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

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ᓇᐃᓈᖅᓯᒪᔪᑦ

ᓇᐸᖅᑐᖃᙱᑦᑐᒥ ᑐᒃᑐᐃᑦ (*Rangifer tarandus groenlandicus*) ᓴᖅᑭᑕᐅᒃᑲᓂᓚᐅᖅᑐᑦ ᓴᓐᓂᕐᒧᑦ ᐊᒃᐸᑰᕐᔪᐊᒥᑦ ᑭᕙᓪᓕᕐᒥᑦ ᓄᓇᕗᒻᒥ 1968–ᖑᑎᓪᓗᒍ, ᐱᑕᖃᕈᓐᓂᑲᓯᓚᐅᖅᑎᓪᓗᒋᑦ ᓴᓪᓕᕐᒥ 1950 ᐱᒋᐊᓕᓵᖅᑎᓪᓗᒍ. ᖃᐅᔨᓴᖅᑕᐅᖏᓐᓇᕐᓂᖔ ᑖᒃᑯᐊ ᐊᑦᔨᒋᔭᐅᙱᑦᑐᖅ ᑐᒃᑐᐃᑦ ᓇᓗᓇᐃᖅᐳᖅ ᐊᓯᔾᔨᕐᔪᐊᖃᑦᑕᕐᓂᖔᓂ ᐅᕙᕐᓂᖏᑦ ᐊᒻᒪ ᓇᓂᑐᐃᓐᓈᕐᓂᖏᑦ ᐊᓯᔾᔨᖅᐸᒃᓗᑎᒃ ᑕᐃᒪᙵᑦ ᓴᖅᑭᑕᐅᒃᑲᓂᓚᐅᖅᑎᓪᓗᒍ ᓴᓪᓕᕐᒥ. ᓴᓪᓕᕐᒥ ᑐᒃᑐᐃᑦ ᐅᕙᖅᓯᒋᐊᓚᐅᖅᑐᑦ 48-ᖑᓚᐅᖅᓗᑎᒃ ᐆᒪᔪᑦ 1968-ᒥᑦ ᓴᖅᑭᑕᐅᓚᐅᖅᑐᑦ 29,425-ᖑᓇᓱᒋᔭᐅᓕᖅᓗᑎᒃ ᐆᒪᔪᑦ ᔫᓂ 1997-ᒥᑦ, ᑲᑎᕝᓗᒋᑦ ᐊᕐᕌᒍᑕᒫᑦ 23%-ᒥᑦ ᐅᕙᖅᓯᒋᐊᖅᓗᑎᒃ. ᑕᐃᒪᙵᑦ ᓴᖅᑭᑕᐅᒃᑲᓂᓚᐅᖅᑎᓪᓗᒋᑦ ᓴᓪᓕᕐᒥ ᑐᒃᑐᑕᐅᔪᑦ ᑭᒡᓕᖃᖅᑎᑕᐅᓕᓚᐅᖅᐳᑦ 1978–ᒧᑦ ᑎᑭᕝᓗᒍ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᑦ 1,138–ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ ᐃᓐᓇᐃᑦ ᐊᒻᒪ ᑎᕿᑐᕋᐃᑦ ᐅᕙᕐᓂᖏᑦ ᐅᑎᕐᔫᒥᓂᕋᖅᑕᐅᓚᐅᖅᐳᑦ ᐃᑲᔪᖅᑐᐃᔾᔪᑕᐅᒃᑲᓂᓕᖅᓗᓂ ᑭᒡᓕᖃᖅᑎᑦᑎᓕᖅᓗᑎᒃ ᓂᕿᒃᓴᕆᓂᐊᖅᑕᒥᓂᒃ ᐊᖑᓇᓱᒃᐸᒃᑐᓂᑦ. ᑐᒃᑐᐃᑦ ᐅᕙᖅᓯᕙᓪᓕᐊᖏᓐᓇᓚᐅᖅᐳᑦ ᐊᒻᒪ 1993-ᒥ ᓯᕗᓪᓕᖅᐹᒥ ᑮᓇᐅᔭᓕᐅᕋᓱᓪᓗᑎᒃ ᐊᖑᓇᓱᒃᑕᐅᔫᓇᖅᑐᑦ ᐊᖏᖅᑕᐅᓚᐅᖅᐳᖅ. 30-ᑲᓴᐃᑦ ᐊᕐᕌᒍᐃᑦ ᓈᓯᒪᓕᖅᑎᓪᓗᒋᑦ, ᐊᒥᓲᓂᖏᑦ ᖃᑦᑏᓐᓇᕈᕆᐊᖅᓯᒪᔪᑦ 29,425–ᖑᓇᓱᒋᔭᐅᓚᐅᖅᓗᑎᒃ ᔫᓂ 1997–ᒥ, 21,277–ᖑᓇᓱᒋᔭᐅᓕᖅᓗᑎᒃ ᔫᓂ 2005–ᒥ, 14,389–ᖑᓇᓱᒋᔭᐅᓕᕆᓪᓗᑎᒃ ᔫᓂ 2007–ᒥ, 13,651–ᖑᓇᓱᒋᔭᐅᓕᕆᓪᓗᑎᒃ ᔫᓂ 2009–ᒥ, 8,467–ᖑᓇᓱᒋᔭᐅᓕᖅᓗᑎᒃ ᔫᓂ 2011–ᒥ, ᐊᒻᒪᐃᖀᒃ 7,287–ᖑᓇᓱᒋᔭᐅᓕᖅᓗᑎᒃ ᒪᐃ 2013–ᒥᑦ. ᑐᒃᑐᐃᑦ ᐅᕙᕐᓂᖏᑦ ᓱᕐᕋᕙᓪᓕᐊᔪᓐᓃᓚᐅᖅᐳᑦ ᐊᒻᒪᓗ ᒪᐃ 2015-ᒥ ᑐᒃᑐᐃᑦ ᐅᕙᖅᓯᒋᐊᓚᐅᖅᐳᑦ 12,370-ᒧᑦ. ᑭᓯᐊᓂ, 2017–ᒥ, ᐅᕙᕈᓐᓃᖅᐹᓪᓕᒃᑲᓂᓚᐅᖅᐳᑦ 9,200–ᒧᑦ. ᐅᕙᕈᓐᓃᖅᐸᓪᓕᐊᑎᓪᓗᒋᑦ ᕿᓚᒥᕈᓗᒃ, ᑐᒃᑐᐃᑦ ᐃᓂᖃᓪᓗᐊᑕᓚᐅᖅᐳᑦ ᓂᒋᖅᐸᓯᐊᓂ ᕿᑭᖅᑕᐅᐸ ᖃᓂᒌᔭᓂᑦ ᑯᖕᒥ 2019–ᒥ ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᑦ ᑐᒃᑐᐃᑦ ᐅᕙᕐᓂᖏᑦ 12,054–ᖑᓚᐅᖅᐳᑦ, ᐊᔾᔨᐸᓗᒋᓪᓗᓂᐅᒃ 2015–ᒥ ᓇᓚᐅᑦᑖᖅᑕᐅᓚᐅᖅᑐᕈᑦ ᓱᕋᓗᐊᙱᓐᓂᕋᖅᑕᐅᓪᓗᑎᒃ. 2023–ᒥ, ᑐᒃᑐᐃᑦ ᐅᕙᕐᓂᖏᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓚᐅᖅᐳᑦ 12,651–ᖑᓇᓱᒋᔭᐅᓪᓗᑎᒃ ᓇᓗᓇᐃᖅᓯᓪᓗᑎᒃ ᓱᕋᓗᐊᙱᓐᓂᖏᓐᓂᒃ. ᐊᖑᓇᓱᒃᑕᐅᕙᒃᑐᑦ ᖃᔅᓯᐅᓇᓱᒋᔭᐅᔪᑦ ᑖᒃᑯᓇᓂᑦᑕᐃᓐᓇᖅ ᐊᔾᔨᒌᒃᐸᙱᓚᑦ. 2011-ᒥ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᑎᓪᓗᒋᑦ, ᐊᕐᕌᒍᓕᒫᒧᑦ ᑲᑎᕝᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔫᓇᖅᑐᑦ 1,000-ᓂᒃ ᑐᒃᑐᓂᒃ ᐊᑐᓚᐅᖅᐳᑦ 2012-ᒥ ᐊᒻᒪ 2013-ᒥ ᐊᖑᓇᓱᙵᓇᒥ, ᐊᒻᒪ ᑲᑎᕝᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔫᓇᖅᑐᑦ 800-ᓂᒃ 2014-ᒥ ᐊᒻᒪ 2015-ᒥ ᐊᖑᓇᓱᙵᓇᓂ. 2015–ᒥ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᑐᒃᑐᐃᑦ

ᐅᓄᖅᓯᒋᐊᖅᑐᑦ ᒪᓕᒃᓱᒋᑦ, ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐅᓄᖅᓯᒋᐊᓚᐅᖅᐳᑦ 1,600−ᓂᒃ ᐊᕐᕌᒍᑕᓖᑦ 2016−ᒥ ᐊᖑᓇᓯᖕᓇᑐᐊᒧᑦ. ᐅᓄᕈᓐᓃᖅᐹᓪᓕᖅᑐᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᒪᐃ 2017−ᒥ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕈᑕᐅᓚᐅᖅᐳᑦ ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᓂᑦ 1,000−ᒧᑦ ᐅᑎᖅᑎᓐᓇᓱᒃᓱᒋᑦ ᐅᕝᕙᓘᓐᓃᑦ ᓱᕐᕌᔫᓐᓃᕐᔫᒥᑐᑎᒃ ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᑦ. ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ, ᓇᑐᓇᐃᖅᑕᐅᔪᓂᑦ ᓱᕐᕌᙱᓐᓂᕋᖅᑕᐅᔪᓂᑦ ᐃᑲᔪᖅᑐᖅᑕᐅᓚᐅᖅᐳᑦ 2019−ᒥ 2023−ᒥᓗ, ᑲᔪᓯᑎᑕᐅᓯᒪᕚᖅ 1,000−ᓂ ᐆᒪᔪᓂᒃ.

ᐊᒃᑐᖅᑕᐅᓴᕋᐃᑦᑐᑦ ᐋᓂᐊᕐᓇᒧᑦ ᖁᐱᕐᕈᓄᑦ ᓯᕗᓕᖅᓱᕈᑎᖏᑦ ᐊᔾᔨᒌᖃᑦᑕᙱᓐᓂᖏᓐᓄᑦ (ᑐᒃᑐᐃᑦ ᐅᓄᖅᓯᒋᐊᓚᐅᖅᐳᑦ, ᐱᒋᐊᕐᓂᖓᓂᑦ, ᖃᔅᔮᓇᑯᓘᓚᐅᖅᓱᑎᒃ ᐊᒃᐸᑑᕐᔪᐊᖅ ᑐᒃᑐᖏᓐᓂᑦ) ᐃᓱᓖᓗᑕᐅᓯᒪᕚᖅ ᓴᖅᑭᑕᐅᒃᑳᓂᓚᐅᖅᑎᓪᓗᒍ ᑐᒃᑐᐃᑦ ᓴᓪᓕᕐᒧᑦ, ᐊᒻᒪᓗ ᐊᐃᑦᑐᕐᔫᑕᐅᑐᐃᓐᓇᓇᑎᐊᖃᕐᔭᒃᓴᓂ ᑐᑦᑐᓄᑦ ᓇᒡᒍᐊᕐᔪᖓᓂ ᓯᕗᓪᓕᖅᐹᒥ ᖃᐅᔨᔭᐅᓚᐅᖅᓯᒪᓪᓗᓂ ᑐᒃᑐᖏᓐᓂ ᕕᕝᕗᐊᕆ 2000-ᒥ. ᓇᒡᒍᐊᕐᔪᖓᓄᑦ ᐊᒃᑐᖅᑕᐅᓂᖏᑦ ᐅᓄᖅᓯᒋᐊᓚᐅᖅᐳᑦ 1.7%-ᒥᑦ ᕕᕝᕗᐊᕆ 2000-ᖑᑎᓪᓗᒍ 58.8%-ᒧᑦ ᒫᑦᓯ 2011-ᒥ ᐅᓄᖅᓯᒋᐊᕐᓂᖏᓐᓄᓪᓗ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕋᓱᒋᔭᐅᕙᑦ ᓄᕐᕆᐅᖅᐹᒃᑐᑦ ᖃᔅᓯᐅᓂᖏᑦ ᑖᒃᑯᓇᓂᑦᑕᐃᓐᓇᖅ. ᓄᕆᐅᖅᑐᑦ ᑲᑕᒋᐊᓚᐅᖅᐳᑦ 93.1%-ᖑᓚᐅᖅᓱᑎᒃ ᕕᕝᕗᐊᕆ 2001-ᒥ 37%-ᑐᐃᓐᓇᐅᓕᖅᓱᑎᒃ ᒫᑦᓯ 2011-ᒥ. ᖃᓄᐃᓕᕙᓪᓕᐊᓂᖏᑦ ᖃᐅᔨᓴᖅᑕᐅᔪᑦ ᓴᓪᓕᕐᒥ ᑐᒃᑐᖏᑦ ᐅᓄᕈᓐᓃᖅᐹᓪᓕᕐᓂᕋᖅᑕᐅᕗᑦ 9%−ᒥ ᐊᕐᕌᒍᑕᒫ 1997 ᐊᒻᒪ 2013 ᐊᑯᓐᓂᖓᓂ, ᑭᖑᓂᐊᒍᓪᓗ ᐅᓄᖅᓯᒋᐊᖅᓱᑎᒃ ᓄᓇᓕᖕᓂ ᐅᖃᐅᓯᐅᔪᑦ ᒪᓕᒃᓱᒋᑦ ᐊᒻᒪᓗ ᓇᑭᙶᕐᓂᖏᑦ ᐃᑲᔪᖅᓱᖅᑕᐅᓪᓗᑎᒃ ᐅᑭᐅᒃᑯᑦ ᐊᓯᐊᓄᐊᖅᐸᓪᓕᐊᔪᑦ 2014 ᐊᒻᒪ 2015 ᐊᑯᓐᓂᖓᓂ. ᑲᑎᓇᓃᑦ ᓱᕐᕌᓗᐊᙱᑉᐳᑦ 2015−ᒥ 2023−ᒧᑦ, ᓱᓕ ᐅᓄᙱᓐᓂᖅᓴᐅᕗᑦ ᑕᐃᔅᓯᒪᓂᓂ. ᐊᑐᓪᓚᖅᑕᕐᓂᖏᓐᓄᑦ ᑖᒃᑯᓂᖓ ᑐᒃᑐᓂᑦ ᓂᕿᒃᓴᓇᓪᓗᓂᔅᔪᒃ ᐊᒻᒪᓗ ᑮᓇᐅᔭᓕᐅᕋᓱᙳᕐᓂᕐᒧᑦ ᐊᖑᔭᐅᔪᑦ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ, ᒫᓐᓇ ᑲᑎᔅᓱᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᖃᐅᔨᓴᒃᑲᓂᕐᓗᑎᒃ ᐅᓄᕐᓂᖏᓐᓂᒃ ᐊᒻᒪᓗ ᑲᑎᓪᓗᒋᑦ ᐊᖑᓇᓱᒃᑕᐅᔪᓐᓇᖅᑐᑦ ᐅᓄᕐᓂᖏᑦ ᐱᓕᕆᐊᖑᔭᕆᐊᖃᖅᐳᑦ.

ᑕᑯᔭᐅᓯᒪᔪᑦ ᓇᒦᓐᓂᖏᑦ 2003−ᒥᑦ ᓇᐃᓴᐃᑎᓪᓗᒋᑦ ᓇᓗᓇᐃᓚᐅᖅᑐᑦ ᐊᓯᔾᔨᓗᐊᖅᓯᒪᙱᓐᓂᖏᓐᓂᑦ ᓇᒦᓐᓂᖏᑦ ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ ᑭᖑᓂᑦᑎᓐᓂ ᖃᐅᔨᓴᖅᑎᓪᓗᒋᑦ ᓄᓇᓕᖕᓂ ᐊᒥᓲᔪᓐᓃᖅᐸᓪᓕᖅᓯᒪᔪᑦ 1997−ᒥᑦ ᖃᐅᔨᓴᖅᑕᐅᓚᐅᖅᓯᒪᔪᑦ ᒪᓕᒃᓱᒋᑦ. ᑕᒪᐃᓐᓂᓪᓕ, ᑐᒃᑐᐃᑦ ᐊᑐᐃᓐᓇᓚᐅᖅᑐᑦ ᕿᑭᖅᑕᐅᑉ ᕿᑎᖅᐸᓯᐊᓂᑦ ᑰᑉ ᓇᖅᓴᖓᒍᑦ ᐊᒻᒪᓗ ᓅᑉᐸᓕᐊᑎᓪᓗᒋᑦ ᐱᖓᓐᓇᖅᐸᓯᐊᓂᑦ ᒪᓂᕋᐅᑉ ᑲᓇᖕᓇᖓᓂᓪᓗ ᖁᑦᑎᒃᑐᒥᑦ ᓄᕕᐱᓇ 1978−ᒥᑦ ᒪᐃ 2023−ᒧᑦ. ᐊᔾᔨᒌᙱᓐᓂᖅᑰᕐᔭᓚᐅᖅᑐᑦ ᐊᕐᕌᒍᓂᑦ ᐱᔾᔪᑎᖃᑐᐃᓐᓇᓇᐊᓕᑦ ᐊᐳᑦ ᖄᖓᓃᑦᑐᖅ ᐊᓯᔾᔨᕐᓂᖓᓄᑦ ᖁᐊᖅᐸᒃᑲᓂᐊᓂᖓᓄᓪᓗ, ᑐᒃᑐᐃᑦ ᓂᕆᖅᑕᕐᓂᖅᓴᐅᓕᖅᓱᑎᒃ ᐊᕐᕌᒍᓂᑦ ᐊᐳᑦ ᐊᐅᒃᐸᓪᓕᐊᓕᖅᑎᓪᓗᒍ ᐊᒻᒪᓗ/ᐅᕝᕙᓘᓐᓃᑦ ᐊᐳᑎᖃᙱᓐᓂᖅᓴᐅᓕᖅᑎᓪᓗᒍ. ᕿᑭᖅᑕᐅᑉ ᕿᑎᖅᐸᓯᐊᓂ, ᕿᑭᖅᑕᐅᑉ ᑰᖓᑕ ᖃᓂᒌᔭᓂ ᐊᒻᒪᓗ ᓅᑉᐸᓕᐊᑎᓪᓗᒋᑦ ᐱᖓᓐᓇᖅᐸᓯᐊᓂ ᒪᓂᕋᐅᑉ ᓄᓇᖏᑕ ᑲᓇᖕᓇᖓᓂ ᕿᑭᖅᑕᐅᑉ, ᐊᑐᖅᑕᐅᕐᔪᐊᖅᓯᒪᔪᑦ ᓴᓪᓕᕐᒥ ᑐᒃᑐᖏᓐᓂ ᐅᐱᕐᖔᒃᑯᑦ ᐊᒻᒪ ᓄᕆᐅᖅᑎᓪᓗᒋᑦ ᒫᓐᓇᒧᑦ ᑎᑭᓪᓗᒍ, ᑖᒃᑯᐊ

ᐃᓂᐅᔪᑦ ᐱᒻᒪᕆᐅᕗᑦ ᐊᑯᓂᐅᔪᒥ ᐊᓐᓇᐅᒪᔫᓐᓇᕐᓂᖏᓐᓂᑦ ᑲᑎᓇᓃᑦ. ᑖᒃᑯᐊ ᓄᓇᐃᑦ ᓴᐳᔾᔭᐅᓯᒪᓗᑎᒃ ᐃᓄᖕᓂᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᔪᓂᑦ ᐸᒡᕕᓴᒃᑕᐅᓗᑎᒃ ᐊᓯᔾᔨᖅᑕᐅᑐᐃᓐᓇᓇᐊᖅᑲᖅᓱᑎᒃ ᐊᒻᒪ/ᐅᕝᕙᓘᓐᓃᑦ ᐊᒃᑐᒃᑕᐅᑐᐃᓐᓇᓇᐊᖅᑲᕐᓗᑎᒃ ᓄᓇᐅᑉ ᑕᐅᑦᑐᖓ ᐃᓚᒋᔭᐅᓪᓚᕆᖕᓂᐊᖅᐳᖅ ᓇᓕᐊᖕᓄᑐᐃᓐᓇᖅ ᐊᑯᓂᐅᔪᒥᑦ ᑐᒃᑐᓂᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᖏᓐᓂ.

