RE-ESTIMATING THE ABUNDANCE OF A RECOVERING POLAR BEAR SUBPOPULATION BY GENETIC MARK-RECAPTURE IN M'CLINTOCK CHANNEL, NUNAVUT, CANADA,

FINAL REPORT

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1. A) EXECUTIVE SUMMARY – ENGLISH

Polar bears (*Ursus maritimus*) are managed across Nunavut, Canada, under a harvest and monitoring system that seeks to ensure harvest is sustainable and identified management objectives are achieved. In recent decades, climatic changes across the Arctic have altered polar bear habitat at unprecedented rates. To retain viable polar bear subpopulations as part of the ecosystem and provide a subsistence resource for Inuit, scientific research and monitoring studies are conducted to evaluate subpopulation status and whether management objectives are being met. Here we report the results of a population study for polar bears inhabiting the M'Clintock Channel (MC) conducted 2014 – 2016. Current samples were collected using less-invasive genetic biopsy darting without immobilizing or physically handling bears. Our analyses included data from the 2014 – 2016 biopsy mark-recapture study, live-capture data collected under a mark-recapture study 1998 – 2000, and limited harvest recovery data over the entire period 1998 – 2016.

   Results of a closed capture-recapture model, implemented in a Bayesian framework and fitted to data for independent animals (i.e., >2 years), suggest a mean abundance estimate of 716 (95% Credible Interval [CRI] = 545 – 955) for the period 2014 – 2016, indicating that the MC polar bear subpopulation increased from the mean abundance in 1998 – 2000 (325 [95% CRI = 220 – 484] in this study; 284 [SE: ± 59.3] in Taylor et al. [2006]). Both the male and female segment of the subpopulation increased between study periods (1998 – 2000 and 2014 – 2016), likely because of a combination of reduced harvest pressure and improved habitat quality. We used a closed population model because data were too sparse for models with more parameters. Estimates of abundance should be interpreted with caution because they reflect the "superpopulation" (e.g., it includes all bears that use the MC management area, some of which spend time in other subpopulations as well) and likely include positive bias due to violation of model assumptions in addition to the negative bias caused by variation in the capture probability. The overall mean litter sizes for the period 2014 – 2016 were 1.70 (SE = 0.09) and 1.61 (SE = 0.11) for cubs-of-the-year and yearlings, respectively. The calculated mean number of yearlings per adult female declined from 0.39 (SE =

0.10) to 0.28 (SE = 0.06) between both study periods, but MC remains a productive polar bear subpopulation despite that decline and observed sea-ice changes. However, given the sparse reproductive data, we are not able to make any substantive inferences. Polar bear body condition (i.e., relative fatness), assessed in the spring, generally increased between the periods 1998 – 2000 and 2014 – 2016. Estimated apparent survival for bears aged 2 and older was 0.88 (SE = 0.02), although this is likely biased downward due to temporary or permanent movement of individual bears with respect to the study area and limited data availability concerning immigration and emigration. This is corroborated by the increase in abundance estimates across periods indicating the survival rate had to be greater than 0.88 to achieve such substantial growth. When we calculated adult survival using the change in abundance estimates between 1998 – 2000 and 2014 – 2016, our estimated rate of 0.93 suggests that the population growth is positive, with a growth rate of 2%. Overall, our findings align with local knowledge that the MC subpopulation recovered from over-harvest that occurred 1979 – 1999 (average harvest 34 bears/yr). Ecologically, we hypothesize that the observed improvements in body condition and strong population growth over time may be related to spatial and temporal reductions in sea-ice type and quantity providing transient benefits to the MC subpopulation due to lighter ice conditions (i.e., a reduction in thick, multiyear ice) and increased biological productivity. However, climate change is the primary long-term threat to polar bears and the threshold beyond which the MC subpopulation could be negatively affected by continued ice loss, like some other polar bear subpopulations, is currently unknown.

Estimating demographic parameters for the MC subpopulation proved to be challenging because small sample sizes, low probability of recapturing the same bear, and lack of movement information constrained analyses in this study such that the estimates of abundance and survival are almost certainly biased. Our estimates represent only the second time the MC subpopulation has been inventoried under a replicable, structured study design and thus offer many opportunities to learn from these experiences in analysis and data collection methodology. For other wildlife populations or ecosystems that share similarities with MC, we recommend collecting additional reproductive data and genetic samples at approximately the midpoint between the

current study and the next comprehensive subpopulation assessment (in Nunavut's case, that would be 5 – 7 years post-field work completion) or increasing study length (e.g. 4 – 5 years), to increase confidence in the survival rates, possible emigration, and reproduction. Further, movement data (satellite telemetry) are recommended. In the absence of satellite telemetry data on polar bear movements, we recommend conducting a meta-analysis to investigate exchange between MC and nearby subpopulations (i.e., Lancaster Sound, and Gulf of Boothia).

1. **B)** ⊲ᐅᒐᑎᓂᕐᒪᓄᑦ ᓇᐃᓈᖕᐸᓕᕐᓂᕐ –

ᓇᓄᐃᑦ (*Ursus maritimus*) ⊲ᐅᒐᑕᐅᓕᑕ ᓄᓇ௨ᑦᒥᒥ, ᑲ௨ᑕᒥ, ⊲ᖕ௨ᕐ⊲ᖅᑕᐅᓂᖕ�ᓂ ⊲ᒪᓗ
ᖃᐅᔨᓴᕐ⊲ᖅᑕᐅᓂᖕ�ᔬᓂ Λᓕᓕᕐᙶᑎᖅᑦ ᖄᖅᐳᔾᔾᓚᕐ⊲ᖅᑎᒥᔾᔭᐅᑦᓗᓂ ⊲ᖕ௨ᕐ⊲ᖅᑕᐅᔾᑦ
⊲ᖅᐳᔾᔾᓗᑎᑕᐅᑎᐊᖤ௨ᖅᑐᒧᑦ ⊲ᒪᓗ ௨ᓄ௨ᖅᑕᐅᕐᒪᓗᒧᖤ ⊲ᐅᒐᑕᐅᓂᖕ�ᔮᓄᑦ Λᓕᓕ⊲ᖕᖅᔪᕐᒪᑦ
⊲ᑐᖅᑕᐅᔮ௨ᔮᑐᕐᖐ. ⊲ᓯᒧᑕᐅᖅᑐᒃ ᖅᒡᓗᐅᖥᓚᐅᖅᑐᒃ ⊲ᖤᔪᒦ, ᒃ௨ᐅᖤ ⊲ᔾᔾᖕᖥ<ᖤᖟ⊲௨ᖐ ᐅᐳᐅᖅᖤᑦᑐ௳ᒦᒦ
⊲ᔾᔾᖤᑎᑎᖥᒪᖕᖐᑦ ௨ᓄᐅᑦ ௨ᔾᖅᖤᑕᖕᖤᓄᑦ ⊲ᔾᔭᐅᖐᑦᔭᐅᖤ ⊲ᖤᔾᔾᓂᖅᖥᔾᓂᖤ. Λᖤᔾᔾᐅᖕᖐ௨ᖥᖟᒧ
⊲ᑐᖅᑕᐅᑎᑎᔭᑦ௳ᖕᖐᑦ ௨ᓄᐃᑦ ⊲ᒥᔾᓂᖅᖤᔾ࿔ᖤᔾᖤᖐᑦ ᐃᓓᖥᐅᓄᖤ ᐅᓚᔾᖐᑭᓕᖤᐊᑦ௳ᓂᖤ ⊲ᒪᓗ
௨ᖕ<<ᖟ௟ᖟ⊲ᖟᖤᑕ௟ᖐᑦ Λᔾᐅᖅᖤᑕᖕᖟᖤ Δᓄᖤᖞᓂᖤ, ᖃᐅᔭᓇ⊲ᖤᖐᖤᖤᖟᔾᑎ ᖃᐅᔭ௟௟⊲ᖤᓂᖤ ⊲ᒪᓗ
ᖃᐅᔾᒪᓚ⊲ᖤᔾᖟᖤᖤ ᖃᐅᔭ௟௟⊲ᐱᑕᐅᖤᑦ Λᖤᖤᕆᖤᔾᑎᖤᖥᓚ ᖃᐅᔭᑦᖤᔾᔾ௟ᐅᖤ ⊲ᒥᔾᓂᖤᖤᖞᖐᑭ
ᖃௗᐊᖤᖥ௟ᖤᖟᖐᖟᖤ ⊲ᒪᓗ ⊲ᑐᖅᑕᐅᑎᑎ⊲ᖟᖟ⊲ᖥᖤᖐᑎ ⊲ᐅᒐᑎᓂᖐᔾ Λᓕᖤ⊲ᖥᖤᔾᔾ ௨ᖥᖤᖤᐅᖤ௟ᖤ.
ᑕᖚᐅᖤᖤ ᐅௗᖤᖟ⊲ᖐᖤ ᔾᖤᒃᔾᖤᑦ ⊲ᒥᔾᓂᖤᖐᑦ ᖃᐅᔭ௟௟⊲ᖤᑕᐅᖐᑦ ௨ᓄᐃᑦ ௨ᔾᖤᔾᔾ Δᖐ⊲ᖤᖞᖟ (MC)
Λᓕᓕ⊲ᖥᑎᓄ⊲ᖤᖞᖤ 2014-2016–ᒦ. Ĺ௳௟ᑮᖥᐊᑦ ᖃᐅᔾᖤᔾᔾᔾ࿔ᖤᖥᖤ Λᔾᐅᖅᖤᑕᓚᐅᖐᖤᖞ ⊲ᖞᖤᖥ௟ᖤᖥ
ᖅᑭᓕᑎᖤᖟ⊲ᖞ࿔ᖤᖥᓂ௟ᖤᖤᖤᖥᖟ⊲ᖥᖟᖤᖤ ᐅᔾᖤᓂᖐᑦ ᖃௗᐊᖞ࿔ᖥᖟᓂᖐᑦ ᑮᐱᔾᖤᔾᑎ⊲ᖥᖟᖤᖤᖞ
⊲ᐅᒐ⊲ᖞௗᐃᑎᑎᐅᖐᑦᖐᖟᖤ ⊲ᖤᖟᓄᑦ ⊲ᖟᑐ⊲ᔾᐅᖟᖞᖤ Λᓕᓕ⊲ᖥᖞᖤᖟᖞᖤ ௨ᓄᐃᑦ. ᖃᐅᔾᖤᔾᓄᖐᖞ
Δᑕᖤᖟ⊲ᖞ ᖃᐅᔾ௟ᖥᑎᖤᖟᓂᖤ ᑕᐅᖟᖥᖞᑦ 2014 – 2016–ᒦ ᖃௗᐃᖞᖟᖞᖤᖟᖤᖤ
௨ᓄ௨ᐃᖤᔾᖤᖞ௟ᖞ–Λᔾᐅᖤᖐᖤᖟᖟᖞᖤᖟᖤ ᖃᐅᔭ௟௟⊲ᐱᑕᖤᖤᖤ, ᐅᖥᓄᖤᖟᖟᖤ–Λᔾᐅᔾᖤ ᖃᐅᔾᖤᔾᑎᑎᖤᖥᖞ ௨⊲ᑕᐅᔾᖤ
ᐅᖞ௟ᖤᖞᖐᖞ ௨ᓄ௨ᐃᖤᔾᖤᖞ௟ᖞ–Λᔾᐅᖤᖐᖤᖟᖟᖤᖟᖤ ᖃᐅᔭ௟௟⊲ᐱᑕᖤᖤ 1998-2000, ⊲ᒪᓗ ᑮᖤᑕᑕᔾᖤᔾᖤ
ᐅௗ௨ᖤᑕᐅᔾᖤᖤᖟ ᑕᒦᖤᔾᖤᖐᑎ ᖃᐅᔾᖤᔾᑎᑎᖤᖥᖐᖤ ⊲ᑐᖤᖟ௨Ĺᖥᖟᖐ 1998-2016.

ᔾᖤᖞ௳ᖟᖐᖞᑦ ᒪᑐᔾᐅᔾᓚᑎ௟ᖤᖞᖤ Λᔾᐅᒐᑕᐅᖤᖤᑐ–Λᔾᐅᖤᖐᖤ௳ᖟᖤᖟ⊲ᖞ௟ᖟ Λᓕᓕᖥᖤᑕᐅᖤᖤᖥ,
⊲ᑐᓕᖤᖐᑎᐅᑕᐅᖐᖤᔾᓚᖥ௟ᖤ ᖃᐅᔾᖤᔾᑎᑎᖤᖥᖞᑕᐅᖐᑎᖤ⊲ᐱᑎᑦ ᖤௗᔾᒦᖤᔾᑎᖞᖐᖞ ⊲ᒪᓗ ௣ᒪᖤᑕᖞᖅᑕᐅᖤᖟᖐᑎ
ᖃᐅᔾᖤᔾᑎᑎᖥ௨ Δᒦᒡᖥᖐᖤ௟ᖐᑦ ᐅᖐᔾᖤᖞᖐᑦ (ᑐᑭᑕᐅᑎᓗᒧ, ᖟ2 ⊲ᖤᖥᔾᖥᖐᖤᖤ), ᐃĹᖞᑐᖞ௳௨ᓕ⊲ᖤᖤ௟ᖟᖐ௟ᖞᖞᖟ
⊲ᒦᔾᖟ௟ᖐᖤᖞ௟ᑕᐅᖅᑕᖐᖤᖟᖤᖤ ᒦᖤᖟᖤᖥᑕᐅᖤᖞᖐ௟ᖤᖟ 716–ᖤᔪᖟᖐᖐᖤ௟ᖞ (95% ᖟᖐᖥᑎᑦ ᒃᖟᑐᐃᖞௗ௨ᓕ⊲ᖐᖤ௟ᖞᖞ
௨ᖟᖟᖟ⊲ᖞᖟᖟᖥᖟᖤ⊲ᖤᖟᖥ [CRI] = 545 – 955) ⊲ᑐᖐᖟᖥᖞ 2014-2016, ᐅᖞᖤᐅᖞᔾᖤᖟᖤ ᑕᐃᖟᖟ௟
Δᖐ⊲ᖤᖞᖞ (MC) ௨ᓄᖞᑎᑦ ⊲ᒦᔾᖟᖞᖤ௳ᖟᖐᑦ ⊲ᒦᔾᖤᓄ⊲ᖞᖟ௳௨ᖞᑕᐅᖥᖞᑕᑕ ᑕᒦᖤᖤᒃᖐ ⊲ᒦᔾᖟᖞᖐᖤᖟ௨ᐅᖟᖞᖅᖤᖞᖟᖤᖤ

ᑕᐅᕙᓂ 1998-2000–ᒥ (325 [95% ᐳᔆᑎᑦ ᓯᓚᑐᐊᖄᖑᐊᖅᖃᖖᒧ ᓇᑦᐊᔪᓕᓇᖖᒐᕝᐊᖓᖓᒡ [CRI] = 220-484] ᑕᑯᑕᓇ ᖃᐅᔨᓴᕐᐊᑉᑎᓂ; 284 [SE: ± 59.3] ᑕᐃᓪ ᐃᑦ ᐊᐅᑦ [2006]). ᑕᒪᑯᐸ ᐊᕿᑎᑦ ᐊᖄᓐᑐᐃᑦᓗ ᖃᐅᔨᓴᖅᑕᐅᕝ ᐊᒥᓱᖃᖕᓂᖓᓂᑦ ᐊᒥᔭᑉᖃᖕᓇᒐᖅᏞᑕ ᐊᖁᓂᖕᓂ ᖃᐅᔨᓴᖅᑕᐅᑎᓪᒍᓇ (1998-2000 ᐊᒪᓗ 2014-2016), ᑕᐃᒪᓇᓛᔾᐊᑎᖃᑦᑐᔆᓴᖅᖆᖅᑎ ᐊᑕᐅᑦᑎᒃᑯᓴᖅ ᖃᐃᖄᓇᐱᐊᖅᖃᖕᓂᖓᓂᑦ ᓇᖅᑐᑕᐅᕙᑐᒡ ᒪᓐᑕᑕᖅᒍᔭᑦᑲᓯ ᐊᒪᓗ ᐱᐅᓂᖅᕲᑕᐅᖕᓂᓗᒡ ᓇᖅᑕᐅᕙᑐᒡ ᐱᐅᓂᖅᒪ. ᐊᑐᐸᕆᑦ ᒪᒍᔾᒪᖖᒍᓂᑦ ᐊᖄᓂᖕᓂᓗᒡ ᐱᓯᓂᒍᑎ ᐱᔪᐊᑎᑦᓗᒍ ᖃᐅᔨᔪᖃᖅᒃᖓᑦ ᓇᑐᓴᖃᔪᐊᑕᐅᖅᏞᑕ ᐱᓯᓂᕿᑎᑦᑲᑦ ᖃᓇᐃᑦᖃᖖᒃᖃᖕᓂᖃᖅᏞᓇᖕᓗᒡ. ᒥᖅᐹᔆᔾᕿᑎ ᐊᖕᒪᓂᖕᓂᓗᒡ ᑐᑭᑕᐅᖅᑕᐅᓯᐊᖃᖅᏞᑕ ᐅᑄᖅᑐᖅᑕᐅᑎᐊᔪᓐᑎ ᐱᕐᐊᒡᓗᒍ ᓴᑭᔨᖅᑎᑦᑎᒪᖕᒡ "ᐊᖄᕐᐊᐧᔾᔪᖃᖕᓂᖓᓂᑦ" (ᑐᑭᐅᑎᓗᒍ, ᐱᖅᑭᐹᔆᖖᒪᖕᒡ ᐃᔪᖃᐧᖕᓂᖓᓂᑦ ᓇᓇᒡᓂᑦ ᐊᑐᖅᑐᖅ ᐃᑐᐊᖅᑭᑦ (MC) ᐊᐅᑕᑕᐧᒥ ᓇᖅᑕᐅᕙᑐᒥ, ᐃᓇᖕᓕᖕᓗ ᑕᐃᑯᐸᐊ ᐃᓇᒥᔭᖃᒪᐧᕿᔾᖖᒃᑐᖕ ᐊᕆᖕᓗᒡ ᐊᖄᕐᔭᐸᖕᓂᐊᑕᓗᒡ) ᐊᒪᓗ ᐱᖅᑭᐹᔆᑐᖅᒪᖕᒐᖅᏞᑕ ᓇᐊᑐᑎᑐᓐᐃ ᐃᓇᐃᖅᔆᓂᖕᒐᕐᑦ ᑕᐃᒪᓚᔾᐊᑎᖅᖖᒧ ᒪᓐᑕᑕᑎᐊᖕᓂᖖᒧᑦ ᐱᓯᓂᖅᑎ ᒥᖅᐹᔆᔾᕿᑎ ᐃᓇᖅᒃᖖᒂᖕᓂᖖᒡ ᐱᐱᕲᖕᓂᓗᒡ ᐃᓇᐃᖅᓂᖅᖅ ᑕᐃᒪᐃᑎᑕᐅᓗᒡ ᐊᔭᒃᖕᓂᖖᒐᒡ ᐱᔭᒃᖕᓂᑦ ᑕᐃᑖᔆᔪᑐᐊᖅᑲᖅᐱᖕᓂᑦ. ᐃᔪᖃᖕᓂᓗᒡ ᐃᖅᐊᔪᐊᑦ ᐊᑎᖅᑕᐃᑦ ᐊᖄᓂᖕᓂᑦ ᐊᑐᖅᑎᓗᒍ 2014-2016 ᐃᒪᓐᑎᑐᑕᐅᖅᖅᑲᑦ 1.70 (SE = 0.09) ᐊᒪᓗ 1.61 (SE = 0.11) ᐊᑎᖅᑕᓐᓂᒡ–ᐊᖅᕲᒍᒥ ᐊᒪᓗ ᐊᖅᒍᔾᔭᐸᐧᔪᒡ, ᐃᑉᐱᕆᑎᐊᖅᕐᐊᖅᑐᖕᒡ. ᓇᖅᐸᕇᐊᖅᔆᑦ ᐊᖄᓂᖅᑕᖕ ᖃᖅᔭᐃᖕᓂᖕᓂᑦ ᐊᖅᒍᔾᔭᐃᑦ ᐊᑐᓇ ᐃᖖᒪᖅᓂᑦ ᐊᖄᑦᓗᖕᓂᑦ ᐅᓇᖅᖕᓂᐊᑕᐅᖅᏞᑕ ᐃᒪᖕᓇᖕᓂᑦ 0.39 (SE = 0.10) ᑎᖅᑐᒡᓂᑦ 0.28 (SE = 0.06) ᑕᒪᑭᖅᓂᖕᓂ ᖃᐅᔨᓴᖅᖃᖅᑕᐅᕿ ᐱᓯᓂᐊᔭᓂᖖᓂᖓ, ᑭᔭᐊᓂ ᐃᑐᐊᖅᑭᑦ (MC) ᐊᔭᖅᑭᑎᒃᕐᖕᒡ ᖅᑐᖕᑎᐅᖅᒪᓇᖕᓂᖓᓂᑦ ᐊᖄᓇᖅᓂᖕᑦ ᑕᒪᖃᐅᖃᓗᐊᖅᑎᓐᓗᒍ ᖃᖅᏞᖃᐳᖃᖕᓂᖕᑦ ᐊᒪᓗ ᖃᐅᔨᒪᕆᐊᖅᑕᐅᖕᓂᐧ ᐃᒪᓕᐅᕿᓯᖅᖕᓂᑦ ᐊᔪᐸᖕᓂᐧ. ᑕᐃᒪᐃᑯᒡᐅᐊᖅᑎᓐᓗᒍ, ᑕᒪᖃᑎᓐᓗᒍ ᓇᐅᑄᖅᑐᐊᖅᖕᓂᐧ ᖅᑐᖕᑎᐅᖕᓂᖕᓂᑦ ᖃᐅᔨᒪᔭᕐᐊᖅᑕᐅᖕᓂᑦ, ᐊᔭᖅᑦ ᑐᖅᖃᑎᐊᖅᑐᒃ ᖃᐅᔨᒪᔭᖕᑕᐅᑎᓯᐊᖅᑲᑦ. ᓇᓐᐃ ᑎᒥᖕᑕ ᖃᓇᐃᖕᓂᖅᑦ (ᑐᑭᐅᑎᓗᒍ, ᖃᑦᓇᓐᓘᐊᑕᖖᒪ), ᖃᐅᔨᓴᖅᑕᐅᕝ ᐅᐱᖖᒪᓪᑯᒡ, ᐊᖅᔆᖃᖅᖅᐹᓇᑕᐅᖅᑐᒡ ᐊᖅᓂᖕᓂᖕ ᐊᑐᖅᑎᓗᒍ 1998-2000 ᐊᒪᓗ 2014–2016. ᒥᖅᐹᔆᔾᕋᑎ ᐅᓚᓇᔭᖕᓂᖕ ᓇᓐᐃ ᐅᑉᐅᖅᑲᓚᖅᑐᐊᖕ 2–ᓇᖕ ᐊᒪᓗ ᐊᖅᐊᐅᓂᖅᖅᐹᖖᒧᖕ ᐃᒪᖅᖅᑐᖅᑐᖅᖕ 0.88 (SE = 0.02), ᑕᒪᐃᑯᓚᖅᖕᓂᖅᓗᒍ ᑕᐃᖅᖃᐅᐃᑐᔆᔪᑐᐊᖖ ᐃᓇᐃᖅᐊᖅᑐᖅᖅ ᐊᐸᓇᐊᖕᓂᓗᒡ ᑕᐃᒪᐃᑯᖅᑲᓚᑐᖅᓇᖅᖅᑐᖅᖕᒍᓂᑦ ᐅᔆᖖᒧᓇᑦ ᐅᑎᖅᔭᑳᖅᑎᓐᓂᑦ ᐃᓕᖅᑯᒃᖕᓂᖕᒡ ᓇᓐᐃ ᖃᐅᔨᓴᖅᖃᖅᑕᐅᑯᖅᔆᖖᓂᖕ ᐊᒪᓗ ᑭᓚᑕᓯᓚᖕᓂᖕ ᖃᐅᔨᒪᔭᖕᑦ

