# CHAPTER 4

# HABITAT AND HABITAT USE IN BAFFIN BAY

## KEY FINDINGS

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

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

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

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

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

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

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

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

## 4.1. Introduction

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

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

## 4.2. Methods

Sea ice Analysis

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

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

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

### Sea-ice Habitat Use and Resource Selection Models

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

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

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

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

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

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

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

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

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

## Terrestrial Habitat Use and Selection

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

## 4.3. Results

### Sea-ice Habitat

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

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

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

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

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

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

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

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

## Terrestrial Habitat Use and Selection

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

## 4.4. Discussion

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

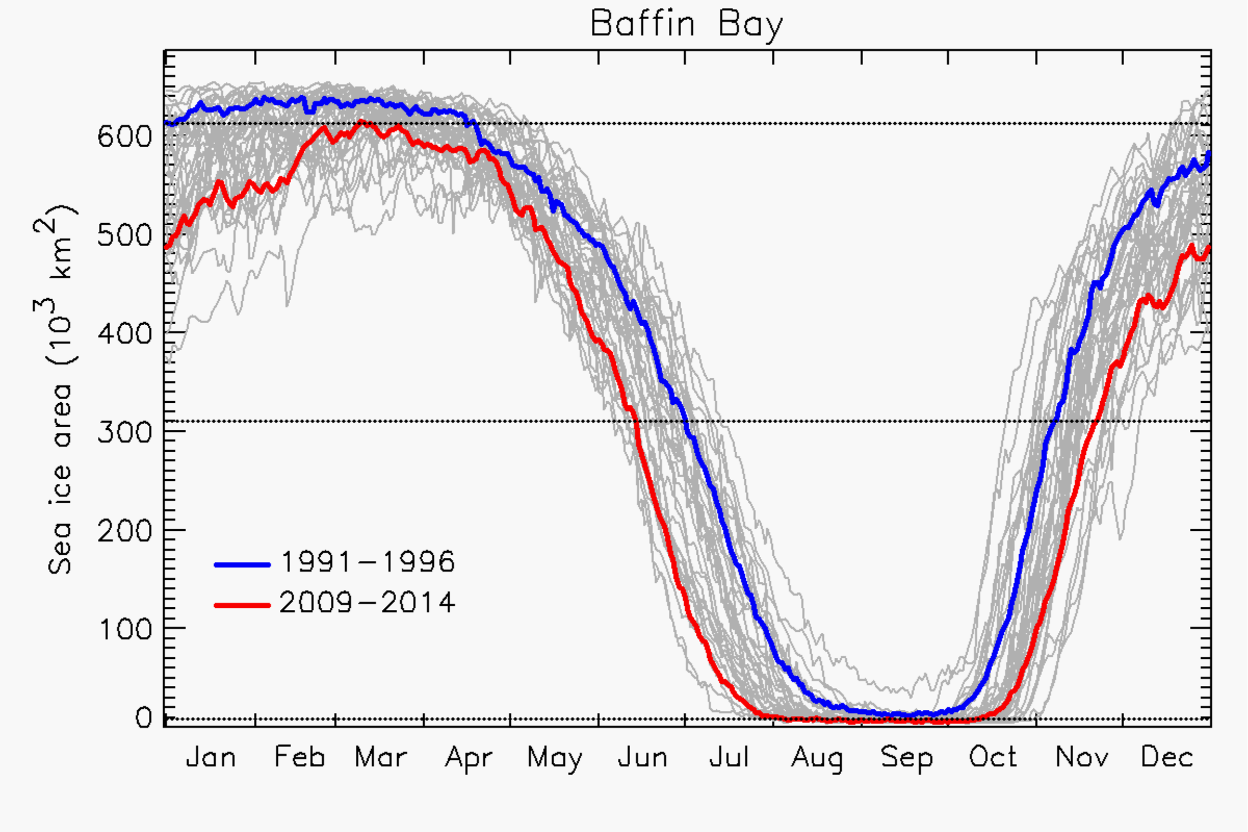
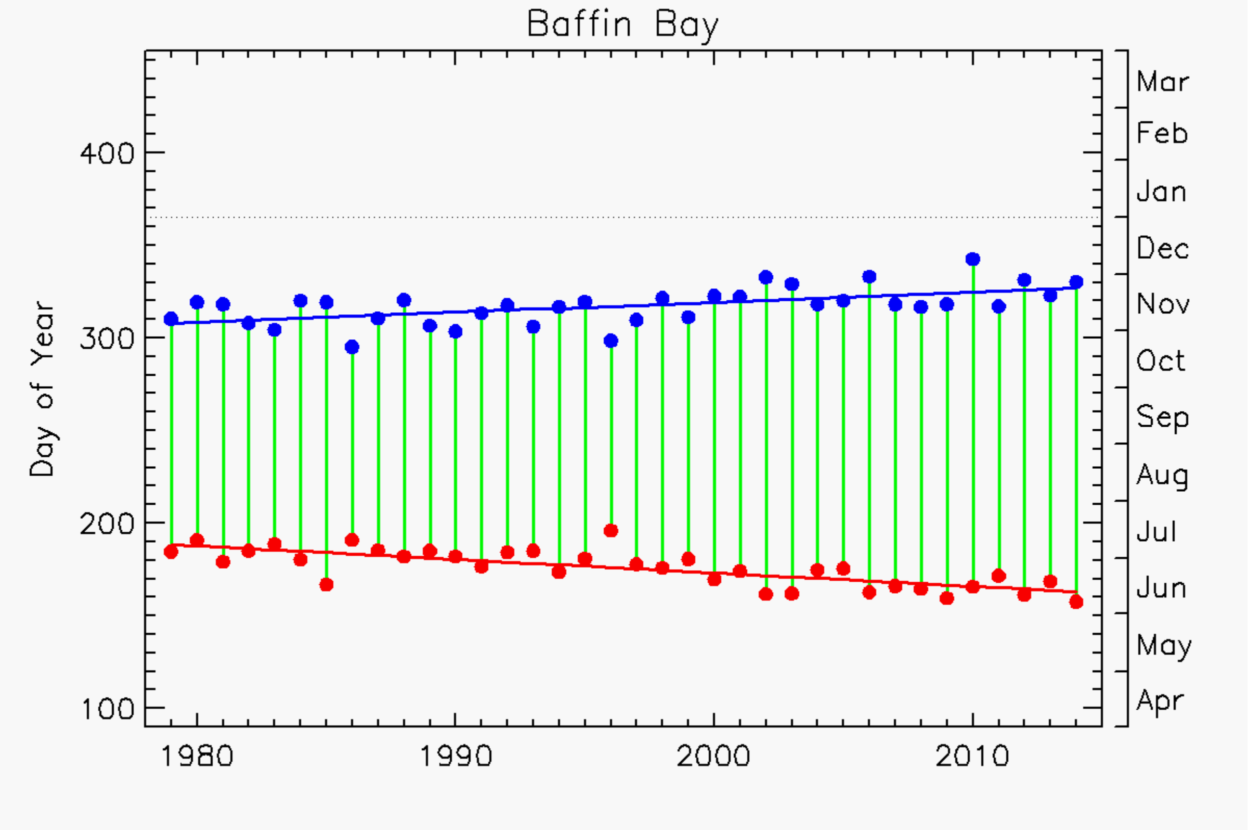


**Figure 4.2.** Day of spring sea-ice retreat (red circles), fall sea-ice advance (blue circles), and the interval between them (green lines), for Baffin Bay (all depths), 1979-2014. Least-squares fits to spring and fall dates are shown (red and blue lines). Trends are given in Table 4.1.



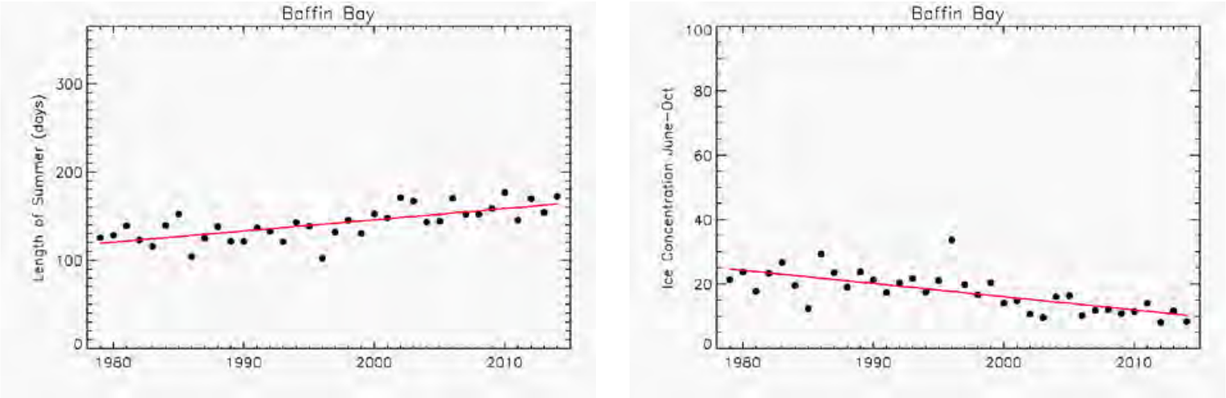


**Figure 4.3.** Length of summer (left) and mean sea-ice concentration during June-October (right) for Baffin Bay (all depths), 1979-2014. Length of summer is the interval from spring sea-ice retreat to fall sea-ice advance (see Figure 4.2, green lines). Least-squares fits are shown (red lines); trends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

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

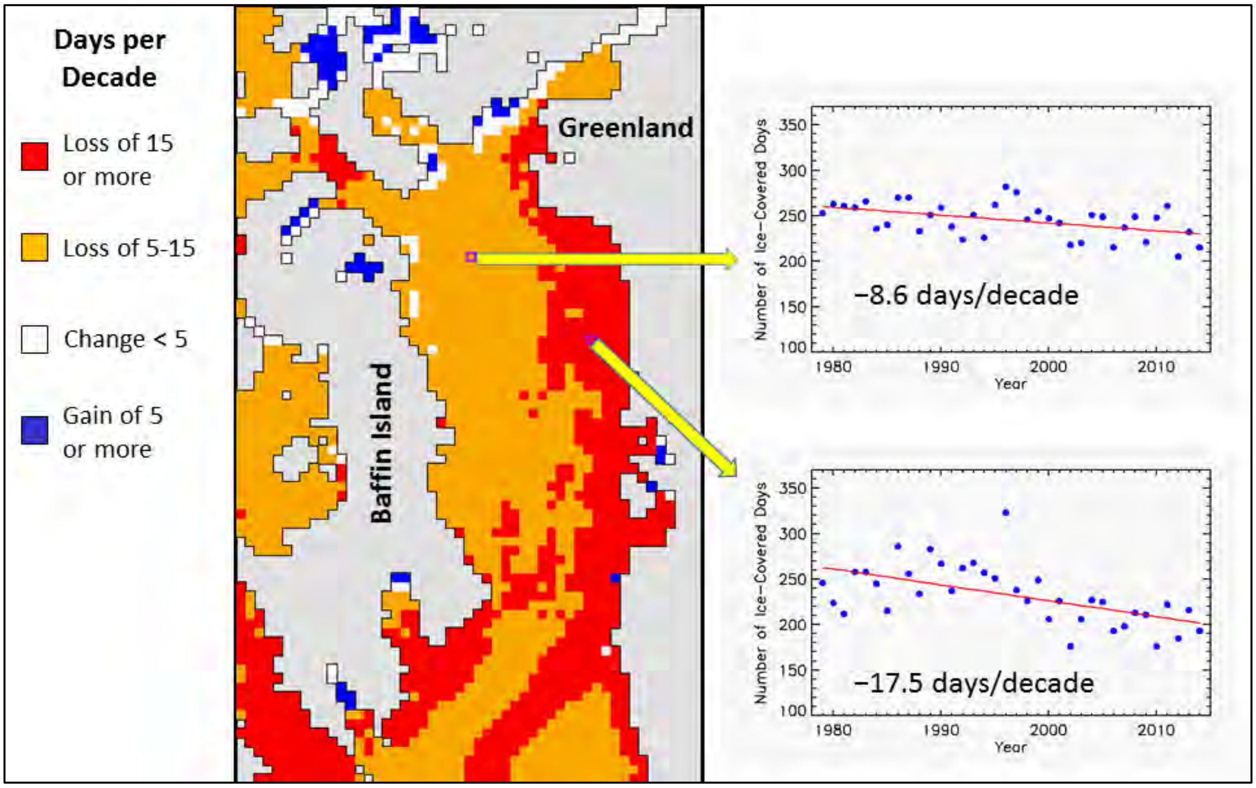


**Figure 4.5.** Sea-ice concentration in Baffin Bay on July 15 of every year from 1979 (upper left) to 2014 (lower right). Color coding: 15-50% (blue), 50-85% (green), 85-95% (yellow), 95-99% (orange), 99-100% (red). Black dots in Baffin Bay indicate shallow depths (< 300 m).

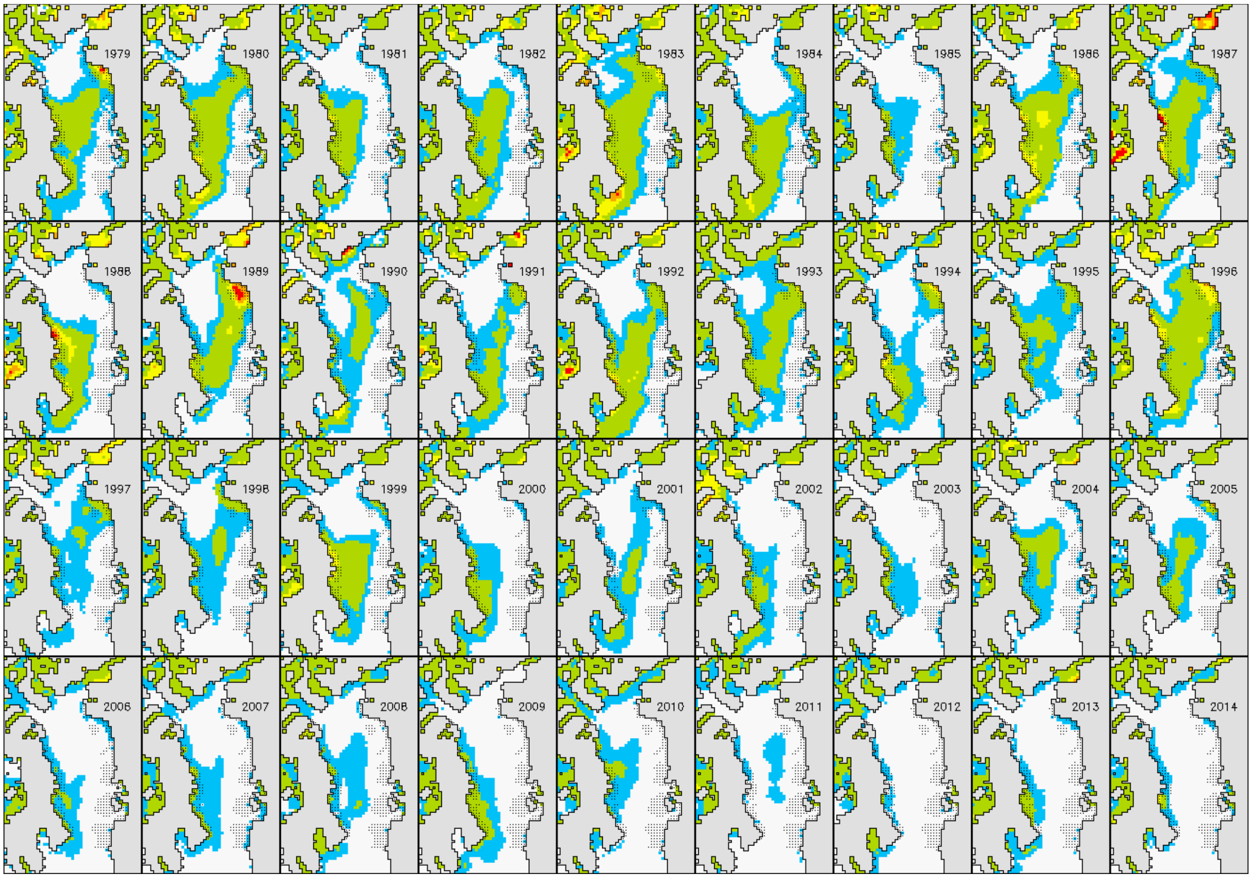


**Figure 4.6.** Location of the leading edge of sea ice (> 50% concentration) along the coast of Baffin Island on November 1 in the years 1991-1995 (top row) and 2009-2013 (bottom row). Color coding of sea-ice concentration is the same as in Figure 4.5. In the top row, black circles mark the leading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

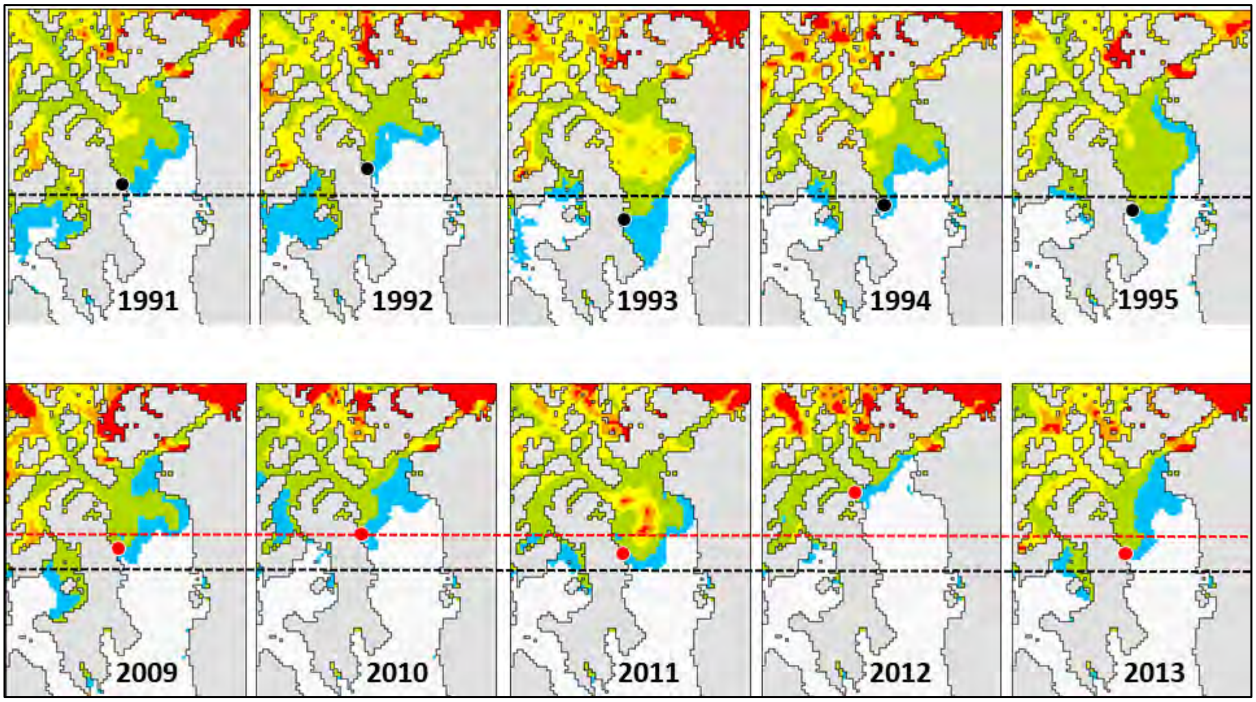


**Figure 4.7.** Movement rate of BB adult female bears (km/day) in the 1990s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

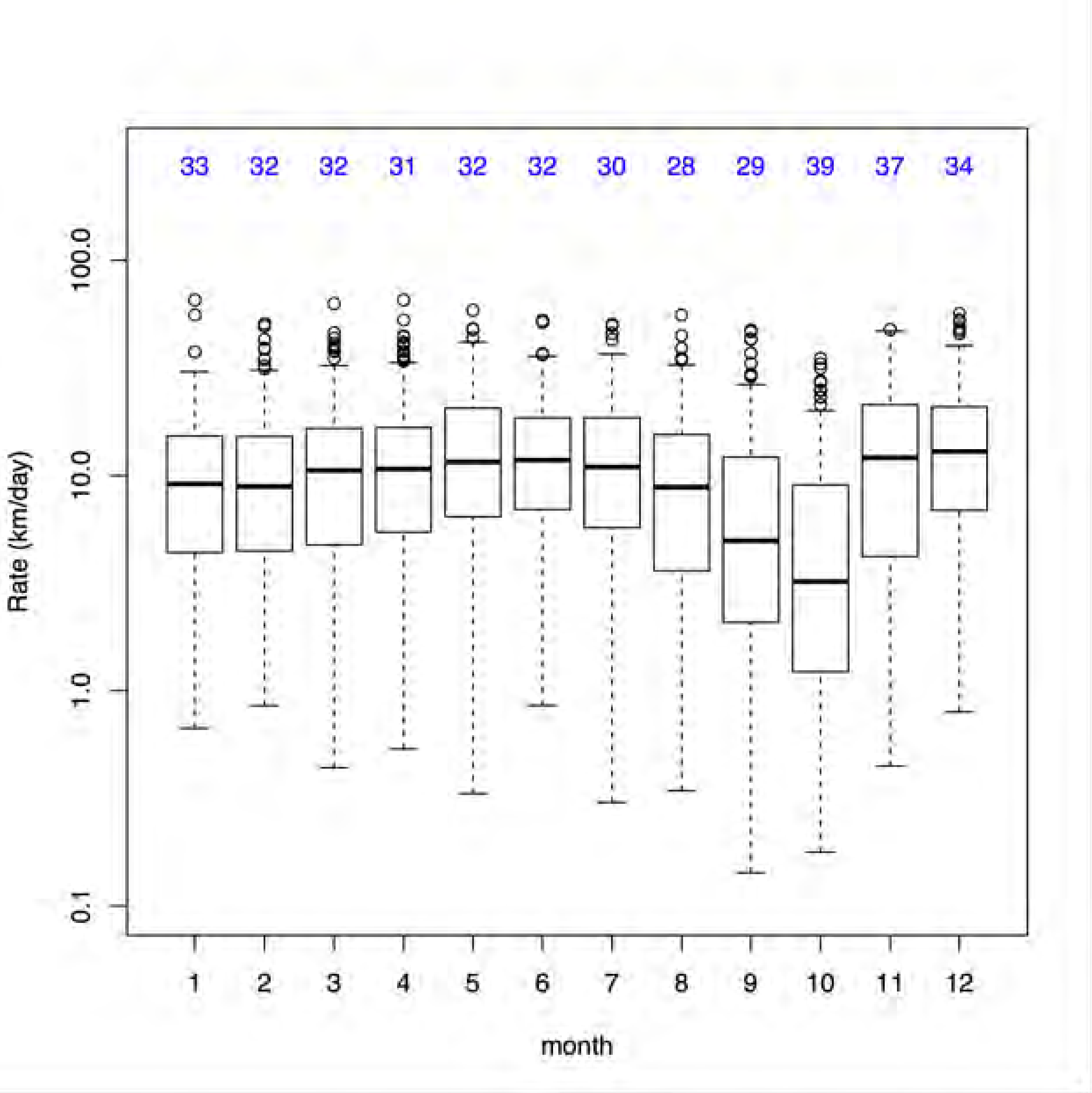


**Figure 4.8.** Movement rate of BB adult female bears (km/day) in the 2000s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

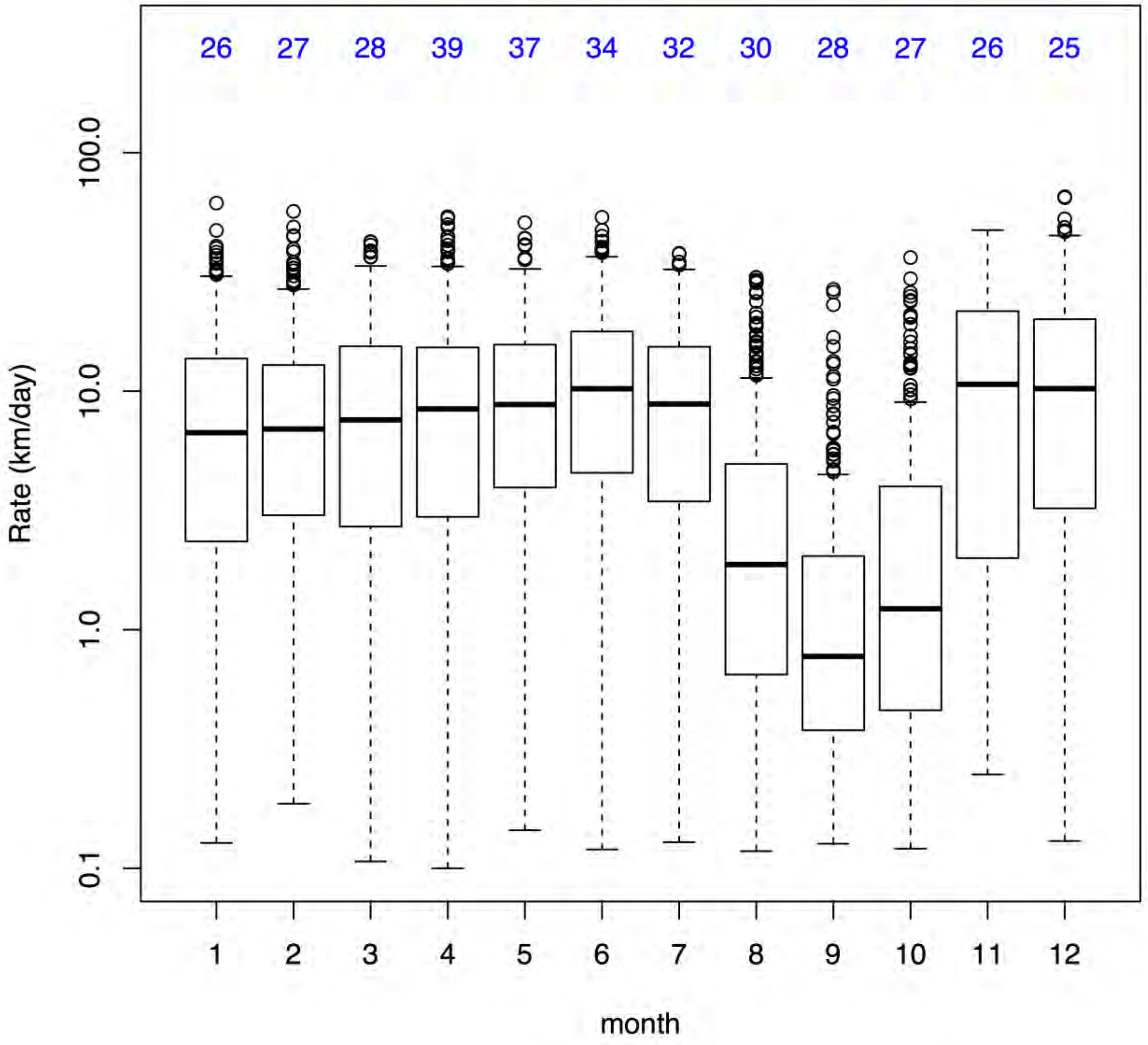


**Fig 4.9.** 1990s and 2000s adult female polar bear habitat use in BB for each of four sea-ice habitat variables: sea-ice concentration in small buffer, distance to 15% sea-ice concentration, distance to 50% sea-ice concentration, and distance to the nearest land. Data from 1990s are shown in red, 2000s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

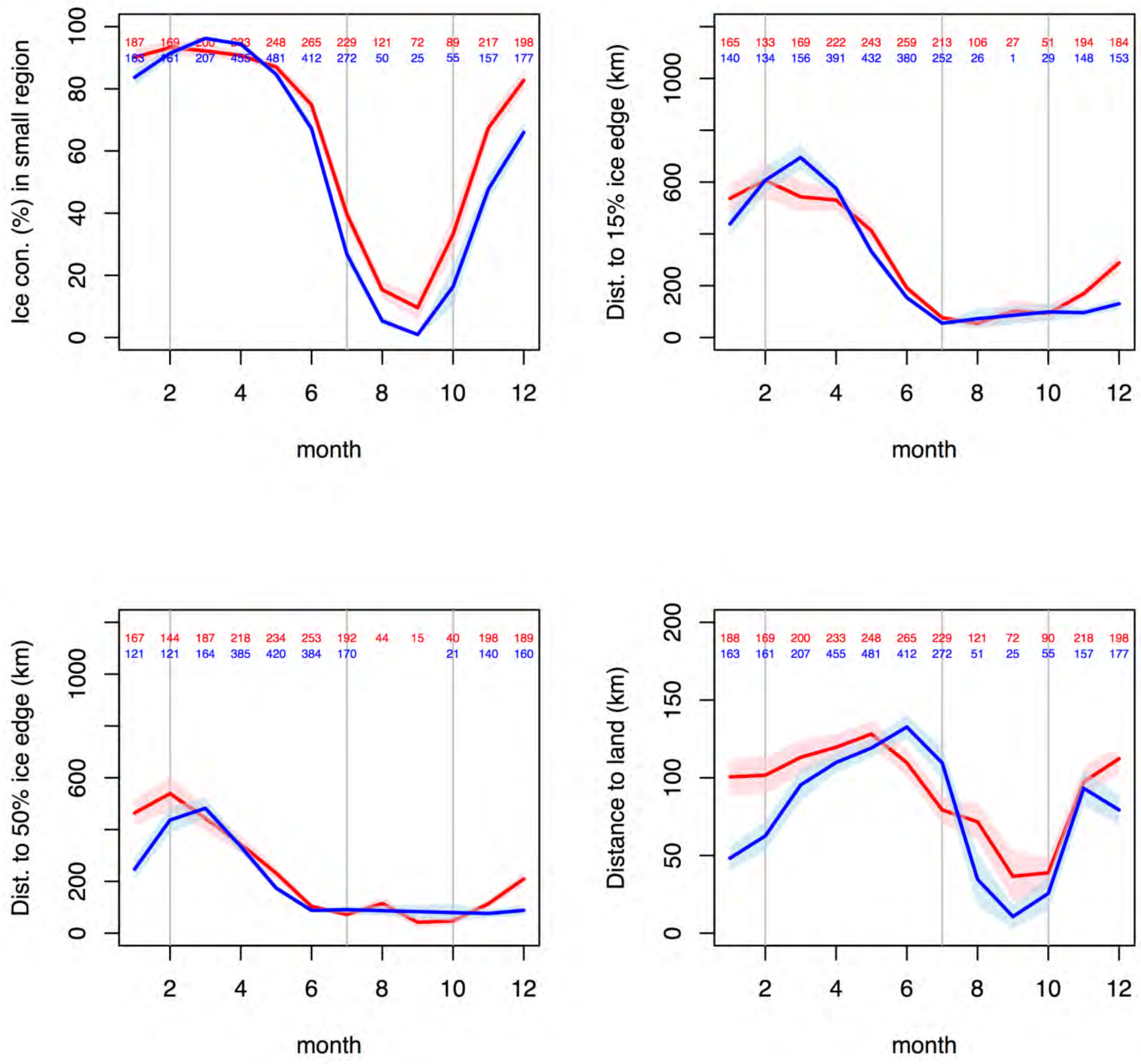


**Fig 4.10.** 1990s and 2000s adult female polar bear habitat use in BB for each of three sea-ice habitat variables: distance to 300 m depth contour, depth (bathymetry), and percentage of observations on the sea ice. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the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