ᐊᐅᓚᔾᔪᑎᑦ ᐊᓯᔾᔨᕈᑕᐅᔪᑦ ᐊᒥᕉᓂᖏᓐᓂ ᑕᑯᔭᐅᖃᑦᑕᖅᑐᓂᑦ ᓇᐃᓴᐃᓂᓕᒫᖏᓐᓂ ᓴᓪᓕᐅᑉ ᑐᒃᑐᖏᓐᓂᑦ ᐊᒥᕈᕗᑦ, ᐊᔪᕐᓇᖅᓱᓂᓗ ᐃᓃᒃᑯᑦ ᑲᒪᒋᔭᓇᐊᒃᓴᖅ ᓈᓴᓇᐊᒃᓴᕐᓗ, ᖃᐅᔨᓴᒃᑳᓂᓇᐊᖅᑲᓕᖅᓱᑎᒃ. ᐱᔾᔪᑕᐅᓪᓗᐊᑕᖅᑰᔨᔪᑦ ᐋᓂᐊᓇᔭᐅᔪᖅ ᓇᒡᒍᐊᕐᓗᖕᓇᖅ ᖃᓄᐃᑦᑑᓂᖓ IV, ᐊᙳᓇᓱᒃᑕᐅᔪᑦ (ᑐᒃᑑᑉ ᓂᕿᖓᓂᒃ ᓂᐅᕕᐊᒃᓴᖃᕈᒪᓪᓗᐊᑕᖅᓱᑎᒃ ᐃᓄᖕᓄᑦ ᑐᓴᐅᒪᔾᔪᑎᒃᑯᑦ), ᐅᑭᐅᕐᓗ ᓯᓚᓗᖃᑦᑕᕐᓂᖓᓄᑦ, ᐱᓗᐊᖅᑐᒥ ᓂᓚᖃᑦᑕᓚᐅᕐᓂᖓ ᐃᓚᖏᓐᓂ ᐊᕐᕋᒍᓂᑦ. ᓇᑐᓇᙱᑦᑎᐊᖅᐳᖅ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᓇᐊᖅᑲᖅᐳᒍᑦ ᐋᓂᐊᓇᔭᐅᕙᒃᑐᒥᒃ ᐱᑕᖃᕐᓂᖏᓐᓂ ᓴᓪᓕᐅᑉ ᑐᒃᑐᖏᓐᓂᒃ ᑐᑭᓯᐅᒪᓂᐊᕈᑦᑕ ᐅᓪᓗᒥ ᐋᓂᐊᖅᑐᑦ ᐅᓄᕐᓂᖏᓐᓂᒃ ᐊᒃᑐᐊᓂᓕᖕᓂᓪᓗ ᐱᕙᓪᓕᐊᑎᑦᑎᓂᕐᒧᑦ ᑲᑎᙵᓂᕐᓄᑦ. ᖃᙳᑦᑎᐊᓵᖅ, ᐊᙳᓇᓱᒃᑏᑦ ᐅᖃᖃᑦᑕᖅᓯᒪᕗᑦ ᐅᓄᙱᓐᓂᖅᓴᐅᓕᕐᓂᖏᓐᓂᒃ ᑐᒃᑐᐃᑦ ᐋᓂᐊᖃᕋᓱᒋᔭᐅᔪᑦ, ᐅᔾᔨᕐᓇᖅᓱᓂᓗ ᐅᓄᖅᓯᕙᓪᓕᐊᓂᖏᑦ ᓄᕋᐃᑦ ᑕᑯᔭᐅᔪᑦ 2015–ᒥ 2023–ᒧᑦ ᐋᓂᐊᖃᖅᐸᒃᑐᑦ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᐊᑐᐃᓐᓇᓇᐊᖃᕐᓂᖏᓐᓂᑦ. ᑕᐃᒪᐃᓐᓂᖅᐸᑦ, ᐊᒻᒪᓗ ᓇᒡᒍᐊᕐᓗᒃᑐᑦ ᐅᓄᕈᓐᓃᖅᐸᓪᓕᐊᔾᔭᑕᐅᓗᐊᕈᓐᓃᖅᓱᑎᒃ, ᑕᐃᒪᓕ ᐊᙳᓇᓱᒃᑕᐅᕙᒃᑐᑦ, ᓯᓚᒧᓪᓗ, ᐊᒻᒪ ᖃᐅᔨᓴᖅᑕᐅᕙᓪᓕᐊᔪᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ, ᑲᒪᒋᔭᐅᖏᓕᓪᓗᐊᖅᑐᑦ ᓯᕗᓂᒃᓴᒥ ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᒌᕐᓗᑎᒃ ᓴᓪᓕᒥ ᑐᒃᑐᖏᑦ. ᐊᒻᒪ ᓯᓚ, ᐊᑑᑎᖃᑦᑎᐊᕐᓂᖅᓴᒥᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᓗᓂ ᐊᙳᔭᐅᕙᒃᑐᓂᒃ, ᐊᒻᒪ ᖃᓄᐃᑦᑐᓇᖅ ᐊᐅᓚᖅᑎᑕᐅᕙᒃᑐᑦ ᑐᒃᑐᐃᑦ ᓂᕿᖏᑦ ᕿᑭᖅᑕᒥ ᐱᒻᒪᕆᐅᓂᐊᖅᐳᖅ ᑐᑭᓯᐊᔭᐅᓇᓱᒡᒍᑎᒃ ᓱᓕᓂᖏᑦ ᐊᙳᔭᐅᕙᒃᑐᑦ ᑕᒪᐃᓐᓄᑦ ᓂᕿᒃᓴᒥᓄᑦ ᐊᒻᒪ ᓂᕿᓄᑦ ᓂᐅᕐᕈᑕᐅᔪᓄᑦ. ᒫᓐᓇᐅᔪᖅ ᑖᒃᑯᐊ ᐱᓕᕆᔾᔪᑏᑦ ᐊᑐᐃᓐᓇᐅᖏᒻᒪᑕ ᒪᓕᒐᓕᕆᔨᑦ ᐱᓕᕆᔨᓂᑦ ᓄᓇᕗᒻᒥ, ᐅᖃᖅᑐᑦ ᐃᓱᒪᒋᔭᐅᑲᓂᓇᐊᓖᑦ ᐋᖅᑭᐊᒃᑲᓂᖅᑐᒋᓪᓗ ᒫᓇᐅᔪᖅ ᐊᙳᓇᓱᒃᑕᐅᔭᓇᖅᑐᑦ ᒪᓕᒐᖏᑦ, ᐃᒻᒪᖄ ᓄᓇᕗᒻᒥ ᐊᖏᖃᑎᒌᒍᑎᒥᑦ, ᐃᓱᒪᒋᔭᐅᑦᑎᐊᓇᐊᓖᑦ ᐆᒪᔪᓂᑦ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᑐᔾᔨᖅᑎᕆᓄᑦ ᓄᓇᕗᑦ ᒐᕙᒪᖕᒐᓂᒡᒍ. ᐊᐅᓚᑦᑎᔾᔪᑕᐅᓇᓱᒃᑐᑦ ᑐᒃᑐᐃᑦ ᓂᕿᖏᓐᓂᑦ ᓂᐅᕕᐊᒃᓴᖅᐸᒃᑐᓂᑦ ᐃᓅᖕᓄᑦ ᑐᓴᐅᒪᔾᔪᑎᑎᒍᑦ ᑲᔪᓯᔪᓐᓇᓚᐅᙱᑉᐳᖅ ᒫᓐᓇ ᐊᐅᓚᑦᑎᔾᔪᑕᐅᔪᖅ ᒪᓕᒃᐳᒋᑦ ᑕᒪᓐᓇᓗ ᐊᑲᐅᙱᓕᐅᕈᑕᐅᔪᖅ ᐃᓱᒪᒃᓴᖅᓯᐅᕈᑕᐅᔭᓇᐊᖅᑲᖅᓱᓂ ᒪᓕᒐᐃᑦ ᐋᖅᑭᒋᐊᖅᑕᐅᓗᑎᒃ. ᑲᑎᒪᖃᑎᖃᓵᓚᐅᖅᑐᑦ ᑭᕙᓪᓕᕐᒥ ᓄᓇᓕᖏᓐᓂ ᐆᒪᔪᕐᓂᐊᑎᒃᑯᖏᓐᓂ, ᓄᓇᓕᓕᒫᑦ ᐅᖃᐅᓯᖃᓚᐅᖅᑐᑦ ᐱᓕᕆᐊᖃᕈᒪᓪᓗᑎᒃ ᐊᑲᐅᙱᓕᐅᕈᑎᒥᒃ ᑕᐃᒪᓇ, ᐅᖃᖅᑐᑎᒃ ᐃᓚᖏᑦ ᐊᖏᖃᑎᒌᒍᑕᐅᔪᑦ ᑎᑭᐅᑎᔭᐅᔭᓇᖅᒪᑕ ᑕᒪᓇ ᐋᖅᑭᒃᓯᒪᐃᓐᓇᓕᖅᑯᓪᓗᒍ. ᖃᓄᐃᓕᐅᖅᑐᖃᖅᐸᑦ ᖃᐅᔨᓴᕐᓗᑎᒃ ᑕᒪᑐᒥᖓ ᐅᓄᖅᓯᕙᓪᓕᐊᔪᓗ ᑐᒃᑐᐃᑦ ᓂᕿᖏᑦ ᓂᐅᕕᐊᒃᓴᖅᐸᒃᑐᑦ, ᐊᑲᐅᙱᓕᐅᕈᑎ ᐱᕙᓪᓕᐊᑐᐃᓐᓇᕐᓂᐊᖅᓵᕆᓯᒪᔭᕐ ᓄᓇᕗᒻᒥ ᓂᕿᖏᓐᓂᒃ ᐊᙳᓇᓱᒃᑕᐅᕙᒃᑐᑦ ᐅᓄᖅᓯᕙᓪᓕᐊᔪᑦ ᑐᒃᑐᐃᑦ ᓄᓇᕗᒻᒥ ᐊᐅᓚᑕᐅᓂᐊᖅᑐᑦ ᐋᖅᑭᒃᓯᒪᓗᑎᒃ ᓄᓇᓕᓕᒫᓂ.

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).

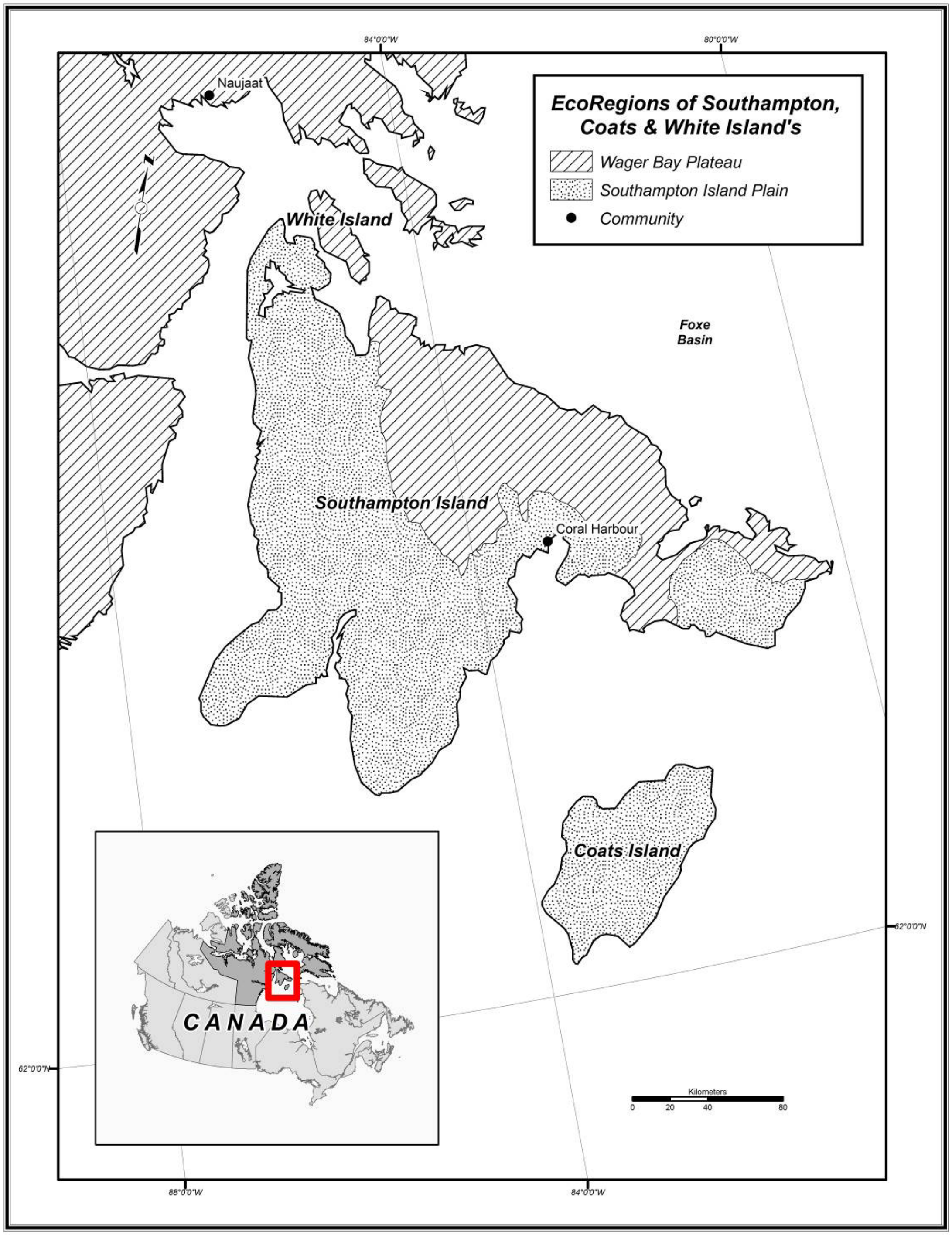

Figure 1 Ecoregions of the Southampton Island, Coats Island, and White Island study areas (Wiken, 1986; Natural Resources Canada, 2001).

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.

<u>March 1990 and 1991</u>

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.

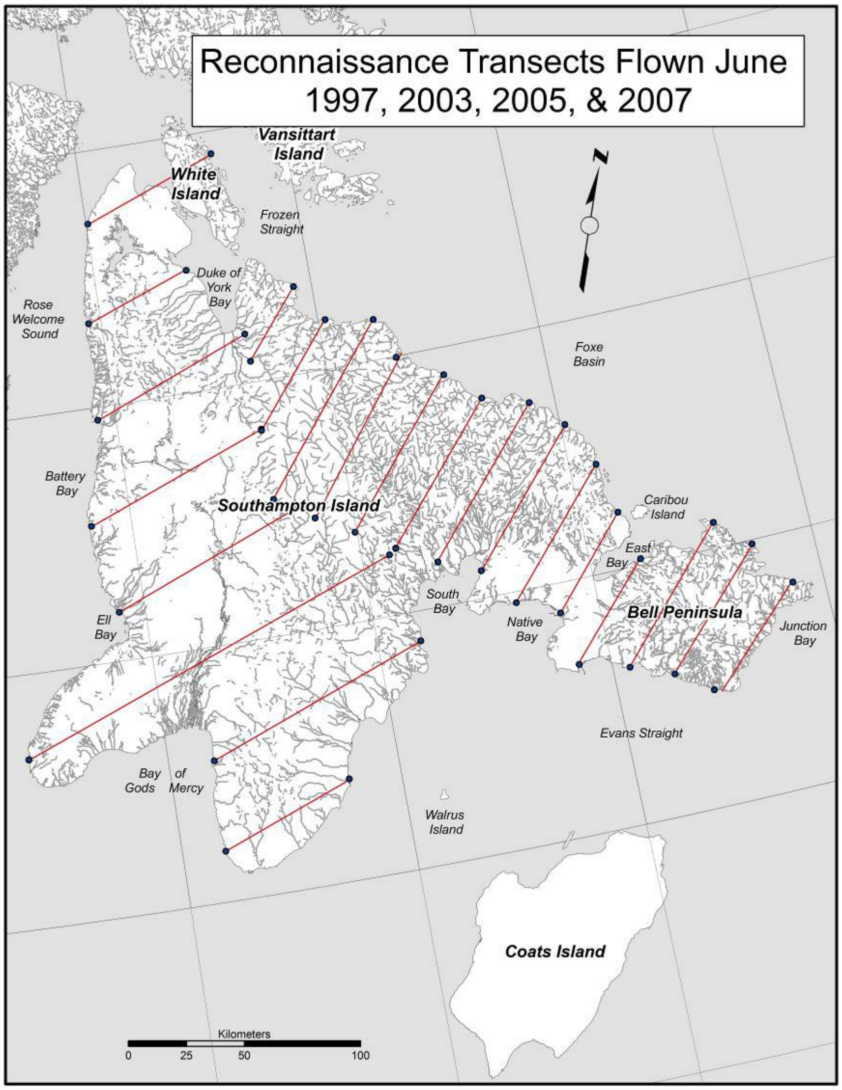

Figure 2. Reconnaissance transects flown in June of 1997, 2003, 2005, and 2007, to delineate abundance strata used to estimate Southampton Islands (including White Island) caribou population.

