013) ᑲᑎᑦᑐᒥᑦ 2,826 ᓇᓄᐃᑦ (95% CI = 2,059-3,593). ᐱᔾᔪᑎᖃᖅᑐᑎ ᐊᒥᓱᒐᓚᓐᓂᑦ ᑭᓪᓕᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐊᑐᐃᓐᓇᐅᔪᓂᑦ ᖃᐅᔨᓴᕈᑎᒥᓂᕐᓂᑦ. ᐅᖃᐅᓯᐅᔪᑦ ᓇᓄᓇᐃᔭᖅᓯᒪᓪᓗᑎ ᐅᕙᓂ ᐅᓂᒃᑳᓂᑦ, ᒥᔅᓴᐅᓴᑦᑕᐅᓯᒪᔪᑦ 1990−ᖏᓐᓂᑦ ᐊᒻᒪ 2000−ᖏᓐᓂᑦ ᑕᕝᕗᖓᑦᑕᐅᑎᒋ ᐊᔾᔨᒌᙳᐊᖅᑎᒐᔅᓴᐅᖏᑦᑐᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᓇᓄᓇᐃᖅᓯᓂᖃᕈᓐᓇᖏᑦᑐᖅ ᐊᓯᔾᔨᖃᑦᑕᖅᑐᓂᑦ ᐊᒥᓲᓂᖏᓐᓂᑦ BB ᓇᓄᖏᓐᓄᑦ. ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᑕᑯᔅᓴᐅᑎᑦᑎᓯᒪᔪᖅ ᐱᑕᖃᕐᓂᖏᓐᓄᑦ ᐊᑖᒍ ᐊᒥᓲᓂᖏᑦ ᓴᓐᓂᕈᑎᐅᑉ ᐃᒪᖓᓂ ᐊᖏᓂᖅᓴᐅᔪᖅ ᓴᓂᐊᓂ ᓇᓄᓇᐃᖅᑕᐅᔪᒥ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ (1,546 ᓇᓄᐃᑦ cf. PBSG 2015); ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐃᓚᒋᓕᓚᐅᖅᑕᖏᑦ PBSG’s (2015) ᓇᓄᓇᐃᖅᑕᐅᓂᖓ BB ᐊᑐᑦᑎᓪᓕᕙᓪᓕᐊᓂᖏᑦ ᐊᒥᓲᓂᖏᑦ. TEK ᑐᑭᓯᒋᐊᕈᑏᑦ ᓇᓄᓇᐃᖅᓯᓯᒪᖏᑦᑐᑦ ᓇᓄᓇᐃᒃᑯᓯᖅᑕᐅᓯᒪᔪᓂᑦ ᐊᑦᑎᓪᓕᒋᐊᖅᑐᓂᑦ BB ᓇᓄᖏᓐᓄᑦ (ᓲᕐᓗ, Dowsley and Taylor 2006, Born et al. 2011).

ᓇᓄᓇᕈᑎᑕᖃᕋᓗᐊᖅᑎᓪᓗᒍ ᖃᐅᔨᔭᐅᔪᓂᑦ BB−ᒥᑦ, ᐊᒥᓱᒐᒪᐃᑦ ᖃᐅᔨᔭᐅᔪᑦ ᐊᑦᑐᐊᓂᓖᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᐃᖏᕋᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᓇᔪᖅᑕᐅᔪᒥ ᐊᑐᐃᓐᓇᐅᓂᖓᓄᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᓂᖓᓄᑦ, ᖃᓄᐃᓘᕐᓂᖓ, ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ, ᐊᒻᒪ ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᑦ ᑕᑯᔅᓴᐅᑎᑦᑎᔪᑦ ᑖᒃᑯᐊ BB ᓇᓄᖏᑦ ᒫᓐᓇᐅᔪᒥᑦ ᐊᖏᔪᒥᑦ ᐊᓯᔾᔨᕐᓂᑰᖅᑐᖅ ᓄᖑᑉᐸᓪᓕᐊᓂᖏᓐᓄᑦ ᓇᔪᖃᑦᑕᖅᑕᖏᑦ. ᒫᓐᓇᐅᔪᒥᑦ ᑕᐃᒪᐃᓕᖓᑐᐃᓐᓇᖅᑲᑕ ᓯᑯᖓ ᓈᒻᒪᑦᑐᒥ ᒥᔅᓴᐅᓴᓯᔅᓴᐅᔪᑦ ᐊᓯᔾᔨᑲᓂᓂᐊᖅᑐᑦ ᑕᕝᕙᓂ ᐊᒥᓲᓂᖏᓐᓂᑦ ᐱᖃᓯᐅᑎᓪᓗᑎ, ᑕᐃᒪᓂᖃᕆᐊᖃᖅᑐᓂ, ᐊᑦᑎᓪᓕᒋᐊᖅᑐᑦ ᐊᒥᓲᓂᖏᑦ ᐊᒻᒪ ᐱᒻᒪᕆᐅᔪᑦ ᖁᕝᕙᓯᓐᓂᕆᐊᖃᖅᑕᖅᑕᖏᑦ, ᑖᓐᓇ ᖃᐅᔨᒪᒋᐊᖃᖅᑎᑦᑎᔪᖅ ᑕᒪᒃᑭᓐᓂ ᓯᕗᓂᔅᓴᒥ ᖃᐅᔨᓴᓂᕐᒥᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ.

ᐊᑐᖅᑐᒥᑦ ᓇᓄᓇᐃᒃᑯᓯᖅᓯᒪᔪᑦ−ᐱᔭᐅᒃᑲᓂᖅᑐᑦ KB−ᒥᑦ, ᓄᑖᖑᔪᖅ (2013-2014) ᒥᔅᓴᐅᓴᑦᑕᐅᔪᑦ ᑖᒃᑯᓄᖓ ᐊᒥᓱᓂᖏᓐᓄᑦ 357 ᓇᓄᐃᑦ (95% CI: 221 – 493). ᐋᖅᑭᑕᐅᒃᑲᓂᖅᓯᒪᔪᖅ 1990 ᐱᔭ.ᐅᒃᑲᓂᖅᑐᒥᑦ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ KB−ᒥᑦ ᒥᔅᓴᐅᓴᑎᑦᑎᓯᒪᔪᑦ 224 ᓇᓄᐃᑦ (95% CI: 145 – 303) ᐊᑯᓐᓂᖏᓐᓄ 1995 – 1997. ᐊᔾᔨᒌᙳᐊᖅᑎᑕᐅᔪᑦ ᑖᒃᑯᓂᖓ ᒪᕿᓐᓂ ᒥᔅᓴᐅᓴᑦᑕᐅᔫᓐᓂᒃ ᓇᓄᓇᐃᖅᓯᔪᑦ ᐱᑕᖃᕐᓂᖓ ᒪᑭᑕᑦᑎᐊᖅᑐᒥ ᐊᒥᓱᕈᖅᐸᓪᓕᐊᔪᓂᓪᓘᓐᓃᑦ, ᒪᓕᓐᓂᖃᖅᑐᖅ ᑐᑭᓯᒋᐊᕈᑎᓂᑦ ᓇᒧᖓᖃᑦᑕᕐᓂᖏᓐᓄᑦ, ᑎᒥᖏᑦ ᖃᓄᐃᓐᓂᖏᓐᓄᑦ ᐊᒻᒪ ᕿᑐᓐᖏᐅᖃᑦᑕᕐᓂᖏᑦ ᐊᒻᒪ ᒪᓕᓐᓂᖃᖅᑐᑎ TEK (Born et al. 2011). ᑖᒃᑯᐊ ᐊᓯᔾᔨᕐᓂᐅᔪᑦ ᓇᓄᓇᐃᖅᓯᔪᑦ ᐊᒥᓲᓂᖏᑦ ᑖᒃᑯᐊ ᓈᒻᒪᑎᐊᖅᑐᑦ ᐊᒻᒪ ᑕᐃᒪᐃᓕᖓᐃᓐᓇᕈᔪᑦᑐᑎ. ᑖᒃᑯᐊ ᖃᐅᔨᓴᓂᐅᔪᑦ ᐊᔾᔨᒌᖏᑦᑕᖏᑦ ᐊᑦᑎᓂᖅᓴᓄᑦ ᒥᔅᓴᐅᓴᑦᑕᐅᔪᓄᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᐱᔭᐅᔪᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᖅᓯᒪᔪᓂᑦ (Taylor et al. 2008); ᖃᓄᐃᓐᓂᕆᔭᖏᑦ ᐃᓚᒋᓕᓚᐅᖅᑕᖏᑦ PBSG’s (2010) ᓇᓄᓇᐃᖅᑕᐅᓂᖓ KB ᐊᑐᑦᑎᓪᓕᕙᓪᓕᐊᓂᖏᑦ ᐊᒥᓲᓂᖏᑦ.

ᑖᒃᑯᐊ ᐊᓯᔾᔨᖅᑐᑦ KB−ᒥᑦ ᖃᐅᔨᑎᑦᑎᒍᓐᓇᖅᑐᑦ ᐱᐅᔪᒥ ᖃᓄᐃᓐᓂᐅᔪᒥᑦ ᐊᑦᑐᖅᑕᐅᒋᐊᙵᐅᑎᓄᑦ ᓯᓚᐅᑉ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᖁᑎᑦᑐᒥ ᐅᑭᐅᑕᖅᑐᒥ ᐃᓱᒪᒋᔭᐅᓯᒪᓚᐅᖅᑐᑦ ᓇᓄᕐᓄᑦ. ᑭᓯᐊᓂᓕ, ᓇᓄᓇᐃᖅᓯᔪᒧᑦ ᐊᓯᔾᔨᕐᓂᖏᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ ᓯᑯᒥ ᑲᔪᓯᑉᐸᑕ ᑖᒃᑯᐊ ᐱᐅᔪ)ᓂᑦ ᐊᑦᑐᐃᓂᖃᕐᓂᐊᖅᑐᒋᔭᐅᔪᑦ ᐋᖅᑕᐅᓗᓂ ᑭᖑᕝᕕᖅᑕᐅᓗᓂᓘᓐᓃᑦ ᐱᐅᖏᑦᑐᓄᑦ ᐊᑦᑐᐃᓂᖅᓴᐅᑎᓪᓗᒍ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᐊᑐᓕᖁᔭᐅᔪᑦ ᖃᐅᔨᓴᐃᓐᓇᖅᑐᑦ ᓯᑯᖓ ᓇᔪᖅᑕᐅᑉᐸᑦᑐᑦ ᐊᒻᒪ ᖃᐅᔨᓴᑲᐃᓐᓇᖃᑦᑕᕐᓗᑎ ᖃᐅᔨᒪᔾᔪᑎᓄᑦ ᐊᓯᔾᔨᐅᖅᑐᓂᑦ ᑖᒃᑯᓂᖓ ᖃᓂᐅᓇᐅᔪᓂᑦ ᐊᒥᓲᓂᖏᓐᓄᑦ.

ᑎᑎᕋᖅᓯᒪᔪᓂᑦ ᐃᓕᓴᕆᔭᐅᔪᑦ

Aars, J., T.A. Marques, S.T. Buckland, M.Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear population size. Marine Mammal Science 25:35-52.

Anon. 2009. Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations, 9 pp.

Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusculanum Press, University of Copenhagen, Copenhagen, Denmark.

Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1, Government of Nunavut, 43 pp.

Dowsley, M., and G. W. Wenzel. 2008. ''The time of the most polar bears'': A co-management conflict in Nunavut. Arctic 61:177–189.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st- century polar bear habitat distribution from global climate models. Ecological Monographs 79:25–58.

Laidre, K. L., E. W. Born, P. Heagerty, Ø. Wiig, H. Stern, R. Dietz, J. Aars, and M. Andersen. 2015. Shifts in female polar bear (*Ursus maritimus*) habitat use in East Greenland. Polar Biology 38:879–893.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302–1320

Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713–1725.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Population status. Status table. http://pbsg.npolar.no/en/status/status-table.html

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.

Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. Marine Mammal Science 32:181–201.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2010. Report of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 18 pp

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

Taylor, M.K., S. Akeeagok, D. Andriashek, W. Barbour, E.W. Born, W. Calvert, D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, F. Messier 2001. Delineation of Canadian and Greenland Polar Bear (*Ursus maritimus)* populations by cluster analysis of movements Canadian Journal of Zoology 79: 690-709.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

ᐊᖑᓇᓱᒃᑕᐅᔪᓂᒃ ᖃᐅᔨᓴᕐᓂᐅᔪᖅ ᓴᓐᓂᕈᐱᐅᑉ ᑕᕆᐅᖓᓂ ᐊᒻᒪ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ ᓇᓄᕐᓂᒃ ᐊᕕᒃᑐᖅᓯᒪᔪᓄᑦ ᑲᑎᙵᐅᖅᑐᓄᑦ: ᓇᐃᓈᕈᑎᖏᑦ ᓇᓂᓯᐅᔪᓂ

ᐱᕗᒌᔭᖅᑕᐅᓯᒪᔪᑦ ᑲᑎᒪᓂᐊᕐᓂᖏᓐᓄᑦ ᑲᓇᑕᒥ-ᑲᓛᔅᓰ ᓄᓈᑦ ᑲᑐᔾᔨᖃᑎᒌᒃᑐᓄᑦ ᑲᑎᒪᔨᓄᑦ ᓇᓄᕐᓄᑦ

ᐋᑐᐋ, ᑲᓇᑕ

ᐅᑐᐱᕆ 25–27, 2017

ᓴᓐᓂᕈᐱᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᓇᓄᐃᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᑲᑐᔾᔭᐅᓪᓗᑎᒃ ᐊᐅᓚᑕᐅᖕᒪᑕ ᑲᓇᑕᒥᑦ (ᓄᓇᕗᑦ) ᐊᒻᒪᓗ ᖃᓛᑦᓰᓂᑦ. *ᑲᓇᑕᒥ-ᑲᓛᔅᓰ ᓄᓈᑦ ᑲᑐᔾᔨᖃᑎᒌᒃᑐᓄᑦ ᑲᑎᒪᔨᓄᑦ ᓇᓄᕐᓄᑦ* (JC) ᐱᓕᕆᐊᕆᔭᕐᓂᖅᓯᑎᑦᑎᖕᒪᑦ ᐊᐅᓚᑦᑎᓂᕐᒥᑦ ᒪᕐᕉᖕᓂ ᐱᓕᕆᔪᓐᓇᕐᓂᖃᕐᕕᐅᔫᖕᓂ. ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑦ (JC) ᐅᖃᐅᔭᐅᒋᐊᓱᖑᒪᑕ ᐅᑯᓇᙵᑦ *ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ* (SWG) ᐃᓚᖃᕐᑐᑎᒃ ᐃᓚᒋᔭᐅᔪᓂᑦ ᑕᒪᒃᑭᖏᓐᓂᑦ ᓄᓇᓕᕐᔪᒥᐅᑕᓂᑦ. ᐱᒋᐊᕐᓗᓂ 2011–2014, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐱᓕᕆᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᑐᑎᒃ ᓴᓐᓂᕈᐱᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐊᒻᒪᓗ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖓᓂ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ. ᐱᔾᔪᑕᐅᔪᖅ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒧᑦ ᑐᑭᓯᓇᓱᐊᕐᓗᑎᒃ ᐊᒃᑐᐃᓂᖓᓂᑦ ᐊᓯᔾᔨᕐᓂᖓᑕ ᑕᕆᐅᑉ–ᓯᑯᖓᑕ ᖃᓄᐃᑦᑑᓂᕆᕙᒃᑕᖓᓂᑦ ᐊᒻᒪᓗ ᐃᓱᒪᓕᐅᕈᑕᐅᔪᒪᓪᓗᓂ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ [TAH]; ᑲᑎᓗᒋᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐃᓄᖕᓂᑦ ᐊᑐᓂ ᐊᕐᕌᒍᒥ ᓄᖑᓗᐊᖅᑎᑕᐅᖏᓪᓗᑎᒃ ᐊᖑᓇᓱᒃᑕᐅᑎᓪᓗᒋᑦ). ᓄᑖᑦ ᖃᐅᔨᓇᓱᐊᕐᓃᑦ ᐱᔭᐅᔭᕆᐊᖃᓚᐅᕐᒪᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᑐᓴᐅᒪᔪᑎᒃᓴᑦ ᐅᑯᓇᖐᖅᑐᑦ ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᑐᑎᒃ ᑕᐅᕙᓂ 1990–ᖏᓐᓂ ᐱᑐᖃᐅᓗᐊᓕᓚᐅᕐᒪᑦ. ᑕᒪᒃᑮᔪᑦ ᓴᕿᑕᐅᓯᒪᔪᑦ ᐱᒋᐊᕐᓗᑎᒃ 2011–2014 ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᑦ ᓴᕿᑕᐅᓯᒪᖕᒪᑕ ᐊᓯᐊᓃᓪᓗᑎᒃ ᐅᓂᒃᑳᓕᐊᕆᓯᒪᔪᓂ.

ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑦ (JC) ᑐᒃᓯᕋᓚᐅᕐᒪᑕ ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᐊᑐᕈᒪᓪᓗᑎᒃ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ 2011–2014-ᒥᖐᖅᑐᓂᑦ ᖃᐅᔨᓇᓱᐊᕈᑎᓂᑦ ᐅᖃᐅᔨᒋᐊᕈᑕᐅᖁᓪᓗᒋᑦ ᐊᖑᓇᓱᐊᕐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᔨᓂᑦ. ᐱᓗᐊᙳᖅᑐᒥᑦ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑦ (JC) ᖃᐅᔨᔪᒪᓚᐅᕐᒪᑕ ᖃᓄᑎᒋᑭᐅᒪᖕᒪᑕ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᓈᒪᒃᑲᔭᕐᑐᑎᒃ ᐱᖓᓱᓄᑦ ᐊᐅᓚᑦᑎᓂᕐᓄᑦ ᑎᑭᑕᐅᔪᒪᔪᓄᑦ: (1) ᐋᖅᑭᐅᒪᑎᑦᑎᓗᑎᒃ ᓄᖑᑉᐸᓪᓕᐊᕐᑯᓇᒋᑦ ᐅᓄᕐᓂᒃᑲᓂᖏᑕ ᐊᖏᔫᓂᑦ; (2) ᐋᖅᑭᐅᒪᑎᓪᓗᒋᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐅᖏᔫᓂᖏᑦ ᐋᖅᑭᑎᐊᖅᓯᒪᖁᓪᓗᒋᑦ ᐅᓄᕐᓂᖏᑦ ᓇᓄᐃᑦ ᐊᕙᑎᑦᑎᓐᓂ ᒪᑭᒪᐃᓐᓇᖅᑎᑕᐅᔪᓐᓇᕐᓂᖏᑦ; ᐊᒻᒪᓗ ᓴᓐᓂᕈᐱᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᑕᒪᐅᖓᑐᐊᖅ, **(3) ᐅᓄᖏᓐᓂᖅᓴᐅᓕᖅᑎᑕᐅᓗᑎᒃ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐊᖏᔫᓂᖓ ᐃᒪᓐᓇᑎᒋᓪᓗᐊᐸᓗᒃ 30% ᐳᓴᓐᑎᑦ 10–15 ᐊᕐᕌᒍᓂ.** ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑦ (JC) ᐃᓕᓴᖅᓯᓚᐅᕐᒪᑕ ᑕᒪᒃᑯᐊ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ

ᖃᐅᔨᓴᓂᑦ ᐃᓚᖃᕐᒪᑕ ᓇᓗᓈᕿᔪᓂᑦ, ᐊᔪᕐᓇᕐᓂᐊᕐᒪᑦ ᑖᒃᑯᓄᖓ ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓇᓚᐅᑦᑎᔪᒥᑦ ᑮᐅᑦᑎᐊᕈᑎᖃᕆᐊᒃᓴᖅ. ᑕᐃᒪᐃᒪᑦ, ᑲᑐᔾᔨᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑦ (JC) ᑐᒃᓯᕋᓕᖅᐸᑦ, ᐊᑐᓂ ᐅᓄᕐᓂᒃᑲᓐᓂᐅᔪᑦ, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓈᓴᐅᓯᕆᖁᓪᓗᒋᑦ ᖃᓄᑎᒋᐅᒐᔭᕐᓂᖏᓐᓂᑦ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᖁᕝᕙᓯᒃᑐᒪᕆᐅᔪᓐᓇᕐᓗᑎᒃ (90% ᐳᓴᓐᑎᓂᑦ) ᐱᔪᓐᓇᕋᔭᕐᓂᖏᑦ ᐊᑐᓂ ᐊᐅᓚᑦᑎᓂᐅᔪᓄᑦ ᑎᑭᑕᐅᔪᒪᔪᓂᑦ (ᑕᒪᓐᓇ ᑐᑭᖃᒃᐳᖅ ᐊᒃᐸᓯᒃᑐᒪᕆᐅᒐᔭᖁᓪᓗᒋᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᖁᓇᒥᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ), ᐱᖃᓯᐅᑎᓗᒋᑦ ᖃᓄᑎᒋᐅᒐᔭᕐᓂᖏᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᐊᑯᓐᓈᕿᓗᑎᒃ (70% ᐳᓴᓐᑎᖏᓂᑦ) ᐱᔪᓐᓇᕋᔭᕐᓗᑎᒃ ᐊᑐᓂ ᐊᐅᓚᑦᑎᓂᐅᔪᓄᑦ ᑎᑭᑕᐅᔪᒪᔪᓂᑦ (ᑕᒪᓐᓇᓗ ᑐᑭᖃᕐᓗᓂ ᐊᑯᓐᓈᕿᔪᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᖁᓇᒥᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓃᑦᑐᑦ).

ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᑭᐅᔪᑎᖃᓚᐅᕐᒪᑕ ᐊᐱᖅᑯᑎᒋᔭᐅᔪᓄᑦ ᐊᑐᕐᓗᑎᒃ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ ᐅᑯᓇᖕᒑᖅᑐᓂᑦ 2011-2014 ᖃᐅᔨᓇᓱᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕈᑎᓂᑦ ᐃᓚᒋᔭᐅᓗᑎᒃ ᐅᓄᕐᓂᖏᓐᓄᑦ ᐋᖅᑭᒃᓱᐃᔪᑎᓂᑦ ᓇᓄᕐᓂᑦ. ᐅᓄᕐᓂᖏᓐᓂᑦ ᐋᖅᑭᒃᓱᐃᔪᑦ ᐱᖃᓯᐅᔾᔨᖕᒪᑕ ᑕᕆᐅᑉ–ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖏᓐᓄᑦ. ᖃᐅᔨᓴᖅᓯᔪᑎᒋᔭᖏᓐᓂ, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᓱᒪᒃᓴᓯᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᐅᓄᖅᑐᓂᑦ ᖃᓄᐃᓕᔪᑕᐅᔪᓐᓇᖅᑐᓂᑦ ᐊᒃᑐᐃᓂᖏᓐᓂᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH), ᐱᖃᓯᐅᑎᓗᒋᑦ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐊᖑᑎᑦ ᐊᕐᓇᓪᓗᐃᓪᓗ ᐊᒻᒪᓗ ᐅᑭᐅᖃᕐᓂᖏᑦ ᓇᓐᓄᒃᑕᐅᕙᒃᑐᑦ ᓇᓄᐃᑦ, ᐊᒻᒪᓗ ᓯᕗᓂᒃᓴᒥ ᐱᓕᕆᓂᐅᒐᔭᖅᑐᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᕐᒥᑦ ᐊᒻᒪᓗ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᕆᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᕐᒍᑦ (TEK) ᖃᐅᔨᓇᓱᐊᕐᓃᑦ. ᓯᕗᓂᒃᓴᒥ ᐱᓕᕆᐊᖑᒐᔭᖅᑐᑦ ᖃᐅᔨᓇᓱᐊᕐᓂᑦ ᐱᒻᒪᕆᐊᓘᖕᒪᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ, ᓄᑖᓂᒃ ᖃᐅᔨᓇᓱᐊᕐᓂᖅᑕᖃᖅᑲᑦᑕᖅᐸᑦ ᕿᓚᒻᒥᐅᓗᑎᒃ, ᐊᒃᐸᓯᒡᓂᖅᓴᐅᒐᔭᕐᒪᑕ ᐅᓄᕈᓐᓃᖅᑎᑕᐅᓗᐊᕋᔭᕐᓂᖏᑕ ᐅᖓᑎᓗᐊᖓᓄᑦ–ᓇᓐᓄᒃᑕᐅᖃᑦᑕᖃᑦᑕᖅᐸᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᖃᐅᓇᓱᐊᕈᑎᑦ ᐱᔭᐅᑎᑦᑎᒐᔭᕐᒪᑕ ᓄᑖᓂᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᓂᑦ ᐊᐅᓚᑦᑎᓯᓄᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᕋᔭᖅᑐᓂᒃ ᐋᖅᑭᒋᐊᖅᓯᔪᑕᐅᔪᓐᓇᕐᓗᑎᒃ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH).

ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓ

2011–2013 ᖃᐅᔨᓴᕐᓇᓯᐊᓂᒃᑯᑦ ᖃᐅᔨᓵᖅᓃᑦ ᒥᒃᓴᐅᓴᒃᓯᓚᐅᕐᒪᑕ ᐃᒪᓐᓇᐸᓘᓇᓱᕆᔭᐅᓪᓗᑎᒃ 2800 ᓇᓄᐃᑦ (95% ᓈᓴᐅᑏᑦ ᐃᓗᐊᓂ [CI]=2,059–3,593) ᑕᒫᓂ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᓇᔭᐅᔪᓂᑦ. ᑕᕆᐅ ᓯᑯᖓ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᒥᑭᓪᓕᒃᑲᓐᓂᖅᓯᒪᖕᒪᑦ ᐊᓂᒍᖅᑐᓂ ᖃᑦᓯᒃᑲᓪᓚᐃᑦ ᖁᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ. ᑕᒪᓐᓇ ᒥᑭᓪᓕᕆᐊᖅᑎᑦᑎᓯᒪᖕᒪᑦ ᓇᔪᖅᑕᐅᓯᒥᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖅᑭᓐᓄᑦ ᐊᑐᖅᑕᐅᓯᒥᑦ, ᐊᒻᒪᓗ ᐱᕆᐊᓕᖅᑎᑦᑎᓯᒪᑐᐃᓐᓇᕆᐊᖃᕐᒪᑦ ᓇᓄᕐᓂᑦ ᓴᓗᒻᓂᖅᓴᐅᓕᖅᑎᑦᑎᓗᓂ ᐊᒻᒪᓗ ᐊᑎᖅᑕᖃᖅᑭᓐᓂᖅᓴᐅᓕᖅᑎᑦᑎᓗᓂ. ᓇᔪᖅᑕᐅᕙᒃᑐᖅ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᒪᑭᒪᑎᑦᑎᓂᐊᖅᑐᒃᓴᐅᓕᕐᒪᑦ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᓂᑦ ᓇᓄᕐᓂᑦ ᓯᕗᓂᒃᓴᒥ ᑕᕆᐅᑉ-ᓯᑯᖓ ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓗᓂ ᑲᔪᓯᑐᐃᓐᓇᕋᔭᖅᐸᑦ. ᑕᐃᒪᐃᒪᑦ, ᑕᕆᐅᑉ-ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᐱᖃᓯᐅᔾᔭᐅᓚᐅᖅᐳᖅ ᐱᐅᓈᖅᕆᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᖃᒃᑎᑕᐅᓗᓂ ᓇᓄᐃᑦ ᐅᓄᕐᓂᖅᑭᓐᓂ ᐅᑯᑎᒎᓇ ᐅᓄᕐᓂᖅᑭᓐᓄᑦ ᐋᖅᑭᒃᓯᐃᔪᑎᖓᓂ.

ᖃᐅᔨᓴᕐᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᖏᑦ SWG ᖃᐅᔨᔪᖅᑕᐅᓚᐅᕐᒪᑦ ᐊᔪᕐᓇᕋᔭᕐᓂᖓᓂᑦ ᐊᑐᕋᓯᕆᐊᖓ ᑎᑭᑕᐅᔪᒪᔪᖅ (1) ᐋᖅᑭᐅᒪᑎᑦᑎᓂᖅ ᓄᖑᑉᐸᓪᓕᐊᖏᓪᓗᑎᒃ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᓯᕗᓂᒃᓴᒥ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᑉ-ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᐅᓕᖅᑎᑦᑎᒐᔭᕐᒪᑦ ᓇᓄᕐᓂᑦ ᖃᓄᑎᒋᐅᒐᔭᕐᓂᕋᓗᐊᖅᐸᑕ ᐊᖑᓇᓱᐊᖅᑕᐅᓂᖏᑦ ᐊᒻᒪᓗ ᐊᑐᓚᓐᖑᐊᖅᑎᑦᑎᓂᓇᔭᐅᓚᐅᖅᑐᖅ ᐃᒪᓐᓇᐅᓂᖃᐃᖕᒪᑦ ᐊᑐᖅᑕᐅᒐᔭᕐᓂᖅᐸᑦ ᑎᑭᑕᐅᔪᒪᔪᖅ (3) 30% ᐳᓴᓐᑎᖏᓐᓂᑦ ᐅᓄᖏᓐᓂᖅᓴᐅᓕᖅᑎᑦᑎᓂᖅ ᐱᔭᒃᓴᐅᒐᔭᖏᓪᓚᑦ ᑕᐃᒪᓐᓇ ᐅᖃᐅᑎᕆᔭᐅᖕᒪᑦ ᐅᓄᕐᓂᖏᑕ ᖁᒃᓴᓇᕐᓂᖏᓐᓂ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᒐᔭᕐᓂᖏᓐᓂᑦ. ᖃᐅᔨᓴᕐᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᖏᑦ (SWG) ᐃᓱᒪᓕᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᖁᕝᕙᓯᒃᑐᒪᓇᐅᓂᖓᓂᑦ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓇᓂᐊᖃᕐᓂᖓᑕ ᑎᑭᑕᐅᔪᒪᔪᑦ (2) ᐊᑐᕐᓗᒍ ᐅᓪᓗᒥᐅᓕᖅᑐᖅ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᑎᑭᐅᒪᑎᓪᓗᒋᑦ 120–ᓄᑦ ᓇᓄᐃᑦ ᐊᑐᓂ ᐊᕐᕌᒍᒥ (ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᑎᑭᐅᒪᓗᑎᑦ 4.3% ᐳᓴᓐᑎᓄᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ), ᐊᒻᒪᓗ ᐊᑯᓐᓈᖅᑭᔪᖅ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓇᓂᐊᖃᕐᒪᑕ ᑎᑭᑕᐅᔪᒪᔪᖅ (2) ᐅᓪᓗᒥᐅᓕᖅᑐᖅ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᑎᑭᐅᒪᓗᑎᒃ 160–ᓄᑦ ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᑐᓂ ᐊᕐᕌᒍᒥ (ᓇᓐᓄᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐅᓄᕐᓂᖅᑲᕐᓗᑎᒃ 5.7% ᐳᓴᓐᑎᓂᒃ ᐅᓄᕐᓂᖏᓐᓂᑦ). ᑕᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᒪᓕᑐᐃᓐᓇᕐᒪᑕ ᖃᑦᓯᒃᑲᓪᓚᖏᑦ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᓂᑦ. ᓯᕗᓪᓕᖅᐹᖅ, ᐃᓱᒪᒋᔭᐅᓚᐅᕐᒪᑕ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐊᖑᑦ ᐊᕐᓇᓪᓗᐃᓪᓘᓐᓂᑦ ᓇᓐᓄᒃᑕᐅᖃᑦᑕᕐᓂᖓᓗᒃᑕᖏᑦ ᑕᐃᒪᓐᓇᖅᓴᐃᓐᓈᕐᓂᐊᖅᐸᑕ ᐊᑐᖅᑕᐅᖅᑕᖅᑕᓚᐅᖅᑐᓂ ᐊᓂᒍᖅᑐᓂ ᖁᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ. ᑕᐃᒪᓕ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᖃᓖᑦᓖᑦ ᓇᓐᓄᒃᐸᒃᑕᖏᑦ ᑲᑎᑕᐅᑎᓪᓗᒋᑦ, ᐃᒪᓐᓇᐅᓕᔪᓚᐅᖅᐸᖅ ᓇᓐᓄᒃᑕᐅᔪᑦ ᐊᑯᓐᓂᖓᓂ 1998 ᐊᒻᒪᓗ 2013 ᒪᑯᐊᖑᓚᐅᕐᒪᑕ 1.25 ᐊᖑᑎᑦ ᐱᔭᐅᑎᓪᓗᒍ 1 ᐊᕐᓇᓪᓗᒃ. ᑕᒪᓐᓇᓕ ᐊᖑᑕᐅᒃᐸᑕ ᐊᕐᓇᓪᓘᒃᐸᑕ ᓇᓐᓄᒃᑕᐅᓯᑦ ᐊᓯᔾᔨᖅᐸᑕ, ᐊᓯᐃᓪᓚᒃᑕᐅᖅ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑐᑦ (TAH) ᐊᓯᔾᔨᕋᔭᖅᑐᒃᓴᐅᖕᒥᔪᑦ. ᐱᖃᑖᓂ, ᑕᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᐃᒪᓐᓇᐅᔭᓇᐊᖃᕐᓂᖅᑎᓐᓇᑕ ᓄᑖᓂᑦ ᖃᐅᔨᓴᕐᓇᓯᐊᓂᒃᑯᑦ ᐊᒻᒪᓗ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᑕᓕᕆᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᑦ (TEK) ᖃᐅᔨᓴᕐᓇᓯᐊᕈᑎᑦ ᐱᓕᕆᐊᖑᖃᑦᑕᕐᓂᐊᕐᓗᑎᒃ ᕿᓚᒻᒥᐅᓂᖅᓴᐅᖃᑦᑕᕐᓗᑎᒃ ᓯᕗᓂᒃᓴᒥ, ᓄᑖᓂᑦ ᖃᐅᔨᓴᕐᓇᓯᐊᕈᑎᒃᓴᓂᑦ ᐊᑐᐃᓐᓇᐅᔪᖅᑕᐅᓂᖏᓐᓂᑦ ᐊᑐᖅᑕᐅᖃᑦᑕᕐᓂᐊᕐᓗᒍ ᓈᓴᐅᓯᓇᔪᑕᐅᓯᒃᑯᓪᓗᒋᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᖅᑐᓄᑦ (TAH) ᓈᑐᐊᕋᒪᑕ 15 ᐊᕐᕌᒍᑦ. ᑕᐃᒪᓕ ᓄᑖᑦ ᖃᐅᔨᓴᕐᓇᓯᐊᓂᒃᑯᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᐃᑦ ᐊᑐᐃᓐᓇᐅᖏᑉᐸᑕ ᓈᓴᐅᓯᓇᐊᔪᑕᐅᓯᒃᑯᓗᒋᑦ

ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᙳᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) 15 ᐊᕐᕌᒍᓂ, ᖁᑭᕙᓯᒡᓂᖓ ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᙳᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᐅᖃᐅᑕᐅᖃᐅᔪᖅ ᑕᑉᐸᒃᑯᓇᓂ ᐊᑐᖅᑕᐅᑎᑦᑎᔪᓐᓇᕐᔭᖏᒪᑕ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑎᑭᑕᐅᔪᒪᔪᒥᑦ ᐅᖃᐅᑕᐅᔪᒥᑦ ᑲᑐᔾᔨᑐᑎᒃ ᐱᓕᕆᖁᔨᔪᑎᓂᑦ (JC) ᐊᒻᒪᓗ ᖁᑭᕙᓯᒡᓂᖅᓵᓗᑐᐃᓐᓇᕆᐊᖃᒃᑯᓂ ᐅᓗᕆᐊᓇᖅᑐᒦᓕᖅᑎᑦᑎᒐᔭᕐᓂᖓᑦ ᐅᖓᑎᓗᐊᖓᓄᑦ−ᓇᓐᓄᒃᑕᐅᖃᑦᑕᕐᓂᕆᒐᔭᖅᑕᖏᑕ ᐊᑐᕐᓂᐊᑐᐃᓐᓇᖅᑐᓂ ᖃᑦᓯᑲᓪᓚᖕᓂ ᖁᓕᐅᔪᖅᑐᑦ ᐊᕐᕌᒍᓂ. ᐱᖓᔪᖓᓐᓂ, ᑕᒪᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᐃᒪᓐᓇᐅᓕᖅᑎᑦᑎᖓᒪᑕ ᐊᒃᐸᓯᒪᑕ ᒥᒃᓴᐅᓴᒃᓯᔪᑕᐅᔪᑦ ᐆᒪᑯᑖᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᐊᑐᖅᑎᓪᓗᒋᑦ 2011-2013 ᓰᓕᓚᐅᖏᒪᑕ ᓴᖅᑭᑕᐅᔪᑦ. ᐊᑐᓚᕆᒃᑐᒥ, ᑕᒪᒃᑯᐊ ᐊᒃᐸᓯᒃᑐᑦ ᒥᒃᓴᐅᓴᒃᓯᔪᑕᐅᔪᑦ ᐃᓚᖓᒍᑦ ᓱᓕᓂᖃᑐᐃᓐᓇᕆᐊᓕᐅᒐᓗᐊᑦ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᖃᕐᓂᖓᓄᑦ ᑕᕆᐅᑉ−ᓯᖁᖃᕈᓐᓅᖅᐸᓪᓕᐊᓂᐅᑉ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᓇᓄᕐᓂᑦ.

ᖃᐅᔨᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓇᑐᓇᐃᖅᓯᓚᐅᕐᒪᑦ ᖃᑦᓯᑲᓪᓚᖕᓂᑦ ᐃᓱᒫᓗᑕᐅᔪᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓃᒃᑲᓐᓂᕆᔭᐅᔪᓂᑦ. ᓯᕗᓪᓕᖅ, ᒫᓐᓇ ᖃᑦᓯᓐᓇᐅᓂᖅᓴᐅᓕᕐᒪᑕ ᐊᙳᑎᑦ ᓇᓄᐃᑦ ᐊᕐᓇᓪᓗᖕᓂᓪᓕ ᐅᓄᕐᓃᒃᑲᓐᓂᕆᔭᐅᔪᓂ, ᐊᒻᒪᓗ ᐊᙳᑎᑦ ᐆᒪᑯᑖᖃᑦᑕᕐᓂᖕᑦ ᐊᒃᐸᓯᒡᓂᖅᓴᐅᓪᓗᑎᒃ ᐊᕐᓇᓪᓗᐃᑦ ᐆᒪᑯᑖᖃᑦᑕᕐᓂᖏᓐᓂᑦ. ᐊᙳᑎᑦ ᓇᓄᐃᑦ ᑲᔪᓯᓗᑎᒃ ᖃᑦᓲᓐᓇᕈᒃᐸᓪᓕᐊᑐᐃᓐᓇᖅᐸᑕ ᐃᒪᓐᓇᐅᓕᖅᑎᑦᑎᔪᓐᓇᕐᒪᑦ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᐅᓕᕐᓗᑎᒃ ᐊᑎᖅᑕᐃᑦ ᐃᕐᓂᐊᙳᔪᑦ, ᐊᒻᒪᓗ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᐅᓕᖅᔭᕐᒪᑕ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐅᓄᕐᓂᖏᑦ. ᑐᑭᓯᓂᖅ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᓐᓂᑦ ᐊᙳᑎᑦ ᓇᓄᐃᑦ ᐱᒪᕆᐊᓘᖕᒪᑦ ᐊᑐᕋᔭᕐᓂᖓᓄᑦ ᓯᕗᓂᒃᓴᒥ ᖃᐅᔨᓇᓯᐊᕐᓂᕐᒧᑦ. ᐱᖃᑖᓂ, ᐊᐅᓚᑦᑎᕚᑦ ᓂᕆᐅᒋᐊᖃᓕᖅᑐᑦ ᑕᒪᓐᓇ ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᙳᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᑕᒥᓂ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂ ᖃᑦᓲᓐᓇᕈᒃᐸᓪᓕᐊᓂᐊᕐᒪᑕ ᐊᑐᕐᓂᐊᖅᑐᓂ ᖃᑦᓯᑲᓪᓚᖕᓂ ᖁᓕᐅᔪᖅᑐᑦ ᐊᕐᕌᒍᓂ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᑕᕆᐅᑉ−ᓯᖁᖃᕈᓐᓅᖅᐸᓪᓕᐊᓂᖓ ᖃᑦᓲᓐᓇᕈᑎᑦᑎᑲᓐᓂᕐᓂᐊᕐᒪᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ ᓇᓄᐃᑦ ᑕᒥᓂ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᓇᔪᖅᑕᐅᕙᒃᑐᒥ ᒪᑭᒪᔪᑕᐅᔪᓐᓇᕐᓂᖓᓂ. ᐱᖓᔪᖓᓐᓂ, ᐃᒪᓐᓇᔪᓐᓇᕐᒥᔪᖅ ᐅᓄᕐᓂᓇᔭᐅᔪᓄᑦ ᐱᓕᕆᔪᑕᐅᕙᒃᑐᖅ ᐊᑐᖅᑕᐅᓯᖅ ᖃᐅᔨᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᖃᑎᒋᓂᑦ (SWG) ᑐᖓᐅᑎᑐᑎᒃ−ᒥᒃᓴᐅᓴᖅᓯᔪᒥᓂᐅᑐᐃᓐᓇᕆᐊᖃᕐᒪᑕ ᐱᐅᓈᖏᑦᑐᓂᑦ ᐊᒃᑐᐃᓂᖃᕋᓯᒋᑐᒋᑦ ᑕᕆᐅᑉ−ᓯᖁᖃᕈᓐᓅᖅᐸᓪᓕᐊᓂᖓᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᓇᓄᐃᑦ. ᑕᒪᓐᓇ ᑕᐃᒪᓐᓇᐃᑉᐸᑦ, ᐊᓯᐃᓪᓚᒃ ᑲᑎᑐᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᙳᓇᓯᐊᖅᑕᐅᔪᓐᓇᕐᑐᑎᒃ (TAH) ᖁᑭᕙᓯᒡᓂᕆᑦ ᐅᖃᐅᑕᐅᖃᐅᔪᑦ ᑕᑉᐸᒃᑯᓇᓂ ᖁᑖᓂ ᐃᒪᓐᓇᐃᓕᑎᑦᑎᔪᓐᓇᕐᒪᑕ ᐅᖓᑎᓗᐊᖓᓄᑦ−ᓇᓐᓄᒃᑕᐅᑎᑦᑎᖃᑦᑕᕐᑐᑎᒃ. ᑕᐃᒪᓇᐅᓗᖅᑎᓪᓗᒍ ᑕᕆᐅᑉ−ᓯᖁᖃᕈᓐᓅᖅᐸᓪᓕᐊᓂᖓ ᐅᓗᕆᐊᓇᕈᑎᖃᕐᓂᐊᕐᒪᑦ ᐅᓄᕐᓂᖅᓴᓄᑦ ᓇᓄᕐᓄᑦ ᐊᑐᕐᓂᐊᖅᑐᒧᑦ−ᑕᐅᕗᖓᑐᖃᒃ, ᐊᓯᖏᑦ ᐅᓄᕐᓂᒃᑲᓅᖏᑦ ᒫᓐᓇ ᐊᑐᖅᐸᓪᓕᐊᓕᕐᒪᑕ ᑕᕆᐅᑉ−ᓯᖁᖃᕈᓐᓅᖅᐸᓪᓕᐊᓂᖓᓄᑦ ᐊᔾᔨᖏᖏᑕᖏᑎᒍ ᖃᓄᐃᓕᕙᓪᓕᐊᓪᓗᑎᒃ, ᐊᒻᒪᓗ ᐊᔪᕐᓇᐸᓗᖕᒪᑦ ᓇᓚᐅᑐᒋᑦ ᖃᐅᔨᒪᕙᖁᕐᓯᒋᐊᖏᑕ ᓯᕗᓂᒃᓴᒥ ᐊᒃᑐᐃᓂᕆᔭᖅᑕᖓ ᑕᒪᑐᒪ ᑕᕆᐅᑉ−ᓯᖁᖃᕈᓐᓅᖅᐸᓪᓕᐊᓂᖓᑕ. ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᑕᒪᒃᑯᐊ ᐃᓱᒫᓗᑕᐅᔪᑦ, ᖃᐅᔨᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᑐᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐊᑐᓕᖁᔨᓕᖁᔨᓕᒪᐅᕐᒪᑕ ᐱᓂᕐᓗᒃᑕᐅᔭᐃᖅᓯᓗᑎᒃ ᐱᓕᕆᖁᔨᓪᓗᑎᒃ ᓇᓐᓄᖃᑦᑕᕐᓂᕐᒧᑦ ᐊᐅᓚᑦᑎᓂᖅ ᐃᑲᔪᖁᓪᓗᒍ ᐊᒃᓵᖏᓐᓂᖅᓴᐅᓕᖅᑎᑦᑎᔪᑕᐅᓗᓂ ᐅᖓᑎᓗᐊᖓᓄᑦ−ᐊᙳᑦᑕᐅᓂᖏᓐᓂᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ (BB) ᐅᓄᕐᓃᒃᑲᓐᓂᕆᔭᐅᔪᓂ.

ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓ

2012–2014 ᖃᐅᔨᓇᓯᐊᓂᒃᑯᑦ ᖃᐅᔨᓴᕐᓃᑦ ᒥᒃᓴᐅᓴᒃᓯᓚᐅᕐᒪᑕ ᐃᒪᓐᓇᐸᓘᓇᓯᒋᔭᐅᓪᓗᑎᒃ 360 ᓇᓄᐃᑦ (95% ᓈᓴᐅᑏᑦ ᐃᓗᐊᓂ [CI]=221–493) ᑕᒦᓂ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ KB ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂ. ᑕᐃᒪᐃᑲᓗᐊᖅᑎᓪᓗᒍ ᑕᓇᐅᑉ ᓯᑯᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓂ ᒥᑭᓂᖅᓴᐅᓕᕋᓗᐊᖅᑎᓗᒍ ᑕᐃᒪᖓᓂᑦ ᐊᓂᒍᖅᑐᓂ ᖃᑦᓯᑲᓪᓚᖓᓂ ᖁᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ, ᑕᒪᓐᓇ ᓱᖅᑭᑎᑦᑎᖏᒪᑦ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᕐᒥᑦ ᒪᑯᓂᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ ᓇᓄᖏᓐᓂᑦ. ᐄᓚᒃ, ᑕᓇᐅᑉ–ᓯᑯᖓᑕ ᖃᓄᐃᑦᑑᓕᕐᓂᖓ ᐃᒪᓐᓇᐃᓕᑎᑦᑎᑐᐃᓐᓇᓇᐊᖃᕐᒪᑦ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᓃᑦᑐᓂᑦ ᖃᑦᔫᓐᓇᕈᒃᐸᓪᓕᐊᑎᑦᑎᖏᓪᓗᓂ ᓇᓄᕐᓂᑦ. ᑕᐃᒪᐃᒪᑦ, ᑕᓇᐅᑉ–ᓯᑯᖃᕈᓐᓃᖅᐸᓪᓕᐊᓂᖓ ᐃᓚᒋᓕᐅᑎᔭᐅᓚᐅᖏᓚᖅ ᐱᐅᓈᖏᑦᑐᒥᑦ ᐊᒃᑐᐃᓂᖓᓄᑦ ᐅᓄᕐᓂᖏᓐᓂᑦ ᓇᓄᐃᑦ ᐱᓕᓇᔪᑕᐅᖃᑦᑕᖅᑐᓂ.

ᖃᐅᔨᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᓱᒪᓕᐅᕈᑎᖃᓚᐅᕐᒪᑕ ᐃᒪᓐᓇᐅᓂᖓᓂᑦ ᐊᔪᕐᓇᕐᒪᑦ ᓇᓗᓇᐃᖅᓯᔭᓇᐊᒃᓴᖅ ᐅᓄᕐᓂᖏᓐᓂᑦ ᑲᑎᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᐃᒪᓐᓇᐅᔪᓐᓇᕋᔭᕐᓂᖅᐸᑕ ᓂᓇᐅᓇᑦᑎᐊᕐᓗᑎᒃ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓇᐊᖃᕐᓂᖏᑦ ᑎᑭᑕᐅᔪᒪᔪᑦ (1) ᐊᒻᒪ (2). ᑖᓐᓇ ᐱᔾᔪᑎᖃᖏᑦᑐᖅ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ ᐱᕗᓪᓕᐊᑦᑎᐊᖏᓐᓂᖏᓐᓂᑦ. ᐃᒪᓐᓇᖕᒎᕐᓕ, ᐱᔾᔪᑎᖃᒃᐳᖅ ᓇᓗᓈᕐᑭᔪᓂᑦ ᑐᓴᐅᒪᔪᑎᒃᓴᑦᑕᖃᓚᐅᕐᒪᑦ ᖃᐅᔨᓇᓯᐊᖅᑎᑦ ᖃᐅᔨᓇᓯᐊᕈᑎᖏᓐᓂ, ᐱᔾᔪᑎᒋᓪᓗᒍ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᖃᑦᔫᓐᓇᐅᖓᑕ ᐊᒻᒪ ᐊᔪᕐᓇᐸᓗᒃᓗᑎᒃ ᖃᐅᔨᓇᓯᒋᐊᖏᑕ. ᖃᐅᔨᓇᓯᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᖃᐅᔨᓚᐅᕐᒪᑕ ᐊᑯᓐᓈᕐᑭᖕᒪᑦ ᐱᔭᐅᔪᓐᓇᑐᐃᓐᓇᓇᐊᖃᕐᓂᖓ ᑎᑭᑕᐅᔪᒪᔫᑉ (1) ᐅᓗᒥᐅᓕᖅᑐᖅ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᑎᑭᐅᒪᓗᑎᒃ 10 ᓇᓄᕐᓄᑦ ᐊᕐᕌᒍᑕᒫᑦ (ᐊᖑᑦᔭᑕᖃᑦᑕᕐᓂᖕᖏᑦ ᑎᑭᐅᒪᓗᑎᒃ 2.8% ᐳᓴᓐᑎᓄᑦ). ᑖᓐᓇ ᖃᐅᔨᓴᔭᐅᔪᖅ ᑐᙵᕕᖃᒃᑎᑕᐅᖕᒪᑦ ᑕᒪᒃᑭᖏᓐᓂᑦ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᒪᔪᑎᑦ ᑕᐅᕙᖓ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᓇᓄᖏᓐᓄᑦ, ᐊᒻᒪ ᐱᖁᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᓇᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᖅ (TEK) ᐃᒪᓐᓇᐅᓂᕋᐃᖕᒪᑕ ᑕᐃᒃᑯᐊ ᐱᕐᓕᕋᕐᕕᒡᔪᐊᑉ ᑕᓇᐅᖓᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᐅᓄᖅᓯᒃᑲᓐᓂᖅᓯᒪᖁᔨᓕᕐᒪᑕ ᑕᐃᒪᙵᓂᑦ ᐅᓄᕐᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ 1990-ᒥᑦ. ᐊᔾᔨᐸᓗᒋᓪᓗᓂᒋᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᑕ ᓴᓐᓂᕈᑎᐅᑉ ᑕᓇᐅᖓᑕ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ, ᑕᒪᒃᑯᐊ ᓴᕿᑕᐅᔪᑦ ᒪᓕᑐᐃᓐᓇᕐᒪᑕ ᖃᑦᓯᑲᓪᓚᖓᓂᑦ ᖃᓄᐃᓕᖓᓂᓇᔭᖏᑕ. ᓯᕗᓪᓕᖅᐹᖅ, ᐃᓱᒪᒋᔭᐅᓚᐅᕐᒪᑦ ᓇᓪᓕᐊᖑᒃᐸᑕ ᐊᖑᑦ ᐊᕐᓇᓪᓗᐃᓪᓘᓐᓂᑦ ᓇᓐᓄᒃᑕᐅᖃᑦᑕᕐᓂᓇᓚᐅᖅᑕᖏᑦ ᑕᐃᒪᓐᓇᖓᐃᓐᓈᕐᓂᐊᖅᐸᑕ ᐊᑐᖅᑕᐅᖃᑦᑕᓚᐅᖅᑐᓂ ᐊᓂᒍᖅᑐᓂ ᖁᓕᐅᔪᖅᑐᓂ ᐊᕐᕌᒍᓂ. ᑕᐃᒪᓕ ᓄᓇᕗᑦ ᐊᒻᒪ ᖃᓛᑦᓯᕐᒥᐅᑦ ᐊᖑᓇᓱᐊᖅᐸᒃᑕᖏᑦ ᑲᑎᑕᐅᔭᕐᒐᖓᑕ, ᐃᒪᓐᓇᑎᑭᐅᒪᔪᒃᐳᑦ 0.94 ᐊᖑᑎᑦ ᐊᑕᐅᓯᕐᒥᑦ 1 ᐊᕐᓇᓪᓗᓕᒐᖅᑐᖅᑳᕐᒐᖓ. ᐱᖅᑖᓂ, ᑕᒪᒃᑯᐊ ᓴᕿᑦᑐᑦ ᐃᒪᓐᓇᐅᔭᓇᐊᖃᒃᑎᑦᑎᖓᑕ ᓄᑖᓂᒃ ᖃᐅᔨᓇᓯᐊᓂᒃᑯᑦ ᐊᒻᒪ ᐱᖁᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᓇᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᖅ (TEK) ᖃᐅᔨᓇᓯᐊᖅᑕᐅᒃᑲᓐᓂᐊᕐᒪᑕ ᐊᑯᓚᐃᑐᒥᑦ ᓯᕗᓂᒃᓴᒥ, ᓄᑖᑦ ᖃᐅᔨᒪᔪᑎᒃᓴᐃᑦ ᐊᑐᐃᓐᓇᐅᖃᑦᑕᕐᓂᐊᕐᒪᑕ ᓈᓴᐅᓯᓇᔪᑕᐅᒃᑲᓐᓂᖃᑦᑕᕐᓂᐊᕐᒪᑕ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᓈᑐᐊᕐᒐᖓᑕ 15 ᐊᕐᕌᒍᑦ. ᑕᐃᒪᓕ ᓄᑖᓂᑦ ᖃᐅᔨᓇᓯᐊᕐᓂᕐᒧᑦ ᖃᐅᔨᒪᔪᑎᒃᓴᓂᑦ ᐊᑐᐃᓐᓇᐅᔪᖃᖏᑉᐸᑦ ᓈᓴᐅᓯᓇᔪᑕᐅᒃᑲᓐᓂᖃᑦᑕᕐᓂᐊᖅᑐᓂᑦ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ

ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) 15-ᓂ ᐊᕐᕌᒍᓂ, ᐅᓄᕐᓂᖏᑦ ᑖᒃᑯᐊ ᑲᑎᓪᓗᒋᑦ ᐱᔭᐅᔪᓐᓇᖅᑎᑕᐅᔪᑦ ᐊᖑᓇᓱᐊᖅᑕᐅᔪᓐᓇᕐᓗᑎᒃ (TAH) ᓴᕿᑎᑕᐅᔪᑦ ᑕᑉᐸᒃᑯᓇᓂ ᐊᒃᓲᓂᖅᓴᖅ ᐱᕕᒡᔪᐊᕈᑕᐅᔪᓐᓇᖅᑐᓂᒃ ᐅᖓᑎᓗᐊᖓᓄᑦ-ᓇᓄᒃᑕᐅᖃᑦᑕᓗᐊᖅᐸᑕ ᐊᑐᖅᑎᓪᓗᒋᑦ ᐊᑐᕐᓂᐊᖅᑐᑦ ᖃᑦᓯᑲᓪᓚᐃᑦ ᖁᓕᐅᔪᖅᑐᑦ ᐊᕐᕌᒍᐃᑦ.

