

CHAPTER 8

POLAR BEAR HARVESTING IN BAFFIN BAY AND KANE BASIN: A SUMMARY OF HISTORICAL HARVEST AND HARVEST REPORTING, 1993 TO 2014

KEY FINDINGS

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- Both Canada (Nunavut) and Greenland harvest from the shared subpopulations of polar bears in Baffin Bay and Kane Basin.
 - During 1993-2005 (i.e., before quotas were introduced in Greenland) the combined annual harvest averaged 165 polar bears (range: 120-268) from the Baffin Bay subpopulation and 12 polar bears (range: 6-26) from Kane Basin (for several of the years, harvest reported from Kane Basin was based on an estimate).
 - During 2006-2014 the combined annual harvest averaged 161 (range: 138-176) from Baffin Bay and 6 (range: 3-9) polar bears from Kane Basin.
 - Total harvest peaked between 2002 and 2005 coinciding with several events in harvest reporting and harvest management in both Canada and Greenland.
 - In Baffin Bay the sex ratio of the combined harvest has remained around 2:1 (male: females) with an annual mean of 35% females amongst independent bears.
 - In Kane Basin the sex composition of the combined harvest was 33% females overall for the period 1993-2014. The estimated composition of the harvest since the introduction of a quota in Greenland is 44% female but the factual basis for estimation of the sex ratio in the harvest is weak.
 - In Greenland the vast majority of bears are harvested between January and June in Baffin Bay and Kane Basin whereas in Nunavut *ca.* 40% of the harvest in Baffin Bay is in the summer to fall (August – November) while bears are on or near shore. In Nunavut, all bears harvested from Kane Basin occurred in the spring.
 - Sport hunting of polar bears is permitted in Canada but not Greenland. Sport hunting activity average 16% of annual harvest and peaked in 2008 coincident with several management actions. This type of hunting is highly selective for older, adult males.
 - Defense-of-life-and-property kills (DLPs) of polar bears in Baffin Bay was highly selective for young (2-3 years old) individuals. We did not find evidence of a trend in the annual number of DLPs between 1993 and 2014. Most DLPs occur during the open water
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period (August-November) when bears are on or near shore. The seasonal frequencies of DLPs in Canada (Nunavut) declined through the summer and fall.

- Genetic validation of the sex of individual bears as reported by the hunters showed that the gender of harvested bears was incorrectly reported in a significant number of cases. Inaccuracies in gender reporting were greatest in Greenland.
 - Based on genetic validation, the presence of physical marks (i.e., ear tags or tattoos) on bears was under reported in the harvest. Detection of marked bears declined with increasing age of marks suggesting that mark loss (especially loss of ear tags) was a problem. This finding has implications for the use of harvest recoveries in MR recapture studies.
 - Management and the history of management of polar bears in Canada and Greenland differs in some respects: (1) quotas were introduced in Canada in 1967 and in Greenland in 2006, (2) sport hunting is allowed in Canada but not permitted in Greenland, (3) Canadian management is designed to achieve a target harvest sex ration of 2:1 (males to females) whereas the harvest of independent polar bears in GL is non-selective, (4) the vast majority of polar bears in Greenland are taken from dog sleds whereas in Canada the vast majority are taken from snowmobiles.
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8.1. Introduction

Climate induced loss of sea-ice habitat has been identified as the ultimate threat to the persistence of polar bears across their circumpolar range (PBSG 2010; Wiig et al. 2015). Other threats to the species, including pollution, industrial development, tourism and over-harvest are considered to be of varying importance amongst the different subpopulations. Three of the 19 recognized subpopulations of polar bears are currently designated as declining by the IUCN/SSC Polar Bear Specialist Group (<http://pbsg.npolar.no/en/index.html>). Of these three (Southern Beaufort Sea, Baffin Bay and Kane Basin), potential over-harvest has been identified as a concern in Baffin Bay (BB) and Kane Basin (KB).

Based on a mark-recapture study, Taylor et al. (2005) estimated the number of polar bears in BB at 2,074 (95% CI: 1544-2604) in 1998. A subsequent population viability analysis (PVA) using the 1998 abundance estimate, associated vital rates and available harvest data projected a decline to 1,564 bears (95% CI: 690-2402) in 2004 (PBSG 2010). Importantly, these

simulations suggested that the combined harvest of bears from BB within Canada and Greenland was unsustainable. Furthermore, this PVA did not take into account on-going sea-ice habitat loss which was predicted to exacerbate the potential impacts of the harvest. In contrast to these scientific findings traditional ecological knowledge (TEK) of polar bears in BB does not suggest this subpopulation is presently declining in response to harvest and / or sea ice (Dowsley and Wenzel 2008, Born et al. 2011). York et al. (2016) have postulated that this disparity between science and TEK may be the result of inaccuracies in the scientific data for BB. In particular, potential under reporting of tagged bears in the harvest and / or over reporting of total harvest in Greenland (both under and over reporting of the harvest may result in bias in mark-recapture estimates of abundance and survival rates). However, the validity of these assertions regarding harvest reporting is unproven.

Based on a mark-recapture study, the size of the KB subpopulation was estimated to be 164 (95% CI: 94-234) polar bears for 1994 - 1997 (Taylor et al. 2008a). Taylor et al. (2008a) also suggested that Kane Basin might act as a sink (i.e., some bears may move from Baffin Bay into Kane Basin) because of unsustainable rates of harvest in KB (ibid.), and lack of genetic differentiation from Baffin Bay (Paetkau et al. 1999). Using the abundance and vital rates estimated from the 1994-97 capture study and catches reported from Nunavut and Greenland, 100% of the PVA simulations resulted in a decline in this subpopulation size after 10 years (PBSG 2010). Similar to BB, this simulated decline in subpopulation size as a result of harvest did not take into account changes in sea-ice habitat. York et al. (2016) postulated that the available TEK for KB, although limited, did not appear to contradict these scientific findings. However, neither of the sources cited by York et al. (2016) – i.e., COSEWIC (2008) and M.K. Taylor (pers. comm. 1986-2008) - bring any documentation for this statement. In contrast, a

detailed study involving the interviewing of Greenland hunters with extensive experience hunting polar bears in the Kane Basin region documented the TEK perception that the occurrence of polar bear in KB has increased (Born et al. 2011).

In response to concerns regarding the projected declines in abundance, the impacts of harvest, on-going sea-ice habitat loss and the apparent disparity between scientific findings and TEK, Canada and Greenland jointly initiated new mark-recapture surveys of the BB and KB subpopulations from 2011-2014. Results from these surveys are presented elsewhere in this report (Chapters 5 and 10). Here we summarize available polar bear harvest information for BB and KB for the period 1993 to 2014; the period spanning both the recent and previous surveys in these subpopulations. Furthermore, a new system of reporting the catch of polar bears was introduced in Greenland in 1993. The objectives of this summary are to provide background information for discussion around future harvest levels and subpopulation status in light of results from the new surveys. We examine the level and composition of the harvest looking for trends over time or in relation to other factors. We also examine the harvest monitoring systems to identify issues that may affect the accuracy of harvest monitoring and the ability of jurisdictions to effectively manage harvest. Finally we make recommendations on improvements to the harvest reporting/monitoring systems.

Both BB and KB are jointly managed by Canada (Nunavut) and Greenland. Systems of harvest management and monitoring differ substantially between these two jurisdictions. Detailed descriptions of the respective systems and relevant history are presented in Appendix D. This information constitutes an important reference for the results presented herein.

8.2. Methods

Canadian Harvest Data

Data on polar bears harvested from BB and KB within Canada (Nunavut) were obtained from the Government of Nunavut's (GN) polar bear harvest database. This database contains records for all bears reported to have been harvested from these subpopulations. The methods of data collection are described elsewhere (Appendix D). From this database we extracted records for bears harvested from Jul 1, 1992 to Jun 30, 2014. The regulatory season for polar bear hunting in Canada (Nunavut) is from July 1 to June 30 the following year. This differs from Greenland where the regulatory season for polar bear hunting is from Jan 1 to Dec 31. Because adjustments in Total Allowable Harvest (quota) are made on a seasonal basis in response to recorded harvest levels in the preceding season, harvest figures were reported by season rather than by calendar year. For example, we used the notation 1992/93 when reporting harvest figures for the season beginning Jul 1, 1992 and ending Jun 30, 1993.

The extracted data included the sex, age class (adult, sub adult, 2-year-olds, yearling and cub-of-the-year [COY]), age (years), date, location and type of harvest for each individual. In reporting total harvest levels we considered all forms of human-caused mortality including illegal harvest and accidental kills but excluded bears killed for humane reasons (i.e., diseased or dying from natural causes). Other analyses were limited to the three main types of harvesting: Regular (subsistence) hunts, sport hunts and the harvest of bears in Defense-Life-and-Property (DLPs). We examined harvest by time (year or season), sex, type, monthly distribution and age. Although known for most individuals, records where sex, age class or date of harvest was unknown were excluded from the corresponding analyses where those parameters were used. Individual ages (years) based on ageing of harvested bears were only available up to June 2010 limiting our analyses of age of harvest to the period 1992/93 – 2009/10.

For DLPs we also examined the relationship with several annual metrics of sea ice to test the hypothesis that the number of human-bear conflicts, hence DLPs, increases in years when sea-ice breaks up earlier and/or forms later thereby forcing bears to remain on land for longer periods without access to their marine mammal prey (Stirling and Parkinson 2006, Stirling and Derocher 2012). As an index of the timing of spring sea-ice break-up, spring transition date was calculated as the date (Julian day) that spring sea-ice cover reached 50% over the continental shelf of BB. As an index of the timing of sea-ice formation in the fall, fall transition was calculated as the date (Julian day) that fall sea-ice cover reached 50% over the continental shelf of BB. The difference between spring and fall transition dates was used as an index for the length of the ice-free season. Additional details of these sea-ice metrics are provided in Chapter 4.

Greenlandic Harvest Data

Magnitude of the Annual Catch from KB and BB – For the period, 1993-2005 (i.e., prior to the introduction of quotas in 2006), data on the Greenland annual catch of polar bears from the KB and BB subpopulations were based on the “Piniarneq” reporting-system (for a description of this system see Appendix D). Information on catches reported via this system is compiled annually and published by the Greenland Government’s Department of Fisheries, Hunting and Agriculture (DFHA) in Nuuk. However, after the introduction of quotas when the polar bear harvest was more closely monitored detailed information on every kill was collected and compiled by the DFHA. Data for this report on the total annual kill of polar bears in 1993-2005 and in 2006-2014 were provided by the DFHA (*in litt.* August 2015)

Seasons of Catch – Information on date (day/month/year) of each polar bear catch was provided by the DFHA (*in litt.* 2014, 2015). This information was used to describe the seasonal

distribution of the Greenland catch of polar bears from the KB and the BB subpopulations during 2006-2014.

Means of Transportation When Hunting Polar Bears – For each bear caught, hunters reported the type of vehicle used during the hunt (i.e., dog sled, skiff or boat; or a combination of these means of transportation). The terms “skiff” describes a boat up to 20 feet in length and usually powered with an outboard engine. Boats larger than 20 BRT/15 BT must not be used during polar bear hunts or for transportation to or from polar bear hunting grounds (Anon. 2005). These boats are small-type fishing vessel made of wood, fiber glass or steel with an inboard engine.

The information reported during 2006-2014 on vehicles used during 445 individual polar bear hunts from the KB and BB subpopulations was used to describe hunting methods both overall and regionally. Due to differences hunting traditions not at least related to stability and duration of the sea-ice cover and therefore distribution of polar bears there are regional variations in hunting methods from north to south in NW and W Greenland (*cf.* also Born et al. 2011). The NW and W coast of Greenland between Kane Basin and the town of Sisimiut was therefore subdivided into six areas: (1) Subarea 1 compasses the area to the north of Ullersuaq/Kap Alexander - 78° 10' N (i.e., Kane Basin), (2) subarea 2 consists of the area between Ullersuaq and Innaaganeq/Kap York (76° 30' N), (3) subarea 3 encompasses the areas between Innaaganeq/Kap York and the peninsula Nuussuaq/Kraulshavn at *ca.* 74° N (i.e., the Melville Bay area), (4) subarea 4 comprises the southern Upernavik area between 74° N and the peninsula Sigguk at 71° 30' N, subarea 5 is the areas between Sigguk and the town of Aasiaat at 68° 45' N (i.e., the Uummannaq, Disko Island and Disko Bay area), and subarea 6 comprises the area between Aasiaat and Sisimiut at *ca.* 66° 55' N.

Sex and Age Composition of the Catch – Before the introduction of quotas information on the sex and age composition of the catch from KB and BB was obtained during interview surveys (Rosing-Asvid and Born 1990, Rosing-Asvid 2002, Born et al. 2011), and few biological samples (Rosing-Asvid 2002). Information on sex and age composition (i.e., independent males and females and dependent cubs) was extracted from these sources.

During an interview survey among experienced polar bear hunters in 2006 in NW Greenland (i.e., the Upernavik and Qaanaaq areas) detailed information was collected on age category (i.e., old, adult, young, and cubs) and sex of the individual bears that the interviewees had shot. Information on the composition of 588 catches (754 individual bears) going back to the early 1950s was presented in Born et al. (2011). Data on age and sex in the Greenlandic catch of polar bears during 1991-2005 (n = 354) were extracted from the interview survey database (Greenland Institute of Natural Resources, GINR, unpublished) and presented separately for the Kane Basin (KB) and Baffin Bay (BB) subpopulations.

Information on sex and age category (cub, young, adult, old) from individual polar bear kills was obtained from the DFHA based on the filled in forms provided by the licensed hunter for each polar bear killed (2006-2014). In case of catches reported by hunters living in the northernmost area (i.e., the Qaanaaq area) a polar bear may have been taken from either the KB or the BB subpopulation. In several cases coordinates of the location of kill was not given whereas the Greenlandic name of the site was noted by the hunter. In most cases it was possible to determine whether a bear was killed inside the KB management unit (i.e., north of the southern border of the KB management unit area on the Greenland side of the mid-sector line in Smith Sound; this border had been placed more or less arbitrarily at exactly 77° N; cf. Derocher et al. 1998, Taylor et al. 2001), or alternatively in the BB management unit. In cases where

neither coordinates nor name of site of kill were stated, it was assumed that if the reporting hunter lived in the northernmost settlement Siorapaluk the bear had been taken in KB. The hunters in this settlement traditionally harvest bears from the KB management unit (Rosing-Asvid and Born 1990, Born et al. 2011). About 10.0% (5 of 49) of the kills (2006-2014) allocated to KB were based on information about hometown of the hunter reporting (or *ca.* 3.5% of a total of a total of 147 catches reported from the Qaanaaq area).

The file contained information on sex and age category of *ca.* 96% of the total catch reported for KB during 2006-2014, and *ca.* 92% of the catches reported for BB during the same period.

Detailed Age Structures – The age (and sex) structure of the Greenland catch of polar bears was examined using several sources of data. Samples (soft tissues and a vestigial tooth, i.e., 1st premolar) were collected from 55 BB polar bears caught by the hunters in 2012 and 2013. This sample represents *ca.* 40% of the reported catch in Greenland from the BB subpopulation in 2012 and 2013. The gender of each sample was determined genetically (Chapter 5). Individual ages were obtained from counting growth layer groups in premolar teeth (Calvert and Ramsay 1998).

In 2009-2013, a total of 131 individual BB-polar bears were live captured by GINR during spring (Mar 23-Apr 23) in W and NW Greenland (i.e., between 70° 14' N and 76° 20' N; i.e., between northern Disko Island and the settlement Savissivik) in connection with a study of movement (GINR unpublished data). These bears were tagged on fast ice and in the offshore pack ice in areas where the hunters usually take polar bears and therefore likely represent the sex and age composition of bears available on the polar bear hunting grounds. The gender of each sample was determined in the field and verified genetically (Chapter 5). Individual ages were

obtained from counting growth layer groups in premolar teeth (Calvert and Ramsay 1998). The sex and age composition of this sample was included for comparison. Also for comparative purposes the age structure of a sample of 63 polar bears collected by the subsistence hunters in NW Greenland during 1988-1996 was inferred from figure 26 in Rosing-Asvid (2002:21).

Pooling of Canadian and Greenlandic Harvest Data

In contrast to Greenland where harvest is reported on a calendar year, harvest in Canada (Nunavut) is reported and quotas are adjusted on a hunting season basis, as described above. This presented some challenges in pooling data from the two jurisdictions. However, as documented later in this report, most harvesting in Greenland (99% in BB and 87% in KB) occurs between January 1 and June 30. We therefore pooled Greenland and Canadian data and report total harvest for BB and KB on the basis of Canadian harvest seasons. For example, harvest for 1998/99 includes bears harvested in Canada from July 1, 1998 to June 30, 1999 and in Greenland for January 1 to December 31, 1999.

Genetic Monitoring of Harvest

From 2011 to 2014, as part of genetic mark-recapture studies, the polar bear harvest in BB, KB and surrounding subpopulations was monitored to detect recoveries of genetically ‘marked’ bears. Tissue samples from harvested bears were submitted for genotyping and genetic sexing as described elsewhere in this report (e.g., Chapter 5). This provided data on genetic sex for comparison with the reported sex of each individual thereby allowing us to examine the accuracy of gender reporting. Additionally, several groups of bears that were physically marked with ear tags and lip tattoos during previous studies were detectable from this genetic monitoring

of harvest. Archived tissues samples from all bears physically marked in BB and KB from 1990 to 1997 were genotyped and genetically sexed with the exception of bears whose ages would have been >35 years when genetic monitoring began in 2011 and a small number of individuals ($n = 15$) for which archived tissue samples were unavailable. Bears physically marked in BB and KB from 2010-2013 were also genotyped. Finally, bears marked during a recent physical mark-recapture study in the neighbouring Davis Strait subpopulation (Peacock et al. 2013) were genotyped. The harvest of one of these genetically and physically marked individuals could be detected by two means; the presence of ear tags or lip tattoos as reported by hunters or by matching the genotype of a harvested bear to that of a previously marked bear. This permitted a comparison of the efficiency of detection of marked bears by these two methods.

8.3. Results and Discussion

Canadian Harvest from BB and KB

Annual Harvest from Baffin Bay – In Canada (Nunavut), a total of 1,633 bears were harvested from the BB subpopulation over the period 1992/93 to 2013/14. Mean annual harvest was 74 bears per year ($SD = 17.13$, range 49-103, $n = 22$). Total Allowable Harvest (TAH) in Nunavut for the BB subpopulation over this period varied from 64 bears per year in the 1990s up to 105 starting in 2004/05 (Table 8.1). This variation reflects harvest management initiatives at the subpopulation level. An increase in TAH was implemented from 2004/05 to 2009/10 as a result of a new memorandum of understanding (MOU) between the Government of Nunavut and communities in BB. Starting in 2010/11, in response to concerns about the sustainability of the combined Canadian and Greenlandic harvest level, the TAH in Canada (Nunavut) was reduced by 10 bears per year over four years and had decreased to 65 by 2013/14. In most years, harvest

from BB remained within the TAH. From 1996 onwards, exceedances of the allowable harvest resulted in adjustments of allowable harvest for the following year, in accordance with the flexible quota system (see Appendix D).

A majority (96%) of the harvest was comprised of independent bears (i.e., ≥ 2 years old) consistent with regulatory prohibitions on harvesting of family groups except in Defense-of-Life-and-Property (DLP). Also in accordance with the management system, the harvest was sex selective. Pooling data across years, the proportion of females in the harvest was 0.34 for independent bears (excludes COY and yearlings) and 0.35 for all bears which is consistent with the target sex ratio of 0.33 (i.e., 2 or more males per female). From 1996 onward, annual variation around this sex ratio (Figure 8.1: range 0.22 – 0.45) was regulated by the flexible quota system. When more than the recommended number of females were harvested, the TAH for the following season was reduced to compensate for the over harvest of females and deviation above the target sex ratio. Consequently, there was no temporal trend in harvest sex ratio.

Annual Harvest from Kane Basin – In Canada (Nunavut), harvest from KB has been minimal over the period 1992/93 to 2013/14 (Table 8.2). Total harvest during this period was 9 bears with a mean annual harvest of 0.4 bears per year (SD = 0.59, range = 0-2, n = 22). The sex ratio of the pooled harvest data (1993-2014) was 33% females (i.e., 2 males per female) consistent with the target sex ratio of the management system. The TAH in Nunavut of 5 bears per year for KB since 1996 has not been exceeded in any year. All bears were harvested in the spring (February to May).

The low level of hunting in KB is in part due to its remoteness and the logistical challenges of travelling in this subpopulation. The nearest Canadian community is Ajuittuq/Grise Fiord on southern Ellesmere Island with a population of <200 people. Access to

KB from Ajuittuq/Grise Fiord involves overland travel across southern Ellesmere Island. Costs for equipment and supplies make this journey less feasible relative to other hunting opportunities in the region (such as Jones Sounds in the Lancaster Sound subpopulation). In addition, spring-time travel into Kane Basin has been increasing in difficulty in recent years due to snow conditions (too little or early melts) on overland trails (M. Akeagok, pers. comm.).

Traditionally the Kane Basin region was a main polar bear hunting ground for the Inuit living in the Qaanaaq area in Northwest Greenland (Vibe 1968) and it is still an important hunting area for them (Born et al. 2011).

Given the sparsity of the Canadian harvest from KB we do not report further on it in this section.

Timing of Harvest from Baffin Bay – Month of harvest was recorded for 1,594 of the 1,633 bears harvested in BB between 1992/93 and 2013/14. Pooling data across years, we examined the distribution of hunting activity across months. Approximately 40% of annual harvesting from Baffin Bay occurred between August and November when bears were on or close to shore. Harvesting activity peaked in October (23% of total).

The three main types of human-caused mortality (i.e., subsistence hunts, sport hunts and the killing of bears in defense, DLP) each varied differently in frequency throughout the year (Figure 8.2). Subsistence hunting activity (n = 1,107), termed “regular” hunting, peaked in October (29% of total hunts of this type) and continued through the fall, winter and spring until sea-ice break-up. A majority (91%) of sport hunting (n = 248) occurred on the sea ice in spring (March-May). DLPs kills (n = 185) peaked in August and declined steadily through the fall. Fifty percent of DLPs occurred in August and September. Notably, this peak in DLPs coincides with the time when bears move to shore from the melting sea ice rather than the timing of freeze-

up (Taylor et al. 2005; Chapter 4). This finding is similar to Dyck (2006) but inconsistent with the hypothesis that human-bear conflicts are driven by poor body condition (Stirling and Parkinson 2006) since body condition will be at a seasonal high point in August, decreasing progressively through the fall (Rode et al. 2011). We speculate that this peak may result from the concentration of bears along the coast as they come off the ice (prior to dispersing inland) combined with the increased range and frequency of boat travel and use of camps by residents of communities along Baffin Island at this time of year. Hence higher rates of DLPs during the summer may be the product of human-bear encounter probability. However, this finding does not discount the hypothesis that bears in poor condition are more likely to come into conflict with people.

Number of Sport Hunts in Baffin Bay – Overall, sport hunting accounted for 16% of the harvest from BB in Canada from 1992/93 to 2013/14. Sport hunting activity increased through the 1990s and early 2000s before sharply declining in 2009 (Figure 8.3). This decline in sport hunting after 2008 coincided with two events. First, the 2008 listing of polar bears as “threatened” under the US Endangered Species Act and the subsequent ban on importation of polar bear hides into the US pursuant to the US Marine Mammal Protection Act (US Department of the Interior: Fish and Wildlife Service 2008). Second, Canada’s issuance in 2010 of a negative non-detriment finding for Baffin Bay under the Convention on International Trade in Endangered Species (CITES) which triggered a ban on Canada’s export of polar bear hides from this subpopulation. These two events seemingly reduced the pool of sport hunters interested in hunting opportunities in BB (Weber et al. 2015).