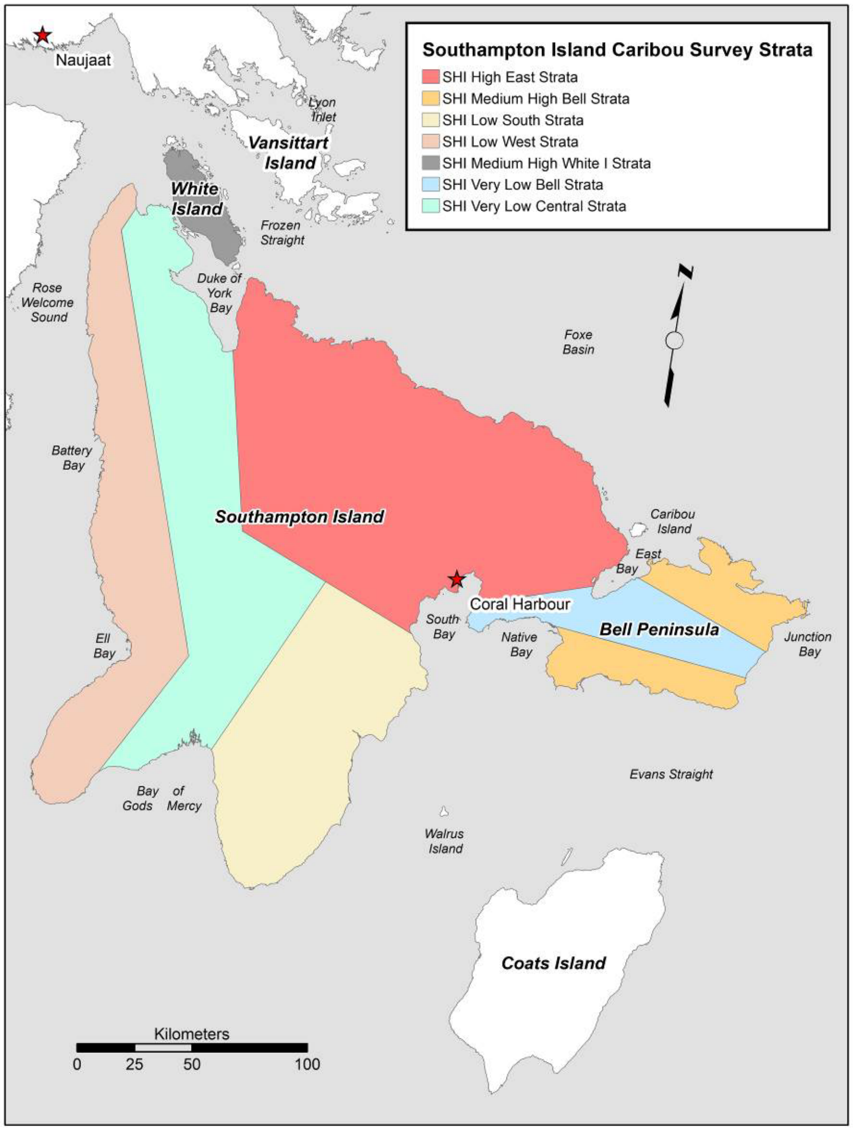

Figure 3. Abundance strata initially delineated using reconnaissance flights to map relative densities of caribou. As caribou distribution changed little across all survey years, these strata were utilized for all surveys post-2007.

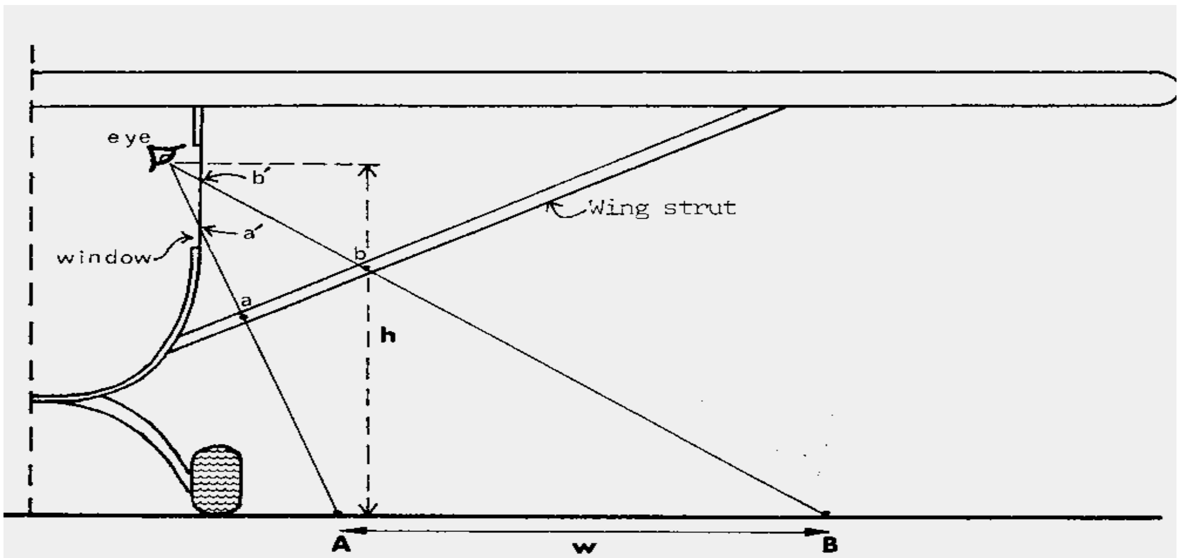

Figure 4. Schematic diagram of aircraft configuration for strip width sampling (Norton-Griffiths, 1978). W is marked out on the tarmac, and the two lines of sight a' – a – A and b' – b – B established. Streamers are attached to the struts at *a* and *b*. a' and b' are the window marks.

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

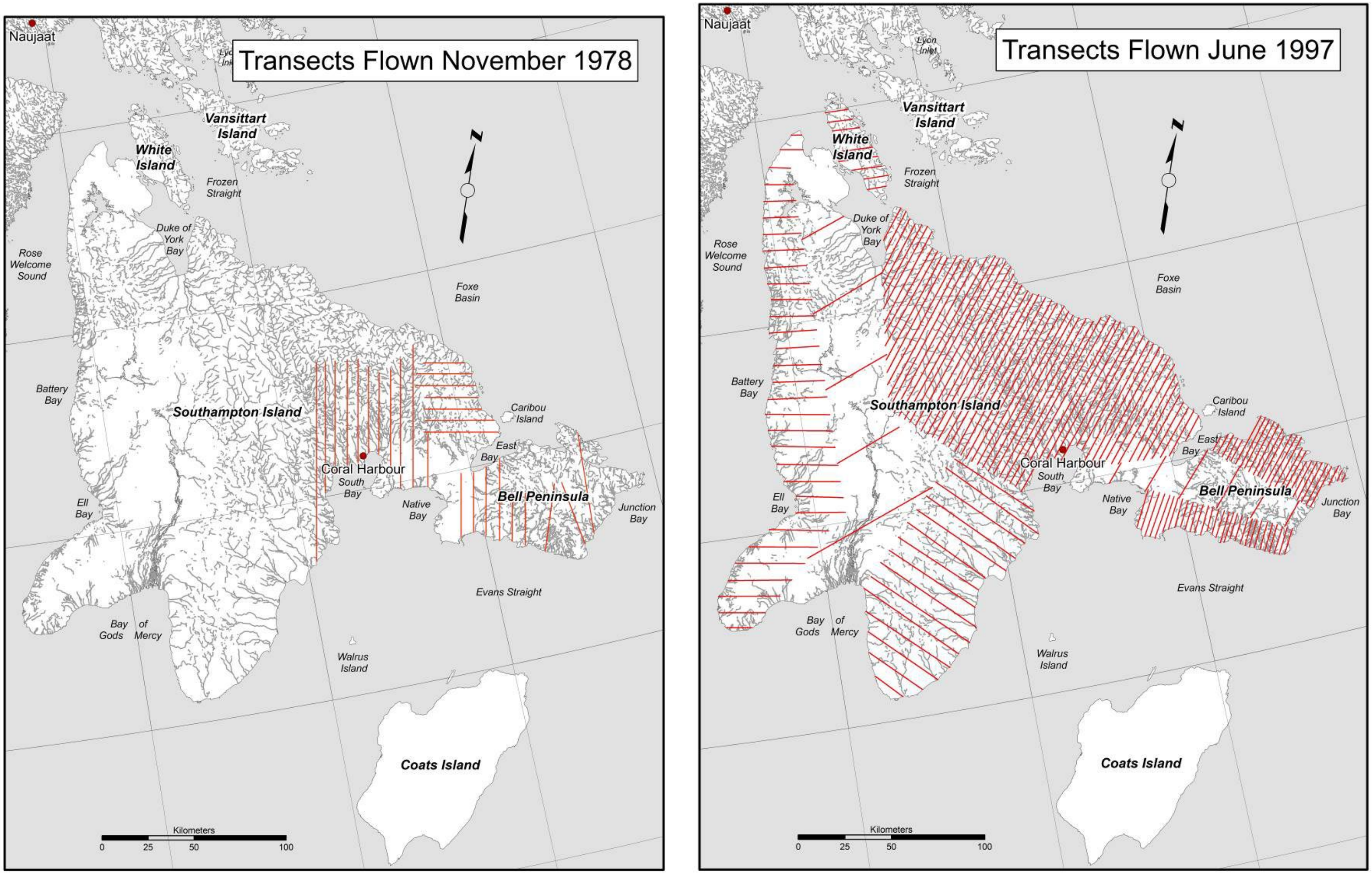

Figure 5. Stratified random transect surveys flown in November 1978 and June/July 1997.

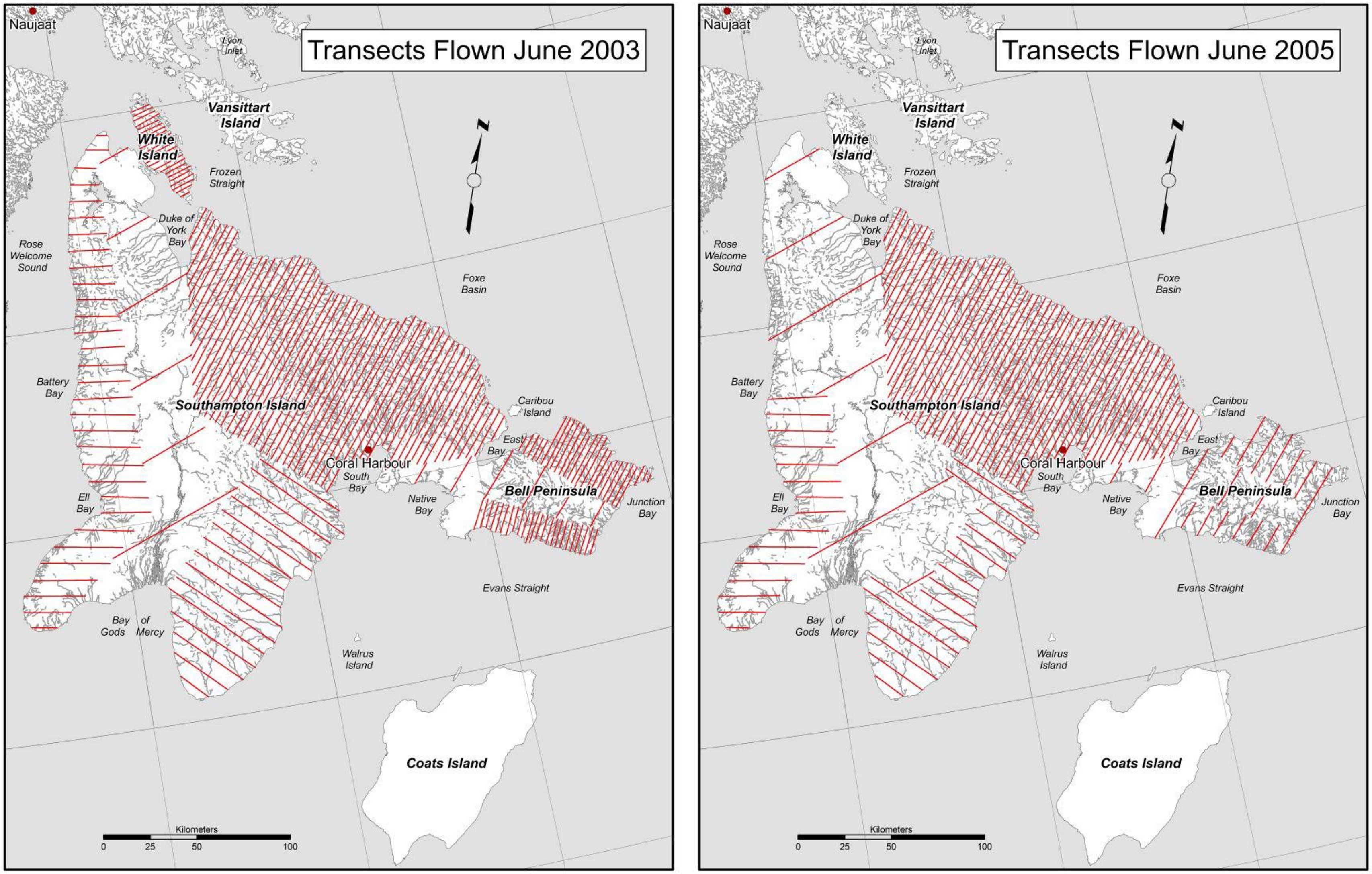

Figure 6. Stratified random transect surveys flown in June 2003 and 2005.

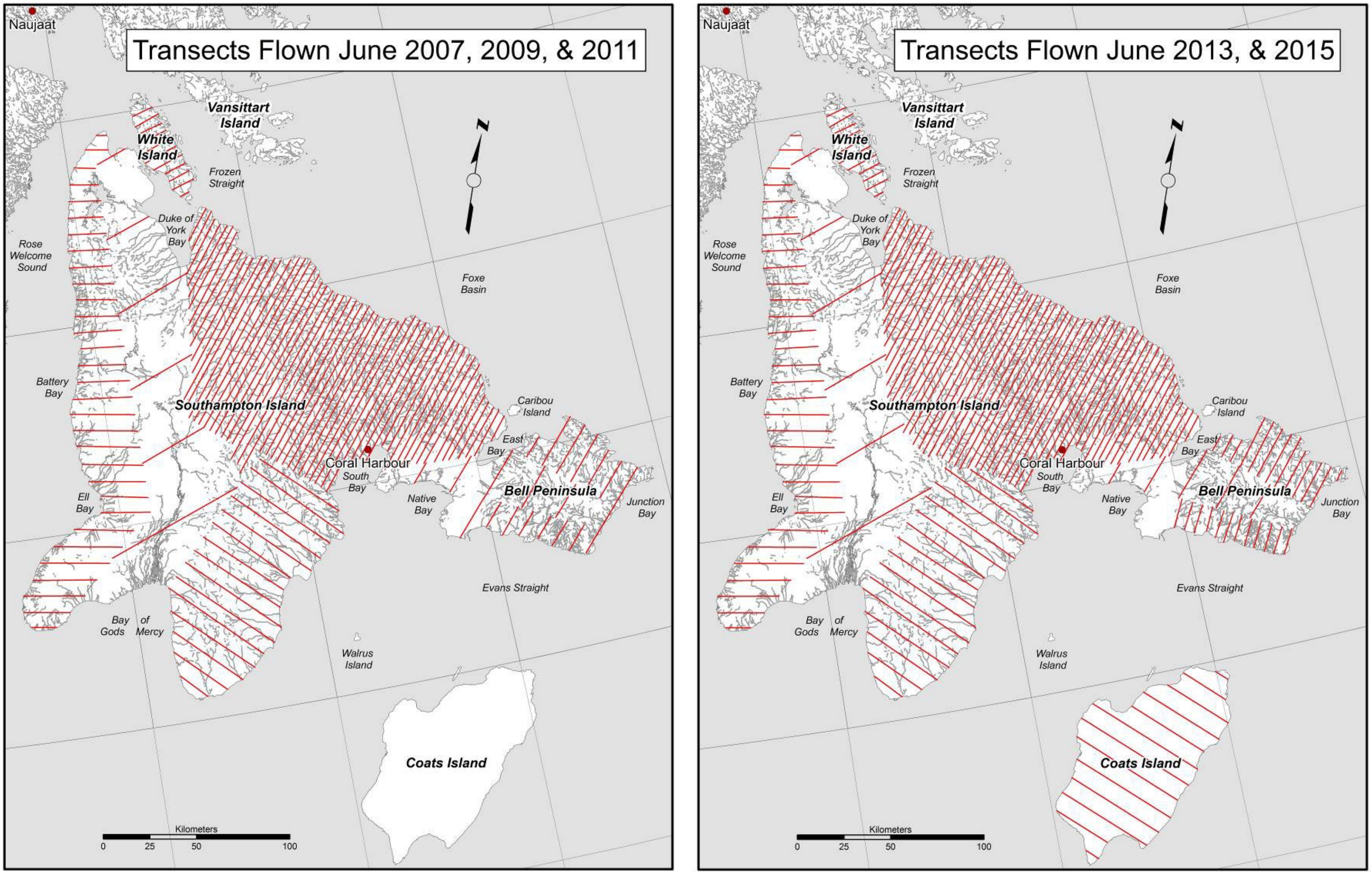

Figure 7. Stratified random transect surveys flown in June 2007, 2009, 2011 and May 2013 and 2015.

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.

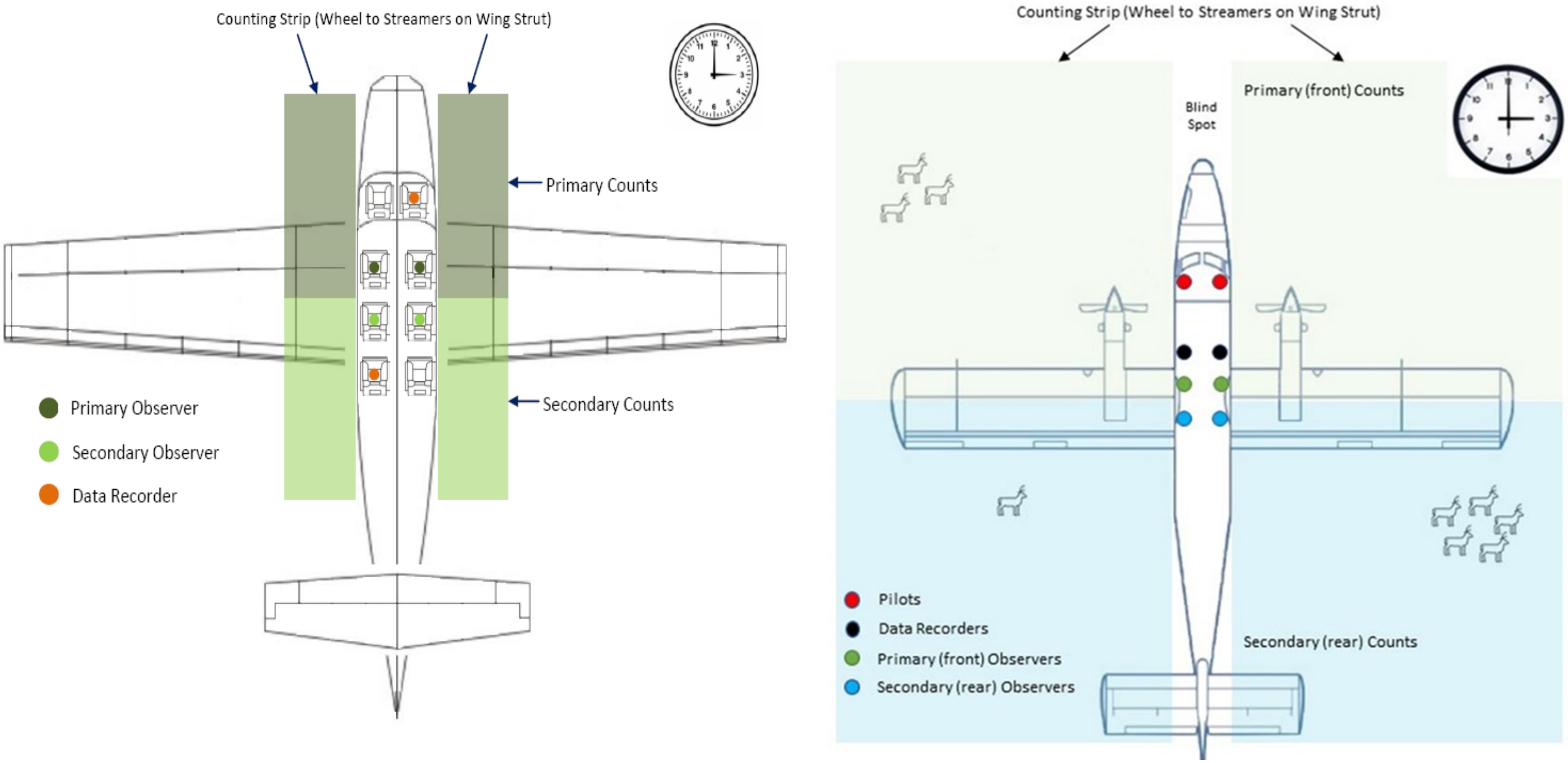

Figure 9. Observer position for the dependent double observer pair method employed on the 2011, 2013, 2015, 2017, 2019, and 2023 Southampton Island caribou abundance surveys for a Cessna caravan (left) and DeHavilland Twin Otter (right). The secondary observer calls caribou not seen by the primary observer 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.