ᐊᑐᐃᓐᓇᐅᔪᑦ ᓄᓇᐊᖕᒪᐊᖅᖃᑦᑕᓂᖕᖔᓄᑦ ᐊᒻᒪᓗ ᓇᕘᑦ᠎ᒍᐊᑕᓕᓐᓯᖕᓂ. ᓄᑕᐅᑎᓂᕐᐊᖕᔾᕐᑕᐅᖕᒪᕐᖃᐅᕝᕆᓂᖕᓴᐃᐅᕐᑏᖕᖔᓄᑦ ᐊᒻᒐᖅᖕᓂᖕᖔᕐᕐᕐ ᒥᖃᐅᖅᑕᕐᐅᑎᓂᖕᖔᓄᑦ ᐊᑐᕐᓂᓕᕐᖓᖕᓂ ᐅᖅᐅᑕᐅᕐᖅ ᐅᒪᑐᑦᖃᕐᑦᑦᓂᖕᖔᕐᕐ ᖃᑍᑎᑐᖕᖔᕐᖕ ᐊᕐᖓᖅᖕᓇᓐᐅᔦᐊᖅᕐᖓᖕᓕᑕᖕ 0.88−ᒐᖕ ᑎᐱᐅᕐᖕᖔ᠎ᕈᕝᐅᒪᖕᑎᕐ᠎ᑐᔥ ᐊᒻᒥᕈᕐᐅᖕᓄᖕ. ᓀᖕᑐᓯᓕᕐᖕᑎᖕᒃᕐᓯᖕᓐᑐ ᐊᖕᓇ᠎ᐃᑦ ᐅᒪᑐᑦᖃᕐᑦᑦᓐᓯᖕ ᐊᑐᕐᓄᖕ ᐊᓐᕐᓂᖕᖔᓐᖕ ᐊᒻᒐᖅᖕᓂᖕᖔᕐᖕ ᒥᖃᐅᖅᖕᖕᐅ᠎ᑎᑐᖕᖔᖕᖕ ᐊᖏᖕᖔᖕᖕ 1998-2000 ᐊᒻᒐ 2014-2016, ᒥᖃᐅᖕᓯᒪᖕᑎ᠎ᑦ ᐊᒻᒐᕐᐅᐊᖕᖔᑎᖕᒃᕐᓕᑕ 0.93 ᐊᓕᖕᔪᖕᕆᐅᐊᖕᖕᐅ᠎ᑕ ᐊᒻᒐᖕᓂᖕ ᐊᒻᒐᖕᖕᓂᖕᖔᖕ ᐱᐅᓐᕐᓂᖕᖔᖕᖕᕐᑑᖕᖔᖕ, ᐊᒻᒐᖕᖕᓂᖕᖕ᠎ᑑᖕᖔᖕ ᐱᖏᖕᖕᐊᓐᕐᖕᖕᕐ 2% ᖔᓴᖕᑎᖕᖔᖕ. ᐃᔾᖃᓇᖕᒃᕐᖕ, ᖃᐅᔦᔥᖕ ᐊᔾᔭᖃᖕᑐᐊᕐᖕᔥᖕᔥᑕ ᓄᓇᖕᓯᓇ ᖃᐅᔦᖕᓯᐅᕐᕈ᠎ᑦ ᑕᒃᑯᐊ ᐃᖕᑐᐊᖅᖕᐹᖕ (MC) ᐊᒻᒐᖕᖕᖔᖕᖕᕐ ᖃ᠎ᓇᐃᖕᑐᖕᖕᐊᐃ᠎ᑕ᠎ᑦ ᓇᖕᖕᖕᖔ᠎ᑐᖕᖕᖕᖃ᠎ᑦᑐᖕᑎᖕ᠎ᑐᖕᑦ ᐊᑐ᠎ᑐᖕᖕᑑᖕᓇ 1979-1999-ᒐ (ᐃᓚᖕᖔᓐᑎᐅᖕ᠎᠎ᔥᑑᖕᑦ ᓇᖕᖕᖔ᠎ᑐᐅᖃᖕ᠎ᑐᖕ 34 ᓇᓄ᠎ᐃ᠎ᕐ᠎᠎ᑦ/ᐊᕐᔣᒐ᠎ᒐ). ᓄᓇᒐᖕᑕ᠎ᓴᓇᖕ, ᖃᖕᕐᔥᑦᑦ ᖃᐅᔦᔥᐅᖕᔥᔥᕐ ᐱᐅᔥᖃᖕ᠎ᓂᕈᖕᖕᑎᖕ᠎᠎ᑦ ᓂᖕᖕᖕᕐᑎᖕ ᖃᓄᐊᖕᓯᖕᖕᒐᖕ ᐊᒻᒐ ᐊᖕᔥᐊ᠎ᔥ ᐊᒻᒐᖕᖕᖕᑦ ᐱᖕᖕᑎᐊᖕᓯᖕᖔᖕᕐ ᐊᑐᖕᑎᖕᖕ᠎ᑐᖕᑦ ᐱᖕᖓᖕᖃᖕ᠎ᓂᖕᖕ ᑐ᠎ᕐᖕᖕᑐᐃᖕᖕ᠎ᓇᖕᑕᖕᑦ ᐃᓂᖕᖃᖕᓂᖕᖔᓄᖕ ᐊᒻᒐ ᐱᖕᖕᖃᖕ᠎ᑐᐊᐱᖕᖕᖃᖕᓂᖕᖔᓄᖕ ᐃᐱᐊᐅᖕᕈ᠎ᒐᖕᓇᖕ ᑕᐃᖕᔪᖕᖕᕝᖕᕐ ᐊᒻᒐ ᐊᒻᒐᖕ᠎ᓂᖕᖕᕐ ᖃᓪᒐᖕᑑᖕᕐᑑᖕ᠎ᓇᖕ ᐱᖏᖕᖕᑐᐱᖕᖕ᠎ᑕ᠎ᓄᖕᖔᖕᕐ ᐃᖕᑐᐊᖅᖕᐹᖕ (MC) ᐊᒻᒐᖕᖕᖔᖕᖕ᠎ᓄᖕ ᑕᐃᐊᖕ᠎ᑎᑐᖕᓇ ᓴᖕ᠎ᓂᖕᖕᐅᖕ᠎ᓂᖕᖕᑎᖕ᠎ᓄᖕ ᔪᖕᕐ ᖃᓄᐃᓇᖕ᠎ᓂᖕ᠎ᑕᖕᑦ (ᑑᑎᐊᐅᖕᑐᖕ, ᖕᕐᖕᖃᖕ᠎ᓂᖕᖕᕐᖕᑦ᠎ᓄᖕᖕᕐ ᐃᔥᕐᖕᖕ᠎ᑕᖕ, ᔪᖕᑐᖕᖕᑐᖕᔥ) ᐊᒻᒐ ᐊᖕᖔᖕᖕᖕᑕᖕᖕ᠎ᓂᖕ ᐅᖕᕐᖕ ᖃᑐᖕᖕᖕᑎᑐᖕᖃᖕᖕᕐᖕᑦ. ᑕᐃᒪᐃᖕᖃᖕ᠎ᑐᖕᑎᖕ᠎ᑐᖕ, ᔪᖕᓇᖕᑦ ᐊᔾᔥᖃᖕᕝᖕᕐᖕᑕᖕᓇᖕ ᐱᖕᑐᐊᖕᑕᐅᖕᖕᕐᑦ ᐊᖕᖔᖕ−ᑕᖕᕈᔥᖕᓇᖕᑑᖕᖃᖕ ᔪᕐᕉᖃᖕᖕᑦᑎᖕ᠎ᔥᖕ ᓇᓐᕐᖕ᠎ᓄᖕ ᐊᒻᒐ ᐱᖕᑐᐊᖕᑦᖃᖕᖕ᠎ᓇᖕᑎᖕ᠎ᑕᖕ᠎ᓄᖕᐅᖕᑎᖕᖕ ᐅᖕᑐᖕᖕᓄᖕᑦ ᐃᖕᑐᐊᖕᑦ᠎ᑕᖕᑦ (MC) ᐊᒻᒐᖕᖕᖔᖕ᠎ᓂᖕᖕᕐ ᐱᐅᖃᖕᔥᑐᖕᑦ ᐊᖕᑐᖕᖕᕝᑕᐅᖕᓂᖕᑐᑐᖕ᠎ᓇᖕᖕᓐᖕᖕᕝᑕᖕᓕᖕ ᔪᖕᖃᖕᔾᖕᖃᖕᕝᖕᕐᖕᓇᖕᑐᐊᖕᓇᖕᖕᖃᖕᕝ, ᑕᐃᒪᖕᖕᓇᖕ᠎ᑕᖕᕐᖕ ᐊᔾᖕᑎᖕ᠎ᑐᖕᑦ ᓇᓄᐃᖕᑦ ᐊᒻᒐᖕᖕᖔᖕ᠎ᓂᖕᖕᕐᑕ, ᒪᖕᖕᓇ ᖃᐅᔦᔥᔥᐅᖕᖕᕐᒐ᠎ᑕ. ᒥᖃᐅᔥᔾᖕᓇᖕᖕ ᐅᓄᖕᕐᔥᕝᕐᖕᖕᐊᖕᓇᓂᖕ᠎ᓇᖕᑦ ᖃᓄᐃᖕᕝᕐᖕᖕᐊᖕᖕᕐᖃᖕᑦᑦᓂᖕᖔᓄᖕ ᐃᖕᑐᐊᖕᑐᒐ ᐊᒻᒐᖕᖕᖔᖕᖕᕐᑕ ᓇᓄᓇᐃᖕᕝᔾᖕᕐᕈᖕᑕ᠎ᑦ ᐱᓄᖕᐊᓄᖕᔥᕐᕐ᠎ᑕᐃᖕᕈᖕᑎᐊᖕᖕᖃᖕᖕᕐᖃᖕᓂᖕᖔᖕᓇ ᐱᕐᑕᖕ᠎ᑎᖕ᠎ᑎᖕᖕ ᒐᖕᕝᕐᕈᖕᔾᖕᑐᖕᖕᕐᖕᓄᖕ ᖃᐅᔦᔥᓇᖕᕝᔥᐃ᠎ᑦ ᐊᖕᕐᖕᖕᑎᖕᖕᕐᑦ, ᐱᔥᐅᖕᖃᖕᖕ᠎ᓂᖕᖕ᠎ᑐᐊᖕᓇᖕᕐᕝᔾᖕᑦᖕᔥᖕ᠎ᓇᖕᖕᕐ᠎ᓄᖕᖕᕐ ᐱᖕᖃᖕ᠎ᓂᖕᖃᖕᑕᐅᖕᔥ᠎ᓂᖕᖕᕐᕐ ᑙᖕᓇᖕᑕᐃᖕᖕᐊᐅᖕ᠎ᔥ ᓇᖕᕝ, ᐊᒻᒐ ᐱᑎᖕᖃᖕᖕᕐᖕᖕᓂᖕᖕᕐᖃᖕ᠎ᓄᖕᖕ ᓄᓇᐊᖕᖕᐊᒐᖕ᠎ᓂᖕᖕ᠎ᑦ ᑐᖕᔥᐅᐃᔾᖕ᠎ᑎᖕ᠎ᕝᔾᐃᐊᖕᑦ ᓄᖕᖕᖃᖕᖕᑎᖕ᠎ᑎᖕ᠎ᔥᖕ᠎ᕐᕐᒐᖕᖕᑦ ᖃᐅᔦᖕᓇᖕᕝᐊᖕᖕ᠎ᓂᖕᖕᒐᖕᕐᖕ ᑕᖕᔥᒐᖕᓇ ᖃᐅᔦᖕᓄᐊᖕᕝᐊᖕ᠎ᓂᖕᖕᒐᖕᕐ ᔥᖕ᠎ᔥᖕ᠎ᑐ ᑕᒃᑯᐊ ᒥᖃᐅᔥᔾᖕ᠎᠎ᕝᑎ᠎ᑦ ᐊᒻᒐᖕ᠎ᓯᖕᖕᕐ᠎ᓂᖕᖔᕐᖕᖕᓇᖕᕐ ᐊᒻᒐ ᐅᒪᑐᑦᖃᕐᑦ᠎ᓂᖕᖔᕐᖕᖕᓇᖕᕐ ᐊᓕᖕᐳᖕ᠎ᑐᖕ᠎ᑎᐅᖕᕝᔾᖕᕐᕐᒐᖕᖕᑕ ᓄᓇᓇᖕᖕᕐ᠎ᑎᖕᓇᖕᕐ᠎ᔥᑑᖕᖕᑖ ᐃᓚᐃᖕᖔᖕ᠎ᓂᖕᖕᕐᖕᕐᑦ. ᒥᖃᐅᔥᔾᖕ᠎ᓇᖕᔥᔾᕐ᠎ᔥ ᑭᓕᖕᖃᖕ᠎ᑐᐊᖕᖕ᠎ᑕ ᐊᐃᖕᕝᕐᖕᖕᖕᓇᖕ᠎ᑐᐊᖕᖕ ᐱᖕᖃᖕᖕ᠎ᓂᖕᖔᒐᖕᕐ ᐃᖕᑐᐊᖕᖕᑐᒐᖕ᠎ᑐᒐᖕᕐ ᐊᒻᒐᖕ᠎ᓯᖕ᠎ᓇᖕᑐᐊᖕᕝᑐ᠎ᑦ ᓀᖕᓯᖕᖃᖕᑐᖕ᠎ᓇᐅᖕᖕᑕᖕᕐᒐᖕᕐᒐᖕᑦ ᐊᑐᖃᖃᖕᓇᖕ᠎ᖃᖕᕐᖕ᠎ᐅᖕᖃᖕᖕ᠎ᔥᖕᖕ᠎ᓇᖕᕝᖕᓇᖕᕐ᠎ᓄᖕᕐᑦ,

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ᐊᖅᑭᒃᓱᐃᖅᑕᐅᓯᒪᓗᒋᑦ ᖃᐅᔨᓴᕐᐊ�ᕐᓂᐅᑉ ᓴᓇᕐᓗᓂᖏᒃ ᐊᒻᒧ ᑕᐅᐳᐊᓂᖏᓐᓄᑦ ᐊᒥᓲᓂᖅ ᐱᓂᐊᕆᖅᐸᑦᑎᑎᓐᑐᓂ
ᐃᓐᖅᑯᕿᐊᖃᕝᖏᑕ ᑕᒃᑯᑎᒃ ᐱᓂᐊᔪᕐᒣᕗᕐᒦᑦ ᖃᐅᔨᓴᕐᐊᕐᑕᐅᕐᓓᕗᕐᑐᕐᔾᓕᔭᓯ ᐊᒻᒧᖅᑭᐳᖁᖃᓕᑕᐃᓯᑦᑯᒥᐃᓴ
ᐂᐊᑕᕿᒪᕐᒦᓄᑦ ᐱᓂᐊᕆᖅᔪᕐᒨᒦᓐ. ᐊᓯᖅᒦᓯᓂ ᐅᒦᕿᑦ ᐊᒻᒧᖅᒥᓂᓯ ᐅᖁᖐᓂᒃ ᐅᕐᒦᒿᖁᒦᓯᐐᕿᓴᒥ
ᐱᔪᖅᓯᖃᓴᕐᖏᒃᓕ ᐊᖓᓐᒃᐊᖁᓯᔪᕐᒥᓂᒃ ᐃᑕᐃᐊᕿᑐ (MC), ᐊᔾᓯᑕᕐᐄᑕᕿᒦᒥᒐᕿᑦ ᒥᐊᖐᕼᐊᖏᕐᔪᕐᖓ ᐃᑎᖁᒦᕼᒦᓯᓴ
ᖅᔾᑕᕆᑐᔭᒦᖁᒦᖏᓯᒃᓂ ᖃᐅᐊᓯᒐᓐᖃᒥᕼᖁ ᐊᒻᒧ ᐅᐊᑐᒦᖅᖐ ᖃᐅᐊᖃᕐᑕᐅᒐᓐᖃᒦᒥᒐᒦᓯᒃ ᐌᕿᐅᒥᖁᒥᒐᓐᒃ
ᖅᐐᖁᒐᐊᕐᒐᐊᒥᕼ ᐊᖃᓐᒥᕼᒦᒥᓂ ᓯᐊᖁᖁᑕᕼ ᖃᐅᐊᕐᒥᖁᑕᐅᕼᓴᖅᖐᒦ ᐊᒻᒧ ᕿᖁᓯᓯᒥ ᑕᓓᐊᒦᓯᒃ
ᐊᒻᒦᖁᒐᖁᒦᑕᐅᖕᐊᒦᕼ ᖃᐅᐊᕼᒦᖁᒦᕼᒐᓐᖃᕐᖏ ᑕᕙᓪᒥ ᐱᓂᐊᕐᔪᕐᖁ, ᐄᒻᖓᓕᓯᕝᕿ 5-7 ᐊᓐᓵᒍᓂ
ᓯᐊᖁᒦᒐᕼ-ᑕᖃᒥ ᐃᕙᐸᐃᕼᔪᓕᖁᔪᑐ ᓯᕼᕼᔭᔭᕼᒐᕐᐊᕐᒥᕼᒦᒐ) ᐅᕐᒦᒐᓐᒃ ᐊᒐᐅᒥᕝᖁᐊᑕᕐᖅᖐᑕᐅᕼᐊᒐᓐᒦᒐ
ᖃᐅᔨᖁᐊᕐᕿᕼᒦᖁᕼ ᐊᒐᐅᕼᐄᒦᖁᕼᒦ (ᑐᕿᑎᐅᖐᓄᒐᐅ 4-5 ᐊᓐᓵᒍᒥᒐᓯ), ᐊᕼᕿᓂᕼᒦᕼᑕᐅᕼᖐᒿᐊᕼᒦᒿᕿᒦᒐ
ᓯᖁᓴᕼᖁᕼᑎᐊᕐᓂᕼᒐ ᐅᒦᖁᒿᖁᕼᑕᕼᓂᕼᒐ ᖃᐅᐄᒿᑎᐊᕼᒦᒿᕼᒦᒐ, ᐊᕼᐊᐅᐊᑐᐊᓐᒦᐊᖁᕼᖐᕼᒦᒐ, ᐊᒻᒧ
ᖅᔾᒿᐅᖅᖐᕼᑕᕼᓂᕼᒐ. ᓯᕼᕼᕼᒐᓐᕼ, ᓴᓐᐊᖁᒥᐊᖁᓂᕼᒦᒐ ᖃᐅᐊᖁᒦᖁᒦᕝᖁᕼ (ᖃᕼᐅᓕᑕᖅᖐᒿᑕᐅᕼᒿᓕᐊᕿᒦᖁ
ᓐᓂᕼᒿᕼᐐᕼᓂᐐᕿᓐ) ᐊᑐᖃᑕᐅᕼᒿᕼᐅᕝᐅᕿᕼ. ᓯᐊᖁᒿᓂᒥᐊᖁ ᖃᕼᒿᐅᓕᑕᖅᖐᑕᐅᕼᒿᓕᐊᕿᒦᖐᒿ ᓐᓂᕼᒿᕼᐐᕼᓂᐐᕼᒿᓕᓐ
ᖃᐅᐊᖁᒦᖁᒥᕼ ᓂᒥᐊᕼ ᓴᓐᐊᖁᒥᐊᖁᓂᕼᒦᒐᓯ, ᐊᑐᖃᑕᐅᕼᒿᖁᒿᒐᕿᒿ ᐱᓂᐊᕐᔪᕐᕼᖁᕼᒿᒐᕝᕼ ᖃᐅᐊᖁᒦᖁᒦᕝᖐᐊᑕᐅᕼᕼᖅ
ᖃᐅᔨᖁᐊᕐᕿᕼᒦᑕᐅᕝᒿᒐᓐ ᑕᐅᕝᒿᖁᕼᓴᕿ ᐅᒐᓐᖁᒥ ᐃᑕᐊᐊᕿᑐ (MC) ᐊᒻᒧ ᖃᓂᕿᖁᓂᕼᖁᑐᕼ ᐊᒻᒿᕼᕿᕼᒿᒿᕼᒐ
(ᑐᕿᑎᐅᖐᓄᒐᐅ, ᐸᓐᓂᕼᖅᑕᕝ ᕼᐄᕼᕝᖁᐊᒐᓂ, ᐊᒻᒧ ᐃᖁᖁᖐᑕᕝ ᐃᒥᓐᖁᓂᕼ).