Number of Defense-of-Life-and-Property Kills (DLPs) in Baffin Bay – Bears killed as DLPs are the only type of polar bear harvest that is not limited by quota in Canada. All DLPs

are deducted from the available quota but the total number of DLPs in a given year is not limited. The number of DLPs in BB varied annually with no apparent trend over the period 1992/93 to 2013/14 (Figure 8.4). DLP-related harvest averaged 8.6 bears per year or about 12% of annual harvest. There was no trend in the proportion of annual TAH allocated to account for DLP kills over the period 1993-2014 (Linear regression, $F_{1,21} = 0.19$, $P > 0.700$). Thus, problems bears do not appear to be using an increasing portion of the available quota in BB.

Restricting the data to independent bears (in this case, independent bears were defined as all individuals except for COYs and yearlings accompanying their mother) during the months of Aug to Nov, when most bears in BB were on land, did not reveal a trend in number of DLPs kills over time (Figure 8.4; Linear regression, $F_{1,21} = 1.404$, $P > 0.200$). The number of DLPs of independent bears during the Aug to Nov period was also unrelated to annual date of spring sea-ice transition (Linear regression, $F_{1,21} = 0.028$, $P > 0.800$), fall transition date (Linear regression, $F_{1,21} = 0.179$, $P > 0.700$) and the number of days between these two dates (used as proxy for length of the ice-free period: Linear regression, $F_{1,21} = 0.121$, $P > 0.700$). This suggests that earlier spring sea-ice break-up, later fall freeze-up and a lengthening ice free period was not associated with increased DLP kills in BB. This finding is inconsistent with the prediction that problem bear kills will increase as sea-ice habitat deteriorates and bears spend more time on land (Stirling and Parkinson 2006, Stirling and Derocher 2012) as has been observed in Western Hudson (Towns et al. 2009). However, we note that the relatively small samples size ($n = 112$) and shorter time series in our study relative to Towns et al. (2009) may have limited the findings. Moreover, unlike the situation in Western Hudson Bay where there is no subsistence hunting of polar bears in the province of Manitoba, some of the bears in BB that were taken by hunters and recorded as part of the subsistence harvest might otherwise have ended up becoming DLPs,

especially those harvested as they approached communities or camps. The number of DLPs recorded amongst our data is thus likely to be an under and somewhat variable estimate of actual or potential DLPs occurring. Our results relating frequency of DLPs to sea-ice conditions should therefore be interpreted with caution.

Grouping data on DLPs into multiyear blocks (1993-95, 1996-00, 2001-05, 2006-10, 2011-13), there was no significant difference in the timing (mean Julian day) of DLPs of independent bears during Aug-Nov (ANOVA, $F = 0.846$, $P > 0.40$).

Sex Ratio of the Harvest in Baffin Bay – As reported above, the sex ratio of the harvest from BB for the period 1992/93 to 2013/14 was 0.34 amongst independent bears and 0.35 for amongst all bears. As expected there were significant differences in sex ratio amongst the different types of hunting ($\chi^2 = 16.03$, $P < 0.001$, $df = 2$) with sport hunting being selective for males (3:1 sex ratio) and DLPs being closer to a 1:1 ratio (Fig 8.5). Sport hunting contributes to the maintenance of a male-selective harvest; compensating for the less selective nature of DLPs.

Age Structure of Harvest in Baffin Bay – Several features of the harvest management system in Canada (Nunavut) that tend to select for or against bears in certain age ranges. While there is a regulatory limit (i.e., the TAH) on the total number of bears harvested each year and adjustments in this limit are made to compensate for the sex ratio of the harvest, there are no specific limitations on the age of bears harvested in BB; with one exception. A prohibition (under the Nunavut Wildlife Act) on the harvesting of family groups (defined as an adult female accompanied by COYs, yearlings or 2-year-olds), except in defense-of-life-and-property, protects most cubs-of-the-year (COY) and yearlings from hunting, as well as some 2-year-olds. This is reflected in the harvest data. Overall, between 1992/93 and 2013/14, *ca.* 20% (37/189) of

DLPs were adult females and their accompanying offspring. Comparable figures for regular hunting and sport hunts were *ca.* 0.4% (44/1104) and 0% (n = 260) respectively.

Sport hunting tends to select for larger, presumably older, adult bears. Looking at the available data on the known or tooth-derived age of harvested bears from BB, there were significant differences in the median ages of harvested female bears amongst the 3 main types of harvest: DLP, regular, sport (Fig 8.6: Kruskal Wallis, $H = 10.97$, $P = 0.004$). Females taken as sport hunts tended to be older than those harvested as DLPs. Similarly, the median age of male bears was significantly different amongst types of hunting (Fig 8.6: Kruskal Wallis, $H = 61.38$, $P < 0.001$). Sport hunting was highly selective for older males relative to both regular hunts and DLPs. DLPs selected for younger males.

Dyck (2006) found that a majority of DLPs occurring in polar bear subpopulations across Nunavut involved bears < 7 years old. Looking more closely at the age distribution of DLPs for Baffin Bay, we see that most bears ($\approx 60\%$) coming into conflict with people are ≤ 3 years of age. Specifically, juveniles aged 2 and 3 years were over-represented amongst the DLPs relative to the ‘population’ age structure derived from mark-recapture sampling, especially amongst males (Figure 8.7; Males: $\chi^2 = 64.55$, $df = 13$, $P < 0.001$; Females $\chi^2 = 30.41$, $df = 13$, $P < 0.005$). Bears aged 2-3 years are at a stage, between weaning and the on-set of sexual maturity, when they may be particularly vulnerable to conflict with humans (Towns et al. 2009). Food availability for these newly independent juveniles may be relatively low since hunting skills are still developing and their relatively small body size limits the ability to compete with larger bears for food. The demands of continued growth during a life-stage of relative food scarcity may lead to poor body condition and reduced survival amongst this age class particularly as environmental conditions deteriorate (Regehr et al. 2007). Poor body condition may increase the tendency for

juveniles to seek food from sites of human activity. Additionally, these individuals may be naïve with respect to risks of interacting with humans; having not yet acquired learned aversions to humans and adopted avoidance behaviors.

Annual variation in the ages of harvested bears was examined by sex and harvest type for the period 1992/93 and 2010; the period for which data were available. The median age of harvested bears did not exhibit trends over time for any of the different types of hunting (for example see Figure 8.8).

Greenlandic Harvest from BB and KB

Annual Harvest from BB – Sport hunting of polar bears is not permitted in Greenland (Anon. 2005). Hence, the only harvest types recorded by the management authorities are (1) regular subsistence harvest under quota, (2) bears killed in Defense-Life-and-Property (DLPs), and (3) illegal hunts.

The Greenlanders' catch of polar bears from the BB subpopulation according to the Piniarneq catch recording system (see Appendix D) during 1993-2005 (i.e., the year before introduction of quotas) are shown in Figure 8.9. The trend in numbers reported per year during 1993-2005 in the Qaanaaq area (i.e., north of the Upernavik area) is not statistically significant ($R = 0.272$, $Z = 0.881$, $P = 0.378$, $n=13$). However, the catch in the Uummannaq-Sisimiut area (i.e., south of Upernavik) increased statistically significantly during the same period ($R = 0.594$, $Z = 2.163$, $P = 0.031$, $n=13$). The catch in these areas amounted to *ca.* 24% of the total catch reported by Greenlanders from BB during 1993-2005. Similarly, the catch reported from the Upernavik area (between *ca.* 74° 35' N and *ca.* 71° 30' N) increased significantly during the same period ($R = 0.794$, $Z = 3.426$, $P = 0.001$, $n = 13$). On average the reported catch of polar

bears in the Upernavik area amounted to *ca.* 57% of the total catch reported in Greenland from BB during 1993-2005 (Figure 8.9). Hence, there are indications that the Greenland catch of polar bears from BB showed a real increase and especially after *ca.* 2000. An increase in availability of polar bears in the Upernavik area during the 1990s and 2000s was also indicated by traditional ecological knowledge, TEK (Rosing-Asvid and Born 1990, Born et al. 2011).

The Piniarneq-data have played an important role in the assessment of abundance and trends in abundance of the BB and the KB subpopulations. Since 1993, the Piniarneq-data on annual catch for the shared subpopulations (BB, KB and DS) as compiled and published by the DFHA have been provided annually to the Canadian Polar Bear Technical Committee and to the meetings of the IUCN/SSC Polar Bear Specialist Group. The data have been incorporated with the reported catch of polar bears in Nunavut from BB in modeling of subpopulation status (e.g., Aars et al. 2006, York et al. 2016). In a PVA (Population Viability Analysis) re-assessment of polar bear subpopulations including BB and KB based on historical data, York et al. (2016) simply assumed that the Greenland catch of polar bears reported in Piniarneq for BB was/are overestimated (“over-reporting”). However, these authors did not present any validation of the Piniarneq-data or new evidence in support of this assumption (Ibid.). As indicated in the previous other evidence suggests that the increase in the Greenlanders’ catch from BB during 1993-2005 was real. Although over-reporting, or under-reporting, to an unknown extent cannot be ruled out.

Annual Harvest from KB – During 1993-2005 (i.e., prior to introduction of quotas in Greenland in 2006) the Greenland annual catch of polar bears from the Kane Basin (KB) subpopulation was 11 (SD = 4.4 bears, range: 6-25/year; n,= 13 years) with no apparent trend.

However, it should be noted that during 1993-1999 the annual catches from KB were estimated at 10 each year during 1993-1999 (PBSG 2002, 2010) based on an interview survey in 1989 (Rosing-Asvid and Born 1990). The statistics on annual catches after 1999 were based on reportings in Piniarneq (2000-2005) and special reporting forms under the quota system (2006-2014).

After the introduction of quotas the Greenlanders' catch of polar bears from the KB management unit (2006-2014) has averaged 6/year (SD = 1.7, range: 2-8/year, n = 9).

Defense-of-Life-and-Property Kills (DLPs) – No defense kills were reported for the Kane Basin subpopulation during 2006-2014. During the same period 7 defense kills were reported for the Baffin Bay subpopulation (2007: 2, 2011: 1, 2012: 4). The kills comprised 1 young male, 3 adult females, 1 young with sex not stated, and 2 with sex and age not stated. The months during which these incidences occurred were: January (n = 1), February (1), July (2), October (2) and December (1). Hence, since the introduction of quotas in 2006 when the recording of defense kills began, there have been no apparent annual or seasonal trends in defense kills from the KB and BB subpopulations.

In Greenland DLPs in one year are not subtracted from next year quota (DFHA, *in litt.* 2016).

Uncertainties in Catch Reporting – Prior to 2006, when quotas were introduced, there was significant uncertainty in the accuracy of the reported polar bear harvest (catch) from BB and KB in Greenland. Reporting occurred via the Piniarneq system. It has been suggested that the polar bear catch reported through the Piniarneq system may be both an under-estimate (i.e., some kills not reported) and an over-estimate (Born 1998, 2002, 2006, Jessen 2002, Rosing-Asvid 2002) of the actual catch. To obtain a hunting license for the coming year, a hunter must

report his catch through the Piniarneq (in practice the catch of all species is reported in a booklet). Inevitably, a hunter may sometimes fail to report a catch of a particular species. Very often several hunters participate in a polar bear hunt (Rosing-Asvid 2002, Born et al. 2011). “Multiple reporting” (*i.e.*, one kill is reported by more than one hunter) leading to an over-estimate may thus occur when several hunters, who have participated in the same hunt and are proud that a bear was taken, each report the same kill via Piniarneq regardless of whether they shot the bear (Jessen 2002).

When evaluating the Piniarneq shortly after it was introduced, Kapel and Rosing-Asvid (1996) wrote that some hunters were not used to paperwork, and they may not see the point of keeping exact notes on the dates and numbers of animals taken. Whether this resulted in under-reporting, over-reporting, or just arbitrary reporting in order to have something to report when renewing the license, was not clear (*Ibid.*). In a study of the Greenland catch of ringed seals, Teilmann and Kapel (1998) identified examples of both under-reporting and over-reporting.

Generally, the numbers reported in Piniarneq are higher than those reported in the previous system of recording catches (*i.e.*, The Hunters Lists of Game, *cf.* Teilmann and Kapel 1998). This apparent difference may be caused by several factors: (1) previous information was incomplete and the estimates of unreported catches too low, (2) the Piniarneq-system overestimates the catch due to “multiple”-reporting, (3) a real increase in the catch, or (4) a combination of all these factors. An example of sources of error in *Piniarneq* is the report in 2004 of 24 and 10 polar bears reported for Sisimiut and Maniitsoq, respectively (Born and Sonne 2006). Some of these (10 and 5) were reported by hunters with a “part-time” hunting license and were suspected to be of muskoxen (O. Heinrich, DFHA, *in litt.* 2005).

Rosing-Asvid (2002) compared information from various sources (trade in hide, information from sampling of biological tissues, and interview survey and Piniarneq) about the catch of polar bears in Greenland for the period 1993–1998. He found cases of under-reporting and of over-reporting. In the Piniarneq, simple errors like ringed seals reported as walruses or polar bears occur. However, validation of the information is not a standard procedure and some over-reporting is found in most of the species where the annual catch is low (Rosing-Asvid 2002). Another type of error may occur because the hunter does not have to report to Piniarneq where the polar bear was shot. The kill is assigned to the municipality in which the hunter lives and is therefore in some cases misplaced if the hunter has taken the bear in another area (ibid.).

According to Rosing-Asvid (2002) validating the Piniarneq-data was (is) not a standard procedure and some over-reporting is found in most of the species where the annual harvest is low. The number of polar bear kills reported in Central Greenland might be overestimated with this new reporting system, however, the trend toward more polar bears caught in West Greenland since the mid 1980's is undoubtedly real. For the period 1970-87 the reported catches only averaged 2/year in Central West Greenland, which is less than reported through the media in the latest years or by forms that for some kills have been filled out at local offices since 1995. The interviewed hunters from Upernavik also reported a marked increase in the number of polar bears in the area since mid-1980s (Born et al. 2011).

As indicated the Piniarneq-data may in some cases represent under-reporting and in other cases “over-“ or “multiple”-reporting. As there has been no standard procedure in place in Greenland for validating to which extent (and/or in which direction) the Piniarneq-system is influenced by these potential errors when comes to polar bears one must be cautious when using and interpreting the data in Piniarneq, as pointed out by Born (2002).

Because of a good correlation between the number of ringed seal hides traded and the number of ringed seals reported via Piniarneq in the Upernavik municipality, Rosing-Asvid (2002) concluded that generally the Piniarneq system worked (works) well in this area (*Ibid.*). Hence, he indicated that the increase in reported catches of polar bears in the Upernavik area from the BB subpopulation was real. The average reported catch of polar bears in the Upernavik area during 1970-1987 (HLG) was 9 bears/year (range: 1-41/year) whereas during 1993-1998 it was 37 bears/year (range: 25-48/year) according to the Piniarneq (*Ibid.*).

Timing of Harvest in Baffin Bay – The Greenlanders catch of polar bears from the Baffin Bay subpopulation is almost exclusively concentrated in winter and spring (Jan-Jun). About 99% (n = 589) of the catches reported with month during 2006-2014 were taken during this period. The remaining *ca.* 1% (n = 5) is caught during Sep-Dec (Figure 8.10). This timing of the catches is in marked contrast to the situation in Nunavut where *ca.* 40% of the harvest is between Aug-Nov with a peak in October.

About 99% of the bears reported from BB were caught during Jan-Jun which is in contrast to pre-2006 when a relatively larger proportion was taken during fall and early winter. The fact that the catches have been more concentrated to the beginning of the year may be a result of the quota for BB (the quota year starts 1 January) being used up fast. According to the interviews conducted in NW Greenland in 2006 the availability of polar bears during spring in BB has increased (Born et al. 2011) which may explain that the quota is used up relatively soon and the catches therefore concentrated to late winter and spring.

Timing of Harvest in Kane Basin – Of 49 catches reported from the Kane Basin subpopulation during 2006-2014, 87.8% (n = 43) were taken during winter-spring (Jan-Jun) with

a clear peak in April. Twenty-six (53.1%) of these catches were taken during April. The remainder (12.2%) of the catches were reported from summer and fall (Aug-Oct); Figure 8.11.

The tendency with a peak in hunting activity in Kane Basin in March-April and a less pronounced peak in fall was also seen prior to 2006 (Born et al. 2011:185). However, after 2006 the seasonal distribution of catches from the KB subpopulation has shown a more conspicuous peak in April. It should however, be noted that the sample size from KB was small.

Means of Transportation – In some cases information on means of transportation used during the polar bear hunt is lacking from the reports on individual catches. However, during 2006-2014 there was information on means of transportation used during 445 individual polar bear hunts from areas between Kane Basin and Sisimiut (i.e., from the KB and BB subpopulations, respectively). Overall, 63.6% of the bears had been caught during a hunt involving dog sled. During 35.3% of the hunts a skiff was used and only in 1.1% (n = 5) of the cases the bear was caught from a <20 BRT/15 BT boat.

Means of Transportation in Baffin Bay – There was regional variation in means of transportation used during polar bear hunts in Baffin Bay. In Subareas 3 and 4 (encompassing the Melville Bay and the Upernavik areas) where ca. 85% of the Greenlanders' annual catch of bears from the BB subpopulation are taken, an average of 71.1% of the bears are caught during dog sled trips (the remainder are taken from skiffs) with a clear difference between the northern part and the southern part of the area (Table 8.3) reflecting differences in density of sea-ice cover and timing spring break-up. In Subareas 5 and 6 (Ummannaq, Disko Bay and areas south to Sisimiut) where only ca. 15% of the bears caught by Greenlanders' from the BB subpopulation are taken, 7.8% of the bears reported during 2006-2014 were killed during dog sled hunts, 88.2% were taken from skiff and 3.9% from a small boat (Table 8.3). These areas have open water (or

light pack ice conditions) during winter and/or early spring (e.g., Buch 2001). Dog sleds were used in 93.5% of the cases in the important polar bear hunting areas 1-3 and in 20.8% of the cases in areas 4-6 (i.e., the areas south of 74° N). This difference in means of transportation between the two overall areas was statistically significant ($\chi^2 = 246.283$, $P < 0.0001$, $df = 1$).

Only in one instance during 2006-2014 was it specified that a polar bear had been caught during a hunt involving the use of a snowmobile (using a snowmobile in connection with hunting polar bears is illegal in Greenland; Anon. 2005). This case involved the illegal kill of a male bear in the Sisimiut area in March 2011.

In none of the areas was there a statistically significant annual trend in fraction of bears taken from skiff or boat during 2006-2014 (linear regressions of weighted percentages of skiff+boat versus year; data not shown).

Means of Transportation in Kane Basin – Of 39 individual polar bear catches (2006-2014) from the Kane Basin subpopulation (i.e., Subarea 1 and 2; only 2 catches reported from Subarea 2), 76.9% were taken from dog sled, 15.4% from skiffs 7.7% from a boat. The catches from skiffs and boats were taken during May ($n = 1$) and June-October ($n = 8$); Table 8.3.

The data on means of transportation showed a marked north-south gradient in the use of sleds vs. skiffs related both to differences in hunting traditions and availability of dense fast or pack ice. In the southern areas where sea-ice conditions to a large extent are influenced by the inflow of relatively warm current from the south (e.g., Buch 2001) the majority of polar bears are taken from skiffs (and in a few cases from small-type fishing vessels) whereas in the areas north of ca. 74° N, where there is fast ice and dense pack ice, the majority of polar bears are caught by dog sled and this means of transportation is still an important element in the traditional way of living and hunting. The 2006-interview survey indicated that there has been an increase in the

use of skiffs for hunting polar bears especially in the Upernavik area since the early 1990s (Born et al. 2011). This development was ascribed to the fact that the sea-ice conditions have become more unstable (for driving a dog sled) and there is an earlier spring ice break-up (ibid.). The same tendency was indicated in the 2006-2014 records of hunting methods (present study) but was not statistically significant.

Hence, in Greenland, the majority of polar bears that are taken from the KB and BB subpopulations are still taken during dedicated polar bear hunts where the dog sled (in the majority of cases) is used for transportation and tracking of polar bears. This maintains an old and traditional way of hunting polar bears.

Since 1968 snowmobiles have been used increasingly in the polar bear hunt. In Arctic Canada, polar bears are nowadays hunted almost exclusively with snowmobiles (except for guided sport hunts, which are required to use a dog team); Slavik (2013 and references therein).

Sex Ratio in Baffin Bay – The sex and age composition of the Greenlanders' catch of polar bears from the BB and KB subpopulations is presented in Table 8.4 and 8.5, respectively.

In Baffin Bay the relative proportions of independent polar bear females and males in three different sets of data (1982-2005; Table 8.4) were not statistically different ($\chi^2 = 1.096$, $P = 0.578$, $df = 2$). Overall, independent females constituted *ca.* 28.3% of the total annual catch (note: including cubs which constituted *ca.* 20% of the catch) prior to 2006.

Independent female polar bears constituted *ca.* 32.5% of the total annual catch during 2006-2014. Overall, the ratio of independent F:M in the catch was *ca.* 1:2 both during 1982-2005 and 2006-2014.

In a sample of 55 bears (2012 and 2013) for which gender was determined genetically and tooth-derived ages were known independent females constituted 45.5% (Table 8.4). The

relative proportions of independent females and males in this sample differed significantly at the 5% level from those in the larger sample (2006-2014) which was based on reports from the hunters ($\chi^2 = 3.972$, $P = 0.046$, $df = 1$).

Sex Ratio in Kane Basin – In Kane Basin the relative proportions of independent polar bear females and males in three different sets of data (1982-2005; Table 8.5) did not differ ($\chi^2 = 4.609$, $P = 0.099$, $df = 2$). Overall, independent females constituted *ca.* 25.4% of the total annual catch (note: including cubs which made up *ca.* 8% of the total catch) prior to the introduction of quotas in 2006. During 2006-2014 independent female polar bears constituted *ca.* 44.9% of the total annual catch (note: after 2005 it has been illegal to catch dependent cubs irrespective of their age). The relative proportions of independent females and males during 1982-2005 and 2006-2014, respectively, differed significantly ($\chi^2 = 5.130$, $P = 0.024$, $df = 1$); independent females constituting a higher proportion of the catch in KB after 2006. However, it must be kept in mind that the basis for data before 2005 is heterogeneous and sample size after 2006 is relatively small.

Age Structure – In a hunter collected sample of a total of 55 polar bears caught in BB Greenland during 2012 and 2013 individuals less than 10 years of age constituted 85.5% and ≥ 10 years olds were 14.5% of the catch. The oldest bears were two 17 year old females (Figure 8.12). In comparison, polar bears less than 10 years of age (i.e., 2-9 years of age) constituted 63.9% and 10+ olds 36.1%, respectively, in the sample of live captured bears (2009-2013) from BB. Hence, the proportion of polar bears ≥ 10 years of age was significantly higher than in the sample from the harvest ($\chi^2 = 8.026$, $P = 0.005$, $df = 1$). The oldest live captured polar bear was a 23 year old male (Figure 8.13).

Adult (i.e., sexually mature) females were defined as greater ≥ 4 years old and adult males as greater than or equal to 5 years old (Molnár et al. 2008). The relative proportion of sexually immature and sexually mature polar bears in the 2012-2013-sample did not differ significantly from a sample of 55 polar bears aged 2+ sampled in NW Greenland during 1988-1996 (Rosing-Asvid 2002: figure 26; $\chi^2 = 1.094$, $P = 0.296$, $df = 1$). COYs and 1 year olds were excluded from this comparison because only the sample from 1988-1996 contained these age groups. The oldest bear in the 1988-1996- sample was a 16 year old female.

Polar bear cubs usually follow the mother for two years and are weaned at 2.5 years of age although some are weaned already during their second spring (range: 1.3-2.3 years; Lønø 1970, Lentfer et al. 1980, Ramsay and Stirling 1986, Amstrup and Durner 1995). Hence for modeling purposes bears 3 years of age and older are considered “independent” or “adult (Taylor et al. 1987). Prior to introduction of quotas (2006) it was prohibited to catch COYs and yearlings in W and NW Greenland whereas after 2005 it became prohibited to take dependent cubs (i.e., cubs demonstrably belonging to a family group) irrespective of their age. Hence, we assume that the 2-year-olds reported in the Greenland catch in recent years (Figure 8.12) had left their mothers. Rosing-Asvid (2002: figure 26) presented an age composition of a sample (1988-1996) that comprises COYs, yearlings and 2-year-olds. Hence, due to differences in hunting regulations during the two periods (i.e., prior and after quotas were introduced) only 2-year-olds and older bears were considered in the comparison of age-structure in the catch. The recent sample from the catch and the sample from 1988-96 did not comprise any bears older than 17 years of age. In the sample from NW Greenland (1988-1996) Rosing-Asvid (2002) found the oldest male to be 14 years and the oldest female 16 years of age. A comparison of the two

admittedly not large data sets indicates no change in age composition of the Greenlanders catch from BB over time.