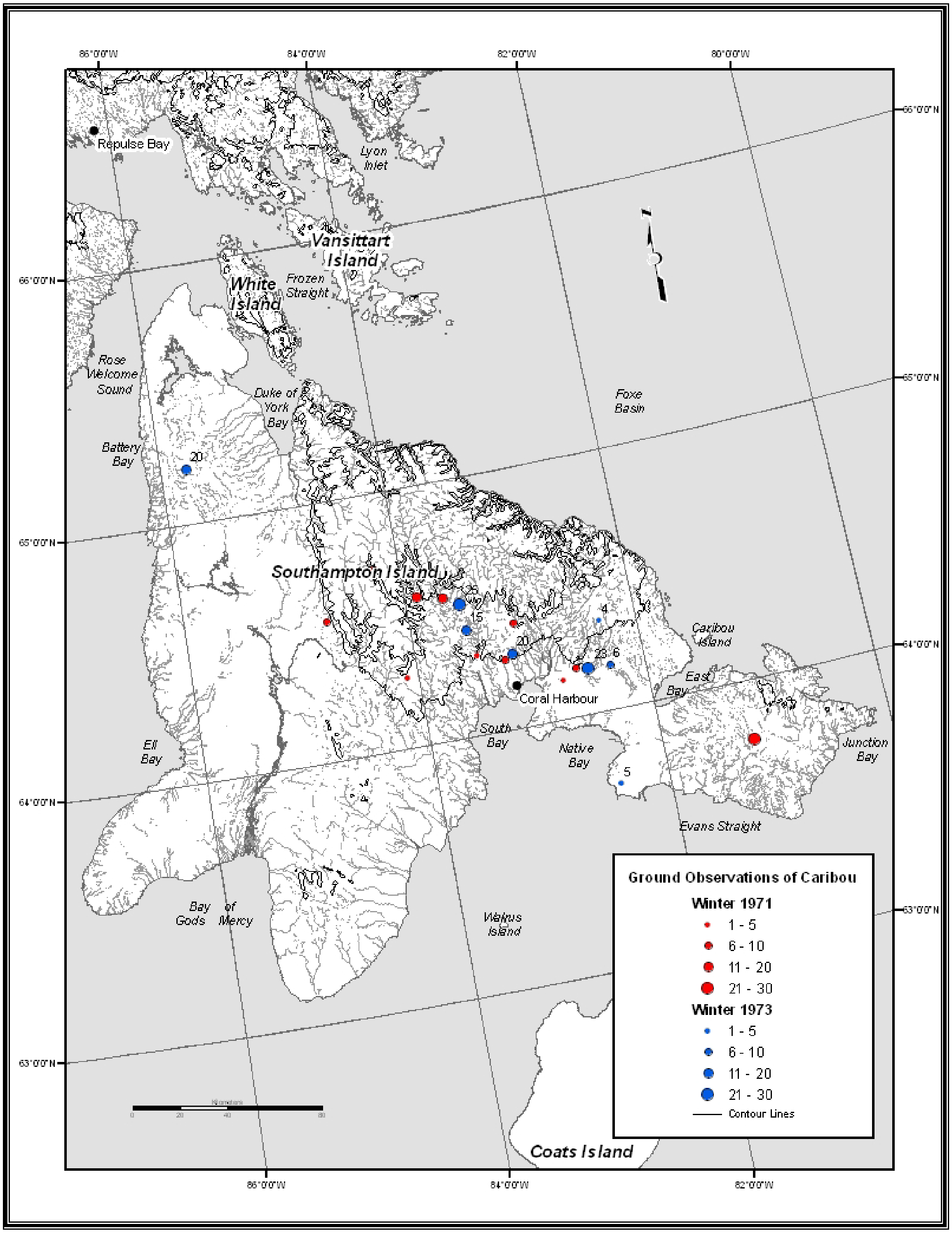

Figure 10. Observations of caribou on Southampton Island (1971 to 1973). Note that group sizes and locations were provided in 1973 reports, but not in 1971.

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%

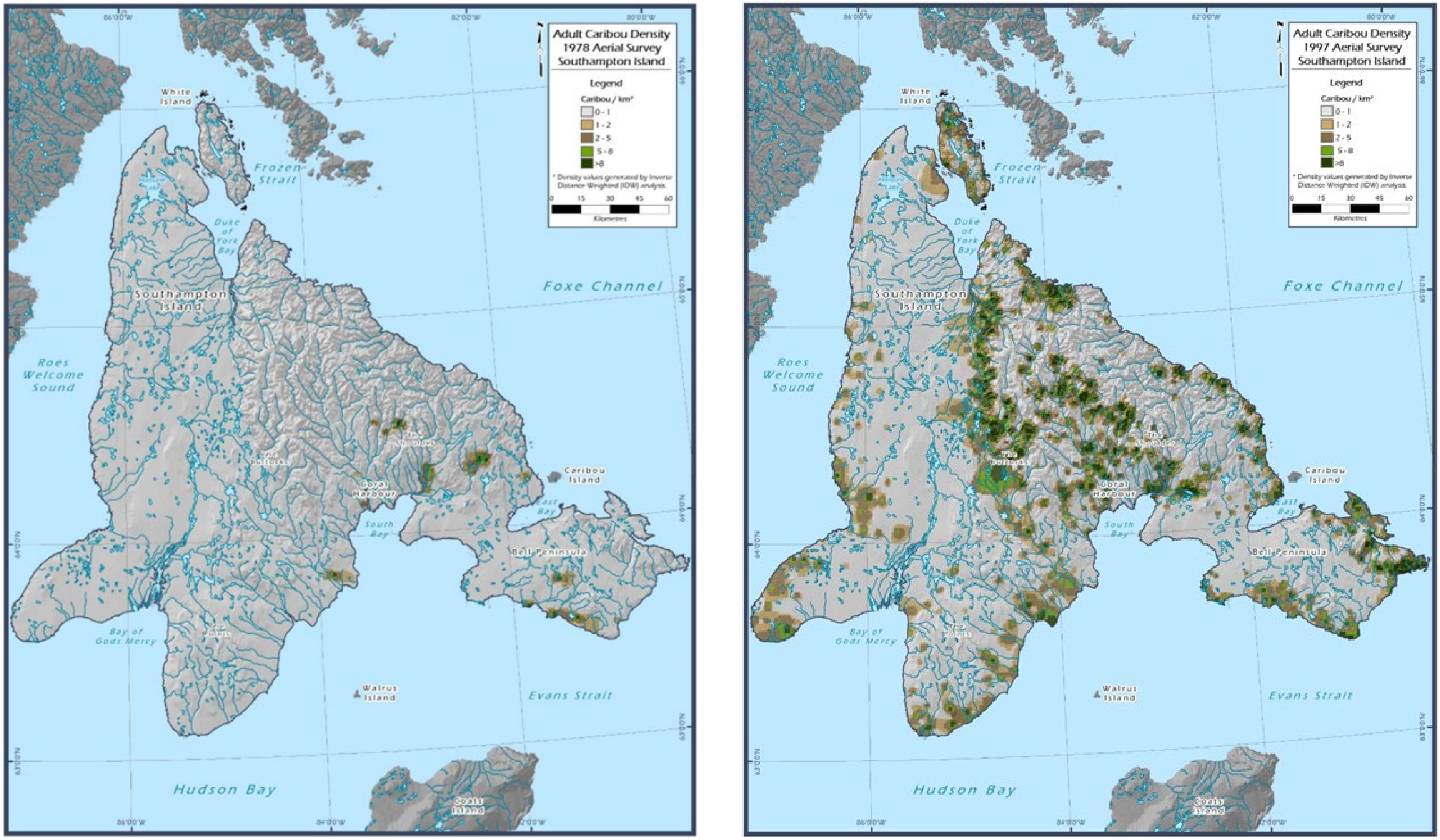

Figure 11. Results of the inverse distance weighted (IDW) interpolation technique applied to the November 1978 and June 1997 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

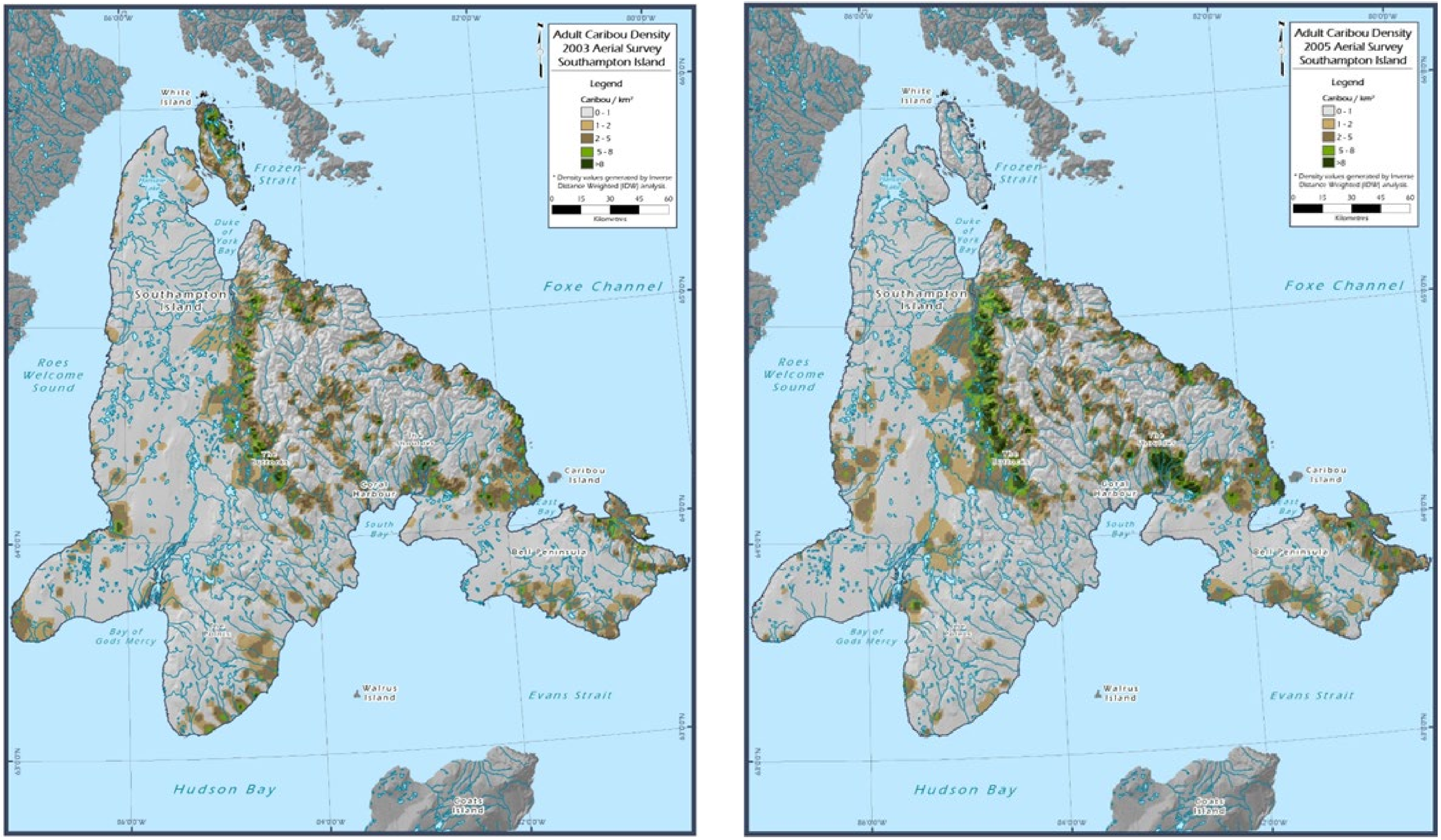

Figure 12. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2003 and June 2005 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

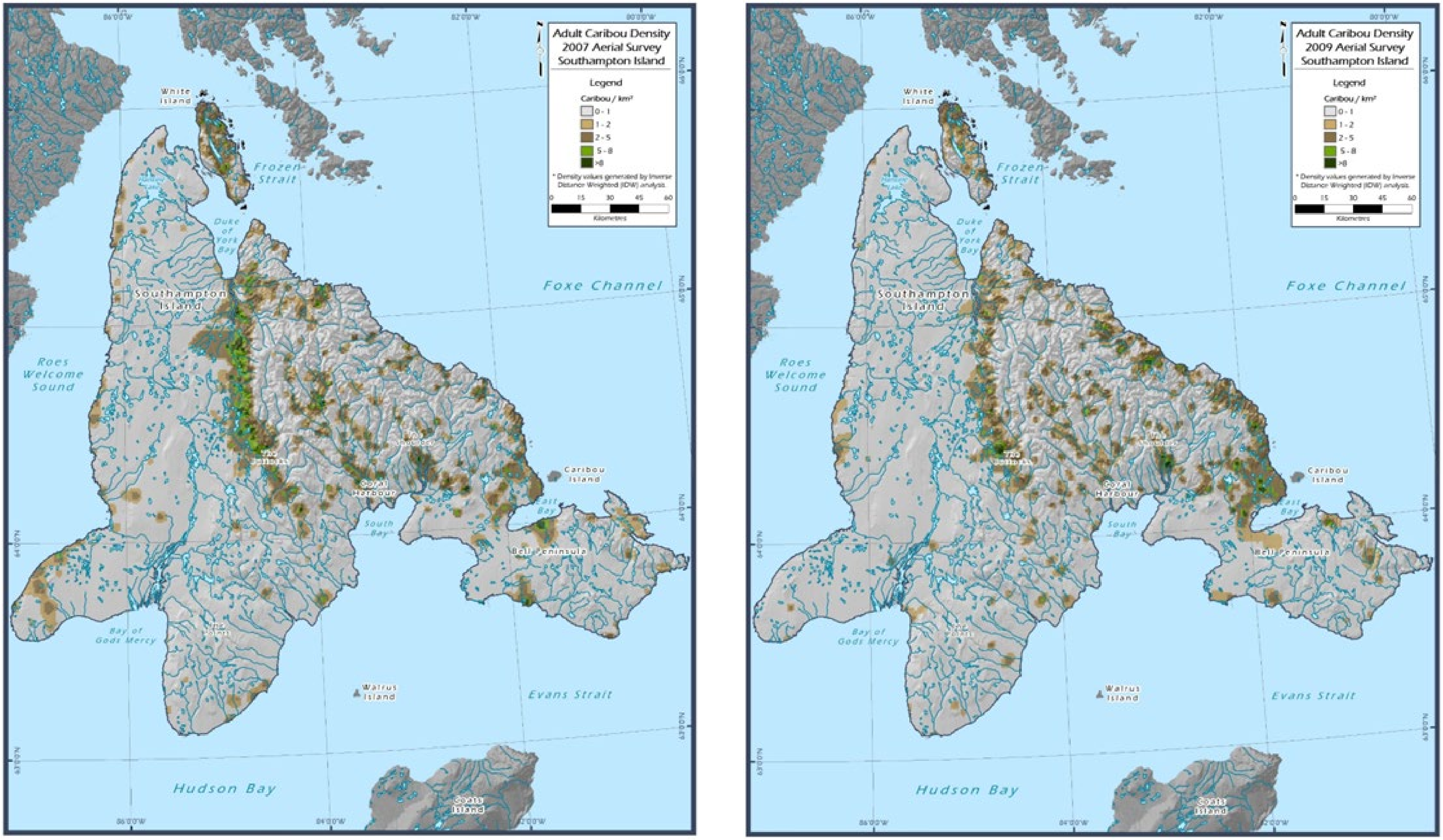

Figure 13. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2007 and June 2009 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

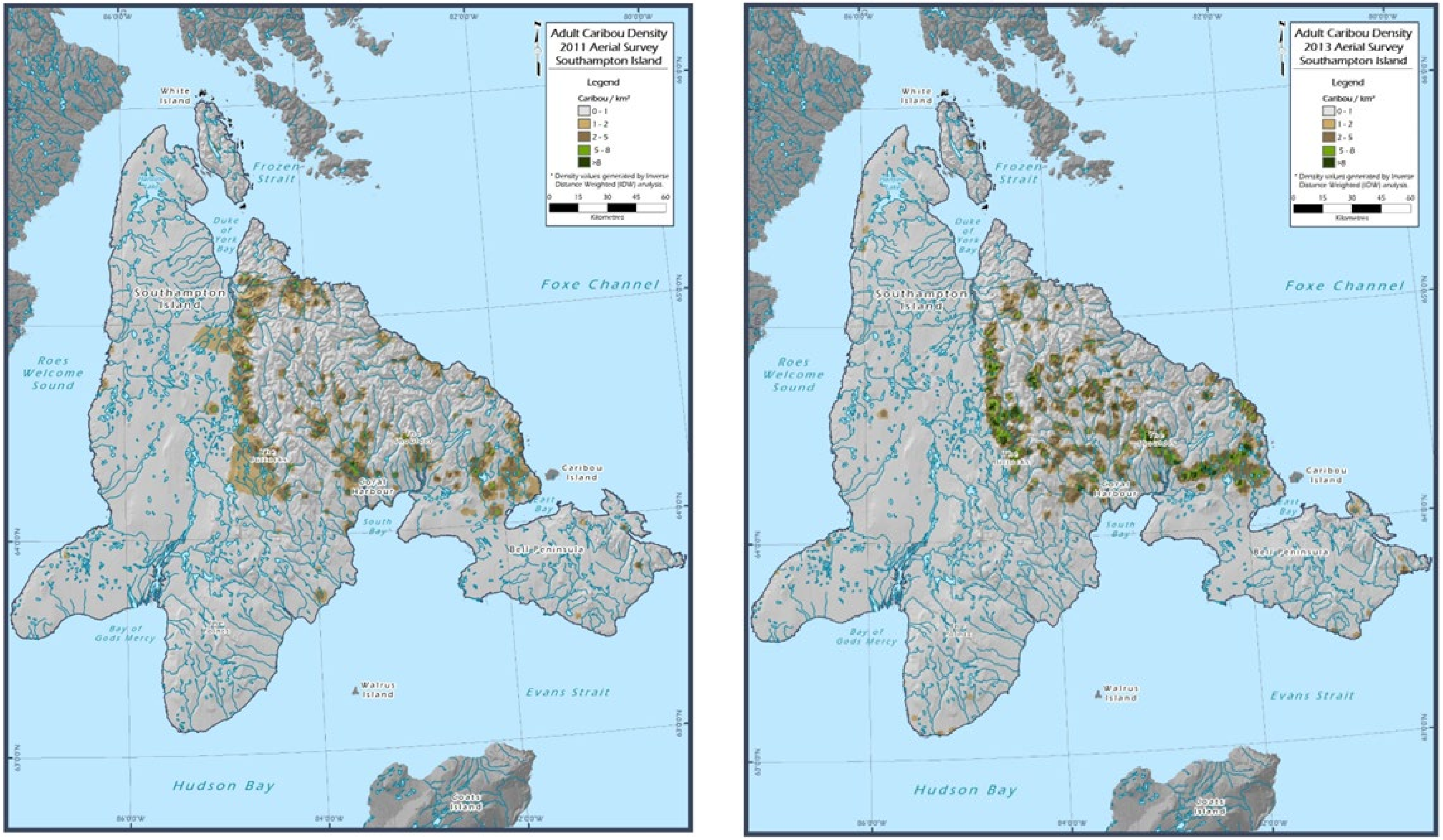

Figure 14. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2011 and June 2013 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