1. C) KAVAMALIQINIRNUT NAINAAQHIMAJUQ –

Nanuit (*Ursus maritimus*) munariyauyut tamainni Nunavunmi, Kaanatami, angunahuarniqmut munariniqmulu qiniqtut naunairiami angunahuarniq munariyauyuq ilitariyauyuqlu munariniqmut piyangit. Taimaa 10nik ukiunik, hilaup aadlangurninnga tamainni Ukiuqtaqtumi aadlanguqtitait nanuit nayugangit aadlatqiiktumik nampanik. Pihimagiami nakuuyumik nanuit amihunik ilagiyanganik avatinganik tuniyuqlu inuujjutikhangit Inungnut, nalunaqtunik naunaiyainiq munarininnganiklu piyait naunaiyariami amihunik qanurittaakhaanik taimaalu munarininngit piliqtait. Hamani uqaqtavut qanurittaakhaanik amihuuninnginnik naunaiyainiq nanuqnut nayugangit Ittuaqtuuq (MC) havaktait 2014-2016. Nutaat uuktuutingit katitiqtauyut aturhutik mikitqiamik-pittailiniq ihariagiyainnik niqinginnik piiyaqtauniq kapuqtauyut nutqaqtihimaittumik akhuraalukluuniit pilugit nanuit. Ihivriurutivut ilaliutihimayuq nampangit uumannga 2014-2016 niqinginnik piiqtauniq naunaiqtait piffaarhugit nanuit naunaiyainiq, tuqutihimaittumik piplugit nampangit katitiqtait uumani naunaiyainiq piffaarhugit naunaiyainiq 1998-2000, kikliqaqtumiklu angunahuarniq piffaarninnga nampangit tamainni uumani 1998-2016.

Qanurittaakhaanik umikhimayumik piyait piffaaqtait uuktuutigiplugu, iliuraqtuq uuminnga Nampanik ihivriuqniq tunngavinga ihuarhaqtauyuqlu nampangit inmikkuuqtunut huradjat (ukunatitut, >2 ukiunik), piyuq piqarninnganik itqurnarutauyuq uuminnga 716 (95pusantmik Itquumayuq Nutqarninnga [CRI] = 545 – 955) uumunnga 2014-2016, naunaiqtait tamna MC nanuit amihuuninngit angikliyuumiqtuq uumannga amihuuninnganik uumani 1998-2000 (325 [95pusantmik CRI = 220 – 484] uumani naunaiyaininngani; 284 [SE: ± 59.3] uumani Taylor aadlallu. [2006]). Tamarmik

anguhalluk arnallatlu ilagiyanga uuminnga amihuuninngit angikliyuumiqtuq naunaiyaqtillugit (1998-2000 uumanilu 2014-2016), aadlatqiiktumik ikikliyuumiqtuq angunahuarniqmut akhuurniq ihuaqhaqhimayuqlu nayugangit qanurittaakhaanik. Atuqtugut qaffit inuuyut angikliyuumiqtut tuqutut uuktuutigiplugu taimaa nampangit piqalluanginmat uuktuutikhamut amihunik kikliqarninnga. Itqurnarninnga amihuuninnganik pipkaijjutauyukhaq qayagilugit taimaa naunaiqmata "amigaininnga" (ukunatitut, ilaliutikmata tamaita nanuit atuqtait MC munarininnga, ilangit nayuqpaktut aadlani amihuuninngit ukuatlu) ilaliutiniaruknaqhiuq nakuuyumik ihuittumik piyuq ulapiqutiyuq uuktuutimut maliktakhangit unalu nakuungittumik ihuittumik pipkaijjutauyuq aadlatqiininnganik pigaangamitkik. Tamainnit nanunnuangit aktikkulaangit uumunnga 2014-2016 ittuq 1.70 (SE = 0.09) unalu 1.61 (SE = 0.11) nanunnuangit-ukiungani unalu atauhiqmik ukiulik, inmikkut. Naunaiyarhimayuq qaffiuyut atauhiqmik ukiulik atauhiqmut arnallakmut mikhiyuq uumannga 0.39 (SE = 0.10) uumunnga 0.28 (SE = 0.06) tamarmiknit naunaiyaininnganik, kihimi MC nakuuyumik piyuq nanuit amihuuninngit humaangittuq mikhiyuq qun'ngiaqtauyuqlu tariup hikunga aadlangurniq. Kihimi, tuttumik nanunnuaqarninnganik nampangit, piliulimaittugut ihariagiyauyumik ihumagininngit. Nanuit timingit qanurittaakhaanik (ukunatitut, puvalaniq), naunaiyarhimayuq upin'ngakhami, angikliyuumiqtuq uumani 1998-2000 uumanilu 2014-2016. Itqurnarutauyut naunaittuq inuujjutingit nanuqnut ukiulgit 2mik avatqumayuqluunit 0.88 (SE = 0.02), taimaa pilimaittuq ihuittuq mikhiyuq tadjakaffukmut ingilrainnaqtullu nanuit naunaiyaininnganut kikliqaqtumiklu nampangit piqarninnga piyuq tikittunik nuutiqtirninngalu. Una naunaiqtauyuq angikliyuumiutinganik amihuuninnganit itqurnarninnga tamainni naunaiyaiyuq inuujjutingit nampangit angitqiyauyukhaq

uumannga 0.88 pigiami angiyumik angikliyuumirniq. Taimaa naunaiyarmatku

angayukhiuyut inuujjutingit aturhugu aadlangurninnga amihuuninnganik itqurnarninnga

uumani 1998-2000 uumanilu 2014-2016, itqurnarutauyut nampavut uuminnga 0.93

pitquyait amihuuninnga angikliyuumiutinga nakuuyuq, piqarhuni angikliyuuminnga

nampanga 2pusantmik. Tamainnit, naunaiqtavut aadjikutariyaa nunallaani ilihimaniq

tamna MC amihuuninngit piffaaqtait amihumik angunahuarniq piyuq uumani 1979-1999

(angunahuarninnga 34 nanuqnik/atauhiqmi ukiumik). Avatininnganut, ihumagiyaqqut

qun'ngiaqtauniq ihuarhiyut timinginnik akhuraaluklu amihuuninnga angikliyumiqtuq

taimaa piyuq inikhanganik qangarnitamik ikikliyuumiqtuq tariup hikunga

qanurittaakhaanik qanuraaluklu tuniyuq tadjakaffuk ikayuutauyuq uumunnga MC

amihuuninngit pikmat uumannga tualihimayuq hikunga (ukunatitut, ikikliyuumiqtuq

hilikninnga, amihunik ukiut hikunga) unalu angikliyuumiqtuq inuujjutinganik

qanurittaakhaanik. Kihimi, hilaup aadlangurninnga hivulliutinga akuniraalukmik

qayangnarutauyuq nanuqnut aullaqtirininnga uumannga MC amihuuninngit

nakuungittumik ayurhautipkaiyuq hikuirninnganit, taimaatut ilangit aadlat nanuit

amihuuninngit, tadja naluyait.

Itqurnarutiyuq piyuq amihuuninnganik kiklikhangit uumunnga MC amihuuninngit

naunaiqtuq akhuurutauyuq taimaa mikkait uuktuutingit aktikkulaangit, piqalluanginmat

piniaruknaqhiyuq piffaarumitkut tamna piyaraluangit nanuq, piqalluanginmallu

ingilrarninnga naunaitkutingit pitquyauyuq naunaiyainiq uumani naunaiyaqtamiknik

taimaa itqurnarninnga amihuuninnganik inuujjutingalu taimaa ihuittuq. Itqurnarutikput

piyaa tuglianganik MC amihuuninngit naunaiqtauyuq uumani aadjikutaliurhimayuq,

ihuarhaqhimayuq naunaiyaininnga piliurninnga talvuuna tuniyuq amihunik

pivikhaqautikhaq ayuiriami tahapkunanit atuqtamiknik naunaiyaininngani nampanganiklu katitiqtut piplugu. Aadlanut huradjat amihuuninngit avatingaluuniit atuqtait aadjikutariikninnga uumunnga MC, katitiquyavut aadlanik nanuliurniqmut nampangit unalu auminganik uuktuqtut qitqani uumannga nutaamit naunaiyainiq aippaangalu iluittuq amihuuninngit naunaiyainiq (Nunavutimi, inniaqtuq 5-7 ukiunik maniqqamungaqtinnagu havaanga iniqtaukpat) angikliyuumirluguluuniit naunaiyaininnga qanuraaluktut piyakhaq (ukunatitut 4-5 ukiunik), angikliyuumiriami ilihimaninnga inuuyunik nampanginnik, unaluuniit ahinunngauyut, nanuliurniqlu. Unalu, ingilraninnga nampangit (saatalaitkut nipiliurniq tunigiamilu taiguqtanginnik) pitquyauyuq. Piqangitkumi saatalaitkut nipiliurniq tunigiamilu taiguqtanginnik nampangit nanuit ingilraninnginnik, pitquyavut pigumik ihivriurninnga nampanganik ihivriuriami himmautingit uumannga MC qanittullu amihuuninngit (ukunatitut, Aqqusiriaq, uumanilu Kangirłurulukmilu).

2. INTRODUCTION

Wildlife managers face complex decisions when seeking to balance conservation and human priorities. Decisions and outcomes must be evaluated periodically so that new information can be fed back into an adaptive management framework (Holling 1978, Lancia et al. 1996, Johnson 1999). Accurate and up-to-date estimates of population abundance are often a key component of informed management decisions (Nichols and Williams 2006). Typically, new estimates of abundance are acquired periodically according to a monitoring interval that is determined by management objectives, species' biology (Gibbs 2008), and available resources. As climatic changes affect many areas around the globe, shortened monitoring intervals may be required to understand the concurrent effects of management interventions and environmental change. Broadly, more frequent monitoring increases the probability of meeting management objectives and reduces the severity of potential negative outcomes (Taylor et al. 2007, Regehr et al. 2017).

One species that has received significant monitoring attention is the polar bear (*Ursus maritimus* Phipps 1774). Polar bears are characterized by having delayed maturation, small litter sizes, and high adult survival rates (Bunnell and Tait 1981). They are at the top of the Arctic food chain and depend on the sea ice for hunting, travel, mating, and in some instances denning (Amstrup 2003). Sea-ice loss resulting from climate change is predicted to impact polar bear subpopulations severely in the future (Derocher et al. 2004, Stirling and Parkinson 2006, Amstrup et al. 2008, Durner et al. 2009, Stirling and Derocher 2012, Atwood et al. 2016, Regehr et al. 2016). The global polar bear population, consisting of 19 subpopulation units, is estimated to be approximately 26,000 polar bears (Obbard et al. 2010, Wiig et al. 2015). There is not currently empirical evidence for declines in global abundance due to sea-ice loss (Regehr et al. 2016). However, accurate assessment of such changes is complicated by poor data for many polar bear subpopulations (Durner et al. 2018, Hamilton and Derocher 2018), spatial and temporal variation in the effects of sea-ice loss, and the fact that some subpopulations have likely recovered in recent decades from overexploitation

prior to the 1973 Agreement on the Conservation of Polar Bears (Honderich 1991, Larsen and Stirling 2009).

Despite on-going research and monitoring efforts on polar bears to date, reliable and updated abundance and demographic information about all subpopulations is still lacking (Obbard et al. 2010, Vongraven et al. 2012, Durner et al. 2018). Polar bear research is expensive and logistically challenging, especially for management jurisdictions that oversee more than one subpopulation. Nunavut, Canada, is home to 12 subpopulations (8 shared with other jurisdictions, 4 entirely within Nunavut; Obbard et al. 2010) and as such, carries the major responsibility of polar bear research and management in Canada. In order to maintain healthy and viable polar bear subpopulations, population studies in Nunavut are carried out on average within a 10 – 15-year rotational cycle, which can vary depending on research needs and priorities (Hamilton and Derocher 2018) along with available resources. Here we present findings from a 2014 – 2016 monitoring study to re-estimate abundance of the M'Clintock Channel (MC) polar bear subpopulation.

M'Clintock Channel is entirely managed by Nunavut, Canada (Figure 1) and an initial physical mark-recapture study was carried out from 1973 – 1978 (Furnell and Schweinsburg 1984) for both MC and the adjacent Gulf of Boothia (GB) subpopulation together as a single demographic unit. The total abundance estimate for both areas was 1081 bears. The estimate was known to be biased by non-representative sampling and was subsequently increased to 900 for GB and 900 for MC based on back-calculations to determine abundance levels that would be necessary to sustain the existing subsistence harvest levels (Aars et al. 2006, Taylor et al. 2006) along with local indigenous knowledge.

In the mid-1990s, the MC estimate was revised downwards to 700 based on hunter reports of reduced densities of polar bears (Aars et al. 2006, Taylor et al. 2006). M'Clintock Channel and GB were later delineated based on movements of satellite radio-collared adult female bears, recoveries of research tags in the harvest (Taylor and

Lee 1995, Taylor et al. 2001), and Inuit knowledge about how local conditions may influence the movements of polar bears (Keith et al. 2005). Genetic analyses based on microsatellite data also suggested some level of differentiation between the MC and GB subpopulations, although the magnitude of population structuring was higher among females than males (Campagna et al. 2013). Past harvests in MC of 34 bears/year from 1979 – 1999 were considered unsustainable (Taylor et al. 2006), resulting in a harvest moratorium from 2001/2002 – 2003/2004 and a reduced harvest of 3 bears annually until 2015. Prior to the current study, the most recent estimate of abundance for the MC subpopulation was 284 bears (SE: ± 59.3) from a physical mark-recapture study conducted 1998 – 2000 (Taylor et al. 2006). In recent decades the subpopulation has been managed to achieve recovery, and local knowledge indicates that more bears have been seen in the 2000s by Gjoa Haven and Taloyoak hunters during their travels across the sea ice (Dyck personal communications with hunters during consultation meetings 2013). This perceived increase in abundance resulted in an increase in the annual harvest from 3 to 12 bears at a 2:1 male to female harvest sex ratio, beginning with the 2015/2016 harvest season.

Nunavut's polar bear co-management system is based on memoranda of understanding (MOU)[1] developed between each community's Hunters and Trappers Association and the government. These MOUs lay out harvest, management and research aspects for each polar bear subpopulation. Under the existing MOU that was co-signed by all parties in 2005, the Government of Nunavut (GN) committed to a new population study for MC. The new study had the objective to estimate the current subpopulation size and composition, and to compare those results to the former study so that this information would be available to responsible management authorities for decision-making. In addition, we sought to obtain data that would provide estimates on survival and reproductive parameters in order to allow population viability analyses. Lastly, by implementing a research method that was minimally invasive but supported by local communities and stakeholders, we sought to evaluate whether genetic mark-recapture can be a useful alternative in population monitoring (Vongraven and Peacock

[1] The MOUs were replaced by the Nunavut Polar Bear Co-Management Plan in September 2019

2011, Vongraven et al. 2012). To address these objectives, we conducted a genetic mark-recapture study from 2014 – 2016.

3. STUDY AREA

The current management boundary for the MC subpopulation (Figure 1) is mainly based on telemetry data for adult female bears that were fitted with radio-collars in adjacent subpopulations, and tag returns from harvested bears (Schweinsburg et al. 1982, Bethke et al. 1996, Taylor et al. 2001). This boundary has also been supported by recent genetic analyses (Campagna et al. 2013, Malenfant et al. 2016) although its validity has been questioned by Inuit (Keith et al. 2005). The MC study area (about 495 000 km^2 including land mass, or 140 000 km^2 of sea-ice; Barber and Iacozza 2004, Hamilton and Derocher 2018) is bound by Victoria Island to the west, Prince of Wales Island to the north, Boothia Peninsula to the east, and the Nunavut mainland to the south (Figure 1). These land barriers are believed to restrict bear movement in and out of the study area. A detailed description of the physiography, currents, and sea ice of the region can be found in Schweinsburg et al. (1981).

Over the past 20 – 30 years, there has been a change in sea-ice quantity and composition. Multi-year sea ice has declined and been replaced by annual ice (Schweinsburg et al. 1981, Rothrock et al. 1999, Comiso 2002, Barber and Iacozza 2004, Keith et al. 2005, Howell et al. 2008, 2009, Sou and Flato 2009, Perovich et al. 2018, Richter-Menge et al. 2018, Environment and Climate Change Canada 2019). This has resulted in a smoother sea-ice platform interspersed with long pressure ridges, with rougher multi-year ice generally limited to localized areas (i.e., M'Clintock Channel proper; Dyck pers. observations). For most of the year, the sea is completely ice covered except for a few small polynyas that attract seals, polar bears, and other species (Hannah et al. 2009, Stirling 1997). From approximately mid-June to July, wide cracks form and extend for miles, providing haul-out habitat for ringed (*Pusa hispida*) and bearded seals (*Erignathus barbatus*) and thus, good foraging habitat for polar bears. From August to early October, much of the sea ice disappears in the southern

and eastern portion of the study area, allowing shipping traffic along the Northwest Passage route (Stewart et al. 2007, Howell et al. 2008, 2009, 2013a, Analyse and Strategi 2011). Currently, some multi-year ice remains in M'Clintock Channel proper year-round – ice that originates and is pushed south from the Queen Elizabeth Islands and M'Clure Strait (Howell et al. 2008, 2013b). How important the contemporary remaining summer multi-year ice in M'Clintock Channel proper is to MC polar bears is currently unknown.

4. METHODS

Sampling – field collections

Our study design followed that of the previous physical mark-recapture study conducted in MC between 1998 – 2000 (Taylor et al. 2006; Figure 2); however, it did not involve the immobilization and physical handling of bears. Inuit co-management partners in Nunavut expressed their concern over wildlife capture and handling during a wildlife symposium in 2009 (Lunn et al. 2010, Department of Environment 2013). As a result, the responsible government management agency explored alternative research methods. Given the presumed low densities of bears (Hamilton and Derocher 2018) and the vast study area, genetic mark-recapture was chosen as the method since it is minimally invasive (Garshelis 2006) and has been successfully applied on various species, including bears (Brown et al. 1991 (right whales (*Eubalaena glacialis*)), Palsbøll et al. 1997 (humpback whales (*Megaptera novaeangliae*)), Boulanger et al. 2004, Olson 2009 (brown bear (*U. arctos*)), Pagano et al. 2014, SWG 2016 (polar bear)). We obtained genetic material for individual bears from a small sample of skin and hair collected via a remote biopsy dart (Pneudart Type C - Polar Bear) fired from a dart gun (Capchur Model 196) from inside a Bell 206 Long Ranger helicopter (Pagano et al. 2014). The extracted DNA was used to identify individual animals without the need for ear-tagging or lip-tattooing (see section "Genetic analysis"). Recaptures occurred when a previously sampled bear was biopsy-darted on a later occasion or when a genetic sample was recovered through the Nunavut polar bear harvest monitoring

program. Every hunter in Nunavut is required to submit samples to the polar bear harvest lab so that age, gender and various other variables can be used in any ecological or demographic assessment (Nunavut Wildlife Act, SNu 2003).