The finding of relatively few polar bears ≥ 10 years of age and no individuals older than 17 years of age in the harvest is perhaps surprising and in contrast to the situation in the sample of live captured bears (2009-2013) from NW Greenland.

The sample from the harvest in Nunavut from Baffin Bay contained several bears older than 17 year of age (*cf.* Figure 8.7).

In contrast, in Central East Greenland where in a sample of 238 polar bears (1983-1996; Rosing-Asvid 2002: figure 8) *ca.* 9% were older than 16 years. The oldest female was 26 years and the oldest male 30 years (*ibid.*).

Combined Canadian and Greenlandic Harvest from BB and KB

Baffin Bay – Data from Nunavut and Greenland were combined to examine overall harvest levels in BB for the period 1993-2014 (Table 8.6). Mean annual harvest was 163 bears (SD = 37.9, range 120-268, n = 22 years). For the period 1993-2005, prior to the introduction of a quota in Greenland, the mean annual total harvest was 165 bears (SD = 48.9, range = 120-268, n = 13 years). For period 2006-2014, after the introduction of a quota in Greenland, the mean annual harvest was 161 bears (SD = 13.6, range = 138-176, n = 9 years).

Total harvest in Baffin Bay peaked between 2002 and 2005 (Figure 8.14). This peak was the result of two events: (1) an increase in allowable harvest in Canada in 2004 (from 64 to 105 bears per year) as part of a new management agreement for the subpopulation and (2) a large increase in reported harvest in Greenland.

Total harvest declined after 2005 through to 2014 as a result of two management initiatives: (1) the introduction of a quota in Greenland in 2006 and a subsequent decrease in annual quotas and (2) implementation of a phased reduction (10 bears per year for 4 years) in Total Allowable Harvest in Canada.

The sex ratio of the reported harvest has remained around 2:1 (male: females) with an annual mean of 0.35 females amongst independent bears.

Kane Basin – Total estimated harvest in Kane Basin for the period 1993-2014 (Table 8.2) was 204 bears with a mean of 9.3 bears per year (SD = 4.63, n = 22, range = 3-26). Prior to the introduction of a quota in Greenland, mean estimated harvest in KB was 11.6 bears/year (1993-2005: SD = 4.61, n = 13, range 6-26). Following the introduction of a quota in Greenland, harvest decreased to a mean of 5.9 bears/year (2006-2014: SD = 1.62, n = 9, range = 3-9).

Thus since the introduction of a quota, the estimated harvest has halved in size and there has been a significant reduction in annual variation. However, the uncertainty of the number of polar bears taken from KB prior to 2006 must be mentioned. The polar bear hunters living in the Qaanaaq area in NW Greenland harvest polar bears both from the Kane Basin and from the Baffin Bay subpopulation (e.g., Born et al. 2011). The annual reports of total catch in the Qaanaaq area during 1993-1999 did not specify whether a bear had been taken from KB or from BB, respectively. Consequently it was assumed that 10 of the total number of polar bears reported from the Qaanaaq annually had been extracted from KB during this period. The remainder was assumed to have been taken from the BB subpopulation. However, the estimate of 10/year for KB represented the upper range of an estimate of 5-10/year which was based on an interview survey conducted in 1989 (Rosing-Asvid and Born 1990, 1995, PBSG 2010). During 2000-2005 the estimates of the fraction of bears reported from the Qaanaaq area that had been

taken in KB were based on location of settlement reporting whereas during 2006-2014 the numbers are based on report of actual site of the kill.

Adding to the uncertainty of the exact number of polar bears that are taken by Greenland from the KB subpopulations is the simple fact that it cannot be determined with any certainty whether a bear taken in the central parts of the Qaanaaq area (i.e., close to the border at 77° N between the KB and BB management zone) belong the KB or the BB subpopulation. This uncertainty will of course have greater implications for the relatively small catch from KB than for BB.

Overall for the period 1993-2014, the estimated sex ratio of bears harvested in Kane Basin was 33% females. However, the sex ratio of the harvest since the introduction of a quota in Greenland has been approximately 44% female (based on pooled data for the period 2006-2014).

Accuracy of Harvest Reporting as Assessed from Genetic Studies of Sex and Individual Identity

Reporting of the Sex of Harvested Bears – During the recent genetic mark-recapture studies in BB and KB (2011-14) bears harvested in BB, KB and surrounding subpopulations were genotyped to establish genetic sex and individual identity in-order to detect recoveries of genetically marked (biopsied) individuals (Table 8.7). Rates of tissue sampling, reporting of sex and genotyping of bears harvested in BB and KB were less than 100%. Overall, 270 (75%) of the 359 bears that were reported as harvested in BB during the mark-recapture sampling period were tissue sampled and genotyped. For Kane Basin, 4 (40%) of the 10 harvested bears were genotyped. Sampling of harvested bears was lower in Greenland than Nunavut.

Using these genotyped bears we compared the gender of harvested bears as reported in harvest records in Canada and Greenland to the gender as determined by genetics. As part of the genetic analyses, conflicts between reported and genetic sex were investigated via additional genotyping to confirm genetic sex (see description of genetic methods in Chapter 5). Thus confidence in the genetic sex data is high. The results indicate there was significant inaccuracy in gender reporting with a bias towards under reporting of females. Pooling data for Canada and Greenland, 16% of genetic females in the harvest were reported as males (Table 8.8). In contrast, 4% of genetic males were reported as females. The bias was greatest in the Greenland harvest, where 39% of genetic females were reported as males and 12% of genetic males were reported as female (Table 8.9 and Figure 8.15a). In Nunavut, 5% of females were reported as males. Two percent of males were reported as females (Table 8.10 and Figure 8.15b). Overall, the sex composition of the genotyped harvest as reported in official harvest records was 37% females. The genetic composition of this harvest was 42% females. For the Greenland harvest, the sex composition of the reported harvest was 39% females. The genetic composition was 54% females. For the Nunavut harvest, the sex composition of the reported harvest was 36% females. The genetic composition was 37% females. Considering only independent bears (subadults & adults), for the Greenland harvest, the sex composition of the reported harvest was 40% females. The genetic composition was 54% females. For the Nunavut harvest, the sex composition of the reported harvest was 36% females. The genetic composition was 36% females.

Assuming these data are representative of the overall harvest, harvest in Greenland appears to be non-selective for sex. Harvest in Nunavut is approximately 2:1 males to females in accordance with target sex ratio of the flexible quota management system. In Nunavut

verification of the sex of harvested bears is a regulatory requirement. Hunters are required to submit the baculum from harvested males. Where proof of sex is not provided sex is verified by genotyping. Our finding that gender reporting in the Canadian (Nunavut) data is accurate was thus expected.

Inaccuracies in reporting the sex of harvested bears is a management issue for these subpopulations. Determination of sustainable harvest levels in part depends on the sex ratio of the harvest (Taylor et al. 2008b; Regehr et al. 2015). The less selective the harvest, the lower the sustainable harvest. Incorrect reporting of gender for harvested bears may also be an issue for some of the past demographic analyses for BB and KB (e.g., PVA's) that have been used to establish quotas and subpopulation status; albeit to an unknown extent at present.

Reporting of Marked Bears in the Harvest – Using data for bears that were physically marked (tagged and tattooed) in either Baffin Bay (1990-1997) or Davis Strait (2005-2007), and subsequently also genotyped, we examined the accuracy of reporting of ear tagged and lip tattooed bears in the harvest relative to the detection of these marked bears via genotyping. Due to small samples sizes we pooled data on recoveries of physically marked bears in Nunavut and Greenland.

In the harvest data for Baffin Bay for the period 2011-2014, 9 recoveries of physically marked bears were detected by genotyping, 4 of which were not reported as tagged or tattooed in official harvest records. Expanding this dataset to the Davis Strait harvest records, resulted in detection of 42 physically marked bears, 12 of which were unreported as marked in harvest records. One of the unreported marks was a Greenland harvest record.

Approximately 29% of recoveries were not reported as being marked. These findings suggest that a significant portion of physically marked bears that are recovered in the harvest are

undetected via the harvest reporting systems currently in place. Although the data are limited, detection probability appears to be a function of the age of a mark; the interval between application of tags and tattoos and subsequent harvest (Fig 8.16). Older marks are less likely to be reported suggesting that loss of tags or fading of tattoos may affect the ability of hunters to detect when they have harvested a marked bear. Indeed, in cases where marked bears were reported in the harvest, 62% were reported based on the presence of lip tattoos only. Examining data for bears physically marked in the Baffin Bay during 1990-97 that were recaptured during the recent Davis Strait inventory (2005-07) we found that 24 of 24 marked individuals whose recapture was detected by genotyping were also detected as marked by field workers. In most cases, notes on the field data sheets indicated that the recaptured bears had lost both ear-tags and were identified by means of their lip tattoo only. Mean capture interval (i.e., age of mark) was 11 years (range 8 to 15) amongst this sample of 24. This suggests that loss of ear-tags is the primary problem affecting detection of marked bears in the harvest. These findings also suggest that the problem of detecting marked individuals may be limited to the harvest data only.

The implications of this finding require careful consideration with respect to past and future mark-recapture studies. The assumption that all marked bears recovered in the harvest are reported, an assumption made in previous polar bear mark-recapture studies in Baffin Bay (Taylor et al. 2005, Peacock et al. 2012), Kane Basin (Taylor et al. 2008a), Davis Strait (Peacock et al. 2013) and elsewhere, appears to be invalid. Under-reporting of marked bears in the harvest may have introduced bias resulting in underestimation of natural survival rates in these studies. However, the extent (significance) of the bias is unknown at present. We recommend further investigation of this issue.

In our admittedly limited sample of harvest recoveries, detection of marks ≤ 5 years old

was > 90%. Moving forward, this finding highlights the importance of maintaining a sample of recently (within 5 years) marked bears in the subpopulation when relying on detection of physical marks to estimate survival rates. Alternatively, we recommend genetic monitoring of the harvest in future studies where detection of ‘old’ marks is anticipated to play an important role. Further research into materials and designs for increasing the endurance of ear-tags may also be warranted.

8.4. Literature Cited

- Aars, J., N. J. Lunn, and A. E. Derocher, editors. 2006. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Amstrup, S. C., and G. M. Durner. 1995. Survival rates of radio-collared female polar bears and their dependent young. *Canadian Journal of Zoology* 73:1312–1322.
- Anon. 1954-87. Sammendrag af Grønlands Fangstlister. Ministeriet for Grønland [Summaries of “Hunters’ Lists of Game”]. Summarized during 1954-1984 by the Danish Ministry of Greenland, Copenhagen, and during 1985-1987 by the Ministry for Fisheries, Hunting and Agriculture, Nuuk. [In Danish].
- Anon. 1973. *Agreement on the Conservation of Polar Bears*. Available at: <http://pbsg.npolar.no/en/agreements/agreement1973.html>.
- Anon. 1976. Bekendtgørelse om ændring af bekendtgørelse om fredning af isbjørne i Grønland [Executive order of preservation of polar bears in Greenland]. *Nalunaerutit-Grønlandsk Lovsamling, Serie A, Nr. I, MJ.G.j. nr. 1701-D8: 109*. [In Danish].
- Anon. 2005. Hjemmestyrets bekendtgørelse nr. 21 af 22. september 2005 om beskyttelse og fangst af isbjørne [Greenland Home Rule Executive order No. 21 of 22 September 2005 on protection and harvest of polar bears], 5 pp. [In Danish].
- Born, E. W. 1998. Research on polar bears in Greenland, primo 1993 to primo 1997. Pages 23–44 in A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. *Polar Bears:*

- Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Born, E. W. 2002. Research on polar bears in Greenland 1997-2001. Pages 67–74 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Born, E. W., and C. Sonne 2006. Research on polar bears in Greenland, 2001-2005. Pages 33–55 *in* J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusulanum Press, University of Copenhagen, Copenhagen, Denmark.
- Buch, E. 2001. The ocean environment. Pages 111–122 *in* E. W. Born, and J. Böcher, editors. The Ecology of Greenland. Atuakkiorfik/Education Publishers, Nuuk, Greenland.
- Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. *Ursus* 10:449–453.
- Derocher, A. E., G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. 1998. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Dowsley, M., and G. W. Wenzel. 2008. “The time of the most polar bears’’: A co-management conflict in Nunavut. *Arctic* 61:177–189.
- Dyck, M. G. 2006. Characteristics of polar bears killed in defense of life and property in Nunavut, Canada, 1970– 2000. *Ursus* 17:52–62.
- Jessen, A. 2002. A note on the management of polar bears in Greenland. Page 65 *in* N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

- Kapel, F. O., and A. Rosing-Asvid. 1996. Seal hunting statistics for Greenland 1993 and 1994, according to a new system of collecting information, compared to the previous Lists-of-Game. NAFO Scientific Council Studies 26:71–86.
- Lentfer, J. W., R. J. Hensel, J. R. Gilbert, and F. E. Sorensen. 1980. Population characteristics of Alaskan polar bears. International Conference on Bear Research and Management 4:109–115.
- Lønø, O. 1970. The polar bear (*Ursus maritimus*, Phipps) in the Svalbard area. Norsk Polarinstitutt Skrifter 149:1–115.
- Molnár, P. K., A. E. Derocher, M. A. Lewis, and M. K. Taylor. 2008. Modelling the mating system of polar bears: a mechanistic approach to the Allee effect. Proceedings Royal Society B 275:217–226.
- Obbard, M. E., G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. 2010. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2002. Status of the polar bear. Pages 21–35 in N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. Arctic 65:391–400.
- Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463–476.
- Ramsay, M. A., and I. Stirling. 1986. On the mating system of polar bears. Canadian Journal of Zoology 64:2142–2151.

- Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. *Journal of Wildlife Management* 71:2673–2683.
- Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.
- Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. *Population Ecology* 54:3–18.
- Rosing-Asvid, A. 2002. The polar bear hunt in Greenland. Greenland Institute of Natural Resources Technical Report 45, 25 pp.
- Rosing-Asvid, A., and E. W. Born. 1990. Fangst af isbjørn (*Ursus maritimus*) i Avanersuaq og Upernavik kommuner: En interviewundersøgelse [Catch of polar bears in the municipalities of Avanersuaq and Upernavik: an interview survey]. Grønlands Hjemmestyres Miljø- og Naturforvaltning Teknisk Rapport Nr. 23, 63 pp. [In Danish with English summary].
- Rosing-Asvid, A., and E. W. Born. 1995. The catch of polar bears in northwestern Greenland. Page 188 in Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Slavik, D. V. W 2013. Knowing *nanuut*: Bankslanders' knowledge and indicators of polar bear population health. MSc thesis, Department of Resource Economics and Environmental Sociology, University of Alberta, Edmonton, Alberta, Canada.
- Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. *Global Change Biology* 18:2694–2706.
- Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. *Arctic* 59:261–275.
- Taylor, M. K., S. Akeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. *Canadian Journal of Zoology* 79:690–709.

- Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. *Arctic* 58:203–214.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. *Wildlife Biology* 14:52–60.
- Teilmann, J., and Kapel, F.O. 1998. Exploitation of ringed seals (*Phoca hispida*) in Greenland. NAMMCO Scientific Publications 1:130–151.
- Towns, L., A. E. Derocher, I. Stirling, N. J. Lunn, and D. Hedman. 2009. Spatial and temporal patterns of problem polar bears in Churchill, Manitoba. *Polar Biology* 32:1529–1537.
- Vibe, C. 1968. Thule-eskimoernes isbjørnejagter [Polar bear hunting of the Thule-Eskimos]. *Tidsskriftet Grønland* 6:175-180. [In Danish].
- Vibe, C. 1985. Miscellaneous information relating to polar bears in Greenland. Pages 44–57 in *Polar Bears: Proceedings of the 8th Working Meeting of the IUCN/SCC Polar Bear Specialist Group*. IUCN, Gland, Switzerland.
- Weber, D. S., T. Mandler, M. Dyck, P. J. Van Coeverden de Groot, D. S. Lee, and D. A. Clark. 2015. Unexpected and undesired conservation outcomes of wildlife trade bans - An emerging problem for stakeholders? *Global Ecology and Conservation* 3:389–400.
- Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. <http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en>.
- York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. *Ecology and Evolution* doi:10.1002/ece3.2030.

Table 8.1. Summary of Canadian polar bear harvest from the Baffin Bay subpopulation for the period 1992/93 to 2013/14.

Harvest Season ¹	Total Harvest	Total Allowable Harvest ²	Proportion Female (Total Harvest) ³	Proportion of Harvest Made up of Independent Bears ⁴	Proportion Female (Independent Bears Only) ⁵
1992/93	62	n/a	0.36	0.94	0.35
1993/94	60	n/a	0.37	0.88	0.34
1994/95	60	64	0.33	0.92	0.35
1995/96	55	64	0.33	0.95	0.35
1996/97	60	64	0.41	0.88	0.42
1997/98	69	64	0.38	0.96	0.38
1998/99	49	64	0.35	0.98	0.36
1999/00	58	64	0.41	0.95	0.40
2000/01	61	64	0.28	0.98	0.28
2001/02	64	64	0.30	1.00	0.30
2002/03	62	64	0.26	0.97	0.22
2003/04	69	64	0.28	0.99	0.25
2004/05	101	105	0.37	0.98	0.38
2005/06	94	105	0.32	0.98	0.28
2006/07	89	105	0.36	0.99	0.38
2007/08	101	105	0.28	0.97	0.26
2008/09	103	105	0.39	0.98	0.39
2009/10	86	105	0.41	1.00	0.41
2010/11	94	95	0.33	0.98	0.34
2011/12	90	85	0.40	0.96	0.37
2012/13	74	75	0.47	0.92	0.45
2013/14	72	65	0.31	0.97	0.29
Mean	74.23		0.35	0.96	0.34

¹ The hunting season in Canada runs from July 1st to June 30th

² Total Allowable Harvest (TAH) is the regulated limit for all human-caused mortalities.

Formerly referred to as the quota

³ Excludes bears of unknown sex (which constituted a mean of *ca.* 1% of harvest annually for the period 1989/90 to 2013/14)

⁴ Excludes bears of unknown dependency (which constituted a mean of *ca.* 4% of harvest annually for the period 1989/90 to 2013/14)

⁵ Excludes bears of unknown sex and dependency

Table 8.2. Canadian and Greenlandic polar bear harvest from the Kane Basin subpopulation for the period 1992/93 to 2013/14.

Harvest Season ¹	Total Harvest	Total Allowable Harvest ²	Harvest (Canada)	Harvest (Greenland) ³	Proportion Female (Independent bears only)
1992/93	10	-	0	10	0.25
1993/94	11	-	1	10	0.23
1994/95	10	-	0	10	0.25
1995/96	10	-	0	10	0.25
1996/97	11	-	1	10	0.23
1997/98	12	-	2	10	0.30
1998/99	11	-	1	10	0.23
1999/00	6	-	0	6	0.25
2000/01	11	-	1	10	0.32
2001/02	12	-	0	12	0.25
2002/03	12	-	0	12	0.25
2003/04	9	-	0	9	0.25
2004/05	26	-	1	25	0.28
2005/06	9	-	1	8	0.40
2006/07	6	15	0	6	0.46
2007/08	7	13	0	7	0.46
2008/09	5	11	0	5	0.46
2009/10	3	11	1	2	0.30
2010/11	6	11	0	6	0.46
2011/12	6	11	0	6	0.46
2012/13	6	11	0	6	0.46
2013/14	5	11	0	5	0.46
Mean	9.3	11.8	0.4	8.9	0.33

¹ Greenland harvest data for a given calendar year were included in the harvest season ending in that calendar year

² No quota in Greenland prior to 2006 thus total quota only presented for period 2006-2014

³ Annual harvest in Greenland (1993-1999) was estimated from an interview survey conducted in 1989 (Rosing-Asvid and Born 1990). Information on annual catch was based on information in Piniarneq during 2000-2005 and thereafter on the special reporting under the quota system (see Appendix D)

Table 8.3. Means of transportation used during 445 polar bears catches in West and Northwest Greenland from Subareas 1 and 2 (i.e., the Kane Basin subpopulation) and Subareas 3-6 (the Baffin Bay subpopulation) during 2006-2014.

Subarea	Dog sled	Skiff	Boat	Total	% Dog sled	% skiff and boat
1+2 ¹	30	6	3	39	76.9	23.1
3	215	8	0	223	96.4	3.6
4	34	98	0	132	25.8	74.2
5	4	25	0	29	13.8	86.2
6	0	20	2	22	0.0	100.0
Total	283	157	5	445	63.6	36.4

¹ Only 2 catches reported from Subarea 2

Table 8.4. Sex and age composition of the Greenland catch of polar bears from the BB subpopulation (1982-2014) based on various sources.

Period	% Females ¹	% Males ¹	Independent F:M ratio	% Cubs	F (n)	Cubs (n)	M (n)	Total (n)	Source	Source
1982-1989	29.9	57.7	1:1.9	12.4	41	17	79	137	Interviews 1989-90	Rosing-Asvid and Born 1990, Rosing-Asvid 2000
1988-1996	33.3	46.0	1:1.4	20.6	21	13	29	63	Samples	Rosing-Asvid ² 2002: table 3
1991-2005	26.8	50.6	1:1.9	22.6	95	80	179	354	Interviews 2006	Born unpublished, and Born <i>et al.</i> 2011
1982-2005	28.3	51.8	1:1.9	19.9	157	110	287	554		
2006-2014	32.2	67.8	1:2.1	0.0	192	n.a. ³	404	596	Special Reporting Forms	2006-13:DFHA ⁴ <i>in litt.</i> (2014), 2014:DFHA <i>in litt.</i> (2015)
2012-2013	45.5	54.5	1:1.2	0.0	25	n.a.	30	55	Samples	Samples collected by hunters, GINR unpublished
2006-2014	32.5	67.5	1:2.1	0.0	193	n.a.	401	594		

¹ Percentage of total annual catch. Adult and subadult females and adult and subadult males = individuals 3 years old and older (*cf.* Taylor et al. 1987)

² A comparison of figure 26 and table 3 in Rosing-Asvid (2002) shows that the percentages of independent bears given in his table 3 are based on 2+ years old bears

³ Since 2006 it has been prohibited to kill dependent cubs irrespective of their age

⁴ DFHA = Department of Fisheries, Hunting and Agriculture (Nuuk)

Table 8.5. Sex and age composition of the Greenland catch of polar bears from the KB subpopulation (1982-2014) based on various sources.

Period	% Females ¹	% Males ¹	Independent F:M ratio	% Cubs	F (n)	Cubs (n)	M (n)	Total (n)	Source	Source
1982-1989	29.2	63.1	1:2.2	7.7	19	5	41	65	Interviews 1989-90	Rosing-Asvid and Born 1990, Rosing-Asvid 2000
1988-1996	36.7	56.7	1:1.5	6.7	11	2	17	30	Samples	Rosing-Asvid 2002: table 3
1991-2005	17.9	73.1	1:4.1	9.0	14	7	57	78	Interviews 2006	Born unpublished, and Born et al. 2011
1982-2005	25.4	66.5	1:2.6	8.1	44	14	115	173		
2006-2014	44.9	55.1	1:1.2	0.0	22	n.a. ²	27	49	Special Reporting Forms	2006-13:DFHA ³ <i>in litt.</i> (2014), 2014:DFHA <i>in litt.</i> (2015)

¹ Independent females as percentage of total annual catch

² Since 2006 it has been prohibited to kill dependent cubs irrespective of their age

³ DFHA = Department of Fisheries, Hunting and Agriculture (Nuuk)

Table 8.6. The combined Canadian and Greenlandic polar bear annual harvest from the Baffin Bay subpopulation for the period 1992/93 to 2013/14. Annual average proportion of independent female polar bears is shown.