ᑕᐅᕙᓂᑦᑎᐊᖅ ᐱᓕᕆᓂᐅᓂᐊᖅᑐᒥ, ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᓇᓗᓇᐃᖅᓯᓚᐅᕐᒪᑕ ᐅᓄᖏᓐᓂᖅᓴᓂᒃ ᐃᓱᒫᓗᑕᐅᔪᓂᑦ ᑕᐃᒃᑯᓄᖓ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖕᒪᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᐅᖃᑕᐅᔪᓂᑦ ᑕᒪᒃᑯᓇᖓᓪᓕ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖕᒪᑕ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ. ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒍ ᐱᑕᖃᕐᒪᑦ ᖃᑦᓯᑲᓪᓚᖕᓂᑦ ᐱᒻᒪᕆᐊᓗᖕᓂᑦ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᔭᕆᐊᓕᖕᓂᑦ. ᐊᔾᔨᓪᓗᐊᐸᓗᒋᓗᓂᒋᑦ ᓴᓐᓂᕈᑎᐅᑉ ᑕᕆᐅᖕᒪᑕ (BB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑕ, ᒫᓐᓇ ᖃᑦᓲᓐᓇᐅᓂᖅᓴᐅᖕᒪᑕ ᐊᖑᑏᑦ ᐊᕐᓇᓪᓗᖕᓂᑦ ᑕᒫᓂ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖕᒥ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂ, ᐊᒻᒪᓗ ᐊᖑᑎᑦ ᐆᒪᑯᑖᒍᓐᓇᖏᓐᓂᖅᓴᐅᖕᒪᑕ ᐊᕐᓇᓪᓗᖕᓂᑦ ᐆᒪᑯᑖᒡᓂᕆᓲᖏᓐᓂᑦ. ᐱᖃᑖᓂ, ᑕᕆᐅᑉ-ᓯᑯᖓ ᓄᖑᓴᕋᐃᓕᕙᓪᓕᐊᒐᓗᐊᖅᑎᓪᓗᒍ ᐱᐅᓈᖅᑐᓂᒃ ᐊᒃᑐᐃᓂᖃᕐᒪᑦ ᑖᒃᑯᓂᖓ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖕᒪᓃᑦᑐᓂᑦ (KB) ᓇᓄᕐᓂᑦ ᒫᓐᓇᕋᑖᖑᓚᐅᖅᑐᖅ ᖁᓕᐅᔪᖅᑐᑦ ᐊᕐᕌᖑᓂ, ᑕᒪᒃᑯᐊ ᐊᒃᑐᐃᓂᖏᑦ ᓂᕆᐅᒋᔭᐅᕚᑦ ᐱᐅᓈᕈᓐᓂᕐᓂᐊᕋᓱᒋᔭᐅᓪᓗᑎᒃ ᐊᑯᓂ ᑕᐅᕗᖓᑐᖃᒃ ᑕᕆᐅᑉ-ᓯᑯᖓ ᓄᖑᓴᕋᐃᓕᕙᓪᓕᐊᑐᐃᓐᓇᖅᑎᓪᓗᒍ. ᐱᖓᔪᖓᓐᓂ, ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖕᒪᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᑦ ᑲᔪᓯᑐᐃᓐᓇᕐᓂᐊᕐᒪᑕ ᐊᔪᕐᓇᐸᓗᐃᓐᓇᕐᓗᑎᒃ ᖃᐅᔨᓇᓱᐊᓇᐊᖏᑕ ᐱᔾᔪᑎᒋᓪᓗᒍ ᐅᓄᖏᑦᑐᑯᓘᖕᒪᑕ ᐊᒻᒪᓗ ᐅᐸᒐᕐᓂᖏᑦᑐᒦᑦᓗᑎᒃ ᐊᐅᓚᑦᑎᓂᒧᑦ ᐱᓕᕆᕕᐅᖃᑦᑕᓱᒥ. ᖃᐅᔨᓇᓱᐊᖅᑎᑦ ᐃᖃᓇᐃᔭᕐᓗᑎᒃ ᐱᓕᕆᖃᑎᒋᑦ (SWG) ᐃᒪᓐᓇᐃᓗᖁᔨᓕᓚᐅᕐᒪᑦ ᑲᑐᔾᔭᐅᓗᑎᒃ ᐱᓕᕆᖁᔨᔪᑎᑦ (JC) ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᖃᕐᓗᒋᑦ ᐱᓕᕆᒋᐊᖃᕐᓂᕈᑎᒃᓴᐃᑦ, ᐱᓕᕆᐊᖑᓴᕋᐃᓐᓂᖅᓴᐅᕗᓪᓗᑎᒃ ᖃᐅᔨᒪᔨᐊᖅᓯᓂᕐᒧᑦ ᐸᕐᓇᒍᑎᑦ ᑕᐃᑯᓄᖓ ᐱᕐᓕᕐᕕᒡᔪᐊᑉ ᑕᕆᐅᖕᒪᑕ (KB) ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓄᑦ, ᓱᕐᓗ ᒪᑯᓇᓂ ᓈᒪᒃᓯᑐᐊᕋᖕᒐᑦ ᖃᖓᑕᓱᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᕙᓪᓗᑎᒃ ᐊᑐᖅᑲᓯᐅᑎᑐᒥᒃ ᐱᖅᑯᓯᒃᑯᑦ ᓄᓇᒥᐅᑕᓕᕆᓂᕐᒥᑦ ᖃᐅᔨᒪᓂᑦ (TEK), ᑕᒪᓐᓇᓗ ᐃᑲᔪᕈᓐᓇᑐᐃᓐᓇᕆᐊᖃᕐᒪᑦ ᐊᐅᓚᑦᑎᔨᓂᑦ ᖃᐅᔨᔪᑕᐅᔪᓐᓇᕐᓗᓂ ᐅᖓᑎᓗᐊᖓᓄᑦ-ᓇᓄᒃᑕᐅᖃᑦᑕᓕᕐᓂᖅᐸᑕ ᐊᒻᒪᓗ ᐊᓯᖏᓐᓄᑦ ᐊᑲᐅᙱᓕᐅᕈᑎᓄᑦ ᓯᕗᓂᖓᒍᑦ ᐊᒃᓯᓗᐊᕌᓗᒃ ᐱᓂᕐᓗᒃᓯᓚᐅᖏᓐᓂᖏᓐᓂᑦ ᐅᓄᕐᓂᒃᑲᓐᓂᖏᓐᓂᑦ.

Harvest Assessment for the Baffin Bay and Kane Basin Polar Bear Subpopulations

Final Report to the Canada-Greenland Joint Commission on Polar Bear

Eric V. Regehr[1], Stephen Atkinson[2,*], Erik W. Born[3,*], Kristin L. Laidre[1,2,*], Nicholas J. Lunn[4,*], and Øystein Wiig[5,*]

[1] Polar Science Center, Applied Physics Laboratory, University of Washington, Seattle, Washington 98105, USA

[2] Box 19, Group 7, RR#2, Dugald, Manitoba, R0E 0K0, Canada

[3] Greenland Institute of Natural Resources, 3900 Nuuk, Greenland

[4] Environment and Climate Change Canada, University of Alberta, Edmonton, Alberta, T6G 2E9 Canada

[5] Natural History Museum, University of Oslo, NO-0318 Oslo, Norway

[*] Scientific Working Group of the Canada-Greenland Joint Commission on Polar Bear

31 July 2017

Please cite as: Regehr, E.V., S. Atkinson, E.W. Born, K.L. Laidre, N.J. Lunn, and Ø. Wiig. 2017. Harvest Assessment for the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear, 31 July 2017: iii + 107 pp.

Table of Contents

Executive Summary

Background

The Baffin Bay (BB) and Kane Basin (KB) polar bear subpopulations are jointly managed by Canada (Nunavut) and Greenland. The Canada-Greenland Joint Commission on Polar Bear (JC) facilitates coordination between the two jurisdictions. In 2011, due to concerns about potentially unsustainable harvest and the demographic effects of observed, long-term changes in sea-ice habitat, the JC tasked its Scientific Working Group (SWG) with reassessing the status of the BB and KB subpopulations. Both subpopulation were surveyed in the 1990s. However, by 2011 these earlier surveys were considered too old to provide a reliable basis for assessment or future harvest management. Consequently, the SWG undertook a research program using genetic capture-recapture (BB and KB), radiotelemetry (BB and KB), and aerial surveys (KB) from 2011–2014 to obtain updated information on subpopulation size, status, delineation, habitat quality, and habitat use (SWG 2016).

Results from these recent studies suggest that, over the last few decades, the BB subpopulation has undergone significant changes in range, movements, habitat use, body condition, and reproductive performance concurrent with a decline in sea-ice extent, duration, and quality. Baffin Bay is a relatively abundant subpopulation with an estimated 2,826 bears in 2011–2013 (95% CI = 2,059–3,593). However, due to differences in capture-recapture (CR) sampling designs, results from the surveys in the 1990s and 2010's cannot be directly compared to assess trends in the size of the subpopulation. For KB, recent studies suggest that the subpopulation is transitioning from a multiyear sea-ice system towards a sea-ice regimen characteristic of the seasonal ice ecoregion, where sea ice melts almost entirely during the summer. The KB subpopulation has responded to changing sea ice by expanding its range since the 1990s, especially during summer. Larger and more variable home ranges, and the use of lower sea-ice concentrations in summer and fall, have also been observed (SWG 2016). The current abundance of the KB subpopulation was estimated at 357 bears in 2012–2014 (95% CI = 221–493), with the available evidence suggesting this subpopulation has been stable or increasing since the 1990s.

Harvest assessment approach

Following a review of research findings for BB and KB (SWG 2016), the JC provided the SWG with three potential alternatives for subpopulation Management Objectives: (1) maintain a relatively stable subpopulation size; (2) maintain a subpopulation size that achieves maximum sustainable yield, with respect to a potentially changing environmental carrying capacity; and (3) reduce subpopulation size by approximately 30% in 10–15 years. Management Objectives 1 and 2 were considered potential objectives for both BB and KB, while an exploration of Management Objective 3 was associated with concerns about human-bear conflicts, and was requested for BB only. The JC requested that the SWG use the best-available information to provide advice on harvest management strategies, including levels of Total Allowable Harvest, under which these objectives could be achieved. The JC provided two levels of risk tolerance (“low” and “medium”) for not meeting each objective.

In this report, we use the ecological and demographic data from SWG (2016) to evaluate a suite of potential harvest strategies for the BB and KB subpopulations. We evaluated all strategies against the specific management objectives and risk tolerances provided by the JC, and for each strategy we also recorded other metrics of biological or management interest (e.g., the probability of severely depleting adult male bears). We interpreted the JC’s request for advice on “low” and “medium” risk tolerance to mean a 90% and 70% chance of successfully meeting a management objective, respectively (alternatively, a 10% and 30% chance of failing to meet a management objective).

We performed a quantitative risk assessment using a demographic model based on the life history of polar bears, which can include the effects of environmental change (Regehr et al. 2017). The potential effects of future changes in sea-ice conditions on subpopulation size and status, and the resulting implications for harvest management, were incorporated in the analyses by using projected trends in carrying capacity (*K*, the capacity of the environment to support a given number of polar bears). In addition to this direct environmental effect, the model included a mechanistic submodel of Allee effects in the mating system, which limited reproduction under conditions of low subpopulation density or imbalanced numbers of adult females and males. We considered several different scenarios of the vital rates (e.g., rates of reproduction and survival) for each subpopulation due to uncertainty and potential bias in some demographic parameters from CR studies for both BB and KB.

For each combination of management objective and vital rates scenario, we evaluated multiple harvest strategies. Harvest strategies were defined in terms of the key elements that can be identified and adaptively managed by authorizing agencies, including harvest rate and harvest level (measured in number of independent bears [i.e., not including cubs-of-the-year or yearlings] removed annually), the sex and age composition of the harvest, the management interval, and the quality (i.e., level of statistical precision) of available survey data. We evaluated 10, 15, and 20-year management intervals, defined as the number of years between successive changes to the harvest based on new data from subpopulation surveys. The management interval often used as an objective in Canada is 15 years. For BB and KB, the interval between the two most recent surveys was 18 years (1993 to 2011), although some management adjustments occurred during this period. Thus, a management interval of 15–20 years approximates current practices for these subpopulations. A management interval of 10 years was used to illustrate the effects of more frequent subpopulation surveys and management changes.

Strategies using three harvest sex ratios (*SR*) were examined; *SR* = 1 (i.e., a 1:1 male-to-female ratio) reflecting conditions where harvest is not selective for either males or females; *SR* = 2 reflecting the target ratio for sex-selective harvest currently implemented in Canada, and reflecting the reported sex ratio of the combined Canada-Greenland harvest 1998–2013; and, for each subpopulation, a '*status quo*' sex ratio of the combined Canada-Greenland harvest 1998–2013, based on results from recent genetic sampling (2011–2013) indicating that sex was incorrectly reported for a substantial number of harvested bears.

All population projections assumed a state-dependent (i.e., dependent on current conditions) management approach, under which harvest levels did not remain constant in the future, but rather were updated according to the management interval. This means that the harvest strategies are tied directly to the timeline for reassessing subpopulation abundance and vital rates.

Baffin Bay

For the BB subpopulation, projections included a proxy for changes in *K* estimated from the number of ice-covered days per year in the BB region, which decline by approximately 5.5% per decade when projected forward in time. Use of a projected, declining trend in *K* is consistent with evidence for range contractions, and changes in nutritional condition and reproductive rates of BB polar bears associated with sea-ice loss (SWG 2016). In the demographic model, we assumed that the current rates of survival and reproduction for BB polar bears reflect a subpopulation that is near its maximum net productivity level (MNPL, the subpopulation size that results in the greatest net annual increment in numbers resulting from reproduction minus losses due to natural mortality). In other words, we assumed that subpopulation size has been held below *K* due to harvest, and that BB bears are not currently experiencing strong density-dependent suppression of survival or reproduction. Our projections did not include potential density-independent effects of sea-ice loss, which could reduce subpopulation resilience and capacity to support harvest in the future. If such changes occur rapidly compared to the schedule for future subpopulation surveys and harvest changes (as determined by the management interval), the risk of negative population outcomes would be higher than estimated from our projections.

We evaluated three scenarios of the vital rates for the BB subpopulation. Each scenario used the same rates of reproduction (litter production rate and cub-of-the-year litter size) as calculated from the recent genetic CR data (2011–2013) but differed in the rates of survival. Scenario 1 used estimates of unharvested survival (*S**) calculated from CR data for the period 2011–2013. Scenario 2 used estimates of *S** calculated from CR data for the period 1998–2010. Scenario 3 used estimates of *S** representing the “average” rates seen amongst polar bear subpopulations; this provided a benchmark for comparison with other subpopulations. Of the three scenarios, we considered Scenario 2 a more likely representation of the status of the BB subpopulation, because it used data specific to BB and could reproduce plausible trends in subpopulation abundance and sex ratio that were consistent with the available scientific information and Traditional Ecological Knowledge. Additionally, unlike Scenario 1, estimates of *S** for the period 1998–2010 were less susceptible to terminal bias (i.e., bias at the end of a time series of estimates, a common problem in CR studies).

Scenario 2 resulted in an unharvested asymptotic population growth rate $\lambda = 1.08$ (SE = 0.02) annually (i.e., 8% per year). For harvest strategies with *SR* = 1.25 (i.e., a 1.25:1 male-to-female sex ratio in the harvest, the estimated *status quo* for BB based on genetic data), an age composition based on historic harvest data, and a 15-year management interval, present-day harvest rates of up to 4.3% and 5.7% were consistent with Management Objective 2 under “low” and “medium” risk tolerances, respectively. We focused on Management Objective 2 for the BB subpopulation because this objective is more relevant to sustainable harvest when *K* is declining. Applying these harvest rates to the current subpopulation size of 2,826 would result in present-day harvest levels of up to approximately 120 and 160 bears per year, depending on risk tolerance. Under this harvest strategy, the present-day harvest level would be maintained for a 15-year period, at which point a new subpopulation survey should have been completed and the harvest should have been re-calculated. Over the next 35 years (approximately three polar bear generations), the harvest level would be expected to decline due to declining *K* with sea-ice loss, and possibly due to other demographic effects.

The harvest strategies listed above (i.e., present-day harvest rates of up to 4.3% and 5.7%) should be interpreted with caution for several reasons. First, strategies at the upper end of this range were associated with up to a 12% probability of severely depleting adult male bears, and up to a 4% probability of extirpation (i.e., reduction to a very small and non-viable subpopulation size) after 35 years. Second, recent subpopulation studies provided evidence for ecological effects of sea-ice loss that could, now or in the future, translate into negative demographic effects above and beyond the declining trend in *K* and Allee effects included in our model. Although there were several reasons to place less confidence in the relatively low estimates of *S** from 2011–2013 (Scenario 1), it is possible that these lower estimates reflected, to some extent, a reduced capacity for subpopulation growth due to sea-ice loss. If that is the case, the risks of negative population outcomes could be much higher than estimated under Scenario 2. Our analyses did not make purposefully-conservative assumptions, and therefore could have understated the future effects of sea-ice loss. Such risks could be reduced through a precautionary approach to harvest management.

Simulations for Management Objective 1 (maintaining a relatively stable subpopulation size) demonstrated that this objective likely cannot be achieved in the mid- to long-term due to projected declines in *K*, which would reduce subpopulation size regardless of harvest level. Simulations also suggested that Management Objective 3 for the BB subpopulation (reduction of 30% in 10–15 years) is probably not feasible at the level of risk tolerance stated by the JC. The largest-possible subpopulation reduction that remained within risk tolerance (with some caveats—see main text) was approximately 25% over 15 years. This required a 1:1 male-to-female sex ratio in the harvest, a 5-year management interval, and improved precision in the vital rates estimated from future subpopulation surveys. Under this harvest strategy, a present-day harvest rate of 8.7% (approximately 245 bears per year) would be applied for a 5-year period, at which point a new subpopulation survey would have been completed and the harvest would be re-calculated. Over a 15-year period, the harvest level would need to decline rapidly as subpopulation size declined. Results suggested that attempting a managed subpopulation reduction without a near-optimal, state-dependent approach—for example, applying a fixed-level harvest of 245 bears per year without new subpopulation surveys—would be associated with high probabilities of severe male depletion and extirpation after 15 years.

Kane Basin

For the KB subpopulation, projections included interannual variation in *K* but no declining trend, reflecting evidence that decreasing sea ice in the multiyear-ice region of KB may have positive ecological effects in the near term (e.g., increased marine productivity as the system transitions to annual sea-ice dynamics; SWG 2016). Similar to BB, we assumed that the estimated rates of survival and reproduction for KB polar bears reflect a subpopulation that is currently functioning near MNPL.

We evaluated two scenarios of the vital rates that differed in terms of estimated survival rates for young bears. Scenario 1 used time-constant estimates of *S** calculated from CR data for the period 1992–2014; and Scenario 2 used similar estimates, but with mean values of *S** for bears less than or equal to 2 years of age modified, to reproduce the estimated increase in subpopulation abundance from 224 bears in the 1990s to 357 bears in the 2010s (SWG 2016). We considered Scenario 2 a more likely representation of the status of the KB subpopulation, because under Scenario 1 the unmodified estimates of *S** for bears age 2 years or less were low (range across age classes of 0.45–0.73) compared to other subpopulations of similar productivity, and had high statistical uncertainty due to sparse data (e.g., less than 4 cubs-of-the-year were sampled per year 2012–2014). Additionally, use of survival rates under Scenario 1 suggested a poor demographic status for the KB subpopulation, yielding an unharvested population growth rate of 1% per year ($\lambda = 1.01$ [SE = 0.04]). This low rate is inconsistent with multiple lines of evidence for the KB subpopulation, including estimated increases in subpopulation size since the 1990s (SWG 2016), the likely positive trends in marine productivity in the region, recent information on nutritional condition and reproduction, and available Traditional Ecological Knowledge.

Scenario 2 for the KB subpopulation resulted in an unharvested population growth rate of 5% per year (λ =1.05 [SE = 0.06]). For harvest strategies with a *SR* = 0.94 (the estimated *status quo*, based on genetic data), an age composition based on historic harvest data, and a 15-year management interval, present-day harvest rates up to 1.7% and 1.1% were consistent with Management Objectives 1 and 2, respectively, at the "medium" level of risk tolerance. At the "low" level of risk tolerance, Management Objectives 1 and 2 could not be met in the absence of harvest, due to variability in subpopulation trajectories resulting from high uncertainty in the vital rates. Applying harvest rates of 1.1% to 1.7% to the current subpopulation size of 357 would result in a present-day harvest levels of up to approximately 4–6 bears per year. Under this harvest strategy, the present-day harvest level would be maintained for a 15-year period, at which point a new subpopulation survey would be completed and the harvest would be re-calculated. Over the next 35 years, the harvest level would be expected to remain stable or increase due to stable or potentially increasing *K*. Harvest strategies at the upper end of this range were associated with up to a 17% probability of severely depleting adult male bears, and up to a 3% increased probability of extirpation compared to projections with no harvest, after 35 years.

Under Scenario 2, harvest rates that met management objectives for the KB subpopulation were lower than the observed harvest rate for the period 1998–2014, during which subpopulation size likely increased (SWG 2016). This inconsistency was due primarily to high statistical uncertainty in estimates of *S** for bears age 2 years or less, a consequence of small sample sizes and relatively short study periods of research in KB. If alternative assumptions were made for uncertainty in estimates of *S** for bears age 2 years or less (e.g., if the precision of these estimates was increased to match the precision of survival estimates for older bears), present-day harvest rates up to 2.2% to 2.8% (8–10 bears per year) were consistent with Management Objective 1 at a "medium" risk tolerance, when following a state-dependent approach with a 15-year management interval.

Considering all available ecological and demographic data for the KB subpopulation, present-day harvest rates up to approximately 2.8% (10 bears per year) seem unlikely to cause negative population outcomes under a state-dependent approach with effective monitoring. It is possible that the logistical challenges of studying the KB subpopulation may lead to continued difficulty in obtaining accurate and precise estimates of vital rates, despite increased survey efforts. We therefore suggest developing a suite of ecological and demographic indicators to monitor subpopulation status, including accurate information on the level and composition of the harvest, marine productivity, habitat availability, reproductive rates, and estimates or indices of subpopulation size (via aerial survey or CR).

Monitoring requirements and further research

The results in this report are intended to help inform and guide subsequent decisions of the JC with respect to determining appropriate levels of harvest for these two shared subpopulations of polar bears. Both BB and KB are experiencing long-term trends in the extent, duration, and quality of sea-ice habitat. Our analyses identify harvest strategies that are designed to maintain subpopulation size near MNPL with respect to a changing *K*, and to limit negative effects of harvest on the probability of subpopulation persistence. All of the harvest strategies presented in this report require the existence of a coupled research-management system under which both the sustainable harvest rate and the harvest level are adjusted periodically, based on new scientific information from subpopulation surveys and other sources. For both BB and KB, our analyses demonstrate that shorter management intervals and more precise data can substantially reduce the risk of negative population outcomes associated with a given harvest strategy. A state-dependent management approach is an effective means of reducing the risk of overexploitation while maintaining opportunities for use. This is especially important if sea-ice loss is currently having ecological effects on polar bears that may signal negative demographic effects in the future (BB); or if a less-conservative harvest strategy is selected when the currently available estimates of vital rates have high uncertainty and appear inconsistent with other lines of evidence (KB). Harvest strategies that appear sustainable over the next three polar bear generations under a state-dependent approach could lead to subpopulation depletion or extirpation under a fixed-level approach that removes the same number of bears annually without reassessment.

In addition to regular, periodic surveys to estimate subpopulation size and vital rates, we recommend more frequent but less intensive monitoring of sea-ice habitat, movement and habitat use, nutritional condition, and reproductive indices based on research and harvest data, and Traditional Ecological Knowledge. Continuous genetic monitoring of the harvest to detect recoveries of genetically marked animals, and improving the accuracy of harvest reporting, are also needed. Systematic analysis of all harvest data, especially for the BB subpopulation, could provide complimentary estimates of harvest rate and other demographic parameters. During the intervening years between scheduled subpopulation surveys, these monitoring programs may provide a mechanism to detect sudden shifts in environmental conditions or subpopulation status that might necessitate a change in harvest strategy (e.g., a shortening of the management interval). Also, these monitoring programs will provide information essential for designing and implementing periodic subpopulation assessments. We suggest that future subpopulation assessment protocols consider the use of integrated population models, which can analyze data from multiple sources (e.g., different types of research, harvest, and Traditional Ecological Knowledge) in a unified framework, potentially leading to improved assessments of overall subpopulation status.

For the BB and KB subpopulations, the harvest sex ratios in recent decades, based on genetic sex determination, indicate weaker selection for males compared to the reported sex ratio and the management goal of a 2:1 male-to-female ratio. In most of our simulations, a harvest strategy with $SR = 2$ (instead of the lower *status quo* values of SR) did not result in higher harvest rates that met management objectives. This is because, for both subpopulations, estimates of S^* were lower for males than females, and females currently comprise approximately 70% of independent bears. These factors, if combined with a strongly male-selective harvest, often led to the severe depletion of adult males in our simulations, which had negative effects on reproduction and increased the probability of extirpation due to Allee effects in the mating system. These findings do not provide evidence against the conservation value of sex-selective harvest for polar bears. Rather, they indicate that depletion of males may be an emerging conservation concern for the BB and KB subpopulations. Given the current regulation of harvest in Canada (Nunavut) based on a sex 2:1 male-to-female sex ratio, it is important to further investigate this issue and particularly the lower estimates of survival for male bears. Our finding of skewed sex ratios in both subpopulations, despite an overall harvest that may not be strongly selective, suggests that these lower survival rates have a biological basis. Concurrent monitoring of the sex ratio in the harvest, the sex and age composition of the subpopulation, and the litter production rate, are necessary to determine the extent to which reduction of male bears could negatively affect the productivity of the BB and KB subpopulations.

Literature Cited in Executive Summary

Regehr, E. V., R. R. Wilson, K. D. Rode, M. C. Runge, and H. L. Stern. 2017. Harvesting wildlife affected by climate change: a modeling and management approach for polar bears. Journal of Applied Ecology doi:10.1111/1365-2664.12864.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear. 31 July 2016: x + 636 pp.

Introduction

Background on the Joint Commission, and the Baffin Bay and Kane Basin subpopulation reassessments

The Canada-Greenland Joint Commission on Polar Bear (JC) was established with the signing of a "*Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*" on 30 October 2009 (Anon. 2009). The primary objectives of this Memorandum of Understanding are to: "(1) to manage polar bear within the Kane Basin and Baffin Bay management units in order to ensure their conservation and sustainable management into the future, and, (2) establish an effective system of management which will include adhering to the principles of conservation". The JC subsequently established a Scientific Working Group (SWG) to provide scientific advice and recommendations with respect to the conservation and management of the Baffin Bay (BB) and Kane Basin (KB) polar bear subpopulations. In 2010, the JC tasked the SWG with using the best-available scientific information to:

(1) Propose Total Allowable Harvest (TAH) levels for the Kane Basin and Baffin Bay subpopulations.

(2) Provide science advice to the Joint Commission for monitoring the effects of habitat changes on polar bears.

The SWG reviewed the available scientific information and reported (SWG 2010) that for both subpopulations the most recent status updates indicated that 100% of population viability analysis (PVA) simulations, using current harvest levels, resulted in subpopulation declines after 10 years (PBSG 2010). The SWG also noted that PVA simulations are typically run 10–15 years beyond the point in time that abundance and vital rates were estimated. Given that the most recent estimates of demographic parameters for the BB and KB subpopulations were from the mid- to late 1990s (Taylor et al. 2005, 2008a; PBSG 2010), the SWG concluded that the available information was outdated and it was unknown whether demographic parameters had changed over time. These factors, in combination with the large-scale environmental changes in BB during recent decades, led the SWG to recommend that a high priority be given to developing new estimates of subpopulation abundance, subpopulation delineation, and vital rates (SWG 2010).

As a result, the JC tasked the SWG with evaluating various methods for assessing the number of polar bears in BB and KB (JC 2010). The SWG considered the pros and cons of physical CR, genetic CR, and aerial surveys; and concluded that physical CR was the preferred method to obtain robust ecological and demographic data (SWG 2011). However, due to the lack of support for physical CR among Inuit in Nunavut, and concerns that variability in sea-ice conditions can make it difficult to obtain accurate abundance estimates from aerial surveys, the JC recommended development of a 3-year research program based on genetic CR methods using biopsy darting. Following this recommendation, field research programs were conducted 2011–2014 as part of comprehensive reassessments of the BB and KB polar bear subpopulations (SWG 2016).

Estimation of sustainable harvest for polar bears

Historically, polar bear management was based on the assumption that sea-ice habitat was relatively stable over the long term and that, once subpopulation size (*N*) had been estimated, conservation could be achieved through harvest management (SWG 2011). Sustainable harvest, therefore, would depend on estimates of abundance and vital rates (e.g., probabilities of survival and reproduction), the harvest level, and the sex and age composition of the harvest. Early modeling suggested that sex-selective harvest at a rate of 4.5% of total population size, was sustainable for polar bears under optimal conditions (Taylor et al. 1987a, b). Recently, Regehr et al. (2015, 2017) also found that a 4.5% harvest rate, with a 2:1 male-to-female sex ratio in the harvest, was reasonable under many biological and management conditions, provided that population surveys were conducted periodically and harvest levels were adjusted when necessary. Regehr et al. (2017) also noted that sustainable harvest rates could be lower or higher than 4.5% under some conditions. This finding is consistent with recent studies that have provided a better understanding of how vital rates vary across subpopulations and change over time (e.g., Amstrup et al. 2001; Taylor et al. 2002, 2005, 2006; PBSG 2006, 2010; Regehr et al. 2007, 2010).

In recent decades, management and conservation of polar bears, particularly in Canada, have been informed by predictive modeling that incorporates subpopulation-specific vital rates.

The development of RISKMAN (Taylor et al. 2001), a stochastic population model, made it possible to perform detailed harvest assessments for multiple subpopulations, providing managers with a better understanding of the risk associated with different harvest strategies (Taylor et al. 2002, 2005, 2006, 2008a). However, RISKMAN was primarily intended to inform near-term management under stable conditions. It did not include a detailed model of density dependence, allow for future changes in environmental conditions or demographic parameters, or provide a way to directly assess how the frequency and intensity of subpopulation surveys can affect the risk of different management actions.

At present, the primary threat to polar bears throughout their range is the reduction in sea-ice habitat area, duration, and quality as a consequence of climate change (Derocher et al. 2004; Laidre et al. 2008, 2015; Wiig et al. 2015; Atwood et al. 2016). In 2009, the Polar Bear Range States (2015) agreed that the impacts of climate change constitute the most important threat to polar bear conservation and recommended that best management practices should “Consider the cumulative effects of climate change and human activities on polar bear subpopulations when making management decisions using tools such as predictive modeling”. Hence, in many situations sound harvest management will no longer rely solely on an estimate of abundance and a fixed annual harvest rate (e.g., 4.5%), or on predictive modeling that assumes stable conditions. The current demographic status of the world’s 19 polar bear subpopulations is variable (PBSG 2017) due to ecological variation, different rates of habitat change, and the influence of anthropogenic stressors (Vongraven and Peacock 2011; Atwood et al. 2016). Furthermore, although national and international agreements provide common standards for polar bear conservation, near-term management goals can vary across jurisdictions as a function of environmental, social, and other factors (Polar Bear Range States 2015). In light of this variability and the primary threat of habitat loss due to climate change, there is a need for improved risk assessment tools that can incorporate a broad range of environmental and direct human-caused factors to address specific management goals.

Harvest assessment for the Baffin Bay and Kane Basin subpopulations

Recent reassessments of the BB and KB polar bear subpopulations provided the ecological and demographic data necessary for harvest risk assessments (SWG 2016). However, the SWG was unable to provide harvest options as part of its final report because the JC had not provided specific guidance on (i) management objectives for each subpopulation, (ii) the expected frequency and intensity of future monitoring, and (iii) risk tolerance with respect to the effects of human-caused removals. Subsequently, the JC requested that the SWG propose TAH levels for both the BB and KB subpopulations based on the abundance estimates in SWG (2016), historical harvest levels, an expected 7–15 year frequency between subpopulation surveys, and that TAH be evaluated relative to the following management objectives (JC 2016):

1a) Maintaining a stable subpopulation at the current subpopulation estimate, with a low tolerance for the risk of declines below 90% of this level.

1b) Maintaining a stable subpopulation at the current subpopulation estimate, with a medium tolerance for the risk of declines below 90% of this level.

2a) A TAH that would ensure a maximum sustainable yield, with a low level of risk tolerance for the subpopulation declining below this level.

2b) A TAH that would ensure a maximum sustainable yield, with a medium level of risk tolerance for the subpopulation declining below this level.

Lacking further guidance, the SWG interpreted “low” and “medium” tolerance for the risk of a subpopulation decline below a specified level, to mean requiring a 90% or 70% probability of maintaining a subpopulation size above the specified level, respectively.

In addition to the management objectives above, the JC subsequently requested that the SWG explore possible methods to achieve a managed reduction of the BB subpopulation, from the current estimate of 2,826 animals (SWG 2016) to approximately 2,000 animals, over a 10–15 year period (JC 2017). Possible reasons for considering a managed reduction could be to reduce human-bear conflicts in circumstances where densities of polar bears have increased or nutritionally-stressed bears are increasingly coming in close proximity to humans, which may occur as sea-ice loss continues (Wilder et al. 2017).

In this report, we used recent estimates of abundance and vital rates (SWG 2016) in a matrix-based demographic model (adapted from Regehr et al. 2015, 2017) to evaluate TAH for the BB and KB polar bear subpopulations, relative to management objectives provided by the JC.

The demographic model is based on the life history of polar bears, and provides several advantages compared to other predictive modeling tools, including (i) an ability to incorporate the effects of a changing habitat (e.g., through a variable or declining *K*), (ii) a species-specific model of density dependence, which is important when evaluating the combined effects of habitat change and human-caused removals; (iii) an integrated model of Allee effects in the mating system, based on the work of Molnár et al. (2008, 2014); and (iv) a direct link between research and management actions, so that harvest strategies can be evaluated in light of the frequency and intensity of future subpopulation surveys.

For each subpopulation, we used data from SWG (2016) to develop alternative scenarios of the vital rates, which either represented plausible conditions for the current status of the subpopulation or provided a useful benchmark (e.g., for comparison with a subpopulation of "average" productivity). We considered multiple scenarios because estimates of demographic parameters from CR studies for polar bears can include uncertainty that is not reflected in the statistical distributions of the parameters, as well as multiple types of bias (Regehr et al. 2009; Chapter 3 in SWG 2016). To obtain robust results from predictive modeling it is important that demographic parameters be evaluated for biological realism and within the framework of other available information (e.g., Skalski et al. 2012).

Our analyses focused on harvest management strategies that were defined in terms of the harvest rate (percentage of the total subpopulation size removed per year), harvest level (measured in number of independent bears removed per year), sex and age composition of removed animals, management interval (number of years between successive subpopulation surveys and management changes), and the precision of demographic parameters estimated from subpopulation surveys. All harvest strategies were evaluated using a state-dependent (i.e., dependent on current conditions; Lyons et al. 2008) management approach, under which both the harvest rate and harvest level were updated periodically according to the management interval. State-dependent management has many features in common with the "adaptive management" approach recommended by the Range States (Polar Bear Range States 2015). It also has many advantages over other management approaches (e.g., a fixed-level harvest), both in terms of mitigating harvest risks and making it possible to maximize long-term yield (Regehr et al. 2017). Because our analyses incorporated multiple types of variability (e.g., statistical uncertainty and environmental variation), results are presented in a probabilistic manner that is consistent with management objectives provided by the JC.

The final results of this analysis are a series of potential harvest strategies for the BB and KB subpopulations. It is intended that these results help inform and guide subsequent decisions of the JC with respect to its determination of appropriate levels of harvest for these two shared subpopulations of polar bears.

Methods

Demographic and management model

We performed population projections using the matrix-based demographic model described in Regehr et al. (2015, 2017). The demographic model is based on the polar bear life cycle (Figure 1), with six female stages representing age and reproductive status, and four male stages representing age (Hunter et al. 2010; Regehr et al. 2010). Transitions between stages are defined by vital rates relative to a post-breeding census from the autumn or spring of year t to the autumn or spring of year $t + 1$, for the BB and KB subpopulations, respectively. Projections were referenced to independent bears (i.e., bears age ≥ 2 years that are not members of a family group) because cubs-of-the-year (C0) and yearlings (C1) are not included as individuals in the life cycle, but rather are used to define the reproductive status of an adult female (adult females with C0, stage 5; adult females with C1, stage 6). Projections were referenced to individual subpopulations and were not designed to consider immigration, emigration, or metapopulation dynamics. Unless otherwise noted, details of the projection model and its application follow from Regehr et al. (2015, 2017). A list of abbreviations, acronyms, symbols, and definitions is provided at the end of this report.

Density dependence

Including density dependence is necessary to evaluate the combined effects of habitat change and human-caused removals (Guthery and Shaw 2013). We constructed density-dependent curves of the vital rates (sample curves shown in Figure BB1) using methods and shape parameters described in Appendix S2 of Regehr et al. (2017). We assumed that available vital rates for the BB and KB subpopulations (section Vital rates) corresponded to an estimated subpopulation size at maximum net productivity level (MNPL; section Abbreviations, Acronyms etc.) under asymptotic population dynamics.

The matrix model was used to project hypothetical polar bear subpopulations forward over annual time steps starting at $t = 1$ (section Population projections). At each time step, density was determined as the sum of metabolic energetic equivalent (*mee*) values in the subpopulation, divided by carrying capacity (K; section Carrying capacity and environmental variation) expressed as energetic equivalents (Regehr et al. 2017). Under this approach, larger bears (e.g., adult males) occupied more energetic space and therefore had a greater density effect than smaller bears (e.g., subadult females). Regehr et al. (2017) found that individual variation in energetic requirements can influence population productivity because a given environment can generally support more females than males. We used *mee* values from Regehr et al. (2017) that were calculated from data on body mass and diet for the Chukchi Sea and Southern Beaufort Sea subpopulations (Table S2 in Regehr et al. 2017) because equivalent data were not available for the BB and KB subpopulations.

The demographic model incorporated a mechanistic submodel for Allee effects in the mating system, following the recommendation of Regehr et al. (2017). Molnár et al. (2008, 2014) proposed that, under some conditions, reproductive rates for polar bears may decline due to limitations in mate finding. Such declines can occur if adult males are depleted relative to adult females, which is possible under sex-selective harvest (McLoughlin et al. 2005; Taylor et al. 2008b); or if polar bear densities are low during the breeding season. Because Allee effects in the BB and KB subpopulations have not been studied directly, the submodel for Allee effects was based on equation 3 from Molnár et al. (2014) with input parameters for a “generic population”. For both the BB and KB subpopulations, we calibrated the Allee submodel by calculating the degree of mating season aggregation that would result in a litter production rate equal to the estimated value from recent subpopulation studies (section Vital rates). This ensured that reproductive rates at $t = 1$ were equivalent to observed values. In subsequent years of projections ($t = 2, 3, \ldots$), the estimate of litter production rate from the Allee submodel was standardized by dividing by its value at $t = 1$. The resulting value was constrained to the interval [0,1] and used to modify the value of litter production rate (β_4) obtained from the density-dependent curves of the vital rates. Under this approach, a subpopulation that did not experience male depletion or significant reductions in abundance, did not experience declines in reproduction due to Allee effects.

For the BB subpopulation, we used an on-ice area of 656,000 km^2 (Stern and Laidre 2016) to calculate the densities of female and male bears available to breed at each time step (stages 4 and 10, respectively), which are inputs to the Allee submodel (Molnár et al. 2014). A mating season aggregation parameter of 0.43 resulted in a litter production rate of 0.93, as estimated from 2011–2013 field data (section Results). For the KB subpopulation, we used an on-ice area of 53,000 km^2 (Stern and Laidre 2016). We set the mating season aggregation parameter to 1.0 because the Allee submodel could produce a litter production rate of 0.71, as estimated from 2012–2014 field data (section Results), without a reduction in the effective subpopulation area.

Carrying capacity and environmental variation

Modeling wildlife populations under climate change required consideration of the effects of variability and trends in the environment (Boyce et al. 2006). We derived a proxy metric to represent potential changes in K using satellite data of sea-ice extent. We calculated separate metrics for the BB and KB subpopulations, based on the number of ice-covered days per year within the management boundary for each subpopulation (Chapters 4 and 9 in SWG 2016). We used the number of ice-covered days because it integrates spatial and temporal variation in sea-ice availability in a manner that is biologically relevant to polar bears (Stern and Laidre 2016). For each subpopulation, we fit a linear model to the time series of ice-covered days from 1979–2014. We then used the fitted model to project correlated values of ice-covered days forward in time, using methods of Gelman and Hill (2007) to simulate uncertainty in the slope coefficient and residual standard errors. Finally, we standardized the metric by dividing the projected values of ice-covered days at year $t = 1, 2, \ldots k$, by the fitted value at year $t = 1$. This resulted in a dimensionless metric (κ) representing proportional changes in K. During population projections, carrying capacity at year t, calculated as $K(t) = K(t = 1) \times \kappa(t)$, operated on vital rates through the density-dependent relationships.

The number of ice-covered days in the BB management area declined over the period 1979–2014 (slope = -1.22 days/year, SE = 0.23, $P < 0.001$), from fitted values of 245 days in 1979 to 203 days in 2014. This is a decline of approximately 5.5% per decade. During harvest assessment analyses for the BB subpopulation, projected values of κ were based on the estimated slope coefficient due to evidence for the effects of sea-ice loss on subpopulation ecology (SWG 2016). Therefore all population projections for BB included a proxy for K that varied from year-to-year, and declined by approximately 23% over three polar bear generations (section Population projections).

The number of ice-covered days in the KB management area declined over the period 1979–2014 (slope = -1.24 days/year, SE = 0.41, $P < 0.01$), from fitted values of 253 days in 1979 to 210 days in 2014. This is a decline of approximately 5.3% per decade. During harvest assessment analyses for the KB subpopulation, projected values of κ were based on a slope coefficient of 0, due to evidence for potential increases in productivity of the KB subpopulation associated with a transition from a multi-year ice region to seasonal ice conditions (SWG 2016). Therefore all population projections for KB included a proxy for K that varied from year-to-year, but remained stable over three polar bear generations (section Population projections). In other words, unlike for the BB subpopulation, projections for KB did not reflect the potential effects of long-term, decreasing trends in sea-ice cover.

In addition to density-dependent variation in the vital rates resulting from variation in K, we subjectively included additional density-independent variation as 25% of total uncertainty (i.e., temporal variation plus sampling uncertainty) in estimated vital rates, following the example of Taylor et al. (2002). Density-independent variation was implemented using the correlation matrix from Regehr et al. (2010), because that analysis estimated vital rates with a multistate CR model that was based on a life cycle graph similar to Figure 1.

Harvest and simulated population assessments

During population projections, harvest was implemented annually at a calculated level. Throughout our analyses the harvest level refers to the number of independent bears removed from a subpopulation by humans (i.e., the combination of subsistence harvest, sport hunting, removals of problem bears, defense kills, etc.). The calculated harvest level was updated every several years, according to the management interval (section Abbreviations, Acronyms etc.). To account for selectivity in human-caused removals and individual variation in the reproductive value of polar bears, harvest was implemented using stage-specific harvest vulnerability vectors. For females and males separately, we estimated harvest vulnerability by comparing the age structure of the harvest to the estimated age structure of the subpopulation (section Population initialization). For the BB subpopulation, age structure of the harvest in Canada was estimated using data from 805 bears for which age had been determined from counts of cementum annuli (Christensen-Dalsgaard et al. 2010) or from individual capture histories, during the period 1998–2013. Because age determination for polar bears is referenced to the spring, whereas stage transitions in the life cycle graph were referenced to autumn for BB, we subtracted 1 from the known age of bears harvested January through August. For example, a 4-year-old male bear in the spring of calendar year *t* remained a member of stage 8 (3 years) until it transitioned to stage 9 (4 years) in the autumn of year *t*. Age structure of the Greenland harvest was determined from 212 bears taken during the period 2012–2015. We assumed this sample was representative of the Greenlandic harvest because cementum ages were not available for other years. Overall harvest vulnerability vectors were derived by averaging the Canadian and Greenlandic vectors, weighted by the total reported harvest in each country 1998–2014 (Table 8.6 in SWG 2016). The resulting harvest vulnerability vectors for females (stages 1–6) and males (stages 7–10) were [0.93, 1.17, 1.10, 1.00, 0.00, 0.00] and [1.25, 1.34, 1.01, 1.00], respectively. During population projections for KB, we used the same harvest vulnerability vectors as were estimated for BB, because harvest data for the KB subpopulation were sparse and the two subpopulations are subject to similar harvest management regimes (Chapter 8 in SWG 2016).

At the beginning of each population projection (i.e., $t = 1$), the harvest level was calculated using the mean values of the vital rates and *N* as estimated from recent subpopulation surveys (section State-dependent management approach). This ensured that starting harvest levels reflected current data for the BB and KB subpopulations. At the beginning of each subsequent management interval, the harvest level was calculated using estimates of vital rates and *N* derived from simulated population assessments. Conceptually, the simulated population assessments represent new subpopulation surveys, performed in the future, to obtain updated data that can be used for management. The simulated population assessments included sampling uncertainty, for which the level and correlation structure were based on recent CR studies (SWG 2016). In other words, each successive simulated population assessment produced demographic parameters of similar precision to the most recent genetic CR studies for the BB and KB subpopulations (SWG 2016). Exceptions were made for some simulations, which included a modified level of precision in simulated population assessments (section Population projections).

This allowed us to evaluate the effects on harvest management of future survey methods that provide more precise or less precise estimates of *N* and the vital rates (section Simulations).

State-dependent management approach

We used a state-dependent management approach to calculate harvest level (Regehr et al. 2017) as a function of *N* and the intrinsic population growth rate (*r*, which depends on the vital rates) as follows:

$$H^{female}(t) = F_O \times \tilde{r}_{MNPL}(t) \times 0.5 \times \tilde{N}(t) \qquad \text{[eqn 1]}$$

and

$$H^{male}(t) = H^{female}(t) \times SR \qquad \text{[eqn 2]}$$

where	
H^{female}	is the number of females that can be removed annually;
F_O	is a factor that directly adjusts the harvest rate to reflect management objectives and the risk tolerance of managers with respect to harvest;
$\tilde{r}_{MNPL}$	is an estimate of the intrinsic population growth rate from subpopulation studies, referenced to population density at MNPL and selected as the 50th percentile of its sampling distribution;
0.5	is a factor to calculate female removals assuming an equal sex ratio in the subpopulation, which serves to protect against excessive female removals when the male segment of a subpopulation is depleted;
$\tilde{N}$	is an estimate of *N* from subpopulation studies and selected as the 50th percentile of its sampling distribution;
H^{male}	is the number of males that can be removed annually; and
SR	is a factor that specifies the male-to-female ratio in removals.

To implement this state-dependent approach, managers must choose input values of the parameters F_O and *SR*. The parameter F_O directly influences the harvest rate: higher values lead to a higher harvest, which can eventually increase the risk of negative population outcomes (e.g., depletion). The parameter SR determines the sex ratio of the harvest.

In our analyses, values of F_O and SR remain constant for the duration of population projections, so that each harvest strategy had a consistent definition. In practice these parameters could be adjusted over time in response to changing biological or management conditions. In contrast, in our analyses the biological parameters in equations 1 and 2 (i.e., the true values of r_{MNPL} and N, as well as their estimated values $\tilde{r}_{MNPL}$ and $\widetilde{N}$) varied during population projections (e.g., N declined over time due to declining K). The notation for time (t) in equations 1 and 2 indicates that the estimated parameters $\tilde{r}_{MNPL}$ and $\widetilde{N}$ are updated periodically, as determined by the management interval. Equations 1 and 2 are written in terms of harvest level for convenience; the harvest rate (in this instance, referenced to the number of independent bears in the subpopulation) for females is the right side of equation 1 before multiplying by $\widetilde{N}$.

Management Objectives

We evaluated harvest relative to three management objectives provided by the JC (section Introduction; Table BB1).

Management Objective 1 was to maintain N above 90% of its starting value. Although we report results for this objective for both subpopulations, it was of limited value for BB because our analyses included projected trends in K that made it unlikely to meet Management Objective 1 even with no harvest (section Carrying capacity and environmental variation).

For Management Objective 2, we interpreted the language "…ensure a maximum sustainable yield" (JC 2016) as a desire to maintain a subpopulation size above MNPL relative to a potentially changing K (Management Objective 2 in Table BB1). To evaluate this objective we used a single value of MNPL corresponding to a subpopulation density (N/K) = 0.70, which is similar to the mean estimate of density at MNPL across a wide range of vital rates (Regehr et al. 2017). Using a single value of MNPL across all population projections, had the benefit of providing a consistent point of reference for management decisions.

Management Objective 3 was specific to BB, and reflected the goal of a managed reduction in total subpopulation size to 2,000 bears in 10–15 years. This corresponds to a reduction of approximately 30% relative to the mean estimate of 2,826 for the period 2012–2013

(SWG 2016). In Table BB1, we express this objective as a desire to maintain a subpopulation size above 70% of its starting value (Table BB1), noting that Joint Commission (2017) indicated that Management Objective 3 should also consider "…not achieving an abundance below the level that would produce maximum sustainable yield" as well as "…a potentially changing environmental carrying capacity". We interpret this as meaning that, to achieve Management Objective 3, the conditions for both Management Objectives 2 and 3 must be met.

