

ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᑲᑎᒪᔨᖏᑦ
ᑲᑎᒪᓵᕐᓂᑦ: ᓇᓕᓕᐊᓱᒪᔨᒥᐅ ᑲᑎᒪᓂᖅ 003-2025
ᑐᐃᒪᐃᓗ 5, 2025 (ᐅᓕᓗᖅ 1)
ᔨᐊᓗᓇᐃᖏ, ᓄᓇᕐᓱᐊᖅ



	ᓇᒪᓐᓄ::	ᑲᒪᒋᔭᓂᖅ:	ᓇᓕᕐᓱᑎᓐᓄ:	ᐅᓂᒃᑲᖏᑦ	ᐱᖃᓂᖅ
9:00 - 9:02 AM	1	ᒪᑐᐃᖏᑦ ᑲᑎᒪᓂᖅ		ᐃᕐᓱᐊᐅᑕᖅ	2 ᒥᓇᕐᓂ
	2	ᓇᓗᓇᐃᖏᓱᑎᑦ ᐊᑕᑐᐊᓗᖏᑐᖃᖏᑦᑕᑦ		ᐃᕐᓱᐊᐅᑕᖅ	
9:02 - 9:05 AM	3	ᑲᑎᒪᔨᓴᓂᓂᑦ: ᖃᒥᓕᕐᓂᑦ ᐊᓚᒪᓗ ᐊᖓᔪᖏᑕᑦ RM003-2025	1	ᐃᕐᓱᐊᐅᑕᖅ	1 ᒥᓇᕐᓂ
	4	ᐊᖏᑭᓱᐃᓂᖅ ᐊᓇᐅᒪᔨᔨᑕᐅᒋᐊᓕᑦ ᖃᓄᐃᓕᓗᐊᖏᓂᖅᑑᓂᖅ ᓴᓕᓕᑦᑕᓗᐊᓂ ᑐᓴᑐᐃᑦ (ᐃᓱᒪᓕᕐᓂᑦ)	2	ᐅᒪᔨᓴᓕᕆᔨᖅᑎᑦ	
		ᖃᖅᖏᑐᑦ			15 ᒥᓇᕐᓂ
	5	ᖃᖓᑐᑕᓴᓂᑦ ᖃᐅᔨᔪᖏᓂᒃ ᖃᐅᔨᔪᖏᑦ ᐅᒥᖓᓂᓂᒃ ᐊᐅᓪᓕᖏᓱᓪᓕᐊᕐᓂ ᓇᓪᓂ, ᕿᓚᓂᒥ ᐱᑕᓂᑦ MX-07 (ᐃᓱᒪᓕᐅᕈᑎᑦᓴᖅ)	3	ᓄᓇᕗᑦ ᑲᓇᖓᓂᓂᓂ ᐊᓇᑎᓕᕆᓂᑦ	
	6	ᓄᑦᖏᐃᐅᐊᖏᓂᖅ ᐊᐅᓪᓕᓱᖃᑎᒋᔭᓂᔪᑦ ᑲᒪᖃᑕᓂᒃ ᐱᔨᓗᑦᑕ ᓄ ᑕᑦ ᐱᑕᖃᖏᑐᓂᒃ ᒥᔪᐅᔪᓱᑎᑦ ᖁᓂᓗᖓᖅ ᑐᑦᑐᕐᓂᒃ (ᑐᓴᕐᓴᓴᑦ)	4	ᓄᓇᕗᑦ ᑲᓇᖓᓂᓂᓂ ᐊᓇᑎᓕᕆᓂᑦ	
	7	2023ᒥ ᑎᕐᓴᐅᔨᑦ ᑐᑦᑐᕐᓂᒃ ᐱᑕᖃᖏᑐᓂᒃ ᒥᔪᐅᔪᓱᑎᑦ ᐊᓚᒪᓗ ᑐᑦᑐᐃᑦ ᖃᓄᐃᓕᒥᖏᓂᑦ (ᑐᓴᕐᓴᓴᑦ)	5	ᓄᓇᕗᑦ ᑲᓇᖓᓂᓂᓂ ᐊᓇᑎᓕᕆᓂᑦ	
		ᐅᑦᓗᑭᓱᓕᓴᖏᑦ			
	8	ᑲᓇᒪᓂᑦ ᓄᓇᕗᑦ, ᐱᒋᔪᖃᖓᑦ ᐊᓇᑎᓕᕆᓂᑦ – ᐊᒥᓱᑦ ᑲᒪᒋᔭᑦ (ᑐᓴᕐᓴᓴᑦ)	6	ᓄᓇᕗᑦ ᑲᓇᖓᓂᓂᓂ ᐊᓇᑎᓕᕆᓂᑦ	

	9	ᐃᖃᓗᓕᒻᒥᐅ ᐊᖏᓚᓗ ᓇᓕᔪᓂᒻᒥᐅ ᑲᒪᒋᔭᓕᖅᓴᓕᒃ ᑲᒪᒋᔭᐅᓂᒃ <ᖁᓇᐃᑎᖓᓕᒃ ᐊᖏᕐᕋᖓᒥ 2 ᓇᐃᓴᓇᑎᓕᒃ ᐊᖏᓚᓗ ᐊᖏ ᕿᐃᒥ 3 <ᖁᓇᐃᓕᒃ (ᑐᐃᕐᑎᑎᕐᕋᒃ)	7	ᓄᓇᕗᑦᒥ ᑲᕿᒪᓵᔨᑦ – ᐃᖃᓗᓕᒻᒥᐅᑦ ᐊᖏᓚᓗ ᓇᓕᔪᓕᒻᒥᐅᑦ	
	10	ᐊᔪᑭᐸᑭᑎᒃ ᐅᔨᓂᖏᓪᓗ ᑲᑎᒪᔨᑎᒃ ᐊᒍᒃᓴᑕᒃ ᐊᖏᔪᓇᓱᒃᑕᐅᓂᖅᑎᓐᓂᒃ ᐊᖏᓚᓗ ᐊᖏᔪᓇᓱᓪᓗᓂ ᒪᓕᑦᒃ ᐊᖏᔪᓇᓱᓪᓗᓂ ᖁᑭᖅᑕᓪᓕᐹᒥ ᑐᑭᒃᑐᓂ 2025 ᐅᕕᖏᓪᓗ 2032 (ᐃᓱᒪᓕᐅᕈᑎᕐᕋᖅᑕ)	8	ᖁᑭᖅᑕᓪᓕᐹᒥ ᐅᒪᓴᓕᒻᒥᐅᑦ ᑲᑎᒪᓂᖅᑕ	
		ᖁᑲᖅᑐᑦ			15 ᒥᓇᕐᕋ
	11	ᓄᓇᕗᑦ ᑐᖏᓚᐅᒃ ᑎᒥᖏᓐᓂᑦ: ᓇᓄᐃᑦ ᐊᖏᔪᓇᓱᐅᕕᓂᕐᓂᑦ ᑭᓪᓕᐊᖏᕐᓂᑐᑦ	9	ᓄᓇᕗᑦᒥ ᑐᖏᓚᐅᒃ ᑎᒥᖏᓐᓂᑦ	
	12	ᖃᐅᔨᖅᑲᐃᔪᑎᑦ: ᐊᐃᒥ ᐊᑦᒍᑦ – ᐊᕐᕕᑦᑐᑦᓭᓚᔪᓂ ᑐᑭᒧᐊᖅᑎᑦᓭᔨᕐᔨᐊᖏᖅ, ᐃᓚᖏᐅᑕᓕᓪᓗᑦ ᐅᑭᐅᖏᑦᑕᑐᒥ (ᑐᐃᕐᑎᕐᕋᑦ)	10	ᐃᓚᖏᐅᑕᓕᓪᓗᑦ	
	13	ᓄᓇᕗᑦᒥ ᐃᓚᖏᐅᑕᖏᓕᒪᖏᓂᒃ ᐊᐅᓪᓕᓭᔪᑎᑦ ᓄᑦᖕᔪᓚᐊᖏᑦᑕᑦ (ᑐᐃᕐᑎᕐᕋᑦ)	11	ᐸᐃᓚᖏᐅᑕᓕᓪᓗᑦ ᐊᖏᓚᓗ ᓄᓇᕗᑦ ᑐᖏᓚᐅᒃ ᑎᒥᖏᓐᓂᑦ	
	14	ᐃᓚᖏᐅᑕᖏᓕᒪᖏᓂᒃ ᐊᐅᓪᓕᓭᖏᓐᓇᑦ ᐊᐅᓪᓕᓭᔪᔨᑎᑦ ᓄᑦᖕᔪᓚᐊᖏᑦᑕᑦ (ᑐᐃᕐᑎᕐᑎᕐᑦ)	12	ᐃᓚᖏᐅᑕᓕᓪᓗᑦ	
	15	<ᖁᓇᐃᓕᒃ ᑐᐃᕐᕕᖏᕐᕋᔨᓗᓂ ᐊᖏᓚᓗ ᐃᓱᓚᓕᐅᕈᑎᒃᓴᐊᖏᕐᓱᐃᓗᓂ ᐱᓪᓗᑎᑦ ᐃᓱᓚᓕᐅᕈᓐᓇᖅᑐᑦ ᐃᓚᔪᑦ ᐅᑭᐅᖏᓪᓗ ᑎᑎᕐᑕᐅᓚᐅᖏᓂᑦ ᐅᓚᔪᑦ ᐊᑦᑕᓇᖅᑐᒍᑎᑦᑐᑦᑐᑦ ᐅᕙᓂ ᐅᓚᔪᓄᓕᒃ ᐊᑦᑕᓇᖅᑐᑎᑦᑐᓗᓕᒃ ᓚᓪᓕᑦᑕᕐᕋᖅ (ᐊᑦᑕᓇᖅᑐᑎᑦᑐᓗᓕᐅᕈᑎᒃ) (ᑐᓇᕐᑎᕐᑦ)	13	ᐃᓚᖏᐅᑕᓕᓪᓗᑦ	



ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑕᑦ ᑲᑎᒪᔨᖏᑦ
ᑲᑎᒡᔪᐊᑎᑦᓴᑦ: ᓇᓪᓕᐅᔪᒪᔨᒥᐅᖅ ᑲᑎᒪᓂᖅ 003-2025
ᑯᐃᒡᐱᓇᐱ 6, 2025 (ᐅᓪᓗᖅ 2)
ᔨᐸᓗᓇᐃᖅ, ᓄᓇᕗᑦ



	ᓇᒡᒪᖏᑐ:	ᑲᒪᒋᔭᕐᓴᕐᓂᖅ:	ᓇᓂᔪᑎᖏᑐ:	ᐅᓂᒃᑲᕐᔾᔪᑎᖅ	ᐱᐃᕐᓴᑦ
	16	ᑲᑎᖅᑐᒥᑦ ᐊᔨᖕᖏᒥᑦ ᑦ ᐱᔭᐅᓂᕐᒥᒃ ᖃᓄᐃᓪᓗᐊᕐᓂᖏᑦ ᖃᓕᕐᓴᓕᑦ ᓇᓪᓕᓂ 0 ᐃᓂ 2026 ᐊᒡᖅᒪᓗ 2027 ᐃᒪᕐᐅᑕᕐᓂᐊᕐᓇᕐᒥ (ᐅᖃᐅᔨᔪᒋᐊᕈᑎᑦ)	14	ᐃᒪᕐᐅᑕᓕᕆᔨᑦ	
	17	ᔪᐅᒧᑦ ᓄᑦᑎᕐᑦ ᐱᑕᖃᕐᑐᓂᐊᖏᕐᓂᒃ ᐃᓂ ᐅᐊᓱᓇᖓᓂ ᐊᑦᓭᓴᑎᒃ ᐃᒪᕐᐅᑕᕐᓂᐊᖏᓕᕆᔨᑦ ᑎᒥᖏᑦ (NAFO) ᓇᓕᓂ 0 ᐃᑭᕐᒥ (ᐅᖃᐅᔨᔪᒋᐊᕈᑎᑦ)	15	ᐃᒪᕐᐅᑕᓕᕆᔨᑦ	
		ᖃᖏᕐᑕᑦ			15 ᒥᓇᑦᔪ
	18	ᑐᑦᔪᕐᐅᑎᑦ ᐃᔪᓕᓕᐅᕈᒪᒃᔪ ᐱᑦᓗᑦ ᐃᔪᓕᑭᕐᑦ ᑎᑎᕐᓯᑕᐅᓂᖏᑦ ᐊᒪᓗᓗ ᓄᑦᖢᔭᕆᐊᓯᑕᐅᓂᖏᑦ Horned Grebe ᐊᑦᒍᔪᑦ ᐅᒪᔨᔪᓄᑦ ᐊᑦᑕᓇᕐᑎᒋᑦᑐᓄᑦ ᒪᓕᒃᖁᔨᐊᖅ (ᐊᑦᑕᓇᕐᑐᒋᑦᑐᓕᓕᐅᑎᑦ) (ᐃᔪᓕᓕᐅᕈᑎᑦᓴᖅ)	16	ᐊᐸᑎᓕᓕᕆᑦ ᐊᒪᓗᓗ ᑭᓕᐅᑦ ᐊᑭᔨᔪᓂᖓᓂᒃ ᖃᓗᑕᒥ	
	19	ᑭᐅᓂᑎᓂᓂ ᐊᖏᑭᓪᑕᐅᓂᖏᑦ ᐊᑐᓕᖏᑕᐅᓂᖏᑦ ᐅᓂᒃᑲᑦ ᐊᒪᓗᓗ ᖃᑎᔾᓄᑦ <ᖅᓇᒍᑎᑦ ᑕᐃᒫᒃ ᓇᓗᓇᐃᔨᖅᑕᐅᒪᒻᒪᑕ ᐅᐸᓂ ᐅᒪᔨᔪᓄᑦ ᐊᑦᑕᓇᕐᑐᒋᑦᑐᓄᑦ ᒪᓕᒃᖁᔨᐊᖅ	17	ᐊᐸᑎᓕᓕᕆᑦ ᐊᒪᓗᓗ ᑭᓕᐅᑦ ᐊᑭᔨᔪᓂᖓᓂᒃ ᖃᓗᑕᒥ	
	20	ᑲᑎᒪᓂᖅ ᓄᖃᕐᑐᖅ ᐅᓇ RM003-2025 ᑲᑎᒪᓂᖅ			

ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂ ᑐᑦᑐᐃᓪ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᖃᓄᐃᓪᓗᐊᖅᓂᖏᓐᓄᓐᓂᓪᑦ ᕿᐃᓴᕿᐊᓂᖅᓯᒪᑦ ᐃᓗᓕᓴᓪᑦ

ᖃᒪᑎᕐᓴᖅ **ᓇᓂᑦᓯᑎᐊ #**

ᕿᐃᓴᕿᐊᓂᖅᓯᒪᑦ ᐃ;ᓴᖏᑦ 2

ᐅᒪᓵᓕᓴᖅᓵᐊᑦ ᓇᓪᓕᐊᓯᒪᓵᕆᑎᓐ ᖃᑎᒪᓂᖏᓐᓄᓐᓂᓪᑦ ᐅᓂᐠᑲᑦ

ᓴᓪᓕᓂ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᖃᓄᐃᓪᓗᐊᖅᓂᖏᓐᓄᓐᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕ;ᐊᓕ ᑐᐠᑐᐃᓪᑦ 2A

ᐅᒪᓵᓕᓴᖅᓵᐊᓂᓪᑦ ᑎᑎᖃᑦ ᐅᓄᓇᓕᓂ ᐊᐅᓪᓚᓯᖃᑕᐅᓵᓯᑦ ᖃᒪᖃᑦᑕᓪᑦ

2024-07-18 – ᐅᒪᓵᓕᓴᖅᓵᐊᑦ ᑐᓴᖅᑎᓯᔪᑎᖏᓐᓄᓐᓂᓪᑦ ᐱᓪᓗᓐ: ᐊᐃᖃᑦᑕᐅᓂᖏᓐᓄᓐᓂᓪᑦ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᖃᓄᐃᓪᓗᐊᖅᓂᖏᓐᓄᓐᓂᑕ ᓄᓇᕗᑦᑐ ᓄᓇᑦᑖᔪᓯᒪᔨᓵᓂ ᓇᓪᓂᓂ 2B

2024-07-24 – ᑲᕿᒪᓵᔨᑦ ᓄᓇᕗᑦᑐ ᒥᓂᑦᑕᖓᑕ ᑎᑎᖃᖏᓐᓄᓐᓂᓪᑦ ᐅᒪᓵᓕᓴᖅᓵᐊᓄᑦ ᐱᓪᓗᑎᓐᓂᓪᑦ: ᐊᐃᖃᑦᑕᐅᓂᖏᓐᓄᓐᓂᓪᑦ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᖃᓄᐃᓪᓗᐊᖅᓂᖏᓐᓄᓐᓂᑕ ᓄᓇᕗᑦᑐᓄᓇᑦᑖᔪᓯᒪᔨᓵᓂ ᓇᓪᓂᓂ 2C

2025-04-07 – ᐅᒪᓵᓕᓴᖅᓵᐊᑦ ᑎᑎᖃᖏᓐᓄᓐᓂᓪᑦ ᐅᐳᖏᓐ ᐊᐅᐃᓵᑦ ᐊᖏᔪᓕᐊᑐᑐᓗᓂᓪᑦ ᐱᓪᓗᑎᓐᓂᓪᑦ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᐊᐃᖃᑦᑕᐅᓂᖏᓐᓄᓐᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂ ᑐᐠᑐᐃᓪᑦ 2D

2025-04-07 – ᑐᖏᓚᐃᓵᓂ ᑎᑎᖃᖏᓐᓄᓐᓂᓪᑦ ᐅᒪᓵᓕᓴᖅᓵᐊᓄᑦ ᐱᓪᓗᑎᓐᓂᓪᑦ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂ ᑐᐠᑐᐃᓪᑦ 2E

2025-04-28 – ᐅᒪᓵᓕᓴᖅᓵᐊᑦ ᑭᐅᓴᕐᑎᓐᓂᓪᑦ ᑐᖏᓚᐃᓵᓂᓗᓂᓪᑦ ᐱᓪᓗᑎᓐᓂᓪᑦ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂ ᑐᐠᑐᐃᓪᑦ 2F

2025-05-07 – ᐅᑎᒪᓴᕐᑎᒋᓂᓪᑦ: ᐊᐃᐅᐃᓵᑦ ᐊᖏᔪᓕᐊᑐᑐᓗᓂᓪᑦ ᐅᑎᒪᓂᖏᓐᓄᓐᓂᓗᓂᓪᑦ ᐱᓪᓗᑎᓐᓂᓪᑦ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂ ᑐᐠᑐᐃᓪᑦ 2G

2025-07-23 – ᑐᖏᓚᐃᓵᓂ ᑎᑎᖃᖏᓐᓄᓐᓂᓪᑦ ᐅᒪᓵᓕᓴᖅᓵᐊᓄᑦ ᐱᓪᓗᑎᓐ ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓗᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂᓐᓄᓐ ᑐᐠᑐᐃᓪᑦ 2H

2025-08-21 – ᐅᒪᓵᓕᓴᖅᓵᐊᑦ ᑎᑎᖃᖏᓐᓄᓐᓂᓪᑦ ᑕᖏᓚᐃᓵᓂᓗᓂᓪᑦ ᐱᓪᓗᑎᓐᓂᓪᑦ: ᐊᓐᓇᐅᒪᓵᕆᑕᐅᓕᐊᓂᓪᑦ ᓴᓪᓂᓪᑦ ᖃᑭᖅᑕᓗᐊᓂ ᑐᐠᑐᐃᓪᑦ 2I

ᐱᖃᓐᓴᐅᓵᖅᓯᒪᑦ ᐅᒪᓵᓕᓴᖅᓵᐊᑦ ᐊᓚᓗᓗ ᓴᓪᓕᐊᓯᒪᔨᓵᓂ ᐃᓱᒪᓕᐅᕐᓗᑎᓐᓂᓗᓂᓪᑦ

2012 ᐊᖏᔪᓱᑎᓐᓴᖅᓯᒪᑦ 1000 ᐱᓱᓴᓴᓴᓪᑦ: ᖃᓄᓕᖏᓐᓄᓕᐅᓕᒪᓂᓪᑦ ᐱᐅᓰᓴᓱᒪᔨᓵᑦ ᕿᓂᓂᑦ ᐊᓚᓗᓗ ᐱᑕᖏᖅᓄᓕᐅᓕᐊᓂ ᐊᓴᓐᓂᐅᑎᐃᑦ ᐊᐃᓴᕿᓰᕿᕕᑕ – ᓄᓇᕗᑦᑐ ᐊᖏᖃᑎᑎᒋᔭᑎᑦ 5.3.24 2J

2013 ᐊᖏᔪᓕᒐᕆᕆᓂ 1000 ᐱᓱᕆᕆᓂ ᑭᓪᓗᖅᖃᓪᓗᓂᑦ -
2012 ᐊᖏᔪᓕᒐᕆᕆᓂ 1000 ᐱᓱᕆᕆᓂ: ᖃᓪᓗᖅᖃᓪᓗᓂᑦ ᐱᑕᕆᔨᓕᔪᑦ ᐸᓚᓂ ᐊᒪᓕᓗ ᐱᑕᖅᖃᓪᓗᓂᓗᓂ ᐊᕐᓇᐃ–ᓄᑦᕎᐃᑦ ᐊᐃᐠᐸᕆᐸᑕ – ᓄᓇᕗᒥ ᐊᖏᖅᑎᑎᒋᔭᑦ 5.3.24_____2K

2014 ᐊᖏᔪᓕᒐᕆᕆᓂ 800 ᐱᓱᕆᕆᓂ ᑭᓪᓗᖅᖃᓪᓗᓂᑦ - ᐱᑕᖅᖃᓪᓗᓂᓗᓂ ᐊᕐᓇᐃᑦ–ᓄᑦᕎᐃᑦ ᐊᐃᐠᐸᕆᐸᑕ – ᓄᓇᕗᒥ ᐊᖏᖅᑎᑎᒋᔭᑦ 5.3.24_____2L

2015 TAH 1600 ᐱᓱᕆᕆᓂ ᑭᓪᓗᖅᖃᓪᓗᓂᑦ - ᐱᑕᖅᖃᓪᓗᓂᓗᓂ ᐊᕐᓇᐃᑦ–ᓄᑦᕎᐃᑦ ᐊᐃᐠᐸᕆᐸᑕ – ᓄᓇᕗᒥ ᐊᖏᖅᑎᑎᒋᔭᑦ 5.3.24 ᐊᒪᓕᓗ 5.6.16_____2M

2019 TAH 1000 ᐱᓱᕆᕆᓂ ᑭᓪᓗᖅᖃᓪᓗᓂᑦ - ᐱᑕᖅᖃᓪᓗᓂᓗᓂ ᐊᕐᓇᐃᑦ–ᓄᑦᕎᐃᑦ ᐊᐃᐠᐸᕆᐸᑕ – ᓄᓇᕗᒥ ᐊᖏᖅᑎᑎᒋᔭᑦ 5.6.16_____2N

2024 ᐊᑯᓂᐅᔨᒥᐅ ᐊᓯᔾᔨᑕᕐᑐᑦ ᖃᑦᔪᐅᓂᖏᓐᓂᐅ ᐊᒪᓕᓗ ᐅᐱᕐᓱᓴᐅᑯᑦ ᓇᒃᓂᖏᓐᓂᐅ ᓴᓪᓕᕐᒥᓗᐊᓂ ᖃᑭᕐᒥᓗᐊᓂ ᑐᑐᐃᓐᓇᖅ: _____2O

ⵛⵣⵖⵐⵔⵗ

ⵒⵙⵛⵐ ⵑⵍⵔⵔⵚⵖⵔⵙ ⵃⵎⵍⵖⵙⵔⵓⵒ

ⵄⵍⵄⵛⵑⵒⵒⵍⵊ

ⵄⵔⵍⵍⵔⵐⵒⵛ: **ⵄⵔⵍⵍⵔⵐⵒⵛⵎⵃⵖⵙ: X**

ⵄⵖⵔⵎⵔ: ⵒⵙⵛⵐ ⵑⵍⵔⵔⵚⵖⵔⵙ ⵃⵎⵍⵖⵙⵔⵔ ⵄⵔⵍⵍⵔⵐⵒⵛⵎⵃⵖⵙⵔ ⵙⵖⵔⵙⵔⵖⵑⵒⵒⵔ ⵖⵔⵙⵔⵔ ⵄⵄⵛ ⵄⵄⵛⵔⵐⵒⵛⵎⵃⵖⵙⵔ ⵄⵄⵛⵔⵐⵒⵛⵎⵃⵖⵙⵔ ⵄⵄⵛⵔⵐⵒⵛⵎⵃⵖⵙⵔ

ⵄⵖⵔⵎ

ⵔⵄⵍⵔⵔ ⵚⵔⵔⵐⵖⵔⵍⵎⵒⵒⵔⵛ ⵒⵒⵍⵊⵐⵔ ⵑⵍⵔⵔⵚⵖⵔⵙ ⵃⵎⵍⵖⵙⵔⵔ ⵑⵍⵔⵔⵚⵖⵔⵙ (“NWMB” ⵐⵙⵒⵒⵔ “ⵃⵎⵍⵖⵙⵔ”) ⵙⵖⵍⵔⵔ ⵍⵎⵖⵔ, ⵒⵙⵛⵐ ⵑⵍⵔⵔⵚⵖⵔⵙⵔⵙⵚⵖⵔⵙ ⵃⵎⵍⵖⵙⵔⵔ ⵙⵖⵔⵙⵔⵖⵑⵒⵒⵔ ⵄⵄⵛⵔⵐⵒⵛⵎⵃⵖⵙ

1993 ᐊᓗᒪᓗ 2009 ᐊᓗᒪᓗ ᓄᓇᕗᒻᒥᒃᓴᓕᓕᐅᔪᓂᖅᓱᓂᒃ ᓄᓇ ᖃᓄᖅᑐᐊᒍᓪᓚᓗᓂ ᓄᓇᐅᑦ ᖁᖏᖅ ᐅᑭᐅᖅᓱᓂ 2010/2011 ᑕᐃᒪᓐᓇ ᑐᓐᑐᑦ ᓂᕆᐹᓕᓕᐅᔪᖓᒪᑕ.

ᐊᔪᖁᔭᐊᓂ 2012 ᐊᓗᒪᓗ 2013, ᐊᑐᓕᖃᓴᓕᐅᓕᐅᖅᑎᓪᓗᑎᑦ ᐊᐃᐃᕐᖅ ᐊᖏᔪᓄᑐᑕᓕᓴᓕᖃᑦ ᑎᑎᕋᓂᖓᑦ, ᓄᓇᕗᒻᒥ ᐱᓕᕆᐃᖓᓱᑦ ᐊᐢᕿᑎᓕᕆᓴᓕᖃᑦ, ᐊᑐᖅᑕᐅᓪᓗᓂ ᒥᓂᔅᑕᐅᑦ ᑎᓕᓕᔭᐅᓂᖅ ᐃᓕᑐᖅᓕᖅᑎᑕᐅᓪᓗᓂᑦ ᑐᑦᑐᖃᖅᑎᑦ 1,000-ᓂᒃ ᐊᓗᒪ ᐊᖏᔪᓕᕐᓂᒃᓄᑦ ᒪᓕᑦᖃᖅᑎᑕᓄᑦᓄᐅᓕᐅᖅᑭᔾᔪᑦ ᐊᓯᐊᓪᓚᕐᓂᒃ ᓄᓇᕆᓕᕐᓂᒃ ᑐᑦᑐᖃᑕᓯᓄᐅᓕᖅᑐᓂ. ᐊᖏᔪᐅᔾᔨᓴᑦ ᑭᓚᓐᓇ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᑐᒃ 800-ᖏᓴᑎᑕᐅᓪᓚᓗᑎᒃ ᒥᓂᔅᑕᐅᑦ ᑎᓕᓕᔭᐅᓂᖅᓱᓚᐅ ᐊᑐᖅᑕᐅᓪᓚᓗᓂ ᐊᓯᖁᔭᐊᓂ 2014-ᖏᑎᓪᓚᔪᑐ.

ᖃᖓᑕᓕᕆᓂᖅᓄᑦ ᐃᓴᐃᑎᓪᓚᓂᑦ ᒪᐃ 2015–ᖏᑎᓪᓚᔪᑐ ᐊᓚᓇᑲᓂᓕᕆᓕᐅᖅᑐᑐᒃ ᐅᓄᓂᓂᖅᑐᑕᑯᐅᓕᕆᓕᐅᖅᑐᑦ 12,297-ᓂᒃ ᑐᓐᑐᑦ, ᐊᓚᓇᐃᖅᓴᓚᓂᑎᒃ ᑐᓐᑐᑦ ᐅᓄᖅᓴᓂᓪᓕᐊᓂᕐᓂᓂᒃ ᑭᖏᓂᕎᓂᒃᓂ ᐅᓄᓂᖅᓂᓕᕆᓕᖅᑕᑦ 7,287–ᓂᒃ ᐃᓴᖅᑕᐅᑎᓪᓚᓂᑦ ᐊᑎᖁᔭᐊᓂ 2013, ᑕᐅᓂᐊ ᐱᔨᑎᖃᓕᐅᖅᑐᑦ, ᐃᓕᓴᒍᔪᑦ, ᑐᓐᑐᑦ ᓄᐃᑎᓕᐅᓄᓂᕐᓂᓂᒃ ᖁᑭᖅᑕᒥᑦ ᓄᓇᓕᐅᕐᔪᒍᑦ. ᑭᖏᔪᓂᖅᓴᓚᔪᑦ 2015 ᐅᓄᓂᓂᑦ ᐃᓴᖅᑕᐅᑎᓪᓚᓂᑦ ᐊᓚᓇᐃᖅᓴᓕᐅᖅᑐᑦ ᐅᓄᖅᓴᓂᐊᓂᓂᒃᓂᓂᒃ ᓴᓪᓕᑦ ᑐᓐᑐᖏᑦ, ᓄᓇᕗᑦ ᐅᒪᓕᔪᓚᓂᔪᓴᐊᑦ ᑲᑎᒪᓂᖓᑦ ᐊᖏᔪ ᒥᓂᔅᑕᐅᑦ ᐊᖏᔪᓄᑐᑕᓕᓴᓕᖃᑦ ᐅᓄᖅᓴᓂᐊᓂᓂᒃᓂᓂᒃ ᑐᓐᑐᖏᑦ, ᓄᓇᕗᑦ ᐅᒪᓕᔪᓚᓂᔪᓴᐊᑦ ᑲᑎᒪᓂᖓᑦ 1,600-ᖏᖅᑎᑕᐅᓪᓚᓗᑎᒃ.

ᐱᓕᓚᐊᖅᓕᓕᐅᖅᑐᑦ ᓴᓵᖏᓐᓂᓂᕆᐊᐅᖅᑎᑦᓚᓂᑦ ᑐᓐᑐᑦ ᐅᓄᓂᓂᖓᑦ, ᐃᓴᖅᑭᐅᐊᖅᑕᐅᓪᓚᓗᑎᒃ ᑯᑕᐅᑦ 2019–ᖏᑎᓪᓚᔪᑐ 1,600-ᖏᓪᓕᐅᖅᑐᑐᓗᓪᓕᐊᑦ ᑐᓐᑐᖏᑦᓂᒃ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᓂᒃ 1,000-ᓄᑦ, ᐃᓚᐃᓕᓕᐅᖅᑎᓪᓚᓂᑦ ᒪᐃ 2017–ᖏᑎᓪᓚᔪᑐ ᓄᓇᕗᒻᒥᒃᓴᓕᓕᐅᓕᖅᑕᒪᑦ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᓂᒃ. ᐅᓪᓚᒥ ᑯᑕᓕᐅᖅᑕᐅᓪᓚᓗᑎᒃ ᐊᑐᓂᓂᐊᖅᑕᒥᓂᓂᒃ ᓴᓪᓕᑦᑎᐅᓄ ᐅᓄᓂᓂᖅᖁᖅᑐᑦ ᑐᓐᑐᖏᑦᓂᒃ 1,000-ᓂᒃ.

ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐃᓗᓕᓂᔨᖅᖁᖅᑐᑦ ᓴᓪᓕᑦᓂ ᑐᓐᑐᑦ ᑯᑕᖅᑎᑕᐅᓂᕐᓂᑦᓂᓄᑦ ᐃᓕᓕᐅᑎᓯᒪᔪᑦ ᐅᑯᓄᖓ ᑐᐱᓂᑦᓯᓯᐊᕐᑎᓄᑦ ᑎᑎᖅᑲᓂ ᓂᕆᐹᓕᓕᐅᖅᑐᑦ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᑦ.

ᑕᐃᒪᓕ ᐃᓄᐃᔾᔨᓴᒪᐊᖅᖁᖅᑐᑦ ᐃᓄᖕᓄᑦ ᐅᓄᓂᖅᖁᖅᑎᑕᐅᓂᖅᑐᓄᓕᑦ ᓴᓪᓕᑦᑎᐅᓄᓕᑦ ᐃᓕᓕᐅᑎᓯᒪᔪᑦ ᐅᑯᓄᖓ ᑐᐱᓂᑦᓯᓯᐊᕐᑎᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑕᒥᑦ, ᐱᖅᑯᓯᖁᖅᑕᑦᓯᓯᐊᕐᑎᓄᑦ ᑯᑕᖅᑕᐅᓂᐊᖅᑕᒥᑦ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᑦ ᓴᓪᓕᑦᑎᐅᓄᓕᑦ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᑦ ᓴᓪᓕᑦᑎᐅᓄᓕᑦ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᑦ ᑯᑕᖅᑕᐅᓂᐊᖅᑕᒥᑦ ᐃᓄᐃᓴᓕᓕᐊᑎᒃᓴᓕᐅᓕᐅᖅᑐᑦ ᓴᓪᓕᑦᑎᐅᓄᓕ

ᓄᓇᒃᓴᓪ ᐅᒪᔨᓕᕆᔨᖅᑎᒋᓴᖅ ᑲᑎᒪᔨᖏᓪᓗ ᐱᓕᕆᖏᑐᐃᓐᓇᖅᑐᓴᖅ ᐊᐅᓪᓚᑦᓯᐊᕐᓇᖅᑎᑦᑎᓯᖏᓐᓄᑦ ᐊᐅᓚᑦᓯᐊᕐᓇᖅᑎᑦᑎᓯᖏᓐᓄᑦ

ᓄᓇᒃᓴᓪ ᐅᒪᔨᓕᕆᔨᖅᑎᒋᓴᖅ ᑲᑎᒪᔨᖏᓪᓗ ᐊᑦᑭᒍᐊᓂ 2024 ᐃᓄᖅᓄᓪ ᐃᓅᓴᔨᓕᐅᕐᓴᖅᓯᓕᕆᓂᖅᓄᓪ ᑭᖑᓪᓕᐅᕙᓚᑦᓴᒍᓪ ᐊᑐᐊᖏᖅ ᑎᑎᖅᑲᖅ

ᔨᓕᐃ 18, 2024–ᖃᐅᑎᓪᓗᒍ, ᓄᓇᒃᓴᓪ ᐅᒪᔨᓕᕆᔨᖅᑎᒋᓴᖅᓯᓪ ᑐᓴᖅᑎᑦᑎᓕᐅᕐᓴᖅᓯᓕᕆᓂᖅᓄᓪ ᐊᐅᓚᑦᓯᑎᐅᕐᓴᖅᑲᑎᒪᔨᖏᓪᓗ ᑕᒪᑐᐃᓐᓇᖅᑐᓴᖅ ᐃᓕᓴᕐᓂᖅᓄᓪ ᖃᐅᔨᓴᕐᓂᖅᓄᓪ ᖃᑦᑭᓴᐅᔭᖅᑎᖅᑕᕐᓂᖅᓄᓪ ᐃᓅᓴᖅ ᐃᓅᓴᔨᓕᐅᕐᓴᖅᓯᓕᕆᓂᖅᓄᓪ ᐊᒻᒪ ᑕᒪᑐᐃᓐᓇᖅᑐᓴᖅᓯᓕᕆᓂᖅᓄᓪ ᐊᐅᓚᑦᓯᐊᕐᓇᖅᑎᒋᓴᖅᓯᓪ ᐊᐅᑐᓂᖅᓴᓕᕆᓂᖅᓄᓪ ᐊᐅᓚᑦᓯᐃᓇᐅᔭᓴᖅᓯᓪ ᖃᖅᑎᑕᐅᔭᖅᑐᓴᖅᓯᓪ ᑐᐊᖓᔪᖅ 2014–ᖃᐅᑎᓪᓗᒍ ᓄᓇᒃᓴᓪ ᐊᒻᒪᓗᓯᓪ ᖃᖅᑎᑕᐅᔭᖅᑐᓴᖅᓯᓕᕆᓂᖅᓄᓪ.

ᑕᐃᒪᓕ ᑭᖓᔪᖏᓚᑦ ᐅᓪᓴᖅᑐᑦᓗᓂᖅᓄᓪ ᐊᓪᓕᓇᐃᐅᔨᓪ ᐃᓗᐊᓄ ᖃᑎᖏᑭᓇᓕᐅᕐᓴᒪᑕ ᐃᓕᓴᕆᔪᖅᖃᓕᑕᐅᔨᓪᓗᒍᓄᓪ ᐃᓗᓴᓇᖏᑦᑐᖅ ᐃᓕᓴᕐᓂᖅᓄᓪ ᐃᓕᓴᕆᓕᕐᓂᖅ, ᐱᖓᓱᐅᑎᓪᓗᒍ ᑭᖓᔪᖏᓚᑦ ᖃᐅᔨᓴᕐᓂᖅᓄᓪ ᐃᓕᓴᕆᓕᕐᓂᖅᓄᓪ ᐊᒻᒪᓗ ᐃᓕᓴᕆᓕᕐᓂᖅᓄᓪ ᑕᐃᓴᖅᓄᖓᓪ ᐊᓐᑐᖅᑕᐅᔨᓕᕆᓂᖅᓄᓪ ᖃᐅᔨᓴᕐᓂᖅᓄᓪ ᐊᐅᓚᑦᓯᐊᕐᓇᖅᑎᒋᓴᖅᓯᓪ ᑕᐃᓴᖅᓄᖓᓪ ᐊᐅᓚᑦᓯᐊᕐᓇᖅᑎᒋᓴᖅᓯᓪ ᐊᐅᓚᑦᓯᐃᓇᐅᔭᓴᖅᓯ

ᓄᓇᒃᓴᑦ ᓱᓕᑦᑐᓄᑦ ᐊᖏᔪᐅᓂᖅ. ᓄᓇᒃᓴᑦ ᑐᓴᖏᓚᖅ ᐊᐃᕐᐳᑎᖅᑕᓕᐅᖅᑐᑦ ᓄᓇᒃᓴᑦ ᐅᒪᔨᓕᓗᓂᔨᕐᓂᐊᕐᓂ ᑲᑎᒪᔨᖏᑦ ᐊᑐᖏᓐᓂᖏᓐᓂᖅ ᓱᓇᓕᕐᒥ ᑐᐱᓕᕐᑐᑎᑕᐅᓕᐅᖅᑐᑦᑐᒃᑯᑦ, ᐃᑲᔪᖅᑐᖅᑕᐅᓕᐅᖅᓭᓪᓗᔨᒥᑦ 2009-ᖏᑎᓕᒍᒍ ᐃᓱᒪᓕᕐᐅᔨᒧᑦ (“Meehan ᐃᓱᒪᓕᕐᖏᓐ”), ᐊᑕᖏᓕᒍᓂᖅ ᐃᓄᐃᑦ ᐊᖏᔪᖏᑦ – ᐱᖃᓱᐅᑎᓕᒍᑦ ᓂᐅᕐᑭᑎᕿᐅᓕᒍᓂᖅ ᐊᖏᔪᐅᒧᑦ ᓲᓕᐅᑕᐅᔨᐃᓐᓇᐅᕐᑯᓕᒍᑦ ᐱᖃᓱᐅᕐᖏᐅᓂᖅ ᐃᓄᐃᑦ ᐃᓅᓴᔨᕐᑎᓴᓂᐊᖅᑕᑕᖏᑕ ᑭᓚᖏᑎᓄᓕ ᐊᒪᒪ ᐊᐱᖅᑯᑎᖃᓕᐅᖅᑐᓪᓗ ᐱᓕᒍᓗ ᓴᓇᓕᓐᓂᖅᑐᑦ ᓄᓇᖃᖅᑐᑦ ᐊᖏᔪᖏᑦ ᓂᐅᕐᑭᑎᕿᐅᓕᒍᓂᖅ ᐱᖃᓱᐅᕐᖏᐅᓂᖅ ᐃᓄᐃᑦ ᐃᓅᓴᔨᕐᑎᓴᓂᐊᖅᑕᑕᖏᑕ ᑭᓚᖏᑎᓄᓕ ᐊᒪᒪ ᐊᐱᖅᑯᑎᖃᓕᐅᖅᑐᓪᓗ ᑲᑎᒪᔨᖏᑦ ᐊᑐᖏᓐᓂᖏᓐᓂᖅ ᐊᖏᔪᖏᑦ ᐊᑐᖏᓐᓂᖏᓐᓂᖅ ᐊᖏᔪᐅᒧ

ᐊᖏᑭᓱᐃᓂᖅ ᐅᓄᖅᓯᓂᐊᖏᑕᖅᑐᓂᓐᓂ ᐃᓄᐃᖅᓴᑎᐅᔪᓂᑎᑐᓂ ᑐᐃᑐᑎᓴᓴᑦ ᓴᓪᓕ᙮ ᖃᑭᖏᓐᓇᓂ ᑐᐃᑐᑎᓐᓄᑦ

ᓄᓇᕗᒻ ᐊᖏᕈᑎᑦᑕᐃᓕᓴᐅᓂᖅ

ᑎᑎᕋᖅᓯᒪᓂᖅᓱ 5.6.23 ᓄᓇᕗᑦ ᐊᖏᕈᑎᑎᓂ ᒪᓕᒪᖅᖢᒪᑕ ᐃᒪᓐᓇ, ᐊᑲᑭᓯᓐᓂᖏᓐᓇᕐᒥᑕ ᑯᑕᓂ ᓂᕆᐊᑎᓄᑦ ᐊᖏᔪᐅᕐᓴᖅᓴᓂ ᑭᖑᓂᕈᑎᓂ ᑯᑕᖅᓴᕐᓚᖏᑦᑐᑐᓄᑦ (ᓯᑦᓴ ᓴᓪᓕᒥᓂᖅ ᑐᐃᑐᑎᓐᓄᑦ)[1] ᓄᓇᕗᑦ ᐅᒪᔪᓕᓀᓴᕆᔨᐊᑦ ᑲᑎᒪᔨᖏᑦ ᐊᑲᑭᓱᓯᕆᐊᖅᑐᖅᓵᑦ ᐃᓄᐃᖅᓴᑎᐅᔪᓂᑎᑐᓂ ᐅᓄᖅᓯᓂᐊᖏᑕᖅᓯᒪᓂᖅᓱ ᐊᑐᑐᓂᑐᓄᑦ ᐅᓄᖅᓯᓂᐊᖏᑕᖅᓯᒪᓂᖅ ᐊᑐᑐᓂᑐᓄᑦ **ᐅᓄᖅᓯᓂᖅᓴᐅᔪᓂᑦ (ᖃᓄᐃᓕᓴᐅᓂᖅᓴᐅᔪᓂᑦ)**:

a) ᐅᓄᖅᓯᓀᓐᓂᑦ ᑐᓴᖓᐃᖅᑐᑦ ᑐᕆᖓᓴᓂᑦ ᖃᓄᐃᑦᑐᖅᓱᓂᑦ ᓄᓇᕗᑦ ᐅᒪᔪᓕᓀᓴᕆᔨᐊᑦ ᓴᓇᐅᓕᓀᓴᕆᔨᐊᑦ ᑲᑎᒪᔨᖏᑦ ᑕᒪᓐᓇᖅᑐᖅ ᐃᓄᐃᖅᓴᑎᐅᔪᓂᑎᑐᓂ ᓄᓇᕗᑦ ᐅᒪᔪᓕᓀᓴᕆᔨᐊᑦ ᑲᑎᒪᔨᖏᑦ, ᐊᑲᑭᓱᓯᕆᐊᖅᑐᖅᓵᑦ ᐅᓄᖅᓯᓂᐊᖏᑕᖅᓯᒪᓂᖅᓱ ᒪᓕᒃᑐᐅᔪᓂᑦ ᐱᐅᓯᐅᔨᓀᐊᖓᖅᑐᖅ ᑎᑎᕋᖅᓯᒪᓂᖅᓱ ᑎᑎᕋᖅᓯᒪᓂᖅᓱᒪᓂᑦ ᐃᓚᖓᓂ 5.6.21 (a) ᓄᓇᕗᑦ ᐊᖏᕈᑎᑎᓂ ᐅᐸᓂᓴᓐᓂ ᐊᑐᐃᓇᓯᑐᓂᓕᕆᔨᓂᑦ ᑎᑎᕋᖅᓯᒪᓂᖅᓱᒪᓂᑦ ᑭᖑᓂᕈᑎᓂ ᓂᕆᐊᑎᓄᑦ ᐊᑐᓯᐅᐸᓂᓴᓐᓂᑦ ᐃᓚᖓᓂ ᐊᑐᓯᐅᐸᓂᓴᓐᓂᑦ ᐊᑐᐃᓇᓯᑐᓂᓕᕆᔨᓂᑦ ᐊᑲᑭᓱᓯᕆᐊᖅᑐᖅᓵᑦ ᐃᓄᐃᖅᓴᑎᐅᔪᓂᑎᑐᓂ ᐊᑐᐃᓇᓯᑐᓂᓕᕆᔨᓂᑦ ᐊᑲᑭᓱᓯᕆᐊᖅᑐᖅᓵᑦ ᐅᓄᖅᓯᓂᐊᖏᑕᖅᓯᒪᓂᖅᓱᒪᓂᑦ ᐃᓚᖓᓂ 5.6.21(b)–ᒥ, ᐊᖓᔪᐅᔨᔪᓂᑦ ᐅᓄᖅᓯᓀᓐᓂᑦᑐᓂᑦ ᐃᓕᓴᓂᑦ ᓇᓗᓇᐃᖅᑕᐅᔪᓴᓂᑦ ᐊᑲᑭᔪᒥ; ᐅᕐᓂᓇᓐᓂᑦ

b) ᐊᑕᖓᑦᓗᒍ ᑲᑎᓐᓄᓗᓂᑦ ᐅᓄᖅᓯᖅᓯᒪᔪᓂᑦ ᐊᖓᔪᐅᔨᔪᐃᓂᑦ ᐊᑲᑭᔪᒥ ᐊᑕᐅᔪᓴᒥ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᓂ ᑕᓪᓕᐊᓂᑦ ᐊᑐᓂᓴᖅᑐᑦ ᐊᑲᑭᔪᒥᑦ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᑕᒫᓐᓇᖅᑐᑦ ᐊᑐᓂᓴᖅᑐᑦ ᐊᑲᑭᔪᓂ ᑕᓪᓕᒪᓐᓇ ᐊᑐᓂᓴᖅᑐᑦ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᑕᒫᓐᓇᖅᑐᑦ ᐊᑐᐃᓇᓯᑐᓂᓕᕆᔨᓂᑦ ᐊᑲᑭᔪᒥ; ᐅᕐᓂᓇᓐᓂᑦ

b) ᐊᑕᖓᑦᓗᒍ ᑲᑎᓐᓄᓗᓂᑦ ᐅᓄᖅᓯᖅᓯᒪᔪᓂᑦ ᐊᖓᔪᐅᔨᔪᐃᓂᑦ ᐊᑲᑭᔪᒥ ᐊᑕᐅᔪᓴᒥ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᒥ ᑕᓪᓕᒪᓐᓇ ᐊᑐᓯᐅᐸᓂᓴᓐᓂᑦ ᐊᑕᐅᓕᒫᓂᖅᑐᑦ ᐊᑲᑭᔪᒥ ᐊᑕᐅᔪᓴᒥ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᑕᒫᓐᓇᖅᑐᑦ ᐊᑐᓂᓴᖅᑐᑦ ᐊᑲᑭᔪᒥ ᑕᓪᓕᒪᓐᓇ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᑕᒫᓐᓇᖅᑐᑦ ᐊᑐᐃᓇᓯᑐᓂᓕᕆᔨᓂᑦ ᐊᑲᑭᔪᒥ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᑕᒫᓐᓇᖅᑐᑦ ᐊᑐᓂᓴᖅᑐᑦ ᐊᑲᑭᔪᒥ ᑕᓪᓕᒪᓐᓇ ᐃᓗᓗᐊᓂ ᐊᑲᑭᔪᑕᒫᓐᓇᖅᑐᑦ ᐊᑐᐃᓇᓯᑐᓂᓕᕆᔨᓂᑦ ᐊᑲᑭᔪᒥ ᑕᓪᓕᒪᓐᓇ

ᓴᓪᓕᖅ ᐃᓄᐃᑦ ᐃᓄᐊᓂᓴᓪᓕᑎᑦ ᐊᖏᔪᐅᓴᖅᓯᓐᓇᖅᑐᑦ ᐅᓄᖅᓂᖅᑎᑦ ᓴᓪᓕᓪᓕ ᑭᑭᖅᑕᖏᓐᓂᒃ ᑐᖅᑐᖅᑎᓄᑦ

ᐊᑕᐃᔪᑦ ᑎᑎᕆᔨᓂᖏᓐᓇᑕ 5.6.21(a): ᑕᐅᔨᐊ ᑲᑎᓐᖏᓐᓇᓵᑦ ᐅᖅᓄᓂᖅᓴᖏᓵᑦ ᐊᖏᔪᐅᓴᐃᓇᓂᑦ ᓇᓕᐊᓐᓂᑐᐃᓐᓇᖅ ᐊᖅᑭᔪᒥ ᑕᐃᑦᔨᒪᓂ ᓄᓇᓕᑦ ᐅᒪᔨᓕᓚᓱᖅᔪᐊᑲᔨᑦ ᐊᖏᔪᓂᒃ ᖃᐅᔨᓴᐃᓂᖃᖅᑎᑦᓴᓕᑦ ᐊᓘᒪᓗ ᖃᑎᐊ ᐊᖅᑭᔪᑕᒪᓐ ᐊᖏᔪᐅᓴᑦ ᐅᓄᖅᓂᖅᑎᑦ ᐃᓗᐊᓂ ᐊᖅᑭᔪᑕ ᑕᓪᓕᒪᓐ ᖃᐅᔨᓴᐃᓂᖃᖅᑎᑦᓴᓕᑦ, ᑕᐅᓐᓇ ᑲᓐᑎᓪᓚᖅ ᐊᐃᓐᑕᐅᓗᓂᓂ ᖃᑎᐊᐃᒪᑦ:

$$\mathrm{BNL} = \frac{\mathrm{Max(Harvest)} + \mathrm{Avg(Harvest)}}{2}$$

ᐃᒪᐃᓕᓚᓱᖅᑐᑦ:

- ᐅᓄᖅᓂᖅᓯᑦ (ᐊᖏᔪᐅᓴᑦ) = ᐅᓄᖅᓂᖅᓯᖅ ᑲᑎᓕᑦ ᐊᖏᔪᐅᓂᓂᑦ ᓇᓕᓐᓂᑕᐃᓐᓇᖅ ᐊᖅᑭᔪᒥ ᐊᑕᐅᓴᒥ ᑕᐃᑦᔨᒪᓂ ᓄᓇᓕᑦ ᐅᒪᔨᓕᓚᓱᖅᔪᐊᑦᓴᓕᑦ ᑲᑎᒪᔨᖏᑦ ᐊᖏᔪᓂᒃ ᖃᐅᔨᓴᐃᓂᖃᖅᑎᑦᓴᓕᑦ.
- ᐊᓂᓂᓐᖓ (ᐊᖏᔪᐅᓴᑦ) = ᐊᓂᓂᓐᖓ ᐊᖅᑭᔪᑕᒪᓐ ᐊᖏᔪᐅᓴᑦ ᐊᖅᑭᔪᒥ ᑕᓪᓕᒪᓐ ᐃᓗᐊᓂ ᓄᓇᓕᑦ ᐅᒪᔨᓕᓚᓱᖅᔪᐊᑦᓴᓕᑦ ᑲᑎᒪᔨᖏᑦ ᐊᖏᔪᓂᒃ ᖃᐅᔨᓴᐃᓂᖃᖅᑎᑦᓴᓕᑦ.

ᐅᓄᖅᓂᖅᓯᖏᓐᓇᖅ ᑐᖅᑐᑕᐅᓂᓂᑦ ᐊᖅᑭᔪᒥ ᑎᑎᕆᖅᑕᐅᔨᒪᔨᑦ ᓄᓇᓕᑦ ᐅᒪᔨᓕᓚᓱᖅᔪᐊᑦ ᐊᖏᔪᓂᒃ ᖃᐅᔨᓴᐃᓂᖃᖅᑎᑦᓴᓕᑦ ᑲᓪᓕᓪᓕ ᐅᓄᖅᓂᖅᖢᐅᓂᖅᑐᑦ 1940-ᓂᒃ (ᓇᓗᓇᐃᖅᓱᓪᓚᑦ 1, ᐊᖅᑭᔪᐊᓂ 1999).

ᐊᓂᓂᓐᐊ ᐅᓄᖅᓂᖏᓐᓇ ᑐᖅᑐᑕᐅᓂᑦ ᐊᖅᑭᔪᒥ ᑕᓪᓕᒪᓐ ᐃᓗᐊᓂ ᓄᓇᓕᑦ ᐅᒪᔨᓕᓚᓱᖅᔪᐊᑦᓴᓕᑦ ᐊᖏᔪᓂᒃ ᖃᐅᔨᓴᐃᓂᖃᖅᑎᑦᓴᓕᑦ ᑲᑎᓪᓕᓪᓕ 1,470-ᖏᓐᓂᑦ (ᓇᓗᓇᐃᖅᓱᓪᓚᑦ1).

$$\mathrm{BNL} = \frac{\mathrm{Max(Harvest)} + \mathrm{Avg(Harvest)}}{2}$$

$\mathrm{Max(Harvest)} = 1940$ and $\mathrm{Avg(Harvest)} = 1470$:

$$\mathrm{BNL} = \frac{1940 + 1470}{2}$$

$$\mathrm{BNL} = \frac{3410}{2}$$

ᐱᖅᓯᐅᑦᓴᐅᓗᑎᒃ ᐊᑐᐃᓐᓇᐅᔨᓴᑦ ᑐᓴᒃᖁᓴᓕ ᐃᓗᐊᓂᓕᑦ ᐋᖅᑭᒃᓯᓱᐊᑎᐅᔨᓴᑦ ᐅᓄᖅᓂᓚᓂᐊᖅᑕᖏᓐᓂᓵᑦ ᐃᓄᐃᑦ ᐃᓄᐊᓂᓴᓪᓕᑎᑦᓵᑦ ᑲᖅᑭᑕᐅᓯᒪᔨᓴᑦ ᒪᓕᓵᔪᓂᑦ ᐊᑕᐅᓂ ᓄᓇᓕᒃ ᐊᓄᐃᑦ ᐃᓄᐊᓂᓴᓪᓕᑎᑦᓵᑦ ᐅᓄᖅᓂᖅᖢᕐᓂᖅᑐᑦ 1,705-ᓱᕆᖅᑐᑦ ᑐᓴᑦ.

ᓄᓇᕗᑦ ᐅᒪᔨᓕᓗᓂᔨᐊᕐᓂᑦ ᑲᑎᒪᔨᖏᑦᑕ ᐃᓱᒪᓕᐅᕈᑎᖓ

ᓄᓇᕗᑦ ᐅᒪᔨᓕᓗᓂᔨᐊᕐᓂᑦ ᑲᑎᒪᔨᖏᑦᑕ ᑕᐃᒪᐃᒪᒻᒪᑦ ᐊᐠᑭᑲᓱᕗᑦ ᐃᓄᐃᑦ ᐃᓄᐠᔨᑎᑲᓗᐊᖅᑕᖏᓐᓄᑦ ᐅᓄᖅᓯᓗᓂᐊᖅᑕᖏᓐᓄᑦᓯᒃ ᓴᓕᖅᓯᓗᓂᐊᖅᑕᖏᓐᓄᑦᓯᒃ ᖃᓄᐃᓕᖅᑐᑦ 1,705-ᖏᑦᑐᒃ.

ᑎᑎᕋᖅᑐᒃ: ᑕᓪᓕᒃ ᐃᐊᓐᑎᓗ, NWMB

ᖃᐅᔨᑭᔨᖅ: ᓱᑎᑦ ᑎᒃ, Lᓕᖅᓴᓕᓚᓕ, NWMB

ᑲᐃᑭᓕ ᓇᑦᓱ, NWMB

ᕿᐃᕆᓕ ᐊᖏᐊᑭᖅ, NWMB

ᐅᓕᒍᐊ: ᓱᑎᐱᓇ 11, 2025

ᓴᓗᓴᐃᖅᓱᐅᑦ 1. ᑐᑲᑐᑕᐅᔨᑦ ᑎᑎᕋᖅᑕᐅᓂᔨᑦ ᕿᓕᓂᒍᓐᓇᓂᒃ ᐊᖏᖅᑐᒍᓐᓇᑦ ᑕᓕᓪᓕᓄᑦ (1997-2001) ᓄᓇᕗᒥ ᐅᒪᔨᓕᓗᓂᔨᐊᕐᓂᑦ ᑲᑎᒪᔨᖏᑦᑕ ᐊᖏᕐᕕᓂᒃ ᖃᐅᔨᕿᐃᐃᓂᖅᖃᖅᑎᓕᓂᑦ.

ᓄᓇᕗᒥ ᐅᒪᔨᓕᓗᓂᔨᐊᕐᓂᑦ ᐊᖏᕐᕕᓂᒃ ᖃᐅᔨᕿᐃᐃᓂᖅᖃᖅᑎᓕᓂᑦ	ᓄᓇᕗᓂᒃ	ᓂᔨᐊᑎ	ᐅᓄᔨᓂᖏᑦ ᐊᖏᕐᕕᐅᔨᑦ

ᐅᑦᓗᐊ			
ᐊᖏᖅᒍ 1 (1997)	ᓴᓪᓕᖅ	ᑐᑲᑐ	1,376
ᐊᖏᖅᒍ 2 (1998)	ᓴᓪᓕᖅ	ᑐᑲᑐ	1,213
ᐊᖏᖅᒍ 3 (1999)	ᓴᓪᓕᖅ	ᑐᑲᑐ	1,940
ᐊᖏᖅᒍ 4 (2000)	ᓴᓪᓕᖅ	ᑐᑲᑐ	1,693
ᐊᖏᖅᒍ 5 (2001)	ᓴᓪᓕᖅ	ᑐᑲᑐ	1,128
		ᐊᖏᖅᒍᓂᑦ ᑕᑦᓯᒪᓂᑦ ᐊᑯᓐᓂᐊ	1,470

ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦᑕ ᑲᑎᒪᔨᖏᑦ
Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐱᓕᕆᑦᑐᐃᓕᓂᖅ ᐅᒪᔨᖏᑦ ᐊᖏᔪᑎᒋᓕᒃ ᐊᑐᕐᑎᐅᓂᖏᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᓂᕐᖏᑦ ᐊᓪᓚᓕ ᖃᐅᔨᓴᕐᑎᓪᓚᓕᒋᑦ ᖃᐅᔨᒪᓂᖏᑦ
Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

ᔮᓇ 18, 2024

ᐱᔨᔪᑎᖏᖕᒪᖏᑐᖅ: ᓴᓇᔨᐅᓂᖓᓂᑦ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᑭᓯᖓᓂᑦ (ᑯᑕᖓᑦ) ᓄᓇᕗᑦ ᐃᓗᐊᓂ

ᓄᓇᕗᑦ ᐅᒪᔨᕆᓚᓕᔨᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ (“NWMB” ᐅᕐᓂᓇᓗᓂᒃ “ᑲᑎᒪᔨᑦ” ᑐᓂᓯᓕᖅᑐᑦ ᐃᓕᓐᓄᓄᑦ ᑐᑭᖅᑎᑦᑎᔨᔪᑎᒃ ᓴᓇᔭᒻᒪᕆᓂᖓᓂᒃᓂᓂᒃ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᐅᓄᔨᓂᓚᓂᐊᖅᑕᖓᓂᒃᓂᓂᒃ (ᑯᑕᒥᓂᐊᖅᑕᖓᓂᒃᓂᓂᒃ) ᑕᒪᒃᑯᓄᖅ ᐅᒪᔨᓄᑦ ᐃᓗᐃᑦᑐᕆᖅᓯᒪᔭᑯᑦ ᐊᖓᔪᐅᔭᕐᓂᖅ ᑭᓯᖓᓂᑦ (ᑯᑕᖅᑐᑐᓄᑦ) (“TAH”) ᓄᓇᕗᑦ ᓄᓇᔨᑯᑎᓯᒪᔨᖅᑎᑕ ᐃᓗᐊᓂ.

ᑐᑭᖅᑎᑕᐅᕗᑎᑦ ᖃᐅᔨᔭᖅᑕᐅᔪᓗᑎᑦ, ᑲᑎᒪᔨᔨᓯᑕᐅᓕᐅᖅᑎᓪᓗᒍ ᑕᒪᓐᓇ ᑐᓐᖓᕆᔨᖅᓴᓗᑎᒃ ᑐᑭᖅᑎᓴᓂᒃ ᐱᓯᒪᔨᐅᔨᓂᒃ ᐊᑐᐃᓐᓇᐅᔨᓂᒃ, ᑲᑎᒪᔨᑦ ᐊᑐᔨᓂᐊᔨᓂᖓᓐᓂᒃ ᖃᓇᑕᐅᔭᒃᑯᑦ ᓄᐊᑕᐅᓯᒪᔨᓂᒃ ᑐᑭᖅᑎᓴᓂᒃ 2004-ᖓᔪᓕᒍ ᐊᖓᔪᐅᔨᓂᒃ ᖃᐅᔨᔪᕿᑕᐅᕆᐅᖅᓯᒪᔨᓂᑦ (“ᐊᖓᔪᓂᒃ ᖃᐅᔨᐃᓂᖃᖅᑎᓕᒍᑦ”) ᐊᑐᖅᑕᐅᓗᑎᒃ ᐋᖅᑭᒃᓯᓗᐃᓂᐊᖅᑐᑦ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᑭᓯᖓᓂᑦ ᓄᓇᕗᑦ ᐃᓗᐊᓂ ᑐᑭᖅᑎᓴᓂᒃ 5.4.5 ᓄᓇᕗᒻ *ᐊᖓᔪᑦᑎᑦ*ᐃᓗᐊᓂ.

ᐅᑯᓂᖓ ᑎᑎᖅᑲᓂᑦ ᐱᖅᓯᐅᑎᓯᒪᔨᑦ ᑎᑎᕋᖅᓯᒪᔨᑦ ᐅᓂᒃᑲᖅᓯᒪᔨᑦ ᖃᓄᖅ ᓄᓇᕗᑦ ᐅᒪᔨᔨᓕᕆᔨᖅᑎᑦᑕ ᑲᑎᒪᔨᖏᑦ ᐃᓯᒪᓕᐅᕆᐅᓪᓚᓕᒪᓐᓇᑦ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᐊᓯᒪᓕᐅᕆᐅᓪᓚᓕᒪᓐᓇᑦ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᐊᖓᔪᐅᔨᓂᖓᓐᓂᒃ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᐊᑐᐃᓐᓇᐅᔨᓂᒃ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᐃᓗᐊᓂᖅᑐᑦᑕ ᑭᓯᖓᓂᑦ ᐃᓄᐃᑦ ᓂᕈᐯᓴᖃᓂᐊᕐᓗᓂᑦ ᐅᒪᔨᕈᓐᓇᖅᑕᖓᑕ ᑭᓯᖓᓂᑦ.

ᑎᑎᕋᖅᑕᖅ,

Ḉᓂᐅᓪᓗ ᓱᐅᓴᒃ
ᐃᒃᓱᐊᐅᑕᖅ
ᓄᓇᕗᑦ ᐅᒪᔨᕆᓚᓕᔨᔪᐊᑦ ᑲᑎᒪᔨᖏᑦ

ᐱᓇᐅᓂᖏᑦ ᐃᓄᐃᑦ ᓂᕈᐱᐊᓂᐊᖅᓯᒪᔪᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖏᑦ ᐊᑐᖅᑕᐅᔪᖅᑎᑦ ᐃᓗᐊᓂ ᓄᓇᕗᒻᒥ ᓄᓇᕗᑦ ᓄᓇᕗᑦᑐᑦᑕᕋᔭᖏᑦ

ᐃᓄᐃᑦ ᓂᕈᐱᐊᓂᐊᖅᓯᒪᔪᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖏᑦ (BNL)

ᑎᑎᕋᖅᓯᒪᓂᖓ 5.6.19 *ᓄᓇᕗᑦ ᐊᖏᕕᑎᖏᓐᓂᒃ* ᒪᓕᒪᖃᖏᒪᑦ ᐃᓄᐃᑦ ᓂᕈᐱᐊᓂᐊᖅᓯᒪᔪᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖓ (ᓯᓚᓕᐅᕐᓴᖅᑐᓄᑦ ᐱᓕᕐᓂᐊᕐᓂᖅᑐᓄᑦ ᐃᓄᐃᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ ᑭᓚᖓ ᐊᐅᐅᑎᓯᒪᔪᑦ ᑭᓚᖓ ᐊᑐᕆᐊᖃᖅᑎᖓᓂᒃ ᐊᓯᖏᓪᓗᓂᒃ ᐃᓄᐃᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ, ᑎᑎᕋᖅᓯᒪᓂᖓ 5.6.16-ᒥᑦ ᑎᑭᓕᔪᐊᖓ 5.6.18 ᐃᓗᐊᓂ *ᓄᓇᕗᑦ ᐊᖏᕕᑎᖏᓐᓂᒃ*.

ᒪᓕᓴᔪᐊᖓ ᑎᑎᕋᖅᓯᒪᓂᖓ 5.6.20 ᐃᓗᐊᓂ *ᓄᓇᕗᑦ ᐊᖏᕕᑎᖏᓐᓂᒃ*, ᐃᓄᐃᑦ ᓂᕈᐱᐊᓂᐊᖅᓯᒪᔪᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖓ (ᓯᓚᓕᐅᕐᓴᖅᑐᓄᑦ), ᑕᐅᔭᓄᑦ ᐃᓗᐊᓂᒃᑯᑦᑎᓐᓂᒃᑐᑦ ᐃᓄᐃᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ ᑭᓚᖓ ᐅᓄᖅᑐᓄᑦ (ᓯᓚᓕᐅᕐᓴᖅᓂᑦ), ᑕᐅᔭᓄ ᐊᑕᖅᑭᔪᓂᑦ ᐊᖓᔪᓚᐅᓴᖅᑎᖅᑐᑦ ᐃᓄᖕᓄᑦ ᐊᑐᑎᓪᓗᒍᑦᑐᓄᒃ ᓴᓂᖅᓭᑕᐅᔭᓕᓪᓗᒍᑦᑐᓄᒃ ᐃᓄᐃᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ.

ᑕᐃᒪᓕᓗ ᐃᓗᐊᑦᑐᐃᓇᖅᓯᒪᔪᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ ᑭᓚᖓ (ᓯᓚᓕᐅᕐᓴᖅᑐᓄᑦ) ᐅᓄᖅᓂᖅᓴᐅᑦᑐᑦᑕ ᐃᓄᐃᑦ ᓂᕈᐱᐊᓂᐊᖅᓯᒪᔪᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖓᓂᒃ, ᐃᓄᐃᑦ ᓯᓚᓕᐅᕐᓴᖅᑐᓄᑦ ᐃᓄᐃᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖓᓂᒃ ᐃᓄᐃᑦ ᓂᕈᐱᐊᓂᐊᖅᓯᒪᔪᑦ ᐃᓄᐃᑦ ᐊᖓᔪᓚᐅᑎᓂᖏᑦ ᐅᒪᔨᕋᓱᒃᑕᕐᕕᒃ ᑭᓚᖓᓂᒃ

ᓄᓇᕗᒻ ᐊᖏᔪᐅᔨᓂᖅ ᖃᐅᔨᓴᐃᓂᐅᓕᐅᖅᓯᒪᔨᖅ (1996-2001)

ᒪᓕᒃᓴᐃᒍ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓂᖏᑦ ᑎᑎᕎᖅᓯᒪᓂᖏᓐᓄᓂ 5.4.5 ᐃᓗᐊᓂ ᓄᓇᕗᑦ ᐊᖏᕈᑎᖏᓐᓂ, ᐱᒃᔨᐊᑎᖏᓐ ᐊᖏᔪᓂᒃ ᖃᐅᔨᓴᐃᓂᖅ ᑐᐱᖁᓂᖅᑯᓗᒃᑐᑦ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐃᑲᔪᖅᓯᒪᔨᐊᕐᓂᖅ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐃᓄᐃᑦ ᓂᕈᐯᕐᓗᒍᓂᖅᑎᓪᓗᑎᑦ ᐃᓕᔨᒃᑯᓐᓂᖅ ᐅᒪᔨᕐᓂᖅ ᐊᒻᒪᓗ ᐃᑲᔪᖅᓯᒪᔨᐊᕐᓂᖅ ᐃᓄᐃᑦ ᐅᖃᓕᒫᓂᖅᑎᑦ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐊᒻᒪᓗ ᐃᓕᓴᕐᓂᐊᕐᓂᖅᑎᑦ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐊᒻᒪᓗ ᐃᓕᓴᕐᓂᐊᕐᓂᖅᑎᑦ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ ᐊᑐᖅᑕᐅᓗᒍᑎᒃ

> “ᐃᓗᓕᒪᖏᑦ ᓄᓇᕗᑦ ᐊᖏᕈᑎᖏᓐᓂᓂ ᐅᖃᖅᓯᒪᔭᒪᓕᒐ ᐃᓄᐃᑦ ᓂᕆᐹᖅᓯᐊᓂᖅᓴᓕᑦᓯᓗᑦ ᐃᒪᓕᕆᔨᒋᑦᑎᐊᖅ
> ᑭᓚᖓᔪ ᐅᓄᕐᓂᒪᔭᖓ ᐊᖏᔪᓄᕆᑕᐅᔫᑦᑐᑦ ᐊᖏᕐᕋᖓᓂᓂ ᐊᒻᒪ ᐃᓚᖅᑯᓂᖅᓴᓂᓂ ᐊᑐᖅᑐᑦ. ᐊᖏᔪᓴᓂ
> ᐅᕝᕙᓚᓐᓂᓂ ᓄᐊᑕᐅᓂᓂ ᐊᒋᑦ, ᖃᓄᐃᔪᐃᑦ, ᖃᐅᒥᑦ ᓂᖅᓂᑦᑐᖅᓂᑦ, ᒥᓄᓕᓐᓂᑦ, ᐊᓂᖅᑎᓪᓗ, ᐃᓚᖅᑭᒪᔭᐅᑎᕐᓗᑎᑦ
> “ᐃᓄᐃᑦ ᐊᑐᕆᐊᖅᑕᖅᑎᓂᓐᓂᑦ” ᐱᖅᓂᐅᕐᓴᐅᓪᓗᑎᑦ ᑕᐅᓂᐊ ᐱᓴᓱᖅᑕᐅᓂᒪᓚᖅᑎᓪᓗᑎᑦ ᐊᖏᔪᓄᕆᑕᐅᑎᒍᑦ
> ᓄᓇᖓᓂᐅᑲᕐᓂᓂᓂᓂᐅᓂᑦ ᐅᕝᕙᓚᓐᓂᓂ ᐃᓚᖅᑭᒪᔭᐅᑎᒍᑦ ᓄᓇᖅᑲᑎᐸᕐᓂᑐᓄᑦ ᓂᐅᕐᑐᑎᐊᖅᑐᑦ. ᑭᓇᑐᐃᓐᓇᐃᑦ
> ᐊᖏᔪᓴᓂ ᐅᕝᕙᓚᓐᓂᓂ ᓄᐊᑕᐅᓇᓱᑦ ᐅᓂᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᓄᓇᕗᑦ
> ᓄᓇᖏᑎᒃᓯᓇᓱᖓᓚᒋᑦ ᓇᓪᒋᑦᓂ ᐱᖅᓂᐅᕐᓴᐅᓚᖅᑎᓂᑐᑦ ᐃᓄᐃᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᐱᒃᐊᖅᑭᓐᓄᑦᓂᓐᓂᑦ.”

ᑐᓐᖓᕕᒋᓪᓗᑎᑦ ᑐᑭᒧᐊᖅᑎᑎᕐᓂᕐᓄᑦ ᑲᑎᒪᔨᖏᓪᓕᐊᓂᖅᓴᓂᑦ ᐃᓇᓚᑎᔪᓂᑦ ᐃᓗᓕᕆᔨᒋᑦᑎᐊᖅᑐᑦ
ᐱᖅᓂᐅᕐᓴᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᐃᓄᐃᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᐅᓇᓚᑎᔪᓂᖅᓴᓂᑦ, ᑭᑭᓪᓗᑎᑦ ᐊᖏᔪᓄᕆᑕᐅᑎᒍᑦ
ᐊᒥᓐᓄᑦ, ᐅᓚᓕᔪᐊᖅᑐᑦ ᐅᓄᖅᑐᓄᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓂᐊᖅᑎᓪᓗᑎᑦ ᔪᓚᑦᑕᐃᑦ ᓄᓇᕗᑦ ᓄᓇᖓᓂᐅᑲᕐᓂᓂᓂᓂᐅᓂᑦ
(ᑭᓇᐅᔨᓚᐅᕆᔪᐊᕈᑕᐅᓪᓗᑎᓗᒃ ᐊᖏᔪᓄᕆᑕᐅᔫᑦᑐᑦ”) ᑕᑯᓂᐊ ᑎᑎᕆᖅᑭᐅᓚᐅᖅᑐᓂᓂᑦᑐᑦ ᐊᖏᔪᓴᓂᑦ
ᖃᐅᔨᓴᐃᓂᓕᖅᑎᓪᓗᑎᑦ ᑎᑎᕆᖅᑭᐅᓚᐅᖅᑐᓂᑦ.

ᐊᕐᕌᔪᓄᑦ ᑕᓪᓕᒪᓄᑦ ᐅᓂᒃᑲᓕᐊᖏᓪᓕᔨᑦ ᑐᓂᐅᖅᑲᖅᑕᐅᓚᐅᖅᓯᓕᔨᑦ ᓄᓇᓪᓕᓐᓄᑦ ᑐᓴᖅᑎᕆᑎᓴᔨᑕᐅᓪᓗᑎᑦ ᐊᒻᒪ
ᖃᓚᑕᐅᓴᒃᑯᑦ ᓄᐊᑕᐅᓯᓕᔨᑦ ᑐᓴᓴᓐᓂᑦ ᔪᓪᓕᒐᓐᓂ ᓇᓗᓇᐃᖅᑎᖅᑕᐅᓴᔪᓯᐊᖅᑐᑎᒃ ᐃᔪᓪᓗᑐᓄᑦ ᐊᖏᔪᓄᕆᑕᐅᑎᒍᑦ
ᐅᖃᖅᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᐅᑭᐅᖅᑕᐅᓚᐅᖅᑐᓂᑦ, ᐱᖅᓂᐅᕐᓴᐅᓪᓗᑎᑦ ᐅᖃᓕᒫᓂᓴᓯᐊᖅᑐᓂᑦ ᐊᐅᓚᓂᖃᖅᓯᒪᔭᖅᑐᑦ
ᑕᒪᓐᓇᐅᔭᓂᑦ ᐊᓐᑐᖅᑕᐅᓚᐅᖅᑐᓂᑦ ᑐᓴᒪᓂᖅᑕᐅᖅᑕᐅᓂᑦ, ᐱᖅᓂᐅᕐᓴᐅᓪᓗᑎᑦ ᐅᖃᓕᒫᓂᓴᓯᐊᖅᑐᓂᑦ ᐃᓚᖅᑭᑎᒃᓴᓂᐊᓯᒪᔪᑦ
2004-ᖁᑎᓪᓗᒍ.

ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᑲᑎᒪᔨᖏᓪᓕᐊᓂᖅᓴᓂᑦ ᐊᑐᓕᖅᑎᓂᓯᓂᖏᓂᑦ ᐃᓇᓚᑎᔪᓂᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᐅᒪᔨᓕᕆᔨᒋᑦᑎᐊᖅᑐᑦ ᑭᓚᖓᔪ (2005–2010)

ᐊᖏᔪᓂᑦ ᖃᐅᔨᓴᐃᓂᓕᖅᑎᓪᓗᑎᑦ, ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᑲᑎᒪᔨᖏᓪᓕᐊᓂᖅᓴᓂᑦ ᐱᐅᓂᖅᑎᓯᓕᐅᖅᑐᓄᑦ ᐊᐅᓚᓂᕐᓂᖅᓴᓕᑦᓯᓗᑦ
ᓴᓇᐅᖅᓯᒪᔭᓂᑦ ᐃᓇᓚᑎᔪᓂᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᐅᒪᔨᓕᕆᔨᒋᑦᑎᐊᖅᑐᑦ ᐊᖏᔪᓄᕆᑕᐅᑎᒍᑦ ᐃᓇᐃᑦᑐᓄᓕᑦ ᐃᓇᐃᑦᑐᓄᑦᓯᓗᑦ
ᑭᓚᓂᐅᑕᐅᓂᒪᓕᓂᔅᓴᐅᑐᓄᓕᑦ ᐅᓚᓕᔪᓕᑦ (ᑯᑕᖅᑲᓂᔅᓴᐅᑐᓄᓕᑦ) ᐱᐅᓂᖅᑕᐅᓚᐅᖅᑐᓂᑦ ᐊᖏᔪᓄᕆᑕᐅᑎᒍᑦ 2005-ᖁᑎᓪᓗᒍ.

ᑎᓄᕐᓴᕐᒥ ᖃᓚᕋᓚᑦᓯᐊᖏᓐᓄᑦ ᐃᓇᓚᑎᔪᓂᑦ ᓂᐅᕐᑐᑎᐊᖅᑎᓪᓗᑎᑦ ᐅᓚᓕᔪᐊᓕᓴᐅᑎᑦᑐᑦ ᑭᓚᖓᔪ

ᐊᖏᔪᐊᓂ 2005, ᓄᓇᕗᑦ ᑐᓐᖓᕕᖅᑯᑦ ᑎᒥᖓᑦ (“NTI”) ᐊᖏᕐᓴᐅᓚᐅᖅᑎᓚᒥᒃ ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᑲᑎᒪᔨᖏᓪᓕᐊᓂᖅᓴᓂᑦ
ᐃᓇᓚᐅᖅᑕᑎᑎᕐᓂᖅᓴᓂᓐᓂᖅ ᐊᑲᑭᓯᓚᑎᓪᓗᑎᑦ ᐃᓚᐃᑦᑐᕆᔨᖅᑯᑦ ᐊᒻᒪᓗᑦᓯᓚᖅᑭᑦ ᑭᓚᖓᔪᓂᖅ 500 ᑕᓐᓂᖅ
ᐅᖃᓕᓴᐅᓂᐅᔪᓂᖅ ᑎᓄᕐᓴᐅᓪᓗᑎᒃᓴᓂᖅ ᓯᓚᖃᓯᒪᔭᖅ ᐊᒻᒪᓗ ᐃᓇᓚᑎᔪᓂᑦ ᐅᒪᔨᓕᕆᔨᒋᑦᑎᐊᖅᑐᑦ ᑲᑎᒪᔨᖏᓪᓕᐊᓂᖅᓴᓂᑦ ᐅᓚᓕᔪᐊᓕᓴᐅᑎᑦᑐᑦ

ᑭᓪᓕᒧᐊᖅᑕᖏᓚᐅᖅ 4.4 ᑕᓪᓕ5ᓂᖅ ᐅᖃᐅᒪᓕᐃᓂᖅ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ <ᓪᓂᓂᖅᑐᑦ ᓄᓇᖏᓪᓂ, ᐊᐃᕐᓂᓗᑎᒃ ᐃᒪᓪᓗ "ᓄᓇᕗᑦ ᐊᖏᑭᑎᑎᓪᓂ ᐱᑕᖅᑭᑦᒪᑦ "ᓂᕆᐅᓴᐅᓂᐊᖅᓯᓗᑎᒃ" ᑐᑭᖏᓚᐅᖅ "ᓂᐅᕿᑭᑎᐅᓴᖅᑭᑦᑐᓄᑦ" ᐅᕐᕕᒃᓯᓂᑦ "ᐊᐅᓪᓚᖅᑎᑕᐅᓂᐊᖅᑭᑦᑐᓄᑦ ᑲᓇᑕᑦ ᓄᓇᕗᑦ, ᑕᓪᓕ ᐃᓕᒪᓂᐅᑕᐅᓚᐅᖅᑐᑦ ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᓪᓂᓄᑦ ᐊᐃᕐᓂᓗᑎᒃ ᑲᑎᒪᔨᖏᑦ ᑕᒪᓐᓇᖅᑐᓄᑦ ᐃᓕᓴᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᑕᒪᓐᓇᖅᑐᓄᑦ ᐃᓕᓴᕆᔨᒃᑯᑦ ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᐃᓕᓴᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᑕᒪᓐᓇᖅᑐᓄ

ᐊᑐᖅᑕᐅᔾᔨᒃᑲᓇᐊᖅᑐᓄᑦ ᐃᓄᓐᓄᑦ, ᓄᓇᖅᑲᑎᒃᓴᖏᑦᑐᑦ ᓂᐅᕗᑎᖅᖃᖅᑕᐅᓂᒃᑎᓪᓗᑎᑦ, ᐊᒻᒪ ᓂᐅᕗᑦᑕᐅᖃᖅᑕᖅᑎᓪᓗᑎᑦ ᓂᕆᐹᒃᓴᐅᓂᒃ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᐃᓕᓴᐊᓂ ᓄᓇᕗᒻ ᓄᓇᖏᑎᑕᓇᒃᓴᐅᓯᒪᔪᓐ.

ᐃᓱᒪᑎᖅᓯᐅᑎᓗᑎᑦ ᐊᐅᓚᐅᑕᐅᔪᑦ ᑐᓂᔭᐅᓪᓚᐅᖅᓯᒪᔪᑦ ᐃᓕᓴᓇᖏᓐᓄᓄᑦ ᐊᐅᓪᓚᑦᑎᓂᐅᔭᖅᑕᐅᔪᓄᑦ, ᓄᓇᕗᑦ ᐅᒪᔨᓴᕐᓂᐊᕐᓂᖏᑦ ᐊᖏᑦᑐᖅᓯᓚᐅᖅᑐᑦ ᐱᖓᓱᒪᓂᖅ ᐃᓱᒪᑎᖅᖏᓄᓄᑦ ᐱᖓᓱᖅᑭᑦᑐᖅᑐᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᓯᐅᑎᓗᑎᑦ ᐱᓕᕆᔨᒃᑲᓐᓂᕐᒥᓄᑦ ᐃᓕᓴᕐᓂᐊᕐᓂᖏᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ, ᓄᓇᕗᑦ ᐅᒪᔨᓴᕐᓂᐊᕐᓂᖏᑦ ᐊᖏᑦᑐᖅᓯᓚᐅᖅᑐᑦ ᐊᒻᒪ ᓂᐅᕗᑦᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ, ᐅᒪᔨᓄᑦ (ᑯᑕᖃᓕᕐᓱᖅᑐᓄᑦ) ᐃᓄᐃᑦᑐᐃᖅᓯᒪᔪᑦ ᐊᑐᐃᓐᓇᖅᓯᒪᔪᑦ ᑭᓪᓕᖅᓯᒪᔪᑦ.

ᑭᓱᓚᐃᑦ ᖃᓄᖅᑐᐊᓂ ᐃᖃᓗᐅᑕᐅᔭᓱᓇᖅᑐᑦ ᐃᓄᐃᑦ ᓂᕆᐹᒃᓴᐅᓂᒃᑎᓪᓗᑎᑦ ᐅᒪᔨᓴᕐᓂᐊᕐᓂᖏᑦ ᑭᓪᓕᖅᓱᒪᓂᖅ

ᐃᑦ 2009-ᖓᔪᓐᓪᓗᒍ, ᓄᓇᕗᑦ ᐅᒪᔨᓴᕐᓂᐊᕐᓂᖏᑦ ᖃᓄᖅᑐᐊᓂ ᐅᓪᓚᑦᑎᓂᐅᓪᓗᑎᑐᒃ ᖃᓄᖅᓱᐊᖅᑎᓪᓚᐅᖅᑐᓄᑦ ᐃᓱᒪᑎᖅᖏᓄᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᑭᓪᓕᖅᓱᒪᓂᖅ (ᑯᑕᐃᑦ) ᐱᓪᓗᑎᑦ ᐊᒻᒪ ᐃᓄᐃᑦ ᓂᕆᐹᒃᓴᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᐅᒪᔨᓴᕐᓂᐊᕐᓂᖏᑦ ᑭᓪᓕᖅᓱᒪᓂᖅ ᐃᖃᓗᐅᑕᐅᔭᖅᑕᐅᓂᒃᑐᖅᑐᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓪᓗᑎᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᕐᓂᐊᕐᓂᖏᑦ ᐊᖏᑦᑐᖅᓯᓚᐅᖅᑐᑦ ᐱᓕᕆᔨᒃᑲᓐᓂᕐᒥᓄᑦ ᐃᓱᒪᑎᖅᖏᓄᓄ

ᓂᐅᕐᑭᑎᖏᖅᑕᐅᓂᖏᓐᓄᑦ, ᐊᑦᒪᓗ (c) ᓂᐅᕐᑭᑎᕿᐅᔪᓂᒃ ᓂᖏᕿᐅᓗᑎᑎᒃ ᐅᐟᙵᔭᓐᓂᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᒃ ᓄᓇᕗᑦ ᓄᓇᕐᔪᑎᑦᑕᓕᕆᖏᓐᓇᑦ ᐃᓗᐊᓂ.

ᒥᓂᑦ ᐅᖅᖃᓐᓂᒃᑐᑦ ᐱᖅᓯᐅᕐᓴᐅᑎᓕᑦᑐᑦ ᓂᐅᕐᑭᑎᕿᑦ ᓄᓇᕗᑦ ᓯᓚᑦᓴᓐᓂᑦ ᐃᓄᐊᓵᑎᓕᑦᑐᑦ ᐃᓄᓇᑦ ᓂᖏᕿᓂᐊᓂᔪᓂᑦ ᐅᒪᔨᕐᓴᐅᑎᓕᑦᑐᑦ ᐃᓄᐃᕐᓂᐊᕐᓂᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐊᒥᐊᓯᐅᑎᓕᑦᑐᑦ ᐃᓄᐊᓵᑎᓕᑦᑐᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᒃ ᑐᓂᐅᕐᑕᐅᓂᒃ ᐊᑐᑦᓗᒍ ᑎᑎᕋᖅᑕᐅᓂᒃ 5.6.31 ᐃᓗᐊᓂ ᓄᓇᕗᑦ ᐊᖏᕐᕉᑎᖏᓐᓂ.

ᒥᓂᑦ ᐊᔪᓚᖅᑎᑎᑎᓐᓗᒍ ᐃᓱᒪᓚᐅᕈᑕᐅᓚᐅᖅᑐᒥᒃ ᓄᓇᕗᑦ ᐅᒪᔨᓚᓕᕆᔨᖅᑐᑦ ᐅᒪᒪᔨᖏᓐᓄᑦ ᐃᓱᒪᓚᐅᕈᑎᑦᑎᓂᕐᓚᐅᖅᓯᒪᔭᖅᑐᓂᒃ, ᑭᖑᓪᓚᖅᑕᖅᓯᓂᒃᓂ ᓄᓇᕗᑦ ᐅᒪᔨᓚᓕᕆᔨᖅᑐᑦ ᐅᒪᔨᖏᓐᓄᑦ ᐃᓱᒪᓚᐅᓚᐅᖅᑐᑦ ᐱᓇᐅᔭᓕᐅᖅᑐᓂᑦ ᐃᓗᐃᑦᑐᓐᓇᖅᓯᒪᔨᓕᖅᑐᑦ ᐊᒻᒪᓗᐅᔪᓕᖅᑐᑦ (ᑯᑕᐃᑦ) ᐊᑦᒪᓗ ᐃᓄᐃᑦ ᓂᖏᕿᓂᐊᓂᔪᓂᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᑭᖑᓪᓚᖅᑕᖅᓯᒪᔭᖅᑐᓂᒃ ᐱᓇᐅᔭᓕᐅᖅᑐᓂᒃ ᐃᓚᐅᖅᑕᐅᓂᒃ. ᑕᐃᑦᓯᒪᓂ ᓇᓚᐅᓂᖏᖅᑎᓐᓗᑦᑐᑦ, ᓄᓇᕗᑦ ᐅᒪᔨᓚᓕᕆᔨᖅᑐᑦ ᐅᖅᑎᒪᓂᕐᑎᒃ ᓄᖅᖅᑎᖅᑎᓐᓗᑦᑐᑦ ᐊᖏᕐᕉᑎᖏᓐᓄᑦ ᐃᓱᒪᑎᐅᓚᐅᖅᑐᓂᒃ ᐊᑦᑭᑭᓯᐅᓚᐅᖅᑐᑦ ᐃᓄᐃᑦ ᓂᖏᕿᓂᐊᓂᔪᓂᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐱᓇᐅᔭᓕᐅᖅᑐᓂᒃ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᒃ ᐃᓄᐃᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑐᓂᒃ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐅᓂᓯᓚᐅᓂᖅᑐᓂᒃ ᐃᓄᐃᑦ ᓂᖏᕿᓂᐊᓂᔪᓂᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐅᒪᔨᓴᖅᑐᓂᑦ.

ᓴᓕᓪᓕ ᖁᑭᖅᑕᖓᓂ ᑐᑦᑐᑦ ᐃᓄᐃᑦ ᓂᖏᕿᓂᐊᓂᔪᓂᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᑦ ᐅᒪᔨᖏᓐᓄᑦ

ᑎᓯᐱᓗ 2013-ᖓᑎᓕᑐᒍ, ᓄᓇᕗᑦ ᐅᒪᔨᓚᓕᕆᔨᖅᑐᑦ ᐃᓄᐃᖅᑎᑦᑎᓂᑦ ᐊᖅᑭᑦᓯᓚᐅᖅᑐᑦ ᐃᓱᒪᓚᐅᓚᐅᖅᑐᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐃᓄᐃᖅᑎᑦᑎᓂᒃᑐᓂᒃ ᐊᓯᔾᔨᖅᑕᐅᑐᒍᒪᔨᓴᒃ ᐃᓗᐊᓂᑦᑐᐃᓇᖅᑐᓂᒃ ᐊᒻᒪᓗᐅᔪᓕᖅᑐᑦ ᐅᒪᔨᖏᓐᓄᑦ (ᑯᑕᑎᓂᐊᖅᑕᖅᑐᓂᒃ) ᓴᓕᓚᓂ ᑐᑦᑐᖅᑐᓄᑦ ᐊᒻᒪᓗ ᐊᖅᑭᑦᓯᓚᐅᖅᑐᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐅᒪᔨᖏᓐᓄᑦ ᓴᓕᓚᑎᐅᓂᒃ ᑐᑦᑐᖅᑐᓄᑦ. ᑭᐃᓕᓐᓂᒃ ᐃᓚᐅᖅᑕᐅᓂᒃ ᐅᒪᒪᔨᓴᖅᑎᑦᑎᓂᒃ, ᓄᓇᕗᑦ ᐅᒪᔨᓚᓕᕆᔨᖅᑐᑦ ᐅᒪᔨᓴᖅᑎᑦ ᓄᖅᑲᖅᑎᑦᓯᓚᐅᖅᑐᓂᒃ ᐱᖓᓱᖅᑯᓂᒃ ᓄᓇᕗᑦ ᓴᖅᓯᒪᔪᖅᑎᑦᑎᓂᒃ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐊᖅᑭᑦᓯᓚᐅᖅᑐᑦ ᐱᓇᐅᔭᓕᐅᖅᑐᓂᒃ ᐊᖅᑭᑦᓯᓚᐅᖅᑐᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐅᒪᔨᖏᓐᓄᑦ.

ᐃᓕᓴ 2014-ᖓᑎᓕᑐᒍ, ᓄᓇᕗᑦ ᓴᖅᓯᒪᔪᖅᑎᑦᑎᓂᒃ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐊᖅᑭᑦᓯᓚᐅᖅᑐᑦ ᐱᓇᐅᔭᓕᐅᖅᑐᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐊᖅᑭᑦᓯᓚᐅᖅᑐᑦ ᐃᓄᐃᑦ ᐅᒪᔨᓴᖅᑎᑦᑎᓂᒃ ᐅᒪᔨᖏᓐᓄ

ᓄᓇᕗᑦ ᑐᓴᒪᕐᒪᑦ ᐊᑭᓴᖅᑐᐃᑎᓕᐅᕙᖅᑐᑦ ᓄᓇᕗᑦ ᓚᕿᒪᔪᖏᑦ ᐃᓱᒪᓕᐅᕈᑎᑎᕐᓴᒪᓂᔪᒃ ᐊᒻᒪ ᓇᓗᓇᐃᖅᓱᓂᓕᐅᖅᑐᑦ ᓄᓇᕗᑦ ᐊᖏᕈᑎᖏᓐᓂ ᐱᖓᓱᐅᔨᕐᓯᒪᖏᓐᓂᔪᒃ ᐃᓄᐃᑦ ᓂᐅᕐᑎᑎᕆᓂᕆᐊᖃᕐᓂᑦ ᑭᓇᐅᔭᓕᐅᕐᓯᐊᕈᑎᒃ ᐱᖓᐅᔨᔨᓱᒪᓂᖏᓐᓂᔪᒃ ᐃᓄᐃᑦ ᓂᐅᕐᑎᑎᕆᓂᕆᐊᖃᕐᓂᑦ ᐅᒪᔪᖏᓐᓇᖅᑐᒥᓂᖏᑦ ᐊᓐᓴᖅᓯᐅᔨᕐᓴᒪᓂᕐᒥᒃ ᐃᓄᐃᑦ ᓂᐅᕐᑎᑎᕆᓂᕆᐊᖃᕐᓂᑦ ᐅᒪᔪᖏᓐᓇᖅᑐᒥᓂᖏᑦ ᑭᓪᓕᔪᐅᓂᒃ.

ᑕᐃᒪᓕᓗ ᐊᐃᓂᐅᑎᖏᑎᓐᓂᑦ ᐃᓱᒪᖏᓂᑦ ᐋᖅᑭᐱᓪᓚᕆᓂᕐᓂᓂᒃ ᑭᓱᓂ ᐱᖓᓱᐅᔨᕐᑐᐅᔭᖏᖅᓯᒪᔾᔪᑎᑦ ᐃᓄᐃᑦ ᓂᐅᕐᑎᑎᕆᓂᕆᐊᖃᕐᓂᑦ ᐅᒪᔪᖏᓐᓇᖅᑐᒥᓂᖏᑦ ᐱᓪᓗᒥᓂᖏᑦ, ᓄᓇᕗᑦ ᐅᒪᔪᓇᓂᔪᒃᓴᒪᓂᖏᑦ ᑲᓇᓪᓚᕆᐅᕙᖅᑐᑦ ᐃᓄᐃᑦᑐᐃᓚᓱᐊᖅᑎᓪᓗᒥ ᐊᐃᓐᓴᐅᔭᓕᐅᕙᖅᑐᑦ ᐃᓄᐃᑦ ᐅᒪᔪᖏᓐᓇᖅᑐᒥᓂᖏᑦ ᐊᐃᓐᓴᐅᔭᓕᐅᕙᖅᑐᑦ ᐅᒪᔪᖏᓐᓇᖅᑐᒥᓂᖏᑦ ᐱᓪᓗᒥᓂᖏᑦ ᐃᓄᐃᑦ ᓂᐅᕐᑎᑎᕆᓂᕆᐊᖃᕐᓂᑦ ᐅᒪᔪᖏᓐᓇᖅᑐᒥᓂᖏᑦ ᐊᐃᓐᓴᐅᔭᓕᐅᕙᖅᑐ

ᐊᖏᑦᖃᑎᒌᒍᓐᓇᖏᓂᖏᑦ ᐊᐅᓚᑦᑎᓯᐅᖃᖅᑕᐅᔪᑦ

ᑕᐃᒪᓕ ᐃᓱᒪᒋᔭᓗᑦ ᐊᖏᑦᖃᑎᒌᓂᐅᔪᓐᓇᖅᑐᒃᓯᑦ ᐊᖃᓯᑎᐅᔪᓐᓇᖅᑐᑦ ᐊᐅᓚᑦᑎᓴᐃᑎᐅᔪᓄᑦ ᐅᒪᔨᓕᕆᓂᖅᓴᓗᑦ ᐊᓯᖏᓐᓄᓂ ᐊᐅᓚᑦᑎᓯᐅᖃᖅᖃᓄᑦᑐᖅ, ᑲᑎᒪᔨᑦ ᓄᖅᑲᓐᓄᑦᓱᒃᑎᒌᒍᓐᓇᓚᐅᖅᑐᑦ ᐊᖃᖏᑐᓱᐃᓂᕆᓴᖏᓂᓂᒃ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᖃᐅᖅᓯᓂᓗᑦ ᑎᓕᐅᕆᓂᓗᑎᒃ ᐊᖏᑦᖃᑎᒌᒍᓐᓇᖅᑐᓄᑦ ᑕᒪᓐᓇ ᐱᓪᓗᒍ ᐱᖓᓱᐅᓴᖅᑲᐅᔪᐊᖏᖅᓯᒪᓕᒃᑐᑦ ᓂᐅᕗᖅᓯᓗᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ.

ᑲᑎᒪᔨᑦ ᑕᐃᒪᓐᓇᑦ ᓴᓂᖅᓭᐃᓕᒪᓐᓇᖅᑎᓯᓗᑦ ᐃᓗᐃᑦᑐᐊᓐᖏᓕᕐᒥᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᑦ ᑭᓯᓚᑦᑐᖅᓂᒃ (ᑯᑕᓂᒃ) ᐃᓄᖅᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓯᓗᑦ ᐃᓗᐊᖅᑕᖏᓂ ᐃᓕᖅᓄᖓᓕᓂ ᐊᖅᑭᑦᑕᐅᓕᕆᓯᓗᓂ ᑕᐃᒪᓐᓇ ᐊᖅᑭᑦᑕᐅᓪᓚᐅᖅᑐᓂᖅᓂᖅᓂ ᐅᓄᖅᓯᕆᓂᑲᖏᑦ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᑭᓯᓚᑦᑐᖅᓂᒃ, ᐃᓱᒪᒋᔭᓗᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ, ᐃᓱᒪᒋᔭᓗᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ.

ᑲᑎᒪᔨᑦ ᑕᐃᒪᓕᓐᓂᑦ ᓴᓂᓴᕿᐃᓕᒪᓐᓇᖅᑎᓯᓗᑦ ᐃᓗᐃᑦᑐᐊᓐᖏᓕᕐᒥᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᑦ ᑭᓯᓚᑦᑐᖅᓂᒃ (ᑯᑕᓂᒃ) ᐃᓄᖅᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓯᓗᑦ ᐃᓗᐊᖅᑕᖏᓂ ᐃᓕᖅᓄᖓᓕᓂ ᐊᖅᑭᑦᑕᐅᓕᕆᓯᓗᓂ ᑕᐃᒪᓐᓇ ᐊᖅᑭᑦᑕᐅᓕᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐅᓄᖅᓯᕆᓂᑲᖏᑦ, ᐃᓱᒪᒋᔭᓗᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ, ᐃᓱᒪᒋᔭᓗᑦ ᑭᓪᓕᒃᑐᓂᓚᐅᖅᑐᓂᖅᓂᖅᓂ ᐃᓗᐃᑦᑐᐊᓐᖏᓕᕐᒥᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ, ᐃᓱᒪᒋᔭᓗᑦ ᐊᖅᑭᑦᑕᐅᓕᕆᓯᓗᓂ ᐃᓗᐃᑦᑐᐊᓐᖏᓕᕐᒥᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ.

ᑲᑎᒪᔨᑦ ᑕᐃᒪᓕᓐᓂᑦ ᓴᓂᓴᕿᐃᓕᒪᓐᓇᖅᑎᓯᓗᑦ ᐃᓗᐃᑦᑐᐊᓐᖏᓕᕐᒥᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᑦ ᑭᓯᓚᑦᑐᖅᓂᒃ (ᑯᑕᓂᒃ) ᐃᓄᖅᓄᑦ ᐊᑐᖅᑕᐅᓂᐊᖅᑎᓯᓗᑦ ᐃᓗᐊᖅᑕᖏᓂ ᐃᓕᖅᓄᖓᓕᓂ ᐊᖅᑭᑦᑕᐅᓕᕆᓯᓗᓂ ᑕᐃᒪᓐᓇ ᐊᖅᑭᑦᑕᐅᓕᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ ᐃᓄᐃᑦ ᓂᖏᒃᖁᓂᐊᖅᓯᓗᑦ ᐊᖓᔪᐅᔨᓐᓇᖅᑐᓄᑦ ᐅᖃᓕᒫᓂᐊᕐᓗᒍ ᐅᒪᔨᓕᕆᓂᖅᓄᑦ ᐅᓄᖅᓯᕆᓂᐊᖅᑕᖏᓂᓂᒃ.

ᐊᖏᔪᑎ ᓯᓚᒻᒪᐊᓘᓐᓃᑦᑐᒻᒪᑦ

ᓄᓇᓕᑦ ᐃᒪᓕᕆᓂᖅᓴᖅᑎᑦ ᑲᑎᒪᔨᖏᑦ ᐅᑭᐱᕐᓯᓕᑦ ᐊᖏᔪᑎᑎᑦᑎᓂᓕᕆᓂᖏᓐᓂᓕᑦ ᐊᐅᓚᑦᑎᔨᐅᖏᑕᐅᔪᑦ ᑭᓯᐊᓂᖏᑦ ᐃᓗᓕᖅᑐᑦ ᑲᑎᒪᔨᖏᑦ ᐅᑭᐱᕐᓯᓕᑦ ᐅᑭᐱᕐᓯᓕᑦᑎᓂᑦ ᐃᓄᓇᓕᑦ ᓂᕈᐯᖓᓄᑦᑎᓂᑦ ᐃᓕᓴᕐᓇᖏᑕᖏᑕ ᐅᓄᖅᓯᓂᖅᓴᖅᑐᓂᖅ ᐊᐅᓚᑦᑎᔨᐅᖏᑕᐅᔪᑦ ᐊᖏᔪᑎ 14 ᐅᖓᑖᓄᑦ ᑕᐃᒪᐃᓕᖓᖓᖏᑕ ᐊᒻᒪ ᑲᑎᒪᔨᑦ ᐃᓐᓇᐅᔭᖅ ᐃᓕᓴᕐᓂᖅᓯᒪᓕᒡᒐᑦ ᑭᖑᓪᓕᖅᑎᑦᑎᓂᖅᑲᖅᓯᓂᖅᐸᑦ ᐊᐅᓚᑦᑎᔨᐅᖏᑕᐅᔪᑦ ᐃᓄᓇᓕᑦ ᓂᕈᐯᖓᓄᑦᑎᓂᑦ ᐃᓕᓴᕐᓇᖏᑕᐅᔪᑦ ᑲᑎᒪᔨᑦ ᑲᑎᒪᔨᖏᑦ ᐃᓄᓇᓕᑦ ᓴᓇᖅᑭᓕᖓᓂᖅᓴᖅᑐᓂᖅ ᐊᐅᓚᑦᑎᔨᐅᖏᑕᐅᔪᑦ ᐃᓕᓴᕐᓂᖅᓯᒪᓕᒡᒐᑦ ᐃᓄᓇᓕᑦ ᐃᓕᓴᕐᓇᖏᑕᐅᔪᑦ ᑲᑎᒪᔨᖏᑦ ᐃᓄᓇᓕ

ᑲᑎᒪᔨ ᓄᓇᕗᑦ ᐱᖃᓐᓇᕐᓂᖅᓴᓂᑦ
Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

ᒥᓂᑦ ᐊᕐᕌᒍᓕᕆᔨᒃᑯᑦ
Minister of Environment
Ministaat Avatiliqiyitkut
Ministre de l'Environnement

July 24, 2024

Mr. Daniel Shewchuk
Chairperson
Nunavut Wildlife Management Board
310-1106 Ikaktuutiak Drive
Iqaluit NU X0A 3H0

Re: Establishment of Basic Needs Level within the Nunavut Settlement Area

Dear Mr. Shewchuk,

I would like to acknowledge your letter dated July 18, 2024 wherein the Nunavut Wildlife Management Board (NWMB) describes how it will address establishing Basic Needs Level (BNL) for wildlife populations or stocks that are subject to a Total Allowable Harvest (TAH).

I note in your letter the long history and the absence of consensus by co-management partners on this issue. I recognize the tremendous efforts the NWMB has made on this issue over the years, and I look forward to working together as co-management partners as we continue to take actions to fully implement Nunavut's wildlife management system.

Sincerely,

Daniel Qavvik
Minister of Environment

Cc. Yvonne Niego, Deputy Minister, Environment

ᑎᑎᖅᑯᕕᐅᑉ 2410
ᐃᖅᓗᐃᑦ, ᓄᓇᕗᑦ X0A 0H0

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ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᑲᑎᒪᔨᖏᑦ
Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐃᓄᐃᑦ ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᐱᓕᕆᔨᒃᑯᑦᑐᑦ ᐅᒪᔨᓕᓐᓂᔨᖏᑦᑐᑦᓗ ᐊᑐᕐᓗᑎᑦ ᐊᓪᒪᒃ ᖃᐅᔨᓴᖏᓐᓂᒪᓕᒃᑕᑦ ᖃᐅᔨᒪᓂᓕᕆᔨᖏᑦᑐᑦ ᐅᒪᔨᓕᓐᓂᔨᓂᖏᑦᑐᑦ ᐅᓪᓗᓕ ᐊᑐᕐᓗᑎᑦ, ᐅᓪᓗᓐ ᐃᓄᖅᑲᐅᑎᒃᓴᖅᑐᑦ ᐅᖃᓕᒫᒃᓴᖅᓯᒪᓕᒃᑕᑦ ᐊᖏᓪᓛᓂᒃ ᓄᓇᕗᑦ.

Havarhait: Munaritjutikhaitnik angutirhat, hivuniktaulutik tahapkunanga angutirhaliqiniqut pimariuyunik ihumagiyaulluaktaitnik, pitiaqattiriqnahuaklutik Inuinait attuqparhimayaitnik angutirhanik atturhimaaklugit, innutiaqutirhait ihumagilugit tahapkua ublumi angunahuaktiuyut innuqutaitlu tatvalu kinguvarhavut tahamani Nunavumi.

Managing wildlife, guided by conservation principles, honouring the key role of Inuit in harvesting, for the well-being of current and future generations in Nunavut.

ᐊᖏᓪᓗ 07, 2025

ᑲᐃᓐ <ᓂᕐᒃ
ᐃᑦᓯᓂᐅᑕᖅ
ᐊᐃᐃᓴᑦ ᐊᖏᔪᓄᑦᑎᑦ ᐊᓚᒪᓗ ᒥᑭᒋᐊᖅᓂᐊᖅᑎᑦ ᑎᒥᖅᑎᑦ
ᓴᖅᓯᑦ, ᓄᓇᕗᑦ
X0C 0G0

ᐃᑦᓯᓂᐅᑕᖅ

ᐱᖅᓗᒍ: ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᓂᐅᕐᓯᓂᖅᑎᑦ ᓴᖅᓯᖅᒍᑦ – ᐃᓄᐃᑎᑎᐅᓄᕆᓕᓂᑦ ᓴᖅᓯᓂ ᑐᑦᑐᐃᑦ

ᐃᓕᐅᖅᓯᓗᓂ ᐅᓂᒃᓂ ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ (ᐅᒪᔨᓕᕆᔨᒃᑯᑦ) ᑲᒪᒋᑎᓕᕆᖏᓐᓄᓄᑦ ᐊᑕᒍᑦ ᓄᓇᕗᑦ ᐊᖏᓪᓄᐸᔪᑎᑦ, ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᐃᖅᓄᐃᐃᔨᕐᑎᓪᓚᑦ ᓂᐅᓕᓄᑦ ᒪᐃ 2025 ᑲᑎᒪᖅᑎᒋᐊᕐᑐᓕᒋᑦ ᐊᐃᐃᓴᑦ ᐊᖏᔪᓄᑦᑎᑦ ᐊᓚᒪᓗ ᒥᑭᒋᐊᖅᓂᐊᖅᑎᑦ ᑎᒥᖅᑎᑦ ᐅᖅᓀᒪᐅᓂᕆᓚᒋᑦ ᐊᐸᑦᑕᐅᓂᖏᑦ ᐃᖅᓄᐃᐃᔨᕐᑎᓪᓚᑦ ᓂᐅᓕᓂᓂ ᑐᑦᑐᐃᑦ.

ᐃᓄᐃᑎᑎᐅᓄᕆᓕᓂᑦ – ᓄᓇᕗᑦ ᐊᖏᖅᑲᑎᒪᔨᑦ 5.6.19 – 5.6.24
ᐊᑕᒍᑦ ᓄᓇᕗᑦ ᐊᖏᖅᑲᑎᒪᔨᑦ, ᐃᓄᐃᑎᑎᐅᓄᕆᓕᓂᑦ ᐃᓕᓐᓂᔨᐅᓂᒪᒃᑕ ᓂᕆᓕᓂᐅᑎᓕᕆᓂᑦ ᑐᓂᔨᐅᓚᓂᑎᑦ ᐅᒪᔨᐃᑦ ᐅᑯᐊ ᑲᑎᑦᑐᑎᑦ ᐊᔪᖏᑎᑦ ᐊᖏᔪᓄᑦᑕᐅᓂᖅᑎᑦ (ᐊᖏᔪᓄᓕᕆᓂᑦ) ᐊᖅᑭᑦᑕᐅᐸᐹᑦ, ᐃᓪᓕᓐᓂᔨᐅᓚᓂᑎᑦ ᐊᖏᔪᓄᑦᑕᐅᓂᖅᑎᑦ ᐊᖏᔪᓄᑦ

ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᑲᑎᒪᔨᖏᑦ
Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐃᓄᐃᑦ ᐅᒪᔨᐅᓂᒃ ᐱᐅᖃᕈᓯᖏᓐᓂᒃ ᐅᒪᔨᔨᓕᓴᐅᓯᖏᓐᓂᒃᓴᓗ ᐊᑐᕐᓗᑎᑦ ᐊᓯᒪ ᖃᐅᔨᓴᖃᑎᒌᒃᓴᑦ ᐅᒪᔨᔨᓕᓴᐅᓯᖏᓐᓂᒃ ᐅᒪᔨᔨᓕᓴᐅᓯᖏᓐᓂᒃ ᐅᒪᔨᔨᓕᓴᐅᓯᖅᑐᑦ, ᐅᓪᓗᓕᒻ ᐃᓄᖅᑲᐅᙵᑦ ᐅᖃᐅᓯᕙᓪᓕᐊᓂᖅᓴᓪᓗ ᐊᖏᓪᓛᓯᐅᓄᑦ ᓇᐃᓐᓇᕐᓂᑦ ᓄᓇᕗᑦ.

Havarhait: Munaritjutikhaitnik angutirhat, hivuniktaulutik tahapkunanga angutirhaliqiniqut pimariuyunik ihumagiyaulluaktaitnik, pitiaqattiriqnahuaklutik Inuinait attuqparhimayaitnik angutirhanik atturhimaaklugit, innutiaqutirhait ihumagilugit tahapkua ublumi angunahuaktiuyut innuqutaitlu tatvalu kinguvarhavut tahamani Nunavumi.

Managing wildlife, guided by conservation principles, honouring the key role of Inuit in harvesting, for the well-being of current and future generations in Nunavut.

ᓄᓇᕗᑦ ᐊᖏᖃᑎᒋᔭᒋᑦ ᐱᑕᖃᖅᑐᑦ ᒪᑭᒋᓐᓂᒃ ᐊᑐᖅᑕᐅᔪᓐᓇᖅᑐᓂᒃ ᖃᐅᔨᓴᓕᐊᖅᓯᓗᑦ ᐃᓅᓴᐸᑎᔨᐅᔨᓕᕆᓂᖅ:

1. ᐊᑐᑐᐃᓐᓇᖅᓯᓗᑦ ᓄᓇᕗᒻᒥ ᐅᒪᔨᖅᑕᓂᒃ ᖃᐅᔨᓴᖃᑎᒌᑦ (1996-2001).
2. ᐊᑐᖅᓯᓗᑦ ᓄᓇᕗᖅᒥ ᐅᒪᔨᖅᑕᓂᒃ ᖃᐅᔨᓴᖃᑎᒌᑦ ᐃᓕᖃᖅᓯᓗᑦ ᓄᓇᓕᒻ ᐊᖏᔪᑕᖏᓐᓄᓕᒃ ᑎᑎᖅᑯᓯᒪᔨᒃ ᐅᓂᖅᓱᒻᒪᑦ 2008 ᐅᕗᖅ 2012 (ᐊᑐᐃᓐᓇᐅᔨᐸᑕ).

ᒪᓕᒃᓯᓗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᐃᓱᒪᓕᐅᕈᑎᖅ, ᑐᐃᑐᑦ ᓂᐅᑦᑕᕐᕿᐅᓯᓗᑦ ᓂᕈᓕᕆᓇᖅᓄᑦ ᐃᓱᒪᒋᐅᖅᓱᐃᖅᑐᑦ ᐊᒋᔪᑦ ᐊᖏᔪᓂᒃ ᐊᖏᔪᓄᑦᓴᓗ ᐱᒪᓗ ᒥᑭᓗ ᑐᑭᐅᐊᖅᓂᐊᖅᑐᑦ ᑎᑎᖅᑯᓯᒪᔨᒃ ᐅᓂᖅᓱᒻᒪᑦ ᐊᖏᔪᓂᒃ ᐊᖏᔪᓄᑦᓴᓗ ᐃᓚᖓᑦ ᐊᑐᐃᓐᓇᐅᔨᐸᑕ.

ᐅᒪᔨᓕᕆᔨᖅᑎᑦ ᐃᖅᓗᐃᓕᖅᑎᖏᓐᓂᑦ ᑭᖑᓂᐊᔪᑦ ᐅᖅᓯᓐᓄᖅ ᐊᐃᓴᑦ ᐊᖑᓪᓚᑦ ᐱᖑᓯᓂᐊᖅᑐᑦ ᐱᓕᓗ ᒥᑭᓗ ᑭᑭᓯᐊᖅᑎᑦ ᑎᒥᖅᑐᑦ ᐃᓕᓴᕐᓂᖅᑐᑦ ᐅᖅᓯᓐᓄᖅᓴᓗ ᐊᑐᖅᑕᐅᓂᖅᓴᓗ ᐊᑐᖅᑕᐅᓂᖅᓴᓗ ᐊᑐᖅᑕᐅᓂ

By Email

Jason Akearok, Executive Director
Nunavut Wildlife Management Board

April 7, 2025

Dear Mr. Akearok:

Re: BNL for Southhampton Island Caribou

I write on NTI's behalf about the Nunavut Wildlife Management Board's ("NWMB") plan to set a Basic Needs Level ("BNL") for the Southampton Island caribou herd using a calculation that would "exclude any commercial harvesting records that may be available": NWMB staff memo of January 7, 2025, *Determination of the Basic Needs Level (BNL) for Southampton Island Caribou*, p. 2. That staff memo references a July 18, 2024 NWMB policy which states that the NWMB will use the Nunavut Wildlife Harvest Study data and thereby exclude from BNL calculations "wildlife harvested in bulk for sale outside of the Nunavut Settlement Area ("Commercial Harvests")": *Re: Establishment of Basic Needs Level within the Nunavut Settlement Area*, ("July 2024 Memo"), p.4.

Whether the NWMB intends to exclude from its BNL calculation for Southhampton Island caribou the harvests sold outside of Nunavut or *all* commercial harvests, NTI is deeply concerned by the methodology and the process leading up to its adoption.

NTI's Concerns

The NWMB started a public hearing in December 2013 to establish the BNL for Southampton Island caribou. It paused that hearing in 2014 to allow time for Nunavut's wildlife co-management partners to try and reach consensus on methodology. The NWMB's stated position was that BNL calculations must include all commercial harvesting, a position that NTI shared. However, the NWMB abandoned that position in its July 2024 Memo. It adopted a policy to exclude Commercial Harvests with no advance notice to NTI and without giving NTI and other co-management partners the opportunity to make submissions on this matter, which the NWMB had parked for a decade.

Moreover, the July 2024 Memo does not clearly explain why the BNL calculations should exclude Commercial Harvests. Does the NWMB's statement at p. 7 of the July 2024 Memo that "there is no authority within the *Nunavut Agreement* for the NWMB to modify the assumptions adopted by the Steering Committee" mean that the NWMB considers itself bound by the Steering Committee's decision? Or does it mean that the NWMB has decided that the Steering Committee interpreted Article 5 correctly? On what basis has the NWMB now rejected the 2009 legal opinion it received from Mr. Meehan, which was that BNL calculations should be based on all Inuit harvests?

BNLs have significant, long-term, food security, social and economic implications for Inuit right-holders. They may also affect the interests of non-Inuit seeking to harvest wildlife. The NWMB's

BNL decision for the Southampton Island caribou herd seems on track to be the first one to apply the policy from the July 2024 Memo. Given the significance of this BNL decision and the precedent it will set for future BNL calculations in Nunavut, procedural fairness is essential. NTI believes that fairness requires resuming and completing the public hearing the NWMB originally started to determine how to calculate BNLs.

Moreover, whether Article 5 of the *Nunavut Agreement* requires some or all Inuit commercial harvests to factor into the BNL calculation is a legal, treaty interpretation question with a correct answer. NTI urges the NWMB to take the time to re-engage with its co-management partners to support the most robust possible decision, as this should reduce the likelihood of a legal challenge and the uncertainty, delays, and expenses that would result from litigation.

NTI's Proposal

NTI requests that the NWMB pause the BNL calculation for Southampton Island caribou and first reconsider its planned exclusion of Commercial Harvests (or, according to NWMB staff, all Inuit commercial harvests) from the BNL calculation. We request that the NWMB take the following steps:

1. Provide NTI and other co-management partners with clear reasoning for its conclusion in the July 2024 Memo that BNL calculations should exclude Commercial Harvests.
2. Provide NTI and the other co-management partners with the opportunity to respond in writing to the NWMB's views and to each other's views, and then reconsider the matter based on all submissions.
3. Provide written reasons for the reconsideration decision.

NTI looks forward to your response on this matter.

Sincerely,



Gabriel Nirlungayuk
Director, Wildlife and Environment Department
Nunavut Tunngavik Inc.

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Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐃᓄᐃᑦ ᐅᒪᔨᓂᐅ ᐱᐅᖏᓪᓗᑭᕐᓂᓂᓂᐅ ᐅᒪᔨᓕᓕᒫᔨᖏᑦᑐᓂᓂᒃ ᐊᑐᑦᓗᑎᒃ ᐊᒻᒪ ᖃᐅᔨᓴᖏᑎᒪᓕᕆᑦ ᖃᐅᔨᓕᓕᒫᔨᖏᑦᑐᓂᓂᒃ ᐅᒪᔨᓕᓕᒫᔨᖏᑦᖃᑦᓗᑎᒃ, ᐅᓪᓗᒥ ᐃᓄᐃᑦᑯᑎᐅᔪᑦ ᑭᖑᓪᓕᐊᔪᓪᓚᑦᓗ ᐊᖏᓂᔪᓄᑦᓗᓪᓚᑦᓗ ᐃᓄᐃᑦᑕᕐᓂᐊᕐᓗᒍ ᓄᓇᕗᒻᒥ.

Managing wildlife, guided by conservation principles, honouring the key role of Inuit in harvesting, for the well-being of current and future generations in Nunavut.

By Email

April 28, 2025

Gabriel Nirlungayuk
Director, Wildlife and Environment Department
Nunavut Tunngavik Inc.

Dear Mr. Nirlungayuk:

Re: BNL for Southampton Island Caribou

I am writing on behalf of the Nunavut Wildlife Management Board (Board) in response to your letter dated April 7, 2025, regarding the Basic Needs Level for caribou on Southampton Island.

The Board's July 2024 Memo followed a comprehensive internal review of the information, evidence, arguments, and perspectives submitted by co-management partners and other parties—regarding what types of harvests are to be included in the calculation of Basic Needs Levels—over the ten years starting from the conclusion of the Nunavut Wildlife Harvest Study ("NWHS") in 2004 and concluding with the adjournment of the Southampton Island caribou public hearing in 2014.

Based on this review, the Board accepted as "correct", and made the decision to rely on, the NWHS Steering Committee's interpretation of the *Nunavut Agreement*, which recommended the exclusion of commercial harvests from consideration in the establishment of a Basic Needs Level (BNL).

The NWHS Steering Committee determined that the "primary application of the Harvest Study results was for the calculation of the basic needs levels" and concluded that:

> the context of the Agreement does indicate that the basic needs level is that amount of harvest which is currently taken for domestic and cultural purposes. …. (sic) Harvesting or gathering for the purpose of obtaining skins, feathers, dog food, craft materials, etc. would all be acknowledged "basic needs" components as long as the end use occurred in the hunter's community or entered into inter-settlement trade. Apparently, not eligible for consideration as an item of "basic need" would be material harvested or gathered for commercial sale as food outside the Nunavut Settlement Area.

The Board's decision to defer to the expertise of the members of the Steering Committee is further supported by the fact that the membership of the Steering Committee was composed of respected and knowledgeable representatives appointed by all three Regional Wildlife Organizations and three government departments. These representatives had ample occasion to voice the positions of their respective organizations at the time the Nunavut Harvest Study was being conducted. Their recommendation reflects their understanding of the intent of these provisions of the *Nunavut Agreement*.

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Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐃᓄᐃᑦ ᐅᒪᔨᓂᐅ ᐱᐅᖏᓱᐊᕐᓂᖅᓂᓂᐅ ᐅᒪᔨᓕᕆᓚᐅᔨᓂᖅᓂᓂᓗ ᐊᑐᑦᓗᑎᒃ ᐊᓚᒪ ᖃᐅᔨᔪᖏᑎᒪᓕᕆᑦ ᖃᐅᔨᒪᓕᕆᓚᐅᔨᓂᖅᓂᓂᐅ ᐅᒪᔨᓕᕆᓚᐅᔨᓂᖏᑦᓗᑎᒃ, ᐅᓪᓗ ᐃᓄᐃᑦᑐᖅᑕ ᑭᖑᓪᓕᖏᓪᓗ ᐊᖏᔪᓄᔪᓐᓇᕐᓂᖏᑦᓗᑎᑦ ᓄᓇᕗᒻᒥ.

Managing wildlife, guided by conservation principles, honouring the key role of Inuit in harvesting, for the well-being of current and future generations in Nunavut.

We trust this clarification addresses your concerns regarding the Board's process and methods that led to the issuance of the July 2024 Memo.

As required under Section 5.6.24 of the Nunavut Agreement, Board staff will consult with the Hunters and Trappers Organization in Coral Harbour to confirm whether the community kept records regarding the levels of non-commercial harvesting activities by Inuit in the five years before the TAH was established.

This meeting is currently scheduled for 6:30 pm local time (EST) on May 07, 2025, at the Hamlet Chambers in Coral Harbour. We welcome Nunavut Tunngavik Inc.'s participation in person or through a link for virtual attendance upon request.

Sincerely,

Jason Akearok
NWMB Executive Director

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ᒪᐃ 7, 2025

ᑲᑎᒪᖏᑎᖏᓂᖅ ᐊᐃᐃᓴᑦ ᐊᖏᔪᓇᓱᑎᑦ ᐊᓚᒪᓗ ᒥᑭᒋᐊᖅᓂᐊᖅᑎᑦ ᑎᒥᖏᑐᓂᖅ
ᐃᖏᓕᒪᐅᓱᓇᖅᓗᔪᒍ ᐋᖅᑭᑕᐅᓂᖏᑦ ᐊᓪᓇᐅᒪᓴᕐᕕᑕᒋᑦ ᖃᓄᐃᖅᓗᐊᖅᓂᖏᑦ
ᓴᖅᓂᑦ ᖁᑭᖅᑕᓗᐊᓂ ᑐᐅᑐᐃᖅ
ᓴᖅᓂᖅ, ᓄᓇᕗᑦ

<u>ᐅᐸᑦᓭᒪᔪᑦ</u>

<u>ᐊᐃᐃᓴᑦ ᐊᖏᔪᓇᓱᑎᑦ ᐊᓚᒪᓗ ᒥᑭᒋᐊᖅᓂᐊᖅᑎᑦ ᑎᒥᖏᑦ</u>
ᑭᓯ <ᓂᕐᔪᐃ: ᐃᑦᓭᔪᕆᐅᑕᖅ
ᒍᓯᓯ ᓇᔨᓭᖅ, ᑐᖏᓕᓪᓚ ᐃᑦᓭᔪᕆᐅᑕᖅ
ᓄᐊᖅᒪᓵ ᓇᔨᓭᖅ
<ᑦᐸᐱ ᓴᕕᐊᖅᔪᐃ
ᐃᓗ ᐃᒥᖅᑕᐅᑦ
ᓄᐊ ᓇᔨᓭᖅ: ᐊᓵᓕᑦᑎ

<u>ᐃᐅᒪᔪᓕᕆᖅᔪᐊᑦ</u>
ᓴᐃᓵᓵ ᐊᖏᐊᑭᖅ
ᑎᐊᓂᕐ ᓵᑎᐊᓗ

<u>ᐅᓕᔪᓕᕆᓕᕆᔨᖅᐊᐊᑦ</u>
ᓴᐃᓴᓵ ᐊᖏᐊᑭᖅ
ᑎᐊᓂᕐ ᓵᑎᐊᓗ

<u>ᑭᓕᓴᐅᓴᐅᔨᔪᑦ ᑕᔨᓵᓇᖅᑎᑦᓵᓂᐅᔨᔪᑦ ᐅᐸᑦᓭᒪᔪᑦ</u>
ᔪᐄᐃᑦ ᑎᒪ (ᐃᐅᒪᔪᓕᕆᓕᕆᔨᖅᐊᐊᓂ ᒪᓕᑐᓕᕆᓕᔪ)
ᐸᒋᓪᓕ ᑲᓕᕐᕕᐅᖅ (ᑐᓴᔪᓕ)
ᐃᓴᓕᓕᕆᓕᕆᔨᖅᐊᐊᑦ ᑕᐃᒪᓐᓇᖅᑐᑦ ᐊᐃᑦᑐᓕᕆᓕᕆᔨᖅᐊᐊ

ᐃᓄᐃᑦ ᓯᒪᓕᕐᓂᐊᕐᓂᖅ ᐊᑐᐃᓐᓇᖅᑕᐊᓕᓪᓕᑦ ᐅᒪᔨᖏᓪᓗ ᐊᑕᐅᓪᑐᑐᓂᓪᓗ ᑲᑎᑦᑐᑦ ᐊᔨᖓᑦᑐᑦ ᐊᓂᒪᓗ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓪᓚᐅᖅ ᐊᓂᒪᓗ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ., ᓴᐃᓴᓐ ᐊᖏᐊᑭᖅ ᐅᖅᐅᐅᓯᖅᓯᐅᐅᓪᓗᖅ ᐃᓕᖓᓐᓂᓪ ᖃᓄᐃᓯᒪᓂᖏᓂᓪ ᐱᓕᕕᑦᓄᓇᕐᑎ ᐅᒪᔨᖏᓪᓗ ᐊᖏᓚᒋᓪᓗ (ᐊᖏᓚᒋᓂᓪᓗ ᖃᐅᔨᓴᖏᓂᓪᓗᖅ) ᐊᓂᒪᓗ ᑲᒪᒋᖏᖓᑦ ᓄᓇᕗᑦ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᖃᓄᐃᓐᓇᓯᐊᕐᑐᑦᑕᑦ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᖃᓄᐃᓕᕕᑦᓂᐊᓂᖓᓂᑦ.

ᒧᐱᓗ ᓇᓀᓪᖅ ᐊᐱᓚᓪᐅᑦᓴᖅ ᖃᓄᖅ ᐊᓯᔨᐅᑎᑦ ᐅᒪᔨᖏᓄᑦ ᐊᓪᓚᑲᖏᑦᑕᐅᑎᓴᖏᓐᓂᑕ ᐅᓭᓂ ᐊᐅᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ. ᓴᐃᓴᓐ ᐊᖏᐊᑭᖅ ᐅᖅᐅᐅᓯᖅᓯᐅᐅᓪᓗᖅ ᐃᓕᑦ ᐅᑎᓂᑐᑦ ᐊᔨᖓᑦᑐᑦ ᐊᓂᒪᓗ ᐊᖏᓚᒋᓂᓪᓗ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᐱᓕᕕᑦᓂᐊᓂᖓᓂᑦ, ᐊᐅᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᐱᓚᖅᑕᐅᓪᓚᐅᖅᓯᐅᐅᓪᓗᖅ ᐃᓕᑦ ᐃᑐᑎᑎᓪᓚᐅᖅ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᐃᓱᐱᓚᓪᐅᑦᓴᖅᓯᐅᐅᓪᓗᖅ ᐃᓕᑦ ᐃᓄᐃᓕᕕᑦᓂᐊᓂᖓᓂᑦ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᐱᓕᕕᑦᓂᐊᓂᖓᓂᑦ ᐃᓕᑦ ᐃᓄᐃᓕᕕᑦᓂᐊᓂᖓᓂᑦ ᐊᓐᓇᐅᒪᓕᕆᔨᑦᑕᐅᓕᐊᓕᓪᓕᑦ ᐱ

ᒍᓱᓗ ᓀᑯᓪᓕᖅ ᐅᖃᓕᐅᔨᑐᖅ ᐃᒧᑦ ᒥᑦᕆ ᑲᒡᓴᓪ, ᐊᑯᓂᐅᔪᒥᑦ ᓄᓀᐅᓪ ᓯᖏᒪᓐᑯᓂᓂ ᐅᒪᔪᓕᓚᑕᓗ ᐱᓕᓚᐊᓕᓐᑦ ᑐᑐᓂᑦ ᑕᒧᓐᓂᑦ, ᐱᑕᖃᑐᐃᓐᓀᓚᐊᖃᖏᓪᒡᑦ ᐊᑐᔨᑐᑐᓂᑦ ᓀᔭᐅᑎᓂᑦ ᐃᓂ 2008 ᐅᓄᖅᓴᓗ 2012. ᑎᐊᓂᕆ ᓇᑎᐊᓗ ᑲᒪᓂᐊᕐᓂᕋᔨᐅᔨᑐᖅ ᖃᐅᔨᔪᒥᐊᖅᐱᒋᓗᐊᔪ ᒥᑦᕆ ᑲᒪᕆᑦ ᐅᔨᔨᑐᔨᖅᓯᓗᓂᓂᑦ ᐃᒧᑦ ᐅᒪᔪᓀᓂᑦᓴᓂᑦ ᖃᐅᔨᓴᓂᖅᓱᒡᒐᓂᑦ ᑎᑎᕐᓂᕐᒪᒪᓂᑦ ᑎᑎᕐᓂᑕᐅᒪᔪᓂᑦ ᐊᖏᔪᑕᑦ. ᐊᑐᐃᓐᓇᐅᐱᒋᓕᐅᔨᓴᐸᐃᑦ ᐊᖏᔪᓂᑦᓴᑦ ᑎᑎᕐᓂᑕᐅᒪᔪᓂᑦ ᐊᑐᐃᓐᓇᐅᓕᖅᑐᑦ ᑎᑎᕐᓂᑕᐅᒪᔪᓂᑦ ᐊᑐᐃᓐᓇᐅᔨᓕᓐᓂᒃᑯᓂᑦ ᐊᑐᐃᓐᓇᐅᓕᖅᑐ

By Email

Jason Akearok, Executive Director
Nunavut Wildlife Management Board

July 23, 2025

Dear Mr. Akearok:

Re: BNL for Southampton Island Caribou

This letter follows up on NTI's letter of April 7, 2025 ("April Letter") about the Nunavut Wildlife Management Board's ("NWMB") plan to set the Basic Needs Level ("BNL") for the Southampton Island caribou herd ("SHI Caribou Herd"). NTI has not yet received a response to the April Letter, but at its June 25, 2025 regular meeting, the NWMB confirmed its plan to set that BNL during its October regular meeting.

As noted in NTI's April Letter, the NWMB has departed from its previous view that Inuit commercial harvests count towards the BNL. The NWMB's approach to calculating the BNL for the SHI Caribou Herd will set a major precedent. Thus, NTI continues to urge the NWMB to be transparent about its proposed approach, including why it has changed so significantly. NTI also reiterates its request that the NWMB hold a public hearing to guide its BNL decision. Such a major decision deserves to be informed by a hearing. Moreover, NWMB never resumed the hearing it adjourned on December 11, 2013, despite repeatedly communicating its intention to do so.

Transparency re NWMB's Proposed Approach

NTI continues to seek clarity on the questions we asked in the April Letter, as well as on a few additional issues:

1. Does the NWMB plan to exclude from its BNL calculation only the harvests sold outside of Nunavut, or *all* commercial harvests?

    - The July 18, 2024 NWMB policy memo, *Re: Establishment of Basic Needs Level within the Nunavut Settlement Area*, ("July 2024 Memo"), states that the NWMB will use the Nunavut Wildlife Harvest Study data and thereby exclude from BNL calculations "wildlife harvested in bulk for sale outside of the Nunavut Settlement Area" (p.4.).
    - However, the NWMB staff memo of January 7, 2025, *Determination of the Basic Needs Level (BNL) for Southampton Island Caribou*, states at p. 2 that the BNL calculation will "exclude any commercial harvesting records that may be available".

2. Why has the NWMB abandoned its original interpretation of Article 5, supported by the legal opinion it secured from Mr. Meehan in 2009, that BNL calculations should be based on ***all*** Inuit harvests? The July 2024 Memo does not provide a clear answer.

3. Why does the NWMB intend to treat 2008-2012 as the five pre-TAH years under s. 5.6.23(b) of the *Nunavut Agreement* (NWMB April 7, 2025 letter to the Aiviit HTO at p. 1)? Given that there was an interim TAH on the SHI Caribou Herd in 2012, which presumably limited Inuit harvesting in that year, why would the NWMB not use the harvest data from 2007-2011 instead?

4. Why has the NWMB only asked the Coral Harbour for its SHI Caribou Herd harvest records, and not Rankin Inlet, Naujaat, and Chesterfield Inlet? Although the harvests by members of those communities might be relatively small, they are relevant to setting the herd's BNL.

Opportunity for Co-Management Partners to Share Views

NTI would welcome an in-person hearing to inform the NWMB's decision on the SHI Caribou Herd BNL. In addition to a hearing, or in the alternative, NTI proposes that the NWMB hold a meeting to discuss the general approach to BNL calculations with Nunavut's co-management partners: NTI, the Government of Nunavut, and the Department of Fisheries and Oceans (which is involved in setting the Kingnait Fiord char BNL).

Conclusion

The SHI Caribou Herd BNL raises major questions about the interpretation of Article 5 that have essentially been dormant for years. Transparency by the NWMB with NTI and Nunavut's other co-management partners ahead of decision-making is critical. There is also real value in providing a forum for the co-management partners to share evidence, context, and interpretations of the relevant Article 5 provisions to help inform the NWMB's decision. A shared interpretation of the Article 5 provisions may still be achievable, thereby avoiding an unnecessary legal dispute, and in any event, dialogue between the co-management partners promotes collaboration and strong working relationships and, by extension, effective wildlife management.

NTI looks forward to your prompt response on this matter.

Sincerely,

Gabriel Nirlungayuk
Director, Wildlife and Environment Department
Nunavut Tunngavik Inc.

cc. Amy Amos, Regional Director General – Arctic Region

amy.amos@dfo-mpo.gc.ca

Drikus Gissing, Director Wildlife and Environment
dgissing@gov.nu.ca

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Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

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Havarhait: Munaritjutikhaitnik angutirhat, hivuniktaulutik tahapkunanga angutirhaliqiniqut pimariuyunik ihumagiyaulluaktaitnik, pitiaqattiriqnahuaklutik Inuinait attuqparhimayaitnik angutirhanik atturhimaaklugit, innutiaqutirhait ihumagilugit tahapkua ublumi angunahuaktiuyut innuqutaitlu tatvalu kinguvarhavut tahamani Nunavumi.

Managing wildlife, guided by conservation principles, honouring the key role of Inuit in harvesting, for the well-being of current and future generations in Nunavut.

By Email

August 21, 2025

Gabriel Nirlungayuk
Director, Wildlife and Environment Department
Nunavut Tunngavik Inc.

Re: NTI Letter Dated July 23, 2025 – Basic Needs Level for Southampton Island Caribou

Dear Mr. Nirlungayuk,

Thank you for your letter of July 23, 2025, concerning the NWMB's process toward the establishment of the Basic Needs Level for the Southampton Island caribou herd. We welcome the opportunity to clarify our position and respond to your specific questions.

Response to Nunavut Tunngavik Inc.'s Questions

1. **Commercial harvests included in the Harvest Study**
   As communicated in the July 18, 2024, NWMB policy memo, the NWMB will rely on the interpretation of the Nunavut Wildlife Harvest Study Steering Committee that:

   "basic needs refers to the amount of harvest which is currently taken for domestic and cultural purposes. …. Harvesting or gathering for the purpose of obtaining skins, feathers, dog food, craft materials etc. would all be acknowledged "basic needs" components if the end use occurred in the hunter's community or entered into inter-settlement trade. Apparently, not eligible for consideration as an item of "basic need", would be material harvested or gathered for commercial sale as food outside the NSA."

   While there may have been confusion in the wording of the NWMB staff memo, I confirm that all data retained in the Harvest Study database—such as fur-bearer harvests and records indicating local commercial sale (e.g., to Hunters and Trappers Organizations or local stores)—remain included in the Basic Needs Level calculations.

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(888) 421-9832

2. **Has the NWMB abandoned its original interpretation of what constitutes a Basic Needs Level.**

   The NWMB has not abandoned its deliberative process on the inclusions in Basic Needs Level calculations and the current process is a continuation of the NWMB work to implement the Basic Needs Level.

   It is important to note that, as no final decision was ever reached through confirmation by the Minister, any preliminary views expressed by the NWMB or any other party—including the Board's 2009 position on the Meehan opinion—were part of the deliberative process and should not be considered to be definitive conclusions of the Board.

   With respect to the Meehan brief, I would like to state that while the Board may from time to time seek additional information, expertise, or arguments that may help it decide a matter, the Board is not bound by any such positions in the exercise of its mandate as the main instrument of wildlife management within the Nunavut Settlement Area.

   The NWMB July 18, 2024, policy memo followed a careful internal review that considered the positions of all the parties, including past NWMB processes and legal opinions. Through this process, the Board determined that moving forward, it will rely on Steering Committee's interpretation of what should be included in the Basic Needs Level.

3. **Five years prior to the imposition of a Total Allowable Harvest**

   I can confirm that Nunavut Tunngavik Inc. is correct that the interim Total Allowable Harvest imposed in 2012 constrained harvest levels that year. Accordingly, the NWMB accepts that the proper reference period under section 5.6.23(b) of the Nunavut Agreement should be 2007–2011, rather than the 2008–2012 period referenced.

4. **Records from other Kivalliq communities**

   Based on NWMB interpretation of S.5.6.19, the Basic Needs Level follows the establishment of a Total Allowable Harvest. In the case of Southampton Island Caribou, a community Total Allowable Harvest for Coral Harbour is the basis for striking the Basic Needs Level.

   While the NWMB is committed to ensuring that all relevant co-management partners are included in these discussions openly and transparently, the only Nunavut community affected by the Southampton Island Caribou Total Allowable Harvest for which the NWMB intends to strike the Basic Needs Level is Coral Harbour.

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Nunavut Wildlife Management Board

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Havarhait: Munaritjutikhaitnik angutirhat, hivuniktaulutik tahapkunanga angutirhaliqiniqut pimariuyunik ihumagiyaulluaktaitnik, pitiaqattiriqnahuaklutik Inuinait attuqparhimayaitnik angutirhanik atturhimaaklugit, innutiaqutirhait ihumagilugit tahapkua ublumi angunahuaktiuyut innuqutaitlu tatvalu kinguvarhavut tahamani Nunavumi.

Managing wildlife, guided by conservation principles, honouring the key role of Inuit in harvesting, for the well-being of current and future generations in Nunavut.

NWMB staff plan to continue public education about the Basic Needs Level and its place in the Nunavut Wildlife Management System, including presentations delivered at Regional Wildlife Board annual general meetings.

I trust that the above addresses your questions.

The NWMB values its collaborative relationship with Nunavut Tunngavik Inc. and all co-management partners and I look forward to continuing our collaboration and work to ensure effective wildlife management.

Sincerely,

Jason Akearok
NWMB Executive Director

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Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

ᒪᐃ 30, 2012

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ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ ᒪᓕᒐᓱᐃᓐᓇᖅᑐᑦ

ᑐᑭᓯᐅᑦ ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᓂᒃ

ᐅᕐᓄ ᐃᑦᓗᓚ 30, 2012 ᑎᑎᕋᒃᓂ ᐅᑐᖓ ᓄᓇᕗᑦ ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᑦ (ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᐅᑦᕐᓵᓐᓂᑦ ᑲᑎᒪᔨᖏᑦ), ᑐᑭᓯᓚᐅᑦᓴᐃᑦ ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᑳᕈᑦᓴᓕᓚᑦ ᐅᓂᒃᓴᓐᓂ ᑲᑎᒪᔨᖏᑦ ᐊᓕᓐᓂᑦ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ (ᐊᖏᓇᓕᓴᒪᓕᐅᔪᓕᖅᑐᑦ) ᐊᓚᓗ ᒪᓕᒃᓂᓂᒃ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ ᒪᓕᒐᓱᐃᓐᓂᒃ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᓂᒃ ᓯᑭᓯᐅᑦ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᓂᒃ, "...ᐃᓱᒪᓂᕆᔪᓚᑦ ᐊᐅᓚᕐᔪᓂᕆᔨᒃᑯᑦ ᐱᒋᐊᕐᓇᓂᒃ ᑲᒪᕆᔭᐅᓂᐊᕐᓂᑦ ᓯᕗᓕᔪᓂ ᐱᒋᐊᓂᖓᓐᓂᑦ ᑭᖑᓪᓕᕐᒥᒃ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ, ᐱᒋᐊᓕᕐᓂᑲᓐ ᔫᓚᐃ *01, 2012*." ᑐᑭᓯᐅᑦ ᐃᓚᐃᓵᑐᓕᕐᓂᑲᓐ ᒥᓂᑦᑕᐅᑦ ᐊᐅᓚᕐᔪᓂᕆᔨᒃᑯᑦ ᐱᒋᐊᕐᓇᖅᑐᓂ, ᑐᓂᔭᓐᓂᑲᓐ ᒪᓕᒐᓱᐃᓐᓇᖅᑐᑦ 5.3.25 ᐅᑯᐊ ᓄᓇᕗᑦ ᓄᓇᕋᓂᔪᓂᕆᔨᒃᑯᑦ ᐊᖑᓐᓄᑦᓴᓴᑦ (ᓄᓇᕋᓂᔪᓴᓴᑦ).

ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᐊᑐᓕᕐᒪᔭᖓᓂᒃ

ᑕᐃᒪᓐᒃ ᓇᓗᓇᐃᕐᑕᐅᓪᓚᖓᓕᒃ ᐊᓪᓚᕐᓂᓐ, ᐅᖃᐅᓇᓇᕐᓂᐅᒃ ᑲᑎᒪᔨᓕᒃ ᐱᔭᕆᐊᖃᓇᔭᖑᖓᓚᒃᑕ ᓄᓇᕋᓂᔪᓴᓴᑦ ᑎᑎᕋᕐᓚᓂᖓᓐᓄᑦ 5 ᐃᓕᓴᕐᒥᐅᑎᖓᓐᒥᒃ-ᑳᕈᑲᓂᕆᔨᒃ ᐱᓪᓚᕆᐊᓕᕆᔨᓕᑐᓂᒃ ᐃᓗᐊᓇ ᕈᓚᕐᑎᐅᔪᓐᓇ ᐱᕙᓪᓕᐅᑎᑕᐅᓪᓚᑦ. ᐸᓪᓇᐊᓇ, ᑕᐃᒪᓐᒃ ᐅᖃᓕᒫᓂᐅᓚᐅᖏᓚᒃᑕ ᑕᐃᑲᓂ ᒪᓕᒃ - ᐃᓱᒪᓂ 2012 ᑎᑎᕋᖅᑭᑲᓂ ᐃᓅᓴᐃᑦ ᐊᓚᓗᓗ ᐃᓕᓴᕐᓯᐅᑕᐅᑦ ᐅᕐᓄ ᐊᐃᓴᐃᑦ ᐊᖏᓇᓕᓐᓂᑦ ᐊᓚᓗᓗ ᒥᑭᓪᓕᐊᓂᓴᓂᑦ ᑎᑎᕋᖅᑭᑦ (ᐊᖏᓇᓕᓴᒪᓕᐅᔪᓕᖅᑐᑦ), ᐊᐅᓚᕐᔪᑎᑕᐅᓴᑎᑦ ᐅᑲᓇ ᐃᓚᖓ 150(1) ᐅᑯᐊ ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᓯᓚᑦᓴᐊᕐᓂᑦ (ᐊᓚᓗᓗ ᐅᑲᓇ ᓄᓇᕋᓂᔪᓴᓴᑦ ᐃᓚᖓ 5.3.24) ᐋᖅᑭᓂᒃ ᐊᓚᓗᓗ ᐊᑐᕐᑎᐅᒃᑲᐃᓂᒃ ᖃᓄᐃᓇᓂᖅᑭᒃᑲᓐᓂᑦ ᐊᓯᔾᔨᓂᖅᓯᒪᑦ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᒥᒃ ᑐᓴᕐᓂᑲᓐᓂᑦ ᐊᓚᓗᓗ ᐊᓂᐅᓂᖅᓱᑎᒃ ᐊᐅᓚᕐᔪᑎᑕᐅᓴᑎᑦ. ᑭᓇᐅᔭᓕ, ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᖃᓄᐃᓇᓂᖅᑭᒃᑲᓐᓂᑦ ᑲᑎᒪᔨᖏᑦ ᐊᓚᓗᓗ ᐊᑐᕐᑎᐅᒃᑲᐃᓂᒃ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᒥᒃ ᐱᓪᓚᕆᐊᓕᕆᔨᓕᑐᓂᒃ ᑐᑭᓯᐅᑦ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᒥᒃ ᐱᓪᓚᕆᐊᓕᕆᔨᓕᑐᓂᒃ ᐊᓯᔾᔨᓴᓂᐊᕐᓂᑦ ᑭᖑᓪᓕᕐᒥᒃ (ᓄᓇᕋᓂᔪᓴᓴᑦ ᐃᓚᖓ 5.3.24 ᐊᓚᓗᓗ ᐃᓚᖓ 158 ᐅᑯᐊ ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ).

ᑕᐃᒪᓐᓇᖅᑐᓚᐅᔪᓕ, ᑲᑎᒪᔨᓂᒃ ᐊᑐᕐᑎᐅᔪᓐᓂᑦ ᐃᓅᓴᐃᑦ ᑲᓪᓚᕐᔪᐃᓚᓐᓂᒃ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ ᓄᓇᕋᓂᔪᓴᓴᑦ ᐃᓚᖓ 5.3.24. ᒪᓐᓇ ᐊᑐᕐᑎᐅᔪᓐᓇᖅ ᑕᒪᐃᓐᓇᓂᑦ ᐃᑲᓪᓚᕐᔪᐃᓕᓐᓂᖅ ᐱᖓᓱᐊᕐᓂᖅ ᐅᑦᕐᓵᓐᓂᑦ ᐅᓪᓗᕆᔨᓂᕆᔨᒃᑯᑦ ᐅᕐᓄ "ᓯᓚᕐᓇᖅᑐᓂᒃ ᓄᓇᕋᓂᔪᓴᓴᑦ ᐅᖃᓕᒫᓂᐅᓚᐅᖏᓚᒃᑕ ᑲᑎᒪᔨᖏᑦ ᐊᐅᓚᕐᔪᑎᑕᐅᓴᑎᑦ ᐱᓪᓚᕆᐊᓕᕆᔨᓕᑐᓂᒃ ᑭᖑᓪᓕᕐᒥᒃ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ", ᐊᐅᓚᕐᔪᑎᑕᐅᓴᑎᑦ ᒪᓐᓇ 16, 2012.[1] ᑲᑎᒪᔨᕐᓇᑎᒥᓂᒃ ᐅᖃᓕᒫᓂᐅᓚᐅᖏᓚᒃᑕ ᑲᑎᒪᓂᕆᑦ ᐃᓕᓐᓂᐅᔪᓚᕆᔨᒃᑯᑦ ᐅᑯᐊ ᓄᓇᕋᓂᔪᓴᓴᑦ.

[1] ᓯᓚᕐᓇᖅᑐᓂᒃ ᓴᓕᓴᕐᓯᓂᐅᓕᒃ ᐅᖃᓕᒫᓂᐅᓚᐅᖏᓚᒃᑕ ᑲᑎᒪᓂᖅ ᐅᕐᓇᐃᐅᓕᓐᓂᑦ ᑭᓱᓕᕐᓂᒃ ᐅᑯᐊ ᐊᐃᓴᐃᑦ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ, ᓄᓇᕗᑦ ᑐᓴᕐᓱᓐᓄᑦ ᑎᑎᕋᖅᑭᑦ, ᑲᕐᒪᓐᓂᑦ ᓄᓇᕗᑦ ᐱᓪᓚᕆᐊᓕᕆᔨᓕᑐᓂᒃ ᑎᑎᕋᖅᑭᑦ, ᐊᓚᓗᓗ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᓂᑦ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ. ᐊᖑᓐᓄᑦ ᐊᓚᓗᓗ ᐊᐃᓴᐃᑦ ᐊᖏᓇᓕᓐᓂᑦᓯᒪᑦ ᐃᓕᓴᕐᒥᐅᑎᖓᓐᓂᑦᓯᒪᑦ ᐃᓕᓐᓂᐅᔪᓚᕆᔨᒃᑯᑦᓯᓚᓚᐅᑦ.

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ᑲᒪᒋᐅᖅᓯᓗᓂ ᐱᕈᖅᓴᐅᒋᐊᖅᑐᑦ ᑎᑎᕋᐅᓚᖅᑐᑦ 5 ᐃᓕᓴᕐᕕᒋᔭᖅᑎᑦ ᐊᑭᑦᓱᐃᓗᓂᓂ

ᐃᓚᐃᓐᓂᐊᖅᑦᑕᑦ ᕿᑯᐊ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐊᒪᓚᓗ ᒥᓂᑦ ᐱᕐᓴᖅᓯᓗᑎᒃ ᐊᐅᓪᓚᕐᔪᐅᖅᑲᑎᒪᓐᓂᖅᓂᒍᑦ ᓄᓇᖅᓯᑎᑕᒋᓴᑦ ᑎᑎᕋᐅᓚᖅᑐᒍᓐᓇᒍᑦ 5 ᐃᓕᓴᕐᕕᒋᔭᖅᑎᑦ ᐊᑭᑦᓱᐃᓗᓂᑎᒃ ᑲᒪᒋᐅᖅᓯᓗᓂ ᐊᑐᖅᑕᕐᕕᐊᓂᖅᓴᖅᓯᒪᑦ ᐊᒪᓚᓗ ᐱᕐᓴᖅᑕᕐᕕᓗᑎᒃ:

1. ᐱᓗᑎᒃ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐃᓚᐃᑦᑐᒥᒃ *ᐃᓕᓴᔪᑕᐅᓕᖅᖢᖅ ᐃᓕᓴᕐᕕᒋᔭᖅᑎᑦ* ᐊᒪᓚᓗ ᐃᓕᐅᕕᓗᑎᒃ ᓇᓚᒪᓂᖅᓯᑦ ᐊᑐᐃᓐᓇᐃᑦ ᑐᓴᖅᓴᓴᖅ, ᐃᓕᖅᖢᓂᕐᒥᑦ ᓇᓚᐃᐃᑉᐳᒪᓕᕆᓂᒃ ᓇᐃᓇᓴᑎᓂᒃ ᐊᑐᖅᑐᓄᑦ ᑐᓴᖅᓴᖅᓯᓂᖅᓴᖅᑎᒃ ᖃᓄᖅᑲᓇᑕᒥ ᐃᓐᓇᐃᓐᓇᖅᑐᓂ ᐅᓪᓗᕆᓂᕐᒧᑦ ᐃᓐᓇᐃᓐᓇᖅᑐᓂ ᐱᓕᕆᔨᒃᑯᑦ ᐃᓚᖅᑎᓄᑦ, ᓇᓂᓕᖅᓂᐊᐃᓐᓇᖅ;

2. ᐊᓕᖅᐳᐃᓗᓂᒃ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐊᑐᖅᓂᓕᒥᒃ ᑭᖅᑯᓕᒍᓂᒃ ᓇᓕᕐᑎᕐᓂᖅᓯᓗᑦ ᑐᓴᖅᑎᕐᓂᖅᓯᒪᓕᒥᒃ (ᐃᓕᐃᒪᓕᕐᑐᖅ, 60 ᐅᓪᓗᐃᑦ) ᐃᓕᐅᓗᓂᒃ ᐊᑐᐃᓐᓇᐅᔨᑎᓂ ᑭᖅᑯᐃᓐᓇᐃᑦ ᐱᖓᑕᐅᓂᖅᑭᓐᓄᑦ;

3. ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐅᖓᐅᓯᖓᓴᓗᓂᒃ ᐃᓱᓐᓇᖅᑭᓐᓂᒃ ᓇᓚᒪᓂᖅᓴᓂᒃ ᐊᑐᐃᓐᓇᕐᓂᒃ ᑐᓴᖅᓴᕐᓴᓂᒃ, ᑐᑭᓯᐅᖅᑕᐅᓕᖅᓴᕐᓂᐊᖅᓂᒃ;

4. ᐊᓕᖅᐳᐃᓗᓂᒃ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐊᑐᐃᓐᓇᐅᔨᔪᑎᒃ ᑲᓚᖓᑕᐅᑐᐃᓐᓇᓕᕆᓂᓐᓄᑦ ᑐᓂᓯᓂᒥᒃ ᑎᑎᕋᐅᓚᖅᑐᓂᒃ, ᑐᑭᓯᐅᖅᑕᐅᓕᖅᓴᕐᓂᒃ ᐅᓂᒃᑲᓂᒃ (ᐃᓕᐃᒪᓕᕐᑐᖅ, 39 ᐅᓪᓗᐃᑦ);

5. ᐊᑐᖅᓗᒍ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ (ᐅᖓᐅᓯᖓᓂᑦ ᐅᓴᖅᓱᓐᓂᖅ ᑎᑎᖓᑲᑎᒍᑦ) ᓇᓕᕐᑎᕐᓂᓯᓂᖅᓂᑦ (ᐃᓕᐃᒪᓕᕐᑐᖅ, 21 ᐅᓪᓗᐃᑦ ᑭᖓᑦᖏᓐᓇᒍᑦ ᑐᓂᓯᓂᕐᒪᖏᑕᕐᕕᐅᑦ ᐅᓂᒃᑲᓂᒃ);

6. ᐊᑐᖅᓗᒍ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐃᓕᓴᕐᕕᒋᔭᖅᑎᒃ-ᑲᑭᖅᑭᐃᓂᖅᓯᑦ ᑲᑎᒪᓂᖅᓂᑦ ᑭᖓᑦᖏᓐᓇᒍᑦ ᓇᓕᕐᑎᕐᓂᓯᓂᐅᑦ;

7. ᑐᓂᓯᓗᓂᒃ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐃᓕᓴᕐᕕᒋᔭᖅᑐᑎᒋᓂᒃ ᒥᓂᑦᑕᓗᑦ (ᓄᓇᖅᓯᑎᑕᒋᓴᑦ ᐃᓕᖓᖏᑦ 5.3.8); ᐊᒪᓚᓗ

8. ᐊᓯᓕᒻᒥᑦ ᐱᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐅᖓᑦᑕᐅᓚᕐᓯᑦ ᓄᓇᖅᓯᑎᑕᒋᓴᑦ ᐃᓕᖓᖏᓐᓂᒃ 5.3.9 ᐅᐅᖏᖓᑦ 5.3.15 (ᖃᐅᔨᓚᒋᐊᓕᒃ: ᑕᐃᒪ ᒥᓂᑦ ᐊᖏᓐᖏᑦᑕᑦ ᐱᒋᐊᓐᖓᔨᒥᒃ ᐅᒪᓕᕆᓂᖅᓴᕐᓯᐊᑦ ᐃᓕᓴᕐᕕᒋᔭᖅᑐᑎᖓᓂᒃ, ᑕᓐᓇ ᐱᑕᖃᓕᖅᖃᓐᓇᕐᓂᑲᓐᓂᖅ ᓄᓇᖅᓯᑎᑕᒋᓴᑦᑎᑐᓂᑦ ᐅᓚᐃᑉᐳᖅᓱᖅ ᐃᓕᐃᒪᓕᕐᑐᖅᓱᖅ ᑕᖅᑭᕐᓇᖓᓐᓂᒃ ᐱᖓᑕᐅᓕᖅᓴᕐᓂᐊᖅᓱᖅ).

ᑕᐃᒪᓕ ᕿᓐᓂᕐᓂᒃ ᐱᖁᑦᓂᖅᐅᓪᓚᕐᓯᑦ ᖃᓕᑎᒥᐅᓂᐅᑦᓂᖅᐅᔪᓐᓇᑐᓗᓂᑦ ᐊᔪᐅᖓᑭᑕᓄᒃ ᐊᓕᖅᖃᖅᑎᑦᑕᑦ, ᑐᑭᓯᓇᑕᓯᐊᓂᑦ ᑕᒪᐃᓄᑦ ᐳᓚᕆᓂᖅᑕᓗᑦ ᐱᖁᑦᓂᖅᓴᖅᑭᑦ ᐃᓕᓴᕐᕕᒋᔭᖅᑎᑦ ᐃᓕᓴᕐᕕᒋᔭᖅᑎᒃ-ᑲᑭᖅᑭᐃᓂᖅᓯᑦ ᐊᑐᐃᓐᓇᐅᔨᓂᒃ ᐃᓕᓴᕐᕕᒋᔭᖅᑎᑦ ᐊᑭᑦᓱᐃᓗᓂᓂᒃ

(c) ᐊᐢᑎᓕᓐᓇᓵᓪ ᑭᓄᑐᐃᓐᓇᓴᓪ ᑐᓂᐅᖏᐃᓕᓗᑎᑦ ᖃᐅᐅᓴᓂᑦ ᖃᐅᐅᓵᕐᓂᑦ ᐊᑐᐃᓐᓇᐅᓂᖅᓯᑕ, ᐊᓚᓗ ᐅᖏᓕᒪᐅᓯᓇᓕᓂᑦ ᖃᐅᐅᓴᓂ - ᐊᓚᓗᓂᑎᖏᖅ ᐃᓱᒪᒋᑦᓯ (ᓄᓇᑦᓯ, ᐊᖏᑭᐊᓂᑦ ᐅᑦᓯᓂᓐᓂᑦ ᑲᔪᓯᔪᑦᓯᒧᑦ ᐃᓱᒪᓕᐅᓯᑎᑐᑎᑦ ᐊᓚᓗᓂ ᑲᒪᒋᓯᑦ - ᐃᓂᓐᓇᖅᓐᓄᑦ ᐊᑎᑦᓯᓄᑦ ᐊᖏᔪᓇᖅᑎᓄᑦ ᐊᓚᓗᓂ ᑕᕆᔨᓄ ᐊᖏᔪᓇᖅᑐᑎᓂᓐᓄᑦ;

(d) ᑭᖑᑦᖏᒍᑦ ᑕᐃᒪᒃ ᑐᒃᖁᖅᖄᓯᐊᖅᓂᑦ, ᐊᐢᑎᓕᓐᓇᓵᓪ ᑎᑎᑦᓯᓗᑎᒃ ᑐᑭᓕᐅᑦᑕᐅᒪᔪᓄᒃ ᑐᒃᖁᖅᖄᓯᐊᖅᓯᓪᓚᑦ ᐅᓄᖅᑦ (ᐃᓕᐅᔨᓗᑎᒃ ᐋᑭᑭᐊᑐᑎᓯ, ᐱᑕᖅᓴᓪᓕ, ᐋᑭᑭᑦᓯ ᖃᐅᐅᓴᐅᓪᓕᓗᑎᒃ ᑐᒃᖁᖅᖄᓯᐊᖅᓯᑦᓂᒃ ᑲᒪᒋᔭᖅᑐᓂᖅᓴᓂᒃ); ᐊᓚᓗᓂ

(e) ᐊᐢᑎᓕᓐᓇᓵᓪ ᑎᑎᑦᓯᓗᑎᒃ ᐊᓚᓗᓂ ᑐᓂᓯᓗᑎᒃ ᐅᒪᔪᓕᓐᓇᓵᓪᓴᔪᐊᓄᑦ ᐃᒪᐃᓕᑐᒥᒃ ᐃᓱᒪᓕᐅᑦᑕᐅᒪᔪᓄᒃ ᐊᓚᓗᓂ ᐃᓕᖅᖄᓯᐊᖅᓂᑦ ᓄᓕᓴᓪᓕᓂᑦ ᐊᑐᐃᓐᓇᓴᓂᒃ ᑐᒃᖁᖅᖄᓯᐊᖅᓯᑦᓂᒃ, ᐃᓂᓐᓇᑎᓄᒃ ᑐᒃᖁᖅᖄᓯᐊᖅᓯᓪᓚᑦ ᖃᓪᓗᓈᑎᑐᑦ ᐅᓪᓴᐅᓂᖅᓴᓂᑦ ᐃᓄᖅᑦᑐᓂᑦ ᐃᓄᖅᑦᑐᓂᑦ ᐅᓪᓴᐅᓂᖅᓴᓂᑦ ᐅᓪᓴᐅᓂᖅᓴᓂᑦ ᐅᓪᓴᐅᓂᖅᓴ

ᑕᐃᒪᓕ ᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᓂᓗᐅᓐᖏᓂᑐᑎᓗᐊᑦ ᐊᖏᕈᐊᖅᓂᓄᐊᕆᓚᐊᕐᓂᒃ ᐊᓐᓇᐅᒪᓐᔨᔪᐃᕐᑐᕈᐊᓂᒃ
ᐃᓕᒪᓕᐅᕐᔅᑎᑕᒧᑦ ᐃᓕᓴᕐᓗᐃᓂ ᓄᓇᕗᑦᑎᑦᑕᑦᑎᓂ ᐃᓪᓚᑦ ᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᑭᖑᓂᖅᓯᒪᔪᑦ
ᐃᓕᒪᓕᐅᕐᔅᑎᒥᑦ ᐊᖏᕈᓱᐃᓂᖅᑎᒥᑦ, ᑐᑭᓯᓇᓚᐅᑎᓪᓗᑎᑦ ᐃᓚᖅ ᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᑭᖑᓂᖅᓯᒪᔪᑦ
ᓇᓕᕆᓂᕐᓂᖏᓂᑦ ᐊᖅᓚᓗ ᐃᓕᒪᓕᐅᕐᔅᑎᒥᑦ ᐊᖏᕈᓱᐃᓂᖅᓯᒪᔪᑦ ᐃᓕᓴᖅᓂᐅᓂᖅᓯᒪᓕᒡ ᐊᖏᕈᑕᐅᓗᑎᑦ
ᐊᓐᓇᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᓴᓪᓕᓂ ᖁᑭᒡᓗᐊᓂ ᑐᑦᑐᑦ. ᑕᐃᒪᐃᓐᓂᖓᓂ, ᐃᑲᔪᕐᓇᐊᖅᓯᓗᑎᑦ
ᐊᖏᓂᔨᓗᑦ ᑲᓴᒪᓐᓂ ᓄᓇᕗᒥ ᖃᓄᐃᓕᓐᓂᖏᑦ ᓇᓗᓇᕐᔪᐊᖅᓯᓗᒍ ᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᓇᓂᒥᑦ
ᖁᒥᑦᓱᓂᖓᓂ ᐊᖅᓚᓗ ᒪᓕᓂᑲᓐᓂᕐᓂᑦ ᐃᓕᓐᓂᐊᕐᓂᖓᓐᓂ, ᑲᑎᒪᔨᑦ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ
ᑎᑎᖃᓂᑦ ᐃᓪᓚᑦ ᐊᓕᓴᖃᑎᒌᓐᓂ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ ᑕᓪᓕᐊᓂᖓᓂᒡᓗᑎᑦ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ
ᐊᓐᓇᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᓴᓪᓕᓂ ᐊᓐᓇᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ ᐊᖏᕈᑕᐅᓗᑎᑦ
ᑎᑎᖃᓂᑦ ᐃᓪᓚᑦ ᓴᓪᓕᓂ ᐊᓐᓇᐅᒪᔨᓕᓗᓂᔨᔪᐊᑦ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ ᐊᖏᕈᑕᐅᓗᑎᑦ
ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ ᐊᖏᕈᓱᐃᓂᖅᓯᒪᔪᑦ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ ᐊᖏᕈᑕᐅᓗᑎᑦ ᐃᓕᓴᕐᓂᐅᓂᖅᓯᒪᓕᒡ
ᐊᓐᓇᐅᒪᔨᓕᓗᓂᔨᔪᐊ

WILDLIFE ACT
SI-001-2012
Registered with the Registrar of Regulations
2012-07-12

SOUTHAMPTON ISLAND CARIBOU HERD TOTAL ALLOWABLE HARVEST INTERIM ORDER (July 1, 2012 to June 30, 2013)

Whereas there has been a precipitous decline in the population of the Southampton Island Caribou Herd since 1997, from more than 30,000 to fewer than 8,000 animals;

And Whereas current harvest levels of more than 2,300 caribou per year combined with a 50% reduction in the breeding success rate of the herd since 2000 and a severe outbreak of brucellosis are likely to result in extirpation of the Southampton Island Caribou Herd within three to five years if conservation measures are not taken immediately;

And Whereas the Southampton Island Caribou Herd is an important food, cultural, and economic resource for the community of Coral Harbour;

And Whereas the Coral Harbour HTO, the Nunavut Wildlife Management Board ("NWMB"), and Nunavut Tunngavik Incorporated agree that immediate conservation measures should be taken to maintain the stability and sustainability of the Southampton Island Caribou Herd until it can be determined if additional long-term conservation measures are necessary;

And Whereas the Minister is of the opinion that urgent and unusual circumstances exist requiring an immediate modification in harvesting activities with respect to the Southampton Island Caribou Herd,

The Minister, in accordance with paragraph 121(b) and section 150 of the *Wildlife Act* and every enabling power, makes the annexed *Southampton Island Caribou Herd Total Allowable Harvest Interim Order (July 1, 2012 to June 30, 2013).*

1. In this Order, "Southampton Island Caribou Herd" means all caribou found on Southampton Island and White Island, being part of Wildlife Management Barren-Ground Caribou Area N/BC/12, as described in the *Wildlife Management Barren-Ground Caribou Areas Regulations*, N.W.T.Reg. R-099-98, as duplicated for Nunavut by section 29 of the *Nunavut Act* (Canada).

2. (1) A total allowable harvest is established for the Southampton Island Caribou Herd for the harvest year of July 1, 2012 to June 30, 2013, in accordance with this Order.

(2) The total allowable harvest from the population of the Southampton Island Caribou Herd for the harvest year is 1,000 caribou.

(3) It is presumed that the Inuit of Coral Harbour need the full amount of the total allowable harvest.

(4) The full amount of the total allowable harvest and basic needs level for the harvest year is allocated to the community of Coral Harbour.

3. The Coral Harbour HTO may issue tags for allocating the basic needs level among its members.

4. For the purposes of conservation, the harvest of the Southampton Island Caribou Herd during the harvest year is subject to the following non-quota limitations:

(a) there shall be no harvesting of cow-calf pairs; and
(b) there shall be no harvesting of mature bull caribou.

5. This Order is subject to review as soon as practicable by the NWMB in accordance with section 158 of the Act, but remains in effect until the earlier of:

(a) June 30, 2013;
(b) registration of an Order to implement an accepted decision of the NWMB concerning the Southampton Island Caribou Herd; or
(c) registration of a subsequent Order of the Minister.

6. This Order comes into force on the later of July 1, 2012 and the date of registration by the Registrar of Regulations.

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Nunavunmi ngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐱᓕᕆᑦᑕᐃᓕᓚᓂᖅ ᐅᒪᔨᓂᒃ ᐊᖏᔪᑎᒋᓕᒃ ᐊᑐᕐᑎᑕᐅᓂᖅ ᐃᓄᐃᑦ ᖃᐅᔨᒪᓂᖅᓯᒪᔪᖅ ᐊᓪᓚᓕ ᖃᐅᔨᓴᕐᑎᑦᑎᓐᓂᒃ ᐊᓪᓚᓕ ᖃᐅᔨᒪᓂᖅ
Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

Hand delivered. May. 15/13 TR.

Minister James Arreak
Government of Nunavut
Department of Environment

May 15, 2013

Re.: Southampton Island caribou herd total allowable harvest interim order (July 1, 2012- June 20, 2013) amendment

Dear Minister Arreak,

This letter is in response to the notification of the Government of Nunavut Department of Environment's proposed amendment to the Southampton Island caribou herd total allowable harvest Interim Order for July 1, 2012- June 30, 2013 received by the Nunavut Wildlife Management Board (NWMB or Board) on May 3, 2013.

As noted in correspondence between senior staff of our relevant organizations dated May 3rd, 2013 the Board was unfortunately unable to provide a formal position on the issue by the end of the business day on May 3, 2013, as requested by your Department to meet Cabinet deadlines. In this correspondence, NWMB staff requested that one of the clauses on the proposed amendment be removed as the Board did not receive the results of population studies conducted by the Department of Environment, originally scheduled for June 2012, or an associated request for decision as requested by the Board in its May 30, 2012 decision letter (attached). The NWMB would like to acknowledge the response by the Department in removing this clause as requested before the proposed amendment was submitted to Cabinet for consideration.

The NWMB formally considered the request to extend the Interim Order until June 30, 2014 during Internal Meeting 005-2013, held on May 9, 2013. After careful consideration of the issue the Board passed the following resolution:

> *RESOLVED that the NWMB inform the Minister of Environment that in consideration of the circumstances the NWMB is in agreement that the Total Allowable Harvest of 1000 being in effect until such time as:*
>
> *1) A 2013 population study is completed;*
>
> *2) The results of that population study are distributed;*
>
> *3) A proposal for NWMB decision is received; and*
>
> *4) That the NWMB has made a procedurally fair decision.*

As you are aware, the NWMB is required to conduct a full review of any interim decision made in response to urgent and unusual circumstances as soon as practicable thereafter (as per NLCA S. 5.3.24 and s. 158 of the Wildlife Act). Regarding a total allowable harvest for the Southampton Island caribou herd, that review will be in the form of a public hearing. The wording of the Board's decision on this matter is intended to reflect this obligation. The

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procedural requirements for NLCA Article 5 decision-making are outlined in detail in the Board's attached May 20, 2012 decision letter.

Should you have any questions or concerns, or require any additional information please feel free to contact the NWMB at your convenience.

Sincerely,

M. Audlakiak

Manasie Audlakiak
A/Chairperson
Nunavut Wildlife Management Board

Attachments (1)

WILDLIFE ACT - LOI SUR

Whereas there has been a precipitous decline in the population of the Southampton Island Caribou Herd since 1997, from more than 30,000 to fewer than 7,000 animals;

And Whereas an increased harvest level combined with a 50% reduction in breeding success rate of the herd since 2000 and a severe outbreak of brucellosis are likely to result in extirpation of the Southampton Island Caribou Herd within three to five years if conservation measures are not continued;

And Whereas the Southampton Island Caribou Herd is an important food, cultural and economic resource for the community of Coral Harbour;

And Whereas the NWMB has not made any formal decision concerning the Southampton Island Caribou Herd since conservation measures were undertaken by the Minister;

And Whereas the Minister is of the opinion that urgent and unusual circumstances continue to exist requiring a continuation of the maximum allowable harvest with respect to the Southampton Island Caribou Herd,

The Minister, in accordance with paragraph 121(b) and section 150 of the *Wildlife Act* and ever enabling power, makes the annexed *Southampton Island Caribou Herd Total Allowable Harvest Interim Order (July 1, 2012 to June 30, 2013), amendment*.

Dated,
Fait le ……..…… 2013.

.....
Hon. James Arreak,
Minister of Environment
Ministre de l'Environment

WILDLIFE ACT

SOUTHAMPTON ISLAND CARIBOU HERD TOTAL ALLOWABLE HARVEST INTERIM ORDER (July 1, 2012 to June 30, 2013), amendment

1. The *Southampton Island Caribou Herd Total Allowable Harvest Interim Order (July 1, 2012 to June 30, 2013)*, registered as statutory instrument numbered SI-001-2012, is amended by this Order.

2. The title of the Order is repealed and the following substituted:

SOUTHAMPTON ISLAND CARIBOU HERD TOTAL ALLOWABLE HARVEST INTERIM ORDER (July 1, 2012 to June 30, 2014)

3. Section 2 is amended:

(a) in subsection (1) by striking out "the harvest year of July 1, 2012 to June 30, 2013**" and substituting "**the harvest years of July 1, 2012 to June 30, 2013 and July 1, 2013 to June 30, 2014**"; and**

(b) in subsection (2) by striking out "the harvest year**" and substituting "**each harvest year**".**

4. Section 5 is amended:

(a) in subsection (1) by striking out "2013**" and substituting "**2014**";**

(b) in subsection (2) by striking out "registration**" and substituting "**coming into force**"; and**

(b) in subsection (3) by striking out "registration**" and substituting "**coming into force**".**

REGULATIONS REGISTER
Registered in the Regulations Register
on Jun 16 . 2014 under
registration number R- 019 - 2014
Filed by

WILDLIFE ACT - LOI SUR LA FAUNE ET LA FLORE

Whereas there has been a precipitous decline in the population of the Southampton Island Caribou Herd since 1997, from more than 30,000 to fewer than 7,000 animals;

And Whereas an increased harvest level combined with a 50% reduction in breeding success rate of the herd since 2000 and a severe outbreak of brucellosis are likely to result in extirpation of the Southampton Island Caribou Herd within three to five years if conservation measures are not continued;

And Whereas the Southampton Island Caribou Herd is an important food, cultural, and economic resource for the community of Coral Harbour;

And Whereas the NWMB has recommended that an interim harvest limit of 800 caribou be enforced for the Southampton Island Caribou Herd until a final decision is made by the NWMB;

And Whereas for the purposes of conservation it is necessary to impose a non-quota limitation prohibiting the harvesting of cow-calf pairs from the Southampton Island Caribou Herd;

And Whereas the Minister is of the opinion that urgent and unusual circumstances exist requiring an immediate modification in harvesting activities with respect to the Southampton Island Caribou Herd,

The Minister, in accordance with paragraph 121(b) and section 150 of the *Wildlife Act* and every enabling power, makes the annexed *Southampton Island Caribou Herd Total Allowable Harvest Interim Order (July 1, 2014 to June 30, 2015).*

Attendu :

le brusque déclin de la population de la harde de caribous de l'île Southampton depuis 1997, qui est passée de plus de 30 000 individus à moins de 7 000;

que l'augmentation de la récolte, ainsi qu'une réduction, depuis 2000, de 50% du taux d'efficacité de la reproduction, et l'apparition d'un important foyer de brucellose dans la harde, entraîneront vraisemblablement la disparition de la harde de caribous de l'île Southampton d'ici trois à cinq ans si des mesures de conservation ne se poursuivent pas;

que la harde de caribous de l'île Southampton constitue une importante ressource alimentaire, culturelle et économique pour la collectivité de Coral Harbour;

que le CGRFN a recommandé qu'une limite de récolte provisoire de 800 caribous soit mise en application pour la harde de caribous de l'île Southampton, jusqu'à ce qu'une décision finale soit prise par le CGRFN;

qu'à des fins de conservation, il est nécessaire d'imposer une limite non quantitative interdisant la récolte de couples mères-petits au sein de la harde de caribous de l'île Southampton;

que le ministre est d'avis qu'il existe des circonstances urgentes et exceptionnelles exigeant la modification immédiate des activités de récolte à l'égard de la harde de caribous de l'île Southampton,

le ministre, en vertu de l'alinéa 121b) et de l'article 150 de la *Loi sur la faune et la flore* et de tout pouvoir habilitant, prend l'*Arrêté provisoire concernant la récolte totale autorisée à l'égard de la harde de caribous de l'île Southampton (1er juillet 2014 au 30 juin 2015),* ci-après.

Dated,
Fait le June 13 2014.

Hon. Johnny Mike,
Minister of Environment
Ministre de l'Environnement

WILDLIFE ACT

SOUTHAMPTON ISLAND CARIBOU HERD TOTAL ALLOWABLE HARVEST INTERIM ORDER (July 1, 2014 to June 30, 2015)

1. In this Order, "Southampton Island Caribou Herd" means all caribou found on Southampton Island and White Island, being part of Wildlife Management Barren-Ground Caribou Area N/BC/12, as described in the *Wildlife Management Barren-Ground Caribou Areas Regulations,* N.W.T.Reg. R-099-98, as duplicated for Nunavut by section 29 of the *Nunavut Act* (Canada).

2. (1) A total allowable harvest is established for the Southampton Island Caribou Herd for the harvest year of July 1, 2014 to June 30, 2015, in accordance with this Order.

(2) The total allowable harvest from the population of the Southampton Island Caribou Herd for the harvest year is 800 caribou.

(3) It is presumed that the Inuit of Coral Harbour have a basic need for the full amount of the total allowable harvest set in this Order.

(4) The full amount of the total allowable harvest for the harvest year is allocated to the community of Coral Harbour.

3. There shall be no harvesting of cow-calf pairs from the Southampton Island Caribou Herd during the harvest year.

4. This Order is subject to review as soon as practicable by the NWMB in accordance with section 158 of the Act, but remains in effect until the earlier of:

(a) June 30, 2015;

(b) the coming into force of an Order to implement an accepted decision of the NWMB concerning the Southampton Island Caribou Herd; or

(c) the coming into force of a subsequent Order of the Minister.

5. The *Southampton Island Caribou Herd Total Allowable Harvest Interim Order (July 1, 2012 to June 30, 2014),* registered as statutory instrument numbered SI-001-2012, is repealed.

6. This Order comes into force on the later of July 1, 2014 and the date of registration by the Registrar of Regulations.

LOI SUR LA FAUNE ET LA FLORE

ARRÊTÉ PROVISOIRE CONCERNANT LA RÉCOLTE TOTALE AUTORISÉE À L'ÉGARD DE LA HARDE DE CARIBOUS DE L'ÎLE SOUTHAMPTON (1er juillet 2014 au 30 juin 2015)

1. Pour l'application du présent arrêté, « harde de caribous de l'île Southampton » s'entend de tous les caribous qui se trouvent sur l'île Southampton et l'île White, faisant partie de la région de gestion du caribou des toundras N/BC/12, telle que décrite dans le *Règlement sur les régions de gestion du caribou des toundras*, Règl. T.N.-O. R-099-98, reproduit pour le Nunavut par l'article 29 de la *Loi sur le Nunavut* (Canada).

2. (1) Une récolte totale autorisée est établie pour la harde de caribous de l'île Southampton à l'égard de l'année de récolte s'étendant du 1er juillet 2014 au 30 juin 2015 en conformité avec le présent arrêté.

(2) La récolte totale autorisée au sein de la population de la harde de caribous de l'île Southampton est de 800 caribous pour l'année de récolte.

(3) Il est présumé que l'intégralité de la récolte totale autorisée qui est précisée dans le présent arrêté constitue un besoin essentiel pour les Inuit de Coral Harbour.

(4) L'intégralité de la récolte totale autorisée pour l'année de récolte est attribuée à la collectivité de Coral Harbour.

3. La récolte de couples mères-petits au sein de la harde de caribous de l'île Southampton est interdite pendant l'année de récolte.

4. Le présent arrêté est assujetti, dans les plus brefs délais, à un examen par le CGRFN conformément à l'article 158 de la Loi, mais demeure valide jusqu'à la première des dates suivantes :

a) le 30 juin 2015;
b) la date d'entrée en vigueur d'un arrêté visant la mise en œuvre d'une décision du CGRFN acceptée concernant la harde de caribous de l'île Southampton;
c) la date d'entrée en vigueur d'un arrêté subséquent du ministre.

5. L'*Arrêté provisoire concernant la récolte totale autorisée à l'égard de la harde de caribous de l'île Southampton (1er juillet 2012 au 30 juin 2014)*, enregistré comme texte réglementaire sous le numéro TR-001-2012, est abrogé.

6. Le présent arrêté entre en vigueur le 1er juillet 2014 ou à la date de son enregistrement auprès du registraire des règlements, selon la date la plus tardive.

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Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

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Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

October 16th, 2015

Honourable Johnny Mike
Minister of Environment
Government of Nunavut

Dear Minister Mike:

Re: Request for Harvesting Limitation Decisions from the Nunavut Wildlife Management Board for the Southampton Island Caribou Population

Request for NWMB Decisions

On August 20th, 2015 the Nunavut Wildlife Management Board (NWMB or Board), received a *Proposal for Decision* (Proposal) from the Government of Nunavut-Department of Environment requesting that the NWMB establish a total allowable harvest of 1600 caribou and a non-quota limitation which prohibits the harvest of cow/calf pairs for the Southampton Island caribou herd (Appendix 1).

NWMB Recommendation

During an Internal In-Camera Meeting (INT 014-2015) on September 23rd, 2015 the Board passed the following resolutions:

RESOLVED that the NWMB write to the Minister of Environment, recommending that he promptly make another reasonable Nunavut Land Claims Agreement Section 5.3.24 interim decision regarding harvesting limitations for Southampton Island caribou – with that interim decision to remain in place until such time as the NWMB completes its public hearing and Nunavut Land Claims Agreement decision-making processes with respect to Southampton Island caribou harvesting limitations.

RESOLVED that the NWMB:

(a) Organize a pre-hearing teleconference with the Southampton Island caribou hearing parties, with the objective being to promptly obtain submissions as to whether the NWMB should divide the current Southampton Island caribou public hearing into two hearings: a written hearing addressing the total allowable harvest and accompanying non-quota limitation for Southampton Island caribou, and an in-person hearing addressing the basic needs level for Southampton Island caribou; and

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(b) *In providing notice of the teleconference, the NWMB inform the parties that, if requested, it is prepared to provide additional time for the preparation and delivery of written submissions – but with the caution that the provision of additional time will necessarily result in a delay in NWMB total allowable harvest and non-quota limitation decision-making, from potentially as early as December 2015 (NWMB Regular Meeting (RM) No. 004-2015) to March 2016 (NWMB RM No. 001-2016) or later.*

In December 2013, the Board held a two day in-person hearing in Iqaluit, Nunavut, to address the level of total allowable harvest, non-quota limitations and the striking of a basic needs level for the Southampton Island caribou herd. At the end of two days, the NWMB decided to adjourn the hearing because more time was required to obtain important evidence concerning the basic needs level from co-management partners. After an adjournment of almost two years, it is still unclear how much longer the second phase of the hearing process will take.

In addition, the last interim order establishing a total allowable harvest expired on June 30th, 2015, leaving the community of Coral Harbour without any harvest restrictions in place. Given the conservation concern created by having no harvest restrictions in place and the uncertainty regarding the amount of time needed for the NWMB to complete the *Nunavut Land Claims Agreement* Article 5 decision-making process regarding Southampton Island caribou, **in these particular circumstances, the Board recommends that you proceed by way of *Nunavut Land Claims Agreement* Section 5.3.24 and make another interim decision regarding harvest limitations for Southampton Island caribou.**

To address the second resolution, NWMB staff held a pre-hearing teleconference on October 7th, 2015 to receive submissions from co-management partners as to whether the NWMB should hold a written public hearing to address the total allowable harvest and accompanying non-quota limitation and a separate in-person public hearing to address the basic needs level at a later date. Co-management partners raised no objection to holding two separate hearings. Therefore on October 9th, 2015 during an In-Camera meeting (IC 006-2015), the Board passed the following resolution:

RESOLVED that the NWMB divide the current Southampton Island caribou public hearing into two hearings: a written hearing addressing the total allowable harvest and accompanying non-quota limitation for Southampton Island caribou, and the continuation of the in-person hearing addressing the basic needs level for Southampton Island caribou.

NWMB-proposed Next Steps

In terms of specific next steps, the Board recommends that you make and implement a reasonable interim decision as soon as reasonably possible regarding harvest management for the Southampton Island caribou herd.

The NWMB will initiate a written public hearing to address the Proposal from the Government of Nunavut-Department of Environment regarding a total allowable harvest and associated non-quota limitation for Southampton Island caribou as soon as possible. It is anticipated that the NWMB will consider the submissions received through the written public hearing and make a decision at its December 2015 In-Camera Meeting. This decision will promptly be forwarded to you for consideration.

If you require further information, please do not hesitate to contact the NWMB.

Yours sincerely,

Ben Kovic
Chairperson of the
Nunavut Wildlife Management Board

Attachments (1)

c.c. Drikus Gissing, Director of Wildlife, Government of Nunavut-Department of Environment;

WILDLIFE ACT - LOI SUR LA FAUNE ET LA FLORE

Whereas there has been a precipitous decline in the population of the Southampton Island Caribou Herd since 1997, from more than 30,000 to approximately 12,000 animals;

And Whereas the Southampton Island Caribou Herd is an important food, cultural, and economic resource for the community of Coral Harbour;

And Whereas the NWMB has recommended that a reasonable interim harvest limit be enforced for the Southampton Island Caribou Herd until a final decision is made by the NWMB;

And Whereas the Minister has determined that a harvest limit of 1600 caribou is sustainable until a final decision is made by the NWMB;

And Whereas for the purposes of conservation it is necessary to impose a non-quota limitation prohibiting the harvesting of cow-calf pairs from the Southampton Island Caribou Herd;

And Whereas the Minister is of the opinion that urgent and unusual circumstances exist requiring an immediate modification in harvesting activities with respect to the Southampton Island Caribou Herd,

The Minister, in accordance with paragraph 121(b) and section 150 of the *Wildlife Act* and every enabling power, makes the annexed *Southampton Island Caribou Herd Total Allowable Harvest Interim Order*.

Attendu :

le brusque déclin de la population de la harde de caribous de l'île Southampton depuis 1997, qui est passée de plus de 30 000 individus à environ 12 000;

que la harde de caribous de l'île Southampton constitue une importante ressource alimentaire, culturelle et économique pour la collectivité de Coral Harbour;

que le CGRFN a recommandé qu'une limite de récolte raisonnable et provisoire soit mise en application pour la harde de caribous de l'île Southampton, jusqu'à ce qu'une décision finale soit prise par le CGRFN;

que le ministre a décidé qu'une limite de récolte de 1 600 caribous constitue une limite durable jusqu'à ce qu'une décision finale soit prise par le CGRFN;

qu'à des fins de conservation, il est nécessaire d'imposer une limite non quantitative interdisant la récolte de couples mères-petits au sein de la harde de caribous de l'île Southampton;

que le ministre est d'avis qu'il existe des circonstances urgentes et exceptionnelles exigeant la modification immédiate des activités de récolte à l'égard de la harde de caribous de l'île Southampton,

le ministre, en vertu de l'alinéa 121b) et de l'article 150 de la *Loi sur la faune et la flore* et de tout pouvoir habilitant, prend l'*Arrêté provisoire concernant la récolte totale autorisée à l'égard de la harde de caribous de l'île Southampton,* ci-après.

Dated,
Fait le ……..…… 2015.

……..

Hon. Johnny Mike,
Minister of Environment
Ministre de l'Environnement

WILDLIFE ACT

SOUTHAMPTON ISLAND CARIBOU HERD TOTAL ALLOWABLE HARVEST INTERIM ORDER

1. In this Order, "Southampton Island Caribou Herd" means all caribou found on Southampton Island and White Island, being part of Wildlife Management Barren-Ground Caribou Area N/BC/12, as described in the *Wildlife Management Barren-Ground Caribou Areas Regulations,* N.W.T.Reg. R-099-98, as duplicated for Nunavut by section 29 of the *Nunavut Act* (Canada).

2. (1) A total allowable harvest is established for the Southampton Island Caribou Herd in accordance with this Order.

(2) The total allowable harvest from the population of the Southampton Island Caribou Herd for the harvest year is 1600 caribou.

(3) It is presumed that the Inuit of Coral Harbour have a basic need for the full amount of the total allowable harvest set in this Order.

(4) The full amount of the total allowable harvest for the harvest year is allocated to the community of Coral Harbour.

3. There shall be no harvesting of cow-calf pairs from the Southampton Island Caribou Herd during the harvest year.

4. This Order is subject to review as soon as practicable by the NWMB in accordance with section 158 of the Act, but remains in effect until the earlier of:

(a) the coming into force of an Order to implement an accepted decision of the NWMB concerning the Southampton Island Caribou Herd; or

(b) the coming into force of a subsequent Order of the Minister.

5. The *Southampton Island Caribou Herd Total Allowable Harvest Interim Order (July 1, 2014 to June 30, 2015)*, registered as statutory instrument numbered R-019-2014, is repealed.

LOI SUR LA FAUNE ET LA FLORE

ARRÊTÉ PROVISOIRE CONCERNANT LA RÉCOLTE TOTALE AUTORISÉE À L'ÉGARD DE LA HARDE DE CARIBOUS DE L'ÎLE SOUTHAMPTON

1. Pour l'application du présent arrêté, « harde de caribous de l'île Southampton » s'entend de tous les caribous qui se trouvent sur l'île Southampton et l'île White, faisant partie de la région de gestion du caribou des toundras N/BC/12, telle que décrite dans le *Règlement sur les régions de gestion du caribou des toundras*, Règl. T.N.-O. R-099-98, reproduit pour le Nunavut par l'article 29 de la *Loi sur le Nunavut* (Canada).

2. (1) Une récolte totale autorisée est établie pour la harde de caribous de l'île Southampton en conformité avec le présent arrêté.

(2) La récolte totale autorisée au sein de la population de la harde de caribous de l'île Southampton est de 1 600 caribous pour l'année de récolte.

(3) Il est présumé que l'intégralité de la récolte totale autorisée qui est précisée dans le présent arrêté constitue un besoin essentiel pour les Inuit de Coral Harbour.

(4) L'intégralité de la récolte totale autorisée pour l'année de récolte est attribuée à la collectivité de Coral Harbour.

3. La récolte de couples mères-petits au sein de la harde de caribous de l'île Southampton est interdite pendant l'année de récolte.

4. Le présent arrêté est assujetti, dans les plus brefs délais, à un examen par le CGRFN conformément à l'article 158 de la Loi, mais demeure valide jusqu'à la première des dates suivantes :

a) la date d'entrée en vigueur d'un arrêté visant la mise en œuvre d'une décision du CGRFN acceptée concernant la harde de caribous de l'île Southampton;

b) la date d'entrée en vigueur d'un arrêté subséquent du ministre.

5. L'*Arrêté provisoire concernant la récolte totale autorisée à l'égard de la harde de caribous de l'île Southampton (1er juillet 2014 au 30 juin 2015)*, enregistré comme texte réglementaire sous le numéro R-019-2014, est abrogé.

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Nunavut Wildlife Management Board

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Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

NWMB - SENT
DATE: Dec 15/15
TF

December 15th, 2015

Honourable Johnny Mike
Minister of Environment
Government of Nunavut

Dear Minister Mike:

Re: Nunavut Wildlife Management Board decisions concerning total allowable harvest and non-quota limitation for the Southampton Island caribou herd.

1. NWMB DECISIONS

On October 16th, 2015, the Nunavut Wildlife Management Board (NWMB or Board) provided notice that it would be conducting a written public hearing to consider the Government of Nunavut-Department of Environment's *Proposal for Decision* (attached). The proposal recommended a total allowable harvest of 1600 caribou and a non-quota limitation prohibiting the harvesting of cow/calf pairs for the Southampton Island caribou herd. The deadline for submissions to the hearing was November 13th, 2015. No submissions were received.

On December 2nd, 2015 at its In-Camera Meeting (007-2015), the Board passed the following resolution:

RESOLVED that (1), pursuant to Sections 5.3.24, 5.6.16, 5.6.17(a) and 5.3.3(a) of the Nunavut Land Claims Agreement (NLCA), the NWMB modify the annual Coral Harbour level of community total allowable harvest for Southampton Island caribou to 1600, from the previous level of 800 established by the Minister of Environment on June 13th 2014, pursuant to NLCA Section 5.3.24; and (2), pursuant to NLCA Sections 5.3.24, 5.6.48 and 5.3.3(a), the NWMB maintain the prohibition on the harvest of Southampton Island caribou cow/calf pairs, also previously established by the Minister of Environment on June 13th 2014, pursuant to NLCA Section 5.3.24.

It is the NWMB's legitimate expectation that – in accordance with your June 13th, 2014 interim decision and your two previous interim decisions in 2012 and 2013, as well as the supporting resolution of the Aiviit Hunters and Trappers Organization relied upon in the Department's *Proposal for Decision* – the full amount of the total allowable harvest is to be allocated to the community of Coral Harbour.

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As per *Nunavut Land Claims Agreement* Section 5.6.19, the Board shall strike a basic needs level once a total allowable harvest has been determined. Therefore, following a pre-hearing teleconference with its co-management partners the Board passed the following resolution during its In-Camera Meeting (006-2015) on October 9th, 2015:

RESOLVED that the NWMB divide the current Southampton Island caribou hearing into two hearing: a written hearing addressing the total allowable harvest and accompanying non-quota limitation for Southampton Island caribou, and the continuation of the in-person hearing addressing the basic needs level for Southampton Island caribou.

The NWMB is planning to resume its in-person hearing to address the basic needs level in 2016.

2. RATIONALE FOR THE NWMB'S DECISIONS

During its decision-making, the NWMB carefully considered the best available scientific information and Inuit Qaujimajatuqangit. A May 2015 aerial survey produced a population estimate of 12,297 caribou, demonstrating that the population had increased from the May 2013 population estimate of 7,287 caribou. This increase is believed to be in part due to an emigration event detected by Coral Harbour hunters in the winter of 2013/2014.

In addition, the Government of Nunavut-Department of Environment staff held a face to face meeting with the Coral Harbour Hunters and Trappers Organization to discuss the survey results. During this meeting the Hunters and Trappers Organization passed a motion supporting a total allowable harvest of 1600 caribou and a non-quota limitation prohibiting the harvest of cow/calf pairs for the Southampton Island caribou herd.

Given that the 2015 population survey indicates an increase in herd size and that the next aerial survey is expected to occur in May 2017, the NWMB does not feel that setting the total allowable harvest at 1600 will cause a conservation concern during this short time frame. In fact, as the most recent Interim Order for this herd expired on June 30th, 2015, leaving no legal harvest restrictions in place, the NWMB feels that setting this total allowable harvest is essential to ensure the sustainable harvesting of this herd.

3. Conclusion

As per Section 5.3.8 of the *Nunavut Land Claims Agreement,* the NWMB is hereby forwarding its decision concerning the harvest management of Southampton Island caribou to you for your review and consideration. As there are currently no legal harvest restrictions in place, the NWMB looks forward to your prompt reply, ensuring that the Article 5 decision making process will be completed as soon as possible.

Should you or your officials have any questions or concerns regarding the contents of this letter, please do not hesitate to contact the NWMB.

Sincerely,

Ben Kovic
Chairperson of the
Nunavut Wildlife Management Board

Enclosures (1)

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Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

Nunavut

January 14, 2015

Mr. Ben Kovic
Chair, Nunavut Wildlife Management Board
Box 379
Iqaluit, NU X0A 0H0

Dear Mr. Kovic:

RE: Nunavut Wildlife Management Board final decisions concerning total allowable harvest and non-quota limitation for the Southampton Island caribou herd.

Thank you for your December 15, 2015 letter outlining the decision of the Nunavut Wildlife Management Board (NWMB) to modify the annual Total Allowable Harvest (TAH) of Southampton Island caribou.

To reiterate, the Board's decision is to increase the community TAH for Southampton Island caribou from 800 to 1600 per year. Further, the Board recommended maintaining a non-quota limitation that prohibits the harvesting caribou cow/calf pairs.

I hereby advise that, pursuant to Nunavut Land Claims Agreement Sections 5.3.9(b) and 5.3.11, I am accepting this decision.

I will direct my staff to implement the decision effective immediately, and until such time as new scientific information and/or traditional knowledge become available that will warrant a new assessment of the Southampton Island caribou TAH.

Sincerely,

Hon. Johnny Mike
Minister of Environment

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Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

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Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

ᔦᓂ 28, 2019

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ᒥᓂᑦ ᓴᙱᐅᑦᖅ:

ᐱᓪᓗᔪ: ᑲᐟᒪᵒᔪᑦ ᓄᓇᕗᑦ ᑐᔾᔪᕈᑎᖏᑦ ᐃᓄᐃᓴᓕᓕᐊᔨᓗᑎᑦ ᐅᑎᕐᑐᑦ ᐊᔨᖓᑦᑐᑦ ᐊᖏᔪᓕᕆᓕᓂᖅ ᓴᓐᓂ ᖃᑭᖅᑕᓗᐊᓂ ᑐᓴᑦ ᐅᐟᖓᖏᑦ 1,600 ᐅᕐᖏᖏᑦ 1,000

ᑕᐃᑲᓂ ᓄᓇᕗᑦ ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᑲᑎᒪᔨᖏᑦ (ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᐅᕐᓂᖅᑕᐅᓂᖅ ᑲᑎᒪᔨᓴ) ᔦᑦ 2019 ᑲᑎᒪᑎᓪᓗᑎᑦ, ᑲᐟᒪᵒᔪᑦ ᓄᓇᕗᑦ – ᐱᓕᕆᖏᓐᓄᒪᓕᒃ ᐊᐟᑎᓕᓇᔨᑦ (ᐊᐟᑎᓕᕆᔨᓴ) ᑐᓴᕐᓕᐅᕐᓕᒪᓕ ᑲᑎᒪᔨᑦ ᐃᓄᐃᓴᓕᓕᐊᔨᓗᑎᑦ ᓄᓇᕗᑦ ᐊᖏᔪᓕᕆᓕᓂᖅ ᐊᖑᑎᒍᑎᓄᑦ ᑲᑎᒪᔨᑦ ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕ (ᐊᖏᔪᓕᕆᓕᐅᔪᓐᓇᖅᑐᑦ) ᑐᑐᐃᑦ ᓴᓐᓂ ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕ ᐅᐟᖓᖏᑦ 1600 ᐅᕐᖏᖏᑦ 1000.

ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦᑲᒪᓕᐅᕐᓂᑦ ᑐᓴᕐᓕᐅᓕᒃᑕ ᑕᐃᑲᓂ ᑲᖏᔪᓂᕐᒥᑦ ᑲᑎᒪᓂᕐᒥᓂ (IC002-2019) ᐅᐟᓂ ᔦᑦ 20, 2019, ᐊᒻᒪᓗ ᐊᖅᑭᑦᓗᑎᒃ ᐅᔨᓂᖏᓐᓄᑦ ᐱᔪᓐᓇᕐᓂᖅᑐᒃ:

ᐊᖅᑭᑕᐅᕐᖓᖅ ᐃᒪᓐᒃ ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᐊᖏᔪᓗᑎᑎᒃ, ᒪᑕᑐᑦ ᐃᓕᖏᑦ 5.2.33(d), 5.6.16 ᐊᒻᒪᓗ 5.6.17(a) ᐅᔨᐊ ᓄᓇᕗᑦ ᐊᖏᔪᖃᑎᒌᓚᑲᓱᑦ, ᓴᓐᓂᑦ ᓄᓇᖃᖅᑐᑦ ᑲᑎᒪᔨᑦ ᐊᖑᑎᒍᑎᓄᑦ ᐊᖏᔪᓕᕆᓕᓂᖅ ᐃᒪᓐᓇᖅ 1000 ᑐᑐᐃᑦ ᓴᓐᓂ ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕ, ᑎᑭᓚᐊᖅᑐᓂᒃᓴᓂᒥ ᓄᑦᑕ ᑐᓴᕐᓴᖅ (ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕ ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᐃᓄᐃᓴᓕᓕᐊᔨᓗᑎᑦ ᐃᒪᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕᓚᐅᖅᑐᓂᖏᑦ) ᐊᑐᐃᓐᓇᐅᓕᕐᓴᑕ.

ᓚᓂ ᐊᖅᑭᑕᐅᕐᖓᖅ ᐃᒪᓐᒃ ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᐊᖏᔪᓗᑎᑎᒃ, ᒪᑕᑐᔪ ᐃᑕᓐᓇ 5.2.33(k) ᐅᔨᐊ ᓄᓇᕗᑦ ᐊᖏᔪᖃᑎᒌᓚᑲᓱᑦ, ᐊᖑᑎᓂᓕᔪᑦ ᒪᑕᑦᑐᐃᓕᑦ ᐊᖑᑎᓂᓕᖅᑐᒃ ᐊᖅᓇᓂᓐᓂᒃ ᓄᑦᑕᓐᓂᒃ (ᐊᖅᓇᐃᑦ/ᓄᑦᑕᐃᑦ ᐊᐃᕐᕿᓱᑦ).

ᑎᑭᐅᑎᓪᓗᑎᒃ ᒡᒃᔨᓄᖓ ᐃᓱᒪᓕᐅᕈᑎᓄᑦ, ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᑲᒪᓕᐅᕐᓂᑦ ᐅᔨᐊᓂᖓ:

- ᐃᔅᓴᓇᒪᓕᑎᒃ ᓇᐸᕐᓗᕐᖓᕐᓂᑐᒥ ᑐᓴᕐᓴᖅ – ᐃᓕᐅᓪᓗᑎᒃ ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕ – ᐃᓄᐃᓴᓕᓕᐊᔨᓗᑎᑦ..
- ᐊᖑᑎᓂᓕᖅᑐᑦ ᐱᕐᖏᖓᑐᓂᒃ ᐊᓴᐃᓂᕐᒥᓂᓂ ᖃᐅᔨᓴᖅᑭ ᓇᓗᓇᐃᔨᓱᒪᒪᓕ ᐃᓕᒃ ᐱᕐᖏᖓᑦ ᖃᑦᔨᓯᐅᓂᕐᒥᑦ ᖃᐅᔨᓴᖏᓐᓄᒪᓕᒃᑕ ᐃᓄᐃᓴᓕᓕᐊᔨᓗᑎᑦ ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ 12,297 ᐅᕐᖏᖏᑦ 8,436 ᑐᑐᐃᑦ ᐅᔨᕐᖏᓂᓂ 2015 ᐊᒻᒪᓗ 2017.
- ᖃᐅᔨᓴᖅᑭᑦ ᒡᒃᔨᑐᓗᑎᒃ ᓄᓇᓕᑦ ᐊᖑᑎᓂᓕᕆᓕᑦ ᒡᒃᔨᓇᖕᒪᑦ ᑐᑐᐃᑦ. ᖃᐅᔨᓴᖅᑭᑦ ᐊᖑᑎᓂᓕᖅᑐᑦ ᐊᒻᒪᓗ ᒥᑭᒃᑎᓂᐊᕐᓂᑦ ᑎᑎᕐᑐᑦ ᐃᑲᔪᖓᑦ ᑐᓴᕐᓴᖅ ᐊᒻᒪᓗ ᓇᓪᓕᓂᖅ ᐊᖑᑎᓂᓕᖅᑐᓕᕆᓕᑦ 1,000ᓂᒃ ᑐᑐᐃᒃ ᐊᖏᔪᓕᕆᓕᓂᖅ.
- ᐊᖑᑎᓂᓕᖅᑐᑦ ᐸᔪᓐᓇᖅᑐᑦ ᐊᓂᒍᖅᓂᒃ ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᐊᒻᒪᓗ ᓇᓪᓕᓂᖅᑐᒃ ᐊᖑᑎᓂᓕᖅᑐᓕᕆᓕᑦ ᐃᓄᐃᓴᓕᓕᐊᔨᓗᑎᑦ 2019ᒥ.

ᐅᒪᔨᖅᑕᒃᓴᓕᓕᐊᔨᓗᑎᑦ ᓂᓚᐅᔪᓐᓇᖅ ᐃᖏᕐᕕᒃᓴᓂᒃ ᐊᒻᒪᓗ ᖃᓄᐃᓕᐅᕈᑎᖏᓐᓄᒃ ᐱᓕᕆᖏᓐᓄᒪᓂᒃ ᐅᕝᕙᓘ *ᓄᓇᕗᑦ ᐊᖏᔪᓕᕆᓕᓂᖅ* ᑎᑎᕐᑐᑦ 5 ᐃᓱᒪᓕᐅᕈᑎᒃ–ᐊᖅᑭᑕᓱᐃᓂᐅᔪᑦ ᐅᒪᒥᓴᐅᓂᒃᓴᓂᒃ.

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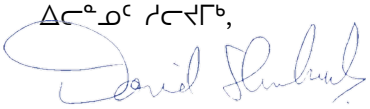
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ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᔨᖅᑎᑦᑕᐅᓯᒪᔪᑦ ᑲᑎᒪᔨᖅᑕᕐᓂ
Nunavunmi Anngutighatigut Aulapkaijitkut Katimajiat
Nunavut Wildlife Management Board

ᐱᓕᕆᑦᑐᐃᓕᓕᒫᖅ ᐅᒪᔨᓴᖅ ᐊᖏᔪᑎᑎᓕᒍ ᐊᑐᕐᑕᐅᓂᖅᑎᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᓂᕐᖏᖅᑎᑦ ᐊᒻᒪᓕ ᖃᐅᔨᓴᕐᑎᒻᒪᕆᒃ ᖃᐅᔨᒪᓂᖅᑎᑦ
Tammaqtailinahuarniriit anngutighat atuqhugit Inuit qaujimajatuqangillu ilihimaniillu ilitquhiannin
Conserving wildlife through the application of Inuit Qaujimajatuqangit and scientific knowledge

ᐃᓗᐊᑦ ᐅᕝᕙᓘᓐᓂᑦ ᐱᓕᕆᔨᑎᑎᑦ ᖃᓄᐃᑲᓘᓐᓄᖅ ᐊᐱᖅᔪᑎᑦᓴᖃᕐᓂᒃ ᐅᕝᕙᓘᓐᓂᑦ ᐃᓯᒪᓕᐊᑎᖅᑕᖃᕐᓂᒃ ᐱᓕᕆᔨᑎᑎᑦ ᐅᔪᐊ ᑎᑎᖅᑕ, ᐊᑲᐅᙵ ᖃᐅᔨᓴᖅᓯᐅᑦᓴᒥᔪᓕᐅᖅ ᐅᒪᔨᓴᕐᑎᒻᒪᕆᒃ.

ᐃᓕᓐᓄᑦ ᔪᓕᐊᑐᖅ,



Ċᓂᐊᑦ ᔪᐅᓴᖅ
ᐃᓱᒪᐅᑕᖅ ᐅᕝᓂ
ᓄᓇᕗᑦ ᐅᒪᔨᓴᕐᑎᒻᒪᕆᒃ ᑲᑎᒪᔨᖅᑕᕐᓂ

ᐱᖅᑕᑦ: ᑐᓴᒃᓴ ᑭᔪᓐ, ᑐᑭᒧᐊᖅᑎᑦᓱᓗ ᐅᒪᔨᓴᖅ ᐊᐅᓕᔪᓂᑉᓗᑦ, ᓄᓇᕗᑦ ᐱᓕᕆᓚᐅᖏᓪᓕᑦ ᐊᒪᑎᓕᓴᐃᓪᓕ

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ᑲᑎᒪᔨᑐ ᓄᓇᕗᑦ ᐱᑭᖅᑕᕐᓂᐊᕐᓂᕗᑦ
Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

Nunavut ᓄᓇᕗᑦ

ᒥᓂᑦ ᐊᕐᑎᓕᓪᓚᕆᔨᓄᑦ
Minister of Environment
Ministaat Avatiliqiyitkut
Ministre de l'Environnement

ᔪᓚᐃ 2, 2019

Ċᓂᐅᑦ ᓯᓴᒃ
ᐃᒃᓯᕈᐅᑕᖅ
ᓄᓇᕗᓴᒥ ᐅᒪᔪᓕᓪᓚᔨᔪᐊᒃᑯᑦ ᑲᑎᒪᓚᒃ
ᑎᑎᖅᑲᖅᐅᑎᖓᒐ 1379
ᐃᖅᑲᐃᑦ, ᓄᓇᕗᑦ X0A 0H0

ᓯᓴᒃᒧᑦ:

ᐱᔪᐊᓕᒃ: **ᓄᓇᕗᑦ ᓴᖅᒪᒃᑯᕐᒥ ᑐᓵᕋᐅᑎᖓ ᖃᑲᓯᐅᓇᑭᖅᑕᐅᔨᑯᓕᒥ ᓴᓕᓂᓂ ᕿᑭᖅᑕᖏᓐᓂ ᑐᑐᑕᐅᔪᐅᖅᑐᑦ 1,600-ᓂᒃ 1,000-ᓄᑦ**

ᖃᐅᔨᐅᓚᑉᒃ ᐃᓕᒪᓕᐅᑎᖃᕐᓕᐅᕋᓱ ᐱᔪᐊᑎᑦᓯᒃᑯᓕᐅᕋᓱ ᖃᑲᓯᐅᓇᑭᖅᑕᐅᔨᑯᓕᒥ ᑲᑎᓕᒥ ᐱᔪᐅᔪᐅᓇᖅᑐᑦ ᐊᖏᓚᐃ ᑯᑕᖃᖅᖏᑦᑐᒃᑯᑦ ᐃᓕᓴᕐᕕᖃᖅᖃᑎᒌᓂᒃ ᓴᓐᓂ ᕿᑭᖅᑕᖏᓐᓂ ᑐᑐᖃᖏᓐᓂᒃ.

ᓇᓗᓇᐃᓚᐊᒃᑲᓂᕐᓗᒍ ᐃᓕᓴᕐᕕᐅᑎᑎᖓᓂᒃ NWMB-ᓄᑦ:

- *"ᐊᑦᖃᑭᓴᓐ NWMB-ᓄᑦ ᐊᖏᕐᖃᓕᓴᒃ, ᓴᓕᑦᑐᑦ ᐃᓭᒃᑯᑦᑐᖅ 5.2.33(d), 5.6.16 ᐊᓚᓗ 5.6.17(a) ᓄᓇᕗᓴᒥ ᐊᖏᕐᕈᑎᓂ, ᓴᓐᓂᕆᐅᑦ ᑲᑎᒪᓚᑦ ᐱᔪᓇᖅᓂᒃ 1000-ᓄᑦ ᑐᑐᓄᑦ ᓴᓐᓂᓂ ᕿᑭᖅᑕᖏᓐᓂ, ᓄᑖᓂᒃ ᑐᕈᓯᕐᕕᕈᑎᓂᒃ (ᖃᐅᔨᓴᓐᖃᖅᑎᓄᑦ ᐅᕙᓪᓕᔭᓇᓂᑦ ᐃᓄᐃᑦ ᖃᐅᔨᓴᓚᕆᑐᖃᖏᓐᓂᒃᓯᒪᔪᑦ) ᐊᑐᐃᓇᕈᓴᐅᕙᖓᓇᓂᓇᓂ.*
- *"ᐊᑦᖃᑭᒃᑯᓇᓂᕐᓂ NWMB-ᓄᑦ ᐊᖏᕐᖃᓕᓴᒃ, ᓴᓕᑦᑐᒍ ᐃᓭᒃᑯᑦᑐᖅ 5.2.33(k) ᓄᓇᕗᓴᒥ ᐊᖏᕐᕈᑎᓂ, ᓄᑕᐅᖏᓐᓂᑐᒃᑯᑦ ᐃᓕᓴᕐᕕᖃᖅᖃᑎᒌᓂᒃᑯᑦᑐᖅᓂᒃ ᐊᖏᓪᓗᓂ ᐃᓄᐃᑦ ᓄᓇᕗᓴᒥ ᐊᖏᕐᕈᑎᓂᒃ ᐊᓯᓚᐃᓂᒃ ᐃᓕᓴᕐᕕᖃᖅᖃᑎᒌᓂᒃ ᐊᓯᓚᐃᓂᒃ ᓄᓇᕗᑦ.*

ᑕᐃᒪᐃᓚᒃᑲᑦ ᐊᖏᕐᕈᑎᓂᖓᓄᑦ ᐃᓕᓴᕐᕕᐅᑎᑎᖓᓂᒃ ᐃᓕᓴᕐᕕᐅᑎᑎᖓᓂᒃ ᐊᖏᕐᕈᑎᓂ ᐱᔪᐊᑎᑦᓯᒃᑯᓕᐅᕋᓱ ᖃᑲᓯᐅᓇᑭᖅᑕᐅᔨᑯᓕᒥ ᑲᑎᓕᒥ ᐱᔪᐅᔪᐅᓇᖅᑐᑦ ᓄᓇᕗᓴᒥ ᓄᑕᐅᖏᓐᓂᑐᒃᑯᑦ ᐃᓕᓴᕐᕕᖃᖅᖃᑎᒌᓂᒃ 1600-ᓂᒃ 1000-ᓄᑦ ᑐᑐᓂᒃ. ᐊᖏᕐᕕᒃᑲᑦ ᐃᓕᓴᕐᕕᐅᑎᑎᓂᓂᒃ ᐃᓕᓴᕐᕕᖃᖅᖃᑎᒌᓂᒃᑯᑦᑐᖅᓂᒃ ᐊᖏᓪᓗᓂ ᓄᓇᕗᑦ ᐃᓄᐃᑦ ᑐᕈᓯᕐᕕᕈᑎᓂᒃ ᐃᓕᓴᕐᕕᖃᖅᖃᑎᒌᓂᒃ.

ᐊᑐᓕᖅᑎᓂᓂᐊᖅᑕᑦ Ċᓄᓇ ᑲᑎᓕᒥ ᐱᔪᐅᔪᐅᓇᖅᑐᓂᒃ ᐃᓕᓴᕐᕕᐅᑎᑎᓂᓂᒃ ᐱᑎᐊᖅᑐᓂ ᐊᐅᓕᖅᑐᓂ ᐃᓄᐃᑦ ᐊᑐᓕᖅᑎᓂᓂᐊᖅᑕᑦ ᐱᔪᐅᔪᐅᓇᖅᑐᓂᒃ ᐃᓕᓴᕐᕕᐅᑎᑎᓂᓂᒃ ᐱᑎᐊᖅᑐᓂ ᐊᐅᓕᖅᑐᓂ ᐊᐅᓪᓚᕆᔨᔪᐅᔪᐅᓇᖅᑐᑦ ᐊᐅᓪᓚᕆᔨᔪᐊᒃᑯᑦ ᐱ

ᐱᓕᐅᖅᑭᓂᕐᓂ ᓄᒋᒥᒃ NWMB-ᑯᓐᓂᒃ ᐃᓱᒪᓕᐅᑭᑎᒥᒃ ᑐᓂᔨᐅᓱᒪᓕᖅᑲᒃᑕ ᓄᒋᖏᓪᓚᐊᖅᑕᐅᓱᒪᔪᑦ ᖃᒧᑉᐅᓕᕐᓂᖅᑭᓐᓄᑦ ᑐᑭᓱᐊᕐᑎᑐᐃᓐᓇᑦ.

ᓱᓕᔪᖅᓴᖅᑐᖓ ᑲᑐᔪᕐᑲᑎᑭᔪᑎᖅᖃᓂᐅᑦ ᑲᔪᓱᐊᕐᓂᖓᓂᒃ ᐊᑐᓕᖅᑎᑕᐅᑎᓕᓗᔪ ᒡᓐᓇ ᐊᐅᓕᕐᓱᓂᕐᒧᑦ ᐃᓱᒪᓕᐅᑭᑎᐅᔪᖅ ᖃᐅᔨᓕᕐᑐᖓᓗ ᐊᕐᓱᐱᑎᖃᖅᓱᒪᓕᕐᓱᓐᓂᒃ ᐋᖅᑭᑕᑕᐅᓂᖓᓗᓄᑦ.

ᑎᑎᕐᖅᑐᖅ,

Joe Savikataaq

ᔨ ᓴᐃᒃᑲᒡᖅ
ᒥᓂᔅᑕ

ᐊᐅᓚᖓᓂᒃ ᑎᑎᖅᒃᑲᒡᖅᑐᖅ: ᐅᒥ ᓄᐳᑦ ᓄᒃᖅᑎ – ᒥᓂᔅᑕᐅᑦ ᑐᖅᑕᐊ, ᐊᕐᑎᓕᓵᓴᒃᑯᑦ, ᓄᓇᕗᑦ ᑲᕐᒪᖓ

Long-term trends in abundance and spring distribution of the Southampton Island caribou herd: 1978 – 2023

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The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment.

ᓇᐃᓐᓇᖅᓯᒪᔪᓂ

ᓇᓱᖅᑐᖏᖅᓯᓂᑐᒥ ᑐᐅᑐᐃᓂ (*Rangifer tarandus groenlandicus*) ᓴᖅᑭᑕᐅᐅᒃᑲᓂᓕᐅᖅᑐᑦ ᓴᓚᓕᔭᒧᑦ ᐊᐅᓱᑐᔪᐊᒥᑦ ᑭᓂᕐᕿᒥᑦ ᓄᓇᕗᓕᒥ 1968–ᖃᐅᔨᓪᓚᐃᔪ, ᐱᑕᖏᑭᓐᓂᒃᑲᓇᐅᖅᑎᓪᓚᓂᑦ ᓴᓚᓕᔭᒥ 1950 ᐱᓂᐊᓕᓴᖅᑎᓪᓚᐃᔪ. ᖃᐅᔨᓴᖅᑕᐅᖏᓐᓇᔭᓂᖓ ᑕᐅᓂᐊ ᐊᐱᔨᓂᐅᐅᖏᓪᑐᖅ ᑐᐅᑐᐃᓂ ᓇᓚᓇᐃᔭᖅᓴᖅ ᐊᓂᔨᔪᐊᖃᖅᑕᑦᑐᓂᖓᓂ ᐅᓄᔭᓂᖓᑦ ᐊᓕᒪ ᓇᓂᑐᐃᓐᓇᓐᓂᖓᑦ ᐊᓂᔨᓴᖅᕿᒃᓚᑎᒃ ᑕᐃᒪᖏᓐᓄᑦ ᓴᖅᑭᑕᐅᐅᒃᑲᓂᓕᐅᖅᑎᓪᓚᓂᐃᔪ ᓴᓚᓕᔭᒥ. ᓴᓚᓕᔭᒥ ᑐᐅᑐᐃᓂ ᐅᓄᖅᓯᒪᔪᓂᑐᒥ 48-ᖏᓪᓕᐅᖅᓯᓚᑎᒃ ᐅᒪᔪᑦ 1968-ᒥᑦ ᓴᖅᑭᑕᐅᓕᐅᖅᑐᑦ 29,425-ᖏᓇᓂᒋᐅᖅᓯᖅᓯᓚᑎᒃ ᐅᒪᔪᑦ ᔪᓂ 1997-ᒥᑦ, ᑲᑎᒪᔨᓂᑦ ᐊᖏᕐᕌᒋᒪᑦ 23%-ᒥᑦ ᐅᓄᖅᓯᒋᐊᖅᓯᓚᑎᒃ. ᑕᐃᒪᖏᓐᓄᑦ ᓴᖅᑭᑕᐅᐅᒃᑲᓂᓕᐅᖅᑎᓪᓚᓂᑦ ᓴᓚᓕᔭᒥ ᑐᐃᑐᑕᐅᔪᑦ ᑭᓪᓕᖅᑲᑎᑕᐅᓕᓕᐅᖅᑐᑦ 1978–ᒪᑦ ᑎᑭᓕᔪᐃ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᑦ 1,138–ᖏᓇᓂᒋᐅᓕᐅᓪᓚᑎᒃ ᐃᓪᓇᐃᐃᑦ ᐊᓕᒪ ᑎᖅᑭᑐᕋᐃᑦ ᐅᓄᖅᓯᒪᔪᑦ ᐅᑎᖅᔨᒥᓂᕋᖅᑕᐅᓕᐅᖅᑐᑦ ᐃᑲᔪᖅᑐᐃᑐᐅᔪᓕᖅᓯᓚᑎᒃ ᓂᖅᑭᖅᓴᓂᑎᓕᖅᑕᐅᓕᐅᖅᑐᑦ ᐊᖏᓇᓂᐊᖅᑕᐅᓕᐅᖅᑐᑦ ᓂᖅᑭᖅᓴᓂᑎᓕᖅᑕᐅᓕᐅᖅᑐᑦ ᐊᖏᓇᓂᐊᖅᑕᐅᔪᓕ

ᐅᓄᖅᓱᒋᐊᖅᑐᑦ ᒪᓕᒃᓴᒋᑦ, ᑲᑎᒪᓴᒋᑦ ᐊᖏᓚᓯᑎᑕᐅᔨᓐᓇᖅᑐᑦ ᐅᓄᖅᓱᒋᐊᓕᐅᖅᓴᑦ 1,600-ᓂᒃ ᐊᖅᑭᒍᑕᒪᑦ 2016-ᒥ ᐊᖏᓚᓯᓐᓇᑐᐊᒧᒪᑦ. ᐅᓄᑭᓐᓂᖅᓴᕐᓗᓂᖅᑐᑦ ᖃᐅᔨᓴᐅᓕᐅᖅᑐᑦ ᒪᐃ 2017-ᒥ ᐅᓄᑭᓐᓂᖅᓴᕐᓗᑕᐅᓕᐅᖅᓴᑦ ᑲᑎᒪᓴᒋᑦ ᐊᖏᓚᓯᑎᑕᐅᔨᓐᓇᖅᑐᓂᒃ 1,000-ᒧᑦ ᐅᑎᖅᑎᓐᓇᓯᓴᒋᑦ ᐅᕐᓂᒍᓐᓂᓂᒃ ᓯᖁᕆᔨᓂᖁᔭᓕᑎᓄᑦ ᖃᓄᐃᓕᓂᕐᓗᐊᓂᖏᓐᓂᑦ. ᑲᑎᒪᓴᒋᑦ ᐊᖏᓚᓯᑎᑕᐅᔨᓐᓇᖅᑐᑦ, ᓇᓗᓇᐃᖅᑕᐅᔨᓂᒃ ᓯᖁᖏᓪᓗᓂᖅᑕᐅᔨᓂᒃ ᐃᑲᔨᖅᑐᑦᑕᐅᓕᐅᖅᓴᑦ 2019-ᒥ 2023-ᒥᓗ, ᑲᔨᓯᑎᑕᐅᓯᒪᒍᖅ 1,000-ᓂ ᐅᒪᔨᓂᒃ.

ᐊᑲᑐᖅᑕᐅᓴᖁᐃᓕᑦᑐᑦ ᐋᓐᓂᐊᖁᓇᒪᑦ ᖁᔭᓕᖁᔪᓐᓇ ᓯᒪᓕᕐᓂᖅᓯᓯᑎᓯᑦ ᐊᐃᒍᖃᓄᖅᑕᖏᓪᓕᐅᓂᖏᓪᓂᓂ (ᑐᑲᑐᐃᓕ ᐅᓄᖅᓱᒋᐊᓕᐅᖅᓴᑦ, ᐱᒋᐊᖁᓂᖓᓂᑦ, ᖃᓴᓯᓂᓐᓄᑦᑐᖅᓴᖅᑐᑦ ᐊᓂᒍᑎᔨᐊᖅ ᑐᑲᑐᖏᓪᓂᒃ) ᐃᓱᒪᓕᑕᐅᓯᒪᔪᖅ ᓴᖅᑭᑕᐅᑲᓂᓂᐅᖅᑎᓪᓗᒍ ᑐᑲᑐᐃᓐ ᓴᓪᓕᓐᓂᒃ, ᐊᒻᒪᓗ ᐊᐃᑕᑐᖁᓕᑕᐅᑐᐃᓐᓇᐊᖃᖅᖁᓂ ᑐᑦᑐᓐᓂᑦ ᓇᓪᓕᖅᑐᒥᓂᖅᓴᕐᓗᒥ ᖃᐅᔨᓴᐅᓕᐅᖅᓯᒪᓪᓗᒍ 2000-ᒥ. ᓇᓪᒍᐊᖁᓇᓐᓇᒪᑦ ᐊᑲᑐᖅᑕᐅᓂᖏᓪᑦ ᐅᓄᖅᓱᒋᐊᓕᐅᖅᓴᑦ 1.7%-ᒥᑦ ᐃᓄᐊᓐ 2000-ᖓᔪᓪᓗᒍ 58.8%-ᒧᑦ ᐃᓪᓕᓯ 2011-ᒥ ᐅᓄᖅᓱᒋᐊᖁᓂᖏᓪᓂᓪᓗᒍ ᐅᓄᑭᓐᓂᖅᓴᕐᓗᕆᓯᐅᔨᐅᓄᑦ ᓄᖁᐅᖅᓴᕐᑐᑦ ᖃᓪᓚᓯᐅᓂᖏᓪᑦ ᑕᕐᔪᓇᓂᑕᑦᑕᐃᓪᓇᖅ. ᓄᖁᐅᖅᑐᑦ ᑲᑕᒋᐊᓕᐅᖅᓴᑦ 93.1%-ᖓᔪᓕᐅᖅᓴᖅᑐᒃ ᐃᓄᐊᓐ 2001-ᒥ 37%-ᑐᐃᓪᓇᐅᓕᓂᖅᓴᖅᑐᒃ ᐃᓕᓴ 2011-ᒥ. ᖃᓄᐃᓕᓂᕐᓗᐊᓂᖏᓪᑦ ᖃᐅᔨᓴᖃᑕᐅᔨᓴᑦ ᓴᓕᓂᒋᐊᖅᑐᑦ ᑐᑲᑐᖏᓪᓂᑦ ᐅᓄᑭᓐᓂᖅᓴᕐᓗᕆᓯᐅᔨᓐᓇᖅᑐᓂᒃᓴᓂᖅᓴᑦ 9%-ᒥ ᐊᑭᑦᑭᒍᑕᒪ 1997 ᐊᒻᒪ 2013 ᐊᑯᓪᓂᖓᓂ, ᑭᖓᓂᐊᔪᓪᓗ ᐅᓄᖅᓱᒋᐊᖅᓴᖅᑐᒃ ᓄᓇᓕᖓᓂ ᐅᖃᐅᓯᐅᔨᓴᑦ ᒪᓕᒃᓴᒋᑦ ᐊᒻᒪᓗ ᓇᑭᖓᓐᓄᑦᑐᖅᑐᑦ ᐃᐅᐅᖅᓴᑯᑦ ᐅᑭᐅᖅᑯᑦ ᐊᓯᐊᓇᓂᖅᓴᕐᓗᓂᕐᓂᐊᔨᓴᑦ 2014 ᐊᒻᒪ 2015 ᐊᑯᓪᓂᖓᓂ. ᑲᑎᑕᓂᑦ ᓯᑭᓗᐊᖏᑦᓴᑦ 2015-ᒥ 2023-ᓗᑦ, ᓯᓚ ᐅᓄᖏᓪᓂᖅᓴᐅᓕᑦ ᑕᐃᔨᓯᒪᓂᓂ. ᐊᑐᓪᓚᕆᖃᑦᑕᓂᖅᑐᓄᑦ ᑕᕐᔪᓂᖓ ᑐᑲᑐᓂᓂ ᓂᕈᐱᓴᓪᓗᓂᔨᐅᒃ ᐊᒻᒪᓗ ᑭᓇᐅᔭᓕᐅᑎᓪᓗᒍᓂᓗᑦ ᐊᖏᓴᐅᔨᓴᑦ ᐱᔨᔪᐅᑎᓪᓗᒍᑦ ᓇᓪᓕᖅᑐᒥ ᐊᖏᓚᓯᑎᑕᐅᔨᓐᓇᖅᑐᑦ ᑲᑎᒪᓴᒋᑦ, ᓯᓚᓐᓇ ᑲᑎᒪᓴᒋᑦ ᐊᖏᓚᓯᑎᑕᐅᔨᓐᓇᖅᑐᑦ ᐊᑭᑦᑭᒍᑕᒪᑦ ᒪᑭᑉᑐᒃ ᐱᓕᕐᓂᓄᑦ, ᑕᐃᒪᓐᓇᖅᑐᒥ ᐊᑭᑦᑭᒍᑕᒪᑦ ᐅᓄᖅᓱᒋᐊᓕᐅᖅᓴᑦ ᐱᓕᕐᓂᓄᑦᑐᖅᑐᑦ

ᐃᓂᐅᔪᓂ ᐱᓕᒪᑕᐅᔾᔪᑕ ᐊᑯᓂᐅᔪᒥ ᐊᓐᓇᐅᒪᔪᓐᓇᖏᓐᓂᖏᓐᓂᑦ ᑲᑎᑕᓄᑦ. ᑕᒃᑯᐊᐊ ᓄᓇᐃᑦ ᓴᓗᐊᐅᓪᓚᑐᑎᒃ ᐃᓄᖏᓂᑦ ᐱᓇᓵᓪᒋᑕᐅᔪᓂᑦ ᕿᓐᓄᓴᑕᐅᓪᓚᑐᑎᒃ ᐊᓯᔨᕐᑕᐅᑐᐃᓐᓇᐊᖅᓯᒪᑎᒃ ᐊᓕᒪ/ᐅᕐᓂᓪᓕᑦ ᐊᒃᑐᒃᑕᐅᑐᐃᓐᓇᐊᖅᓴᓚᑎᒃ ᓄᓇᐅᑦ ᑕᐅᑦᑐᖓ ᐃᓕᒋᐅᓐᓂᓪᓚᖅᓯᒪᓂᖅᓴᖅ ᓇᓕᐊᓄᑐᐃᓇᖅ ᐊᑯᓂᐅᔪᓂᑦ ᑐᒃᑐᓄᑦ ᐊᐅᓚᑦᑎᓂᓪᓗᑦ ᕿᓂᓇᐅᑎᓪᓗᑦ.

ᐊᐅᓚᔨᔪᑎᑦ ᐊᓯᔨᔾᑕᑕᐅᔪᑦ ᐊᒥᓱᓂᖏᓪᓂ ᑕᑯᐅᐅᖅᑎᑦᑕᖅᑐᓂᑦ ᓇᐃᓴᐃᓂᓕᒫᖅᑐᓂ ᓴᓕᑦᑐᖓᓂᑦ ᐊᒥᓱᓂ, ᐊᔪᓐᓇᖅᓯᓂᑲᓐᓂᕐᒪᑦ ᐃᓄᖏᓪᓗᑎᒃ ᐃᓄᖏᓪᓗᑎᒃ ᐊᐃᓕᖅᑐᓂᑦ ᐃᓄᖏᓪᓗᑎᒃ. ᐱᔨᔪᑕᐅᓚᓪᐊᑕᖅᑯᓱᔪᑦ ᐋᓵᓂᐅᓚᔨᐅᔪᑦ ᓇᓱᓴᔪᓐᓇᖅ ᖃᓄᐅᔾᔨᓂᖅ IV, ᐊᖏᔪᓗᑕᐅᔪᑦ (ᑐᓵᔨᑦ ᓂᕈᖏᓂᓂᑦ ᓂᐅᕐᐃᐊᖅᓴᖏᓐᓂᖅᑐᓂᑦ ᐃᓄᖏᓂᑦ ᑐᐱᒪᔨᔪᐊᑎᒃᑯᓐᓂᑦ), ᐅᑭᐅᔪᓪᓚ ᓱᓕᓚᖅᑕᖏᑦᑐᓂᖏᓐᓄᓕᓂᑦ, ᐱᓚᐊᖅᑐᑎ ᓂᓕᖅᑕᖏᑦᑎᓕᐅᔪᓐᓄᑦ ᐃᓕᖏᓂᓂᑦ ᐊᖅᑭᒍᔪᑦ. ᓇᓚᓇᖏᑎᐊᖅᓴᖅ ᖃᐅᔨᔪᖅᑕᓕᓂᖏᓂᖏᓂᓂᑦ ᐃᓕᖏᓂᓂᑦ ᐊᐃᓕᖅᑐᓂᑦ ᐊᐃᓕᖅᑐᓂᑦ ᐊᓯᔨᕐᑕᐅᑐᐃᓐᓇᐊᖅᓴᖅᓄᑦ ᐅᑭᓚᐅᑦ ᐊᐃᓕᖅᑐᖅᑕᐅᑦ ᐃᓄᖏᓪᓗᑎᒃ ᐃᓄᖏᓪᓗᑎᒃ ᐃᓄᖏᓪᓗᑎᒃ ᐊᐃᓕᖅᑐᓂᑦ ᐃᓄᖏᓪᓗᑎᒃ ᐃᓄᖏᓪᓗᑎᒃ ᐃᓄᖏᓪᓗᑎᒃ

SUMMARY

Barren-ground caribou (*Rangifer tarandus groenlandicus*) were reintroduced onto Southampton Island (SHI) from Coats Island in the Kivalliq Region of Nunavut in 1968, following their extirpation from SHI in the early 1950s. The ongoing assessment of this unique herd suggests large fluctuations in abundance and distributional changes since its re-establishment across SHI. The SHI caribou herd grew from the 48 animals introduced in 1968 to an estimated population of 29,425 animals by June 1997, yielding an annual rate of increase of approximately 23%. Since its re-introduction the SHI herd harvest was restricted up until 1978 when an estimate of 1,138 adult and yearling caribou suggested that numbers had recovered to a level that could once again support a limited subsistence harvest. The herd continued to grow and by 1993 the first commercial harvest was approved. After nearly 30 years of growth, herd abundance declined from the estimated high of 29,425 in June 1997, to 21,277 by June 2005, 14,389 in June 2007, 13,651 in June 2009, 8,467 in June 2011, then to 7,287 in May 2013. Herd abundance began to stabilize and by May 2015 the population had increased to 12,370. However, by 2017, the population had declined again to 9,200. During this short period of decline, caribou distribution concentrated into a core area within the south-central portion of the Island in the vicinity of the Kirchoffer River. By 2019 the estimate of herd abundance was 12,054, which was similar to the 2015 estimate suggesting stability. In 2023, herd size was estimated to be 12,651 indicating continued stability. Harvest estimates over the same periods varied widely. Following the 2011 survey, an annual Total Allowable Harvest (TAH) of 1,000 caribou was applied over the 2012 and 2013 harvesting seasons, and a TAH of 800 over the 2014 and 2015 harvesting seasons. Following the population increase detected in 2015, the TAH was increased to 1,600 caribou annually for the 2016 harvesting period only. A reduction in abundance detected in May 2017 led to the reduction in TAH back to 1,000

in an attempt to reverse or stabilize the trend. The TAH, bolstered by indications of stability in 2019 and 2023, has been maintained at 1,000 animals.

Susceptibility to disease and parasites due to low genetic heterogeneity (the population grew, from introduction, from a very small group of Coats Island caribou) has been a concern since the reintroduction of caribou to SHI, and was a likely catalyst to the wide spread infection of caribou with *Brucellosis suis* which was first detected in the population in February 2000. Prevalence of Brucellosis climbed from 1.7% in February 2000 to 58.8% in March 2011 and this increase is thought to have contributed to decreased pregnancy rates over the same period. Pregnancy rates dropped from a high of 93.1% in February 2001 to a low of 37% in March 2011. Trend analysis suggests that the SHI caribou population decreased at a rate of 9% per year between 1997 and 2013, followed by an increase due to a locally reported and genetically supported winter immigration event between 2014 and 2015. While the herd has been stable from 2015 to 2023, it is still below historic levels. Given the reliance of users on this population for subsistence and commercial harvesting purposes, continuation of the current TAH is recommended to maintain stability and continue to promote recovery over the next 2 to 3 years, at which time a reassessment of herd abundance and TAH levels should be undertaken.

Observed distributions from the 2003 survey indicated little change from distributions observed during previous surveys though localized densities had decreased when compared to 1997 results. Overall, caribou continued to heavily use the central portions of the Island along the Kirchoffer River valley and along the transition between the western flats and the eastern highlands from November 1978 through to May 2023. Small variations between years were likely the result of changes in snow cover and associated icing, with caribou feeding at higher elevations during years of early snow melt and/or reduced snow cover. The central portion of the Island, in the vicinity of the Kirchoffer River and along the general transition from the western flats to the more topographically rugged eastern highlands of the Island, have received high use by SHI caribou in the spring and during calving from introduction to present, making these areas extremely important to the long-term viability of the herd. The protection of these

areas from anthropogenic disturbance that may modify and/or impact the landscape will be a critical component of any long-term herd management plans.

The mechanisms driving the changes in abundance observed over the entire survey history of the Southampton Island caribou population are multiple, and difficult to isolate and quantify, suggesting that further research is required. It appears that the main drivers have been the disease *Brucella suis* Type IV, harvest (with emphasis on the sale of caribou meat through social media), and poor winter weather, primarily in the form of icing events in some years. Clearly the need to continue monitoring disease prevalence in SHI caribou is required if we are to understand present day infection rates and associated productivity for the herd. Recently, hunters have reported fewer caribou with signs of disease, and a noticeable increase in the number of calves observed in 2015 through 2023 which suggests that disease prevalence may be decreasing further. If this is the case, and Brucellosis no longer represents a significant mechanism of decline, then harvest, along with weather, and condition monitoring, should become the focus of future monitoring for the SHI herd. Additionally, more effective means of monitoring the harvest, and any exports of caribou meat off the island will be critical in understanding the true extent of the harvest for both subsistence and meat sales. At present these tools are not available to enforcement officers within Nunavut, suggesting that further thought and required amendments to current harvesting regulations, and perhaps the Nunavut Agreement itself, should be seriously considered by wildlife management organizations and the Government of Nunavut. Attempts to control the sale of caribou meat through social media have failed under the current Management regime and consideration should be given to addressing this issue through amendments to legislation. In recent consultations with Kivalliq community HTOs, all communities expressed a willingness to address the problem in this way, suggesting that some mutual agreement could be reached to more permanently resolve this issue. If nothing is done to monitor this novel and growing mechanism of caribou meat sales, we fear the problem will grow more serious for Nunavut's subsistence harvesters as more and more caribou populations within Nunavut will be managed through the establishment of a TAH.

ABSTRACT

Barren-ground caribou (*Rangifer tarandus groenlandicus*) were reintroduced onto Southampton Island (SHI) from Coats Island in the Kivalliq Region of Nunavut in 1968, following their extirpation from SHI in the early 1950s. The ongoing assessment of this unique herd suggests large fluctuations in abundance and distributional changes since its re-establishment across SHI. The SHI caribou herd grew from the 48 animals introduced in 1968 to an estimated population of 29,425 animals (+/- 3,050, 95% CI; CV=5.5%) by June 1997, yielding an annual rate of increase of approximately 23%. Since its re-introduction the SHI herd harvest was restricted up until 1978 when an estimate of 1,138 (95% CI = 419.7-1,856.3; CV=63.0%) adult and yearling caribou suggested that numbers had recovered to a level that could once again support a limited subsistence harvest. The herd continued to grow and by 1993 the first commercial harvest was approved. After nearly 30 years of growth, herd abundance declined from the estimated high of 29,425 in June 1997, to 21,277 (95% CI = 18,098-24,896, CV=8.0%) by June 2005, 14,389 (95% CI = 12,684-16,325; CV=6.4%) in June 2007, 13,651 (95% CI = 12,091-15,412; CV=6.1%) in June 2009, 8,467 (95% CI = 7,558-9486, CV = 5.7%) in June 2011, then to 7,287 (95% CI = 6,580-8,071, CV = 5.0%) in May 2013. Herd abundance began to stabilize and by May 2015 the population had increased to 12,370 (95% CI = 11,140-13,736, CV = 5.1%). However, by 2017, the population had declined again to 9,200 (95% CI = 7,755-10,915; CV = 8.7%). During this short period of decline, caribou distribution concentrated into a core area within the south-central portion of the Island in the vicinity of the Kirchoffer River. By 2019 the estimate of herd abundance was 12,054 (95% CI = 10,354-14,032, CV = 7.5%), which was similar to the 2015 estimate suggesting stability. In 2023, herd size was estimated to be 12,651 (95% CI = 11,044-14,493, CV = 6.7%) suggesting continued stability. Harvest estimates over the same periods varied widely. Following

the 2011 survey, an annual Total Allowable Harvest (TAH) of 1,000 caribou was applied over the 2012 and 2013 harvesting seasons, and a TAH of 800 over the 2014 and 2015 harvesting seasons. Following the population increase detected in 2015, the TAH was increased to 1,600 caribou annually for the 2016 harvesting period only. A reduction in abundance detected in May 2017 led to the reduction in TAH to 1,000 in an attempt to reverse or stabilize the trend. The TAH, bolstered by indications of stability in 2019 and 2023, has been maintained at 1,000 animals.