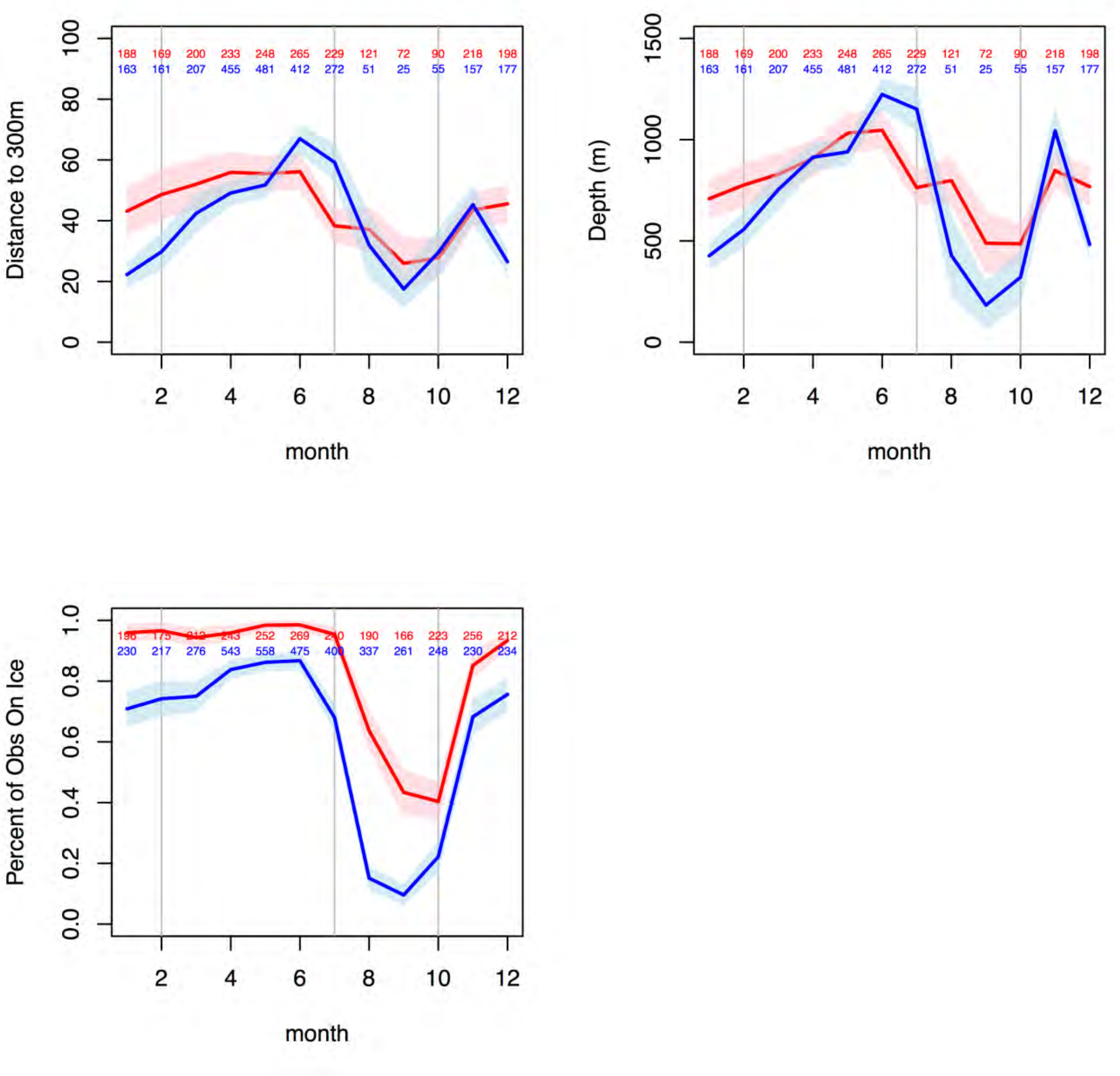


**Figure 4.11.** 1990s and 2000s adult female polar bear habitat use for each of four land habitat variables: Elevation, Slope, Aspect and Distance to the Baffin Island coast (from inland). Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. The small numbers in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

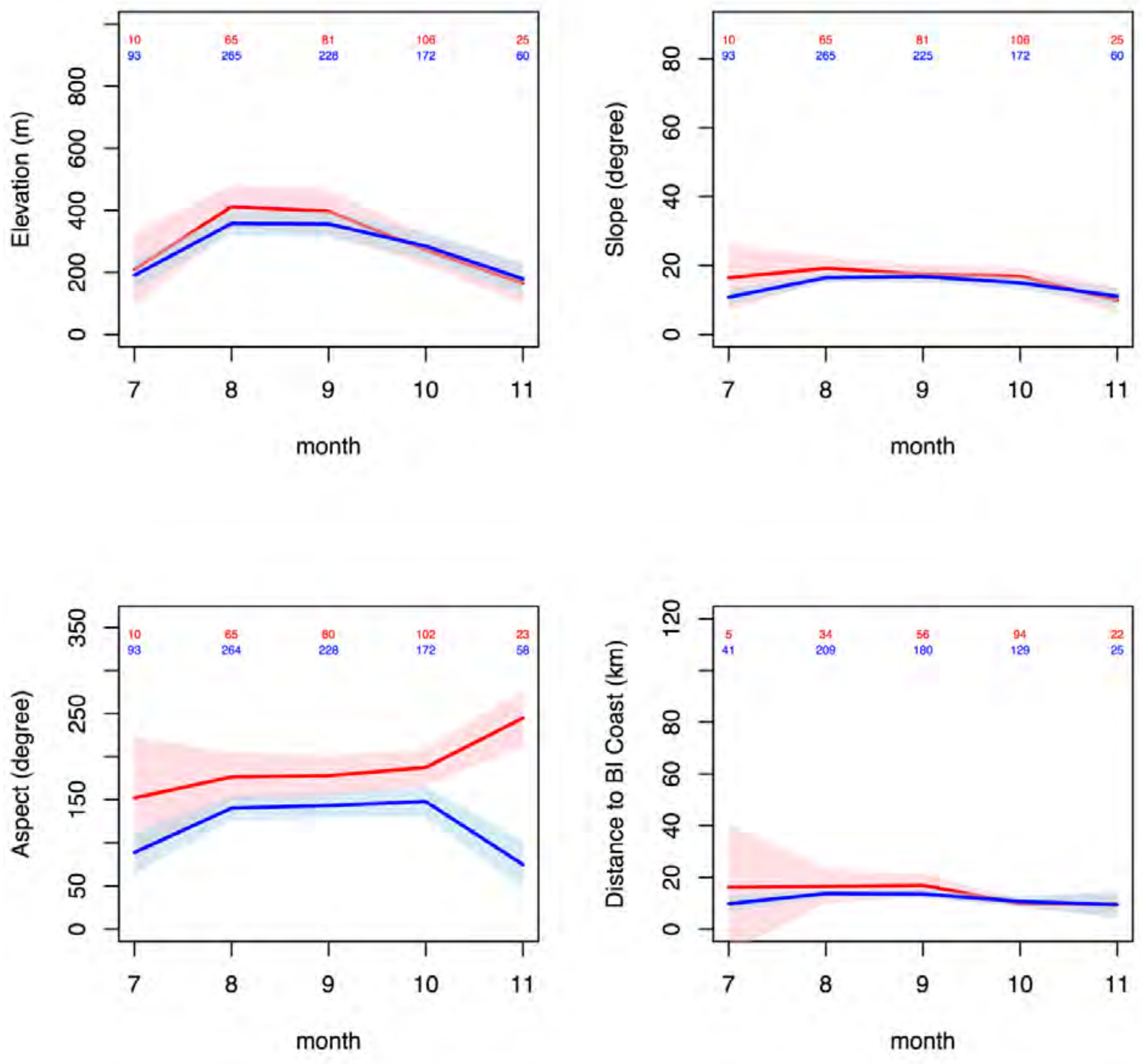


**Figure 4.12.** Map of ASTER Digital Elevation Model (DEM) data and IBCAO bathymetry shown for Baffin Island used in the RSF analyses. Partial tracks from a single adult female (68005) collared in 2009 in West Greenland are shown for reference.

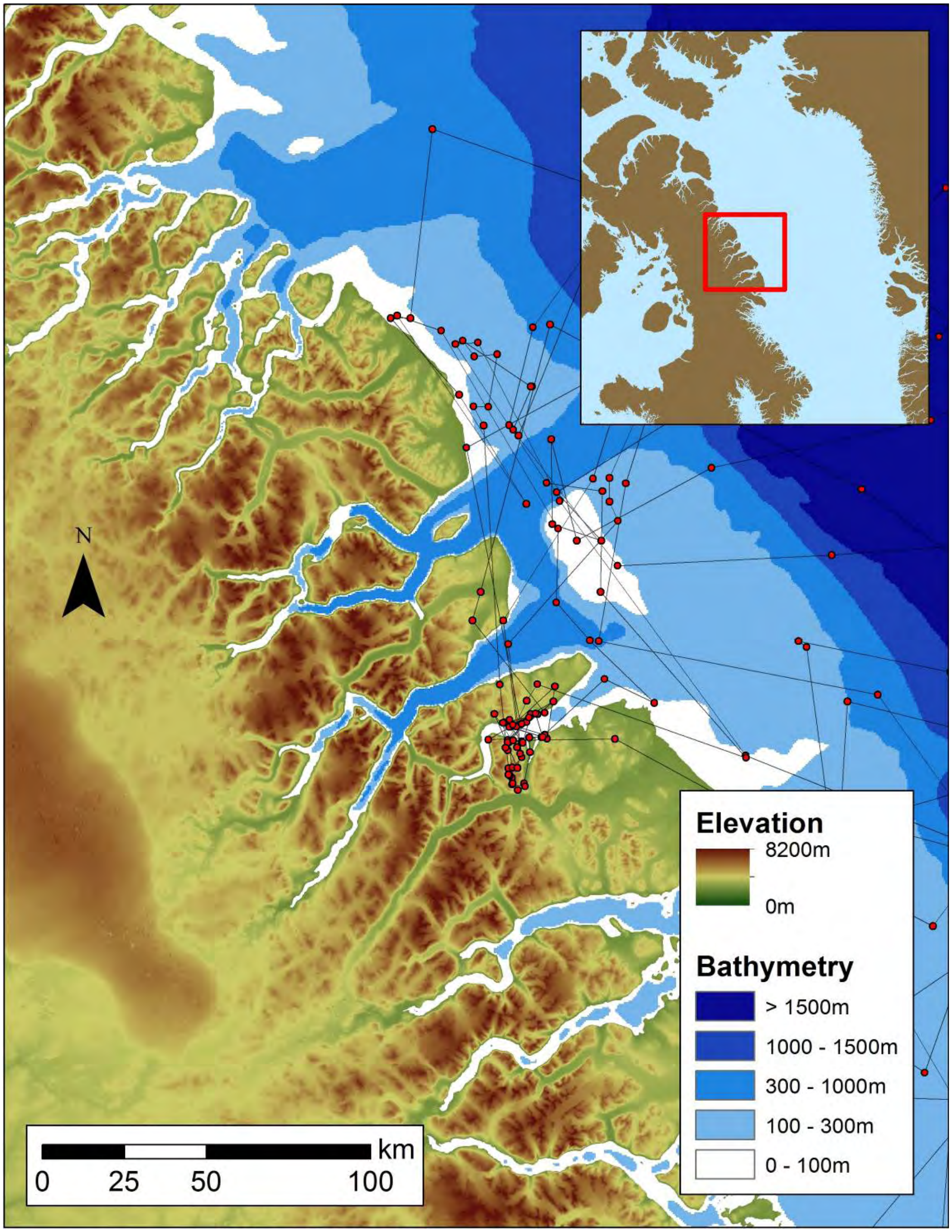


**Figure 4.13.** Arrival dates (on land) for adult female BB polar bears in the 1990s and 2000s. This does not include two BB bears arriving on Ellesmere Island in the 1990s. Also bears in glacial fronts in Melville Bay not included as they remained in coastal habitat year-round.

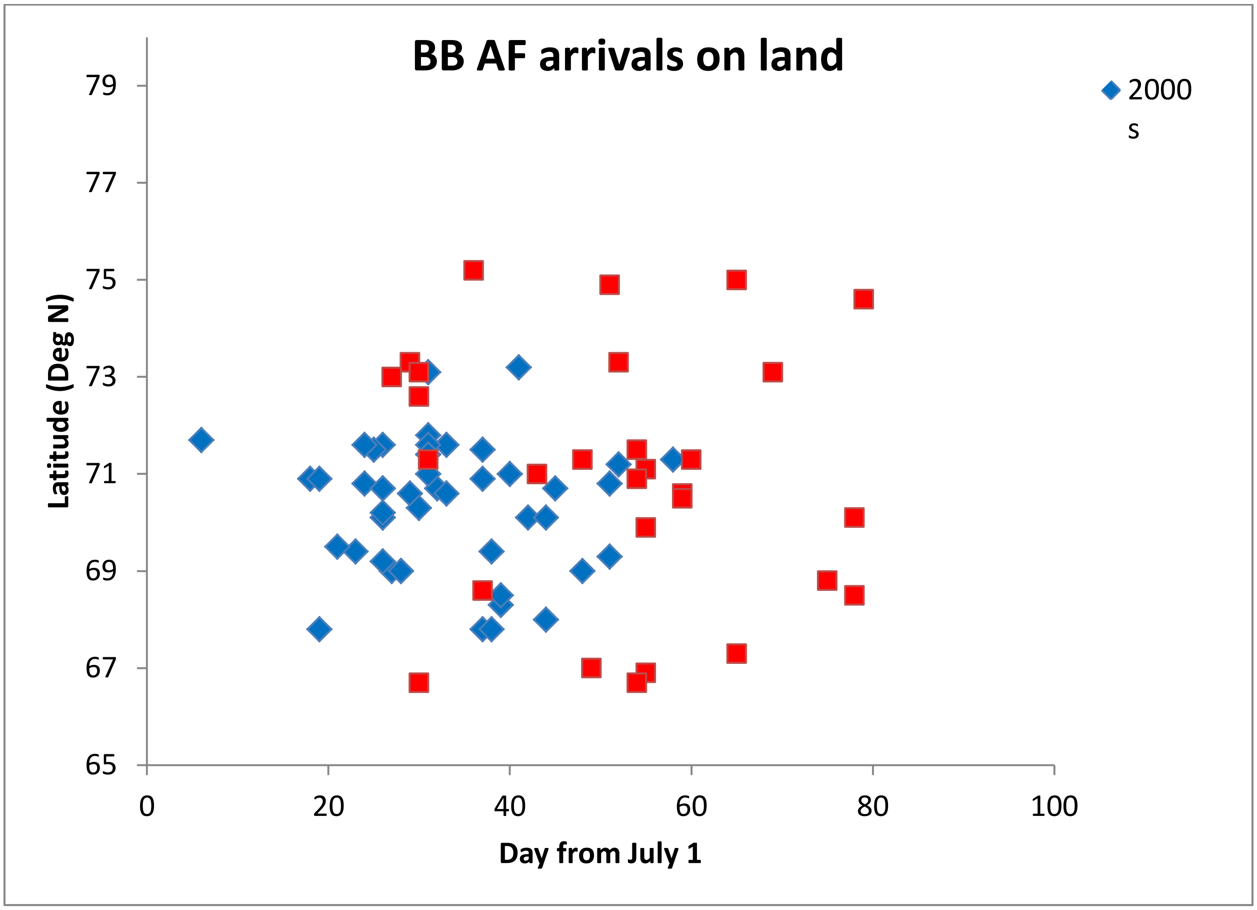


**Fig 4.14.** Departure dates in fall (off land) for adult female BB polar bears in the 1990s and 2000s.

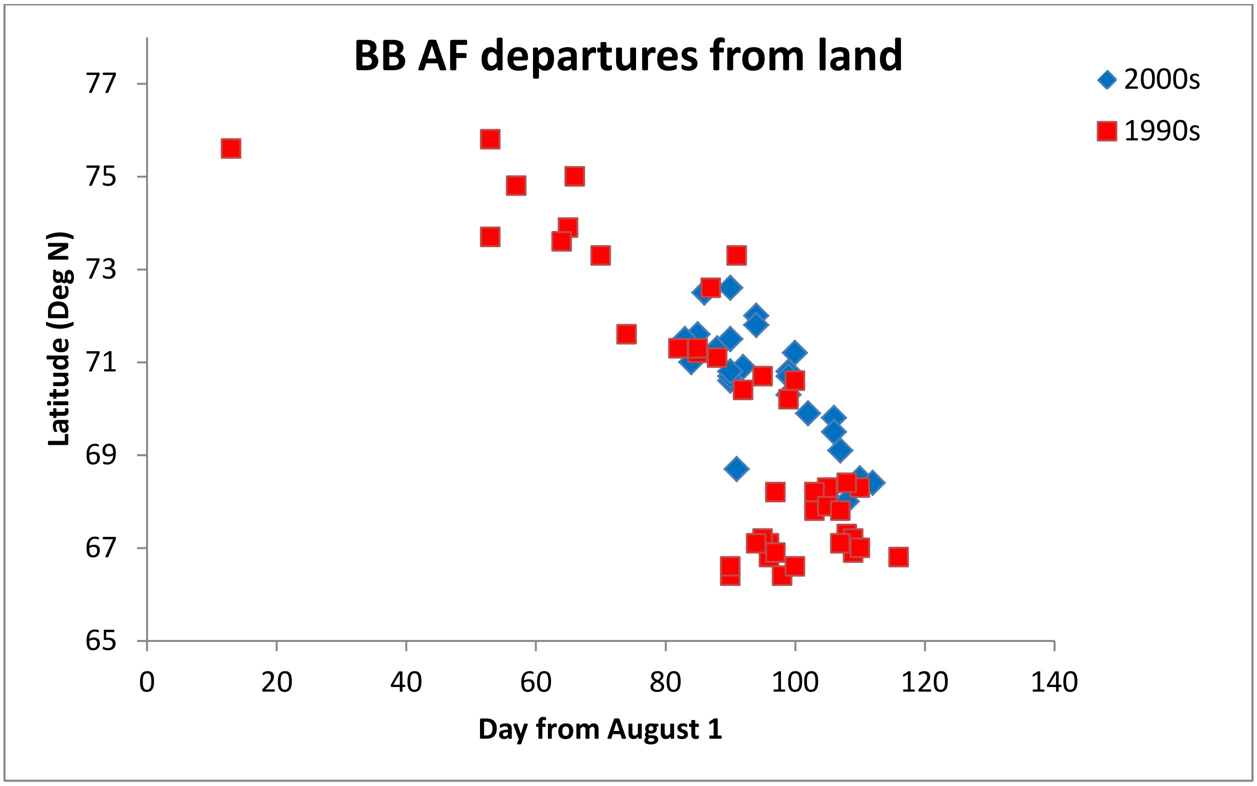


**Figure 4.15.** Correlations between the sea-ice retreat and advance metrics (see methods) and the arrival and departure dates on Baffin Island for adult females in both decades.

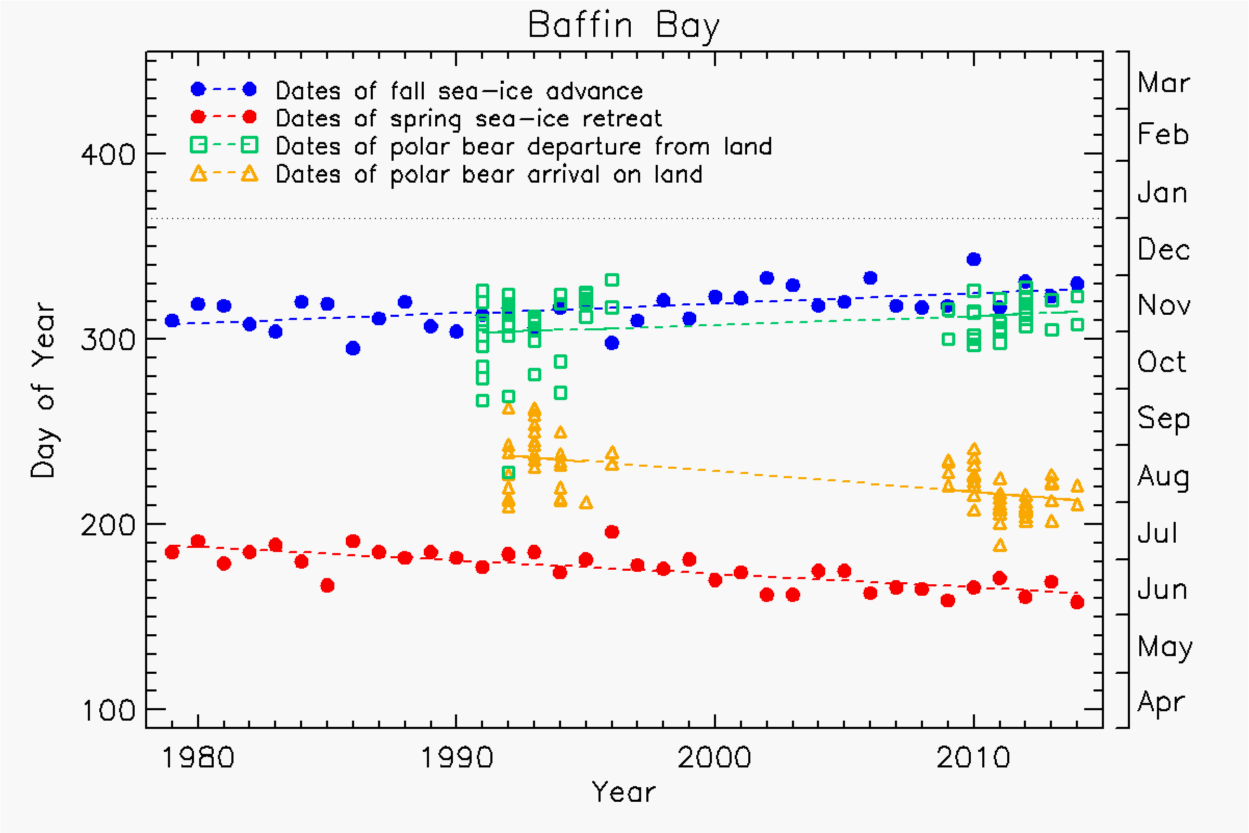


**Figure 4.16.** Distance to land vs. time plot for polar bear captured in 1992 (PTT 14411) showing a potential swimming event in September 1993. Purple squares denote the departure date from land on to sea ice and purple triangles denote the arrival date from sea ice/water on to land for that individual.

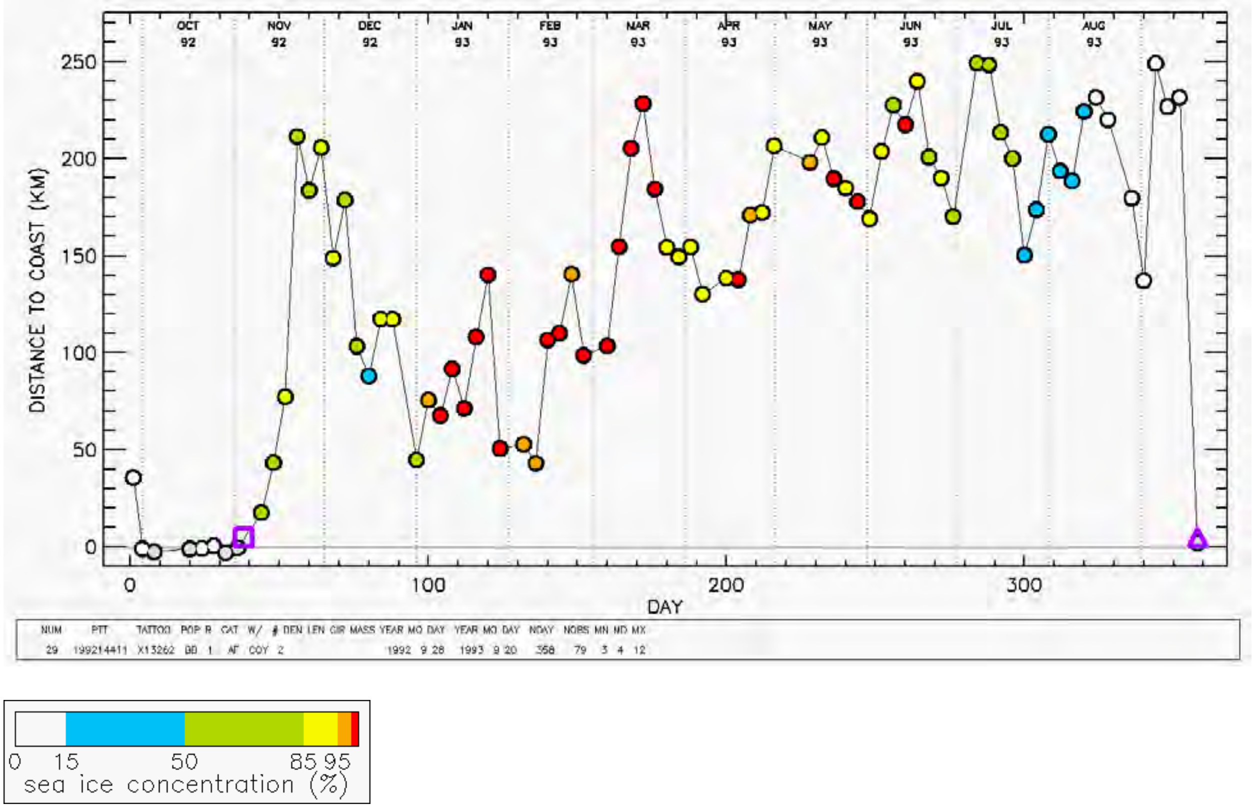


**Figure 4.17.** Distance to land vs. time for polar bear captured in 2011 (PTT 105808) showing an example of a swimming event in both July 2011 and July 2012. Purple squares denote the departure date from land on to sea ice and purple triangles denote the arrival date from sea ice/water on to land for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