In Table BB1 there are two versions of each Management Objective, reflecting "low" and "medium" levels of risk tolerance for not meeting the objective (section Introduction). We interpreted Management Objectives 1 and 2 as mid- to long-term objectives, and therefore evaluated them at the final time step $t = 36$ years, corresponding to approximately three polar bear generations in the future (section Population projections). Management Objective 3 was a short-term objective and was evaluated at $t = 15$. When reporting which harvest strategies met the management objectives, we included an additional condition requiring that the increased probability of a subpopulation being extirpated ($P_{extirpation}$) due to harvest, compared to an identical projection without harvest, not to exceed 0.05. In other words, this condition sought to ensure that harvest alone would not result in more than a 1-in-20 chance of extirpation. Applying this condition and Management Objective 2 together, over a sufficiently long time period, is consistent with the definition of "sustainable harvest" suggested by Regehr et al. (2107). Harvest strategies that met Management Objectives 1 and 2 generally were not associated with a high probability of extirpation, which meant that the condition on $P_{extirpation}$ had only a minor influence on the results. The exception was for some harvest strategies associated with Management Objective 3 (section Results).

Vital rates

We parameterized the matrix-based projection model using estimates of vital rates for the BB and KB subpopulations from recent genetic CR studies (SWG 2016). The published vital rates were adapted to the matrix-based projection model using methods described in Appendix S1 of Regehr et al. (2017).

For both the BB and KB subpopulations, litter production rate (equivalent to the parameter β_4 in Figure 1, for bears age ≥ 5 years) was not reported in SWG (2016) due to uncertainty in the age of bears that were observed from the air but not physically captured, and therefore did not provide a tooth for subsequent age determination. Also, there was uncertainty in the sex of some animals for which genetic samples were not obtained. We used simulation methods to estimate reproductive parameters for each subpopulation in a manner that accounted for this uncertainty. First, we created 10,000 bootstrap datasets by resampling, with replacement, from the 2010s field data. In each bootstrap dataset, animals with known reproductive status (i.e., adult females with dependent young), or of known age (from counts of cementum annuli) and known sex (from genetic analysis of tissue samples), were deterministically assigned to a life cycle stage. For animals of known sex but unknown age, a value for numeric age was sampled from a multinomial distribution created for bears of the same field-estimated age class. The multinomial distribution used probabilities calculated from Table B3 in SWG (2016), which compares sex and age classes as estimated from the air, with known sex and age for bears observed in BB and KB during the period 2011–2014. For animals of unknown sex, a similar procedure was used that considered both sex and age, with probabilities calculated from Table B2 in SWG (2016). For each bootstrap dataset, we estimated mean litter production rate as the number of females with C0 (stage 5) in year $t + 1$ divided by the product of adult female survival and the number of females available to breed (stage 4) in year t, taking into account annual sample sizes (Taylor et al. 1987b). The number of bears in each stage was calculated directly from the field data because CR modeling did not identify differences in recapture probabilities among female bears (Chapters 5 and 10 in SWG 2016). Standard error in the reproductive parameters was estimated as the standard deviation of point estimates from the 10,000 bootstrap datasets.

Baffin Bay

During recent subpopulation studies for BB, there was uncertainty and concern about bias in estimates of survival probability, particularly during the period 2011–2013 (Chapter 5 in SWG 2016). Therefore, we considered three alternative scenarios for the vital rates of BB polar bears, which represented a potential range of conditions (i.e., from low to high) for the current status of the subpopulation. We performed population projections using the vital rates for all three scenarios (section Simulations) and attempt to provide guidance about which results are most applicable to management.

Reproductive parameters

All three scenarios of the vital rates for the BB subpopulation used reproductive parameters estimated from CR studies 2011–2013, based on field data described in Chapter 6 of SWG (2016), and using the methods described above. To calculate litter production rate we used $S = 0.95$, the estimate of total survival for females ≥ 2 years during the period 1998–2010 (Table 5.8 in SWG 2016). The resulting litter production rate for bears age ≥ 5 years (β_4) was 0.93, which suggests relatively high mating success and cub production. Other reproductive parameters were similar to values presented in Chapter 6 of SWG (2016), with minor differences in the point estimates and variances due to the simulation methods used here (Table BB1). The parameter β_4 is the most important breeding parameter in the matrix-based projection model (Hunter et al 2007). Due to the lack of age data for most observations in BB during the period 2011–2013, we were unable to estimate litter production rate for 4-year-old bears (β_3) with an acceptable degree of accuracy, and therefore used the value 0.10 for this parameter from Taylor et al. 2005. We set the value for β_5 to 0, because females that are observed with C0 in the autumn of year t are not able to subsequently lose their cubs, re-breed, and be observed with a new litter of C0 in year $t + 1$.

Scenarios for survival

We evaluated three scenarios for survival rates of BB polar bears, representing alternative hypotheses for the current status of the subpopulation. Scenario 1 used estimates of natural (i.e., unharvested) survival (S^*) for the period 2011–2013 (page 261 in SWG 2016). Scenario 1 represents the hypothesis that estimates of S^* for 2011–2013 are accurate, in contrast to the higher estimates for BB polar bears during the period 1998–2010 and the higher mean estimates for most other subpopulations (Appendix S1 in Regehr et al. 2017). This scenario suggests a relatively poor status for the BB subpopulation (section Results).

For Scenario 2, we calculated S^* from estimates of total survival (i.e., including harvest) and harvest reporting probabilities for the period 1998–2010 (Table 5.8 in SWG 2016), using the equations for natural survival on page 257 of SWG (2016). Scenario 2 represents the hypothesis that the 1998–2010 estimates of S^* for BB are accurate, whereas the 2011–2013 estimates were negatively biased. SWG (2016) proposed that such bias may occur due to heterogeneity in recapture probability, non-random patterns of temporary emigration, of other factors. Terminal bias (i.e., bias at the end of a time series) in survival estimates commonly occurs in CR studies of long-lived, mobile animals (Peñaloza et al. 2014). Similar to our rationale in using Scenario 2, other studies have elected not to use terminal estimates $S*$ in population viability analyses due to concerns about bias (e.g., Langtimm 2009).

Scenario 3 used hypothetical survival rates representing an “average” polar bear subpopulation. To derive mean estimates of $S*$ for Scenario 3, we started with the estimates of $S*$ from Scenario 1 for female bears, and set estimates of $S*$ for male bears equal to 99% of these values. This reflects observations from other case studies that natural survival is generally similar for females and males (Table S1 in Regehr et al. 2017). Next, we increased all estimates of $S*$ in equal increments, until they produced an intrinsic population growth rate (r) of 0.05 under deterministic and asymptotic population dynamics. The value $r = 0.05$ is the mean estimate across case studies for polar bears as reviewed in Appendix S2 of Regehr et al. (2017). For Scenario 3, we used an amount of sampling uncertainty equivalent to data precision level 3 in Regehr et al. (2017). This represents the 50th percentile of estimated sampling uncertainty in case studies for polar bears, and is therefore typical of recent studies for the species (Appendix S4 in Regehr et al. 2017). Scenario 3 provides a benchmark for comparison with scenarios 1 and 2; it does not represent a data-based hypothesis for the current status of the BB subpopulation.

Estimates of $S*$ corresponding to the three scenarios of the vital rates are presented in Table BB3. For each scenario, we used the matrix-based projection model to calculate basic demographic parameters under asymptotic population dynamics (Table BB4). These parameters provide a general sense of the capacity for subpopulation growth under each scenario, but do not fully describe how the subpopulations behaved in the demographic model, because the model includes multiple types of stochasticity and can produce transient dynamics.

Kane Basin

During recent subpopulation studies estimates of $S*$ were derived from CR and harvest data collected during the period 1992–2014 (Chapter 10 in SWG 2016). Due to sparse data, SWG (2016) did not consider estimation models that allowed for temporal variation in survival. This presents a challenge for harvest assessment because all available estimates of $S*$ represent average values over a 23-year period, and do not reflect potential changes in survival in recent years. Furthermore, exploratory population reconstruction using the time-constant estimates of S^* and observed harvest patterns, resulted in a declining subpopulation trend over the period 1998–2014 (section Population initialization). This is inconsistent with estimated increases in abundance of the KB subpopulation from CR modeling, from 224 (SE = 40) for the period 1995–1997, to 357 (SE = 92) for the period 2013–2014 (Chapter 10 in SWG 2016). SWG (2016) provide several caveats for the estimated increase in abundance, including potentially inconsistent sampling frames between the 1990s and 2010s. However, other lines of evidence from subpopulation ecology and Traditional Ecological Knowledge also suggest that the KB subpopulation is currently healthy and has been stable or increasing in recent years (SWG 2016). We considered two alternative scenarios of the vital rates for the KB subpopulation to reflect uncertainty in current subpopulation status.

Reproductive parameters

The two scenarios of the vital rates for KB used reproductive parameters estimated from CR studies 2012–2014, based on field data described in Chapter 21 of SWG (2016) and using the methods described above. To calculate litter production rate we used $S = 0.95$, the estimate of total survival for females ≥ 3 years during the period 1992–2014 (Table 10.3 in SWG 2016). The resulting litter production rate for bears age ≥ 5 years (β_4) was 0.71 (Table KB1). Because we were unable to estimate litter production rate for 4-year-old bears (β_3) using data from genetic sampling conducted 2012–2014, we set this parameter to 0 based on the finding by Taylor et al. (2008) that 4-year-old bears did not reproduce. We set the value for β_5 to 0.10 based on Regehr et al. (2010), which is the only study to directly estimate the probability that a female observed with C0 in the spring of year *t*, conditional on losing her cubs, will re-breed and produce a new litter of C0 in the spring of year $t + 1$. This likely had a minor effect on results, due to the relative unimportance of β_5 to population growth (Hunter et al. 2007).

Scenarios for survival

We evaluated two scenarios for survival rates of KB polar bears. Scenario 1 used un-modified estimates of S^* as reported on page 496 of SWG (2016). Scenario 1 represents the hypothesis that time-constant estimates of S^* are accurate and represent the current status of the

KB subpopulation, despite being too low to reproduce the estimated increase in abundance over the period 1998–2014 (section Population initialization). Scenario 1 suggests a relatively poor status for the KB subpopulation (section Results).

For Scenario 2, we modified values of S* for some sex and age classes until the vital rates were sufficiently high to reproduce the estimated increase in abundance over the period 1998–2014. Specifically, for female and male polar bears age ≤ 2 years, we created 10 equal-increment values of *S** from a minimum corresponding to the point estimate for that sex and age class, to a maximum corresponding to the estimate of *S** for bears age ≥ 3 years of the same sex. This approach retained the lower values of unharvested survival for males compared to females, a pattern that was apparent for both the BB and KB subpopulations (SWG 2016). We chose to modify values of *S** for bears ≤ 2 years, rather than for adults, because CR and dead-recovery sample sizes were small for younger bears (Table 10.1 in SWG 2016), which resulted in high sampling uncertainty and increased potential for bias (Pollock et al. 1990). We performed population reconstructions for each set of equal-increment values of *S**, to determine the magnitude of increases in *S** necessary to achieve a 50% probability of reproducing the estimated increase in abundance of the KB subpopulation (section Population initialization). Thus, Scenario 2 represents the hypothesis that estimates of *N* for the KB subpopulation are accurate and provide a valid basis for inference about the subpopulation’s capacity to grow and support harvest. Uncertainty in estimates of *S** for Scenario 2 was calculated from the relative standard deviations for Scenario 1, which meant that even though estimates of *S** for bears ≤ 2 years were increased under Scenario 2, the corresponding level of data precision was not improved.

Estimates of *S** corresponding to the two scenarios of the vital rates are presented in Table KB2. Basic demographic parameters, calculated under asymptotic population dynamics, are presented in Table KB3.

Population projections

We performed population projections to evaluate the dynamics of the BB and KB polar bear subpopulations and to investigate the effects of different harvest strategies. For a given projection, the main biological inputs were: a starting value of *N* (expressed as a number of independent bears); mean values from a scenario of the vital rates, referenced to MNPL;

estimates of sampling variation, process variation, and the correlation structure of the vital rates; a starting stage distribution; a starting subpopulation density, expressed as the ratio N/K; stage-specific *mee* values; and a stochastic projection of the dimensionless metric κ, representing future variation in K. The main management inputs were: a value of F_O for use in Equation 1 to calculate harvest rate; a value of *SR* for use in Equation 2 to calculate harvest level; stage-specific harvest vulnerability vectors for females and males; a value for the management interval; and *rsd.mod*, the modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty, which was applied to future subpopulation assessments (section Abbreviations, Acronyms, etc.).

Populations were projected 35 years into the future (i.e., from $t = 1, 2, \ldots 36$), which is equivalent to approximately three polar bear generations (Regehr et al. 2016). At each time step $t = 2, 3, \ldots k$, the following operations were performed. First, subpopulations were projected forward 1 year using a stage-structured matrix model: $\boldsymbol{n}(\mathrm{t}+1) = \boldsymbol{A}(\mathrm{t}) \times \boldsymbol{n}(\mathrm{t})$, where $\boldsymbol{n}(\mathrm{t})$ is a stage distribution vector representing the number of animals in each life cycle stage at time step t, and $\boldsymbol{A}(\mathrm{t})$ is a 10×10 projection matrix (Caswell 2001). Entries in $\boldsymbol{A}(t)$ were defined in terms of vital rates in the life cycle graph (Figure 1). Demographic stochasticity was not included, because it is considered relatively unimportant at subpopulation sizes typical of polar bears (White 2000). Second, harvest was allocated among stages using a multinomial distribution with the probability for each stage calculated as the product of its proportional stage distribution and harvest vulnerability vector. For some projections, selective harvest led to the depletion of bears in one or more stages. If the specified harvest level exceeded the number of bears in a stage, the excess harvest was applied to adult bears of the same sex (i.e., stages 4 or 10). If the specified harvest exceeded the total number of one sex, the excess harvest was applied to adult bears of the other sex. Third, subpopulation density was calculated by summing *mee* values across animals in the subpopulation, then dividing by the total *mee* values available at carrying capacity. The survival and reproductive rates corresponding to this density were determined from the density-dependent curves, with modifications applied to the parameter β_4 based on the Allee submodel. Fourth, these vital rates were subject to density-independent stochastic variation. Finally, the resulting vital rates were used to construct a projection matrix for the next time step $\boldsymbol{A}(t+1)$.

During population projections, we defined persistence as maintaining a subpopulation size greater than a pre-determined quasi-extinction threshold. We used a threshold of 100 independent bears for the BB subpopulation, which is similar to values that have been used for brown bears (Wielgus 2002). We used a threshold of 25 independent bears for KB, because it is a smaller subpopulation and 25 bears was likely high enough to avoid negative small-population dynamics resulting from demographic stochasticity (Morris and Doak 2002). We note that Regehr et al. (2017) used higher quasi-extinction thresholds, calculated as 15% of starting *N*. We did not follow that approach because our analyses incorporated an Allee submodel, which provided a mechanistic description of small-population dynamics that have been suggested as important for polar bears (Molnár et al. 2014). During projections, subpopulations that crossed below the quasi-extinction threshold were considered extirpated and could not recover.

Population initialization

For both BB and KB we performed exploratory population reconstruction, which consisted of retrospective projections that used historic biological and management conditions. This helped to evaluate the vital rates, establish reasonable initial conditions for the main population projections (e.g., a subpopulation composition that was consistent with the history of sex-selective harvest), and reduce transient dynamics in early years of projections (Caswell 2001).

Baffin Bay

We performed population reconstruction for the period 1998–2010 using vital rates from Scenario 2. We did not include 2011–2013 in the reconstruction due to concerns about bias in the survival estimates for those years (SWG 2016). For the population reconstruction, the mean value of starting *N* was set to 1,968 independent bears. This was calculated from the estimated total subpopulation size (i.e., including C0 and C1) of 2,826 for the period 2011–2013 (Chapter 5 in SWG 2016). We used this starting value because of potential bias in the 1993–1997 estimate of *N* due to limited geographic sampling in the 1990s (Chapters 3 and 6 in SWG 2016). Methods to convert from total subpopulation size, to the number of independent bears, are described below. Starting N/K (i.e., at $t = 1$) was set to 0.67, which corresponded to MNPL as estimated from the density-dependent curves of the vital rates for Scenario 2. Harvest was implemented at a fixed level of 162 independent bears per year, which was the mean reported harvest for the BB subpopulation during the period 1998–2010 (Table 8.4 in SWG 2016).

We used a male-to-female sex ratio in the harvest (*SR*) of 1.25 for population reconstruction. This value was derived by averaging Canadian and Greenlandic harvest sex ratios, weighted by the total reported harvest in each country. The Canadian sex ratio was calculated directly from hunter-reported sex as it agreed with the genetically-determined sex, which indicated that 0.34 of harvested bears were female (Chapter 8 in SWG 2016). The Greenlandic sex ratio was estimated separately for this analysis, due to apparent discrepancies between hunter-reported sex and genetically-determined sex (Chapter 8 in SWG 2016). Specifically, we used the genetic sex data from tissue samples collected from 77 polar bears harvested in Greenland during the period 2011–2013, to estimate that 0.53 of the reported harvest was female. Lacking genetic sex data for other years, we assumed that this proportion was representative of the Greenlandic harvest 1998–2010. Harvest was implemented using the stage-specific harvest vulnerability vectors for the BB subpopulation. For population reconstruction, we used a deterministic proxy metric for *K* that was based directly on the observed time series of ice-covered days 1998–2010. Other specifications for population reconstruction were the same as for the full suite of projections (section Simulations).

The reconstructed subpopulation, using vital rates from Scenario 2 and a fixed-level harvest of 162 bears per year, exhibited a relatively stable trajectory during the period 1998–2010. The observed population growth rate was 1.01 (SE = 0.10) per year as calculated using the methods of Humbert et al. (2009). The ending proportion of females in the subpopulation was 0.69. For comparison, the mean proportion of females in the BB subpopulation during the period 2011–2013 was 0.66, as calculated using sex- and age-specific estimates of abundance for bears age ≥ 2 years from the most-supported CR model (Chapter 5 in SWG 2016). The similarity between the proportions 0.69 and 0.66, and the ability of population reconstruction to produce plausible population dynamics (i.e., a stable subpopulation under the observed harvest), provide a degree of confidence in the vital rates of Scenario 2.

To obtain a starting stage distribution for the full suite of population projections, we adjusted the final stage distribution from the population reconstruction until the proportion of females was 0.66, keeping the within-sex stage distributions constant. This produced the 10-stage distribution vector [0.07, 0.05, 0.05, 0.17, 0.18, 0.14, 0.07, 0.05, 0.05, 0.17]. When combined with estimates of C0 and C1 litter size for Scenario 2, this produced a ratio of independent bears to total bears of 0.70. Therefore, all projections for the BB subpopulation started with a mean value of $N_{t=1}$ = 2,826 × 0.70 ≈ 1,968 independent bears, where 2,826 is the estimated total subpopulation size for 2011–2013 from SWG (2016). To reduce transient dynamics, all projections started at a subpopulation density N/K = 0.81, the median estimated density at the final year of population reconstruction.

Kane Basin

We performed population reconstruction for the period 1998–2014, to evaluate the vital rates from Scenario 1 and to identify values of S^* for Scenario 2 (section Vital rates). For the population reconstruction, the mean value of starting N was set to 153 independent bears. This was calculated from the estimated total subpopulation size of 224 for the period 1995–1997 (Chapter 10 in SWG 2016). Methods to convert from total subpopulation size, to the number of independent bears, are described below. Starting N/K was set to 0.73, which corresponded to MNPL as estimated from the density-dependent curves of the vital rates for Scenario 1. Harvest was implemented at a fixed level of 8 independent bears per year, which is the mean harvest reported for the KB subpopulation during the period 1998–2013 (Tables 8.2 and 8.5 in SWG 2016).

We used SR = 0.94 for population reconstruction. This value was derived by averaging Canadian and Greenlandic harvest sex ratios, weighted by the total reported harvest in each country. The Canadian sex ratio was calculated directly from hunter-reported sex, which indicated that 0.33 of harvested bears were female (Chapter 8 in SWG 2016). The Greenlandic sex ratio was based on genetically-determined sex for bears harvested from the KB subpopulation during the period 2011–2014, which indicated that 0.53 of harvested bears were female (Chapter 8 in SWG 2016). Lacking genetic sex data for other years, we assumed that this proportion was representative of the Greenlandic harvest 1998–2014. Harvest was implemented using the stage-specific harvest vulnerability vectors that were calculated for the BB subpopulation.

For KB population reconstruction, we used a deterministic proxy metric for K with interannual variation based on the observed time series of ice-covered days 1998–2014, but with an increasing trend of approximately 3% per year. We subjectively included this trend in *K*, rather than using observed values of the sea-ice metric, because an increasing trend would be necessary to allow *N* to increase from 224 bears in 1995–1997 to 357 bears in 2012–2014 (i.e., $N_{t=17} / N_{t=1} \approx 1.6$; Chapter 10 in SWG 2016), conditional on vital rates that were sufficiently high to produce such an increase. Other specifications for population reconstruction were the same as for the full suite of projections (section Simulations).

An initial KB population reconstruction used vital rates from Scenario 1 and a fixed-level harvest of 8 bears per year. This resulted in a declining trajectory during the period 1998–2014, with an observed population growth rate of 0.98 (SE = 0.40) per year. The large variance in the observed growth rate was due primarily to high sampling uncertainty in vital rates for the KB subpopulation (Chapter 10 in SWG 2016). The fact that the growth rate was negative, despite an increasing proxy for *K*, suggests that a subpopulation with vital rates similar to Scenario 1 would be unlikely to support a harvest of 8 bears per year, even in the absence of density-dependent regulation.

Subsequent KB population reconstructions used the same conditions as described above, but with the estimates of $S*$ from Scenario 1 modified to include incremental increases in survival for bears age ≤ 2 years (section Vital rates). We found that an average proportional increase in $S*$ of 38% (i.e., $S^*_{modified} = 1.38 \times S^*_{original}$) produced an increasing trajectory during the period 1998–2014, with an observed population growth rate of 1.03 (SE = 0.33). This corresponded to a median increase in abundance of $N_{t=17} / N_{t=1} = 1.59$ (SE = 0.71), which is similar to the estimated increase in abundance for the KB subpopulation from 1995–1997 to 2012–2014. Therefore, Scenario 2 of the vital rates for the KB subpopulation included these modified estimates of $S*$ for bears ≤ 2 years (section Results).

The reconstructed subpopulation, using vital rates from Scenario 2 and a fixed-level harvest of 8 bears per year, produced an ending proportion of females in the subpopulation of 0.70. For comparison, the mean proportion of females in the KB subpopulation during the period 2012–2014 was 0.71, as calculated from sex- and age-specific estimates of abundance for bears ≥ 2 years from the most-supported CR model (Chapter 10 in SWG 2016). To obtain a starting stage distribution for the full suite of projections, we adjusted the final stage distribution from the population reconstruction until the proportion of females was 0.71, keeping the within-sex stage distributions constant. This produced the 10-stage distribution vector [0.06, 0.06, 0.05, 0.24, 0.16, 0.14, 0.05, 0.04, 0.03, 0.17]. When combined with estimates of C0 and C1 litter size for Scenario 2, this leads to a ratio of independent bears to total bears of 0.68. Therefore, all projections for the KB subpopulation started with a mean value of $N_{t=1} = 357 \times 0.68 \approx 244$ independent bears, where 357 is the estimated total subpopulation size for 2012–2014 from SWG (2016). To reduce transient dynamics, all projections started at a subpopulation density $N/K = 0.69$, the median estimated density at the final year of population reconstruction.

Simulations

We define a “simulation” as multiple replicates of a population projection, where each replicate has the same mean biological inputs and the same management inputs (section Population projections). For each simulation, we used a parametric bootstrap procedure that generated 250 correlated random samples of the input vital rates and starting value of N, for the purpose of representing sampling variation in the vital rates (White 2000). We subjectively included sampling variation as 75% of total uncertainty (i.e., temporal variation plus sampling uncertainty) following the example of Taylor et al. (2002). Samples of the vital rates were generated using either a multivariate beta distribution or a stretched beta distribution (Morris and Doak 2002), as described in Appendix S3 of Regehr et al. (2017). When vital rates were near the boundary conditions [0,1] and variances were large, shape parameters for the beta distribution occasionally could not be determined. When this occurred, we generated a sample for the vital rate in question using a truncated normal distribution. The correlation structure for sampling variation in the vital rates was informed by the most-supported CR model for the BB subpopulation (Table 5.7 in SWG 2016). Specifically, we used a correlation coefficient of 1 within the following sets of parameters, and a correlation coefficient of 0 between the sets: [σ_1, σ_2, σ_3, σ_4, σ_5, σ_6], [σ_7, σ_8, σ_9, σ_{10}], [σ_{L0}, σ_{L1}], [β_3, β_4, β_5], and [N]. This correlation structure was also used for simulated population assessments, based on the assumption that future estimation methods would be broadly similar to SWG (2016).

For each random sample of the vital rates and starting N, we ran 50 projections, each with a different stochastic projection of κ (section Carrying capacity and environmental variation). Also, the projections included stochastic, density-independent variation at each time step. Therefore, for each simulation the resulting 250 × 50 = 12,500 replicate projections reflected both sampling and environmental variation. Although 12,500 is a relatively small number of replicates for population viability analysis (White 2000), this number was computationally feasible (section Software) and gave reproducible results at the levels of precision we report.

For each subpopulation, we performed simulations designed to evaluate a range of biological conditions and harvest strategies relevant to polar bear management (see below). For each simulation, we report the probability of meeting management objectives, calculated as the number of replicates that met the corresponding population condition (Table BB1) divided by the total number of replicates. At specific time steps, we also report $P_{extirpation}$, defined as the proportion of replicates for which N declined below the quasi-extinction threshold at any time step prior to time step $t = k$; and the probability of male depletion ($P_{male.dep}$), defined as the proportion of replicates for which the number of adult males (stage 10) was below 50% of the quasi-extinction threshold at $t = k$. The metric $P_{male.dep}$ is relevant because lower values of S^* for males compared to females, combined with sex-selective harvest, led to severe depletion of adult male bears under some conditions. Finally, we report the median change in subpopulation size ($N_{t=k}$ / $N_{t=1}$), the median subpopulation density ($N_{t=k}$ / $K_{t=k}$), and the mean realized harvest level ($H_{t=k}$). These values were calculated over all replicates, including those that led to extirpation. The metric H is relevant because some simulations included declining N (to declining K or to high harvest) and declining r (due to Allee effects caused by male depletion), which led to declining values of H over time under state-dependent approach.

Baffin Bay

We performed a primary set of simulations for the BB subpopulation to evaluate sustainable harvest for the three scenarios of the vital rates. All primary simulations used a management interval of 15 years and *rsd.mod* = 1. For each scenario, we performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Five, 11, and 8 values of the management factor F_O for Scenarios 1, 2, and 3, respectively. These values of F_O corresponded to starting harvest levels that differed by 20 bears per year, and encompassed the estimates of maximum sustainable yield for each scenario based on asymptotic dynamics and non-selective harvest (section Results).

2. Three values of sex ratio in the harvest, corresponding to *SR* = 1.0, 1.25, and 2.0. The value of *SR* = 1.25 represents the current status quo for the BB subpopulation under the assumptions made during population reconstruction (section Population initialization). We included *SR* = 2 because it is a common management objective for polar bears (Taylor et al. 2008b). We included *SR* = 1 to evaluate non-sex selective harvest, which might be a strategy for managed population reduction.

We performed a secondary set of simulations using the vital rates for Scenario 2 only, which we considered the most likely representation of the current status of the BB subpopulation. The objectives were to evaluate the effects of changes in the management interval and the precision of data obtained from future subpopulation assessments. All secondary simulations used *SR* = 1.25. We performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Nine values of the management factor F_O, corresponding to starting harvest levels that differed by 10 bears per year, and encompassed the range of harvest that met management objectives for Scenario 2 during primary simulations.
2. Three values for the management interval corresponding to 10, 15, and 20 years.
3. Three levels of precision in subpopulation data, corresponding to *rsd.mod* = 0.5, 1.0, and 1.5.

We also performed several *post hoc* simulations that were focused on the issue of managed population reduction.

Kane Basin

We performed a primary set of simulations for the KB subpopulation to evaluate sustainable harvest for the two scenarios of the vital rates. All primary simulations used a management interval of 15 years and *rsd.mod* = 1. For each scenario, we performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Five and 9 values of the management factor F_O for Scenarios 1 and 2, respectively. These values of F_O corresponded to starting harvest levels that differed by 2 bears per year, and encompassed the estimates of maximum sustainable yield for each scenario based on asymptotic dynamics and non-selective harvest (section Results).

2. Two values of sex ratio in the harvest, corresponding to *SR* = 0.94 and 2.0. The value of *SR* = 0.94 represents the current *status quo* for the KB subpopulation under the assumptions made during population reconstruction (section Population initialization).

We performed a secondary set of simulations using the vital rates for Scenario 2 only, to evaluate the effects of changes in the management interval and the precision of data obtained from future subpopulation assessments. Scenario 2 was more useful for this investigation because, unlike Scenario 1, it led to non-zero harvest levels that met management objectives. All secondary simulations used *SR* = 0.94. We performed simulations over 36 annual time steps for all combinations of the following inputs:

1. Nine values of the management factor F_O, corresponding to starting harvest levels that differed by 1 bear per year, and encompassed the range of harvest that met management objectives for Scenario 2 during the primary simulations.
2. Three values for the management interval corresponding to 10, 15, and 20 years.
3. Three levels of precision in subpopulation data, corresponding to *rsd.mod* = 0.5, 1.0, and 1.5.

Finally, we performed several *post hoc* simulations focused on the ramifications of high uncertainty in estimates of S^* for bears age ≤ 2 years.

Software

Computations were performed in the R computing language (version R 3.4.0; The R Project for Statistical Computing; http://www.r-project.org). Simulations were run the Amazon Elastic Compute Cloud (http://aws.amazon.com/ec2/) using an Amazon Machine Image for RStudio Server (RStudio 2016) developed by L. Aslett (http://www.louisaslett.com/RStudio_AMI/). Each simulation took approximately 60 minutes using a Memory Optimized r4.xlarge computing instance.

Results

Baffin Bay

Primary simulations

We performed a primary set of simulations to evaluate population dynamics and sustainable harvest, for three scenarios of the vital rates, over a period of three polar bear generations during which K declined. Projections used a 15-year management interval, a baseline level of data precision (i.e., *rsd.mod* = 1.0) based on recent subpopulation surveys (SWG 2016), and a mean starting subpopulation size $N_{t=1}$ = 1,968 (SE = 236) independent bears with a stage distribution and subpopulation density determined from population reconstruction (section Population initialization). Tables BB5–BB7 present the highest harvest strategies that met management objectives for each scenario of the vital rates (see below). The harvest rate (h) in these tables is presented as the percentage of total subpopulation size (i.e., subpopulation size including C0s and C1s) that is removed each year as independent bears, because this definition of h is commonly used in polar bear management. Strategies with lower values of F_O than appear in Tables BB5–BB7, but otherwise similar inputs (e.g., the same harvest sex ratio), also met management objectives. Some of the harvest strategies in Tables BB5–BB7 could result in the depletion of adult male bears; probabilities of extirpation, compared to projections with no harvest, that approach the upper limit of 0.05; or declines in the calculated harvest level over time. Detailed results from the primary simulations are presented in Appendix S1.

Scenario 1

Scenario 1 of the vital rates resulted in an asymptotic intrinsic growth rate at MNPL (r_{MNPL}) of 0.03 (Table BB4), suggesting a limited capacity for growth and low resilience relative to other polar bear subpopulations (Regehr et al. 2017). Statistical uncertainty in estimates of S^* for the period 2011–2013 (Table BB3) contributed to high uncertainty in the estimate of r_{MNPL}, with approximately 26% of its sampling distribution below 0 (i.e., corresponding to a negative intrinsic growth rate). Management Objective 1 was not achievable even in the absence of harvest, due to the combined effects of low r, declining K, and high uncertainty in the vital rates (Table BB5). Management Objective 2 could be met using F_O = 0 to 0.41, depending on the value of *SR* and risk tolerance. This corresponds to a starting harvest rate $h_{t=1}$ = 0 to 0.7%, and a starting harvest level $H_{t=1}$ = 0 to 20 bears per year. Upper limits on F_O that met Management Objective 3 were 0 to 1.22, depending on *SR* and risk tolerance. However, due in part to high uncertainty in the vital rates, harvest strategies at the upper end of this range simultaneously increased $P_{extirpation}$ toward the upper condition of 0.05 at $t = 15$, while being unlikely to reduce the median subpopulation size by 30% (Table S.BB1). A subpopulation similar to Scenario 1 would have little capacity to support harvest, and would risk a 2 to 3% chance of extirpation at $t = 36$ in the absence of harvest (Table S.BB1).

Scenario 2

Scenario 2 resulted in r_{MNPL} = 0.08 (Table BB4), suggesting a strong capacity for growth and relatively high resilience. Due to declining *K*, Management Objective 1 was either not achievable or only achievable with no harvest (Table BB6). Management Objective 2 could be met using F_O = 0.43 to 1.03, depending on *SR* and risk tolerance. This corresponds to a starting harvest rate $h_{t=1}$ = 3.6 to 5.7%, and a starting harvest level $H_{t=1}$ = 100 to 160 bears per year. At the upper end of this range, a harvest strategy using F_O = 0.92 and the *status quo* value of $SR = 1.25$ corresponds to $h_{t=1}$ = 5.7% and $H_{t=1}$ = 160 bears per year, which is similar to harvest of the BB subpopulation in recent decades (Chapter 8 in SWG 2016). A sample of replicates from population projections can help visualize the effects of this harvest strategy on subpopulation trajectories. Figure BB3 shows that the median subpopulation size declines in parallel with (but not faster than) declining *K*, which is a consequence of a state-dependent management approach that meets Management Objective 2. The color-coding in Figure BB3 identifies the potential for male depletion or subpopulation extirpation in later years (Table S.BB2). For replicates that experienced male depletion, reproductive rates declined due to Allee effects in the mating system. This reduced the subpopulation's capacity for growth and resulted in lower calculated harvest levels under the state-dependent approach. Figure BB4 illustrates these effects, for the same harvest strategy that was shown in Figure BB3. Other harvest strategies that were more selective for males (i.e., $SR = 2$), including some strategies that met Management Objective 2 (Table BB6), had higher probabilities of causing male depletion (e.g., up to 0.25 at $t = 36$; Table S.BB2). Evaluation of Management Objective 3 is presented with results from the secondary simulations (see below).

Scenario 3

Scenario 3 was based on hypothetical vital rates that resulted in r_{MNPL} = 0.05 (Table BB4), representing a subpopulation with average capacity for growth and resilience. This scenario provided a benchmark for comparison with the data-based Scenarios 1 and 2. Similar to Scenario 2, Management Objective 1 was either not achievable or only achievable with no harvest, due to declining K (Table BB7). Management Objective 2 could be met using F_O = 0.53 to 0.89, depending on SR and risk tolerance. This corresponds to a starting harvest rate $h_{t=1}$ = 2.1 to 3.6%, and a starting harvest level $H_{t=1}$ = 60 to 100 bears per year. Harvest strategies under Scenario 3 that used $SR = 2$ were less likely to cause male depletion than similar strategies under Scenario 2 (Table S.BB3), because under Scenario 3 values of $S*$ were similar for adult females and adult males. Upper limits on F_O that met Management Objective 3 were 0.66 to 1.40, depending on SR and risk tolerance. In some cases, the condition requiring that $P_{extirpation} < 0.05$ at $t = 15$, compared to a similar projection without harvest, was the limiting factor for Management Objective 3. In other words, a harvest strategy could achieve an acceptable level of risk with respect to the population condition $N_{t=15} > (0.7 \times N_{t=1})$ but also result in $P_{extirpation} > 0.05$. Harvest strategies that met Management Objective 3, in terms of both $N_{t=15} > (0.7 \times N_{t=1})$ and the condition on $P_{extirpation}$, were unlikely to reduce the subpopulation size by 30% (Table S.BB3). These findings suggest that a managed population reduction of 30% is unlikely to be achieved, within the guidelines for risk tolerance provided by the JC, when using a 15-year management interval for a subpopulation with vital rates and a level of data precision similar to Scenario 3.

Secondary simulations

We performed a secondary set of simulations for Scenario 2 to evaluate the effects of management interval and data precision. Table BB8 shows the highest harvest strategies that met Management Objective 2b as a function of these factors, illustrating the potential impact of different management conditions. For example, $h_{t=1}$ is 54% higher for a 10-year management interval and $rsd.mod$ = 0.5 (which corresponds to a relative standard deviation of 0.01 in σ_4 due to sampling uncertainty), compared to a 20-year management interval and $rsd.mod$ = 1.5 (which corresponds to a relative standard deviation of 0.03 in σ_4). Table S.BB4 provides detailed results for the simulations summarized in Table BB8. In Table S.BB4, results for the expected value of $H_{t=36}$ require additional explanation. It appears counterintuitive that $H_{t=36}$ is higher for a 20-year management interval than for a 15-year management interval. This occurs because the harvest strategies in Table S.BB4 are fairly aggressive, and can result in moderate degrees of male depletion and reduced capacity for growth at $t = 36$. Using a 15-yr management interval, these negative effects result in a reduced harvest level at the second subpopulation assessment, which occurs at $t = 32$. In contrast, using a 20-year management interval, the second subpopulation assessment does not occur until $t = 42$, which is beyond the duration of projections. The ramifications are that using a 20-year management interval (i) leads to higher probabilities of extirpation at $t = 36$, and (ii) would be expected to result in large reductions to the calculated harvest level at $t = 42$.

We used results from the secondary set of simulations to evaluate Management Objective 3, because the primary simulations suggested that achieving a subpopulation reduction, within the specified risk tolerance, would require a short management interval and improved data precision. In the secondary simulations, the harvest strategy with the highest harvest and best management conditions was $F_O = 1.15$, a 10-year management interval, and $rsd.mod = 0.5$. This corresponded to $h_{t=1} = 7.1\%$ and $H_{t=1} = 200$. For this strategy, the median value of $N_{t=15} / N_{t=1}$ was 0.86 at $t = 15$ (i.e., a median reduction of 14% in starting subpopulation size; Table S.BB4). Although this strategy met Management Objective 3b as stated in Table BB1, it did not reduce the subpopulation by 30%. Also, it led to a 0.38 probability that subpopulation size was below MNPL at $t = 15$, which exceeded the risk tolerance for the condition on maximum sustainable yield that was associated with Management Objective 3 (section Management Objectives).

Post hoc simulations

We performed two *post hoc* simulations to inform future considerations for managed population reduction. First, we identified a state-dependent harvest strategy (BB_S1) that came as close as possible to achieving a subpopulation reduction of 30% in 15 years, while meeting the population condition for Management Objective 3b as stated in Table BB1, but without the additional condition related to maximum sustainable yield. Harvest strategy BB_S1 used $F_O = 1.58$, $SR = 1.0$, a 5-year management interval, and $rsd.mod = 0.5$. Use of $SR = 1$ promoted subpopulation reduction by removing more females, compared to a sex-selective harvest. This strategy corresponded to $h_{t=1}$ = 8.7% and $H_{t=1}$ = 245. At t = 15, the median value of $N_{t=15}$ / $N_{t=1}$ was 0.75 (i.e., a median reduction of 25% in starting subpopulation size) and Management Objective 3b was met (Table S.BB5). The mean harvest level declined at each management interval, concurrent with declining subpopulation size (e.g., $H_{t=15}$ = 180 bears per year; Table S.BB5). Conceptually, BB_S1 represents a near-optimal harvest strategy that would require nearly continuous surveys and rapid management response.

For comparison with BB_S1, we evaluated a second harvest strategy (BB_S2) that used the same starting harvest level, but did not follow a state-dependent approach and used the *status quo* value SR = 1.25. Strategy BB_S2 applied a fixed-level harvest of 250 bear per year, for a period of 15 years, without new subpopulation assessments or changes to management during this period. At t = 15, the median value of $N_{t=15}$ / $N_{t=1}$ was 0.55, indicating that the subpopulation was depleted beyond the desired 30% reduction. Also, strategy BB_S2 did not meet Management Objective 3 and resulted in a 0.30 probability of male depletion and a 0.23 probability of subpopulation extirpation at t = 15 (Table S.BB5).

Kane Basin

Primary simulations

We performed a primary set of simulations to evaluate population dynamics and sustainable harvest, for two scenarios of the vital rates, over a period of three polar bear generations during which K remained stable. Projections used a 15-year management interval, the baseline level of data precision (i.e., *rsd.mod* = 1.0), and a mean starting subpopulation size $N_{t=1}$ = 244 (SE = 41) independent bears with a stage distribution and subpopulation density determined from population reconstruction (section Population initialization). Tables KB4–KB6 present the highest harvest strategies that met management objectives for each scenario of the vital rates. Strategies with lower values of F_O, but otherwise similar inputs, also met management objectives. Some of the harvest strategies in Tables KB4–KB6 could result in the depletion of adult male bears or increased probabilities of extirpation, compared to projections with no harvest, that approached the upper limit of 0.05. Detailed results for the primary simulations are presented in Appendix S1.

Scenario 1

Scenario 1 of the vital rates resulted in r_{MNPL} = 0.01 (Table KB3), suggesting a very limited capacity for growth and low resilience. Statistical uncertainty in estimates of S^* for the period 2012–2014 (Table KB2), especially for bears ≤ 2 years, contributed to high uncertainty in the estimate of r_{MNPL}, with approximately 29% of its sampling distribution below 0 (i.e., corresponding to a negative intrinsic growth rate). Due to low values of r and high uncertainty in the vital rates, most management objectives could not be met even with no harvest (Table KB4). This is illustrated by Figure KB1, which shows a sample of replicates from population projections with $F_O = 0$ (i.e., no harvest). Although the median N increases gradually over time, the subpopulation trajectories are highly variable. The color-coding in Figure KB1 indicates that male depletion is possible due to lower estimates of S^* for males compared to females (e.g., $P_{male.dep} = 0.10$ at $t = 15$; Table S.KB1). A subpopulation with vital rates similar to Scenario 1 would have little guarantee of supporting harvest, and would face a 2 to 4% chance of extirpation at $t = 36$ in the absence of harvest (Table S.KB1). The ramifications of high uncertainty in vital rates for the KB subpopulation were evaluated in the secondary and *post hoc* simulations (see below).

Scenario 2

Scenario 2 resulted in r_{MNPL} = 0.05 (Table KB3), suggesting a medium capacity for growth and resilience. Management Objectives 1a and 2a (i.e., the version of the objectives with low risk tolerance for not achieving the population condition; Table BB1) could not be met with no harvest, due to variability in subpopulation trajectories arising from uncertainty in the vital rates (Table S.KB2). Management Objective 1b could be met using F_O = 0.31 to 0.48, depending on the value of SR (Table KB5). This corresponds to a starting harvest rate $h_{t=1}$ = 1.7% and a starting harvest level $H_{t=1}$ = 6 bears per year, which is similar to recent harvest of the KB subpopulation (SWG 2016). Management Objective 2b could be met using F_O = 0.21 to 0.31, depending on the value of SR. This corresponds to a starting harvest rate $h_{t=1}$ = 1.1% and a starting harvest level $H_{t=1}$ = 4 bears per year. The harvest strategy that meets Management Objective 2 is lower than the strategy that meets Management Objective 1, because projections for the KB subpopulation included a stable trend in K (i.e., $K_{t=36} \approx K_{t=1}$) Therefore, at $t = 36$, Management Objective 1 requires a subpopulation size that is greater than $0.9 \times N_{t=1} = 0.90 \times (0.69 \times K_{t=1}) \approx 0.62 \times K_{t=1}$; whereas Management Objective 2 requires a subpopulation size that is greater than $0.70 \times K_{t=36} \approx 0.70 \times K_{t=1}$ (i.e., a lower subpopulation size). Harvest strategies that met Management Objectives 1 and 2 were associated with increases in median N of up to 21% at $t = 36$ (Table S.KB2). This indicates that uncertainty in the vital rates, rather than the mean values of the rates, was a limiting factor in meeting management objectives. At $t = 36$, the harvest strategies in Table KB5 were associated with probabilities of causing male depletion of up to 0.27, due in part to lower S^* of males; and increased probabilities of extirpation, compared to projections with no harvest, of up to 0.03 (Table S.KB2).

Secondary simulations

We performed secondary simulations for Scenario 2 to evaluate the effects of management interval and data precision. Table KB6 shows the highest harvest strategies that met Management Objective 1b as a function of these factors, indicating the potential impact of different management conditions on harvest strategies for the KB subpopulation. For example, the highest harvest strategy under improved management conditions (i.e., a 10-year management interval and $rsd.mod = 0.5$) corresponded to $h_{t=1} = 2.2\%$ and $H_{t=1} = 8$. This harvest rate is 57% higher than the rate for a 20-year management interval and $rsd.mod = 1.5$.

In contrast to the BB subpopulation, the highest starting harvest level for the KB subpopulation, under improved management conditions, was lower than the expected value of maximum sustainable yield based on asymptotic population dynamics (i.e., 13 bears per year; Table KB3). This finding is largely due to high uncertainty in estimates of S^* for bears age ≤ 2 years (Table KB3). The reason is that improved data precision (i.e., $rsd.mod = 0.5$) only reduced sampling variation for simulated population assessments that occur in the future. Therefore, all subpopulation trajectories were highly variable during the first management interval (i.e., for the first 10, 15, or 20 years) due to high uncertainty in the baseline vital rates for Scenario 2. Because we only evaluated strategies with time-constant values of F_O, the range of harvest strategies that met management objectives was constrained by high uncertainty in the currently-available data for the KB subpopulation. A consequence of this effect is that, for some harvest strategies, the calculated harvest level increased after the first management interval, and remained 1-2 bears higher than the starting value at $t = 15$ and 36 (Table S.KB3).

Post hoc simulations

We performed two *post hoc* simulations to explore the ramifications of high uncertainty in estimates of S^* for bears age ≤ 2 years. First, we reduced sampling variation in these estimates by 75%, which resulted in levels of uncertainty similar to bears age ≥ 3 years (e.g., the reduced relative standard deviation due to sampling uncertainty was 0.04 for the vital rate σ_1, which is equivalent to the un-modified value for σ_4). Conceptually, this permitted exploration of how a higher level of confidence in estimates of S^* for bears age ≤ 2 years might affect sustainable harvest. After reducing the uncertainty in estimates of S^*, we performed simulations under conditions similar to current harvest practices for the KB subpopulation (i.e., $SR = 0.94$, a 15-year management interval, and $rsd.mod = 1.0$) and identified the highest strategy that met Management Objective 1b. The resulting strategy (KB_S1) corresponded to $F_O = 0.80$, $h_{t=1} = 2.8\%$, and $H_{t=1} = 10$ bears per year (Table S.KB4). Interpretation of these results requires caution, because there was not an analytical basis for reducing uncertainty in estimates of S^* for bears age ≤ 2 years. Nonetheless, this *post hoc* simulation can provide guidance on what a sustainable harvest strategy might be, if it was assumed with an increased degree of confidence that survival rates of bears age ≤ 2 years have been sufficiently high to produce the estimated increase in abundance for the KB subpopulation.

Second, we explored uncertainty in estimates of S^* for bears age ≤ 2 years using a different approach. For all other simulations in this report, total uncertainty in the vital rates was partitioned as 75% sampling variation and 25% process variation, following the example of Taylor et al. (2002). Because time-constant estimates of S^* for KB bears were referenced to a 23-year period from 1992–2014, this partitioning may not be accurate for younger animals, which often exhibit higher interannual variation in survival compared to adults (Eberhardt 2002). We were not able to perform an analysis of variance components in S^* (e.g., Cooch and White 2016) due to sparse data. Therefore, for bears age ≤ 2 years, we subjectively repartitioned total uncertainty as 25% sampling variation and 75% process variation. After making this change, we performed simulations under conditions similar to current harvest practices for the KB subpopulation (i.e., $SR = 0.94$, a 15-year management interval, and $rsd.mod = 1.0$) and identified the highest harvest strategy that met Management Objective 1b. The resulting strategy (KB_S2) corresponded to $F_O = 0.64$, $h_{t=1} = 2.2\%$, and $H_{t=1} = 8$ bears per year (Table S.KB4).

Discussion

We used subpopulation data from SWG (2016) in a demographic model adapted from Regehr et al. (2017) to evaluate a suite of potential harvest strategies for the BB and KB polar bear subpopulations, relative to management objectives and risk tolerances provided by the JC.

Demographic and management model

Additional details of the demographic model are presented in Regehr et al. (2015, 2017), including caveats and topics for future work. In this report we integrated the demographic model with a mechanistic submodel of Allee effects in the mating system (Molnár et al. 2008, 2014). In simulations for both the BB and KB subpopulations, lower estimates of S^* for males than females, combined with a sex-selective harvest, produced stage distributions that were skewed toward females. If the demographic model did not incorporate Allee effects, such subpopulations could exhibit unrealistically high values of r (i.e., in excess of the theoretical r_{max} under asymptotic population dynamics) because most adults were female and litter production rates could potentially remain high even in the near-absence of adult males. This effect could be compounded by the fact that the model tracked subpopulation density in terms of metabolic energetic equivalents, which allowed a given resource base to support a larger number of females compared to males (section Density dependence). We suggest that deriving parameters of the Allee submodel for the BB and KB subpopulations, and validating model-based predictions against field data, are areas for future work.

The demographic model differed from RISKMAN (Taylor et al. 2001) in several ways, including its ability to model temporal changes in vital rates and K. Also, we used a correlation structure for sampling variation that was informed by the CR models used to estimate vital rates (Chapter 6 in SWG 2016), whereas RISKMAN assumes sampling errors are independent (Taylor et al. 2006). This can influence PVA results, with correlated vital rates generally resulting in more variable subpopulation trajectories and higher risks of negative outcomes. To illustrate, we used the demographic model to calculate an unharvested, asymptotic population growth rate (λ)

using vital rates for the BB subpopulation as reported in Taylor et al. (2005). Our estimate of λ = 1.053 (SE = 0.022) was similar to the geometric mean estimate of unharvested λ = 1.055 (SE = 0.011) in Taylor et al. (2005), but our estimated standard error was twice as large. We suggest that our approach is more consistent with recommended practices in population viability analysis (Morris and Doak 2002).

Application of the demographic model required several key assumptions. First, to create density-dependent curves of the vital rates it was necessary to specify a subpopulation density (N/K) at which the vital rates were estimated. In practice, it is not possible to directly estimate K for wildlife populations (e.g., Gerrodette and Demaster 1990). Therefore, we inferred that N/K corresponded to a subpopulation size in the vicinity of MNPL, based on evidence that harvest in recent decades had been near maximum sustainable yield (Regehr et al. 2017). If actual N/K corresponded to a subpopulation size below MNPL, our estimates of intrinsic growth rate (r) could be positively biased. That is, the value of r corresponding to the vital rates would be closer to r_{max}, whereas we assumed it was equivalent to r_{MNPL}. If actual values of N/K corresponded to a subpopulation size significantly above MNPL, bias would be in the opposite direction. Second, we initialized population projections at time step $t = 1$ at a subpopulation size close to MNPL. This assumed that the BB and KB subpopulations are currently not experiencing strong density-dependent suppression of demographic parameters, which could be inaccurate for BB given evidence of range contraction (Chapter 4 in SWG 2016) and links between sea-ice and nutritional condition and reproductive rates (Chapters 6 and 7 in SWG 2016). Although we placed low confidence in the low estimates of S^* for the BB subpopulation from 2011–2013 (Scenario 1; see below), it is possible these estimates partially reflected negative effects of sea-ice loss. In combination, the short duration of recent subpopulation assessments in BB and KB, statistical uncertainty and potential bias in demographic parameters, and interannual variation, precluded direct estimation of subpopulation density. Our modeling approach did not make purposefully conservative assumptions about current density effects, especially for the BB subpopulation, and therefore could have understated the current and future effects of sea-ice loss. We recommend that future predictive modeling include sensitivity analyses with respect to key assumptions (e.g., Zabel et al. 2006).

