ASSESSMENT OF ABUNDANCE FOR THE GULF OF BOOTHIA POLAR BEAR SUBPOPULATION USING GENETIC MARK-RECAPTURE

Final Report

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1.A) EXECUTIVE SUMMARY – ENGLISH -

Polar bears (*Ursus maritimus*) are managed across Nunavut, Canada, under a quota system that seeks to ensure harvest is sustainable. 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 ensure continued availability of 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 Gulf of Boothia (GB) conducted 2015 – 2017. Current samples were collected using less-invasive genetic biopsy darting without immobilizing or physically handling bears. Our analyses included 2015 – 2017 biopsy sampling data, live-capture data collected under a designed study 1998 – 2000, live-capture data collected opportunistically 1976 – 1997, and harvest recovery data over the entire period 1976 – 2017. Results of live-capture dead-recovery models fitted in Program MARK suggest that a mean abundance estimate of 1525 (standard error [SE] = 294) for the period 2015 – 2017 was similar to mean abundance in 1998 – 2000 (1610 [SE = 266] in this study; 1592 [SE = 361] in Taylor et al. [2009]). Mean cub-of-the-year and yearling litter sizes for the period 2015 – 2017 were 1.61 (95% confidence interval [CI] = 1.51 – 1.70) and 1.53 (95% CI = 1.41 – 1.64), respectively, with no apparent trend compared to 1998 – 2000. The mean number of yearlings per adult female for the period 2015 – 2017 was 0.36 (95% CI = 0.26 – 0.47) which suggests that GB is currently a productive polar bear subpopulation, despite sea ice change. This is consistent with our finding that polar bear body condition (i.e., fatness) in the spring increased between the periods 1998 – 2000 and 2015 – 2017. We detected sex- and age-specific variation in total survival rate (i.e., including harvest mortality) with higher estimates for adult females (0.95; 95% CI = 0.81 – 0.99) than adult males (0.85; 95% CI = 0.74 – 0.92) for the period 2005 – 2017. A potentially related effect was detected as an increase in the proportional abundance of females from 0.57 in 1998 – 2000 to 0.61 in 2015 – 2017. The asymptotic, intrinsic population growth rate calculated using a matrix projection model with estimates of total survival was 0.06 (95% CI = -0.06 – 0.12) for the period 2005 – 2017, suggesting strong

potential for growth. However, our results for subpopulation size and trend should be interpreted with caution because our estimate of abundance reflects the "superpopulation" (e.g., it includes all bears that use the GB management area, some of which spend time in other subpopulations as well) and our estimate of population growth rate does not account for permanent emigration from the GB management area. Overall, our findings suggest that the demographic status of the GB subpopulation is currently healthy, although we recommend that lower estimates of total and un-harvested survival for male bears warrant further investigation. We hypothesize that spatial and temporal reductions in sea ice may have provided transient benefits to the GB subpopulation due to increased biological productivity. Climate change is the primary long-term threat to polar bears and the threshold beyond which the GB subpopulation could be negatively affected by continued ice loss, like some other polar bear subpopulations, is currently unknown. This study represents the second structured population assessment in 22 years for the GB subpopulation. Based on experience garnered through this study and analysis, we submit several recommendations for consideration when planning future polar bear population studies. We suggest collecting additional data at approximately the midpoint between planned subpopulation assessments. In this case, that equals approximately 5 – 7 years from the 2017 completion of field work. Additionally, while the recommendation for movement data is not new, it continues to be highly recommended for subpopulations with known exchanges of bears between areas. In the absence of satellite telemetry data on polar bear movements, conducting a meta-analysis to investigate exchange between GB and nearby subpopulations (i.e., Lancaster Sound, GB, and M'Clintock Channel) may help alleviate some of the uncertainty around individual subpopulation estimates for these areas. Finally, when time, resources, and management objectives warrant it, we recommend conducting a quantitative harvest risk assessment to inform sustainable harvest levels.

1.B) EXECUTIVE SUMMARY INUKTITUT

ᖃᐅᔨᓴᖅᓯᓂᖅ ᐅᓄᕐᓂᖕᒐᓄ� Gulf of Boothia–ᒥ ᓇᓄᕆᔭᐅᖃᑦᑕᓂᖅᑐᓄ ᐊᑐᖅᑐᖅ�ᒡᑐᓂ ᑎᒥᖕᑎᑎᒍ ᓇᓄᖃᐃᑯᑦᑎᖅᓯᓂᒡᒃ–ᐱᒃᑫᓂᖅᓯᓂᒡᒃ

ᐊᐅᓚᑦᑎᔨᓂᖕᓴᒡᑐ ᓇᐃᓇᖅᓯᒃᒐᕐᖅ

ᓇᓄᐃ (*Ursus maritimus*) ᐊᐅᓚᑦᑎᐃᐅᕐᒃᐳᒃ ᓄᓇᖃᒡᒥᒑ, ᑲᓇᑕᒥ, ᖃᕐᔾᒡᑡᐸᓄᖕᓂᓚᒡ ᐊᑐᖅᑕᐅᕐᖅᑐᑎᒃ ᖁᓂᖅᑐᖃᒡ ᓇᓄᖅᑎᑎᐊᒐᒪᓂᓚᒡ ᑕᒪ ᐊᒡᓂᕆᔭᐅᕐᖅᑐᖅ ᑲᔾᒡᓇᑐᒪᓇᒡᓚᖕᒢ. ᐊᕐᒡᒐᓴᖅᑐᒐ ᖂᑕᓂ, ᓯᓚᐅᒡ ᐊᕐᒡᒃᓯᓂᐅᒢᒃᒡ ᐅᑉᐅᖅᑕᖅᑐᓪᐱᒐ ᐊᕐᒡᒃᑎᑎᑎᓯᓚᖅ ᓇᓄᐃ ᐃᓂᒓᖅᒄᖕᒧᓂᒃ, ᑕᒪᓇᒪ ᓯᒃᑮᒡᑖᓚᓂᐅᒢᒦᓂ. ᐱᓯᕿᓛᒡᒄ ᑲᔾᒡᒄᒐᒡᑎᐊᖅᑐᒡᒃ ᓇᓄᖕᒧᒢᒃ ᓇᓄᖃᑎᒡᒄᐱᒄᖕᒃᖅᑐᒄ ᐃᑎᕆᔭᐅᓄᒪᒣ ᓄᓇᒄᒦ ᓇᓄᖕᒡᑎᑎᐊᖅᒐᒄᐅᕐᖅ ᑲᔾᒡᒐᖕᑐᒦ ᒪᒓᓚᕐᖕᒧᒦᒓ ᐊᑐᖅᑕᐅᕐᒐᖅᑐᒡ ᐃᓄᖕᓂᒄ, ᖃᐅᔨᓴᓂᒦᒄᒡ ᓇᐅᑎᖅᓯᕐᒃᓯᓂᒦᒄᒥ ᐱᓯᓂᐊᖕᒡᐅᖅᑐᒦ ᖃᐅᔨᓴᖅᓯᒥᓄᓂᒡᒡᒦᒦ ᓇᒓᖅᑎᑎᐱᖕᒡᓂᖅᑐᒃ ᖃᓄᐃᓯᓚᓂᖕᒧᒄᒥᒃ ᐊᒡᓚᒄ ᐊᐅᓚᑦᑎᓂᒡᒡᒄ ᑐᕕᕐᒡᐱᐅᒡᒔ ᐊᐱᖕᑎᓚᕐᐅᒦᓄᐊᒡᓚᖕᒢᒄ. ᐅᕙᓂ ᐅᓂᖕᑲᐅᒡᖅᓯᖅᒃᐳᒦᒡ ᖃᓄᐃᓯᓚᓂᖕᒧᒦᒃ ᖃᕐᔾᐅᓂᖕᒧᓂᒄ ᖃᐅᔨᓴᖅᓯᓂᒦᒡ ᓇᓄᒦᒄ ᓄᓇᓚᖕᒧᒄ Gulf of Boothia–ᒥ ᑲᔾᒡᒄᓚᐅᕐᖅᐳᖅ 2015-2017-ᒥ. ᒣᖕᒪᒄ ᖃᖕᓄᓚᖕᒄᐃᒦ ᑲᑎᐅᓇᓚᐅᕐᖅᐳᒦ ᐊᑐᖅᑐᖕᒡᖕᒄᑐᒦ ᐸᐅᒦᓴᖕᒡᒢᓄᖅᖕᒄᒦ ᑎᒓᖕᓚᓂᒄ ᐱᖅᒓᔭᕐᖅᑐᒦ ᓇᐅᒃᖅᑕᐅᖕᒧᒡᓚᒦ ᐊᐅᓚᕐᒧᓇᐃᓯᑎᖕᒡᖕᒡᒦᒣᒄᒡ ᑎᒓᖕᑎᑎᒍᒦᒢᖕᒦᓂᒡ ᑲᖕᒥᒄᑲᖕᒄᐅᖕᒢᒦᒡᒄᖕᒄ. ᖃᐅᔨᓴᓂᖕᒄᔭᒦᒡ ᐃᑎᖃᑕᐅᖅᖕᒄᒦᒣᒡ 2015-2017-ᒦ ᑎᒓᖕᒄᓚᓂᒄ ᖃᐅᔨᓴᓚᖕᒄᖕᒡᒡᒦᒄ ᑎᑎᕐᖅᒥᓚᕐᖕᓂᒦᒃ, ᐅᒃᐱᒓᖕᒄ–ᐱᖅᑎᒓᒄᒡᒄ ᑎᑎᕐᖅᒥᓚᕐᖕᓂᒄ ᑲᑎᐅᕐᖕᓂᒦᒢ ᐊᖅᖕᒡᒃᔭᒡᑎᒄᒢᖕᑆᒓᒦ ᖃᐅᔨᓴᓂᖕᒄᒣᒄ 1998-2000-ᒥ, ᐅᒦᒓᖕ–ᐱᖅᑎᒓᖕᓂᒄ ᑎᑎᕐᖅᒥᓚᕐᖕᓂᒄ ᑲᑎᐅᓚᐅᕐᖅᑐᒡ ᐱᖕᒃᓯᖕᒄᓂᓚᒡ 1976–1997–ᒥ, ᐊᕐᐅᖅᒃᖅᐸᒄᖕᒓᓚᓄᒡᒄᒄᒡ ᐅᑎᖅᑎᑎᓂᖕᒡᒡᒄ ᑎᑎᕐᖅᒥᓚᕐᖕᓂᒄ ᐅᐹᖕᓗᒡᒦᖅ 1976-2017–ᒡᒦᒡ. ᖃᐅᔨᔭᐅᒡᒄ ᐅᒪᒓᖕ–ᐱᓯᓂᖕᒄ, ᑐᒦᕐᔭᒓ–ᐱᖕᓚᖕᒃᓂᖕᒦᒄ ᐊᑐᖅᑕᐅᓄᒡᖅᑐᒦᒃ ᐊᖅᖕᒡᒃᔭᒄᖕᑎᐅᖅᒦᓚᕐᒧ MARK–ᒥ ᐃᓚᐅᑎᑎᖅᑯᓚᒄᒓᒡ, ᖃᔾᒡᐅᓂᖕᒧᓂᒡ ᖃᔾᒡᐅᓇᕐᒓᔭᐅᐅᒦᒓ 1525–ᓂᒡ (ᑕᔾᒦᖅᑕᐅᓚᕐᖅᖕᒓᖅ [SE] = 294) 2015 – 2017–ᒡᒡᒦ ᐊᕐᖅᒃᒃᓚᖕᓚᐅᕐᖅᐳᖅ ᐅᓄᖕᒄᓇᒦᒓᖕᒡᒄᒧᓂᒡ 1998 – 2000–ᒥ (1610 [SE = 266] ᑕᖕᒄᐅᓂ ᖃᐅᔨᓴᓇᒃᔭᐅᖅᒥ; 1592 [SE = 361]ᐅᖕᒄᓂ, Taylor et al. [2009–ᒥ]).