Susceptibility to disease and parasites due to low genetic heterogeneity has been a concern since the reintroduction of caribou to SHI, and was a likely catalyst to the wide spread infection of caribou with *Brucellosis suis* which was first detected in the population in February 2000. Prevalence of Brucellosis climbed from 1.7% in February 2000 to 58.8% in March 2011 and this increase is thought to have contributed to decreased pregnancy rates over the same period. Pregnancy rates dropped from a high of 93.1% in February 2001 to a low of 37% in March 2011. Trend analysis suggests that the SHI caribou population decreased at a rate of 9% per year between 1997 and 2013, followed by an increase due to a locally reported and genetically verified immigration event between 2014 and 2015. While the herd has been stable from 2015 to 2023, it is still below historic levels. Given the reliance of users on this population for subsistence and commercial harvesting purposes, continuation of the current TAH is recommended to maintain stability and continue to promote recovery over the next 2 to 3 years, at which time a reassessment of herd abundance and TAH levels should be undertaken.

Key words: *Commercial harvest, barren-ground caribou, caribou, Rangifer tarandus, Southampton Island, Coral Harbour, Kivalliq, disease, Brucellosis suis, Nunavut, population survey, demographic studies.*

Table of Contents:

1.0 INTRODUCTION 12

2.0 STUDY AREA 16

3.1 Caribou Introduction (1967)–An Historical Account 19

3.2 Aerial Surveys (1978-2023) 20

3.2.1 Surveys pre-1990 20

3.2.2 Single Observer Pair Method 21

***3**.2.3 Dependent Double Observer Pair Method 32*

3.3 Distribution 37

3.4 Statistical Analysis of Abundance and Trend 40

3.4.1 Strip transect surveys (1997-2023) 40

3.4.2 Double observer/strip transect analyses (2011-2023) 40

3.4.3 Distance sampling analyses (2017) 43

3.4.4 Trend analyses 44

3.5 Disease Monitoring 45

3.6 Genetic Analysis – Movement 46

4.0 RESULTS AND DISCUSSION 49

4.1 Population Distribution 49

4.2 Strip Transect Surveys 61

4.2.1 Dependent double observer analyses (2011-2015,2019, 2023) 66

4.2.2 Distance sampling/double observer pair sampling in 2017 75

4.2.3 Comparison of estimates from strip transect, dependent double observer pair, and distance sampling 81

4.3 Trend estimates: 85

4.3.1 Trend from 1978 to 1997(the increase phase) 85

4.3.2 Trend from 1997 to 2023 (the decline phase) 87

4.4 Effect of Disease on Abundance: 91

4.4.1 Brucellosis and heard trend 93

4.5 Harvest 97
4.5.1 Harvest Management and Planning 2011 to present 100
4.6 Population Genetics 106

5.0 CONCLUSIONS & RECOMMENDATIONS 112
5.1 Aerial Survey Methods 112
5.2 Herd Trend 113
5.3 Future Management 116

6.0 ACKNOWLEDGEMENTS 118

7.0 LITERATURE CITED 119

1.0 INTRODUCTION

Following the extirpation of caribou from Southampton Island (SHI) in the early 1950s, there was much discussion regarding their reintroduction as well as recognition of the careful husbandry that must go hand in hand with such a program (MacPherson, 1967). Discussions continued up until 1967 at which time Northwest Territories Commissioner Stuart Hodgson along with D.S. Munro, Director of the Canadian Wildlife Service (CWS), made the decision to move forward with the reintroduction of caribou to SHI. The target group for the source population was the Coats Island Herd due to its close proximity and ecological and environmental similarities to Southampton Island. Regional Superintendent A. G. Loughery and Research Supervisor A.H. MacPherson began implementation of the program on June 7th, 1967. The very first caribou on Southampton Island following their extirpation were 48 caribou captured and transported from Coats Island.

From their start on Southampton Island caribou were watched closely by wildlife officials. The first evidence of the success of the introduction was communicated by the game Management Officer Ed Bowden who estimated between 100 and 125 caribou ranging over the southern half of the island in the winter of 1971 (Game Management Files, 1971, 1972, 1973). In November 1978 Kraft (1978) initiated the first quantitative aerial survey since the introduction, estimating 1,138 (95% CI = 420-1,856; CV = 63.0%) adult and yearling caribou. The results of the 1978 survey suggested that numbers had recovered to a level that could once again support a limited subsistence harvest. From 1978 to 1999 the Government of the Northwest Territories managed the progress of the 1967 reintroduction of caribou. From 1999 onward, the Government of Nunavut (GN) took over this responsibility. The current GN management strategy follows a management plan developed in partnership with the Coral Harbour Hunter and Trappers Organization (HTO) and consists of a program

relying upon regular aerial surveys and an extensive health monitoring program. Due to confirmed declines following the 2011 population estimate, the health monitoring component of population studies of the SHI herd, which included a one hundred animal harvest for the assessment of health and condition, has been suspended to allow all available tags to go to the hunters of Coral Harbour.

Following introduction, caribou had steadily increased to an estimated population of 29,425 animals (95% CI = 26,375-32,475; CV = 5.5%) by 1997, which represented the highest number of caribou ever recorded on the island. The 1997 estimate suggested an annual rate of increase of 23% since introduction. An aerial population survey conducted in 2003 detected the first decline of caribou since their introduction, showing a population estimate of 17,981 (95% CI = 15,854-20,108; CV = 6.0%) caribou. The population remained relatively stable between June 2003 and a follow-up survey flown in June 2005. The June 2005 abundance survey estimated 20,582 (95% CI = 17,526-23,638; CV = 7.5%) but the observed increase from the 2003 mean estimate was not found to be statistically significant. The first evidence of a significant drop in abundance since the 1997 estimate was recorded in June 2007 when survey results estimated 15,452 (95% CI = 13,594-17,310; CV = 6.0%) caribou. This confirmed a 14% decline over this 10 year period (1997-2007) (Campbell, 2015). The SHI caribou population continued its decline to 13,953 (95% CI = 12,163-15,743; CV = 7.0%) in June 2009, and to 7,902 (95% CI = 6,641-9,163; CV = 8.0%) by June 2011. The June 2011 result prompted the setting of a Total Allowable Harvest (TAH) for the 2012 harvesting season. The establishment of the TAH effectively slowed the decline and by May 2013 abundance was statistically stable at an estimated 7,287 (95% CI = 6,242-8,332; CV = 7.0%) caribou (Campbell, 2015).

A genetic study, based on the confirmation of a movement event between the mainland and SHI and first reported by local hunters in the winter of 2013, was initiated in spring 2014 and completed a year later. The study confirmed that mainland barren-ground caribou DNA came onto SHI for the first time since introduction sometime during the winters of either 2013 or 2014, following the May 2013 survey but prior to May 2015. This movement is believed to have increased the population some time prior to the May 2015 abundance survey where 12,368 (95% CI = 10,518-14,542, CV = 8.1%)

adult and yearling caribou were estimated. Since May 2015 the herd appears to have remained stable though still below historic levels. However, the potential for a founder effect for this introduced population, leading to low genetic heterogeneity and associated increased susceptibility to disease and parasites was a concern. Initial concerns surfaced in February 2000 when the reproductive disease *Brucellosis suis* was detected for the first time, growing to a prevalence of 58.8% by February 2011. High rates of *Brucellosis* in the population are thought to have been the main catalyst behind recorded declines observed following the June 1997 survey (Campbell, 2015).

Reproductive disease is thought to have been a major contributor to overall population declines since the early 2000's. Pregnancy rates declined from approximately 80% in 1997, to 60% in 2003, reaching a low of 36.3% in 2008, then climbing to 55.6% in 2010, to decline again to 37.0% in 2011 (Campbell, 2015). The reproductive disease *Brucellosis suis* (Brucella) was first detected in February 2000 at a prevalence of 1.7% (of sampled caribou) and by March 2011, rates of infection had risen to 58.8% (Campbell, 2015). High Brucella infection rates raised concerns regarding human health, as well as the ability of the SHI caribou herd to sustain and recover from substantial commercial harvesting and subsistence harvesting pressures.

Health studies were first initiated in March 1993 and continued as a linked program alongside the annual commercial harvest up to and including the 2009 harvesting year. Following the cessation of the commercial harvest in 2009, health studies continued as a standalone 100 caribou harvest up until 2011. Following continued declines in SHI caribou abundance, health screening after 2011 continued in the form of harvester sample kits up to Spring 2015. Results from these studies suggest body condition did not change significantly over most years. One notable exception was that in February and March of 2011 the SHI herd was in the poorest condition reported since the initiation of the program (Campbell, 2015). During the winters of 2010 and 2011, hunters reported numerous freezing rain events and extensive icing across the island. These icing events likely made winter forage less accessible to caribou (Tyler, 2010). Icing events that reduced accessibility to food were likely associated with the observed declines in condition, which further reduced reproductive success (Cameron et al.

1993, Gerhart et al. 1997). Support for this hypothesis stemmed from numerous local reports of starving and dead caribou during mid to late winter 2011 (Campbell, 2015).

Brucellosis and icing events are not the only issues threatening the SHI caribou population. Over-harvest has become a dominant threat to the long-term sustainability of this population. In particular, ongoing caribou harvest for trade or sale within Nunavut territory, is suspected of driving harvest levels beyond subsistence requirements, though as of spring 2024, the extent of this harvest relative to the current TAH is unconfirmed. Elements of this unregulated sale of caribou meat are also driving increased harvest pressure on breeding females: customers offer higher payment for fat caribou, which during the winter and spring seasons are predominantly pregnant females.

In this report we summarize the findings of 45 years of monitoring of the SHI caribou herd whose range covers the extents of Southampton and White Islands, Nunavut. We discuss trends in abundance, disease, harvest, and other long-term threats to the herd and their implications for management of this barren-ground caribou population.

2.0 STUDY AREA

At 43,000 km^2 Southampton Island is the largest island in Hudson Bay. The island is divided into the Northern and Southern Arctic ecozones. The Northern Arctic ecozone covers White Island, and the northeastern third of Southampton Island including northern Bell Peninsula and can be further divided into the Boothia-Foxe Shield eco-province and the Wager Bay Plateau ecoregion (**Figure 1**).

The Wager Bay Plateau ecoregion covers the northeastern Kivalliq Region, extending westward from the northern portion of Southampton Island on Hudson Strait to Chesterfield Inlet in the south, and as far west as the Back River (Wiken, 1986; Natural Resources Canada, 2001). The mean annual temperature of this ecoregion is approximately -11°C with a summer mean of 4.5°C and a winter mean of -26.5°C. The mean annual precipitation ranges from 200 to 300 mm. This ecoregion is classified as having a low Arctic ecoclimate and is characterized by a discontinuous cover of tundra vegetation, consisting mainly of dwarf birch (*Betula glandulosa*), willow (*Salix spp*.), northern Labrador tea (*Ledum decumbens*), mountain avens (*Dryas integrafolia)*, and *Vaccinium* spp. Taller dwarf birch, willow, and alder (*Alnus spp.*) occur on warm sites, while wet sites are dominated by willow and sedge (*Carex spp*.). Lichen-covered rock outcroppings are prominent throughout this ecoregion. This ecoregion is composed of massive Archean rocks of the Canadian Shield that form broad, sloping uplands, plains, and valleys. It rises gradually westward from Chesterfield Inlet to an elevation 600 meters above sea level (ASL), where it is deeply dissected. Soils of this ecoregion are typically Turbic and static cryosols developed on discontinuous, thin, sandy moraine and alluvial deposits, while large areas of regosolic static cryosols are associated with marine deposits along the western coast. Permafrost is continuous with low ice content. Naujaat and Baker Lake are the main settlements within the ecoregion (Wiken, 1986; Natural Resources Canada, 2001).

The Southampton Island Plain ecoregion covers the remainder of Southampton Island and all of Coats and Mansel Islands (**Figure 1**). The mean annual temperature within this ecoregion is approximately -11°C with a summer mean of 3°C and a winter mean of -24.5°C. The mean annual precipitation ranges from 200 to 300 mm (Wiken, 1986; Natural Resources Canada, 2001). This ecoregion is classified as having a low Arctic ecoclimate and is characterized by a nearly continuous cover of low Arctic shrub tundra vegetation, consisting of dwarf birch, willow, northern Labrador tea, mountain avens, and *Vaccinium* spp. Wet sites are dominated by willow, sedge, and moss. The region is composed of the partly submerged blanket of flat-lying Paleozoic carbonate rocks and is generally less than 90 meters ASL in elevation. Bedrock outcrops are common as are Static and turbic cryosol soils developed on level to undulating morainal and marine deposits. The maritime influence is limited to the late summer and early fall as coastal ice and fog persist for long periods in the summer when the sea ice is absent. The ecoregion is underlain by continuous permafrost with medium ice content composed of ice wedges. Coral Harbour is the largest settlement within this ecoregion (Wiken, 1986; Natural Resources Canada, 2001).

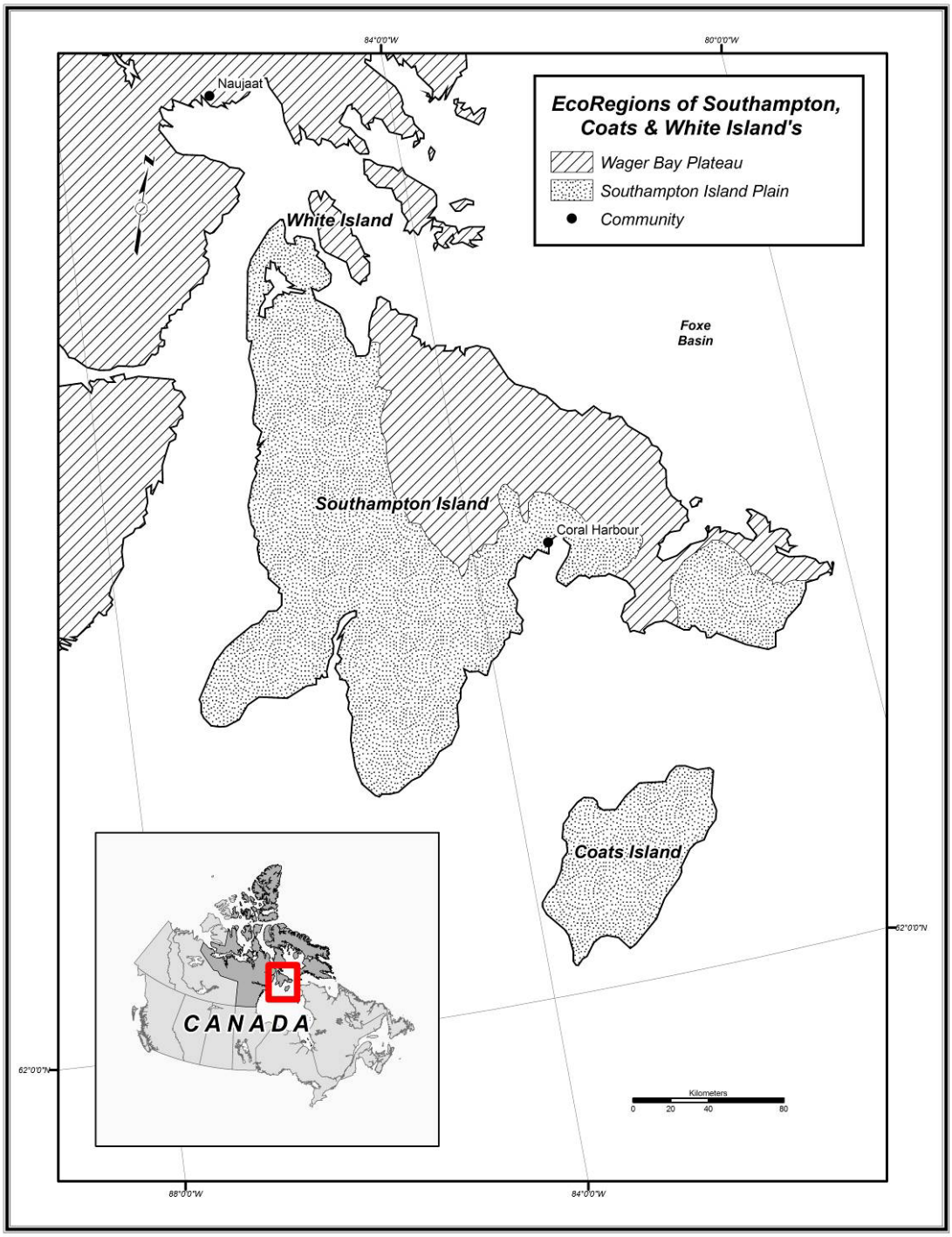


Figure 1 Ecoregions of the Southampton Island, Coats Island, and White Island study areas (Wiken, 1986; Natural Resources Canada, 2001).

3.0 METHODS

3.1 Caribou Introduction (1967)–An Historical Account

Caribou reintroduced to Southampton Island from the Coats Island herd were initially immobilized from a G2 helicopter using a CO_2 gas-operated Palmer 'Cap-chur' gun and both 2 cc and 5 cc darts. The darts used during the initial capture contained a pre-measured dose of crystalline succinylcholine ('Anectine') dissolved in isotonic water at a concentration of 5mg/cc and administered at a rate of 5 mg per 100 pounds (MacPherson and Manning, 1968). The tranquillizer 'Largactil' at a concentration of 25 mg/cc and a dosage of 125mg per 100 pounds was used to maintain immobility. Up to seven animals were captured in this way, per day. Captured animals were taken to a base camp with an enclosure on Coats Island where they were weighed, medicated with Vitamin E and Selenium as well as an antihistaminic and anti-biotic, injected into the shoulder. Animals were held in the enclosure for up to one week. From the enclosure animals were re-captured for transport by roping or tackling, tied up in slings, tranquillized and placed in single and twin Otter fixed wing aircraft for their final transport and release onto Southampton Island in the vicinity of the Coral Harbour airport. In total, 66 caribou comprising 12 bulls, 26 cows (one pregnant), and 10 calves (8 male and 2 female), were captured and released onto Southampton Island. Of the original 66, 18 animals died, two from dart wounds, two from broken legs, and six from what appeared to be capture myopathy (CWS correspondence, 1969; MacPherson and Manning, 1968). Reasons for the remaining eight deaths do not appear in the available records. In total, 48 animals survived the reintroduction, making up the founding group of Southampton Island's reintroduced caribou herd.

3.2 Aerial Surveys (1978-2023)

Survey data from all surveys were initially analyzed using Jolly's Method 2 for unequal sample sizes (Jolly 1969 *In* Norton-Griffiths 1978). Only counts of adults and yearlings were used for the final population estimates as calves are not considered fully recruited into the population until they have survived their first winter. Lake areas were not subtracted from the total area calculations used in density calculations.

3.2.1 Surveys pre-1990

Following their reintroduction onto Southampton Island, caribou were monitored periodically by both local and Government wildlife officials, primarily using ground-based methods. An aerial survey flown by Kraft in November 1978 was the first scientific population estimate made since the reintroduction of caribou onto SHI (Kraft, 1978; Kraft, 1981). Kraft used a stratified transect survey method to cover three (3) strata that were believed to represent the full extent of the Southampton Island Herd's fall distribution. The survey was flown between November 22nd and 25th, 1978 and utilized one observer on each of the left and right side of the single engine high wing DeHavilland Single Otter aircraft. Transects were placed 6.44 km apart for a total of 12.5 % coverage of the entire survey area. Effective strip width was a total of 800 meters, 400 meters out each side of the aircraft, while survey elevation was 122 meters AGL (above ground level) with a mean survey speed of 140 kph (Kraft, 1981). Population estimates were derived by calculating the density of caribou observed for all transect strips, and multiplying density by the total stratum area.

A second survey method was employed in June 1986 and consisted of a stratified random block survey design (Heard and Grey, 1987). The census zone was divided

into 5 strata which received differential coverage ranging from 11% to 54%. The stratification into census zones was based on a pre-survey reconnaissance, habitat and range preference delineations (Parker, 1975), and timely observational data from both local hunters, wildlife service personnel, and previous survey observations of caribou. A Bell 206B helicopter was used as the survey vehicle at variable speeds and altitudes. The survey personnel consisted of two rear seat observers, a front left seat navigator, and a pilot. Sightings from all personnel were recorded. Each caribou was approached and circled so that its sex and age class could be determined. Heard and Grey also attempted to determine sightability through the re-surveying of portions of three blocks at three times the initial survey intensity to determine the differences between the two surveys. This method of determining sightability was, however, unsuccessful due to the movement of animals between survey zones. A third survey flown June 1991 aimed at improving the 1987 survey effort. The June 1991 survey followed, for the most part, the same methodology employed in 1987 by Heard and Grey (Ouellet, 1992). The main modifications made to the 1987 methods were made to ensure complete coverage of the island, involving the delineation of two strata defined as low density, which were surveyed using an aerial strip transect survey flown with a Cessna 337 fixed-wing aircraft. Sampling intensity varied from 11% to 51% over 48 transects and/or blocks flown.

3.2.2 Single Observer Pair Method

The March 1990 and 1991, July 1995, June 1997, June 2003, June 2005, June 2007, and June 2009 surveys were flown using a single observer pair stratified systematic aerial strip transect method. The specifics of each of these surveys are discussed.

March 1990 and 1991

The March 1990 survey was flown using a Cessna 337 fixed wing aircraft at 120 meters above ground level (AGL) at various speeds between 185 and 222 kilometers per hour. The survey crew included two rear seat observers, a front right seat navigator, and the pilot. The strip width on each side of the aircraft was 400 meters. The survey covered the entire Island using 18 transects, which yielded 4% coverage, leading to low survey intensity and precision (Ouellet, 1992). Because of the low precision of the 1990 estimate, the survey was repeated in 1991 utilizing a quadrat method. The 1991 survey estimate being of greater precision, has been used as representative of this period. Unfortunately little of the March 1990 Southampton Island (SHI) caribou survey method has been documented (Ouellet, 1992).

July 1995

A single observer pair stratified systematic aerial strip transect survey was flown in late June and early July of 1995, however, there were serious problems with sightability due the caribou's darker summer coats following the late spring molt (May/June) of their light-colored sun-bleached winter coats (Mulders, 1995). Because of this Biologists of the time believed the 1995 estimate of 18,275 (95% CI = 16,885-19,665) to be a significant underestimate, and therefore results from this survey have been excluded from this reports summary of historical population trends. Due to the sightability issues with the 1995 survey, a specific recommendation was made to conduct surveys earlier in June or before, prior to molting (Mulders, pers. comm.). The survey to re-estimate the population was later flown in June 1997.

June 1995 to 2009

Aerial surveys flown from 1997 to 2009 utilized a stratified systematic aerial strip transect method flown with a high wing single engine turbine or gas, fixed wing aircraft. During these survey years the survey crew included a pilot (front left seat), a data recorder/navigator (front right seat), a left rear seat observer and a right rear seat observer. The pilot monitored air speed and altitude while following transects using a

Trimble GPS (June 1995, June/July 1997), remaining survey years were navigated using cloned pre-programmed routes on two Garmin C-176 (2003 through 2009) geographic positioning system (GPS) units set to WGS 1984 datum and true north. The data recorder/navigator was responsible for recording observer observations, assisting in the navigation of transects, and monitoring a second identically programmed GPS unit for the purposes of double-checking the position, altitude, distance from transect, and ground speed (all other surveys). Geographic coordinates (waypoints) and numbers of adult and calf caribou were either recorded on compact tape recorders with associated positions marked on a map (1997), or recorded on data sheets (all other surveys). The responsibilities of the left-side and right-side observers were to monitor their 400-meter strips and call out numbers of caribou separated by adults and calves, both on and off transect as indicated by wing strut markers. The 2003, 2007, and 2009 air crews remained the same throughout the survey, while during the 2005 survey, one or more observers were changed part way through the survey. Information on the 1995, and 1997 surveys concerning consistency in air crews could not be found. Survey aircraft included a Cessna 337 during both the 1995 and 1997 surveys, a DeHavilland Turbo-Beaver during the June 2005 survey, and a Cessna grand caravan during the 2003, 2007, and 2009 surveys. Reconnaissance surveys used to delineate strata extents were flown in June of 1997, 2003, 2005, and 2007 (**Figure 2**). Transect spacing remained similar between all surveys up to 2009 (**Figure 3**). The largest single modification to strata occurred within the Low South strata in 2005 as a result of extensive flooding along the Boas River, which travels through the strata. In this case transects over the Boas River area were shortened to avoid flooded areas where caribou would not be found. Strip width (w) for all surveys were established using dowels, or for post-2005 surveys, streamers, attached to the wing struts, based on calculations described in Norton-Griffiths (1978) (**Figure 4**).

Strip width calculations were confirmed by flying perpendicularly over runway distance markers or other fixed distance markers periodically throughout the survey. The strip width area for all abundance surveys was 400 meters per side.

Standardized reconnaissance transects with a total observation strip of 800 meters (400 meters per side) were flown during the June 1997, 2003, 2005, and 2007 surveys

and used to stratify caribou into areas of similar relative densities used later to allocate effort for the abundance phase (Heard 1987). A stratified random transect method was then used during the abundance phase of all surveys from 2009 to present (**Figures 5 to 8**). Evidence of a distributional constancy in SHI caribou's use of spring range led to the cessation of reconnaissance surveys and the re-direction of survey effort into established abundance strata following the June 2007 survey.

Attempts were made to maintain a constant altitude of 122 meters during all surveys prior to 2003, while a radar altimeter was employed during all remaining surveys to increase altitude precision between transects and survey years. Transect design for all surveys randomly placed the first transect within each of three strata (Low, Medium and High) along a line of latitude or otherwise randomly selected, with each sequential line being evenly spaced. Once established, the same survey transects (with small modifications during some years), were flown as caribou distributions changed little between years.

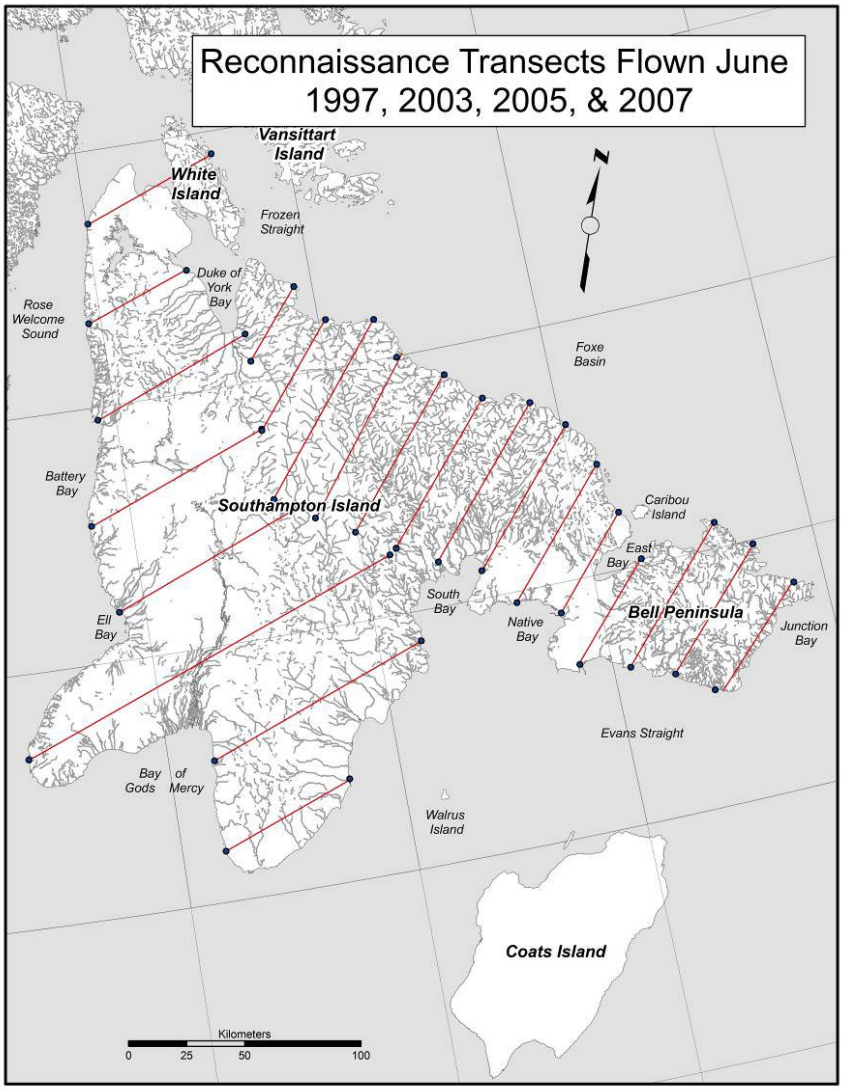


Figure 2. Reconnaissance transects flown in June of 1997, 2003, 2005, and 2007, to delineate abundance strata used to estimate Southampton Islands (including White Island) caribou population.

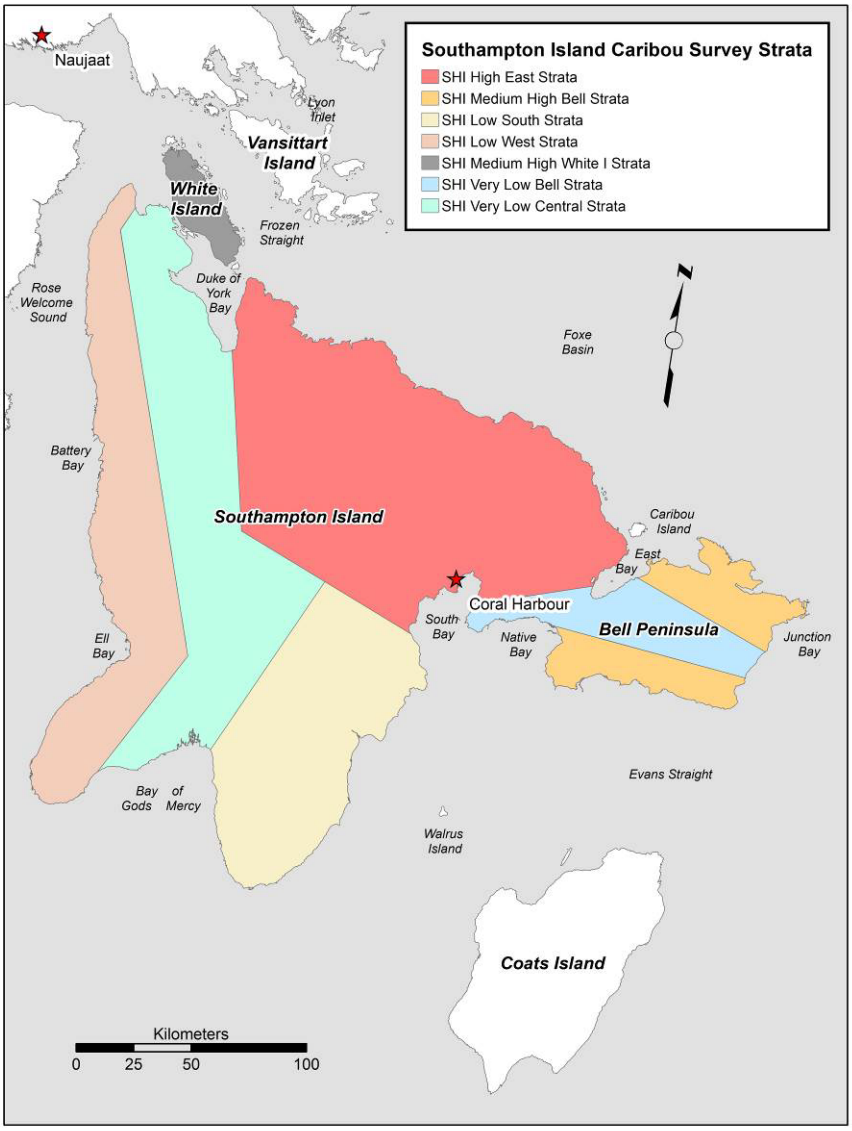


Figure 3. Abundance strata initially delineated using reconnaissance flights to map relative densities of caribou. As caribou distribution changed little across all survey years, these strata were utilized for all surveys post-2007.

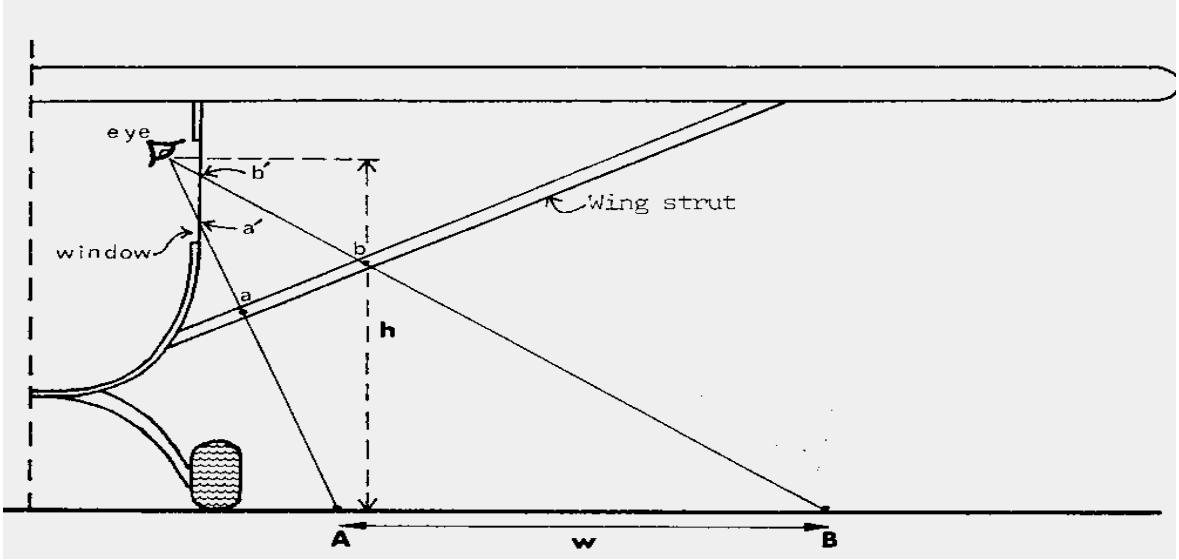


Figure 4. Schematic diagram of aircraft configuration for strip width sampling (Norton-Griffiths, 1978). W is marked out on the tarmac, and the two lines of sight a' – a – A and b' – b – B established. Streamers are attached to the struts at *a* and *b*. a' and b' are the window marks.

Where:

w = W * h/H

W = the required strip width**;**

h = the height of the observer's eye from the tarmac; and

H = the required flying height

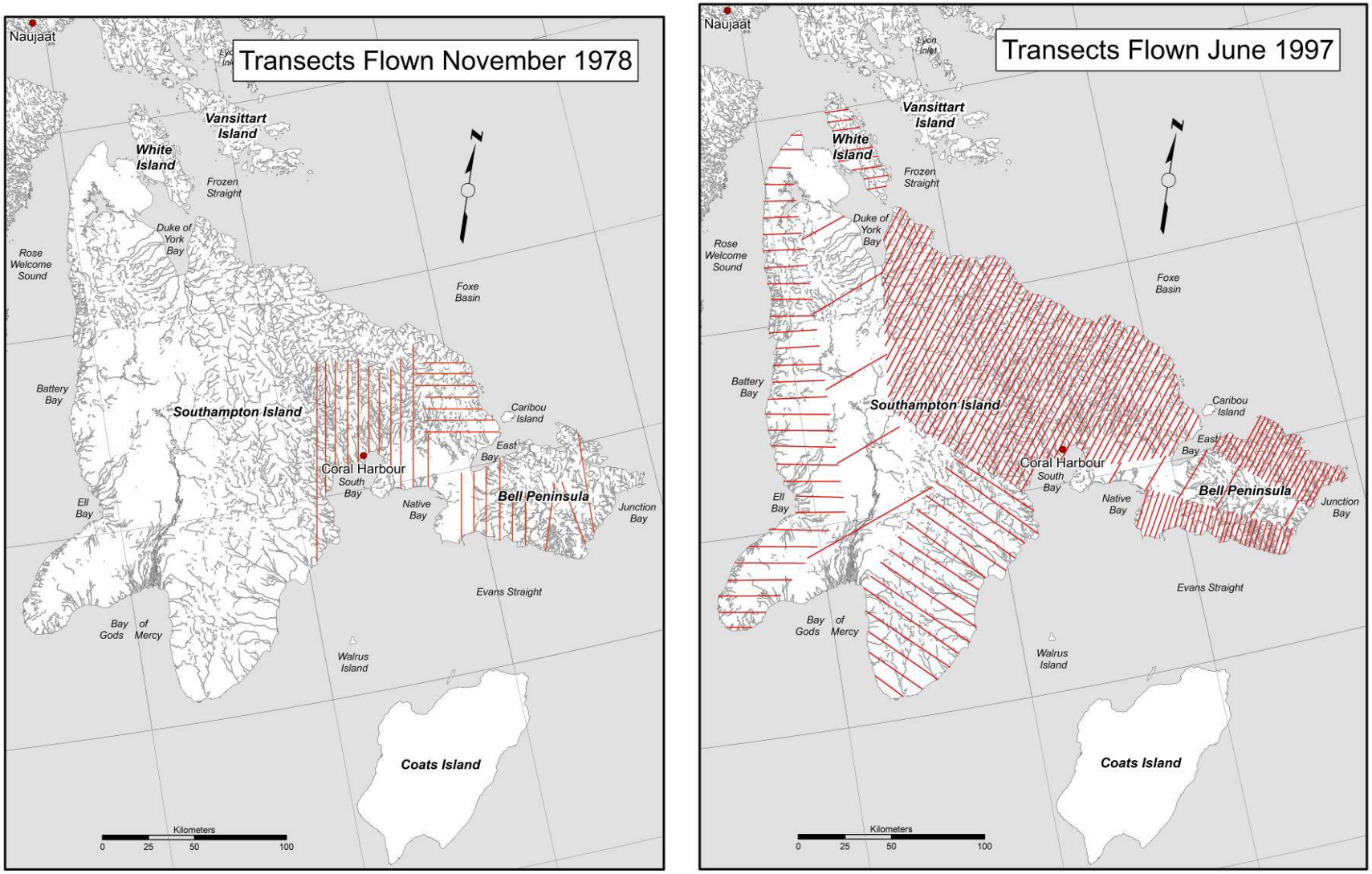


Figure 5. Stratified random transect surveys flown in November 1978 and June/July 1997.

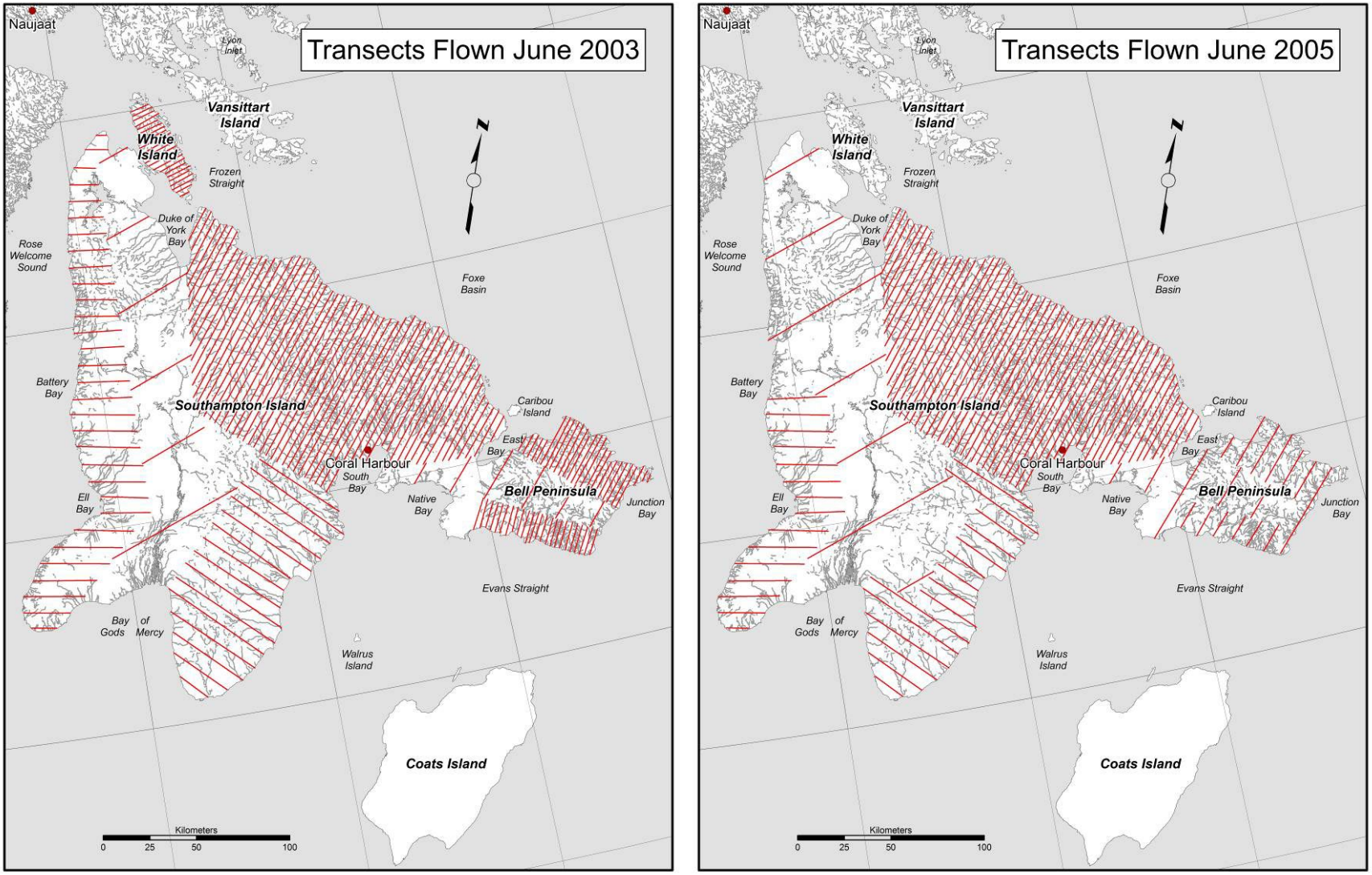


Figure 6. Stratified random transect surveys flown in June 2003 and 2005.

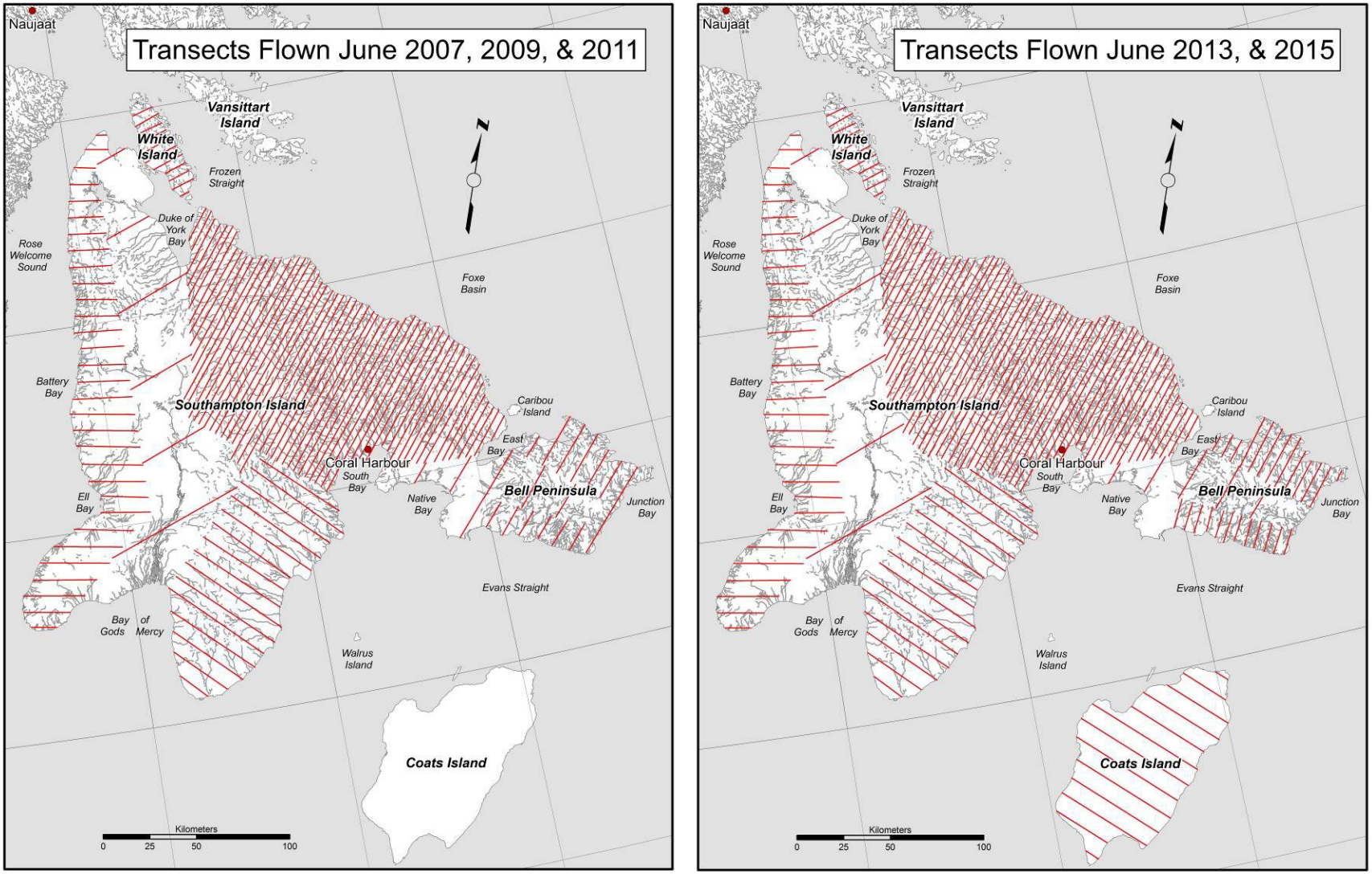


Figure 7. Stratified random transect surveys flown in June 2007, 2009, 2011 and May 2013 and 2015.



Figure 8. Stratified random transect survey flown in May 2017, 2019, and 2023 (Note: Coats Island survey was planned but not flown in 2023 due to logistic constraints.).

2011 to 2023

The June 2011 and May 2013, 2015, 2017, 2019, and 2023 surveys followed a similar methodological set up as those discussed for surveys between 1995 and 2009 with the following notable exceptions: **1**- Survey timing changed from early June to early May beginning in 2013 as weather modelling indicated more "flyable" days and the provision of more continuous snow cover for improved sightability, while maintaining distributions similar to June based strata. These changes proved successful leading to a permanent change in survey scheduling to mid to late May; **2**- The 2017, 2019 and, 2023 air crews remained the same throughout the survey, while the 2011 and 2015 surveys recorded one or more observers had changed part way through the survey; **3**- A Cessna grand caravan was used up to May 2017 at which time the aircraft type changed to a DeHavilland Twin Otter from 2019 to present; **4**- Between 2011 and 2023, strata remained similar between surveys (**Figure 3**). During this period transect spacing did increase with decreasing relative densities within the Bell Peninsula and White Island strata between 2011 and 2015, but was increased back to higher coverage from 2017 to present due to detected increases in caribou relative density; **5**- All surveys from 2011 to present were upgraded from a single-pair observer platform to a dependent double-pair observer platform to increase survey accuracy and precision, and to involve more HTO and community based participation within GN survey programs.

3.2.3 Dependent Double Observer Pair Method

The June 2011, and May 2013, 2015, 2017, 2019, and 2023 surveys, were marked by a change in visual survey method. An additional two (2) observers and one (1) data recorder were added to the survey crew increasing the crew to seven (7) individuals including the pilot/pilots. This method, termed the dependent double observer pair method, was adopted to all Kivalliq regional ungulate surveys following its development

for Barren-ground caribou abundance surveys in June 2011 (Campbell et al. 2013). Pilot studies conducted on Muskox abundance in 2010 and barren-ground caribou abundance in 2011, confirmed fewer animals were being missed while using this new visual observation methodology. Additionally, more HTO representatives could be involved in the survey whereby four (4) dedicated observers, of which at least two (2) were experienced observers split between the left and right sides on the plane, could be involved in the aerial survey.

The dependent double-observer pair method involves one “primary” (front) observer who sits in the front seat of the plane and a “secondary observer” (rear) observer who sits behind the primary observer on the same side of the plane (**Figure 9**). One data recorder sitting on the right-hand side of the aircraft is assigned the right primary and secondary observers while the second data recorder, sitting on the left side of the aircraft is assigned the left primary and secondary observers. The method adhered to five basic rules; ***1 -*** The primary observer called out all groups of caribou (number of caribou and location) he/she saw within the 400-meter-wide strip transect after they passed halfway between the primary and secondary observer (approximately at the wing strut). This counting included caribou groups that were between approximately 12 and 3 o’clock for right side observers and 9 and 12 o’clock for left side observers. The main requirement was that the secondary (rear) observer be given time to call out all caribou seen before the primary (front) observer called them out; ***2 -*** The primary observer called out whether he/she saw the caribou that the secondary observer saw or did not see, and both would call out any observations of any <u>additional</u> caribou groups the other did not see. Both the primary and secondary observers waited to call out caribou until the group observed passed half way between observers (between 3 and 6 o’clock for right side observers and 6 and 9 o’clock for left side observer); ***3 -*** The observers discuss any differences in group counts to ensure that they are calling out the same groups and/or different groups and to ensure accurate counts of larger groups; ***4 -*** The data recorder categorized and recorded counts of caribou groups into “primary only”, “secondary only”, and “both”, entered as separate records; ***5 -*** The observers switched places approximately half way through each survey day (i.e. during

re-fueling) to monitor observer ability based on position (front vs rear). The recorder noted the names and positions of the primary and secondary observers.

The sample unit for the survey was "*groups of caribou*" not individual caribou. This group-based observation method created problems for the data recorder trying to determine when a group of caribou started and ended and/or differed from individual caribou that were separated by short distances. To resolve this issue, recorders and observers were instructed to consider individuals to be those caribou that were observed independent of other individual caribou and/or groups of caribou through an estimated separation of 100 meters.

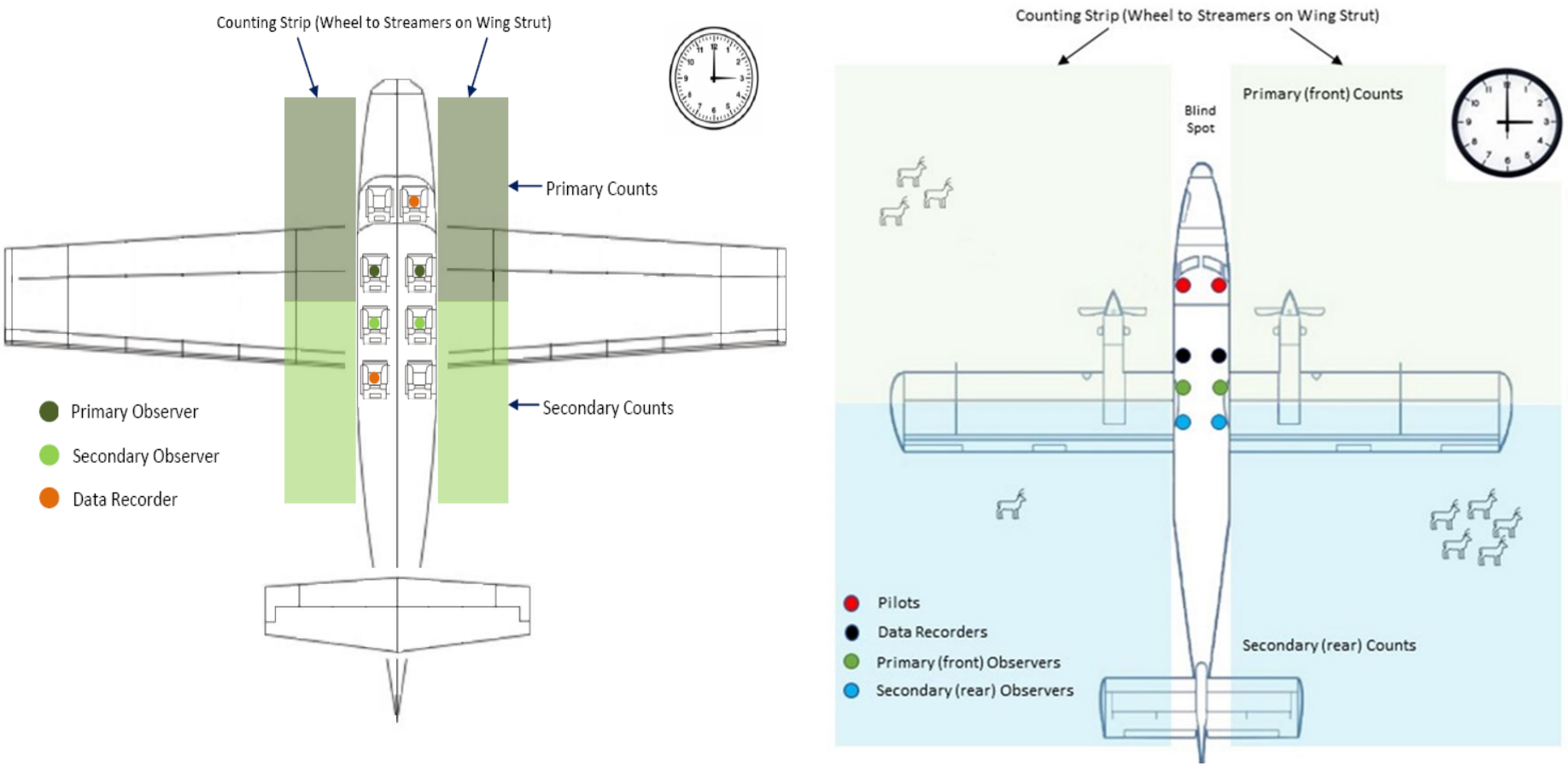


Figure 9. Observer position for the dependent double observer pair method employed on the 2011, 2013, 2015, 2017, 2019, and 2023 Southampton Island caribou abundance surveys for a Cessna caravan (left) and DeHavilland Twin Otter (right). The secondary observer calls caribou not seen by the primary observer after the caribou have passed through the main field of vision of the primary observer. The small hand on a clock is used to reference relative locations of caribou groups (e.g. "Caribou group at 3 o'clock" would suggest a caribou group 90º to the right of the aircrafts longitudinal axis).

3.3 Distribution

Distribution maps were developed to graphically summarize survey data for survey observations up to and including May 2023. The distribution maps were generated through an interpolation which provided an estimate of the number of caribou present at un-sampled locations based on the known values gathered at surrounding locations. In this study an Inverse Distance Weighted (IDW) interpolation technique was used within ArcMap's Spatial Analyst extension for surveys not employing double observer pair methods. IDW is an effective means of interpolating scattered data points. It estimates values by calculating a weighted average. The farther a sampled point is from the cell being evaluated, the less weight it has in the calculation of the cell's value (Watson and Phillip, 1985). This type of analysis generates a surface consisting of cells, each with an attribute (in this case, population density), used to interpret the spatial distribution of geographic points and then convert them into a continuous distribution reflecting estimates of point densities.

To account for null data, all survey observations were first buffered to ten kilometers. To account for nil records, those portions of the transect not covered by the observation buffers were then divided into 1-kilometer (km) segments with the first starting 1-km from the edge of the nearest buffered observation or transect starting point. At each division between 1-km segments, a point with an observed value of zero was inserted. The analysis was then run using the survey observation values as well as the newly populated zero values. The analysis requires that a series of parameters be defined. The parameters, along with a description and the settings used are summarized in **Table 1**.

The resulting surfaces were themed by the population density attribute and overlaid on a base map of Southampton and White Islands. Density class or "bins" are developed to reveal the most visual information and highlight and estimate

distributional changes between surveys. As area estimates of relative densities on Southampton Island are mostly aggregated in the 0 to 5 caribou/km^2 class, the bins were developed to accentuate these lower relative densities.

Table 1. Inverse distance weighting (IDW) analysis parameters employed in the analysis of Southampton Island caribou densities from 1978 through 2023.

Parameter	Description	Settings
Z Value Field	The Z value is the attribute being used to derive the interpolated surface.	The population attribute stored in the field Number.
Power	The higher the Power value, the greater the influence of values closest to the interpolated point. The most common value for the Power parameter is 2.	A value of 2 was selected.
Search radius type	The search radius can be either variable or a fixed distance.	As the sample points were not evenly distributed, a variable search radius was used that assessed the data points nearest to the particular cell of interest.
Search radius setting	The search radius setting specifies either the maximum distance of a fixed radius search or the number of points for a variable type.	The number of points considered in each of the analyses was 12.
Output cell size	The resolution (or cell size) of the grid (the surface) resulting from the analysis.	An out-cell size of 100 m2 was specified resulting in a population density of animals per hectare.

3.4 Statistical Analysis of Abundance and Trend

3.4.1 Strip transect surveys (1997-2023)

The standard Jolly estimator (Jolly 1969, Krebs 1998) was applied to the strip transect data for all years using an assumed strip width of 800 meters with the single exception of the 2017 survey year where the strip width was 918 meters. Strip transect data for 2017 was created using distance sampling, where the first 2 observation bins of data for both sides of the aircraft, equaled an 800-meter strip. For all year's strata were estimated separately and then combined for a total estimate of the SHI herd. Coats Island was also surveyed in 2013, 2015, and 2017 and was excluded from the South Hampton Island estimate. Log-normal confidence limits were generated for all estimates (Thompson 1992).

3.4.2 Double observer/strip transect analyses (2011-2023)

Because this method assumes equal sightability between observers it is essential that the observers switch seats over the course of a survey day (preferably) or periodically (Cook and Jacobsen 1979). Estimates of herd size and associated variance were measured using the mark-recapture distance sampling (MRDS) package (Laake et al. 2012) in the program R (R Development Core Team 2009). In MRDS, a full independence removal estimator, which models sightability using only dependent double observer pair information (Laake et al. 2008a, Laake et al. 2008b) was used, making it possible to derive dependent double observer pair strip transect estimates. Strata-specific variance estimates were calculated using the formulas of Innes et al.

(2002). Data were explored graphically using the ggplot2 (Wickham 2009) package in R.

Modelling of sighting probability variation

One assumption of the dependent double observer pair method is that each caribou group observed had an equal probability of being sighted. To account for differences in sightability we also considered the following sightability covariates in the MRDS analysis (**Table 2**). Each observer pair was assigned a binary individual covariate and models were introduced that tested whether each pair had a unique sighting probability. Previous analyses (Campbell et al. 2012; Boulanger et al. 2014) suggested that the size of the group of caribou had strong influence on sighting probabilities and therefore we considered linear and log-linear relationships between group size and sightability (**Table 2**). Cloud and snow cover were recorded by data recorders as they changed and were included in the analysis as ordinal rankings. We suspected that sightability was most likely lowest in mixed snow cover conditions and therefore we considered both categorical and linear models to describe variation in sightability caused by snow cover. Cloud cover could also influence sightability by causing glare, flat light, or variable lighting. We used the same basic strategy to model cloud cover variation as we did for snow cover variation.

The fit of models was evaluated using the Akaike Information Criterion and corrected for small sample size (AIC_c). The model with the lowest AIC_c score was considered the most parsimonious, thus minimizing estimate bias and optimizing precision (Burnham and Anderson 1998). The difference in AIC_c values between the most supported model and other models (ΔAIC_c) was also used to evaluate the fit of models when their AIC_c scores were close. In general, any models with a ΔAIC_c score of less than 2 between them were considered to have equivalent statistical support.

Table 2. Covariates used to model variation in sightability of caribou for the dependent double observer pair analysis conducted on the abundance survey of the Southampton Island caribou herd.

Covariate	Acronym	Description
Observer pair	observers	each unique observer pair
Group size	size	size of caribou group observed
	Log(size)	Natural log of group size
Snow cover	snow	snow cover (0,25,75,100 %)
	snowc	continuous
Cloud cover	cloud	cloud cover (0,25,75,100 %)
	cloudc	continuous

3.4.3 Distance sampling analyses (2017)

For the 2017 survey only, distances of caribou groups from the survey planes were binned into intervals (0-200m, 201-400m, 401-600m, 601-1000m, and 1001m-1500m), based upon markers on the wing struts of the survey plane, as was done in the 2014 Baffin Island caribou survey (Campbell et al. 2015). In addition, the dependent double observer pair also assessed sightability of caribou in the 0–200-meter strip closest to the aircraft.

A combined distance sampling and mark-recapture approach was used to estimate abundance for the 2017 data set. The basic approach involved using mark-recapture analytical methods to estimate the probability of detection of caribou at 0 distance from the survey plane, and distance sampling methods to estimate the decrease in probability of detection at greater distances from the plane. This approach ensured a more robust estimate than using distance sampling methods alone which assume that the probability of detection of caribou groups at 0 distance from the plane is 1 (Borchers et al. 1998, Buckland et al. 2004, Laake et al. 2008a, Laake et al. 2008b, Buckland et al. 2010, Laake et al. 2012).

As with the dependent double observer pair analysis, the MRDS R package (Laake et al. 2012) was used to build mark-recapture and distance sampling models. The general approach used was to build distance sampling models with the mark-recapture model parameters held constant. Once a parsimonious distance sampling model was identified, the mark-recapture model was built to further assess sightability of caribou in immediate proximity to the aircraft. The same general set of covariates used in the dependent double observer pair analysis (**Table 2**) were used for both the dependent double observer pair and distance sampling models. As with the dependent double observer pair analysis, AIC methods were used to assess model fit. Overall model fit was also assessed using goodness of fit tests as well as graphical comparison of detection functions with histograms of frequencies of observations from the survey.

3.4.4 Trend analyses

We used log-linear models to analyze trends for the increase and decrease phase of the caribou abundance dataset (McCullough and Nelder 1989, Thompson et al. 1998, Williams et al. 2002). Our models assumed an underlying quasi-Poisson distribution of estimates with population change occurring on the exponential scale. Abundance survey estimates were weighted by the inverse of their variance therefore giving more weight to the more precise estimates. A log-link was used for the analysis allowing direct estimates of yearly rate of change as one of the regression β terms. Additive terms were used to estimate phase-specific trends and the effect of a possible immigration event, likely occurring between May 2013 and May 2015 surveys, on SHI herd trend.

3.5 Disease Monitoring

The health and condition of SHI caribou was monitored through the collection of harvest samples, beginning in 1995 and ending in 2015 (Campbell, 2015). Disease monitoring began as a standard commercial harvest protocol whereby the Canadian Food Inspection Agency (CFIA) randomly collected between 300 and 400 blood samples from commercially harvested animals from 1993 through 2007. From 2007 to 2011, blood samples were obtained by GN biologists from animals collected for health and condition harvests from remaining ventricular and/or arterial blood. Sampled blood was drained into red topped vacutainers, left to stand approximately two hours at between five and ten degrees Celsius, then spun down in a centrifuge for approximately ten to fifteen minutes to separate the serum from cellular material. Individual serums were poured off into new sterile red-topped vacutainers, carefully packed and allowed to freeze at approximately -20^{o} to -30^{o} degrees Celsius. Frozen blood serums were then transported first to labs in Lethbridge Alberta for Brucellosis and Tuberculosis screening, then to the CFIA lab in Ontario for further disease testing. Adult female reproductive tracts were also collected in 2005 for the purposes of identifying reproductive stress and/or disease. All sampling pre-2009 was carried out in conjunction with the commercial harvest which collected samples between mid-February and early April up to 2009 and continued as a smaller research based harvest up to 2011. The GN did not have access to all CFIA test results. A smaller hunter based sampling program continued from 2011 to spring 2015. Samples collected during the hunter based sampling program included the lower jaw, the left kidney including fat, filter paper and vacutainer sampling of ventricular and/or arterial blood, sex, estimated age, and reproductive status, and any evidence of disease and parasites.

Additional variables measured included a ratio of bare kidney to kidney fat index, the recording and sampling of any apparent disease and/or diseased tissue, the recording

and sampling of parasitic infections, the measurement of back fat, bone marrow condition (in some years), pregnancy rates, fetal sex (in some years), and age through the analysis of cementum-annuli from the sampling of I-1 (the first incisor) from the lower jaw. In the case of the GN health studies, all anatomical components of an individual caribou being sampled and/or measured were recorded along with a common tag number and the associated harvest year. This common tag number allowed for the pooling of analysis results to provide a comprehensive description of the health, age, and sex of the individual being sampled. From 1995 through 1999, and in association with a community based annual commercial harvest, approximately 400 animals per year were sampled in this way up to 2009.

Sampling across February and March 2000 through 2009 was reduced to approximately 200 to 300 animals (excluding 2001, 2002 and 2003). Following the closing of the commercial harvest in 2009, disease monitoring was reduced to 100 animals in 2010 and 2011. Following the 2011 survey results and subsequent application of a total allowable harvest (TAH) in 2012, the community of Coral Harbour requested that the health monitoring harvest be suspended so that all TAH allocations could be provided to the community. Only Brucella results are provided in this report while a summary of condition through time can be obtained in Campbell (2015).

3.6 Genetic Analysis – Movement

Over the winter of 2014, Coral Harbour hunters reported caribou tracks crossing the ice from the mainland across to the northwestern extents of SHI. Though no estimates of the total number of caribou involved in this crossing were communicated beyond “hundreds”, local hunters had observed more calves in June 2014 and increased densities of caribou in the following harvesting year compared to preceding hunting seasons. Results from the May 2015 abundance survey estimated a significant

increase in SHI caribou abundance of both adults and calves compared with 2013 results. This population increase was theorized by both the community of Coral Harbour and Wildlife officials' to be related to the immigration of mainland caribou onto SHI. We set out to further investigate this hypothesis using population genetics.

We engaged Wildlife Genetics International (WGI) to pursue this question using the clustering programs Structure and Genetix, which produce accessible visual summaries of the results (Paetkau, 2015; Paetkau 2003). Caribou tissue collected on SHI in 2004, and 2014, as well as tissue samples collected from hunters in the Naujaat area in 2014, were compared for assessments of ancestry. Additionally, WGI used archived samples from the Qamanirjuaq caribou herd for added comparative analysis with the SHI herd. WGI used GeneClass2 to assess ancestry hypotheses explicitly (Paetkau, 2003; Paetkau, 2004; Paetkau, 2015). Initial explorations included data from South Baffin Island, Melville Peninsula, and the Ahiak and Beverly herds of barren-ground caribou, but these explorations did not identify any associations of relevance to SHI. Genotyping used a standard set of 18 highly variable microsatellites that they had consistently employed for other barren-ground caribou genetic analyses in Nunavut and the Northwest Territories. The analysis proceeded in two rounds of 9 markers (including gender markers), as all 18 markers cannot be loaded into a single sequencer lane. After completing a first pass with all 18 markers, WGI did a round of reanalysis ('cleanup') of individual data points that were scored with low confidence (1) during the first pass (Paetkau, 2015). This reanalysis used 5 µL of DNA per reaction, up from the 3 µL used for first pass. In some cases, multiple attempts were made to confirm problematic data points. At the end of the cleanup phase, 6 samples from SHI still had low-confidence scores in their genotypes (Paetkau, 2015). In total, WGI was able to successfully genotype complete 18-locus genotypes for 37 samples from Naujaat, and 131 from SHI. With genotyping completed, WGI defined an individual for each unique multilocus genotype, taking identifiers from the first sample to be assigned to each individual, of which 37 samples from Naujaat were assigned to 34 individuals (10M:24F), and the 131 samples from SHI in 2014 were assigned to 127 individuals (76M:51F). None of these animals had previous detections in the greater Nunavut

dataset including samples from the 2004 harvesting season. Paetkau (2015) then used resampling in the software GeneClass2 to generate 10,000 simulated mainland and island genotypes, and plotted the distribution of the island/mainland likelihood ratio to produce critical values for statistical testing (Paetkau *et al.* 2004 *Mol. Ecol.*). By way of example, 99% of simulated island genotypes had a log likelihood ratio in excess of 9.

4.0 RESULTS AND DISCUSSION

4.1 Population Distribution

It is important to note that although an island population, some exchange of SHI caribou with the mainland likely occurred on a very small scale during rare winters when an ice bridge had formed across Roes Welcome Sound (Local Knowledge). According to island residents, during most winters, Roes Welcome sound does not freeze over completely creating an effective barrier to caribou movement. If such an ice bridge was to form however, this exchange would most likely have been with the Wager Bay population of caribou occupying the Lyon Inlet area due to its closer proximity to SHI (**Figure 10**). Though caribou tracks have been observed on the ice of Roes Welcome Sound in late winter in this same area (going both east to the island and west from the island), there has been no direct evidence of large-scale successful crossings prior to 2014.

Between 1968 and 1978, the first ten years of caribou occupancy on SHI following re-introduction, monitoring was mainly conducted using ground observations. During this period observations of caribou taken during patrols, whether by ground or by air, suggested caribou had spread extensively across the Island (**Figure 11**). Using Inverse Distance Weighting (IDW), it was determined that caribou were largely aggregated in the shoulder area east and northeast of Coral Harbour, the south shore of Bell Peninsula, and along the coast just south of the town of Coral Harbour, all areas still occupied by the herd to date (**Figure 12**). No animals were observed by hunters or aerial reconnaissance conducted in previous years anywhere further north or west of the areas indicated in **Figure 12**.

After the examination of all available observational data up to 1987, Heard and Gray (1987) concluded that there always appeared to be caribou in the core areas of Bell Peninsula and the Kirchoffer River uplands northeast of Coral Harbour prior to and since re-introduction (Heard and Grey, 1987; Heard and Ouellet, 1994). Observations made during their 1987 aerial population estimate showed an expansion of the herd further north and west, and throughout the coastal habitats of Bell Peninsula. Unfortunately, point data are not available from this survey for a more precise summary of caribou distribution. Observations of caribou distribution on SHI in June 1991 was similar to that recorded in 1987. Ouellet (1992) suggested that caribou range did not appear to expand between 1987 and 1991 even though their numbers increased substantially, suggesting that to accommodate growth, densities simply increased within the existing seasonal range. Once again, point data is not available for either of Ouellet's 1990 or 1991 surveys. All surveys conducted from 1997 to present have point data from which to base more detailed spatial assessments of distribution. Aerial survey results from June 1997 also showed little in the way of distributional change since Ouellet's observations in 1992 although caribou densities had continued to increase significantly. Densities and associated abundance estimates tracked during surveys are further displayed and substantiated using Inverse Distance Weighting (IDW) methods (**Table 3**) (**Figure 12**).

A continued assessment of distributional change following introduction was made using IDW methods on aerial survey point data (**Figures 12, 13, 14, 15, 16, and 17)**. These analyses suggests that caribou distribution increased across the Island from the point of introduction, up to the 1987 survey year at which time the now rapidly increasing population stopped expanding its range, suggesting that the herd had reached full occupancy of usable caribou habitat on the island. Caribou had occupied the southern portions of the island including Bell Peninsula and inland toward the central portions of the island along the Kirchoffer River watershed by as early as 1983. As observed by Ouellet (1992), densities increased over the same geographic areas from 1987 up until a period between 1998 and 2002, at which time densities over the same areas decreased

significantly. The most dramatic decrease in densities was on Bell Peninsula between the 2005 and 2007 survey years (**Figure 13 and 14**). According to survey density estimates, this declining trend on Bell Peninsula continued through to June 2015, and was also reported by local hunters. Caribou had all but abandoned the area, likely as a result of overgrazing from the previous years of higher relative densities. Today the central portion of the island remains the most highly used habitat by SHI caribou. Ecologically, this area is where the western flats meet the eastern highland, creating an ecotone between the Wager Bay Plateau and Southampton Island Plain Ecoregions. This area and its west south western exposed hills tend to melt out and expose vegetation in spring well before surrounding habitats.

Observed distributions from the 2003 survey indicated little change from distributions observed during previous surveys though localized densities had decreased. Overall, caribou continued to heavily use the central portions of the Island along the Kirchoffer River valley and along the transition between the western flats and the eastern highlands from November 1978 through to May 2023 (**Figures 12 to 17**). Small variations between years were likely the result in changes in snow cover and associated icing, with caribou feeding at higher elevations during years of early snow melt and/or reduced snow cover.

In summary, the central portion of the Island, in the vicinity of the Kirchoffer River and along the general transition from the western flats to the more topographically rugged eastern highlands of the Island, have received high use by SHI caribou in the spring and during calving from introduction to present, making these areas extremely important to the long-term viability of the herd. The protection of these areas from anthropogenic disturbance that may modify and/or impact the landscape will be a critical component of any long-term herd management plans.



Figure 10. Likely routes used by both emigrating and immigrating caribou both off of, and onto Southampton Island (based on hunter reports of tracks across sea ice). Most notable track observations were in winter 2014.

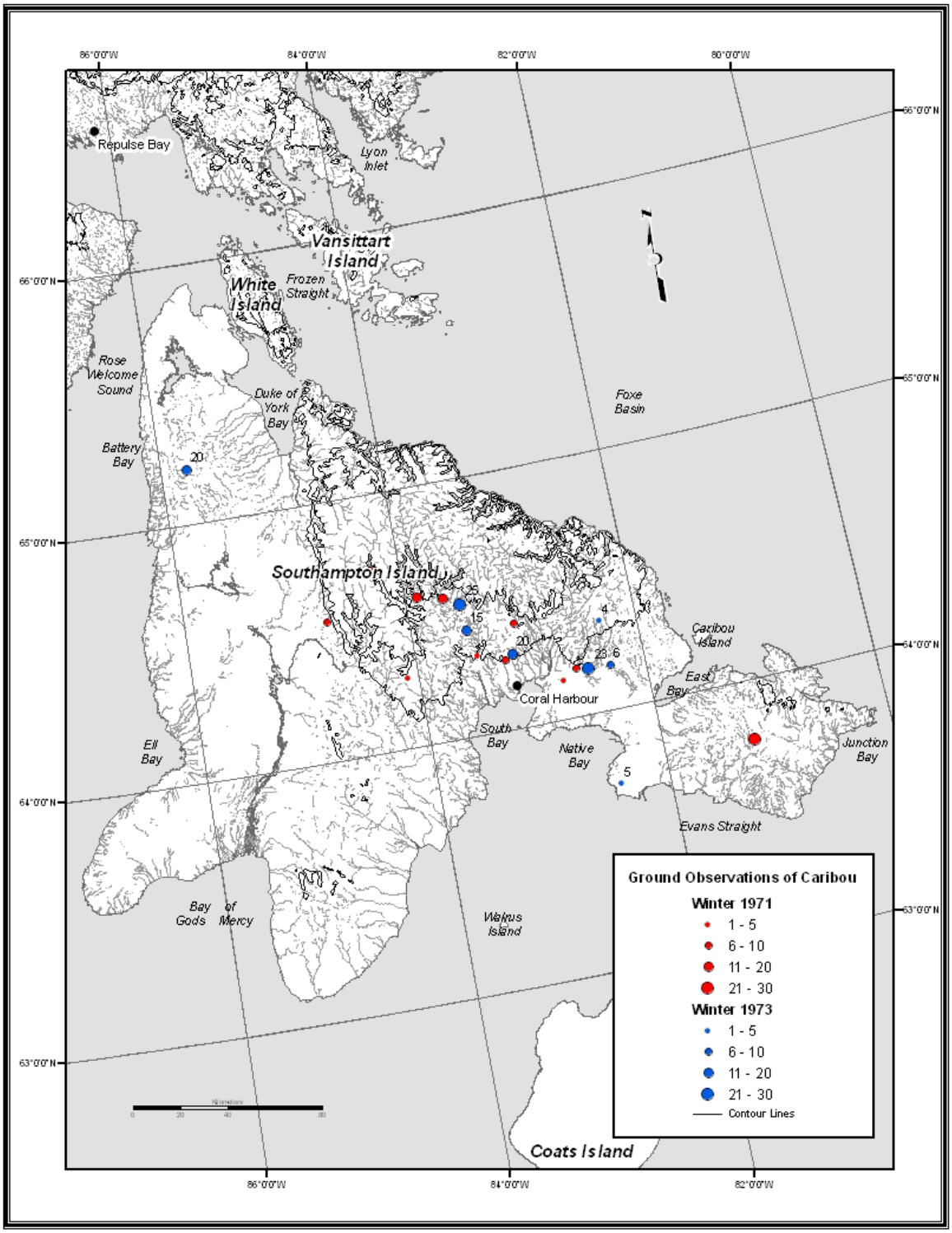


Figure 10. Observations of caribou on Southampton Island (1971 to 1973). Note that group sizes and locations were provided in 1973 reports, but not in 1971.

Table 3. Inverse distance weighting (IDW) values for the entire Southampton Island study area including White Island and Bell Peninsula showing changes in adult caribou density through time. Results from June 2005 were removed as White Island was not surveyed in that year.

Caribou/km²	Percent Area By Class										
	1978	**1997**	**2003**	**2007**	**2009**	**2011**	**2013**	**2015**	**2017**	**2019**	**2023**
0-1	98.2	69.8	72.6	80.8	83.3	88.9	89.2	85.8	82.0	86.7	85.2
1-2	0.9	11.0	12.0	9.2	8.9	7.0	4.0	5.2	6.7	5.2	6.1
2-5	0.6	11.5	12.2	8.2	6.9	3.7	4.8	7.1	8.4	5.7	6.2
5-8	0.2	3.7	2.5	1.5	0.7	0.3	1.4	1.4	2.2	1.5	1.6
>8	0.1	4.0	0.7	0.4	0.2	0.0	0.6	0.5	0.7	0.8	0.9

Caribou/km²	Percent Change										
	1978 to 1997	**1997 to 2003**	**2003 to 2007**	**2007 to 2009**	**2009 to 2011**	**2011 to 2013**	**2013 to 2015**	**2015 to 2017**	**2017 to 2019**	**2019 to 2023**	**1997 to 2023**
0-1	-28.4	2.8	8.2	2.5	5.6	0.3	-3.4	-3.8	4.7	-1.5	15.4
1-2	10.1	1.0	-2.8	-0.3	-1.9	-3.0	1.2	1.5	-1.5	0.9	-4.9
2-5	10.9	0.7	-4.0	-1.3	-3.2	1.1	2.3	1.3	-2.7	0.5	-5.3
5-8	3.5	-1.2	-1.0	-0.8	-0.4	1.1	0.0	0.8	-0.7	0.1	-2.1
>8	3.9	-3.3	-0.3	-0.2	-0.2	0.6	-0.1	0.2	0.1	0.1	-3.1
Net Change > 1 caribou/km²	28.4%	-2.8%	-8.1%	-2.6%	-5.7%	-0.2%	3.4%	3.8%	-4.8%	1.6%	-15.4%

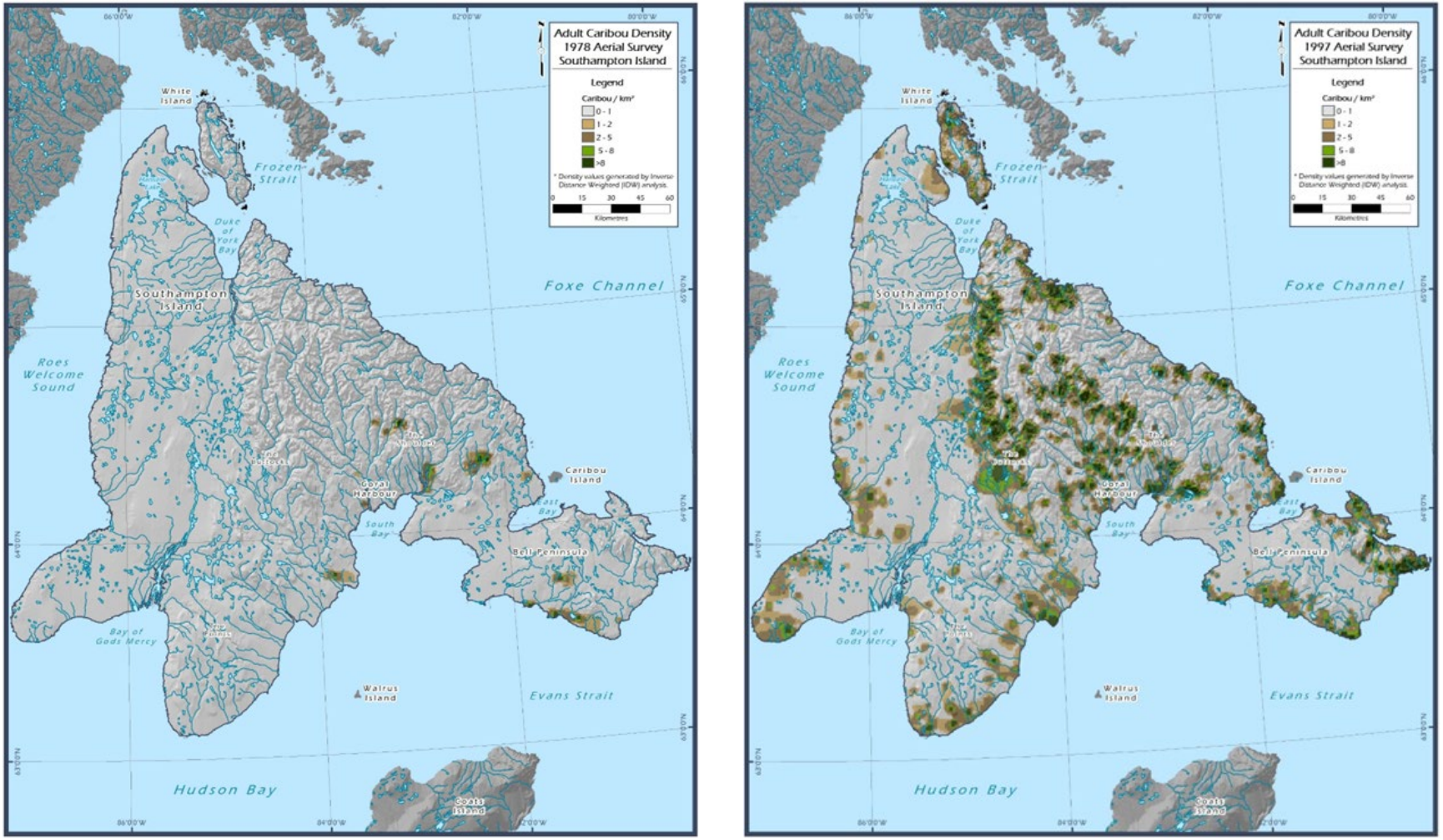


Figure 11. Results of the inverse distance weighted (IDW) interpolation technique applied to the November 1978 and June 1997 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

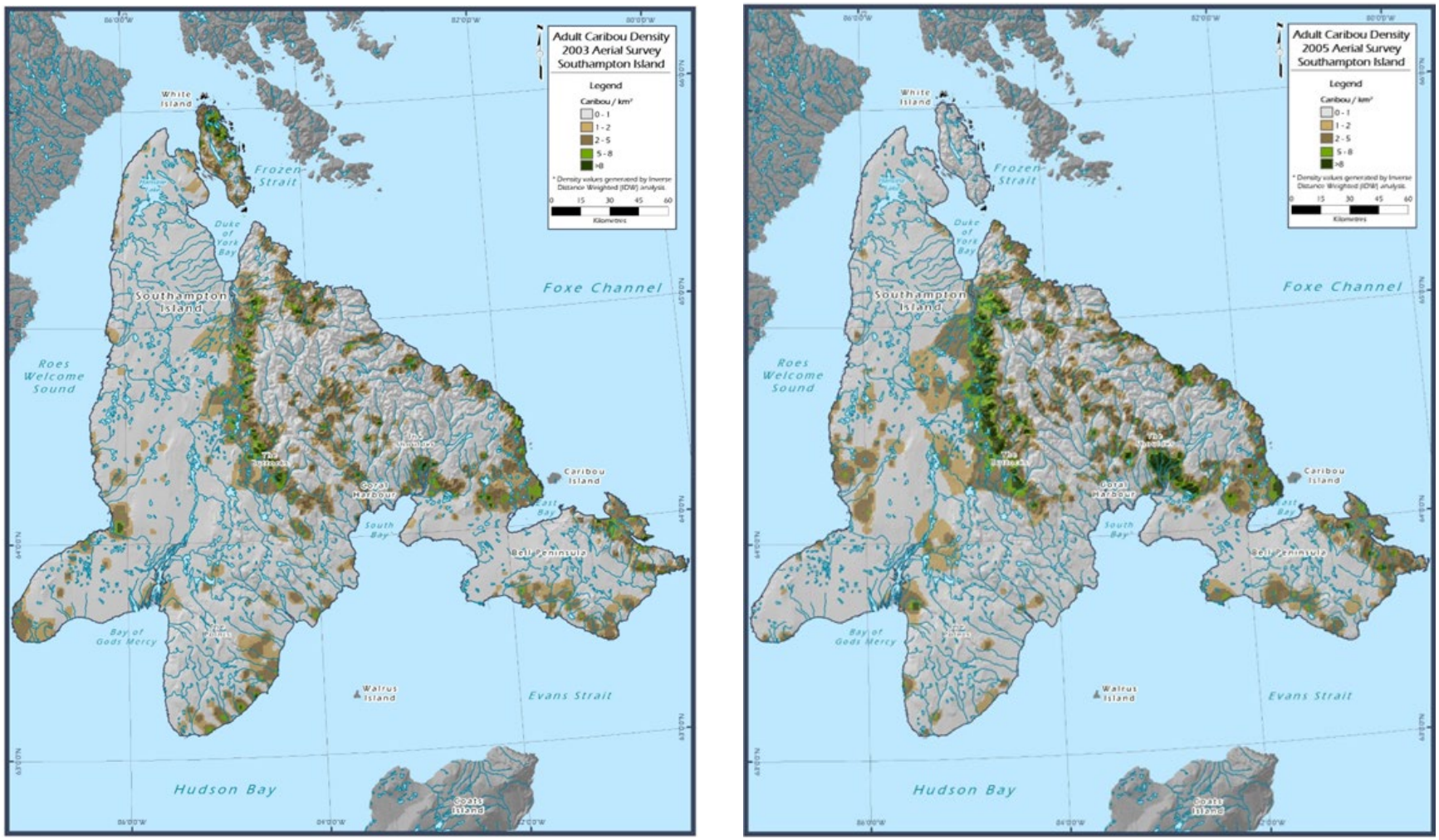


Figure 12. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2003 and June 2005 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

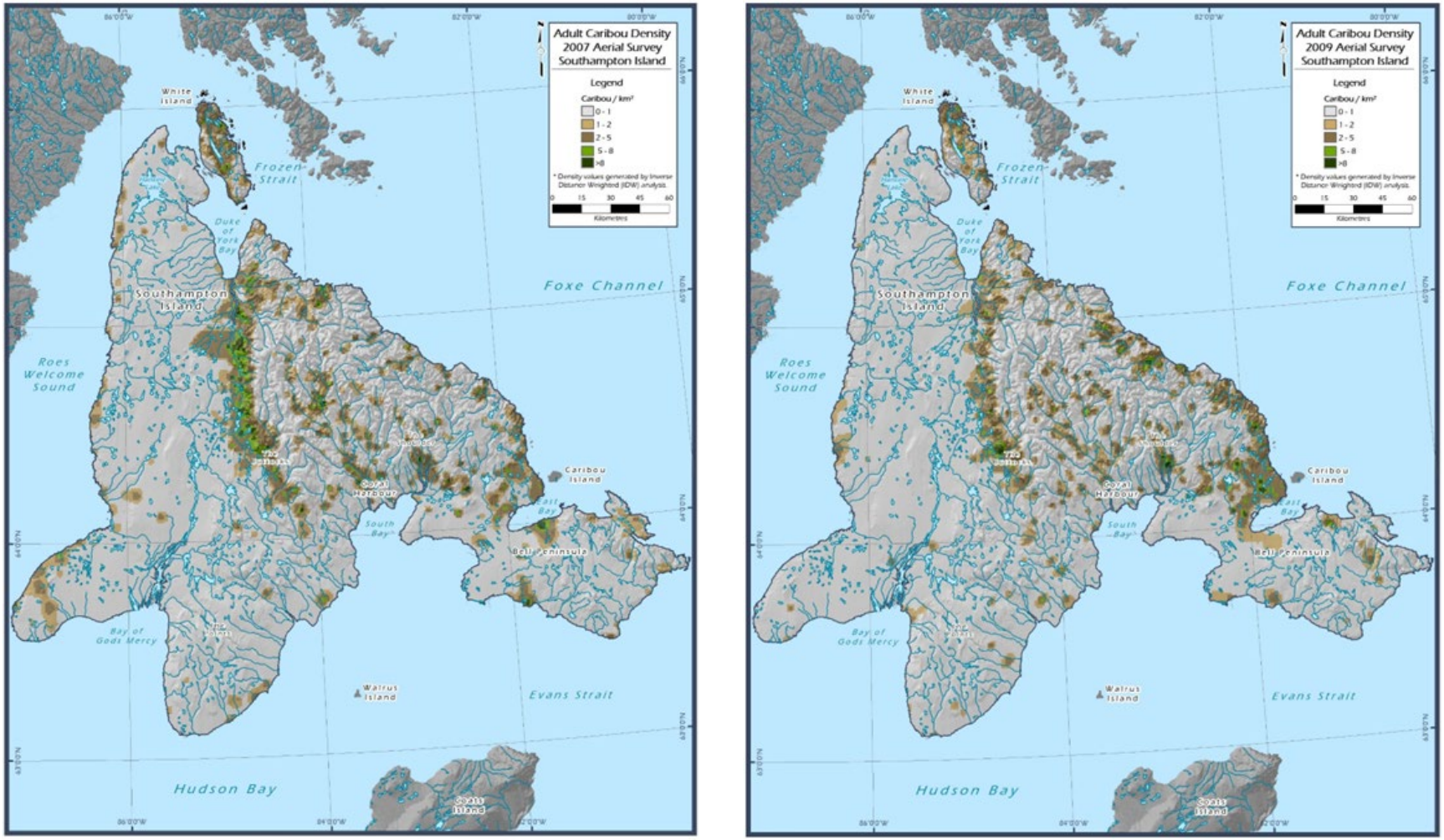


Figure 13. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2007 and June 2009 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

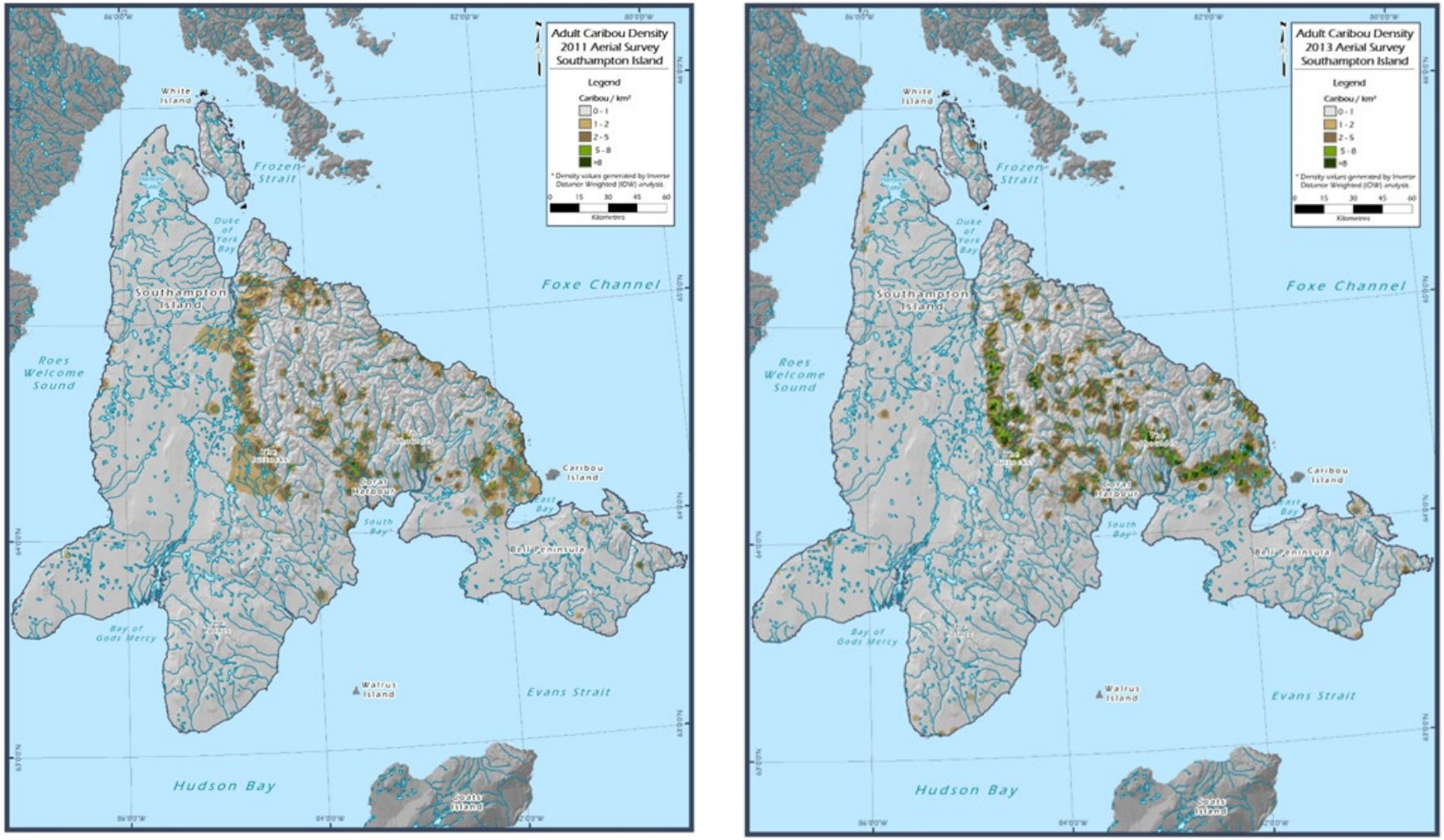


Figure 14. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2011 and June 2013 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

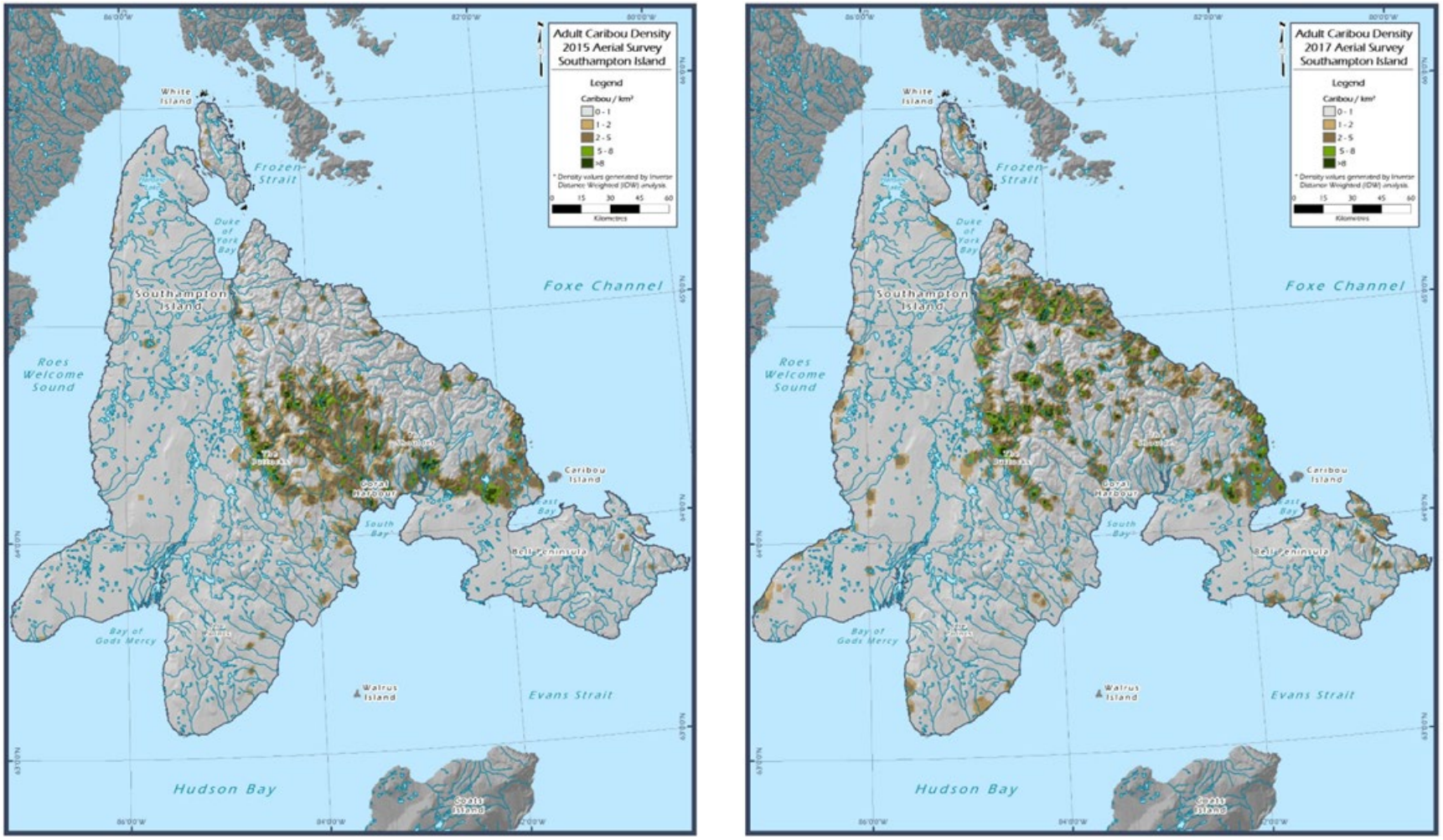


Figure 15. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2015 and May 2017 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

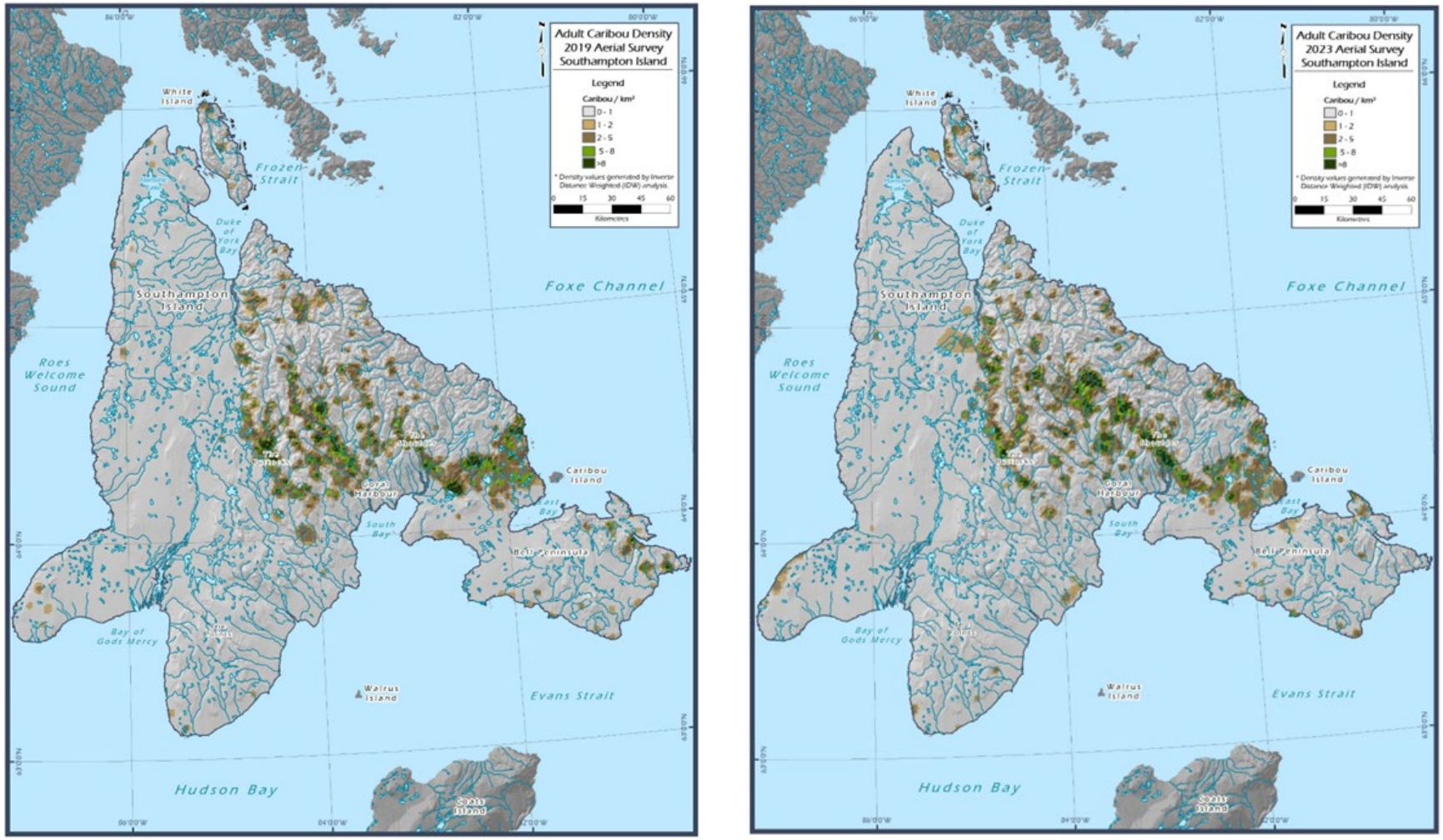


Figure 16. Results of the inverse distance weighted (IDW) interpolation technique applied to the May 2019 and May 2023 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

4.2 Strip Transect Surveys

Population estimates from 1997 to 2023 were reasonably precise, with Coefficients of Variation (CVs) of less than 10% in all years (**Table 4, Figure 18**). **Figure 18** is a visual representation of **Table 4** while strata-specific estimates are shown in **Figure 19**. Coats Island is included in **Figure 19** based on the Coral Harbour Hunters and Trappers organizations (HTO) requests to include, however, it is important to note that it was not included in overall Southampton Island estimates. Population declines occurred in all strata from 1997 to 2013, started increasing between 2013 and 2015, then remained relatively stable between 2015 and 2023. The use of different scales on the graph in **Figure 19** aids in interpretation of stratum-specific trends but it is also misleading in terms of the relative abundance of caribou in each stratum. For this reason, the same estimates of caribou numbers are plotted on the same scale (**Figure 20**), clearly indicating that the majority of caribou on SHI occurred on the High Eastern SHI strata in all years.

Table 4. Strip transect estimates of caribou on Southampton Island, showing the number of strata sampled each year, the number of caribou counted on transect, and population estimates with descriptive statistics (SE = standard error, CV = coefficient of variation) are given for each year of surveys from 1997 through 2023.

Year	Strata	Caribou	Strip transect estimates				
	sampled	Counted	N	SE	Confidence Limits		CV
1997	7	5777	29,425	1622.5	26,375	32,827	5.5%
2003	7	3833	18,479	1099.8	16,420	20,797	6.0%
2005	6	4079	21,227	1701.8	18,098	24,896	8.0%
2007	7	2689	14,389	914.6	12,684	16,325	6.4%
2009	6	2521	13,651	833.1	12,091	15,412	6.1%
2011	7	1667	7,937	580.4	6,861	9,182	7.3%
2013	7	1597	7,284	525.3	6,307	8,413	7.2%
2015	7	3068	12,319	931.6	10,591	14,328	7.6%
2017	7	1685	8,436	680.8	7,184	9,906	8.1%
2019	7	2551	11,944	1080.8	9,972	14,305	9.1%
2023	7	2610	12,565	801.9	11,066	14,266	6.4%

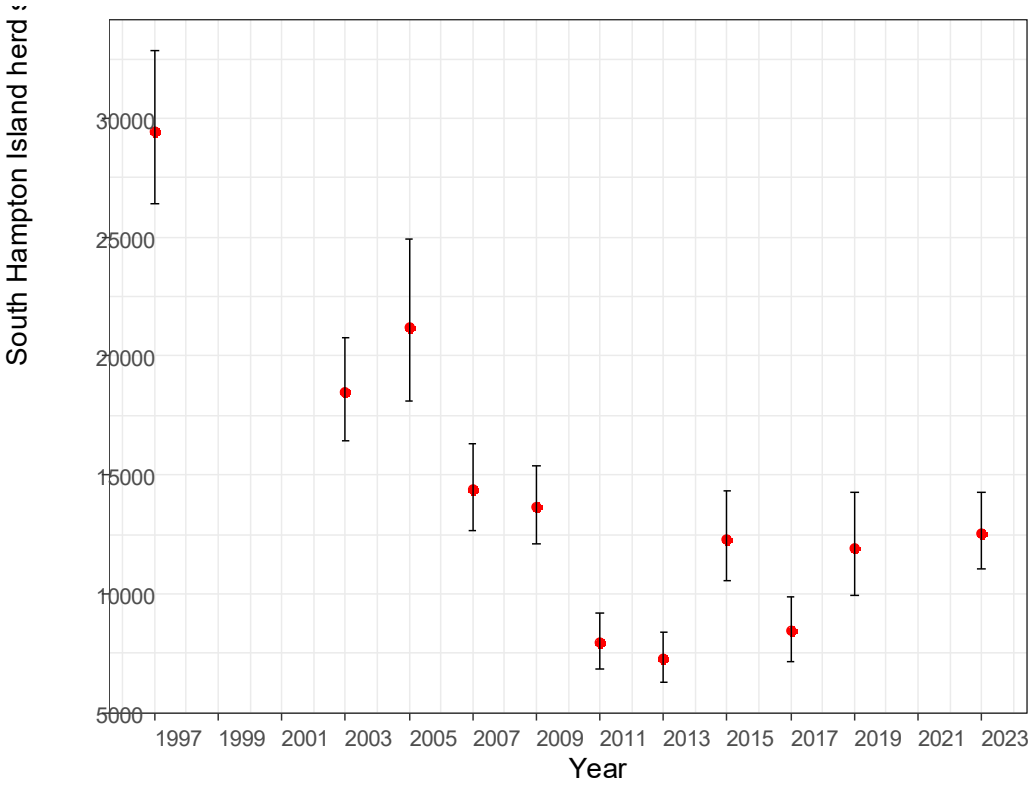


Figure 17. Population abundance estimates of the Southampton Island caribou herd using a strip transect estimator, according to strata listed in Table 4.

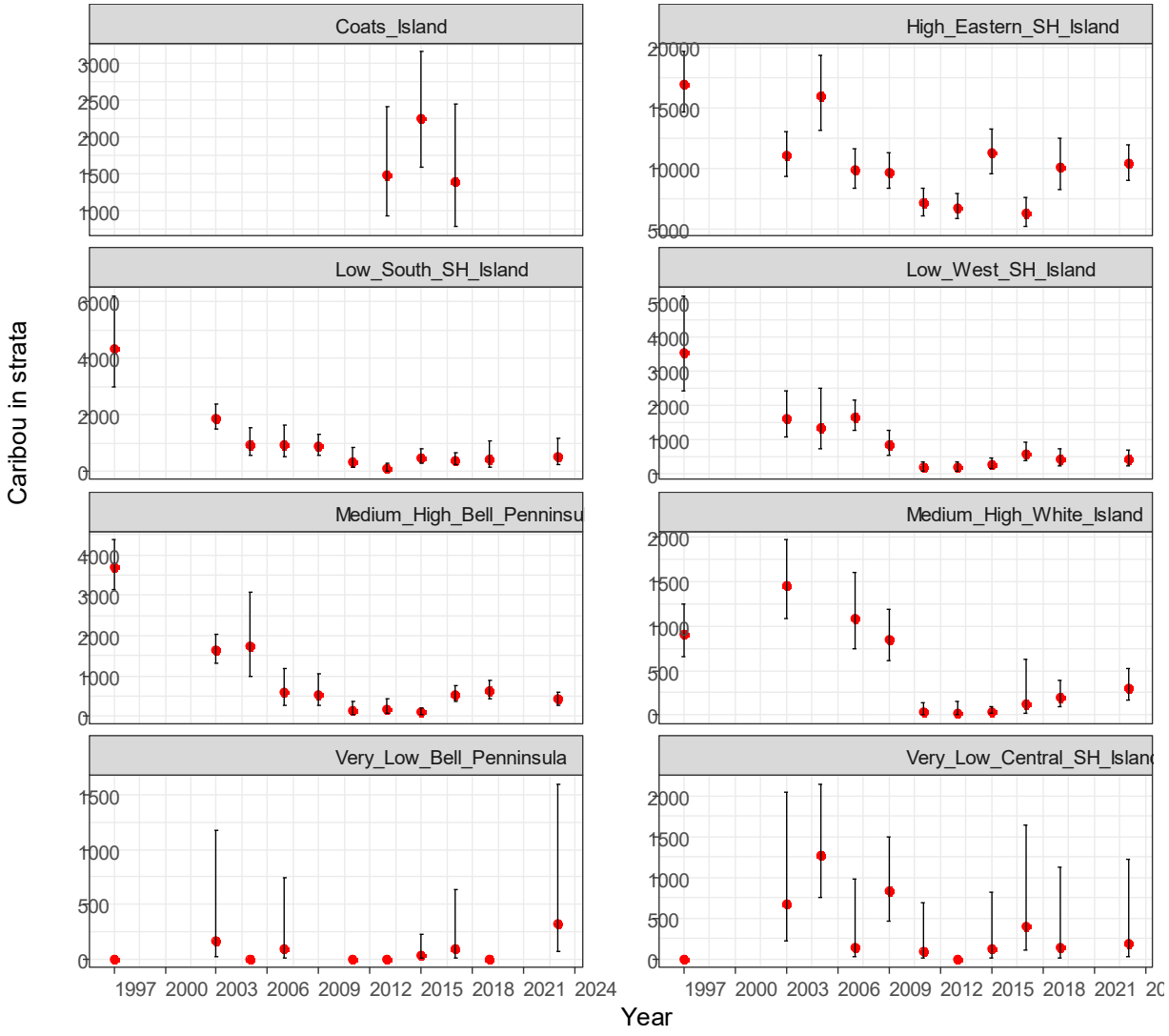


Figure 18. Strata-specific estimates of strata sampled using a strip transect estimator. Note that the y-scales are different for each graph.

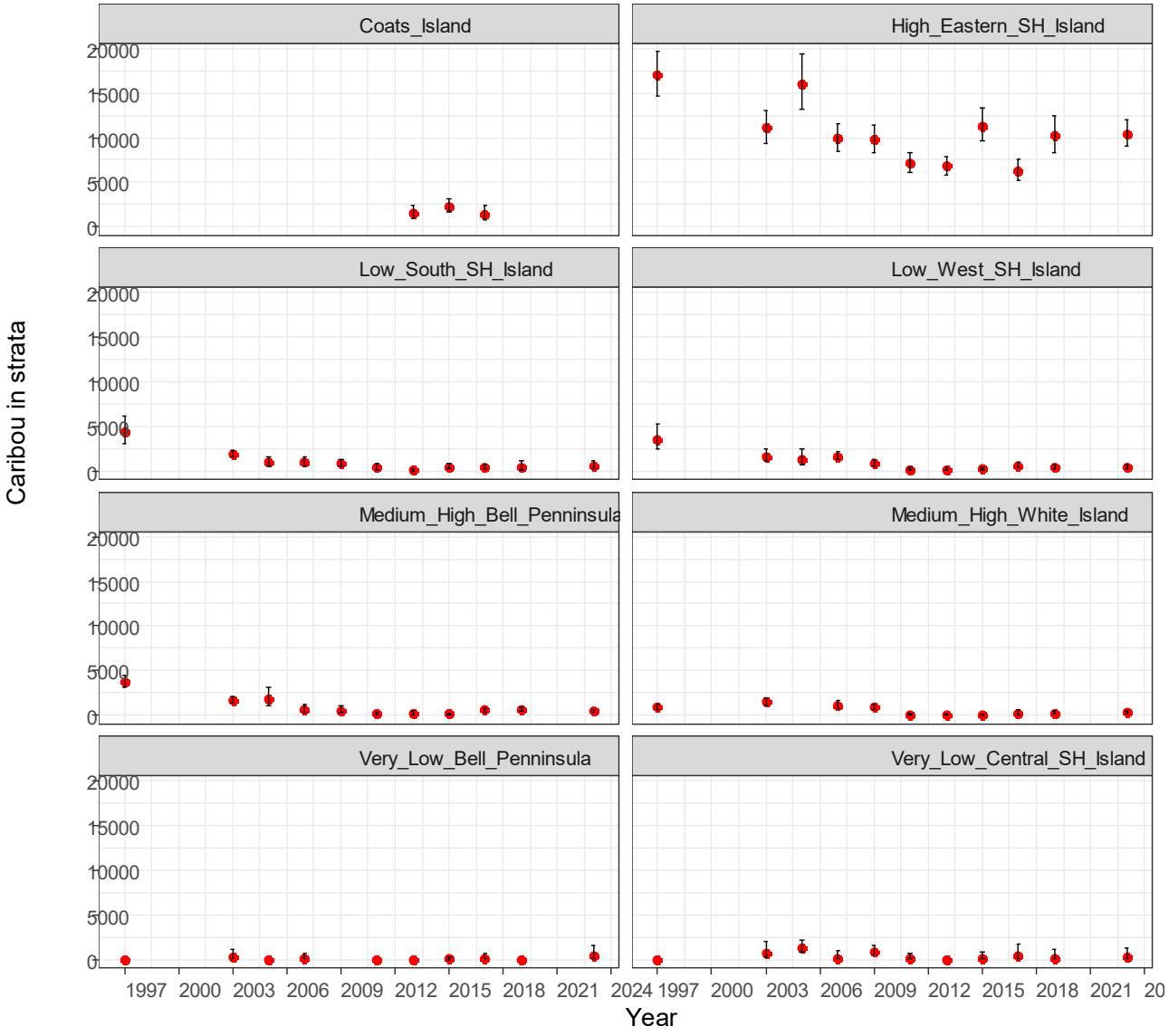


Figure 20. Strata-specific estimates of strata sampled using a strip transect estimator with the same scale used on each graph.

4.2.1 Dependent double observer analyses (2011-2015,2019, 2023)

Dependent double observer pair data were collected using fixed-wing aerial surveys in 2011, 2013, 2015, 2019, and 2023. In 2017, we used binned distance markers on wing struts to allow for distance sampling methods, as described in the methods section of this report. Survey conditions, group sizes, and observer efficiency varied between each survey year. These data were explored graphically to help assess dominant forms of variation prior to identifying a statistical model for population estimates derived from the dependent double observer pair method. The distribution of group sizes was relatively similar during each survey year with larger groups observed in 2015 (**Figure 21**).

In general, smaller group sizes were more likely to be seen only by a single observer. Observers were placed into 19 pair combinations of which 11 observer pairs switched between primary and secondary roles, and 8 did not. The assumption of the dependent double observer pair method is that the two same-side observers have similar sighting probabilities and therefore, estimates may be biased when observers do not switch places during the survey. The sighting probability of pairs varied between observers for some pairs (i.e. in particular for pair 7) showing a higher relative frequency of only one observer seeing a group of caribou (**Figure 22**).

Between 1997 and 2011, all surveys were flown in early June close to or during the onset of spring melt. From 2013 to present, survey deployment was changed to early to mid-May (see methods) though no detectable variation in relative densities and their related strata were found. Regardless, snow cover varied each survey year with 2011 having a full range of snow cover and other years showing primarily high elevation snow cover, particularly 2013 and on, following the change in survey timing (approximately 20 days earlier) to May (**Figure 23**). Cloud cover also varied for each year (**Figure 24**), with no discernable patterns. Sighting probabilities were lower in 2011 as shown by higher frequencies of single observer sightings (**Figure 25**).

Dependent double observer pair model selection, performed by sequentially calculating differences in Akaike's information criterion (AICc), suggested that sighting probabilities varied according to a combination of observer, year, size of caribou groups observed, and cloud categorized in 25% intervals, and an interaction of snow cover and group size (**Table 5**). Two models were supported by differences in AICc values of less than 2. The support for year as a sightability term suggested that there were year-specific factors affecting sightability that were not accounted for by other covariates. Observer pairs in the analyses were reduced to the main pairs that exhibited lower sighting probabilities, given that a model with all observer pairs parameterized did not converge. Using this strategy, the main observer pairs that displayed lower or higher probabilities were accounted for with other observer pairs set to a mean value. The predictions of the most supported model (model 1 in **Table 5**) are shown graphically, demonstrating that sightability was lower in only 2011 and 2019, for both observer pairings and as a function of cloud and snow cover (**Figure 26**).

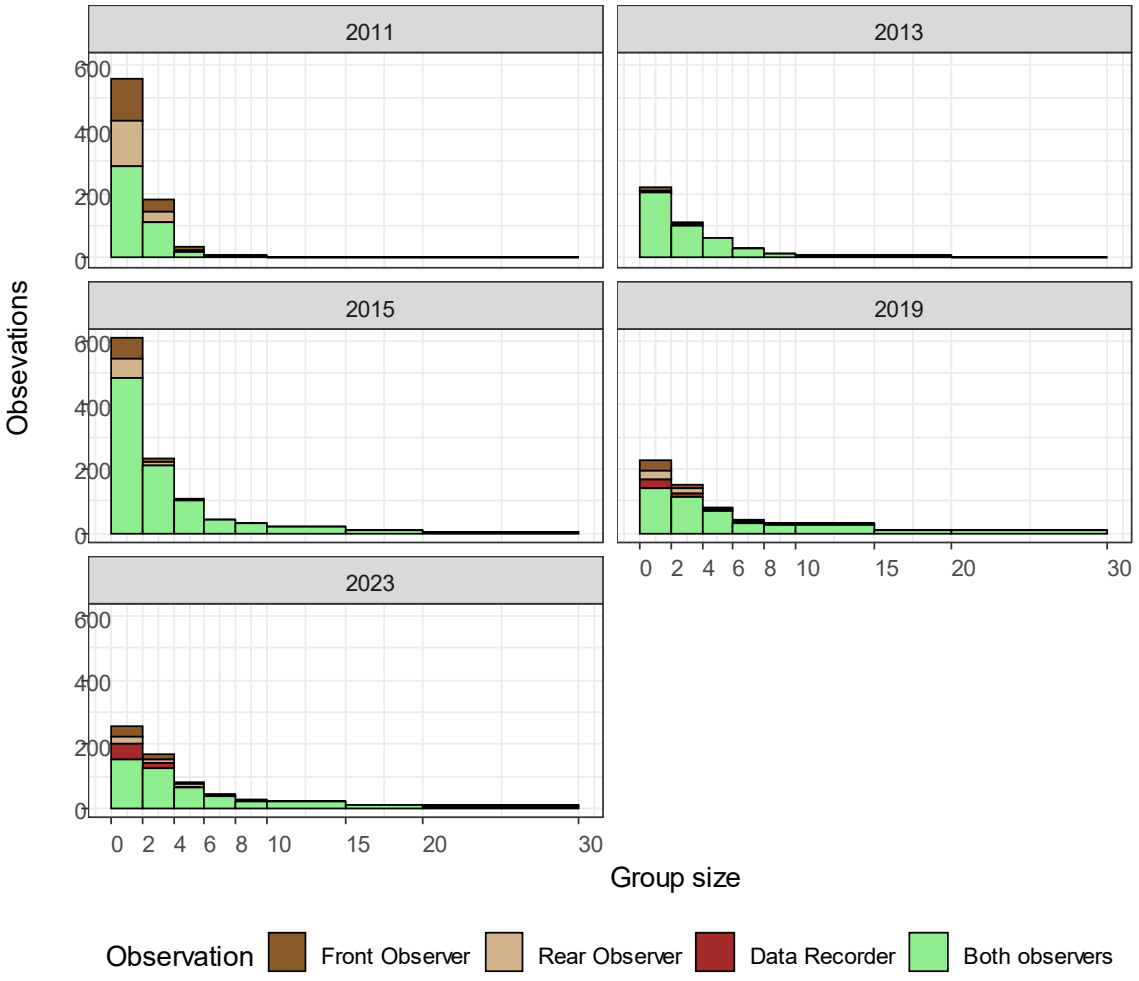


Figure 21. Group sizes of caribou observed each year for surveys conducted in 2011, 2013, 2015, 2019, and 2023, with frequency of sightings made by front, rear, data recorder and both observers.

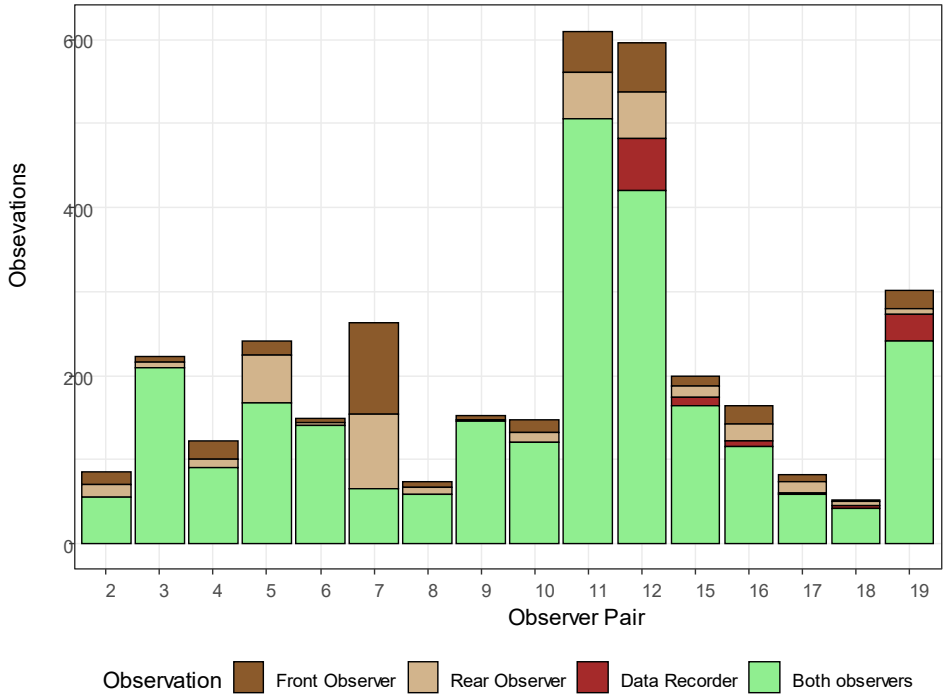


Figure 19. Observer pairings with frequencies of sighting by front, rear, data recorder, and both observers (2011-2023).

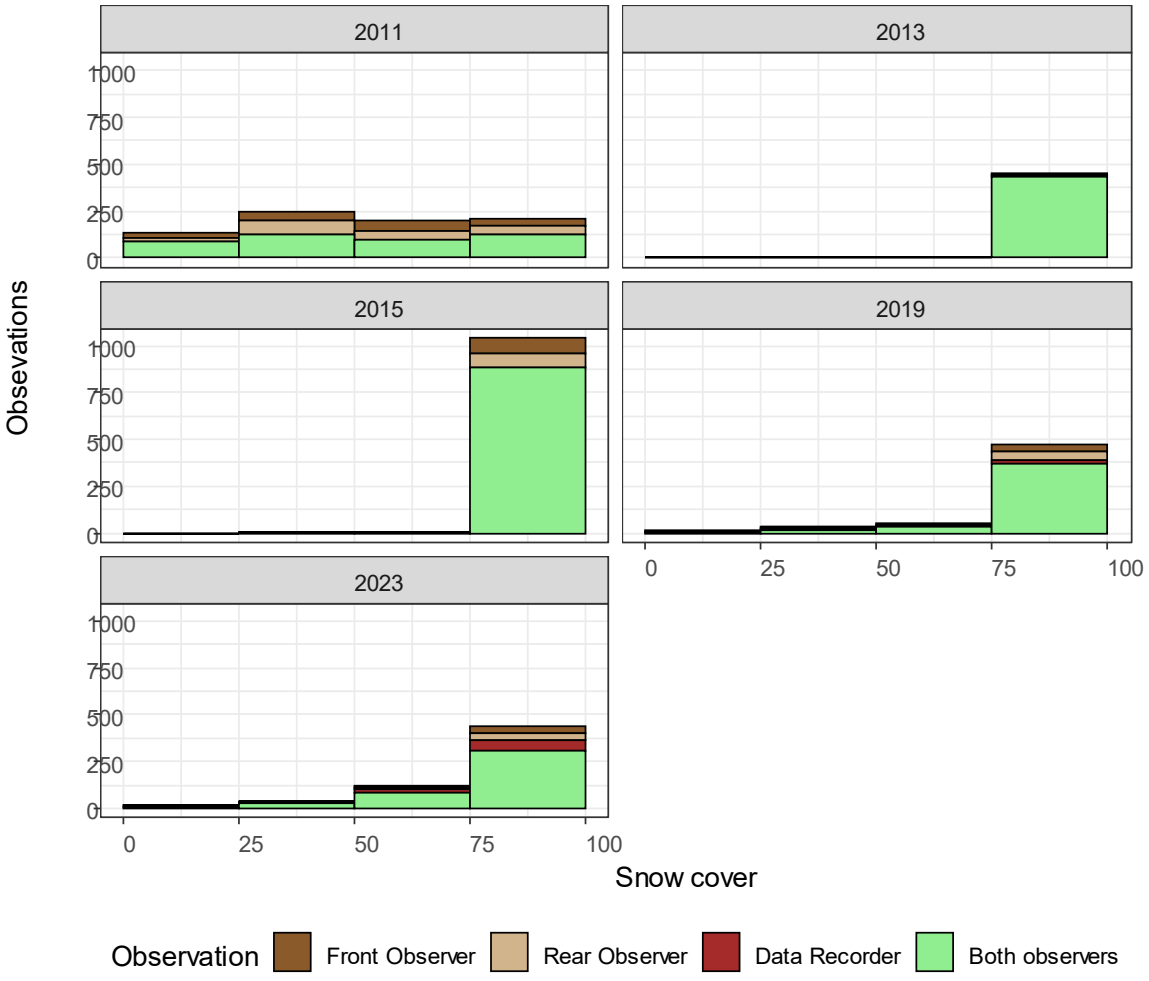


Figure 20. Snow cover during each year of the survey with frequencies of sightings by front, rear, data recorders, and both observers (2011-2023).

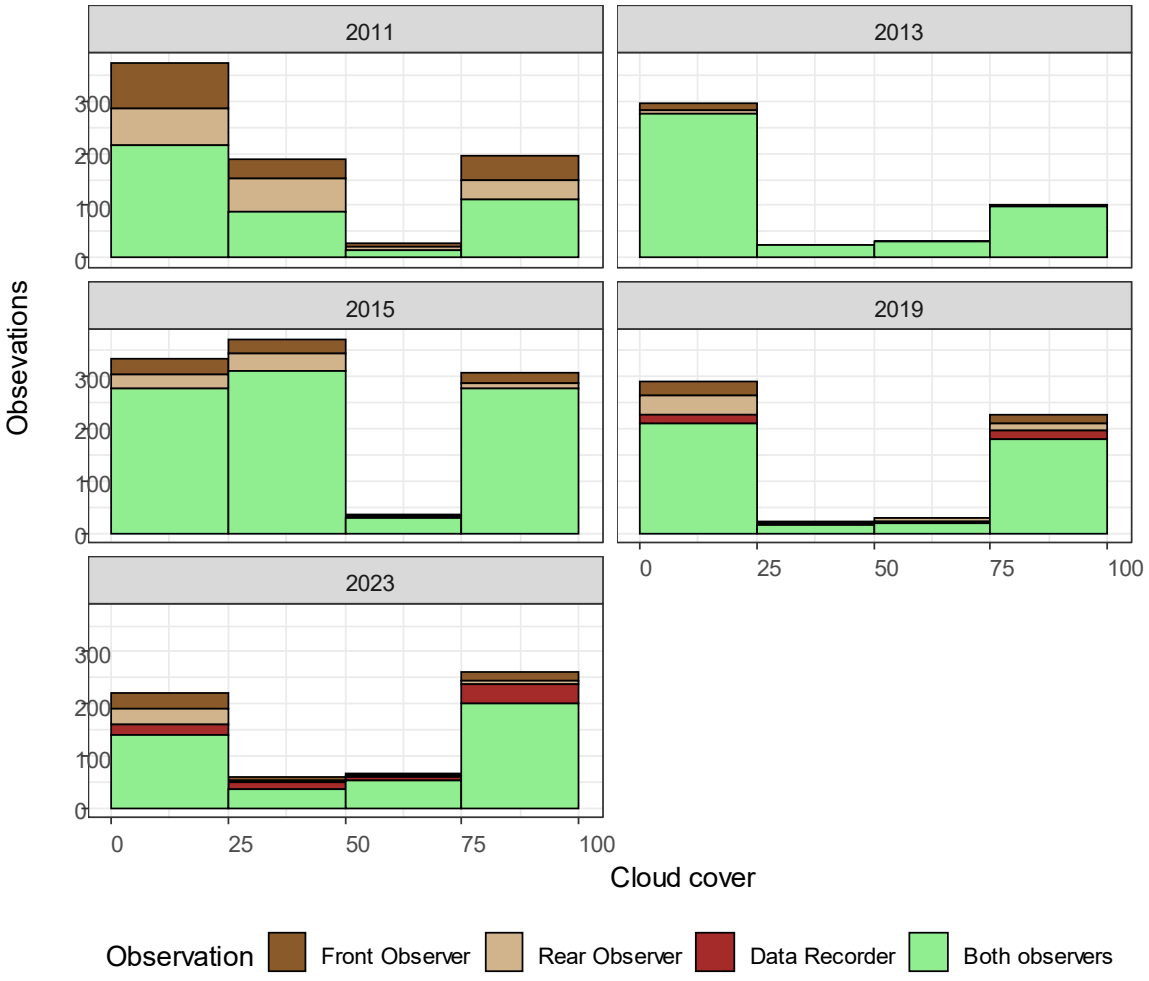


Figure 21. Cloud cover during each year of the survey with frequencies of sightings by front, rear, data recorders and both observers (2011-2023).

Table 5. Dependent double observer model selection results (2011-23). Sample size adjusted Akaike Information Criterion (AICc), the difference in AICc between the most supported model. For each model (ΔAIC_c), AIC_c weight (w_i), number of model parameters (K), and deviance is given. See **Table 2** for covariate definitions

No	Model	AICc	ΔAIC_c	w_i	K	LL
1	observers (reduced) +Year + size + cloud +snowc*size	1916.82	0.00	0.47	16	-942.3
2	observers (reduced) +Year + size + cloud	1917.00	0.19	0.43	15	-943.4
3	observers (reduced) +Year + size + cloud+Year*size	1919.93	3.11	0.10	18	-941.8
4	Year + size + cloud + snow	1946.30	29.48	0.00	11	-962.1
5	observers (all)	2007.52	90.70	0.00	15	-988.7
6	size + snow + cloud	2018.84	102.02	0.00	4	-1005.4
7	YearF	2020.53	103.72	0.00	8	-1002.2
8	snow + cloud	2049.87	133.06	0.00	7	-1017.9
9	snow	2060.00	143.18	0.00	4	-1026.0
10	snowc + cloudc + snowc * cloudc	2092.56	175.74	0.00	4	-1042.3
11	size	2132.15	215.33	0.00	2	-1064.1
12	logsize	2138.00	221.18	0.00	2	-1067.0
13	cloud_factor	2166.28	249.46	0.00	4	-1079.1
14	constant	2180.59	263.78	0.00	1	-1089.3

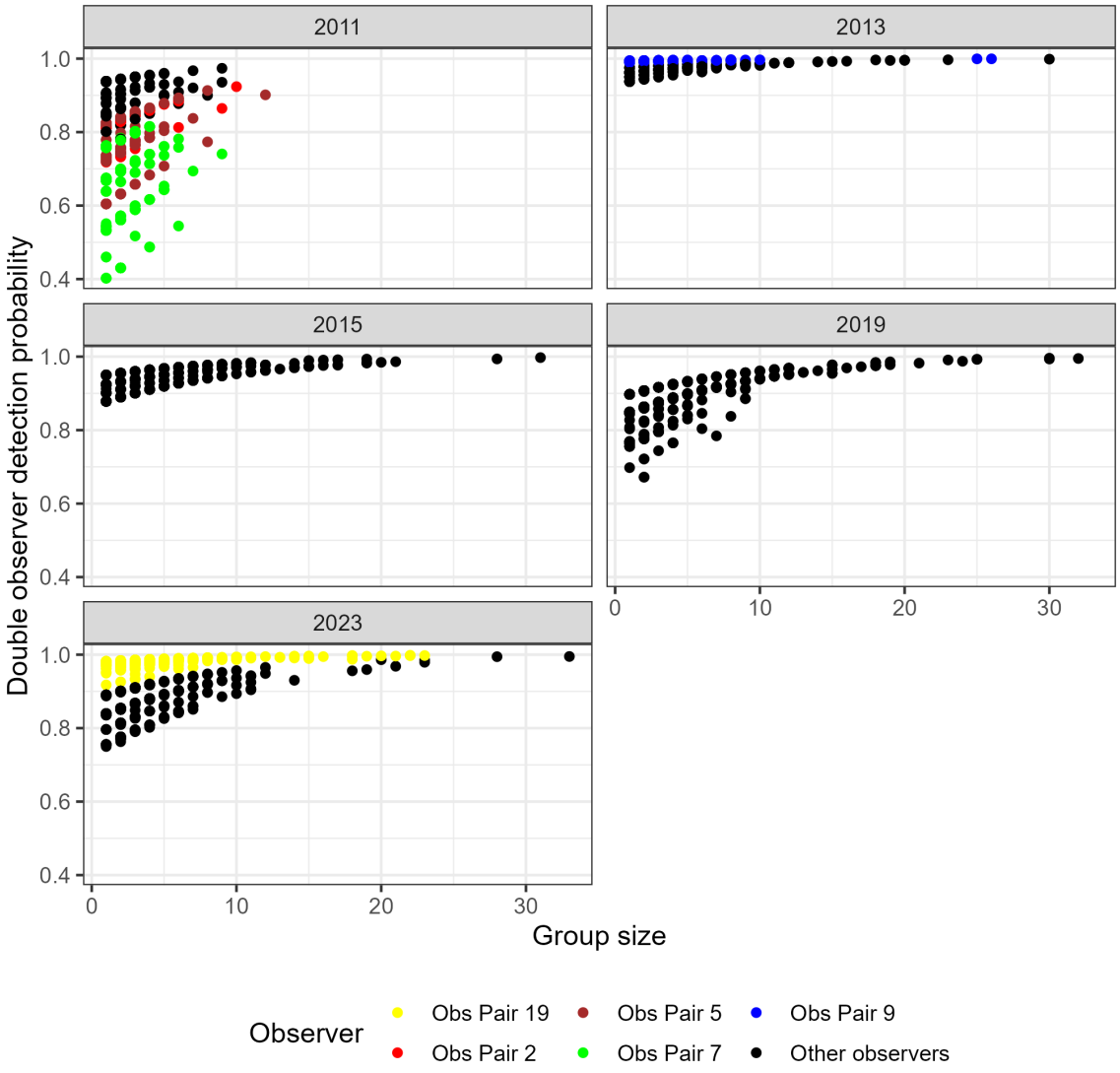


Figure 22. Dependent double observer detection probabilities as a function of year, group size, for selected observer pairs (2011-2023).

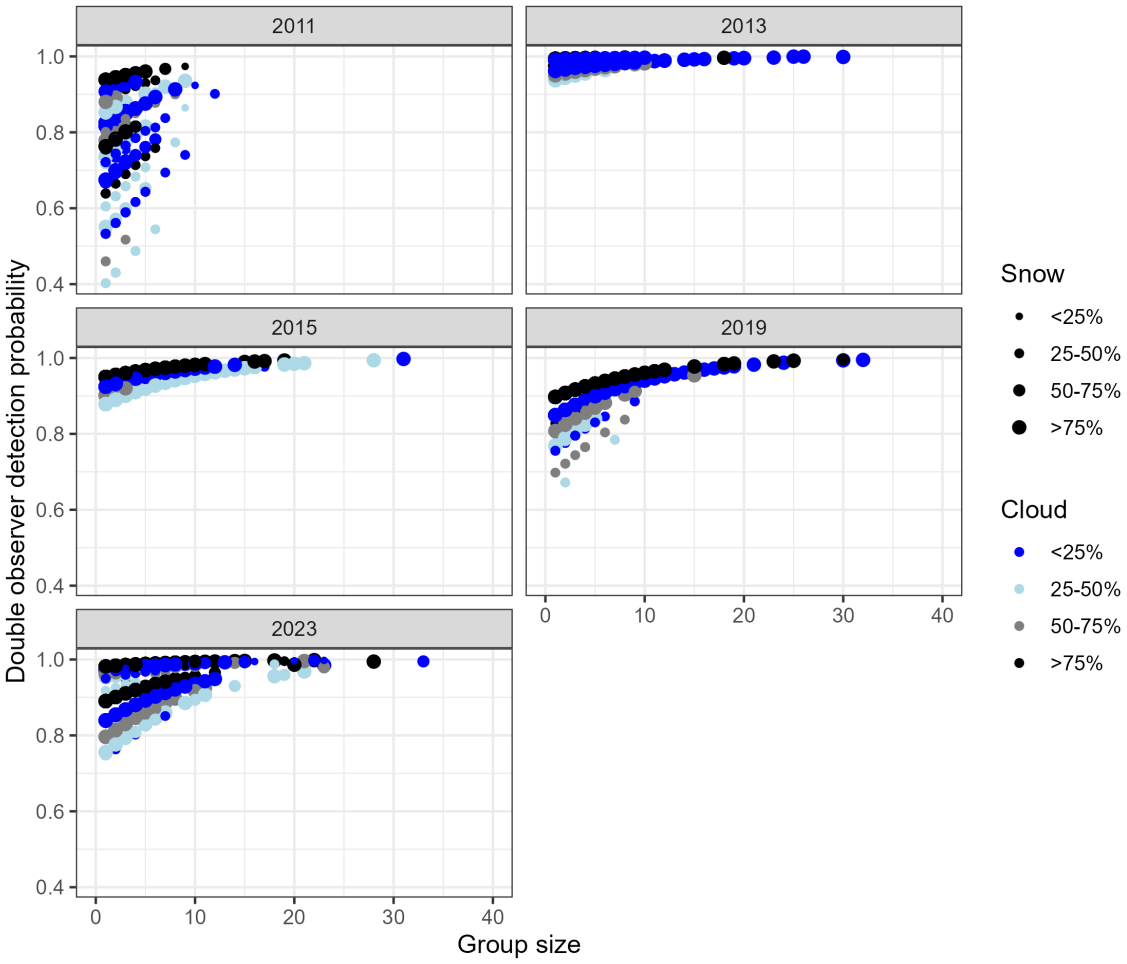


Figure 23. Dependent double observer probabilities as a function of year, group size, snow and cloud cover (2011-2023).

4.2.2 Distance sampling/double observer pair sampling in 2017

During the 2017 survey, frequencies of observations by distance bins revealed different detection probability curves between observer pairs 1 and 2. Observer pair 1 had a higher frequency of observations near the aircraft whereas observer pair 2 had a higher frequency away from the plane. Compared to previous years of dependent double observer pair sampling, there was a higher frequency of observations from data recorders in 2017, suggesting a higher level of observation experience by the data recorders. To utilize these recorder observations, we categorized them as single observer observations and assumed that the data recorder had similar sighting probabilities to the other observers (**Figure 27**). Snow cover was greater than 50% in the area of most observations, and the results of the 2017 observations suggested that sightability was lower when snow cover conditions were below 50% (**Figure 28**).

Model selection was proceeded by building distance sampling models with the mark-recapture model parameters held constant, and by initially comparing half normal and hazard rate models. Of these, the hazard rate model was the most supported, with observer pair and snow (continuous cover) as covariates. Once this model was selected, dependent double observer pair mark-recapture models were compared with observer pair and snow, as well as the most supported covariates. Group size (log transformed) was also supported as a distance sampling covariate (Model 1, **Table 6**). Goodness-of-fit for model 1 was marginal (chi-square=19.8, df=7, p=0.006), however most of the lack of fit came from the 600-1,000-meter distance bin which would have less influence on estimates given low observation frequency rates in this bin (**Figure 29**). An additional analysis was conducted, which used the first 2 distance bins of data to fit dependent double observer pair only models to the data, without the distance component (**Table 7**). The same suite of dependent double observer pair models was applied to the data set as used in previous years analysis and as listed in **Table 6**. According to this

subsequent analysis, a model with observer, snow (continuous) and the log of group size was most supported. Population abundance estimates from this model were thus compared to the distance sampling and strip transect estimates from other survey years.

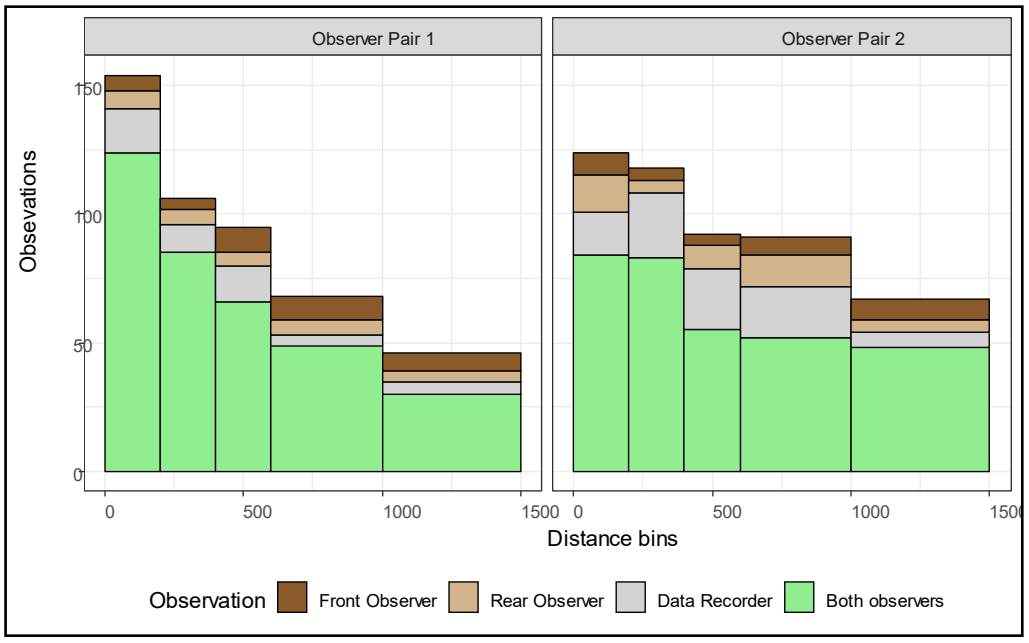


Figure 24. Frequencies of observations by distance bin for the 2 observer pairs in May 2017.

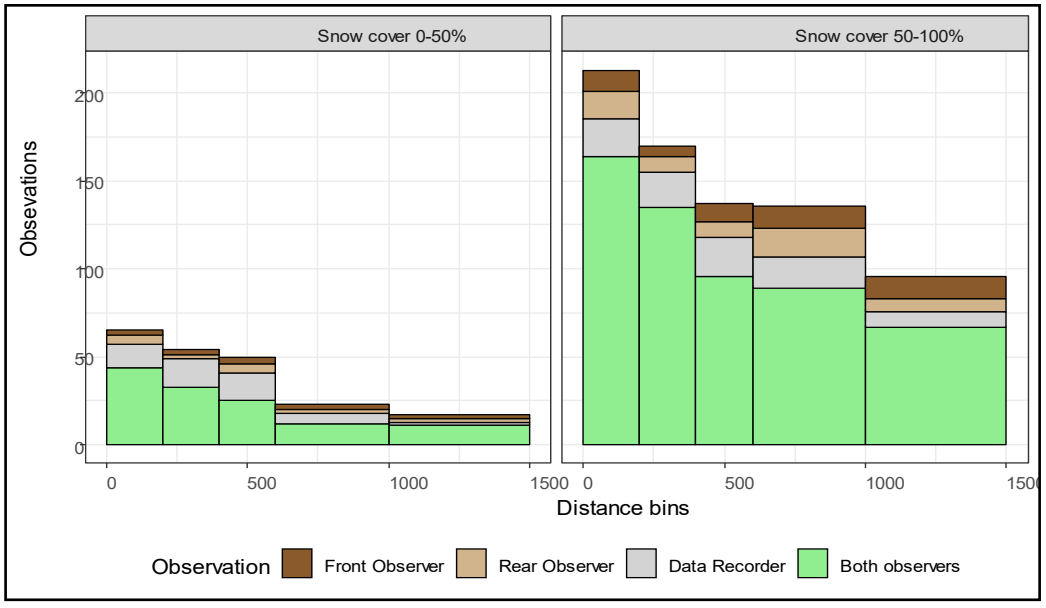


Figure 25. Frequencies of observations by distance bin for 2 levels of snow cover in May 2017.

Table 6. Dependent double observer model selection results for May 2017. Sample size adjusted Akaike Information Criterion (AICc), the difference in AICc between the most supported model for each model (ΔAICc), AICc weight (wi), number of model parameters (K) and deviance is given. See **Table 2** for covariate definitions.

		Distance sampling	2x observer	Model fit				
No.	**DF**	**Distance covariates**	**covariates**	AIC_c	ΔAIC_c	w_i	K	LL
		Distance /Double observer models						
1	HR	obs+snowc+log(size)	obs+snowc	4000.0	0.00	0.95	8	-1992.
2	HR	obs+snowc+log(size)	obs+snow	4006.6	6.59	0.04	10	-1993.
3	HR	obss+snowc+size	obs+snow	4009.3	9.24	0.01	10	-1994.
4	HR	obs+snowc	obs+snow	4010.8	10.75	0.00	9	-1996.
5	HR	obs+log(size)	obs+snowc	4012.7	12.66	0.00	7	-1999.
6	HN	obss+snowc+logsize	obs+snow	4019.2	19.16	0.00	9	-2000.
7	HN	obs+snowc	obs+snow	4019.8	19.71	0.00	8	-2001.
8	HR	obss+snowc	obs	4020.1	20.04	0.00	6	-2004.
9	HR	obss+snowc	size	4028.0	27.94	0.00	6	-2008.
		Distance sampling models						
10	HR	obss+snowc	constant	4031.2	31.14	0.00	5	-2010.
11	HN	obss+snowc	constant	4040.2	40.11	0.00	4	-2016.
12	HN	obs+snowc+size	constant	4040.2	40.15	0.00	5	-2015.
13	HR	obs	constant	4041.7	41.66	0.00	4	-2016.
14	HN	snowc	constant	4045.3	45.28	0.00	3	-2019.
15	HR	size	constant	4047.0	46.96	0.00	4	-2019.
16	HR	constant	constant	4047.3	47.27	0.00	3	-2020.
17	HN	obs	constant	4061.1	61.09	0.00	3	-2027.
18	HN	constant	constant	4067.3	67.22	0.00	2	-2031.
19	HN	size	constant	4067.9	67.89	0.00	3	-2031.

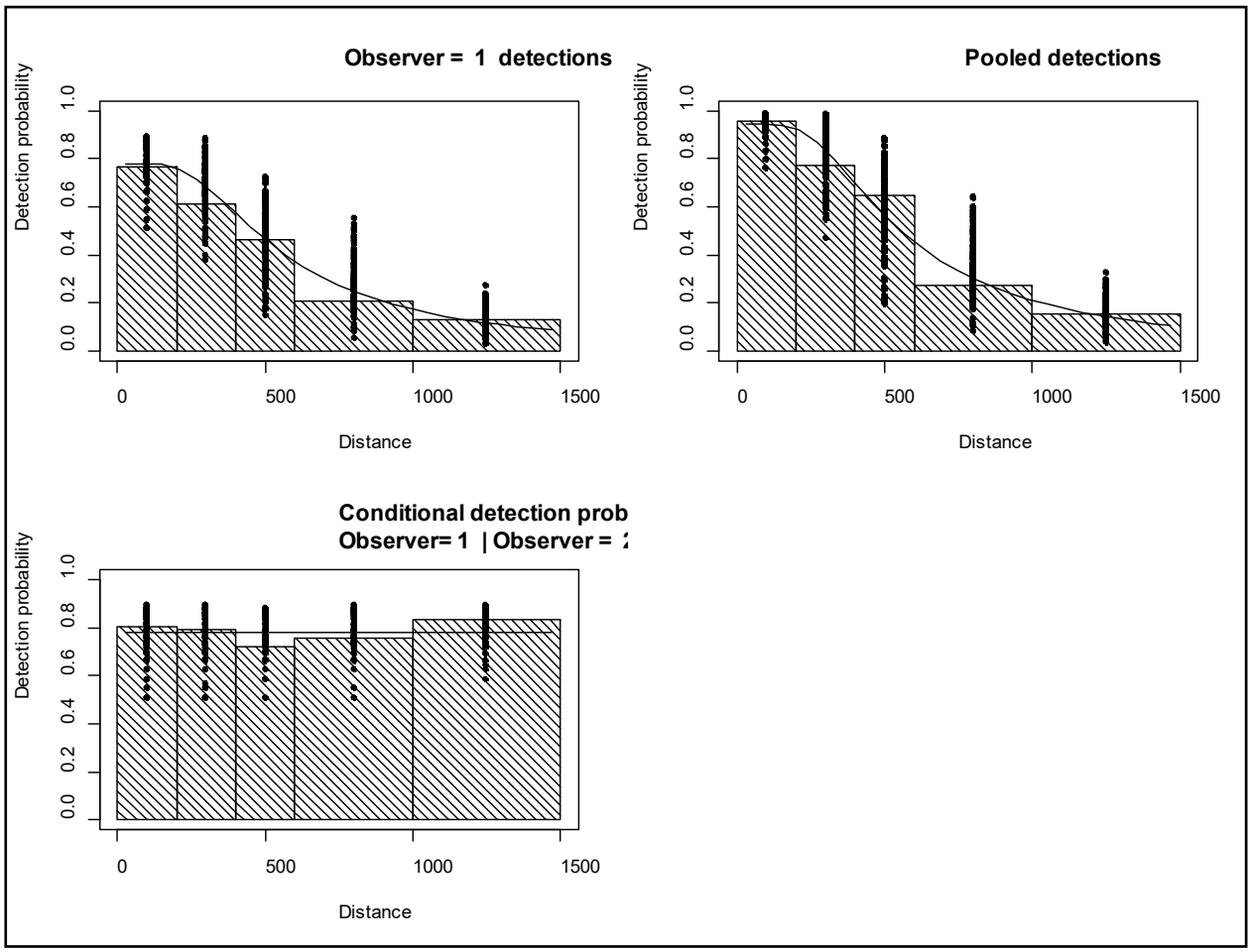


Figure 26. Graphical representation of goodness-of-fit of the most supported double observer model for May 2017 (Model 1, Table 7).

Table 7. Dependent double observer model selection results for 2017. Sample size adjusted Akaike Information Criterion (AICc), the difference in AICc between the most supported models for each model (ΔAIC_c), AIC_c weight (w_i), number of model parameters (K), and deviance is given. See **Table 2** for covariate definitions

No	Model	AIC_c	ΔAIC_c	w_i	K	LL
1	obs+snowc+log(size)	1,193.0	0.00	0.40	4	-592.5
2	obs+snowc+size	1,193.1	0.07	0.38	4	-592.5
3	obs+snowc	1,194.7	1.70	0.17	3	-594.3
4	obs+snow_factor	1,197.8	4.75	0.04	5	-593.8
5	obs+snow_factor+cloud_factor	1,200.3	7.26	0.01	8	-592.0
6	constant	1,208.7	15.73	0.00	1	-603.4

4.2.3 Comparison of estimates from strip transect, dependent double observer pair, and distance sampling

Comparison of strip transect single observer pair, dependent double observer pair, and distance sampling estimates suggests reasonable agreement between estimates, with the confidence intervals from each method all overlapping.

Further inspection of estimates suggests that the assumption of perfect sightability on the 400-meter survey strip was met in only 2013 and 2015 with estimates being close for dependent double observer pair and strip transect estimates (**Figure 30**). In 2011, variability in observers and snow cover reduced the strip transect estimates compared to the dependent double observer pair estimates. In this context, the dependent double observer pair method provided a test of assumptions of the strip transect method and corrected estimates when the assumption of perfect sightability was violated. Estimates from the dependent double observer pair method when compared with the single observer pair jolly estimator were 6% higher in 2011, similar in 2013 and 2015, and 4% higher in 2017. A similar comparison for 2019 and 2023 showed double observer pair estimates 5.0 % higher in May 2019 and only 1% higher in May 2023, suggesting sightability was likely improved during the May 2023 survey.

In 2017, distance sampling estimates were higher than dependent double observer pair and strip transect estimates. This may have been due to one of the observer pairs not putting enough survey effort into the distance bins closer to the aircraft (**Figure 27**), as indicated by different shapes of the detection histograms for the two observer pairs. This would have caused a negative bias in both strip transect and dependent double observer pair estimates and illustrates a potential issue with distance sampling; observers spending too much time looking out at further bins which are often easier to view than the closer bins. In the case of conditions of excellent sightability, this can lead to a significant over estimate. The dependent double observer pair method partially accounted for this by also estimating the sighting probabilities of observers near the survey line. The dependent double observer pair method assumes that the

two observers in a pair have equal sighting probabilities. It is therefore essential that observers switch places half way through the day to ensure robust estimates from this method. Of the 19 observer pairings across all surveys, 11 switched places which may have affected the overall quality of the dependent double observer pair estimates. If observers cannot switch places, then an independent observer method should be considered especially when caribou density is not high.

Table 8. Comparison of estimates of Southampton Island caribou using strip transect, double observer, and distance sampling/double observer (2017 only).

Year and Method		Caribou counted	N	SE	Conf. Limit		CV
2011	Strip transect	1667	7,937	580.4	6,861	9,182	7.30%
2011	2x Observer strip transect	1667	8,467	479.9	7,558	9,486	5.67%
2013	Strip transect	1597	7,284	525.3	6,307	8,413	7.20%
2013	2x Observer strip transect	1597	7,287	365.6	6,580	8,071	5.02%
2015	Strip transect	3068	12,319	931.6	10,591	14,328	7.60%
2015	2x Observer strip transect	3068	12,370	681.3	11,140	13,736	5.14%
2017	Strip transect	1685	8,436	680.8	7,184	9,906	8.10%
2017	Distance 2x observer	1653	9,200	796.4	7,755	10,915	8.70%
2017	2x Observer strip transect	1665	8,752	759.5	7,365	10,399	8.70%
2019	Strip transect	2512	11,521	1063.3	9,583	13,852	9.20%
2019	2x Observer strip transect	2512	12,054	901.9	10,354	14,032	7.48%
2023	Strip transect	2610	12,565	802.0	11,066	14,266	6.40%
2023	2x Observer strip transect	2610	12,651	851.0	11,044	14,493	6.70%

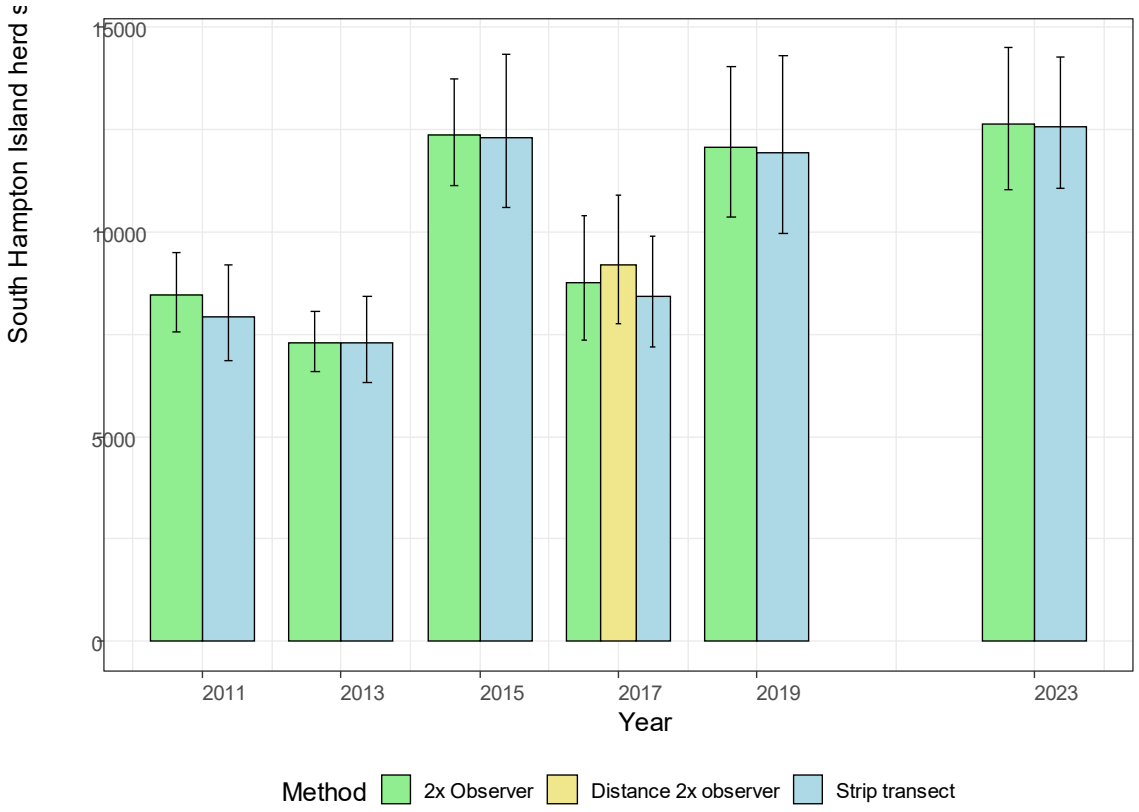


Figure 30. Comparison of more recent Southampton Island caribou herd abundance estimates between June 2011 and May 2023, from strip transect, dependent double observer, and 2017 only distance sampling/double observer analyses.

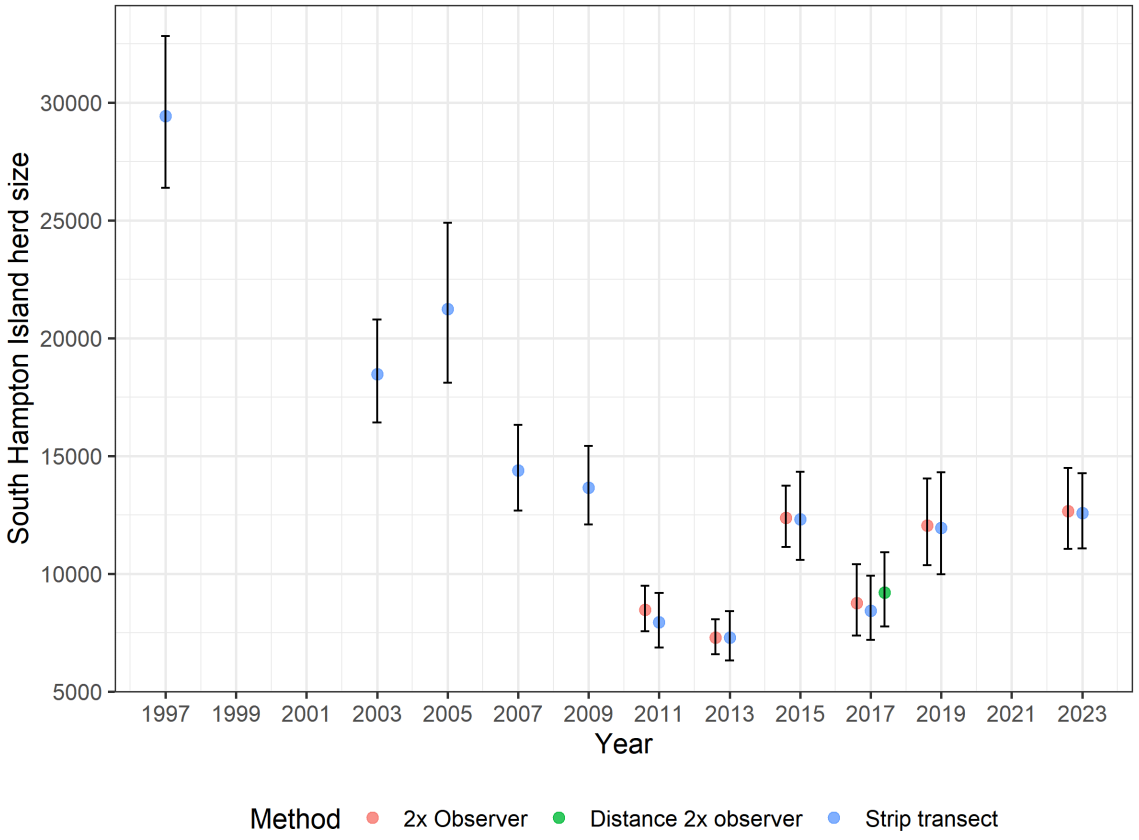


Figure 31. A long-term comparison of strip transect, dependent double observer pair, and distance sampling (2017 only)/double observer pair estimates from June 1997 through May 2023.

4.3 Trend estimates:

Our trend analyses cover three separate phases of Southampton Island caribou abundance: **1**- prior to 1997 when herd abundance was increasing; **2**- from 1997 to 2017 when the herd was declining, and from 2017 to 2023 when estimates indicated stability.

4.3.1 Trend from 1978 to 1997(the increase phase)

The historic data set (1978-1991) was added to the analysis to obtain an estimate of trends in the SHI caribou population during the phase of increase that occurred from 1978 to 1997. This was accomplished by adding terms to account for the decrease phase, which allowed us to estimate an annual rate of increase of 1.18 (CI=1.12-1.25), or, 19% (CI=16-22%), from 1978 to 1997 (**Table 9**). A plot model for these predictions is shown in **Figure 32**.

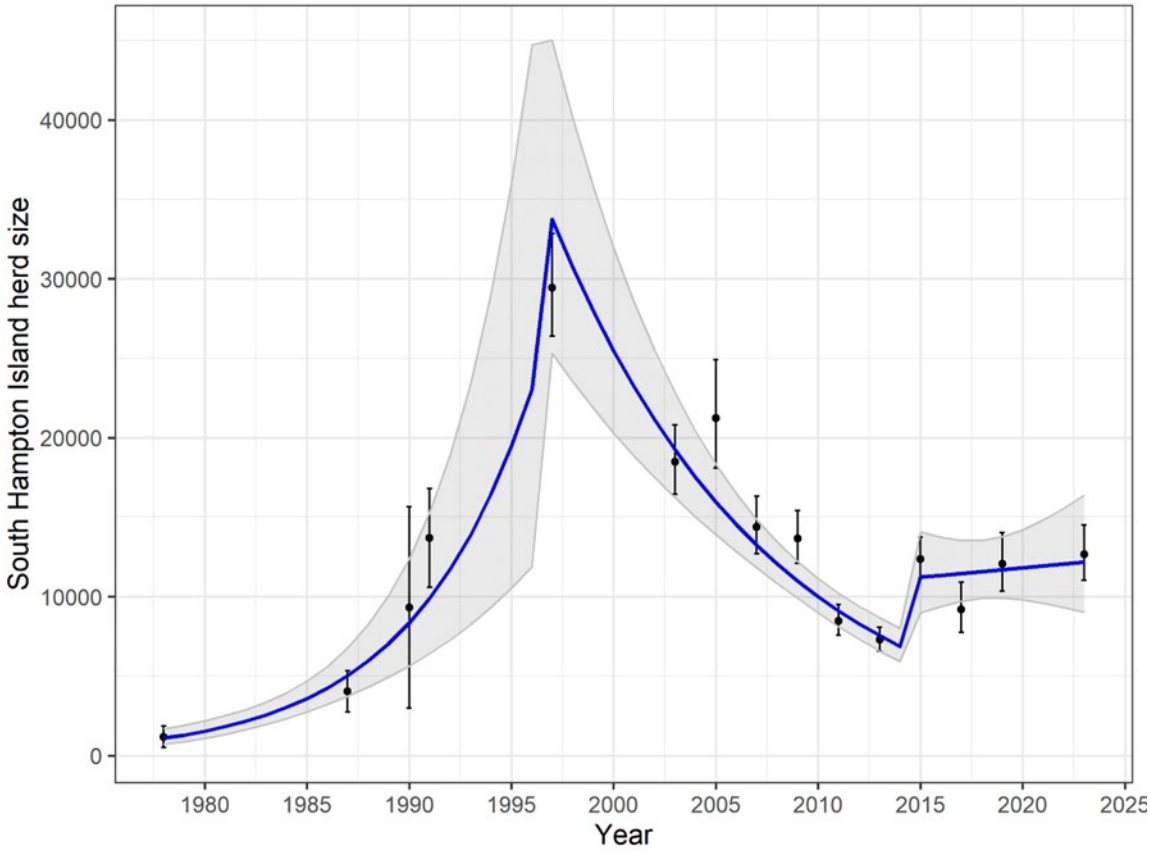


Figure 27. Predictions of herd size of the Southampton Island from the log-linear model (Table 11) which assumes a constant decline in population size after 1997 with an immigration event that occurred before the 2015 survey.

Table 9. Log-linear model parameter estimates for trend analysis (1978-2023).

Term	β	SE (β)	t	p-value	Conf. Limit	
(Intercept)	927.28	0.24	28.30	0.0000	558.53	1444.60
Trend (1978-1997)	1.18	0.03	6.31	0.0001	1.12	1.25
Decrease-intercept	236.37	0.44	12.41	0.0000	98.93	558.71
Immigration (2015)	0.04	1.15	-2.90	0.0175	0.00	0.34
Trend (1998-2014)	0.77	0.03	-9.02	0.0000	0.73	0.81
Trend (2015-2023)	1.01	0.03	3.59	0.0059	0.95	1.07

4.3.2 Trend from 1997 to 2023 (the decline phase)

Data from 1997 to 2023 included strip transect, dependent double observer pair, and distance sampling surveys. The use of different methods had minimal effects on the overall abundance trends identified. However, the best estimates for 2011, 2013, and 2015, based on model fit and lowest CV's, were dependent double observer pair estimates which accounted for sightability, especially in 2011. For 2017, the distance sampling estimate was least biased because of observer error. For this reason, we used strip transect data for estimates from surveys up to 2009, followed by dependent double observer pair estimates for 2011 to 2015, 2019 and 2023, and distance sampling estimates for 2017.

T-tests were initially used to compare the significance of the difference between sequential estimates (**Table 10**). Of the 8 survey estimate comparisons the 1997 to 2003, 2005 to 2007, 2009 to 2011, 2013 to 2015, 2015 to 2017, and 2017 to 2019 periods showed statistically significant change, with no significant change being detected between 2019 and 2023. Of these comparisons, only the 2013 to 2015, and 2017 to 2019 estimates showed a statistically significant increase in the SHI caribou population, all others, with the exception of 2019 to 2023, displayed significant declines. Annual change in population size, based on a year-to-year comparison of estimates (expressed as ratios), varied between 0.79 and 1.30. SHI caribou abundance estimates from 1997 to 2023 are shown graphically in **Figure 33**.

A log-linear model was then applied to further assess trend. To estimate the effect of a potential immigration event on the overall trend, prior to the 2015 survey, an additive term was applied to model use to generate the 2015-2017 survey estimates. This term basically assumed that the SHI population was increased by a constant amount during this time due to immigration. These terms were both found to have a significant effect on the trend in caribou abundance (**Table 11)**. The year term provided an estimate of long-term annual rate of change for the SHI population (0.91 CI=0.89-0.93) which was not, overall, affected by the immigration event. This translates to a 9% (CI=7-11%)

decline in caribou abundance each year, from 1997 through 2017. A trend term for the period of 2015 to 2023 was not significant.

A plot of model predictions reveals good fit of the model to estimates with predictions intersecting the confidence limits of all 10 estimates (**Figure 33**). Namely, the model suggests that the herd declined at a constant rate from 1997-2014, followed by an immigration event sometime between May 2013 and May 2015 (Paetkeau, 2015), and then continued to decline at a similar rate as it had previously, from 2015-2017 (**Figure 33**). Using this model, and assuming a constant rate of decline (9%) over the period, we estimated that approximately 5,024 caribou would have had to immigrate to SHI between May 2013 and May 2015 to account for the increased number of animals observed in May 2015.

Table 10. Estimates used for the 1997 to 2023 trend analysis of Southampton Island caribou abundance, with the results of t-tests comparing the estimates of successive surveys. Also shown are estimates of gross and annual change based on the ratios of successive estimates.

Year	method	N	SE	CV	df	t-test	df	p-value	Gross change	Annual change
1997	Strip transect	29,425	1622.5	5.5%	93					
2003	Strip transect	18,479	1099.8	6.0%	90	-5.58	163	0.0000	0.63	0.93
2005	Strip transect	21,227	1701.8	8.0%	76	1.36	132	0.1774	1.15	1.07
2007	Strip transect	14,389	914.6	6.4%	88	-3.54	117	0.0006	0.68	0.82
2009	Strip transect	13,651	833.1	6.1%	80	-0.60	168	0.5514	0.95	0.97
2011	2x Observer	8,467	479.9	5.7%	52	-5.39	122	0.0000	0.62	0.79
2013	2x Observer	7,287	365.6	5.0%	32	-1.96	83	0.0539	0.86	0.93
2015	2x Observer	12,370	681.3	5.5%	31	6.57	48	0.0000	1.70	1.30
2017	Distance 2x observer	9,200	796.4	8.7%	134	-3.02	122	0.0030	0.74	0.86
2019	2x Observer	12,054	901.9	7.5%	33	2.37	91	0.0198	1.31	1.14
2023	2x Observer	12,651	851.0	6.7%	39	0.48	71	0.6314	1.05	1.01

Table 4. Log-linear model parameter estimates for trend analysis (1997-2019).

Term	β	SE (β)	t	p-value	Conf. limit	
Intercept	37053.14	0.15	68.31	0.0000	27104.24	49599.61
Trend λ (1997-2015)	0.91	0.01	-8.41	0.0001	0.89	0.93
Immigration (2015)	0.25	0.60	-2.31	0.0540	0.08	0.82
Trend λ (2015-2019)	1.01	0.03	0.39	0.7112	0.96	1.06

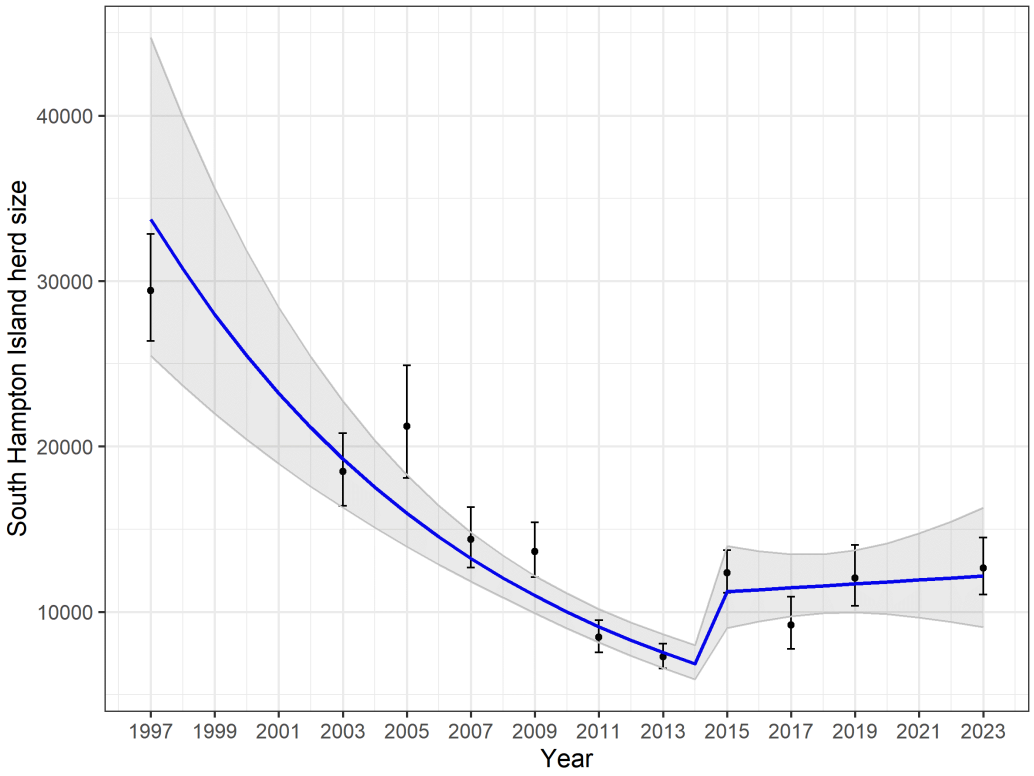


Figure 28. Predictions of herd size of the Southampton Island caribou population from the log-linear model (Table 11), which assumes a constant decline in population size with an immigration event that occurred before the 2015 survey. Confidence limits are provided as shaded regions on the plots.

4.4 Effect of Disease on Abundance:

Brucellosis is an infectious disease caused by the Bacteria Genus *Brucella*. Many different animal species including humans can become infected. The form of Brucellosis known to occur in wild caribou is *Brucella suis* Type IV. In caribou this bacterium occurs primarily within tissues of the reproductive system but also commonly occurs within leg joints (Williams and Barker, 2001; O'Reilly, 1992; Corbel, 2006). The bacteria can also be found in the milk, blood, urine and semen of infected animals (O'Reilly, 1992; Corbel, 2006). Animals can be infected by the bacteria by either oral ingestion, direct contact with the mucus membranes of the eyes, nose, or mouth, or through breaks in the skin. *Brucella* can also be transmitted by contaminated objects (fomites) (Corbel, 2006). Some animals are carriers and can have the bacteria without showing signs of the illness. Animals in these cases can shed the bacteria into the environment for long periods, infecting other animals in the herd. Brucellosis can cause reproductive problems such as abortions, still birth and infertility. Other signs can include arthritis, swelling of the joints and testicles, and udder infections (mastitis) (Williams et al. 2001; CDC 2016; Corbel, 2006). Tissues and fluids associated with the amniotic sac, drainage of fluid from swollen joints, vaginal discharge, fetal fluids, and semen can be highly infective and can spread the bacterium into the immediate environment where uninfected animals can become infected through the ingestion of infected tissues and contaminated objects such as plants. The potential for environmental concentration of this disease makes Brucellosis a density-dependent disease. Areas of concentration such as migratory corridors, rutting areas and particularly calving grounds, would represent some of the higher risk seasonal range for the spreading of this disease (Williams et al. 2001; O'Reilly, 1992; Corbel, 2006). Predation and scavenging of diseased tissue can also contribute to the bacterium's spread throughout the environment.

By 1995, the condition and productivity of the herd had changed little, an assessment that would remain up until the 2000 harvesting season when CFIA random blood

testing identified the beginning of what would become a rapid induction of the bacterial disease *Brucella suis* serovar 4 in the SHI caribou herd (**Figure 34**). There is no evidence of this disease within this population prior to the 2000 harvesting season. Susceptibility to disease and parasites due to low genetic heterogeneity has been a concern for the SHI herd since the introduction of caribou to SHI, and was a likely catalyst to the wide spread infection of caribou with *Brucellosis suis* first detected in the population February 2000, and followed by a recorded decline in abundance. Prevalence of Brucellosis climbed from 1.7% in February 2000 to 58.8% by March 2011 and this increase is thought to have contributed to decreased pregnancy rates over the same period. Pregnancy rates dropped from a high of 93.1% in February 2001 to a low of 37% in March 2011.

Health monitoring of the SHI barren-ground caribou had its beginnings in 1988 when D. Heard (departmental correspondence) sampled 20 cows in March to determine their reproductive status and general condition. These small condition studies continued through 1991 (Jan Adamczewski and Douglas Heard unpublished data) at which time the condition studies were discontinued. The analysis of condition was started up again in February 1996 in association with the initiation of the large-scale commercial harvest on the Island in March 1993. Due to the small sample sizes collected during this period little weight could be put on their influence on overall trends in condition. The first samples did, however, give results that were consistent with hunter reports of caribou on SHI in excellent health and condition during this period.

Based on the CFIA testing of 400 caribou annually between 2000 and 2009, and 100 caribou tested annually between 2009 and 2011, the first cases of *Brucella suis* were reported during the 2000 harvest year (1.7% of 400 animals tested) and had reached a prevalence of 19.5% in 2003, 28.6% in 2005, 48.8% by 2007, 39.1 % by 2009 and 58.8 % by 2011. Pregnancy rates initially dropped from 93.1% in 2001 to 37.9% in 2005, and then increased to 64.4% in 2007. The hopes that the disease was declining in the population were dashed when a 2009 screening showed pregnancy rates dropping further to 44.3%. The last major condition study conducted in March 2011, prior to the application of a TAH, recorded pregnancy rates of 37% (**Figure 34**).

In 1992 the Canadian Polar Commission released a status report on Brucellosis in the Circumpolar Arctic (O'Reilly, 1992). In the report, O'Reilly summarized the incidence of Brucellosis across the Circumpolar arctic (**Table 12**). Brucellosis prevalence within the Southampton Island population reached a high of 58.9% in 2011 which represents the highest prevalence amongst any caribou and/or reindeer populations' worldwide (O'Reilly, 1992). Currently levels are unknown due to a cessation of the annual caribou condition harvest. With the human health issues associated with Brucellosis through either the consumption or handling of infective tissues, Coral Harbour residents are concerned over the future of their caribou herd. Though disease screening has not taken place since 2011, an increase in population abundance was recorded in June 2015. This finding, coupled by hunter reports of fewer infected caribou and more calves, suggests that the disease prevalence has been dropping from 2015 to present. Disease screening is planned to be continued beginning in the 2025 harvesting season.

4.4.1 Brucellosis and herd trend

Concurrent with the rising prevalence of the reproductive disease *Brucella suis* was the reported declines in abundance from 1997 through 2013 (**Figure 34**). It appears clear that Brucellosis was a contributing factor to the steady declines observed in this population of caribou. However, with high commercial harvest rates of the SHI herd up to 2009, it is likely that both commercial hunting pressure and disease together, contributed significantly to a declining trend in caribou abundance up to 2013. By 2003, three years following the first confirmed cases of Brucellosis in SHI caribou, pregnancy rates were still over 85% and the population was still over the hypothesized carrying capacity of the island, theorized to be 15,000 animals (Ouellet et al. 1996). With Brucellosis being a density dependent disease, it was decided by all co-managers that a further reduction in caribou abundance would be beneficial to the long-term viability of the SHI population hence the continuation of the commercial harvest up to the 2009

harvesting season. Abundance monitoring every two years was continued up to 2019 and again most recently in 2023, and associated adjustments to a Total Allowable Harvest (TAH) was utilized to help recover the declining trend of the herd. Relative stability between 2013 and 2015 suggested the situation was improving and that the disease was having less of an impact on the herd. We believe that there is definitive evidence of an immigration event onto SHI that likely occurred during the winter of 2014. We further believe that this genetically verified influx of more disease resistant mainland caribou played an important role in the reduction in hunter observed disease prevalence as evidenced by hunter reports and the cessation in herd decline first realized in 2015.

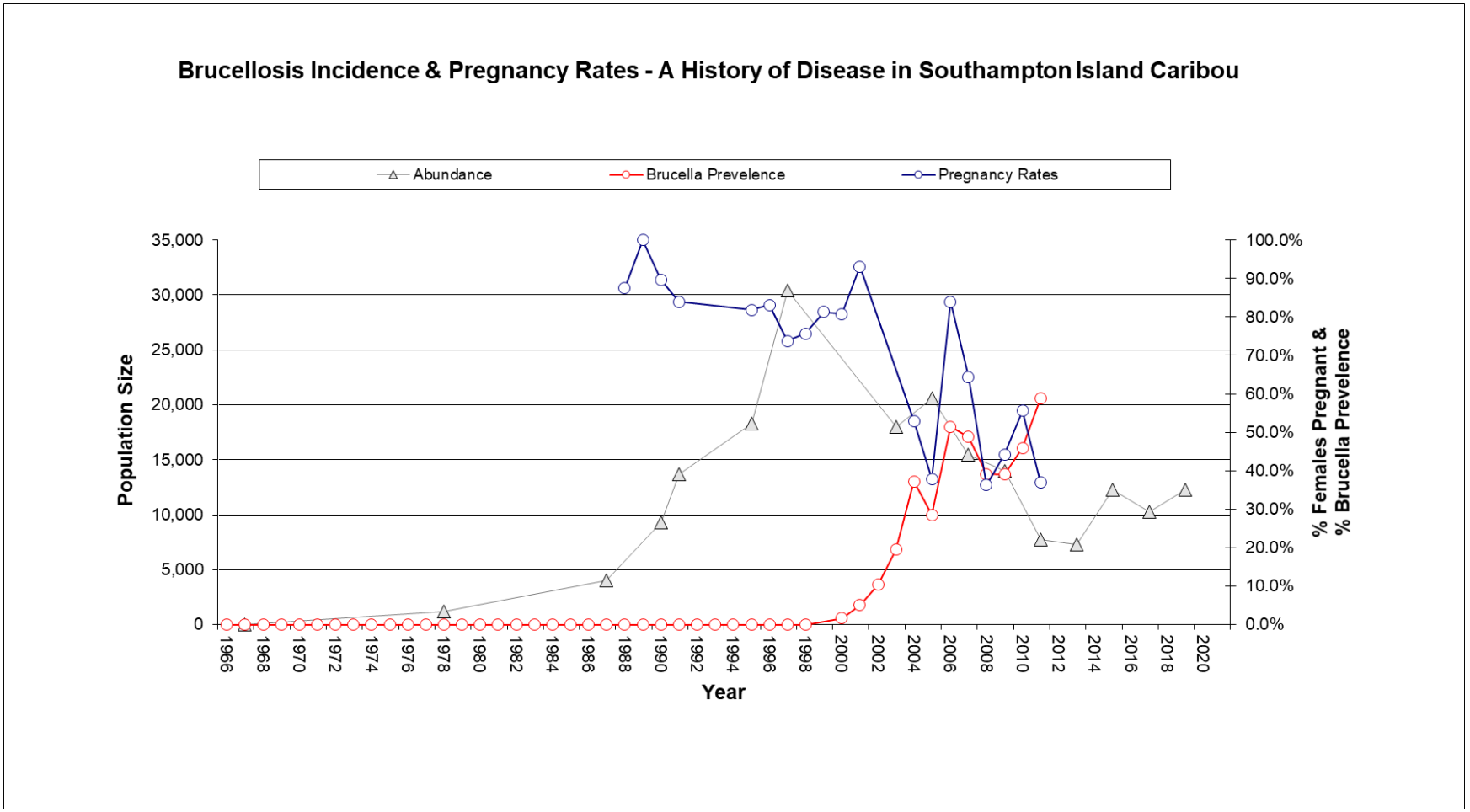


Figure 29. A history of abundance, pregnancy rates and *Brucellosis suis* prevalence for the Southampton Island caribou herd originally introduced onto the island from Coats Island in 1968.

Table 5. Circumpolar Incidence of Brucellosis in barren-ground caribou and reindeer across arctic North America (after O'Reilly, 1992).

Herd	Incidence (%)	Date	Remarks	Source
Southampton	Not Present	1990	75 samples	(NWT Wildlife notes)
Qamanirjuaq	4%	1966-68		(NWT Wildlife notes, 1983)
Beverly	< 2%	1983	118 samples	(Goldfarb, 1990)
Bathurst	Present	1981-1983	3 samples	(NWT Wildlife notes, 1983)
Baffin Island	14-35%	Mid-1980s	N Baffin highest	(O'Reilly, 1992)
Melville/Boothia	20-35%	1980s	17 samples	(O'Reilly, 1992; Gunn et al. 1991)
Ahiak	?			
Porcupine	15-20%	1980s	?	(O'Reilly, 1992)
Central Arctic	15-20%	1980s	?	(O'Reilly, 1992)
Western Arctic	</= 30%	1960-1980	?	(O'Reilly, 1992; Neiland et al. 1968)
Nechina	1-6.5%	1962-65	?	(Neiland et al. 1968)
George River	Not Present	1987-88	?	(Forbes 1991; Greenberg et al. 1958)
QEI Peary	Present	1980s	1 sample (P. of W. Island)	(Forbes, 1991)

4.5 Harvest

Throughout the reintroduction of barren-ground caribou to SHI, wildlife managers of the time were vigilant in their on-going management of the herd. Management recommendations were, in all cases, based on research results, and in particular, quantitative population estimates. In February 1978, the first caribou hunt since the 1968 introduction was carried out on SHI. The quota was set at 25 bulls and was based on observations from a reconnaissance survey flown in 1977 that sighted a total of 172 caribou, 79 of which were adult males, 54 adult females, and 39 yearlings, suggesting a sex ratio skewed towards males (Kraft, 1978; Gates, 1988) (**Table 13**) (**Figure 35**). In August 1979, the TAH (quota) for bulls was increased to 50 based largely on the findings of the November 1978 population survey which estimated 1,138 adult and yearling caribou (Kraft, 1978). Early in 1983 the first cow harvest was approved with a TAH set at 20. Regulations were developed along with this new TAH stipulating those 10 cows be harvested in the spring and the remaining 10 in the fall. The TAH was then raised from 50 to 250 bulls, and from 20 to 50 cows, based on recommendations generated following the 1987 population estimate though the estimate was inconclusive (Heard and Grey, 1987; Heard and Ouellet, 1994).

During the 1988 harvesting year, concerns regarding the accidental harvesting of females seem to have led to the removal of the female quota and an increase in the male quota to 300 animals sometime in 1988. At this time, it was clearly indicated in the regulations that; “hunting zone J/2 (Southampton Island) was restricted to 300 male caribou.” In 1989 recommendations to increase the TAH to 400 caribou, of which 100 could be female, were made. These recommendations were supported by GNWT Biologist Doug Heard who indicated the proposed increases were based on sound ecological principles and quantitative survey results (Renewable Resources Official Correspondence 140 007 005 & 150 001 005, October, 1989). Seasons for this new quota were recommended to be from October 1st to October 31st for males and April 1st to May 31st for females. By 1993, and in response to rapid population growth

reported by Ouellet in 1991, the TAH was removed (Ouellet, 1992) (**Table 14**). From 1993 up until the 2012 harvesting season subsistence harvest was not accurately monitored. In Nunavut, monitoring of caribou harvest in the absence of a TAH is not mandatory. Although the 1991 Nunavut Wildlife Management Board (NWMB) Harvest Study attempted to quantify wildlife harvest through hunter interviews, it is generally agreed that the final harvest study estimates may not be entirely accurate in some cases. For SHI, however, accurate records of commercial harvest numbers and sex ratios were kept from 1992 through to 2009.

The first commercial quotas were established in 1992 and were set at 250 animals (gender breakdown unknown). Despite the 1992 commercial allocation, it was not until 1993 that the first five caribou (of unknown gender) were harvested commercially. Commercial quotas continued to rise to 1,000 animals in 1993, 5,000 in 1994, and 6,000 by 1997 (Junkin, 2003). Since 1993 there have been annual commercial harvests up to and including the 2009 harvesting season. Interestingly, a non-sex-selective subsistence quota of 1,000 animals was re-instated in 1994 in an effort to offset an increase in the commercial quota from 1,000 to 5,000 over the same period (Junkin, 2003). By 1997, in response to survey results indicating the continued rapid growth of the population to 30,381 animals (Mulders, 1995), concerns about the caribou population having exceeding the Islands hypothesized carrying capacity of 15,000 caribou were being realized (Ouellet et al 1994, Ouellet et al 1993). In response to these concerns, the wildlife regulations were once again amended to allow an unlimited subsistence harvest and a non-sex-selective commercial quota of 6,000 caribou.

Overall, the commercial harvest was successful in reducing the population to the estimated carrying capacity of the Island of 15,000 caribou (Ouellet et al. 1996). Concerns of the time, however, were that continued high harvest rates, in excess of 6,500 caribou over the 2006 and 2007 harvesting seasons, would drive the population too low to sustainably maintain the estimated subsistence harvest rate of 1,500 to 2,000 caribou annually. Additionally, there was the concern of rising *Brucella* prevalence and its observed impact on the reproductive potential of the SHI herd. The

continued decline of SHI caribou following the 2003 survey estimate only heightened these concerns, and by 2007, when the population had dropped further to an estimated 14,389 adult and yearling caribou, discussions on ending the commercial harvest had begun. However, the harvest employed many local people and the political will to continue the harvest was high. Despite these pressures, the harvest was cancelled by the Coral Harbour HTO in 2008 and only a small harvest of 843 was undertaken in March 2009. Between 1978 and 2009 an estimated total of 27,400 caribou had been harvested for subsistence purposes and 42,000 for commercial purposes yielding a total harvest of 69,400 caribou, of which 61% were taken for commercial purposes. Since 2009 there has been no commercial caribou harvest. Results from the 2009 aerial abundance estimate showed no significant change between survey periods suggesting that the cessation of the harvest was having the net effect of slowing and/or stabilizing the population decline.

Unfortunately, the stabilizing effect lasted only a short period and by June 2011 estimates of population abundance dropped further to 8,442 adults and yearlings. With the commercial harvest having been stopped, and the subsistence harvest remaining relatively constant at an estimated 1,500 to 2,000 caribou annually there was little more managers could do address anthropogenic causes of decline other than reduce the subsistence harvest. The reasons for this rapid decline appeared to now be related to the reported high prevalence of the reproductive disease Brucellosis. By March 2011, Brucellosis disease prevalence had reached a high of 58.8% and spring pregnancy rates had declined to 37% (**Figure 34**). In addition to high rates of disease, around this time, and despite the cessation of the commercial harvest, a new method of selling country foods was gaining popularity and increasing harvests of SHI caribou. This new harvest pressure was developing from the growing demand for the sale of caribou meat on social media within Nunavut. A lucrative market had opened up on Baffin Island where Baffin communities were struggling with declining caribou populations as well. When sales of caribou meat from SHI first took hold on social media, 24,764 kilograms of caribou meat, representing an estimated 710 caribou, were sold and shipped from SHI in the first 8 months of sales (**Figure 36**). Unfortunately, the data provided by the airline to assess the shipment of caribou meat off SHI was cutoff in January 2012,

removing our ability to further quantify and monitor the internet sales and harvest totals, through export traffic, for the months of heaviest harvesting (March, April, and May).

4.5.1 Harvest Management and Planning 2011 to present

Meetings in the summer and fall of 2011 between the Government of Nunavut (GN) Department of Environment (ENV) and the Coral Harbour HTO, and additional meetings with all stakeholders in the winter of 2012, led to a formal request by the Coral Harbour HTO to the GN and the NWMB to apply a TAH of 4 caribou per household (1,000 caribou) in an attempt to stabilize the decline through subsistence harvest management.

Another product of these meetings was the development of the *Southampton Island Barren-ground Caribou Population Management Plan* (2012), which was submitted to the NWMB for decision in March 2012. The plan outlined an agreement to establish a TAH of 1,000 caribou and a Non-Quota Limitation (NQL) protecting cow/calf pairs. Also in the plan was the specification of continued harvester-supported monitoring, and the continued assessment of SHI caribou population abundance every 2 years. The urgency of the situation led to the NWMB supported, and community requested, establishment of a Ministerial Management Initiative (through the Nunavut *Wildlife Act*) to immediately assign a temporary TAH of 1,000 caribou for the SHI caribou population.

By June 2013, the herd had further declined to an estimated 7,287 adult and yearling caribou, prompting the GN ENV to recommend a further reduction to 2 caribou per household (500 caribou) with 100 caribou held back for the HTO to use as deemed appropriate, for a total of 600 caribou. The community rejected this recommendation, preferring to wait until the May 2015 abundance estimate had been completed to make a final decision. The community based its decision on hunter observations of reduced signs of Brucellosis within their catch and a general thought that herd health and

pregnancy rates were improving. Continued reports of healthy caribou, fewer signs of disease, several reports of a possible movement of caribou onto the Island over the winters of 2014 and 2015, and a noticeable increase in calves in June 2014, preceded the May 2015 abundance survey. Consistent with community reports, the 2015 survey estimated a significant increase in adult and yearling caribou. In two years, the population had increased by 5,081 animals to 12,368 caribou, an estimate believed higher than could be accounted for by reproduction alone. The community of Coral Harbour was not surprised with the result, attributing the increase to what they believe was the movement of a large group of caribou from the mainland onto the north end of the island during the winter of 2014. In an attempt to verify these accounts, the GN conducted a genetic analysis using SHI hunter provided tissue samples from 2014 and then comparing them to SHI samples from 2004 and samples collected on the mainland in the vicinity of Naujaat in 2014. This analysis quantitatively confirmed, for the first time, the presence of mainland barren-ground caribou genetic markers on SHI in 2014.

The observed increase documented in 2015 lead to a temporary increase in TAH to 1,600 caribou with a NQL protecting cow/calf pairs **(Table 14**). This increased TAH continued through the 2016 and 2017 harvesting years after which the 2017 survey results, indicating a significant decline in abundance, saw the SHI TAH lowered back to 1,000 caribou/year which has remained to present.

Table 6. History of the Southampton Island assigned subsistence harvest quotas (TAH) prior to 1992. Harvest management prior to the first commercial allocation in 1992 (subsistence harvest estimated using government reports, HTO correspondence and personal communications with wildlife staff).

YEAR	Regulated Quotas (TAH)						Total Allowable Harvest (TAH)
	Subsistence				Commercial		
	Female (#)	Male (#)	No Sex Selection (#)	Total (#)	No Sex Selection (#)	Total	
1978	0	25	0	25	0	0	25
1979	0	50	0	50	0	0	50
1980	0	50	0	50	0	0	50
1981	0	50	0	50	0	0	50
1982	0	50	0	50	0	0	50
1983	20	50	0	50	0	0	50
1984	20	50	0	50	0	0	50
1985	20	50	0	50	0	0	50
1986	20	50	0	50	0	0	50
1987	50	250	0	250	0	0	250
1988	0	300	0	300	0	0	300
1989	100	300	0	300	0	0	300
1990	0	400	0	400	0	0	400
1991	0	400	0	400	0	0	400

Table 7. History of the Southampton Island harvest assigned commercial and subsistence Quotas (TAH) from 1992 to present (subsistence harvest estimated using government reports and submissions, NWMB meeting correspondence and Harvest Study (2004), HTO correspondence, and personal communications with wildlife staff).

YEAR	Regulated Quotas (TAH)						Total Allowable Harvest (TAH)
	Subsistence				Commercial		
	Female (#)	Male (#)	No Sex Selection (#)	Total (#)	No Sex Selection (#)	Total	
1992	0	400	0	400	250	250	650
1993	no limit	no limit	no limit	no limit	1,000	1000	no limit
1994	NA	NA	1,000	1,000	5,000	5,000	6,000
1995	NA	NA	1,000	1,000	5,000	5,000	6,000
1996	NA	NA	1,000	1,000	5,000	5,000	6,000
1997	no limit	no limit	no limit	no limit	6,000	6,000	no limit
1998	no limit	no limit	no limit	no limit	6,000	6,000	no limit
1999	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2000	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2001	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2002	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2003	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2004	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2005	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2006	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2007	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2008	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2009	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2010	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2011	no limit	no limit	no limit	no limit	6,000	6,000	no limit
2012	NA	NA	1,000	1,000	0	0	1,000
2013	NA	NA	1,000	1,000	0	0	1,000
2014	NA	NA	1,000	1,000	0	0	1,000
2015	NA	NA	1,000	1,000	0	0	1,000
2016	NA	NA	1,600	1,600	0	0	1,600
2017	NA	NA	1,600	1,600	0	0	1,600
2018	NA	NA	1,000	1,000	0	0	1,000
2019	NA	NA	1,000	1,000	0	0	1,000
2020	NA	NA	1,000	1,000	0	0	1,000
2021	NA	NA	1,000	1,000	0	0	1,000
2022	NA	NA	1,000	1,000	0	0	1,000
2023	NA	NA	1,000	1,000	0	0	1,000

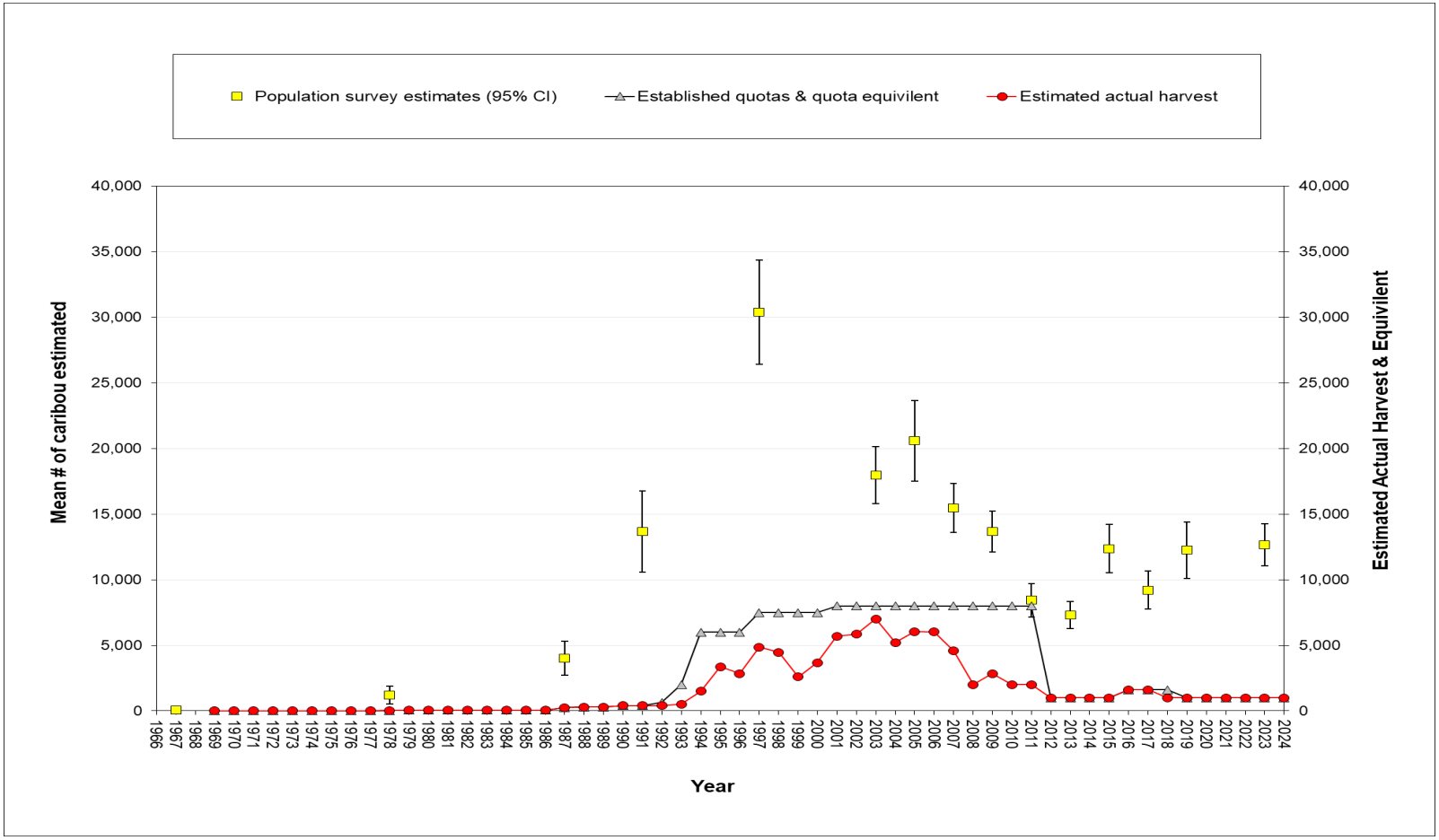


Figure 30. An examination of quota adjustment and actual harvest based on population estimates (Quota equivalents = estimated maximum subsistence harvest substituted for “no-limit” quota allowance values, Tables 13 and 14).

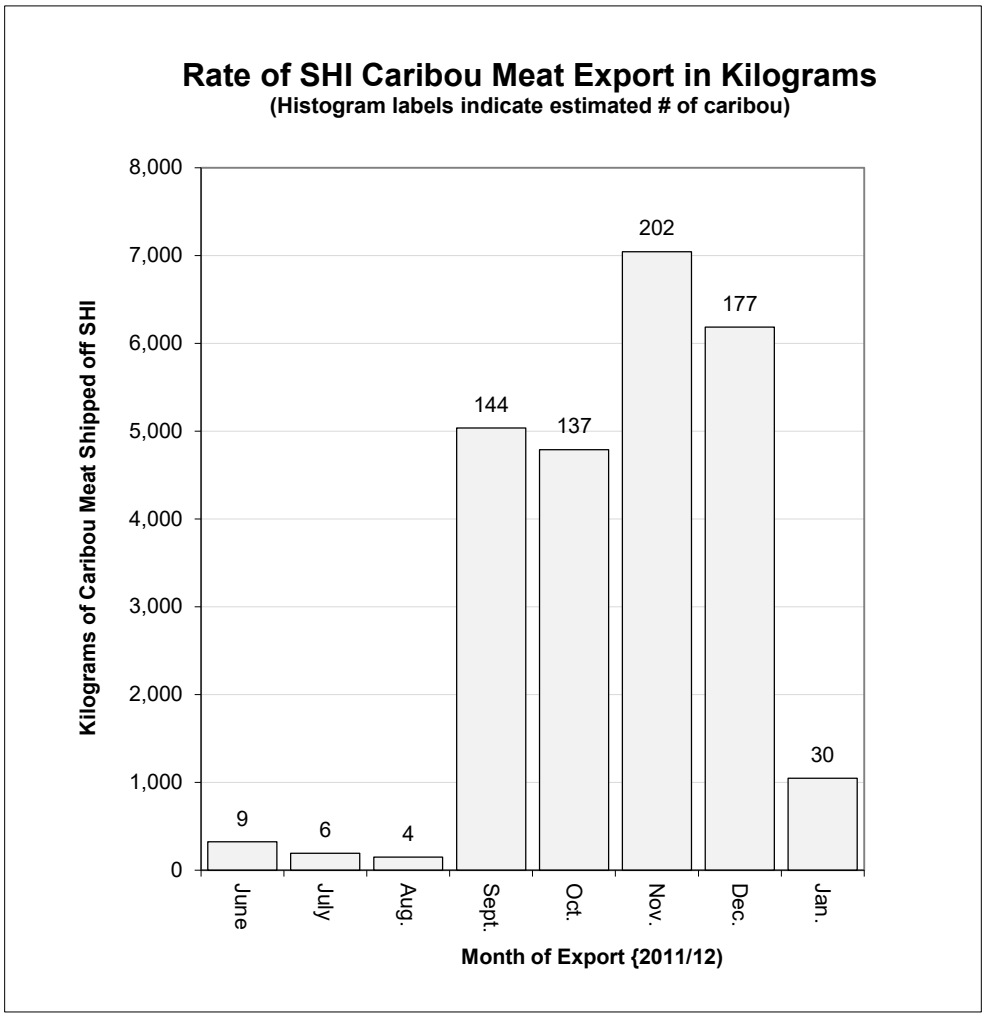


Figure 31. Caribou exports off Southampton Island primarily to Baffin Island communities. Data collected over an 8-month period in 2011/12.

4.6 Population Genetics

The 2015 abundance survey results showed a statistically significant mean increase of 5,081 caribou from the previous survey in 2013, an increase that would be difficult to account for through reproductive rates alone. The GN, in partnership with the Coral Harbour HTO, set out to try and confirm the possible mechanism of this increase. Based on information collected over two meetings with the Coral Harbour HTO, the primary mechanism forwarded by the HTO was the movement of caribou onto SHI. Hunter reports of many tracks coming onto the Northwest end of the island from across the sea ice suggested immigration was likely the mechanism causing the increase in caribou abundance. We sought to verify these observations through a genetic analysis of SHI tissue samples from 2014 (collected just following the reported movement) and 2004 (collected a decade prior to the suspected movement). Both these samples would then be compared with archived Qamanirjuaq caribou samples collected in 2012, and caribou samples collected in the vicinity of Naujaat during the winter of 2014, both used to represent eastern Kivalliq mainland herds. The Naujaat samples represented barren-ground caribou closest to SHI though separated by Rose Welcome Sound. We employed Wildlife Genetics International (WGI) to analyze the samples and test the validity of this immigration event hypothesis as the cause of the observed increase in caribou abundance on SHI.

Using Qamanirjuaq Herd samples and samples collected in the vicinity of Naujaat (Repulse Bay) to represent the mainland populations, and starting out by using only the Southampton data from 2004, WGI noted that the dramatic separation of mainland and island populations was not perfectly reflected across all individuals, even in 2004 (Paetkau, 2015) (**Figure 37**). Specifically, a Qamanirjuaq individual C45 (partially red bar in group 2) and an SHI individual 155 (partially green bar in group 4) were estimated to have ~ 35% ancestry in the 'wrong' population. These unusual individuals were previously dismissed as outliers, but that may have been premature: the stark

differences in allele frequencies should have allowed accurate assessments of ancestry using 18 markers (Paetkau, 2015). Upon examining the 2014 samples, WGI found a marked shift between the 2004 and 2014 SHI genotypes, with 3% of the 2004 caribou being estimated to have < 90% SHI ancestry, versus 35% of individuals collected between 2013 and 2014 having < 90% SHI ancestry. Assuming that this shift is not the result of a change in sampling location — the NW region of SHI might show more mainland influence than the south — this change in the genetic composition of the population over the course of a decade is dramatic (Paetkau, 2015).

According to Paetkau (2015), the temporal shift was strong enough to leave little doubt that geneflow had occurred from the mainland to the island sometime after 2004 but prior to 2015. To address the question of ancestry, Paetkau (2015) calculated the likelihood that each genotype in the dataset would have been drawn from either the mainland (using Qamanirjuaq and Naujaat caribou herd DNA samples for allele frequencies) or the Southampton Island group (using 2004 data for SHI) (Paetkau, 2003) (**Figure 38**). Paetkau concluded that with $P < 0.01$ that any genotype with a lower ratio did not have pure island ancestry, while ratios in excess of -7.8 ($P < 0.01$) had ancestry other than pure mainland.

With consideration to the number of tests conducted and associated hypothesis testing framework, WGI assessed the risk that the outliers are simply Type I errors. Having tested 86 individuals from the mainland, and 58 SHI individuals from 2004, a correction for multiple tests indicated critical values of 0.0006 and 0.0009, respectively, in order to achieve an 'experimentwise' $P = 0.05$, suggesting a genotype with a more extreme P than those that would be expected to occur through Type I error in 5% of similar datasets (Paetkau, 2015). The P-values estimated by GeneClass2 for C45 and 158 were 0.0003 and 0.0000, respectively, so these 2004 outliers cannot be explained by chance, even after correcting for the number of individuals tested (Paetkau, 2015). Paetkau therefore concluded that the evidence of movement in both directions (onto and off of the mainland) by 2014, was statistically meaningful. Indeed, both SHI individuals are statistically excluded as purebred members of either source population

(mainland or island), indicating that they are members of the F1, or subsequent, hybrid generation (Paetkau, 2015).

Moving forward a decade, Paetkau (2015) found that 19 of the 127 new 2014 SHI caribou had a likelihood of $P < 0.01$ that they were from "pure" SHI caribou as represented by the 2004 samples. According to Paetkau 23 individuals produced a P between 0.05 and 0.01 which individually could be explained as outliers (Type I error). As a group, however, Paetkau believed there were too many outliers to be so easily dismissed, as a Type I error for a dataset of 127 pure SHI animals. In total, Paetkau observed 19 individuals beyond the critical ratio for $P = 0.01$, and 42 beyond $P = 0.05$ suggesting a substantial mainland influence present in 2014 but not present in 2004.

Though the results do not support that a pulse of mainland individuals had moved onto Southampton Island recently, they also do not support that genetic isolation of the island herd has been maintained. Paetkau (2015) points out that samples collected on SHI between 2013 and 2015 did not appear to include any F0 (parental generation) immigrants from the mainland. Paetkau concluded that the analysis has documented that a large proportion of 2014 SHI caribou samples (about 1/3 of the current set) are of F1 (offspring generation) or subsequent-generation hybrid ancestry.

One possible explanation of the absence of apparent F0 immigrants from the mainland could be that such individuals arrived at the northwest corner of the island and took a generation or more to reach as far south as the region where the hunter samples were collected, which is more towards the southcentral extents of SHI. This however, cannot explain the statistically significant increase in caribou abundance along with the local reports of mainland caribou migrating onto SHI between the May 2013 and 2015 surveys. Possible reasons for this finding could be related to a sampling bias whereby hunter samples collected from early 2014 could have missed an immigration event occurring later in the winter. Though unlikely, consideration must also be given to the mainland comparative samples. Most of the samples were collected from areas close to Naujaat creating a second possible sampling bias that could have excluded more northern groups of caribou as potential source populations, such as caribou in the

vicinity of Lyon Inlet. The overall findings seem to suggest that at some point after 2004, SHI caribou likely emigrated to the mainland returning to SHI at some point prior to 2015.

Clearly, additional genetic analysis needs to be undertaken to more accurately determine the cause of the hybridization event documented sometime between 2004 and 2015. Overall, we suggest that local hunter knowledge, and scientific evidence to date, all point to the arrival of a large contingent of caribou onto SHI from an area or areas on the mainland not covered by SHI aerial survey extents.

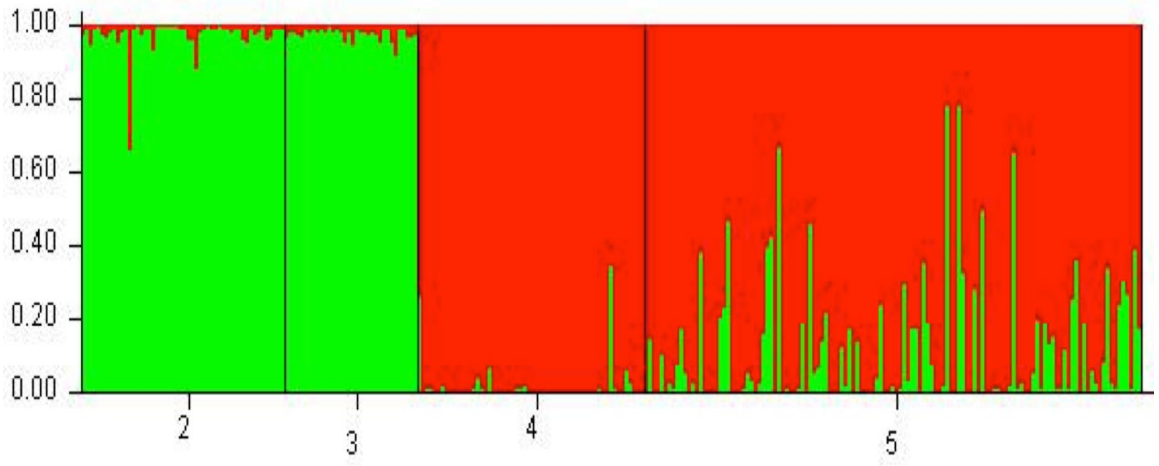


Figure 32. Structure results using 2014 genetic samples. Each column represents an individual, with its estimated proportion of mainland ancestry coded green, and SHI ancestry red. The 'populations' are Qamanirjuaq (2; w9741), Naujaat (3; g1616), SHI 2004 (4; w9741) and SHI 2014 (5; g1616) (Paetkau, 2015).

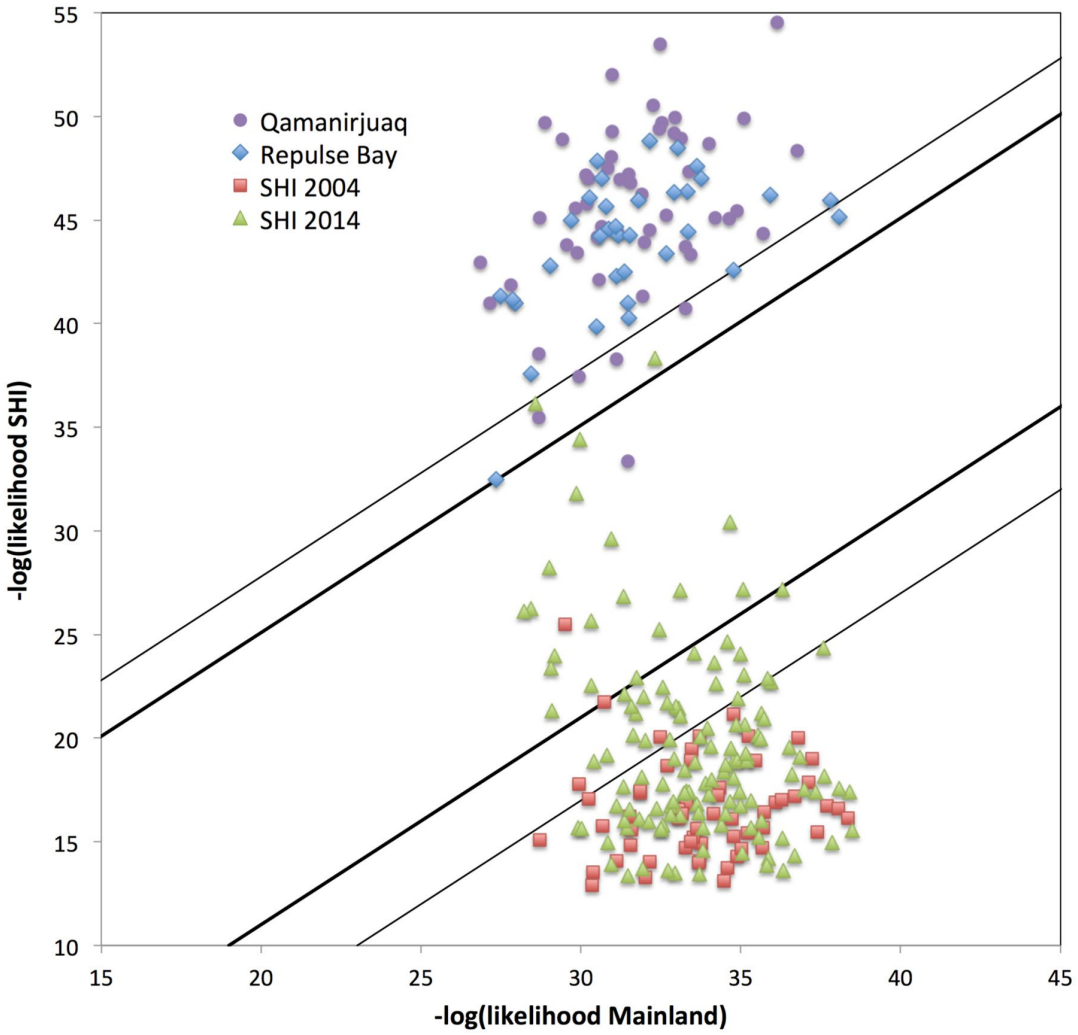


Figure 33. Likelihoods of occurrence based on mainland and (2004) island allele frequencies of caribou according to genetic analysis from different populations and years. Resampling in GeneClass2 indicated that 95% of purebred individuals are expected to have likelihood ratios outside the light lines, while 99% should sit beyond the heavy lines. Individuals between the heavy lines, including C45 (purple circle) and 155 (orange square) have genotypes that are rarer than 99% of individuals of either pure mainland or pure island ancestry. These include seventeen 2013–2014 SHI caribou (Paetkau, 2015).

5.0 CONCLUSIONS & RECOMMENDATIONS

5.1 Aerial Survey Methods

Overall, survey efforts from 1997 to 2023 were relatively precise (CV = 0.055 to 0.087) and were able to track two decades of decline followed by a period of stability within the Southampton Island caribou population. Methods changed over the period, namely from single observer pair configurations from 1997 through 2007, to dependant double observer pair configurations in 2009 to 2015, 2019, and 2023, and to a composite of dependant double observer pair and distance sampling configurations in 2017.

The dependant double observer pair configuration proved to be the most advantageous method, given that front and rear observers switch positions half way through each survey day, and that both front and rear observers are given the prescribed opportunities (see methods) to see the groups while flying along transects. Ensuring that at least one of the observers is experienced for each pair helps ensure unbiased estimates using this approach. The method reduced sightability errors common to the single observer pair method, and provides more precise estimates of wildlife populations. This method was the most effective at correcting estimates when the assumption of perfect sightability was violated. The dependant double observer pair method had other advantages. Incorporating more involvement of community members in research builds local support for the method and survey results, increases training opportunities for observers, improves research capacity in the territory, and incorporates co-management partners in research aspects of wildlife management.

Although the addition of distance sampling methods can further improve survey precision, the task of the observers becomes more challenging, and problems can

arise when using observers with limited experience. In 2017, distance sampling estimates were higher than dependent double observer pair and strip transect estimates. This may have been due to one of the observer pairs not putting enough survey effort to the bins near the aircraft (**Figure 27**) as indicated by different shapes of the detection histograms for the 2 observer pairs. This would cause a negative bias in both strip transect and dependent double observer pair estimates. This illustrates a potential issue with distance sampling, observers spending too much time looking out at further bins which are often easier to view than the closer bins, rather than surveying one strip more thoroughly. The dependant double observer pair method partially accounted for this by also estimating the sighting probabilities of observers near the survey line. In the 2017 case, the observer was identified using dependant double observer records and the error addressed.

Based on our analyses and experience, we suggest that the dependant double observer pair method is the most appropriate method to meet the rigours of quantitative caribou abundance assessments while promoting collaboration with co-management partners. Distance sampling methods, though exceptional in many respects, should only be deployed when experienced observers occupy all observer positions, and, in combination with the dependant or independent double observer pair configuration. If abundance was to decline further on Southampton Island, greater consideration should be given to incorporating distance sampling into survey methods. This may mean working closely with community HTOs to ensure only experienced observers are chosen, to reduce errors which contradict the assumptions of statistical models used in population estimates.

5.2 Herd Trend

The SHI caribou population peaked sometime between 1995 and 2000, and has since then declined by an estimated 9% annually up until the 2017 survey estimate. A probable immigration event sometime between May 2013 and May 2015 likely helped to significantly increase abundance by an estimated 5,082 caribou, however, by May 2017 the population trajectory seems to have fallen back into the 9% annual rate of decline trend that was documented up until 2013. This decline turned around between May 2017 and May 2019, when abundance increased to levels similar to 2015 results suggesting stability across the period. Reasons for the decline detected in May 2017 are likely related primarily to three separate mechanisms including harvest, Brucellosis prevalence, and icing and its effects on forage availability during some winters. The increases detected in May 2019 were likely related to the HTO directed reduction in harvest over that period, and possibly a second immigration event, though studies have yet to confirm such an event. Conditions over the 2-year period were favorable with no indications of icing or other extreme weather events. Any one or part of these metrics could have led to the increases observed in 2019.

Brucellosis likely had little influence on abundance trend until 2004 when disease prevalence reached an estimated 40%. As a result, we believe harvest was the main mechanism of decline between 1997 and 2004. One must keep in mind, however, that the reduction in abundance was the goal during this period, as the population was believed to be well beyond the island's carrying capacity of 15,000 caribou (Ouellet, 1993). Since 2004, both the reproductive disease Brucellosis and harvest combined, were likely the main mechanisms of decline. Unfortunately, at this point we are unable to ascribe which of these two primary mechanisms may have had the greater effect on the decline in abundance of SHI caribou. This being said, by 2005, abundance was still above the hypothesized carrying capacity of SHI (Ouellet, 1993), so the management goal of reducing the SHI caribou population remained unchanged. By 2007, herd estimates were below the estimated carrying capacity of 15,000 caribou, however, declines in abundance seemed to slow between 2007 and 2009, based on survey results. Additionally, Brucellosis prevalence was declining by 2009 and, based on hunter reports, general condition/health was increasing. As Brucellosis prevalence

had been steadily decreasing from 2006 through 2009, and the declines over the same period were slowing, the management goals were amended by the Coral Harbour HTO to reduce the Islands commercial harvesting rather than impact the subsistence harvest. Agreement was reached amongst all co-management partners to suspend the commercial harvest after 2009, in an attempt to further stabilize the decline and maintain an abundance that could support the subsistence harvest. Between 2009 and 2011, however, the caribou population significantly dropped by 5,209 animals, the greatest observed decline over any 2-year period. During this period trends in Brucellosis prevalence reversed and climbed to the highest recorded, and pregnancy rates dropped to below 40%, the second lowest recorded since 2000. Additionally, the unanticipated sale of caribou meat through social media, a new form of commercial harvesting protected as a right under the Nunavut Agreement, began in 2010 and reached levels believed to have exceeded the subsistence harvest over the 2011/2012 harvesting season. It appears that during this period, disease and harvest together were driving the population down. With the formal commercial harvest already stopped in 2009, the Coral Harbour HTO and GN had little option but to apply a TAH to reduce the subsistence harvest as an attempt to control the sale of caribou meat, primarily to Baffin communities, through social media.

The statistically significant increase in the SHI caribou population between May 2013 and May 2015, subsequent decline of an estimated 9% between 2015 and 2017, and 9% increase between 2017 and 2019, has been difficult to explain based on survey results and/or reproductive rates alone. Genetic studies conducted as a follow-up to hunter observations of an immigration event between the winters of 2013 to 2015, suggest a large group of caribou had come onto the island from the mainland. Further analysis found that the caribou that immigrated onto SHI maintained SHI lineage, suggesting an emigration event of SHI caribou off SHI to the mainland prior to the 2013-1014 sampling program but after the 2004 sampling. Further analysis will be required to provide insight as to whether a similar migration event could have had a similar effect on the increases detected between 2013 and 2015, and 2017 and 2019.

However, the genetic work did indicate that sometime between 2004 and 2015, a significant mixing of mainland and SHI caribou occurred.

The recent 2023 results suggest little has changed since 2019 with the population showing stability. More analysis comparing consecutive years of SHI genotypes, with a more geographically broad collection of caribou genetic samples from coastal areas bordering SHI, will be necessary in order to more effectively explore possible mainland connections and reduce potential sampling bias that may be masking actual events. Although it is only a remote possibility, we believe that SHI caribou reproductive potential alone is unlikely to have accounted for the 41% increase estimated between 2013 and 2015, though would not be out of the realm of possibility to have accounted for the increases observed between 2017 and 2019.

5.3 Future Management

Another survey proposed for May 2026 will further assess the maintenance of the detected stability between May 2015 and May 2023. Should a renewed decline be observed, discussions with the Coral Harbour HTO and other stakeholders regarding the consideration of a further reduction in TAH will have to be arranged shortly following the surveys completion, in an attempt to try and safeguard against associated long-term hardship to the residents of Coral Harbour. Should survey results suggest continued stability, or an increase in caribou abundance, consultations on maintaining the current TAH (in the case of stability), or increasing and/or removing the TAH, (wholly dependant on the magnitude of any detected increase), will be discussed with all stakeholders.

The mechanisms driving the changes in abundance observed over the entire survey history of the Southampton Island caribou population are multiple, and difficult to isolate and quantify, suggesting that further research is required. It appears that the

main drivers have been the disease *Brucella suis* Type IV, harvest (with emphasis on the sale of caribou meat through social media), and poor winter weather, primarily in the form of icing events in some years. Clearly the need to continue monitoring disease prevalence in SHI caribou is required if we are to understand present day infection rates and associated productivity for the herd. Recently, hunters have reported fewer caribou with signs of disease, and a noticeable increase in the number of calves observed in 2015 through 2023 which suggests that disease prevalence may be decreasing further. If this is the case, and Brucellosis no longer represents a significant mechanism of decline, then harvest, along with weather, and condition monitoring, should become the focus of future monitoring for the SHI herd. Additionally, more effective means of monitoring the harvest, and any exports of caribou meat off the island will be critical in understanding the true extent of the harvest for both subsistence and meat sales. At present these tools are not available to enforcement officers within Nunavut, suggesting that further thought and required amendments to current harvesting regulations, and perhaps the Nunavut Agreement itself, should be seriously considered by wildlife management organizations and the Government of Nunavut. Attempts to control the sale of caribou meat through social media have failed under the current Management regime and consideration should be given to addressing this issue through amendments to legislation. In recent consultations with Kivalliq community HTOs, all communities expressed a willingness to address the problem in this way, suggesting that some mutual agreement could be reached to more permanently resolve this issue. If nothing is done to monitor this novel and growing mechanism of caribou meat sales, we fear the problem will grow more serious for Nunavut's subsistence harvesters as more and more caribou populations within Nunavut will be managed through the establishment of a TAH.

6.0 ACKNOWLEDGEMENTS

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September 26, 2025

NTI Submission for NWMB Regular Meeting 003-2025, Agenda item 4:

Establishing the Basic Needs Level for Southampton Island Caribou

Overview of NTI's Position

Nunavut Tunngavik Inc. (NTI) maintains its request that the Nunavut Wildlife Management Board (NWMB or Board) postpone the Southampton Island Caribou Basic Needs Level (BNL) decision in order to resume the hearing for this matter that it started in December 2013 and adjourned in 2014. NTI originally made this request in its April 7, 2025 letter to the NWMB and in its follow up letter of July 23, 2025.

NTI understands and respects the NWMB's intent to resume making BNL decisions for terrestrial wildlife and fish. However, there is no urgency to the Southampton Island Caribou BNL decision, particularly since the 2019 Total Allowable Harvest remains in place to address conservation concerns. In contrast, making the BNL decision now will have two detrimental consequences:

1. Proceeding to a major, precedent-setting decision about how to calculate BNLs under Article 5 of the *Nunavut Agreement* without first completing the public hearing that the NWMB paused in 2014 will breach procedural fairness.

2. It will preclude the opportunity for NTI and the Governments of Nunavut and Canada to seek to reach consensus on a proposal, for the NWMB's consideration, on how to interpret and implement Article 5 BNL provisions, and instead put this matter on track for potential litigation, which entails costs, uncertainty and delays.

Procedural Fairness

As the NWMB's July 18, 2024 "Basic Needs Level Policy Memo" (2024 BNL Memo) notes, the question of how to calculate BNLs has been contentious. The Board's own views on the matter have changed over time. Its previous position, prior to the 2024 BNL Memo, was that all Inuit commercial harvesting is included in the BNL, regardless of where wildlife is sold: the NWMB adopted that interpretation of Article 5 in its 2009 BNL decision for Kinngait char (which the Minister of the Department of Fisheries and Oceans then rejected).

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While the 2024 BNL Memo communicates the NWMB's changed position on how to calculate BNLs, it does not explain why it has adopted the Steering Committee's interpretation of what the BNL includes. In NTI's view, this lack of transparency on such a major issue as well as the failure to complete the paused Southampton Island Caribou BNL in order to inform the NWMB's decision, breach the principles of procedural fairness.

The NWMB has also not yet indicated how it intends to address the widely acknowledged shortcoming of the Nunavut Wildlife Harvest Study (Harvest Study), namely its inaccurate capture of Inuit harvest levels, even with respect to the Steering Committee's own direction of which harvests were to be recorded for the purpose of informing BNL calculations. NTI urges the NWMB to invite submissions from all parties on how to establish BNLs that implement Inuit priority access to wildlife, a fundamental right that is critical to Inuit food security, economic security, and well-being in a way that fully achieves the Article 5 objectives in light of the limitations of the Harvest Study.

Creating the Opportunity for Consensus-Building

The NWMB should create space and opportunity for NTI and the Governments of Nunavut and Canada to develop a shared proposal for how to interpret and implement the Article 5 BNL provisions, for the NWMB's consideration. A consensual outcome on this critical aspect of wildlife management would be far preferable to litigation and the costs, uncertainty, and delays associated with resolving the matter in court.

Litigation is not inevitable. Although it is true that NTI, the Government of Nunavut, and the Government of Canada have not yet reached a shared understanding of how to calculate the BNL since the NWMB suspended the Southampton Island Caribou hearing in early 2014, this is not because wildlife co-management partners hit an impasse. Rather, the issue went dormant after the NWMB suspended the Southampton Island Caribou hearing and the parties turned their attention to other wildlife management issues. Over a decade has passed, and it is worth creating a final opportunity for consensus-seeking, especially in light of changed circumstances and the progress that has been made in Crown-Inuit reconciliation. For example:

- NTI, the Government of Nunavut, and the Department of Fisheries and Oceans have established a collaborative and productive working relationship co-developing Nunavut Fishery Regulations, and they are building consensus on challenging Article 5 implementation issues. This gives the parties a strong foundation upon which to explore their respective views, interests and concerns in relation to the BNL methodology. NTI is keen for dialogue, as confirmed in our correspondence to the NWMB and in informal outreach to the Government of Nunavut and the Department of Fisheries and Oceans (DFO). DFO may also be willing to explore this issue, and hopefully the Government of Nunavut is as well, at least once its election period is over.

- In 2023, the federal Crown adopted *Canada's Collaborative Modern Treaty Implementation Policy*, which provides new guidance for interpreting and implementing modern treaties. Sections 3.8, 3.9, 3.10, 4.1 and 4.3(d) of that Policy in particular suggest Canada may be willing to return to the BNL issue and

seek to reach consensus with Inuit on how to interpret and implement those Article 5 provisions.

The NWMB would only need to defer its BNL decision for a relatively brief time period to provide the opportunity for consensus building on this significant Article 5 issue. NTI would welcome the NWMB setting a time limit within the coming months on the window for consensus-seeking to keep this matter on track for decision.

NTI's Requests

In summary, NTI requests that the NWMB:

1. postpone its BNL decision for Southampton Island Caribou;
2. resume the hearing on this matter that it paused in 2014 to hear submissions from its wildlife co-management partners; and
3. encourage NTI, the Government of Nunavut and the Government of Canada to seek to build consensus on a proposal for interpreting and applying the Article 5 BNL provisions in advance of the hearing.

Respectfully submitted by:



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ᓄᓇᕗᕋ ᐅᒪᔨᓪᓗᓂᔨᐊᕐᓂ ᑲᑎᒪᔨᖏᑦ

ᑎᑎᕋᖅᑭᒋᑦ ᐱᓕᕆᑎᑦᓯᐊ ᓄᓇᕗᑦ ᑐᙳᓚᓱᑦ ᑎᒥᖅᑕ ᑎᑎᕋᓪᕈᐅᖅᑕᖅᓯᑦ ᑲᑎᒪᔨᓄᑦ ᔫᑎᐱᓂ 26, 2025–ᖃᐅᑦᓯᐊ, ᐱᓕᓴᐃᔨᓂᑦ ᐊᓪᓚᖅᓴᓂᑦ ᑐᓂᓯᔪᓐᓇᐅᑉ.

ᑲᓇᑕᐅᔪᑦ ᐃᒪᕋᐅᑕᓕᓴᓱᒋᖏᑦ ᓇᓗᓇᐃᖅᓯᔪᓇᖅᑐᑦ ᐱᓇᓯᐊᕋᖅᑕᐅᔪᒪᓪᒪᑕ ᐊᒪᒪ ᐃᓕᑎᕐᐅᔪᒪᓪᓗᑎᑦ ᓰᓴᕋᖅᑕᐅᑲᔪᓂᖅᓂᖅᓴᖅᑎᑦ ᐊᔪᓇᐅᒪᔨᓴᕆᔭᑦ ᓂᕋᑭᓴᓂᔪᐅᔪᕐᓂᑐᖅ ᒪᓕᓪᑐᑐᒍ ᓄᓇᕗᑦ ᐊᖏᓕᖅᑐᒍᖓ. ᐊᖏᓕᕋᖅᑎᑎᓪᓗᓯ ᐅᕋᖅᑕᖅᑎᖅᑲᖅᓂᓕᓴᐊᕋᖅᑕᖅᑐᒍᑦ ᑲᑎᒪᔨᓄᑦ ᑲᑎᕋᖅᑕᖅᑐᒍᑦ ᓇᓗᓇᐃᖅᓯᒪᔨᔪᐊᕋᖅᑐᓂ ᑐᓂᓯᔪᒪᓂ ᐊᒪᒪ ᐊᔨᓰᖏᓕᑦᑐᒍᑦ ᑲᑐᔨᕋᖅᑕᖅᑐᒥᑦ ᕿᑲᓇᐅᓂᖅᓴᓂᖓᓕᒫᑦ ᐱᖓᓱᐊᕆᔭᖅᑐᑦ ᑐᓴᕐᓯᒪᓂᖓᓂ.

ᑲᑐᔨᕋᖅᑎᑎᓪᓗᑎᑦ ᐃᖅᑲᓇᐃᔨᕋᖅᑎᕋᖅᑎᐃᓄᖅᓯᒪᔪᑦ ᓄᓇᕗᕋ ᐅᒪᔨᓪᓗᓂᔨᐊᕐᓂ ᑲᑎᒪᔨᖏᑦᓂᑦ ᐊᒪᒪ ᑲᑐᔨᓂᑎᑦ ᐊᐅᓚᕆᔨᒪᓕᒃᑐᖅᓂᑦ ᑲᑐᔨᕋᖅᑎᕐᓂᑦᓂ ᐃᑲᔪᖅᓯᓪᓗᑎᑦ ᐊᖏᓪᓕᐊᓂᖅᓴᓂᑦ ᐃᓱᒪᓕᐅᕈᓐᓇᖅᑎᑦᓂ ᐱᓕᕆᑎᑦᓯᐊ ᐊᓪᓚᖅᓴᓂᑦ ᐃᓱᒪᓕᐅᕈᓐᓇᖅᑎᑦᓂ ᐃᓱᒪᓕᐅᕈᓐᓇᖅᑐᓂᑦ ᐃᑲᔪᖅᓯᓪᓗᑎᑦ ᐃᓱᒪᓕᐅᕈᓐᓇᖅᑎᑦᓂ ᐃᓱᒪᓕᐅᕈᓐᓇᖅᑐᓂ

ᐅᐻᓇᖅ ᑐᓂᔨᐅᔪᖅ

ᓄᓇᕗᒻᒥ ᐅᒪᔪᓕᕆᔨᖅᑎᐊᖅᓯᑦ ᖃᑎᒪᔨᖏᓪᓗᓂ

ᐅᐻᓇᖅ

ᑐᓴᖅᓴᓂᑦ: **ᐃᓱᒪᓕᐊᖏᓪᓚᔪᖅ: X**

ᐱᔪᓐᑕ: ᖃᓄᖅᑕᓱᖅᓯ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᖃᓇᐃᓕᓕᖅᓂᖏᓪᓕᖏᑦ ᐅᒥᖏᒪᓐᓂᖅ ᐊᐅᓚᑦᑎᓂᖅᑐᒍᑦ ᑭᒡᓕᑭᐅᐅᔪᖅ, ᑭᓚᓕᖅ, MX-07

ᐱᓕᕆᐊᖏᓚᐅᒪᔪᑦ

- ᑭᓕᓂᖅᒥ ᖁᑭᖅᑕᒪᓴᐃᓕᑦ, ᐅᒥᖏᒪᓐᓂᖅ ᐊᐅᓚᑦᑎᓂᖅ Zone MX-07, ᖃᐅᔨᓴᖅᑕᐅᓴᓚᐅᖅᑐᖅ ᒪᖅᑮᓐᓇᓐᓂᑦ ᑭᖏᓕᓵᓐᓂᑦ ᐊᖏᖁᔭᐃᓂᑦ, 2013 ᐊᒻᒪ 2014. ᑐᑭᒧᑦ ᐃᓐᓇᖏᑦᑕᓚᓂᖅ, 1,296 ᐅᒥᖏᒪᓐᓂᑦ ᓄᓴᐃᓚᐅᖅᑐᑦ, ᓇᓚᐅᑦᑕᖅᓯᓚᑎᖅ 10,026 ± 597 (SE) (CV = 6%) ᐅᒥᖏᒪᖃᖅᓂᖏᓂᑦ.
- ᐃᓐᓇᐅᔪᖅ ᑲᑎᒪᓚᑦ ᐊᖏᔪᓵᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᕐᓭᖏᓂᑦ MX-07 ᑕᒪᐸᑭᓚᑦ 400–ᖏᔪᑦ.
- ᐅᖃᐅᔪᕆᓚᒪᔪᖅ ᐊᖏᔪᐅᓭᐅᑦ ᐅᓇᖅᑭᑐᑐᒍᓕᒫᑦᑕᖃᖅᓯᒪᓕᒡᓕ ᐊᑦᑕᔪᓂ ᐊᑐᖅᑕᐅᓴᖅᑐᓂ ᖁᔭᓚᖅᑐᑦ ᐊᖏᔪᓵᑎᓂᑦ ᐱᔪᐅᔪᖅ 0–ᖏᑦᓚᓂᖅ 2019–ᖏᑎᒃᓚᔪ, ᐃᖃᓗᐊᖅᑐᑎᑕᐅᔨᓚ ᐊᖏᔪᓵᑎᓂᖅᑐᑦ ᐊᖏᔪᓯᓚᐅᖅᓯᒪᓂᕆᖅᓱᓂᖅ 15 ᑐᖓᔪᓂ ᐅᒥᖏᒪᓐᓂᖅ 2018–ᖏᓚᐅᖅᓯᒪᖏᒪᑦ ᑕᐃᔨᓯᒪᖏᖅᓂᑦ.

ᐃᓐᓇ ᖃᓄᐃᑕᖏᓪᓕᖏᓪᓕ

- ᓇᐃᔨᐃᓚᐅᖅᑐᑦ ᒪᕐᕉᐃᑕᖅᖃᖏᓪᓕᔪᓕᒃᑯᑦ, ᓂᒋᐊᓂᑦ ᐅᑐᐱᓚ 23 ᐊᒻᒪᓗ ᓄᕐᐱᐱᓚ 1, 2023 ᐊᑯᓂᖏᓪᓕᓂᑦ, ᐅᐊᖏᓇᖏᓕᓪᓕᓂ ᓇᐃᔨᐃᔪᑦ ᐋᒋᔪ 26 ᐊᒻᒪ ᔪᑎᐱᓚ 1, 2024 ᐊᑯᓂᓂᐊᓂᓚᐅᖅᑐᑦ. ᐊᖏᔪᓵᑎᑐᓕᕆᔨᓄᑦᑎᓂᒃᑯᑦ ᖃᓄᖅᑭᐊᖅᑎᑦ ᐃᓚᐅᑦᑎᐊᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᑎᓂᓚᑦ.
- ᐅᒥᖏᒪᐃᑦ ᑕᑯᐅᔪᑦ ᑐᑭᒡᓕᔪᑦ ᐅᓄᖅᑲᐊᓚᐅᖅᒪᑕ (n = 43) ᓇᓚᐅᑦᑕᑭᑎᓚᐅᕆᐊᖅᓴᖅ ᔪᓚᓪᓚᔨᓴᖅᑐᒥᑦ ᔪᑦᓗ ᐅᑦᐱᖏᓐᓇᖅᑐᒥᑦ ᖃᓴᔪᐅᓂᖏᓐᓂᑦ ᑕᐃᒪᐃᓚᐅᖅᑐᑦ ᑕᒪᐦᓇᓗ ᓴᖅᑭᑎᑎᑐᓚᐅᖅᓯᒪᓂᖅ ᓇᓚᐅᑦᑕᑭᓯᐊᓂᖅᑐᑦ ᓇᐃᔨᐅᓇᓴᖏᓐᓇᖅ. ᐅᑭᐅᒥᑦ ᐃᓐᓇᐃᓂᖅ ᐊᑐᖅᑕᐅᓚᐅᖅᓴᖅ ᑕᐃᒪᐃᓐᓇᖅᑐᖃᖅᓯᒪᓂᖅ ᑕᐃᒪᐃᖅᑎᑦ ᑕᒪᐦᓇᓗ ᓴᖅᑭᑎᑎᑐᓚᐅᖅᓯᒪᓂᖅ ᔪᓚᓪᓚᔨᓴᖅᑐᒥᑦ ᓇᓚᐅᑦᑕᑭᓯᐊᓂᖅᑐᑦ ᐃᓚᐅᑦᑎᐊᓚᐅᖅᑐᑦ ᓇᐃᔨᐅᓇᓴᖏᓐᓇᖅ. ᐅᓇᖅᑐᓂᖅᓴᐅᑦ ᓇᓚᐅᑦᑕᐅᔪᑦ 180–ᓚᐅᖅᑐᑦ ᐅᒥᖏᒪᐃᑦ.
- 2013-2014–ᒥᑦ ᑎᑭᔪᓚᒍ ᐅᑭᐅᖅ 2023-2024–ᒧᑦ ᐅᒥᖏᒪᐃᑦ MX-07–ᒥ ᐅᓇᑭᓐᓂᖅᓴᓪᓕᔪᖅᑕᖅᓯᒪᓕᕆᔨᑦ. ᑎᒥᐅᑦ ᐋᓐᓂᐊᑭᑎᑎᑲᓕᐃᑦ ᖃᓇᐃᓕᓕᖅᓂᖏᓪᓚᓪᓚ (ᐃᔪᓂᐅᖏᓚᔨᐊᖅᑐᑦ, ᔪᑦᑎᑲᐃᑎᑦᑐᑦ ᔪᑏᐱᔪᐃᓕᑦ, Erysipelothrix, ᖃᓂᔪᓚᖏᓂᑦ, ᑭᒍᑎᖃᑎᐊᖏᑦᑐᑦ, ᓇᑭᐱᐅᑎᐱᖃᖏᓚᔨᐊᖅᓂᖅ), ᐅᓇᔪᓂᖅᐱᐅᓕᖅᑐᑦ ᐅᓂᐅᖅᑎᕆᔨᓴᕆᔨᑦ ᐊᐅᓯᐃᓕᑦ ᐊᒪᖅᑯᓚᓪ ᑕᒪᐦᓇᐊᖏᓪᓕᖏᑦ, ᓄᑦᑕᖏᓂᖏᑦᓚᓪ ᐅᒥᖏᒪᐃᑦ ᓇᓇᖅᑕᔪᓚᑦ ᓄᓇᓕᖅᑐᑦ ᐊᐅᓚᖏᑦᑐᓂᑦ ᐊᐅᑐᖅᑐᑦᑎᓇᓂᑦ ᖃᐅᔨᓴᖅᑎᓂᓚᑦ.