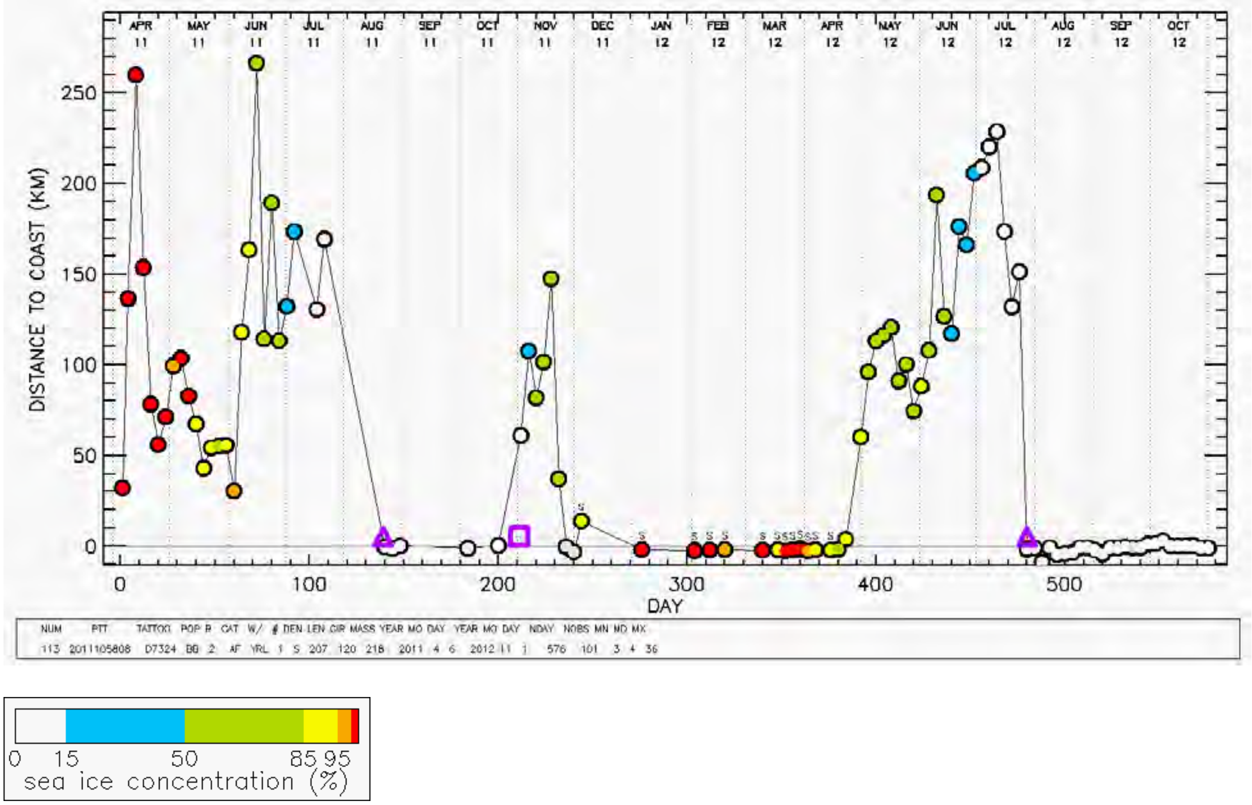
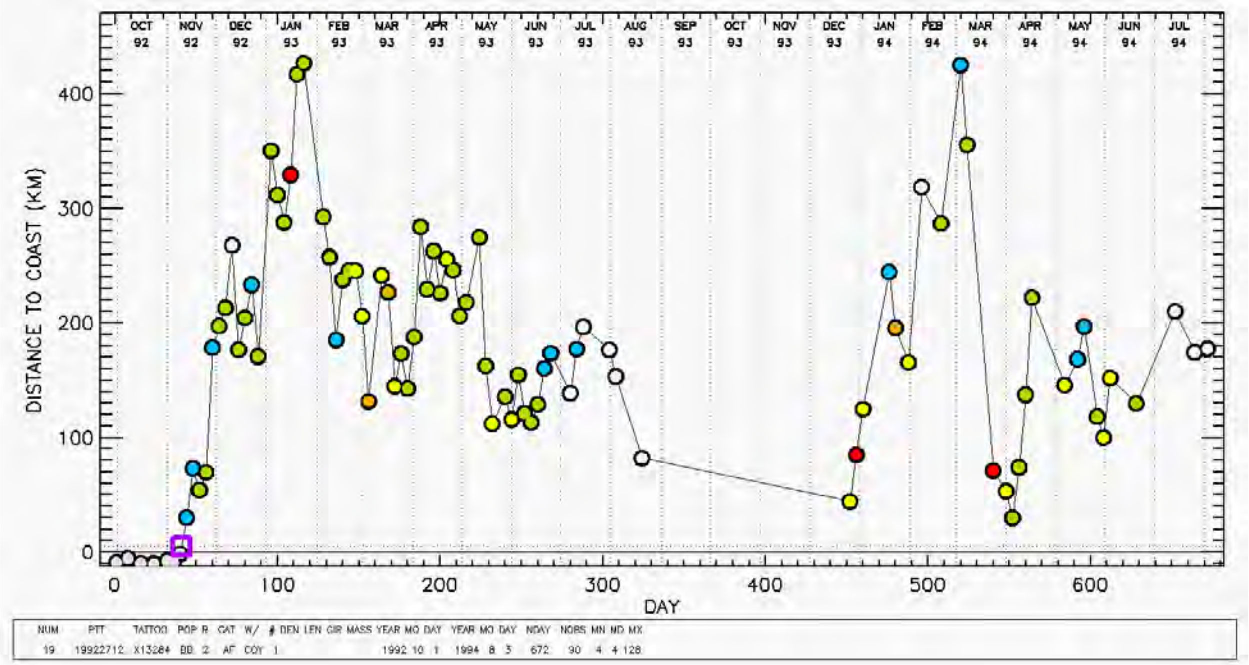
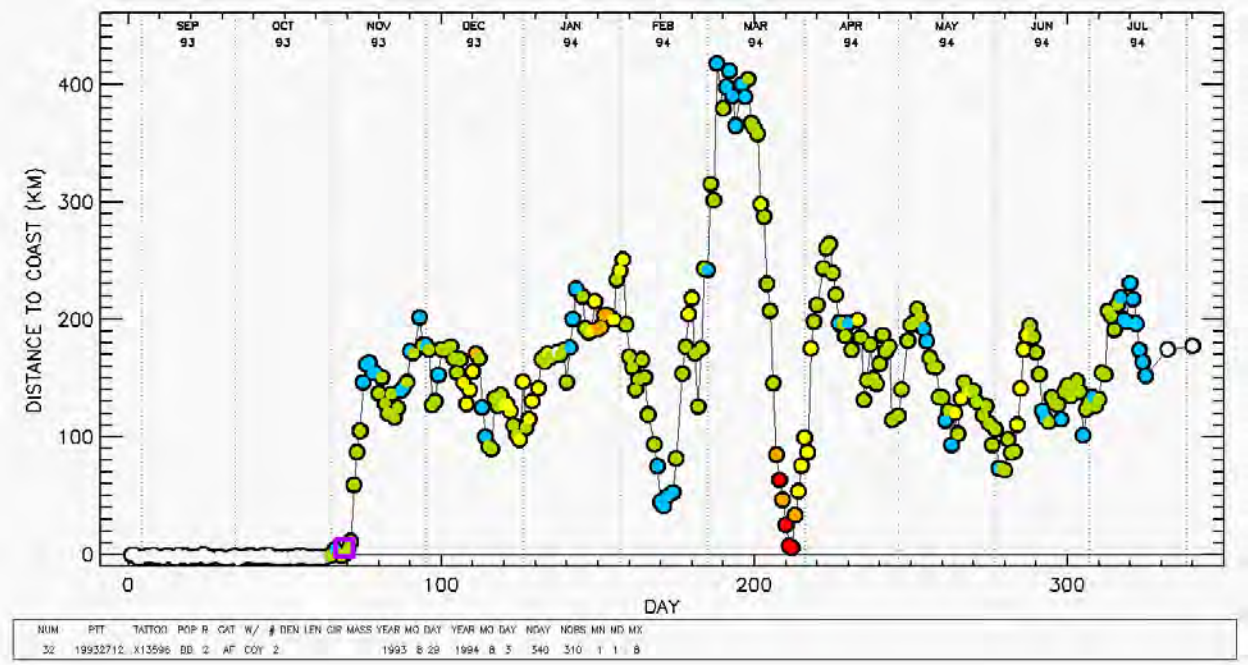


**Figure 4.18.** **a**) Track of adult female bear tagged in 1992 where the last three locations from the individual occur between July-August 1994 in open water 180 km from land; **b)** track of adult female bear collared in 1993 through August 1994, where the last two locations occur in open water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

**a)**



**b)**



**c)**

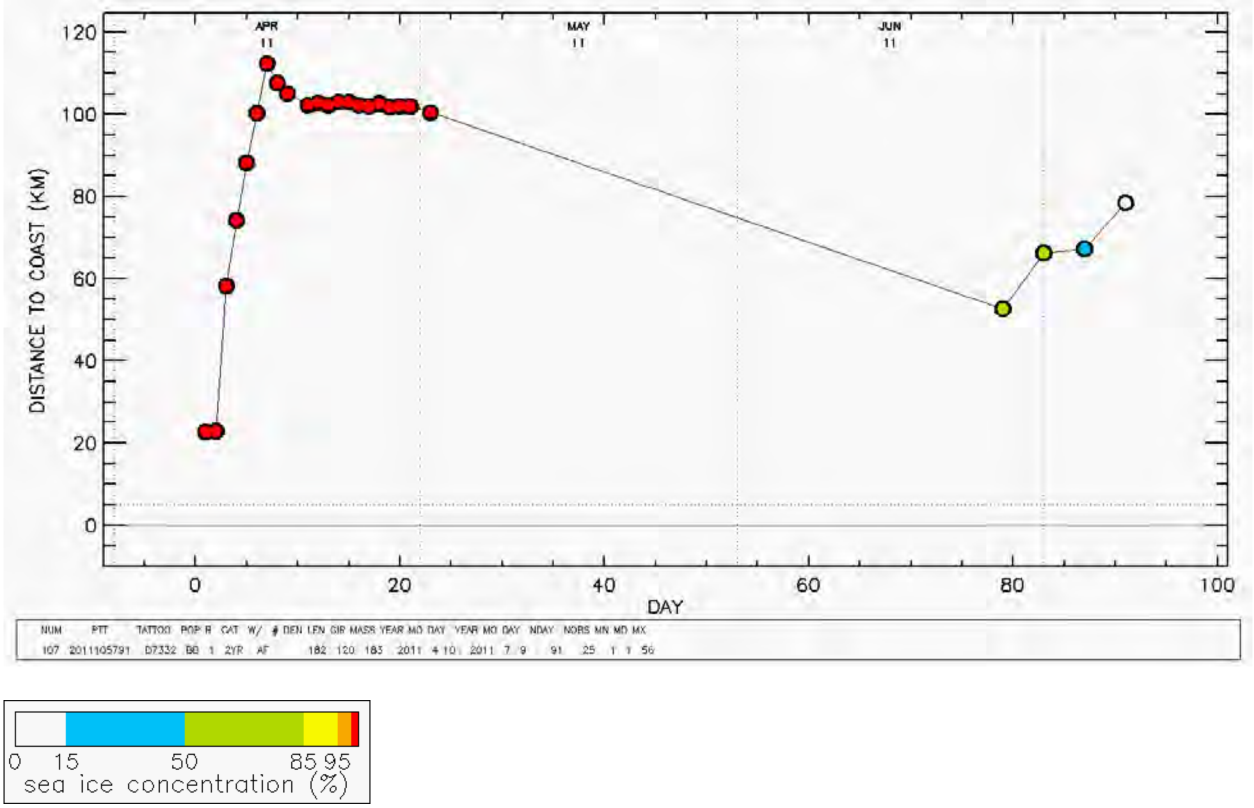

APR 11
MAY 11
JUN 11
DISTANCE TO COAST (KM)
120
100
80
60
40
20
0
0
20
40
60
80
100
DAY
NUM PTT TATTOO POP R CAT W/ # DEN LEN GIR MASS YEAR MO DAY YEAR MO DAY NDAY NOBS MN MD MX
107 2011105791 D7332 B0 1 2YR AF 182 120 183 2011 4 10 2011 7 9 91 25 1 1 56
0 15 50 85 95
sea ice concentration (%)

**Figure 4.19.** Maternity den locations in BB by decade.

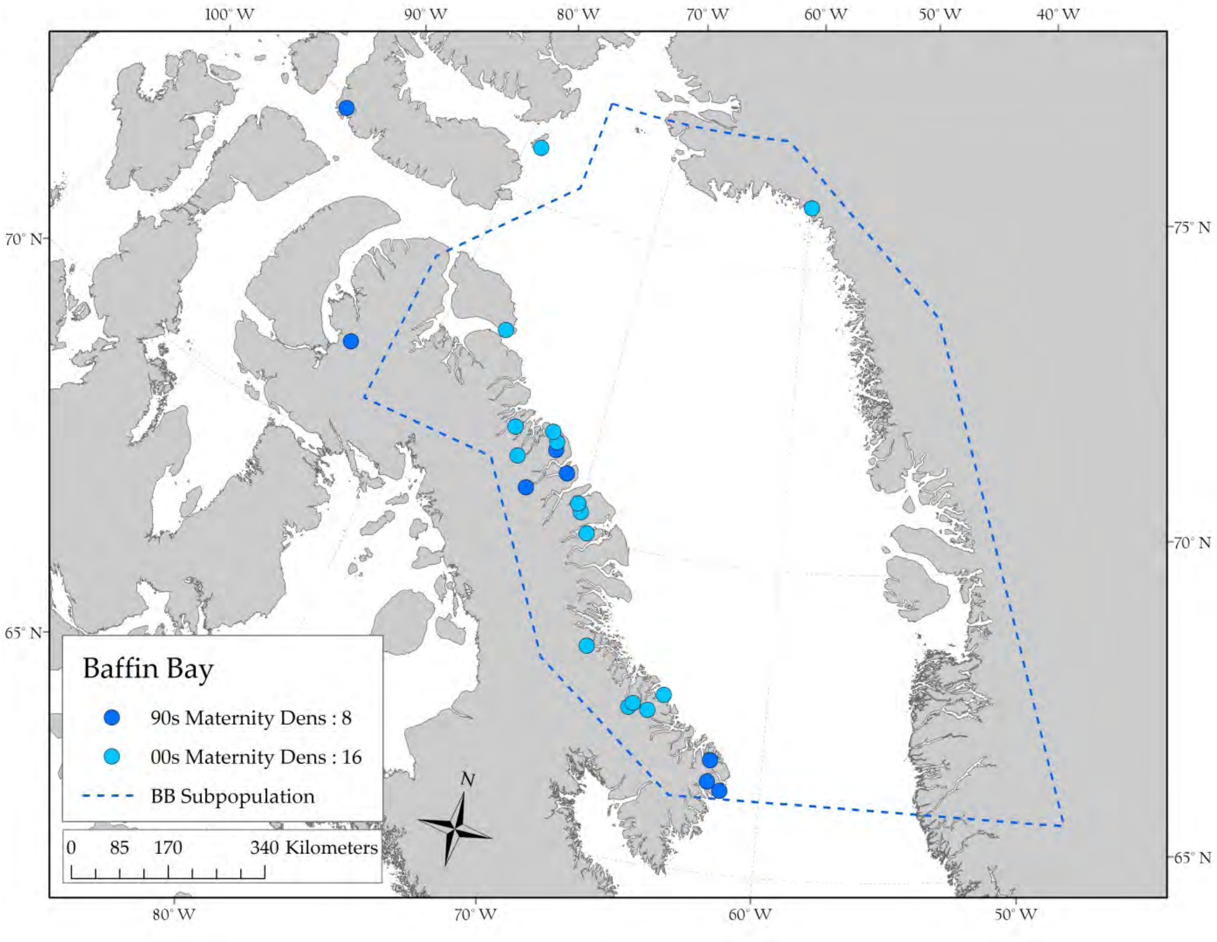


**Figure 4.20.** Shelter den locations in BB by decade.

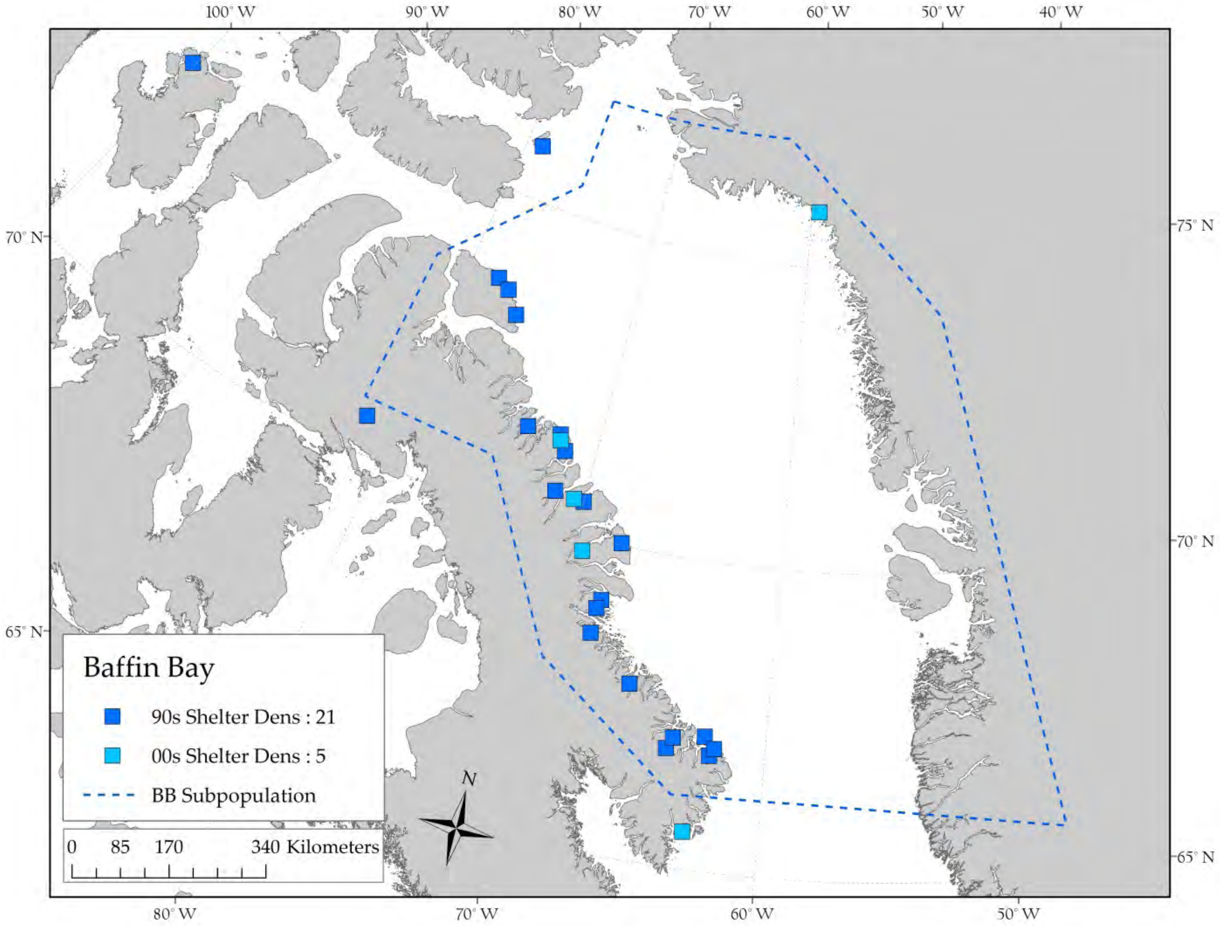


**Figure 4.21.** Boxplots comparing maternity den duration of Baffin Bay maternity dens ($p$ = 0.017) (1990s: $n$ = 8; 2000s: $n$ = 16).

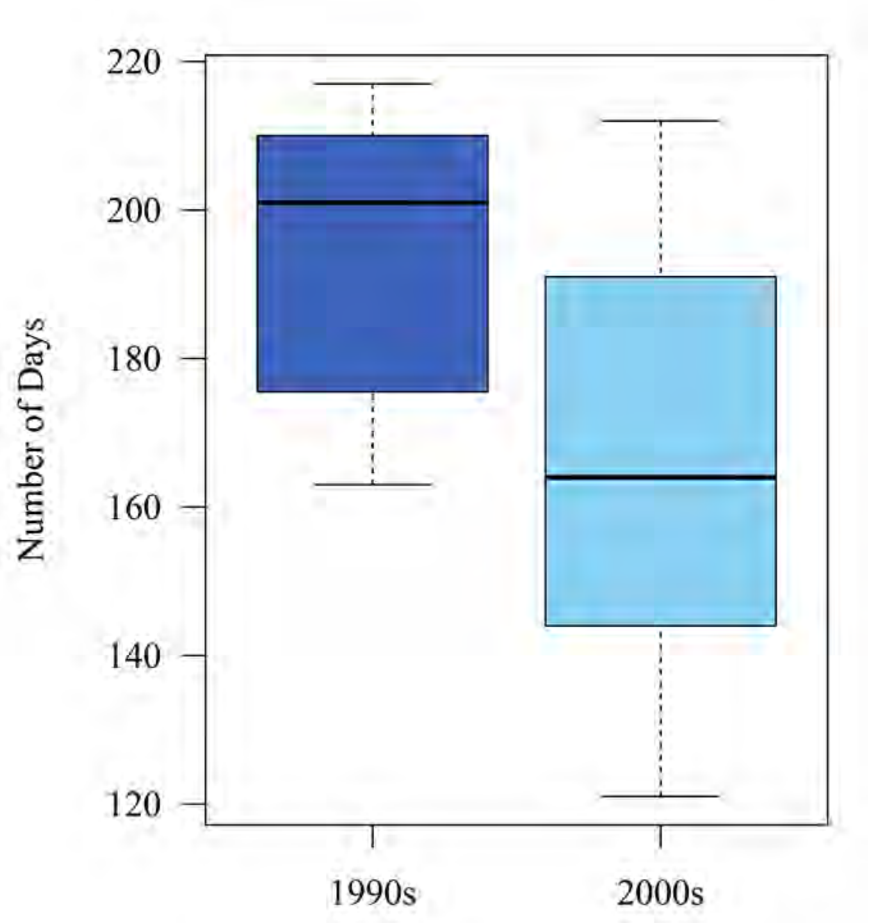


**Figure 4.22.** Boxplots comparing entry ($p = 0.018$) and exit dates ($p = 0.399$) of Baffin Bay maternity dens (1990s: $n = 8$; 2000s: $n = 16$)

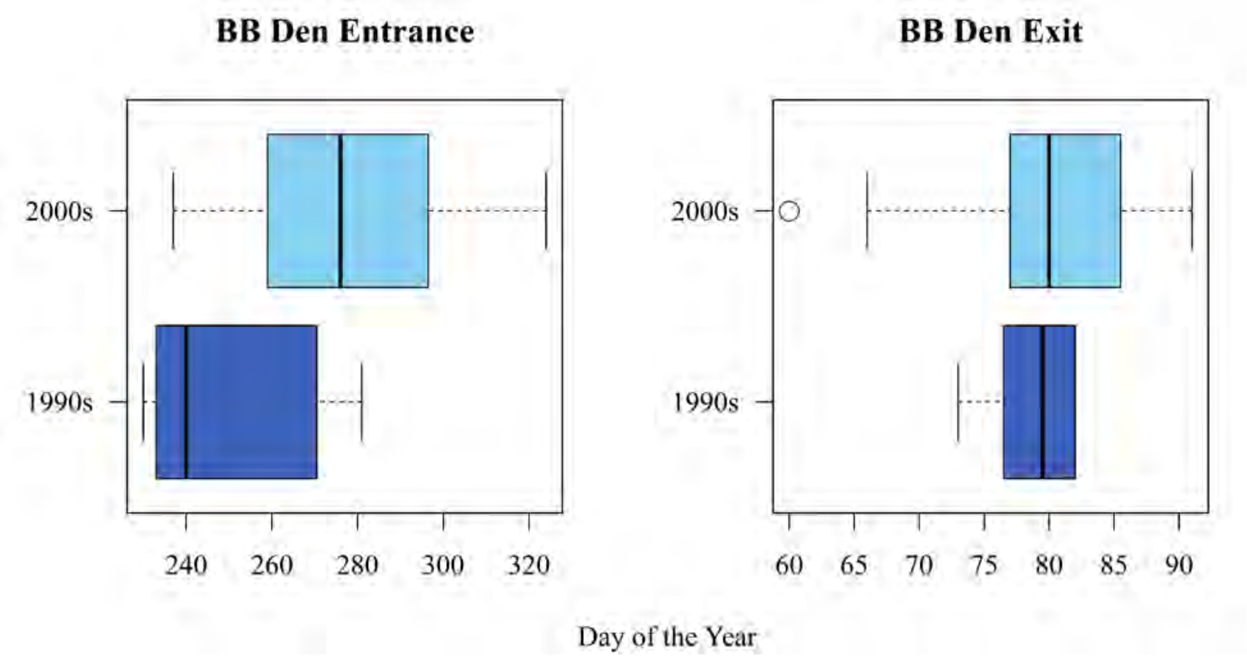


**Figure 4.23.** Boxplots comparing the first date on land (FDOL) of pregnant females in BB in the 1990s (*n* = 8) and 2000s (*n* = 16) (First FDOL used; *p* = 0.002).

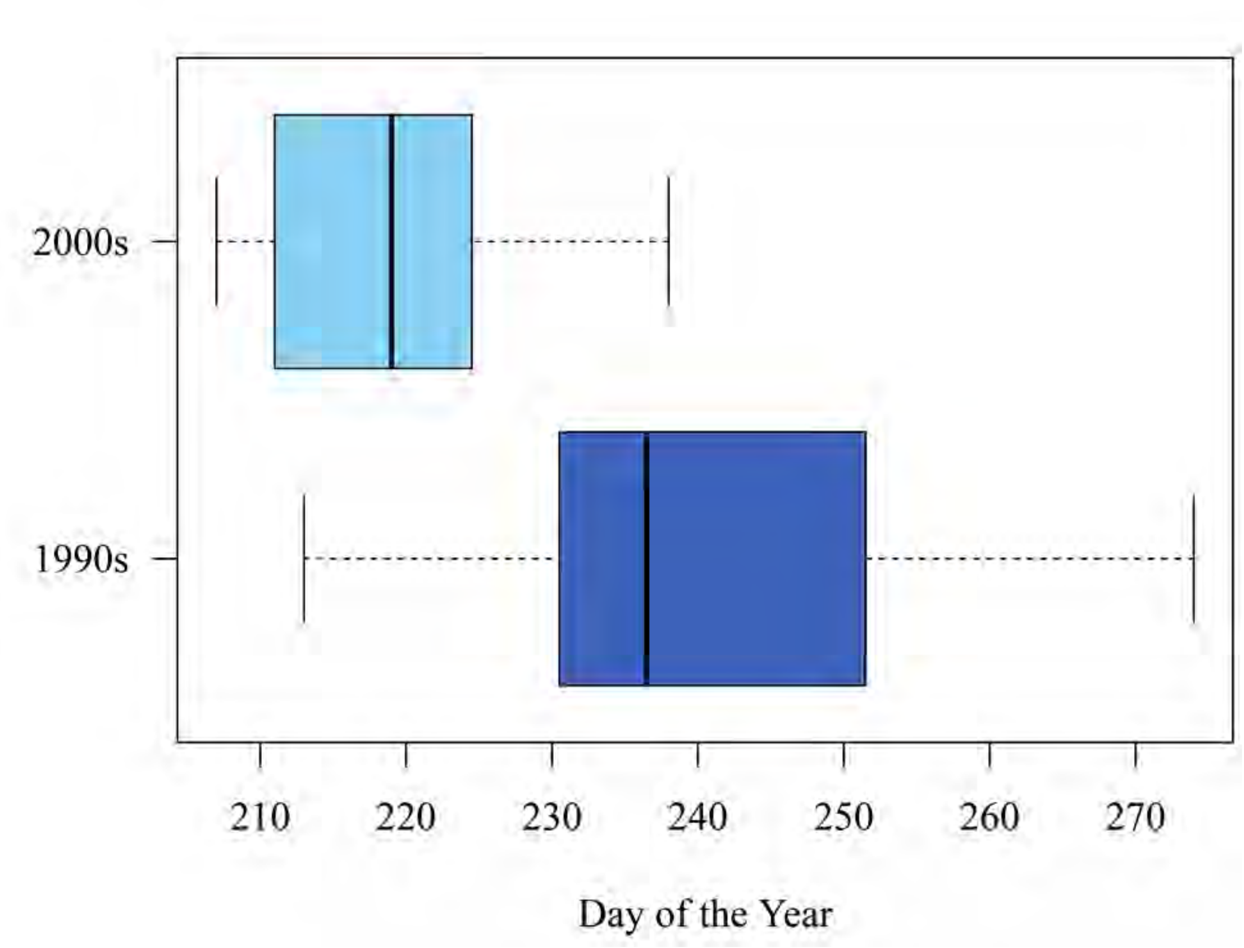


**Figure 4.24.** Plots comparing the aspect, slope, elevation, and distance to coast of the 1990s ($n$ = 8) and 2000s ($n$ = 15) maternity dens in Baffin Bay (the den on Greenland was omitted). The aspect plot consists of a compass face with lines marking the directions that dens faced. The lines are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