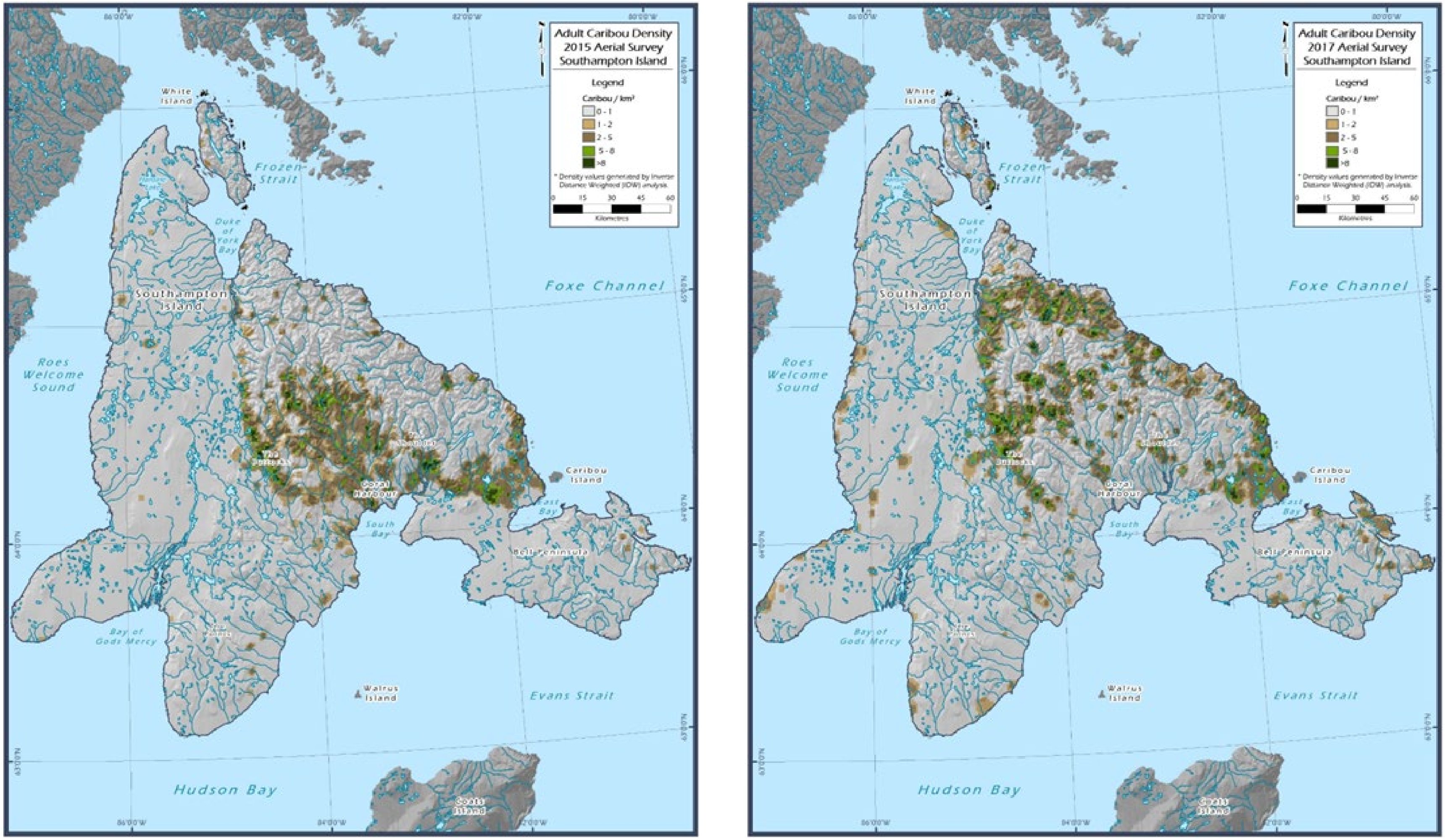

Figure 15. Results of the inverse distance weighted (IDW) interpolation technique applied to the June 2015 and May 2017 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

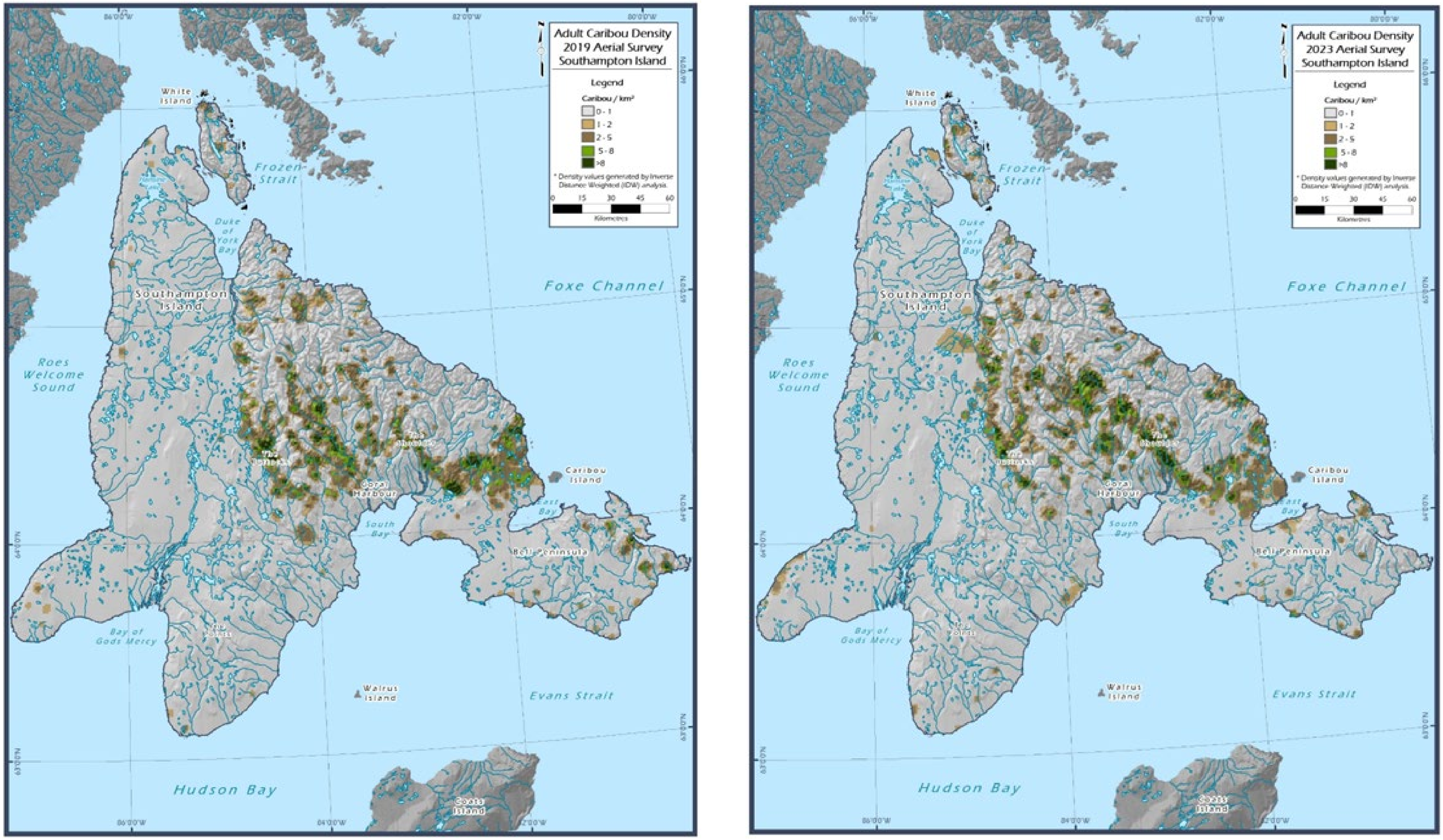

Figure 16. Results of the inverse distance weighted (IDW) interpolation technique applied to the May 2019 and May 2023 abundance survey observations showing relative density of barren-ground caribou (*Rangifer tarandus*) on Southampton Island.

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%

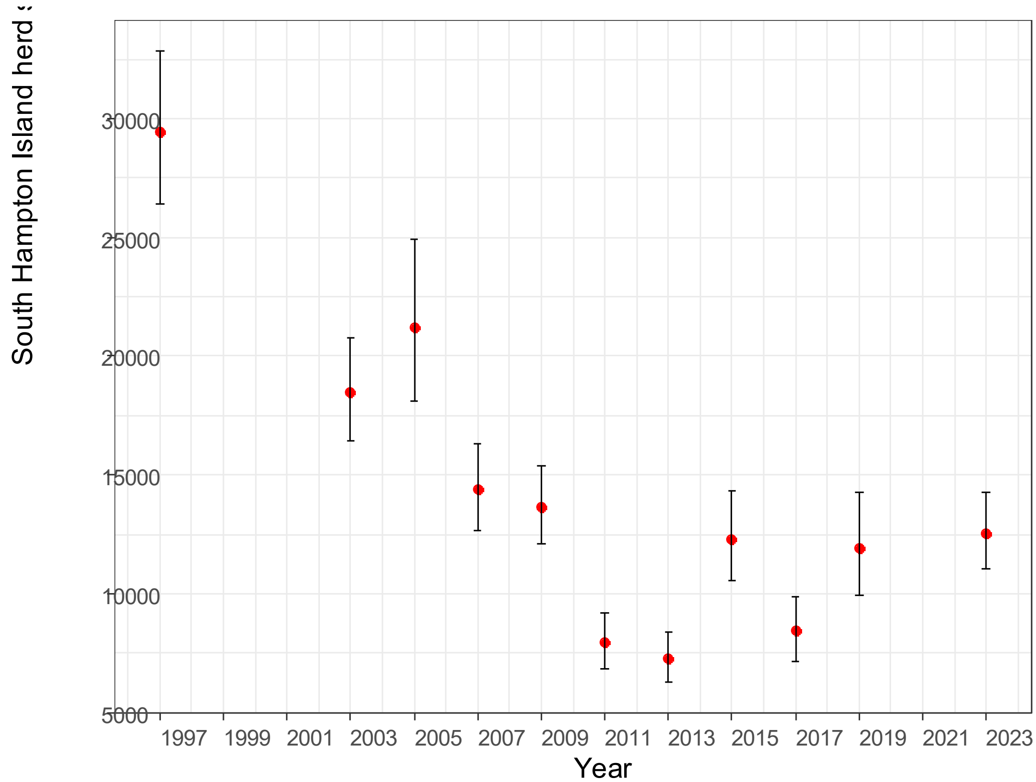

Figure 17. Population abundance estimates of the Southampton Island caribou herd using a strip transect estimator, according to strata listed in Table 4.

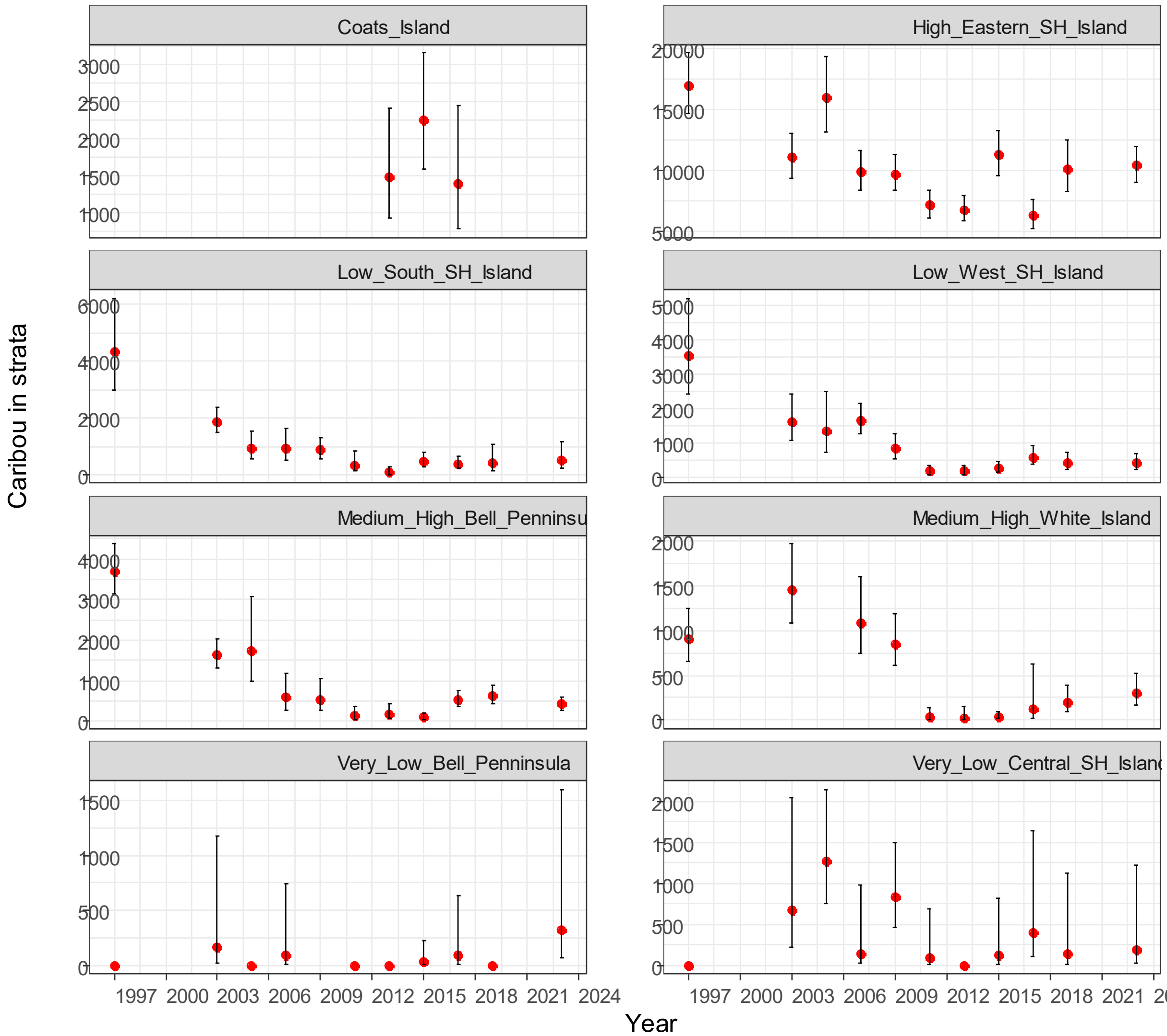

Figure 18. Strata-specific estimates of strata sampled using a strip transect estimator. Note that the y-scales are different for each graph.

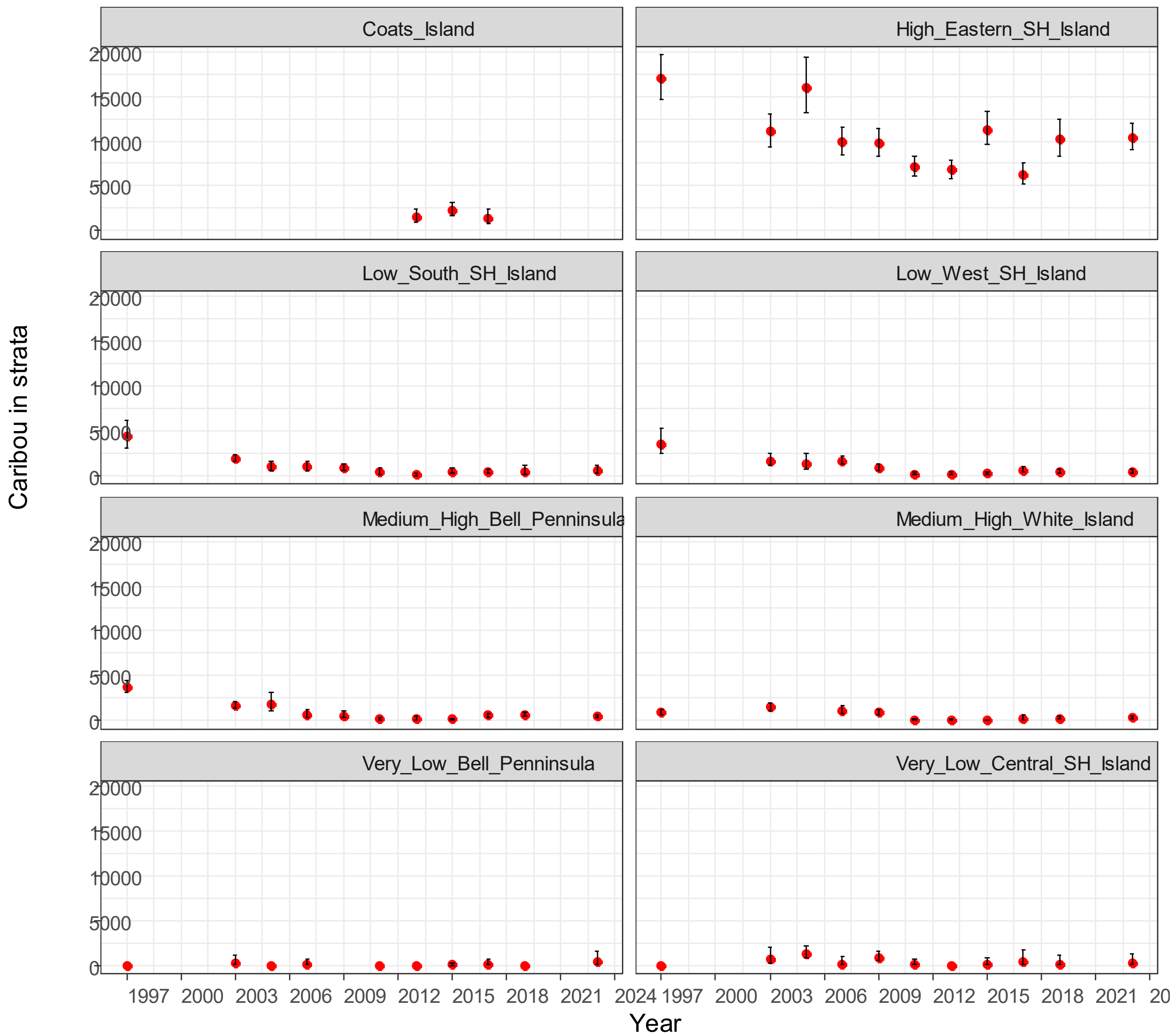

Figure 20. Strata-specific estimates of strata sampled using a strip transect estimator with the same scale used on each graph.

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**).