We initially intended to begin field work in early April, but poor spring weather forced us to wait until late April to early May each year between 2014 and 2016. Field work usually was completed by mid-June. Approximately 80% of the entire MC study area was searched every year though poor weather and unsafe flying conditions prevented us from searching the entire study area during each field season, and we were not able to sample M'Clintock Channel proper in any year. In mid-April 2016, we used a Twin Otter for a reconnaissance flight over M'Clintock Channel proper to assess bear presence and sign. This allowed us to infer whether this portion of the study area potentially contained animals that were unlikely to be exposed to sampling effort unless they moved into areas that were searched by field crews.

Searches for bears were conducted at approximately 100 – 120 m above sea level, and at average speeds between 120 – 150 km per hour. Search areas were initially discussed with hunters and local Hunters' and Trappers' Associations during pre-study consultation meetings to gain insight about sea-ice conditions and bear distribution. Also taking past capture locations (Taylor et al. 2006) into account, we searched the sea ice, adjacent coastal areas, and small islands of Coronation Gulf, Dease Strait, Victoria Strait, Franklin Strait, James Ross Strait, Larsen Sound, Rae Strait, and Queen Maud Gulf during 2014 and 2015 (Figure 1). We decided to forego the Coronation Gulf and Queen Maud Gulf areas in 2016 because we observed very low bear activity and presence during our survey flights, and local knowledgeable hunters also indicated and confirmed that bears are rarely seen in those areas.

In order to minimize potential sampling bias, and to allow replication of this study, we used a "semi-structured" sampling approach. Generally, we flew transect lines across the sea ice and small islands with search intensity proportional to apparent bear activity (or bear presence). When signs of bears (e.g., tracks, bears, seal kills) were rare

or plentiful, search transect lines were spaced further apart (i.e., 11 – 16 km), or closer to each other (i.e., 7 – 10 km), respectively. In that fashion, we were able to cover large sections of the study area efficiently (Figure 3). We decided to fly our survey transects from east to west and vice versa whenever possible (e.g., perpendicular to suspected density gradients based on past capture and harvest locations).

Once we located a bear, a small sample of tissue (<5 mm diameter), mostly skin with some adipose tissue attached to it (Pagano et al. 2014), was taken from the rump area at an approximate distance (or altitude) of 3 – 7 m using a biopsy dart (5CC Polar Bear Biopsy DNA Dart, Pneu-Dart Inc., Williamsport, PA). All bears except cubs-of-the-year (C0) were sampled. Cubs-of-the-year in early spring are still small and easily confused (Atkinson and Ramsay 1995, Robbins et al. 2012), so we decided not to dart C0s to avoid possible injury and the splitting-up of family groups. Every bear that was biopsied received a unique field identification number so that the genetic results and our field data could be cross-referenced and linked.

The biopsy darts are designed to fall to the ground after impact and are retrieved without physically handling a bear (see Appendix A for images). The effectiveness of these darts for sampling polar bears has been previously demonstrated (Pagano et al. 2014, SWG 2016, Dyck et al. 2020). The darts are quick and easy to use and require less pursuit time of bears than during capture operations. On average, it took less than 4 minutes from when a bear was initially spotted to the time when the dart was picked up after sampling a bear (GN, unpublished data). The design and relatively low velocity of the dart means that risk of injury to a bear is minimal. Typically, bears show no or very little response to the impact of the dart and are left with no obvious visible mark. In order to facilitate easy spotting of darts on the ice or in deeper snow, a 10 – 15 cm long and ~2 cm wide strip of brightly colored flagging tape (C.H. Hanson, Naperville, IL; or Johnson, Montreal, PQ) was tied and wrapped around the distal end of the dart.

We recorded the date, time, location of each observed bear (or group of bears), body condition based on aerial inspection using a subjective standard fat index (Stirling

et al. 2008; a scale from 1 – 5 with 1 being skinny, 3 average and 5 obese), specific markings or characteristics, group size or litter size, the estimated field age class (C0, yearling (C1), 2-year old, subadult, adult) and field sex both with a confidence qualifier (a = high confidence; b = low confidence). Field age class and sex where assessed remotely from the helicopter at altitudes between 3 – 7 m by the same observer. When we encountered mothers and their dependent offspring we distinguished C0s, C1s, and 2-year old bears based on their size and physical features (e.g., blood or fecal/urine stains, scars) or their behavior to a) assign them to a field age class, and b) avoid sampling the same individual more than once. Additional cues such as body size of the individual bear in relation to its surrounding or group members, body shape and proportions, presence of scars, secondary sexual characteristics, observation of urination, and gait were all used to determine sex and age class (SWG 2016, Laidre et al. 2020).

When field age class and sex of a bear were initially assessed with low confidence, additional field notes were taken. For example, young subadult male bears and younger adult females are at times difficult to discern from the air when they are solitary. If we thought that the encountered bear was a young adult female, but were uncertain (confidence classifier "b") then we also noted what this bear could be as alternative – in this case "maybe a young subadult male". We used genetic results to confirm the field-recorded sex and age classes. Lastly, we recorded factors that may have influenced detection probability during sightings, including weather conditions (e.g., cloudy, clear, sun glare), bear activity when first observed, and sea-ice characteristics in general and within the immediate vicinity (~ 30 m) of an individual bear that may impede detection (e.g., sea-ice type: flat, intermediate, rough multi-year ice).

Our work combined data collected during the genetic biopsy sampling sessions from 2014 – 2016, considered the *late period,* and data from the previous capture-mark-recapture study conducted 1998 – 2000, or the *early period*.

Sampling – recovering previously marked bears through harvest

To detect the recovery of previously 'marked' bears (e.g., when bears were marked either during the initial mark-recapture study from 1998 – 2000, or from a previous biopsy-darting field season), we asked hunters to return any ear tags or lip tattoos from their harvested bears. To detect recoveries for bears in which a tattoo may be too faded to see or bears that had been marked through biopsy sampling (which leaves no physical marking), small muscle tissue samples were collected from all bears harvested in MC and surrounding subpopulations such as GB, Lancaster Sound (LS) and Viscount Melville Sound (VM) throughout the duration of the study (May 2014 – June 2016). These samples were stored in 2 ml cryovials (ThermoScientific, Nalgene long-term storage cryogenic tubes) at - 20˚C until sample preparation and analyses.

Sampling - recaptured bears from past population study

Because the initial subpopulation inventory for MC (1998-2000) was conducted using physical capture-mark-recapture methods in which a physical tag or tattoo was used for identification, we had no genetic database for these bears. In order to identify recaptures of bears during our 2014-2016 study that were originally marked during the 1998-2000 study, we examined captures and recaptures from the 1998 – 2000 population inventory, removed bears that we knew were dead (e.g., through a recovered ear tag or tattoo by harvest) and selected the remaining individuals that could be still alive (≤ 34 years of age) in 2014 for genetic analyses. Samples (ear plugs from punching a hole through the pinna so that unique identification ear tags can be applied) of captured and re-captured bears from the initial study had been stored in cryovials at - 20˚C until preparation for genetic analyses.

Sample preparations

We used the same method to prepare all field and laboratory tissues or biopsy samples. A lentil-size piece of skin (~1 – 1.5 mm thick) or tissue was cut from either the biopsy

sample, the ear plug, or the muscle tissue with a new scalpel blade (# 20) and transferred onto a shipping card (Avery, 70 x 35 mm) and attached with scotch tape. Each sample card was labelled with the unique bear identification number and placed into a coin envelope (57 x 89 mm) and left to dry at room temperature for up to 3 days. The dried specimens where then sent to Wildlife Genetics International Inc. (Nelson, British Columbia) for individual genotyping and sex determination.

Genetic analysis

DNA was extracted from tissue with QIAGEN DNeasy Blood and Tissue Kits (Qiagen, Inc.). The tissue samples were genotyped at eight previously published dinucleotide microsatellite loci (REN145P07, CXX20, MU50, G10B, G10P, G10X, MU59, G10H; Paetkau and Strobeck 1994, Paetkau et al. 1995, 1998; Taberlet et al. 1997, Breen et al., 2001, Ostrander et al. 1993). Analysis of individual identity followed a 3-phase protocol previously validated for bears and described elsewhere (Paetkau 2003; Kendall et al. 2009).

First, to select markers for the analysis of individual identity, we used allele frequency data from approximately 1700 polar bears for which complete 20-locus genotypes existed before the genetic mark-recapture study began (GN, unpublished data). We ranked the 20 microsatellite markers in the dataset by expected heterozygosity. The eight most variable markers that could be analyzed together in a single sequencer lane were selected for use. These surpassed the required standard for marker variability (Paetkau 2003). In addition to the eight microsatellite markers, we analyzed sex on every sample, using a *ZFX*/*ZFY* marker. We searched the dataset for genotype matches that seemed unlikely based on our field data. In each case, three extra markers were added to the genotypes to lower the probability of chance matches between individuals. The extra loci confirmed all of these matches. Once the genotyping and error-checking was complete, we defined an individual for each unique eight-locus genotype.

Abundance

We estimated abundance using a closed-population mark-recapture model (Otis et al. 1978) in a Bayesian framework (Kéry and Schaub 2011) for independent animals (>2 years old) encountered during 2 primary sampling periods which occurred during the spring field seasons for the years 1998 – 2000 (early sampling period) and 2014 – 2016 (late sampling period). We used annual time-steps referenced to the springtime field seasons, resulting in three capture occasions within the early and late period (1998, 1999, 2000 and 2014, 2015, 2016, respectively). The model allowed for capture probability to vary by sex but was held constant across capture occasions within each primary sampling period. We fit separate models for the early and late periods. Abundance estimates for the two sampling periods were derived separately without any shared parameters. Furthermore, we split each abundance estimate by sex to obtain separate estimates of detection and abundance by sex. We make no assumptions about the change in population between the periods, nor do we assume equal capture probability. We fit a total of four separate closed-population models, one for each sex and time period.

The model assumed that the MC subpopulation was a geographically and demographically closed population within each three-year period of data collection. Therefore, the model assumes there is no movement in or out of the study area and no birth or death. Polar bear survival is generally higher for adults (Amstrup and Durner 1995), which should reduce bias associated with violation of the demographic closure assumption. However, lower survival rates for younger polar bears (Regehr et al. 2010) and recruitment of juvenile animals into the study population could be expected to cause positive bias in estimates of abundance (Pollock et al. 1990). Furthermore, potential violations of the geographic closure assumption due movement of animals in and out of the study area mean that the estimate of abundance does not represent the number of animals within the study area at any given time, but rather represent the total number of bears available for capture across the three-year period (i.e., the "superpopulation"; Kendall et al. 1997). We estimated abundance using closed models,

despite potential biases, because the data were sparse and insufficient to parameterize an open population model. Moreover, because the survey area changed annually with changing weather and effort, common estimators such as the Horvitz-Thompson for N from each year's sampling were not appropriate because the estimator's results would conflate changing survey area with population size in unknown ways. We chose to estimate abundance using data from the 2 primary sampling periods rather than from all data from 1998 – 2016 because the 13-year gap between the 1998 – 2000 and the 2014 – 2016 surveys will overstate permanent emigration causing survival (φ) to go down and capture probability (p) to go up. Therefore, the estimate of p will be too high underestimating population size. While not ideal, using the closed models provides the best estimate with the available data.

To fit the closed-population capture-recapture models, we performed Markov chain Monte Carlo (MCMC) analysis using JAGS (Plummer 2003) through the R package *R2jags*. Each model was run for 20,000 iterations with the initial 2,000 discarded for burn-in. We used diffuse normal prior distributions on a logit link for all parameters. We checked for model convergence using $\hat{R}$ statistics and by examining MCMC chain plots (Gelman et al. 2013).

Survival

We estimated annual, apparent survival for independent bears >2 years old using all encounters from 1998 – 2016 by grouping our data into the 2 capture-mark-recapture sampling periods (1998 – 2000 and 2014 – 2016) and using available dead-recoveries from 1998 – 2016. Data were sparse with respect to live-recaptures and dead-recoveries and there was a 13-year gap (i.e., 2001 – 2013) in sampling between the capture-mark recapture studies. The gap period was characterized by a very low harvest rate resulting in minimal dead-recovery opportunities (e.g., 3 bears per year as harvest). Additionally, p is essentially equal to zero because the closed population model does not allow any recaptures during the gap period between sampling efforts. Because we did not have radiotelemetry data, and very few or no data on recoveries of

previously marked animals, we could not estimate fidelity (*F;* the probability that an animal does not permanently emigrate from the sampling area and remains available for live observation in the future*)* to our study area. Therefore, estimates are not true survival but rather apparent survival, which is the probability of a bear remaining alive and available for capture, given it was alive at the previous sampling time. Bears that permanently leave the study area and remain alive, but are unavailable for recapture cannot be separated from mortality when estimating apparent survival. Therefore, apparent survival will likely be lower than true survival due to emigration.

We used a Cormack-Jolly-Seber (CJS) model (Cormack 1964, Jolly 1965, Seber 1965) and considered apparent survival (ϕ) varying by sex (i.e., male or female) or remaining constant, and capture probability (*p*) varying by sex, study period (i.e., early versus late period), or remaining constant. The commonly used Burnham model was not applicable for estimating MC survival rates because the harvest rate changed, yet there are insufficient data to estimate multiple recovery probabilities. Therefore, the survival from the Burnham model would be unreliable and would reduce to a CJS model in the absence of additional data. We fit six models representing all combinations of ϕ and *p* in Program Mark (White and Burnham 1999) through the *Rmark* (Laake 2013) package in R. We used AIC to rank models with the lowest AIC value suggesting the strongest support (Burnham and Anderson 2002). All models differed by a single, nested parameter, therefore we evaluated whether that parameter resulted in a model improvement based on ΔAIC and parameter estimates.

Reproduction

We calculated reproductive indices for MC polar bears using data for the early and late study periods by using reproductive metrics that have been identified as important for monitoring (Vongraven et al. 2012). The annual observations of dependent young during the sampling periods were few and variable which limited our ability to estimate many reproductive indices. We calculated the mean number of C0 and C1 per adult female (AF) by year and study period (± SE) using the observed sampling data. Adult

females were a) bears identified genetically as females, and b) bears classified in the field with the age class "adult" with high confidence. We also calculated mean C0 and C1 litter size by study period, although the data were too sparse to evaluate patterns in litter size as function of biological, environmental and temporal factors.

Population growth

We estimated population growth rate in two ways to understand differences between observed changes in abundance and demographic rates. First, we estimated the empirical growth rate as the ratio of the late period abundance over the early period abundance for males and females. We then computed an average annual growth rate (λ) by taking the 17th root of the growth rate to account for the length of time between the two study periods. We estimated separate growth rates for males and females because the abundance estimates differed by sex. Second, we computed an asymptotic growth rate from a 4-stage matrix model based on the demographic rates estimated in this study (Mills 2012). For rates that were not available from our study, we used values from Taylor et al. (2006). The population matrix was defined as:

$$L = \begin{bmatrix} 0.00 & 0.00 & 0.00 & 0.17 \\ 0.62 & 0.00 & 0.00 & 0.00 \\ 0.00 & 0.88 & 0.00 & 0.00 \\ 0.00 & 0.00 & 0.88 & 0.88 \end{bmatrix},$$

where C0 survival probability = 0.62 (Taylor et al. 2006), 2+ year old survival probability = 0.88 (present study), and recruitment = 0.39 C0 per AF (present study). The value of 0.17 in the upper right of the matrix is the product of AF survival (0.88), recruitment (0.39), and sex ratio at birth (0.5). We solved for asymptotic growth rate by calculating the dominant eigenvalue of the matrix (L) assuming a stable stage distribution.

Body condition

We compiled body condition score (BCS) data for the early and late study periods. Bears were assigned a BCS on a scale of 1 – 5 with 1 being skinny and 5 being obese

(Stirling et al. 2008) through physical handling and capture (early period; 1998 – 2000) or aerial observation during biopsy sampling (late period; 2014 – 2016) from April to June. Sex, age, and reproductive classes were assigned during physical handing during the early period and ages were determined based on previous capture history, known birth year, or from tooth analysis (Calvert and Ramsay 1998). During the late sampling period, classification was done during biopsy sampling while flying approximately 3 – 7 m above the ground with sex verified by subsequent genetic analysis (SWG 2016, Laidre et al. 2020). Observers who participated in classifying age class and sex during biopsy sampling had either participated in both study periods, or were experienced in physical capture-mark-recapture studies.

The BCS raw scores were binned into 3 classes: 'poor' (1 – 2), 'average' (3), and 'good' (4 – 5) in order to follow recommended monitoring schemes (Stirling et al. 2008, Vongraven et al. 2012) and facilitate comparison between previous studies (SWG 2016, Laidre et al. 2020). Like previous studies, we did not include dependent offspring in the BCS analyses because their body condition is dependent on maternal condition (SWG 2016) and we excluded within-year observations of the same individual.

We modeled BCS using ordinal logistic regression and included *period* as an indicator of sampling period (early = 1998 – 2000 or late = 2014 – 2016). Reproductive status, age, and sex were combined into one 4-level categorical variable, *reproclass* (ADM = adult male, ADFI = independent adult female, ADFWO = adult female with offspring, and SUB = subadults of both genders) and Julian date of sampling (*jul_cap_day)* was included as a continuous covariate to reflect the amount of time a bear had on their preferred sea-ice hunting platform before being sampled in year t. The sampling seasons (April-June) in this study also coincided with the annual seal pupping period, which is known to be prime feeding period for bears (Pilfold et al. 2012, Reimer et al. 2019). Thus, we predicted that increased time on the ice prior to sampling would be associated with higher BCS. The number of days between sea-ice retreat and advance (*icetm1$_{t-1}$)* was included to evaluate the hypothesis that interannual variation in BCS was related to sea-ice availability in the previous year. We selected a global model

that reflected biological and environmental variables we hypothesized, or that have been shown in other studies, to have effects on BCS (Rode et al. 2012, SWG 2016, Laidre et al. 2020). Finally, given our interest in evaluating whether different reproductive classes and age classes had varying BCS based on the amount of time they spent on the sea ice during the months immediately prior to observation (*jul_cap_day)* and whether this relationship was different between our two sampling periods (*period)*, we included a 3-way interaction for *reproclass, jul_cap_day,* and *period.* Once the global model was selected, we performed a backwards and forwards model comparison (stepAIC; Package *MASS* with AICc criteria functionality added) to obtain the most supported model which included main effects for *epoch*, *reproclass, jul_cap_day, icetm*, and interactions for *epoch* and *icetm* and *epoch* and *reproclass* (Table 1). On the final model, we performed ordinal regression-specific goodness of fit test (Pulkstenis-Robinson test; $p > 0.1$; Fagerland and Hosmer 2017). Covariates were considered significant at $p < 0.05$ and predicted probabilities were calculated for significant covariates.

We hypothesized that BCS would be correlated with ice conditions based on previous studies suggesting decreased body condition with increased ice-free days and lower sea-ice concentrations (Rode et al. 2012, Laidre et al. 2020). Thus, we calculated the number of days between the summer sea-ice retreat and fall sea-ice advance in *sampling year t - 1* (Environment and Climate Change Canada 2018). Sea-ice retreat and advance in MC were defined as the point in which the sea-ice concentration for a given year fell below, or exceeded, respectively, the halfway point between the averaged 1979 – 2016 March sea-ice concentration (representative of annual sea-ice maximum) and the average September sea-ice concentration (annual minimum). For MC, that transition threshold sea-ice concentration was 59%.

Seal observations

There is little recent information about seal abundance across the Canadian Arctic. We therefore recorded all seal locations encountered along our flight paths while searching

for polar bears. These data were collected only during the 2014 field operations due to logistical challenges and can potentially provide baseline information on relative abundance for help in assessing ecosystem productivity.

5. RESULTS

General overview

During research operations 2014 – 2016, we spent between 72.5 and 97.5 hours flying each season in search of polar bears across the sea ice, with an average distance flown per year of 12,300 km (Table 2, Figure 3). The highest bear encounter rate occurred during 2014. Each field season was conducted generally between May and early to mid-June. Due to poor weather conditions a large portion of M'Clintock Channel proper could not be surveyed (Figure 3) and a reconnaissance flight via plane into the Channel in 2016 found few signs of bears (e.g., only one bear was observed).

The MC study area is vast and bears occur at low densities (Hamilton and Derocher 2018). In addition, polar bears were not distributed evenly across the study area (Figure 4). Most bears across all study periods were encountered from Franklin Strait southward to Victoria Strait and Jenny Lind Island. The sea ice in Queen Maud Gulf, Dease Strait and Coronation Gulf did not exhibit many signs of polar bears, at least not during the time of the survey (i.e., April to early June). The low coverage of M'Clintock Channel proper by Twin Otter did not suggest high bear density, however, it remains uncertain if sea ice is used by bears with higher intensity north of Gateshead Island during early spring at times when we were not present.

Due to logistical constraints we were only able to record seal observations during 2014. During that field season, work continued into June when ambient temperatures were sufficiently high for seals to haul out on the sea ice and bask in the sun. On our search flights we observed 2,236 seals distributed across the area where most bears were encountered (Figure 5).