ᑐᓴᐊᒻᒪᔪᖅᓴᖅᑐᑦ

- ᑭᖓᓪᓕᖅᑰᖅ ᖃᐅᔨᓴᕐᓂᖓᓪᓗᑎᑦ ᐅᓂᐅᒃᑲ ᑐᓂᐅᖅᑲᖅᑕᐅᓕᐅᖅᑐᖅ ᓄᓇᕗᑦ ᑐᖓᔪᖅᑐᕐᓄᑦ, ᖃᐅᔨᓪᓗᖅᑐᑦ ᐊᖓᔪᓂᐊᖅᑎᑦ ᑲᑐᔾᔨᖅᑎᑎᖅᑕᑦ (KAA), ᐃᖃᓗᓕᔅᑐᑎᐊᖅ ᐊᖓᔪᓇᔨᑐᑕᓂᕆᔨᒃᑯᑦ, ᐊᓯᒪ ᖁᑎᖏᑕᐅᑦ ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖅᑕᑦ (KRWB) ᐃᓕᓴ 20, 2025–ᒥ.
- ᑕᐃᔨᓴᒪᓂ ᐊᐸᑎᓕᕆᔨᒃᑯᑦ ᖃᐅᔨᓴᖅᑎᖓᓱᒃ ᑲᑎᒪᖃᑎᖅᑲᒪᒪᓕᐅᖅᓴᖅ ᖁᑎᖏᑕᐅᑦ ᐅᒪᔨᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖅᑕᑦᓂᓴᒃ, ᐃᖃᓗᓕᔅᑐᑎᐊᖅ ᐊᖓᔪᓇᔨᑐᑕᓂᕆᔨᒃᑯᖅᑕᓂᓴ ᐊᓯᒪ ᖃᓄᖏᓪᓗᑕ ᐅᖃᖅᑐᑎᑦᑎᓐᓂᒃ ᖃᐅᔨᓴᖅᑭᑐᐃᓐᓇᓂᑦ ᐊᓯᒪ ᐊᐅᓪᓚᑎᓂᓪᓗᑦ ᐱᓕᕆᔨᖓᔪᓐᓇᖅᑐᓂᑦ ᐱᓇᓱᐊᕐᓯᓂᖅᓴᒃ ᐊᑦᑕᖅᑐᓂᒃ.
- ᓄᓇᕗᑦ ᑲᐊᒪᔨᒃᑯᑦ ᐃᖃᓗᐃᔭᖅᑎᖅᑕᑦ ᑲᑎᖅᑎᖅᑕᓚᐅᖅᓴᑦ ᖃᓄᐊᕋᓚᓐᓂᒃ ᐃᓚᒋᔭᐅᔪᓂᒃ ᐃᖃᓗᓕᔅᑐᑎᐊᖅᑎᒋᐊᖅᓂᒃ ᐊᐃᑦᓚ 3, 2025–ᒥ, ᑭᓱᐊᓂ ᑲᑎᒪᔨᖅᑐᖅᑕᐅᔪᑦ ᐅᓄᖅᑲᓕᐊᓚᐅᖅᓯᒪᑕ ᑲᑎᒪᔪᑦ ᐃᓚᒋᔭᐅᔪᓐᓇᖅᑐᓂᒃ.
- ᐊᐸᑎᓕᕆᔨᒃᑯᑦ ᐃᖃᓗᐃᔭᖅᑎᖅᑕᑦ ᐅᖃᖅᑎᑎᖅᑕᓚᐅᕆᔪᓂᑦ ᖃᓄᖏᓪᓗᑎᑦᑎᓐᓂᐅᓂᒃ ᐋᒥ 18, 2025–ᒥ ᐅᖃᐅᓯᖃᖅᓯᒪᔾᔪᑐᓗ ᖃᐅᔨᓴᕐᓂᖓᓪᓗᑦ ᖃᐅᔨᓴᒪᔭᐅᓕᖅᑐᓂᒃ. ᖃᓄᖏᓪᓗᑎᑦ ᐊᖓᔪᓇᐃᑦᑕᑦ ᑲᑎᒪᔨᖅᑕᑦ ᓇᓗᓇᐃᖃᓯᒪᔨᖅᑐᓂᒃ ᐅᒥᖓᒪᐃᑦ ᑭᓕᓂᓪᓗᑦ ᐃᓴᓇ ᐅᓄᖅᑲᓗᐊᓂᖅᑐᓂᒃ.
- ᑭᖓᓪᓕᓂᒥᒃ ᑐᑭᓯᒋᐊᖅᑎᑎᓴᓂᓪᓂᓪᓗᑦ ᐃᓂᒃᓴᓕᐅᖅᑐᖅᑕᐅᖅᓯᒪᔪᖅ ᐊᐃᑦᓯ 19, 2025–ᔪᑦ ᐃᖃᓗᓕᔅᑐᑎᐊᖅᑎᒋᐊᖅᑎᑦ ᐊᓯᒪᔨᖅᑕᓚᐅᕆᔨᒃᑯᑦᓴᑎᒃ ᐅᓄᓂᖅᓴᒃᑯᓂᒃ ᐊᖓᔪᓇᔨᒃᑯᐸᒃᑐᑐᓂᒃ ᐅᐸᒃᑐᑐᖅᑲᓴᖅᓯᒪᓕᖅᓱᓕ ᑕᐃᔨᓴᒪᓂ ᐊᐃᑦᓚ 3-ᒥ ᐅᐸᔪᑎᔨᓇᓕᐅᖅᓯᒋᐅᑐᓄᑦ. ᑕᓐᓇ ᑲᑎᒪᓂᖅ ᑲᔨᓴᓕᐅᖅᓯᒋᐅᑐᖅ ᐊᑦᖁᑦᑕᐅᓴᒪᔨᒃᑯᑦ.

ᐊᑐᓕᖅᑯᐅᔨᑦ

- ᑐᖅᑲᓚᕆᖅᓴᓂ ᖃᖅᑎᑕᑎᐊᑦᓴᖅ ᖃᖅᑎᑕᕐᔨᖅᑎᖅᓴᓂ ᖃᐅᔨᓴᕐᓂᓱᓴᐅᓕᐅᖅᑐᑐᓂᒃ, ᐃᓕᓴᖅᔨᑐᑐᖅᑲᓂᕆᔨᒃᑯᑦ/ᐃᓄᐃᑦ ᖁᐅᔨᓴᒪᔭᑐᑐᖅᑕᑎᒃᑯᓂᒃᑎᑦᑕᖅᑐᕐᔨᓕᕐᓂᑦ ᐱᓕᓱᐊᕐᓴᐅᑦ <ᖅᓇᔪᑕᐅᓂᓕᒫᑦᓂᒃ, ᓄᓇᕗᑦ ᑲᑎᒪᔨᒃᑯᑦ ᐊᑐᓕᖅᑯᐅᔨᑦ ᐃᓕᓴ ᑲᑎᒪᔭᑦ ᖃᖅᑎᕐᕋᐅᓇᖅᑎᑕᐅᓂᖅ 400–ᓂᒃ ᐅᒥᖓᒪᖅᓂᒃ ᐅᓄᕐᓂᓗᓕᐊᖅᑕᐅᖅᑐᕆᔨᓴᓕᑎᒃ ᑕᓕᓕᒪᓄᑦ ᐅᒥᖓᒪᖅᓄᑦ.

ᐅᑐᓗᓕ ᓄᓇᕗᑦ ᐱᖃᓐᓇᕐᓂᐊᕐᓂᖅ
Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

ᐊᓂᒍᓐᓇᐃᓕᓴᐅᑎᑦ
Department of Environment
Avatiliqiyikkut
Ministère de l'Environnement

Muskox (*Ovibos moschatus*) abundance estimate on Victoria Island Group, MX-07

2023 - 2024

Report

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Department of Environment
Government of Nunavut
Kugluktuk, NU

TABLE OF CONTENTS

LIST OF FIGURES 3
LIST OF TABLES 4
ABSTRACT 5
INTRODUCTION 7
OBJECTIVES 9
MATERIALS AND METHODS 9
Study Area 9
Integration of Inuit Qaujimajatuqangit (IQ) in the Survey Design 11
Design of Distance Sampling Surveys 11
Survey Design 11
Distance Sampling and Field Setting 13
Data Analysis 14
Muskox Group Size 14
Muskox Herd Distribution, Dispersion 14
Minimum Count of Muskoxen Observed 14
RESULTS 15
Survey Conditions 15
Muskox Group Size 17
Muskox Herd Distribution, Dispersion 17
Minimum Count of Muskoxen Observed 19
DISCUSSION 20
ACKNOWLEDGEMENTS 23
REFERENCES 24

LIST OF FIGURES

Figure 1: Map of the Victoria Island Group Muskox Management Zone, MX-07. A Digital Elevation Model (DEM) was used to depict the very subtitle change in elevation from 0 green to 760 red in metres (Arctic DEM, Porter, 2023). 10

Figure 2 :Stratification and transect lines design for the 2023 Dolphin and Union caribou survey (dark grey) and 2024 muskox abundance survey (light grey) of the Victoria Island Group, MX-07..... 12

Figure 3: In the south section of MX-07, flight tracks (grey) done between October 23rd and November 1st, 2023. The 2024 flight tracks were represented by a different colour for each survey day between August 26th to September 1st, 2024. 16

Figure 4: Frequency distribution of muskox group size (n = 48) observed during the 2023 and 2024 surveys combined, where the bins ranged from 1 to 19. 17

Figure 5: Geo-referenced locations of muskox herd seen “on” and “off” transects between October 23rd and November 1st, 2023 (Strata 2023) and between August 26th to September 1st, 2024 (Strata 2024)..... 18

LIST OF TABLES

Table 1: Summary of the survey design with the number of transect lines, distance between transects, transect length, and the percentage of coverage. 13

Table 2: Summary of the survey progress between August 25th to September 2nd, 2024, as well as the general weather encounter affecting visibility. 15

Table 3: Summary of the minimum count of muskox in each surveyed stratum based as to whether the observations off the transect strip were included or not. 19

ABSTRACT

Victoria Island (Canada) was once the habitat of a dense muskox (*Ovibos moschatus*) population. In the Muskox Management Zone Victoria Island Group, MX-07, the number of muskoxen declined to 10,026 in 2013-2014. In 2023, there was a need to conduct an abundance estimate, as hunters have not seen sign of recovery and were adapting to this new reality by changing their harvest location to the Canadian mainland. An aerial survey was designed using Distance Sampling paired with a dependent double observer configuration. The aim was to have a new abundance estimate for this management zone. This vast landmass was surveyed in two parts, the south section between October 23rd and November 1st, 2023 and the north section between August 26th to September 1st, 2024. In total, 48 muskox groups were seen, and the average group size was 6.13 (sd = 4.11) excluding lone bulls. There were 17 lone bulls. The number of muskoxen seen on transect was too low to derive an abundance estimate (n = 43). Most of the muskoxen were observed in Stratum 13 with 11 groups and a minimum count of 47 muskoxen. For the totality of the management zone, the minimum count was 180 muskoxen in 60,384 km^2 of survey area for a total density of 0.003 muskox/km^2 during the 2023-2024 survey. The health conditions of the muskox, increase number of grizzly bear and wolf sightings, migration of muskox off the island to the mainland might have contributed to the observed decline of muskox in MX-07. The limited number of muskox left is worrisome, and warrant more research to explain the cause of decline and work with co-management partners to seek opportunities to promote recovery.

The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment. Reference to any specific type of gear and/or equipment does not mean that it is endorsed by the authors or the institutions they represent.

INTRODUCTION

Two subspecies of muskox (*Ovibos moschatus*) roam across the Canadian Arctic; the subspecies *Ovibos moschatus wardi* found in the Arctic Archipelagos and the subspecies *Ovibos moschatus moschatus* found on the Canadian mainland. Hunters from Kugluktuk and Cambridge Bay (Nunavut) can differentiate between these two muskoxen types. *Ovibos moschatus wardi* has a slight physical differentiation with their whiter faces, saddles, stocking horns, as well as longer tooth row and styles on the upper molars (van Coeverden de Groot, 2001). This separation is supported by recent genetic analysis that showed distinct clustering between these two groups (Lok et al., 2024). However, muskox overall have a very low genetic diversity and this genetic clustering between the two subspecies remains unclear compare to other subspecies with higher genetic diversity. More studies are needed to confirm the designation of muskox subspecies (Lok et al., 2024)

Because of the immensity of Victoria Island (Northwest Territories and Nunavut), the second largest arctic island, it is difficult to estimate wildlife abundance. Until 1976, no systematic survey for wildlife, including muskox, was done on Victoria Island (Spencer, 1976). Surveys were usually limited to a very low coverage and only a part of the island was surveyed at a time, because of its size and associated logistical challenges. In August 1980, the first transect survey occurred on the southeastern part of Victoria Island, by flying only three percent of its coverage. They counted 91 muskoxen to derive an estimate of 1,760 (CV = 68%) (Jakimchuk and Carruthers, 1980). Three years later, the same area was surveyed at a coverage of 20%, which provided an estimate of 3,300 muskoxen (CV = 10%) (Jingfors, 1984). These results were consistent with local hunter reports of an increasing trend in muskoxen.

To monitor the change in muskox abundance overtime, three surveys in 1988, 1993, and 1999 occurred in southeastern portion of Victoria Island (Gunn and Patterson, 2012). The 1988 survey estimated 12,372 ± 1,064 (SE) muskoxen in a study area of 44,390 km^2, while in 1993 and 1999 the estimates were 12,563 ± 1,254 (SE) and 18,290 ± 1,100 (SE) muskoxen respectively in a study area of 39,100 km^2 (Gunn and Patterson, 2012). Muskoxen density was used as an indicator of trend, and a positive change in density was found significant between 1993 and 1999 (Gunn and Patterson, 2012). The study area was not representative of the real population boundary and interpreting this trend was ambiguous, as it was impossible to refute possible immigration from the rest of the island into the study area.

In fact, muskoxen were documented to inhabit the southwestern and the northeast of Victoria Island. In 1983, a survey in the southwestern part of Victoria Island resulted in an estimate for the area of 135 muskoxen. At this location, muskox appeared sparsely distributed on the landscape and at a very low density (Poole, 1985). Muskoxen were also known to be distributed to the northeast of the island. So in August 1990, the east and south of Hadley Bay, as far south as Washburn Lake, was systematically surveyed. On transect, 700 muskoxen were counted for an estimate of 5,452 ± 521 (SE) muskoxen (Gunn and Lee, 2000). There again, the muskoxen were relatively evenly dispersed, and the density appeared lower than the southeastern part of the island. The heterogeneity of vegetation cover and the climatic variations across Victoria Island might have been significant factors in determining muskox density, but such did not appear to be playing limiting factors to their distribution extent.

In 2015, the harvest regulations were repealed and new comprehensive regulations package was brought into force, implementing the Nunavut *Wildlife Act* (S. Nu. 2003, c.26). This included new muskox management zones (R-026-2015) that were defined via extensive consultations with co-management partners and approved via the Nunavut Wildlife Management Board processes outlined in Article 5 of the *Nunavut Agreement*. In 2013 and 2014, MX-07 was surveyed in two consecutive years. On transect, 1,296 muskoxen were counted, which resulted in an estimate of 10,026 ± 597 (SE) (CV = 6%) muskoxen. Muskox density across the island varied from 0.0617 to 0.1321 muskox/km^2, where the density was consistently higher in the southeastern part of the island (Leclerc, 2015a). Since it was the first time that such an extensive area was surveyed, it was difficult to compare this result with the previous surveys of Victoria island. Nonetheless, the 2013-2014 estimate highlighted a decline in the muskox abundance, which was consistent with local observations.

While abundance surveys are usually point-events characterized with change in survey area and method through time, traditional knowledge provides a longer and smoother time series. Traditional knowledge collected from Cambridge Bay allowed to represent the muskox abundance from 1960 to 2014. While the muskox abundance was low in the 1960s, it slowly increased to peak around 1990 to 2000, and soon after, declined to reach the lowest number in 2014 (Tomaselli et al., 2018). The participants described an 85% decline in muskoxen. Reasons behind this decline included climate changes, an increase in predators, and a decrease in health conditions. Local hunters have observed muskox carcasses on the land (acute mortalities), which they associated with disease outbreaks (Tomaselli et al., 2018).

Since 2013-2014, no additional aerial survey occurred. Since then, a continuous decline in muskox number was observed by local hunters with no sign of recovery. Throughout a community-based health surveillance program (2012 to 2024), emerging diseases were detected, such as brucellosis (*Brucella sp.)* and orf virus (genus *Parapoxvirus*), while low pregnancy rates and deficiencies in important trace minerals, such as Selenium, were documented. All these health conditions negatively affected the reproduction and survival rate of individuals, which affected the population dynamics (Tomaselli et al*.,* 2014; Dalton et al., 2024; Aguilar et al., 2024; Hee et al., 2024; Dickinson et al.,2025). In response to the low muskox abundance, the Ekaluktutiak Hunters and Trappers Association (EHTA) has been proactive by voluntarily stopping sport harvesting in MX-07 and establishing a period of harvest moratorium (April 15th to July 30th) in an attempt to increase calf survival (EHTA pers. com). With the muskox scarcity in MX-07, both the ETHA and Kugluktuk Angoniatit Association (KAA) have shifted their muskox harvest to the Central Kitikmeot Group, MX-11, to ease pressure and to promote recovery. The KAA have stated that the muskox population follows a cyclical pattern of population highs followed by population lows, like caribou, and that the population is at present very low (KAA pers. com.).

Since the documented decline in muskox numbers identified in 2013-2014, the decline seems to have continued. As a result of suspected continued population decline, we completed an aerial survey of MX-07. The results from this survey will be used to inform actions to address the muskox recovery on Victoria Island.

OBJECTIVES

The main objectives of this study were:

1) To establish a new abundance, estimate of MX-07 and identify the population trend.
2) To review the current management regime and propose new recommendations if necessary.

MATERIALS AND METHODS

Study Area

The Victoria Island Group, MX-07, is surrounded by water: Hadley Bay to the north, M'Clintock Channel to the east, and to the south: the Dolphin and Union Strait, Coronation Gulf, and Dease Strait (Figure 1). The boundary of the west side of the muskox management zone is defined by the Northwest Territories border. The total land area of MX-07 is 141, 677 km^2 including Stefansson Island and Jenny Lind Island or 136,430 km^2 not including island. Muskox were previously distributed widely over most of the management zone, where variation in muskox density was observed across the island, because of the heterogeneity of the landscape (Poole, 1985, Gunn and Lee, 2000, Gunn and Patterson, 2012, Leclerc, 2015). Victoria Island topography is characterized by its predominantly low elevation that ranges from 100 to 200 metres above sea level (asl), except the Shaler Mountains in the central part of Victoria Island, where the elevation reaches up to 760 metres (Environment Canada, 1995). After the Wisconsin glaciation, the retreat of the Laurentide ice sheet left many glacial landforms (i.e. drumlins, till plains, moraines, eskers, dried river valleys and meltwater channels) on Victoria Island (Saarela et al., 2020). The southwestern coast of Prince Albert Sound, the northeast coast of Hadley Bay and an extensive area between Namaycush and Washburn lakes are xeric and sandy, with very poorly vegetated to unvegetated areas. The ground is rough and mostly composed of rocky and gravelly substrates. In this denuded landscape, arctic aven (*Dryas integrifolia*) is usually the dominant species that colonizes scoured substrate, as this shrub grows in cold and wet habitat (Environment Canada, 1995). Hygric sedge and sedge fen communities are found on the coast adjoining the M'Clintock Channel and Lady Franklin Point. The landscape of Victoria Island is well-known for its many ponds, lakes, and rivers and the vegetation north of Namaycush Lake seems to be restrained to the edge of water bodies and drainage areas.

There are two bio-climate subzones on Victoria Island, Subzone C and D (CAVM Team, 2003), and the boundary between these two zones runs from southeast to northwest approximately. To the north, zone C, the cold summer temperature of 5 to 7ºC limit the cover of vascular plants (5 to 50%) and the species richness from 75 to 150 species of local floras (Saarela et al., 2020). The warmer summer temperature of the southern zone D, 7 to 9 ºC, allows for an increase in cover of vascular plants (50 to 80%), as well as an increase in species of local floras from 125 to 205 species (Saarela et al*.,* 2020). Plant communities live primarily on a calcareous substrate that is found across the island.

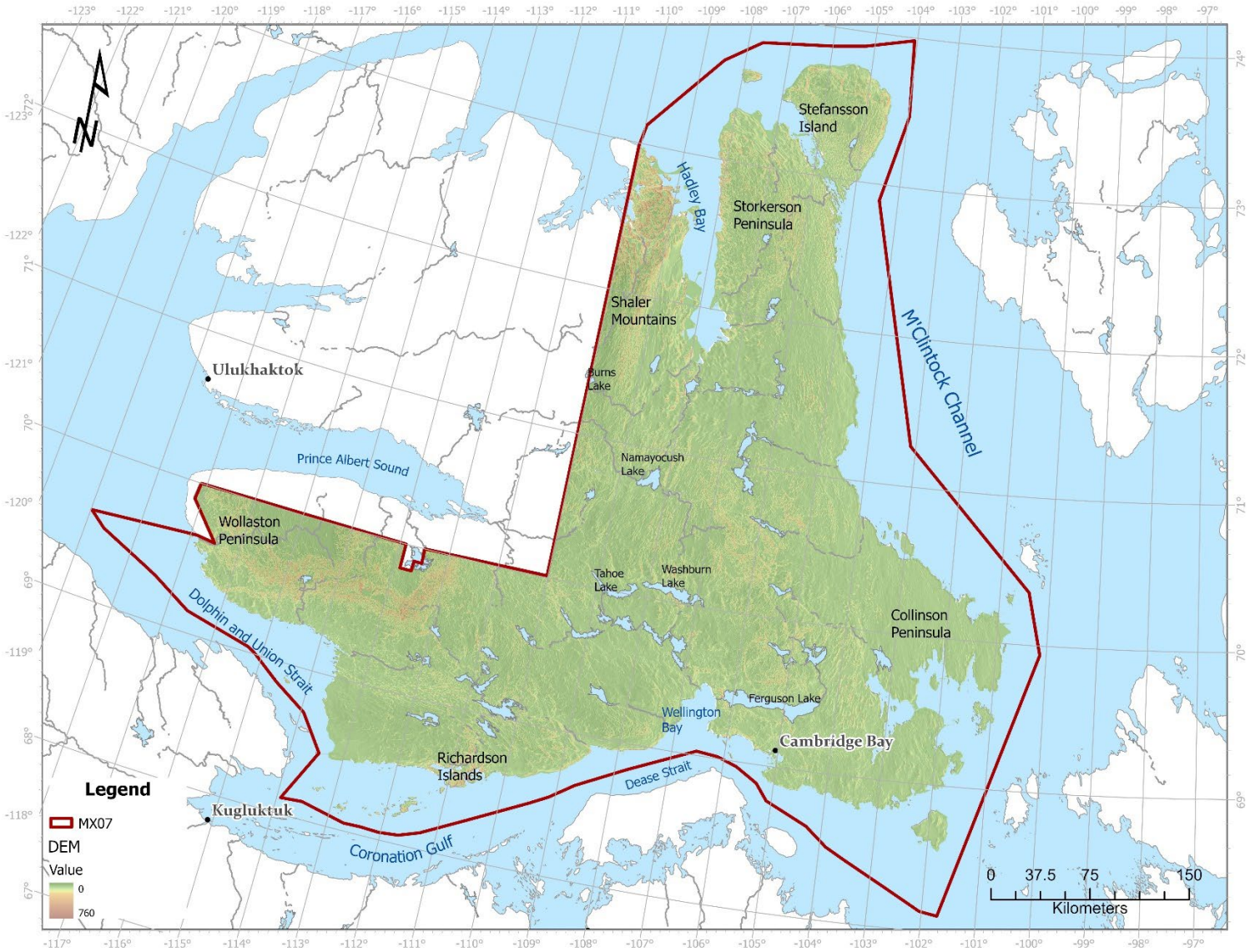


Figure 1: Map of the Victoria Island Group Muskox Management Zone, MX-07. A Digital Elevation Model (DEM) was used to depict the very subtitle change in elevation from 0 green to 760 red in metres (Arctic DEM, Porter, 2023).

Integration of Inuit Qaujimajatuqangit (IQ) in the Survey Design

Meetings with the EHTA, the KAA, and the Kitikmeot Regional Wildlife Board (KRWB) occurred on January 5th and August 7th, 2024. During these meetings, the KAA suggested that there was no need to survey MX-07, since *IQ* indicated that the muskox number was at the low point in their population cycle. They suggested that surveying MX-07 in totality would be an expensive endeavour where the funds could be used for alternative programs. As an alternative to a full island survey, information from the 2023 Dolphin and Union caribou (DU) herd survey was used in combination with a new survey in 2024 covering the remaining portion of the island that was not surveyed in 2023. Although the focal species for the DU survey was caribou, others species observed, such as muskox, were recorded as non-target species (Leclerc et al*.,* 2024). Both Hunters and Trappers Organizations were not comfortable with a multi-species survey approach, but saw advantages of using the muskox observations from 2023 to derive a muskox estimate and to avoid flying the same survey area a second time at lower effort (10-15%) to get the same results. All parties supported this modified survey methodology.

The north part of Victoria Island, the unsurveyed area during the 2023 DU survey, was completed in August-September, 2024. Since hunters seldomly travel to the north of Victoria Island, there was more confidence in using the muskox distribution from the last 2013-2014 survey to stratify this area than rely solely on hunter observations. There, the survey coverage was originally set at ~36% in the three main strata (12, 13, and 15) in aim to have a minimum of 60 muskox observations to derive an estimate. However, after discussing with the air charter company, the coverage was lowered to approximately 25% to 27%, because of the remoteness of the survey location and associated logistic challenges involved. Still, the coverage was set higher than the 1990 and 2014 surveys, that had 13% and 10% coverage, respectively (Gunn and Lee, 2000; Leclerc, 2015). In the final survey design, a similar coverage for Stratum 12, 13, and 15 was established. In addition, the orientation of the transect lines for the Stratum 15 was adjusted based on recommendations from EHTA. By working closely together, early in the design phase, *IQ* and all perspectives were considered to ensure confidence in the results and more effective management outcome.

Design of Distance Sampling Surveys

Survey Design

Because of the immensity of MX-07, the management unit was surveyed less than a year apart in two separate surveys. The southern strata (Figure 2, dark grey area) were surveyed in October-November, 2023. This area was divided into 11 strata of unequal coverage allocations, where a stratum had a minimum of 8 to 59 transect lines, or an effort varying from 21% to 74% (Table 1). The coverage allocations and direction of the transect lines were done based on the best available knowledge of the Dolphin and Union caribou herd fall distribution, to maintain the orientation of the sides of the strata, and to assure transects of relatively equal length (Buckland et al., 2001). The remaining part of the management zone was surveyed in August-September, 2024 (Figure 2, light grey area). The northern area was separated into four strata and was labeled stratum 12 to 15 (Table 1). The lowest effort (12%) was allocated on the coast of M'Clintock Channel, while the

three others strata had relatively the same percentage of effort, about 25% to 27%, as muskox density was expected to be consistent across these areas. Stefansson Island was not included, as the soil was made up principally of sand, lacked vegetation, and consequently no muskox was observed in 2014 (Leclerc, 2015). The package *dssd* in the R Statistical Platform (R Development Core Team 2009, Marshall and Rexstad, 2023) was used to draw the transect lines with a random starting point in each stratum.

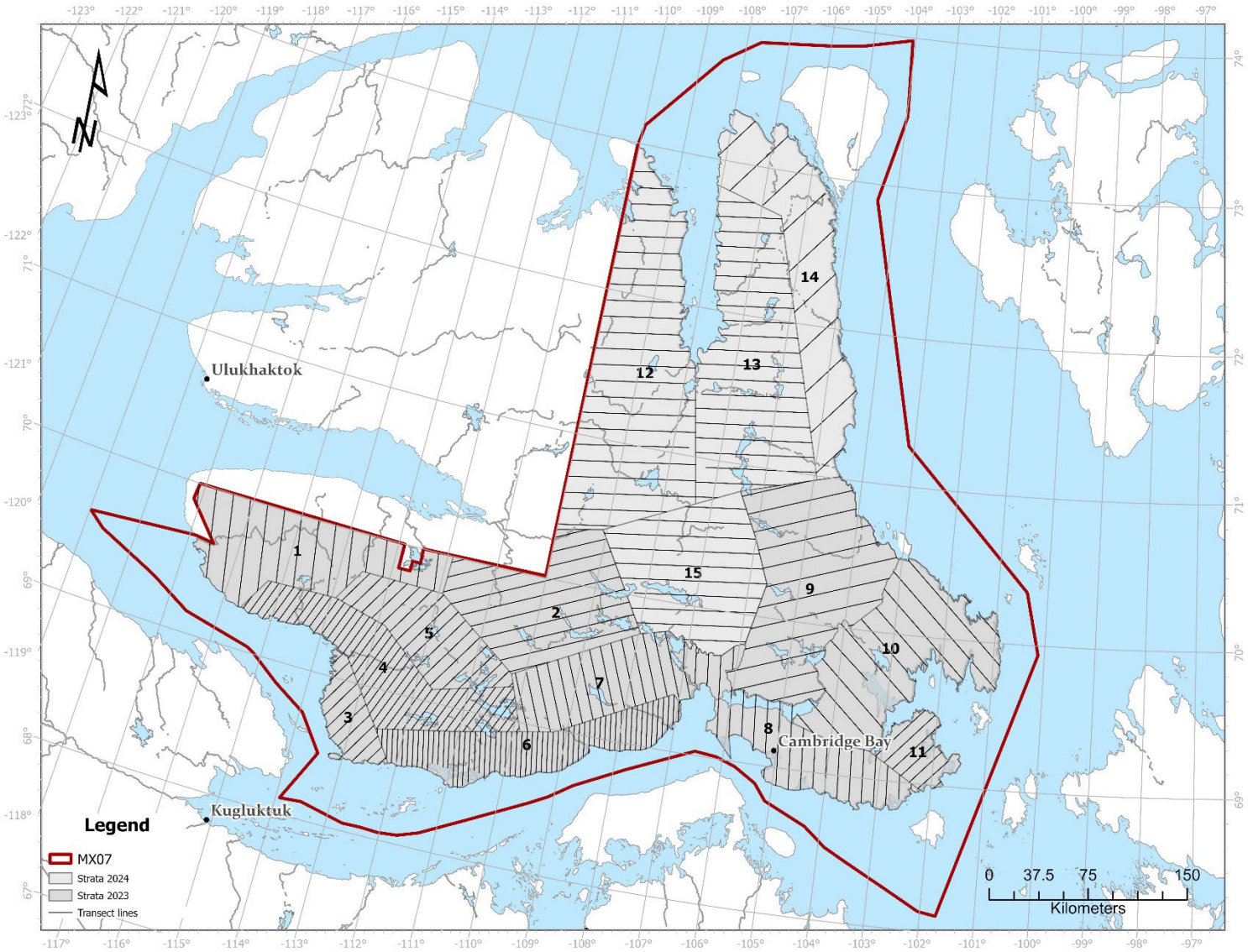


Figure 2 :Stratification and transect lines design for the 2023 Dolphin and Union caribou survey (dark grey) and 2024 muskox abundance survey (light grey) of the Victoria Island Group, MX-07.

Table 1: Summary of the survey design with the number of transect lines, distance between transects, transect length, and the percentage of coverage.

Year	Stratum	Area (km)	Number of Transects	Distance between Transects (km)	Transect Length (km)	Coverage (%)
2023	1	9,328	16	13.5	1,558	21.89
	2	9,787	10	10	1,383	29.50
	3	2,375	11	8	495	36.36
	4	8,101	42	5.5	1,990	53.94
	5	5,622	17	8.5	958	34.93
	6	6,972	59	4	2,258	74.32
	7	6,581	16	8.5	1,036	35.24
	8	7,577	21	9.5	1,310	30.67
	9	13,297	12	14	1,336	21.51
	10	11,064	12	14	1,177	21.30
	11	2,151	8	8	426	38.07
2024	12	18,340	24	12	2,113	24.89
	13	10,415	22	11	1,890	27.07
	14	10,871	10	25	946	11.97
	15	13.950	12	11	1,252	26.82
	---	136,430	292	---	20,128	---

Distance Sampling and Field Setting

The 20,128 km of transect lines were flown by Twin Otter at a speed of 160-180 km/hr and at an altitude of 122 metres using a radar altimeter to maintain a constant altitude, while following the topography of the terrain. Using Gunn and Patterson's (2012) recommendations, the Distance Sampling method was used to decrease the bias of the estimate and compare the density estimated between strata. The wing struts were divided into five bins marked with a different color of electric tape to identify the distance of the observations. The bins represented a distance from the survey line of 0-200 metres, 200-400 metres, 400-600 metres, 600-1,000 metres, and 1,000-1,500 metres. Precaution was taken to extend the 0 mark beyond the blind spot directly under the aircraft and the total sampling distance was three kilometers (Leclerc et al., 2024). Observations made farther than 1,500 meters or during the ferry were recorded as "off" transect. A dependent double-pair observer configuration was used, where the experienced trained observers work together to detect and confirm their observations (Buckland et al., 2005). Muskox group was used as the sampling unit, and if the group moved, the original distance before the movement was recorded. The number of adults and calves were both recorded. In 2024, other species sightings of wolverine (*Gulo gulo*), wolf (*Canis lupus*), polar bear (*Ursus maritimus*), and caribou (*Rangifer tarandus*) were also acknowledged and logged. Two data recorders, left and right, were recording the observations of the front and back observers sitting on their respective side of the airplane. Recorders used ArcGIS Survey123 application to record date, time, location, species, group size, distance of the observations, and survey conditions (snow cover, presence of fog, and cloud cover). The data were secured and saved to the online cloud to be retrieved later by logging on to the password protected website.

Data Analysis

Since the field design and the data collection were consistent in 2023 and 2024, the two muskox observation datasets were merged for the analysis. The "on" and "off" transect observations (n = 48) were used to increase the sample size in order to infer on the muskox spatial distribution pattern and group size across the MX-07 muskox management zone.

Muskox Group Size

Muskox group dynamics is a fundamental part of the ecology and evolutionary process of the species, including mating and predator protection. Risk of predation in muskox, an open-land species, has been found to affect group size as it promotes fitness and protects offspring by decreasing the probability of capture (e.g. group formation) (Heard, 1992). Due to the importance that group size plays for this social ungulate, the variation in group sizes encountered was represented by a histogram. Group or cluster was defined by muskox occurring closely to each other, where the extent of the group could be well defined by the lack of animal in the surrounding. The number of lone muskoxen, assumed to be bulls, was also reported, but not included in the group size averaging. In the event that only one of the two observers detected the group that count was used, if both observers agreed on the same amount of muskox that count was used, and if both observers did not see the same amount of muskox an average of the two counts was done to determine the group size to keep the group size unbiased. The minimum and the maximum of group size were reported and the mean group size was calculated. The proportion of calves per adult muskox was provided for 2024 only.

Muskox Herd Distribution, Dispersion

The observed muskox locations, where all the muskox groups ("on" and "off" transects) were seen in 2023 and 2024, were at priory plotted on a map using ArcGIS Pro version 3.3 (ESRI, 2024) to determine the distribution (presence and absence) of muskox in MX-07. Since muskox dispersion on the landscape is usually influenced by the quality of the habitat and physiological needs (Wilkinson et al., 1976, Smith 1989, Gunn 1990), the spatial pattern of the group distribution was further investigated by using a spatial point-pattern analysis test, Moran's I. By determining the distance from one group relative to the others, it was possible to determine if the pattern expressed by the group dispersal on the landscape depicted a random or nonrandom pattern. If the pattern was not random, this pattern can be further described as clustered or dispersed (Oyana, 2020). The Moran's Index value was calculated, as well as the z-score and p-value.

Minimum Count of Muskoxen Observed

Since the number of observations on transect was low in all strata and in the study area (n = 43), there were not enough observations to parameterize an individual model for the muskox, as a sample size of at least 60 to 80 observations is necessary (Buckland et al., 2005). Therefore, it was assumed that all muskoxen were detected on transect, and the muskox total minimum count were reported as muskoxen observed within each stratum and in the management zone. Predators, grizzly bear (*Ursus arctos*), polar bear, and arctic wolf, were reported using the predator index

(Heard, 1992). The predator index reported all sighting against the total number hours of flying per 100 hrs on transect.

RESULTS

Survey Conditions

Since the 2023 survey condition and progress were described in Leclerc et al., 2024, emphasis was attributed to describe the 2024 survey condition on the remaining portion of MX-07. The Twin Otter departed on August 25th from Yellowknife to Cambridge Bay. The survey was completed between August 26th to September 1st, 2024, where Table 2 shows the survey progress and weather. The survey was performed in great weather. The visibility on transects was ideal, and no fog was encountered, except in the high hills along the west coast of Hadley Bay. On August 31st, the aircraft encountered some isolated showers, but the visibility was not impacted because of the minimal amount of precipitation (Table 2). Since there were weather delays, the northernmost four transects of Stratum 9 that were left unsurveyed in 2023 were flown (Leclerc et al., 2024). In sum, between 2023 and 2024, the entire management zone on Victoria Island was covered.

Since both surveys were completed in under 10 days, there is confidence that muskoxen did not move between strata during the survey periods nor between the two study areas, since muskox moved over a limited area in a year (Schmidt et al., 2016; Gunn and Fournier, 2000). Victoria Island terrain is open and relatively flat and dark brown muskoxen are easily detected, as they cannot hide under the tree canopy, or behind hills. Therefore, the availability bias was relatively low.

Table 2: Summary of the survey progress between August 25th to September 2nd, 2024, as well as the general weather encounter affecting visibility.

Date	Survey Progress	Weather
August 25th	Departure from Kugluktuk and Fuel Cache	Sunny, partially overcast
August 26th	Strata 4 and 1	Sunny
August 27th	Strata 3 and 2	Partially overcast, no fog
August 28th	Stratum 1- North	Sunny, fog in high coastal hills
August 29th	Stratum 2- North	Sunny, no fog
August 30th	Off	---
August 31st	Strata 1, and 2-North	Overcast, isolated showers, no fog
September 1st	Strata 9 and 2	Overcast, no fog
September 2nd	Departure from Cambridge Bay	Sunny, partially overcast

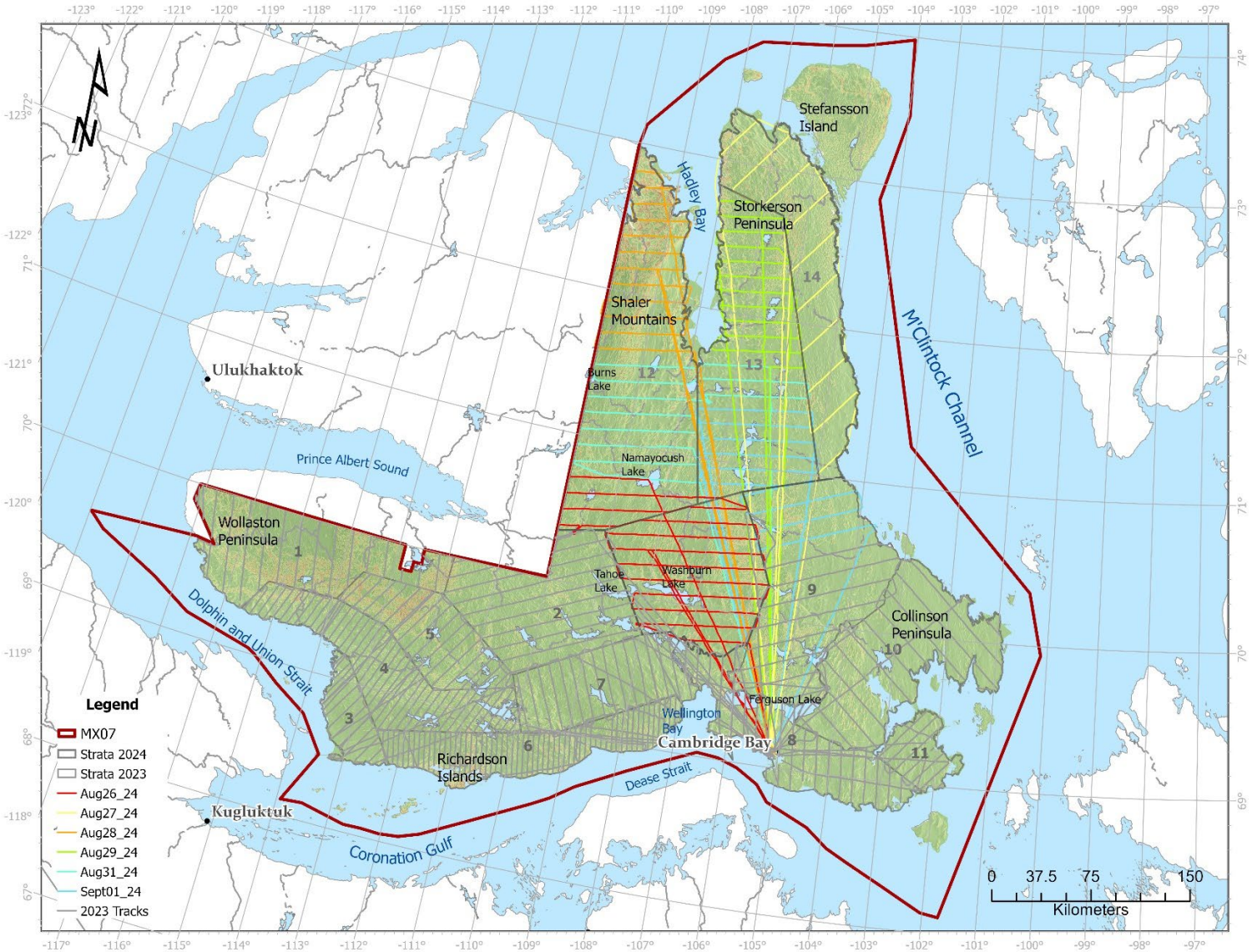


Figure 3: In the south section of MX-07, flight tracks (grey) done between October 23rd and November 1st, 2023. The 2024 flight tracks were represented by a different colour for each survey day between August 26th to September 1st, 2024.

Muskox Group Size

The largest group size observed was 18 muskoxen. Out of the 48 observations, 17 of them were single animals, most likely lone bulls since the surveys were performed during the rut (Figure 4). Without taking the single observations into account, the average group size was 6.13 (sd = 4.11) Groups with over 10 muskoxen were infrequent and accounted for less than 15% of the total number of the groups encountered. The calves to adults ratio was 5.3 calves: 100 adults including all number of calves and muskox seen.

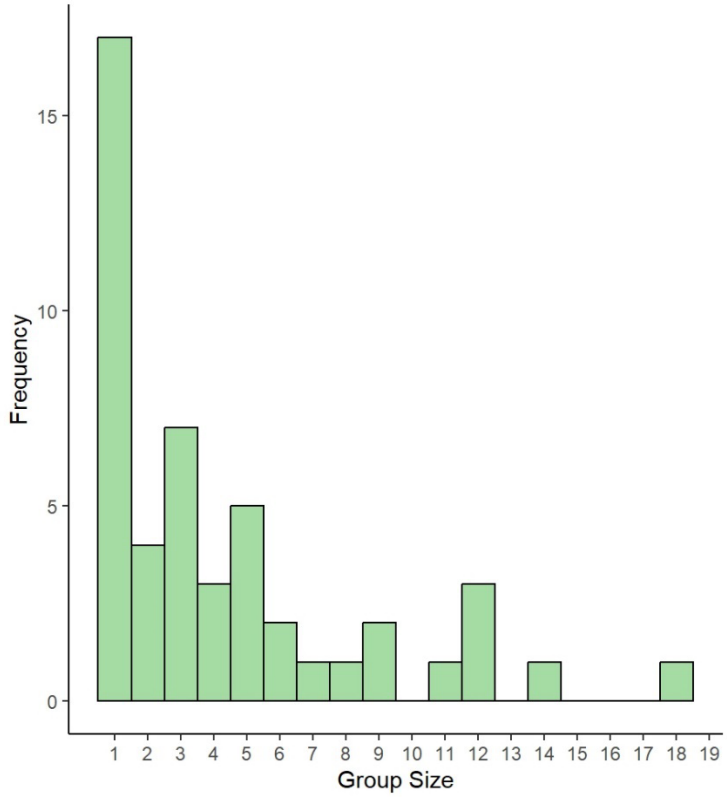


Figure 4: Frequency distribution of muskox group size (n = 48) observed during the 2023 and 2024 surveys combined, where the bins ranged from 1 to 19.

Muskox Herd Distribution, Dispersion

During the two surveys combined, 48 groups of muskoxen "on" and "off" transect were detected (Figure 5). No muskox was observed in Stratum 14 along the M'Clintock Channel and in Stratum 11 east of Cambridge Bay. Other than at these two locations, muskox groups were distributed continuously on the landscape. To shed light on the spatial pattern of the muskox groups, dispersion of the muskox on the landscape was further investigated by a spatial-point-pattern analysis. Such analysis revealed that muskoxen groups were clustered on the landscape and their pattern was not random (Moran's Index = 0.315, z-score = 2.885, p-value = 0.0039).

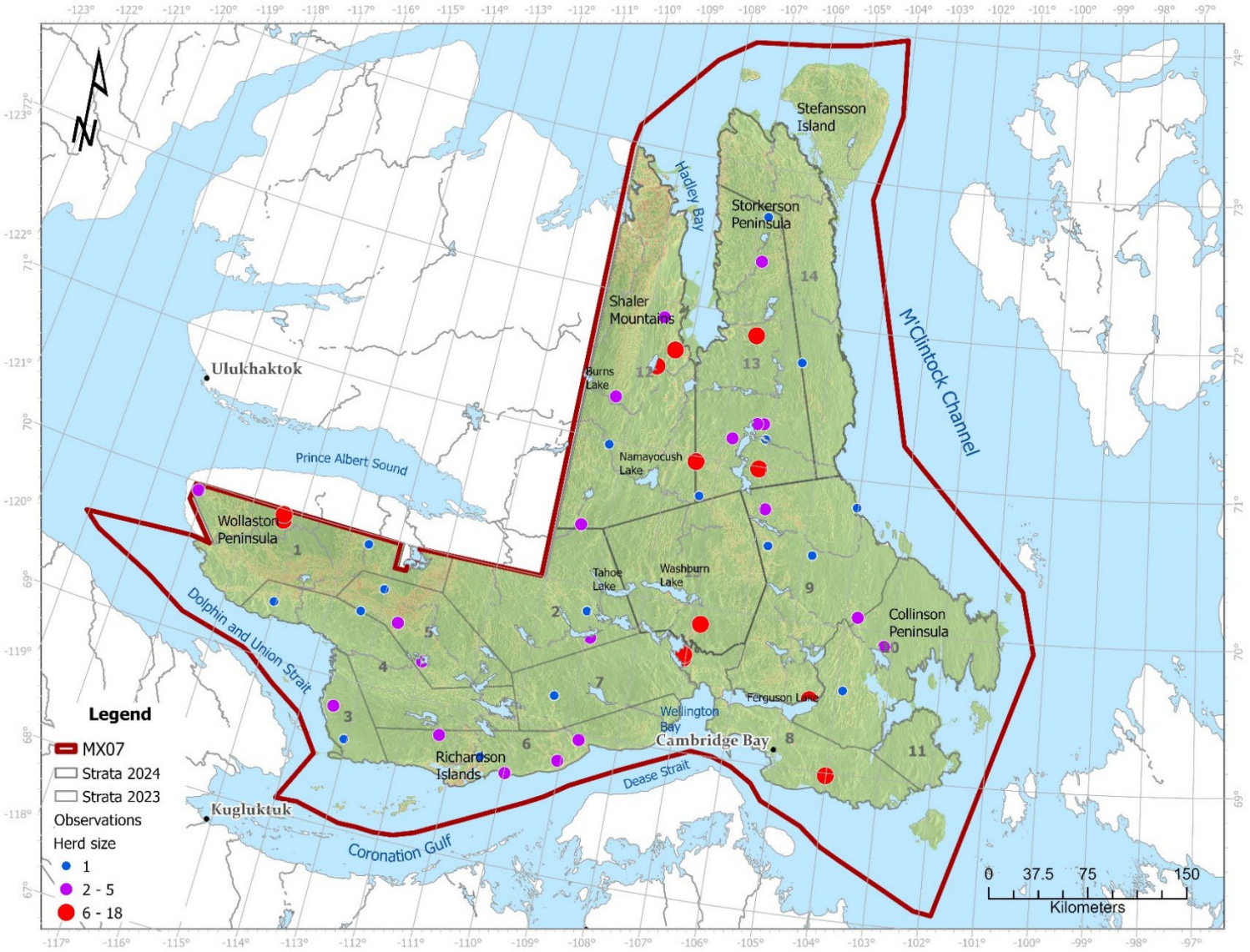


Figure 5: Geo-referenced locations of muskox herd seen "on" and "off" transects between October 23rd and November 1st, 2023 (Strata 2023) and between August 26th to September 1st, 2024 (Strata 2024).

Minimum Count of Muskoxen Observed

All the muskox observations made "on" and "off" transect were considered. Among the 15 Strata in MX-07, the number of muskox group varied from 0 to a maximum of 11 groups. The average number of groups per stratum was 3.69 (sd = 2.83). Over the entire muskox management zone MX-07, we identified a minimum count of 207 muskoxen; 124 in 2023 and 83 in 2024. Stratum 13 had the highest number of muskoxen observed, with 47 muskoxen. If we removed the observation "off" transect, 43 groups were "on" transects for a minimum count of 180 muskox in 60,384 km^2 of survey area for a total density of 0.003 muskox/km^2. The scarcity of muskox across the entire MX-07 management zone prohibited us from doing any further analysis, such as Distance Sampling, and from deriving an abundance estimate with adequate confidence, even for rough management decision (CV = 25.5%) (Pollock et al., 1990; Skalski et al., 2005).

Table 3: Summary of the minimum count of muskox in each surveyed stratum based as to whether the observations off the transect strip were included or not.

Year	Stratum	Area (km^2)	Number of Group	Number of Muskox (on and off transect)	Number of Muskox (on transect)
2023	1	9,328	4	25	25
	2	9,787	2	3	3
	3	2,375	2	5	1
	4	8,101	1	1	1
	5	5,622	4	9	9
	6	6,972	5	15	15
	7	6,581	1	1	1
	8	7,577	3	34	20
	9	13,297	6	25	23
	10	11,064	2	6	5
	11	2,151	0	0	0
2024	12	18,340	6	24	24
	13	10,415	11	47	41
	14	10,871	0	0	0
	15	13.950	1	12	12
		136,430	**48**	**207**	**180**

Although the observers were tasked to record predators, no grizzly bears or wolves were observed during the 2023 nor the 2024 survey. Nonetheless, in the southern part of the management zone (2023 Strata) 15 groups of polar bears were observed for a total of 30 individuals, while in the north (2024 Strata) three individual polar bears were detected.

DISCUSSION

Victoria Island offers a rich landscape diversity. Stefansson Island and the Storkerson Peninsula are known for their bare vegetation and a rock outcrop dominated flat landscape where semi-aquatic sedge and grasses have a tendency to concentrate around ponds, lakes, and in the drainage system. As herbivore, muskox rely strongly on the distribution of plant communities they feed upon. Muskox are not likely to adopt a random dispersion pattern on this heterogeneous landscape. Indeed, the dispersion pattern of muskox in the management zone was clustered. The highest number of muskoxen was found in Stratum 13 (n = 47), which may indicate a more optimal muskox habitat at this location, although some muskoxen were found distributed across the management zone (excepted Strata 11 and 14).

During the 2023 Dolphin and Union caribou survey, a considerable southern section of MX-07 was flown at very high coverage, 30% to 70%, and the number of muskox groups observed was minimal, only accounting for 30 groups. For comparison, over 210 groups were observed in the same area in August 2013-2014 (Leclerc, 2015). Local knowledge from Cambridge Bay corroborates this observation, as hunters have seen fewer groups and they have to travel longer distances between groups in their traditional hunting areas (EHTA pers.com).

Muskox group size seems to vary independently of the muskox population trend. During the declining trend, 2000 to now (Tomaselli et al., 2018), two surveys performed revealed that the number of muskoxen per group is small, with a mean of 5.7 (sd = 4.32) in 2013-2014 (Leclerc, 2015) and 6.13 (sd = 4.11) in 2023-2024 (Figure 4). A majority of small groups were still encountered at a time where the muskox number was found to be increasing. When the muskox number was increasing, from 1960 to 1990 (Tomaselli et al., 2018), the mean group size ranged from 9.9 (SE = 1.4) (Gunn and Lee, 1990) to 5.1 (SE = 0.2) (Gunn and Patterson, 2012) depending on the survey year and survey area. However, the range in groups size seems to contract in a declining phase. The largest group size was 18 in this study and 25 in 2013-2014 (Leclerc, 2015). Local knowledge affirmed that groups larger than 20 individuals were seen before the decline. In fact, group up to 37 muskoxen were reported during the increasing phase (Gunn and Patterson, 2012).

The muskox defense mechanism is characterized by the adult's behavior to form a tight circle facing outward, protecting the calves in the centre (Heard 1992). Forming large group can be a dissuasive and be used as an intimidating tactic to deter predators. When the group is small, muskoxen are more likely to form a line and individuals, especially calves, might become an easier target. Since the first sighting of grizzly bears at Rymer Point around 1978, their sightings on Victoria Island continue to increase as the years progressed (KAA pers. com.). Local knowledge suggests that muskox calves are an easy target for the grizzly that seems to kill, and leave most of the carcasses untouched (KAA and EHTA pers.com). Some hunters have followed grizzly bear tracks and found seven muskox carcasses (EHTA pers.com). However, no grizzly bears or wolves were observed during the survey of MX-07, while a total of 18 groups of polar bear were observed for a total 33 individuals. Their absence in the management zone seems reasonable based on the low number of prey.

Victoria Island had the largest endemic muskox number in the circumpolar Arctic at its peak in the 1990s (Cuyler et al., 2019). The results from this survey can be compared with the 2013-2014 survey results, as the same area was surveyed. In 2013-2014, 1,296 muskoxen were counted on transect, which resulted in an estimate of 10,026 ± 597 (SE) (CV = 6%) muskoxen and a density of 0.0743 muskox / km^2 (Leclerc, 2015a) In 2023-2024, a minimum zone of 180 muskoxen resulted from our effort, but the observations were too scarce to have confidence in an abundance estimate for the management zone. Results suggest that between these two MX-07 surveys, a continuous decline has occurred, with 86% less on transect observations then in 2013-2014, even with a higher survey effort in 2023-2024. In May 2023, a survey on the Northwest Territories portion of Victoria Island resulted in 380 adults and 25 calves observed on transect, with preliminary estimates showing that the muskox at this location also continued to decline (Davison et al., 2023). Due to the length of time between estimates, it is impossible to determine a more detailed picture of the trajectory of the trend within these 10 years. For example, to determine if the decline has stabilized over the last two to three years. Therefore, an increase in the monitoring frequency is warranted for this management zone, as well as the development of indices that can be used in between costly surveys to monitor trends more closely.

Muskox population cycles are not well understood. This species faced extinction in the early 1900s, because of an overexploitation of this resource by explorers during the fur trade (Barr, 1991). Since then, small pockets of muskox were recovering and recolonized their habitat. Such a pattern has been monitored on King Williams Island and the Boothia Peninsula, locations previously characterized and documented by the absence of muskox, where muskoxen are now thriving and increasing (Leclerc, 2015b; Leclerc, 2019). On the west side of the Canadian Arctic, on Banks and Victoria Islands, muskox reached their peak in abundance between 1990 and 2000, but underwent a rapid, drastic decline (Tomaselli et al., 2018; Leclerc, 2015; Davison and Baryluk, 2021). These examples might support that the muskox cycle is characterized by periods of boom and bust across its range.

During the bust, the concern goes beyond the decline in abundance, as the very few individual lefts are imperiled by a loss of genetic diversity making them less resilient to diseases and brings additional concern on their capacity to adapt to accelerated rapid warming in the Arctic (Lok et al., 2024). This is a concern because muskox genetic diversity is naturally very low and, in some cases, even lower than a species which have bene identified a risk. For example, muskox have lower diversity than the Tasmanian devil (*Sarcophilus harrisii*), a species that recently experienced a population bottleneck (Lok et al., 2024, Morris et al., 2012). The very low number of muskoxen left on Victoria Island increases the conservation concern for the fate of muskox as a species. A recent genetic study of the whole muskox genome showed that muskox underwent repeated population bottlenecks (Prewer et al., 2022). This suggests their past ability to survive "the climate crash" and glaciation events (Bartoli et al., 2005), as well as their ability to survive the hypothesis of the boom-bust muskox population dynamics.

The boom-bust dynamics are characterized by a period of almost exponential growth that overshoots the species carrying capacity, which leads to a subsequent population crash (Doebeli et al., 2021; Haller-Bull and Bode, 2021). Such eruptive dynamics are common for species that are free of limiting factors (predators) in the new colonized environment. Understanding the cause of muskox population fluctuations such as primary exogenous (environment stochastic with weak

density dependance) or endogenous (over-compensatory dependence) could help manage muskox population to minimize boom-bust dynamics, so the species can reach a stable equilibrium dynamic and to lessen the extinction risk (Boor et al., 2017).

Many contributing factors, acting in concert, might have played a role in causing this drastic decline of muskox on Victoria Island. Wildlife populations are regulated by different mechanisms; either density-dependent or independent. The risk of diseases can increase with the host population density. The community-based health surveillance program allowed us to monitor the diseases infection rate in muskox as the population grew. In 2008-2013, an increase in morbidity and mortality were documented, as hunters found muskox carcasses on the land. The post-mortem investigation of dead muskoxen resulted in the carcasses testing positive for *Erysipelothrix rhusiopathiae*. In addition, a fecal sample collection also confirmed the northward spread of the muskox lungworm, *Umingmakstrongylus pallikuukensis,* and finally*,* in more recent years, Orf and Brucellosis were also detected (Tomaselli et al., 2014; Dalton et al., 2024; Aguilar et al., 2024). These observations allow to detect change in population trend earlier, acting as surveillance warning system. However, these increases in outbreaks could be because of an alteration of the environment from anthropogenic climate changes (Lok et al., 2024).

The northern part of Victoria Island, an arctic desert, receives less than 250 mm of precipitation annually. Climate change is occurring at an unprecedented rate (Rantanen et al., 2022). Shift in the precipitation regime, such as an increase in the phase of liquid over solid precipitation and potential increase in evaporation rate would drive complex ecological changes in the arctic terrestrial ecosystem affecting plant diversity, abundance, and distribution. These major changes can impact and disrupt muskox habitat. Recently, hunters have seen an influx of muskox on the sea ice migrating to the mainland (KAA and EHTA pers.com), perhaps seeking a more optimal habitat. Although these changes in the muskox habitat remain highly speculative, additional effort should be allocated to investigate if change in the environment occurred during the decline.

The Victoria Island management zone, MX-07, has been managed for a harvested population since the muskox numbers were large and the probability of extinction in the short term was negligible. Now, with such a small number remaining, the management regime should shift from focusing on harvest potential to minimizing the risk of extirpation and promoting timely recovery. Such models should take into consideration the possible immigration/emigration factors from the neighboring population from surrounding islands, such as Banks Island, King William Island, or Prince of Wales Island, and including other sources of variation (i.e. genetic effect, environment stochasticity). Since the muskox on Victoria Island has been subjected to no or low harvest pressure for over the past 5 years, attention should be geared to the threats currently affecting reproduction rates by preserving calves. Co-management partners' discussion should focus on developing management strategies with long-term goals. To assure species persistence, the establishment of management strategies to promote the long-term maintenance of the species on the landscape and minimizing extinction threat is primordial. It is also important to continue gathering information on muskox population dynamics.

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ᑐᑭᓯᐊᖅᑎᑦᑎᐅᔨᑎ ᓇᐃᔭᖅᓯᒪᔨᖅ

ᐅᒥᖏᒪᑦ (*Ovibos moschatus*) ᖃᓄᖅ ᐅᓄᖅᑎᑭᓂᖏᓐᓂᒃ ᓇᓕᐅᑦᑕᕐᓂᖅ ᐃᖃᓗᒃᑐᒥᑦ, MX-07<3

2023 – 2024

ᐃᖃᓗᒃᑐᖅ (ᑲᓇᑕ) ᐊᒥᓱᒪᓕᓐᓂᒃ ᐅᒥᖏᒪᖃᖅᖃᑦᑕᓕᐅᖅᓴᖅ (Ovibos moschatus). ᐱᐅᔨᓐᓂᖅᐸᑦᓂᐊᓂᖏᑦ ᐅᒥᖏᒪᐃᑦ ᖃᓄᐃᙱᖏᑦᑎᐊᕐᓂᖏᑦ, ᐅᓄᕐᓇᓐᓂᐊᓂᖏᑦ ᐊᒃᓚᐃᑦ ᐊᒪᖃᐅᑦᓗ ᑕᑯᐅᓐᓇᒃᑐᑦ, ᓄᑦᐸᒃᑐᑦ ᐅᒥᖏᒪᐃᑦ ᖁᑭᖅᑕᒥᑦ ᓄᓇᐃᓐᓇᕐᒧᑦ ᐃᑲᔪᕐᓂᖅᑕᐃᓐᓇᕆᖅᓴᖅᑐᑦ ᖃᐅᔨᕗᐅᔨᓇᓄᑦ ᐅᓄᕐᓇᓐᓂᖅᐸᑦᓂᐊᓂᖏᓐᓄᓇᓄᑦ ᐅᒥᖏᒪᐃᑦ MX-07–Γ.

ᑕᓐᓇ ᓄᓇᑐᔪᖅ, MX-07, ᖃᐅᔨᔪᖅᑕᐅᓇᐅᖅᑐᖅ ᐅᑭᐅᑦ ᒪᖅᑲᓐᓄᑦ, 2013 ᐊᒻᒪᓗ 2014. ᑐᑭᒧᑦ ᐃᓕᓴᕐᕋᓗᑎᑦ, 1,296 ᐅᒥᖏᒪᓐᓂᑦ ᓱᓕᐃᓇᐅᖅᑐᑦ, ᓇᓇᐅᑦᑕᖅᓴᓗᑎᑦ 10,026 ± 597 (SE) (CV = 6%) ᐅᒥᖏᒪᖃᖅᓂᖏᓂᑦ. ᑕᐃᒪᖏᓂᑦ ᖃᐅᔨᔪᓇᐅᖅᒪᑕ 2013-2014–Γᑦ, ᐅᒥᖏᒪᑦ ᐅᓄᕐᓂᖅᐸᑦᓂᐊᓕᕐᓂᓇᖅᑯᔨᐅᒃᑦ.

ᖃᐅᔨᔪᖅᓂᐅᔪᒥᑦ ᑐᕋᓚᓴᐅᓪᓗᐊᑕᖅᑐᖅ ᓄᑕᒥᒃ 2023-2024 ᖃᓄᖅ ᐅᓄᖅᑎᑭᓂᖏᓐᓂᒃ ᓇᓇᐅᑦᑕᕈᒪᓪᓗᓂᒃ ᐅᒥᖏᒪᖏᓂᒃ ᐃᖃᓗᒃᑐᖅᑎᑦᑎᑦ, MX-07. ᑕᐃᒪᐃᓚᒪᑦ, ᒪᑯᓂᖏᒐ ᑐᕋᓚᖃᓪᓗᐊᑕᓇᐅᖅᑐᑦᑦ: 1) ᓄᑕᒥᒃ 2023-2024 ᖃᓄᖅ ᐅᓄᖅᑎᑭᓂᖏᓐᓂᒃ ᓇᓇᐅᑦᑕᕈᒪᓪᓗᓂᒃ MX-07-Γᑦ ᓇᐅᑦᑎᖅᓯᓪᓗᒍᓗ ᖃᓄᐃᙱᖏᑦᑎᐊᕐᓂᖏᓂᑦ ᐊᒻᒪᓗ 2) ᖁᒥᖅᑭᓪᓗᑦ ᐃᓐᓇ ᐊᐅᓪᓚᑎᑦᓯᔨᐅᔨᑦ ᓄᑕᖅᓴᓂᒃᓗ ᐊᑐᓕᖅᓴᓪᓗᒍᑦ ᐱᓵᓱᐊᖃᓄᑦ.

ᖃᐅᔨᔪᓇᐅᖅᑐᑦ ᒪᖅᑭᐃᓕᓴᐅᓕᖅᑯᑦ, ᓂᒋᐊᓂ ᐅᑐᐱᓗ 23 ᐊᒻᒪᓗ ᓄᕕᐱᓗ 1 ᐊᑯᓂᖏᓂᑦ, 2023 ᐊᒻᒪᓗ ᐅᐊᖏᓇᖏᓂᑦ ᐊᑲᑭᓯ 2 ᐊᒻᒪᓗ ᔨᑎᐱᓗ 1 ᐊᑯᓂᖏᓂᑦ, 2024.

ᐊᖏᕈᑲᔨᖅᑕᐅᔨᒪᔭᑯᑦ ᔨᓪᓚᑐᒍᓪᓗ, 48–ᓂᑲ ᑲᑎᖏᓐᓇᐅᖅᑐᐃᓂᑲ ᑕᑯᓪᓚᐅᖅᑐᑦ 17–ᖏᓪᓗᓂᒃ ᐸᖏᓂᑦ ᐅᒥᖏᒪᑐᐊᖏᓪᓗᓂᒃ. ᐅᓄᖅᑎᑭᓵᓴᓂᒃ 6.13 (sd = 4.11) ᐃᓚᐅᑎᓂᓪᓚᑦ ᐸᖏᓂᑦ. ᐊᒥᓱᓂᖅᓴᑦ ᐅᒥᖏᒪᑦ ᑕᑯᐅᓚᐅᖅᑐᑦ ᐃᖏᕐᕋᕐᕕᐅᔨᖅ 13–ᑯᑦ 11 ᑲᑎᖏᓐᓇᐅᖅᓴᓂᒃ ᐅᓄᖏᓪᓕᖅᓴᓂᒃᓗ 47 ᐅᒥᖏᒪᖏᓂᒃ.

2023-2024 ᖃᐅᔨᔪᓂᒥᑦ, ᑲᑎᖏᓐᓇᐅᖅᑐᑦ ᐅᒥᖏᒪᑦ ᖃᓴᓱᐅᓂᖏᓂᑦ ᐅᓄᖏᓗᐊᓕᐅᖅᑐᑦ ᖃᓄᖅ ᐅᓄᖅᑎᑭᓂᖏᓐᓂᒃ ᓇᓇᐅᑦᑕᑦᑎᐊᕈᒪᓪᓗᓂᒃ (n = 43). ᑕᐃᒪᐃᓚᒪᑦ, ᑲᑎᑕᐃᓵᓇᓪᓗᓂᑦ ᐊᐅᓪᓚᑎᑦᓴᑦ ᓱᓕᖅᑕᖏᓂᑦ, ᐅᓄᖏᓪᓕᓂᒃ ᓱᓕᐃᓇᐅᖅᑐᑦ 180 ᐅᒥᖏᒪᖏᓂᒃ.

ᑕᓐᓇ ᖃᓴᓵᓇᐅᓂᖏᓂᑦ ᐅᒥᖏᒪᑦ MX-07–Γᑦ ᐃᓱᒪᓗᓇᖅᑐᖅ. ᐅᒥᖏᒪᑦ ᐊᓵᓇᐅᒪᔭᑯᓪᓗᓂᑦ, ᐱᓕᒪᓇᐅᔪᖅ ᐊᐅᓪᓚᑎᓂᔭᓕᑦ ᖃᓄᖅᑖᑭᑎᓪᓗᐅᔭᓂᖅ ᐊᑯᓂᐅᔨᖅ ᐅᒥᖏᒪᖃᖅᑯᑦᓗᓂᑦ ᓄᓇᖏᓐ ᒥᓪᓂᐅᐊᖅᓗᓗᓗ

ᓄᖅᑐᐃᓵᓇᕆᐊᖅᓯᓂᖅᓴᖅ ᐅᑎᖅᑎᓵᓇᓕᐊᖅᑎᓪᓗᓕᖅᓴ. ᖃᐅᐅᔨᕿᕕᐅᔪᓴ ᐊᑯᓵᓂᖅᓂᓵᓂᖅ, ᑐᕿᓕᒪᑦᔪᕐᑎᓂᑦ ᓄᐊᖅᑐᓵᓇᕆᐊᖅᓯᐊᖅᓯᐳᒍᖅ ᐅᓄᖅᑎᑎᓂᓂᖅᓴᖅ ᖃᓄᐃᐸᓴᓕᓪᓚᐃᓂᖅᓴᖅ, ᖁᑎᕿᑐᓕᓪᓕᓗ ᑐᑭᔪᐅᒪᓂᕐᒃ ᐊᑐᑭᔪᓂᑎᓵᓂ ᐊᐅᓪᓚᕆᑎᑎᐊᖅᐸᓪᓕᖅᒪᓕᓗᓕᒃ ᔪᕕᓂᕆᓂᓵᓂ.

Kugluktuk Angoniatit Association ● Hunters' & Trappers' Organization
PO Box 309, Kugluktuk NU X0B 0E0 ● Phone: (867) 982-4908 ● Email: kugluktuk@krwb.ca

PITJUTAOYOK: Kugluktuk Angoniatit Katimayiit (HTO-kut) titigakhimayot kiotjutigivlugit ukkua Nunavut Kavamatkut kiotjutigihimayaitnik tahapkununga umingmait munagitjutaitnik mighagot tahamani nunaoyami MX-07-mi.

HIVUNIGHITJUTAT:

- Umingmait kangagalok nikkigiyaokataktot hapkua Kugluktukmiut Inuinaknin.
- Kugluktumiut anguniaktit atokataktait tahapkua pingahut aviktoghimaniit nunaoyami anguniakvigivlugit tahapkua: MX-11 kivatani Kugluktumi; MX-07 Kiilikinikmi MX-09, tatvalu una MX-11 tahapkua atogaoloaktot anguniaktiit anguniakvigivlutjuk.
- Anguniakviohimangitok tahamna aviktoghimaninga nunaoyamin MX-07-mi Kugluktukmiot anguniaktitnin tatvanganin 2019-min kinguaniitlo, ikittunuanik umingmakviohimayok
- Kugluktukmiut anguniaktit takkuhimaliktot ikkilivalianinganik tahapkua amigaitilangit umingmait tahamani (anniakutikakpaliktuniklo) taimatot ahiani aviktoghimayuni nunauyami angutiakpaliktot.
- Kugluktumi HTO-kut ikkayokhiotihimangitat ukkua Nunavut Kavamatkut parnaiyaotatnik naunaiyagomavlutjuk tahamna aviktokhimayomi nunaoyami MX-07.

UBLUMI KANOGININGANIK:

- Ukkua Kugluktumi HTO-kut katimakatigihimayat ukkua Nunavut Kavamatkut havaktiit tatvani June 18, 2025-mi ukkaotigivlugit tahapkua naunaiyaotaohimayot.
- Ukkaotikakhimayot tahapkuninga Kititaohimayot Anguniakutikhat Ataatait ikkilivaligomangmatjuk 400-mit talimanut ataatakhanik tamatkiklugo tahamna aviktoghimayomi nunaoyami.
- Ukkua Kugluktumi HTO-kut ikkayoghiotingitaat tahamna tatvalu uktokuyaat ukkua Nunavut Kavamatkut katimakatigitkovlugit nunaliitni nallaktitlugit.
- Ukkua Kugluktumi HTO-kut atokataktat una MX-11-mi nunaoyami hapkua nunaliitni anguniagangata Nunakatigit Nikkighakhiokhugit Avakuktakhanik.

UKTOKUYAOYUT:

- Ukkua Kugluktuk-mi HTO-kut pihimayomayat ukkua Kititaohimayot Anguniakutikhat Ataataghat aulaakuvlugit 400-nik ataataghat tatvalu tamaita hapkua anguniaktit katimayiiit ilikut angutighanik munagitjutighaitnik ihuaghaitkoyait tahapkuani aviktoghimayomi nunaoyami, taimatotaok inmingnik havagihimayaitnik.
- Ukkua Kugluktumi HTO-kut katimakatigiyomayait hapkua allaat Anguniaktit Katimayiit tatvalu Ikkayoktigikhutik Munaghiyioyot kiotjutighanik hivunighami tahapkua Nunavumi Anguhighalikiyiit Munaghiyiolutik Katimayiit katimakatigiyomavlugit ahivakulugit tahapkua Kititaohimayot Anguniakutikhat Ataatakhat umingmakhiokutit tatvani Aviktoghimayomi Nunaoyami MX-11-mi.

Titigakhimaya uma

Amanda Dumond, Ataniata

ᐅᐻᓄᖅ ᑐᓂᓴᐅᔪᖅ

ᓄᓇᕗᑦ ᐅᒪᔪᓕᓇᔨᕆᐊᖃᕐᓂᖅ ᑲᑎᒪᔨᖏᑦᓯᓄᑦ

ᐅᔨᒪᖏᒃ

ᑐᑭᓯᐊᕐᕕᑎᓂᖅᓴᖅ: X **ᐃᓱᒪᓕᐅᕕᑐᑎᑦ:**

ᐱᓵᔪᑦ: ᑐᓴᖅᑎᑦᑎᓂᖅ ᐊᐅᓪᓚᑦᑎᖅᑎᑦᑐᓂᑦ ᓄᓕᒥᑦ ᐅᒪᔪᑦ ᐊᒥᓱᓂᖏᓐᓂᖅ ᓇᓚᐅᑦᑕᖅᑕᐅᔪᓂᑦ ᖃᐅᖅᑕᓗᒃ ᑐᑭᓯᐊᕐᓗᒃ ᑐᑐᖏᓐᓂᑦ.

ᐱᓕᕆᐊᖏᑐᒪᔪᑦ

- ᖃᐅᖅᑕᓗᒃ ᑐᑐᖏᑦ ᐊᑦᔨᐅᖓᖏᑦᑐᐅᓐᓂᑦ ᐊᓪᒪ ᐅᓄᖓᑦᓴᓗᑎᒃ ᑐᑐᑐᔪᑦ (ᑕᐃᓵᓱᒪᓂ 20,000 40,000-ᖃᐅᓚᐅᖅᓱᒪᔪᑦ ᓂᕿᔪᑎᑦ) ᐊᖏᖕᓇᓱᑕᐅᔭᕐᓴᑎᒃ ᐊᖏᖕᓇᓱᑎᓂᑦ ᖃᑎᖅᑕᐅᓂ ᖃᓄᐃᓱᐅᑦᓯᓂᑦ ᓄᓇᓕᖓᓂ ᐊᒪᒪ ᓄᓇᓕᖓᓂ ᑲᓇᖕᓇᖅᑐᓱᓕᒥ ᓄᓇᑦᓱᐊᒥ.
- 2018–ᒥ ᖃᐅᖅᑕᓗᒃ ᑐᑐᖏᑦ ᓇᓚᐅᑦᑕᖅᑕᐅᓚᐅᖅᑐᑦ 4,105-ᖅᓯᓱᑦᓴᐅᑦᓯᑎᑎᒃ (95% CI = 2,931 - 5,759). ᑕᑯᐊᐊ ᖃᐅᔨᓴᕿᑕᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᒪᔭᑦ ᐅᓄᕿᓂᖅᓴᕐᓂᕐᓂᖏᓐᓂᑎᒃ 88%–ᒥ 1997–ᒥ ᓇᓚᐅᑦᑕᖅᑕᐅᓚᐅᖅᑐᑦ 34,558–ᖅᓯᓗᑎᒃ (95% CI = 30,275 – 38,841).
- ᐊᖅᑭᒋᐊᖅᑕᐅᓚᐅᖅᑎᑦᓗᒍ ᖃᐅᔨᓴᕿᑕᐅᓂᐊᖅᑐᖅ ᐱᖅᓯᐅᑎᑦᓗᓕᑦ ᑎᒍᒥᐊᖅᑎᐅᖅᑕᐅᔪᑦ ᐊᑐᕐᔪᒃᓯᖏᓐᓂᑦ, ᑎᑎᖅᑲᖅᓯᒪᔪᑦ, ᖃᓄᖕᓂᕿᓂᖅᓱᐃᓕᐅᓚᐅᖅᑐᑦ, ᐊᒥᓱ ᓄᓇᕗᒻᒥ ᖃᐅᔨᓴᓴᐅᔪᑦ ᖃᐅᔨᓴᖅᓯᒪᔪᑦᖃᑎᒌᓂᒍ, ᖃᐅᔨᓴᐅᔪᑦ 2020–ᒥ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑐᑦ ᓇᓚᐅᑦᑕᖅᑕᐅᔪᑦ 3,815–ᖅᓯᓗᑎᒃ (95% CI = 2,930 – 4,966, CV= 13%) ᑐᑦᑐᐃᑦ, ᒪᓕᓚᐅᖅᓴᑦ ᐅᓄᕿᓂᖅᓴᕐᓂᕐᓂᐊᕐᓂᑦ ᖃᐅᔨᓴᐅᓚᐅᖅᑐᑐᒃ 2018–ᒥ ᖃᐅᔨᓴᖅᑕᐅᔪᓂᑦ.
- ᑎᓯᐱᓗ 2020–ᒥ, ᓄᓇᕗᑦ ᐅᒪᔪᓕᓇᔨᕆᐊᖃᕐᓂᖅ ᑲᑎᒪᔨᖏᑦᓯᓄᑦ ᖃᑎᖅᑭᒋᓚᐅᖅᓴᑦ ᓯᕗᓕᓂᖅᓴᕐᓂᖏᓐᓂᑎᒃᑎᒃ ᐊᖅᑭᒃᓯᒪᓕᖅᑎᑕᐅᔪᓂᑦ ᐊᑐᓕᖅᑎᑦᓯᐊᓕᓴᓂᖅ ᑲᑎᒪᔨᖏᑦᓯᓄᑦ ᐊᖏᖕᓇᓱᑕᐅᔪᓂᑦ 42–ᓂᒃ ᑐᑦᑐᓂᒃ ᖃᐅᖅᑕᓗᒃ ᑐᑦᑐᖏᓐᓂᒃ. ᑲᑎᒪᔨᑦ ᐃᓱᒪᓕᐅᓚᐅᖅᑐᑦ ᐅᓄᖅᓯᒪᔪᓂᑦ ᐊᖏᖕᓇᓱᑕᐅᔪᓂᑦ 105–ᖅᓯᓕᓴᓗᑎᒃ ᑐᑦᑐᑦ ᐊᒪᒪ ᓄᓇᕗᑦ ᓴᖇᒪᓕᒃᓂᑦ ᑕᑯᓂᓂᖅᓴᖅ ᒪᓕᓚᖅᓴᖅᑎᑦᑎᓕᐅᖅᑐᑦ.
- ᐃᑲᔪᖅᑐᕈᒪᓕᓴᓂᑦ ᐅᑎᖅᑎᑦᑎᐊᓕᓕᓕᐊᓂᖅ ᑐᑐᑐᓂᒃ, ᓄᓇᕗᑦ ᓴᖇᒪᖓ ᐊᑐᓕᖅᑎᑦᑎᓂᓗᒪᔭᖅ ᓂᕈᑐᖅᑭᓴᑐᑐᓂᑦ ᐊᐅᓪᓚᑦᑎᑦᑎᓂᐅᔪᓂᖅᑐᑦ ᐃᓕᓴᐅᑦᓂᓕᖅᑐᑦ ᑲᑎᒪᔨᖏᑦᓯᓄᑦ ᐊᐅᓚᑦᑎᑦᑎᓂᐅᔪᓂᖅᑐᑦ ᑲᒪᓂᖅ ᐊᒪᓕᖅᑐᖅᑐᑦ ᑭᓇᐅᔭᖅ ᒪᓂᓕᑎᑎᓕᐅᖅᑐᑦ. ᐊᐅᑐᐊᓂᖅᑭᓓᓂ ᐅᓄᖅᓯᒪᔪᓂᑦ ᖃᐅᖅᑕᓗᒃ ᑐᑐᖏᓐᓂᑦ, ᐊᒪᓕᖅᑐᑦ ᑲᑎᒪᔨᖏᑦᓯᓄᑦ ᐊᖏᖕᓇᓱᑕᐅᔪᓂᑦ $900–ᖅᓄᖅ ᐊᑕᐅᓱᖅ ᖃᐅᔨᓴᓕᕆᓴᖅ ᐊᖅᑭᒃᓯᒪᓕᖅ (ᐊᓱᖏᓐᓂᓂᓪᓕ $500–ᖅᓯᓗᓂ).
- ᖃᐅᖅᑕᓗᒃ ᑐᑐᖏᓐᓂᒃ ᖃᓄᖕᓂᓱᑐᖅᓱᐅᑕᓚᐅᖅᑐᑦ ᐊᐃᑎᓪ 2021–ᒥ (36 ᐊᖏᓐᓇᓗᐃᑦ) ᐊᒪᒪ ᐃᑦᓱᓗ 2024–ᒥᑦ (35 ᐊᖏᓐᓇᓗᐃᑦ). ᑎᒍᒥᐊᖅᑎᐅᖅᑕᐅᔪᑦ ᐃᓕᐅᓕᐅᖅᑐᑦ, ᐊᒪᒪ ᐊᖏᖕᓇᓱᑐᑐᓂᑦ, ᐊᒪᒪ ᐊᖏᖕᓇᓱᑐᑐᓂᖅᓴᖅ ᑭᓴᖅᑐᐃᐃᓕᓴᖅᓯᒪᓂ ᐃᓕᐅᓕᐅᖅᑐᑦ ᑕᒪᐃᓐᓇᖅᑐᑦ ᐱᓕᕆᔨᒃᑯᑦ ᑕᒪᓐᓇᖅᑐᑦ ᐊᐅᓚᑦᑎᑦᑎᓂᐅᔪᓂᖅᑐᑦ ᐊᑐᖅᑕᐅᓂ.

ᐃᓐᓇ ᖁᓄᐃᓕᓐᓄᓐᓂ

- ᐃᓐᓇ ᖃᐅᖅᑕᓗᒃ ᑐᑐᖏᑦᑕ ᐊᒥᓱᓂᖏᑦᑐ 8,000 ᑐᖓᔪᓂᑦᑐᑦ, 20%–ᖏᑦᓂᓕᓴᓗᑎᒃ ᑕᐃᓵᓱᒪᓕᓕᓴᐃᑦ. ᑕᐃᒪᐃᓱᒪᑦ, ᐃᓐᓇ ᐊᐅᓪᓚᑦᑎᖅᑎᑦᑎᓂᐅᔪᓂᖅᑐᑦ ᑲᑎᒪᔨᖏᑦᓯᓄᑦ ᐊᖏᖕᓇᓱᑕᐅᔪᓂᑦ ᐱᓵᔪᑦᑎᓇᓕᖅᑎᓂᖅᑐᑦ ᑐᑐᖏᓐᓂᑦ ᐅᑎᖅᑎᑦᑎᓂᐅᔪᓂᑦ.
- ᓄᑦᖃᖅ ᑎᓪᓗᓯᓂᑦ ᐅᓄᖅᓯᒪᔪᓂᖅ ᖃᐅᔨᓴᖅᑐᑦ ᐱᐊᓂᓚᐅᖅᑐᑦ 23–ᒥᑦ ᓄᕐᐱᐊᓗ 2–ᔪᑦ, 2023. ᖃᐅᔨᓴᖅᓂᕆᔪᑦ ᐸᖅᓴᐅᓄᖅ ᐱᖅᓯᐅᑎᑦᓗᒍ ᐅᖃᐅᓯᐅᓚᐅᖅᑐᑦ ᑎᒍᒥᐊᖅᑎᐅᖅᑕᐅᔪᓂᑦ ᐊᑐᖅᓴᓗᑎᒃᓗ ᐊᑦᔨᔪᓗᖓᓂ

ᐊᖁᑭᒃᓯᒪᔪᒥᑦ 2020–ᒥᑦ ᐊᑐᖅᑕᐅᓕᐅᖅᑐᒥᑦ. ᐊᖏᓚᓴᐃᑐᓗᓵᓴᐅᓐᓂᖅ–ᓂᕈᐊᖅᑕᐅᓯᒪᔪᑦ ᓄᓇᓕᖅᒥᐅᑦ ᐃᓕᐅᓕᐅᖅᓴᑦ ᖃᐅᔨᖃᑕᐅᓕᓪᓗᑎᒃ.

- ᖃᐅᔨᓴᕈᑎᐃᓂᑦ 2023–ᒥᑦ ᓇᐃᓴᐃᑎᓕᓪᓗᒥᑦ ᓇᓕᐅᑦᑕᓕᐅᖅᑐᑦ 5,229 (95% CI = 3,985 – 6,473, CV= 12%) ᑭᓕᓂᐅᓴ ᖁᑭᖅᑕᖏᓂ ᓄᓇᐃᓐᓇᒥᓕᓪᓗ. ᑕᐃᓇ ᓄᑕᖅ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᖅ ᓯᓕ 8,000 ᑐᐢᑐᖃᑎᒌᓄᑦ ᑐᖏᓕᓂᔨᓴᖅ, ᑭᓯᐊᓂ, ᓇᓗᓇᐃᖅᓯᖃᓕᐅᖅᓯᒪᓕᖅ ᐅᓄᔪᓂᓚᖏᑦ ᓯᓕ ᐅᓄᑭᓐᓂᖅᓴᓕᓕᐊᓂᖏᓐᓂᓂᖅ.

- ᓄᓇᒃᑕ ᓴᙵᒪᔪᑦ ᐱᓕᓚᖃᑎᖃᖅᓯᒪᒃᑕ ᐱᓕᓚᖃᑎᖃᕆᐊᖏᓐᓇᖅᓱᓗᓐᓗ ᐊᔪᑐᖅᑕᐅᓯᒪᔪᓂᖅ ᓄᓇᓕᖅᓂᓕ, ᐊᐅᓕᑦᑎᐅᐅᖃᖅᑎᑕᐅᑦ ᐊᖏᒪ ᓄᓇᑦᓯᐊᐸᓴᑦ ᐱᓕᓚᖃᐊᖏᓐᓂᔪᓂᖅ ᐊᖏᒪ ᖃᐅᔨᓴᖃᑦᓯᓕᕆᔨᖅ ᖁᑭᖅᑕᑦ ᑐᔪᔪᖅᑕᐅᓯᒪᔪᓂᖅ.

ᑐᓴᕆᓂᖅ:

- ᑲᑎᖏᓐᓄᑦ ᑐᓴᕆᓯᓕᐅᖅᑐᑦ ᐊᖏᓚᓴᐃᑐᓗᓵᓴᐅᓐᓂᖅ ᐃᖃᓗᓴᑐᑎᐊᒥ (ᐃᓕᐊᓗ 14, 2025), ᖃᓄᐃᓗᖅᑐᒥ (ᐃᓕᐊᓗ 17, 2025), ᐊᖏᒪ ᖁᑎᖏᐅᑦ ᐅᒪᓴᓯᓗᓵᓴᐅᓐᓂᑦ ᑲᑎᒪᓴᖏᑦ (ᐃᓕᐊᓗ 28, 2025). ᐅᖃᖃᑎᖃᕆᓯᓕᐅᖅᑐᑦ ᐳᓵᕿᐃᑦ ᐊᖏᒪ ᐅᒥᖏᓚᑲᑐᖅ ᐊᖏᓚᓴᐃᑐᓗᓵᓴᐅᓐᓂᓂᖅ, ᑭᓯᐊᓂ ᑲᔪᓯᓵᓕᓕᐅᖅᓯᒪᔪᓂᑐᖅ. ᓇᐃᓇᖅᑭᓯᒪᔪᓂᖅ ᑐᓴᕆᓵᑎᑦᑎᓕᐅᖅᑐᑦ ᐃᓕᓴᕐᓇᖅᑐᓂᖅ ᐊᑎᖅᑕᐅᓕᐅᖅᓯᒪᓕᖅᑐᑦ ᐳᓵᕿᐃᑦ ᐊᖏᒪ ᐅᒥᖏᓚᑲᑐᖅ ᐊᖏᓚᓴᐃᑐᓗᓵᓴᐅᓐᓂᓕᓂᖅ ᑎᑎᕋᖅᑭᒪᔪᓂᖅ ᑐᓂᓯᔪᓚᓪᓗᑎᒃ. ᑐᑭᖅᑎᑕᐅᓕᐅᖅᓯᓂᑐᒍᑦ ᓯᓕ ᐃᓕᓴᕐᓇᖅᑐᓂᖅᑕᐅᑦᑎᓕᐅᖅᑎᑕᐅᓕᓪᓗᑎᑦ ᐃᓕᑦ 28, 2025–ᒥ ᐱᑎᐊᖅᓴᓂ. ᐃᓕᓇ ᐊᖅᑭᑦᓯᐊᓇᓚᐅᒍᑦ ᑐᐃᓯᕆᓂᑎᓂᐊᓂᖅᑐᓂᓂ ᑭᓯᐊᓂ ᑐᓴᓯᕐᓇᖅᑐᓂ ᐊᖏᓚᓴᐃᑐᓗᓵᓴᐅᓐᓂᓂᖅ, ᑲᑎᒪᓕᖅᑐᑦ ᑐᓂᓯᓕᐅᖅᑎᓪᓗᒥᑦ ᐃᓯᓕᖅᓴᕐᓂᓚᖅᓱᓂᑦ.

ᐊᑐᓕᕐᓂᐅᔪᖅ:

- ᑐᖓᓚᐱᓕᓪᓚᔪ ᐃᓕᓇ ᓇᓕᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ, ᑕᒪᕐᒥᓂᑎᐊᖅ ᐊᑐᐃᓐᓇᐅᔪᑦ ᖃᐅᔨᓴᖅᑐᑕᐅᓗᓵᓴᐅᓐᓂᓂ ᑐᑭᓯᐊᖅᑐᑦᑎᐊᓂᖅ, ᖃᐅᔨᓪᓚᔪᐅᑐᖅᑐᖃᖅᑐᑦᑕᖅᐃᓕᓴᐅᓕ/ᐃᓄᐃᑦ ᖃᐅᔨᓯᓚᕐᔪᐊᖅᑐᑦᑕᖅᑐᑦ, ᓄᓇᓕᖅᑐᑦ ᓴᙵᒪᔪᓂᖅ ᐊᓯᔾᔨᐯᒪᖓᓂᑦᑐᑦ ᐃᓕᓇ ᐊᓯᔾᔨᖏᓚᖓᓂᑦᑐᑦ ᑲᑎᓕᓂᑦ ᐊᖏᓚᓴᐃᑐᓗᓵᓴᐅᓐᓂᑦ 105–ᓂᖅ.

ᐅᑐᓗᑐᔪ ᓄᓇᕗᑦ ᐱᖃᓐᓇᕐᓂᐊᕐᓂᖅ
Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

ᐊᐅᓚᓕᕆᔨᒃᑯᑦ
Department of Environment
Avatiliqiyikkut
Ministère de l'Environnement

Abundance estimates for the Dolphin and Union caribou herd (*Rangifer tarandus groenlandicus x pearyi*)

Fall 2023

Report

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The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment. Reference to any specific type of gear and/or equipment does not mean that it is endorsed by the authors or the institutions they represent.

TABLE OF CONTENTS

LIST OF FIGURES 5
LIST OF TABLES 6
ABSTRACT 7
INTRODUCTION 8
OBJECTIVES 9
MATERIALS AND METHODS 9
Dolphin and Union Caribou Herd Seasonal Range 9
Integration of IQ in the Survey Design 11
Dolphin and Union Daily Movement Rates 12
Design of Distance Sampling Surveys 12
Sampling Design 12
Field Setting 14
Data Analysis 15
Mark-Recapture Distance Sampling (MRDS): Detection Function 16
Density Surface Model (DSM) 16
Population Changes and Overall Trend 18
RESULTS 19
Dolphin and Union Daily Movement Rates 19
Survey Conditions 20
Data Exploration 21
Distribution 21
Group Size 23
Distance Sampling Analysis 23
Choice of Covariates 23
Mark-Recapture Distance Sampling (MRDS): Model Selection 26
Estimates 28
Density Surface Model 29
Model Selection 30
Abundance estimation and variance estimation 30
Population Changes and Overall Trend 32

DISCUSSION 34
ACKNOWLEDGEMENT 37
REFERENCES 38

LIST OF FIGURES

Figure 1: Dolphin and Union caribou herd utilization distribution (%) during the rut, October 13th to November 7th (Nagy et al., 2011). 10
Figure 2: Left: 2020 Dolphin and Union strata placement and effort (dark grey= low effort, blue= medium effort, yellow = high effort, and pink= very high effort). Strata were designated based on historic survey observations, cumulative caribou telemetry data, IQ, and budgetary constraint. Right: Based on IQ, the revised 2023 strata boundaries. Note that the light grey is the Northwest Territories and the medium grey is Nunavut. 12
Figure 3: Final 2023 Dolphin and Union caribou herd survey design, with A to O strata and the transect lines..... 14
Figure 4: Work flow of the Density Surface Model (DSM) adapted from Miller et al., (2013). . 18
Figure 5: Left: Collar locations (n = 21) during the rut, which as has been established with a start date of October 13th (dark blue) and a final date of November 7th (yellow) for 2023. Right: Dolphin and Union collar daily movement rate from October 13th to November 7th, where the shaded grey highlights the period of the abundance survey. 19
Figure 6: Flight tracks (blue) between October 23rd and November 1st, 2023. A section of G stratum was re-flown (light pink) on November 2nd, because of poor visibility during the first attempt..... 21
Figure 7: Overview of the 148 Dolphin and Union caribou groups' locations observed on transects during the survey..... 22
Figure 8: Histogram displaying the Dolphin and Union group size (n = 148) seen during the survey, where the bins ranged from 1 to 17..... 23
Figure 9: Box plot showing the variation in group size in function of the binned distance. 24
Figure 10: Histogram of detections in function of the binned distance, where events detected by both observers (grey) and events missed by one observer (light blue) were plotted..... 24
Figure 11: Using histogram, exploration of the different covariates (Snow Cover, Fog, and Cloud Cover) in function of the binned distance. 25
Figure 12: Fitted relative detections probability for Model 8 overlaid on a histogram of the true detection probability. 28
Figure 13: Proportion of success of the two observers at different distance bins with fitted relative detection probability for Model 8. 28
Figure 14: Predicted density surface with NDVI and Slope as bivariate smooth for x and y location as predictor. The resulting abundance of caribou is shown within each predicted grid cell of size 9 km^2. The smaller map shows the coefficient of variation per predicted grid cell over the study area to inform areas of higher uncertainty..... 31
Figure 15: Left: Estimates for the Dolphin Union caribou herd on Victoria Island from 1997 to 2023. Right: Population estimates for the Dolphin Union caribou herd on Victoria Island and mainland for 2020 and 2023. 33

LIST OF TABLES

Table 1: Summary of the proposed survey design by stratum, including the stratum area, number of transects per stratum, distance between transect, transect length, and the per cent coverage (effort) of the stratum with a survey strip of 1,500 meters. 13
Table 2: Distance and mark-recapture model covariates. 16
Table 3: Environmental covariates used in the DSM model. 17
Table 4: Timeline of the Dolphin and Union survey as each stratum was completed between October 23rd and November 2nd, 2023. 20
Table 5: Model specifications and model selections results for point independence analysis with removal configuration of observer. Note, hr indicates a hazard-rate model and hn shows a half-normal model, other covariates symbology are explained in Table 2. 27
Table 6: Summary of the estimated number of caribou in each surveyed stratum derived from the detection function fitted with Model 8..... 29
Table 7: Model specification and model selection results for the DSMs, where the model with the highest deviance explained was selected. Note that the covariates were detailed in Table 3..... 30
Table 8: Estimates of the Dolphin Union herd from 1997-2023 for Victoria Island (VI). The 2020 and 2023 estimates for the Dolphin Union for the Victoria Island and mainland (VI/Mainland) are in two separate rows..... 32

ABSTRACT

Fall is the traditional time to survey the Dolphin and Union caribou herd. This is when they gather along Victoria Island's southern coast, awaiting sea ice formation. Since the herd reached a population low in 2018, the survey frequency has increased. In 2023, effort was made to update the herd abundance. The two main goals of this study were: 1) to provide an estimate of abundance, and 2) to detect a trend in the population trajectory. Over a study area similar to 2020, an aerial survey was performed from October 23rd to November 2nd, 2023 using three Twin Otters. The survey effort resulted in 148 caribou groups seen on transect, for 718 caribou. Groups were ranging from 1 to 17 caribou, with a mean of 4.85 caribou in a group. The Dolphin and Union caribou herd formed two aggregations, one close to Read Island and the other inland north of Richardson Island. Since the caribou were clustered in the study area, a Density Surface Model (DSM) was used to estimate the number of caribou on Victoria Island and mainland. This resulted in an estimate of 5,229 caribou (95% Cl = 3,985–6,473, CV = 0.12). The population trend between 2020 and 2023 estimates for Victoria Island and mainland were not significantly different, and the overall finite rate of change was above 1. This study provided an accurate estimate of the abundance of the Dolphin and Union caribou for 2023, which can be deemed stable at best. The Dolphin and Union caribou herd with an estimate of 5,229 caribou is still considered in a low vulnerable phase (less than 8,000 caribou). Due to the importance of this herd in Inuit subsistence and culture, management actions should aim to continue to support its recovery.

INTRODUCTION

The Canadian Arctic is the home of caribou subspecies, such as the Peary (*Rangifer tarandus pearyi*), barren-ground (*Rangifer tarandus groenlandicus*), and the Dolphin and Union (*Rangifer tarandus groenlandicus x pearyi*) caribou. The Dolphin and Union caribou herd uniquely migrates twice yearly across sea ice, connecting its summer range on Victoria Island to its winter range on the Canadian mainland (Poole et al., 2010). Ecological monitoring of caribou abundance is a prerequisite for decision-making in wildlife management and conservation, and helps to describe population dynamics (Hauser et al., 2006). Having current caribou herd estimates and trends help to redirect management actions to either promote herd growth or reduce the herd to a desired level. Indigenous people still rely on wildlife abundance for their subsistence and the continuation of their traditional practices. Caribou herd estimates are deeply significant to Indigenous people subsistence, whose cultural practices rely on abundant wildlife.

Concomitant to other caribou herds facing a circumpolar decline, the Dolphin and Union caribou herd declined from 34,558 (95% CI = 27,757-41,359; CV = 0.12) caribou in 1997 to 18,413 (95% CI = 11,644-25,182; CV = 0.17) caribou in 2015 (Nishi and Gunn, 2004; Dumond and Lee, 2013; Leclerc and Boulanger, 2018). In 2017, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessed the Dolphin and Union caribou herd as Endangered due to continued population decline and numerous threats to the herd. In 2022, the Nunavut Wildlife Management Board (NWMB) and the Federal Minister of Environment recommended up listing the status of the subspecies to Endangered. Since the beginning of the listing process, the Dolphin and Union caribou herd continued to decline, with an estimate of 4,105 caribou in 2018 (Leclerc and Boulanger, 2020). In 2020, the herd stabilized with 3,815 caribou in a study area, including Victoria Island and part of the Canadian mainland (Campbell et al., 2021). The low abundance of the Dolphin and Union caribou sparked some conservation and management concerns (Worthington et al., 2018). Culturally central to the Western Canadian Arctic communities of Kugluktuk, Cambridge Bay, Ulukhaktok, and outpost camps of Bathurst Inlet and Bay Chimo, community members not only witnessed the decline, but were directly impacted by the consequence of reduction in country food (Tomaselli et al., 2018, Hanke and Kutz, 2020).

Monitoring frequency increased if the caribou herd was at or near a critical management threshold or declining. Abundance estimates rely on two main variables, a geographic area and detectability. The calving ground survey method for estimating abundance of barren-ground caribou herds was difficult to perform for the Dolphin and Union caribou herd (Nishi and Buckland, 2000). Then, an alternative fall abundance survey method was established by taking advantage of the herd aggregation on the coast of Victoria Island, while the entire herd await the sea ice to form (Nishi and Gunn, 2004). The fall survey method was revised in 2020, by using Distance Sampling instead of strip-transect method, and by increasing extensively the survey area (Campbell et al., 2021). In a declining herd, the species-level detectability is prone to vary over time, as caribou become more scarce and harder to detect. As a result, survey effort that was once reliable may become inadequate (Kéry, 2002). Decreasing detectability can warrant an increase in survey effort and coverage to ensure that shifts in distribution are captured and absence data can be distinguished from non-detections (McCarthy et al., 2013; Burns et al., 2019; Buckland et al., 2005).

Distance Sampling method models the detection probability of a focal species (caribou), as a function of the distance from a transect to generate an abundance estimate (Buckland et al., 2005). This method is cost-effective to monitor abundance for species that occur over larger area, by extrapolating abundance between transect lines of a defined region. Previous efforts have shown that part of the Dolphin and Union caribou herd can be found inland on Victoria Island (Figure 1). The very low density of caribou in a sparse study area paired with known coastal caribou aggregation was a challenge for Distance Sampling method, Mark-Recapture Distance Sampling (MRDS). Therefore, a model-based inference approach that also includes environmental parameters, Density Surface Model (DSM), was used to increase the accuracy of the estimate (Miller et al., 2013). Although the methods used to monitor the Dolphin and Union caribou herd have evolved with cutting-edge analysis, the surveys were consistently performed in late October, taking advantage of their seasonal aggregation on the coast of Victoria Island.

It was recommended to increase the monitoring frequency of the Dolphin and Union caribou herd abundance (Campbell et al., 2021), so co-management partners can be provided with more control over management actions. Timely revision and change of harvest management, based on up-to-date herd assessments, will help to ensure that a potentially declining caribou herd is not left in a vulnerable state.

OBJECTIVES

The main goal of this study was to monitor the Dolphin and Union caribou herd to support co-management partners, government agencies, and management boards in informing management actions aimed at increasing herd abundance while promoting its recovery. The two main objectives of this survey were: 1) to provide an estimate of Dolphin and Union caribou herd abundance, and 2) to detect a trend in the herd trajectory.

MATERIALS AND METHODS

Dolphin and Union Caribou Herd Seasonal Range

As recommended (Nishi and Gunn, 2004), the Dolphin and Union caribou have been consistently surveyed during the fall. Although the timing of staging period needs to be defined, the period of the rut, October 13th to November 7th (Nagy et al., 2011), was used in this study to be consistent with Campbell et al., (2021). The rut was an optimal time of the year to survey the Dolphin and Union caribou herd based on their distribution (Figure 1). During this period, the Dolphin and Union caribou herd migrates to the coast of Victoria Island and aggregates in higher densities, while both sexes mixed and their movements were reduced (Figure 1) (Leclerc and Boulanger, 2020). The range, at the rut, was more logistically feasible to survey than the entire range. To take into consideration caribou that could not complete their spring migration to Victoria Island, areas on the Canadian mainland were included in the study area as Campbell et al., (2021).

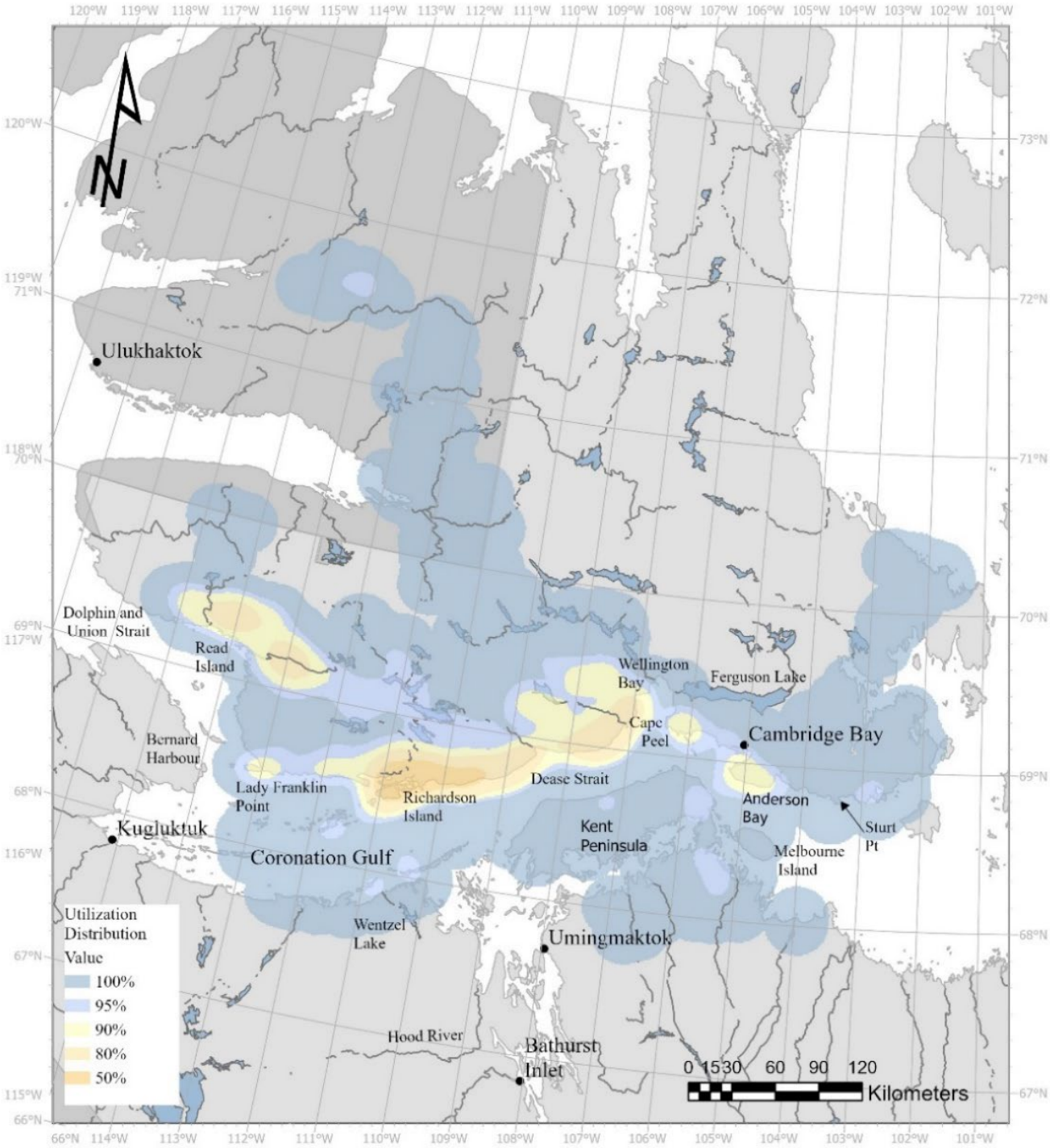


Figure 1: Dolphin and Union caribou herd utilization distribution (%) during the rut, October 13th to November 7th (Nagy et al., 2011).

Integration of IQ in the Survey Design

Consultations with the affected Hunters' and Trappers' Organizations (HTOs) and the Hunters' and Trappers' Committee (HTC) occurred on May 1st, June 1st, July 14th, and September 7th, 2023. The aim was to discuss the proposed survey design and solicit feedback. Since the intent was to use a design similar to the 2020 survey (Campbell et al., 2021), the HTOs and HTC were asked to review the 2020 design to determine if it was still accurately representative of the fall distribution of the Dolphin and Union caribou herd to design the 2023 survey. Because telemetry data provide limited information, which may not represent fully the herd seasonal distribution, incorporating IQ is valuable considering recent shifts in the Dolphin and Union caribou herd distribution (Hanke and Kutz, 2020; Kuptana 2022; Campbell et al., 2021).

During consultation, feedback received from the HTOs and HTC regarding the survey design was captured by creating digital polygons from the meeting map. Consistent with previous surveys (1997 to 2020), there was a consensus that the bulk of the distribution of caribou would be on the south coast of Victoria Island, from Cape Peel to Lady Franklin Point (Figure 2, pink). Due to the longer delay in sea ice formation, Kugluktukmiut and Ekaluktutiak hunters have seen Dolphin and Union caribou abandon their migration and turn back inland (Hanke et al., 2021). These observations reinforced the need to survey inland, leading to a strong recommendation to survey a considerable extent of Victoria Island (Figure 2, blue). IQ has mentioned that the Dolphin and Union caribou was present on the east side of Victoria Island. To acknowledge this information, this area was given a medium-low survey effort (Figure 2, blue). Since hunters have observed Dolphin and Union caribou at the Storkenson Peninsula and Collinson Peninsula, Ekaluktutiak knowledge holders recommended once more to include these areas in the 2023 survey design. Justification remained to include Bathurst Inlet and the Kent Peninsula, as hunters have seen repeatedly some caribou trying to cross back to Victoria Island when the sea ice was no longer adequate, which trapped them on the coast of the mainland in the summer (Haniliak et al., in press). Knowledge holders agreed that the northern and southern strata (Figure 2, dark grey) should be surveyed with minimal effort. The 2023 proposed survey design, representing the HTOs/HTC areas of interest (Figure 2), can be compared to the 2020 design map (Figure 2).

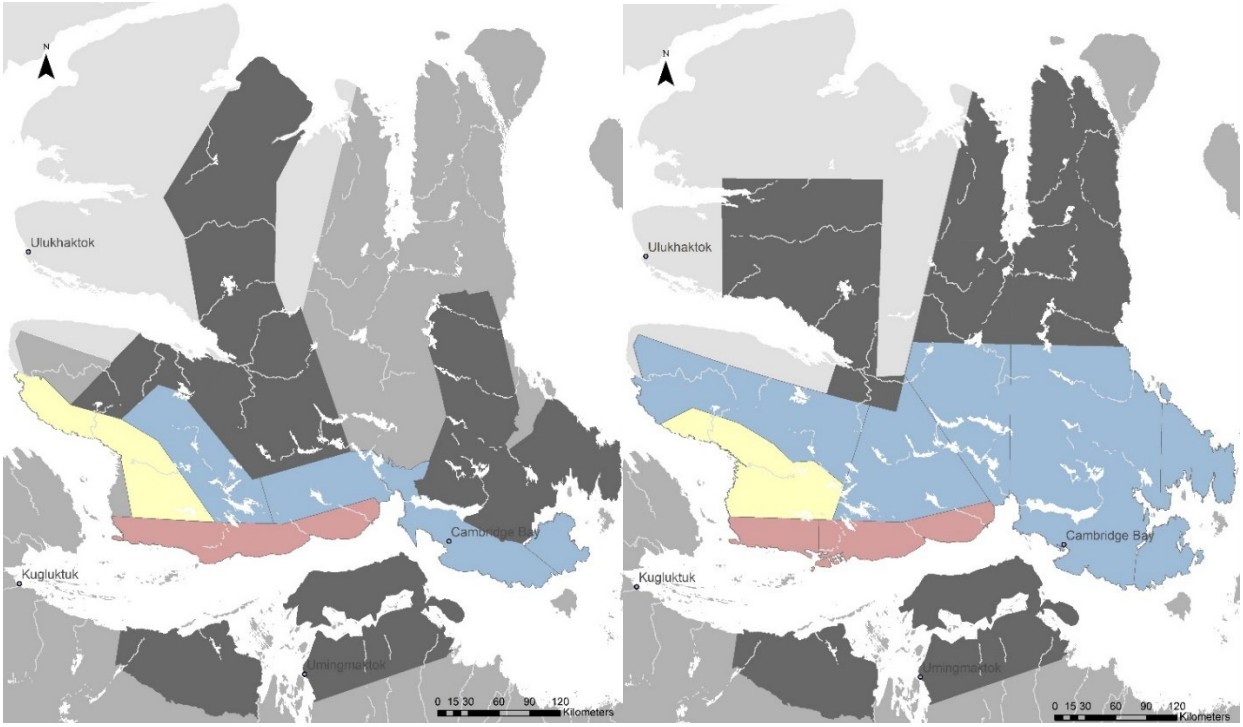


Figure 2: Left: 2020 Dolphin and Union strata placement and effort (dark grey= low effort, blue= medium effort, yellow = high effort, and pink= very high effort). Strata were designated based on historic survey observations, cumulative caribou telemetry data, IQ, and budgetary constraint. Right: Based on IQ, the revised 2023 strata boundaries. Note that the light grey is the Northwest Territories and the medium grey is Nunavut.

Dolphin and Union Daily Movement Rates

The Dolphin and Union caribou telemetry programs ran from 1996 to 2006 and from 2015 to present. Following the seasonal breakdown of barren-ground caribou of Nagy et al. (2011), the period of rut for 2023 was extracted from the entire data set. From October 13th to November 7th, the daily collar locations and movement rates were monitored using the package *TuktuTools* (Curriot et al., 2022) in the R software environment (R Development Core Team 2009). Low movement rates were ideal to time the survey and count caribou, as the animal movement between strata was limited (Leclerc and Boulanger, 2020).

Design of Distance Sampling Surveys

Sampling Design

Identified through IQ and scientific knowledge, the final 2023 study area was 137,549 km^2 (Figure 3). Known caribou density and IQ guided the stratification of this area into 15 strata of unequal effort allocation, 12 strata on Victoria Island and three strata on the mainland. Simulations were used to estimate bias, variance, and confidence interval that a change in survey design could cause, while other parameters (area, population size, detection function) remained constant (Buckland et al., 2005). Therefore, the effort and orientation of the transect lines were tested with respect to

known gradients of caribou density to develop the best design. The true abundance value for each stratum was based on the results of the 2020 abundance estimate (Campbell et al., 2021). The probability of detecting caribou was modeled from a half-normal and hazard rate function, a scale parameter of 350, a truncation of 1,500 meters, and no covariate was used to model the detection probability. Models were tested and evaluated using an Akaike Information Criterion (AIC). Simulations were performed with the package *dsims* (Marshall, 2022a) called for R (R Development Core Team 2009). The best design was picked based on the best precision with: sd of means = 217.76, a mean SE = 249.49, RMSE = 216.68, and per cent bias = 0.06, while having the lowest amount of transect lines. Finally, since the Dolphin and Union caribou herd would aggregate on the coast in higher density, the number of transect lines was increased in the coastal strata.

An unequal amount of effort was set in each stratum following a limit of 19,000 km of transect lines. This led to allocate eight to 59 transect lines in each stratum, with at least 19% coverage in the low-density stratum and up to 74% in the high-density stratum, taking into consideration a maximal distance of 1,500 meters on each side of the plane. The transects, spanning the full width of the stratum, were set perpendicular to known density, to maintain the axes of the stratum, and to assure transects of relative equal length (Buckland et al., 2001). A total of 277 transect lines were randomly drawn using the package *dssd* in the R Statistical Platform (R Development Core Team 2009, Marshall and Rexstad, 2023) for 18,973 km of transects and 28,74% effort in the overall study area (Figure 3, Table 1).

Table 1: Summary of the proposed survey design by stratum, including the stratum area, number of transects per stratum, distance between transect, transect length, and the per cent coverage (effort) of the stratum with a survey strip of 1,500 meters.

Stratum	Area (km^2)	Number of Transects	Distance between Transect (km)	Transect Length (km)	Effort (%)
A	15,439	16	13.5	1,558	21.89
B	8,101	42	5.5	1,990	53.94
C	9,787	10	10	1,383	29.5
D	25,224	17	14	2,310	20.94
E	5,622	17	8.5	958	34.93
F	6,581	16	8.5	1,036	35.24
G	6,972	59	4	2,258	74.32
H	7,577	21	9.5	1,310	30.67
I	13,297	12	14	1,336	21.51
J	11,064	12	14	1,177	21.3
K	2,151	8	8	426	38.07
L	5,716	13	12	820	24.76
M	9,402	11	15	953	19.8
N	8,248	12	15	962	19.55
O	2,375	11	8	495	36.36
Study Area	137,549	277	---	18,973	28.74

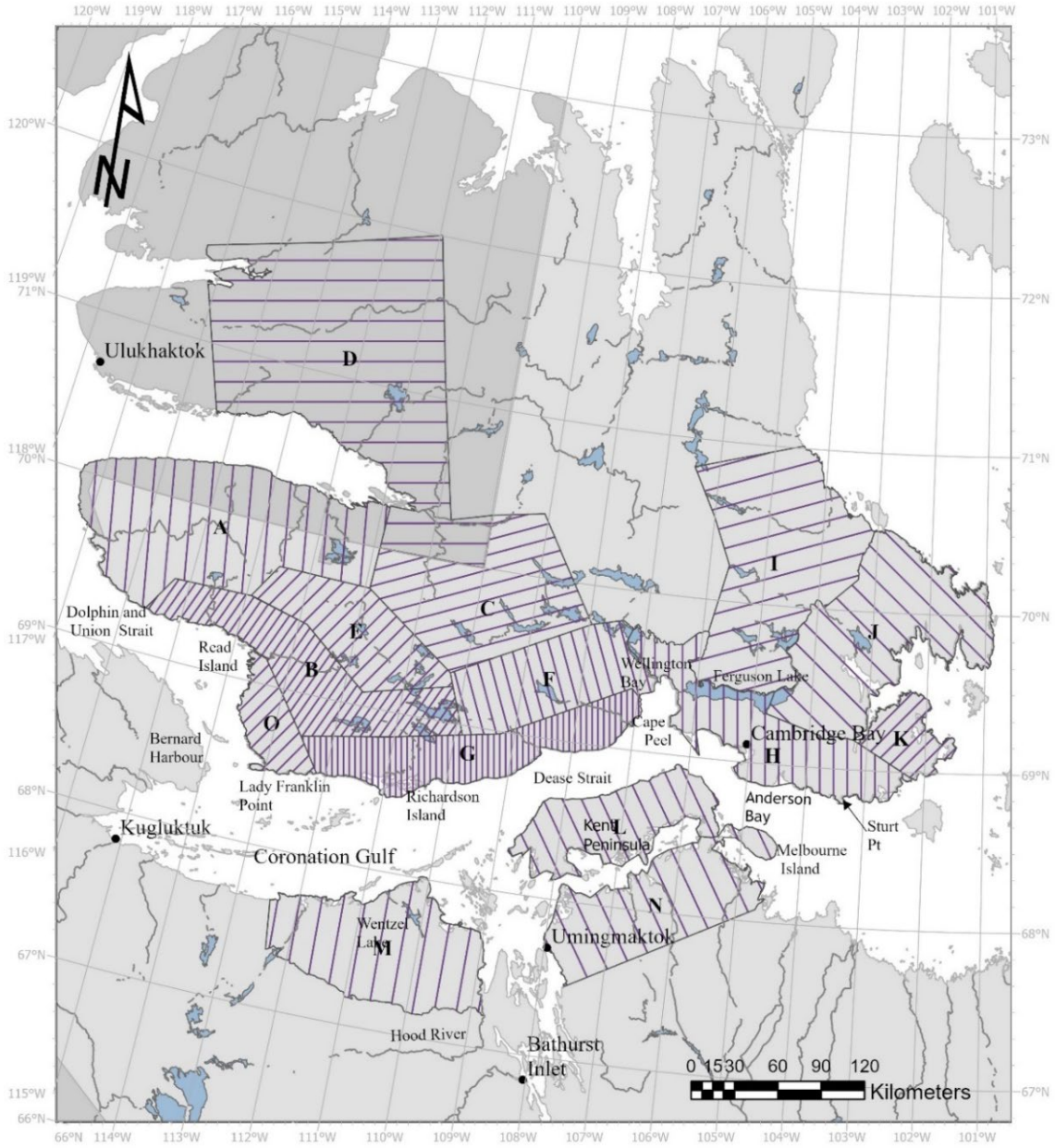


Figure 3: Final 2023 Dolphin and Union caribou herd survey design, with A to O strata and the transect lines.

Field Setting

Three Twin Otters were used to fly the survey, where one was based in Kugluktuk, Cambridge Bay (Nunavut), and the last one in Ulukhaktok (Northwest Territories). The transects were flown at a speed of 160-180 km/hr, while maintaining an altitude of 122 m (400 feet) taking into consideration the topography. Wing struts were equipped with rope to set spacing bins at: 0-200 meters, 200-400 meters, 400-600 meters, 600-1,000 meters, and 1,000-1,500 meters for a maximum sampling distance of 1,500 meters on each side of the plane (Campbell et al., 2021). Bins were calculated using the formula of Northon-Griffiths (1978):

$$w = W*h/H$$

Where W was the required strip width; h the height of the observer's eye from the tarmac; and H the required flying height. Since the observers could not see under the plane, the start of the first bin was extended beyond the wheel of the Twin Otters.

Since Distance Sampling needed at least 60 to 80 observations to fit a detection function, the dependent double-pair observer configuration was used to increase the probability that observers detect less conspicuous lone caribou and to assess the detection probability (Buckland et al., 2005; Campbell et al., 2021). In this configuration, both observers were fully aware of each other's observations, making the roles of the observers asymmetric. The detection pattern was: {1,0}, {0,1}, and {1,1} depending if the first observer only, the second observer only, or both observers detected the group. Observations were made perpendicular to the transect lines, and the observations were based on the groups of caribou observed. If the plane triggered a movement response from the caribou, the observers were required to call the bin before the movement had occured (Buckland et al., 2005). The first and second observers alternated place after each flight.

The overall detection probability was expected to be very high close to the aircraft and to decrease with increasing in distance (Buckland et al., 2001). To make sure this assumption was met, mandatory observers' training, recording the distance before caribou were disturbed and moved, and daily plotting of the data for immediate observers' behaviour correction were implemented in the field.

The first recorder logged the observations for both left front and back observers, and the second recorder noted the observations for right front and back observers. The recorder's task was restricted to data entry. Information documented included the GPS locations of the observation (group), the binned distance, the number of caribou, and sighting conditions, to name a few. Observations beyond 1,500 meters or made during ferry flights between transects were recorded as "off transect" and not included in the final analysis. Covariates influencing detections were recorded, such as snow cover, cloud cover, and presence/absence of fog. During this survey, other species sightings were recorded, such as, but not limited to: wolverine (*Gulo gulo*), wolf (*Canis lupus*), polar bear (*Ursus maritimus*), moose (*Alces alces*), and muskox (*Ovibos moschatus*).

Data Analysis

Despite some poor visibility conditions, the double observer pair structures were maintained during the survey, ensuring that the probability of an observer to detect a group was consistent across all events. Some transects were flown a second time to confirm that no additional groups and lone caribou were missed. The observation generated from the second flight was not used to derive the detection function or estimate analysis, because of the collapse in the double-pair observer structure. The pilots and recorders were assigned with different tasks and these distractions were making them unavailable to be committed to observe full time, impacting negatively their detection probability. In addition, observations made by the pilots and recorders were not taken into consideration, because the limitation of the current model in integrating a third pair of observers to compute detectability.

Mark-Recapture Distance Sampling (MRDS): Detection Function

First, a routine data visualization was performed to provide a firm rationale in the choice of covariates for the model. The distribution of the groups observed on transect was mapped in relation to the strata and a histogram was plotted to explore the frequency of observation for each group size. The frequency of distributions of each covariate (cloud cover, snow cover, group size, presence/absence of fog) as a function of distance was plotted to capture visually any trends between these variables. Since some observers had no observations, and observers spend an unequal amount of time surveying due to a high turnover rate, it was not possible to use observer as a covariate.

As both observers were aware of each other observations, the removal configuration was implemented in the MRDS analysis (method = rem), which limits the future use of alternative methods. The program R with the packages *Distance* and *mrds* were used use as a sequential approach for model building and to fit a detection function to the observations (Laake et al., 2012). The final covariates taken into consideration in the model were: distance, group size, cloud cover, presence/absence of fog (Table 2). Each model was evaluated using an Akaike Information Criterion (AIC) (Burnham and Anderson, 1998). In addition, a goodness-of-fit test (or chi-square test (χ^2)), was performed to test for adequate fit (Buckland et al., 1993; Buckland et al., 2005). The overdispersion (c) was calculated by using the observed chi-squared for a global model divided by its degrees of freedom. If there is no overdispersion, then c should be equal to 1. Selection of the best model was based on lowest AIC score, goodness-of-fit test, and overdispersion (Burnham and Anderson, 2001).

Table 2: Distance and mark-recapture model covariates.

Covariate	Abbreviation	Type
Distance bin from plane	D	ordinal
Group size	S	continuous
Cloud cover	C	ordinal
Visibility (fog)	F	ordinal

As some transects were not surveyed in strata D and I, the lengths of these transects and area were adjusted to reflect the discrepancy with the planned design. The abundance estimate was then derived from the best model with strata-specific estimates. Variance was estimated in the *mrds* package, which considered the distance sampling, mark-recapture, and encounter rate variation (Innes et al., 2002). Two MRDS estimates were performed, one for Victoria Island/Mainland and one for Victoria Island only.

Density Surface Model (DSM)

Following a two-step approach, the spatial habitat components were added to the best MRDS model for Victoria Island/Mainland (Miller, 2012, 2013; Winiarski et al., 2013). Since the final 2020 abundance estimate was for Victoria Island/Mainland, the DSM analysis was performed only

for this area. To help guide covariate selection, the covariates that were known to influence the spatial distribution of other caribou herds were first considered (MacNearney et al., 2016). Since the timing of the survey coincided with the fall migration, ecological factors that might have impacted their distribution were considered. Caribou have been found to express fidelity to their migration route, and therefore the distance from the historical crossing locations (XingDist) was taken into consideration, as well as the distance to lake (LakeDist) and river (RivDist) (Duquette, 1988; Nicholson et al., 2016). Since caribou are ruminant, patches of adequate foraging quantity and quality across the landscape can similarly affect their distribution pattern. The Normalized Difference Vegetation Index (NDVI) of July was used to inform areas with high photosynthetic activity and productivity. Since the survey occurred at the end of October, outside the peak of plant productivity, the Normalized Difference Tillage Index (NDVT) was also used to determine the non-photosynthetic biomass, crop residue, that would be available as winter forage for caribou. For the topographic variables, the topographic roughness index (TRI) and slope (Slope) were considered (Table 3). To facilitate the interpretation of the variables and avoid multi-collinearity, the collinearity between the covariates was assessed using a pearson correlation test with a threshold of 0.75.

Table 3: Environmental covariates used in the DSM model.

Covariate	Abbreviation	Description
Distance from crossing location	Xing Dist	Based on collar data 2015 to 2022 crossing location along the coast of Victoria Island
Distance to river	RivDist	Distance to large streams calculated using Canada National Atlas 2 million Hydrology
Distance to lake	LakeDist	Distance to large streams calculated using Canada National Atlas 2 million Hydrology
Slope	Slope	Slope based on 30 m digital elevation model (DEM)
Terrain Roughness Index	TRI	
Green Vegetation	NDVI	Normalized Difference Vegetation Index to measure photosynthetic biomass
Dry Vegetation	NDTI	Normalized Difference Tillage Index to measure non-photosynthetic biomass

Using the fitted detections functions from the MRDS, a Density Surface Model (DSM) was built by fitting the selected environmental covariates (Table 3). Beside deriving a more precise abundance estimate, the DSM provided clues to the environmental covariates that were potentially driving the spatial distribution observed during the survey. Thus, the workflow of the DSM can be illustrated as per Figure 4 (Adapted from Miller et al., 2013). The fitted model with the higher deviance explained was selected, checked for goodness-of-fit, and the randomized quantile residuals were plotted for visual exploration. The best DSM model was used to calculate the abundance estimate and to derive the uncertainty of the estimation (coefficient of variation).

The *Distance* and *dsm* packages were used as a sequential approach for model building to fit a detection function to the observations (Laake et al., 2012; Miller et al., 2013). The final density and CV maps were plotted over a Digital Elevation Model (DEM) to facilitate visualization (NRCan, 2017).

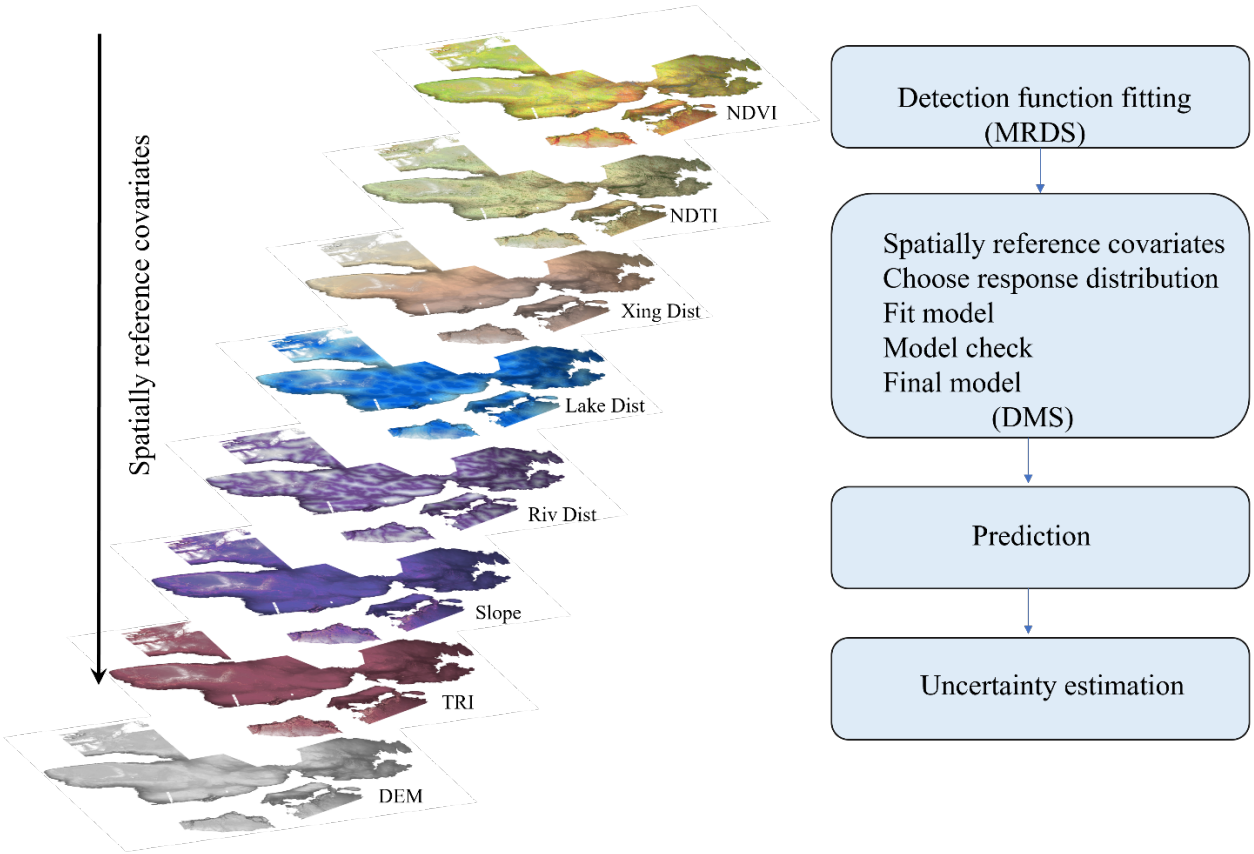


Figure 4: Work flow of the Density Surface Model (DSM) adapted from Miller et al., (2013).

Population Changes and Overall Trend

The 2023 estimate was initially compared to the 2020 estimate using a t-test ($\alpha = 0.05$) to determine if they were significantly different. In addition, the rate of change (lambda (λ)) between survey years was calculated following this equation:

$$\lambda = N_{t+1}/N_t$$

If λ is higher than 1, the population increased, below 1 the population decreased and very close to 1 the population is stable (Gasaway et al., 1986).

RESULTS

Dolphin and Union Daily Movement Rates

Daily, the collar locations of 21 cows were monitored to track their position in relation to the strata, hoping that the majority would have reached and aggregated in the high-density coastal Stratum G by the end of October (Figure 5). The 5 km/day threshold is usually used during calving ground survey to detect the peak of calving where cows' movement become very low. Early in the survey, some caribou movement rates were still above 5 km/ day and the movement of the third quartile rated became below the threshold as the strata were surveyed (Figure 5). Since the caribou showed low movement rates, caribou did not move out of the survey area or stratum. The movement rates increased slightly after the survey, but the median remained below the threshold. The displayed movement rates did not increase as expected, showing no sign that their fall migration resumed despite entering the month of November. In fact, the last day of the survey, November 2nd, none of the collars resumed their migration and none left Victoria Island for the sea ice (Figure 5). Therefore, the Dolphin and Union caribou that summered on the island were confidently captured on the island during the survey.

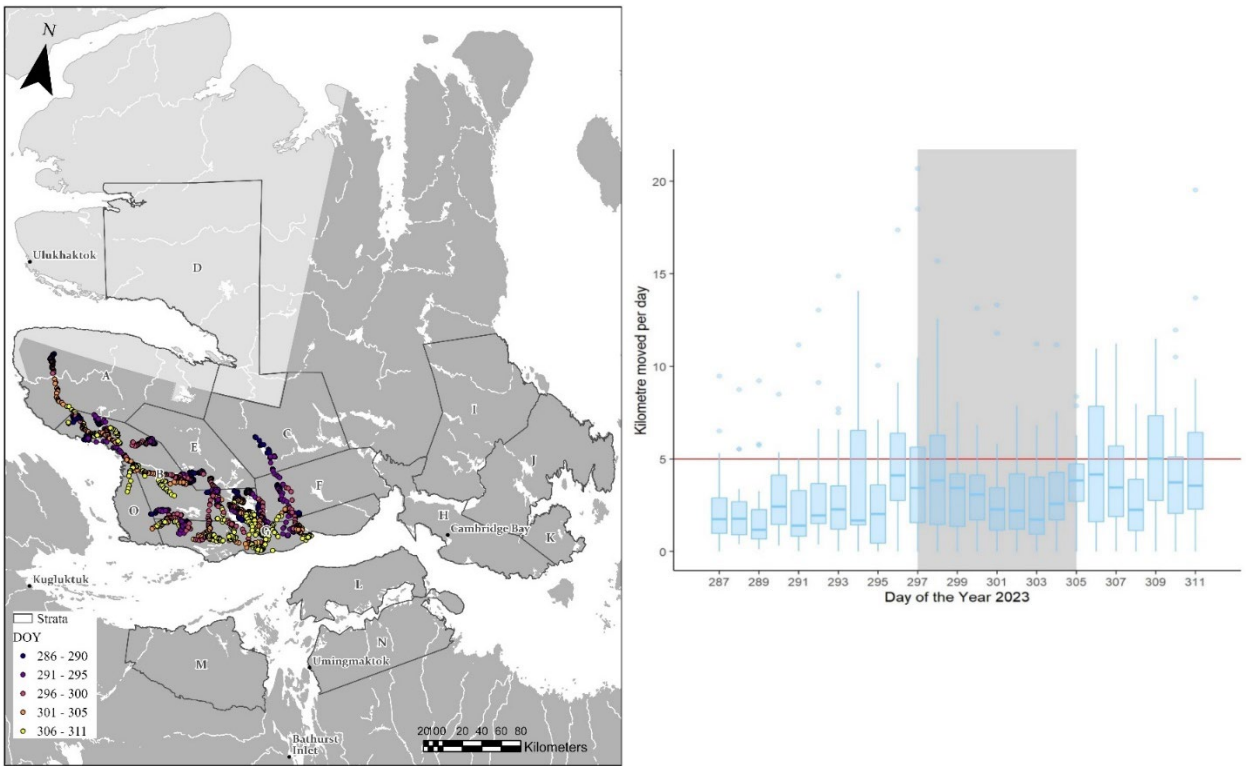


Figure 5: Left: Collar locations (n = 21) during the rut, which as has been established with a start date of October 13th (dark blue) and a final date of November 7th (yellow) for 2023. Right: Dolphin and Union collar daily movement rate from October 13th to November 7th, where the shaded grey highlights the period of the abundance survey.

Survey Conditions

The Dolphin and Union survey occurred from October 23rd to November 2nd, 2023, with three Twin Otters (C-GNPS, C-ATO, and C-FATM). The aircraft arrived at Kugluktuk on October 23rd. After being fitted, two planes departed, one to Cambridge Bay and the other to Ulukhaktok. On October 24th, the very-low-density stratum to the extreme north (D) and on the mainland strata (M, N, and L) (Table 4) were surveyed.

Strata A, E, O were progressively completed and the strata to the east of Cambridge Bay where no collars were present. On October 31st, the three planes surveyed simultaneously strata B, G, and F where the collars were located. The following day, the unfinished areas of those three strata were completed. Some transects were surveyed twice to make sure that no large groups were potentially missed because of poor visibility. The transect numbers 158 to 169 (Figure 7, light pink) in stratum G were flown twice, where nine groups of caribou were seen the first time and five groups were seen the second time. Since fewer groups were seen on the second attempt, the data were discarded. During the survey, two days of poor weather (October 25th and 30th) prohibited the planes from flying. There was a small section of stratum D and the northernmost four transects of stratum I that were left not surveyed (Figure 6).

Table 4: Timeline of the Dolphin and Union survey as each stratum was completed between October 23rd and November 2nd, 2023.

Date	Aircraft and Strata		
	C-GNPS	**C-ATO**	**C-FATM**
October 23	Arrival Kugluktuk	Arrival Cambridge Bay	Arrival to Ulukhaktok
October 24	Stratum M	Strata N&L	Stratum D
October 25	Stratum L	Strata H&I	Weather
October 26	Strata M&O	Strata I&K	Stratum D
October 27	Stratum A	Strata K&J	Stratum D
October 28	OFF	OFF	OFF
October 29	Stratum E	Stratum C	Stratum A
October 30	Weather	Stratum F	Weather
October 31	Stratra B&G	Strata B,G&F	Strata B&G
November 1	Stratum B &Departure	Strata G&F	Departure
November 2	----	Stratum G and Departure	----

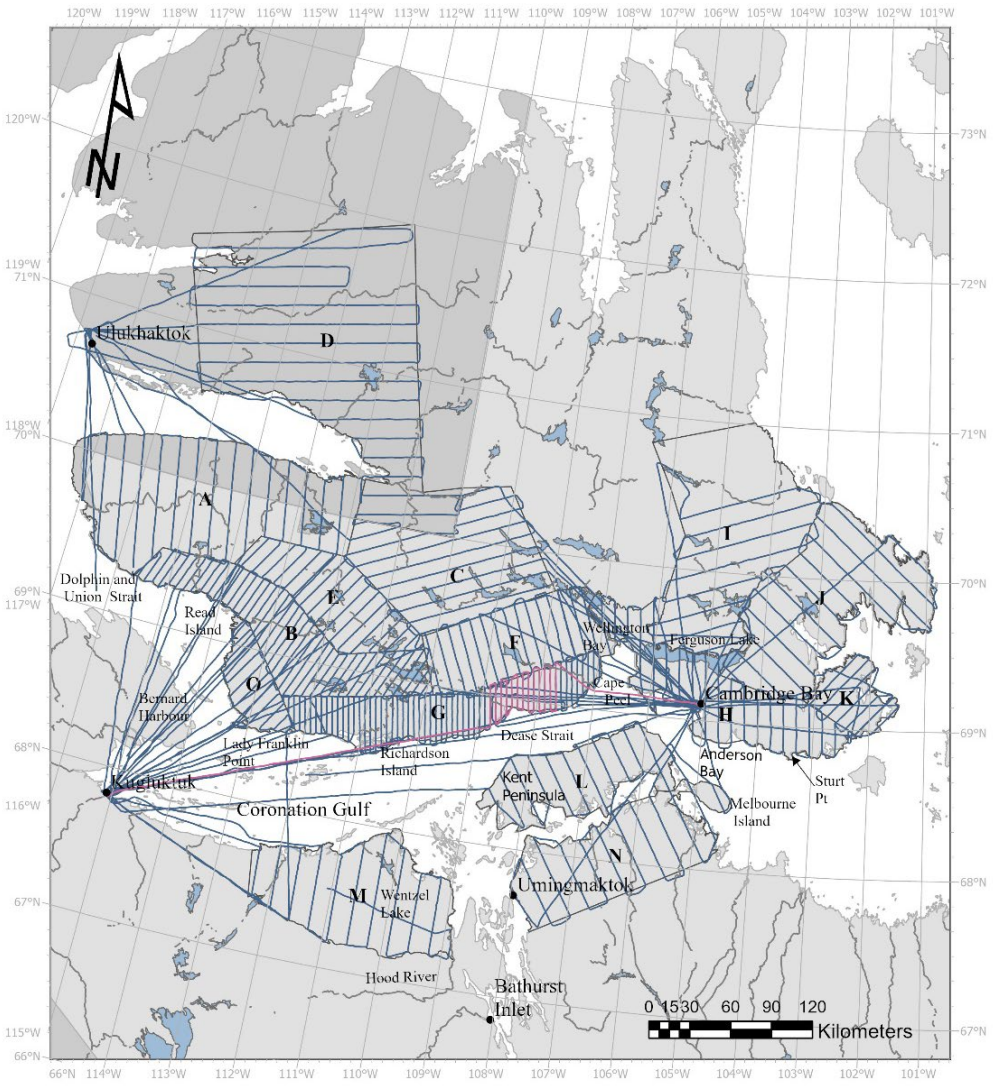


Figure 6: Flight tracks (blue) between October 23rd and November 1st, 2023. A section of G stratum was re-flown (light pink) on November 2nd, because of poor visibility during the first attempt.

Data Exploration

Distribution

During the survey, no caribou were seen in strata I, K, and M, whereas the bulk of the observations were in strata B, G, and F (Figure 7). Two separate aggregations were seen in the study area: one close to Read Island and the other 39 km north of Richardson Island inland. In this aggregation, most caribou groups were seen at the north boundary of the Stratum G and to the west in the Stratum B. Some caribou groups were seen on the east side of Bathurst Inlet and other groups were observed on the Kent Peninsula. Although there might be a possibility that the Dolphin and Union caribou summer on the Kent Peninsula, some observers noticed a mix of smaller, whiter caribou and some bigger, browner ones at this location. In the Northwest Territories (Stratum D), four caribou groups were observed. There were 148 groups of caribou seen on transect, summing to 718 Dolphin and Union caribou.

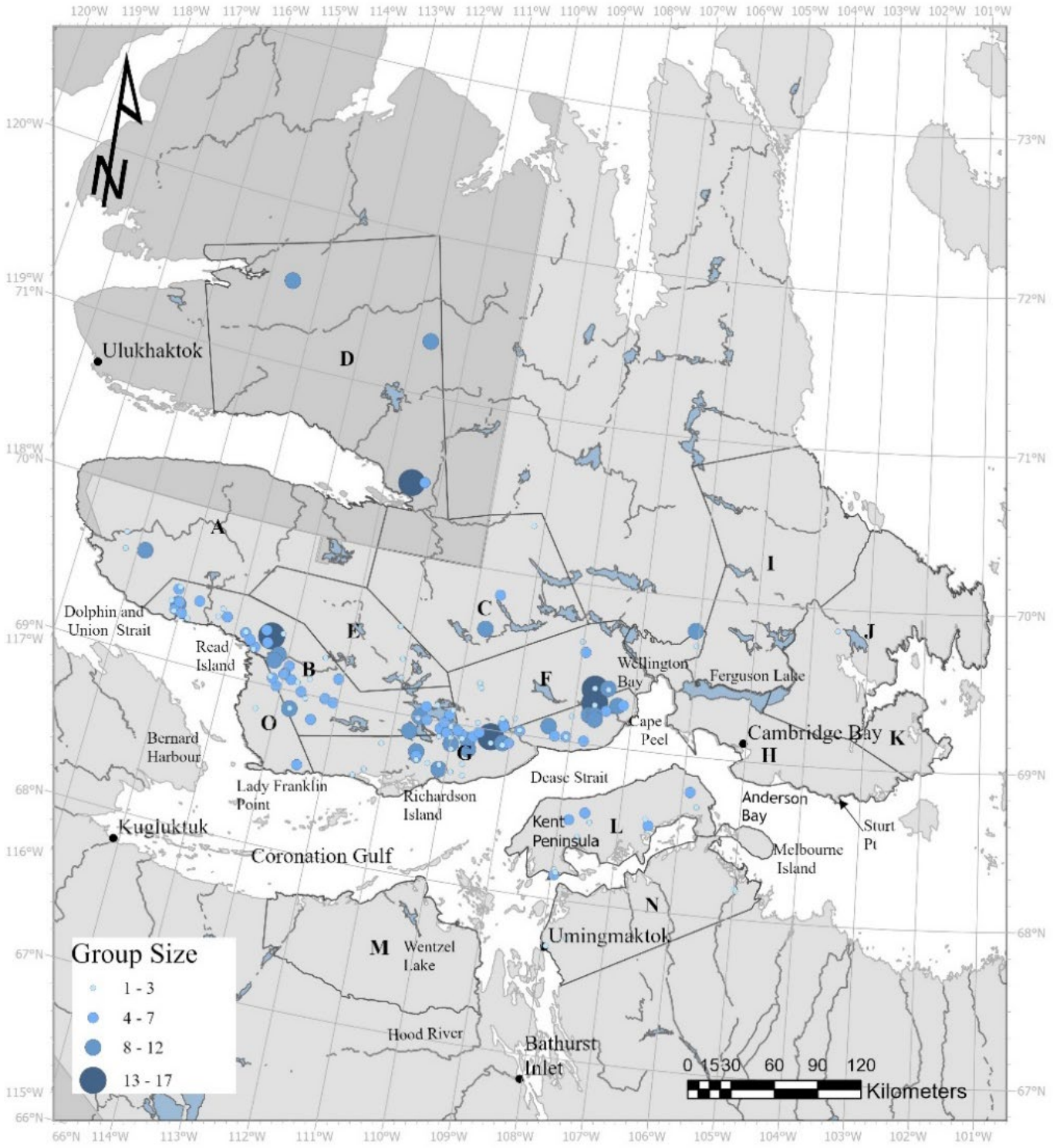


Figure 7: Overview of the 148 Dolphin and Union caribou groups' locations observed on transects during the survey.

Group Size

The group sizes encountered varied from a minimum of 1 to a maximum of 17, where most of the groups were small, with a mean of 4.85 (sd = 3.47) (Figure 8). As the maximum group size was 17, this suggests that the Dolphin and Union caribou did not form large groups in 2023. The data were skewed toward the first bin, 1-3, which reflects the asymmetry of the dataset.

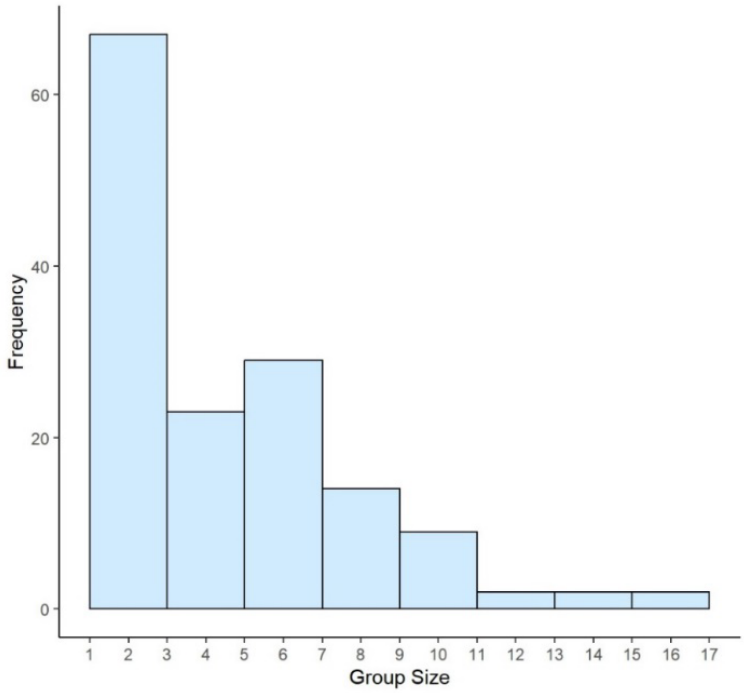


Figure 8: Histogram displaying the Dolphin and Union group size (n = 148) seen during the survey, where the bins ranged from 1 to 17.

Distance Sampling Analysis

Choice of Covariates

Initially, the relationship between each parameter and distance was visualized. Even if the distribution of the group size ranged from 1 to 17, smaller groups were easier to detect closer to the plane, while larger groups were easier to detect away from the plane. As shown by the box plot, the first two bins had a median of three individuals per group, increasing to six in the 4th and 5th bins (Figure 9). Small caribou groups were detected in the last bin confirming ideal detection conditions. To illustrate the underlying distribution in each bin, the individual observations are represented by points, which allowed to distinguish outliers.

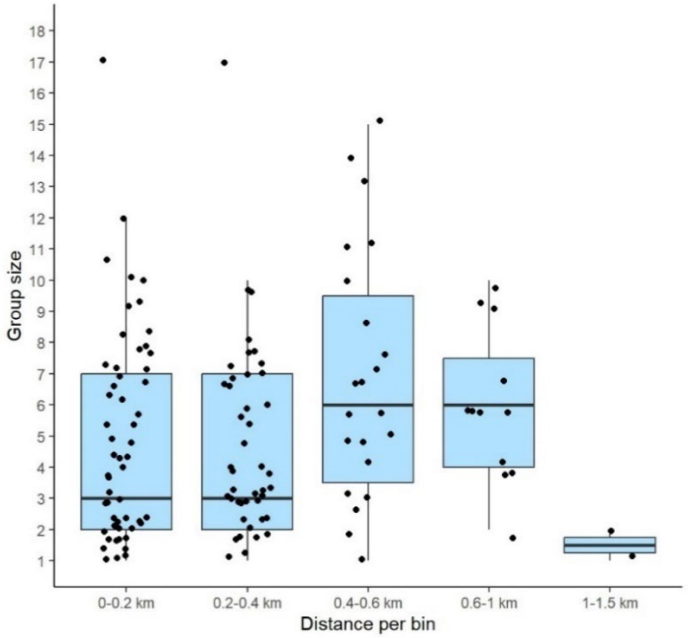


Figure 9: Box plot showing the variation in group size in function of the binned distance.

This plot confirmed that the assumption of the model was met (Figure 10). The number of observations varied as function of distance, as the number of detected events gradually decreased from the first bin. Thus, observers were detecting more caribou in the first bin.

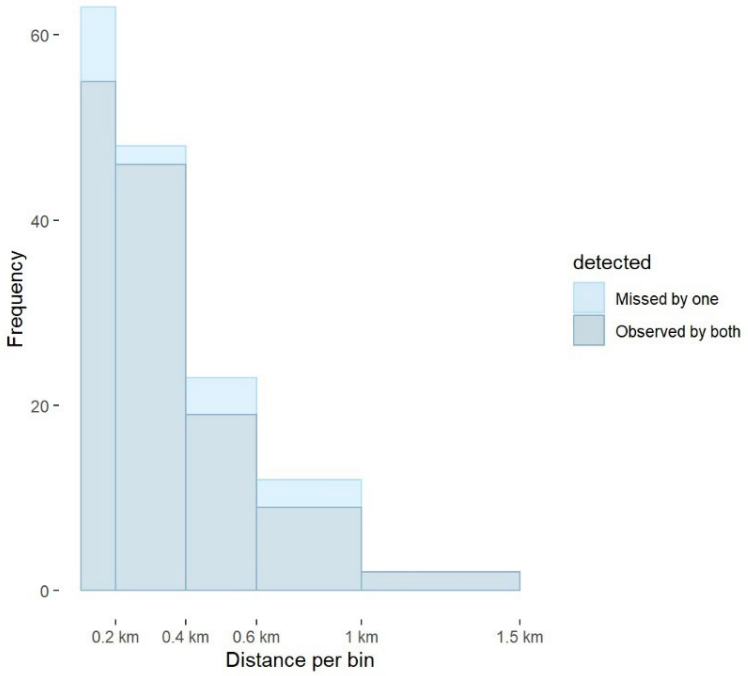


Figure 10: Histogram of detections in function of the binned distance, where events detected by both observers (grey) and events missed by one observer (light blue) were plotted.

The continuous variables, snow cover and cloud cover, were transformed into discrete variables based on a predefined category (Figure 11). Since the percentage of snow cover varied arbitrarily based on the judgment of different recorders, they were reclassified in four categories, where data were only available for two categories (50-75% and 75-100%). Only four observations were made in the 50 to 75% category, and the remaining (n = 144) fell into the 75-100% category. Since the snow cover was uniform in the study area, consistent snow cover would have had no effect on the detection probability (Figure 11). This parameter was no longer considered in the analysis.

Fog was classified as present or absent. Most of the observations (n = 118) were made in the absence of fog, which reinforces our confidence that the caribou groups were available to be detected and were less likely to have been missed. However, when the fog was present, a spike in the first bin was observed, and the number of observations quickly decreased as a function of distance. Fog can impair the ability to detect caribou, as the fog layer can be thick and obstruct the visibility. This parameter was kept and investigated in the model.

Finally, the cloud cover as a function of distance was also visualized. The survey occurred either in low (less than 25%) or high cloud cover (over 75%). Observer's perception could be affected by the intensity of the sun or the flat light created by a variation in cloud cover. Thus, this variable was considered.

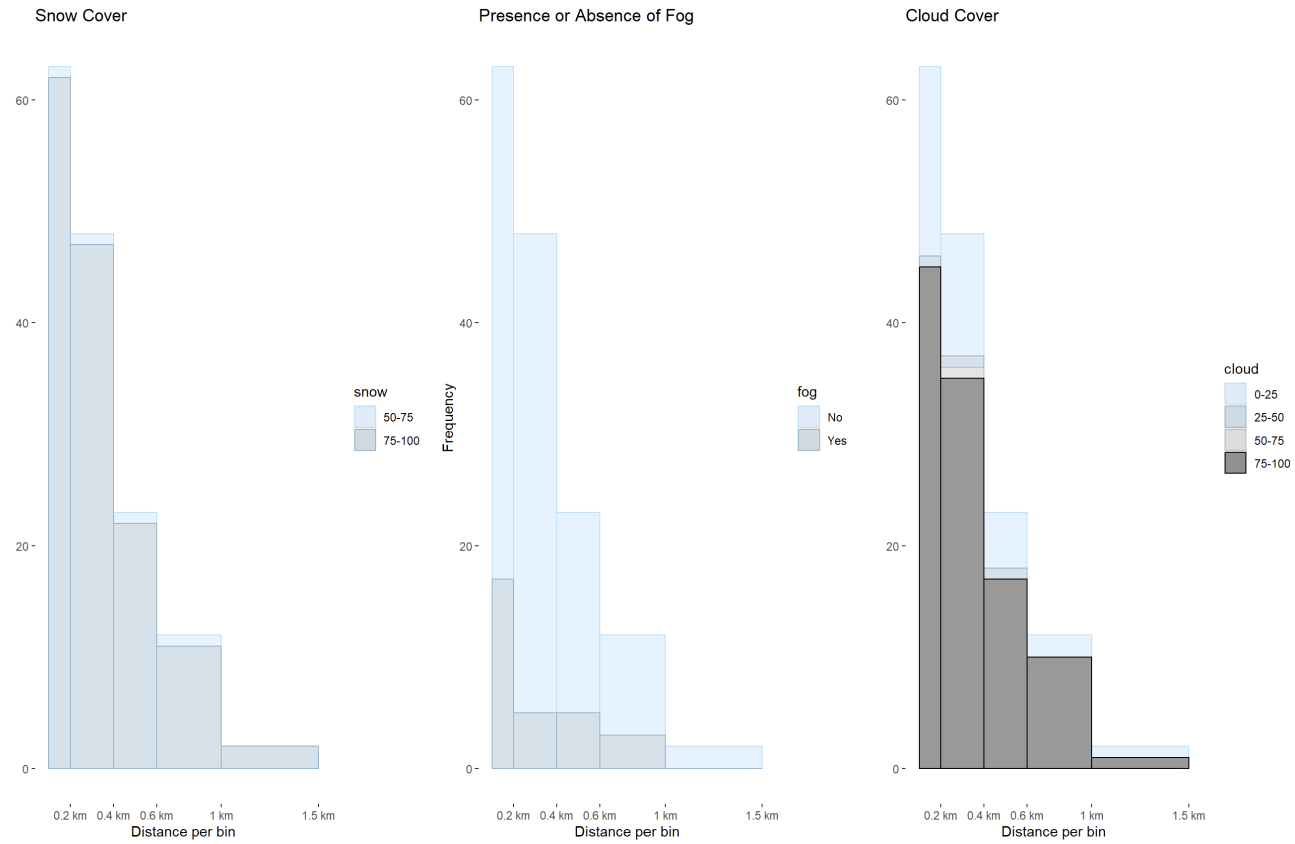


Figure 11: Using histogram, exploration of the different covariates (Snow Cover, Fog, and Cloud Cover) in function of the binned distance.

Mark-Recapture Distance Sampling (MRDS): Model Selection

Since the observers worked together to detect caribou, the caribou observed by the front observer was detected by the back observer with certainty. This resulted in our inability to estimate the detection probability of the back observer, rendering it not a mark-recapture event (Buckland et al., 2007). Therefore, a slight modification to the calculation for likelihood was applied and the "removal configuration" method was used. A forward stepwise model selection procedure based on the AIC score was implemented, starting by investigating the Distance Sampling (DS) model and Mark-Recapture (MR) model with no predictor covariate for the half-normal (hn) and hazard-rate (hr) key function (Model 00 and 01). A combination of different variables was iteratively introduced until all were included. In many ecological situations, multiple models in the candidate set may be reasonable, and the "best" model was not apparent. This is valid when all the model with AIC difference is less than 2 (see Model 00 and 01) (Burnham and Anderson, 2001). A goodness-of-fit assessment (χ^2) was performed to further test the fit of the model on the true values. The Model 03, with the lowest AIC score, had an acceptable fit ($\chi^2 = 9.42$, df = 4, p = 0.051), but an overdispersion of 2.33 showing a lack of fit (Burnham and Anderson, 2001). The fit between the true values and the Model 8 was acceptable ($\chi^2 = 8.00$, df = 5, p = 0.091) with an overdispersion c = 1.6. From the distribution of the data, the half-normal represented a better visual fit, as it assumed that sightability decreased quickly with distance, which is particular to aerial survey because of the difficulty of detecting animals from higher altitude (Clancy et al., 1997). Based on its lower AIC score, goodness-of-fit, overdispersion closer to 1, the model that included the terms distance, groups size, and fog in the MR model (Model 8) was selected as the best model (Table 5). Model 8 was characterized by a point estimate of p (0) for both platforms combined of 0.98 (SE = 0.010).

Table 5: Model specifications and model selections results for point independence analysis with removal configuration of observer. Note, hr indicates a hazard-rate model and hn shows a half-normal model, other covariates symbology are explained in Table 2.

Model	Key	DS model $g.(\underline{z})$	MR model $p_{j\|3-j}(y,\underline{z})$	K	AICc	ΔAICc	w_i	likg	GOF p-value
03	hr	logS	S	5	484.5401	0.000	0.517	1.00	0.051
02	hr	S	1	5	487.1513	2.611	0.140	0.271	0.196
8	hn	1	D+S+F	5	487.3154	2.775	0.129	0.249	0.0915
7	hn	1	D+F	4	489.1471	4.607	0.052	0.099	0.2775
01	hn	1	1	2	490.2732	5.733	0.029	0.057	0.2791
9	hn	1	D+C+F	5	490.7420	6.202	0.023	0.045	0.1624
10	hn	1	D+S+C	5	491.0470	6.507	0.019	0.039	0.0478
5	hn	1	D+S	4	491.1175	6.577	0.019	0.037	0.1124
00	hr	1	1	3	491.5933	7.053	0.015	0.029	0.2714
1	hn	1	D	3	491.7291	7.189	0.014	0.027	0.2791
6	hn	1	D+C	4	491.9118	7.372	0.013	0.025	0.1705
2	hn	S	D	4	492.5159	7.976	0.009	0.019	0.2206
0	hr	1	D	4	493.0493	8.509	0.007	0.014	0.7073
4	hn	F	D	4	493.4028	8.863	0.006	0.012	0.1879
3	hn	C	D	4	493.7184	9.178	0.005	0.010	0.1945

A visual exploration of the plot of the relative detection probability for Model 8 (Figure 12) did not suggest any lack of model fit, as the estimated probabilities were all below the true values. Figure 13, showing the proportion of success at different distance of the line, the smooth line is going down slowly, showing that the heterogeneity of detection has been considered and modelled. However, there was still some unmodelled heterogeneity for the further bins, as the line would have expected to come down more (Buckland et al., 2007). If heterogeneity is not adequately modelled with the MR model, large bias can ensue with a positive bias in estimation of the detection probability and negative bias in estimation of the abundance (Link, 2003). Heterogeneity was not properly modelled in Model 03 and Model 02, as the smooth line was horizontal across all distances (data not shown), inferring that using the covariate size alone was not enough to model heterogeneity. Since there was still some heterogeneity remaining in Model 8, a point independence model was used, so the model was less sensitivity to this remaining unmodelled heterogeneity (Fewster and Pople, 2008). This resulted in an estimate overall average detection probability ($\hat{p}$) of 0.3.

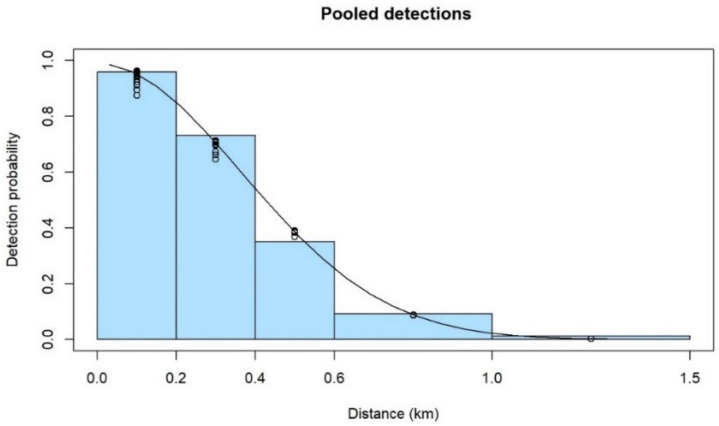


Figure 12: Fitted relative detections probability for Model 8 overlaid on a histogram of the true detection probability.

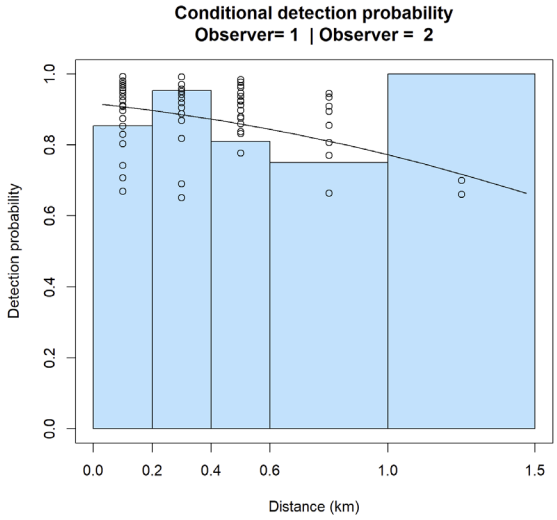


Figure 13: Proportion of success of the two observers at different distance bins with fitted relative detection probability for Model 8.

Estimates

the best MRDS model, Model 8, the estimate was 4,589 (95% Cl = 3,391-6,210, CV = 0.15) caribou for Victoria Island and 5,363 caribou (95% Cl = 4,061-7,081, CV = 0.14) for Victoria Island and the mainland (Table 6). The bulk of the observations were made in the Strata B, F, and G accounting for over 60% of the total estimate (Table 6). The stratum in the Northwest Territories (Stratum D) and the Kent Peninsula had a sizable number of caribou, 649 and 620 respectively. However, the confidence in these numbers was low, as in Stratum D only four groups were seen over a very large area (24,518 km^2) and in Stratum L barren-ground caribou was confirmed by observers. On the mainland, a large majority of the observations were made on the Kent Peninsula (Stratum L), but the number was still minimal, with only 47 caribou counted. In both cases the coefficients of variation (CV) were higher than the recommended threshold of 0.20 (Pollock et al., 1990). These two MRDS estimates (Victoria Island and Victoria Island and the mainland) were relatively precise, with a coefficient below 0.20, providing confidence in the total caribou estimated.

Table 6: Summary of the estimated number of caribou in each surveyed stratum derived from the detection function fitted with Model 8.

	Stratum	Caribou	N	SE	CIL	CIU	CV	Area (km^2)
Victoria Island								
	A	12	181.63	125.83	47.89	688.82	0.69	15,432
	B	272	1,667.72	353.63	1,093.80	2,542.79	0.21	8,101
	C	19	217.01	119.44	68.02	692.30	0.55	9,787
	D	39	648.52	388.40	200.88	2,093.68	0.60	24,518
	E	7	66.83	38.94	21.28	209.88	0.58	5,622
	F	70	568.03	262.24	223.10	1,446.25	0.46	6,581
	G	239	1,030.63	237.13	654.85	1,622.05	0.23	6,972
	H	10	105.91	102.39	19.47	576.19	0.97	7,577
	I	0	0	0	0	0	0	8,539
	J	2	31.42	31.18	5.09	194.11	0.99	11,064
	K	0	0	0	0	0	0	2,151
	O	8	71.67	44.26	20.25	253.58	0.62	2,375
Victoria Island:		**662**	**4,589.38**	**705.01**	**3,391.32**	**6,210.67**	**0.15**	**108,719**
Mainland								
	L	47	620.43	212.22	301.82	1,275.40	0.34	5,716
	M	0	0	0	0	0	0	9,402
	N	9	153.81	95.66	43.57	539.98	0.62	8,248
Total Mainland:		56	773.81	234.05	415.49	1,441.16	0.30	23,366
Victoria Island/Mainland								
Victoria Island/Mainland:		**718**	**5,363.19**	**757.62**	**4,061.91**	**7,081.34**	**0.14**	**132,085**

Density Surface Model

This vast study area spanned across high and less suitable caribou habitat, as the distribution of caribou in the survey area was not uniformly distributed (Figure 7). Most caribou were aggregated in three strata (B, F, G), which inferred a relationship between variation of density and specific environmental covariates in the study area. Therefore, the MRDS model (Model 8) was further modelled using a Density Surface Model (DSM) to provide inference about caribou abundance and habitat. A pearson correlation test was run on the covariates to investigate the strength and the direction of the linear relationship between them. Two pairs of variables, NDVI and NDTI as well as TRI and Slope, had a correlation coefficient (r) higher than 0.75, and were not taken into consideration together in a model. Choice between these covariates was made by selecting the DSM providing the highest percentage of deviance explained (Table 7).

Model Selection

After estimating the abundance for each transect segment, a generalized model was fitted to the resulting estimate using a quasi-Poisson structure. Since a Tweedie distribution provides adequate flexibility and better captures the over-dispersion in the data, it was used to create a smooth function. Each covariate was investigated separately. The preferred model was selected based on the higher deviance explained. Thus, Model 10 with the highest deviance explained, 28.5, was selected. This model was also tested using goodness-of-fit test (NDVI p = 0.94; Slope p = 0.12). Randomized quantile residuals were investigated by transforming the residuals to be normally distributed, and the residuals versus a linear predictor were plotted, visually explored, and interpreted with caution to assure model fit.

Table 7: Model specification and model selection results for the DSMs, where the model with the highest deviance explained was selected. Note that the covariates were detailed in Table 3.

Model #	Model name	Description	Deviance explained
1	Dsm.xy	Bivariate smooth of location, quasi-Poisson	23.6
2	Dsm.xy.tw	Bivariate smooth of location, Tweedie	26.4
3	Dsm.xy.twL	Bivariate smooth of location, Tweedie, smooth of LakeDist	26.7
4	Dsm.xy.twR	Bivariate smooth of location, Tweedie, smooth of RivDist	26.3
5	Dsm.xy.twX	Bivariate smooth of location, Tweedie, smooth of XingDist	26.4
6	Dsm.xy.twS	Bivariate smooth of location, Tweedie, smooth of Slope	27.3
7	Dsm.xy.twT	Bivariate smooth of location, Tweedie, smooth of TRI	27.2
8	Dsm.xy.twV	Bivariate smooth of location, Tweedie, smooth of NDTI	26.7
9	Dsm.xy.twND	Bivariate smooth of location, Tweedie, smooth of NDVI	27.8
10	Dsm.xy.twNDS	Bivariate smooth of location, Tweedie, smooth of NDVI+Slope	28.6

Abundance estimation and variance estimation

The best performing model, Model 10, was used to predict the abundance for Victoria Island and the mainland. Having made a prediction for each cell (9 km^2), the caribou abundance and respective coefficient of variation were both mapped (Figure 14). There were two main locations where the abundance was high. One of them was the north coast of the Dolphin and Union Strait around Read Island and the second location was between Richardson Island and Cape Peel 39 km inland from the coast (Figure 14). The DSM model resulted in an abundance estimate of 5,229 caribou (95% Cl = 3,985–6,473, CV = 0.12), which was not statistically different from the 5,363 caribou (95% Cl = 4,061-7,081, CV = 0.14) estimate for Victoria Island and the mainland using MRDS ($t' = 0.14 < t_{0.05, 287} = 1.96$). As DSM was more precise than the MRDS estimate and accounted for caribou aggregation in the study area, the use of the DSM abundance estimates as the 2023 Dolphin and Union caribou herd estimate for Victoria Island and the mainland was strongly justified.

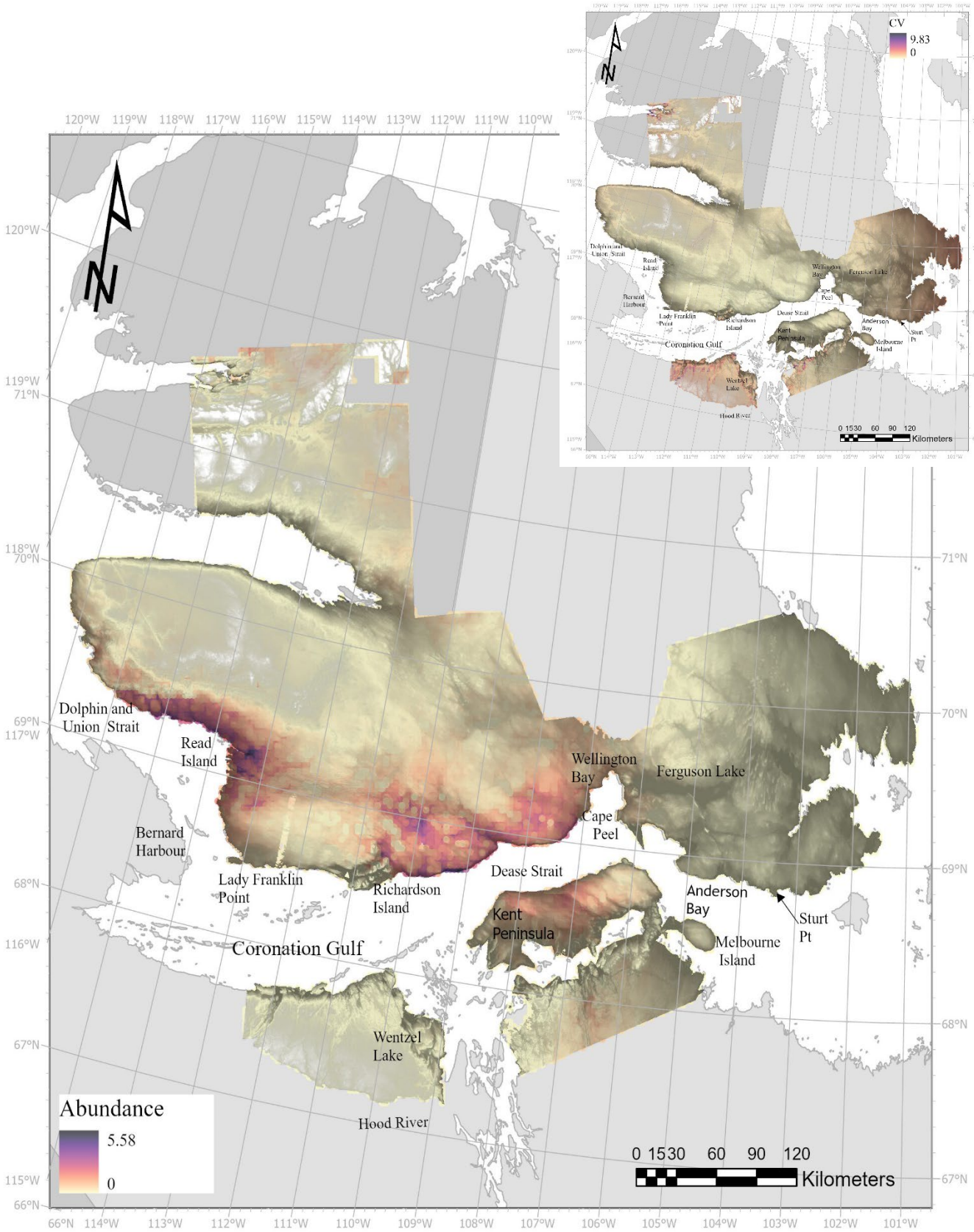


Figure 14: Predicted density surface with NDVI and Slope as bivariate smooth for *x* and *y* location as predictor. The resulting abundance of caribou is shown within each predicted grid cell of size 9 km^2. The smaller map shows the coefficient of variation per predicted grid cell over the study area to inform areas of higher uncertainty.

Population Changes and Overall Trend

Table 8 summarized the estimates resulting from the Dolphin and Union caribou herd monitoring program conducted between 1997 and 2023. Comparison between the 2020 and 2023 estimates was conducted for both Victoria Island and the Victoria Island and mainland. The results were compared using a two-way t-test ($\alpha = 0.05$) to determine if the estimates were statistically significantly different.

The number of caribou on Victoria Island increased from 3,579 in 2020 to 4,589 in 2023, but this difference was not significant ($t' = 1.19 < t_{0.05, 246} = 1.96$). Even when the 2018 and 2023 estimates were compared, there was no statistical difference detected between estimates ($t' = 0.49 < t_{0.05, 216} = 1.96$) upholding status quo. For Victoria Island and mainland, the 2023 estimate was not different from the 2020 estimate ($t' = 1.73 < t_{0.05, 308} = 1.96$) (Table 8).

The overall finite rate of change between survey year (lambda (λ)) was calculated as the ratio of the population size in year plus 1 to the population size (Gasaway et al., 1986). Since the time interval is over two years, a constant rate is assumed between this interval. Thus, the estimated finite rate of change was below 1 before 2020, and was higher than 1 in 2023, showing positive growth (Table 8). The overall finite rate of change (lambda (λ)) were provided (Table 8).

Table 8: Estimates of the Dolphin Union herd from 1997-2023 for Victoria Island (VI). The 2020 and 2023 estimates for the Dolphin Union for the Victoria Island and mainland (VI/Mainland) are in two separate rows.

Year	N	SE	Conf. Limit		CV	df	t-test	λ
1997 (VI)	34,558	4283.0	27,757	41,359	0.12			
2007 (VI)	27,787	3613.0	20,250	35,324	0.13	21	-1.21	0.80
2015 (VI)	18,413	3133.8	11,644	25,182	0.17	55	-1.96	0.66
2018 (VI)	4,105	694.8	2,931	5,750	0.17	54	-4.46	0.22
2020 (VI)	3,579	476.5	2,758	4,644	0.13	379	-0.62	0.87
2023 (VI)	4,589	705	3,391	6,211	0.15	246	1.19	1.28
2020(VI/Mainland)	3,815	513.7	2,930	4,966	0.13			
2023 (VI/Mainland)	5,229	635.0	3,985	6,473	0.12	308	1.73	1.37

Figure 15 shows the Dolphin and Union caribou herd estimates between 1997 and 2023 on Victoria Island, with the latest 2023 survey resulted in an estimate of 4,589 caribou (95% Cl = 3,391-6,210, CV = 0.15). The survey frequency between surveys was wider before 2015 until a declining trend was confirmed, and the monitoring increased to every 2-3 years. Between 2018 to 2023, the herd abundance was low and the abundance estimate remained at best consistent with no further decline recorded (Figure 15). Since 2020, a portion of the Canadian mainland and the Kent Peninsula are included in the Dolphin and Union study area to derive the herd estimate (Figure 15). Consistently with Campbell et al., (2021), an abundance estimate for Victoria Island and the mainland was provided, resulting in 5,229 caribou (95% Cl = 3,985–6,473, CV = 0.12). The estimates for Victoria Island and the mainland are higher than if Victoria Island only is considered. Although, for consistency with 2020, this report provided the final herd estimate for Victoria Island and the mainland, caution with this estimate should applied because: this area of the Canadian mainland is known to potentially overlap with other barren-ground herds, the lack of sea ice prohibited the caribou to start their crossing during the survey, and no collared caribou were trapped at this location the previous spring.

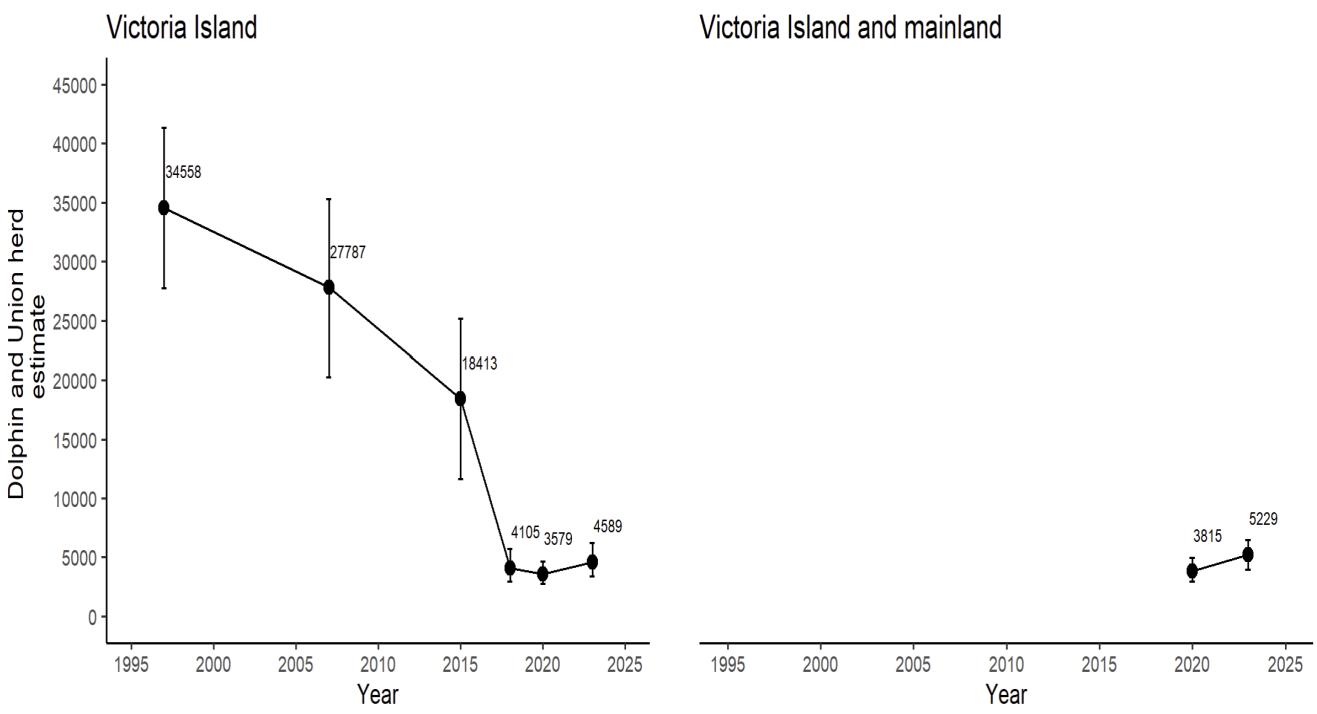


Figure 15: Left: Estimates for the Dolphin Union caribou herd on Victoria Island from 1997 to 2023. Right: Population estimates for the Dolphin Union caribou herd on Victoria Island and mainland for 2020 and 2023.

DISCUSSION

The same monitoring effort as the 2020 survey was used to generate an abundance estimate in 2023, as the same area was surveyed with minor modifications (Hauser et al., 2006). Prior to 2015, groups with over 100 individuals were seen (Leclerc and Boulanger, 2018). In 2023, the mean group size was 4.85 (sd = 3.47), and no group larger than 17 was observed (Figure 8). The greater frequency of small group observed could be explained by the caribou having not reached the coastline, where groups combined to form larger ones along the shoreline. Since groups were small, although hard to detect, when detected observers could accurately count the number of caribou in each group (Heard 1985; Heard and Jackson 1990).

Typically, 60 to 80 observations are needed to fit a detection function (Buckland et al., 2005). The number of groups seen in each stratum was less than 60, even in the two high density strata B and G. All the observations in the study area resulted in 148 groups of caribou across all strata. This is less than in 2020 with 202 groups, but more than in 2018 with 91 groups (Campbell et al., 2021; Leclerc and Boulanger, 2020). To meet the main assumption of Distance Sampling, mandatory observers' training, recording the distance before caribou are disturbed were implemented (Buckland et al., 2001). It was possible to have good detection in the first bin even when using Twin Otter, which did not provide an optimal seating position and bubble windows. Most of the observations were made in the closest bins to the aircraft and declined gradually with distance (Figure 10). The shape of the histogram of detection was an improvement from the 2020 survey (Campbell et al., 2021).

Dolphin and Union caribou herd distribution in the study area was informed by the collars (n = 21) and caribou observation gathered during the survey. At the time of the survey, no collar started to resume their migration and venture outside of the study area since the sea ice did not form and the daily movement rate was below 5 km/day (Figure 5), providing confidence in the survey results, as a minimum movement within the study area occurred. Where no collars were observed, the aerial survey allowed us to detect additional caribou inland within the Strata D, C, A, and E, as well as on the Kent Peninsula (Stratum L), and a negligible number of caribou in Strata O and J (Figure 7). Collars and observations both confirmed caribou aggregated in two distinct clusters; one close to Read Island and the other north of Richardson Island, 39 km inland (Figure 7). The distribution of caribou in 2023 differed from what was recorded in previous coastal surveys, as then the majority of caribou were within 20 km inland from the coast (Stratum G). The unexpected distribution created additional caribou clustering in some Strata B, G, and F (Figure 7).

MRDS method assumed uniform distribution of animal, but this assumption was challenged with the 2023 aggregation of caribou within stratum. A model allowing for this consequential heterogeneity of the landscape, such inference between the habitat variation and abundance, became preferable. The DSM, which allows for integrating the roles of environmental factors in abundance estimate (Miller et al., 2013), was therefore used to provide a more precise and robust estimate of the Dolphin and Union caribou herd for Victoria Island and the mainland. In fact, this method estimates the relationship between the distribution of animals, as a function of environmental covariates while estimating abundance.

The 2023 Dolphin and Union survey effort was successful in generating an accurate abundance estimate for the herd. The analysis using DSM for Victoria Island and the mainland resulted in an estimate of 5,229 caribou (95% Cl = 3,985–6,473, CV = 0.12). DSM was a robust method to model estimate, especially because caribou are clustered in two main areas. The abundance was modestly explained (28.6), by two environmental characteristics: NDVI and slope (Winiarski et al., 2013). The Dolphin and Union caribou herd selected areas that had a gentle slope and where plant productivity was high during the growing season. In the future, there is still a need to better determine the habitat characteristics that contribute to the Dolphin and Union caribou herd distribution during the rut by exploring the inclusion of other covariates, such as snow depth. The coefficient of variation (CV) of the best DSM model (Model 10) was less than 0.20 (Pollock et al., 1990). Skalski et al., (2005) suggest that a coefficient of variation below 0.128 is accurate for management, while 0.255 is acceptable for rough management. The predictive distribution and abundance mapping (Figure 14) could better stratify the sampling and increase efficiency of future survey.

Having the inability to tell confidently other subspecies of caribou apart from the aircraft, it is possible that some other caribou subspecies were counted as Dolphin and Union caribou in areas of range overlap. Some strata overlapped with the distribution of Peary caribou on Victoria Island (Stratum D), while others on the Canadian mainland (i.e. Strata L and N) overlapped with tundra wintering herds. The rate of mixing at these locations of range overlap is currently unknown. It is more likely that including caribou belonging to other subspecies may have inflated the Dolphin and Union counts. This contributed to potentially overestimate the true Dolphin and Union caribou herd estimate. Therefore, careful consideration should be taken when making harvest recommendations based on the total estimate presented in this report for Victoria Island and the mainland. A more conservative study area might be warranted.

Although the past two survey results were for the Victoria Island and the mainland, the estimate of Dolphin and Union caribou only for Victoria Island goes back to 1997. The Dolphin and Union caribou herd experienced a slow decline from 1997 to 2015, which precipitated rapidly between 2015 and 2018 (Figure 15). The last three surveys conducted in 2018, 2020, and 2023 for Victoria Island only, showed that the herd has remained stable, as the estimates were not statistically significant between 2018-2020 (Campbell et al., 2021), and between 2020-2023 ($t' = 1.19 < t_{0.05, 246} = 1.96$) (Figure15). For the last two surveys, a larger study area was set. The increase in number from 3,815 (95% Cl = 2,930–4,966, CV=0.13) to 5,229 caribou (95% Cl = 3,985–6,473, CV = 0.12) is also not statistically significant ($t' = 1.73 < t_{0.05, 308} = 1.96$). Although the 2023 estimate was precise (CV < 0.20), a potential small true biological change in the Dolphin and Union caribou herd trend trajectory was not detected with confidence within the 3-year frequency interval between survey (Conroy et al., 2018). Inclusion of local knowledge and health monitoring information would provide complementary information to the Dolphin and Union caribou herd trend. In conclusion, either for Victoria Island or for Victoria Island and mainland, the estimated number of caribou was at best stable based on the 2023 results.

Before the next survey, it is recommended to determine the minimal change in abundance estimate (i.e., minimum biologically significant effect size) that would cause change in the management strategy of the Dolphin and Union herd. Statistical power analysis would be useful in the research planning effort to design a survey with the CV needed to detect a management threshold with more

confidence (Conroy et al. 2018). Such an effort should be strategically located where only the Dolphin and Union caribou are likely to be found in high density to calculate a reliable detection function. However, as community confidence in the survey design and result is paramount, there could be an interest to continue to fly a large survey area, such as the full range at the rut.

In conclusion, although the population estimate of the Dolphin and Union herd has been low since 2018, there have been no further declines reported. Meanwhile, the current estimate, 5,229 caribou (95% Cl = 3,985–6,473, CV = 0.12) for Victoria Island and the mainland, still felt below 8,000 caribou, which is the threshold used to determine the low point in their cycle (Worthington et al., 2018). Emphasis between survey years should be put forward to monitor the herd demographic indicators, such as reproduction rate, calf production, recruitment rate, and cow survival. Dolphin and Union caribou are adapting to unpredictable climatic conditions under a warming Arctic and the herd is still in a state of vulnerability. Preserving this herd is a critical conservation priority, as it is central to the preservation of Inuit subsistence and preservation of the traditional values of the communities in the western Kitikmeot and the eastern Inuvialuit communities.

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ᑐᑭᓯᐊᖅᑎᑦᑎᐅᔪᓐ ᓇᐃᔨᖅᓯᒪᔪᖅ

ᐅᓄᖅᓂᒃᑯᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᑦ ᖃᑭᖅᑕᓗᒍ ᑐᑦᑐᒃᑯᑦ

ᐅᑭᐊᐅᓴᖅᑎᑦ 2023

ᑲᓇᑕᐅᑉ ᐅᑭᐅᖅᑕᖅᑐᒃ ᑐᑦᑐᖏᖅᓯᐅᑉᐳᖅ, ᓄᖅᓗ ᖃᑭᖅᑕᓂ ᑐᑦᑐᐃᑦ (*Rangifer tarandus pearyi*), ᓇᓱᖅᑐᖏᖅᑐᒥᑦ (*Rangifer tarandus groenlandicus*), ᐊᓴᒪᓗ ᖃᑭᖅᑕᒍᓗᑉ ᑐᑦᑐᑦᑕ (*Rangifer tarandus groenlandicus x pearyi*). ᖃᑭᖅᑕᒍᓗᑉ ᑐᑦᑐᑦᓂᓂ ᓇᔪᖅᑕᐅᓴᖅᑐᑦ ᐅᑯᓇᓂ ᐃᖏᓗᑐᑎᑕᐊᑉ ᖃᑭᖅᑕᒍᓕᒫᓴᓴᓐ ᐊᓴᒪᓗ ᑲᓇᑕᐅᑉ ᓄᓇᖅᓱᓄᐊᓂ ᐅᐊᖏᓇᖏᓐ.

ᑲᑎᒪᔨᕿᓂᑦ ᐊᑕᑦᓇᖅᑐᒥᑦᑐᓂᑦ ᐄᒪᔪᓂᑦ ᑲᓇᑕᒥ (COSEWIC) ᖃᐅᔨᓴᓪᓕᐅᖅᑐᑦ ᖃᑭᖅᑕᒍᓗᑉ ᑐᑦᑐᖏᓂᓂ ᓄᖓᑐᐃᓇᖏᖅᑕᓂᕆᔨᕐᓂᕆᐃᔪᑦ. ᑭᖓᓂᐊᒍᓕᓐᓂ, 2022-ᒥ, ᓄᓇᕗᓕᒥ ᐄᒪᔪᓕᕐᓗᑎᐅᓴᖅᑎᑦ ᑲᑎᒪᔨᖏᑦ ᑲᓇᐃᖏᓐᓂᒃᑯᓐᓂ ᒥᓂᑦᑕᖓ ᐊᐃᑉᐸᖅᑕᐅᓴᖅᑐᓂᒃ ᑎᑎᕋᖅᑕᐅᔪᓴᓕᐅᖅᓯᒪᓂᒃ ᐊᐃᔨᓴᖏᑦᑐᑦ ᐊᑦᑕᓇᖅᑐᖅᓯᓕᐅᖅᓯᒪᔪᖅ.

ᑎᑎᕋᐅᓴᖅᑎᑦ ᐅᓄᖅᓂᒃᑯᑦ ᖃᐅᔨᓴᖅᑕᐅᓴᐅᖅᑐᑦ 2018-ᒥ ᐅᓄᕐᓂᖅᓯᒪᔪᓕᕐᓂᖏᓐᓂᒃ ᑕᑯᓴᐅᖅᑐᑦ 4,105-ᓯᓚᓄᑦ ᑐᑦᑐᓄᑦ ᐃᖃᓗᐃᑦᑎᑕᐊᑉ ᖃᑭᖅᑕᖏᓐᓂ. ᑭᓯᐊᓂ, ᐅᓄᖅᓂᒃᑯᑦ ᐊᓯᔾᔨᓗᐊᓴᐅᖅᓯᒪᔪᑦᑐᑦ 2020-ᒥ, ᖃᐅᔨᓴᖅᑕᐅᕆᐹᓂᓴᐅᖅᑐᑦ ᓴᓴᑕᐅᖅᑐᖅᓯᓗᑎᒃᓗ 3,815-ᖓᓴᓯᕐᔭᐅᓗᑎᒃ ᑐᑦᑐᐃᑦ, ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᐃᖃᓗᐃᑦᑎᑕᐊᑉ ᖃᑭᖅᑕᖏᓐᓄᓂᑦᑐᑦ ᓄᓇᐃᓐᓇᕐᑎᑦᑐᓪᓗ.

ᐅᔪᐊ ᒪᑦᑐᒃ ᑐᖃᓐᓇᔭᐅᓪᓗᐊᑕᓴᐅᖅᑐᑦ ᓴᓇᐃᑎᓪᓗᒋᑦ: 1) ᐅᓄᖅᓂᒃᑯᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ ᓄᖅᑭᑎᓪᓗᒋᑦ ᖃᑭᖅᑕᒍᓗᑉ ᑐᑦᑐᖏᓂᓂ ᐊᓯᔾᔨᓗ 2) ᖃᓄᐃᓪᓚᕐᓂᖓᖏᓂᑦ ᖃᐅᔨᓇᓯᓗᒋᑦ ᖃᑭᖅᑕᒍᓗᑉ ᑐᑦᑐᖏᓂᓂ.

ᖃᐅᔨᓴᕿᑎᓕᓴᓂᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓴᐅᖅᑐᑦ 4,589-ᖓᓯᓕᐅᖅᑐᑦ ᐃᖃᓗᐃᑦᑎᑕᐊᑉ ᖃᑭᖅᑕᖏᓐᓂ ᐊᓴᒪᓗ 5,229 ᑐᑦᑐᐃᑦ ᐃᖃᓗᐃᑦᑎᑕᐊᑉ ᖃᑭᖅᑕᖏᓐᓂᑦ ᓄᓇᐃᓐᓇᕐᑎᑦ. ᐊᒥᓯᓂᖏᑦ ᖃᓄᐃᓪᓚᕐᓂᖓᖏᓂᑦ ᓯᕆᕐᓂᖅᓇᓯᑎᐅᓇᓴᖅᑐᑦ 2023-ᒥ ᐊᒥᓯᓂᖏᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓴᐅᖅᑐᑦ ᑕᐅᑐᓴᓯᑦ ᒪᑦᑐᒃᓄᑦ ᖃᐅᔨᓴᖅᑕᐅᓴᐅᖅᓯᒪᓕᒃᓄᑦ.

ᖃᑭᖅᑕᒍᓗᑉ ᑐᑦᑐᖏᓂᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓴᑦ 5,229-ᖓᓇᓯᑎᐅᓴᑦ ᑐᑦᑐᐃᑦ, ᓯᓚ ᐊᑦᑎᓗᐊᕐᓯᑎᐅᓴᐅᖅᑐᑦ (8,000

ↄᓱᓚᓂ). ᑦᑲᐸᐊ ↄᑲↄᐃᑦ ᐱᓴᒪᐅᐳᓂᓕᓵᓄᑦ ᐃᓄᓵᓄᑦ ᐊᐢᓇᐅᒪᔨᔾᑕᐳᑦᓗᑎᑲ ᐃᓕᖅᐸᓱᓚᓄᑦᓗ, ᐊᐅᓪᑎᔨᑕᐳᔾᑦ ᐃᑲᔨᖅᑐᐃᓕᓵᓇᐊᖅᖅᓴᑦ ᐅᑎᕋᑦᓕᐊᓂᓕᓵᓂᑦ.

Kugluktuk Angoniatit Association ● Hunters' & Trappers' Organization
PO Box 309, Kugluktuk NU X0B 0E0 ● Phone: (867) 982-4908 ● Email: kugluktuk@krwb.ca

PITJUTAOYOK: Kugluktuk Angoniatit Katimayiit (HTO-kut) titigakhimayot kiotjutigivlugit ukkua Nunavut Kavamatkut kiotjutigihimayaitnik tahapkununga umingmait munagitjutaitnik mighagot tahamani nunaoyami MX-07-mi.

HIVUNIGHITJUTAT:

- Umingmait kangagalok nikkigiyaokataktot hapkua Kugluktukmiut Inuinaknin.
- Kugluktumiut anguniaktit atokataktait tahapkua pingahut aviktoghimaniit nunaoyami anguniakvigivlugit tahapkua: MX-11 kivatani Kugluktumi; MX-07 Kiilikinikmi MX-09, tatvalu una MX-11 tahapkua atogaoloaktot anguniaktiit anguniakvigivlutjuk.
- Anguniakviohimangitok tahamna aviktoghimaninga nunaoyamin MX-07-mi Kugluktukmiot anguniaktitnin tatvanganin 2019-min kinguaniitlo, ikittunuanik umingmakviohimayok
- Kugluktukmiut anguniaktit takkuhimaliktot ikkilivalianinganik tahapkua amigaitilangit umingmait tahamani (anniakutikakpaliktuniklo) taimatot ahiani aviktoghimayuni nunauyami angutiakpaliktot.
- Kugluktumi HTO-kut ikkayokhiotihimangitat ukkua Nunavut Kavamatkut parnaiyaotatnik naunaiyagomavlutjuk tahamna aviktokhimayomi nunaoyami MX-07.

UBLUMI KANOGININGANIK:

- Ukkua Kugluktumi HTO-kut katimakatigihimayat ukkua Nunavut Kavamatkut havaktiit tatvani June 18, 2025-mi ukkaotigivlugit tahapkua naunaiyaotaohimayot.
- Ukkaotikakhimayot tahapkuninga Kititaohimayot Anguniakutikhat Ataatait ikkilivaligomangmatjuk 400-mit talimanut ataatakhanik tamatkiklugo tahamna aviktoghimayomi nunaoyami.
- Ukkua Kugluktumi HTO-kut ikkayoghiotingitaat tahamna tatvalu uktokuyaat ukkua Nunavut Kavamatkut katimakatigitkovlugit nunaliitni nallaktitlugit.
- Ukkua Kugluktumi HTO-kut atokataktat una MX-11-mi nunaoyami hapkua nunaliitni anguniagangata Nunakatigit Nikkighakhiokhugit Avakuktakhanik.

UKTOKUYAOYUT:

- Ukkua Kugluktuk-mi HTO-kut pihimayomayat ukkua Kititaohimayot Anguniakutikhat Ataataghat aulaakuvlugit 400-nik ataataghat tatvalu tamaita hapkua anguniaktit katimayiiit ilikut angutighanik munagitjutighaitnik ihuaghaitkoyait tahapkuani aviktoghimayomi nunaoyami, taimatotaok inmingnik havagihimayaitnik.
- Ukkua Kugluktumi HTO-kut katimakatigiyomayait hapkua allaat Anguniaktit Katimayiit tatvalu Ikkayoktigikhutik Munaghiyioyot kiotjutighanik hivunighami tahapkua Nunavumi Anguhighalikiyiit Munaghiyiolutik Katimayiit katimakatigiyomavlugit ahivakulugit tahapkua Kititaohimayot Anguniakutikhat Ataatakhat umingmakhiokutit tatvani Aviktoghimayomi Nunaoyami MX-11-mi.

Titigakhimaya uma

Amanda Dumond, Ataniata

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ⵔⵙⵣⵏⵔ ⵉⵍⵔⵔⵓⵙⵏⵔⵔⵄⵄⵜⵔ ⵃⵎⵍⵏⵛⵔⵙⵔ

ⵉⵔⵍⵛⵍ

ⵊⵏⵛⵃⵔⵉⵔⵔⵏⵙⵔ: X **ⵄⵔⵍⵔⵔⵉⵔⵊⵔⵔⵜ:**

ⵏⵛⵔⵔⵉⵔⵔⵜⵙ: 2023-ⵔ ⵎⵛⵏⵔⵔⵛ ⵊⵛⵊⵛⵔ ⵔⵛⵏⵉⵔⵎⵔⵍⵔⵔ ⵄⵍⵍ ⵃⵎⵍⵏⵜⵙⵊⵔ ⵜⵃⵔⵍⵄⵔⵔⵛⵛⵔⵙ.

ⵏⵔⵔⵄⵛⵊⵔⵍⵔⵔ:

- ⵛⵜⵔⵔⵄ ⵊⵜⵊⵄⵔ ⵊⵜⵊⵔⵔⵉⵔⵔⵔⵍⵔⵔⵔⵛⵔ ⵉⵔⵔⵔⵍⵛⵜⵔ ⵔⵔⵔⵍⵛⵔ ⵄⵔⵍⵔⵔⵛⵔ ⵄⵛⵔⵔⵜⵔ ⵄⵛⵔⵔⵜⵔ ⵄⵛⵔⵔⵜⵔ, ⵔⵛⵔⵔⵔⵔⵊⵔⵔⵜⵙ, ⵔⵔⵔⵍⵍ ⵛⵃⵍⵔⵔ'ⵊⵔⵔⵛ, ⵛⵜⵔⵔⵔⵛⵔⵔⵔⵔⵔⵔ ⵔⵔⵔⵍⵛⵔ ⵔⵔⵔⵍⵛⵔ ⵔⵔⵔⵍⵛ

ᑐᓴᑐ (95% CI = 124,704-179,558; CV = 8.5%). ᓇᓕᐅᑦᑕᖅᑕᐅᔪᖅ ᐊᒥᓱᓂᒃ ᐅᓄᖅᓯᐅᐊᖅᓂᕆᐊᔾᔨᖅ ᑎᓴᓕᐅᑦ ᐊᖅᓇᓪᓗᖅᓯᑎᕐᓂᒃ.

- ᐋᓂ 2023–ᒥ ᐊᑕᖏᖅᓗᓂᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᑦ ᐅᓄᖅᓂᖅᓴᓪᓚᐅᓕᐅᖅᓴᑦ ᑲᑎᖅᓗᓂᑦ ᑐᓴᑐᐃᐅᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓯᒪᔪᓂᑦ 2018–ᖃᓪᓗᓂ 103,372 (95% CI = 93,684-114,061; CV = 4.9%) ᑎᑎᕋᖅᐸᓪᓕᐊᓪᓗᓂ ᐊᖅᑭᒍᑕᒪᑦ 8% ᐅᓄᖅᓯᐅᐊᖅᑐᑦ ᑕᒪᐃᓐᓂᐅ ᐊᖅᓇᖅᓗᐃᑦ ᐊᒻᒪ ᑕᒪᖏᑲ ᑐᓴᑐᐃᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᑦ. ᐅᓄᖅᓯᓭᓪᓕᖅᓯᒪᔪᑦ 2018–ᒥ 2023–ᒧᑦ ᐅᑎᕆᐊᖅᓴᑦ ᐊᖅᑭᒍᑕᒪᑦ 4-5%–ᒥ ᐅᓄᑭᓐᓂᖅᐸᓪᓕᖅᓗᓂᒃ 2018–ᒥ ᐊᒻᒪ 2011–ᒥ ᑕᒪᐃᓐᓄᑦ ᑐᓴᑐᓄᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ ᒪᓕᒃᓗᓂᑦ 136,608 ᐊᒻᒪ 103,372, ᐊᑐᓂ.

- ᐃᖏᓂᐅᑎᐊᖏᓂᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓕᐅᖅᓴᑦ ᑎᓴᓕᐅᖅᑐᒧᑦ ᖃᓄᐃᓕᓪᓕᐊᓂᖓᓐᓂᒃ ᐅᓄᖅᓯᐅᐊᑭᑕᐅᓪᓚᐊᑕᓕᐅᖅᓴᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᑦ ᖏᓂᖅᓯᓕᕐᓂᖅᓯᒪᔪᑦ ᐊᓐᓇᐅᓯᒪᔪᑦ ᑕᐅᑐᑦᓗᓂᑦ ᐃᓐᓇᕐᓕᐅᖅᓴᑦ.

- ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᐊᖏᓇᓪᓚᐃᑦ ᖏᓂᖅᓯᑭᖅᓯᒪᔪᑦ ᓇᓂᓂᓂᑦ ᑕᔪᑎᑦᑎᐅᖅ ᑐᓴᑐᐃᑦ ᐊᐅᓕᓂᓂᑦ ᖇᖏᑲᐅᑦ ᑐᓴᑐᓂᓐᓂ ᑎᓴᓕᐅᑦ ᓄᖏᐅᖏᓪᓗᓂᓕ ᐊᒻᒪᓗ ᓇᒧᓐᓇᑭᖅᑐᑐᑦ ᑎᓴᓕᐅᑦ ᑐᓴᑐᓂᑦ ᐊᒻᒪ ᐊᕼᐃᐊᖏᑎᐅᑦ ᐅᓴᑐᓂᑦᑕᑦ ᓄᖏᐅᖏᓪᓗᓂᓂᓐᓂᒃ. ᐃᓕᓴᐅᑎᓯᒪᔪᖅ ᐅᓄᖅᑎᑎᓕᓂᓂᓂᓐᓂᒃ ᐊᑐᐃᓚᖅᓴᖅ (IPM) ᓇᓗᓇᐃᓚᐅᖅᓯᒃ ᑎᓴᓕᐅᑦ ᐅᓴᑐᓂᑦᑕᒪ ᐃᓂᑦᕋᓂᖅᑐᑦ 2023–ᒥᑦ ᐊᒥᓱᓕᐅᖅᓂᕐᓂᓂᓕᓂᑦ.

- ᑲᑎᓴᓕᑦ ᐊᕼᐃᐊᐅᒃ ᐊᒻᒪ ᑎᓴᓕᐅᑦ ᓇᓂᓂᓂᑦ ᓇᓕᒧᑉᐸᓗᓂᓂᖅᓯᐊᖅᑐᒃ ᐊᐅᓕᓂᓂᓂᓐᓂᒃ ᐃᓐᓇᐃᓐᓇᖅᑐᑦ ᐊᑐᖅᑕᒥᑦᓗᓂᓂᓐᓂᒃ ᑎᓴᓕᐅᑦ ᑐᓴᑐᓂᑦ ᐊᒻᒪ ᐊᕼᐃᐊᐅᖏᓪᓗᓂᑦ ᑐᓴᑐᓂᑦᑕᒪ ᐃᓂᑦᕋᓂᖅᑐᑦ.

- ᓇᓂᓂᓂᑦ ᖃᓄᐃᓕᓴᓪᓕᐊᓂᖓᓐᓂᓪᓗᑦ ᐊᑐᖅᑕᐅᓕᐅᖅᓴᑦ, ᓇᓕᐅᑦᑕᖅᓯᒪᔪᑦ ᐃᓕᓴᐅᑎᓯᒪᔪᑦ ᐅᓄᖅᓯᓂᓂᑦᑕᒪ ᖃᐅᔨᓴᖏᑎᓯᓂᑦ ᖃᐅᔨᓴᓪᓚᐅᑎᓯᐊᖅᑐᓂᒃ ᑭᐅᓂᑦᑕᖅᓯᒪᔪᑦ 2023–ᒥ ᐊᖅᑭᒍᑕᒪᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᑦ ᓇᓗᓇᐃᖅᓯᖅᑯᓯᔾᔨᑦ ᓇᓂᓂᓂᑦ ᐊᒻᒪ ᖃᐅᔨᓴᓂᓂᑦ ᓇᓗᓇᐃᖅᓯᓯᒪᓕᑎᐊᖅᓴᑦ ᐊᖏᓪᓕᓯᒪᓂᖓᓐᓂᒃ 2023–ᒥ ᐅᓄᖅᓂᖏᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ ᒪᓕᒃᓗᓂᑦ 2018–ᒥ ᐅᓄᖅᓂᖏᑦ ᓇᓕᐅᑦᑕᖅᑕᐅᓯᒪᔪᑦ.

ᑕᕐᓴᐊᖅᑐᖃᖅᓂᖅ:

- ᓯᑲᓪᓕᖅᒥ ᑕᕐᓴᐊᖅᑐᖃᓕᐅᖅᓴᑦ ᖃᑎᖅᑎᐅᑦ ᑭᓴᓪᓕᖅᑎᐅᓪᓗ ᐅᒪᔨᓕᓚᐅᓵᕆᔨᓂᓂᑦ 2023 ᐱᑎᓕᓵᕐᓂᖅᑎᓪᓗᒍᑦ ᓇᓗᓇᐃᖅᓴᓂᑎᓪᓗ ᐃᑲᔪᖅᑕᖅᓂᑦᑕᐅᖅᓴᖅᑎᓯᓂᒃ ᖃᐅᔨᓴᖅᓂᐅᔪᑦ ᐃᑲᔪᖅᑕᖅᓂᑦᑕᐅᓪᓚᐅᖅᓴᖅᑎᓯᓂᒃ ᑕᑐᓂᒃ ᓇᐅᑦᑕᖅᑐᑦ ᐱᓕᕆᔨᖅᑕᐅᓂᖅ, ᑲᒪᒋᔭᖅ, ᑕᓗᖅᓴᐃᓐᓇᖅ, ᑲᑎᖅᑯᓂᖅ, ᖃᓄᖅᑲᐅᑎᖅ ᐊᒻᒪᓗ ᐃᓚᖃᑦᑕᕐᓂᖅ ᐊᑐᖅᑕᒥᑦᓗᓂᑦ ᐃᑲᔪᖅᑕᖅᑕᐅᓯᒪᔪᑦ. ᐃᓚᖃᑦᑕᕐᓂᖅ ᑕᓗᖅᓴᐃᓐᓇᖅᑎᑦ, ᐃᖃᓗᐃᑦ ᑕᓗᖅᓴᐃᓐᓇᖅ, ᑲᑎᖅᑯᓂᖅ, ᖃᓄᖅᑲᐅᑎᖅ ᐊᒻᒪᓗ ᐃᓚᖃᑦᑕᕐᓂᖅ ᐊᑐᖅᑕᒥᑦᓗᓂᑦ ᐃᑲᔪᖅᑕᖅᑕᐅᓯᒪᔪᑦ ᐱᓕᕆᔨᖅᑕᐅᓂᖅ ᐃᓚᖃᑦᑕᕐᓂᖅ ᑕᓗᖅᓴᐃᓐᓇᖅᑎᑦ, ᐃᖃᓗᐃᑦ ᐊᒻᒪᓗ ᓴᓱᑦᑕᖅᑐᑦ ᐊᑐᖅᑕᒥᑦᓗᓂᑦ ᐃᑲᔪᖅᑕᖅᑕᐅᓯᒪᔪᑦ ᐱᓕᕆᔨᖅᑕᐅᓂᖅᓯᒪᔪᑦ. ᖃᐅᔨᓴᖅᓂᖅᓯᒪᓕᖅᑎᓪᓗᒍᑦ ᑕᓴᓴᐃᓚᖅᑕᐅᓪᓚᐅᖅᓴᖅᑎᓯᓂᒃ ᖃᑎᖅᑎᐅᑦ ᐅᒪᔨᓕᓚᐅᓵᕆᔨᓂᓂᑦ ᐊᐅᓚᑦᑎᐊᕐᓂᖅᓯᒪᔪᑦ ᐅᓂᒃᑲᓪᓗᓗ ᑕᓂᓯᐅᓕᐅᖅᓴᓂᓂᑦ ᐊᐅᓕᑦᑎᐊᕐᓂᖅᓯᒪᔪᑦ ᐅᓄᖅᓯᐅᐊᑭᑕᐅᓂᖅᓴᓂᓕ Ḷᓴᒃ 31, 2025–ᖃᐅᑎᓪᓗᒍᑦ.

ᐊᑐᓪᓚᕆᐊᖅᓴᑦ:

- ᐊᐸᑎᓕᓂᔨᐅᑦ ᖃᓄᑐᐃᓐᓇᖅ ᐊᓯᔾᔨᕐᑯᔨᖏᑦ ᐃᓐᓇᐅᔭᕐᓂᖅ ᐊᐅᓪᓚᑎᐅᔪᕐᓂ ᑎᑎᓕᐅᕐᓴᑦ ᑐᑐᑦᓂᓕᖅ.

Abundance and Trends of the Beverly Mainland Migratory Subpopulation of Barren-Ground Caribou (*Rangifer tarandus groenlandicus*) June 2011 – June 2023

FILE REPORT

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ABSTRACT

The Beverly barren-ground caribou herd migrates each spring from their winter ranges in northern Saskatchewan and southeastern Northwest Territories to their annual concentrated calving area (ACCA). The ACCA has historically been concentrated in the vicinity of Beverly Lake but in more recent years has shifted to the vicinity of the Queen Maud Gulf and Adelaide Peninsula, Nunavut. Abundance estimates developed over the last three decades suggested that the herd has significantly declined from an estimated 276,000 adults and yearlings in June 1994, to 136,608 by June 2011, and further to 103,372 by June 2018. Additionally, survey observations and collared cow telemetry locations have also documented a general shift in the calving distribution east from the Perry River area, more towards the Adelaide Peninsula's western and southern extents. These continued declines and shifts in ACCA extents provided the impetus for an updated estimate of the Beverly subpopulations abundance in June 2023. In June 2023, we estimated the abundance of the Beverly tundra wintering barren-ground caribou herd using both video and visual methods, and based on the estimated numbers of breeding and non-breeding female barren-ground caribou within the herd's ACCA extending from the Queen Maud Gulf coastline to the eastern shores of Chantrey Inlet. We further examined demographic trends for the Beverly herd using an Integrated Population Model (IPM).

Similar to the June 2011 and 2018 surveys, the June 2023 survey effort was deployed in five main stages including a collar reconnaissance, reconnaissance survey, abundance survey, calving ground composition survey, and fall composition survey. Unlike the 2011 and 2018 surveys, we also used a video survey modification to the visual method to address significant clumping of calving caribou not previously

encountered. We used a systematic aerial transect visual survey technique for reconnaissance surveys to inform the stratification of the survey area by caribou density. Following reconnaissance, we flew a combination of a stratified systematic aerial transect visual and video surveys to estimate the number of adult and yearling female and breeding female caribou within the Beverly ACCA. Our survey protocol employed a dependent double observer pair method, and a novel video method used to record high density clumps of calving caribou for later assessment using newly developed digital counting techniques. Survey effort focused on estimating the number of adult female caribou during peak calving. Additionally, we conducted composition surveys within all abundance survey strata to estimate the proportion of breeding and non-breeding females in each stratum. To obtain estimates of females, breeding females, males, and overall adult and yearling caribou within the survey area, the estimated number of adult caribou (1+ year old) for each survey stratum was multiplied by the sex and age class proportions of that stratum using composition survey results. Finally, whole herd estimates were extrapolated using sex ratios quantified during fall composition studies.

The June 2023 abundance survey yielded a breeding female estimate of 69,900 (95% CI = 59,135-82,837; CV = 9.2%) and a total adult female estimate of 88,236, (95% CI = 75,564-103,033; CV = 8.3%). The resulting estimate documents a significant increase in Beverly caribou breeding female and female abundance and reversal in the previously observed declining trend documented during June 2011 survey estimating 67,414 (95% CI = 61,257-74,190; CV = 4.8%) breeding females and 80,705 (95% CI = 73,636-88,452; CV = 4.6%) females, and June 2018 survey estimating 48,977 (95% CI = 44,056-54,448; CV = 5.3%) breeding females and 61,070 (95% CI = 55,583-67,099; CV = 4.7%) females.

The 2023 extrapolated whole herd estimate for the Beverly herd was 152,131 (95% CI = 124,704-179,558; CV = 8.5%) adults. This estimate was significantly larger than the 2018 whole herd estimate of 103,372 (95% CI = 93,684-114,061; CV = 4.9%) documenting a yearly 8% increase for both female and whole herd estimates. The

increase from 2018 to 2023 reverses a yearly 4-5% decline based on the 2018 and 2011 whole herd estimates of 136,608 and 103,372, respectively.

Analysis of cow collar locations document directional movement of caribou from the Bathurst herd to the Beverly and multi-directional movement between the Beverly herd and Ahiak herds calving ground. An integrated population model (IPM), which considers all data sources (collars, calving ground surveys, and compositions surveys) was applied to explore demographic factors as well as the relative influence on movements, herd status, and trend. Combinations of Ahiak and Beverly movements suggest that movements were relatively balanced between the two calving ground areas with only a slight net movement into the Beverly calving-ground. However, the ability of the IPM to detect annual variation in movements was limited due to sparse data. Bathurst directional movement into the Beverly did occur in many years however the overall impact to Beverly herd size was relatively low given the reduced size of the Bathurst herd. High productivity levels were estimated for the Beverly which was one of the main drivers of demographic increase given that estimated collar-based survival levels were moderate. When movement and demography was modeled, estimates from the IPM were within the confidence limits of the 2023 adult female herd estimate suggesting that movement and demography could explain the increase in magnitude of the 2023 abundance estimate when compared to the 2018 abundance estimate.

We make recommendations for the enhancement of future calving ground surveys including the use of video assisted counts, focused IPM analyses that consider neighboring herds in the same analysis, and a hybrid distance sampling visual survey method to improve robustness of estimates to sight ability bias.

Key words: Calving ground visual survey, Caribou calving ground, Kitikmeot region, Double observer pair method, Barren-ground caribou, Beverly Subpopulation, Ahiak subpopulation, Northeast Mainland, Queen Maud Gulf, Adelaide Penninsula, Nunavut, *Rangifer tarandus groenlandicus*, abundance, population survey, decline, Integrated Population Model.

TABLE OF CONTENTS

ABSTRACT **3**

1.0 INTRODUCTION **9**

2.0 STUDY AREA **18**

2.1 QUEEN MAUD GULF LOWLAND ECOREGION **19**

2.2 CHANTREY INLET LOWLAND ECOREGION **20**

3.0 METHODS **25**

3.1 RECONNAISSANCE AND ABUNDANCE SURVEYS **26**

3.1.1 Double Observer Pair Method 27

3.1.2 Video Assisted Counts of High-Density Strata 28

3.1.3 Survey Timing 29

3.1.4 Reconnaissance Survey 29

3.1.5 Allocation of Effort 31

3.1.6 Abundance Survey 33

3.1.7 Composition Survey of Abundance Strata 33

3.2 SURVEY ANALYSIS **41**

3.2.1 Video Interpretation and Analysis 41

3.3 ANALYSIS OF ABUNDANCE **44**

3.3.1 Modelling of Visual Sighting Probability Variation 44

3.3.2 Estimates of Breeding Females, Adult Females, and Adults on the Calving-Ground 45

3.3.3 Analysis of Fall Composition Data 46

3.3.4 Estimation of Herd Size 47

3.4 COLLAR ANALYSIS 48

3.4.1 Movement Between Calving-Grounds 48

3.4.2 Survival Rate Analysis 48

3.5 INTEGRATED POPULATION MODEL ANALYSIS 49

3.6 AERIAL WILDLIFE SURVEY–OBSERVATION COLLECTOR (AWS-OC) .. 52

4.0 RESULTS & DISCUSSION 55

4.1 RECONNAISSANCE SURVEY 59

4.2 ALLOCATION OF EFFORT 60

4.3 ABUNDANCE SURVEY 66

4.3.1 Analysis of collar movements 66

4.3.2 Video Assisted Counts in High-Density Strata 72

4.3.3 Estimation of abundance using Jolly strip transect estimator with video-assisted counts 79

4.3.4 Double observer modelling of sightability 83

4.3.5 Composition survey results 91

4.3.6 Estimation of breeding females, adult females, and bulls 95

4.3.7 Extrapolated herd estimates 98

4.4 ABUNDANCE TRENDS 104

4.5 COLLAR ANALYSIS 109

4.5.1 Movements between calving grounds 109

4.6 COLLAR SURVIVAL ANALYSIS 120

4.7 INTEGRATED POPULATION MODEL - DEMOGRAPHIC ANALYSIS 124

4.7.1 Data sources 124
4.7.2 Model scenarios considered 128
4.7.3 Comparison of estimates from IPM model scenarios.....140

5.0 CONCLUSIONS..... - 142 -

5.1 GENERAL COMMENTS ..... - 142 -

5.2 HERD DEMOGRAPHY AND MOVEMENTS FROM OTHER HERDS ... - 143 -

5.3 SURVEY METHODOLOGY ..... - 146 -
5.3.1 Video approach to confront large group sizes in 2023..... - 146 -
5.3.2 Sightability issues in previous surveys..... - 146 -

6.0 RECOMMENDATIONS..... - 149 -

7.0 ACKNOWLEDGEMENTS.....151

8.0 LITERATURE CITED.....153

9.0 APPENDICES.....163

***9.1* APPENDIX A – INTEGRATED POPULATION MODEL DETAILS.....163**

***9.2* APPENDIX B-MULTI-STATE MODEL ESTIMATES OF MOVEMENT BETWEEN CALVING GROUNDS. 170**

***9.3* APPENDIX C-- USE OF A REDUCED DISTANCE SAMPLING AND DOUBLE OBSERVER METHODOLOGY TO IMPROVE ROBUSTNESS OF CALVING GROUND SURVEY ESTIMATES 171**

1.0 INTRODUCTION

Following the last glacial period, caribou (*Rangifer tarandus*) in North America recolonized their range from several refugia, resulting in the emergence of multiple ecotypes (Yannic et al., 2014). Although Inuit have relied on several caribou subpopulations and ecotypes for survival over centuries, the first written reference to barren-ground caribou was likely that of Martin Frobisher in 1576 (Banfield, 1951). Samuel Hearne recorded the earliest detailed account of migratory behavior, distribution and movements, and the use of caribou by subsistence harvesters, in 1795 (Banfield, 1951). Early reports and interviews with residents, however, yielded little insight into the dynamic nature and distributions of barren-ground caribou (*Rangifer tarandus groenlandicus*) subpopulations west of Hudson Bay **(Figure 1)**.

It wasn't until the mid-1900's that that an increase in demographic studies of the barren-ground caribou herds east of the Hudson Bay was realized (Calef, 1979). Eight major barren-ground caribou herds were identified within the then Northwest Territories (NWT), now the NWT and Nunavut (NU), during this period. Together these herds likely exceeded 600,000 caribou (Calef, 1979). The most prominent mainland migratory herds during this period included the Bathurst, Beverly, and Porcupine herds (then thought to be stable), and the Qamanirjuaq herd (then thought to be declining). It was during this period of heightened research that more poorly understood subpopulations including the Melville Peninsula, Wager Bay, Cape Bathurst, and Bluenose herds (then thought to be increasing) were being more thoroughly defined (Calef, 1979; Heard and Jackson, 1990; Thomas, 1969; Rippin, 1971; Moshenko, 1974; Gunn and Decker, 1982; Stephenson et al., 1984; Gunn, 1984; Heard, 1982; Gunn and Sutherland, 1997; Williams and Heard, 1990; Williams et al., 1989; Thomas and Kiliaan, 1985; Thomas and Barry, 1990).

Our study focuses on the Beverly subpopulation, which migrates annually into Nunavut from winter ranges in northern Saskatchewan and the southeastern Northwest Territories (**Figure 2**). Abundance estimates suggest that the herd has declined from an estimated 276,000 individuals in 1994 to approximately 103,372 adults and yearlings by 2018 (Campbell et al. 2019). Since 2011, reconnaissance surveys conducted in 2013, 2016, and 2018, have indicated further declines in relative densities as well as a general shift in the calving distribution east toward the Adelaide Peninsula.

The survey history of the Beverly herd has been irregular, and most recently complicated by apparent distributional shifts within the herds spring, calving, and post-calving seasonal range. For example, a June 2007 calving ground survey found too few breeding females on the "traditional" Beverly calving area near Beverly and Garry lakes (175 observed on transect; relative density of 0.40 caribou/km^2) to conduct a photo-survey (Johnson et al., 2008). In the following years, the Government of the Northwest Territories (GNWT) continued to observe lower densities of caribou during reconnaissance surveys flown over the same area in June 2008, 2009, and 2010 (90 - 100 caribou observed on transect in June 2010; relative density of 0.20 caribou/km^2, unpublished GNWT data). At the time, these results suggested a severe decline in the Beverly subpopulation. Despite all indications from reconnaissance surveys up to June 2010 suggesting a population crash with the threat of extirpation, local knowledge and an assessment of collar movements over the same period suggested another possible reason for the decline. Collar relocations suggested a shift in the Beverly annual concentrated calving area (ACCA) of some 200 to 250 km north of their previous ACCA (also known as their traditional calving grounds), to the western Queen Maud Gulf Lowlands (QMGL) and south and western extents of Adelaide Peninsula (Nagy et al. 2011; Campbell et al., 2012; Campbell et al., 2019). The knowledge of local hunters (Baker Lake, Gjoa Haven, and Kugaaruk Hunters and Trappers Organization [HTO] meetings and pers. comm.) agreed that the Beverly herd had been calving further north from 2007 to present. Still, competing views suggested that the primary mechanism was a major decline coupled with a distributional shift, ending with a switching to the QMGL

ACCA to maintain the advantages of gregarious calving (Gunn et al, 2010; Gunn et al. 2012, Adamczewski et al. 2015). Small sample sizes of collars deployed prior to 2002, and the lack of reproductive assessments associated with these initial captures, render a quantitative assessment of this period unreliable, making it difficult to conclude which mechanisms were responsible for the numbers observed on the traditional calving area prior to 2007. However, quantitative evidence from more recent telemetry and aerial survey work, combined with local knowledge, strongly support the theory of a distributional shift in the ACCA, and provides an explanation for the observed increases in breeding female and female abundance in the QMGL during calving. Though inconclusive, we believe that the movement northward from the southern calving area began much earlier than 2005. Reconnaissance data from June 2016 suggested that the shift was not temporary as observations continued to show no re-establishment of calving within the previously recognized ACCA near Beverly and Garry lakes.

In addition to monitoring movements of individuals from the surveyed herd, it is also important to consider the potential for movements of animals from neighboring herds into the Beverly survey study area during a survey effort. This is particularly true for surveys in the QMGL area, where historically other caribou subpopulations have also calved. The Bathurst herd has previously calved annually within the western extents of the current Beverly QMGL ACCA. Prior to the shift of the Bathurst calving area to the west of Bathurst Inlet (Williams and Heard, 1990; Sutherland and Gunn, 1996; Gunn et al, 2000), the Bathurst herd calved across an area west of the Perry River extending to the eastern shore of Bathurst Inlet (Gunn, 1996 Sutherland and Gunn, 1996). Bathurst adult cow telemetry data collected by the GNWT from 2007 to present has shown a small proportion of Bathurst collared cows returning to the east side of Bathurst Inlet and mixing with calving Beverly cows over most years. Furthermore, a small number of caribou from the Ahiak Herd (a tundra wintering caribou subpopulation which is part of the Northeast Mainland group including the Wager Bay and Lorillard herds) also calve in close proximity to the Beverly ACCA along its eastern extents in the vicinity of Adelaide Peninsula. Overall, however, analyses of collar movements suggest that in most years the majority of the Ahiak

herd tend to calve further to the east of Adelaide Peninsula (Sutherland and Gunn, 1996; Gunn et al., 2000; Gunn, 1996; Gunn et al, 2008; Campbell et al., 2019, Campbell et al., 2022). Nagy et al. (2011) and Nagy and Campbell (2012) delineated caribou subpopulations calving east of the Beverly subpopulation and within the eastern part of the QMGL. The analysis of Northeast Mainland (NEM) caribou telemetry suggests the main calving area for the Ahiak subpopulation extends from the Adelaide Peninsula to the west coast of Simpson Peninsula, with the majority of calving occurring east of Chantrey Inlet, suggesting considerable separation between the herds. Though variable from year to year, there has been some spatial overlap between the adjacent Beverly and Ahiak subpopulations as well as between the Beverly and Bathurst subpopulations during the calving season since 2017 (Campbell et al. 2019).

Calving ground aerial survey methods have been improving since the first barren-ground caribou surveys were flown in the mid to late 1960s. Early estimates often varied in reliability, making precise comparisons through time challenging. Photographic methods were first deployed for Beverly calving-ground abundance surveys in 1982, and were then used consistently thereafter (June 1984, 1988, 1993, and 1994). Photographic methods improved count accuracy and abundance estimate precision where high animal densities made accurate counts by observers difficult or unmanageable. We first deployed the dependent double-observer pair visual observation method for caribou in June 2011 to estimate the abundance of the Beverly herd following several years of reconnaissance surveys indicating consistent low relative densities of Beverly caribou on the calving grounds (Campbell et al. 2012; Campbell et al., 2019). This survey method has been incorporated into used in all Beverly caribou surveys (both reconnaissance and abundance) to present. Where densities permit, this method employs more local survey observers, is less complicated logistically, and improves precision by correcting visual counts for sightability biases thereby allowing efficient, unbiased estimates without the use of the photo plane. This visual method can effectively be used when densities of less than 15 to 20 caribou/km^2 are encountered (Cook and Jacobsen 1979, Buckland et al. 2010). When caribou densities fall within the suggested limits, the dependent

double-observer pair visual method has proven to be more cost effective than traditional photographic methods without compromising accuracy or precision. However, when caribou densities are in excess of 15 to 20 caribou/km^2, either through population growth or clumping, visual methods are more likely to be insufficient in terms of their ability to accurately and/or precisely estimate abundance within effected strata. This condition can be difficult to predict when planning an abundance survey as it can be influenced by a number factors including changes in productivity and associated abundance between surveys, changes in spring melt and runoff along migratory routes, and/or other environmental or anthropogenic changes or disturbances that can modify caribou movements and distributions. For this reason, communication with aerial photography charter companies is always developed and maintained during all planned Beverly calving-ground abundance surveys.

Reconnaissance survey observations flown by the GNWT between 2007 and 2010 all reported group sizes and associated densities below the 15-20 caribou/km^2 threshold with very rare exceptions, while Government of Nunavut (GN) reconnaissance surveys flown between June 2010 and 2018 showed similar densities with only rare groupings of greater than 15 to 20 caribou/km^2 observed. For this reason, and with all quantitative evidence confirming a statistically significant declining trend up to June 2018, the 2023 Beverly calving ground survey was planned using the double observer pair visual method with the request to have a photo plane standing by in case of any unexpected change that may drive general observational densities above the recommended threshold.

Our main objective for the June 2023 survey was to obtain an estimate of adult female Beverly caribou calving within the QMGL from the eastern shore of Kent Peninsula to the western shore of Chantrey Inlet and the Back River, including Adelaide Peninsula (**Figure 3**). We used retrospective analysis and published studies both prior to, and following the survey for the purposes of delineating subpopulations from the survey strata. The main contents of this report are the survey results and the environmental and logistic challenges that had to be

overcome to meet the assumptions and objectives of the survey method deployed. We emphasize that the main objective of this study is to provide an abundance estimate for the Beverly herd to add to the assessment of caribou subpopulations in the region, and to inform interjurisdictional and jurisdictional co-management efforts.

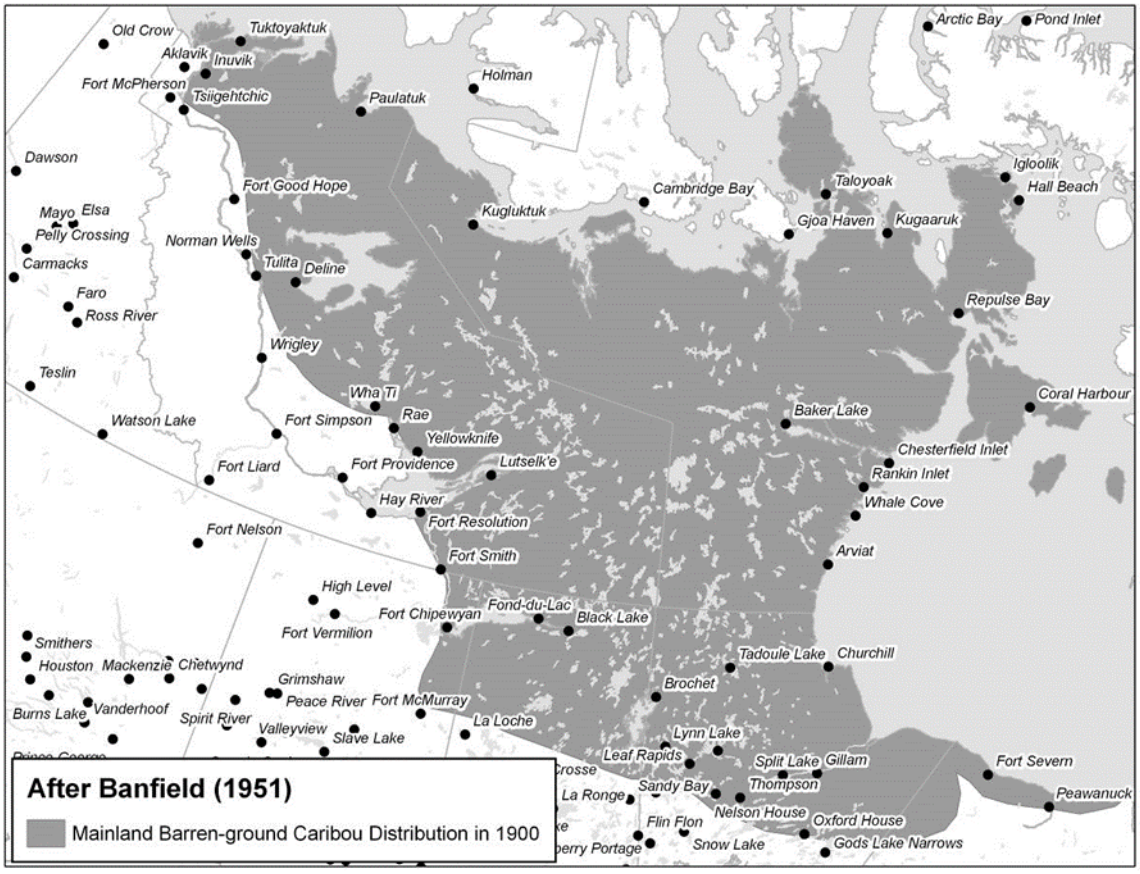


Figure 1. Mainland barren-ground caribou distribution based on local observations and studies from the early 1900s (after Banfield, 1951).

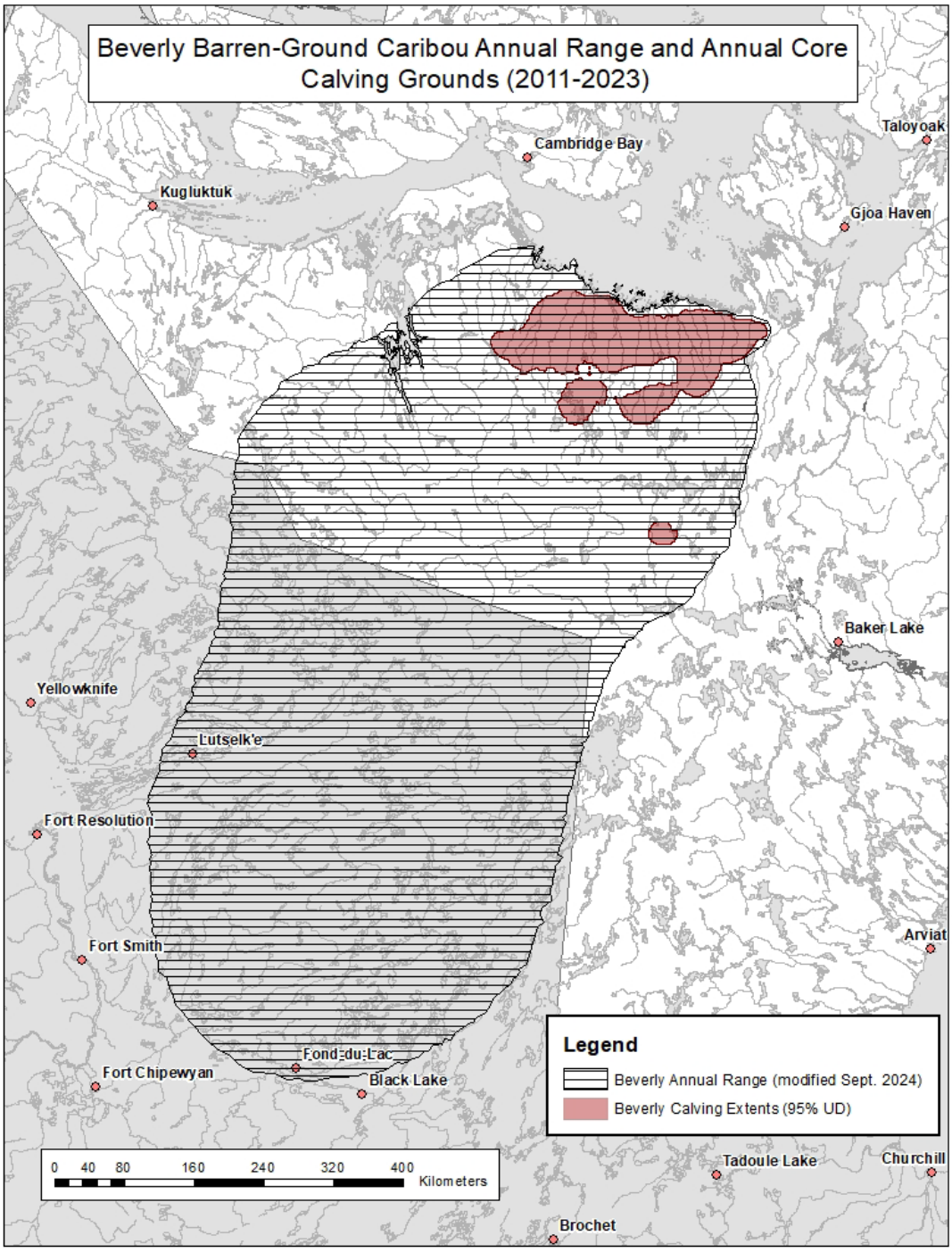


Figure 2. The annual range and concentrated calving grounds of the Beverly barren-ground caribou subpopulation. Calving polygons based on kernel analysis of all available telemetry data current to July 2023 and representing the 95% utilization distribution. Annual range extents modified from Nagy and Campbell (2012) to reflect calving data current to July 2023.