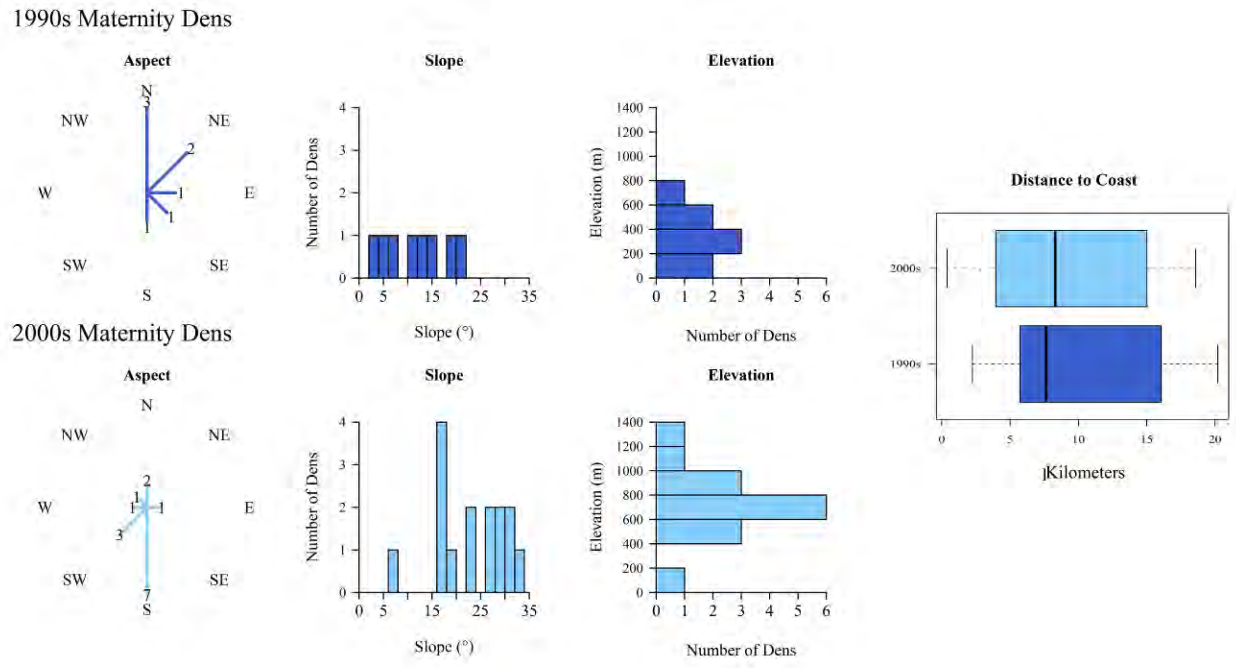
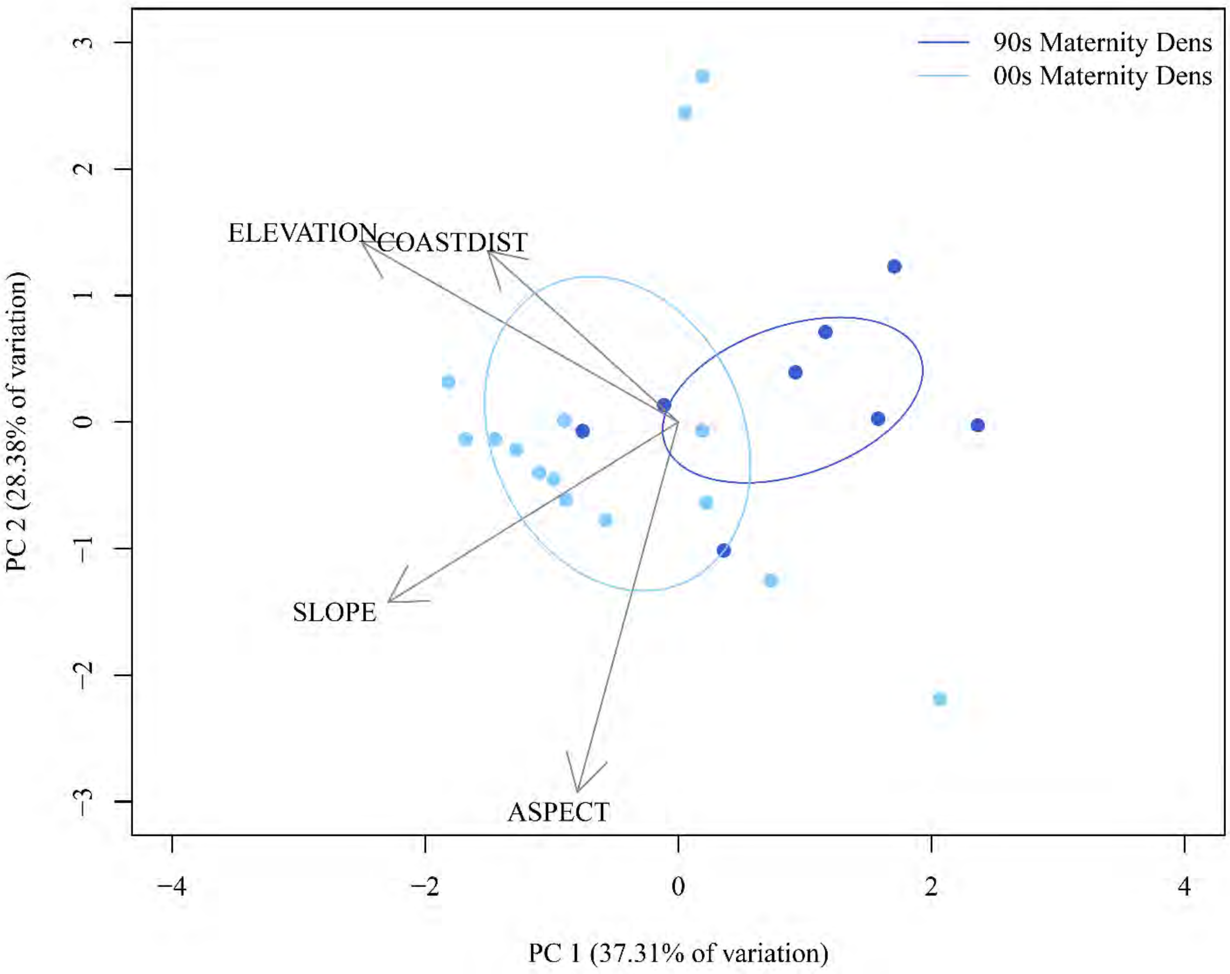


**Figure 4.25.** Biplot symbolizing the results for the principal component analysis (PCA) ordination of the Baffin Bay maternity dens and their habitat descriptors (elevation, slope, aspect, and distance to coast or 'coastdist'), with ordiellipses drawn around year groups (1990s and 2000s; confidence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



# CHAPTER 5

# GENETIC MARK-RECAPTURE STUDY OF POLAR BEARS IN BAFFIN BAY

## KEY FINDINGS

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

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

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

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

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

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

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

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

## 5.1. Introduction

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

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

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

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

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

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

## 5.2. Materials and Methods

### Study Area

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

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

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

## Mark-Recapture Sampling Design

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

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

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

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

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

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

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

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

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

### Field Methods

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

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

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

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

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

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

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

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

## Harvest recoveries

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

## Genetic Analysis

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

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

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

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

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

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

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

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

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

### Statistical Analysis

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

## 5.3. Results

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

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

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

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

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

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

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

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

## 5.4. Discussion

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

**Table 5.8.** Model averaged (<Δ 4 $\mathrm{QAIC}_c$) parameter estimates obtained from mark-recapture study of polar bears in the Baffin Bay subpopulation, 1993 – 2013.

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

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

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

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

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

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

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

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

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

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

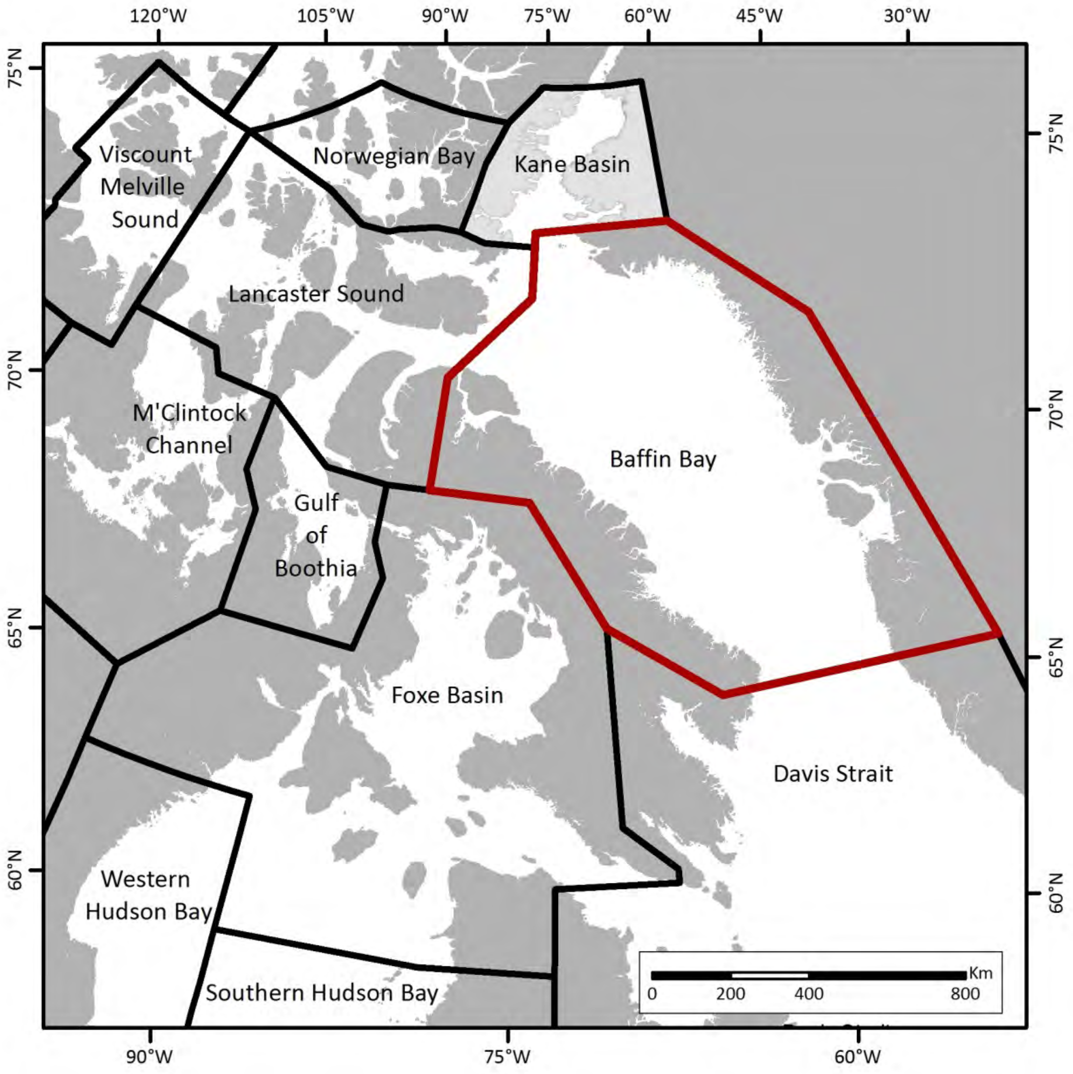


**Figure 5.2.** Locations of polar bears captured in Baffin Bay during the 1970s, 1980 - 1985, and 1989 – 1993.

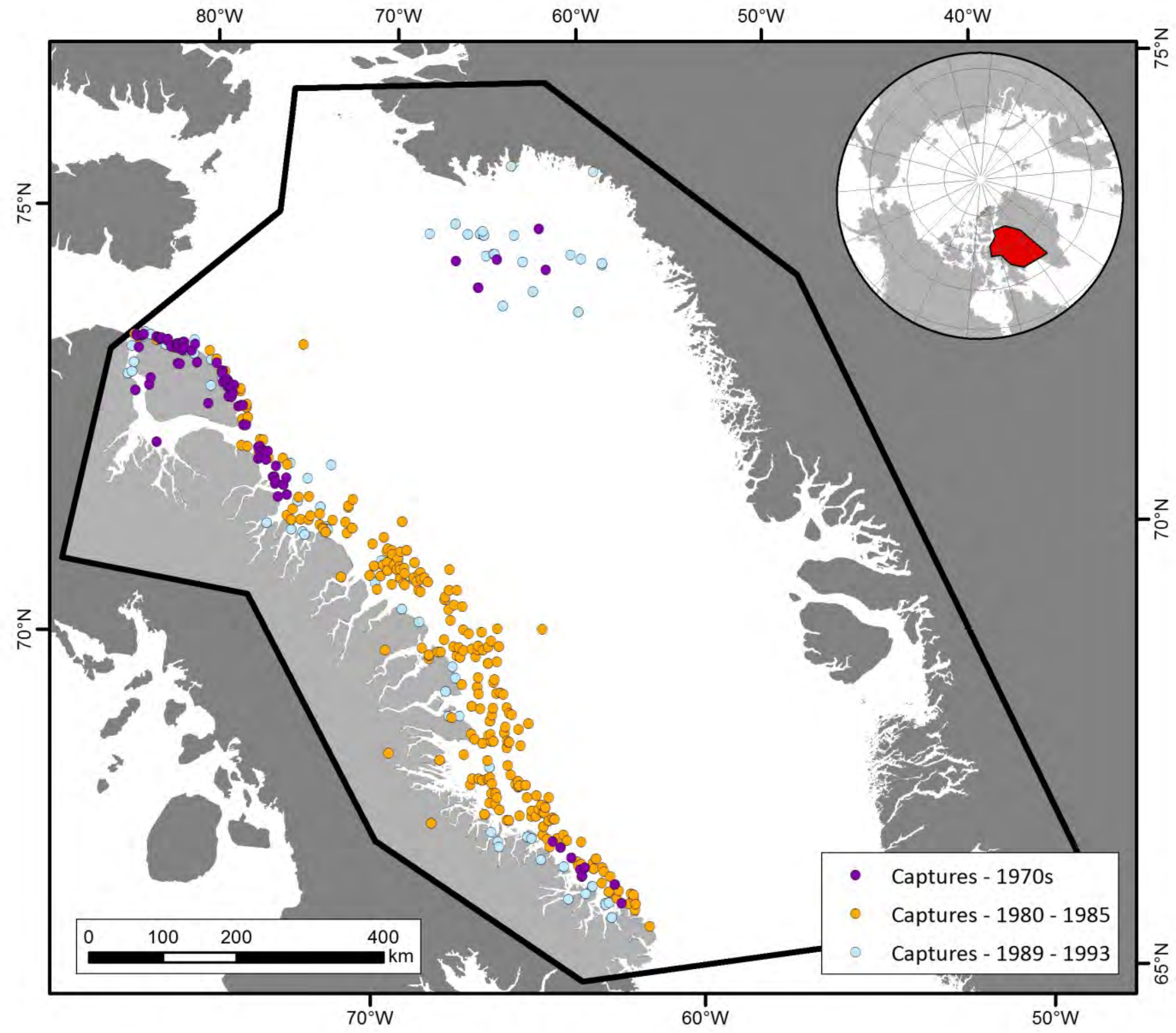


**Figure 5.3.** Locations of polar bears captured in Baffin Bay during August – October, 1993 – 1995 and 1997.

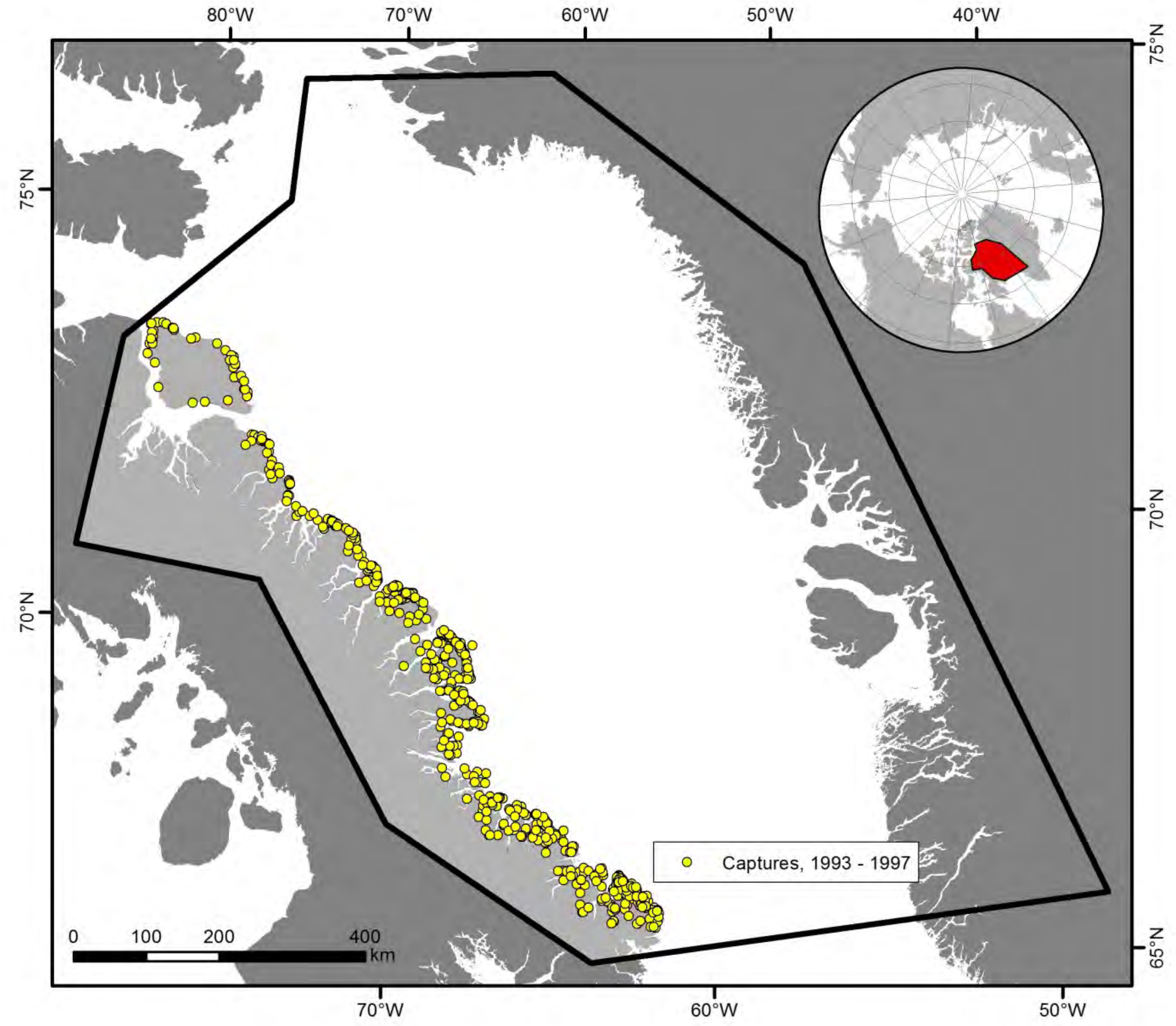


**Figure 5.4.** Sampling strata delineated on Baffin Island, Canada, for genetic mark-recapture study completed during 2011 – 2013. Fall sampling also was completed in the nearshore regions around Melville Bay, Greenland, denoted by the yellow star.

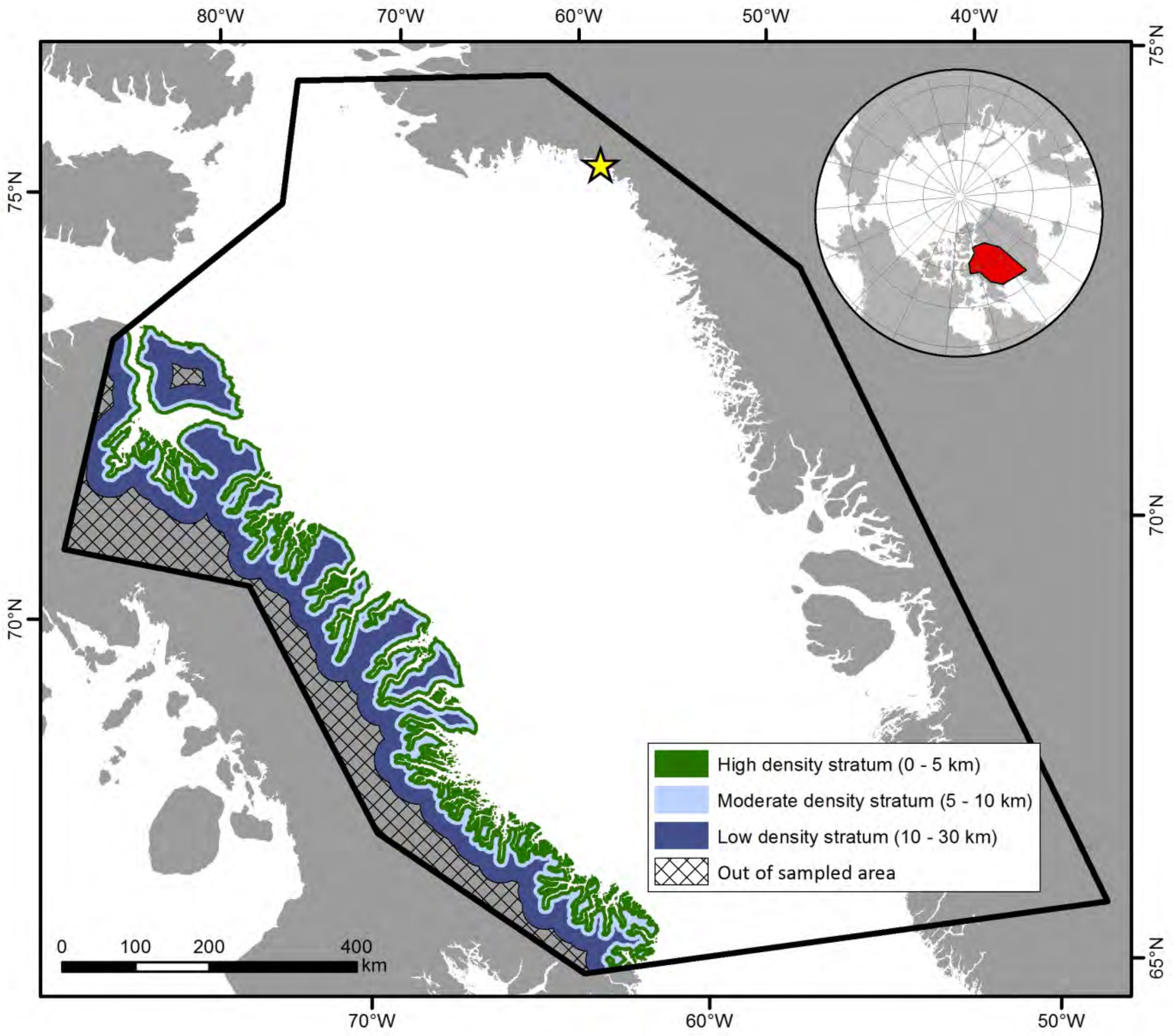


**Figure 5.5.** Left: Pneudart Inc. DNA (top) and Biopsy (bottom) darts used during the 2011 genetic-mark recapture in Baffin Bay. Right: A sample of skin and fat provide by a DNA dart.

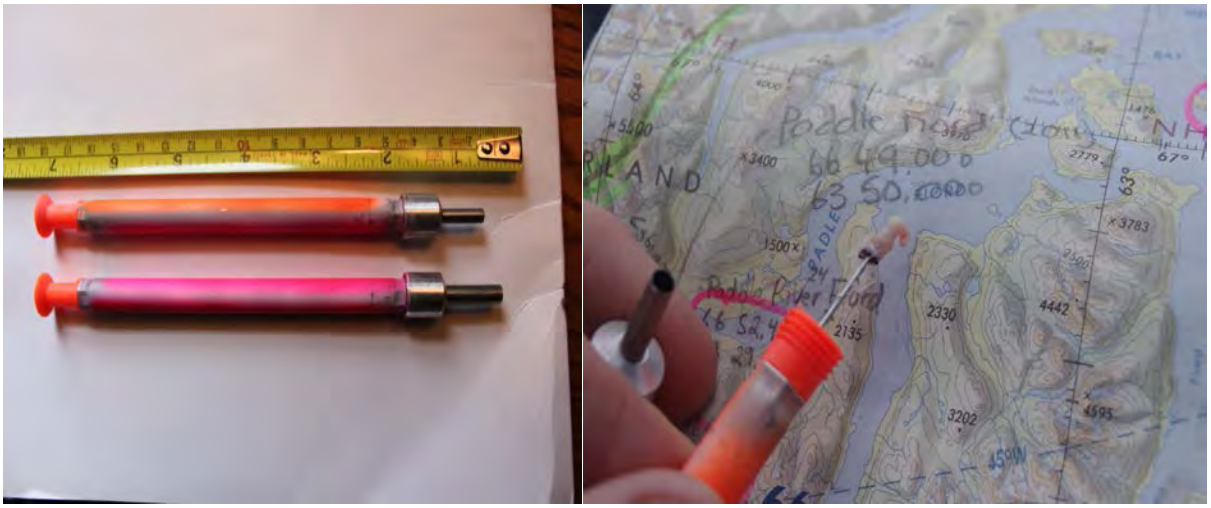


**Figure 5.6.** 9-locus mismatch distribution for 4,657 polar bears from Nunavut and the Greenland side of BB.

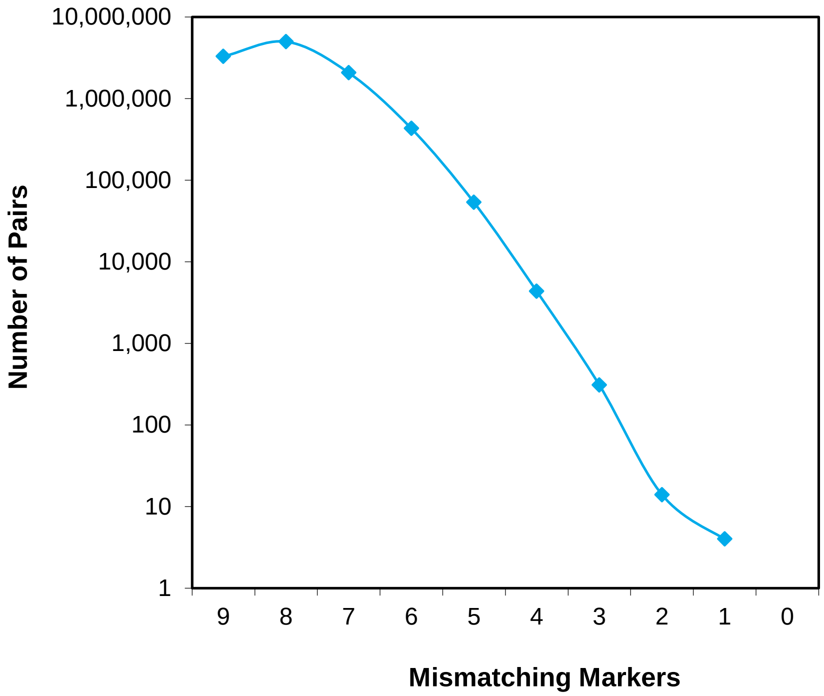


**Figure 5.7.** Locations of marked BB polar bears recovered in the harvest in BB and surrounding subpopulations during 1993 – 2013.

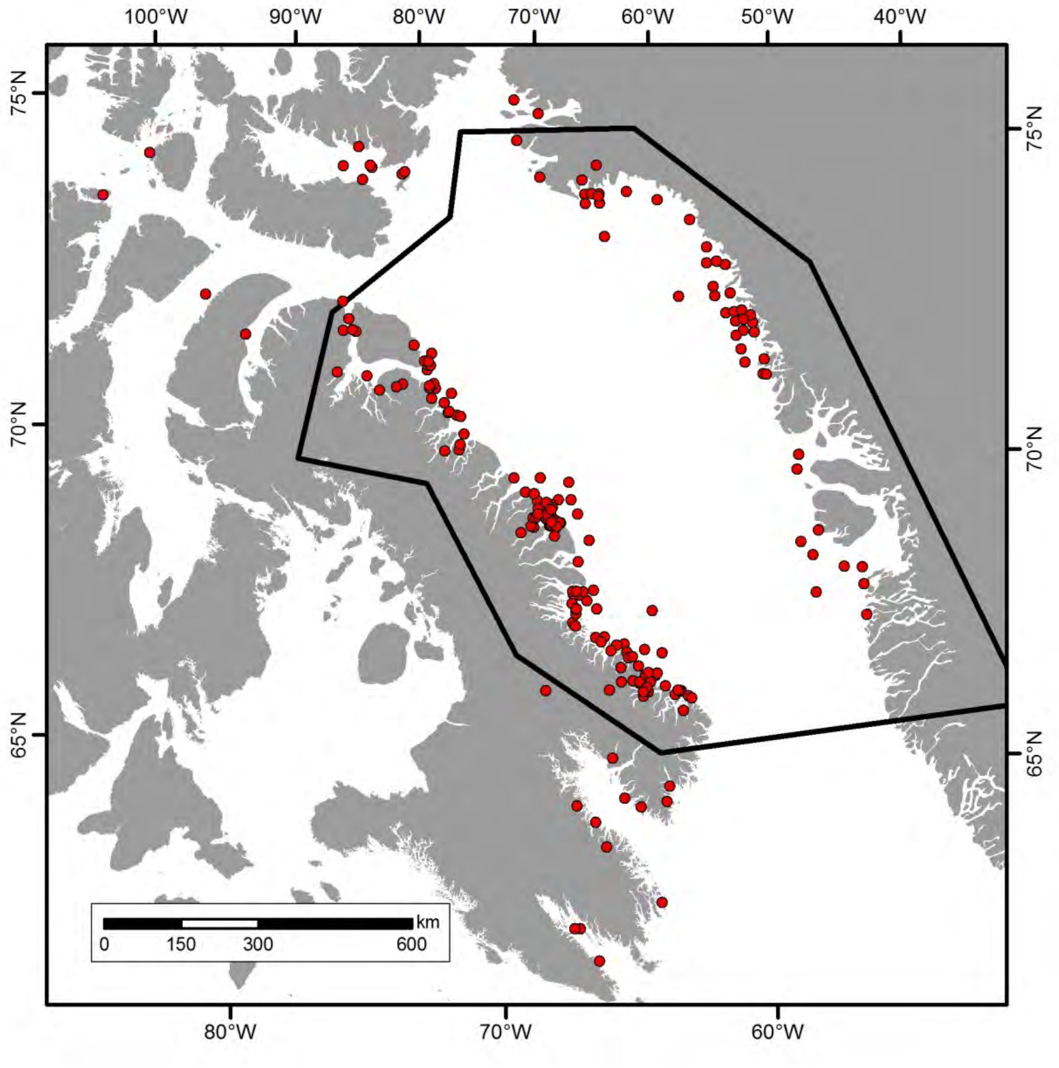


**Figure 5.8.** Locations of polar bears sampled in Baffin Bay during August – October, 2011 – 2013. Sampling in Greenland occurred near Melville Bay.