Samples examined

For the entire study period 1998 – 2016 we analyzed a total of 953 (319 biopsy, 102 physical capture, and 532 harvest) tissue samples for genetic identification. We identified 244 individual bears through the biopsy sampling activities 2014 – 2016. All 102 tissue samples from physical captures during the early study period (1998 – 2000) were successfully analyzed and of the 532 harvest samples, 99% produced reliable genetic results. Overall, the success rate of extracting DNA material from all study samples (research and harvest) was 97.8%.

Dead recoveries of marked bears through the harvest resulted in 22 bears being identified, 7 of these were recovered in subpopulations outside MC (4 in LS, 1 in Northern Beaufort Sea, 1 in Foxe Basin, and 1 in Viscount Melville Sound). Sixty-eight percent of all recoveries occurred in MC, and no recoveries were made in GB. We live-recaptured 6 bears marked in 1998 – 2000 and 33 bears 2015-2016. As a note of interest, 7 bears that were originally marked via biopsy in MC 2014 – 2016 were live re-captured in Gulf of Boothia during the GB population study conducted 2015 – 2017 (Dyck et al. 2020).

Population demographic information

Abundance – Estimated total (males and females combined) abundance was 325 (95% Credible Interval (CRI) = 220 – 484) for the period 1998 – 2000, and 716 (95% CRI = 545 – 955) for the period 2014 – 2016 (Figure 6). The CRIs around the total abundance do not overlap across the two study periods providing substantial evidence for an increase. Estimated recapture probability was 0.13 in both periods with higher precision in the later period (95% CRI = 0.03 – 0.19 (early), 95% CRI = 0.10 – 0.18 (late)).

We estimated sex-specific abundance to obtain additional insight into population dynamics. Between the two study periods, the female segment of the MC subpopulation increased from 219 (95% CRI = 124 – 405) to 327 bears (95% CRI = 230 – 487, Figure

7). The males showed a larger increase from 134 (95% CRI = 74 – 256) to 360 bears (95% CRI = 244 – 550, Figure 6).

Survival – We estimated apparent survival for independent bears aged 2 and older from 1998-2016. The best-supported model included constant survival and detection probabilities across time and sex (Table 3). All other models showed no support given that the AIC values increased when a single parameter was added. Apparent survival from the top model was 0.88 (SE = 0.02) and detection probability was 0.17 (SE = 0.02). Cub-of-the-year survival was not possible to estimate because they were not sampled due to their small physical size (see Methods above).

Reproduction and Recruitment – During the 1998 – 2000 mark-recapture sampling efforts, 23 family groups (5 with single C0, 7 with 2 C0, 3 with single C1, 8 with 2 C1) were sampled, representing a total of 38 (19 C0 and 19 C1) dependent offspring. Through the genetic biopsy sampling study 2014 – 2016, we sampled 27 family groups with 46 C0 (8 with single C0, 19 with 2 C0), and 18 family groups with 29 C1 (7 with single C1, and 11 with twins; Table 4). For the 1998 – 2000 period, C0 and C1 mean litter sizes were 1.58 (SE = 0.14) and 1.71 (SE = 0.14), respectively. Calculated mean C0 and C1 litter sizes for 2014 – 2016 were 1.70 (SE = 0.09) and 1.61 (SE = 0.11), respectively. We calculated the number of C0 and C1 per AF across the two study periods (Table 4). The mean C0 and C1 per AF from 1998 – 2000 was 0.38 (SE = 0.02) and 0.39 (SE = 0.10), respectively. The mean C0 and C1 per AF for 2014 – 2016 was 0.43 (SE = 0.10) and 0.28 (SE = 0.06), respectively. The overall mean C0 recruitment was 0.39 (SE = 0.11).

Population Growth – Based on the estimated increases for the female and male proportions of the subpopulation between the two time periods, the average annual growth rate (λ) was 1.02 for females and 1.05 for males. We built a 4-stage matrix population model to describe the polar bear subpopulation with C0, C1, subadult, and adult as life stages. We included vital rates estimated above. For parameters not included in this analysis, we used estimates from Taylor et al. (2006). Specifically, adult

survival and C0 per AF were estimated in this analysis and therefore those estimates were used. Cub-of-the-year survival was taken from Taylor et al. (2006) because we were not able to estimate this value reliably using this study's methodology. Using our calculated recruitment value of 0.39 for C0 per AF, a survival rate of bears older than 2 of 0.88, and C0 survival of 0.62 the matrix model results suggest a declining subpopulation ($\lambda = 0.97$). This represents a discrepancy between observed demographic rates and calculated abundance. Our estimate of apparent survival is biased low compared to true survival due to unknown emigration. Furthermore, unmodeled heterogeneity in recapture probability is a well-known source of bias in estimates of survival from CJS-type models (Devineau et al. 2006). To explore this, we calculated what level of adult survival would be needed to achieve the estimate of female $\lambda = 1.02$ based on changes in abundance across study periods. The new adult survival probability of 0.93 provides a population growth of $\lambda = 1.02$ (Figure 8). That survival value is consistent with survival in the absence of harvest from Taylor et al. (2006). Flat population growth ($\lambda = 1.0$) occurs when survival is 0.91.

Body condition

We analyzed a total of 380 BCSs from the two study periods (Table 5). The most supported model included *period*, *reproclass*, *jul_cap_day*, and *icetm$_{t-1}$* and interactions *period:reproclass* and *period:icetm$_{t-1}$* (Table 1). Body condition of bears improved for all reproductive classes from the early period to the late (P$_{Poor\ early\ adult\ females\ and\ subadults}$ = 0.50 vs P$_{Poor\ late\ adult\ females\ and\ subadults}$ = 0.14), except for adult males (X^2 = 10.81, P = 0.01; P$_{Poor\ early\ ADM}$ = 0.15 vs P$_{Poor\ late\ ADM}$ = 0.17; Figure 9). Later sampling in the year was associated with better body condition (X^2 = 9.38, P < 0.01; Figure 10). As the number of days between sea-ice retreat and advance increased (*icetm*), the predicted probability of a bear being in poorer condition increased but this was more pronounced in the early period (X^2 = 3.86, P < 0.05; P$_{Poor\ early\ icetm136}$ = 0.58 vs P$_{Poor\ late\ icetm136}$ = 0.19).

6. DISCUSSION

General

This study reports population abundance, survival, population growth, reproductive indices and body condition using the data from surveys conducted in the MC polar bear subpopulation between 2014 – 2016 and 1998 – 2000 along with dead-recoveries of harvested bears from 1998 – 2016. After more than 15 years of a reduced harvest and a moratorium that were implemented because of overharvest (Taylor et al. 2006), the subpopulation has recovered to the determined mid-1990s level (Aars et al. 2006; Taylor et al. 2006). Without the support and participation of community co-management partners from Taloyoak, Cambridge Bay and Gjoa Haven, this subpopulation would not have recovered over the past 15 years, hence this report and results should be welcomed as good news.

The recovery of this subpopulation may have been aided not only by concerted conservation actions by communities and management authorities, but also, counterintuitively, by climate-induced sea ice changes occurring in this region. While some polar bear subpopulations are showing negative impacts from climate change, (Regehr et al. 2007, Regehr et al. 2010, Lunn et al. 2016, Obbard et al. 2016, 2018), the short-term narrative may be different in terms of MC. Historically, the study area, and in particular M'Clintock Channel proper, had an abundance of multi-year sea-ice that remained mostly throughout the year (Schweinsburg et al. 1981, Barber and Iacozza 2004; Howell et al. 2008, 2009; Environment and Climate Change Canada 2018, 2019; Sou and Flato 2009). However, recent evidence suggests that the open-water extent in the western Arctic (including the study area) has been increasing between 1968 and 2005 (Stewart et al. 2007) and sea-ice cover during the summer has declined (Stern and Laidre 2016, Rothrock et al. 1999, Comiso 2002). In addition, some heavy multi-year sea-ice has been already replaced by annual ice (Barber and Iacozza 2004, Environment and Climate Change Canada 2018, 2019, Marz 2010, Perovich et al. 2018, Richter-Menge et al. 2018) and an even greater shift is expected (Sou and Flato

2009; Hamilton et al. 2014). The observed changes from multi-year to annual sea ice result in declining sea-ice thickness. Younger and thinner sea ice is more mobile and susceptible to mechanical wind forcing and is also more vulnerable to complete melting in the summer which contributes to the observed decrease in summer sea-ice extent. (Richter-Menge 2018, Perovich et al. 2018). This reduction in sea ice results in the absorption of more heat by the upper ocean (Richter-Menge 2018). While sea-ice loss overall is considered very detrimental to the persistence of polar bears, in the short term, it may have beneficial effects since many of the observed sea-ice changes have been associated with greater Arctic marine productivity (Derocher et al. 2004, Häder et al. 2014, Frey et al. 2018). This increased productivity and dynamic ice may have played a role in the observed improvement in body condition of bears in MC between the late 90s and the recent study period (Derocher et al. 2004).

Currently, it is uncertain when continued reductions in sea-ice availability may cross a threshold such that limited time to hunt seals on the ice begins to have a negative effect on the MC subpopulation which have been documented for more southerly polar bear subpopulations (Bromaghin et al. 2015, Lunn et al. 2016). Over the long term, progressive loss of Arctic sea ice is a primary threat to the species (Atwood et al. 2016, Regehr et al. 2016).

Abundance

Polar bear abundance has increased across the two study periods with the male segment of the subpopulation increasing more rapidly than females. This may reflect recovery of the male segment after depletion due to general harvest overexploitation, which, when coupled with a sex-selective harvest (2 males for each female in Nunavut) could have been further exacerbated (Taylor et al. 2005, McLoughlin et al. 2005, Taylor et al. 2008a). Male abundance was almost half of female abundance in the early period but grew to be equal to or slightly larger than female abundance during the late time period. Increases in male abundance over females reflect a higher growth rate for males

during the study period, potentially from reductions in harvest pressure and/or immigration of males into MC from other subpopulation units.

We estimated abundance of MC polar bears using closed-population mark-recapture models. Yet, our sampling occurred across 3-year periods suggesting that the assumptions of demographic and geographic closure are almost certainly violated. The limited numbers of bears detected and the sparse recaptures within a year precluded fitting models that can potentially reduce bias in parameter estimates, such as the 'robust design' or 'spatially-explicit' capture-recapture models. Moreover, 'open' population models that include an abundance estimate (e.g. forms of the Jolly-Seber model) require more years of data with more recaptures than our data allowed. Thus, despite evidence from the few harvest-recovered marked bears that the subpopulation is not, in fact, 'closed' to emigration, our limitations with available data prevented these more highly-parameterized models from being fitted. These more complex models better reflect biological and ecological systems. However, they are 'data-hungry' (Bromaghin et al. 2015, Lunn et al. 2016, Regehr et al. 2018) meaning if less data are available due to low densities or other constraints, then these approaches, such as multistate capture-recapture models and integrated population models, are generally not options to estimate abundance, despite the potential advantages of these models in estimating demographic parameters of interest and reducing bias.

The basis for capture-mark-recapture studies rely on the marks that are initially put out into the population during the study's first field season and the subsequent recovery of those marks (recaptures) through harvest recovery or re-sampling (Caughley et al. 1977, Amstrup et al. 2010). This approach has worked relatively well for larger, denser subpopulations that allow for relatively large sample sizes (Regehr et al. 2007, Peacock et al. 2013). However, when populations are small, occur at low densities, have low harvest levels, and/or are located in remote regions that are difficult to access, recapturing or recovering marked individuals from the population is difficult (e.g. M'Clintock Channel). This process is made more complicated when it is unknown if

the subpopulation is open or closed (Kendall 1999), which can only be determined through radiotelemetry to examine long-term movement patterns of individual bears.

The consequences of assumptions violations in closed-population models are well known (Otis et al. 1978) and affect both the actual abundance and what that abundance geographically represents. A lack of demographic closure results in underestimated detection probability (for example, bears that die are no longer available for detection). The underestimated detection probability leads to an overestimated abundance for any given year. Despite these caveats, for this study, the total number of bears in the study area available for detection across the three years appeared to have been unbiased (i.e., a similar number of bears frequented the study area while the study was conducted) and our estimated abundance for MC using a closed population model for the early period with 325 bears was similar to Taylor et al. (2006; 284 bears). A lack of geographic closure blurs the boundaries of the study area. If bears move in and out of the sampled area, then the estimated abundance refers to an area larger than the area sampled (e.g., estimating the "superpopulation"). A superpopulation is defined as all the animals with a chance (non-negligible probability) of occurring within the MC management boundary, regardless of where the animals were located at any given sampling occasion (Schwarz and Arnason 1996). Thus, estimates of superpopulation size in year t likely reflect some animals that were temporary emigrants in year t. We were not able to estimate temporary emigration directly from the sampling area (Cooch and White 2019) because our sample sizes were not sufficiently large to do so, and there are no recent radiotelemetry data to provide location and movement data. However, recoveries of previously marked bears in other subpopulations through the harvest sampling program indicate that movement into and out of MC is occurring.

Lastly, the fact that we were not able to survey the entire study area – namely the portions of M'Clintock Channel proper – contributes to the uncertainty surrounding our abundance estimate. Although we did not detect many signs of bear activity while conducting our reconnaissance flight, it is unknown how many bears (e.g., bears that may temporarily move into this area from the neighboring LS subpopulation, or resident

MC bears) may utilize this section of the study area throughout the timing of our surveys since we were able to conduct only one limited survey flight due to poor weather conditions. Because we have no information on how many bears could have been in this area, we are unable to determine whether or not our abundance estimate would be affected negatively or positively.

Taken together with the effects of demographic and geographic closure violation, the estimate of abundance is almost certainly larger than the actual number of animals within the MC subpopulation boundary at any given time. This should be taken into consideration when using these findings to inform management decisions. For example, if capture-recapture analyses are performed independently for multiple adjacent subpopulations that experience exchange of animals, the sum of the estimates of superpopulation size will be larger than the actual total number of bears in the subpopulations (i.e., there will be "double counting" of some bears). This could lead to cumulative TAH levels that result in removal of a larger proportion of polar bears each year than was intended based on the TAH levels for the individual subpopulations.

Survival

We estimated apparent survival of polar bears from 1998 – 2016. The resulting survival probability (0.88) is lower than biological survival estimated from other studies (Taylor et al. 2006). This is likely due to a combination of factors such as emigration away from the study area, which will cause apparent survival to be lower than biological survival (Lebreton et al. 1992). Further, capture-recapture data were collected intensively for 3 years in 2 distinct study periods separated by 13 years. Therefore, few observations of bears exist between 2001 and 2013. The missing sampling years greatly reduce the power to estimate survival or estimate variation in survival across time, sex, or age classes.

Survival is known to differ among sex and age classes; however, none of the models including differences in survival by sex were supported by the data (Table 3). In addition we were not able to test for differences in survival by age class. It is very likely that by pooling age classes and sexes the overall mean natural survival rate was also biased low (SWG 2016). Furthermore, unmodeled heterogeneity in recapture probabilities can introduce substantial negative bias into estimates of survival (Regehr et al. 2009). Unfortunately, with live capture-recapture data, limited harvest data, and no contemporary information on animal movements (e.g., from satellite radiocollars), there are few options to estimate biological survival. Our data were too sparse for joint live/dead models and capture probability was too low for known fate models. These challenges were also recognized by Taylor et al. (2006).

Reproduction

Our field observations of C0 and C1 litter sizes revealed inter-annual variation with mean values similar to other subpopulations within the Canadian Arctic Archipelago (Table 4; Durner et al. 2009; dated estimates – Lancaster Sound and Norwegian Bay: Taylor et al. (2008b); Gulf of Boothia: Dyck et al. (2020); Kane Basin: SWG (2016)), although our sample sizes were small. We estimated reproduction based on counts of C0 and C1 observed with adult females. Reproduction rates were very similar across our study periods and were within the ranges estimated by Taylor et al. (2006).

It is difficult to draw definite conclusions about whether all reproductive parameters differ between the two studied periods because of limited data. Estimating the number of C1 per AF is considered a key reproductive parameter (Vongraven et al. 2012, Regehr et al. 2015) because it integrates cub production and cub survival. The C1 per AF of the recent period of 0.28 was lower than during the earlier period (0.39 in this study). Whether this decline is real or represents an artifact of sample size is unknown. Nevertheless, our observed number of C1 per AF appears to be sufficient to maintain a viable subpopulation, provided that survival is within the normal range for healthy subpopulations (Regehr et al. 2015). Continued monitoring of MC to obtain

improved estimates of survival and reproductive rates is prudent to determine whether this subpopulation remains healthy.

Population Growth

We estimated population growth rate both empirically based on changes in abundance and using a matrix population model to compare observed changes in abundance to theoretical population growth rates arising from the vital rates. For the purpose of estimating an asymptotic population growth rate based on the vital rates, we used a simplified matrix projection model that does not accurately represent the multiyear reproductive cycle of polar bears (Regehr et al. 2017). Although such a simplified model would not be suitable for stochastic projections (Taylor et al. 1987), we believe it was sufficient for a general assessment of consistency between empirical and matrix-based estimates of population growth rate. The changes in abundance suggest that growth was approximately 2% per year for females and 5% per year for males for the period 1998 – 2016. Conversely, the estimated vital rates suggested a population growth rate of -3% per year (i.e., that a subpopulation with these vital rates would decline by 3% per year). Therefore, the demographic rates and abundance estimates are not internally consistent. The most likely explanation is negative bias in estimates of true survival for adult females. We estimated apparent survival rather than biological survival. In addition, we pooled independent bears (subadults and adults) in order to obtain survival rates during this study. Adult polar bear survival rates are higher than subadults (Regehr et al. 2007, 2010), and pooling them would bias the result negatively. Finally, there was likely unmodeled heterogeneity in recapture probabilities that introduced additional negative bias into survival estimates (Regehr et al. 2009). If we replace estimated survival from Taylor et al. (2006), the model shows growth similar to our observed female population growth.

The discrepancies in abundance and survival provide insight into the utility of each data type. Abundance data appear to be providing stronger inference into population dynamics of this polar bear population. The survival information contains too

much bias relative to biological survival to be meaningful for polar bear management. If capture-recapture data were collected over a longer time period, then survival may become a more useful parameter. Other data-based and simulation studies for polar bears have documented that, although mean percent relative bias can be higher for estimates of abundance than survival, the resulting challenges to demographic inference are actually larger for bias in adult female survival because it is a primary driver of population growth for long-lived species like polar bears (Eberhardt 1990).

One question that remains is the amount of potential bias in estimates of abundance and survival for the two study periods. The abundance estimates use data across 3-year periods, therefore some bears included in the estimate died before the end of the period. Our estimated apparent survival rate (0.88) would suggest a declining subpopulation, however, to achieve abundance estimates derived for this study, survival rates would need to be 0.92, which is reasonable for polar bears. Therefore, the total mortality during a 3-year abundance estimate is expected to be about $0.15 = (1 - 0.92^2)$. An increase in abundance is also supported by other lines of evidence. First, the MC subpopulation was managed for recovery and had a restricted harvest for 15 years that was designed to nurture population increase (Taylor et al. 2006) and likely led to a recovery of the depleted male proportion. Second, body condition of bears improved between the two studies, which could be an indication that the habitat improved as multi-year ice decreased over the past 15 years resulting in increased productivity, enhancing seal habitat which may be reflected in a larger carrying capacity.

Similar to estimates of abundance and survival, potentially high and variable levels of immigration and emigration across subpopulation boundaries can directly affect estimation and interpretation of population growth rate (Peňaloza et al. 2014). In some other subpopulation studies, radiotelemetry data have been critical to resolving these issues (Regehr et al. 2018). For regions where radiotelemetry is not available, we recommend that the best way to reconcile these interpretation challenges and provide accurate information to inform management is to perform a meta-analysis of the

capture-recapture and harvest recovery data for all subpopulations within the region that are known to exhibit substantial levels of exchange (e.g., GB, MC, and LS).

Body condition

Polar bears observed during the recent study period were in better body condition compared to the late 1990s with the exception of adult males, which is not unexpected given that during April – June, males are often intent on searching for mates and breeding rather than only feeding (Stirling et al. 2016). Further, rapid changes in sea-ice characteristics in the last 15 years from multi-year to more annual ice, which is less thick and prone to experiencing leads and cracks, may facilitate increased opportunities for hunting during the annual seal pupping period that occurs in mid-April. These conditions potentially account for our finding that body condition improves later in the year (Stirling and Archibald 1977, Pilfold et al. 2014, Reimer et al. 2019).

It is less likely that sampling method is responsible for changes in the observed BCS between time periods. Raw BCS scores were binned into 3 general categories to account for any potential small biases in observer classifications (Laidre et al. 2020). Furthermore, there have been varied results in other studies in which earlier time period BCS classification was done by physical handling and compared to later time period BCS classifications based on aerial observations, suggesting that there is not an inherent bias in aerial observation versus physical handling body condition classification (Kane Basin: no change in BCS, Baffin Bay: decrease in BCS, Gulf of Boothia: increase in BCS; SWG 2016, Dyck et al. 2020, Laidre et al. 2020). Many of the same observers and biologists that participated in the early physical capture and handling studies also participated in the aerial observation studies which supports reliability and consistency between study methods for BCS. The general application of our index during physical handling has been shown to be a reliable indicator in the assessment of body condition (Stirling et al. 2008). Moreover, there is the potential to assess the lipid content of the extracted adipose tissue from the biopsy darts (Pagano et al. 2014, McKinney et al. 2014) which could be used to verify the aerial condition assessments.