ᓇᓇᖅᑡᖅᑕᖁᔭᐅᔪᑦ ᐊᕐᕈᒐᒡᒥ ᐊᕐᕉᓇᑎᐊᓐᓗ ᐊᖑᑎᒥᓂᖕᒋᑦ 2015 – 2017−ᒧᑦ 1.61−ᖑᒐᐅᖅᔪᑦ (95%−ᒥᑦ ᑕᖃᖏᓇᐅᓯᖕᓂᖕᓂᑦ [CI] = 1.51 – 1.70) ᐊᒡᒧ 1.53 (95% CI 1.41− 1.64), ᑭᖑᓕᖏᔭᒍᑎᒃ, ᐅᔾᐦᓇᖅᑑᒥᒃ ᖃᓄᐃᓚᕿᓐᓇᐃᒃᓈ ᑕᑯᓪᒧᒍ 1998 – 2000. ᖃᔭᐅᑎᓂᔭᖕᒋᑦ ᐊᕐᕈᓇᑎᐊᑦ ᐃᖕᓇᔾᑦ ᐊᕚᒧᑦ 2015 – 2017−ᒧᑦ 0.36−ᖑᒐᐅᖅᔪᖅ (95% CI = 0.26 – 0.47) ᑕᒪᖕᓇ ᐃᓚᐃᓐᑎᖅᑯᒥᔾᓈᔪᓂ, ᐸᖕᓇ GB ᒥᖕᓇ ᖄᑐᖁᒥᐅᖅᒪᐅᓐᓗᓂ ᓇᓇᔾᓂᑦ, ᓱᒐᖕᓗ ᐊᔾᔾᖃᔾᒪᓚᕐᓗᐊᖅᑎᓪᓗᒍ. ᑕᒪᖕᓇ ᐊᔾᖃᖅᔭᖅ ᖃᐅᔨᔾᑎᓂᑦ ᓇᓇᐃᑦ ᑎᒥᖕᒋᑦ ᖃᓄᐃᓚᕈᓇᖕᒋᑦ (ᒫᓗ, ᖄᐃᓯᓇᖕᒋᑦ) ᐅᐱᖕᒪᐢᑯᑦ ᐊᖕᓕᓕᒐᓚᖅᔪᒥᖖᔭᖅ ᐅᑕᓇ, 1998-2000−ᒥ ᐊᒡᒧ 2015-2017-ᒥ. ᖃᐅᔨᓚᐅᖅᔪᑦ ᑭᖑᖕᒋᖕᓂᒃ ᖃᔭᓇᕉᓐᓗ ᐊᕐᕈᖃᖕᓂᖕᓂᒃ−ᐱᔾᑎᖅᓈᔪᐊᖅᑑᒥᒃ ᐊᔾᐊᖕᒋᖕᓂᒋᒥᒃ ᑲᓇᖕᓇᒥᑦ ᐅᒪᔾᖃᓯᓂᔭᖕᒋᓂ (ᒫᓗ ᐃᒫᖃ, ᐃᓇᕆᔭᐅᓇ ᐊᖕᓂᐅᔾᓚᕙᑦ ᑑᒡᑕᐅᔭᑦ) ᓱᒡᑎᖕᒋᖕᓴᒐᒃ ᖃᔭᐅᓇᒃᒥᖕᓈᔾᖃᖅᓗᓂ ᐃᖕᓇᔾᓂᒃ ᐊᕐᓂᓂᑦ (0.95;95% CI=0.81-0.99) ᐃᖕᓇᖕᒋᓂᒃ ᐊᖕᐃᓐᒋᓂᒃ (0.85;95% CI=0.74-0.92 ᐊᕐᕈᒧᓂᑦ 2005-2017. ᐊᑉᑐᐊᖕᑲᑐᐃᖕᓴᓇᐊᓚᖕᒥᒃ ᖃᓄᐃᓚᖕᓚᕐᓂᒡᒥᒃ ᖃᐅᔨᔭᐅᕐᕛᑲᓚᐅᖅᔪᖅ ᐅᓇᖕᔾᓕᐊᓇᐅᓐᓗᓂ ᖃᔭᐅᑎᓂᔭᖕᒋᓂᒃ ᐊᕐᓇᐃᑦ 0.57−ᒥᑦ 1998−2000-ᒥ 0.61−ᒧᑦ 2015−2017−ᒥ. ᖃᔭᐅᑎᓂᖕᒋᓂᒃ ᐅᓇᖕᔾᖁᓚᕐᐊᓂᖕᒋᑦ ᖃᔭᐅᑎᓂᔾᔭᐅᓂᖕᒋᑦ ᐊᑑᖅᑑᖃᖅᓗᓂ matrix ᖃᐅᑐᐊᖅᒪᖕᓂᐅᓂᒡᒧᑦ ᐊᑑᖅᑕᐅᓇᖅᑑᒥᒃ ᖃᔭᐅᑎᓂᖕᓚᖅᒧᓂ ᑲᓇᖕᒋᑦ ᐅᒪᓇᓂᔭᖕᒋᑦ 0.06-ᖑᒐᐅᖅᔪᖅ (95% CI = 0.06 - 0.12) 2005-2007−ᒧᑦ, ᐃᓚᐃᓐᑎᖅᒃᒃᑯᔾᐢᓂᒃ ᓴᖕᒥᓕᒥᒃ ᐊᔾᒫᓂᖕᑐᐃᖕᓴᓇᐊᖅᖃᓂᒃ ᐱᔾᖃᐊᓚᖕᐊᓂᒥᒃ. ᑭᒃᐊᓇᓗᑦᑕᖅ, ᖃᔭᔭᐅᑦ ᖃᔭᐅᓂᖕᓚᑕ ᐅᓇᖕᑎᓐᖕᒋᑦ ᖃᓄᐃᓚᕈᓇᐊᖕᒋᓐᓗ ᑐᑭᖃᖕᑎᐅᓐᓚᐊᖅᔪᑦ ᐅᔾᔾᔭᖕᑎᐊᖕᑐᖅᖃᒥᒃ ᐃᓚᐃᖕᒋᓇᒃᒥᒃ, ᖃᔭᐅᑎᓂᖕᓚᓂᕐ ᐅᓇᕐᓂᑦ ᐊᑉᑐᐊᖕᖃᒪᑦ "ᐅᓇᕐᐊᕐᓂᖕᒧᑦ" (ᒫᓗ ᐃᒫᖃ, ᑕᒪᖕᓇ ᐃᓚᖃᖅᔪᖅ ᓇᓇᒫᓂᒃ ᐊᑐᖅᐊᑐᒃᑖᒥᒃ GB-ᒥᒃ ᐃᓂᒃ ᐊᐅᓪᑎᓇ ᐅ ᒥᒃ, ᐃᓚᖕᒥᓐᓗ ᐊᓱᒋᓇᑦᑕᖅ ᓇᐆᖃᓂᖕᒋᔾᓪᑖᖕᒥᑕᐅᖅ) ᖃᔭᐅᓇᒫᖁᓂᖅᔪᖕᓗ ᖃᔭᐅᓂᖕᒋᑕ ᐅᓇᖕᔾᖁᓚᕐᐊᓂᔭᖕᒋᓂᒃ ᐃᓚᕐᖕᒋᓂᒃ ᐊᔾᐊᓇᐊᕐᒋᓂᒡᒥᒃ ᑕᐃᖄᕈᖕᒋᓇᓂᕐᓗᓇᓗ GB−ᒥᑦ ᐊᐅᓪᑎᓂᒐᒧᑦ ᐃᓂᒥᑦ. ᐊᑕᖕᒌᓂᒐ, ᖃᐅᔾᔭᖅ ᑕᑯᑎᑦᑯᔾᔮᖅ ᓇᓇᐅᓂᖕᒋᓂᒃ ᖃᓄᐃᓚᖕᓂᖕᒋᑦ GB−ᒥ ᓇᓂᕈᔭᖅᑲᖕᓂᖅᑐᒃ ᒫᓇ ᖃᓄᐃᖕᒋᓂᒃ, ᐃᓚᐃᑲᑐᐊᖅᑎᓐᓗᒍ, ᐃᓚᐃᑐᐢᑯᔾᓇᔾᖅᑎᓐᓗᑕ ᐊᖕᑎᓂᖕᓴᓂᒃ ᖃᔭᐅᓇᒫᖁᓂᒍᔾᖁᓂᑎᒃ ᑲᓇᖕᒋᑦ ᐊᖕᔭᐅᔾᓚᖕᒋᑐᓂᓗᓗ ᐅᓇᓇᓇᐊᓇᐅᑎᖕᒋᓂᒃ ᐊᖕᑎᖕᒋᑦ ᓇᓇᐃᑦ ᒫᓇ

ᖃᐅᔨᓴᖅᑕᐅᖃᑦᑕᓂᐊᖅᖃᑦᑎᑎᑦᑐᑦ. ᐃᓕᖃᐅᐅᑲᓯᒐᖅᓂᖁᐅᔪᑦ, ᐃᒥᕈ ᓇᓂᖁᓂᖕᒥᖁᓂ
ᒥᑭᖕᓯᕋᖅᑐᖅ ᑐᓇᖓ ᐱᑎᖁᑎᐃᖁᓇᐃᖃᖅᖂᐅᖅ ᓅᓂᒍᑦ ᐱᕿᑖᓛᔭᑕᐅᔩᒪᐅᓯᓂᖅ GB-ᒥ
ᓇᓄᕆᐅᖃᖕᓂᖅᑐᓂᑦ ᑕᐃᒪᐃᔭᐅᑎᖃᖅᑐᓂᑯ ᓄᓕᕐᑕᐸᑕᐅᖅᒍᑦ
ᐅᓇᖅᔫᖕᓚᖅᒐᖕᓂᒐᔪᕈᖕᓂᒍᑦ. ᓵᐅᐸᑦ ᐊᕐᔭᖅᐸᑦᓚᖕᓈᓕ ᐱᓚᖓᐊᖐᖅ ᐊᖑᖃᑲᓐᑭ
ᖃᑎᖅᓯᓇᖅᑎᑎᓐᓄ ᓇᓂᕐᓂᑦ ᐊᑐᖅᑕᐸᓚᖅᖃᑐᔩᓄ ᑕᒃᑯᐊ GB-ᒥ ᓇᓄᕆᐅᖃᖕᓂᖅᑐᑦ
ᐱᐅᖕᒪᑐᖐᑯᑦ ᐊᑐᖅᑕᐸᓂᖅᑕᖕᒪᖕᓂᒐ ᑲᕈᔫᓇᖅᑐᖐᑯ ᑐᐁᖅ ᓄᔾᐸᕐᓚᑕᐊᖓᕐᕿᑖᖕᓄ, ᓕᖕᓄ
ᐊᓚᕈᒐᓄ ᓇᐊᖅᑲᑎᐡᓄ, ᒪᑐᓄᑦ ᑎᒃᕐᓱ ᖃᐅᔭᒪᕐᑎᒐᖅ. ᑕᒪᓇ
ᖃᐅᔨᐊᓅᕋᐅᖕᕐᓇᓂᕈᑭᖅ ᑭᓕᖅᑐᐃᐅᖅ ᐊᐃᕐᕘᓕᓂᖅ ᐊᖅᑭᑭᖅᑕᐅᔩᒪᐊᖅ
ᖃᕐᕈᐅᓂᖕᓄ ᖃᐅᔨᓴᖅᕐᓯᓂᖅ 22-ᓂ ᐊᕐᖁᑎᓂ GB-ᒥ ᓇᓄᕆᐅᖃᖕᓂᖅᑐᑦ.
ᑐᖕᓕᐃᓕᖕᓄ ᐱᑎᓂᖅᑕᖅᔩᓚᓂᖅ ᐱᓴᐸᖅ ᑖᕿᓇ ᖃᐅᔭᓇᕐᖕᑯᑦ ᖃᐅᔨᓴᖕᑯᑦᓂ,
ᑐᓇᕐᐸᒍᑦ ᖃᔭᕕᖕᓚᖕᓂᖅ ᖃᓄᐊᑎᐅᖅᑯᔂᓂᖅ ᐃᓯᒪᖕᓱᕐᐸᐅᑐᑭᐅᓇᖅᑐᖕ
ᐱᖁᕐᔭᖅᑐᖅᑲᓕᖅᖃᑦ ᓯᐹᓂᖕᒥ ᓇᓄᖕ ᖃᔭᐅᓂᖕᓄ ᖃᐅᔨᓴᖕᓂᑎᐸᕐᓇ.
ᐃᓕᖃᐃᓕᑕᐅᖅᑯᐸᓇᕐᐳᑦ, ᑲᑎᖅᓯᐃᕙᖅᖕᓄ ᑎᑎᖅᔩᒪᑲᖅᖓᖕᑐᖕ ᖁᑎᖕᐸᑯᔲᓇ
ᐱᖁᕐᔭᖅᑕᐅᔩᒪᕈᖕ ᖃᔭᐅᖃᖕᓂᓇᕐᖕᓄ ᖃᐅᔨᓴᖕᓂᓇ. ᑕᐃᒪᐃᖕᓇᖕᓂᐸᖕᓄᖕ, ᑕᒪᓇ
ᓇᓕᒍᕐᔭᖅᐸᖕᕈᖅ 5-7-ᐸᖕᑐᖕᖕ ᐊᕐᖁᑎᖕ 2017-ᒥ ᐱᕐᓄᖃᖅᑕᐅᓕᐅᖅᑐᒥ ᓵᓕᒥ
ᐱᓚᓂᓇᕐᐸᕐᒥ. ᐊᒪᓗᖕᖃᖕᓂᖅᑕᐅᖅ, ᑕᒪᓇ ᖃᓄᐃᑕᐅᓕᐈᖅᑭᐊᓂᖅ ᓄᕐᖃᖅᐸᖕᓂᕐᖕᓄᑦ
ᑎᑎᖅᔩᒪᕈᖕ ᓄᒾᓚᖕᑎᐈᖕᓄᒍ, ᐊᑐᒃᔭᐸᕐᓚᑎᐊᖅᔩᒪᕈᖅ ᓇᐊᖅᑲᑎᕐᖕᑎᕐᖕᓂᖅᑐᓂᕐ
ᖃᐅᔨᓚᕐᐸᐈᑐᒍᑦ ᐃᓂᒥ ᐊᒥᖅᑲᑭᖅᖃᖅᐸᕐᖕᓂᕐᖕᓂᕐ ᓇᓄᐃᑦ ᑖᑯᓇᓂ ᐃᓇᓂ.
ᐱᓕᖃᕐᖕᓂᖕᕈᓚᓂᕐ ᖃᕐᑭᕐᑕᖅᑎᑕᐅᔩᒪᕈᑎᒍᑦ ᑎᑎᖅᖃᑕᐅᔩᒪᕈᐟᖕᓂ ᓇᓄᐃᑦ
ᓄᕐᑕᓂᓇᕗᖃᑎᕐᖕᓂᕐᖕᓄᑦ, ᑲᕈᑎᖃᑎᓂᖅ ᖃᐅᔨᓴᖕᓚᓂᕐᒥ ᖃᐅᔨᖏᕐᓚᓂᕐᖕᓄᑦ
ᐊᒥᖅᖃᖕᑎᐡᕈᑭᖕᓇᓂᕐᖕᓄᑦ GB-ᒥ ᖃᓂᓕᕐᖕᓚᓇᓇ ᓇᓄᕆᐅᖃᖕᓂᖅᑐᑦ (ᓕᖕᓄ
ᑕᕐᓄᔭᖐᐸᕐᖕ ᐃᓚᖕᖕᑕ, GB, ᐊᒪᓗ M'Clintock Channel) ᐃᑲᕈᑐᐈᓇᕈᐅᐃᖃᖕᓂᑭ ᒥᑭᖕᓯᕋᖅᖕᑎᖃᑎᐈᓄ ᐃᕐᖕᓚᓂᕐ ᓇᓇᐅᖅᕐᔭᕐᖕᑐᕐᒥᕐ ᐃᓂᑭᒍᖅᑯᑐᓂᕐ ᓇᓄᕆᐅᖃᖕᓂᖅᑐᑦ
ᖃᕐᕈᐅᓇᓯᕐᓇᐡᓂᕐ ᑕᒃᑯᓂᖕᕆ ᐃᓇᓄᕐ.   ᑭᖕᓕᓇᖅᕈᒥᕐ, ᐱᕕᖕᖕᓂᕐ, ᐊᑐᖅᑕᐅᕐᓂᖅᑐᕐᑦ,
ᐊᐅᓚᕐᓂᕐᔭᕐᒥᕐ ᑐᕐᓂᕐᐸᐃᕐ ᑕᐃᒪᐈᑐᕐᖕᑎᑎᕐᕈᐸᑕ, ᐃᓕᖃᐅᖅᑯᔂᕈᕐᐸᕐᐳᑦ,
ᑲᕈᑎᖃᑎᕐᖃᕐᖕᓇ ᖃᕐᕈᐅᓂᖕᓄᑦ ᐊᕈᓂᖕ ᕐᑭᐊᖕᓚᐡᓇᑐᐈᓇᕐᐸᑐᕐᖕᓕ ᖃᐅᔨᓴᖕᓚᓂᕐᒥᕐ
ᑐᐡᐳᓚᑎᕐᑎᐊᓚᕈᐟᓕᕐ ᑲᕈᔫᓂᕐᕈᐡᖕᓚᐡᑐᓂᕐ ᐊᕈᓄᖕᓄᒍ ᖃᕐᕈᔯᕐᐡᓇᓂᕐᐸᐈᕈᕐ.