Polar bears are distributed throughout the circumpolar Arctic in 19 subpopulations (PBSG 2010). Their life history is dependent on sea ice (Laidre and Regehr 2017), which is used as a platform from which to hunt their primary prey, ringed seals (*Pusa hispida*) and bearded seals (*Erignathus barbatus*). Earlier sea-ice breakup and reductions in optimal ice habitat have been linked to reductions in polar bear body condition, survival, reproduction, and abundance in some subpopulations (Stirling et al. 1999; Regehr et al. 2007, 2010; Rode et al. 2012; Bromaghin et al. 2015; Lunn et al. 2016; Obbard et al. 2016). Additional studies have documented use of less optimal sea ice habitat in several polar bear subpopulations (e.g. Durner et al. 2009, Wilson et al. 2014, Laidre et al. 2015, McCall et al. 2015). Our population projections for the BB and KB subpopulations included environmental variation primarily through the proxy metric for K, which was calculated from remote-sensing data for sea ice. Sea-ice metrics from other case studies for polar bears are reviewed by Stern and Laidre (2016). We did not consider rapid, non-linear declines in K or potentially catastrophic ecological or demographic effects due to climate change (e.g., Derocher et al. 2013), although the modeling framework could readily be adapted to include such effects. If subpopulations experience negative density-dependent effects that are larger or more abrupt than represented by the proxy metric for K, or negative density-independent effects that occur rapidly with respect to the management interval (i.e., so that multiple years elapse before such effects are detected), the harvest strategies identified in this report might cease to meet management objectives, resulting in increased risk of negative outcomes. Following a state-dependent management approach with a relatively short management interval (e.g., 10–15 years) can mitigate such risks, because reductions in N and the vital rates, whatever their cause, could be detected in future subpopulation surveys, and harvest strategies adjusted accordingly. Population dynamics and harvest strategies for declining populations are reviewed in detail in USFWS (2016).

Management objectives

We evaluated Management Objectives 1 and 2 at $t = 36$, corresponding to three polar bear generations (Regehr et al. 2016) in the future, a common time reference for population projections (e.g., IUCN 2017). We also report results at $t = 15$ years to provide insight into near-term population dynamics and identify potential metrics for monitoring (e.g., the proportion of females, see below).

Management Objective 1, which desired to achieve $N \geq 90\%$ of its current value, is more relevant to harvest assessments when habitat is stable or increasing. Under conditions of declining *K*, it is not possible to meet Management Objective 1 over the long term, even in the absence of harvest. For example, population projections for BB only met Management Objective 1b under cessation of harvest, which resulted in transient subpopulation increases as *N* approached *K*, followed by declines as *N*/*K* reached 1 (Tables BB6 and S.BB2).

Management Objective 2, which desired to keep $N \geq$ MNPL with respect to a changing *K*, is more relevant to harvest assessments when habitat is declining. The goal is to maintain a constant ratio of *N*/*K*, such that subpopulation size and carrying capacity decline in parallel. If *N* remains far enough below *K* due to harvest (e.g., at MNPL), density effects are alleviated and there is a harvestable surplus. Under a harvest strategy that fulfills these conditions, long-term declines in *N* are driven primarily by declines in *K*. Regehr et al. (2017) proposed that such strategies are possible for polar bears, as long as habitat loss affects subpopulations primarily through density-dependent mechanisms (e.g., increased crowing and competition for limited resources), or if a state-dependent management approach is followed and the management interval is short enough to respond to density-independent reductions in *r* (e.g., reduced reproductive success because bears have insufficient time to hunt seals on the sea ice, regardless of density). Figure BB3 shows sample replicates from population projections that illustrate this concept, except toward the end of the projections when the probability of severe male depletion increases and causes reproductive failure (see below).

Management Objective 3 desired to achieve, but not exceed, a 30% reduction in *N* in 10–15 years, while maintaining subpopulation size above the level necessary to achieve maximum sustainable yield (i.e., above MNPL). Simultaneously meeting these two population conditions is likely not possible. That is because MNPL for polar bears occurs at approximately $N/K = 0.70$ (Regehr et al. 2017). Unless a subpopulation started at $N/K = 1$, a 30% reduction in *N* would necessarily result in a density $N/K < 0.70$ (i.e., below the subpopulation size that would produce maximum sustainable yield).

Harvest and subpopulation sex ratio

For both subpopulations, we performed projections with multiple values of sex ratio in the harvest (*SR*). Male-biased harvest is a common wildlife management and conservation tool (e.g., Mysterud 2011). For polar bears, seeking to harvest at $SR = 2$ (i.e., a 2:1 male-to-female ratio) is intended to protect adult females (Taylor et al. 2008b), which have the highest reproductive value (Hunter et al. 2007). For the BB and KB subpopulations, harvest data from recent decades suggested that *SR* = 1.25 and 0.94, respectively. These estimates were based on hunter-reported sex in Canada, which genetic testing suggests is highly accurate; and genetic sex determination of harvest samples from Greenland in the 2010s, which was assumed to represent the long-term sex ratio in the Greenlandic harvest due to apparent inaccuracies in hunter-reported sex (Chapter 8 in SWG 2016). For most of our simulations, harvest strategies that used *SR* = 2, instead of the lower *status quo* values of *SR*, did not result in higher harvest rates that met management objectives. This should not be interpreted as evidence against the conservation value of sex-selective harvest. Rather, it is a consequence of lower estimates of *S** for males than females in both subpopulations which, in conjunction with a sex-selective harvest, often led to the depletion of males. This had negative effects on reproduction via Allee effects in the mating system, translating into lower realized values of *r* and lower sustainable harvest. Taylor et al. (2008b) suggested that a 2:1 male-to-female harvest designed to achieve maximum sustainable yield, is unlikely to reduce the abundance or mean age of male bears to the point of reducing mating success. However, that analysis used equal survival rates for males and females, and the harvest rate was calculated relative to a different interpretation of maximum sustainable yield than is used here. Accurate monitoring of the sex ratio in the harvest, as well as the sex composition of the subpopulation and the litter production rate, are necessary to determine the extent to which reduction of male bears might affect the productivity of the BB and KB subpopulations. We suggest that it is important to investigate the analytical and biological reasons for lower estimates of *S** for male bears. Our finding that females comprise approximately 70% of independent bears in both subpopulations, despite harvest that may not be strongly selective for males, suggests that there is a biological basis for this finding, which could signal an emerging conservation concern and have demographic consequences not considered in our analyses.

We estimated stage-specific harvest vulnerability vectors for females and male using age data from the BB subpopulation (section Population initialization). Results suggested a slight preferential selection for juvenile bears of both sexes, compared to their representation in the subpopulation. For example, male two-year-olds (stage 7) were 25% more likely to be harvested compared to what would be expected based on their relative abundance. Strong selection against adult female bears with dependent young (stages 5 and 6) was consistent with the protection of family groups in BB (Chapter 8 in SWG 2016). All population projections assumed that harvest vulnerability will remain constant in the future. If this is not the case (e.g., if selection becomes stronger for adults and weaker for subadults), the harvest strategies that meet management objectives might change.

Baffin Bay

We suggest that demographic modeling results for both subpopulations should be interpreted within the context of other available information. Sea-ice habitat in the BB region significantly declined between the previous subpopulation assessment in the 1990s and the recent reassessment in the 2010s (Chapter 4 in SWG 2106). The length of summer (i.e., the number of days from sea-ice retreat in spring to sea-ice advance in fall) increased by 12 days/decade since 1979. The mean sea-ice concentration during June–October decreased by 4% per decade. The general pattern of melt occurs about 3–4 weeks earlier in the 2010s than in 1990s. In general, BB has incurred large changes in the sea-ice regime experienced by polar bears and this has resulted in habitat loss (Stern and Laidre 2016; SWG 2016), which has translated to biological changes in the subpopulation. BB bears used significantly lower sea-ice concentrations in winter and spring in the 2010s than the 1990s (SWG 2016). Adult female bears are significantly closer to land in all months than in the 1990s, except at the end of breakup (June–July) when they stay on offshore sea ice as long as possible, likely to maximize feeding. Arrival dates on Baffin Island in summer are one month earlier in the 2010s than in the 1990s, and therefore the amount of time bears spent on land has increased by 20–30 days since the 1990s. There is a significantly shorter maternity den duration in the 2010s and maternity dens occur at higher elevations and steeper slopes than maternity dens in the 1990s, likely due to reduced snow cover (Escajeda et al. 2017).

Body condition declined in BB between 1993 and 2013, and declines were in close association with the duration of the ice-free period and spring sea ice transition dates. Reproductive metrics indicate that, from 1993 to 2013, an annual index of C0 recruitment declined concurrent with a trend towards earlier spring sea-ice break-up (SWG 2016). There has also been a significant reduction in the size of the 2010s BB 95% kernel range (i.e., a measure of the area used by bears fitted with radiocollars) in all months and seasons compared to the range in the 1990s. The most marked reduction is a 60% decline in subpopulation range size in summer. With respect to movements across subpopulation boundaries, BB bears in the 2010s were significantly less likely to leave BB than in the 1990s. In particular, there was a reduction in the number of collared bears moving into Davis Strait and Lancaster Sound, apparently due to reduced winter sea-ice coverage. This suggests the BB subpopulation has become more discrete, with less exchange between it and other subpopulations.

For the BB subpopulation, litter production rate for females age ≥5 years (β_4) was estimated to be 0.93 (SE = 0.08) from field data collected 2011–2013. Taking into account statistical uncertainty, this value is similar to the values of 0.88 for bears age 5 years, and 1.00 for bears age ≥ 6 years, reported for BB by Taylor et al. (2005); and higher than the mean value of 0.80 for bears age ≥ 6 years across 11 other subpopulations (range = 0.44 to 0.98; Table S1 in Regehr et al. 2017). Relatively high litter production is consistent with our modeling assumption that, despite evidence for ecological change, the BB subpopulation is currently not experiencing strong density-dependent limitation in demographic parameters. We estimated β_4 directly from the sample of observed bears (section Methods), which was assumed to reflect the subpopulation because CR modeling did not identify differences in recapture probabilities (Chapter 5 in SWG 2016). However, during the autumn single adult females were more likely to be inland or at high elevations (SWG 2016), which could have led to heterogeneity in recapture probabilities that was not detected in the modeling process. If this was the case, single adult females could have been under-represented in the observation sample, which could lead to positive bias in estimates of β_4 because the number of single adult females appears in the denominator of the equation for litter production rate.

The three scenarios of the vital rates for BB corresponded to significant differences in subpopulation status (Table BB4) and therefore in harvest strategies. We placed less confidence in Scenario 1, because estimates of S^* for the period 2011–2103 were based primarily on three years of sampling, and bias in survival during the terminal years of a CR study is common when there is un-modeled heterogeneity in recapture probabilities or non-random temporary emigration from the sampling area (Peñaloza et al. 2014). Estimates of S^* for Scenario 2 were referenced to 1998–2010, a longer period that was bracketed by intensive sampling in the 1990s and 2010s, and throughout which research marks were returned in the harvest. Furthermore, population reconstruction suggested that a subpopulation with the vital rates from Scenario 2 could exhibit a stable trajectory over the period 1998–2010, when subject to the observed harvest of approximately 162 bears per year and observed variation in sea-ice conditions. We started the population reconstruction in 1998 at an assumed abundance of 2,826 (i.e., the estimate for the period 2011–2013), because SWG (2016) indicated that lower estimates of *N* from the 1990s included an unknown level of negative bias, and that trends in the size of the BB subpopulation could not be reliably determined. Our finding that the vital rates of Scenario 2 were capable of maintaining a stable subpopulation does not constitute evidence that this occurred. We did not complete a comprehensive suite of population reconstructions, and other combinations of factors (e.g., higher starting *N* and lower vital rates) might reproduce equally plausible histories. Nonetheless, the vital rates of Scenario 2 appear consistent with available information on the history of the BB subpopulation, to the extent this can be determined given uncertainties and potential biases in the data. Also, population reconstruction from 1998–2010 led to an ending proportion of females in the subpopulation that was similar to the value estimated from sex- and age-specific abundance estimates for the period 2011–2013.

For the reasons discussed above, we considered Scenario 2 to be the more likely representation of the current status of the BB subpopulation. Harvest strategies in Table BB6 that met Management Objective 2 are likely to satisfy the definition of sustainable harvest proposed by Regehr et al. (2017), which requires maintaining a subpopulation size above MNPL with respect to a changing *K*, and limiting the negative effects of harvest on persistence. For harvest strategies using the *status quo* value of SR = 1.25 and a 15-year management interval, the upper limits on present-day harvest rate ($h_{t=1}$) were 4.3 and 5.7% for “low” and “medium” risk tolerances, as stated by the JC. This corresponds to present-day harvest levels of up to 120 and 160 bears per year, respectively, which would be applied for a period of 15 years and then updated. This range encompasses current TAH of 132 for the BB subpopulation (SWG 2016). The sustainability of these harvest strategies is conditional on the input data and assumptions of our modeling approach, including (1) that Scenario 2, the most optimistic scenario of the vital rates, is an accurate representation of the current and future status of the BB subpopulation; and (2) adherence to a state-dependent management approach over the next 35 years, with a 15-year management interval and future subpopulation assessments that provide a level of precision similar to the 1998–2010 estimates of S^* (SWG 2016). The harvest strategy corresponding to “low” risk tolerance (i.e., $h_{t=1}$ = 4.3%) is associated with lower probabilities of male depletion and extirpation in later years of the projection. For the BB subpopulation, nearly all harvest strategies can be expected to require reductions in the harvest level over time, due primarily to declining *K*, but also potentially due to harvest if there are inaccuracies in the input data or our modeling approach. It is also possible that the harvest rate will decline over time due to Allee effects in the mating system or to density-independent reductions in *r* that were not considered in our analyses.

We included Scenario 3 of the vital rates, to provide a means of comparison with an "average" polar bear subpopulation. Table BB7 indicates that, using the *status quo SR* = 1.25 and a 15-year management interval, harvest strategies with $h_{t=1}$ = 3.6% could meet Management Objective 2 at the "medium" level of risk tolerance. This corresponds to a present-day harvest level of up to 100 bears per year. The upper limit on $h_{t=1}$ was also 3.6% for a harvest strategy with *SR* = 2, which is lower than the historic standard 4.5% harvest rate when using a 2:1 male-to-female sex ratio, for subpopulations experiencing positive environmental conditions (Taylor et al. 1987a). This difference is partially due to our inclusion of a declining trend in *K* for the BB subpopulation. It also suggests that our demographic modeling approach, when used in conjunction with Management Objective 2 and a "medium" risk tolerance as stated by the JC, may be slightly more conservative than previous predictive modeling for polar bears.

For the BB subpopulation, the challenges of meeting Management Objective 3, as it was stated by the JC, were presented above. To inform future discussion of subpopulation reduction, we identified a harvest strategy that resulted in a 25% reduction in starting subpopulation size over 15 years, while remaining with the stated risk tolerance for not exceeding a 30% reduction (Management strategy BB_S1 in Table S.BB5). This strategy required *SR* = 1, a 5-year management interval, and improved precision in the vital rates estimated from future subpopulation surveys. The starting harvest rate was 8.7%, corresponding to a present-day harvest level of 245 bears per year. It is unlikely that the near-optimal management conditions required by this strategy are feasible in practice, suggesting that either the management objectives or risk tolerances associated with a managed subpopulation reduction require reconsideration. Another practical challenge of managed reduction is that harvest must be rapidly reduced from very high levels in early years, to much lower levels once the target subpopulation size has been achieved. The risks of not reducing harvest in this manner were demonstrated by harvest strategy BB_S2, which maintained a fixed-level harvest of 245 bears per year for 15 years, without new subpopulation assessments or adjustments to the harvest. That strategy resulted high probabilities of extirpation (Table S.BB5), emphasizing the critical importance of monitoring and responsive management under aggressive harvest regimens.

Kane Basin

The annual cycle of sea-ice habitat in KB has shifted from a largely year-round ice platform (>30% coverage in summer) to a cycle that resembles the seasonal ice ecoregion (Amstrup et al. 2008) with complete melt-out in summer (<5% coverage; SWG 2016). The KB subpopulation has responded to changing sea-ice conditions with broad movement and habitat use patterns that are more similar to those of bears in seasonal sea-ice ecoregions (e.g., expanded seasonal home ranges). Apparent improvement in body condition in the 2010s, and no evidence of changes in reproductive performance in KB between the 1990s and 2010s, may reflect natural variation or a response to long-term changes in sea-ice dynamics in KB (SWG 2016). These observed changes reflect general differences in habitat use of bears occupying the archipelago vs. seasonal ice ecoregions: bears inhabiting seasonal ice regions have larger and more variable home ranges as they temporally track sea ice, whereas bears in archipelago regions have smaller home ranges with less variation.

For the KB subpopulation, litter production rate for females age ≥5 years (β_4) was estimated to be 0.71 (SE = 0.16) from field data collected 2012–2014. Considering statistical uncertainty, this suggests reproductive success similar to, or slightly lower than, other subpopulations (Table S1 in Regehr et al. 2017). The two scenarios of the vital rates for KB corresponded to significant differences in subpopulation status (Table KB3) and therefore in harvest strategies. Scenario 1 was characterized by low capacity for growth and high uncertainty ($\lambda = 0.01$ [SE = 0.04]), due largely to low and uncertain estimates of S^* for bears age ≤ 2 years (Table KB2). We placed less confidence in Scenario 1 because survival estimates of young bears were based on very small sample sizes during the period 2012–2014 (e.g., < 4 C0 marked per year). Furthermore, population reconstruction suggested that a subpopulation with vital rates from Scenario 1 would exhibit a declining trend from 1998–2014, when subject to the observed harvest of approximately 8 bears per year and an increasing trend in K. This is inconsistent with the estimated increase in abundance from 224 in the 1990s to 357 in the 2010s (Chapter 10 in SWG 2016), and with other evidence for productivity of the KB subpopulation (Chapters 9, 12, and 13 in SWG 2016).

Scenario 2 for KB included modified values of S^* for bears age ≤ 2 years, as necessary to reproduce the estimated subpopulation trend between the 1990s and 2010s, keeping other vital rates (e.g., adult survival) identical to Scenario 1. Vital rates in Scenario 2 retained the same amount of sampling variation as Scenario 1. Therefore, trajectories during population reconstruction were highly variable, corresponding to an 80% probability that the ending subpopulation size was at least one bear larger than the starting subpopulation size (i.e., that $N_{2014} > N_{1998}$). This stochastic representation of the history of the KB subpopulation was slightly more pessimistic (i.e., more likely to correspond to a declining subpopulation) compared to findings in SWG (2016), which suggested that the probability of a positive subpopulation change between the 1990s and 2010s was 95%. For Scenario 2, the modified estimates of S^* for bears age ≤ 2 years (Table KB2) were lower than the corresponding estimates for BB (noting that survival has a different time reference for the two subpopulation based on spring vs. autumn sampling), and within the range of juvenile survival estimates for other subpopulations with spring sampling (Table S1 in Regehr et al. 2017). The estimate of $\lambda = 1.05$ for Scenario 2 was equivalent to the mean estimate of unharvested population growth rate for other subpopulations (Regehr et al. 2017). Considering other lines of evidence for increasing productivity in the KB region, we suggest that Scenario 2 is a plausible representation of the current demographic status of this subpopulation.

For Scenario 2 of the vital rates and using the *status quo* value of $SR = 0.94$ and a 15-year management interval, the highest harvest strategy that met Management Objective 1 at "medium" risk tolerance corresponded to $h_{t=1} = 1.7\%$ and $H_{t=1} = 6$ bears per year. This finding is conditional on the input data and assumptions of the modeling approach, including (1) that Scenario 2 is an accurate representation of the KB subpopulation; (2) that K will remain stable for the next 35 years; and (3) adherence to a state-dependent management approach over the next 35 years, with a 15-year management interval and future subpopulation assessments that provide a level of precision similar to the estimates of S^* in SWG (2016). Sustainable harvest strategies for KB exhibited slight increases in mean harvest level over time (Table S.KB2), due in part to stability in K. If increasing biological productivity in the KB region leads to increasing trends over time in K and the intrinsic population growth rate, sustainable harvest levels would be expected to increase as well.

The relatively low estimate of sustainable harvest for KB was largely due to high uncertainty in vital rates, particularly estimates of S^* for bears age ≤ 2 years. This is evidenced by comparing a harvest level of up to 6 bears per year, calculated from the stochastic model projections (see above), with the estimated maximum sustainable yield of 13 bears per year based on asymptotic population dynamics (Table KB3). To investigate further, we performed two *post hoc* simulations with alternative assumptions for sampling uncertainty. The first simulation, which reduced uncertainty in estimates of S^* for bears ≤ 2 years to match the level of uncertainty in other survival estimates, resulted in a sustainable harvest of up to 2.8% (10 bears per year) under Management Objective 1 with "medium" risk tolerance, when using a 15-year management interval. The second simulation, which reallocated total uncertainty between sampling and process variation, resulted in a sustainable harvest of up to 2.2% (8 bears per year) under the same management conditions. These simulations were relevant because it is difficult to obtain precise and accurate estimates of vital rates for small and remote subpopulations such as KB. Without such estimates, the options available to managers include (1) inferring subpopulation status and sustainable harvest based on data other than the estimated vital rates (e.g., by modifying some estimates of S^* based on other information, similar to Scenario 2 and the *post hoc* simulations); or (2) adopting a conservative harvest strategy (e.g., Taylor et al. 2002).

Considering all available ecological and demographic data for the KB subpopulation, we suggest that present-day harvest rates up to approximately 2.8% (10 bears per year) are unlikely to cause negative population outcomes, if coupled with effective monitoring under a state-dependent approach. Use of a 10-year management interval would reduce the risks of harvest associated with high uncertainty in the currently available vital rates. If the challenges of studying the KB subpopulation lead to continued difficulty in obtaining accurate and precise estimates of vital rates, despite increased survey efforts, supplementary monitoring that is more frequent but less intensive may be valuable. We suggest developing a suite of ecological and demographic indicators to monitor subpopulation status, including accurate information on the level and composition of the harvest, marine productivity, habitat use and availability, reproductive rates, and estimates or indices of subpopulation size (via aerial survey or CR).

Monitoring

All of the harvest strategies considered in this report require the existence of a coupled research-management system under which both the sustainable harvest rate and the harvest level are adjusted periodically, based on new scientific information from subpopulation surveys and other sources. For both subpopulations, shorter management intervals and more precise estimates of N and vital rates, can substantially reduce the risk of negative population outcomes associated with a given harvest strategy. Results from the secondary simulations can help managers balance trade-offs between monitoring frequency and intensity (and therefore cost), the sustainable harvest rate, and harvest risks (Tables BB8 and KB6).

In our simulations, the management interval corresponded to the exact number of years between changes to the harvest level. For example, during population projections a 15-year management interval meant that new simulated population assessments were completed, and changes to the harvest level implemented, every 15 years. In practice, time lags in the coupled research-management system will likely result in departures from this simplified representation. For example, even if on-the-ground changes to TAH are implemented every 15 years, each change might be based on data from subpopulation surveys that were completed 2–3 years earlier. Application of the findings in this report should consider major differences, if they exist, between the definition of the management interval in practice and the definition used here.

Periodically obtaining new estimates of N and the vital rates (which determine r) is a central feature of a state-dependent management approach (Regehr et al. 2017). These parameters can be difficult and expensive to collect for wildlife populations (e.g., Williams et al. 2002), although both field methods (e.g., genetic CR; SWG 2016) and analytical approaches (e.g., Bayesian implementation of multistate models; Lunn et al. 2016) continue to evolve. Our analyses highlight the challenges of using estimated demographic parameters in harvest assessments for polar bears, even when the parameters were obtained from well-designed CR studies (SWG 2016). We sought to address these challenges primarily through consideration of multiple scenarios of the vital rates, which were developed based on (i) the estimated vital rates, referenced to different time periods; (ii) insights into the magnitude and directionality of potential bias (e.g., Schaub et al. 2004; Peñaloza et al. 2014); (iii) population reconstruction; and (iv) comparison with other case studies for polar bears. Although the magnitude of bias is generally lower in estimates of survival from CR studies compared to estimates of abundance

(Williams et al. 2002), the ramifications of relatively small bias in survival (e.g., 1–2%) can be profound when survival rates are used for population projections (e.g., Regehr et al. 2009). Skalski et al. (2012) recommend that biological realism should serve as a fundamental check for estimated demographic parameters and trends, and that auxiliary information should be used whenever possible to validate the results from predictive modeling. Peacock et al. (2011) recommend that management decisions for polar bears also include assessments of changes in body condition, habitat, population, and genetic delineation, and simultaneous surveys on Traditional Ecological Knowledge and human dimensions. Most of these subjects were successfully addressed during recent studies in BB and KB (SWG 2016). In this report, analyses relied heavily on estimates of subpopulation abundance and vital rates from SWG (2016), with limited interpretation of these estimates for the purpose of developing alternative scenarios and *post hoc* simulations.

Detailed recommendations on how to improve future estimates of N and r for the BB and KB subpopulations are beyond the scope of this report. Analyses to optimize the design of CR studies are being conducted under implementation of the Circumpolar Action Plan for polar bears (Polar Bear Range States 2015). For BB and KB, we suggest considering CR studies with a modified sampling scheme (e.g., sampling every other year for a period of six years, rather than sampling annually for three years), which could be combined with ongoing, less-intensive monitoring (see below). Also, we suggest that future study plans consider collecting and analyzing multiple types of data under the framework of an integrated population model (e.g., Frederiksen et al. 2014). Integrated population models can offer benefits for precision, accuracy, and the number of relevant parameters that can be estimated. For example, Regehr et al. (In preparation) concurrently analyzed radiotelemetry and CR data, which allowed direct estimation of temporary emigration and likely reduced bias in estimates of survival. Integrated population models can lead to increased consistency among demographic parameters (e.g., such that estimates of survival from studies of individually-marked animals, are consistent with estimates of subpopulation trend from aerial surveys). Finally, these models could allow integration of multiple types of research data with information from the harvest, local observations, and Traditional Ecological Knowledge. This has the potential to provide an improved assessment of overall subpopulation status that represents multiple perspectives.

Together with the conclusions and recommendations in SWG (2016), our analyses highlight important ecological and demographic indices for monitoring the BB and KB subpopulations. Accurate knowledge of the number, and sex and age composition, of human-caused removals is critical to understanding the effects of harvest on a subpopulation. Incomplete harvest reporting can lead to subpopulation depletion and other undesired outcomes, including negative bias in estimates of S^* that result in pessimistic population projections and lower estimates of sustainable harvest. We recommend that all harvested bears for the BB and KB subpopulations be genetically monitored, to detect recoveries of animals that were genetically marked during research. When used in CR models, accurate recovery data provide important information on survival and can reduce bias compared to studies with live recaptures only (Kendall et al. 2013). Furthermore, a systematic analysis of the sex and age composition of all harvest data (i.e., not only from research-marked bears) could provide complementary estimates of harvest rate and other demographic parameters (Skalski et al. 2005). Although such analyses can be limited by low statistical power and untestable assumptions, these problems are reduced if sample sizes are large (e.g., for the BB subpopulation) and recent data are available from a comprehensive subpopulation assessment. We suggest that systematic analysis of harvest data can be a useful monitoring tool between subpopulation surveys.

Nutritional condition and reproductive rates should be monitored as key indicators of subpopulation productivity (e.g., Vongraven et al. 2012). Changes in these parameters may precede or occur at the same time as other demographic changes (e.g., declining survival). Estimated relationships between time series of reproductive rates (or any other vital rate) and environmental conditions can be used in population projections to evaluate the future effects of habitat loss (e.g., Hunter et al. 2010). When such relationships are available, this represents an empirical alternative to our approach of projecting future trends in K that operated on vital rates through the density-dependent functions.

For a subpopulation that is harvested near maximum sustainable yield, and therefore in theory should function at a density well below carrying capacity (e.g., $N/K = 0.70$), declining nutritional condition and reproductive rates may signal negative density-independent effects. Density-independent limitation can result in lower values of r_{MNPL}, thus reducing the sustainable harvest rate that is calculated from equations 1 and 2. Under strong density-independent limitation, continued harvest without adjustment could lead to predominately additive mortality, with the potential to accelerate subpopulation declines compared to what would be expected under sea-ice loss only (USFWS 2016). Given that there is uncertainty about the extent to which sea-ice loss affects polar bears via density-dependent vs. density-independent mechanisms (e.g., Rode et al. 2012; Lunn et al. 2016), a high priority should be placed on monitoring spatial and temporal changes in habitat availability (e.g., as estimated from remote-sensing data of sea ice; Stern and Laidre 2016) along with indices of nutritional status and reproduction. SWG (2016) presented multiple reproductive indices for the BB and KB subpopulations, and we suggest that the number of yearlings per adult female may be particularly useful because it integrates litter production rate and juvenile survival (Rode et al. 2014; Regehr et al. 2015). Furthermore, concurrent monitoring of reproduction and the proportion of females in the subpopulation, is important to detect potential declines in subpopulation productivity due low male survival, skewed sex ratios in the subpopulation, and potential Allee effects in the mating system. Our analyses highlight these issues as potential conservation concerns for both the BB and KB subpopulations.

Acknowledgements

This report was prepared with contract support for EVR from the Greenland Institute of Natural Resources (Nuuk), through support by a grant from the Danish Cooperation For Environment In The Arctic (DANCEA). H. Stern (University of Washington) provided sea-ice data. M. Runge (U.S. Geological Survey) provided advice on technical details of the demographic model. A complete list of acknowledgements for the research conducted in BB and KB can be found in SWG (2016).

Abbreviations, Acronyms, Symbols, and Definitions

Adult – A polar bear age ≥ 5 years.

Allee effect – In this report, Allee effects refer to changes in reproductive rates due to density effects in the mating system. Declining reproductive rates can occur if adult males are depleted relative to adult females, or if overall subpopulation density is low during the spring on-ice breeding season.

Capture-recapture (CR) – A type of research study in which animals are individually marked, often through biopsy darting to obtain a genetic sample, or chemical immobilization to apply a physical mark (e.g., ear tags and lip tattoos). Over multiple years, data on individually marked animals can be used to estimate abundance and vital rates.

Carrying capacity (*K*) – The maximum number of individuals in a subpopulation that can be supported by the environment. This limit reflects the availability of food, habitat, and other resources. In this report, *K* is measured in the number of independent bears. Within the demographic model *K* is converted to metabolic energetic equivalents for the purpose of tracking subpopulation density over time.

Coefficient of variation (CV) – Ratio of the standard deviation to the mean of a statistical distribution of values. The CV reflects the level of uncertainty in an estimate, compared to the value of the estimate.

Confidence interval (CI) – A range of values that describes the uncertainty surrounding an estimate. Estimates of abundance and vital rates are often accompanied by a 95% CI.

Cub-of-the-year (C0) – A polar bear cub less than one year of age. In the polar bear life cycle it is assumed that C0 are born on 01 January of each year.

Density dependence – Demographic processes that change the birth or death rates as subpopulation density (i.e., the number of individuals per unit of habitat) changes.

Dependent young – A polar bear age ≤ 2 years that is accompanied by its mother.

Extirpation – The functional extinction of a subpopulation, which occurs in the population projections when a subpopulation size falls below the quasi-extinction threshold. In our analyses, extirpation is an irreversible condition that cannot be recovered from once the quasi-extinction threshold is crossed.

Harvest – In this report, harvest refers to all types of human-caused removals (i.e., subsistence harvest, sport hunting, removal of problem bears, defense kills, etc.).

Harvest level (*H*) – The number of independent bears removed each year through harvest.

Harvest rate (*h*) – Percentage of the total subpopulation size (i.e., the number of all bears, including dependent young) that is removed as independent bears each year through harvest.

Harvest strategy – A particular set of management and research conditions that define how harvest is conducted within the context of a state-dependent management approach. A harvest strategy is specified by inputs to equations 1 and 2, which determine the level and sex ratio of the harvest; as well as by the management interval and the level of precision in subpopulation data.

Independent bears – Polar bears age ≥ 2 years that are not with their mothers. Includes all polar bears in a subpopulation except for yearlings, cubs-of-the-year, and dependent two-year olds.

Intrinsic population growth rate (*r*) – The intrinsic population growth rate in the absence of human-caused removals. The maximum intrinsic growth rate ($\boldsymbol{r_{max}}$) occurs at a low density relative to carrying capacity. The intrinsic growth rate at a density referenced to maximum net productivity level is denoted $\boldsymbol{r_{MNPL}}$. Both r_{max} and r_{MNPL} are unharvested, potential growth rates that provide measures of the resilience of a subpopulation.

$\boldsymbol{\kappa}$ – A dimensionless metric representing proportional changes in carrying capacity (K), calculated from the number of ice-covered days per year. During projections, carrying capacity at year t, calculated as $K(t) = K(t = 1) * \kappa\ (t)$, operated on vital rates through the density-dependent relationships.

Litter production rate (β_4) – The proportion of adult females that are available to breed in year t, which produce a litter of cubs-of-the-year in year t+1.

Management interval (*mgmt.interval*) – Duration (in years) of the interval between successive changes to the harvest level based on new data from completed subpopulation surveys. For example, under a 10-year management interval, a harvest level would calculated in year $t = 1$ and then applied each year $t = 1, 2, ... 10$. During the later years of this period, a subpopulation survey would be completed to provide updated estimates of abundance and the vital rates. A new harvest level would be calculated using these data equations 1 and 2, and the new harvest level would be applied in each year $t = 11, 12, \dots 20$.

Management Objective – An overall goal for management of a subpopulation, as stated by the responsible management agencies. In this report, management objectives are presented as a desired population condition (e.g., maintaining a relatively stable subpopulation size) along with a risk tolerance for not meeting the population condition.

Maximum net productivity level (MNPL) – The subpopulation size that results in the greatest net annual increment in subpopulation numbers resulting from reproduction minus losses due to natural mortality. The value of MNPL depends on how density dependence operates in a subpopulation. Regehr et al. (2017) suggested that for polar bears MNPL occurs at approximately 70% of the maximum number of animals the environment can support on average (i.e., MNPL ≈ 0.70 × *K*).

Metabolic energetic equivalent value (*mee*) – The energetic requirements of an individual bear, expressed relative to the energetic requirements of an average adult female. Larger bears (e.g., adult males) have higher *mee* values that smaller bears (e.g., subadult females), and therefore occupy more "energetic space" and make a greater individual contribution to density effects.

$P_{extirpation}$ – The probability of extirpation for a subpopulation.

$P_{male.dep}$ – The probability of severe male depletion, defined as the number of adult males in a subpopulation (stage 10 in Figure 1) falling below 50% of the quasi-extinction threshold.

$P_{Objective}$ – The probability of meeting the population condition corresponding to a Management Objective as defined in Table BB1.

Population growth rate (λ) **–** The rate of change of subpopulation size, measured in numbers of individuals per unit time.

Population projection – A simulated process in which the matrix-based model is used to project the size and composition of a subpopulation forward over a certain number of annual time steps. Each projection was defined by a specific set of biological and management conditions.

Population reconstruction – In this report, population reconstruction refers to retrospective population projections that used historic biological and management conditions. Population reconstruction was used to explore the past performance of the BB and KB subpopulations.

Quasi-extinction threshold – The size below which a subpopulation is considered to be extirpated. Population viability analyses often use quasi-extinction thresholds that are larger than one animal, because at very low numbers there can be negative small-population dynamics that reduce viability and accelerate extirpation. In this report, the quasi-extinction thresholds were 100 and 25 independent bears for the BB and KB subpopulations, respectively.

Risk tolerance – The attitude toward risk of the responsible management agencies. In this report, risk tolerance is expressed as the required probability of meeting the population condition associated with a Management Objective (e.g., the required probability, as stated by managers, of maintaining subpopulation size above a desired level).

***rsd.mod* –** A modifier on the baseline relative standard deviation (SD) of the vital rates due to sampling uncertainty. Using different values of *rsd.mod* in population projections, allowed evaluation of the effects of different levels of precision in the data obtained from future subpopulation surveys. For example, a projection with *rsd.mod* = 0.5 meant that simulated population assessments would produce estimates of the vital rates and subpopulation size with approximately 50% less sampling variation, compared to the actual amount of sampling variation for the corresponding scenario of the vital rates.

Scenario of the vital rates – A specific set of vital rates assumed to represent the current status of a subpopulation. In this report, multiple scenarios of the vital rates were considered because of uncertainty and potential bias in estimates of certain demographic parameters from CR studies for both the BB and KB subpopulations (SWG 2016).

Stage – Stages in the life cycle graph representing bears of different sex, age, and reproductive status (Figure 1).

Standard deviation (SD) – A statistical measure that quantifies the amount of variation of a set of numbers around the mean (i.e., average) value. A low standard deviation means that most numbers are very close to the mean.

Standard error (SE) – A statistical measure that quantifies the amount of variation associated with an estimated parameter. The standard error is the standard deviation of a parameter's sampling distribution (i.e., its probability distribution, as estimated from a random sample of data).

Sex ratio (S*R*) – A factor that specifies the male-to-female ratio in the harvest. For example, *SR* = 2 is equivalent to a 2:1 male-to-female sex ratio.

Simulated population assessments – Simulated subpopulation surveys that were performed during population projections, on a schedule according to the management interval. The simulated population assessments provided updated estimates of subpopulation size and vital rates, which were used in equations 1 and 2 to calculate an updated harvest level, which was applied for the subsequent management interval.

State-dependent management – An approach under which management actions are based on the current state (status) of the subpopulation. In this report, state-dependent management refers to a coupled research-management system under which both the harvest rate and the harvest level are adjusted periodically, based on new scientific information from subpopulation surveys.

Subadult – Independent polar bear aged 2–4 years

Subpopulation – One of the 19 polar bear subpopulations recognized by the International Union for the Conservation of Nature (in the present case the Baffin Bay and Kane Basin subpopulations).

Subpopulation size (*N*) – The number of bears in a subpopulation.

***t* –** Annual time step in a subpopulation projection. Quantities labeled with a subscript *t* are referenced to a specific time step. For example, $H_{t=1}$ is a harvest level at year 15.

Vital rates – Demographic parameters such as reproductive rates and survival rates, which define transitions in the life cycle graph (Figure 1) and determine the composition and growth of a subpopulation.

Yearling (C1) – A polar bear cub between one and two years of age.

Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. *in* E. T. DeWeaver, C. M. Bitz, and L.-B. Tremblay, editors. Arctic Sea Ice Decline: Observations, Projections, Mechanisms, and Implications. American Geophysical Union, Washington, D.C., USA.

Amstrup, S. C., T. L. McDonald, and I. Stirling. 2001. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. Journal of Agricultural, Biological, and Environmental Statistics 6:221–234.

Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations,* 9 pp.

Atwood, T. C., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. F. Bromaghin. 2016. Forecasting the relative influence of environmental and anthropogenic stressors on polar bears. Ecosphere 7:1-22.

Boyce, M. S., C. V. Haridas, C. T. Lee, and N. S. Demography. 2006. Demography in an increasingly variable world. Trends in Ecology & Evolution 21:141–148.

Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634-651.

Caswell, H. 2001. Matrix Population Models, 2nd edition. Sinauer Associates Inc., Sunderland, Massachusetts, USA.

Christensen-Dalsgaard, S. N., J. Aars, M. Andersen, C. Lockyer, and N. G. Yoccoz. 2010. Accuracy and precision in estimation of age of Norwegian Arctic polar bears (*Ursus maritimus*) using dental cementum layers from known-age individuals. Polar Biology 33:589–597.

Cooch, E. G., and G. C. White, editors. 2016. Program MARK: A Gentle Introduction. 16th Edition. Available from: http://www.phidot.org/software/mark/docs/book/.

Derocher, A. E., J. Aars, S. C. Amstrup, A. Cutting, N. J. Lunn, P. K. Molnar, M. E. Obbard, I. Stirling, G. W. Thiemann, D. Vongraven, O. Wiig, and G. York. 2013. Rapid ecosystem change and polar bear conservation. Conservation Letters 6:368-375.

Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, O. Wiig, E. DeWeaver, M. C. Serreze, S. Belikov, M. Holland, J. A. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25-58.

Eberhardt, L. L. 2002. A paradigm for population analysis of long-lived vertebrates. Ecology 83:2841-2854.

Escajeda, E., K. L. Laidre, E. W. Born, Ø. Wiig, S. Atkinson, M. Dyck, S. H. Ferguson, N. J. Lunn. 2017. Identifying shifts in maternity den phenology and habitat characteristics of polar bears (*Ursus maritimus*) in Baffin Bay and Kane Basin. Polar Biology DOI: 10.1007/s00300-017-2172-6

Frederiksen, M., J.-D. Lebreton, R. Pradel, R. Choquet, and O. Gimenez. 2014. Identifying links between vital rates and environment: a toolbox for the applied ecologist. Journal of Applied Ecology 51:71-81.

Gelman, A., and J. Hill. 2007. Data analysis using regression and multilevel/hierarchical models. Cambridge University Press, New York, New York, USA.

Gerrodette, T., and D. P. Demaster. 1990. Quantitative determination of Optimum Sustainable Population level. Marine Mammal Science 6:1-16.

Guthery, F. S., and J. H. Shaw. 2013. Density dependence: applications in wildlife management. Journal of Wildlife Management 77:33–38.

Humbert, J.-Y., L. S. Mills, J. S. Horne, and B. Dennis. 2009. A better way to estimate population trends. Oikos 118:1940–1946.

Hunter, C. M., H. Caswell, M. C. Runge, S. C. Amstrup, E. V. Regehr, and I. Stirling. 2007. Polar bears in the southern Beaufort Sea II – Demography and population growth in relation to sea ice conditions. U.S. Geological Survey Administrative Report, U.S. Geological Survey Alaska Science Center, Anchorage, Alaska, USA.

Hunter, C. M., H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. Ecology 91:2883–2897.

IUCN Standards and Petitions Subcommittee. 2017. Guidelines for Using the IUCN Red List Categories and Criteria. Version 13. Prepared by the Standards and Petitions Subcommittee. Available from http://iucnredlist.org./documents/RedListGuidelines.pdf

JC [Canada-Greenland Joint Commission on Polar Bear]. 2010. Request for review of survey methods. Letter from the Canada-Greenland Joint Commission on Polar Bear to the Scientific Working Group to the Joint Commission on Polar Bear, 26 October 2010.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2016. Response to the SWG from the Joint Commission re: Harvest risk analysis. Email correspondence from C. Ladanowski, 7 October 2016.

JC [Canada-Greenland Joint Commission on Polar Bear]. 2017. Response to the SWG from the Joint Commission re: Harvest risk analysis. Email correspondence from C. Ladanowski, 20 February 2017.

Kendall, W. L., R. J. Barker, G. C. White, M. S. Lindberg, C. A. Langtimm, and C. L. Penaloza. 2013. Combining dead recovery, auxiliary observations and robust design data to estimate demographic parameters from marked individuals. Methods in Ecology and Evolution 4:828-835.

Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications 18:S97–S125.

Laidre, K. L., and E. V. Regehr. 2017. Arctic marine mammals and sea ice. Pages 516-533 *in* D. N. Thomas, editor. Sea Ice, 3rd Edition. Wiley-Blackwell, West Sussex, UK.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737.

Langtimm, C. A. 2009. Non-random temporary emigration and the robust design: conditions for bias at the end of a time series. Pages 745–761 *in* D. L. Thomson, E.G. Cooch, and M. J. Conroy, editors. Modeling Demographic Processes in Marked Populations. Environmental and Ecological Statistics, Volume 3. Springer Verlag, New York, New York, USA.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range: impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302-1320.

Lyons, J. E., M. C. Runge, H. P. Laskowski, and W. L. Kendall. 2008. Monitoring in the Context of Structured Decision-Making and Adaptive Management. Journal of Wildlife Management 72:1683-1692.

McCall, A. G., A. E. Derocher, and N. J. Lunn. 2015. Home range distribution of polar bears in western Hudson Bay. Polar Biology 38:343-355.

McLoughlin, P. D., M. K. Taylor, and F. Messier. 2005. Conservation risks of male-selective harvest for mammals with low reproductive potential. Journal of Wildlife Management 69:1592–1600.

Molnár, P. K., A. E. Derocher, M. A. Lewis, and M. K. Taylor. 2008. Modeling the mating system of polar bears: a mechanistic approach to the Allee effect. Proceedings of the Royal Society B: Biological Science 275:217–226.

Molnár, P. K., M. A. Lewis, and A. E. Derocher. 2014. Estimating allee dynamics before they can be observed: Polar bears as a case study. PLoS ONE 9:e85410. doi:10.1371/journal.pone.0085410.

Morris, W. F., and D. F. Doak. 2002. Quantitative Conservation Biology: Theory and Practice of Population Viability Analysis. Sinauer Associates Inc., Sunderland, Massachusetts, USA.

Mysterud, A. 2011. Selective harvesting of large mammals: how often does it result in directional selection? Journal of Applied Ecology 48:827-834.

Obbard, M. E., M. R. I. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15-32.

Peacock, E., A. E. Derocher, G. W. Thiemann, and I. Stirling. 2011. Conservation and management of Canada's polar bears (*Ursus maritimus*) in a changing Arctic. Canadian Journal of Zoology 89:371-385.

Peñaloza, C. L., W. L. Kendall, and C. A. Langtimm. 2014. Reducing bias in survival under nonrandom temporary emigration. Ecological Applications 24:1155-1166.

Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 Agreement on the Conservation of Polar Bears. Available from http://naalakkersuisut.gl/~/media/Nanoq/Files/Attached%20Files/Fiskeri_Fangst_Landbrug/Polarbear%202015/CAP/CAP%20Book.pdf

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:1-97.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world’s polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T.

D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

PBSG [IUCN/SSC Polar Bear Specialist Group]. 2017. Summary of polar bear population status per 2017. Published 30 March 2017. Available from http://pbsg.npolar.no/en/status/status-table.html

Regehr, E. V., M. Ben-David, S. C. Amstrup, G. M. Durner, and J. S. Horne. 2009. PhD Dissertation. Chapter 4. Quantifying bias in capture-recapture studies for mobile species: a case study with polar bears. University of Wyoming, Laramie, Wyoming USA.

Regehr, E.V., N. Hostetter, R. Wilson, K. Rode, A. Von Duyke, G. Pendleton, and S. Converse. (In preparation). Estimating demographic parameters for mobile species with sparse data: a Bayesian capture-recapture model for polar bears in the Chukchi Sea.

Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117-127.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673–2683.

Regehr, E. V., K. L. Laidre, H. R. Akçakaya, S. C. Amstrup, T. C. Atwood, N. J. Lunn, M. Obbard, H. Stern, G. W. Thiemann, and Ø. Wiig 2016. Conservation status of polar bears (*Ursus maritimus*) in relation to projected sea-ice declines. Biology Letters 12:20160556, doi:10.1098/rsbl.2016.0556.

Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.

Regehr, E. V., R. R. Wilson, K. D. Rode, M. C. Runge, and H. L. Stern. 2017. Harvesting wildlife affected by climate change: a modeling and management approach for polar bears. Journal of Applied Ecology doi:10.1111/1365-2664.12864.

Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768-782.

Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and O. Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3-18.

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76-88.

RStudio. 2016. RStudio: Integrated Development for R. RStudio, Inc., Boston, Massachusetts, USA. Available from http://www.rstudio.com/

Schaub, M., O. Gimenez, B. R. Schmidt, and R. Pradel. 2004. Estimating survival and temporary emigration in the multistate capture-recapture framework. Ecology 85:2107-2113.

Skalski, J. R., M. V. Clawson, and J. J. Millspaugh. 2012. Model evaluation in statistical population reconstruction. Wildlife Biology 18:225-234.

Skalski, J. R., K. E. Ryding, and J. J. Millspaugh. 2005. Wildlife Demography: Analysis of Sex, Age, and Count Data. Elsevier Academic Press, San Diego, California, USA. 636 pp.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere 10:2027–2041.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2010. Report of the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear, 28-31 May 2010, Ilulissat, Greenland, 18 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2016. Re-Assessment of the Baffin Bay and Kane Basin Polar Bear Subpopulations: Final Report to the Canada-Greenland Joint Commission on Polar Bear. 31 July 2016: x + 636 pp.

Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294-306.

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987a. Modelling the sustainable harvest of female polar bears. Journal of Wildlife Management 51:811–820.

Taylor, M., F. Bunnell, D. DeMaster, R. Schweinsburg, and J. Smith. 1987b. ANURSUS: a population analysis system for polar bears (*Ursus maritimus*). International Conference on Bear Research and Management 7:117-125.

Taylor, M. K., J. Laake, H. D. Cluff, M. Ramsay, and F. Messier. 2002. Managing the risk from hunting for the Viscount Melville Sound polar bear population. Ursus 13:185–202.

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. Polar Biology 31:491–499.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. Journal of Wildlife Management 70:1667-1673.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. Wildlife Biology 14:52–60.

Taylor, M.K., Obbard, M., Pond, B., Kuc, M., and Abraham, D. 2001. RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations. Iqaluit: Government of Nunavut.

Taylor, M., M. Obbard, B. Pond, M. Kuc, and D. Abraham. 2006. A guide to using RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations, Version 1.9.003. The Queen’s Printer for Ontario. 58 pp.

USFWS [United States Fish and Wildlife Service]. 2016. Polar Bear (*Ursus maritimus*) Conservation Management Plan, Final. U.S. Fish and Wildlife, Region 7, Anchorage, Alaska, USA, 104 pp.

Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Vongraven, D., and E. Peacock. 2011. Development of a pan-Arctic monitoring plan for polar bears: background paper. Circumpolar Biodiversity Monitoring Programme, CAFF Monitoring Series Report No. 1, CAFF International Secretariat, Akureyri, Iceland.

White, G. C. 2000. Population viability analysis: data requirements and essential analyses. Pages 288-331 *in* L. Boitani, and T. K. Fuller, editors. Research Techniques in Animal Ecology: Controversies and Consequences. Columbia University Press, New York, New York, USA.

Wielgus, R. B. 2002. Minimum viable population and reserve sizes for naturally regulated grizzly bears in British Columbia. Biological Conservation 106:381-388.

Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

Wilder, J. M., D. Vongraven, T. Atwood, B. Hansen, A. Jessen, A. Kochnev, G. Yowrk, R. Vallender, D. Hedman, and M. Gibbons. 2017. Polar bear attaches on humans: implications of a changing climate. Wildlife Society Bulletin: DOI: 10.1002/wsb.783.

Williams, B. K., M. J. Conroy, and J. D. Nichols, editors. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA. 817 pp.

Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. Ecosphere 5:1-24.

Zabel, R. W., M. D. Scheuerell, M. M. Mcclure, and J. G. Williams. 2006. The Interplay Between Climate Variability and Density Dependence in the Population Viability of Chinook Salmon. Conservation Biology 20:190-200.

Tables and Figures

Table BB1. Potential Management Objectives for the Baffin Bay (1-3) and Kane Basin (1-2) polar bear subpopulations. Population size (*N*) and carrying capacity (*K*) are measured in the number of independent bears and referenced to an annual time step (*t*) during population projections.

Management Objective	Population condition	Required probability of meeting objective
1a	$N_{t=36} > (0.9 \times N_{t=1})$	≥ 0.90
1b	$N_{t=36} > (0.9 \times N_{t=1})$	≥ 0.70
2a	$N_{t=36} > (0.7 \times K_{t=36})$	≥ 0.90
2b	$N_{t=36} > (0.7 \times K_{t=36})$	≥ 0.70
3a	$N_{t=15} > (0.7 \times N_{t=1})$	≥ 0.90
3b	$N_{t=15} > (0.7 \times N_{t=1})$	≥ 0.30

Table BB2. Reproductive parameters for the Baffin Bay polar bear subpopulation estimated from field data collected 2011-2013. Dependent young are cubs-of-the-year (C0) and yearlings (C1). Adult females are ≥ 5 years. The mean and standard error (SE) we calculated using simulations methods described in the main text.

Parameter	**Mean**	**SE**
Litter production rate for adult females (β_4)	0.93	0.08
C0 per adult female	0.58	0.04
Proportion of adult females with C0	0.38	0.02
C0 litter size	1.55	0.04
C1 per adult female	0.35	0.03
Proportion of adult females with C1	0.24	0.02
C1 litter size	1.47	0.05

Table BB3. Estimates (mean and standard error [SE]) of unharvested survival (*S**) for three scenarios of the vital rates for the Baffin Bay polar bear subpopulation. The scenarios are described in the main text.

†The life cycle graph (Figure 1) does not include separate stages for cubs-of-the-year (C0) and yearlings (C1), but survival rates for these age classes contribute to transition probabilities between reproductive stages for adult females.