The replacement of multi-year with annual ice in our study area may have also provided improved seal habitat and contributed to an increase in the polar bear prey base. To our knowledge, there are no quantitative data about seal abundance from our study area available; however, during our investigations and observations, it became apparent that ringed and bearded seals appear relatively abundant and demonstrated a preference for annual sea ice (GN, unpublished data reports). These longer-term changes in habitat may be in part responsible for the fact that we found BCS of bears sampled in the late period to be relatively unaffected by the number of days between sea-ice retreat and advance, which wasn't the case in the early period, suggesting that over time, the ecosystem has become more productive. It is important to note that our study periods encompass a relatively short period of time, with 3 years in the early period and 3 years for the late period. Inter-annual variation could significantly affect BCS for such a limited temporal window. Thus, we caution over-interpretation beyond general trends for BCS. It is likely that the potential enhanced productivity brought on by changes in sea-ice dynamics may be a short-lived advantage to polar bears if access to their prey is reduced by a declining sea-ice hunting platform, though the time scale of these events remains unknown.

7. MANAGEMENT IMPLICATIONS

The need for continued monitoring

In the past 20 years, polar bear population studies in Nunavut were generally conducted over a 3-year period, which is a relatively short time considering polar bear life spans. In many studies, survival rates tend to be biased low because of limited study length, low recapture probabilities, unmodeled heterogeneity in recapture probability due in part by prohibitive weather to cover the entire study area, and movements of animals with respect to the sampling area (Taylor et al. 2008b, Regehr et al. 2009, SWG 2016, Dyck et al. 2020). In the case of MC, several of these factors are true, including unknown emigration rates, low density of bears (fewer bears receive marks), and potential heterogeneity in recapture probability resulting in a likely low-biased survival rate.

What we have learned from this process of studying and analyzing MC data is that continued monitoring, in the form of increasing study length or adding an intermittent marking session, would reduce the type of bias we encountered in estimating population parameters like survival and abundance (Peacock et al. 2012).

Further, the MC subpopulation area has experienced drastic sea-ice changes since the 1990s with multi-year sea-ice diminishing and being replaced by annual ice (Stern and Laidre 2016, Environment Canada 2018, 2019). It is currently unknown what importance the little remaining multi-year ice plays for MC polar bears, especially during the summer months (e.g., as feeding platform or summer retreat areas).

The need for improved data

Concomitant to adding intermittent marks or increasing study length, is the need to obtain an understanding of the movement into, and out of, the MC subpopulation boundaries, especially in light of continuing sea-ice changes. The results of this study were affected by the lack of available data to inform even the simplest population models, leading to abundance, survival, and population growth estimates that are known to be biased. Emigration rates are vital to accurately estimating survival.

The delineation of this subpopulation is inferred based on movement of collared bears in adjacent subpopulations from the 1990s, prior to the large-scale ice changes in the region (Taylor et al. 2001). Tag recoveries of captured and harvested bears, and some genetic analyses, indicate that MC likely is a distinct unit, but that has been disputed by local hunters and community members (Taylor et al., 2001; Bethke et al. 1996; Schweinsburg et al. 1982; Campagna et al. 2013; Malenfant et al. 2016; Keith et al. 2005, Dyck and Bohling, in prep.) and the current study provided evidence that bears tagged in the MC region were harvested in adjacent subpopulations (see Results Section – samples examined). With continued reduction in multi-year sea-ice, and sea-ice in general predicted to decline, (Sou and Flato 2009; Hamilton et al. 2014), understanding the behavior of bears and their ecology in MC is critical to maintaining a

healthy population (Vongraven et al. 2012). Very little about the movement patterns and habitat use of MC polar bears is known under the current environmental conditions since there has not been a satellite telemetry study to monitor movements and habitat use. At the direction of community co-partners representing Inuit societal values and concerns over physically handling wildlife, the GN Department of Environment, did not carry out any collaring for telemetry data in MC, despite efforts to garner support. In the future, the GN will have to make decisions on how to continue monitoring polar bears in this subpopulation in order to provide adequate information to decision makers.

Harvest management and considerations

The MC subpopulation represents a unique polar bear management unit in that bears are sparsely distributed (low density) over a large geographic area. This requires adaptive harvest management and considerations. The MC polar bear subpopulation saw a harvest of approximately 32 ± 10 bears (range: 12 – 55) between 1970 and 2001 (roughly 19.5 males and 12.0 females; GN, unpublished data) which was not sustainable over the long-term (Taylor et al. 2006) and led to a moratorium and harvest reduction. Harvest levels in the past were based on vague abundance data with high uncertainty and expert opinion (Aars et al. 2006, Taylor et al. 2006). Our study suggests that MC abundance increased since 2000, although with significant caveats and high uncertainty (e.g., biased survival rates and biased abundance; unknown emigration). Future research and monitoring should seek to understand the role emigration plays in this subpopulation so that estimates of survival can be re-assessed.

Here we provide several considerations to aid in harvest management decisions:

- The mean abundance estimate of 716 bears (95% CRI = 545 – 955) for the period 2014 – 2016 is for independent bears 2 years and older and includes substantial caveats and uncertainty, including the knowledge that this estimate is positively biased. Furthermore, this estimate applies to a group of bears that use

the MC region but may also use other management units (e.g., superpopulation; see Discussion Section).

- Data for this subpopulation are sparse and a quantitative harvest risk assessment using subpopulation-specific estimates of vital rates (Regehr et al. 2018) is not possible with the available data.

- A conservative approach to harvest will reduce the probability of subpopulation declines, especially in light of uncertainty in the available information and the documented changes in the sea ice regime.
  - Attempts to reduce subpopulation abundance without effective monitoring and a coupled research-management system increase the probability of negative biological effects on the subpopulation (e.g., reduction to a small size).

- Although recovery of this subpopulation from previous overexploitation appears successful, it came at a high cost to communities during the recovery period from reduced hunting opportunities and knowledge transfer to new hunters of polar bear hunting practices. To prevent this from recurring, we recommend focusing on the considerations above and additional recommendations below to achieve long-term sustainability and subsistence use of this subpopulation.

Additional specific recommendations for MC

1. Seek support from co-management partners to implement a radiotelemetry study to collect movement data in MC to obtain emigration estimates, resolve boundary issues, collect missing demographic data, and evaluate changes in habitat use and denning in light of the ongoing sea ice changes. Before starting such a study, it would be possible to identify the sample size and

duration required to address information needs so that no more bears are physically captured than necessary;

2.　a) Increase monitoring activities by sampling bears (i.e., introduce more marks into the MC subpopulation) 5 – 7 years post-completion of the field portion of the last study (e.g., in 2023 or 2024) for a 1 year injection of marks until the next comprehensive population study will be conducted (~10 – 15 post-completion of last inventory; 2027 – 2032) to increase the number of marked individuals, recaptures and recapture probability of marked individuals. These factors will assist in determining more realistic survival rates when the next comprehensive study is undertaken (note that a power analysis will likely aid in determining whether additional marks really provide more data, and if this endeavor is cost-effective);

b) Monitor reproductive metrics at the time of mark introduction to assess reproductive performance of MC, and if there are significant changes in reproduction consider whether the timing of the next comprehensive subpopulation assessment should be changed;

3.　Increase population study length to 4 – 5 years to ensure that it covers a full reproductive cycle and reduces potential biases and assumptions that are required during the modeling process;

4.　Consider any TAH recommendation above the current TAH allocation with caution and as an interim harvest level until a) the meta-analysis is performed and/or b) the boundary issue has been resolved which can assist in resolving the caveats of whether MC is a closed or open subpopulation, and to what degree emigration (either temporary or permanent) is affecting vital rates.

8. ACKNOWLEDGEMENTS

9. LITERATURE CITED

Analyse and Strategi. 2011. Marine traffic in the Arctic: a report commissioned by the Norwegian Mapping Authority. Oslo, Norway, 23 pp.

Aars, J., Lunn, N.J., and Derocher, A.E. (eds.). 2006. *Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 20-24 June 2005, Seattle, Washington*. Gland, Switzerland and Cambridge, UK.

Atkinson, S.N., and Ramsay, M.A. 1995. The effect of prolonged fasting on the body composition and reproductive success of female polar bears (*Ursus maritimus*). Functional Ecology 9:559–567.

Amstrup, S.C., McDonald, T.L., and Manly, B.F.J. 2010. Handbook of capture-recapture analyses. Princeton University Press.

Amstrup, S.C. 2003. Polar bear, *Ursus maritimus*. In: G.A. Feldhamer, B.C. Thomson and J.A. Chapman (eds), Wild Mammals of North America: Biology, Management, and Conservation, pp. 587–610. John Hopkins University Press, Baltimore, MD, USA.

Amstrup, S.C., and Durner, G.M. 1995. Survival rates of radio-collared female polar bears and their dependent young. Canadian Journal of Zoology 73:1312-1322.

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

Atwood, T.C., Peacock, E., McKinney, M.A., Lillie, K., Wilson, R., Douglas, D.C., Miller, S., and Terletzky, P. 2016. Rapid environmental change drives increased land use by an Arctic marine predator. *PLoS One* 11: e0155932.

Barber, D.G., and Iacozza, J. 2004. Historical analysis of sea ice conditions in M'Clintock Channel and the Gulf of Boothia, Nunavut: implications for ringed seal and polar bear habitat. Arctic 57:1-14.

Bethke, R., Taylor, M., Amstrup, S., Messier, F. 1996. Population delineation of polar bears using satellite collar data. Ecological Applications 6:311-317.

Breen, M., Jouquand, S., Renier, C., Mellersh, C.S., Hitte, C., Holmes, N.G., Cheron, A., Suter, N., Vignaux, F., Bristow, A.E., Priat, C., McCann, E., Andre, C., Boundy, S., Gitsham, P., Thomas, R., Bridge, W.L., Spriggs, H.F., Ryder, E.J., Curson, A., Sampson, J., Ostrander, E.A., Binns, M.M., and Galibert, F. 2001. Chromosomespecific single-locus FISH probes allow anchorage of an 1800-marker integrated radiation-hybrid/linkage map of the domestic dog genome to all chromosomes. Genome Research, 11:1784-1795.

Bromaghin, J.F. et al. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecological Applications 25:634-651.

Brown, M.W., Kraus, S.D., and Gaskin, D.E. 1991. Reaction of North Atlantic right whales (*Eubalaena glacialis*) to skin biopsy sampling for genetic and pollutant analysis. Report of the International Whaling Commission (Special Issue 13):81–89.

Boulanger, J. et al. 2004. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British Columbia. Canadian Journal of Zoology 82:1267–1277.

Bunnell, F.L. and Tait, D.E.N. 1981. Population dynamics of bears—implications. In C. W. Fowler and T. D. Smith (eds.), Dynamics of large mammal populations, pp. 75–98. John Wiley and Sons, New York.

Burnham, K.P. and Anderson, D.R. 2002. Model selection and multi-model inference: an information theoretic approach. Springer-Verlag, New York, NY, USA.

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

Campagna, L., Van Coeverden de Groot, P.J., Saunders, B., Atkinson, S., Weber, D., Dyck, M.G., Boag, P.T., Lougheed, S.C. 2013. Extensive sampling of polar bears (*Ursus maritimus*) in the Northwest Passage (Canadian Arctic Archipelago) reveals population differentiation across multiple spatial and temporal scales. Ecology and Evolution 3:3152-3165.

Caughley, G. 1977. Analysis of vertebrate populations. Princeton University Press.

Cooch, E.G., and White, G.C. 2019. Program MARK: a gentle Introduction. Colorado State University, Fort Collins, Colorado, USA.

Comiso, J.C. 2002. A rapidly declining perennial sea ice cover in the Arctic. Geophysical Research Letters 29:1956, doi 10.1029/ 2002GL015650.

Cormack, R.M., 1964. Estimates of survival from the sighting of marked animals. Biometrika, 51:429-438.

Department of Environment. 2013. Statutory Report on Wildlife to the Nunavut Legislative Assembly. 75 pp. Available at: http://assembly.nu.ca/library/ GNedocs/2013/001029-e.pdf. Accessed 24 December 2015.

Derocher, A.E., Lunn, N.J., and Stirling, I. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163–176.

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

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

Durner, G. M., Laidre, K.L., and York, G.S. (eds.). 2018. *Polar Bears: Proceedings of the 18th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 7–11 June 2016, Anchorage, Alaska*. Gland, Switzerland and Cambridge, UK: IUCN. xxx + 207 pp.

Dyck, M., Regehr, E.V., and Ware, J.V. 2020. Assessment of abundance for the Gulf of Boothia polar bear subpopulation using genetic mark-recapture. Final Report, Government of Nunavut, Department of Environment, Igloolik, Nunavut. 72 pp.

Eberhardt, L.L. 1990. Survival rates required to sustain bear populations. Journal of Wildlife Management 54:587-590.

Environment and Climate Change Canada. 2018. Sea ice summary McClintock Channel and Gulf of Boothia [internal report]. Edmonton, AB. 11 pp.

Environment and Climate Change Canada. 2019. Canadian Environmental Sustainability Indicators: Sea ice in Canada. Accessed 12 February 2019. Available at: www.canada.ca/en/environment-climate-change/services/environmental-indicators/seaice.html.

Fagerland, M.W., and Hosmer, D.W. 2017. How to test for goodness of fit in ordinal logistic regression models. The Stata Journal 17:668-686.

Frey, K.E., Comiso, J.C., Cooper, L.W., Grebmeier, J.M., and Stock, L.V. 2018. Arctic Ocean Primary Productivity: The response of marine algae to climate warming and sea ice decline. NOAA Arctic Report Card 2018.

Furnell, D.J., Schweinsburg, R.E. 1984. Population dynamics of central Arctic polar bears. Journal of Wildlife Management 48:722-728.

Garshelis, D.L. 2006. On the allure of noninvasive genetic sampling – putting a face to the name. Ursus 17:109-123.

Gelman, A., Stern, H.S., Carlin, J.B., Dunson, D.B., Vehtari, A. and Rubin, D.B., 2013. *Bayesian data analysis*. Chapman and Hall/CRC Press.

Gibbs, J.P. 2008. Monitoring for Adaptive Management in Conservation Biology. Synthesis. American Museum of Natural History, Lessons in Conservation. Available at http:// ncep.amnh.org/linc

Häder, D.-P., Villafañe, V.E., and Helbling, E.W. 2014. Productivity of aquatic primary producers under global climate change. Photochemical and Photobiological Sciences 13:1370-1392.

Hamilton, S.G., and Derocher, A.E. 2018. Assessment of global polar bear abundance and vulnerability. Animal Conservation 22:83-85.

Hamilton, S.G., Castro de la Guardia, L., derocher, A.E., Sahanatien, V., Tremblay, B., and Huard, D. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. PLoS One, DOI:10.1371/journal.pone.0113746.

Hannah, C.G., Dupont, F., and Dunphy, M. 2009. Polynyas and tidal currents in the Canadian Arctic Archipelago. Arctic 62:83-95.

Holling, C.S. 1978. Adaptive environmental assessment and management. Willey & Sons.

Honderich, J.E. 1991. Wildlife as a hazardous resource: an analysis of the historical interaction of humans and polar bears in the Canadian Arctic 2000 B.C. to A.D. 1935. Master of Arts, University of Waterloo, Waterloo, ON, Canada.

Howell, S.E.L., Tivy, A., Yackel, J.J., and McCourt, S. 2008. Multi-year sea-Ice conditions in the western Canadian Arctic Archipelago region of the Northwest Passage: 1968–2006. Atmosphere-Ocean 46:229-242.

Howell, S.E.L., Duguay, C.R., and Markus, T. 2009. Sea ice conditions and melt season duration variability within the Canadian Arctic Archipelago: 1979–2008. Geophysical Research Letters, DOI: 10.1029/2009GL037681.

Howell, S.E.L., Wohlleben, T., Komarov, A., Pizzolato, L., and Derksen, C. 2013a. Recent extreme light sea ice years in the Canadian Arctic Archipelago: 2011 and 2012 eclipse 1998 and 2007. The Cryosphere 7:1753-1768.

Howell, S.E.L., Wohlleben, T., Dabboon, M., Derksen, C. Komarov, A., and Pizzolato, L. 2013b. Recent changes in the exchange of sea ice between the Arctic Ocean and the Canadian Arctic Archipelago. Journal of Geophysical Research Oceans, 118, DOI:10.1002/jgrc.20265.

Johnson, B.L. 1999. The role of adaptive management as an operational approach for resource management agencies. Conservation Ecology 3: 8. [online] URL: http://www.consecol.org/vol3/iss2/art8.

Jolly, G.M., 1965. Explicit estimates from capture-recapture data with both death and immigration-stochastic model. Biometrika 52(1/2):225-247.

Keith, D.J. Arqvig, L. Kamookak, J. Ameralik and the Gjoa Haven Hunters" and Trappers" Organization. 2005. Inuit Qaujimaningit Nanurnut, Inuit Knowledge of Polar Bears. Gjoa Haven Hunters" and Trappers" Organization and CCI Press, Edmonton, Alberta.

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

Kendall, W.L. et al. 2009. Sampling design considerations for demographic studies: a case of colonial seabirds. Ecological Applications 19:55-68.

Kendall, W. 1999. Robustness of closed capture-recapture methods to violations of the closure assumption. Ecology 80:2517-2525.

Kéry, M. and Schaub, M., 2011. *Bayesian population analysis using WinBUGS: a hierarchical perspective*. Academic Press.

Laake, J.L. 2013. RMark: An R Interface for Analysis of Capture-Recapture Data with MARK. AFSC Processed Rep 2013-01, 25p. Alaska Fish. Sci. Cent., NOAA, Natl. Mar. Fish. Serv., 7600 Sand Point Way NE, Seattle WA 98115.

Laidre, K.L., Atkinson, S., Regehr, E.V., Stern, H.L., Born, E.W., Wiig, O., Lunn, N., Dyck. M. 2020. Interrelated ecological impacts of climate change on an apex predator. Ecological Applications doi.org/10.1002/eap.2071

Lancia, R.A., Braun, C.E., Collopy, M.W., Dueser, R.D., Kie, J.G. et al. 1996. ARM! For the future: adaptive resource management in the wildlife profession. Wildlife Society Bulletin 24:436-442.

Larsen, T.S., and I. Stirling. 2009. The agreement on the conservation of polar bears - it's history and future. Norsk Polarinstitutt, Tromso, Norway.

Lebreton, J.-D., Burnham, K.P., Clobert, J., and Anderson, D.R. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies.Ecological Monographs 62:67–118.

Lunn, N.J. et al. 2010. Polar bear management in Canada, 2005-2008. Pages 87-114 *in* M.E. Obbard, G.W. Thiemann, E. Peacock, and T.D. DeBruyn (eds.) *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Copenhagen, Denmark, 29 June–3 July 2009.* Gland, Switzerland and Cambridge, UK: IUCN.

Lunn, N.J., Servanty, S., Regehr, E.V., Converse, S.J., Richardson, E., and Stirling, I. 2016. Demography of an apex predator at the edge of its range: impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26: 1302-1320.

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

Marz, S. 2010. Arctic Sea Ice Ecosystem: a summary of species that depend on and associate with sea ice and projected impacts from sea ice changes. CAFF Assessment Series Report, Anchorage, AK, 64.pp. Available at: http://www.caff.is/assessment-series/97-arctic-sea-ice-ecosystem-a-summary-of-species-that-depend-on-and-associate-with. Accessed 29 December 2016.

McKinney, M. A., Atwood, T., Dietz, R., Sonne, C., Iverson, S.J., and Peacock, E. 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution 4:516–527.

McLoughlin, P. D., Taylor, M.K., and Messier, F. 2005. Conservation risks of male-selective harvest for mammals with low reproductive potential. Journal of Wildlife Management 69:1592-1600.

Mills, L.S., 2012. Conservation of wildlife populations: demography, genetics, and management. John Wiley & Sons.

Nichols, J.D., and Williams, B.K. 2006. Monitoring for conservation. Trends in Ecology and Evolution. 21:668-673.

Nunavut Wildlife Act, SNu. 2003. c 26, URL http://canlii.ca/t/51x1n retrieved on 2020-01-14.

Obbard, M.E., Thiemann, G.W., Peacock, E. and DeBruyn, T.D (eds). 2010. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Copenhagen, Denmark, 29 June–3 July 2009. pp. 235. IUCN, Gland, Switzerland and Cambridge, UK.

Obbard, M.E., Stapleton, S., Szor, G., Middel, K.R., Jutras, C., and Dyck, M. 2018. Re-assessing abundance of Southern Hudson Bay polar bears by aerial survey: effects of climate change at the southern edge of the range. Arctic Science 4:634-655.

Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15–32.

Olson, T. 2009. Remote biopsy dart sampling of brown bears. National Park Service, U.S. Department of Interior. Katmai National Park and Preserve. Alaska Region Natural Resources Technical Report NPS/AR/NRTR-2009-74.

Otis, D.L., Burnham, K.P., White, G.C. and Anderson, D.R., 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62:3-135.