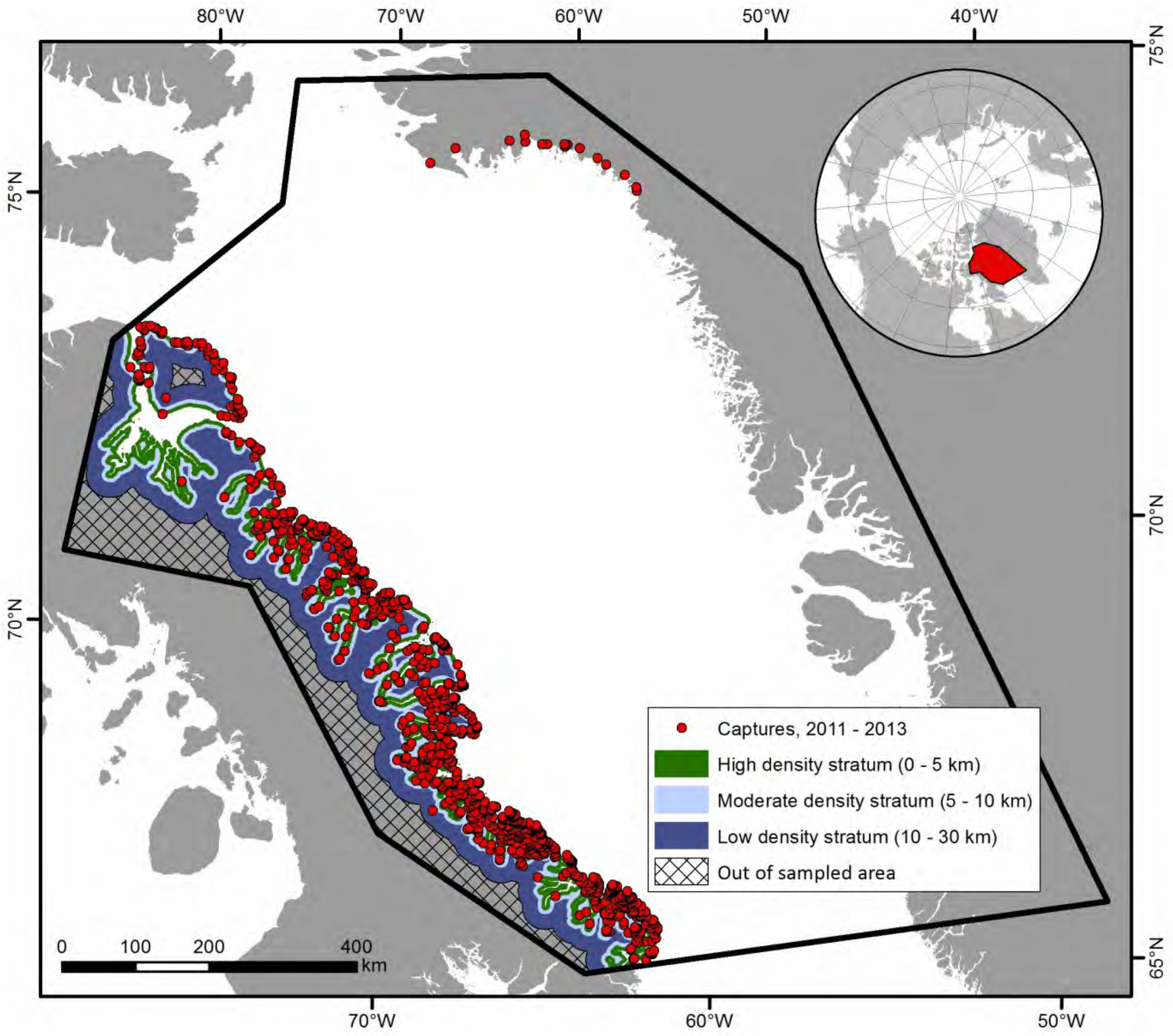
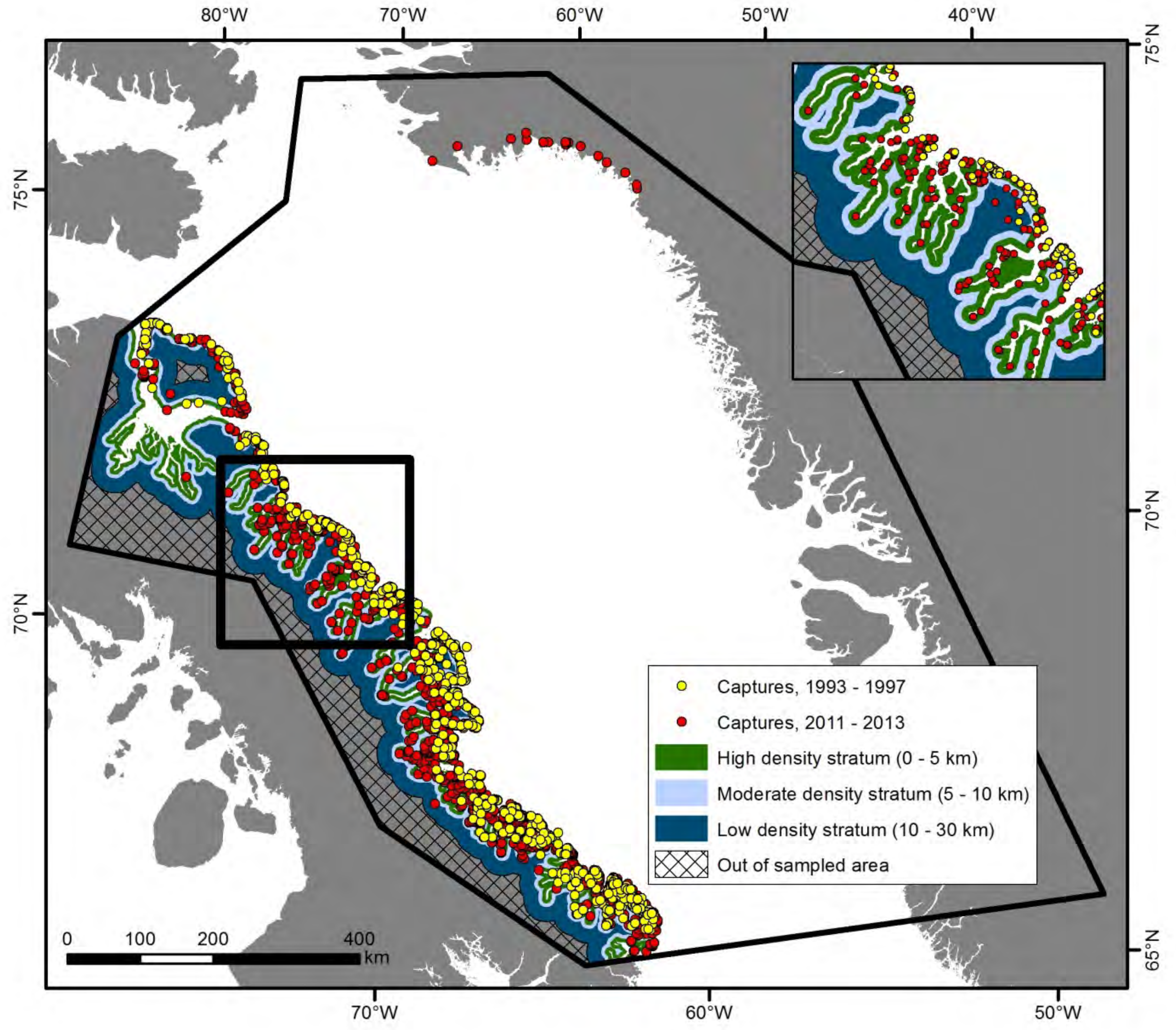


**Figure 5.9.** Locations of polar bears sampled in Baffin Bay during August – October, 1993 – 1995, 1997 and 2011 – 2013. The region bounded by the black square is enlarged in the inset. Note the absence of captures in fjords on Baffin Island and in northwestern Greenland during the 1990s.



# CHAPTER 6

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

## KEY FINDINGS

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

## 6.1. Introduction

For populations of large, long-lived mammals, changes in reproductive performance can be one of the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1981, 1987). In populations approaching carrying capacity, declines in reproductive performance are likely to occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring indices or metrics of reproduction may therefore provide a useful tool for detecting potential population trends that may warrant more in-depth study. This is especially true for populations in which cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring reproductive metrics may provide a form of surveillance that can be used to trigger more intensive study.

Reproductive metrics have been identified as an important component for monitoring polar bears across their circumpolar range (Vongraven et al. 2012). These metrics may be used to track long-term trends in the status of subpopulations, parameterize population viability models and support harvest risk assessments (Regehr et al. 2015). In particular, changes in reproduction are predicted to be amongst the first subpopulation-level effects of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Indeed, declines in reproduction have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (Regehr et al. 2007, Rode et al. 2010, Rode et al. 2014).

Changes in reproductive metrics can signal significant changes in subpopulation status of polar bears. However, observations of poor reproductive performance alone do not necessarily imply a decline in subpopulation status. Studies of several polar bear subpopulations have documented declines in reproduction associated with increases in abundance that may be the result of density dependence (Derocher 2005, Peacock et al. 2013). In other cases, variation in reproductive performance within or amongst subpopulations has been attributed to geographic or annual variation in biological productivity and prey availability (i.e., variable carrying capacity; Stirling and Øritsland 1995, Stirling and Lunn 1997, Stirling 2002, Rode et al. 2014). Information on reproduction must therefore be considered alongside other measures of subpopulation performance in-order to properly assess status.

The Baffin Bay (BB) subpopulation is part of the seasonal ice ecoregion as defined by Amstrup et al. (2008) in which sea ice melts almost entirely in the summer and bears are forced ashore for extended periods of time, during which they have no or reduced access to food. Baffin Bay has experienced a long-term reduction in sea-ice cover and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapter 3). As a result, bears are spending an increasing amount of time on land. Previous studies have documented a decline in body condition amongst BB bears in association with these trends in sea ice (Rode et al. 2012), and similar trends were found in the current study (see Chapter 7), however changes in reproductive metrics in BB have not been reported.

We summarized reproductive metrics for the Baffin Bay polar bear subpopulation using data collected during two periods of mark-recapture sampling from 1993 to 1997 (Taylor et al. 2005) and 2011 to 2013 (this report). Annual variation in reproduction was examined to assess trends over time and to evaluate the hypothesis that reproductive performance varied with sea-ice conditions. Because Baffin Bay has been infrequently monitored, we also sought to assess the utility of reproductive metrics as a surveillance tool for monitoring subpopulation status between periods of more in-depth demographic study, especially since some metrics can be obtained by methods that do not require physical capture (e.g., aerial surveys, harvest monitoring). Finally, we compared reproductive metrics for BB with other subpopulations to make inferences about the relative performance of this subpopulation. Results from these analyses provide context for understanding the status of BB polar bears.

## 6.2. Materials and Methods

### Study Area

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

### Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in BB, Canada. Sampling occurred along eastern Baffin and Bylot Islands during the ice-free season from late August to mid-October in 1993-95, 1997 and 2011-2013. Most bears in Baffin Bay move onto land on Baffin and Bylot Islands in late summer as the sea ice breaks up and remain on land until freeze-up in the late fall (Taylor et al. 2005). Sampling was extensive across this on-land study area during both periods (1990s and 2000s) of the study (Figure 6.1). The remote biopsy sampling in Greenland conducted in 2012 and 2013 were not included in this analyses.

Various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that a presumably relatively small proportion of the BB subpopulation summers in the Qimmusseriarsuaq / Melville Bay area of NW Greenland rather than moving with the retreating sea ice and summering on Baffin and Bylot Islands.

During the 1990s, bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2005). Data on the sex, estimated age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred by biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. We remotely estimated sex and age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) from the air at a range of 10-20 feet above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males, e.g., fore-leg guard hairs), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B).

### Reproductive Metrics

We calculated annual reproductive metrics that have been previously recommended (Vongraven et al. 2012) or used in studies of polar bears (e.g., Stirling et al. 1980, Derocher and Stirling 1995, Rode et al. 2010, Peacock et al. 2013, Regehr et al. 2015), including mean litter sizes (± SD) for cubs-of-the-year (COY) and yearlings. Because we did not have estimated ages for adult females sampled during 2011-2013, we calculated a pooled mean for each year rather than age-specific values. Recruitment indices were calculated as the total number of COYs or yearlings divided by the total number of adult females in the sample (Derocher and Stirling 1995, Regehr et al. 2015). Calculation of reproductive metrics considered only those COY and yearlings accompanying their mother at time of observation. During the 1990s, approximately 6% of yearlings were found to be independent of their mother during the ice-free period (GN unpublished data). These independent yearlings were not included in the calculation of yearling recruitment since we were not able to identify independent yearlings encountered during the 2000s sampling period with a known degree of accuracy.

Although Taylor et al. (2005) reported that search effort during the 1990s was uniform and systematic across the coastal regions, islands, and inland reaches of Baffin Island, recent examination of mark-recapture and telemetry data collected in the 1990s suggest sampling was spatially restricted to a portion of the BB subpopulation’s seasonal range and did not sample bears located farther inland or on the sea ice (GN unpublished data, Chapter 3). In contrast, sampling during 2011 – 2013 was more comprehensive and systematic on onshore areas, and the amount of un-searched sea ice during the sampling period was greatly reduced. To explore the potential impact of this difference in sampling between epochs on the calculation of reproductive metrics, we estimated and delineated the minimum extent of the sampling frame from the 1990s using capture locations in a GIS. We subset the 2000s data using this layer to create a dataset collected over the same sampling area in both time periods. We then recalculated reproductive metrics using this geographic subset. We expected that reproductive metrics calculated for the 2000s using the subset sampling area would reduce potential bias by adjusting for bears that may have been functionally missed by the limited geographic scope of sampling on land in the 1990s, given that satellite telemetry data indicated that onshore habitat use did not vary between epochs. However, we note that this geographic subset exercise would not correct for bears that may have been missed during the 1990s due to their location on the sea ice during the fall sampling period. Within season recaptures of individuals were excluded from both data sets for analyses.

### Sea-ice Metrics

Based on findings from other studies (Stirling et al. 1999, Durner et al. 2004, 2006, 2009, Regehr et al. 2007, Rode et al. 2010, 2014), we hypothesized that timing of sea-ice break-up over the continental shelf (< 300m depth) of BB may influence the annual reproductive performance of polar bears as indexed from data collected during the ice-free period. We therefore calculated the annual sea-ice metric, spring transition date, as day of the year (1-365) when ice area over the continental shelf of BB reached 50% in the spring (see Chapter 2 for methodology). Whereas some researchers have used lower sea ice thresholds for studying polar bears (e.g., Cherry et al. 2013, Rode et al. 2014, Obbard et al. 2016), the exact threshold level is less important in Baffin Bay because the changes in sea-ice area during spring and fall occur quickly, such that relatively small differences in transition dates result from small changes in the threshold values (Laidre et al. 2015). For sea ice in Baffin Bay, spring transition date is also correlated with other sea-ice metrics including fall transition date (negative correlation) and the interval between spring and fall transitions (Laidre et al. 2015).

We examined annual variation, temporal trends, and relationships to sea ice for reproductive metrics. Similar to Rode et al. (2014), we did not include both time and sea ice in the same analyses since long-term trends in the spring transition date were well-established *a priori* (Laidre et al. 2015, this report) and these two parameters were correlated over the 7 years of sampling (Pearson coefficient = -0.846, $p = 0.017$). We also examined relationships between metrics for COYs and those for yearlings in the following year for periods when sampling occurred in successive years.

To examine temporal trends in reproductive metrics and associations with sea ice, we used weighted least squares regression (General Linear Model procedure), with number of litters sampled each year as the weighting variable. Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

## 6.3. Results

During mark-recapture sampling in 1993-1995, 1997, and 2011-2013, we sampled 251 family groups consisting of an adult female and 1-3 dependent COY (400 COY in total; Table 6.1). During this period we also sampled 152 family groups consisting of an adult female and 1-3 dependent yearlings (231 yearlings in total). The mean number of family groups sampled annually was 35.9 (range: 16-62) and 21.7 (range: 8-44) for COY and yearling families respectively. From these data we created a second subsampled dataset base on geographic location to exclude bears that were encountered outside the standardized sampling area as described previously. This geographic subset consisted of 191 COY family groups containing 1-3 dependent COY (300 COYs in total) and 105 yearling family groups containing 1-3 dependent yearlings (160 yearling in total; Table 6.2). The mean number of family groups sampled annually was 27.3 (range: 16-48) and 15.0 (range: 8-25) for COY and yearling families respectively. Annual reproductive metrics for these two datasets are presented in Tables 6.1 and 6.2.

### Litter Size

There was annual variation in mean litter size amongst adult females with COYs but differences amongst years were non-significant for both the full (Kruskal-Wallis, $P = 0.153$) and geographic subset (Kruskal-Wallis, $P = 0.069$) data (Figure 6.2). For adult females with yearlings, differences in mean annual litter size were not significant for the full dataset (Kruskal-

Wallis, *P* = 0.051). For the subset data, differences in mean yearling litter size were significant (Kruskal-Wallis, *P* = 0.035) with both 1997 and 2013 having significantly lower litter sizes than 1994 and 1993 (Bonferonni correction for multiple comparisons, $P < 0.050$ in each case; Figure 6.3).

There were no statistically significant temporal trends in annual COY or yearling mean litter size from 1993 to 2013 (Table 6.3). Similarly, mean litter sizes were not associated with date of spring sea-ice transition.

Recruitment

COY recruitment showed a significant negative temporal trend from 1993 to 2013 for the geographic subset data, but not the full dataset (Table 6.3). COY recruitment also exhibited a positive association with date of spring sea-ice transition (Figure 6.4) for both the full and geographic subset data (i.e., later spring break-up was associated with higher COY recruitment). For yearlings, annual recruitment was not associated with either time or spring transition date (Figure 6.4).

There were 4 instances where sampling occurred over 2 successive years: 1993-94, 1994-95, 2011-2012, and 2012-2013. For these back-to-back years we compared reproductive metrics for COY (time t) to those of yearlings in the following year (time t+1). There was no relationship between annual mean litter size for COY in year *t* and yearling litter size in year *t + 1* (Table 6.4; Figure 6.5). COY recruitment was positively associated with yearling recruitment the following year for the geographic subset data but not the full dataset. COY litter size was closely associated with yearling recruitment in the following year for both datasets (Table 6.4; Figure 6.5), such that higher mean COY litter size in year *t* resulted in higher yearling recruitment in year *t + 1*. We also examined the relationship between yearling reproductive metrics and spring ice transition date the previous year finding no association for either the full ($F_{1,\,6} = 0.128$, $r^2 = 0.025$, $P = 0.735$) or subset data sets ($F_{1,\,6} = 0.095$, $r^2 = 0.019$, $P = 0.771$).

## 6.4. Discussion

Calculating annual reproductive metrics from mark-recapture field data is subject to several potential sources of error, including non-random sampling with respect to the overall study subpopulation. Similar to other seasonally ice-free subpopulations (e.g., Derocher and Stirling 1990), polar bears in Baffin Bay are known to exhibit a degree of spatial segregation by sex, age class and reproductive status with respect to the use of terrestrial habitat during the ice-free period (Ferguson et al. 1997; Chapter 4). For example, adult females with cubs tend to select fjords and avoid offshore islands and coastal regions where densities of adult males are higher (Ferguson et al. 1997). Pregnant bears select inland and upland denning habitats (Chapter 3). Mark-recapture sampling in the 1990s was more restricted in geographic extent relative to the 2010s (Chapter 3). The extent to which this difference in sampling between the two time periods introduced error and more importantly, systematic bias into our estimates of reproduction, cannot be fully evaluated. However, sampling bias is unlikely to account for the results of our analyses for several reasons. First, we attempted to account for differences in sampling by restricting some analyses to data collected within a standardized sampling area. This made little difference to the results. Results based on full and subset data were very similar. Second, sampling bias between the 1990s and 2000s would not account for the temporal trends in reproduction or associations with the timing of spring sea-ice transition unless sampling bias varied in proportion to these factors; something that is unlikely. Third, under-sampling of fjord habitats in the 1990s may have introduced bias in sampling of adult females with offspring. However, the most likely impact of this bias would have been underestimation of recruitment indices since fewer adult females with offspring would have been sampled relative to adult females overall. Fourth, sampling bias would not account for the association observed between reproductive metrics in successive years and the closeness of this association in some cases.

Another source of error in estimation of reproductive parameters in our study originated from the misclassification of bears that were observed from the air rather than handled during 2011-2013. However, when combined with genetic sexing, the accuracy of this method of classification is high even for lone adult females[1] (Appendix B). Furthermore, this source of error does not necessarily introduce systematic bias. Lone adult females could only have been misclassified as lone subadult females (and vice-versa), and there is no evidence to suggest inaccuracy in this area favours one age-class versus the other.

Finally, analyses of telemetry data collected from collared adult females suggest that a proportion of collar bears remained on the remnant sea ice in some years during the 1990s and were unavailable for sampling (Chapter 3). Although this could have introduced bias if certain classes of bears tended to remain on the ice while others moved to shore, we were unable to correct for this potential source of bias. However, the most likely effect would have been over-representation of lone (pregnant) adult females in our sampling data since this is the class of adult females that has an obligate need to come ashore to look for suitable denning habitat. This, in turn, would have led to underestimation of recruitment in the 1990s relative to the 2000s.

## Litter Size

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[1] Approximately 84% of lone adult females were correctly classified (GN unpublished data; see chapter 5, appendix 1.)

We found little evidence of changes in litter size amongst COY or yearlings in BB. Annual variation in litter size was for the most part not significant and there were no trends over time or in association with spring transition date. This finding is consistent with Molnar et al. (2011), who found that although litter size is predicted to vary in response to changes in maternal body condition and environmental conditions, it is a relatively insensitive reproductive metric. Large changes in maternal condition and environment are necessary to produce statistically significant differences in litter size. Litter size does, however, remain an important reproductive metric for monitoring polar bear subpopulations (Vongraven et al. 2012). Changes in litter size have been associated with temporal and geographic variation in ecosystem productivity (Stirling and Lunn 1997, Peacock et al. 2013), and long-term trends have been detected in association with changing subpopulation status (Derocher and Stirling 1995).

### Trends in COY Recruitment and Association with Sea-ice Conditions

We found evidence of a temporal trend in our index of COY recruitment that was also closely associated with variation in sea-ice conditions. From 1993 to 2013, cub recruitment declined concurrent with a trend towards earlier spring sea-ice break-up. Similar declines in reproduction over time and in association with sea-ice conditions have been previously reported for polar bear subpopulations (Derocher and Stirling 1995, Derocher 2005, Rode et al. 2010, Peacock et al. 2013, Rode et al. 2014), but not in Baffin Bay. Earlier spring break-up (also associated with later fall freeze-up) presumably decreases feeding opportunities for polar bears, thereby resulting in poorer maternal body condition and reduced investment in reproduction. This, in turn, will be manifested as reduced natality rates and / or lower offspring survival. Our index of cub recruitment incorporates both of these parameters reflecting to an unknown degree a blend of decreased cub production and lowered cub survival over the first 8 months of life. The association between cub recruitment and spring transition date in our study suggests that lower cub survival from birth to the first ice-free season may be a primary mechanism driving lower reproduction in Baffin Bay. However, we have not demonstrated a causal relationship; other factors may play an important role in cub recruitment, particularly since recruitment was associated with both time (year) and spring transition date. These two parameters are correlated with one another and may also be associated with other parameters that we did not consider. As such, there is some uncertainty as to the extent to which declining reproduction in BB is mediated by sea-ice conditions.

Declining reproduction and body condition are amongst the first subpopulation level effects predicted occur in polar bears as a result of climate change (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Our evidence of a decline in reproduction in BB from 1993 to 2013 is accompanied by evidence of concurrent declines in body condition amongst bears in this subpopulation over the same period (Rode et al. 2012, Chapter 6). These changes may signal a reduction in the carrying capacity of BB. Although the point estimate of abundance from our recent genetic mark-recapture was higher than the 1990s estimate, the difference between estimates was not statistically significant (Chapter 5). Additionally, differences in these point estimates may be largely explained by differences in sampling design between the two time periods. Regardless of whether density effects are at play, if the observed association between sea ice and reproduction is real and the well documented trend in sea-ice continues, it is reasonable to expect that this subpopulation will experience significant changes in reproductive performance as a result of declining habitat (Laidre et al. 2015).

Yearling Recruitment

Recruitment calculated as the number of yearlings per adult female has been identified as an important reproductive metric to monitor in polar bear subpopulations, incorporating both natality and survival of COY (Vongraven et al. 2012). Regehr et al. (2015) suggest that variation in yearling recruitment may be a primary mechanism driving changes in subpopulation growth, with values ranging from 0.1 to 0.3 necessary for subpopulation persistence. Estimated annual recruitment values for Baffin Bay during 1993 to 2013 ranged from 0.24 to 0.51, suggesting that BB continues to function as a viable subpopulation, according to this metric. Interestingly, in contrast to previous studies (e.g., Rode et al. 2010, 2014), we did not find evidence that yearling recruitment was associated with sea-ice conditions. Instead, our results suggest that recruitment of yearlings is largely determined by reproductive metrics for COYs in the previous year. We found an association between COY recruitment in year t and yearling recruitment in year t + 1, as well as a strong association between COY litter size and yearling recruitment to following year. These findings suggest that a majority of mortality of polar bear cubs occurs during the first 8 months of life, such that recruitment of yearlings is heavily dependent on the number of COY that are born and survive up to the first ice-free period. Since COY recruitment itself appears to be associated with spring transition date, yearling recruitment up to the ice-free period may be influenced to a greater degree by ice conditions the previous year than by ice conditions in the current year. We did not find evidence of this lag effect but our sample size was very small.

Although values in the 2000s tended to be lower than those observed in the 1990s, we did not find a significant temporal trend in annual yearling recruitment. However, given the association between COY reproductive metrics and sea ice, and the long-term trends in sea ice in BB, we suspect that a decline in yearling recruitment would be evident with a larger and longer-term data set. In this context, we suggest that monitoring annual litter size of COY may be a useful tool for tracking trends in recruitment in the absence of more intensive subpopulation studies. This metric can be readily acquired from aerial surveys without capture or biopsy of bears, without the need for extensive observer experience in identifying age-sex classes, and at relatively low cost. Our very limited data suggest that mean annual COY litter size is closely related to yearling recruitment the following year. However, we acknowledge that the robustness of this relationship has not been validated with a larger data set and under a range of environmental conditions.

### Comparison with other Subpopulations

Indices of reproduction for BB were comparable to other polar bear subpopulations in the seasonal ice ecoregion (Amstrup et al. 2008) that have been recently studied by mark-recapture or aerial survey (Table 6.5). Mean litter sizes in Baffin Bay and the proportions of COY and yearlings were similar to those observed in Foxe Basin (Stapleton et al. 2016) and Southern Hudson Bay (Obbard et al. 2015); two subpopulations classified as stable (PBSG 2010). In contrast, metrics for Baffin Bay were notably higher than those estimated for Davis Strait (Peacock et al. 2013) and Western Hudson Bay (Lunn et al. 2014). These comparisons suggest that BB remains a relatively productive subpopulation despite apparent recent declines in reproduction.