			Scenario 1		Scenario 2		Scenario 3	
Sex	**Age class**	**Stage**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
female	C0	†	0.88	0.06	0.88	0.06	0.88	0.06
female	C1	†	0.89	0.06	0.89	0.06	0.89	0.06
female	2-4 year	1-3	0.91	0.05	0.96	0.02	0.93	0.05
female	≥5 year	4-6	0.91	0.05	0.96	0.02	0.93	0.05
male	C0	†	0.88	0.06	0.88	0.06	0.88	0.06
male	C1	†	0.89	0.06	0.89	0.06	0.89	0.06
male	2-4 year	7-9	0.83	0.06	0.91	0.02	0.92	0.06
male	≥5 year	10	0.83	0.06	0.91	0.02	0.92	0.06

Table BB4. Demographic parameters (mean and standard error [SE]) for the Baffin Bay polar bear subpopulation, corresponding to the three scenarios of the vital rates, based on asymptotic population dynamics. The parameters are: unharvested population growth rate (λ); subpopulation density (i.e., N/K) corresponding to maximum net productivity level (MNPL); intrinsic population growth rate at MNPL (r_{MNPL}); intrinsic population growth rate at low population density (r_{max}); and maximum sustainable yield (MSY) measured in numbers of independent bears under non-selective harvest.

	Scenario 1		**Scenario 2**		**Scenario 3**	
Sex	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
λ	1.03	0.05	1.08	0.04	1.05	0.03
MNPL	0.72	0.04	0.67	0.03	0.70	0.03
r_{MNPL}	0.03	0.06	0.08	0.03	0.05	0.03
r_{max}	0.03	0.06	0.10	0.04	0.06	0.04
MSY	49	67	156	50	100	59

Table BB5. Summary of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 1 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 1.0**			***SR* = 1.25**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
2a	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
2b	0.41	20	0.7%	0.36	20	0.7%	0.27	20	0.7%
3a	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
3b	1.22	60	2.1%	1.08	60	2.1%	1.08	80	2.8%

Table BB6. Summary of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 1.0**			***SR* = 1.25**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
2a	0.78	120	4.3%	0.69	120	4.3%	0.43	100	3.6%
2b	1.03	160	5.7%	0.92	160	5.7%	0.60	140	5.0%
3a	1.03	160	5.7%	0.92	160	5.7%	0.60	140	5.0%
3b	1.16	180	6.4%	1.15	200	7.1%	0.78	180	6.4%

Table BB7. Summary of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 3 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 1.0**			***SR* = 1.25**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.00	0	0.0%	0.00	0	0.0%	0.00	0	0.0%
2a	0.60	60	2.1%	0.53	60	2.1%	0.53	80	2.8%
2b	0.80	80	2.8%	0.89	100	3.6%	0.66	100	3.6%
3a	1.00	100	3.6%	0.71	80	2.8%	0.66	100	3.6%
3b	1.40	140	5.0%	1.24	140	5.0%	0.93	140	5.0%

Table BB8. Summary of secondary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Rows are different levels of *rsd.mod*, a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty. Columns are different management intervals. Values in the cells represent the upper limits that meet Management Objective 2b (Table BB1); where F_O is a factor to calculate the harvest rate, and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with SR = 1.25.

	Management interval (years)		
rsd.mod	**10**	**15**	**20**
	(Results reported as values of F_O)		
0.5	1.15	0.98	0.86
1.0	1.03	0.92	0.80
1.5	0.92	0.80	0.75
	(Results reported as values of $h_{t=1}$)		
0.5	7.1%	6.0%	5.3%
1.0	6.4%	5.7%	5.0%
1.5	5.7%	5.0%	4.6%

Table KB1. Reproductive parameters for the Kane Basin polar bear subpopulation estimated from field data collected 2012-2014. Dependent young are cubs-of-the-year (C0) and yearlings (C1). Adult females are ≥ 5 years. The mean and standard error (SE) we calculated using simulations methods described in the main text.

Parameter	**Mean**	**SE**
Litter production rate for adult females (β_4)	0.71	0.16
C0 per adult female	0.55	0.10
Proportion of adult females with C0	0.34	0.06
C0 litter size	1.64	0.10
C1 per adult female	0.22	0.06
Proportion of adult females with C1	0.17	0.04
C1 litter size	1.23	0.12

Table KB2. Estimates (mean and standard error [SE]) of unharvested survival (*S**) for two scenarios of the vital rates for the Kane Basin polar bear subpopulation. The scenarios are described in the main text.

†The life cycle graph (Figure 1) does not include separate stages for cubs-of-the-year (C0) and yearlings (C1), but survival rates for these age classes contribute to transition probabilities between reproductive stages for adult females.

			Scenario 1		**Scenario 2**	
Sex	**Age class**	**Stage**	**Mean**	**SE**	**Mean**	**SE**
female	C0	†	0.45	0.15	0.74	0.25
female	C1	†	0.74	0.15	0.87	0.15
female	2 year	1	0.74	0.15	0.87	0.15
female	3 year	2	0.97	0.04	0.97	0.04
female	4 year	3	0.97	0.04	0.97	0.04
female	≥5 year	4-6	0.97	0.04	0.97	0.04
male	C0	†	0.45	0.15	0.70	0.23
male	C1	†	0.54	0.17	0.74	0.23
male	2 year	7	0.54	0.17	0.74	0.23
male	3 year	8	0.90	0.06	0.90	0.06
male	4 year	9	0.90	0.06	0.90	0.06
male	≥5 year	10	0.90	0.06	0.90	0.06

Table KB3. Demographic parameters (mean and standard error [SE]) for the Kane Basin polar bear subpopulation, corresponding to the two scenarios of the vital rates, based on asymptotic population dynamics. The parameters are: unharvested population growth rate (λ); subpopulation density (i.e., *N*/*K*) corresponding to maximum net productivity level (MNPL); intrinsic population growth rate at MNPL (r_{MNPL}); intrinsic population growth rate at low population density (r_{max}); intrinsic population growth rate at low population density in the absence of human-caused removals (r_{max}); and maximum sustainable yield (MSY) measured in numbers of independent bears under non-selective harvest.

	Scenario 1		**Scenario 2**	
Sex	**Mean**	**SE**	**Mean**	**SE**
λ	1.01	0.04	1.05	0.06
MNPL	0.73	0.05	0.69	0.04
r_{MNPL}	0.01	0.04	0.05	0.06
r_{max}	0.02	0.05	0.07	0.06
MSY	3	6	13	13

Table KB4. Summary of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 1 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 0.94**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.00	0	0.0%	*NA*	*NA*	*NA*
2a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
2b	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*

Table KB5. Summary of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. For each row in the table, parameters are shown for the maximum starting (i.e., $t = 1$) harvest strategy that meets the corresponding Management Objective (Table BB1). F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with a 15-year management interval and baseline data precision (i.e., *rsd.mod* = 1). *NA* indicates that a Management Objective was not met with no harvest.

Management Objective	***SR* = 0.94**			***SR* = 2.0**		
	F_O	$H_{t=1}$	$h_{t=1}$	F_O	$H_{t=1}$	$h_{t=1}$
1a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
1b	0.48	6	1.7%	0.31	6	1.7%
2a	*NA*	*NA*	*NA*	*NA*	*NA*	*NA*
2b	0.32	4	1.1%	0.21	4	1.1%

Table KB6. Summary of secondary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Rows are different levels of *rsd.mod*, a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty. Columns are different management intervals. Values in the cells represent the upper limits that meet Management Objective 1b (Table BB1); where F_O is a factor to calculate the harvest rate, and $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears. All simulations followed a state-dependent management approach with SR = 0.94.

	Management interval (years)		
rsd.mod	**10**	**15**	**20**
	(Results reported as values of F_O)		
0.5	0.64	0.64	0.56
1.0	0.56	0.48	0.48
1.5	0.56	0.40	0.40
	(Results reported as values of $h_{t=1}$)		
0.5	2.2%	2.2%	2.0%
1.0	2.0%	1.7%	1.7%
1.5	2.0%	1.4%	1.4%

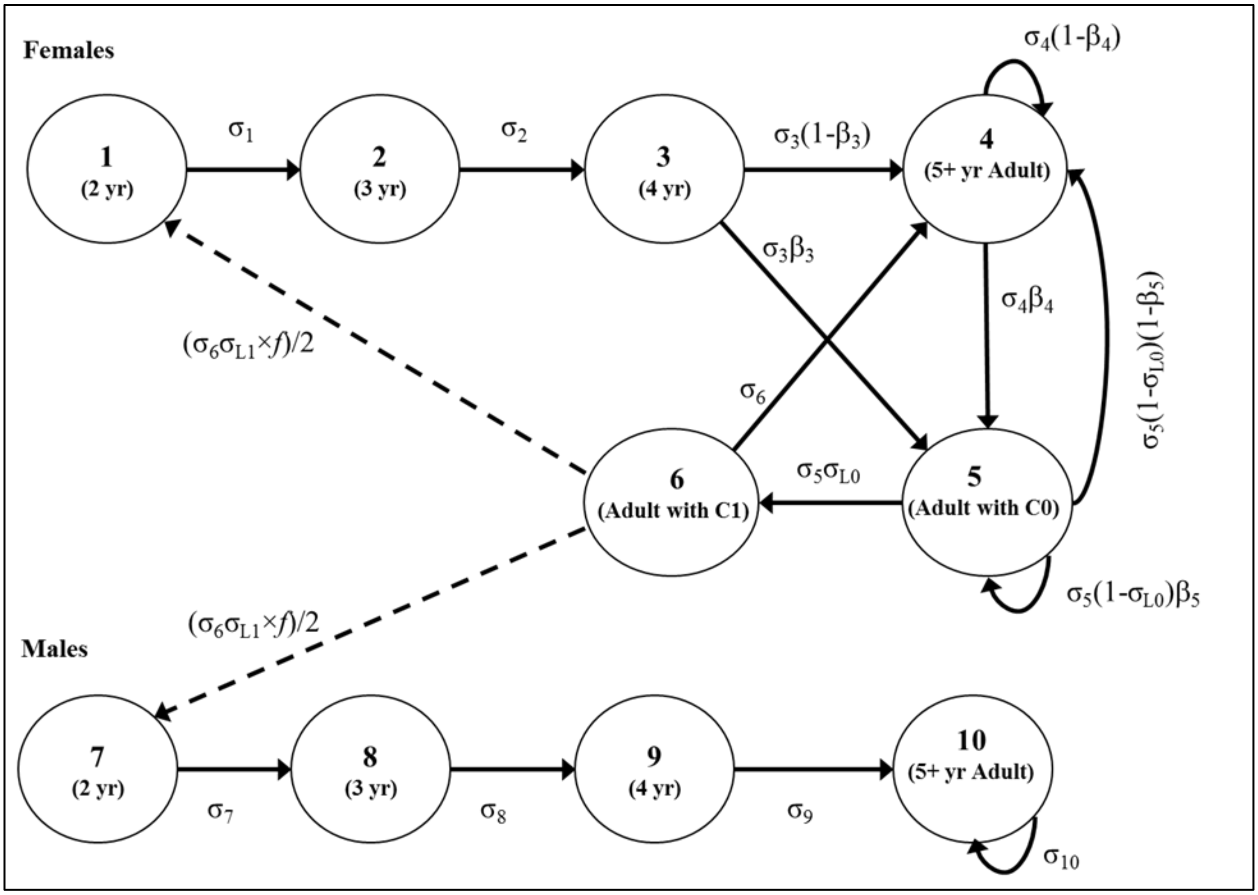


Figure BB1. The polar bear life cycle graph underlying the matrix-based projection model, reproduced from Figure 1 in Regehr et al. (2017). Stages 1–6 are females and stages 7–10 are males; σ_i is the annual probability of survival of an individual in stage *i*, σ_{L0} and σ_{L1} are the probabilities of at least one member of a cub-of-the-year (C0) or yearling (C1) litter surviving, *f* is the expected size of C1 litters that survive to 2 years, and β_i is the probability, conditional on survival, of an individual in stage *i* breeding, thereby producing a C0 litter with at least one member surviving. Solid lines are stage transitions and dashed lines are reproductive contributions.

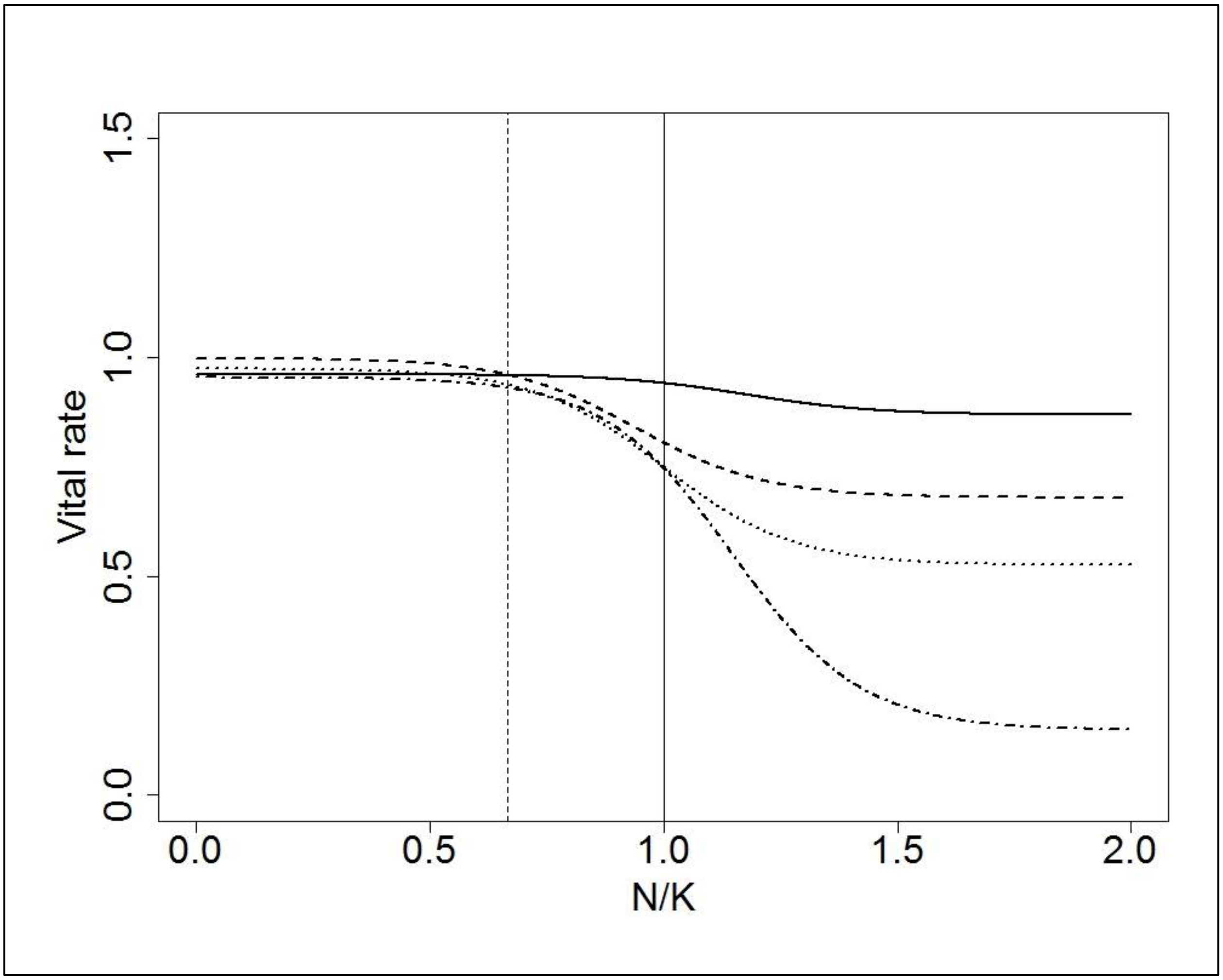


Figure BB2. Sample density-dependent curves of the vital rates for the Baffin Bay polar bear subpopulation for Scenario 2. Vital rates shown are survival probability for stage 4 (σ_{4}, solid line), survival probability for stage 1 (σ_{1}, dashed line), survival probability for cub-of-the-year litters (σ_{L0}, dotted line), and breeding probability for stage 4 (β_{4}, dash-dot line). Density on the x-axis is expressed as the ratio of population size (*N*) to carrying capacity (*K*). The solid vertical line corresponds to *N*/*K* = 1 at carrying capacity. The vital rates at this density would result in a stable subpopulation (i.e., intrinsic population growth rate [*r*] = 0) assuming asymptotic dynamics. The dashed vertical line corresponds to maximum net productivity level (MNPL). The vital rates at a subpopulation size equivalent to MNPL are the mean parameter values for Scenario 2 (Tables BB2 and BB3).

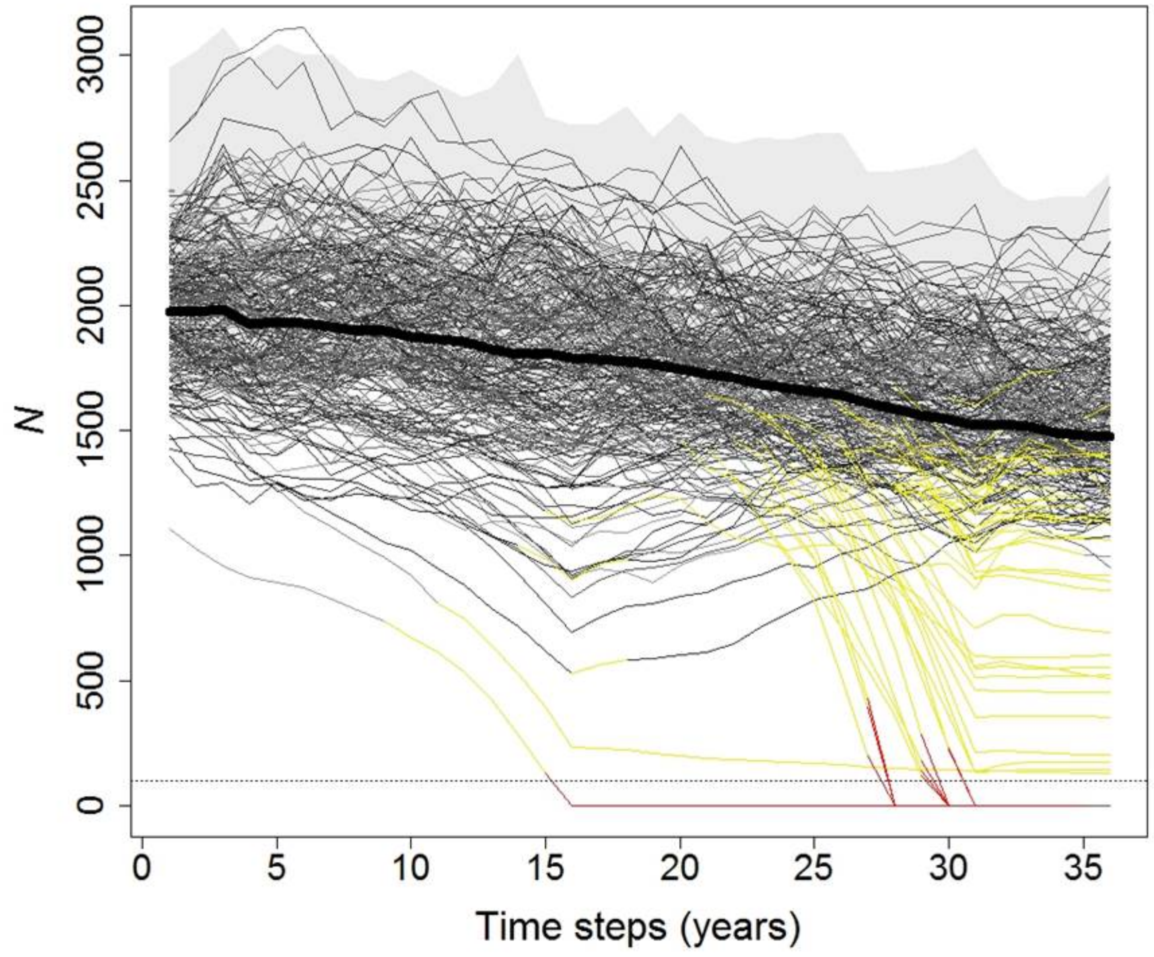


Figure BB3. Sample replicates (black lines) from population projections for the Baffin Bay subpopulation, using vital rates from Scenario 2. The grey shaded area in the background represents the upper 95% confidence interval for carrying capacity, measured in number of bears, which declined at a rate of approximately 5.5% per decade. The y-axis *N* is subpopulation size referenced to independent bears, and the heavy black line is median subpopulation size. Replicates are shaded yellow and red for time steps at which they experienced male depletion or extirpation, respectively. Projections are for a harvest strategy with F_0 = 0.92, *SR* = 1.25, a 15-year management interval, and *rsd.mod* = 1.0 (management inputs are defined in the main text). This harvest strategy equates to a starting (i.e., *t* = 1) harvest level of 160 bears per year.

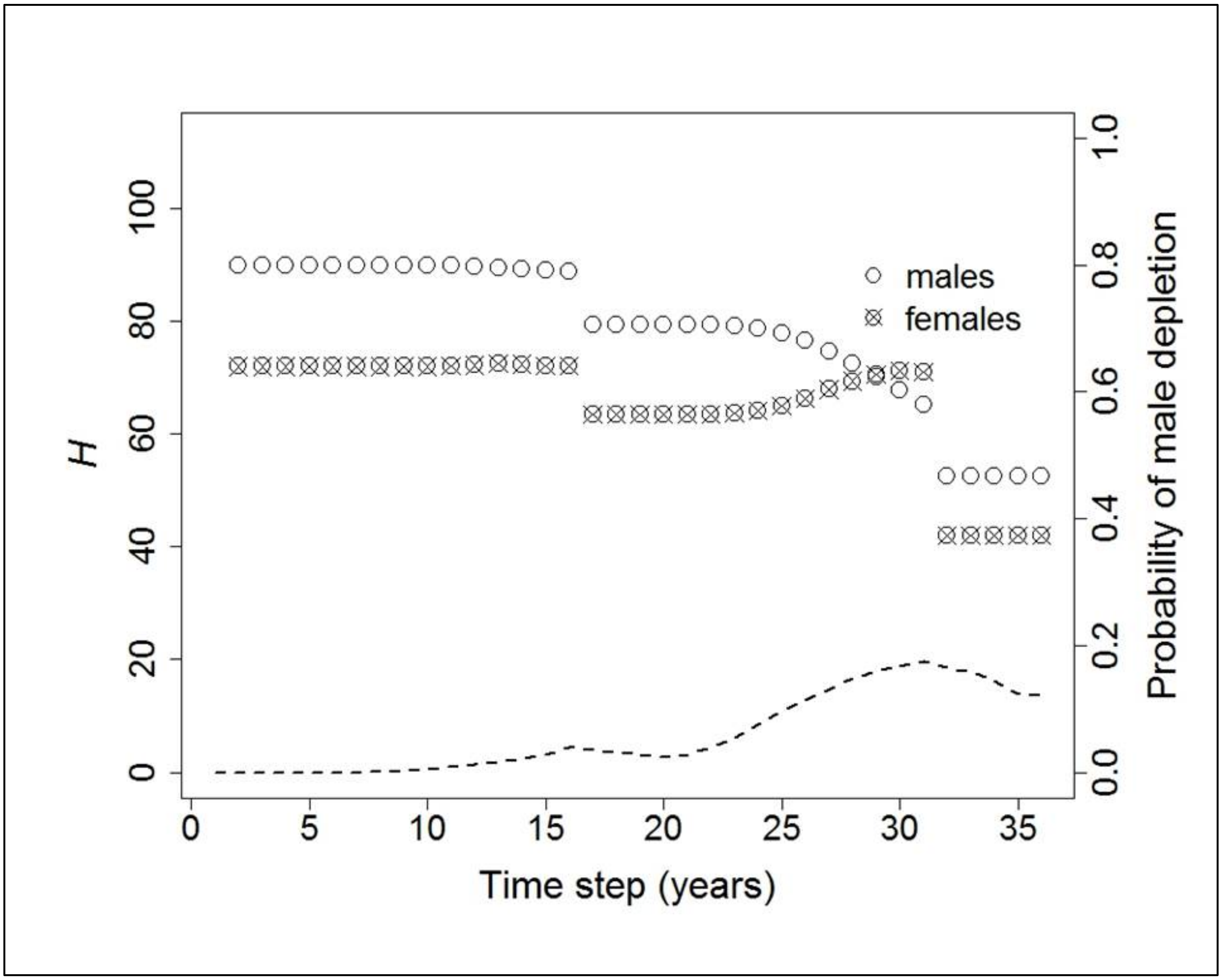


Figure BB4. Example results from population projections for the Baffin Bay polar bear subpopulation, using vital rates from Scenario 2. The left y-axis *H* is the harvest level, with the circles and cross-circles representing the average number of independent male and female polar bear removed per year, respectively, under a state-dependent management approach. The right y-axis is the probability of severe male depletion, values of which are plotted as the dashed line. Projections are for a harvest strategy with F_0 = 0.92, *SR* = 1.25, a 15-year management interval, and *rsd.mod* = 1.0 (i.e., the same harvest strategy as Figure BB3; management inputs are defined in the main text). This harvest strategy equates to a starting (i.e., *t* = 1) harvest level of 160 bears per year.

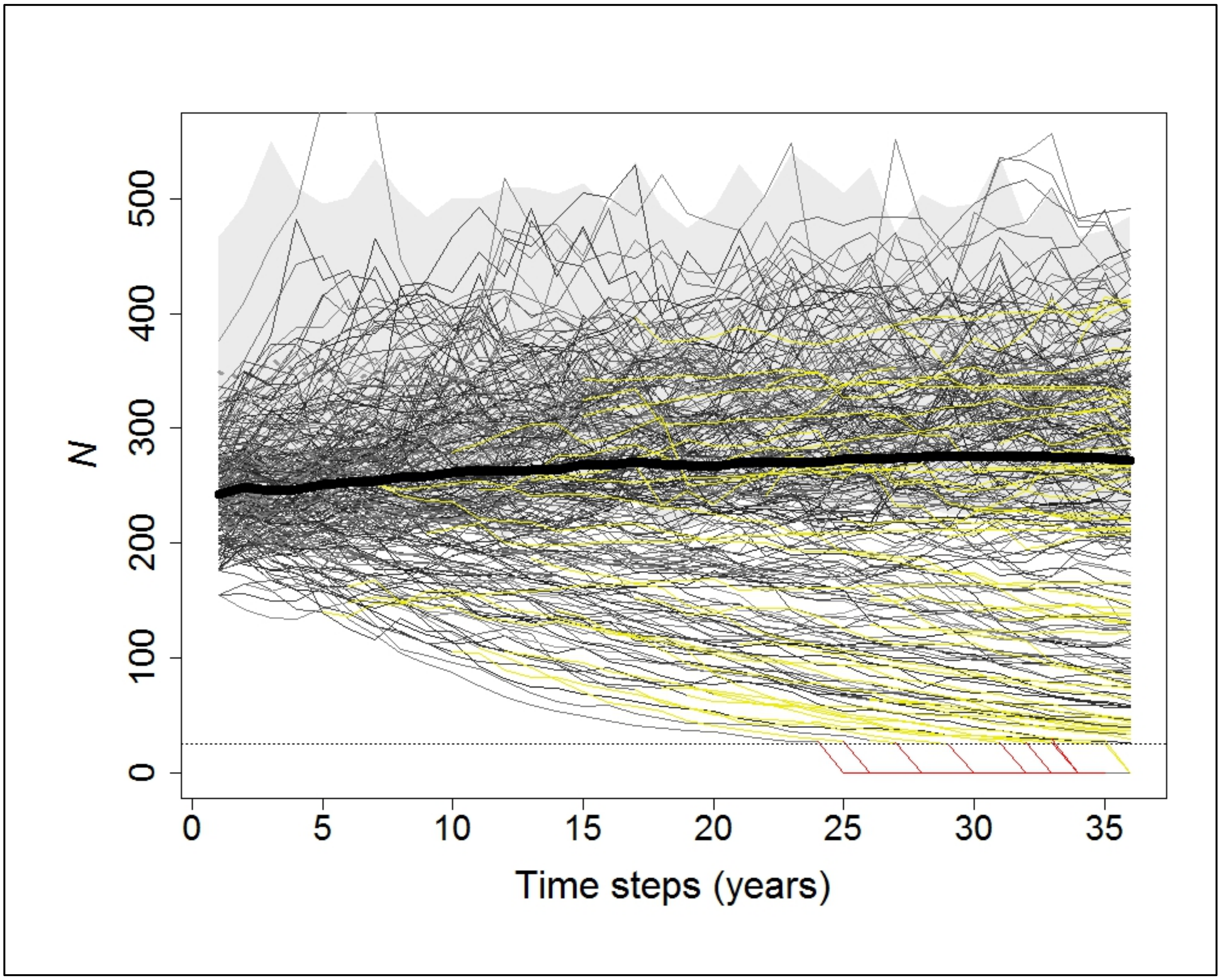


Figure KB1. Sample replicates (black lines) from population projections for the Kane Basin subpopulation, using vital rates from Scenario 1. The grey shaded area in the background represents the upper 95% confidence interval for carrying capacity, measured in number of bears, which declined at a rate of approximately 5.5% per decade. The y-axis *N* is subpopulation size referenced to independent bears, and the heavy black line is median subpopulation size. Replicates are shaded yellow and red for time steps at which they experienced severe male depletion or extirpation, respectively. Projections are for a subpopulation with no harvest.

Appendix S1.

Table S.BB1. Detailed results of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 1 of the vital rates. Each column corresponds to one simulation, with results reported at time steps $t = 15$ and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; N is population size referenced to independent bears; and K is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

	***t* = 1**					***t* = 1**					***t* = 1**				
F_O	0.00	0.41	0.81	1.22	1.63	0.00	0.36	0.72	1.08	1.45	0.00	0.27	0.54	0.81	1.08
$h_{t=1}$	0.0%	0.7%	1.4%	2.1%	2.8%	0.0%	0.7%	1.4%	2.1%	2.8%	0.0%	0.7%	1.4%	2.1%	2.8%
$H_{t=1}$	0	20	40	60	80	0	20	40	60	80	0	20	40	60	80
SR	1.00	1.00	1.00	1.00	1.00	1.25	1.25	1.25	1.25	1.25	2.00	2.00	2.00	2.00	2.00
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	***t* = 15**					***t* = 15**					***t* = 15**				
N_t/N_1	1.05	1.05	0.95	0.90	0.89	1.08	1.03	0.99	0.93	0.87	1.07	1.04	0.98	0.97	0.90
N_t/K_t	0.94	0.91	0.85	0.80	0.78	0.95	0.91	0.87	0.82	0.76	0.94	0.93	0.87	0.86	0.81
H_t	0	20	40	59	77	0	20	40	58	77	0	20	39	59	78
$P_{extirpation}$	0.00	0.00	0.01	0.03	0.04	0.00	0.00	0.00	0.02	0.04	0.00	0.00	0.01	0.01	0.02
$P_{male.dep}$	0.00	0.00	0.01	0.04	0.05	0.00	0.00	0.00	0.04	0.07	0.00	0.01	0.01	0.06	0.11
$P_{Objective1}$	0.78	0.73	0.66	0.58	0.53	0.81	0.78	0.69	0.60	0.51	0.81	0.79	0.69	0.68	0.57
$P_{Objective2}$	0.83	0.79	0.72	0.67	0.63	0.86	0.82	0.77	0.69	0.60	0.87	0.82	0.76	0.76	0.67
$P_{Objective3}$	0.86	0.83	0.77	0.71	0.69	0.89	0.86	0.82	0.74	0.66	0.91	0.85	0.80	0.81	0.73

Table S.BB1. Continued.

			***t* = 36**					***t* = 36**					***t* = 36**		
N_t/N_1	0.88	0.83	0.65	0.42	0.37	0.90	0.82	0.69	0.45	0.30	0.89	0.83	0.63	0.39	0.25
N_t/K_t	0.93	0.85	0.68	0.45	0.38	0.94	0.87	0.72	0.47	0.32	0.94	0.88	0.66	0.41	0.27
H_t	0	27	39	41	46	0	28	39	40	38	0	27	31	35	33
$P_{extirpation}$	0.03	0.05	0.07	0.21	0.22	0.02	0.03	0.06	0.20	0.26	0.03	0.04	0.06	0.25	0.33
$P_{male.dep}$	0.01	0.04	0.11	0.18	0.20	0.02	0.03	0.12	0.20	0.22	0.01	0.05	0.23	0.26	0.25
$P_{Objective1}$	0.53	0.30	0.13	0.07	0.05	0.55	0.34	0.15	0.08	0.05	0.55	0.37	0.15	0.09	0.07
$P_{Objective2}$	0.78	0.72	0.49	0.31	0.26	0.81	0.77	0.53	0.33	0.24	0.81	0.77	0.48	0.33	0.26
$P_{Objective3}$	0.78	0.71	0.47	0.28	0.23	0.80	0.76	0.50	0.29	0.21	0.80	0.76	0.45	0.29	0.22

Table S.BB2. Detailed results of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; N is population size referenced to independent bears; and K is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

(a) male-to-female sex ratio in harvest (*SR*) = 1.0

						***t* = 1**					
F_O	0.000	0.516	0.646	0.775	0.904	1.033	1.162	1.291	1.420	1.549	1.679
$h_{t=1}$	0.0%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%	7.1%	7.8%	8.5%	9.2%
$H_{t=1}$	0	80	100	120	140	160	180	200	220	240	260
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
						***t* = 15**					
N_t/N_1	1.14	1.03	1.01	0.97	0.95	0.87	0.85	0.77	0.73	0.57	0.47
N_t/K_t	1.00	0.91	0.89	0.86	0.83	0.78	0.75	0.68	0.64	0.51	0.41
H_t	0	82	102	122	142	162	180	198	220	220	225
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.04	0.03	0.16	0.21
$P_{male.dep}$	0.00	0.00	0.00	0.00	0.00	0.01	0.03	0.10	0.12	0.21	0.26
$P_{Objective1}$	1.00	0.98	0.92	0.83	0.68	0.45	0.31	0.17	0.09	0.04	0.03
$P_{Objective2}$	1.00	1.00	0.99	0.95	0.91	0.79	0.66	0.45	0.34	0.19	0.14
$P_{Objective3}$	1.00	1.00	1.00	0.97	0.96	0.90	0.80	0.62	0.55	0.36	0.27
						***t* = 36**					
N_t/N_1	0.96	0.86	0.84	0.80	0.78	0.72	0.69	0.62	0.57	0.39	0.20
N_t/K_t	1.00	0.90	0.88	0.85	0.81	0.77	0.72	0.65	0.59	0.41	0.21
H_t	0	72	86	97	104	105	107	105	108	94	90
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.01	0.03	0.07	0.14	0.18	0.35	0.43
$P_{male.dep}$	0.00	0.00	0.00	0.00	0.02	0.04	0.05	0.08	0.11	0.13	0.13
$P_{Objective1}$	0.75	0.32	0.22	0.13	0.08	0.05	0.03	0.02	0.02	0.01	0.00
$P_{Objective2}$	1.00	1.00	0.99	0.95	0.84	0.71	0.55	0.40	0.30	0.18	0.11
$P_{Objective3}$	1.00	0.99	0.97	0.91	0.80	0.66	0.50	0.33	0.24	0.14	0.08

Table S.BB2. Continued

(b) male-to-female ratio in the harvest (SR) = 1.25

$t = 1$											
F_O	0.000	0.459	0.574	0.689	0.804	0.918	1.033	1.148	1.263	1.378	1.492
$h_{t=1}$	0.0%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%	7.1%	7.8%	8.5%	9.2%
$H_{t=1}$	0	80	100	120	140	160	180	200	220	240	260
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
$t = 15$											
N_t/N_1	1.12	1.06	1.00	0.96	0.96	0.92	0.88	0.80	0.75	0.59	0.41
N_t/K_t	1.01	0.92	0.89	0.87	0.85	0.82	0.78	0.71	0.66	0.53	0.37
H_t	0	81	101	122	142	161	181	198	211	212	205
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.03	0.07	0.18	0.28
$P_{male.dep}$	0.00	0.00	0.00	0.00	0.01	0.03	0.05	0.14	0.23	0.34	0.39
$P_{Objective1}$	1.00	0.98	0.93	0.86	0.75	0.59	0.43	0.26	0.16	0.07	0.03
$P_{Objective2}$	1.00	0.99	0.98	0.96	0.93	0.85	0.73	0.55	0.42	0.25	0.12
$P_{Objective3}$	1.00	1.00	0.99	0.98	0.97	0.92	0.85	0.72	0.58	0.39	0.24
$t = 36$											
N_t/N_1	0.94	0.88	0.85	0.81	0.78	0.75	0.71	0.65	0.56	0.37	0.14
N_t/K_t	1.00	0.91	0.89	0.87	0.82	0.79	0.74	0.68	0.58	0.39	0.15
H_t	0	71	81	89	93	94	98	100	99	96	93
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.02	0.04	0.11	0.16	0.25	0.36	0.45
$P_{male.dep}$	0.00	0.00	0.01	0.02	0.08	0.12	0.14	0.20	0.22	0.21	0.23
$P_{Objective1}$	0.76	0.39	0.28	0.21	0.13	0.09	0.06	0.04	0.03	0.02	0.01
$P_{Objective2}$	1.00	1.00	0.99	0.95	0.84	0.74	0.64	0.51	0.37	0.25	0.17
$P_{Objective3}$	1.00	0.99	0.97	0.93	0.81	0.69	0.57	0.42	0.29	0.18	0.12

Table S.BB2. Continued

(c) male-to-female ratio in the harvest (*SR*) = 2.00

$t = 1$											
F_O	0.000	0.344	0.430	0.516	0.603	0.689	0.775	0.861	0.947	1.033	1.119
$h_{t=1}$	0.0%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%	7.1%	7.8%	8.5%	9.2%
$H_{t=1}$	0	80	100	120	140	160	180	200	220	240	260
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
$t = 15$											
N_t/N_1	1.12	1.04	1.02	1.01	0.99	0.94	0.85	0.78	0.60	0.41	0.15
N_t/K_t	1.01	0.94	0.91	0.89	0.87	0.83	0.76	0.69	0.54	0.36	0.13
H_t	0	81	102	122	142	161	181	195	198	197	181
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.08	0.16	0.26	0.42
$P_{male.dep}$	0.00	0.00	0.00	0.01	0.03	0.17	0.32	0.46	0.57	0.60	0.51
$P_{Objective1}$	1.00	1.00	0.96	0.92	0.81	0.63	0.39	0.24	0.09	0.06	0.02
$P_{Objective2}$	1.00	1.00	0.99	0.98	0.94	0.83	0.69	0.51	0.31	0.20	0.11
$P_{Objective3}$	1.00	1.00	1.00	1.00	0.97	0.89	0.79	0.61	0.41	0.27	0.15
$t = 36$											
N_t/N_1	0.94	0.88	0.85	0.83	0.79	0.73	0.68	0.61	0.48	0.22	0.00
N_t/K_t	1.00	0.93	0.90	0.87	0.82	0.77	0.72	0.64	0.51	0.23	0.00
H_t	0	64	66	68	67	73	87	94	101	92	77
$P_{extirpation}$	0.00	0.00	0.00	0.01	0.05	0.09	0.14	0.19	0.27	0.40	0.55
$P_{male.dep}$	0.00	0.02	0.08	0.17	0.25	0.29	0.34	0.34	0.33	0.34	0.27
$P_{Objective1}$	0.76	0.47	0.37	0.28	0.20	0.14	0.10	0.06	0.04	0.02	0.01
$P_{Objective2}$	1.00	0.99	0.93	0.85	0.74	0.66	0.60	0.47	0.36	0.21	0.11
$P_{Objective3}$	1.00	0.98	0.92	0.83	0.72	0.62	0.53	0.39	0.28	0.15	0.08

Table S.BB3. Detailed results of primary simulations for the Baffin Bay polar bear subpopulation, using Scenario 3 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

(a) male-to-female ratio in the harvest (*SR*) = 1.0

	$t = 1$							
F_O	0.000	0.598	0.797	0.996	1.196	1.395	1.594	1.793
$h_{t=1}$	0.0%	2.1%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%
$H_{t=1}$	0	60	80	100	120	140	160	180
mgmt.interval	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	$t = 15$							
N_t/N_1	1.14	1.05	1.00	0.98	0.94	0.87	0.82	0.73
N_t/K_t	1.02	0.93	0.90	0.87	0.83	0.78	0.72	0.66
H_t	0	60	80	100	119	136	153	172
$P_{extirpation}$	0.00	0.00	0.00	0.01	0.01	0.04	0.06	0.06
$P_{male.dep}$	0.00	0.00	0.00	0.01	0.01	0.01	0.05	0.06
$P_{Objective1}$	0.97	0.94	0.88	0.80	0.68	0.48	0.35	0.21
$P_{Objective2}$	0.99	0.97	0.94	0.89	0.84	0.73	0.60	0.48
$P_{Objective3}$	1.00	0.98	0.97	0.92	0.89	0.81	0.70	0.59
	$t = 36$							
N_t/N_1	0.94	0.84	0.79	0.73	0.68	0.64	0.61	0.57
N_t/K_t	0.99	0.88	0.83	0.77	0.72	0.68	0.64	0.60
H_t	0	51	66	76	79	80	82	80
$P_{extirpation}$	0.00	0.00	0.00	0.02	0.04	0.07	0.12	0.15
$P_{male.dep}$	0.00	0.00	0.00	0.01	0.01	0.02	0.02	0.02
$P_{Objective1}$	0.72	0.26	0.17	0.11	0.07	0.06	0.05	0.03
$P_{Objective2}$	0.97	0.94	0.86	0.68	0.54	0.46	0.41	0.36
$P_{Objective3}$	0.97	0.93	0.83	0.64	0.49	0.41	0.36	0.31

Table S.BB3. Continued

(b) male-to-female ratio in the harvest (*SR*) = 1.25

				t = 1				
F_O	0.000	0.531	0.709	0.886	1.063	1.240	1.417	1.594
$h_{t=1}$	0.0%	2.1%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%
$H_{t=1}$	0	60	80	100	120	140	160	180
mgmt.interval	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
				t = 15				
N_t/N_1	1.15	1.05	1.02	0.99	0.94	0.87	0.83	0.79
N_t/K_t	1.01	0.93	0.91	0.87	0.83	0.78	0.74	0.70
H_t	0	60	80	99	119	136	156	167
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.02	0.03	0.05	0.09
$P_{male.dep}$	0.00	0.01	0.00	0.00	0.03	0.04	0.07	0.06
$P_{Objective1}$	0.96	0.92	0.88	0.80	0.69	0.52	0.42	0.33
$P_{Objective2}$	0.97	0.94	0.92	0.86	0.82	0.71	0.64	0.56
$P_{Objective3}$	0.98	0.95	0.94	0.89	0.86	0.79	0.73	0.66
				t = 36				
N_t/N_1	0.95	0.85	0.82	0.77	0.73	0.69	0.66	0.67
N_t/K_t	0.99	0.89	0.86	0.80	0.76	0.73	0.70	0.70
H_t	0	52	63	74	79	82	86	89
$P_{extirpation}$	0.00	0.01	0.00	0.01	0.06	0.08	0.10	0.13
$P_{male.dep}$	0.00	0.01	0.01	0.02	0.02	0.03	0.03	0.04
$P_{Objective1}$	0.70	0.33	0.23	0.17	0.12	0.09	0.07	0.06
$P_{Objective2}$	0.96	0.93	0.88	0.75	0.66	0.58	0.52	0.51
$P_{Objective3}$	0.96	0.92	0.86	0.72	0.62	0.53	0.47	0.45

Table S.BB3. Continued

(c) male-to-female ratio in the harvest (*SR*) = 2.00

	***t* = 1**							
F_O	0.000	0.399	0.531	0.664	0.797	0.930	1.063	1.196
$h_{t=1}$	0.0%	2.1%	2.8%	3.6%	4.3%	5.0%	5.7%	6.4%
$H_{t=1}$	0	60	80	100	120	140	160	180
mgmt.interval	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	***t* = 15**							
N_t/N_1	1.14	1.07	1.05	1.02	0.98	0.94	0.91	0.84
N_t/K_t	1.0	0.9	0.9	0.9	0.9	0.8	0.8	0.7
H_t	0	60	80	99	118	136	153	171
$P_{extirpation}$	0.00	0.00	0.00	0.01	0.02	0.03	0.06	0.06
$P_{male.dep}$	0.00	0.00	0.00	0.03	0.04	0.07	0.12	0.17
$P_{Objective1}$	0.97	0.94	0.91	0.84	0.80	0.67	0.59	0.42
$P_{Objective2}$	0.99	0.96	0.95	0.89	0.86	0.80	0.72	0.60
$P_{Objective3}$	0.99	0.97	0.97	0.91	0.89	0.85	0.77	0.69
	***t* = 36**							
N_t/N_1	0.93	0.89	0.86	0.82	0.77	0.74	0.72	0.71
N_t/K_t	0.98	0.93	0.91	0.86	0.81	0.79	0.76	0.74
H_t	0	50	65	68	73	75	79	80
$P_{extirpation}$	0.00	0.00	0.01	0.03	0.07	0.09	0.14	0.14
$P_{male.dep}$	0.00	0.00	0.02	0.05	0.09	0.13	0.13	0.14
$P_{Objective1}$	0.70	0.47	0.38	0.29	0.22	0.18	0.16	0.14
$P_{Objective2}$	0.96	0.96	0.94	0.82	0.74	0.68	0.62	0.59
$P_{Objective3}$	0.96	0.95	0.93	0.80	0.71	0.64	0.58	0.54

Table S.BB4. Detailed results of secondary simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. Results are presented for the highest strategies that meet Management Objective 2b, for each unique combination of *mgmt.interval* and *rsd.mod*. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; N is population size referenced to independent bears; and K is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1). All results assume a state-dependent management approach.

					***t* = 1**				
F_O	1.148	0.976	0.861	1.033	0.918	0.804	0.918	0.804	0.746
$h_{t=1}$	7.1%	6.0%	5.3%	6.4%	5.7%	5.0%	5.7%	5.0%	4.6%
$H_{t=1}$	200	170	150	180	160	140	160	140	130
SR	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25
mgmt.interval	10	15	20	10	15	20	10	15	20
rsd.mod	0.50	0.50	0.50	1.00	1.00	1.00	1.50	1.50	1.50
					***t* = 15**				
N_t/N_1	0.86	0.89	0.93	0.88	0.90	0.96	0.92	0.96	0.96
N_t/K_t	0.76	0.78	0.82	0.78	0.81	0.85	0.81	0.84	0.85
H_t	159	173	153	163	161	142	154	142	132
$P_{extirpation}$	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
$P_{male.dep}$	0.03	0.02	0.01	0.01	0.03	0.00	0.01	0.01	0.00
$P_{Objective1}$	0.25	0.47	0.62	0.41	0.55	0.77	0.55	0.75	0.79
$P_{Objective2}$	0.62	0.77	0.87	0.75	0.80	0.93	0.85	0.92	0.93
$P_{Objective3}$	0.89	0.88	0.95	0.92	0.89	0.98	0.96	0.96	0.97
					***t* = 36**				
N_t/N_1	0.74	0.77	0.78	0.74	0.75	0.76	0.76	0.77	0.77
N_t/K_t	0.77	0.80	0.81	0.77	0.79	0.79	0.79	0.80	0.81
H_t	126	97	122	115	94	130	107	90	113
$P_{extirpation}$	0.00	0.03	0.01	0.01	0.04	0.02	0.01	0.05	0.03
$P_{male.dep}$	0.10	0.11	0.12	0.14	0.10	0.18	0.10	0.10	0.13
$P_{Objective1}$	0.02	0.06	0.12	0.07	0.09	0.14	0.12	0.14	0.18
$P_{Objective2}$	0.76	0.80	0.83	0.71	0.76	0.77	0.73	0.76	0.78
$P_{Objective3}$	0.68	0.76	0.76	0.65	0.71	0.70	0.68	0.71	0.72

Table S.BB5. Detailed results of *post hoc* simulations for the Baffin Bay polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time step $t = 15$. Management strategies BB_S1 and BB_S2 are described in the main text. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-3 (Table BB1).

	Management strategy	
	BB_S1	**BB_S2**
	***t* = 1**	
F_O	1.58	-
$h_{t=1}$	8.7%	8.7%
$H_{t=1}$	245	245
SR	1.00	1.25
mgmt.interval	5	-
rsd.mod	0.50	-
	***t* = 15**	
N_t/N_1	0.75	0.55
N_t/K_t	0.67	0.49
H_t	180	212
$P_{extirpation}$	0.00	0.23
$P_{male.dep}$	0.01	0.30
$P_{Objective1}$	0.05	0.07
$P_{Objective2}$	0.22	0.24
$P_{Objective3}$	0.73	0.36

Table S.KB1. Detailed results of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 1 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

	***t* = 1**					***t* = 1**				
F_O	0.000	0.645	1.289	1.934	2.579	0.000	0.417	0.834	1.251	1.668
$h_{t=1}$	0.0%	0.6%	1.1%	1.7%	2.2%	0.0%	0.6%	1.1%	1.7%	2.2%
$H_{t=1}$	0	2	4	6	8	0	2	4	6	8
SR	0.94	0.94	0.94	0.94	0.94	2.00	2.00	2.00	2.00	2.00
mgmt.interval	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	***t* = 15**					***t* = 15**				
N_t/N_1	1.10	1.02	0.90	0.79	0.79	1.09	1.02	0.98	0.83	0.75
N_t/K_t	0.78	0.72	0.64	0.56	0.56	0.76	0.70	0.69	0.58	0.53
H_t	0	2	4	6	7	0	2	4	6	7
$P_{extirpation}$	0.00	0.01	0.01	0.02	0.06	0.01	0.00	0.01	0.01	0.07
$P_{male.dep}$	0.10	0.16	0.26	0.34	0.37	0.10	0.15	0.34	0.39	0.46
$P_{Objective1}$	0.74	0.64	0.52	0.46	0.43	0.71	0.61	0.63	0.48	0.42
$P_{Objective2}$	0.59	0.46	0.37	0.33	0.30	0.56	0.45	0.45	0.34	0.28
	***t* = 36**					***t* = 36**				
N_t/N_1	1.12	0.82	0.44	0.28	0.22	1.12	0.79	0.54	0.29	0.24
N_t/K_t	0.80	0.59	0.31	0.20	0.16	0.79	0.55	0.38	0.21	0.17
H_t	0	3	4	4	4	0	3	3	3	3
$P_{extirpation}$	0.02	0.09	0.19	0.28	0.38	0.04	0.08	0.17	0.31	0.36
$P_{male.dep}$	0.20	0.36	0.45	0.47	0.43	0.27	0.45	0.59	0.50	0.51
$P_{Objective1}$	0.70	0.48	0.21	0.13	0.10	0.68	0.43	0.25	0.14	0.10
$P_{Objective2}$	0.64	0.33	0.14	0.08	0.07	0.61	0.31	0.16	0.09	0.06

Table S.KB2. Detailed results of primary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

	***t* = 1**									***t* = 1**								
F_O	0.000	0.159	0.319	0.478	0.638	0.797	0.956	1.116	1.275	0.000	0.103	0.206	0.309	0.412	0.516	0.619	0.722	0.825
$h_{t=1}$	0.0%	0.6%	1.1%	1.7%	2.2%	2.8%	3.4%	3.9%	4.5%	0.0%	0.6%	1.1%	1.7%	2.2%	2.8%	3.4%	3.9%	4.5%
$H_{t=1}$	0	2	4	6	8	10	12	14	16	0	2	4	6	8	10	12	14	16
SR	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00
mgmt.interval	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15	15
rsd.mod	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	***t* = 15**									***t* = 15**								
N_t/N_1	1.28	1.31	1.26	1.28	1.19	1.16	1.10	1.06	0.99	1.34	1.28	1.26	1.23	1.20	1.16	1.13	1.08	1.01
N_t/K_t	0.92	0.91	0.88	0.88	0.83	0.81	0.78	0.75	0.70	0.92	0.91	0.89	0.86	0.84	0.82	0.78	0.76	0.70
H_t	0	2	4	6	8	10	11	13	13	0	2	4	6	8	9	11	13	13
$P_{extirpation}$	0.00	0.00	0.02	0.02	0.04	0.03	0.04	0.09	0.18	0.00	0.00	0.00	0.02	0.03	0.04	0.06	0.09	0.18
$P_{male.dep}$	0.06	0.10	0.13	0.12	0.16	0.20	0.24	0.26	0.18	0.05	0.08	0.14	0.15	0.23	0.28	0.31	0.32	0.36
$P_{Objective1}$	0.85	0.85	0.78	0.78	0.73	0.74	0.69	0.63	0.57	0.86	0.85	0.78	0.78	0.72	0.73	0.70	0.63	0.57
$P_{Objective2}$	0.77	0.77	0.72	0.69	0.64	0.64	0.58	0.53	0.48	0.79	0.78	0.71	0.70	0.64	0.65	0.61	0.54	0.49
	***t* = 36**									***t* = 36**								
N_t/N_1	1.26	1.29	1.20	1.19	1.07	0.97	0.84	0.72	0.59	1.33	1.26	1.21	1.14	1.03	0.91	0.78	0.54	0.48
N_t/K_t	0.92	0.90	0.85	0.83	0.75	0.68	0.60	0.52	0.42	0.93	0.90	0.87	0.81	0.73	0.65	0.55	0.39	0.34
H_t	0	3	5	7	8	9	9	9	10	0	3	4	6	6	7	7	6	7
$P_{extirpation}$	0.03	0.03	0.05	0.06	0.07	0.08	0.14	0.22	0.29	0.03	0.03	0.05	0.06	0.10	0.12	0.17	0.26	0.32
$P_{male.dep}$	0.11	0.16	0.18	0.17	0.22	0.29	0.32	0.33	0.25	0.13	0.15	0.22	0.27	0.36	0.43	0.47	0.47	0.44
$P_{Objective1}$	0.81	0.83	0.75	0.73	0.66	0.60	0.50	0.42	0.37	0.84	0.83	0.75	0.72	0.61	0.56	0.46	0.36	0.33
$P_{Objective2}$	0.78	0.79	0.70	0.66	0.57	0.48	0.39	0.31	0.28	0.80	0.78	0.70	0.65	0.53	0.46	0.36	0.28	0.25

Table S.KB3. Detailed results of secondary simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time steps t = 15 and 36. Results are presented for the highest strategies that meet Management Objective 1b, for each unique combination of *mgmt.interval* and *rsd.mod*. F_O is a factor to calculate the harvest rate using equation (1); *SR* is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; *N* is population size referenced to independent bears; and *K* is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

					$t = 1$				
F_O	0.638	0.638	0.558	0.558	0.478	0.478	0.558	0.398	0.398
$h_{t=1}$	2.2%	2.2%	2.0%	2.0%	1.7%	1.7%	2.0%	1.4%	1.4%
$H_{t=1}$	8	8	7	7	6	6	7	5	5
SR	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94	0.94
mgmt.interval	10	15	20	10	15	20	10	15	20
rsd.mod	0.50	0.50	0.50	1.00	1.00	1.00	1.50	1.50	1.50
					$t = 15$				
N_t/N_1	1.12	1.19	1.22	1.15	1.25	1.24	1.17	1.19	1.24
N_t/K_t	0.80	0.82	0.86	0.82	0.89	0.88	0.84	0.84	0.87
H_t	10	8	7	9	6	6	8	5	5
$P_{extirpation}$	0.01	0.02	0.03	0.00	0.02	0.01	0.01	0.01	0.02
$P_{male.dep}$	0.17	0.15	0.11	0.13	0.11	0.14	0.10	0.09	0.09
$P_{Objective1}$	0.74	0.76	0.77	0.79	0.82	0.80	0.79	0.77	0.79
$P_{Objective2}$	0.61	0.66	0.69	0.69	0.74	0.72	0.69	0.67	0.70
					$t = 36$				
N_t/N_1	1.08	1.10	1.11	1.10	1.17	1.15	1.12	1.15	1.18
N_t/K_t	0.78	0.77	0.79	0.80	0.84	0.82	0.81	0.82	0.84
H_t	9	9	9	8	7	7	7	5	6
$P_{extirpation}$	0.04	0.06	0.07	0.05	0.05	0.07	0.05	0.05	0.08
$P_{male.dep}$	0.23	0.21	0.19	0.21	0.18	0.21	0.16	0.14	0.15
$P_{Objective1}$	0.70	0.72	0.72	0.75	0.79	0.75	0.74	0.72	0.74
$P_{Objective2}$	0.59	0.61	0.64	0.66	0.71	0.67	0.66	0.66	0.68

Table S.KB4. Detailed results of *post hoc* simulations for the Kane Basin polar bear subpopulation, using Scenario 2 of the vital rates. Each column corresponds to one simulation, with results reported at time step $t = 15$ and $t = 36$. Harvest strategies KB_S1 and KB_S2 and the simulation conditions on which they are based, are described in the main text. F_O is a factor to calculate the harvest rate using equation (1); SR is a factor that specifies the male-to-female ratio in the harvest; $H_{t=1}$ is the starting harvest level measured in the number of independent bears per year; $h_{t=1}$ is the starting harvest rate, defined as the fraction of total population size (i.e., including dependent young) that is removed each year as independent bears; *mgmt.interval* is the management interval (years); *rsd.mod* is a modifier on the baseline relative standard deviation of the vital rates due to sampling uncertainty; N is population size referenced to independent bears; and K is carrying capacity expressed as numbers of bears. Results for $P_{condition}$ are the estimated probabilities of meeting population conditions corresponding to: extirpation, male depletion, and Management Objectives 1-2 (Table BB1). All results assume a state-dependent management approach.