1.C) EXECUTIVE SUMMARY INNUINAQTUN

Naunaiyaqni Amigaitpiaqni tapkuat Tariunga Boothia Nannut amigaitni Ilangi Atuqtauyut Aqnallut Anguhallut Titiqni-Angutqiktauyut

Aulapkaiyini Naittuq

Nannut (*Ursus maritimus*) aulatauyut humiliqak Nunavut, Kanata, atuqhugit haviktakhat havagutai pinahuat atuqpiaqni angutauyut ihuaqhihimanit. Taimaa 10nik ukiunik, hilaup aadlangurninnga tamainni Ukiuqtaqtumi aadlanguqtitait nanuit nayugangit aadlatqiiktumik nampanik. Pitariangi naamaktumik nannut amigaitni ilangi ilaunit tapkununga uumatyutit atuqpiaqni piyaunginnalaqnit niqikhanut piqaqnit tahapkununga Inuit, naunaiyainiq naunaiyaut munarinilu naunaiyautit havariyauyut naunairiangi amigaitni ilangi qanuritni aulatauninutlu ihumagini piyakhai. Hamani tuhaqhitautivut tapkuat qanuritni amigaitni naunaiyaut tapkununga nannut nayuqpaktat Tariunga Boothia (GB) havariyauyuq 2015-2017. Nutaat uuktuutingit katitiqtauyut aturhutik mikitqiamik-pittailiniq ihariagiyainnik niqinginnik piiyaqtauniq kapuqtauyut nutqaqtihimaittumik akhuraalukluuniit pilugit nanuit. Qauyihainivut ilalik 2015-2017 uumatyutit naunaiyautit tuhagakhat, uumatitlugit-tiguyauni tuhagakhat katitiqni atuqhugit hanatyuhikhat naunaiyaqni 1998-2000, uumatitlugit-tiguyauni tuhagakhat katitauni pilalirangata 1976-1997, angutauyutlu utiqtitni tuhagakhat tamaitnut pivigiyaini 1976-2017. Qanuritni uumatitlugit-tiguyaunituqungayut-utiqtitni pityuhit ihuaqhihimayut tapkunani Havagut MARK piniraqtai anginiqhamik amigaitni mikhautni tapkuat 1525 (atuqpakni ulamniqni [SE] = 294) pivigiyanut 2015-2017 ayyikkutapyagiya anginiqpaq amigaitni talvani 1998-2000 (1610 [SE = 266] uumani naunaiyaut; 1592 [SE = 361] talvani Taylor et al. [2009]). Anginiqpaq piarait-ukiumun tapkuatlu ukiulgit piarait aktilangi pivigiyanut 2015-2017 tapkuanguyut 1.61 (95% nalungitninut akunit [CI] = 1.51-1.70) tamnalu1.53 (95% CI = 1.41-1.64), tuklirinut, pitquhiqaqungitnit hutqikni tapkuat 1998-2000. Tamna anginiqpaq qaphiuni ukiulgit atuni iniqnit aqnallut pivigiyanut 2015-2017 tamnauyuq 0.36 (95% CI = 0.26-0.47) tapkuat piniraqtai tamna Tariunga

Boothia tatya piruttiaqtut nannut amigaitni ilangi, pigaluaqtitlugu tariup hikua allanguqnia. Una malikhaqmiya naunaiqtavut tapkuat nannut timingi qanuritni (naunaipkutariplugu, uqhuqaqnit) upingami ilagiaqtut akungani pivigiyai 1998-2000 tamnalu 2015-2017. Naunaiqtavut aqnallut anguhallut- ukiungilu-tainit allatqit katitlugit annaumani aktilat (naunaipkutariplugu, ilautitlugit angutat tuqutaunit) puqtutqiyautitlugit mikhautni iniqnit aqnallut (0.95; 95% CI = 0.81-0.99) tapkunangaunganit iniqnit anguhallut (0.85; 95% CI = 0.74-0.92) pivigiyanut 2005-2017. Atulaq turangayuq aktuania naunaiqtauyuq ilagiaqni avikhimaninut amigaitni qnallut talvanga 0.57 talvani 1998-2000 tikitlugu 0.61 talvani 2015-2017. Tamna ayyikkiquqni, taittiaqni amigaitni aglivaliani aktilat kititni atuqhugit kitityutit pinahuginit uuktut mikhauttaqnigut katitlugit annaktut tamnauyuq 0.06 (95% CI = -0.06-0.12) pivigiyanut 2005-2017, piniraqhugit akhut aglivalialaqni. Kihimik, qanuritnivut amigaitni ilangi aktilat pitquhitlu tukiliuqtakhat munarilugit piplugu mikhautnivut amigaitninut pihimani tapkuat "amigaitniqpanguni" (naunaipkutariplugu, ilalgit tamaita nannut atuqtat Tariunga Boothia aulatauvia inaa, ilangi nayuqtat ahii amigaitni ilangiluttauq) mikhautavutlu amigaitni aglivaliani aktilat piyaungittut ahiningartaqnit taphumanga Tariunga Boothia aulatauvia inaa. Tamaitnut, nalvaqtavut piniraiyut tapkuat amigaitni qanuritnit taphuma Tariunga Boothia amigaitni ilangi tatya nakuuyut, pinahuaquigaluaqhuta pukkitqiyat mikhautnit katitninut angutaungittutlu annaumani anguhalluit nannut naunaiyatqikharialgit. Pinahugiyavut tapkuat akuttuni mikhivallilaknilu tariup hikua piqarutaulat nuktiraqninut ikayuqtat tamna Tariunga Boothia amihuni ilangi piplugu ilagiaqni uumatyutit piaraniktaqni. Hilap allanguqnia tamna pityutauniqhaq hivituyumun hivuranauta nannut nayuqpaknitlu avataanut Tariunga Boothia amigaitni ilangi ihuittumik aktualaqni hikuiqpalianginnaqat, taimattauq ilai nannut amigaitni ilangi, tatya naunaqmata. Una naunaiyaut kivgaqtuta aipanik hanatyuhit amigaitni naunaiyaqni tapkunani 22 ukiut tahamunga Tariunga Boothia amigaiti ilangi. Piplugit atuqhimani piyauyut atuqhugu una naunaiyaut qauyihaqnitlu, tuniyavut qaphit aturahuaquni ihumagiyauyukhat parnaiyaititlugit hivunikhami nannut amigaitni naunaiyautit. Aturahuaquyavut katitiqni ilagiarutit tuhagakhat mikhaani qitqani akungani parnakhimayat amigaitni ilangi naunaiyaqni. Uumani piplugu, tamna piya mikhaani 5-7 ukiut talvanga 2017 iniqtauni maniqami havat. Ilagiaqhugu, pigaluaqtitlugit aturahuaquni nuktiraqnit tuhagakhat nutaungittut, huli

pinahuaquyauqpiaqtuq tapkununga amigaitni ilangi ilihimayqnut himmiqtautai nannut akungani inait. Piqangititlugu qangattaqhimayunik takukhautitni tuhagakhat nannut nuktiraqnit, havarinia angiyumik-qauyihaqni naunaiyautit himmiqtautai akungani Tariunga Boothia hanianilu amigaitni ilangi (naunaipkutariplugu, Lancaster Hanikgakhik, Tariunga Boothia, tamnalu M'Clintock Kangikhuakyuk) ikayulat naunairutai ilai naunaqtut piplugu ilikkut amigaitni ilangi tahapkuat inait. Kingulliqpamik, pikpat pivikhait, piqaqni, aulataunilu ihumagiyauyut piyaqaliqturini, aturahuaquyavut havarini amigaitninut angutat hivuranaqni naunaiyaqni tuhaqhittangi ihuaqhihimani angutat puqtunit.

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, resource availability, and species' biology (Gibbs 2008). 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 can increase the probability of meeting management objectives and reduce the severity of potential negative outcomes resulting from mis-specified management interventions (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 apex predators and as such bioaccumulate environmental contaminants (e.g., Derocher et al. 2003, Fisk et al. 2009, McKinney et al. 2009, 2011, Letcher et al. 2010, Routti et al. 2019). As a circumpolar species that depends 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 (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). Currently there is no empirical evidence for declines in global abundance due to sea-ice loss (Regehr et al. 2016). However, some subpopulations have exhibited negative effects resulting from

climate change (e.g., Bromaghin et al. 2015, Lunn et al. 2016) and accurate assessment of global 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 (Rode et al. 2014), 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 the on-going research and monitoring efforts, reliable and updated abundance and demographic information about all subpopulations is still lacking (Obbard et al. 2010, Vongraven et al. 2012). Polar bear research is expensive and logistically challenging, especially for management jurisdictions that oversee multiple subpopulations. 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 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, priorities, and available resource (Hamilton and Derocher 2018). Here we present findings from a 2015 - 2017 study to estimate abundance and evaluate the demographic status of the Gulf of Boothia (GB) polar bear subpopulation.

Gulf of Boothia (GB) is a relatively small polar bear subpopulation area that is entirely managed by Nunavut (Fig. 1). An initial physical mark-recapture study was carried out from 1973 - 78 for the M'Clintock Channel (MC) and the adjacent GB subpopulations, although at the time it did not identify these as separate management units. The total abundance estimate for both areas was 1081 bears (Furnell and Schweinsburg 1984, Urquhart and Schweinsburg 1984). The estimate was known to be biased by non-representative sampling and was subsequently increased to 900 for GB and 900 for MC (Furnell and Schweinsburg 1984, Aars et al. 2006) based on the fact that the entire area was sampled, and the knowledge of Inuit local hunters about polar bear abundance in the broader study area (Derocher et al. 1998, Aars et al. 2006).

The GB and MC subpopulations 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), Inuit knowledge about how local conditions may influence the movements of polar bears (Keith et al. 2005), and genetic analyses (Paetkau et al. 1999, Campagna et al. 2013, Malenfant et al. 2016).

Prior to this study, the most recent population inventory work for GB was completed in 2000, where abundance (mean ± SE) was estimated to be 1592 ± 361 polar bears (Taylor et al., 2009). Based on those results, the population was considered stable or very likely increasing during the early 2000s due to a high intrinsic growth rate and relative low harvest levels (Taylor et al. 1987, 2009, Durner et al. 2018). However, harvest rates for GB increased from an average of 40 bears per year (with a Total Allowable Harvest [TAH] of 41) as reported by Taylor et al. (2009), to 62 bears per year (22 females and 40 males on average annually with a TAH of 74 starting in 2004/2005; Government of Nunavut (GN), unpublished data), between 2005 and 2017 (GN, unpublished data). How this change in harvest may have affected the GB subpopulation abundance and status is unclear.

Polar bears in Nunavut are managed through a co-management system and memoranda of understanding (MOU) between each community's Hunters and Trappers Association and the territorial government[1]. These MOUs lay out harvest, management and research aspects for each polar bear subpopulation. Under the existing 2005 MOU, the GN committed to begin a new population study for GB in 2015. The new study had the objective to estimate the current subpopulation size and composition, and to compare these results to the former study. In addition, we sought to obtain data that would provide estimates on survival and reproductive parameters that can be used in population viability analyses and a quantitative harvest risk assessment. Lastly, by implementing a research method that was minimally-invasive and supported by local communities and stakeholders, we sought to evaluate whether genetic mark-recapture

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[1] As of September 2019 the Nunavut Polar Bear Co-Management Plan is replacing the Memoranda of Understanding.

can be compared with traditional capture mark recapture studies previously done in GB in order to establish longer term trends for population monitoring (Vongraven and Peacock 2011, Vongraven et al. 2012).

3. STUDY AREA

The GB polar bear subpopulation lies entirely within Nunavut and encompasses an area of approximately 67 000 km^2 (excluding land; Taylor et al. 2001, 2009, Barber and Iacozza 2004, Hamilton and Derocher 2018; Fig. 1). The management unit is bound by the Boothia Peninsula to the west, and Brodeur Peninsula to the east. The geography of the study area is described in Schweinsburg et al. (1981). The current management boundary is mainly based on telemetry data for adult female bears that were fitted with radio-collars, tag returns from harvested bears (Schweinsburg et al. 1982, Bethke et al. 1996, Taylor et al. 2001), and genetic analyses (Campagna et al. 2013, Malenfant et al. 2016). Validity of the current boundary has been questioned by Inuit local knowledge (Keith et al. 2005).

Sea ice generally begins to form in early October and persists until July or August in most areas of GB (Schweinsburg et al. 1981). The most southerly area of GB, namely Committee Bay, remains mostly ice-covered throughout the year (Barber and Iacozza 2004). The presence of various ice types such as mobile, multi-year rubble, and first-year ice creates diverse seal habitat across GB (Barber and Iacozza 2004). Recent sea ice and climate data analyses indicate that the Arctic sea ice quality and abundance has changed during the past 30 years and that in most polar bear subpopulations, the sea ice melts sooner and forms later than in the 1980s (Stroeve et al. 2012, Stern and Laidre 2016, Regehr et al. 2016, Environment and Climate Change Canada 2019). Currently, sea ice persists across GB to various degrees throughout the year, but it is predicted that GB may be ice-free for 5 months each year by the late 21st century (Hamilton et al. 2014).

4. METHODS

Sampling – field collections

Our 2015 - 2017 study design was informed by the previous physical mark-recapture study conducted in GB 1998 - 2000 (Taylor et al. 2009; Fig. 2), although our study did not involve the immobilization and physical handling of bears. Inuit co-management partners in Nunavut expressed 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 generally low densities of bears on the sea ice and the vast study area, genetic mark-recapture was selected 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)). From 2015 - 2017, our biopsy darting sampling sessions occurred between April to late-May each year where we searched the sea ice and near-shore areas for bears across the entire study area. We allocated approximately 100 hours of helicopter time for each field season to search for bears. 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, which are typical methods for individual identification during live-capture studies (see section "Genetic analyses"). 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 from each polar bear harvest so that age, gender and various other variables can be used in ecological and demographic assessments (Nunavut Wildlife Act, SNu 2003).

Search areas were initially discussed with hunters and local Hunters' and Trappers' Associations during pre-study consultations to gain insight about sea-ice conditions and bear distribution. We also took past capture locations (Taylor et al. 2009) into account when searching the sea ice, adjacent coastal areas, and small islands of our study area (Figs. 2b and 3).