Harvest Season ¹	Total Harvest	Total Allowable Harvest ²	Harvest (Canada)	Harvest (Greenland) ³	Proportion Female (Independent bears only)
1992/93	134		62	72	0.35
1993/94	120		60	60	0.35
1994/95	124		60	64	0.35
1995/96	122		55	67	0.35
1996/97	139		60	79	0.38
1997/98	165		69	96	0.36
1998/99	146		49	97	0.36
1999/00	126		58	68	0.37
2000/01	158		61	97	0.33
2001/02	182		64	118	0.33
2002/03	268		62	206	0.32
2003/04	225		69	156	0.32
2004/05	236		101	135	0.36
2005/06	173		94	79	0.30
2006/07	165	178	89	76	0.35
2007/08	176	176	101	75	0.29
2008/09	174	173	103	71	0.36
2009/10	150	171	86	64	0.37
2010/11	165	160	94	71	0.33
2011/12	165	152	90	75	0.35
2012/13	137	142	74	63	0.39
2013/14	146	132	72	74	0.31
Mean	163	161	74.23	89.00	0.35

¹ Greenland harvest data for a given calendar year were included in the harvest season ending in that calendar year

² No quota in Greenland prior to 2006 thus total quota only presented for period 2006-2014

³ Harvest in Greenland is estimated from reported harvest in west Greenland and the estimated portion of this harvest that occurs in Baffin Bay

Table 8.7. Genotyping of bears harvested in Canada and Greenland, 2011-2014. Data presented as the percentage of individuals in the reported harvest that were sampled and genotyped. Total number of individuals reported as harvested is presented in parenthesis. Data are organized by Nunavut hunting seasons which run from July1 to June 30.

Subpopulation	2011/12		2012/13		2013/14	
	Canada	Greenland	Canada	Greenland	Canada	Greenland
BB	75 (85)	64 (69)	90 (78)	42 (60)	100 (67)	n/a ¹
DS	74 (38)	-	92 (60)	-	-	-
LS	75 (92)	-	91 (92)	-	-	-
KB	0 (0)	50 (6)	0 (0)	25 (4)	-	-
FB	86 (107)	-	91 (109)	-	-	-
NW	0 (0)	-	33 (3)	-	-	-

¹ Greenland harvest during this season occurred after mark-recapture sampling ceased and is therefore not reported.

Table 8.8. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Data from Canada (Nunavut) and Greenland harvest.

Reported Sex	Genetic Sex	
	Male	Female
Male	156	19
Female	6	97

Table 8.9. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Greenland harvest only.

Reported Sex	Genetic Sex	
	Male	Female
Male	29	15
Female	4	24

Table 8.10. Comparison of the reported versus genetic sex of bears harvested in Baffin Bay and Kane Basin, 2011-2014. Data from Canada (Nunavut) harvest only.

Reported Sex	Genetic Sex	
	Male	Female
Male	127	4
Female	2	73

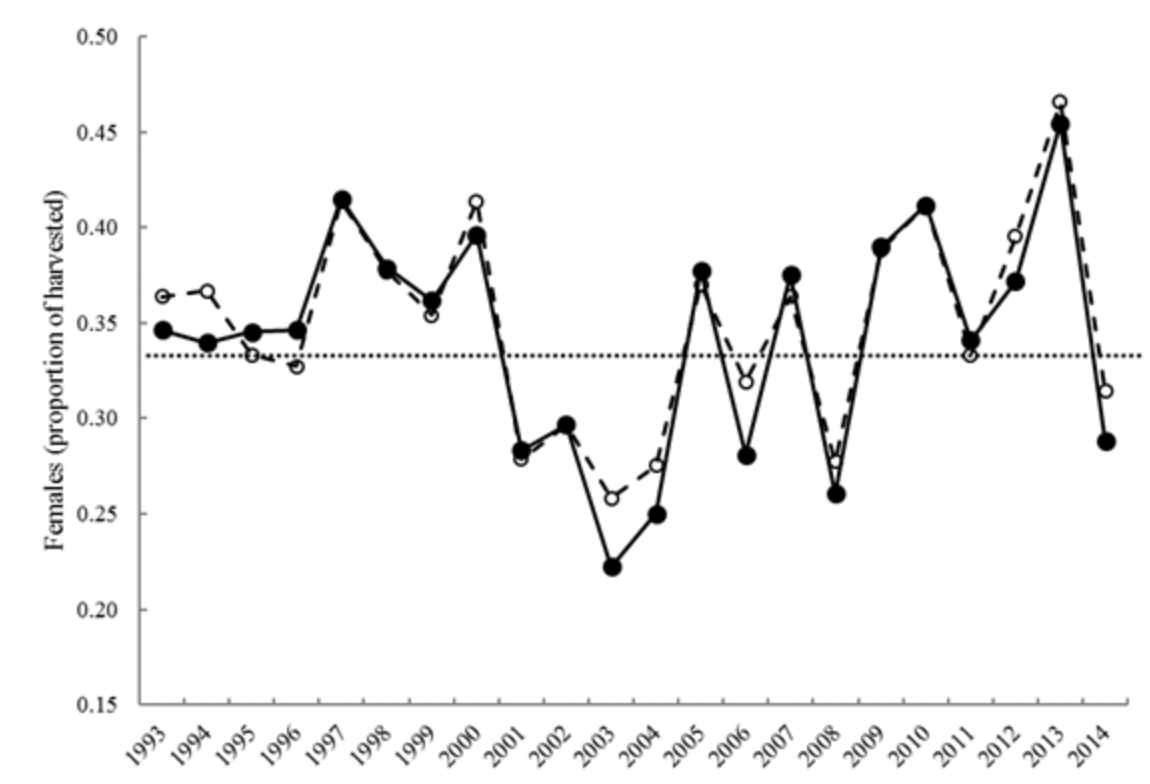


Figure 8.1. Harvest of female polar bear as a proportion of total Canadian harvest from Baffin Bay. All females (dashed line) and independent females only (solid line). Dotted line indicates target sex ratio (0.33).

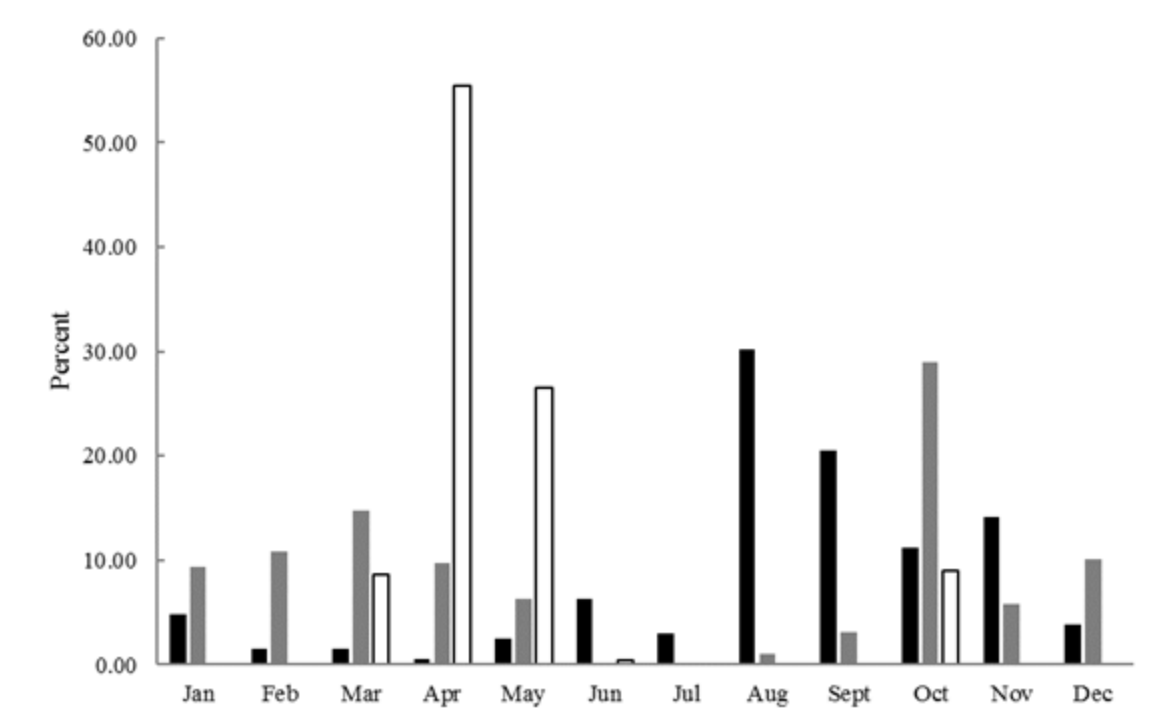


Figure 8.2. Monthly distribution of polar bear harvesting in Baffin Bay by type of harvest (1992/93-2013/14). Regular hunts (grey), defense-of-life-and-property kills (black) and sport hunts (white). Bars represent the percentage of hunting of a given type that occurred each month.

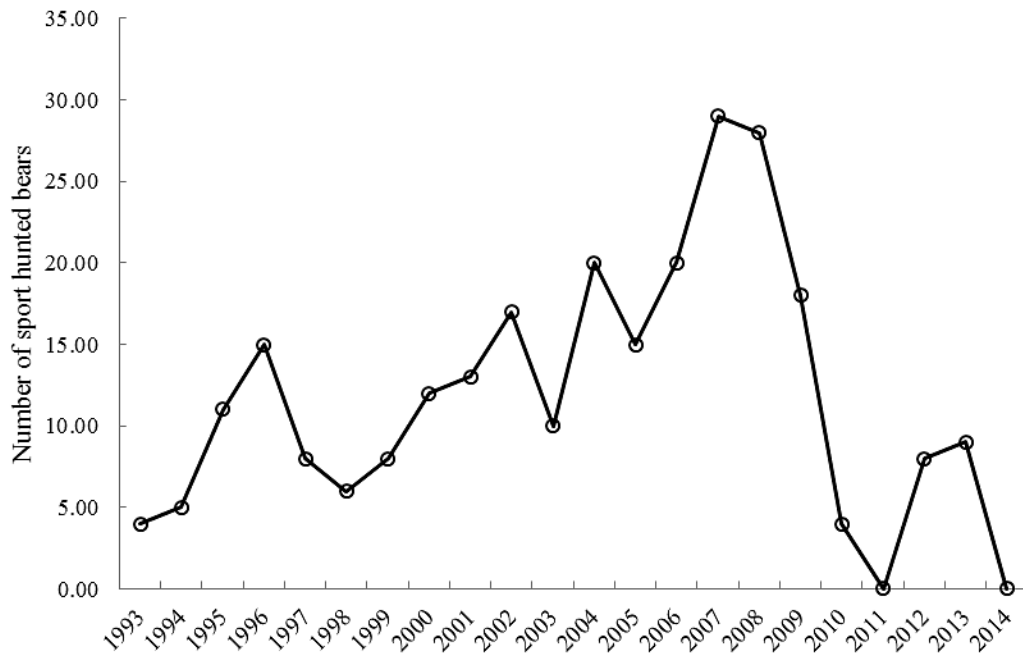


Figure 8.3. Numbers of polar bear taken by sport hunters in Baffin Bay, Canada.

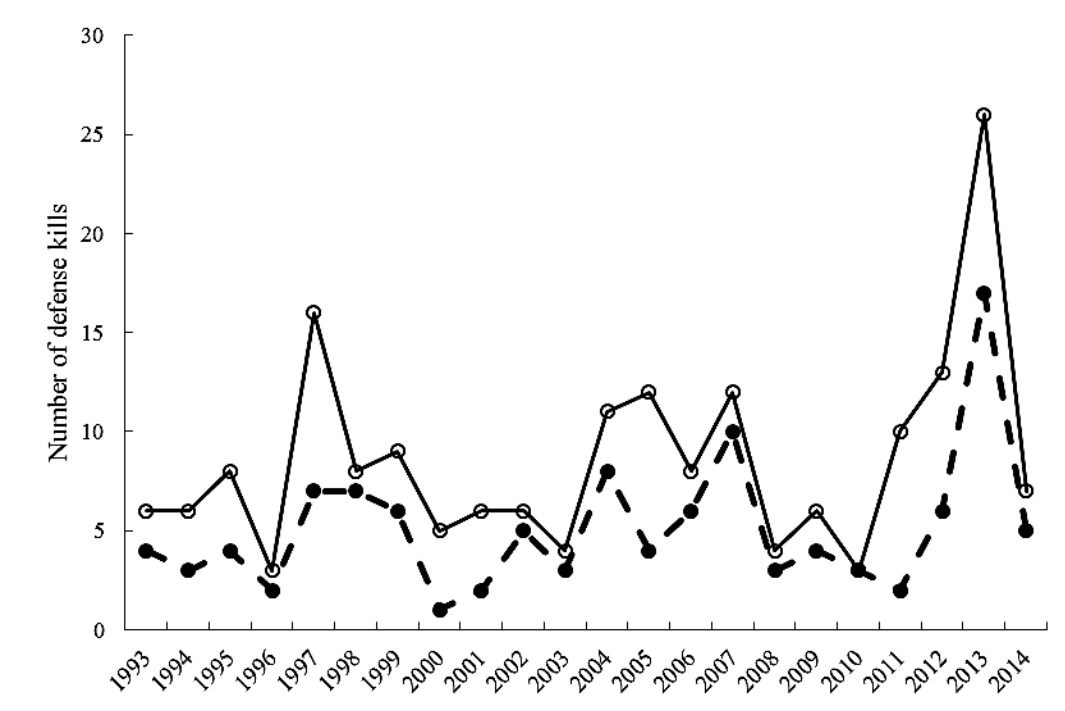


Figure 8.4. Number of polar bears harvested in defense-of-life-and-property (DLP) in Baffin Bay, Canada. Total DLPs per year (solid line) and DLPs of independent bears during August–November each year (dashed line).

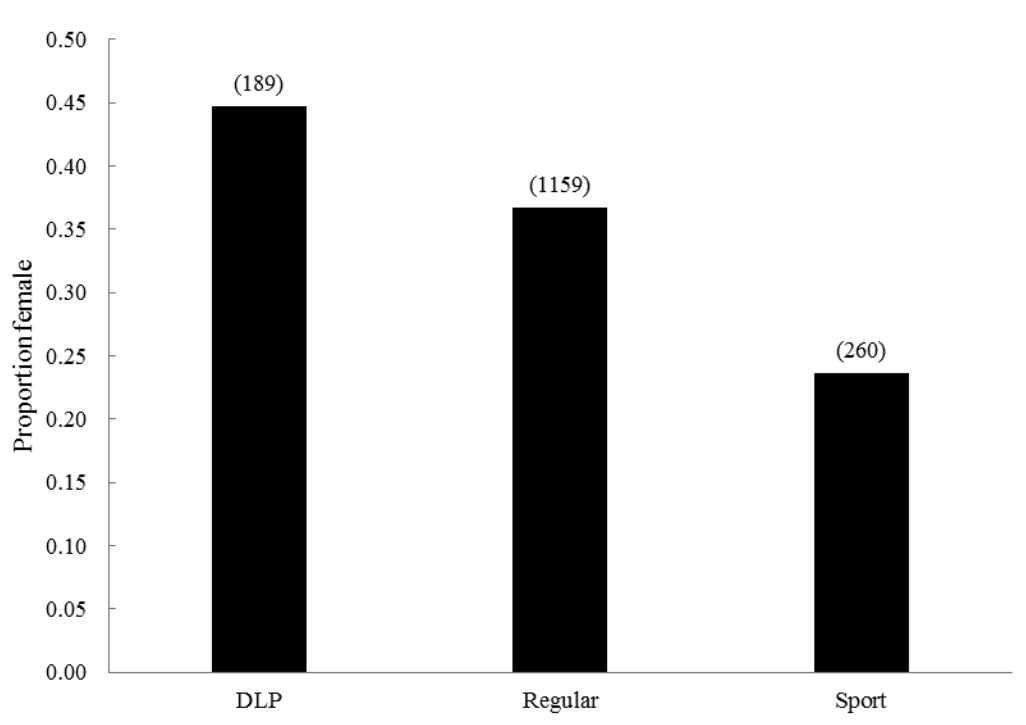


Figure 8.5. Sex composition of the three main types of polar bear harvesting in Baffin Bay, Canada, for the period 1992/93 to 2013/14. Bars represent the proportion of harvest that was female. Data are for independent bears only. Sample sizes in parentheses.

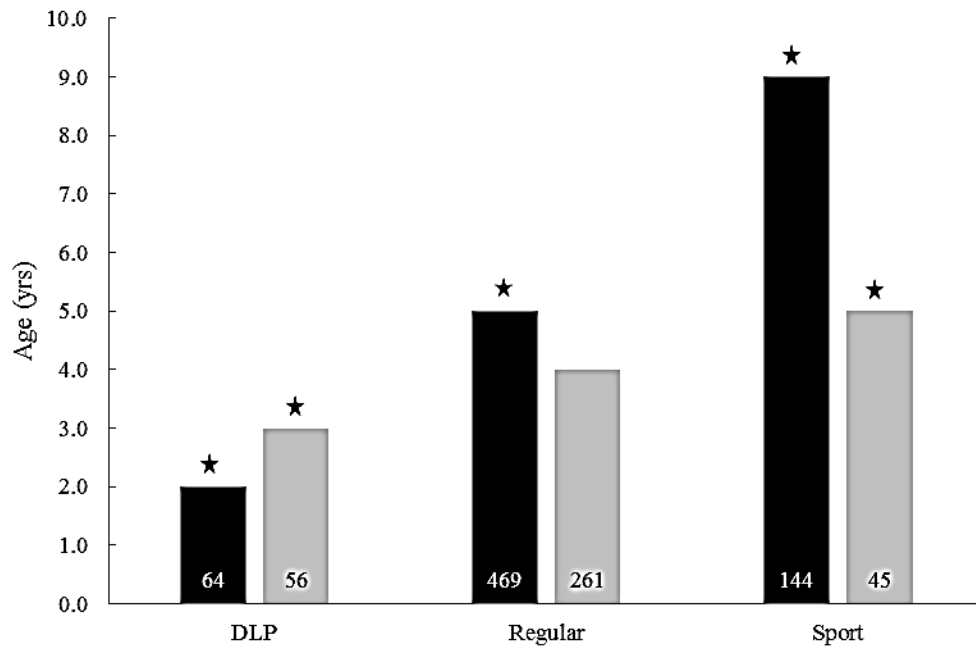
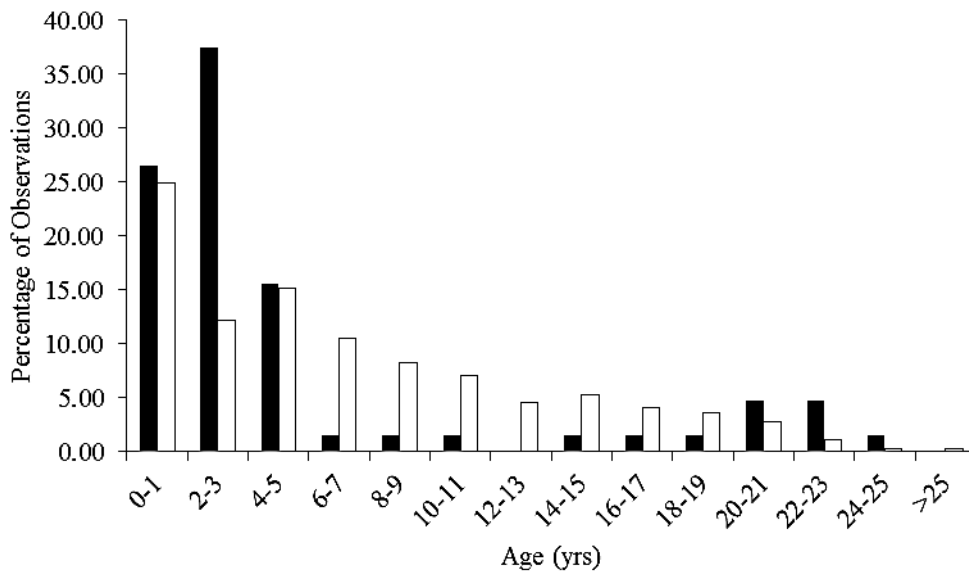


Figure 8.6. Median age of male (black) and female (grey) bears harvested for defense-of-life-and-property (DLP), regular (subsistence) and sport hunting purposes in Baffin Bay, Canada, 1993-2010. Within sexes significant differences denoted by * (Based on Mann-Whitney *U* test with Bonferroni correction for multiple comparisons, two-tailed, alpha = 0.05). Sample sizes within bars.

(a)



(b)

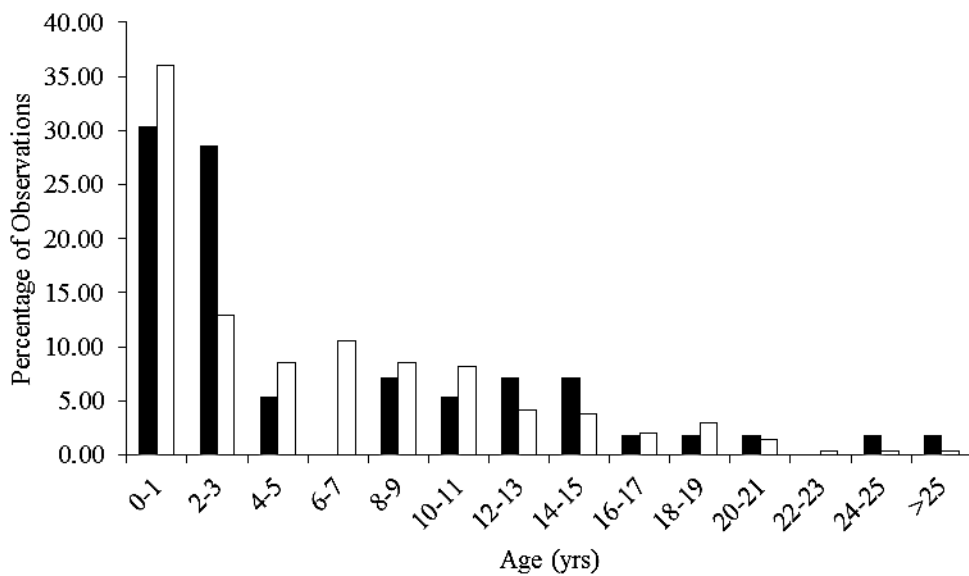


Figure 8.7. The age distribution of (a) 64 male and (b) 56 female bears harvested in defense-of-life-and-property (black) in Baffin Bay (Canada), 1993-2010, relative to the age distribution of 778 bears captured during mark-recapture sampling (white), 1993-1997 (GN unpublished data).

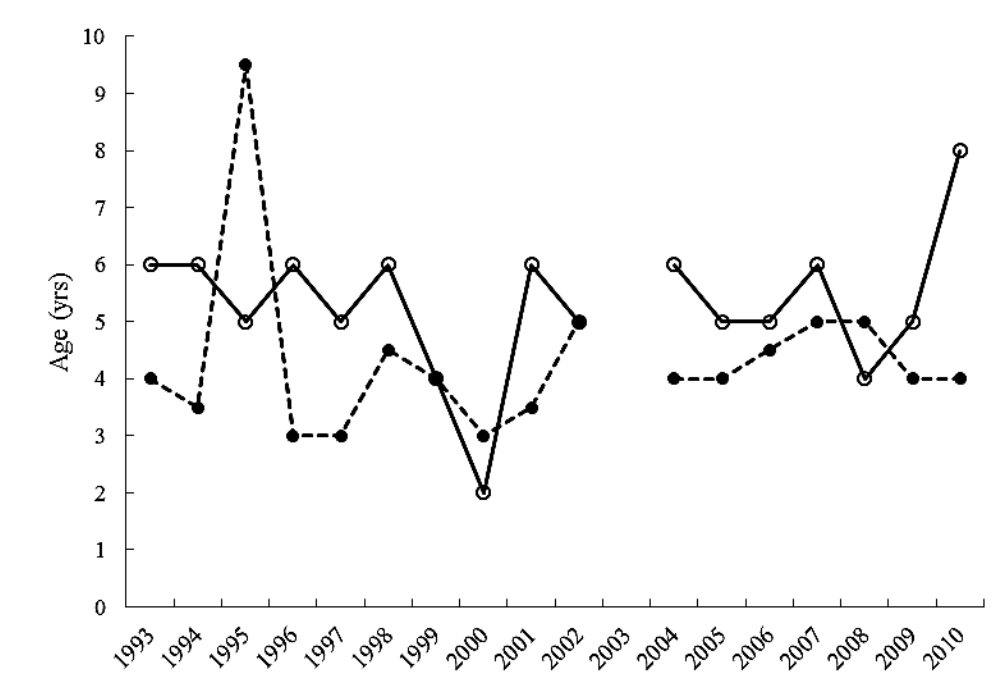


Figure 8.8. Median age of harvested male (solid line) and female (dashed line) polar bears in Baffin Bay, Canada. Regular hunts only.