Ostrander, E. A., Sprague, G. F. J. & Rine, J. 1993. Identification and characterization of dinucleotide repeat (CA)n markers for genetic mapping in dog. Genomics, 16: 207- 213.

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

Paetkau D, Shields GF, Strobeck C. 1998 Gene flow between insular, coastal and interior populations of brown bears in Alaska. Molecular Ecology 7:1283 – 1292.

Paetkau, D. and Strobeck C. 1994. Microsatellite analysis of genetic variation in black bear populations. Mol Ecol. 3:489-495.

Paetkau, D. W., Calvert, C., Stirling, I. and Strobeck, C. 1995. Microsatellite analysis of population structure in Canadian polar bears. Mol Ecol 4:347-354.

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

Palsbøll, P. J., Allen, J., Berube, M., et al. 1997. Genetic tagging of humpback whales. Nature 388:767–769.

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

Peacock, E., Taylor, M.K., Laake, J., and Stirling, I. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management, Doi:10.1002/jwm.489.

Perovich, D., Meier, W., Tschudi, M., Farrell, S., Hendricks, S., Gerland, S., Haas, C., Krumpen, T., Polashenski, C., Ricker, R., and Webster, M. 2018. Sea ice. NOAA Arctic Report Card. Available at: https://arctic.noaa.gov/Report-Card/Report-Card-2018/ArtMID/7878/ArticleID/780/Seanbsplce

Pilfold, N.W., Derocher, A.E., Stirling, I., Richardson, E., and Andriashek, D.S. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. *PLoS One* 7: e41429.

Pilfold, N. Derocher, A.E., Stirling, I. & Richardson E. 2014. Polar bear predatory behaviour reveals seascape distribution of ringed seal lairs. Population Ecology 56:129-138.

Plummer, M., 2003. February. RJAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. In *Proceedings of the 3rd international workshop on distributed statistical computing* (Vol. 124, No. 125.10).

Pollock, K.H., Nichols, J.D., Hines, J.E., and Brownie, C. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:1-97.

R Core Team. 2019. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing. Vienna, Austria. www.r-project.org.

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

Regehr, E.V., Ben-David, M., Amstrup, S.C., Durner, G.M., and Horne, J.S. 2009. Chapter 4. Quantifying bias in capture-recapture studies for mobile species: a case study with polar bears; in Polar Bear (*Ursus maritimus*) demography in relation to Arctic sea ice decline. PhD thesis, University of Wyoming, WY.

Regehr, E.V., Wilson, R.R., Rode, K.D., and Runge, M.C. 2015. Resilience and risk—A demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029, 56

Regehr, E.V., Laidre, K.L., Akcakaya, H.R., Amstrup, S.C., Atwood, T.C., Lunn, N.J., Obbard, M., Stern, H., Thiemann, G.W. & Wiig, Ø. 2016. Conservation status of polar bears (*Ursus maritimus*) in relation to projected sea ice declines. Biology Letters 12:1–5.

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

Regehr, E.V., Wilson, R.R., Rode, K.D., Runge, M.C., and Stern, H.L. 2017. Harvesting wildlife affected by climate change: a modelling and management approach for polar bears. Journal of Applied Ecology 54:1534 –1543.

Regehr, E.V., Hostetter, N.J., Wilson, R.R., Rode, K.D., Martin, M.S., and Converse, S.J. 2018. Integrated population modeling provides the first empirical estimates of vital rates and abundance for polar bears in the Chukchi Sea. Scientific Reports 8:16780.

Reimer, J.R., Brown, H., Beltaos-Kerr, E., and de Vries, G. 2019. Evidence of intraspecific prey switching: stage-structure predation of polar bears on ringed seals. Oecologia 189:133-148.

Richter-Menge, J., Jeffries, M.O., and Osborne, E. Eds. 2018. The Arctic In "State of the Climate in 2017". Bulletin of the American Meteorological Society 99:S143-S173.

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

Robbins, C.T., Ben-David, M., Fortin, J.K. and Nelson, O.L. 2012. Maternal condition determines birth date and growth of newborn bear cubs. Journal of Mammalogy 93:540-546.

Rothrock, D.A., Yu, Y., and Maykut, G.A. 1999. Thinning of the Arctic sea-ice cover. Geophysical Research Letters 26:3469–3472.

Schwarz, C.J., and Arnason, A.N. 1996. A general methodology for the analysis of capture–recapture experiments in open populations. Biometrics 52:860– 873.

Schweinsburg, R.E., Furnell, D.J., and Miller, S.J. 1981. Abundance, distribution and population structure of polar bears in the lower central Arctic islands. Government of the Northwest Territories File Report, Yellowknife, N.W.T. 92 pages.

Schweinsburg, R.E., Lee, L.J., Latour, P.B. 1982. Distribution, movement and abundance of polar bears in Lancaster Sound, Northwest Territories. Arctic 35: 159-169.

Seber, G.A., 1965. A note on the multiple-recapture census. Biometrika 52(1/2):249-259.

Sou, T., and Flato, G. 2009. Sea Ice in the Canadian Arctic Archipelago: modeling the past (1950–2004) and the future (2041–60). Journal of Climate, DOI: 10.1175/2008JCLI2335.1

Stern, H.L., and Laidre, K.L. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere, 10:1-15.

Stewart, E.J., Howell, S.E.L., Draper, D., Yackel, J., and Tivy, A. 2007. Sea ice in Canada's Arctic: implications for cruise tourism. Arctic 60:370-380.

Stirling, I. Spencer, C. and Andriashek, D. 2016. Behavior and activity budgets of wild breeding polar bears (*Ursus maritimus)*. Marine Mammal Science 32:13-37.

Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694–2706

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

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

Stirling, I. 1997. The biological importance of polynyas in the Canadian Arctic. Arctic 33:303-315.

Stirling, I., and Archibald R. 1977. Aspects of predation of seals by polar bears. Journal of the Fisheries Research Board of Canada 34:1126-1129.

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

Taberlet, P., Camarra, J.-J., Griffin, S., Uhres, E., Hanotte, O., Waits, L. P., Dubois-Paganon, C., Burke, T. & Bouvet, J. 1997. Noninvasive genetic tracking of the endangered Pyrenean brown bear population. Molecular Ecology, 6:869-876.

Taylor, B.L., Martinez, M., Gerrodette, T., Barlow, J., and Hrovat, Y.N. 2007. Lessons from monitoring trends in abundance of marine mammals. Marine Mammal Science 23:157-175.

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

Taylor, M. K., Laake, J., McLoughlin, P.D., Born, E.W., Cluff, H.D., Ferguson, S.H., Rosing-Asvid, A., Schweinsburg, R., and Messier, F. 2005. Demography and viability of a hunted population of polar bears. Arctic 58:203–215.

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

Taylor, M.K., Laake, J.L., McLoughlin, P.D., Cluff, H.D., Messier, F. 2008b. Mark-recapture and stochastic population models for polar bears of the high Arctic. Arctic 61:143-152.

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

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

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

Vongraven, D., and Peacock, E. 2011. Development of a pan-Arctic monitoring plan for polar bears: a background paper. Circumpolar Biodiversity Monitoring Programme, CAFF Monitoring Series Report No. 1. CAFF International Secretariat, Akureyri, Iceland.

Vongraven, D., Aars, J., Amstrup, S., Atkinson, S.N., Belikov, S., Born, E.W., DeBruyn, T.D., Derocher, A.E., Durner, G., Gill, M., Lunn, N., Obbard, M.E., Omelak, J., Ovsyanikov, N., Peacock, E., Richardson, E., Sahanatien, V., Stirling, I., and Wiig, Ø. 2012. A circumpolar monitoring framework for polar bears. Ursus 5:1–66.

Wiig, Ø., Amstrup, S., Atwood, T., Laidre, K., Lunn, N., Obbard, M., Regehr, E. and Thiemann, G. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490.

White, G.C. and Burnham, K.P., 1999. Program MARK: survival estimation from populations of marked animals. Bird study, 46(sup1):S120-S139.