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**Table 6.1.** Reproductive metrics for annual mark-recapture sampling data from Baffin Bay. Full dataset.

| Year | Mean Litter Size (n, SD) | | Recruitment Index[2] (Offspring/adult female) | |
|---|---|---|---|---|
| | COY | Yearling [1] | COY | Yearlings [1] |
| **1993** | 1.63<br>(16, 0.50) | 1.75<br>(8, 0.46) | 0.79 | 0.42 |
| **1994** | 1.87<br>(23, 0.51) | 1.82<br>(11, 0.38) | 0.83 | 0.38 |
| **1995** | 1.52<br>(21, 0.51) | 1.64<br>(14, 0.63) | 0.71 | 0.51 |
| **1997** | 1.64<br>(25, 0.49) | 1.25<br>(12, 0.45) | 0.65 | 0.24 |
| **2011** | 1.57<br>(58, 0.50) | 1.53<br>(34, 0.51) | 0.68 | 0.39 |
| **2012** | 1.47<br>(62, 0.50) | 1.55<br>(44, 0.54) | 0.47 | 0.35 |
| **2013** | 1.65<br>(46, 0.49) | 1.34<br>(29, 0.49) | 0.60 | 0.31 |

[1] Dependent yearlings only. Capture data for 1993-97 suggests up to 6% of yearlings are independent of their mother in the fall

[2] Calculated per Derocher and Stirling (1995), Vongraven et al. (2012), Regehr et al. (2015)

**Table 6.2.** Reproductive metrics for annual mark-recapture sampling data from Baffin Bay. Data for 2011-13 were filtered to exclude individuals encountered outside the area where sampling was estimated to have occurred in the 1990s.

| Year | Mean Litter Size (n, SD) | | Recruitment Index[2] (Offspring/adult female) | |
|---|---|---|---|---|
| | COY | Yearling [1] | COY | Yearlings [1] |
| **1993** | 1.63<br>(16, 0.50) | 1.75<br>(8, 0.46) | 0.79 | 0.42 |
| **1994** | 1.87<br>(23, 0.51) | 1.82<br>(11, 0.38) | 0.83 | 0.38 |
| **1995** | 1.52<br>(21, 0.51) | 1.64<br>(14, 0.63) | 0.71 | 0.51 |
| **1997** | 1.64<br>(25, 0.49) | 1.25<br>(12, 0.45) | 0.65 | 0.24 |
| **2011** | 1.48<br>(33, 0.51) | 1.60<br>(20, 0.50) | 0.61 | 0.40 |
| **2012** | 1.42<br>(48, 0.50) | 1.48<br>(25, 0.51) | 0.55 | 0.30 |
| **2013** | 1.64<br>(25, 0.49) | 1.27<br>(15, 0.46) | 0.55 | 0.26 |

[1] Dependent yearlings only. Capture data for 1993-97 suggests up to 6% of yearlings are independent of their mother in the fall

[2] Calculated per Derocher and Stirling (1995), Vongraven et al. (2012), Regehr et al. (2015)

**Table 6.3.** Results of least squares regressions for annual reproductive metrics derived from mark-recapture sampling of polar bears during the ice-free period (August-October) in Baffin Bay. Number of litters was used as a weighting variable. Significant relationships in bold.

| Reproductive Metric | Litter Age | Explanatory Variable | Dataset | $F_{1,6}$ | $r^2$ | $P$ |
|---|---|---|---|---|---|---|
| Litter Size | COY | Year | Full | 1.64 | 0.25 | 0.256 |
| Litter Size | COY | Year | Subset | 3.37 | 0.40 | 0.126 |
| Litter Size | COY | Spring Ice Transition | Full | 3.42 | 0.41 | 0.124 |
| Litter Size | COY | Spring Ice Transition | Subset | 3.92 | 0.44 | 0.105 |
| Litter Size | Yearling | Year | Full | 1.22 | 0.20 | 0.319 |
| Litter Size | Yearling | Year | Subset | 1.16 | 0.19 | 0.331 |
| Litter Size | Yearling | Spring Ice Transition | Full | 1.64 | 0.25 | 0.256 |
| Litter Size | Yearling | Spring Ice Transition | Subset | 1.38 | 0.22 | 0.293 |
| Recruitment | COY | Year | Full | 5.34 | 0.52 | 0.069 |
| Recruitment | COY | Year | Subset | 22.43 | 0.82 | **0.005** |
| Recruitment | COY | Spring Ice Transition | Full | 53.90 | 0.92 | **0.001** |
| Recruitment | COY | Spring Ice Transition | Subset | 11.60 | 0.70 | **0.019** |
| Recruitment | Yearling | Year | Full | 0.72 | 0.13 | 0.434 |
| Recruitment | Yearling | Year | Subset | 1.37 | 0.21 | 0.295 |
| Recruitment | Yearling | Spring Ice Transition | Full | 0.64 | 0.11 | 0.460 |
| Recruitment | Yearling | Spring Ice Transition | Subset | 1.91 | 0.28 | 0.225 |

**Table 6.4.** Results of least squares regressions for annual reproductive metrics derived in year *t* and year *t+1* from mark-recapture sampling of polar bears during the ice-free period (August-October) in Baffin Bay, Canada. Number of litters in year *t+1* was used as a weighting variable.

| Reproductive Metric (year t) | Reproductive Metric (year t+1) | Dataset | $F_{1,3}$ | $r^2$ | $P$ |
|---|---|---|---|---|---|
| COY Litter Size | Yearling Litter Size | Full | 0.80 | 0.29 | 0.465 |
| COY Litter Size | Yearling Litter Size | Subset | 0.92 | 0.29 | 0.431 |
| COY Recruitment | Yearling Recruitment | Full | 3.88 | 0.66 | 0.188 |
| COY Recruitment | Yearling Recruitment | Subset | 20.33 | 0.91 | **0.046** |
| COY Litter Size | Yearling Recruitment | Full | 474.43 | 0.99 | **0.002** |
| COY Litter Size | Yearling Recruitment | Subset | 1854.94 | 0.99 | **0.001** |

**Table 6.5.** Comparison of reproductive metrics for polar bear subpopulations in the seasonal ice ecoregion. Sampling occurred during ice-free periods.

| Subpopulation | Mean Litter Size | | Proportion of Total Observations[2] | | Source |
|---|---|---|---|---|---|
| | COY | Yearling | COY | Yearlings | |
| Baffin Bay (1993-97) | 1.67 | 1.60 | 0.16 | 0.09 | Taylor et al. (2005) |
| Baffin Bay (2011-13)[1] | 1.55 | 1.48 | 0.15 | 0.09 | This study |
| Davis Strait (2005-07) | 1.49 | 1.22 | 0.08 | 0.09 | Peacock et al. (2013) |
| Foxe Basin (2009-2010) | 1.55 | 1.48 | 0.13 | 0.10 | Stapleton et al. (2016) |
| Southern Hudson Bay (2011) | 1.56 | 1.49 | 0.16 | 0.12 | Obbard et al. (2015) |
| Western Hudson Bay (2011) | 1.43 | 1.22 | 0.07 | 0.03 | Stapleton et al. (2014) |

[1] Based on sampling across study area

[2] Some of these recent studies relied on aerial survey which is less accurate in identifying adult females (without genotyping to determine sex). For this reason we used published data on mean litter sizes and the proportion of COY and yearlings within these studies rather than calculating indices of recruitment used in the present study

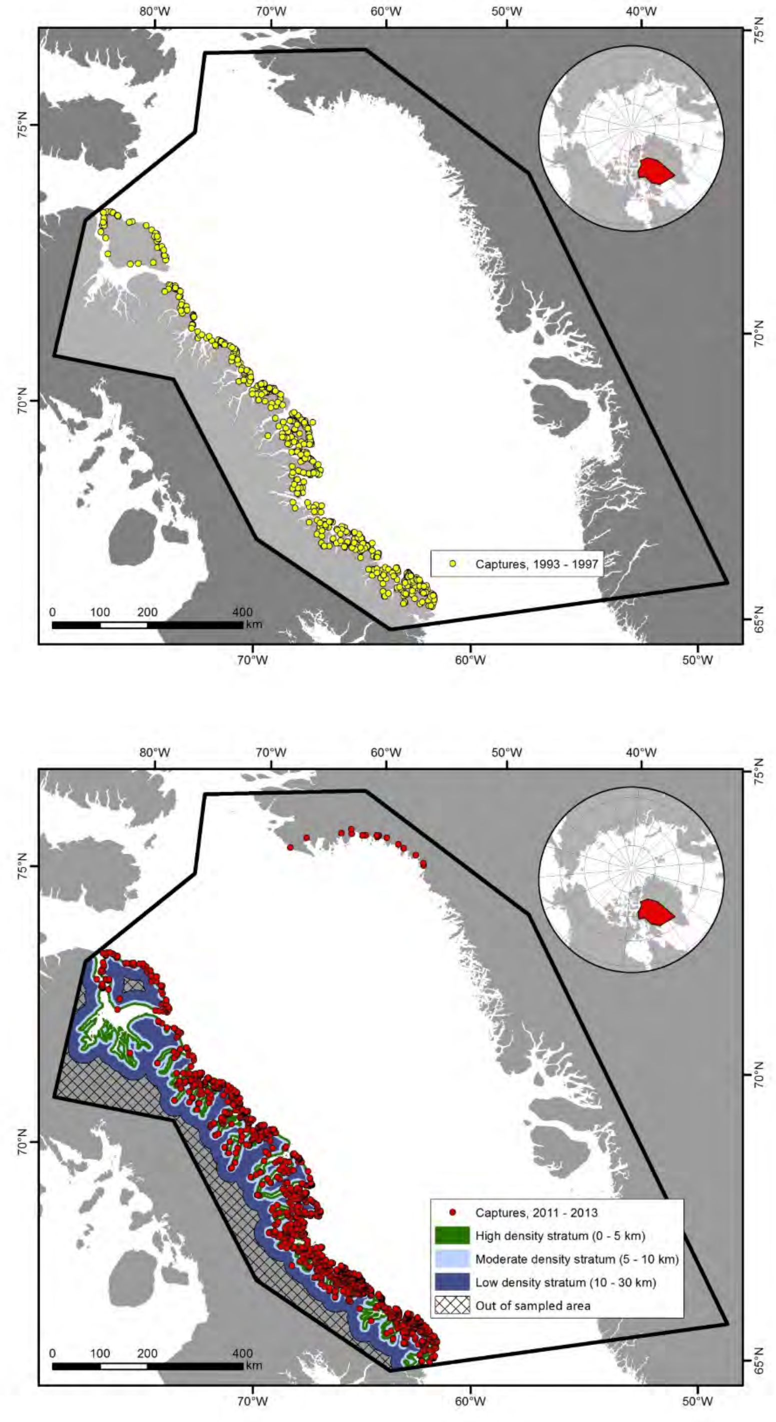
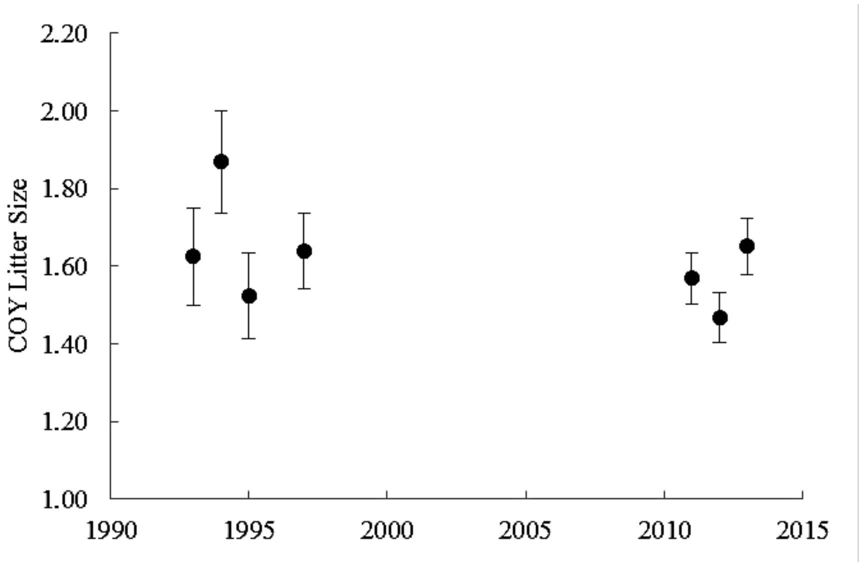


**Figure 6.1.** Spatial distribution of bears recorded during sampling in the Baffin Bay polar bear subpopulation, 1993 – 1995, 1997 (top), and 2011 – 2013 (bottom).

(a)



(b)

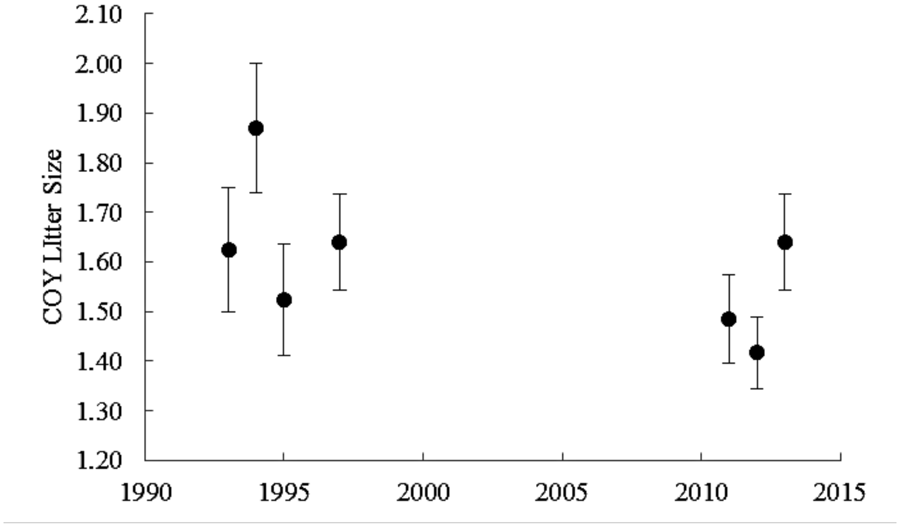


**Figure 6.2.** Mean litter sizes (+/- SE) for adult females with cubs-of the year (COY) during the ice-free period in Baffin Bay. Calculated using the (a) full and (b) geographically subset data (see text).

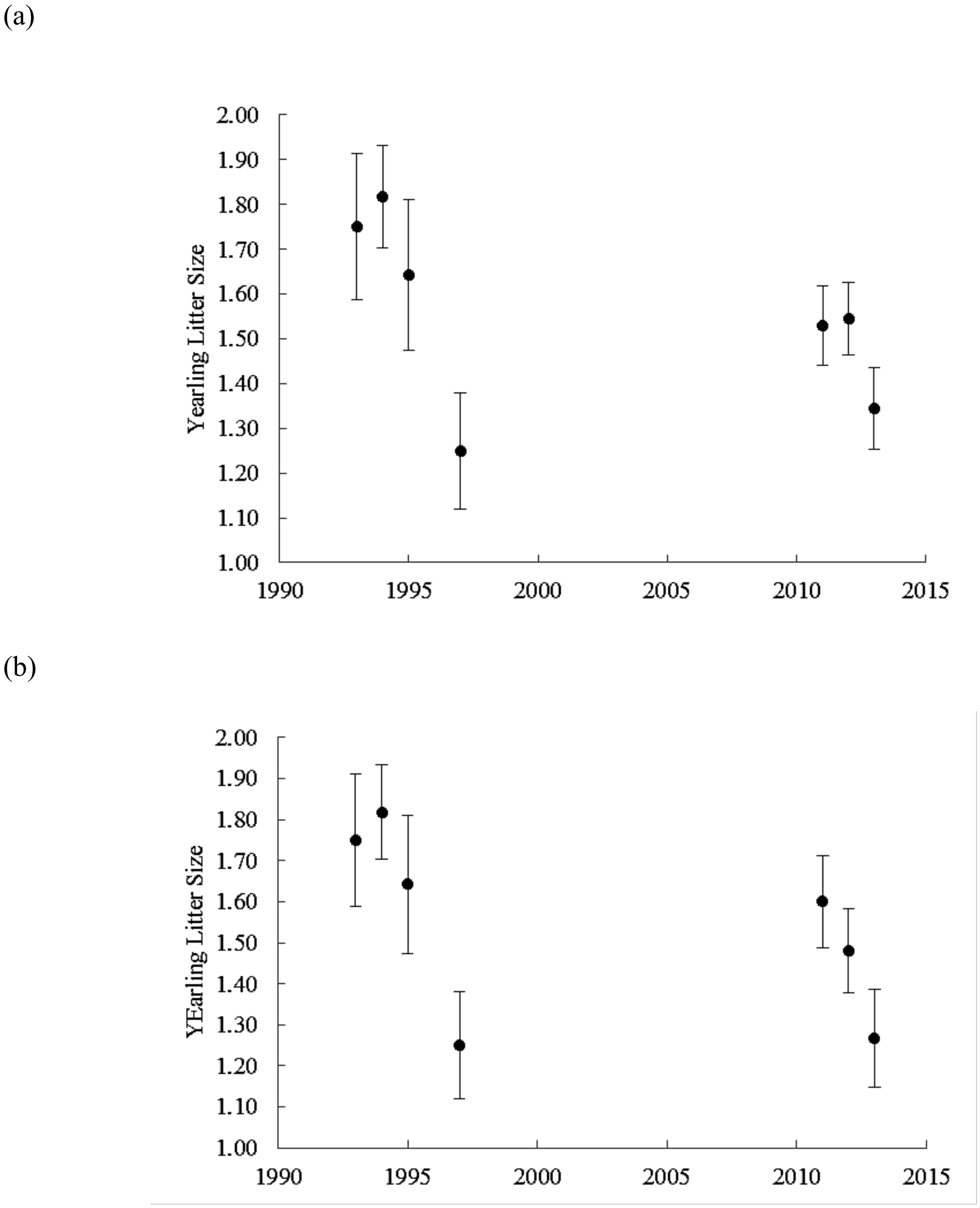


**Figure 6.3.** Mean litter sizes (+/- SE) for adult females with yearlings during the ice-free period in Baffin Bay. Calculated using the (a) full and (b) geographically subset data (see text).

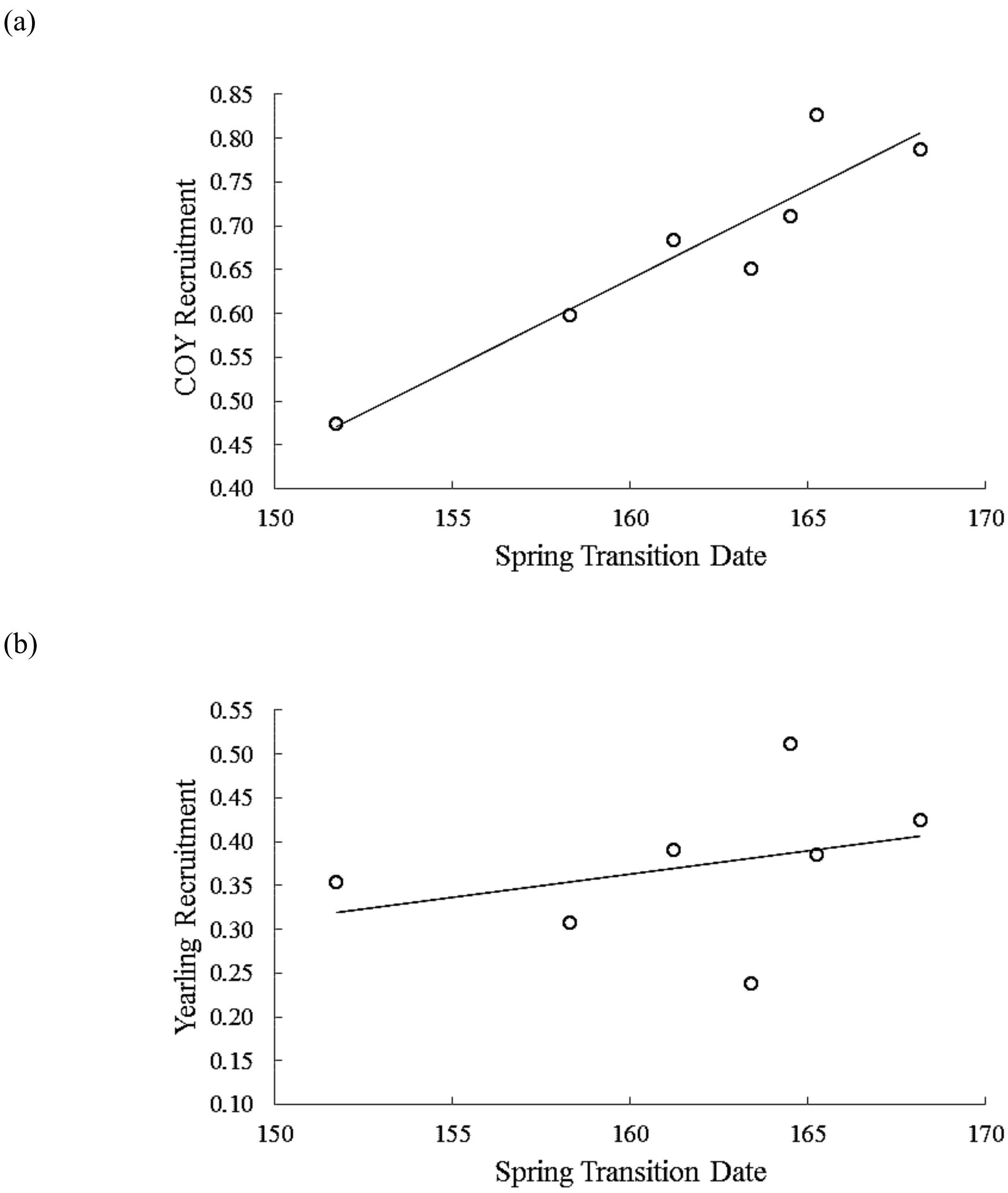
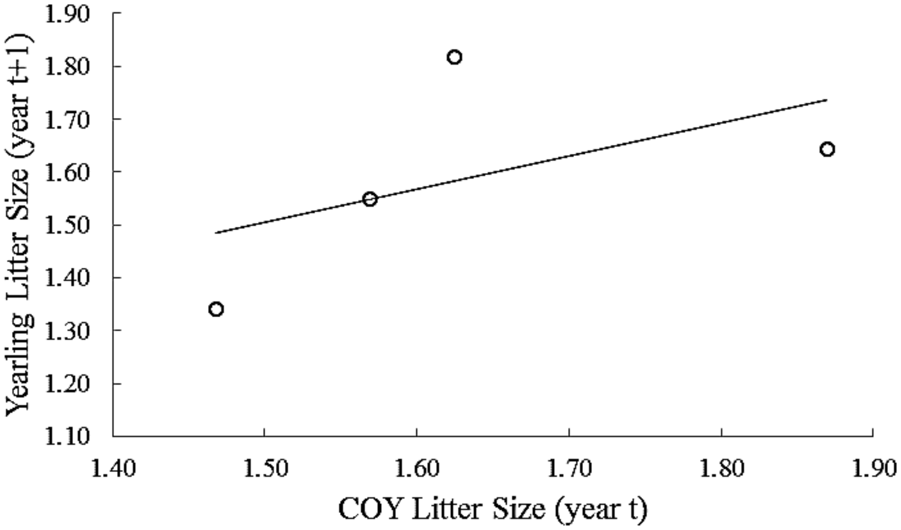


**Figure 6.4.** The relationship between spring transition date and annual recruitment index for (a) cub-of-the-year and (b) yearling polar bears during the ice free period (August-October), in Baffin Bay. Recruitment calculated as the number of COY or yearlings per adult female in the sample subpopulation using the full mark-recapture dataset.

(a)



(b)

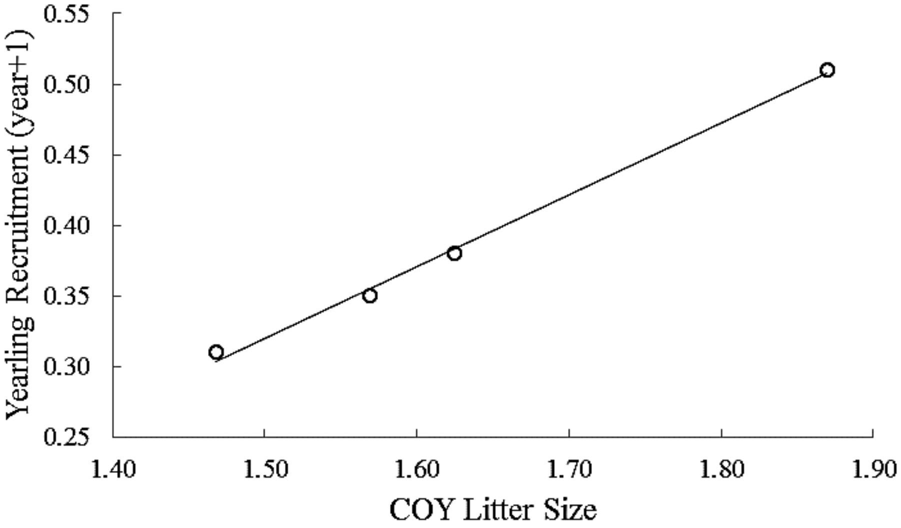


**Figure 6.5.** The relationship between mean annual litter size amongst cub-of-the-year litters and (a) yearling litter size and (b) yearling recruitment in the following year (year +1). Data are for polar bears sampled during the ice free period (August-October), in Baffin Bay. Metrics calculated using full mark-recapture dataset.

# Chapter 7

# Body Condition of Baffin Bay Polar Bears

## Key Findings

- Fatness index (FI) scores were collected during two periods of mark-recapture sampling in BB, 1993-1995, 1997 and 2011-2013. We examined trends in this metric of body condition across both sampling periods in relation to sea-ice conditions.
- We found evidence of declines in body condition amongst bears in BB over the period 1993 to 2013. Body condition in BB polar bears declined in close association with the ice-free period and spring sea-ice transition dates. This is consistent with the hypothesis that reduced time on the sea ice is a primary mechanism driving this decline.
- Our results follow with previous studies that showed similar results through 2010 with different metrics derived from physical handing of bears (Rode et al. 2012).
- The springtime aerial survey was successfully implemented due to the small geographic These findings are consistent with available traditional knowledge suggesting that body condition of polar bears in BB was poorer in the early 2000s relative to the 1990s (Dowsley and Wenzel 2008, Born et al. 2011).
- We found evidence of recent foraging in approximately 9% of the BB bears observed during the on-land period. Marine mammals, in particular seals and walrus made up almost half of the identifiable food sources.

## 7.1. Background

For populations of large, long-lived mammals changes in body condition will be among the early indicators of density-dependent regulation and / or changes in environmental carrying capacity (Fowler 1987, 1990, Zedrosser et al. 2006). In populations approaching K, declines in condition will occur before declines in adult survival. From both wildlife management and species conservation perspectives, monitoring body condition may therefore provide a useful tool for the early detection of population trends that warrant more in-depth study. This is especially true for populations where cost or logistical constraints limits the capacity to undertake on-going, intensive demographic studies. In these cases, monitoring condition may provide a form of surveillance that can be used to trigger periods of more intensive study.

The annual life-cycle of polar bears (*Ursus maritimus*) is characterized by large seasonal changes in body condition (Watts and Hansen 1987, Ramsay et al. 1992, Ramsay and Stirling 1988, Atkinson and Ramsay 1995). Throughout most of their circumpolar range, bears are thought to gain condition during the spring and early summer when juvenile seals are abundant and relatively susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). This period of hyperphagia is followed by a scarcity of food in the late summer and fall when sea ice reaches a minimum throughout the Arctic. During this season, bears in some regions are forced onto land by the melting sea ice where access to seals and other marine mammal prey is greatly reduced (Stirling et al. 1977, Derocher and Stirling 1990a,b, Ramsay et al. 1991). In other regions, bears remain on off-shore pack-ice but likely also have reduced access to and/or less success in catching seals (Amstrup et al. 2000, Stirling 2002, Atwood et al. 2015, Rode et al. 2015). Some individuals utilize terrestrial foods sources during the summer and fall. However, the extent to which this occurs and the significance of terrestrial foods to energy budgets remains the subject of on-going debate and research (e.g., Derocher et al. 1993a, Hobson et al. 2009, Gormezano and Rockwell 2013, 2015, Rode et al. 2015, Tartu et al. 2016). Regardless, it is well documented that many bears especially those in the seasonal ice ecoregion like Baffin Bay (Amstrup et al. 2008) rely on nutrients and energy stored within adipose and other body tissues to meet a significant portion of their maintenance requirements for survival during this period. Consequently, body condition amongst most individuals declines progressively through the summer and fall until access to sea ice increases in the late fall and early winter (Atkinson and Ramsay 1995, Atkinson and Ramsay 1996, Derocher and Stirling 1995, Rode et al. 2011, Obbard et al. 2016). Additionally, adult females rely on body stores to support reproductive activities. Mothers with cubs continue to lactate during the summer and fall if their condition and/ or available food is sufficient (Derocher et al.1993b, Derocher and Stirling, 1996). Pregnant females enter dens in the fall where they rely exclusively on body stores to support gestation and early to mid-lactation over a period of 6-8 months (Atkinson and Ramsay 1995, Derocher and Stirling 1995). Given this dynamic cycle of feeding and fasting, body condition attained during the spring and early summer is expected to exert a significant influence on the survival, reproductive performance and thus status of polar bear subpopulations (Atkinson and Ramsay 1995, Derocher and Stirling 1995, 1996, Molnar et al. 2010, Molnar et al. 2011).

Tracking long-term trends in body condition has been identified as an important component of the monitoring scheme for polar bears across their circumpolar range (Vongraven et al. 2012, Patyk et al. 2015). In the absence of more intensive studies, simple body condition metrics may be useful indices for monitoring subpopulations and detecting responses to changing environmental conditions (Amstrup et al. 2006, Stirling et al. 2008a, Vongraven et al. 2012). Of particular concern, changes in body condition are predicted to be amongst the first subpopulation-level impacts of climate change evident in this species (Derocher et al. 2004, Stirling and Parkinson 2006, Wiig et al. 2008, Stirling and Derocher 2012). Indeed, declines in condition have been documented in several polar bear subpopulations in association with long-term changes in sea-ice conditions that appear to be climate induced (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016). Although these trends in body condition can signal significant changes, observations of declining condition alone do not necessarily imply a decline in subpopulation status. Studies of some polar bear subpopulations have documented declines in condition in association with increases in abundance that may be the result of density dependence (Stirling et al. 1999, Rode et al. 2011, Peacock et al. 2013).

Information on body condition must therefore be interpreted alongside other measures of subpopulation performance in-order to properly assess status.

A variety of quantitative and qualitative body condition indices have been used on polar bears including body weight estimated from girth (e.g., Derocher and Stirling 1995, Rode et al. 2011), body mass indices standardized for length (e.g., Stirling et al. 1999, Cattet et al. 2002, Obbard et al. 2016), skull width (Rode et al. 2010, 2011), percent body fat determined by isotopic dilution or bioelectrical impedance analysis (Atkinson and Ramsay 1995, 1996, McKinney et al. 2014), percent lipid content of adipose tissue biopsies (Stirling et al. 2008b, McKinney et al. 2014) and a visually assigned fatness index (Amstrup et al. 2006, Stirling et al. 2008a,b). Most of these condition indices require the handling of bears to collect measurements. However, the fatness index (FI) and potentially the lipid content of adipose tissue (Pagano et al. 2014, McKinney et al. 2014) may be obtained without handling thus making them suitable for use in subpopulations monitored by less invasive methods such as aerial survey or genetic mark-recapture.