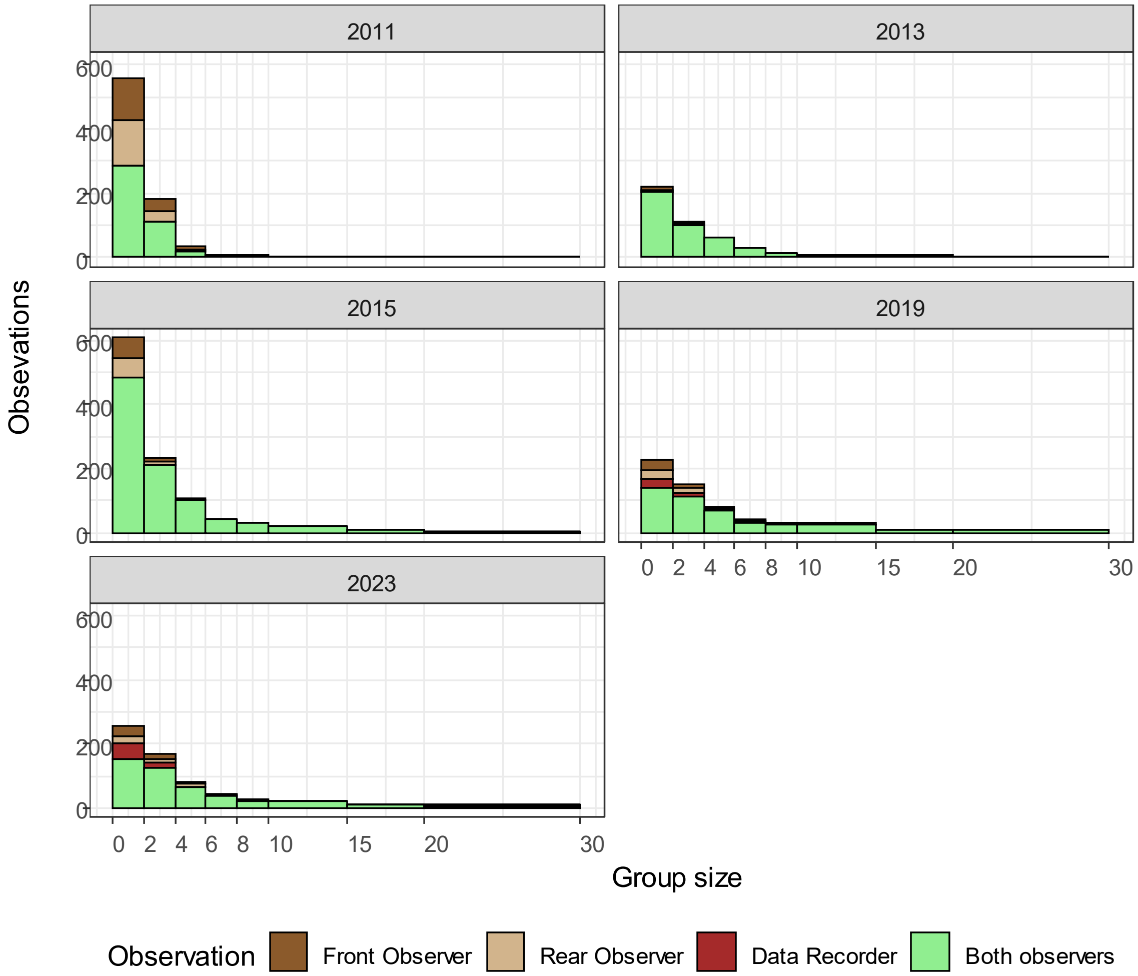

Figure 21. Group sizes of caribou observed each year for surveys conducted in 2011, 2013, 2015, 2019, and 2023, with frequency of sightings made by front, rear, data recorder and both observers.

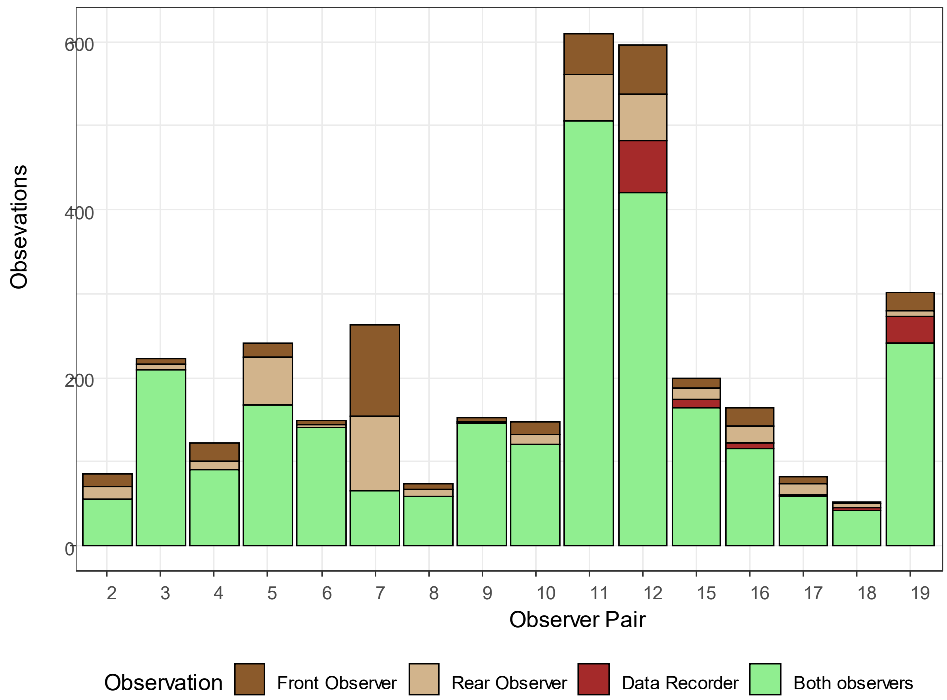

Figure 19. Observer pairings with frequencies of sighting by front, rear, data recorder, and both observers (2011-2023).

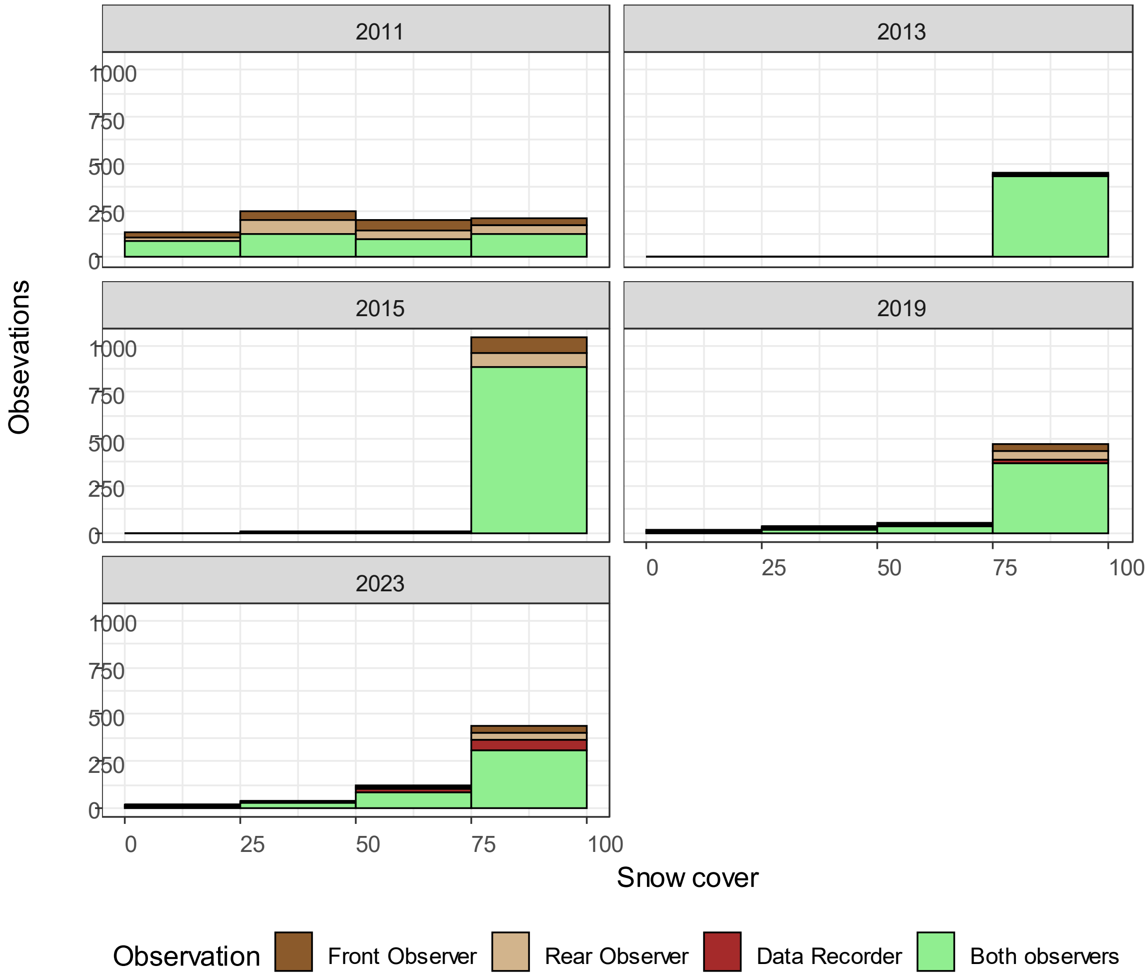

Figure 20. Snow cover during each year of the survey with frequencies of sightings by front, rear, data recorders, and both observers (2011-2023).

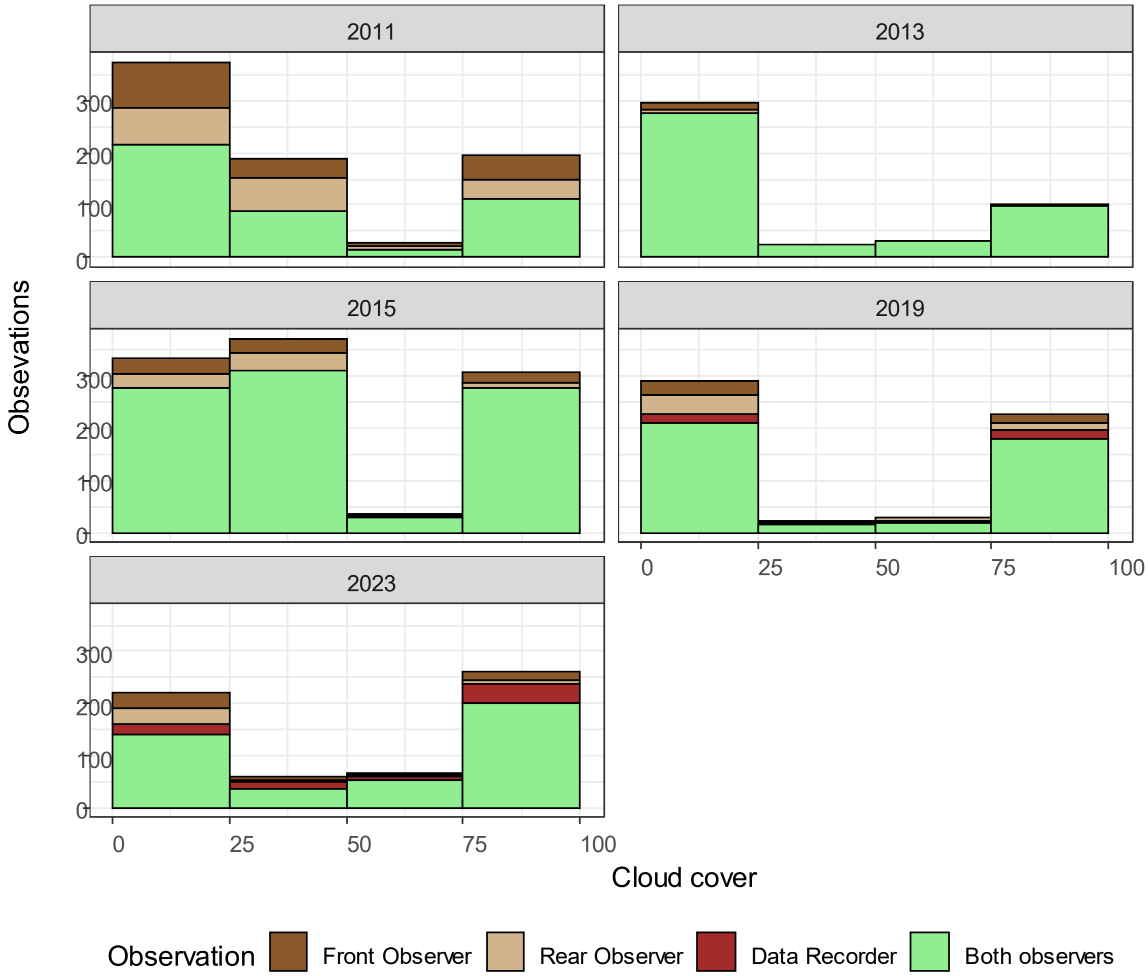

Figure 21. Cloud cover during each year of the survey with frequencies of sightings by front, rear, data recorders and both observers (2011-2023).

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

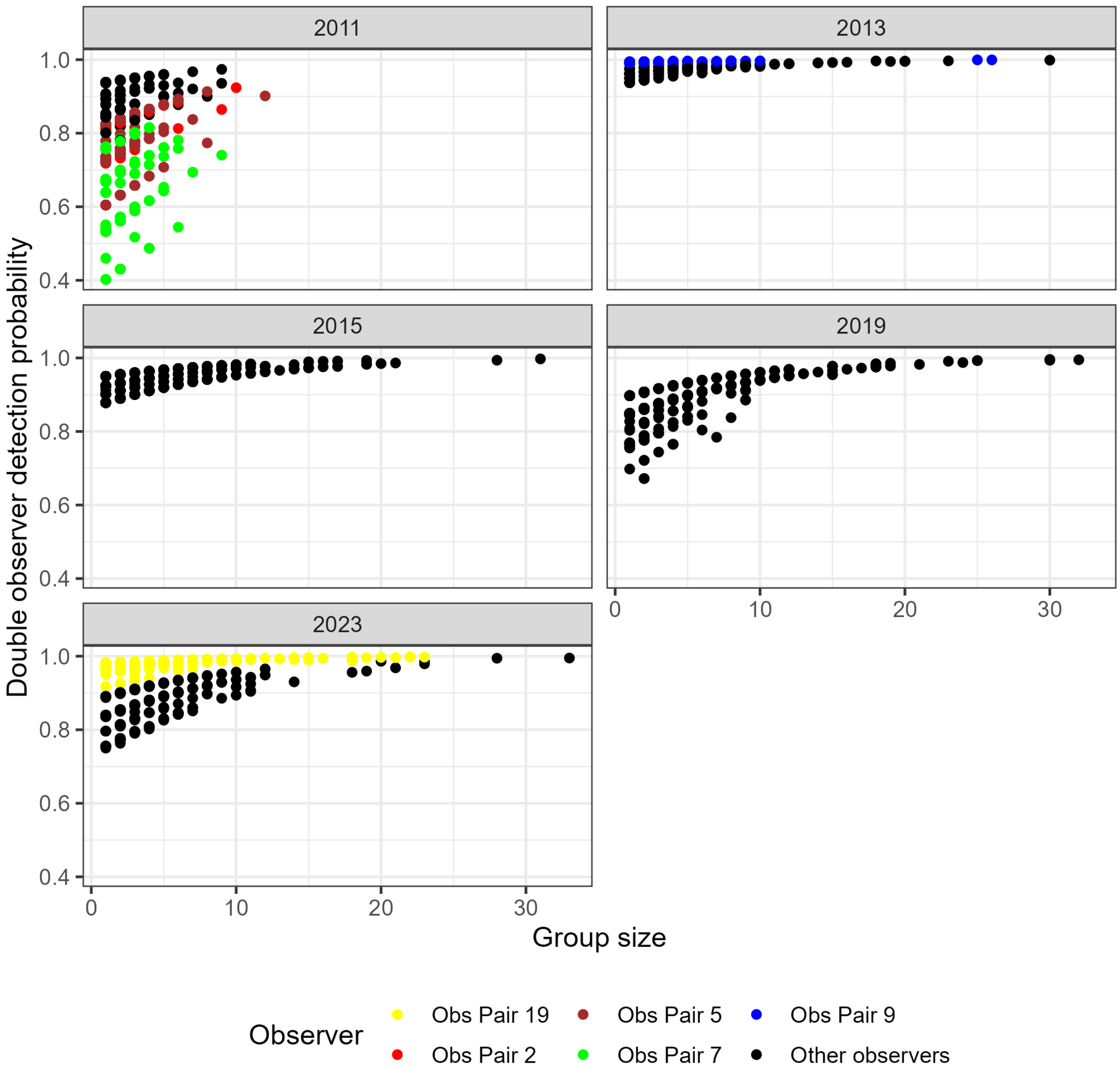

Figure 22. Dependent double observer detection probabilities as a function of year, group size, for selected observer pairs (2011-2023).

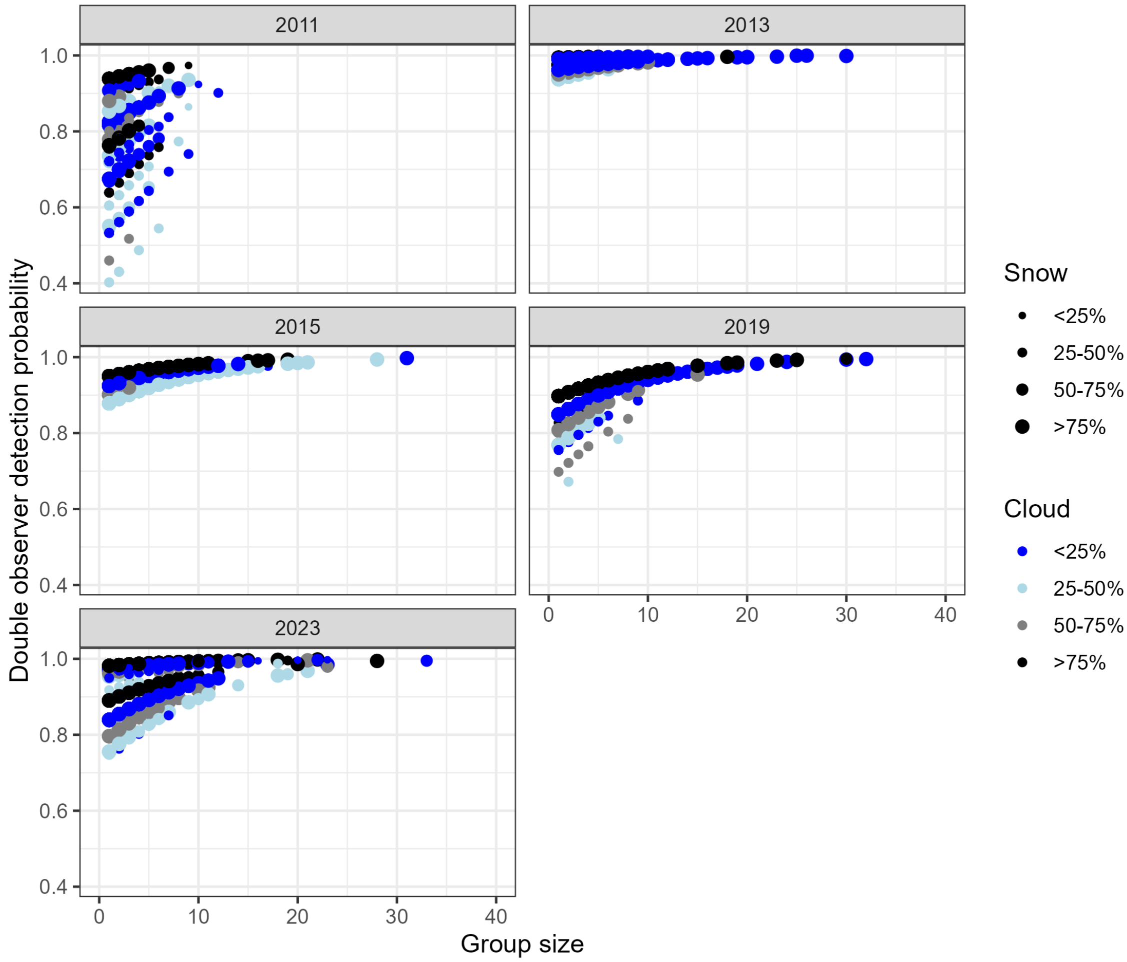

Figure 23. Dependent double observer probabilities as a function of year, group size, snow and cloud cover (2011-2023).