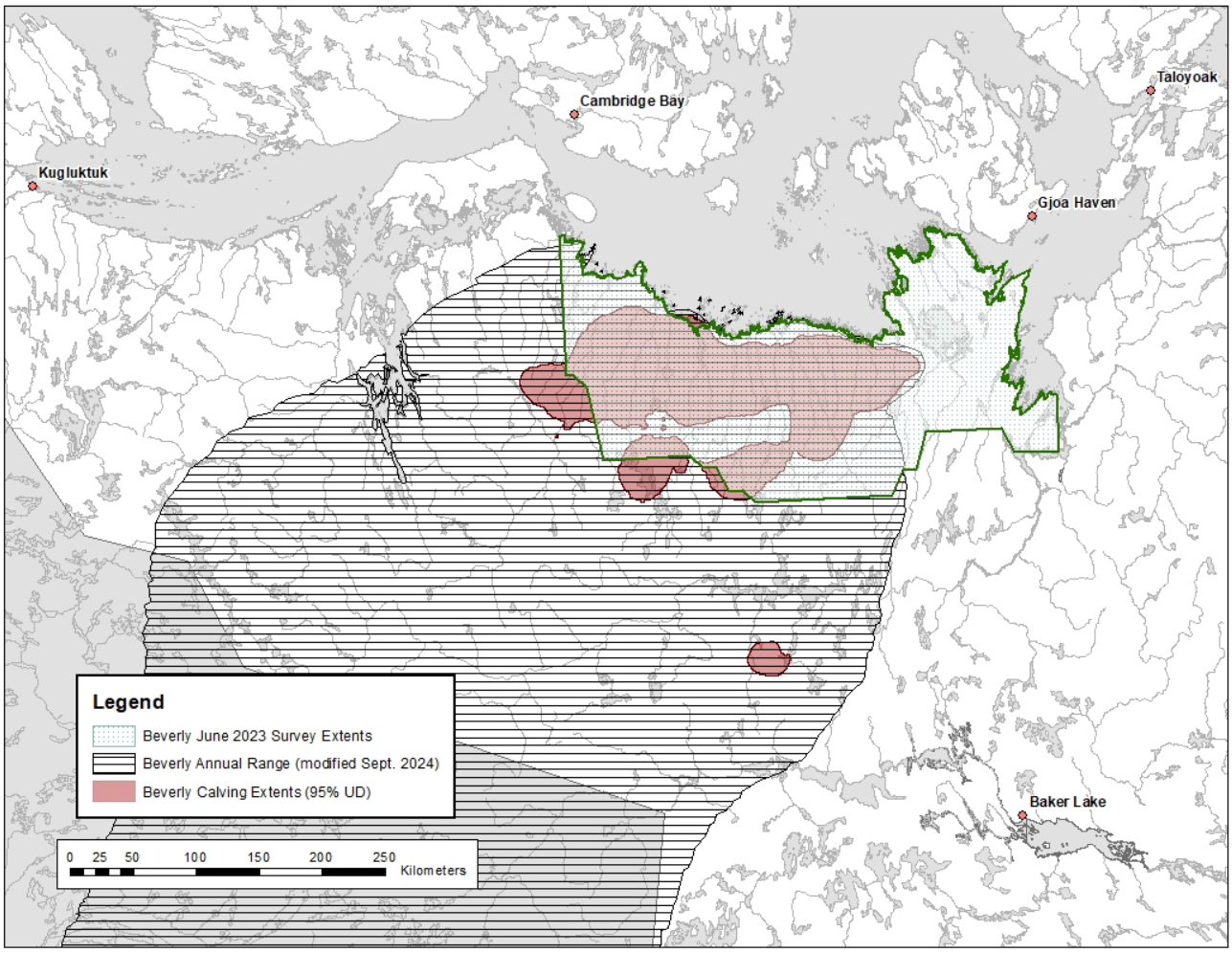


Figure 3. The June 2023 calving-ground abundance survey extents (Green) including annual core calving grounds (red) and annual range (Black). Calving polygons based on kernel analysis of all available telemetry data current to July 2023 and representing the 95% utilization distribution. Annual range extents modified from Nagy and Campbell (2012) to reflect calving data current to July 2023.

2.0 STUDY AREA

The estimated annual range of the Beverly herd, based on satellite-collar location data collected between 2000 and 2011, is approximately 426,160 km^2 (Nagy et al. 2011, Nagy 2011, Nagy and Campbell 2012, Campbell et al. 2014). The Beverly 2011 to 2023 annual concentrated calving area (ACCA), including both the Garry Lakes and Queen Maud Gulf (QMG) calving extents, was estimated using kernel analysis, and found to be 38,491 km^2. By way of comparison, the historical pre-2011 Beverly Lakes ACCA was an estimated 16,131 km^2 or 41% of post 2011 QMG extents (Nagy et al. 2011, Nagy and Campbell 2012, Campbell et al. 2014). The majority of the ACCA, fall and spring range, including the spring and fall migratory corridors, and the majority of the post-calving habitat, lie within Nunavut **(Table 1)**. The annual range of the Beverly subpopulation spans across Nunavut, Saskatchewan, and the NWT. The communities of Black Lake and Fond-du-Lac in Saskatchewan, Lutselk'e in the Northwest Territories, and Baker Lake, and Gjoa Haven, in Nunavut, within the Beverly subpopulation's annual range.

The June 2023 Beverly calving ground survey area covered an estimated 73,184 km^2. It extended south from the shores of the Queen Maud Gulf and northern and eastern shores of Adelaide Peninsula to a latitude of approximately 66.5˚N, and east from the Ellice River to the western shore of Chantrey Inlet and the Back River (Wiken, 1986). The Beverly subpopulation's annual range extends from the Southern Arctic Ecozone south through the Taiga Shield Ecozone (Wiken, 1986) crossing a total of nine Ecoregions including the Queen Maud Gulf Lowland, the Takijua Lake Upland, the Garry Lake Lowland, the Back River Plain, the Coppermine River Upland, the Dubawnt Lake Plain/Upland, the Kazan River Upland, the Tazin

Lake Upland and the Selwyn Lake Upland (Wiken, 1986; Ecological Stratification Working Group, 1996) (**Figure 4**).

The Beverly subpopulation's late-winter range lies predominantly within the Tazin Lake Upland and Selwyn Lake Upland Ecoregions; spring and fall migration corridors lie wholly or partially within the Kazan River Upland, the Dubawnt Lake Plain/upland, the Takijua Lake Upland (western extents) and the Garry Lake Lowland (Campbell et al. 2012, Campbell et al. 2014). Post-calving range varies but lies predominantly within the Garry Lake Lowland, the Back River Plain, and the Takijua Lake Upland to the west.

2.1 QUEEN MAUD GULF LOWLAND ECOREGION

The majority of the survey area covering the Beverly annual concentrated calving area (ACCA), lies within the Queen Maud Gulf Lowland Ecoregion with its eastern boundary extending into the Chantrey Inlet lowland within the last 15 years **(Figure 5)**. The Queen Maud Gulf Lowland extends eastward along the Arctic slope, from Bathurst Inlet to near Chantrey Inlet and is associated with the lowlands south of the Queen Maud Gulf. The mean annual temperature of this ecoregion is approximately -11°C with a summer mean of 5.5°C and a winter mean of -27°C. The mean annual precipitation of the ecoregion varies according to latitude, ranging from 125 mm within its northern extents, to 200 mm within its southern extents.

The Queen Maud Gulf Lowland Ecoregion is classified as having a low Arctic ecoclimate and is characterized by a cover of shrub tundra vegetation, consisting of dwarf birch (*Betula glandulosa*), willow (*Salix spp*.), northern Labrador tea (*Ledum decumbens*), mountain avens (*Dryas spp.)*, and Ericatious shrubs (*Vaccinium* spp).

Tall dwarf birch, willow, and alder (*Alnus crispa*) occur on warm sites; while wet sites are dominated by sphagnum moss (*Sphagnum spp.*) and sedge (*Carex spp.*) tussocks. Geologically the region is composed of massive Archean rocks that form broad, sloping uplands that reach about 300-m above sea level (ASL) in the south, and subdued undulating plains near the coast. The coastal areas are mantled by silts and clay of postglacial marine overlap. Bare bedrock is common, and turbic and static cryosols, developed on discontinuous thin sandy moraine with level alluvial marine deposits, are the dominant soils. Permafrost is continuous and deep with low ice content. The Queen Maud Gulf Lowlands are an important habitat for waterfowl and shorebirds, and the Queen Maud Gulf Bird Sanctuary covers most of the ecoregion offering protection to much of the Beverly ACCA (Wiken, 1986; Ecological Stratification Working Group, 1996).

2.2 CHANTREY INLET LOWLAND ECOREGION

The eastern extents of the Beverly ACCA lie within the Chantrey Inlet Lowland Ecoregion **(Figure 5)**. The Chantrey Inlet lowland is associated with lowlands surrounding Chantrey Inlet and Adelaide Peninsula. The mean annual temperature of this ecoregion is -12°C, with a summer mean of 4.5°C and a mean winter low of -28°C. The mean annual precipitation is similar to the western extents of the Beverly ACCA, and ranges from 125 mm to 200 mm. The Chantrey Inlet Lowland Ecoregion is classified as having a low Arctic ecoclimate characterized by large areas of exposed, sparsely vegetated bedrock in association with shrub tundra vegetation, consisting of dwarf birch, willow, northern Labrador tea, *Dryas spp.*, and *Vaccinium* spp. Tall dwarf birch, willow, and alder occur on warm sites while wet sites are dominated by sphagnum moss and sedge tussocks.

Near the coast of this ecoregion the surface is mantled by silts and clay of postglacial marine overlap, and is underlain by massive Archean rocks that form a level to undulating plain that reaches about 300-m ASL within its southern extents. Turbic and static cryosols developed on discontinuous, thin, sandy moraine, and level alluvial and marine deposits, are the dominant soils in the ecoregion. The east and west sides of Chantrey Inlet are underlain by continuous permafrost with low ice content. The northern half of the Adelaide Peninsula is characterized by continuous permafrost with medium to high ice content in the form of ice wedges and massive ice bodies (Wiken, 1986; Ecological Stratification Working Group, 1996).

Table 1. Beverly mainland migratory barren-ground caribou seasonal range areas within the Northwest Territories and Nunavut based on telemetry data current to 2013 (Campbell et al. 2014). Though the annual range of the Beverly subpopulation crosses into Saskatchewan, the combined 95% utilization distribution of all Beverly seasonal ranges do not.

Season	Total Area (km^2)	NU Area (km^2)	NWT Area (km^2)	NU %	NWT %
Spring	53,287	36,858	16,428	69%	31%
Calving	16,131	15,951	179	99%	1%
Post-calving	35,119	34,808	311	99%	1%
Summer	176,940	151,380	25,560	81%	19%
Fall Migration	27,781	8,344	19,437	32%	68%
Rut	96,953	24,581	72,372	25%	75%
Winter	91,459	19,024	72,436	21%	79%

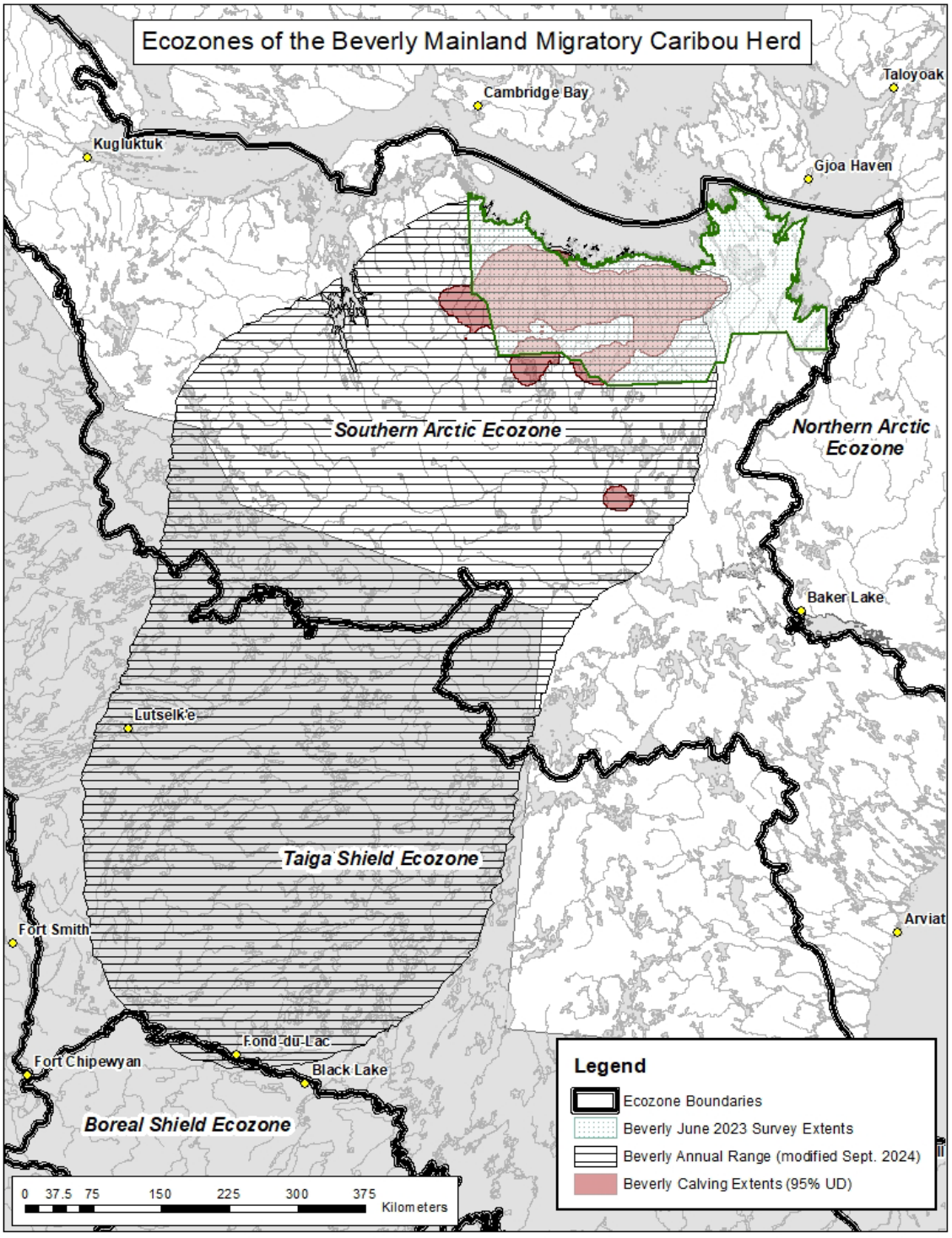


Figure 4. Ecozones of the Beverly barren-ground caribou subpopulations annual range extents and annual concentrated calving areas (ACCA) (Wiken, 1986, Ecological Stratification Working Group 1996, Campbell et al. 2014 & 2019).

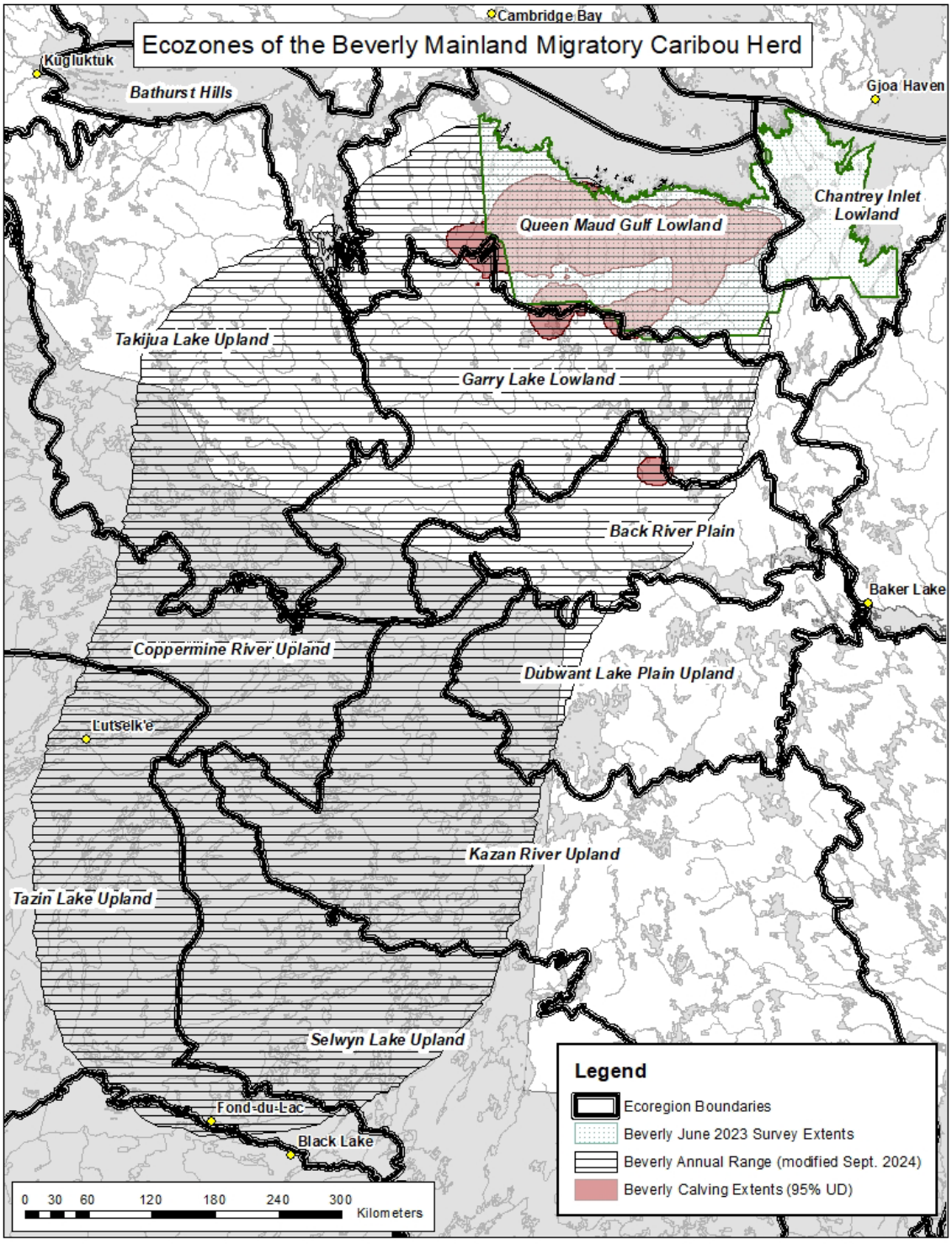


Figure 5. Ecoregions of the Beverly barren-ground caribou subpopulations annual range extents and annual concentrated calving areas (ACCA) (Wiken, 1986, Ecological Stratification Working Group 1996, Campbell et al. 2014).

3.0 METHODS

The June 2023 Beverly barren-ground caribou herd stratified random transect survey was designed to be based out of the communities of Cambridge Bay, and Gjoa Haven, however, aviation fuel shortages in Gjoa Haven in June 2023 necessitated a redesign of survey logistics with Cambridge Bay as the sole base of operations. This change did not impact survey quality but increased survey costs and ferrying time at the beginning and end of each flying day.

The abundance estimate used a cooperative double observer pair visual method, and a novel videographic method, to count caribou along pre-determined transect lines. The inclusion of a photographic method was necessary due to a previously undocumented and uncharacteristic clumping of calving Beverly caribou observed on the 2023 calving ground. Though a contingency to use a photo-plane should such an event occur was in place, the plane that was required was unable to position to Cambridge Bay due to mechanical issues, necessitating the development of an alternative photographic option.

Both the visual and videographic portions of the reconnaissance and abundance surveys used two high-wing, turbine, fixed wing Twin Otter aircraft for transect counts, and one Eurocopter B-2 rotary wing aircraft for abundance strata composition counts. The multiple aircraft were used to shorten the survey period, take maximum advantage of good weather windows, reduce the probability of either double counting and/or under counting (due to movement over time) caribou within the study area, and to involve more HTO representation within survey aircraft. At the same time as the video and visual surveys, the rotary wing aircraft was used to classify both visual and video groups of breeding and non-breeding female caribou,

yearlings, calves, young bulls, and mature bulls. To extrapolate to a whole herd estimate, we used fall composition data collected during the rutting period in October 2022 by GNWT Biologists (Adamczewski et al. 2024). The Fall data provides information on herd sex ratio as all ages and sexes of caribou gather together within their breeding range in and around mid to late October.

The visual portion of the survey employed a dependent double-observer pair method. The typical configuration was comprised of the pilot, two data recorders (rear left and front right) and four observers (two on the left side of the aircraft and two on the right side). Only caribou observed within the strip, as defined by the inner and outer streamers attached to the left- and right-wing struts, were recorded. Survey timing relied predominantly on a daily assessment of collared cow movement rates.

3.1 RECONNAISSANCE AND ABUNDANCE SURVEYS

The June 2023 Beverly calving-ground survey used both reconnaissance and abundance surveys, which in turn, were comprised of two observational methods to count caribou. These methods included a double observer pair visual method for moderate to low densities, and a videographic method to address higher densities (above the ability to accurately count visually) due to clumping. Both surveys used two Twin Otter Dehaviland high wing survey aircraft equipped with radar altimeters to ensure that an altitude of 121.92 m (400 feet) above ground level (AGL) was maintained. The strip width on each side of the aircraft was 400 meters, for a total transect width of 800 m. Survey strip widths were marked by streamers attached to the wing struts (Campbell et al. 2012) and were calculated using the formula from Norton-Griffiths (1978):

$$w = W*(h/H)$$

where W is the strip width, H is the flight height, h is the observer height when the plane is on the ground and w is measured and marked on the ground to position wing strut marks (**Figure 6**).

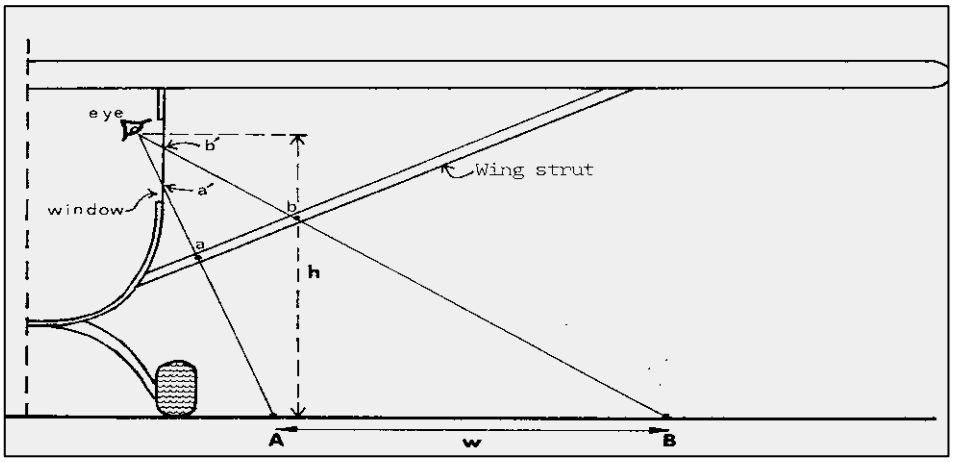


Figure 6. Schematic diagram of aircraft configuration for strip width sampling (Norton-Griffiths, 1978).

3.1.1 Double Observer Pair Method

The double-observer method involves one “primary” or “front” observer who sits in the front left- or right-hand side of four observer seats centered around the wing struts, and one “secondary” or “rear” observer who sits behind the primary observer on the same side of the plane (**Figure 7**). The method adhered to five basic steps; ***1 -*** The primary observer called out all groups of caribou (number of caribou and location) he/she saw within the 400-meter-wide strip transect before they passed halfway between the primary and secondary observer (approximately at the wing strut). This included caribou groups that were between approximately 12 and 3 o’clock for right side observers and 9 and 12 o’clock for left side observers. The main requirement was that the primary observer be given time to call out all caribou

seen before the secondary observer called them out; ***2 -*** The secondary observer called out whether he/she saw the caribou that the first observer saw and observations of any additional caribou groups. The secondary observer waited to call out caribou until the group observed passed half way between observers (between 3 and 6 o'clock for right side observers and 6 and 9 o'clock for left side observer); ***3 -*** The observers discussed any differences in group counts to ensure that they are calling out the same groups or different groups and to ensure accurate counts of larger groups; ***4 -*** The data recorder categorized and recorded counts of caribou groups into "primary only", "secondary only", and "both", entered as separate records; ***5 -*** The observers switched places approximately half way through each survey day (i.e. during re-fueling) to monitor and account for observer ability. The recorder noted the names of the primary and secondary observers.

The sample unit for the survey was "*groups of caribou*" not individual caribou. Recorders and observers were instructed to consider individuals to be those caribou that were observed independent of other individual caribou and/or groups of caribou. If sightings of individuals were influenced by other individuals, then the caribou were considered a group.

3.1.2 Video Assisted Counts of High-Density Strata

Unlike previous Beverly calving-ground surveys (both reconnaissance and abundance 2008 to 2018), larger than expected group sizes (>100 caribou/km^2) of caribou were observed during the 2023 survey which challenged the ability of observers to visual count caribou within the 400m survey strip. Though a back-up photo plane was put in place as a contingency prior to the start of the abundance survey, it was not available during the abundance phase due to mechanical problems necessitating an alternative means of accurately counting the observed clumped aggregations of caribou. To accomplish this a video-assisted strategy was implemented. The basic procedure involved data recorders utilizing cell phones so that observations were recorded using video when large groups were encountered

(**Figure 8**). To collect video data, we mounted two Samsung S22 smartphones in plane 1, and two iPhone 12 smartphones in plane 2, on each on the left and right sides of each plane. The field of view of the video spanned the entire strip width and included the leading edge of the struts with associated strip width markings. This video, or secondary counting method, allowed for a simultaneous visual count of the same groupings of caribou (**Figure 9**).

3.1.3 Survey Timing

We used daily movement rates of collared Beverly caribou cows, and to a lesser extent, collar reconnaissance observations, to identify the onset of peak calving for the purposes of beginning the reconnaissance survey. From collars, we estimated peak calving as the dates where female daily movement rates were at their lowest. The examination of 20 years of daily movement rates of collared Beverly females have been shown to accurately predict the beginning of peak calving when movement rates drop below 5 km per day (Campbell et al. 2012; Boulanger et al. 2018). Collar reconnaissance flights provided an additional index of the onset of peak calving when the proportion of calves per 100 females observed across the extents of the Beverly ACCA reach 15% or higher (Campbell et. al. 2012, Boulanger et. al. 2018).

3.1.4 Reconnaissance Survey

We initiated the reconnaissance survey when daily movement rates dropped below 5 km per day and/or observed calves per 100 cows equaled or exceeded 15 percent. The reconnaissance survey is a low coverage survey (9%) used to survey known calving extents, as well as beyond known calving extents when justified, to ensure all possible aggregations of females are located and included in the abundance survey to follow. This phase of the study collects data to generate geographically

explicit relative densities of caribou and their general composition (such as breeding and non-breeding females). The results of the reconnaissance survey are then used to calculate and plot relative densities of females for the purposes of stratification, with areas of similar density grouped together into strata for the abundance phase of the survey. Defining strata in this way increases precision of the population estimate and informs on the counting method required to maximize accuracy.

In total, fifty-one north-south oriented reconnaissance survey transects ranging from 50 to 180 km long were distributed systematically at 10-km spacing across the northern mainland from Ellice River to Committee Bay **(Figure 10)** using UTM coordinates and the WGS 84 datum. Reconnaissance transects covered 7,570 linear kilometers and observed an area estimated to cover 6,056 km^2. Each transect had associated transect station points that were located at 10-kilometer intervals **(Figure 10)**. Each station had an alphanumeric identifier (e.g. Bv83) allowing it to be easily referenced. Each 10-kilometer transect segment was named after its northern station. Transects were created using Environmental Systems Research Institute (ESRI) ArcMap Geographic Information System (GIS) software and were based on the UTM zone 15 World Geographic System and the (WGS) 1984 coordinate system.

Following the systematic reconnaissance but prior to the initiation of the visual abundance survey, we entered all observations into ESRI ArcMap GIS software to calculate relative densities of breeding females using a tool utility. The tools allowed us to calculate the relative density of observed caribou locations along the sample transects and display these results on a map. We used vector-based analysis methods based on the following steps:

1. The survey transect segments were buffered by a user-specified width (1,000m in this survey; i.e., 800m strip width and 200m blind spot under the aircraft) yielding polygons that were 10 km^2 (i.e., 1.0 km wide x 10 km long).

2. The survey observation points were intersected with the derived buffer polygons.

3. The density was calculated for each polygon by dividing the number of 1+ year-old caribou by the area of the buffer polygon (# of 1+ year old caribou/km²).

4. The relative density (number of observations per km²) was thematically displayed on a map based on pre-defined classes or bins.

We then used the resulting graphics to stratify the breeding female distribution into high, medium and medium/low-density strata.

3.1.5 Allocation of Effort

The main objective of the survey was to obtain a precise and accurate estimate of female caribou on the Beverly herd calving ground and assess their breeding status. To achieve this, the survey area was stratified following the results of a systematic reconnaissance survey. Survey stratification involved grouping neighboring segments with similar density into contiguous areas termed strata, so that each stratum enveloped distributions of similar caribou densities. In addition, stratification was used to determine if stratum density summaries where high enough to require the use of a photo survey plane (> 20 caribou/km^2), or if visual observation methods could be used. In this survey, two higher-density strata were identified; these strata were planned to be addressed, at least in part, using photographic methods. Other strata that had lower densities of caribou (< 20 caribou/km^2) were flown using visual survey methods alone. Given that the objective of the survey was to estimate breeding females and total females, priority, as defined by survey effort, was given to those delineated strata showing evidence of female caribou.

Once the survey strata were delineated, an estimate of caribou numbers was derived from the reconnaissance data (Jolly 1969). The relative population size of each

stratum and the degree of variation of each estimate were used to allocate the number of transects within each stratum. Within each stratum, transects were aligned at right angles to the longitudinal axis of the stratum to maximize the total number of transects (N). Transect placement for each stratum began with the initial random placement of the first transect perpendicular to a line running through the stratum parallel to its longest axis, with the remaining transects systematically placed at regular intervals according to the allocation of survey effort. During the allocation of effort process, we also had to consider available resources in the final determination of strata total coverage (Heard 1987, Campbell et. al. 2012).

Two potential strategies for allocation were considered for this aerial survey. First, optimal allocation of survey effort was considered based on sampling theory (Heard 1987, Thompson 1992, Krebs 1998). Optimal allocation basically assigned more effort to strata with higher densities given that the amount of variation in counts is proportional to the relative density of caribou within the stratum. Optimal allocation was estimated using the above-mentioned derived estimates of population size for each stratum and associated survey variance.

If strata were reasonably small, then optimal allocation was further adjusted to ensure an adequate number of transect lines. In particular, a power analysis of previous survey data suggested that there should be a minimum of 10 transects per stratum with closer to 20 transects being optimal for higher density areas. In general, coverage should be at least 15% with coverage starting at a minimum of 25% for higher density strata (Boulanger 2020). In the context of sampling, increasing the number of lines in a stratum minimizes the influence of any one transect on estimate precision. As populations become more clustered/clumped, a higher number of transect lines is required to achieve adequate precision (Thompson 1992, Krebs 1998). Transect orientations within strata, transect shape files, and coverage estimates, were generated and cross-validated using the *dssd* R package (Marshall 2021).

3.1.6 Abundance Survey

The abundance survey and coordinated abundance strata composition surveys immediately followed the reconnaissance survey. This quick turnaround was necessary to minimize the effects of the movement of caribou between newly delineated abundance strata based on reconnaissance observations. The abundance survey began June 13th, and was completed June 16th, following the completion of the reconnaissance survey on June 12th **(Table 2)**. The abundance survey used the same survey methods deployed during the reconnaissance survey, with the exception that the collection of composition data from the fixed wing aircraft was optional. Both the abundance and composition surveys were completed as quickly as possible while prioritizing good sightability. The study area within which all survey phases were flown, covered 288,312 km² and encompassed the known extent of caribou calving in the area of the Queen Maud Gulf and Adelaide Peninsula ACCA (Johnson and Mulders 2002; Johnson et al. 2008; Johnson and Williams 2008; Kelly in prep. 2010; Nagy et al. 2011, Campbell et al. 2012).

3.1.7 Composition Survey of Abundance Strata

June composition surveys were timed to begin concurrently with visual abundance surveys from 13 – 16 June 2023 to ensure minimal movement of animals occurred between strata. We used a B2 Eurocopter rotary aircraft to complete abundance strata composition. Sampling was structured to begin at a fuel cache and then proceed to a predetermined transect station within a minimum of two (2) kilometers from the strata corner/boundary. From this transect station the aircraft would proceed to the next nearest transect station to the north and/or south, prioritizing the sampling of the next nearest caribou group (including individuals) encountered on route. Transect stations (each 10 km apart) were selectively used to fly rough transects aimed to equally distribute composition effort (Campbell et al. 2012). At times, observed groups of caribou drew the aircraft away from the pre-planned flight

path. When sampling caused deviation from the preplanned flight path, the aircrew, when possible, would stop sampling caribou groups that were seen greater than 5 kilometers perpendicular to the original flight path between transect stations. From this point, only caribou groups observed within this five-kilometer buffer would be sampled and an attempt to rejoin the original flight path was made while sampling any groups of caribou encountered. During re-positioning flights from the stratum to the fuel caches, caribou encountered within a minimum of 2 km inside of target stratum boundaries were classified opportunistically and variation of flight paths was held to within 2 km to reduce deviation from the planned re-fueling routes.

During composition surveys caribou were classified as yearlings (≥ 1 year and < 2 years of age), bulls, cows with calves (calves < one month old), calves (< 1 year of age or short yearlings), cows with udders, udderless (udders not visible) cows with antlers, and udderless cows without antlers. Breeding cows were tallied as cows with calves, cows with udders, and udderless cows with antlers. Non-breeders were tallied as udderless cows with no antlers, yearlings, and bulls. Using this information, we estimated the proportions of breeding females, adult females, calves, yearlings, and young and mature bulls, for each stratum surveyed on the calving ground.

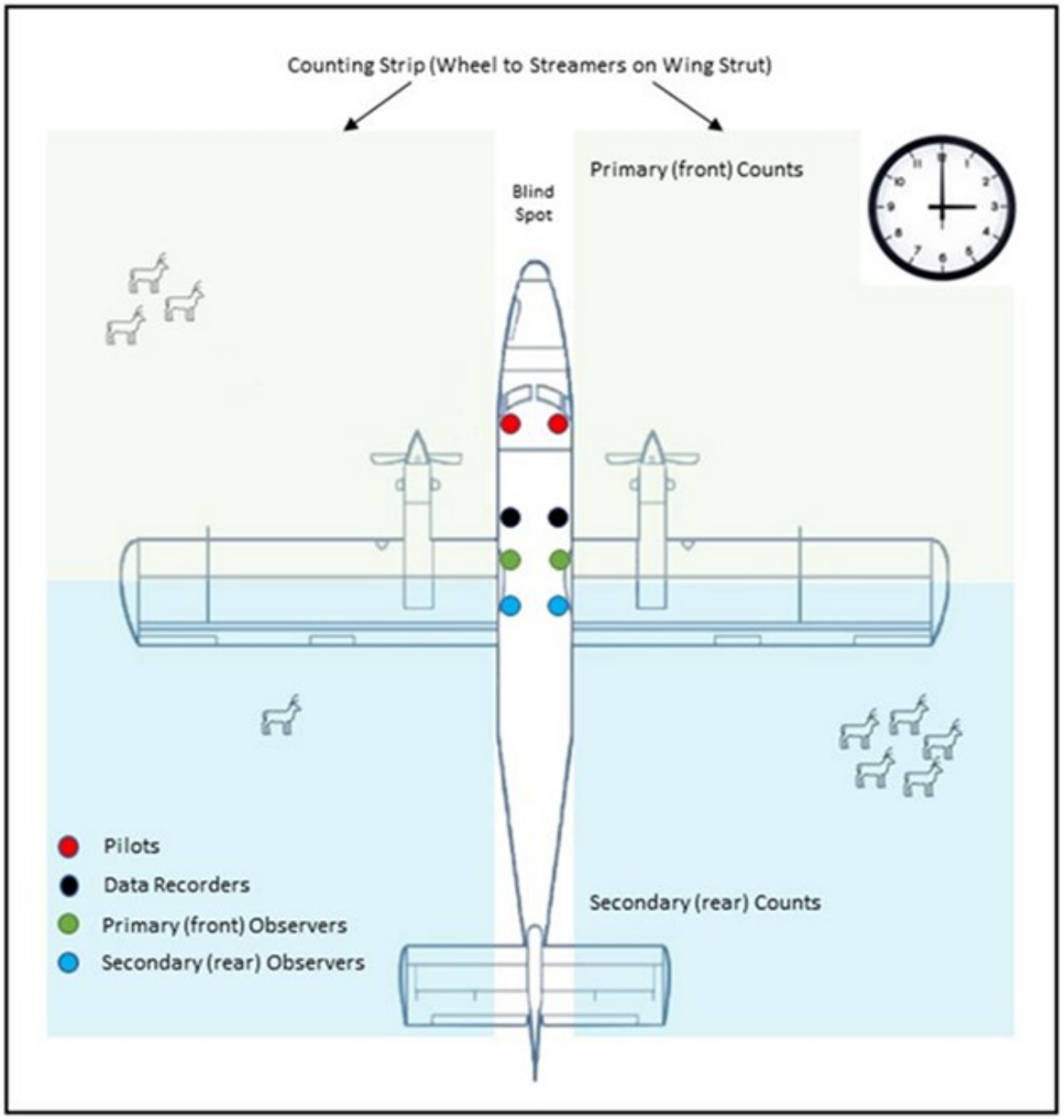


Figure 7. Observer position for double observer methods. The secondary (rear) observer calls caribou not seen by the primary (front) observer after the caribou have passed the main field of vision of the primary observer. Time on a clock is used to reference relative locations of caribou groups (e.g. “Caribou group at 1 o’clock”).



Figure 8. A Samsung S22 smartphone mounted on the window on the right side of the aircraft at the data recorders seat.



Figure 9. Screen shot from videos used for secondary video counts of caribou (note caribou within both video frames). This image taken from the data recorders seat on the right-hand side of the aircraft (note the inclusion of the leading edge of the wing strut and transect width markers). Excellent sightability conditions are evident with clear skies and no snow cover.

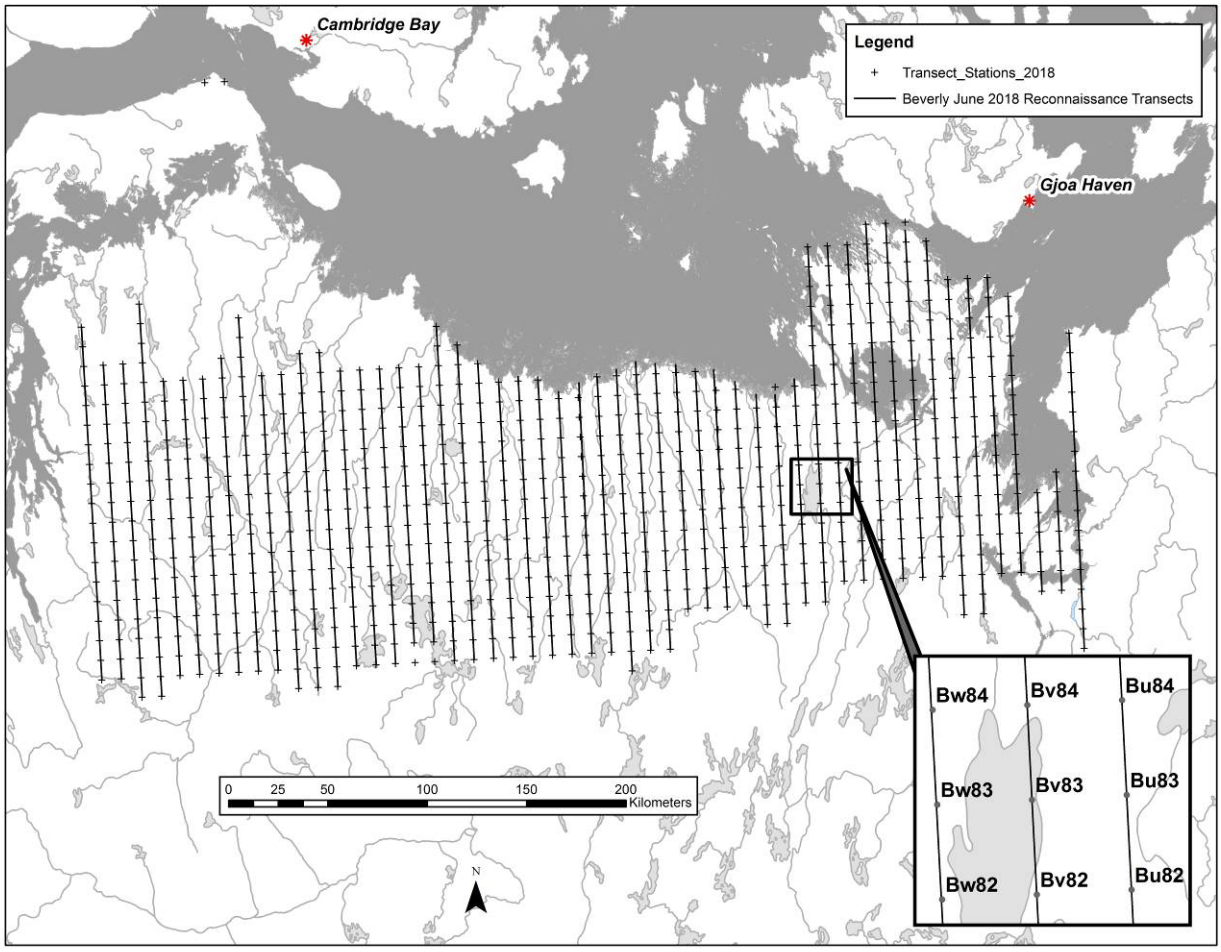


Figure 10. Reconnaissance transects and transect stations (lower right) of the Beverly 2023 calving ground abundance survey. Transects placed to cover the known extents of female caribou based on previous surveys and collar telemetry.

Table 2. A comparison between the June 2011, 2018, and 2023 Beverly mainland migratory caribou subpopulation abundance survey timing. Survey timing based on an observed reduction of daily collared caribou cow movement rates below 5 km/day.

Survey Activity	Date (2011)															
	Jun-04	Jun-05	Jun-06	Jun-07	Jun-08	Jun-09	Jun-10	Jun-11	Jun-12	Jun-13	Jun-14	Jun-15	Jun-16	Jun-17	Jun-18	Jun-19
Systematic Reconnaissance						X	X	X	X		*Ice Fog & Freezing Rain*					
Abundance										X		X	X	X		
Composition												X	X	X	X	

Survey Activity	Date (2018)															
	Jun-04	Jun-05	Jun-06	Jun-07	Jun-08	Jun-09	Jun-10	Jun-11	Jun-12	Jun-13	Jun-14	Jun-15	Jun-16	Jun-17	Jun-18	Jun-19
Systematic Reconnaissance		*Ice Fog & Freezing Rain*			X	X	X	X	X							
Abundance										X	X	X	X			
Composition										X	X	X	X	X		

Survey Activity	Date (2023)															
	Jun-04	**Jun-05**	**Jun-06**	**Jun-07**	**Jun-08**	**Jun-09**	**Jun-10**	**Jun-11**	**Jun-12**	**Jun-13**	**Jun-14**	**Jun-15**	**Jun-16**	**Jun-17**	**Jun-18**	**Jun-19**
Systematic Reconnaissance			*Ice Fog & Freezing Rain*	*Ice Fog & Freezing Rain*	X	X	*Ice Fog & Freezing Rain*	X	X							
Abundance									X	X	X	X				
Composition											X	X	X			

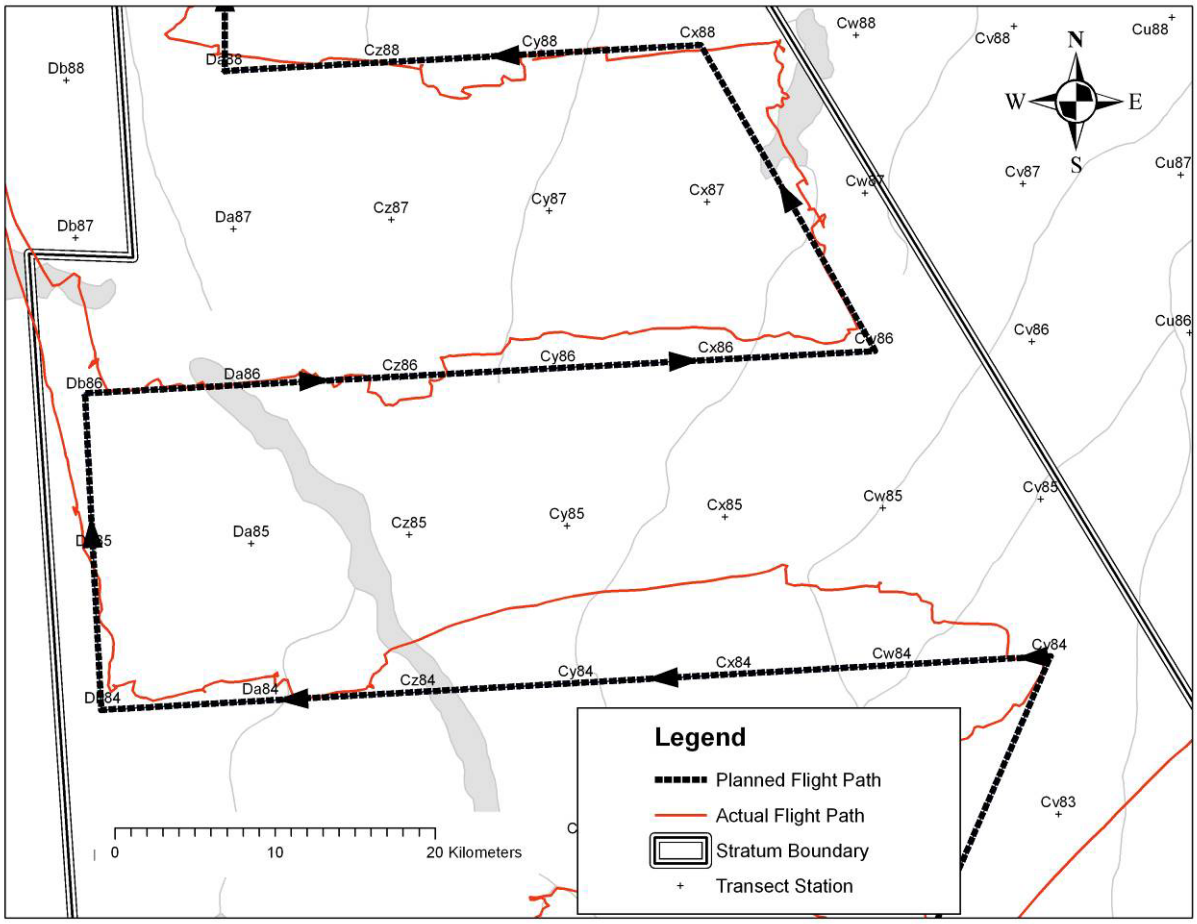


Figure 11. Stratum composition flight methods used during the 2023 Beverly calving ground survey vs. planned routes. Actual track deviations (red line) away from planned routes (black lines) were required to classify all observed caribou groups. The next nearest group would be classified up to a maximum of 5 km perpendicular to the planned route (half way between transect stations).

3.2 SURVEY ANALYSIS

3.2.1 Video Interpretation and Analysis

Transect videos were converted to orthomosaic images using a process that extracted overlapping video frames that were then imported into Agisoft Metashape for photogrammetric processing. These orthomosaic images were generated without georeferencing information due to the data capture method so manual georeferencing was achieved using landscape features from basemap imagery, and an approximate location from the video capture time and aircraft GPS data. The orthomosaic images and videos were then reviewed by interpreters in tandem to locate caribou, resulting in the point dataset with a waypoint for each counted caribou. Due to the low resolution of the video, methods to increase the accuracy in caribou identification were developed. One of the key ways interpreters increased their confidence in identifying caribou was through observing movement in the video and identifying the same suspected caribou point in the orthomosaic image. One challenge was that the video cameras mounted in the survey aircraft windows were not stabilized with a fixed field of view and as a result, the orthomosaic image was influenced by the aircraft pitch, roll, and other movements, unlike a human observer that can compensate on the fly for these three-dimensional movements and maintain a relatively geographically stable observation strip in their mind. As a result, it was possible that some caribou waypointed on videos fell outside of the actual 400m survey strip or were “Off Transect”. For this reason, a 400m survey strip was overlaid on the orthomosaic images to account for potential caribou counted on videos outside the survey strip. Both the left and right side 400m survey strips included a 182-meter blind spot under the survey plane. A tally of caribou counted within and outside the survey strip was then calculated for each orthomosaic image (**Figure 12**).

An additional challenge with comparison and reconciliation of visual and video counts was that developed polygons often spanned many visual observations which precluded exact matching of visual observations with counts on each orthomosaic image. As in the visual double observer survey method, orthomosaic images often did not overlap the actual transect line due to the blind spot under the survey plane. Therefore, waypoints of visual observations did not occur in the ortho polygon. In addition, it was likely there was a time lag between the time an observation occurred and the time it was entered in the tablet by the data recorder leading to some visual waypoints occurring past end of an orthomosaic image. To confront this uncertainty, the distance from each tablet waypoint to the nearest orthomosaic image (on the same side of the plane of the waypoint) was estimated. Tablet waypoints were then paired with the orthomosaic image location based on a series of threshold distances of the waypoint from the polygon ranging from 200 to 1,500 meters. Sensitivity analyses were then conducted to assess the optimal cutoff threshold.

Once an appropriate distance threshold was determined, transect counts were tallied for each survey line using counts within each orthomosaic image, and visual counts that were outside the orthophotos. The resulting counts for each line were then used to estimate abundance for the video/photo strata. The proportion of counts from photos and visual observations, were also tallied for each line to assess the proportion of caribou on the line that were counted in the photos versus visual counts. It was not possible to apply double observer methods to lines that had both visual and photo counts. For these strata, the Jolly strip transect estimator was applied (Krebs 1998).

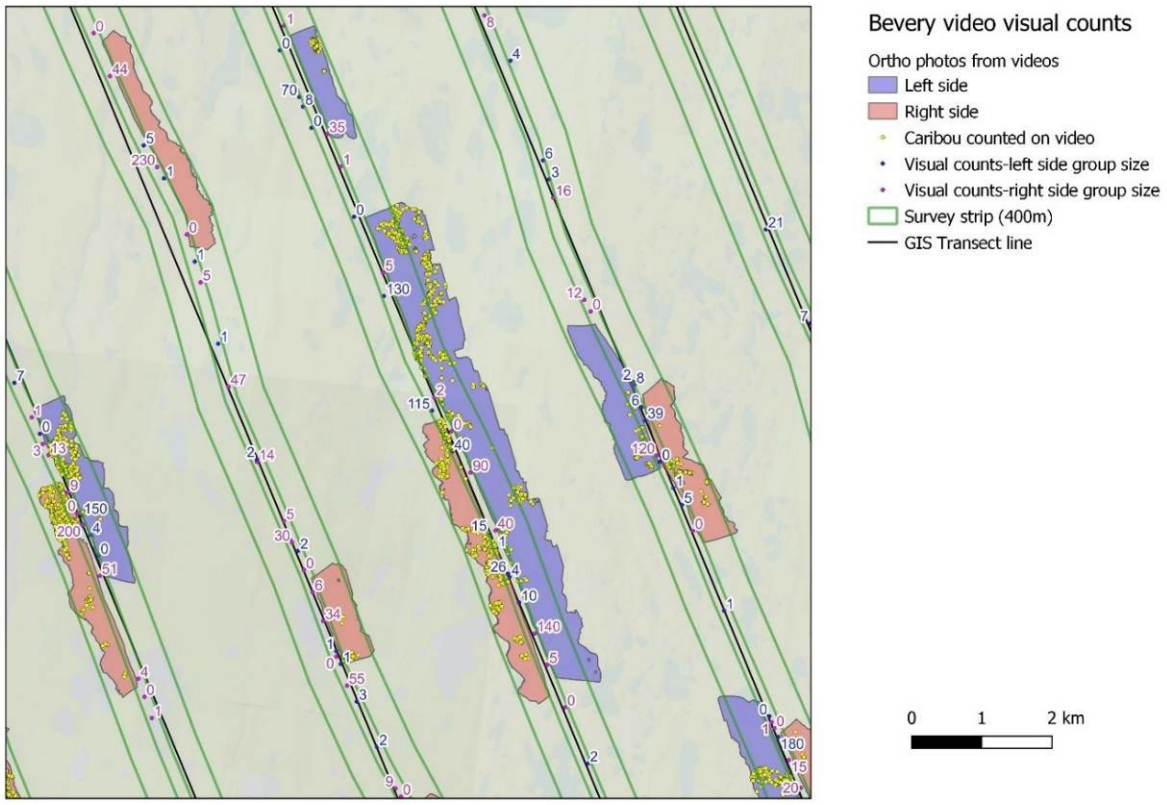


Figure 12. Example of orthomosaic polygons compared to visual observations. The 400m survey strip is outlined in green. The count of caribou groups from the survey plane is labelled next to each waypoint with caribou counted on orthophotos delineated by yellow points.

3.3 ANALYSIS OF ABUNDANCE

Estimates of herd size and associated variance were developed using the mark-recapture distance sampling (MRDS) package (Laake et al. 2012) in program R program (R Development Core Team 2009). In MRDS, a full independence removal estimator which models sightability using only double observer information (Laake et al. 2008a, Laake et al. 2008b) was used therefore making it possible to derive double observer strip transect estimates. Strata-specific variance estimates were calculated using the formulas of (Innes et al. 2002). Estimates from MRDS were cross checked with strip transect estimates (that assume sightability=1) using the formulas of Jolly (1969)(Krebs 1998). Data was explored graphically using the ggplot2 (Wickham 2009) R package and QGIS software (QGIS Foundation 2020).

3.3.1 Modelling of Visual Sighting Probability Variation

For the visual component of the survey one assumption of the double observer method is that each caribou group observed had an equal probability of being sighted. To account for differences in sightability we also considered the following sightability covariates in the MRDS analysis (**Table 3**). Each observer pair was assigned a binary individual covariate and models were introduced that tested whether each pair had a unique sighting probability. Previous analyses (Campbell et al. 2012, Boulanger et al. 2014) suggested that the size of the group of caribou had strong influence on sighting probabilities and therefore we considered linear and log-linear relationships between group size and sightability (**Table 3**). Changes in cloud and snow cover were recorded by data recorders as ordinal rankings. We suspected that sightability was most likely lowest in mixed snow cover conditions and therefore we considered both categorical and linear models to describe variation in sightability caused by snow cover. Cloud cover could also influence sightability

by causing glare, flat light, or variable lighting. We used the same basic strategy to model cloud cover variation as snow cover variation.

Table 3. Covariates used to model variation in sightability for double observer analysis.

Covariate	Acronym	Description
observer pair	observers	each unique observer pair
group size	size	size of caribou group observed
	Log(size)	Natural log of group size
snow cover	snowcat	snow cover (0,25,75,100)
	snow	continuous
cloud cover	cloudcat	cloud cover (0,10,25,75,100)
	cloud	continuous

The fit of models was evaluated using the Akaike Information Criterion (AIC) index of model fit. The model with the lowest AIC_c score was considered the most parsimonious (involving the fewest assumptions), thus minimizing estimate bias and optimizing precision (Burnham and Anderson 1998). The difference in AIC_c values between the most supported model and other models (ΔAIC_c) was also used to evaluate the fit of models when their AIC_c scores were close. In general, any model with a ΔAIC_c score of less than 2 was worthy of consideration.

3.3.2 Estimates of Breeding Females, Adult Females, and Adults on the Calving-Ground

Calving-ground composition surveys were conducted concurrently with visual surveys. Breeding cows were tallied as cows with calves, cows with udders, and

udderless cows with antlers. Non-breeders were tallied as udderless cows with no antlers, yearlings and bulls. Using this information, the proportion breeding females, adult females, bulls, and adults was estimated for each stratum surveyed on the calving ground. Bootstrap methods were used to obtain variance estimates. In this case, 1,000 resampling's of the data were generated, and the mean and standard deviation from resampling were used as point estimates for the proportion of breeders, and the associated standard error (Manly 1997).

Estimates of proportions of each target group (breeding females, adult females, and bulls) were then multiplied by the visual estimate of all caribou for each stratum to obtain an estimate of the abundance of the given target group. Variances for combined visual strata were obtained using program MRDS therefore accounting for covariances introduced by the double observer sightability models. Variances for photo and visual strata, or composition survey and strata estimates, were obtained for the combined estimates using the delta method (Seber 1982, Williams et al. 2002), assuming no correlation between the two estimates. Degrees of freedom for combined estimates were estimated using the formulas of Buckland et al (1993).

3.3.3 Analysis of Fall Composition Data

Composition surveys were conducted in the fall of 2022 by the Government of the Northwest Territories (Adamczewski et al. 2024). These surveys were used to determine bull-cow ratios and the proportion adult cows, needed for extrapolated whole-herd population estimates. The bull-cow ratio was simply the count of bulls divided by the count of cows, whereas the proportion of adult cows was the number of cows divided by the number of adult cows and adult bulls. As with the calving ground composition survey data, a bootstrap procedure was used for point estimates, standard error, and percentile-based confidence limits. For this, a bootstrap process using 1,000 resamples was conducted on the original data set (Manly 1997).

3.3.4 Estimation of Herd Size

Total herd size (adults at least two years old) was estimated by using a recent estimate of the bull:cow ratio from a fall composition survey conducted in October 2022 (Adamczewski et al. 2024) to extrapolate or “add on” the bulls to the estimate of adult females (Adamczewski et al. 2017). This method of extrapolation was first used in the 2014 Qamanirjuaq caribou herd survey (Campbell et al. 2016), and has been used in other recent calving photo surveys for the Bluenose-East and Bathurst herds (Adamczewski et al. 2022, Boulanger et al. 2022). This estimator uses the estimate of total adult females divided by the proportion of adult females in the herd (sex ratio) from one or more fall composition surveys. This accounts for the bulls in the herd, very few of which are on the calving grounds in June. It makes no assumption about the pregnancy rate of the females and does not include the yearlings.

An alternative herd size estimate was also derived by adding the estimate of adult cows and bulls in the survey strata. This estimate assumes that the majority of bulls occurred on the surveys strata and that all bulls were from the Beverly herd. This assumption was investigated by comparing the locations of collared bulls known to be Beverly (from previous years and seasons) during the calving ground survey (mid-June), post-calving (mid-July) and during the rut (mid-October). The question of interest was how the locations of bulls compared on the calving ground versus mid-July when herds are more commonly segregated, and the rut when composition surveys occur, when the herds are the least segregated (to obtain bull-cow ratios for extrapolated estimates). This analysis was done in collaboration with Jan Adamczewski and Judy Williams (Environment and Climate Change (ECC), Government of Northwest Territories) using bull data retrieved from the ECC database (Adamczewski et al. 2020, Adamczewski et al. 2024).

3.4 COLLAR ANALYSIS

3.4.1 Movement Between Calving-Grounds

Further analyses were conducted to assess overall movements of caribou between the Bathurst, Beverly, Ahiak, Lorillard, and Wager Bay herds to assist in interpretation of estimates. For this analysis, the June locations of collared females were classified based on calving ground location. Using this information, movements between calving grounds as well as fidelity were assessed graphically. Multi-state models (Hestbeck et al. 1991, Brownie et al. 1993, White et al 2006) in program MARK (White and Burnham 1999) were then used to estimate movement (transition probabilities) between calving ground areas for females that were monitored for more than one calving season.

3.4.2 Survival Rate Analysis

Collar data collected for female caribou from 2011-2023 were compiled for the Beverly caribou herd by GNWT ECC staff. Fates of collared caribou were determined by assessing the movement of collared caribou cows, with mortality being assigned to collared caribou based on lack of collar movement over an extended period unrelated to collar failure or device drop-off. The data were then summarized by month as live or dead caribou. Caribou whose collars failed or were scheduled to drop off were censored from the analysis. Data were grouped by “caribou years” that began during calving of each year (June) and ended during the spring migration (May). The Kaplan-Meier method was used to estimate survival rates, accounting for the staggered entry and censoring of individuals in the data set

(Pollock et al. 1989). This approach also ensured that there was no covariance between survival estimates for the subsequent demographic model analysis.

3.5 INTEGRATED POPULATION MODEL ANALYSIS

The most direct measure indicating the status of breeding females is their survival rate, which is the proportion of breeding females that survive from one year to the next. This metric, along with productivity (recruitment of yearlings to adult breeding females) determines the overall population trend. For example, if breeding female survival is high then productivity in previous years can be relatively low and the overall trend in breeding females can be stable. Alternatively, if productivity is consistently high, then some reductions in adult survival rate can be tolerated. The interaction of these various indicators can be difficult to interpret. A population model can help increase our understanding of these various indicators and in turn, of herd demography.

We used a Bayesian state space Integrated Population Model (IPM) (Buckland et al. 2004, Kery and Schaub 2012, Schaub and Kery 2022) based upon the original (OLS) model (White and Lubow 2002) developed for the Bathurst herd (Boulanger et al. 2011) to further explore demographic trends for the Beverly herd. This IPM has been applied to both the Bathurst and Bluenose-East herds since 2012 (Boulanger et al 2024). A state space model allows separate modelling of field sampling estimates and demographic processes. This work was in collaboration with a Bayesian statistician/modeler (Joe Thorley-Poisson Consulting) (Thorley 2017, Ramey et al. 2018, Thorley and Boulanger 2019). Details of the IPM are given in **Appendix A**.

We used adult female estimates, as well as calf-cow ratios, bull-cow ratios, estimates of the proportion of breeding females on the calving ground, and adult female and bull survival rates from collared caribou to estimate the most likely adult female survival values that would result in the observed trends in all of the demographic indicators for the Beverly herd. Calf cow ratios were recorded during fall (late October) and spring (late March to April) composition surveys whereas the proportion of breeding females was measured during composition surveys conducted on the calving ground. Breeding female proportions were estimated as the ratio of breeding females to adult females from each calving ground survey used in the analysis.

The Bayesian IPM model is a stage based model that divides caribou into 3 age-classes, with survival rates determining the proportion of each age class that makes it into the next age class (**Figure 13**); this structure is identical to the OLS modeling done previously on the Bathurst and Bluenose-East herds (Boulanger et al. 2019). We note that the underlying demographic model used for the Bayesian state space model is identical to the previous OLS model. However, the Bayesian IPM method provides a much more flexible and robust method to estimate demographic parameters that takes into account process and observer error. One of the biggest differences is the use of random effects modelling to model temporal variation in demographic parameters. For random effects models, it is assumed that there is a central mean value for a parameter (i.e. Cow survival) with a distribution of values created over time based on temporal variation. This contrasts with the OLS method where temporal variation was often not modelled or modelled with polynomial terms which assumed an underlying directional change over time.

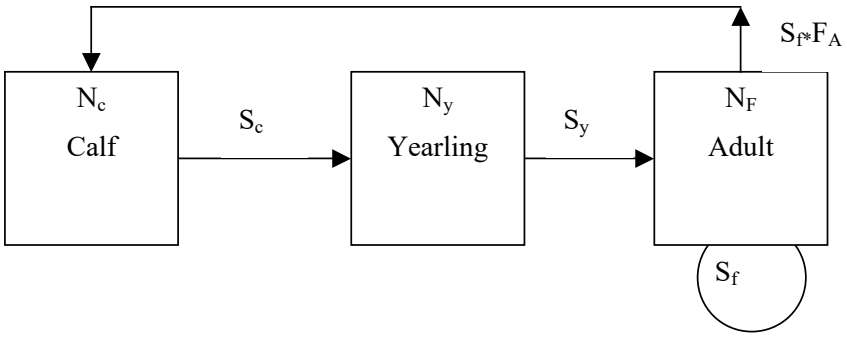


Figure 13. Underlying stage matrix life history diagram for the caribou demographic model used for Beverly caribou. This diagram pertains to the female segment of the population. Nodes are population sizes of calves (N_c), yearlings (N_y), and adult females (N_F). Each node is connected by survival rates of calves (S_c), yearlings (S_y) and adult females (S_f). Adult females reproduce dependent on fecundity (F_A) and whether a pregnant female survives to produce a calf (S_f). The male life history diagram was similar with no reproductive nodes.

3.6 AERIAL WILDLIFE SURVEY–OBSERVATION COLLECTOR (AWS-OC)

To increase data entry speed without reducing accuracy, and to reduce the time required to perform preliminary analysis of reconnaissance data for abundance stratification, a digital data entry system, termed the "Aerial Wildlife Survey – Observation Collector" (AWS-OC), developed and deployed in June 2011, was used for this survey (Campbell et. al. 2012). The software was originally developed by the Government of Nunavut, Wildlife Research Division, in collaboration with Integrated Ecological Research, Caslys Consulting Ltd, and Nunavut Tunngavik Inc. (NTI), in 2011, and originally deployed on the June 2011 Beverly mainland migratory barren-ground caribou calving ground abundance survey (Campbell et al. 2012). Since its original launch, improved hardware, and some enhancements to the AWS-OC software had been undertaken prior to its deployment in June 2018, and again in 2023 (Boulanger et al. 2018).

The AWS-OC software operates with Windows editions 7 through 10 and was developed specifically for use in both independent and dependent double-observer pair aerial caribou surveys, including distance-sampling applications, to facilitate the collection of field data, and the subsequent management of the resultant observation dataset. This tablet-based system allows for the instantaneous entering of caribou group waypoints (observations) directly into a digital database. Data entry time was cut by approximately 50% over standard hand written datasheets, with the added benefits of continuous back up onto a USB drive into a digital database with no additional data entry required. The application includes two modules:

1- The ***AWS-OC Field Collection Module*** is designed for collecting observation data while airborne. The application is spatially enabled to connect with a Global Positioning System (GPS), and displays the current location on maps that are compatible with ESRI's ArcGIS software. Minimal training is required to operate

the system;

2- The ***AWS-OC Data Manager Module*** is designed for use on the ground or in the office for data management and field planning tasks. Advanced user functionality is focused on tabular data accessible with MS Access database software and integration with ESRI ArcGIS.

The AWS-OC is designed for use on windows touch screen tablets and has been designed and tested to integrate with the internal (integrated) GPS signal of the Xplore (Motion) R12 touch screen tablet. Configuration still allows for external GPS connections if required. For added durability and stability in severe turbulence, the tablets have been equipped with solid-state hard drives. The tablets also included swappable batteries that allow for uninterrupted operation during a flight, and USB ports to allow for data transfer following field collection. Additional equipment and tools that complete the AWS-OC field kit include a spare battery to provide added insurance for power supply for a full day of fieldwork, USB flash memory stick, and two software utility applications to merge text files and merge shapefiles to assist with data management tasks.

The data entry page of the *Survey Session Details* form (**Figure 14**) allows the entry of common details (i.e., unique aircraft ID, crew assignments, and appropriate transect file, which enables the auto-completion of transect details based on the GPS signal). Additionally, the software automatically records altitude, ground speed. Input fields for the entry of co-variate data such as cloud cover, snow cover, alternate species, and habitat type are also provided.

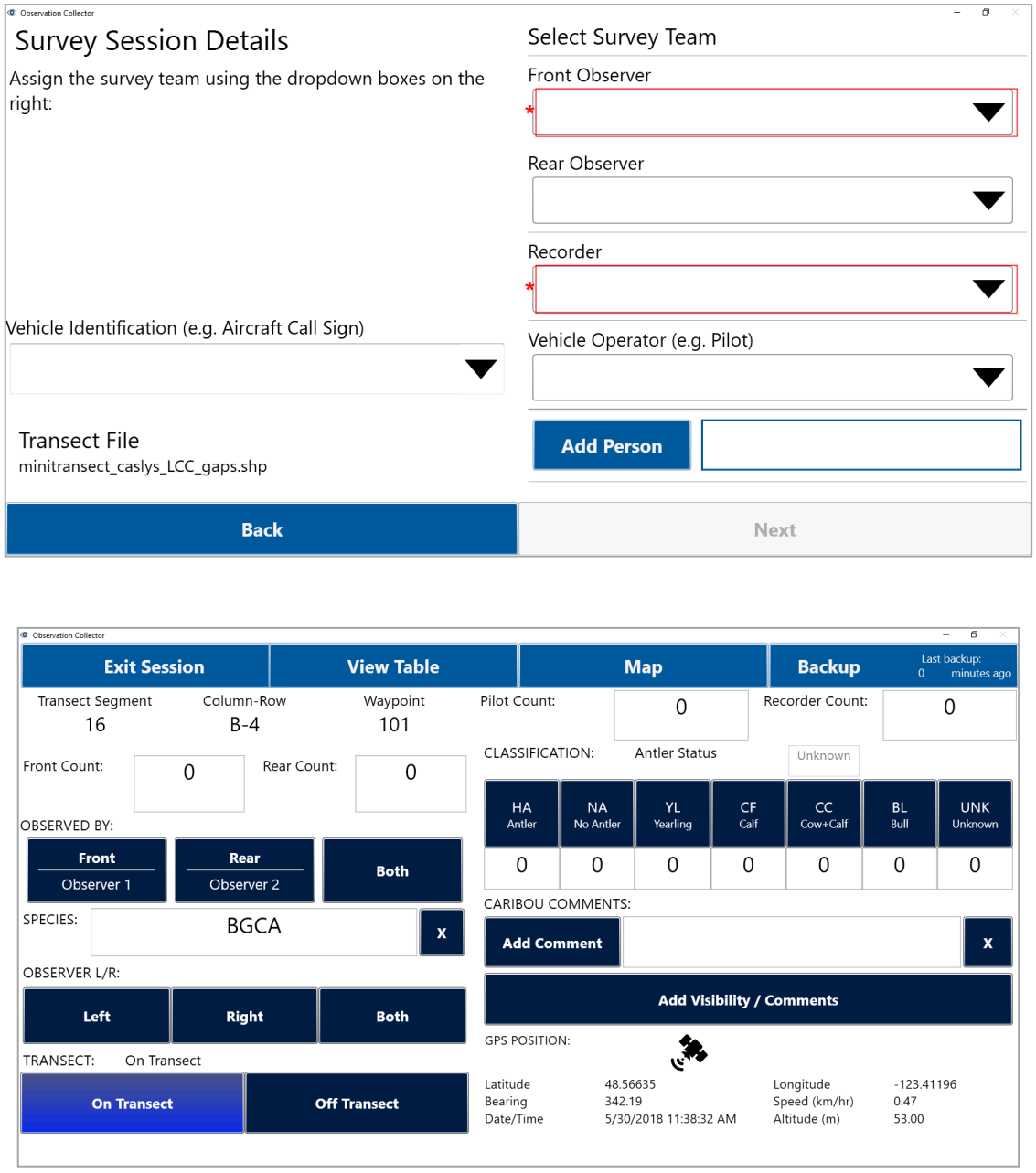


Figure 14. The data entry screens of the AWS-OC tablet interface used during the June 2023 Beverly mainland migratory barren-ground caribou abundance survey. Screen shots include the s urvey session details (Top), and primary data collection display (Bottom).

4.0 RESULTS & DISCUSSION

The geographic distribution of calving caribou observed during the June 2023 Beverly calving-ground abundance survey was atypical when compared to June reconnaissance and abundance surveys flown between 2008 and 2018. Between June 2008 and 2018, Beverly caribou cows had arrived onto the Queen Maud Gulf ACCA, between the Ellice River east to McNaughton River near Adelaide Peninsula, across a much larger geographic area (**Figure 15**). This larger area of occupation lead to much lower relative densities across the calving ground. Densities encountered over these survey years were consistently lower than the 15 to 20 caribou/km^2 threshold that would trigger the need for photographic counts. As a result, and coupled with a confirmed declining trend up to 2018, Beverly herd reconnaissance and abundance surveys have been set up as visual surveys since 2008 (Campbell et al. 2012; Campbell et al. 2019). However, in June 2023 densities of caribou were significantly higher within high density strata. During the 2023 June survey effort, unseasonably high air temperatures and an associated early spring melt of the QMG lowland snow pack and river and lake ice covers, caused Rivers and creeks within the calving grounds to surge, creating high-water volumes and dangerous currents, causing rivers and creeks to overflow their banks. These high volumes of fast-moving water created an effective barrier to the movement of migrating calving caribou cows onto their known annual core calving areas in early June, with many caribou clumping up into high-density groups within the forks of surging creeks and rivers. This reduction in the spread of calving caribou into their extended annual calving extents in June 2023 is also represented in the lower overall area of the known calving extents (an estimated 30% reduction) occupied by calving caribou when compared to June 2018 and prior calving extents. This situation was

moderated on or about June 14th and 15th as the water levels started to drop. This clumping posed counting problems, whereby some groups were too large and dense (in some instances in excess of 100 caribou/km^2) to accurately count using visual methods alone. Though a photo plane was kept on standby for such events, the aircraft unavailable due to mechanical issues. With a steady reduction in river water levels as the survey progressed past the reconnaissance phase, and the possibility of the movement of calving caribou out of developed strata as water levels dropped, we were unable to wait for the mechanical issue to be resolved so proceeded with a video method to specifically assess the high-density clusters of caribou. The collection and counting of video data represented a novel approach, requiring the development of new spatial and quantitative methods both in the field and post-survey, to produce accurate counts.

The start of the reconnaissance survey was initially scheduled for June 6th but was postponed to June 8th due to low cloud and fog in the survey area over June 6th and 7th (**Table 3**). The earlier proposed start of the reconnaissance survey compared to the June 2011 and 2018 surveys is thought to have been a result of the earlier snowmelt in June 2023. The June 2023 calving grounds in the vicinity of the Armark River, approximately 30 kilometers east of the Simpson River and within the center of the Beverly annual concentrated calving area, was relatively snow free in early June 2023 when compared to the same general area in mid-June 2011 (**Figure 16**).

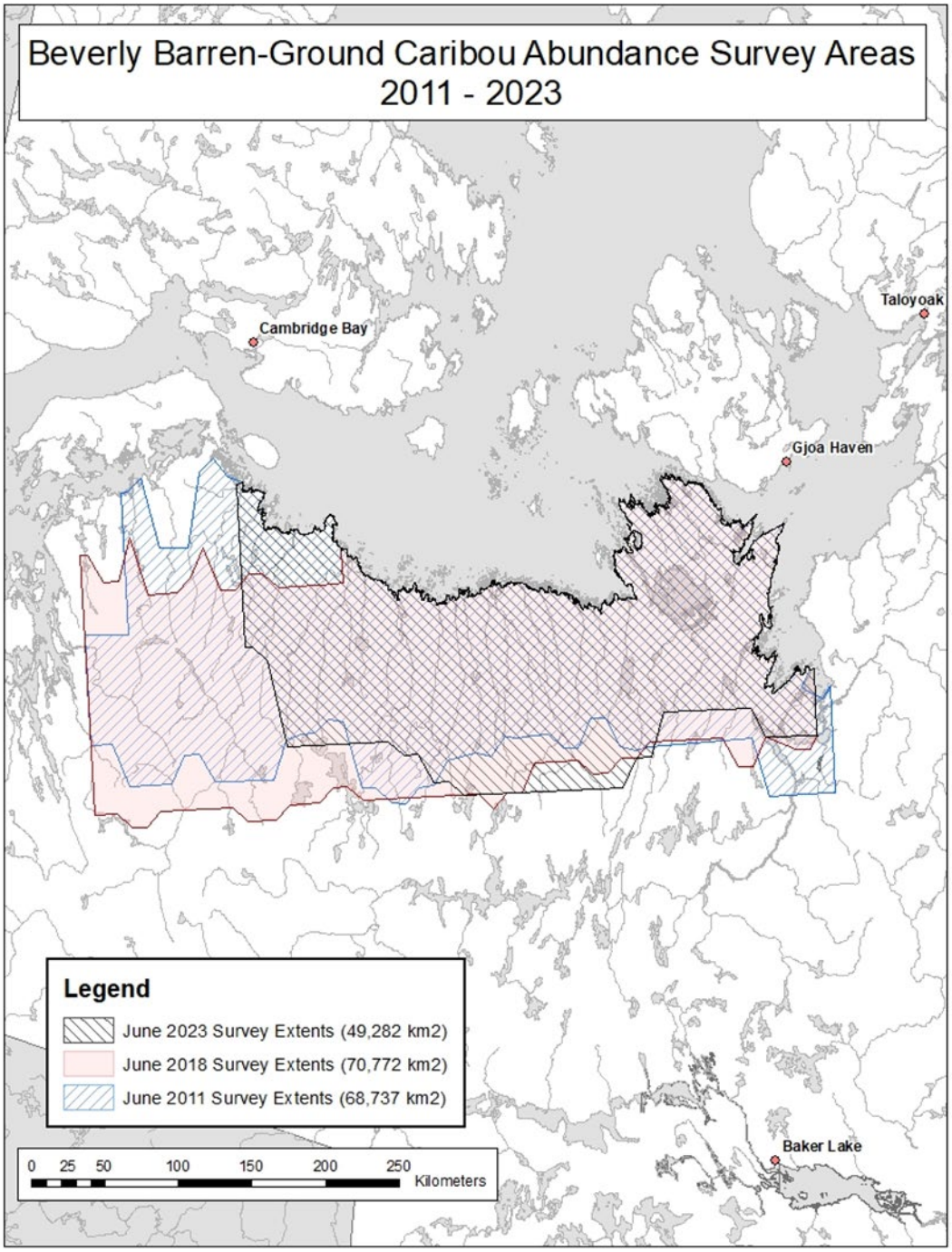


Figure 15. Beverly Herd Reconnaissance survey extents for the June 2011, 2018, and 2023 Beverly Herd calving-ground surveys (June 2023 survey extents in black).



Figure 16. The Queen Maud Gulf Lowlands in the vicinity of Armark River June 12th, 2011 (Above), and in the same general vicinity June 8th, 2023 (Below).

4.1 RECONNAISSANCE SURVEY

Reconnaissance flights occurred from June 8th to June 12th 2023 at which time large aggregations of caribou were observed to the southwest of the Adelaide Peninsula. Recon crews observed high water levels and fast currents in the Simpson River and associated creeks and smaller rivers within the watershed, due to the early spring melt and early breakup. These high-water conditions created a barrier to the late spring migration of calving cows east of the Simpson River, funneling caribou between large river systems and restricting movement to the more northern extents of their annual calving grounds (**Figure 17**).

Reconnaissance data was summarized by 10 km segments revealing many segments with high densities and 6 segments with extremely high densities (>50 caribou/km^2) (**Figure 18**). Strata development was achieved by first overlaying strata boundaries over reconnaissance segment based relative densities, and secondly by adjusting based on recon summary composition observations (**Figure 19**). Two strata (Very-high-density-east and Very-high-density-west) had segments with densities equal to or exceeding 50 caribou/km^2 with the majority of caribou being breeding females. Other strata, notably to the southwest, had high densities, however, the majority of caribou in these strata were bulls and non-breeding caribou (**Figure 20**). Remaining strata had medium to low densities of caribou and were delineated accordingly. Inspection of segment composition revealed that the Very-high-density strata dominantly contained breeding females with mixed breeding females and non-breeders in most remaining medium and low-density strata.

4.2 ALLOCATION OF EFFORT

Strata effort was defined based on survey logistics and relative densities of caribou within strata (**Table 4**). The total number of transect kilometers was restricted to 5,000 kilometers (km) based on survey logistics. An additional constraint was ensuring that the Very-high-density strata could be flown in less than two survey days given the increasing risk of high level of movement based on the movements of collared caribou in the area. The highest effort (km) was given to the two Very-high-density strata which also had the highest concentration of breeding females. Effort to the remaining strata (that contained a lower proportion of breeding females) was set to coverage levels of approximately 15% to accommodate higher effort within the Very-high-density strata. Two additional Very-low-density strata based on reconnaissance flights, were added for reference but were not re-flown due to the lack of evidence of movement within these strata apparent from composition observations made concurrent with abundance survey observations. The final strata dimensions are shown in **Table 5**. The Very-high-density-east stratum was further split into two strata, Very-high-density-east-1 and Very-high-density-east-2, to accommodate a change in aircraft navigation systems and revisions to the Very-high-density-east strata to ensure it could be flown in a single survey day.

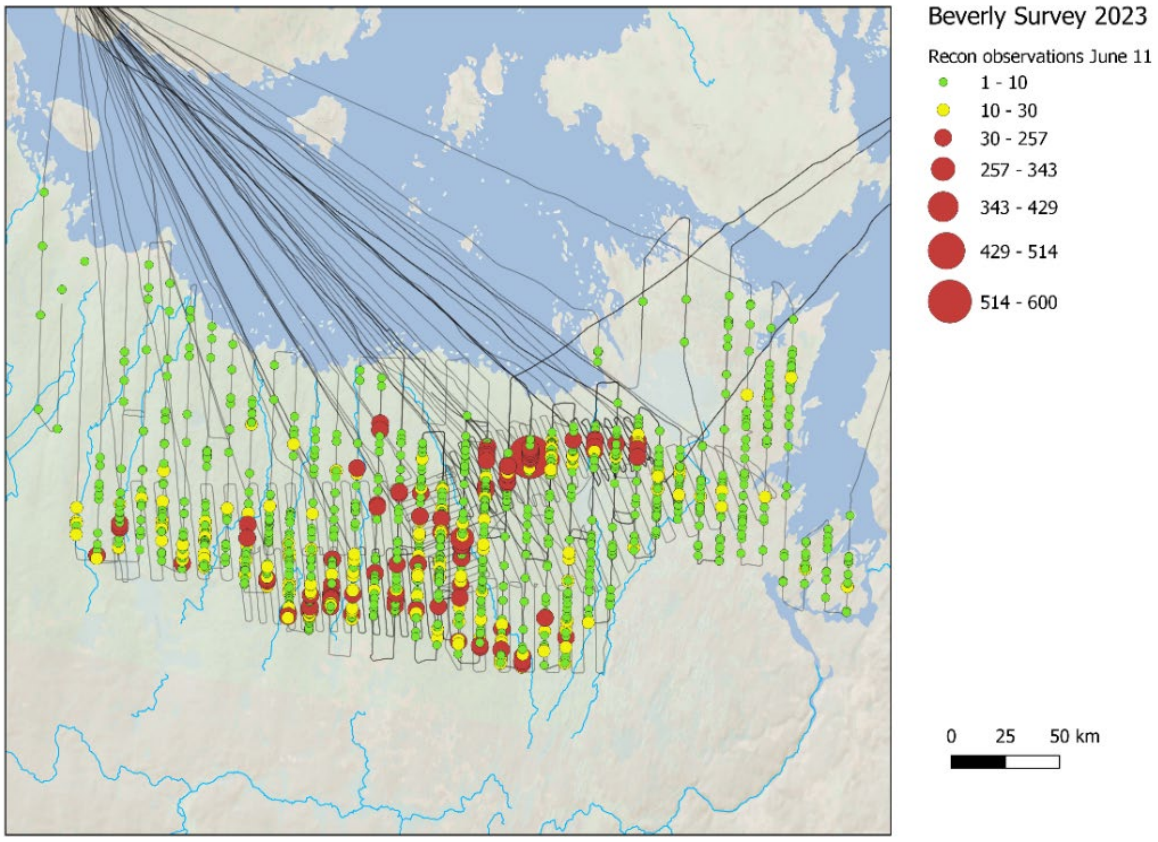


Figure 17 Flight lines and reconnaissance observations. Observations are color and size coded based on the number of caribou observed.

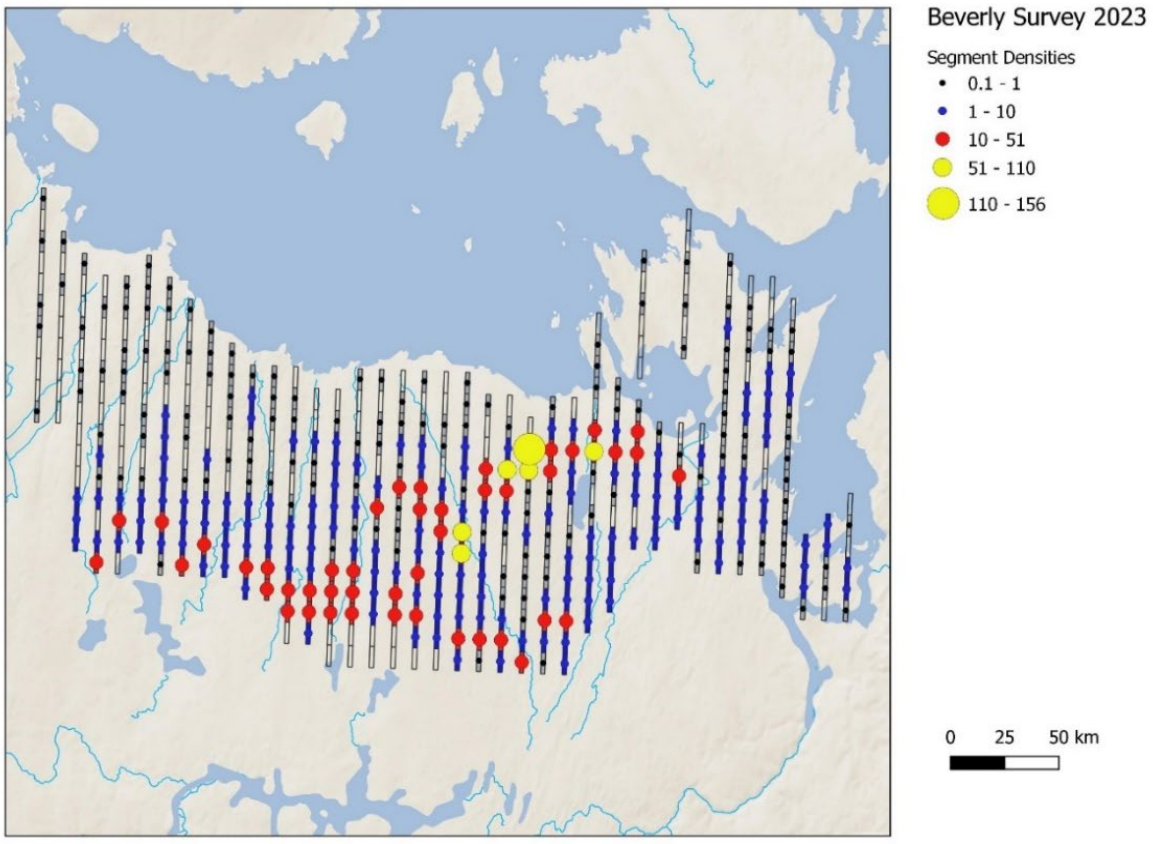


Figure 18. Summary of reconnaissance survey segments.

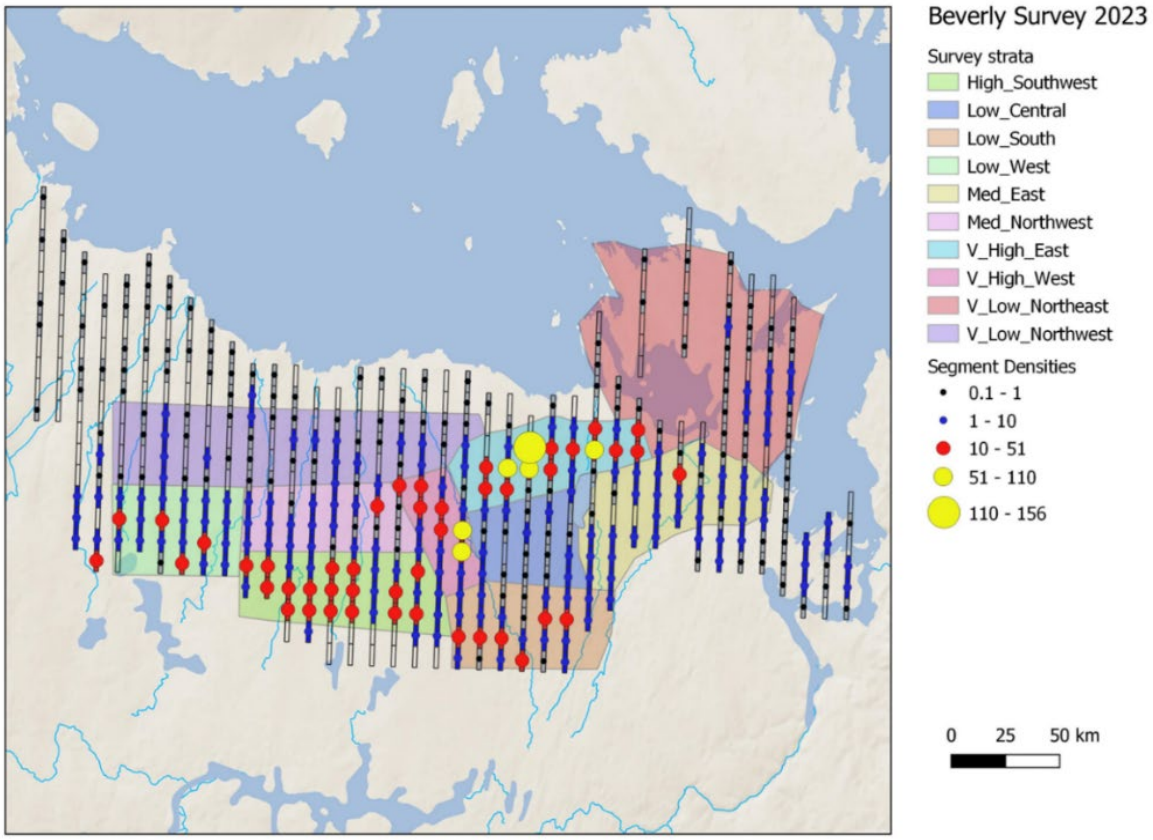


Figure 19. Summary of reconnaissance survey segment density with developed abundance strata overlaid.

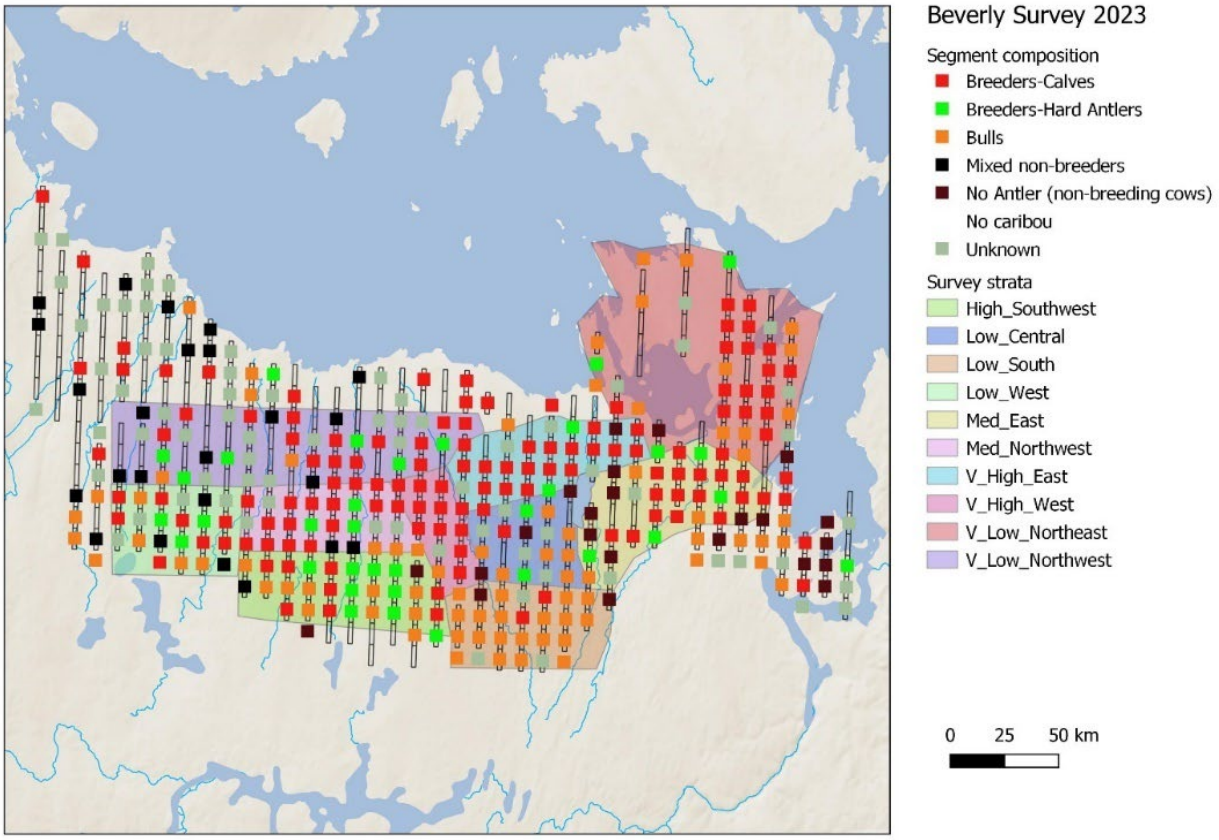


Figure 20. Summary of reconnaissance survey segment composition.

Table 4. Allocation of effort for strata. Allocation was based on Density/abundance or variation CV in density/abundance. Allocation lines were adjusted to ensure adequate coverage for all strata.

Strata	Dimensions		Recon Density		Allocation (transects)				
	baseline	Average Length	D	cv(D)	Using CV(D)	Using D	Adjusted	km	coverage
V_High_East	89	31.7	24.63	0.24	59	45	36	1066	0.30
V_High_West	57	24.0	20.17	0.35	25	27	19	439	0.25
High_Southwest	96	34.1	14.93	0.11	19	28	30	816	0.25
Med_East	82	42.1	2.48	0.22	5	4	18	639	0.15
Low_South	56	45.3	7.09	0.06	2	7	14	446	0.14
Med_Northwest	80	30.4	4.61	0.28	10	8	20	428	0.19
Low_Central	49	35.2	1.15	0.28	1	1	10	374	0.16
Low_West	60	42.0	5.58	0.15	5	6	12	468	0.15

Table 5. Strata dimensions for the June 2023 Beverly calving-ground survey.

Strata	Area (km^2)	# of transects	Average transect width (wi)	Baseline length	Total transect km	Total area surveyed (km^2)	Coverage
V_High_East1	1947.5	23	32.5	59.9	747.6	598.1	30.7%
V_High_East2	716.5	14	23.2	30.8	325.3	260.3	36.3%
V_High_West	1376.8	19	23.1	59.7	438.4	350.7	25.5%
High_Southwest	3236.7	30	33.5	96.7	1004.6	803.7	24.8%
Med_East	3416.9	18	35.5	96.2	639.5	511.6	15.0%
Low_South	2822.0	14	34.6	81.6	484.0	387.2	13.7%
Med_Northwest	2425.8	20	28.4	85.5	567.4	453.9	18.7%
Low_West	2487.6	12	38.6	64.4	463.4	370.7	14.9%
V_Low_Northwest	5595.5	17	34.1	164.1	579.7	463.7	8.3%
V_Low_Northeast	8245.1	11	75.7	108.9	832.7	666.2	8.1%
Low_Central	1885.5	10	37.3	50.5	373.1	298.5	15.8%

4.3 ABUNDANCE SURVEY

Visual surveys were conducted in a second “abundance survey” phase, excluding all the Very Low-density strata and the Low-density South stratum (**Figure 21**) which used June 2023 reconnaissance data (**Figure 20**). High densities occurred in the two Very-high-density strata. A group of higher density caribou also occurred in the Low-density-central stratum, however, this group likely moved from the Very-high-density-east strata based on collar movements discussed later in this report.

4.3.1 Analysis of collar movements

Migration paths of collars reveal a north-eastern path for female caribou with most ending up in the two Very-high-density strata while bulls mainly occurred within southern strata (**Figure 22**). In contrast to previous abundance surveys, very little northern movement within the Beverly herds ACCA was observed in June 2023. The monitoring of caribou movements onto the calving-ground suggests that there was no defined period of reduced movements in June 2023 that would typically indicate the peak of calving (**Figure 23**). There was a drop in movement rates from June 2nd to 6th which corresponded to the approximate time when some caribou arrived on the main calving ground, which may explain the reduction in movement rates. We suggest that calving cows, stopped at swollen rivers and creeks during their eastward migration onto the calving-ground, explains the atypical movement rates during calving and the lack of movement cues typically representative of peak calving for this herd. Median movement rate of caribou varied from 7 to 10 kilometers per day from June 12 to 15 when sampling was occurring. During this time movement in the high-density areas was within the strata (**Figure 24**) with the exception of a burst of movement out of the Very-high-density-east strata into the Low-density-central stratum on June 15 when this area was flown. For this reason,

estimates from the Low-density-central stratum were not used in estimates of adult females due to likely double counting of caribou.

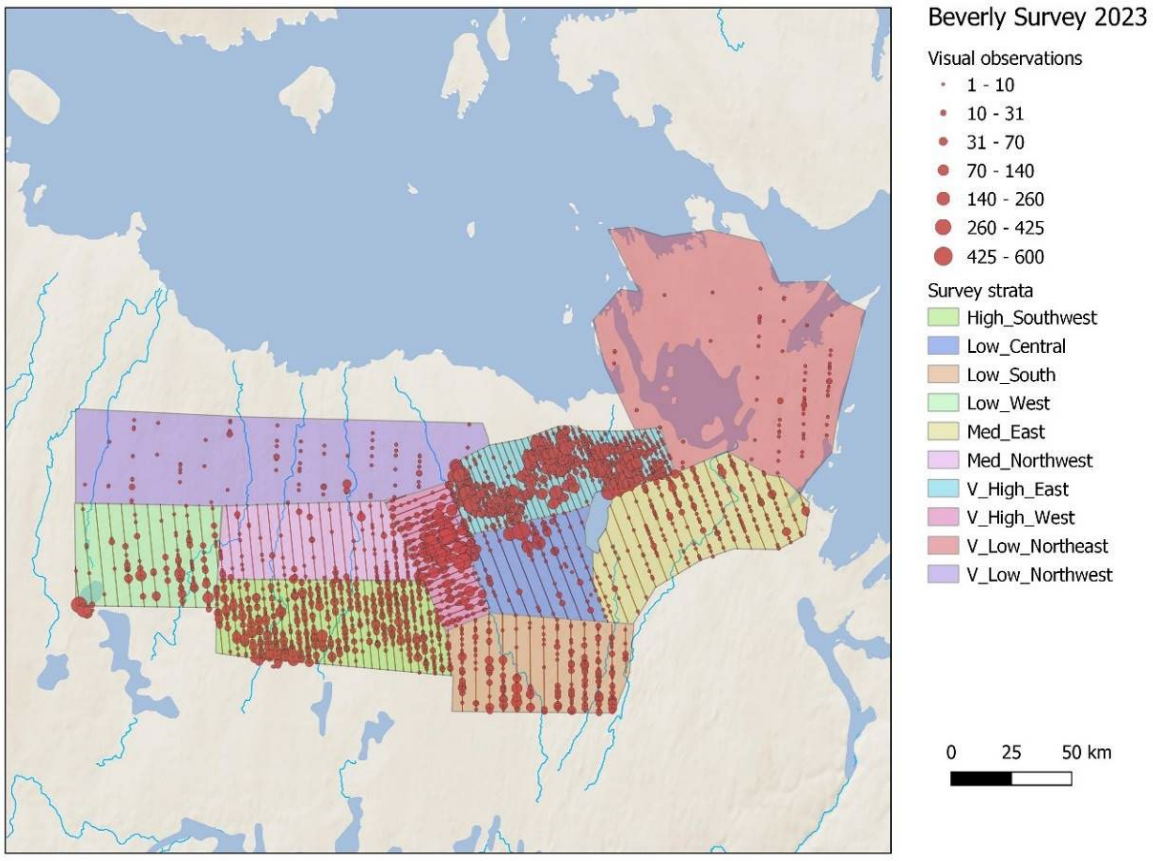


Figure 21. Visual survey transect lines and observations.

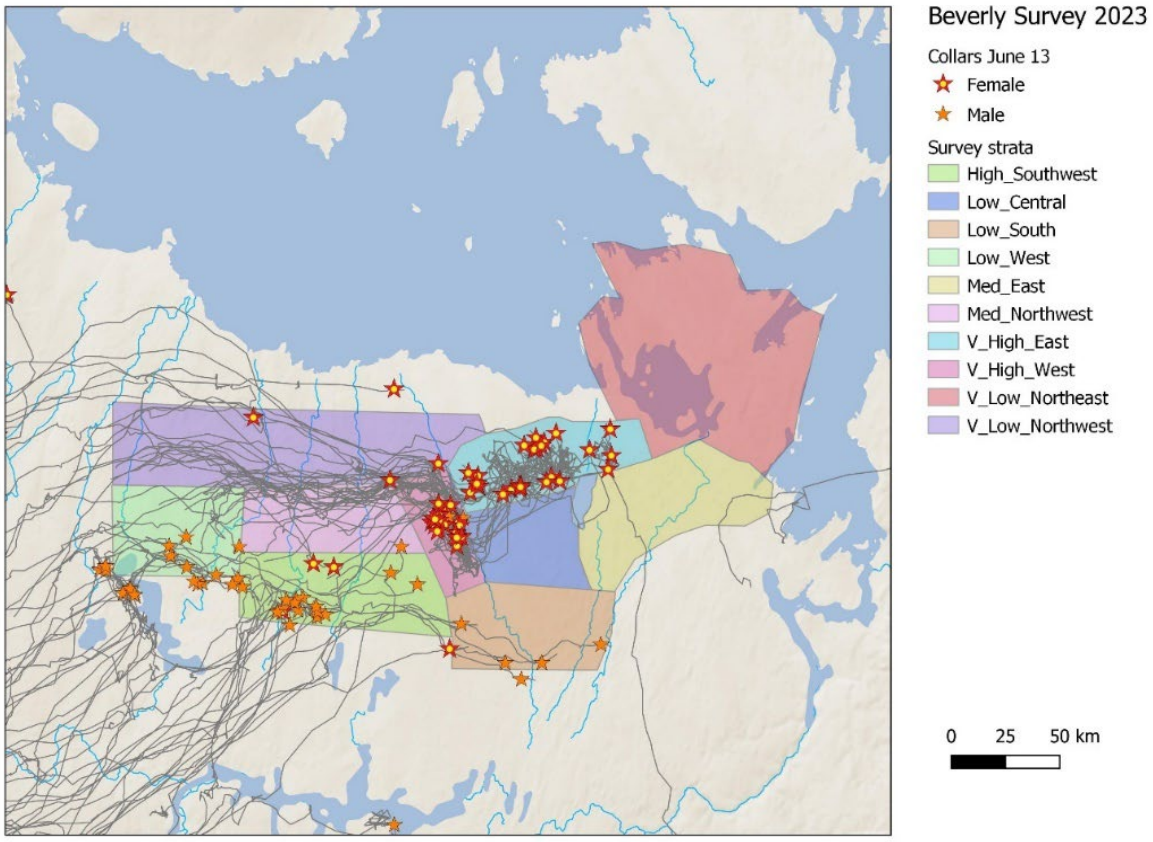


Figure 22. Collar migration paths up to June 13th when visual surveys occurred.

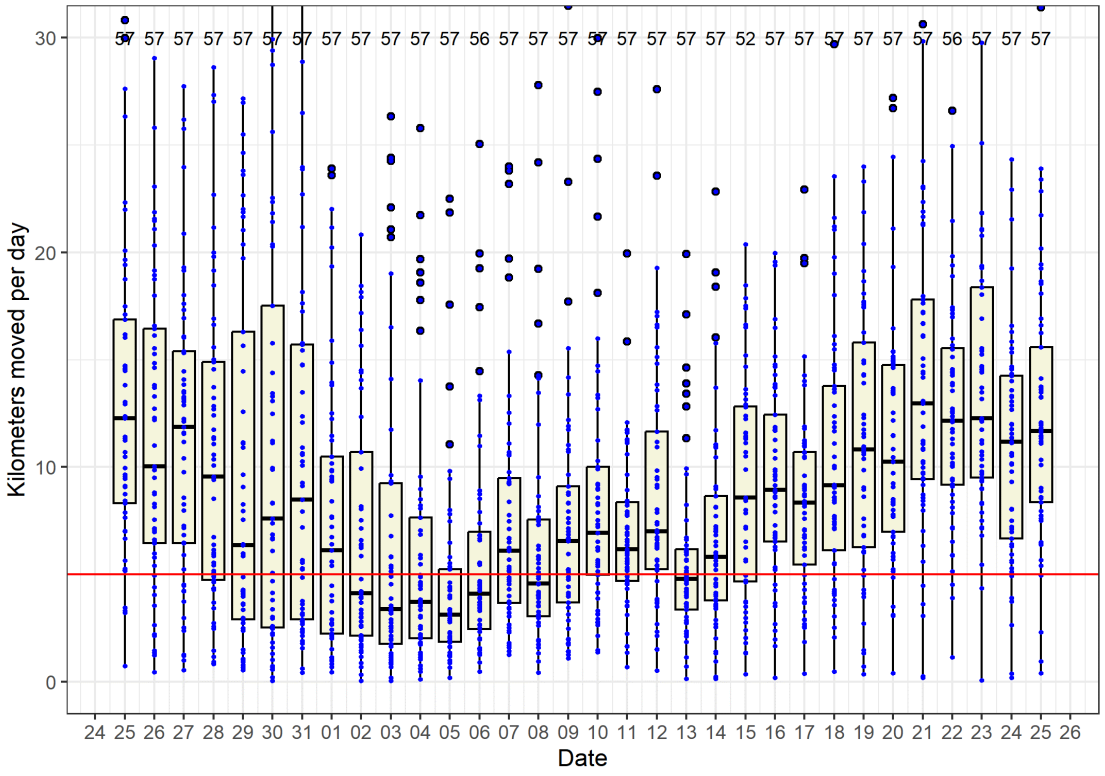


Figure 23. Movement rates of Beverly caribou during the June 2023 Beverly calving-ground survey.

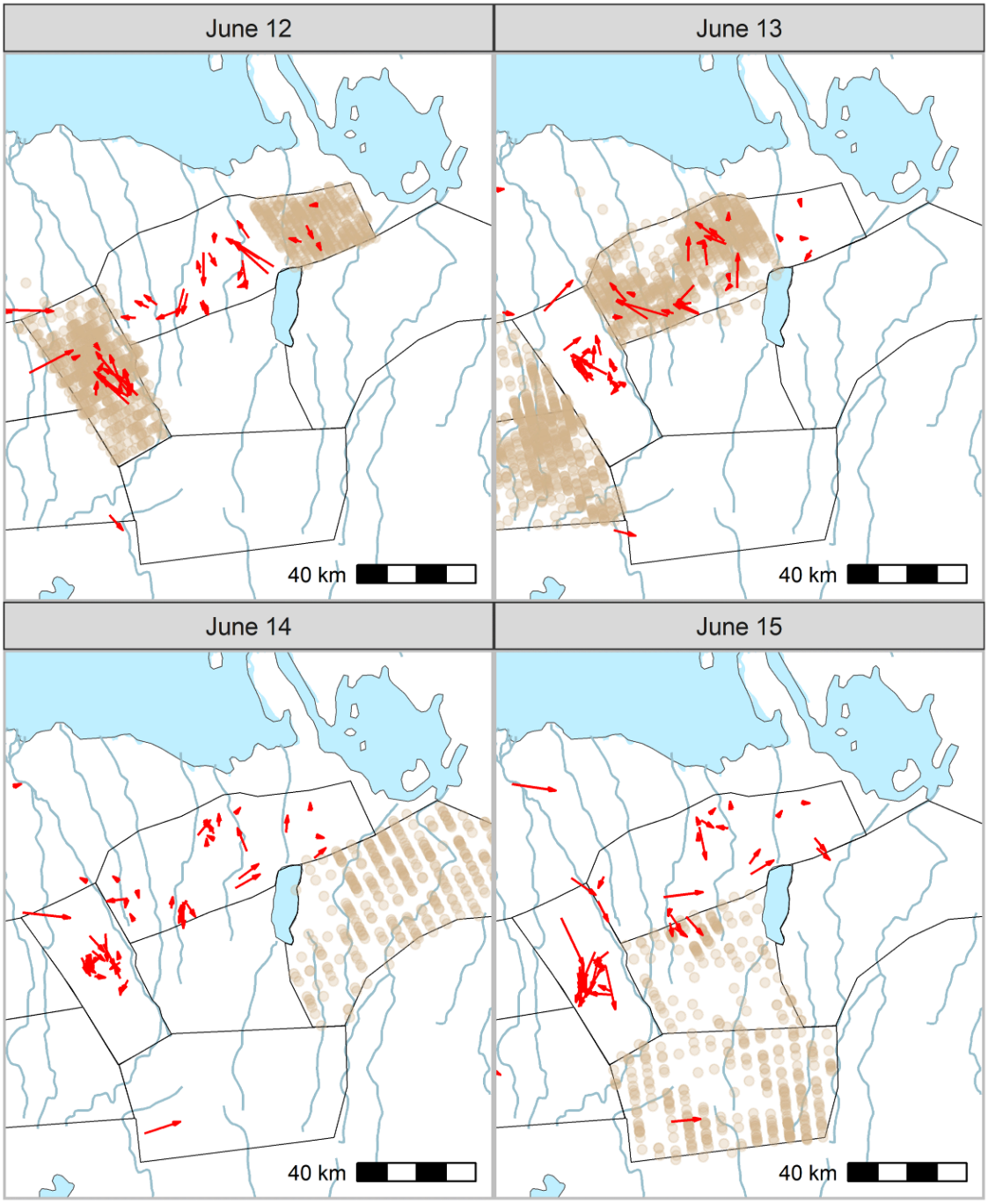


Figure 24. Directional daily movements of collared females relative to transects flown (brown circles) for the main days in which central high-density strata were flown. The direction of daily movement of caribou is indicated by arrows.

4.3.2 Video Assisted Counts in High-Density Strata

In total, 125 video/orthomosaic images of the three Very-high-density survey strata were taken during the survey (**Figure 25**). Of the 125 orthomosaics, 61, 18, and 44 occurred in the Very-high-density-east-1 (VHDE-1), Very-high-density-east-2 (VHDE-2), and Very-high-density-west strata respectively (VHDW). Comparison of visual observation group sizes and ortho-photo locations suggests that in excess of 98% of the larger groups seen on transect were within the orthomosaics.

During the review of visual and video caribou counts, waypoints from visual observations were counted as the same caribou as orthophotos based on an assumed distance of the waypoints from the edge of the orthophotos. Reconciliation of orthophotos and visual observations was considered across a range of buffer distances. In general, the buffer distance did not affect the general relationship between visually counted caribou, and caribou counted on photos (**Figure 26**). The difference in estimates between the two approaches varied by approximately (plus or minus) 5% across the range of buffer distances with visual counts increasing with increasing buffer distances. We chose a buffer distance of 1,000 meters which balanced inclusion and exclusion of visual counts from each of the orthophotos. At a survey speed of 160 kilometers per hour, 1,000 meters would represent approximately 20 seconds of time. Therefore, using 1,000-meter buffers assumes that recorders recorded observations within approximately 20 seconds of the end of the video recording. When 1,000 meters was used as the buffer distance the mean number of locations observations within a photo was 3.7 (SD = 2.7; min = 1, max = 16, n=121). The mean group size of caribou counted for each visual observation waypoint within the 1,000-meter buffer was 42.3 caribou (SD = 68.6; min = 1, max = 440, n = 450).

Caribou were counted on photos that were approximately within the 800-meter (400 meters per side) survey strip, however, pitch and movement of the plane affected the survey strip distance and thus the total area within photos within which caribou were counted. A comparison of visual and photocounts using all the caribou on the

photos suggested that visual counts were slightly higher when total counts were less than 100 but then became less than photo counts as higher numbers (densities) of caribou were encountered (**Figure 27**). When the 400-meter strip width was imposed (for each of the left and right-hand sides of the aircraft) (**Figure 28**), visual counts were higher than photo counts up to counts of 400 caribou. This result suggests that an additional factor influencing visual counts was inclusion and exclusion of caribou within the survey strip, suggesting that visual observers had difficulty maintaining their outer strip boundaries when caribou numbers were high. This result reflects the increased work load of an observer when high numbers of caribou have to be counted causing less attention to be directed to strip boundaries. This finding could also be due to factors such as pitch of the plane, difficulties in discerning proportions of groups within the strip, and short-term observer fatigue. We also note that this comparison assumes that sightability of caribou on the photos was similar to visual observations from the plane. Using the 400-meter strip-corrected photo counts reduced the count of caribou by 4,353 (approximately 13% lower) compared to uncorrected visual counts.

Comparison of visual surveys in 2011, 2018, and 2023 reveals large differences in the contribution of different group sizes to overall caribou counts (**Figure 29**). In 2011 and 2018, the majority (>95%) of caribou counted were in relatively small group sizes (<30 caribou). In contrast, the majority (70%) of caribou counted in 2023 were larger groups (>30 caribou) illustrating the large difference in densities observed during the 2023 survey and the necessity to augment visual counts with photo-counts. By design the majority of larger group sizes occurred within video/orthomosaic polygons, meaning that the more accurate and precise video-based counts were used instead of visual counts for these observations and for the final estimates of abundance.

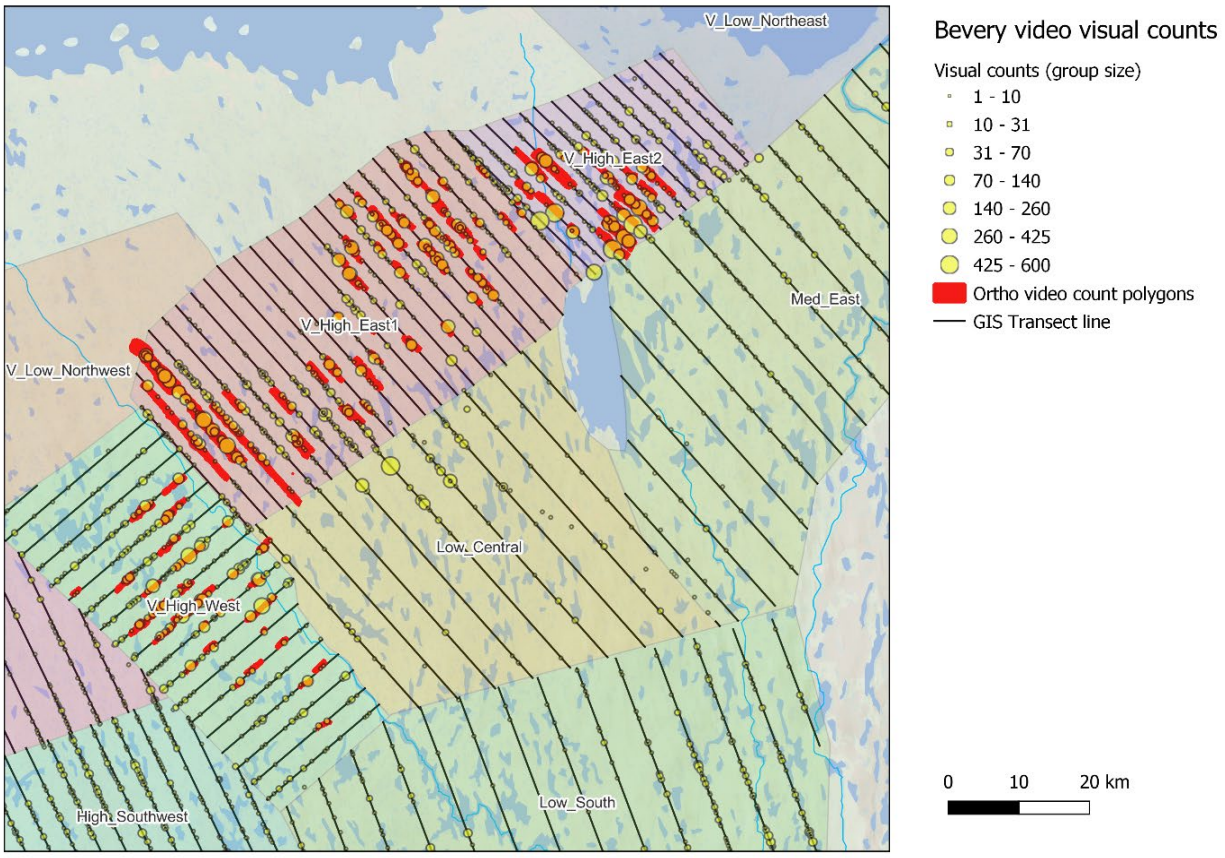


Figure 25. Coverage of ortho photos relative to visual observations in the Very-High-Density West and East strata.

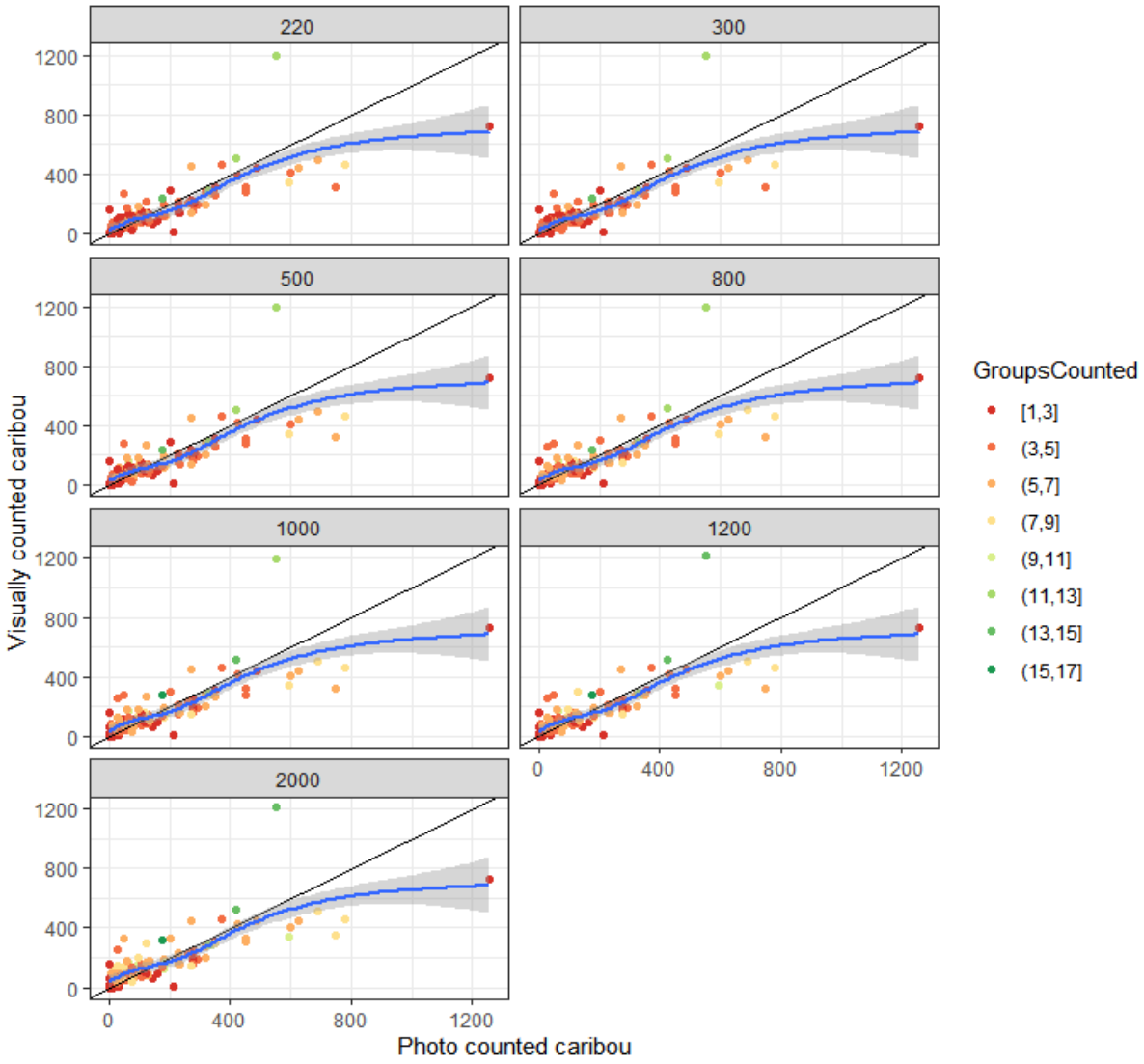


Figure 26. Relationship between visually counted caribou (assumed to be within the orthomosaics photo-video) and the number of caribou counted on each orthophoto. The number of visual group waypoints within each photo is also displayed by color of point.

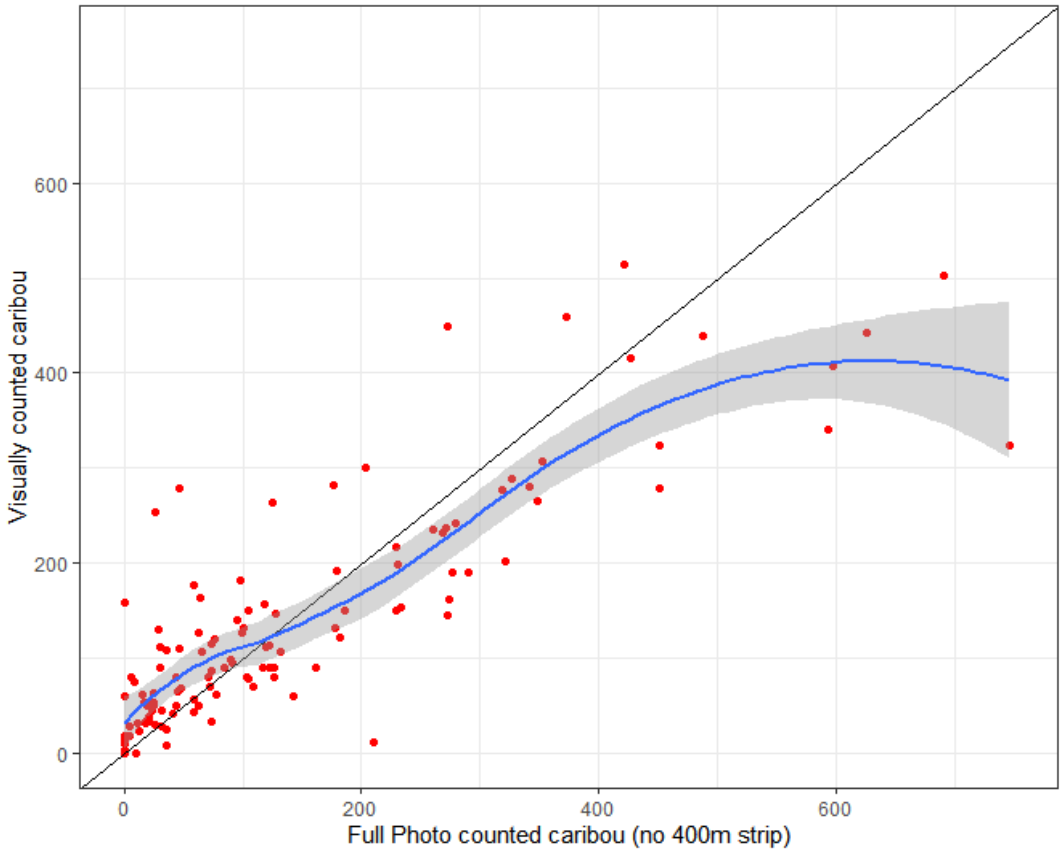


Figure 27. Relationship between caribou counted visually versus caribou counted on photos using all the caribou countable on photos with no strip width. A smoothed (blue line) is used to indicate dominant trends in the data set. Confidence limits are shown as shaded areas around the smoothed line.

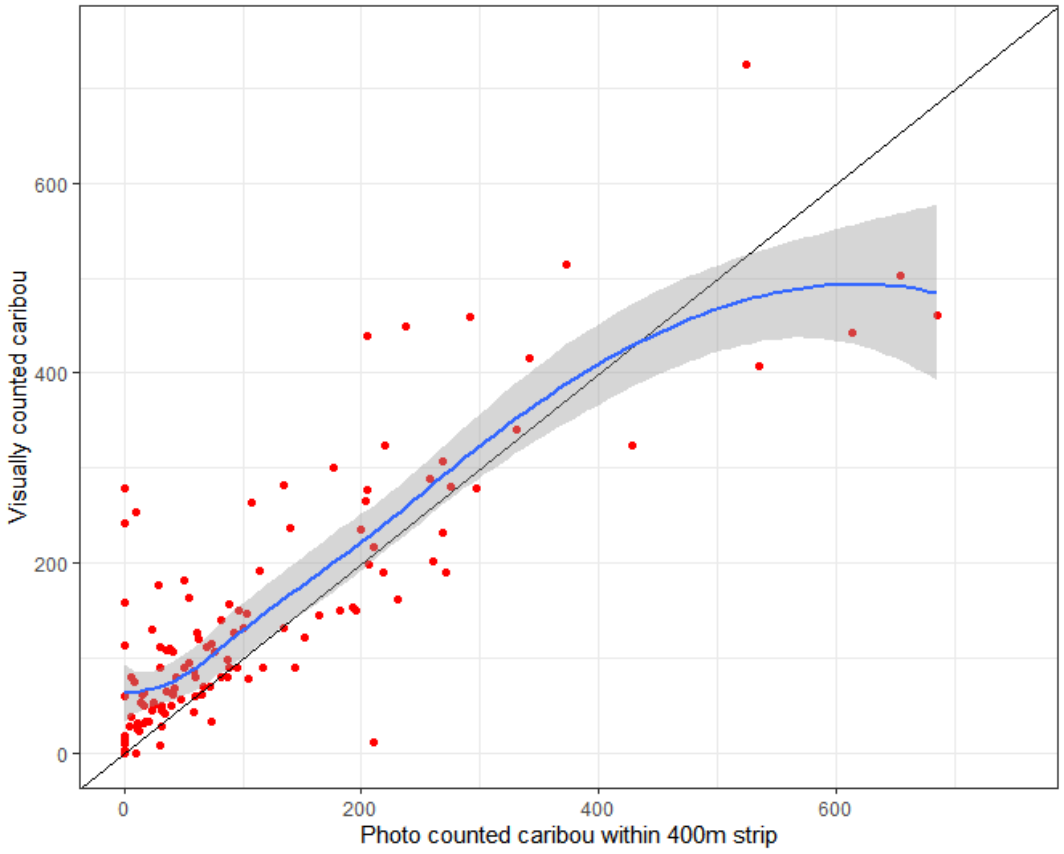


Figure 28. Relationship between caribou counted visually versus caribou counted on photos using all the caribou on photos with a 400-meter strip width applied.

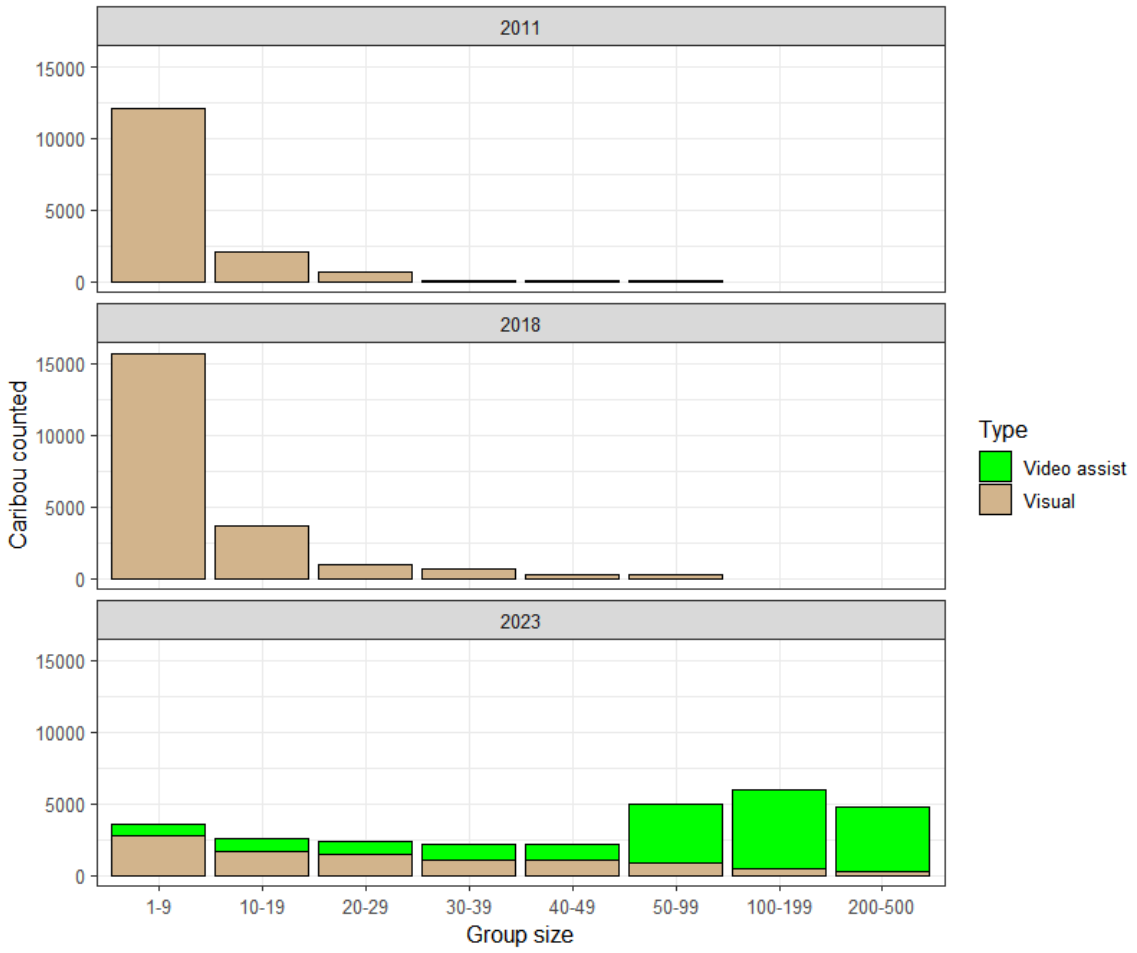


Figure 29. Comparison of contribution of group sizes observed to the total count of caribou on the 2011, 2018 and 2023 surveys. Sub-bars denote if visual counts were replaced in 2023 with video/orthomosaic counts.

4.3.3 Estimation of abundance using Jolly strip transect estimator with video-assisted counts

Abundance was initially estimated with the Jolly strip transect estimator. For the very-high-density strata, ortho-photo data was combined with visual counts (in areas where ortho photos were not used) to provide mutually exclusive estimates for each transect line. Transect densities were above 15 to 20 caribou per km^2 for many of the transect lines in the Very-high-density strata. **Figure 30** shows the proportion of counts in these strata that came from photo counts versus visual counts. For most transect lines with a density above 10, the majority of caribou counted came from the photos as opposed to visual counts.

Transect densities within the remaining survey strata were below 10 caribou per km^2 with the exception of the High-density-southwest (HDSW) and Low-density-south (LDS) strata (**Figure 31**). Higher densities in these strata were more due to bulls and non-breeders and therefore had less influence on adult female estimates. As noted earlier, higher densities in central lines of the Low-density-central (LDC) were likely due to a spillover of caribou from the VHDE-1 stratum with lower densities on other survey lines. For this reason, the LDC stratum was not used in the estimate of adult females due to likely double-counting of caribou on the central lines.

Estimates of abundance and density demonstrated higher densities, in excess of approximately 20 caribou per km^2, in all of the very-high-density strata (**Table 6**). In contrast, densities were 12 caribou per km^2 in the High-density-southwest strata and less than 9 caribou per km^2 in the rest of the strata. Of most interest were the estimates from the Very-high-density strata, within which, the use of video/ortho photo data reduced estimates from by approximately 14,000 caribou (17%) compared to visual count data. Estimates from other strata were derived using double observer methods as discussed in the next section.

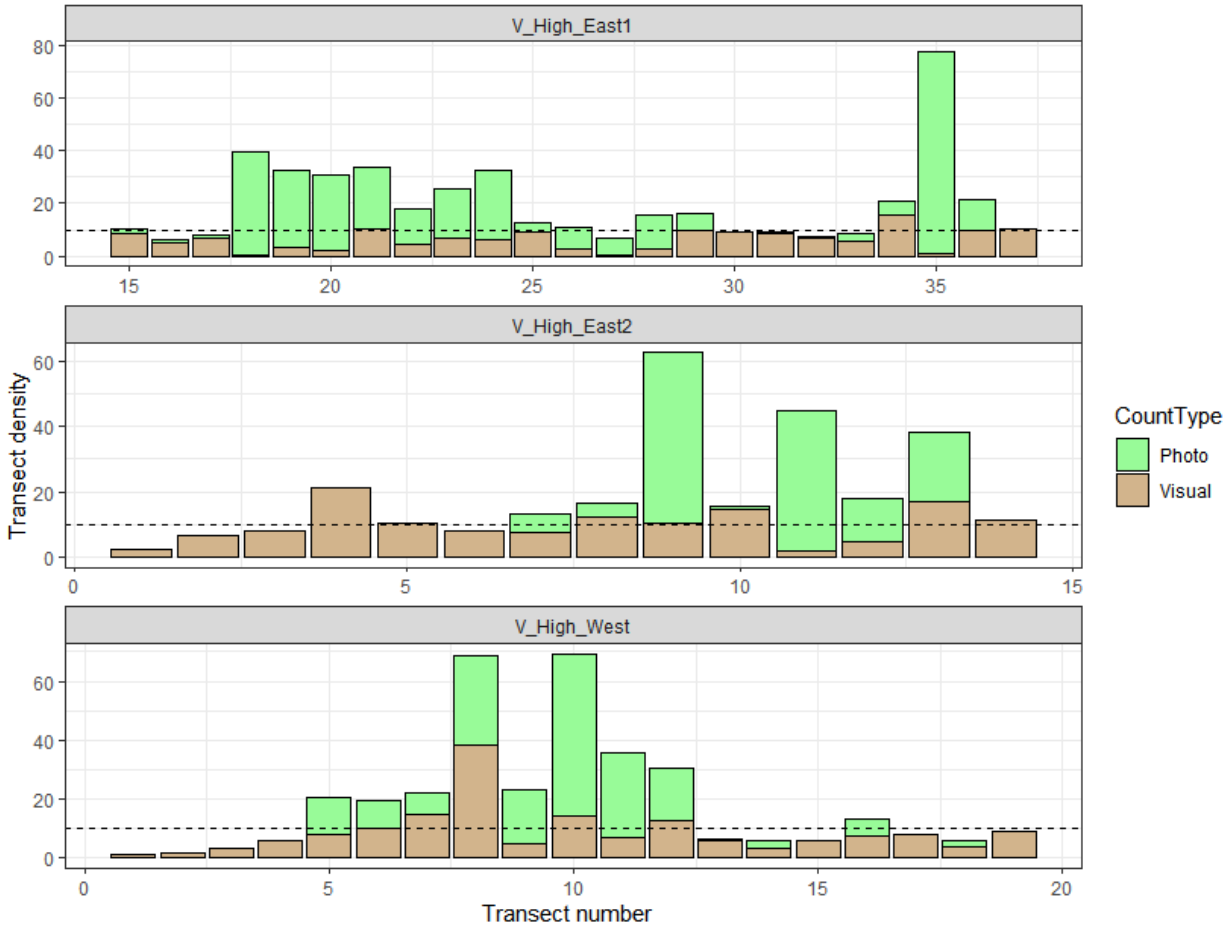


Figure 30. Transect densities for the very high strata with proportions of counts based on photo/orthomosaics or visual source delineated for each stratum.

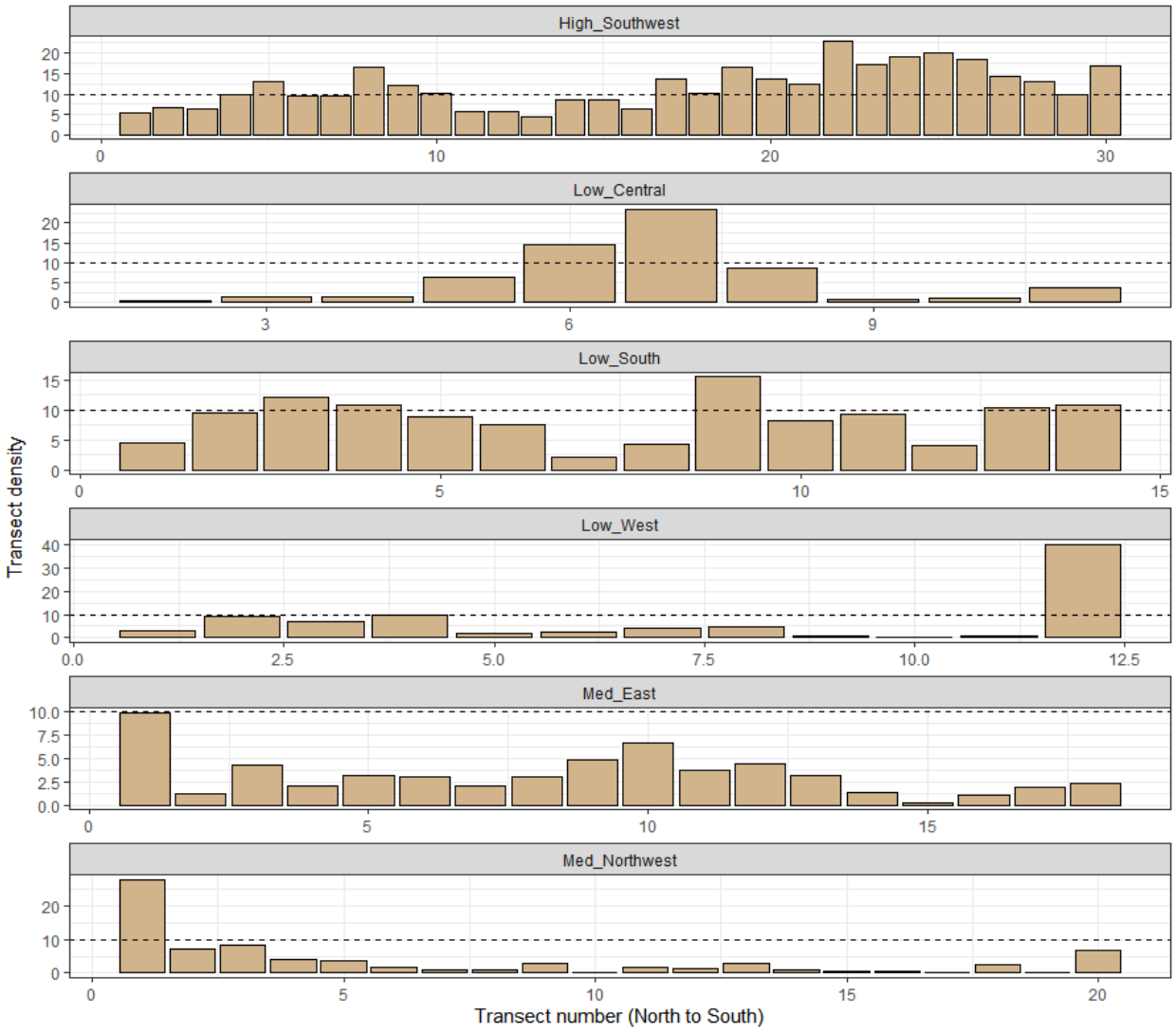


Figure 31. Transect densities (caribou/km^2) for visual strata. Note that most transects were well within visual observation limits (15 to 20 caribou/km^2).

Table 6. Estimates of density and abundance using the standard Jolly 2 strip transect estimator. Video-corrected data was used for High East1, High East2, and High West strata.

strata	Caribou counted	N	SE	Conf. Limit		CV	Density (caribou/km^2)
Video-assisted counts							
V_High_East1	12,075	39,319	5251.9	29,842	51,805	0.134	20.2
V_High_East2	5,444	14,988	2685.0	10,209	22,003	0.179	20.9
V_High_West	6,929	27,198	5758.9	17,517	42,230	0.212	19.8
Visual counts							
High_Southwest	9,596	38,645	2473.5	33,908	44,044	0.064	11.9
Med_East	1,533	10,239	1292.3	7,853	13,349	0.126	3.0
Low_South	3,272	23,847	2720.9	18,653	30,489	0.114	8.5
Med_Northwest	1,090	5,825	1161.3	3,854	8,806	0.199	2.4
Low_West	1,931	12,957	3843.4	6,838	24,552	0.297	5.2
V_Low_Northwest	483	5,828	1182.2	3,807	8,920	0.203	1.0
V_Low_Northeast	324	4,010	1323.8	1,959	8,211	0.330	0.5
Low_Central	2,045	12,917	4405.5	6,099	27,356	0.341	6.9

4.3.4 Double observer modelling of sightability

Double observer modelling was conducted to generate estimates of abundance for all strata except VHDE-1, VHDE-2, and VHDW strata where video-assisted counting was employed. It was not possible to use visual methods with video-assisted counting since video observations needed to be pooled in orthophoto areas.

Data was initially summarized based on 11 observer pairs who switched front (primary) and rear (secondary) places intermittently during the survey. Of 5,766 observations, 5,348 were for paired observers with only 418 (7%) occurring for non-paired observers. The non-paired observers were pooled into observer pair 12. Summaries from all observer pairs suggest high detection probabilities with single observer detection probabilities greater than 0.9 and double observer probabilities approaching 1 (**Table 7** and **Figure 32**). Frequencies of observation by group size suggests most cases of single observer sightings occurred when group sizes were lower as shown in previous studies (**Figure 33**).

Summaries of snow and cloud cover indicate that the majority of observations occurred with 0 snow cover and 0 cloud cover which illustrates good sighting conditions during the survey (**Figure 34**). For model selection, cloud and snow cover were considered as categorical (i.e. one parameter (minus 1) estimated for each category) and continuous (assumes a linear relationship between cloud or snow cover and sighting probabilities). In general, the categorical form was more supported. In addition, the effect of observers and group size, whereby detectability increased with increasing group size, was also suggested as indicated by the most supported model (Model 1, **Table 8**). Another way to view model predictions is by observer pair (**Figure 35**), which demonstrates that some observer pairs had higher sighting probabilities than others when group size was lower but, in most cases, sighting probabilities were close to 1 when groups sizes were larger.

Double observer estimates were only produced for visual strata that did not have video-assisted counts (**Table 9**). Using these most supported models produced a

total estimate of 101,767 caribou with reasonable precision (CV=6.6%). In contrast, estimates using just the Jolly N strip transect estimator were 101,352 (CV=6.0%) which was only 415 caribou lower than the double observer estimate. The similarity in estimates was due to the high sightability conditions, with most groups over five caribou being observed by both observers.

Table 7. Summary for pooled pairs given in previous table. Naïve single (p1x) and double (p2x) probabilities are given.

Observer pair	Frequencies				Single observer probabilities		Double observer Probabilities	
	Front	Rear	Both	Data recorder	No data recorder	Data recorder	No data recorder	Data recorder
1	9	20	333	17	0.94	0.90	1.00	0.99
2	5	5	887	20	0.99	0.97	1.00	1.00
3	4	4	268	1	0.99	0.98	1.00	1.00
4	12	25	683	6	0.97	0.96	1.00	1.00
5	28	34	490	7	0.94	0.93	1.00	0.99
6	4	14	190	10	0.93	0.89	1.00	0.99
7	11	15	202	14	0.93	0.88	1.00	0.99
8	86	89	1007	58	0.92	0.88	0.99	0.99
9	90	35	174	21	0.88	0.83	0.99	0.97
10	4	9	337	14	0.97	0.94	1.00	1.00
11	4	5	96	1	0.95	0.94	1.00	1.00
12	1	17	394	6	0.96	0.94	1.00	1.00

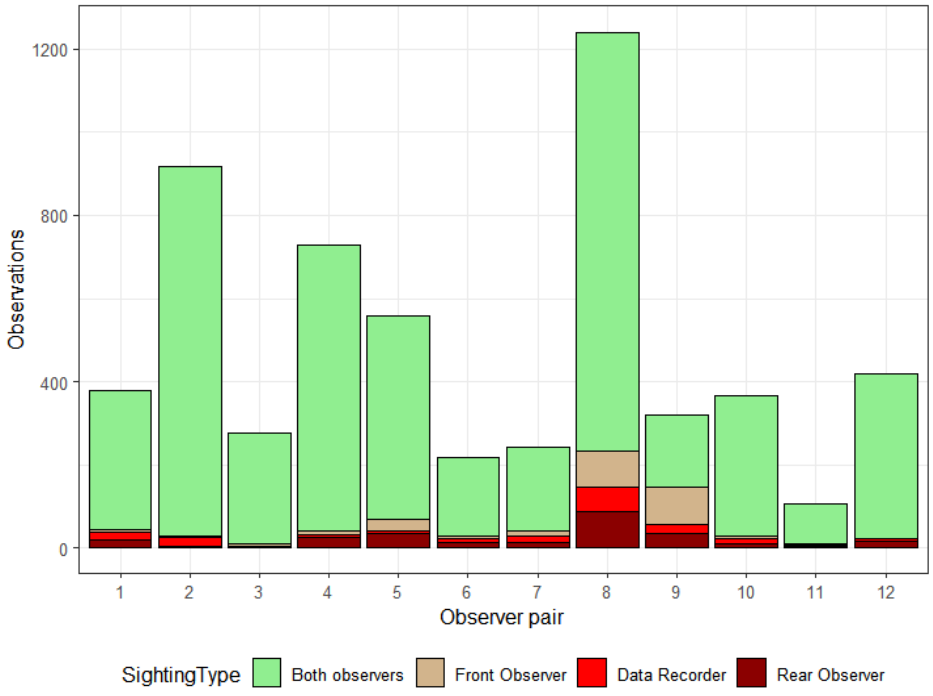


Figure 32. Sighting types by observer pair.

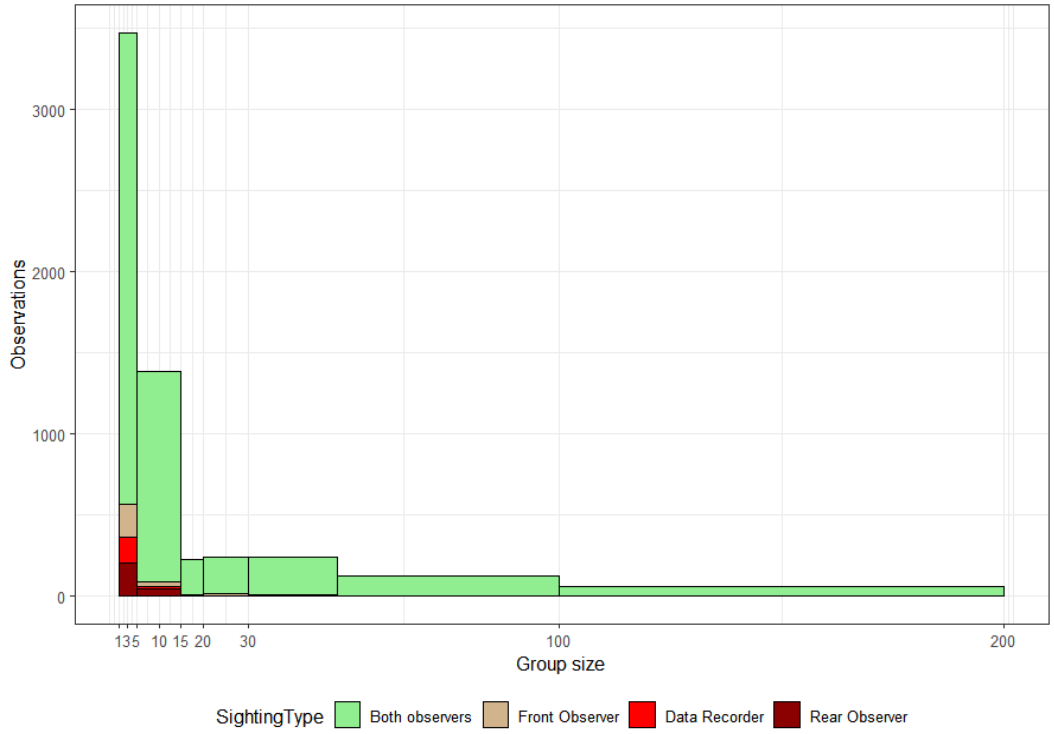


Figure 33. Frequencies of observation by group size as a function of observation type (B=Both, F=Front, R=Rear).

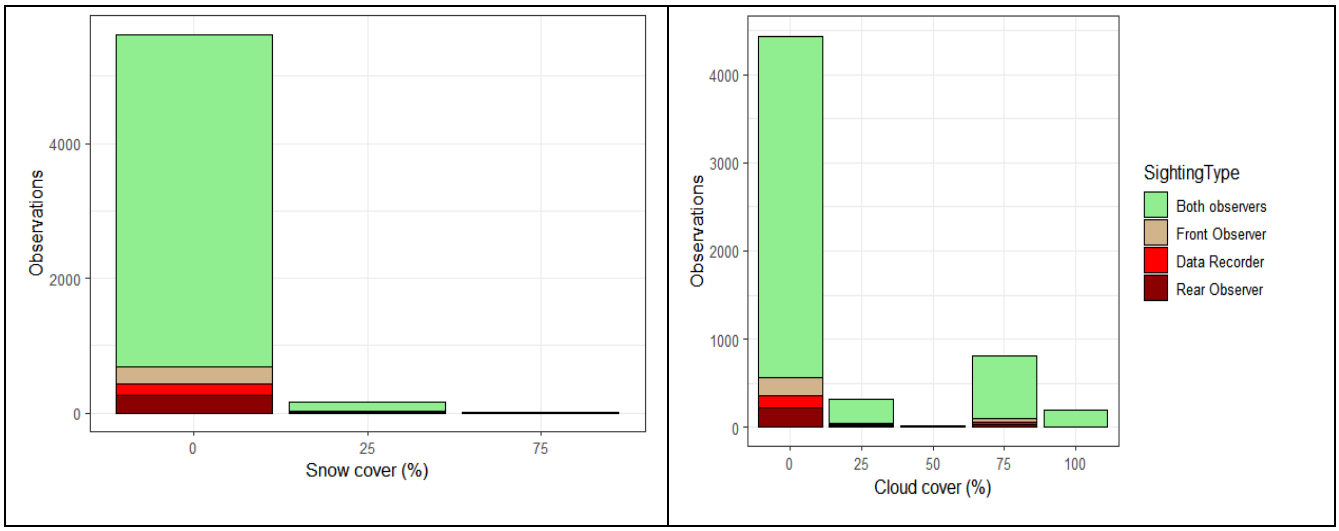


Figure 34. Observation frequencies by snow and cloud cover as a function of observation type (B=Both, F=Front, R=Rear).

Table 8. Double observer model selection results. Sample size adjusted Akaike Information Criterion (AICc), the difference in AICc between the most supported model for each model (ΔAIC_c), AIC_c weight (w_i), number of model parameters (K), and deviance is given.

No	Model	AIC_c	ΔAIC_c	w_i	K	LL
1	logsize + obs + cloudF	2877.77	0.00	0.91	17	-1420.1
2	logsize + obs	2882.48	4.72	0.09	13	-1427.2
3	logsize + cloudF	2971.64	93.88	0.00	6	-1479.6
4	logsize + snowc + cloudc	2987.39	109.62	0.00	4	-1489.6
5	logsize + cloudc	2989.05	111.28	0.00	3	-1491.5
6	logsize	2998.87	121.10	0.00	2	-1497.4
7	logsize + snowc	3000.28	122.51	0.00	3	-1497.1
8	logsize + PhaseF	3000.92	123.15	0.00	3	-1497.4
9	PhaseF * logsize	3001.88	124.11	0.00	4	-1496.8
10	obs	3027.07	149.30	0.00	12	-1500.6
11	size	3109.50	231.73	0.00	2	-1552.7
12	cloudF	3128.93	251.16	0.00	5	-1559.3
13	StrataF	3138.85	261.08	0.00	12	-1556.5
14	snowF + cloudc	3142.22	264.45	0.00	4	-1567.0
15	cloudc	3146.39	268.62	0.00	2	-1571.2
16	constant	3150.46	272.69	0.00	1	-1574.2
17	PhaseF	3150.90	273.13	0.00	2	-1573.4
18	snowF	3151.07	273.30	0.00	3	-1572.5
19	snowc	3151.37	273.61	0.00	2	-1573.7

Table 9. Double observer visual abundance estimates from Model 1 (**Table 7**) for strata excluding VHD-1, VHDE-2, and VHDW that were estimated using video counts.

Strata	Counted	N	SE	Conf. Limit		CV
V_Low_Northeast	324	4,032	759.8	2,553	6,368	0.188
High_Southwest	9,616	38,825	1853.1	35,072	42,980	0.048
V_Low_Northwest	483	5,881	1372.5	3,493	9,901	0.233
Med_East	1,533	10,285	1003.9	8,252	12,819	0.098
Low_South	3,272	23,866	3269.9	17,288	32,948	0.137
Med_Northwest	1,104	5,910	1244.4	3,715	9,400	0.211
Low_West	1,931	12,968	5100.6	5,127	32,803	0.393
Total	**18,263**	**101,767**	**6720.1**	**88,461**	**117,075**	**0.066**

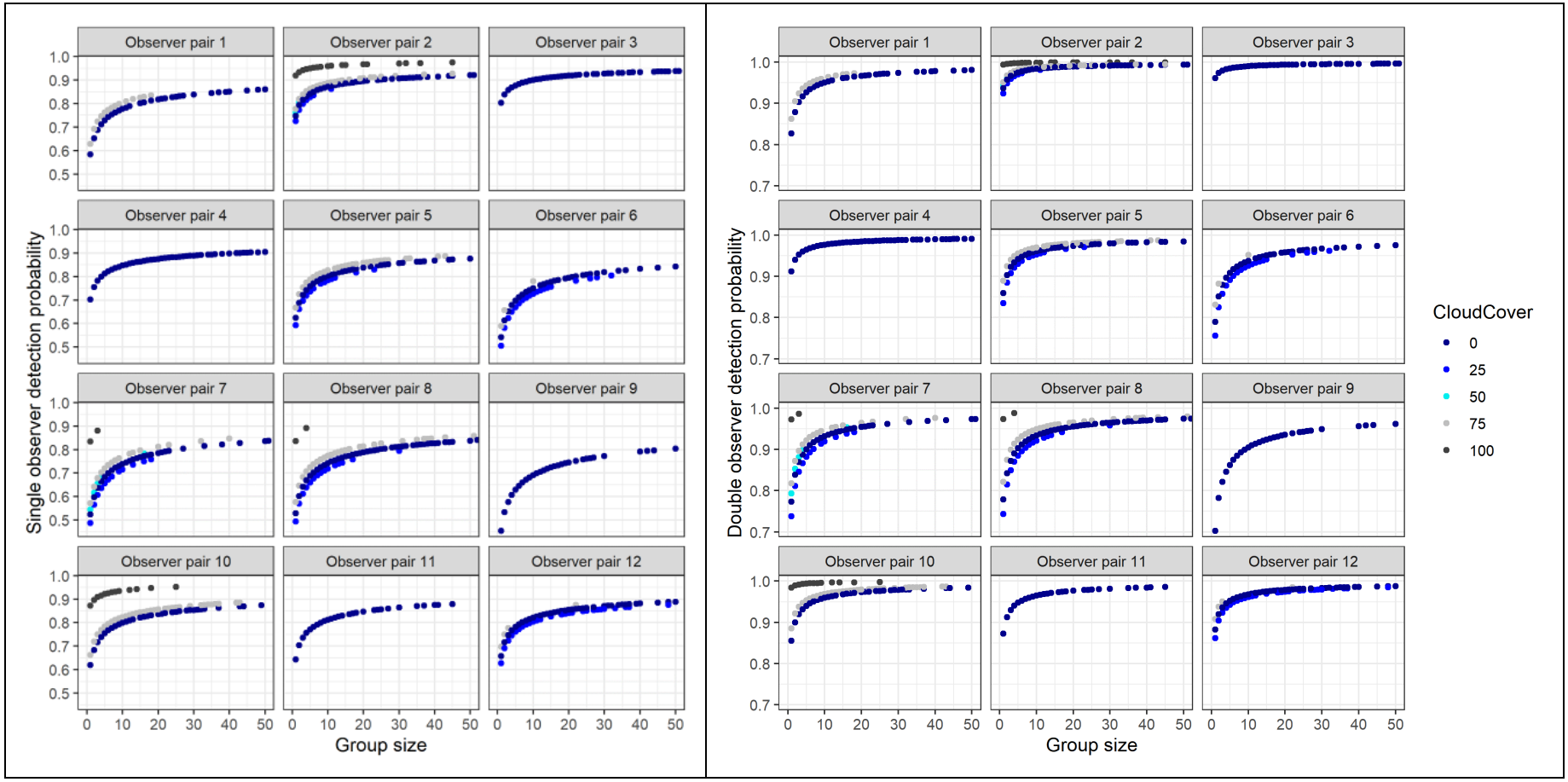


Figure 35. Predicted single and double observer sighting probability as a function of group size and observer pair.

4.3.5 Composition survey results

Calving ground composition surveys were flown from June 14th to 16th 2023 to coincide with the main visual abundance survey (**Table 2**). Results revealed expected segregation of females and bulls with the majority of females occurring in the VHDE and VHDW(1+2) strata, with bulls dominating the more southern strata, a finding consistent with bull telemetry location and movement data (**Figure 36**). Of note was a sub-segregation within the VHDW stratum where considerable mixing between breeding and non-breeding females, yearlings, and young and mature bulls occurred within the southern extents of the strata, likely the result of the migratory delays and barriers imposed by the advanced melt and associated high water levels. Overall sample sizes of groups were reasonably high in the main strata sampled, yielding reliable assessments of abundance strata composition (**Table 10**). Proportions of breeding cows, bulls, and cows (breeding and non-breeding) were estimated for each survey strata (**Table 11**).

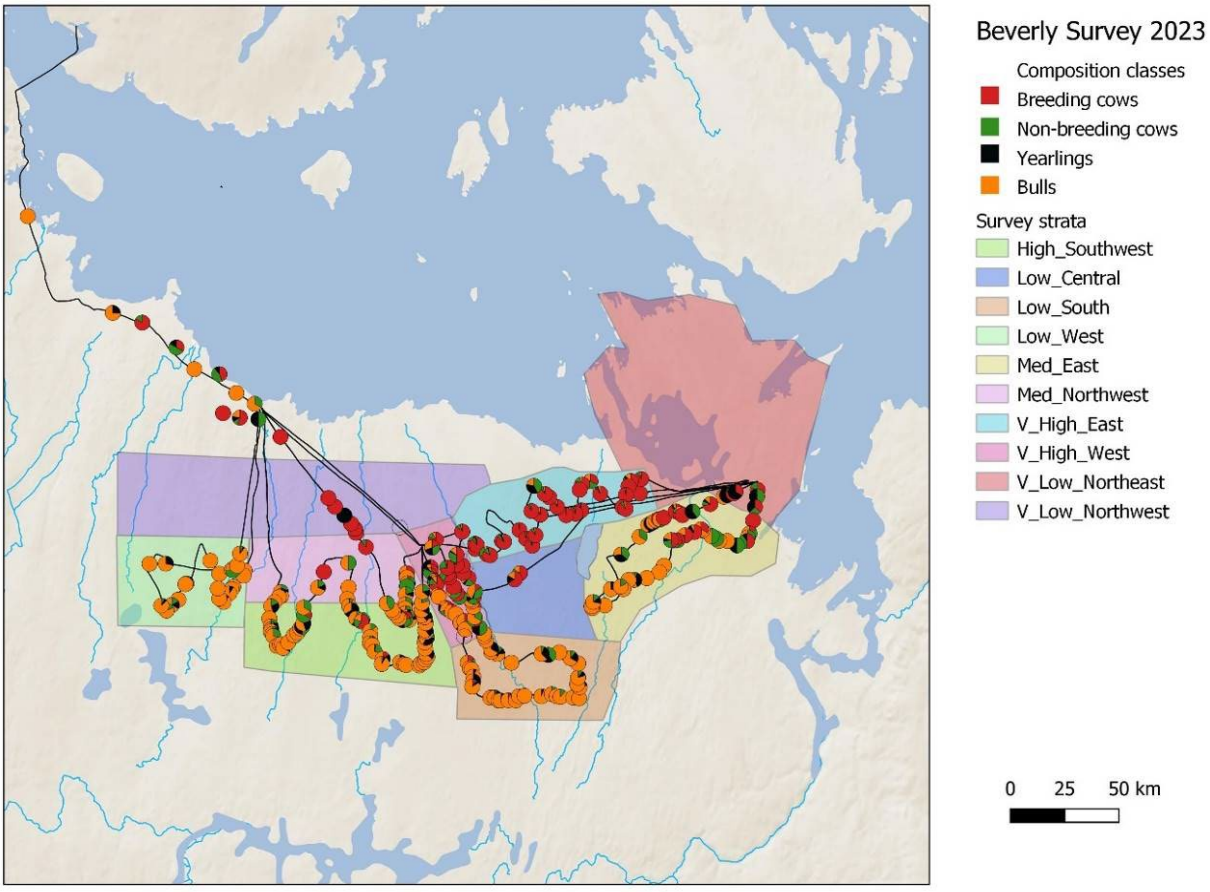


Figure 36. June 2023 composition survey flight paths with pie charts depicting each sample.

Table 10. Summary of observations during the June 2023 Beverly calving-ground composition survey.

Strata	Breeding cows	Non-Breeding Cows	Calves	Bulls	Yearlings	Total 1+ Caribou	groups
V_High_East1	4510	378	4046	153	72	5113	22
V_High_East2	1178	109	1060	64	19	1370	10
V_High_West	1430	481	993	473	134	2518	38
High_Southwest	26	148	12	1078	155	1407	62
Med_East	264	71	227	145	51	531	42
Med_Northwest	13	28	7	91	9	141	18
Low_Central	32	2	28	7	3	44	2
Low_South	14	43	0	538	61	656	34
Low_West	3	18	1	292	17	330	22
V_Low_Northeast	12	6	9	3	10	31	7
V_Low_Northwest	17	0	16	0	1	18	6

Table 11. Estimated proportion of breeding females (breeding females/total) and adult females (breeding+non-breeding females)/total within the Beverly June 2023 calving-ground abundance strata. Standard errors and confidence limits were based on bootstrap resampling methods.

Strata	Proportion	Estimate	SE	Conf. Limit	
V_High_East1	Breeding Cows	0.882	0.012	0.858	0.905
V_High_East1	Bulls	0.030	0.004	0.021	0.037
V_High_East1	Cows	0.956	0.005	0.948	0.965
V_High_East2	Breeding Cows	0.860	0.041	0.753	0.909
V_High_East2	Bulls	0.047	0.009	0.031	0.067
V_High_East2	Cows	0.939	0.015	0.902	0.962
V_High_West	Breeding Cows	0.568	0.054	0.446	0.650
V_High_West	Bulls	0.188	0.032	0.140	0.265
V_High_West	Cows	0.759	0.037	0.674	0.815
High_Southwest	Breeding Cows	0.018	0.007	0.008	0.033
High_Southwest	Bulls	0.766	0.022	0.721	0.807
High_Southwest	Cows	0.124	0.016	0.095	0.159
Med_East	Breeding Cows	0.497	0.103	0.259	0.637
Med_East	Bulls	0.273	0.066	0.180	0.444
Med_East	Cows	0.631	0.079	0.441	0.735
Med_Northwest	Breeding Cows	0.092	0.068	0.006	0.252
Med_Northwest	Bulls	0.645	0.061	0.518	0.752
Med_Northwest	Cows	0.291	0.067	0.179	0.433
Low_Central	Breeding Cows	0.727	0.037	0.667	0.769
Low_Central	Bulls	0.159	0.005	0.154	0.167
Low_Central	Cows	0.773	0.065	0.667	0.846
Low_South	Breeding Cows	0.021	0.008	0.007	0.036
Low_South	Bulls	0.820	0.019	0.780	0.857
Low_South	Cows	0.087	0.016	0.059	0.123
Low_West	Breeding Cows	0.009	0.005	0.000	0.020
Low_West	Bulls	0.885	0.019	0.850	0.922
Low_West	Cows	0.064	0.014	0.039	0.094
V_Low_Northeast	Breeding Cows	0.387	0.125	0.125	0.618
V_Low_Northeast	Bulls	0.097	0.100	0.000	0.333
V_Low_Northeast	Cows	0.581	0.134	0.269	0.806
V_Low_Northwest	Breeding Cows	0.944	0.080	0.714	1.000
V_Low_Northwest	Bulls	0.000	0.000	0.000	0.000
V_Low_Northwest	Cows	0.944	0.080	0.714	1.000

4.3.6 Estimation of breeding females, adult females, and bulls

Estimates of breeding females were derived by multiplying the overall estimates for the calving ground by the proportion breeders in each stratum (**Table 12**). Estimates of adult females were derived by multiplying total caribou by proportions of adult females (**Table 13**). Estimates of bulls were derived by multiplying the proportion bulls in each stratum by the total caribou (**Table 14**). An estimate was also derived for all caribou (bulls, cows, and yearlings) which included the very low strata (where no composition surveys were conducted) which increases the estimates by 9,913 caribou (CI = 6,046-16,270) (**Table 15**). It is assumed that the majority of these caribou were non-breeding caribou. All results were used in combination with fall composition data, data that is used to determine sex ratios as all sexes and ages gather during the fall rut, to develop a whole herd estimate.

Table 12. Estimates of breeding females in the Beverly June 2023 calving-ground core stratum.

Strata	N total caribou	CV	Prop. Breeding females	CV	N breeding females	SE	Conf. Limit		CV
V_High_East1	39,319	0.134	0.882	0.013	34,679	4655.5	26,284	45,755	0.134
V_High_East2	14,988	0.179	0.860	0.048	12,890	2390.8	8,664	19,178	0.185
V_High_West	27,198	0.212	0.568	0.095	15,448	3587.5	9,545	25,003	0.232
High_Southwest	38,825	0.048	0.018	0.363	699	255.9	328	1,488	0.366
Med_East	10,285	0.098	0.497	0.207	5,112	1168.2	3,069	8,516	0.229
Low_South	23,866	0.137	0.021	0.358	501	192.1	209	1,203	0.383
Med_Northwest	5,910	0.211	0.092	0.735	544	415.9	120	2,467	0.765
Low_West	12,968	0.393	0.009	0.573	117	81.2	25	542	0.694
Total	**173,359**	**0.061**			**69,990**	**6473.5**	**59,135**	**82,837**	**0.092**

Table 13. Estimates of adult females in the Beverly June 2023 calving-ground core stratum.

Strata	N total caribou	CV	Prop. Adult F	CV	N Adult females	SE	Conf. Limit		CV
V_High_East1	39,319	0.134	0.956	0.005	37,589	5024.0	28,524	49,535	0.134
V_High_East2	14,988	0.179	0.939	0.016	14,074	2531.8	9,571	20,695	0.180
V_High_West	27,198	0.212	0.759	0.048	20,643	4483.7	13,148	32,409	0.217
High_Southwest	38,825	0.048	0.124	0.132	4,814	674.0	3,577	6,479	0.140
Med_East	10,285	0.098	0.631	0.125	6,490	1028.1	4,545	9,266	0.158
Low_South	23,866	0.137	0.087	0.186	2,076	480.1	1,210	3,562	0.231
Med_Northwest	5,910	0.211	0.291	0.230	1,720	536.8	872	3,393	0.312
Low_West	12,968	0.393	0.064	0.220	830	373.8	290	2,376	0.450
Total	**173,359**	**0.061**			**88,236**	**7343.3**	**75,564**	**103,033**	**0.083**

Table 14. Estimates of bulls in the Beverly June 2023 calving-ground core stratum.

Strata	N total caribou	CV	Prop. Bulls	CV	N Bulls	SE	Conf. Limit		CV
V_High_East1	39,319	0.13	0.030	0.140	1,180	228.0	793	1,755	0.19
V_High_East2	14,988	0.18	0.047	0.201	704	189.7	397	1,247	0.27
V_High_West	27,198	0.21	0.188	0.170	5,113	1387.4	2,921	8,951	0.27
High_Southwest	38,825	0.05	0.766	0.029	29,740	1656.6	26,413	33,486	0.06
Med_East	10,285	0.10	0.273	0.243	2,808	735.9	1,567	5,030	0.26
Low_South	23,866	0.14	0.820	0.023	19,570	2718.2	14,113	27,136	0.14
Med_Northwest	5,910	0.21	0.645	0.095	3,812	880.6	2,294	6,336	0.23
Low_West	12,968	0.39	0.885	0.022	11,477	4520.9	4,531	29,069	0.39
Total	**173,359**	**0.61**			**74,404**	**5822.5**	**61,855**	**89,499**	**0.08**

Table 15. Estimates of total 1+ year old caribou *including* very low-density strata for the Beverly June 2023 calving-ground survey.

Strata	N	SE	Conf. Limit		CV
V_High_East1	39,319	5251.9	29,842	51,805	0.134
V_High_East2	14,988	2685.0	10,209	22,003	0.179
V_High_West	27,198	5758.9	17,517	42,230	0.212
High_Southwest	38,825	1853.1	35,072	42,980	0.048
Med_East	10,285	1003.9	8,252	12,819	0.098
Low_South	23,866	3269.9	17,288	32,948	0.137
Med_Northwest	5,910	1244.4	3,715	9,400	0.211
Low_West	12,968	5100.6	5,127	32,803	0.393
V_Low_Northwest	5,881	1372.5	3,493	9,901	0.233
V_Low_Northeast	4,032	759.8	2,553	6,368	0.188
Total	**183,272**	**10635.6**	**161,417**	**208,084**	**0.058**

4.3.7 Extrapolated herd estimates

The most recent fall composition survey for the Beverly herd occurred in October 2022 (Adamczewski et al. 2024) and was used to determine the sex ratio of the Beverly herd as typically all sexes and ages gather together in the fall during the October rut. Sampling for the survey occurred in 2 periods (**Figure 37**) with the first period (October 20-23) targeting Bathurst collars and the second period (October 24-November 3) targeting Beverly collared caribou. Further inspection of collar locations revealed that there were 3 Beverly cow collars mixed in with Bathurst cows in the first sampling period and observations around these collars were used to estimate bull cow ratios. In contrast, the second sampling period mainly targeted Beverly bulls and the remainder of the Beverly cow collars and therefore these observations were also used for estimates. The resulting data set had 251 groups (2,661 bulls:3,675 cows) classified. The resulting estimate of bull cow ratio was 0.724 (SE=0.037, CI=0.659-0.799) and the adult cow proportion was 0.580 (SE=0.012, CI=0.556-0.605). If only the October 23-November 4 data is used the bull-cow ratio was 0.628 (SE=0.042, CI=0.55-0.71), and an adult cow proportion of 0.614 (SE=0.0183, CI=0.579-0.650) which are the ratios reported in Adamczewski et al. (2024).

As the Beverly June 2023 calving-ground survey was designed to assess the abundance of adult female caribou, the total or “extrapolated” herd size was estimated using the estimated number of adult females developed during the abundance phase, divided by the proportion of cows in the herd using 2022 fall composition results (herd size = $N_{\text{adult females}}$ / proportion females in herd (fall)) (**Table 16**).

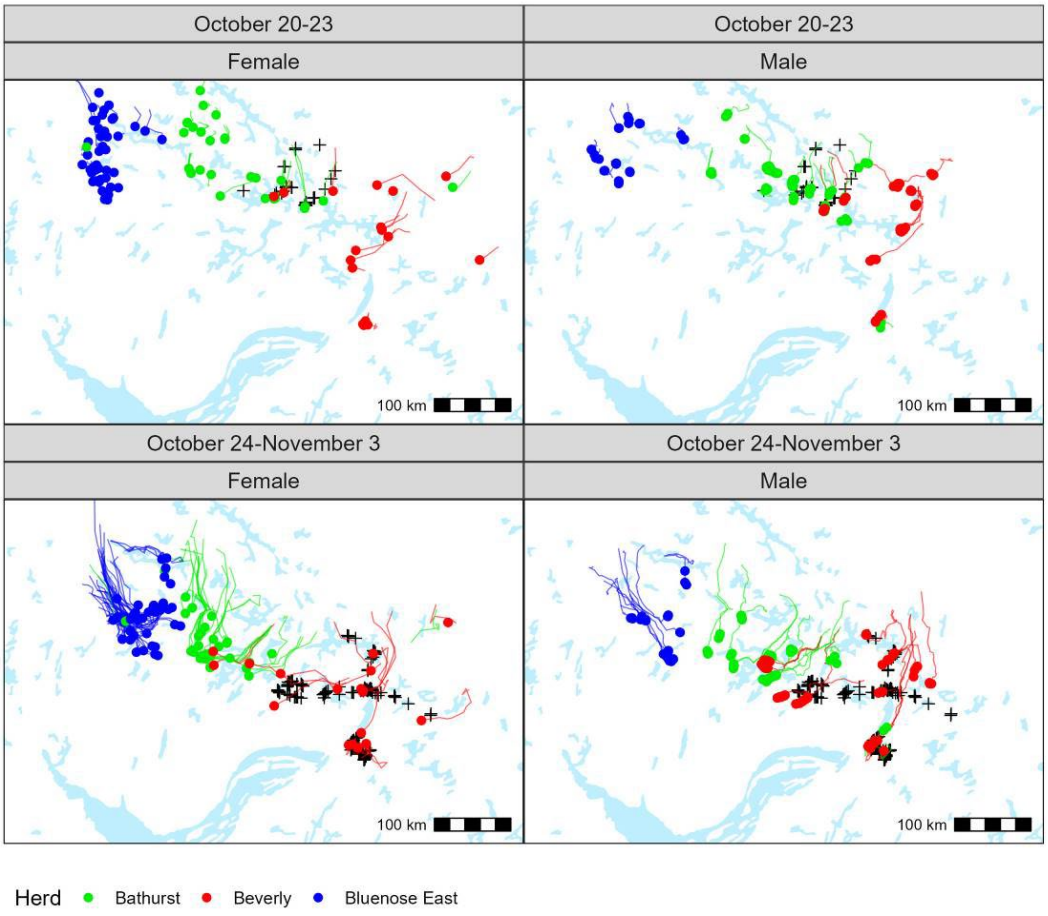


Figure 37. Collar locations and composition observations (+ signs) for 2 survey periods during the Fall 2022 Beverly composition survey. The first period (October 20-23) mainly targeted the Bathurst herd but did sample in the proximity of 5 Beverly cows. The second period primarily targeted the remainder of the Beverly herd. Paths of collared caribou during each sampling period are also shown.

One pattern evident during the 2023 Beverly calving-ground survey was high densities of bulls in most of the southern strata. Inspection of bull locations during the survey reveals that bulls were either in the proximity of, or within the 2023 survey strata, a condition not previously observed when compared to past surveys and telemetry data (**Figure 38**). More exactly, of 44 collared Beverly bulls, 28 (or 63%) were within the survey strata and only 16 were outside. Of the 16 bulls outside, only 2 (or 4.5%) were a substantial distance from the survey area (**Figure 39**). We believe that the un-seasonably early spring melt, and associated overflowing and fast-moving rivers, slowed and/or stopped the movement of breeding females into the more northern extents of their annual concentrated calving area. This barrier to movement held migrating breeding females within areas typically occupied by bulls based on telemetry and previous calving-ground survey observations. This condition reduced bull and cow segregation, causing breeding females to overlap into areas typically occupied by bulls during calving. **Figure 38** also suggests there was some mixing of the Bathurst and Beverly herds during the 2022 rut when the composition survey occurred. These observations suggest that most of the Beverly Herd (males and females) were within the June 2023 survey strata at the time of abundance flights. Using these observations, an alternative estimate of herd size was derived as the summation of bulls and cows (**Table 17**). This estimate is higher than the extrapolated herd estimate but lower than the total estimate of caribou on the calving ground. The remainder of caribou in this case would be yearling caribou which amounted to 10,731 (SE = 874.3, CI = 8,313 - 13,853) yearlings across all strata (excluding the low-density strata).

Table 16. Estimates of extrapolated herd size for the 2023 Beverly calving-ground survey based on estimates of adult females from the survey and estimates of proportion cows in the herd from the 2022 fall composition survey.

Method	N	SE	Conf. Limit		CV
Proportion females	152,131	13071.9	124,704	179,558	0.085

Table 17. Estimates of adult herd size as the summation of bulls and cows estimated in core survey strata during the June 2023 Beverly calving-ground survey.

Group	N	SE	Conf. Limit		CV
Bulls	74,404	5822.5	61,855	89,499	0.078
Cows	88,236	7343.3	75,564	103,033	0.083
Total	**162,640**	**9371.5**	**143,375**	**184,494**	**0.058**

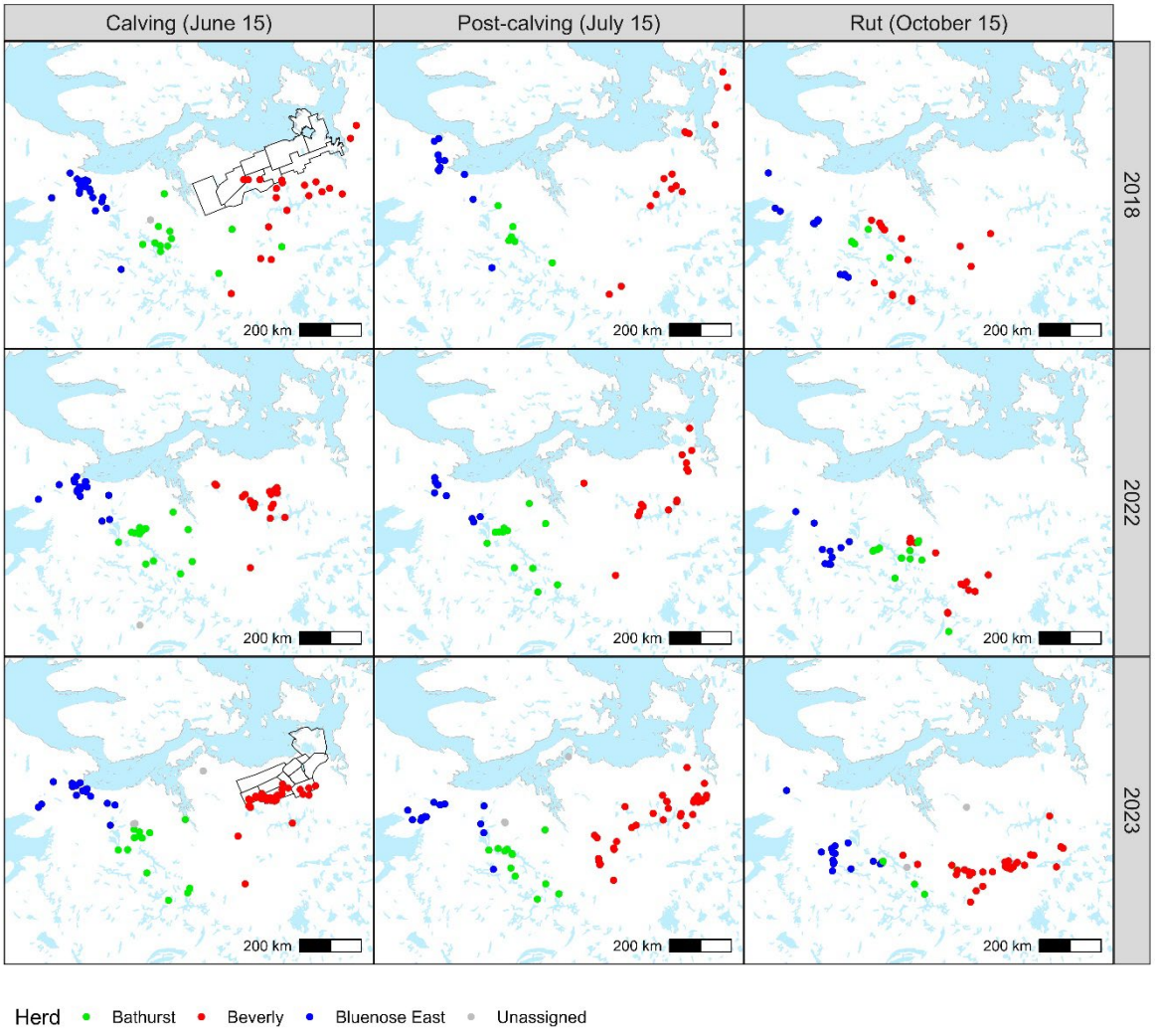


Figure 38. Locations of bulls for calving, post-calving, and rut in 2018, 2022 and 2023. Strata areas are shown for 2018 and 2023 when calving ground surveys occurred. Composition surveys occurred during the fall of 2022 during the rut.

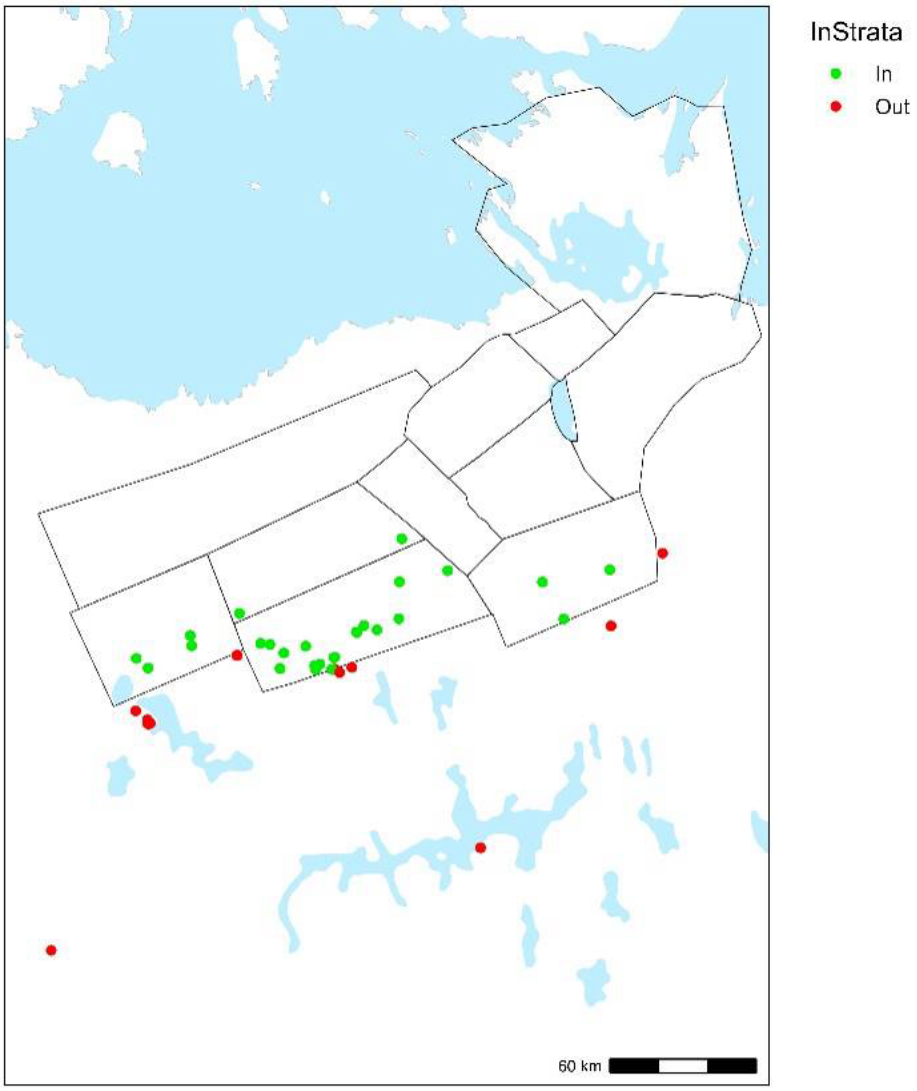


Figure 39. Closeup of locations of 44 Beverly bull collars on June 15, 2023 relative to survey strata. Locations are classified as to whether they are within or outside of the survey strata. One collar to the south is not shown.

4.4 ABUNDANCE TRENDS

Trends in adult females suggest an increase in 2023 from the 2018 survey and approximate stability compared to the 2011 survey (**Figure 40**). Estimates of yearly change for adult females indicate a significant yearly increase of 8% per year (CI=4-12%) of adult females between 2018 and 2023 (**Table 18**).

Reconnaissance data from abundance survey years (2011, 2018, and 2023) and reconnaissance survey years (2013 and 2016) provides a secondary assessment of trend and spatial distribution of caribou on the calving ground. A plot of segment densities indicates higher segment densities in 2013 and 2023 (**Figure 41**). However, very high segment densities are more prevalent in 2023 with segment densities above 50 caribou/km^2. A plot of transect densities from west to east also reveals notably higher transect densities in 2023 but also reduced coverage to western areas. Despite the difference in coverages, the overall number of caribou observed during the reconnaissance survey was two to four times higher than observed in any of the other survey years. Some areas of higher density were video surveyed, along with the visual counts, for the June 2023 reconnaissance survey that preceded the abundance phase. Similar cross-referencing methods (to the abundance survey video assessments) were used to reconcile visual and video polygons within the June 2023 reconnaissance. The proportion of caribou counted on orthomosaic polygons along reconnaissance transects is delineated in **Figure 42**. Peak densities (approximately 40 caribou/km^2) on some lines are at levels also observed during the visual survey.

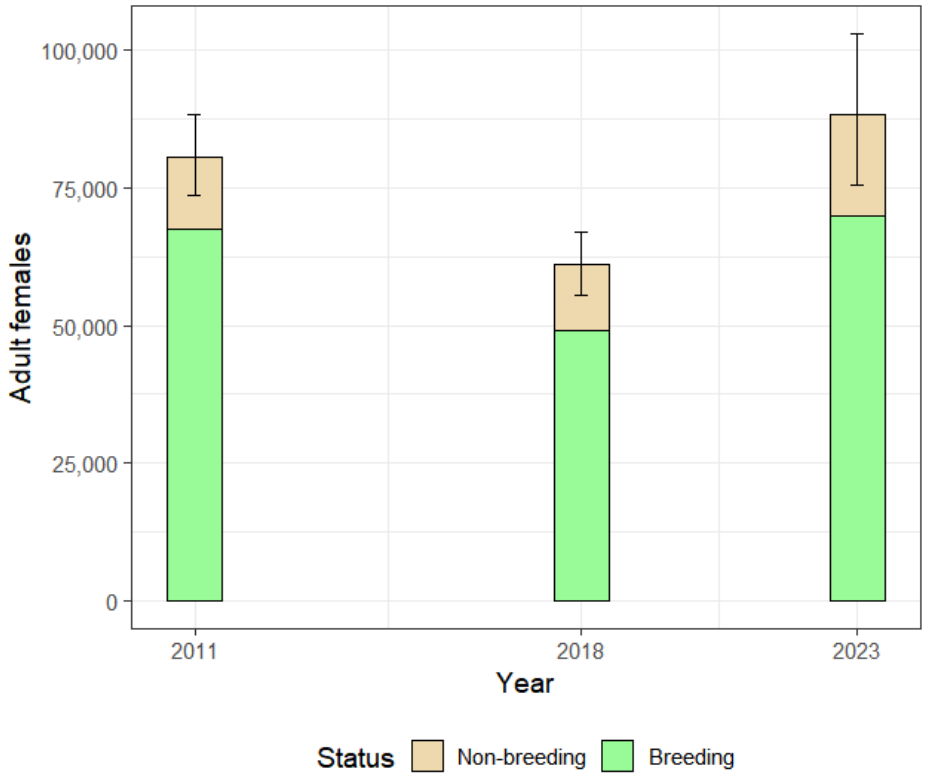


Figure 40. Adult female estimates by breeding status from 2011-23 Beverly abundance surveys.

Table 18. Estimates of gross and annual change for adult females between Beverly June calving-ground abundance survey efforts.

Years	Year 1		Year 2		Gross change				Yearly change (lambda)			
	N	SE	N	SE	GC	SE	Conf. Limit		λ	SE	Conf. Limit	
2018-23	61070	2887.8	88236	7343.3	1.44	0.14	1.20	1.74	1.08	0.02	1.04	1.12
2011-18	80705	3724.3	61070	2887.8	0.76	0.05	0.66	0.86	0.96	0.01	0.94	0.98
2011-23	80705	3724.3	88236	7343.3	1.09	0.10	0.91	1.31	1.01	0.01	0.99	1.02

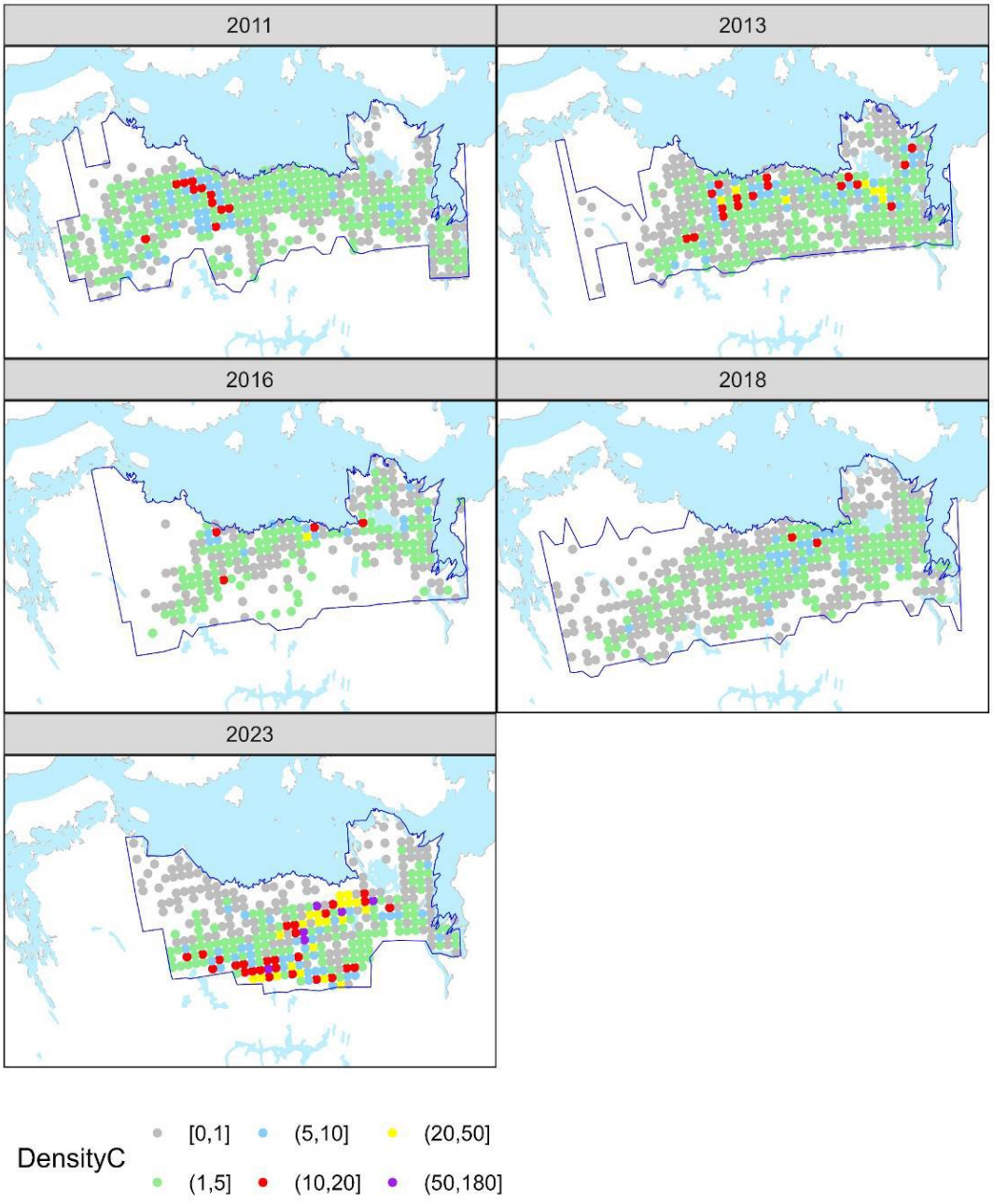


Figure 41. Segment densities (caribou/km^2) from June Beverly reconnaissance surveys. Densities are color coded based on ranges of densities observed at each segment flown. The extent of surveys is delineated by a blue polygon for each year.

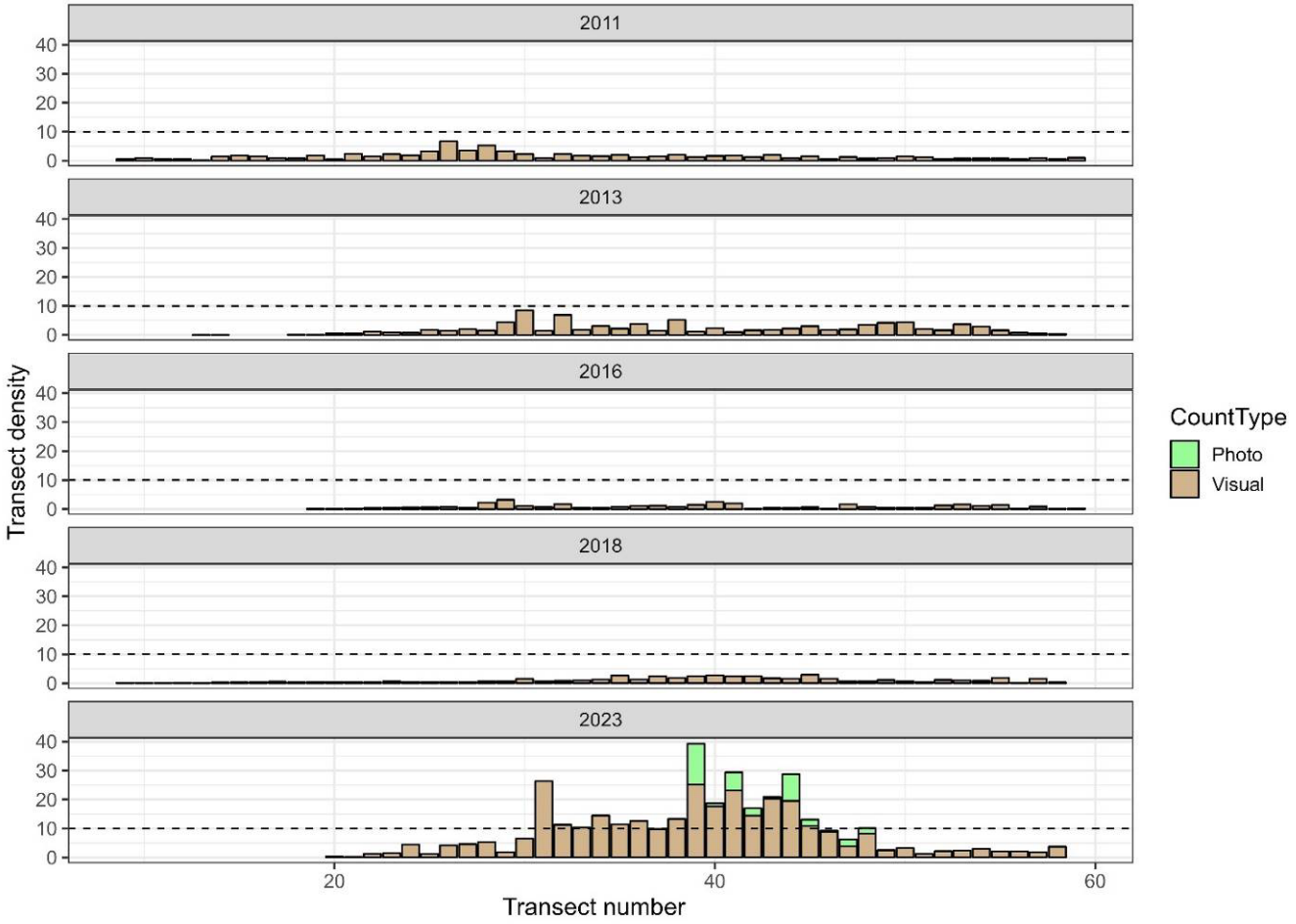


Figure 42 Caribou density (caribou/km^2) by reconnaissance transect line for each survey year. The proportion of caribou counted on photos is given as sub-bars for 2023 (**Figure 18**).

4.5 COLLAR ANALYSIS

Collar data was used to estimate movement between calving grounds and derive survival rate estimates for Beverly cows and bulls. All collar data for the Beverly herd was provided by the Government of the Northwest Territories and included both cow and bull collared caribou. During the Beverly June 2023 calving-ground survey, collared caribou from the Lorillard, Wager Bay, Ahiak, Qamanirjuaq, Beverly, Bluenose, and Bathurst herds were monitored daily. Collared bull data was only available for the Bathurst and Bluenose herds. All collar data was used primarily to help predict reconnaissance and abundance survey extents and timing. Due to the overlap between the Beverly, Bathurst and Ahiak herds observed during the calving season over the last 10 years, collar data was also used to assess the degree of overlap between the Beverly survey area and the Bathurst herd to the west, and Ahiak herd to the east. Additionally, collar data was used to assess any irregularities in the herds spring movements onto their annual concentrated calving areas when compared to historically documented spring migratory movements and associated distribution, throughout delineated seasonal calving extents.

4.5.1 Movements between calving grounds

From 2018 to present, late winter (April) telemetry locations and migration paths of Nunavut's mainland migratory caribou herds, reveal recent intermixing, which may have influenced movements of adult females between calving grounds up to and including June 2023 (**Figure 43**). A challenge with assessing movements between calving grounds is that caribou need to be monitored more than one year. Sample sizes of adult females with known herd membership, based on whether they were monitored over consecutive years, are presented in **Table 19**. It can be seen that

sample sizes are limited, below 20 for many herds, which ultimately limits the power to detect inter-herd movements (Boulanger et al. 2024).

Observations of mean calving locations of adult female caribou monitored more than one year reveals movement between the Beverly and Ahiak annual calving-grounds in most years, as well as movement of caribou from the Bathurst to the Beverly calving-ground starting in 2018 (**Figure 44**). Within the Ahiak calving-grounds fidelity was relatively low, with most caribou moving to another calving ground (Wager Bay or Beverly herds) each year. In 2022 and 2023, a small number of Ahiak cows were documented to have calved just to the east of Bathurst Inlet, and to the west, but not in the proximity, of the majority of calving Beverly caribou.

The calving extents of the Beverly or Bathurst herds were roughly based on which side of Bathurst Inlet caribou were located on June 10th. In some cases, Bathurst caribou calved just to the east of Bathurst Inlet which was further west than the majority of Beverly caribou which typically calved in the central Queen Maud Gulf. However, those cows calving to the east of Bathurst Inlet were anomalous as they were well east of the core Bathurst calving ground located, since the late 1990s, west of Bathurst Inlet. Therefore, there is some uncertainty in classifying caribou to a calving ground in this area of overlap (**Figure 45**). Caribou that calved just to the east of Bathurst Inlet were therefore also assessed based on their post-calving movements. For example, in 2021 Bathurst collared caribou cows identified as BGCA-19370, BGCA-20105, and BGCA-21162 calved to the east of Bathurst Inlet then moved further east to join other Beverly caribou cows later in June. These individuals were therefore classified as Beverly caribou. In contrast, Bathurst cow number BGCA-19370 calved to the east of Bathurst Inlet in 2019 but remained in this area for the remainder of June (away from other Beverly cows) and therefore was classified as Bathurst. In reality, travel conditions and other seasonal factors likely influence where caribou calve. Therefore there will be a segment of caribou that occur away from the central Bathurst and Beverly caribou calving grounds leading to inexact classifications given the relatively low sample size of collars of known herd status.

Along the western boundary of the Beverly annual calving-ground, movements in and out of each calving ground for a given year, as well as the number of caribou showing fidelity to a given calving ground in a given year, are displayed in **Figure 46**. The complexity of calving-ground use overlap was exemplified in June 2023, whereby 16 Bathurst collared caribou from June 2022 returned to the Bathurst calving ground while 4 moved to the Beverly calving ground, with 1 Bluenose-East caribou moving into the Bathurst calving ground. This figure also shows how the Ahiak herd calving-ground is a confluence with movement into and out of the calving ground each year from both Beverly caribou cows in the west, and Wager Bay caribou cows in the east. The frequencies of movement in and out of calving grounds can also be shown spatially by period (2011-2018 and 2019-2023) (**Figure 46**). Of most interest is whether there was directional movement into the Beverly calving-ground from 2019 to 2023 which could have contributed to the increase in Beverly abundance observed in June 2023. It can be seen that from 2019 to 2023, 12 Bathurst, and 8 Ahiak caribou, moved onto the Beverly calving ground. At the same time two Beverly caribou moved to the Bathurst calving ground, however, these caribou occurred east of Bathurst Inlet in the ambiguous area between the two herds as shown in **Figure 45**, and therefore this movement may not add up to an actual increase in the adult cows in the Bathurst herd but rather be an extra-limital movement by a Beverly cow.

The challenge with interpreting frequency data comes when attempting to estimate a proportion or probability of switching given different sample sizes of collars for each herd. To help better inform this process a multi-state model was used to estimate switching probabilities which are displayed graphically in **Figure 48**. Period-specific (2011-2018 and 2019-2023) switching probabilities were estimated for the Bathurst, Beverly, and Ahiak herds. Sparse data precluded estimation of period-specific probabilities for the Ahiak, Lorillard, and Wager Bay herds. When using Beverly, Bathurst, and the limited Ahiak telemetry data, estimates suggest directional movement from the Bathurst calving-ground to the Beverly, and from the Ahiak calving-ground to the Beverly in the 2019 to 2023 period. Estimates were run using Markov Chain Monte Carlo methods to ensure robust estimates. The resulting

estimates suggested that fidelity for the Beverly herd was relatively high (0.90 and 0.96) for both periods while fidelity for the Ahiak herd was relatively low over the same period (0.42 and 0.62). The general conclusion from the collar analysis is that there has been net movement of caribou into the Beverly from both the Bathurst and the Ahiak calving grounds. Therefore, increases in abundance are likely due to both immigration of caribou into the Beverly calving ground as well as demographic increase. This topic was explored further using an Integrated Population Model.

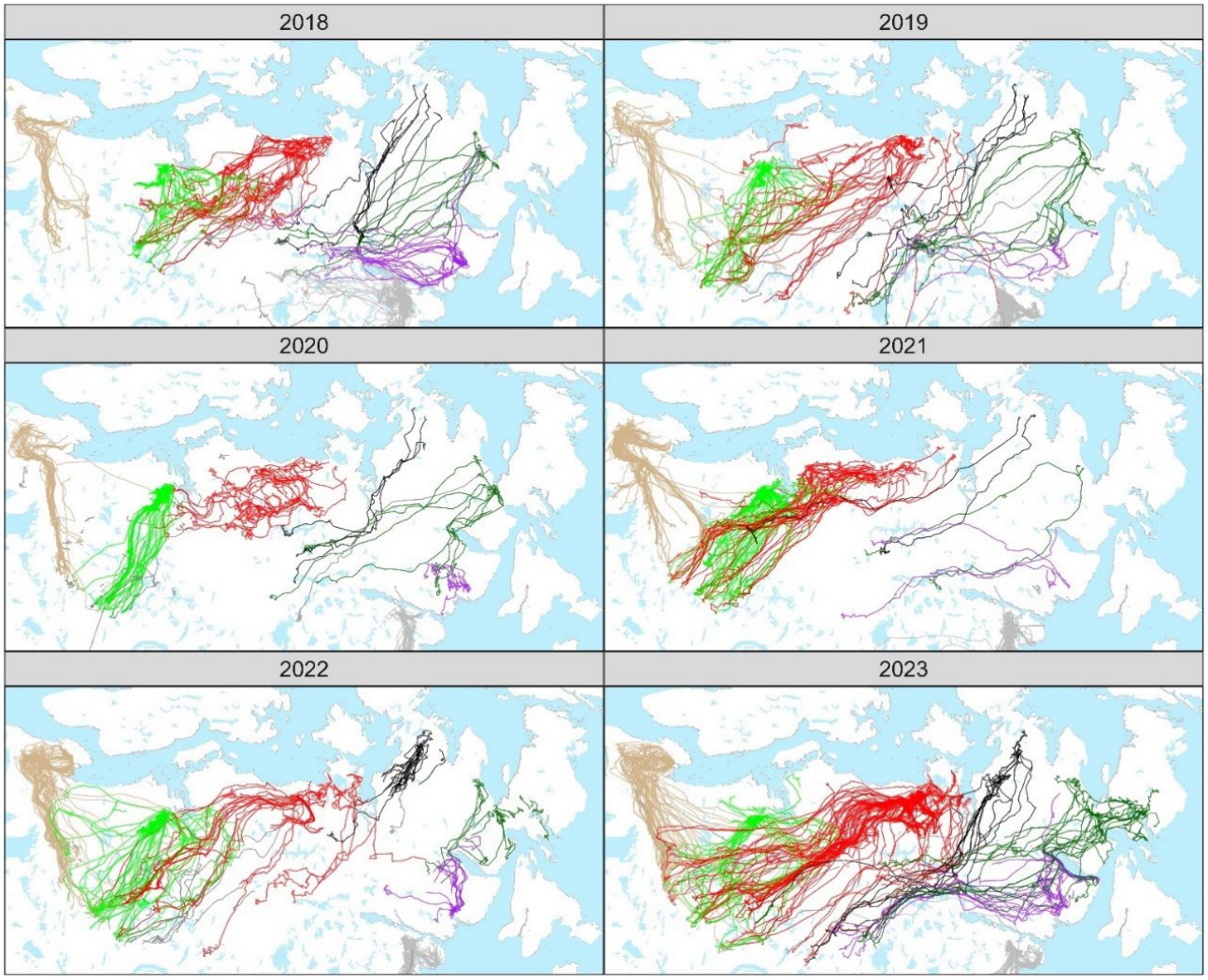


Figure 43. Migration paths (March to June 15) for the Bluenose-East (brown), Bathurst, (green), Beverly (red), Ahiak (black), Wager Bay (dark green), and Lorillard herds (purple). Collars were classified based on the calving ground that they calved in during 2022 and 2023.

Table 19. Sample sizes of adult female caribou that were monitored more than 1 complete year to determine known calving ground locations over a two year period.

Year	Bathurst	Beverly	Ahiak	Wager Bay	Lorrilard
2011	10	7	7	0	6
2012	15	16	6	0	5
2013	10	11	4	1	4
2014	14	28	1	1	3
2015	22	29	1	4	5
2016	26	27	3	6	9
2017	28	26	2	5	8
2018	18	30	7	8	13
2019	22	29	5	10	5
2020	36	16	3	7	3
2021	37	29	2	2	3
2022	37	18	13	15	7
2023	16	22	9	11	9
Totals	**291**	**288**	**63**	**70**	**80**

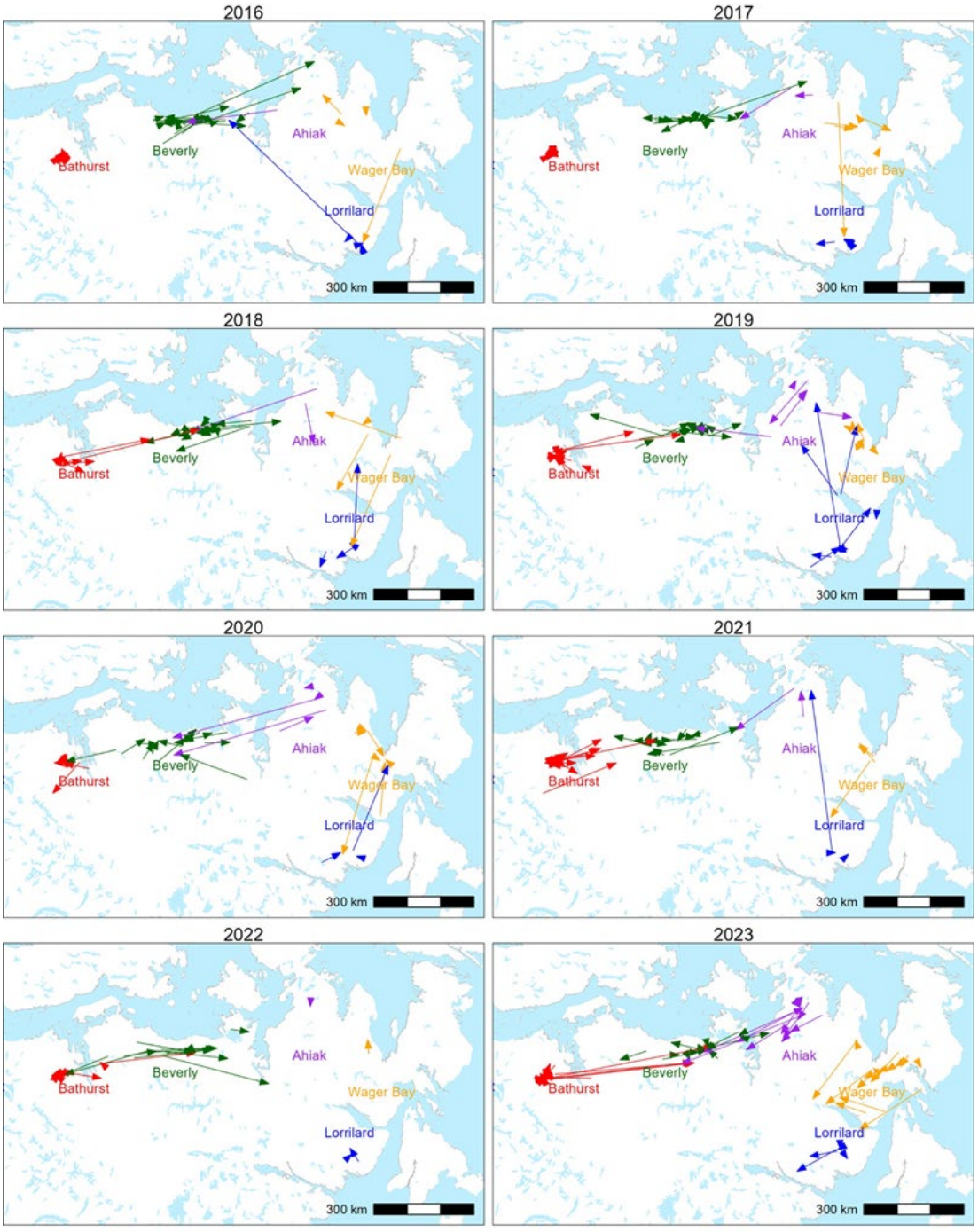


Figure 44. Mean calving ground locations for individual caribou in the previous year (tail of arrow) and current year (head of arrow) with the arrow colored according to the previous year calving ground. Each line and arrow indicates one collared caribou

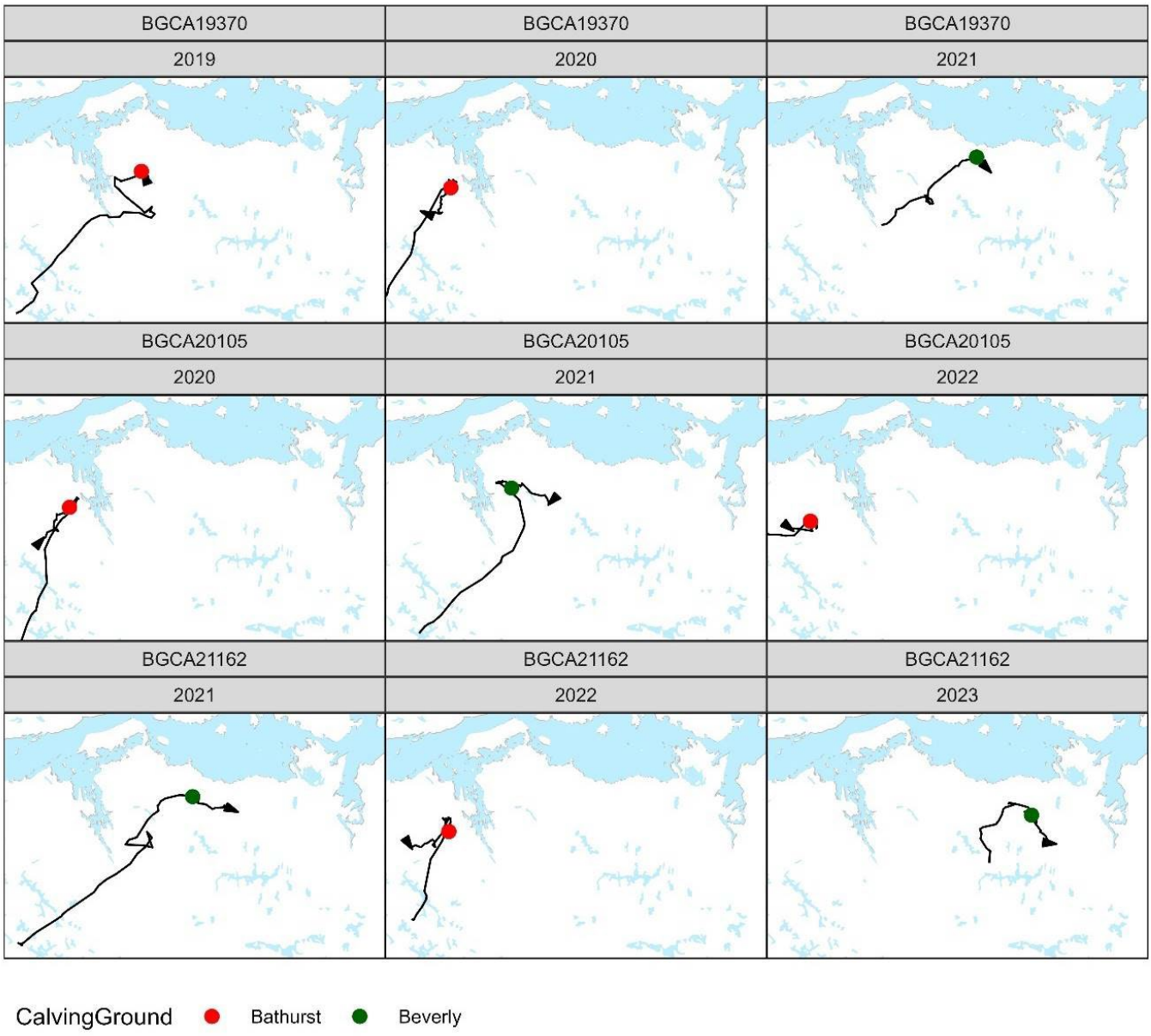


Figure 45. Calving locations (on approximately June 10th) and paths (May 1-June 30) for 3 collars that calved in-between the core Bathurst and Beverly calving grounds.

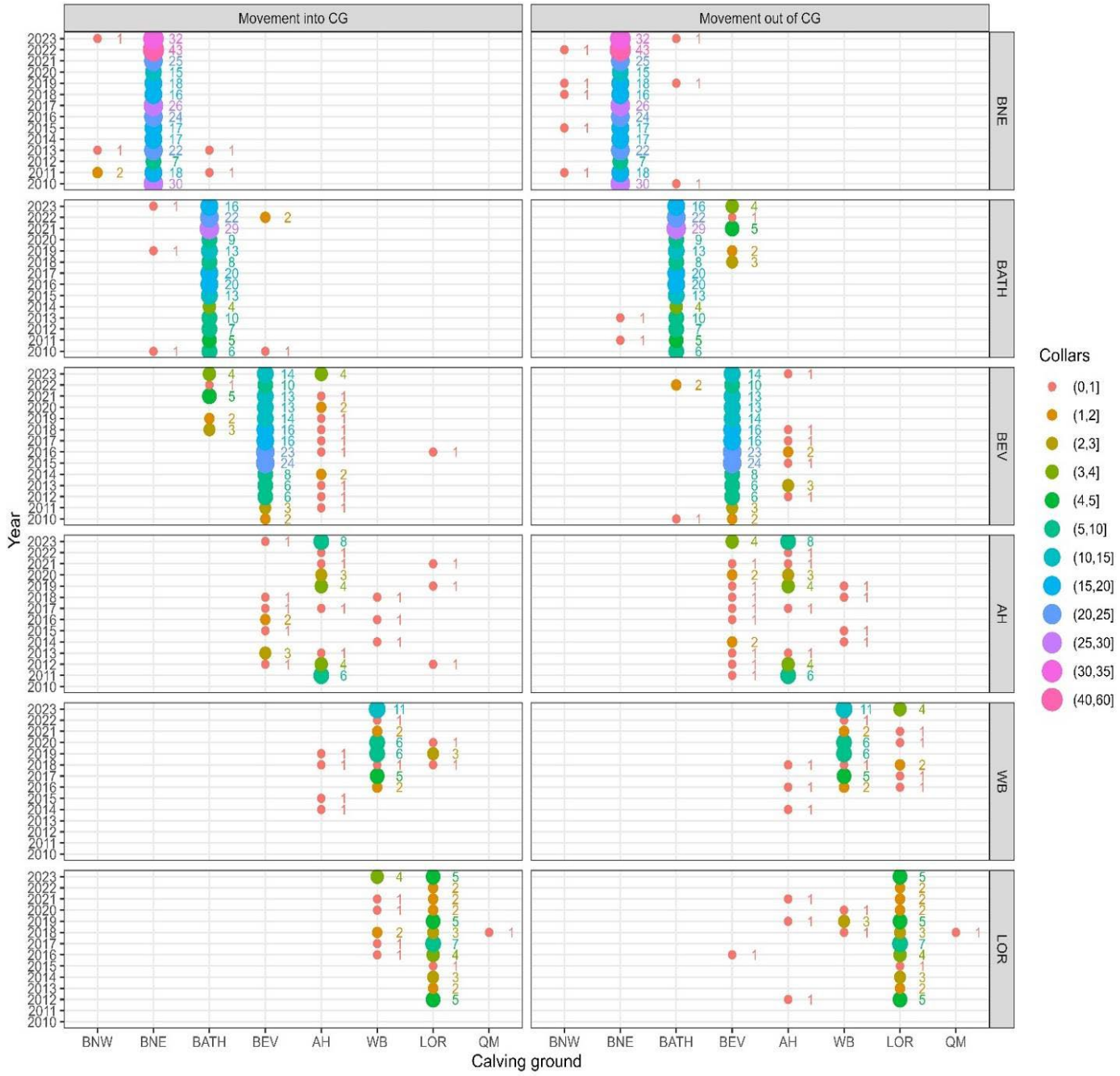


Figure 46. Graphical summary of movement events in and out of calving grounds. The left plot displays movements into the cg each year and the right are movements out, with columns indicating the herd. If a column is the same as the herd (i.e. LOR column in LOR herd plot) then sample sizes indicate caribou returning to the calving ground from the previous year (fidelity). This column is repeated for movements into and out of calving grounds.

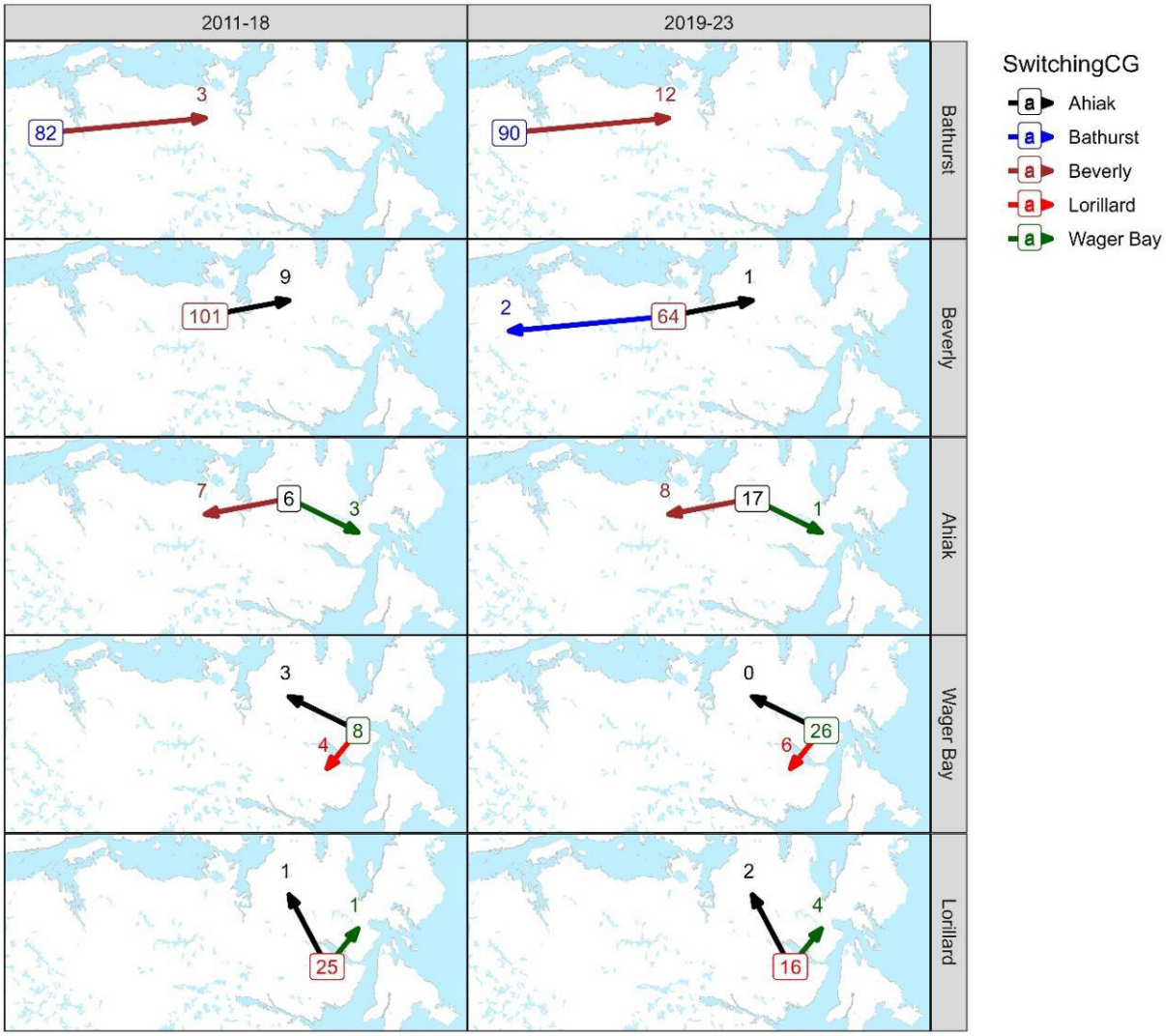


Figure 47. Frequencies of fidelity (numbers in boxes) and switching events of adult females from 2011-2018 and 2019-2023 for the Beverly and surrounding caribou herds.

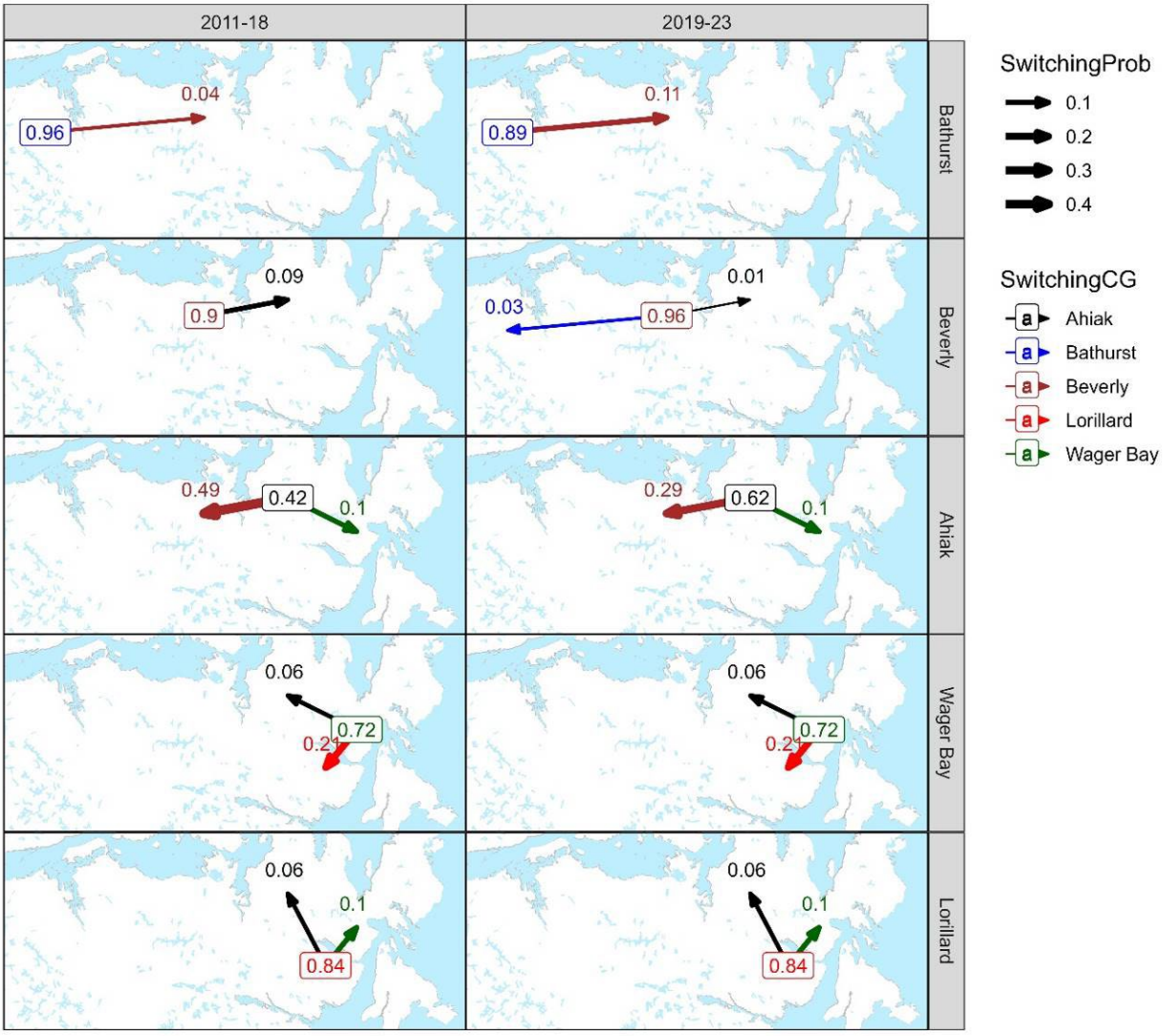


Figure 48. Estimates of fidelity to specific calving-grounds (numbers in boxes) and switching probabilities (arrows) of adult females switching calving-grounds from 2011-2018 and 2019-2023 for the Beverly and surrounding caribou herds. Estimates with confidence limits are given in Appendix B.

4.6 COLLAR SURVIVAL ANALYSIS

One of the key demographic parameters that can be used to help understand changing barren-ground caribou abundance is adult female survival. We used data sets of collared adult female caribou maintained by the Government of the NWT to estimate survival. Caribou collar histories were constructed using location data and collar fate data. In addition, caribou were classified by the calving ground that they calved on in a given year. A caribou that calved on the Beverly or Adelaide Peninsula area was considered a Beverly cow for this analysis (**Figure 49**). The resulting collection of associated data was then used to estimate monthly mortality rates of caribou based on the number of collar mortalities divided by the total collars monitored for a given month (**Figure 50**). Yearly survival rates were then estimated based on the product of 1-monthly mortality rate for a given caribou year, which is the Kaplan-Meir survival rate estimator (Pollock et al. 1989). Estimates, however, were of lower precision due to lower collar sample sizes (**Figure 51**). The survival rates were used as an input into the Integrated Population Model (IPM) which also utilized other data sources to refine survival rate estimates. The results of the survival analysis and the IPM within which it was used are discussed in the following section.

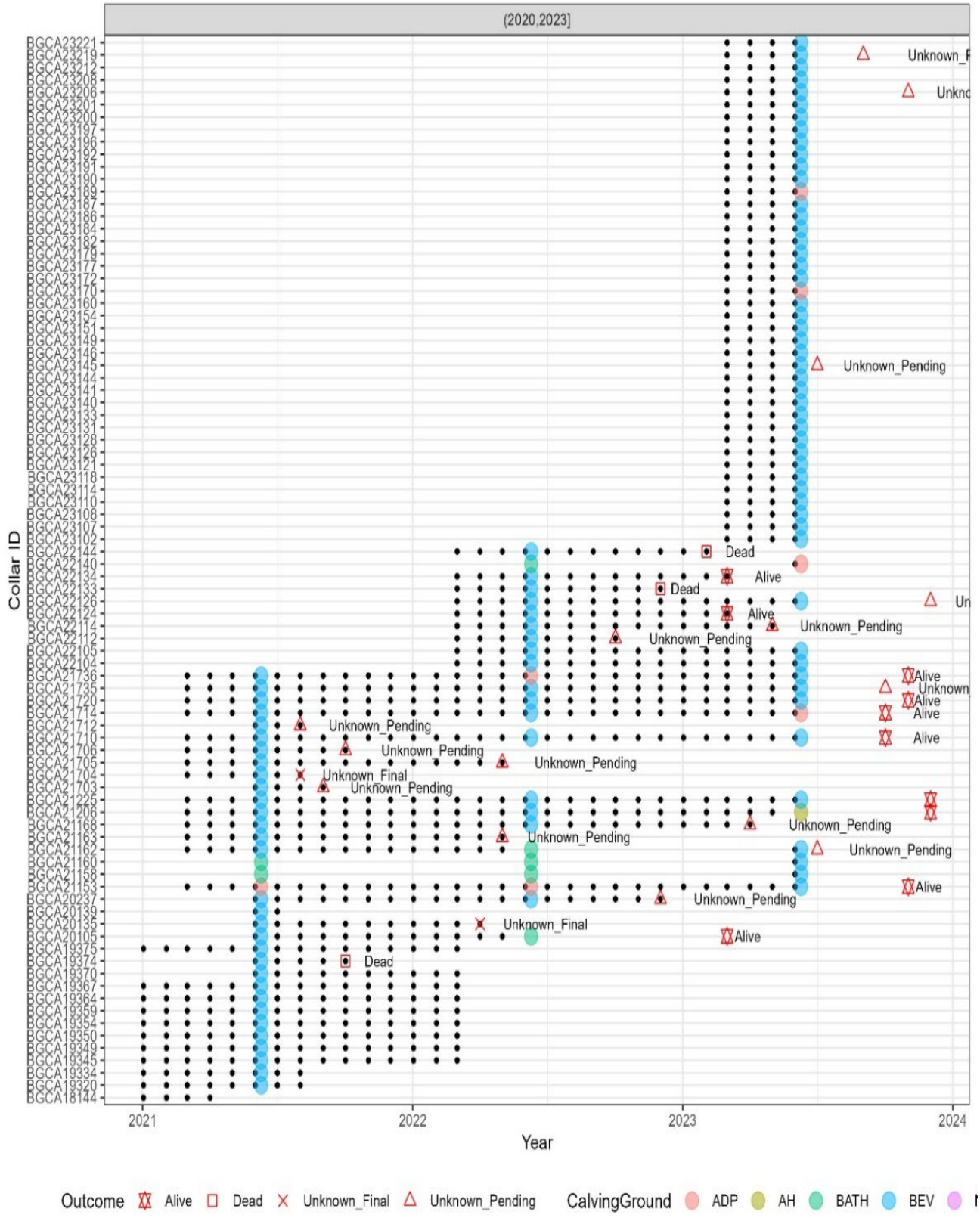


Figure 49. Collar histories of Beverly cows. Included is fate of each collar and the calving ground (colored dot) it utilized each June.

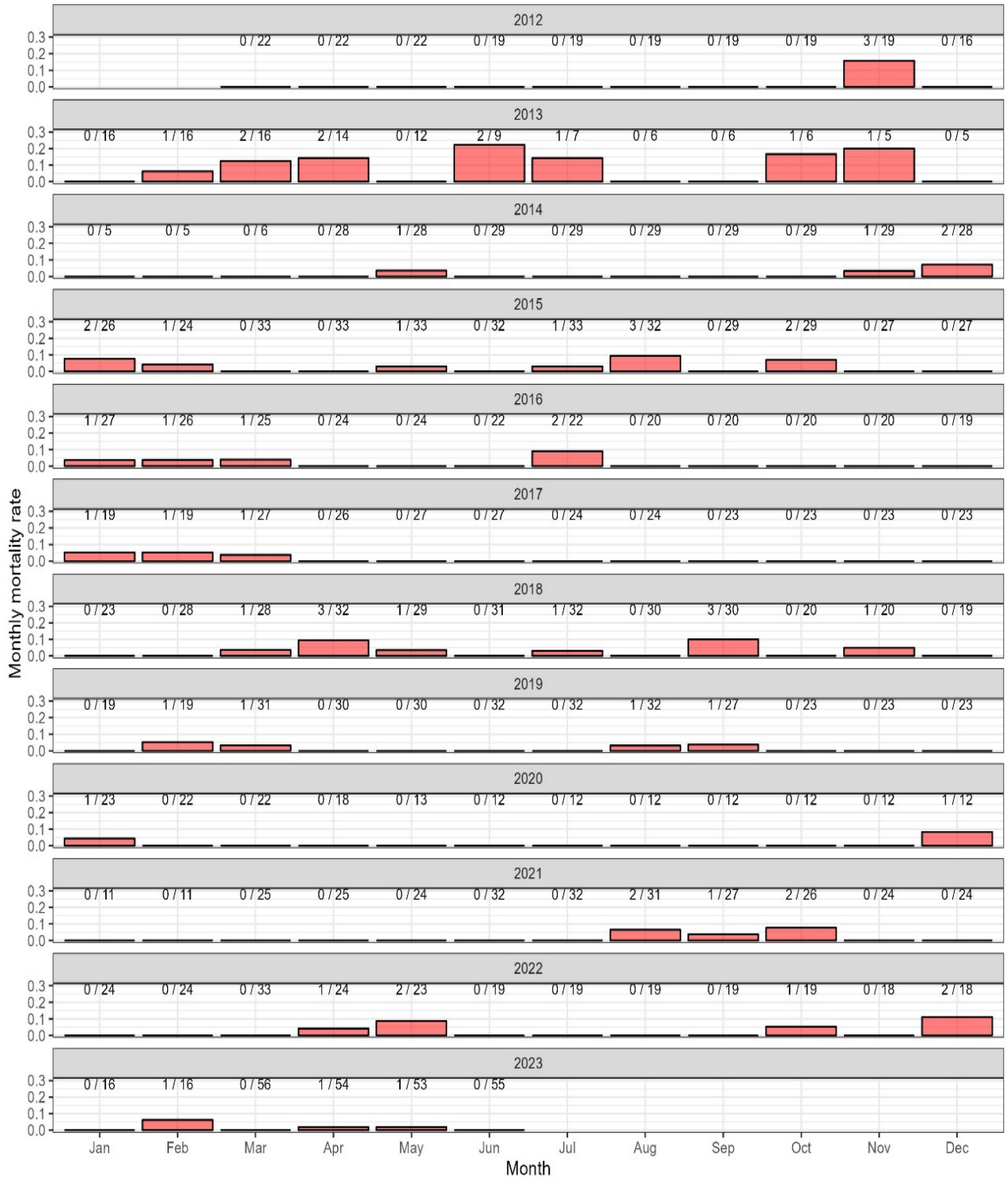


Figure 50. Monthly mortality rates for collared caribou from 2012 to 2023. Sample sizes of mortalities and collars monitored is given above each bar.

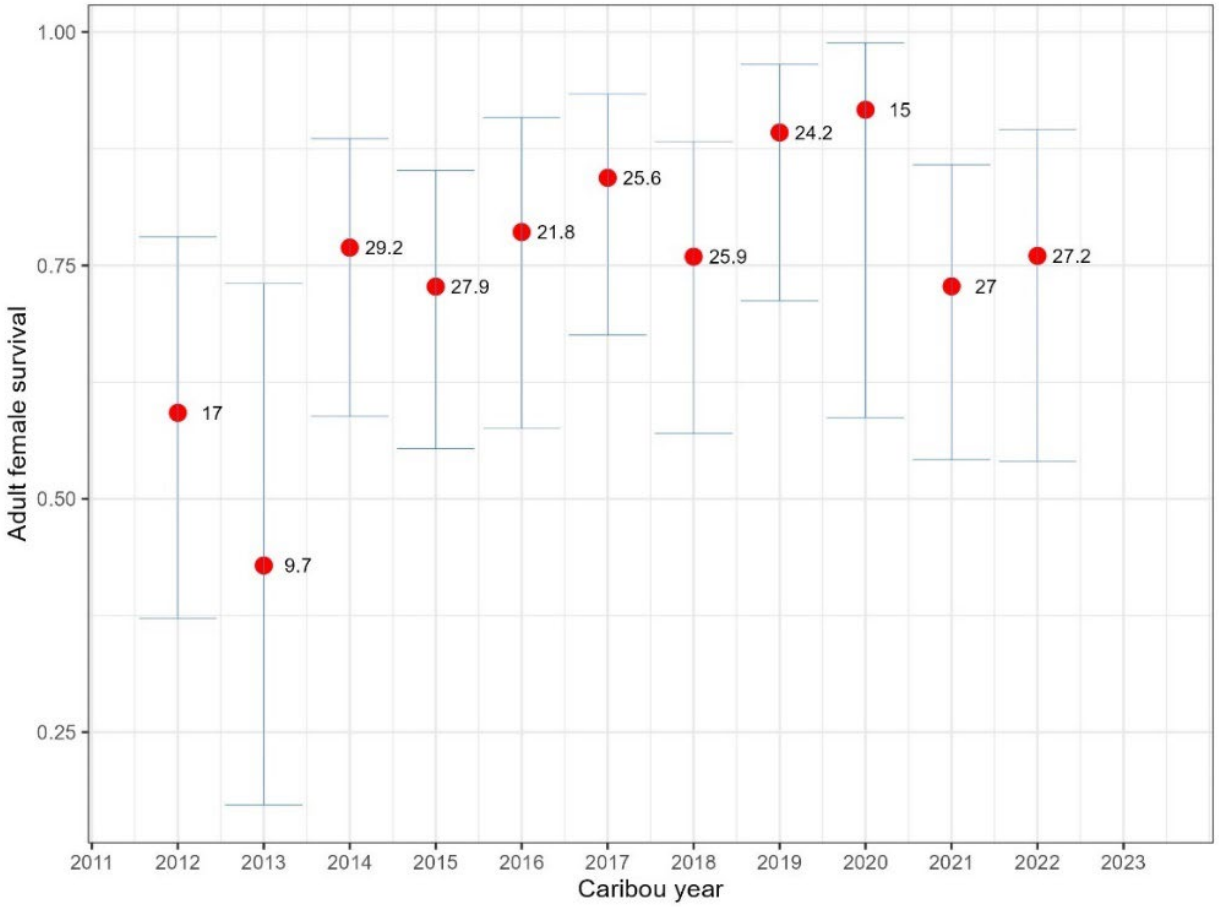


Figure 51. Yearly survival rates for collared adult females. The mean number of collars monitored is given next to each estimate.

4.7 INTEGRATED POPULATION MODEL - DEMOGRAPHIC ANALYSIS

4.7.1 Data sources

In addition to adult survival, other data sources for the IPM included spring and fall composition surveys, survival rates for bulls, and composition survey results from calving grounds (Adamczewski et al. 2023, Adamczewski et al. 2024; Campbell et al. 2013; Campbell et al. 2019). While limited, the data set does suggest some trends such as reasonable productivity (spring CC ratios > 0.4) and cow survival rates in the range of 0.8 or higher for many of the years measured (**Figure 52**). Also considered is the estimate of Ahiak adult females in 2011 and 2021 (Campbell et al. 2022) in unison with frequencies of collars moving to the Beverly (**Figure 46**).

An additional source of information was the estimated immigration of the Bathurst to the Beverly based upon the Bathurst IPM which models fidelity and movement of the Bathurst herd (Adamczewski et al. 2022, Boulanger et al. 2024). For this model fidelity of Bathurst cows to the Bathurst calving ground was estimated by the frequency of cows that calved on the Bathurst calving ground in successive years (**Figure 53**). As discussed previously, a proportion of cows emigrated to the Beverly calving ground most notably after 2018. Using estimates of fidelity, emigration/immigration, and adult cows, the number of cows present on the calving ground with and without immigration/emigration was estimated. The difference in these two numbers was then considered an estimate of cows from the Bathurst emigrating to the Beverly calving ground. This number was usually low but did increase up to nearly 2,000 adult Bathurst cows in 2018 with levels varying between 0 and 1,000 from 2019 to 2022. Note that these Bathurst cows are assumed to continue having calves based on fecundity rates of the Bathurst herd on the given year (**Figure 53**). Additionally, the actual impact of Bathurst adult cows immigrating to the Beverly herd is compounded

through time as the newly recruited adult females continue to produce calves in successive years, further bolstering demographic growth within the Beverly herd as female calves themselves become breeding females and so on.