The Baffin Bay (BB) subpopulation is part of the seasonal ice-free ecoregion as defined by Amstrup et al. (2008) in which sea ice melts almost entirely in the summer and bears are forced ashore for an extended period of time, during which they have no or reduced access to food. Baffin Bay has experienced a long term reduction in sea-ice cover and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015). As a result, bears are spending an increasing amount of time on land. Examining data on the morphometric measurements of BB polar bears (girth, length and skull width) for the period 1977 to 2010, Rode et al. (2011) detected a decline in body condition concurrent with declining sea-ice cover. However, geographically restricted sampling and uncertainty about trends in subpopulation density during the latter years of this study limited the interpretation of these findings and conclusions regarding subpopulation status (York et al. 2016).

Here we summarize information on the body condition of polar bears in BB using a different measure of condition; the fatness index (FI). FI scores were collected during two periods of mark-recapture sampling in Baffin Bay from 1993 to 1997 and 2011 to 2013. During the latter period of sampling bears were surveyed by genetic mark-recapture using biopsy darts. Because biopsy darted bears were not handled our collection of body condition data was limited to visually assigned FI scores only. The FI has been validated as a measure of condition in polar bears, being closely correlated with more quantitative condition indices (Stirling et al. 2008b, McKinney et al. 2014) and other biological factors (e.g., Henricksen et al. 2001, Amstrup et al. 2006). Our study examined body condition using a different index of condition collected over a different (albeit overlapping) temporal and spatial sampling frame to that of Rode et al. (2011). We examined trends in condition in relationship to sea ice. During part of this study, we also collected information on the foraging habits of BB polar bears to assess the range of food sources utilized by bears during the ice-free period. Our results provide supplementary information for interpreting the results of the recent genetic mark-recapture in BB and for understanding the present status of this subpopulation.

## 7.2. Materials and Methods

### Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km$^2$ in Baffin Bay, covering portions of Baffin Island and Bylot Island (66.2°N to 73.8°N) in Nunavut/Canada and parts of West and Northwest Greenland (66.0°N to 77.0°N; Taylor et al.

2005). BB is bounded by Greenland to the east, Baffin Island to the west, the North Water polynya in the north and Davis Strait to the south (Figure 7.1). Three communities in Nunavut and 37 communities in Greenland harvest bears from BB, although the majority of the Greenland harvest is taken between *ca.* 72° and 76° N. Baffin Bay is ice-covered in winter but typically ice-free in summer. During late spring and summer break-up, sea ice recedes from Greenland westward across Baffin Bay; the last remnants of ice typically occur off the coast of Baffin Island. Historically sea ice also remained in Melville Bay, NW Greenland (Born 1995). Most polar bears remain on the sea ice as it recedes and then come ashore to spend the ice-retreat period on Baffin and Bylot Islands (Taylor et al. 2005). A presumably small number of bears remain on land in northwestern Greenland throughout the ice-retreat period (Born 1995, this study).

### Field Sampling

Data for the study were collected during two periods of systematic mark-recapture sampling on Baffin Island in Baffin Bay. Sampling occurred along eastern Baffin and Bylot Islands during the ice-free season from late August to mid-October in 1993-95, 1997 and 2011-2013. Most bears in Baffin Bay move onto land on Baffin Island and Bylot in late summer as the sea ice breaks up and remain on land until freeze-up in the late fall (Taylor et al. 2005). Sampling was spatially extensive across this on-land study area during all years (Figure 7.1) although there were some noted differences in sampling strategy between the two periods (1990s and 2000s) of the study (Chapter 3).

Using helicopters we searched for bears across the study area. During the 1990s, bears were sampled by physical capture on Baffin Island and examination using methods previously described (Taylor et al. 2005). Data on the sex, estimated age-class and reproductive status of each individual were recorded. Age of individuals was determined based on previous capture history, known (in the case of cubs and yearlings) or estimated from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998). Individuals were identified by means of uniquely numbered ear tags and lip tattoos. Group size, family status, location and date were also recorded.

During the 2000s, sampling occurred by remote biopsy darting (Pagano et al. 2014) and subsequent genetic analysis to determine genetic sex and identify individuals. We remotely estimated sex, age class (cub-of-the-year, yearling, subadult [ages 2 – 4], and adult) and reproductive status from the air at a range of 3-10 m above ground. Sex was later confirmed via genetic analysis. In estimating age-class and sex, the observer used multiple cues, including the size of an individual relative to its surrounding environment or accompanying bears, membership in a family group (mothers and cubs or yearlings), secondary sexual characteristics (adult males), body shape and proportions, the presence of scars (which are most often seen on adult males) and observations of urination (i.e., urine dribbling from under tail in females). Fields notes also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. Age-class was later verified in some bears from previous or future captures in which an individual was captured and physically examined or where an individual was matched via DNA to membership in a known family at some past or future point. We assessed the accuracy of this system for estimating the age-class and sex of polar bears using a sample of BB bears of known age-class (Appendix B).

During both sampling periods, all encountered bears were assigned a FI score on a scale of 1-5 where 1 and 5 represent the leanest and most obese bears, respectively (Stirling et al. 2008a).

During the 1990s, this score was based on physical examination of captured bears. For bears in the 2000s, FI scores were assigned based on examination from the air at a distance of 3-7 m above ground. Additional information collected for all bears at the time of observation included the identity of the observer, date, and location (coordinates).

Additionally various sources of information including traditional ecological knowledge (TEK), expedition reports and unpublished data (Born 1995, Born et al. 2011, GINR unpublished data) indicate that small, albeit unknown, proportion of the BB subpopulation summers in the Qimmusseriarsuaq / Melville Bay area of NW Greenland rather than moving with the retreating sea ice and summering on Baffin and Bylot Islands. For comparative reasons estimates of body condition index from bears biopsy dated in NW Greenland are presented separately from that on Baffin Island. During 1-14 September 2012 and 2013 a total of 20 biopsies (2012: 6, 2013: 14) were sampled on land and along glacier fronts in the Melville Bay area using the same methods described above in Nunavut. During sampling the FI index was scored for each bear by three observers with extensive experience in judging body condition of polar bears both during examination from the air and during subsequent physical handling. The sex of all biopsied individuals was determined genetically post hoc.

### Body Condition Scoring

Bears were initially scored according to the standard FI on a scale of 1 to 5 (Stirling et al. 2008b). This scoring system was subsequently simplified to a 3 point scale of poor (FI = 1 or 2), fair (FI = 3) and good (FI = 4 or 5) condition; hereafter termed the Body Condition Score (BCS). Similar modifications of the FI for polar bears have been employed in other studies to facilitate analyses (Stirling et al. 2008a) or have been recommended for use in general monitoring schemes for polar bears (Vongraven et al. 2012). In our case, this refinement was made in part due to the lower frequencies of bears scored as 1 and 5, but also to address concerns about potential bias. The assumption was made that a simplified scale would be subject to less bias resulting from different observers and / or distance from bear at time of scoring. Experienced observers should be able to discriminate a bear in poor, fair or good condition even at distances of up to 7 m.

Bears coming off the ice in summer are thought to be at or near their annual peak in body condition having recently gone through a period of hyperphagia when juvenile seals are relatively abundant and susceptible to predation (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). For our analyses, therefore, we calculated the proportion of bears rated as being in good condition (BCS = 3) as a metric of annual body condition within each sex and age class. Repeated observations of the same individual (as identified by physical mark or genotype) within a given year were excluded from the analyses. Observations of the same individual in different years were included. Similar to Stirling et al. (2008a), we assumed that observations of the same individual in different years were statistically independent given the dynamic nature of body condition in polar bears (Watts and Hansen 1987, Atkinson and Ramsay 1995, 1996) and it's response to annual variation in environmental conditions.

Sea-ice Metric

Based on findings from other studies (Stirling et al. 1999, Durner et al. 2004, 2006, 2009, Regehr et al. 2007, Rode et al. 2010, 2014), we hypothesized that timing of sea-ice break-up over the continental shelf (< 300m depth) of BB may influence the body condition of polar bears during the ice-free period. We therefore calculated the annual sea-ice metric, spring transition date, as day of the year (1-365) when ice area over the continental shelf of BB reached 50% in the spring (Chapter 4). We used *Sea-ice Concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data* (Cavalieri et al., 1996) available from the National Snow and Ice Data Center (NSIDC) in Boulder, Co (See Laidre et al. 2015 Appendix S1 for additional details). While some researchers have used lower ice cover thresholds for studying polar bear relationships to sea ice (e.g., Cherry et al. 2013, Rode et al. 2014, Obbard et al. 2016), Laidre et al. (2015) reported that spring transition dates in Baffin Bay were not sensitive to the choice of threshold because usually the decrease of sea-ice area in the spring and the increase of sea-ice area in the fall proceed relatively quickly. A small change in the threshold results in a small change in the transition dates (Laidre et al. 2015). For sea ice in Baffin Bay, spring transition date is also correlated with other sea-ice metrics including fall transition date (negative correlation) and the interval between spring and fall transitions (Laidre et al. 2015).

We examined annual variation in our body condition metric, trends over time, and relation to spring sea-ice transition date. Similar to Rode et al. (2014), we did not include both time and sea ice in the same analyses since long term trends in our sea-ice metric were well established *a priori* (Laidre et al. 2015, Chapter 4) and these two parameters were closely correlated over the 7 years of sampling (Pearson coefficient = -0.846, $P = 0.017$). Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

Foraging Observations

Observations of bears feeding or evidence that they had recently fed were collected during the second sampling session, 2011-13, only. In 2011, systematic observations of the feeding activities were not collected. In 2012 and 2013, all bears encountered were examined from the air for evidence of feeding activity. Evidence of feeding included the presence of kills and other obvious food items, fresh oil or blood staining around the mouth, pendulous/distended abdomens (full stomachs), the production of black tar-like feces during pursuit (normally seen in bears that have been eating marine mammals) and the production of feces containing visible berries.

## 7.3. Results

Body Condition Scores

In total, 2500 polar bears were assigned a BCS during mark recapture sampling. Six bears assigned a BCS were not assigned to a sex-age class. These were excluded from the further analyses. Amongst independent bears, samples sizes were largest for adult males (n = 783), adult females with offspring (423) and lone adult females (225). Our analyses focussed on these three groups of bears for the following reasons: (1) Sample sizes were relatively large and/or (2) the accuracy of classifying bears into sex and age class at the time of aerial observation and BCS assignment was relatively good (Appendix B)[1]. Amongst dependent offspring we examined BCS for cubs-of-the-year (COY) and yearlings both of which can be identified with good accuracy from the air.

During the period 1993-97, all observations were made by a single individual (Table 7.1). During the period 2011-13, observations were made by 3 individuals but a majority (79%) were made by a single individual. Of the 2496 bears in our study, four were observed by two observers in the same year. In each case the condition scores assigned by the observers were the same (adult male in poor condition, adult female in fair condition, adult female in poor condition,

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[1] Accuracy for adult males, adult females with offspring, lone adult females was 95%, 100% and 74%, respectively. In comparison accuracy for sub-adult males and females was <40%.

subadult female in fair condition). Because the observers collected BCS data on bears in different years and / or different parts of the study area from each other, it was not possible to distinguish observer effects from other factors.

In general, body condition was better amongst adult males and lone adult females than other age classes (Table 7.2). For example, pooling data across years, 25 and 32 % of individuals were classified as being in good condition amongst adult males and lone adult females, respectively. In contrast, amongst adult females with offspring, subadult females and subadult males, the percentage of bears in good condition was 9, 4 and 4% respectively.

There was annual variation in body condition. For example, during the period 1993-97, when all observations were collected by a single individual, the distribution of adult male BCS varied significantly from year-to-year ($\chi^2 = 24.01$, d.f. = 6, $P < 0.001$) with more males in good condition in 1993 (58.8%) compared to other years, in particular 1997 (32.0%). Annual variation in condition was similar, although not significant, for lone adult females ($\chi^2 = 11.51$, d.f. =6, $P = 0.070$) and females with offspring ($\chi^2 = 9.94$, d.f. =6, $P = 0.132$) during this period. For the period, 2011-13, when a majority of observations were made by a single but different observer, condition was also found to vary significantly from year-to-year amongst adult males ($\chi^2 = 24.31$, d.f. =4, $P < 0.001$) but not lone females ($\chi^2 = 4.89$, d.f. =4, $P = 0.300$) or females with offspring ($\chi^2 = 7.71$, d.f. =4, $p < 0.100$). During this recent sampling period, 2011 tended to be a better year for body condition.

Our annual body condition metric was associated spring sea-ice transition date amongst some sex and age classes of bears (Table 7.3). The proportion of adult males assigned a BCS of 3 (good) in a given year was closely associated with the timing of spring sea-ice transition. A higher proportion of adult males were in good condition in years with a later spring transition date (Figure 7.2a). A similar association was evident for adult females with offspring (Figure 7.2b) but not lone adult females (Figure 7.2c). These associations between condition and sea ice were also evident for adult males (Exponential curve; $F_{1,3} = 101.27$, $r^2 = 0.98$, $P = 0.010$) and females with offspring (Exponential curve; $F_{1,3} = 72.12$, $r^2 = 0.97$, $P = 0.014$) when analyses were limited to the period 1993 to 1997; the 4 years when data were collected by a single observer.

For COY, the proportion of bears in good condition was unrelated to the timing of spring sea-ice transition (Table 7.3). In contrast, later spring transition was associated with a higher proportion of yearlings in good condition. Similar to adult males and females with offspring, this association was also evident when analysis was limited to the period 1993-97 when analyses were limited to the period 1993 to 1997; the 4 years when data were collected by a single observer (Linear regression; $F_{1,3} = 51.30$, $r^2 = 0.96$, $P = 0.019$).

Body condition showed a negative trend over time amongst some sex and age classes (Table 7.3). For adult males and adult females with offspring the proportion of bears in good condition during the ice-free period declined from 1993 to 2013. As similar trend, although not significant ($P = 0.065$), was evident for yearlings.

In Melville Bay, NW Greenland in 2012 and 2013 the adult bears were generally in good body condition. The samples collected from this area included 10 adult females, 5 adult males, 3 subadults (1 F, 2 M) and 2 female yearlings. Three adult females and 1 adult male scored FI = 4 and 7 adult females and 4 adult males scored F = 3. Three subadults (1 F, 2 M) and 2 female yearlings all scored F = 3. Hence, although sample size in Greenland was low, BB bears in this area were in good body condition despite an on-land period which is longer than BB polar bears that summer on Baffin Island.

Foraging During the Ice-Free period

One hundred and seven (9.3%) of the 1146 bears observed in 2012 and 2013 showed evidence of feeding. Prevalence of feeding was lowest amongst adult males (8%) and dependent offspring (7%), and highest amongst subadults (13%). Across sex and age classes, 50% of feeding observations were among adult males and subadults (Figure 7.3). The distribution of feeding observations amongst sex and age-classes did not differ significantly from the sex and age-class composition of all bears observed (feeding and not feeding); although there was a tendency for subadult bears to be over-represented amongst those observed feeding ($\chi^2 = 5.607$, d.f. = 4, $P = 0.23$). There was no seasonal trend in prevalence of feeding observations during the sampling period (Figure 7.4).

Bears were observed feeding on a range of food items including seals (species unknown), walrus (*Odobenus rosmarus*), narwhal (*Monodon monoceros*), Arctic char (*Salvelinus alpinus*), Greenland shark (*Somniosus microcephalus*) and berries (Table 7.4). Where food source was known, marine mammals comprised 47% of the observations of bears feeding. At two locations, congregations of bears were encountered along streams where char were observed to be running in large numbers. Fish carcasses found at these sites and the presence of bears standing in close quarters along these watercourses suggested bears were actively fishing. Seventeen seal kills were noted while searching for bears. Most were located along the shores of fjords rather than the main open coastline (Figure 7.5).

In Melville Bay polar bears were observed feeding on seals on patches of floes of fast ice and more or less consolidated bergy bits at glacier fronts. During September in both 2012 and 2013 numerous narwhal, ringed, bearded, harp and hooded seals were observed close to glacier fronts and land in Melville Bay (Born et al. 2012, 2013) indicating that suitable food for BB polar bears is abundant in this area during the open-water season.

## 7.4. Discussion

Body Condition of Baffin Bay Bears

Our results demonstrate that body condition amongst BB polar bears declined over the period 1993 to 2013. The close association between condition during the ice-free period and spring sea-ice transition date is consistent with the hypothesis that reduced time on the sea ice and presumably declining access to prey during the important spring to early summer feeding period is a primary mechanism driving this decline. However, with our qualitative body condition data and simplistic analysis we have not demonstrated cause and effect. Annual variation in body condition was associated with both time (year) and spring sea-ice transition date. These two parameters are correlated with one another and may also be associated with other parameters that we did not consider. The extent to which declining condition in BB is mediated by ice conditions therefore remains uncertain.

Using body condition metrics different from those used in the present study, Rode et al. (2011) detected a decline in the condition of BB polar bears between 1990 and 2010 concurrent with declining sea-ice cover. Our findings are consistent with this earlier study except that we did not find associations between body condition and sea-ice cover amongst all sex and age classes of bears. This may be due to limitations of the qualitative condition data used in the present analyses and sample size issues. The BCS is a qualitative and thus less precise measure of condition than the quantitative metrics used by Rode et al. (2011) that were derived from morphometric measurements acquired during physical capture and handling of bears.

Additionally, during the latter years of our study (2011-2013) the BCS for each bear was assessed based on examination from a distance rather than capture and physical examination. Inaccuracies in classifying bears by age class and sex from the air, combined with a lack of direct physical handling to assess condition may have introduced more error in assigning BCS. Given these limitations, BCS data are likely a less robust and less sensitive means of detecting changes in body condition over time or in response to ecological parameters (Vongraven et al. 2012, McKinney et al. 2014). Direct, quantitative measurement of body condition by morphometry or adipose tissue lipid content (McKinney et al. 2014) remains the most reliable and precise means of monitoring condition.

Nevertheless, our study extends the findings of Rode et al. (2011) in three notable ways. First, we find that trends in body condition and the association with sea-ice conditions have continued beyond 2010. Second, Rode et al. (2011) suggest that important trends in body condition that can affect reproduction might not be detectable from on the ground observations without capture and physical measurement of bears. Our findings suggest that long-term trends in body condition can be detected without handling of bears albeit with less sensitivity. As found in other studies (Amstrup et al. 2006, Stirling et al. 2008a), visually assigned body condition scores are a useful means of monitoring body condition and polar bear responses to environmental conditions. In the absence of physical capture programs and / or more intensive monitoring schemes, the collection of condition scores provides a simple and low cost means to track general trends in BB and likely other polar bear subpopulations. Finally, York et al. (2016) maintain that the evidence linking reduced body condition to sea-ice decline in Baffin Bay (Rode et al. 2011) is ambiguous because the body condition data used in the analyses were collected in varying parts of the subpopulation area over the period of study rather than range wide throughout. For example, during 1990s data on condition during the ice-free period were collected from bears across the subpopulation's seasonal range on eastern Baffin Island. In contrast, sampling in the 2000s was restricted to a relatively small southern portion of the range near the boundary with Davis Strait. Consequently, condition data for the 2000s may not have been representative of the subpopulation as a whole. However, sampling in our study was range wide during both time periods (1993-1997 and 2011-2013) as illustrated in Figure 7.1 suggesting that the findings of Rode et al. (2012) were representative of BB.

Several sources of bias were possible in our study. BCS data were collected by several observers in different years and different parts of the study area. Notably a single observer collected all data during the 1990s. Several different observers collected data during the 2000s and in different parts of the study area from one another. To reduce potential observer bias in assigning qualitative condition scores, we employed a simplified body condition scoring system that required observers to discriminate between bears in poor, fair and good condition. All observers in the study were experienced polar bear biologists who had previously handled hundreds or thousands of bears in varying condition and should have been capable of easily discriminating bears in good condition. However, since individual bears were not scored by more than one observer, teasing out observer effects is challenging because differences in scored condition may reflect real temporal or spatial differences in the bears sampled. Never-the-less, several lines of evidence suggest observer bias was likely not a significant factor in our study. First, a majority of observations were made by a single observer within each time period (1990s and 2000s). Differences between these two observers in scoring body condition are unlikely to explain the linear and non-linear trends in condition we observed or the close association between condition and sea ice. Additionally, the statistical significance of these associations was maintained when analyses were restricted to a single time period when all observations were collected by a single observer. Finally, as noted by Stirling et al. (2008a) although the FI from which our condition metric was derived is a qualitative index and thus subjective, it has been found to be "*repeatable between individual biologists when blind comparisons are done in the field over both short and long time periods*." In other studies, FI data collected by multiple observers have been found to correlate closely with quantitative indices of condition (e.g., Stirling et al. 2008b, McKinney et al. 2014). Therefore, while we cannot exclude the possibility of observer bias in our study, we suggest that this potential source of bias is unlikely to account for our results.

Body condition scores in the 1990s were collected from bears captured and physically examined. In contrast scores in the 2000s were from bears observed from the air without handling. The effect of close-up versus distance examination on the scoring of condition is unknown. McKinney et al. (2014) found that remotely assigned FI ratings did not correlate with the % lipid content of adipose tissue; another measure of condition. However, their sample sizes were small and limited to comparisons of bears of FI 3 and 4 only whereas bears in our study had FI ratings ranging from 1 to 5. Remotely scoring FI may be a less robust (precise) index of condition but is not necessarily inherently biased relative to physical examination. For many of the same reasons discussed previously concerning observer effects, we suggest that this potential source of bias is unlikely to account for our results. The use of a simplified scoring system (poor, fair, good) in our study should have helped to reduce errors in scoring for bears observed from the air. Trends in condition over time and the close association with sea-ice metrics cannot be explained by differences in examination distance.

Another source of error in our study associated with differences in sampling between the

1990s (physical capture) and 2000s (aerial observation) was in the classification of bears by sex and age-class based on aerial observation rather than handling. Classifying bears from the air is without doubt less accurate than physical examination. However, aerial classification is accurate in most instances (Appendix B), especially amongst adult males and adult females with offspring; the two classes exhibiting the strongest trends in condition in our study. Additionally, despite being less precise we have no evidence to suggest that aerial classification results in a bias in age and sex classifications amongst a group of bears. This source of measurement error thus seems unlikely to account for our results.

An assumption of our study was that bears sampled within our study area were representative of the BB subpopulation. Although sampling during both the 1990s and 2000s was extensive across the seasonal range of BB bears, the proportion of bears in the subpopulation exposed to sampling may have differed between these two periods (Chapter 3). In the 1990s, a high proportion of collared bears did not come ashore on Baffin Island during the sampling windows but instead remained on remnant offshore sea ice where they could not be sampled. This observation suggests that a significant portion of the subpopulation was not sampled in the 1990s. Whether this biased our estimates of body condition is unknown. However, we note that bears remaining out on the ice were likely still able to hunt seals to some extent and may therefore have been in better condition than those coming ashore. Consequently, any bias in our sampling would have resulted in underestimation of condition in the 1990s or in years when spring transition occurred later. This would therefore not account for the trends in condition we observed over time or in association with date of spring sea-ice transition.

Relative to the 2000s, sampling in the 1990s was also more concentrated near the coast with less inland sampling (Chapter 3). Similar to other seasonally ice-free subpopulations (e.g., Derocher and Stirling 1990) polar bears in Baffin Bay are known to exhibit a degree of spatial segregation by sex, age class and reproductive status with respect to the use of terrestrial habitat during the ice-free period (Ferguson et al. 1997, Chapter 4). Adult females with cubs tend to select fjords and avoid offshore islands and coastal regions where densities of adult males are higher. Pregnant bears select inland and upland denning habitats. While limited inland sampling in the 1990s may have resulted in under sampling of certain sex, age and reproductive classes we are unaware of any evidence to suggest that this would also have biased body condition data. However, to explore the potential impact of this difference in sampling between epochs on the body condition data, we estimated and delineated the minimum extent of the sampling frame from the 1990s using capture locations in a GIS. We subset the 2000s data using this layer to create a dataset collected over the same sampling area in both time periods and repeated our analyses of trends in body condition. The results were essentially the same (Appendix C), suggesting that this sampling difference between epochs did not influence our findings.

Bears in BB lose condition through the summer and fall while on land in BB (Rode et al. 2011). Differences amongst years in the timing of sampling could therefore have affected our annual body condition metric. Sampling occurred between late August and late October but varied somewhat in timing from year-to-year. We did not consider timing of sampling in our analysis. During preliminary exploration of the data we noted that the 3 years where the median date (Julian day) of sampling was earliest were the best (1993) and two worst (2012 and 2013) years for body condition amongst both adult males and adult females with offspring, as measured by our metric. Median date of sampling was also not associated with our condition metric for any of the sex and age classes of bears. Similarly, looking at the number of days between spring transition date and the date of sampling for each bear as an index of timing of sampling relative to seas-ice breakup we found similar results suggesting that timing of sampling did not account for the annual variation in body condition that we were observing at a broad scale with our somewhat crude measure of condition. However, we acknowledge that a more sophisticated analysis such as a polynomial logistic regression could incorporate sampling date as a covariate.

Declining body condition and reproduction are amongst the first subpopulation level effects predicted occur in polar bears as a result of climate change (Derocher et al. 2004, Stirling and Parkinson 2006, Molnar et al. 2011, Stirling and Derocher 2012). Our evidence of a decline in condition in BB from 1993 to 2013, along with similar findings from a previous study (Rode et al. 2011), is accompanied by evidence of a concurrent decline in reproduction in this subpopulation. These findings are also consistent with available traditional knowledge suggesting that body condition of polar bears in BB was poorer in the early 2000s relative to the 1990s (Dowsley and Wenzel 2008, Born et al. 2011). These changes may signal a reduction in the carrying capacity of BB.

The bears that were biopsied in Melville Bay in September 2012 and 2013 generally appeared to be in good body condition. However, during an interview survey among experienced polar bear hunters in NW Greenland ca. 24% of the 72 interviewees noted that polar bears had generally become thinner (Born et al. 2011).

Similar to recent observations in the Davis Strait subpopulation (Rode et al. 2011, Peacock et al. 2013) we cannot rule-out possible density effects on body condition and reproduction resulting from a declining sea-ice platform. Regardless of whether density effects are at play, if the observed association between sea ice and body condition is real and the well documented trend in sea ice continues it is reasonable to expect that this subpopulation will experience an on-going decline in condition as a result of declining habitat (Laidre et al. 2015). This in turn is predicted to lead to demographic consequences including reduced adult survival (Molnar et al. 2010, 2011).

Measures of body condition have been identified as one of the most important metrics needed to evaluate polar bear health (Patyk et al. 2015). Similar to previous studies (e.g., Amstrup et al. 2006, Stirling et al. 2008a,b, McKinney et al. 2014) we have demonstrated the utility of a simple qualitative metric for monitoring trends in body condition in polar bear subpopulations where more detailed quantitative measures of condition may not be available. In circumstances where demographic studies are conducted periodically rather than on an on-going basis or where the selected methods of survey do not involved capture and handling, collection of visually assigned body condition scores from harvested bears or from opportunistic observations of free ranging animals offers a useful means of surveillance. Such surveillance may be carried out by government agencies but there is also potential for implementation as part of a community-based ecosystem monitoring scheme. Changes in condition detected through this method of monitoring may serve as a trigger to initiate more intensive studies.

We acknowledge that results from analyses of FI scores, including those of the present study, must be interpreted cautiously given the many potential biases associated with this type of data. While many of these potential biases can be mitigated through study design and analyses, further work is needed to examine the robustness of these data before this method of monitoring is implemented more widely in government or community-based monitoring schemes.

### Foraging During the Ice-Free period

Polar bears have been found to opportunistically exploit a wide variety of food sources while on land during the summer and fall (e.g., Derocher et al. 1993a, Brook and Richardson

2002, Dyck and Romberg 2007, Gormenzano and Rockwell 2013, Iverson et al. 2014, Rogers et al. 2015, Tartu et al. 2016). Most of these foods are terrestrial in origin including berries, bird eggs, birds, small mammals and occasionally large mammal prey such as caribou or reindeer. Although bears have been observed catching fish (Dyck and Romberg 2007) and seals in open water (Furnell and Oolooyuk 1980), and scavenging the bone piles of human hunted bowhead whales (Rogers et al. 2015), the available evidence suggests marine mammals generally represent a small portion of the diet during this period of minimum sea ice. In Western Hudson Bay, for example, Gormenzo et al. (2013) found evidence of marine mammal remains in less than 5% of polar bear fecal samples collected during the on-land period. In contrast, terrestrial foods such as vegetation and eggs made up the majority of material in these samples. Similar dietary habits have been documented in Southern Hudson Bay (Russell 1975)[2].

In Baffin Bay we found evidence of recent foraging in approximately 9% of the bears observed during the on-land period. The type of food consumed was known for approximately half of these individuals. Marine mammals, in particular seals and walrus made up almost half of the identifiable food sources. While these are opportunistic observations and therefore subject to numerous potential biases, these findings suggest that bears in Baffin Bay may make greater use of marine mammals during the ice-free period than bears in some other subpopulations. This may be the result of differences in habitat and / or the availability of marine mammal prey. In contrast to the lowlands of the Hudson and James Bays, the east coast of Baffin Island is characterized by rugged coastline with high mountains, long, deep fjords and glaciers some of which run directly into the marine environment. Of the seal kills documented during our study most were located along the shores of fjords rather than the main open coastline (Figure 7.5). As suggested by Derocher et al. (2004) fjords may offer preferred seal hunting habitat for polar

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[2] Russell (1975) found seal remains in 9% of polar bear scats collected on-land during the summer.

bears for several reasons. Stable sea ice at tide water glaciers provides prime breeding habitat for ringed seals (Lydersen et al. 2014). Additionally, remnant sea ice that persists longer into the summer, the shedding of ice from glaciers, the early formation of new sea ice around freshwater outflows and the availability of Arctic char and other food sources near the mouths of rivers may make fjords good habitat for seals. These same features may also make fjords good polar bear habitat. Stable and persistent ice provides a platform to hunt from and the steep sides of fjords give polar bears easy access to the deeper waters in which seals may be swimming during the open water period.