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.

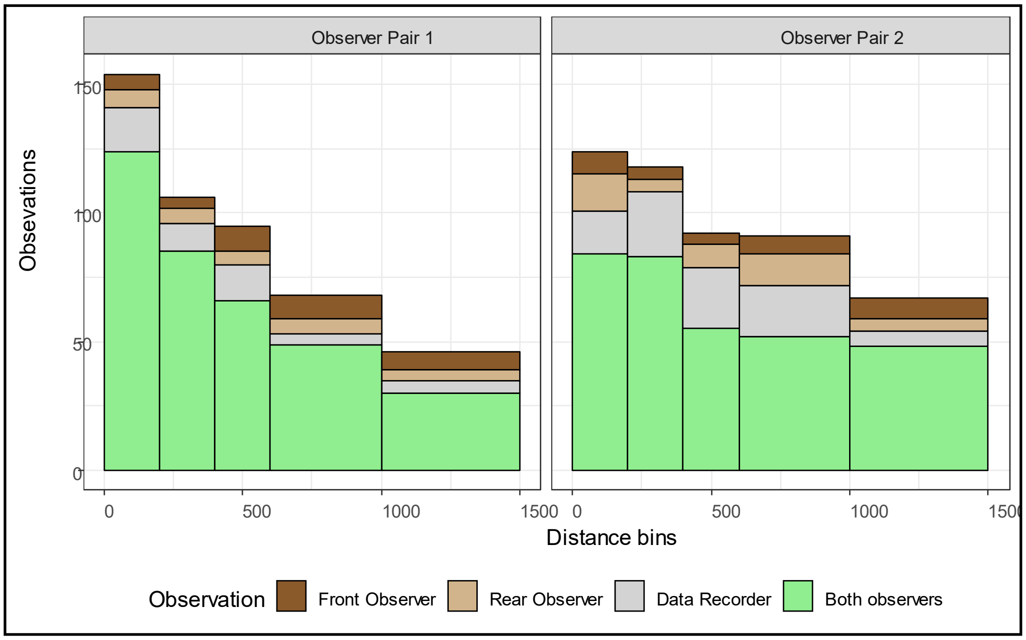

Figure 24. Frequencies of observations by distance bin for the 2 observer pairs in May 2017.

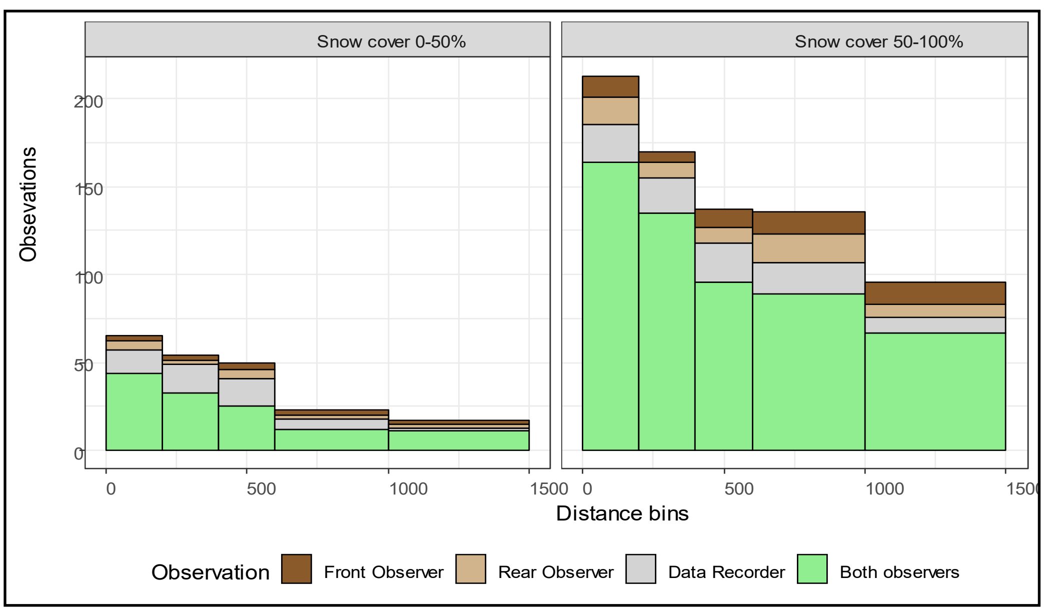

Figure 25. Frequencies of observations by distance bin for 2 levels of snow cover in May 2017.

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

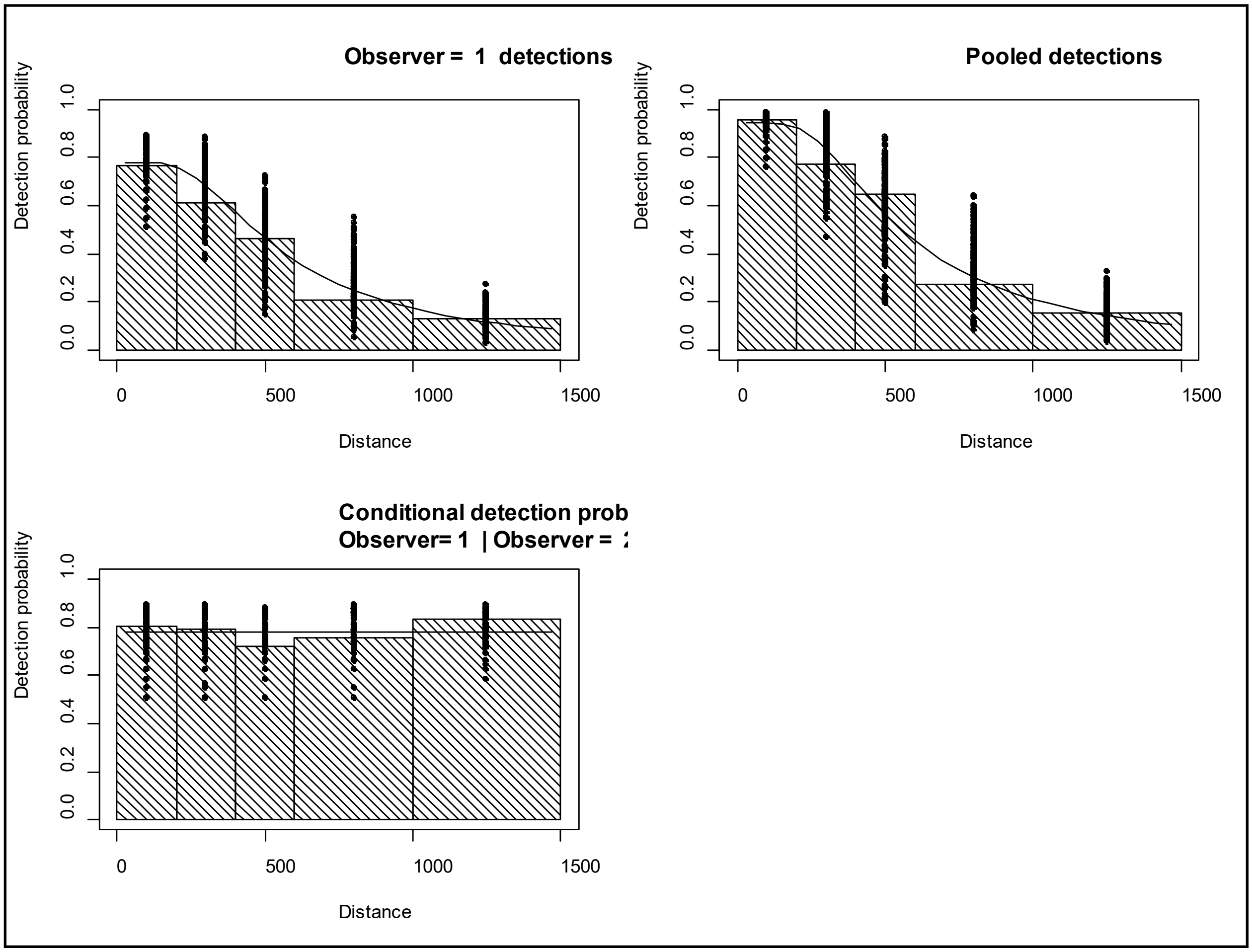

Figure 26. Graphical representation of goodness-of-fit of the most supported double observer model for May 2017 (Model 1, Table 7).

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%

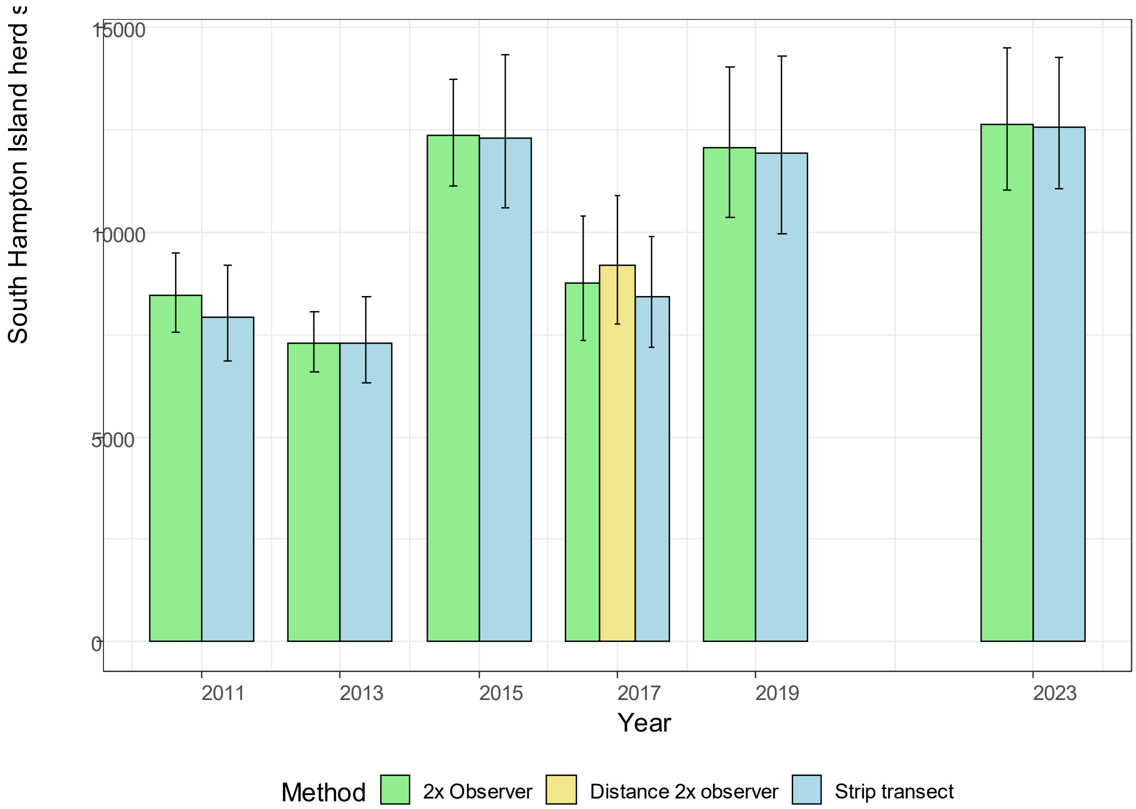

Figure 30. Comparison of more recent Southampton Island caribou herd abundance estimates between June 2011 and May 2023, from strip transect, dependent double observer, and 2017 only distance sampling/double observer analyses.

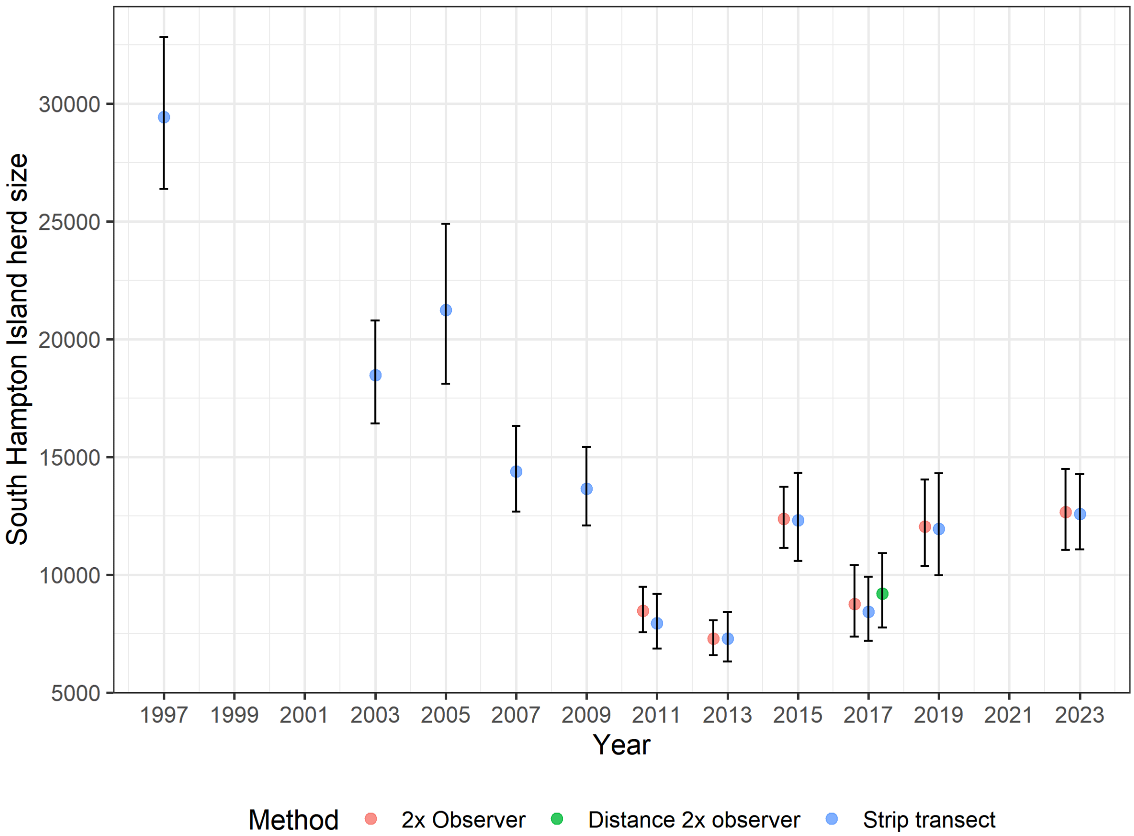

Figure 31. A long-term comparison of strip transect, dependent double observer pair, and distance sampling (2017 only)/double observer pair estimates from June 1997 through May 2023.

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**.

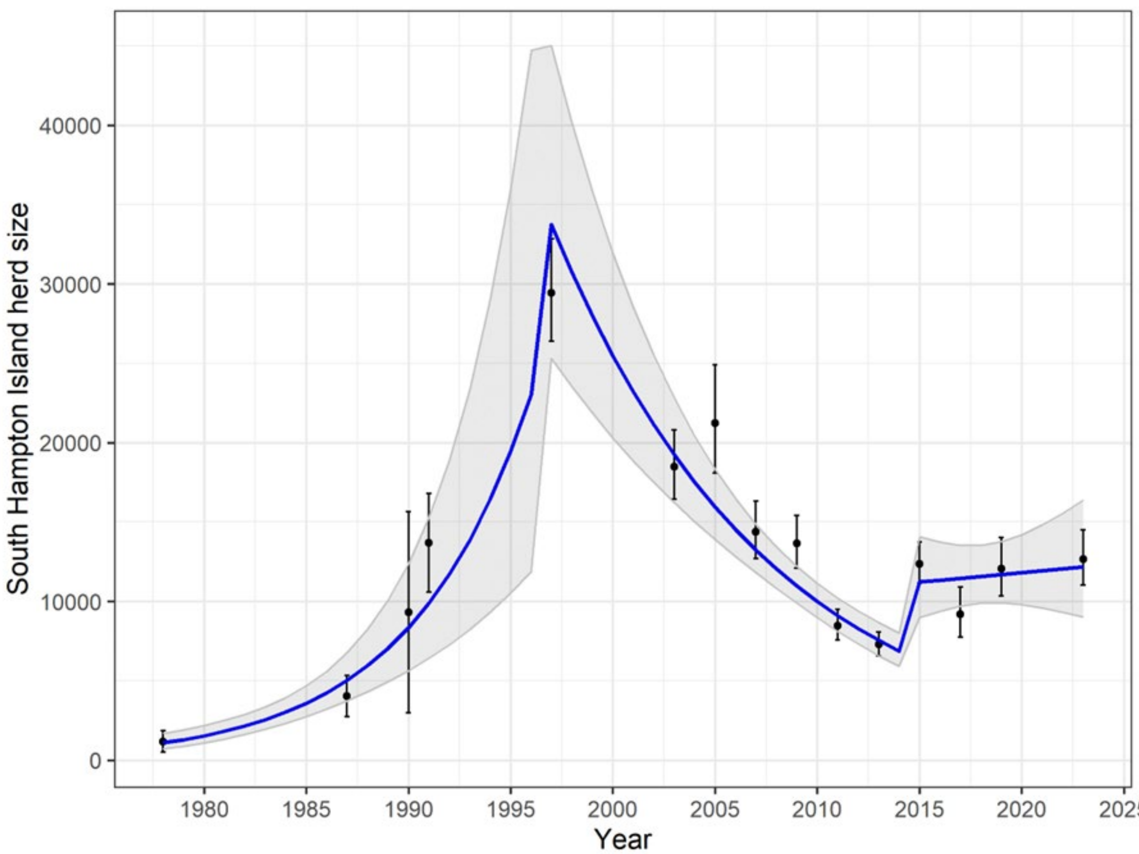

Figure 27. Predictions of herd size of the Southampton Island from the log-linear model (Table 11) which assumes a constant decline in population size after 1997 with an immigration event that occurred before the 2015 survey.