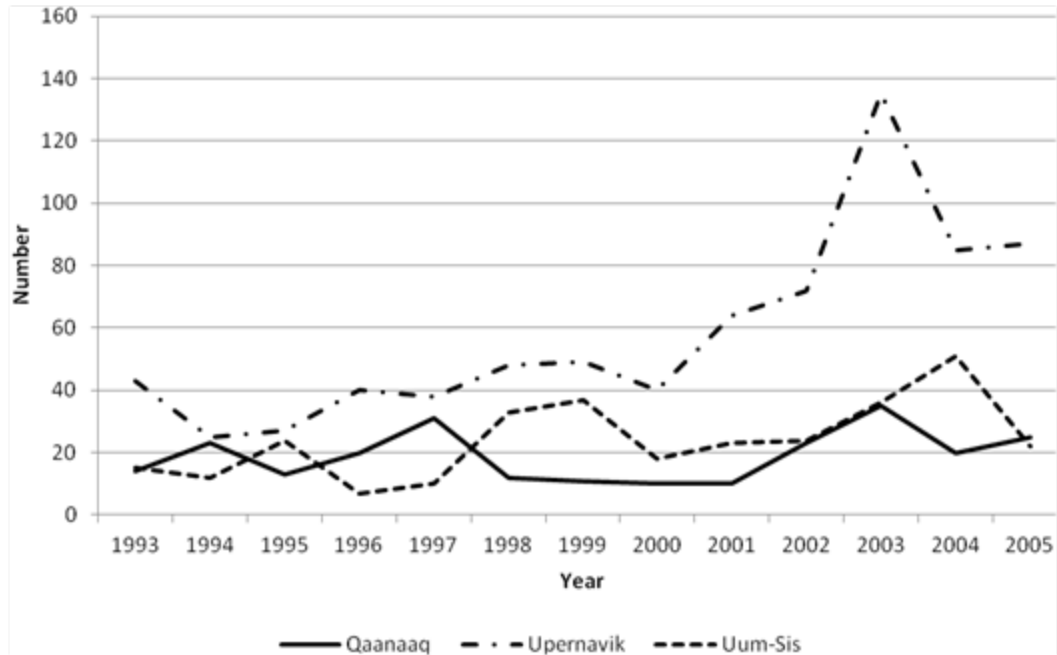


Figure 8.9. The Greenland catch of polar bears from the Baffin Bay subpopulation reported in Piniarneq (1993-2005). The catch is shown for three different regions: (1) The Qaanaaq region representing polar bears taken between $74^{\circ} 35' \text{ N}$ and $76^{\circ} 20' \text{ N}$ (i.e., the Melville Bay region *sensu lato*), (2) the Upernavik area between *ca.* $71^{\circ} 30' \text{ N}$ and $74^{\circ} 35' \text{ N}$, and (3) the areas between Uummannaq and Sisimiut between $66^{\circ} 55' \text{ N}$ and $71^{\circ} 30' \text{ N}$.

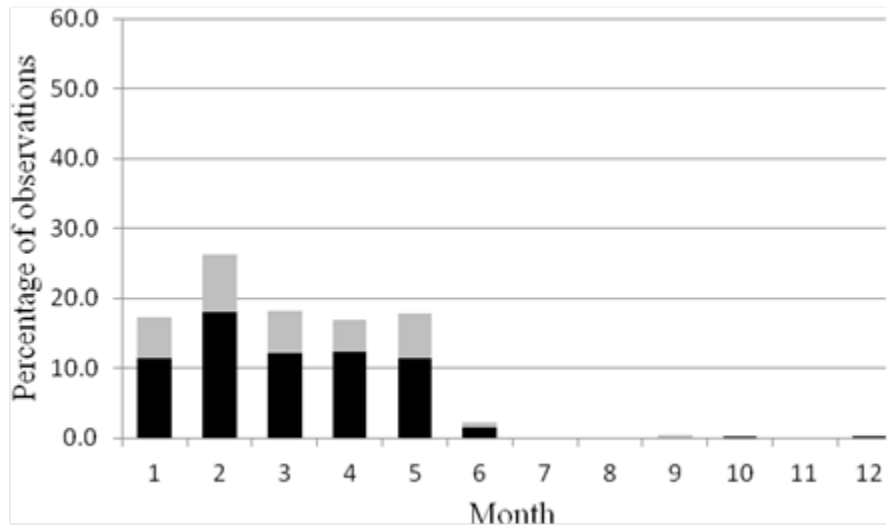


Figure 8.10. Seasonal distribution of catches of polar bears ($n = 594$; 587 legal and 7 illegal catches) taken in NW and W Greenland from the Baffin Bay subpopulation based on reports from the licensed hunters (2006-2014). Black = Males. Grey = Females. Source: DFHA (in litt 2014 and 2015).

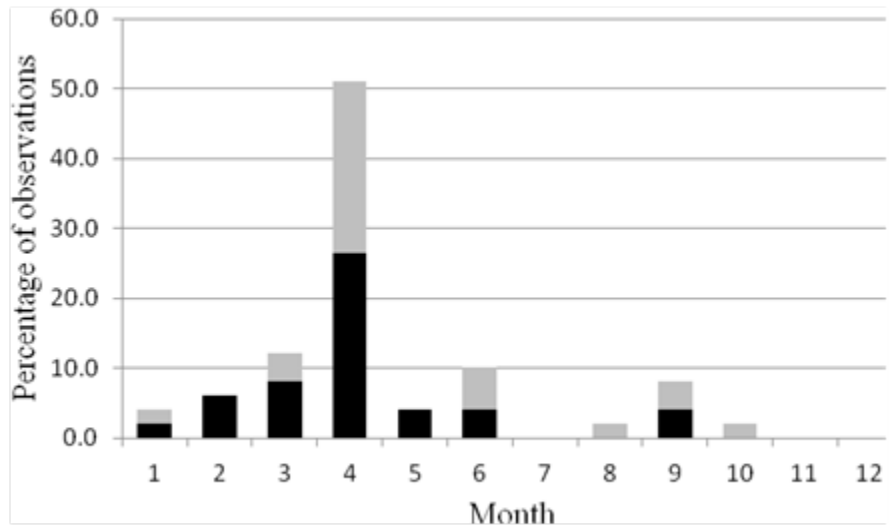


Figure 8.11. Seasonal distribution of catches of polar bears ($n = 49$; 48 legal and 1 illegal catches) taken in NW Greenland from the Kane Basin subpopulation based on reports from the licensed hunters (2006-2014). Black = Males. Grey = Females. Source: DFHA (*in litt.* 2014 and 2015).

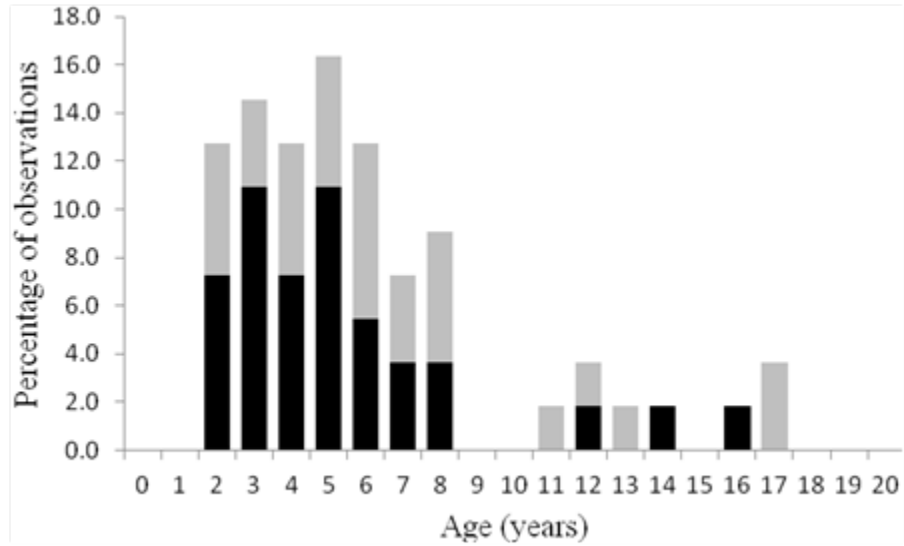


Figure 8.12. Age distribution of a total of 55 polar bears (Grey = Females: $n = 25$; Black = Males: $n = 30$) that were taken from the BB subpopulation by Greenland subsistence hunters in 2012 ($n = 33$) and 2013 ($n = 22$). Sex was determined genetically.

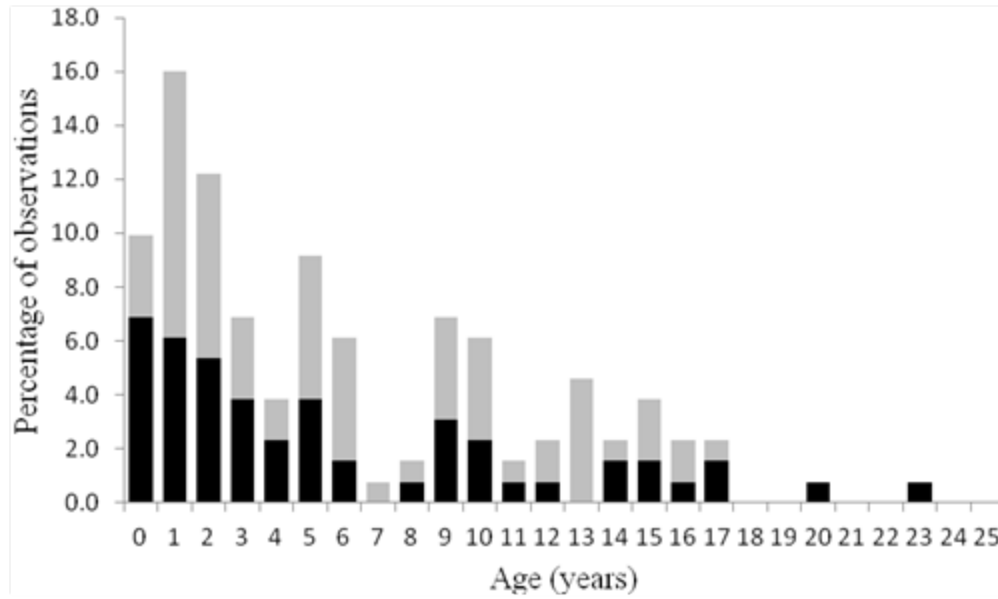


Figure 8.13. Age distribution of 131 polar bears (Grey = Females: $n=73$; Black = Males: $n=58$) that were live captured in NW Greenland north of *ca.* 70° N from the BB subpopulation during spring 2009-2013 in connection with the present study (GINR unpublished data).

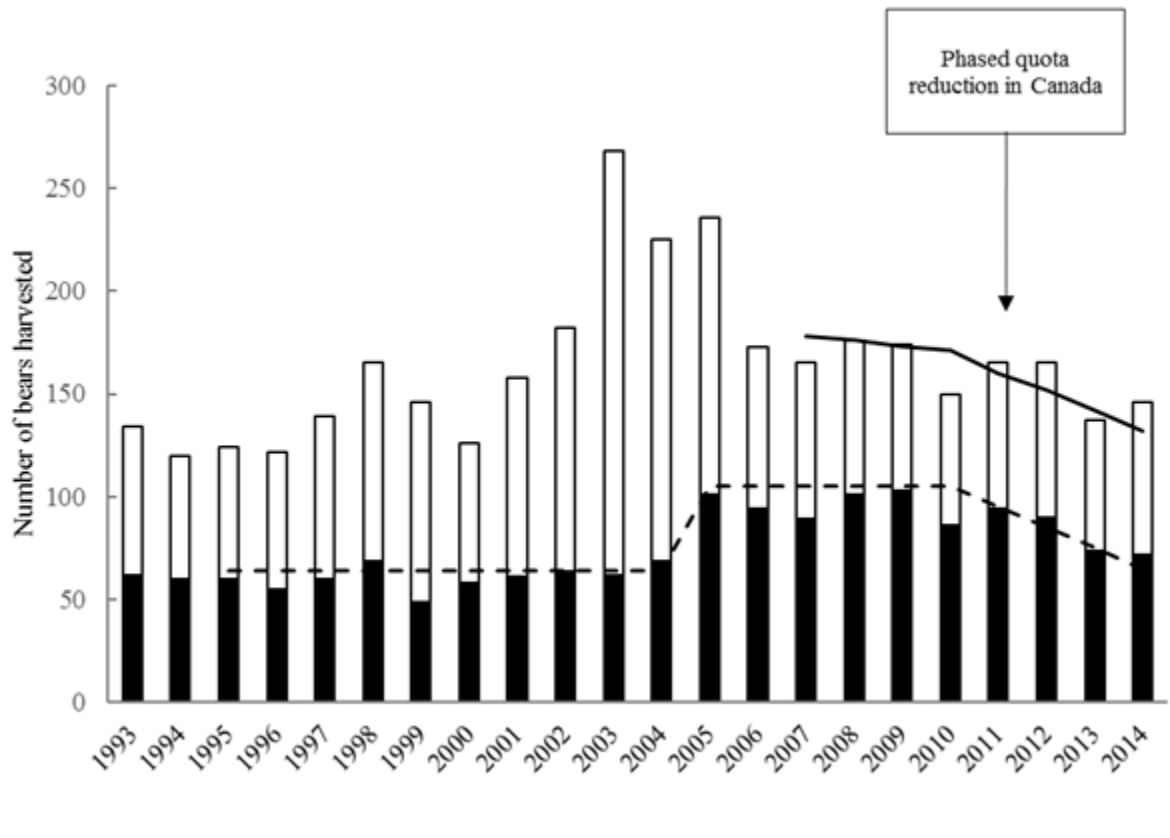
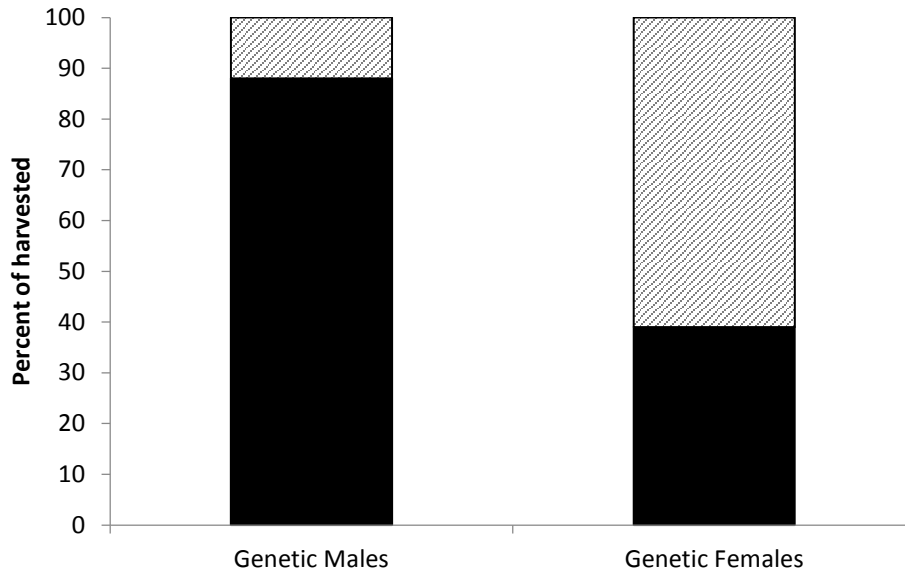


Figure 8.14. Combined Canadian (black) and Greenlandic (white) harvest of polar bears from the Baffin Bay subpopulation. Total allowable harvest in Canada (dashed line) and Canada-Greenland combined total permitted harvest (solid line) levels are also shown.

(a)



(b)

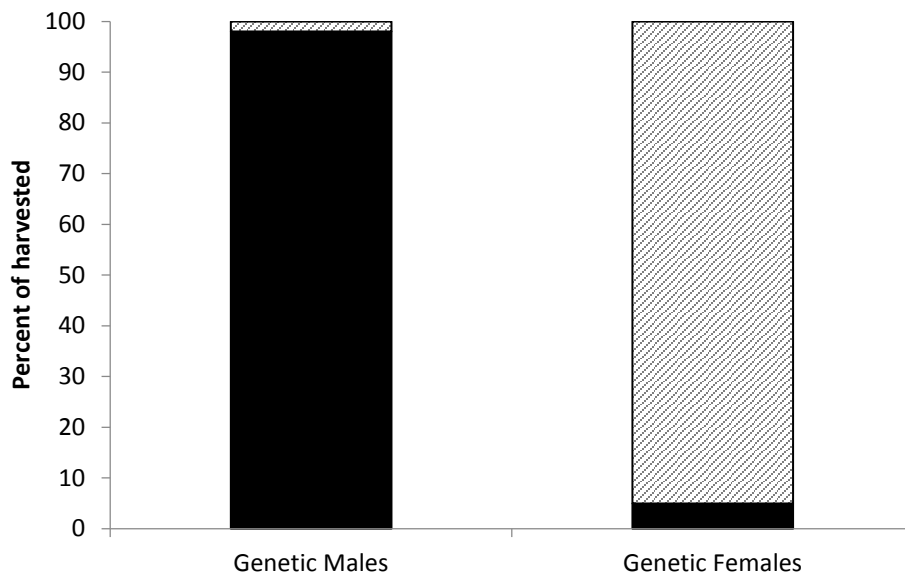


Figure 8.15. Accuracy of gender reporting for polar bears harvested in Baffin Bay and Kane Basin (2011-14) in (a) Greenland and (b) Nunavut. Bears reported as males and females are indicated in black and hatched, respectively

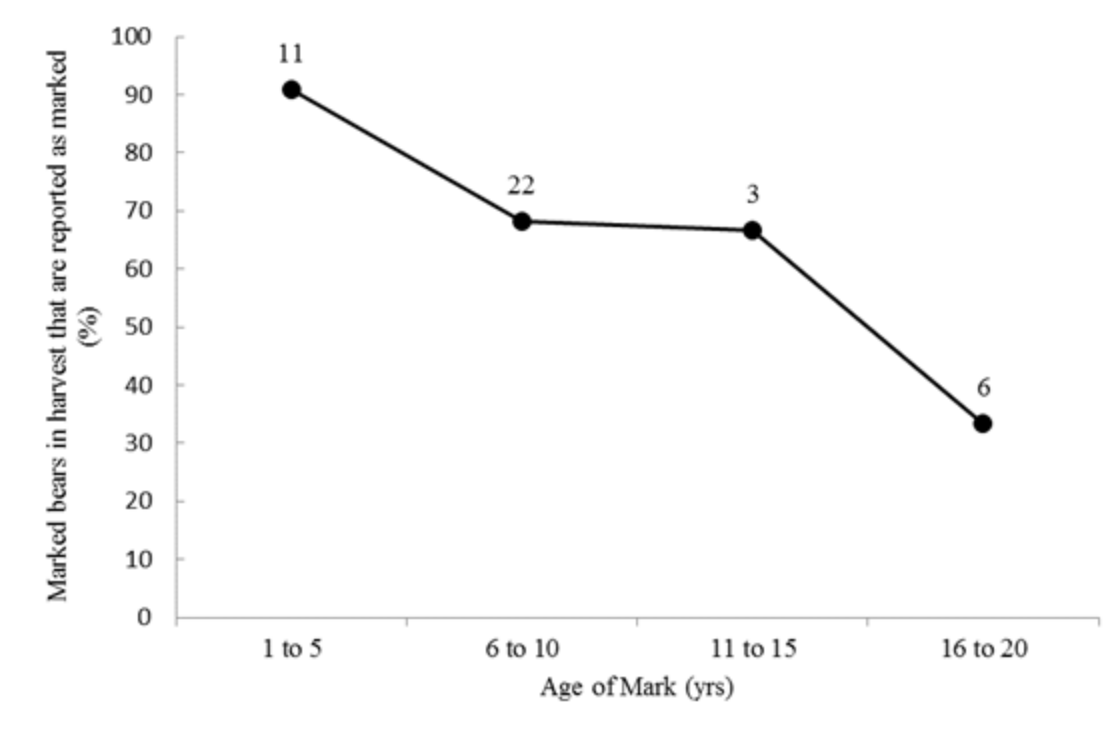


Figure 8.16. Accuracy of reporting of physically marked (tag and tattooed) bears in the harvest in Canada and Greenland. Data are for bears physically marked in Baffin Bay (1990-97) and Davis Strait (2005-07) that were recovered in the harvest 2011-14. Sample sizes above points.

CHAPTER 9

HABITAT AND HABITAT USE IN KANE BASIN

KEY FINDINGS

- The annual cycle of sea-ice habitat in KB has shifted from a largely year-round ice platform (>30% coverage in summer) to a cycle that resembles the seasonal ice ecoregion with complete melt-out in summer (<5% coverage).
 - Sea ice is retreating earlier in Kane Basin spring by 7 days/decade, and advancing later in fall by 5-6 days/decade and length of summer (number of days from retreat to advance) is increasing by 12 days/decade. The mean sea-ice concentration during June-October is decreasing by 5-6 percent/decade.
 - The KB subpopulation has responded to changing sea-ice conditions with broad movement and habitat use patterns that are more similar to those of bears in seasonal sea-ice ecoregions (e.g., expanded seasonal home ranges, see Chapter 2 and use of lower sea-ice concentrations in summer and fall).
 - Four-day movement rates in KB are lower than those in BB and have a less pronounced seasonal cycle. There are no significant differences in movement rates between the 1990s and 2000s except in October where rates were higher.
 - Land use in KB during summer remains intermittent because some sea ice remains in fjords and coastal areas. No on-land arrival and departure dates could be determined from satellite telemetry.
 - Three maternity dens were found in KB in the 2000s. All were located on Ellesmere Island. There was no significant difference in maternity denning duration, entry dates, or exit dates between the 1990s and 2000s.
 - Overall, the movement (Chapter 2) and habitat results combined with reproductive metrics (Chapter 11) and body condition (Chapter 12), indicate that KB bears are experiencing more seasonal sea-ice ecoregion-like conditions, which since the 1990s may have increased overall biological productivity of the area.
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9.1. Introduction

The Kane Basin (KB) subpopulation of polar bears occurs between the Canadian Arctic Archipelago and Northwest Greenland, referred to by some as the Arctic Archipelago ecoregion, historically characterized by year-round sea-ice habitat in islands in the Canadian High Arctic

and in the Kane Basin region in Northwest Greenland (Hamilton et al. 2014). Ice in this ecoregion is largely multi-year, except in Kane Basin where much of the ice is annual though partially present year-round. In contrast to the Canadian Arctic Archipelago, the Kane Basin region is also characterized and influenced by influx of multi-year ice flowing to the area from the Arctic Ocean. In these areas, sea ice remains along coastal areas in summer providing temporally-stable hunting opportunities for polar bears during summer. This is in contrast to the seasonal ecoregion (Baffin Bay), where sea ice disappears almost completely in summer and bears are forced onshore (Amstrup et al. 2008, this study).