Searches for bears were conducted at approximately 100 - 120 m above sea level, and at average speeds between 120 - 150 km per hour. 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 reflected that with further (i.e., 11 - 16 km) or nearer spacing (i.e., 7 - 10 km), respectively. In that fashion, we were able to cover large sections of the study area efficiently (Fig. 3). We decided to fly our survey transects from east to west and vice versa whenever possible, and to be perpendicular to suspected density gradients based on local knowledge, past capture and hunter-provided 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 using a biopsy dart. All bears except cubs-of-the-year (C0s) were darted in the rump area from an approximate distance (or altitude) of 3 - 7 m. C0s in early spring are still small and easily confused (Atkinson and Ramsay 1995, Robbins et al. 2012), and therefore were not darted 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 can be retrieved without handling a bear. The effectiveness of these darts for sampling polar bears has been previously demonstrated (Pagano et al. 2014, GN, unpublished data and reports, SWG 2016). The darts are quick and easy to use and require less pursuit

of bears than live-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 darting 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.

In addition to collecting the biopsy sample, we recorded the date, time and location of each observed bear (or group of bears), body condition based on visual assessment using a standardized fat index (e.g., 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 (e.g., C0, yearling (C1), 2-year old, subadult [approx. 2 - 4 years], adult [approx. ≥ 5 years]) and estimated gender. Both field age-class and gender estimated included a confidence qualifier (i.e., a = high confidence; b = low confidence). Field age-class and gender throughout this project were assessed remotely from the helicopter at altitudes between 3 - 7 m by four experienced observers. When we encountered mothers and their dependent young, we distinguished C0s, C1s, and 2-year old offspring based on their size relative to their mother and physical features (e.g., blood or fecal/urine stains, scars) 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 estimate gender and age-class. Genetic microsatellite analysis was used later to confirm the gender of each sampled bear (see section Genetic analysis).

When field age class and gender 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 (e.g., confidence classifier "b") then we also noted what this bear could be as alternative – in this case "maybe a young subadult male". When genetics confirmed the field estimate of sex, we assessed the identity of the bear as recorded initially. If the genetics returned a different sex, we reviewed our notes and concluded that the bear, in this example, must have been a young subadult male. 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 affect detection (e.g., sea ice type: flat, intermediate, rough multi-year ice).

Our work combined data collected during the genetic biopsy sampling sessions from 2015 - 2017, data from the previous capture-mark-recapture study conducted between 1998 - 2000, sporadic live-captures conducted from 1976 - 1997, and harvest recovery data for the entire period 1976 - 2017 (Peacock et al. 2012).

Sampling – recovering previously marked bears through harvest

To detect the recovery of previously individually identified 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) by hunters, small muscle tissue samples were collected from all bears harvested in GB and surrounding subpopulations such as MC, Lancaster Sound (LS) and Foxe Basin (FB) throughout the duration of the current biopsy darting study (i.e., April 2015 - May 2017). Polar bear harvesting occurs throughout the year and these samples were stored in 2 ml cryovials (ThermoScientific, Nalgene long-term storage cryogenic tubes) at - 20˚C after submission to our laboratory until sample preparation and analyses.

Sampling - recovered bears from past population 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 (e.g., ≤ 34 years of age) in 2015 for genetic analyses. Samples (e.g., 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. Briefly, a lentil-size piece of skin (~ 1 - 1.5 mm thick) or tissue was obtained from either the biopsy sample, the ear plug, or the muscle tissue using a scalpel blade (# 20) then 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, 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).

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, 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 these matches. Once the genotyping and error-checking was complete, we defined an individual for each unique eight locus genotype.

Sea-ice metrics

Other population studies have identified relationships between the spatial and temporal availability of sea ice and demographic parameters for polar bears (Regehr et al. 2007, Rode et al. 2012, Laidre et al. 2020). March and September mean ice concentrations were calculated for the entire GB area for each day sea-ice data were available and then averaged across 1979 - 2016 (Environment and Climate Change Canada 2018). We calculated the number of days between the sea ice retreat and sea ice advance in calendar year *t* using the transition dates when ice concentration dropped below, and exceeded, respectively, the midway point of sea ice concentration between the March and September mean (Environment and Climate Change Canada 2018). For the GB area, this transition sea-ice concentration was 63% (Environment and Climate Change Canada 2018). We describe the annual interval that sea-ice concentration was below the transition threshold as the "low-ice days" (Fig. 4). To evaluate the potential relationships between sea ice and the status of GB polar bears, we analyzed several metrics (e.g., body condition, recruitment, and survival) of bears in year *t* as a function of the duration of low-ice days in year *t*-1.

Body Condition Score

We compiled body condition score (BCS) data from two distinct time periods of mark-recapture population sampling in GB. 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 (1998 - 2000) or aerial observation during biopsy sampling (2015 - 2017). All BCS observations occurred in April and May. Sex, age, and reproductive classes were assigned during physical handing during 1998 - 2000 and ages were determined based on previous capture history, known birth year, or from tooth analysis (Calvert and Ramsay 1998). During the biopsy sampling period, classification was done at approximately 3 - 7 m above the ground with sex verified by subsequent genetic analysis (SWG 2016). Observers who participated in classifying age class and sex during biopsy sampling had either participated in both sampling 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) to follow recommended monitoring schemes (Stirling et al. 2008, Vongraven et al. 2012) and facilitate comparison with other 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). We excluded within-year observations of the same individual but retained observations of the same individual in different years.

We modeled BCS using ordinal logistic regression (Venables and Ripley 2002) and included *period* as an indicator of sampling period (early = 1998 - 2000 or late = 2015 - 2017). Reproductive status, age, and sex were combined into the four-level categorical variable *reproclass* (ADM = adult male, ADFI = independent adult female, ADFWO = adult female with offspring, and SUB = subadults of both sexes), and sampling day of year (*jul_cap_day)* were included as a continuous covariate to reflect the amount of time bears had on their preferred sea ice hunting platform before being sampled in year *t*. The sampling periods 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 low-ice days ($icetm1_{t-1}$) 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 be related to BCS (Rode et al. 2012, SWG 2016, Laidre et al. 2020). Finally, given our interest in evaluating whether different reproductive classes and genders 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 three-way interaction between *reproclass, jul_cap_day,* and *period.* Once the global model was selected, we performed a backwards and forwards model comparison (stepAIC; Package MASS in the R programming language [R Core Team 2019]) to obtain the best-supported final model (ΔAIC < 2) (Table 1). We performed Lipsitz and Hosmer-Lemeshow tests to evaluate fit of the global ordinal regression model ($p > 0.1$; Fagerland and Hosmer 2017). Best-supported model covariates were considered significant at $p < 0.05$ (Wald X^2 tests) and predicted probabilities for each BCS class were calculated based on the suite of final-model covariates.

Reproduction

We evaluated reproductive indices for polar bears in GB using data from physical captures 1998 - 2000 and biopsy sampling 2015 - 2017. We used reproductive metrics that have been identified as important for monitoring polar bears (Vongraven et al. 2012). First, we C0 and C1 litter size as a function of biological, environmental, and temporal factors using logistic regression. We considered litter size (*ls*) for adult female *i* in year *t* to be a binary response variable (i.e., ls_{it} = 1 or 2). Analyses for C0 and C1 litters were performed separately using a three-step modeling approach, although we note that the C0 and C1 litter size data were not independent due to potential repeated measures and correlations (i.e., C1 litter size in year *t* is likely a function of C0 litter size

in year t-1). We created a general model that included the main hypothesized sources of variation in the data. General models were simple due to small sample size. To ensure the general model was a suitable starting point for model selection, we evaluated goodness-of-fit (GOF) using Hosmer and Lemeshow tests (Hosmer et al. 2013). Second, we developed a candidate model set representing all combinations of main effects and interaction terms in the general model, with a marginality constraint to ensure that interactions were only included if the corresponding main effects were included. Third, we performed model selection using Akaike's Information Criterion adjusted for small sample size (AIC_c) and then estimated model-averaged parameters for all models with $\Delta AIC_c < 4$ (Burnham and Anderson 2002). Modeling was performed in the R programming language version 3.5.2 (R Development Core Team 2016) using package *MuMIn* (Bartón 2018) for multi-model inference.

The general model for C0 litter size was $ls_{it} = \beta_0 + \beta_1\ period_{it} + \beta_2\ icetm1_{it} + \beta_3\ BCS_{it} + \beta_4\ month_{it} + \beta_5\ period_{it} \times month_{it}$, where $period_{it}$ is a two-level factor indicating whether the observation of adult female i in year t was in the early or late period (1998 - 2000 and 2015 - 2017, respectively); $icetm1_{it}$ is the duration of the low-ice days in calendar year t-1 (see section Sea-ice Metric) for a polar bear observed in calendar year t; BCS_{it} is a three-level factor representing the body condition score of the adult female at the time of observation (see section Body Condition Score); $month_{it}$ is a two-level factor indicating whether a bear was observed in April or May; and $period_{it} \times month_{it}$ is an interaction term allowing the month effect to potentially differ between the early and late periods (e.g., because within-year temporal variation in litter size could change due to changes in sea-ice conditions, den emergence date, etc.). We hypothesized that litter size would be negatively correlated with $icetm1$ (Laidre et al. 2020), positively correlated with BCS (Derocher and Stirling 1998), and negatively correlated with $month$ because observations later in the spring reflected additional time in which cubs could die.

The general model for C1 litter size was $ls_{it} = \beta_0 + \beta_1\ period_{it} + \beta_2\ icetm1_{it} + \beta_3\ BCS_{it}$, where definitions of the predictor variables are the same as in the model for C0s.

We did not include the predictor $month_{it}$ because individual C1 survival is generally high (e.g., Regehr et al. 2017) and we did not expect litter size to change between April and May.

After evaluating patterns in litter size, we calculated the mean number of dependent young (C0 or C1) per adult female and evaluated differences between time periods. We also evaluated litter production rate, defined as the proportion of adult females that are available to breed in year t that produce a litter of C0 in year $t+1$ (Taylor et al. 1987). These metrics have been used as indices of productivity for other polar bear subpopulations (e.g., Peacock et al. 2013, Regehr et al. 2015). We quantified uncertainty using a nonparametric bootstrap procedure with 1,000 iterations during which observations of individual polar bears were resampled with replacement and the three reproductive metrics were calculated from the resampled data.

Survival

We used the Burnham capture-recapture model (Burnham 1993) in Program MARK (Cooch and White 2019) to analyze live-observation and dead-recovery data for the GB subpopulation. Live observations consisted of physical captures during which bears were assigned an individual identification number, or the identity of a previously captured bear was recorded; and biopsy sampling during which individual identification was determined from genetic analysis of a tissue sample (see sections above about recovering samples of bears through harvest and from the previous study). Live observations were conducted under random sampling protocols that attempted to search the entire area within the GB subpopulation boundary in 1998 - 2000 (physical captures) and 2015 - 2017 (biopsy sampling). Additionally, bears were physically captured and released each year 1976 - 1978, and sporadically during the period 1979 - 1997. Because research conducted from 1976 - 1997 did not follow a sampling protocol designed to evaluate demography, we included initial captures from this period but did not include recaptures of previously marked bears. This approach has been used in other analyses (e.g., Taylor et al. 2009) to increase the number of marked bears without

introducing heterogeneity into recapture probabilities, which can result in biased parameter estimates (Peňaloza et al. 2014). Because recaptures were excluded or did not occur in some years, within the Burnham model we fixed recapture probability to 0 in 1976 - 1997 and 2001 - 2014. Throughout the entire study period 1976 - 2017, dead-recovery data were obtained from hunter reports of research-marked bears and genetic analysis of tissue samples from bears that were harvested.

The Burnham model is a common choice for estimating survival and abundance of polar bears (SWG 2016). Parameters in the model are survival (S; the probability of surviving interval t to t+1), recapture probability (p; the probability of re-observing a live marked animal), dead reporting probability (r; the probability that an animal which dies is killed by humans and reported to authorities), and fidelity (F; the probability that an animal does not permanently emigrate from the sampling area and remains available for live observation in future years). We limited our analyses to bears age ≥ 1 year (i.e., C1s and older) because in the 2010s most C0s were not biopsy darted or individually identified.

We developed a candidate model set based on combinations of parameter-specific submodels, with the structure of each submodel informed by hypotheses about polar bear biology and study design. We considered 16 submodels for S (Table 2). The temporal factor *year* allowed survival to differ between 1976 - 2004 and 2005 - 2017. We chose these year blocks to evaluate the potential influence of habitat changes in the past decade (Environment and Climate Change Canada 2018) and because total allowable harvest (TAH) for the GB subpopulation was increased in 2004 (see section Introduction). The two-level factors *sex* (female vs. male) and *sub* (C1s and subadults [2 - 4 year] vs. adults [age ≥ 5 year]) were included to allow sex- and age-specific variation in survival (e.g., Regehr et al. 2007). The covariate *icetm1*, calculated the same as for reproductive analyses, was included to evaluate the hypothesis that interannual variation in survival was related to sea-ice availability in the previous year. We considered five submodels for *r* that included *sex* and *year* to reflect sex-specific harvest and potential changes in harvest mortality associated with changes in harvest

level. The four submodels for *p* included *sex* to allow potential variation in recapture probability resulting from sex-specific habitat selection or movement patterns (Laidre et al. 2013), and *year* to accommodate different levels of sampling effort in the 1990s and 2010s. We did not include a submodel with annual variation in *p* because sample sizes were similar within each three-year block of intensive capture-recapture research. The four submodels for *F* included *sex* and *year*. Unlike Taylor et al. (2009), we estimated *F* rather than fixing it to 1 because bears captured in the GB management unit have been harvested in adjacent subpopulations, suggesting some degree of permanent emigration (see section Discussion - Abundance). Each submodel was constructed as a linear function, on the logit scale, of the various factors, covariates, and interaction terms discussed above. We fitted all possible combinations of the parameter-specific submodels in Program MARK (Cooch and White 2019) accessed through the R programming environment (R Core Team 2019) using the package RMark (Laake 2013).

We performed model selection and multimodel inference using $QAIC_c$ (Burnham and Anderson 2002). We used the overdispersion factor $\hat{c}$ = 1.2, calculated as the ratio of live observations of dependent cubs (i.e., C1s and two-year-old cubs still accompanying their mothers) to total live observations (Taylor et al. 2009). For validation, we derived a separate estimate of $\hat{c}$ using the parametric bootstrap procedure in Program MARK (Cooch and White 2019) with the general model *S(year+sex+year:sex)r(year+sex+year:sex)p(year+sex)F(sex)*, where "+" represents an additive effect and ":" represents an interaction. The bootstrap estimate of $\hat{c}$ was 1.2, suggesting that our empirical estimate adequately reflected extrabinomial variation in the data. Model-averaged parameter estimates were derived from all candidate models with $\Delta QAIC_c$ < 4. Our estimates of *S* reflected harvest mortality, so we derived estimates of un-harvested survival as $S^* = S + r \times (1 - S)$ (Peacock et al. 2013) and estimated variance via the delta method (Taylor et al. 2008). This equation assumes that harvest of all marked bears is reported, and that harvest mortality is additive (i.e., that no harvested bears would otherwise have died during a given interval).