	Management strategy	
	KB_S1	**KB_S2**
	***t* = 1**	
F_O	0.80	0.60
$h_{t=1}$	2.8%	2.2%
$H_{t=1}$	10	8
SR	0.94	0.94
mgmt.interval	15	15
rsd.mod	1.00	1.00
	***t* = 15**	
N_t/N_1	1.20	1.19
N_t/K_t	0.85	0.83
H_t	10	8
$P_{extirpation}$	0.02	0.00
$P_{male.dep}$	0.07	0.11
$P_{Objective1}$	0.82	0.79
$P_{Objective2}$	0.69	0.65
	***t* = 36**	
N_t/N_1	1.09	1.08
N_t/K_t	0.80	0.77
H_t	10	8
$P_{extirpation}$	0.04	0.04
$P_{male.dep}$	0.21	0.18
$P_{Objective1}$	0.71	0.70
$P_{Objective2}$	0.56	0.55

ᑐᓂᔭᐅᔪᖅ
ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᑦ ᔪᐊᒃ ᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ
ᒫᑦᓯ 2018

ᐅᑯᓇᖓ

ᑐᑭᓯᐅᒪᒍᑎᒃ ᓴᖅ: X ᐃᓱᒪᓕᐅᕈᑦ :

ᐱᔾᔪᑎᖓ : ᐱᕗᓪᓕᐊᑎᑕᐅᔭᓇᐊᖅᑐ ᓂᖕᒥ ᓄᑦ ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ ᐱᖁᔭᑦ

ᐃᒪᓐᓇᐃᓯᒪᕗᖅ:

ᐱᕗᓪᓕᐊᑎᑕᐅᔭᓇᐊᖅᑲᓕᓚᐅᖅ ᓯᒪᕗᑦ ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ ᐱᖁᔭᑦ ᑲᒪᒋᔭᐅᓇᓵᖅ ᒪᑕ ᓄᓇᕗᒻᒥ ᐊᖕᑭᑎᑖᖅᓯᒪᓕᖅᓗᐊᖅ ᑎᓪ ᓗᑕ. ᐅᑯᐊᑦ ᐊᑑᑎᒋᓗᐊᖅ ᑕᖕᒥ ᑐᑭᓯᓯᒪᒍᑎᒃ ᓴᐃᑦ ᑲᒪᒋᔭᐅᒃ ᑲᓐᓂᓇᐊᖅ ᓯᒪᓕᖅ ᓂᖕᒥ ᓄᑦ ᐊᒻ ᒪᓗ ᐱᓕᕆᐊᖅᑎᕆᒍᑕᐅᓕᕈᓐᓇᖅ ᒪ ᓄᓇᕗᒻᒥ ᐆᒪᔪᓕᕆᔨᑦ ᔪᐊᒃ ᑯᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᒻ ᒪᓗ ᓄᓇᕗᑦ ᒐᕙᒪᒃ ᑯᖕᒥ ᓄᑦ ᑕᖅᑭᓂᖃᐃᔪᓂ.

ᒥᓂᔅ ᑕ ᐊᒻ ᒪᓗᐊᐃᒃ ᐹᑐᖕᕆᖕᖑᒥᓂᔅ ᑕᐅᑉ ᑲᑎᒪᖃᑎᖃᖅᓯᒪᓕᖅ ᒪᑕ ᐊᖅᖢᖄᒥᒃ ᐊᒻ ᒪᓗ ᐊᐅᓚᑦ ᓯᓴᒻ ᒪᓇᖅᑯ ᓂᒃ ᓄᓇᕗᑦ ᑐᙵᕕᒃ ᑯᑦ ᐅᖃᐅᓯᖄᖅᑐᑎᒃ ᓯᕗᒧᒋᐊᕈᑕᐅᔪᓇᖅ ᑐᓂᒃ ᐱᕗᓪᓕᐊᑎᑕᐅᔭᒍᑎᒃ ᓴᖕᒥ ᓂᒃ ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ ᐱᖁᔭᓄᑦ.

ᐊᖕᒥᖃᑎᒌᒍᑕᐅᓚᐅᖅ ᒪ ᒪᖅ ᕈᐃᓂᒃ ᐱᓕᕆᖃᑎᒌᓐᓂᐊᖅ ᑐᓂᒃ ᐋᖅᑭᒃ ᓯᔪᒪᓪ ᓗᑎᒃ (ᐊᖅᖢᖅᑲᐅᑏᑦ ᐊᒻ ᒪᓗ ᖃᐅᔨᒻ ᒪᔨᒻ ᒪᓇᓂᒃ) ᐃᑲᔪᖅ ᓂᖃᖅᓴᔪᖅ ᑐᓂᒃ ᐱᕗᓪᓕᐊᑎᑕᐅᓕᖅ ᐸᑕ ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ ᐱᖁᔭᑦ.

ᖃᐅᔨᒻ ᒪᔨᒻ ᒪᕆᐅᓪ ᓗᑎᒃ ᐱᓕᕆᖃᑎᒌᒃ ᑐᑦ ᑲᑎᒪᔨᐅᓗᑎᒃ ᐊᑕᐅᓯᐊᕐᕋᑎᒃ ᐅᖃᓚᐅᑎᒃ ᑯᑦ ᑲᑎᒪᓯᒪᓕᖅ ᑐᑦ ᐊᒻ ᒪᓗ ᑲᑎᒪᓪ ᓚᑦ ᑖᓛᖅ ᑐᑎᒃ ᔭᓐᓄᐊᕆ 30-ᒥ. ᑕᕝᕙᓂ ᑲᑎᒪᓂᐅᔪᒥ ᐱᕖᓪ ᓕᖅ ᓂᖃᓚᐅᖅ ᑐᑦ ᐊᒥᓱᑲᓪ ᓚᓐ ᓂᒃ ᐃᓱᒫᓗᓐ ᓇᑐᓂᒃ ᑲᒪᓪ ᓗᑎᒃ. ᑭᖑᓂᐊᒍᑦ ᑲᑎᒪᓪ ᓚᑖᖅ ᓂᐊᔪᒫᖅ ᑐᖅ ᐱᓇᓱᐊᕈᓯᖕᒥᒃ ᕕᕗᐊᕆ 26 ᐃᖃᓐ ᓂ.

ᐊᖏᓪ ᓗᐊᑲᓪ ᓚᓐ ᓂᒃ ᐱᕖᓪ ᓕᐹᓪ ᓕᕈᑎᖁᒥ ᓇᖅ ᐳᑦ ᐱᕗᓪᓕᐊᑎᑕᐅᑎᓪ ᓗᒋᑦ ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ ᐱᖁᔭᑦ ᐃᒪᓐᓇᐃᖁ ᓗᒍ ᐊᑑᑎᒋᔪᒫᖅ ᓂᐊᖅ ᑕᖕᒥ ᐱᖁᔭᑦ ᑐᓴᖅ ᑕᐅᒃ ᑲᕆᑕᐅᔭᓐᓇᓵᖅ ᒪᑦ ᒥᓂᔅ ᑕᒧᑦ /ᐊᖅᖢᖄᖅᑐᖅᑎᓄᑦ ᓄᓇᕗᑦ ᑐᙵᕕᒃ ᑯᑦ ᔪᓚᐃᑦ 9, 2018-ᒥ.

ᑎᑎᕋᖅᑕᐅᔪ ᐅᑯᓇᖓ ᓯᑲᑦ ᑭᐅᓪ ᐳᑦ, ᑐᑭᒧᐊᒃ ᑎᑦ ᑎᔨ ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ
ᑲᒪᒋᔭᖅᑯ ᑎᐊᓂᖏ ᒧᑦ
ᐃᒪᖅ ᒥᐅᑕᓕᕆᔩᒃ ᑯᑦ ᑲᓇᑕᒥ
ᔩᐊ1 ᒨᓇᔅ, ᐃᖅᑲᓗᓕᕆᓂᖏ ᒧᑦ ᐊᑐᐊᒐᓂᒃ ᖃᐅᔨᒪᔨ
ᓄᓇᕗᑦ ᑐᙵᕕᒃ ᑎᒥᖓ

ᐅᓪ ᓗᖓ ᕕᕝ ᕗᐊᕆ 2, 2018

ᒪᓂᔾᔪᑎ

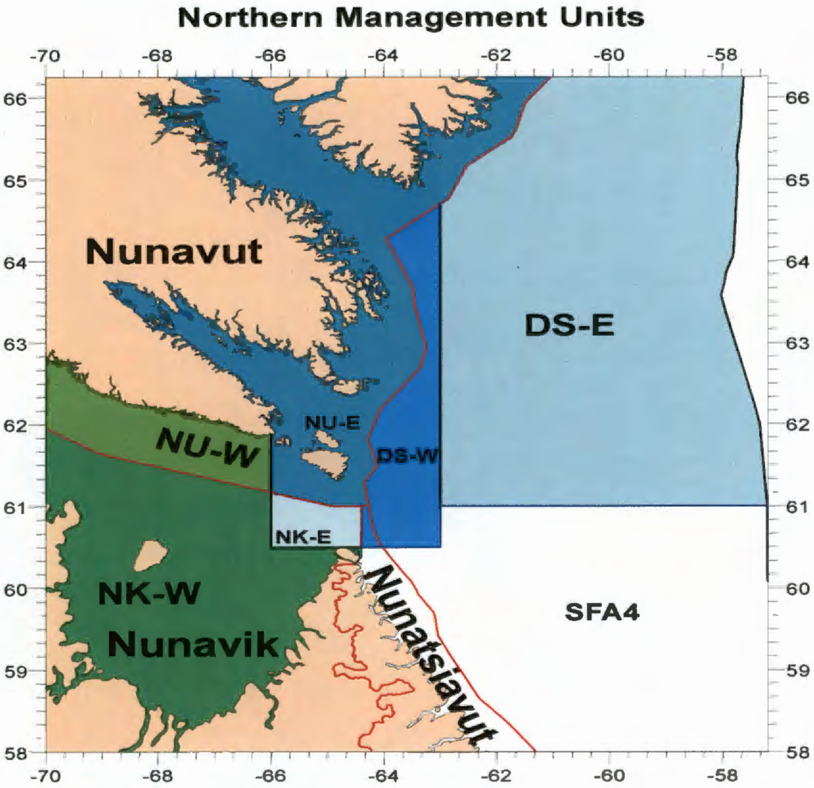
ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᓂᕐᒧᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ ᐊᒻᒪᓗ ᓄᓇᕕᒻᒥ ᑕᕆᐅᕐᒥᐅᑕᓕᕆᔩᑦ ᑲᑎᒪᔨᖏᓐᓄᑦ

ᐱᑦᔪᑎᖓ

ᖃᐅᔨᒪᔭᑦᓴᖅ: ᑐᑭᑖᕐᓂᖅ: X ᐋᖅᑭᑕᐅᖅᑯᔨᓂᖅ: X

ᑲᒪᒍᑕᐅᔪᖅ: ᑲᑎᓪᓗᒋᑦ ᐊᖏᖅᑕᐅᒪᔪᑦ ᐱᒍᑎᑦᓴᑦ ᖃᓄᐃᓪᓗᐊᓂᖏᑦ ᐃᓚᖓᔪᑦ ᑕᕐᕋᒥᐅᑕᕐᓄᑦ (*Pandalus Borealis*) ᐊᒻᒪᓗ ᖁᐸᑎᓕᓐᓄᑦ (*Pandalus Montagui*) ᑭᖑᕐᓚᓄᑦ 2018/19-ᒥ ᐃᖃᓗᓐᓂᐊᓇᒥ, ᐊᒻᒪᓗ 2). ᐃᖃᓗᓐᓂᐊᓇᒧᑦ ᐃᑳᖅᓯᕈᑏᑦ ᒪᓕᒐᖅᒋᑦ ᑐᕌᖅᑐᑦ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᓄᓇᕕᒃ ᐊᕕᑦᑐᕈᑎᖏᓐᓄᑦ

<u>ᓄᓇᖕᖑᐊᖅ:</u>
ᑐᖑᔪᖅᑕᐅᓂᖏᑦ – ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ ᖃᐅᔨᓴᖅᕕᒃ
ᓄᓇᐅᔭᐅᓂᖏᑦ – ᓯᖅᑭᓂᐅᑉ ᓂᐱᖕᕕᐊᒍᑦ ᖃᐅᔨᓴᖅᕕᒃ



ᑲᑎᓪᓗᒋᑦ ᐊᖏᖅᑕᐅᒪᔪᑦ ᐱᒍᑎᑦᓴᑦ ᖃᓄᐃᓪᓗᐊᓂᖏᑦ ᐃᓚᖓᔪᑦ ᑕᕐᕋᒥᐅᑕᕐᓄᑦ (*Pandalus Borealis*) ᐊᒻᒪᓗ ᖁᐸᑎᓕᓐᓄᑦ (*Pandalus Montagui*) ᑭᖑᕐᓚᓄᑦ 2018/19-ᒥ ᐃᖃᓗᓐᓂᐊᓇᒥ

<u>ᐅᖓᑎᒥᐅᑕᖏᑦ</u>

ᒪᕐᕈᐃᑦ ᑭᖑᕐᓚᓴᔭᑦ (ᖁᐳᑎᓖᑦ - *Pandalus Montagui* ᐊᒻᒪᓗ ᑕᕐᕋᒥᐅᑕᑦ *Pandalus Borealis*) ᐱᖃᖃᓲᖑᕗᑦ ᐅᑭᐅᖅᑕᑐᒥ ᑭᖑᕐᓚᓂᐊᕐᕕᒥ ᑕᓪᓗᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᐊᒻᒪᓗ ᐃᑭᕋᓴᕐᔪᐊᒥ ᓯᕿᓂᐅᑉ ᓄᐃᕙᐊᒍᕗᓯᑉ ᑕᒪᓐᓇ ᖃᓄᓯᑎᓪᓗᒍ ᐃᓚᒋᔭᐅᔪᓂᒃ ᓄᓇᕗᒻᒥ ᓄᓇᑖᖑᓯᒪᔪᓄᑦ (NSA) ᐊᒻᒪᓗ ᓄᓇᕕᐅᑉ ᑕᕆᐅᖁᑎᖓᓄᑦ (NMR). ᑲᑎᑦᓯᒋᑦ ᐊᖏᖅᑕᐅᒪᔪᑦ ᐱᒍᑎᑦᓴᑦ (TAC) ᐊᑐᓂ ᑭᖑᕐᓚᓴᔭᓄᑦ ᐋᖅᕿᑕᐅᓯᒪᔪᑦ ᒪᕐᕈᐃᓄᑦ ᐃᓖᒃᑯᑦ ᐃᓯᒪᑕᕈᑎᑎᒍᑦ ᖃᐅᔨᓴᕐᕕᓄᑦ (ᓯᕿᓂᐅᑉ ᓄᐃᕙᐊᒍᑦ ᐊᒻᒪᓗ ᓂᐱᖕᕙᐊᒍᑦ), ᑌᒪ ᐊᕕᑦᑐᓯᒪᒻᒥᓯᑎᑉ ᐊᐅᓚᑦᓯᔭᐅᕕᓐᓄᑦ ᒪᓕᑦᓯᓂ ᓇᓗᓃᕐᓯᒪᔪᓂᒃ ᐊᒥᖅᑳᕈᑎᖃᕐᓂᒧᑦ ᐊᖏᖃᑎᒌᒍᑎᓂᒃ.

ᑐᖅᑲᑕᕐᕕᐅᑉ ᑐᑭᑖᖏᑦ ᐱᑦᔪᑎᓖᑦ 2017/18-ᒥ TAC-ᓂᒃ ᐊᑐᕐᓯᒪᔪᑦ ᐃᓯᒪᑕᕈᑏᑦ ᐃᓯᒪᓕᐅᕐᓂᒍᑎᖏᓐᓂᒃ, ᑖᒃᑯᐊᓗᑦᑕᐅᖅ ᐃᓯᒪᒻᒥᓯᑎᑉ 2016-ᒥ 2015-ᒥᓗ ᑭᓯᓐᓂᑯᑦ ᖃᐅᔨᒪᒍᑎᑖᕕᓂᕐᓂᒃ, ᑐᑭᑖᖑᓯᒪᔪᓂᓪᓗ ᐊᒻᒪᓗ ᐋᖅᕿᑕᐅᖁᔭᕕᓂᕐᓂᒃ NWMB-ᑯᓄᑦ ᐊᒻᒪᓗ NMRWB-ᑯᓄᑦ, ᐊᒻᒪᓗ ᐃᓯᒪᑦᓴᓯᐅᕈᑎᕕᓂᕐᓂᒃ ᐅᑭᐅᖅᑕᑐᒥ ᑭᖑᕐᓚᓂᒃ ᖃᐅᔨᒪᔨᐅᓂᕐᒧᑦ ᑲᑎᒪᔨᐊᐱᓐᓂᒃ EAZ-ᒥ. 2017/18-ᒥ ᐃᖃᓗᓐᓂᐊᓇᒧᑦ, TAC-ᖏᑦ ᐊᒻᒪᓗ ᐊᒡᒍᕈᑏᑦ ᑕᒪᖏᓐᓂᒃ ᑭᖑᕐᓚᓂᒃ WAZ-ᒥ ᐊᒻᒪᓗ EAZ-ᒥ ᒪᑭᑌᓐᓇᑎᑕᐅᓯᒪᔪᑦ.

ᓯᓇᒥᑦᓯᔪᑦ ᑭᖑᕐᓚᓂᒃ ᖃᐅᔨᓴᐅᑏᑦ ᐊᑐᕐᑕᐅᓲᖑᕗᑦ ᐊᕐᕌᒍᒃ ᒪᕐᕉᒃ ᑕᒫᑦ, ᑭᖑᓪᓕᖅ ᓯᓇᒥᓪᓗᒋᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓂᕆᐅᓐᓇᑎᓪᓗᒍ 2019-ᒥ. ᐃᓯᒪᑕᕐᓂᖅ ᓄᐃᑦᓯᒍᑕᐅᓯᒪᔪᖅ ᑭᖑᖅᑐᑦ ᖃᓄᐃᒻᒪᖕᒫᑕ ᑐᓴᒃᑲᓂᕐᑎᓯᒍᑎᓂᒃ ᖃᐅᔨᒪᒍᑕᐅᓚᖓᑎᓪᓗᒋᑦ TAC-ᓂᒃ ᑐᑭᑖᕈᑎᓄᑦ ᐊᑐᓚᖓᔪᓄᑦ ᖀᔪᒥ ᐃᖃᓗᓐᓂᐊᓇᒥ.

ᑐᑭᑖᕐᓃᑦ ᐊᒻᒪᓗ ᐋᖅᕿᑕᐅᖁᔨᒍᑏᑦ ᐊᑦᑐᐊᔪᑦ ᖃᓄᐃᓪᓗᐊᑐᓂᒃ TAC-ᖃᕐᓂᒧᑦ ᓯᕿᓂᐅᑉ ᓂᐱᖕᕙᐊᒍᑦ ᖃᐅᔨᓴᕐᕕᒥ (WAZ) ᐊᒻᒪᓗ ᓯᕿᓂᐅᑉ ᓄᐃᕙᐊᒍᑦ ᖃᐅᔨᓴᕐᕕᒥ (EAZ) ᐃᓖᒃᑯᑦ ᐃᓕᖓᔪᑦ ᐊᑐᓂ ᑭᖑᕐᓚᓴᔭᓄᑦ 2018/19-ᒥ ᐃᖃᓗᓐᓂᐊᓇᒥ ᐱᒋᐊᖃᓕᕐᒥᔪᑦ ᑕᒪᖏᓐᓂᒃ ᑲᑎᒪᔨᐅᕕᓐᓂᒃ.

ᑌᒣᒐᓗᐊᕐᑎᓗᒍ, ᐱᐅᓯᐅᖃᑦᑕᓯᒪᑦᓯᓂᓗ ᓯᕗᓂᕐᒥ, ᐃᓯᒪᑦᓴᓯᐅᕐᓃᑦ ᐊᒻᒪᓗ ᐊᐅᓚᑦᓯᔨᐅᖃᑎᒌᑦ ᐱᕐᕈᓯᖏᑦᑕ ᖃᖓ ᑲᔪᓯᑎᑕᐅᓂᖏᑦ ᐊᒻᒪᓗ ᐊᑐᐃᓐᓇᓯᒍᓐᓇᓂᖅ ᓄᑖᓂᒃ ᐃᓯᒪᑕᕈᑎᓄᑦ ᖃᐅᔨᒪᒍᑎᑖᓂᒃ ᑐᑭᑖᕐᓂᒧᑦ ᑐᑭᒧᐊᒍᑎᑦᓴᓂᒃ ᓯᓚ ᐃᓯᒫᓗᓐᓇᓖᕐᒥᔪᑦ ᑕᒻᒪᓂ ᐃᖃᓗᓐᓂᐊᓇᒥ. ᐃᓯᒪᑕᕈᑎᓄᑦ ᑐᓴᒃᑲᓂᕈᑏᑦ ᖃᒻᒥᓂᑕᐅᓂᕐᐹᓂᒃ 2017-ᒥ ᑭᓯᒍᑎᓂᒃ ᓂᕆᐅᓐᓇᑐᑦ ᕕᕈᐊᕆᐅᑉ ᕿᑎᖓᓂ ᐃᓯᐊᓂᓘᓐᓃᑦ ᑭᓯᐊᓂ, 2018-ᒥ. ᐃᓯᒪᑦᓴᓯᐅᖃᑎᖃᕐᓂᖅ NSAC-ᑯᓂᒃ ᓯᕗᓂᓕᐅᕐᓯᒪᔪᖅ ᒫᑦᓯᐅᑉ ᐱᒋᐊᖕᖏᓂᖓᓄᑦ. NWMB-ᑯᑦ ᐊᒻᒪᓗ NMRWB-ᑯᑦ ᑲᑎᒪᓂᑐᐃᓐᓇᖏᑦ ᐱᐅᓯᒃᑯᑦ ᐊᑑᑎᓲᖑᕗᑦ ᒫᑦᓯᐅᑉ ᕿᑎᖓᓂ ᐊᒻᒪᓗ ᑭᖑᕐᓚᓂᒃ ᐃᖃᓗᓐᓂᐊᓇᐅᑉ ᐅᒃᑯᐃᓂᖓ ᑐᕌᐳᖅ ᐁᕆᓕ 1-ᒧᑦ.

ᑌᒣᒻᒪᑦ, ᐃᓂᑦᓚᓯᐊᓂᕐᓴᐅᖁᓗᒍ ᐃᖃᓗᓐᓂᐊᕕᒃ 2018/19-ᒥ ᐃᖃᓗᓐᓂᐊᓇᒧᑦ ᐊᒻᒪᓗ ᑭᖑᕋᕈᑎᓂᒃ ᐊᑐᖁᔭᖕᖏᒧᑦ ᑕᒪᑐᒪ ᐱᕕᖃᕐᓂᓕᒫᖓᓄᑦ, ᐱᑦᔪᔨᕕᒃ ᐱᔪᒪᕗᖅ ᑲᑎᒪᔨᐅᕖᑦ ᑐᑭᑖᕐᓂᖏᓐᓂᒃ ᐊᒻᒪᓗ ᐃᓯᒪᓕᐅᕐᓂᒍᑎᖏᓐᓂᒃ ᐊᑦᑐᐊᔪᓂᒃ ᐊᖏᖅᑕᐅᒌᕐᓯᒪᔪᓂᒃ ᐱᕐᕈᓯᖃᕐᑎᓯᓂᕐᒧᑦ TAC-ᓂᒃ 2018/19-ᒥ ᐃᖃᓗᓐᓂᐊᓇᒥ WAZ-ᒥ ᐊᒻᒪᓗ EAZ-ᒥ.

ᓯᕿᓂᐅᑉ ᓂᐱᖕᕙᐊᒍᑦ ᖃᐅᔨᓴᕐᕕᒃ - ᑐᑭᑖᕆᐊᓕᒃ

WAZ ᐃᓘᓐᓇᓂ ᐃᓚᐅᕗᖅ NSA/NMR ᐃᓗᐊᓂ. ᐱᒋᐊᕐᓯᒍ 2014-ᒥ, ᐃᓯᒪᑕᕈᑎᓄᑦ ᖃᐅᔨᓴᕐᓂᖅ WAZ-ᒥ ᐊᑐᕐᑕᐅᓯᒪᔪᖅ ᐅᑭᐅᖅᑕᑐᒥ ᑭᖑᕐᓚᓂᒃ ᖃᐅᔨᓴᐅᑎᓄᑦ ᖅᑮᓇᐅᔭᖃᐅᑎᒧᑦ (NSRF) ᐊᒻᒪᓗ ᐱᑦᔪᔨᕕᑦᑎᓄᑦ. ᑌᒣᒻᒪᑦ, ᐅᒥᐊᖅ ᐊᒻᒪᓗ ᐱᓇᓲᑏᑦ ᐊᑐᕐᑐᑦ ᖃᐅᔨᓴᕐᓂᒥᒃ ᐊᓂᒍᕐᑎᓯᒍᑕᐅᑦᓯᑎᑉ ᐊᓯᑦᔨᓯᒪᔪᑦ, ᐊᒻᒪᓗ ᖃᖓ ᐊᕐᕌᒍᑉ ᐃᓗᐊᓂ ᖃᐅᔨᓴᕐᐸᓂᖓᑦ ᐊᓯᑦᔨᓯᒪᒻᒥᓯᓂ. ᑭᖑᓂᐅᑎᓪᓗᒍ ᑕᒪᑐᒧᖓ, ᖃᖓᒃᑯᑦ ᑭᖑᓕᕇᕐᑎᑐᑦ ᑕᒃᒐᓂ ᖃᐅᔨᓴᕐᕕᒥ ᐱᒋᐊᓯᐊᑎᑕᐅᓯᒪᔪᑦ 2014 ᐊᕐᕌᒍᖓ ᓯᕗᓪᓕᐸᐅᓯᑎᓪᓗᒍ. ᐃᓯᒪᑕᕈᑏᑦ ᓇᓗᓃᕐᓯᓯᒪᔪᑦ ᐃᑭᓐᓂᐹᖓᓄᑦ 3—5-ᓂᒃ ᑕᕐᓯᕈᑎᓂᒃ ᖃᖓᐅᓂᕐᒥᒍᑦ ᑭᖑᓕᓈᓂᒃ ᐱᑕᖃᕆᐊᖃᕐᑎᓗᒍ ᐃᑉᐱᒍᓯᑦᓯᐊᓗᓂ ᑐᑭᑖᕐᑐᖃᕐᓂᐊᐸᑦ ᐋᖅᕿᒋᐊᕈᑎᓂᒃ TAC-ᓄᑦ. WAZ ᐃᓗᐊᓂ, ᐊᓯᑦᔨᕈᑏᑦ ᐃᖃᓗᓐᓂᐊᒐᓪᓗᐊᑦ ᐊᒥᒉᓐᓂᖓᑎᒍᑦ ᖄᖏᐅᑎᔪᑦ 25%-ᒥᒃ (ᐃᓯᒪᒋᔭᐅᔪᒥᒃ ᑲᑕᑦᑐᒪᕆᐊᓘᑎᓪᓗᒍ) ᓄᐃᖃᑦᑕᓯᒪᕗᑦ ᑕᒪᖏᓐᓂ ᑭᖑᕐᓚᓴᔭᓂ ᐊᕐᕌᒍᑕᒫᑦ ᑌᒪᖕᖓᓂᒃ ᑖᒃᑯᐊ ᓄᑖᑦ ᖃᐅᔨᓴᖃᑦᑕᓃᑦ ᐱᒋᐊᓕᐅᔪᒻᒪᑕ, ᑌᒣᖕᖏᑐᑑᑎᓪᓗᒋᑦ ᑕᕐᕋᒥᐅᑕᑦ 2013-ᒥ ᐊᒻᒪᓗ 2014-ᒥ ᖃᐅᔨᓴᕐᑕᐅᒍᑖᓐᓂ. 2016-ᒥ ᖃᐅᔨᓴᕈᑎᕕᓃᑦ ᐊᕐᕌᓂ ᓇᓗᓃᕐᓯᓚᐅᔪᔪᑦ ᐃᖃᓗᓐᓂᐊᒐᓪᓗᐊᑦ ᐊᒥᒉᓐᓂᖓᑕ ᑲᑕᕐᑐᕕᓂᐅᓂᖓᓂᒃ ᑕᕐᕋᒥᐅᑕᕐᑎᒍᑦ (54%) ᐊᒻᒪᓗ ᖁᐳᑎᓕᑦᑎᒍᑦ (42.5%). ᐊᑐᕆᐊᓪᓚᑎᑦᓯᓂᖅ TAC-ᓂᒃ (2,080t ᐊᒻᒪᓗ

6,138t) ᓄᐃᑎᕐᓯᓯᒪᔪᖅ ER-ᒥᒃ 15.9%-ᒥᒃ ᑕᕐᕆᐅᑕᕐᓄᑦ ᐊᒻᒪᓗ 19.3%-ᒥᒃ ᖁᐳᑎᓕᓐᓄᑦ. 2016-ᒥ ᐃᖃᓗᓐᓂᐊᓇᐅᑉ ᓯᕗᓂᐊᒍᑦ, ER-ᖏᑦ 15% ᐊᑖᓂᖃᑦᑕᓚᐅᔪᔪᑦ ᑌᒪᖕᖓᐅᓂᕐᓴᖅ.

ᒪᓕᑦᑎᒋᓕᐅᑎᕐᓯᒍ ᑰᑕᓂᒃ ᐊᒻᒪᓗ ᑲᑎᑦᑐᒋᑦ ᐊᖏᕐᑕᐅᓯᒪᔪᓂᒃ ᐱᒍᑎᑦᓴᓂᒃ (TAT) ᑐᑭᑖᕐᓂᖏᓐᓄᑦ ᑲᑎᒪᔨᐅᕙᑦ 2017/18-ᒥ, ᐊᖏᕐᑕᐅᓚᐅᔪᔪᓄᑦ ᑐᕐᖃᑕᕐᕙᒍᑦ, ᑲᔪᓯᑎᕐᓯᓚᐅᔪᒻᒥᔪᑦ 50/50-ᒥᒃ ᐊᕙᒍᑎᒥᒃ ᐃᓘᓐᓈᓂᒃ TAC-ᒥᒃ ᓄᓇᕗᑦᒍᑦ ᐊᒻᒪᓗ ᓄᓇᕕᒻᒍᑦ.

ᐱᑦᔪᐳᕙᒃ ᐱᔪᒪᕗᖅ ᑕᒪᖏᑦᑕ ᑲᑎᒪᔨᐅᕙᑦ ᑐᑭᑖᖏᓐᓂᒃ ᐱᔭᐅᒍᓐᓇᑐᑦ ᖃᓄᐃᓪᓗᐊᓂᖏᓐᓂᒃ NU/NK-ᒥ ᓯᕿᓂᐅᑉ ᓂᐱᕝᕙᐊᒍᑦ (ᐃᒫᒃ, ᐃᓘᓐᓈᒍᑦ TAC-ᒥᒃ), ᐊᒻᒪᓗ ᓱᕐᖁᐃᑎᕐᓯᓂᖕᒪᓂᒃ ᐊᕙᒍᑎᒥᒃ ᐃᓘᓐᓈᒍᑦ TAC-ᒥᒃ ᐊᑯᓐᓂᒥ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᓄᓇᕕᒃ ᐃᓄᖏᓐᓄᑦ.

ᐋᕐᕿᑕᐅᖁᓇᕐᑐᑦ WAZ-ᒥ:

ᐃᓱᒪᕐᓱᓂ ᐃᓱᒪᑕᕈᑏᑦ ᓯᑕᒪᓂᒃ ᖃᐅᔨᒪᒍᑏᑦ ᑕᕐᓯᒍᑎᖏᓐᓂᒃ ᐱᑕᖃᕐᓂᖓᓂᒃ ᖃᖓᐅᓂᕐᒥᒍᑦ ᑭᖑᓕᕇᓂᒃ, ᐊᒻᒪᓗ ᐃᓱᒪᕐᓱᓂ ᑲᑕᕐᑐᒪᓇᐊᓘᖃᑦᑕᓯᒪᓂᖏᓐᓂᒃ ᐃᖃᓗᓐᓂᐊᒐᓪᓗᐊᑦ ᐊᒥᒉᓐᓂᖏᑦ ᑕᒪᖏᓐᓂ ᑭᖑᕐᓚᓴᔭᓂ ᐊᕐᕌᒍᓂ ᖃᒻᒥᖅ, ᓱᒐᓗᐃᖕᖏᑐᒥᒃ ᐱᐅᓯᖃᕈᑎᑦᓴᖅ ᓱᒐᓗᐃᓕᖓᓗᓂ ᑭᖑᕐᓕᓂᐅᑉ ᓄᓇᑖᕐᕙᐅᓯᒪᔪᑦ ᐃᓂᖏᑦᑕ ᐃᓗᐊᓂ ᖄᖏᐅᑎᖕᖏᓂᖕᒪᓂᒃ ᒪᑭᑕᔭᓪᓗᐊᓃᒃ ᐊᓐᓂᑐᕐᑐᑦ ᐱᒍᑎᖏᓐᓂᒃ ᐊᑦᑐᐊᕗᖅ ᑌᒣᓕᖕᒉᓐᓇᑎᕐᓯᓂᕐᒧᑦ ᐅᓪᓗᒥ TAC-ᓂᒃ WAZ ᐃᓗᐊᓂ ᐊᒻᒪᓗ ᐱᓂᕐᒧᑦ ᐊᑕᐅᓯᒋᐊᓪᓚᒥᒃ ᖃᐅᔨᒪᒍᑏᑦ ᑕᕐᓯᒍᑎᖕᒪᓂᒃ ᖃᖓᐅᓂᒃᑯᑦ ᑭᖑᓕᕇᓂ ᐋᕐᕿᒋᐊᕐᑎᓯᔪᖃᖕᖏᓂᖕᒪᓂ TAC-ᓂᒃ. ᑌᒣᖏᓐᓇᑎᕐᓯᓂᖅ ᐅᓪᓗᒥ TAC-ᓂᒃ 2018/19-ᒥ ᓱᒐᓗᐃᒐᔭᖕᖏᑐᖅ ᐃᓱᒪᑕᕈᑎᑎᒍᑦ ᑕᑯᓐᓇᓯᒍ ᐃᓱᒪᕐᓱᓂ ᖃᐅᔨᓴᕈᑏᑦ ᖃᒻᒥᓂᑖᐱᐅᓂᖕᒪᓂᒃ. ᑭᓯᐊᓂ, ᑌᒣᓕᖓᔪᒥᒃ ᑲᔪᓯᑎᕐᓯᓂᖅ ᓄᐃᕐᓯᓴᑐᐃᓐᓇᓇᐊᖃᕐᒥᔪᖅ ER-ᖓᑕ ᐊᑕᐅᓰᑦ ᑕᒪᖏᑦᑕᓘᓐᓃᑦ ᑭᖑᕐᓗᑦ ᐱᓕᐅᒥᓂᖕᒪᓂᒃ ᓯᑲᓐᓂᓴᐅᓗᓂ ᐅᓪᓗᒥ 15.9%-ᒥᒃ ᑕᕐᕆᐅᑕᕐᑎᒍᑦ ᐊᒻᒪᓗ 19.3%-ᒥᒃ ᖁᐳᑎᓕᕐᑎᒍᑦ. ᑲᑎᒪᔨᐅᕙᑦ ᐃᓱᒪᒍᓐᓇᑐᑦ, ᒪᑲᒌᒃᑯᑕᐅᑎᓪᓗᒍ, ᒪᑭᕐᓯᓂᕐᒥᒃ TAC-ᓂᒃ ᑐᖕᖓᕙᖃᕐᑎᓗᒋᑦ 15%-ᒥᒃ ER-ᒥᒃ, ᑕᒪᓐᓇᓗ ᓄᐃᕐᓯᒍᓐᓇᓯᓂ ᓇᓪᓕᐊᓂᒃ ᐊᒥᓱᖕᖑᔩᒥᓂᕐᒥᒃ ᐅᕝᕙᓘᓐᓃᑦ ᐃᑭᓕᐅᒥᓂᕐᒥᒃ ᐊᑕᐅᓰᓄᑦ ᑕᒪᖏᓐᓄᓘᓐᓃᑦ ᑭᖑᕐᓕᓄᑦ. 2018-ᒥ ᖃᐅᔨᓴᕈᑏᑦ ᐊᕐᕌᒍ ᓄᐃᕐᓯᑖᕐᑐᑦ ᐊᕐᕌᒍᐃᑦ ᑕᓪᓕᒪᖓᓐᓂᒃ ᖃᖓᐅᓂᒃᑯᑦ ᑭᖑᓕᕇᓂ, ᐱᒍᑎᑦᓴᕈᕐᓗᑎᒃ ᐳᕿᐊᖕᖏᓂᕐᓴᐅᓂᕐᒥᒃ ᓇᐅᑐᐃᓐᓂᑦ ᐊᓯᕐᕙᕈᑏᑦ TAC-ᓄᑦ ᑐᖕᖓᕙᖃᕐᓂᖕᒪᓂᒃ ᐃᓗᐃᑦᑐᓕᐅᕐᓂᓴᒥᒃ ᑐᑭᓯᒪᓂᕐᒥᒃ ᑖᒃᑯᓂᖓ ᐃᖃᓗᓐᓂᐊᒐᕐᓂᒃ.

TACs:
ᓇᓪᓕᑳᕈᑎᑦᓴᖅ 1:
ᑌᒣᖏᓐᓇᑎᕐᓯᓂᖅ ᑕᒪᖏᓐᓂᒃ TAC-ᓂᒃ ᑕᒣᓐᓄᑦ ᑭᖑᕐᓕᓄᑦ ᑕᓪᓕᒪᖕᖑᕈᑏᑦ ᖃᐅᔨᒪᒍᑎᓂᒃ ᑕᕐᓰᒍᑎᓄᑦ ᐱᖕᖏᓂᖏᓐᓂ ᐊᕐᕌᒍ ᖃᐅᔨᓴᐅᑎᓂᒃ, ᑌᑲᓂ ᓯᓇᒥᑦᑐᑦ ᑭᖑᕐᓕᓂᒃ ᖃᐅᔨᓴᐅᑏᑦ ᑲᔪᓯᑎᑕᐅᓛᕐᑎᓗᒋᑦ.

ᓇᓪᓕᑳᕈᑎᑦᓴᖅ 2:
ᑌᒣᖏᓐᓇᑎᕐᓯᓂᖅ TAC-ᓂᒃ ᑕᒣᓐᓄᑦ ᑭᖑᕐᓕᓄᑦ ᐅᓪᓗᒥ ER-ᖏᑦ (15.9% ᑕᕐᕆᐅᑕᕐᓄᑦ ᐊᒻᒪᓗ 19.3% ᖁᐳᑎᓕᓐᓄᑦ) ᖄᖏᐅᑎᔭᐅᖕᖏᓚᐅᕐᐸᑕ 2018-ᒥ ᖃᐅᔨᓴᐅᑏᑦ ᓇᓗᓀᕐᓯᔅᓂᐊᐸᑕ ᐃᖃᓗᓐᓂᐊᒐᓪᓗᐊᑦ ᐊᒥᒉᓐᓂᖓᑕ ᑲᑕᓐᓂᖕᒪᓂᒃ.

ᓇᓪᓕᑳᕈᑎᑦᓴᖅ 3: **(ᐊᑐᖁᓇᕐᑐᖅ)**
ᒪᑭᕐᓯᓂᖅ ᐃᓘᓐᓈᒍᑦ TAC-ᓂᒃ ᑐᖕᖓᕙᖃᕐᓗᓂ 15%-ᒥᒃ ER-ᒥᒃ ᑕᒪᖏᓐᓄᑦ ᑭᖑᕐᓕᓄᑦ ᒪᑲᒌᒃᑯᑕᐅᑎᓪᓗᒍ.

ᓯᕿᓂᐅᑉ ᓄᐃᕙᐊᒍᑦ ᖃᐅᔨᓴᕐᕕᒃ – ᐋᕐᕿᑕᐅᖁᔨᐅᑎᓂᒃ ᐊᒻᒪᓗ ᑐᑭᑖᕐᓂᒥᒃ ᐱᒍᑕᐅᒥᐊᓕᒃ

EAZ ᑐᕐᖁᓯᒪᕗᖅ ᑕᒪᖏᓐᓄᑦ NSA/NMR ᐃᓗᐊᓄᑦ ᐊᒻᒪᓗ ᓯᓚᑖᓄᑦ. ᑖᒃᑯᐊ ᑭᖑᕐᓕᖏᑦ ᐊᐅᓚᑕᐅᒍᑎᖃᕐᓯᒪᔪᑦ ER-ᒥᒃ 15%-ᒧᑦ ᖃᓂᑦᑑᖏᓐᓇᓕᒫᕐᑐᒥᒃ. ᑕᕐᕆᐅᑌᑦ ᑭᖑᕐᓗᑦ ᖃᓄᐃᖕᖏᑐᒌᖕᓇᕐᓴᓗᐊᕐᑎᑐᒋᑦ ᐱᑦᔪᐳᕙᑦᑕ PA-ᓄᑦ ᐃᓂᓪᓚᑎᕈᑎᖏᓐᓂ, ᓯᓕ ᓱᕐᖁᐃᖏᒍᑎᓂᒃ ᐱᑕᖃᕐᓴᓇᕐᑐᖅ ᖁᐳᑎᓕᓐᓄᑦ ᐊᑦᑐᐊᔪᓂᒃ.

EAZ-ᒥ, TAC-ᖏᑦ 2017/18-ᒥ ᐃᖅᑲᓗᓐᓂᐊᓇᒥ ᑌᒣᖏᓐᓇᑎᑕᐅᓚᐅᔪᔪᑦ ᑕᖅᑫᒥᐅᑕᖅᓄᑦ (9,488t) ᐊᒻᒪᓗ ᖁᐳᑎᓕᓐᓄᑦ (840t), ᓄᐃᑦᓯᓯᑎᑉ ER-ᒥᒃ 14.47%-ᒥᒃ (ᑕᖅᑫᒥᐅᑕᖅᓄᑦ) ᐊᒻᒪᓗ 6.1%-ᒥᒃ (ᖁᐳᑎᓕᓐᓄᑦ), ᑕᒪᓐᓇᓗ ᐊᖏᖅᑲᑕᐅᕗᖅ ᐊᐅᓚᑦᓯᒍᑎᓄᑦ ᑕᒫᓂ ᐃᓂᒥ 15%-ᒥᒃ ER-ᒥᒃ ᒥᑦᓴᐅᓯᒍᑎᓕᒻᒥ. ᓱᓕᐅᒻᒥᔪᖅ, ᐃᓱᒪᓇᖅᑎᑕᐅᔪᖅ ᑰᑕᓂᒃ ᐊᒻᒪᓗ TAT-ᓂᒃ ᑐᑭᑖᕈᑎᕕᓂᖏᓐᓄᑦ ᑲᑎᒪᕕᐅᔪᑦ ᐊᑐᓂ 2015/16 – 2017/18 ᐃᖅᑲᓗᓐᓂᐊᓇᖅᓂ, ᑐᖅᖁᑕᖅᕕᒧᓪᓗ ᐊᖏᖅᑕᐅᓚᐅᔪᑦᓯᓂ, ᐊᑦᑐᐊᒥᔪᖅ ᑲᔪᓯᑎᑦᓯᓂᕐᒧᑦ ᐊᕕᑦᑐᕈᑎᒥᒃ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᓄᓇᕕᒃ ᐃᓄᖏᓐᓄᑦ NU/NK-ᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ, ᐊᒻᒪᓗ ᐊᕕᑦᑐᕈᑎᒥᒃ ᐊᑯᓐᓂᒥ NU /NK-ᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᒍᑦ ᐊᒻᒪᓗ ᑕᑦᑐᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᐊᐅᓚᑦᓯᕕᓄᑦ.

EAZ-ᒥ, ᖁᐳᑎᓖᑦ ᑑᖅᑕᐅᓯᑎᑉ ᐃᖅᑲᓗᓐᓂᐊᒐᐅᕗᑦ ᐊᒻᒪᓗ ᑕᖅᑫᒥᐅᑌᑦ ᐱᖅᑲᓯᐅᑎᑦᓯᒋᑦ ᐱᔭᐅᓯᖑᕗᑦ ᓄᓇᕗᖅᒥ ᐊᒻᒪᓗ ᓄᓇᕕᒻᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ ᐊᐅᓚᑦᓯᔭᐅᕕᓐᓂ, ᒧᒥᐊ ᓯᓕᒻᒥᑎᑦᑐᒍ ᐃᒪᖅᐱᒥ ᑕᑦᑐᕈᑎᐅᑉ ᑕᕆᐅᖓᓂ ᐊᐅᓚᑦᓯᔭᐅᕕᓐᓂ. ᖃᓄᑐᐃᓐᓇᖅ ᐃᓱᒪᓂᖅ ᐊᖏᑦᓕᑎᓇᓂᖅᒥᒃ ᑕᖅᑫᒥᐅᑕᖅᓄᑦ TAC-ᓂᒃ ᐃᓱᒪᒍᑕᐅᒋᐊᖅᑲᖅᖁᖅ ᓱᖅᖁᐃᓇᐊᖅᑲᖅᓂᒥᒃ ᖁᐳᑎᓖᑦ ᐊᑐᐃᓐᓇᐅᓂᖓᑕ ᓈᒻᒪᓚᖓᓂᖓᓂᒃ; ᓱᓇᒥᑦᑐᑦ ᖁᐳᑎᓕᓐᓂᒃ ᐱᖅᑲᓯᐅᑎᒍᑏᑦ ᐱᔭᐅᓴᓂᐊᖅᐸᑕ, ᑕᖅᑫᒥᐅᑕᖅᓂᒃ ᐃᖅᑲᓗᓐᓂᐊᓂᖅ ᐅᒃᑯᐊᑕᐅᒐᔭᖅᒪᑦ ᐊᒻᒪᓗ ᒧᒥᑦᑐᒥᑦ ᑌᒣᒐᔭᖅᐸᑕ ᑕᒪᓐᓇᓴᓐᓇᖅ ᓯᓚᒐᔭᖅᒥᓯᓂ.

ᐱᑦᔪᔨᕕᒃ ᐱᔪᒪᕗᖅ ᐋᖅᑭᑕᐅᖅᑯᔭᖏᓐᓂᒃ ᑲᒪᕕᐅᔪᑦ ᐱᑦᔪᑎᓕᓐᓂᒃ TAC-ᓂᒃ ᑕᒪᖏᓐᓄᑦ ᑭᖑᑦᓚᓴᔭᓄᑦ EAZ ᐃᓗᐊᓂ. ᑲᑎᒪᔨᐅᕖᑦ ᐃᓱᒪᓕᐅᖅᓂᒍᑎᖏᑦ TAC-ᒥᒃ ᐱᑦᔪᑏᑦ ᐃᓱᒪᓇᖅᑎᓯᓚᖕᒐᔭᑦ_ᐃᒪᖅᐱᒧᑦ (ᑕᑦᑐᕈᑎᐅᑉ ᑕᕆᐅᖓᓄᑦ) ᐊᒻᒪᓗ NSA/NMR-ᒧᑦ (NU/NK-ᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ ᐊᐅᓚᑦᓯᔭᐅᕕᓐᓄᑦ) ᐊᕕᑦᑐᑎᓯᒍᑎᐅᑉ ᓱᖅᖁᐃᓂᖅᑲᓂᒃ. ᓱᓕᒥᐊᑦᓚᒃ, ᐱᑦᔪᔨᕕᒃ ᐱᔪᒪᒻᒥᔪᖅ ᑲᑎᒪᕕᐅᔪᑦ ᑐᑭᑖᖏᓐᓂᒃ ᐊᑦᑐᐊᔪᓂᒃ ᐊᓐᓂᑐᕈᑏᑦ ᖃᓄᐃᓪᓗᐊᓂᖓᓄᑦ NU/NK-ᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ ᐊᐅᓚᑦᓯᔭᐅᕕᓐᓂ.