10. FIGURES AND TABLES
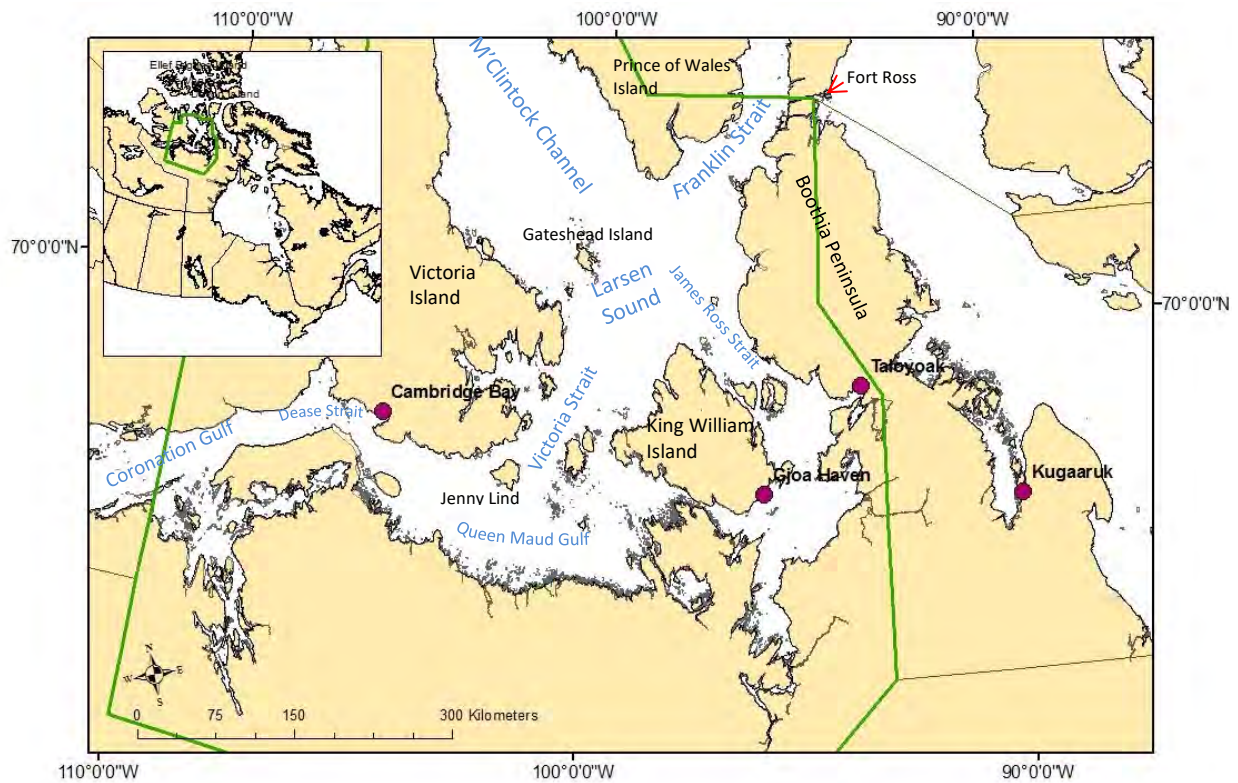
Figure 1.   Overview and location of the M'Clintock Channel polar bear subpopulation with major geographical features and water bodies.
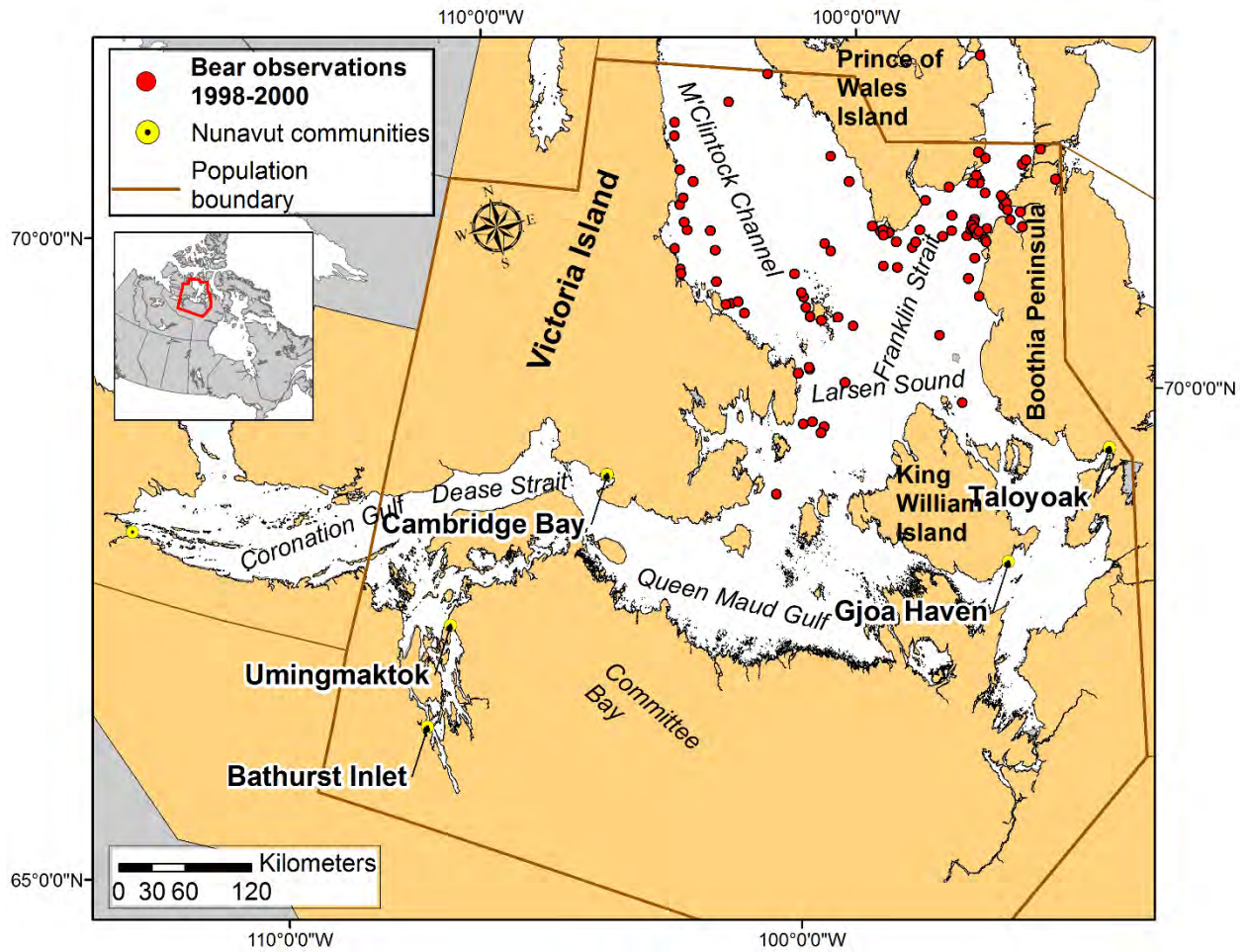
Figure 2. *Capture and re-capture locations for the 1998 – 2000 M'Clintock Channel polar bear study.*
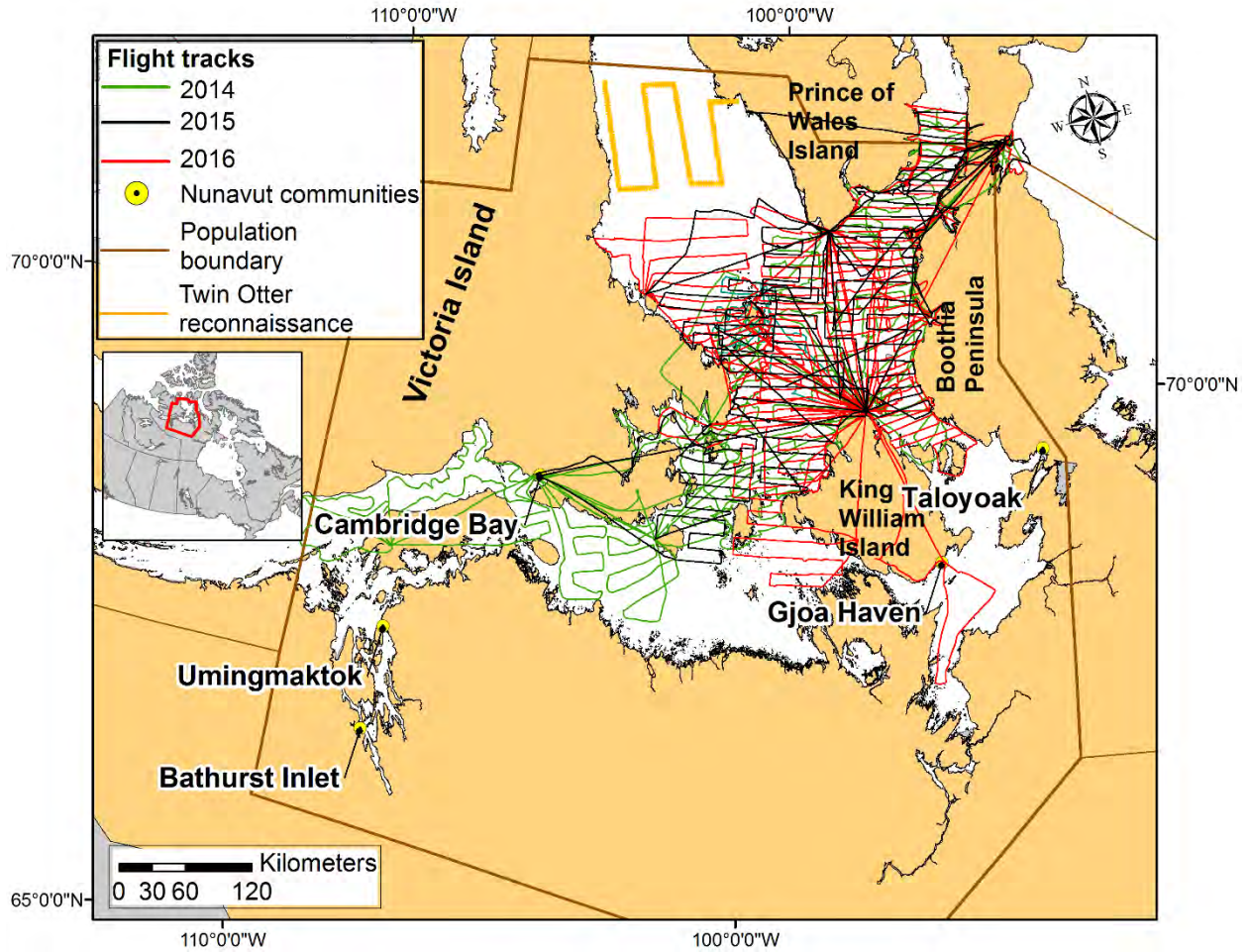
Figure 3. *Helicopter paths flown in search for polar bears in M'Clintock Channel, Nunavut, Canada, during April/May-June 2014 – 2016. The golden path represents the Twin Otter reconnaissance flight during April 2016.*
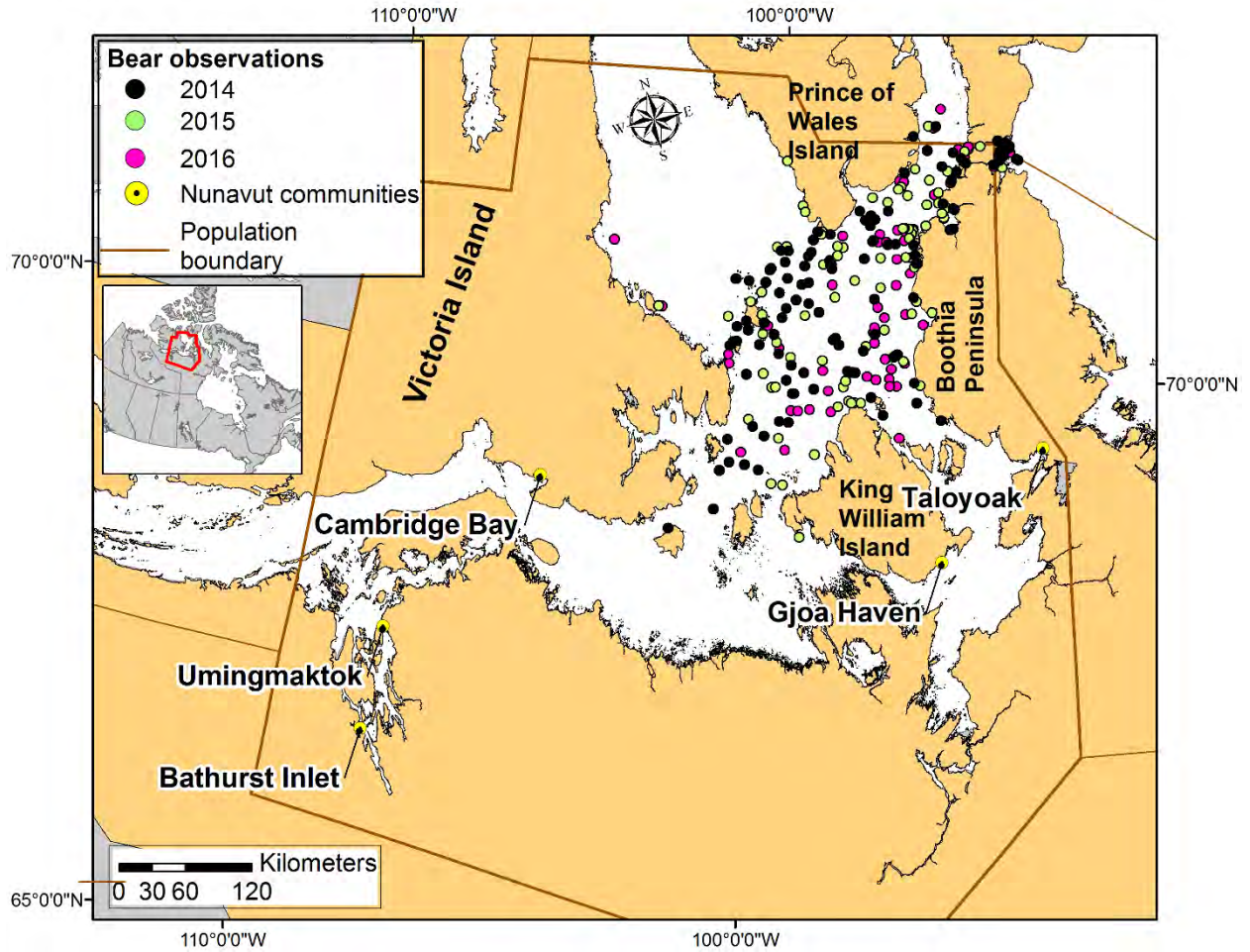
Figure 4.   Locations of polar bear encounters in the M'Clintock Channel polar bear subpopulation during April – June of 2014 – 2016.
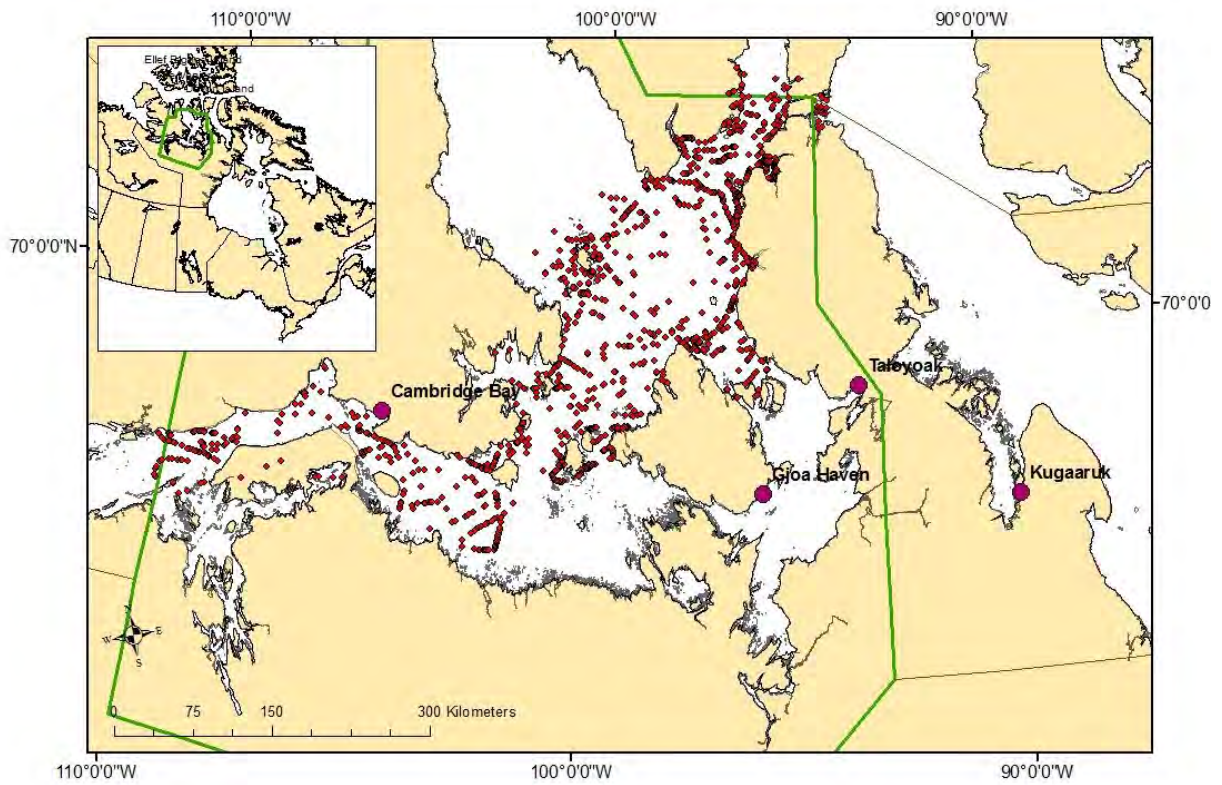
Figure 5.   *Seal observations for May – June 2014 in M'Clintock Channel (n = 2,236) recorded during search for polar bears.*
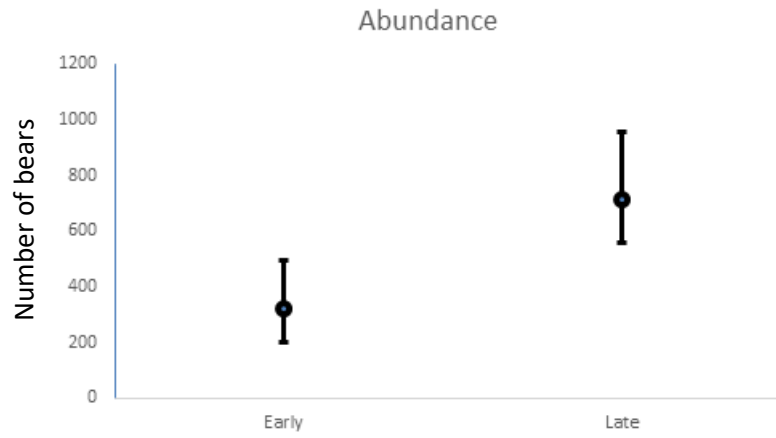
Figure 6. Estimated polar bear abundance in M'Clintock Channel during the early (1998 – 2000) and late (2014 – 2016) study periods.
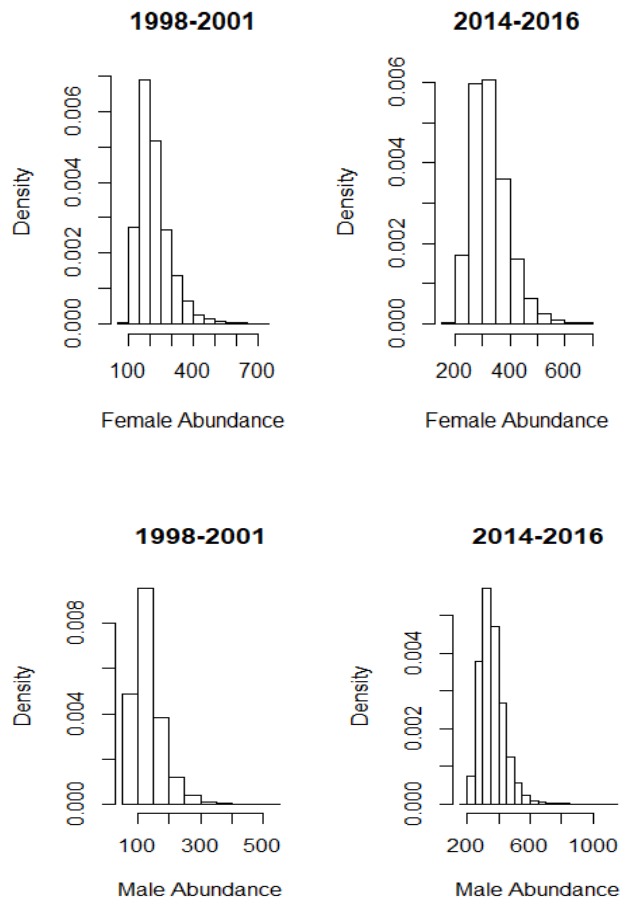
Figure 7. Posterior distributions for abundance estimates of female (top) and male (bottom) M'Clintock Channel polar bears.
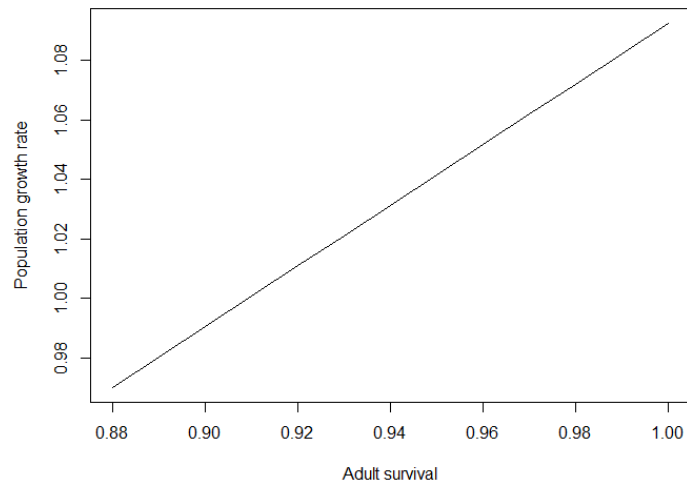
Figure 8. Population growth rate (λ) as a function of adult female survival. The observed growth rate is achieved when survival is approximately 0.92.
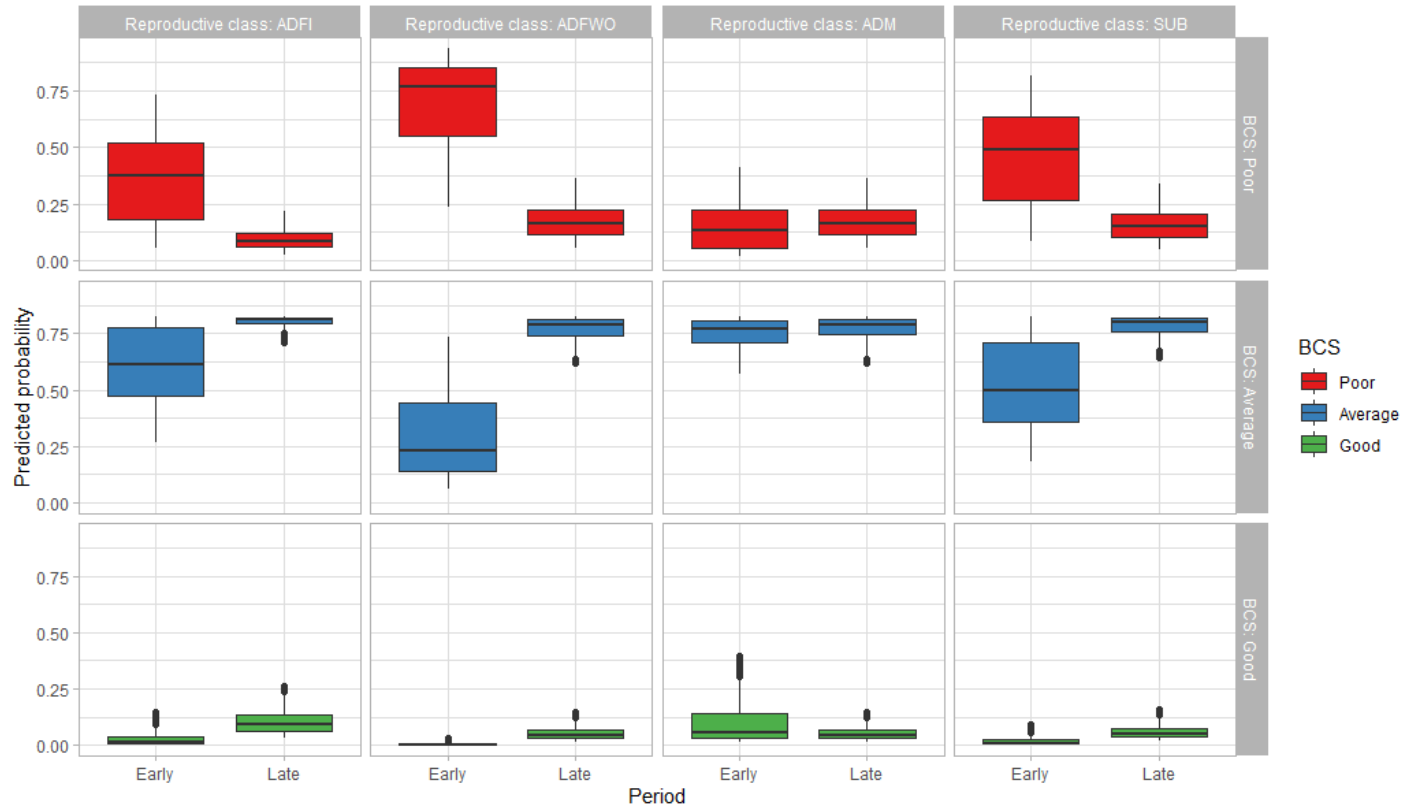
Figure 9. Predicted probabilities of bears being classified in Poor, Average, or Good condition in the early (1998 – 2000) and late (2014 – 2016) sampling periods. ADFI = adult, independent female, ADFWO = adult female with offspring, ADM= adult male, SUB = subadults of both sexes
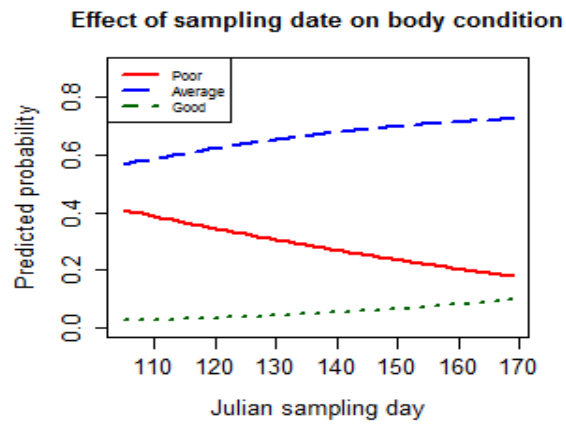
Figure 10. *Predicted probabilities of a bear being in Poor, Average, or Good body condition when sampled at different dates.*

Table 1. Parameter estimates for best fit ordinal logistic regression model for body condition analysis of the M'Clintock Channel polar bear subpopulation

Parameter	Estimate	SE	p
periodEarly	1.27	1.54	0.41
reproclassADFWO	-0.74	0.47	0.12
reproclassADM	-0.73	0.40	0.07
reproclassSUB	-0.62	0.50	0.21
jul_cap_day	0.02	0.01	<0.01
$icetm_{t-1}$	-0.01	0.01	0.11
periodearly: $icetm_{t-1}$	-0.03	0.01	0.05
periodearly:reproclassADFWO	-0.97	0.82	0.24
periodearly:reproclassADM	2.07	0.96	0.03
periodearly:reproclassSUB	0.14	0.77	0.86

Table 2.   Overview of field statistics of the M'Clintock Channel polar bear study 2014 – 2016.

Field Year	Search time (hrs)	Number of bears/hr	Bears encountered[a]	Flown distance (km)	Duration
2014	97.5	1.90	155	12,600	4 May – 18 June
2015	72.5	1.68	122	10,100	5 May – 8 June
2016	94.0	1.00	95	14,200	19 April – 7 June

[a] The number of bears encountered does not represent the number of unique individuals (e.g., some bears have been resampled within same sampling period)

Table 3.   Model selection results for Cormack-Jolly-Seber models of polar bear capture-recapture data from 1998 – 2016 used to estimate apparent survival of independent bears > 2 years. K is the number of parameters in the model.

Model	K	AICc	ΔAICc	Weight	Deviance
Phi(constant) p(constant)	2	425.53	0.00	0.28	26.46
Phi(sex)p(constant)	3	426.22	0.69	0.19	420.15
Phi(constant)p(period)	3	426.40	0.87	0.18	25.30
Phi(sex)p(period)	4	427.02	1.49	0.13	418.90
Phi(sex)p(sex)	4	427.25	1.72	0.12	419.13
Phi(constant)p(sex)	3	427.46	1.94	0.10	421.39

Table 4. Mean numbers for cubs-of-the-year (C0) and yearlings (C1) per adult female and litter size for the M'Clintock Channel polar bear subpopulation, 1998 – 2000 and 2014 – 2016.

Year	Offspring per adult female			Litter size*			
	C0	C1		C0	n	C1	n
1998	0.40	0.25		2.00	4	1.67	3
1999	0.40	0.33		1.20	5	1.67	3
2000	0.33	0.60		1.67	3	1.80	5
2014	0.41	0.15		2.00	8	1.50	4
2015	0.61	0.35		1.50	14	1.71	7
2016	0.26	0.32		1.80	5	1.57	7

*Litter sizes of zero (whole litter loss) are not listed; all litters depend on at least one offspring being present.

Table 5.   Body condition scores (BCS) for polar bears (n = 380) in the M'Clintock
Channel subpopulation 1998 – 2000 and 2014 – 2016. Poor BCS
corresponds to a thin bear and Good BCS corresponds to a fat/obese bear.
Age classes are adult (≥ 5 years) and subadult (2 – 4 years).

	Body condition scores						
	1998 – 2000				2014 – 2016		
	Poor	Average	Good		Poor	Average	Good
Adult female without offspring	6	12	1		4	52	8
Adult female with offspring	22	8	1		4	44	1
Adult male	2	9	1		18	78	11
Subadult	24	31	2		2	38	1

Appendix A Study activities

Ice habitat images from the field work, in addition to some images of the genetic biopsy darting activities are presented in this appendix to demonstrate the harsh environment, field activities and the non-invasiveness of the technique.
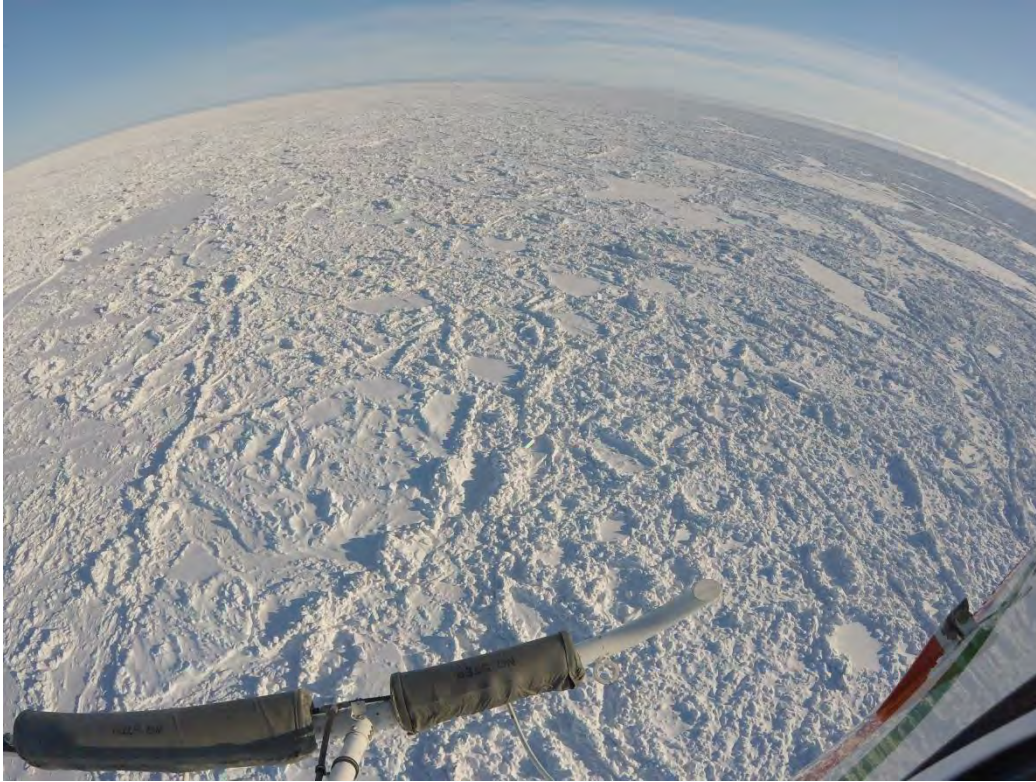

Plate A1. Image from the helicopter directly facing the sea ice. Rough ice, pressure ridges, and ice pans are visible (M. Dyck, Government of Nunavut).

Plate A2. *A polar bear being genetically sampled from the air. The **orange color at the left rump area is the flagging tape from** the mid-air dart as it hits the bear and falls to the ground (M. Dyck, Government of Nunavut).*

Plate A3. *View of the sea ice with pressure ridges and a wind-blown and snow-encrusted surface. A polar bear is visible in the red circle (M. Dyck, Government of Nunavut).*

Plate A4.   Discoloured multi-year ice pushed together to form high pressure ridges and rubble ice fields (M. Dyck, Government of Nunavut).

Plate A5.   Five adult male polar bears along a crack in the sea-ice. These bears were observed feeding together on a bearded seal carcass in May 2014. The sixth bear is not pictured (M. Dyck, Government of Nunavut).

Plate A6. Genetic biopsy sampling is very minimally invasive. A male polar bear is pictured lying down after being darted, with the dart in the background (M. Dyck, Government of Nunavut).

Plate A7. View of flatter sea-ice areas with a polar bear circled on the flat portion of the sea ice (M. Dyck, Government of Nunavut).