Abundance

We used Horvitz-Thompson type estimators (McDonald and Amstrup 2001) to derive abundances in year t as $\widehat{N}_t = n_t / \hat{p}_t$, where n_t is the number of individually identified animals observed alive in year t, and $\hat{p}_t$ is a model-averaged estimate of recapture probability in year t. To estimate abundance of bears age ≥ 1 year we stratified the subpopulation by sex and summed the female and male estimates, which was necessary to accommodate sex effects in recapture probability. Finally, we adjusted annual abundances to include approximate numbers of C0s by adding the product $(\widehat{N}_t^{AFC0} \times \overline{ls}^{C0})$, where $\widehat{N}_t^{AFC0}$ is the estimated number of adult females with C0 litters in year t, and $\overline{ls}^{C0}$ is overall mean C0 litter size. We used the delta method to construct variance estimates for annual estimates of total N and for average estimates of total N over several years. In doing so, we assumed that estimates of recapture probability and C0 litter size were independent. Note that abundance estimates from a capture-recapture framework that allows permanent emigration, but not temporary emigration, may not represent the number of animals within the sampling area at a given point in time. Specifically, abundance estimates from the current study represent the "superpopulation", defined as the group of animals that are alive and have a non-negligible probability of occurring within the sampling area, regardless of their actual location at a particular time. In other words, the superpopulation estimate in year t reflects temporary emigrants (i.e., animals that are outside of the GB management unit in year t but may return in future years).

Population growth

We used estimates of S and S^* from live-recapture dead-recovery modeling, together with estimates of litter production rate and C0 litter size, to estimate intrinsic population growth rate (gr) using a 10-stage matrix-projection model based on the life history of polar bears (Regehr et al. 2017). Because we did not estimate C0 survival in the current study, we used the mean estimate of 0.889 (SE = 0.179) for the period 1976 - 2000 from Taylor et al. (2009) for all matrix calculations. We estimated var(gr) by generating

10,000 correlated samples of the input vital rates using the model-averaged variance-covariance matrix for sex- and age-specific estimates of survival. We assumed that the correlation structure for C0 survival was the same as for subadults, that litter production rate and C0 litter size had a correlation coefficient of 1, and that there was no correlation between survival and reproductive parameters. Estimates of *gr* represent asymptotic intrinsic growth rate at a stable stage distribution.

5. RESULTS

General overview

During research operations in 2015 - 2017, we spent an average of 103 hours of flying in April and May each year in search of polar bears across the sea ice, with an average distance flown per year of about 12,200 km (Table 3, Figs. 2 and 3). The number of bears encountered during each survey season was similar, with a mean of 170 observed bears per field season.

The GB study area is vast and consists of differing ice types (Barber and Iacozza 2004). The distribution of bears during the 2015 - 2017 study appeared to be more uniform across the study area as compared to 1998 - 2000 when bears were encountered in higher concentrations east of the Boothia Peninsula and near the west shore of Melville Peninsula (Figs. 1 - 3). Moreover, there appeared to be no bear encounters directly north of Committee Bay during the 1998 - 2000 study, in contrast to our recent observations. During both studies no bears were encountered in the lower section of Committee Bay (Fig. 2).

Samples examined

We collected a total of 406 biopsy samples during research operations in 2015 - 2017. Of these, 397 (97.8%) contained sufficient material for genetic analysis. We

identified 10 GB bears that were previously captured during the 1998 - 2000 study (Taylor et al. 2009), and 1 LS bear that was 22 years old in 2017 when it was sampled. We also identified 7 individuals that were previously sampled during the MC study between 2014 - 2016. Overall, 324 individual bears were identified from these field samples. Some bears were resampled within the same season: 18 bears were sampled twice, 2 bears were sampled three times, and 1 bear was sampled four times (representing 5% of all successful samples). Re-sampling of the same individual within the same field season was low and likely occurred because weather prevented coverage of a large area within a short time frame, allowing bears to move over longer distances. Biopsy sampling leaves no visible marks on the individual animal as is the case with traditional mark-recapture studies (e.g., Peacock et al. 2013) thus it is impossible to avoid some re-sampling.

Through the harvest sampling program, we submitted 1704 samples between 2005 - 2017 from GB and neighboring subpopulations (338 GB, 701 FB, 402 LS, 47 MC, and 216 with unknown subpopulation) for genetic analyses. Twenty-five bears from the biopsy sampling sessions were harvested and recovered, as well as 8 previously marked bears from the 1998 - 2000 study. Those 8 bears were recovered in GB (6), MC (1) and LS (1). The 6 recovered bears in GB were identified through genetic testing because no ear tags and tattoos were reported.

Field sampling activities

Biopsy sampling activities on the sea ice went very well. The darts do not leave a mark when bears are darted in the rump, and most bears do not react to the impact of the dart. Many of the adult males move very slowly away once darted, if at all. The colored flagging tape attached to the end of the dart makes dart retrieval easy and quick.

During our survey flights, additional observers besides the pilot and biologist were on board the helicopter. In order to safely maneuver during darting, some observers had to be safely dropped off once a bear was seen to reduce weight, but

before the darting activities began. It took the crew, on average, 4.3 min (± SE; 0.19; range: 2 - 8 min; n = 62) from the time a bear was observed for the first time (e.g., at times > 1 km from the helicopter) and when the additional observer was picked up again. The direct darting activities involving the safe approach of the bear, darting the bear, and dart retrieval took an average of 2.0 min (± SE; 0.11; range: 1 - 5 min; n = 62; GN, unpublished data).

Body condition score

Body condition scores were higher between 2015 - 2017 compared to 1998 - 2000 (n = 626; χ^2 = 5.5, *p* = 0.02; Fig. 5, Table 4). This was reflected in a decrease in the proportion of bears in poor condition (P_{poor}) and an increase in the proportions of bears in average and good condition (i.e., P_{poor} = 0.31 for early period vs P_{poor} = 0.07 for the late period; Fig. 5; Table 4). Adult females with offspring (P_{poor} = 0.28) and subadults (P_{poor} = 0.26) were more likely to be in poor body condition compared to other age and reproductive classes (mean P_{poor} for ADFI and ADM = 0.11; χ^2 = 11.4, *p* < 0.01, Fig. 6). For females with dependent offspring, increasing amounts of time on the ice before being sampled (*jul_cap_day*) was associated with higher BCS (χ^2 = 9.0, *p* < 0.05).

In the early period, bears were more likely to be in poor condition as $icetm_{t-1}$ increased (*icetm* = 70 d: $P_{poor\ early\ period}$ = 0.24 and *icetm* = 104 d: $P_{poor\ early\ period}$ = 0.39; χ^2 = 13.5, *P* < 0.001). The opposite was true in the late period; the probability of being in poor condition decreased as $icetm_{t-1}$ increased (*icetm* = 70 d: $P_{poor\ late\ period}$ = 0.12 and *icetm* = 104 d: $P_{poor\ late\ period}$ = 0.03).

Reproduction

We observed 99 adult females with C0 litters during intensive capture-recapture studies conducted in 1998 - 2000 and 2015 - 2017 (Table 5). The general model for C0 litter size provided an adequate fit to the data (Hosmer and Lemeshow test: X^2 = 6.91, *df* = 8, *P* = 0.55). The candidate model set included eight models with ΔAICc < 4, from which

model-averaged parameter estimates were derived (Table 6). Low importance scores (i.e., sums of normalized AIC_c weights for models that included a variable) indicated a lack of support for variation in C0 litter size as a function of our proposed predictor variables (Table 6). The low-AIC_c model included one parameter (i.e., intercept only; β = 0.43, SE = 0.21, P = 0.04). Overall mean C0 litter size was 1.61 (95% CI = 1.51 - 1.70).

We observed 80 adult females with C1 litters during intensive capture-recapture studies conducted 1998 - 2000 and 2015 - 2017 (Table 5). The general model for C1 litter size provided an adequate fit to the data (Hosmer and Lemeshow test: X^2 = 5.96, df = 7, P = 0.54). The candidate model set included five models with ΔAIC_c < 4, from which model-averaged parameter estimates were derived (Table 7). Low importance scores indicated a lack of support for variation in C1 litter size as a function of our proposed predictor variables (Table 7). The low-AIC_c model included one parameter (i.e., intercept only; β = 0.10, SE = 0.23, P = 0.65). Overall mean C1 litter size was 1.53 (95% CI = 1.41 - 1.64).

The other reproductive metrics for GB polar bears were similar, or slightly lower, in 2015 - 2017 compared to 1998 - 2000. Mean number of C0s per adult female was 0.51 (95% CI = 0.39 - 0.64) for the 1990s and 0.43 (95% CI = 0.32 - 0.44) for the 2010s, which corresponds to a probability of 0.85 that values were smaller in the 2010s. Mean number of C1s per adult female was 0.37 (95% CI = 0.27 - 0.48) for the 1990s and 0.36 (95% CI = 0.26 - 0.47) for the 2010s, which corresponds to a probability of 0.54 that values were smaller in the 2010s. Mean litter production rate was 0.76 (95% CI = 0.48 - 1.0) for the 1990s and 0.64 (95% CI = 0.41 - 0.98) for the 2010s, which corresponds to a probability of 0.71 that values were smaller in the 2010s. Note that the ratio estimator we used to calculate litter production rate was different from the estimator used by Taylor et al. (2009), which required assumptions about litter loss and population growth rate.

Demographic analyses

Survival - The capture-recapture data contained 987 live observations of individually identified polar bears and 139 dead recoveries of research-marked bears during the period 1976 - 2017 (Table 8). The candidate model set included 1280 live-recapture and dead-recovery models representing combinations of the parameter-specific submodels. Of these, 104 models had $\Delta QAIC_c < 4$, indicating relatively high model-selection uncertainty. To evaluate the explanatory power of the various factors, covariates, and interaction terms in each parameter-specific submodel, we calculated importance scores defined as the sum of $QAIC_c$ weights for all submodels containing a given term (Table 9). Importance scores for survival (*S*) suggested strong support for a sex effect and for a step change between the year blocks 1976 - 2004 and 2005 - 2017, relatively weak support for an age effect, and little or no support for interannual variation in survival in relation to our sea-ice metric. Importance scores for recovery probability (*r*) provided weak to moderate support for a sex effect and a step change between year blocks. Finally, importance scores for recapture probability (*p*) and site fidelity (*F*) provided little or no support for sex or temporal effects.

Our model-averaged parameter estimates were consistent with patterns that would be expected based on the importance scores for the various terms (Table 10). Point estimates of un-harvested survival (*S**) increased for females, and decreased for males, between the year blocks 1976 - 2004 and 2005 - 2017. Point estimates for *r* decreased slightly for females and increased slightly for males. Point estimates of *F* ranged between 0.93 - 0.99, suggesting relatively high fidelity to the GB management unit. Due to sampling uncertainty and potential process variation, no temporal changes in parameter estimates were statistically significant at an alpha level of 0.05.

Abundance - Mean model-averaged estimates of total subpopulation abundance, including numbers of C0s, were 1610 (SE = 266) for 1998 - 2000 and 1525 (SE = 294, 95% CI = 949 - 2101) for 2015 - 2017. Based on a randomization procedure, this corresponds to a probability of 0.57 that abundance of the GB subpopulation was

approximately stable or increasing (subjectively defined as $N_{2015-2017} \geq 0.9 \times N_{1998-2000}$), and a probability of 0.43 that abundance was declining (defined as $N_{2015-2017} < 0.9 \times N_{1998-2000}$). Our estimate of mean abundance for 1998 - 2000 was very close to the estimate of 1592 (SE = 361) for the same period from Taylor et al. (2009).

Population Growth – The time-constant estimate of asymptotic intrinsic population growth rate (gr) for the period 2005 - 2017, calculated using estimates of total survival (S), was 0.06 (95% CI = -0.06 - 0.12). The estimate of un-harvested growth rate for the period 2005 - 2017 was gr = 0.07 (95% CI = -0.05 - 0.13). This suggests a strong potential for growth in the absence of harvest, although precision was low. For the period 1976 - 2004, estimates of harvested and un-harvested gr were 0.03 (95% CI = -0.07 - 0.09) and 0.05 (95% CI = -0.04 - 0.10), respectively. Although comparison is complicated by different model structures and datasets, these values are similar to the corresponding point estimates of gr = 0.02 and 0.06 for the period 1976 - 2000 reported in Taylor et al. (2009).

6. DISCUSSION

General

The GB study area experienced drastic sea ice changes over the past decades (Barber and Iacozza 2004, Stern and Laidre 2016, Environment and Climate Change Canada 2018). The quantity of multi-year sea ice has declined across the Canadian Archipelago (Mudryk et al. 2018, Perovich et al. 2018, Richter-Menge et al. 2018) and the fall freeze and spring thaw cycles in GB changed significantly, extending the period between sea-ice retreat and sea-ice advance by 16 days per decade (Stern and Laidre 2016). Moreover, the mean summer sea-ice concentration (June to October) has been decreasing by 9% per decade (Stern and Laidre 2016). As recently as the 1980's, the GB region was characterized by 40 - 50% multi-year ice during the summer, but this amount has declined to less than 10% between 2011 and now (Environment and

Climate Change Canada 2018) and the shift is predicted to continue (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. Annual sea ice 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 detrimental to the persistence of polar bears, in the short term, it may have beneficial effects in some parts of the high Arctic since many of the observed sea ice changes have been associated with greater marine productivity (Derocher et al. 2004, Häder et al. 2014, Frey et al. 2018).

Abundance

Our estimate of mean abundance for the period 1998 - 2000 was 1610 (SE = 266), which is very similar to the estimate of 1592 (SE = 361) for the same period from Taylor et al. (2009). The new mean abundance estimate of 1525 (SE = 294) for the period 2015 - 2017 corresponds to a probability of approximately 0.57 that the GB subpopulation has remained approximately stable or increased despite observed sea-ice changes. We suggest that abundance estimates from 1998 - 2000 and 2015 - 2017 are likely an accurate portrayal of trends in abundance given the consistent methodology between the intensive capture-recapture efforts. Taylor et al. (2009) suggested that the subpopulation could sustain a quota increase from 40 to 74 bears per year which was instituted in 2004/2005. The 74-bear quota was rarely filled over the past 14 years with an average of 62 bears per year (22 females and 40 males) removed from the subpopulation. The sex ratio of removed bears was 64.3% male in keeping with the 2:1 sex selective harvest management system in place in Nunavut during that time (range: 56.7 - 72.1% male for the 2004/2005 – 2016/2017 harvest seasons; GN, unpublished data).