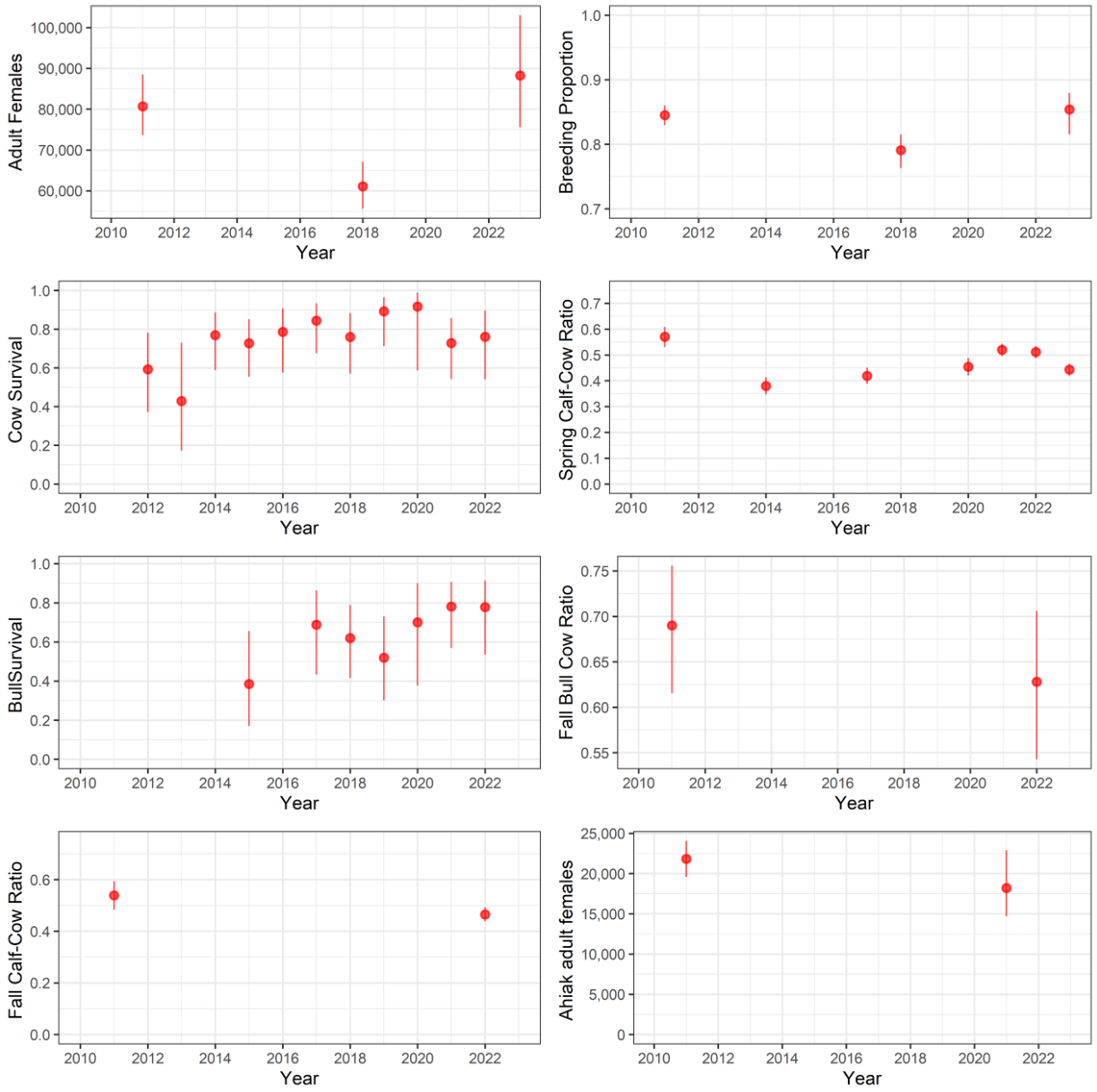


Figure 52. Field data used as input into the Integrated Population Model.

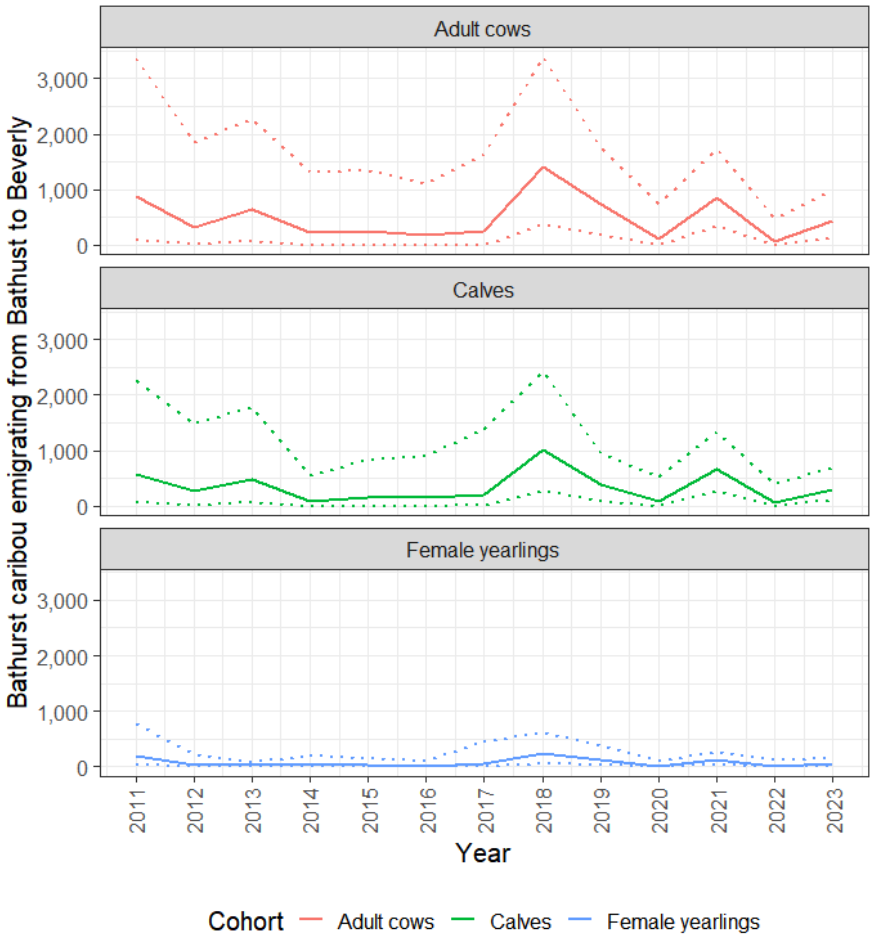


Figure 53. Estimated number of Bathurst caribou emigrating to Beverly based on IPM developed for the Bathurst herd (Boulanger et al 2024). The number of Bathurst adult cows emigrating is based on collar movements and other IPM data sources. The number of calves emigrating is the product of cows emigrating X the estimated pregnancy rate for a given year. The number of female yearlings emigrating assumes similar emigration rates for adult females. Dotted lines are the 95% confidence limits on predictions.

4.7.2 Model scenarios considered

The primary questions addressed by the IPM analysis were whether the observed increase in Beverly herd abundance could be due to demographic factors (survival and productivity) in comparison to effects of potential emigration from the neighboring Ahiak herd to the east, and Bathurst herd to the west. Three model scenarios were run to address these questions:

1. *Trend in Beverly due to demographic factors only:* For this model, trend and abundance in 2023 was estimated based solely on adult female and male survival rates and productivity from composition surveys as shown in **Figure 52**.
2. *Trend in Beverly due to demography + movement between Ahiak and Beverly:* For this model run the effect of emigration and immigration of adult females and accompanying calves to and from the Ahiak herd was added to the base demographic model. Demography (survival and productivity) of the Ahiak herd was assumed to be similar to the Beverly herd. The actual trend in the Ahiak was further constrained by estimates in 2011 and 2021 (**Figure 52**).
3. *Trend in Beverly due to demography + movement between Ahiak and Beverly and movement from the Bathurst herd:* The effect of movement of adult females and accompanying calves from the Bathurst herd (**Figure 53**) was then added to the Beverly-Ahiak demographic model developed in step 2. The actual Bathurst IPM was not incorporated into the analysis, only outputs were used. As discussed later, future models could incorporate the Bathurst IPM to better consider likely non-independence of movements between herds.
4. *Trend in Beverly due to demography + movement between Ahiak and Beverly and movement from the Bathurst herd + yearlings emigrating:* Scenario 3 was extended with the allowance of yearlings (in addition to adult females and accompanying calves) moving from the Beverly, Bathurst, and Ahiak calving grounds.

The main metrics used to compare scenarios are estimates of adult females and breeding females which are the main cohorts monitored by calving ground surveys as well as other demographic indicators. Adult females are proportional to overall herd size and therefore are an indicator of overall herd trend. We also show the fit to breeding females (adult females X proportion adult females breeding) which corresponds to the core segment of caribou targeted during calving ground surveys and therefore may be the best group to compare IPM predictions with field estimates.

*Base demographic model (**scenario 1**)*

The fit of the field data to the base Beverly demographic model was adequate with all IPM estimates (blue lines in **Figure 54**) falling within confidence limits of field estimates (red dots and confidence lines). The ability of a herd to increase depends on whether caribou are surviving (adult, yearling, and calf survival) and whether productivity (fecundity and calf survival) is high enough to offset caribou mortality. Adult female survival rates were approximately 0.8 in most years which is adequate for herd stability but will limit herd increase (Boulanger at all 2011). Productivity was high as indicated by calf-cow ratios of 0.5 or greater in most years as well as higher fecundity values. Fecundity or pregnancy rate was modelled for cows aged 2 and aged 3 and older therefore accounting for potential age structure effects on productivity with lower pregnancy rates for younger cows (**Figure 55**). Calf survival was 0.6 or higher for most years once again suggesting higher productivity.

An additional source of mortality was from the harvesting of adult cows and bulls (**Figure 56**). There are no solid estimates of harvest of caribou for the Beverly herd within either Nunavut, Northwest Territories, or Saskatchewan, the main harvesters of the herd. We assumed nominal levels of harvest with high uncertainty for females and males. This allowed the model to estimate harvest based on other data sources.

All estimates developed for the abundance of Beverly Herd adult females suggests a reasonable fit to all data points except the June 2023 adult and breeding female abundance estimates (this report), where the IPM estimate is below or just meets the lower confidence limit from the field estimate (**Figure 57**). We also show the fit to breeding females (adult females X pregnancy rate) which are more likely to include the core segment of older adult females which displays a slightly better fit than adult females. These results suggest that a demographic increase was plausible, however, demography alone does not entirely explain the higher 2023 estimate.

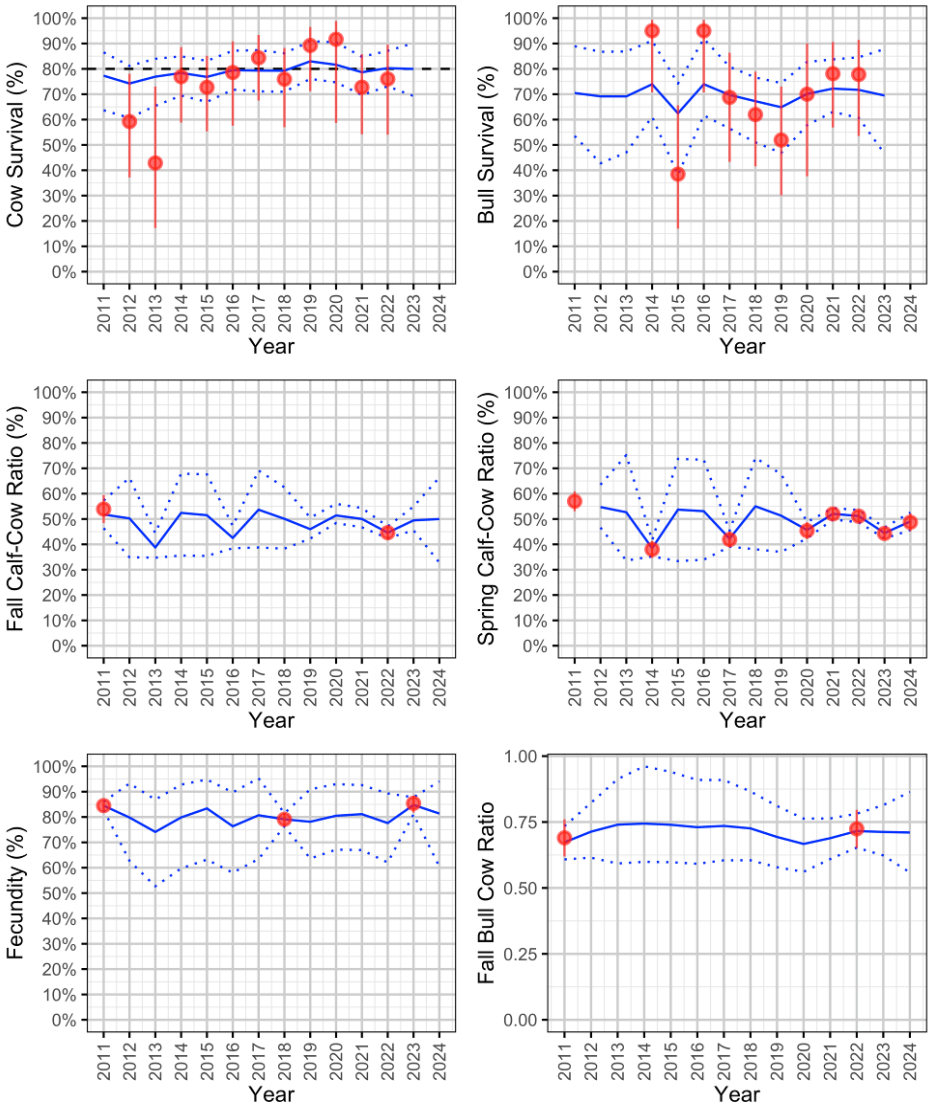


Figure 54. Fit of base demographic model to field measurements of Fecundity, fall composition, Cow survival, Bull survival, Spring composition, and Fall bull to cow ratio. Dotted lines are 95% confidence limits on predictions.

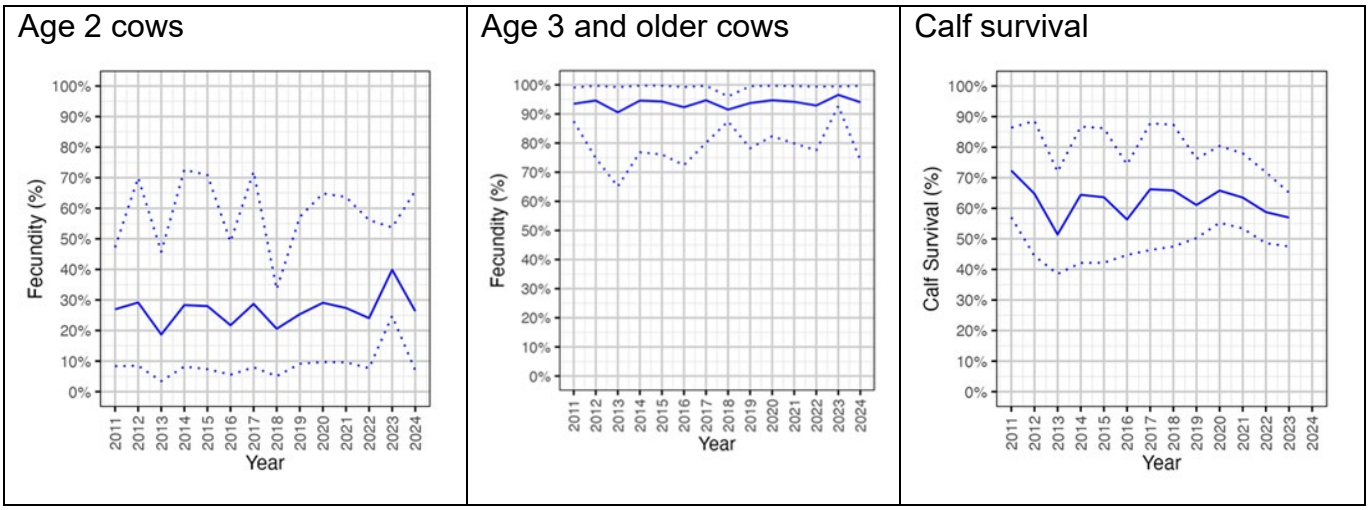


Figure 55. Estimates of fecundity (pregnancy rate) for cows age 3. Dotted lines are 95% confidence limits on predictions.

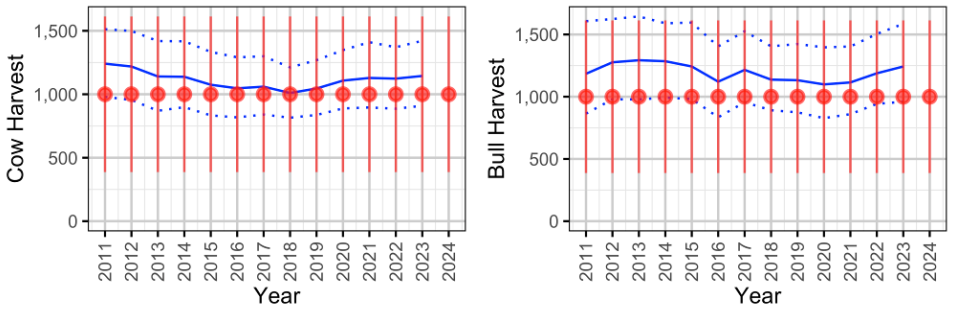


Figure 56. Estimates of harvest from IPM (blue line) compared to rough field estimates. Dotted lines are 95% confidence limits on predictions.

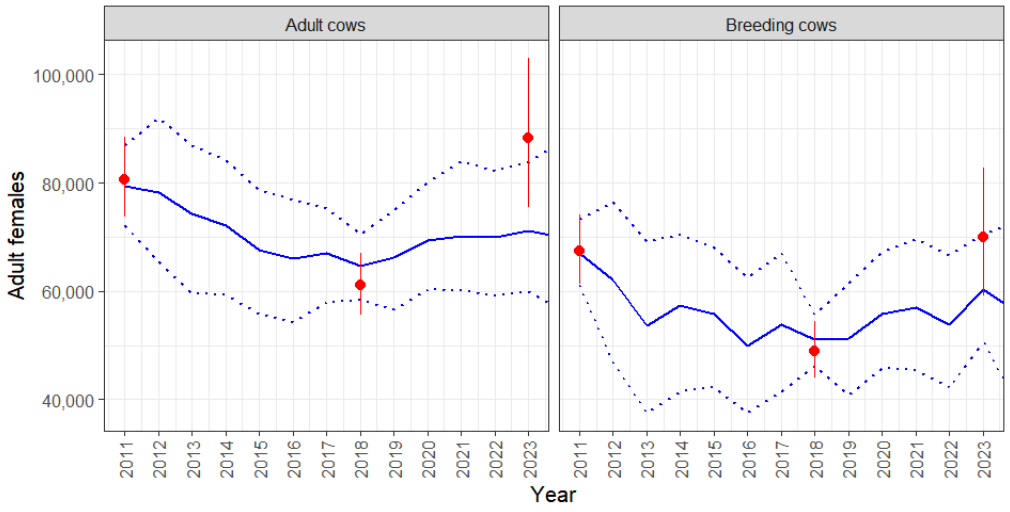


Figure 57. Estimates of adult and breeding females from the base Beverly demographic model. Dotted lines are 95% confidence limits on predictions.

*Base demographic model+Ahiak (**scenario 2**)*

IPM Survival and productivity rates from the base Beverly model were applied to the Ahiak herd to fit estimates of the Ahiak cows in 2011 and 2021. The resulting model predictions fit the field estimates well suggesting that sharing of demographic parameters between the Beverly and Ahiak was a reasonable strategy.

Estimates of % movement between the Beverly and Ahiak (and vice versa) was based on yearly collar estimates (**Figure 58**). Given low sample sizes, estimates for the Ahiak to Beverly movements were variable due to low sample sizes of collars (**Figure 46**) leading to very wide confidence limits on yearly estimates (**Figure 59**). For example, most estimates of movement for the Ahiak to Beverly were based on 2-5 known collars each year. In this case the IPM model gave very little weighting to these yearly estimates. Mean IPM estimates of movement were approximately 5% for Beverly to Ahiak and 30% for Ahiak to Beverly. Unfortunately, the IPM models suggest that there is only enough information to indicate a mean trend from these data sets.

We note that the relative difference in herd sizes (Beverly adult females being approximately 4.5 times more abundant than Ahiak adult females) means that the higher movement rate from Ahiak to Beverly does not necessarily mean that the net movement of caribou from the Ahiak calving-ground to the Beverly calving-ground was higher. To illustrate this, the relative number of adult females between calving grounds was estimated for each herd (**Figure 61**). In most years, a higher number of Ahiak females (black line) moved into the Beverly calving-ground compared to Beverly moving into the Ahiak calving-ground (brown line). However, when summed across years, the net movement of Ahiak caribou into the Beverly calving-ground involved approximately 3,600 Ahiak caribou. As a result, the estimate of adult females in June 2023 for the *demography + Ahiak herd* model was slightly higher than the *demography alone* model, suggesting a slight increase in abundance could be attributed to the movement of Ahiak herd females from the Ahiak calving-ground into the Beverly calving-ground (**Figure 61**).

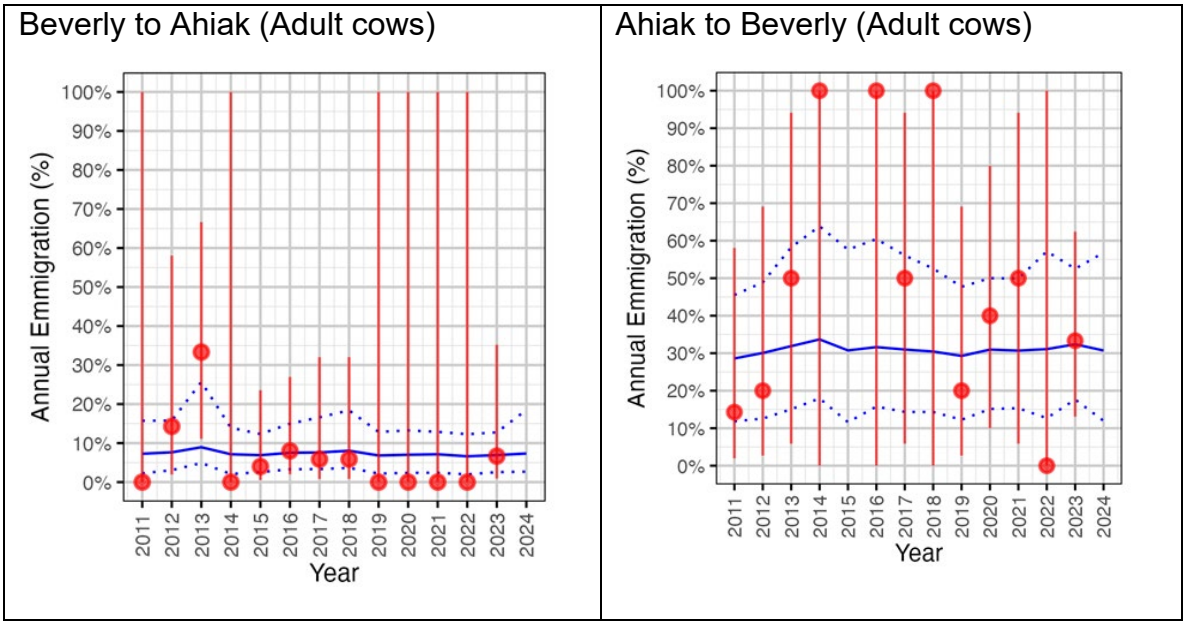


Figure 58. Estimates of percent (%) movement from Beverly to Ahiak (left) and Ahiak to Beverly (right) by year. Dotted lines are the 95% confidence limits on predictions.

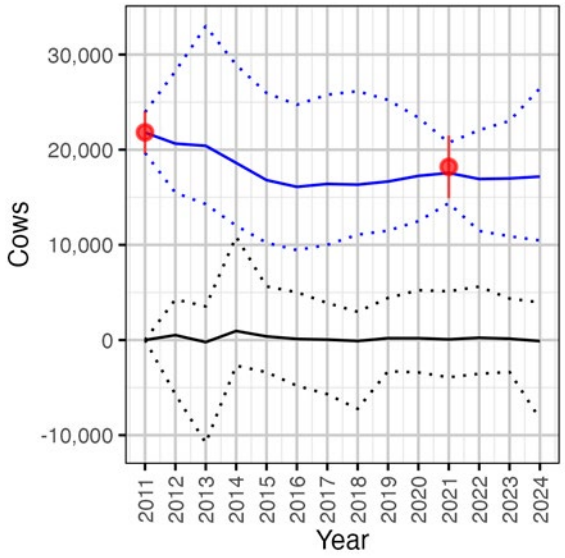


Figure 59. Estimates of adult females for the Ahiak herd (blue line) compared to field estimates. Also shown are estimates of emigration of the Ahiak to the Beverly herd for each year (black lines). Dotted lines are the 95% confidence limits on predictions.

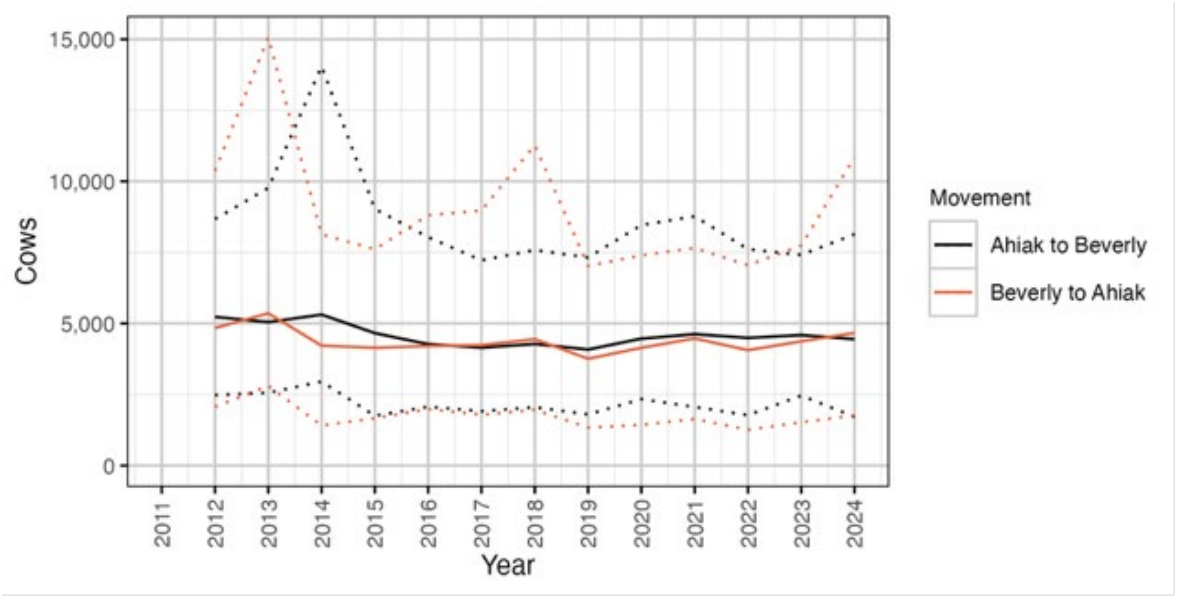


Figure 60. Estimates of adult females moving between Ahiak and Beverly calving grounds each year. Dotted lines are the 95% confidence limits on predictions.

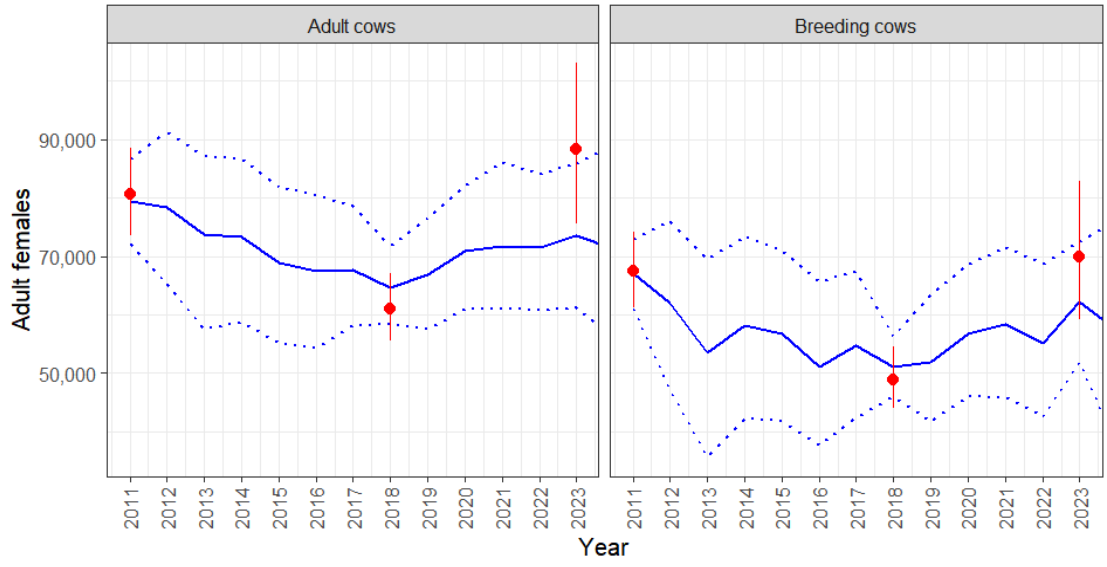


Figure 61. Estimates of adult and breeding females based on demography and movement between Ahiak and Beverly calving grounds. Dotted lines are the 95% confidence limits on predictions.

*Demography+Ahiak+Bathurst (**scenario 3**)*

The estimate of adult females entering the Beverly calving-ground from the Bathurst calving-ground was based solely on the Bathurst IPM as shown in **Figure 53**. A zoomed in version of adult females emigrating into the Beverly calving-ground is shown in **Figure 62**. Roughly 500-1,000 adult females were estimated to have emigrated into the Beverly herd since 2011. The net result of this IPM was a slight increase in the estimated abundance of adult female Beverly caribou in 2023 (**Figure 63**).

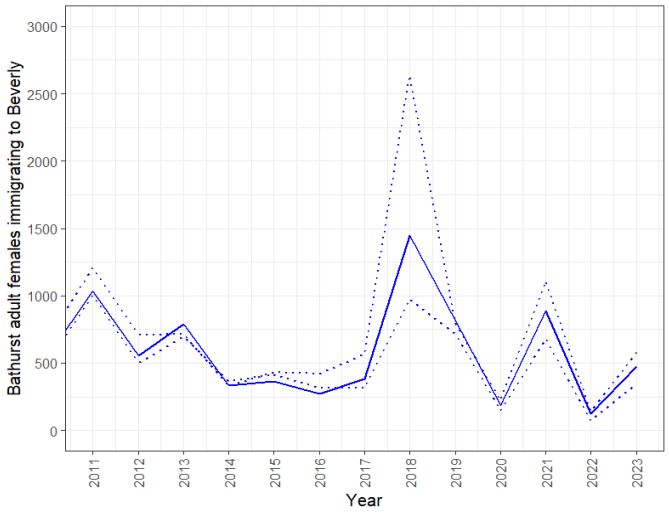


Figure 62. Estimate of Bathurst cows emigrating to the Beverly herd. Dotted lines are the 95% confidence limits on predictions.

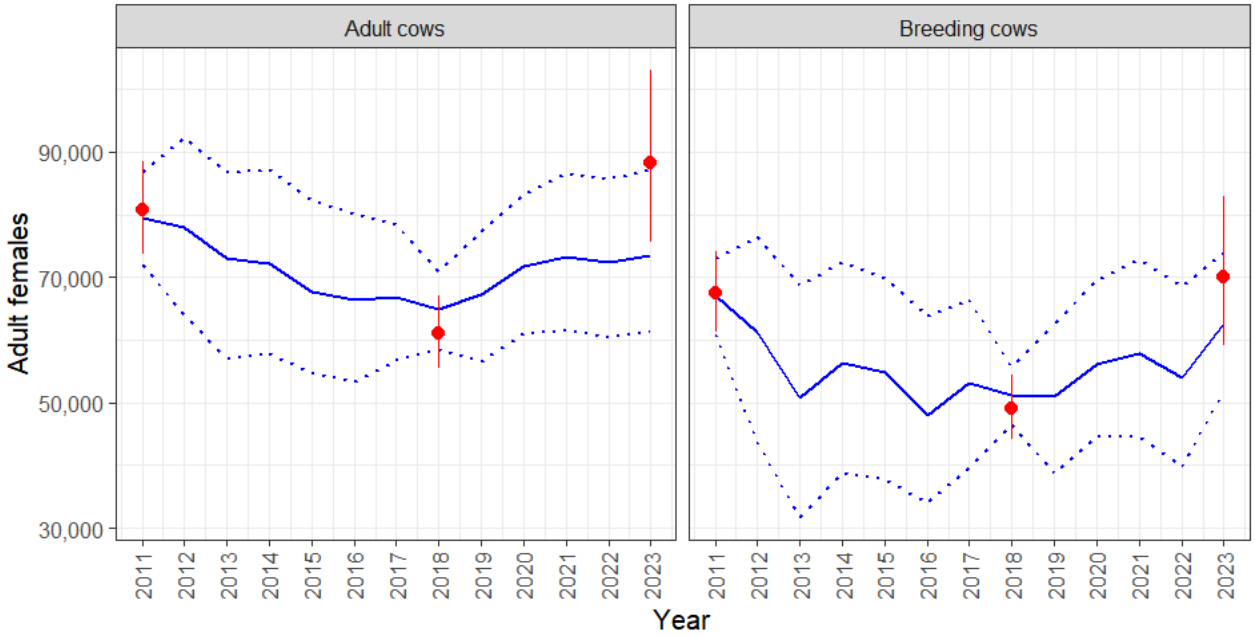


Figure 63. Estimates of Beverly adult and breeding females from the demography+Ahiak+Bathurst model. Red symbols = abundance estimates and associated 95% confidence intervals. Dotted lines are the 95% confidence limits on predictions.

Demography+Ahiak+Bathurst with female yearlings emigrating (scenario 4)

An additional scenario was run where female yearlings followed adult females to the calving grounds (at the same rate of fidelity/movement as the adult females). There was minimal difference in adult female estimates due to the fact that Ahiak and Beverly movement was roughly equal so the net movement would have mainly been yearling from the Bathurst herd **(Figure 64).**



Figure 64. Estimates of Beverly adult and breeding females from the demography+Ahiak+Bathurst model with yearlings emigrating. Red symbols = abundance estimates and associated 95% confidence intervals. Dotted lines are the 95% confidence limits on predictions.

4.7.3 Comparison of estimates from IPM model scenarios

Comparisons of IPM scenario estimates of adult females in 2023 suggests a slight increase in estimates can be attributed to movement of caribou from the Ahiak and Bathurst herds with IPM estimates intercepting the lower confidence estimates of field estimates when movement was considered (**Figure 65**). The fit of the IPM to the breeding female estimate in 2023 is slightly better than adult females. One potential reason for this is that the breeding females have a higher probability of all occurring within the surveyed calving ground area. In contrast a small proportion of adult females may not be in the survey due to non-breeder status or inability to travel to the core calving ground area. Regardless, from a statistical perspective this result suggests that movement and demography could explain the unexpected increase in magnitude of the 2023 abundance estimate when compared to the 2018 abundance estimate. As discussed later, further enhancements to the IPM could be considered to more fully consider each scenario and its impact on the increase in Beverly herd abundance documented between June 2018 and June 2023.

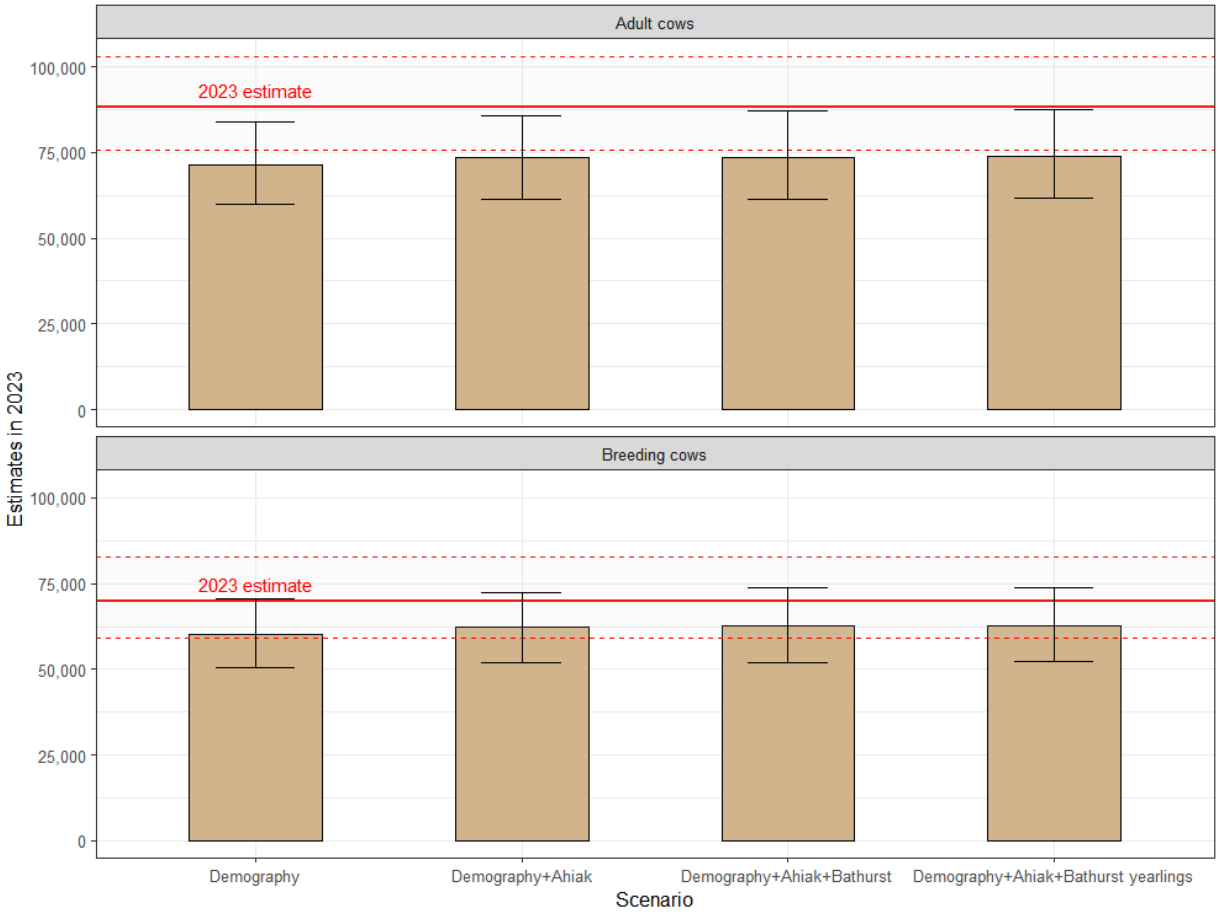


Figure 65. Comparison of estimates of adult and breeding females in 2023 from 4 IPM scenarios. Dotted lines are the 95% confidence limits on predictions.

5.0 CONCLUSIONS

5.1 GENERAL COMMENTS

Results from the Beverly 2023 survey suggest that the herd has increased significantly since the last survey in 2018 with an annual rate of increase of 8% (CI. 4-12%) for adult females. Demographic analyses suggest that this increase is likely due to higher productivity of caribou as well as potential movement from the neighboring Bathurst and Ahiak calving ground areas. Reconnaissance survey densities (**Figure 42**) and group sizes of caribou observed were larger than in 2018 also indicating an increase in overall herd size.

One of the main challenges of the survey was large group sizes that had not been observed in previous surveys, and were difficult to count using solely observers. We developed a video-assisted approach to obtain counts of caribou in high density strata which successfully corrected visual count estimates (**Figure 30**). The survey also occurred during a time of slightly higher movement levels (7 to 10 km per day). Inspection of net movements does not reveal any movements from high density strata when they were sampled. However, a group of collars did move into the central low from high east before the central low was sampled therefore inflating densities and abundance estimates in this stratum (**Figure 24**). Therefore, estimates from this stratum were excluded in the analysis.

The herd size estimate based on the summation of bulls and cows on the calving ground survey area is 6% higher (≈10,000 caribou) than the extrapolated herd estimate. This difference could simply be due to survey imprecision given overlap of confidence limits. The location of bulls on the calving ground in 2023 was unique compared to other survey years (**Figures 38 and 39**). It could be argued that the summation of bulls and cows provides a more direct estimate of herd size than the extrapolation estimate especially given challenges of assessing herd membership of bulls and cows during the fall composition surveys. However, in most years bulls are south of the calving ground survey area and therefore it is not possible to estimate herd size using the summation of bulls and cows.

5.2 HERD DEMOGRAPHY AND MOVEMENTS FROM OTHER HERDS

The Integrated Population Model analysis provided a method to assess various factors that potentially caused the increase in abundance. Overall analysis results suggest that while IPM estimates were lower than the 2023 field estimates, the confidence limits did overlap between field and IPM estimates suggesting results were not statistically different (**Figure 65**). IPM results suggested an increase in the Beverly which was due to both demographic factors and movements from adjacent calving grounds.

A demographic increase was suggested which was primarily due to higher productivity as indicated by higher calf cow ratios as opposed to higher survival rates. Relatively high productivity levels are suggested for the Beverly which was one of the main drivers of demographic increase given that the survival levels were

moderate. Analysis of Bluenose-East and Bathurst herds demography relative to environmental covariates during this time period did suggest increased productivity due to cooler and wetter spring and summers (Boulanger et al. 2024). This type of analysis could also be repeated (using MERRA covariates) to assess if productivity was related to range and environmental conditions.

Field estimates of survival rate were averaging 0.8 which is low for an increasing herd. Based on a matrix model analysis, mean adult female survival levels ranging from 0.85 to 0.92 are required to produce an 8% annual increase in population size (Boulanger et al. 2024). One potential issue is that collar survival rates have been shown in some studies to be biased low (Rasiulis et al. 2014). The IPM partially offsets this issue by using other data sources to estimate survival in the herd. However, some data sources such as abundance estimates are sparse (3 surveys for the entire analysis) compared to annual survival rate estimates, and therefore the influence of these data sources is limited. We suspect that IPM estimates of survival would potentially increase if more indicators of positive growth, such as the 2023 survey estimate, were present in the data set. We also note that the use of environmental covariates, as was done for the Bathurst and Bluenose-East herds (Boulanger et al. 2022) could better inform variation in survival rates than the simple random effects model used in this analysis.

Movements of caribou from the Bathurst and Ahiak also contributed to the increase of the Beverly. Combinations of Ahiak and Beverly movement suggest that movements were relatively balanced between the 2 calving ground areas with only a slight net movement into the Beverly calving-ground (**Figure 60**). However, this still had a small effect on trend. One challenge is the low number of collared caribou available to estimate movement which led to very imprecise estimates of movement (**Figure 46**). The IPM estimated roughly stable movements of Ahiak to Beverly (**Figure 58**: i.e. 30%) which was offset by movements of Beverly to the Ahiak (**Figure 60**). The lack of variation detected in yearly movement rates may have been due to sparse data obscuring trends in the data set. The Bathurst also contributed to the

increase; however, the amount was limited due to the relatively small size of the Bathurst herd (**Figure 62**).

One of the largest challenges of understanding movements and demography of caribou is the limited sample size and selective sampling associated with collared caribou. In general, collaring targets adult caribou and therefore survival, movements, and locations of younger caribou (2-3 years old) is not indicated within collar data. This issue combined with limited sample size of collars challenges estimates of movement rate between herds. For example, a power analysis conducted by Boulanger et al. (2024) suggested that power of collared caribou to switch between calving grounds was low unless collar sample sizes of caribou that had known herd membership (calved the previous year on a calving ground) was at least 20 to 30 caribou. This sample size often is not achieved for many herds (**Table 19**). This issue and the increasing overlap of herds on winter ranges which can lead to higher switching rates (Boulanger et al. 2024) creates an additional source of variation in herd trend and status.

The IPM approach attempts to use all the data sources available to estimate and understand herd status and trend. In this survey we combined the Ahiak and Beverly under the assumption of similar underlying demography between the 2 herds. Estimates from the Bathurst IPM were also used to estimate likely movement of the Bathurst into the Beverly. Eventually adding the Bathurst IPM into the Beverly IPM would more efficiently model movements between the 2 herds by considering likely covariance in movements between the 2 herds. For example, the increase in the Beverly and decrease in the Bathurst would be used to further inform and estimate movement/switching rates between calving grounds.

5.3 SURVEY METHODOLOGY

The 2023 survey was unique from previous surveys in that group sizes were too large in high density strata to allow visual counting creating a potential counting bias in estimates. This contrasts with previous surveys (i.e. 2018) where there may have been sightability bias due to poor survey conditions.

5.3.1 Video approach to confront large group sizes in 2023

The video/orthophoto approach provided a way to allow a secondary count of larger group sizes in high density strata. One challenge with this approach was that it was not possible to link visual observations with video observations. In addition, the video counts were sporadic meaning that survey lines contained both visual and video counts. We developed an iterative method that allowed pairing of observations with video counts within each orthophoto polygon. The main finding from this comparison is that visual counts underestimated larger groups (**Figure 27**), however, this was offset by confining counts to within the survey strip. Once the survey strip was considered, visual counts were higher than the orthophoto counts (**Figure 28**). So, this finding suggests that visual counts overestimated caribou within the survey strip when group size was very large.

5.3.2 Sightability issues in previous surveys

Survey conditions were less than optimal in 2018 which may have resulted in a negatively biased estimate if there were groups of caribou with very low sighting probabilities. While it is not possible to ascertain bias from field surveys, a few indirect sources indicate potential for slight negative biases. IPM estimates for adult

cows in 2018 were 6% higher than the 2018 field estimates suggesting a potential negative bias in 2018 (**Figure 57**), however differences between the IPM and field estimate were within confidence limits of the field estimate so this difference could be due to statistical uncertainty. Empirical comparison of double observer and distance sampling estimates from past surveys suggest that double observer approach may underestimate abundance by 5-10% as detailed in Appendix C. Other comparisons also suggest that double observer estimates may be biased low. A comparison of paired photo and double observer survey lines during the Bluenose-East surveys in 2013 and 2023 surveys suggested that photo estimates were 7 to 14% higher, however, further analysis suggested that the main bias occurred on lines with higher densities of caribou where photo counts were higher with similar estimates at moderate densities (Boulanger et al. 2024).

The dependent double observer method attempts to account for lower sightability by estimating probability of detection of groups. However, one assumption of this approach is that caribou groups have equal sightability and any unequal sightability is accounted for by the use of covariates such as group size. Unaccounted heterogeneity (unequal sightability) could be caused by poor survey conditions causing some caribou groups to being easily observed with others showing low sightability. A particular challenge are ”salt and pepper” conditions which can cause some caribou to have much reduced sighting probabilities. Finally, unequal observer probabilities can create bias especially if the observers do not switch during the survey.

We note that negative biases (of 5-10%) in strip transect/double observer estimates from 2018 cannot explain the increase documented in the 2023 survey. Use of the IPM provides a way to offset biases with any particular technique. For example, the IPM estimate of herd size in 2018 (**Figure 57**) which is based on all data sources and provides a potentially improved estimate of herd size since it considers both past and future data points, and from this, estimates the most likely herd size.

We propose a hybrid distance sampling/double observer approach as detailed in Appendix C that will be more robust to sightability bias. This approach which involves also collecting binned distance data, should be possible to implement with relatively minor changes in field protocol. The main premise behind this approach is that distance sampling approach will be more robust to unequal sighting probabilities between caribou groups. Using this approach in combination with double observer methods should provide more robust survey estimates especially when sighting conditions are marginal. We also note that the video method developed in this report also can be used to cross check sightability and counts from observers.

6.0 RECOMMENDATIONS

It is suggested that future surveys consider surveying the core Bluenose East, Bathurst, Beverly, and Ahiak calving grounds within the same survey year. Both collar and IPM analyses demonstrate varying movement between these calving areas which complicates interpretation of stand-alone abundance estimates of these herds. If funding is limited, then the Beverly and Ahiak should be surveyed during the same year given the close linkage between caribou in these 2 herds.

One result from this survey is that it may not be possible to rely on visual only methods in future surveys if higher densities of caribou are observed. The video assistance approach developed in this project should be further developed to allow better linkage between observer and video observations. This could be done by Bluetooth linkage of phones or video recording devices with tablets used for observations. Alternatively, and with appropriate financial resources, photo planes could be stationed on site to photo survey areas of high density as is done for the Qamanirjuaq, Bathurst, and Bluenose East herds. Maintaining aircraft on standby outside of Nunavut, as in this survey, carries with it availability risks and associated holding costs that could be logistically and financially restrictive.

A reduced distance sampling double observer approach, as detailed in Appendix C, should be considered, to allow estimates with greater robustness from visual surveys. To implement this approach, we propose a 600m strip would be surveyed with 3 distance bins (0-200m, 200-400m, 400-600m). The main change from the present methodology would be a slightly wider survey strip. The only additional piece of information that would be required is what survey bin the majority of any

caribou group occurs in. Also, observers would need to be reminded to pay the most attention to the closest survey bins.

In this study we developed an Integrated Population Model to help assess herd demography and trend. The current model utilizes a random effect approach to model temporal variation in demographic parameters. Environmental covariates could be used to refine the modelling of temporal variation as has been done for the Bathurst and Bluenose East herds. In addition, one finding of this analysis and the collar movement analysis is that there is likely some linkage between the Ahiak, Beverly, Bathurst, and Bluenose East herds given overlap between their winter ranges. This likely causes movement which may not be detected by examining collared caribou alone given low sample sizes as well as lack of representation of younger age classes. Both the Bluenose-East and Beverly herds exhibited increases that could not be easily explained by demography alone and one possible explanation is movement between the herds. One method to better assess movement and demography would be to link the Bluenose-East, Bathurst, Beverly, and Ahiak integrated population model. This model would then allow joint modelling of movements between all 4 calving grounds which would provide a more comprehensive assessment of the effect of movement as well as demography of all 4 herds.

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9.0 APPENDICES

9.1 APPENDIX A – INTEGRATED POPULATION MODEL DETAILS

The suggested citation for this analytic appendix is:

Thorley, J.L. and Boulanger, J. (2024) Beverly Caribou Herd Population Analysis 2024. A Poisson Consulting Analysis Appendix. (www.poissonconsulting.ca)

Background

Monitoring surveys of the Beverely barren-ground caribou (*Rangifer tarandus groenlandicus*) herd have been conducted intermittently since 2011. The surveys include June calving ground aerial surveys (Boulanger et al. 2017), as well as fall and spring sex and age composition surveys and monitoring of cows with satellite or GPS collars.

Data Preparation

The estimates of key population statistics with standard errors (SEs) were provided in the form of an csv spreadsheet and prepared for analysis using R version 4.4.1 (R Core Team 2024).

Statistical Analysis

Model parameters were estimated using Bayesian methods. The estimates were produced using JAGS (Plummer 2003). For additional information on Bayesian estimation the reader is referred to McElreath (2020).

Unless stated otherwise, the Bayesian analyses used weakly informative prior distributions (Gelman, Simpson, and Betancourt 2017). The posterior distributions were estimated from 1500 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of 3 chains (Kery and Schaub 2011,

38–40). Model convergence was confirmed by ensuring that the potential scale reduction factor $\hat{R} \leq 1.05$ (Kery and Schaub 2011, 40) and the effective sample size (Brooks et al. 2011) ESS ≥ 150 for each of the monitored parameters (Kery and Schaub 2011, 61).

Where computationally practical, the sensitivity of the posteriors to the choice of prior distributions was evaluated by doubling the standard deviations of the priors and then using $\hat{R}$ to evaluate whether the samples were drawn from the same posterior distribution (Thorley and Andrusak 2017).

The parameters are summarized in terms of the point *estimate*, *lower* and *upper* 95% compatibility limits (Rafi and Greenland 2020) and the surprisal *s-value* (Greenland 2019). Together a pair of lower and upper compatibility limits (CLs) are referred to as a compatibility interval (CI). The estimate is the median (50th percentile) of the MCMC samples while the 95% CLs are the 2.5th and 97.5th percentiles. The s-value indicates how surprising it would be to discover that the true value of the parameter is in the opposite direction to the estimate (Greenland 2019). An s-value of > 4.32 bits, which is equivalent to a p-value < 0.05 (Kery and Schaub 2011; Greenland and Poole 2013), indicates that the surprise would be equivalent to throwing at least 4.3 heads in a row on a fair coin.

The results are displayed graphically by plotting the modeled relationships between individual variables and the response with the remaining variables held constant. In general, continuous and discrete fixed variables are held constant at their arithmetic mean and first level values, respectively, while random effects are held constant at their typical value (Kery and Schaub 2011, 77–82). Unless stated otherwise the typical value is the arithmetic mean. When informative the influence of a particular variables is expressed in terms of the *effect size* (i.e., relative change in the response variable) with the 95% CI (Bradford, Korman, and Higgins 2005).

The analyses were implemented using R version 4.4.1 (R Core Team 2024) and the embr family of packages.

Model Descriptions

The data were analyzed using a state-space population model (Newman et al. 2014).

Population

The cow and bull harvest, fecundity, breeding cow abundance, cow survival, bull survival, fall bull cow, fall calf cow and spring calf cow ratio data complete with standard errors were analyzed using a stage-based state-space population model similar to Boulanger et al. (2011). Key assumptions of the female stage-based state-space population model include:

- Calving occurs on the 11th of June (with a year running from calving to calving)
- Cow natural survival from calving to the following year varies continually and randomly by year.
- Bull natural survival from calving to the following year varies randomly by year.
- Cow and bull natural survival is constant throughout the year in any given year.
- Harvest of cows and bulls occurs on the 15th of January.
- Yearling survival to the following year is the same as cow natural survival.
- Calf survival varies between the summer and winter seasons and randomly by year.
- The calf sex ratio is 1:1.
- Fecundity varies randomly by year.
- In an average year two-year-old cows have a fecundity of 0.25.
- Female yearlings are indistinguishable from cows in the fall and spring surveys.
- The uncertainty in the number of cows in the initial year is described by a positively truncated normal distribution with a mean of 75,000 and a standard deviation of 25,000.
- The number of bulls in the initial year relative to the number of cows is described by a beta distribution with an alpha of 11 and a beta of 11.
- The number of calves in the initial year relative to the number of cows is described by a beta distribution with an alpha of 19 and a beta of 3.
- The number of yearlings in the initial year relative to the number of cows is described by a beta distribution with an alpha of 11 and a beta of 11.
- The number of two-year-old cows in the initial year relative to the number of cows is described by a beta distribution with an alpha of 3 and a beta of 19.
- The uncertainty in each data point is normally distributed with a standard deviation equal to the provided standard error.

Further details on the model can be obtained upon request from the authors.

Tables

Population

Table 1. Parameter descriptions.

Parameter	Description
Annual	The year as a integer starting at 1
BreedingCowsSE[i]	The standard error for BreedingCows[i]
BreedingCows[i]	The data point for the number of breeding cows in the i^{th} year
BreedingProportionSE[i]	The standard error for BreedingProportionSE[i]
BreedingProportion[i]	The data point for the log-odds proportion of cows breeding in the i^{th} year
BullSurvivalSE[i]	The standard error for BullSurvival[i]
BullSurvival[i]	The data point for log-odds bull survival from the $i\text{-}1^{th}$ year to the i^{th} year
CowSurvivalSE[i]	The standard error for CowSurvival[i]
CowSurvival[i]	The data point for log-odds cow survival from the $i\text{-}1^{th}$ year to the i^{th} year
FallBullCowSE[i]	The standard error for FallBullCow[i]
FallBullCow[i]	The data point for the bull cow ratio in the fall of the i^{th} year
FallCalfCowSE[i]	The standard error for FallCalfCow[i]
FallCalfCow[i]	The data point for the calf cow ratio in the fall of the i^{th} year
HarvestedBullsSE[i]	The standard error for HarvestedBulls[i]
HarvestedBulls[i]	The data point for the number of bulls harvested in January of the i^{th} year
HarvestedCowsSE[i]	The standard error for HarvestedCows[i]
HarvestedCows[i]	The data point for the number of cows harvested in January of the i^{th} year
SpringCalfCowSE[i]	The standard error for SpringCalfCow[i]
SpringCalfCow[i]	The data point for the calf cow ratio in the spring of the i^{th} year
bBreedingCows1	The number of breeding cows in the initial year
bBullsCows1	The bull to cow ratio in the initial year
bFecundityCow	The log-odds proportion of three year and older cows breeding in a typical year
bFecundityTwolingCow	The log-odds proportion of two year old cows breeding in a typical year
bHarvestRateBull	The log-odds probability of a bull being harvested in a typical year
bHarvestRateCow	The log-odds probability of a cow being harvested in a typical year

Parameter	Description
bSurvivalBullAnnual[i]	The random effect of the i^{th} Annual on bSurvivalBull
bSurvivalBull	The log-odds bull survival in a typical year
bSurvivalCalfAnnual[i]	The random effect of the i^{th} Annual on bSurvivalCalfSummerAnnual and bSurvivalCalfWinterAnnual
bSurvivalCalfSummerAnnual	The log-odds summer calf survival if extended for one year
bSurvivalCalfWinterAnnual	The log-odds winter calf survival if extended for one year
bSurvivalCowAnnual[i]	The random effect of the i^{th} Annual on bSurvivalCow
bSurvivalCow	The log-odds cow (and yearling) survival in a typical year
sSurvivalBullAnnual	The SD of bSurvivalBullAnnual
sSurvivalCalfAnnual	The SD of bSurvivalCalfAnnual
sSurvivalCowAnnual	The SD of bSurvivalCowAnnual

Table 2. Model coefficients.

term	estimate	lower	upper	svalue
bBullsCows1	0.597	0.493	0.707	10.55
bCalvesCows1	0.755	0.685	0.824	10.55
bCows1	79229.291	72304.779	86920.280	10.55
bFecundityCow	2.853	1.778	5.032	10.55
bHarvestRateBull	-3.529	-3.808	-3.250	10.55
bHarvestRateCow	-4.127	-4.343	-3.940	10.55
bSurvivalBull	0.784	0.436	1.249	8.23
bSurvivalCalfSummerAnnual	-0.325	-0.841	0.398	1.66
bSurvivalCalfWinterAnnual	1.720	0.560	4.229	10.55
bSurvivalCow	1.442	1.161	1.738	10.55
bTwolingsCows1	0.136	0.043	0.257	10.55
bYearlingsCows1	0.570	0.372	0.748	10.55
sFecundityAnnual	0.684	0.239	1.901	10.55
sSurvivalBullAnnual	0.347	0.023	1.113	10.55

term	estimate	lower	upper	svalue
sSurvivalCalfAnnual	0.452	0.085	1.041	10.55
sSurvivalCowAnnual	0.352	0.051	0.795	10.55

Table 3. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
14	16	3	500	100	702	1.01	TRUE

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9.2 APPENDIX B-MULTI-STATE MODEL ESTIMATES OF MOVEMENT BETWEEN CALVING GROUNDS.

Period	Current year calving ground	Previous year calving ground	events	Estimate	LCI	UCI
2011-18	Ahiak	Ahiak	6	0.418	0.124	0.696
2011-18	Ahiak	Beverly	9	0.085	0.045	0.155
2011-18	Ahiak	Lorillard	1	0.061	0.02	0.173
2011-18	Ahiak	Wager Bay	3	0.064	0.021	0.18
2011-18	Bathurst	Bathurst	82	0.965	0.896	0.989
2011-18	Bathurst	Beverly	0	0	0	0
2011-18	Beverly	Ahiak	7	0.487	0.259	0.721
2011-18	Beverly	Bathurst	3	0.035	0.011	0.104
2011-18	Beverly	Beverly	101	0.9	0.804	0.953
2011-18	Lorillard	Lorillard	25	0.837	0.604	0.937
2011-18	Lorillard	Wager Bay	4	0.213	0.118	0.352
2011-18	Wager Bay	Ahiak	3	0.095	0.036	0.228
2011-18	Wager Bay	Lorillard	1	0.102	0.043	0.223
2011-18	Wager Bay	Wager Bay	8	0.723	0.468	0.861
2019-23	Ahiak	Ahiak	17	0.615	0.422	0.845
2019-23	Ahiak	Beverly	1	0.015	0.002	0.098
2019-23	Ahiak	Lorillard	2	0.061	0.02	0.173
2019-23	Ahiak	Wager Bay	0	0.064	0.021	0.18
2019-23	Bathurst	Bathurst	90	0.886	0.819	0.939
2019-23	Bathurst	Beverly	2	0.027	0.004	0.074
2019-23	Beverly	Ahiak	8	0.29	0.153	0.48
2019-23	Beverly	Bathurst	12	0.114	0.061	0.181
2019-23	Beverly	Beverly	64	0.958	0.828	0.994
2019-23	Lorillard	Lorillard	16	0.837	0.604	0.937
2019-23	Lorillard	Wager Bay	6	0.213	0.118	0.352
2019-23	Wager Bay	Ahiak	1	0.095	0.036	0.228
2019-23	Wager Bay	Lorillard	4	0.102	0.043	0.223
2019-23	Wager Bay	Wager Bay	26	0.723	0.468	0.861

9.3 APPENDIX C-- USE OF A REDUCED DISTANCE SAMPLING AND DOUBLE OBSERVER METHODOLOGY TO IMPROVE ROBUSTNESS OF CALVING GROUND SURVEY ESTIMATES

Introduction

This analysis explores the use of a reduced distance sampling combined with double observer methodology to improve the estimation of sightability and subsequent abundance estimates from calving ground surveys. Some of the recent calving ground surveys have been challenged by poor survey conditions (Boulanger et al. 2019, Campbell et al. 2019, Boulanger et al. 2022) which created potential challenges with comparison of surveys such as the 2023 calving ground surveys that had ideal survey conditions. In this paper we assess potential negative bias in double observer methods and propose a reduce distance sampling / double observer methodology that should improve the robustness of calving ground survey abundance estimates.

The current calving ground survey methodology uses a 400 meter strip transect combined with a double observer mark-recapture modelling effort method to estimate sightability and abundance. The two observers in this case represent an initial detection of caribou (the mark) and a potential re-detection of the caribou group (the recapture) hence double observer methods are analyzed using a mark-recapture analysis approach. This approach was adopted in 2011 to accommodate higher densities of caribou observed on calving grounds which prevented the use of full distance sampling methods, which require greater observer training and the ability to observe caribou across a relatively wide survey strip (1500m) with classification of groups into 5 distance bins (Campbell et al. 2012).

Distance sampling methods have been applied in Nunavut for caribou surveys (Campbell et al. 2015, Campbell et al. 2019, Campbell. et al. 2021) that occurred in spring or fall periods and therefore had lower densities of caribou making it possible to apply the distance sampling method. Results from all of these surveys estimate that detection probability declines with distance within the 400m survey strip. Comparison of distance sampling estimates with double observer 400m strip transect estimates suggest that that the double observer approach does not adequately estimate sightability with estimates being 5-11% lower than double observer distance sampling estimates (Campbell et al. 2015, Campbell et al. 2019). The likely reason for this is that heterogeneity (unequal detection probabilities) of caribou groups within the survey strip creates a positive bias in estimated detection probabilities from double observer surveys leading to a negative bias in estimates. This issue is especially relevant for the dependent double observer method used in many surveys that allows observers to communicate finding but also places restrictive assumptions on detection probability variation such as each observer in a pair having equal detection probability (Buckland et al. 2004). The general mechanism for bias in double observer detection probability with distance is that as distance from the plane increases and detection probability decreases observers become more likely to just observe the more obvious caribou groups and both miss the less obvious groups leading to non-independence of observer detection probabilities (Buckland et al. 2010). This non-independence basically creates heterogeneity variation leading to a positive bias in detection probability (only the easy to observe groups are detected usually by both observers) and a resulting negative bias in abundance. The amount of bias likely varies with overall sightability. If survey conditions are good (low snow years such as 2023) the bias is likely minimal, however, in years of poor sightability such as in some previous calving ground surveys then higher levels of bias are possible.

One unique attribute of distance sampling is that it is relatively robust to heterogeneity variation as long as sightability on the survey line is 1 or is

estimated using double observer methods due to pooling robustness of the distance sampling approach (Buckland et al. 1993, Rexstad et al. 2023, Laake and Collier 2024). For this reason, the general approach used in distance sampling surveys is to focus mark-recapture/double observer methods on estimating detection on the survey line with distance sampling used to model and estimate the decline in detection probability with distance (Laake et al. 2008a, Laake et al. 2008b). We suggest that it is advantageous to collect and include distance sampling data during aerial surveys as it provides a way to confront likely heterogeneity bias that challenges double observer methods.

Methods

The general approach we propose is the use of distance sampling with a reduced number of bins and overall strip distance surveyed. This approach should be less overwhelming than a full distance sampling approach but still may allow a better estimate of sightability and more robust estimates. To assess this approach, we used empirical comparisons and a simulation study.

For the empirical comparisons, we subset distance sampling data sets from the Baffin 2024 (Campbell et al in prep) and South Hampton Island surveys (Campbell et al. 2019) to create reduced data sets and reran the analyses with the reduced data sets to produce estimates. These estimates were then compared with full data set estimates as well as double observer estimates that do not include distance sampling using the *mrds* R (Laake et al. 2012) package. Detailed analysis results are given in survey reports from each project and are not detailed in this report.

We also conducted a brief simulation study to assess whether unbiased estimates could be derived using reduced bins. For these analyses, we simulated the detection function from the Baffin 2024 and then analyzed the simulated data using the 2 and 3 bin designs. Simulations were focused on the distance sampling component of the analysis and assumed detection on the

survey line was 1 with no variation in detection functions. We considered hazard rate and half normal models for estimation and assessed whether there was adequate power to select the appropriate detection function model with each design. Simulations were evaluated in terms of overall estimator bias, confidence interval coverage (proportion of simulations where confidence limits from the estimate overlapped the true value), and precision. Sample sizes of observations were generated to roughly approximate sample sizes from the Baffin Island survey. Five hundred simulations were conducted for each design. Simulations were conducted using the *dsims* R package (Marshall 2022).

Results

Southampton Island

The 2017 Southampton Island caribou survey employed a double observer distance sampling approach with 5 survey bins as shown in Figure 1. The data set consisted of 962 observations used to fit the detection function. A hazard rate detection function was selected which allows a “shoulder” of constant detection which in this case extended for the 0-200m survey bin before declining in subsequent bins. Detection also varied by observer order, snow cover and the log of group size for the distance detection function. For the double observer model detection varied by observer order and snow cover. Detection on the survey line was estimated to be 0.94 (CV=0.013).

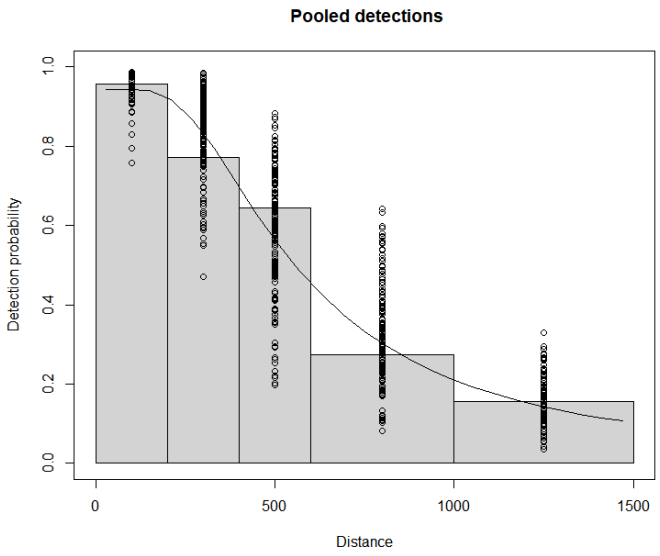


Figure 1. Distance sampling detection function for the full MRDS analysis of Southampton Island.

For the mark-recapture portion of the analysis it is possible to plot double observer sightability estimates for each distance bin to assess if double observer detection probabilities were reduced with distance (Figure 2). It can be seen that there is a slight decline up to the 600m bin but minimal change overall. This plot illustrates the effect of heterogeneity variation. With no heterogeneity variation this plot would be similar to Figure 1 with a decline in detection with distance. Instead, observers both only observe the more sightable groups with increasing distance leading to an overestimate of detection probabilities and a negative bias in double observer-only estimates.

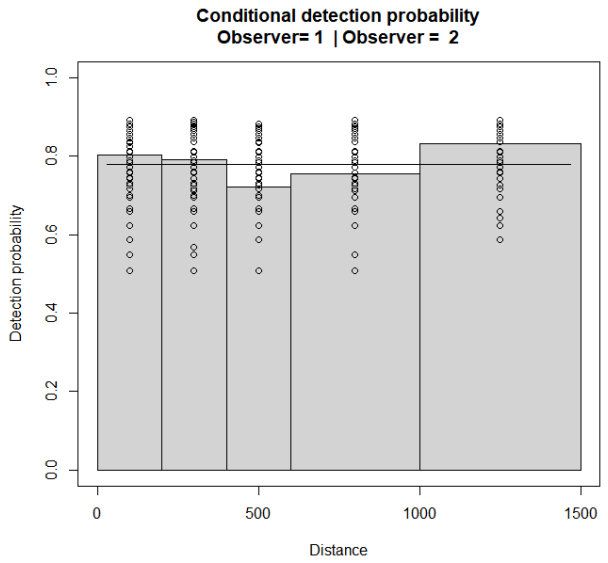


Figure 2. Distance sampling detection function for the full MRDS analysis of Southampton Island. Each bar is the conditional double observer detection probability based on the ratio of caribou groups observed by the secondary observer to overall groups detected.

Figure 3 shows estimated based on subsetting the full data set. The first bar represents abundance from the full data set for comparison with the 1, 2, and 3 bin estimates. Hazard rate (the detection function that allows a larger shoulder area) and half normal (a simpler detection function that restricts the shoulder distance) were considered along with mark-recapture/double observer models with 1 bin (as used in calving ground surveys) and 2 bins (with distance as a covariate to estimate decline in detection across the 2 bins). In general, reasonably similar estimates occur when 2 and 3 bins were used with the best agreement for the hazard rate (HR) model with 3 bins. The hazard rate model did not converge when only 2 bins were used, however, in this case the half-normal still provided an estimate of reasonable agreement. The double observer/mark-recapture only estimates were 5% lower than the full MRDS estimates for this survey.

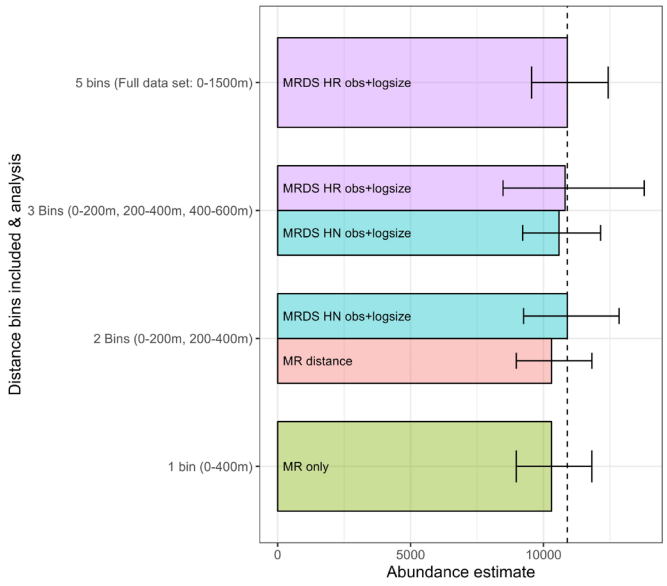


Figure 3. Southampton Island 2017 analysis. The same double observer/mark-recapture model (obs+snow) was used for all analyses shown. The dashed line indicates the estimate using the full 5 bin (0-1500m) data set.

Baffin 2024 fixed wing survey

The fixed wing only data set (a portion of the survey was conducted via helicopter which was less relevant) which consisted of 827 observations was used for this analysis. The detection function for the South Baffin Island 2024 survey was relatively similar to the Southampton Island survey with a hazard rate detection function estimating a shoulder area of approximately 200m before declining at successive distances (Figure 4). Sightability on the survey line was estimated as 0.96 (CV=0.029). Plotting double observer estimates of sightability with distance revealed a decline in estimated detection probabilities, however, detection probability estimates at further survey bins were still much higher than distance sampling estimates.

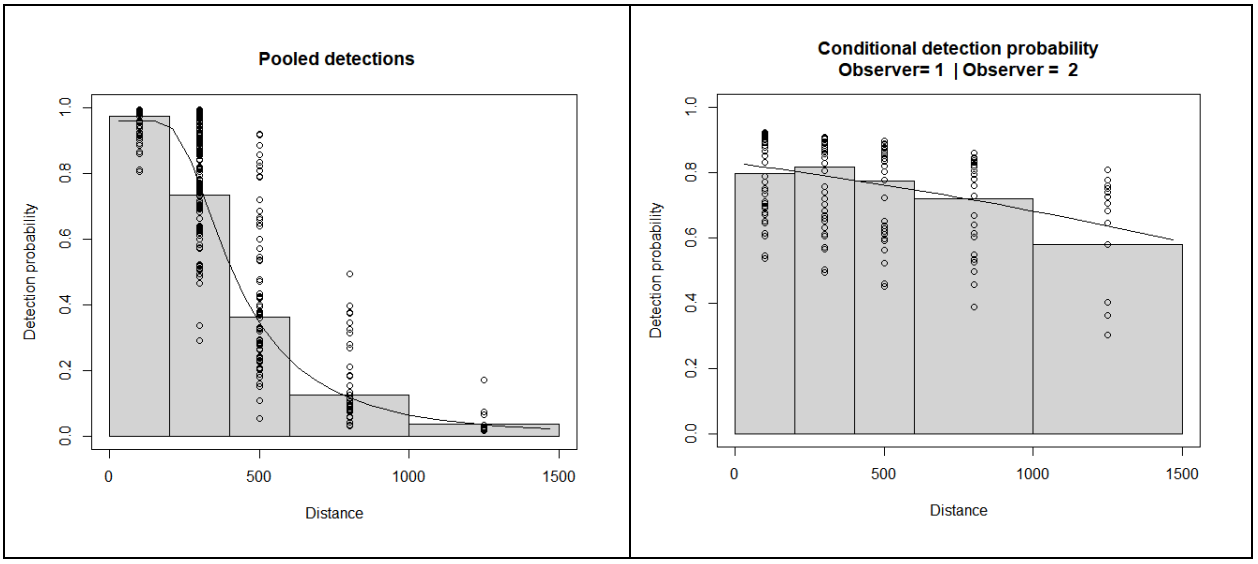


Figure 4. Distance sampling detection function for the full MRDS analysis of Baffin Island (left) and double observer estimates of detection with distance (right).

At the time of this report, estimates for the Baffin Island project had not been released so we use relative abundance, which is the estimate for each model divided by the estimate of the full MRDS model (Figure 5) for comparisons. Estimates when the data was subsetted to reduced numbers of bins were different than South Hampton Island with 3 bins required to obtain an estimate using a hazard rate model that was similar to the full estimate. At 2 survey bins, only the half-normal model converged leading to higher (+10%) estimates presumably due to the half-normal model underestimating the area of constant sightability in the vicinity of the plane. If only double observer model methods were used, estimates were 10% lower than the full MRDS model estimates.

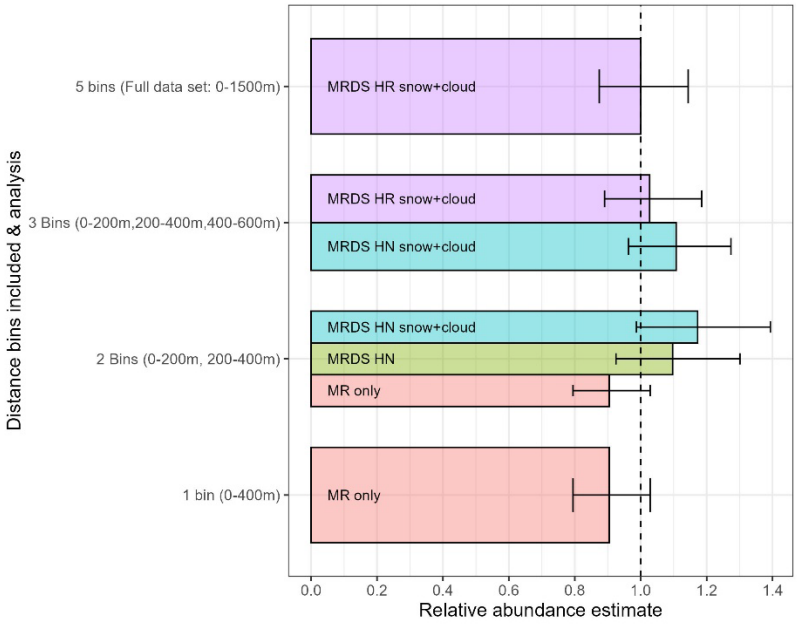


Figure 5. Baffin Island 2024 analysis. Estimates are given as relative abundance, which is the estimate for each model divided by the estimate of the full 5 bin MRDS model. The same double observer/mark-recapture model (ob6+ob11+distance+logsize) was used for all analyses shown. The dashed line indicates the estimates using the full 5 bin (0-1500m) data set.

Simulation study

One of the main questions addressed by the simulation study was whether the reduced bin designs could produce unbiased estimates of abundance. The main source of bias would be the inability of distance models to adequately estimate the shoulder area of constant sightability near the plane and/or the reduction in detection across the range of distances used in the survey. An additional source of bias could be lack of resolution in the data to allow selection of the hazard rate model compared to the simpler half-normal model. For simulations a hazard rate generating model used was based on the Baffin Island detection function with a shoulder of approximately 200m of constant sightability in the vicinity of the plane (Figure 4). A hypothetical study area based on the High East Beverly strata was simulated with a population of 3000

caribou. Figure 6 shows an example of one run of the simulations. For estimation, the data was binned in 5 distance bins (0-200, 200-400, 400-600, 600-1000, and 1000-1500m). Single caribou were simulated with no group size variation in detection probabilities.

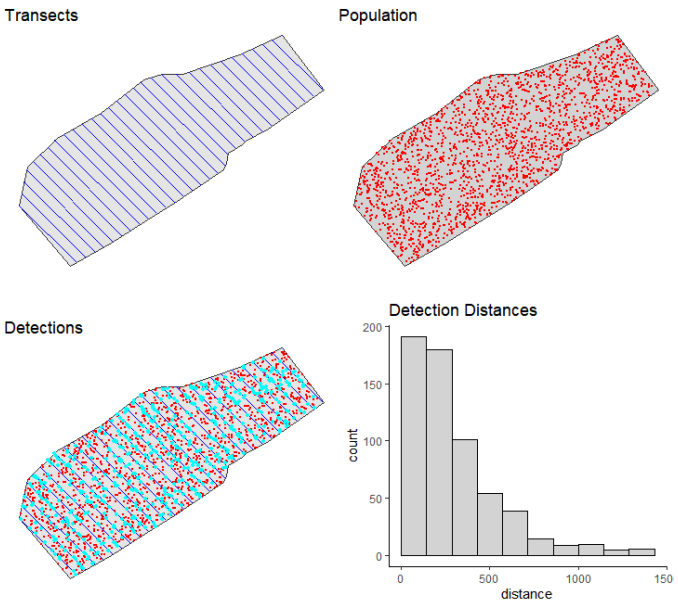


Figure 6 Example of one simulation run with transects, simulated caribou (red dots), detections (blue dots) and simulated distance data. The actual bins used for analyses were (0-200,200-400,400-600,600-1000, and 1000-1500m).

Simulations generated 686 to 919 detections of caribou depending on the number of bins included in the simulation (Table 1). These sample sizes were approximately similar to the South Hampton and Baffin Island data sets. Simulation study results suggested that the hazard rate model could provide unbiased estimates with 97% confidence interval coverage when 3 bins of sampling were used for the analysis (Figure 7 and Table 1). When 2 bins were used, estimates were slightly positively biased with occasional large estimates with non-model convergence in 3% of simulations, however confidence interval coverage was still adequate (94%). In contrast, estimates from the half-normal detection function were positively biased in most simulations with 2 or 3 bins of sampling with lower confidence interval coverage. When AIC model selection

was used to choose between the hazard rate and simpler half normal model the hazard rate model was chosen in 61% of simulations with 2 or 3 bins sampled. As a result, estimates from simulations using AIC model selection resulted in an overall positive bias due to the half normal model being selected in simulations. If 5 bins were simulated, the hazard rate model was selected in 100% of simulations and estimates were unbiased.

Table 1. Simulation results. The mean number of caribou detected, method used to obtain estimates (AIC selection of HR or HN, or HN and HR models), % mean bias, confidence interval coverage (% of simulations where the confidence interval included the true value), mean CV of estimates, proportion of simulations using AIC where the hazard rate (HR) model was selected, and proportion of simulations where models did not converge.

Distance Bins used	Sampling width (m)	Mean caribou detected	Model method	Bias (%)	Conf. Interval Coverage	Mean estimate CV	Proportion AIC HR selected	Proportion failures
5	1500	919.3	AIC	0.2	0.96	0.046	1.00	0.00
3	600	803.2	AIC	4.4	0.73	0.079	0.61	0.00
2	400	684.9	AIC	3.5	0.90	0.055	0.61	0.04
3	600	805.5	HN	9.0	0.57	0.094		0.00
2	400	686.0	HN	5.0	0.87	0.074		0.00
3	600	805.0	HR	0.3	0.97	0.049		0.00
2	400	689.2	HR	3.4	0.93	0.063		0.03

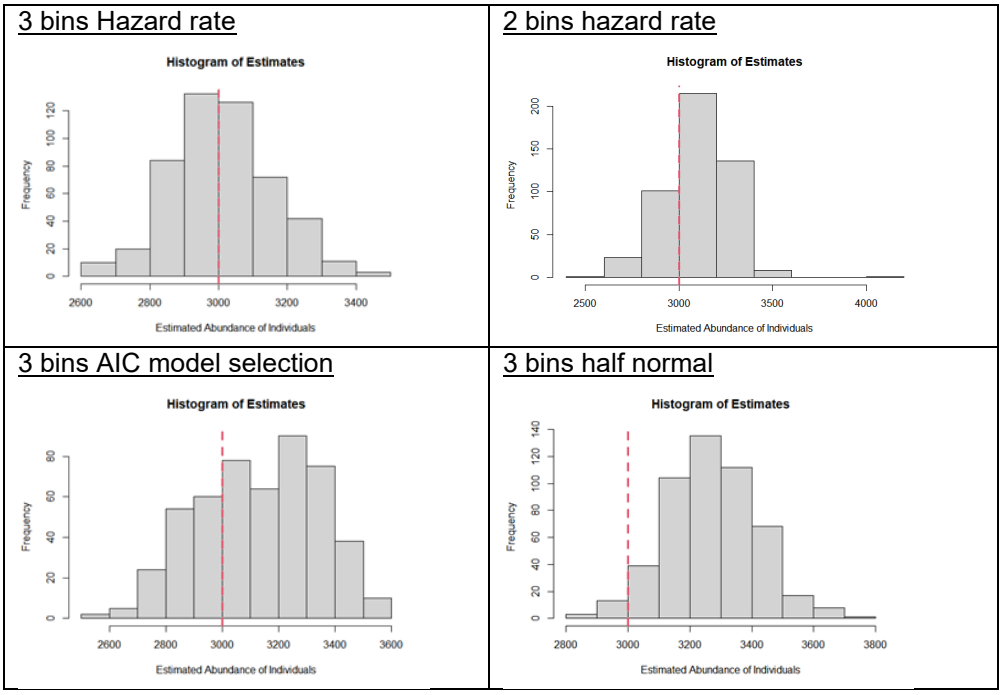


Figure 7. Examples of the distributions of simulated estimates compared to true value (red line) for each simulation run.

Discussion

Results from this analysis suggest that it is possible to get robust estimates using a reduced distance sampling and double observer approach. Of the data formulations considered the 3-bin (0-200m, 200-400m, 400-600m) provided the most reliable estimates based on both empirical comparisons and the simulation study. One reason for this is that it was possible to fit a hazard rate model that assumes an area of constant sightability near the plane (which is likely) with 3 bins of data. If 2 bins are used, then the half-normal model had to be used which is more restrictive in shoulder width leading to a reduced shoulder area and an overestimate of abundance.

Empirical comparisons provided a useful first step in assessing this methodology, however they were limited in that only one outcome of the survey is considered, and the true abundance was not known. The simulation study allowed an estimate of actual bias which further supported empirical comparisons. The simulation study was a simplification of field efforts given that detectability on the

line was assumed to be 1 and group size variation was not simulated. Therefore, the levels of precision from the simulation study will be optimistic. However, simulation results provide the same general conclusion as empirical comparisons which is reassuring.

Empirical comparison of double observer and distance sampling estimates from past surveys suggest that double observer approach may underestimate abundance by 5-10% which is a reasonable difference but not enough to change general survey conclusions. Other comparisons also suggest that double observer estimates may be biased low. For example, IPM analysis of the Beverly 2023 survey also suggests that adult female estimates were approximately 5% lower in 2018 that IPM predictions. A comparison of paired photo and double observer survey lines during the Bluenose-East surveys in 2013 and 2023 surveys suggested that photo estimates were 7 to 14% higher, however, further analysis suggested that the main bias occurred on lines with higher densities of caribou where photo counts were higher with similar estimates at moderate densities (Boulanger et al. 2024). Regardless, negative biases in strip transect/double observer estimates cannot explain the increase documented in the 2023 survey or the increase documented in between 2018 and 2023 for the Bluenose-East herd. However, it is possible that double observer estimates showed a greater degree of negative bias in years such as 2021 Bluenose-East survey when survey conditions were poor. In this case, the proposed approach may provide a better estimate of abundance when survey conditions are challenging.

One additional consideration with visual survey estimates is potential counting bias when densities of caribou are higher. In the Beverly 2023 survey it was found that visual estimates may be biased high when densities are very high due to challenges of determining whether caribou were within the survey strip. In this case, the video-assisted approach provides a secondary way to count caribou

and can complement the use of double observer and distance methods. In general, it is assumed that visual surveys will be applied at moderate densities of caribou (<10 caribou/km^2) with video or photo plane methods used when densities are higher.

Simulation results suggest that unbiased estimates can be obtained with 2 or 3 bin sampling, however the 3-bin approach (0-200,200-400, 400-600m) is preferable to reduce the risk of not having enough data to adequately determine the shoulder area of constant sightability. For example, the 2-bin design only selected the hazard rate model in 61% of simulations leading to a positive bias if the half-normal model was used for estimates. One strategy to confront this issue would be to only consider a hazard rate model for the analysis. The hazard rate model is considered to be "model robust" in that it can still produce unbiased estimates when the width of the shoulder of constant sightability near the survey line is not large (Buckland et al. 1993). Further simulations could be conducted to explore a greater range of survey conditions.

The main difference between the South Hampton and Baffin Island surveys in this analysis and calving ground surveys is the reduced densities of caribou which made it easier to apply distance sampling methods. For example, on the Baffin Island survey densities were less than 1 caribou per km^2 and therefore would be considered low density in a calving ground survey. During calving ground surveys observers and data recorders are more preoccupied with counting caribou groups and therefore successful application of this approach requires minimal additional complexity for observers and data recorders. However, we suggest the only additional piece of information that would be required is what survey bin the majority of any group occurs in. Also, observers would need to be reminded to pay the most attention to the closest survey bins (if the 3 bin approach is used). We propose that this additional complexity is worth considering given the likely robustness of this approach compared to just using the double observer method for calving ground survey estimates.

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PIJUTAOYOK: Kugluktuk Angoniatit Katimiyiit (HTO-kut) titigakimayot kiotjutaitnik Nunvavut Kavamatkut kiotjutigihimayaitnun tatvunga 2023-mi Ahiakmiut tuktuit amigaitilangitnik kititjutaitnik tatvalu tuktuit kanogininganik.

HIVUNIGHITJUTAIT:

- Kugluktumi anguniaktit anguniakpaliktait tahapkua Ahiakmiut tuktuit tatvanganin 2019-min. Kihiani kaoyimayaoyok tahapkua Ahiakmiut tuktuit anguniaktaovakhimayot kangagalok.
- Immakjuap Kivatani tahamani kaningani tuktuit Kugluktumiut anguniaktit tuktuhiokluakpaktait. Ukkua tuktuit ikiklihimalgakhutik amigaikjumikhimayok ukkiuni kangituni.
- Kilohiktomitaok Tuktuit ikiklihimalikmiyot.
- Kugluktumiut anguniaktit ukakatakpaktut tahapkua Kilohiktokmiot Tuktuit illaliotikatalikmiyot tahapkunani Akhiakmiut Tuktuitni.

UBLUMIOYOK KANOGININGIT:

- Ukkua Nunavut Kavamatkut naunaiyaihimayot amigaitilangitnik tuktuit kanogininingitniklo tatvani 2023-mi. tahapkua tuktuit amigaitilangit takkunakhiyot amigaikpalianyot tatvanganin kingolikmi naunaiyakataohimangmata.

Uktutaotkuyaoyot:

- Ukkua Kugluktumi HTO-kut pikataoyomainaktot katimatjutaoyuni pitjutigivlugit tahapkua Ahiakmiut Tuktuit.
- Ukkua Kugluktuk-mi HTO-kut pikkuyagaluangit tamakmik tahapkua Nunavut Kavamatkut tatvalu Nunatiami Kavamatkut ihuakhaitkovlugit ahiakmi tuktuit naunaiyaitkovlugit. Ataotikut naunaiyaktaoyagiakaktot naunaitiangiangita tamaita kaffiotilangitnik nahautait takkuktaohimayot.

Titigakhimaya uma
Amanda Dumond, Ataniat

ᐅᐻᓄᖅ ᑐᓂᔨᐅᔪᖅ

ᓄᓇᕗᑦ ᐅᒪᔪᓕᕆᔨᖅᑎᑦᑕᐃᓐᓇᑦ ᑲᑎᒪᔨᖅᑭᑐᓂᓪᓗᑦ

ᐅᔨᒪᖅ

ᑐᓴᓴᖏᑦ: X **ᐃᓱᒪᓕᐅᕈᑎᑦ:**

ᐱᓵᖅᑕ: ᓄᓇᕗᑦ ᓴᖅᒪᖅ, ᐊᖅᑎᓕᕆᔨᐃᓐᑦ–ᐊᐳᔨᑭᖅᑭᑦᑐᑦ ᐱᓕᕆᐊᖓᔪᓕᑦ ᓄᑕᖅᐳᕆᐊᖅᓭᒪᔪᓕᑦ

ᐊᐳᖏᑕᐅᔪᓕᑦ ᖃᐅᔨᓴᓴᖏᓴᐃᓪᓗ

- ᓄᓇᕗᑦ ᑭᓴᖅᑐᐃᕐᕕᖅ ᐅᐊᖓᓇᖅᐸᓂᐊᓂᐊᓂ ᐊᐳᖏᓪᓗᑦ ᐅᓇᑎᐅᓂᐊᓂ (ᐊᐳᓪᐃᐧᐣ) ᓇᓂᑐᐃᓴᓇᖅᑐᑦ ᑲᓇᑕᒥ.
- ᐊᐳᖏᐃᑦ ᑎᑎᕋᖅᑕᐅᓂᒪᔪᓕᑦ ᐃᓂᒪᓗᑕᐅᔪᓪᓗᑎᑦ ᑲᓇᑕᒥ *ᐅᒪᓂᔪᐃᑦ ᐊᑦᑕᖏᓇᖅᑐᑦᑐᑦ ᐱᖏᔪᖅᓴᑎᑎᔪᑦ* (SARA) ᓇᓄᐃᔪᓪᓗ ᓇᓵᓄᐳᑕᐅᓴᐅᓴᓐᓂᑦ ᐊᐳᓇᐅᒪᓴᐅᔪᑕᐅᔪᓪᓗᑎᑦ, ᐃᓄᓂᕐᒥᑦ ᐊᒻᒪ ᐱᖏᔪᓂᓄᑎᑦ ᐊᒻᒪ ᑭᓇᐅᔪᓂᓴᐅᓴᑦ ᐊᖏᓇᓂᐅᓴᖅᑎᑎᓂᑎᓂᓴᐊᑎᑦ ᐅᓄᖓᑦᓂᖅᓴᖅᓂ. ᐅᓄᖓᑦᑐᑦ ᓴᖅᑭᑎᑦᑎᓇ

ᑐᑭᓯᐊᑭᑎᕆᔨᐊᓪᓚ ᐊᑐᖅᑕᐅᓗᑎᑦ ᐃᑲᓴᑎᐅᓂᐊᖅᑐᒥᑦ ᖃᐅᔨᓴᖅᑐᑕᒪᓕᕆᓂ ᑎᑎᕋᖅᓯᒪᔭᕐᓂᓂ ᑐᑦᐅᒪᑎᑦᑎᓗᑎᑦ ᐊᐅᓚᑦᑎᑦᓴᑦᑕᐅᔭᓂ.

- ᖃᐅᔨᓴᖅᑐᑦ ᐊᑐᓚᐅᖅᑐᑦ ᐊᖏᖅᑭᑦᓯᒪᔭᓂᒃ ᐊᐱᖅᑯᑎᖅᓯᒪᔭᓂᒃ ᐊᒻᒪᓗ ᐱᓕᕆᖃᑎᒌᓚᑎᑦ ᓄᓇᖓᔪᐊᒥᑦ ᐊᑐᖅᓴᑎᑦ ᐅᓂᒃᑲᐅᔨᖅᓴᖅᑐᑎᑦ, ᑎᑭᓚᑦ ᑎᑎᕋᖅᑕᐅᓂᖅᑐᑦ ᐃᓕᓴᕐᓂᑦ ᐃᓕᓴᕐᓂᑦ ᐊᐅᓚᑦᑎᑦᓴᑦᑕᐅᔭᓂᑦ ᐅᓄᖅᓯᓕᕆᓂᖅᑐᑦ, ᐊᓕᐅᑦᑕᖅᑕᐅᔭᓂᑦ ᓄᓇᑐᐃᓐᓇᕆᔨᐅᑦᑕᐅᔭᓂᑦ ᐅᓄᖅᓯᓕᕆᓂᖅᑐᑦ, ᐊᒻᒪᓗ ᖃᐅᔨᓴᓴᐅᑦᑕᐅᔭᓂᑦ ᐊᒻᒍᐃᑦ ᓄᓇᐃᑦ ᐊᖏᔪᐊᓕᕆᓂᖅᑐᑦ ᐃᓐᓇᖅᓯᓚ.
- ᐃᓕᐅᔭᑦ ᓄᓚᓇᐃᖅᓯᓕᐅᖅᑐᑦ ᐅᓄᖅᓯᓂᒋᓯᓂᖅᑐᑦᓯᒪᓂᖅᑐᑦ ᐊᒻᒍᐃᑦ ᑭᓴᓕᕐᑎᐅᓂ ᑕᐃᒪᓐᓕᒫᓂᑦ 2000 ᓯᓚᐅᔪᓂ, ᑕᓯᐅᐅᓚᔨᐅᐅᓐᓯᓪᓚᑎᑦ 1960 ᑐᓴᓂᓂ ᑕᓯᐅᐅᓚᔨᓕᕆᔨᓪᓚᑎᓪᓚᓗ 1990–ᒥᑦ ᐅᓴᒻᒋᔪᓕᓪᓚᓗ.
- ᐅᑎᒪᖅᑕᐅᔭᑦ ᐅᖅᑕᓯᐅᖅᑐᑦ ᐊᒻᒍᐃᑦ ᑕᐃᔨᓯᒪᓂ ᐅᓄᓚᐊᓯᐅᐅᓐᓂᑭᓂᓂᑐᒃ ᑭᓴᓕᕐᒥᑦ, ᓄᓇᑐᐃᓐᓇᕆᔨᓂᕐᖅᑐᑎᑦᑕᐅᓕᕐᓂᑎᑦ ᐊᒻᒪᓗ ᐅᑎᖅᑑᓂᑎᑦ.
- ᐃᓕᐅᔭᑦ ᓄᓚᓇᐃᖅᓯᓕᐅᖅᑐᑦ ᐅᓄᖅᓯᓴᓕᓪᓕᖅᓯᓂᖅᑐᑦ ᐊᒻᒍᐃᑦ ᑭᓴᓕᕐᒥᑦ ᑕᐃᒪᓐᓕᒫᓂᑦ 2000 ᓯᓚᐅᔪᓂ, ᑕᓯᐅᐅᓚᔨᐅᐅᓐᓯᓪᓚᑎᑦ 1960 ᑐᓴᓂᓂ ᑕᓯᐅᐅᓚᔨᓕᕆᔨᓪᓚᑎᓪᓚᓗ 1990–ᒥᑦ ᐅᓴᒻᒋᔪᓕᓪᓚᓗ.
- ᑲᑎᒪᖅᑕᐅᔭᑦ ᐅᖅᑕᓯᐅᖅᑐᑦ ᐊᒻᒍᐃᑦ ᑕᐃᔨᓯᒪᓂ ᐅᓄᓚᐊᓯᐅᐅᓐᓂᑭᓂᓂᑐᒃ ᑭᓴᓕᕐᒥᑦ, ᐊᓂᐅᓚᓴᖅᑐᑎᑦᑕᐅᓕᕐᓂᑎᑦ ᐊᒻᒪᓗ ᐅᓄᖅᓯᓂᒋᓯᓂᖅᑐᑎᑦᑕᐅᓕᕐᓂᑎᑦ ᐊᒻᒪᓗ ᐅᓄᖅᓯᓂᒋᓯᓂᖅᑐᑎᑦᓯᓪᓚᓗ ᐅᓄᖅᓯᓂᒋᓯᓂᖅᑐᑎᑦ ᐅᑎᖅᑑᓂᑎᑦ ᑲᑎᒪᖅᑕᐅᔭᑦ ᐊᒻᒍᐃᑦ ᐅᓄᖅᓯᓂᒋᓯᓂᖅᑐᑎᑦ (Awan et al. 2019) ᐊᑐᖅᓯᓗᑎᑦ ᓇᓂᕐ ᐅᖅᑯᒪᐃᓕᓴᐅᓚᓂᒃ ᐊᒻᒪᓗ Empirical Bayesian Kriging (2.88-4.07 ᓇᓂᐃᑦ/1000km^2) ᑲᑎᒪᔨᑦ ᐅᓄᖅᓯᓂᖅᑐᑦ 1081-1515 ᓇᓂᐃᑦ ᖃᐅᔨᓴᖅᑎᑦᓚᐅᔪᓂᑦ.
- ᑕᒪᖅᑎᒃ ᓇᓕᐅᖅᑕᐅᔭᑦ 95–ᓴᓵ CI ᐊᐳᔨᖏᓪᓂᔨᓚᓐᓂᕐᖅᑐᑦ Awan et al–ᒧᑦ. (2019) 2.1-6.1 ᓇᓂᐃᑦ/1000km^2 ᐊᐳᔪᓕᓂᕐᓗᓂ ᐊᓯᓂᓪᓂᑦ ᖃᐅᔨᓴᖅᑕᐅᖃᖅᑕᐅᔭᑦ ᐊᐳᔪᓕᓂᕐᓂᓪᓂᒃ ᓇᔨᖃᓂᓪᓂᒃ.
- ᐃᓇᖃᑎᑎᓂᒃᑲᐅᑉᓯᒧᑦ ᐃᓕᓴᕐᓂᖓ ᑭᓴᓕᕐᒥᑦ ᐅᖅᑕᐅᔭᐅᓪᓚᕐᓂᑐᖅ ᐊᓯᓂᓪᓂᑦ ᐊᐃᔅᑐᖅᓯᒪᔭᓂᑦ ᐊᐃᔅᑐᖅᓯᓚᐅᔭᓂᑦ ᐊᐃᔅᑐᖅᓯᓚᐅᔭᓕᓪᓚᓗ

- 2019-20–Γ, 658–σᵇ ᐊᒪᖅძσᵇ ᐊᖏᔪᓕᐸᵇᑐᑦ ᐅᖅᐅᓂᐅᓕᐅᖅᐳᑦ ᓄᓇᕗᓕᒻΓ, 64–ᐳᓴᖅᑎᑦ ᐊᖏᔪᐅᓂᐊᕐᓂ ᐊᕐᕕᐊσ, ᖃᒪσ'ᑐᐊᖅΓ ᐊᒪᒪ ᖁძᕐᓗᖅᑐᖅΓ ᐊᖏᔪᓕᐸᵇᑎσᵇ.
- 2020-21–Γ, 699–σᵇ ᐊᒪᑭᖅᑕᐅᔪσᑦ ᐅᖅᐅᓂᐅᓕᐅᖅᐳᑦ, 68–ᐳᓴᖅᑎᑦ ᐊᖏᔪᐅᓂᐊᕐᓂ ᐊᕐᕕᐊσ, ᖃᒪσ'ᑐᐊᖅΓ ᐊᒪᒪ ᖁძᕐᓗᖅᑐᖅΓ. ᑕᒪᐃᓐσ ᐊᕐᕆᔨσ, ᖃᓴᓂᐅσᓂᑦ ᐱᑭᓱᖅᓂᒪᓂᑐᑦ ᐊᖏᔪᓕᐸᵇᑕᐅᕙᒃᑐᑦ ᐅᓄᑭᓐᔪᖅᐸᓕᓕᐅᖅᐳᑦ ᖃᐅᔨᖏᓐᓂᑐᒍ ᐊᒪᒪ ᐅᓄᖅᑦᓗᑎᑐᑦ ᐊᖏᔪᓕᐸᵇᑎᑦ ᐃᑲᔪᕐᔪᐊᓕᐅᖅᐳᑦ ᐱᐳᐅᔪσᑦ.
- ᐊᑦᔨᑎᖅᑎᑲᔪᐊᖅᑎᓕᑐᑦ ᓄᓇᕗᒻσ, ᐊᒪᖅძᑦ ᓄᓇᕗᒻᓄᑦ ᖃσᓄσᖅᓴᐅᓕᑐᑎᵇ ᓇᓄᵇᑕᐅᖅᑎᑦᑕᓕᐅᖅᐳᑦ ᖁძᙵᐊᓕᓴᐃᕐᐸ ᑐᓴᓂᒥ (75 ᑭᓕᒋᑕσᵇ ᐅᖅᓱᓕᵇᑎᑕᓕᑐᑎᵇ) ᐊᒪᒪ ᓄᓇᕗᒻσ ᐅᖅᓱᓕᓄᑎᑕᓕᑐᑎᵇ ᐅᑭᐅᵇძᑦ, ᐅᖅᓱᓕᓭᒋᓕᑐᑎᵇ ᐊᐃᑦᕆᓕΓ ᐊᒪᒪ ᒪᐃΓ (500 ᑭᓕᒋᑕᓄᑦ ᑎᑭᓕᑐᑎJ).
- ᐊᐅᖃᑎᓕᓚᵇძᑦ ᖃᐅᔨᓴᖅᐸᓕᓕᐊᑭᓚᓇᖅᐳᑦ ᖃᐅᔨᓴᓱᑲᵇᓴσᵇ ᑐᓇσᖅძᑕᐅᔪσᵇ ᑐᓴᖅᑎᑦᑎᖅᵇσᓭᓇᖅᓗᑎᑲᓗ ᐊᐅᓭᑦᑎᔨᐅᖅᖃᑎᑲᓂᑦ ᑭᒋᓕᓕᐅᖅᐸᓕᒻᒥᑦ ᐊᐃᑦᕆᒪᓕΓ ᒪᐃΓ ᒪᐃΓ (500 ᑭᓕᒋᑕᓄᑦ ᑎᑭᓕᑐᑎJ).

ᐊᒪᑭᖅᑕᐅᔪᑦ ᖃᐅᔨᓴᓱᑲᵇᓴᓕᑐᑎ

- ᓄᓇᕗᓴΓ ᖃᐟᖃᐃᓐᓴᔪᐃᑦ (ᖃᐟᖃᐢ) ᐃσᓂᖃᑦᓗᑎᵇ ᑕᒪᐃᓐσᓂᑲᓴᵇ ᐊᐸᖅᑎᑦᑐσ ᓄᓇᕗᓴΓ ᓇᓗᓇᐃᖅᑕᐅᔪᓕᓕᐅᓗᑎᑐᓗ ᑕᒪᐃᓐᓄᑦ ᐊΓᑦᓄᑦ ᐊᒪᒪ ᐊᖏᑎᔪᓄᑦ σᑦᔪᑎᓄᑦ *ᓄᓇᕗᓴᒥ ᐊᖏᑎᔭᑎᖏᓐᓄσ*. ᑑᵇძᐊ ᐱᓪᒪᓇᐅᓕᓗᑎᵇ Γᖅძᔪᓕᑦ ᓄᓇᕗᓴΓ ᐃᓕᖅძᔪᖏᓐᓄσ ᑭᓇᐅᔭᓕᐅᑎᑎᵇᓴᖏᓐᓄσᓕᓗ.
- ᖃᐟᖃᐃᓐᓴᔪᐃᑦ ᑎᑎᕋᖅᑕᐅᔪᒪᓐᔪᔪᑦ ᐃᔪᒧᓗᑕᐅᔨσᖏᓐσᵇ ᑲᓇᑕΓ *ᐅᒪᔪᑦ ᐊᑦᑕᑦᓇᖅᑐᐅᑦᑐᑦ ᐱᖃᔪᖏᓐᓄσ* (SARA).
- ᑭᖓσᓕᑎᓐσ ᖃᐅᔨᓴᖅᑕᐅᓕᐅᖅᑐσᓕ ᓇᓕᐅᑦᑑᖅᑕᐅᔪᑦ ᐅσᓄᑦσᖏᓕ 1.6–σᑦ 4.4–σᓄᑦ ᖃᐟᖃᐃᓐᓴᔪᐃᑦ/1,000 km² ᑭᓭᓕᔭᑎᐃᑦ 4.14 ᖃᐟᖃᐃᓐᓴᔪᐃᑦ/1,000 km² ᓇᐸᑦᑐᓕᑲ ᑕᓗᖓσ ᖁᑎᑦᑎᐅσ.
- ᓄᓇᕗᑦ ᒐᓭᒪᵇძᑦ ᑲᑎᖅᓗᐃᓗᖓᕗᑦ ᖃᐟᖃᐃᓐᓴᔪᐃᑦ σᐊᑦძᖏᓕᵇσᵇ ᐊᒪᒪ ᑐᑦძᑕᐅᔪᐸᑦ Γᵇᓵᓄᑦ ᑎᑎᕋᖅᓗᒪᔪᑕᑦ ᐊᖏᔪᐅᓂᐊᕐᓂ ᓇᓕᐅᑦᑕᖅᑕᐅᔪᓕᑐᑎᵇ ᐊᒪᒪ ᖃᐅᔨᓴᑦᓗᑕᐅᓕᑐᑎᵇ ᐊᒪᒪ ᖃᐅᔨᓴᖅᑕᐅᔪᓕᑐᑎᵇ ᐊᒪᒪ ᑭᓗᓂᖏᓐᓄσ ᐊᖏᔪᐅᓂᐊᕐᓂ.
- ᐊᖏᔪᐅᓂᐊᕐᓂ ᑐᓂᓗᓂᒪᔪᑦ ᓂᐊᐅᑦᑐᖅᑕᐅᔪᑦ ᐊᖏᔪᐅᓂᐊᕐᓂ 691–σᵇ ᖃᐟᖃᐸᑕᐅᔪᑦ 2018-2019 ᐊᒪᒪ 2023-2024 ᐊძᵒσᓪᓚσ.
- ᐅᓄᑦσᖅᓴᐃᑦ ᖃᐟᖃᐸᑕᐅᔪᑦ ᖁᑎᑦᑎᐅσᵇ ᐊᖏᔪᐅᓂᐅᓕᐅᖅᓭᒪᔪᑦ, ᑐᑦᔪᓂᖅᑐᑦ ᖃσᓂᑲσᐳᓂᵇ, ᐅᓄᖅᑭᵒσᖅᓴᐃᑦ ᐊᖏᔪᐅᓂᐅძᑦᓂᓂᑐᑎᵇ ᑲᓇᐟᖅᓴᕐᔪᓕᑦ.
- ᐊᑦᖃᐊᑦ ᐊᒪᒪ ᖃᒪσ'ᑐᐊᖅ ᐊᖏᔪᖏᓂᑦ ᐅᓄᓭᓕᔪᖅᑕᐅᓕᐅᖅᑐᑦ ᓇᐃᔨᖅᑕᐅᔪᑦ.
- ᐊᖏᔪᐅᔪᑦ ᑎᑎᕋᖅᓭᒪᔪᑦ ᑕძᑎᑦᑎᐱᑦ 84–ᐳᓴᵒ ᖃᐟᖃᐸᑕᐅᓕᐅᖅᑐᑦ ᖁძᑭᐅᑎᒧᑦ, 16–ᐳᓴᖅᑎᑦᓕᓕ Γᑭᑭᐊᑦᓄᑦ/ᑭᔪᖃᑦᑕᓄᑦ ᐱᔪᐅᑦᓂᑎᵇ.
- ᖃᐟᖃᐃᓐᓴᔪᐃᑦ ᐊᖏᔪᑦᓂᑦᓂᑎᵇ ᒪᑭᑐᐃᖅᓭᓂᐊᕐᓇᓂ ᐊᖏᔪᑦᓂᐃᑦ ᐊᖏᔪᑭᐅᔪᑦ ᐅᓄᑦσᖅᓴᐅᑦᓂᑎᵇ ᐊᑦᓇᑦᓂᓐσᑦ ᐱᐸᖅᓴᓭᓭᓕᔭᑎᐃᑦ ᐊᑎᑎᖅᑕᓄᓂᐃᑦ ᐊᖏᔪᐅᑦᑑᖅᑕᐅᔪᑦ (63 ᐳᓴᵒ ᐊᖏᔪᐅᔪᑦᓂᒪσᑦ.
- ᐊᖏᔪᐅᔪᑦ ᐃᵒᓇᑭᓂᑦσᖅᓴᑦ ᐱᔪᐅᖃᑎᑦᑕᓕᐅᖅᓇᓂᵇ ᐅᑭᐅᖅᖏᔪᓂᑦᓴᖅᑎᑦᑎᑦ , ᐃᵒᓇᑭᖅᓭᒪᔪᑦσᑦᓂ ᐊᖏᔪᐅᔪᑦ ᐅᓄᖅᓭᓭᐊᖅᓇᓂᵇ ᐅᑭᐅᖅᖏᔪᑎᑦᓂᑐJ.
- ᓇσᑐᐃᵒᓇᓐᓂσᑦ, ᐅᑭᐅᖅᑦᑦ, ᐊᖏᓇᖅ/ᐊᖏᔪᓂᑐᖅ ᐱᔪᐅᔪᑦ ᖃᓄᐃᖅᑲᓇᕗᐅᖅᑐᑦ.
- ᑭᓭᐊσ, ᐅᓄᖅᓭᑐᐊᖅᓇᓂᵇ ᐊᖏᓇᑦᓂᐃᑦ ᐊᖏᔪᐅᔪᑦσᑦ ᐅᵒᔨᓚᑦᐅᔪᑦᓚᐊᕐᑲᵇ ᖃᐅᔨᓴᖅᑕᐅᑦᑭᓭᓇᖅᑐᐅᑦᓂᑎᵇ ᐊᖏᔪᐅᑦᑑᖅᑕᐅᔪᑦ.

ᐊᐸᑎᓕᕐᓂᖅᐸᑦ
Department of Environment
Avatiliqiyikkut
Ministère de l'Environnement

GRIZZLY BEAR (*Ursus arctos*) HARVEST MONITORING IN NUNAVUT

Summary report

August 2025

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The opinions in this report reflect those of the author and not necessarily those of the Government of Nunavut, Department of Environment. Reference to any specific type of gear and/or equipment does not mean that it is endorsed by the authors or the institutions they represent.

Summary

The purpose is to monitor the geographic distribution, sex, and age composition of grizzly bear harvest in Nunavut. Hunters were asked to return the lower jaw or whole skull, a piece of muscle, a small piece of the skin with hair (2x5cm), and to provide kill information (date, location, etc.) to their local Conservation Officer on a voluntary basis. Tooth samples were processed to determine age of the individuals. A total of 200 grizzly bears were reported harvested from 2020 to 2024, 109 from the Kitikmeot and 91 from the Kivalliq region. Grizzly bear harvest is tightly linked to hunter accessibility; in late winter/early spring hunters use snowmachines to access areas further from communities, and in summer ATVs are used to access areas around communities as well as boats to extend hunting off water channels. Grizzly bear harvest was concentrated in the western Kitikmeot, in the vicinity of Kugluktuk, Cambridge Bay and along the traditional travel route from the Cambridge Bay to the Bathurst Inlet area. Arviat and Baker Lake were the next highest contributors to the total harvest in the Kivalliq region.

Between 2019 and 2024, the harvest was dominated by a younger age class (<5 years) in the Kivalliq region and the proportion of adults in the harvest was 38%, while proportion of adults in Kitikmeot harvest was 56%. Long-term reported harvest trend (2010-2024) shows that in the Kitikmeot region, the number of bears harvested averaged 15 bears annually (SD = 7.3, range 4–32), and was significantly higher in 2021 (32 bears). Grizzly bear harvests in the Kivalliq have increased substantially since 2008. From 2010 to 2024, the harvest averaged 20 bears annually (SD = 6.9, range 9–34). The number of grizzly bears currently being harvested within the Kivalliq region may not be sustainable over the long term and may cause a population decline, highlighting the need to better understand the status of the population and determine viable harvest levels.

ᐊᐅᓪᓗᐃᑦ (ᑲᔪᖅ ᓇᓂᖅ) ᐱᓕᐅᓴᑦᑐᓂᑦ ᖃᐅᔨᓴᕐᓂᖅ ᓂᓇᓄᓕᒥᒋᑦ

ᓇᐃᓱᖅᓯᒪᔪᖅ

ᐱᔭᕐᑕᐅᔪᖅ ᓇᐅᑎᖅᓯᐃᓂᕐᒧᑦ ᓂᓇᒋᑦ ᓇᑭᖏᕐᒪᖏᑕ, ᐊᕐᓇᓪᓚᒪᖏᑕ ᐊᖏᓴᓪᓚᒪᖏᑕ ᖃᓄᐃᓂ ᐅᑭᐅᖃᓕᒪᖏᑕ ᐊᖏᓴᐅᑎᓪᓗᑎᑦ ᓂᓇᓄᓕᒥᒋᑦ. ᐊᖏᓴᓇᓯᑎᑦ ᑐᓂᓯᖃᑦᑕᕐᔾᔨᐅᔪᑦ ᐋᓪᓕᓪᓗᓂᑦ, ᓂᑭᒋᑦ, ᐊᒥᐊᕐᔨᔾᔨᒃ ᒥᖅᔾᖃᖅᕐᓗᓂ (2x5cm), ᑐᓴᖅᑎᑎᑐᓗᑎᒃᓗ ᖃᖅᖏᓪ ᓇᓪᓂᓇᐅᕐᒪᖏᓪᓕᐊᑉᐳᓪᓂᑦ (ᐅᓪᓗᖅᓯᓂ, ᓇᒥᑦ, ᑕᒪᒃᔾᐊ) ᐅᒪᔭᕐᓂᐊᖅᑎᑦᔾᔨᓪᓂᑦ ᐃᔾᓯᖅᓯᕐᓗᔾ. ᑭᔪᑎᑦ ᖃᐅᔨᓴᕐᑕᐅᓕᐅᖅᑐᑦ ᖃᐅᔨᓴᐅᐅᓇᓯᓗᑎᒃ ᖃᓂᓂᒋ ᐅᑭᐅᖃᓕᒫᖏᓐᓂᑦ. ᑲᑎᓪᓗᒋᑦ 200 ᐊᒃᓴᒃᑕᐅᔾᔪᒪᓂᕆᖅᑕᐅᔪᑦ 2020–ᒥ 2024–ᒪᓂ, 109–ᓂ ᖃᑎᒌᒥᐅᓂᑦ 91–ᓗ ᑭᓪᓕᓪᓚᒥᑦ. ᐊᒃᓴᒃᑕᐅᔪᓱᓇᖅᑐᑦ ᐊᖏᓴᓇᓯᑎᒪᑦ ᐱᓕᐅᔪᓱᓇᓕᔾᔪᑎᖅᓴᑦ; ᐅᑭᐅᑐᖃᐅᓪᓕᖅᑎᓪᓗᒍ/ᐅᐱᕐᖏᓕᕐᓴᖅᑎᓪᓗᒍ ᐊᖏᓴᓇᓯᑲᑦ ᖃᒍᑕᐅᑦᔨᓂᒃ ᐊᑐᖃᒃᑕᑕᖅᓴᑦ ᓂᓇᓕᓱᓂ ᐅᖏᓪᓗᓯᓂᖅᓴᒍᑦ, ᐅᐱᕐᖏᓐᓄᑦᓯᓪᓗ ᓯᑕᒪᓕᓪᓕ (ᑎᓴᒪᓪᓕ) ᐊᑐᖃᒃᑕᐅᓕᐅᖅᓯᓄᑦ ᓂᓇᓕᓱᓂ ᖃᓄᐅᑦᔨᖅᑭᑦᑐᓪᓂᑕᓂ ᐊᒪᒪ ᐅᒥᐊᓂᑦ ᐊᖏᓴᓇᓯᐊᕐᕕᐅᓪᓚᑎᒃ ᐃᒪᓄᑦ. ᐊᒃᓴᒃᑕᐅᔪᑦ ᖃᑎᖃᐅᑦ ᐱᖏᓐᓇᖏᓪᓕᐊᓪᓚᑕᐅᒪᐅᖅᑐᑦ, ᖁᓯᓪᓚᖅᑐᑦᑐ ᐊᕐᓂᑦ, ᐃᖅᓚᑐᑎᐊᒥ ᐊᖅᓯᒋᑎᑦᔨᐅᓂᖅᑐᑦᑕᓪᓚᑦ ᐃᖅᓚᓴᑐᑎᐊᒥᑦ ᖃᑎᓕᑦᑐᐅᑦ. ᐊᕐᓇᓕᑦ ᐊᒪᒪ ᖃᒪᓂ'ᑐᐊᖅ ᐊᖏᓴᖅᑎᑦ ᐅᓄᓭᓯᖃᑕᐅᓕᐅᖅᑐᑦ ᓇᐃᔨᖅᑐᑐᑦᑎᓪᓗᑎᑦ.

ᐊᓯᓂᓕᓴᓂ 2019 ᐊᒪᒪᓗ 2024, ᐊᖏᓴᓇᓯᑕᐅᓪᓚᐊᑕᓕᐅᖅᑐᑦ ᒪᑲᓯᓂᖅᓴᐃᑦ (ᑕᓪᓕᒪᒪᐃᑦ ᐊᕐᕋᔪ) ᑭᓪᓕᓪᓚᒥᑦ ᐃᓂᓇᐅᓂᖅᓴᐃᓪᓚᓗ ᐱᔭᑦ 38 ᐳᓴᓂᑎᐅᓪᓕᓪᓚᖅᓯᑎᑲ, ᐃᓂᓇᐅᓂᓕᕆᓪᓕ ᖃᑎᖅᑎᐅᓂᑦ 56 ᐳᓴᓂᑎᐅᓕᐅᖅᓯᑎᑲ. ᐊᓯᓂᐅᔨᓯᑦ ᐅᖃᐅᓯᐅᓯᒪᔨᑦ ᓇᓂᓄᑕᐅᓪᓓᑐᑦ ᖃᓄᐃᓕᓪᓓᓪᓕᐊᓂᕆᑦ (2010-2024) ᑕᓯᐊᓴᐅᑎᑦᓯᐅᓚᖅ ᖃᑎᖅᑎᑎᐅᓂ, ᖃᑦᓯᐅᐅᓂᕆᑦ ᓇᓂᓄᑕᐅᔨᑦ 15-ᖓᔪᓴᓪᓕᐅᖅᑐᑦ ᓇᓄᐃᑦ ᐊᖏᖅᑭᒧᒪᒃ (SD = 7.3, ᐊᓯᓂᓂᕆᑦ 4-32), ᐊᒪᒪᓗ ᐅᓄᖅᓂᖅᓴᐅᑐᓗᑎᒃ 2021-ᒥ (32 ᓇᓄᐃᑦ). ᐊᒃᓴᒃᑕᐅᓪᓓᑦᑐᑦ ᑭᓪᓕᓪᓚᖅᒥ ᐅᓄᖅᓯᓪᓓᓪᓚᓪᓚᖅᓴᑦ ᑕᐃᒪᓐᓇᑦ 2008–ᖓᔪᓕᐅᖅᑎᑦᓗᒍ. 2010–ᒥ 2024–ᒧᓄ, ᓇᓂᓄᑕᐅᓪᓓᑦᑐᑦ 20–ᓂ ᓇᓄᓂᓂᑦ ᐊᖏᖅᑭᒧᒪᒃ (SD = 6.9, ᐊᓯᑦᑐᓂᓐᓄ 9-34). ᐊᒥᓱᓂᓕᑦ ᐊᒃᓴᒃᑕᐅᓪᓓᑦᑐᑦ ᑭᓪᓕᓪᓚᒥᑦ ᐱᐅᖏᓪᓚᕿᓴᐅᔨᓯᓂᐃᓂᓇᑕᓚᐅᓕᓚᓂᒃ ᐊᓯᓂᐅᔨᐊᒥ ᐅᓄᑭᓯᖅᓴᓪᓚᑦᑎᓪᓚᑕᐅᓇᖏᓱᓂᓪ, ᑐᑭᓱᓪᓓᓪᓚᓇᐊᖅᓴᒪᓇᓚᓪᓗᑐ ᖃᓄᐃᑕᓐᓄᓪᓗᓂᕆᓂᓂᑦ ᐊᒥᓱᓂᓕᑦ ᖃᖅᖏᓐᓇᐅᓪᓚᐊᑕᐅᓕᐅᖅᑐᑦ ᐊᖏᓴᓇᓯᑐᓗᑎᒃᓴᓂᑦ.

Akhait (*Ursus arctos*) an'ngujauhimajunik munariniq Nunavunmi

Naittumik Titiraqhimajuq

Pidjutikhaq munagiyaangat humiliqaak nunami nayugangit, qanuritmangaat, ukiungitlu akhat anguyauvakhimayut Nunavunmi. An'nguniaqtut apigijaujut utiquplugit agliguit niaquaplu hauninganik, ilagani nukiip, mikkamik amirmit nujaqaqtumik (2x5cm) aktilanga, tunihijuqhatlu an'ngujarnik naunaitkutinik (ublua, humi, qanuqlu) An'nguhiqiijimun Havaktianut piumaluni piluni ilikkut. Kigutingnik uuktuutikharnik hanaqiyauvaktun naunaiyaiyaangat ukiungit akhat. Atauttimun 200 ngujut aqhait unniudjutaujut an'nguniaqtaujut talvanga 2020 min 2024 mun, 109 Kitikmeonit 92 lu Kivallirmin. Aqhanik an'nguniaqtaujut pidjutijut an'nguniaqtip pijaangnikkut; nunggunianut ukiumi/atulihaaliqtumi upin'ngaqhami an'nguniaqtit atuqhutik sikiituunik un'ngahiktuliariami nunamingnit, aujamilu haantaqhutik (ATVs) atuqtauvaktut pijaaginnik avataani nunallaanggit imaalu qajaqtuqhutiklu an'ngunahuariamingnik imakkut. Aqhanik an'nguniarnik ihumagigilluaqtaujut uataani Kitikmeot, haniani Kugluktup, Iqaluktuuttiaq in'ngilraangnitatlu aullaaqviujait Iqaluktuuttiarmin Kingaukmun nunainni. Arviat hamanilu Qamanittuaq tugliuyut amigainirit ikayuutait atautimun anguniaqtamingnun Kivallirmi.

Qitqani 2019 2024 milu, an'nguniarniq amigaitqijaujut nukaqhiinni (<5 ukiuni) iKivallirmi ilanggattauq inirnirit an'nguniaqtut 38 pusanngujuq, ilanganittauq inirni Kitikmeoni an'nguniarniq 56 pusanngujuq. Hivitunikkut unniudjaujut an'nguniarnirmik auladjutait (2010-2024) takupkaqtittijut Kitikmeoni, nampait aqhanik an'nguniaqtaujut imaa ittut 15 aqhaliqivaktut ukiuq tamaat (SD = 7.3, naunaitkutaa 4–32), imaalu amigaitqijaupluni 2021 mi (32 aqhat). Aqhat an'nguniaqtaujut Kivallirmi amigaiqjuummiktut talvamin 2008. Talvanga 2010 talvunga 2024, tahapkuat aqhaliqivaktut qitqanit 20 nik ukiuq tamaat (SD = 6.9, naunaitkut 9–34). Nampanggit aqhait hadja an'nguniaqtaujut Kivallirmi aulalimaittungnaqhijut hivitunirmi ikiqlijuummingniagungnaqhijutlu, takupkaqtitaqhunilu ihariangnia kangiqhittiaqtuqhatigut qanurinmanggaatigut qaffiutigut ihumaliurnikkutlu ihuaqtuniklu an'nguniaqtaujuqhatigun aadlatkiingni.

Surveillance de la récolte de grizzli (*Ursus arctos*) au Nunavut

Sommaire

Le but de cette initiative est de surveiller la distribution géographique des récoltes de grizzli au Nunavut ainsi que leur répartition en ce qui concerne le sexe et l'âge. Pour y parvenir, nous avons demandé aux adeptes de la chasse de grizzli de remettre à leur responsable de la conservation le crâne entier de leur prise, sinon sa mâchoire inférieure, ainsi qu'un échantillon de muscle et un petit morceau de fourrure (2 cm x 5 cm) prélevés sur celle-ci, de même qu'à fournir, sur une base facultative, des renseignements sur leur récolte (date, emplacement, etc.). L'âge des sujets a été déterminé grâce à l'analyse des échantillons dentaires. Pour la période de 2020 à 2024, la récolte déclarée s'élève à 200 grizzlis, dont 109 proviennent de la région du Kitikmeot et 91 de la région du Kivalliq. La récolte de grizzlis dépend étroitement de la facilité d'accéder à leurs populations. Les adeptes de la chasse utilisent la motoneige pour accéder aux populations éloignées de leur localité à la fin de l'hiver et au début du printemps; en été, on recourt au véhicule tout-terrain (VTT) pour la récolte à proximité, et à l'embarcation pour étendre les zones de chasse aux abords des cours d'eau. Les prises ont majoritairement été récoltées dans l'ouest du Kitikmeot, aux alentours de Kugluktuk et de Cambridge Bay, de même que le long du parcours traditionnellement emprunté pour se déplacer entre la région de Cambridge Bay et celle de Bathurst Inlet. Quant à Arviat et à Baker Lake, dans la région du Kivalliq, elles figuraient au deuxième rang parmi les grands contributeurs à la récolte totale.

Entre 2019 et 2024, la majeure partie de la récolte de grizzli dans le Kivalliq représentait la catégorie des jeunes (<5 ans) et les adultes comptaient pour 38 % des prises régionales, alors que dans la région du Kitikmeot, les sujets adultes représentaient 56 % de la récolte. Les données recueillies sur une longue période (2010-2024) concernant les récoltes déclarées démontrent que dans la région du Kitikmeot, une moyenne de 15 grizzlis était récoltée par année (écart-type : 7,3; intervalle de variation : 4 à 32), ce qui était considérablement inférieur aux données pour 2021 (32 grizzlis). La récolte de grizzlis dans le Kivalliq a considérablement augmenté depuis 2008. Entre 2010 et 2024, sa moyenne était de 20 grizzlis par année (écart-type : 6,9; intervalle de variation : 9 à 34). Les actuelles habitudes de récolte du grizzli dans la région du Kivalliq pourraient ne pas être durables à long terme et entrainer un déclin de la population, d'où la nécessité de bien en comprendre la situation et de déterminer des quotas de récolte viables.

Table of Contents

Summary ii
ᓄᐃᓕᖅᑲᓇᕐᓂᖅ iii
Naittumik Titiraqhimayuq iv
Sommaire v
1.0 Introduction 1
2.0 Methods 3
3.0 Results and Discussion 4
4.0 Acknowledgments 15
5.0 Literature cited 16

List of Figures

Figure 1. Proportion of males and females in reported harvest in Kitikmeot (a) and Kivalliq (b) from 2020 to 2024. 5

Figure 2. Distribution of reported grizzly bear harvest in Nunavut, from 2020 to 2024. 7

Figure 3. Reported grizzly bear harvest in Kivalliq and Kitikmeot regions between 2010 and 2024... 8

Figure 4. Proportion of males and females in reported harvest in the Kitikmeot region, from 2010 to 2024.. 8

Figure 5. Proportion of males and females in reported harvest in the Kivalliq region, from 2010 to 2024. 9

Figure 6. Age and sex structure of the reported Kitikmeot grizzly bear harvest, from 2019 to 2024. 10

Figure 7. Age and sex structure of the reported Kivalliq grizzly bear harvest, from 2019 to 2024.. 11

Figure 8. Proportion of age classes in the Kitikmeot and Kivalliq reported harvest, from 2019 to 2024.. 12

List of Tables

Table 1. Reported grizzly bear harvest in Kitikmeot and Kivalliq regions between 2020 and 2024. 6

1.0 Introduction

Nunavut represents the northeastern edge of grizzly bear (*Ursus arctos*) distribution in Canada. Inuit observations, harvest records and research indicate an increase in numbers and range expansion eastward and northward (Clark 2007, Nirlungayuk 2011, Dumond et al. 2015, Awan et al. 2019, 2023, Barrueto et al. 2023, Harding et al. 2025, Awan et al. 2025). Grizzly bears are listed as a species of Special Concern under the federal Species at Risk Act (SARA) and bears are regularly harvested for subsistence, and a relatively small amount of revenue through trophy hunting and the sale of hides and parts. There are limited baseline data on grizzly bear distribution and density within Nunavut, in part because of the cost and challenge of surveying bears at low densities in vast and remote areas.

In Nunavut, grizzly bears are classified as both a furbearer and a big game animal under the *Nunavut Agreement*, and are classed as a Presumption as to Needs species where the full level of harvest is dedicated to Inuit (see 5.6.5(a) of the *Nunavut Agreement*). Currently, Inuit are able to harvest grizzly bears for subsistence use with no restrictions. The local Hunters and Trappers Organizations (HTOs) have the power to approve sport hunter harvesting of furbearers under 5.6.13(c) of the *Nunavut Agreement*. Therefore, the responsibility to limit harvesting of grizzly bears, fall strictly under the roles of HTOs and Regional Wildlife Organizations (RWOs). For a sport hunter (NU resident or a non-resident/non-resident foreigner) to hunt a grizzly, they would need HTO approval, along with a harvesting licence and a fur tag issued from the Government of Nunavut. They also need the services of an outfitter, with some specific exceptions under the Act/regulations.

For several years, hunters in the Kitikmeot have reported an increase in bear sightings and occurrences of bears in new areas (e.g. Victoria Island). Some Inuit hunters have suggested predation as a potential contributing factor in current caribou declines. Increasing grizzly bear populations was identified as a threat to caribou recovery. Similarly, hunters from Arviat, Baker Lake and Rankin Inlet have observed increasing grizzly bear numbers and voiced their concerns over problem bears around

communities, at cabins, meat caches and regard grizzly bears as a problem animal (Awan 2021, Harding et al. 2025). However, the number of cabins has increased significantly over the past 30 years, likely increasing conflicts compared to temporary tent camps (Awan et al. 2025).

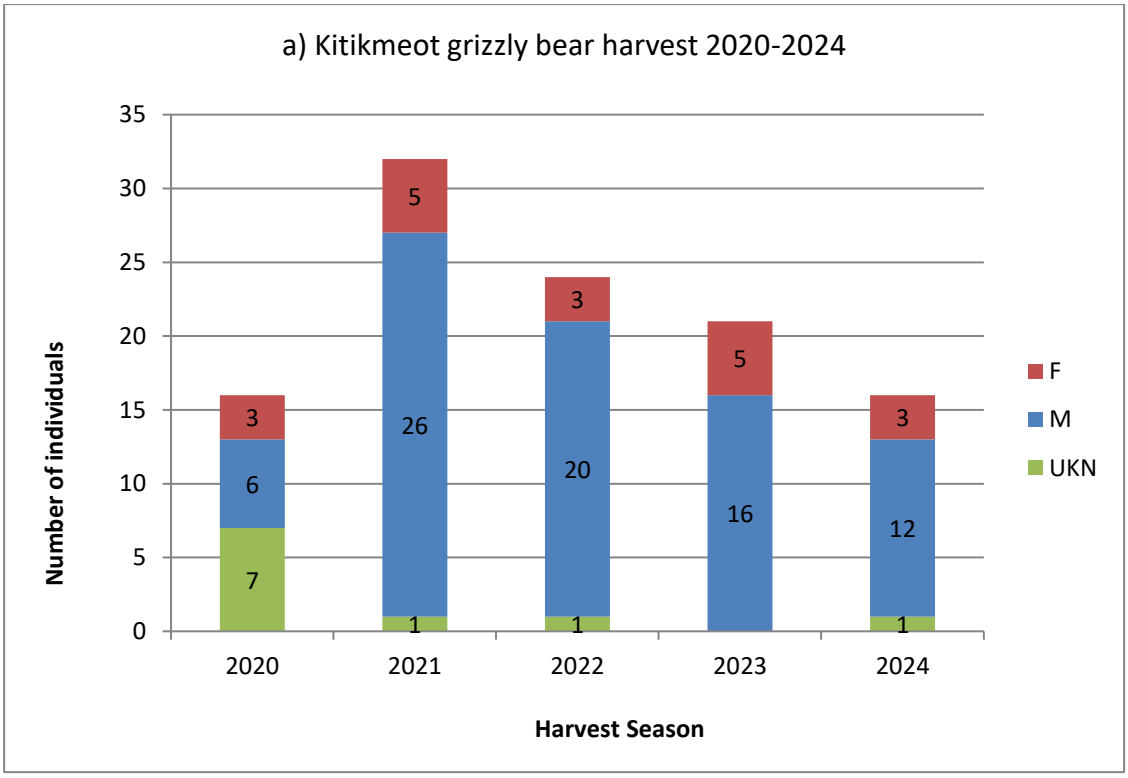
Although territory wide surveys have not been conducted, in 2016 and 2017, Awan et al. (2019) sampled 4 grids (49 km × 49 km) in the Kivalliq Region of Nunavut using DNA-hair tripods and estimated a density of 3.5 bears/1,000 km^2 (95% CI = 2.1–6.1 bears/1,000 km^2). We sampled the western mainland of the Kitikmeot Region using DNA hair snagging methods from 2021 to 2023 and updated the earlier results (2008/09, Dumond et al. 2015) and sampled the areas west of Bathurst Inlet in 2022 and area around and south of Bathurst Inlet in 2023. The overall density estimate from our 2021–2023 study is 5.92 bears per 1,000 km^2 (CI = 5.02–6.83, Awan et al. 2025). Monitoring of grizzly bear harvest numbers, sex and age composition of the harvested bears is fundamental to the conservation of the species and its sustainable use.

2.0 Methods

Subsistence harvesting reports are voluntary and, with few exceptions, hunters have been reporting the harvest with good return rates to-date. Hunters in the Kivalliq and Kitikmeot regions were asked to return the lower jaw or whole skull, a piece of muscle, a small piece of the skin with hair (2x5cm), and to provide kill information (date, location, etc.) to their local Conservation Officer for each harvested grizzly bear. To determine the age of the harvested individuals, we sent the first premolar (lower PM1) to Matson's Laboratory LLC (Milltown, MT, USA) for cementum analysis. This technique is based on the cyclic nature of cementum growth in teeth forming annular patterns of different darkness depending on the season (Matson 1981). Age results from the 2018-2019 season (2019) have been included in all analysis presented in this report which take into consideration the age of the individuals. The harvest year was assigned as the year at the end of the prescribed harvesting season, for example, 1 July 2019—30 June 2020 was the 2020 harvest year/season.

3.0 Results and Discussion

A total of 200 grizzly bears were reported harvested from the 2020 to 2024 harvest seasons, 109 (males = 80, females = 19 and unknown sex = 10) from the Kitikmeot and 91 (males = 57, females = 33 and unknown sex = 1) from the Kivalliq region (Fig. 1).



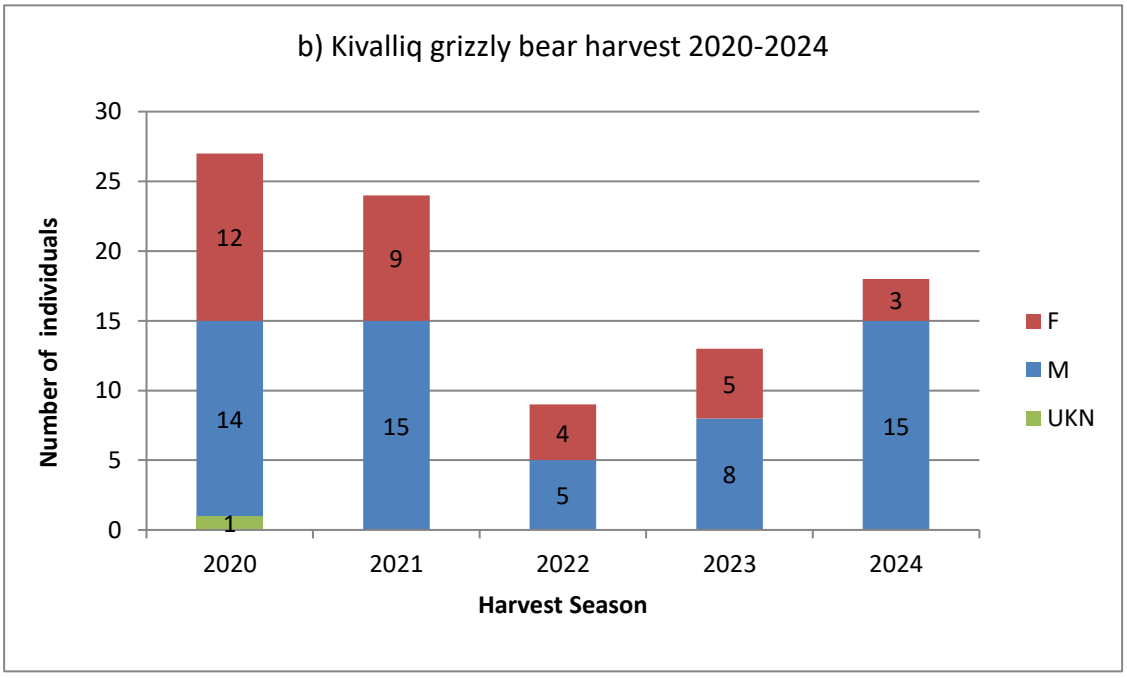


Figure 1: Proportion of males and females in reported harvest in Kitikmeot (a) and Kivalliq (b), from 2020 to 2024.

The known harvest, by region and community is summarized in Table 1. Most grizzly bears were harvested in the western Kitikmeot, in the vicinity of Kugluktuk, Cambridge Bay and along the traditional travel route from the Cambridge Bay to the Bathurst Inlet area, and few in the eastern communities. Arviat and Baker Lake were the highest contributors to the total harvest in the Kivalliq region (Fig. 2, Table 1). Grizzly bear harvest is tightly linked to hunter accessibility; in late winter/early spring hunters use snowmachines to access areas further from communities, and in summer all terrain vehicles (ATVs) are used to access areas around communities as well as boats to extend hunting off water channels.

Table 1: Reported grizzly bear harvest in Kitikmeot and Kivalliq regions between 2020 and 2024. RH = Regular Hunt, SH = Sport Hunt.

Community	2020		2021		2022		2023		2024	
	RH	SH	RH	SH	RH	SH	RH	SH	RH	SH
Cambridge Bay	7		9		9	5	5	4		7
Gjoa Haven					1					
Kugaaruk							1			
Kugluktuk	9		23		5	4	9	2	7	2
TOTAL KITIKMEOT	16		32		15	9	15	6	7	9
Arviat	15		20		3		6		7	
Baker Lake	12		2		4	1	3	1	7	1
Naujaat					1					
Rankin Inlet			1				3		3	
Whale Cove				1						
TOTAL KIVALLIQ	27		23	1	8	1	12	1	17	1

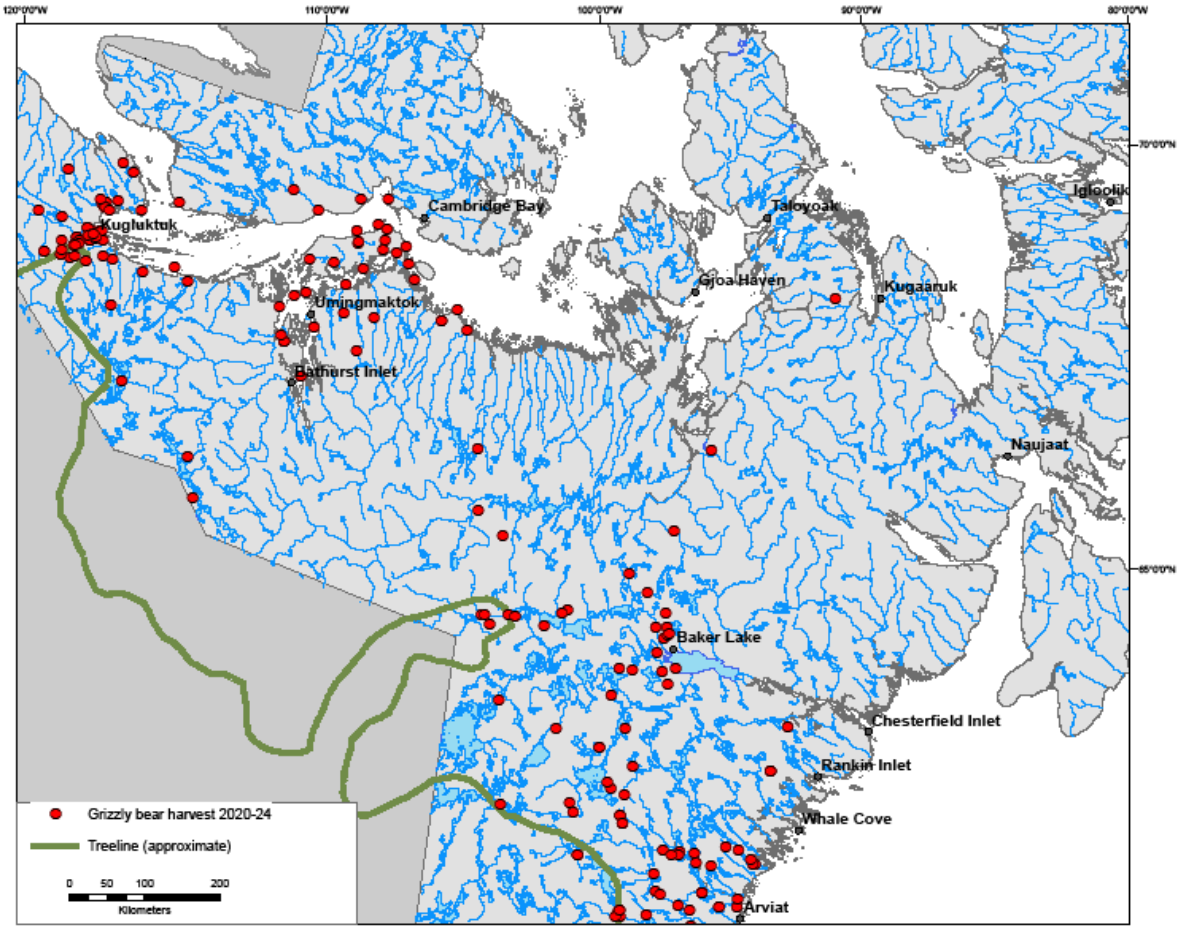


Figure 2: Distribution of reported grizzly bear harvest in Nunavut, from 2020 to 2024.

Long-term reported harvest trend of 15 years (2010-2024) shows that in the Kitikmeot region, the number of bears harvested averaged 15 bears annually (SD = 7.3, range 4–32; Fig. 3), and was significantly higher in 2021 (32 bears). Reported grizzly bear harvests in the Kivalliq have increased substantially since 2008. From 2010 to 2024, the harvest averaged 20 bears annually (SD = 6.9, range 9 –34; Fig. 3).

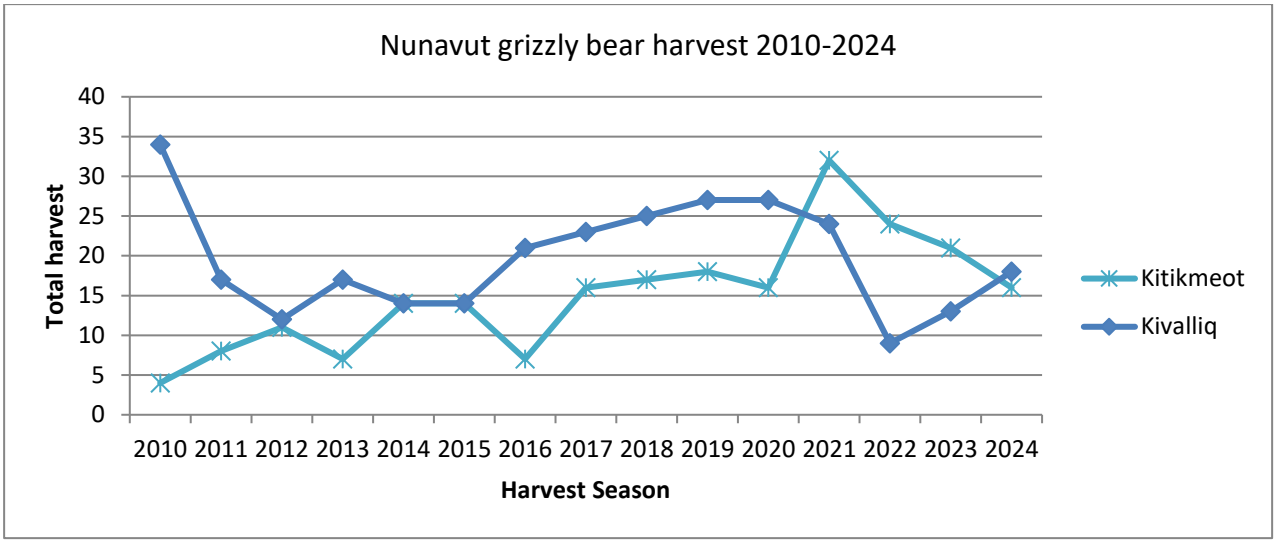


Figure 3: Reported grizzly bear harvest in Kivalliq and Kitikmeot regions between 2010 and 2024.

The male to female ratio in the harvest has been relatively stable for the last 15 years in the Kitikmeot region with males and females representing an average of 85% and 15% of the total harvest respectively (Fig. 4). Males composed >80% of the harvest in most of the last 15 years, indicating a stable and healthy population.

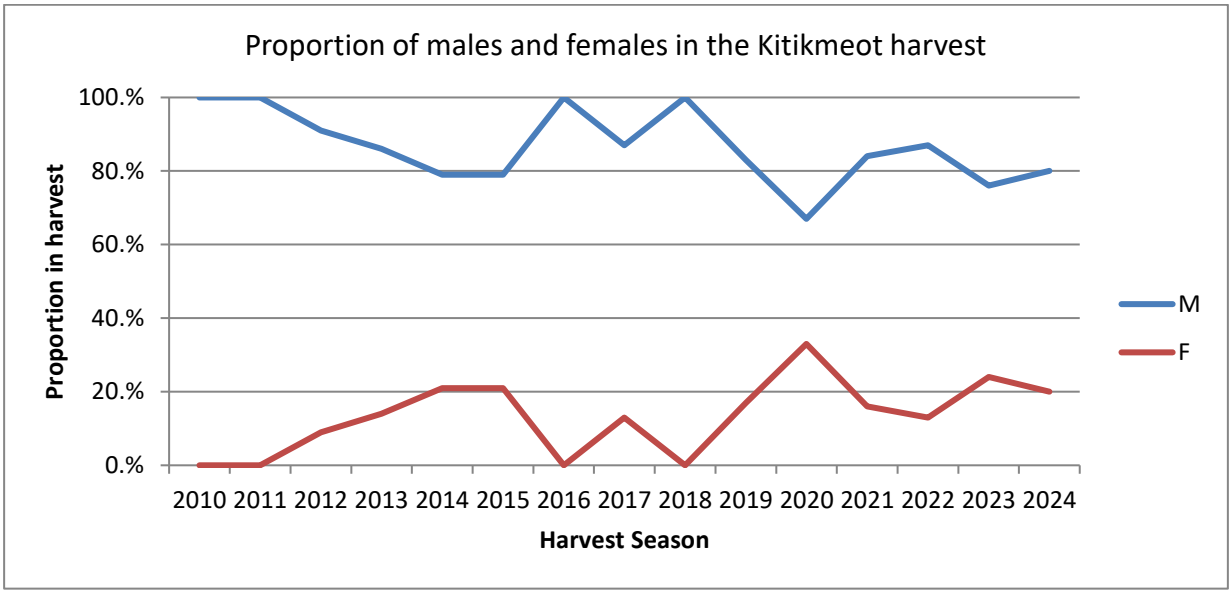


Figure 4: Proportion of males and females in reported harvest in the Kitikmeot region, from 2010 to 2024.

In the Kivalliq, a similar pattern was up to 2009 harvest season, with an average of 82% and 18% of males and females respectively in the harvest (Awan 2021). However, the proportion of females in the Kivalliq harvest increased to 32% and 71%, for the 2010 and 2011 seasons respectively, before dropping to average 29% from 2012 to 2024 season. A decline in the percentage of males taken in the 2011 harvest suggests fewer males are present (Fig. 5).

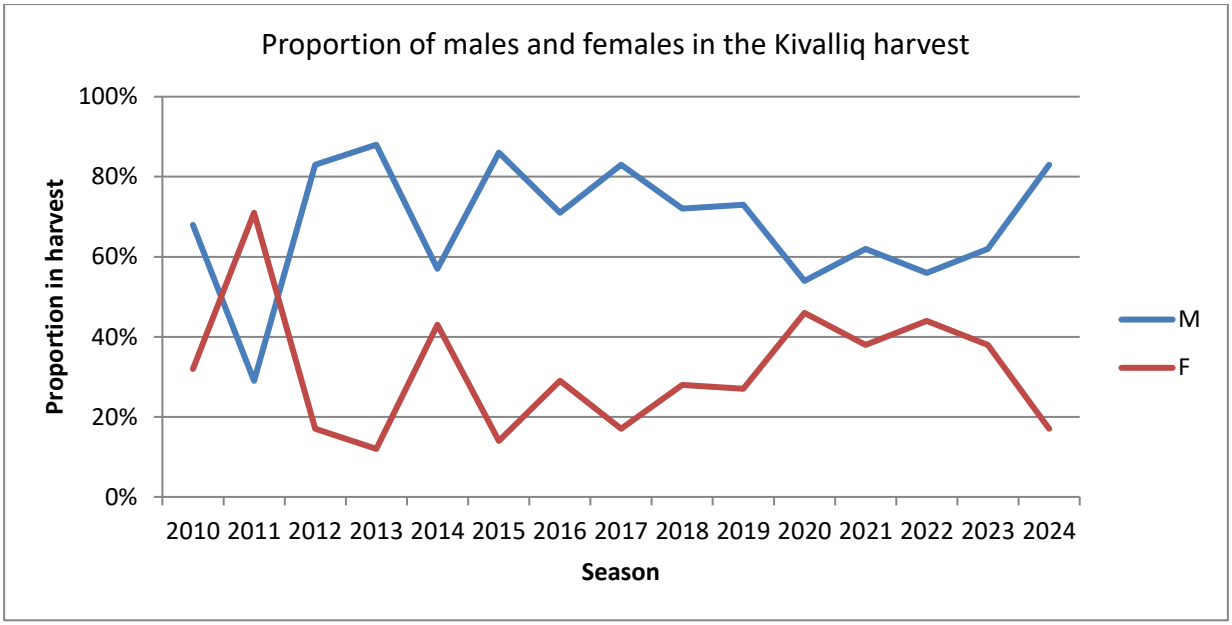


Figure 5: Proportion of males and females in reported harvest in the Kivalliq region, from 2010 to 2024.

The ages of harvested grizzly bears in Kitikmeot region ranged from 1 year to 23 years (Fig. 6). The oldest female (23 years) was killed by a hunter in 2020 in the Kugluktuk area.

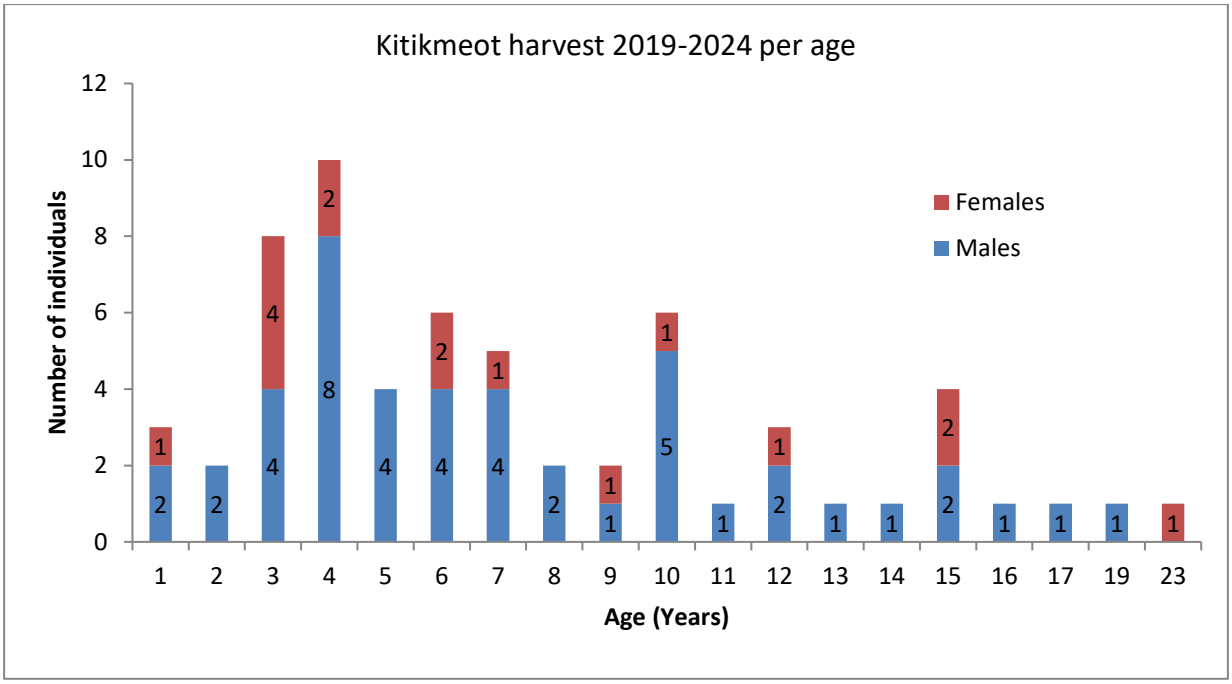


Figure 6: Age and sex structure of the reported Kitikmeot grizzly bear harvest, from 2019 to 2024.

The ages of harvested grizzly bears in Kivalliq region ranged from 1 year to 24 years (Fig. 7). The oldest male (24 years) was harvested in September 2021 about 80 km northeast of Arviat near the coast.

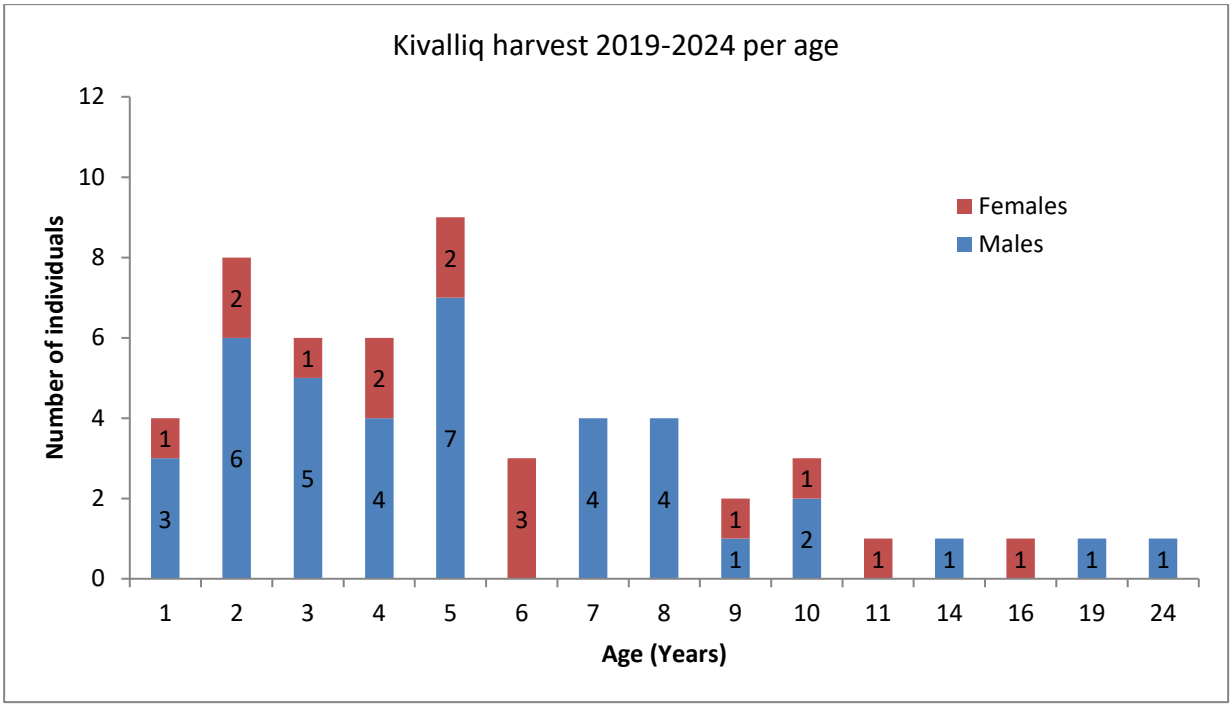


Figure 7: Age and sex structure of the reported Kivalliq grizzly bear harvest, from 2019 to 2024.

The average age of the grizzly bears harvested from 2019-2024 in the Kitikmeot was 7.48 years (n = 62) old, compared to 5.81 years (n = 55) old in the Kivalliq for the same period. The proportion of adults in the Kivalliq harvest was 38%, while proportion of adults in Kitikmeot harvest was 56% (Fig. 8). In both regions, the average age in the harvest slightly increased compare to 2013-2018 harvest period (Kitikmeot = 7.97 years, Kivalliq = 5.89 years; Awan 2021).

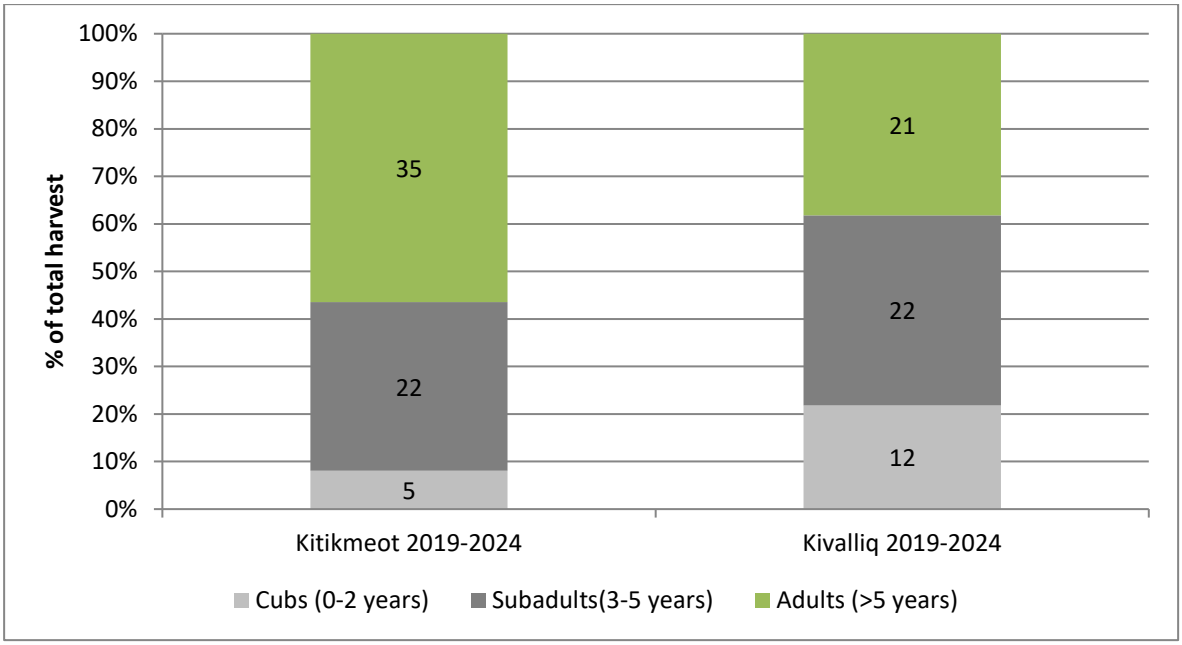


Figure 8: Proportion of age classes in the Kitikmeot and Kivalliq reported harvest, from 2019 to 2024.

In Nunavut, grizzly bears are being harvested both as part of traditional/subsistence (regular hunts) activities and as part of commercial activities (sport hunts). In the Kitikmeot region, approximately 28% of harvested grizzly bear harvest from 2010-2024 were taken in sport hunts, 93% of which were males. For the same period, in the Kivalliq region, approximately 3% were taken as sport hunts, 86% of which were males. Apparently, this low sport hunts success, seems attributed to logistic constraints, non-availability of sport hunters or dearth of high trophy quality. Regular harvest is increasing, grizzly bears on tundra have low reproductive potential (McLoughlin et al. 2003) and cannot afford high subsistence and sport hunt at the same time.

The large home range of males compared to females, and the fact that their highest movement rates happen during spring probably increase their vulnerability to harvest which occurs at the same time of year (den emergence to end of June) (McLoughlin et al. 1999). The continuous high proportion of males in the Kitikmeot harvest indicates

that the population seems to support relatively well the current harvest despite the continuous strong bias towards the removal of males.

The average harvest in the Kitikmeot between 2020 and 2024 was 22 bears per year. It is usually agreed that an annual human-caused mortality of 2-3% is a safe management goal for most northern grizzly bear populations (Sidorowicz and Gilbert 1981, McLoughlin et al. 2003). Awan et al. (2025) population estimate was over 900 grizzly bears in the 156, 490 km2 study area of the western mainland Kitikmeot Region. Overall density estimate from our 2021-2023 study is 5.92 bears/1,000 km^2 in the western mainland of the Kitikmeot Region (Awan et al. 2025). Based on those densities and from the demographic parameters of barren ground grizzly bears in the Kitikmeot region (McLoughlin and Messier 2001), the current harvest thus appears to be within the recommended limits and should result in a stable population.

The average age and age distribution of the 2019-24 harvest is comparable to the long-term average and appears to be relatively well distributed among the whole range of age. This also seems to point towards a stable population and a sustainable current harvest. However, in 2021 the Kitikmeot harvest was 32 bears (Table 1), suggesting careful monitoring of harvest is warranted.

Similarly to the Kitikmeot region, the sex ratio in the Kivalliq harvest has historically been biased towards males representing approximately >80% of the total harvested individuals up to 2009 harvest season, however, the proportion of females increased to 32% in 2010 and 71% in 2011. The recent increase in the proportion of females in the Kivalliq harvest is concerning, but it seems that this occasional harvest is not a long-term conservation concern. The grizzly bear harvest in the Kivalliq should however continue to be monitored closely to ensure the durability of this population for future generations.

Due to limited food resources, barren-ground grizzly bears on the tundra have very large spatial requirements (McLoughlin et al. 2002a) and later ages of maturation than elsewhere, making them more susceptible to over-harvest and disturbance, given their

overall low productivity and slow recovery potential (McLoughlin et al. 2003, McLellan et al. 2017). There is concern that the cumulative effects of various human-caused mortalities and increasing development on the land may cause the grizzly bear population to decline in Nunavut.

4.0 Acknowledgments

Thank you to everyone involved in the collection of the harvest information presented here – the hunters, the HTOs, and the Conservation Officers.

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Participatory assessment of aklak (Grizzly bear, *Ursus arctos*) abundance and distribution in the Kivalliq Region, Nunavut

Final Report, ECCC contract # 3000713560

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Contents

Abstract 3,4
Plain language summary 7
Background 10
Objectives 11
Methods 12
Individual semi-structured interviews 12
Participatory mapping 12
Focus groups 14
Recruitment and participation 15
Community validation 15
Inuit Qaujimajatuqangit in this study context 16
Quantitative abundance assessment methods 17
Results 25
Participant observation of changes in aklak abundance and distribution 25
Participant observations of aklak ecology 26
Participant attitudes towards aklak 29
Geography of aklak observations 30
Quantitative assessment of aklak abundance 30
Conclusions 34
Aklak distribution, abundance, and implications 34
Quantitative estimation of aklak abundance 34
Aklak ecology 34
Recommendations for further investigation 35
Implications for co-management and planning 36
Methodological recommendations for future wildlife research involving Inuit Qaujimajatuqangit 37
Dissemination of study results 38
Acknowledgements 38
References 40

List of Tables

Table 1. Process for ascribing proxy values to aklak sightings without a discrete number noted 21
Table 2. Parameters used for IDW and EBK interpolation processes. 25
Table 3. Summary density statistics from each of the four output raster datasets. 33
Table 4. Estimated abundance of aklak within the study area using IDW and EBK. 33

List of Figures

Figure 1. EXAMPLE OF PARTICIPATORY MAPPING EXERCISE. 13
Figure 2. ARVIAT FOCUS GROUP PROPORTIONAL PILING EXERCISE. 15
Figure 3. THE STUDY AREA IS DELINEATED BY THE GREEN BOUNDARY, WITH ITS SOUTHERN EDGE DEFINED BY NUNAVUT BORDER AND ITS NORTHERN LIMIT EXTENDING TO THE NORTHERNMOST INTERVIEW DATAPOINT. 18
Figure 4. THE DISTRIBUTION OF THE THREE INPUT DATASETS, INCLUDING ALL INTERVIEW DATAPOINTS, HARVEST DATA, AND THE STUDY BLOCKS USED FROM THE AWAN ET AL. (2019) MARK-RECAPTURE STUDY. 19
Figure 5. 10X10 KM GRID OVERLAID ONTO THE STUDY AREA, SHOWING THE GRID CELLS THAT INTERSECTED WITH THE OBSERVATION ZONE IN RED, WITH GRID CELLS CONTAINING OBSERVATIONS IN DARKER VALUES PROPORTIONAL TO THE NUMBER OF SIGHTINGS IN THAT AREA. 24
Figure 6. SIGHTINGS REPORTED BY PARTICIPANTS CATEGORIZED ACCORDING TO TIME PERIOD. 31
Figure 7. RESULTS OF INVERSE DISTANCE WEIGHTING SPATIAL INTERPOLATION, USING INTERVIEW DATA ONLY AS INPUT (MAP A) AND INTERVIEWED DATA COMBINED WITH DNA MARK RECAPTURE STUDY DATA (AWAN ET AL. 2019) AS INPUT (MAP B) 32
Figure 8. RESULTS OF EMPIRICAL BAYESIAN KRIGING SPATIAL INTERPOLATION USING INTERVIEW DATA ONLY AS INPUT (MAP A) AND INTERVIEW DATA COMBINED WITH DNA MARK RECAPTURE STUDY DATA (AWAN ET AL. 2019) AS INPUT (MAP B) 33

ᐃᓱᒪᑎᕆᐅᔾᔨᑦ

ᐊᐳᓪᓚᐳ (ᐊᐳᓪᓕᐃᑦ, ᐅᖃᓗᑕᖏᓚᐅᖅᓯᑎᑎᒃ) ᑕᐅᑐᑕᐅᓱᑦ ᑭᓴᕐᓂᕆᐅᓂᑦ, ᓄᓇᕗᒻᒥᑦ, ᐊᒥᓱᑭᖅᐸᕐᓂᐊᔨᑦ ᑭᓴᕐᓂᖏᑦ. ᐊᑐᓪᓚᐅᖅᑐᑐᒍᑦ ᐃᓅᖅᑭᓱᖅᓯᒪᔨᓂᒃ ᐊᐱᖅᓱᐃᓂᖅᓂᒃ ᐊᒻᒪᓗ ᐃᓪᓚᐅᖅᑕᐅᓂᖅᓯᒪᑦ ᓄᓇᖕᒪᐊᓪᓚᐅᑎᓂᖅᓯᒪᑦ ᐊᔨᑭᓐᓂᖅᓴᐅᑎᓂᒃ ᐅᖅᐅᓱᖅᓴᓂᖅᓯᒪᑦ ᐅᓂᕐᑲᖅᑕᐅᔨᓂᒃ, ᓇᓪᐅᑦᖅᑕᐅᔨᓂᒃ ᐃᓂᐅᔨᓂᒃ ᓇᓐᓂᓂᖏᓐᓂᒃ ᐊᒻᒪᓗ ᖃᓄᖅ ᐅᓄᖅᑎᑎᓂᖏᓐᓂᒃ, ᐊᒻᒪᓗ ᖃᐅᔾᒪᓂᖃᖅᓂᖅ ᐊᐳᓴᐃᑦ ᓇᔪᖏᓐᓂᒃ ᐃᓄᐃᑦ ᐊᖃᐃᓇᖅᑎᖏᓐᓂᑦ ᐊᒻᒪᓗ ᐃᓐᓇᓂᑦ. ᐃᓪᓚᐅᔨᑦ ᐅᖃᐅᔾᖃᓪᓚᓪᐅᖅᓴᑦ ᐅᓄᖅᔪᑎᐊᖅᓂᖏᓐᓂᒃ ᖃᖏᔪᐅᓂᑦᓴᖏᑦ ᐊᐳᓴᐃᑦ ᑭᓴᕐᓂᕆᐅᓂ ᑕᐃᒪᖓᓂᑦ ᐱᑎᐊᓪᓚᐅᖅᓯᒪᓂᑦ 2000–ᖏᓐᓂᓂ, ᑕᑯᓪᒪᓇᖓᓕᓴᑎᒃ 1960–ᖏᓐᓂᓂ ᑭᔪᐊᓂ ᖃᐅᔨᕆᐅᒪᔨᐳᓕᑎᒃ 1990–ᖏᓐᓂᓂ ᑕᐅᓄᖓᖏᓐᓂᐸᕐᓚᐊᓕᑎᒃ. ᑲᑎᒪᖃᑕᐅᔨᑦ ᐅᖃᓪᓚᐅᖅᓯᑦ ᐊᐳᓴᐃᑦ ᐅᓄᖓᓕᐊᖅᓯᖏᓐᓂᑦ ᑭᓴᕐᓂᕆᑎ, ᐃᓄᓕᐊᔨᓴᕐᓂᓚᓂᖏᓐᓂᑦ ᐃᒪᓐᓇᐅᖓᓐᓄᖅᑐᒍᖅ ᐃᓄᓕᐊᔨᓴᕐᓂᐊᓂᖏᓐᓂᑦ. ᐊᐳᓴᐃᑦ ᓄᓇᖅᐸᔪᑐᒍᓕᐊᓂᖏᓐᓂᑦ ᑕᑯᐊᓂᑦ ᑲᑎᒪᓂᖅ; ᑭᔪᐊᓂ, ᔪᓚᖅᐸᔪᑐᒍᒥ ᑕᑯᐅᑲᓚᔾᔪᓂᖅᓴᐅᓕᐅᖅᓯᑦ. ᐊᐳᓴᐃᑦ ᐊᒥᐃᓂᖏᓐᓂᑦ ᓰᓴᐳᖅᑕᐅᓕᐅᖅᓯᑦ ᑕᒪᐳᑭᓂᑦ ᐊᐱᖅᓱᓂᒥᑦ ᑎᑎᕋᖅᓯᒪᔨᑦ ᖃᐅᔨᓴᕐᓂᕆᑕᐅᓂᑯᐃᑦ ᐃᓄᑐᓪᓚᑎᒃ ᐊᒻᒪ ᐊᐱᖅᓱᓂᒥᑦ ᑎᑎᕋᖅᓯᒪᔨᑦ ᖃᐅᔨᓴᕐᓂᕆᑕᐅᓂᑯᐃᑦ ᐊᒻᒪ ᑭᓇᐅᔾᔨᖅᑎᖏᑦ ᖃᐅᔨᓴᕐᓂᕆᑕᐅᓂᑯᐃᑦ ᐊᒻᒪ Empirical Bayesian Kriging (2.88-4.07 ᓇᓄᐃᑦ/1000km^2) ᑲᑎᓪᓚᓂᑦ ᐅᓄᓂᖏᓐᓂᑦ 1081-1515 ᓇᓄᐃᑦ ᖃᐅᔨᓴᖅᑲᐅᔨᒥ. ᑕᒪᖏᑎ ᒥᓴᐅᑕᑎᓯᒪᔨᑦ 95% CI–ᒥ ᒥᓴᐅᑕᑎᓯᒪᔨᑦ Awan et al–ᒧᑦ. (2019) 2.1-6.1 ᓇᓄᐃᑦ/1000km^2 ᐊᐳᔨᓚᖏᓐᓚ ᐊᓯᖏᓐᓄᓕ ᖃᐅᔨᓴᖅᑕᐅᖃᑕᖅᓯᒪᔨᓄᑦ ᐊᐳᔨᓚᖏᓐᓂᒃ ᓇᔪᖏᓐᓂᒃ. ᖃᓄᖅᑲᓂᕐᓂᖅ ᑕᔾᔪᓯᖏᓓ ᓄᑕᒥᒃ ᓇᓪᑕᐅᑕᑦᑕᒥᒃ ᐊᑐᑎᖅᑎᖅᑐᖅ. ᐃᓅᖅᑎᑎᓕᕐᓂᖅᑐᑦ ᐃᓐᓇᑎᖅᑎᖅᑐᖅ ᐃᓄᐃᑦᑎᒍᑎᓯᒪᔨᓄᑦ ᑲᑎᒪᓂᖅᓴᐅᓕᐅᖅᓯᑦ ᐊᐳᔨᓚᖏᓐᓂᒃ ᐊᒻᒪ ᐊᐱᖅᓱᓂᒥᑦ ᑎᑎᕋᖅᓯᒪᔨᑦ ᑎᑎᕋᖅᓯᒪᔨᑦ ᖃᐅᔨᓴᕐᓂᕆᑕᐅᓂᑯᐃᑦ ᐊᒻᒪ ᑭᓇᐅᔾᔨᖅᑎᖏᑦ ᖃᐅᔨᓴᕐᓂᕆᑕᐅᓂᑯᐃᑦ ᐊᒻᒪ ᑭᓇᐅᔾᔨᖅᑎᖏᑦ ᖃᐅᔨᓴᕐᓂᕆᑕᐅᓂᑯᐃᑦ (Awan et al. 2019) ᐊᑐᕐᓚᑎᒃ ᓇᓀᓵᐅᔾᔨᓂᖅᓴᒪᓂᒃ ᐊᒻᒪ Empirical Bayesian Kriging (2.88-4.07 ᓇᓄᐃᑦ/1000km^2) ᑲᑎᓪᓚᓂᑦ ᐅᓄᓂᖏᓐᓂᑦ 1081-1515 ᓇᓄᐃᑦ ᖃᐅᔨᓴᕐᓂᕆᐅᔨᒥ. ᑕᒪᕐᒥᒃ ᒥᓴᐅᑕᑎᓯᒪᔨᑦ 95% CI–ᒥ ᒥᓴᐅᑕᑎᓯᒪᔨᑦ Awan et al–ᒧᑦ. (2019) 2.1-6.1 ᓇᓄᐃᑦ/1000km^2 ᐊᐳᔨᓚᖏᓐᓚ ᐊᓯᖏᓐᓄᓕ ᖃᐅᔨᓴᖅᑕᐅᖃᑕᖅᓯᒪᔨᓄᑦ ᐊᐳᔨᓚᖏᓐᓂᒃ ᓇᔪᖏᓐᓂᒃ. ᖃᓄᐅᑲᓂᕐᓂᓂᖅ ᑕᓴᓯᒥᓴ ᓄᑕᒥᒃ ᓇᓪᑕᐅᑕᑦᑕᒥᒃ ᐊᑐᖅᑐᒥᒃ ᐊᑐᑎᖅᑎᖅᑐᖅ. ᐃᓄᖅᑎᑎᓯᒪᔨᑦ–ᐅᒪᔨᓪᓚᓂᕐᓂᒧᑦ ᐃᓚᓂᓴᓕ ᑭᓴᕐᓂᕆᒥ ᐅᖅᐅᓱᓯᓯᒪᓕᖅᓂᑕᖅ ᐊᓯᖏᓐᓄᓕ ᐊᕐᓴᑐᖅᓯᒪᓂᐅᔨᓄᑦ ᑲᓇᑕᒥ, ᖃᐅᔨᓴᕐᓂᓂᒧᑦ ᐃᓪᓚᐅᖅᑕᐅᔨᓄᑦ ᓄᓇᓕᖕᒪᓂᖅ ᐊᐱᖅᓱᓂᒥᑦ ᐊᑐᓚᐅᖅᓯᒪᓂᖅ ᐊᑐᕐᓴᐅᓂᑦ ᐊᑐᕐᓴᐃᓐᓇᓪᓚᓂᖅ ᐊᓯᖏᓐᓄᓕ ᐊᐳᔨᓚᖏᓐᓂᒃ ᐊᐱᖅᓱᓂᒥᑦ ᐊᐱᖅᓱᓂᒥᑦ ᐊᐱᖅᓱᓂᒥᑦ

nalautaaqniganik pigiarutip aturniqatiaqniaqtuq. Tamna inuuhiliqinirmut-pijuq pijuq Kivallirmi naunaingittuq aadlanut avikturhimajunut Kaanatami, naunaijainiqlu ilaujut naunaiqtait atan'ngujat maligangit atuqtangillu uumunnga akhat pijuq aadlangajuq uumunnga aviktuqhimaninnganut pipluni.

Abstract

Aklak (grizzly bear, *Ursus arctos*) are viewed by residents of the Kivalliq region, Nunavut, as increasing in abundance across the region. We used semi-structured interviews and participatory mapping exercises to elicit narratives of encounters, estimates of spatial distribution and abundance, and knowledge of aklak ecology from Inuit hunters and elders. Participants uniformly emphasized an increase in the relative abundance of aklak across the Kivalliq region since the beginning of the twentieth century, with rare encounters prior to the 1960s but frequent encounters from the 1990s onward. Participants emphasized that aklak were not historically abundant in the Kivalliq, suggesting a range expansion rather than a population in recovery. The distribution of aklak was primarily inland, where encounters frequently occurred along lakes and rivers; however, sightings in coastal areas have become more common. Aklak densities were calculated from both interview data alone and interview data plus DNA sampling grid hits (Awan et al. 2019) using inverse distance weighting and Empirical Bayesian Kriging (2.88-4.07 bears/1000km^2) for a total abundance of 1081-1515 bears in the study area. Both estimates are within the 95% CI range estimated previously by Awan et al. (2019) of 2.1-6.1 bears/1000km^2 and are similar to other previous studies in similar habitat. Further exploration of this new estimation method would be useful. The socio-ecological context of the Kivalliq is not analogous to other regions in Canada, and study participants emphasized that management policies and practices for aklak should be distinctive to the region's context.

Résumé

Selon les résidents de la région de Kivalliq, au Nunavut, l'abondance d'aklaks (grizzli, *Ursus arctos*) a augmenté dans l'ensemble de la région. Nous avons eu recours à des entrevues semi-structurées et à des exercices de cartographie participative pour obtenir de chasseurs inuits et d'aînés des récits de rencontres, des estimations de la répartition spatiale et de l'abondance et des connaissances du milieu écologique de l'aklak. Les participants ont rapporté de façon unanime une augmentation de l'abondance relative de l'aklak dans l'ensemble de la région de Kivalliq depuis le début du vingtième siècle, les rencontres ayant été rares avant les années 1960, mais fréquentes depuis les années 1990. Les participants ont insisté pour dire que l'aklak n'a jamais été abondant dans les environs de Kivalliq, ce qui suggère une expansion de l'aire de répartition plutôt qu'une population en voie de rétablissement. La répartition de l'aklak se limitait surtout à l'intérieur des terres, et les rencontres se produisaient fréquemment le long des lacs et des rivières; cependant, les observations dans les zones côtières sont devenues plus fréquentes. Les densités ont été calculées à partir des données recueillies lors des entrevues elles-mêmes et des données tirées des entrevues en plus des occurrences de la grille d'échantillonnage de l'ADN (Awan et coll. 2019) en utilisant la pondération de la distance inverse et le krigeage bayésien empirique (2,88 à 4,07 ours/1 000 km^2) pour une abondance totale de 1 081 à 1 515 ours dans la zone étudiée. Les deux estimations se situent dans l'IC de 95 % estimée précédemment par Awan et coll. (2019) de 2,1 à 6,1 ours/1 000 km2 et sont similaires à celles des autres

études semblables pour un habitat semblable. Un approfondissement de cette nouvelle méthode d'estimation serait utile. Le contexte socioécologique de Kivalliq ne s'apparente en rien aux autres régions du Canada, et les participants de l'étude ont souligné que les politiques et pratiques en matière de gestion de l'aklak devraient être adaptées au contexte particulier de la région.

ᐅᖃᐅᓯᑐᐃᓐᓇᖅᑐᑦ ᓇᐃᔨᖃᖅᓯᒪᔨᖅ

- ᖃᐅᔨᓴᓂᕐᒧᑦ ᑐᖏᖅᖅ: ᖃᐅᔨᓂᕐᑎᓗᓂ ᓄᓇᓕᖅᑎ ᖃᐅᔨᒪᓂᕐᒥᐅ ᐊᑯᐅᓕᐃᑦ ᐅᓄᕐᓂᖅᑭᓂᐅᓂᐅᑦ, ᓇᓂᐅᓂᖅᑭᓂᐅᓂᐅᑦ ᐊᓪᒪ ᐅᒪᔨᑦ ᓇᔨᖏᖅᑎᑕᓗ ᖃᓄᐃᑦᑐᓂᕐᑎᑦ ᑭᓴᕐᓗᑎᓕᓗ ᖃᐅᔨᔪᔨᑕᐅᔨᑦ ᐃᓄᐃᑦ ᖃᐅᔨᒪᔪᑐᖃᖅᓂᐅᑦ (IQ) ᐃᑲᔨᕐᓗᑎᑦ ᐱᑕᖃᖅᑐᑦ ᑭᐅᓕᓂᐊᑭᑎᓄᑦ ᑲᑎᖅᓯᖅᓯᒪᔨᓂ ᖃᐅᔨᑎᓕᓗᑎᑦ ᐅᒪᔨᓕᕆᓂᕐᒧᑦ ᐊᐅᓪᓚᑎᑦ ᐊᑐᕋᓯᑎᓂᖅ.

- 2022-2023–ᒥ, ᓄᓇᓕᖅᓂᑦ ᐃᖅᑲᓇᐃᔪᖅᑎᑦᑕᓪᓚᐅᖅᑐᑦ ᐃᓄᖅᓂᑦ ᖃᐅᔨᔪᖅᑎᓂᑦ ᐊᐱᖅᓯᖅᓴᑎᖅ ᐊᕐᕕᐊᓂᑦ ᖃᒪᓂ'ᑐᐊᒥᓕᓗ, ᐃᓕᓴᓂᐊᕐᑎᑦᑎᓕᑎᖅ 2023 ᐱᑎᐊᖅᓂᖓᓂᑦ. ᐊᓪᒪ ᓯᓕ, ᑭᖓᓂᖓᑦᑎᓴᓂ ᐊᐱᖅᓯᓂᐅᖃᖅᑕᓪᓚᐅᖅᑐᑦ ᖃᒪᓂ'ᑐᐊᒥᑎ ᐊᖓᔪᓯᖅᑐᑦᑕᓪᓚᐅᖅᑐᑦ ᐃᓕᓕᐅᔪᖅᑐᑦ ᖃᐅᔨᔪᖅᑕᐅᔪᓂᐅᑦ. ᑕᒪᕐᐳᖅ ᖃᒪᓂ'ᑐᐊᒥᑎ ᐊᓪᒪ ᐊᕐᕕᐊᕐᒥᐅᑦ ᐃᓕᓕᐅᔪᑦ ᐃᓕᓴᕐᓗᑎᓂᖅᑐᑦ ᐅᓄᖅᓯᓂᓕᕐᓂᓕᕆᓂᖅᑐᑦ ᐃᓄᐃᑦ ᑕᑯᖅᑕᐅᑎᖅᑕᐅᖅᑐᑦ ᐊᖅᑲᓂᑦ ᑭᓴᕐᓗᑎᑦ ᐱᑕᕐᓂᖅᑐᑦ ᐃᓕᓴᕐᓗᑎᑦ ᑕᐃᒪᓐᖓᑦ 1990–ᓂᐅᓂ ᐃᓚᖓ 1960–ᓂᐅᓂᒋᑎᓪᓗᒍ.

- ᐃᓄᐃᑦ ᐊᓪᒪ ᐊᖅᑲᐃᑦ ᐊᑯᓂᓕᓂ ᑯᖅᑯᓂ ᐊᓪᒪ ᑕᓯᖅᑯᑦ.

- ᐊᖏᔪᓯᖅᑕᐅᓂᖅᑐᑦ ᐊᖅᑲᓐᓂ ᐃᓯᒪᒋᔭᐅᓂᒃ ᑲᔪᓯᑎᑕᐅᔪᓂᖅᑐᑦ ᐃᓕᓴᕐᓗᑎᓂᑦ ᐅᓄᖅᓯᓂᓕᕆᓂᐊᑎᓪᓚᐅᑦ.

- ᐱᔭᐅᔪᓂᕐᓇᖅᑐᑦ ᑭᖑᓪᖅᑎᑕᐅᔪᓪᓚᐅ ᐊᖅᑲᓗᑕᐅᓂᖅᑐᑦ ᐃᓯᒪᒋᔭᐅᓂᒃ ᐱᐅᓚᐊᖏᓂᖓᓂᖅᑐᑦ ᐅᓂᒃᓯᓚᓂᖅ ᐅᓄᖅᓯᓂᓕᕆᓂᐊᑎᓪᓚᐅᑦ ᐊᖅᑲᓐᓂ ᐅᓄᖅᓯᓂᓕᕆᓂᐊᑎᓪᓚᐅᑦ ᐊᓪᒪ ᐅᒪᔨᑦ ᐃᓄᐃᑦ ᑕᑯᖅᑕᐅᔪᓂᖅᑐᑦ ᐊᖅᑲᓐᓂ ᐅᓄᖅᓯᓂᓕᕆᓂᐊᑎᓪᓚᐅ

- Atuqhugit qitqani inuit aklaklu hilait atuqtaujut kuukap uvalu tahiitlu.
- Angunnahuaknirmun aklak tautukhimajunik uumajunik ilaujunik pihimajunik amigaitilaangit angiklivalialiqtun.
- Qaffiutilaangit pittailijakhaillu aklak angunahuarniq ihumagijaungittut akhuraaluk piiqtaulimaittuqluuniit nunamiknit.
- Qaffiunigit aklak nunaujarmit Inuit takujainik nalaumajunik nanijaujunik hivuanit ilituqhaunmit naunaijagaujut DNA-nik aklak amiqmit mitquinik qiuknik napaqutinik atuliqhimajut nunami.
- Inuit atuqhimaaqtangit anguniarniq, angunahuarniq, pirlukhutiklu tuktu, iqhitaaqtaujut amigairjumiqtumit nunami amigaitni, atuqattaqhugit amiqhaat pirujat igluqpannuallu qajangnaqhivaktut pittailiplugillu hapkuat pirujat.
- Havaanga aklak inuuhiqmut-pijuq nunanga Kivallirmi avikturhimajuq aadlangajuq aadlanit aviktuqhimajunit Kaanatami.
- Amigaiqpalianigit atuqtait, ilaujut takuvaktut aklak-mik nuutqaqtitiniganik inuuhiqmi inuujuhiqmilu atuqtumik, ukuagugituq aalatqiit ihariagijauniginik angunahuarutikharnik nuutqaqtitijutinik.

Plain language summary

- Study objective: elicit local knowledge of aklak abundance, distribution and ecology in the Kivalliq region and use insights from Inuit Qaujimajatuqangit (IQ) to complement existing scientific data to inform wildlife management practices.
- In 2022-2023, locally hired Inuit research assistants conducted interviews in Arviat and Baker Lake (Qamani’tuaq), with workshops held in early 2023. Additionally, previous interviews provided by the Baker Lake Hunters and Trappers Organization (HTO) were included in the analysis. Both Baker Lake (Qamani'tuaq) and Arviat participants attested to increases in human encounters with aklak across the Kivalliq region since at least the 1990s and perhaps as early as the 1960s.
- Encounters between humans and aklak tend to occur along rivers and lakeshores.
- The harvest of aklak is viewed as sustainable by participants as the population is increasing.
- Quotas and restrictions on aklak harvest are considered not pragmatic or desirable by community members.
- The number of aklak from mapping Inuit observations aligns with findings from a previous study that sampled DNA from aklak hair snagged on wooden tripod posts deployed on the land.

- Inuit practices around hunting, harvesting, and caching caribou, are threatened by an increase in aklak abundance, as more frequent encounters with aklak scavenging caches and cabins pose safety risks and disrupt these practices.
- The role of aklak in the socio-ecological landscape of the Kivalliq region is distinct from that in other regions in Canada.
- With increasing encounters, participants view aklak as a disruption to the socio-ecological system, rather than as a species in need of conservation interventions.

Résumé en langage simple

- Objectif de l'étude : recueillir des connaissances locales sur l'abondance de l'aklak, sa répartition et son milieu écologique dans la région de Kivalliq et tirer parti des perspectives issues de l'Inuit Qaujimajatuqangit (IQ) pour bonifier les données scientifiques actuelles afin d'orienter les pratiques en matière de gestion des espèces sauvages.
- En 2022-2023, des assistants de recherche inuits embauchés localement ont mené des entrevues à Arviat et à Baker Lake (Qamani'tuaq) et ont animé des ateliers au début de 2023. De plus, des entrevues réalisées précédemment par l'organisation de chasseurs et de trappeurs (OCT) de Baker Lake ont été incluses dans l'analyse. Les participants de Baker Lake (Qamani'tuaq) et d'Arviat ont tous confirmé une augmentation du nombre d'humains ayant rencontré un aklak dans l'ensemble de la région de Kivalliq depuis au moins les années 1990, et peut-être même depuis les années 1960.
- Les rencontres entre un humain et un aklak ont tendance à se produire le long des lacs et des rivières.
- La récolte de l'aklak est perçue comme étant durable par les participants puisque la population augmente.
- Selon les membres des collectivités, les quotas et les restrictions visant la récolte de l'aklak ne sont ni pragmatiques ni souhaitables.
- Le nombre de spécimens de l'aklak, d'après la cartographie tirée des observations inuites, est conforme aux conclusions d'une étude précédente qui avait prélevé des échantillons d'ADN des poils de l'aklak accrochés à des trépieds en bois déployés dans les terres.
- Les pratiques inuites en matière de chasse, de récolte et de cache pour le caribou, sont menacées par une augmentation de l'abondance de l'aklak, puisque les rencontres plus fréquentes avec l'aklak détruisant les caches et les cabanes présentent un risque pour la sécurité et perturbent ces pratiques.
- Le rôle de l'aklak dans le paysage socioécologique de la région de Kivalliq est différent de celui qu'il joue dans d'autres régions du Canada.

- En raison de l’augmentation des rencontres avec l’aklak, les participants le perçoivent comme une perturbation du milieu socioécologique et non comme une espèce devant faire l’objet d’efforts de conservation.

Background

Members of the HTOs of Baker Lake (Qamani'tuaq) and Arviat have shared that they have witnessed an increase in aklak[7] abundance in the Kivalliq region for approximately the last fifty years (Clark and Slocombe 2011; Lokken et al. 2019). Community members have expressed concerns about human-wildlife conflict, as aklak cause damage to hunting cabins, dig up caches of caribou meat, and can pose risks to community safety (Nirlungayuk 2011; Manning 2022).

Climate change is impacting northern wildlife population dynamics (Humphries et al. 2004) and Indigenous communities are well-positioned to observe and interpret these changes (Etiendem et al. 2020; Henri et al. 2020; Jessen et al. 2022; Tomaselli et al. 2018). Furthermore, as pointed out by McLoughlin and Stenhouse (2021, 51), more information is needed to understand how climate change may be contributing to improved ecological conditions for aklak and shaping the dynamics of their range expansion in this region. The increasing population in the Kivalliq region provides a contrast to other areas in Canada with the implication that management strategies for this region ought to significantly differ from other regions (McLoughlin and Stenhouse 2021, 48-51).

Like all communities in Nunavut, the hamlets of Arviat and Baker Lake were formed as a result of Canadian government-driven relocations of Inuit people in the 1950s and 1960s (Tester and Kulchyski 2011). Arviat is now home to descendants of the Paallirmiut, who traditionally hunted in both inland and coastal areas, and Ihalmiut, whose territories lay inland near Ennadai Lake. Qamani'tuaq is home to eleven Inuit groups, including the coastal Utkuhiksalingmiut, Qaingningmiut, Paallirmiut, Natsilingmiut, Killirnirmiut, Iluilirmiut, Hanuniqtuurmiut and the inland Harvaqtuurmiut, Haningajurmiut, Akilinirmiut, and Ahiarmiut. The latter inland-oriented groups were known in the ethnographic literature as the Caribou Inuit due to their inland orientation and reliance on terrestrial mammals like caribou for subsistence (Tester and Kulchyski 2011). Inuit culture is deeply rooted in hunting, with knowledge of wildlife behaviour and distribution vital for Inuit survival.

Many of the Elders interviewed for this study were born on the land before the settlement of the hamlets and are fluent in their group's particular dialect of Inuktitut. Thus, they have access to knowledge gained from both their own long experiences on the land and knowledge passed on through oral traditions. Arviat, the southernmost mainland community in Nunavut, and Baker Lake, the only inland community. Prior to forced relocation in the 1950s, many of the Elders in the Kivalliq were members of inland-oriented Inuit groups. This is in contrast to other communities in Nunavut, which are coastal and oriented towards marine resources (while also still using inland resources). All participants asserted that aklak were more likely to be found inland, therefore community members in Baker Lake and Arviat have greater familiarity with the inland areas where aklak are more often found. Historically, aklak encounters were rare, but they were still known to Inuit communities: "*living and seeing with grizzly bears have always been in part of Inuit life*" (Norman Attungala, Baker Lake 2003).

[7] The terms "aklak" and "grizzly bear" will be used interchangeably throughout this report.

Objectives

The overall purpose of this study is to estimate grizzly bear abundance and distribution in the Kivalliq region by consulting Inuit Qaujimajatuqangit on aklak with existing scientific data and presenting the findings in ways that inform grizzly bear co-management. These two biological parameters are identified as monitoring priorities in the Co-Management Plan (Government of Nunavut 2017, 14).

Objectives of this study were to:

1. Generate estimates of grizzly bear abundance and changes in distribution through participatory exercises with HTO members in Arviat and Baker Lake.
2. Document Inuit observations and understanding of grizzly bear population trends, distribution, ecology, and the social, economic, cultural, and biological aspects of coexisting with grizzly bears— (including perceptions of risk, hunting practices, predation on caribou, human-bear conflicts, and effects of climate change).
3. Compare new data with past data from Baker Lake, specifically HTO's 2002 interviews (if available), Clark's 2003 interviews, and Loken's 2014 TUNDRA interviews to identify changes and shifts in social tolerance for grizzlies in the region.

Methods

Individual semi-structured interviews

Semi-structured interviews were selected as appropriate given the cross-cultural context of research, small sample size, and our research goal to both elicit and respect Inuit Qaujimajatuqangit. Semi-structured interviews were conducted locally in Baker Lake (n=6) and Arviat (n=16). Questions revolved around participant observations of aklak abundance, distribution, ecology, and conflict with humans. Individual interviews were conducted by Emil Arnalak in Arviat and Russell Toolooktook in Baker Lake from April 2022 to February 2023 a local community member in either English or Inuktitut, based on participant preference, with most conducted in Inuktitut. Interviews were translated by local research assistant, Emil Arnalak, while Dr. Lauren Harding transcribed those conducted in English. Inuktitut interviews of 2002 from Baker Lake are still undergoing translation at the time of this report.

In addition to the data from community-based interviews in 2023, this study includes past interviews conducted in Qamani'tuaq in 2002 by Natasha Thorpe with Golder & Associates, translated by Lizzie Iblauk (n=1), and interviews conducted by Douglas Clark during his doctoral research in 2003 (n=12; Clark and Slocombe 2011). More interviews are undergoing translation, which once received will expand the temporal range and scope of our data set from Qamani'tuaq. NVivo software was used to conduct thematic analysis, identifying key themes and observations that emerged from the qualitative interview data.

Participatory mapping

Participatory mapping was a key method used in this study to capture Inuit knowledge and perceptions of aklak distribution. During each individual interview participants used markers to draw on a topographic map with a clear overlay to mark locations where they had encountered aklak and indicating areas where they believed aklak were likely or unlikely to be found (Figure 1). Standard topographic maps were used as base maps, and spatial categories emerged through dialogue with the participants and were not pre-determined by researchers. This approach aligns with established participatory mapping methods that seek to represent local perceptions of the landscape and allow for the cultural contextualization of data when possible and appropriate (Ernoul et al. 2018; Riddell et al. 2022; Robinson et al. 2016). The maps also provided a useful tool for the elicitation of place-based qualitative data. Qualitative spatial data was collected and marked on the map throughout the interview process, including areas of human-wildlife conflict (i.e., cabin break-ins) and known denning sites. The map overlays were digitized and integrated into spatial database by a geospatial analyst at Caslys Consulting (B.C.), ensuring the data was aligned with the geospatial framework of the study.

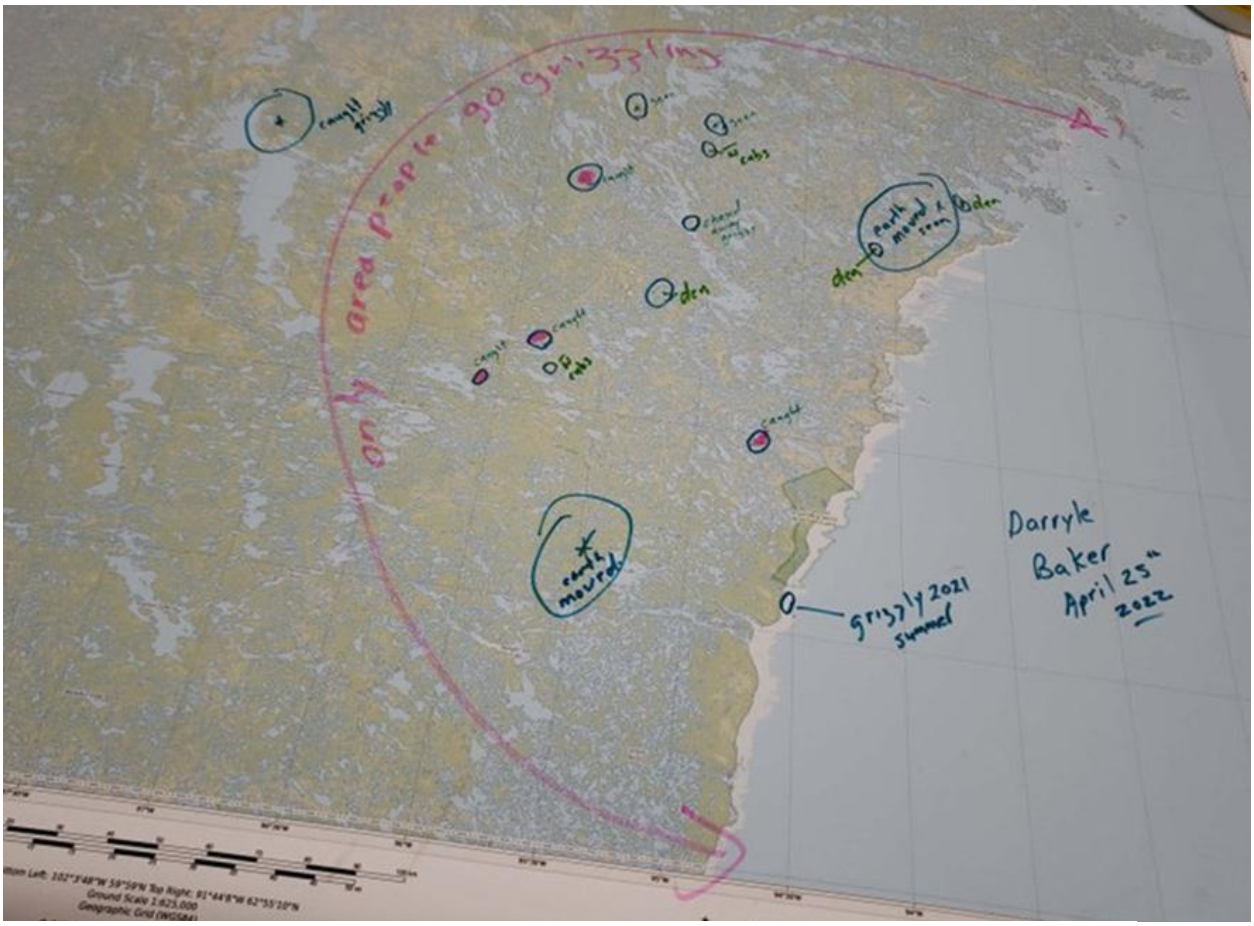


FIGURE 1. EXAMPLE OF PARTICIPATORY MAPPING EXERCISE.

Focus groups

The second phase of data collection employed focus groups and proportional piling methods to validate initial findings and gain deeper insights into aklak distribution and abundance. Focus group interviews were conducted in Arviat (n=6) and Baker Lake (n=2) in February 2023 during a community visit by Dr. Harding. In addition to the continued collection and confirmation of qualitative data on aklak, a second participatory mapping exercise was carried out oriented towards capturing relative abundance and spatial distribution.

Using proportional piling methods, participants collectively estimated aklak density on a topographic map (Figure 2), placing beads to represent relative abundance categories of 'few' or 'many'. Photographs of estimated aklak density from the group interviews was combined with participatory mapping data from individual interviews. A second proportional piling method was attempted in Arviat to graph change in population over time, but participants rejected this method in favour of a narrative approach to discuss changing abundance. Focus group sessions were recorded, transcribed, and thematically analyzed using NVivo software.

Proportional piling has been used effectively for estimating abundance of wildlife that congregate in groups like ungulates (Tomaselli et al. 2018) but may not be an effective method for estimating abundance of wildlife species like aklak which are encountered sporadically and individually. However, this method remains useful for this study as it enables participants to convey their perceptions of relative abundance across the landscape, offering valuable qualitative insights into aklak distribution patterns.

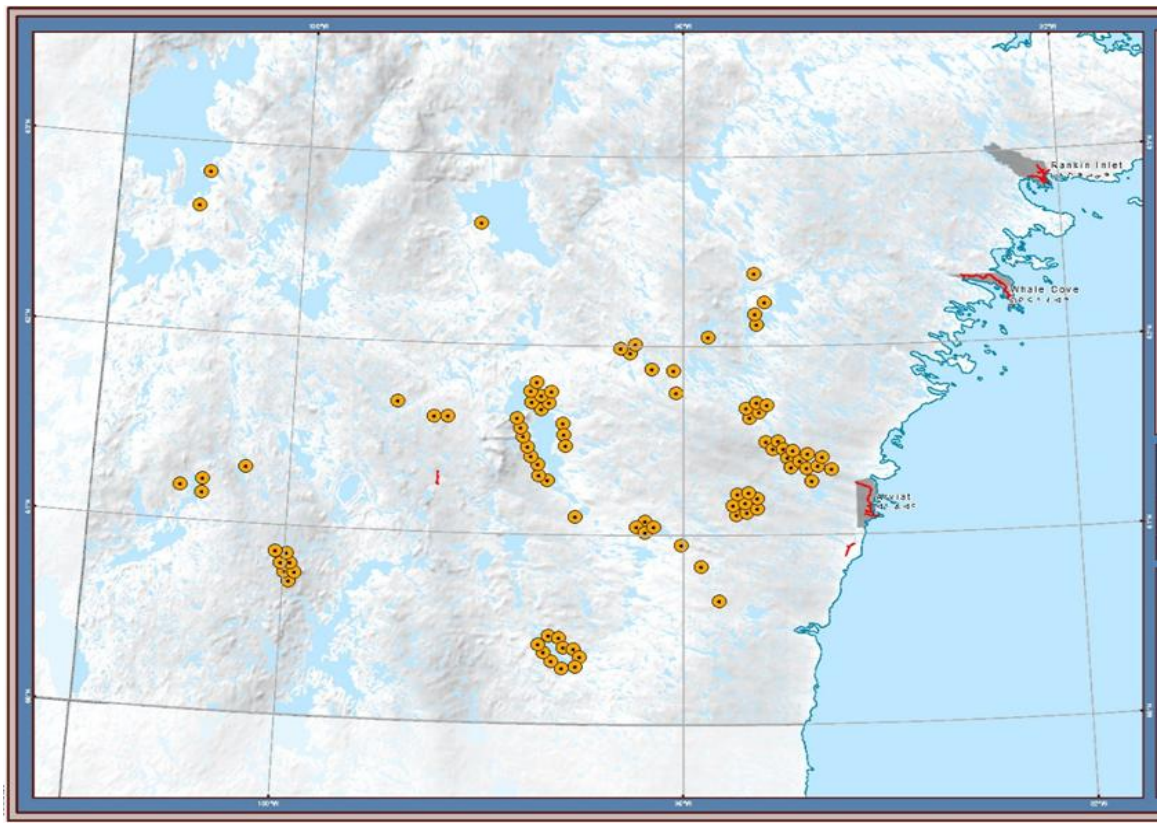


FIGURE 2. ARVIAT FOCUS GROUP PROPORTIONAL PILING EXERCISE.

Recruitment and participation

A list of potential participants in each community was created by the Government of Nunavut (GN) Department of Environment (ENV) and HTO board, who were then invited to give interviews in the wildlife office. This study faced several delays due to the need for face-to-face interviews, which were difficult to conduct during the COVID-19 pandemic. Focus group interview participation in Arviat was moderate with ten participants and participation in Baker Lake was low with two participants. The latter may be attributed to poor weather conditions, as it was communicated to the researchers that several Elders did not want to leave their homes in the high winds and low temperatures (windchill of -67 degrees Celsius) present during the one day allocated for the focus group.

Community validation

Several strategies have been employed in this study to ensure that participant contributions are accurately represented. This includes the use of a local interviewer in each community, which helps preserve the linguistic and cultural context of knowledge communication, as well as the fact that participants were invited from a list of acknowledged community experts generated by the GN, ENV and Inuit HTO.

Prior to visiting Arviat and Baker Lake, Harding reviewed the collected interviews and prepared a list of clarifying questions for focus group participants. During the focus groups, these questions were asked to

clarify or expand on statements from the semi-structured interviews, as well as to better understand whether there was any disagreement or consensus among participants on the major themes that emerged. It should be noted that, as an 'outsider,' Dr. Harding was unlikely to observe dissenting opinions or internal community conflicts during the short interactions she had with participants. At the same time, a second attempt at community validation will occur when the contents of this report are shared with both participants and the wider community in Arviat and Baker Lake. We hope to circulate the contents of the final report in both English and Inuktitut and make use of digital storytelling tools to disseminate the study results in forms that are accessible and culturally appropriate.

Inuit Qaujimajatuqangit in this study context

This study explicitly engaged with Inuit Qaujimajatuqangit (IQ), in keeping with the Government of Nunavut's objective to integrate Inuit values across government programs and services. This approach also supports the specific goal of using IQ as expressed in the GN's Grizzly Bear Co-Management Plan (Government of Nunavut 2017).

The term Inuit Qaujimajatuqangit encompasses all aspects of traditional Inuit culture including values, worldview, language, social organization, knowledge, life skills, perceptions and expectations (Wenzel 2004, 240). It is a purposefully holistic and fluid concept, reflecting the intertwining of values, tradition, skill, and knowledge in Inuit expertise, as well as the continued expansion of Inuit knowledge through lived experience (Greene 2021, 198). It cannot simply be equated with "traditional ecological knowledge" (TEK) as it includes knowledge disseminated through ongoing oral traditions, knowledge accumulated through personal experience, and the interpretation of knowledge through the lens of Inuit values (Greene 2021, 207; Oosten and Miller 2018). Furthermore, IQ is relational, meaning the social relationship between speaker and listener influences its content and so cannot be separated from the social context in which it is articulated (Tester and Irniq 2008). For wildlife management, it is important to recognize that, for Inuit people, relations with non-human animals are both social and subsistence-based (Laugrand and Oosten 2022; Rasing 2018, 6-8).

It should be noted that IQ, as a formalized concept, has "predominately pertained to the interface between foreign-imposed non-Inuit governance and Inuit culture" (Greene 2021, 209). As such, our interpretation of IQ and the articulation of Qaujimajatuqangit by our research participants are artefacts of cross-cultural encounters. What this means is that we, as the authors of this study, are not attempting to conclusively represent IQ on aklak, but rather are collaboratively articulating knowledge on aklak that study participants considered useful, important, and of interest to non-Inuit researchers, policy makers, co-management and conservation practitioners.

Much of the interview data was collected in Inuktitut and the process of translation can impact the interpretation of qualitative data, especially in a cross-cultural context (Hennink 2008, 21). We wish to acknowledge these barriers to interpretation, but also highlight our attempts to counter it with community validation, as this reflexivity and transparency is key to scholarly rigor in qualitative methodologies (Hennink 2008, 22).

An important methodological consideration in this research is our efforts to address to what Pfeifer (2018, 29) has termed the 'credibility gap,' where Inuit knowledge "does not have [the same] credibility compared to Western academic knowledge". Pfeifer argues that there is a need to recognize Inuit knowledge and everyday practices on the land as "a qualitative mode of inquiry producing scientific

evidence" and Inuit communities (hunters/harvesters/knowledge keepers) as equally valid references as those esteemed by academic publication standards (Pfeifer 2018, 31). This is particularly important given the impetus to decolonize research relations and the proven insights of Indigenous science in understanding complex socio-ecological systems (Reid et al. 2022; Turner et al. 2022). While the interview data collected in this study is descriptive and qualitative, it provides important insights that can:

1. be investigated quantitatively (below),
2. complement existing scientific data, and
3. direct future research priorities.

Furthermore, Inuit Qaujimajatuqangit is useful for managers attempting to understand temporal and spatial patterns in wildlife harvest, establish baselines to understand the rapid change happening in Arctic ecosystems, and understand community impacts of ecological change (Etiendem et al. 2020).

Quantitative abundance assessment methods

Defining the study area

The study area was defined by the geographic extent of all mapped data collected during the interview process (Figure 3). It includes the southern portion of Nunavut, with its northern boundary set by drawing a horizontal line from east to west at the latitude of the northernmost mapped interview observation.

Abundance estimation process

Data overview

The purpose of this segment of the project was to estimate the total number of aklak within the study area using knowledge collected on physical maps during interviews with community members and hunters. In addition to this primary data source, we incorporated two additional datasets on aklak locations within the study area for the period of interest. The first was derived from a DNA capture-recapture study conducted in 2016 and 2017 (Awan et al. 2019), and the second comprised GN, ENV harvest data spanning from 2008 to 2023. While all datasets fell within the defined study area, the coverage of aklak locations within each dataset differed. Figure 4 displays the distribution of each of the three datasets across and within the study area, with the DNA capture-recapture study represented by blocks, while the other two datasets are depicted as points.

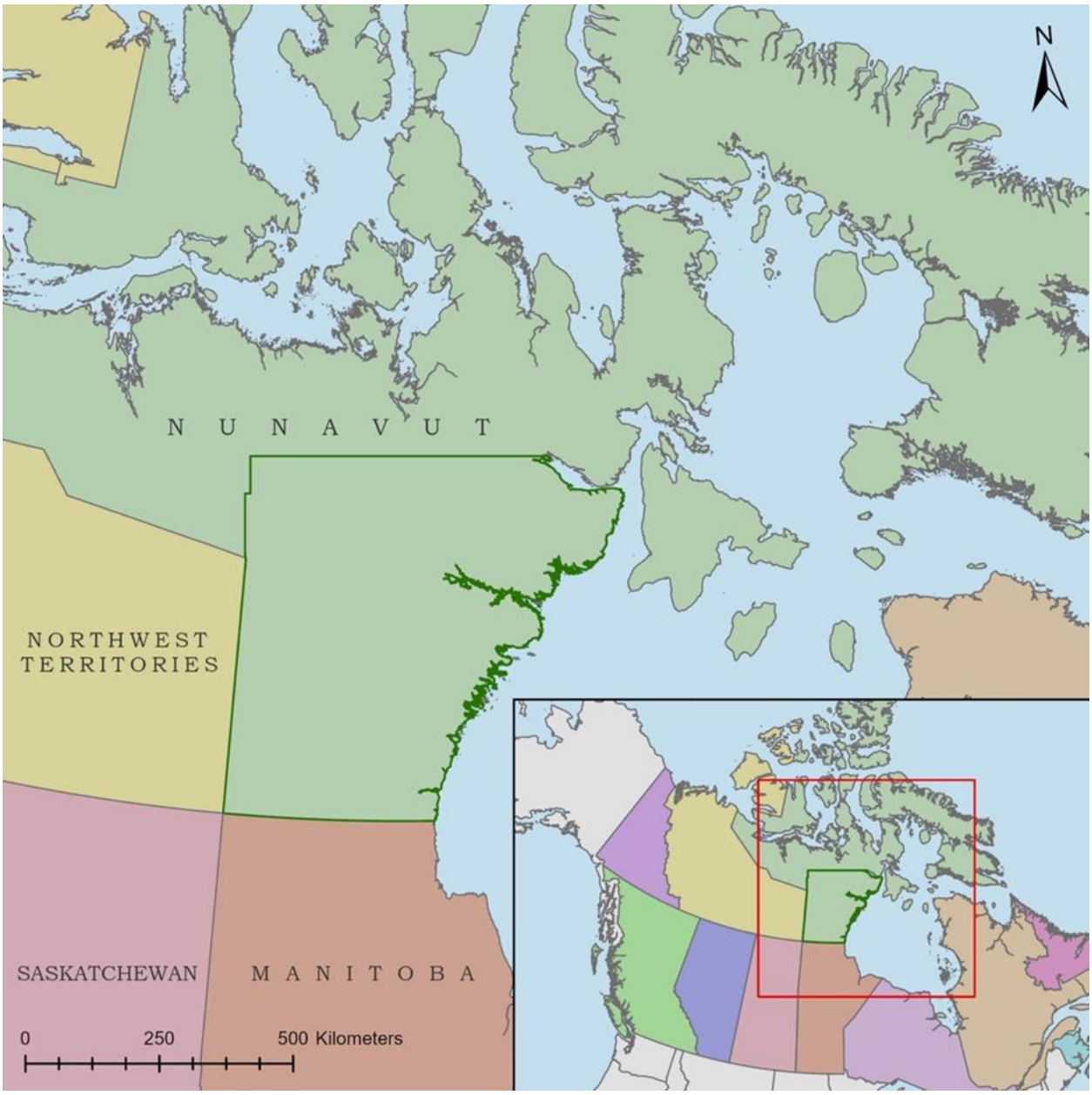


FIGURE 3. THE STUDY AREA IS DELINEATED BY THE GREEN BOUNDARY, WITH ITS SOUTHERN EDGE DEFINED BY NUNAVUT BORDER AND ITS NORTHERN LIMIT EXTENDING TO THE NORTHERNMOST INTERVIEW DATAPOINT.

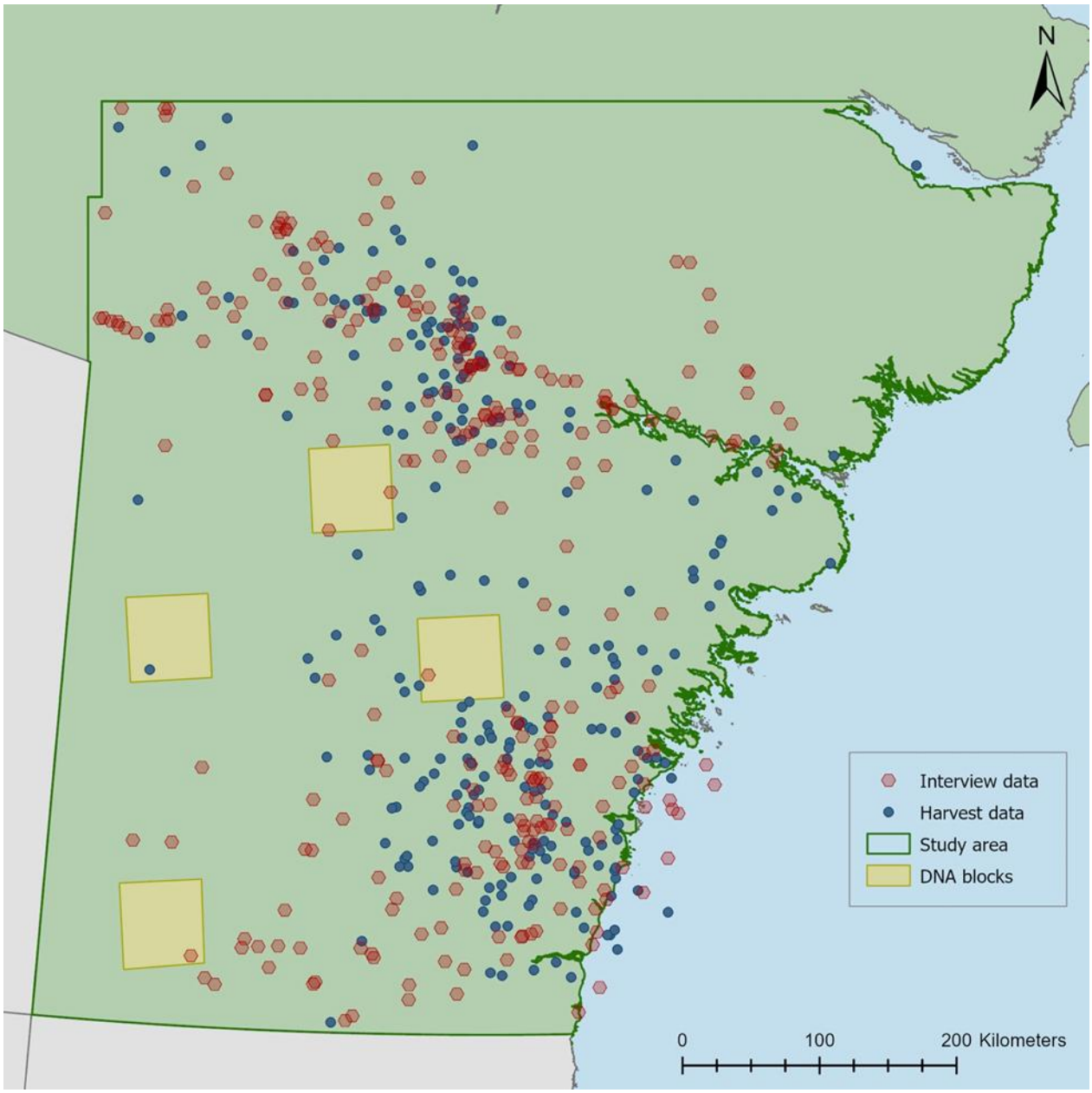


FIGURE 4. THE DISTRIBUTION OF THE THREE INPUT DATASETS, INCLUDING ALL INTERVIEW DATAPOINTS, HARVEST DATA, AND THE STUDY BLOCKS USED FROM THE AWAN ET AL. (2019) MARK-RECAPTURE STUDY.

Notably, data collected based on people's reported land use practices (the interview data and harvest data) largely overlaps, while data collected by Awan et al. (2019) using DNA capture-recapture does not intersect with the distribution of the other two datasets. Based on the distributions of the datasets, we identified the DNA study data as having the potential to complement the interview data in the spatial interpolation process. We determined that the harvest data was likely to overlap with the interview data not only in spatial extent, but also in individual aklak sightings (as interviewees reported places where aklak were harvested as well as observed in other contexts). Therefore, while the harvest data was not appropriate for use in the spatial interpolation process, it could be used as an alternative data source for abundance estimation via spatial interpolation.

Cleaning and processing interview data

The first step in the abundance estimation process involved cleaning and processing the interview data. All interview data were digitized using ArcGIS Pro software and saved into a geodatabase. Data points were digitized in two ways. First, vector shapefiles were created to accurately capture all map elements added by participants, including point, line, or polygon features. This required that interview data were digitized and saved into three separate feature classes (point, line, and polygon) depending on the feature type. Attribute tables were used to ascribe additional data to each feature, including an anonymized participant code; a description of the feature class (point, line, or polygon); and any comments that were added to the map for contextualization along with the feature. Second, all datapoints were compiled into a single feature class. This required that line and polygon features were transformed into points by calculating the centroid of each shape. Once all datapoints were compiled into a single feature class, the attribute table was exported, and data processing continued using Microsoft Excel.

The original maps produced in the interviews were consulted along with the feature comments to further categorize the data in Microsoft Excel. First, features were categorized according to whether they identified aklak signs, sightings, or something else (e.g., some participants noted typical aklak travel routes or geological features like eskers). Signs were further categorized by type, and columns were created to account for temporal data, including one for the year that the observation was made and one for the season (e.g., spring, summer, fall, or winter). Finally, a numeric column was created to store an estimate of the number of bears seen for each datapoint that constituted a sighting. For datapoints that suggested sightings but did not explicitly provide information about the number of aklak seen, conservative estimates were made using proxies (Table 1). Because the interviews produced qualitative data, we were conscientious in assigning numeric values to non-quantitative data points, including by keeping track of all assumptions with the justifications that informed decision-making in assigning proxy values throughout this process. Data points containing locations of aklak sightings (and locations of aklak signs that were "counted" as sightings) were taken forward for use in the spatial interpolation process.

Importantly, interviewees consistently observed that the abundance and locations of grizzly bears in the region had notably changed during the decade of the 1990s. As we are interested in recent aklak abundance, we removed aklak sightings made prior to the year 1990 from the dataset to be used for interpolation.

Using the single feature class containing all spatial data collected in interviews, we applied a kernel density calculation in ArcGIS Pro to produce a raster that contained information about the density of interview observations across the study area. The raster was classified using natural breaks into ten

classes, and the bottom three classes (areas with the lowest observation density) were removed. The resulting area represented our "observation zone," which was used to differentiate between regions of the study area for which there was substantial local knowledge about aklak abundance and regions for which aklak abundance was largely unknown to local populations. In essence, the observation zone represents participants' land use patterns. The goal of the abundance estimation process thus requires using data contained in the observation zone (or "known" regions) to project aklak distribution in the areas of the study area outside of the observation zone (the "unknown" regions) using spatial interpolation, and then summing aklak abundance inside and outside the observation zone to produce a total abundance estimate within the study area.

TABLE 1. PROCESS FOR ASCRIBING PROXY VALUES TO AKLAK SIGHTINGS WITHOUT A DISCRETE NUMBER NOTED.

Key Words	Proxy	Number Used	Justification
grizzly with cubs; family; bear with cubs	Yes	3	Cub of the year litter size average: 2.23. Yearling litter size average: 1.86. Two-year-old litter size average: 1.85. Average family grouping will thus contain 3 bears (1 mother, 2 cubs). (McLoughlin and Messier, 2001).
lots of; more; grizzly area; grizzlies like it here; core area; main area	Yes	3	More than one sighting, but no quantity given. No mention of cubs or family. Presumably sightings were made separately, but interviewee includes a descriptor that suggests a number higher than two.
cubs	Yes	2	Cub of the year litter size average: 2.23. Yearling litter size average: 1.86. Two-year-old litter size average: 1.85. (McLoughlin and Messier, 2001).
grizzlies; sightings	Yes	2	More than one sighting, but no quantity given. No mention of cubs or family. Presumably sightings were made separately.
den	Yes	1	Grizzlies may dig multiple dens but only use one, so there is a chance that not all dens seen indicate a bear. In any given year a female of cub-bearing age (typically 5-8 years of age) will have an average of 1.5 cubs with her. Thus, the average number of bears per den can be rounded down to one (McLoughlin and Messier, 2001).
digging; tracks	Yes	1	Signs that clearly indicated grizzly presence were counted as sightings.
less; fewer; don't see	No	0	When interviewees indicated a lack of sightings, no sightings were counted.
cabins; smoke	No	0	Some datapoints referred to things other than grizzly signs or sightings.

Refining input data for spatial interpolation

The application of spatial interpolation strategies is limited by the format of the input data. To maximize our ability to apply various spatial interpolation strategies, we set some parameters on the format of our input data. Due to the nature of the interview approach and the scale of the maps used, the spatial

precision of participant observations was limited. We created grid using the "Create Fishnet" tool in ArcGIS Pro that spanned the study area, aggregated the sightings within each grid cell, and then used grid centroids containing total number of aklak sightings within each 100 km ² cell (10km by 10km) as the input dataset for the interpolation process (Figure 5). Values were only ascribed to those grid cells that overlapped with the observation zone; cells within the observation zone that did not contain sightings made by interviewees were ascribed a value of zero. Cells that fell outside of the observation zone were only ascribed a value if there were sightings recorded. We chose a grid cell size of 10 x 10 km for two reasons. First, because participant observations can be expected to demonstrate precision within approximately 10 km. Second, because the DNA study data, which we planned to use to complement the interview data in spatial interpolation calculations, was collected and reported on at 10 km intervals within each study block (sampling grid). This allowed us to seamlessly integrate the data collected via DNA capture into the grid within the grid cells that overlapped with the DNA study blocks. Grid cells that fell outside the interview observation zone and the DNA study blocks were not ascribed to any numeric values (null), to be populated based on the results of the spatial interpolation processes. After sighting data from interviews and DNA study data was ascribed to the relevant grid cells, the centroid of each grid cell with an ascribed value was exported into a point feature class to be used as input data for the spatial interpolation.

Exploring weighting variables

Before applying spatial interpolation, we explored whether there were potential variables that could be used to increase the accuracy of the abundance estimation by weighting the spatial interpolation. In their study examining grizzly bear habitat factors in the northwestern United States, Sells et al. (2023) used a normalized difference vegetation index (NDVI) to indicate abundance of food, while Chetkiewicz and Boyce (2009) found that greenness positively correlates with grizzly population in Canadian Rocky Mountains. NDVI is a metric used to quantify vegetation health and density using aerial imaging and sensor data. We determined that it would be worthwhile to explore whether a correlation exists in Nunavut between NDVI and aklak distribution by conducting geostatistical analyses using our various data sources on aklak presence. We decided to test for correlation between aklak sightings and NDVI within the observation zone, under the premise that if a correlation existed within the "known" regions of the study area, we could use NDVI data in the "unknown" to predict abundance outside of the observation zone more accurately.

We collected NDVI data from two sources: one raster dataset containing average NDVI values over the months of June, July, and August from 2016–2020 from Google Earth Engine (Earth Engine Data Catalog, 2024), and another containing trends in NDVI values between 1984–2012 (where significant negative values represent browning and significant positive values represent greening) from NASA's Distributed Active Archive Center for Biogeochemical Dynamics (NASA Earth Data, 2019). Both data products had a resolution of 30 m and were collected by different generations of the LANDSAT satellites.

The NDVI raster files were added to ArcGIS Pro, and the datasets were cleaned by removing outlier values. The pixel values were aggregated to the 10 x 10 km grid cells by calculating the mean value within each cell to match the degree of precision of our data on aklak sighting. Three statistical tests were used to determine whether there was a correlation between NDVI and aklak abundance: local bivariate analysis in ArcGIS Pro, and Spearman and Pearson correlation tests in SPSS. All tests were run multiple times to assess the correlation between both NDVI datasets with all three data sources

(interviews, DNA capture, and harvest), within their respective observation zones. We reran the local bivariate analysis test with a wide range of neighbour parameters.

There were no significant correlations between NDVI and any of the datasets, or between NDVI change and any of the datasets. Therefore, we did not take NDVI forward as a weighting variable for spatial interpolation.

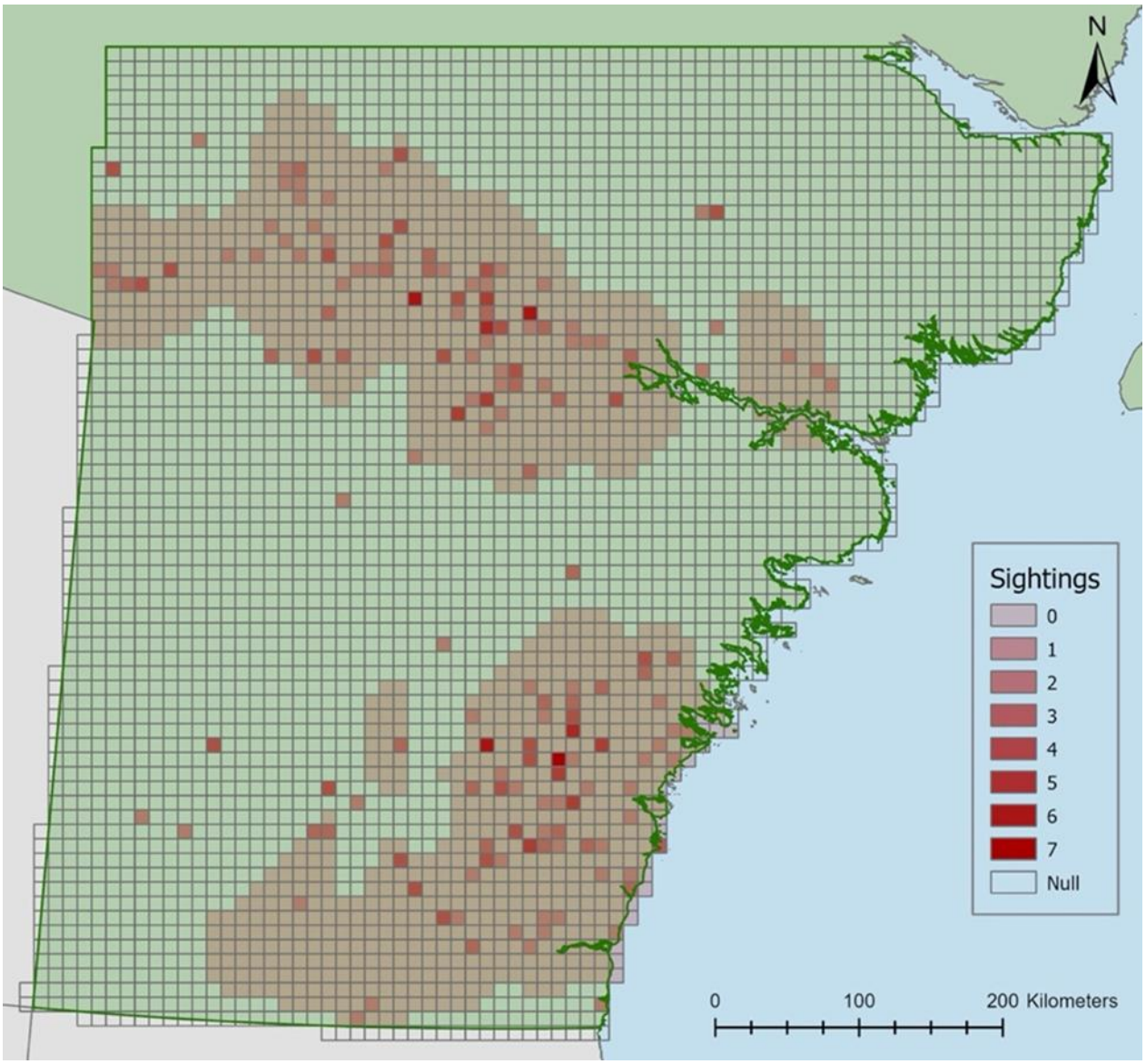


FIGURE 5. 10X10 KM GRID OVERLAID ONTO THE STUDY AREA, SHOWING THE GRID CELLS THAT INTERSECTED WITH THE OBSERVATION ZONE IN RED, WITH GRID CELLS CONTAINING OBSERVATIONS IN DARKER VALUES PROPORTIONAL TO THE NUMBER OF SIGHTINGS IN THAT AREA.

Interpolation strategies and total abundance calculation

Two interpolation strategies were applied: Inverse Distance Weighted (IDW) interpolation and Empirical Bayesian Kriging (EBK). Both interpolation processes function on the fundamental geographic premise that "everything is related to everything else, but near things are more related than distant things" (sometimes known as "Tobler's First Law of Geography") (Tobler 1970, 236). Both interpolation strategies were applied using the interview dataset alone (signs and sightings of aklak after 1990) as the input data, as well as using the interview dataset combined with the DNA study capture locations as input data. As discussed previously, the DNA capture locations offered spatial complementarity to the interview and harvest observation zones, allowing for a more spatially robust input dataset. Table 2 shows the parameters set for each interpolation method. For EBK, additional model parameters limited the maximum number of points in each local model to 100, the local model area overlap factor to 1, and the number of simulated semivariograms to 100. All interpolation calculations were done within the study area boundaries.

TABLE 2. PARAMETERS USED FOR INVERSE DISTANCE WEIGHTING AND EMPIRICAL BAYESIAN KRIGING INTERPOLATION PROCESSES.

Interpolation Method	Power	Search Neighbourhood	Max Neighbours	Min Neighbours
IDW	2	Standard	15	10
EBK	N/A	Standard circular	15	10

Both processes output continuous (floating point) raster files, with a single non-integer value ascribed to each cell. Based on our input dataset, the value ascribed to a given cell essentially represents the likelihood of aklak being observed in a discrete location at a given time. Because our input values were either zeros or positive integers, the majority of cell values in the output rasters ranged from zero to one. Total abundance was calculated by summing all cell values across the study area.

Results

Participant observation of changes in aklak abundance and distribution

This section summarizes the key themes reported by participants, including factors influencing aklak movement patterns and perceived population growth. The results are organized by descriptive codes that emerged from thematic analysis.

Relative abundance of aklak has rapidly increased since at least the 1990s:

Both Baker Lake (Qamani'tuaq) and Arviat participants attested to increases in human encounters with aklak across the Kivalliq region. Aklak are a known species within Inuit Qaujimajatuqangit, but prior to the mid-20th century encounters with aklak were rare. Aklak are viewed as increasing in abundance and distribution across the Kivalliq since at least the 1990s and perhaps as early as the 1960s. While one participant suggested that hunting may be limiting further population growth, all other translated interviews (n=18) indicated that the population is increasing rather than stable.

"There were not much grizzlies before the 1990s that I know of. I started catching grizzlies around the year 2000." (Ken Avaala, Baker Lake 2022)

Aklak prefer inland habitats, particularly around lakes, rivers, wetlands, and eskers:

While aklak are found everywhere in the Kivalliq mainland, encounters between humans and aklak tend to occur along rivers and lakeshores inland. In contrast to data the study authors collected in Northern Manitoba (Clark et al. 2022), participants in the Kivalliq all indicated that aklak were more likely to be found inland, but that there were more found near the coast than in the past.

"*I think the coast is where is where you would rarely see grizzly bears, along the whole coast.*" (Joe Savikataaq, Arviat, 2022)

"I think there's grizzlies all over the tundra. They're hardly close to the Hudson's Bay, they're mostly inland." (Ken Avaala, Baker Lake, 2022)

"They follow the rivers more for the fish." (Ken Avaala, Baker Lake, 2022)

Climate change is expanding aklak habitat northward:

Study participants perceive the rise in aklak populations in the Kivalliq as driven by climate change, rather than a sign of natural abundance. For example, one participant described changes in habitat and vegetation as bringing aklak further north:

"It is probably from what they had eaten. I think that their vegetations are starting to be different compared to in the past. It is from global warming that their behaviour is changing and where there vegetations are changing and it's affecting them from the changes and the growing population of grizzly bears are improving." (Norman Attungala, Baker Lake 2003)

"We're beginning to see some animals and birds that we were unheard of in earlier days. Just I think that probably has a lot to do with what is known as global warming you know, you know, you know as it warms up. It's not just the grizzlies that are beginning to move up here. So, I don't know, I guess it's hot. I believe that has a lot to do with it. They're just moving up and realizing that they can survive up here. But like I said, it's just not the grizzlies increasing, everything is increasing all your birds, mammals, even moose, we're even beginning to see moose. So that's probably it, they're just moving up, and probably will continue to do so." (Hugh Ikoe, Baker Lake, 2023)

Habitat destruction is driving wildlife north:

Participants in Arviat described wildfires in regions south of Nunavut as causing aklak to migrate into the Kivalliq. They described aklak as disliking smoke, and as moving in Nunavut to avoid the smoke from wildfires in Saskatchewan, Manitoba, and the Northwest Territories.

"*They are all over the world, they are in the world, in land, everywhere, but in white man's land, but the trees are on fire, right now, they have really increased.*" (Paul Kablutsiak, Arviat, 2022)

Participant observations of aklak ecology

Participant observations highlight aklak adaptability, intelligence, and ecological roles, offering valuable insights into their diet, habitat, and interactions with humans in the Kivalliq region.

Defending scavenged food:

Inuit caches are well-designed to keep out common scavengers like the Arctic fox, but participants noted that the traditional way of building a cache would not keep out aklak due to its size, strength and digging ability. When aklak dig up a cache, they will stay nearby and defend it:

"*they don't finish it in one day, they make a bed, look like this, I seen one, by the willow, they sleep there and the next day they start eating again."* (Alex Iglookyouak, Baker Lake, 2003)

Hiding from humans:

Participants described aklak as tricky, and as having the ability to hide themselves well despite their large size. Participants in Arviat spoke of how aklak were known to make a whistling noise, while Baker Lake participants did not have this attribute included in their traditional knowledge. Participants from both communities emphasized that traditional knowledge warned Inuit to be alert and wary if they suspect an aklak was around, as it could hide from humans. One participant described that in his own experience and from knowledge passed on to him by elders, he knew that aklak:

"*…pretend to run away and they look at you from someplace else.*" (Tom Mannik, Baker Lake, 2003)

Known to scavenge carcasses as well as hunt:

One participant observed aklak following wolf tracks and stated that the aklak intended to steal carcasses from wolves. See also Gunther and Smith (2004) for similar observations in Yellowstone National Park.

Known to den from November to April/May:

Females with cubs were observed to leave their dens later than males. One report of a winter-active aklak (February) from Baker Lake (focus group 2023) that was caught by a hunter and was very thin.

Regular diet includes bird eggs, geese, fish, caribou, musk-ox, and ground squirrel:

One participant observed an aklak attempting to drown a muskox. Muskox were also observed fleeing from aklak.

"*The muskox did not form a defensive circle, as they usually do, but ran and ran and ran.*" (Joe Savikataaq, Arviat, 2022)

Diggings for sik-sik (Arctic ground squirrel) were commonly observed.

"*Seems like loader had been there, part of it, maybe after ground squirrels*." (Ryan St. John, Arviat, 2022)

Observed feeding on vegetation:

"*They also like to be in the damp hummock in the land I guess they like plants that are moist. We mostly see them in the flatlands around hummock area."* (Norman Attungala, Baker Lake, 2003)

"*I shot a grizzly bear and there was nothing but berries inside its stomach.*" (Tony Uluadluak, Arviat, 2022)

Sexual dimorphism:

Females were known to be smaller than males, and one participant noted that the hind leg tracks of females were more triangular in shape.

"*You can tell by its size, their body, or by claws, yes, and by the face.*" (Daniel Kablutsiak, Arviat, 2022)

Observations of bears congregating in groups:

One participant described encountering five adult bears at one time, which may indicate the presence of an abundant and attractive food source.

Behavioural plasticity, intelligence, individual agency:

"All grizzly bears are different. Some will come attack a person, a human being, or some other grizzly bears run away." (Emily Alerk, Baker Lake, 2003)

"The way of living in their habitat is similar to us humans. We hunt where we can find food and we go to the hunting area and start having cabins there because we will know we will get what we want or because of way the land makes us feel or go to the place that we used to live, it is same as the animals they know where to get food, where to stay from their comfort zone. It is parallel living where an animal stays where they can find food just like humans do. They do not talk or communicate like humans do but they know what they eat, they know where to get food, they know what to do to get food and they know how the weather will be." (Norman Attungala, Baker Lake, 2003)

Maternal investment:

Participants noted that female aklak usually bore two cubs, with three being rare. It was described how aklak, like caribou, did not have cubs every season but waited at least two years between litters.

Avoiding conflict with humans:

"*Like grizzly, if it is not grizzly, it will flee, if it is a fox, it sees, that it is not a fox, it will flee. Animals were like that.*" (Dorothy Aglukark, Arviat, 2022)

"*The woman was going to face a grizzly, head on. She is petrified, the grizzly bear will have no match along with her carried child. Using Inuit Knowledge, she was told not to move and do not speak, when we see a large animal. I know this for a fact, because I have been told about this. She remembers this. "I will just stop, stop walking, just being still." She was being quiet along with her child, making sure she does not speak. Grizzly bear, there, was just coming. She was just like that, it just crossed by. She survived. She did not move even though the grizzly bear crossed very close by*." (Dorothy Aglukark, Arviat, 2022)

Inuit taxonomy based on colour and behaviour:

Different types of aklak were described by participating Elders.