TACs:
ᓇᓪᓕᑳᕈᑎᔅᓴᖅ 1: **(ᐊᑐᖅᑯᓇᖅᑐᖅ)** ᒪᑭᑕᑦᓯᓂᖅ ᑕᒪᖏᓂᒃ TAC-ᓂᒃ ᐅᓪᓗᒥ ᖃᓄᐃᓪᓗᐊᓂᖏᑦ ᑌᒣᓕᖓᔅᓴᓇᖅᑎᑐᕐᒋᑦ, ᐃᓕᑕᕐᓯᒪᑐᓂ ᑖᒃᑯᐊ ᑭᖑᕐᒧᑦ ᐊᐅᓚᑕᐅᓕᓂᐅᓂᖓᓂᒃ 15%-ᒥᒃ ER-ᖅᑲᖅᑎᓯᒋᑦ, ᓯᕗᑦᓕᖅᑲᓯᒪᑎᑦᑐᒍ ᓯᖅᑭᓂᕙᓯᓐᓂᖏᓐᓂ ᐅᓄᖅᓂᐹᖓᓄᑦ ER-ᒥᒃ 20%-ᒥᒃ, 15%-ᒥᓪᓗ ᐊᒻᒪᓗ 10%-ᒥᒃ ᑖᒃᑯᐊ ᖃᓄᐃᖕᖏᑐᑦ, ᐃᑉᐱᒋᑦᓯᐊᕆᐊᓖᑦ ᐊᒻᒪᓗ ᐊᑦᑕᓇᓕᖅᑐᑦ ᐃᓂᖏᓐᓂᑎᓪᓗᒋᑦ ᐃᓈᒃᑯᑦ.

ᐃᓱᒪᑕᕈᑎᓄᑦ ᑲᖑᓇᖅᑭᐅᑏᑦ ᓇᑐᓂᕐᓯᐸᑕ “ᓯᑲᑦᑐᒪᓇᒻᒥᒃ ᐊᓯᔾᔨᕈᑎᒥᒃ ᐃᖅᑲᓗᓐᓂᐊᒐᓪᓗᐊᑦ ᐊᒥᒉᓐᓂᖓᑎᒍᑦ” (ᖁᒻᒧᑦ ᐊᒻᒧᑑᓐᓃᑦ, ᐊᖏᓂᖅᓴᐅᐸᑦ 25%-ᒥᒃ) ᓇᓪᓕᑐᐃᓐᓇᖏᓐᓄᑦ ᑭᖑᖅᓚᓄᑦ WAZ-ᒥ ᐅᕝᕙᓘᓐᓃᑦ EAZ-ᒥ, DFO-ᑯᑦ ᑐᓯᕌᖅᑐᑦ ᐋᖅᑭᒋᐊᕈᑎᒥᒃ ᑲᑎᒪᔨᐅᕖᑦ ᑕᒪᑐᒨᓇ ᑐᑭᑖᖅᓯᒪᔨᖏᓐᓄᑦ ᐊᒻᒪᓗ ᐋᖅᑭᑕᐅᖅᑯᓯᒪᔭᖏᓐᓄᑦ.

ᐅᖅᑲᓇᐊᑭᑦᓕᑎᕈᑎ ᑐᓯᖅᐅᑎᒧᑦ

ᑕᒪᓐᓇ ᐃᓱᒪᒋᑦᓱᒍ, ᐱᑦᔪᔨᕕᒃ ᐱᔪᒪᕗᖅ ᑲᑎᒪᔨᐅᕕᓐᓂᒃ, 2018/19-ᒥ ᐃᖅᑲᓗᓐᓂᐊᓂᕐᒧᑦ:

1) ᐊᖏᖅᑲᑕᐅᓂᖓᓂᒃ ᒪᑭᑕᑦᓯᓂᖅᒥᒃ ᐃᓘᓐᓈᒍᑦ TAC-ᓂᒃ ᑐᖕᖓᕕᖅᑲᖅᑎᑐᕐᒋᑦ 15%-ᒥᒃ ER-ᒥᒃ ᑕᒪᖏᓐᓄᑦ ᑭᖑᖅᓚᓄᑦ WAZ-ᒥ (ᓇᓪᓕᑳᕈᑎᔅᓴᖅ 3), ᐃᓱᒪᓗᓂ ᐊᓯᔾᔨᑕᖅᑐᓂᒃ ᐃᖅᑲᓗᓐᓂᐊᒐᓪᓗᐊᑦ ᐊᒥᒉᑦᓂᖓᑎᒍᑦ ᐊᖅᕿᒍᓂ ᖃᒻᒥᖅ ᐊᒻᒪᓗ ᐊᑕᐅᓯᒋᐊᑦᓚᐅᑉ ᖃᐅᔨᓴᐅᑏᑦ ᑕᖅᓯᕈᑎᖓᑕ ᖃᖓᐅᓂᒃᑯᑦ ᑭᖑᓪᓕᕆᓂ ᓄᐃᓯᒐᔭᖅᓂᖓᓂᒃ ᑐᑭᓯᒪᓯᐊᓂᖅᐅᑐᓂ TAC-ᓕᐅᕈᑎᓂᒃ.
2) ᐋᖅᑭᑕᐅᖅᑯᔭᖓᓂᒃ ᐱᑦᔪᑎᓕᒻᒥᒃ TAC-ᓂᒃ ᑕᒪᖏᓐᓄᑦ ᑭᖑᖅᓚᓄᑦ EAZ-ᒥ, ᐃᓱᒪᓗᑎᒃ ᐃᓱᒣᓐᓈᓗᒃ ᐊᓯᔾᔨᑕᖅᓂᖓᓂᒃ ᐊᒻᒪᓗ ᓄᐃᓚᖅᑭᔭᐅᔭᒥᒃ ᓱᖅᖁᐃᓯᖕᖏᓂᖅᒥᒃ ᐊᑦᑐᐊᔪᒥᒃ ᖁᐳᑎᓕᓐᓄᑦ ᑭᖑᖅᓚᓄᑦ, ᐊᒻᒪᓗ ᐅᓪᓗᒥᐅᔪᖅ ᑕᖅᑫᒥᐅᑌᑦ ᑭᖑᖅᑐᑦ ᖃᓄᐃᖕᖏᓂᐅᑉ ᐃᓂᖓᐅᓐᓂᖓᓂᒃ. ᐱᑦᔪᔨᕕᒃ ᐊᑐᑦᑕᐅᖅᑯᔨᒐᔭᖅᑐᖅ ᓇᓪᓕᑳᕈᑎᔅᓴᒥᒃ 1; ᐊᒻᒪᓗ
3) ᐊᒥᖅᑳᕈᑎᖅᑲᖅᓂᒧᑦ ᑐᑭᑖᖏᓐᓂᒃ, ᐃᓱᒪᓇᖅᑎᑕᐅᒍᓐᓇᑎᓪᓗᒋᑦ TAC-ᓄᑦ ᐊᒻᒪᓗ ᐊᓐᓂᑐᕈᑎᔅᓴᑦ ᖃᓄᐃᓪᓗᐊᓂᖏᓐᓄᑦ;

   a. ᑐᑭᑖᕐᓂᒥᒃ ᐊᒥᖅᑳᕈᑎᖅᑲᖅᓂᐅᑉ ᓄᓇᕗᑦ ᐊᒻᒪᓗ ᓄᓇᕕᒃ ᐃᓂᓪᓚᕈᑎᖏᓐᓂᒃ WAZ ᐃᓗᐊᓂ
   b. ᑐᑭᑖᕐᓂᒥᒃ ᐊᒥᖅᑳᕈᑎᖅᑲᖅᓂᐅᑉ ᐃᓂᓪᓚᕈᑎᖏᓐᓂᒃ NU/NK-ᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ ᐊᐅᓚᓯᕙᓐᓂ
   c. ᐋᖅᑭᑕᐅᖅᑯᔭᖕᒐᓂᒃ ᐊᒃᒍᑎᑦᓯᒍᑎᓄᑦ EAZ-ᒥ TAC-ᓂᒃ NU/NK-ᒥ ᓯᖅᑭᓂᐅᑉ ᓄᐃᕕᐊᒍᑦ ᐊᒻᒪᓗ ᑕᓪᓗᕈᑎᐅᑉ ᑕᓇᐅᖕᒪᓂ ᐊᐅᓚᓯᕙᓐᓄᑦ

4) ᑐᑭᑖᖏᓐᓂᒃ ᐊᑦᑐᐊᔪᓂᒃ ᐃᖅᑲᓄᓐᓂᐊᓇᒍᑦ ᐃᑳᖕᓯᕈᑎᓂᒃ WAZ-ᒥ, ᐊᒻᒪᓗ ᐃᓯᒪᓕᐅᖅᓂᒍᑎᖏᓐᓂᒃ ᐊᑦᑐᐊᔪᓂᒃ ᖃᓄᖅ ᑲᔪᓯᑎᑦᓯᔪᑦᓴᐅᓂᖅᒍᑦ ᑖᑦᓯᒥᖕᒐ ᒪᓕᒐᖅᒥᒃ EAZ-ᒥ, ᑲᒪᒋᔭᑦᓴᐅᑎᑦᓗᒍ ᐃᓛᒃᑰᓗᓂ ᑐᑦᓯᕋᐅᑎᓂᒃ 1—3, ᖁᓛᓂ ᑕᒃᑲᓂ.

ᐊᓪᓚᓂᒃ ᐋᖅᑭᓯᐃᔪᑦ: ᐆᒪᕕᖕᓂᒃ ᐊᒻᒪᓗ ᐃᖅᑲᓄᓐᓂᐊᕕᖕᓂᒃ ᐊᐅᓚᑦᓯᔩᑦ, ᐃᖅᑲᓄᓐᓂᐊᕖᑦ ᐊᒻᒪᓗ ᐃᒪᖅᐲᑦ ᑳᓇᑕ

ᐅᓪᓗᖓ: ᕕᕈᐊᕆ, 2018

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ
ᐅᓂᒃᑳᓕᐊᖅ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᖅᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖕᒋᓐᓄᑦ (NWMB)

ᒫᔾᔨ 7, 2018

ᑲᑎᒪᔾᔪᑎᒃᓴᑦ

- ᓄᓇᕗᒻᒥ ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᖃᓄᐃᑉᐸᓚᐅᕐᓂᖏᓐᓂ
- ᖁᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ ᐱᕈᖅᐸᓪᓕᐊᓂᖓᓂ ᐅᓂᒃᑳᖅᑐᐊᖅ
- 2017–ᒥ ᐊᐅᓚᑕᐅᓂᖓᓄᑦ ᐊᑲᐅᙱᓕᐅᕈᑕᐅᔪᑦ ᐊᒻᒪᓗ ᑎᑭᐅᒪᔭᐅᔪᓐᓇᕐᓂᖏᑦ
- ᓄᓇᓕᖕᓂ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐃᓂᖅ
- ᓯᒡᔭᒥ ᐃᖃᓗᓚᕿᐊᕐᓃᑦ
- ᐅᒥᐊᕐᔪᐊᓕᕆᔨᑦ
- ᓂᐅᕕᐊᒃᓴᓕᕆᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ
- ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᐊᑐᐊᒐᖕᒐᓐᓂ ᖁᒥᕐᕈᓂᖅ

ᓄᓇᕗᒻᒥ ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᖃᓄᐃᑉᐸᓚᐅᕐᓂᖏᓐᓂ

- ᓯᕗᓪᓕᖅᐹᒥ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ ᑲᒻᐸᓂᐅᔪᑦ, ᕿᑭᖅᑖᓗᒃ ᑯᐊᐳᕆᓴᒃᑯᑦ, ᐱᓚᐅᖅᐳᑦ ᑕᐃᒪᐃᓕᖓᐃᓐᓇᕐᓂᐊᖅᑐᒥ ᑭᖑᒃᐸᓂᑦ ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓂ (ᐱᓕᕆᐊᖑᔪᒧᑦ ᓴᓂᕐᕙᒃᑕᐅᓯᒪᓂᖓ) 1987–ᒥ.

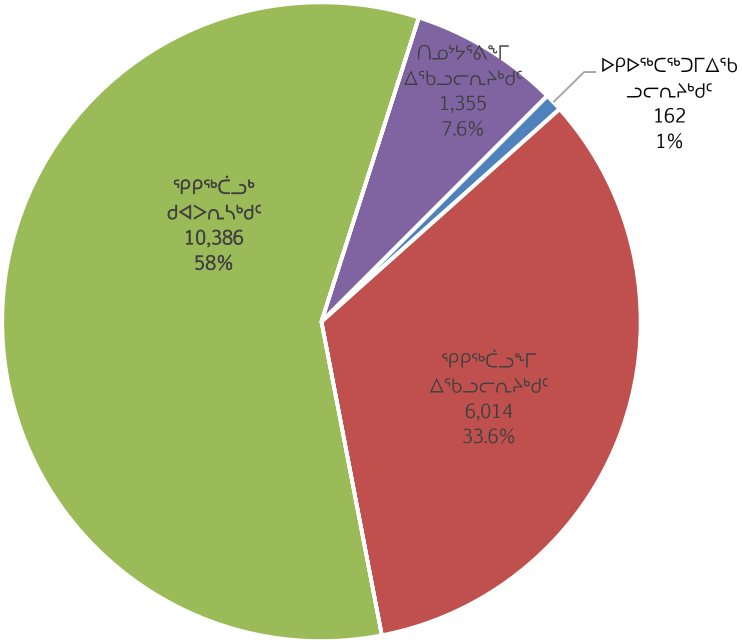
- ᕿᑭᖅᑖᓗᒃ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑎᒥᖁᖅᑎᑕᐅᔪᑦ 2001–ᒥ, ᑲᑎᑎᕆᓪᓗᑎᒃ 0A ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓂ 6–ᓂ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᑦ ᒥᑭᒥᐊᕐᓂᐊᖅᑎᒃᑯᓐᓂᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓂᑦ, ᐱᖓᓱᓂ ᑲᒻᐸᓂᐅᔪᓂᑦ, ᐊᒻᒪᓗ ᕿᑭᖅᑖᓗᒃ ᑯᐊᐳᕆᓴᒃᑯᑦ.

- ᕿᑭᖅᑖᓗᒃ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐋᖅᑭᒃᓱᖅᑕᐅᓚᐅᖅᑐᖅ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ ᐱᓕᕆᓪᓗᑎᒃ ᑲᑎᙵᔪᓄᑦ (ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ, ᓄᓇᕗᒻᒥ ᑐᙵᕕᒃᑯᓐᓂ, ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᓐᓂᓪᓗ) ᐊᔭᐅᕆᓂᕐᒧᑦ ᐅᒥᐊᕐᔪᐊᓂ ᓇᖕᒥᓂᖃᕐᓂᕐᒧᑦ.

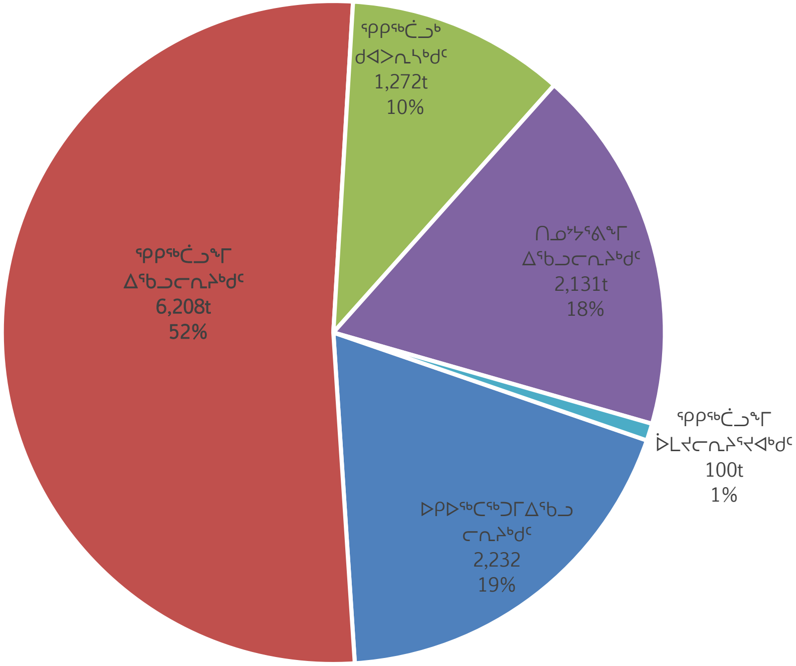
- ᕿᑭᖅᑖᓗᒃ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᓂᕈᐊᓚᐅᖅᐳᑦ ᐱᕈᖅᐸᓪᓕᐊᑎᑦᑎᒃᑲᓐᓂᕐᓂᕐᒧᑦ 70%–ᒥ ᑮᓇᐅᔭᓕᐊᓂᑦ ᑲᒻᐸᓂᒧᑦ, ᕿᑭᖅᑖᓗᒃ ᑯᐊᐳᕆᓴᒃᑯᑦ ᐊᒻᒪ ᓇᖕᒥᓂᕆᔭᐅᔪᓂ ᑲᒻᐸᓂᐅᔪᓂ ᐱᔪᒪᖃᑕᐅᓚᐅᙱᓚᑦ ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂᑦ.

ᓄᓇᕗᒻᒥ ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ 2017-ᒥ

ᑭᖕᒍᒃᐸᑦ



ᖃᓚᕋᓚᑦ



ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᓲᒃᑯᓐᓂ ᐱᕈᖅᐸᓪᓕᐊᓂᖓᓂ ᐅᓂᒃᑳᖅᑐᐊᖅ

- ᓱᕗᓪᓕᐅᓪᓗᑎᒃ ᓅᓐᓂᕐᒧᑦ ᓂᖏᖅᑕᕐᓂᕐᓂᑦ, ᐊᒻᒪᓗ ᖃᓪᓗᓈᑦ ᓄᓇᖓᓐᓂ/ᑲᓇᑕᐅᑉ ᓯᓚᑖᓂ ᐱᓕᕆᖃᑎᖃᕐᓂᕐᒥᑦ
- ᓱᕗᓪᓕᖅᐸᐅᓪᓗᑎᒃ ᐊᒻᒪ ᐱᔪᑑᓪᓗᑎᒃ 100%–ᒥ ᐃᓄᖕᓂ ᓇᖕᒥᓂᕆᔭᐅᔪᓂ ᖃᔅᓯᒐᓚᖕᓂ ᐆᒪᔪᓄᑦ ᐊᔾᔨᒌᙱᓐᓂᖏᓐᓂ ᐃᖃᓗᒐᓲᑎᓂᑦ (2)
- ᓱᕗᓪᓕᖅᐸᐅᓪᓗᑎᒃ ᐱᓂᕐᒧᑦ 100%–ᒥ ᐃᓄᖕᓂ ᓇᖕᒥᓂᖃᕐᓂᖏᓐᓂ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒥ ᐅᒥᐊᕐᔪᐊᓂ ᐊᐅᓚᑕᐅᓂᖏᓐᓂ, ᑮᓇᐅᔭᓕᐊᓂᑦ, ᓂᐅᕕᐊᒃᓴᓕᕆᓂᕐᒥ, ᐊᒻᒪ ᖃᑭᓂᐅᔪᓂᑦ.

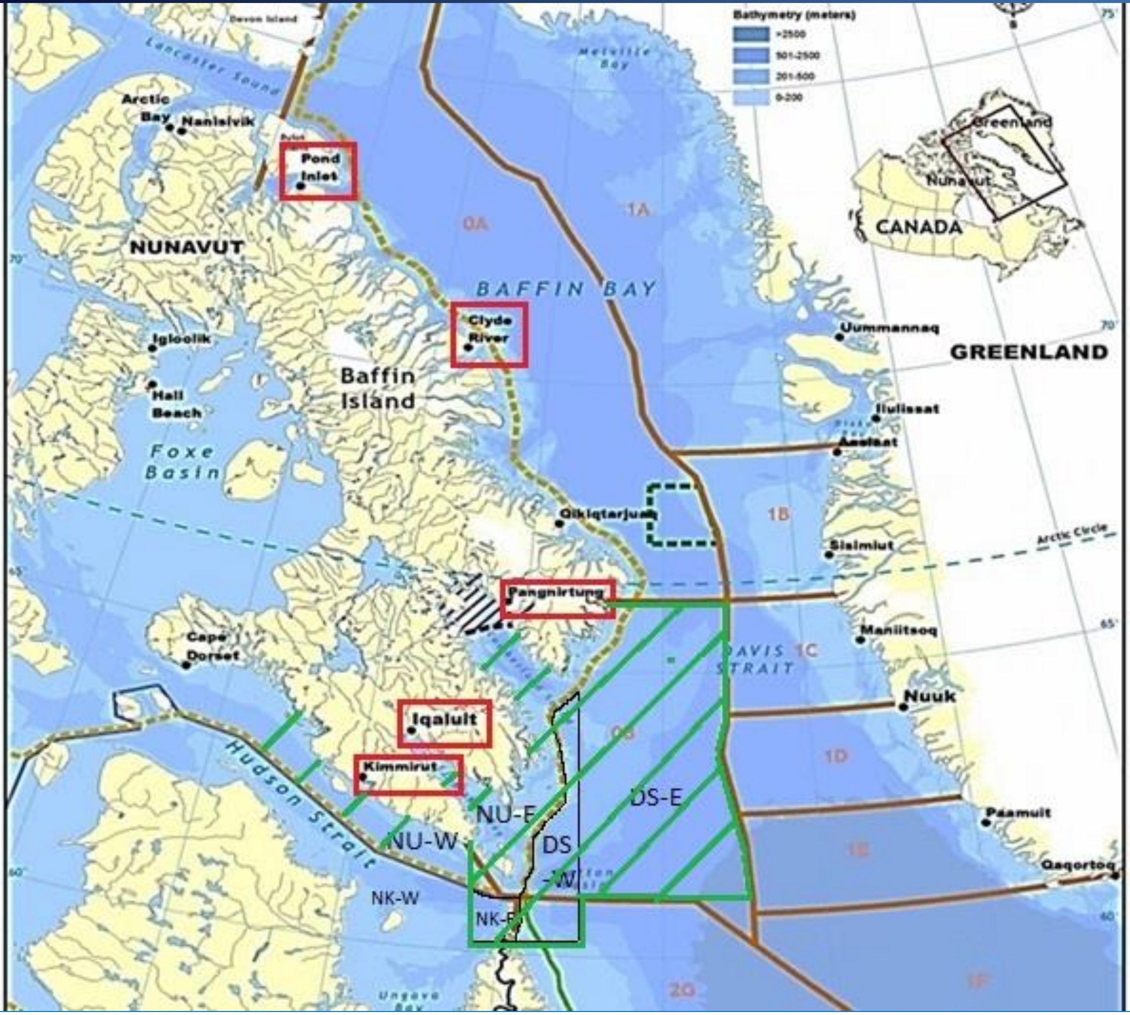
ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ ᐱᕈᖅᐸᓪᓕᐊᓂᖕᒐᓂ ᐅᓂᒃᑳᖅᑐᐊᖅ, 2

- ᓱᒃᑲᓕᕆᐊᖅᑐᒥ ᓄᓇᓕᖕᓂ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ 2016–ᒥ
- ᐊᐅᓚᑦᑎᓂᖃᓕᓚᐅᖅᑐᑦ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᕐᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖕᒋᓐᓄᑦ 2017–ᒥ
- ᒪᓕᒃᐸᓪᓕᐊᔪᒥ ᐸᕐᓇᐃᓂᖅ ᐊᒻᒪᓗ ᓅᑦᑎᓂᕐᒥ ᐊᐅᓚᑕᐅᓂᐅᔪᓂ ᐃᖅᑲᓇᐃᔮᓂᑦ ᓄᓇᒋᒻᒥ ᐱᕙᓪᓕᐊᒍᑦ
- $10 ᒥᓕᐊᓐᓂ ᑮᓇᐅᔭᓕᐅᕆᓱᐊᖅᑎᑕᐅᓂᐅᔪᓂᑦ ᐸᕐᓇᒍᑎᖕᒐ 2018-2020–ᒧᑦ

ᕿᑭᖅᑖᓗᒻᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂ ᑮᓇᐅᔾᔭᒃᓴᓂᕐᒧᑦ ᐱᕙᓪᓕᕈᑎᑦ

- ᐱᓕᒻᒪᒃᓴᓂᖅ, ᐃᖅᑲᓇᐃᔭᖅᑎᓯᐅᕐᓂᕐᒧᑦ, ᐃᓄᖕᓂ ᐃᖅᑲᓇᐃᔮᖅᑖᕐᓂᕐᒧᑦ, ᓄᓇᓕᖕᓂ ᐊᑐᖅᐸᒃᑕᖏᓐᓂ ᐊᐅᓚᔪᓐᓇᖁᓪᓗᒋᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖓᓂ ᐊᒻᒪ ᖃᑭᓂᕐᒧᑦ ᑐᓐᓂᖅᓴᐃᓂᖓᓂ
- ᐊᖏᒡᓕᒋᐊᖅᑐᒥ ᓄᓇᕗᒻᒥᐅᓂ ᐃᖅᑲᓇᐃᔮᓂᑦ ᐱᕙᒃᓴᖃᕐᓂᐅᔪᓂᑦ ᑕᒪᒃᑭᓂ ᑭᒡᓕᐅᔪᓂᑦ ᑲᒻᐸᓂᐅᑉ ᐱᖁᑎᒋᔭᖏᑦ ᐊᒻᒪᓗ ᑮᓇᐅᔭᓕᐊᖏᑦ ᐊᖏᒡᓕᕙᓪᓕᐊᑎᓪᓗᒋᑦ
- ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᒃᑲᓐᓂᐅᔪᑦ ᖃᑭᓂᕐᓄᑦ ᓄᓇᓕᖕᓄᑦᑐᙵᕕᓕᖕᓄᑦ, ᒪᑭᒪᑎᑦᑎᔪᓐᓇᖅᑐᓂ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᖃᓄᐃᓕᐅᕐᓂᐅᔪᓄᑦ
- ᑮᓇᐅᔾᔭᒃᓴᓂᕐᒧᑦ ᐃᒻᒥᒃᑰᕈᓐᓚᕐᓂᖏᓐᓄᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᑦᓗ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ.

ᑫᑭᖅᑖᓗᒻᒥ ᐃᖃᓗᓕᕆᔾᑯᓐᓂ ᐊᖑᓇᓱᒃᑐᓕᕆᔾᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᐊᒻᒪᓗ ᐃᖃᓗᖃᓱᒡᕕᓄᑦ ᐃᓂᐅᔪᑦ



ᐅᖃᓇ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔩᖁᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑎᒍᑦ, ᑫᑭᖅᑖᓗᒻᒥ ᐃᖃᓗᓕᕆᔾᑯᑦ ᑭᖑᒃᐸᕐᓂᑦ ᐊᒻᒪ ᖃᓕᕌᓕᖕᓂᑦ ᑰᑕᖃᖅᐳᑦ ᐅᑯᓇᓂ:

- ᐃᓂᐅᔪᒥ 0A–ᒥ
- ᐃᓂᐅᔪᒥ 0B–ᒥ
- ᓄᓇᕗᒻᒥ–ᑲᓇᖕᓇᒥ ᐱᓇᖕᓇᒥᓪᓗ
- ᑫᑭᖅᑖᓘᑉ ᐃᑭᖓ–ᑲᓇᖕᓇᒥ ᐱᓇᖕᓇᒥᓪᓗ
- ᑭᖑᒃᐸᓄᑦ ᐱᓇᓱᐊᕐᕕᐅᓲᑦ ᐃᓂᖓ-1

ᐊᐅᓚᑕᐅᓂᖓᓄᑦ ᐊᑲᐅᙱᑦᓯᐅᑭᑕᐅᔪᑦ 2017–ᒥ

- ᐱᔪᓐᓇᕐᑎᐊᖅᓯᒪᒐᓗᐊᖅᑎᓪᓗᒋᑦ ᐊᒻᒪ ᓄᓇᓕᖕᓂ ᕿᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᓂᑦ, ᑲᑎᒪᔨᖏᑦ ᐃᓱᒫᓘᑎᖃᓚᐅᖅᐳᑦ ᐊᖏᒡᓕᒋᐊᖅᑐᒥ ᖃᓪᓗᓈᑦ ᓄᓇᖓᓐᓂ ᐃᖅᑲᓇᐃᔮᖃᕐᓂᐅᔪᓂ ᐊᒻᒪ ᑭᖑᕙᕆᐊᕐᓂᖓᓂ ᑐᓂᓯᓂᐅᔪᒥ ᕿᓇᐅᔭᓄᑦ ᐋᖅᑭᒃᓱᖅᑕᐅᓯᒪᓂᖏᓐᓂ

- ᑲᑎᒪᔨᑦ ᑐᐊᕕᕐᓇᖅᑐᒥ ᑲᑎᒪᓂᖃᓚᐅᖅᐳᑦ ᓯᑎᐱᕆ 21-ᒥ, ᓄᖅᑲᖓᑎᑦᑎᓪᓗᑎᒃ ᐊᐅᓚᑦᑎᔨᒻᒪᕆᐅᔪᓂ ᐊᒻᒪᓗ ᐱᒋᐊᖅᑎᑦᑎᓪᓗᑎᒃ ᐊᐃᕙᐅᑕᐅᔪᓐᓇᖅᑐᒥ ᕿᓇᐅᔭᓕᐊᕆᕙᒃᑕᑦ ᖃᐅᔨᓴᖅᑕᐅᒻᒪᕆᖕᓂᖏᓐᓂ

- ᑲᑎᒪᔨᑦ ᓇᓂᓯᓚᐅᖅᐳᑦ $1.4 ᒥᓕᐊᓐᓂ ᐸᓴᒋᔭᕐᓂ ᐊᑐᕐᓂᕐᓗᒍᑕᐅᓯᒪᔪᓂ ᕿᓇᐅᔭᓂᑦ, ᒪᕐᕈᖓᓂ ᑐᐊᕕᕐᓇᖅᑐᒥ ᑲᑎᒪᖁᔨᓚᐅᖅᐳᑦ ᐅᒃᑐᐱᕆᒥ, ᐲᖅᓯᓪᓗᑎᒃ ᐊᐅᓚᑦᑎᔨᒻᒪᕆᐅᔪᓂ, ᐱᒋᐊᖅᑎᑦᑎᓪᓗᑎᒃ ᐃᖅᑲᖅᑐᒐᒃᓴᐅᓂᐅᔪᓂ ᐅᑎᖅᑎᕆᓂᕐᒧᑦ ᕿᓇᐅᔭᓂᑦ

- ᑲᑎᒪᔨᑦ ᑲᑎᒪᖃᑎᖃᖅᖢᑎᒃ ᐊᑐᖅᑐᐊᖅᑎᓂᑦ, ᑐᓂᓯᔨᐅᔪᓂᑦ, ᕿᓇᐅᔭᓕᐊᕆᕙᒃᑕᑦ ᖃᐅᔨᓴᖅᑕᐅᒻᒪᕆᓕᕆᔨᓂᑦ ᓇᓗᓇᐃᖅᓯᓂᕐᒧᑦ ᐱᖁᑎᓄᑦ ᐊᑦᑐᖏᖅᑕᐅᓂᖏᓐᓂ ᐊᒻᒪ ᓴᙱᒃᑎᒋᐊᖅᑕᐅᓂᖓᓂ ᑲᒻᐸᓂᐅᔪᒥᑦ

- BMO ᕿᓇᐅᔭᒃᑯᕕᓕᕆᔨᒃᑯᑦ: *“ᐅᓇ ᑕᑯᕙᕗᑦ ᖃᓄᐃᓐᓂᕐᓘᑕᐅᔪᒥ ᑎᒥᕈᖅᓯᒪᔪᓄᑦ ᐊᐅᓚᑕᐅᓂᖓᓂ ᐱᓕᕆᓂᖓᓄᑦ, ᖁᕕᐊᑉᐳᒍᑦ ᓴᙱᔪᒥ ᐱᓕᕆᖃᑎᖃᕐᓂᒥᑦ ᑲᑎᒪᔨᓄᑦ.”*

ᓄᑖᖑᔪᒥ ᐊᐅᓚᑕᐅᓂᖓᓄᑦ ᑎᑭᐅᒪᔭᐅᔪᓐᓇᕐᓂᖏᑦ

- ᓄᑖᖑᔪᑦ ᐃᒻᒥᒃᑰᖅᑐᓂ ᑮᓇᐅᔭᓕᐊᕆᕙᒃᑕᑦ ᖃᐅᔨᓴᖅᑕᐅᒻᒪᕆᖕᓂᖏᓐᓂ ᑲᑎᒪᔨᕋᓛᑦ ᑲᑎᖦᖢᒍ ᑲᒪᑦᑎᐊᙱᓐᓂᐅᔪᒥ
- 50% ᐅᖓᑖᓂ ᐊᐅᓚᑦᑎᔨᐅᔪᓄᑦ ᐊᒻᒪ ᐊᓪᓚᕕᓕᕆᓂᕐᒧᑦ ᑮᓇᐅᔾᔭᒃᓴᓂᖏᑦ ᐊᒻᒪ ᐊᑭᓕᒐᒃᓴᖏᑦ ᒫᓇᐅᔪᒥ ᓄᓇᕗᒻᒥ
- ᑕᒪᕐᒥᒃ ᐅᒥᐊᕐᔪᐊᓕᕆᔨᑦ ᐊᐅᓚᑕᐅᔪᑦ ᐃᓗᐊᒍᑦ, ᐱᖓᓱᑦ ᑐᕌᖓᓂᓕᖕᓂ ᐃᖅᑲᓇᐃᔭᖅᑎᓖᑦ ᓄᓇᕗᒻᒥ
- ᑕᒪᕐᒥᒃ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ ᐊᐅᓚᓂᖓ ᐃᓗᐊᒎᓕᖅᐸᑦ, ᓇᒃᓴᖅᑐᒥ ᖄᒃᑲᓐᓂᐊᒍᑦ $1.8 ᒥᓕᐊᓐᓂ ᐊᕐᕌᒍᑕᒫᒥ ᑮᓇᐅᔭᓕᐅᕐᓂᕐᒧᑦ 2018 ᐅᖓᑖᓄᑦ
- ᐱᔭᕇᖅᓯᓂᖓ ᒥᑦᑎᒪᑕᓕᖕᒥ ᑎᑎᕋᕐᕕᖓᓂ ᐃᒡᓗᕐᔪᐊᒥ ᐱᔨᕆᐊᕐᒍᓂᐊᖅᑐᒥ 100%–ᕐᔪᐊᓄᑦ ᐃᓄᖕᓂ ᐱᓕᕆᔨᓂᑦ (ᐱᓕᕆᔨᐅᔪᑦ 12-ᓂ), ᑎᑎᕋᐅᔭᖅᑕᐅᓂᖓᓄᑦ 100% ᓄᓇᕗᒻᒥ ᑲᒻᐸᓂᐅᔪᒥ (ᐃᖃᓗᐃᑦ)

ᓄᑖᖑᔪᒥ ᐊᐅᓚᑕᐅᓂᖓᓄᑦ ᐊᖏᖅᓱᒪᓂᐅᔪᓂᑦ

- ᐸᕐᓇᒍᑎ ᐱᓕᕆᐊᖑᕙᓪᓕᐊᕗᖅ ᓅᓐᓂᕐᒧᑦ ᑕᓪᓕᒪᓂ ᐊᐅᓚᑦᑎᔨᒻᒪᕆᐅᔪᓂ ᐃᓂᐅᔪᓂᑦ ᓄᓇᕗᒻᒧᑦ 2018-2019–ᒥ
- ᓴᖅᑭᑎᑕᐅᓂᐅᔪᒥ ᓄᑖᓂ ᐊᖏᒡᓕᒋᐊᖅᑎᑦᑎᓂᕐᒧᑦ ᓂᐅᕕᐊᒃᓴᖃᕐᓂᐅᔪᓄᑦ/ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ ᐊᐅᓚᓂᖓᓐᓂ ᐊᒻᒪ ᓄᓇᓕᖏᓂ ᐱᕈᖅᓴᐃᓂᖓᓄᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖓᓂ
- ᐊᖏᖅᓱᒪᓂᐅᔪᒥ ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᓂᑦ ᒥᑭᓛᖑᓂ $15 ᒥᓕᐊᓐᓂ ᓄᓇᓕᖕᓂ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᕐᒥ ᐊᕐᕌᒍᓄᑦ ᑕᓪᓕᒪᓄᑦ.
- ᓄᑖᒥᒃ ᐅᒥᐊᕐᔪᐊᒥ ᐊᖏᖅᓱᒪᓂᐅᔪᒥ ᕿᒥᕐᕈᔭᐅᓂᐊᕐᓂᖓ ᐃᓱᒻᒥᖅᑐᐃᔨᓂᑦ ᐊᒻᒪ ᑲᑎᒪᔨᓂᑦ

ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᒃᑲᓐᓂᕐᓂᐅᔪᒥ ᓄᓇᓕᖕᓂ

2016	ᓄᑖᒥ ᑎᑎᕋᕝᕕᖕᒥ ᐃᒡᓗᕐᔪᐊᒥ ᓇᑉᐸᖅᑎᓚᔭᐅᓗᓂ ᐸᖕᓂᖅᑑᒥ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖕᒍᔪᓄᑦ 2016–ᒥ	ᐸᖕᓂᖅᑑᖅ	$1,400,000
2017	ᓄᑖᒥ ᑎᑎᕋᕝᕕᖕᒥ ᐃᒡᓗᕐᔪᐊᒥ ᓇᑉᐸᖅᑎᓚᔭᐅᓗᓂ ᒥᑦᑎᒪᑕᓕᖕᒥ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖕᒍᔪᓄᑦ 2017–ᒥ	ᒥᑦᑎᒪᑕᓕᒃ	$1,300,000
2017	ᐃᖃᓗᓕᕆᓂᖏᑦ ᐃᒡᓗᕐᔪᐊᖅᓅᓇᓕᖕᓂ ᖁᐊᒃᑯᕕᒃ ᐃᑲᔪᖅᓱᖅᑕᐅᔪᖅ ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂᑦ ᒥᑦᑎᒪᑕᓕᖕᒥ	ᒥᑦᑎᒪᑕᓕᒃ	$715,000*
2017	ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖕᒍᔪᓄᑦ ᐊᑐᖅᑐᐊᖅᑎᑕᐅᓂᖓᓄᑦ ᐃᑲᔪᖅᓯᐃᓂᖏᑦ ᐊᒪᕈᖏᑦ	ᐃᖃᓗᐃᑦ	$310,000
2018	ᐋᖅᑭᐅᒪᐃᓐᓇᖅᑎᑦᑎᓂᖅ ᐃᒡᓗᕐᔪᐊᖓ ᐊᒻᒪ ᑎᑎᕋᕝᕕᖓ	ᑲᖏᖅᖢᒑᐱᒃ	$1,100,000
2019	ᓄᓇᓕᖕᓂ ᐃᒡᓗᕐᔪᐊᖅ	ᑭᒻᒥᕈᑦ	$600,000
2020	ᐊᓯᖏᑦ ᓄᓇᓖᑦ	ᓄᓇᕗᑦ	$600,000
	ᑲᑎᓪᓗᒋᑦ		**$6,035,000**

"ᓄᓇᓕᖕᓂ ᖁᐊᒃᑯᕕᖓ ᐱᓕᕆᐊᒃᓴᖅ ᑮᓇᐅᔭᖃᖅᑎᑕᐅᔪᖅ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᓐᓂ, ᐃᑲᔪᖅᓱᖅᑕᐅᔪᖅ ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᓐᓂᑦ

ᑰᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᖅ ᓇᐃᓈᖅᓯᒪᔪᖅ ᐊᒻᒪᓗ ᓇᓚᐅᑦᑖᕆᔭᐅᔪᖅ

ᓇᓚᐅᑦᑖᕆᔭᐅᔪᖅ

	2016	2017	2018	2019	2020	ᑲᑎᓪᓗᒋᑦ
ᓄᓇᓕᖕᓂ ᐊᑐᖅᐸᒃᑕᖏᓐᓂ ᐊᐅᓚᔪᓐᓇᕐᑯᓪᓗᒋᑦ	$1,400	$1,710	$1,000	$600	$1,100	**$5,810**
ᓯᒡᔭᒥ ᐃᖃᓗᒐᓱᐊᕐᓃᑦ	$50	$110	$250	$250	$250	**$910**
ᓂᖏᖅᑕᕐᓂᑦ	$1,450	$1,400	$1,500	$1,750	$2,000	**$8,100**
ᑲᑎᓪᓗᒋᑦ	**$2,900**	**$3,140**	**$2,750**	**$2,600**	**$2,750**	**$14,810**

ᐊᖏᓂᐅᔪᑦ $000–ᒦᑦᑐᑦ

ᓯᒡᔭᒥ ᐃᖃᓗᒐᓱᐊᕐᓂᓂᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖓ

- ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᖅ PFL–ᒧᑦ 2017–ᒥ ᐊᖏᓪᓕᒋᐊᖅᓯᓂᕐᒧᑦ ᓇᖕᒥᓂᕆᔭᐅᓂᖓᓂ 32%–ᒧᑦ
- ᐱᕙᓪᓕᐊᑎᑕᐅᑦᑎᐊᕐᓂᖓᓂ ᓄᓇᓕᖕᓂ ᖁᐊᒃᑯᕕᖓᓐᓂ ᐊᒻᒪ ᒥᑭᑦᑐᒥ ᐃᖃᓗᓕᕆᕝᕕᖕᒥ ᐃᔨᓄᕐᔪᐊᒥ ᒥᑦᑎᒪᑕᓕᖕᒥ
- ᑐᓂᓯᓂᖅ ᐃᑲᔪᖅᓱᐃᓂᕐᒥ ᐊᒻᒪ ᕿᓂᕐᓂᕐᒧᑦ ᐊᔾᔨᐸᓗᐊᓂ ᓄᓇᓕᖕᓂ ᖁᐊᒃᑯᕕᖕᒥ ᐱᓕᕆᐊᒃᓴᓂᑦ ᐊᓯᖏᓐᓂ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ ᓄᓇᓕᐅᔪᓂᑦ
- ᐃᖅᑲᓇᐃᔮᖅᑲᖅᑎᑦᑎᓂᖅ ᐅᓪᓗᓕᒫᖅᓯᐅᒻᒥ ᐃᖃᓗᓕᕆᔨᒥ ᐱᓕᕆᔨᐅᔪᒥ ᓄᓇᕗᒻᒥ 2018–ᒥ
- ᐅᖃᓪᓚᖃᑎᖃᕐᓂᒃᑯᑦ ᓄᓇᕗᒻᒥ ᒐᕙᒪᒃᑯᖓᓂ, ᑳᓐᓄᐊᕐ, ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓄᑦ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᒃᑯᓐᓄᓪᓗ ᑲᑐᔾᔨᖃᑎᒌᖑᔪᓄᑦ, ᐊᒻᒪ ᐃᒫᓗᖕᒥ ᖃᐅᔨᓴᖅᑎᐅᔪᓄᑦ ᑐᓴᐅᒪᖃᑦᑕᐅᑎᓕᕆᔨᐅᔪᓄᑦ ᑭᓪᓕᖃᖅᑎᑦᑎᓂᕐᒧᑦ ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᒥ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ ᓂᐅᕕᐊᖅᓴᐅᓂᐊᖅᑐᓄᑦ ᓯᒡᔭᒥ ᐃᖃᓗᒐᓱᐊᕐᓂᕐᓂᑦ ᒥᑦᑎᒪᑕᓕᖕᒥ 2018–ᒥ.

ᒥᑦᑎᒪᑕᓕᖕᒥ ᓯᔾᔭᒥ ᐃᖃᓗᒐᓱᐊᖑᓂᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖕᒪᓄᑦ ᐸᕐᓇᒍᑎ 2018/2019

ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓯᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᖅ	ᐅᓂᒃᑳᕐᓂᖕᒋᑦ	ᐊᑭᖓ
ᓯᔾᔭᒥ ᐃᖃᓗᒐᓱᐊᖑᓂᑦ ᑕᑭᔫᓪᓗᓂ ᐅᒥᐊᕐᔭᖓ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᒫᓐᓇᐅᔪᒥ ᕿᒥᕐᕈᕗᑦ ᓴᖑᑎᑦᑎᓂᖏᑦ 43' ᓴᕕᕋᔭᐅᔪᒥ ᐃᓗᐊᒍᑦ ᐅᒥᐊᕐᔪᐊᒥ ᐊᑐᖅᑕᐅᓂᐊᕐᓂᕐᒧᑦ ᒥᑦᑎᒪᑕᓕᖕᒥ ᐊᒻᒪ ᑲᖏᖅᖢᒑᐱᖕᒥ	$400,000
ᑐᓂᓯᓂᕐᒥ ᓯᑯᒃᑯᑦ ᐃᖃᓗᒐᓲᑎᒃᓴᓂᑦ	• ᐃᕐᙲᓐᓇᑲᐅᑎᒋ ᐋᖅᑭᐊᖅᑕᐅᓂᖏᓐᓂ ᐱᑕᖅᑳᓂᒃᑐᓂ ᑭᕕᒍᑎᓂᑦ • ᐹᖅᑲᖅᑕᐅᓂᖏᓐᓄᑦ ᓱᓇᒃᑯᑖᑦ ᐊᒻᒪ ᐱᓕᕆᔾᔪᑎᑦ • ᑐᓂᓯᓂᕐᓂ ᓂᒃᓯᓂᑦ, ᐊᐅᓚᓴᐅᑎᓂᑦ, ᐊᒻᒪ ᐱᔾᔪᑎᓕᖕᓂ ᐱᓕᕆᔾᔪᑎᓂᑦ • ᓂᕿᑕᑦ, ᐅᒥᐊᕐᔪᐊᓂᑦ, ᑐᕌᖓᔪᒥᒡᓘᓐᓃᑦ ᑕᓇᐅᕐᒥ ᐅᒥᐊᕐᔪᐊᒥ ᐃᓄᒃᓯᖕᒥ	$100,000
ᓂᐅᕕᐊᒃᓴᓕᕆᔨ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑮᓇᐅᔭᓂᒃ ᑐᓂᓯᓂᐊᖅᐳᑦ ᐊᑭᓖᓂᖏᑦ ᐃᖃᓗᒐᓱᒃᑎᓂᑦ ᖃᓕᕐᓕᖕᓂᑦ ᓄᓇᒧᐊᑐᐊᕈᑎᒃ; ᐃᑲᔫᓯᕈᑎᒃᓴᒥ ᑐᓂᔭᐅᓂᐊᖅᑐᑦ ᐱᔪᒪᓂᖃᕐᓂᐊᕐᒪᑕ ᐊᒻᒪ ᐱᕙᓪᓕᐊᑎᑦᑎᓗᑎᒃ ᐃᖃᓗᓕᕆᓂᕐᒥᑦ	$25,000
ᑎᑎᕋᕝᕕᒃᓴᒥ ᐃᓂᒃᓴᖃᕐᓂᖅ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑐᓂᓯᓂᐊᖅᐳᑦ ᑎᑎᕋᕝᕕᒃᓴᒥ ᐃᓂᒃᓴᖃᕐᓂᕐᒥᑦᑕᒪᒃᑭᓂ ᓄᓇᓕᖕᓂ ᐃᑲᔪᖅᓯᐃᓂᕐᒧᑦ ᓯᔾᔭᒥ ᐃᖃᓗᒐᓱᐊᖑᓂᑦ	ᐅᓪᓗᖅᑖᖅᑎᑕᐅᓂᐊᖅᑐᖅ

---

*ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐃᓚᖓᒍᑦ ᑮᓇᐅᔭᓂᒃ ᑭᐳᓯᓂᖃᕐᓂᐊᖅᑐᑦ ᓇᓕᒧᑎᑦᑎᓂᕐᒧᑦ ᑮᓇᐅᔭᖃᖅᑎᑕᐅᓂᖏᓐᓂ

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ
BAFFIN FISHERIES

ᓯᒃᔭᒥ ᐃᖃᓗᒐᓱᐊᕐᓂᓂᑦ ᐱᕚᓪᓕᐊᑎᑕᐅᓂᖓ

ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᖅ	ᐅᓂᒃᑳᕐᓂᖏᑦ
ᖃᐅᔨᒪᓂᖅ	ᒥᑦᑎᒪᑕᓕᖕᒥ ᐃᓕᑦᑎᓂᐅᔪᑦ ᐊᒻᒪ ᐱᓕᕆᑦᑎᐊᕈᓐᓇᕐᓂᖕᒪᓂ ᐊᑐᖅᑕᐅᓂᐊᖅᐳᖅ ᐊᓯᖏᓐᓄᑦ ᓄᓇᓕᖕᓄᑦ ᓄᓇᕗᒻᒥ
ᐱᓕᒻᒪᒃᓴᐃᓂᖅ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᖃᑎᖃᕐᓂᐊᖅᑐᑦ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐊᒻᒪ ᐃᒪᕐᒥᐅᑕᓄᑦ ᐱᓕᒻᒪᒃᓴᓂᕐᒧᑦ ᑲᓯᒪᔨᒃᑯᓐᓂ ᐃᑲᔪᖅᓱᐃᓂᕐᒧᑦ ᐱᓕᒻᒪᒃᓴᒃᑲᓐᓂᕐᓂᐅᔪᒥ ᑲᒪᒋᔭᕐᑲᕐᓂᕐᒧᑦ, ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐊᒻᒪ ᑐᖅᑯᐃᓂᕐᒧᑦ ᖃᓕᕋᓕᖕᓂᑦ, ᑕᑭᔪᒥ ᐊᐅᓚᓴᐅᑎᓂᒃ ᕿᓚᒃᓯᓂᕐᓄᑦ ᐊᒻᒪ ᓂᕿᑕᓂᑦ ᐸᕐᓇᐃᓂᕐᒧᑦ.
ᑲᓇᑕᒥ ᓂᕿᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᓕᕆᔩᑦ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑲᒪᒋᔭᕐᑲᕐᓂᐊᖅᑐᑦ ᐱᓕᒻᒪᒃᓴᐃᓂᕐᒥ ᐊᒻᒪ ᓇᓗᓇᐃᔭᐅᔾᔪᐃᓂᕐᒥ ᐱᔭᕆᐊᓕᖕᓂ ᓴᖅᑭᑎᑦᑎᓂᕐᒧᑦ ᑲᓇᑕᒥ ᓂᕿᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓂᖏᓐᓄᑦ ᐱᓕᕆᔨᓂᑦ ᐊᖏᖅᑕᐅᓯᒪᔪᓄᑦ ᐃᖃᓗᓕᕆᓂᕐᓄᑦ ᒥᑭᑦᑑᑕᐅᓂᖕᒪᓂ ᐱᓕᕆᕝᕕᖕᓂᑦ
ᓂᐅᕙᐊᒃᓴᓕᕆᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕙᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑕᑖᖅᑐᓄᑦ ᑐᓴᐅᒪᑎᑦᑎᓂᐅᔪᓂ ᓂᐅᕕᖅᐸᒃᑐᓄᑦ ᓂᐅᕙᐊᒃᓴᕐᑲᕐᓂᕐᒥ ᓯᒃᔭᒥ ᐃᖃᓗᒐᓱᐊᕐᓂᓂᑦ ᖁᑦᑎᓛᖕᒪᓂ ᐱᐅᓂᖕᒪᓂ, ᐊᒡᒐᒧᑦ–ᐱᔭᐅᔪᓄᑦ ᐃᓄᖕᓂ ᐃᖃᓗᒐᓱᒃᑎᓂᑦ.