The mean point estimate of the proportion of females among independent polar bears (i.e., age ≥ 2 years) increased from 0.57 for the period 1998 - 2000 to 0.61 for the period 2015 - 2017. This appears consistent with the estimates of harvest recovery probability and the estimated differences in total, and un-harvested, survival between females and males. This finding may suggest that the selective harvest of polar bears at a 2:1 male-to-female ratio has resulted in a gradual depletion of adult males in the subpopulation, which is consistent with model-based predictions of declining male numbers under a sex-selective harvest (McLoughlin et al. 2005, Taylor et al. 2008, Regehr et al. 2015). We suggest that this effect could be mitigated by lowering the TAH while maintaining a sex-selective harvest. Alternatively, maintaining the current TAH, but switching to a 1:1 sex ratio for several years could also mitigate the gradual depletion of males but would increase the risks of overharvest given that adult female bears are the most important contributors to population growth (Eberhardt 2002, Hunter et al. 2010). We recommend that a more thorough harvest risk assessment be conducted to further investigate this and other issues related to the sustainability of current removal levels from the GB subpopulation (e.g., change in carrying capacity and environment over time; Regehr et al. 2017).

The GB study area has an estimated density of 8.9 bears per 1000 km^2 based on the current abundance estimate, which is the highest, currently known, density of polar bears within the subpopulation boundaries recognized by the IUCN Polar Bear Specialist Group (Durner et al. 2018). It is more than 5 times the median density of 14 subpopulations for which abundance estimates exist (Hamilton and Derocher 2018). It is also important to note that our estimates of abundance from the current study, as well as from the past study (Taylor et al. 2009), represent the "superpopulation". A superpopulation is defined as all the animals with a chance (non-negligible probability) of occurring within the GB management boundary, regardless of where the animals were located at any given sampling occasion (e.g., Schwarz and Anarson 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 directly estimate temporary emigration from the sampling area (Cooch and White 2019) because our sample sizes

were not sufficiently large to do so, and there are no recent radio-telemetry 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 GB is likely occurring (Fig. 7). Therefore, our estimates of abundance are likely larger than the actual number of animals within the GB 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.

Population Growth

Our estimates of the population growth rate (*gr*) for the period 2005 - 2017 based on total survival (gr = 0.06) and un-harvested survival (gr = 0.07) for the 2010s are high for polar bears, suggesting strong capacity for growth. Our estimates of *gr* for the 1990s were similar to estimates from Taylor et al. (2009), although a direct comparison is complicated by statistical uncertainty and different modeling structures and datasets. Note that our estimates of *gr* for the 1990s had more statistical uncertainty than that of Taylor et al. (2009) because we accounted for covariance among demographic parameters, whereas it appears that Taylor et al. (2009) considered variation in the different demographic parameters to be independent.

The high estimates of *gr* from this study should be interpreted with caution because they are based on estimates of total survival. Therefore, they reflect the potential for biological population growth but not necessarily the trend in the numbers of polar bears that remain within the GB subpopulation boundary. Indeed, when the harvested population growth rate for the period 2005 - 2017 is recalculated using

estimates of apparent survival (i.e., the probability of remaining alive and not permanently emigrating from the GB management unit) the point estimate is negative (gr = -0.024; i.e., suggesting that the number of bears within the GB subpopulation boundary may be decreasing). Direct interpretation is complicated by statistical uncertainty (e.g., the coefficient of variation for the estimate of gr based on total survival was 0.79). However, this may suggest that emigration from the GB region is one explanation for the apparently contradictory findings of (1) a lower point estimate of abundance for 2015 - 2017 compared to 1998 - 2000 and (2) high point estimates of gr for 2005 - 2017 that suggest the GB subpopulation was growing during this period. In other words, it is possible that high estimates of gr based on total survival do indeed reflect increasing numbers of bears (i.e., there are more births than deaths), but that a substantial proportion of these bears are permanently emigrating from the GB management area. As the ice becomes more dynamic in GB and the surrounding areas, bears may be more dynamic in their movements. 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, radio-telemetry data have been critical to resolving these issues (e.g., Regehr et al. 2018). For regions where radio-telemetry 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).

Reproduction

Our estimates of reproductive indices (e.g., litter size, offspring per female) are on the higher end of the range of expected values for polar bears (Baffin Bay: SWG 2016, Foxe Basin: Stapleton et al. 2016, Western Hudson Bay: Dyck et al. 2017, Southern Hudson Bay: Obbard et al. 2018, Chukchi Sea: Regehr et al. 2018), suggesting that the GB subpopulation is currently capable of healthy reproduction. During our genetic biopsy sampling we were not able to collect data on the numeric age of most bears (i.e.,

through counting cementum annuli in teeth; Calvert and Ramsay 1998), hence we cannot comment on age of first litter for females or inter-birth intervals. However, our estimated number of C1 per adult female of 0.36 in 2015 - 2017 appears to be sufficient to maintain a viable subpopulation, provided that survival is within the normal range for healthy subpopulations (Regehr et al. 2015). The number of C1 per adult female (0.36 in this study) is considered a key reproductive parameter (Vongraven et al. 2012, Regehr et al. 2015) because it integrates cub production and cub survival. This is especially important when C0s cannot be sampled or handled, as in this study (see Method section above). Our estimates for 1998 - 2000 and 2015 - 2017 suggest that no significant change in recruitment occurred over time. Declines in reproductive performance in association with sea ice deterioration have been documented for some polar bear subpopulations (Derocher and Stirling 1995, Derocher 2005, Rode et al. 2010, Peacock et al. 2013, Rode et al. 2014). As spring sea ice break-up occurs earlier (which is also associated with later fall freeze-up; Stern and Laidre 2016, Regehr et al. 2016) feeding opportunities for polar bears presumably decrease, leading to poorer maternal body condition and reduced investment in reproduction. Despite changes in sea ice conditions over the past decades we did not detect any significant changes in reproductive output for GB polar bears, although if climate change continues as predicted (IPCC 2014) there will likely be a threshold beyond which reproduction declines (Laidre et al. 2020).

Survival

Opposite to what Taylor et al. (2009) found in their study, our estimated survival rates (total and un-harvested) demonstrated lower survival rates for males than females (Table 10). Estimates of total (i.e., including harvest mortality) survival for adult females of 0.95 for the period 2005 - 2017 were high relative to other subpopulations for which survival estimates are available (Regehr et al. 2018, their Table S3). However, direct comparison is complicated because most other estimates are of apparent survival which includes permanent emigration. Similar to our findings for the GB subpopulation, a recent study documented male survival rates to be reduced for the Baffin Bay

subpopulation (SWG 2016). We are unaware of why un-harvested male survival may be declining for GB bears and we recommend this as an important area for research and monitoring. There also was moderate support for a time-period effect on survival, with total survival increasing for females and decreasing for males. This should be interpreted with caution because confidence intervals had substantial overlap. There was relatively low support for an age class effect in survival, with point estimates of survival lower for subadults than for adults, although again the CIs overlapped. No support for variation in survival as a function of the sea-ice covariates we explored was detected.

Estimates of un-harvested survival for adult females for the period 2005 - 2017 (0.97) were also high. When considered along with the reproductive indices, these findings suggest that the GB subpopulation remains capable of strong growth. As a note, estimates of total survival (S) reflect the probability of remaining alive. Estimates of S directly from the Burnham models are not estimates of apparent survival (i.e., the probability of remaining alive and not permanently emigrating) because the Burnham model directly estimates the fidelity parameter F. Unlike Taylor et al. (2009), we did not fix the fidelity parameter (F) to 1 (i.e., no assumed permanent emigration) based on the evidence of some movement from GB garnered from harvest recoveries. These factors suggest that there is some permanent emigration, which should be estimated to reduce potential bias in estimates of survival and abundance. Estimates of the parameter F ranged between 0.93 and 0.99 depending on sex and time period, with very large confidence intervals. Collecting movement data through radiotelemetry would provide better understanding of the movement into and out of the GB boundaries allowing more precise estimation of survival and abundance.

Body condition

Bears in GB were in better body condition in the most recent survey from 2015 - 2017 compared to the previous survey in 1998 - 2000. This is in direct contrast to some other

subpopulation studies that have found decreasing body condition of bears in recent years (Rode et al. 2012, Stirling and Derocher 2012, SWG 2016, Laidre et al. 2020). However, polar bear subpopulation ecosystems vary widely. Within GB, multi-year sea ice predominated until recently (e.g., mid-1990s) when a shift to thinner, annual ice has occurred (Schweinsburg et al. 1981, Barber and Iacozza 2004, Howell et al. 2008, 2009, Sou and Flato 2009, Environment and Climate Change Canada 2018). This shift to annual ice may facilitate a short-term boost in hunting opportunities for bears as the ice is thinner and more prone to leads and cracks allowing access to bears' preferred prey, ringed seals (*Pusa hispida*). Indeed, we saw that in the recent time period, as the duration of low-ice days increased, bears were more likely to be in better condition. This is counterintuitive when thinking about polar bears' reliance on sea ice as a hunting platform. However, the GB ecosystem does not currently experience 100% ice-free periods and the low-ice days represented concentrations that were 63% or lower (see Methods: Sea-ice metrics) which are still within the range of preferred polar bear ice concentrations (Durner et al. 2009). It is worth noting that during the period 2009 - 2014 (Stern and Laidre 2016), the sea-ice area dipped to ~10%. Polar bears come onshore at concentrations of around 10-15% ice (Cherry et al. 2013) and thus, if sea ice coverage declines further, we may see a similar negative relationship of body condition and low sea ice concentration or extent as has been reported for other subpopulations (Regehr et al. 2007, Rode et al. 2012, SWG 2016, Laidre et al. 2020).

More favorable ice conditions relative to seal hunting, coupled with the seal pupping period that occurs roughly around mid-April, may account for our finding that body condition improved for bears sampled later in the field season (Stirling and Archibald 1977, Pilfold et al. 2014, Reimer et al. 2019). Females with offspring were much more likely to be in poor body condition compared to the other reproductive groups. When they were sampled earlier in the year, their probability of being in poor condition was highest which is unsurprising given the increased nutritional stress this reproductive class faces due to lactation and parturition. As time progressed, the likelihood of being in poor condition declined and they were more likely to be rated as

'average' suggesting that access to prey during the prime feeding period in the spring was beneficial for accumulating nutritional stores.

Similar to previous studies (SWG 2016, Laidre et al. 2020, GN unpublished data report MC 2020), the differences in body condition we observed are not likely related to the sampling method. Raw BCS scores were binned into 3 general categories to account for any potential small biases in observer classifications. Furthermore, in other similar studies in which comparisons in BCS were made for an earlier time period that used physical capture to determine BCS and a later time period in which aerial classifications were done, there were no trends of either method for BCS, suggesting that there is not an inherent bias in either method for BCS classification (e.g. Kane Basin: no change in BCS over time, Baffin Bay: decrease in BCS over time, M'Clintock: increase in BCS over time; SWG 2016, Laidre et al. 2020, GN unpublished data). In this study, the observer with the most sampling observations participated in both the early sampling period and recent one. The other observers were experienced and had participated both in physical capture studies and in aerial observation studies. The general application of our body condition index during physical handling has been shown to be a reliable indicator (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.

7. MANAGEMENT IMPLICATIONS

The need for continued monitoring
Climate change has affected the sea ice in every polar bear management unit (subpopulation) (Stern and Laidre 2016; Regehr et al. 2016), including GB. Over time, ice concentrations and thickness have declined, and the break-up and freeze-up dates have advanced and delayed, respectively (Stern and Laidre 2016). These changes in sea ice dynamics can elicit behavioural, nutritional, and demographic changes in bears. For example, studies in Baffin Bay documented that bears have reduced their home

range size and are spending more time on shore during the ice-free period with reduced denning periods (SWG 2016). In other subpopulations, the effects of climate change on polar bears have been exhibited through reduced body condition, survival rates, and litter sizes (Regehr et al. 2007, Stapleton et al. 2014, Lunn et al. 2016, Dyck et al. 2017, Obbard et al. 2016, 2018). These sea ice changes and their impact on bears have only become apparent because of concerted monitoring efforts of both sea ice and bear movements over long periods of time.

Body condition, reproduction, and survival may reflect changes on a finer temporal scale than abundance and can help understand the mechanisms through which environmental change affects polar bears. The GB subpopulation currently has several knowledge gaps that present challenges for informed decision making. It is currently unknown how bears in GB spend their time during the sea-ice minimum (e.g., July to October) due to the lack of movement data. Also, the delineation of this subpopulation is inferred based on movement of collared female bears during the 1990s (Bethke et al. 1996, Taylor et al. 2001), prior to the large-scale changes in sea-ice habitat. Recoveries of previously captured, and subsequently harvested, bears indicate that there is emigration into LS, MC, and FB (Fig. 7), although whether this is permanent or temporary is difficult to determine without movement data. Note also that our abundance estimate is for the superpopulation (see Discussion section) which likely reflects more animals than occur within the GB management boundary.

In respecting Inuit societal values and concerns over physically handling wildlife, the GN, Department of Environment, did not carry out any collaring to collect radio-telemetry data in GB, despite efforts to garner support for a collaring program and the associated valuable data. The GN, together with other co-management partners, will have to decide on how monitoring polar bears in this subpopulation will continue in order to provide adequate information to decision-makers.

Harvest management and considerations

The GB polar bear subpopulation experienced a mean annual harvest of approximately 62 bears between the harvest years 2004/2005 and 2016/2017 (roughly 40 males and 22 females; GN, unpublished data) with a TAH of 74 bears per year. Our current abundance estimate for the superpopulation, together with other demographic data, suggest that the subpopulation has likely remained stable or only declined slightly given the removal rates and observed climatic sea ice changes. We suggest that taken together this study provides evidence that the GB subpopulation is currently healthy and productive. We documented a potential decline in the male proportion of the subpopulation, which may reflect the harvest system in place (i.e., 2 males for every female). However, similar to the Baffin Bay subpopulation (SWG 2016), we also found evidence for a decline in un-harvested survival for males, which we cannot currently explain. Future research and monitoring should seek to understand the causes and potential ramifications of male survival rates.