"*He says there are three types of grizzlies. There [are] the brown ones. And the one that looked like ground squirrels, and the dark ones. The ones that look like ground squirrels, they're the mean ones, the vicious ones.*" (Arviat focus group 2023)

This taxonomic distinction differs from the Linnaean system used by biologists but as Polfus et al. (2016) found for caribou, Indigenous taxonomies can encompass features identifiable through focused scientific investigation. Phenotypic plasticity characterizes *Ursus arctos* across its circumpolar range and the history of taxonomic classifications of this species is a dynamic one that is not yet settled (Kitchener 2010). We therefore recommend that any future genomic research on grizzlies in the Kivalliq specifically look for population substructure. Moreover, knowledge about behavioural variation, whatever its potential relationship with physiognomy, is important for understanding and reducing human-wildlife conflict.

Participant attitudes towards aklak

The increasing abundance of aklak in the Kivalliq poses challenges to intangible cultural heritage, particularly Inuit practices surrounding hunting, harvesting, and caching caribou (Bortolotto 2007; Lenzerini 2011; UNESCO 2003). In February 2023, Hugh Ikoe of Baker Lake described how his family would make caches of caribou meat every season. He described how, in the past, wolves, foxes and wolverines might destroy up to a few of his family's caches, but the last time his family cached caribou meat, every single cache was destroyed by aklak. He stressed that this was a loss of approximately 15 caches full of caribou meat, which was significant to his community both financially—due to the time, fuel and labour invested—and culturally, as cached caribou is viewed as the most culturally important food for inland Inuit groups.

"When a grizzly bear finds a piruyaq (caribou meat cache) it thinks that it belongs to the grizzly bear. The bear can become aggressive to a person who went to go get its catch and starts to attack the person. What I heard and seen before is when a grizzly bear finds a caribou caches it will think that the food it his and will protect what it is find from any animal or from any human." (Norman Attungala, Baker Lake, 2003)

"The only reason why we survived up here you know, far in native country, is that it's cool enough that we can cache it [meat]. That was the only way to survive, bury it with rocks. And it will not go bad. You have to cache as much are you can, because if you didn't you're facing certain starvation, which did happen in 1958. Lot of people starved to death." (Hugh Ikoe, Baker Lake, 2023)

"So that [caching meat] was very much part of our culture. You know, we have to cache but today we can't. As you've probably heard, we can't do it anymore because of the grizzlies. This caching was such a part of our culture because it was like sausage, you know, like Italian or Ukrainian, everybody makes a sausage and everybody loved it, whether you are polish or German everybody made their own sausage and that. To us, it was the same thing. [Word for cached caribou in Inukitut] had a certain taste of its own. So basically, I called it our sausage. It was very much part of our culture. We would like to keep it but now you can't have cache anymore." (Hugh Ikoe, Baker Lake, 2023)

Participants also noted that it was more dangerous to camp in tents out on the land, and that many people were now scared to camp without a hard-sided cabin. Concerns about keeping young children and family members safe while camping out on the land were expressed in both communities. Some participants stated they no longer felt safe camping in inland areas once considered safe due to the absence of polar bears, as the increasing presence of aklak has changed this.

The harvest of aklak is viewed as sustainable by study participants as the population is generally viewed to be increasing. Quotas and restrictions on aklak harvest are not pragmatic or desirable for community members. Hunters who harvested intentionally rather than opportunistically did so in the spring, when aklak had just emerged from their dens. While aklak meat and pelts would be used when harvested, aklak were not a preferred food source. The harvest of aklak was viewed similarly to pest control. While some community members would hunt aklak in the spring, most aklak were harvested after a conflict with human. Participants characterized aklak negatively, using terms such as aggressive, dangerous, and nuisance as descriptors.

Geography of aklak observations

- **Higher observed density inland than in coastal areas.** This contrasts our data from northern Manitoba (Clark et al. 2022), suggesting either an observation bias in that data or different habitat preferences. Behavioural differences may also play a role, as all confirmed grizzly bear observations in northern Manitoba are of lone adult bears, often males, indicating a dispersing population at the edge of its range (Clark et al. 2022), rather than an established breeding population, as in the Kivalliq. Further investigation into this difference would be worthwhile.

- **Concentration of encounters around river corridors, lakeshores, and wetlands.** Participatory mapping exercises revealed most encounters with aklak occurred near lakes and rivers. While this may be due to the use of bodies of water as landmarks, it also was noted by participants that aklak preferred vegetation in wetter areas and with trees over the open tundra**.**

- **No differences noted in sex distribution of aklak.** Participants communicated no other observations about sex-specific space use differences and answered negatively when asked if they knew of any differences in distribution between sexes.

Quantitative assessment of aklak abundance

Cleaning and processing data

For those datapoints with temporal data included, observations made by participants spanned from the 1950s to the present. Total aklak sightings by interviewees since 1990 was 296. Figure 6 shows the distribution of sightings made by interviewees categorized by time period (pre-1990s, during the 1990s, and after 2000). Aklak sighting density within the 122,900km^2 observation zone (based on interview data alone) was 2.41 bears/1000km^2. While the observation zone does not overlap with the study area used in the DNA capture study (Awan et al. 2019), we compared this result to the DNA study findings, which estimated a population density of 3.5/1000km^2 within a 95% confidence interval of 2.1–6.1 bears/1000km^2.

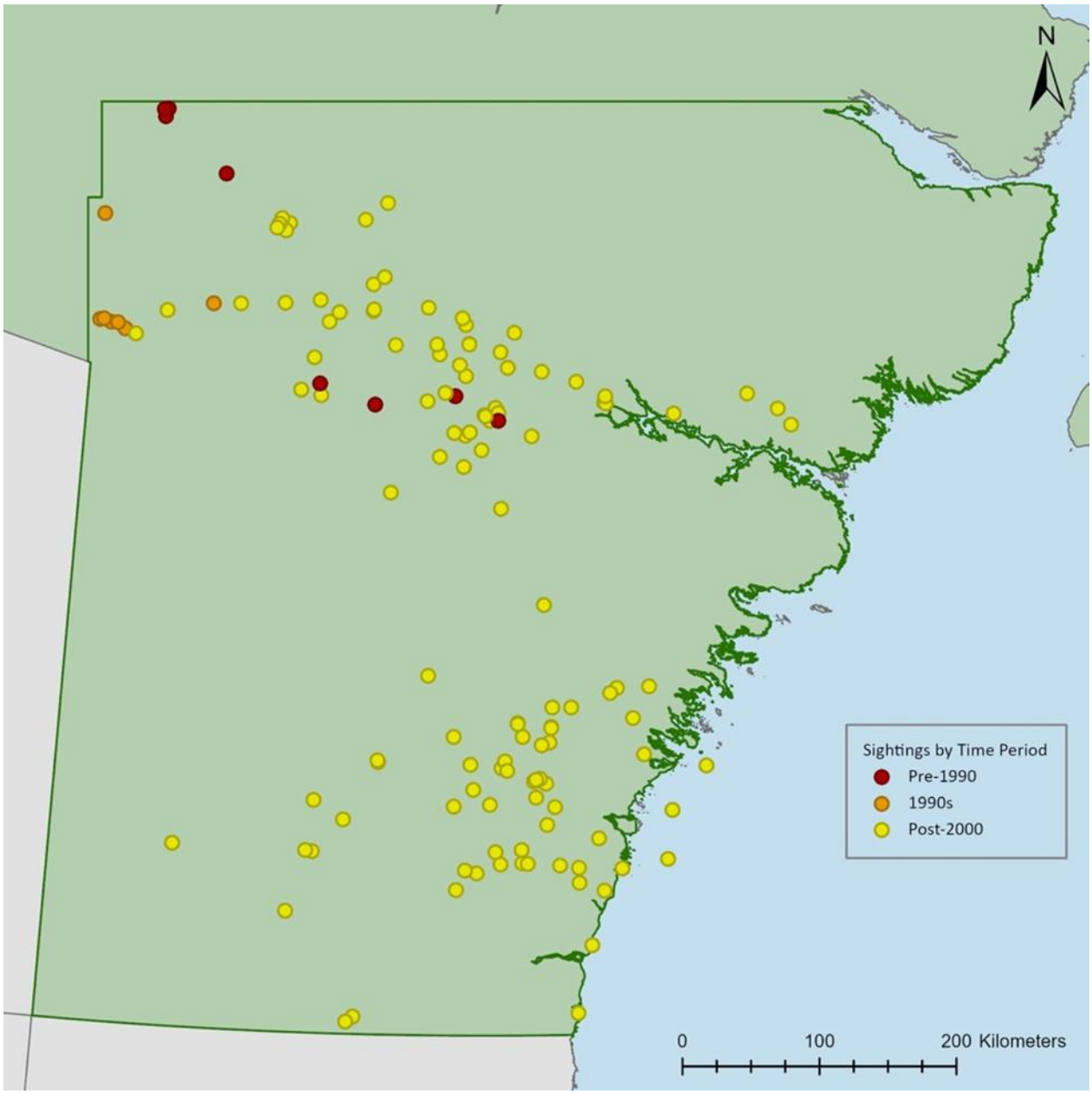


FIGURE 6. SIGHTINGS REPORTED BY PARTICIPANTS CATEGORIZED ACCORDING TO TIME PERIOD.

Spatial interpolation and total abundance

Figures 7 and 8 show the raster outputs for the IDW and EBK interpolation processes, respectively. The results of the interpolation using only the interview dataset as input are displayed in map A (on the left), while the results using the interview and DNA recapture study datasets as input are displayed in map B (on the right). Table 3 shows a selection of summary statistics from each of the four output rasters. Table 4 summarizes the abundance estimation results from the two spatial interpolation processes, including a total abundance estimate within the 371,600km^2 study area, and a density estimate of aklak per square kilometer within the study area. 95% confidence intervals were calculated for density and

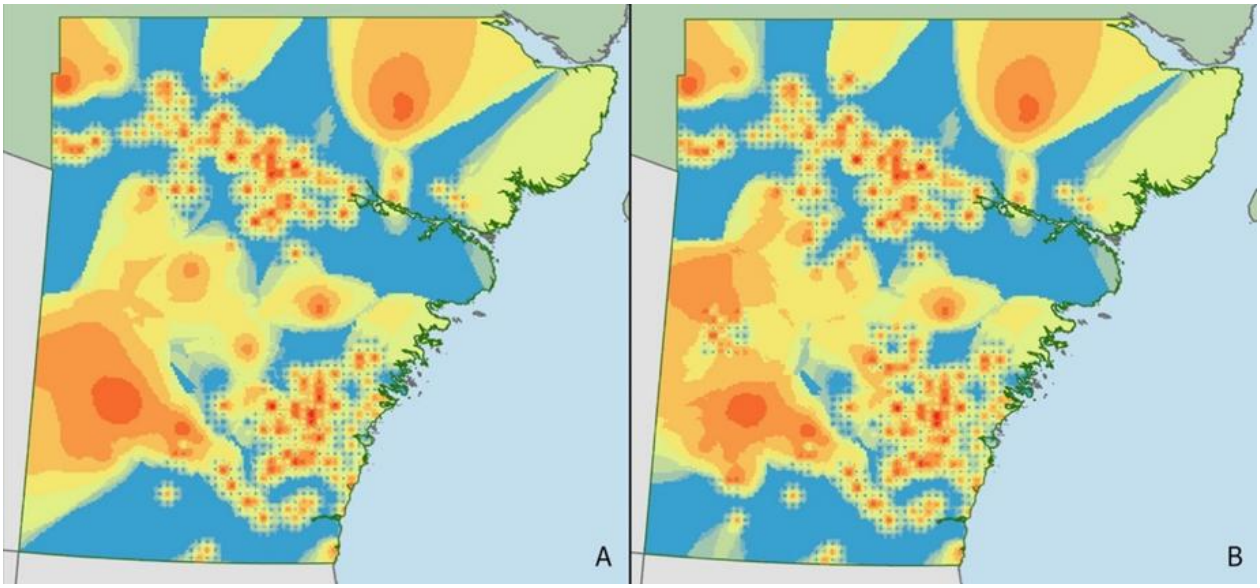


FIGURE 7. RESULTS OF INVERSE DISTANCE WEIGHTING SPATIAL INTERPOLATION, USING INTERVIEW DATA ONLY AS INPUT (MAP A) AND INTERVIEWED DATA COMBINED WITH DNA MARK RECAPTURE STUDY DATA (AWAN ET AL. 2019) AS INPUT (MAP B).

abundance using SPSS and are also shown in those tables.

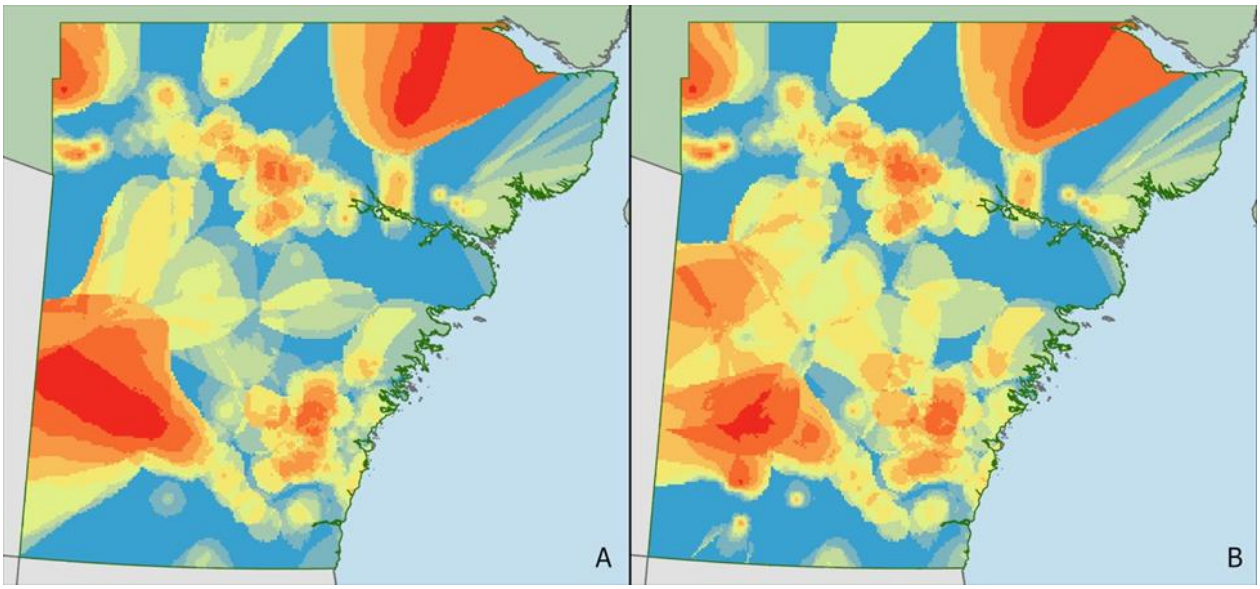


FIGURE 8. RESULTS OF EMPIRICAL BAYESIAN KRIGING SPATIAL INTERPOLATION USING INTERVIEW DATA ONLY AS INPUT (MAP A) AND INTERVIEW DATA COMBINED WITH DNA MARK RECAPTURE STUDY DATA (AWAN ET AL. 2019) AS INPUT (MAP B).

TABLE 3. SUMMARY DENSITY (BEARS/1000KM^2) STATISTICS FROM EACH OF THE FOUR OUTPUT RASTER DATASETS.

		Inverse Distance Weighting		*Empirical Bayesian Kriging*	
Input Data		*Interview*	*Interview and DNA*	*Interview*	*Interview and DNA*
Maximum		*3.93*	*3.93*	*2.58*	*2.21*
Minimum		*0*	*0*	*-0.05*	*-0.04*
Mean		*0.29*	*0.29*	*0.41*	*0.36*
Mean 95% CI	***Lower***	*0.28*	*0.28*	*0.39*	*0.35*
	Upper	*0.30*	*0.30*	*0.43*	*0.38*
Median		*0.14*	*0.14*	*0.18*	*0.21*
Standard Deviation		*0.41*	*0.39*	*0.54*	*0.43*

TABLE 4. ESTIMATED ABUNDANCE OF AKLAK WITHIN THE STUDY AREA USING INVERSE DISTANCE WEIGHTING AND EMPIRICAL BAYESIAN KRIGING.

		Inverse Distance Weighting		**Empirical Bayesian Kriging**	
Input Data		*Interview*	*Interview and DNA*	*Interview*	*Interview and DNA*
Total		1081	1073	1515	1351
95% CI	**Lower**	1032	1027	1450	1300
	Upper	1130	1120	1580	1404
Aklak per 1000 km^2		2.91	2.88	4.07	3.64
95% CI	**Lower**	2.77	2.76	3.90	3.50
	Upper	3.04	3.01	4.25	3.78

Conclusions

Aklak distribution, abundance, and implications

Mapped observations precisely indicate the southward spread of aklak observations since the 1990s, with the earliest documented observations from the region of the Thelon Game Sanctuary (Figure 6). Inuit Qaujimajatuqangit strongly suggests that aklak are undergoing a population increase and subsequent range expansion across the Kivalliq region that began decades ago. This situation is distinctive from other regions in Canada where grizzly bear populations are increasingly under threat from habitat destruction or are in recovery from a population decline (McLoughlin and Stenhouse 2021). Study participants are aware of this difference and recommend that management strategies should account for the unique social and ecological contexts that aklak experience in the Kivalliq region.

Quantitative estimation of aklak abundance

The estimated aklak densities from both interview data (2.91-4.07 bears/1000km^2) and interview data plus DNA grid hits (2.88-3.64 bears/1000km^2) are all within the 95% CI range estimated for our study area by Awan et al. (2019) of 2.1-6.1 bears/1000km^2. Our estimates are also consistent with results from previous studies in similar habitats: 3.5 bears/1000km^2 in the 1990s from the nearest other grizzly bear study area in Nunavut/NWT, which adjoins the Kivalliq (McLoughlin and Messier 2001); 5.2 – 6.7 bears/1000km^2 from the same area in 2012-2014 (Barrueto et al. 2023); and 5 to 6.6 bears/1000km^2 from the Kitikmeot region, Nunavut (Awan et al. 2023; Dumond et al. 2015). Further, they are even consistent with estimates at the lower range of the large-scale habitat- and mortality-based density projections by Mowat et al. (2013) and 3.9 bears/1000km^2 extrapolated from abundance estimates for the Arctic Coastal Plains grizzly bear zone by Banci et al. (1994). Such similarities with these geographically broader estimates are not unexpected, assuming that the habitat productivity/home range size relationships for grizzlies described elsewhere hold here too (McLoughlin et al. 2003).

While these similarities are promising it would be premature to consider it definitive validation of our study (or those other ones), for several reasons. First, our estimates are based on cumulative observations over 33 years, so it is not yet clear how comparable they are to those generated from short-duration mark-recapture studies. Nonetheless, the lack of change over two decades between the estimates in McLoughlin and Messier's original study area, despite industrial development there, suggests such comparisons may not be *a priori* invalid. Second, Awan et al. (2019) provided only provisional estimates, recommending a much larger sampling effort for greater accuracy and reliability. Third, recalling that our initial plan for abundance estimation was to use proportional piling exercises, we lack a theoretical framework for explaining how these observational data actually indicate abundance. Besides further methodological development (discussed below), it will be necessary to build such a framework, which should itself be done in a participatory way. Participants' observations about the limited applicability of proportional piling for solitary animals shows both the need and the value of further Inuit involvement in such research.

Aklak ecology

Participants provided detailed observations of aklak food habits and interspecific interactions, habitat selection, litter size and timing, and denning. Unsurprisingly, Inuit observations correspond with findings

from scientific studies conducted in similar Arctic habitats (e.g. Arthur et al. 2017; Edwards et al. 2009, 2011; Gau et al. 2002; McLoughlin et al. 1999, 2002; Reynolds et al. 1987; Young and McCabe 1998).

We were surprised that neither NDVI nor NDVI change explained the spatial distribution of aklak observations given its strong predictive value for grizzly bears elsewhere (e.g. Apps et al. 2016; Mowat et al. 2013). While it's possible that NDVI might have stronger explanatory power at larger spatial scales, it's also possible that the species' range expansion in the region may not necessarily be driven by vegetation productivity, or perhaps not directly. Since Gau et al. (2002) found grizzlies adjacent to our area were predominantly carnivorous (on caribou), such a habit could explain the lack of dependence on vegetation productivity as measured by NDVI. Petzold and Goward (1988) demonstrated that NDVI doesn't accurately measure lichen productivity, indicating that if aklak distribution was governed by caribou – which are dependent on lichens – NDVI measures could well be insufficient to detect it. Consequently, more precise insights into what aklak eat in the Kivalliq, and whether their diet is comparable to other parts of the species' Arctic range, would be helpful.

Recommendations for further investigation

An important area for further inquiry is the investigation of solutions to the problem of caribou cache destruction by aklak. As noted above, aklak were viewed as the primary reason that Inuit were no longer able to cache caribou meat, a practice which is viewed by community members as vital to Inuit culture. If a way could be found to deter aklak from digging up caribou caches this could allow the practice of caching meat to continue and support Inuit cultural continuity. Participants also expressed a desire to deter aklak from entering their camps and cabins, particularly at night. Deterring bears from damaging property is a primary issue in reducing human-bear conflict across species and ecosystems. Further investigation into what deterrence methods are feasible, affordable, accessible, and efficient in remote communities in the Kivalliq region would be welcomed by communities.

Improving methods for estimating density and abundance from participatory mapping would be benefit aklak co-management and help bridge traditional and local knowledge with broader wildlife management goals. As Mowat et al. (2013) noted, conducting detailed abundance estimations is not feasible for most regions due to high costs and competing priorities. Since the cost of this project was considerably less than a mark-recapture field study, the potential benefits of fully and appropriately utilizing this kind of data are substantial. To be clear, this line of methodological inquiry is not a question of scientifically judging the reliability of IQ about aklak, which would be epistemologically inappropriate even if it was possible (Brook and McLachlan 2005). Rather, what's required is understanding of how comparable this observation-based approach to grizzly bear density and abundance estimation is to other established methods which use different information inputs. For example, many observations mapped by our participants were associated with lakes and rivers. Awan et al. (2019) sampled lakeshores specifically (as well as grids) where they detected a high number of individuals. They were concerned this could lead to erroneous density/abundance estimation, yet that doesn't appear to be the case with our data despite the same spatial associations.

Implications for co-management and planning

Unique situation of "threats" in the Kivalliq

Study results suggest that the positioning of aklak in the socio-ecological context of the Kivalliq region is not analogous to other regions in Canada as the threat they pose to Inuit food systems is the paramount concern of communities facing human-aklak conflict and the abundance and distribution of aklak is increasing. Aklak are perceived as a threat to the existing socio-ecological system, rather than as a species under threat and in need of protection.

Need for extended deterrence strategies

While harvest was presented as a primary means for reducing conflict with aklak, strategies for reducing encounters through deterrence and preventing habituation were also suggested by participants as something they'd like to know more about. Participants expressed a desire for strategies to deter aklak from damaging property and for more community education about the importance of proper food storage and garbage disposal. Participants were resigned to the increased presence of aklak on their territories, stating "*they're here to stay"* (Hugh Ikoe, Baker Lake, 2023). Participants expected aklak to continue to be a problem for the foreseeable future.

Participants do not desire a quota system and are skeptical of Canadian science-based wildlife management

Participants were uniformly adamant that a hunting quota system should not be reinstated for grizzly bears, and that Inuit should have the ability to judge for themselves the best course of action according to the conditions of a particular encounter. As noted previously, participants described aklak as varying in behaviour and temperament, which means that it was highly dependent on the individual bear and the specific context of the encounter whether lethal action was deemed necessary or not. Participants emphasized aklak intelligence and agency, and how during an encounter aklak also assessed the situation and decided on a course of action. It was emphasized that generalized rules and regulations such as quota systems are not responsive to the unique conditions of each aklak-human encounter. As stated by one participant commenting on the previous quota system: "*when a grizzly bear is going to be dangerous it will not think of the legislation"* (Simeon Mikkingwak, Baker Lake, 2003).

Participants in Clark's 2003 interviews indicated they believed the quota system was instated to protect bears due to concerns of low bear populations. Even though there is no longer a quota system, our data from 2022 suggests that community members continue to perceive the goal of aklak management to be an aklak population increase. For example, participants stated that in the past the government was "*trying to get more grizzlies*" (Avaala 2022). This was consistent with data from the 2003 interviews conducted by Dr. Douglas Clark. Participants expressed the view wildlife management regulations are designed to protect wildlife, not prevent human-wildlife conflict or protect Inuit communities and culture. Participants are aware that their cultural attitudes towards aklak differs from those of conservationists from southern Canada, where grizzly bears may be viewed with reverence (Harding 2014) or as cultural keystone species (Clark et al. 2021). It is important that non-Inuit managers and Conservation Officers are aware of the views expressed by participants that aklak are increasing partially due to protections introduced by a colonial government.

"…they are increasing in our land. When they could be caught, I am alright with it, and it they were able to catch the cubs too, I would be okay with it." (Ludovic Onerk, Arviat, 2022)

"…we hunters, inside, one hundred miles, radius, are stabilizers of grizzly bears, like, we will not make them increase." (Daniel Kablutsiak, Arviat, 2022)

"*It's right, if it is caught. There seems to be many grizzly bears, if they are caught, it is alright.*" (Dudley Copland, Arviat, 2022)

"*No! I do not want it [the lack of a harvest quota] changed. If they do, your cache, your cache had been taken again on September. They keep saying that, I want them shot down. What good is it if they try and keep them alive? What good is it?*" (Paul Kablutsiak, Arviat, 2022)

"The thing I don't like about the grizzly is that it has completely taken away a very important part of our culture. Caching caribou is our culture. And what made it that way was at some point we were told that don't kill them, don't kill the grizzly or you're gonna go to jail and some people did." (Hugh Ikoe, 2023)

Methodological recommendations for future wildlife research involving Inuit Qaujimajatuqangit

A more effective interview context for future studies involving qualitative data collection and IQ could be achieved by implementing the following approaches:

1. It is recommended that future interviews take place in a community setting other than the wildlife office. While Conservation Officers were supportive of this research, the association between the wildlife office and government oversight of hunting, particularly given the politically complex history of wildlife regulation in Nunavut, may have influenced some participants' willingness to engage fully.

2. Interview questions should be translated into Inuktitut in consultation with speakers of multiple dialects before interviews take place, pre-emptively addressing issues of dialect difference and translation that came up during the interview process.

3. Community validation would ideally take place over several days, to accommodate changing local conditions (adverse weather, overlapping community events) that affect participation.

4. Simultaneous translation was found to be more efficient than post-interview translation, with the researcher, interviewer/translator, and participant creating a dialogue discussing the meaning of participant statements as issues came up.

5. Greater involvement of community members in research design (question creation), data analysis, and the communication of results is recommended for future studies. While efforts were made in this study, the pandemic presented multiple barriers. Given the accessibility of the methods and the community's interest for intergenerational knowledge transfer, the potential for capacity building is considerable (Greene and Zawadski 2022). This would require more involvement in the research process from community educators and organizations as facilitators.

Dissemination of study results

One of the challenges of this project is the cross-cultural context of the communication of not only the study data, but also the results. We have found throughout the data collection process that topographic maps are excellent tools for eliciting qualitative data from participants. Disseminating the study results to the community, which is an imperative for ethical research practice with Indigenous communities, can be a challenge given the bilingual and cross-cultural context. Maps are a valuable cross-cultural communication tool, complementing written reports and oral presentations. The digitized maps generated from the participatory mapping exercises can be uploaded onto a digital platform, along with participant descriptions, to create a 'storymap' interface (e.g. Kopatz et al. 2021). This approach locates study results within the landscape, emphasizing place-specific context, community connections to the conservation area, and the experiential elements integral to Inuit Qaujimajatuqangit.

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Baker Lake (Qamani'tuaq):

Barney Aaruaq
David Aksawnee
Emily Alerk
Norman Attungala
John Avaala
Ken Avaala
Harold Etegoyuk
Timothy Evviuk
Alex Iglookyouak
Hugh Ikoe
Joedee Joedee
Tom Mannik
Simeon Mikkingwak
Joe Niego
David Owingayak
Philip Putumiraqtuq
Joan Scottie
Hugh Tulurialik

Arviat:

Dorothy Aglukark
Peter Alareak

Thomas Siatalaaq Alikaswa
Darryl Baker
Dudley Copland
Joe Savikataaq
Leo Ikakhik
Louis Irkok
Cedric Manik
Arden Nibgoarsi
Ludovic Onerk Issumatardjuak
Daniel Kablutsiak
Paul Kablutsiak
Joe Savikataaq
Dominic Sinnisiak
Jordan St.John
Tony Uluadluak

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Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

WOLVERINE (*Gulo gulo*) HARVEST MONITORING IN NUNAVUT

Summary report

February 2025

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The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment. Reference to any specific type of gear and/or equipment does not mean that it is endorsed by the authors or the institutions they represent.

Summary

Overall, Nunavut hunters submitted skulls and kill information from 691 harvested wolverines between 2019 and 2024 harvest years (1 July 2023 –30 June 2024 = 2024 harvest year). Most wolverines were harvested in the Kitikmeot region, in the vicinity of Kugluktuk, while relatively fewer were harvested in the eastern communities. Arviat and Baker Lake were the next highest contributors to the total harvest. Harvest data shows that 84% of wolverines were hunted with a rifle, while 16% were trapped. Sex ratio of the harvest was male biased with approximately two times as many males harvested than females. Juveniles and yearling predominated as 63% of the harvest while the other 37% were adults. The harvest was dominated by younger age classes during the early winter, while the harvest of adult individuals increased later in the winter. The spatial distribution, age, and sex structure of the harvest were typical of a healthy population. However, the relatively low number of mature females in the harvest is important to note and should be monitored closely.

ᖃᐟᐱᐣ (ᔪᓗ ᔪᓗ) ᐱᓴᐅᔪᓂᐣ ᖃᐅᔨᓴᖏᓂᖅ ᓄᓇᕗᒻᒥ

ᓇᐃᓭᖅᓯᒪᔪᑦ

ᐊᒋᖏᓪᓕᒪᖅᓴᓂᑦ, ᓄᓇᕗᒻᒥ ᐊᖓᔪᓇᓯᑎᑦ ᑐᓂᓯᓪᓕᐅᖅᓯᒪᔪᑦ ᓂᐊᕆᔨᖏᓪᒋᐅᓂᖅ ᑐᑎᖅᒋᐅᔪᒪᔪᓂᓪᓗ ᖃᐟᐃᒍᕆᔨᐃᑦ ᒥᐅᓴᓄᑦ 691–ᓂᐅ ᐊᖓᔪᖃᐃᓂᕆᓂᐅ ᐅᑭᐅᑦ 2019 ᐊᒻᒪ 2024 ᐊᑯᓂᓪᐊᓂ (1 ᔪᓕᐃ 2023 - 30 ᔫᓂ 2024 = 2024 ᐊᖓᔪᓇᓯᓇᒥ). ᐅᓄᕆᓂᖅᓴᐃᑦ ᖃᐟᐃᐅᑕᐅᔪᑦ ᖃᑎᖃᒥᐅᓂᐅ ᐊᖓᔪᐅᓕᐅᖅᓯᒪᔪᑦ, ᕆᑯᓪᓗᖅᑐᑦ ᖃᓄᓯᒋᓂᔪᓪᓚᐅᑦ, ᐅᓄᖖᑎᓂᖅᓴᐃᑦ ᐊᖓᔪᐅᓂᑯᓪᓚᐅᑎᑦ ᑲᓇᖅᑕᐅᔪᓂ. ᐊᕆᔨᖏᑕᑎᑦ ᐊᒻᒪ ᖃᒪᓂᑐᐊᖅ ᐊᖓᔪᖏᑎᑦ ᐅᓄᓭᖓᖃᑕᐅᓕᐅᖅᑐᑦ ᓇᐃᓴᖅᑕᐅᔪᓂ. ᐃᑎᖅᓯᒪᓂᖅ 84% ᖃᐟᐃᒍᕆᔨᕐᑕᐃᑦ ᕿᐱᐅᑎᒧᑦ ᐱᖁᐅᔪᐃᓂᕐᓂ, ᑕᐃᑲᑯᐊ 16% ᒥᑭᒋᐊᖅ/ᑭᖏᖃᑦᑕᓐᓄᑦ ᐱᖁᐅᓪᓗᑎᑲ. ᐊᖓᔪᐅᔪᓂ ᐊᖓᔪᓪᓗᐃᑦ ᐊᕆᓭᓪᓗᐃᓪᓗ ᓇᐃᓴᖅᑕᐅᑎᓪᓚᑦ ᐊᖓᔪᓪᓗᐃᑦ ᐅᓄᕆᓂᖅᓴᐅᓭᐅᖅᑐᑦ ᒪᕐᕈᐃᓪᓗᐊᖅᓯᒪᑲᑎᑲ. ᐃᓕᓭᔨᓴᖅᓯᒪᙵᑦᑐᑦ ᐊᒻᒪ ᑎᓪᑐᑐᐃᑦ ᐅᓄᕆᓂᖅᓴᐅᓭᐅᕐᒥᔪᑦ 63% ᐊᖓᔪᐅᔪᓭᓂᓕᓂ ᑕᐃᑲᑯᐊ ᐃᓕᓭᔨᐊᓂᑲᑦᑐᑦ 37%–ᖓᓪᓚᑎᑲ. ᐅᑭᐅᖓᓵᕆᔨᖏᑎᓪᓚᔭᒍ ᐃᓕᓭᔨᓪᓘᔨᓂᖅᓴᑦ ᐱᖁᐅᖅᑕᑦᑕᐅᓕᐅᖅᑐᑦ, ᐃᓕᓭᔨᖅᓯᒪᓂᖅᓴᓪᓚᓭᑦ ᐅᑭᐅᖓᓪᓚᓭᓪᓗᒍ ᐱᖁᐅᓪᓚᑎᑲ. ᓄᓇᓂ ᐅᖓᑕᓯᑲᑦᑕᐃᓂᖅ ᐱᖁᐅᓪᓚᑎᑲ, ᐅᑭᐅᑦ, ᐊᕆᓇᖅ/ᐊᖓᔪᓪᓚᔫᓴᒋᑎᑕ ᓇᐃᓴᐃᓵᔨᓯᑦ ᑕᒪᐃᓕᓂᓂ ᐅᓂᒃᑯᐃᓵᔨᐅᑕᐅᓕᐅᖅᑐᑦ. ᑭᓯᐊᓂᑕᐅᖅ, ᖃᑐᖅᑕᐅᑐᓂᓇᖅᑐᓂᓂᓂ

Qalvik (*Gulo gulo*) anguniaqtunun munaridjutikhaq Nunavunmi

Naittuq Naunaijauti

Tamainun, Nunavunmi anguniaqtut tujuqtut niaqunik humilu tuquhimajatigut kangiqhidjutit hamanga 691 anguniaqtaujut qalviit qitqani 2019 unalu 2024 anguniarutini ukiuni (1 Taaqhivalirvia 2023 –30 Imaruqtirvia 2024 = 2024 anguniaqvingmi ukiungani). Tamavjaita qaviit angujaujut Kitikmeot aviktungnianin, qanilruani Kugluktup, ikitut angujaujut kivataanin nunallaanin. Arviat hamanilu Qamanittuaq tugliujut amigainirit ikajuutait atautimun anguniaqtamingnun. Angunianikkut naunaijautit takunaqtut tapkua 84 pusat qalviit hiquutikkut tuquhimajut, imaalu 16 pusat naniriaqhimajut. Qanuriniitigut amigainirit angujaujut angutiluat naamavjaktuq malruiqtughugu amigainiit angutit angujauluaqtut angnalurnin. Qalviat angujaujut imaa 63 pusat anguniaqtaujut aallat 37 pusat ininirit. Angujaujut amigaitqijaujut qalviat ukiungitigut naunaijaqhimajut ukiuq atulihaaqtilugu, angujaujut ininirnit atautit amigairjuumijun nungutinnagu ukiumi. Ilaujut tunijaujut, ukiungit, qanuriniit qanuqtut anguhimajut aanniaqtuqangitut. Kihimi, ikitpalaat qaffiuniit angnaluit angujaujuni aghuungnaqtut ilihimalugit munarijaujukhallu qanitukkut.

Contrôle de la récolte de carcajou (G*ulo gulo*) au Nunavut

Résumé

En tout, les adeptes de la chasse du Nunavut ont remis les crânes et les données provenant de 691 prises de carcajous lors des années de récolte de 2019 à 2024 (à titre indicatif, l'année de récolte de 2024 s'étend du 1er juillet 2023 au 30 juin 2024). La majeure partie de la récolte a eu lieu dans la région du Kitikmeot, aux environs de Kugluktuk, alors que le nombre de carcajous récoltés dans les localités de l'est était relativement peu élevé. Arviat et Baker Lake figuraient au deuxième rang parmi les grands contributeurs à la récolte totale. Les données de récolte révèlent que 84 % des carcajous ont été chassés au fusil et 16 % ont été piégés. Le ratio mâle-femelle de la récolte était biaisé en faveur des mâles, qui en représentaient environ les deux tiers. Les jeunes carcajous et les petits d'un an constituaient la majeure partie de la récolte, représentant 63 % des prises alors que les prises adultes en représentaient 37 %. En préhiver, les carcajous récoltés provenaient surtout de la population de jeunes, et c'est aux mois de mars et d'avril qu'on a vu une croissance de la récolte d'adultes. La répartition spatiale, l'âge et la répartition par sexe de la récolte étaient caractéristiques d'une population en bonne santé. Cependant, on souligne que la récolte comportait un nombre relativement peu élevé de femelles adultes, et cette population devrait faire l'objet d'une surveillance étroite.

Table of Contents

1.0 INTRODUCTION.....1
2.0 METHODS.....3
3.0 RESULTS AND DISCUSSION.....4
4.0 ACKNOWLEDGMENTS.....12
5.0 LITERATURE CITED.....13

List of Figures

Figure 1. Reported wolverine harvest in Nunavut, from 2019 to 2024. 4

Figure 2. Distribution of reported wolverine harvest in Nunavut, from 2019 to 2024..... 5

Figure 3. Wolverine reported monthly harvest in Nunavut, from 2019 to 2024. Label indicates total number of wolverines harvested per month.. 6

Figure 4. Age and sex structure of the reported wolverine harvest in Nunavut, from 2019 to 2023.. 7

Figure 5. Monthly reported wolverine harvest per age class in Nunavut, from 2019 to 2023. 10

List of Tables

Table 1. Male to female sex ratio by age in the reported wolverine harvest in Nunavut, from 2019 to 2023. 8

Table 2. Age and sex distribution of reported wolverine harvest in Nunavut, from 2019 to 2023. 9

1.0 Introduction

For thousands of years, Inuit have relied on the harvest of wildlife for food, clothing, and trade. In Nunavut, the furbearer harvest for clothing and for income is a seasonal and traditional activity, where opportunities for other employment are scarce. Under the Nunavut Agreement, furbearer harvest rights are held by Inuit beneficiaries, non-Inuit who harvested furbearers legally in Nunavut settlement area prior to 1981, and non-Inuit whose application has been approved and recommended by local Hunters and Trappers Organizations (HTOs).

In Nunavut, wolverines (*Gulo gulo*) occupy almost all areas of the territory and are classified as both a furbearer and a big game animal under the Nunavut Agreement. They are an important furbearer in Nunavut's culture and economy and have traditionally been considered a vital resource for hunters because of the beauty and frost resistant properties of the fur, which make it unique and quite valuable (Hash 1987). Wolverine fur is highly prized in local communities as ruffs or trims on parkas (Cardinal 2004). Unlike most provinces, hunters and trappers in Nunavut do not have registered or traditionally exclusive family trap lines or hunting areas, so wolverines are generally harvested opportunistically wherever people travel (Mulders 2000) or harvested while hunting other game. Wolverine hunting in Nunavut is mainly by firearms from snow machines while in other provinces trapping is the major method (Lee 2016).

Wolverine densities are believed to be moderate in the western mainland of Nunavut but low on the Arctic islands and in the eastern mainland (Slough 2007, Species at Risk Committee 2014). Using DNA-based mark-recapture, Awan and Boulanger (2016), Awan et al. (2018) estimated density from 1.6 to 4.4 wolverines/1,000 km² in the Kivalliq region and 4.14 wolverines/1,000 km^2 in the Napaktulik Lake area in the Kitikmeot region (Awan et al. 2020). Inuit observations and recent harvest reports suggest that wolverine numbers in Nunavut are either stable or slightly increasing, and the species may be expanding its range eastward and northward (Awan et al. 2012, COSEWIC 2014). The wolverine has been assessed as a species of Special Concern in Canada

and in 2018 was listed as the same under the *Species at Risk Act* (*SARA*). The wolverine is generally described as a scavenger (Banci 1987) and an opportunistic predator throughout its range, occurs in low densities, have low birthing and recruitment rates, are sensitive to human disturbance, and require large secure areas to maintain viable populations (Magoun 1985, Mattisson et al. 2011). It is considered a wilderness species and potential indicator of ecosystem health (Carroll et al. 2001). Mulders (2000) suggests that the arctic tundra contains large undisturbed tracts of habitat that may act as reservoirs to maintain harvestable populations of wolverine in Nunavut.

Nunavut contributes substantial numbers to the national harvest even though ecological data for tundra wolverine are sparse, especially in the northeastern edge of the species distribution. Currently, there is no quantitative limit on their harvest by Inuit and there is no requirement for hunters to report their harvest. The only mechanism for tracking the harvest or pelts is the Government of Nunavut's Department of Environment (GN-ENV) fur-pricing program, which misses pelts, which are used locally or sold as raw frozen wolverine hides privately; the actual harvest is unknown. The current collection of wolverine skulls and kill information is an attempt to estimate the harvest, to monitor the age and sex of the harvest.

2.0 Methods

Each year we obtained the skull and a small piece of skin (~2.5 x 2.5 cm) with fur from hunters with the assistance of HTOs and Conservation Officers (COs). Hunters across Nunavut voluntarily reported and provided skull and skin samples from harvested wolverines via their local COs. A $50 subsidy was provided to hunters for each skull brought back to the COs office to encourage the reporting and return of samples. Recently, increased activity in the online buying and selling of wolverine skulls has led to less reporting under this program. The online skull buyers offer higher prices for skulls than our subsidy amount of $50. From January 2025 GN-ENV increased the subsidy amount to $100 per wolverine.

For each wolverine sample collected during the course of this study, COs filled out sample collection forms with information from hunters about the harvest location, date, wolverine sex, and method of harvest, and their opinion of the current abundance trend of the local wolverine population (decreasing, stable or increasing). The skinned skulls were stored frozen, and were thawed at room temperature for examination in early May each year. We extracted a lower canine and sent to Matson's Laboratory LLC (Milltown, MT, USA) for age estimation using cementum analysis. This technique is based on the cyclic nature of cementum growth in teeth forming annular patterns of different darkness depending on the season (Matson 1981).

Following Banci and Harestad (1988) and Vangen et al. (2001), individuals were then grouped into three age classes: juvenile (0-1 year, date of birth is set to March 1st), yearling (1-2 years) and adult (≥ 2 years).

3.0 Results and Discussion

A total of 691 wolverines were reported as harvested from July 2018 to June 2024, 397 from the Kitikmeot, 290 from the Kivalliq and 4 from the Qikiqtaaluk region (Igloolik and Sanirajak) (Fig. 1). According to our sample collection program, reported wolverine harvest in Nunavut has been fluctuating and the harvest report rate for this period (2019-24) appears to be lower compared to early 2000s in the Kitikmeot region (Awan et al. 2012).

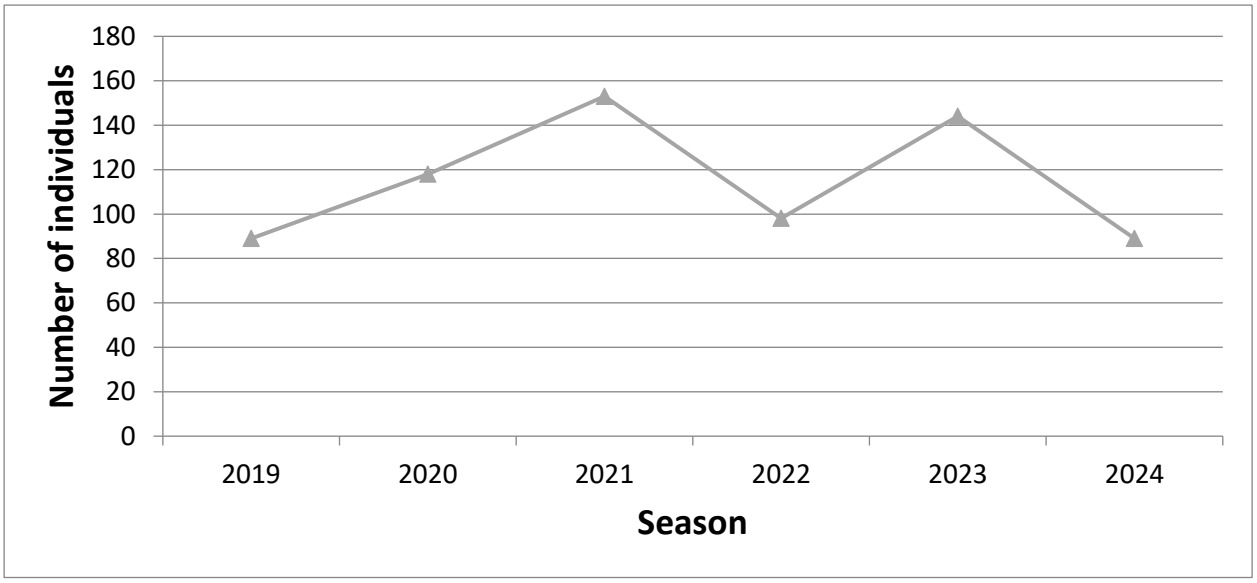


Figure 1. Reported wolverine harvest in Nunavut, from 2019 to 2024. Note that the harvest year is assigned as the year at the end of the regulations' harvesting season, for example, 1 July 2018–30 June 2019 = 2019 harvest year.

The harvest was unevenly distributed throughout the territory, with most wolverines having been harvested in the western Kitikmeot. Harvest in the vicinity of Kugluktuk was particularly intense, and along the traditional travel route from the Kugluktuk to the Contwoyto Lake. Relatively fewer wolverine were harvested in the eastern communities,

while Arviat and Baker Lake were the next major contributors to the total harvest (Fig. 2). Comparatively, high wolverine harvest happening out of Kugluktuk, when the caribou herds wintered nearby, was due to hunters spending more time on the land to hunt caribou. While hunting caribou, harvesters usually pursued wolverine when they saw wolverine or found fresh tracks.

Wolverine harvest distribution shows that harvest took place over a wider range on the Nunavut mainland, but it tended to be concentrated near communities. The surrounding areas without hunting may act as refugia or reservoirs that produce wolverines that move into the hunted areas and sustain the harvest (Mulders 2000, Lee 2016).

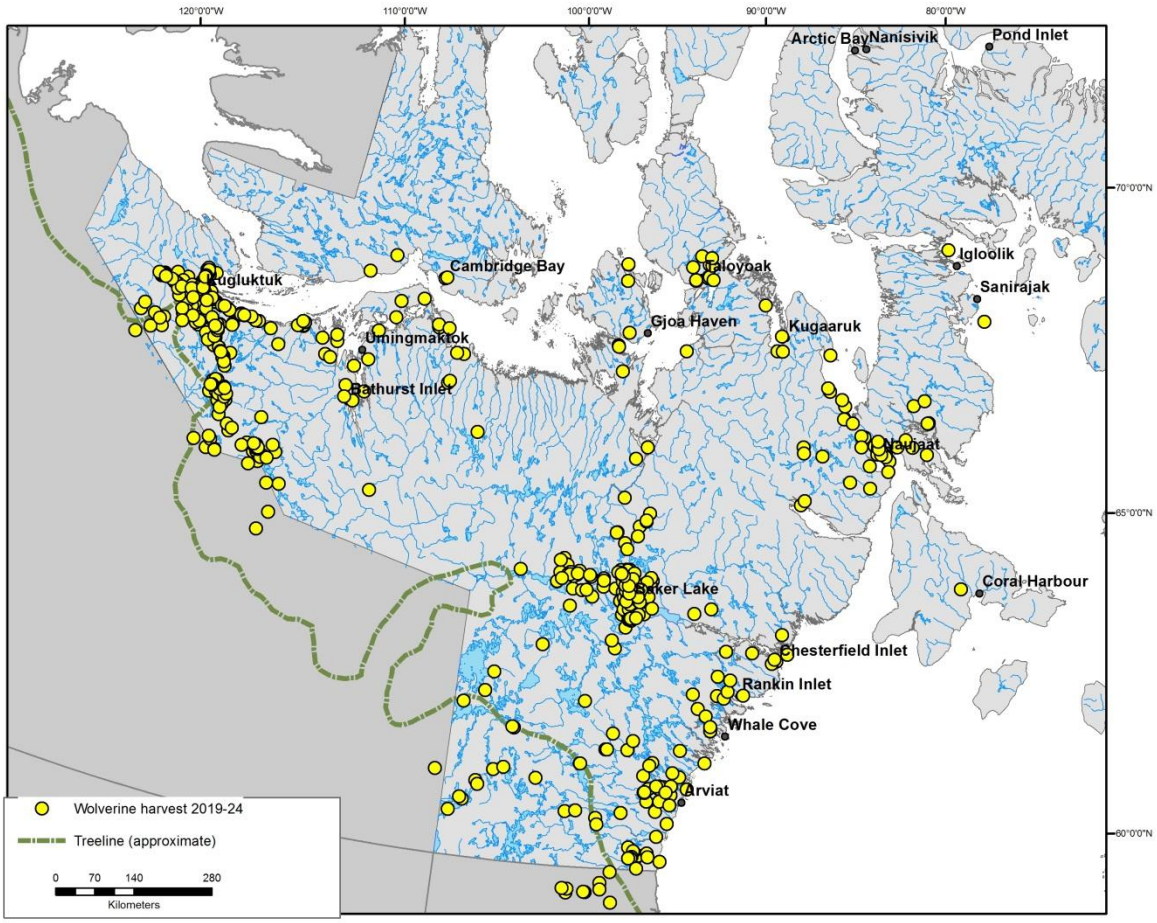


Figure 2. Distribution of reported wolverine harvest by Nunavut hunters from 2019 to 2024. Symbol may represent >1 wolverine harvested at that location.

Most wolverines were harvested between November and April each year (Fig. 3), when the fur is in prime condition. Harvest data shows that approximately 84% of wolverines were hunted with a rifle and 16% animals were trapped. Of the trapped animals (n = 109), 94% were killed in quick-kill traps and 6% in leg hold traps. In the Kivalliq region, only five hunters trapped 12 wolverines, while in the Kitikmeot region 97 wolverines were trapped by 15 hunters.

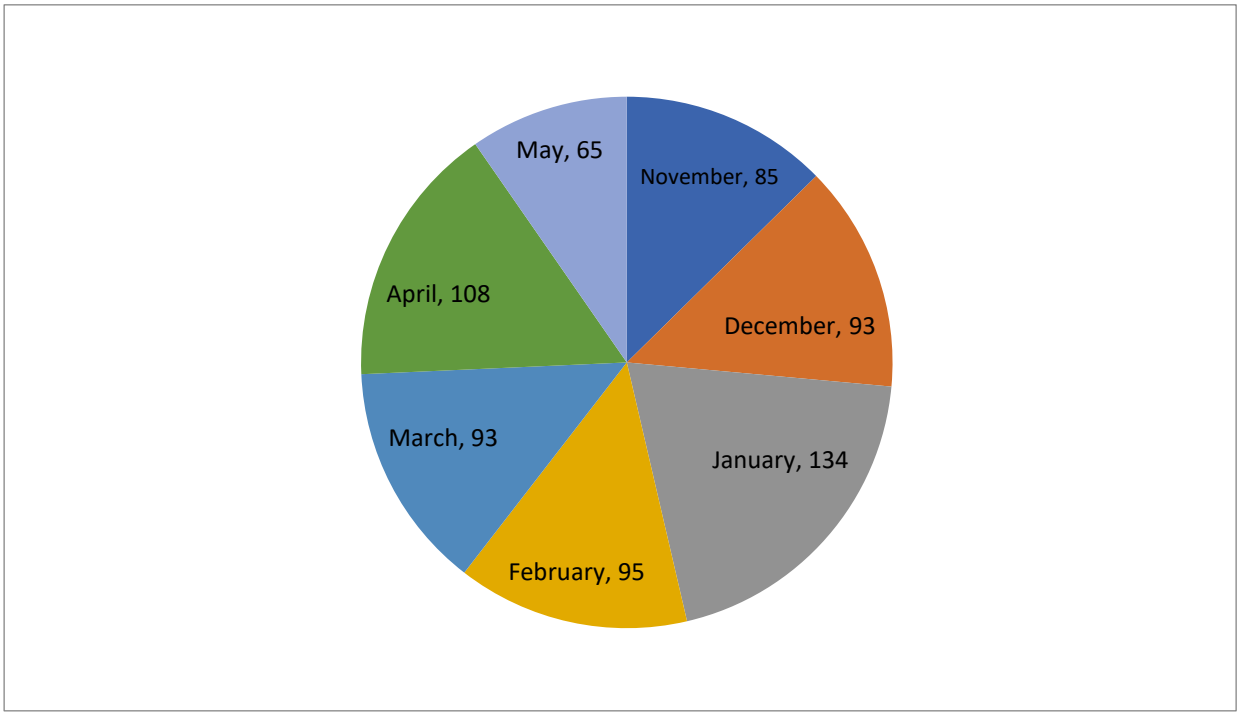


Figure 3. Wolverine reported monthly harvest in Nunavut, from 2019 to 2024. Label indicates total number of wolverines harvested per month.

The male:female ratio of the harvest was highly biased towards males (Fig. 4) with approximately twice as many males harvested than females (ratio = 1.7 and 2.1 in the Kitikmeot and the Kivalliq respectively), a typical figure for northern Canada (Mowat et al. 2020). We assumed sex ratio at birth in the population is 1:1, however, it is difficult to

distinguish if this reflects the actual sex ratio of the population or a difference in vulnerability by sex. Lee (1994) also observed a 2:1 male to female ratio in the western Kitikmeot harvest in the early 1990s. However, his observations showed a strong male bias in the younger age classes (< 2 yo) while the adult sex ratio was not different from a 1:1 ratio. Examining the sex ratio per age class since 2019 seems to show the opposite with a male to female ratio higher in the older individuals (Table 1), possibly pointing towards a diminution in mature females in the population, which will need to be monitored closely.

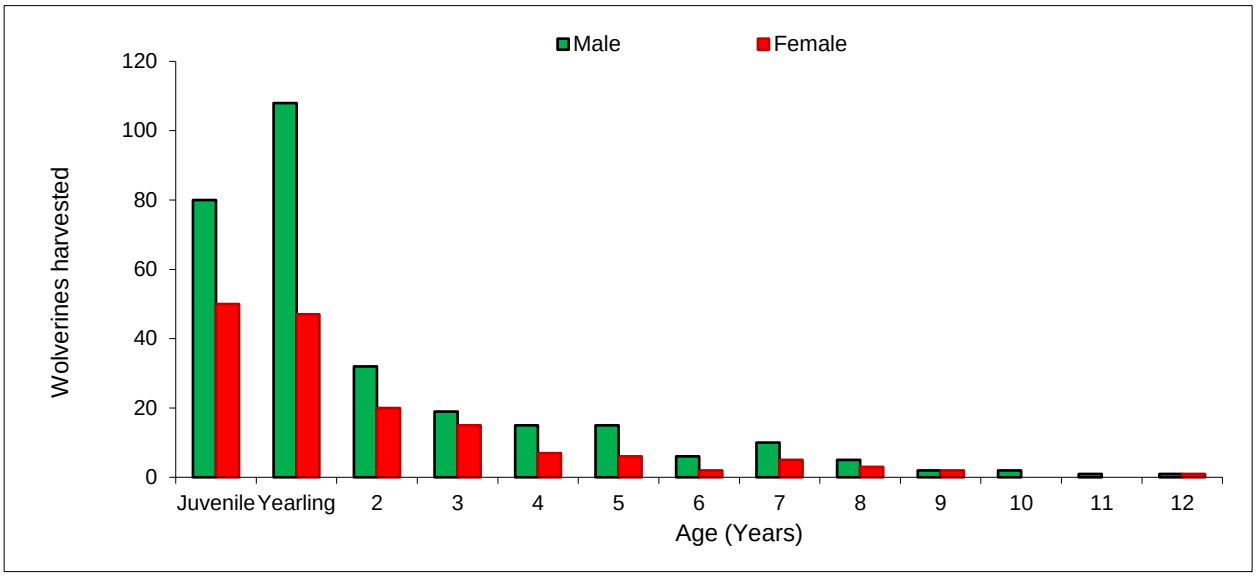


Figure 4. Age and sex structure of the reported wolverine harvest in Nunavut, from 2019 to 2023.

Table 1. Male to female sex ratio by age in the reported wolverine harvest in Nunavut, from 2019 to 2023.

Age	M:F total harvest	M:F sex ratio
Juveniles	80:50	1.6
Yearlings	108:47	2.3
2	32:20	1.6
3	19:15	1.3
4	15:7	2.1
5	15:6	2.5
6	6:2	3
7	10:5	2
8	5:3	1.7
9	2:2	1
10	2:0	-
11	1:0	-
12	1:1	1

The age distribution of the reported wolverine harvest from 2019 to 2023 is weighted more towards juvenile/yearling animals (Fig. 4). The ages of harvested wolverines ranged from <1 year to 12 years. The oldest female and male (12 years) were killed (shot) in Kugluktuk (n = 2) area in 2021 and 2022, respectively. Overall, 29% of the wolverines harvested in Nunavut during that period were juveniles, 34% were yearlings and 37% were adults (Table 2). A greater proportion of young animals in the harvest moderate the conservation risk of harvest (Mowat et al. 2020). However, the proportion of adult females in the harvest increased (13%) for this period (2019-23) compared to 2014-18 (9%, Awan 2020). Wolverine harvest appears to be biased towards younger males (Rausch and Pearson 1972, Lee 1994) probably due to their higher movement rates and vulnerability (Scrafford et al. 2024). Kukka et al. (2017, 2023) describe the high proportions of young males in the harvest to be because vacant areas created by the harvest of resident animals may be filled by dispersing young males (Magoun 1985). Others have reported long dispersal movements in yearlings from their natal area before reaching sexual maturity (Copeland 1996, Mulders 2000, Vangen et al. 2001, Inman et al. 2012) and movement of wolverines from areas with lower mortality to those

with higher mortality (Gervasi et al. 2015, 2016). The high proportion of juveniles and yearlings (63%) and the low proportion of females (36%) among adults in the harvest may signal the importance of dispersal in the population dynamics of this species within the traditional harvesting areas. Yearlings and juveniles represented 64% of the known age male harvest, in contrast to 81% reported by Lee (1994) in the Kitikmeot region in the early 1990's, and by Rausch and Pearson (1972) who also reported wolverine harvest biased toward younger males in Alaska and Yukon.

Table 2. Age and sex distribution of reported wolverine harvest in Nunavut, from 2019 to 2023.

Age Class	Sex		Total (% of total)
	Males (% of males)	Females (% of females)	
Adult (≥ 2 years)	108 (36.5%)	61 (39%)	169 (37%)
Yearling	108 (36.5%)	47 (30%)	155 (34%)
Juvenile (< 1 year)	80 (27%)	50 (31%)	130 (29%)
Total	296	158	454

Figure 5 illustrates the temporal variation in the harvest according to age class. The harvest was dominated by younger age class during the winter while the harvest of adult individuals was comparatively high in March/April, which is similar to other wolverine harvest studies (Lee 2016, Kukka et al. 2017). For males, this corresponds to the time when maximal size of testes and highest levels of testosterone are attained (Pasitschniak-Arts and Larivière 1995), probably marking the start of the breeding season and resulting in increased movements and increased vulnerability of sexually mature males for harvest.

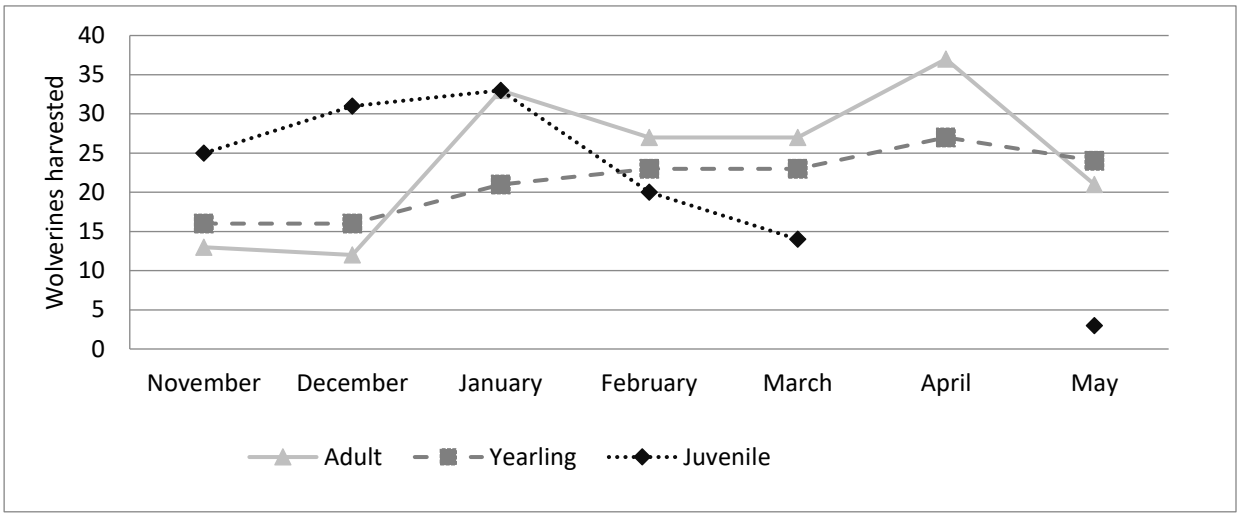


Figure 5. Monthly reported wolverine harvest per age class in Nunavut, from 2019 to 2023.

Wolverine populations in Nunavut are considered healthy. This seems to be supported by the sex and age distribution of the harvest. However, the relatively low number of mature females (13%) in the harvest is important to note and should be monitored closely.

In an effort to obtain information on wolverine population relative abundance, wolverine harvesters (a hunter may provide >1 harvested wolverine to the program within and among years) were asked to answer the following question when bringing back samples to their wildlife office: “Do you think the wolverine population is decreasing, stable or increasing?”. Harvesters’ perception about trends in wolverine populations differed among regions, but majority believed that numbers were stable. In the Kitikmeot (n = 371), most respondents believed the population was stable (67% stable, 32% increasing) while most respondents of the Kivalliq (n = 255) also believed their local population was stable (61% stable, 39% increasing). However, over the long term, the increasing resource development on the tundra and climate change may adversely affect the species, which suggests a need for continuous monitoring of the health of the wolverine population. Monitoring of the wolverine population is also important as part of

predator research and management as it relates to ungulate species management in both the Kivalliq and Kitikmeot regions.

4.0 Acknowledgments

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Building Nunavut Together
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Department of Environment
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Ministère de l'Environnement

SPATIAL AND TEMPORAL PATTERNS OF WOLF HARVEST IN RELATION TO CARIBOU HERD MOVEMENTS IN NUNAVUT

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July 2023

Summary

Enhanced wolf (*Canis lupus*) management actions are often employed to support recovery of declining caribou (*Rangifer tarandus*) herds. Most herds of migratory tundra caribou (*R. t. groenlandicus*) in Nunavut and adjacent Northwest Territories (NWT) are either in decline or at historically low population levels. Beginning in 2018-19 in the Kitikmeot Region and expanding to all of Nunavut in 2019-20, the Government of Nunavut (GN) Department of Environment (ENV) offered compensation through the Support for Active Harvesters program to local hunters to harvest wolves and provide sample and location data. We examined wolf harvest data from 1,500 wolves killed by hunters within Nunavut between 2018-19 and 2020-21 to describe the spatial and temporal patterns of the Nunavut wolf harvest in relation to caribou densities based on collared individuals. We examined the age and sex structure of the wolf harvest, and the spatial and temporal clustering of the harvest (i.e., whether hunting is effective at removing entire packs). We also provided rough estimates of the number of wolves associated with the Qamanirjuaq caribou herd and of the proportion of these wolves harvested by Nunavut hunters.

During 2018-19, 146 wolves were recorded harvested by 52 hunters within the Kitikmeot, primarily by hunters from Kugluktuk. Harvest numbers peaked in December and February–May, with a high proportion of the harvest during October–December comprised of juveniles and a wider distribution of ages harvested as the season progressed.

During 2019-20, 658 wolves were recorded harvested across Nunavut, with 64% of the harvest from Arviat, Baker Lake and Kugluktuk hunters. Harvest peaked in November and March–May. During 2020-21, 699 wolves were recorded harvested, with 68% of the harvest from Arviat, Baker Lake and Kugluktuk hunters. In both years, the proportion of juveniles in the harvest declined over time and a relatively small number of hunters contributed a large portion of the harvest. Nearly all (99%) of the wolves were shot and the remainder were trapped.

Although patterns differed among communities, wolves were generally harvested closer to communities prior to Christmas (generally within 75 km) and further from communities later in the winter, with peak distances in April and May (up to 500 km). Thus, even though the hunters were far from their communities, necessitating camping over day-trips, they were effective at harvesting larger numbers of wolves. Arviat hunters harvested wolves over a broad distance later in the winter, including over 450 km from the hamlet in 2019-20, but the distribution of harvest was higher closer to higher densities of caribou. The distribution of high densities of Qamanirjuaq caribou and hence the wolf harvest differed between years. The proportion of packs harvested differed among communities and generally decreased as pack size increased and increased as the season progressed.

Rough estimates of the number of wolves associated with the Qamanirjuaq caribou herd suggest that hunters annually removed approximately 12–25% of the wolves associated with the late winter and spring movements of the Qamanirjuaq herd. Estimates of wolf numbers and proportion of wolves harvested are based on assumed rather than observed relationships between caribou and wolf numbers and therefore should be interpreted very cautiously. More work is required to better estimate how many wolves may be associated with the migratory herds, especially the Qamanirjuaq herd, during winter and spring, but given high harvests of wolves south and southwest of Arviat during March, April and May, it is likely that

numbers of wolves and wolf-killed caribou are greatly reduced as the herd migrates to their calving grounds west of the coast of Hudson Bay, resulting in low numbers of wolves and reduced wolf predation on the calving and early post-calving grounds.

Wolf harvesting is a traditional part of the culture of Nunavut hunters, as well as a source of seasonal income. The effectiveness of Nunavut wolf hunters appears high, with large annual harvests from some communities, especially Arviat, Baker Lake and Kugluktuk. Nunavut wolf hunters appear able to target older and presumably potentially breeding portions of the population later in the winter after most natural losses would have occurred, increasing the additive impacts of these ground-based harvest efforts. In some areas wolf harvest during spring migration result in low numbers of wolves and reduced wolf predation on the calving and post-calving grounds. Since wolf populations can quickly rebound when treatment is removed due to immigration from surrounding areas and high reproductive rates, the incentive program should be continued.

ᐃᓂᒋᖅᑕᕐᓂᖅᓯᓐᓄᑦ ᐊᒻᒪᓗ ᖃᓄᐃᑦ ᐊᑯᓂᐅᑎᓂᖅᓯᓐᓄᑦ ᖃᓄᐃᓕᐅᖅᑭᓚᐅᕆᓂ ᐊᒪᑭᖅᑕᐅᔪᑦ ᐱᔪᓐᖃᖅᑐᑦ ᑐᖅᑐᐃᑦ ᐊᐅᓚᓕᓂᖅᓯᓐᓄᑦ ᓄᓇᕗᒻᒥᑦ

ᐊᐅᓚᑦᑎᐊᓂᖅᓴᖅᑐᑦ ᓇᐃᔨᖅᓱᔪᑦ

ᐱᐅᓱᒋᐊᖅᓱᒪᔪᖅ ᐊᒪᖅᖁᑯᓂᒃ (*Canis lupus*) ᐊᐅᓚᑦᑎᐅᑦ ᖃᓄᐃᓕᓕᐅᑕᐅᔪᑦ ᐃᑲᔪᖁᓂᐊᕐᓗᑦ ᐊᒥᓱᔪᓂᕐᕿᑭᓚᐅᓂᖅᑐᑦ ᐅᑎᖅᑎᑦᑐᒪᑦᓗᑦ ᑕᐅᑐᐃᑦ (*Rangifer tarandus*) ᐊᒥᕋᐃᑦ. ᑕᒪᓕᐅᑦᓯᓚᐅ ᐊᒥᖁᕋᐃᑦ ᐅᑎᖅᑕᖃᖅᑕᖅᑐᑦ (*R. t. groenlandicus*) ᓄᓇᒧᑦ ᓄᓇᕗᒻᒥᑦ ᐊᒻᒪᓗ ᖃᓄᑦᑕᓂ ᓄᓇᑦᑎᐊᖏᑦ ᐊᒥᓱᔪᓂᕐᕿᑭᓚᐅᔪᓂ ᐅᕐᖤᔭᓂᐅᑦ ᑕᐃᒪᓐᓇᓂᒥᑦ ᐊᒥᓱᑎᑐᓂᖅᑐᑦ ᐊᑦᑎᓯᔪᓚᒥᖅᑐᑦ. ᐱᕙᐊᓴᖅᑎᓐᓗᒍ 2018-19 ᖃᑎᖅᑎᐅᓂᑦ ᐊᒻᒪᓗ ᑎᑭᐅᑎᖤᓚᐊᓕᔪᓂ ᓄᓇᕗᔭᑦᑕᖏᑦ 2019-20-ᒥᑦ, ᓄᓇᕗᒻᒥ ᒪᖤᒪᔭᑯᑦ ᐊᖤᑎᓕᓐᓇᔭᑯᓂᑦ ᐅᑎᖅᑎᑦᑎᖅᑕᑦᑕᕐᒪᓕᓕᐅᖅᑐᑦ ᐃᑲᔪᖁᓂᒍ ᒪᖅᐃᖅᑕᖅᑐᓄᑦ ᑐᖅᑐᐃᑦ ᓄᓇᕗᑦᑕᐃᓕᓐᓂᑦ ᒪᖅᐃᑦᑕᓂᖅᑐᓄᑦ ᓄᓇᕗᓕᒥᐅᑕᐃᑦ ᒪᖅᐃᑎᑦ ᐊᒪᑭᖅᑐᓂᒃ ᐃᓕᓴᕐᓂᒃ ᐊᒪᓕᓐᓂᒃ ᐊᑐᖅᑐᐃᑦ ᐊᒻᒪᓗ ᑐᓂᓯᒪᓂᖅ ᐊᒪᑭᖅᑕᐅᔪᓂᒃ/ᑎᑎᖅᖢᓂᒃ. ᖃᒥᖅᑭᓕᐅᖅᑕᕐᓂᖅ ᐊᒪᑭᖅᑕᐅᔪᓕᔪᓂᑦ ᓴᓇᕐᑎᑦᑎᓂᑦ/ᑎᑎᖅᖢᐃᑦ 1,500-ᖏᔪᔪᓂᑦ ᐊᒪᑭᐃᑦ ᑐᖁᑕᐅᔪᓂᒃ ᒪᖅᐃᑎᓄᓂᑦ ᓄᓇᕗᒻᒥᑦ ᐊᑯᓂᔪᓂᕐᓂᑦ 2018-29 ᐊᒻᒪᓗ 2020-21 ᐅᖅᐅᓱᓚᓂᐊᖅᓗᑦ ᐃᓂᒋᖅᑕᕐᓂᖅᓯᓐᓄᑦ ᐊᒻᒪᓗ ᖃᓄᐃᑦ ᐊᑯᓂᐅᑎᓂᖅᓯᓐᓄᑦ ᖃᓄᐃᓕᐅᖅᑭᓚᐅᕆᓂ ᓄᓇᕗᒻᒥ ᐊᒪᑭᖅᓱᐅᖅᑎᓂᓂ ᐱᔪᓐᖃᖅᑐᓄᑦ ᑐᖅᑐᐃᑦ ᐊᑯᓂᐅᑎᓂᖅᓯᓐᓄᑦ ᐃᑲᔪᖁᓂᒍ ᐊᑐᖅᓯᓐᓄᑦ ᐊᒻᒪᓗ ᖃᓄᐃᑦ ᐊᑯᓂᐅᑎᓂᖅᓯᓐᓄᑦ ᑲᒪᓂᓱᐊᖅᑕᐅᔪᓂᒃ, ᐊᒻᒪᓗ ᐃᓂᒋᖅᑕᕐᓂᖅᓯᓐᓄᑦ ᐊᒻᒪᓗ ᖃᓄᐃᑦ ᐊᑯᓂᐅᑎᓂᖅᓯᓐᓄᑦ ᑲᑎᓂᕐᓂᑦ ᐊᒪᑭᖅᑕᐅᔪᓂᒃ (ᐅᐅᑐᑎᑎᓕᔪ, ᐊᒪᑭᖅᑕᐅᖅᑕᕐᓂᖅᓯᓐᓄᑦ ᐃᑲᔪᐃᓂᖅᓴᖅᑎᖏᑦ ᐱᓕᕐᓂᖅᓯᓐᓄᑦ ᑕᒪᑎᓕᓐᓂᑦ ᐊᒪᑭᐃᑦ ᐃᑐᐊᓂᖅᑐᖅᑐᑦ ᐊᒻᒪᓗ ᐊᓯᔾᔨᑎᑦᑐᓂᒃ ᐅᑭᐅᖅᑕᖅᑐᓂᒃ ᐊᒪᑭᖅᑕᐅᖅᑕᕐᓂᖅᑐᓂᒃ ᐱᖓᓱᒪᒋᔭᖅᑐᓂᒃ ᑕᒪᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ ᐊᒪᑭᖅᑕᐅᔪᓂᒃ ᓄᓇᕗᒻᒥ ᒪᖅᐃᑎᓄᑦ.

2018-19-ᒥᑦ, 146 ᐊᒪᑭᖅᑕᐅᔪᑦ ᑎᑎᕋᐅᓯᐅᓕᐅᖅᑐᑦ 52 ᒪᖅᐃᑎᓄᑦ ᖃᑎᖅᑎᐅᓂᑦ, ᐱᓗᐊᖅᑐᒥᑦ ᒪᖅᐃᑎᓂᑦ ᓴᐃᔪᖅᑐᑎᒥᑦ. ᐊᒪᑭᖅᑕᐅᔪᑦ ᖃᓚᓯᐅᓂᖅᑐᑦ ᐊᒥᓱᓄᐃᖅᓱᓕᐅᖅᑐᑦ ᐃᓐᓇᓕᓴᕐᓗᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓂᖅᓴᖅᑎᖏᑦ ᓄᓕᖅᑕᐃᓕᒥᑦ, ᐊᒻᒪᓗ ᐃᓂᒋᖅᑕᕐᓂᖅᓯᓐᓄᑦ ᐊᒻᒪᓗ ᖃᓄᐃᑦ ᐊᑯᓂᐅᑎᓂᖅᓯᓐᓄᑦ ᑲᑎᒪᔨᖅᑐᑦ ᐊᒪᑭᖅᑕᐅᔪᓂᒃ (ᐅᐅᑐᑎᑎᓕᔪ, ᐊᒪᑭᖅᑕᐅᖅᑕᖃᖅᑕᖅᑐᑦ ᐃᑲᔪᖁᓂᐊᕐᓗᑦ ᐅᑭᐅᖅᑕᖅᑐᑦ ᐊᒪᑭᖅᑕᐅᖅᑕᕐᓂᖅᑐᓄᑦ ᐱᓕᕐᓂᖅᓯᓐᓄᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓂᖅᓴᖅᑎᖏᑦ ᐃᓐᓇᓕᓴᕐᓗᑦ ᐊᒻᒪᓗ ᐊᒪᑭᖅᑕᐅᔪᓂᒃ ᐃᓂᒋᖅᑕᕐᓂᖅᓯᓐᓄ

ᓄᓇᓕᑦ ᐊᑯᓂᖅᑭᓂᑦ ᖃᓄᐃᓕᓚᐅᖅᐸᓪᓚᐊᓂᖅᑭᑦ ᐊᕐᓴᒋᓚᐅᙵᖅᑭᑦᑐᑎᓗᐃᑦ, ᐊᒪᑕᖅᑕᐅᖃᖅᑎᑦᑕᖅᑐᑦ ᓄᓇᓕᖅᓚᑦ ᖃᓄᓂᓂᖅᓴᐅᔨᖃᖅᑎᑦᑕᓚᐅᖅᑐᑦ ᐊᓂᐟᕕᐊ ᖃᖅᑲᓚᕆᖅᑭᑦ (75 ᑭᓯᒥᑕᓂᒃ ᖃᓄᑎᒋᔨᒥᑦ) ᐊᒻᒪᓗ ᓄᓇᓕᖅᓂᓂᑦ ᐅᖅᓱᓂᖅᓴᐅᔨᓂᑦ ᐅᑭᐅᖅᑎᑦᓗᒍ, ᐅᖅᓱᓂᓯᓂᖅᓴᐅᓕᖅᓂᒃ ᑎᓪᓗᓈᑦ-ᒥᑦ ᐊᒻᒪᓗ ᓄᖏᐃᑦ-ᒥᑦ (500 ᑭᓯᒥᑕᓄᑦ ᑎᑭᓕᒍ). ᑕᐃᒪᐃᓇᒧᑦ, ᒪᖃᐃᐃᑲᑦ ᐅᖅᓱᓂᒃᓚᐊᖅᓴᑎᒃ ᓄᓇᓕᖅᑐᓂᑦ, ᑕᖅᓯᒪᓕᖅᖃᖅᑕᖅᓴᓕᑎᒃ, ᑕᐃᒪᐃᓇᒧᑦ ᐊᒥᓱᓂᖅᓴᒃ ᐊᒪᑕᖅᑕᑕᓯᐅᖅᑕᖅᑐᑦ. ᐊᖏᔪᐊᓂᑦ ᒪᖃᐃᐃᑲᑦ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᑭᓐᓂᖅᓴᐅᓯᐅᖅᑐᖅ ᐅᑭᐅᖅᑎᑦᑕᓂᒃ, 450 ᑭᓯᒥᑕᓂᒃ ᐅᖅᓱᓂᑎᒥᔪᒥᑦ ᐊᖏᔪᐊᓂᑦ 2019-20-ᒥᑦ, ᑭᓱᐊᓂ ᓇᒧᖅᐅᓱᒪᓂᖅᑭᑦ ᐊᒪᑕᖅᑕᐅᔨᑦ ᖁᓕᑎᖅᓂᖅᓴᐅᓕᐅᖅᑐᖅ ᖃᓄᓂᓂᖅᓴᐅᔨᑦ ᐊᒥᓱᑎᑎᓂᐅᔨᓂᑦ ᑐᑦᑐᓂᑦ. ᓇᒧᖅᐅᖃᖅᑕᕐᓂᖅᑭᑦ ᐊᒥᓱᒃᑲᓕᐅᔪᓚᑎᒃ ᖃᒪᓂᖏᔪᐊᖅ ᑐᑦᑐᖅᑭᑦ ᐊᒻᒪᓗ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᕐᓴᒋᓯᖅᑭᖅᑕᖅᑐᑦ ᐅᑭᐅᖅ ᐊᑯᓂᖅᑭᑐᓂᑦ. ᐃᓕᖅᑭᑦ ᑲᑎᒪᔨᑦ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᕐᓴᒋᓯᖅᑭᖅᑕᖅᑐᑦ ᓄᓇᓕᖅᑐᓂᑦ ᐊᑯᓂᖅᑭᑐᓂᑦ. ᐊᒥᓱᓂᖅᑐᑦ ᐊᒥᓱᒃᑲᓕᐅᔪᓚᑎᒃ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᒻᒪᓗ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᕐᓴᒋᓯᖅᑭᖅᑕᖅᑐᑦ ᐅᑭᐅᖅᑭ ᐊᑯᓂᖅᑭᑐᓂᑦ. ᐊᓪᓚᕆᐊᖓᑲᖅᓯᐅᓕᖅᑐᑦ ᓄᓇᓕᖅᑐᓂᑦ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᒻᒪᓗ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᑐᖅᑐᑦ ᐊᒻᒪᓗ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᕐᓴᒋᓯᖅᑭᖅᑕᖅᑐᑦ ᓄᓇᓕᖅᑐᓂᑦ ᐊᑯᓂᖅᑭᑐᓂᑦ. ᐊᒻᒪᓗ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᕐᓴᒋᓯᖅᑭᖅᑕᖅᑐᑦ ᐊᒻᒪᓗ ᐊᒪᑕᖅᑕᐅᔨᑦ ᐊᕐᓴᒋᓯᖅᑭᖅᑕᖅᑐᑦ ᓄᓇᓕᖅᑐᓂᑦ ᐊᑯᓂᖅᑭᑐᓂᑦ.

ININGINNI TADJALU QANURILIURUTAIT AMARUNI ANGUNIARNIRNI PIDJUTIKKUT TUKTUT INGILRADJUTAINNI NUNAVUNMI

Aulapkaijini Naittuq Naunaijaruti

Pijuummiqtiqhimajunik amaruni (*Canis lupus*) munaridjutainni qanuriliurutait atuqtauvaktut ikajurnirmun ikikliliqtunun tuktuni (*Rangifer tarandus*). Amigaitqijaujun ingilrainnaqtuni ukiuqtaqtumi tuktuit (*R. t. groenlandicus*) Nunavunmi haniliriiktumilu Nunatsiami (NWT) naliani ikilivallialiqtun imaaluunniit ingilraarnitamin ikikliqpiaqtun amihuarjuutainni. Aullaqtiqhuni 2018-19mi Kitikmeoni pivaliqhunilu tamaat Nunavunmun 2019-20mi, Nunavut Kavamanga (GN) Avatiliqijikkut Havagvia (ENV) hailihimapkaijun manikhanik talvuuna Ikajuutikkut Anguniaqtinun pinhuarutimimi ukununga nunallaami anguniaqtinun anguniarnirnun amarunik tunilutiklu timaanin ihivriuqtakhamik humiinniinnilu naunaitkutainnik. Ihivriuqtavun amaruit angujauhimajun naunaitkutainnik ukunanga 1,500 amarunin anguniaqtinin Nunavunmi uvani 2018-19min uvanilu 2020-21mun naunaijariami humiinninginnik qangalu qanuriliurutait Nunavunmin amarunik angujaujunin pidjutaitigun tuktunun amigainniinnik tunnganiqarhuni qunguhirmiaqaqtunin tuktuinnin. Ihivriuqtavun ukiungit anguhaluujaakhaita arnarluujaakhaita amaruni angujauhimajuni, ukuallu humiittaakhaitat qangalu katihimadjutait anguniaqtaujuni (imaa, naliak anguniarniit ihuarutaukpat nungupkarutainnik tamainnik amaruqatigiingnik). Tunihimajugullu nalautinniarhimalugin qaffiujaakhainnik amaruit pidjutilgit uvani Qamanirjuaq tuktuinni avvainnilu hapkunani amaruni angujauhimajunik Nunavunmin anguniaqtiinnin.

Atuqtilugu 2018-19, 146 amarut titiraqtaujut angujaujut 52nik anguniaqtinin Kitikmeonin, taapkunanngalluarlu anguniaqtinin Kugluktumin. Anguniarnikkut qaffiuniit amigaitqijaujun Tisaipami Fibjualimi–Maymilu, amigaitqijaupluni anguniarniq uvani Aktuupami–Tisaipami ilaujut amaruat aallatqiillu ukiuqarutait angujaujun anguniarvik hivumuutillugu.

Talvani 2019-20, 658 amarut titiraqtauhimajun angujaujun tamaani Nunavunmi, pipluni 64% avvaanik anguniarutinin Arvianin, Qamanittuamin Kugluktuminlu anguniaqtiinnin. Anguniarniit amigaitqijaujun Nuvaipami uvanilu Maatsimi–Maymilu. Talvani 2020-21mi, 699 amarut titiraqhimajun angujaujun, pipluni 68% avvaanik anguniaqtaujun Arvianin, Qamanittuamin Kugluktumillu anguniaqtinin. Taapkuangni ukiuni, amigainniit amaruani angujaujuni ikiklilaaqtuq hivitunirmi taimaalu ikitqijaujullu anguniaqtiit pidjutijun amigaitqijaujumik avvaanik anguhinirmi. Tamatqiutivjaktun (99%) amarunik hiquqtauhimajun ilaquillu naniriaqtauhimajun.

Taimaitkaluaqhuni qanurinniit aallangajun naliinni nunallaani, amarut angujauvaktun qanitqijaujumik nunallaanun hivuagun Quviahukvingmi (imaakiaq iluani 75ni km) ungahitqijaujunilu nunallaanin atukhaarniani ukiup, piqarhuni ungahitqijaujunik Aprilmi Maymilu (talvunga 500nik km). Taimaatun, taapkuat anguniaqtiit ungahikkaluaqhutik nunallaamingin, piplutik tangmaaqpakhutik, nakuujumik anguniaqtun amigaitqijaujunik amarunik. Arvianin

anguniaqtiit angujun amarunik ungahiktuni atukhaaqtumi ukiumi, ilaujuqlu avatquhimajumik 450 km haamlanganin 2019-20mi, kihimi pidjutaa anguniarniup amigaitqijauvaktuq qanitiqjaani amigaitqijauniqarnirni tuktuni. Una pidjutaa amigaitqijauniqarnirni uvani Qamanirjuaq tuktuni taimaidjutaa uvani amaruni angujaujuni aallakkiiktun ukiuni. Amigainniit amaruqatiinni angujaujuni aallangajuq nunallaani taimaalu ikiklijuummiqtun amaruqatiit amigairaangata amigaiqhutiklu ukiup hivumuuniani.

Nalautinniaqhimajun qaffiujaakhaita amaruit pidjutilgit uvani Qamanirjuaq tuktuni ihumagijun anguniaqtiit ukiuq tamaat anguvaktait imaakiaq 12-25% avvaani amaruni pidjutilgit atukhaarniani ukiumi upingaamilu ingilradjutainni uvani Qamanirjuaq tuktuinni. Nalautinniarhimajun amaruni qaffiujaakhainni ukunanilu qaffiujuni amarut angujauvaktun tunnganiqaqtun ihumagijaujuni kihiani tautukhimanikkut pidjutainni tuktuni amarunilu qaffiunirni taimaalu uqaqtaujukhat ihumagidjutiinnakkut. Havagjuummirniit ihariagijaujun nakuutqijaujumik qaffiujun amarut pidjutiqarniinni ukunani ingilrainnaqtuni tuktuinni, ukuallaunilu Qamanirjuaq tuktuinni, talvani ukiumi upinnaamilu, kihimi pidjutaani amigaittuni angudjutini amaruni hivuraani uvanilu hivuraani-ualliani Arviani Maatsimi, Aprilmi Maymilu, pijungnaqhijuq qaffiuniit amaruni ukunanilu amarunin-tuqutauvaktun tuktuit ikiklijuummiqpiaqpaktun taimaatun tuktui ingilradjutainni nuriviinnun nunanun ualliani hinaani Kangiqsualuup, pidjutingani qaffiunirnik ikittunik amarunik uvanilu ikittunik amarunin tuqutirutainnik nurrivingni uvanlu pilihaaliqtuni nurrihuirutimi nunainni.

Amarunik anguniarniit pitquhikkut ilaujuq pitquhiinni Nunavunmin anguniaqtiinnin, pidjutiujuqlu qakunguraanga maniliurutait. Nakuuniqarniit uvani Nunavunmin amarunik anguniaqtiinnin qulvahiktutun ittuq, amigaittuni ukiuq tamaat anguniarutinin ilanginnin nunallaanin, ukualluanin Arvianin, Qamanittuamin Kugluktumillu. Nunavunmin amarunik anguniaqtiit tautungnarniqaqtun turaangajait angajukhitqijanik najjijaaqtuni ilainni amaruni atukhaaqniani ukiumi kinguagun amigaitqijanik aularaanginnaqtukkut tuqudjutainnik pikpata, amigaiqjuummiqtiqhugin hapkuat ilaudjutaujun hulqutit hapkunani nunami-tunnganiqaqtun anguniarnikkut havagutit. Ilanginni nunani amaruni anguniarniit upingaami ingilradjutainni pidjutivaktun ikittunik amarunik imaalu ikiklijunik amarunin tuqutirutinik uvani nurrivingni uvanlu nurrihuirutimi nunainni. Taimaatun amaruit amigairniit qilamik amigaiqtaaqmata hanaqidjuti nutqaqtitaugaangat pidjutipluni qaidjutainnik hanianiittunin nunanin ukunangalu amigaittunin najjijunin aktilaanginnin, una pilaqhuutikkut pinahuaruti aulapkaqtitauhimmaaqtukhaq.

TENDANCES SPATIALES ET TEMPORELLES DE LA CHASSE DU LOUP PAR RAPPORT AUX DÉPLACEMENTS DES HARDES DE CARIBOUS AU NUNAVUT

Résumé

Des mesures de gestion du loup (*Canis lupus*) améliorées sont souvent utilisées pour appuyer la restauration des hardes de caribous (*Rangifer tarandus*) en déclin. La plupart des hardes de caribous migrateurs de la toundra (*R. t. groenlandicus)* au Nunavut et dans les Territoires du Nord-Ouest (T.N.-O.) sont en déclin ou à des niveaux de population historiquement bas. Depuis 2018-2019 dans la région de Kitikmeot, et depuis 2019-2020 dans l'ensemble du Nunavut, le ministère de l'Environnement du gouvernement du Nunavut (GN) offre une incitation dans le cadre du programme de soutien aux chasseurs actifs aux chasseurs locaux pour chasser le loup et fournir des échantillons et des données de localisation. Nous avons examiné les données de 1 500 loups tués par des chasseurs au Nunavut entre 2018-2019 et 2020-2021 afin de décrire les tendances spatiales et temporelles de la chasse du loup au Nunavut par rapport aux densités de caribous, fondées sur les animaux munis d'un collier. Nous avons examiné la structure par âge et par genre des loups chassés, ainsi que le regroupement spatial et temporel de la récolte (c.-à-d. si la chasse peut efficacement éliminer des meutes entières). Nous avons également fourni des estimations approximatives du nombre de loups associés à la harde de caribous de Qamanirjuaq et de la proportion de ces loups récoltés par les chasseurs du Nunavut.

En 2018-2019, 146 loups ont été récoltés par 52 chasseurs dans le Kitikmeot, principalement par des chasseurs de Kugluktuk. Le nombre de prises a atteint un sommet en décembre et en février-mai, avec une forte proportion de la récolte d'octobre à décembre ayant étant composée de jeunes loups. La distribution des âges des prises s'est élargie au fur et à mesure que la saison a avancé.

En 2019-2020, 658 loups ont été récoltés dans l'ensemble du Nunavut, dont 64 % par des chasseurs d'Arviat, de Baker Lake et de Kugluktuk. La récolte a atteint un sommet en novembre et en mars-mai. En 2020-2021, 699 loups ont été récoltés dans l'ensemble du Nunavut, dont 68 % par des chasseurs d'Arviat, de Baker Lake et de Kugluktuk. Au cours de ces deux années, la proportion de jeunes loups dans la récolte a diminué au fil du temps et un nombre relativement faible de chasseurs a contribué à une majorité de la récolte. Presque tous les loups (99 %) ont été abattus et le reste a été piégé.

Les tendances diffèrent d'une collectivité à l'autre, mais avant Noël, les loups étaient généralement récoltés près des collectivités (dans un rayon d'environ 75 km) et plus tard en hiver, ils l'étaient loin des collectivités, les plus grandes distances ayant été en avril et en mai (jusqu'à 500 km). Ainsi, même si les chasseurs étaient loin de leurs collectivités, nécessitant de camper sur site lors d'excursions de plus d'une journée, ils pouvaient efficacement récolter un plus grand nombre de loups. Les chasseurs d'Arviat ont chassé les loups sur une grande distance plus tard au cours de l'hiver, y compris à plus de 450 km du hameau en 2019-2020, mais la distribution des prises était supérieure plus près des densités élevées de caribous. La distribution des fortes densités de caribou de Qamanirjuaq et, par conséquent, la récolte de loups, a différé d'une année à l'autre. La proportion de meutes récoltées variait d'une collectivité à l'autre et diminuait généralement à mesure que la taille des meutes augmentait, et augmentait au fur et à mesure que la saison avançait.

Des estimations approximatives du nombre de loups associés à la harde de caribous de Qamanirjuaq suggèrent que les chasseurs ont éliminé annuellement environ 12 à 25 % des loups associés aux déplacements de la harde de Qamanirjuaq à la fin de l'hiver et au printemps. Les estimations du nombre de loups et de la proportion de loups récoltés sont fondées sur des relations présumées plutôt qu'observées entre le nombre de caribous et de loups et doivent donc être interprétées avec beaucoup de prudence. Du travail supplémentaire est nécessaire pour mieux estimer le nombre de loups qui pourraient être associés aux hardes migratrices, en particulier la harde de Qamanirjuaq, en hiver et au printemps, mais compte tenu des prises élevées de loups au sud et au sud-ouest d'Arviat en mars, en avril et en mai, il est probable que le nombre de loups et de caribous tués par les loups diminue considérablement à mesure que la harde migre vers ses aires de mise bas à l'ouest de la côte de la baie d'Hudson, entrainant ainsi un faible nombre de loups et une réduction de la prédation des loups dans les aires de mise bas et de début de la période postvêlage.

La chasse du loup fait partie intégrante de la culture des chasseurs du Nunavut et constitue une source de revenus saisonniers. L'efficacité des chasseurs de loups du Nunavut semble élevée, compte tenu des récoltes annuelles importantes dans certaines collectivités, en particulier Arviat, Baker Lake et Kugluktuk. Les chasseurs de loups du Nunavut semblent être en mesure de cibler les parties plus âgées et vraisemblablement reproductrices de la population plus tard au cours de l'hiver, après que la plupart des pertes naturelles se soient produites, ce qui augmente les effets cumulatifs de ces efforts de récolte au sol. Dans certaines régions, la récolte du loup pendant la migration printanière réduit le nombre de loups ainsi que la prédation des loups dans les aires de mise bas et de postvêlage. En raison du fait que les populations de loups peuvent rapidement rebondir lorsque le traitement est retiré en raison de l'immigration en provenance des régions environnantes et des taux de reproduction élevés, le programme d'incitation devrait être poursuivi.

Table of Contents

Summary ii
ᐊᐅᓪᓗᒍᓂᖅᑐᑦ ᓇᐃᔭᖅᓴᓂᑦ iv
Aulapkaijini Naittuq Naunaijaruti vi
Résumé viii
Introduction 1
Study Area 2
Study Design and Methodology 3
Spatial data acquisition 3
Wolf harvest data 3
Caribou collar data 4
Data preparation 5
Proportion of wolf packs harvested 6
Results 8
Wolf harvest 8
Kitikmeot 2018-19 8
Nunavut 2019-20 10
Nunavut 2020-21 14
Harvesters' perception of wolf population trends 17
Wolf harvest relative to distance from communities and caribou 17
Proportion of wolf packs harvested 24
Kitikmeot 2018-19 24
Nunavut 2019-20 25
Nunavut 2020-21 26
All data combined 28
Wolf numbers and proportion harvested 29
Discussion 31
Limitations 33
Recommendations 34
Acknowledgements 35
Literature cited 35
Appendix 1: Caribou collar sample size 40
Appendix 2: Monthly caribou and wolf harvest maps October 2018 – June 2021 42

List of Tables

Table 1. Number and sex of recorded wolf harvests by community, Kitikmeot Region, 2018-19. 9
Table 2. Distribution of wolf harvest by individual hunters, Kitikmeot Region, 2018-19. 9
Table 3. Distribution (*n* = 146), mean age (*n* = 139) and proportion of males (*n* = 128) in the monthly wolf harvest, Kitikmeot Region, 2018-19. 9
Table 4. Number and sex of recorded wolf harvests by community, Nunavut, 2019-20. 12
Table 5. Distribution (*n* = 653), mean age (*n* = 604) and proportion of males (*n* = 592) in the monthly wolf harvest, Nunavut, 2019-20. 12
Table 6. Distribution of wolf harvest by individual hunters by region, Nunavut, 2019-20. 13
Table 7. Number and sex of recorded wolf harvests by community, Nunavut, 2020-21. 15
Table 8. Distribution (*n* = 699), mean age (*n* = 599) and proportion of males (*n* = 685) in the monthly wolf harvest, Nunavut, 2020-21. 15
Table 9. Distribution of wolf harvest by individual hunters by region, Nunavut, 2020-21. 17
Table 10. Perception of wolf population trend (%) of wolf hunters providing carcasses for the Support for Active Harvesters program, 2018-19 – 2020-21. 17
Table 11. Collar sample size by caribou herd and month, December 2018 to June 2021, and most recent caribou herd estimates. 40

List of Figures

Figure 1. Map of Nunavut and communities; the Support for Active Harvesters program applies to the entire territory. 3
Figure 2. Median daily movement distances (m) by caribou for the GNWT Environment and Climate Change (ECC), and Government of Nunavut Department of Environment (Nunavut) data sets. 6
Figure 3. Distribution of reported wolf harvest in the Kitikmeot Region, 2018-19. Symbols may represent >1 wolf harvested at that location. 8
Figure 4. Distribution of age of harvested wolves by 3-month period, Kitikmeot Region, 2018-19 (*n* = 139). 10
Figure 5. Distribution of reported wolf harvest in Nunavut, 2019-20. Symbols may represent >1 wolf harvested at that location. 11
Figure 6. Distribution of age of harvested wolves by 3-month period, Nunavut, October 2019 – June 2020 (*n* = 596). 13
Figure 7. Distribution of reported wolf harvest in Nunavut, 2020-21. Symbols may represent >1 wolf harvested at that location. 14
Figure 8. Distribution of age of harvested wolves by 3-month period, Nunavut, October 2020 – June 2021 (n = 591). 16
Figure 9. Proportion of young wolves within the annual harvest by Nunavut region, 2019-21. 16

Figure 10. Wolf harvest locations plotted by distance from communities (km) during the A) 2018-19 (Kitikmeot), B) 2019-20 (Nunavut) and C) 2020-21 (Nunavut) harvest seasons. Smoothed lines provide an estimate of overall trend in the data set. 18

Figure 11. Relationship between wolf harvest distance from caribou centers of activity (higher densities) and communities, Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21. 19

Figure 12. Number of wolves observed (A) and number of wolves harvested (B) relative to the distance from caribou centers of activity (higher densities), Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21 data combined. “Lone” = no group size reported but harvested at the same location on the same date; “NoObs” = no group size reported but harvested ≤5 km apart on the same date; “Observed” = pack size observation. 20

Figure 13. Number of wolves harvested related to distance from community and binned distances from higher densities of caribou, Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21. 21

Figure 14. Locations of wolves harvested grouped by distance from areas of higher caribou density (DistanceCaribou), Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21. 22

Figure 15. Collared caribou distribution by herd and wolf harvest collection locations, March 2019. 23

Figure 16. Collared caribou distribution by herd and wolf harvest collection locations, April 2020. 23

Figure 17. Collared caribou distribution by herd and wolf harvest collection locations, April 2021. 24

Figure 18. Mean proportion of wolf packs (≥2 wolves) harvested by month generated by QGIS assessment, Kitikmeot, 2018-19. Sample sizes shown above bars. Linear trendline added. 25

Figure 19. Mean proportion of wolf packs (≥2 wolves) harvested by community generated by QGIS assessment, Nunavut, 2019-20. Sample sizes shown above bars. 26

Figure 20. Mean proportion of wolf packs (≥2 wolves) harvested by month generated by QGIS assessment, Nunavut, 2019-20. Sample sizes shown above bars. Linear trendline added. 26

Figure 21. Mean proportion of wolf packs (≥2 wolves) harvested by community generated by QGIS assessment, Nunavut, 2020-21. Sample sizes shown above bars. 27

Figure 22. Mean proportion of wolf packs (≥2 wolves) harvested by month generated by QGIS assessment, Nunavut, 2020-21. Sample sizes shown above bars. Linear trendline added. 27

Figure 23. The proportion of wolf packs harvested by pack size (wolves observed) using a rules-based assessment, data combined for 2018-21. Points are jittered to increase interpretation. Predictions of mean percent harvested from logistic regression analysis with log(wolves observed) as a predictor are shown (red line). Confidence intervals are shaded around the prediction line. 28

Figure 24. The proportion of wolf packs harvested by date using a rules-based assessment for 3 communities across different harvest periods. Points are jittered to increase interpretation. Predictions of mean percent harvested from logistic regression analysis with date as a predictor with log(wolves observed) set at mean values (4 wolves; red lines). Confidence intervals are shaded around the prediction line..... 29

Figure 25. Collared caribou distribution and centers of activity, and wolf harvest collection locations, October 2018 to June 2021 (see legends for month and year). 42

Introduction

Enhanced wolf (*Canis lupus*) management actions are often employed to support recovery of declining caribou (*Rangifer tarandus*) herds. These actions may include hunter training, support or incentives for ground-based harvesting, sterilization, poisoning, and aerial shooting, among other wolf removal options (Adams et al. 2008, Farnell 2009, Russell 2010, Hayes 2013, McLaren 2016, Boertje et al. 2017, WFATWG 2017). Experience in many jurisdictions suggests that wolf removal must be greater than 55–60% of the population annually for ≥4 years and carried out over a large area to maintain reduced wolf densities sufficiently to elicit a demographic response in an ungulate population (National Research Council 1997, McLaren 2016, Nishi et al. 2020). These levels of wolf harvest are often difficult for ground-based harvesting efforts to attain (Webb et al. 2011, McLaren 2016). Aerial shooting (often coupled with collaring of ≥1 member of a pack) has been effective in some areas (Russell 2010, Hayes 2013, McLaren 2016), but can be compromised by comparatively low wolf densities and difficulties in locating wolves, and high costs (WFATWG 2017, Nishi et al. 2020). Predator management strategies on migratory wolves following migratory herds are even more challenging (McLaren 2016). By late fall when young wolves are large and strong enough to travel with the pack, most wolves within the range of migratory caribou herds resume their close association with the migratory caribou and maintain it throughout winter (Musiani et al. 2007, Clark et al. 2021).

Most herds of migratory tundra caribou (*R. t. groenlandicus*) in Nunavut and adjacent Northwest Territories (NWT) are either in decline or at historically low population levels (COSEWIC 2016). Evidence from traditional and scientific sources indicates that caribou herds periodically increase and decrease at some interval (COSEWIC 2016, Bongelli et al. 2020). Although not fully understood, cyclical changes are likely largely due to trends in range quality, habitat disturbance (i.e., forest fires), climate, predation, disease and anthropogenic influences (COSEWIC 2016). Wolf predation is often singled out as a significant contributing factor (Hayes and Russell 2000, Russell 2010). Wolves are known to predate both adults and calves (Bergerud et al. 2008), and managing the wolf populations to reduce wolf predation on caribou is one of the few management levers available.

To support recovery of the Bathurst and Bluenose-East caribou herds, the Government of the NWT (GNWT) Environment and Climate Change (ECC) and Tłı̨chǫ Government (TG) began an aerial shooting program to supplement community wolf harvest incentive programs in late winter 2019-20, but the effectiveness of this program to reach wolf reduction targets (60–80% removal levels) within winter range has yet to be clarified (Nishi et al. 2020). Beginning in 2018-19 in the Kitikmeot Region and expanding to all of Nunavut in 2019-20, the Government of Nunavut (GN) Department of Environment (ENV) through the Support for Active Harvesters program offered compensation to local hunters to harvest wolves and provide samples and location data. Nearly 150 wolves were harvested by local hunters in the Kitikmeot in 2018-19, and over 650 and about 700 wolves were harvested Nunavut-wide during 2019-20 and 2020-21, respectively, the vast majority from within the Kitikmeot and Kivalliq regions. Through sample collection forms, carcass examination, and age analysis, we determined the location, date, sex and age of these harvested wolves. These data, coupled with GN and GNWT collar data of caribou herds in Nunavut, provide an opportunity to examine the spatial and temporal patterns of the wolf harvest in relation to caribou herd movements and distance from communities. Nunavut hunters are in a unique position as

they are able to effectively hunt wolves on good snow conditions in the open tundra, as opposed to within the treeline where soft snow and forests often hamper hunter travel and effectiveness.

Although most migratory caribou herds within mainland NWT and Nunavut have shown steep rates of decline and low abundance, for example the Bathurst and Bluenose-East herds (Adamczewski et al. 2022, Boulanger et al. 2022), the Qamanirjuaq herd in the Kivalliq Region of Nunavut has instead demonstrated a moderate rate of decline of 2% based on the 2017 calving ground survey (Boulanger et al. 2018). Of key interest is whether relative stability of the Qamanirjuaq herd can be correlated to the more intensive wolf harvest measures undertaken on this herds' range.

The objectives of this analysis were:

1. Using GIS, describe the spatial and temporal patterns of the wolf harvest in Nunavut in relation to inferences of caribou herd locations/densities from collared caribou and distance from communities;
2. Temporally document the age and sex structure of the wolf harvest, including examination of spatial and temporal clustering of the harvest (i.e., whether hunting is effective at removing entire packs); and
3. Estimate shifts in age structure of wolves harvested as an index of the effectiveness of the ground-based wolf control program.

Study Area

While the Support for Active Harvesters program applied to all of Nunavut, our analysis of wolf harvest locations in relation to caribou distribution focussed on the Kitikmeot and Kivalliq regions within Nunavut, primarily mainland Nunavut and Victoria Island (Fig. 1). Other islands and peninsulas adjacent to and north of the Nunavut mainland (e.g., King William Island and Boothia Peninsula) had no supporting caribou collar data. Comparatively few wolves were harvested in the Qikiqtaaluk region during the study and there was no reported harvest on Baffin Island.

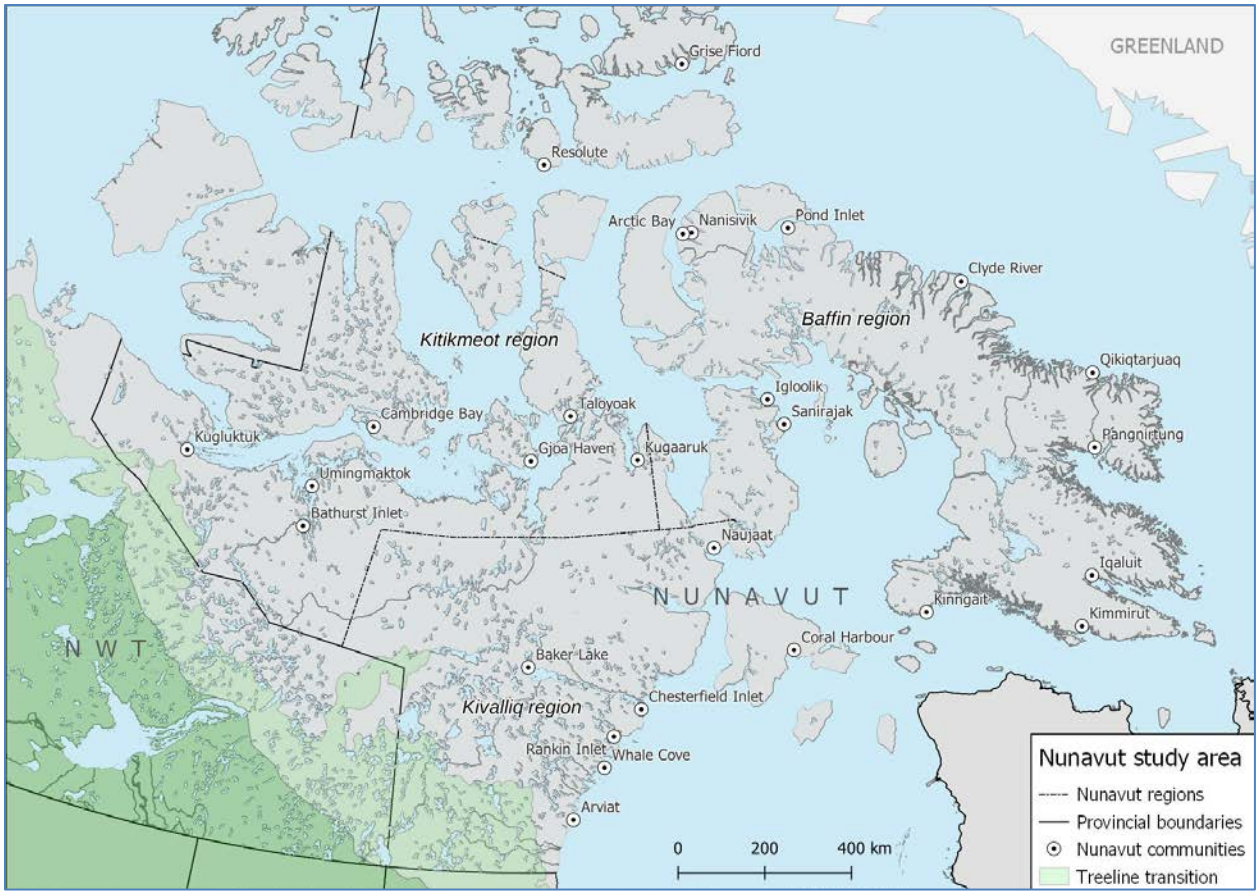


Figure 1. Map of Nunavut and communities; the Support for Active Harvesters program applies to the entire territory.

Study Design and Methodology

Spatial data acquisition

Wolf harvest data

Wolves occupy almost all of Nunavut, and are classified as both a furbearer and a big game mammal under the Nunavut Agreement (https://www.tc.canada.ca/sites/default/files/migrated/nunavut_land_claims_agreement.pdf). Under the Nunavut Agreement, furbearer harvest privileges are held by Inuit beneficiaries, non-Inuit who harvested furbearers legally in the Nunavut settlement area prior to 1981, and non-Inuit whose application has been approved by local Hunters and Trappers Organizations (HTOs). Unlike most provinces, hunters and trappers in Nunavut do not have registered or traditionally exclusive family trap lines or hunting areas.

Wolf hunting in Nunavut is mainly by firearms from snowmobiles, with few wolves trapped. With a very limited road network in the territory, snowmobiles are a vital form of transportation which hunters use to harvest wolves because it is the only practical method given the wide-open spaces of the Nunavut tundra. Early in the season (October to February) most hunters harvest wolves on day or weekend trips

while harvesting caribou or muskoxen (*Ovibos moschatus*). From March onward hunter groups generally take trips 1–2 weeks in length specifically for wolf hunting.

There are no estimates of wolf numbers within Nunavut. Both migratory and resident wolves occur within the territory, with most individuals moving with migrating caribou herds. Additional large prey for wolves includes populations of muskoxen which are expanding throughout most of the mainland (with lower densities further east and northeast within Nunavut; Campbell and Lee 2019) and moose (*Alces alces*) around Bathurst Inlet and along the treeline, areas supporting quantities of shrubs. Other carnivores in the area include Arctic fox (*Vulpes lagopus*), red fox (*V. vulpes*), wolverine (*Gulo gulo*), grizzly bear (*Ursus arctos*) and black bear (*U. americanus*) along the treeline. Polar bears (*U. maritimus*) occur along coastal areas.

Each year we obtained the skull and a small piece of skin (~2.5 x 2.5 cm) with fur from hunters with the assistance of HTOs and Conservation Officers (CO). A $300 subsidy was provided to hunters for each skull brought back to their wildlife office to encourage the reporting and return of samples. For each wolf sample collected during the course of this study, COs filled out sample collection forms with information from hunters about their name (known only to the senior author) and home community, the harvest date, location, wolf sex, and method of harvest, and their opinion about the current abundance trend of the local wolf population: "Do you think the wolf population is decreasing, stable or increasing". The skinned skulls were stored frozen, and were thawed at room temperature for examination in early May each year. We sent the first premolar to Matson's Laboratory LLC (Milltown, MT, USA) for age estimation using cementum analysis. This technique is based on the cyclic nature of cementum growth in teeth forming annular patterns of different darkness depending on the season (Matson 1981). Following Mech (1970) and Gese and Mech (1991), individuals were then grouped into three age classes: juvenile (0-year-old), yearling (1-year-old) and adult (≥2 years).

Kitikmeot 2018-19: To support harvesting activities and contribute to the knowledge of wolf ecology and species management in Nunavut, ENV launched a pilot project of skull collections within the Kitikmeot Region as part of the Support for Active Harvesters Program and in response to community concerns, to provide samples for scientific research and examine wolf harvest effort and use of wolf harvest as a management tool in caribou recovery. There was no limit to the number of wolves submitted.

Nunavut 2019-20 and 2020-21: The Kitikmeot program was expanded Nunavut-wide in September 2019, again with no limit to the number of skulls submitted. Nunavut hunters killing wolves within the NWT North Slave Wolf Harvest Incentive Area received an additional $900 payment from the GNWT.

Caribou collar data

Caribou collar locations from October 2018 to June 2021 from collars deployed by ENV were obtained for the Qamanirjuaq (QM), Lorillard (LR), Wager Bay (WB), Ahiak (AH) and Dolphin and Union (DU) herds within Nunavut (Caslys Consulting Ltd., Saanichton, BC). Collar data from the DU herd ran from April 2015 to early December 2020, and resumed in April 2021 with additional collaring. Collars were deployed by ECC on the Bluenose-East (BE), Bathurst (BA) and Beverly/Ahiak (BV) herds (ENR 2021). Here we consider the Beverly and Beverly/Ahiak herds to be synonymous. Within the Nunavut data set all animals with Herd = UK designation were visually confirmed as QM caribou. Within the ECC data set, 13 of 18 caribou with

Herd = NA (not available) were assigned to herd based on movement paths or association with other collared caribou. Both sexes were represented in the collar data, with increasing numbers of bulls collared after 2015.

Collar fix rate varied, with most collars providing from 1 to several GPS locations daily. Over 1.5 million caribou collar locations were obtained. To facilitate GIS computations, to prevent more frequent fixes from skewing (unduly weighting) collar density maps, and because we did not require fine-scale temporal resolution to the data, we thinned/rarefied the data sets to one location daily and subset the collar data sets to cover the period from October 2018 to June 2021 (covering the 3 years of wolf harvest data). The ECC collar data had time and datetime stamp but Nunavut and DU data did not have a time component.

Data preparation

We conducted spatial analysis in QGIS (QGIS Development Team 2020), GRASS (GRASS Development Team 2020) and MapInfo (MapInfo 2000) using the Canada Lambert Conformal Conic (LCC) projection. We first conducted telemetry density (heat) mapping (kernel density estimation; Downs and Horner 2012) to display monthly patterns of caribou distribution. To determine heat map kernel radius, we estimated the median daily movements among the Nunavut and ECC data sets by month:

1. We added LLC coordinates to the files.
2. We sorted the data sets by caribou identification and date, and calculated the distance (in metres) between successive locations.
3. We calculated the number of days between successive fixes and applied a filter to remove all times >1 day, which accounts for missed fixes where a daily location was not obtained.
4. We calculated the mean distance moved per day by animal and month.
5. We calculated the animal sample size and median distance moved by month. Not unexpectedly, the median daily movement distances differed somewhat between the Nunavut and ECC data sets, and we used the average between data sets for heat mapping (Fig. 2).

Caribou herds often overlapped spatially and temporally; thus, the caribou heat map was run on the combined collar data set from all herds. However, sample sizes of collars varied among herds and over time as does the recent population estimate for each herd. For example, in December 2018 there were 27 collars on BA caribou, out of a population of approximately 8,200 (Adamczewski et al. 2019), thus each collar “represented” about 300 caribou. In contrast, there were 49 collars on QM caribou out of an estimated 288,000 caribou (Boulanger et al. 2018) – roughly one collar for every 5,900 caribou. To account for these differences, we weighted each caribou collar location by a relative factor of representativeness, ranging from 1 (DU or BA herds) to a high of 87 for the QM herd, based on the number of collared individuals and the most recent herd estimate (Appendix 1).

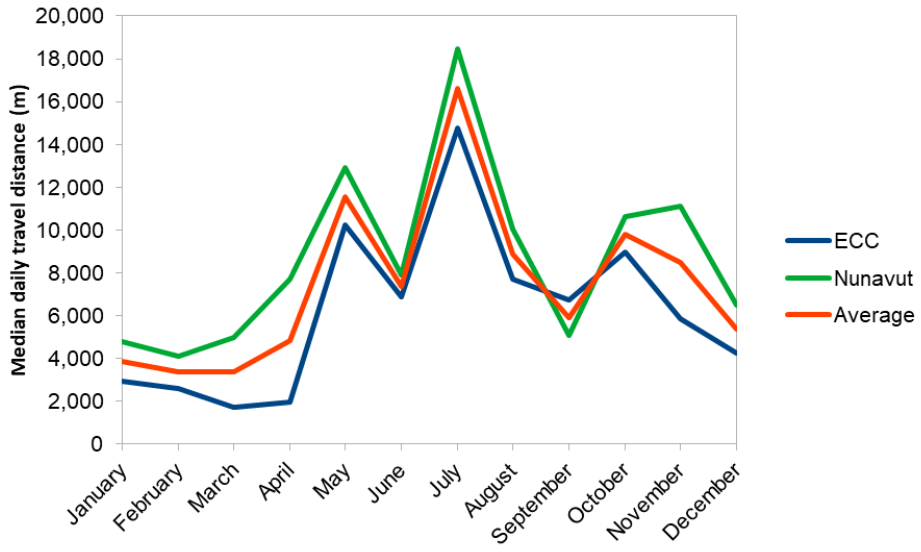


Figure 2. Median daily movement distances (m) by caribou for the GNWT Environment and Climate Change (ECC), and Government of Nunavut Department of Environment (Nunavut) data sets.

We used heat mapping to demonstrate the relative distribution of caribou collars on a monthly basis using the monthly median for kernel radius and Epanechnikov kernel shape (QGIS Heatmap plugin). To highlight areas where caribou collar density (and presumably caribou distribution) was greatest we mapped centers of activity as collar density above the mean map average (the average density across all mapped caribou distribution).

We produced monthly maps showing caribou distribution and the distribution of reported wolf harvests. Given the scale of mapping, adjacent wolf kill locations within ~25 km were shown on the maps as a single dot representing the cumulative number of wolves harvested.

For each reported wolf kill location we calculated:

1. The distance to the nearest caribou heat map boundary (the mapped boundary between caribou distribution and no caribou);
2. The distance to the nearest caribou center of activity (collar densities greater than the map average); and
3. Distance to the community from the kill data, where the hunting party originated.

To demonstrate the distribution of caribou herds in relation to the wolf harvest we also produced a single month's map from all seasons (March 2019, April 2020 and April 2021) using un-weighted heat mapping for each herd. We used a 10-km kernel for each herd.

Proportion of wolf packs harvested

We examined the spatial and temporal clustering of the harvest in an effort to assess whether hunting was effective at removing entire packs. We used two methods to examine this question. In the "QGIS assessment" method we examined the locations and dates of wolf harvest by community. Within the database we used reported data on whether the harvested wolf was alone, the number of individual

wolves observed and comments to determine what proportion of entire groups were removed, as well as spatial and temporal associations to better interpret the data. For example, if two different wolf harvests on the same day within reasonable proximity to each other (often <5 km to a maximum of 15 km) both said pack size was two, we assumed the entire pack of two was removed. To a large extent this depended on accurate reporting of wolf harvest locations and whether the number of individual wolves in the pack was recorded. If no pack size was recorded the number of wolves in the pack prior to harvest was unknown.

In addition to the visual grouping, we also conducted a "rules-based" date and distance data assessment to 'assign' pack size using a hierarchical approach to group observations, initially grouping by community, date and location, assigning 6 apparent lone wolves to pairs (same date and latitude-longitude). We found some issues with the data where observed pack size and whether more than a lone wolf was observed and harvested were not consistently recorded. In addition, observations of harvest on the same day within a 5–10 km were considered a pack. If no observations of pack size were noted then the number observed was set equal to the number harvested for a given group. This may underestimate the number observed, however, again, it was not possible to evaluate this potential bias with the information available. We acknowledge that this approach might not classify packs or all harvest events correctly. For example, if a pack was hunted over a long distance with different latitude-longitude for each kill then it is possible that the pack would be divided into multiple harvest groups. As discussed later, to mitigate these issues we suggest that future data collection should include a "pack ID" of similar on each set of records in the database to delineate whether the same group of wolves was targeted among records.

Logistic regression (Hosmer and Lemeshow 2000) was used to test for factors that affected harvest success, with harvest success estimated as wolves harvested divided by wolves observed. An event/trials input format was used (i.e., wolves harvested/wolves observed) for logistic regression models, therefore accounting for different sample sizes of wolves observed in the estimation of percent harvested. Logistic regression analysis was conducted using the *glm* function included in program R (R Development Core Team 2009). Results from all analyses were plotted using the *ggplot2* package (Wickham 2009) with GIS analyses conducted using the *sf* package (Pebesma 2018) in *R.* Data manipulation was conducted using the *dplyr* (Wickham 2011) and *lubridate* (Grolemund and Wickham 2011) *R* packages.

Results

Wolf harvest

Seasonal caribou movements generally matched known patterns of calving in northern and eastern mainland Nunavut followed by movements south and west away from calving areas towards and into treeline for the winter (see monthly heat maps in Appendix 2). Collared DU caribou mostly calved and summered on Victoria Island and wintered on the Nunavut mainland.

Kitikmeot 2018-19

During 2018-19, 146 wolves were recorded harvested within the Kitikmeot, primarily by hunters from Kugluktuk (69% of the regional harvest; Table 1). Two main concentrations of harvest were detected from Kugluktuk hunters, around Napaktulik (Takijuq) Lake (*n* = 41) and around the northern portion of Contwoyto Lake (*n* = 44; Fig. 3). Cambridge Bay hunters primarily harvested wolves on the south coast of Victoria Island <100 km from the community and on Kent Peninsula. All wolves were reported as shot.

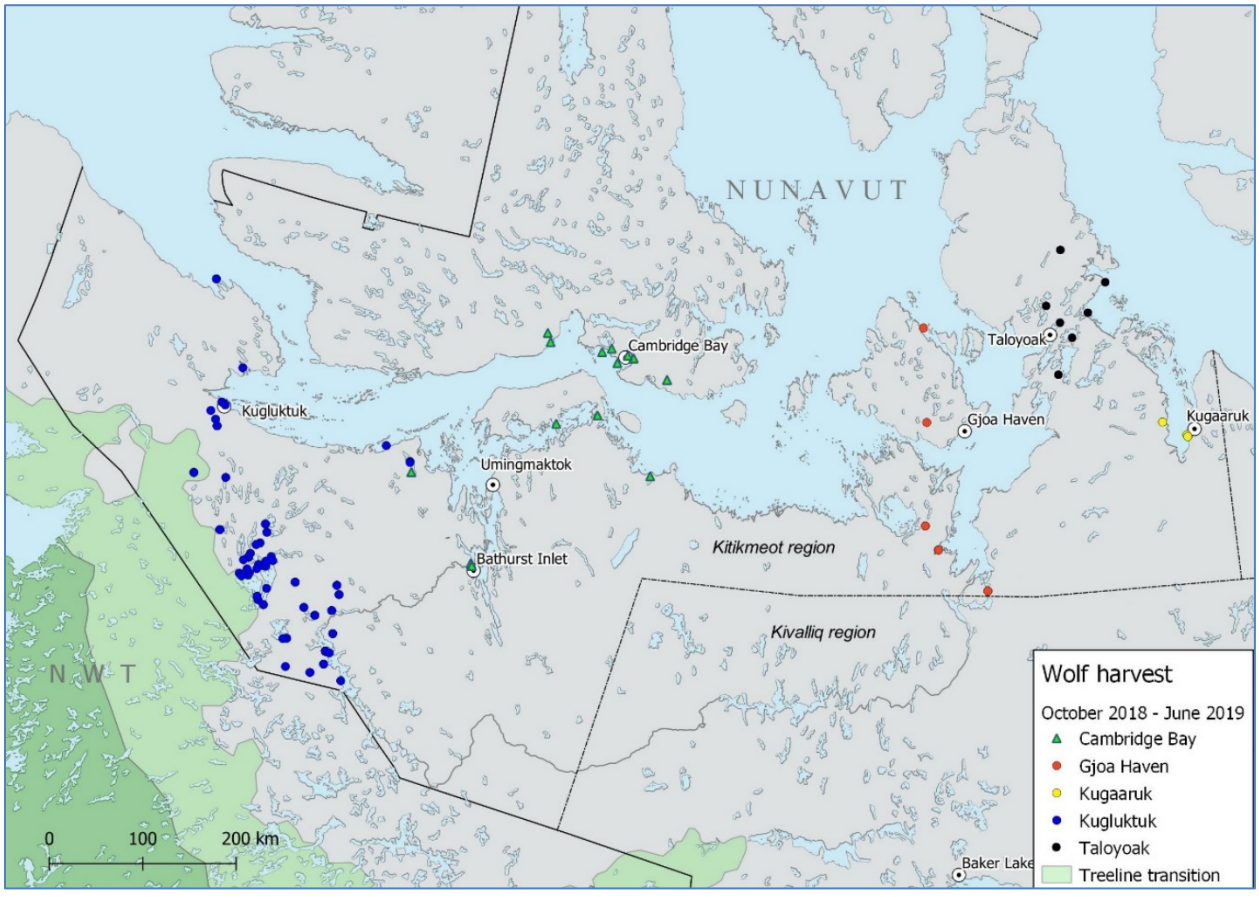


Figure 3. Distribution of reported wolf harvest in the Kitikmeot Region, 2018-19. Symbols may represent >1 wolf harvested at that location.

Table 1. Number and sex of recorded wolf harvests by community, Kitikmeot Region, 2018-19.

Community	Female	Male	Unknown	Total
Kugluktuk	36	50	15	101
Cambridge Bay	10	11	0	21
Gjoa Haven	2	11	0	13
Taloyoak	1	5	2	8
Kugaaruk	2	1	0	3
Total	51	78	17	146

A total of 52 hunters reported harvesting wolves. A single hunter from Kugluktuk (resident of an outpost camp) accounted for 32% of the regions' harvest (Table 2).

Table 2. Distribution of wolf harvest by individual hunters, Kitikmeot Region, 2018-19.

Number of wolves	1	2	3	4	5	6	12	46
Number of hunters	31	11	2	2	3	1	1	1
% of harvest	21	15	4	6	10	4	8	32

Four wolves were recorded as harvested in October and November 2018 (all relatively closes to communities), with increased harvest in December (Table 3). Nearly three-quarters (71%) of the regional harvest occurred during February to May.

Table 3. Distribution (*n* = 146), mean age (*n* = 139) and proportion of males (*n* = 128) in the monthly wolf harvest, Kitikmeot Region, 2018-19.

	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
Number	2	2	21	11	31	25	27	21	6
Age (yrs)									
Mean	0.5	1.0	1.7	2.8	3.5	3.6	1.5	3.0	2.2
SE	0.49		0.52	0.60	0.47	0.52	0.35	0.48	0.90
n	2	1	21	10	29	25	25	20	6
Sex									
% males	100	100	67	55	52	60	52	68	100
n	2	2	6	11	31	25	27	19	5

Males comprised 60% of the harvest, with a slight peak in males in the harvest early and late in the season (Table 3). The age distribution of harvested wolves (*n* = 139) changed as the season progressed with juveniles comprising 54% of the harvest during October to December, with higher proportions of 1–3-

year-olds later in the season (Table 3, Fig. 4). Juveniles were heavily harvested early in the season and a wider distribution of ages harvested as the season progressed. The mean age of harvested wolves peaked in February and March (Table 3).

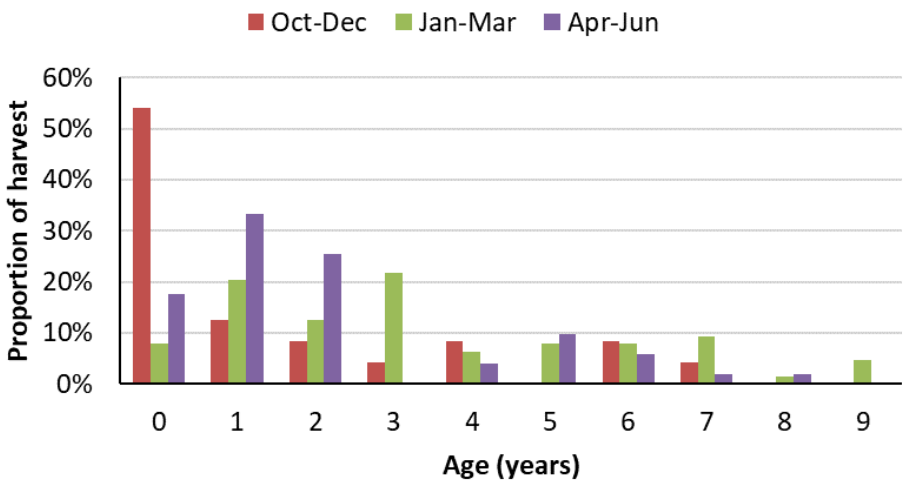


Figure 4. Distribution of age of harvested wolves by 3-month period, Kitikmeot Region, 2018-19 (*n* = 139).

Nunavut 2019-20

During 2019-20, 658 wolves were recorded harvested within Nunavut, with 64% of the harvest from Arviat, Baker Lake and Kugluktuk hunters (Fig. 5; Table 4). Only 34 wolves were reported harvested within the Qikiqtaaluk region (within the High Arctic), 5% of the Nunavut collection. Concentrations of harvested wolves included west and southwest of Arviat, near Dubawnt Lake, about 350 km west of Arviat, 50 km north of Baker Lake, and north of Aberdeen Lake. Rankin Inlet hunters occasionally travelled to meet the returning QM herd in late spring, to areas traditionally hunted by Arviat hunters. The distribution of peak harvest was bi-modal, with a small peak in November and a higher peak in March-May (Table 5). Six of the wolves (1%) were reported as trapped (4 of these from Baker Lake) and the remainder were shot.

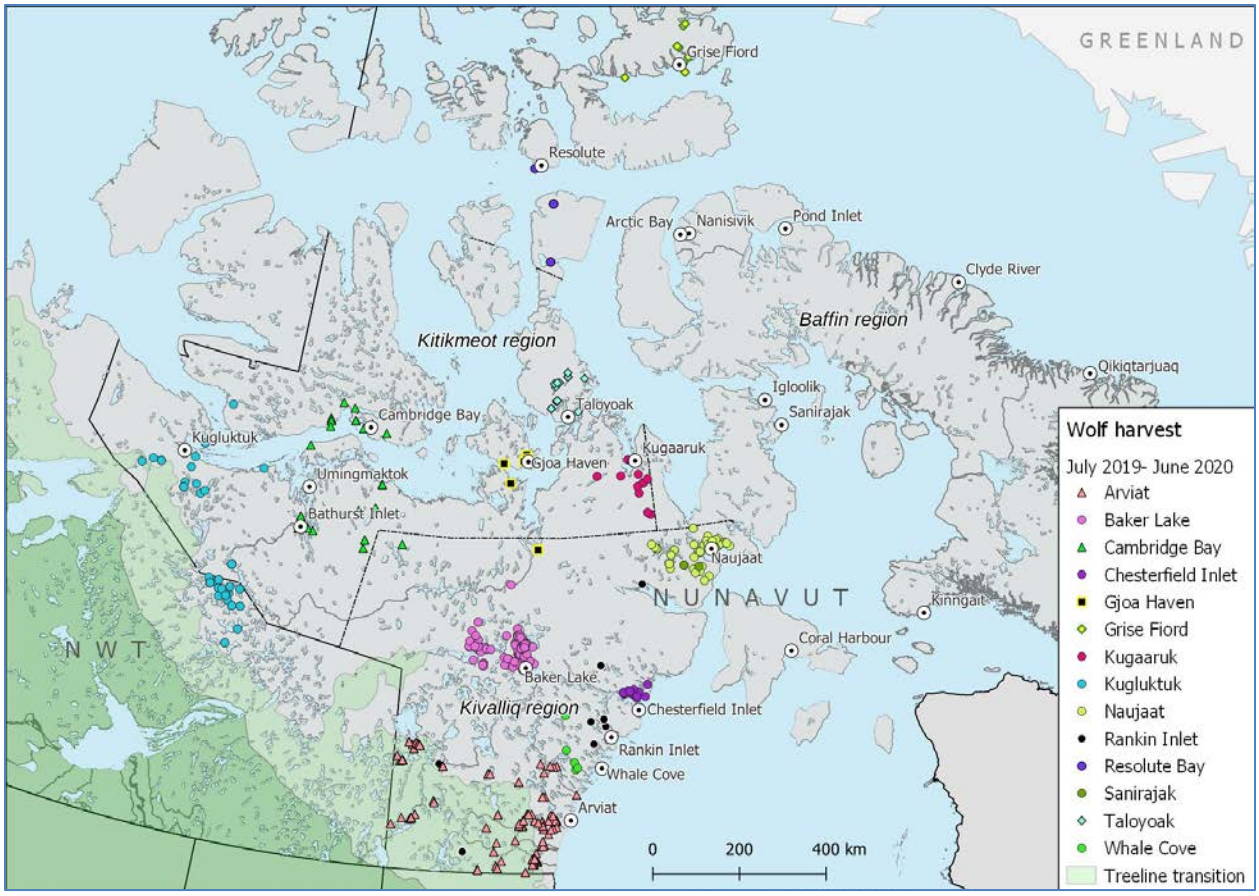


Figure 5. Distribution of reported wolf harvest in Nunavut, 2019-20. Symbols may represent >1 wolf harvested at that location.

Table 4. Number and sex of recorded wolf harvests by community, Nunavut, 2019-20.

Region	Community	Female	Male	Unknown	Total
Kitikmeot		**69**	**83**	**1**	**155**
	Cambridge Bay	10	25		35
	Gjoa Haven	4	5	1	10
	Kugaaruk	2	13		15
	Kugluktuk	29	35		64
	Taloyoak	24	5		29
Kivalliq		**188**	**226**	**57**	**471**
	Arviat	70	97	57	224
	Baker Lake	63	67		130
	Chesterfield Inlet	10	8		18
	Naujaat	28	30		58
	Rankin Inlet	13	23		36
	Whale Cove	4	1		5
Qikiqtaaluk		**13**	**18**	**3**	**34**
	Grise Fiord	1	7	3	11
	Sanirajak	2	1		3
	Resolute Bay	10	10		20
	Total	**270**	**327**	**61**	**658**

Table 5. Distribution (*n* = 653), mean age (*n* = 604) and proportion of males (*n* = 592) in the monthly wolf harvest, Nunavut, 2019-20.

	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
No.	2	2	7	17	95	41	39	53	128	182	83	4
Age (yrs)												
Mean	4.0	0.5	0.5	1.7	1.3	1.3	0.9	1.5	1.5	1.4	1.8	1.0
SE	3.00	0.50	0.50	0.55	0.24	0.27	0.24	0.27	0.18	0.15	0.19	0.00
n	2	2	4	17	93	40	39	52	116	174	63	2
Sex												
% males	100	100	86	47	54	34	69	51	54	60	55	25
n	1	2	7	17	95	41	39	53	113	145	75	4

Male wolves comprised 55% of the Nunavut harvest in 2019-20, with higher proportions of the males in the harvest from January to May (Table 5). The age distribution of younger harvested wolves changed as the season progressed with juveniles comprising 53% of the harvest during October to December and declining, and yearlings comprising 23% of the harvest and increasing (Fig. 6). The proportion of harvested adult wolves changed little as the season progressed. The mean age of harvested wolves peaked early and late in the winter (Table 5).

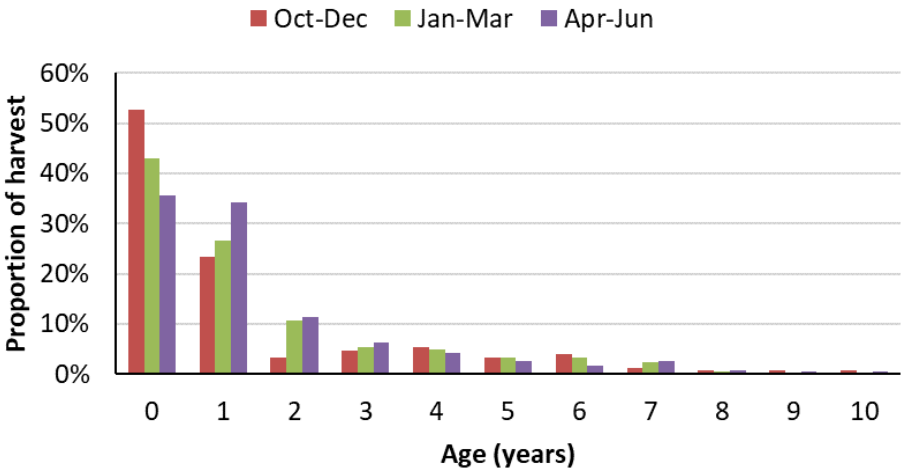


Figure 6. Distribution of age of harvested wolves by 3-month period, Nunavut, October 2019 – June 2020 (*n* = 596).

A total of 471 wolves were harvested by 123 hunters in the Kivalliq region during the 2019-20 season. (Table 6). Ten hunters (8%) each harvested over 10 wolves, contributing 38% of the wolf harvest, and 18 hunters (15%) took just over 50% of the wolves. A similar pattern was evident in the Kitikmeot region, with 8 (13%) of the 60 hunters harvesting 44% of the 153 wolves. Seven of the 21 hunters (33%) contributed nearly 60% of the 34 wolves harvested in the Qikiqtaaluk region.

Table 6. Distribution of wolf harvest by individual hunters by region, Nunavut, 2019-20.

Kitikmeot Region											
Number of wolves	1	2	3	4	5	6	7	8	11	18	
Number of hunters	36	6	4	5	1	4	1	1	1	1	
% of harvest	23	8	8	11	3	16	5	6	8	12	
Kivalliq Region											
Number of wolves	1	2	3	4	5	6	7	8	9	10	11-26
Number of hunters	49	22	16	5	13	2	1	1	2	2	10
% of harvest	10	9	10	4	14	3	1	2	4	4	38
Qikiqtaaluk Region											
Number of wolves	1	2	3	5							
Number of hunters	14	3	3	1							
% of harvest	41	18	26	15							

Nunavut 2020-21

During 2020-21, 699 wolves were recorded harvested within Nunavut, with 68% of the harvest from Arviat, Baker Lake and Kugluktuk hunters (Fig. 7; Table 7). Eight wolves were reported harvested within the Qikiqtaaluk region, 1% of the Nunavut total. Concentrations of harvested wolves included southwest of Arviat (but not as far west as in 2019-20), north, west and south of Baker Lake, and north of Point Lake in the NWT. Seven of the wolves (1%) were reported as trapped and the remainder were shot.

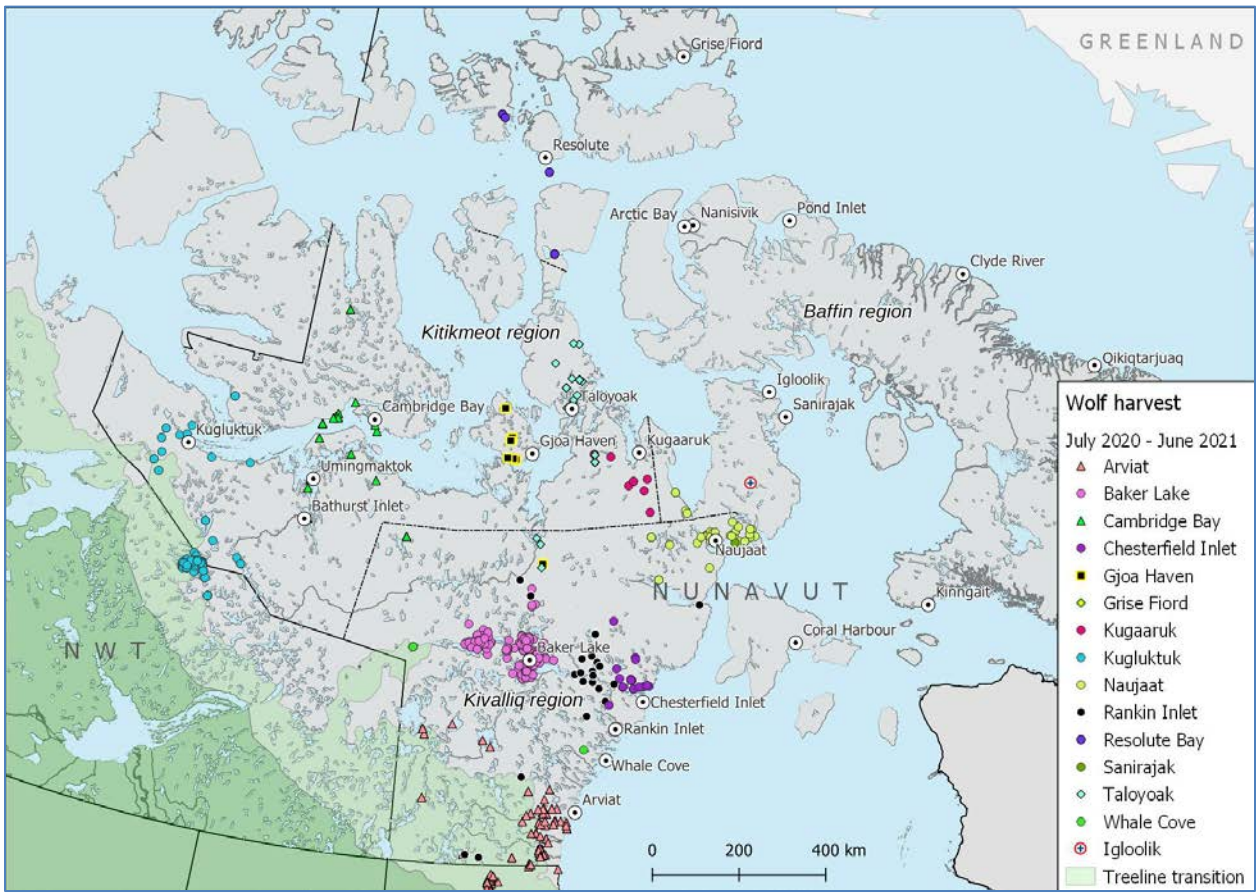


Figure 7. Distribution of reported wolf harvest in Nunavut, 2020-21. Symbols may represent >1 wolf harvested at that location.

Male wolves comprised 56% of the Nunavut harvest in 2020-21 (Table 7), with higher proportions of the males in the harvest in December and March (Table 8). The age distribution of younger harvested wolves changed as the season progressed with juveniles comprising 56% of the harvest during October to December and declining, and yearlings comprising 26% of the harvest and increasing (Fig. 8). The proportion of harvested 2-year-olds increased as the season progressed, and more 3–5-year-olds were harvested mid-winter. The mean age of harvested wolves peaked early and late in the winter (Table 8).

Table 7. Number and sex of recorded wolf harvests by community, Nunavut, 2020-21.

Region	Community	Female	Male	Unknown	Total
Kitikmeot		**84**	**120**	**10**	**214**
	Cambridge Bay	15	21		36
	Gjoa Haven	4	5		9
	Kugaaruk	5	7		12
	Kugluktuk	48	67	10	125
	Taloyoak	12	20		32
Kivalliq		**215**	**261**	**1**	**477**
	Arviat	58	74		132
	Baker Lake	108	113		221
	Chesterfield Inlet	8	15		23
	Naujaat	20	29	1	50
	Rankin Inlet	21	26		47
	Whale Cove		4		4
Qikiqtaaluk		**6**	**2**		**8**
	Sanirajak	1			1
	Igloolik		1		1
	Resolute Bay	5	1		6
	Total	**305**	**383**	**11**	**699**

Table 8. Distribution (*n* = 699), mean age (*n* = 599) and proportion of males (*n* = 685) in the monthly wolf harvest, Nunavut, 2020-21.

	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
No.	4	1	5	7	113	81	85	78	142	113	60	6
Age (yrs)												
Mean	1.3	8.0	1.8	0.4	1.1	1.3	0.8	0.9	1.4	0.8	2.1	0.8
SE	0.33	0.00	1.18	0.20	0.19	0.22	0.17	0.18	0.16	0.18	0.28	0.17
n	3	1	4	7	113	80	79	53	114	81	58	6
Sex												
% males	50	100	60	86	53	59	46	54	64	55	47	83
n	4	1	5	7	113	81	85	78	141	104	60	6

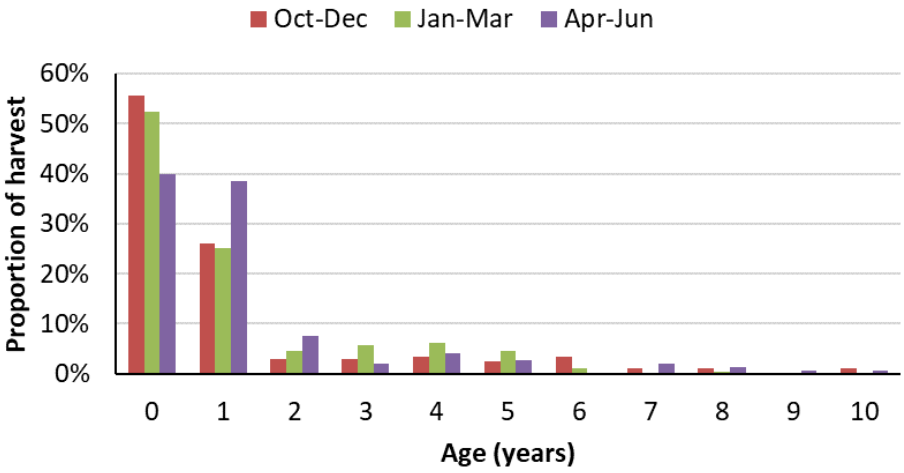


Figure 8. Distribution of age of harvested wolves by 3-month period, Nunavut, October 2020 – June 2021 (n = 591).

Summing among years, young wolves (aged 0–1 years) comprised 70–80% of the annual harvest in the Kitikmeot and Kivalliq regions in 2019-20 and 2020-21, increasing from 43% of the Kitikmeot harvest in 2018-19 (Fig. 9).

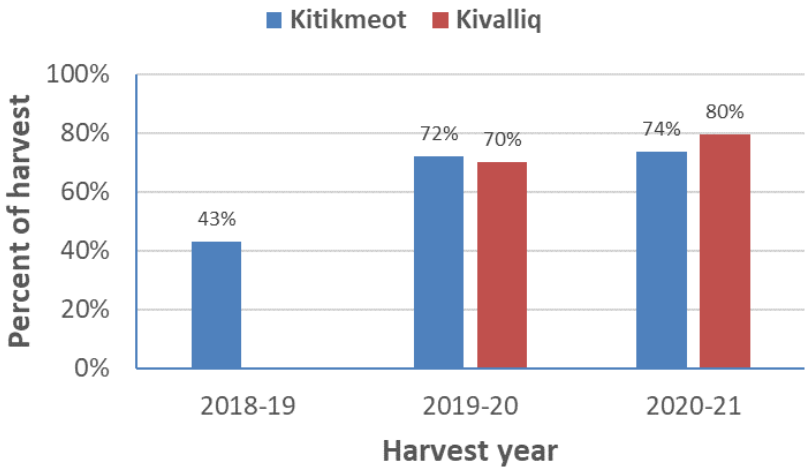


Figure 9. Proportion of young wolves within the annual harvest by Nunavut region, 2019-21.

A total of 477 wolves were harvested by 122 hunters in the Kivalliq region during the 2020-21 season. (Table 9). Nine hunters (7%) each harvested from 11-54 wolves, contributing 38% of the wolf harvest. One hunter from Baker Lake harvested 54 wolves, fully 11% of the regions harvest. In the Kitikmeot region, 4 (6%) of the 64 hunters harvesting 39% of the 214 wolves harvested, and one hunter accounted for 12% of the regions harvest.

Table 9. Distribution of wolf harvest by individual hunters by region, Nunavut, 2020-21.

Kitikmeot Region											
Number of wolves	1	2	3	4	5	6	8	10	11	26	
Number of hunters	31	7	13	3	1	2	2	1	2	1	
% of harvest	14	7	18	6	2	6	7	5	10	12	
Kivalliq Region											
Number of wolves	1	2	3	4	5	6	7	8	9	10	11-54
Number of hunters	51	22	16	8	1	8	1	4	2	0	9
% of harvest	11	9	10	7	1	10	1	7	4	0	38
Qikiqtaaluk Region											
Number of wolves	1	2									
Number of hunters	4	1									
% of harvest	67	33									

Harvesters' perception of wolf population trends

Harvesters' perception about trends in wolf populations differed among regions (χ^2 $P < 0.001$), but almost all believed that numbers were either stable or increasing (Table 10). About 16% more respondents believed wolf numbers were increasing in the Kitikmeot compared with the Kivalliq region. Note that the sample size does not reflect individual hunters, since a hunter may provide >1 harvested wolf to the program within and among years.

Table 10. Perception of wolf population trend (%) of wolf hunters providing carcasses for the Support for Active Harvesters program, 2018-19 – 2020-21.

Region	*n*	Decreasing	Stable	Increasing
Kitikmeot	504	0	40	60
Kivalliq	921	3	53	44
Qikiqtaaluk	24	0	33	67

Wolf harvest relative to distance from communities and caribou

Wolves were generally harvested closer to communities prior to Christmas (generally within 75 km) and further from communities later in the winter, with peak distances in April and May (Fig. 10). This pattern was less evident in 2018-19 within the Kitikmeot, where the harvest came predominantly from Kugluktuk

hunters. However, in 2018-19 much of the Kugluktuk wolf harvest came from an outpost camp hunting near caribou concentrations around Napaktulik and Contwoyto lakes (M. Awan, unpubl. data).

Spatial and temporal patterns of wolf harvest differed among communities. Arviat hunters harvested wolves close to the community during October–December, and over a broad distance later in the winter, including over 450 km from the hamlet in 2019-20. Baker Lake showed a similar pattern at a reduced scale, harvesting within 100 km prior to Christmas (largely along the Meadowbank all-weather access road north of the hamlet), and further away during mid to late winter. Hunters from Kugluktuk largely harvested wolves further from the hamlet during the winter, depending on the winter distribution of caribou herds.

These patterns of harvest generally reflected the patterns of movement of higher densities of caribou (Appendix 2) and the restricted travel conditions early in the year.

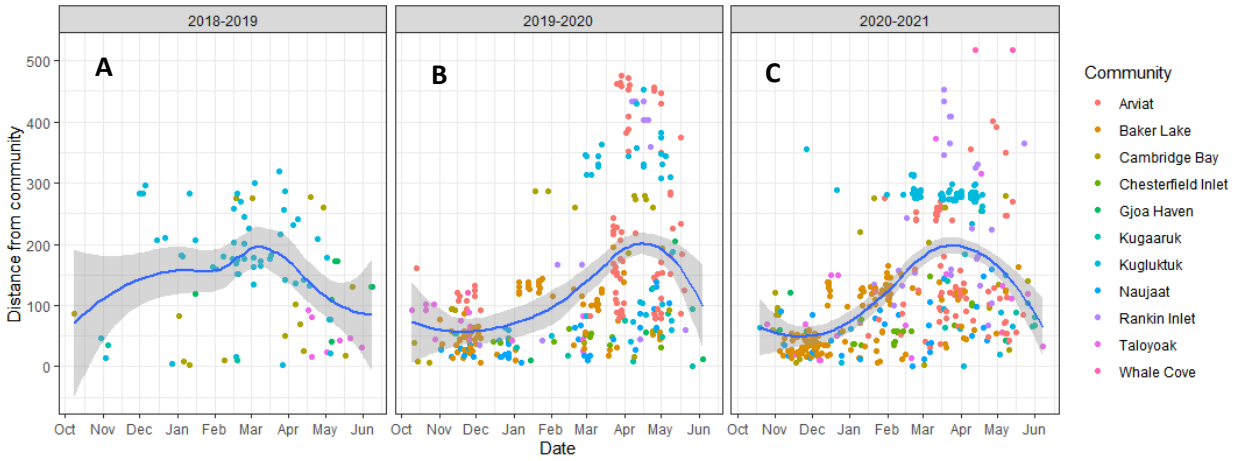


Figure 10. Wolf harvest locations plotted by distance from communities (km) during the A) 2018-19 (Kitikmeot), B) 2019-20 (Nunavut) and C) 2020-21 (Nunavut) harvest seasons. Smoothed lines provide an estimate of overall trend in the data set.

Hunters travelled up to 500 km from their communities to harvest wolves, but generally harvested wolves within 200 km of higher densities of collared caribou (Fig. 11). The communities that harvested wolves >200 km from higher density caribou occurrence were located where caribou collaring and hence caribou distribution within Nunavut were likely incomplete, mainly within the northeast Nunavut mainland: Kugaaruk, Naujaat, Taloyoak and Gjoa Haven, and to a lesser extent Cambridge Bay.

Estimated pack size and numbers of wolves harvested were higher closer to higher densities of caribou (Figs. 12, 13). Thus, even though the hunters were far from their communities, necessitating camping over day-trips, they encountered larger packs of wolves and were effective at harvesting larger number of wolves. Hunters from the communities of Kugaaruk, Naujaat, and Taloyoak did not follow this pattern, again likely related to limited collaring of caribou in those areas.

Communities such as Arviat and Kugluktuk, whose hunters move to areas with higher densities of caribou, were able to harvest large numbers of wolves close to those caribou. A spatial plot of this data set by year and distance from higher densities of caribou showed that areas of higher wolf harvest were often further from communities (Fig. 14). An exception appeared to be Baker Lake where most wolf harvest occurred within 150 km of the community.

In many cases the harvested wolves could be reliably "assigned" to a particular caribou herd when herd overlap was limited, but in some cases overlap of collared caribou from different herds meant assigning caribou herds to the wolf harvest was uncertain and problematic (Figs. 15, 16, 17).

Hunters appeared to focus their wolf harvest efforts where caribou densities were high. For example, the QM caribou herd was further from Arviat in April 2020 compared to April 2021, and the distribution of the wolf harvest mirrored this pattern (Figs. 16, 17).

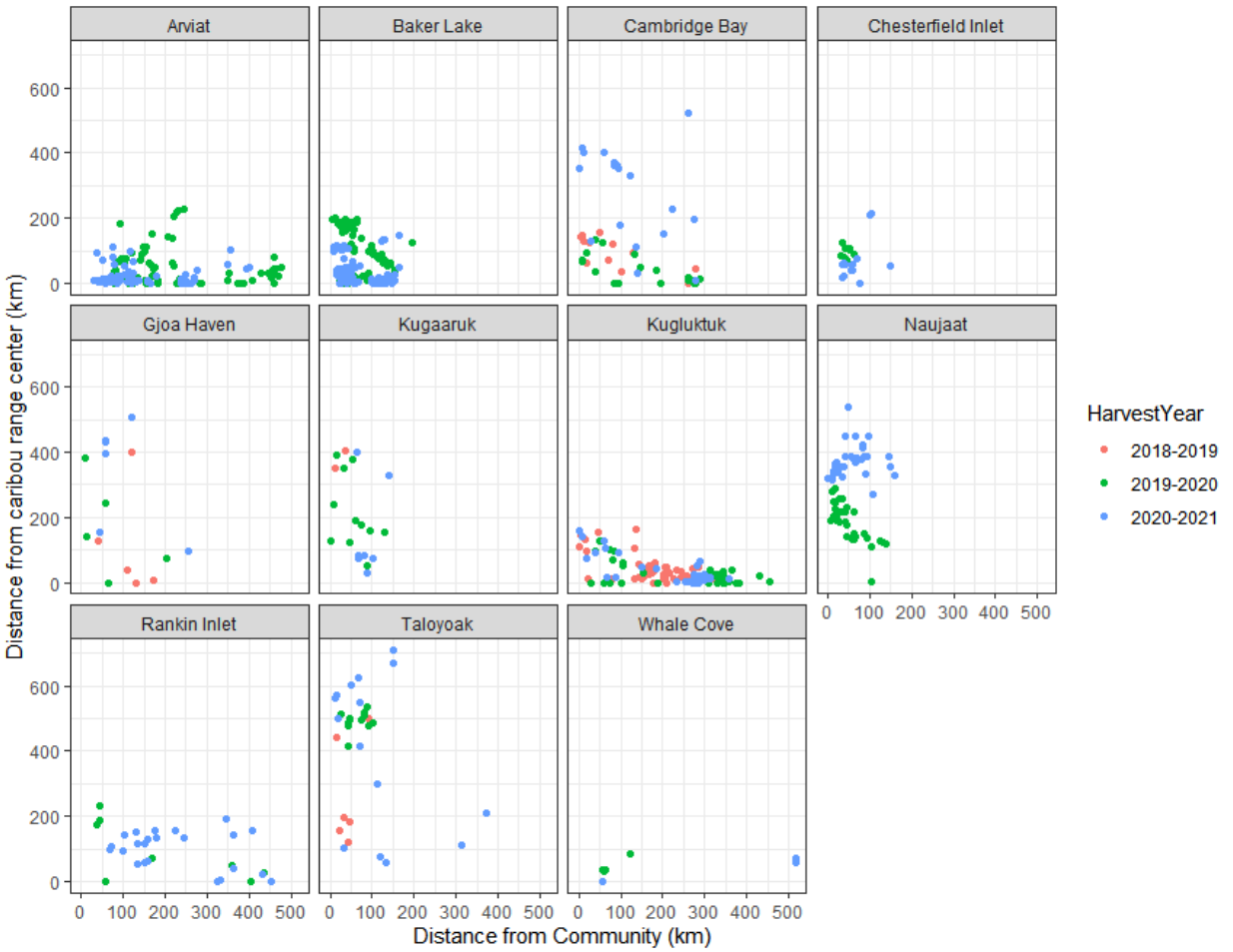


Figure 11. Relationship between wolf harvest distance from caribou centers of activity (higher densities) and communities, Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21.

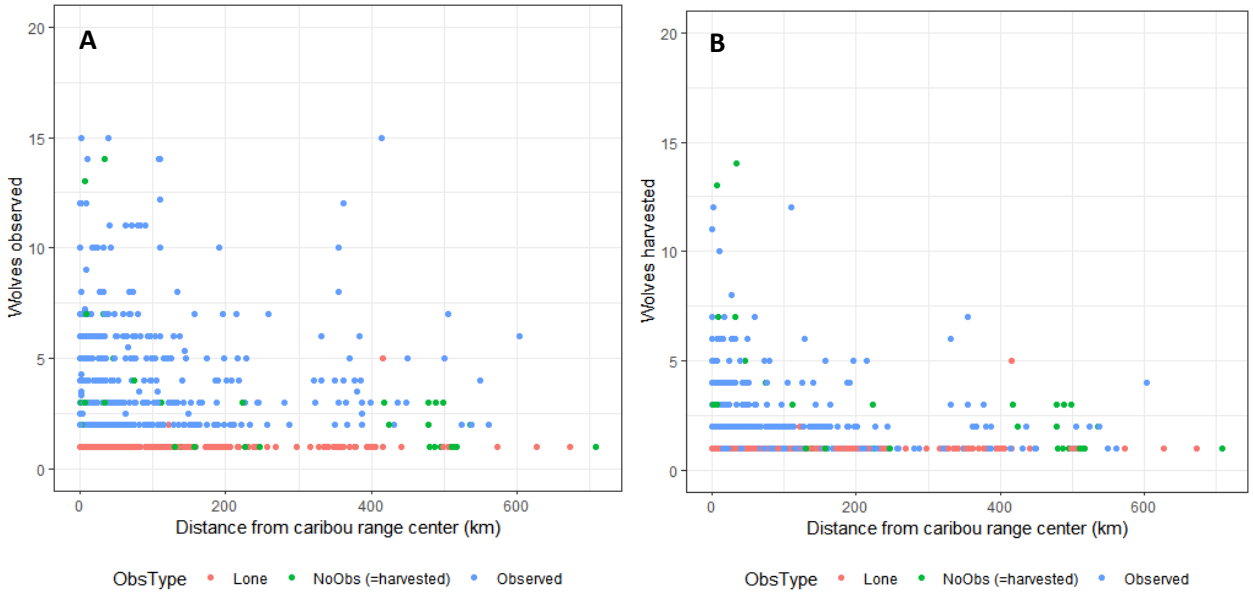


Figure 12. Number of wolves observed (A) and number of wolves harvested (B) relative to the distance from caribou centers of activity (higher densities), Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21 data combined. "Lone" = no group size reported but harvested at the same location on the same date; "NoObs" = no group size reported but harvested ≤5 km apart on the same date; "Observed" = pack size observation.



Figure 13. Number of wolves harvested related to distance from community and binned distances from higher densities of caribou, Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21.

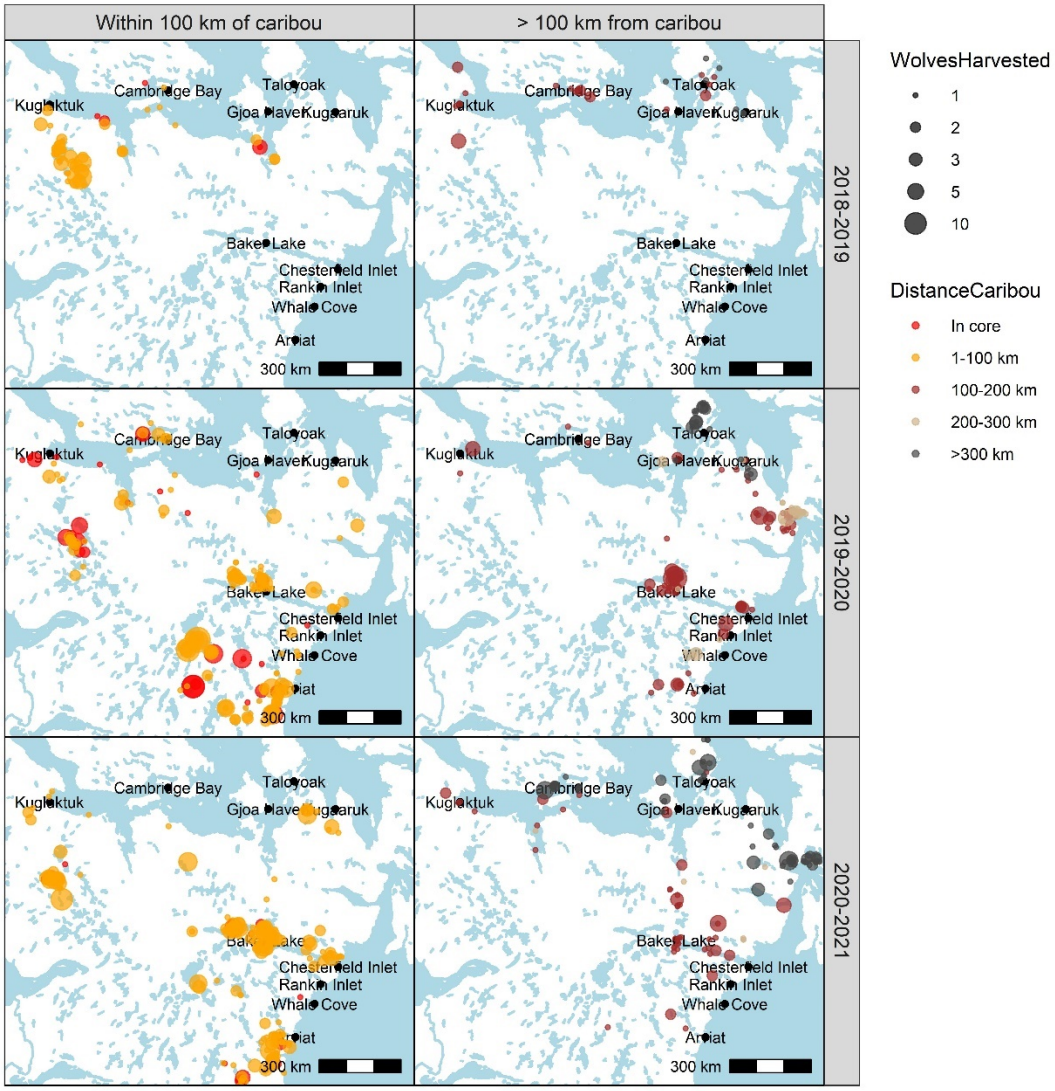


Figure 14. Locations of wolves harvested grouped by distance from areas of higher caribou density (DistanceCaribou), Kitikmeot 2018-19 and Nunavut 2019-20 and 2020-21.

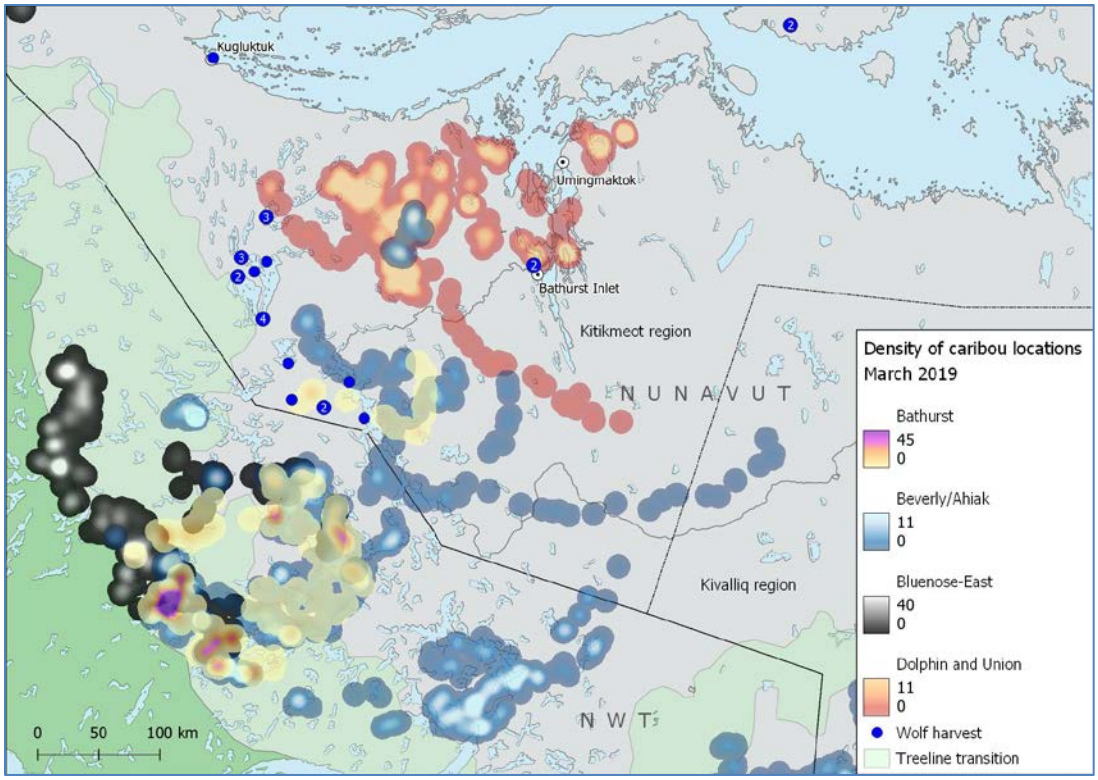


Figure 15. Collared caribou distribution by herd and wolf harvest collection locations, March 2019.

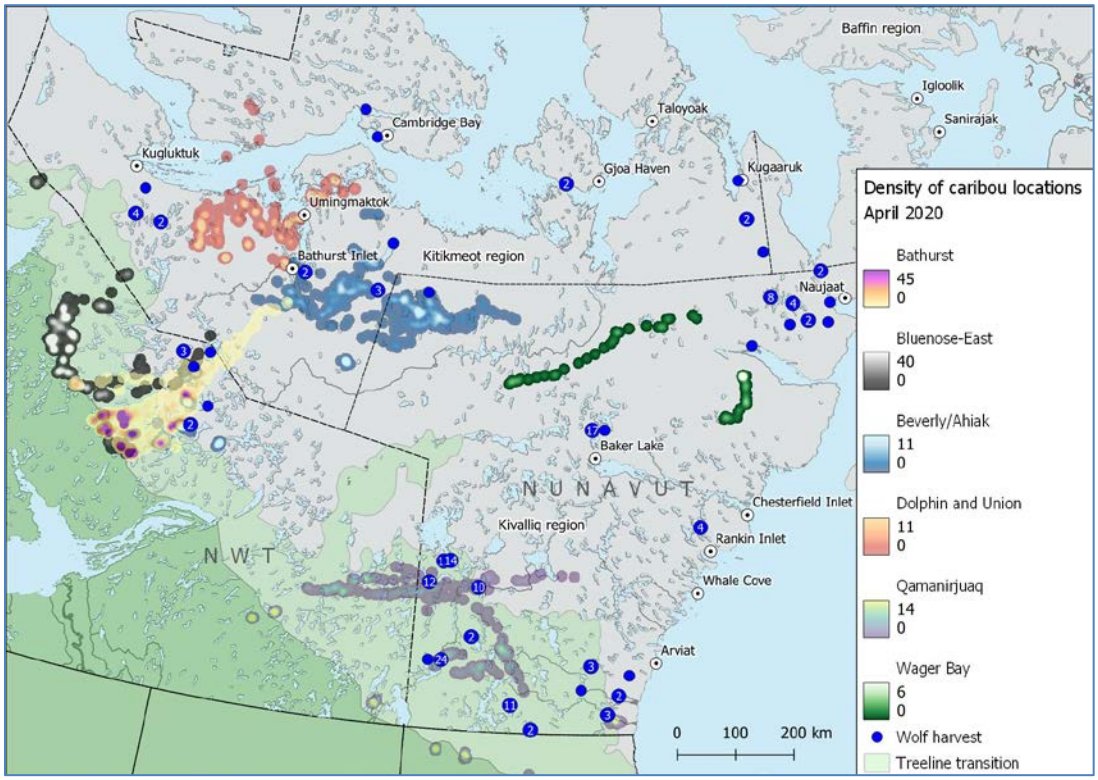


Figure 16. Collared caribou distribution by herd and wolf harvest collection locations, April 2020.

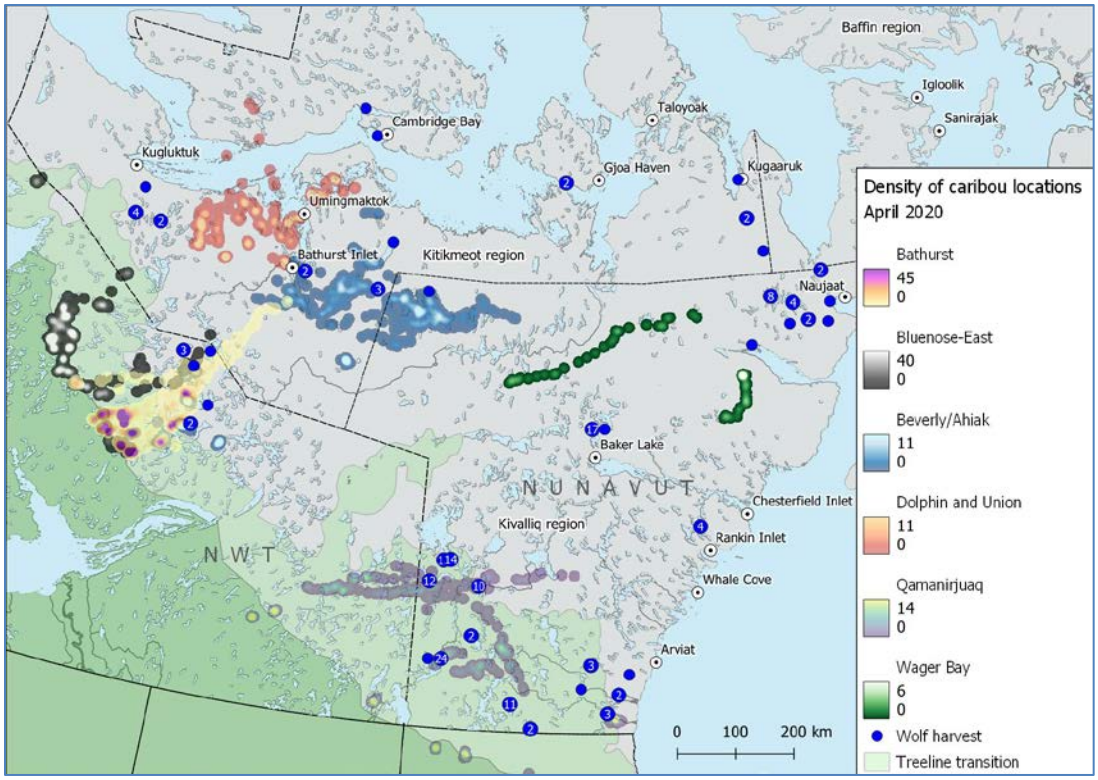


Figure 17. Collared caribou distribution by herd and wolf harvest collection locations, April 2021.

Proportion of wolf packs harvested

Kitikmeot 2018-19

Within the Kitikmeot in 2018-19 only Kugluktuk had sufficient samples sizes to examine the proportion of wolves harvested from packs observed (94% overall; *n* = 23 packs). The mean proportion of each pack harvested was high and trended higher across the winter, but monthly sample sizes were low (Fig. 18). Full packs were harvested on 87% of occasions.

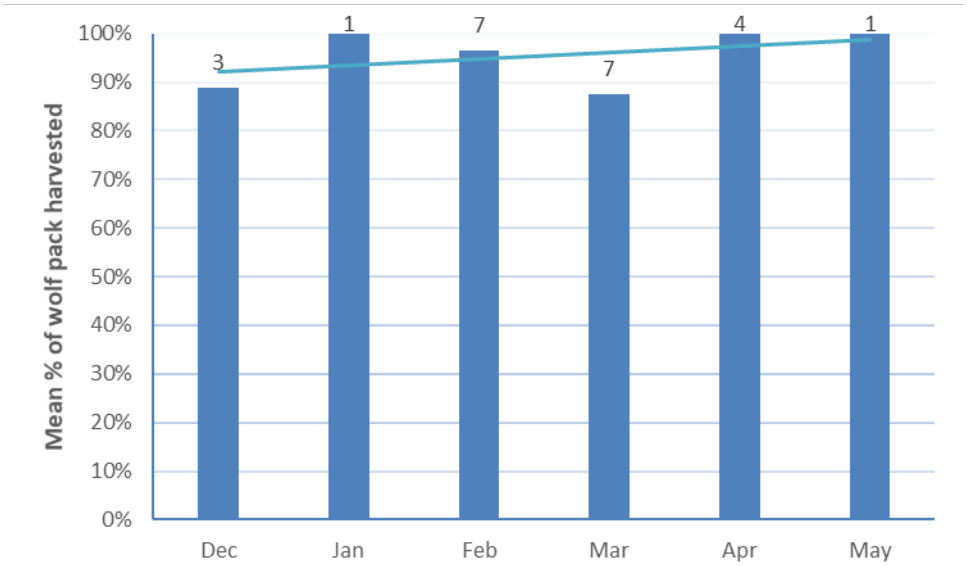


Figure 18. Mean proportion of wolf packs (≥2 wolves) harvested by month generated by QGIS assessment, Kitikmeot, 2018-19. Sample sizes shown above bars. Linear trendline added.

Nunavut 2019-20

Hunters from Arviat and Kugluktuk reported harvesting the greatest proportion of wolves from packs observed during 2019-20, roughly 75% (Fig. 19). Across Nunavut, the mean proportion of each pack harvested increased as the winter progressed from 50% in November to 75% in May (Fig. 20). Full packs were harvested on 35% of occasions across the season, with an increasing trend as the season progressed: 16% during October–December (*n* = 63), 39% during January–March (*n* = 76), and 50% during April and May (*n* = 68). It is likely that wolf packs in a given area were collectively harvested over time by different hunting parties until there were none left, but it is difficult to verify this assumption.

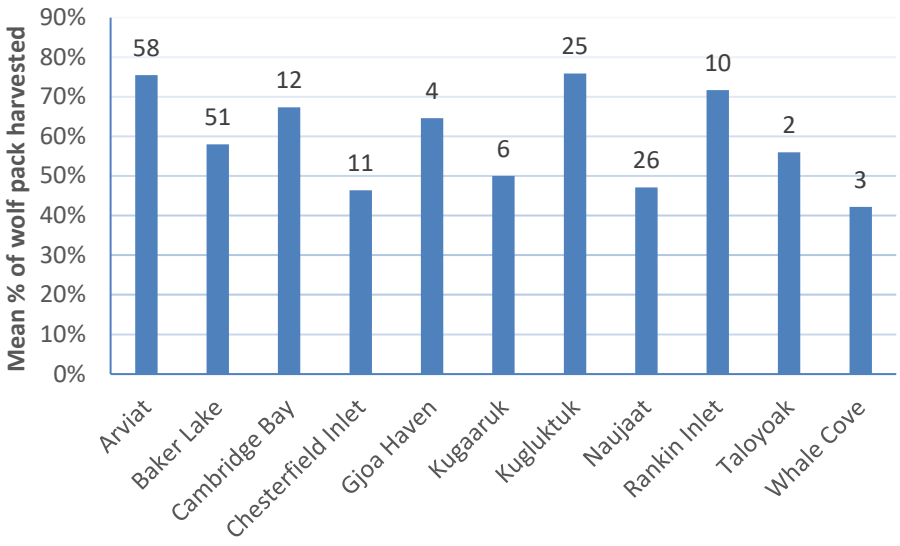


Figure 19. Mean proportion of wolf packs (≥2 wolves) harvested by community generated by QGIS assessment, Nunavut, 2019-20. Sample sizes shown above bars.

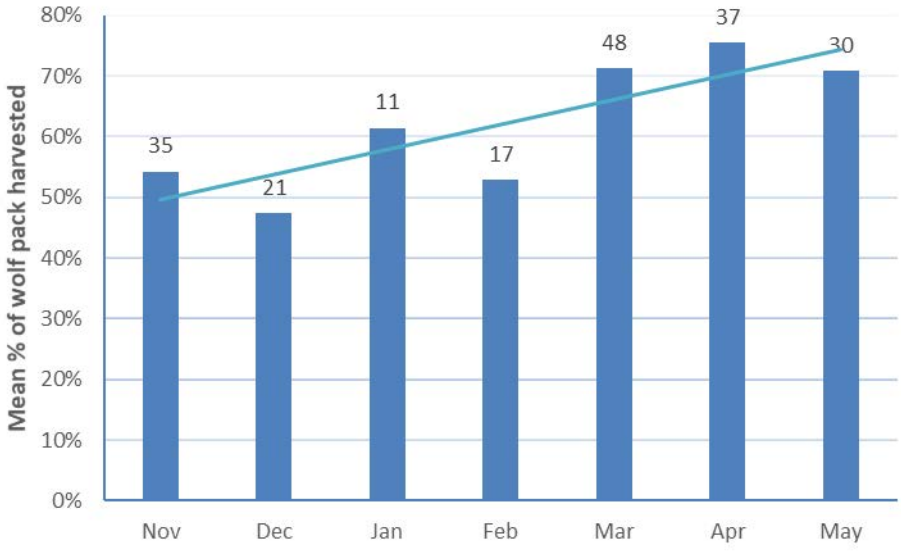


Figure 20. Mean proportion of wolf packs (≥2 wolves) harvested by month generated by QGIS assessment, Nunavut, 2019-20. Sample sizes shown above bars. Linear trendline added.

Nunavut 2020-21

Hunters from Arviat reported harvesting the greatest proportion of wolves from packs observed during 2020-21 (85%), with most harvesting 50–70% of packs (Fig. 21). Across Nunavut, the mean proportion of each pack harvested again increased as the winter progressed (Fig. 22). Full packs were harvested on 40%

of occasions across the season, with an increasing trend as the season progressed: 22% during October–December (*n* = 64), 42% during January–March (*n* = 102), and 56% during April and May (*n* = 54). Arviat hunters harvested full packs 73% the time overall.

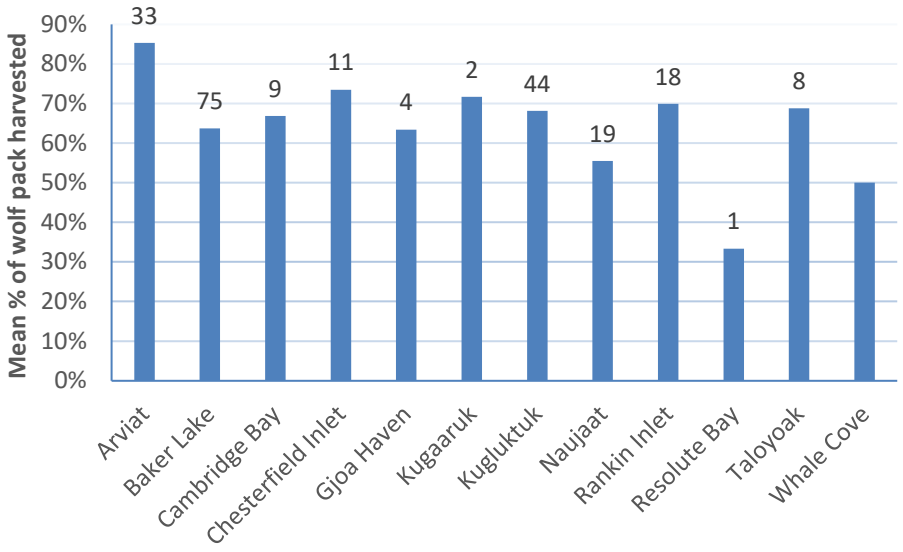


Figure 21. Mean proportion of wolf packs (≥2 wolves) harvested by community generated by QGIS assessment, Nunavut, 2020-21. Sample sizes shown above bars.

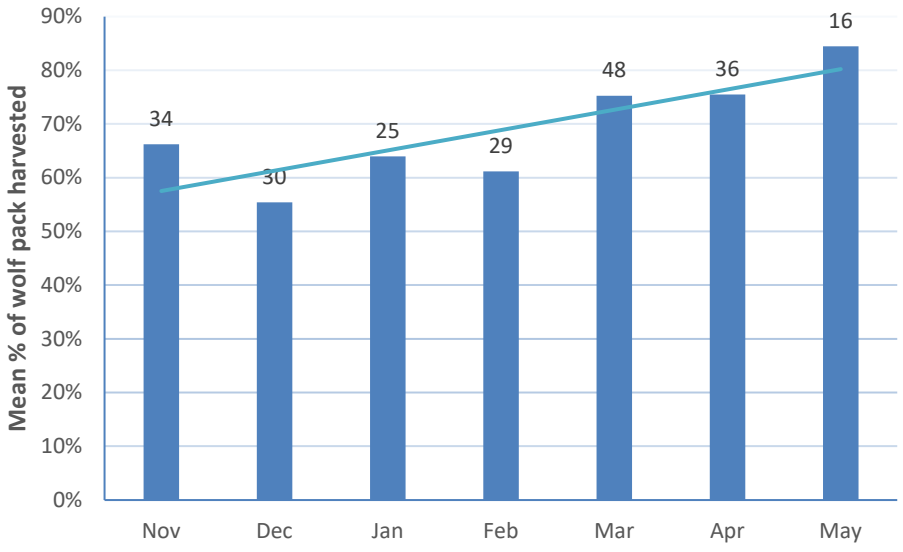


Figure 22. Mean proportion of wolf packs (≥2 wolves) harvested by month generated by QGIS assessment, Nunavut, 2020-21. Sample sizes shown above bars. Linear trendline added.

All data combined

Combining all data and using the rules-based assessment we tested the effect of the number of wolves observed, an index of pack size, on percent harvested. The resulting data set had 841 packs or individuals defined, 301 of which were single wolf observations and could not be used to define percent of pack or group harvested, leaving 540 groups of ≥2 wolves to assess harvest success. Wolves observed (log-transformed) was a significant predictor of percent harvested ($Z = -5.7$, $P < 0.0001$), with lower percentage of wolf packs harvested as pack size increased (Fig. 23). This result is likely because it is easier to harvest most or all of small packs than large packs.

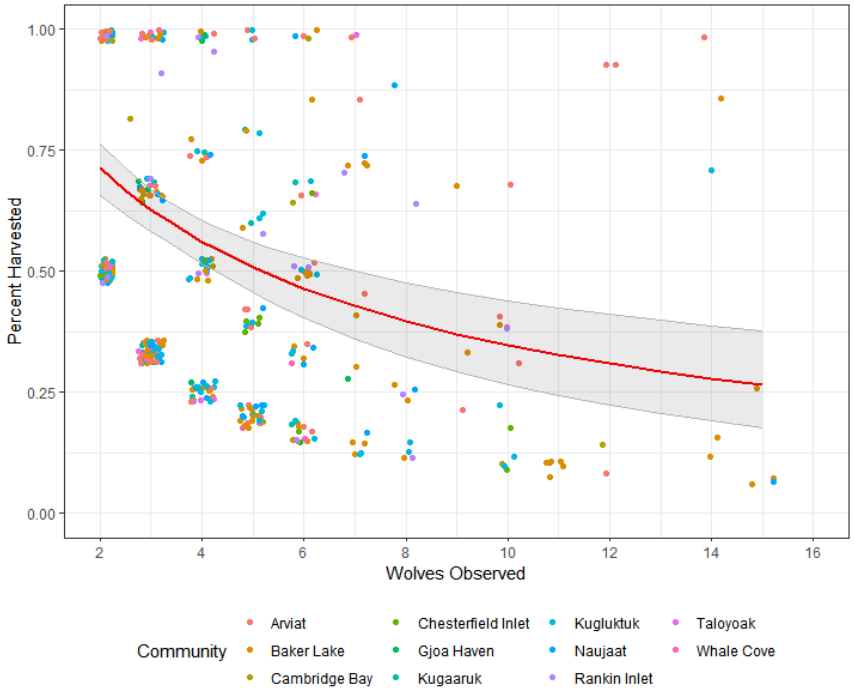


Figure 23. The proportion of wolf packs harvested by pack size (wolves observed) using a rules-based assessment, data combined for 2018-21. Points are jittered to increase interpretation. Predictions of mean percent harvested from logistic regression analysis with log(wolves observed) as a predictor are shown (red line). Confidence intervals are shaded around the prediction line.

Of additional interest was whether a temporal trend in harvest could be detected. For this analysis the data set was reduced to the 3 communities with the most harvest data – Arviat, Baker Lake, and Kugluktuk. Models were considered with community, wolves observed, and date for each harvest year. A model with harvest year, date of harvest, and wolves observed was used to control for harvest year and the effect of the number of wolves on harvest rate. Interactions between community and date of harvest were tested with no significant differences detected. A model with an additive effect of date across all communities had some support with date of harvest being significant ($Z = 2.2$, $P = 0.02$) in addition to pack size ($Z = -$

4.1, $P < 0.0001$; Fig. 24). A plot of predictions (standardized for a mean pack size of 4 wolves) revealed a weak trend with harvest rate increasing as the season progressed.

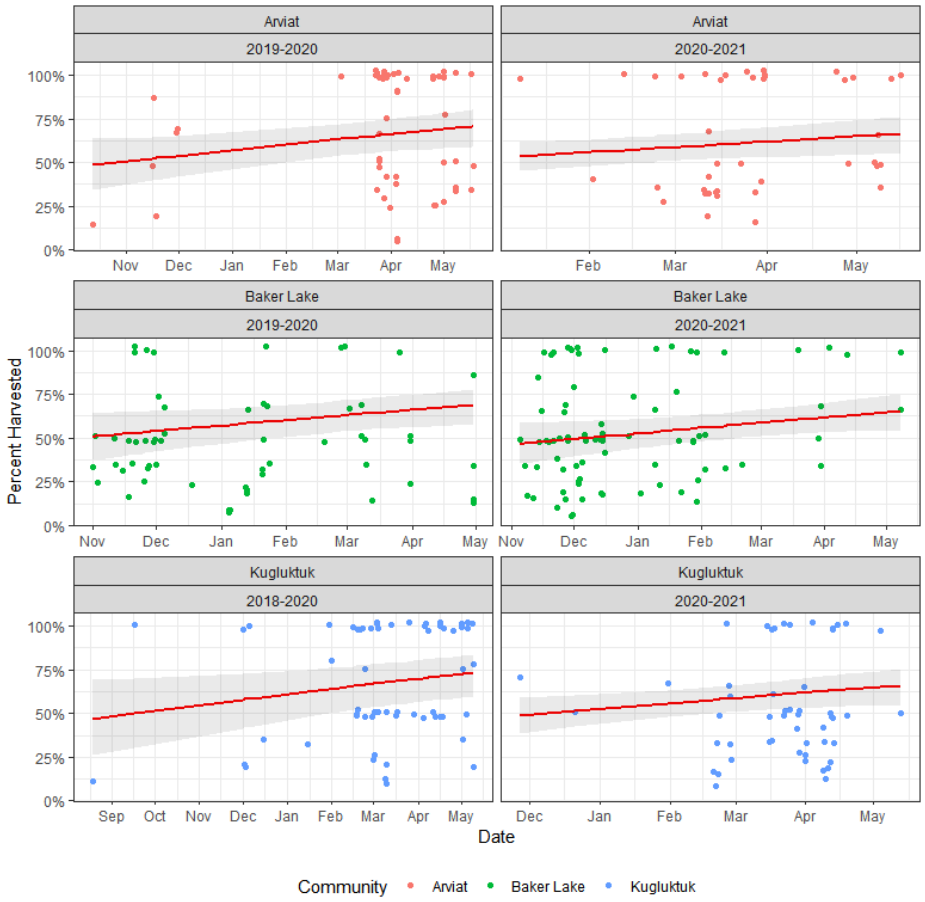


Figure 24. The proportion of wolf packs harvested by date using a rules-based assessment for 3 communities across different harvest periods. Points are jittered to increase interpretation. Predictions of mean percent harvested from logistic regression analysis with date as a predictor with log(wolves observed) set at mean values (4 wolves; red lines). Confidence intervals are shaded around the prediction line.

Wolf numbers and proportion harvested

One of our objectives was to attempt to estimate how many wolves may be associated with the migratory QM herd during winter, and thus what proportion of the wolves is being harvested by community hunters. We focussed on the QM herd because recent estimates of herd size were available (Boulanger et al. 2018) and there was little overlap with adjacent herds. No field-based estimates of wolf numbers in Nunavut are available. Methods to estimate wolf numbers using ungulate biomass (Kuzyk and Hatter 2014, Nishi et al. 2020) are complicated by the need to calculate wolf density over the massive range of the migratory

QM caribou herd, as well as to consider the substantial number of muskoxen present in portions of the range. The ungulate biomass method makes numerous assumptions and therefore this estimate may not pertain to the current wolf population in migratory caribou ranges. Ungulate biomass could be used to explore wolf numbers; however, we strongly caution against using these estimates to define management targets.

Nishi et al. (2020) used a 'ungulate biomass index' (UBI) regression equation (Kuzyk and Hatter 2014) to derive wolf density estimates applied to 95% utilization distribution of caribou winter range size.

$Y = 5.4x - 0.166x^2$

where y = wolf density (wolves/1,000 km^2) and x = ungulate biomass index/km^2. Barren-ground caribou were assigned a relative biomass value of 2, therefore estimates of caribou density (caribou/km^2) were multiplied by a factor of 2 to generate the UBI biomass index. Muskoxen and moose were not considered in this current 'back of the envelope' calculation.

Assuming approximately 250,000 QM caribou within a winter range covering about 70,000 km^2 (mean annual 95% utilization distribution from 2017-18 to 2021-22; Caslys Consulting, unpubl. data), the UBI calculation estimated approximately 2,100 wolves in total associated with the QM herd during winter. Alternatively, ECC used a ratio of roughly 10 wolves/1,000 caribou on winter range (Nishi et al. 2020: Tables 24–26), which means (assuming ~250,000 QM caribou) there would be ~2,500 wolves in total associated with the winter range of the QM herd. Parker (1972) estimated a ratio of 1 wolf per 114 caribou, equating to an estimate of approximately 2,200 wolves associated with the herd. This approach also assumes availability of caribou to wolves at all times and does not consider the impact of predator control or other measures on wolf abundance. All these factors likely will cause variation in actual wolf abundance and for this reason the UBI index and other ratios should be interpreted cautiously.

These rough calculations of wolf abundance associated with the QM caribou winter range suggest that, assuming the communities of Whale Cove, Arviat and Rankin Inlet hunt wolves associated with the QM herd and that the wolf harvest levels from 2019-20 (265 wolves) and 2020-21 (183 wolves) are usually attained, Nunavut hunters are harvesting ~9–13% of the wolf population associated with the QM herd during winter and spring each year. However, these calculations overlook that a portion of the wolf population will not follow the QM herd onto the tundra and become potentially available to NU hunters. The proportion of the wolf population associated with the QM herd that is available to NU harvesters is unknown, but based on movements of a limited number of collared wolves in the NWT and data from Russia, perhaps 50–70% of wolves migrate with herds in spring (Kolpaschikov et al. 2015, Clark et al. 2021), meaning that NU hunters are taking a larger proportion of the wolves associated with the QM and available to harvesting on the tundra, perhaps in the 12–25% range.

Discussion

Wolf harvesting is a traditional part of the culture of Nunavut hunters, as well as a source of seasonal income. During 2018-19 to 2020-21, the number of harvested wolves fluctuated but the number of hunters remain more or less the same. Some hunters hunt wolves occasionally while hunting caribou and/or muskoxen, and some hunt wolves on a more regular basis. Almost all wolves (~99%) were shot with few animals trapped. The effectiveness of Nunavut wolf hunters appears high, with large annual harvests from some communities, especially Arviat, Baker Lake and Kugluktuk. Within the North Slave region of the NWT, the catch per unit effort by Nunavut harvesters was 2 magnitudes higher than that of NWT harvesters (based on harvester questionnaires; Nishi et al. 2020). Nunavut hunters travel long distances to harvest wolves, often later in the winter and generally towards higher densities of caribou (as indexed by collared individuals), similar to other Arctic areas (Hayes et al. 2016).

The proportion of packs harvested differed among communities, but generally decreased as pack size increased and increased as the season progressed. Effectiveness of harvesting small packs of wolves was high, with roughly 50–60% harvest of packs with ≥3 individuals; the proportion of each pack removed increased as winter progresses, possibly related to longer daylight and better travel conditions. Removal of entire wolf packs may reduce caribou predation rates since partial removal can cause pack splitting and formation of more numerous packs (number of packs is a strong determinant on predation rates) and smaller packs (per capita kill rates are higher in smaller packs; Ballard et al. 1997, Hayes et al. 2000).

The proportion of immature individuals in a harvest is a valuable indicator of the impact of removal programs on carnivore populations (Kelsall 1968, Robinson et al. 2008). Largely based on the intensive wolf control programs (primarily wide-spread poisoning) conducted in the NWT during the mid-1950s to early 1960s, Kelsall (1968) and Fuller and Novakowski (1955) suggested that the proportion of young wolves in the harvest would shift from 10–20% in unexploited wolf populations, to 65–75% in heavily exploited populations. The age structure observed in both the Kitikmeot and Kivalliq regions in 2019-20 and 2020-21 (70–80% young wolves) indicate high levels of exploitation of the segment of the wolf population accessible to harvesters.

Many caribou herds within mainland NWT/Nunavut overlap during portions of the year, making it difficult to “assign” a harvested wolf to a particular herd. In recent years there has been extensive overlap north of Yellowknife among the BA, BE and BV herds (ECC, unpubl. data). ECC used an elaborate approach to assign harvested wolves to caribou herds when examining ground and aerial wolf harvest conducted in late winter 2019-20 (Nishi et al. 2020). However, hunters from Arviat, Whale Cove and Rankin Inlet target only what could be considered QM wolves, harvesting 265 and 183 wolves from the QM caribou herds’ range in 2019-20 and 2020-21, respectively. Although not a large proportion of the total number of wolves associated with the entire QM herd, assuming each wolf may consume approximately 29 caribou annually (Hayes and Russell 2000, WFATWG 2017: Appendix F), this represents about 5,300–7,700 caribou not predated or approximately 2–3% of the current QM population estimate.

For the QM herd where “assigning” harvested wolves to a particular herd is possible, approximately 12–25% of wolves associated with the herd during spring migration and pre-calving movements may be harvested by Nunavut hunters each year. These values, while impressive, are still far lower than the >55–

60% annual harvest required to maintain reduced wolf densities sufficiently to illicit a demographic response in an ungulate population (National Research Council 1997, McLaren 2016, Nishi et al. 2020). Regardless, given high harvests of wolves south and southwest of Arviat during March, April and May, it is likely that both the numbers of wolves and number of wolf-killed QM caribou are greatly reduced as the herd migrates during late winter and spring to their calving grounds west of the coast of Hudson Bay. This results in hunters targeting wolves following migrating caribou to their calving grounds and the removal of most resident wolves from the calving grounds, and therefore low numbers of wolves and reduced wolf predation on the calving and post-calving grounds. Low predation rates of new-born QM calves in the early 2010s were attributed to high harvest rates of wolves during pre-calving migration (Szor et al. 2014).

This level of harvest pressure requires annual effort since adjacent wolf populations can quickly respond to reduced wolf density by long-distance dispersal and increasing natality rates (Schmidt et al. 2017). It is possible that the traditional harvest of large numbers of wolves by coastal communities may have influenced demographic trends in the QM herd. Demographic analysis of the QM herd is being conducted in unison with the 2022 calving ground survey (J. Boulanger, pers. comm.). Results from this survey should provide further inference on QM status and associated demographic trends.

From 1995 to 1998, an average of 146 wolves were harvested annually in the Qikiqtaaluk Region, with about 80% taken from Baffin Island (Krizan 1999, unpubl. data). Caribou numbers on Baffin Island were likely high to declining during the 1990s (Campbell et al. 2015). These wolf harvests are minimum counts; a hunter survey in the Qikiqtaaluk region in 1998 showed that approximately one half of the harvest was used privately for clothing or sold locally (and hence not enumerated in the system), with the other half was sent to fur auctions (Krizan 1999). No wolves harvested from Baffin Island were detected by the Nunavut-wide Support for Active Harvesters program over 2 years, during a period of very low caribou numbers (Campbell et al. 2015, Ringrose 2018) and no significant secondary prey to maintain wolf populations. Where alternative large prey is absent, such as on Baffin Island, wolf populations will generally decline with declining primary prey (Mech 2005). Despite low and/or declining populations of caribou within the central mainland Nunavut (Kitikmeot Region), high wolf abundance can be sustained by alternative large prey, such as muskoxen and moose.

Except for hunters from Baker Lake, from October to late February Nunavut hunters mainly harvested wolves while hunting other game, leading to similar hunting effort from early to mid-winter among years. The number of wolves harvested appears to fluctuate depending on caribou numbers (and hence wolf numbers) in the area. Beginning in early March, hunters generally travelled to specifically hunt wolves, resulting in higher hunting effort and harvests.

Ground-based harvest efforts often remove the most naïve and vulnerable segments of the population (Adams et al. 2008), whereas control programs are thought to remove wolves more randomly since lethal control is less selective and breeders are more likely to be killed than they would be otherwise (Webb et al. 2011, Schmidt et al. 2017). If young or dispersing individuals make up a large portion of the harvest, harvest mortality may be compensatory because young naïve wolves exhibit the highest rates of natural mortality and dispersal (Boyd and Pletscher 1999; Hayes and Harestad 2000; Fuller et al. 2003; Smith et al. 2010). However, Nunavut wolf hunters appear able to target older and presumably potentially

breeding portions of the population later in the winter after most natural losses would have occurred, increasing the additive impacts of these ground-based harvest efforts.

Hunters in the Kitikmeot Region, mainly from Kugluktuk, harvested wolves primarily from winter ranges. The high proportion of young in the Kitikmeot sample during 2019-20 and 2020-21 indicate relatively intense harvest within the areas accessed by hunters. However, few wolves were harvested in April and May during caribou spring migrations to the mainland calving grounds of the BA and BE herds (Appendix 2). Logistics and access may be limiting the ability of Kitikmeot harvesters to reduce wolf numbers as caribou approach the calving grounds.

Wolf populations can quickly rebound when treatment is removed due to immigration from surrounding areas and high reproductive rates (National Research Council 1997, Schmidt et al. 2017). Within Nunavut, wolves are an annual resource for hunters, and wolf harvest for clothing and for income is a seasonal and traditional activity. For example, in the range of the BE, BA and BV caribou herds, migratory wolves generally den around and below treeline (Heard and Williams 1992). Although this likely also occurs on QM range where these wolves would be distant and unharvestable by hunters from coastal communities on Hudson Bay, many wolves associated with the QM herd also den above treeline closer and within the calving grounds (Heard and Williams 1992). Either way, “crops” of wolves therefore follow caribou residing or moving onto the tundra where they are accessible to coastal hunters on an annual basis.

Limitations

Our analyses were of course limited by numbers and distribution of collared caribou among herds. It was evident that collar distribution was thin to non-existent in some areas, especially in the northeast Nunavut mainland (Appendix 2). In other areas, the attrition in collars was evident, for example collar sample size on the Lorillard and Wager Bay herds was greatly reduced by April 2020 when extensive wolf harvest associated with these two herds occurred north of Baker Lake (Fig. 5, Appendix 2).

Portions of our analyses and summaries were dependent on assumptions made on the wolf harvest data, largely related to the definition of pack size and numbers harvested. In some cases, it was easy to determine that, for example, 6 wolves were harvested from a pack of 6 wolves at the same location. Many other groupings were not so obvious. These data could be rendered clearer, facilitating more robust analyses, if the database were able to clarify which harvested wolves belonged to which recorded pack size and the number of wolves harvested from that particular pack -cross-referencing harvest ID numbers would be helpful. Also, harvest of individual wolves needs to be clarified that they were indeed lone wolves, and not associated with other individuals. In some cases, the database observations did not agree with the spatial records. In general, we consider the proportion of wolf pack harvested to be a crude measure of harvest effectiveness.

A measure of catch per unit effort (CPUE) by Nunavut wolf hunters would provide another indication of the effectiveness of harvest within the territory. The challenge with the current wolf harvest data set is that annual harvest effort is unknown and therefore it is difficult to ascertain if annual harvest success for some communities is due to greater harvest effort or higher harvest effectiveness. However, there are many variables that may affect measures of harvest effort: for example, effort likely varies with snow conditions and the winter distribution of caribou and hence the associated wolves; wolves are harvested

during harvest trips for other species (e.g., caribou, muskoxen) as well as trips primarily focussed on hunting wolves; unsuccessful wolf harvest trips are not recorded; and harvest effectiveness likely varies between experienced and young hunters. Regardless, a data set that provides even a crude measure of effort, possibly the number of excursions and number of days of hunting and the number of kilometres travelled (and possibly details of routes taken) could provide a better overall estimate of harvest effectiveness (Clake et al. 2021).

More work is required to estimate how many wolves may be associated with the migratory QM herd (and possibly other herds) during winter. We suggest that using the UBI equation to estimate wolf numbers may be problematic for migratory barren-ground caribou for a number of reasons (Kuzyk and Hatter 2014). This equation was developed for wolf populations in British Columbia that prey on a variety of relatively stationary southern ungulate species, rather than northern wolves that primarily prey upon more ephemeral prey sources like populations of caribou and muskoxen. An inherent issue with the UBI calculation is that it does not consider that wolves do not have access to caribou across winter or seasonal ranges especially during the denning season (late winter) when movements by wolves are more restricted. Therefore, it is likely that only a proportion of caribou populations are "available" to wolves at any one time. Annually variable overlap among adjacent herds on winter ranges further complicates this relationship. Due to migratory caribou movements and restrictions of wolves to denning sites, the year-round availability of caribou to wolves is lower in northern populations and therefore the inherent relationship between wolf abundance and caribou biomass is different. Studies on functional response of predators to varying levels of prey density have shown that various relationships are possible, all of which have implications on the underlying relationship between prey and predator density (Zimmerman et al. 2005). Therefore, this likely produces an over-estimate of wolves given that only a proportion of caribou are available to wolves across the large seasonal range. We attempted to partially correct for this using crude estimates of the proportion of wolves that may be following the QM herd during spring migration. The UBI method also likely overestimates wolf abundance by assuming that the effectiveness of wolves in preying upon northern caribou is similar to the multitude of ungulate species available to southern wolves. It also assumes human-caused mortality rates do not exceed sustainable limits and wolf numbers have had time to adjust to prey biomass. For all these reasons we suggest estimates from this index are very crude, are likely overestimates, and should be treated cautiously.

Recommendations

1. The Support for Active Harvesters program facilitates harvest of wolves associated with concentrations of caribou within Nunavut, and appears to be an effective wolf management tool to remove a large proportion of accessible wolves while simultaneously providing additional support to hunters. Since wolf populations can quickly rebound when treatment is removed due to immigration from surrounding areas and high reproductive rates, the incentive program should be continued.
2. The Support for Active Harvesters program should ensure prompt payment to harvesters that submit skulls. The faster harvesters are paid, the more likely they will restock fuel and supplies and return to the land to hunt additional wolves.

3. Although challenging to obtain, we recommend that some form of hunter effort should be collected to supplement the wolf harvest data currently collected. The NWT program considers distance travelled for hunting and time spent hunting to estimate hunter effort (Clark et al. 2021). A data set that provides effort, possibly the number of excursions and number of days of hunting (and possibly details of routes taken) would provide a better overall estimate of harvest effectiveness.
4. The effectiveness of the wolf harvest program could be better examined by conducting a demographic analysis of caribou survival using the collar data. This would give a relative indication of how survival rates of the QM herd, where the proportion of wolves harvested during late winter and spring is thought to be comparatively high, compared to other herds.
5. Within the mainland Kitikmeot Region, much of the focus of wolf harvest incentives current target the winter ranges of the BE and BA herds. The effectiveness of harvest and resulting impacts on the demography of these herds would likely be enhanced by increased logistical support to harvesters from both sides of the border. Support and incentives to harvesters to continue to hunt wolves during migration from BE and BA winter ranges to calving ranges would enhance the effectiveness of the overall program; establishing forward operating camps and fuel caches may be useful. Expansion of the NWT Wolf Management Incentive Area into the Kitikmeot Region would provide additional incentives to Nunavut harvesters, further increasing the harvest of wolves associated with the BE and BA herds. It may be beneficial to engage and support experienced wolf hunters to set up harvester camps and focus on areas not normally accessible during spring caribou migration (*cf* Kolpaschikov et al. 2015, Clark et al. 2021).

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Appendix 1: Caribou collar sample size

Table 11. Collar sample size by caribou herd and month, December 2018 to June 2021, and most recent caribou herd estimates.

Year-Month	BA	BE	BV	QM	LR	WB	AH	DU
2018-Oct	27	43	42	49	16	9		3
2018-Nov	27	43	42	49	16	9		3
2018-Dec	27	43	42	49	16	9		33
2019-Jan	27	42	41	49	15	8		31
2019-Feb	28	42	40	48	13	8		31
2019-Mar	38	56	60	52	6	4		30
2019-Apr	39	57	59	55	3	4		28
2019-May	37	55	58	51	3	4		28
2019-Jun	36	51	53	44	2	4		24
2019-Jul	35	50	52	42	2	4		24
2019-Aug	35	49	51	29	1	3		24
2019-Sep	29	40	43	26	1	3		23
2019-Oct	26	39	33	23	1	3		21
2019-Nov	25	38	32	23	1	3		21
2019-Dec	25	36	31	23	1	3		20
2020-Jan	25	36	30	23	1	3		20
2020-Feb	31	32	29	24	1	3		21
2020-Mar	70	71	32	29	1	2		18
2020-Apr	71	67	26	23	1	2		16
2020-May	59	59	22	16	1	2		13
2020-Jun	57	55	22	12	1	2		12
2020-Jul								8
2020-Aug								7
2020-Sep	45	53	25	38	6	5	4	4
2020-Oct	43	53	25	38	6	5	4	4
2020-Nov	42	50	25	36	6	4	4	4
2020-Dec	41	49	25	36	5	4		3
2021-Jan	40	47	24	36	5	4		
2021-Feb	44	47	26	34	5	3	1	
2021-Mar	55	82	61	33	5	3	2	
2021-Apr	55	78	61	33	4	3	4	36
2021-May	55	78	60	32	5	3	4	35
2021-Jun	53	78	59	30	5	3	4	35
Herd size	8,200	22,350	103,400	288,250	41,000	41,000	71,340	3,800

Year of estimate and source of herd size estimates:

Bathurst (BA): 2018; Adamczewski et al. 2019.

Bluenose East (BE): 2018; Boulanger et al. 2019.

Beverly (BV): 2018; Campbell et al. 2019.

Qamanirjuaq (QM): 2017; Boulanger et al. 2018.

Lorillard (LR): 2002; COSEWIC 2016.

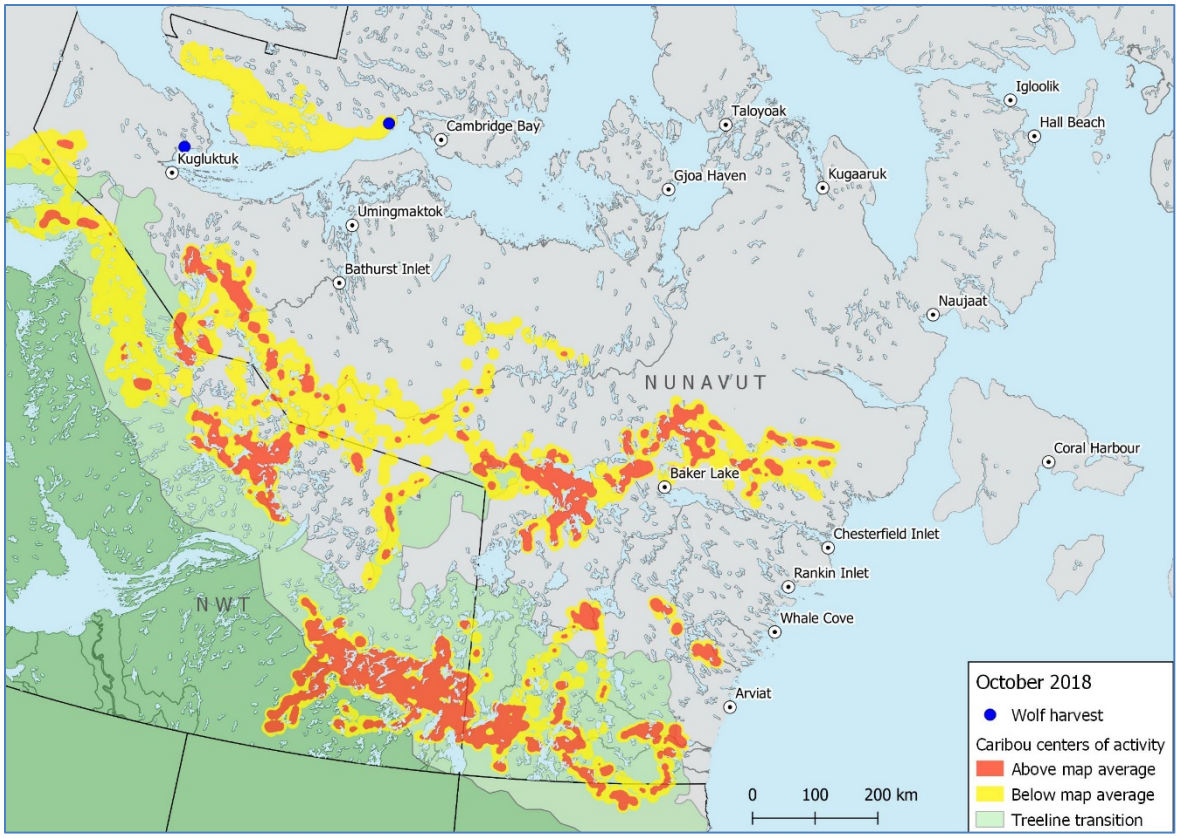
Wager Bay (WB): 2002; COSEWIC 2016.

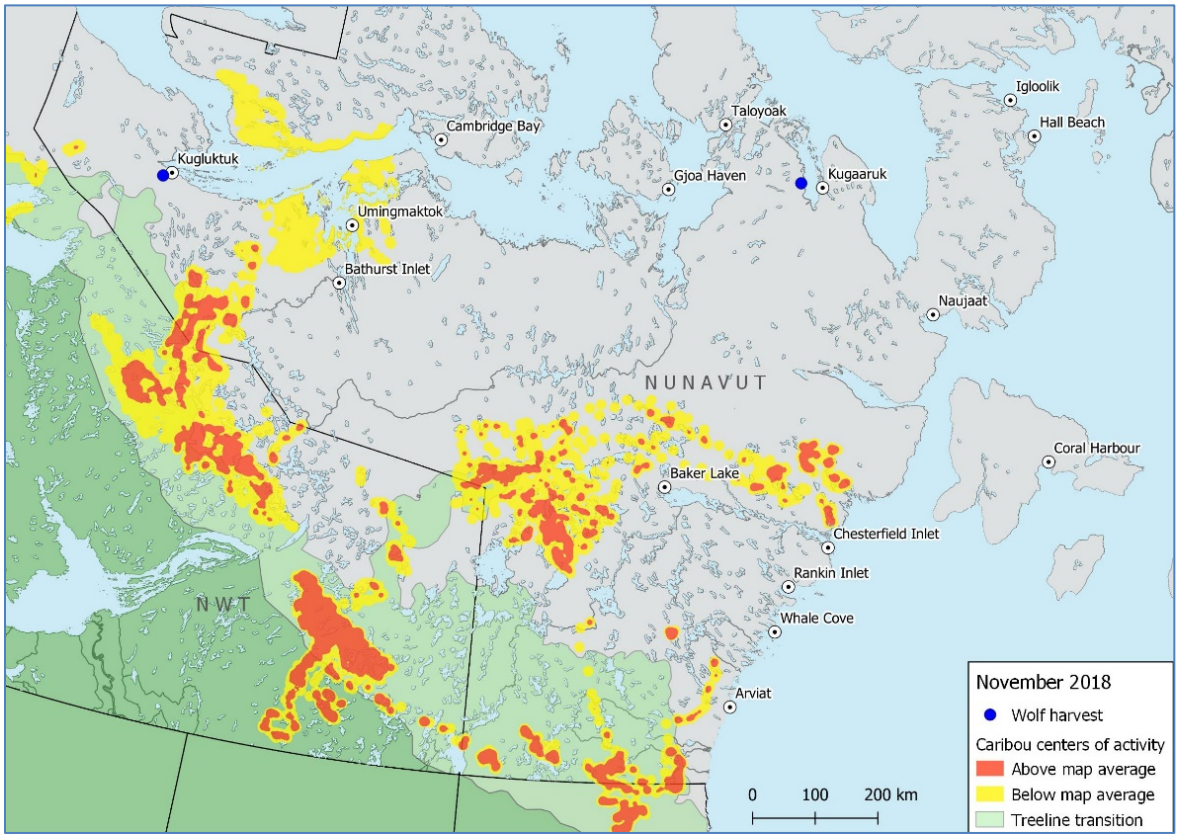
Ahiak (AH): 2011; Campbell et al. 2014.

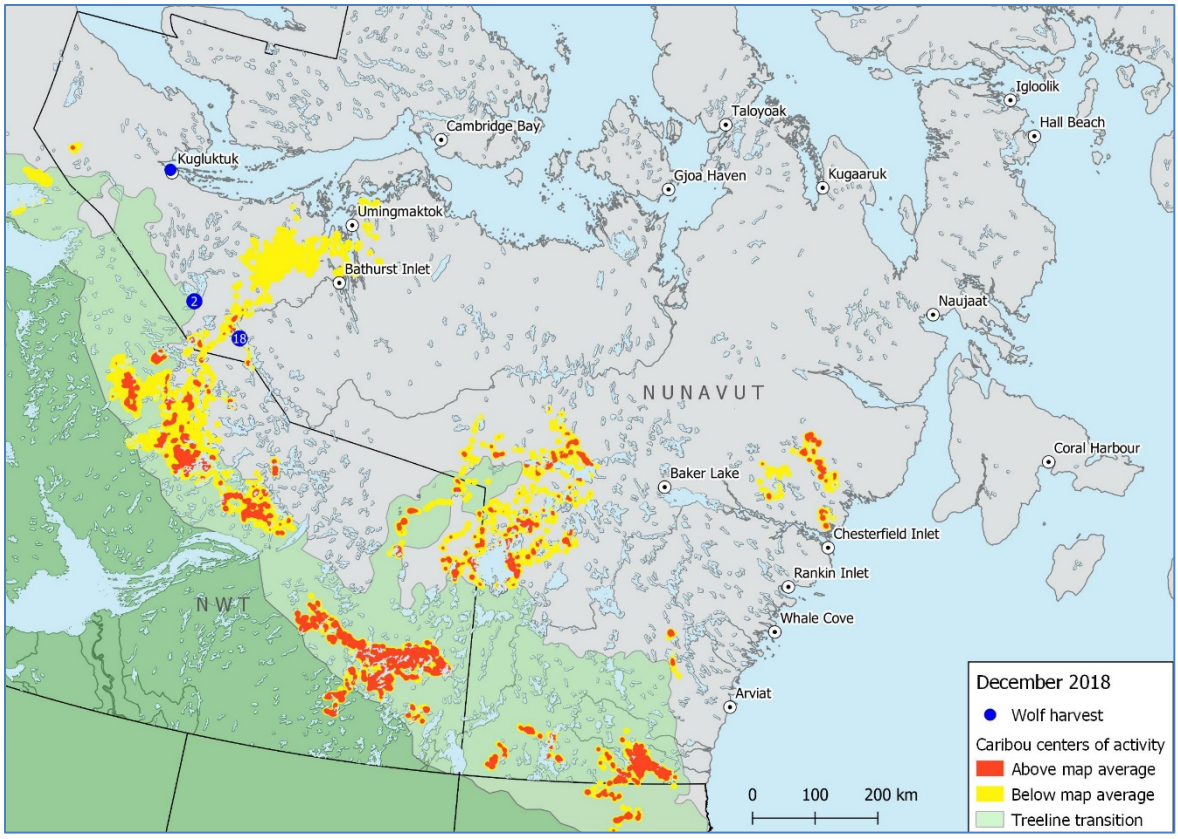
Dolphin and Union (DU): 2020; Campbell et al. 2021.

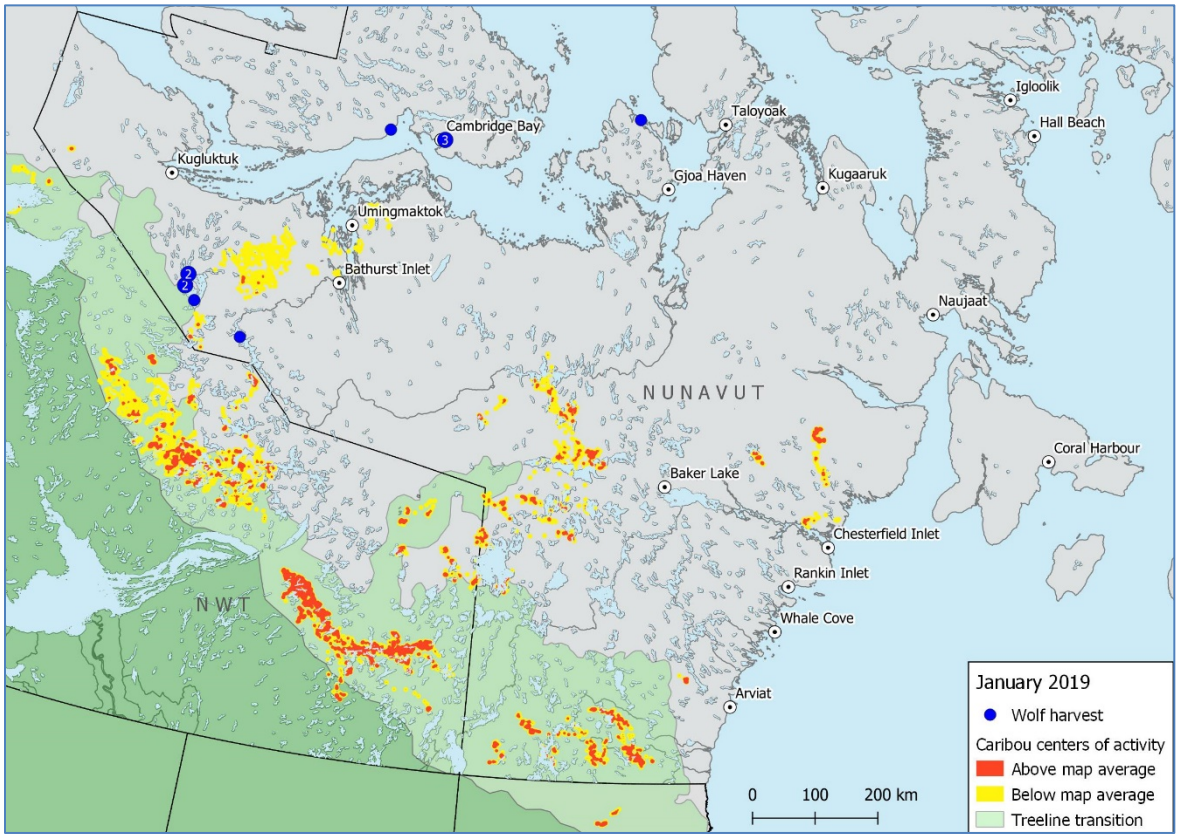
Appendix 2: Monthly caribou and wolf harvest maps October 2018 – June 2021

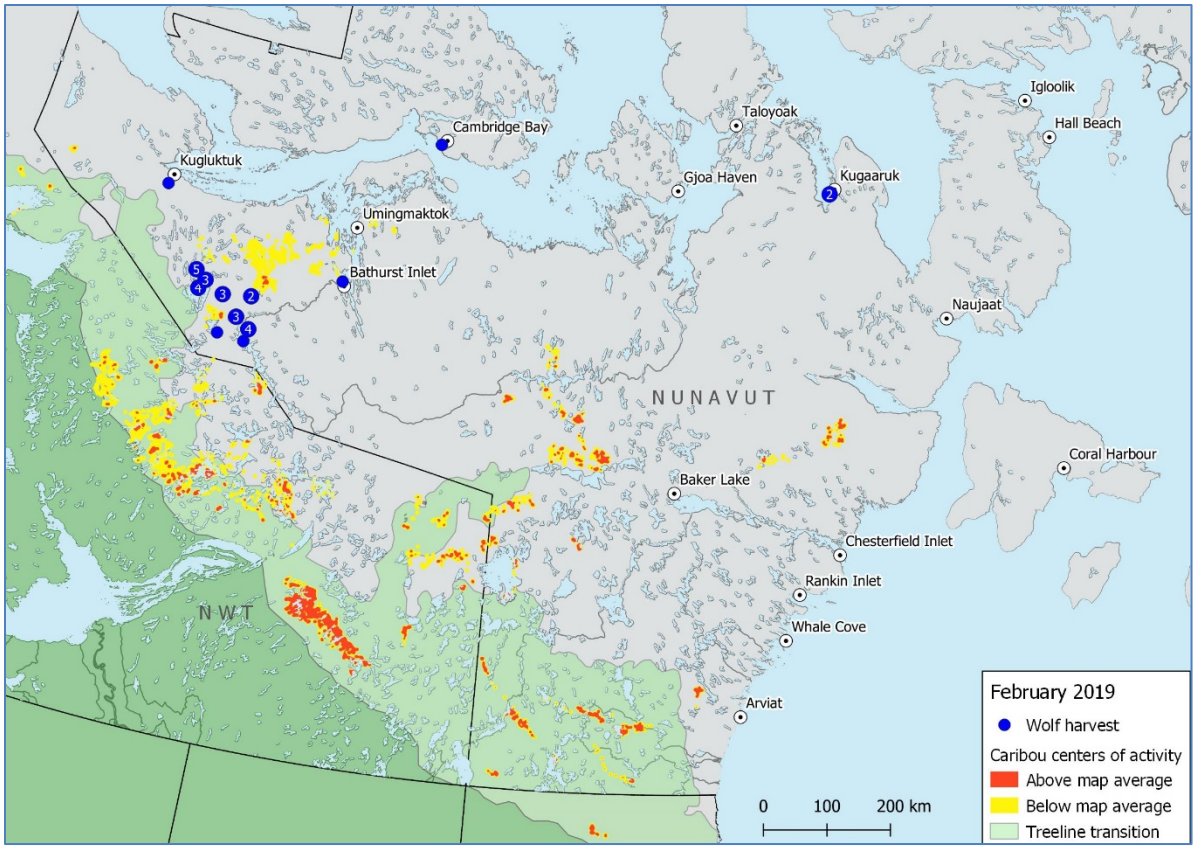
Figure 25. Collared caribou distribution and centers of activity, and wolf harvest collection locations, October 2018 to June 2021 (see legends for month and year).