ᓱᒡᓴᒥ ᐃᖃᓗᒐᓱᐊᕐᓂᓂᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖕᒐᓄᑦ ᐸᕐᓇᒍᑎ

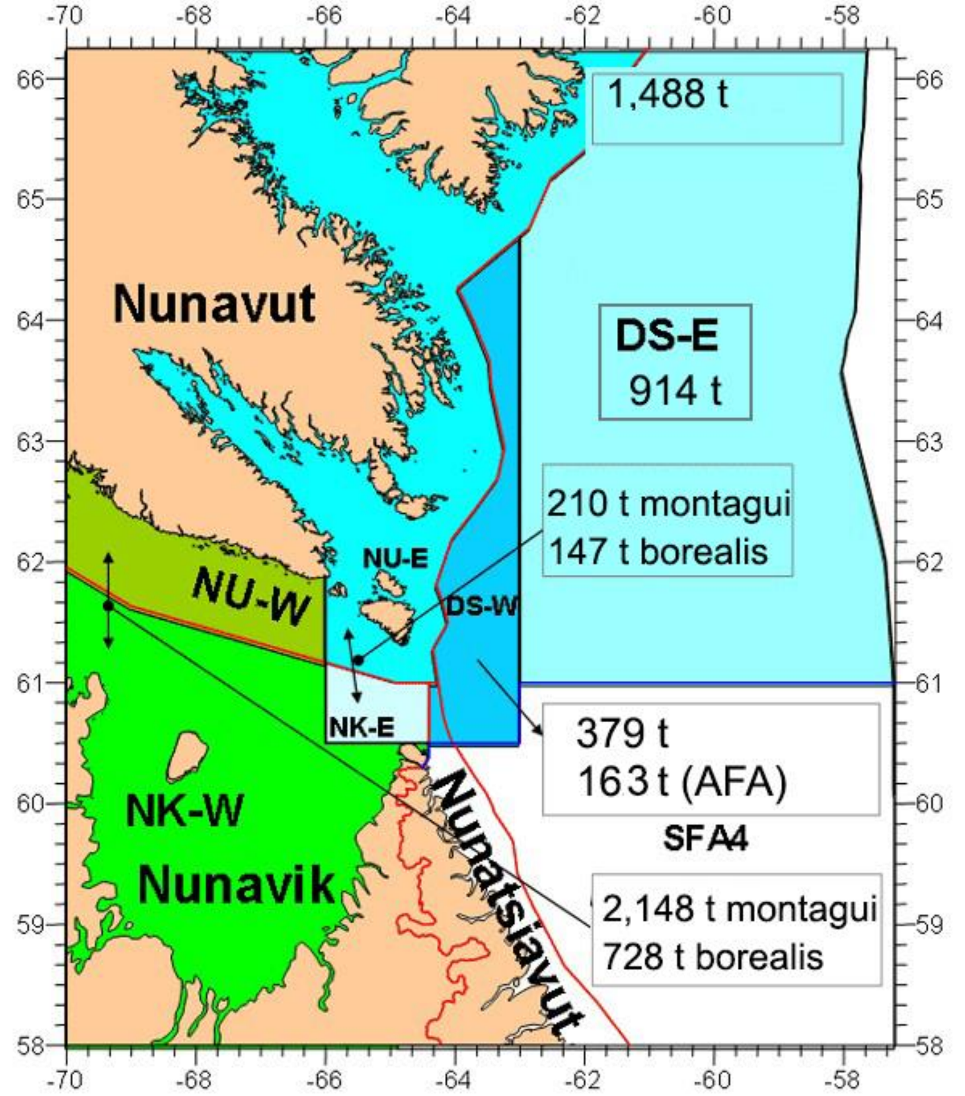
ᑮᓇᐅᔭᓂᒃ ᐱᕈᖅᑎᑦᑎᓇᓱᖕᓂᕐᒧᑦ ᑐᖅᑯᖅᓯᓂᐅᔪᖅ	ᐅᓂᒃᑳᕐᓂᖕᒋᑦ
ᓇᒦᓐᓂᖕᒋᓐᓂ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᓚᐅᖅᐳᑦ ᓂᐅᕕᐊᒃᓴᐅᔭᖅᑐᕐᓂᐊᕐᓗᓂ ᖃᐅᔨᓴᖅᑕᐅᓂᖕᒐᓂ ᐃᓱᒪᓕᐅᕆᓂᕐᒧᑦ ᐱᔪᓐᓇᕋᔭᕐᓂᖕᒐᓂ ᐅᒥᐊᕐᔪᐊᒃᑯᑦ/NEAS ᐅᓯᑲᑦᑕᖅᑏᑦ ᐅᓯᖃᑕᐅᔪᓐᓇᕐᓂᖕᒐᓂ ᐸᕐᓇᒍᑎ ᐅᓯᓂᕐᒧᑦ ᖁᐊᖑᔪᓂ ᖃᓕᕋᓕᖕᓂᑦ ᓂᐅᕕᐊᒃᓴᐅᔭᖅᑐᕐᓂᐊᖅᑐᓄᑦ
ᖃᐅᔨᓴᕐᓂᖅ ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᓂᖕᒐ	OTC ᐅᑭᐅᖅᑕᖅᑐᒥ ᐱᓕᕆᖃᑎᖃᕐᓂᖅ ᐃᑲᔪᖅᓱᐃᓂᕐᒧᑦ ᒪᑭᒪᑎᑕᐅᔪᓐᓇᕐᓂᖕᒐᓂ, ᓇᓗᓇᐃᖅᓯᓂᕐᒧᑦ ᐱᐅᓛᓂ ᐃᖃᓗᒐᓱᒡᕕᐅᔪᓂᑦ ᐃᓂᐅᔪᓂᑦ, ᐱᓇᓱᐊᙱᑕᒥᓂᑦ ᐱᑦᑕᐃᓕᓂᕐᒧᑦ ᐊᑲᐅᙱᓕᐅᕈᑕᐅᔪᑦ.
ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᓅᑕᐅᓂᖕᒐᓂ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᓅᑎᕆᔪᓐᓇᕐᓂᕋᐃᔪᑦ ᑕᕆᐅᕐᒥ ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᓱᓵᒧᑦ ᓄᓇᓕᐅᔪᓄᑦ ᐃᑲᔪᖅᓱᐃᓂᕐᒧᑦ ᑮᓇᐅᔭᓕᐅᕈᑕᐅᔪᓐᓇᕐᓂᖕᒋᓐᓄᑦ.
ᑲᑎᖅᓱᐃᔨᐅᔪᑦ ᐅᒥᐊᕐᔪᐊᑦ	ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᕿᒥᕐᕈᓂᐊᕆᕗᑦ ᐃᓯᒪᓂᐅᔪᒥ ᑲᑎᖅᓱᐃᓂᐊᖅᑐᒥ ᐅᒥᐊᕐᔪᐊᒥ ᐅᓯᓂᕐᒧᑦ ᐊᖑᓇᓱᒃᑐᓕᕆᔨᒃᑯᓐᓂ ᒥᑭᒋᐊᕐᓂᐊᖅᑎᓂᓪᓗ ᑲᑐᔾᔨᖅᑎᕌᖑᔪᓂᑦ ᐊᖑᓇᓱᖕᓂᐅᔪᓂᑦ ᐊᒻᒪ ᐃᖃᓗᒐᓱᖕᓂᕐᓄᑦ ᓱᓇᒃᑯᑖᖕᒋᓐᓂ ᓄᓇᓕᖕᓄᑦ, ᐊᐅᓪᓚᖅᑎᑦᑎᓗᑎᒃ ᖃᓕᕋᓕᖕᓂᑦ ᓂᐅᕕᐊᒃᓴᐅᔭᖅᑐᖅᑎᓪᓗᒋᑦ

ᓂᐅᕕᐊᒃᓴᐅᓯᖅᑐᕐᓂᐊᖅᑐᓂ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ ᐸᕐᓇᒍᑎᖓ ᑐᖀᒐᒃᓴᐃᑦ

ᑐᖀᒐᒃᓴᐃᑦ:

1. ᐊᖑᓇᓱᒃᐸᒡᓗᑎᒃ ᑕᒪᒃᑭᓂ ᓴᓂᕐᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔭᒃᑯᓐᓂᑦ–ᓇᖕᒥᓂᕆᓯᐅᔪᓄᑦ ᐅᒥᐊᖅᔪᐊᓄᑦ
2. ᐊᖏᓛᖑᔪᒥ ᐊᑐᕐᓗᓂ ᐃᓄᖕᓂᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖃᕐᓂᖓᓂ
3. ᐊᖏᒡᓕᒋᐊᕐᓗᒍ ᑮᓇᐅᔭᓕᐊᕆᔭᖏᓐᓂᑦ

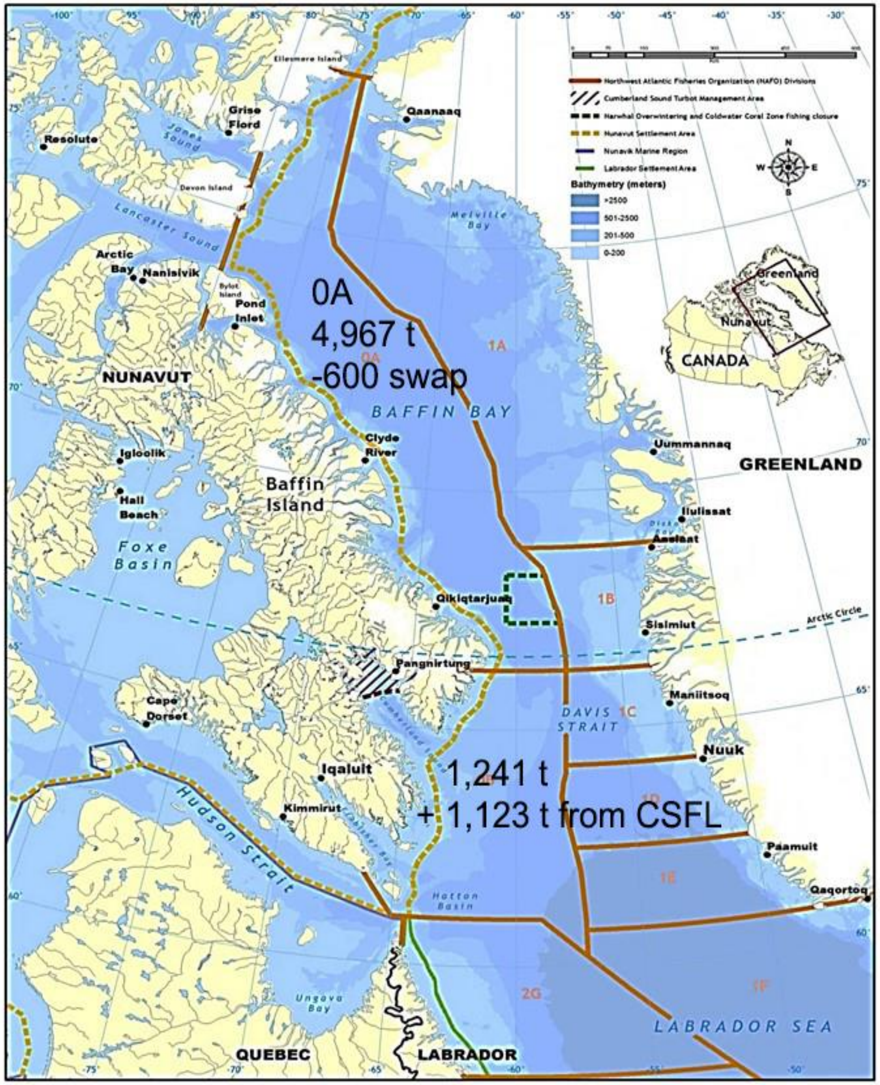
ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᓂᒃᑯᓐᓂᑦ ᑭᖑᒃᐸᐃᑦ 2017



- 100%-ᑲᓴᖕᒥ ᐱᓚᐅᖅᐳᑦ; ᐃᓚᐅᑎᑕᐅᓇᓂ: ᐊᑦᑎᒃᑐᒥ-ᐱᐅᓂᖓᓂ ᐅᖓᕙ ᒫᓐᑕᐃᒡᒥ
- ᐃᑲᔪᖅᓱᐃᓂᖅ ᐅᑭᖅᑕᖅᑐᒥ ᐃᖃᓗᒐᓱᐊᕐᓂᒃᑯᑦ ᑲᑐᔾᔨᖃᑎᒋᑦ ᕿᑭᖅᑖᓗᑉ ᐃᑭᖓ-ᐱᓇᖕᓇᒥ-ᒥ 163 ᑕᓐᓂᑦ
- ᒪᑭᕕᖕᒥ ᐱᓕᕆᖃᑎᖃᖅᓗᑎᒃ ᐃᑲᔪᕐᓂᕐᒧᑦ ᐱᓇᓱᐊᕐᓂᖏᓐᓂ 150 ᑕᓐᓂ
- ᑭᖑᒃᐸᓄᑦ ᐱᓇᓱᐊᕐᕕᐅᓲᑦ ᐃᓂᖓ 1 ᒫᓐᓇᐅᔪᒥ ᑮᓇᐅᔭᓕᐅᕈᑕᐅᙱᑦᑐᖅ, ᑭᓯᐊᓂ ᕿᓂᕐᓂᕐᒥ ᐸᕐᓇᐃᔪᑦ

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᖃᓕᕋᓕᑦ 2017

- ᐱᓚᐅᖅᐳᑦ 100%–ᑲᓴᖕᓂ ᓴᓂᕐᕙᒃᑕᐅᓯᒪᓂᖏᓐᓂ
- 600 ᑕᓐᓂ 0A ᖃᓕᕋᓕᖕᓂᑦ ᓛᐸᑐᐊᒥ ᐃᖃᓗᒐᓱᒃᑎᓂ ᑲᑐᔾᔨᖃᑎᒌᖏᑦ ᑭᖑᒃᐸᓄᑦ ᑲᒻᐸᓂ–ᓄᑦ 1350 ᑕᓐᓄᑦ ᑭᖑᒃᐸᓄᑦ
- ᐃᑲᔪᖅᓱᐃᓂᖅ ᑎᓄᔾᔪᕐᕕᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ–ᑯᓐᓂ ᓇᓗᓇᐃᖅᓯᓂᕐᒧᑦ ᐱᔪᓐᓇᖅᑕᐅᓂᖏᓐᓂ ᐃᓄᖕᓄᑦ–ᓇᖕᒥᓂᕆᔭᐅᔪᓄᑦ ᑲᒻᐸᓂᐅᔪᒥ



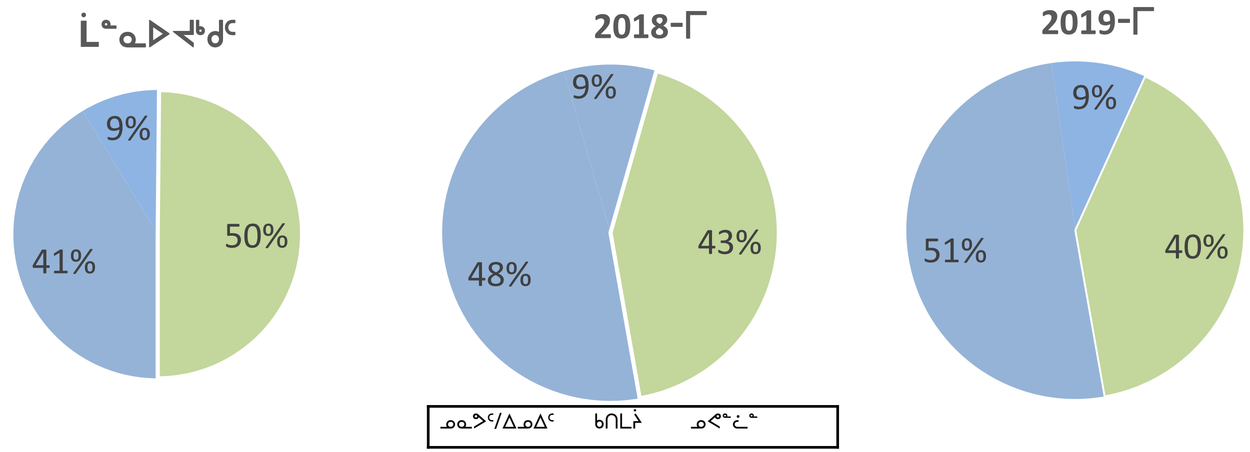
ᐅᒥᐊᕐᔪᐊᓕᕆᔨᑦ

ᑐᕌᒐᒃᓴᐃᑦ:

1. ᐊᖏᓪᓕᕆᐊᕐᓗᒍ ᐅᓄᕐᓂᖏᓐᓂ ᓄᓇᕗᒻᒥᐅᓂ ᐃᖅᑲᓇᐃᔮᖅᑲᖅᑐᓄᑦ ᓯᔮᒥ ᐊᒻᒪ ᑕᕆᐅᕐᓂᑦ
2. ᐱᐅᓯᕆᐊᕐᓗᒍ ᕿᑭᖅᑖᓗᖕᒥ ᐃᖅᑲᓗᓕᕆᔩᒃᑯᓐᓂ ᐱᓕᕆᓕᒪᓂᖏᓐᓂᑦ

ᐸᐳᓚᑦ+ᓘᑦ ᐸᒻᓚ ++ᕋᕿᕕᖕ! ᐱᖅ?ᓇᐱᔭᖅ+ᑦ ᐸ$ᓛᖅHᐳᕚᖕᓛᕆᑦ

ᑕᑯᓗᒋᑦ ᓄᓇᕘᒥ ᓄᕙᖕᓛᒥᓪᓗ ᕿᓐᓇᐅᔾᔭᒃᓵᓇᔭᐅᕙᒃᑐᑦ*



ᐃᓚᐅᖃᓯᐅᑎᖏᑦᑐᖅ $286,000 ᐃᓄᐃᑦ ᕿᓐᓇᐅᔾᔭᒃᓴᖏᑦ ᒥᑦᑎᒪᑕᓕᐅᑉ ᐃᒡᓗᖁᑎᖓᓂ

ᒪᕐᕉᒃ ᓄᑖᒃ ᐃᖅᑲᓇᐃᔮᒃ ᓴᖅᑭᑕᐅᓂᖏᑦ ᓄᓇᕘᒥ

ᒪᕐᕉᒃ ᐃᓚᒋᐊᕈᑕᐅᔫᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐃᖅᑲᓇᐃᔮᒃ ᓄᓇᕘᒧᑦ

---

*ᐃᓚᐅᖃᓯᐅᑎᕙᑦ ᖃᐅᔨᒪᔨᑕᓕᒫᑦ ᑭᓯᐊᓂ ᐃᓚᒋᔭᐅᓇᑎᒃ ᕿᓇᐅᔭᓂᒃ ᖃᐅᔨᓴᖅᑏᑦ ᐊᒻᒪ ᐱᕋᔭᓄᑦ ᒪᓕᑲᓕᕆᔩᑦ

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᓂᕐᒧᑦ **BAFFIN FISHERIES**

ᓯᕗᓪᓕᐅᔾᔪᐅᖁᔭᐅᕙᑦ

ᐊᑦᑕᓇᔾᔭᐃᖅᓯᒪᓂᖓᓂ
ᐃᖅᑲᓇᐃᔭᖅᑎᓯᐅᕐᓂᕐᒧᑦ
ᐸᐸᑦᑏᓐᓇᕐᓂᕐᒧᑦ
ᐊᑭᓕᖅᓱᖅᑕᐅᓂᖓ
ᖁᕝᕙᖅᓴᐃᓂᖓ

ᐊᑦᑕᓇᔾᔭᐃᖅᓯᒪᓂᖓᓂ

- ᐃᖅᑲᓇᐃᔭᖅᑎᓄᑦ ᐊᑦᑕᓇᖅᑐᒦᑎᑦᑕᐃᓕᒪᓂᖅ ᓱᕋᕐᓕᐅᖏᓐᓇᐅᔭᖅᐳᖅ. ᖃᖓᓕᒫᑦᑎᐊᖅ.
- ᓄᑖᑦ ᐱᓕᕆᐊᖑᔪᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᔪᑦ:
    - ᐋᓐᓂᐊᖃᖅᑕᐃᓕᒪᑎᑦᑎᓂᕐᒧᑦ ᐊᒻᒪ ᐊᑦᑕᓇᔾᔭᐃᖅᓯᒪᓂᖓᓂ
    - ᓲᐳᖅᑐᓕᑦ ᐋᖓᔮᕐᓇᖅᑐᑦ ᐊᒻᒪ ᐃᒥᐊᓗᒃ
    - ᖃᓄᐃᙱᑦᑎᐊᕐᓗᓂ ᐃᖅᑲᓇᐃᔭᕐᓂᕐᒧᑦ
- ᓄᑖᒥ ᓇᓗᓇᐃᔭᐅᔾᔪᐃᓂᕐᒥ ᐱᓕᕆᐊᖑᔪᓂ ᐱᕆᐊᖅᑎᑕᐅᓗᑎᒃ
- ᑐᕌᒐᒃᓴᖓ 0–ᒥ ᐱᕈᓗᐊᕿᓂᐅᔪᓂᑦ ᐋᓐᓂᕐᓂᐅᔪᓂᓪᓗ ᐃᖅᑲᓇᐃᔭᕐᕕᖓᓂ.

ᕿᑭᖅᑖᓗᒃ ᐃᖃᓗᓕᕆᔩᒃᑯᑦ ᐊᑦᑕᓇᔾᔭᐃᖅᓯᒪᓂᖓᓂ ᐱᓕᕆᑦᑎᐊᕐᓂᖏᑦ

	ᐊᓯᐅᔨᓂᐅᔪᒥ ᐊᑯᓂᐅᓂᖓᓂ ᖃᓄᐃᓐᓂᐅᔪᓂ ᐅᓄᕐᓂᖓ	ᑎᑎᕋᖅᑕᐅᔪᓐᓇᖅᑐᐃᑦ ᖃᓄᐃᓐᓂᐅᔪᓂ ᐅᓄᕐᓂᖓᓂ	ᐊᕿᑦᑐᒥ ᑕᖕᒋᓕᑦ ᖃᓄᐃᓐᓂᐅᔪᓂ ᐅᓄᕐᓂᖓᓂ
ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔩᒃᑯᓐᓂ	1.70	1.70	1.70
ᓅᕙᓐᓛᓐ ᐊᒻᒪ ᓛᐸᑐᐊ ᐃᖃᓗᒐᓱᒃᑕᐅᓯᒪᔪᑦ	2.10	3.20	1.10
ᓅᕙᓐᓛᓐ ᐊᒻᒪ ᓛᐸᑐᐊ ᐃᖃᓗᓕᕆᓂᑦ	3.30	4.70	1.90
ᓅᕙᓐᓛᓐ ᐊᒻᒪ ᓛᐸᑐᐊ ᑕᐃᒪᐃᒐᔪᖕᓂᖓ	1.50	2.20	1.00

ᐃᖅᑲᓇᐃᔭᖅᑎᓯᐅᕐᓂᕐᒧᑦ

ᐃᖅᑲᓇᐃᔮᖅᑖᖅᑎᑦᑎᓗᓂ ᐃᖅᑲᓇᐃᔭᖅᑎᓯᐅᕐᓂᕐᒧᑦ ᐱᓕᕆᔭᐅᔪᓂᑦ ᓄᓇᕗᒻᒥ ᓄᓇᓕᐅᔪᓂᑦ	✓
ᐊᖏᒡᓕᒋᐊᖅᓯᓗᓂ ᐅᔾᔨᕈᓯᖕᓂᕐᒥ ᑐᓴᐅᒪᑎᑦᑎᓇᓱᐊᕐᓂᕐᒧᑦ, ᓈᓚᐅᑎᒃᑯᑦ ᑐᓴᖅᓴᐅᓂᐅᔪᓄᑦ ᐊᓯᖏᑦᓗ	✓
ᐊᐅᓚᑦᑎᖃᑎᖃᕐᓗᓂ NFMTC−ᑯᓐᓂ ᓇᓗᓇᐃᖅᓯᓂᕐᒧᑦ ᐊᒻᒪᓗ ᐃᖅᑲᓇᐃᔭᖅᑎᑖᕋᓱᐊᕐᓂᕐᒧᑦ ᓄᑖᓂ ᐃᓱᓕᑦᑎᓂᑯᓂ	✓
ᐊᖏᓛᖑᑎᓐᓇᓱᐊᕐᓗᒍ ᐅᓄᕐᓂᐅᔪᓂ ᐃᖅᑲᓇᐃᔭᕐᓂᑰᙱᑦᑐᓂ ᐊᖏᒡᓕᒋᐊᖅᑎᑦᑎᓂᕐᒧᑦ ᑎᑎᕋᖅᓯᒪᓂᐅᔪᓂ ᐊᑐᐃᓐᓇᐅᔪᓂ ᐃᓄᖕᓂ ᐃᖅᑲᓇᐃᔭᖅᑎᒃᓴᓂᑦ	✓
ᐱᐅᓯᒋᐊᕐᓗᒍ ᐸᐸᑦᑏᓐᓇᕐᓂᕐᒧᑦ ᐅᖃᖃᑎᖃᕐᓂᒃᑯᑦ ᐊᒻᒪᓗ ᐱᓕᕆᔭᒧᑦ−ᐊᖏᖅᑕᐅᓯᒪᔪᓂ ᐃᖅᑲᓇᐃᔭᕐᕕᒋᕙᖕᓂᐊᖅᑕᖏᓐᓂᑦ	✓

ᐃᖅᑲᓇᐃᔭᖅᑎᓯᐅᕐᓂᕐᒧᑦ ᓄᓛᓂᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᒃᓴᓂᑦ

2016

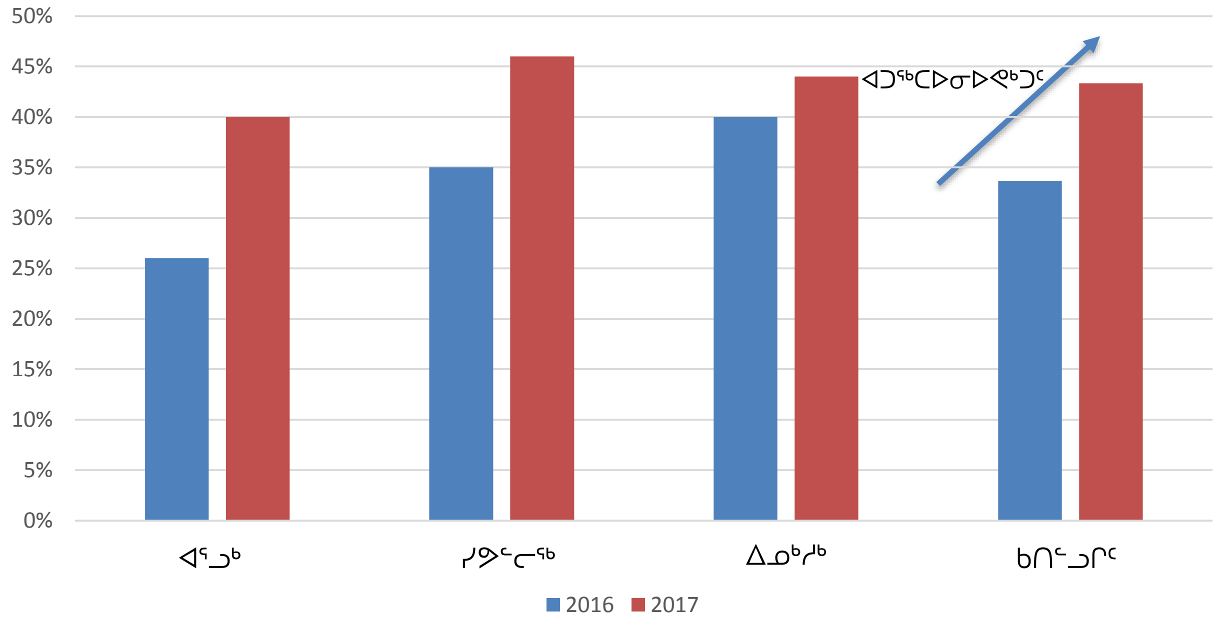
- ᑕᓪᓕᒪᓂ ᓄᓛᓂᒃ ᐃᖅᑲᓇᐃᔭᖅᑎᓯᐅᕐᓂᕐᒧᑦ ("ᐃᖅᑲᓇᐃᔭᓚᐅᖅᓯᒪᙱᑦᑐᓂ") ᐊᐅᓪᓛᕐᓂᐅᔪᓂᑦ, ᒪᕐᕉᖕᓂ ᐃᓚᐅᔪᖅᑲᐃᓐᓇᓚᐅᖅᐳᖅ ᐃᖅᑲᓇᐃᔭᖅᑕᐅᖏᓐᓇᖅᑐᓂᑦ

2017

- 15 ᐊᐅᔾᓛᕈᓐᓇᖅᓯᓚᐅᖅᐳᑦ, 12–ᖑᔪᑦ ᐃᖅᑲᓇᐃᔭᖅᑕᐅᖏᓐᓇᖅᐳᑦ ᒫᓐᓇᒧᑦ ᑎᑭᖦᖢᒍ
- ᐱᖓᓱᑦ ᓱᓕ ᐃᓚᐅᖅᑕᐅᓕᕐᓂᐊᖅᑐᑦ ᑕᒪᑐᒪᓂ ᐊᕐᕌᒍᒥ

ᐱᐅᓯᒋᐊᖅᑕᐅᓂᐅᔪᑦ 2017−ᒥ

ᐃᓄᐃᑦ ᐅᒥᐊᕐᔪᐊᓕᕆᔨᐅᔪᑦ ᐃᖅᑲᓇᐃᔮᖃᕐᓂᐅᔪᑦ %−ᖑᓪᓗᓂ ᑲᑎᓪᓗᒍ
ᐅᒥᐊᕐᔪᐊᓕᕆᔨᐅᓪᓗᑎᒃ*



*ᐃᓚᖃᖅᐳᖅ ᐃᓂᐅᔪᓂᑦ ᐃᓄᐃᑦ ᐱᖃᑕᐅᓯᒪᓂᐅᔪᓄᑦ ᓄᓇᕗᖃᕐᓂᕐᒧᑦ ᒫᓐᓇᐅᔪᒥ ᐱᓕᒻᒪᒃᓴᖅᑕᐅᕗᑦ ᐊᒻᒪ ᐸᐃᑉᐹᖅᑖᖅᑎᑕᐅᓂᕐᒧᑦ. ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐱᓕᕇᓐᓇᕆᐊᖃᖅᐳᑦ ᓇᓗᓇᐃᖅᓯᓂᕐᒥ ᐃᓄᐃᑦ ᐱᖃᑕᐅᓯᒪᔪᓄᑦ ᓄᓇᕗᖃᕐᓂᕐᒧᑦ ᐱᓕᒻᒪᒃᓴᖅᑕᐅᓂᖏᓐᓂ ᐱᓕᕆᔨᒻᒪᕆᐅᓂᐅᔪᓄᑦ ᐃᓂᐅᔪᓄᑦ.

2017–ᒥ ᐃᖅᑲᓇᐃᔮᓄᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓂᐅᔪᑦ

- ᐃᓄᐃᑦ ᐃᖅᑲᓇᐃᔮᕐᑲᕐᓂᖏᓐᓄᑦ ᐅᓄᖅᓯᒋᐊᖅᑐᑦ ᑕᒪᒃᑭᓄᑦ ᐅᒥᐊᕐᔪᐊᓄᑦ
- ᓄᑖᑦ ᐃᖅᑲᓇᐃᔮᖅᑎᑖᖑᔪᑦ ᒪᕿᐃᓪᓗᐊᖅᑐᒥ ᐅᓄᖅᓯᒋᐊᖅᐳᑦ 2017–ᒥ
- ᓯᕗᓪᓕᖅ ᐊᐅᓪᓛᕐᓂᖅ 5–ᒥ 73%–ᖑᔪᑦ ᐃᓄᖕᓂ ᐃᖅᑲᓇᐃᔭᖅᑎᕐᑲᓚᐅᖅᐳᑦ (11–ᖑᔪᑦ 15–ᓂ ᐱᔪᓐᓇᖅᑐᓂ ᐃᓂᐅᔪᓂᑦ)
- ᓄᑖᖅ ᑐᕌᒐᒃᓴᖕᒐ 75%–ᖑᕙᖅ ᐃᓄᖕᓂ ᐃᖅᑲᓇᐃᔭᖅᑎᐅᔪᓂᑦ ᑕᐃᒪᐃᑉᐸᓚᐅᖅᑐᓂ 50%–ᖑᔪᒥ
- ᐸᐸᑦᑏᓐᓇᕐᓂᕐᒧᑦ ᕐᑯᑦᑏᓐᓇᖅᐳᖅ 2017–ᒥ

ᓂᐅᕕᐊᒃᓴᓕᕆᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ

2017–ᒥ, ᕿᑭᖅᑖᓗᒻᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᑎᑭᑎᑦᑎᔪᓐᓇᖅᓯᓚᐅᖅᐳᑦ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ ᐃᓗᐊᓃᑦᑐᓄᑦ ᐅᑯᖓ:

1. ᐲᖅᓯᓂᕐᒧᑦ $1.8 ᒥᓕᐊᓐᓂ ᐱᖕᒐᓱᖕᒐᓐᓂ–ᐱᓕᕆᖃᑕᐅᔪᓄᑦ ᐊᑭᓕᖅᓱᕈᑕᐅᒃᑲᓐᓂᖅᑐᒥ
2. ᓂᐅᕕᐊᒃᓴᖃᕐᓂᐅᔪᒥ ᐊᔾᔨᐅᙱᑦᑐᖅ, ᐱᐅᓪᓚᕆᒃᑐᐊᓗᖕᓂ ᕿᑭᖅᑖᓗᒻᒥ ᓴᓇᐅᒪᓂᑦ
3. ᖃᐅᔨᒪᓂᕐᒥ ᐱᓗᑎᒃ ᐊᒻᒪᓗ ᖃᐅᔨᒪᕐᔪᐊᕐᓂᕐᒧᑦ ᓄᑖᖑᔪᒥ ᐱᓕᕆᓂᐅᔪᒥ; ᓅᑎᕆᓗᑎᒃ ᖃᐅᔨᒪᓂᕐᒥ ᐊᒻᒪᓗ ᖃᐅᔨᒪᕐᔪᐊᕐᓂᖏᓐᓂ ᐅᑭᐅᖅᑕᖅᑐᒧᑦ
4. ᓴᖅᑭᑦᑎᓗᑎᒃ ᐱᕕᒃᓴᖃᕐᓂᐅᔪᒥ ᐱᖕᒐᓱᖕᒐᓐᓂ–ᐱᓕᕆᖃᑕᐅᓗᑎᒃ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ ᑲᒻᐸᓂᐅᓗᑎᒃ ᐊᓯᖏᓐᓄᑦ ᐊᖑᓇᓱᒃᑎᐅᔪᓄᑦ.

ᓯᕗᓪᓕᖅᐹᖅ ᑕᒪᒃᑭᖅᑐᖅ ᐊᖅᑮᒍ

- ᐱᓕᕆᑦᑎᐊᕈᓐᓇᖅᑐᒥ ᐊᖅᑮᒍᒥ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᕐᒧᑦ ᐱᕈᖅᐸᓪᓕᐊᓂᖕᒪᓄᑦ
- ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔩᒃᑯᑦ ᐊᐅᓚᑦᑎᓚᐅᖅᐳᑦ 100%–ᒥ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᕐᓄᑦ ᑕᒪᒃᑭᓄᑦ ᑕᕆᐅᕐᒥᐅᑕᓄᑦ ᓂᕿᓄᑦ ᐱᔭᐅᔪᓄᑦ ᐊᖅᑮᒍᒥ
- ᐊᐅᓚᑦᑎᓚᐅᖅᐳᑦ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᕐᓂᑦ ᑕᕆᐅᕐᒥᐅᑕᓄᑦ ᓂᕿᓂᑦ ᐅᕚᓇ ᐱᖓᓱᖓᓐᓂ ᐱᓕᕆᖃᑕᐅᔪᓂ ᓚᐃᓴᓐᓯᖓᒍᑦ ᑕᐅᖅᓲᓂᕐᒧᑦ ᐊᖏᕈᑕᐅᔪᓂᑦ
- ᒥᒃᓵᒋᐊᖅᑐᒥ ᓂᒡᓕᓇᖅᑎᑦᑎᓗᑎᒃ ᑐᖅᑯᐃᓯᒪᕕᓕᓂᑦ ᐊᑭᓕᐅᑎᓂᑦ

ᐅᓯᔭᐅᓂᖓ ᐅᒥᐊᕐᔪᐊᒥ

- ᑕᒪᕐᒥᒃ ᑕᕆᐅᕐᒥᐅᑕᐃᑦ ᓂᕿᑦ ᓂᐊᕕᐊᒃᓴᐅᓚᐅᖅᐳᑦ ᐅᓯᔭᐅᓂᖓ ᐅᒥᐊᕐᔪᐊᒥ ᑐᓚᒃᑕᕐᕕᓂᑦ ᓂᐅᕙᓐᓛᓐᒥ ᐅᕝᕙᓘᓐᓃᑦ ᑲᓛᔅᓴᐅᑉ ᓄᓈᓐᓂ
- ᓂᐅᕕᖅᑐᑦ ᐱᕗᑦ ᐊᒻᒪ ᐊᑭᓖᕗᑦ ᐃᖃᓗᖕᓂ ᓄᓇᒧᐊᑐᐊᕌᖓᒥ
- ᓂᐅᕕᖅᑐᖅ ᑲᒪᒋᔭᖅᑲᖅᐳᖅ ᐃᒪᐅᑉ ᐃᓯᔭᐅᔾᔪᑎᖓᓂ, ᑲᓇᑕᐅᑉ ᓯᓚᑖᓄᑦ ᑐᓚᒃᑕᕐᕕᖕᒥ ᐅᓯᖕᑮᔭᐅᖅᑕᐅᓂᖓᓂ, ᓂᒡᓕᓇᖅᑎᑦᑎᓗᑎᒃ ᑐᖅᑯᐃᓯᒪᕕᓕᖕᓂᑦ, ᑐᑖᕐᓂᖓᓄᑦ, ᓇᓪᓕᐅᒃᑯᒫᓂᒃ, ᐊᓯᖏᓐᓄᑦ ᑮᓇᐅᔭᓄᑦ ᐅᓗᕆᐊᓇᕐᓂᖏᓐᓂ ᐊᓯᖏᓪᓗ
- ᑕᒪᓐᓇ ᐱᔪᓐᓇᖅᑎᑦᑎᕗᖅ ᐊᑭᓕᖅᑕᐅᓴᕋᐃᓐᓂᖓᓄᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖏᓐᓄᑦ.

ᓄᑖᓂ ᑭᖑᒃᐸᓂᑦ ᐱᓕᕆᓂᐅᔪᒥ

- 2017–ᒥ, ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐃᓚᓯᓚᐅᖅᐳᑦ ᔮᐸᓃᓯᒥ ᑭᖑᒃᐸᓄᑦ ᐱᓕᕆᓂᖃᕐᓂᐅᔪᒥ ᓯᕗᓪᓕᖅ ᐃᖃᓗᓕᕆᕝᕕᖓᓄᑦ
- ᓴᖅᑭᑎᑦᑎᓂᐅᔪᒥ ᓲᓯᒥ ᐱᐅᓂᖓᓂ ᑭᖑᒃᐸᓂᑦ ᐊᑐᐃᓐᓇᐅᕗᖅ
- ᐊᖏᓪᓕᒋᐊᖅᑎᑦᑎᔪᒥ ᐱᐅᓂᖓᓂ ᐊᑐᓂ ᑭᓗᒍᕌᒻᒥ ᐃᓚᖏᓐᓄᑦ ᑭᖑᒃᐸᓄᑦ ᓴᓇᓯᒪᔪᓄᑦ
- ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐃᓚᒋᐊᖅᓯᓚᐅᖅᐳᖅ ᒪᕐᕉᖕᓂ ᐊᖏᓛᖑᔪᓂ ᓂᐅᕕᖅᑎᐅᔪᓂᑦ ᔮᐹᓐᒥ ᑎᑎᕋᖅᓯᒪᓂᐅᔪᓂ ᐱᓕᕆᔾᔪᔾᔭᐅᕙᒃᑐᓄᑦ.

ᓂᐅᕕᐊᒃᓴᓕᕆᔨᑦ ᑲᑎᙵᓂᐅᔪᑦ

- ᐃᓚᒋᔭᐅᔪᑦ ᓂᐅᕕᐊᒃᓴᓕᕆᓂᕐᓂᑦ ᐊᒻᒪᓗ ᓂᐅᕕᐊᒃᓴᕆᔭᐅᔪᓐᓇᕐᓂᖏᓐᓄᑦ ᑲᑎᙵᓂᐅᔪᓂ ᑲᑎᖃᑎᖃᓚᐅᖅᐸᑦ ᓂᐅᕕᖅᑎᐅᔪᓂᑦ ᓯᓚᕐᔪᐊᓕᒫᒥ
- ᓴᖅᑭᑎᓯᒪᔪᒥ ᐱᓕᕆᐊᙳᔪᒥ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᖃᐅᑕᒫᒥ ᐱᐅᔪᒥ ᐱᔪᓐᓇᑦᑎᐊᕐᓂᐅᔪᓂ ᐅᓂᒃᑳᓕᐊᓂᑦ
- ᑲᑎᒪᖃᑎᖃᖅᓱᑎᒃ ᐱᖓᔪᖓᓐᓂ–ᐱᓕᕆᖃᑕᐅᓂᐊᖅᑐᓂ ᐱᓕᕆᔾᔪᑎᓂᑦ ᕿᓂᕐᓂᕐᒧᑦ ᐱᕕᒃᓴᖃᕐᓂᐅᔪᓄᑦ ᐃᓚᒋᐊᖅᓯᓂᖓᓄᑦ ᐱᐅᓂᖓᓂ
- ᐱᐅᑦᑎᐊᖅᑐᒥ ᓯᕗᓪᓕᖅᐹᒥ ᐊᕐᕌᒍᖓᓂ

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖅᑲᓗᓕᕆᔨᒃᑯᑦ ᓴᓂᖅᕙᖅᑕᐅᓯᒪᔪᓄᑦ ᑐᕌᒐᒃᓴᐅᔪᓄᑦ

1. ᐱᓕᕆᖃᑎᖃᖅᓗᑎᒃ ᑕᒪᒃᑭᓂ ᑎᓴᒪᓂᒃ ᓄᓇᕗᒻᒥ ᐃᖅᑲᓗᓕᕆᔨᐅᔪᓄᑦ ᐊᖏᒡᓕᕆᐊᖅᑎᑦᑎᓂᖅᒧᑦ *ᑲᑎᕐᓗᒍ* ᑳᑕᐅᔪᓂᑦ
2. ᓇᓗᓇᐃᖅᓯᓗᓂ ᐃᓄᐃᑦ ᓴᙱᔪᒥ ᓂᐱᖃᖅᓂᖕᒋᓐᓂ ᐆᒪᔪᓕᕆᓂᖅᒧᑦ ᐊᐅᓚᑕᐅᓂᖕᒋᓐᓂ
3. ᓴᖅᑭᑦᑎᓗᓂ ᓯᓚᐅᔪᒥ ᓇᖕᒥᓂᖃᖅᓂᖅᒧᑦ ᐱᔪᓐᓇᑦᑎᐊᖅᓂᖕᒐᓂ.

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᓴᓂᕐᕙᒃᑕᐅᓯᒪᔪᓄᑦ ᐊᑐᓕᖁᔭᐅᔪᓂᑦ

1. ᐊᔪᕐᓇᙱᑎᒋᐊᕐᓚᒃᑐᒍ ᐱᓕᕆᓂᐅᔪᒥ
2. ᓴᖅᑭᑎᑦᑎᓗᓂ ᑐᖂᒐᒃᓴᐅᔪᓂ ᑎᑭᐅᒪᔭᐅᔪᒃᓴᓂ ᑲᔪᓯᑎᑦᑏᓐᓇᕐᓂᕐᒧᑦ ᑰᑕᖕᒐᓂ
3. ᒥᑭᓛᖕᒐᓂ ᑕᓪᓕᒪᑦ (5) ᐊᕐᖂᒍᓄᑦ ᓴᓂᕐᕙᒃᑕᐅᓯᒪᔪᓄᑦ ᐱᓕᕆᓂᐅᔪᒥ
4. ᓅᑕᐅᓗᓂ ᐊᑦᑐᖕᒋᖅᓯᒪᑦᑎᐊᖅᑐᒧᑦ ᓴᓂᕐᕙᒃᑕᐅᓯᒪᔪᓄᑦ
5. ᐊᕕᒃᑐᕐᓗᒍ ᓯᕗᓂᒃᓴᒧᑦ ᑲᑎᓪᓗᒍ ᐱᔭᐅᔪᓐᓇᖅᑐᑦ ᐊᔾᔨᒌᖑᓂᐅᔪᓂᑦ

ᕿᑭᖅᑖᓗᖕᒥ ᐃᖅᑲᓗᓯᓕᔩᑦ

BAFFIN FISHERIES

ᑐᓂᔭᐅᔪᑦ
ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᖏᑕ ᑲᑎᒪᔨᖏᓐᓄᑦ
ᒫᔾᔨ 2018

ᐆᒧᖓ

ᑐᑭᓯᒋᐊᕈᑎᔅᓴᖅ: X

ᐃᓱᒪᓕᐅᕆᐊᓕᒃ:

ᐱᔾᔪᑎᓕᒃ: ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐅᓪᓗᒥᒧᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓂ -*ᐃᒪᕐᒥᐅᑕᓄᕐ ᐱᖁᔭᐃᑦ ᐸᖅᑭᔭᐅᔪᑦ ᐃᓚᖏᑦ*

ᐅᓪᓗᒥᒧᑦ ᖃᓄᐃᓕᖓᓕᕐᓂᖏᓐᓂ:

ᔫᓂᒥ 2016, ᑲᓇᑕᒥ ᓇᓗᓇᐃᖅᓯᓚᐅᕐᖓᑕ ᑕᓪᓕᒪᐅᓕᖅᑲᖓᔪᓂᑦ ᐸᕐᓇᐅᑎᓂᑦ ᑲᓇᑕᒥ ᓯᓚᕐᔪᐊᒥᓗ ᐃᒪᕐᓂᑦ ᐱᐅᖅᓯᐊᕈᒪᓂᕐᒥᑦ (MCTs), ᐱᖃᓯᐅᔾᔨᒐᔭᖅᑐᖅ ᐋᖅᑮᒋᐊᖃᕐᓂᑎᓐᓂ ᐃᒪᕐᒥᑦ ᐸᖅᑭᔭᕆᐊᓕᑦᑎᓐᓂ (MPAs). ᐃᒪᕐᒥ ᐸᖅᑭᑦᑎᓂᐊᕐᓂᑎᓐᓂ ᐊᑭᑐᓗᐊᓐᖏᑦᑐᓂ ᐱᐅᖅᓯᐊᕋᓯᐊᕐᓗᑎᑦ ᐃᖃᓗᐃᑦ ᐃᒪᕐᒥᐅᑕᐃᑦᑐᓂ ᐃᓂᖏᑦ, ᐅᓗᕆᐊᓇᖅᑐᒦᑦᑐᑦ ᐃᒪᕐᒥᐅᑕᐃᑦ ᐃᖏᑦᑐ, ᐊᔾᔨᐅᖏᑦᑐᓄᑦᑐ ᐱᓕᕆᐊᖑᔭᕆᐊᓖᑦ, ᐊᒻᒪᓘᓐᓃᑦ ᖁᑦᑎᓂᖅᐹᒃᑯᑦ ᐱᕈᖅᑐᕐᕕᑦ ᐸᖅᑭᔭᐅᔭᕆᐊᖃᕐᓂᖏᑦ. ᓯᕗᓪᓕᖅᐹᒥ ᓇᓗᓇᐃᖅᓯᖅᑳᕆᐊᖃᖅᑐᒍᑦ ᓇᓗᓇᐃᕈᓴᒐᔭᖅᑕᑎᓐᓂ (AOIs).

ᑲᑎᒪᑎᑦᑎᓂᖅ ᓇᓗᓇᐃᖅᑕᐅᒍᒪᒐᔭᖅᑐᒥᑦ (AOIs):

- ᐅᓂᒃᑳᕐᕕᒋᕙᑖᓚᐅᕙᑦᑎᒍᑦ ᓄᓇᕗᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ, ᓇᓗᓇᐃᕈᒪᔭᖓᓂ ᐸᕐᓇᑦᑏᑦ ᑲᑎᒪᑎᑦᑎᓚᐅᖅᓯᒪᓪᓗᑎᑦ ᐆᒪᔪᕐᓂᖅᑎᒃᑯᓐᓂ ᓄᓇᓕᓐᓂᓪᓗ 9-ᓂᑦ ᓇᓗᓇᐃᖅᑕᐅᒍᒪᒐᔭᓐᖑᐊᖅᑐᓂᑦ (ᐱᖓᓱᑦ ᐊᑐᓂ ᓄᓇᕗᒻᒥ ᐊᕕᑦᑐᖅᓯᒪᔪᓂᑦ).
- ᐅᓂᒃᑳᕐᕕᐅᕙᑖᓚᐅᖅᑐᑎᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᑦ, ᐃᒪᒥᐅᑕᓕᕆᔩᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖏᑦ ᓇᓗᓇᐃᖅᓯᒋᐊᖃᖅᑐᑦ ᐅᓪᓗᒥᒧᑦ ᐅᓂᒃᑳᕐᓗᑎᑦ ᐊᕐᕋᒍᑕᒫᖅᓯᐅᑎᖓᓐᓂ ᑲᑎᒪᕕᔾᔪᐊᓕᖅᑲᑕ ᑭᕙᓪᓕᖅ ᐆᒪᔪᓕᕆᔨᖏᑦ ᑲᑎᒪᔨᖏᑦ ᐊᒻᒪᓗ ᕿᑎᕐᒥᐅᑦ ᐆᒪᔪᓕᕆᔨᖏᑕ ᑲᑎᒪᔨᖏᑦ. ᑕᐃᒪᓐᖓᓂ ᑭᖑᓪᓕᖅᐹᒥ ᐅᓂᒃᑳᕐᕕᐅᓚᐅᕐᑕ, ᕿᑭᖅᑖᓗᒃ ᐆᒪᔪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐃᓱᒪᒋᔭᖓᓐᓂ ᐅᓂᒃᑲᓚᐅᖅᑐ ᐅᓪᓗᒥᒧᓪᓗ ᐃᓱᒪᒋᔭᖓᓐᓂ ᐅᓂᒃᑳᖅᑐᑎᑦ.
- ᐅᑭᐊᔅᓵᖑᓚᐅᑎᑐᖅ ᐅᑭᑭᐅᕐᓗ, ᐃᒪᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᐃᖅᑲᓇᐃᔭᖅᑎᖏᑦ ᑕᐃᒫᔅᓴᐃᓐᓇᐸᓗᒃ ᐅᓂᒃᑲᓚᐅᕐᒥᔪᑦ ᐅᓪᓗᒥᒧᓗ ᐅᓂᒃᑳᖅᑐᑎᑦ ᑭᖑᓪᓕᖅ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓄᑦ ᐊᒻᒪᓗ ᕿᑭᕐᒥᐅᑦ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᓐᓄᑦ ᒫᓐᓇᕈᓗᒃ. ᑖᒃᑯᐊ ᐅᓂᒃᑲᓚᐅᖅᑐᑦ ᐃᖃᕐᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑐᔅᓯᕚᖏᓐᓄᑦ ᐱᔾᔪᑎᖃᖅᑐᑎᑦ ᐱᔭᐅᔪᒪᒐᔭᖅᑐᓂᑦ ᐊᑐᓕᖁᔭᖏᓐᓂ ᑲᑎᒪᔩᑦ, ᑕᐃᒫᒃ ᐅᖃᖅᓯᒪᒻᒪᑕ ᓇᓗᓇᐃᕈᒪᒐᔭᓐᖑᐊᖅᑕᖓᓐᓂ ᐸᕐᓇᑦᑎᓄᑦ. ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᓱᓕ ᐱᓕᕆᖃᑎᖃᕆᐊᖃᖅᑐᑦ ᐊᕕᑦᑐᖅᓯᒪᔪᑦ ᐃᓄᐃᑦ ᑲᑐᔾᔨᖃᑎᒌᖏᑦᑕ ᑲᑎᒪᔨᖏᓐᓂ ᐊᑐᓕᖁᔭᖏᓐᓂ ᑐᔅᓯᕙᕐᕕᒋᓗᓂᒋᑦ.
- ᐋᖅᑭᒋᐊᕆᐊᓕᓐᓂᑦ ᓇᓗᓇᐃᔭᖅᓯᒪᔪᓂᑦ ᑖᒃᑯᓄᖓ ᐅᓪᓗᒥᒧᑦ ᐅᓂᒃᑳᕐᕕᐅᓯᒪᔪᑦ ᑲᖏᖅᖡᓂᕐᒥ ᐊᑦᑑᑉᐱᕆᑎᓪᓗᒍ 2017, ᐊᒻᒪᓗ ᑐᓂᔭᐅᓚᖅᑐᑎᑦ ᐆᒪᔪᓕᕆᔨᕐᔪᐊᒃᑯᓐᓄᑦ.

ᑎᑎᕋᖅᑐᖅ: ᕿᑎᖓᓄᑦ ᐅᑭᑭᐅᖅᑕᖅᑐᒧᓪᓗ – ᐃᖃᓗᓕᕆᔨᒃᑯᑦ, ᐃᒪᕐᒥᐅᑕᓕᕆᓂᖅ
ᐅᓪᓗᖓ: ᕕᕝᕗᐊᕆ 2, 2018