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

- Conduct a meta-population analysis that includes all possible subpopulations where some exchange of bears occurs (e.g., with LS and MC). This is important because the current abundance estimate for the GB subpopulation of 1525 bears (SE = 294) likely includes bears that also spend time in other management units. Assessing each subpopulation individually could lead to overestimating the total number of bears available and increases the risk of overharvest.

- Determine harvest management objectives (e.g., to maintain, reduce, or increase the subpopulation), taking into account possible changes in environmental carrying capacity in the future and the observed reduction in male proportion and survival rates. Perform a quantitative harvest risk assessment so that scientific information is available to help inform and justify management decisions.

Research recommendations for GB

These recommendations reflect both newly gained insight from the experience of conducting and analyzing the GB data as well as continued awareness of the importance of certain research methods.

1. Seek support from co-management partners to implement a radio-telemetry study to collect movement data in GB to obtain emigration estimates, resolve boundary issues, collect missing demographic data, improve precision and accuracy of demographic estimates, and evaluate changes in habitat use and denning in light of the 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) Sample bears (i.e., introduce more marks into the GB subpopulation) 5 - 7 years post-completion of field portion of last study (e.g., in 2023 or 2024) until the next comprehensive population study will be conducted (~10 – 15 yrs 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 GB, and if there are significant changes in reproduction consider whether the timing of the next comprehensive subpopulation assessment should be changed;

3.   Or, 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;

8. ACKNOWLEDGEMENTS

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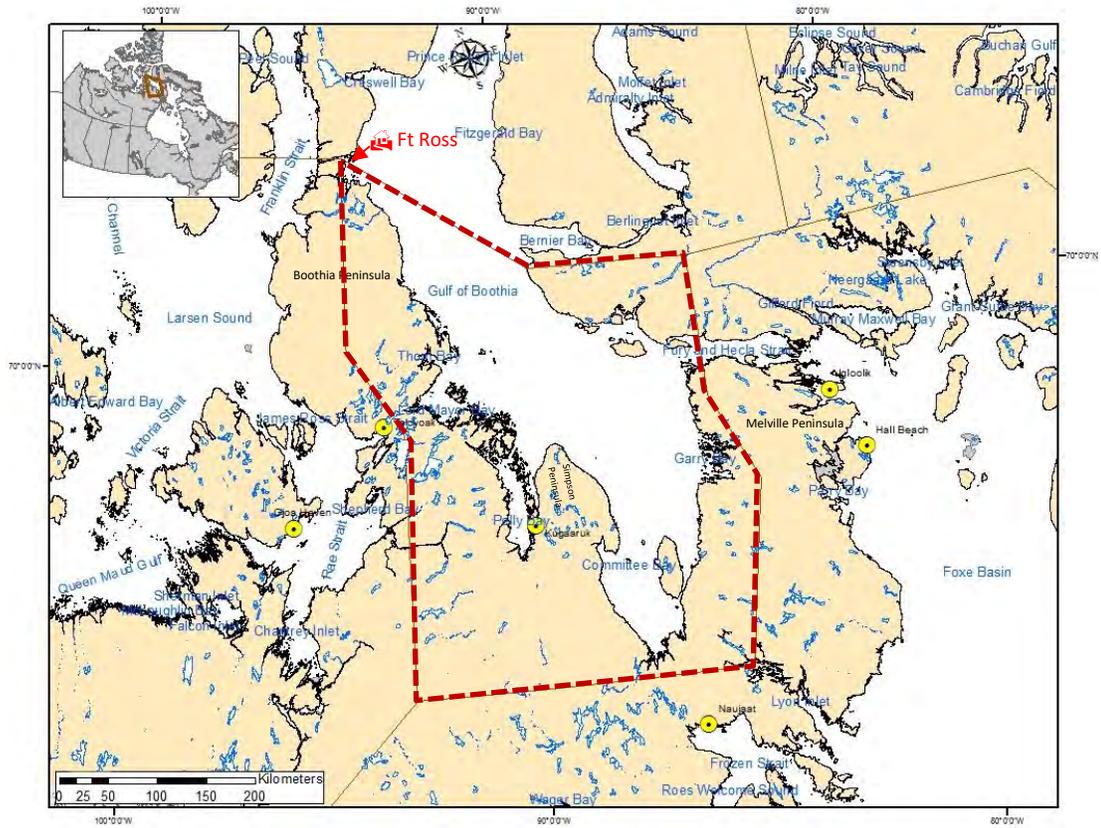
Figure 1.   Basic overview and location of the Gulf of Boothia polar bear subpopulation delineated by red dashed line.

a)

b)
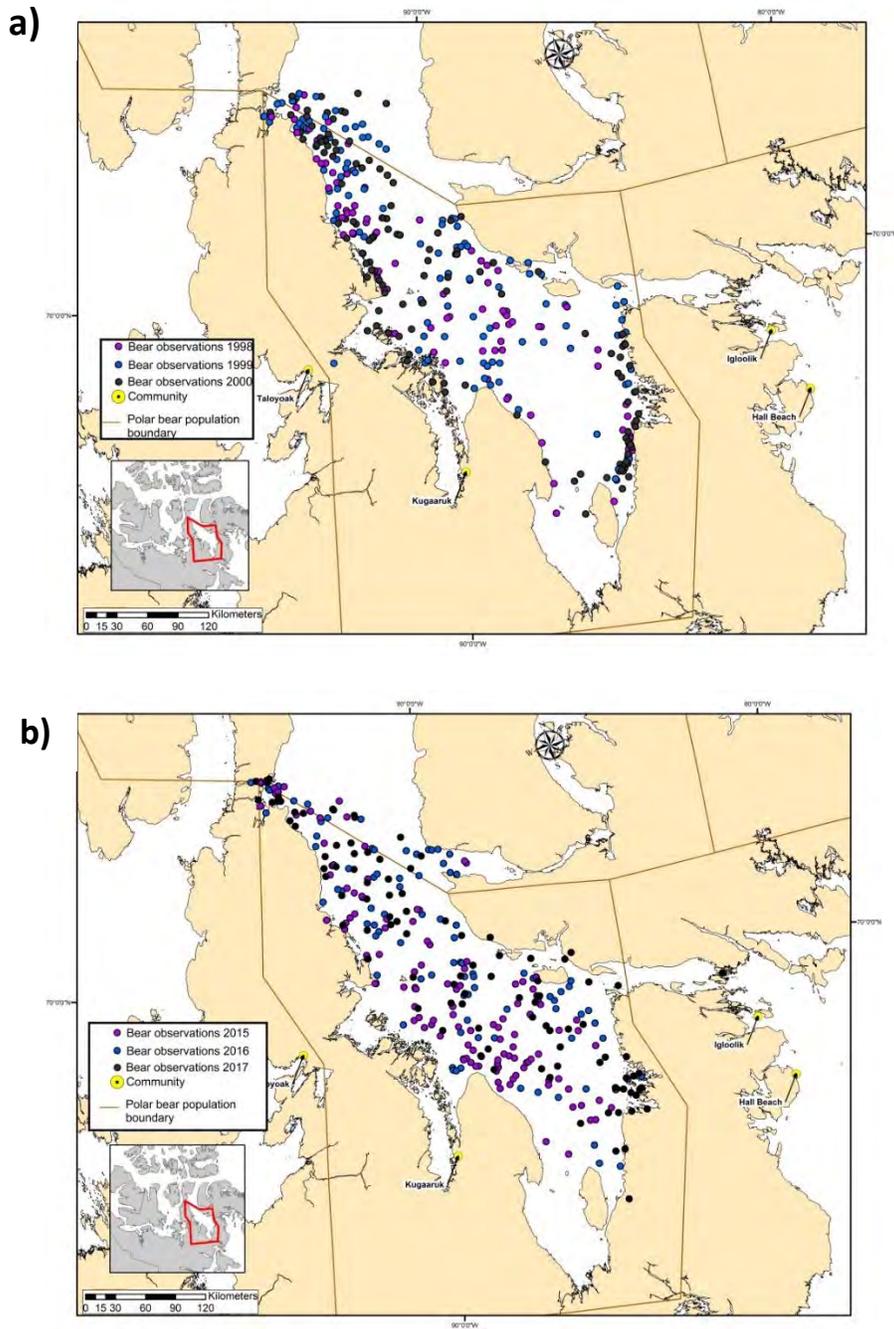
Figure 2.   Locations of observed polar bears within the Gulf of Boothia study
area during the 1998 - 2000 (a) and 2015 - 2017 (b) studies.
Different colored dots indicate different years. Inset shows
subpopulation boundary in red.
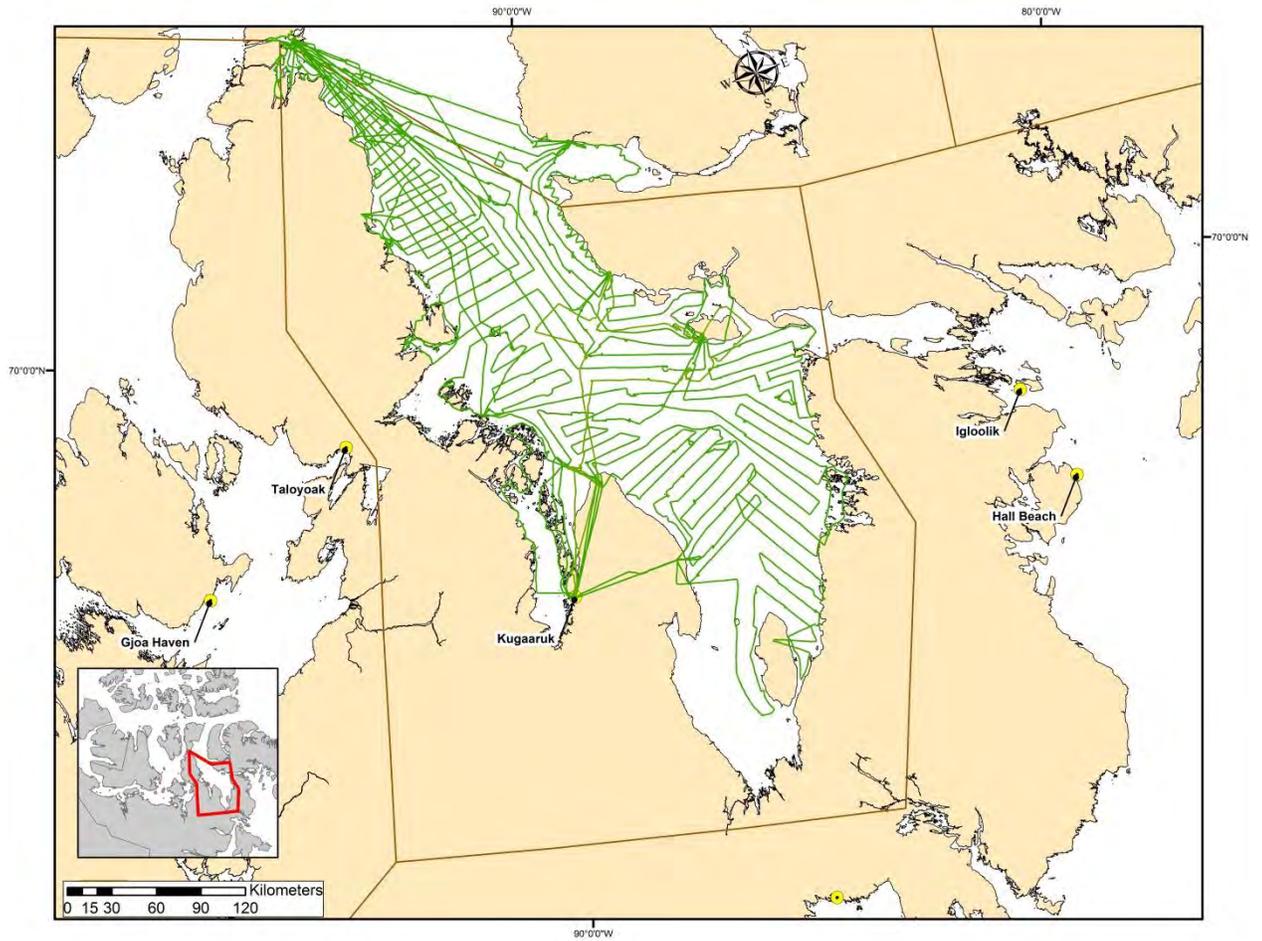
Figure 3.   Flight tracks (green lines) of helicopter flown in search for polar bears in Gulf of Boothia, Nunavut, Canada, during April/May 2017. Inset shows subpopulation boundary in red.
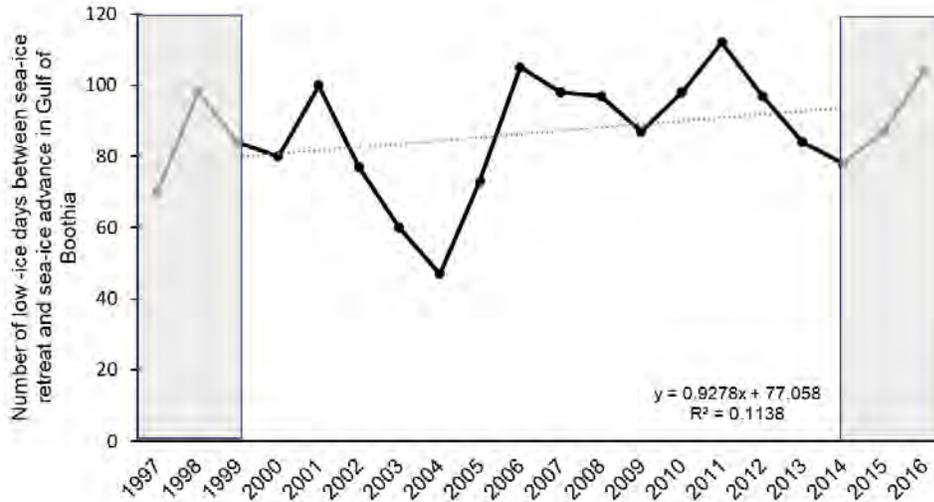
Figure 4. Sea-ice metric of 'low-ice days' calculated as the number of days between the sea ice retreat and sea ice advance in calendar year t *using the transition dates when ice concentration dropped below, and exceeded, respectively, the midway point of sea ice concentration between the March and September mean (Environment and Climate Change Canada 2018). Shaded boxes indicate sampling periods used in this study and intervening years are shown for context. Gray dotted line indicates the linear trend of low-ice days from 1997-2016.*
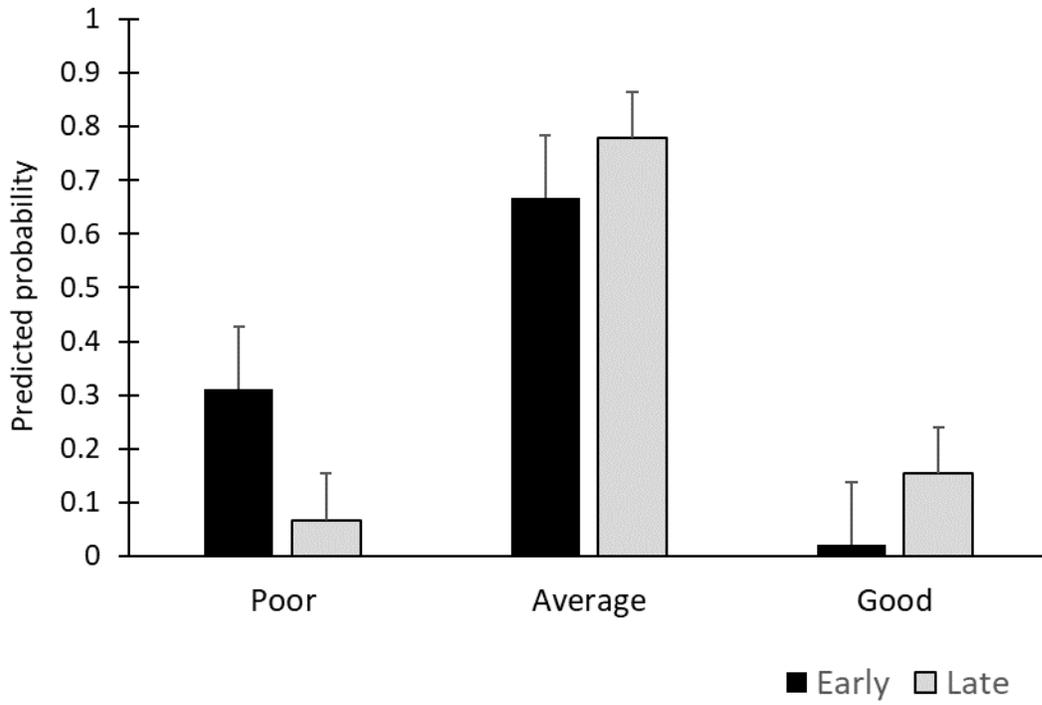
Figure 5.   *Predicted probability based on best-fit model parameter estimates of a bear being classified as poor, average, or good body condition for each time period (Early = 1998 - 2000; Late = 2015 - 2017).*
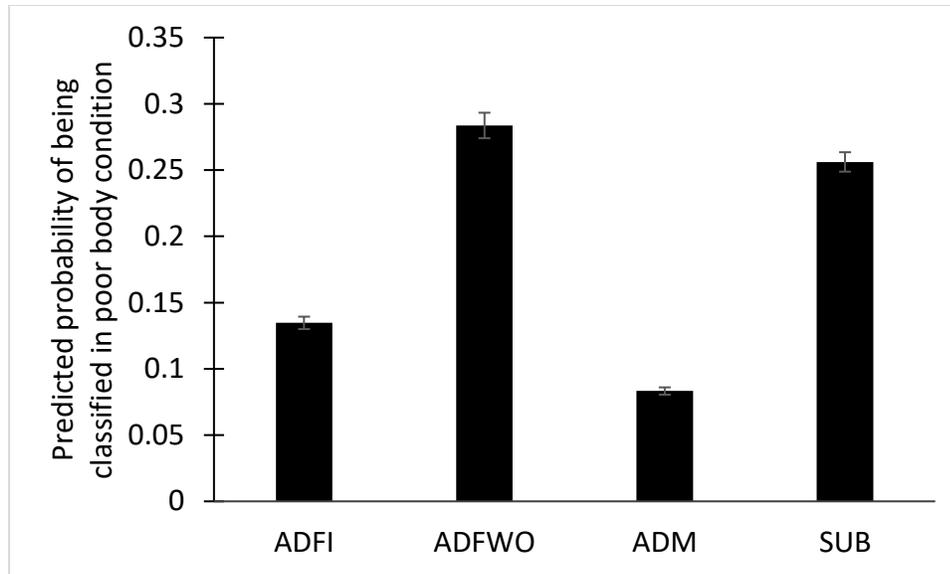
Figure 6. *Predicted probability based on best-fit model parameter estimates of a bear being classified in poor body condition for each reproductive age class across both time periods. Adult females with offspring and subadults were more likely than other reproductive age classes to be classified in poor body condition at the time of sampling (ADFI = independent adult female, ADFWO = adult female with offspring, ADM = adult male, SUB = subadults of both genders).*
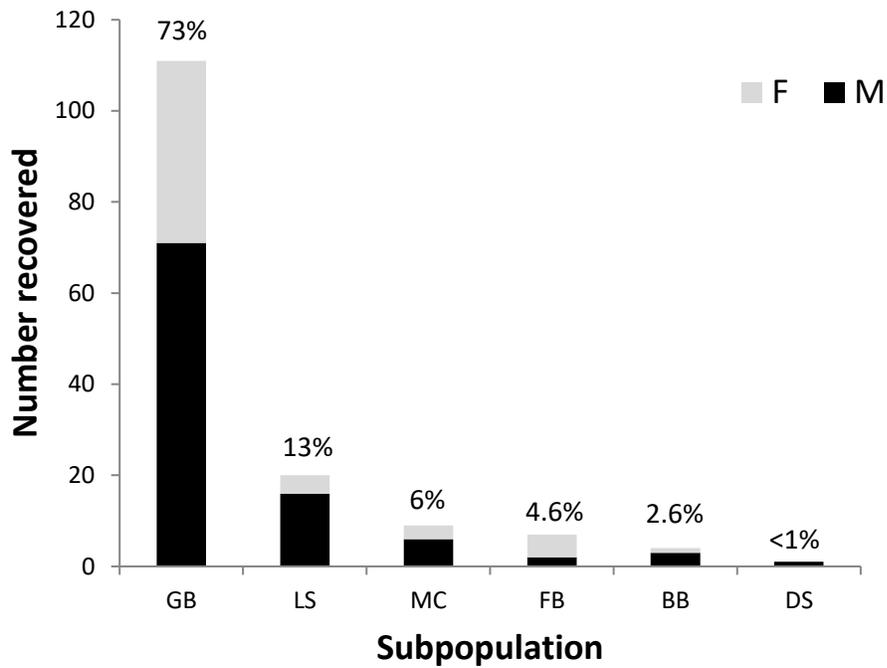
Figure 7.   *Number of polar bear tags that were initially deployed within the Gulf of Boothia subpopulation boundary and subsequently recovered through the harvest between 1972 and 2017. Percentages indicate the proportion of total recoveries that occurred in a given subpopulation (GB=Gulf of Boothia; LS = Lancaster Sound; MC=M'Clintock Channel; FB=Foxe Basin; BB=Baffin Bay; DS=Davis Strait).*