Earlier studies comparing movements and habitat selection of polar bears in the archipelago and seasonal ecoregions documented differences in the influence of sea-ice regimes on movements and habitat use. Ferguson et al. (2000) showed that bears in the archipelago regions (including KB) are strongly influenced by the temporal and spatial distribution of land fast ice around islands providing a consistent habitat across the annual cycle, though possibly lower density year-round prey base. This was in contrast to seasonal ecoregion bears that are strongly influenced by the availability of seasonal sea ice, resulting in a more productive but temporally limited the duration of feeding. Bears in these two ecoregions also demonstrate contrasting movement patterns: bears in the archipelago region (e.g., KB) tended to have smaller home ranges and greater irregularity in movement patterns because they utilize small coastal areas around the complex land masses (or in case of the Kane Basin region fast ice in fjords or fields of pack ice in Nares Strait-Kane Basin), whereas bears in the seasonal sea-ice region (e.g., BB) had large home range sizes (Ferguson et al. 1998) and more regular movement patterns associated with the seasonal growth and recession of sea ice.

Recent work examining Global Climate Model projections of sea-ice habitat for polar bears in the Canadian Arctic Archipelago suggest that conditions will shift towards a seasonal-sea-ice ecoregion before the end of the 21st century (Hamilton et al. 2014). This shift is expected to initially increase productivity in the archipelago system, as thicker ice is replaced by thinner annual ice increasing production blooms and prey platforms for ice seals. Such a change would be expected to be associated with changes in movement patterns and habitat use similar to that exhibited by bears in the seasonal ecoregion. To date, no studies have quantified such behavioral changes. In this chapter we document changes in sea-ice habitat of KB over the satellite record, and quantify change in habitat use using satellite telemetry data collected from collared adult females in the 1990s and 2000s.

9.2. Methods

We refer to “Kane Basin” as the region within the boundaries of the Kane Basin (KB) polar bear management unit (PBSG 2010; Figure 1.1.) that encompasses the northern part of Smith Sound, Nares Strait and Kane Basin and the southern part of Kennedy Channel and adjacent fjords on eastern Ellesmere Island and in NW Greenland.

KB polar bears were captured and tagged between mid-April and early May 2012-2013 (Figure 9.1). Field operations were based out of the Alexandra Fjord station on Ellesmere Island. A total of 34 bears were furnished with satellite-transmitters in KB. Twenty were adult females who received a satellite collar (Table 9.1) and 14 were adult males or subadults (given satellite radio ear tags). Data from adult females were combined with a historical data set from 12 adult females collared between 1992 and 1994 on the west side of KB in the fjords and fast ice (Taylor et al. 2001). Only bears captured within the KB subpopulation boundaries (PBSG 2010) were

included in the analysis. The eastern side of the Kane Basin region was surveyed during 1994-1997 but no female polar bears for collar deployment were found in these areas (Taylor et al. 2001). Hence, radio collars were only deployed on the west side of KB in the 1990s (ibid.). Given the different distribution of collar deployments between decades we tested for differences in area use and mean latitude using only bears captured in West KB in the 2000s and found no differences (see Chapter 2). Therefore, all bears from the 1990s were compared to all bears in the 2000s in habitat models. Bears in all decades largely utilized KB, however in both decades bears moved from KB into BB, LS and in the 2000s out of KB into the Arctic Basin.

Sea ice

Data sources and methods for sea-ice analysis in KB are the same as those described for BB (see Chapter 4 and Stern and Laidre 2016). The entire KB region, as defined by PBSG (Figure 1.1), was used for the sea-ice habitat analyses. The area was roughly divided into two distinct parts. The northern part, or Kane Basin proper, is bounded on the south by Smith Sound, consisting of almost entirely of shallow (< 300 m) water. The southern part contains the northern part of the North Water Polynya, and is bounded on the south by roughly 77° N latitude where it adjoins northern Baffin Bay. The southern part of Kane Basin consists mostly of deep (> 300 m) water on the Ellesmere Island side and shallow water on the Greenland side.

Habitat Use Analyses

Methods for KB movement rates, habitat covariates, and multivariate RSF modeling are the same as those described in Chapter 4 for BB. In the multivariate terrestrial RSF we did not include the variable pertaining to the distance to the smoothed Baffin Island coastline. In the

multivariate sea-ice RSF we excluded the covariates representing the distance to the 15 and 50% sea-ice concentration because in some months in KB the distance resulted in unrealistic potential movements of bears.

We attempted to quantify the dates of arrival and departure on land in KB as described for BB in Chapter 4. This analysis was confounded by the fact that KB bears have access to sea ice much of the summer. Thus KB bears used land intermittently and it was not possible to quantify a specific date where bears arrived on shore and did not leave. There were no potential swimming events identified in KB. Maternity denning analyses were conducted with the same methods as described in Chapter 4 for BB.

9.3 Results and Discussion

Sea-ice habitat

Kane Basin consisted of 81 SSMI sea ice grid cells ($53 \times 103 \text{ km}^2$); 68% had a mean depth $< 300 \text{ m}$, 32% had a mean depth $> 300 \text{ m}$. The seasonal cycle of the sea ice in KB has changed dramatically since the 1990s (Figure 9.2). In the 1990s sea ice did not disappear from KB and in summer months $>50\%$ of KB was ice covered. In the 2000s, there has been greater extent of sea-ice loss and KB reaches $\sim 5\%$ coverage in summer. The sea-ice loss has been most pronounced from May and through the late fall, and there are few differences in sea-ice coverage between January and April. Trends in the four sea-ice metrics (described in Chapter 4) are provided in Table 9.2. All trends are statistically significant and show a loss of sea-ice habitat. In Kane Basin as a whole, sea ice is retreating earlier in spring by 7 days/decade, and advancing later in fall by 5-6 days/decade (Figure 9.3). 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 5-6 percent/decade (Figure 9.4ab). The trend in the date of spring sea-ice retreat is apparently stronger for the shallow depths of KB than for all depths. The trend in the date of fall sea-ice advance is the same for both depth categories. The downward trend in the June-October sea-ice concentration is apparently stronger for the shallow depths.

The year-to-year variability in sea-ice metrics for Kane Basin was larger than for Baffin Bay, i.e., the scatter about the trend lines was larger. There are several reasons for this variability: (1) the North Water Polynya is an area of dynamic sea-ice activity that affects Kane Basin; (2) there is typically an “ice arch” north of Kane Basin that determines whether ice lingers in the basin (arch intact) or is flushed out (arch collapses); and (3) Baffin Bay is much larger than Kane Basin and so is less affected by such relatively small-scale phenomena as (1) and (2).

The trend in the annual number of ice-covered days in Kane Basin is between -5 and -15 days/decade for most of the areas with shallow depths (Table 9.2). For the southern portion of Kane Basin, the trend is steeper than -15 days/decade on the Greenland side and there is almost no trend on the deeper Ellesmere Island side. Thus, the pattern of extreme sea-ice loss in Baffin Bay along the coast of Greenland (see Chapter 4) extends northward into the southeast portion of Kane Basin.

Spring sea-ice melt in the Kane Basin region begins in May in the North Water Polynya, which generally becomes ice-free by July. Kane Basin proper, to the north, generally holds some sea ice all summer. Figure 4.5 shows that on July 15, Kane Basin proper is almost always ice-covered, often with 50% or more sea-ice concentration. The year 2009 was exceptional, when all the ice in Kane Basin was swept out in May and June. In October, sea ice advances from north to south through Kane Basin, but the date of advance is generally trending later (Table 9.2 and Figure 9.2).

Movement rates – In Kane Basin, mean monthly movement rates for adult females were overall lower than in Baffin Bay. In KB in the 1990s, mean monthly movement rates ranged from a low of 3.4 km/day (in August) to a high of 9.4 km/day (in February) (Figure 9.5, Table 9.3). Rates for adult females in the 2000s were similar and ranged from 4.7 km/day (in September) to 6.94 km/day (in November) (Figure 9.5, Table 9.3). Compared to Baffin Bay, there was a substantially less pronounced cycle to movement rates over the year in KB. There were no significant differences in movement rates between decades except in October in the 2000s where rates were higher than the 1990s (Table 9.3).

RSF sea-ice models – The sample sizes of adult females in KB in the 1990s and 2000s were smaller than in BB (12 and 20 bears, respectively). Collars deployed between 2012 and 2013 were removed in April 2014 thus tracking durations in the later period were also shorter. We examined univariate relationships for multiple habitat covariates over the annual cycle of sea ice (Figure 9.7 and 9.8). KB bears in the 1990s used similar sea-ice concentrations as bears in the 2000s between January and May (Figure 9.7). Starting in late spring (June) and continuing through December, KB bears in the 2000s used significantly lower sea-ice concentrations than in the 1990s. This was most pronounced in August-October. Distances from bears to the 15% or 50% sea-ice concentration thresholds varied widely across the annual cycle and were similar between decades, though in the 2000s bears were significantly closer to the 50% sea-ice edge in March and April. There were no large differences in bears' distance to land in either decade, other than bears being closer to land from October-December in the 2000s.

The multivariate RSF model in winter demonstrated adult female polar bears in the 1990s had a strong preference for higher ice concentrations. This preference was not present in the

2000s. In both decades bears had a similar strength of preference for the distance to the 300 m depth contour and preferred shallower depths (more strongly and significantly in the 2000s) (Table 9.4). The multivariate RSF model for spring showed that bears also had a strong significant preference for higher ice concentrations in the 1990s (Table 9.5). The preference was reduced in the 2000s but there was no significant difference between decades. In spring in the 1990s, bears were farther from the shelf break (300 m contour), whereas in the 2000s they were closer to 300 m and this change was significant between decades. There was no preference for depth in either decade in spring. In both decades, there was a preference not to move to land, but this was significantly stronger in the 2000s.

RSF terrestrial models – Adult female use of land was intermittent in KB, thus land use models reflect use of land largely near the shoreline as bears moved on and off sea ice (Figure 9.9, Figure 9.10). The terrestrial models demonstrated that KB bears preferred lower elevations, a preference which has significantly increased in the 2000s. Bears tend to avoid steep slopes in both decades and were significantly less likely to move to sea ice once they were on land (Table 9.6).

Arrival and departure dates – KB is part of the Archipelago ecoregion, which in contrast to the seasonal ice ecoregion, historically does not melt out completely each year. Bears in KB exhibit fundamental differences in their habitat use because of the availability of sea ice between systems. In general, KB bears had access to sea ice for most of the summer, especially in the 1990s though this has been significantly reduced in the 2000s. Some bears utilized fjord ice for most of the summer and never arrived on land, while others spent intermittent time on land. Overall patterns of land use among individuals were not consistent and thus it was not possible to quantify on-land arrival and departure dates. No long-distance swimming events were observed,

though one of the swimming events recorded for BB in July 2010 resulted in a BB collared bear arriving on Ellesmere Island after a long distance swim from offshore pack ice in Northern BB.

Kane Basin denning – Nine dens were found from 2012 to 2015 in KB: three maternity dens (Figure 9.11) and six shelter dens (Figure 9.12). In the 1990s data, Ferguson et al. (1997) also found nine dens, of which three were maternity dens and six were shelter dens (Table 9.7). All dens were on land with the exception of one 1990s shelter den that was located on landfast ice nine kilometers from the shore of Ellesmere Island. Most of the dens were located on Ellesmere Island except for three dens on Devon Island. None of the adult females from KB denned on Greenland. The minimum latitude for the 1990s dens was 77.94° N, and 77.04° N for the 2000s dens.

There was no significant difference in maternity denning duration ($p = 1$) (Table 9.8, Figure 9.13), entry dates ($P = 0.6$) and exit dates ($P = 1$) (Figure 9.14). Only four of the KB bears in the 2000s provided useable temperature data for inferring exit dates and no temperature data were available from the 1990s. There was no significant correlation between latitude and maternity den entry dates ($\tau = 0.138$, $P = 0.848$) or duration ($\tau = 0.2$, $P = 0.707$). The median first date on land for the $n=3$ pregnant females in the 1990s was 18 September (SD = 31 days) and in the 2000s was 23 August (SD = 20.8 days; Figure 9.15). The difference between the two time periods was not significant despite the median FDOLs being 27 days apart. The sample size was small and there was considerable variability. Habitat characteristics among maternity dens did not significantly differ between decades (Figure 9.16; Table 9.9, 9.10).

9.4. Literature Cited

Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 in E. T.

- DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea-ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.
- Ferguson, S. H., M. K. Taylor, E. W. Born, and F. Messier. 1998. Fractals, sea-ice landscape, and spatial patterns of polar bears. *Journal of Biogeography* 25:1081–1092.
- Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. *Ecology* 81:761–772.
- Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. *PLoS ONE* 9(11): e113746. doi:10.1371/journal.pone.0113746.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. *The Cryosphere Discussion Papers*, in review.
- Taylor, M. K., S. Akeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. *Canadian Journal of Zoology* 79:690–709.

Table 9.1. Breakdown of adult females (AF) collared in the Kane Basin region 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	KB	3	0	5	3	1	12
2000s	KB	5	1	7	3	4	20

Table 9.2. Trends in date of spring sea-ice retreat, fall sea-ice advance, fall – spring dates, and summer (June-Oct) sea-ice concentration in Kane Basin (all depths, and depths < 300 m). All trends are statistically significant at the 99% level according to a 2-sided F test, except the date of spring retreat (all depths), which is significant at the 95% level.

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	-6.8	+5.6	+12.4	-5.4
Depths < 300 m	-9.7	+5.5	+15.2	-6.9

Table 9.3. Mean monthly movement rates in KB for radio-collared adult female polar bears (AFs) in the 1990s and 2000s. One SE about the mean is given. We conducted a parametric test of significance between decades; bold = significant at the 5% level.

Month	Mean	SE	N	N	Mean	SE	N	N	t-test P value
	1990s	1990s	1990s	steps	2000s	2000s	2000s	steps	
1	3.62	3.78	5	21	4.87	7.59	12	98	0.8691
2	9.39	9.74	4	16	5.9	6.99	11	93	0.4504
3	8.52	9.17	7	35	5.63	6.73	13	117	0.3802
4	5.79	7.28	13	80	5.66	6.57	13	138	0.5197
5	7.35	10.19	12	92	6.75	5.87	21	230	0.6035
6	5.31	7.2	12	85	5.22	4.47	18	193	0.6012
7	3.6	3.19	11	81	5.75	4.13	14	166	0.1011
8	3.41	3.45	10	62	4.84	4.66	14	132	0.5136
9	3.91	5.9	10	63	4.5	4.57	14	132	0.1466
10	4	3.07	9	51	6.42	7.1	14	141	0.0394
11	7.87	8.96	9	40	6.94	6.57	13	118	0.986
12	7.87	8.82	8	36	4.68	5.22	12	116	0.3835

Table 9.4. Sea-ice resource selection function (RSF) model coefficients for the Winter season in KB 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 such by a certain number of units for more meaningful 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		p-value	2000s		p-value	p-value
	coef	SE	1990s	coef	SE	2000s	delta
Mean ice conc.10	0.479	0.564	0.3963	0.074	0.192	0.699	0.4976
Dist to 50%.100	-0.014	0.134	0.918	-0.032	0.115	0.7823	0.9181
Dist to 300 m.100	1.82	0.824	0.0272	1.738	0.826	0.0353	0.9436
Depth.100	-0.034	0.078	0.6579	-0.065	0.084	0.4387	0.7892
Land	3.377	4.895	0.4903	1.108	1.504	0.4615	0.6577

Table 9.5. Sea-ice resource selection function (RSF) model coefficients for the Spring season in KB 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 such by a certain number of units for more meaningful 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. Bold = significant at the 5% level.

	1990s		P-value	2000s		P-value	P-value
	coef	SE	1990s	coef	SE	2000s	delta
Mean ice conc.10	0.255	0.062	<0.001	0.223	0.058	0.0001	0.7107
Dist to 50%.100	-0.007	0.051	0.8867	-0.118	0.071	0.0949	0.2031
Dist to 300 m.100	1.324	0.502	0.0084	-0.997	0.483	0.0392	<0.001
Depth.100	-0.159	0.043	0.0002	-0.102	0.051	0.0451	0.4
Land	-0.349	0.476	0.4639	-0.588	0.469	0.2107	0.7208

Table 9.6. Terrestrial resource selection function (RSF) model coefficients for the Summer season in KB 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 such by a certain number of units for more meaningful 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. “Not Land” is the tendency of a bear to move from land on to sea ice. Note not all bears used land in summer and land-use was intermittent. Bold = significant at the 5% level.

	1990s		P-value	2000s		P-value	P-value
	coef	SE	1990s	coef	SE	2000s	delta
elev.100	-0.452	0.07	<0.001	-0.74	0.054	<0.001	0.001
slope.10	0.256	0.136	0.0594	0.316	0.07	<0.001	0.692
aspect.10	-0.019	0.013	0.1621	0.007	0.007	0.3166	0.088
NotLand	-0.443	0.317	0.1619	-1.186	0.166	<0.001	0.03

Table 9.7. Number of polar bear maternity and shelter dens in Kane Basin in the 1990s and 2000s.

All Dens		Maternity Dens		Shelter Dens	
1990s	9	1990s	3	1990s	6
2000s	9	2000s	3	2000s	6
Total	18	Total	6	Total	12

Table 9.8. Summary table of the phenology for Kane Basin polar bear maternity dens.

	1990s			2000s		
	Maternity Dens (<i>n</i> = 3)			Maternity Dens (<i>n</i> = 3)		
	Entry DOY	Exit DOY	Duration (# days)	Entry DOY	Exit DOY	Duration (# days)
Mean	279	78.3	164.3	274	77.7	168.7
Min	274	69	145	252	65	144
Max	289	89	180	301	88	184
Median	274	77	168	269	80	178
SD	8.7	10.1	17.8	24.9	11.7	21.6

Table 9.9. Summary table of the habitat characteristics for Kane Basin polar bear maternity and shelter dens. Elev. = elevation (meters), Asp. = aspect (degrees), CoastDist = distance to nearest coastline (kilometers).

	All Maternity Dens (<i>n</i> = 6)				All Shelter Dens (<i>n</i> = 12)			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	327	165.8	13.4	5.2	366.6	141.8	15.4	3.4
Min	9	28.3	3.7	0.7	6	5.5	1.8	0.1
Max	506	229	17.8	12.5	855	350	36.9	8
Median	408	186.8	15.9	3.4	318	168.3	12	2.8
SD	188.4	69.8	5.4	4.8	274.8	126.3	10.8	2.4
	1990s Maternity Dens (<i>n</i> = 3)				1990s Shelter Dens (<i>n</i> = 6)			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	207.7	201	11.9	2.1	386.7	134	12.9	2.6
Min	9	185.9	3.7	0.7	6	12	1.8	0.1
Max	422	229	16.1	4.3	855	349.7	36.9	5.5
Median	192	188.1	15.8	1.2	257	105.7	9.9	2.3
SD	206.9	24.3	7.1	1.9	378.5	137.4	12.4	2.1
	2000s Maternity Dens (<i>n</i> = 3)				2000s Shelter Dens (<i>n</i> = 6)			
	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)	Elev. (m)	Asp. (°)	Slope (°)	CoastDist (km)
Mean	446.3	130.6	15	8.3	346.5	149.5	17.9	4.2
Min	394	28.3	10.7	2.6	149	5.5	7.2	1.6
Max	506	187.7	17.8	12.5	500	350	32.5	8
Median	439	175.9	16.5	9.7	355.5	168.3	16.4	4.1
SD	56.4	88.8	3.8	5.1	148.2	126.7	9.3	2.5

Table 9.10. Results of the two-sample Mann-Whitney U tests comparing habitat characteristics for Kane Basin polar bear maternity and shelter dens.

	Maternity Dens (<i>n</i> = 6)		Shelter Dens (<i>n</i> = 12)	
	W	<i>p</i>-value	W	<i>p</i>-value
Elevation	8	0.2	20	0.818
Slope	7	0.4	25	0.31
Aspect	1	0.2	17	0.937
Distance to Coast	8	0.2	24	0.394

Figure 9.1. Distribution of capture locations for polar bears in Kane Basin (KB) in each decade. See Table 9.1 for sample sizes in each year. Note in 1990s bears in KB were captured on the west side of KB whereas 2000s bears were captured on both the east and west side.

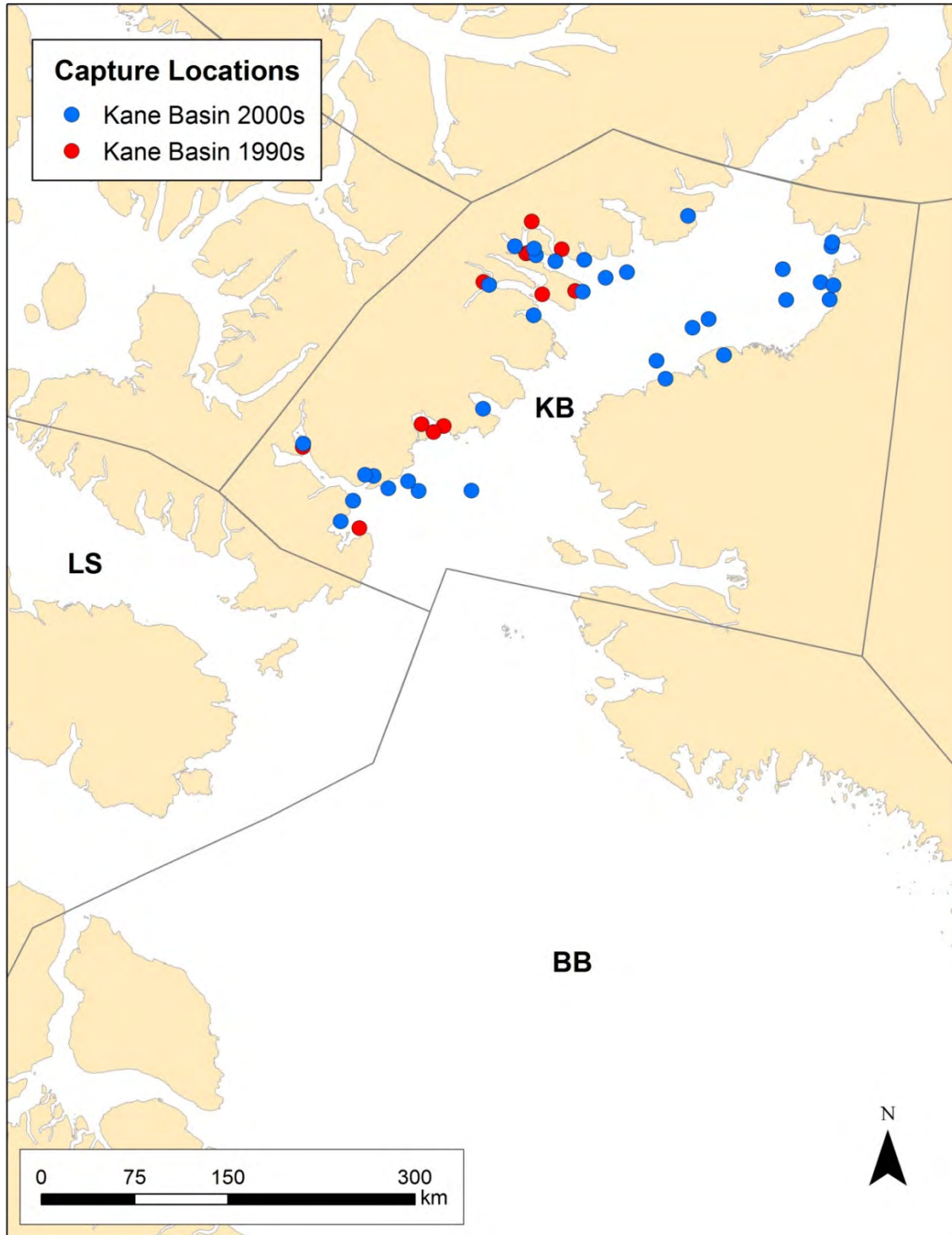


Figure 9.2 Sea-ice area in Kane Basin (all depths) for the years 1979-2014 (gray curves). Two six-year averages are also shown (colored curves). The threshold for defining the dates of sea-ice retreat and advance (middle horizontal dotted line) is halfway between the average March sea-ice area (upper dotted line) and the average September sea-ice area (lower dotted line).