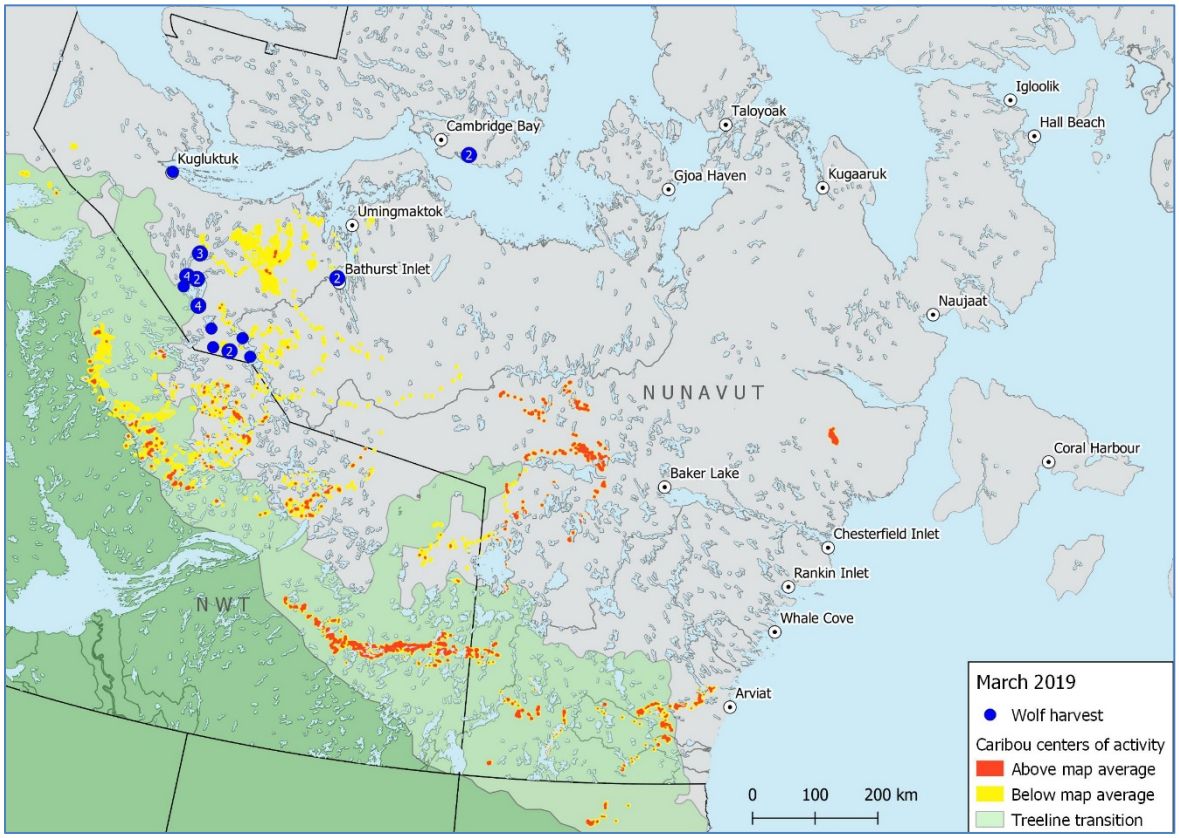

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October 2018
Wolf harvest
Caribou centers of activity
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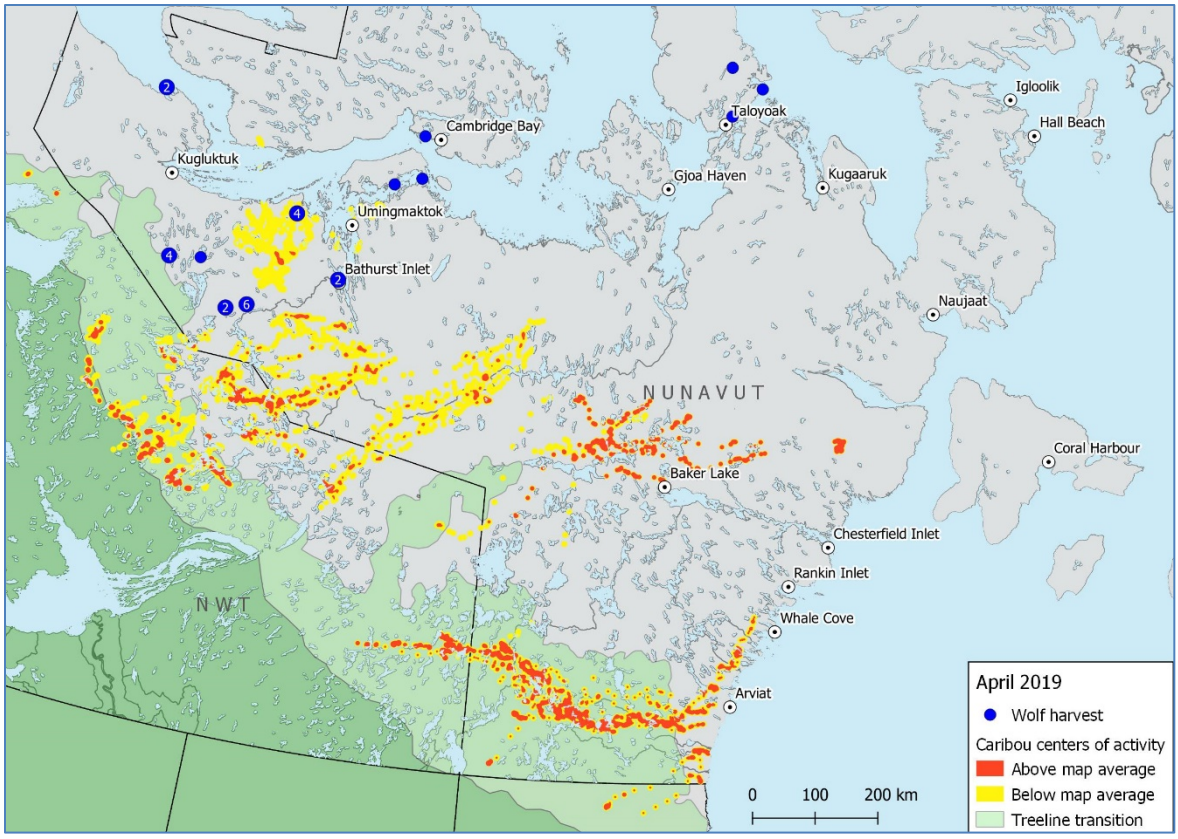

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Wolf harvest
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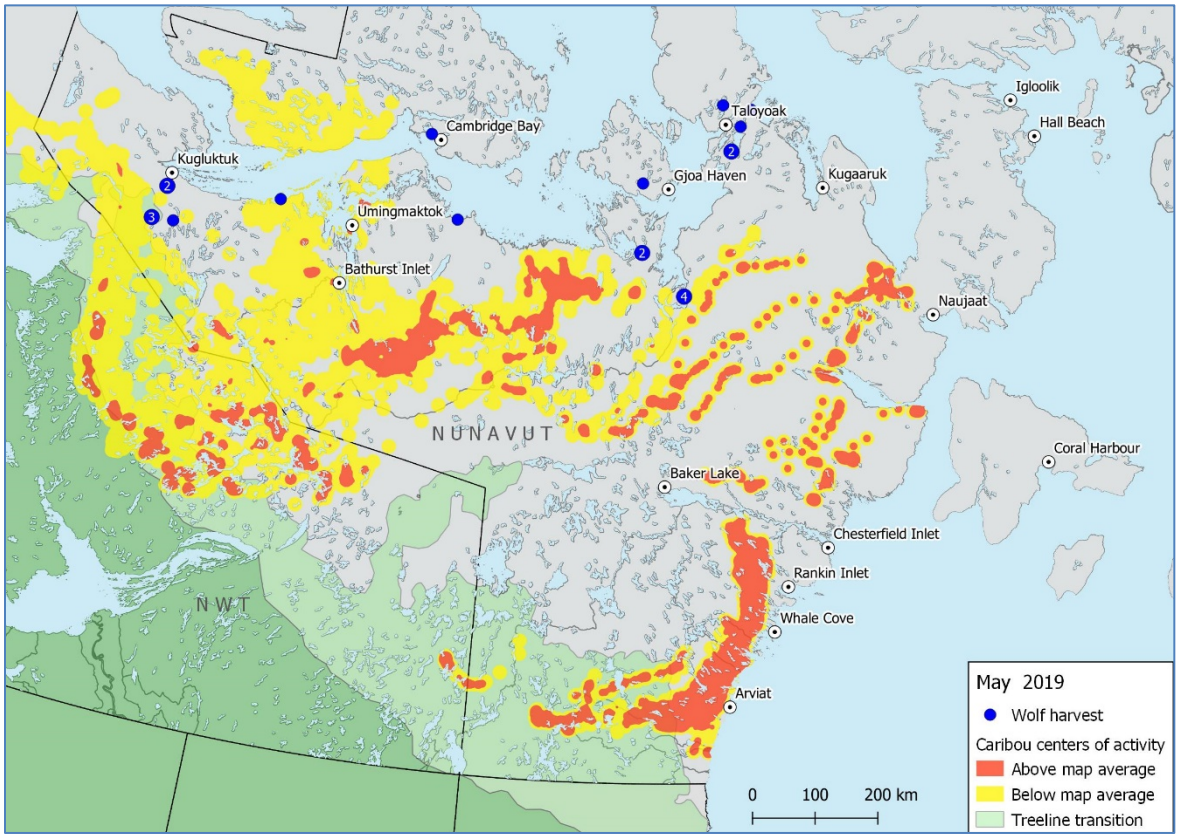

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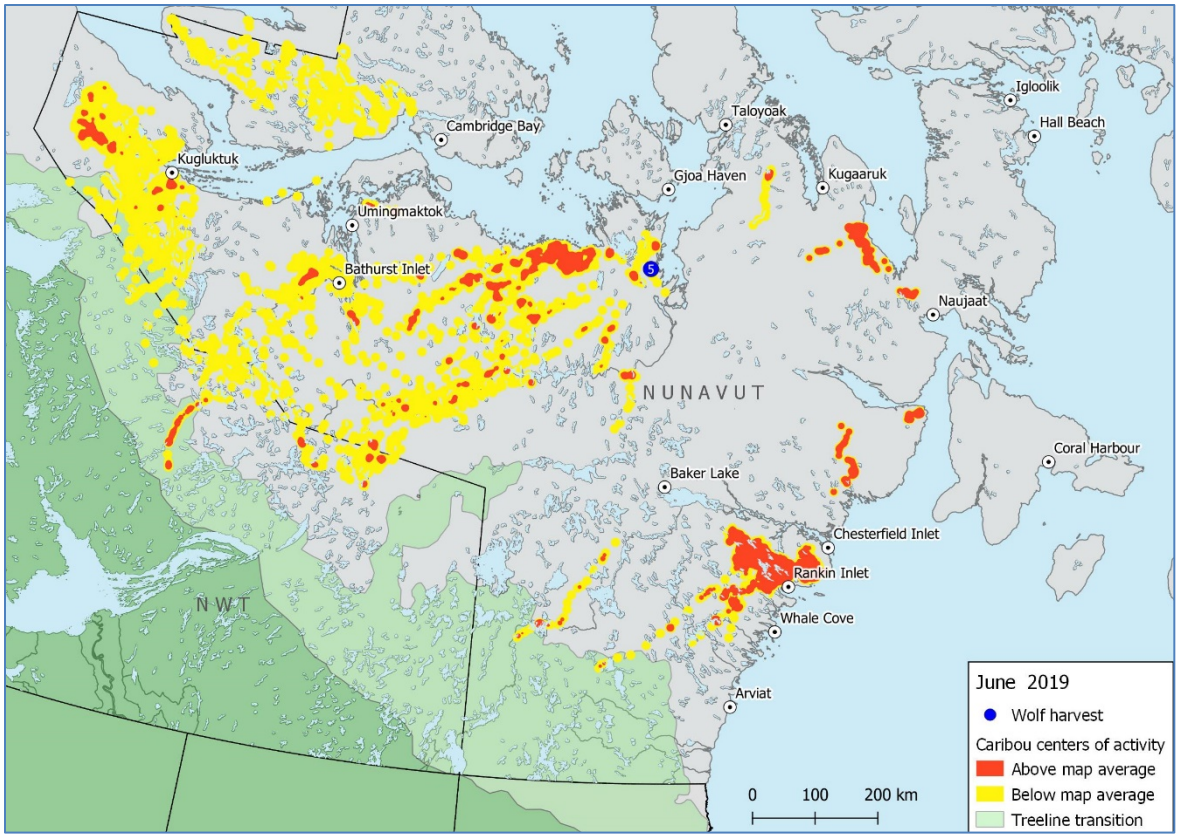

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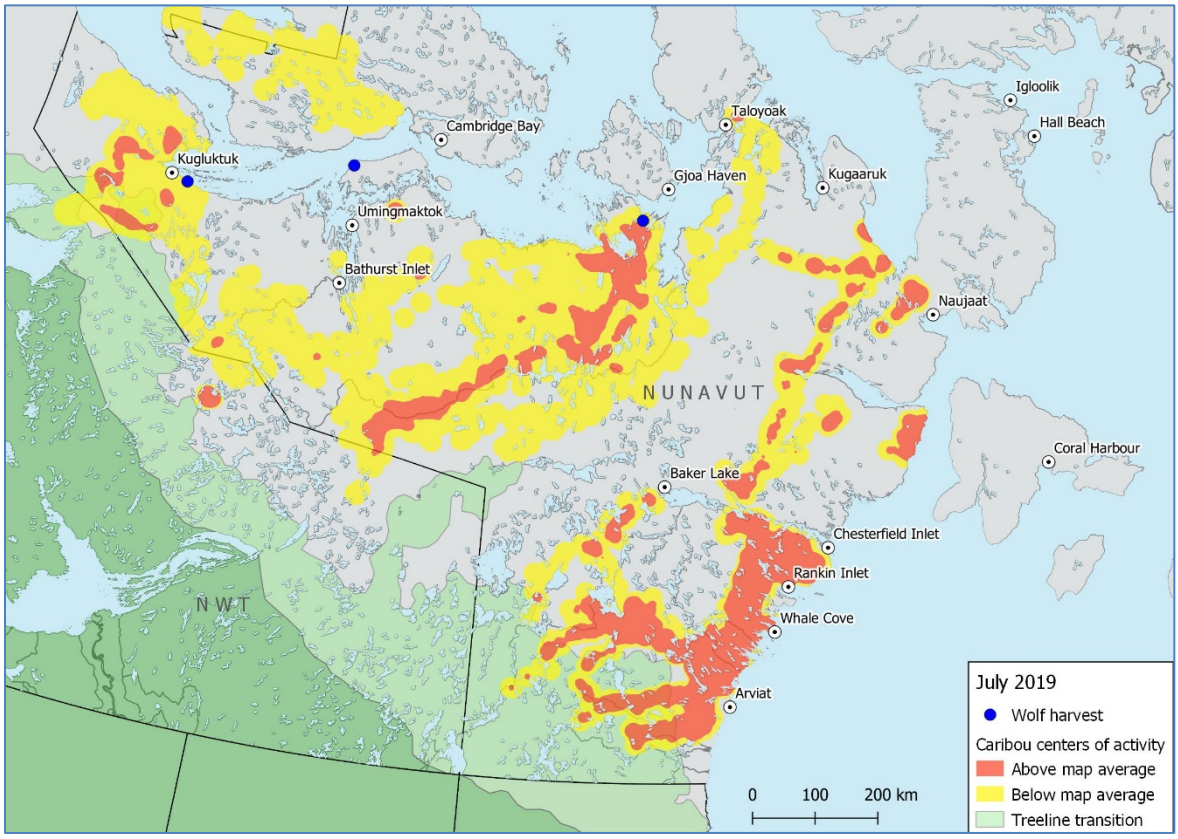

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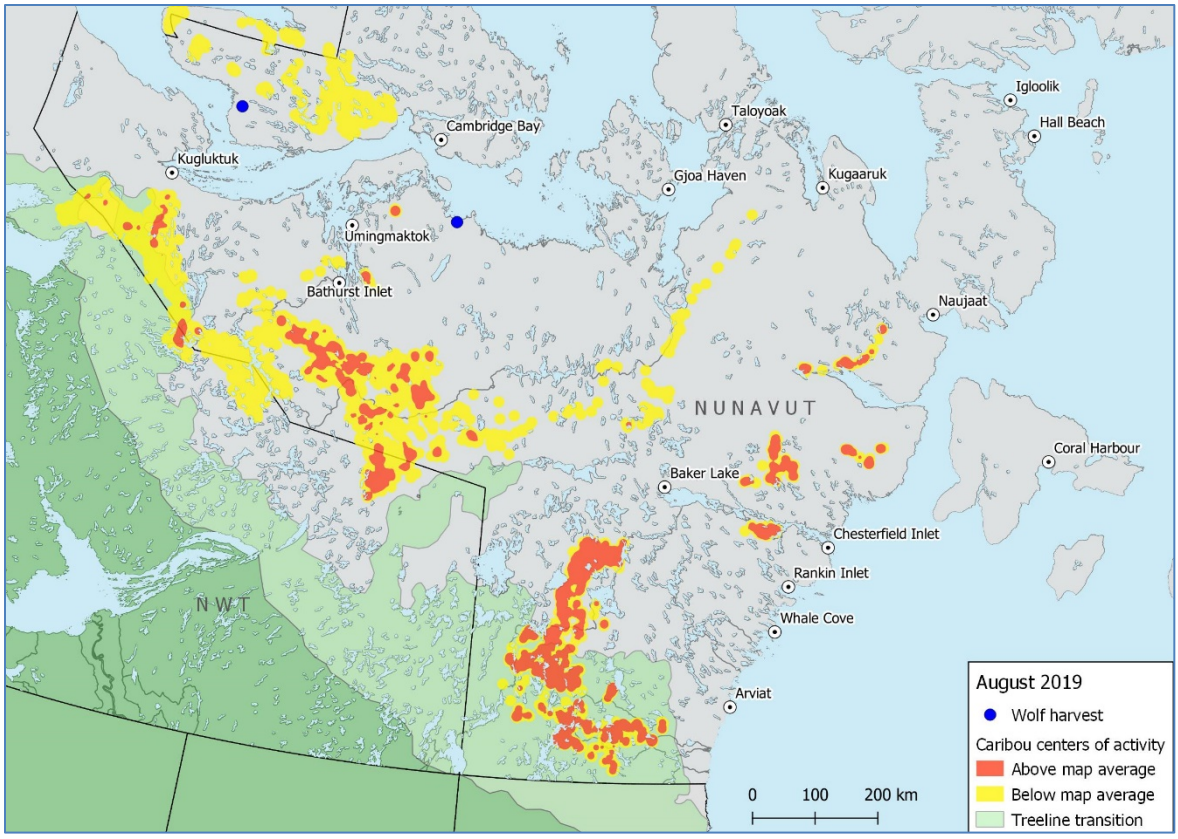

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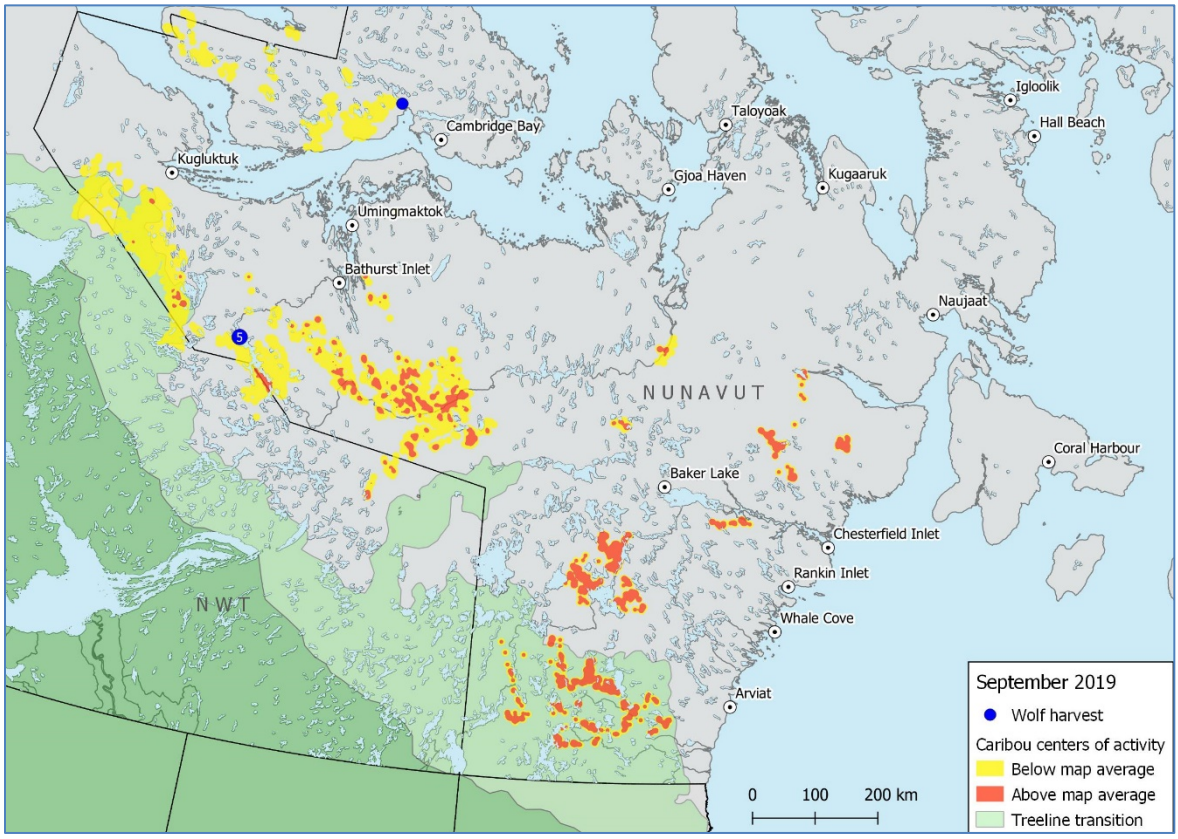

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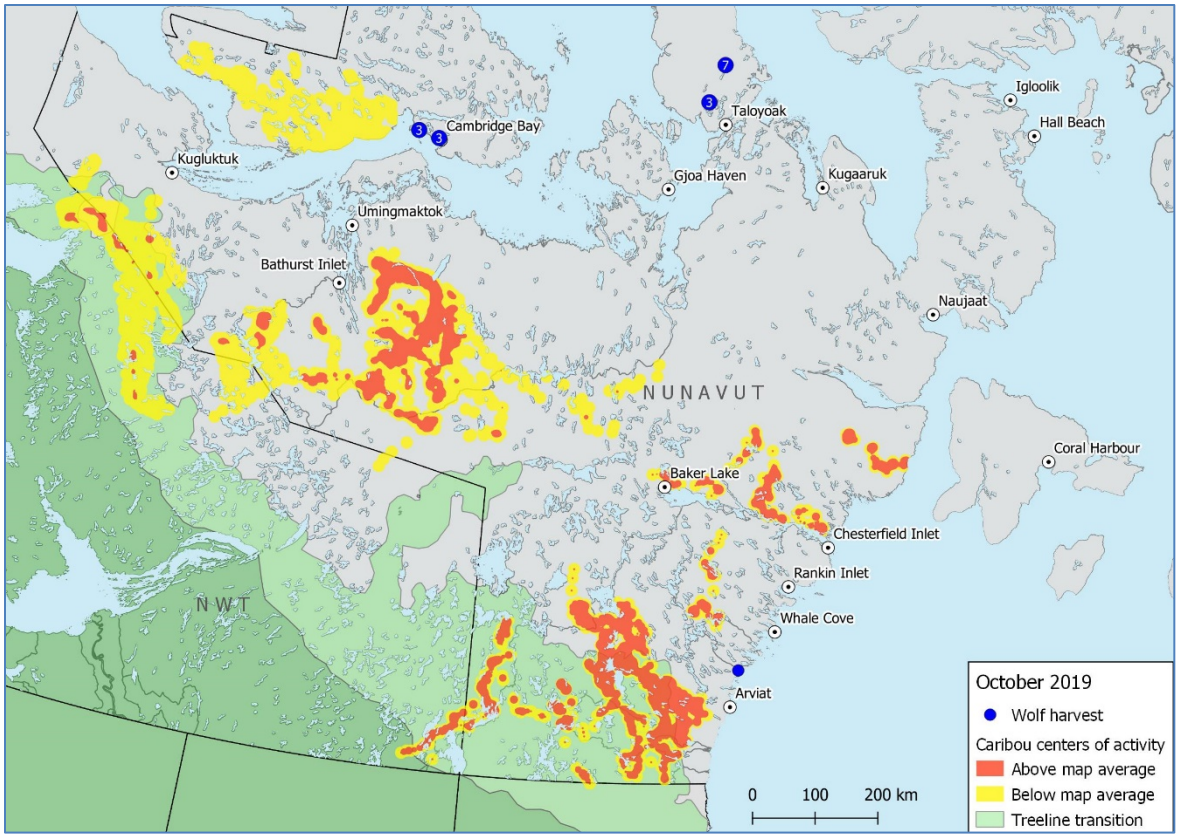

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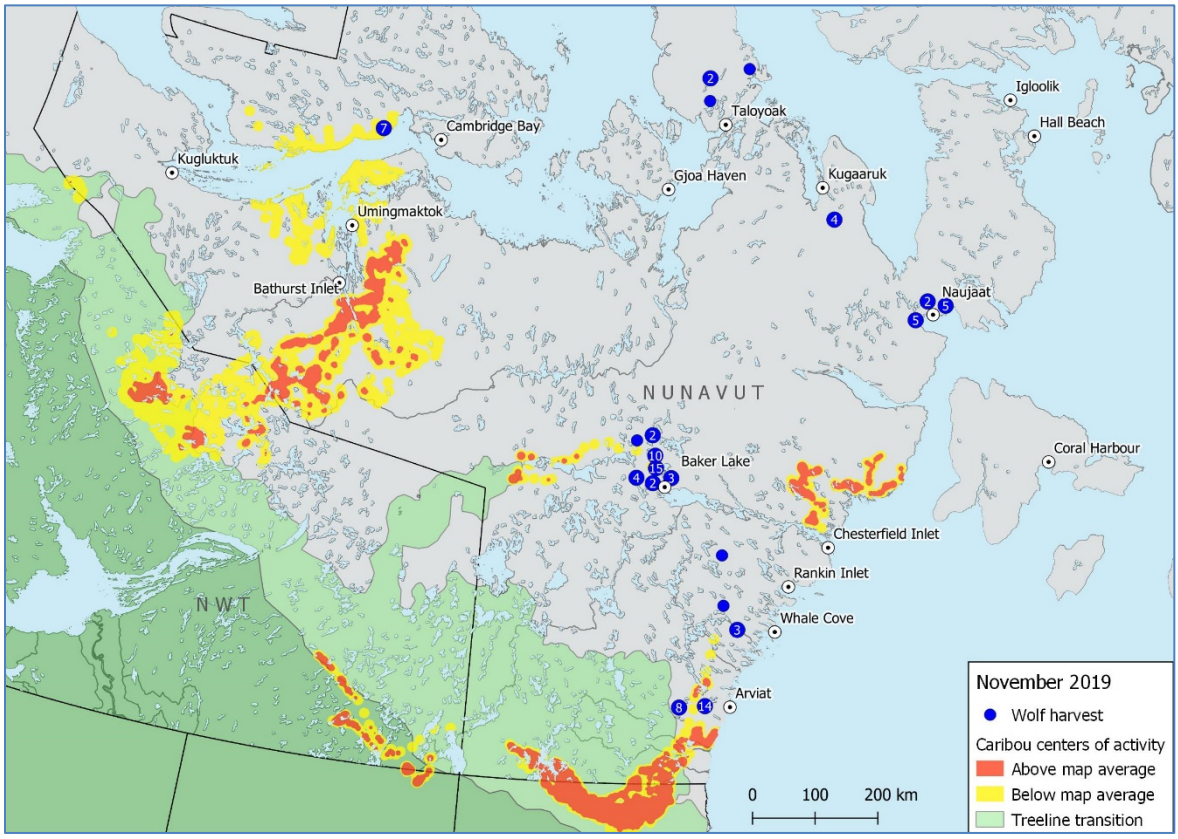

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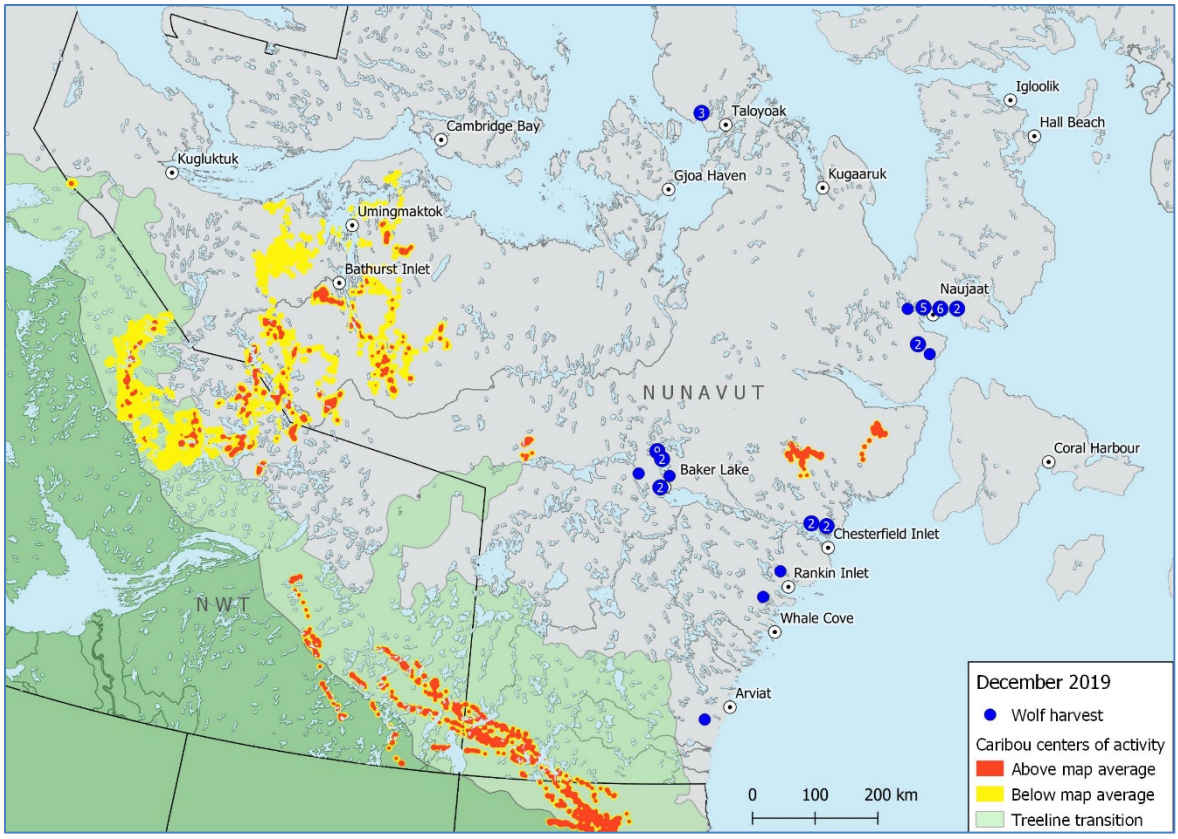

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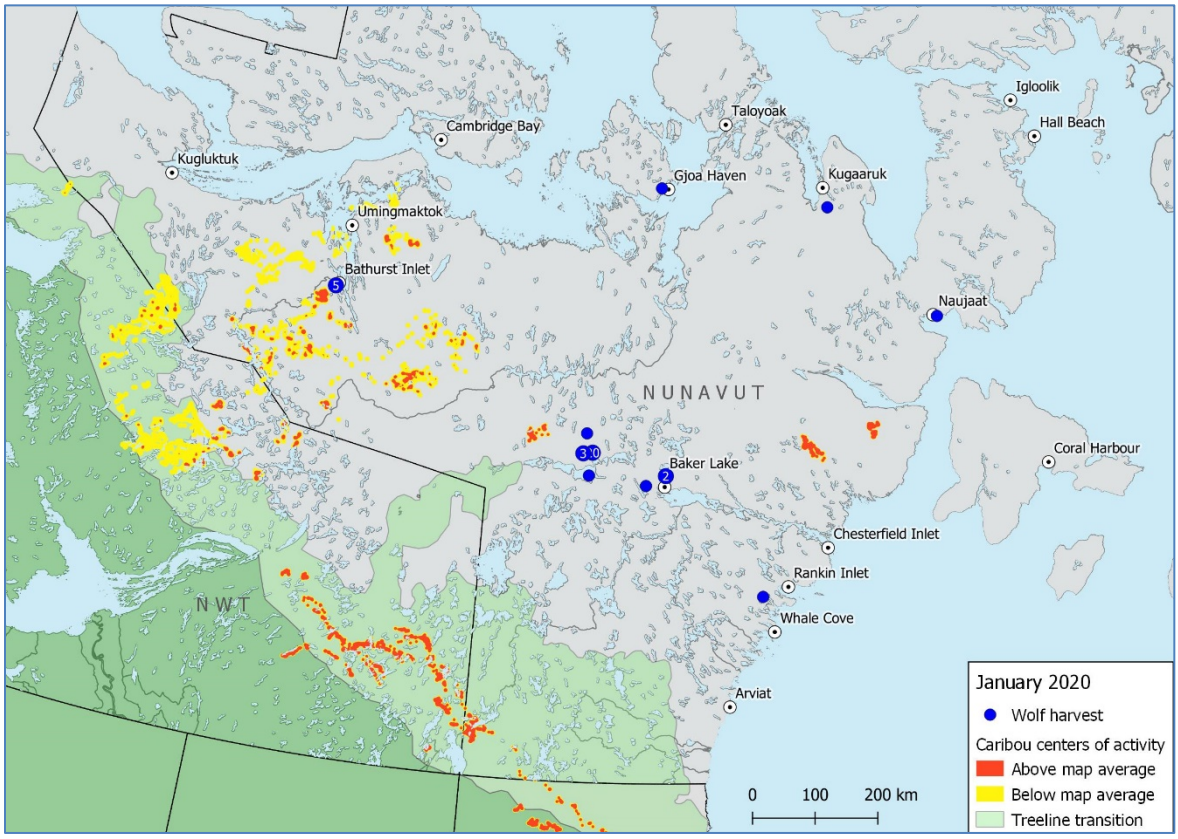

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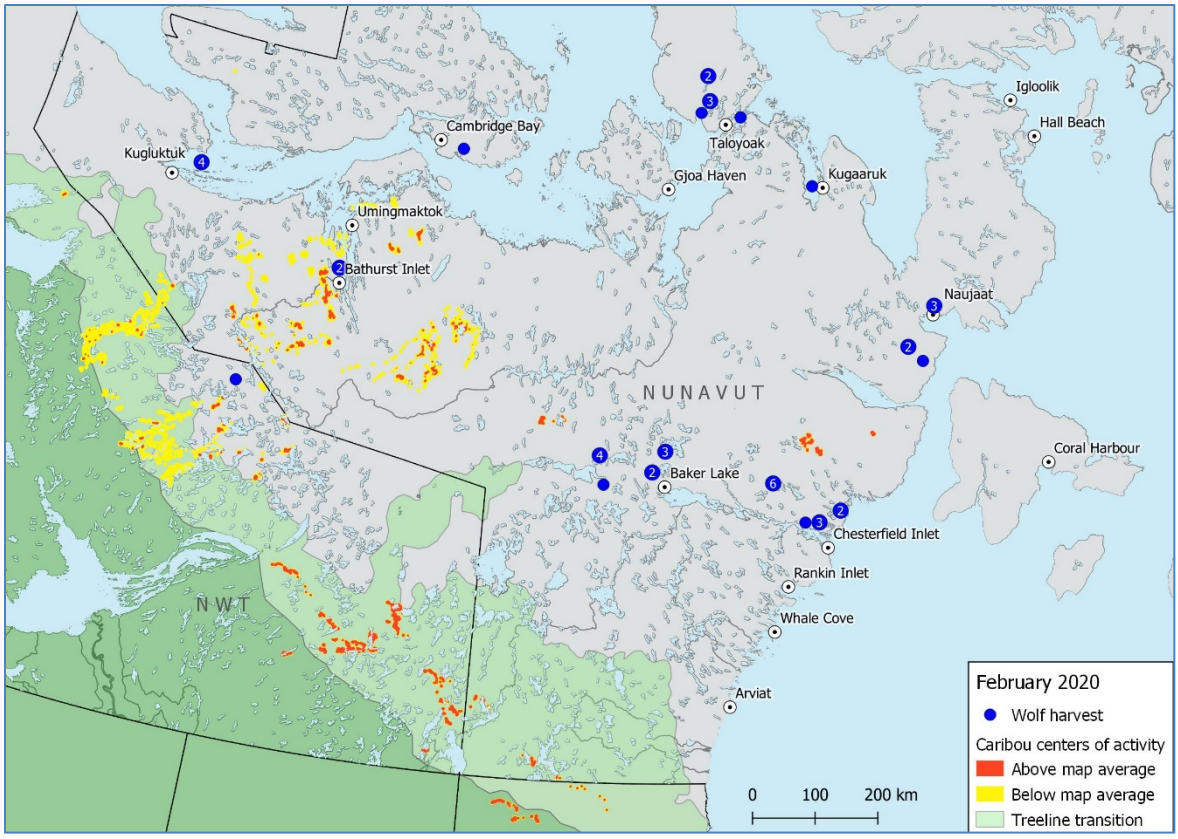

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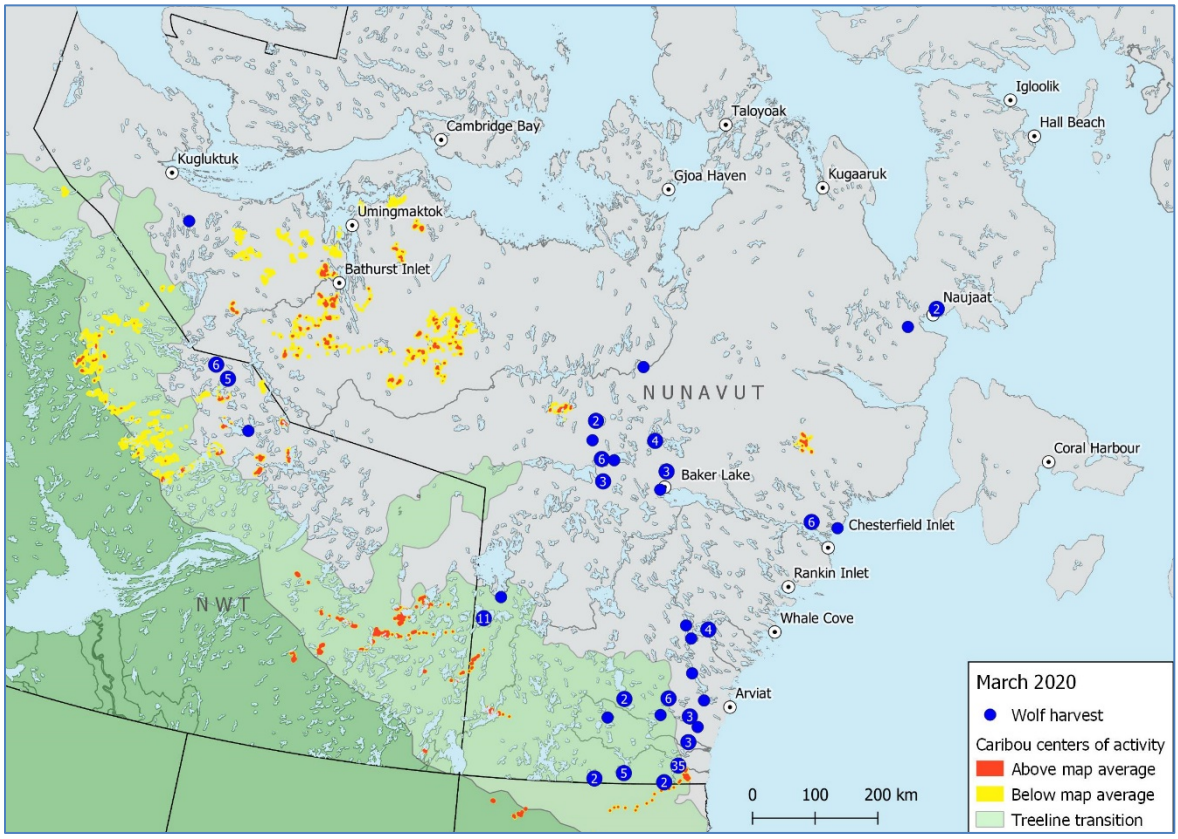

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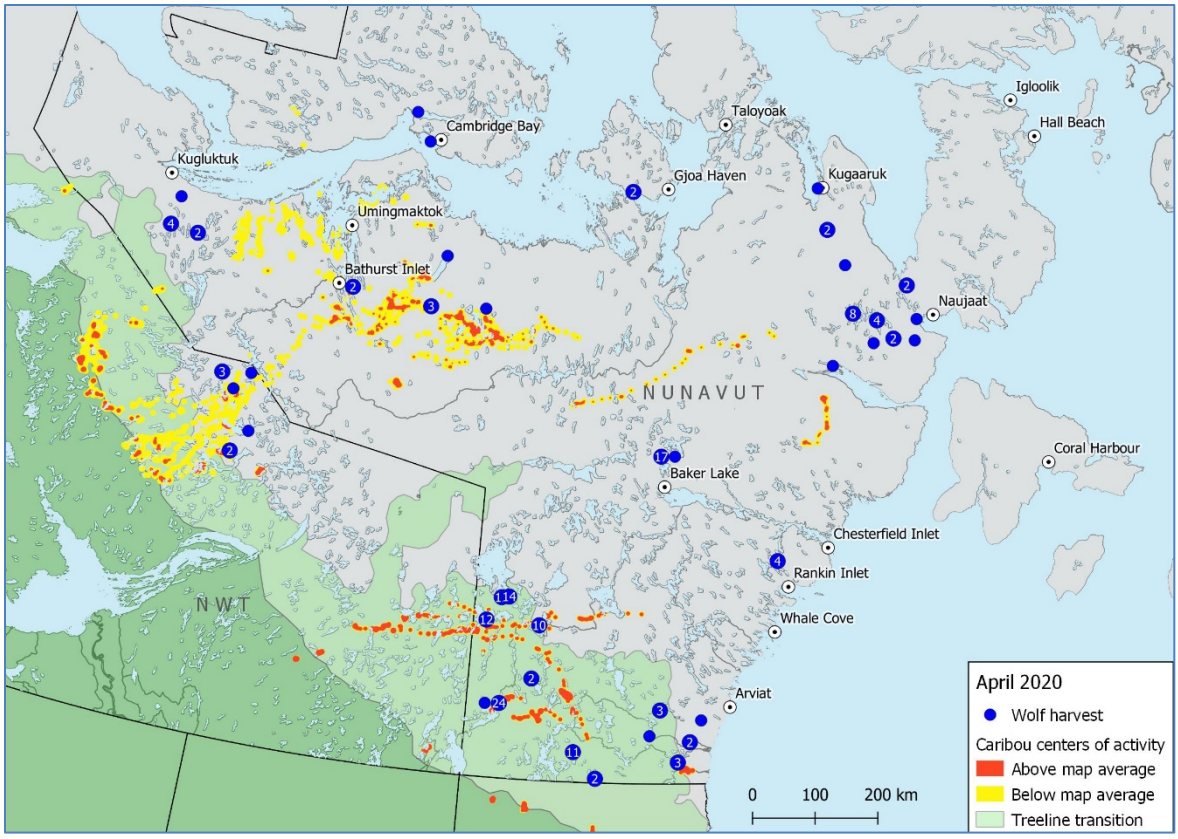

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November 2019
Wolf harvest
Caribou centers of activity
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Treeline transition

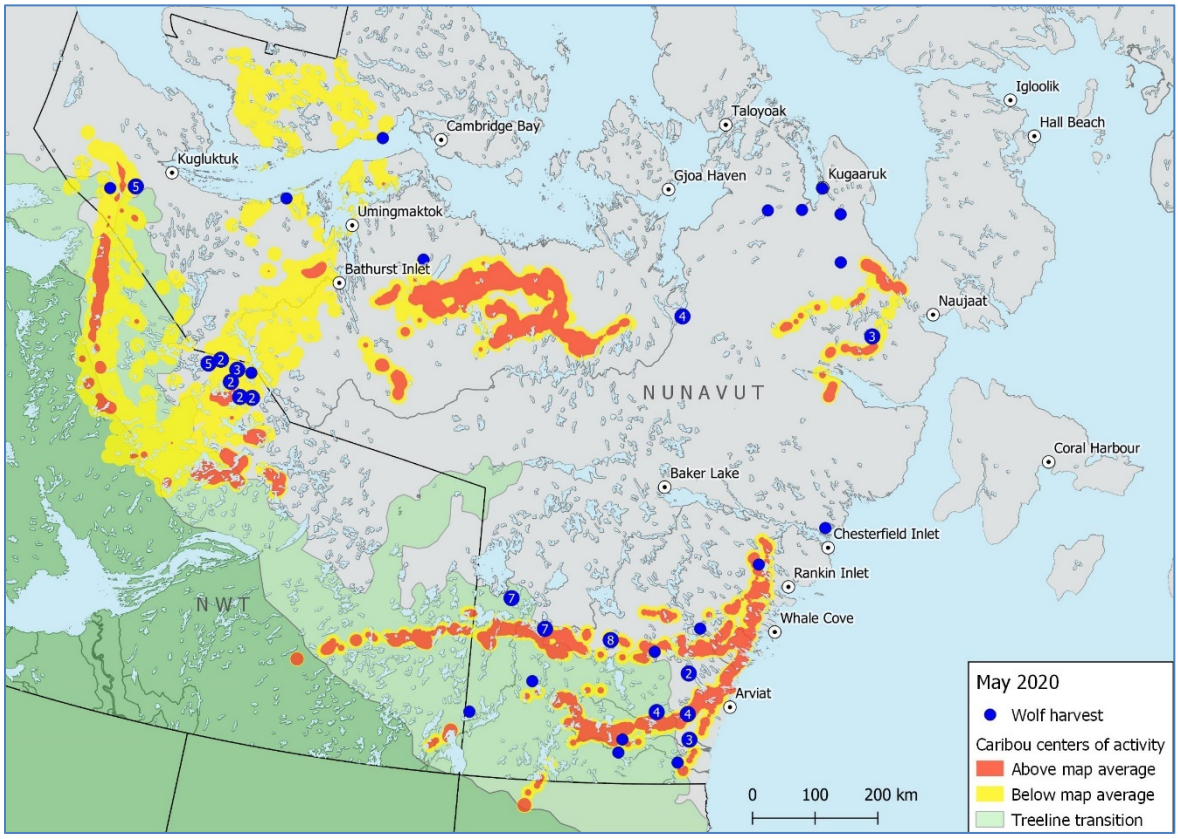

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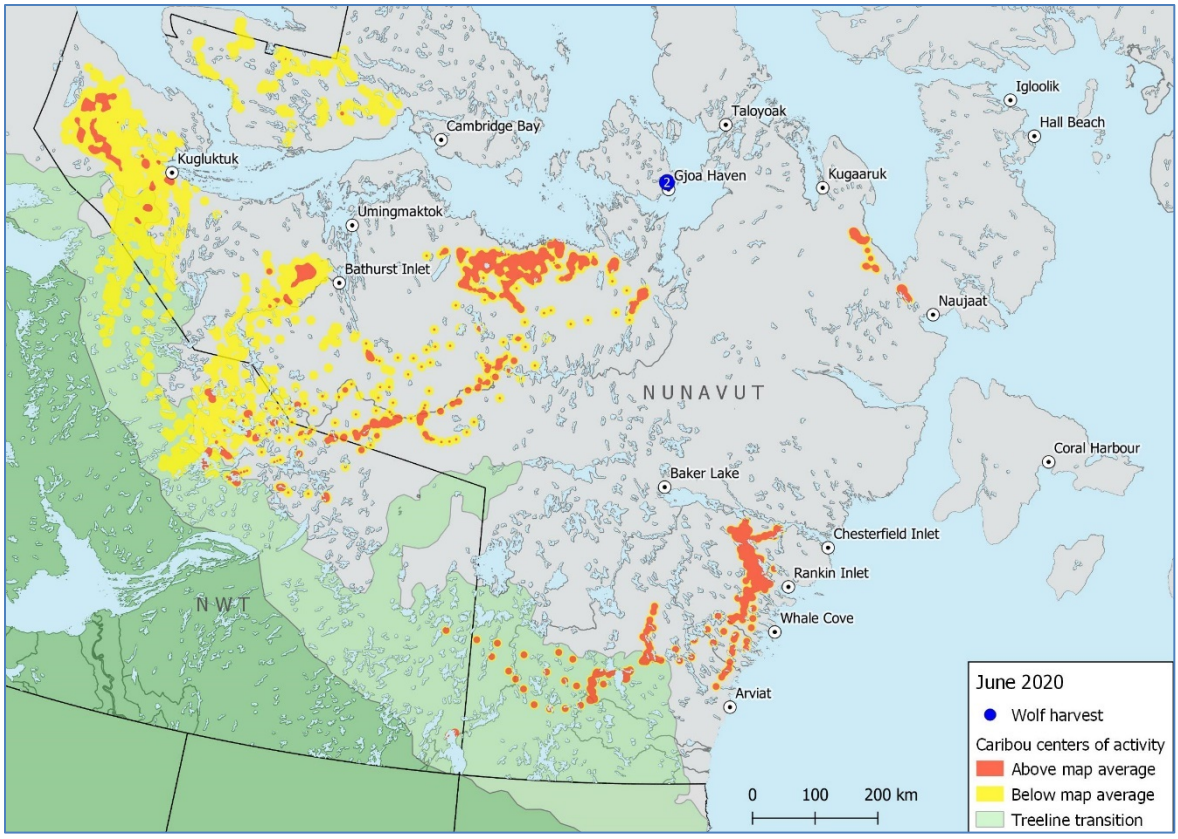

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Wolf harvest
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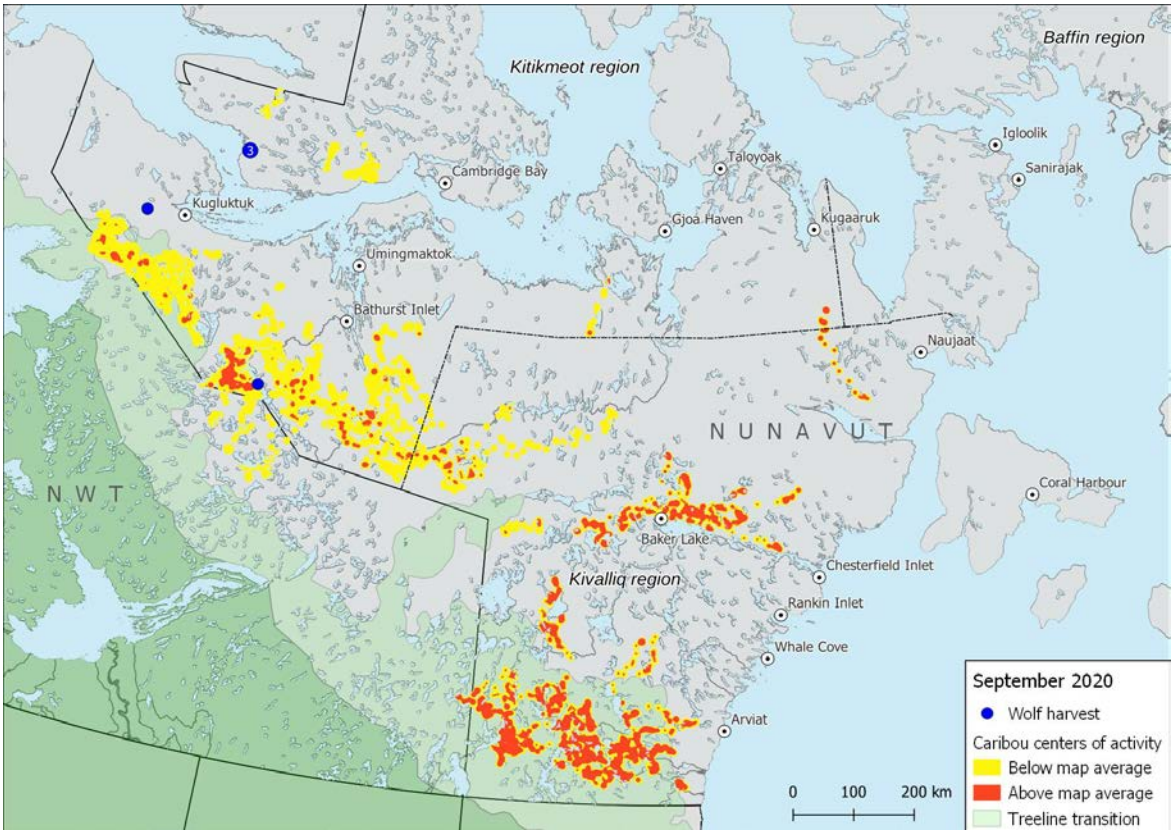

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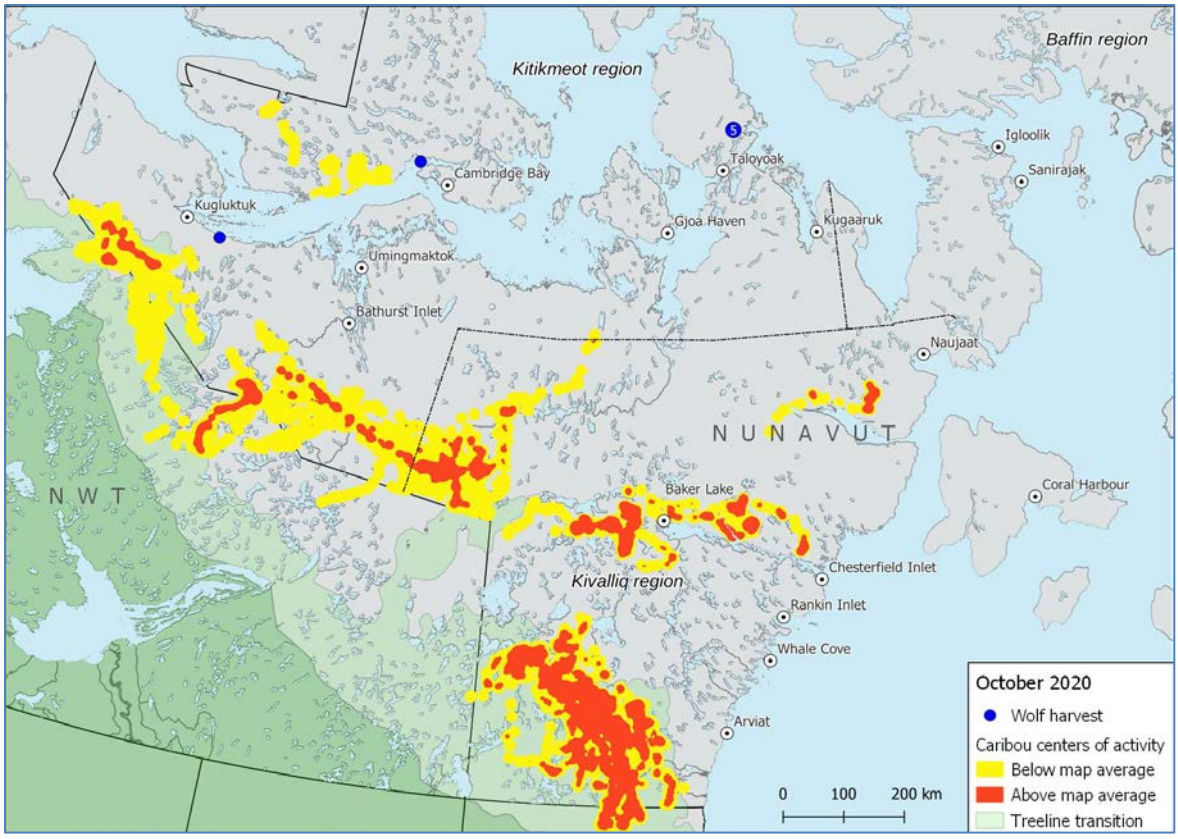

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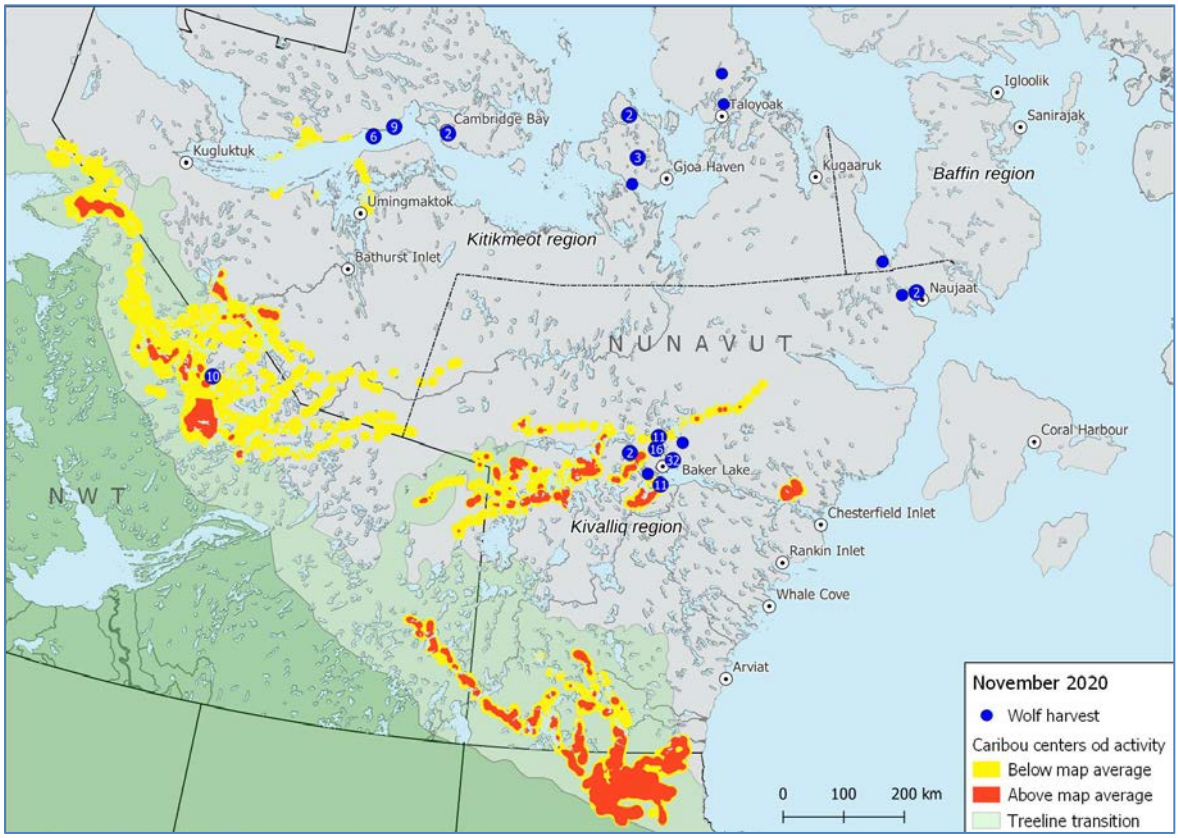

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April 2020
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Caribou centers of activity
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Treeline transition

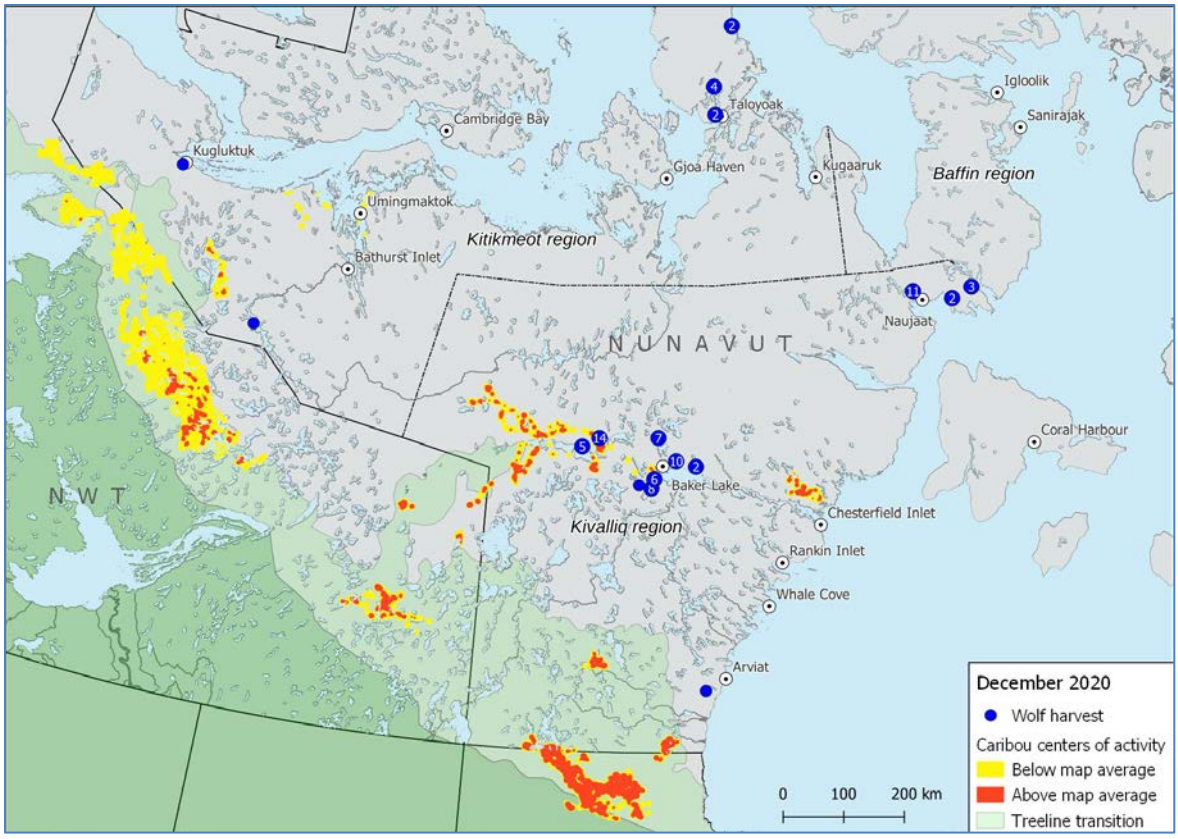

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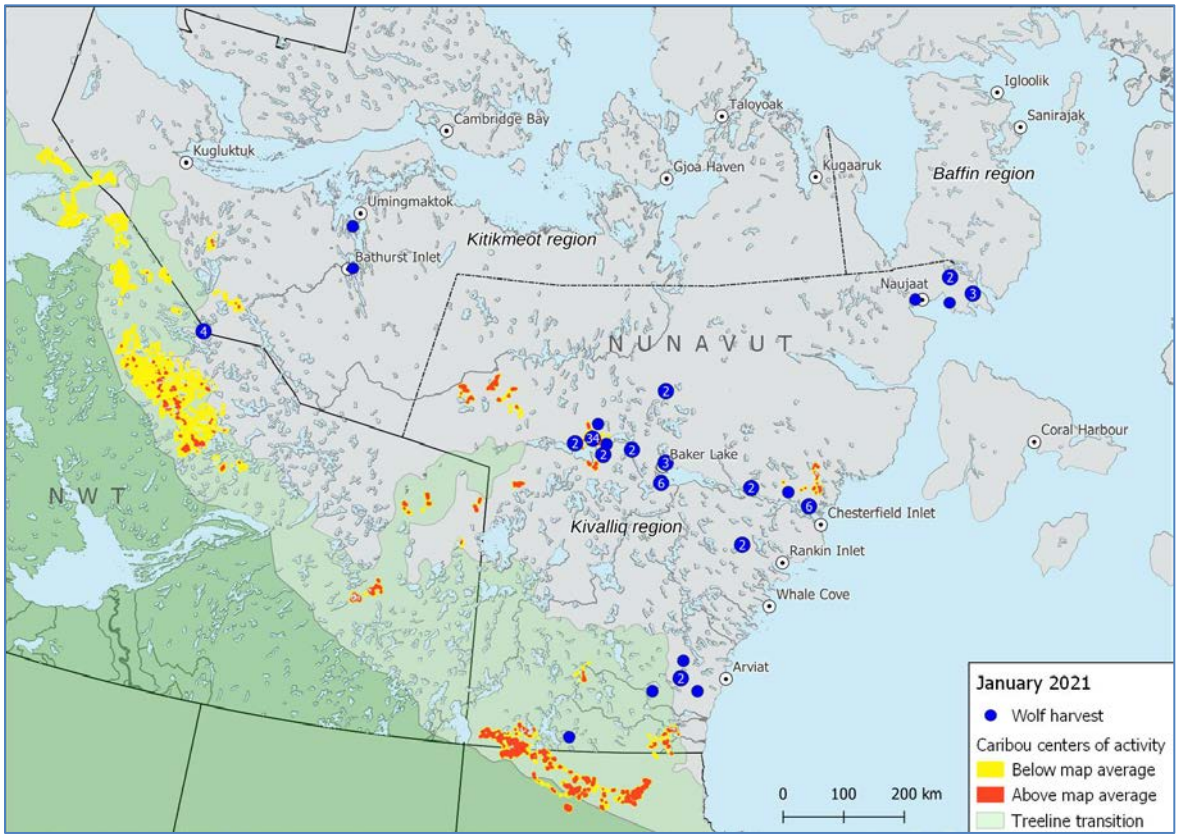

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June 2020
Wolf harvest
Caribou centers of activity
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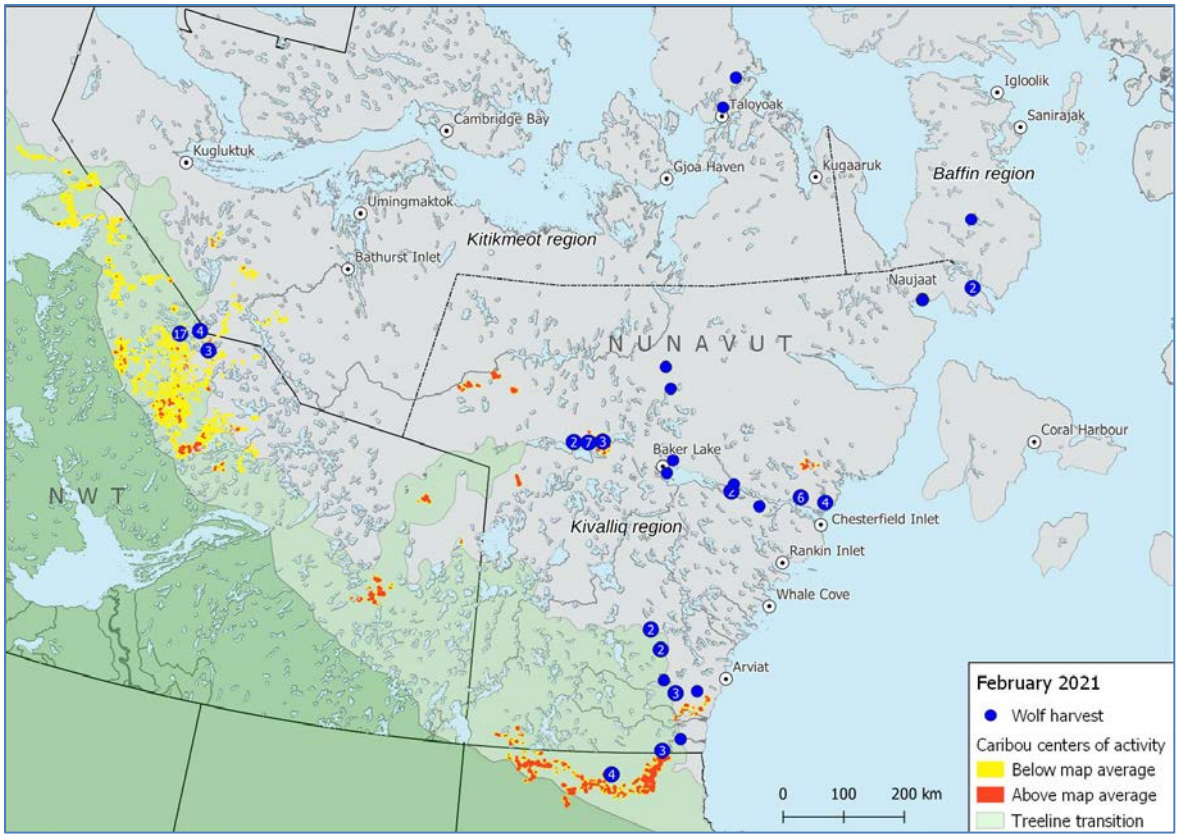

Kitikmeot region
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Bathurst Inlet
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Coral Harbour
Baker Lake
Kivalliq region
Chesterfield Inlet
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September 2020
Wolf harvest
Caribou centers of activity
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Above map average
Treeline transition

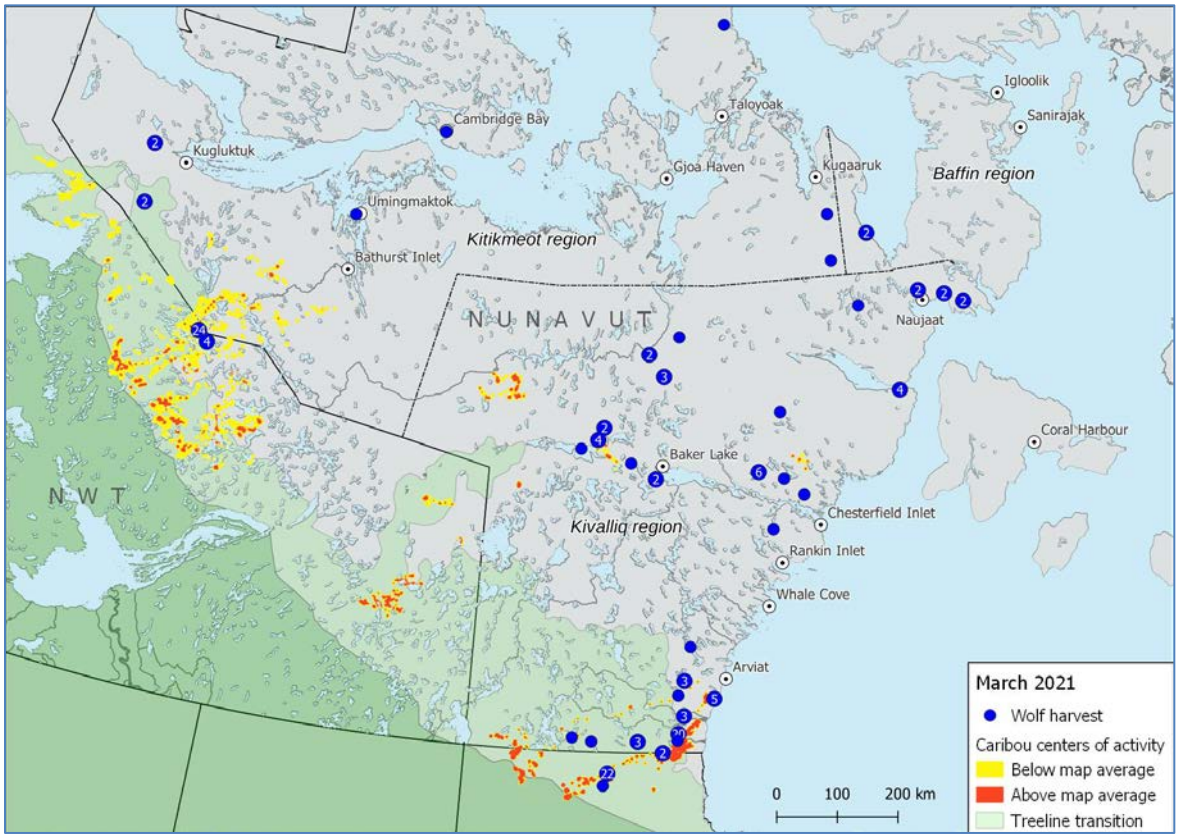

Kitikmeot region
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Sanirajak
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Baker Lake
Coral Harbour
Chesterfield Inlet
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Whale Cove
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October 2020
Wolf harvest
Caribou centers of activity
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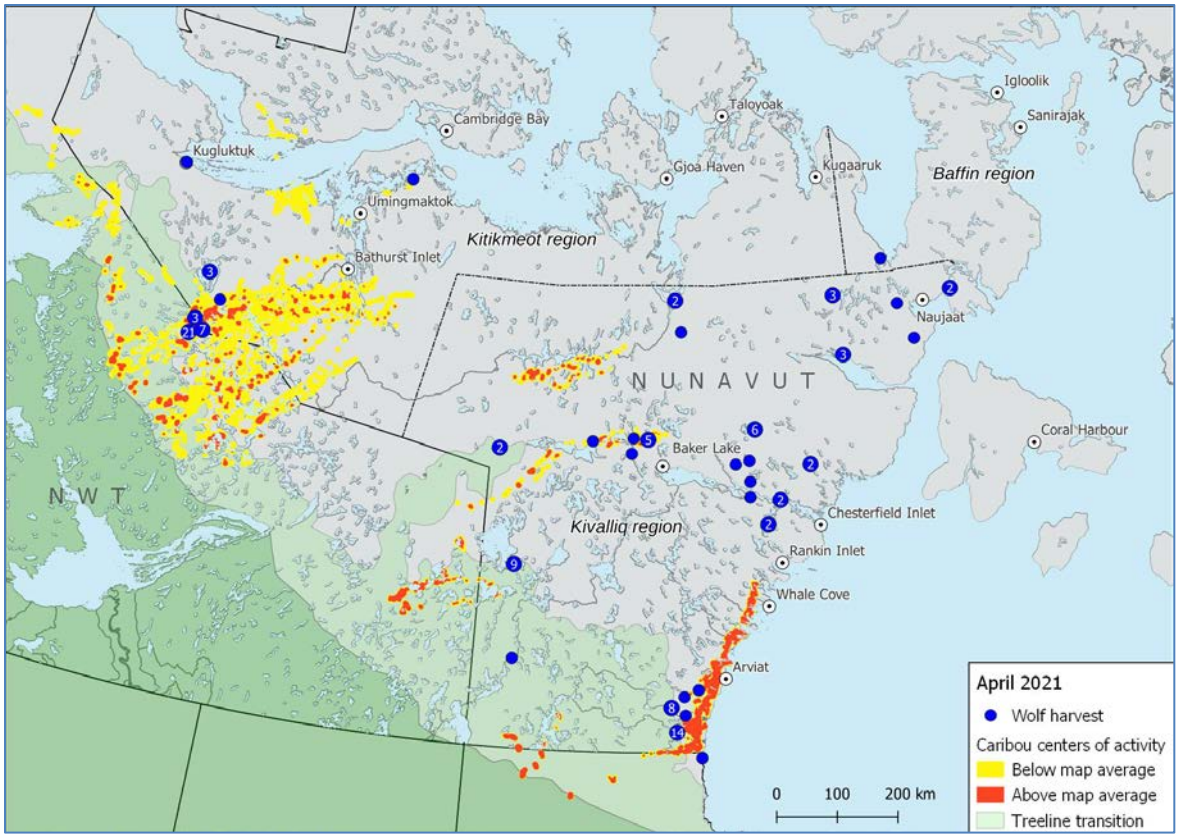

Kitikmeot region
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November 2020
Wolf harvest
Caribou centers od activity
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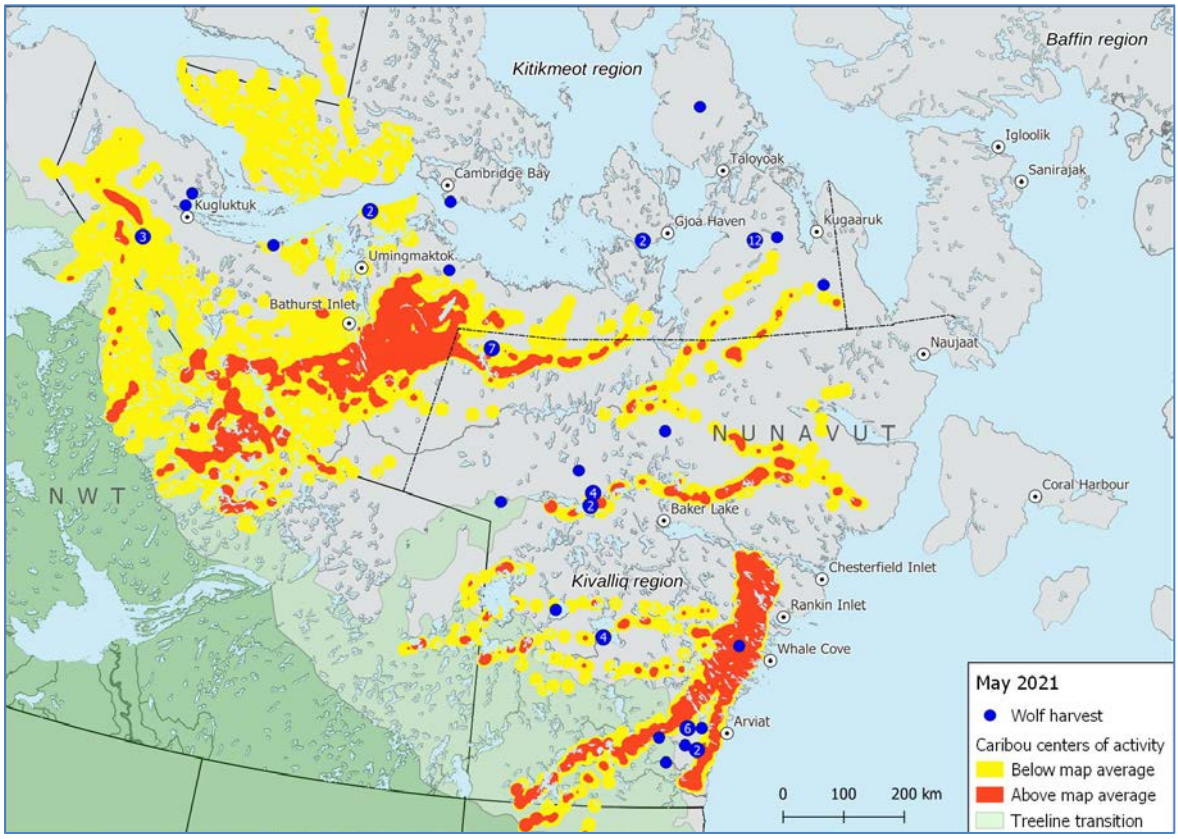

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Wolf harvest
Caribou centers of activity
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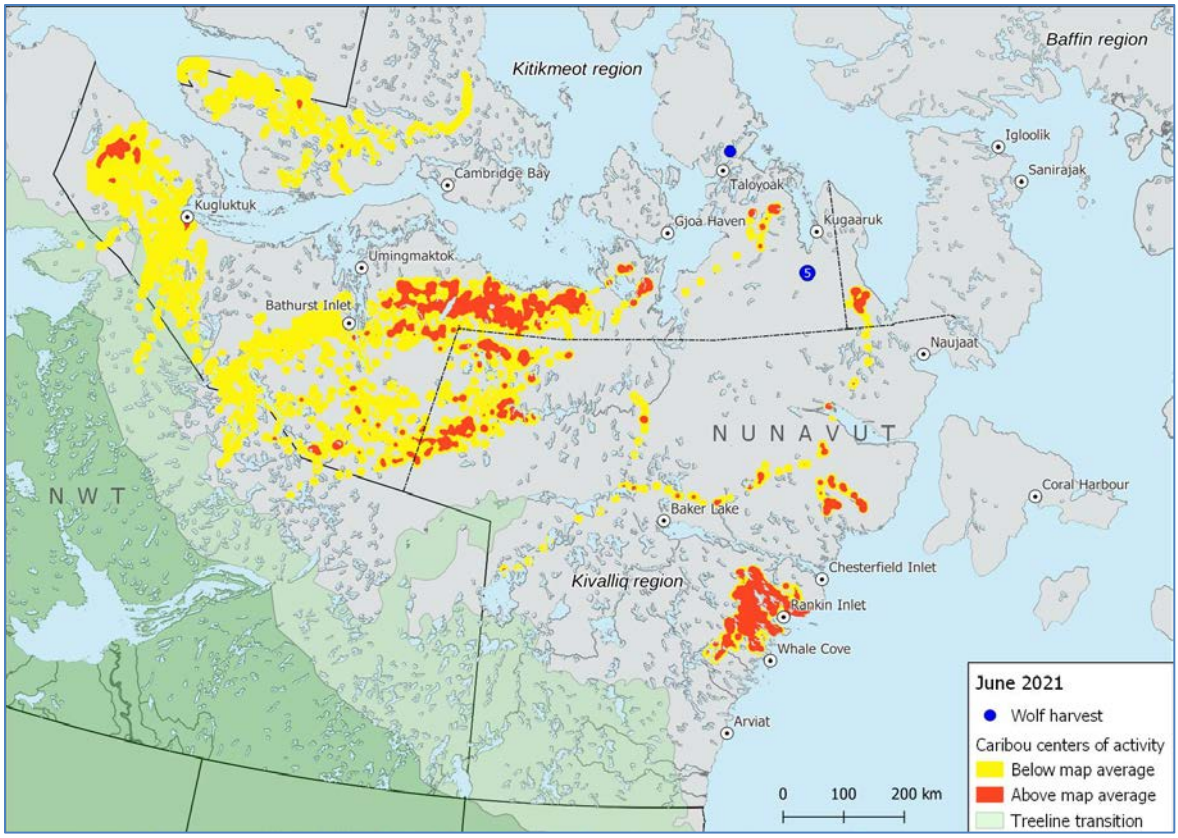

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Caribou centers of activity
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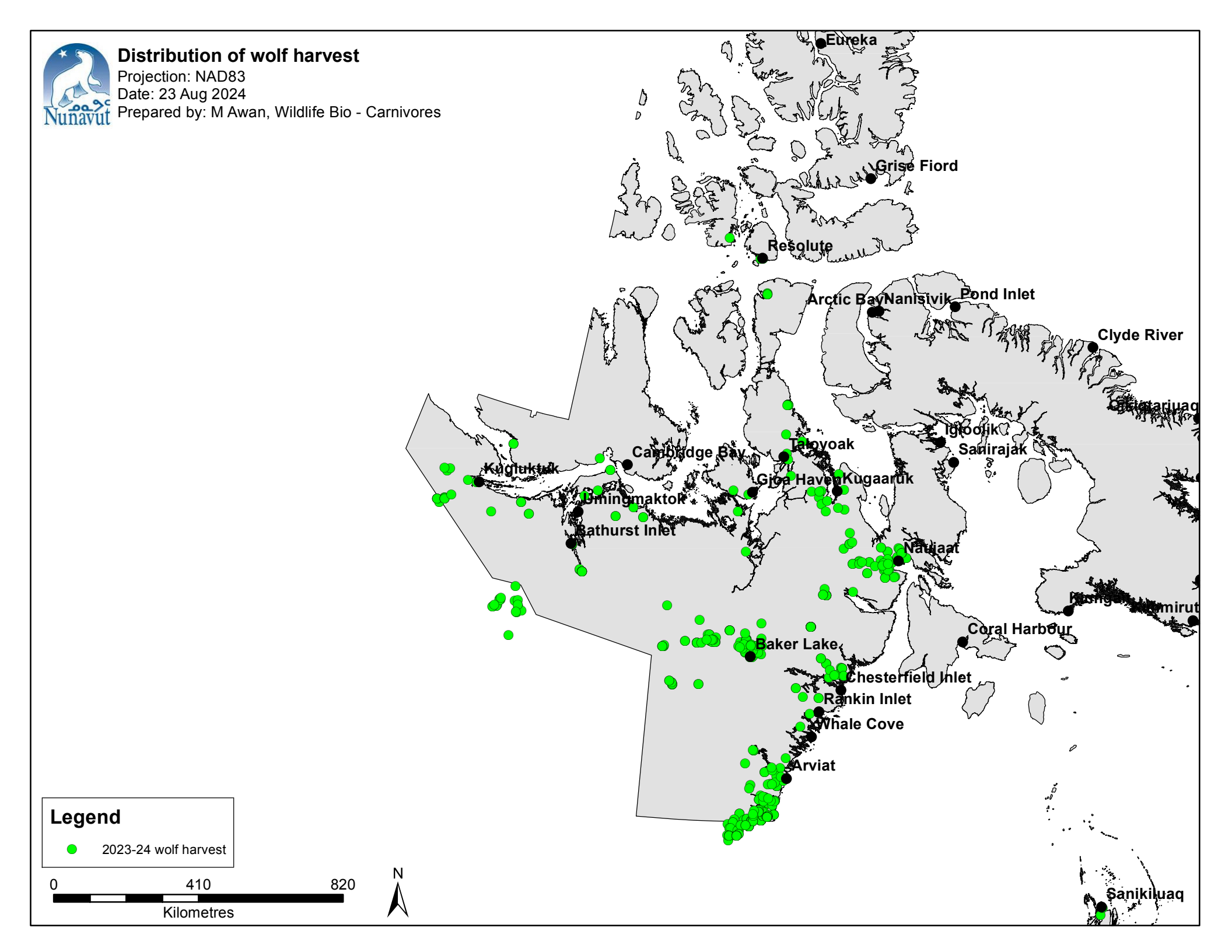

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February 2021
Wolf harvest
Caribou centers of activity
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Above map average
Treeline transition
0 100 200 km


Igloolik
Taloyoak
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Cambridge Bay
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Bathurst Inlet
Naujaat
NUNAVUT
Coral Harbour
Baker Lake
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Chesterfield Inlet
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Rankin Inlet
Whale Cove
Arviat
March 2021
Wolf harvest
Caribou centers of activity
Below map average
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Treeline transition
0 100 200 km


Igloolik
Taloyoak
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Chesterfield Inlet
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April 2021
Wolf harvest
Caribou centers of activity
Below map average
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Treeline transition
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Baffin region
Kitikmeot region
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Bathurst Inlet
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Coral Harbour
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May 2021
Wolf harvest
Caribou centers of activity
Below map average
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Kitikmeot region
Baffin region
Kivalliq region
Cambridge Bay
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Taloyoak
Gjoa Haven
Kugaaruk
Igloolik
Sanirajak
Naujaat
Coral Harbour
Baker Lake
Chesterfield Inlet
Rankin Inlet
Whale Cove
Arviat
N W T
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0 100 200 km
June 2021
Wolf harvest
Caribou centers of activity
Below map average
Above map average
Treeline transition


Distribution of wolf harvest
Projection: NAD83
Date: 23 Aug 2024
Prepared by: M Awan, Wildlife Bio - Carnivores
Eureka
Grise Fiord
Resolute
Arctic Bay
Nanisivik
Pond Inlet
Clyde River
Igloolik
Taloyoak
Cambridge Bay
Kugluktuk
Gjoa Haven
Kugaaruk
Umingmaktok
Bathurst Inlet
Naujaat
Coral Harbour
Baker Lake
Chesterfield Inlet
Rankin Inlet
Whale Cove
Arviat
Sanikiluaq
Legend
2023-24 wolf harvest
0
410
820
Kilometres
N

ᑲᑎᒪᔨᑦ ᓄᓇᕗᑦ ᐱᖃᓐᓇᕐᓂᖅᓴᓂᖅ
Building Nunavut Together
Nunavuliuqatigiingniq
Bâtir le Nunavut ensemble

WOLVERINE (*Gulo gulo*) HARVEST MONITORING IN NUNAVUT

Summary report

February 2025

Malik Awan
Department of Environment
PO Box 120
Arviat, Nunavut
X0C 0E0

The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment. Reference to any specific type of gear and/or equipment does not mean that it is endorsed by the authors or the institutions they represent.

Summary

Overall, Nunavut hunters submitted skulls and kill information from 691 harvested wolverines between 2019 and 2024 harvest years (1 July 2023 –30 June 2024 = 2024 harvest year). Most wolverines were harvested in the Kitikmeot region, in the vicinity of Kugluktuk, while relatively fewer were harvested in the eastern communities. Arviat and Baker Lake were the next highest contributors to the total harvest. Harvest data shows that 84% of wolverines were hunted with a rifle, while 16% were trapped. Sex ratio of the harvest was male biased with approximately two times as many males harvested than females. Juveniles and yearling predominated as 63% of the harvest while the other 37% were adults. The harvest was dominated by younger age classes during the early winter, while the harvest of adult individuals increased later in the winter. The spatial distribution, age, and sex structure of the harvest were typical of a healthy population. However, the relatively low number of mature females in the harvest is important to note and should be monitored closely.

ᖃᐟᐱᐣ (ᔪᓗ ᔪᓗ) ᐱᓴᐅᔪᓂᐣ ᖃᐅᔨᓴᖏᓂᖅ ᓄᓇᕗᒻᒥ

ᓇᐃᓭᖅᓯᒪᔪᑦ

ᐊᒋᖏᓪᓕᒪᖅᓴᓂᑦ, ᓄᓇᕗᒻᒥ ᐊᖓᔪᓇᓯᑎᑦ ᑐᓂᓯᓪᓕᐅᖅᓯᒪᔪᑦ ᓂᐊᕆᔨᖏᓪᒋᐅᓂᖅ ᑐᑎᖅᒋᐅᔪᒪᔪᓂᓪᓗ ᖃᐟᐃᒍᕆᔨᐃᑦ ᒥᐅᓴᓄᑦ 691–ᓂᐅ ᐊᖓᔪᖃᐃᓂᕆᓂᐅ ᐅᑭᐅᑦ 2019 ᐊᒻᒪ 2024 ᐊᑯᓂᓪᐊᓂ (1 ᔪᓕᐃ 2023 - 30 ᔫᓂ 2024 = 2024 ᐊᖓᔪᓇᓯᓇᒥ). ᐅᓄᕆᓂᖅᓴᐃᑦ ᖃᐟᐃᐅᑕᐅᔪᑦ ᖃᑎᖃᒥᐅᓂᐅ ᐊᖓᔪᐅᓕᐅᖅᓯᒪᔪᑦ, ᕆᑯᓪᓗᖅᑐᑦ ᖃᓄᓯᒋᓂᔪᓪᓚᐅᑦ, ᐅᓄᖖᑎᓂᖅᓴᐃᑦ ᐊᖓᔪᐅᓂᑯᓪᓚᐅᑎᑦ ᑲᓇᖅᑕᐅᔪᓂ. ᐊᕆᔨᖏᑕᑎᑦ ᐊᒻᒪ ᖃᒪᓂᑐᐊᖅ ᐊᖓᔪᖏᑎᑦ ᐅᓄᓭᖓᖃᑕᐅᓕᐅᖅᑐᑦ ᓇᐃᓴᖅᑕᐅᔪᓂ. ᐃᑎᖅᓯᒪᓂᖅ 84% ᖃᐟᐃᒍᕆᔨᕐᑕᐃᑦ ᕿᐱᐅᑎᒧᑦ ᐱᖁᐅᔪᐃᓂᕐᓂ, ᑕᐃᑲᑯᐊ 16% ᒥᑭᒋᐊᖅ/ᑭᖏᖃᑦᑕᓐᓄᑦ ᐱᖁᐅᓪᓗᑎᑲ. ᐊᖓᔪᐅᔪᓂ ᐊᖓᔪᓪᓗᐃᑦ ᐊᕆᓭᓪᓗᐃᓪᓗ ᓇᐃᓴᖅᑕᐅᑎᓪᓚᑦ ᐊᖓᔪᓪᓗᐃᑦ ᐅᓄᕆᓂᖅᓴᐅᓭᐅᖅᑐᑦ ᒪᕐᕈᐃᓪᓗᐊᖅᓯᒪᑲᑎᑲ. ᐃᓕᓭᔨᓴᖅᓯᒪᙵᑦᑐᑦ ᐊᒻᒪ ᑎᓪᑐᑐᐃᑦ ᐅᓄᕆᓂᖅᓴᐅᓭᐅᕐᒥᔪᑦ 63% ᐊᖓᔪᐅᔪᓭᓂᓕᓂ ᑕᐃᑲᑯᐊ ᐃᓕᓭᔨᐊᓂᑲᑦᑐᑦ 37%–ᖓᓪᓚᑎᑲ. ᐅᑭᐅᖓᓵᕆᔨᖏᑎᓪᓚᔭᒍ ᐃᓕᓭᔨᓪᓘᔨᓂᖅᓴᑦ ᐱᖁᐅᖅᑕᑦᑕᐅᓕᐅᖅᑐᑦ, ᐃᓕᓭᔨᖅᓯᒪᓂᖅᓴᓪᓚᓭᑦ ᐅᑭᐅᖓᓪᓚᓭᓪᓗᒍ ᐱᖁᐅᓪᓚᑎᑲ. ᓄᓇᓂ ᐅᖓᑕᓯᑲᑦᑕᐃᓂᖅ ᐱᖁᐅᓪᓚᑎᑲ, ᐅᑭᐅᑦ, ᐊᕆᓇᖅ/ᐊᖓᔪᓪᓚᔫᓴᒋᑎᑕ ᓇᐃᓴᐃᓵᔨᓯᑦ ᑕᒪᐃᓕᓂᓂ ᐅᓂᒃᑯᐃᓵᔨᐅᑕᐅᓕᐅᖅᑐᑦ. ᑭᓯᐊᓂᑕᐅᖅ, ᖃᑐᖅᑕᐅᑐᓂᓇᖅᑐᓂᓂᓂ

Qalvik (*Gulo gulo*) anguniaqtunun munaridjutikhaq Nunavunmi

Naittuq Naunaijauti

Tamainun, Nunavunmi anguniaqtut tujuqtut niaqunik humilu tuquhimajatigut kangiqhidjutit hamanga 691 anguniaqtaujut qalviit qitqani 2019 unalu 2024 anguniarutini ukiuni (1 Taaqhivalirvia 2023 –30 Imaruqtirvia 2024 = 2024 anguniaqvingmi ukiungani). Tamavjaita qaviit angujaujut Kitikmeot aviktungnianin, qanilruani Kugluktup, ikitut angujaujut kivataanin nunallaanin. Arviat hamanilu Qamanittuaq tugliujut amigainirit ikajuutait atautimun anguniaqtamingnun. Angunianikkut naunaijautit takunaqtut tapkua 84 pusat qalviit hiquutikkut tuquhimajut, imaalu 16 pusat naniriaqhimajut. Qanuriniitigut amigainirit angujaujut angutiluat naamavjaktuq malruiqtughugu amigainiit angutit angujauluaqtut angnalurnin. Qalviat angujaujut imaa 63 pusat anguniaqtaujut aallat 37 pusat ininirit. Angujaujut amigaitqijaujut qalviat ukiungitigut naunaijaqhimajut ukiuq atulihaaqtilugu, angujaujut ininirnit atautit amigairjuumijun nungutinnagu ukiumi. Ilaujut tunijaujut, ukiungit, qanuriniit qanuqtut anguhimajut aanniaqtuqangitut. Kihimi, ikitpalaat qaffiuniit angnaluit angujaujuni aghuungnaqtut ilihimalugit munarijaujukhallu qanitukkut.

Contrôle de la récolte de carcajou (G*ulo gulo*) au Nunavut

Résumé

En tout, les adeptes de la chasse du Nunavut ont remis les crânes et les données provenant de 691 prises de carcajous lors des années de récolte de 2019 à 2024 (à titre indicatif, l'année de récolte de 2024 s'étend du 1er juillet 2023 au 30 juin 2024). La majeure partie de la récolte a eu lieu dans la région du Kitikmeot, aux environs de Kugluktuk, alors que le nombre de carcajous récoltés dans les localités de l'est était relativement peu élevé. Arviat et Baker Lake figuraient au deuxième rang parmi les grands contributeurs à la récolte totale. Les données de récolte révèlent que 84 % des carcajous ont été chassés au fusil et 16 % ont été piégés. Le ratio mâle-femelle de la récolte était biaisé en faveur des mâles, qui en représentaient environ les deux tiers. Les jeunes carcajous et les petits d'un an constituaient la majeure partie de la récolte, représentant 63 % des prises alors que les prises adultes en représentaient 37 %. En préhiver, les carcajous récoltés provenaient surtout de la population de jeunes, et c'est aux mois de mars et d'avril qu'on a vu une croissance de la récolte d'adultes. La répartition spatiale, l'âge et la répartition par sexe de la récolte étaient caractéristiques d'une population en bonne santé. Cependant, on souligne que la récolte comportait un nombre relativement peu élevé de femelles adultes, et cette population devrait faire l'objet d'une surveillance étroite.

Table of Contents

1.0 INTRODUCTION.....1
2.0 METHODS.....3
3.0 RESULTS AND DISCUSSION.....4
4.0 ACKNOWLEDGMENTS.....12
5.0 LITERATURE CITED.....13

List of Figures

Figure 1. Reported wolverine harvest in Nunavut, from 2019 to 2024. 4

Figure 2. Distribution of reported wolverine harvest in Nunavut, from 2019 to 2024..... 5

Figure 3. Wolverine reported monthly harvest in Nunavut, from 2019 to 2024. Label indicates total number of wolverines harvested per month.. 6

Figure 4. Age and sex structure of the reported wolverine harvest in Nunavut, from 2019 to 2023.. 7

Figure 5. Monthly reported wolverine harvest per age class in Nunavut, from 2019 to 2023. 10

List of Tables

Table 1. Male to female sex ratio by age in the reported wolverine harvest in Nunavut, from 2019 to 2023. 8

Table 2. Age and sex distribution of reported wolverine harvest in Nunavut, from 2019 to 2023. 9

1.0 Introduction

For thousands of years, Inuit have relied on the harvest of wildlife for food, clothing, and trade. In Nunavut, the furbearer harvest for clothing and for income is a seasonal and traditional activity, where opportunities for other employment are scarce. Under the Nunavut Agreement, furbearer harvest rights are held by Inuit beneficiaries, non-Inuit who harvested furbearers legally in Nunavut settlement area prior to 1981, and non-Inuit whose application has been approved and recommended by local Hunters and Trappers Organizations (HTOs).

In Nunavut, wolverines (*Gulo gulo*) occupy almost all areas of the territory and are classified as both a furbearer and a big game animal under the Nunavut Agreement. They are an important furbearer in Nunavut's culture and economy and have traditionally been considered a vital resource for hunters because of the beauty and frost resistant properties of the fur, which make it unique and quite valuable (Hash 1987). Wolverine fur is highly prized in local communities as ruffs or trims on parkas (Cardinal 2004). Unlike most provinces, hunters and trappers in Nunavut do not have registered or traditionally exclusive family trap lines or hunting areas, so wolverines are generally harvested opportunistically wherever people travel (Mulders 2000) or harvested while hunting other game. Wolverine hunting in Nunavut is mainly by firearms from snow machines while in other provinces trapping is the major method (Lee 2016).

Wolverine densities are believed to be moderate in the western mainland of Nunavut but low on the Arctic islands and in the eastern mainland (Slough 2007, Species at Risk Committee 2014). Using DNA-based mark-recapture, Awan and Boulanger (2016), Awan et al. (2018) estimated density from 1.6 to 4.4 wolverines/1,000 km² in the Kivalliq region and 4.14 wolverines/1,000 km^2 in the Napaktulik Lake area in the Kitikmeot region (Awan et al. 2020). Inuit observations and recent harvest reports suggest that wolverine numbers in Nunavut are either stable or slightly increasing, and the species may be expanding its range eastward and northward (Awan et al. 2012, COSEWIC 2014). The wolverine has been assessed as a species of Special Concern in Canada

and in 2018 was listed as the same under the *Species at Risk Act* (*SARA*). The wolverine is generally described as a scavenger (Banci 1987) and an opportunistic predator throughout its range, occurs in low densities, have low birthing and recruitment rates, are sensitive to human disturbance, and require large secure areas to maintain viable populations (Magoun 1985, Mattisson et al. 2011). It is considered a wilderness species and potential indicator of ecosystem health (Carroll et al. 2001). Mulders (2000) suggests that the arctic tundra contains large undisturbed tracts of habitat that may act as reservoirs to maintain harvestable populations of wolverine in Nunavut.

Nunavut contributes substantial numbers to the national harvest even though ecological data for tundra wolverine are sparse, especially in the northeastern edge of the species distribution. Currently, there is no quantitative limit on their harvest by Inuit and there is no requirement for hunters to report their harvest. The only mechanism for tracking the harvest or pelts is the Government of Nunavut's Department of Environment (GN-ENV) fur-pricing program, which misses pelts, which are used locally or sold as raw frozen wolverine hides privately; the actual harvest is unknown. The current collection of wolverine skulls and kill information is an attempt to estimate the harvest, to monitor the age and sex of the harvest.

2.0 Methods

Each year we obtained the skull and a small piece of skin (~2.5 x 2.5 cm) with fur from hunters with the assistance of HTOs and Conservation Officers (COs). Hunters across Nunavut voluntarily reported and provided skull and skin samples from harvested wolverines via their local COs. A $50 subsidy was provided to hunters for each skull brought back to the COs office to encourage the reporting and return of samples. Recently, increased activity in the online buying and selling of wolverine skulls has led to less reporting under this program. The online skull buyers offer higher prices for skulls than our subsidy amount of $50. From January 2025 GN-ENV increased the subsidy amount to $100 per wolverine.

For each wolverine sample collected during the course of this study, COs filled out sample collection forms with information from hunters about the harvest location, date, wolverine sex, and method of harvest, and their opinion of the current abundance trend of the local wolverine population (decreasing, stable or increasing). The skinned skulls were stored frozen, and were thawed at room temperature for examination in early May each year. We extracted a lower canine and sent to Matson's Laboratory LLC (Milltown, MT, USA) for age estimation using cementum analysis. This technique is based on the cyclic nature of cementum growth in teeth forming annular patterns of different darkness depending on the season (Matson 1981).

Following Banci and Harestad (1988) and Vangen et al. (2001), individuals were then grouped into three age classes: juvenile (0-1 year, date of birth is set to March 1st), yearling (1-2 years) and adult (≥ 2 years).

3.0 Results and Discussion

A total of 691 wolverines were reported as harvested from July 2018 to June 2024, 397 from the Kitikmeot, 290 from the Kivalliq and 4 from the Qikiqtaaluk region (Igloolik and Sanirajak) (Fig. 1). According to our sample collection program, reported wolverine harvest in Nunavut has been fluctuating and the harvest report rate for this period (2019-24) appears to be lower compared to early 2000s in the Kitikmeot region (Awan et al. 2012).

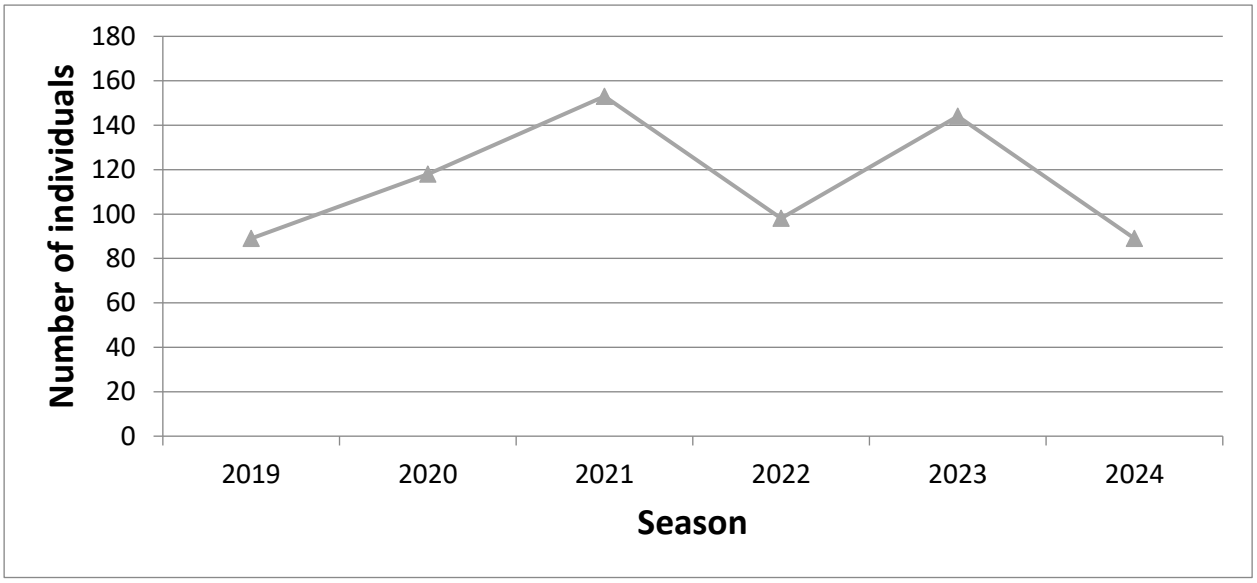


Figure 1. Reported wolverine harvest in Nunavut, from 2019 to 2024. Note that the harvest year is assigned as the year at the end of the regulations' harvesting season, for example, 1 July 2018–30 June 2019 = 2019 harvest year.

The harvest was unevenly distributed throughout the territory, with most wolverines having been harvested in the western Kitikmeot. Harvest in the vicinity of Kugluktuk was particularly intense, and along the traditional travel route from the Kugluktuk to the Contwoyto Lake. Relatively fewer wolverine were harvested in the eastern communities,

while Arviat and Baker Lake were the next major contributors to the total harvest (Fig. 2). Comparatively, high wolverine harvest happening out of Kugluktuk, when the caribou herds wintered nearby, was due to hunters spending more time on the land to hunt caribou. While hunting caribou, harvesters usually pursued wolverine when they saw wolverine or found fresh tracks.

Wolverine harvest distribution shows that harvest took place over a wider range on the Nunavut mainland, but it tended to be concentrated near communities. The surrounding areas without hunting may act as refugia or reservoirs that produce wolverines that move into the hunted areas and sustain the harvest (Mulders 2000, Lee 2016).

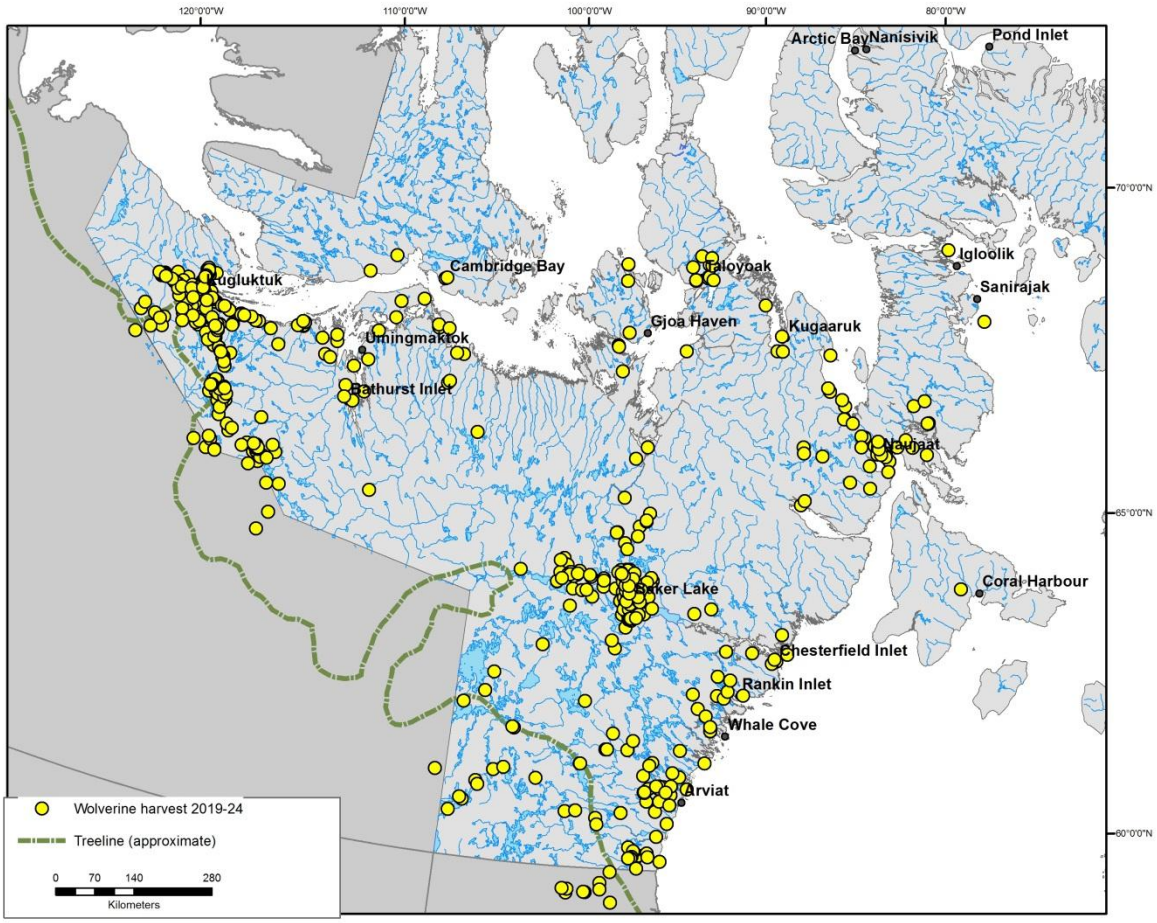


Figure 2. Distribution of reported wolverine harvest by Nunavut hunters from 2019 to 2024. Symbol may represent >1 wolverine harvested at that location.

Most wolverines were harvested between November and April each year (Fig. 3), when the fur is in prime condition. Harvest data shows that approximately 84% of wolverines were hunted with a rifle and 16% animals were trapped. Of the trapped animals (n = 109), 94% were killed in quick-kill traps and 6% in leg hold traps. In the Kivalliq region, only five hunters trapped 12 wolverines, while in the Kitikmeot region 97 wolverines were trapped by 15 hunters.

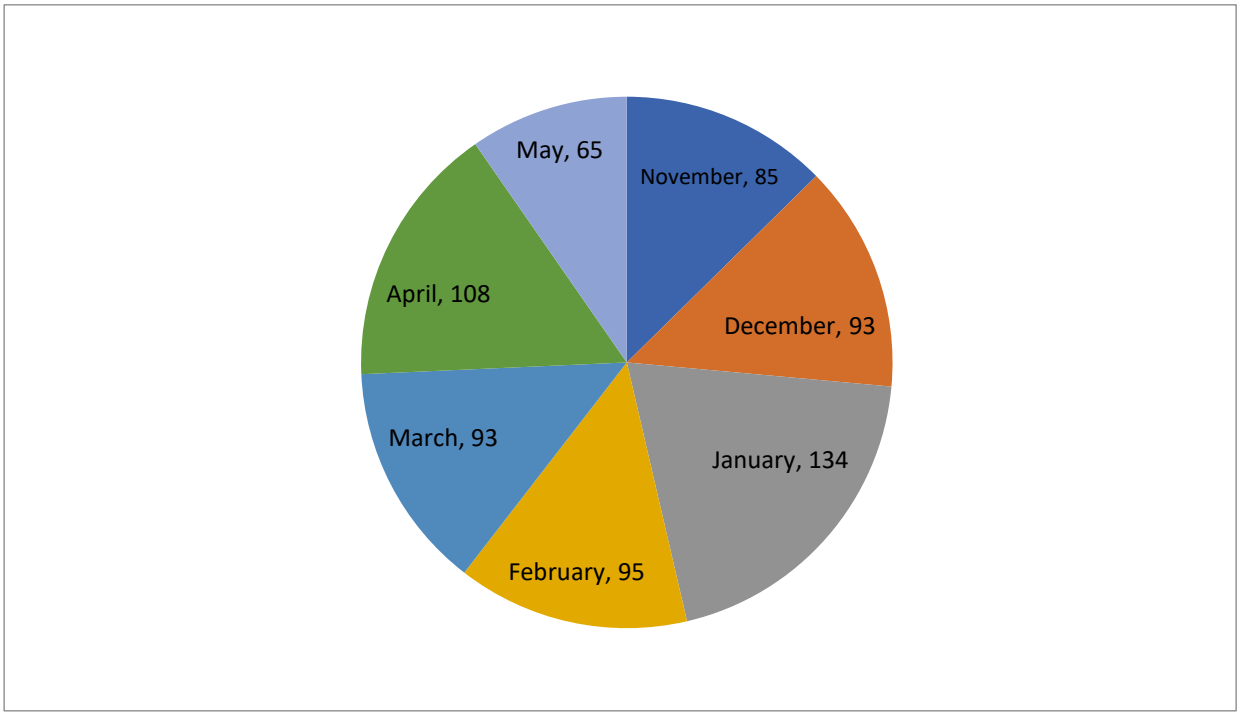


Figure 3. Wolverine reported monthly harvest in Nunavut, from 2019 to 2024. Label indicates total number of wolverines harvested per month.

The male:female ratio of the harvest was highly biased towards males (Fig. 4) with approximately twice as many males harvested than females (ratio = 1.7 and 2.1 in the Kitikmeot and the Kivalliq respectively), a typical figure for northern Canada (Mowat et al. 2020). We assumed sex ratio at birth in the population is 1:1, however, it is difficult to

distinguish if this reflects the actual sex ratio of the population or a difference in vulnerability by sex. Lee (1994) also observed a 2:1 male to female ratio in the western Kitikmeot harvest in the early 1990s. However, his observations showed a strong male bias in the younger age classes (< 2 yo) while the adult sex ratio was not different from a 1:1 ratio. Examining the sex ratio per age class since 2019 seems to show the opposite with a male to female ratio higher in the older individuals (Table 1), possibly pointing towards a diminution in mature females in the population, which will need to be monitored closely.

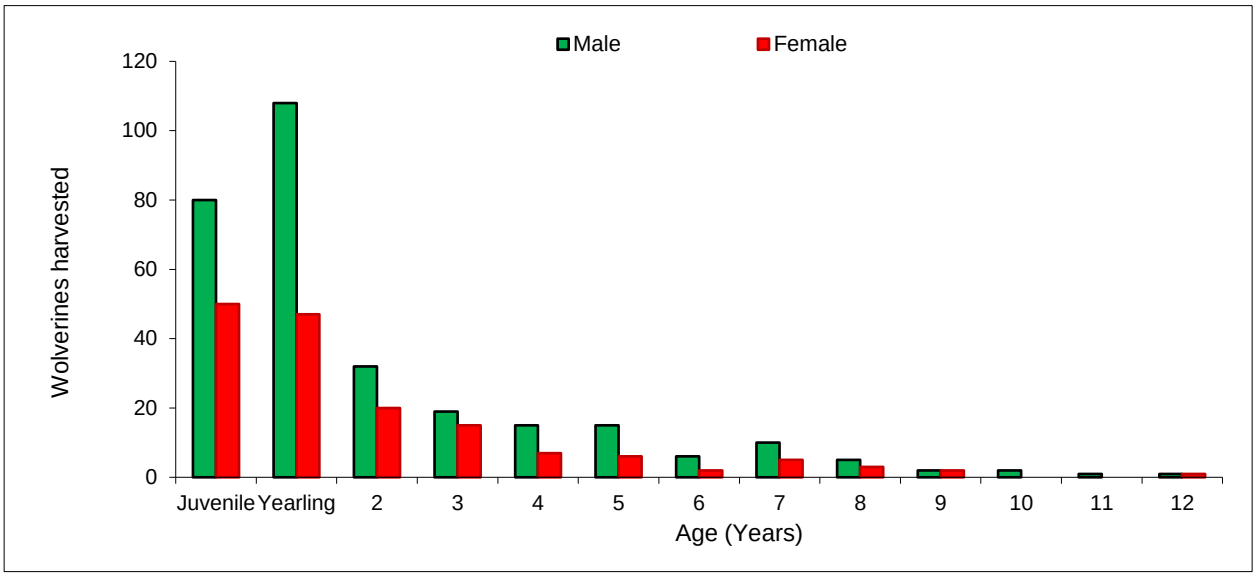


Figure 4. Age and sex structure of the reported wolverine harvest in Nunavut, from 2019 to 2023.

Table 1. Male to female sex ratio by age in the reported wolverine harvest in Nunavut, from 2019 to 2023.

Age	M:F total harvest	M:F sex ratio
Juveniles	80:50	1.6
Yearlings	108:47	2.3
2	32:20	1.6
3	19:15	1.3
4	15:7	2.1
5	15:6	2.5
6	6:2	3
7	10:5	2
8	5:3	1.7
9	2:2	1
10	2:0	-
11	1:0	-
12	1:1	1

The age distribution of the reported wolverine harvest from 2019 to 2023 is weighted more towards juvenile/yearling animals (Fig. 4). The ages of harvested wolverines ranged from <1 year to 12 years. The oldest female and male (12 years) were killed (shot) in Kugluktuk (n = 2) area in 2021 and 2022, respectively. Overall, 29% of the wolverines harvested in Nunavut during that period were juveniles, 34% were yearlings and 37% were adults (Table 2). A greater proportion of young animals in the harvest moderate the conservation risk of harvest (Mowat et al. 2020). However, the proportion of adult females in the harvest increased (13%) for this period (2019-23) compared to 2014-18 (9%, Awan 2020). Wolverine harvest appears to be biased towards younger males (Rausch and Pearson 1972, Lee 1994) probably due to their higher movement rates and vulnerability (Scrafford et al. 2024). Kukka et al. (2017, 2023) describe the high proportions of young males in the harvest to be because vacant areas created by the harvest of resident animals may be filled by dispersing young males (Magoun 1985). Others have reported long dispersal movements in yearlings from their natal area before reaching sexual maturity (Copeland 1996, Mulders 2000, Vangen et al. 2001, Inman et al. 2012) and movement of wolverines from areas with lower mortality to those

with higher mortality (Gervasi et al. 2015, 2016). The high proportion of juveniles and yearlings (63%) and the low proportion of females (36%) among adults in the harvest may signal the importance of dispersal in the population dynamics of this species within the traditional harvesting areas. Yearlings and juveniles represented 64% of the known age male harvest, in contrast to 81% reported by Lee (1994) in the Kitikmeot region in the early 1990's, and by Rausch and Pearson (1972) who also reported wolverine harvest biased toward younger males in Alaska and Yukon.

Table 2. Age and sex distribution of reported wolverine harvest in Nunavut, from 2019 to 2023.

Age Class	Sex		Total (% of total)
	Males (% of males)	Females (% of females)	
Adult (≥ 2 years)	108 (36.5%)	61 (39%)	169 (37%)
Yearling	108 (36.5%)	47 (30%)	155 (34%)
Juvenile (< 1 year)	80 (27%)	50 (31%)	130 (29%)
Total	296	158	454

Figure 5 illustrates the temporal variation in the harvest according to age class. The harvest was dominated by younger age class during the winter while the harvest of adult individuals was comparatively high in March/April, which is similar to other wolverine harvest studies (Lee 2016, Kukka et al. 2017). For males, this corresponds to the time when maximal size of testes and highest levels of testosterone are attained (Pasitschniak-Arts and Larivière 1995), probably marking the start of the breeding season and resulting in increased movements and increased vulnerability of sexually mature males for harvest.

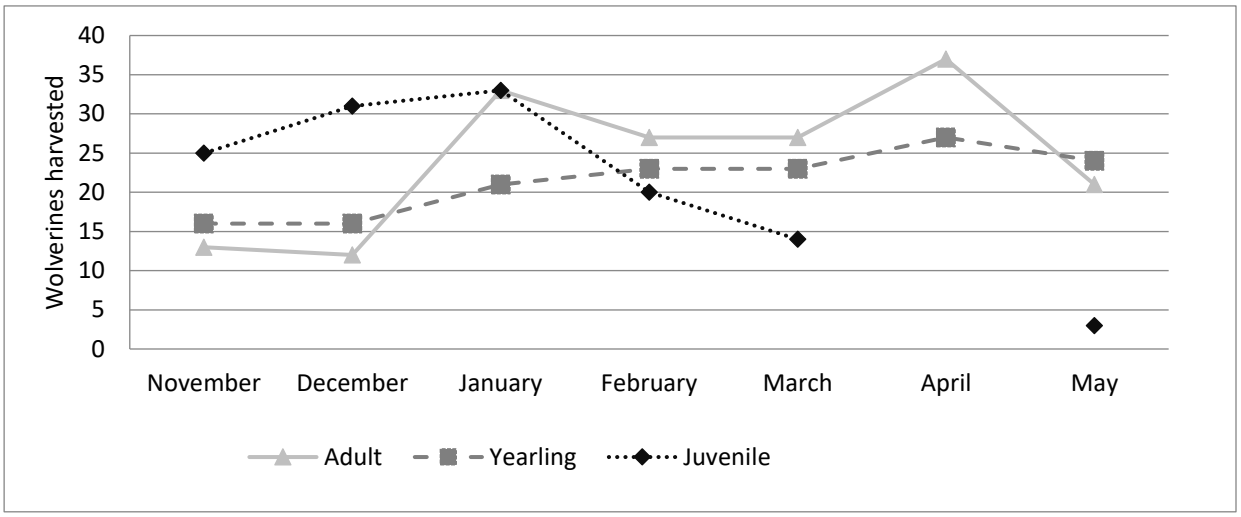


Figure 5. Monthly reported wolverine harvest per age class in Nunavut, from 2019 to 2023.

Wolverine populations in Nunavut are considered healthy. This seems to be supported by the sex and age distribution of the harvest. However, the relatively low number of mature females (13%) in the harvest is important to note and should be monitored closely.

In an effort to obtain information on wolverine population relative abundance, wolverine harvesters (a hunter may provide >1 harvested wolverine to the program within and among years) were asked to answer the following question when bringing back samples to their wildlife office: “Do you think the wolverine population is decreasing, stable or increasing?”. Harvesters’ perception about trends in wolverine populations differed among regions, but majority believed that numbers were stable. In the Kitikmeot (n = 371), most respondents believed the population was stable (67% stable, 32% increasing) while most respondents of the Kivalliq (n = 255) also believed their local population was stable (61% stable, 39% increasing). However, over the long term, the increasing resource development on the tundra and climate change may adversely affect the species, which suggests a need for continuous monitoring of the health of the wolverine population. Monitoring of the wolverine population is also important as part of

predator research and management as it relates to ungulate species management in both the Kivalliq and Kitikmeot regions.

4.0 Acknowledgments

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ᐃᖃᓗᓯᓂᐊᖏᖅᑐᑕᕆᓂᖅᑯᓗᒃ ᐃᒪᒥᐅᑉᓱᓂᓱᓂ ᐱᐅᓯᖏᓐᓄᑦ ᐊᖏᕐᒍ 2-ᖏᓂᒃ ᐱᑕᕆᔨᐊᓕᖏᓂᖅᑯᓗᒃ ᐸᖏᐅᑎᖅᓱᓂ ᐊᑐᓴᖅᑎᖅᓯᒪᓂᖅ, ᐱᓴᓵᖅᓯᔨᓪᓚᒃ ᐱᓕᕆᐊᖓᔪᓂᒃ ᓯᑭᒪᑎᑎᔨᓵᓐᓂ ᐊᑯᓂᐅᔾᔨᖅᑕᐅᓕᕆᓂᐅᓂᒃ ᐱᖓᓱᓕᕆᓂᖅᑐᐃᓐᓇᖅᑐᓂᒃ ᐱᖃᓐᓇᖅᑐᖅᑐᓂᒃ ᐱᖓᓱᐃᓐᓇᖅᑐᖅᑐᓂᒃ: ᐃᖃᓗᓯᓂᐊᖏᓂᖅᑐᖅ, ᓇᕋᑎᕋᓯᖏᓂᖅᑐᖅ ᐊᒻᒪᓗ ᐊᔪᕈᓐᓇᖅᑎᕋᓯᖏᓪᓗᑎᒃ.

ᐊᑯᓂᖅᑕᐅᓯᒪᓕᕆᔨᖅᑕᒃᑭᑦ:

ᐱᓕᕆᖓᒃᑲ ᓴᖅᑭᑐᓂᖅᓴᐅᓕᕆᖅᑎᑎᑕᐅᔨᖅᑐᑦ ᓄᓇᕋᔨᐊᖏᑦ ᑲᑐᔾᔨᖅᑎᑎᕐᑐᓂᖅᑭᑐᓂᖅᑐᒃ ᐃᓅᓂᖅᑎᖅᑎᕋᓯᖏᓂᖅᑯᓗᒃ ᐊᑯᓂᑐᑦ ᑲᒪᒋᔭᖅᑐᓂᒃ ᐊᑯᓂᖅᑐᑦᑐᒃ ᑲᑐᔾᔨᖅᑎᑎᕐᑐᓂᒃ ᐊᑯᓂᖅᑐᑦᑐᒃ ᐃᓅᓄᑦ ᑲᑐᔾᔨᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ. ᑐᓴᕈᑦᓇᖅᑐᖅ ᑲᔪᓯᖏᓂᖅᑎᕋᓯᖏᓕᕆᔨᖅᓱᒪᓂᖅᓱᓂ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓯᓂᐊᖏᖅᑐᑕᕆᓂᖅᑯᓗᒃ ᖃᐅᔨᓴᖅᑎᑎᕐᑐᓂᖅᑭᑐᓂᖅᑐᒃ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ. ᐱᓕᕆᖓᒃᑲ ᐊᖏᕐᒍ 2025-ᒥ, ᐱᐅᓯᔾᔨᓕᕆᓂᖅᑎᑎᕐᑐᓂᖅᑐᖅ ᑕᐃᓴᕆᐅᓂᖅᑐᖅ ᐊᒻᒪᓗ ᐱᓕᕆᖓᒃᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᑦ ᓄᓇᕗᑦᐅᔨᓂᖅᑐᑦ ᓄᓇᖃᓐᓂᒪᒥ. ᐱᓕᕆᖓᒃᑲ ᐊᖓᑎᓕᕆᓂᐊᕆᐊᓕᖅᑎᖅᑐᖅ ᐊᐃᓴᑐᖅᓯᒪᓂᖅᑐᖅ ᑕᐃᓴᕆᐅᓂᖅᑐᖅ ᐅᐸᒍᑎᓕᕆᓂᖅᑐᖅ ᑕᒪᖅᑎᒃ ᐱᖓᓱᓂ ᐊᐃᓴᑐᖅᓯᒪᓂᖅᑐᖅ ᐊᒻᒪᓗ ᐃᖅᑲᖅᑐᖅᑐᓂᒃ ᓴᓇᓕᖅᑐᖅᑐᓂᒃ ᐊᓕᑎᖅᑐᖅ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐱᓕᕆᖓᒃᑲ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᓄᓇᕗᒻᒥ ᐃᖃᓗᓯᓂᐊᖏᖅᑐᑕᕆᓂᖅᑯᓗᒃ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐊᑐᓴᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐱᓕᕆᖓᒃᑲ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐊᑐᓴᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐃᓅᓂᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐊᑐᓴᖅᑎᖅᑎᓂᖅᑭᑐᓂᖅᑐᒃ ᐃᓅᓂ

ᐱᓴᐊᑎᓪᓗᒍ ᐊᓕᖅᑲᓴᐊᕐᓂᖏᑦ ᐃᖅᑲᓗᑎᓴᐊᖅᑐᑕᖅᓂᕐᒧᑦ ᐊᓴᐸᔨᓂᖏᑦ, ᑲᓴᓴᐸᖏᑦ ᑲᑐᔪᐱᑦᑲᖅᓂᕐᓂᖏᑦ ᐊᑯᑭᑐᑐᓂ ᑲᓪᓚᐅᓂᕐᓂᖏᑦ. ᐊᑕᕆᔨᒪᔪᖅ IQaluk (ᐃᖅᑲᓗᓂ) ᐱᑕᕆᐊᖅᓴᑯᓂ, ᐊᒻᒪᓗ ᓄᓇᑕᐅᓴᐸᑎᓂᐅ ᐱᖅᑯᑎᐅᐊᓂ ᐱᓕᕐᓂᐊᑎᕐᑎᓂᖏᑦ. ᐱᑕᕆᔨᒪᔪᖅ ᑲᓴᓂᓇᖅᑐᖅ ᐅᒥᐊᖅᓄᑦ ᑐᒡᓕᑦᑐᓂ ᓂᐱᖅᑎᑎᕐᓂᖏᑦ ᐊᒻᒪᓗ ᑲᓴᓂᓇᖅᑐᒥᑦ ᐃᒪᖅᑯᑎᐅᐊᓂ ᐃᖅᑲᓗᑎᓴᐊᖅᑐᑕᖅᓂᕐᒧᑦ ᓄᓇᑐᓪᓚᒍ.

ᐊᖅᑯᒍ 3 (2025-2026) ᐊᑐᖅᑎᕐᑎᓂᕐᒧᑦ <ᖅᓇᐅᑦᑎᓪ:

ᐊᖅᑯᒍ 3 ᑐᕆᓚᖅᖄᑐᖅ ᑲᓴᓂᓇᖅᑐᒥᑦ ᐊᖅᑯᒍ 2 ᑲᓴᓂᑎᑕᐅᕐᑎᐊᖅᑐᓂ ᓂᐅᓕᑎᕐᑎᓂᓇᓂᓪᓕᑦ ᓴᖓᔪᑎᑕᐅᓴᐸᓂ ᓂᐅᑦᑐᓴᐅᓴᕆᓚᐅᓐᓂᓂ ᐊᒻᒪᓗ ᓄᒋᑦ ᖃᓄᖅᑐᓂᕐᓂᖏᑦ ᐱᑕᕆᐊᖅᓂᑦ ᐅᐅᓂᒡᓕᑦ.

ᐱᓕᕆᐊᑕᐅᓴᑦ ᐃᖅᑲᓗᑎᓴᐊᖅᑐᑕᖅᓂᕐᒧᑦ ᐱᑕᕆᓴᐅᓴᑦ:

ᐃᖅᑲᓗᑎᓴᐊᖅᑎᑦ ᖃᐅᐅᓚᓂᖓᓂᐅᓂᒃ ᑕᐅᖅᓯᖅᑎᕐᑎᓂᕐᒧᑦ ᐊᑯᑭᑐᑐᓂ <ᖅᓇᐅᑕᓚᓂᓕᕆᖅ 2025-2026-ᒧᑦ ᐊᑐᐃᓇᐅᓴᒋᑦᑐᖅ 2026-2027-Γ, ᐃᓕᓴᐅᓚᓂᕆᓕᕆᔨᓂ ᐊᖏᕐᖃᑎᑎᖅᑎᓂᐅᓂᐅᕐᓴᖅ ᑐᐅᓯᐅᒪᖃᑎᑎᖅᑎᓂᐅᓂᕐᒧᑦ ᑎᑎᕋᖅᑕᐅᖅᓯᒪᓕᐅᖅᑐᖅ ᐊᖅᑯᒍ 2-Γ. ᐱᑕᕆᔨᒪᔪᖅ ᑲᓴᓯᓕᒋᑐᖅ ᓂᐅᖅᑕᑎᖃᖅᓂᕐᒧᑦ ᑕᑯᑦᓯᐊᖅᓂᕐᑎᓂᖏᑦ ᖃᓇᑕᒥ ᐃᖅᑲᓗᑎᓴᐊᖅᑐᑕᖅᓂᕐᒧᑦ ᓯᓇᖃᑦᑕᒥ ᐊᒻᒪᓗ ᐃᖅᑲᓗᑎᓴᐊᖅᑐᑕᕆᔨᓐᓂ ᑲᑎᒪᓂᕐᑎᓂᖏᑦ ᐅᑭᐊᖅᓱᖃᓐᓄᓂ 2025-Γ ᑐᑭᓯᐅᒪᓂᖅᓴᐅᓂᕐᒧᑦ ᐊᖅᓯᔨᑕᐅᔪᓂᒥ ᐊᒻᒪᓗ ᐊᖏᕆᐊᑭᑕᐅᔪᓂᒥ ᐱᑕᕆᔨᕆᐊᓐᓂᐅᒃ ᐊᐅᑕᓯᑎᓂᕐᒧᑦ.

ᓇᓂᕐᓯᐊᓂᕐᒧᑦ ᐱᖅᑯᑎᐅᐊᓂᖅ:

ᑐᕆᓚᖅᑕᓚᓇᖅᑐᑦ <ᖅᓇᐃᓂᕐᑎᑦ ᐊᒻᒪᓗ ᑲᓴᓂᓇᖅᑐᑎᓂᕐᑎᑦ ᓇᓂᑎᑕᐅᓂᕐᒧᑦ-ᑐᕋᖏᓚᐸᑎᑦ ᑲᑎᒪᓂᕐᑎᑦ 2026-2027-Γ ᑲᑐᔪᖅᑎᑎᖅᓯᒪᓂᖅᓴᐅᓂᕐᒧᑦ ᐊᒻᒪᓗ ᓄᓇᓕᐅᓂᕐᒧᑦ. ᐱᑕᕆᔨᒪᔪᖅ ᑐᕋᓯᖅ ᐋᖅᑭᑲᓯᐊᕐᓂᖏᑦ ᓄᓇᓕᒫᓂ ᐊᒻᒪᓗ ᐱᓕᕐᓂᐊᑎᕐᑎᓂᖏᑦ ᐃᓕᓴᐅᓂᕐᒧᑦ ᑲᑐᔪᖅᑎᑎᖅᑎᓂᐅᓂᕐᒧᑦ ᓄᓇᓕᒫᓂ ᐊᒻᒪᓗ ᐊᓯᖏᑦ ᐱᑕᕆᔨᐅᑭᑲᓕᐅᔪᓂᒥ. <ᖅᓇᐅᑦᑎᓪ ᑲᓴᓯᕐᓂᐊᓂᕐᒧᑦ ᐊᕐᑎᑎᕆᐅᓂ ᖁᑎᖅᑭᓂᕐᒧᑦ ᖁᑕᐃᓚᑲᖏᓐᓂᓂᕐᒧᑦ ᓄᓇᓕᒫᓂ ᑐᖓᔪᓚᐅᖅᑕᐅᓚᖅᑐᓂᕐᒧᑦ ᐱᑕᕆᔨᕐᓯᒪᓕᖅᑐᖅ.

ᐱᑕᕆᐊᖅᑭᐅᓇᐅᑎᓂᒃ ᐱᓕᑭᐅᓂᖅ ᐊᒻᒪᓗ ᓄᓇᑕᐅᓴᐸᑦᑎᓪ:

ᐱᑕᕆᔨᒪᔪᖅ ᑲᓴᓂᓇᖅᑐᖅ ᐊᖅᑯᑭᓂᖏᑦ ᐃᓐᓄᑦᑐᐊᓂᕐᒧᑦ ᐊᐃᔪᑐᖅᑭᒪᔪᓂᒥ ᐃᓐᓄᑦᑐᐊᓂᕐᒧᑦ, ᐃᓕᑎᑐᓂᕐᒧᑦ ᐱᑕᕆᐊᖅᑎᑎᑎᓂᕐᒧᑦ ᓇᓂᕐᑎᓂᕐᒧᑦ ᐃᓐᓄᑦᑐᐊᓂᕐᒧᑦ. ᐃᓗᐊᓂᓪᓕᐊᑐᖅᑐᓂ ᐱᑕᕆᔨᒪᔪᖅ ᐋᖅᑭᑲᓯᐊᕐᓂᖏᑦ ᐊᑯᑭᑐᑐᓂ ᐊᑐᐃᓇᐅᓴᒋᑦᑐᖅ ᓄᓇᓕᒫᓂ ᐊᒻᒪᓗ ᑲᓇᑕᒥ ᐱᑕᕆᔨᒪᔪᖅ ᐊᑐᐃᓇᐅᓴᒋᑦᑐᖅ <ᖅᓇᑕᐅᓂᒪᓕᕆᔨᖅ 2025-Γ, ᑲᑐᔪᖅᑎᑎᖅᑎᓂᐅᓂᒃ ᐱᑕᕆᔨᒪᔪᖅ ᑲᑐᔪᖅᑎᑎᖅᑎᓂᐅᓂᕐᒧᑦ ᐃᓐᓄᑦᑐᐊᓂᕐᒧᑦ ᐱᑕᕆᔨᒪᔪᖅ ᐃᓐᓄᑦᑐᐊᓂᕐᒧᑦ ᐊᑐᐃᓇᐅᓴᒋᑦᑐᖅ <ᖅᓇᐅ

ጋσʔቦᖏϟΔᑦ

ᓄᓇᕗᒻᒥ ᐅᒪᔨᓕᕆᔨᖅᑕᐃᒪᓕᒃᑯᑦ ᑲᑎᒪᔨᖏᑐᓄᑦ

ᑲᑎᒪσᓱᐅᓇᓴᖏσᒃ ᑲᑎᒧᑐᓂᐅᒃ ᐃϟᐅᑎᓂᒃ 003-2025

ᒪጋᒥᖏᒐ ᐱᔨᔪᑎᓕᒃ

ʔᑲᐅᔪᑎʔᑎᑭᕗᑦ: ☐ Δᓗᒪᓕᐅᕗᑎϟʔᖅ: ☒

Δᓗᒪᓂᑕᐅᔨᖅ: *ᐱᑎᕐᖏᑎᑕᐅᔪᒪᔨᓂ 2025-2026-ᒥ ᐊʔᖏᑎᕐᖅᑕᐅσᓂᖏᓂ ᑕᒪΔᵒσᒃ ᑲᑎᒧᑦᓂᑎᓂ ᐊᖓᐃᓇᓂᑕᐅᔨᵒᓇᖅᑐᓂ TAH-ᖏᓂ ᐊᓪᒪᓗ ᑯᑕᑯᖓᒐᖏᑐᒥᑲᒃ Δᓇᖅᑲᖏᑎᑎᐱᔨᑎᖏᓂ ᖃᓄᖅᑕᓂᔭᓪᓄ ᑕᑐᖅᑕᐅᓇᖅᑐᓂ ᑕᑐᖅᑕᐅᓇᖅᑐᓄᑦ ᐱᔨᔪᑎᖅᑐᓂᓗ (ᐅᑎᓴᵒ Δᓕᑎᓕᐅᖅᑕᖏᑦᓗ 1998, ΔᓄΔᑦᓗ ΔᵒᓇᑐᖅᑲΔᑦ). ᑭᒃᑯᐊ ᑲᑎᒪᔨᖏᑦ ᖃᓄᖅᑕᓂᔭᑦ ᐊᖓᐃᓇᓴᑎᓕᒫᑦ ᑲᑎᒪᔨᖏᑦ QWB-ᑯᑦ ᐱᑎᐊᖅᑎᑎᑯᔨᓕᐅᖅᑐᑦ ᑐᖅᑐΔᑦ ᐊᐅᓕᑕᐅᖅᑲᑕᑎᓂᖏᑦ ᑲᓱᔨᑯᓕᑐᓂᑦ ᖃᔨᓄΔᑦ 10-ᓄᑦ ᐊᖏᖅᔪᓄᑦ, ᑎᑭᓕᑐᒧ ᑕΔᑯᖓᒪ 2032-ᒧᑦ, ᑐᖏᓚᖅᖏᑐᒃ ᐊᑐᕈᑯᖅᑎᑕᐅᔨᵒᓇᖅᓂᖏᵒσᒃ ᐊʔᖏᑎᐊᖅᑕᐅᖅᑲᑕᕈᵒᓇᖅᓂᖏᓂᑐᓄᒃ ᓇᓕᐊᑐΔᵒᓇΔᑦ ᑕᒪΔᵒσᒃ ᑲᑎᒪᔨᑦ ᑐᑐᖅᑕᐅᓇᖅᓂᖏᵒσᒃ ᖃᑎᖅᑕᐅᔨᖅᑕᐅᔨᓕᐅᖅᑐᑦ Δᓕᑦᐸᑦᑕᓗᓄᑦ.

ᑕᒧᑎσ 2022-ᒥ, ᐅᒪᔨᓕᕆᔨᖅᑕᐃᒪᓕᒃ ᐊᒥᔨᕈᑎᐊᖅᑕᐅᖅᑐᑦ ᐊᒥᔨᕈᓚᐊᖅᑕᐅσᓂᖏᵒσᒃ 100-σᒃ ᖃᓄᖅᑕᓗᓂᒥ ᑐᖅᑐσᒃ ᑕΔᑲᖏᑦ ᑕᒪΔᵒσᒃ ᑲᑎᒧᑦᓂᑎᓂ ᐊᖓᐃᓇᓕᕆᔨᓇᵒᓇᓴᓂᖏᵒσᒃ, ᑭᖓσᐊᔪᑦᓗ ᐊᖏᖅᔪᒧᑕᒻᒪᖅ ᐊᒥᔨᕈᓚᐊᖅᑕᐅᖅᑲᑕᐃᓇᖅᑐᒃ ᑕᒪσᒃ 50-σᒃ ᑐᖅᑐᑕᐅᔨᵒᓇᖅᑐᑦ ᐊᖏᖅᔪᒧᑕᒻᒪᖅ ᐱᑎᐊᖅᑎᑑᑎᑯᔨᓕᐅᖅᑐᑦ 2023-ᒥᒃ ᑎᑭᓕᑐᒧ 2032-ᒧᑦ. 2022-ᒥ, QWB-ᑯᑦᑕᐅᖅ ᐱᑎᐊᖅᑎᑎᑯᔨᓕᐅᖅᑐᑦ ᐱᓕᑦᓕᐊᓇᓴᖅᑐᒥᒃ ᐊᒥᔨᕈᓚᐊᖅᑕᐅᔨᵒᓇᓴᖏᵒσᒃ ᐊᖏᓇᓕᓂᖅᑕᐅᔨᵒᓇᓴᖏᓂᑦ ᑐᖅᑐΔᑦ ᑎᑭᐅᑎσᐊᖅᑐᑦ 50% ᐳᓴᑎᖏᑦᑯᓇᓴᖅᑐσᒃ ᑕᒪΔᵒσᒃ ᑲᑎᒧᑦᓂᑎᓂ ᐊᖏᖅᔪᑕᒻᒪᖅ ᑎᑭᐃᒧᑕᒻᒪᖅ ᐊᖓᐃᓇᓕᕆᔨᐅᑦᓴᖏᵒσᒃ ᑎᑭᑦ 2032-ᒧᑦ. ᑲᑎᒪᔨᓇᓇᖏᑦ ᐅᒪᔨᓕᕆᔨᖅᑕᐃᒪᓕᒃᑯᑦ ᑲᑎᒪσᖏᵒσ, Δᓗᒪᓕᐅᓕᐅᖅᑐᑦ ᑭᒃᑯᐊ ᐊᖏᕐᕕᒪᓂᖅ ᐊᖏᓇᓕᓂᖅᑕᐅᔨᵒᓇᓴᖏᵒσᒃ ᑕᒪσᓕᒫᑎᒃ 20% ᐳᓴᑎᖏᑎσ ᑕᒪΔᵒσᒃ ᑲᑎᒧᑦᓂᑎᓂ ᑐᖅᑐᓗᖏᓚᐊᖅᑕᐅᔨᵒᓇᓴᖏᵒσᒃ ᐊᖏᖅᔪᒧᑕᒻᒪᖅ ᐱᑎᐊᖅᑎᒃ 2022-ᒥᒃ ᑎᑭᐃᒧᑕᒻᒪᖅ ᑕᒪᐅᖅᓗ ᐊᖏᖅᔪᒪᑦ 2032. ᑕᒪᒃᑯᐊ Δᓗᒪᓕᐅᕈᑕᐅσᖏᓂ ᐱᓴᓰᖅᑕᐅᓕᐅᖅᑐᑦ ᐊᖓᓂᖅᑕᐅᓕᐅᖅᑐᑦ ᑲᒪᓕᓂᑦ ᑕᒪᒃᑯᐊ ᑕᒪᑐᑦ ᑎᑭᐃᒧᑕᒻᒪᖅ ᐊᖏᖅᔪᒪᑦ ᑕᒪᒃᑯᐊ ᑕᒪᑐᑦ ᑲᑎᒪᔨᖏᑦ ᑐᖅᑐᑕᐅᔨᵒᓇᓴᖏᓂᑦ ᐊᖓᐃᓇᓕᕆᔨᖅᑕᐅᖅᑐᑦ ᐊᖏᖅᔪᒪᑦ ᑕᒪᒃᑯᐊ ᑕᒪᑐᑦ ᑲᑎᒪᔨᖏᑦ ᑐᖅᑐᑕᐅᔨᵒᓇᓴᖏᵒσᒃ ᐊᖏᖅᔪᒪᑦ ᑕᒪᒃᑯᐊ ᑕᒪᑐᑦ ᑲᑎᒪᔨᖏᑦ ᑐᖅᑐᑕᐅᔨᵒᓇᓴᖏᵒσᒃ, ᑕᒪᒪᑕᒻᒪᖅᑐᑦ ᐱᓕᑦᓕᐊᓇᓴᖅᑐᑦ ᐊᖏᖅᔪᒪᑦ ᑕᒪᒃᑯᐊ ᑕᒪᑐᑦ ᑯᑕᑯᖓᒐᖏᑐᒥᑲᒃ ᐊᖓᐃᓇᓕᕆᔨᐅᑦᓴᖏᵒσᒃ ᑲᑎᒪᔨᖏᑦ ᑐᖅᑐᑕᐅᔨᵒᓇᓴᖏᓂᑦ ᐊᖓᐃᓇᓕᕆᔨᖅᑕᐅᖅᑐᑦ ᑲᑎᒪᔨᖏᑦ ᑐᖅᑐᑕᐅᔨᵒᓇᓴᖏᵒσᒃ ᐊᖏᖅᔪᒪᑦ ᑕᒪᒃᑯᐊ ᑕᒪ

QWB-ժ՞ ᐃᓕᓴᕐᓂᖅᑭᓪᓗᒍ ᑕᒪժᓂ ᐃᓄᖅᓄ՞ ᑕժᔪᐅᖅ ᖃᓄᖅᑎᓂᖅ ᐊᖏᓪ ᑐᑭᓯᐅᔪᐊᑐᐃᓐᓇᖅᑕ ᐃᓕᓴᕐᐅᑭᓄᖅ ᑕᐃᖁժᐊ ᐊᖅᓇᐅᐅᔪᖅ ᐃᓕᓴᖅᑕᐅᐅᔪ՞ ᑐᖏᑐᔪᐅᓇᓄᖅᑎᓂᖅᓂ ᐱᖏᓪᓗᐅᔪᓂᖅ ᑐᖅᑎᓂᖅᓄᑦᑎᓪᓗᒍ ᓴᖅᑭᓪᓗᐅᔪᓂᖅᑎᓂᖅᓄᑦ ᑲᑎᒪᔨᖅᑕᐅᔪᓂᖅᑎᓂᖅᓄᑦ ᓴᖅᑭᓪᓗᐅᔪᓂᖅᑎᓂᖅᓄᑦ ᐅᖃᓕᓴᐅᔪᓂᖅᑎᓂᖅᓄᑦ, ᑐᖅᑲᐃᖅᑕᐅᔪᓂᖅᑎᓂᖅᓄᑦ ᐅᖅᑕᐅᔪᓂᖅᑎᓂᖅᓄᑦ ᐊᒥᐊժᓂᖅᑐᐊᓕᒍ ᐅᖅᑲᐅᔪᐃᓕᓴ

ᓱᕐᓯᑦ ᓱᓕ ᒥᑭᔨᓂᖅᓴᓂ, ᐊᒥᓱᓴᓂᕈᐊᒪᓕᒃᑯᑦ ᑐᖅᑐᓄᑦ ᑕᐅᕐᓂ ᖃᑐᖓᓴᖅᓭᕆᔨᑎᑎᖅᑕᐅᖅ ᑲᓐᐊᑦᓴᐃᓐᓇᐃᑦ ᓱᕐᓯᑦ, ᓇᖅᒥᓂᖅ ᓂᓚᕆᕋᖅᓴᖅᑎᓇᐊᖅᑕᐅᒪᓕᒃᑯᑦ ᐃᓇᐊᖅᑐᓇᕐᒥᐅᓇᐃᓐᓇᑦ ᐅᑎᖅᑕᐅᑦ, ᐊᒥᓱᓕᕆᔨᖏᓪᓕᐊᓂᐃᑦ ᑐᖅᑐᑦ. ᐃᓇᐃᑦ ᖃᐅᔨᓴᓱᑦ ᐊᒻᒪᓗ ᐃᓕᓴᕐᓱᓕᖅᑐᑦ ᐊᒥᓱᓂ ᓇᓗᓇᐃᖅᑲᒪᓚᕆᐅᕙᖅᑐᓂᖅ ᖃᓄᖅᑭᑐᓂᖃᖅᑐᑦ ᖃᐅᔨᓴᓂᕐᓂᖅᓴᓂᖅ ᐊᓱᔪᓇᖅᓱᒪᙵᖅᑐᓂᖅ ᖃᐅᔨᓴᓂᕐᓂᖅᓴᓂᖅ

ᓴᖅᑭᑦᓯᔨᕐᓂᖅ ᑕᐃᒧᖏᓐᓄᑕᒪᖅ ᐊᐅᓚᑦᑐᓂᒃ ᒪᓕᒃᑐᓂᒃ ᐃᓄᐃᑦ–ᖃᐅᔨᓴᒪᔭᑐᑦᓂᒃ <ᖃᓄᐅᓯᐅᖃᔨᓂᖅ ᖁᑭᖅᑕᒋᔪᑦ ᐅᐊᖓᓄᓚᓄᑦ, ᑭᓯᐊᓂ ᓯᓕᒃᓂ ᐃᓯᒪᑎᔭᐅᓕᖅᑐᑦᓂᕐᒪᓂᕐᓂᓂᕐᑦ.

ᒪᓕᒃᑕᐅᓪᓗᑎᒃ ᐃᓄᐃᑦ ᖃᐅᔨᓴᒪᔭᑐᖃᖓᑦᓯᒃᑐᑦ ᐊᒻᒪᓗ ᐃᓄᐃᑦ ᐅᓴᓴᖃᑦᑕᖅᑕᖓᑦ ᑐᑐᑐᓕᔪᓂᕆᒃ ᐊᒥᓯᐅᓴᖅ, ᐊᐞᓂᐅᔨᔪᖅ ᖃᓄᐃᓕᒥᐅᔨᓴᖅᑐᑦᓂᖅ ᐱᑎᒪᓕᐅᓂᕆᒃ ᐊᑐᓕᖅᑐᑦᓂᖅ>ᑦ ᐱᒻᒪᓕᓕᐅᓂᕆᒃ ᐊᑐᓕᐊᖅᑕᐅᔨᓴᖅᑐᑦᓂᖅᑐᑦ ᐃᓄᐃᑦ ᖃᐅᔨᓴᒪᔭᑐᖃᖓᑦᓯᒃᑐᑦ ᐃᓄᐃᑦ ᐅᓴᓴᖃᑦᑕᖅᑕᖓᑦ ᐊᐅᓚᑦᑐᓂᒃ ᐊᐞᓂᐅᔨᔪᖅᑐᑦᓂᖅ

ᑐᑭᓯᓂᐊᖅᐸᑦ:

ᑐᑭᓯᓂᐊᖅᐸᒥᓱᓕᕆᓂᖅ ᐊᖏᔪᓚᑦ ᑕᒪᓂ 2018, 2019 ᐊᓪᒪᓗ 2020 QWB–ᓄᑦ ᐱᑎᐊᖅᓄᓯᒥᓂᖅᑐᑦ ᐊᒥᓱᓗᐊᖏᑦᑐᓂᑦ ᐊᒥᓱᑕᐊᑕᐅᑎᖅᓴᓂᑦ 2015–ᒥ ᑕᒪᐃᓐᓂᑦ ᑲᑎᒪᓂᖅᑐᑦ ᑕᐃᒪᐃᓕᐊᓂᖅᓯᒪᓐᓂᑦ ᑭᑭᖅᑕᓗᓐᓂᒥ. ᐅᒪᔨᓕᕆᓚᕐᔨᐊᕐᓄᑦ ᐊᖏᑎᑭᑎᖅᑕᓕᖅᖏᓂᑐᑦ ᑐᓴᕐᔪᐅᑎᖅᓂᖅᑐᑦ ᑕᐃᒪᐃᓕᓕᖏᓴᕐᓂᑦ, ᒥᑭᔪᒥᖅ ᐃᓕᑭᑎᐊᖅᑕᕆᓕᕆᓂᕐᓗᐊᖅᓄᑦ ᐊᖏᓐᓂᒃᓂᖅᑐᑦ ᑕᐃᒪᓐᓂ 2020–ᒥ.

2021–ᒥ ᐊᖏᕿᙵᒪᓴᖅ ᑲᑎᒪᖏᐊᖏᓂᖏᓐᓂᑦ QWB–ᓄᑦ ᐅᖅᑲᐅᔪᖅᑕᕆᓕᖅᑐᑦ ᐃᓄᐃᑦ ᖁᑲᐅᓚᒪᕐᖅᑐᖅᑐᑦ ᐃᓄᐃᑦᓗ ᑭᑭᖅᑕᓗᓐᓂᒥ ᑐᑭᑐᓂ, ᐅᖅᑲᐅᔾᔨᑎᐊᖅᑕᕆᓕᔪᓗᑎᑦ ᐃᖅᑲᓇᐃᓕᖅᑎᑐᓂᓄᑦ ᐃᓄᐃᑦᓗ ᑐᓴᕐᔪᐅᑎᖅᓴᓂᑦ ᑐᓂᔭᖅᑎᓂᒃᑐᑦ ᐊᖏᓐᓂᒃᓂᖅᑐᑦ ᐊᖏᔪᓚᑦ ᐊᒥᓱᑕᐊᑕᐅᑎᖅᓴᓂᑦ ᐊᖏᓐᓂᒃᓂᖅᑐᑦ ᐊᖏᔪᓚᑦ ᐊᒥᓱᓕᕆᓂᖅᑐᑦ ᐊᖏᓐᓂᒃᓂᖅᑐᑦ ᐊᖏᔪᓚᑦ ᐊᒥᓱᑕᐊᑕᐅᑎᖅᓴᓂ

5. QWB–ᓄᑦ ᐊᐅᓪᓚᕐᒪᕆᐊᑦ ᑲᑎᒪᔨᕆᓪᓕᖅᑐᑦ ᑲᒪᔨᐅᓂᐊᖅᑐᑦ ᐊᖏᕐᓇᓱᖅᑎᑦ ᑲᑐᔨᖅᑎᑎᒋᐊᕐᓂᓄᑦ ᐊᐃᑦᑐᓱᐊᖅᑕᓪᓚᕆᐅᓂᐊᖅᑐᓄᖅ ᒪᓕᖅᑕᐅᓗᑎᖅ ᐊᑐᖅᑕᐅᑲᐃᐅᓇᖅᓂᑦ ᖃᐅᖅᑕᓴᖅᑐ ᑐᑐᓱᐅᑭᓇᖅᓯᒪᑦ ᒪᓕᑦᑎᕆᓇᓱᐅᓂᖅᑕᑦ.

6. ᐊᑐᕋᓂᖅᓯᓂ 2031-2032–ᒥ ᐊᖏᕐᕕᒍᖅ, QWB ᖃᐅᔨᒋᐊᕈᒪᖅᑐᑦ ᐊᑐᖅᓂᐊᖏᓚᖅᓴᓕᓂᑦ ᐊᑐᖅᑕᐅᓴᕐᓂᓪᓚᖅᓴᓕᓂᑦ ᑕᒪᐃᐅᓂᓄ ᑲᑎᑦᓂ ᑐᑐᑕᐅᔨᐅᓇᖅᓂᑦ ᐊᓚᒪᓗ ᑯᑕᑯᖅᓯᖅᓂᐅᓂᑦ ᐃᓱᖅᑲᕐᓂᖅ ᑕᒪᒃᑯᐊ 1, 2032–ᒥ, ᐅᕐᖓᓗᓂᑦ ᑕᐃᒃᑯᐊ ᑕᒪᐃᐅᓂᐊᑦ ᑲᑎᑦᓂ ᐊᓚᒪᓗ ᑯᑕᑯᖅᓯᖅᓂᐅᓂᑦ ᐃᓱᖅᑲᕐᓂᖅᑐᑎᑦ ᐃᓱᖅᑲᕐᓂᖅᑐᑦ ᑲᔪᓱᑎᐅᕐᓂᐊᖏᓚᖅᓴᓕᓂᑦ ᑲᖅᑭᑦᐊᖅᑕᐅᓂᐊᖏᓚᖅᓴᓕᓂᑦ ᓱᕐᓂᓴᒃᑲᓂ, ᑐᖅᑲᕕᖅᓂᐊᖅᓴᑦ ᑕᒪᐃᐅᓂᐊᑦ ᐊᑐᐃᐅᓇᐅᔨᓂᐊᖅᑐᑦ ᖃᐅᔨᒪᓕᕆᓂᐊᖅᑐᑦ ᐃᓱᖅᑲᕐᓂᖅᑐᑦ ᐃᓱᖅᑲᕐᓂᖅᑐᑦ ᑲᑎᒪᔨᖅᑕᐅᓂᐊᖏᓚᖅᓴᓕᓂᑦ ᓱᕐᓂᓴᒃᑲᓂ, ᑐᖅᑲᕕᖅᓂᐊᖅᓴᑦ ᐊᑐᐃᐅᓇᐅᔨᓂᐊᖅᑐᓄᖅ ᖃᐅᔨᒪᓕᕆᓂᐊᖅᑐᑦ ᑐᑐᑕᐅᔨᐅᓇᖅᓯᒪᑦ ᐊᓱᔨᑕᐅᓂᖅᓯᒪᓕᖅᑐᑦ. QWB–ᓄᑦ ᑐᓱᔨᕈᑎᖅᓴᒥᓂᖅᑐᓄᖅ ᑐᓂᓱᐊᒻᒪᓕᕐᓂᑦ ᐅᒪᓕᔨᓕᕆᓇᖅᓴᕐᐳᐊᓂᑦ ᖃᓄᐃᓇᐅᔪᖅ ᐊᑐᕐᕋᔨᓇᖅᑐᓂᕈᑎᓄᖅ ᐊᓱᔨᑕᐅᓂᖅᓯᒪᔪᖅᑐᓄᖅ.

ᐅᖃᓕᒫᓚᐅᑦ ᐊᑐᖅᑕᐅᓱᒪᓂᖅᓯᑦ ᑕᑯᓂᓂ ᐅᓂᒃᑲᓂ:

Ferguson ᐃᓕᖅᑎᓗ 1998. ᐃᓄᐃᑦ ᖃᐅᔨᒪᔨᑐᑐᖅᑐᓂᒃ ᐊᑯᓂᐅᔨᐳᑯᑦ ᐊᖃᓱᔨᖅᕿᑦᓕᒥᐊᖃᕈᑦᓂᖅᓯᑦ ᐊᒥᓱᓂᖅᓯᑦ ᐅᑭᐅᖅᑕᖅᑐᒥ ᓇᕿᖅᑐᖃᖅᑎᖅᓯᑐᒥ ᑐᓴᖅᑐᖅᓂᓂ. ᐅᑭᐅᖅᑕᖅᑐᒥ 51: 201–219.

ᑎᑎᕋᖅᑕᐅᓱᒪᔪᑦ ᐅᑯᓄᖅ: ᒪᐃᑯᑦ ᐅᓴᓴᓂ ᐊᓚᒪᓗ ᑯᓗᓪᓕ ᐱᕋᓱᐅᓪᓕᖅ, ᖃᑭᖅᑕᓪᓗᒥ ᐊᖏᓚᖅᑎᓪᓗᒪᓕᒃ ᑲᑎᒪᔨᖅᑐᓂᓄ

ᐅᓪᓗᒥ: ᓱᑎᐱᓇ 5, 2025

SUBMISSION TO THE

NUNAVUT WILDLIFE MANAGEMENT BOARD (NWMB)

Regular Meeting No. RM 003-2025

FOR

Information: ☐ Decision: ☒

Issue: *Proposed 2025-2026 Revision of the Total Allowable Harvest (TAH) and Non-Quota Limitation for Harvesting of Baffin Island Caribou (BIC) from 2025 to 2032*

Background:

In 2022, the Qikiqtaaluk Wildlife Board (QWB) recommended an Inuit system of caribou harvest management on Baffin Island to start on July 1, 2022, in accordance with sub-section 5.1.2(e) of the Nunavut Agreement. This harvest management system was based on past practices of Inuit harvest management and Inuit Qaujimajatuqangit (IQ) regarding natural long-term changes of caribou populations on Baffin Island (Ferguson et al 1998, and Inuit Elders). The QWB proposed that its caribou management system should continue for 10 years until 2032, with the aim to remove or modify any Total Allowable Harvest {TAH) and any Non-Quota Limitation (NQL) after a review with co-management partners.

In 2022, the NWMB approved an increase of 100 Baffin Island caribou in the Total Allowable Harvest (TAH), to be followed by an annual increase of 50 caribou from 2023 to 2032. In 2022, the QWB had also proposed a gradual increase in the maximum harvest of females to reach about 50% of the total annual harvest by 2032. In a closed NWMB session, the NWMB decided to set the maximum harvest of females at 20% of the total harvest throughout the period from 2022 to 2032. These decisions were finalized by their acceptance by the responsible Minister.

In 2022, the QWB proposed to reassess the schedule of increases in the TAH after five years; however, many HTOs and Inuit in the communities are expressing concerns about the speed at which the caribou population is increasing in many parts of Baffin Island. The consensus among Inuit appears to be that the TAH is too low and the NQL is not needed and unnatural, leading some Inuit to advocate for elimination of both the TAH and NQL already, some seven years before 2032.

The QWB is very concerned that these observations by Inuit make it reasonable to conclude that the current limits on caribou harvesting may jeopardize the key objectives of the QWB's 2022 proposal, which were based on the principles of conservation in section 5.1.5 of the Nunavut Agreement. The QWB reasserts its objectives for the conservation of Baffin Island caribou through Inuit systems of caribou management and Inuit Qaujimajatuqangit, as stated below:

- Maintain the natural balance of the lichen-caribou-Inuit ecological system that has existed since time immemorial

- Protect caribou winter habitats, and enable the regrowth of ample lichen needed for future caribou populations,
- Maintain a vital, healthy caribou population across Baffin Island capable of sustaining Inuit harvesting into future decades
- Restore and revitalize a depleted population of caribou and their habitat over the long term.

Assertion of Primacy of Inuit Systems of Wildlife Management in Decisions in Nunavut

In December 2020, the Qikiqtaaluk Wildlife Board (QWB) adopted the position to assert that the Nunavut Agreement, a constitutionally protected treaty between the Inuit of Nunavut and the Crown of Canada, provides primacy to Inuit Systems of Wildlife Management with respect to decision-making processes and outcomes regarding wildlife and wildlife harvesting by Inuit. This primacy extends to Inuit Qaujimajatuqangit (IQ) because IQ is the basis for Inuit Systems of Wildlife Management. In this context, "primacy" refers to what comes first and remains most important. Inuit Systems of Wildlife Management are specifically recognized in sub-section 5.1.2(e) of the Nunavut Agreement. Other sections of Article 5 of the Nunavut Agreement point out the special rights and effective roles that Inuit have in wildlife management and harvesting in Nunavut. Science and scientific wildlife management are not mentioned in Article 5 of the Nunavut Agreement. The QWB asserts that any wildlife management plan, recommendation, decision or other measure established or implemented regarding caribou or other wildlife in Qikiqtaaluk Region needs to give primacy to Inuit rights, Inuit Systems of Wildlife Management, and Inuit Qaujimajatuqangit. This view is supported by both the Nunavut Agreement and the United Nations Declaration on the Rights of Indigenous Peoples.

Inuit System of Long-term Management of Caribou on Baffin Island

Since time immemorial, Inuit on Qikiqtaaluk (i.e., Baffin Island and nearby islands) have known about and managed the long-term population caribou cycles that occur over the lifetime of an elder, or about 70-90 years. Children are taught by elders that if there are many caribou in their young years, there will be very few caribou when they have children of their own to feed, but if they live to become elders, there will be many caribou again. Inuit know and recognize many more specific signs of shorter-term changes that predict how the caribou population will change in local areas and across the island in the near future throughout the 70-90-year cycles.

During each phase of the long-term cycle, Inuit have harvest management strategies that are stated very simply, but include a huge amount of knowledge and deep understanding of the factors and processes that cause, regulate and affect each phase. For example:

"Snow is no problem unless there have been too many caribou for too long."

This statement reflects Inuit understanding that the amount of lichen food for caribou in winter is a major factor in the growth and decline phases of caribou populations on Qikiqtaaluk, and that over time, caribou can deplete their lichen food resources when there has been "too many caribou for too long". Lichens have no roots and grow very slowly. If caribou find ample lichen after digging through snow, they can maintain their strength and dig through deep, hard snow. If they do not find ample lichen, they become weaker, cannot get enough food, move to other areas, feed in riskier areas like cliff faces, become more susceptible to wolf predation, and

produce fewer calves that may not survive, resulting in gradual but dramatic declines in abundance and distribution over about 15 years in one phase of the cycle. According to IQ, one such period of rapid decline occurred from about 1930 to 1945.

Once there have been "too many caribou for too long" and while lichen food resources have not yet recovered, males can become important for the entire population because, with their heavier and stronger bodies, they may break through hard surface snow and sometimes ice when digging craters for feeding. Thereby, males can enable females and calves to expand the craters that they started, so females and calves can also access more food during winter.

The decline phases of the long-term cycles are all very normal and predictable for Qikiqtaaluk caribou. In the late 1990s and early 2000s after decades of too-many caribou, Inuit across Qikiqtaaluk started to see signs of impending decline, and predicted the next great decline phase of the long-term cycle, which actually occurred from about 2000 to 2015, as predicted by knowledgeable Inuit, who did not need complicated mathematical or scientific models.

The QWB and southern Qikiqtaaluk HTOs called for an IQ-based management plan for the decline phase expected by Inuit during 2005 – 2020. Based on Inuit Qaujimajatuqangit and elders' predictions, the plan was developed but not implemented by government. The QWB also called for development of a similar IQ-based plan for northern Qikiqtaaluk, but this again was not undertaken.

Based on Inuit Qaujimajatuqangit and Inuit observations of caribou in many areas, the long-term cycle has now entered a new critical phase that requires a flexible Inuit system of harvest management that is distinct from that advocated by the Government of Nunavut at least until 2022. The depleted lichen food that caribou need during winter has not yet had time to replenish itself. Caribou have the potential to increase in abundance faster than depleted lichen can grow.

Until 2022, the GN managed the caribou for immediate maximum population increases. By limiting harvesting to mainly males, the caribou may increase in some years, but they will potentially eat as much lichen food as grows each year, but the lichen is already depleted. Such a harvest strategy could keep the caribou's food in an over-eaten, depleted condition for many years, perhaps perpetually depleted so that the abundant Qikiqtaaluk caribou population of the 1970s – 2000s may never return.

In the 1950s and 1960s, Inuit intentionally managed caribou using a very different strategy, one that applies again today! Even though there were few caribou in the 1950s, Inuit elders knew that:

> *"We had to keep harvesting the caribou. The land needed to rest."*

In the 1950s and 1960s, important Inuit elders knew that there had been *"too many caribou for too long"* in previous decades, and caribou abundance declined, just as Inuit had predicted in the 1930s and 1940s because they did not have enough lichen to eat. Inuit knew that "*the land had to rest"* with a low abundance of caribou maintained through harvesting by Inuit for two or three decades in the late 1940s, 1950s and 1960s. Inuit understand how large Baffin Island is and realize that it is very difficult cause extensive caribou depletion across the island. Continued harvesting of caribou by Inuit during the late 1940s, '50s and '60s allowed the slow-

growing winter food of caribou, lichen, to recover in large amounts over large areas again, so it could support great abundances of caribou seen in subsequent decades.

The great abundance of caribou on Qikiqtaaluk from about 1970 to the early 2000s was not an unexplainable accident. It occurred because of the Inuit System of Caribou Management and intentional harvesting by Inuit near local communities, camps and elsewhere during previous decades.

This proven and ancestral Inuit System of c\Caribou Management on Baffin Island, based on Inuit Qaujimajatuqangit, must be enabled, implemented and adjusted as needed and as advised by Inuit using their own IQ over the next 10-20 years, so that the critical food of caribou, lichen, may recover in sufficient quantities to support the next great abundance of Qikiqtaaluk caribou, consistent with all of the principles of conservation in the Nunavut Agreement and the QWB's objectives for the conservation of Baffin Island caribou.

Consultation:

After consultations with the HTOs, in 2018, 2019 and 2020 the QWB proposed modest increases in the 2015 Total Allowable Harvest (TAH) for Baffin Island caribou. The NWMB did not approve any of these requests, although a minor adjustment in the harvest of females was allowed in 2020.

The 2021 Annual General Meeting (AGM) of the QWB discussed the IQ and the Inuit systems of Baffin Island caribou management, with advice and input from staff of NTI. During subsequent AGMs of the QWB and at other meetings, most HTO delegates have repeatedly recommended larger annual increases in the TAH, and increases the NQL to allow for more females to be harvested.

The QWB has passed interim regional caribou harvesting by-laws so that community allocations may be adjusted from year to year so that communities that harvest in excess of their allocations do not put other communities at a disadvantage.

"The land needs to rest."

Recommendation:

The QWB Executive recommends that the Nunavut Wildlife Management Board (NWMB) approves and implements the following revisions to the Inuit system of caribou harvest management on Baffin Island starting on January 1, 2026, or sooner:

6. a) An increase of the Total Allowable Harvest (TAH) for Baffin Island caribou of 50 on January 1, 2026 or before, for a total of 550 caribou for the harvest year from July 1, 2025 to June 30, 2026; and

   c) Then subsequent annual increases of the TAH for Baffin Island caribou of 100 in each of the next six (6) years to reach a total of 1,150 on July 1, 2031.

7. Adjust the Non-Quota Limitation (NQL) so that the maximum harvest of females will be 40% females of the annual TAH (e.g., up to 220 females in 2025-2026, up to 460 females in 2031-2032);
8. Until at least June 30, 2032, the Basic Needs Level for Inuit will equal the entire annual TAH.
9. The QWB may reassess changes in the TAH and NQL based on input received from HTOs and any other available and relevant information. If a change in subsequent TAHs and NQLs appear warranted, the QWB will make a proposal to the NWMB for any recommended changes. Input from other co-management partners will be solicited through NWMB consultations and other processes
10. The QWB Executive Committee will be responsible for HTO allocations in accordance with its interim regional Baffin Island caribou harvesting by-laws.
6. During 2031-2032, the QWB will assess whether or not the TAH and NQL may be eliminated on July 1, 2032, or if the TAH and NQL should be continued or modified in future, based on all available information. The GN will be consulted during this assessment process. The QWB will make a proposal to the NWMB for any recommended changes.

Literature Cited:

Ferguson et al. 1998. Inuit Knowledge of Long-term Changes in a Population of Arctic Tundra Caribou. Arctic 51: 201–219.

Prepared by: Michael Ferguson and Kolola Pitsiulak, Qikiqtaaluk Wildlife Board

Date: Septer 5, 2025

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ᓄᓇᕗᒻᒥ ᐅᒪᔨᓕᕆᔨᖏᔪᐊᖅᑯᓘᑎᒃ ᑲᑎᒪᔨᖅᑭᓐᓇᓕᒃ

ᐅᔪᖅᑐ

ᑐᓴᖅᑎᓴᔪᑎᑎ: ᐃᑦᐸᐃᑦ **ᐃᓄᐃᑦ:** *X*

ᐱᓵᒋᑦ: ᓄᓇᕗᒻᒥ ᐊᓂᓕᕆᓂᖏᓐᓄᑦ ᐊᐅᓪᓴᓴᖅᑎᑎᒋᑦ <ᖃᓄᐅᑎᖅᑐ (<ᖃᓄᐅᑦ) ᐋᖅᑭᑦᑕᐅᓪᓕᐅᖅᑐᖅ ᓄᓇᕗᑦ ᑲᖏᒪᖅᓯᓄᑦ (GN) ᐱᓕᕆᔨᖅᑎᖅᑲᖅᑐᑎᒃ ᐊᐅᓪᓴᓴᖅᑎᑎᒋᓄᑦ ᐊᒻᒪᓗ ᑐᑭᒧᐊᔪᑎᒃᓴᒻᒥᓪᓕᐅᖅᑐᖅ ᐊᓐᓄᖅᑭᑦᑕᓂᖏᓪᓕᒃ ᐊᒻᒪᓗ ᐊᐅᓪᓴᓴᑦᑕᐅᓪᓗᓂ ᓄᓇᕗᓪᓕᒧᑦ. ᑭᓱᐊᓂᓪ, ᐊᐅᓪᓴᓴᖏᓪᓕᒃ ᐊᐅᓪᓴᓐᖓᑦ ᐱᓵᓕᕆᔨᓴᓴᐊᒻᒪᓕᕆᐊᔭᖑᓐᓇᖅᑐᖅᑐᖅ ᓴᖅᑭᖅᑕᐅᓂᖓᓕᒫᑦ ᐊᖏᓇᓱᓐᓂᖏᓪᓕᒃ ᐊᐅᓪᓴᓴᖏᔭᓐᓄᑦ ᐊᒻᒪᓗ ᓇᓐᐃᓚᐅᑎᒃᓴᒃᑯᑦ (HACCS). ᐊᐅᓪᓴᓐᖓᑦ ᑲᓱᓱᓴᒫᑎᓴᐅᑦᑕᐅᓪᓴᖅᑐᖅ ᐊᐅᓪᓴᓴᖏᓪᓕᒃ ᑐᑭᒧᐊᔪᑎᒃ, ᑕᒪᑭᑲᓐ <ᖃᓄᐅᑎᑦ ᐊᒻᒪᓗ HACCS ᓇᓪᓴᓂᓱᓴᒪᓱᐊᖅᑯᑦ ᐃᓕᓴᒃᑯᑦ 5 *ᓄᓇᕗᑦ ᐋᖅᑭᒃᑯᑎᖅᑯᓘᓐᓄᑦ.* ᐃᓕᖅᑭᑦ <ᖃᓄᐅᑎᐅᓵᕐᑐᑦ ᐊᒻᒪᓗ HACCS ᓴᓂᐊᑎᑦᓱᒪᓕᕆᐊᖅᑭᑦᑕᐅᓪᓴᖅᑐᖅ RWO ᐊᒻᒪᓗ HTO ᐃᓱᒪᓕᐅᑭᓄᓇᕗᑦᑎᓐᓄᑦᓂᖅ ᓴᐳᕐᐅᓱᒪᔪᓂᖅ *ᓄᓇᕗᑦ ᐋᖅᑭᒃᑯᑎᖅᑯᓘᓐᓄᑦ.*

ᑲᒪᒋᐊᕐᓇᖅᑐᓂᖓ:

ᐋᖅᑭᖅᑕᐅᓕᐅᖅᑎᓄᓐᓇᒧᑦ ᐊᒻᒪᓗ ᐊᑐᓕᖅᑕᐅᓗᓂᑐ HACCS, ᐋᖅᑲᓇᓱᑕᐅᔪᓄᓐᓇᖅᑐᓂᖅ ᓱᓴᐃᕐᔪᐊᑎ ᐊᐅᓕᑕᐅᓕᐅᖅᑐᖅᑐᖅ ᑐᑭᓯᒪᓴᖅᑎᑦᓴᓂᖅᑐᓂᖅ ᐋᖅᑭᕕᑎᖅᑯᓘᓐᓄᑦ (MOU) ᓄᓇᕗᑦ ᑲᖏᒪᖅᓯᓄᑦ ᐊᒻᒪᓗ RWO. RWO ᓴᓂᔨᐃᓱᒪᓕᓴᖅᑐᑦ ᐃᓗᓕᖅᑯᓘᓐᓄᑦ ᐊᐅᓪᓴᓴᖏᓪᓕᒃ ᐊᐅᓪᓴᓴᖏᓪᓕᒃᑯᓘᓐᓄᑦ GN-ᓂᑦ ᐊᐅᓪᓴᓴᖏᓪᓕᒃᓂᓕᕆᓂᑦ ᓄᓇᕗᑦᑕᐅᔪᓄᓐᓇᖅᑐᖅᑐᖅ ᑲᒪᖓᖅᓱᓕᓴᖅᑯᓘᑎᒃ.

HACCS ᐊᑐᑲᐃᓄᐅᓂᐊᖅᑕᒥᓂᖅ ᐋᖅᑭᖅᑕᐅᓕᐅᖅᑐᖅᑐᖅ 2019 ᐊᒻᒪᓗ ᒥᓂᑕᒧᑦ ᐋᖅᑭᖅᑕᐅᓗᓂ ᕆᐊᖅᑕᐅᓗᓂ 2022. ᓄᓇᕗᑦ ᑐᖓᔪᓕᒃ ᑎᒥᖓ ᐃᑲᔨᖅᓴᐃᓚᑐᐊᖅᑎᓪᓗᑎᑦ ᐊᑐᓂᓪᓕᒥᐅ ᓯᑕᒪᑦ ᐃᓚᖅᑭᖅᓯᑐᑦ ᑭᓚᖅᑐᑦᑕᐅᓪᓴᖅᑐᖅ (NQL), HACCS ᐅᖓᑖᓂᒃᓴᖅᓴᒪᔨᑦ ᐊᐅᓪᓴᓴᖏᓪᓕᒃᑯᓘᓐᓄᑦ RWO ᐊᒻᒪᓗ HTO ᐃᓱᒪᓂᑦ 5 ᓄᓇᕗᑦ ᐋᖅᑭᒃᑯᑎᖅᑯᓘᓐᓄᑦ. ᐱᓗᐊᖅᑐᒥ, ᑯᑕᒃᖅᑎᑕᐅᓴᒪᓕᓴᖅ ᐊᒻᒪᓗ ᑲᔨᓴᑎᑕᐅᓕᓪᓕᒥ ᐅᒪᔨᔨᑎᑕᖅᓴᓕᐅᖅᑐᖅ ᐅᒪᔨᖅᑕᐅᔪᓄᓐᓇᖅᑐᑦ (BNL) ᐊᒻᒪᓗ ᐋᖅᑭᒃᑭᑦᑕᐅᓴᒪᔨᑦ ᐅᒪᔨᔨᑎᖅᓴᓕᐅᖅᑐᖅ (ᐋᖅᑭᒃᑭᑦᐊᖅᓴᒪᔨᑦ BNL) HTO-ᖏᑎᒍᑦ ᐊᐃᑲᖅᑐᐊᖅᓴᒪᓕᓪᓕᒥ, ᓄᑭᖅᑲᑕᐅᔨᔨᖅ ᐊᐅᓪᓴᓴᖏᔭᓐᓂᓗᓗ RWO-ᓄᑦ, ᐊᐃᑲᐃᖅᓴᒪᓕᐅᖅᓱᒪᑦ 5.7.6 (b) ᐋᖅᑭᒃᑯᑎᓂ.

ᑐᓴᖅᑕᐅᑐᓂᖅᓱᖅ:

ᓄᓇᕗᑦ ᑐᖓᔪᓕᒃ ᑎᒥᖅᓱᖅ (ᑐᖓᔪᓕᒃ) ᐱᓕᕆᔨᖏᓪᓕ ᐃᓕᐅᖅᑲᓕᑕᖅᓴᒪᔨᑦ ᐅᖅᑲᐅᓴᕆᔨᐅᑎᓪᓗᒍ, ᐃᓕᑎᕆᐅᓪᓗᓂᒃ ᓇᓗᓇᐃᖅᓴᒪᔨᐅᑎᒃᑯᓘᓐᓄᑦ ᐅᓂᐅᑲᖅᑐᑎᖅᑲᖅᑐᑦᑕᓂᖅᑐᒃᑯᓗ ᐊᑦᑕᕐᓂᑐᖅᑲᖅᑐᑦᑕᓂᖅᑐᒃᑯᓘᓗ ᐊᖅᑭᒍᓕᒫᖅᑐᖅᑎᑦᓗ ᐃᓕᓴᕐᔪᐊᖅᑐᒃᑯᓘᓐᓄᑦ ᐊᐃᑉᐸᓂᖅᑐᖅᑐᒥ ᐅᑉᐳᖅᑕᓗᐊᓕᒐ ᐱᓱᓕᓂᓂ ᐊᐃᖅᑐᓂᖅᑐᖅᑐᒥ ᐅᓂᐅᑲᖅᑐᑎᖅᑲᖅᑐᑦᑕᓂᖅᑐᒃᑯᓘᓐᓄᑦ. ᖃᐅᖅᓱᒍᑦ, ᐃᓚᖓᑕᐅᑕᖅᑐᑦᑕᓂᖅᑐᒃᑯᓘᓐᓄᑦ ᐊᐃᑉᐸᓂᖅᑐᖅᑐᖅ ᐃᓕᓴᕐᔪᐊᖅᑐᒃᑯᓘᓐᓄᑦ ᐅᒪᔨᖅᑕᐅᔪᓄᓐᓇᖅᑐᖅᑐᖅ – ᖃᑭᐃᐅᑕᖅ, ᐅᒪᔨᖅᑕᒃ ᐅᑯᓇᓐᓴᖅᓯᓕᕆᔨᖅᑐᖅ 3 RWO ᐊᒻᒪᓗ 3 ᐊᐃᑲᔨᖅᓯᓕᕆᓂᖅ ᐃᓄᐃᑦ ᐅᑐᐊᖅᑎᑦᓴᖅᑐᓂᖅᑐᖅ.

ᐊᑐᕐᑯᐅᓴᓱᑦ:

ᓄᓇᕗᑦ ᑐᖓᔪᓕᒃ NWMB-ᑯᓘᑐᒃ ᐊᒻᒪᓗ ᓄᓇᕗᑦ ᓱᓴᒪᓂᖅᑐᖅ ᐱᓕᕆᔨᖅᑎᓪᓗᑎᑦᓴᖅᑐᖅ ᑐᖓᔪᓕᒃᑯᑯᓘᓐᓄᑦ ᐊᒻᒪᓗ RWO ᐃᓇᖅᑭᖅᑕᐅᓗᑎᒃ HACCS RWO ᒪᓕᑦᓴᓕᖅᑭᓘᓐᓄᑦ ᐊᐅᓪᓴᓴᖏᓪᓕᒧᑦ ᑯᑕᐅᑐ ᑐᓂᐅᖅᓴᒃᐃᐃᓪᓕᖅᑎᑦ, ᓇᓗᓇᐃᖅᓴᒪᓕᐅᖅᑐᖅ ᒪᓕᑦᓴᓕᖅᑭᓘᓐᓄᑦ ᐃᓕᓴᕐᔪᐊᖅᑐᒃᑯᓘᓐᓄᑦ RWO-ᓄᑦ.

ᑎᑎᕆᖅᑕᖅᑐᖅ:

ᑭᐅᓚᐅᑦ ᓂᕆᓇᖅᓴᕆᔨᖅ
ᑐᑭᒧᐊᖅᑎᑎᒃᓴ, ᐅᒪᔨᓕᕆᓚᓂᖏᓪᓕᒧᑦ ᐊᖓᑎᓕᕆᓚᓂᖏᓪᓕᒧᑦᓗ
ᓄᓇᕗᑦ ᑐᖓᔪᓕᒃ

ᐅᑦᓗᖅ: ᓴᕐᑎᑎᒪᓚ 2nd, 2025

Fisheries and Oceans Canada (DFO)
Arctic Region

Nunavut Wildlife Management Board (NWMB) Regular Meeting
November 5, 2025
Amy Amos, Regional Director General Arctic Region

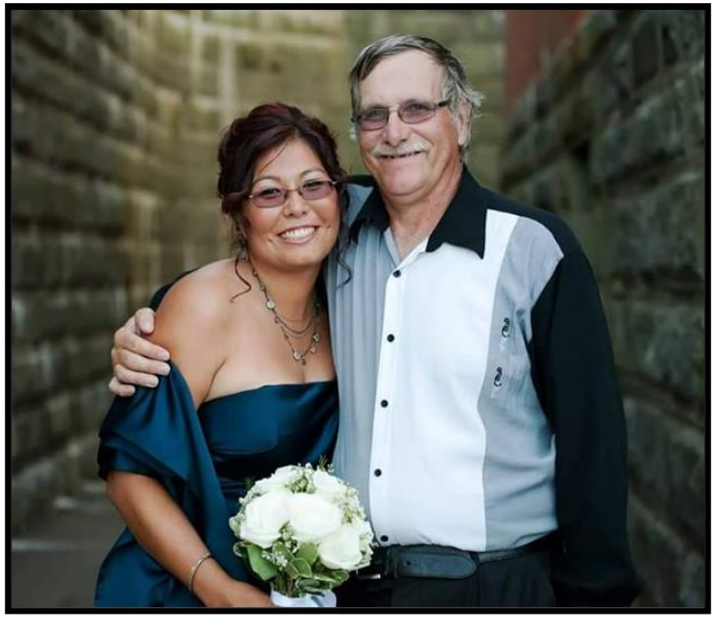


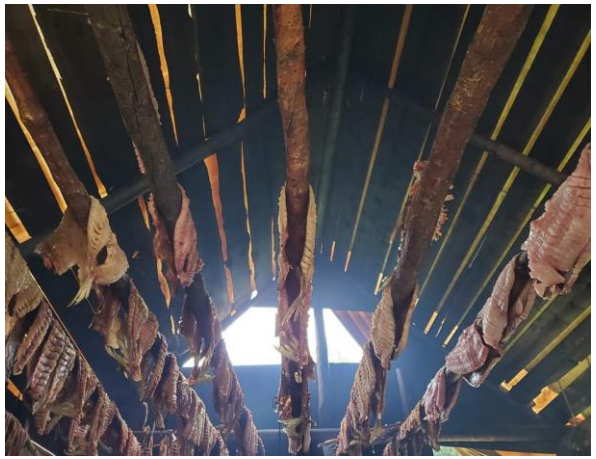
Canada

Introduction

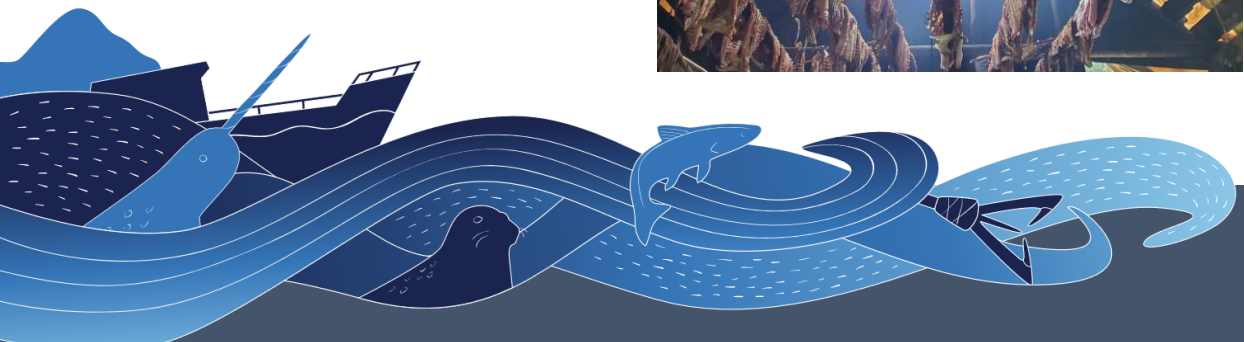












The Creation of Arctic Region: Background

2018: Announcement of the Arctic Region

Jonathan Wilkinson, then Minister of Fisheries, Oceans and the Canadian Coast Guard, and Natan Obed, President of ITK, announced the creation of a stand-alone Arctic Region, including the four regions of Inuit Nunangat.

2018 – 2020: Engagement Period

50+ sessions held with Indigenous organizations, Northern partners, and provincial/territorial governments focused on refining service delivery and regional priorities

2021: New Regions Established

The DFO "Central and Arctic" Region is dissolved and two new regions were created 1) the Arctic Region and 2) Ontario and Prairies Region

2021: Boundary Announcement

Spanning from the Yukon North Slope, Northwest Territories, Nunavut, Nunavik, Nunatsiavut, Hudson Bay, James Bay, and all of Inuit Nunangat, reflecting key principles from partner engagement.

2021: New Executive Hires in the Arctic, Strengthening Leadership

Andrew Randall, Arviat

Amy Amos, Inuvik

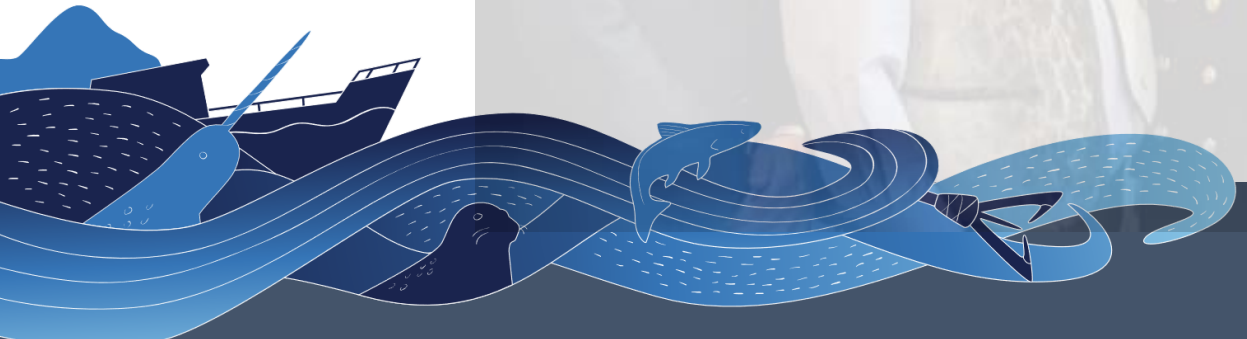
2023: Leadership Changes

Jody Proctor, Regional Director General, replacing Gabe Nirlungayuk

2024: Core Programs transitioned to the Arctic Region + Strengthening Leadership

All remaining programs transitioned to Arctic Region

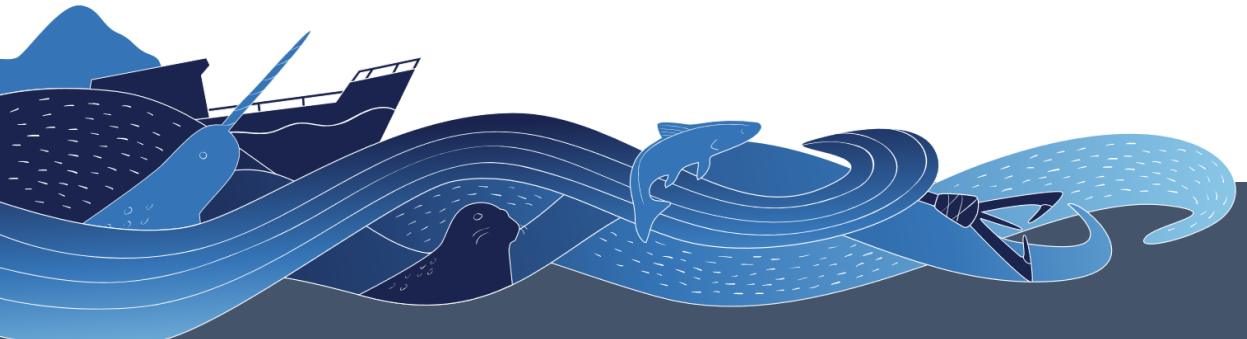
Amy Amos announced as RDG, Arctic Region



Canada

Arctic Region – Senior Executive Structure

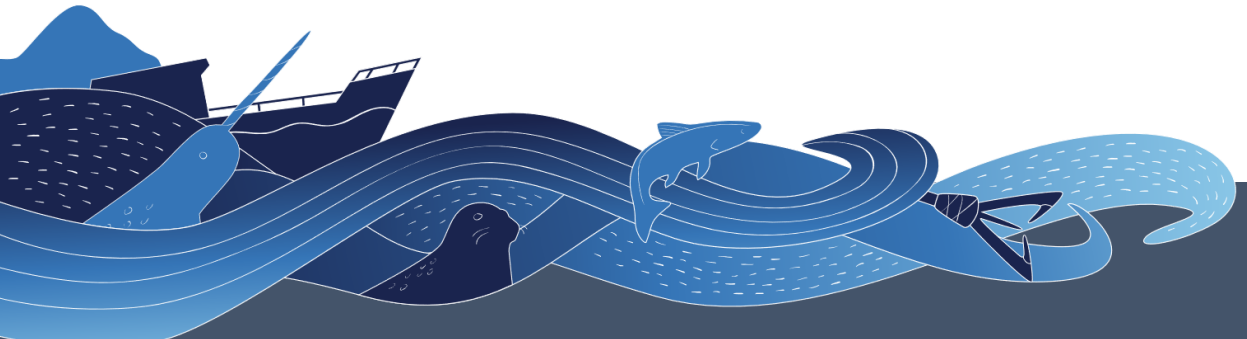
- Regional Director General
  Amy Amos
  (Inuvik)
  - Director, Arctic Operations
    Micheline Kilabuk-Cote
    (Iqaluit)
  - Director, Science
    Joe Justus
    (Winnipeg)
  - Director, Fisheries Management
    Christian Marcoux
    (Yellowknife)
  - Director, Ecosystems Management
    Andrew Randall
    (Arviat)
    - Director, Oceans Management
      Georgina Williston
      (Yellowknife)
    - Director, Fish and Fish Habitat
      Cal Wenghofer
      (Rankin Inlet)
  - Director, Conservation and Protection
    Gerald Fillatre
    (Yellowknife)



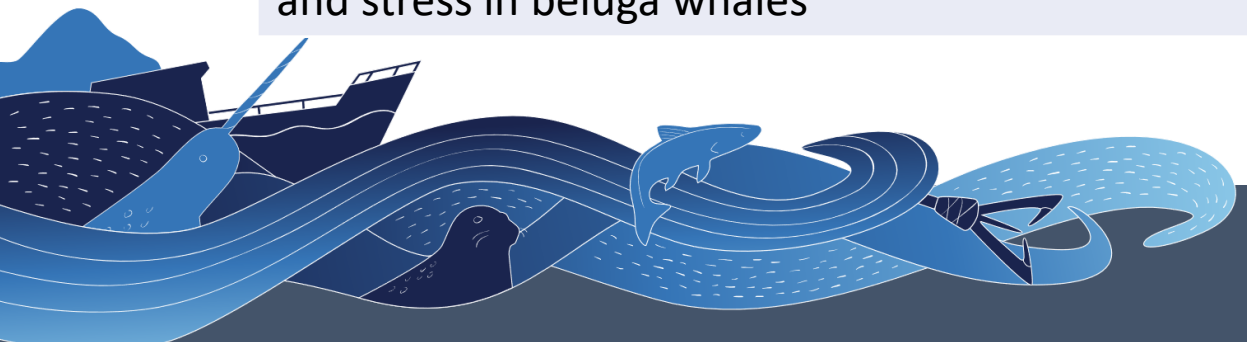
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Overview of Arctic Region Employment
Implementation of Nunavut Agreement Article 23

- In 2023, 17 positions were encumbered by Nunavut Inuit. As of March 2025, 24 positions are encumbered by Nunavut Inuit, which represents a 9% increase (encumbered positions only).
- In the 2023-2033 Inuit Employment Plan, DFO-CCG committed in its short-term goal (by March 2026) to increase its percentage of Inuit in encumbered position to 40%.
- As of March 2025, DFO-CCG currently stands at 42%.



Projects supported using Nunavut Implementation Funds in 2025/26
Age Validation study of Arctic Char in Cambridge Bay
Developing the Inshore Greenland Halibut Fishery
Cumberland Sound Turbot Management Area Fish Plan Sampling
Fisheries Research of key Cumberland sound Arctic Char stocks
Monitoring Interactions of subarctic whale species with Baffin Bay Fisheries
Collaborative Fieldwork to advance ECWG Bowhead Whale population using genetics and drone imagery
Monitoring of key Arctic Char stocks near Qikiqtarjuaq using community-based monitoring
Narwhal and Beluga Tagging, biopsy, and drone work in Admiralty Inlet, Foxe Basin, and Hudson Strait
Community Based Assessment of Walrus Vulnerability to disturbance
Determining of epigenetic and chronological aging methods correlate with environmental change and stress in beluga whales



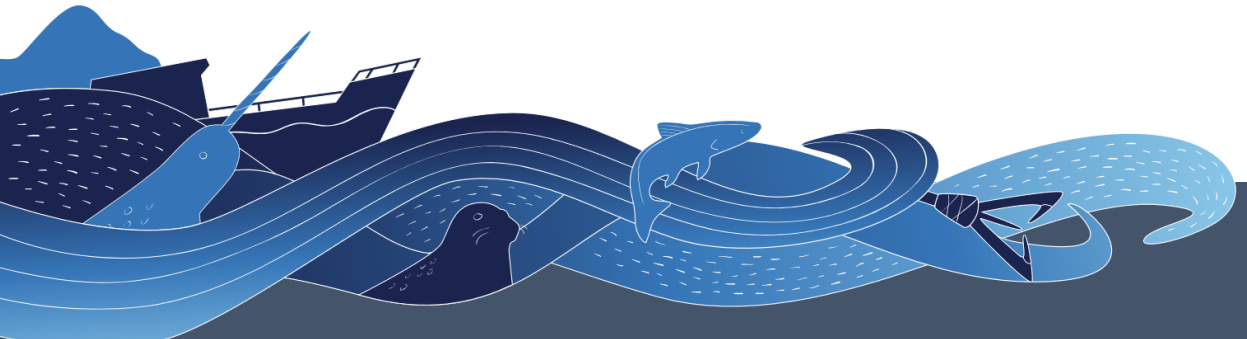
Board & DFO Relationship



Our continued collaboration is key to our shared objectives

Thank you for your time

Mársı | Kinanāskomitin | Thank you | Merci | Hąį’ | Mutna | Quana | Qujannamiik | Nakurmiik | Quyanainni | Máhsı | Máhsı | Mahsì | Gunalchéesh | Kwǟnäs̈chis | Máhsin cho | Tsin’jj choh | Sógá sénlá’ | Quanaqqutit | ᓇᔨᖏᑦ



Canada

ᑐᓂᔨᐅᕐᖓᖅ
ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᓂᖅᐸᐃᑉᐳᑦ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ

ᑐᕐᖏᓚᐅᖅ

ᑐᓴᖅᑎᕐᔪᒍᑎ: X ᐃᓕᓴᕐᐅᕐᑭᑦ: ᐊᑐᓕᖅᑯᓂᕐᔪᐃᖅᑐᑦ:

ᐱᓕᕐᓯᑦ: ᑐᓴᖅᑎᕐᔪᒍᑎ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓕᕐᓯᑦ ᓄᓇᕗᑦ ᐃᖅᑲᐅᓕᕆᓂᖅᓱᒃᑐᑦ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ

ᑲᒪᒋᐅᔨᓕᕆᔨᓐᓇᓚ:

ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᖅᑲᒋᐅᔨᓕᕐᓯᑦ ᓄᓇᕗᑦ ᐃᖅᑲᐅᓕᕆᓂᖅᓱᒃᑐᑦ ᐃᓄᐃᑦ ᑲᒪᒋᐅᔨᒪᐃᑦᑐᑐᖅᑎᖅᑕᖅᑲᒥᖅ ᐊᓐᓂᑎᑕᐅᔨᓕᕆᔨᒥᖅ ᑲᓇᕗᑦ ᓱᓕᓐᓄᑦ. ᐃᓴᖅᑲᐅᓕᕆᓂᖅᓱᒃᑐᑦ ᓄᓇᕗᑦ ᓄᓇᓐᓄᑦ ᓇᓗᓄᐃᖅᑕᐅᔨᓕᕆᓂᓐᓇᓂ (NSA) ᓂᓪ ᐊᐅᓪᓚᑕᐅᔨᑦ ᐊᑐᖅᑕᐅᔨᔪᓂᖅ ᐊᐅᓪᓚᕐᔪᕐᑎᓂᖅ ᑕᕝᕙᓐᓇᖅᑎᖅᑲᐊᓐᓂᑕᑐᓂᖅ ᐊᓪᓚᓪᓂ ᑲᔪᓯᑕᐃᓐᓇᐅᔨᓐᓂᑕᑐᓂᖅ ᐃᓄᐃᑦ ᐱᓕᔪᓐᓄᑦᑐᓂᖅ ᐊᓐᓚᓗ ᐊᐳᐅᓐᓂᑕᑐᖅ ᐃᓴᖅᑲᓪᓚᕐᔪᖅᑐᑦ <ᖃᓇᑕᐅᔨᓕᕆᔨᓂᖅ *ᓄᓇᕗᑦ ᐊᖏᑕᑎᖏᓐᓂᔅᓴᑐᓂᖅ (NA).* ᓄᑕᓂᖅ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᐅᖅᑲᒪᒋᑕ ᐅᖓᓚᐅᖅᑭᓂᕐᔪᐃᓕᕐᓂᑦᑕᑐᓂ ᐊᖏᔪᓂᕐᓂᑦᑐᖅ ᐱᓕᕐᓯᑦ ᐅᓴᐅᓕᕆᓂᖅ ᐊᓚᓗ ᐅᓴᓕᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅᑐᖅ *NA,* ᓂᕐᔭᓗᐊᔪᑎᖅᓴᐅᑎᖅᑐᓂᖅ ᑲᐃᓚᒋᖅᑎᑦᓯᖅᑐᓂᖅ ᐱᓇᓂᐊᕆᓚᐅᖅᓯᓐᓂᓄᑦ ᓄᓇᖅᑲᖅᑲᖅᓂᓚᕐᓯᑦ ᓄᓇᓕᖅᑎᓂ, ᐃᑲᔪᑎᓂᖅ ᐃᓄᐃᑦ ᓄᓇᖅᓯᓄᑦ ᐊᑐᐊᓴᖅᑲᖅᑐᓂᖅ ᐅᓂᒃᑳᖅᑲᖅᓯᓐᓂᓄᑦ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅᑐᖅ, ᐊᓚᓗ ᓴᖅᑭᑎᓕᖅᑐᖅ ᐅᓪᓗᒥᓂᐅᔨᑎᓂᖅ ᐃᓴᖅᑲᓪᓚᕐᔪᐊᑎᓂᖅ <ᖃᓇᓂᓚᕐᔪᖅᑐᑦ ᒪᑭᒪᓇᕐᓂᖅᓯᓐᓂᓄᑦ ᐃᓴᖅᑲᓪᓚᕐᔪᖅᑐᑦ NSA–ᒥ.

ᐃᒪᓕᖅᑎᑕᓚᓚᐅᑉᐳᑦ ᑲᓇᑕᒥ (DFO) ᐊᓚᓗ ᓄᓇᕗᑦ ᑐᖏᓚᓪᓕᐅᖅ ᑎᒥᖏᑲ (ᑐᖏᓚᓪᓕᐅᖅ) ᑐᖏᓚᓴᐃᔪᑦ ᑕᕐᓂᒥᖏᑲ ᐱᔪᐊᑎᖅᓴᖅᑲᒥᖅ ᐊᖏᔪᑎᑎᑕᐅᑭᓴᑦᑐᓂᑦ ᖃᐅᔨᐃᑕᖏᑲᖏᑲ ᑐᓂᔪᕐᓂᑦ ᓄᓇᕗᑦ ᐃᖅᑲᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅᓱᒃᑐᑦ (NFR). ᑕᑯᓗ ᐊᖏᔪᑎᑎᓂᓂᑭᑐᑎ ᐱᓯᓕᓪᓚᑎᓂᓂᔨᑕᑉᐳᖅ ᐊᖏᔪᑎᑎᑕᐅᑭᓴᖅᓂᒪᔨᓕᕆᓂᖅ ᓴᖅᑭᖅᑲᑕᐅᓕᐅᖅᑲᒥᖅ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ ᐅᕐᑐᓵᐱᓚ 2024 ᐊᓚᓗ ᔪᓐᓂ 2025 ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ.

ᓄᓇᕗᑦ ᐃᓴᖅᑲᓪᓚᕐᔪᕐᑎᑦ ᐊᐅᓪᓚᕐᔪᕐᑎᓄᑦ ᐱᓴᓚᐊᑭᓪᓚᑎᓚᔅᓴᑦ (NFRWG) ᑕᒃᑯᓚᖏᓐᓇᖅᑐᑦ ᐃᓚᒋᐅᔨᓂᖅᓱᒃᑐᑦ ᐃᓴᖅᑲᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᑕᓚᐅᑕᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ ᐃᓴᖅᑲᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᑕᓚᐅᑕᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅᑦᓗ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ, ᑐᖏᓚᓪᓕᐅᖅ, ᒪᑭᕕᖅ, ᐃᓴᖅᑭᓪᓗ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ ᓴᖅᑭᒪ (CNG; ᐃᓚᒋᐅᔨᓚᓚᐅᖅᑐᑦ ᐅᑭᐊᑦᓴᖏᓐᓂᓄᑦ 2024) ᐊᓚᓗ ᓄᓇᕗᑦ ᓴᖅᑭᒪᖏᓐ (GN). ᓄᓇᕗᑦ ᐅᒪᔨᓕᕆᓚᓚᐅᖅᑕᐅᓂᖅᓱᒃᑐᑦ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ (NWMB), ᓄᓇᕕᖅ ᑕᓚᐅᖅᑎᑕᐅᓂᖅ ᐅᒪᔨᓕᕆᓚᐅᖅᑕᐅᓂᖅ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ (NMRWB) ᐊᓚᓗ ᐊᐅᐅᑕᓚᓚᐅᖅᑕᐅᓂᖅ ᐅᒪᔨᓕᕆᓚᐅᖅᑕᐅᓂᖅᓱᒃᑐᑦ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ (EMRWB; ᐃᓕᓴᕐᐅᕐᑭᑦᓴᖅᑐᑦ ᐱᑎᐊᓕᔪᖅᓯᓐᓂᓄᑦ 2025) ᑐᔪᓴᖅᑐᑭᓚᖅᑕᖅᑲᒥᖅ ᑲᑎᒪᓂᖅᓯᓐᓂᓄᑦ. NFRWG ᑲᔪᓯᑎᕐᔪᐊᓕᐅᖅᑐᑦ ᐋᖅᑭᐸᓴᖅᑕᐅᔨᔪᓂᖅ ᐊᑐᐊᑦᓴᖅᑐᑦ ᐱᔪᐊᑎᖅᓴᖅᑲᒥᖅ (PIP), ᐅᓗᐅᐃᖅᑲᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᐊᑐᐊᑦᓴᖅᑐᑦ ᐊᓚᓗ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓄᑦ ᑐᕐᖏᓚᓪᓕᐅᖅᑕᓄᑦ ᑕᕐᓂᒥᖏᑲᓪᓚᕐᔪᐊᑎᓄᑦ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓄᑦ, ᑎᓂᓪᓚᓇᓗ 2024.

ᐃᓕᐅᑎᑕᐅᓂᖅᓱᒃᑐᑦ ᐱᔪᐊᑎᖅᓴᖅᑲᒥᖅ–ᑎᐃᒥᐊᖅᑳᑦ ᐊᓚᓗ ᑎᐃᒥᐊᖅᑲᑕᐅᔨᔪᓂ

ᐱᐊᓂᖅᑕᐅᓂᕐᔪᐃᓕᕐᓯᑦ PIP ᑐᓂᐅᖅᑲᖅᑕᐅᓕᐅᖅᑐᖅᑐᖅ ᓂᓕᒃᓄᑦ ᐱᑎᐊᓕᔪᖅᓯᓐᓂᓄᑦ 2025 ᑐᔪᖅᑎᑦᓂᓕᕆᑕᐅᓪᓗᒥᓂ ᑐᔪᓚᐊᖅᖃᐅᑎᓪᓗᒥᓂ ᐱᔪᐊᑎᖅᓴᖅᑲᒥᖅ ᑎᐃᒥᐊᖅᑳᑦ, ᑎᐃᒥᐊᖅᑲᑕᐅᔨᔪᓂ, ᐊᓚᓗ ᐊᓂᔪᓂ ᑲᒪᒋᐅᔨᒪᓚᖅᑐᑦ. ᐊᖃᑦᑐᖅᓂᕐᔪᖅᑐᓂ ᐃᓕᐅᑎᑕᐅᓂᖅᓱᒃᑐᑦ ᐅᑎᖅᑕᐅᓂᖅᓱᒃᑐᑦ ᐃᓚᒋᐅᔨᓂᖅᓱᒃᑐᑦ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ ᑎᐃᒥᐊᖅᑳᑦ ᐃᓚᒋᐅᔨᓂᖅᓱᒃᑐᑦ ᐊᐅᓪᓚᕐᔪᕐᑎᑦᑕᓂᖅ

ᐃᒪᖏᐅᑕᓕᕆᔨᒃᑯᑦ ᑐᓴᖅᑎᕋᒍᑕᐅᔭᓂᐊᖅᑐᑦ NFR ᐊᖑᐃᓚᒃᑯᓐᓄᑦ ᓄᓇᕕᓗᒥ ᐊᓚᓗᓐ ᓄᓇᕗᓗᒥ ᐊᕕᑐᖅᓯᒪᔪᑦ ᐅᒪᔪᓕᕆᔨᑦ ᑲᒪᓚᕕᓐᓄᑦ AGM–ᖏᓐᓂᒃ ᐊᑯᓂᖅᓴᓂᒃ ᐅᑦᑐᕐᐱᓗ ᐊᓚᓗᓐ ᑲᓱᓪᓗᓀ 2025.

ᐃᒪᖏᐅᑕᓕᕆᔨᒃᑯᑦ ᐃᑕᐅᑎᕋᓱᓂᐊᖅᑐᒥᔪᑦ ᐊᓱᖏᑦᓂᒃ ᓱᓕᑦᓂᖏᓐᓇᖅᑐᓂᒃ NFRWG ᖃᐅᔨᒪᔭᒪᓇᓴᖏᐊᖅᑐᓂᒃ ᓄᑦᑕᓂᒃ ᐊᑐᐊᑦᓂᒃ ᐊᓚᓗᓐ ᓱᕐᓂᕆᑎᓐᓂ ᐊᐅᑕᔪᓕᐅᕙᓂᒃ ᐅᓪᓗᓈᓕᐅᕙᑯᓕᒃᑕᒥ, ᑐᑭᓱᒪᓕᐅᔭᓗᑎᒃ, ᐊᓚᓗᓐ ᐊᕐᑐᐊᔪᓂᒃ ᐅᖃᐅᔨᓴᖏᓂᒃ ᑐᓴᓈᖏᐊᐅᔭᓗᑎᒃ, ᐃᑕᒥᓕᐅᑕᓗᑎᒃ ᔪᑎᐅᐅᐊᐅᓂᒃ ᑯᑉᓴᓕᐅᔭᓗᑎᒃ ᐊᓚᓗᓐ ᒪᓕᐅᔪᖅ ᐅᑎᒪᔨᓂ, ᐊᑎᑕᐅᐊᓕᓴᐃᓂ ᐊᕐᓚᐃᑦ ᐅᑎᒪᔨᖏᓐᓂ ᐊᓚᓗᓐ ᒪᓕᐅᔪᖅᖏᓐᓂ.

ᐃᒪᖏᐅᑕᓕᕆᔨᒃᑯᑦ ᐅᖃᕈᓐᓇᖅᑐᓂᕐᒥᓂ ᐊᓕᐸᓐᖏᓐᓂᑐᓄᓂ ᖃᐅᔨᒪᔨᑕᑦ ᐅᑎᒪᔨᕐᓯᓄᓂ ᐅᑎᒪᓂᕐᓄᓂ 2026 ᑐᓴᖅᑎᕋᓱᓀᒃ ᑲᓇᓐᖏᑦ ᐅᑭᐅᖅᑕᖅᑐᒥ ᐃᓅᒥᐊᖅᑎᐅᑕᓂᒃ ᑎᒍᒥᐊᖅᑎᐅᑕᓂᒃ ᖃᐅᔨᒪᔨᑕᑦ ᐅᑎᒪᔨᕐᓯᓄᓂᖏᓐᓂ ᐊᓚᓗᓐ ᐅᑭᐅᖅᑕᖅᑐᒥ ᑭᖑᓪᓗᒍ ᐃᓕᓴᕐᓂᖏᓐᓂ ᐊᓚᓗᓐ ᒪᓕᐅᔪᖅᖏᓐᓂ.

NFRWG ᐊᐸᖅᑭᕋᐃᓂᐊᖅᑐᑦ “ᑐᕿᒪᐅᖅᑕᑭᓕᓂ” ᐅᓂᒃᑲᓂᒃ ᐃᒪᒋᐅᔨᓂᒃ ᐊᑐᓂ ᐱᔨᓇᐅᑎᓂᒃ–ᑎᒍᒥᐊᖅᑎᓂᓪᓗᐊᑕᖅᓂᒃ ᐊᓚᓗᓐ ᑎᒍᒥᐊᖅᑲᓂ ᑲᑎᓴᖅᑎᑎᓂᕐᓂᒃ, ᐅᓂᒃᑲᖅᓗᓀᒃ ᑕᒋᔭᖏᓐᓇᓇᓇᓂᒃ ᐱᒋᔭᐅᓇᓂᓪᓗ. ᓱᕐᓂᓕᖅᐸᖓᓂᐊᖅᑐᖅ ᓄᓇᕗᓂ ᐃᒪᒋᐅᔨᓂ ᑲᑎᓴᖅᑎᑎᓂᒃ ᐱᐅᓂᖅᑐᑦ ᐃᓚᖓᔪᖅ 2025 ᓄᖓᒍᓪᓗᐊᓂ.

<u>ᓇᓗᓇᐃᖅᑐᖅᓯᒪᔨᓂᒃ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐱᓇᓱᒪᓕᐅᖅᑕᐅᓂᖏᓐ</u>

ᑕᑯᒋᐊᕈᑎᓪᓗᐊᑕᖅ NFR ᑲᑐᔭᔨᓄᑦ ᐊᐸᖅᑭᓯᖅᑕᐅᓂᖏᓐᓂᓄᑦ ᑕᐃᔨᐅᔨᖅ “ᓇᓗᓇᐃᖅᑐᖅᓯᒪᔨᖅ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐱᓇᓱᒪᓕᐅᖅᑕᐅᓂᖏᓐᓂ” (DRD) ᐊᐅᓪᓚᓂᖏᓐ. ᑕᑯᓇ ᐱᓪᓚᖅᑎᑎᑦᑐᓄᓂ ᑲᒪᒋᔨᐅᔨᖅ NFRWG ᓇᓗᓇᐃᖅᑐᖅᓯᒪᓕᕆᔨᓂᖅᓴᓂᒃ ᐃᓗᓪᓂᕐᓂᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦᑎᓐᓂᕐᒥᔪᑦ. ᐸᖅᓇᖅᑐᓂᖏᓐᓂ DRD ᐊᐸᖅᑭᓯᖅᑕᐅᑎᓪᓚᓇᓂᒃ ᓇᓗᓇᐃᖅᑐᖅᓯᒪᓕᕆᔨᖅᓴᓂᒃ ᐊᑐᐊᑦᓂᖅ ᐱᓪᓗᐊᑕᖅᓄᑦ NFR ᓴᓇᒪᖏᑐᓄᑦ ᐊᐸᖅᑭᓯᒪᓕᓂᓚᓂᐊᖅᑕᖏᓐᓂ, ᓱᓴᓗ ᑲᑐᔭᐅᔨᑦ ᐃᖅᑲᓗᓂᓚᓂᖅᓯᒪᔪᑦ <ᖃᓇᐅᑎᑦ (CFP), ᐊᒪᓚᓗ ᐱᐊᓂᑕᐅᓂᖏᓐᓇᑦ ᐊᑕᖅᑭᒋᐊᖅᑐᒃ ᑎᑎᕋᖅᓯᒪᓕᕆᓚᐅᖅᑐᓂᒃ ᖃᐅᔨᓴᖅᑎᓂᖅᓴᓂᒃ ᐊᑐᖅᑕᐅᔨᓂᒃ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ *(ᓄᓇᕗᒻᒥ ᐃᖃᓗᓕᕆᓂᖅᓯᒪᔪᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ, ᕿᓚᐅᖅᑕᑭᓄᑦ ᐅᓪᓗᓈᓕᐅᕙᑯᓕᒃᑕᒥ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ, ᐊᐸᓂᔪᓂᒃ ᐃᖃᓗᓕᕆᓂᖅᓯᒪᔪᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ)* ᐊᒪᓚᓗᓐ ᑐᖓᔪᓂᒃ ᑲᒪᒋᐊᖅᑐᓂᒃ ᑐᓴᖅᑎᑎᓂᕐᓂᒃ ᑐᑭᖓᓚᓐᓂᒃ ᐃᓕᓴᕐᓂᖏᓐᓂ ᑐᑭᓯᓚᐅᖅᑐᑦ NFR ᐱᓇᓱᐊᕈᑎᓴᓇᓄᑦ, ᑎᑭᐅᔪᑎᑎᓪᓗᐊᑕᖅᓄᑦ ᐊᔪᕐᓴᖅᑎᑎᓂᕐᒥᔪᑦ ᒪᓕᑦᓴᓴᓂᕐᓂᑦ ᒧᐊᖅ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐊᒪᓚᓗᓐ ᑐᖓᔪᓂᒃ ᑲᑎᒪᔨᖅ/ᒥᓂᑦ ᐃᓱᒪᓕᐅᕈᑎᔪᓂᑦ ᑐᑲᖓᔪᓂᑦ NFR ᐱᓇᓱᐊᕈᑎᓴᓇᓄᑦ, ᑎᑭᐅᑎᒍᑎᓴᓇᓄᑦ ᐊᔪᕐᓴᖅᑎᑎᓂᕐᒥᔪᑦ ᒪᓕᑦᓴᓴᓂᕐᓂᑦ ᐃᓐᓇ ᐊᑐᖅᑕᐅᔨᑦ ᐊᒪᓚᓗᓐ ᖃᓄᐃᓕᐅᕈᑎᓂᖅᓯᒪᔪᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐊᑐᐊᑦᓂᖅ NFR, ᑕᕐᓇᓂ “ᐅᑭᐅᖅᑕᖅᑐᒥ ᐊᔪᕐᓴᖅᑎᑎᓂᒃ ᐃᓚᒋᐅᔨᓕᕆᔨᓂᒃ ᑎᑎᕋᖅᓯᒪᔪᑦ” ᑕᐃᔨᓴᖅᑎᑦ (ᑕᐃᔨᐃᑦ ᒪᓕᑦᓴᓴᓂᕐᒧᑦ ᐃᓕᓴᕐᓂᖏᓐᓂ ᐊᔪᕐᓴᖅᑎᑎᓂᕐᒥᔪᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ), ᐅᕐᓇᓗᓐᓂᑦ ᐃᓕᓐᓂᐊᖅᑐᑦ DFO ᒪᓕᑦᓴᓕᓐᓂᒃ ᑲᔪᓯᑎᓴᓇᓐᓂᕐᓂᑦ ᐅᖃᐅᔨᓴᖏᓐᓂᑦᑎᓐᓂᕐᓂᑦ ᑕᑯᓇᐃᓐᓂ ᑎᑎᕋᖅᓯᒪᔪᑦ.

1. ᐊᑐᖅᑕᐅᑐᐃᓇᓗᓐᓂ ᒪᓕᑦᓴᓴᖅᑐ ᐊᒪᓚᓗᓐ ᐊᑐᖅᑕᐅᑐᐃᓇᓂᐊᖅᑕᑦ, ᐃᓱᒪᓕᐅᔨᑕᐅᑐᐃᓇᓗᓐᓂ ᐃᓂ “ᐃᓂᖅᑎᓂᐊᖅᓯᒪᓕᖅᑎᓗᒍ. ᐃᓕᓴᕐᓇᓕᑎᓴᓇᓄᑦ ᐃᓂᓴᖓᓗᓄᓂ ᒪᓕᑦᓴᓴᖅᑐᓂ ᐃᓕᒋᔭᐅᔪᓇᓂᒃ ᑐᑭᖓᓚᓐᓂᒃ ᐅᓪᓗᓈᓕᐅᕙᑯᓕᒃᑕᒥ ᑎᑎᕋᖅᓯᒪᔪᑦ ᐱᓕᕆᔨᒃᑯᑦ ᐊᑐᖅᑕᐅᔨᑦ ᐊᒪᓚᓗᓐ/ᐅᕐᓇᓗᓐᓂᑦ ᐊᐸᖅᑭᓯᖅᑕᐅᓂᖏᓐᓂᑦ ᐅᖃᐅᔨᓴᖅᑎᑎᓂᕐᓂᑦ ᐃᓕᓴᕐᓂᖏᓐᓂᑦ (ᐃᓕᓴᕐᓂᑦ ᐊᔪᕐᓴᖅᑎᓕᒪᓂᖅᑎᓪᓗᓂᑦ ᑐᓂᔭᐅᓕᖅᑕᖅᑐᓂᒃ ᑲᑎᒪᔨᖅᑐᓄᓂᒃ ᐅᒪᔪᓕᕆᔨᑦ ᐅᑎᒪᔨᖏᓐᓂᒃ ᐃᓕᓴᕐᓂᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐊᑐᖅᑕᐅᔨᑦ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ ᐊᑐᖅᑕᐅᔨᓂᒃ ᐊᑐᖅᑕᐅᔨᓂᒃ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ).
2. ᐊᑐᖅᑕᐅᓕᖅᑐᐃᓇᓗᓐᓂ ᒪᓕᑦᓴᓴᖅᑐ ᑭᓱᐊᓂ ᐅᓪᓗᒥᔭᖏᓐᓇᖅᑎᓪᓗᑕᒍ ᐊᒪᓚᓗᓐ/ᐅᕐᓇᓗᓐᓂᑦ ᐃᖃᓗᓕᕆᔨᑦ ᐅᖃᐅᔨᓴᖅᑕᖏᓐᓂᑦ ᐊᒪᓚᓗᓐ/ᐅᕐᓇᓗᓐᓂᑦ ᐃᓚᓕᖏᓐᓂᑦ (ᐃᓚᓕᖏᓐᓂᑦ ᐊᓱᔪᖅᓱᒪᓕᖅᑎᓪᓗᓂᑦ ᑐᓂᔭᐅᓂᖏᓐᓂᑦ ᑐᓴᐅᔪᓐᓇᖅᑐᑦ ᑲᑎᒪᔨᖅᑐᓄᓂᒃ ᐅᒪᔪᓕᕆᔨᑦ ᐅᑎᒪᔨᖏᓐᓂᒃ ᐃᓕᓴᕐᓂᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐊᑐᖅᑕᐅᔨᑦ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ ᐱᖅᑕᐅᔪᑎᖓᓐᓂ ᐊᑐᖅᑕᐅᔨᖅ ᒪᓕᑦᓴᓴᖅᑐᖅ.
3. ᐊᑐᖅᑕᐅᑎᑕᐅᓇᓗᓐᓂ ᒪᓕᑦᓴᓴᖅᑐᑦ ᐅᓪᓗᓈᓕᐅᕙᑯᓕᒃᑕᒥ ᒪᓕᑦᓴᓴᖅᑐᓂᒃ ᐱᖅᑕᐅᔪᑎᖓᓐᓂᒃ ᐊᑐᖅᑕᐅᔨᖅ ᒪᓕᑦᓴᓴᖅᑐᖅ.

ᓂᓚᐅᑦᓴᐅᔨᖅ DRD ᐱᐊᓂᕐᓯᒪᓪᓚᕆᔨᖓᓐᓂ ᐃᓕᕐ 2026.

ᓱᕐᓂᕆᑎᓕᕐᓂᖅ ᑲᑐᔭᐅᔨᒥᒃ ᐃᖃᓗᓐᓂᑕᓂᕐᓗᑦ ᕿᓇᐅᑎᑎᐅᕐᑦ ᑐᓂᔭᐅᓂᒃ

ᑕᑯᓇ CFP ᓄᑕᖅ ᐃᖅᑲᓇᐃᔨᐅᑎ NFRWG ᐃᓄᐃᓂᒃ ᐊᑐᓵᔨᑦ ᐊᐅᓪᓚᕆᔨᓂᒃ ᓇᖕᒥᓂᖅ ᐃᖃᓗᓐᓂᑕᓂᕐᓗᑦ. ᑐᓵᔭᐅᑕᐅᔨᖅ ᐊᓱᐊᓄᓐᓂᕐᓗᑦ ᐃᖃᓗᓐᓂᓱᓄᑦ ᕆᐃᔨᓄᑦ, ᐱᓕᕆᔨᑦᑐᓄᓂ ᐃᖃᓗᓐᓂᓱᓄᑦ. HTO ᐊᒪᓚᓗᓐ RWO ᐊᑐᓂᖅᑎᐅᔨᖅᑕᖏᓐᓄ CFP ᐊᐅᓪᓚᕆᔨᕐᓂᖅ ᐃᓄᐃᑦ “ᐊᖏᕐᓇᖅᑐᒥᒃ ᓂᐅᔭᕐᑎᓂᖅᑎᓪᓗᒃ” ᐃᖃᓗᓐᓂᓱᓄᑦ ᐃᓄᐃᑦ ᐱᓯᓂᐅᓇᓂᖅᑎᑎᓂᒃ ᐱᐅᓂᖅᑐᑦ ᐃᖃᓗᓐᓂᓱᓄᑦ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ ᐊᑐᖅᑕᐅᔨᑦ ᐊᐅᓪᓚᕆᔨᕐᓂᑎᑦ ᐊᑐᖅᑕᐅᔨᓂᒃ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ ᐊᑐᖅᑕᐅᔨᓂᒃ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ ᐃᓕᓴᕐᓂᖏᓐᓂᒃ ᐃᓕᓴᕐᓂᖏᓐ

ᐊᓚᓗ ᐁᖅᑭᓯᓂᒪᓗᑎᒃ ᐅᒪᔨᖅᑕᐅᔨᓐᓇᖅᑐᑦ ᑭᖅᑕᖅᑭᓐᓂᒃ ᓄᑦᓄᑦ ᐃᖏᓗᑕᓚᐅᕐᓄᑦ ᐅᕐᓀᔅᓂᓕᑦ ᐊᖏᑕᑎᐊᖅᑐᑦ ᐊᑐᖅᑕᐅᔨᑦ ᐃᖏᓗᑕᓚᐅᕐᓂ, ᑐᓂᔭᐅᑦ NWMB-ᑯᓐᓄᓚᐊᖅᑎᔭᖅᑐᖅ ᓂᕆᒧᐊᖅᑎᑕᐅᖅᑯᓐᓇᒍ ᐅᐃᒍ V ᓄᓇᕋᐊᒥ ᐃᖏᓗᑕᓚᐅᖅᓯᒪᑦ ᐊᐅᓪᓚᕋᕐᓯᒪᑦ NFR-ᓄᑦ, ᐊᓂᑕᒪᖅᑭᓐ ᓄᑦᓗᐃᑦ ᐊᖏᑕᖅᑕᕈᐊᑕᖅᑭᓐᓄᑦ ᒪᑕᑐᓴᖅᑭᑦ. ᐅᐃᒍ V-ᒥ ᑯᑕᑲᐃᑦ ᐊᓚᓗ ᐃᖏᓗᑐᓂᕆᘐᓴᐅᑎᑦᓗᒍ ᐱᑕᕐᑎᑦᑭᓐᓂᖅᑐᖅ ᐃᖏᓗᑐᓂᕆᘐᓴᖅᑭᓐᓄᑦ ᑐᓂᔭᐅᑎᒃ ᐃᓂᒪᑕᐅᖅᑐᓂᓗ ᐅᓪᓗᕆᐊᒃᑲᓐᓂᖅᑐᒃ ᓂᕆᔨᖅᑎ DFO-ᑯᓄᓂᑦ ᐅᖅᑲᐅᔭᕆᐊᕈᑎᓂᒃ ᐃᓂᒪᑕᐅᕈᑎᓂᓕᓗ ᐲᓇᐅᔨᑕᐅᕋᕆᐊᖅᓂᖅᓗᑦ ᐃᖏᓗᑐᓂᓄᓕᖅᑎᒃ, NFR ᑕᐃᒪᐃᑕᖅᓴᐃᓇᐅᑎᓯᓗᑦ ᓄᓗᐊᑦ ᓄᑦᓗᖅᑕᑦ ᐊᖏᑕᓂᖅᑕᑦ ᒪᓇᓇᓗ ᐊᑐᖅᑕᐅᔨᖅᓴᖅ.

ᐊᐅᓚᑦᓂᐅᖅᑲᐅᔨᒥ, DFO, NTI, ᐊᓚᓗ <ᓇᓂᖅᑐᖅ HTO ᑲᑐᔨᓗᑎᒃ ᐅᑕᑐᕈᐅᖅᑐᑦ CFP ᐃᓕᖏᓄᓂᑦ ᓄᓇᕆᓇᓂ ᐃᖏᓗᔪᐱᑦᓂᓄᓂᖅᑎᒃ. CFP ᐁᖅᑭᓂᖅᑕᐅᓕᐅᖅᑐᖅ ᐊᓚᓗ DFO ᑲᑎᖅᑎᖅᑕᐅᖅᑐᖅ ᒪᓇᓗ ᒪᕆᑕᓚᖅᑭᐊᕆᓇᓂᑦ ᐊᖏᖅᑕᐅᖅᓯᔨᓗᒍ ᐃᖏᓗᑕᓚᓂᓂᖅ ᑲᑐᔪᐅᔨᖅ ᐃᖏᓗᑕᓚᓂᓂᖅᓯᒪᔪᑦ ᓕᐃᓴ. ᐱᔪᒋᑦ ᐅᑕᑐᕈᑎᐅᓴ ᐃᓕᓴᐅᓂᕐᒪᑦᓗᑎᒃ, ᐊᑐᑦᓕᒋᑦᓗᒍ ᐅᑕᑐᕈᑦ, ᑐᕽᒪᖏᓐ ᐱᓕᓚᐊᖏᖅᑕᐅᔨᔪᑦ, ᐊᑭᑎᓇᖅᑕᖅᑭᖏᑦ, ᐊᓚᓗ ᐊᓂᖏᑦ HTO/ᐊᖏᓇᓂᑎᑦ ᐊᑐᑕᐊᓕᓴᐅᖅᑐᖅ, ᐊᓚᓗ ᓇᓗᓇᖅᑐᐃᐅᓂᔪᑦ ᐊᑦᓂᕐᖅᑐᖅᓯᒪᑦ ᓴᖅᑭᕐᖅᑐᖅᓯᒪᖅ ᑐᖏᓕᒥ NFRWG ᐃᓂᖅᑕᐅᖅᑭᓂᖅᓴᖅᑭᖅᓂ CFP ᐊᑐᐊᑦ. ᓂᕆᔨᖅᑎ ᐅᑕᑐᕈᐅᑎ ᐃᓂᒪᔨᖅᓂᕆᐅᕈᐅᔪᓇᖅᑐᖅ ᐊᓂᖅᑭᓂᖅ ᐊᖏᖅᑕᐅᖅᓯᒪᑦ ᓄᓇᕆᓇᓂ ᐊᓚᓗ ᐊᓂᖅᑭᓂᖅᓄᑦ ᐅᒪᔨᓄᑦ. ᑐᑦᓄᓕᖅ ᑲᔪᓂᔪᓇᖅᓂᐅᑉᓗᒍ ᐊᓂᖅᑭᓂᖅᓄᑦ ᐊᑐᖅᑕᐅᖅᑐᖅ ᐃᓕᓴᐅᓂᖅᑐ

ᑐᓂᓴᐅᔨᖅ
ᓄᓇᕗᒻᒥ ᐅᒪᔨᓪᓚᕆᔨᐊᓕᒃᑯᓂ ᑲᑎᒪᔨᖏᕐᓄᓂ
ᔪᑎᐱᓚ 2025

ᒪᓕᓴᖅ

ᑐᑭᓯᐊᕐᑭᑎ: X ᐃᒪᓐᓇᖅᑐᑎᓂ: ᐃᓯᒪᓪᓕᕐᑭᑎ:

ᐱᓇᐃᓗᑦᖅ: ᐃᒪᖏᑕᓪᓚᕆᐅᒃᑯᓂ ᑲᓇᑕᒥ (DFO) – ᐃᖅᑲᓗᓪᓚᑐᓂᖏᓐᓂ ᐊᐅᓪᓗᑎᓂᖏᓐᓂ (FM) ᐊᐅᓪᓚᐅᖏᓐᓄᓂ ᑐᑭᓯᐊᕐᑭᑎᓂ

ᓴᐃᓇᓵᓂ:

1. ᑐᐃᓂᓪᓂ

- ᑲᑎᕐᑐᑎᓂ ᑐᓴᖅᑎᕐᑎᒍᑕᐅᓯᒪᔨᓂ ᓄᓇᖅᑕᐅᔨᓂ ᑐᐃᓂᓪᓄᓂ ᐊᐅᓪᓚᕆᐊᑎᕐᔨᐅᔨᓕᒥᓂ ᐅᕐᓂ 2024/2025 ᑐᐃᓂᓪᓯᐊᓂᖏᓐᓄᑦ: ᐅᐸᓴ ᐃᒪᓐᓗ 45, ᔪᒥᑦ ᐃᒪᓐᓗ 0, ᐅᐊᓄᓇᓐᓂᓂ ᑕᓯᐅᖅᑳᐊᖅ 130, ᑯᐃᓇᖏᐅᑦ ᖃᑭᖅᑕᓐᓗ 81, ᑲᓇᓄᓇᓐᓂᓂ ᖃᑭᖅᑕᒃᓚᓱᒃ 148, ᓴᓄᓂᑐᑎᐅᑦ ᑐᓇᓂᑐᓯᐅᓚᓄ ᐃᒪᓐᓗ 195, ᐊᓗᒪ ᖃᐅᖏᓇᖅ 123.
- 2025-ᒥ ᑲᓇᑕᒥ ᖃᐅᐅᓴᖏᓪᓗᒍ ᐅᖃᐅᓪᓴᐅᓕᐊᖅᑲᑦ ᐊᓕᓚᓐᓇᖅᓗ ᐱᓕᕆᓇᖅᑐᑦᑕᒍ ᖃᑎᖏᑕᓇᓕᓴᓂᑐᓂ ᑲᑎᒪᓂᖅᑐᓂ ᓄᓇᕗᑦᑕᐅᓚᐊᖅᑲᑦ ᐊᒥᓱᓂᖏᓂᑕ ᓇᓕᓪᓕᓂᖏᓂ ᐊᓚᓗᓐ ᒪᑭᒪᔨᓴᕆᓇᖅᑐᓂ ᐊᒥᓱᓂᖏᓂ ᐊᖓᔪᖅᐅᔨᓴᖅᑐᓂ ᐅᖃᐅᓪᓴᐅᓕᐊᖅᑐᑎ ᖃᑎᖏᑕᓇᓕᓴᓂᑐᒃ ᑐᐃᓂᓪᓂ ᐊᒥᓱᓂᖏᓐᓄᑦ. ᓄᓇᕗᒻᒥ ᑐᐃᓂᓪᓂ ᐱᓪᓗᒍ ᐱᓕᕆᓂᕐᓂᑦ ᐱᓕᕆᓇᖅᑐᑦᑕᒍ ᖃᑎᖏᑕᓇᓕᓴᓂᑐᓂ ᐊᓚᓗᓐ ᓄᓇᕗᑦᑕᐅᓚᐊᖅᑲᑦ ᐊᑐᓕᕆᓂᖅᓴᓂᒃ ᐃᖅᑲᓗᓪᓚᑐᓂᖏᓐᓂ ᐊᐅᓪᓚᕆᐊᑎᕐᔨᐅᔨᓕᒥᓂ *ᐸᖅᓴᐅᑎᖓᓂᑐᒃ* (IFMP) *ᑐᐃᓂᓪᓂ ᐱᓪᓗᒍ ᓄᓇᕗᒻᒥ ᓄᓇᕕᑎᑕᐅᔨᓚᐅᖅ ᐊᖏᕐᕉᑎᖓᓂᑐ.* ᐃᔨᓪᓕᕐᓚᕐᓇᖅᑎᑦᓗᒍ ᒪᓕᕐᓂ 2025-ᒥ, ᓄᓇᕗᒻᒥ ᑐᐃᓂᓪᓂ ᐱᓪᓗᒍ ᐊᓚᓗᓐ ᐊᐅᓪᓚᕆᐊᑎᕐᔨᐅᔨᓂ ᑲᑎᒪᓂᖅᑐᓂ ᑲᑎᒪᓂᐅᕐᖏᓚᐅᖅ. ᐃᖅᑲᓗᓪᓚᑐᓂᖏᓐᓂ ᐊᒥᓱᓗᓐ ᐃᒪᓕᕐᓂᑕᐅᓪᓚᑐᓂᖏᓐᓂ ᑲᑎᒪᓂᖅᑐᓂ ᖃᐅᔨᓴᖅᑎᑦᓗᒍᑦ ᐃᓕᓴᕐᓂᖅᑕᐅᓂᖏᓐᓂᓕᓂᒃᑐᓂᖏᓐᓄᓂᒃᑐᓂ ᐊᒥᓱᓂᖏᓐᓄᓂ ᐊᓚᓗᓐ ᐃᒪᓕᕐᓂᑕᐅᓪᓚᑐᓂᖏᓐᓂ ᑲᑎᒪᓂᖅᑐᓂ ᖃᐅᔨᓴᖅᑎᑦᓗᒍᑦ ᐃᓕᓴᕐᓂᖅᑕᐅᓂᖏᓐᓂᓕᓂᒃᑐᓂᖏᓐᓄᓂᒃᑐᓂ ᐊᒥᓱᓂᖏᓐᓄᓂ ᐊᓚᓗᓐ ᐃᒪᓕᕐᓂᑕᐅᓪᓚᑐᓂᖏᓐᓂ ᑲᑎᒪᓂᖅᑐᓂ ᖃᐅᔨᓴᖅᑎᑦᓗᒍᑦ

3. ᖃᓄᓗᒍᑦ

- ᐸᖅᓂᖅᑐᑦ ᑲᖏᖅᑐᐊᓗᐊᕐᓂᐅᓂ ᖃᓄᓗᒍᑦ (CSB) ᐱᓕᓗᑎᑦ ᐱᓪᓗᒃᓴᑦ ᑲᑎᒪᔨᕆᓚᖏᑦ ᑲᑎᒪᖅᑎᑦᑕᐃᖅᓯᖅᑐᑦ ᓴᒪᓕᐊᓚᐅᖅᑐᑦ ᒪᑭᒪᔨᓇᖅᑐᓕᕆᔨ ᐊᒥᓱᓂᖏᑦ ᖃᑎᖅᑐᓇᐅᔭᖅᐳᖅᑐᓂᑦ ᐊᐅᓚᕋᓂᖅᓴᖅ ᖃᓄᓗᒍᓕᑦ ᖃᒥᖅᖃᓇᓕᖅᑐᑦ ᐅᐊᕐᓇᕋᑎᑦ ᑎᑎᕋᕐᓂᑎᑦ ᐊᐅᓚᕋᓂᖅᑐᑦ, ᓇᓗᓇᐃᖅᓯᓗᑎᑦ ᐅᑦᓗᒥᐅᔨᖅ ᖃᐅᔨᓴᐅᑎᑦᑐᓴᒍᑦ, ᐊᒪᓚᓗ ᐊᑐᓕᖅᑎᑦᑕᐅᓂᖓᓄᑦ ᐱᐅᓪᓗᐊᓚᔨᑦ ᐃᓄᐃᑦ ᖃᐅᔨᓴᖅᑐᖓ (IQ) ᐊᒪᓚᓗ ᖃᐅᔨᒪᓂᖅᓯᓚᑦ ᖃᐅᔨᓴᐅᓕᖅᑐᐊᓗᐊᕐᓂᐅᑎᑦ ᖃᐅᔨᓴᓂᖅᓯᒪᔪᑦ.
- ᓱᓂ 19, 2025-ᒥ ᐃᓐᓇᐃᓚᒋᔭᐅᓕᖅᑐᖅ ᑲᑎᒪᓂᖏᑦ ᐸᖅᓂᖅᑐᑦ ᖃᓄᓗᒍᑦ ᐱᓪᓗᑎᑦ ᐱᓪᓗᒃᓴᑦ ᑲᑎᒪᔨᕆᓚᖏᑦ. ᑭᒧᑦᓪᓕᖅᑐᖅ ᐸᖅᓂᖅᑐᑦ ᖃᓄᓗᒍᑦ ᐱᓪᓗᑎᑦ ᐱᓪᓗᒃᓴᑦ ᑲᑎᒪᔨᕆᓚᖏᑦ ᑲᑎᒪᓕᕆᓂᖏᑦ ᐸᖅᓂᖅᑐᑦᑎᒋᕆᓕᖅᑐᑦ, ᓄᓇᕗᑦ ᐊᒪᓚᓗ ᑭᖑᓂᐊᓕᖅᑐᑦ ᐃᓄᐃᑦ ᐊᓪᓚᖅᓯᓂᖅᑐᑦ ᐃᓕᓴᕐᓂᖅᑐᑦ ᐃᓕᑦᑐᖅᑕᐅᓪᓗᓂᖏᓪᓗᓂᒃ ᓄᓇᕗᒻᒥᐅᑦ ᐅᓪᓗᒥᐅᓕᕆᓚᖏᑦ ᑲᑎᒪᓕᕆᓂᖏᑦ ᐸᖅᓂᖅᑐᑦ ᖃᓄᓗᒍᑦ ᐱᓪᓗᑎᑦ ᐱᓪᓗᒃᓴᑦ ᑲᑎᒪᔨᕆᓚᖏᑦ ᐊᒪᓚᓗ ᓄᓇᕗᑦᑎᐊᕐᓂᐅᑎᑦ ᑲᑎᒪᓕᕆᓂᖏᑦᓯᒪᔪᑦ.
- ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐊᒪᓚᓗ ᐃᓚᒋᑎᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ ᓂᕆᐅᑕᒧᑦ ᐱᓪᓗᒃᑲᒪᒋᔭᐅᓕᖅᑐᖓᓄᑦᓯᒪᓂᖅᑐᑦ ᐊᐅᓚᕋᓂᖅᑐᑦᑎᒋᕆᓕᖅᑐᑦ ᖃᐅᔨᓴᖅᑐᑦ ᐃᓕᓴᕐᓂᖅᑐᑦ ᐊᒪᓚᓗ ᓄᓇᕗᒻᒥᐅᑦ ᑲᑎᒪᓕᕆᓂᖏᑦ ᐊᒪᓚᓗ ᓄᓇᕗᑦᑎᐊᕐᓂᐅᑎᑦ ᑲᑎᒪᓕᕆᓂᖏᑦᓯᒪᔪᑦ.
- ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐊᒪᓚᓗ ᐃᓚᒋᑎᑕᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᓂᖏᑦ ᓂᕆᐅᑕᒧᑦ ᐱᓪᓗᒃᑲᒪᒋᔭᐅᓕᖅᑐᖓᓄᑦᓯᒪᓂᖅᑐᑦ ᐊᐅᓚᕋᓂᖅᑐᑦᑎᒋᕆᓕᖅᑐᑦ ᐃᓕᓴᕐᓂᖅᑐᑦ ᐊᒪᓚᓗ ᓄᓇᕗᒻᒥᐅᑦ ᑲᑎᒪᓕᕆᓂᖏᑦ ᐊᒪᓚᓗ ᓄᓇᕗᑦᑎᐊᕐᓂᐅᑎᑦ ᑲᑎᒪᓕᕆᓂᖏᑦᓯᒪᔪᑦ.
- ᐃᖃᓗᓕᕆᔨᒃᑯᑦ ᐊᒪᓚᓗ ᐃᓚᒋᑎᑕᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᓂᖏᑦ ᐱᓪᓗᒃᑲᒪᒋᔭᐅᓕᖅᑐᖓᓄᑦᓯᒪᓂᖅᑐᑦ ᐊᐅᓚᕋᓂᖅᑐᑦᑎᒋᕆᓕᖅᑐᑦ ᐃᓕᓴᕐᓂᖅᑐᑦ ᐊᒪᓚᓗ ᓄᓇᕗᒻᒥᐅᑦ ᑲᑎᒪᓕᕆᓂᖏᑦ ᐊᒪᓚᓗ ᓄᓇᕗᑦᑎᐊᕐᓂᐅᑎᑦ ᑲᑎᒪᓕᕆᓂᖏᑦᓯᒪᔪᑦ.

4. ᐊᖏᕕᑦ

a. 2024-ᒥ ᖃᓄᓗᒍᑦᓯᒪᓂᖅᑐᑦ:

- ᑲᑎᓕᒻᒥᖅᓯᑦ ᐊᖓᔪᐅᔨᓴᖅᑐᑦ (TAH) ᑲᓇᑐᓇᓂᓂ ᑲᓇᑕ–ᐱᖓᑐᓇᖓᓂ ᑲᓱᓐᓂ ᓄᓇᕗᑦ (ECWG) ᐊᖏᔪᓂᒃᑯᑦᓯᒪᓕᖅᑐᑦ ᓄᓇᕗᑦ ᐃᒪᓕᖅᓂ 5–ᖓᕗᑦ ᐊᖏᕐᕌᒍᑦ. ᑲᑎᓕᒻᒥᖅᓯᑦ ᐊᖓᔪᐅᔨᓴᖅᑐᑦ ᑲᑐᔪᐅᕗᑦ: 2 ᐊᖏᕐᕌᒍᑦ ᐊᑐᓂ ᖃᐅᔨᓴᓇᖅᑎᑦ ᐊᒪᓚᓗ

ᑭᓲᓕᖏᑎᓗ, ᐊᓚᒪᓗ 1 ᐊᑕᐅᓯᖅ ᐊᖏᕐᕌᒍ ᖃᑎᖏᑕᐅᓂ. ᑭᓯᐊᓂᓕ, 2025-ᖁᒧᑦᓗᒍ, ᐊᑕᐅᓯᑐᐊᔪᓗᒃ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅ ᓄᓇᕗᑦ ᐃᒪᖅᑎᖅ.

- ᑕᒪᒃᑭᒃ ᑭᓲᓕᖏᑎ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ (KWB) ᐊᓚᒪᓗ ᖃᑎᖏᑕᐅᑦ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ (KRWB) ᑐᓴᖅᑎᕌᓯᒪᓕᖅᑐᑦ ᐃᖅᑲᓇᕐᓂᓱᐊᔪᓂ ᐊᓚᒪᓗ ᐃᒪᖏᑕᐅᓇᕐᓂᓱᐊᔪᓂ ᑲᓇᑕᒥ ᐱᓇᐅᖏᓐᓂᖏᑦᑐᓂᒃ ᓇᓕᐊᓂᐅᖏᑐᐃᓇᖅ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ ᑐᓴᖅᑎᑲᓂᒃᑯᓂᖅ ᓴᐳᑎᖅᑕᐅᑦᑎᓇᓂ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ ᐃᓕᓴᕐᓂᐅᑉᐳᖅ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ. ᑕᐃᒪᐃᓕᒫᑦ, ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅᑕᐃᓇᖅᑐᒥᓂᕐᓂᖅ ᐃᓕᓴᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ 2025-ᖁᒧᑦᓗᒍ.
- 2025-ᒥ, ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅ ᐸᖅᑐᑎ ᐋᖅᑭᒃᖢᖅᑕᐅᓯᒪᔪᖅ ᐸᒃᓂᖅᑐᑦᑎᒥ ᐊᒃᑕᖅᑕᐅᓕᐅᖅᑕᐅᓯᒪᔪᖅ ᖃᐅᖅᑕᒋᐊᒃ ᐅᒪᓕᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᒃᓄᑦ (QWB). ᓄᓇᕗᑦ ᑐᙵᓚᓕᒃ ᑎᒥᖅᓚ (NTI) ᓇᓗᓇᐃᖅᑲᓇᐅᖅᑎᑦᓯᒪᓂ ᐊᖏᕐᕋᐃᓂᐊᕐᓵᑦ ᖃᐅᔨᓴᕐᓂᑦ ᐊᑐᖅᓴᓂᑦ ᐱᓕᒪᓐᓇᐃᓂᕐᒥᒃ ᐱᓵᓱᖅᑕᐃᓇᐃᓂᐅᑎᓂᒃ, ᐃᓕᓴᕐᓂᓱᐊᔪᓂ ᐊᓚᒪᓗ ᐃᒪᖏᑕᐅᓇᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ ᖃᐅᔨᓴᕐᓂᐅᑉᐳᖅ ᐃᓕᓴᕐᓂᐅᑉᐳᖅᑐᑦ ᐃᓄᐃᑦ ᐃᓕᓴᕐᓂᓱᐊᔪᓂ ᐅᐱᒍᓐᓇᖅᑐᑦ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅᑐᑦ ᐅᐱᒍᓐᓇᖅᑐᑦ ᐃᓕᓴᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ.
- ᐋᓚ 26, 2025-ᖁᒧᑦᓗᒍ, ᐸᒃᓂᖅᑐᑦᑎᒥ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅᑎᑦ ᐊᖏᕐᕋᓚᐅᖅᑐᑦ 45–ᐃᓱᒪᑦ ᑕᑭᓂᓚᓂᖅ (13.7 ᑎᑕ) ᓇᓵᐊᒥᒃ ᖃᓇᒃ ᐊᖏᕐᕋᖅᑎᒃ ᖃᑭᑭᖅᑕᓂ. ᐱᓕᒃᑐᓕᐅᖅᑐᑦ ᓭᕐᕋᐃᓴᐃᓕᓴᕐᓂᓗ ᑎᓂᓚᕐᓂᑦ ᐋᓚ 26 ᑎᑭᕐᑐᒍ ᐋᓚ 29. ᑕᒃᑕ ᐱᐅᖏ–ᐃᓚᓵ ᑕᐅᓴᑦ ᐊᓚᒪᓗ ᑕᒃᑕ ᑐᐊᕐᓄ ᑭᒃ (ᑲᓇᑕᒥ ᐅᒪᓕᒃ ᐊᒃᑕᖅᑕᐃᓇᓕᓂᖏᑦᑐᓂ ᑯᐊᐸᑯᓂᖅᓯᒪᓂᖅ) ᐃᑲᔪᓕᐅᖅᑐᑦ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅᑎᓂᒃ ᓄᐊᕐᑎᓂᑦᑐᓂ ᑎᒥᖅᓚᓂ ᑕᑭᓂᖅᑕᒥ ᐊᓚᒪᓗ ᐃᓪᓚᖅᓯᐅᓕᐅᖅᑐᑦ ᐃᓕᓴᕐᓂᓱᐊᔪᓂ ᖃᐅᔨᓴᕐᓂᐅᑉᐳᖅᑎᓂᒃ ᑕᒪᒃᑭᒃᖢᖅᑐᒥ ᐊᑯᓇᒥ 3-ᓂᒃ ᐃᓱᒪᓂ ᑕᑭᓂᓕᒃ ᑕᑭᓂᓕᓴᒃ (94 ᓱᐊᓐᕈᑎᑦᑕ) ᓇᓵᐊᖏᑦᑐᒃ.
- ᐃᓕᓴᕐᓂᓱᐊᔪᓂ ᐊᓚᒪᓗ ᐃᒪᖏᑕᐅᓇᕐᓂᓱᐊᔪᓂ ᑲᑎᒪᔨᖅᑎᑦ ᖃᔪᓐᓇᐅᑉᐸᒪᓕᕐᓂ ᐸᒃᓂᖅᑐᑦᑎᒥ ᐊᖏᕐᕋᐃᓂᐅᑉᐳᖅᑎᓂᒃ ᐊᐖᓴᖏᓂᕐᑎᓂᒃ ᑐᓂᓯᓚᐅᑉᐸᒪᓂᒃ ᑕᑭᓂᖅᑕᒥ ᐊᑐᐊᓕᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᕐᓂᐅᑉᐳᖅᑎᓂᒃ ᑲᓇᑕᒥ ᐃᓚᖏᑕᐅᓇᕐᓂᓱᐊᔪᓂ ᖃᐅᔨᓴᕐᓂᐅᑉᐳᖅᑎᓂᒃ ᑐᓂᓯᓚᐅᑉᐸᒪ

ᑲᓇᑕᑦ ᑲᓇᖅᓇᖅᑐ–ᑲᓪᓚᓐᓂᑦ ᓄᓇᑦ ᐱᖅᑭᖅᓇᐊᖅᑎᓂ ᐊᖏᓴᑦ ᐊᒥᓯᓂᖅᓂᑦ ᖃᑲᐅᓴᖏᓂᖏᖅᑐᖅᑐᑦ ᑭᓴᓐᖃᖅᑎᑕᐅᔪᑦ ᓇᓗᓇᐃᕈᑎᖅᓂᑦ ᑎᑎᕋᖅᑲᓯᒪᔪᓄᑦ.

- ᑲᓇᑕᒥ ᖃᑲᐅᓴᖏᓂᒧᑦ ᐅᖃᑲᐅᓴᕈᑎᖅᑳᑦ ᐊᑦᓚᓯᐃᖅ ᐊᑐᓂᖏᑲᓂᑦ ᐊᓚᒪᓗ ᐃᓯᒪᓕᐅᖅᓯᒪᓂᖅᓂᑦ ᐊᑐᐃᖅᓇᐅᒪᓂᖅᑐᑦ ᐃᓄᖅᑐᑐᓕᖅᓯᒪᔪᑦ ᐱᓴᖅᑕᐅᓯᒪᓕᖅᐸᑕ, ᐊᓚᒪᓗ ᐅᓂᖅᑲᐅᑕᐅᓗᑎᑦ ᖃᑭᐅᕐᕕᖅᑕᐅᕐᓴᖅᑕᒥ ᐊᑐᐃᖅᓇᐅᒪᓂᖅᑐᑦ ᐃᓕᓴ

ᖃᐅᔨᓴᐃᔨᖅᑎᑦ ᑎᒥᖅᑐ, ᑭᐸᑯᑐᐃᓇᐃᑦ ᐊᐅᓂᐊᖅᑕᐃᓕᓚᓂᕐᒧᑦ ᑎᒥᖅᑐ ᑲᓇᑕᒥ, ᐊᓪᒪᓗ ᐊᐊᑎᑐ ᐊᓪᒪᓗ ᓯᓚᑦ ᐊᓯᖏᑦᑎᔨᓂᖅᑐᑦ ᑲᓇᑕᒥ – ᑲᓇᑕᒥ ᐅᒪᔪᓚᓗᓐᓂᑯᑦ ᐱᔨᑎᖅᑎᖅᑎᑦ.

- ᓄᓇᕗᓪᒥ, ᑭᐸᑯᑐᐃᓇᐃᑦ ᑐᓴᐅᒪᑎᑕᐅᓚᐅᖅᓯᒪᔪᑦ ᑎᖅᒥᐊᑦ ᐊᐢᓂᐊᓚᖅᓂᖅᑎᑦᑕ ᖃᐅᔨᓴᖏᓐᓇᔪᐊᑎᖅᑎᑦ ᑲᒪᑎᕐᕋᐅᖅᑐᐊᖅᑕᒪᑦ ᓄᓇᕗᑦ ᓴᖅᒪᖅᑎᖅᑎᑦ (ᐊᐅᑎᓚᓗᓐᓂᑯᑦ ᐊᓪᒪᓗ ᐊᐢᓂᐊᓚᖅᑲᖏᖅᑕᑐᓚᓗᓐᓂᑯᑦ) ᐊᓪᒪᓗ ᑲᓇᑕᒥ ᓂᔪᐃᑦ ᐊᐢᓂᐊᖅᑕᐃᓚᓕᓂᖅᑎᖅᓄᑦ ᑯᐊᐸᑯᐢᓄᑦ.
- ᒪᐃ 2025-ᖓᒍᓐᓄᑐ, ᐃᖃᓄᑕᓚᓗᓐᓂᑯᑦ ᐊᓪᒪᓗ ᐃᒪᖏᐅᑕᓚᓗᓐᓂᑯᑦ ᑲᓇᑕᒥ ᐱᓴᖃᖅᓯᒪᔪᑦ ᐱᑕᓚᖏᐊᔪ ᐃᓗᐊᓂ ᑐᓴᑯᓯᓚᐅᖅᓴᓚᐊᑕᓐᓂᖅ ᐃᑲᔪᖃᑕᐅᔪᓐᓇᖅᑐᑦ ᑕᓐᑲᐅᑦᑎᐊᓂᖅ ᓄᓇᕗᑦ ᓴᖅᒪᓕᖅᑎᑦ ᖃᓄᐃᓕᑕᐅᕐᓂᖅᑎᑦ ᐅᓂᖅᑲᐅᑕᐅᔪᓐᓇᖅᑐᑦ ᐊᐢᓂᐊᖅᑕᐃᓚᓕᓂᖅᑎᖅᓄᑦ/ᑐᓴᐃᖅᓴᔪᑦ >ᐃᐸᑦ. ᑕᒪᓐᓇ ᐃᓕᖅᑯᓯᓚᖅᑲᓚᐊᑦᑐᖅ ᑐᓴᐅᒪᑎᑎᑐᓚᓗᓐᓂᑯᑦ ᐊᐢᓂᐊᖅᑕᐃᓚᓕᓂᖅᑎᖅᓄᑦ ᐃᒪᐃᖅᑕᐅᓯᖅᑐᐊᖅᑕᐅᓯᓯᓚᐃᔨᓂᖅᓄᑦ ᓄᓇᕗᑦ ᓴᖅᒪᓕᖅᑎᖅᓄᑦ ᐃᓕᐅᑎᑕᐅᓪᓗᓂᑦ ᑎᑎᕐᕋᐅᖅᑎᑎᑕᐅᓚᓂᖅᓄᑦ ᒪᓕᑕᑕᐅᓪᓗᓂᑦ ᑲᑎᑕᐅᑦᑎᓂᑦ ᓄᓇᕗᑦ ᑎᒥᖅᑐ ᐊᐢᓂᐊᖅᑐᑦ ᐊᐃᑕᑐᑕᒪᑦ ᓇᐅᑦᓯᐊᖅᓯᒪᔪᑦ ᐃᓚᖅᑎᑕᐅᓯᒪᔪᑦ.