Table 1. Parameter estimates for best-fit ordinal logistic regression model (reference level = "poor"/BCS = 1) for body condition score analysis of the Gulf of Boothia subpopulation.

Parameter	Estimate	SE	p
periodlate	3.77	1.61	0.02
reproclassADFWO	-5.70	3.12	0.07
reproclassADM	3.74	3.03	0.22
reproclassSUB	2.07	3.22	0.52
jul_cap_day	0.03	0.02	0.14
periodearly:icetm	0.04	0.01	0.001
periodlate:icetm	-0.02	0.01	0.08
reproclassADFWO:jul_cap_day	0.04	0.03	0.14
reproclassADM:jul_cap_day	-0.03	0.02	0.29
reproclassSUM:jul_cap_day	-0.02	0.03	0.35

Table 2. Parameter-specific submodels used to analyze live-recapture dead-recovery
data for the Gulf of Boothia polar bear subpopulation.

Submodel name	Submodel structure
S1	.
S2	year
S3	icetm1
S4	sex
S5	sub
S6	year + sex
S7	year + sex + year:sex
S8	year + sub
S9	year + sub + year:sub
S10	icetm1 + sub
S11	icetm1 + sub + icetm1:sub
S12	sex + sub
S13	year + sex + sub
S14	year + sex + sub + year:sex + year:sub
S15	icetm1 + sex + sub
S16	icetm1 + sex + sub + icetm1:sex + icetm1:sub
r1	.
r2	year
r3	sex
r4	year + sex
r5	year + sex + year:sex
p1	.
p2	year
p3	sex
p4	year + sex
F1	.
F2	year
F3	sex
F4	year + sex

(*S* = survival; *r* = dead reporting probability; *p* = recapture probability; *F* = fidelity)

Table 3. Overview of descriptive field statistics of the Gulf of Boothia polar bear study 2015 - 2017.

Field Year	Search time (hr)	Number of bears/hr	Bears encountered[a]	Flown distance (km)	Duration
2015	96.0	1.90	185	11,737	29 April - 26 May
2016	99.3	1.62	161	12,867	20 April - 14 May
2017	115.0	1.40	162	12,200	26 April - 15 May

[a] The number of bears encountered does not represent the genetically corrected number of bears (e.g., some bears have been re-sampled within same sampling period)

Table 4. Body condition scores (BCS) for polar bears in the Gulf of Boothia
subpopulation 1998 - 2000 and 2015 - 2017. 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			2015 - 2017		
	Poor	Average	Good	Poor	Average	Good
Adult female without offspring	17	28	3	2	60	19
Adult female with offspring	30	40	2	5	86	4
Adult male	19	104	4	1	64	28
Subadult	25	34	2	4	43	2
Total	91	206	11	12	253	53

Table 5. Numbers and mean sizes of cub-of-the-year (C0) and yearling (C1) litters observed during capture-recapture studies on the Gulf of Boothia polar bear subpopulation.

	1998	1999	2000	2015	2016	2017
Number of C0 litters	20	13	20	12	22	12
Mean C0 litter size	1.60	1.54	1.70	1.75	1.50	1.58
Number of C1 litters	13	17	10	18	9	13
Mean C1 litter size	1.31	1.53	1.80	1.56	1.44	1.62

Table 6. Model-averaged parameter estimates for a binomial logistic regression on cub-of-the-year (C0) litter size for the Gulf of Boothia polar bear subpopulation.

Parameter	Estimate	SE	P	Importance
(Intercept)	0.78	1.12	0.49	NA
icefree.tm1	0.00	0.01	0.75	0.31
periodearly	0.02	0.19	0.90	0.18
month05	-0.01	0.18	0.98	0.17
BCS (level 1)	-0.07	0.27	0.79	0.15
BCS (level 3)	0.11	0.43	0.80	0.15

Table 7. Model-averaged parameter estimates for a binomial logistic regression on yearling (C1) litter size for the Gulf of Boothia polar bear subpopulation.

Parameter	Estimate	SE	P	Importance
(Intercept)	-0.74	1.53	0.63	NA
icefree.tm1	0.01	0.02	0.57	0.41
periodearly	-0.05	0.24	0.86	0.26
BCS (level 1)	0.02	0.13	0.91	0.06
BCS (level 3)	0.00	0.25	1.00	0.06

Table 8. Numbers of live-observations and dead-recoveries (in parentheses) of individually identified polar bears in the Gulf of Boothia subpopulation used in survival estimation.

Years	AFNC[a]	AFC0[b]	AFC1[c]	AM[d]	C1[e]	SF[f]	SM[g]
1976 - 1997	21 (18)	17 (0)	10 (0)	49 (23)	15 (0)	13 (4)	21 (0)
1998 - 2000	75 (3)	53 (0)	40 (0)	128 (6)	68 (0)	49 (3)	44 (5)
2001 - 2017	88 (5)	46 (0)	40 (0)	94 (19)	61 (0)	21 (1)	34 (5)

([a]AFNC = adult female no cubs; [b]AFC0 = adult females with cubs-of-the-year; [c]AFC1 = adult females with yearlings; [d]AM = adult male; [e]C1 = yearlings; [f]SF = subadult females; [g]SM = subadult males)

Table 9.   Importance scores for the various factors and covariates within the parameter-specific survival submodels. Importance scores for interaction terms (e.g., year:sex) should be interpreted with caution because interactions can only appear in models with the corresponding main effects.

Factor or covariate	S	r	p	F
sex	0.82	0.33	0	0
year	0.71	0.35	0.06	0.16
year:sex	0.67	0.33	NA	NA
sub	0.23	NA	NA	NA
year:sub	0.23	NA	NA	NA
icetm1	0.05	NA	NA	NA
icetm1:sex	0	NA	NA	NA
icetm1:sub	0	NA	NA	NA

Table 10.   Model-averaged parameter estimates for the Burnham model for survival and abundance.

Parameter	Class	Year block	Estimate	lci	uci
$S*$	Adult female	1976-2004	0.94	0.90	0.98
$S*$	Adult male	1976-2004	0.93	0.90	0.95
$S*$	Subadult female	1976-2004	0.93	0.86	0.99
$S*$	Subadult male	1976-2004	0.91	0.85	0.96
$S*$	Adult female	2005-2017	0.97	0.91	1.00
$S*$	Adult male	2005-2017	0.90	0.83	0.96
$S*$	Subadult female	2005-2017	0.95	0.86	1.00
$S*$	Subadult male	2005-2017	0.87	0.75	0.99
S	Adult female	1976-2004	0.92	0.86	0.96
S	Adult male	1976-2004	0.89	0.85	0.93
S	Subadult female	1976-2004	0.90	0.80	0.95
S	Subadult male	1976-2004	0.87	0.77	0.92
S	Adult female	2005-2017	0.95	0.81	0.99
S	Adult male	2005-2017	0.85	0.74	0.92
S	Subadult female	2005-2017	0.94	0.69	0.99
S	Subadult male	2005-2017	0.81	0.59	0.92
r	All female	1976-2004	0.26	0.17	0.38
r	All male	1976-2004	0.29	0.22	0.37
r	All female	2005-2017	0.22	0.08	0.46
r	All male	2005-2017	0.33	0.21	0.47
p	All female	1976-2004	0.11	0.08	0.15
p	All male	1976-2004	0.12	0.08	0.16
p	All female	2005-2017	0.10	0.07	0.14
p	All male	2005-2017	0.10	0.07	0.15
F	All female	1976-2004	0.95	0.71	0.99
F	All male	1976-2004	0.99	0.38	1.00
F	All female	2005-2017	0.93	0.79	0.98
F	All male	2005-2017	0.95	0.59	1.00

($S*$ = unharvested survival; S = total survival; r = dead reporting probability; p = recapture probability; F = fidelity)