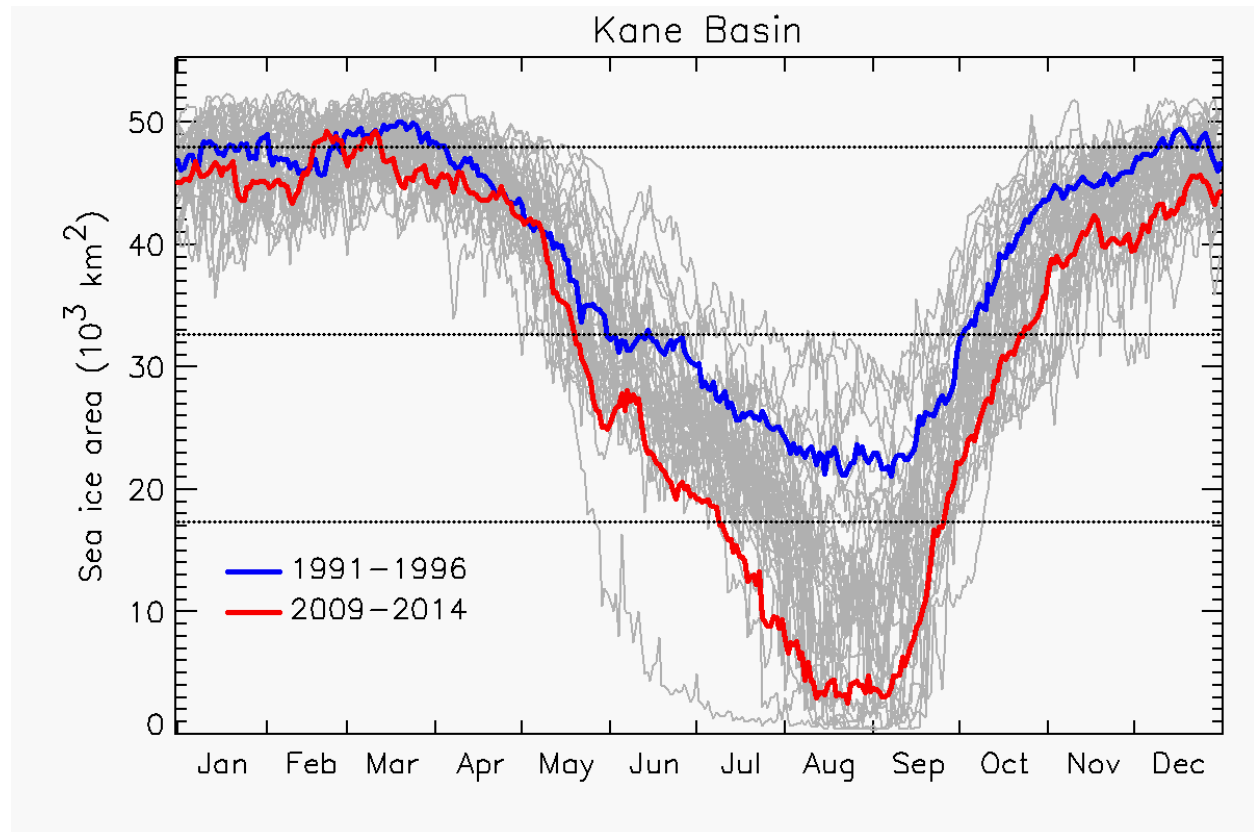


Figure 9.3 Day of spring sea-ice retreat (red circles), fall sea-ice advance (blue circles), and the interval between them (green lines), for Kane Basin (all depths), 1979-2014. Least-squares fits to spring and fall dates are shown (red and blue lines). Trends are given in Table 9.2.

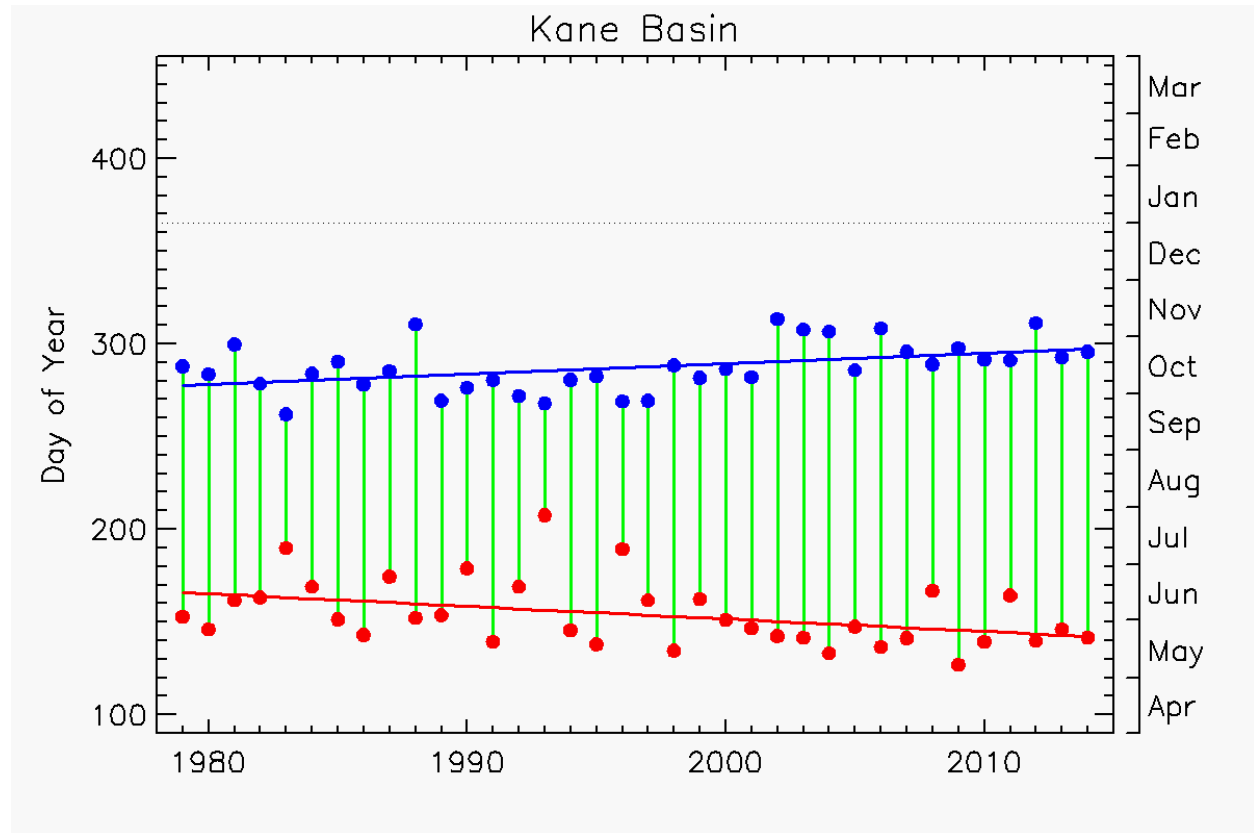
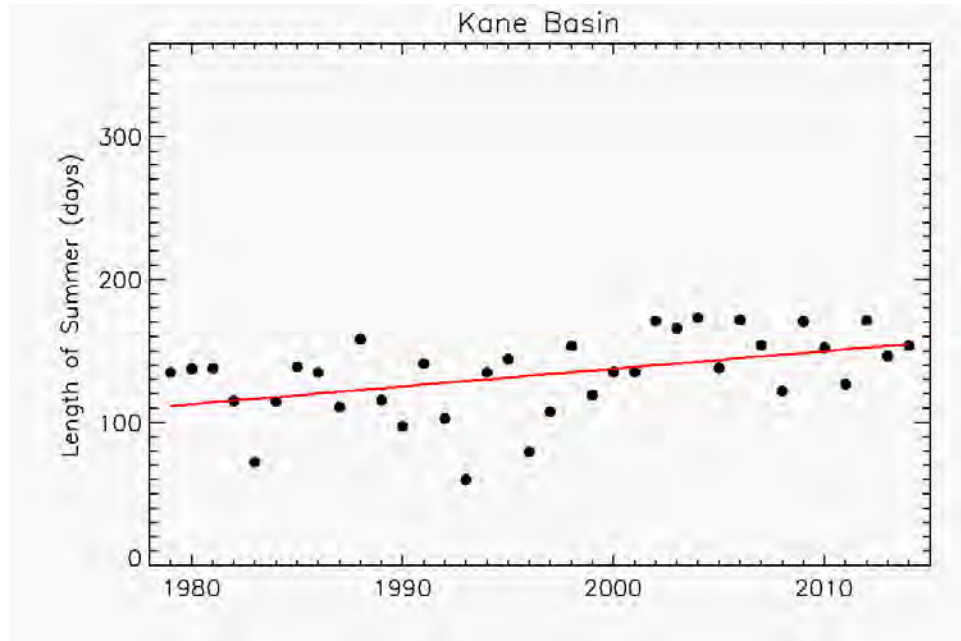


Figure 9.4. Length of summer (a) and mean sea-ice concentration during June-October (b) for Kane Basin (all depths), 1979-2014. Length of summer is the interval from spring sea-ice retreat to fall sea-ice advance (see Figure 9.2, green lines). Least-squares fits are shown (red lines); trends are given in Table 9.2.

(a)



(b)

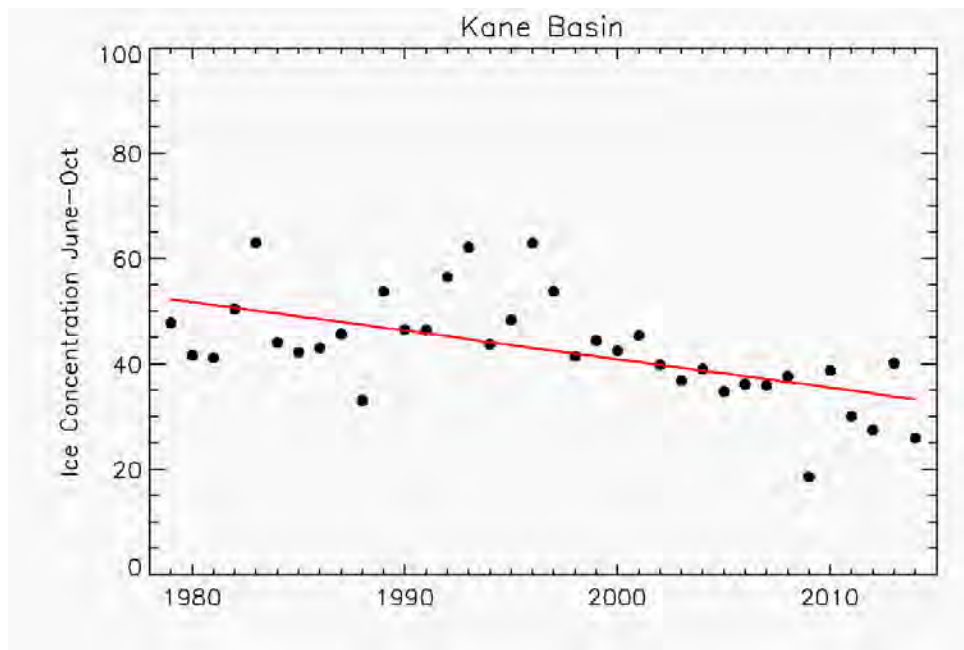


Figure 9.5. Movement rate of KB adult female bears (km/day) in the 1990s Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

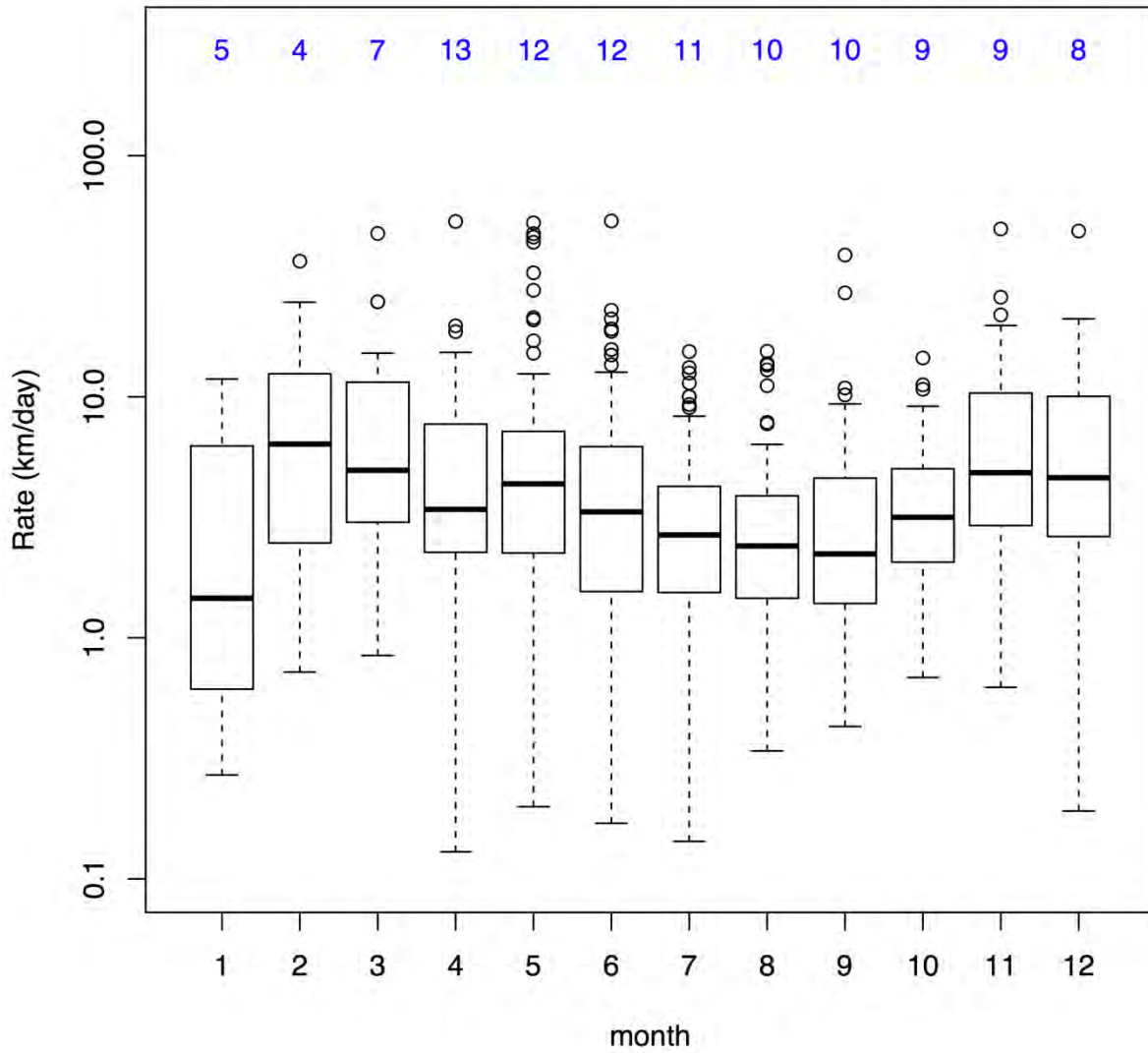


Figure 9.6. Movement rate of BB adult female bears (km/day) in the 2000s. Y axis is on a log scale and labels are listed as raw values. Blue numbers indicate the number of individual bears in each month.

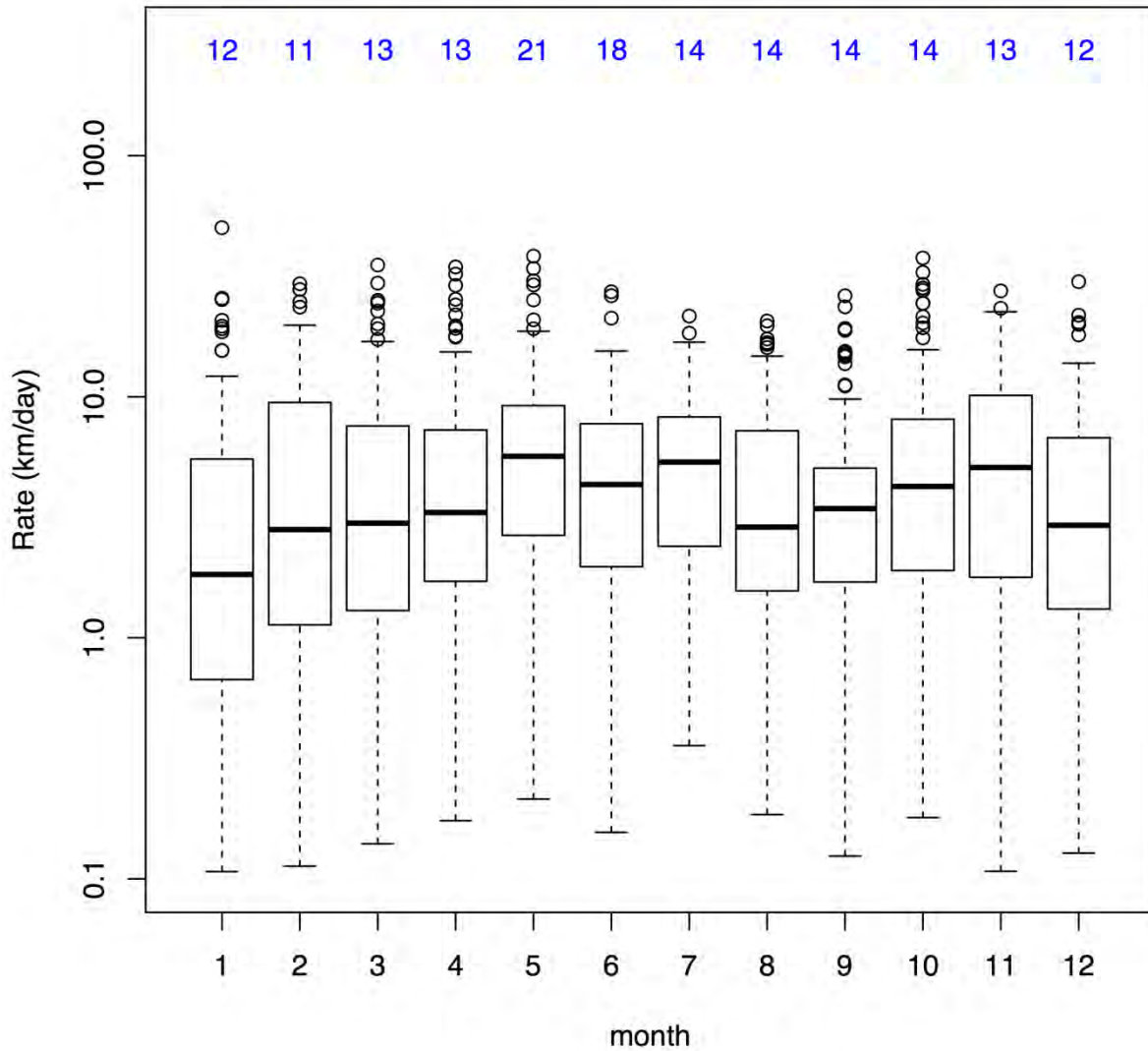


Figure 9.7. 1990s and 2000s adult female polar bear habitat use in KB for each of four sea-ice habitat variables: sea-ice concentration in small buffer, distance to 15% sea-ice concentration, distance to 50% sea-ice concentration, and distance to the nearest land. Data from 1990s are shown in red, 2000s 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. Months 8 -10 also represent land use by bears. The small numbers indicate the number of movements captured within each months.

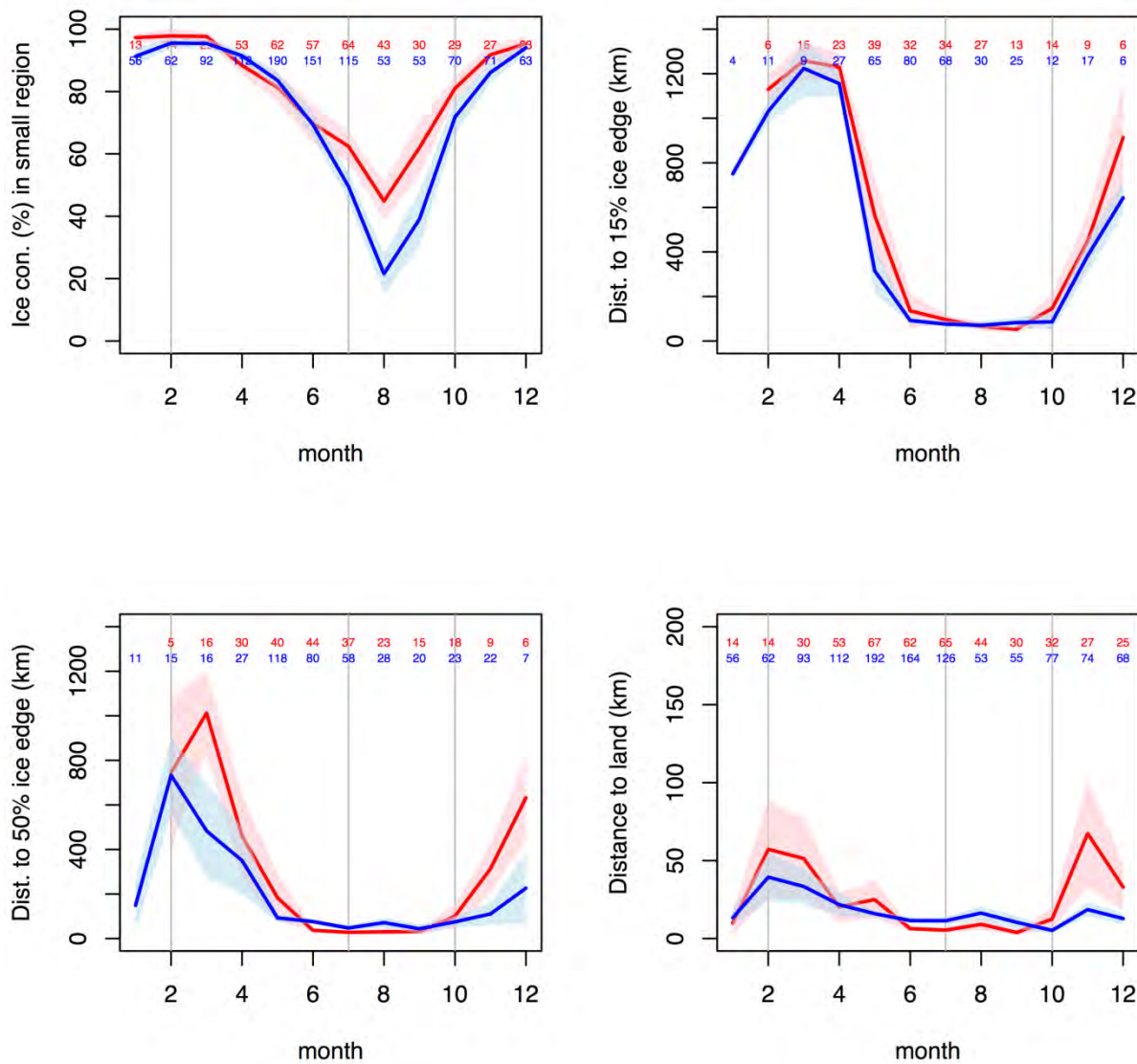


Figure 9.8. 1990s and 2000s adult female polar bear habitat use in KB for each of three sea-ice habitat variables: distance to 300 m depth contour, depth (bathymetry), and percentage of observations on the sea ice. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Vertical lines indicate monthly boundaries for seasons (winter, spring summer). SSM/I sea-ice concentration is used in both decades. Months 8 -10 also represent land use by bears.