In Melville Bay polar bears were observed in September near or at glacier fronts where numerous ringed seals were also observed (Born et al. 2012, 2013). Satellite telemetry has shown that some bears remain in this habitat throughout the year for several years (Chapter 2). Hence, clearly some bears are able to sustain year round on prey (likely mainly ringed seals) taken in this type of habitat.

We observed a number of bears that appeared to be feeding on anadromous Arctic char during their seasonal runs into creeks and lakes. Observations of polar bears feeding on Arctic char have been previously reported (Dyck and Romberg 2007, Dyck and Kebreab 2009) and traditional knowledge of certain Arctic char runs that are visited annually by polar bears is present amongst residents of Nunavut (S. Atkinson pers. comm.). In Baffin Bay, the significance of Arctic char to polar bear nutritional budgets is unknown. However, we note there are numerous char runs and the available biomass is potentially high. While this food source is only available during a short seasonal window in late summer, the timing and location of char runs is highly predictable. This makes it a reliable source of food for some bears (at least locally) during the ice-free period when other foods are scarce. In some grizzly bear populations, access to anadromous fish has been shown to directly affect body mass, litter size and population density (Hildebrand et al. 1999, 2004). Whether Arctic char hold similar significance for polar bears is unknown but is worthy of further investigation. Based on energetics modelling, Dyck and Kebreab (2009) speculated that polar bears with access to char could in theory maintain or gain body weight during the ice-free period. In contrast, Rode et al. (2010b) suggested that the use of char by polar bears was limited by the availability of suitable water bodies (creeks and rivers) in which bears could capture anadromous fish with an energetic efficiency high enough to permit maintenance or gains in weight. To date, however, there have been no direct empirical studies of the significance of Arctic char in the diets and energetics of polar bears. Arctic char have not been included in prey models for quantitative free fatty acid signature analysis (QFFASA) studies of polar bear diet (e.g., Thiemann et al. 2008, 2009, Galicia et al. 2015). Consequently, this prey's signature would not have been detected in dietary studies conducted to date. We suggest that QFFASA models of polar bear diets should be calibrated to include the signatures of Arctic char sampled from the same regions as the polar bears being studied.

Sea-ice conditions are changing in Baffin Bay (Laidre et al. 2015, Chapter 4). Studies suggest this is affecting the movements and distribution (Chapter 2), habitat use (Chapter 4), body condition (Rode et al. 2011, this study) and reproductive performance (Chapter 6) of polar bears in the region. Declining condition and reproduction is presumably mediated by reduced per capita food intake but precisely how availability of food for polar bears is changing is unknown in part due to lack of knowledge about trends in marine mammal populations (Laidre et al. 2015). Changes in prey abundance and vulnerability to predation are both potential mechanisms. Changes in prey diversity are also possible. One such change may be increased access to sub-Arctic seals such as harp seals (*Pagophilus groenlandicus*). Baffin Bay is part of the summer range of the western north Atlantic harp seal population. The near 2.5 fold increase in this seal population over the last 30 decades is one of the mechanisms postulated to have supported an increase in polar bear abundance in the neighbouring Davis Strait (DS) subpopulation (Peacock et al. 2013). Similarly, McKinney et al. (2013) attributed improving body condition (expressed as adipose tissue lipid content) amongst polar bears in East Greenland (EG) to increased access to sub-Arctic seals including harp seals. Unlike DS and EG, however, bears in BB do not have access to harp seal whelping areas and are therefore not able to access this prey species during its most vulnerable season. For bears in BB, hunting of harp seals is limited to late spring through to fall when predation success rates amongst polar bears hunting in low ice cover or open water are likely relatively low. Polar bears in Svalbard are known to prey on harp seals in the summer (Derocher et al. 2002). Bears in BB may have similar summer foraging opportunities. Indeed, using QFFASA, Galicia et al. (2015) found that adult male polar bears from BB had a higher proportion of harp seal in their diet relative to bears from other subpopulations. This suggests that polar bears in BB have been able to benefit to some extent from the availability of this species. However, the observation that body condition amongst BB bears has declined over the last 3 decades suggests that any shift in prey availability associated with harp seal population expansion has not offset the effects of declining sea-ice conditions on access to other food sources.

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**Table 7.1**. Frequency of observations for body condition scores of polar bears in Baffin Bay. Proportion of within-year observations in parentheses.

| | Observer | | | | |
|---|---|---|---|---|---|
| **Year** | **1** | **2** | **3** | **4** | **Total** |
| 1993 | - | - | - | 149 (1.0) | 149 |
| 1994 | - | - | - | 220 (1.0) | 220 |
| 1995 | - | - | - | 243 (1.0) | 243 |
| 1997 | - | - | - | 285 (1.0) | 285 |
| 2011 | 31 (0.06) | 415 (0.87) | 36 (0.07) | - | 482 |
| 2012 | - | 529 (0.79) | 142 (0.21) | - | 671 |
| 2013 | - | 316 (0.70) | 134 (0.30) | - | 450 |
| Total | 31 | 1260 | 312 | 897 | 2500 |

**Table 7.2**. Frequency of body condition scores (BCS) assigned to polar bears on Baffin Island. BCS scores were derived from Fatness Index (FI) scores (1-5) assigned to polar bears during field observations (following Stirling et al. 2008b). FI scores of 1-2, 3 and 4-5 were assigned BCS of poor, fair and good respectively.

| | | | **BCS** | | | |
|---|---|---|---|---|---|---|
| **Sex-Age Class** | | | **Poor** | **Fair** | **Good** | **Total** |
| Adult Male | Year | 1993 | 7 | 13 | 30 | 51 |
| | | 1994 | 14 | 30 | 27 | 71 |
| | | 1995 | 21 | 28 | 31 | 80 |
| | | 1997 | 13 | 72 | 40 | 125 |
| | | 2011 | 16 | 88 | 35 | 139 |
| | | 2012 | 52 | 123 | 21 | 196 |
| | | 2013 | 23 | 86 | 12 | 121 |
| | Total | | 133 | 440 | 185 | 783 |
| Adult Female (Lone) | Year | 1993 | 2 | 2 | 6 | 10 |
| | | 1994 | 3 | 6 | 3 | 12 |
| | | 1995 | 0 | 4 | 2 | 6 |
| | | 1997 | 2 | 20 | 5 | 27 |
| | | 2011 | 3 | 26 | 14 | 43 |
| | | 2012 | 8 | 44 | 19 | 72 |
| | | 2013 | 5 | 26 | 24 | 55 |
| | Total | | 21 | 128 | 61 | 225 |
| Adult Female (w/offspring) | Year | 1993 | 4 | 13 | 8 | 25 |
| | | 1994 | 15 | 19 | 6 | 40 |
| | | 1995 | 15 | 22 | 5 | 42 |
| | | 1997 | 11 | 24 | 3 | 38 |
| | | 2011 | 7 | 75 | 9 | 91 |
| | | 2012 | 18 | 91 | 3 | 112 |
| | | 2013 | 8 | 63 | 4 | 75 |
| | Total | | 76 | 307 | 38 | 423 |
| Subadult Female | Year | 1993 | 3 | 7 | 2 | 12 |
| | | 1994 | 6 | 6 | 0 | 12 |
| | | 1995 | 4 | 14 | 0 | 18 |
| | | 1997 | 4 | 10 | 1 | 15 |
| | | 2011 | 3 | 31 | 3 | 37 |
| | | 2012 | 13 | 62 | 0 | 75 |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | 2013 | 4 | 35 | 3 | 42 |
| | Total | | 36 | 165 | 9 | 211 |
| Subadult Male | Year | 1993 | 1 | 7 | 1 | 9 |
| | | 1994 | 4 | 11 | 1 | 16 |
| | | 1995 | 14 | 13 | 2 | 27 |
| | | 1997 | 8 | 9 | 0 | 17 |
| | | 2011 | 4 | 19 | 1 | 24 |
| | | 2012 | 8 | 45 | 1 | 54 |
| | | 2013 | 8 | 27 | 1 | 36 |
| | Total | | 40 | 131 | 7 | 183 |
| COY | Year | 1993 | 2 | 22 | 2 | 26 |
| | | 1994 | 16 | 28 | 0 | 44 |
| | | 1995 | 15 | 15 | 4 | 34 |
| | | 1997 | 22 | 19 | 0 | 41 |
| | | 2011 | 4 | 80 | 12 | 96 |
| | | 2012 | 8 | 85 | 1 | 94 |
| | | 2013 | 5 | 64 | 8 | 77 |
| | Total | | 67 | 313 | 25 | 412 |
| Yearling | Year | 1993 | 1 | 8 | 6 | 15 |
| | | 1994 | 3 | 16 | 6 | 25 |
| | | 1995 | 7 | 18 | 5 | 30 |
| | | 1997 | 3 | 18 | 1 | 22 |
| | | 2011 | 0 | 43 | 5 | 48 |
| | | 2012 | 5 | 51 | 1 | 57 |
| | | 2013 | 5 | 35 | 2 | 42 |
| | Total | | 24 | 190 | 25 | 239 |
| 2-Year-Olds | Year | 1993 | 0 | 1 | 0 | 1 |
| | | 1995 | 2 | 3 | 1 | 6 |
| | | 2011 | 2 | 0 | 0 | 2 |
| | | 2012 | 0 | 8 | 1 | 9 |
| | Total | | 4 | 12 | 1 | 18 |
| Total | Year | 1993 | 18 | 73 | 51 | 149 |
| | | 1994 | 59 | 116 | 37 | 220 |
| | | 1995 | 71 | 117 | 45 | 243 |
| | | 1997 | 60 | 172 | 48 | 285 |
| | | 2011 | 38 | 362 | 76 | 480 |
| | | 2012 | 100 | 510 | 41 | 669 |
| | | 2013 | 55 | 336 | 53 | 448 |
| | Total[1] | | 401 | 1686 | 351 | 2494 |

[1] Excludes 6 bears of unrecorded sex-age class

**Table 7.3**. Regression results for an annual body condition metric for polar bears on Baffin Island. The metric, proportion of bears in good condition, was derived from observed frequencies of Fatness Index (FI) scores ranging from 1 to 5 (Stirling et al. 2008). Bears of FI 4 or 5 were in good condition. Spring ice transition was the decimal day (1-365) when ice cover over the continental shelf of BB reached 50%. Regressions were performed in the Curve Estimation procedure of SPSS (Version 24.0).

| **Sex-Age Class** | **Dependent Variable** | $F_6$ | $r^2$ | $P$ | ***Curve Type*** |
|---|---|---:|---:|---:|---:|
| Adult Male | Spring Ice Transition | 102.99 | 0.98 | ≤ **0.001** | Quadratic (2nd order) |
| Adult Male | Year | 18.50 | 0.79 | **0.008** | Linear |
| Adult Female (alone) | Spring Ice Transition | 0.65 | 0.12 | 0.456 | Linear |
| Adult Female (alone) | Year | 0.03 | 0.01 | 0.863 | Linear |
| Adult Female (with offspring) | Spring Ice Transition | 53.29 | 0.91 | **0.001** | Exponential |
| Adult Female (with offspring) | Year | 7.31 | 0.59 | **0.043** | Exponential |
| Yearling | Spring Ice Transition | 21.57 | 0.81 | **0.006** | Exponential |
| Yearling | Year | 5.526 | 0.53 | 0.065 | Exponential |
| COY | Spring Ice Transition | 0.10 | 0.02 | 0.760 | Linear |
| COY | Year | 0.334 | 0.06 | 0.587 | Linear |

**Table 7.4**. Food sources used by bears on Baffin Island during Aug to Oct, 2012 and 2013.

| Food Source | Number of Bears Observed |
|---|---|
| Berries | 10 |
| Arctic Char | 14 |
| Walrus | 10 |
| Seal | 11 |
| Narwhal | 1 |
| Greenland Shark | 1 |
| Unknown | 60 |
| Total | 107 |

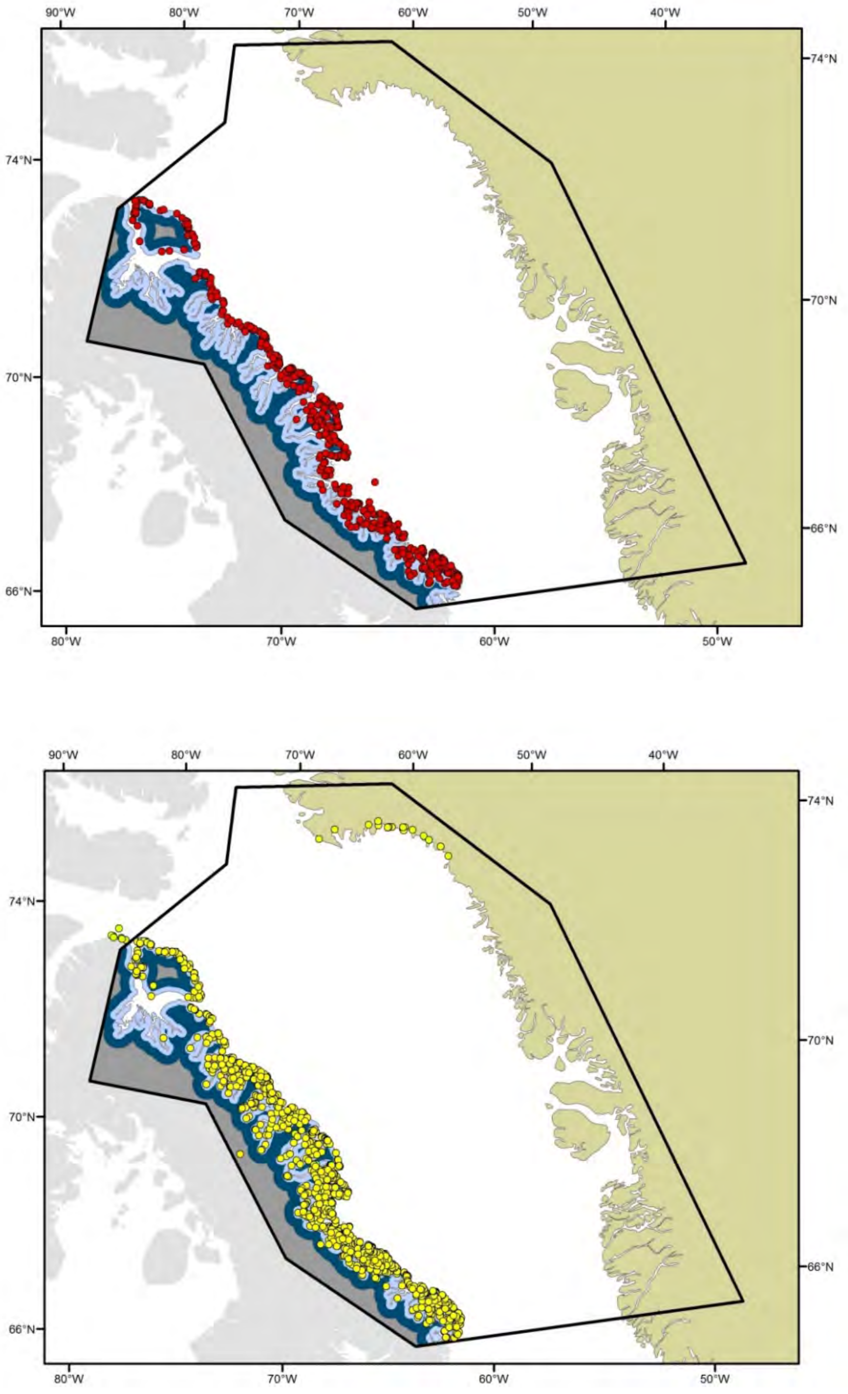
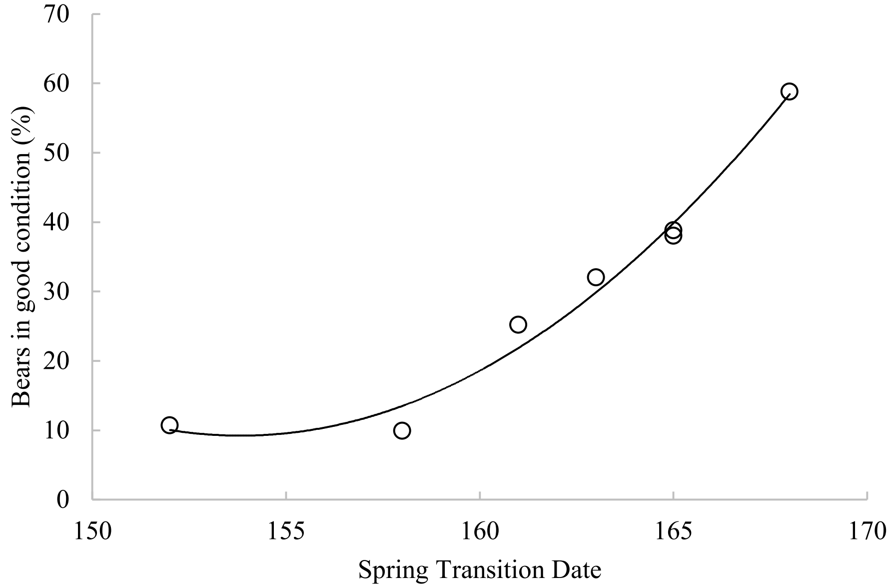


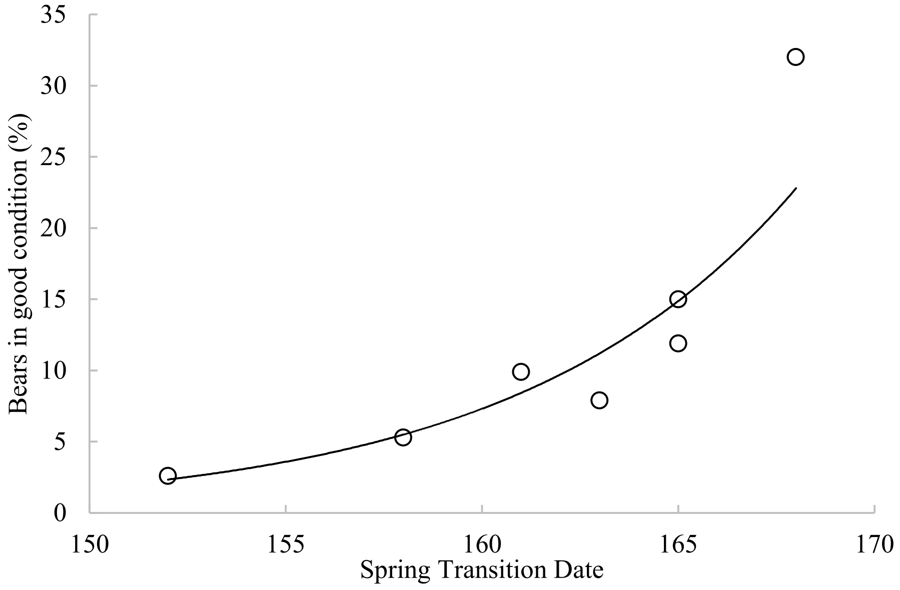
**Figure 7.1.** Spatial distribution of live captures recorded during sampling in the western parts of the range of the Baffin Bay polar bear subpopulation, 1993 – 1995, 1997 (top), and 2011 – 2013

(bottom). Red, 1990s. Yellow, 2010s. During both periods polar bears were also live captured in the eastern parts (i.e., the Melville Bay area) of the subpopulation´s range (data not shown).

(a)


70
60
50
40
30
20
10
0
Bears in good condition (%)
150
155
160
165
170
Spring Transition Date

(b)


35
30
25
20
15
10
5
0
Bears in good condition (%)
150
155
160
165
170
Spring Transition Date

(c)

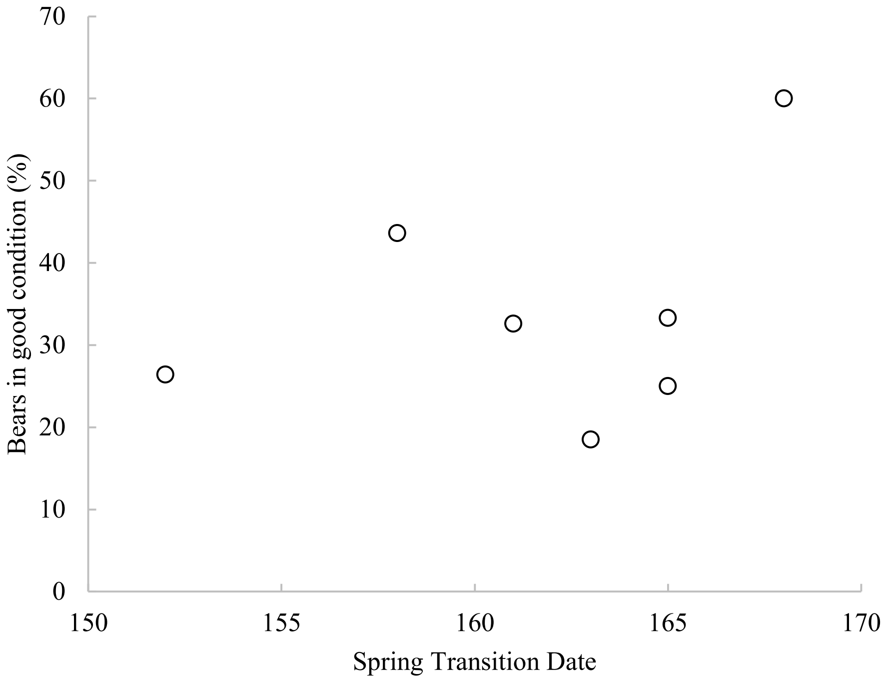


**Figure 7.2.** The association between the percent of bears in good body condition in western Baffin Bay and the timing of spring sea-ice transition date for (a) adult males, (b) adult females with dependent offspring and (c) lone adult females.

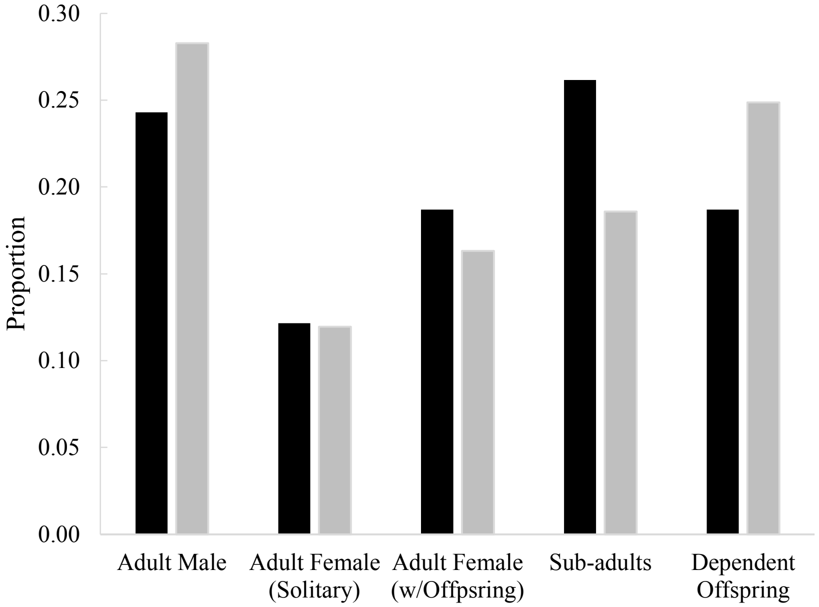


**Figure 7.3**. The sex and age class distribution of polar bears showing evidence of recent feeding (black) as compared to all the bears observed (grey) on Baffin Island in Baffin Bay during August-October, 2012 and 2013.

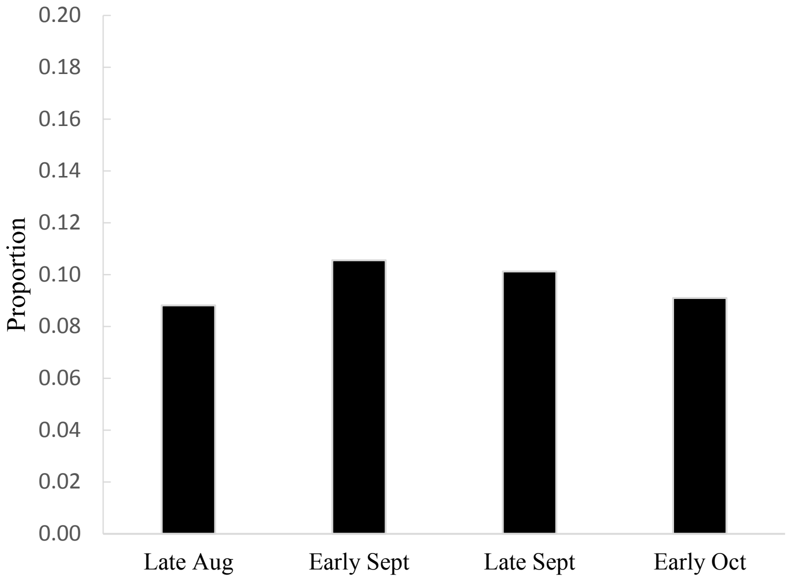


**Figure 7.4**. Prevalence of feeding evidence amongst bears on Baffin Island in Baffin Bay, 2012 and 2013. Data presented bimonthly.

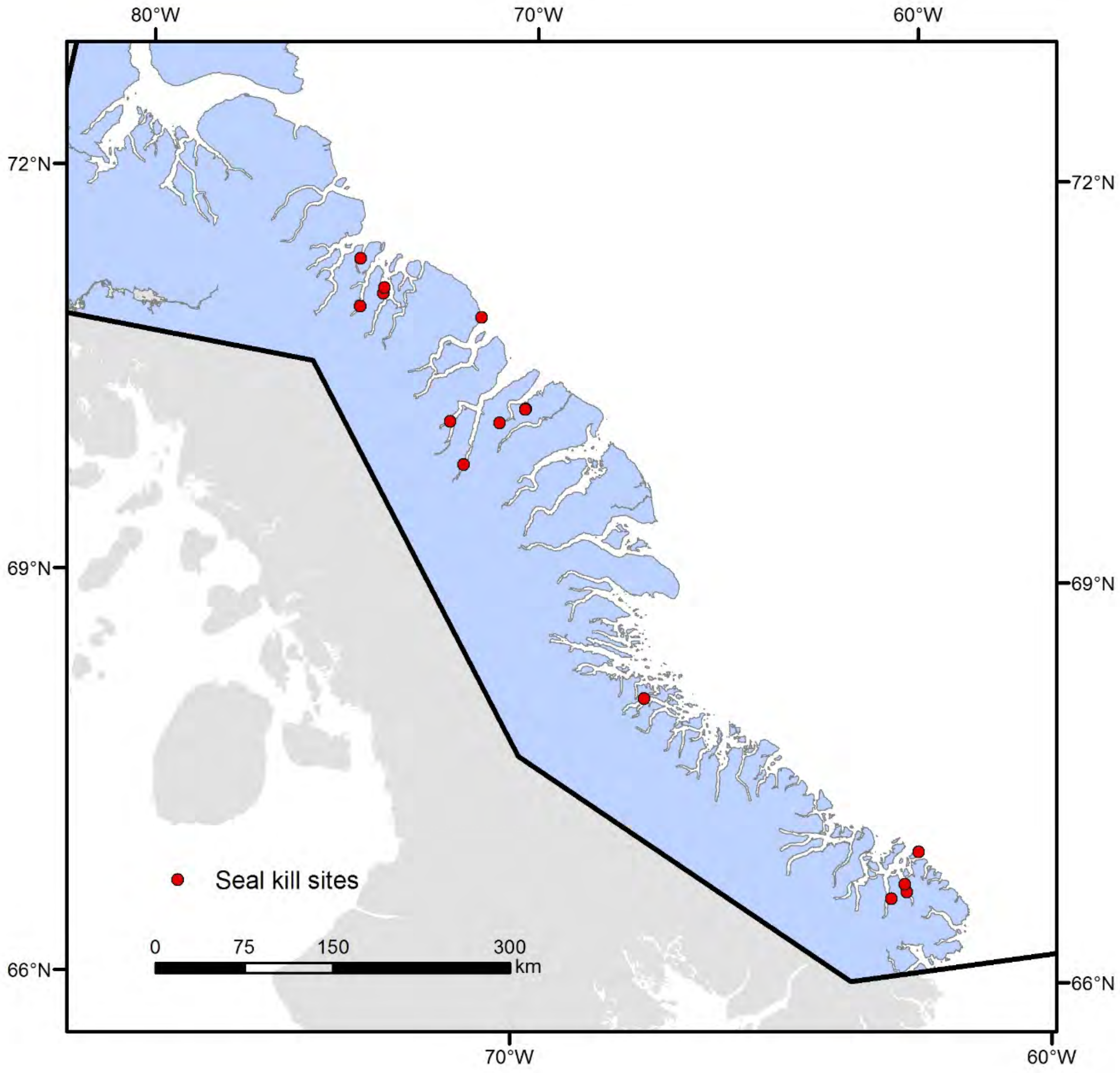


**Figure 7.5**. Distribution of seal kill sites observed during polar bear biopsy darting along eastern Baffin Island, Aug-Oct 2011-13.