ᑲᓇᑕᒥ ᓄᓇᕗᑦ ᓇᑦᓴᓂᖅ ᓇᒃᑎᖅᓂᖅ (ᖃᓄᑕᖅᑐᓂᖅ):

1. **ᑎᓄᖅᑲᖏᖅᓱᒥ ᖃᓄᑕᖅᑕᓚᓗᓐᓂᑦᑐᑦ ᐊᐅᓪᓚᕆᑎᐅᓂᖅ (CSTMA):**

- 2025-ᒥ ᐅᑭᐅᖅᒥ ᖃᓄᑕᖅᑐᓂᐅᓪᓚᓪᒪᓕᖅᓯᒪᔪᑦ <ᖏᓂᖅᑐᒃ ᑲᖏᖅᑐᐊᓚᓪᓚᐊᓂᖅ ᐃᐦᖏᐊᓚᓐᓂᒥ ᐃᓱᒋᓪᓗᒥ, ᑲᑎᑕᐃᒪᖅᓴᓂᖅ 219 ᑕᐢᓴ ᐅᖅᑯᒪᐃᓂᓪᒃᑐᑦ ᖃᓄᑕᖅᑕᓄᕆᐊᖅᓂᓪᓚᑎᑦ (ᐅᖅᑯᒪᐃᓂᓪᒃᖏᓐᓄᑦ) ᓄᓇᕗᕆᐊᖅᓂᓪᓚᑎᑦ <ᖏᓂᖅᑐᒃ ᐃᖃᓄᑕᓚᓐᓇᖅᓱᓚᓄᑦ.
- ᐊᐠᑎᓯ 2025-ᒥ, ᐃᖃᓄᑕᓚᓗᓐᓂᑯᑦ ᐊᓪᒪᓗ ᐃᒪᖏᐅᑕᓚᓗᓐᓂᑯᑦ ᑲᓇᑕᒥ ᐱᓇᓯᐊᑭᑎᒥᒃ ᐱᑕᐅᖅᓯᒪᔪᑦ ᓯᑯᖅᑲᖏᕐᑎᓪᓗᒍ ᐃᒪᖏᒥ ᐊᐅᓴᑯᑯᓂ ᐃᖃᓄᑕᓚᐱᓯᐊᑭᑎᒥᒃ <ᖏᓂᖅᑐᒃ ᑲᖏᖅᑐᐊᓚᓪᓚᐊᓂ ᐊᓪᒪᓗ ᐅᑕᖅᑭᔪᑦ ᐅᒥᐊᖅᓯᐊᖅ ᖃᐅᔨᓴᖅᑕᐅᓯᖅᓱᓚᖅᓂᖅᑐᒥᒃ ᑲᑎᒪᔨᖅᑎᑦᑕ ᐃᓕᓴᐅᔪᓂᑦ ᑐᓴᐅᒪᑎᑎᑕᐅᓚᓂᖅᑐᑦ ᐃᓕᓴᐅᔪᓂᑦ ᑲᑎᑕᐅᑦᑎᓂᑦ ᓄᓇᕗᑦ ᑎᒥᖅᑐ ᐃᖃᓄᑕᓚᐱᓯᐊᑭᑎᒥᒃ ᐃᓕᓴᐅᔪᓂᑦ ᐃᓕᓴᐅᔪᓂᑦ ᐃᓕᓴᐅᔪᓂᑦ

ᐃᖅᓗᓚᒋᑕᐅᓂᖏᓕ ᐃᓕᒍᑕᖏᓕ:

- ᓄᓇᕐᓂᐅ 9, 2022-ᖓᑎᓪᓗᐊ ᐃᖅᑲᓗᓀᓴᖅᑯᑦ ᐊᓚᒪᓗ ᐃᒪᖏᐅᑕᓀᓴᖅᑯᑦ ᑲᑉᓗᓂ ᑲᑐᔾᓚᐅᖅᓯᒪᔪᑦ ᑐᓴᐅᒃᑐᓂᖅᓯᐊᕐᓂᑦ ᐊᖏᔪᖅᑕᐅᖏᑦᑐᓂᑦ *ᐃᖅᑲᓗᓀᓂᖏᓪᓗᑎᑦ (ᑕᒪᐃᐅᓄᑦ)* ᐊᑐᐊᒐᕆᑦᑕᓂᖅ ᓇᓗᓇᐃᖅᓯᒪᔪᓕᒃᑐᑎᑦ ᑭᖑᓪᓕᖏᖅᑎᑦ ᐊᒥᓱᑦ ᐃᖅᑲᓗᖅᑲᐃᐅᔪᑦ ᒪᓕᒥᐊᖅᑲᖅᑎᑕᐅᓴᖅᑐᑦ ᐃᖅᑲᓚᓱᐊᖅᑕᐅᔪᕐᓂᑦ ᐊᒥᓱᓂᕐᓂᑦ ᑎᑎᕕᖅᓯᒪᓂᖅ (ᑎᑎᕕᖅᓯᒪᓂᖅᓂᑦ ᐃᓕᖅᓂᑦ 6.1-6.3) *ᐃᖅᑲᓗᓀᓂᖅ ᐱᕕᓂᖅ* ᒪᓕᓪᓗᐊ. ᐱᖅᓯᐅᓴᓕᐅᖅᑲᓯᒪᔪᑦ ᐃᖅᑲᓚᐃᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐃᖅᑲᓴᑐᓱᐊᕐᒥ ᐊᒪᓚᓗ <ᓴᓂᖅᑐᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐊᒥᓱᓂᕐᓂᑦ ᐃᖅᑲᓚᐃᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐅᑭᐅᖅᑕᖅᑐᑦ ᖃᐅᔨᓴᖅᑐᑦ ᐊᒥᓱᓂᕐᓂᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐅᓇᓕᐸᓂᖅ ᐊᒥᓱᓂᕐᓂᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐊᒥᓱᓂᕐᓂᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐃᖅᑲᓗᐃᑦ ᐊᒥᓱᓂᕐᓂᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐃᖅᑲᓗᐃᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐃᖅᑲᓗᐃᑦ ᑲᑎᒪᓂᖅᑐᑦ ᐊᒥᓱᓂᕐᓂᑦ ᐃᖅᑲᓗᐃᑦ ᑲᑎᒪᓂᖅᑐ

ᖃᐅᔨᒪᓕᕐᖄᓂᓗᑎᓯᑦ, ᑐᓴᕆᐊᖏᕐᖃᖅᑐᖃᖅᑕᐊᓕᒃ ᑎᒍᒥᐊᖅᑎᐅᔨᓂᒃ ᐊᓯᒪᓗ ᐱᔨᓇᐅᑎᖃᖅᑐᑦᓂᒃ ᓇᓗᓇᐃᖅᓯᓕᐅᕐᓂᕐᓂᑎᓂᒃ ᑕᒪᐃᓄᐊ ᒥᓂᔅᓂᕆᑭᓇᖏᓐᓇᓂᑦ ᓄᑦᓂᑦ ᑐᓴᕆᐊᑎᑦ ᐊᑎᖅᑭᐅᖅᑐᑦ ᐃᖃᓗᐃᑦ. ᑕᐃᒪᐃᓐᓂᖏᓐᓄᑦ, ᐅᑭᐅᖅᑕᖅᑐᒥ ᐊᓯᒪᓗ ᖃᕐᓂᖅᑕᒥᐅᒃ ᓴᓂᒥᑦ ᐊᒥᐊᓕᓐᓂᒃ ᑭᖑᓂᖅᑐᑦ ᐊᒥᓂᑦ ᐃᖅᑲᓗᐃᑦ ᐃᓕᓴᕐᑕᐅᔨᓕᕐᓂᖅᑐᑦ ᐊᒥᓂᑦ ᐃᖅᑲᓗᐃᑦ ᐊᒥᐊᓕᓐᓂᒃ ᑭᖑ

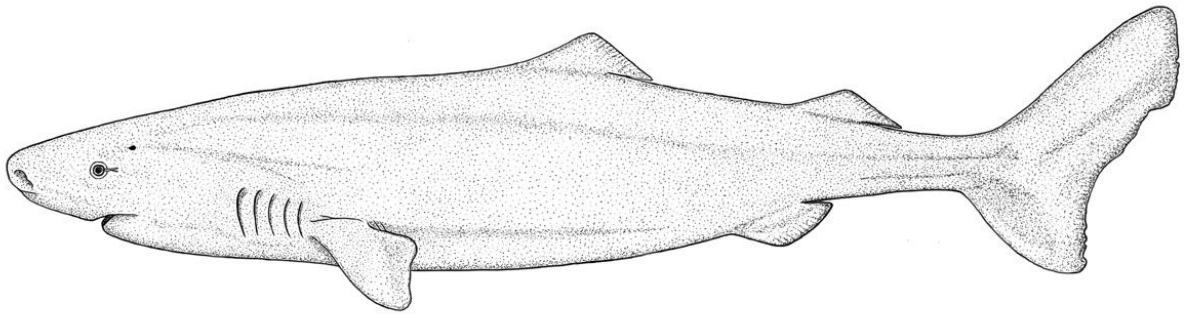
ᑐᓂᔭᐅᔨᑦ ᓄᓇᕗᒻᒥ ᐅᒪᔨᓕᓚᔨᖏᔪᐊᖅᓯᑦ ᑲᑎᒪᔨᖓᑐᓂᓄᑦ ᐊᒍᓂ 2025

ᐅᕗᖓ

ᑐᐃᒍᖅᓴᖅ: X **ᐃᓱᒪᓕᐊᖅᓴᖅ:**

ᐱᔭᕇᐅᔨᖅ: ᑐᐃᒍᖅᓴᑦ <ᖏᓇᐅᑎᓄᑦ ᑐᑭᓯᓇᓱᖅᑎᑦᕕᖅᓴᖅᓄᑦ ᐊᒪᒪᓗ ᐃᓱᒪᓕᐊᓚᖅᐅᓗᓂᓂ ᐃᓕᓕᐅᑎᖅᐅᑐᐃᓇᓚᐊᓕᑦ ᐃᖃᓗᔨᔨᐊᖅ (*Somniosus microcephalus*) ᐱᖅᐅᖅᓚᐊᖃᖅᓵᖏᑦᑐᓄᓄᑦ ᐅᒪᔨᑦ ᓄᖅᒍᑕᐅᓕᖅᑐᑦ ᐊᑦᑕᖏᖅᓯᑦᑐᑦ ᐊᑐᐅᓗᒍ ᐅᒪᔨᑦ ᐃᓄᐃᑦᑕᐃᓇᓱᐊᕐᓂᖅᓯᒪᔪᑦ ᐱᖁᔭᖅ.

ᖃᓄᐃᓕᓴᕐᓚᓱᒪᖅᓯᒪᓕᖅ:
ᑎᑎᕋᖅᓱᒪᒃᑯᖅ 3.5 ᐃᓅᓴᓗᐊᓕᓕᖅᑐᓄᐅᖅ ᐅᓄᖅᓱᓗᐊᖅᑕᑐᓄᓕᓗᓂᓗ ᑎᑎᕋᖅᑕᐅᓱᒪᔨᔨᓱᕐᑐᑦ ᐱᖅᐅᖅᓚᖃᕋᑎᑎᖅ, ᐅᔨᐊᐃ ᐃᖃᓗᓕᓚᓚᔨᖅᑎᑦ ᐃᒪᖏᐅᑕᓕᓚᓚᔨᖅᑎᓪᓗ ᑲᓇᑕ ᑐᓴᖅᑎᑎᕐᔪᑦ ᓄᓇᕗᒻᒥ ᐅᒪᔨᔨᓱᕐᑐᑦᕕᖅᑎᑦ ᑲᑎᒪᔨᖓᑐᓂ ᑲᑎᒪᔨᕐᓯᒪᓂᖅ ᐅᒪᔨᓄᑦ ᑲᑎᒪᔨᖓᑐᓂᓄᑦ ᑐᓴᖅᑎᑎᕐᔪᑦ ᐃᓅᓴᓗᐊᖅᑕᑐᓄᓕᓗᓂᓗ ᑎᑎᕋᖅᑕᐅᓱᒪᔨᔨᓱᕐᑐᑦ ᐱᖅᐅᖅᓚᖃᕋᑎᑎᖅ, ᐊᒪᒪᓗ ᐃᖃᓗᓕᓚᓚᔨᖅᑎᑦ ᐃᓅᓴᓗᐊᖅᑕᑐᓄᓕᓗᓂᓗ ᑎᑎᕋᖅᑕᐅᓱᒪᔨᔨᓱᕐᑐᑦ ᐃᓅᓴᓗᐊᖅᑕᑐᓄᓕᓗᓂᓗ ᑲᑎᒪᔨᖓᑐᓂᓄᑦ ᐊᒪᒪᓗ ᐃᓅᓴᓗᐊᖅᑕᑐᓄᓕᓗᓂᓗ ᑲᑎᒪᔨᖓᑐᓂᓄᑦ



ᓴᖏᓱᖅᑐᖅ 1: ᐃᖃᓗᐊᔨᐊᕐᖅ

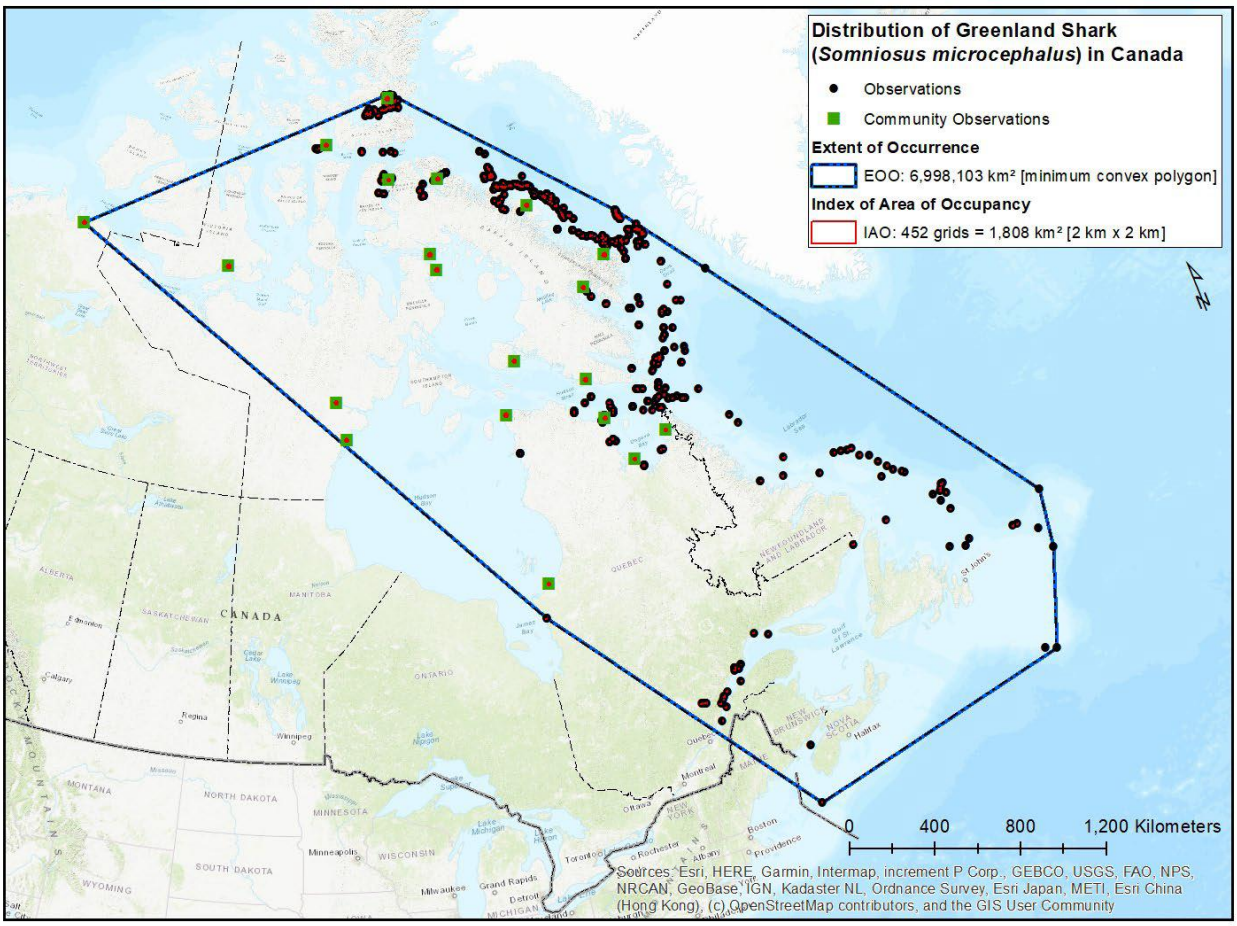
ᓇᒧᓕᕆᔨᒪᓂᖅᓯᑦ
ᐃᖃᓗᐊᔨᐊᕐᖅ ᓇᓂᕆᐅᔨᔪᓇᖅᑐᖅ ᑕᒪᐃᓂᓂ ᓄᐊᑦ ᐊᑦᓭᓪᑎᑦ ᐃᒪᘐᓕᓂ ᐊᓚᒪᓗ ᖃᓄᑭᖅᓯᓂᓂᑦ ᐅᑭᐅᖅᑕᖅᑐᖅ ᐃᒪᖅᓂᓂ. ᑕᐅᔪᓂᑐᑐᑦ ᐃᑎᔨᓂ ᐃᒪᘐᓕᓂ ᐃᒪᘐᓕᓂ Γᐅᔨᑯ ᐊᓚᒪᓗ ᑲᑎᐱᐊᔪ, ᐅᐊᖅᓇᓕᓄᑦ ᑲᓇᖅᓇᓕ ᔨᓇᐃᑎᑦ ᔨᑕᐃᑦᑦ ᐊᓚᒪᓗ ᑎᑭᐅᒪᓪᑐᑎᑦ ᑕᒪᐃᓂᓂ ᐊᑦᓭᓪᑐᑎᑦ ᑲᑎᐱᐊᓂ ᐊᓚᒪᓗ ᑲᑎᐱᐊᓂ ᐊᐃᓪᓭᓪᑦ, ᐊᓚᒪᓗ ᑲᑎᐱᐊᓂ ᐅᓭᓪᑦ ᐊᓚᒪᓗ ᑎᑭᐅᒪᓪᑐᑎᑦ ᐅᐊᖅᓇᓕᓄᑦ ᐅᐊᖅᓇᓕᓄᑦ ᐅᓭᓪᑦ. ᑲᑎᐱᐊᓂ Svalbard ᑲᑎᐱᐊᓂ ᐅᐊᖅᓇᓕᓄᑦ ᐅᐊᖅᓇᓕᓄᑦ ᐅᓭᓪᑦ. ᑲᑎᐱᐊᓂ, ᐊᖅᖃᒍ ᐃᓗᐊᓂ ᓀᓪᓭᐊᖅᓂ ᐅᓄᖅᓂᖅᓴᐅᑦᑕᖅᑎᓂᖅᓯᑦ ᐃᖃᓗᐊᔨᐊᕐᑦ ᑕᑯᐅᑦᓭᓪᓕᔨᓂᑦ ᓄᓇᖅᐸᓪᓕᒥ ᐃᒪᖅᓂ ᓄᓇᓄᓪᒥ ᐊᐅᐅᖅᑯᑦ. ᓴᖏᓱᖅᑐᖅ 2 ᑕᑯᑎᑦᑎᐊᖅ ᓇᒧᓕᕆᔨᒪᓂᖅᓯᑦ ᐃᖃᓗᐊᔨᐊᕐᑦ ᑲᓇᖅᓇᓕᓄᑦ ᐊᓚᒪᓗ ᑲᑎᐱᐊᓂ ᐃᒪᘐᓕᓂ ᐃᒪᖅᓯᑦ.

ᑐᑭᔨᓇᔨᖅᑎᑭᔅᔪᐊᓀᓪᓕᓂᑦ ᐊᒪᓚᓗ ᐊᓕᕆᖅᑕᐅᔪᐊᖅᖃᖅᑐᑦ
ᐃᓪᓚᒐ 5.2.34 (f) ᓄᓇᓄᑦ ᐊᕐᕕᑎᑦᐅᖅᖃᔨᓂᑦ ᑭᐅᑯᐊ ᓄᓇᓄᓪᒥ ᐅᒪᔨᓪᓗᓵᔨᐊᐅᑯᑦ ᑲᑎᒪᔨᕆᑦ ᐃᒪᓕᓇᐅᓂᐊᖅᑐᑦ, ᓇᕕᒥᓂᖅ ᐃᒪᓕᐅᓂᐊᖅᑐᑦ ᐅᓄᐊᓇᓇᖅᑐᑦ ᐅᕐᓇᑦᓯᓂᕕᓕᓂ ᐃᓅᔨᓪᓗᓂᖅ, ᐊᕕᒃᔨᓇᐃᓪᓗᓂᖅ ᑭᐅᑯᑐᓂᖅᓯᑦ ᐅᓄᐊᓇᓇᖅᑐᑦ ᐃᓅᔨᓪᓗᓂᖅ ᐃᓕᓴᐅᔪᓂᖅ ᐅᓄᐊᓇᓇᖅᑐᑦ ᐅᓄᐊᓇᓇᖅᑐᑦ ᐅᓄᐊᓇᓇᖅᑐᑦ

ᐊᓐᓕᖅᑕᐅᑎᑦᑎᓂᖅ
ᐱᓴᓰᖅᑎᓪᓗᒍ ᑭᒃᑯᑐᐃᓪᓗᓯᓂᒃ ᑐᑭᓯᓇᓯᖏᑎᖏᓴᕐᓂᖅ. ᓇᐃᓇᓰᖅᓯᒪᔨᒃ ᑎᑎᕋᖅᓯᒪᔨᑦ ᑐᑭᓯᓇᖏᑭᒍᑎ ᑐᓂᓴᐅᓂᐊᖅᑐᑦ ᑲᑎᒪᓂᒃᑯᓐᓂᒃ ᐅᑎᖅᑕᐅᓂᒃ ᐊᑲᐅᓂᖅᑐᑦ. ᒥᓂᑦᑕ ᐊᐢᑎᑎᓕᕆᔨᒃᑯᑦ ᐊᑦᒪᓗ ᓯᓚᐅᑦ ᐅᑉᑯᓯᐢᓕᓂᐊᓂᖓᓗᓂ ᑲᓇᑕ (ᐃᓱᔨᖅᑕᐅᓗᑎᒃ ᑕᒃᑯᓇᖓᓕᑦ ᐃᖏᓗᓕᕆᔨᒃᑯᑦ ᐃᒪᖏᐅᑕᓕᕆᔨᒃᑯᑉᓗ ᑲᓇᑕ) ᖃᓄᑦᓂᐊᖅᑐᑦ 5.2.34(f) ᐃᓯᒪᓕᐊᖓᓂᒃ ᑲᑎᒪᔨᓂᑦ ᐱᔭᐸᑎᐊᕐᓂᓗ ᐱᔭᐅᔪᓐᓇᐊᖅᑐᕐᓂᓗ ᐱᒪᖏᓐᓇᖅᑐᑦ.

ᑎᑎᕋᖅᑐᖅ:
ᑯᓕᔭᑎ <ᔪᕐᕋ, ᐅᒪᔨᓂᒃ ᐃᓅᓴᓗᐊᓚᐅᓂᐊᖅᑐᓂᒃ ᐅᒪᔨᓕᓚᕕᔨ, ᐃᖃᓗᓕᕕᔨᑯᑦ ᐃᒪᖏᐅᑕᓕᕕᔨᑯᓕᓗ ᑲᓇᑕ, ᐅᑭᐅᖅᑕᖅᑐᒥ

ᐅᓪᓗᒥ:
ᐊᐃᔪ 28, 2025



ᓴᖑᕐᑐᖅ 2. ᓇᒧᓐᓄᓕᓂᔅᓱᑦ ᐃᖃᓗᐊᕋᖅ. ᓴᖏᑎᑕᐅᓕᕆᓂᑦ ᑕᒃᑯᓇᖓᑦ ᑲᑎᒪᔨᖏᑦᑕᓂᓪᓗ ᓄᖅᓯᓂᖅᑐᓄᑦ ᐅᓪᓗᕆᓄᑦ ᑲᓇᑕᒥ (2025).

ᑐᑭᓯᐊᖏᑦᑐᑦ:

ᑲᑎᒪᔨᖏᑦ ᓄᖅᓯᓂᖅᑐᓄᑦ ᐅᓪᓗᕆᓄᑦ ᑲᓇᑕᒥ. 2025. **ᑐᓴᒃᖏᑦ.** ᑲᑎᒪᔨᖏᑦ ᓄᖅᓯᓂᖅᑐᓄᑦ ᐅᓪᓗᕆᓄᑦ ᑲᓇᑕᒥ ᖃᐅᔨᖏᖅᓯᓗᕆᓂ ᐊᐱᒪᓗ ᖃᓄᐃᓕᓂᖅᓯᒪᓕᒃᑕᒥ ᐅᓂᒃᑳᖅ ᑕᒃᑯᓂᖅ ᐃᖃᓗᐊᕋᖅ *Somniosus microcephalus* ᑲᓇᑕᒥ. ᑲᑎᒪᔨᖏᑦ ᓄᖅᓯᓂᖅᑐᓄᑦ ᐅᓪᓗᕆᓄᑦ ᑲᓇᑕᒥ. ᐊᑐᓵ. xiii + 50 pp. (https://www.canada.ca/en/environment-climate-change/services/species-risk-public-registry.html).

ᑐᓂᓴᐅᔨᓪ ᒡᖏᔪᓐᓇ ᓄᓇᕗᑦ ᐅᒪᓚᑦᑕᓇᖅᑐᔪᓂ ᑲᑎᒪᔨᖏᓐᓂᒃᑯᓂ ᐃᒪᓕᕐᒥᔪᓂ

ᑐᑭᓯᐅᕐᑭᑎ: ᐃᓱᒪᓕᐅᖅᑕᐅᔪᖅ: ᐃᒪᓐᓇᖅᑐᑎᑦ: X

ᐱᓇᐃᓗᑕᖅ: ᑲᑎᕐᑎᑦ ᐃᖅᑲᓗᑕᐅᔪᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᓂᖏᑦ ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑦ ᐅᑯᓇᓂ ᐊᓚᑐᖅᑭᓯᒪᔪᖅ 0 ᐅᑯᓄᖅ 2026 ᐊᒻᒪ 2027 ᐃᖅᑲᓚᓱᐊᖅᓇᐅᑎᓕᑐᑐᒍ.



ᑎᑎᖅᑐᑦᑐᖅ 1. ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑦ (*Reinhardtius hippoglossoides*).

ᐅᖅᑲᐅᓯᖅᑭᒪᓂᖏᑦ

ᓂᐅᖅᑭᑕᐅᕙᒃᑐᑦ ᐃᖅᑲᓕᑕᕆᓂᐅᔪᖅ ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑐᓂᑦ (GHL) ᐱᑕᖅᑲᖅᑐᖅ ᓱᓕᒋᓂ ᓄᓇᕗᒻᒥ ᐊᖏᕐᕿᑎᑎᒋᒪᒍᒋᑦ ᓄᓇᕐᔪᑎᑎᑭᐅᔪᓄᑦ (NSA) ᐅᐊᖏᓇᒥᒃ ᐊᑦᓯᖏᑎᒥ ᐃᖅᑲᓕᑕᕆᓂᖅᓗᑦ ᑲᑐᔾᕿᑎᑎᑭᖏᑦ (NAFO) ᐊᒻᒪᑐᖅᓱᒪᔪᖅ (SA) 0 ᐊᒻᒪᑐᖅᓱᒪᔪᖅ ᐅᐊᓂᓇᒍᑦ, ᐊᕕᑕᖅᓱᒪᔪᖅ 0A (ᓴᓂᓂᕈᑎᑎᐅᑦ ᐃᒪᓐᓴ) ᐊᒻᒪ ᓂᑎᓴᓗ, ᐊᕕᑕᖅᓱᒪᔪᖅ 0B (ᑕᓪᓗᕈᑎᑎᐅᑦ ᐃᒪᓐᓴᓂᓂ). ᑭᓇᐅᔾᑕᐅᕋᓱᐊᓂᖓᓕᒍ ᐃᖅᑲᓚᓱᐊᖅᓇᐅᑎᓕᑐᑐᖅ ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑐᓂᓄᑦ ᐱᑎᐊᖅᓂᖅᑲᖅᑐᑦ ᐅᓴᐃᒪᓕᖅ 1 ᐊᒻᒪ ᐃᓱᓕᓂᖅᑲᖅᑐᑎᑦ ᑎᓱᐱᓇ 31. ᓄᓇᖓᐊᖅᖢ ᑕᑯᓴᐅᑎᕐᑎᕐᓴᖅᖢ ᐅᐊᖏᓇᒥᒃ ᐊᑦᓯᖏᑎᒥ ᐃᖅᑲᓕᑕᕆᓂᖅᓗᑦ ᑲᑎᕐᑎᑦ ᐊᕕᑕᖅᓱᒪᔪᓂ ᐊᑐᑎᓂ ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑐᓂᑦ ᐃᖅᑲᓕᑕᕆᓂᖅᓗ ᓇᓂᐅᑦᑕᓇᖅᑐᖅ ᐃᓕᒍᑕᖓ 1–ᒥ.

ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑦ ᐃᖅᑲᓗᐃᑦ ᐅᕙᓂ ᐊᒻᒪᑐᖅᓱᒪᔪᖅ 0 ᐃᓕᒥᔭᖓ ᑭᓕᓕᖏᑐᓂᑦ ᐅᖓᓗᑕᐅᔾᔨᖅᑎᑦᑕᐅᑎᔪᓂ ᐃᖅᑲᓗᐃᑦ ᑲᑐᑎᔭᐅᔪᓂ ᐅᑯᓂᖓ ᑲᓇᑕ (ᐊᕕᑕᖅᓱᒪᔪᓂ 0A ᐊᒻᒪ 0B) ᐊᒻᒪ ᐊᑯᑭᑐᑦ (ᐊᕕᑕᖅᓱᒪᔪᓂ 1A ᑎᑭᑕᒍ F ᐃᒪᐃᓐᓇᕆᑦ). ᑐᓴᕐᖏᔪᓂᑦ ᑕᒪᑭᑭᓐᓄᒃ ᓄᓇᓕᔪᐊᓐᓄᒃ, ᐅᐊᖏᓇᒥᒃ ᐊᑦᓯᖏᑎᒥ ᐃᖅᑲᓕᑕᕆᓂᖅᓗᑦ ᑲᑐᔾᕿᑎᑎᑭᖏᑦ ᑭᓚᓕᓂᔪᓂᖅᓗᑦ ᑲᑎᒪᔨᖏᓐᓂᒃᑯᓂ (SC) ᐱᑕᖅᑲᑎᑎᒋᔪᑦ ᐅᖅᑲᐅᔾᔨᑎᐊᕐᒪᑦᑎᓂ ᐃᖅᑲᓗᐃᑦᑕᐅᔪᑦ ᖃᓄᐃᓇᓕᔪᓂᖏᑦ ᐊᑦᓯᖏᑎᒥ ᐃᖅᑲᓕᑕᕆᓂᖅᓗᑦ ᑲᑐᔾᕿᑎᑎᑭᖏᑦ ᑭᓚᓕᓂᔪᓂᖅᓗᑦ ᑲᑎᒪᔨᖏᓐᓂᒃᑯᓂ ᒪᓕᓂᔪᓂᖅᓗᑦ ᑲᑎᒪᔨᖏᓐᓂᒃᑯᓂ ᑭᓚᓕᓂᔪᓂᖅᓗᑦ ᐃᒪᓐᓴᓂᓂ ᐊᑯᑭᑐᑦ ᐊᑯᑭᑐᑦ ᖃᓕᕆᓕᖏᑐᓂᑦ ᐃᖅᑲᓗᐃᑦ ᐊᑯᑭᑐᑦ ᐊᑦᓯᖏᑎᒥ ᐃᖅᑲᓕᑕᕆᓂᖅᓗᑦ ᑲᑐᔾᕿᑎᑎᑭᖏᑦ ᐃᒪᓐᓴᓂᓂ ᐊᑯᑭᑐ

ᓇᓗᓇᐃᖅᑎᑎ 1. ᐊᐃᕐᑐᖅᓯᒪᓂᖓ ᑕᕐᓯᒧᖓ 2025 ᐊᑯᑭᑐᑦ ᑲᑎᒪᓗᑦ ᐱᓕᕆᓇᖅᑎᑕᐅᔪᑦ ᐱᓗᐅᔪᕆᓇᖅᑐᑦ ᑲᓂᖏᖅᑎᑕᐅᓯᒪᓂᖅᑎᑦ ᑕᕆᓇᒃᑐᒃ (t) ᑕᒃᑯᓇᖓ ᐅᐊᖓᓇᒥᒃ ᐊᑦᑉᓇᑎᒃᑎ ᐃᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ ᑲᑐᔾᔨᖅᑎᑕᐅᓯᒪᓂᖅᑎᑦ ᐊᐃᕐᑐᖅᓯᒪᔪᓂᖅ ᐊᓚᒍᖅᓯᒪᔪᖅ 0.

ᐃᖅᑲᓇᓯᐊᖏᖅᑕᐅᔪᑦ	ᐅᒥᐊᖏᔪᐊᑦ/ᐱᔪᒪᓂᖅᑲᖅᑐᑦ	2025 ᐃᓕᓴᖅᑲᖅᑕᐅᔪᑦ (ᑕᓐᓇ)
ᐅᐊᖓᓇᒥᒃ ᐊᑦᑉᓇᑎᒃᑎ ᐃᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ ᑲᑐᔾᔨᖅᑎᑦᑕᓂᖅ ᐊᐃᕐᑐᖅᓯᒪᔪᖅ 0A	ᓄᓇᕗᑦ ᐊᓯᔾᔨᖅᑎᑦᑕᔭᐅ ᑐᓂᓯᔪᓐᓇᖅᑐᑦ	8,604.99
	ᓄᓇᖅᑭᓯᒪᒥ ᐃᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ ᐱᖓᓱᓇᐊᑎᑎᓯᒪᓕᒫᑦ	100.00
	ᑲᑎᑦᑕᑦ ᐃᓕᓴᐅᔪᒥ 0A ᑯᑕᐃ	**8,704.99**
ᐅᐊᖓᓇᒥᒃ ᐊᑦᑉᓇᑎᒃᑎ ᐃᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ ᑲᑎᑦᑕᑦ ᐅᐃᖅᑲᖅᑎᑦᑕᓂᖅ ᐊᐃᕐᑐᖅᓯᒪᔪᖅ 0B	ᓄᓇᕗᑦ ᐊᓯᔾᔨᖅᑎᑦᑕᔭᐅ ᑐᓂᓯᔪᓐᓇᖅᑐᑦ	3,840.59
	ᓄᓇᓕᒃ ᐊᓯᔾᔨᖅᑎᑦᑕᔭᐅ ᑐᓂᓯᔪᓐᓇᖅᑐᑦ	402.82
	ᐱᓕᕆᓕᐅᔪᑦ ᐊᒻᒪᓗ ᐊᓯᔾᔨᖅᑎᑦᑕᔭᐅ ᑐᓂᓯᔪᓐᓇᖅᑐᑦ	2,654.10
	ᐋᖅᑭᔾᔪᑕᓄᑦ ᐃᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ ᐊᑭᑐᑕᕐᕕᒃᑐᑦ	900.00
	ᑲᑎᑦᑕᑦ ᐃᓕᓴᐅᔪᒥ 0B ᑯᑕᐃ	**7,797.51**
ᐊᓚᒍᖅᓯᒪᔪᖅ 0 ᑲᑎᖅᓱᓕᑦ	**ᐊᐃᕐᑐᖅᓯᒪᔪᖅ 0A + 0B ᐱᖓᓱᓇᐊᑎᑎᓯᒪᓂᖅᑎᑦ ᐃᓕᓴᖅᑲᓕᑦ**	**16,502.50**

2024–ᒥ, ᐃᓕᓴᐅᔪᒥ ᐊᓚᒍᖅᓯᒪᔪᖅ 0 ᑲᑎᖅᓱᓕᑦ ᐃᓕᓴᖅᑲᐅᑎᓇᖅᑐᑦ ᑕᒪᐃᓐᓂ ᐃᓕᓴᖅᑲᐅᑎᓇᖅᑐᑦ.

ᓂᐅᕐᕈᑕᐅᓕᖅᑐᑦ ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓂᓕᒫᑦ ᐊᑯᑭᑐᑦ ᖃᐅᓕᕆᓕᖅᑎᓂᓐᓂ ᑕᐃᑲᓂ ᐊᐃᕐᑐᖅᓯᒪᔪᖅ 0A ᑐᑦᖃᖅᑕᑦᓱᓕᕆᓂᕐᒧᑦ ᓄᓇᕗᒻᒥ (NU) ᐱᓕᐅᒍᓕᕆᓂᖅ. 2006–ᒥ ᓄᓇᕗᒻᒥ ᐅᒪᔪᓕᓇᔪᕐᓇᐅᑯᓂ ᑲᑎᑦᑕᑦ (ᑲᑎᒪᔨᖅᑎᑦ) ᐱᕐᑯᐊᕗᖅᑐᐅᑉᐳᓕᐅᖅᑐᑦ ᐃᓕᓴᐅᔪᖅᑕᐅᓕᖅᑐᓗᑎ 100 ᑕᕆᓂᖅᑐᒃ ᐊᐃᕐᑐᖅᓯᒪᔪᓕᒧᑦ 0A ᐃᒪᐃᐅᓪ ᑯᑕᖅᑭᖅᑕᐅᓕᖅᑐᑦ ᓄᓇᖅᑭᓯᒪᓂᖅᑎᑦ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓂᖅᑎᑦ ᐱᖓᓱᓇᐊᑎᑎᓯᒪᓂᖅᑎᑦ ᓄᓇᕗᒻᒥ ᓇᔪᖅᑕᖏᓚᓂ; ᐱᕐᑯᐊᕗᖅᑲᓇᑕᖅᑐᑦ ᐃᑲᔪᓂᖅᑕᐅᓕᐅᖅᑐᑎᓂᖅᑐᒃ ᒥᓂᑕᒧᑦ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓂᖅᑎᑦ ᐃᒪᖏᐅᑕᓇᑕᖅᓯᒪᓂᖅᑎᑦᓗ ᑲᓇᑕᒥ (ᒥᓂᑕ) ᐊᓚᒪ 100 ᑕᕆᓂᖅ ᐃᓕᓴᐅᔪᖅᑕᐅᓕᐅᖅᑐᑦ 2006 ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓇᐅᑎᓂᓗᒍ. ᑕᐃᑲᖕᖏᓂᖅ ᐃᓂᒪᓕᐅᕈᑕᖕᖏᓚᐅᓕᖅᑎᓂᓗᒍ, 100 ᑕᕆᓂᖅ ᓄᓇᖅᑭᓯᒪᒥ ᑯᑕᐅᔪᓕᕆᔨ ᒪᓕᒃᓴᐅᔾᔨᕕᖅᑕᕆᓇᖅᑎᓂᓗ ᐊᓚᒪᓗ ᐊᑦᑎᓐᓄᑦᑐᖅᑕᕆᓇᖅᑎᓂᓗ ᐊᐃᕐᑐᖅᓯᒪᔪᓂᒃ 0A ᐃᒪᐃᐅᓪ ᐱᖓᓱᓇᐊᑎᑎᓯᒪᓂᖅᑎᑦ ᐃᓂᒪᓕᐅᕈᑕᖅᑐᑦ ᖃᐅᔨᓴᖅᑎᑦ ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓂᓕᒫᑦ ᐱᓕᕆᓂᖅᑎᑦ. ᑲᑎᒪᔨᖅᑎᑦ ᑎᑎᕋᖁᓪᓗᓂᖅᑐᑦ ᐱᓕᒪᓐᓇᖅᑎᓪᓗ ᑲᑎᑦᑕᑦ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓂᖅᑎᑦ ᐃᓂᒪᓕᐅᕈᑕᖅᑐᑦ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓂᖅᑎᑦ ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓂᓕᒫᑦ ᐃᓕᖓᔪᖅᑕᐅᓯᒪᓂᖅᑎᑦ ᐊᓚᒍᖅᓯᒪᔪᖅ 100 ᑕᕆᓂᖅ ᐊᑦᑉᓇᑎᒃᑎ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ 0A ᐃᒪᐃᐅᓪ ᐱᖓᓱᓇᐊᑎᑎᓯᒪᓂᖅᑎᑦ ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓂᓕᒫᑦ. 2024–ᒥ ᐊᑕᐅᓯᖏᑦ ᓯᑯᒥ ᑲᑎᒪᔨᖅᑎᑦᑕᖏᑦ ᐱᓕᕆᓇᐊᖓᔪᖅ II ᓕᐃᓴᓐᓯ ᑐᓂᓯᔪᓕᐅᖅᑐᑎᓂᖅ ᐊᑐᐃᓐᓇᖅᑎᖏᓐᓂᕐᒧᑦ 100 ᑕᕆᓂᖅ ᐊᐃᕐᑐᖅᓯᒪᔪᖅ 0A ᐃᒪᓕᐊᑎᑎᓂᒥ ᑲᓇᑕᐅᓯᒪᓂᖅᑎᑦ ᐱᖓᓱᓇᐊᑎᑎᓯᒪᓂᖅᑎᑦ ᐃᓯᓴᖅᑲᓕᑦ 3 ᑕᕆᓂᖅ, ᐊᕐᓇᓕᓐᓂᐊᖓᔪᓂ 50 >ᓴᓇᑎ ᐱᖓᐅᓕᐅᖅᑐᑦ. ᑎᑎᕋᖁᓪᓗᓂᖅᑎᑦ ᑲᑎᑦᑕᑦ ᐃᓕᓴᖅᑲᓇᑕᖅᑐᑎᓂᓕᒫᑦ ᐊᑐᐃᓐᓇᖅᑎᖏᓐᓂᕐᒧᑦ ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓂᓕᒫᑦ ᐊᑦᑉᓇᑎᒃᑎ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓕᒫᑦ ᐃᓂᒪᓕᐅᕈᑕᖅᑐᑦ ᐃᓕᓴᖅᑲᓇᑕᖅᓯᒪᓂᖅᑎᑦ ᐃᓕᓴᖅᑲᓚᓯᐊᕐᓂᓕᒫᑦ ᐃᓕᖓᔪᖅᑕᐅᓯᒪᓂᖅᑎᑦ ᐃᓯᓴᖅᑲᓕᑦ

ᖃᑦᑕᖅᓯᒪᓂᖅ ᐃᖃᓗᓯᑎᑕᐅᔪᓐᓇᓂᖅ ᐊᒻᒪᓗ ᐊᑐᖃᖅᑕᐅᔪᓐᓇᓗᒥ ᖃᐅᔨᓴᖅᓯᒪᔪᓐᓇᖅ ᐃᖃᓗᓯᑎᑕᐅᔪᓐᓇᖅ ᖃᓄᐃᑦᑐᖅ ᓴᓇ 2024 ᐊᒻᒪᓗ 2025 ᑕᖃᕈᓐᓇᓂ ᐅᐊᖏᓐᓇᒃ ᐊᑦᓯᓐᓄᖅᒥ ᐃᖃᓗᓯᑎᑕᐅᔪᓐᓇᖅᓯᒪᓂᖅ ᑲᑎᑦᓯᐊᖅᑐᑦ ᐱᑕᖃᖅᑐᑦ ᐃᓕᓴᕐᓂᖅ ᐱᑕᖃᓕᕐᓂᖅᑎᑦᓗᓂᖅ. ᐱᖓᐃᐅᔪᓂ ᑕᒪᑐᐊᑎᒧᑦ ᐱᖃᓗᐊᖅᑐᒥᑦ ᐃᖃᓗᓯᑎᑕᐅᔪᓐᓇᖅ ᓄᓇᕗᑦ ᑎᑎᖃᖅᖃᓯᐅᒃᑐᓂ ᐊᒻᒪᓗ ᐊᑐᖃᖅᑕᐅᖓᓚᖅᑐᑦ ᓇᓗᓇᐃᔪᕐᓂᖅᑐᑦ ᑕᐅᔭᕐᓂ ᐅᓇᖅᑐᖅ ᐃᓕᓴᕐᓂᖅᑎᑦᓗᓂᖅ ᐅᒪᓐᓇᖅᑐᑦ ᐊᖏᔪᖅ ᐅᑭᐅᖅᑕᐅᔪᓐᓇᖅ ᐃᓕᓴᕐᓂᖅᑎᑦᓗᓂᖅ ᐅᒪᓐᓇᖅᑐᑦ ᐊᑐᖃᖅᑕᐅᔪᓐᓇᖅ ᐃᓕᓴᕐᓂᖅᑎᑦᓗᓂᖅ

ᑭᖅᑕᓯᓂᐊᕐᓂᖅ ᑲᑎᒪᔨᖏᑦ ᓂᐱᖅᑎᑎᕆᔨᐅᙱᖏᑦᑐᖅ “ᓯᐊᖅᓇᒧᑦ ᑭᒃᑯᓗᒃᑐᒧᑦ” ᐊᑐᕐᕆᔨᐅᕐᖅ ᑕᐃᒪᐃᕐᒪᑦ ᐊᖅᑐᒍᑦᑐᖅᓂᑦ. ᑐᑭᓯᓇᖅᑐᒥᔨᑐᓂᓂ, “F_{target}” ᑐᑭᖅᑲᖅᑎᑕᐅᖓᒪᑦ ᓯᐊᖅᓇᖅᑐᒥᑦ ᑭᒃᑯᓗᒃᑐᒥᑦ ᑭᖑᓂᓯᒪᔪᑦ ᐅᓂᒃᑲᓕᐊᒥᑦ ᐊᑐᕐᓂᐊᕐᙱᖏᑦᑐᖅ, ᐅᑦᑐᑎᐅᕐᕕᖅ ᐊᓂᐊᑎᑎᓂᒥᑦ ᐊᒋᔪᑦ ᐅᐊᖓᓇᒃ ᐊᑦᕿᓐᓂᒃᑎ ᐃᖅᑲᓕᑕᓂᖅᓗᑦ ᑲᑎᕐᔪᖅᑎᑎᑦᑕᕐᓂᖅᓗᑦ ᓯᓚᐅᑐᐊᔪᑦ ᐊᑐᖅᑕᐅᕐᔨᓗᑦ ᐊᑦᑕᐅᖅᑕᐅᕐᓯᒪᔨᓄᑦ ᓂᕈᐊᖅᑕᐅᕐᔨᖅ ᓴᖅᑭᐅᖅᑕᐅᕐᓯᒪᔨᓄᑦ ᐃᓅᓯᕆᔨ

ᓂᓚᐅᔭᓐᓄᑦ ᖃᐅᔨᓴᓂᐊᓄᑦ ᑕᐅᑐᖓᐊᖅᑕᐅᔪᑦ ᑐᓂᔾᐅᓭᓴᖅᑐᑦ ᐱᓇᓱᐊᕕᓱᓂᒃ ᐊᑎᕐᕕᒃᑯᓕᕐᓂᑦ ᐊᒥᓱᓕᕐᓂᑦ ᐃᓕᓴᕐᓇᓵᖅᐊᓂ ᐅᑎᓕᓴᖅᑎᑦ ᐅᑎᒍᑎᓕᕐᓂᑦ ᐃᓐᓇᐅᑉ ᐅᓪᓗᓕᕐᓂᑦ ᐊᑐᐊᓂ ᐅᑎᓕᕐᓂᑦ ᐃᓕᓴᕐᓇᓵᖅᐊᓂ ᐅᑎᓕᓴᖅᑎᑦ ᐅᑎᒍᑎ

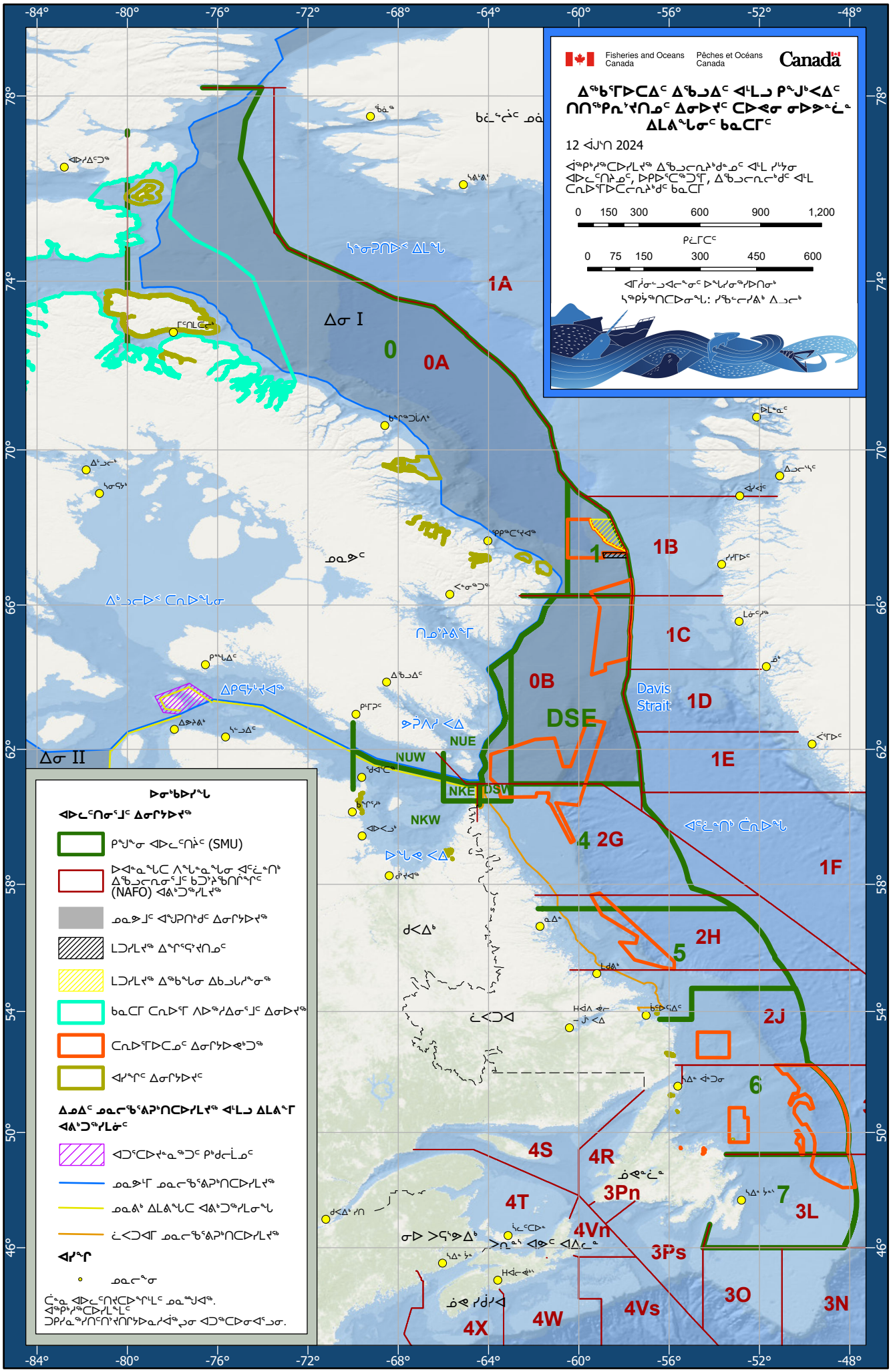
3. ᐃᓕᐅᖅᑲᖅᑕᐅᓂᖏᓐᓂ ᑯᑕᖏᓂ ᐅᒥᐊᖅᐊᓄᑦ, ᐱᖅᓱᐅᑎᓪᓗᑎ ᓄᓇᖅᐸᓱᒻᒥ ᑯᑕᖏᓂ ᐃᖅᑲᓕᓚᓂᖅᓴᑦ ᐱᓕᓕᓕᐊᑎᕆᑎᓂᖅᓴᑦ.

ᐱᓕᕆᑕᖅᑕᖏᓐᓄ ᐅᒪ: ᐃᓕ ᐃᒡᓕᓐᓄ, ᐊᐅᓪᓚᕆᑎᓕᓚᓱᒃ ᐃᖅᑲᓕᓚᓂᖅᓴᒧ ᐱᓕᓚᔪ, ᔪᓄ ᐊᑦᑕᒻ, ᐊᕐᕉᑐᖅᓯᒪᔨᓂ ᐃᖅᑲᓕᓚᓂᖅᓴᒧ ᐊᐅᓪᓚᕆᑎᔭᓄᑦ ᐱᓕᓚᔪᓕᓚᓱᒃ, ᐊᓕᒪᓗ ᑭᐃᓐ ᕿᐃᔪᔪ, ᖃᐅᐅᔨᖅᑎᓄᓇ ᑭᖑᓪᓗᓯᐊᖅᑎ, ᐅᑭᐅᖅᑕᖅᑐᑦ ᐊᐃᕐᑐᖅᓯᒪᓂᔨᓂ, ᐃᖅᑲᓕᓚᓂᖅᓴᑦᓯᓗ ᐃᒪᖏᑦᐅᑕᐅᓂᖅᓴᑦ ᐃᓕᖏᑕᐅᑕᓂᖅᓴᓪᓗ ᑲᓇᑕᒥ.

ᐅᓪᓗᖅ: ᔮᓐᓄᐊᕐᓇᖅ 2, 2025

ᐃᓕᓴᕐᕕᖏᓂ

ᐃᓕᓴᕐᓂᑲ 1 – ᓄᓇᖕᖏᐊᕐᕋᖅ ᐃᖅᑲᖏᑕᐅᑕᐃᓕ ᐃᖅᑲᓚᐃᓕ ᐊᓚᒪᓗ ᑭᖕᖏᔪᕐᕈᐃᓕ ᑎᑎᖅᑭᑦᓯᕐᓗᓂᓄᑦ ᐃᓂᐅᔨᓕ ᑕᐅᖃᓂ ᓂᐅᓚᐅᓂᒃᑯᓐᓄᑦ ᐃᒪᓕᐃᓐᓇᓂᓕ ᑲᓇᑕᒥᓕ
ᐃᓕᓴᕐᓂᑲ 2 – ᐅᐊᖏᓇᒥᒃ ᐊᑦᑯᓐᓄᒥ ᐃᖅᑲᓕᓚᓂᖅᓴᑦ ᑲᑐᔪᖅᑎᑎᒋᖏᓂ ᑭᖑᓕᓚᓂᐊᕐᓂᖅᓴᒧ ᑲᑎᒪᔨᖏᓂ ᓇᓕᓇᐃᔪᖅᓯᒪᓂᖏᓂ ᑎᑎᕋᖅᓯᒪᔨᓕ
ᐃᓕᓴᕐᓂᑲ 3 – ᑲᑎᒪᓂᖅᓴᑦ ᓇᓕᓇᐃᔪᖅᓯᒪᓂᖏᓂ ᑲᓇᕋᖏᓚᓂ ᐃᖅᑲᖏᑕᐅᑕᓚᓂᖅᓴᑦ ᖃᑲᐅᔨᒪᕆᓚᕆᐅᔨᓕ ᑲᑎᒪᔨᕋᑦᑕᑦ ᐃᓕᑎᕐᔪᐅᔨᓕ ᑕᑯᐊᓇᖅᑕᖏᓂ 2026 ᐊᓚᒪᓗ 2027 ᐊᑯᑭᑐᑦ ᖃᑲᑕᕋᓕᖏᐊᓂ ᑲᑎᑦᓗᑎ ᐃᖅᑲᐳᑕᐅᔨᐊᓇᖅᑐᑦ
ᐃᓕᓴᕐᓂᑲ 4 – ᑲᑐᔪᓂᑲᑯᑦ ᑭᐅᔪᐅᓂᖏᓂ ᑕᐃᑲᖕᖏᑦ ᓄᓇᕗᓪᓕ ᐃᖅᑲᓕᓚᓂᖅᓴᑦ ᑲᑐᔪᖅᑎᑎᑦ ᐊᓚᒪᓗ ᓄᓇᕗᑦ ᑲᕐᒪᓴᑯᓂᖅᓂᓄᑦ ᑭᐳᑯᓄᖕᖏᑦ 2026 ᐊᓚᒪᓗ 2027 ᐊᑯᑭᑐᑦ ᖃᑲᓕᕋᖏᓐᓂᓄᑦ ᑲᑎᑦᑕᒍ ᐱᑎᑕᐅᔨᐊᓇᖅᑐᓂᑲ ᐱᔪᐅᓂᖏᓐᓂᓄ (English version attached)



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ᐃᓕᒍᑕᖏᑐ 2

ᐊᑯᑭᑐᑦ ᖃᓕᕋᓕᖅᑭᑦ ᓇᒃᓂᖅᖏᑐ 0+1 (ᐃᒪᐃᓐᓇᕆᑦ)
ᐅᖃᐅᓯᐅᒋᐊᑭᑦ ᐋᑉᓂ 2025 ᐅᑯᓇᖏᑐ 2026-2027

ᐊᑐᓕᖅᑯᐅᔪᖅ ᐅᑯᓇᖏᑐ 2026 ᐊᒻᒪ 2027

ᖃᐅᓯᓴᖅᑐᑕᓗᓂᔪᑦ ᑲᑎᒪᓯᑦ ᐃᖃᓗᐃᑦ ᐱᑕᖃᖅᓂᐊᖅᓂᖅᑭᑦ ᐃᓱᒪᑎᕐᐅᔪᑦ ᓇᓗᓇᐃᖅᑕᐅᓱᒪᓂᖅᖏᓐᓄᑦ ᖃᓄᓱᐅᓂᖅᑭᑦ ᐅᖓᓂ ᖃᓇᐃᖅᑐᑎᐊᖅᑐᓗᑦ ᐃᓂᐅᔪᒥ ᐅᖓᓂ ᐅᓯᓴᓱᓕᐊᑦᑕᐅᔪᖅᑯᑦ, ᐊᒻᒪᓗ ᐱᖓᓱᓂ ᐃᓴᓂᐅᖅᖓᖓᑐᓂ ᐱᓯᐅᔪᓐᓂ ᓱᓴᒋᑦᑐᓗᑦ ᓄᓇᕗᖓᑦ. ᑕᐅᑐᖓᔪᐊᖅᑕᐅᔪᓴᖅᑯᑦ ᐃᓴᖃᓗᐃᑦ ᑐᖓᓄᐅᑎᓱᒪᖅᑕᑕ ᐃᒪᐃᓐᓇ B_{lim} 2028-ᒥ ᐃᒪᐃᓐᓇᐅᑕᐅᓴᖅᑐ <1% ᐊᒻᒪ ᑐᖓᓕᓂ $B_{trigger}$ ᑐᖓᓕᓅᓐᓂᔪᓐᓇᕆᑕᐅᓴᖅᑐ 10%-ᒥ.

F_{target} ᒪᑕᕆᑎᑕᐅᓱᒪᔪᖅᑐ ᐃᓴᖃᓴᑕᐅᓱᒪᔪᓂᖅ 30,243 ᑕᓐᓇ ᐊᒻᒪ 30,153 ᑕᓐᓇ ᐅᖓᓂ 2026 ᐊᒻᒪ 2027. ᐅᑯᐊ ᐱᖁᐅᔪᓄᑦ ᐊᖖᑕᓂᖅᑎᑕᐅᔪᑦ ᖃᖏᑕᐅᑎᓱᒪᑐᐃᓐᓇᓕᐊᑦᓯᓕ 35%-ᒥ ᐊᒻᒪ 36%-ᒥ ᖃᖏᑕᐅᑎᓱᒪᓂᖅᓱᓄᑦ F_{lim}.

ᐊᐅᓕᕆᓂᖅᓗᑦ ᐱᓕᓕᐊᖏᔪᑦ

ᑲᓇᑕ ᐊᒻᒪ ᑎᐊᖑᒧᖅ (ᑭᓱᖃᖅᑐᐃᓕᓗᑎᒃ ᑲᓪᓚᓕᓂ ᓄᓈᓂ) ᐱᔪᒪᓕᐅᖏᒪᑕ ᖃᐅᓰᓴᖃᖅᑕᐅᖏᑦᑐᓕᓗᑎᒃ ᐃᓴᖃᓗᐃᑦ ᖃᓄᐃᓕᓴᖅᖓᑦᓯᓂᖅᑐᓂᓯᓂᖅᑐᒃ ᐊᒻᒪ ᐅᖃᐅᓰᓴᓗᑎᒃ ᒪᓕᓐᓂᔪᓐᓇᐅᑉ ᐅᑯᐊ NAFO-ᑯᑦ ᐅᓰᓴᓂᔪᓕᓕᓐᓂᒃ ᑲᒪᒋᐅᔾᔨᓕᓕᖅᖃᖅᓯᓂᖅᑭᑦ ᐱᓕᓐᓄᔪᐊᑎᐊᖅᓴᓂᒃ (PAF).

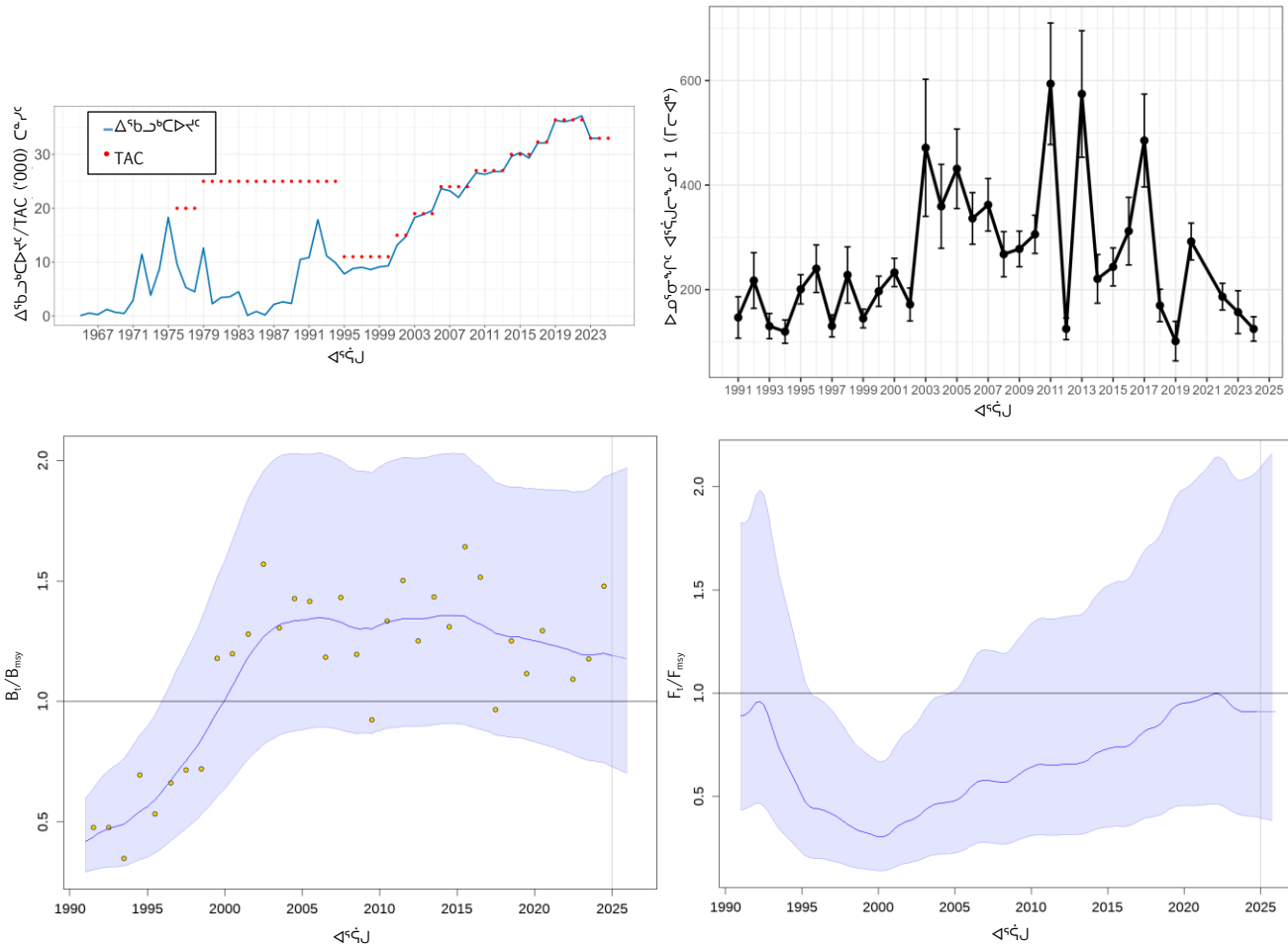
ᑲᑎᒪᔨᐊᕐᓂᖅᓗᑦ ᑐᖓᓕᕐᓂᖅ	ᖃᓄᐃᓕᓴᖅᖓᓕᓄᑦ ᐅᖃᐅᓯᓴᖅᑭᑦ	
ᐅᖃᑎᒌᓗᑦ ᐅᕐᓯᑉᔅᓂᑦ ᑕᐃᒪᐃᓕᓴᖅᓱᑎᓇᖅᑐᓗᑦ ᑕᐃᒪᒃᑯ Bmsy	B > Bmsy	ᖃᓄᐃᓕᓴᖅᖓᓂᖅᑐᑦ
ᐱᑕᖃᓄᐅᓂᖅᑎᒌᓗᑦ ᐃᓴᖃᓗᓯᓗᓇᐊᕐᓂᑦ (ᐱᑕᖃᓴᖅᖃᐅᔪᒥ)	F < Flim	ᓇᓗᓇᐃᖅᑐᑎᓱᒪᓂᖅᑐᑦ
ᐱᑕᖃᓄᐅᓂᖅᑎᒌᓗᑦ ᐃᓴᖃᓗᓯᓗᓇᐊᕐᓂᑦ (ᐅᒪᓂᖅᖃᖅᑐᑦ ᓇᔨᓴᒥᓂᑦ)	TCI ᓇᓗᓇᐃᔭᖅᑕᐅᓯᒪᓕᕐᓂᖅᑐᑦ	ᐊᑲᓇᖅᓯᒪᕐᓂᖅᑐᑦ
ᐊᑐᓗᓂ ᐊᖏᕈᓇᐃᖅᓯᒪᓂᖅᓱᑦ ᐊᑐᖅᑕᐅᓂᖓᖓ	ᑕᒪᐃᓐᓂ ᑲᒪᒋᐅᔾᔨᓕᓕᖅᖃᖅᓯᓂᖅᑭᑦ ᐱᖁᓘᓐᓃᑦ ᖃᓄᐃᓕᓴᖅᖓᓕᐅᑐᐃᓐᓇᕐᓂᖅᑐᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓕᕐᓂᖅ	ᖃᓄᐃᓕᓴᖅᖓᓂᖅᑐᑦ
ᐊᖏᑎᒋᓕᐊᑐᒍ ᐊᖏᕈᓇᐅᓐᖓᓇᓂᑦ ᐊᖏᑕᖃᕐᕕᐊᓂᑦ ᐅᒪᓕᐊᓂᑦ ᐃᒪᒋᐅᑕᓂᑦ ᐊᒻᒪ ᐊᖖᑎᓐᓇᓂᑦ	ᑐᑭᒧᐊᖅᑎᑕᐅᓂᑦ ᐃᓴᖃᓗᓕᓴᐅᓂᑦ VME ᒪᑐᓂᒧᑦ ᐊᑐᖅᑕᐅᓂᑦ, ᐊᑐᑎᖅᑎᑎᐊᖅᑐᑦ ᐊᖓᔪᓂᖅᑕᐅᓂᑦ ᒪᓕᒐᖓᑦ ᓇᓗᓇᐃᖅᑐᑦ	ᐊᑲᓇᖅᓯᒪᕐᓂᖅᑐᑦ
ᓴᓗᐃᑦᑕᐃᓕᓕᓂᑦ ᑕᓇᐅᒥ ᐅᒪᓕᐅᓂᖅᑐᓂᓯᓂᖅᑐᑦ	ᖃᓄᐃᓕᓴᖅᖓᓂᖅᑐᑦ ᓇᓗᓇᐃᖅᑕᐅᓴᖅᓯᒪᕐᓂᖅᑐᑦ	ᐊᑲᓇᖅᓯᒪᕐᓂᖅᑐᑦ

ᐊᐅᓕᕆᓂᐊᒻᒪᓴᐅᓂᖅᑐᑦ

ᐊᑯᑭᑐᑦ ᖃᓕᕋᓕᖅᑭᑦ ᐃᓴᖃᓗᖅᑭᑦ ᐅᖓᓂ ᐊᓴᑐᖅᓯᒪᔪᖅ 0 + 1 (ᐃᒪᐃᓐᓇᕆᑦ) ᐃᓕᑎᒥᖓ ᐊᖏᑎᓂᖅᓴᓂᑦ ᐊᒥᓱᓂᖅᑎᓐᓂᑦ ᓇᒧᖓᓴᖅᑕᕐᓂᖅᑭᑦ ᐅᐊᓐᓇᖓᑕ ᐱᖓᓐᓇᖓᓂ ᐊᑦᓭᓐᑎᒃ.

ᐃᓴᖃᓗᖅᑭᑦ ᖃᓄᐃᓕᓴᖅᖓᓂᖅᑭᑦ

ᑲᑎᓕᒻᒥᖅᓯᑎᒃ ᖃᑦᓭᓂᑦ B_{msy} (B/B_{msy} = 1.2), ᐊᑕᓂᓐᓂᔪᐅᑐᐃᓐᓇᕆᑦᑕᖃᖅᑎᓂᖅᑭᑦ B_{lim} (30% B_{msy}) ᑐᖓᓕᓂᒃᑲᑦ 1%, ᐊᒻᒪᓗ ᑐᖓᓕᓂᐅᑐᐃᓐᓇᕆᑦᑕᖃᖅᑎᓂᖅᓱᒪ ᐅᓇᖅᓱᒪ $B_{trigger}$ (75% B_{msy}) ᐃᒪᐃᑐᖅᑲᑦ 3%. ᐃᓴᖃᓗᓯᒃᓴᓂᖅ ᑐᖓᓕᓂᒃᑲᑦ F_{msy} (F/F_{msy} = 0.91), ᖃᓄᐃᓂᓂᐅᑐᐃᓐᓇᕆᑦᑕᖃᖅᑎᓂᖅᓱᒪ F_{lim} (F_{msy}) ᓲᓂᓕᓂᐅᓂ 42%, ᐊᒻᒪᓗ ᖃᓄᐃᓂᒃᑲᑦ ᓲᓂ F_{target} (85% F_{msy}) ᐃᒪᐃᑐᖅᑲᑦ 57%.



ᓇᓗᓇᕐᕕᖏᓂᖅ ᓇᑭᐃᓴᖅᓂᖅ

B_{lim} ᐃᒪᐃᑐᖅ 30% *B_{msy}*. *$B_{trigger}$* ᐃᒪᐃᑐᖅ 75% *B_{msy}*. *F_{lim}* ᐃᒪᐃᑐᖅ *F_{msy}*. *F_{target}* ᐃᒪᐃᑐᖅ 85% *F_{msy}*.

ᓇᓕᐅᕈᑎᑎᐅᔪᓂ

ᐊᑯᓂᖅᓯᕐᓗᖅ-ᐊᑯᓂᐅᓂᖅᓴᓇᓂ ᑐᕐᓴᐅᔪᓂ ᑲᔪᓯᐅᕐᓂᕐᓕᐅᔪᖅᑐᑦ ᐊᖏᕐᒪ 2028-ᒧᑦ ᐊᖅᓯᓂᖅᑕᐅᔪᓇᓂ ᐋᖅᑭᖅᓯᐊᖅᑕᐅᓯᒪᔪᓂ = ᑲᑎᓇᑎᑦ ᐊᖅᓯᓂᖅᑕᐅᔪᓄᓇᖅᑐᑦ (TAC) 33,005 ᑕᓪᓯᑦ 2025-ᒧᑦ. ᐃᒪᐃᖅᑎᓄᓇᖅᑐᑦ *F* ᐅᓇᕐᓂᖅᓯᑎᑐᒧᑦ ᐃᓕᓕᐅᑎᑕᐅᓕᐅᖅᑐᑦ ᐅᖃᖅᑐᑦ 2026-2027 ᐊᐳᑲᑎᖅᓂᓂᑎᑎᒧᑦ ᐅᖃᖅᑐᑦ *F* ᐃᓕᐅᑦᓗᓂ *F*=0, ᐅᑯᐊ ᐱᖁᑦᓂᖁᔨᑦ ᕿᑯᓂᓴᑐᑎᑕᐅᔪᓂ ᐅᓇᕐᕆᕐᓂᖅᓯᑎᑦ ᖃᓄᐃᖅᑎᑎᑕᐅᑐᒧᑦ ᐃᓂᐅᔪᓂ (75% *F_{msy}*, 85% *F_{msy}*, ᐊᒪᒪ *F_{msy}*,) ᐊᒪᓚᓂ ᐱᖁᑦᓂ ᕿᑯᓂᓴᑐᐃᑎᒥ ᐱᓕᐅᔪᓕᓂᔪᓂᒃ ᓂᓕᒻᒥᑎᓂᓇᓂ ᓇᓇᕐᓂᓇᓂ (*$F_{status\ quo}$*, ᐃᖏᓇ ᑲᑎᑦᓇᓂᑦ ᐊᖏᕐᒪᓇᓂᑕᐅᔪᓄᓇᖅᑐᑦ (TAC), ᐊᒪᒪ 2019-2022 ᑲᑎᑦᓇᓂᑦ ᐊᖏᕐᒪᓇᓂᑕᐅᔪᓄᓇᖅᑐᑦ (TAC)).

ᐅᒧᖏᓚᓂ $F_{status\ quo}$ ᐃᓂᒪᑎᐅᔪᓇᓂᑦ, ᐃᒪᐃᑐᐃᖏᓇᓚᐊᖅᑎᓂᖏᓐᓄ ᐅᖃᓂ *F* > *F_{lim}* 2027 ᐃᓱᓚᐅᔾᔨᑐᖅ 51%. ᐅᖃᓂ 75% *F_{msy}*, ᐃᒪᐃᑐᐃᖏᓇᓚᐊᖅᑎᓂᖏᓐᓄ ᐅᓇ *F* > *F_{lim}* ᐅᖃᓂ 2027 ᐃᓱᓚᐅᔾᔨᑐᖅ 26%. ᑐᕐᖅᑕᐅᔪᓂ ᖃᓄᐃᖅᑎᑎᑕᐅᑐᑎᓂ 85%-ᖏᓪᕐᖅ *F_{lim}*, ᑕᐃᒪᐃᐃᖏᓇᖅᑎᓂᖏᓐᓄᓗ ᑕᓪᓕ *F* > *F_{lim}* 2027 ᐃᒪᐃᓂᒪᓕᓂᓂᐅ 36% ᐊᒪᓚᓗ ᑕᒪᑐᐃᖏᓐᓄ *F_{msy}* ᑐᕐᖅᑕᐅᔪᓂᑦ, ᑕᐃᒪᐃᐃᖏᓇᖅᑎᓂᖏᓐᓄ ᐱᑎᕐᓂᒪᓂᕐᒥᐅᑦ 50%-ᒧᑦ. ᐃᓂᒪᑎᐅᔪᓇᑎᑐᒧᑦ ᐃᖏᓇ ᑲᑎᑦᓇᓂᑦ ᐊᖏᕐᒪᓇᓂᑕᐅᔪᓄᓇᖅᑐᑐᓂ TAC (33,005 ᑕᓪᓕ) ᐊᒪᒪ ᐅᖃᓂ 2019-2022 ᑲᑎᑦᓇᓂᑦ ᐊᖏᓂᓇᓂᑕᐅᔪᓄᓇᖅᑐᑐᓂᑐᒃ (TAC) (36,370 ᑕᓪᓕ) ᐃᒪᓇᐅᑐᑐᐃᓇᓚᐊᖅᑎᓂᖏᓐᓄ ᐅᓇ*F* > *F_{lim}* ᐅᖃᓂ 2027 ᐃᓱᓚᐅᔾᔨᑐᖅ 44% ᐊᒪᒪ 54%. ᐅᓇᕐᓂᖅᑎᑎᒧᑦ ᐃᓂᒪᑎᐅᔪᓇᓂᑦ, ᖃᓄᐃᓕᓂᖏᓚᓕᒥᓇᕐᓂᖅᑎᑎᑦ ᐅᓇᕐᓂᓗᓚᐅᔪᓂᑦ ᐊᑕᐅᓂᓇᓚᕐᓂᖅᑎᑎᑦ *B_{lim}* ᐅᖃᓂ 2028 ᑐᖏᓕᐅᔾᔨᑐᑦ 1%. ᖃᓄᐃᓕᓂᖏᓚᓕᒥᓇᕐᓂᖅᑎᑎᑦ ᐅᓇᕐᓂᓗᓚᐅᔪᓂᑦ ᐊᑕᐅᓂᓇᓚᕐᓂᖅᑎᑎᑦ *$B_{trigger}$* ᐅᖃᓂ 2028 ᐃᓱᓚᐅᔾᔨᑐᖅ 4% ᒪᐅᖏᓐ 9% 6-ᓂ ᖃᓄᐅᓴᖅᑕᐅᔪᓇᓂᖅ ᐃᓕᓴᓕᓐᓂᓂᒧᑦ, ᑭᓇᐊᓂ ᐅᒪ *F*=0 ᖃᓄᐃᓇᓕᖏᓐᓄᓇᓇᕐᓂᖅᓯᒪᔪᓂ.

ᖁᑯᐃᓕᓐᓄᒪᓂᖅᑎᑦ ᐅᓄᖅᑎᓚᐅᐊᓱᑦ ᐅᖅᑎ 2028 ᖁᑯᓯᓂᖅᑎᖅᖁᖅ ᐅᓄᖅᑎᓂᑦᑎᔪᑦ ᐅᖅᑎ 2025 ᖁᑎᐊᐃᓂᑐᖅ 27% ᐊᓱᒪ 52% ᑕᒪᐃᓐᓄᑦ ᐃᓱᒪᑎᕆᐅᐊᓕᐊᑎᓂ, ᐅᓇ ᐃᓕᐅᖕᖏᑦᓴᓂ ᐅᓇ *F*=0 ᖁᑯᐃᓕᓐᓄᓂᖅᓇᓵᖅᑎᓂᖅᓄᑦ.

ᐃᓱᒪᑎᓴᐅᓴᐊᑎᐅᓴᓪ ᐃᔅᖅᓗᵇᑕᐅᓴᵊᓐᓇᔅᖅᓯᓂᓂ$_{2025}$ = 33,005 ᑕᵊᔪ		
ᐊᔅᖏᔪ	ᐱᑕᖅᔅᓂᖏᑎᓪ (‘000t) ᐅᓄᔅᓂᓗᓴᐅᑎᒍᑦ	ᑐᖁᖅᑕᐅᓴᓪ ᐊᵇᑐᐊᓂᓕᓪ ᐊᑐᵇᖅᵊᓂᑭᵊᓐᓇᔅᖅᑐᑦ ᓄᓇᒥᖏᓕᖅᑐᑦ ᓂᔅᓴᑎᓂᖏᓕᖅᑐᓂ (B/Bmsy) ᐅᓄᔅᓂᓗᓴᐅᓴᓪ (80%CI)
F=0		
2025	33005	0.87 (1.19, 1.64)
2026	0	0.84 (1.18, 1.65)
2027	0	0.97 (1.30, 1.74)
2028	-	1.10 (1.42, 1.83)
F = 0.75F_{msy}		
2025	33005	0.87 (1.19, 1.64)
2026	26811	0.84 (1.18, 1.65)
2027	27043	0.84 (1.19, 1.67)
2028	-	0.85 (1.20, 1.69)
F = 0.85F_{msy}		
2025	33005	0.87 (1.19, 1.64)
2026	30243	0.84 (1.18, 1.65)
2027	30153	0.83 (1.17, 1.66)
2028	-	0.82 (1.17, 1.68)
F = F_{msy}		
2025	33005	0.87 (1.19, 1.64)
2026	35328	0.84 (1.18, 1.65)
2027	34618	0.80 (1.15, 1.65)
2028	-	0.77 (1.13, 1.65)
F = $F_{status\ quo}$		
2025	33005	0.87 (1.19, 1.64)
2026	35502	0.84 (1.18, 1.65)
2027	34768	0.80 (1.15, 1.65)
2028	-	0.77 (1.13, 1.65)
ᐃᓵᓀᐅᐊᓴᖅ ᑲᑎᑦᓗᒍ ᐊᖓᐃᓀᓯᑲᑕᐅᐊᓴᓀᓇᔅᖅᑐᓀᓄ		
2025	33005	0.87 (1.19, 1.64)
2026	33221	0.84 (1.18, 1.65)
2027	32789	0.81 (1.16, 1.65)
2028	-	0.79 (1.15, 1.66)
2019-2022 ᑲᑎᑦᓗᒍ ᐊᖓᐃᓀᓯᑲᑕᐅᐊᓴᓀᓇᔅᖅᑐᓀᓄ		
2025	33005	0.87 (1.19, 1.64)
2026	36834	0.84 (1.18, 1.65)
2027	35906	0.81 (1.16, 1.65)
2028	-	0.79 (1.15, 1.66)

		ᐃᖃᓗᐊᑕᐅᔪᓂ 2025 = 33,005 ᑕᕐᓂᓂ						
		F=0	ᖃᓄᐃᖅᑭᑎᐊᖃᖅᑐᒍᓂ ᐃᓕᑦᑎᔪᖅ			ᓂᕿᓯᑦᑐᓄᓕᓂ ᓄᓇᒥᖅᓂᓄᓕᓂ ᐱᓕᕆᔨᓕᔪᓂ		
			F = $0.75F_{msy}$	F = $0.85F_{msy}$	F = F_{msy}	F status quo	ᒪᐃᓇᐅᔪᖅ ᑲᑎᒪᔨᒍ ᐊᖏᔪᓕᐅᕆᔨᓇᖅᑐᓄᓕᓄ	2019-2022 ᑲᑎᒪᔨᒍ ᐊᖏᔪᓕᐅᕆᔨᓇᖅᑐᓄᓕᓄ
ᐱᑕᖃᕐᓂᖅᓯᑦ (50%)	2025	33005	33005	33005	33005	33005	33005	33005
	2026	0	26811	30243	35328	35502	33005	36370
	2027	0	27043	30153	34618	34768	33005	36370
$P(F>F_{lim})$	2025	42%	42%	42%	42%	42%	42%	42%
	2026	<1%	25%	35%	50%	51%	44%	54%
	2027	<1%	26%	36%	50%	51%	44%	54%
$P(B<B_{lim})$	2025	<1%	<1%	<1%	<1%	<1%	<1%	<1%
	2026	<1%	<1%	<1%	<1%	<1%	<1%	<1%
	2027	<1%	<1%	<1%	<1%	<1%	<1%	<1%
	2028	<1%	<1%	<1%	<1%	<1%	<1%	<1%
$P(F>F_{target})$	2025	57%	57%	57%	57%	57%	57%	57%
	2026	<1%	39%	50%	65%	65%	59%	68%
	2027	<1%	39%	50%	64%	64%	59%	68%
$P(B<B_{trigger})$	2025	3%	3%	3%	3%	3%	3%	3%
	2026	4%	4%	4%	4%	4%	4%	4%
	2027	<1%	4%	5%	6%	6%	6%	7%
	2028	<1%	4%	6%	8%	8%	7%	9%
$P(B_{2028}>B_{2025})$		96%	52%	42%	30%	29%	34%	27%
$(B_{2028}-B_{2025})/B_{2025}$		96%	1%	-12%	-31%	-32%	-23%	-37%
$P(B_{2027}>=B_{2025})$		89%	48%	42%	33%	33%	36%	31%
$P(B_{2027}<B_{2025})$		12%	52%	59%	67%	67%	63%	69%

ᖃᐅᔨᓴᖅᑕᐅᓂᖅᓴᒃ

ᐊᑦᖁᑭᓱᐊᔨᔪᐃᑦ ᑲᔪᓯᖏᓪᓕᐊᖅᓯᖅᓄᓇᖅᑐᒍᓂ (SPiCT) ᐊᑐᖅᑕᐅᓯᒪᔪᖅ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᓗᑎᓂ ᒡᒃᑯᐊ ᐱᑕᖃᕐᓂᐊᕐᓇᖅᓯᓂ. ᐱᖓᐊᕐᓇᓂ ᑕᒪᑐᒃᖠᖅ ᐱᓕᕆᔨᒃᑯᑦ ᐱᖃᓗᐊᖅᑐᑦ ᐱᖃᓄᐃᖅᑲᓇᔨᓂ ᓄᓇᕗᒻᒥᔨᖅ ᐊᒻᒪᓗ ᐊᑐᖅᑕᐅᖅᓯᒪᔪᑐᒍᓂ ᓇᓗᓇᐃᔨᐅᑦᑐᓗ ᑎᑎᖅᑲᓇᕐᓂᖅ ᐱᖓᓱᖏᑦ ᐱᑕᖃᕐᓂᐊᕐᓇᖅᓯᒪᔪᓂ ᓇᓗᒥᒃᓯᖅᑐᒍᓂ ᓂᕿᔪᒪᓂᖅᓯᒪᔪᑐᒍᓂ ᐅᑎᕐᓂᐊᕐᓇᖅᑐᒍᓂ ᐃᓄᐃᑦᑐᑦ ᓄᓇᕗᑦᑐᒥᓂ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᓗᑎᓂ ᑲᑎᒪᔨᒍ ᐊᖏᔪᓕᐅᕆᔨᓇᖅᑐᓄᓕᓄ ᓄᓇᕗᑦᑐᒥᓂ.

ᑭᖑᓪᓕᒥᓂ ᖃᐅᔨᓴᖅᓂᐅᓪᓚᕆᖅᑐᖅ ᓂᓚᐅᑎᕐᔪᐊᖅᓯᖅ 2027.

ᐃᓇᓐᓂᓂ ᐊᔪᑐᖅᑕᐅᔪᓂ

ᐃᓕᖅᑐᓪᓕᓂᐅᓂᐅᑎᔪᓪᑐᓂ ᐊᔪᑐᐊᓂᓪᓕᓂ ᑐᖃᐅᖅᑎᑦᑕᖅᓯᒪᔪᓂ ᑎᑎᕆᖅᑕᐅᓯᒪᔪᑎᑦᓪᓗᑎᓂ. ᐊᓯᖏᓂ ᓇᐱᓐᓂᓴᓕᖅᖢᓂ (ᒧᐃᖅᓗ. ᓯᕐᓐᓇᖅᑐᓂ, ᐅᒥᐊᖅᐊᓪᓚᕆᓂᖅ, ᐅᖅᑯᓯᐊᑐᓴᓕᐅᓂᖅ) ᑎᑎᕆᖅᑕᐅᓯᒪᔨᖏᑦᑐᓂ.

ᐅᒪᒪᓂ ᖃᐅᔨᓴᖅᓂᖅ ᐊᒻᒪ ᐊᐅᖓᓯᓄᓂ ᖃᓄᐃᓕᑐᖅᑕᐅᑎᓂᖅᓯᒪᓂ

ᓇᓗᓇᐃᖅᑲᓱᒪᔪᓂ ᖃᐅᔨᓴᖅᓂᐅᔪᓂ ᖃᑎᒌᕐᕕᐅᔪᖅᑲᓇᐅᖏᑦᑐᖅ ᑕᖅᐅᖅᖃᓂ ᖃᐅᔨᓴᖅᓂᐅᔪᓕᒫᓂ.

ᐅᒪᓂᖅᖁᑦᑐᓂ ᓇᔪᑎᒥᓂ ᑲᔪᓯᓐᓇᕐᓇᖅᑐᐅᓂ ᐊᖏᓂᑕᐅᔪᓂ

ᐊᐅᑐᐃᓂ ᐊᑎᑎᓪᓚᑦ ᐃᖅᑲᓗᓯᓄᖅᓂᒥᑦ ᑕᒪᑐᒍᓐᓄᑦ VME-ᓂᑦ SA 0 ᖃᐅᔨᓴᖅᑕᐅᓯᒪᔪᖅ 2016-ᒥ. ᐱᖓᓱᓂᑦ ᐃᓂᑦ ᑎᑯᐊᖅᑕᐅᓯᒪᔪᓂ ᑕᒪᐅᒥ ᖃᓄᐃᓕᑐᖅᑕᐅᑎᓪᓗᑎᓂ ᐃᖅᑲᓇᐃᔨᖅᑕᐅᑎᓂᖅᓯᒪᓂ ᐃᖅᑲᓇᐅᓂᒥᓂ: ᑎᑦᑯ ᖁᓪ, ᑕᐃᒪᓕᑦ ᐃᑭᖓᒃ, ᑎᒪᔨ

Hᐊᑕᵒ ᑲᖏᑯᓂᐊ.ᑕᑦᓗᑭᑎᐅᐸ ᐃᒪᖏᓱᓂᐦ ᐃᓂᑦ SA 1-ᒥ ᑐᒪᓴᐅᓯᒪᓴᑦ ᐃᑯᑲᖏᓂᓯᖏᓂᖅᓭᒧᑦ ᓴᐳᓴᐅᓯᒪᔪᓂᓗᑦ ᐃᑯᑲᖏᓂᓱᓂ

2026 ᐊᓱᒪ 2027 ᐊᑯᑭᑐᓂ ᖃᓕᕆᓂᑦ (GHL) ᑲᑎᒪᑎ ᐊᖏᓂᖅᑕᐅᔪᓐᓇᖅᑐᑦ (TAC)
ᑲᓇᖅᓇᖏᓂ ᐃᖅᑲᖅᑎᐅᑕᓕᕆᓂᖅᓯᑦ ᖃᐅᔨᒪᓕᕆᓚᐅᔪᑦ ᑲᑎᒪᓕᕆᓂᑦᓂ (EAGSAC)
ᐃᓕᑭᐅᔪᑐᓂ ᐃᓱᒪᑭᐅᔪᑦ
ᑲᑎᒪᓂᕆᓂᑦ ᐅᖅᑲᖅᑎᑎᑦᒍᑕᐅᔪᑦ ᓇᐃᓇᓂᒃᓯᒪᓕᖅ

ᐊᒃᓱᖏᐅᑕᖅ – ᐃᓕᑦ ᐃᒡᓗᓴ, ᐊᐃᒃᑐᖅᓱᒪᔪᓂ ᐃᖅᑲᓕᕆᓚᐅᖅᓯᑦ ᐊᐅᓕᕆᑎᓕᒪᓕᒃ (FM), ᐅᑭᐅᖅᑕᖅᑐᒥ ᐊᐃᒃᑐᖅᓱᒪᔪᓂ (AR), ᐃᒪᖅᒥᐅᑕᓕᕆᔨᒃᑯᑦ ᑲᓇᑕᒥ (DFO)

ᑲᓇᖅᓇᖏᓂ ᐃᖅᑲᖅᑎᐅᑕᓕᕆᓚᐅᖅᓯᑦ ᖃᐅᔨᒪᓕᕆᓚᐅᔪᑦ ᑲᑎᒪᓕᕆᓂᕐᓂ ᐃᓕᑭᐅᔪᑦᓂ ᐃᓕᑭᐅᖅᑲᑕᐅᔪᑦ

ᑭᓇᐅᔭᑕᐅᕆᔨᒃᓚᖅᓯᑦ ᐃᖅᑲᓕᕆᓚᐅᖅᓯᑦ ᑲᑐᔨᖅᑲᑎᑭᓐᓂᑦ:

ᓄᓇᕗᒻᒥ ᐃᖅᑲᓕᕆᓚᐅᖅᓯᑦ ᖃᑐᔨᖅᑲᑎᑭᑦ (NFA) – ᑎᓪᓗᒃ ᖁᑦᓗᐊᖅ, ᑭᓯᖅᑕᐃᔪᖅ:

- ᐅᑭᖅᑕᖅᑐᒥ ᐃᖅᑲᓕᓯᐊᕆᓚᐅᖅᓯᑦ ᑲᑐᔨᖅᑲᑎᑭᑦ (AFA)
- ᖃᑭᖅᑕᒋᓐᖏᒻ ᐃᖅᑲᓕᕆᓚᔨᑦ (BFC)
- ᑎᓄᔨᖏᕆᒻ ᐃᖅᑲᓕᕆᓚᔨᒃᑯᑦ ᑭᓯᕆᕆᓯᒪᔪᑦ (CSFL)
- ᖃᑭᖅᑕᒋᓐᓄᒃ ᑯᐊᐳᓚᓴᓗ (QC)

- ᐅᑭᐅᑕᖅᑐᒥ ᑲᑐᔨᖅᑲᑎᑭᑦ (NC) – ᐊᑦᓕᔭᑎᐅ ᐅᕆᐃᓕᒪ, ᑭᓯᖅᑕᐃᔪᖅ:
  - ᒪᑭᕐᕕᒃ
  - ᒋᐸᑐᐊ ᐃᖅᑲᓯᓯᐊᖅᑭᑦ ᑲᑐᔨᖅᑲᑎᑭᕐᓂᑦ ᑭᖓᔪᕐᓕᕆᓚᐅᖅᓯᒪᓕᑦ ᑎᒥᖓᑦ (Labrador Fishermen’s Union Shrimp Company Ltd.) (LFUSC)
  - ᔉᖅᖏᓚᐃᑦ ᐃᖅᑲᑐᓕᕆᔨᒃᑯᑦ ᑯᐊᐸᒃᑯᑦ ᑎᒥᖓᑦ (Torngat Fish Producers Co-operative Society Ltd.)
  - ᓄᓇᓕᔪᐊᕐᓂ ᑎᒥᖅᑯᑎᕐᓂᑦ (Nunatsiavut Group of Companies)
  - ᖃᑭᖅᑕᒋᓐᓄᒃ ᑯᐊᐳᓚᓴᓗ (QC)
- ᐊᑦᓕᓐᓂᒃ ᔪᕆᐅᓕᕐᓗᓄ ᑲᑎᒪᔨᓂ (AGC) – ᔪᑎᓕᒪ ᑎᓕᑦ, ᑭᓯᖅᑕᐃᓪᓕᑐᓂ:
  - ᕼᐊᐸᐳ ᔪᕆᐃᓄ ᑭᖓᔪᐸᓕᕆᓚᐅᖅᓯᒪᓕᑦ ᑎᒥᖓᑦ (Harbour Grace Shrimp Co. Ltd)
  - ᐅᓴᓐ ᔪᐃᓄ ᐃᓐᑐᐃᓴᓄ (Ocean Choice International)
  - ᓅᖏᐅᓐᓂᓐ ᐱᓕᕆᖓᐃᔾᔨᓂᑦ ᐱᖏᑦᓕᐊᔪᓕᕆᓚᐅᖅᓯᒪᓕᑦ ᑯᐊᐸᓴᓇᖓᑦ/ᐊᐃᓄᐅᐊᑐ (Newfoundland Industrial Development Corporation/Icewater)
  - ᑭᓪᓐᐅᐊᑕ ᑕᓚᐅᕆᐅᑕᐃᑦ ᓂᕆᑭᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd) (CW)
  - ᓄᐊᑎᒃ ᑎᒥᖓᑦ (Nordic Ltd.)

ᐃᓕᐅᔪᑦ

- AFA – ᕼᐃᐅᓚ ᐅᑭᑦ, ᔪᐃᓕᓂ ᖃᒪᓂᖅ, ᓴᑎ ᑐᒪᔪ
- LFUSC – ᑕᒪ ᐱᐅᓚ
- ᑐᕆᐃᓴᐟ (TriNav) ᐃᖅᑲᓕᕆᓚᐅᖅᓯᒪᓕᑦ ᑐᓴᕆᔪᐊᖅᑭᑦ ᑭᓯᖅᑕᐃᔪᑦ AFA-ᒥᒃ – ᑕᐃᐟ ᐅᓕᐟ
- BFC – ᔪᓚᔨ ᐸᓴᓂᑲᓐ, ᐱᑕ ᐊᑕᒪᔨ, ᐸᑦ ᓴᑐ
- QC – ᐱᑕ ᑭᓴᐃᓐᓴᖅ, ᔭᐊᓚ ᐅᐊᑦ
- ᒪᑭᕐᕕᒃ – ᑐᓂ ᕿᐃᑦ
- ᓄᓇᕗᒻᒥ ᑲᑎᓐᓄᑐᖅᑐᑦ ᑲᒪᐸᓂᐅᑦ – ᐃᐅᕿᓐ ᐊᐅᕿᑦ
- ᕼᐊᐸᐳ ᔪᕆᐃᓄ ᑭᖓᔪᐸᓕᕆᓚᐅᖅᓯᒪᓕᑦ ᑎᒥᖓᑦ (Harbour Grace Shrimp Co. Ltd.) – ᑎᐅᓚᒃ ᒋᑕ
- ᑭᓪᓐᐅᐊᑕ ᑕᓚᐅᕆᐅᑕᐃᑦ ᓂᕆᑭᑦ ᑎᒥᖓᑦ (Clearwater Seafoods Ltd) – ᑲᑐᓚᓐ >ᐃᑦ

ᐊᐅᓪᓗᖅᑎᑦᑕᕐᓂᖅ ᑲᑐᔾᔨᖅᑎᑎᑦ

- ᓄᓇᕗᒻᒥ ᐅᒪᔨᓪᓚᕆᔨᕐᐊᖅᑯᑦ ᑲᑎᒪᔨᖏᑦ (NWMB) – ᓂᓪ ᑲᓪᑦᖏᓴᐅ

ᐊᖏᑎᓪᓚᕐᓂᖅ ᑲᖏᒪᐅᖕᖏᑦᑐᓂ ᑲᑐᔾᔨᖅᑎᑎᑦᓴᒻᒪᔨᓄᑦ

- ᐃᐊᒐᔪ ᑐᓄᕆᐊᖅ – >ᓚᐤ ᑕᕆᐤ

ᑲᖏᒪᔪᑯᑦ ᑲᑐᔾᔨᖅᑎᑎᑦᓴᖏᑦ

- ᑲᖏᒪᖏᑦ ᓂᐅᕐᓴᐅᓕᕐᒥ ᓕᓴᐃᑐᑐᓕᓪᓗ – ᓴᐊᒋᑕ ᑯᓘᓴ
- ᑲᖏᒪᖏᑦ ᓄᓇᕗᒻᒥ – ᐋᒃ ᐊᑎᐤ, ᑕᓕᐊ ᔭᖅ

ᐃᒪᖃᐅᑕᓕᓚᔪᑯᓂᔪᓄ ᑭᖑᖏᑐᐃᓐᓇᑦ

- ᑭᖑᓂᑦ ᓂᐅᕐᓴᐅᓕᕐᒥ ᓕᐸᑐᐃᓕᑦᓗ – ᓴᐊᒋᑕ ᑯᓘᓴ
- ᑲᖏᒪᖏᑦ ᓄᓇᕗᒻᒥ – ᐋᒃ ᐊᑎᐤ, ᑕᓕᐊ ᔭᖅ

ᐃᒪᖃᐅᑕᓕᓚᔪᑯᓂᔪᓄ ᑭᖑᖏᑐᐃᓐᓇᑦ

- ᑭᖑᓂᓕᕐᓂᖅ – AR
  - ᔪᔪᑕᖏᖅ ᑭᐃᐤ ᐄᐃᔨᔪ, ᑐᖏᐃᓕᓕ ᔪᐤᑎᐅᑦ, ᐊᔪᖏᓐᓇᐤ, ᓕᓪᓕ ᐋᑦᑭᓐᓇᐤ
- ᓄᓇᖏᐊᒥ ᐃᖏᓴᑐᓕᓪᓚᕐᓂᖅ ᐊᑐᐊᒐᖏᖅ
  - ᔭᕐᑐᓇ ᔭᒐᒻᑎᑦ
- ᐃᖏᓴᑐᓗᓕᓂᔪᓄ ᐊᐅᓪᓕᑕᐅᓂᖅᑎᑦ
  - ᐊᐤᑐᑐᖏ ᒧᐊ AR, ᔨᕽ ᐋᑕᒪ AR, ᑲᐃᑦᓂᐤ ᒪᐃᓴ AR, ᔨᐊᓓᑲ ᑎᐱᖏᐃᕽ AR, ᖏᐱᐤ ᓂ (ᓂᐅᐃᐤᓀᐤ [NL]), ᖏᐱᑦ ᒧᓚᒻ NL
- ᐊᑐᐊᒐᖏᓄᑦ ᐊᒪᒪ ᒪᑭᒪᔭᓇᓓᓂᑦ (P&E) – ᐋᐤᑎᐅᓚᐅ ᐊᒪᒪ ᓄᓇᑭᑐᑦ (OP)
  - ᔨᒻ ᕼᐊᐅᓚ, ᑲᓕᐤ ᑕᐃᔨᑦ, ᐃᐅ ᓕᑭ
- ᓴᑕᐃᑦᑕᐃᓕᓕᖏᓴᑦ ᐊᒪᒪ ᖃᐅᔨᓇᓕᓕᖏᓴᑦ
  - ᑕᐃᒻᖏᐤ ᖏᓇ AR, ᕿᐊᓚ ᒪᖏᑉᓄᑦ AR, ᑎᐊᐤ >ᖏᐅᐤ (ᓄᓇᖏᔨᐊᓕᒥᒪᓗᑦ)
- ᑎᑎᖕᑲᓂᖅ ᐊᐅᓕᖏᑎᓂᖕᖏ ᐊᒪᒪ ᓕᐃᒻᐤᓇᓕᓚᖕᖏ AR
  - ᖏᔭᔨ ᕼᐊᐤᒻᒻ

ᑲᑎᒪᓂᐅᐸ ᐱᔨᔪᑎᖏᓚ:

ᑑᑲᔪᓄᖏᓚ EAGSAC ᐃᓕᓕᑎᐅᔨᓄᑦ ᑐᒻᖏᑲᑎᑭᔪᑎᖏᑦᓴᓕᑦ ᐃᓇᒪᑕᑎᖏᓐᓂᐤᑲ ᐅᐊᖏᓐᓄᐤᓂᓂ ᑕᐊᖏᓐᓇᑦ ᐃᖏᓴᑐᓕᓪᓚᕐᓂᖅ ᑲᑐᔾᔨᖅᑎᑎᑦᓴᒻᒪᔨᓄᑦ (NAFO) ᐃᓂᐲᑲᓂᓂᐅᔭᖏᐤ 0 ᐊᔪᑭᑦᑐᓂ ᖏᑦᕋᓕᑉᑦ (GHL) ᑲᑎᓐᓕᑦ ᐊᖓᔪᑦᑕᐅᔨᓇᓐᓄᖏᑦ (TAC) ᖏᑲᓄᑎᐅᓂᖏᑦ 2026 ᐊᒪᒪᓗ 2027-ᒧᑦ. ᑲᑎᒪᓂᕆᒥ ᐱᐚᕆᖏᑎᑎᐱᓇᓐᓄᓕᐅᐅᖏᖅᑐᑦ EAGSAC-ᔪᓇᓂᑦ ᐃᓕᑎᔭᐅᔨᑦ ᐱᐚᖏᑦᑕᐅᔨᓇᓐᓄᓕᐅᐅᖏᖅᑐᑎᓐ ᐃᒪᖃᐅᑕᓕᓚᔪᑯᓂ ᑭᖑᓂᔪᓄᖏᓚᖓᓇᓂᓂ ᐊᐱᖏᔪᑎᓐ ᐊᐱᖏᔪᑎᓐ ᑭᖑᓂᔪᓄᖏᓚᖓᓇᓂᓂ ᑲᑎᒪᓂᐅᐸ ᑐᒻᖏᑲᑎᑭᔪᑎᖏᑦᓴᓕᑦ ᐅᖏᐅᔭᓕᐊᑭᑎᓐ ᑐᒻᖏᑲᑎᑭᔪᑎᖏᑦᓴᒻᒪᔨᓄᑦ (NAFO) ᑭᖑᓂᔪᓄᖏᓐᓂᖏᑦᓴᓕᑦ ᑲᑎᒪᔨᓄᑦ (SC) ᑕᒪᑐᒧᖓ SA0+1 (ᐃᒪᐱᖓᒥ) ᐊᔪᑭᑦᑐᓂ ᖏᑦᕋᓕᑉᑦ (GHL) 2026-ᒥ ᐊᒪᒪ 2027-ᒥ.

ᐅᖏᑲᐅᔪᐅᑐᓂᓄ:

ᑭᖑᓂᔪᓄᖏᓚᕆᒪᔨᑦ ᐅᖏᑲᑎᔭᓕᐊᑭᑎᓴᑦ

ᐃᒪᖃᐅᑕᓕᓚᔪᑯᓂ ᑲᓇᑕᒥ (DFO) ᑭᖑᓂᔪᓄᖏᓚᖓᑦ (ᑭᐱᖓ ᕼᐃᔨᔪ) ᑭᐅᑕᐅᖏᑐᖏᖅ ᓂᖏᑯᐃᓇᖏᓚᖓᓂᑦᓗᓐ ᐱᖏᖏᑲᑕᐅᔨᑦ ᐊᐱᖏᑯᐃᓐᓇᑦ ᐅᖏᑲᐅᓂᓚᖏᑦᑕᓐᓂᑦᓗᓐ ᐱᔨᔪᑎᑦᑕᓂᓐᓂᑦᓗᓐ ᐅᐊᖏᓐᓄᐤᓂᓂ ᐊᐅᓪᓕᑕᐅᓂᖅᑎᑦ ᐃᖏᓴᑐᓕᓪᓚᕐᓂᖅ ᑲᑐᔾᔨᖅᑎᑎᑦᓴᒻᒪᔨᓄᑦ (NAFO) ᑭᖑᓂᔪᓄᖏᓐᓂᖏᑦᓴᓕᑦ ᑲᑎᒪᔨᓄᑦ (SC) ᐅᖏᑲᐅᔨᑎᐊᑭᑎᓐᓇᖏᓂᔪᓄᑦ.

ᐊᐱᖅᑯᑎᑦ	ᑭᐅᔭᓯᑎᑦ
ᐅᐊᖏᓇᖓᓂ ᐊᑦᓯᓂᐅᑎᒥ ᐃᖅᑲᓗᓪᓚᓂᖅᓴᓪᓗᑦ ᑲᑐᔪᖅᑎᑎᑦᓴᖓᔪᓄᑦ (NAFO) 2+3. ᓇᓗᓇᐃᑭᖏᓇᖅᐱᓯᐅᒃ ᑕᕆᔨᒃ ᐃᑎᔨᒧᐊᑭᖏᓇᖏᓕᖓᑦ ᐃᓂᓄᑦ ᐊᒥᓇᓂᓕᕆᔨᒃᑐᑦ ᖃᐅᔨᓂᓪᑕᐅᓂᖅᑐᓂᓄᑦ?	ᐊᑲᐅᖏᑦᑐᐅᕐᑎᑕᕆᑲᖏᑦᑐᖅ ᑕᕆᔨᖅ ᑕᐅᓇᖏᓴᓗᓂ ᐱᓕᕆᔨᖅᑕᐃᓂᓄᑦ ᐊᓯᖏᓪᓗᓂ ᐃᖅᑲᓗᐃᑦ ᐊᒻᒪ ᒪᑎᑦᑕᐅᑎᖓᑦ ᐃᖅᑲᖖᖓᓂᖏᓪᓕᓗᑎᒃ. ᐅᒥᐊᕐᕕᐊᖅ ᐱᓕᓚᑎᐊᓕᒡᓗᓇᓱᒧᒪᕐᕉᖅ.
ᐊᓕᐊᐳ ᖃᐅᔨᓴᕐᓂᕐᒥ ᓇᓗᓇᐃᕿᑎᑦ ᐅᕐᖤᓕᓂᓐᓂᑦ ᖃᐅᔨᓴᕿᑎᑦ ᐃᓕᓴᖅᑲᖅᑕᐅᓯᖓᖤᑦ ᐃᓐᓇᕿᖅᓯᒪᙳᕐᑕᓂᑐᓂᖅ ᐱᓇᓱᐊᕐᓂᕐᒧᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᑦ?	ᒪᓐᓇ ᐱᓇᓱᐊᕐᓂᕐᒧᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᑦ ᐱᓕᐅᓯᑦ ᐊᑯᐳᑕᓂ ᑎᕋᖅᑑᑎᑦ ᐃᖅᑲᓗᐃᑦ ᐊᒻᒪ ᑭᖓᓴᐅᓂᖅ ᖃᐅᔨᓴᕐᓂᕐᒥᑦ ᑕᐃᑲᖅᑕᐅᓯᒪᔪᑦ 1C-ᒥᑦ ᐊᒻᒪ 1D-ᒥᑦ. ᐊᑐᖅᑕᐅᖏᓪᓗᓂᖅᐸᖅᓯᑎᑲᓂᖅ ᑕᕆᔨᖅ ᐊᒻᒪ <ᑲᓪᐅ ᑲᓕᑕᐅᓯᑦ ᐃᓚᖃᐅᑕᓪᓚᕕᖅᑯᓂᑦ ᐃᓕᑎᐅᔪᓂᖅᓴᓂᑦ 200-ᒥᑦ 400-ᒧᑦ ᑭᒡᓘᓂᑦ ᐃᑎᓕᓵᓂ ᐱᓕᕆᖏᐃᐅᕐᖤᖅᑐᒥ. ᑐᑯᓯᒪᓕᖅᑎᒧᑦ ᐊᒥᓯᓂᖅᓴᓂ ᑎᑎᕋᖅᑐᓂᖅ ᐃᖅᑲᓗᖅᓂᑦ ᐊᒻᒪ ᑭᑲᓇᖅᑐᑦᑕᐅᑎᓂᖅᓴᓂᖅᓴᐅᓕᖅᑐᑎᒃ ᑎᑎᕋᖅᑐᓂ ᐊᑯᐳᑕᓂ ᖃᐅᔨᓴᕐᓂᕐᒥᑦ (GHL) ᑲᓇᑕᐅᕐᑦ ᐃᒪᖅᓂᓂ, ᑭᓯᐊᓂᓕ ᐱᖓᓱᓯᐃᓇᓕᓂ ᐊᖏᕐᕋᔪᓂ ᖃᐅᔨᓴᕿᑎᑦᑎᐊᖅᑐᑦ. ᑲᑎᑕᐅᓂᖅ ᖃᐅᔨᓴᕿᑎᑦ ᑎᑎᕋᖅᑐᓂᓄᑦ ᐊᑯᐳᑕᓂ ᖃᐅᔨᓴᕐᓂᕐᒥᑦ (GHL) ᑲᓇᑕᐅᕐᑦ ᐃᒪᖅᑎᓂᓂ ᐊᑐᖅᑕᐅᓯᐅᒪᔭᖅᑎᑦ ᓯᓚ, ᑭᓯᐊᓂᓕ ᐱᓂᐊᖏᓂᖃᖅᑭᒪᓂᖅ ᐊᑐᓕᔪᓂᓕᕆᓂᖅ ᓯᕗᓂᖃᖅᑎ. ᐊᓐᓇᐅᓂᖅ, ᐊᑯᐳᑕᓂ ᑎᑎᕋᖅᑐᓂ ᖃᑎᖃᓗᐃᑦ ᐊᒻᒪ ᑭᖓᓴᐅᓂᖅᓴᓂᖅ ᖃᐅᔨᓴᕿᑎᑦ ᐊᑐᐃᓐᓇᐅᑎᕐᓂᕐᓂᑦ ᐅᑭᐅᖏᓂ 1 ᐱᓇᓱᐊᕐᓂᕐᒥᑦ ᑎᑎᕋᖅᑕᐅᓯᒪᔪᓂᓴ.
ᑕᒪᐃᓇ ᐃᖅᑲᓂᑐᑐᐊᖓᔪᒃ, ᐅᕐᖤᓕᓂᓐᓂᑦ ᐱᖅᑲᓯᐅᔨᔪᒃ ᐃᒪᓕᐃᕐᓂᑕᐅᖅ?	ᐃ, ᑕᒪᐃᓇ ᐱᖅᑲᓯᐅᔨᔪᖅ ᐃᒪᕐᕆᑦ.
ᑭᓯᓂᖅ ᓯᑕᓂᕆᐃᓴᐸᑎᖅᑲᖅᓯᑦ ᑕᐅᔪᓇᖖᓗ ᐅᐊᖏᓇᖓᓂ ᐊᑦᓯᓂᐅᑎᒥ ᐃᖅᑲᓗᓪᓚᓂᖅᓴᓪᓗᑦ ᑲᑐᔪᖅᑎᑎᑦᓴᖓᔪᓄᑦ (NAFO) ᑭᖑᓪᓕᔭᓂᓪᓗᑦ ᑲᑎᒪᔨᖏᑐᓂ (SC) ᐊᑐᓕᔪᓂᓪᓗᑦ ᑲᑎᒪᔨᖏᑐᓂ "ᖃᐅᔨᓴᒪᓕᐅᕐᐸᑐᓇᓂᓴᓂ ᖃᐅᔨᓴᕿᑎᑦ" ᐊᑐᓕᔪᓂᐅᕐᐸᓂᖅ *ᐅᓂᒃᑰᒥ ᑭᖑᓪᓕᔭᓂᓪᓗᑦ ᑲᑎᒪᔨᑦ ᑲᑎᒪᔨᖏᑐᓂᓪᓗ, 29 ᒪᐃ – 12 ᓴᒃᑐ 2025* ᑕᔪᐅᕐᐸᓂᖅᓴᓂ [ᐅᓱᓇ](#) (ᒪᐅᐱᓚᖅ 53). "F target" ᐊᑐᓕᔪᓂᐅᕐᖤᖅᑎᑦᑕᖅ, ᐊᑲᑭᖤᖓᐊᖅᓯᒪᔪᖅ ᐊᑐᖅᑕᐅᕐᐸᓂᖅᓴᔪᑕᒍ ᐅᐊᖏᓇᖓᓂ ᐊᑦᓯᓂᐅᑎᒥ ᐃᖅᑲᓗᓪᓚᓂᖅᓴᓪᓗᑦ ᑲᑐᔪᖅᑎᑎᑦᓴᖓᔪᓄᑦ (NAFO) ᐅᓴᓯᕋᓯᐊᖅᓴᓪᓗᑦ ᐊᑐᕋᓯᐊᖅᑕᖏᑐᓂᓴᓂ (PA) ᐊᑲᑭᑉᓯᒪᓂᐅᕐᐸᓂᑦ ᓴᖅᑭᑲᖅᑕᕆᑎᖅᓴᖅ ᐃᖅᑲᓗᓯᓯᒋᓐᓂᑦ ᖃᑲᓇᐃᑎᑐᑎᒃᓴᖅᑐᒃ ᐊᑲᑐᐃᓐᓇᐅᑎᕐᓂᕐᓂᑦ ᓯᕗᓂᕐᑯᓇᑕᒍ. ᑭᖑᓪᓕᔭᓂᓪᓗᑦ ᐅᖅᑲᐅᓴᓕᐊᕐᑕᐃᓂᑦ ᐊᑕᖏᓂᖅᓯᒪᔪᓂᑦ ᓂᕆᔨᖅᑲᓯᐅᔨᓪᓗᑦ/ ᐅᓗᓚᐊᓇᕐᓇᖅᑐᓗᓂᑦ ᑎᑎᕋᖅᑲᓯᒪᔪᑦ, ᓇᓗᓇᐃᓴᐃᓯᒪᖏᑕᒪᓚᑦ ᐅᕙᓚᐊᓇᕐᓇᖅᑐᓗᓂᑦ ᐊᒻᒪ ᐊᑐᖅᑕᐅᕐᐸᓂᖅᓴᓪᓗᑦ (TAC). ᐊᑯᓂᓂᖅᑲᑐᑉᐊᑯᓂᐅᑕᓂᑦ ᑐᕆᓚᐅᔨᑦ ᑎᑎᕋᖅᑲᓯᒪᔪᑦ (ᒪᐅᐱᓚᖅ 56) ᐱᔨᓇᓇᖅᑎᑎᕋᖅᓴᖅ ᐃᖅᑲᓗᓪᓚᓂᖅᓴᓪᓗᑦ ᐊᐅᓪᓚᕆᐊᐅᔨᓂᖅ ᐅᓗᓚᐊᓇᕐᓇᖅᑐᓗᓂᑦᑐᑭᓯᐊᓕᓗᑎᒃ ᐃᓯᒪᓕᐅᕐᓇᓗᑎᒃ.
ᐅᖅᑲᓕᒻᒥᓕᕐᒪᙱᑦ ᑎᑎᖅᑲᒥᐅ ᐃᓯᒪᔨᓇᖅᑐᓂᑎᑦ ᑕᒪᐃᓇ ᐊᑐᓕᔪᓂᐅᕐᐸᓂᖅᓴᓪᓗᑦ ᖃᓄᐅᔪᖅᑎᑎᑦᓴᖓᔪᓄᑦ ᓂᕆᔨᖅᑲᓯᐅᔨᓪᓗᑦ/ᐅᓗᓚᐊᓇᕐᓇᖅᑐᓗᓂᑦ ᓇᓗᓇᐃᓴᐃᓯᒪᖏᑕᒪᓚᑦ ᐊᑐᖅᑕᐅᕐᐸᓂᖅᓴᓪᓗᑦ	ᑕᒪᐃᓇ ᑕᐃᒪᐃᖓᓕᖅᑐᓂᖅ. ᐅᐊᖏᓇᖓᓂ ᐊᑦᓯᓂᐅᑎᒥ ᐃᖅᑲᓗᓪᓚᓂᖅᓴᓪᓗᑦ ᑲᑐᔪᖅᑎᑎᑦᓴᖓᔪᓄᑦ (NAFO)

ᐊᑐᕐᓂᐊᖅᑐᓂ ᒪᕐᑭᖅᓂ ᐊᕐᕌᔪᓯᓂ.	ᑭᓴᓪᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᖅᑎᑦ (SC) ᐊᐱᓚᓪᒪᓴᐅ ᐱᓕᕐᓕᐊᑎᑕᐅᕐᕋᑦᓗᓕᓂᑦ ᓇᒋᑦ ᐅᔨᔪᓪᓕᐊᓂᕐᒧᑦ ᐊᑐᕋᓪᐊᖅᑕᖅᑎᓐᓄᑦ (PA) ᐃᖅᑭᖅᓪᒪᓂᐅᓴᐅᓴᐅᐸᑦ ᓇᓗᓇᐃᔭᐃᓴᐅᑕᖅᑕᒻᒪᒪᑦ. ᐅᐊᖅᓇᖅᓴᓂ ᐊᑦᓭᓄᑎᒥ ᐃᕐᑲᓗᓕᓂᕐᒧᑦ ᑲᑐᔪᓂᑲᑎᑦᓴᓂᑦ (NAFO) ᑭᓴᓪᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᓯᑦ (SC) ᑕᑯᓄᖅᓴᒪᑦᑕᕐᑲᖅ ᐊᑐᖅᑕᐅᓪᒪᓴᐅ ᐅᕐᖅᐅᓪᓕᖅᑕᒃ ᐊᑐᓕᕐᓄᔨᓴᐅᑎᒥ. ᐅᕐᖅᐅᔨᔪᑎᐊᓕᕐᓂᒧᑦ ᐊᕐᐳᑐᖅᑕᐅᓴᐅᖅ F target ᒪᓕᒃᑐᑦ ᐊᖓᔪᓂᖅᑕᐅᓴᐅᑦ 30,000 ᑕᕐᓪᓂᒃ. ᐅᓄᐊ ᐱᓴᐅᓴᓄᑦ ᐊᖑᑎᓂᕐᖅᑕᐅᓴᐅᑦ ᖃᒃᖑᑎᐅᑎᓪᓕᒪᑐᐃᓐᓇᕐᓕᐊᓯᑦ 35%-ᒥ ᐊᓖᒪ 36%-ᒥ ᖃᒃᖑᑎᐅᑎᓪᓕᓂᐅᔪᑦ F lim ᐅᓂᒃᑲᖅᑕᕐᕆᕐᓂᑦ ᑕᑯᓄᐊ ᐅᐊᖅᓇᖅᓴᓂ ᐊᑦᓭᓄᑎᒥ ᐃᕐᑲᓗᓕᓂᕐᒧᑦ ᑲᑎᒪᔨᖅᑎᑦ (NAFO) ᑭᓴᓪᓂᐊᕐᓂᕐᒧᑦ ᑲᑎᒪᔨᖅᑎᑦ (SC) ᐊᑐᕐᓄᔨᓴᐅᑎᖅᑕᐅᓂᓂᒃ ᐃᕐᑲᒃᓯᓪᓂᕐᒧᑦ ᑕᐃᒪᓐᓇᑎᑎᑦ. ᓯᑕᒫᓐᓂᐅ ᐅᓂᒃᑲᖃᐃᓐᓇᓐᓇᖅᑐᑦ ᑭᓪᓗ ᓂᕿᑭᖅᓴᖅᑭᐅᓂᒧᑦ ᖃᓄᐅᑎᐅᓂᕐᒥᓂᒃ ᐊᒻᒪ ᐅᓂᒪᓐᓇᖅᑐᑦ ᐅᖃᓕᑕᐅᓐᓇᖅᑐᑦ ᐅᖅᑲᑕᐅᔪᓐᓇᖅᑐᑦ F lim ᐃᒪᐃᑐᐃᓐᓇᖅᓴᓂᕐᒥᓂᒃ F target.
ᓂᕿᑭᖅᓴᖅᑭᐅᓂᒧᑦ/ᐅᓂᕆᐊᓐᓇᖅᑐᓐᓇᓄᑦ ᐃᖅᑭᕐᖿᐃᖅᑕᐅᔪᓕᕐᓂᑦ ᑎᑎᕋᖅᑭᓕᕐᓂᑦ (ᐊᑯᓂᕋᖅᑭᓂᑦ-ᐊᑯᓂᐅᓂᕐᑎᑦ ᑐᕌᓗᕆᔨᐅᓕᕐᓂᑦ ᑎᑎᕋᖅᑭᓕᕐᓂᑦ), ᑭᑭᑎᑭᓕᖅᒪᒪᑕ P-ᒥᒃ ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᓕᒫᓂᒃ ᐱᖅᒪᔪᐊᓂᓂ, ᐊᒻᒪ ᑕᐃᒪᐃᖕᒥᑯᑯᓂ, ᐅᓂᒃᑲᑭᓐᓇᐃᐅᓂ ᑕᒪᓐᓇ?
ᑕᐃᒪᓐᓇᖅᑐᓂᕐᑎᑦ ᐱᑕᖅᑳᐃᐅᓕᕐᓂᒧᑦ ᐅᒪᓂᕐᒥᖅᑎᑦᓴᓂᑦ ᐅᓄᕋᓂᖅᓴᐃᓐᖓᑦ ᑕᑯᓛᑦᓕᓂᕐᑎᑦ ᑎᕋᑭᑐᑕᐅᓂᖅᓴᓂ ᑎᑭᑎᑦᓴᐅᑎᑦ?
ᑕᐃᒪᓐᓇᖅᑐᓂᕐᑎᑦ ᐱᑕᖅᓯᓂᕐᒧᑦ ᑭᓴᓪᓂᐊᕐᓂᑦ ᑕᒪᑐᐃᓐᓇᖅᑐᓂᕐᒥᓂᒃ ᐅᓄᕋᓂᖅᓴᐃᓐᖓᑦ ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᓕᒫᓂᒃ ᐊᑎᖅᑕᐅᓂᕐᒥᓂᒃ ᐱᖅᐳᑎᓪᓕᐊᓂᕐᒥᓂᒃ ᑎᑎᕋᖅᑭᓕᕐᓂᑦ ᐃᓕᓴᐃᓐᓇᖅᑐᑦ ᐊᑐᖅᑕᐅᓂᖅᓴᓂ ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᓕᒫᓂᒃ ᐊᑎᖅᑕᐅᓂᕐᒥᓂᒃ ᐱᖅᐳᑎᓪᓕᐊᓂᕐᒥᓂᒃ ᑎᑎᕋᖅᑭᓕᕐᓂᑦ ᐊᑐᖅᑕᐅᓂᖅᓴᓂ ᑕᐃᒪᐃᑐᐃᓐᓇᕆᐊᓕᒫ

	ᐃᖃᓗᓕᓴᐅᓂᖏᑦ ᑐᖁᖃᑦᑕᖅᑐᓄᑦ – ᐊᐅᓚᓴᐅᓂᖅᓴᐅᔪᑎᖅ ᐱᑕᖃᕆᐊᐅᓪᓚᑦ ᐅᓄᕐᓂᖅᓯᓪᓕᐊᑎᓪᓗᑎᑦ, ᐅᓄᕐᓂᖅᓴᐅᔪᑎᖅ ᐱᕈᖅᓯᓪᓕᐊᔪᑎᖅ.
ᑭᓴᓂᒃ ᐁᖅᑭᖓᔪᐊᖅᑕᐅᓴᒪᓪᓚᑦ ᐃᓕᐃᐅᓇᓴᒪᓐᓇᑦ ᑎᑎᕋᐅᓂᖅ?	ᐃᐅᐱᐅᓂᖅᓴᕋᐃᐅᓂᖏᓪᓗᑦ ᖃᑎᖃᑭᓂᑦ ᑕᐃᑲᑭᓂᑦ ᐊᑕᐅᑎᖅᓴᓴᓂ ᐊᑦᓯᓚᐅᑎᒥ ᐃᖃᓗᓯᓚᕆᓂᖏᓪᓗᑦ ᑲᑎᕐᓂᖅᑎᑦᑎᓂᖅ (NAFO) ᖃᑎᖃᑭᓂᖏᓪᓗᑦ (ᓇᐃᐅᖅᓴᓴᒥ <ᐃᐢᐢᒋᖅᓯᑦᑐᑦ). 2024-ᒥ ᖃᑎᖃᑭᑭᓂ ᖃᑎᖃᑭᓴᓴᑦ ᐱᐅᑦᓚᐊᓴᓂᐅᐱᖅᓴᖅ SPiCT ᐁᖅᑭᖓᔪᐊᖅᑕᐅᓴᒪᓪᓚᑦ ᐊᑐᓚᖅᑕᐅᖅᑲᖏᑎᖅ. ᐊᖁᑦᖢᒋᒪᑦ ᓄᑖᖢᔪᐊᖅᑕᐅᔪᑭᓂᒃ ᖃᐅᔨᓴᖁᓂᖏᓪᓗᑦ ᐱᓚᓇᐅᑦᓚᐊᑭᓂ ᓄᑖᑭᓂᒃ ᖃᐅᔨᓴᕿᑎᓄᑦ ᑭᓴᐊᓂᓚ ᑐᖓᔪᓄᐅᓄᑦ ᖃᑎᖃᑭᓴᓂᖅ ᐱᓚᓇᐅᑦᓚᐊᑭᓂᖏᓪᓗᑦ.
ᑕᑯᓗ ᐁᖅᑭᖓᔪᐊᖅᓯᒪᔪᖅ ᑐᖓᑕᐅᑎᔪᖅ ᐊᓪᒪ ᐅᖓᓚᑕᐅᔨᓴᓚᓂ ᑐᖁᖅᑕᖅᑑᑐᓂ. ᖃᒃᓄᐃᓪᓚᑦ ᑕᐃᒪᐃᓕᓴᕐᓂᖅ ᐊᓪᒪ, ᖃᒃᓄᓕᓐ ᐊᖏᑎᑎᑭᓪ?	ᖃᒃᐅᔨᓴᕿᑎᓪᓗᑎᑦ ᓚᒪᑯᓗ ᐃᓚᐃᓂᖅᓴᖅᓯᒪᓂᖅᓴᒃ ᑕᖅᖢᓂ SPiCT ᖃᒃᓄᓕᐅᑐᖓᓕᒫᓂ ᐁᖅᑭᖓᔪᐊᖅᓯᓚᐅᓂ. ᑕᑯᓗ ᓇᓗᓇᐃᖅᑕᐅᑐᐃᓇᖅᓯᒪᔪᖅ ᐊᒥᐊᒃᓯᑦ ᐱᖁᓇᖅᑕᐅᓚᐅᖅᓯᒪᓂᖅ (SPiCT) ᐁᖅᑭᖓᔪᐊᖅᑕᐅᓯᒪᔪᖅ ᐅᓴᖅᓯᑕᐅᓇᓂᖅᓴᓪᓗᑦ ᐁᖅᑭᒃᑕᐅᓯᒪᔪᖅ, ᑭᓯᐊᓂᓕ ᐅᖓᓚᑕᐅᓴᓯᒪᓂᖅᓯᑦ ᐊᖓᑎᓲᖅᓯᑦ. ᑕᒪᒃᑯᐊ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓕᓪᑐᐊᖅᑐᑦ ᑐᑭᓯᐅᑕᑎᓪᓗᓂᑦ ᖃᒃᐅᔨᓴᐅᔪᓂᒃ.
ᑕᐃᒪᐃᑐᐃᓇᐅᑕᐊᓕᓂᖓᓂᒃ ᑭᖓᔪᓕᐅᔨᓴᓂ ᑎᑎᕋᖅᓯᒪᔪᓂᒃ (ᑕᐃᒪᐃᑐᐃᓇᐅᑕᐊᓕᒡᓕᑦ ᑕᐢᖁᓂ F > F lim), ᐃᓯᒪᑦᓕᓂᑦ 2022 ᑲᑎᒪᓂᑦ ᐊᖓᔪᓂᐅᑕᐅᔨᓴᖅᑐᑦ (TAC) ᖃᒃᓄᐃᑎᑎᐅᓂᖓᓂᑦ, ᐱᑕᖃᓕᐅᖅᑐᑦ 54%-ᒥᒃ ᑕᐃᒪᐃᑐᐃᓇᐅᑕᐊᓕᓂᖓᒥᒃ F-ᒥ (ᑐᖁᖃᑭᑕᖅᑐᑦ) ᐅᓄᕐᓂᖅᓴᐅᓂᐊᖅᑐᑦ F lim-ᒥᑦ. ᑭᒃᓕᒧᖅᓱᓂᖅ ᓯᓕᓂᒃ ᑕᒪᓐᓇ ᐱᑕᖃᓄᐅ 46%-ᒥᒃ ᑕᐃᒪᐃᑐᐃᓇᐅᑕᐊᓕᓂᒃ ᑕᑯᓗ F ᑐᖓᓚᓂᓇᓴᓂᖓᓂᑐᒃ F lim-ᒥᑦ? ᑕᒪᓐᓇ ᑕᓪᓕᖅᓯᒪᓕᕐᓂᓗᓂ ᑐᑭᓕᐊᖓᓯᒪᓂᒃ?	ᐃᓚ, ᓯᖅᑲᐃᓪᓕᓕ ᐊᐃᑲᓯᒪᔪᖅ ᑕᐃᒪᐃᑐᐃᓇᐅᑕᐊᓕᖅᓴᓂᖓᓚᓄᑦ ᐊᐃᑲᑐᖅᓯᒪᓂᖓᓂᑦ.
ᐱᓕᐅᑐᐊᖅᑕᓪᓕᐊᓂᖓᓯᒪᓂᒃ ᐊᖓᔪᓂᖅᓯᒪᔪᓂᒃ ᑐᖁᖅᑕᖅᑑᑐᓂ ᐃᓕᓴᕐᓂᖅᓯᒪᔪᓂᒃ ᐊᑐᖅᑐᑦ

ᓀ ᓴᓪᒋᖅᓴᓄ (NWMB)	ᐅᖃᐅᓯᐊᖃᕐᕆᔨᒃᑯᓐᓂᖅ ᐊᐱᖃᑯᑎᐊᖃᕐᕆᔨᒃᑯᓐᓂᑐ ᐃᓐᓇᐅᔪᖅ.
ᑯᓚᓴ <ᓭᓂᔪᓄ (BFC)	ᐅᖃᐅᓯᐊᖃᕐᕆᔨᒃᑯᓐᓂᖅ ᑕᒪᑐᒧᖅ ᑲᑎᓀᓂ ᐊᖏᔪᓂᐅᔪᓇᖅᑐᑦ (TAC) ᖃᓄᐃᑎᐅᓂᖅᓯᓄᓐᓄᑦ.
ᓴᓴᒥᓐᑕ ᑯᒪ (ᓚᖓᒪᖅᓯᓂ ᓂᐅᔨᓇᓕᖅᑐᓂ ᐅᓂᒡᓗᒍ ᓭᒪᓐᓇ ᓭᓐᓄᑦ)	ᐊᐱᖃᑯᑎᑕᕆᔨᒃᑯᓐᓂᖅ; ᑐᓂᔪᓭᖅᑐᑦ ᑎᑎᕋᖅᓯᓪᓚᔪᓂᒃ ᐅᖃᐅᓯᕆᓂᓄᑦ.
ᑕᒪ ᐱᐅᓚ (LFUSC)	ᑲᑎᓀᓂ ᐊᖏᔪᓂᐅᔪᓇᖅᑐᓄᑦ (TAC) ᐅᖃᐅᓯᕆᓂᖃᕐᕆᔨᒃᑯᓐᓂᖅ ᑲᑎᒪᓂᖏᓪᓕ; ᐊᑐᐃᓐᓇᖅᐸᐃᓂᔪᓇᖅᑎᑦ ᑎᑎᕋᖅᓯᔪᓇᒥᒃ ᑐᓂᔫᓕᕆᓂᖅᐸᑕ (ᑐᖏᓚᐅᔪᖅ ᐊᔪᓴᑎ 27).
ᐊᓕᓴᑎᐅ ᐅᓪᕿᐃᓕᓕ (NC)	ᐱᔪᖅ ᑕᑯᓪᓗᓂ ᐊᑐᖅᑕᐅᓇᓕᖅᑐᓂ ᐃᓐᓇ ᐃᓯᒪᓕᓪᓗᓂ ᓇᓗᓇᐃᓴᐃᓂᓪ ᐊᖅᐸᕐᓂᕐᑐᓂ ᓂᕆᑲᓴᖅᓯᐅᔾᔨᓪᓚᑦ ᐊᖏᑲᖅᑕᐅᓯᑯᓪᓗᓂ, ᓂᕆᑲᓴᖅᓯᐅᔾᔨᓂᐅᓇᓴᖅᑐᑭᓴᖅᑐᓂ ᓴᒪᒪ ᑕᐃᒪᐃᑐᐃᓐᓇᕐᓂᓕᕆᓂ. ᐊᑐᐃᓐᓇᐅᑎᑎᔪᖅ ᐊᑐᑎᖃᖅᑐᑐᒃ ᑎᑎᕋᖅᓯᒪᔪᓂᒃ ᐱᑎᐊᖅᑕᐅᖃᓪᓚᑐᖅᑕᓂ ᐅᖃᐅᓯᐊᕐᓇᓂᓂ ᐱᓇᖅᓴᓂᓂ ᐊᒪᒪ ᐊᐅᑕᓪᓂᕐᒥ ᐃᓯᒪᑕᐅᖅᑕᐅᓂᒃ. ᓇᑕᒧᐊᑕᖅᑎᑕᐅᓴᓇᕐᓂᓕᕆᓂ ᐅᖃᐅᓯᐅᓴᐅᕐᓂᒃ ᐃᓕᓴᕐᓂᓇᐅᑕᐅᓴᓇᕐᓂᓕᕆᓂ ᓴᖅᑭᑎᓪᓗᓂ ᑐᓴᕐᓗᐅᑎᓪᓗᓂ; ᓴᖅᑭᑎᓪᓗᓂ ᓴᓇᓕᓕᒫᓂ ᐃᓕᓴᕐᓂᓇᐅᑕᐅᓴᓇᕐᓂᓕᕆᓂ ᓴᓇᓕᓕᒫᓂ ᐃᓕᓴᕐᓂᓇᐅᑕᐅᓴᓇᕐᓂᓕᕆ

- ᑎᑎᕆᖅᓯᒪᔪᓂ ᑐᓂᔨᐅᔪᓂ ᑕᕆᔨᓐᓄᑦ EAGSAC-ᑯᓘᓐᓂ ᐃᓕᑎᔨᐅᔪᓂᒃ ᐱᔪᓕᐅᕗᑦᓯᑎᒃ ᑲᑎᒪᓂᑦ ᐊᖏᓂᒃᑕᐅᔪᓐᓇᖅᑐᓐᓂ (TAC) ᐊᑐᕆᑯᓕᔨᕿᑦ ᐃᓱᓕᖅᓴᖅᑕᓂ ᓴᓇᖅᓚᐅᔨᔪᒻ **ᐱᖓᒪᔪᐊᑦ, ᐊᔪᕐᑎ 27, 2025-ᒥ;** ᑭᖑᓂᐊᔪᓂ ᐊᔪᕐᑎ 27-ᒥ ᓴᓇᖅᓚᐅᔨᖅ ᐱᔪᓐᓇᖅᑕᕆᓂᐊᖅᑐᖅ ᓱᓕ ᑎᑎᕆᖅᓯᒪᔪᓂᒃ ᑐᓂᔨᐅᔪᓂᒃ ᑭᓱᓂᓕ ᑐᓴᖅᑎᑎᔪᓐᓇᖅᑐᑦ ᐅᕆᖅᑭᓱᓂᒃ ᑕᕆᑕᓕᓐᓂᔪᐊᖅᑐᑦ ᑲᑎᒪᔨᖅᑐᓂᒃ ᑕᕆᓚᐅᖅᑕᑦ (NWMB) ᑲᑎᒪᔨᑦ ᑲᑎᒪᓂᖏᑐᓂ ᓄᐊᐱᓇᓗ 5-ᒥ.
- ᐃᓚᖏᐅᑕᓕᓗᐅᑦ ᐱᔪᓐᓇᐃᓐᓇᕆᐊᖅᑐᑦ ᑎᑎᕆᖅᓯᒪᔪᓂᒃ ᑐᓂᔨᐅᔪᓂᒃ ᓴᓇᖅᓚᐅᔨᔪᒻ ᐃᑲᔪᑎᖅᑎᕆᐊᖅᑐᑦ ᒥᓂᑕᒧᑦ ᑲᑎᒪᓂᑦ ᐊᖏᓂᒃᑕᐅᔪᓐᓇᖅᑐᓂᒃ (TAC) ᐃᓕᓴᕐᐅᕐᑕᐅᔪᓐᓇᖅᑐᑦ ᑐᐳᓕᖅᑎᑎᒍᓐᓇᐅᑦ ᑎᑎᕆᖅᓯᒪᔪᓂᒃ **ᐅᓪᓗᑐᐃᓐᓇᖅ, ᓄᑎᐱᓗ 12, 2025.**
- ᐃᓚᖏᐅᑕᓕᓗᐅᑦ-ᑭᓴᓕᓄᑦᓯᐊᖅᑎᖅᑎᓂᒃ ᖃᐅᔨᓕᐊᖏᓂᐊᖅᑐᖅᑐᓂᒃ ᖃᐅᓇᑎᑕᓐᓇᖅᑐᑦ ᐅᑭᐅᖃᐅᖅᑕᖅᓯᓚᖓᑕ ᐊᑯᑭᑐᑐᓂ ᖃᐅᑦᓯᓪᓚᕆᐊᖅᑎᓪᓗᖓᓂᒃ (ᐊᑲᑭᐅᑕᐅᓕᓚᐅᑲᓂᒃ ᑕᐃᓇ 35 ᓄᐊᓂᑎᑦ ᓴᖅᐱᐅᒋᔭᖅᓯᓂᒃ ᑕᑭᓂᒃᑕᐅᓪᓗᖓ ᐊᑐᖅᑕᐅᓪᓗᖓ ᐊᑲᑭᖅᑕᐅᓪᓗᖓ ᐊᑲᑭᖅᑭᖖᒪᔪᓂᒃ ᐱᔾᔨᐅᓄᐃᓕᓐᓂᒃ).
- 35 ᓄᐊᓂᑎᑦᑕᒥᐅᒃ ᑲᑎᒪᓂᑦ ᑕᑎᓂᔭᓂᑦ, ᐊᑯᑭᑐᑐᓂ ᖃᐅᑕᖅᑕᔭᓂᑦ (GHL) ᐃᓚᐃᑦᑐᔪᑦ ~6-7 ᐅᑭᐅᔭᓂᑦ (ᑕᑯᓗᑐᐃ ᑎᑎᕆᖅᓯᒪᔪᓂᖅ 6 ᐅᕐᓂᓂ: ᖃᑭᒥᖅᑭᓂᖅ ᐊᑯᑭᑐᑐᓂ ᖃᐅᑕᖅᑕᔭᓂᑦ ᐱᑕᖅᖃᓕᕐᓂᓄᑦ ᐃᓗᓕᕐᓂᓂᒃ ᑕᑭᓂᓂ ᐅᐊᖅᓯᓚᐅᖅᑕᖅᑐᓂᒃ ᐊᖏᔪᓂᒃᑕᐅᓪᓗᖓ

August 27, 2025

Hon. Joanne Thompson, MP, PC
Minister of Fisheries
Fisheries and Oceans Canada
200 Kent St
Station 15N100
Ottawa ON K1A 0E6

Via email: DFO.Minister-Ministre.MPO@dfo-mpo.gc.ca

Dear Minister:

We are pleased to write to provide our views on the Subarea 01 TAC levels for 2026 and 2027 for Greenland Halibut, in response to DFO's Eastern Arctic Groundfish Stakeholder Advisory Committee (EAGSAC) update of August 12, 2025 requesting our views.

The NAFO Scientific Council Report of June 2025 ((NAFO SCS Doc. 25/13, Serial No. N7664) provides clear evidence that the Greenland Halibut stock remains in excellent health. As the report demonstrates, across all catch scenarios provided there is less than a 1% chance of the stock falling below B*lim*, the threshold at which serious harm to the stock could occur. Furthermore, there is less than a 7% chance of the stock declining below B*trigger* in 2027, the level at which it would move into the cautious zone within the NAFO Precautionary Approach Framework. Current biomass remains well above B*msy*, with the B/B*msy* ratio estimated at 1.20, placing the stock firmly within the Healthy Zone of the Precautionary Approach Framework.

Additionally, the assessment model consistently under-estimates biomass while over-estimating fishing mortality, as documented in the NAFO SC June Report. This inherent conservatism means all TAC options provide additional protections beyond the stated risk levels. Additionally, fisheries performance continues to remain robust, with broadly distributed and historically high catch rates demonstrating the stock's continued productivity.

Given the above, a fishery Total Allowable Catch (TAC) based on F*msy* represents the optimal balance between conservation and sustainable utilization. This approach would ensure continued economic and social benefits to adjacent and dependent Nunavut communities, who are majority Inuit. An F*msy*-based TAC remains entirely consistent with both Canada's and NAFO's Precautionary Approach Frameworks. Notably, even at F*msy* fishing levels, projections indicate a one-third probability that biomass will actually increase by 2027-2028 compared to 2025 levels.

For the above reasons, and the stock's placement in the healthy zone, a reinstatement to the 2022 TAC level is a plausible option. That is clearly defensible.

Our position, however, is to recommend that the 2026 and 2027 TAC levels be set at F*msy* (35,328t and 34,618 mt respectively) as contained in Table 1.2 at page 105 of the NAFO SC June meeting report. This recommendation is grounded on compelling scientific evidence that demonstrates the stock's robust health and minimal risk profile.

The F*msy* TAC level effectively balances conservation objectives with sustainable resource utilization while avoiding any material risk of stock harm. Given the robust health of this resource, there are no scientific grounds for constraining its optimal economic use. Conversely, there are compelling reasons to fully utilize this healthy and productive stock for the benefit of dependent communities.

Thank you for your consideration.

Qujannamiik,

Hon. David Akeeagok
Minister of Community Services, Government of Nunavut

Joeli Qamanirq, Arctic Fishery Alliance

Kelly Kilabuk, Baffin Fisheries

Julai Alikatuktuk, Chair of NFA Board, Cumberland Sound Fisheries

Jerry Ward, Qikiqtaaluk Corporation

Derek Butler, Nunavut Fisheries Association

OCEANS NORTH

1459 Hollis St, Suite 101, Halifax, NS, B3J 1V1

www.oceansnorth.org

August 25, 2025

Submission on 2026/27 TAC recommendations for GHL Subarea 0

Attn: Nunavut Wildlife Management Board

Introduction

Oceans North is a charitable organization that fosters science– and community-based conservation in the Arctic and Atlantic regions of Canada, as well as western Greenland. Our goal in all fisheries processes in which we participate is to ensure that depleted fish populations are rebuilt wherever possible and that they support future generations of fishers while contributing to a functional marine ecosystem. We appreciate the opportunity to provide input on the Total Allowable Catch for Greenland halibut in NAFO Subarea 0 +1 offshore for the 2026/2027 fishing seasons.

Goal of long-term sustainability

We are pleased that this offshore Greenland halibut stock now has an accepted model assessment tool and acknowledge the hard work of NAFO Scientific Council (SC) members in achieving this goal. The 2025 NAFO SC assessed the stock as being in the healthy zone according to the NAFO Precautionary Approach Framework, with a low (<1%) probability of the stock being below B_{lim} and risk of fishing mortality being above F_{msy} at 42%.

Greenland halibut are of significant value to the region in many ways. As a species, they play a critical role in Arctic ecosystems as a bentho-pelagic predator, helping transport nutrients throughout the water column and acting as an important prey item for narwhal, beluga, and seals. The fishery further adds tremendous economic value to northern communities, providing employment opportunities and financial benefits as one of the most lucrative groundfish fisheries in the region.

We appreciate that with the stock in the healthy zone there may be strong desires to ramp up the fishery to maximize short-term profits. However, given the significant value of the fishery combined with below-average recruitment indices in recent years,

we suggest following lower-risk TAC recommendations that would benefit the sustainability of the stock in the long-term.

We are therefore recommending **TAC be set at F_{target} fishing levels that result in the long-term stability of the stock**, corresponding to catch levels of 30,243 t and 30,153t in 2026 and 2027, respectively.

Thank you for your consideration of this submission.

Sincerely,

Katie Schleit
Fisheries Director
Oceans North

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ᑕᕐᓯᒧᖏᑐᒃ

ᑐᑭᓯᐅᕐᕕᑭᑎ: ᐃᓕᓴᕐᕕᖏᓂ: ᐃᓚᖓ: X

ᐃᓯᒪᓂᐅᑎᑎᓂᖅ: ᓯᖏᒧᐊᓕᖅᑎᓂᑎᓂᖅ ᐱᔨᐅᖏᕐᕿᓂᖅ ᒪᓕᕐᕉᓂ ᐊᑯᖏᕿᓂ ᕿᑎᓕᕐᓂᖅ ᐃᕐᓴᓂᕆᓂᖓᓂᒃ ᐅᐊᖏᓇᖅᓂ ᐊᕐᓇᖅᑎᖅ ᐃᒪᖓᑕᐅᓂᖓᓂᒃᑐᒥᓂᒃ ᐅᐊᖏᓇᖅᓂ ᐊᕐᓇᖅᑎᖅ ᐃᒪᖓᑕᐅᓂᖓᓂᒃᑐᒥᓂᒃ ᐃᓕᖓᔪᖅᑐᑦ (SA) 0–ᒥ. ᐱᕐᓴᖅᑎᐅᕿᓴᕆᓕᕐᕉᓂ ᐃᓯᒪᑎᖏᖅᓯ ᓯᖓᒧᐊᓕᖅᑎᓂᑎᓂᖅ ᐱᔨᐅᖏᕐᕿᓂᖅ ᐊᑐᑎᒃᓴᖅᓴᐅᓂᖅᓴᖅᑐᖅᓂᖓᓂᒃᑐᒥᓂᒃ ᐊᐅᓪᓚᕐᓂᑎᐊᖏᓂᖅᓴᖅᑐᖅᓂᖓᓂᒃᑐᒥᓂᒃ ᐃᓕᖓᔪᖅᑐᑦ ᐱᓇᐅᖏᓕᕐᓂᖓᓂᒃᑐᒥᓂᒃ ᐃᓕᖓᔪᖅᑐᑦ ᐱᓇᐅᖏᓕᕐᓂᖓᓂᒃᑐᒥᓂᒃ ᐃᓕᖓᔪᖅᑐᑦ ᐃᓕᖓᔪᖅᑐᑦ ᐃᓕᖓᔪᖅᑐᑦ

ᑯᑕᓪᓗᐅᐳᑦ ᐊᖏᕐᒍᖅᓯᓄᑦ (ᓄᖏᓗ, ᓯᕗᓪᓕᐸᑎᑕᐅᔪᑦ ᐊᓚᓗ ᐊᑐᖅᑐᐊᖏᓂᖅ) ᑕᒪᐃᓐᓄᑦ ᐊᑯᑭᑐᖏᑦᑕᐅᓄᑦ ᐃᖏᕐᕕᖅᓯᓄᑦ, ᐃᓕᓴᐅᕗᖅᓯᒪᓕᒐᓗᓐ ᐊᑯᑭᑐᑎᓂ ᖃᓄᖏᓕᓪᓂ ᐅᐊᖅᓇᓚᑎᓂ ᐊᑭᓐᓄᖅ ᐃᒪᖏᐅᑕᖏᓂᐊᖅᓯᕗᓪᓕᓴᐅᓂ ᑎᑎᖏᑎᖅᓚᖅᑐᑦ SA 1 (ᐃᒪᐃᖅᑎᑦ). ᐳᓴᓂᖅᓚ ᐱᔪᓇᖅᑎᑕᐅᔪᑦ ᐊᑯᑭᑐᖏᑦᑕᐅᓄᑦ ᐱᖏᑯᖅᓚᒍᑦ ᖁᑎᖅᓚᐊᑎᔪᓇᖅᓯᒪᓕᒫᑦ ᓯᕗᓪᓕᐸᑎᑕᐅᓂᕐᒥᐅᖅᓯᖏᓐᓄᑦᑐᖅ ᐱᓴᐅᖅᓯᒡᒎᖅᑐᑦ *ᖁᐅᐅᓯᒪᔪᓂᑎᓂ* (ᓄᖏᓗ 15 ᐳᓴᓂᑦ ᐃᖁᓗᑕᓪᓗᑎᓂᐅᔪᑦ ᑕᒪᐅᑭᖅᓱᑕᑦ ᑯᑕᖅᓚᑦ), ᐱᑕᓪᓚᕐᓇᖅ ᐃᓄᒪᑕᐅᖅᓄᒪᔪᑦ ᐊᖏᕐᑭᑎᑕᐅᖁᓂᖅᑕᖅᑐᓂᖅᓯᒪᓕᒫᑦ ᐊᑯᑭᑐᖏᑦᑕᐅᓄᑦ ᑲᔪᓄᑎᑕᐅᓂᖅᓯᒪᓕᒫᑦ ᓯᕗᓪᓕᐸᑎᑕᐅᓂᖅᓯᒪᓕᒫᑦ ᐱᓴᐅᖅᓯᒡᒎᖅᑐᑦ ᒪᑕᒪᓕᒡ ᐃᓄᖅᓴᓕᖅᑐᑦ SA 0+1 (ᐃᒪᐃᖅᑎᑦ) ᐊᑯᑭᑐᓂ ᖃᓄᕐᕕᒃᓴᓄᑦ.

ᐱᔪᓀᑎᑦᓴᐃ ᐃᖃᓄᐃᓕᓚᑲᑦ ᐱᔪᒪᓂᖅᑭᓂᒃ ᐃᖅᑎᕐᕌᖅᑎᑦ ᐊᓕᔪᐃᑲᐅᑎᑎᔪᓇᖅᓯᖅᑭᓂᒃ ᓯᕗᓪᓕᐸᑎᑕᐅᓂᖅᓯᒪᓕᒫᑦ ᐱᖏᕐᕕᖅᓯᒡᒎᖅᑐᑦ, ᐃᓕᓴᐅᑲᖏᓯᕗᐅᑎᑎᓪᓕᐃᓗᐃ ᑕᒪᐃᑎᒍᑦ ᐃᑲᔪᕐᓂᖅᑭᓂᒃ ᖃᓄᐃᓕᐅᕐᖅᑭᓇᖅᓯᖅᑭᓂᒃ ᐃᖃᓄᐃᓕᓴᐅᓂᕐᒥᐅᖅᓯᖏᓐᓄᑦᑐᖅ ᐃᓄᒪᑕᐅᓂᖅᓯᒪᓕᒫᑦ ᐃᒪᓕᖅᑭᐅᓂᖅᓯᒪᓕᒫᑦ ᐱᔪᒪᓂᖅᑭᓂᒃ ᐃᖃᓄᐃᓕᓴᐅᓂᕐᒥᐅᖅᓯᖏᓐᓄᑦᑐᖅ ᐊᑯᑭᑐᖏᑦᑕᐅᓄ

ᐅᖃᐅᓯᕆᓕᕆᐅᔪᑦ ᐅᑦᓯᒥᒃ ᖃᖅᑭᖅᑎᑎᔪᑦ ᐃᑲᔪᖅᓯᖅᑕᐅᓂᖓᓂᖅ ᓯᖏᒧᐊᖅᑎᑎᓂᖅ.

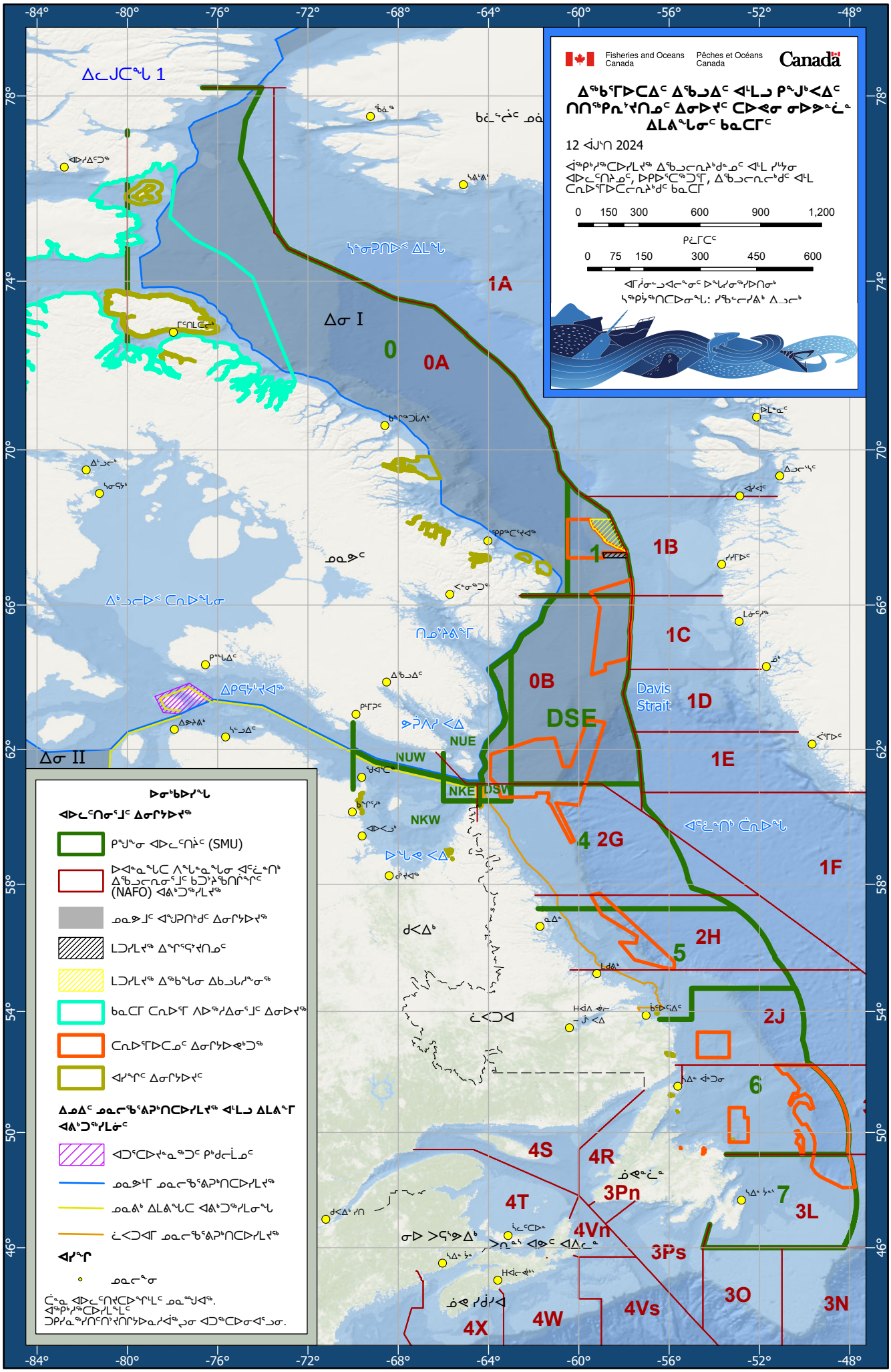
ᖃᐅᐸᒪᕆᐅᑦ, ᓄᓇᖏᑦ ᑲᖏᒪᖓᑐ ᐃᑲᔪᖅᓯᓂᕋᒧᐅᖅᑐᑦ ᐊᑐᖅᑕᐅᓂᖓᑐ ᓯᖏᒧᐊᖅᑎᑎᓂᖅ ᐱᕕᐅᖓᓂᖅ ᐃᑦᑲᓕᑕᓚᓂᐅᔪᒪᑦ, ᑭᓯᐊᓂᑕᐅᖅ ᓴᓇᖅᑯᔪᑦ ᓯᖏᒧᐊᖅᑎᑎᓂᕐᒧᑦ ᐱᕕᐅᖓᓂᕐᒥᑦ ᐊᑐᐊᒐᖓᑦ ᑐᓂᒃᑯᐊᓕᔪᑦᑐ ᓄᓇᖏᒪᓕᕆᓂᐊᖅᑐᑦ (ᐊᑐᖓᓂᖅᓱᖓᓗᑎᒃ ᑐᖓᓂᒃᐃᖅᑲᖅᑐᑦ ᐊᑦᓀᓐᓂᑦ ᑲᑎᒪᓂᖅ ᑐᑭᓯᐅᑎᒃ ᐊᑦᓀᓐᓂᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ ᐊᑐᐊᒐᖓᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ ᐊᑐᐊᒐᖓᑦ ᐃᓚᒋᑦ *ᖃᐅᔨᓴᕐᔪᐊᓘᑦ* ᓴᖅᑭᖅᑎᑎᓂᑎᑎᐊᖓᓕᒫᑦ ᐊᑦᔨᐅᖓᓂᓐᓂᐅᖅᑐᑦ, ᐊᐅᓚᓂᖅᑲᑦ ᐱᑦᓚᖓᖓᓂᐅᖅᑐᑦ ᐊᒻᒪᓗ ᐊᐅᓚᓂᐊᕐᓇᖅᓯᓂᖓᓂᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ ᐊᑐᐊᒐᖓᑦ ᐊᐅᓚᓂᐊᖅᑐᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ SA 0–ᒥ.

ᐅᖃᐅᓯᕆᓕᖅᓯᒪᓂᖅ ᓄᓇᖏᒪᒥ ᐃᖃᓗᐃᑦᑕᕐᓂᖅ ᑲᑐᔾᔨᖃᑎᒋᑦ (NFA) ᐃᓕᓴᖅᓯᓚᐅᖅᑐᑦ ᓯᖏᒧᐊᖅᑎᑎᓂᖅ ᐱᖓᓂᖅᑐᑦ ᐊᓯᔾᔨᒪᓕᐅᕕᖓᓂᖅᓯᒪᓂᖅ ᑭᓯᐊᓂᖅᑐᑦ ᐊᑐᐊᒐᖓᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ ᐱᖓᓂᖅᑐᑦ 15%–ᒥᑦ 25%–ᒧᑦ ᐊᑦᔨᑎᖅᐅᓕᕆᓂᖅᓯᐊᖅᓚᑦ ᐊᑯᑭᑐᖅᑎᒋᐅᓄᑦ ᒪᓕᑎᐅᓕᖅᑐᑦ.

ᐊᑐᓕᖅᑯᐅᔪᖅ

ᐱᔨᐊᑎᒋᓕᑐ ᐃᖃᓄᓕᓯᓴᑎᑦ ᐱᔨᒪᓂᕐᓂᓂᖅ ᐃᕐᓂᐊᓕᖅᑎᑦ ᐊᓯᔾᔨᑲᖏᐅᑎᑎᔨᓕᖅᓯᕐᓂᖅᓂᖅ ᓯᖏᒧᐊᖅᑎᑎᓂᖅᑐᑦ ᐱᖓᓂᖅᑐᑦ, ᐃᓯᒪᔭᖏᖅᓯᐅᑦᑎᐊᕐᓂᓕᒫᖅᑐᑎᒃᓴᓂᖅᓯᒪᓂᖅ ᑭᒪᐃᐅᑦᑐᖅᓯᒪᓂᖅ, ᐃᓚᒋᔭᖏᖅᓯᐅᑦᑎᐊᕐᓂᓕᒫᖅᑐᑎᒃᓴᓂᖅᓯᒪᓂᖅ ᐃᖃᓄᓕᓯᓴᑎᑦ ᐊᑐᐊᒐᖓᑦ ᐊᐅᓚᓂᐊᖅᑐᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ SA 0–ᒥ ᐊᑐᓕᖅᑎᑎᓂᖅᑐᑦ (ᒪᓕᓚᖏᖅᑐᑦ ᐅᓂᐊᓚᐅᓂᖓᓂᖅᑐᑦ ᐊᒻᒪᓗ ᐊᑐᓚᐅᖏᖅᑎᑕᖅᑐᑦᓯᒪᓂᖅᑐᑦ ᐃᓕᓴᐅᔪᖅᑐᑦ *ᖃᐅᔨᓴᕐᔪᐊᓘᑦ*) ᐱᑎᐊᓕᓗᑎᒃ 2026–ᒥ ᐃᓕᓴᐅᔪᖅᓯᒪᔪᖅᑕᐅᓂᖓᓂᓗᓂ.

ᐱᓗᐊᖅᑐᒥᒃ, ᐱᔨᓐᓇᖅᑎᑕᐅᔨᓂ ᓯᖏᒧᐊᖅᑎᑎᓂᖅᓯᒪᓂᑦ ᐊᖅᑭᒧᐅᔨᓂ, ᐱᔨᓐᓇᐅᑎᓂᖅᑲᖅᑎᑎᐅᔨᓕ ᓯᖏᒧᐊᖅᑎᑎᓂᐱᔨᓐᓇᓇᕐᓂᖅᑐᑦ ᐱᖓᓂᖅᑐᑦ ᒥᑭᓐᓇᖅᑐᑦᓯᒪᓂᖅ 15 >ᓴᓐ ᐊᑯᓐᓂᖓᓂᓂ ᐊᒥᐊᖅᑯᑎᓐᓂᖅᑐᑦ ᓴᓂᖅᑭᖅᑕᐅᔨᓂᓂ (ᑕᒪᖅᑯᖅᓯᒪᓗᑎᒃᑕᑦ ᑕᒪᖅᑐᑦ ᐃᓯᖅᓴᓕᓂᐊᖅᑐᑦ ᐊᑐᓂᓕᓂᐊᖅᑐᓗ ᐃᓕᓴᐅᔪᖅᓯᐅᓇᓐᓂᑦᓗ ᑯᑎᒃᓴᓕᒫᖅᑐᑎᒃᓴᓂᖅᓯᒪᓂᖅ ᓯᖏᒧᐊᖅᑎᑎᓂᕐᒥᑦ ᐱᖓᓂᖅᑐᑦ ᐊᒥᐊᖅᑯᑎᓐᓂᖅᑐᑦ ᐊᑐᓂᓕᓂᐊᖅᑐᑦ ᐊᑐᐊᒐᖓ


Fisheries and Oceans Canada
Pêches et Océans Canada
Canada
ᐃᖃᓗᐃᑦ ᐃᓄᒃᑎᑐᑦ ᐃᓄᒃᓚᐃᑦ ᐊᑦᓚᓗ ᑭᓱᒍᐸᐃᑦ ᑎᑎᕐᑭᓂᖅᑐᑦ ᐃᓕᐊᓐᓂᑦ ᐅᑭᐅᖅᑎᑦ ᐊᑦᓚ ᑭᓱᓂ ᐊᐅᑉᐳᒍᓐᓇᖅᑐᑦ ᐃᓕᐊᓐᓂᑦ ᑲᓇᑕᒥ
12 ᐊᔪᓐ 2024
ᐊᖏᑭᓱᖏᑕᐅᓯᒪᔪᖏᓪ ᐃᖏᓄᑕᒥᓵᑦᑯᓄᑦ ᐊᑦᓚ ᑭᓱᓂ ᐊᐅᑉᐳᓪᓚᑦ, ᐅᑭᐅᖅᑕᖏᑐᒃᑎᒥ, ᐃᖏᓄᑕᒥᓵᑦᑯᓄᑦ ᐊᑦᓚ ᑲᓇᐅᖅᑎᑕᒥᓵᑦᑯᓄᑦ ᑲᓇᑕᒥ
0 150 300 600 900 1,200
ᑭᓵᒥᑦ
0 75 150 300 450 600
ᐊᒥᓱᓕᓪᓚᐊᓕᕆᓂᖅ ᐅᖏᑭᓱᖏᑎᑕᐅᓂᖅ
ᓴᖏᑭᖏᑎᑕᐅᓂᖅᓯᒪᓕᖅ: ᓱᖏᓕᓕᓱᕐᕋᖅ ᐃᓗᓕᖅ
ᐃᓕᒍᑦᑐᖅ ᑕᓪᓕᐊᑲᓂᖅ
1A
ᐃᓂ I
0
0A
1
1B
1C
0B
DSE
Davis Strait
1D
1E
ᐃᓂ II
NUE
NUW
NKE
DSW
NKW
4 2G
1F
2H
5
2J
6
4S
4R
4T
3Pn
7
3L
4Vn
3Ps
3O
3N
4Vs
4W
4X
ᐅᖃᐅᓯᖅ ᑕᓪᓕᐊᑲᓂᖅ
ᐊᑦᓚᓕᕆᓂᖅ ᐅᖃᐅᓯᖅ (SMU)
ᐅᐊᖏᓐᓄᑦ ᐱᖓᓐᓄᑦ ᐊᑦᓚᓂᖅ (NAFO) ᐊᖏᔪᒃᑯᑦ
ᓄᓇᒃᓯᒪᓂᑦ ᐊᖓᔪᖅᑎᓯᒪᓂᑦ ᐃᓂᒋᔭᖅ
ᒪᑐᐸᓪᓚᖅ ᐃᓕᓴᕐᕋᔭᖅᑎᓂᑦ
ᒪᑐᐸᓪᓚᖅ ᐃᖏᓄᒃᓱᐃᓐᓂᖅ
ᑲᓇᑕᒥ ᑕᓪᓕᐊᑲᓂᖅ ᐱᐅᓪᓚᐊᓂᖅ ᐃᓂᒋᔭᖅ
ᐃᓄᐃᑦ ᐃᓚᖏᓐᓇᑦ ᐊᑦᓚ ᐃᓚᖏᓪᓕᐊᓂᖅ ᐊᖏᔪᒃᑯᑦ
ᐊᓯᖏᑦ
ᓄᓇᓕᖅ
ᑕᑯᓗ ᐊᐅᑉᐳᓪᓚᑦᑎᓪᓗᒍ ᓄᓇᖃᖅᑐᑦ.
ᐊᖏᔪᒃᑯᑦᑐᐅᓯᒪᔪᖏᓪ
ᑐᑭᓯᓗᖏᓐᓄᑦᓯᐅᑎᑦᓴᐅᓇᓱᒋᐊᖏᓪᓗᓂ ᐊᑐᖏᑦᑐᐅᓂᐊᓪᓚᖅ.
© Minister of Fisheries and Oceans Canada 2023 (ᑎᓂᒃᑕ ᐃᖏᓄᑕᒥᓵᑦᑯᓄᑦ ᐊᑦᓚ ᑕᓪᓕᐊᑎᑦᑐᑕᒥᓵᑦᑯᓄᑦ ᑲᓇᑕᒥ 2023)
ᐊᖏᔪᒃᑯᑦᑐᐅᓯᒪᔪᖏᓪ ᐃᖏᓄᑕᒥᓵᑦᑯᓄᑦ ᐊᑦᓚ ᑕᓪᓕᐊᑎᑦᑐᑕᒥᓵᑦᑯᓄᑦ ᑲᓇᑕᒥ, ᐅᑭᐅᖅᑕᖏᑐᒃᑎᒥ, ᐃᖏᓄᑕᒥᓵᑦᑯᓄᑦ ᐊᐅᑉᐳᓪᓚᑦ

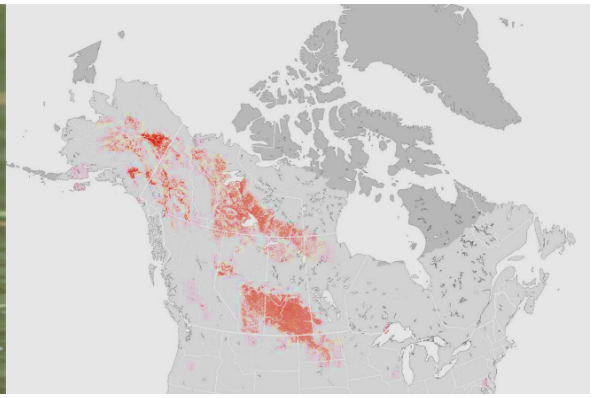
ᑐᓂᔨᐅᓯᒪᔪᑦ ᓄᓇᕗᑦ ᐅᒪᔪᓕᓗᐊᔪᖅᑕᐅᔪᑦ ᑲᑎᒪᔨᖏᑦᓴᓄᑦ

ᐱᓕᕆᖃᖅᑐᑎᑦ

ᑐᑭᓯᒋᐊᑭᑎᓂᖅ **ᐃᓱᒪᓕᐅᓇᐊᓕᐳᖅ: X**

ᐱᔪᑎᓕᖅ: ᐃᓱᒪᓕᐅᐸᔪᒍᑦ ᑎᑎᕐᓂᐅᕐᓯᓚᓕᑦ ᐊᓯᐸᕐᓚᑦᓚ ᒥᑎᐅᕆᐅᕈᖅᑐᑦ ᐃᒪᒥᐅᑕᐅᓯᖅ (Horned Grebe) ᐃᓗᐊᓂᑦ ᑎᑎᕐᓂᓯᒪᔪᖅ ᓂᕆᔨᑎᓂᑦ ᐅᓗᓇᐊᓇᖅᑐᑐᓗᓂᑦ ᐱᖓᓱᖅᑐᓂᖅ (SARA)





ᖃᓄᐃᑦᑐᓯᒪᓕᒃᑲᖅ:

- ᒥᑎᐅᕆᐅᕈᖅᑐᑦ ᐃᒪᒥᐅᑕᐅᓯᖅ (Horned Grebe) ᓯᓚᓪᓕᖅᐸᒥ ᑎᑎᕐᓂᑕᐅᓪᓕᐅᕐᓯᒪᔪᑦ ᐃᓪᓕᖅᑐᓂᒃ 1–ᒥ ᑕᐃᑲᓂᒃ ᓂᕐᔪᑎᓂᑦ ᐅᓗᓇᐊᓇᖅᑐᑦᑕᓗᓂᑦ ᐱᕐᓂᖅᑐᓂᒃ (SARA) 2011–ᒥ ᒪᕐᕉᓕᖅᑲᖅᓯᓪᓗᑎᑦ ᐊᐃᕐᑐᖅᓯᒪᓂᖅᑎᑦ:
  - ᐱᖅᓯᓄᖅᓯᓂᒃ (ᐃᓯᒪᓗᑎᒃᖅᑐᑦ)
  - ᒪᑲᑦᓪ ᖁᑭᖅᑕᖅᓂᒃ (Magdalen Island), ᓂᐸᑲ (ᓄᖅᔪᑐᐃᓄᓕᓇᓚᐅᕐᑲᑦ)
- 2023–ᒥ, ᑕᑲᓂᐊ ᑲᑎᒪᔨᕿᓪᓚᕐᓂᖅᓴᓂᒃ ᓄᖅᔪᑐᐃᓇᓚᕐᓂᓕᕆᓂᖅᑕᐅᓪᓚᐅᕐᓂᑦ ᓂᕐᔪᑎᓂᑦ (COSEWIC) ᑐᓴᕐᓴᖅᑐᑦ ᒪᕐᕈᑲᑦ ᐊᒥᓱᓂᖅᑎᑦ ᑲᑎᒪᓕᑦᑎᑦ ᓄᑕᖅᑎᓕᑦᑎᑦ ᓇᓗᓇᐃᑲᓂᖅᑎᑦ ᖁᒥᕐᖁᔨᐅᓪᓚᕐᓂᑦ ᓂᐅᓪ ᐃᓱᒪᓗᓇᖅᓂᓗᑎᑦ.
- ᐊᑐᓕᕐᓂᔨᓄᑦ ᑲᑎᒪᓕᑦᑎᑦ ᑕᑲᓂᐊ ᖁᒥᕐᖁᔨᐅᓗᑎᑦ ᐊᒋᐅᓯᐅᓕᕐᓗᑎᑦ ᐱᐅᓂᖅᓴᐅᔨᓕᕐᓂᖅᓯᒪᑦ ᐃᓕᓴᐲᖅᓯᒪᓂᖅᑎᓯᒪᓕᕐᓗᑎᑦ.
- ᑲᑎᒪᓕᑦᑎᑦ ᐊᒥᓱᓂᖅᑎᑦ, ᓄᑕᕐᒥᑦ ᓇᓗᓇᐃᓴᐲᓂᖅᓯᒪᓕᕐᓗᑎᑦ ᐃᓱᒪᓕᐅᕐᓗᑎᑦ, ᑭᓱᓕᓕ ᐊᓯᐸᖅᑐᐊᓗᓂᖅᑕᑎᓗᐊᑦ.

- ᑎᑎᕆᖅᑕᐅᕐᕕᑕ, ᓄᑦᖃᒍᓗᑎᑦ ᑲᓇᑕᒥ ᐊᐅᓪᓚᑎᓂᕐᒧᑦ ᕿᖏᐅᑎᓂᑲ ᑕᐃᒪᐃᒋᐊᖅᑕᕆᔨᖅᑐᑦ ᑦᑲᔪᓇᖅ ᓂᖏᐊᑎᓄᑦ.
- ᓄᓇᕗᑦᒥ, ᓂᖏᐊᑦ ᑕᑯᐅᐅᓯᒪᑐᐃᓇᖅᑐᑦ ᑭᓯᐊᓂ ᓂᒋᖅᕿᓯᐊᓂ ᑎᓇᖅᑯᐊᓂ, ᑭᓪᓕᖅᑭᓂ ᓄᓇᑕᑎᐊᑦ, ᓴᓴᑲᑦᓯᐊᓵ ᐊᓪᒪᓗ ᐃᓂᑐᑦᒥᑦ.
- ᑐᑭᓯᒋᐊᕈᑎᑕᖅᑲᑲᓂᖅᑐᑦ ᖃᓄᓕᕆᓪᓕᖅᑭᓂᓂᓂᓂᓂ ᓂᖏᐊᑦ, ᐱᔨᐅᖅᑕᑦᓂᖅᑭᓂᓂ, ᓀᐊᒃᖅᑎᑦ ᐊᓪᒪᓗ ᐅᓗᓇᐊᓇᖅᑐᒃᑯᑦᑐᓂᖅᑎᑦ ᑕᐴᓂᓂᓕᕆᓂᖅ A–Γ

ᑲᑎᒪᑎᑦᑎᖅᑕᑦᓂᓂᖅ:

ᐅᑦᓗᓐᖓ	ᐃᐴᐅᐊᓇ 5, 2022	ᐊᖏᑭᓵᓯ 2025	ᓯᑎᑦᐱᓇ 2025 (ᐃᓯᒪᑭᔨᐅᐊᖅᓂᖅ ᑕᐃᒧᖅ)
ᖃᓄᖅ ᖃᐅᔨᓯᐊᖃᑦᑕᑦᓂᐊᖏᒪᖓᒪᑕ	ᑲᑎᒪᔨᐊᑎᓴᓂᑦ ᑐᓂᓯᓗᓇᑎᑦ ᐃᑦᑭᑦᓭᓗᓂᖅᑕᐅᑎᒃᓯᑦ	ᑲᑎᒪᔨᐊᑎᓴᓂᑦ ᑐᓂᓯᒃᓂᓂᓗᑎᑦ ᐃᑦᑭᑦᓭᓗᓂᖅᑕᐅᑎᒃᓯᑦ	ᖃᐅᔨᓯᐊᓕᕆᓂᓗᑎᑦ ᐅᖃᓗᓕᐅᑎᒃᓯᑦ ᑭᖑᓂᖓᓂ
ᑭᓚᓗ ᖃᐅᔨᓯᐊᖃᑕᐅᓂᐊᖏᒪᖓᓕᖅ	ᑭᓴᕐᓕᕆᖅ ᐃᓄᐃᑦ ᑲᑐᔨᖃᑎᒋᔭᖅᑐᑦ, ᑭᓴᕐᓕᕆᖅ ᓂᖏᐊᑕᑕᓚᓂᕐᒧᑦ ᑲᑎᒪᔨᖅᑭᑦ (NWMB), ᑐᓴᖓᕕᒃᓯᑦ (NTI), ᓄᓇᕗᑦ ᓴᖅᓪᓕᒃᓯᑦ (GN)	ᑭᓴᕐᓕᕆᖅ ᐃᓄᐃᑦ ᑲᑐᔨᖃᑎᒋᔭᖅᑐᑦ, ᑭᓴᕐᓕᕆᖅ ᓂᖏᐊᑕᑕᓚᓂᕐᒧᑦ ᑲᑎᒪᔨᖅᑭᑦ, ᐅᓇᓱᖅᑕᕐᓂᐊᖏᒪᖓᓕᖅᑎᑦ ᐊᖏᑭᓵᓯ 2025 (NWMB), ᑐᓴᖓᕕᒃᓯᑦ (NTI), ᓄᓇᕗᑦ ᓴᖅᓪᓕᒃᓯᑦ (GN)	ᑕᐃᒃᑯᐊ ᑭᐅᓂᐅᖅᑎᑦᑐᑦ

- ᑲᑎᒪᔨᐊᑎᓴᓂᑦ ᑎᑎᕋᓯᐊᖃᖅᑎᓗᓂ ᐊᓪᒪᓗ ᐊᐱᖅᑯᓯᕐᓂᑦ (ᐃᓄᑦᑎᑐᑐᖓᓕᐊᑦ ᖃᐅᔨᓗᓀᑎᑐᑦᓗ; ᐃᓕᓴᐅᑎᓯᒪᓗᑎᑦ B–Γ), ᑐᑭᓯᓯᐊᖓᓕᕆᓂᖅᑐᑦ ᑭᓚᐅᑦᓴᖅᑯᓯᓂᖅ ᓂᖏᐊᑕᑕᓚᓂᕐᒧᑦ [ᑐᑭᓯᒋᐊᖓᓕᖅᑐᑦ ᓂᖏᐊᑦ ᑕᑯᐅᐅᓯᒪᑐᐃᓇᖅᑐᑦ ᐊᐅᓪᓚᑎᓂᕐᒧᑦ ᑲᑎᒪᔨᖅᑭᑦ ᐊᐱᖅᑯᓯᕐᓂᑦ ᐅᓂᒃᑳᑦ ᐃᓄᐃ

- ᖃᐅᔨᒪᔭᐅᔪᑦ ᓇᒫᓂᕐᓂᓂ ᒥᑎᐅᔭᐅᓴᖅᑐᑦ ᐃᒪᖏᑕᐅᓕᕆᖅ (Horned Grebe) ᓄᓇᒃᓚᒥ ᐃᓗᐊᓂᓐᓄᑦᑐᓗᓪᓗᒍ 50–ᑭᓚᒫᑕᓂᕐ ᐅᒪᔪᔾᓂᐊᖅᑎᕐᑯᓂᕐ ᕿᐊᓘᒪᓚᕐᑯᓄᓂᕐᓂᓪᓂᕐ, ᑕᐃᒪᐃᓂᕐᓂᕐᓄᑦ ᐱᓕᕆᓂᖅᓄᓂᕐᑐᑦ ᖃᐅᔨᓴᒥᖏᓐᓇᓘᒥᑦ ᑕᒪᑐᒍᓐᓇᖅ.
- ᓄᓇᒃᓚᒥ, ᑲᓇᑕᒥ ᓂᖏᐊᑎᓕᕆᓂᖏᓐᓇᑦ ᐱᔨᑦᑎᖏᖅᑎᓄᑦ (CWS) ᐃᑲᔪᖅᑐᖅᑕᐅᓚᐅᕐᖅᑐᑦ ᐊᓯᔨᖅᑕᐅᔪᒪᓂᕐᓂᓂ ᓄᓇᒃᓚᑦ ᓴᖅᓪᓚᑲᓂᕐᓂᓂᖅ. ᐊᓯᐊᐃᑦ ᑭᐅᓚᐅᓂᕐᓂᑐᑦ ᐃᓐᓇᒧᑦ.
- ᐅᑭᐅᖅᑕᖅᑐᓂ, ᑲᓇᑕᒥ ᓂᖏᐊᑎᓕᕆᓂᖏᓐᓇᑦ ᐱᔨᑦᑎᖏᖅᑎᓄᑦ (CWS), ᑎᑎᕋᖅᑕᐅᓂᒃᓵᓚᓕᐊᓂᖅᑎᓵᓐᓄᐃ, ᐃᑲᔪᖅᑐᐃᓚᐅᖅᑐᑎᓐᓄᐊᓕᑦ ᐊᓯᔨᖅᑕᐅᔪᓚᔪᒥᑦ ᑕᐃᑲᓂᓕᓐ ᐅᐃᑭᐅ (Weekeezi) ᓄᓇᓕᕆᓂᖏᓐᓇᑦ ᑲᑎᒪᔨᖏᑦ ᓄᓇᑦᑎᖏᒥ. ᐊᐅᒥ, ᓂᓪᓕᓚᐅᓂᕐᓂᑐᑦ ᑐᐱᖏᓐᓇᕐᓂᕐᓂᐅᓕᕆ ᓇᓪᓕᐊ ᓂᖏᐊᑎᓕᕆᓂᖏᓐᓇᑦ ᐅᖃᐅᔨᒪᔭᐅᓴᖅᑐᑦ ᑲᑎᑦᓂᖅᓄᑦ (North Slope) ᐅᕝᕙᓚᓂᓂᑦ ᐊᐅᖅ ᐃᖃᓗᓕᕆᓂᖏᓐᓇᑦ ᓂᖏᐊᓕᕆᓂᖏᓐᓇᓪᓗᓐᓂ ᑲᑎᒪᔨᖏᑦ.

ᖃᓄᐃᓕᓴᕐᓂᐊᓕᕆᓂᖏᓐᓂᑦ:

ᐅᑯᐊ ᑲᑎᒪᔨᑦ ᐃᓱᒪᓕᕆᓱᒪᖏᑦ, ᒥᓂᔨᑦ ᐊᑐᓕᖏᔨᓐᓇᓂ ᓴᕝᓄ ᑲᑎᒪᔨᖏᑎᓄᑦ ᐃᓱᒪᑎᖃᓱᐅᑎᓵᓇᑦ ᑲᑎᒪᔨᕋᓴᒃ ᖃᓄᐃᓕᓴᕐᓂᐊᓕᕆᓂᖏᓐᓂᑦ ᓄᓇᕗᑦ ᑲᑎᒪᔨᖏᑦ (COSEWIC) ᖃᑎᖏᑕᐅᓂᖏᓂᑦ, ᑲᑎᒪᑎᑦᑎᓂᖏᓂᑦ, ᑕᐃᔾᔨᐊᓐ ᓂᖏᐊᓕᕆᓂᖏᓐᓇᓪᓗᑦ ᐊᐅᓪᓚᑎᓕᑦ ᑲᑎᒪᔨᖏᑦ ᐊᖏᓂᖅᓱᓐᓂᑎᑦ ᓂᖏᐊᓂᓂ ᓄᓇᑕᑐᑎᐅᓵᑦ ᐊᖏᑐᑎᖏᓐᓄᑦᓂᓂ (ᐱᖃᓱᐅᑎᓂᑦ ᓄᓇᓕᕆᓂ ᐅᒪᔪᓚᔪᖅᐊᓕᐳᔨᑦ ᑲᑎᒪᔨᖏᑦ), ᐊᓯᒪᓐ ᒪᓕᑎᓂᑦ ᐊᑦᑐᐃᓂᖃᖅᓂᖏᓐᓂᓂ ᖃᑎᖏᑕᓂᓪᓗᑦ ᑎᑎᕋᖅᑕᐅᓱᒪᔪᑦ. ᑭᖑᓪᓕᖅᑕᒥᓕᓂ ᐃᓱᒪᓕᐅᔾᔨᓂᖏᓂᓂ 5.3.16 ᓄᓇᓕᕆᑦ ᐊᖏᑕᑐᖓᓂᓂᓂ, ᑕᒪᓐᓇ ᒪᓕᕐᑕᐅᒋᐊᓕᓐ ᒥᓂᔨᑦ ᑕᐃᔾᔨᐊᓐ ᐊᑦᑐᐃᓂᖃᖅᓂᖏᓐᓂᓂ ᐊᑲᓴᔨᖅᑕᐅᓚᐅᖅᑐᓕᖅ, ᑕᐃᑲᓂ ᒪᓕᕐᑕᑎᑦ ᒪᓕᑎᑦᑎᑦ 5.3.16 ᓄᓇᓕᕆᑦ ᐊᖏᑕᑐᖓᓂᓂᓂ, ᑕᒪᓐᓇ ᒪᓕᕐᑕᐅᑭᐊᓕᕆᓂᖅ ᒥᓂᔨᑦ ᐃᓱᒪᓕᕆᓂᖏᓐᓂᓂ.

ᐃᓕᑭᖅᐅᓱᑎᒪᔭᒍ ᓴᖅᓚᑐᖅᑲᓂᑦ ᐃᓕᓴᖅᔪᓚᖅᓴᕐᑕᕐᑐᓂᓂ, ᐅᓪᓗᐃᑦ 30 ᐅᖃᖅᐳᖃᖅᑎᑕᐅᓕᒪᑕ ᑭᐳᔪᑐᐃᓇᐃᑦ ᐃᓱᒪᓕᐅᒥᓂᖅᑎᑦ ᑲᓇᑕᒥ ᓴᖅᓚᐳᔪᓂᓂ, ᐃᓚᖓ 1. ᑭᖑᓪᓕᓴᕐᑕᒥᓕ ᑲᒪᓕᓪᓗᑎᓂ ᓴᖅᓄᓂ ᑲᑎᒪᔨᖏᑦ ᑎᑎᕋᖅᑭᔪᓕᓪᓗᑎᓂ ᐃᓱᒪᓕᐅᒥᓂᖅᒥᓂᓂ. ᓴᖅᓄᓂ ᑲᑎᒪᔨᓄᑦ ᑎᑎᕋᖅᑭᔪᓂᐊᖅᓂᖅᑲᑦ ᓂᖏᐊᑎᓂᓂ, ᓴᖅᓚᕐᑐᖅᓯᒪᓕᖅᓂᖅᓴᖅ ᓴᖅᑭᖅᑐᓕᕆᓂᖅ ᐃᓱᒪᓕᐅᒥᓂᖅᑎᑦ ᐃᓄᐃᑦᑕᐅᓂᖅᑎᑦ ᓴᖅᑭᖅᑐᓕᕆᓕᓪᓗᑎᓂ ᑲᑎᒪᔨᖏᑦ ᓴᖅᓄᓂ II–ᒥ.

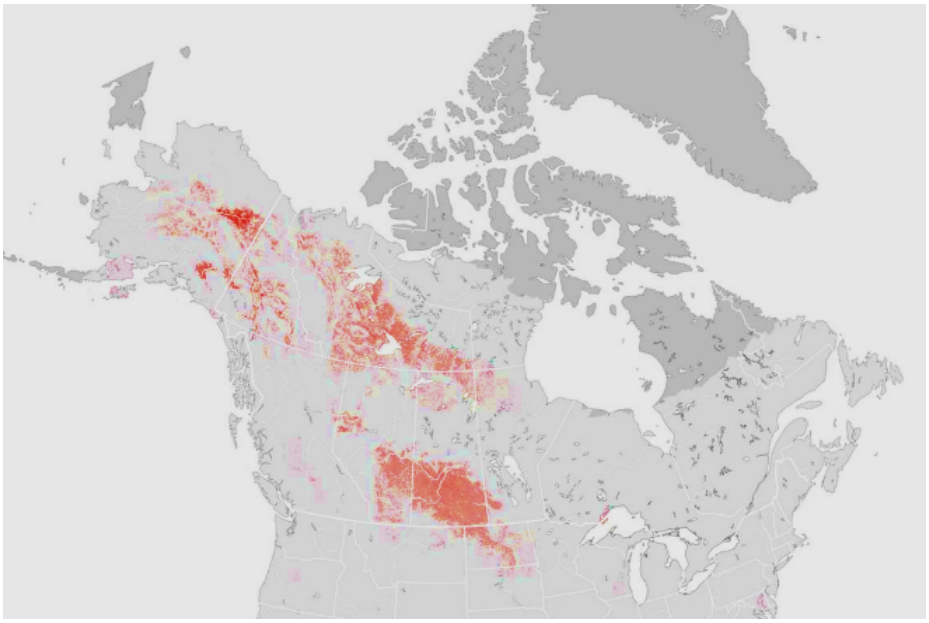
ᐃᓱᒪᓕᐅᕆᐊᓕᖅᓴᑦ:
ᐃᓱᒪᓕᐅᔾᔨᓴᓕᑕ ᐅᒪᔪᓕᕆᓚᓪᓕᔾᔨᐊᓕᕆᔨᓂᓂ (NWMB) ᑎᑎᕋᖅᑕᐅᔾᔨᓚᓕᑦ ᑲᑎᑦᓂᓕᑦ ᐊᒥᓱᓂᖏᓐᓇᑦ ᒥᑎᐅᔭᐅᓴᖅᑐᑦ (Horned Grebe) ᓴᖅᓚᒪᑐᖅᓴᔾᔨᓕᑐᓂᓂ ᓂᖏᐊᑎᓂᓂ ᐅᓄᕆᓂᕐᓄᑦᓂᓂ ᐃᓚᖏᑕᐅᓕᕆᓂᖅ s.5.2.34(f) ᐊᓯᒪᓚ 5.3.16-5.3.23.

ᑎᑎᕋᖅᑕᐅᔪᒪᔨᖅ: ᑲᓇᑕᒥ ᓂᖏᐊᓕᕆᓂᖏᓐᓇᑦ ᐱᔨᑦᑎᖏᖅᑎᓄᑦ, ᐅᑭᐅᖅᑕᖅᑐᒥ
ᐅᓪᓚᓐᓂ ᑎᑎᕋᖅᑕᐅᓂᖅᓴᓐᓄᑦ: 2025-08-22

ᐃᓕᒍᑕᖏᑦ A

ᐅᖏᓪᓗᓂᖏᑦ ᐃᓴᐅᔨᓐᓇᕐᓂᖏᓪᓗ

- ᑕᒪᓂᖅᐃ 92% ᑭᐳᔪᐊ ᒥᑎᐅᓴᐅᓴᖅᑐᑦ ᐃᒪᖏᐅᑕᐅᓯᖅ (Horned Grebe) ᐃᕐᓂᐅᕐᓇᖏᑦ ᑲᓇᑕᒫᒥᔪᑦ ᓇᒥᑕᐅᓐᓇᖅ
- ᐱᑕᖅᓯᑦ ᐊᐅᔾᑕᒥ, ᐳᓚᑎᑦ ᑲᓪᓚᐱ, ᒪᓂᑐᑦ, ᓄ ᐳᖏᔨᐃᐳ, ᓄᓇᑎᑎᐊᖅ, ᓄᓐᓂ ᔨᔪᓴ, ᓄᓇᓂᑦ, ᐊᓪᑎᐅᓚᔪ, ᔪᑦᐸ, ᓴᓴᑦᔨᐊᓪ, ᔫᑲᓪ, ᐊᑦᓯᓪᑎᐳ ᐃᒪᐃᖏᓂᓂ ᐊᒪᒪᓗ ᐳᔨᐃᐸᐃᖏᓂ.
- ᑕᒪᓂᖅᐃ ᓄᓇᓂᒥ ᓂᒋᐊᓂᒫᔨᓯᑦ, ᑭᓪᓕᖏᓪᓂ ᓄᓇᑎᑎᐊᖅ ᐊᒪᒪᓗ ᒪᓂᑐᑦ.
- ᑲᑎᑕᐅᓕᐅᖅᑎᓪᓇᒋᑦ ᒪᖅᑭᐳ ᐊᐃᖅᑕᑐᖅᔨᒪᔭᐣ, ᐱᖏᓪᓇᖏᓂᓕᑦ (ᐃᔨᒪᓗᒋᑲᐅᔪᑦ) ᓴᖅᑭᖅᑕᓂ ᓄᓇᓂᒥᒋᑦ.



ᑎᑎᖅᑐᑦᓴᖅᔨᔪᖅ 1 ᐃᕐᓂᐅᕐᓇᖅ ᐱᐅᓂᒃᑐᓪᓗ ᒥᑎᐅᓴᐅᓴᖅᑐᑦ ᐃᒪᖏᐅᑕᐅᔪᑦ ᑲᓇᑕᒥ.

ᓇᔨᖏᓪᓕᑦ

- ᒥᑎᐅᓴᐅᓴᖅᑐᑦ ᐃᒪᖏᐅᑕᐅᓯᖅ (Horned Grebe) ᐃᕐᓂᐅᕎᔨᒪᑕ ᓄᓇᑐᐃᓐᓇᖏᑦ ᓇᓵᖅᑐᖅᑭᖏᓪᑐᓂᖅ ᐊᔨᖏᓐᓂᓂᑐᑦ ᐊᔨᑦᑐᓐᓂ ᓇᓵᖅᑐᑕᓕᓐᓂᓪᓗ.
- ᑕᒪᑲᔨᐊ ᐊᔨᓐᓇᖏᔨᓂᖅ ᑕᓇᐅᖅᑭᓐᓂᑎᑦ ᒪᓂᒋᑎᐅᑕᐅᓕᒪᑕ ᐱᑭᖅᑐᖅᑭᖏᓪᑐᓂᖅ ᐱᓕᕐᓂᐊᖅᑐᓂᖅ ᐊᑐᖅᑐᖅᑭᖏᓪᑐᓂᖅ ᑕᔨᓂᐊᕐᓂᖅᑐᑦ ᓂᕐᑭᓂᔨᐅᓂᖅᑐᑦ.

ᐅᓗᓇᐊᓇᖅᑐᑎᑦᑐᑦ ᐃᕐᓂᐅᕐᓇᖏᑦ

- ᒪᓂᑦᑕᖅᓐᖏᑦᑎᐊᓕᖅᑐᓂ ᐱᑭᖅᑐᐃᑦ ᐱᔪᔨᑎᑦᓗᓐᖏᑦ, ᐃᒪᐃᓐᓗ, ᐅᑭᖅᑲᖅᔨᓂᐅᖅᑐᑦ ᐊᑐᖅᑐᖅᑭᖏᓪᑐᓂᖅ
    - ᓄᓇᖅᐸᔨᓱᑦᑐᓂᓂᓪ ᐃᒪᖏᑦ ᐅᖏᓪᓗᑦᑐᓂᑦ

- ᒪᑭᒪᔨᐱᑎᓴᐅᑦ ᐊᓯᔨᔪᖅᑐᑎᑦ, ᐱᓗᐊᖅᑐᒥᑦ ᒪᓴᐅᑦ ᓄᓇᐅᑦ
    - ᐊᔨᔪᐅᖏᑦᑐᓂᓗᓐᓗ ᐃᒪᖏᒥᑦ ᐱᑭᖅᑐᖃᓕᖅᑐᓂ ᐊᒥᓯᓐᓗ ᐃᒪᑐᐃᓇᖏᒥᑦ ᓂᖏᓴᓴᖅᓯᐅᖅᓯᑦᑐᑦ
    - ᐊᒥᓯᓐᓗ ᓂᖏᑦᑎᐊᓂᐊᕐᐅᑦ ᒪᓴᓄᓕᓚᖅᑐᑦ ᐊᔨᔪᐅᖏᑦᑐᑦ ᐱᑭᖅᑐᖃᖏᓂᖏᓄᓂ
- ᓯᓚ ᐊᓯᔨᔪᑦᓚᐊᓂᖅᖢᓗ
    - ᐸᓂᖅᐸᑦᓚᐊᓚᖅᑐᓂ ᒪᓴᐅᓂᖅᖢᓗ ᓄᓇ
- ᑐᖏᓂᓚᐊᖃᑦᑕᖅᑐᑎᑦ ᐅᐊᕐᓂᑦ ᓇᐸᑕᓄᑦᑐᓂᑦ ᐱᔨᔪᑎᖃᖅᑐᑎᑦ ᐊᓄᓇᒧᑦᓗ ᓄᑭᓴᐅᑎᓚᐅᖅᑐᐊᓕᑦᓗᓐᓗ

ᑐᓂᔨᐅᓯᒪᔪᑦ ᓄᓇᕗᑦ ᐅᒪᔪᓕᓗᐸᔪᕐᓂᐊᕐᓂᖅ ᖃᑎᒌᖏᑐᓄᑦ

ᐱᔪᔨᑎᖅᖅᑐᑎᑦ

ᐃᓱᒪᓕᐅᓗᐊᓕᖅ: X ᑐᑭᓯᓯᐊᑭᑎᕐᓂᖅ

ᐱᔪᑎᓕᖅ: ᓯᕗᓂᑎᓂᓂ ᐊᖏᑭᓯᓯᑕᐅᔪᓂᕐᓗᖅ ᐊᑐᓕᖅᑎᑕᐅᓂᕐᒧᑦ ᐅᓂᒃᑲᓂ ᐊᒪᒪᓗ ᐱᑎᑕᐅᑎᑦᓴᓄᑦ ᐸᖏᓐᓄᑦ ᑎᑎᕋᖅᓯᒪᔨᓪᓗᓂ ᓂᕐᔪᑎᓂ ᐅᓪᓗᓐᓄᑦᑐᐃᓐᓇᖅᑐᓂ ᐱᖏᓂᖅᓯᒪᔪᓄᑦ (Species at Risk Act)

ᖃᓄᐃᑦᑐᓯᒪᓕᖏᓐᓄᖅ:

- ᑕᐃᒪᓐᖏᓂᓂ ᑎᒍᓯᓚᐅᖅᓯᒪᓕᒪᑕ ᑲᙳᒪᑐᖅᑲᑦ ᓂᕐᔪᑎᓂᑦ ᐅᓪᓗᓐᓇᓚᖅᑐᐃᓐᓇᖅᑐᓄᑦ ᐱᖏᓂᖅᓯᒪᔪᓄᑦ (SARA; 2002), ᐅᖏᓪᒐᓱᓪᓗᐊᖅᑐᖅ 40–ᓂᑦ ᐊᒥᓯᖅᑎᑦᑎᐅᓐᓂᕐᓂᒧᑦ ᐅᓂᒃᑲᓚᐅᖅᓯᒪᔪᑦ (ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᖏᓐᓄᑦ ᐊᒥᓯᖅᑎᑦᑎᐅᓐᓂᕐᓂᒧᑦᓴᓪ ᐸᖏᓐᓄᑦ) ᐅᑭᐅᖅᑕᖅᑐᑉ ᓂᕐᔪᑎᖅᓯᒪᓄᑦ ᐅᓪᓗᓐᓇᓚᖅᑐᐃᓐᓇᖅᑐᓄᑦ ᐊᖅᑭᐅᓴᓯᐃᓯᒪᔪᑦ ᑎᑎᕋᖅᑐᓂᑎᓪᓗ ᐱᓕᕆᖅᑎᖅᑲᖅᑐᓂ ᐅᑭᐅᖅᑕᖅᑐᒥ ᓄᓇᖅᑲᐅᖅᑕᐅᓂᕐᓗᓪᓗ ᐊᐅᓚᑦᑎᓂᖅᑕᐅᔪᓂᖅ.
- ᑎᑎᕋᖅᑕᐅᔪᖅᑕᓚᐅᖅᑎᑦᓴᐃᒍ ᑕᒪᐃᓄᓐᖓ, <u>ᐱᔪᑎᓕᖅᑎᑦᓂᖅ ᐸᖏᓐᓄᑦ</u>, ᓇᓴᓇᐃᔨᐃᓚᐅᕐᒥᔪᑦ ᖃᓄᖅ ᐊᒥᓯᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᓂᐊᖏᓪᓗᒪᓐᖓ ᐸᖏᓐᓇᓴᑎᑦᓴᓪ, ᐊᒪᓚᓴᓐᓂᑦ <u>ᐊᑐᓂᖅᑎᑦᑎᓂᕐᒧᑦ ᐅᓂᒃᑲᓂ</u>, ᐅᓂᒃᑲᑦ ᖃᓄᖅ ᐊᑐᓂᖅᑎᑦᑎᓂᕐᒥᑦ ᐱᓇᓂᓕᒪᓐᖓ ᐊᒥᓯᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᓂᐊᖏᓪᓗᒪᓐᖓ ᑎᑎᕋᖅᓯᒪᔪᑦ, ᑕᐃᒪᐃᑦᑐᓂᐅᖅᑕᖅᑐᑦᑕᐊᖅᖏᓐᓄᑕ ᐊᖏᖅᑭᒍᑦ ᑕᑦᓂᓕᑦ ᐊᓂᔪᖏᓂᖅ.
- ᑕᒃᑯᐊ ᖃᓇᑕᒥ ᓂᕐᔪᑎᓕᓚᐅᖅᓯᒪᔪᑦ ᐱᓕᑎᑎᖅᑎᓄᑦ (CWS) ᐃᓕᓐᓇ ᖃᒪᓇᓂᓕᐊᔪᑦ ᐱᓕᓚᐊᓚᕕᐊᓕᓂᕐᓂᓐᓂ ᑕᒪᒃᑯᓂᖓᓪᓚᖅ.

<u>ᐱᔪᐊᑭᑎᖅᓴᐃᑦ ᐸᖏᓐᓄᑦᖏᑦ</u>

- ᐱᔪᐊᑭᑎᖅᓂᕐᓂᒧᑦ ᐸᖏᓐᓄᑦᑎᑕᖃᖅᑐᒃ ᒪᓕᑦᑐᑎᑦ 47 ᑕᐃᒃᑐᓂ ᓂᕐᔪᑎᓂᑦ ᐅᓪᓗᓐᓇᓚᖅᑐᐃᓐᓇᖅᑐᓄᑦ *ᐱᖏᓂᖅᓯᒪᔪᓄᑦ (2002)* ᐊᒥᓯᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᓂᐊᖏᓪᓗᒪᓐᖓ ᐸᓚᐅᖅᑎᑦ, ᐊᒥᓯᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᑐᓚᖅᑕᖅᑐᑦᑕᐊᖅᖏᓐᓄᑕ.
- ᐱᔨᔪᓂᕐᓂᑦ ᐱᔪᐊᑭᑎᓄᑦ ᐸᖏᓐᓄᑦ ᐃᑲᔪᖅᑐᐃᐱᒪᓴᓪᓗᓂᑦ ᑎᑎᕋᖅᑕᐅᔪᓂᕐᒧᑦ ᑭᐅᓂᖓᓄᑦ ᐊᑐᐃᓚᐅᓕᓂᖅᑐᐃᓐᓇᖅᑐᓂ. ᐊᑐᓚᐃᑲᓂᕐᒪᑦ ᐊᑐᐃᓚᐅᓕᓂᖅᑐᐃᓐᓇᖅᑐᓂ ᐸᖏᓐᓄᑦ ᐊᑐᓚᐅᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᓂᐊᖏᓪᓗᒪᓐᖓ ᐸᖏᓐᓄᑦ, ᐱᖓᓱᓕᔪᓂᑦ ᐱᔪᐊᑭᑎᓂᒃ ᐃᓚᓪᓕᑎᑦᑎᓂᕐᒧᑦ ᐊᑐᐃᓚᐅᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᓂᐊᖏᓪᓗᒪᓐᖓ ᐸᖏᓐᓄᑦ, ᐊᒪᓕᒪᓐᓇᓂᑦ ᐱᔪᐊᑭᑎᓂᒃ ᐅᓂᒃᑲᓂ ᐅᓪᓗᓐᓇᓚᖅᑐᐃᓐᓇᖅᑐᓄᑦᑐᓂᒃ, ᐅᓇᓱᖅᓯᓚᐅᓂᓕᓂᒃ ᓂᕐᔪᑎᓂᑦ ᐱᔪᐊᑭᑎᓂᒃ ᐊᑐᐃᓚᐅᖅᑎᑦᑎᐅᓐᓂᕐᓂᐊᓯᐊᖏᑦᑎᐅᓐᓂᕐᓂᑦ, ᐊᒪᓚᓪ

<ᖃᖅᑭᐅᓗᓂᑦ ᓇᐋᒃᖁᑦ ᑕᐃᒫᒃ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᓕᒪ ᓂᕆᐊᑎᑦ ᐱᑎᐊᑭᑎᓄᑦ <ᕆᓇᐅᑎᑦ ᐱᖃᓯᐅᓴᐃᓕᒥᐊᕐᓂᑦ ᖃᐅᔨᓴᖅᑕᑕᓚᐊᖅᕆᓂᖅᓂᓪᓗ ᐃᓄᐃᑦ ᒪᑭᒪᐊᑎᓴᖅᑎᑐᓄᑦ ᐊᑭᖅᑎᓂᖅ ᐃᑲᐋᑎᑎᓴᖅᑎᓪᓗ ᑕᒪᓐᓇ ᐊᑐᓕᖅᑎᑕᐅᑎᓪᓗᒍ.

- ᐱᑎᐊᑭᑎᓴᓄᑦ <ᕆᓇᐅᑎᑦ ᒪᓕᒫᒥᐅᑦ ᐊᐅᓴᐃᐅᓇᖅᑎᑐᓂᓪᓗ ᐁᖅᑭᒃᓯᒪᓕᒫᓂᑦ ᑲᑎᒪᔨᑎᑦᑎᓕᖅᑎᑦᑎᓂᑦ ᐊᐅᓚᑦᑎᔪᓐᓇᑦ <ᕆᓇᐅᑎᑎᓂᖅᓱᓕᑦᑐᑦ ᐊᒻᒪᓗ ᐊᒥᓯᑦᑎᑦᑎᑲᓂᖅᓱᓕᑦᑐᑦ ᐃᓕᓴᕐᓂᖅ ᐃᓅᓴᖅᑕᓗᒃᑐᑦ ᐊᒻᒪᓗ ᑐᓴᕐᓂᖅᓱᓕᑦᑐᑦ ᐱᑎᑦᑎᓂᖅ <ᐃᓚᒋᖅᑐᑦ ᐃᑲᔪᖅᑕᐅᓂᖅᓱᓕᑦᑐᑦ (NWMB) ᐅᖃᓕᒫᓐᓇᑐᑦ ᑕᐃᑲᓂ ᓂᕆᐊᑎᑦ ᐅᓇᓱᓕᑦᑐᑦ (SAR) ᑭᒃᑯᑐᐃᓐᓇᓕᖅᑎᑦ ᐊᑎᓕᐅᖅᓯᒪᓕᐊᓕᓐᓂᖅᑎᑦ.

<u>ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓯᓐᓂᖅᑎᑦ ᐅᓂᒃᑲᑦ</u>

- ᐊᑐᓕᖅᑎᑦᑎᓂᓕᕆᓂᑦ ᐅᓂᒃᑲᑦ ᐱᓵᓚᐊᓪᓕᑦ ᒪᓕᓕᑐᑎᑦ ᒪᓕᑎᕆᓂᑦ 46, 55, ᐊᑦᒪᓗ 72 ᑕᐃᒃᓂᓂ *ᓂᕆᐊᑎᓂᑦ ᐅᓗᓇᓚᓇᖅᑐᑐᑐᓄᑦ ᐱᕐᓂᖅᓴᓐᓂᖅ (2002)* ᐊᑦᒪᓗ ᐃᓄᑐᐃᓐᓇᓕᖅᑎᑦ ᓴᖅᑭᖅᑕᐅᐅᓕᓪᓕᑦ ᐅᑭᐅᓄᑦ ᑕᓪᓕᒪᑦ ᐃᓗᐊᐅ ᐊᒥᓯᑦᑎᑦᑎᓂᖅᓱᓕᑦᑐᑦ ᑎᑎᕆᖅᓯᒪᔪᓂᖅ ᓴᖅᑭᑦᑕᐅᐅᓕᓪᓕᑦ.
- ᑕᒃᑯᐊ ᐅᓂᒃᑲᑦ ᓇᐃᓇᖅᓯᒪᔪᑦ ᐱᓂᕐᓚᐊᓂᖅᑎᓐᓄᑦ ᐃᒪᒋᔭᕆᑎᓂᖅᓱᓕᑦᑐᑦ ᑲᑎᒪᔨᑎᑦᑎᓂᓕᕆᓂᑦ ᐊᒥᓯᑦᑎᑦᑎᓂᓕᕆᓂᑦ ᑎᑎᕆᖅᓯᒪᔪᓂᖅ, ᐊᑐᓕᖅᑎᑦᑎᓂᓕᕆᓂᑦ ᐱᑎᑦᑎᓂᖅ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᐊᒻᒪᓗ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ, ᒪᓕᑎᕆᓂᑦ ᐃᓅᓴᖅᑕᓗᒃᑐᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑎᑦ, ᐊᑦᒪᓗ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᓴᐅᓂᑎᓂᓂᓂ ᑐᓂᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᐊᐅᓚᑦᑎᔪᓐᓇᑦ ᐊᐅᓚᑦᑎᓂᖅᓱᓕᑦᑐᑦ, ᒪᓕᑎᕆᓂᑦ ᐃᓅᓴᖅᑕᓗᒃᑐᑦ ᐊᐅᓚᑦᑎᓂᖅᓱᓕᑦᑐᑦ.
- ᐅᓂᒃᑲᑕᐅᓂᖅᓯᓐᓂᓪᓗᒍ ᐊᑐᓂ, ᑐᓴᕈᒪᓂᑎᓂᓂ ᑐᓴᐃᒪᓴᖅᑐᒍᑦ ᐊᐅᓚᑦᑎᓂᖅᖢᑕᒥᓚᒪᓂᑲ ᓄᓇᕐᓂᓂᓪᓗ, ᐅᒪᔨᓂᐅᓂᖅᓱᓕᑦᑐᑦ ᐱᖓᓱᖅᑐᑦ (HTO/HTA), (ᐱᖓᓱᓂᓂ ᐊᕐᕌᒍᑦᑐᓂᖅᓱᓕᑦᑐᑦ ᐅᒪᔨᓕᕆᓂᑦᑎᑦ ᐅᒪᔨᓕᕆᓂᑦ ᑲᑎᒪᔨᑎᑦᑎᓂᓕᕆᓂᑦ (RWO), ᐅᒪᔨᓕᕆᓂᑦᑎᓂᓕᕆᓂᑦ (NWMB) ᑲᑎᒪᔨᖓᓂᑦ ᐃᓅᖅᓴᓇᐃᓴᖅᑎᖓᓂᑦ, ᑐᓵᖓᕐᓂᑦ (NTI) ᐊᑦᒪᓗ ᓄᓇᕗᑦ ᓴᖅᓯᒪᓂᑦ (GN) ᐃᓅᖅᓴᓇᐃᓴᖅᑐᑦ ᑐᑭᓯᓂᐊᕐᓂᖅᓯᒪᓂᑦ ᑐᓂᓯᒪᔨᖅᑐᖅᓯᒪᓂᑦ, ᑕᐃᒪᐃᑦ ᑐᑦᓯᑦᑎᓂᑎᒍᓐᓇᑦ ᖃᑎᖅᑕᒍᓐᓇᑦ ᐱᑎᐊᑭᑎᓂᓕᕆᓂᑦᑎᓂᖅᓱᓕᑦᑐᑦ, ᑕᐃᒪᐃᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᐱᑎᑦᑎᓂᖅᓱᓕᑦᑎᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᑕᐃᒪᐃᑦ ᑐᑦᓯᑦᑎᓂᑎᒍᓐᓇᑦ ᐃᓕᓴᕐᓂᖅᓱᓕᑦᑐᑦ ᑕᐃᒪᓐᓇ ᑕᐃᒪᐃᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᐱᑎᑦᑎᓂᖅᓱᓕᑦᑎᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᖅᓱᓕᑦᑐᑦ ᐊᐅᓚᑦᑎᔪᓐᓇᑦ ᐊᐅᓚᑦᑎ

ᐱᓕᓇᐊᖏᓂᐊᖅᑐᑦ ᓂᓪᓕᑦᓯᓐᖏᖅᑐᓕᓪᓗ ᐱᖅᑕᐅᔪᓂ). ᑕᒪᐅᑯᐊ ᓇᐃᓭᖅᓂᒪᓕᕐᓂ ᐃᓕᓴᐅᑎᓂᒪᖅᑕᕐᓂᐊᖅᑐᑦ.

ᐱᑎᐊᑭᑎᓴᓇᓂ ᐸᖏᓇᐅᑎᖓᑦ

- ᐱᑎᐊᑭᑎᓴᓇᓂ ᐸᖏᓇᐅᑦᑐᑦ ᑎᑎᕆᖅᑕᐅᓭᓕᔪᓂ ᓂᖏᐸᓕᓇᓂᒃᑎᑦ ᑕᐃᑲᓂ ᐃᓕᑎᔭᐅᔪᓂᐅᑲ ᓄᓇᓕᒻᒥ ᓄᓇᖅᐸᓗᒍᑦᑐᑦ (Peary) ᑐᑦᑐᐃᑦ.
- ᓄᓇᓕᑦ, ᐅᒪᔪᖏᓂᐊᖅᑎᓄᑦ (HTO/HTA), ᐊᐃᕐᑐᖅᓀᒪᔪᓂ ᓂᖏᐸᓕᓇᓂᖏᓐᓄᑦ, ᑲᑎᒪᑲᑦᑐᓂ ᐃᓗᐊᓂᐊᖅᑎᑦ, ᑲᓚᒪᑲᓂᓗ ᐃᓗᐊᓂᑦᑐᑦ ᖃᓄᑦ ᓄᓇᖅᐸᓗᒍᑦᑐᑦ (Peary) ᑐᑦᑐᐃᑦ ᖃᑎᖏᑭᔪᓇᖅᑎᑕᐅᓪᓚᕐᓂᑕ ᑎᑎᕆᖅᑕᐅᓀᒪᔪᓂᐅᑲ ᑐᓴᖏᖅᑐᓂᒪᓕᓇᓂᒃᑎᑦ ᐊᓐᓂᖅᑐᑦᑐᓂᒪᓕᓇᓂᒃᑎᑦ ᓄᓇᓕᒻᒥ ᐃᓐᓇᐃᓐᓇᖅᑐᑦ ᖃᑎᖏᑭᐅᔪᓂᐅᑲ ᐅᖃᓕᕐᓂᐊᖅᑐᑦ. ᑕᑯᓇ ᓂᓪᓕᖅᑕᐅᓀᒪᔪᖅ ᑐᓂᔅᐅᑦᑐᓂᔅᑐᖅ ᐅᒪᔪᓂᐅᑲ ᐃᓕᔪᓕᓇᕐᔪᐊᑲᓀᓂᒪᓂ (NWMB) ᐃᓱᔪᖅᑐᖅᑕᐅᔪᓚᓂᑦ ᓴᖅᑭᖅᑕᐅᓀᒪᔪᓄᑦ ᐃᓄᑐᐃᓇᓕᓄᑦ ᐅᖅᓱᐊᖅᑎᑦᑕᐅᓚᓂᓪᓚᑦ, ᒪᓕᓪᓚᓂᑦ ᒪᓕᑎᐊᓕᓇᓂᑦ ᓂᓪᓕᖅᑕᐅᓂᖓᓇ ᐊᐅᓪᓕᑎᓂᔭᓕᑦ ᐸᖏᓇᐅᓂᓄᑦ ᐊᑲᒪᓗ ᐊᒥᓪᓚᖅᑕᐅᓂᖏᓐᓇᐅᔪᓂᐅᑲᓕᓂᔭᓂᖓᓂᑦ ᐸᖏᓇᐅᓂᓄᑦ.

ᐊᑐᓕᖅᑎᑕᐅᓂᖓᓄᓇᓄᑦ ᐅᓂᒃᑲᑦ

- ᐊᑐᓕᖅᑎᑎᓂᔭᓕᑦ ᐅᓂᒃᑲᑦ ᓇᐅᔨᓴᓇᓄᑦ ᐊᒪᓚᓗ ᓇᐃᔨᑦ ᐱᔨᓵᔨᖅᑕᐅᓚᓂᑦ ᑎᑎᕆᖅᑕᑦ ᓴᖅᑭᖅᑕᐅᓚᓂᓕᓗ ᑕᒃᑯᓇᖓ ᓂᓪᓕᐊᑎᓂᒃ ᐅᓚᓗᐊᖅᑐᖅᑐᑎᓂᒃ (SAR) ᑭᒃᑯᑐᐃᓇᖅᑐᓄᑦ ᐊᑎᓕᐅᖅᓯᒪᔪᕆᐊᓕᖓᓄᑦ. ᐱᔨᓵᔨᖏᓚᓂᑦ ᖃᐅᔨᒃᐱᐃᓕᓂᑦ ᐃᓐᓇᐃᓐᓇᖅᑐᑦ ᐃᒪᔪᖏᓂᐊᖅᑎᓄᑦ (HTO/HTA), ᐊᐃᖅᑐᑦᑐᓂᒪᓕᓇᓂᒃᑎᑦ ᓂᓪᓕᐊᑎᓂᒃ ᑲᑎᑦᓴᖃᖅᑎᓂᒃᑐᓄᑦ, ᑲᓚᒪᑲᓂᓗ ᖃᐅᔨᓴᖃᖅᑎᓂᒃᑐᓄᑦ ᑕᒪᑲᑦᑐᓂᒪᓕᓇᓂᒃᑎᑦ.
- ᑐᑭᓯᓚᐊᑭᑎᓂ ᐱᔨᐅᓯᒪᓕᖅᑐᑦ ᑐᓴᔪᖏᓐᓄᐅᓯᒪᔪᓂ ᑕᒃᑯᓇ ᑕᐸᓕ (Dolphin) ᐊᒪᓚᓗ ᓱᓕᔨᐊᓕ (Union) ᑐᑦᑐᐃᑦ ᐃᓕᓴᐅᓯᒪᓕᕆᔨᖅᑐᑦ ᐊᑐᖅᑕᐅᓚᓂᑦ. ᑕᐃᒃᑯᓇ ᑐᓂᔨᓯᒪᔪᓂ, ᐅᑭᐅᖅᑕᖅᑐᑉᐸᑦ ᑲᓚᒪᑯᓕᖅᑐᓄᑦ (GN) ᐊᒪᓚᓗ ᐊᐃᖅᑐᖅᓀᒪᔪᓂ ᓂᓪᓕᐊᑎᓂᒃᑎᑦ ᑲᑎᑦᓴᖃᖅᑎᓂᒃᑐᑦ ᑲᑎᒪᔨᖏᓕᓂᑦ (ᐱᖓᓱᓂᒃ ᐊᐃᖅᑐᑦᑐᓂᒪᓕᓇᓂᒃᑎᑦ ᑲᑎᑦᓴᖃᖅᑎᖏᑦᑐᑦ (RWO), ᐅᒪᔪᓕᓇᕐᔪᐊᑲᓀᓂᒪᓂ (NWMB) ᖃᑎᖏᑭᐅᔪᓚᕐᓂᑕᕆᔨᓂᑦ ᑭᓱᖅᑐᐃᓇᓕᖅᑎᑦᑕᐅᓚᓂᑦ. ᖃᑎᖏᑭᑭᖅᑲᑕ, ᐊᑦᓚᐃᓂ ᐃᓕᓴᐅᓚᓂᑦ ᐱᔨᓵᔨᖏᓚᓂᑦ ᑕᒃᑯᓇᖓ ᓂᓪᓕᐊᑎᓂᒃ ᐅᓚᓗᐊᖅᑐᖅᑐᑎᓂᒃ (SAR) ᑭᒃᑯᑐᐃᓇᖅᑐᓄᑦ ᐊᑎᓕᐅᖅᓯᒪᔪᕆᐊᓕᖓᓄᑦ ᐱᔨᓵᔨᖅᑕᐅᓚᓂᑦ.

ᑎᑎᕆᖅᑕᐅᓀᒪᔪᖅ: ᑲᓇᑕᒥ ᓂᓪᓕᐊᑎᓂᒃᑎᑦ ᐱᓕᑎᑎᓂᖅᑎᓄᑦ, ᐅᑭᐅᖅᑕᖅᑐᑐᒥ
ᐅᓪᓚᖓ ᑎᑎᕆᖅᑕᐅᓂᖓᓇᓄᑦ: 2025-08-22

ᐃᓕᒍᑦ A

ᓇᐃᓴᖅᓯᒪᔪᑦ ᐱᑎᐊᑭᑕᐅᔾᔨᓇᐊᓕᓪᓕᑦ ᐸᖏᓐᓄᐅᑎᖓᑦ (AP)

ᐱᑎᐊᑭᑎᐊᓂᓄᑦ ᐸᖏᓐᓄᑐᑦ ᑕᐃᒪᐃᑎᐊᓕᓪᓕᑦ ᒪᓕᒪᖅ ᒪᓕᑦᑐᒍ 47 ᑕᕐᓴᓂ ᓂᖏᔪᓂᑦ ᐅᓗᓇᐊᓇᖅᑐᒋᑦᑐᓄᑦ ᐱᖏᓂᖅᓯᓐᓂ (2002) ᐃᓕᒥᖅᓯᐅᑎᒻᒪᑦᑕ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᑎᑎᕋᖅᓯᒪᔪᓕᕐᓂᓂ ᒪᓂᒋᑐᖅᓯᓂᓂ ᓂᖏᔪᓂᓂ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ, ᑕᐃᒪᐃᑎᐊᖅᑐᑐᒍᑎᓪᓗᑦᑕ. ᑎᑎᕋᖅᑕᐅᔾᔨᓇᓕᖅᑐᑦ ᓴᖅᑭᖅᑕᐅᔾᔨᒪᔪᑦ ᒪᓕᑦᑐᒍᑎᓂᓄᑦ ᑕᐃᒫᓱᓐᓇᕐᓕᖅ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ, ᐅᕐᓴᓕᔾᔪᓂᑦ ᓱᓕᔪᑎᒍᑦ. ᐱᔾᔪᑎᖅᑐᑐᒍᑦ ᒎᓕᒋᐊ ᐱᑎᐊᑭᑎᐊᓂᓄᑦ ᒪᓕᑦᑐᒋᓕᒪᓂᖏᑦ ᐸᖏᓐᓄᑐᑦ ᑎᑎᕋᖅᑕᐅᔾᔨᒪᔪᓂᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᐸᖏᓐᓄᑐᑦ ᐊᒻᒪᓗ ᓇᓗᓇᐃᔾᔨᖅᓯᒪᓂᖏᓐᓂᒃᑐᓂᓂᑦ ᐸᖏᓐᓄᑐᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ. ᒎᓕᒋᐊ ᓇᓗᓇᐃᔾᔨᐃᓯᒪᔪᑦ ᓱᓚᒋᓕᖅᐸᐅᑎᓕᖅᑎᓂᖏᓐᓂᒋᓕᓗ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᐱᑎᐊᑭᑎᓂᒃ ᑲᔪᓯᖏᓪᓂᒍᑦ ᑕᐃᑲᓂ ᓇᓗᓇᐃᔾᔨᖅᓯᒪᔪᓂᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ, ᐱᖓᓱᐅᓴᓱᒪᔪᑦ ᑲᒪᒋᔭᓐᓇᑦᑎᐊᓂᑦ ᐅᓗᓇᐊᓇᖅᑐᒋᑦᑐᓄᑦ, ᐅᓴᖅᓯᓗᓇᐊᓕᕐᓂᑦ ᓂᖏᔪᓂᑦ ᐊᒥᓱᑭᖅᑎᑕᐅᑲᑲᑎᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ, ᐸᖏᑭᓂᔭᓐᓇ ᓇᔪᒃᑲᓐᓇ ᓂᖏᔪᓂᓄᑦ

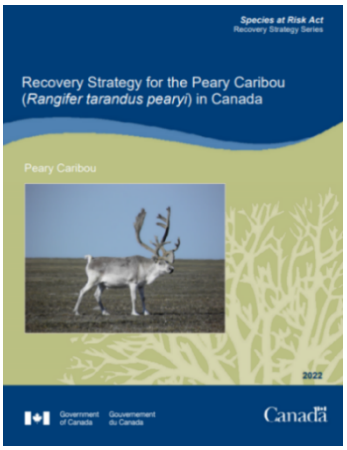
ᑲᔪᓯᕐᓴᐊᓂᔾᔨᓂᖅᓯᓐᓄᑦ ᓇᓗᓇᐃᔾᔨᖅᓯᒪᔪᑦ

- ᐱᑎᐊᑭᑎᐊᓂᓄᑦ ᐸᖏᓐᓄᑐᑦ ᑎᑎᕋᖅᑕᐅᔾᔨᒪᔪᑦ ᑲᓇᑕᒥ ᓂᖏᔪᓂᓚᓂᔾᔨᑦ ᐱᔾᔪᑎᕋᖅᑎᓄᑦ (CWS), ᒪᓕᑦᑐᒍᑎᓂᓄᑦ ᓴᖅᑭᖅᑕᐅᔾᔨᒪᔪᓂᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᑎᑎᕋᖅᓯᒪᔪᓕᕐᓂᑦ
- ᓱᓚᒋᖅᐸᒥ ᐃᖅᑲᖅᑐᐃᐃᓚᓚᓂᓗᑦᓯᑦ ᖃᐅᔨᓴᕆᔨᒪᓂᑦ (JR1) ᑎᑎᕋᖅᑕᓱᒪᔪᑦ ᐱᑎᐊᑭᑎᓄᑦ ᐸᖏᓐᓄᑐᑦ, ᑐᓴᕐᓇᖅᑕᒪᓕᑦᑐᒍᓐᓂᑦ
- ᑲᑎᑦᑎᓚᓂᓂᑦ ᐱᔾᔪᑎᕋᖅᑎᓄᑦ (CWS) ᐃᓕᓴᐃᒪᒍᑦ ᑲᒪᒋᐅᑎᒍᓐᓂᒃ ᐅᖅᑲᐅᓱᒥᓗᑦ
- ᐊᐃᐸᖏᓐᓄᑦ ᐃᖅᑲᖅᑐᐃᐃᓚᓚᓂᓗᑦᓯᑦ ᖃᐅᔨᓴᕆᔨᒪᓂᑦ (JR2) ᑎᑎᕋᖅᑕᐅᔾᔨᒪᔪᑦ ᐱᑎᐊᑭᑎᓄᑦ ᐸᖏᓐᓄᑐᑦ, ᐃᑲᔪᖅᑐᖅᑕᐅᔾᔨᔪᒍᓐᓂᒃ ᓴᖅᑭᖅᑕᐅᔾᔨᒪᔪᓕᕐᓄᑦ ᐃᓄᕐᓄᑦ ᐅᖅᑕᓄᓱᒪᓕᑦᑐᒍᓐᓂᑦ ᓇᓴᒪᓕᖅᑲᑦ
- ᑎᑎᕋᖅᑕᐅᔾᔨᒪᔪᑦ ᐱᑎᐊᑭᑎᓄᑦ ᐸᖏᓐᓄᑐᑦ ᓴᖅᑭᖅᑕᐅᔾᔨᒪᓂᑦ ᒎᓕᒋᓄᒃ ᐅᓗᓇᐊᓇᖅᑐᒋᑦᑐᓄᑦ (SAR) ᑭᐅᔨᑐᐃᓐᓇᓄᑦ ᐊᑎᓕᐅᖅᓯᒪᔪᐊᓕᕐᓄᑦ ᐅᕐᓗᓄᑦ 60–ᓄᑦ ᑭᐅᔨᑐᐃᓐᓇᐃᑦ ᐅᖅᑲᓐᓇᖅᑎᑕᐅᔾᔨᒪᓂᑦ
- ᑲᑎᑦᑎᓚᓂᓂᑦ ᐱᔾᔪᑎᕋᖅᑎᓄᑦ (CWS) ᑲᒪᒋᐅᑎᒍᓐᓂᑦ ᐃᓕᓴᐅᔾᔨᒪᓂᑦ ᐅᖅᑲᐅᓱᒥᓗᑦ. ᐱᓴᖅᓯᒪᓕᕐᓂᖅᑕᓗ ᓴᖅᑭᑦᓗᓂᑦ ᓂᖏᔪᓂᒃ ᐅᓗᓇᐊᓇᖅᑐᒋᑦᑐᓄᑦ (SAR) ᑭᐅᔨᑐᐃᓐᓇᓕᓄᑦ ᐊᑎᓕᐅᖅᓯᒪᔪᐊᓕᕐᓂᑦ ᐱᓴᖅᓯᒪᓕᕐᓂᓚᓂᑦ

*ᐅᓴᐅᓴᑦ ᑎᑎᕋᖅᓯᒪᔪᑦ ᐃᓕᓴᐅᓴᐅᔪᓐᓇᕐᓂᖅᓯᓐᓄᑦ ᐱᓕᕆᖅᑕᐅᔾᔨᒪᓂᑦ

ᓇᓗᓇᐃᔾᔨᖅᑕᐅᔾᔨᒪᔪᑦ ᑕᐃᑲᓂᒋᐊ ᓂᖏᔪᑎᑦ. ᐱᑎᐊᑭᑎᐊᓂᓄᑦ ᐸᖏᓐᓄᑐᑦ ᐱᖓᓱᐅᓴᐅᓴᒪᒥᔪᑦ ᐅᓴᐅᓴᕆᓕᕐᓂᖅᑐᓂ ᐃᓄᕐᓄᑦ ᐅᖅᑲᐅᓱᒥᓗᑦ ᐃᓕᓴᐅᔾᔨᒪᓂᑦ ᒪᑭᐅᑦᑎᓂᓚᓂᑦ ᐊᑭᐅᖅᓯᒪᔪᓂᑦ ᐃᓚᖓᑕᐅᓂᖅᓯᒪᔪᓂᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᐊᒻᒪᓗ ᓇᓗᓇᐃᔾᔨᖅᓯᒪᔪᓂᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᐸᖏᓐᓄᑐᑦ ᐊᒻᒪᓗ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᐸᖏᓐᓄᑐᑦ.

ᐱᑎᐊᑭᑎᐊᓂᓄᑦ ᐸᖏᓐᓄᑐᑦ ᒪᓕᑦᑐᖅᓯᓂᑦ ᑕᐃᒪᐃᑦᑐᓂᖅᓯᓐᓂᑦ ᐊᐅᓪᓕᑦᑎᓂᑦ ᐸᖏᓐᓄᑐᑦ ᐊᒻᒪᓗ ᐊᒥᓱᑭᖅᑎᑦᑎᐅᖅᓯᐊᓂᖏᓂᖅᓯᒪᓂᖏᑦ ᐸᖏᓐᓄᑐᑦ.



ᐱᕙᓪᓕᐊᓂᐊᓕᖅᑐᑦ ᐱᒋᐊᕈᑎᓄᑦ ᕿᖏᐅᑎᑦ

ᓄᓇᖅᕿᓱᒃᑎᑦ (Peary) ᑐᑦᑐᑦ – ᐱᒋᐊᕈᑎᓴᓇᓄᑦ ᕿᖏᐅᑎᑦ ᒪᓕᑦᑎᐊᖅᑐᑦ 2022 ᐊᒥᓱᕈᖅᑎᑦᑎᒃᓴᓂᕆᓱᐊᖏᓂᒍᑦ ᐅᓂᒃᑲᑦ

ᖃᓄᐃᑦᑐᓂᑦ ᓂᓚᐅᑎᓪᖃᖅᖏᓂᐊᖏᒪᖓᓕᖅᐱᑦ:

ᑕᒪᓐ ᖃᓂᑎᖓᓂᓗᐊᕐᓂᑦ ᑎᑎᕆᖅᓯᒪᔨᓂᓗ ᒪᓄᓄᖅᓯᒪᔨᓂᓗ ᐱᓄᓇᓕᖅᑐᑦ, ᐊᐱᓚᓴᐅᑎᓴᖅᑐᑎᑦ ᖃᑎᖅᓯᔨᓐᓇᖏᓪᓗᖓᓂᖅᐱᑦ ᐋᖅᑭᒃᑯᐊᓚᐊᒪᓂᓂᓂᑦ ᐱᒋᐊᕈᑎᓴᓇᓄᑦ ᕿᖏᐅᑎᓂᓕᖅ.

ᓇᐃᓴᖅᓯᒪᔪᓂ ᐊᑐᓕᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐅᓂᒃᑲᑦ (IR)

ᐊᑐᓕᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐅᓂᒃᑲᑦ ᑕᐃᒪᐃᑭᐊᓪᓕᑦ ᒪᓕᒐᖅ ᒪᓕᑦᑐᒍ 46, 55, ᐊᖏᒪᓗ 72 ᑕᐃᑲᓂ ᓂᕆᐊᑎᓂᑦ ᐅᓗᓕᐊᓇᖅᑐᒋᑦᑐᓗᓄᑦ ᐱᕆᐻᖅᓯᓂᐅᓂ ᐅᓂᒃᑲᓕᐅᑕᓪᓕᓪᓕᑦ ᐊᑐᓕᖅᑎᑦᑎᐅᓂᖏᓐᓄᑦ ᐊᒥᓱᑭᖅᑎᑦᑎᐊᖅᓂᓂᖏᓐᓄᑦ ᑎᑎᕋᕐᓂᒃᑲᐃᓐᓇᑦ (ᐊᐅᓪᓚᑦᑎᓂᖏᓐᓄᑦ ᕿᖏᐅᑎᓵᑦ, ᐊᒥᓱᑭᖅᑎᑦᑎᐊᖅᓂᓂᖏᓐᓄᑦ ᕿᖏᐅᑎᓵᑦ, ᐅᕐᓂᐊᔪᓂᒃ ᐱᒋᐊᕈᑎᓴᓄᑦ ᕿᖏᐅᑎᓵᑦ) ᐊᖏᒪᓗ ᐱᓂᓪᓕᐊᓂᕐᓂᓂ ᑲᓱᓂᔨᓪᓕᑦ. ᒪᓕᑦᑐᒍᑦ ᓂᕆᐊᑎᓂᑦ ᐅᓗᓕᐊᓇᖅᑐᒋᑦᑐᓗᓄᑦ ᐱᕆᐻᖅᓯᓂᐅᓂ (SARA), ᐊᑐᓕᖅᑎᑕᐅᓂᖏᓐᓄᑦ ᐅᓂᒃᑲᑦ (IR) ᓴᖅᑭᖅᑕᐅᖅᑕᑕᒪᐊᓪᓕᑦ ᑎᑎᕋᕐᓂᒃᑲᓗᑎᑦ ᐅᑭᐅᓄᑦ ᑕᑦᓪᓕᒪᑦ ᐃᓗᐊᓂ ᐊᒥᓱᑭᖅᑎᑦᑎᐊᖅᓂᓂᕿᓂᐊᓂᖏᓐᓄᑦ ᑎᑎᕋᕐᓂᒃᑲᐃᓐᓇᑦ ᓴᖅᑭᖅᑕᐅᖅᑕᓗᑎᑦ.ᑦ ᑕᒪᑲᑯᐊ ᐅᓂᒃᑲᑦ ᓇᐃᓴᖅᓯᒪᔪᓂ ᐱᓂᓪᓕᐊᓂᕐᓂᓂ ᐊᒥᓱᑭᖅᑎᑦᑎᐊᖅᓂᓂᕿᓂᐊᓂᖏᓐᓄᑦ ᑐᖓᔪᖅᑐᓂ, ᐊᓴᓱᑦᓗᑎᑦ ᐊᑐᓕᖅᑎᑎᓂᕐᓂᑦ ᐱᐅᓂᖅᓯᐊᓂᖏᓐᓄᑦ ᖃᖅᖢᓱᑯᓂᖅᓯᓂᒃ (ᑕᐃᒪᐃᑭᐊᖅᑕᖅᑲᒃᑕ) ᓇᓗᓇᐃᖅᓯᒪᔪᓂ ᐊᒥᓱᑭᖅᑎᑦᑎᐊᖅᓂᓂᕿᓂᐊᓂᖏᓐᓄᑦ, ᐅᖃᐅᓯᓪᓗᓂ ᐱᐅᓂᖅᑕᖅᑲᒃᑕᓪᓗᓂ ᑎᑎᕋᕐᓂᒃᑲᐃᓐᓇᑦ ᐊᑐᓕᖅᑎᑎᓂᕐᓂᑦ ᓂᕆᐊᑎᓂᑦ ᐊᑐᓕᖅᑎᑦᑎᐊᖅᓂᓂᕿᓂᐊᓂᖏᓐᓄᑦ ᐊᑐᓕᖅᑎᑎᓂᕐᓂᑦ ᐃᓚᖓᑕᐅᓂᖏᓐᓄᑦ, ᐅᖃᐅᓯᓪᓗᓂ ᐱᐅᓂᖅᑕᖅᑲᒃᑕᓪᓗᓂ ᑎᑎᕋᕐᓂᒃᑲᐃᓐᓇᑦ ᐊᑐᓕᖅᑎᑦᑎᐊᖅᓂᓂᕿᓂᐊᓂᖏᓐᓄᑦ ᐊᑐᓕᖅᑎᑎᓂᕐᓂᑦ ᐱᐅᓂᖅᑕᖅᑲᒃᑕᓪᓗᓂ ᐊᑐᓕᖅᑎᑦᑎᐊᖅᓂᓂᕿᓂᐊᓂᖏᓐᓄᑦ ᐊᑐᓕᖅᑎᑎᓂᕐᓂᑦ

ᖃᓄᐃᑦᑐᓂᑦ ᓂᓗᐅᑎᖅᖃᓂᐊᖏᓐᓇᖅᐱᑦ:

ᑕᒫᓂ ᖃᓂᑎᖅᓚᓕᕕᐊᕐᑎᑦ ᑎᑎᕆᖅᓭᒪᔪᓂᒃ ᓇᓪᓕᐊᓐᓂ ᓂᕆᔨᑎᓄᑦ, ᑐᑭᓯᑎᐊᑭᑎᓂᑦ ᑐᓂᓯᔾᐅᖃᖅᑕᑦᑐᑕᒃᖅᑐᑎᑦ ᓯᕗᓂᑎᓐᓂ ᐱᑎᐊᑭᑎᑦᓴᓂᒃ ᒪᓕᓪᑐᓄᓂ ᒪᓕᓪᓚᑎᓪᓚ ᐊᑲᑭᑎᐊᓇᐊᓕᓐᓂᒃ ᑭᖅᔪᓪᓕᐊᓂᖅᑐᓄᒃ ᐊᑐᓕᖅᖢᖅᐳᖅᓂᒃ ᐊᑐᓕᖅᑎᑕᐅᓂᕆᑦ ᐅᓂᒃᑲᑦ (IR) ᖃᑎᖅᑭᔨᓇᖅᓚᑦᑎᓴᖅᐱᑦ.