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

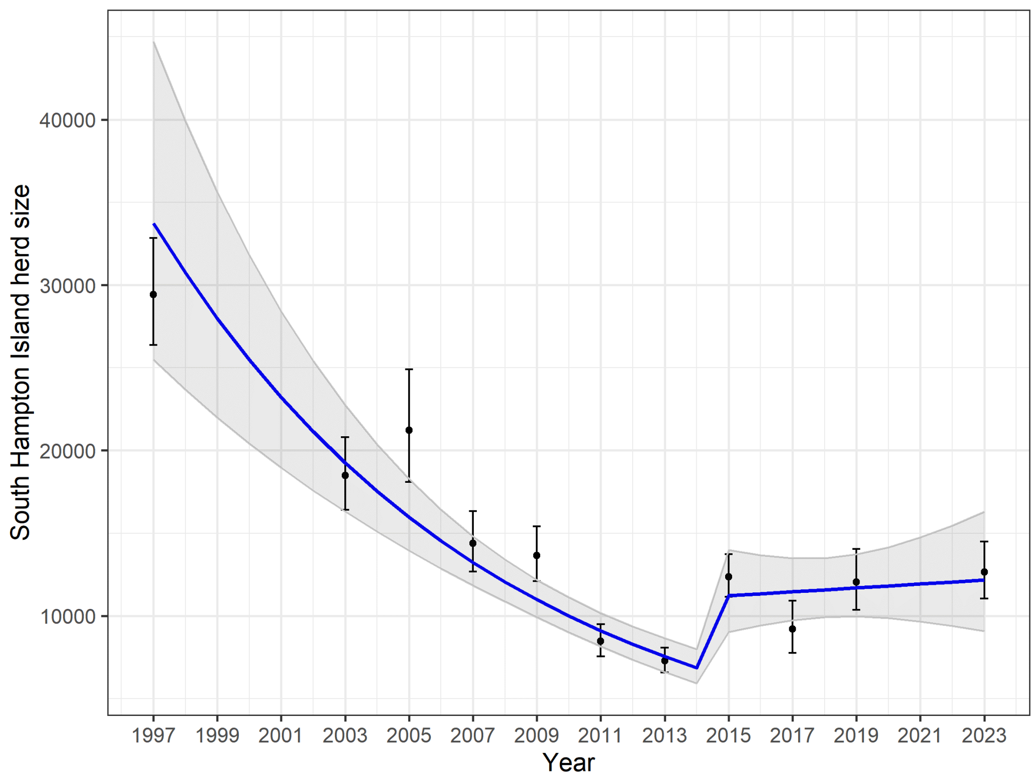

Figure 28. Predictions of herd size of the Southampton Island caribou population from the log-linear model (Table 11), which assumes a constant decline in population size with an immigration event that occurred before the 2015 survey. Confidence limits are provided as shaded regions on the plots.

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.

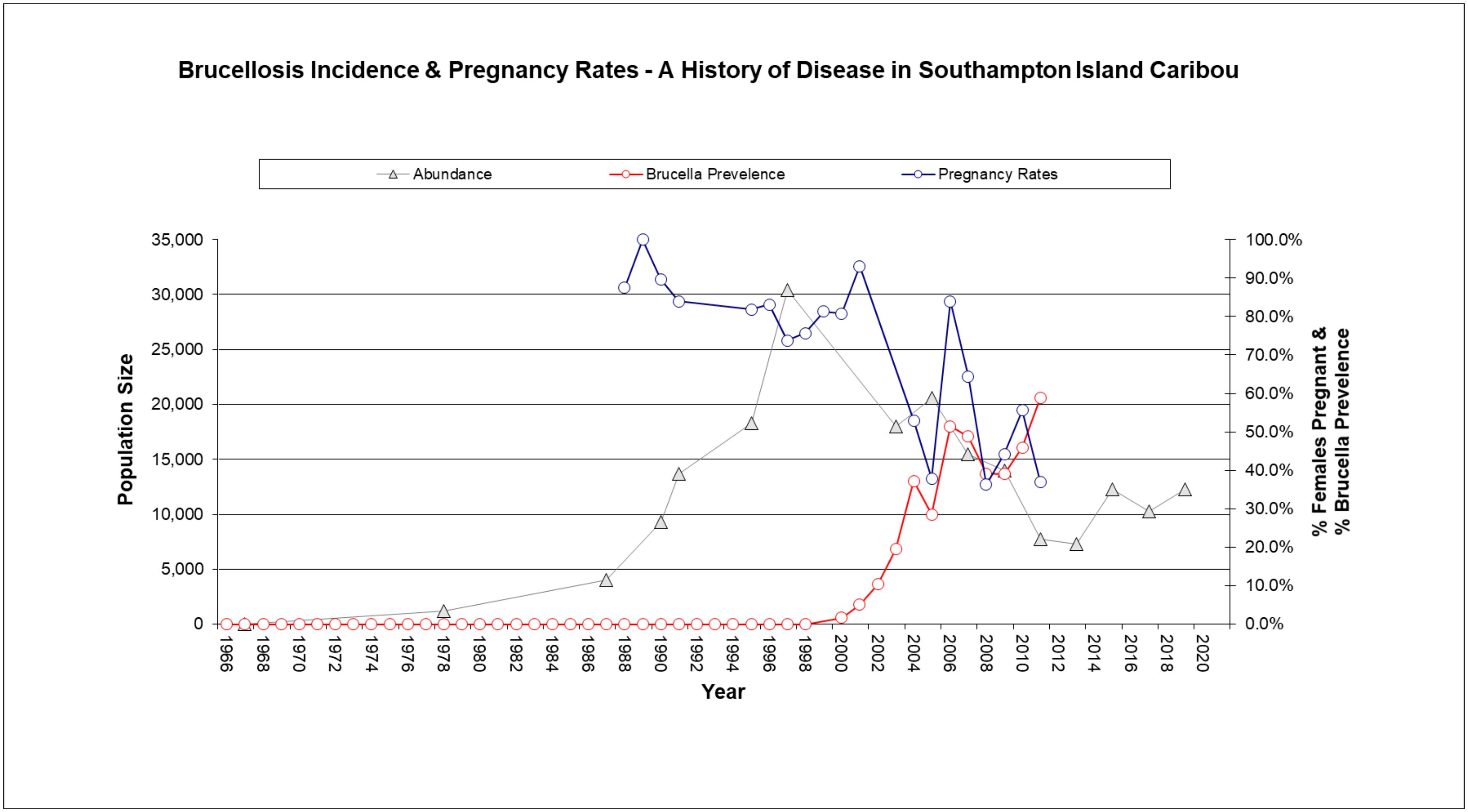

Figure 29. A history of abundance, pregnancy rates and *Brucellosis suis* prevalence for the Southampton Island caribou herd originally introduced onto the island from Coats Island in 1968.

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

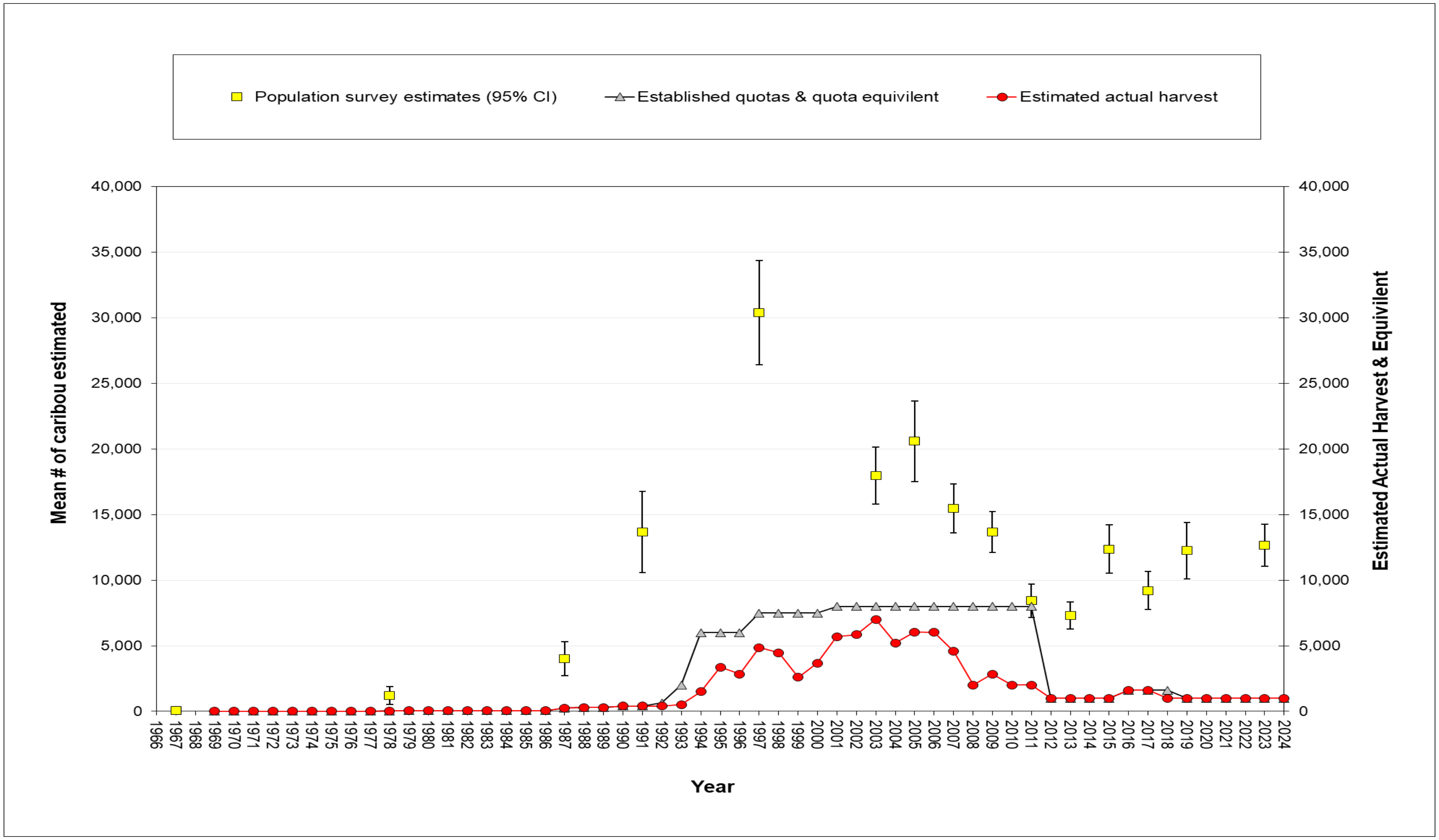

Figure 30. An examination of quota adjustment and actual harvest based on population estimates (Quota equivalents = estimated maximum subsistence harvest substituted for "no-limit" quota allowance values, Tables 13 and 14).

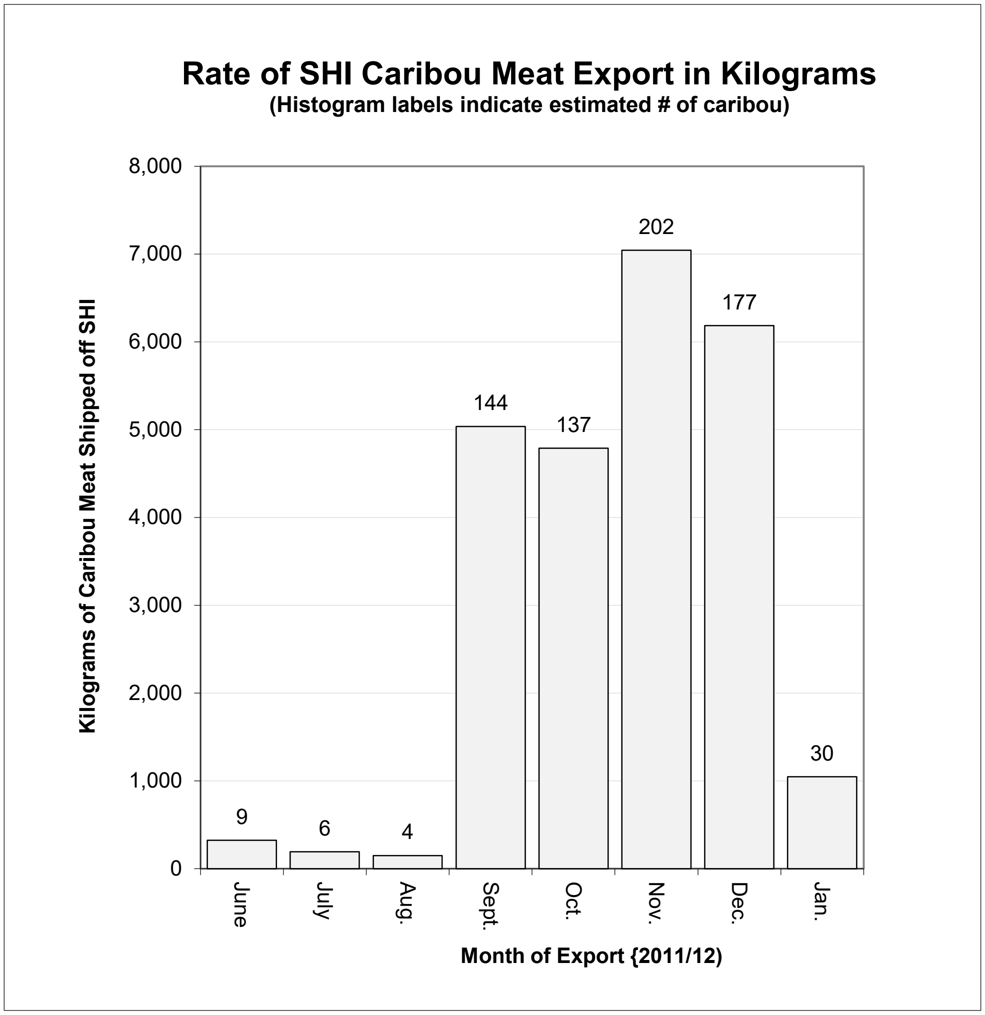

Figure 31. Caribou exports off Southampton Island primarily to Baffin Island communities. Data collected over an 8-month period in 2011/12.

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.

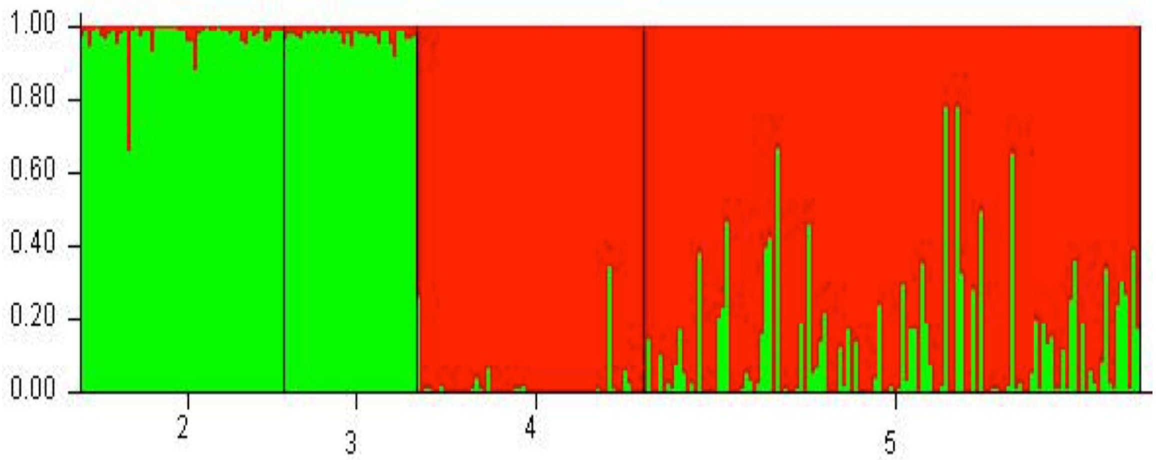

Figure 32. Structure results using 2014 genetic samples. Each column represents an individual, with its estimated proportion of mainland ancestry coded green, and SHI ancestry red. The ‘populations’ are Qamanirjuaq (2; w9741), Naujaat (3; g1616), SHI 2004 (4; w9741) and SHI 2014 (5; g1616) (Paetkau,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

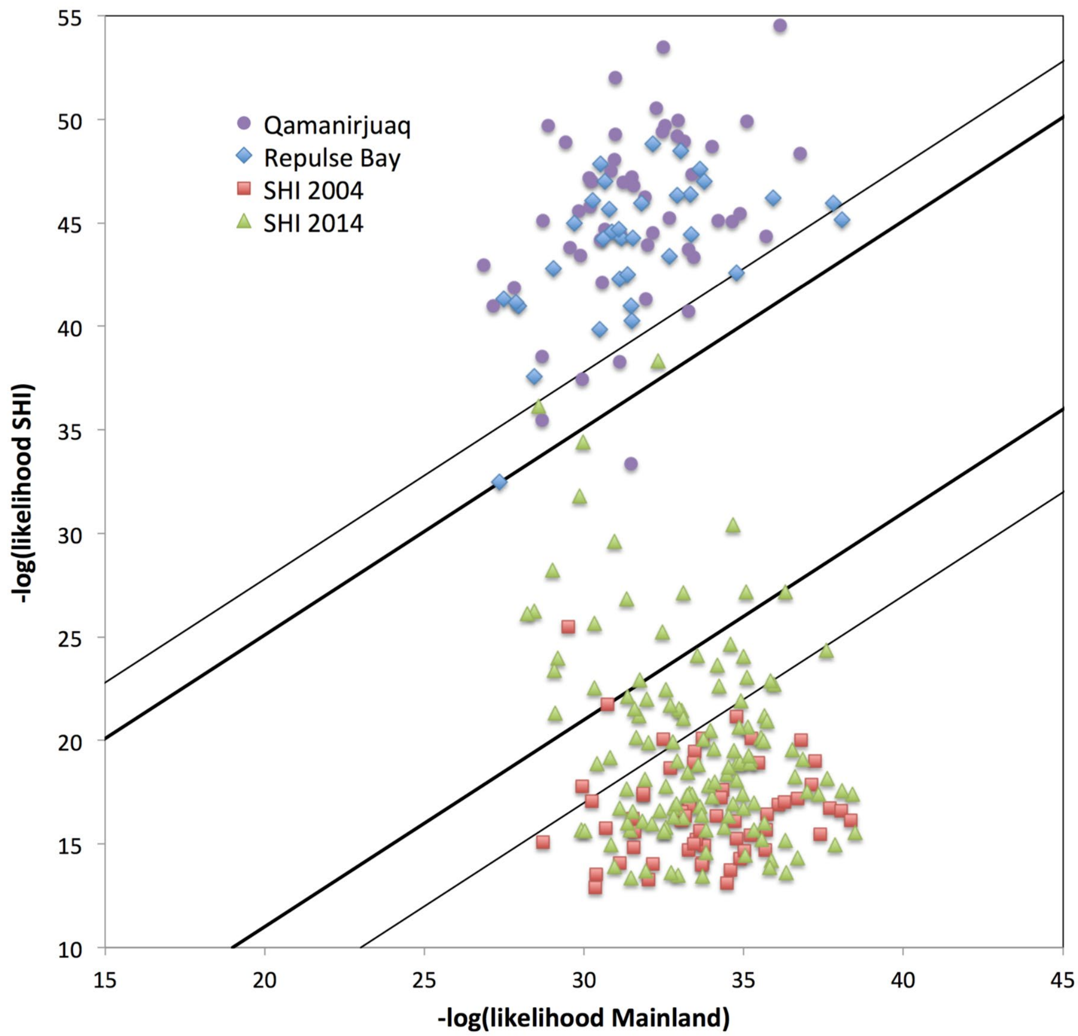

Figure 33. Likelihoods of occurrence based on mainland and (2004) island allele frequencies of caribou according to genetic analysis from different populations and years. Resampling in GeneClass2 indicated that 95% of purebred individuals are expected to have likelihood ratios outside the light lines, 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.

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