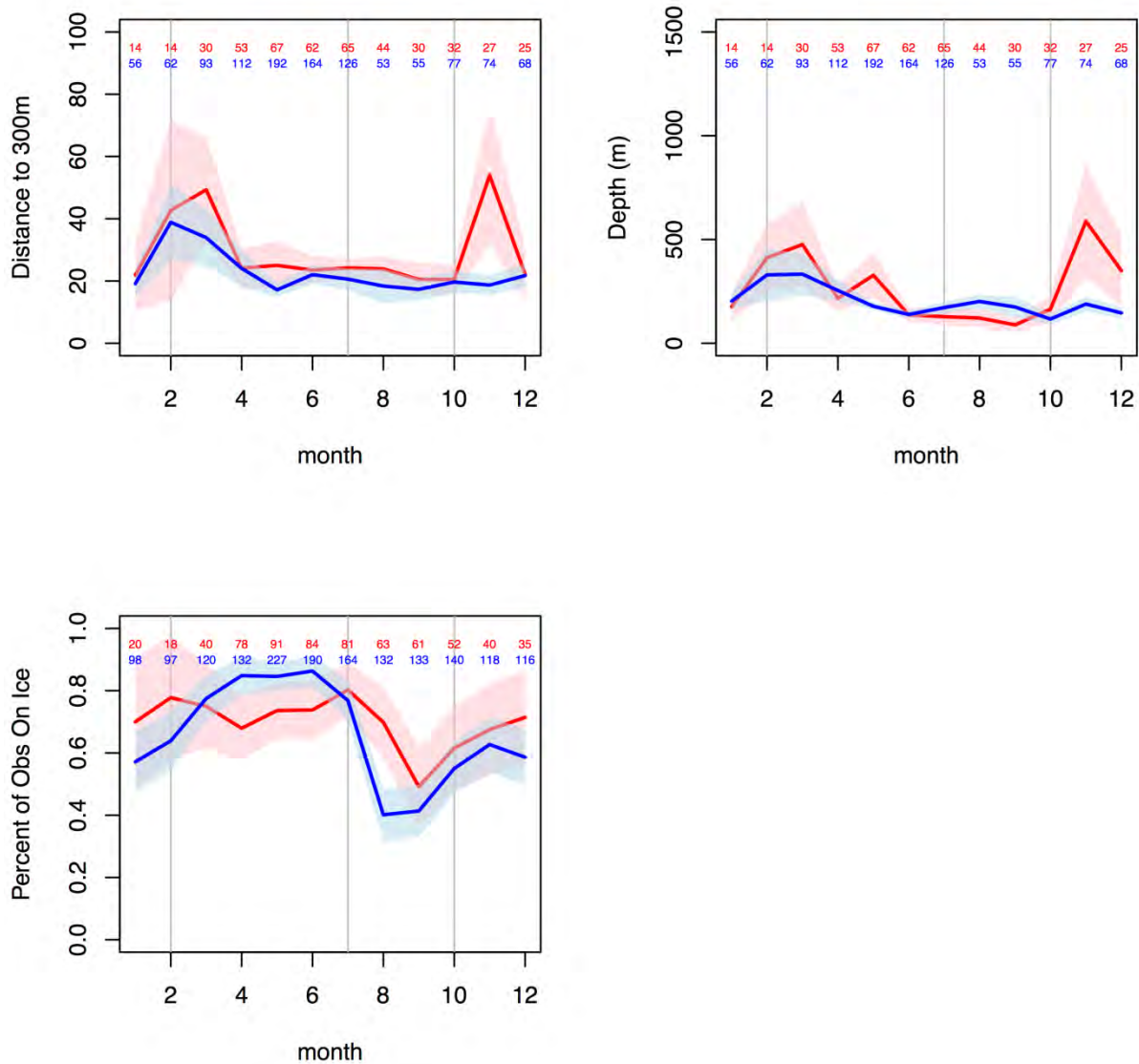


Figure 9.9. 1990s and 2000s adult female polar bear habitat use in KB for each of three terrestrial habitat variables: elevation, slope, and aspect. Data from 1990s are shown in red, 2000s in blue. Shaded regions represent 2 SE of the mean. Months 8 -10 also represent land use by bears.

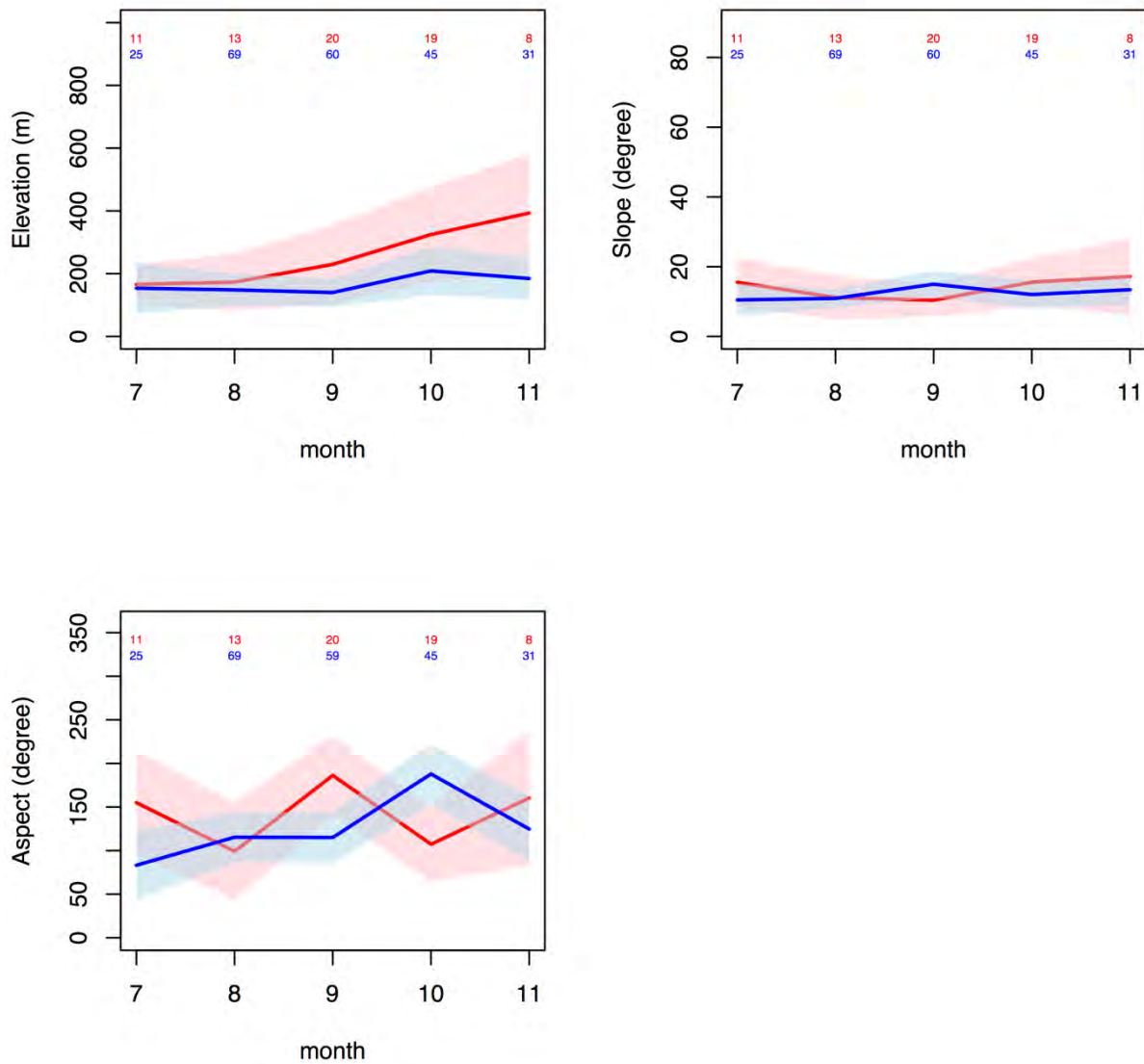


Figure 9.10. ASTER Digital Elevation Model (DEM) used in KB with trackline of a single bear tagged in the 2000s.

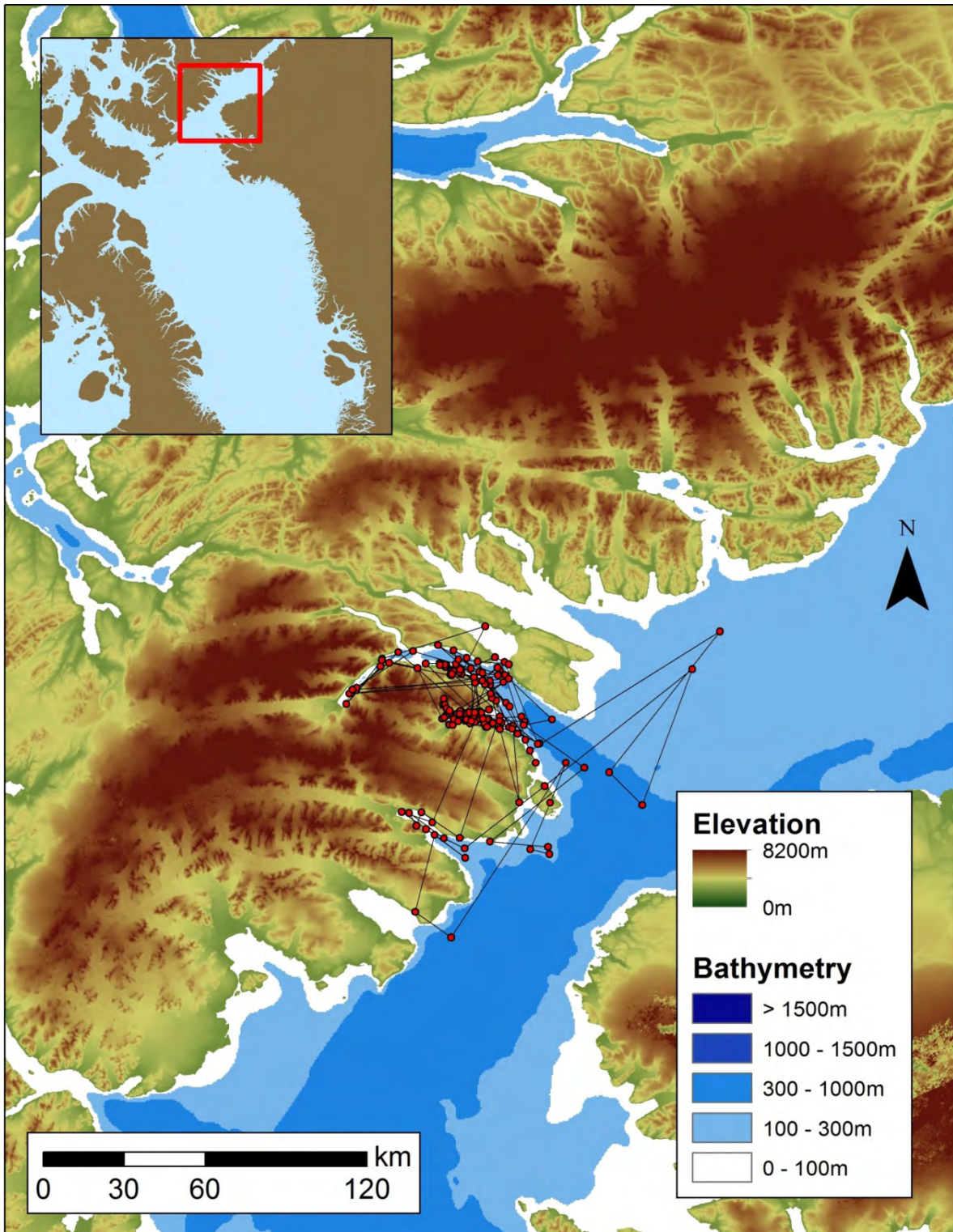


Figure 9.11. Distribution of KB polar bear maternity dens in the 1990s and 2000s.

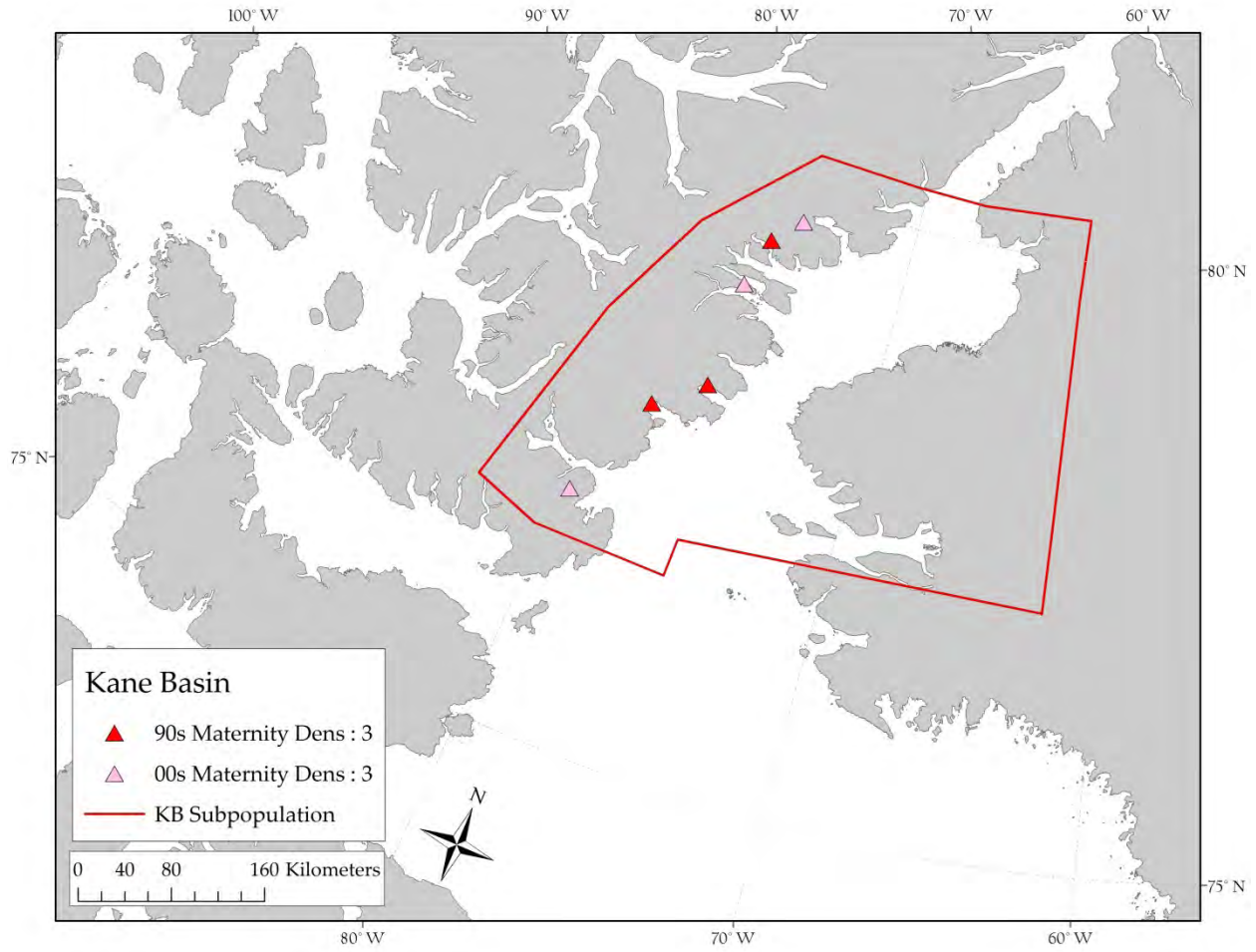


Figure 9.12. Distribution of KB polar bear shelter dens in the 1990s and 2000s.

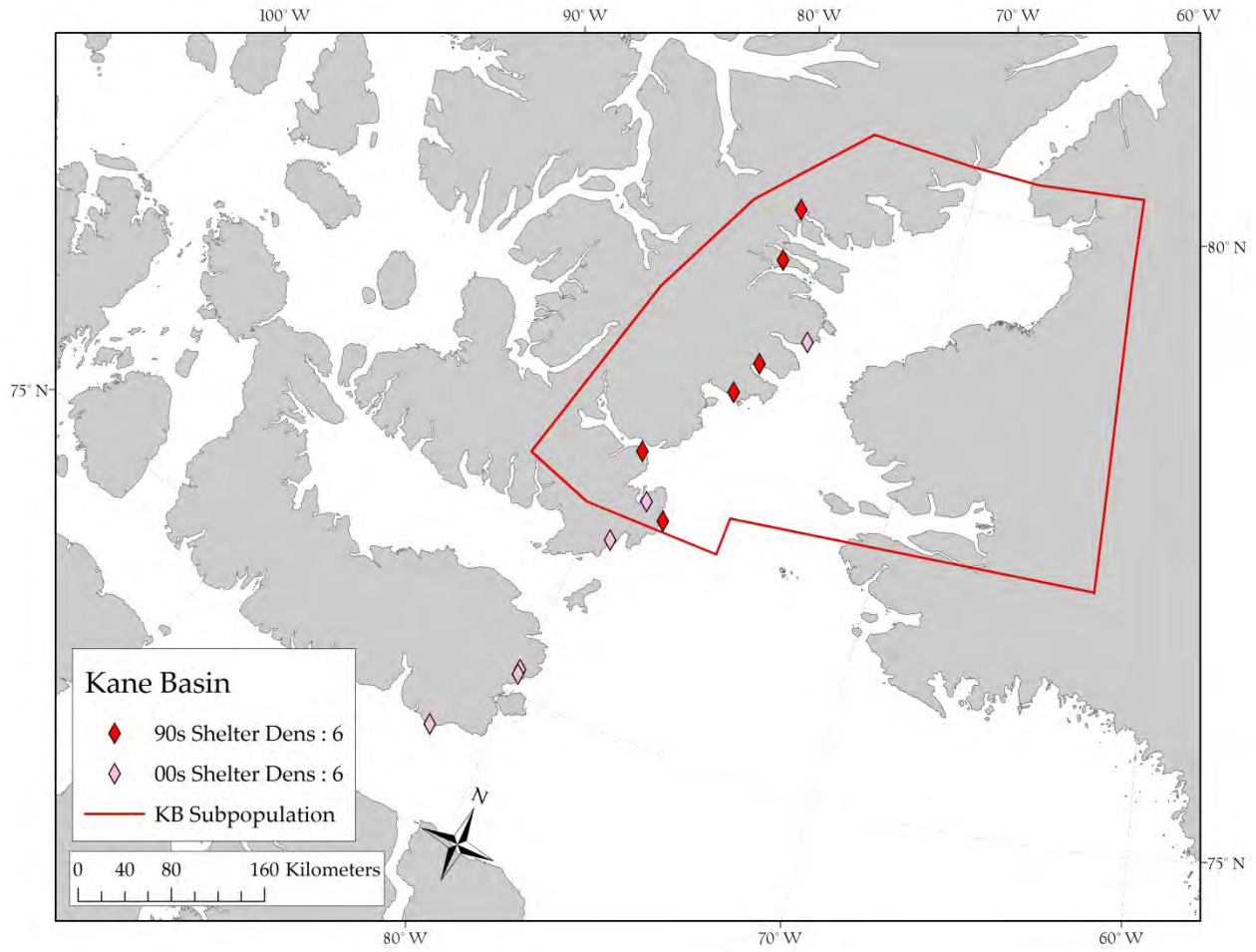


Figure 9.13. Boxplots comparing den duration of Kane Basin (KB) polar bear maternity dens ($P = 1$) (1990s: $n = 3$; 2000s: $n = 3$).

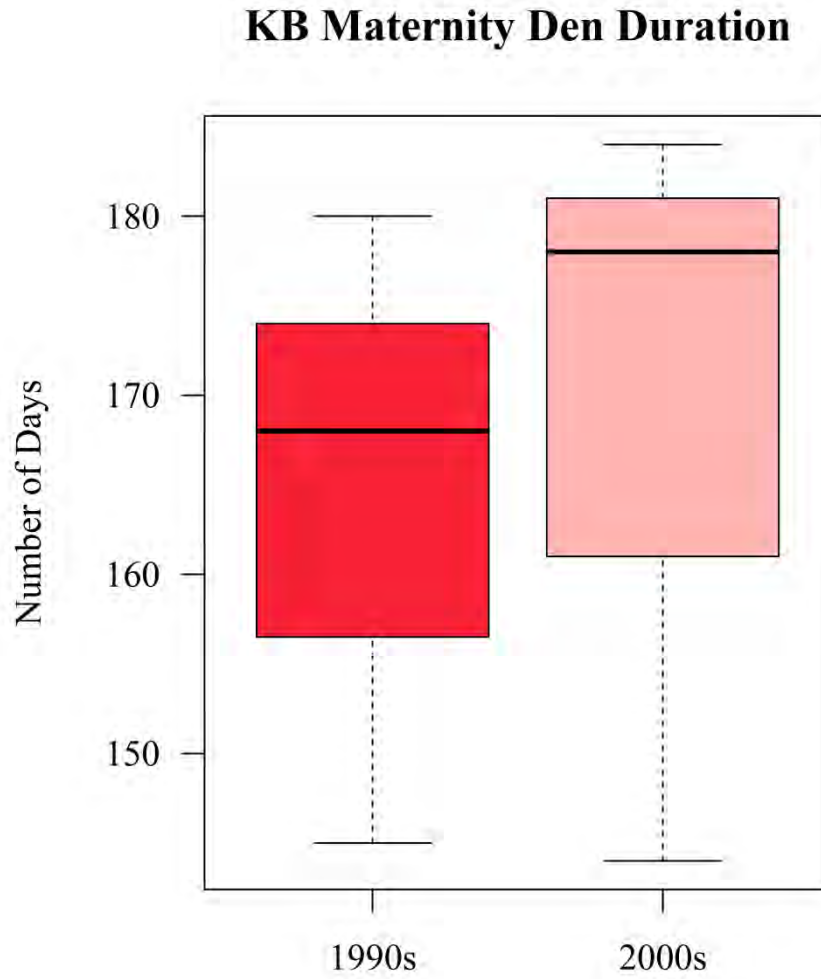


Figure 9.14. Boxplots comparing entry ($P = 0.6$) and exit dates ($P = 1$) of Kane Basin (KB) polar bear maternity dens (1990s: $n = 3$; 2000s: $n = 3$).

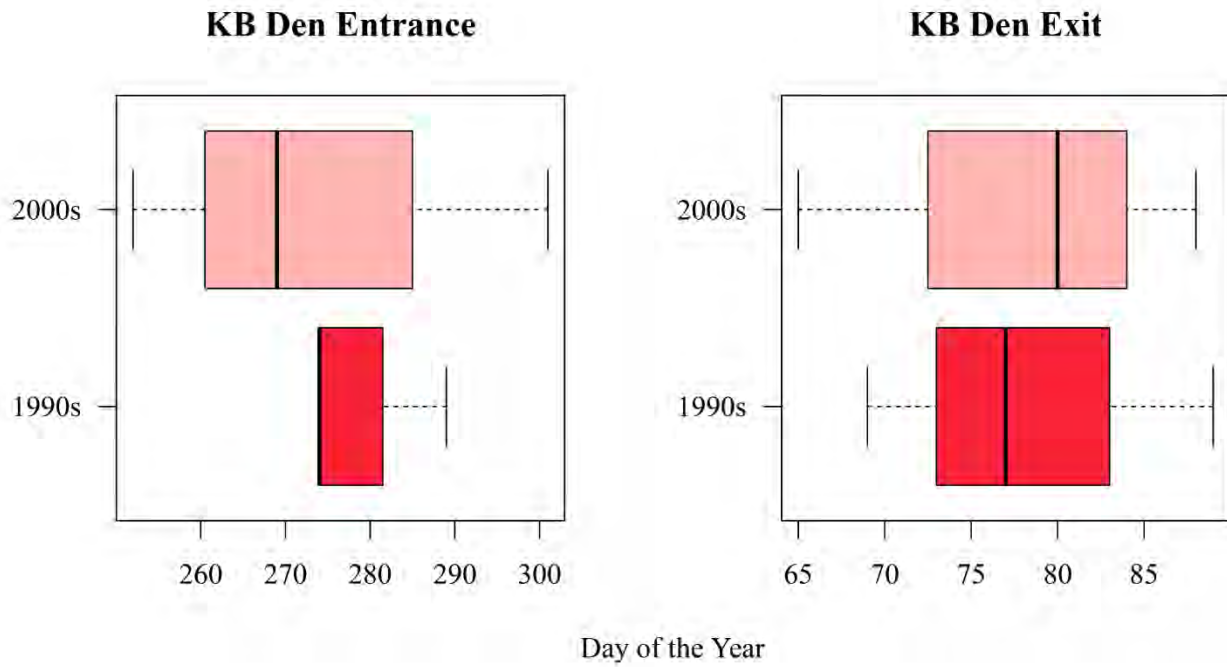


Figure 9.15. Boxplots comparing the first date on land (FDOL) of pregnant female polar bears from the 1990s ($n = 3$) and 2000s ($n = 3$) in Kane Basin (KB) ($P = 1$).

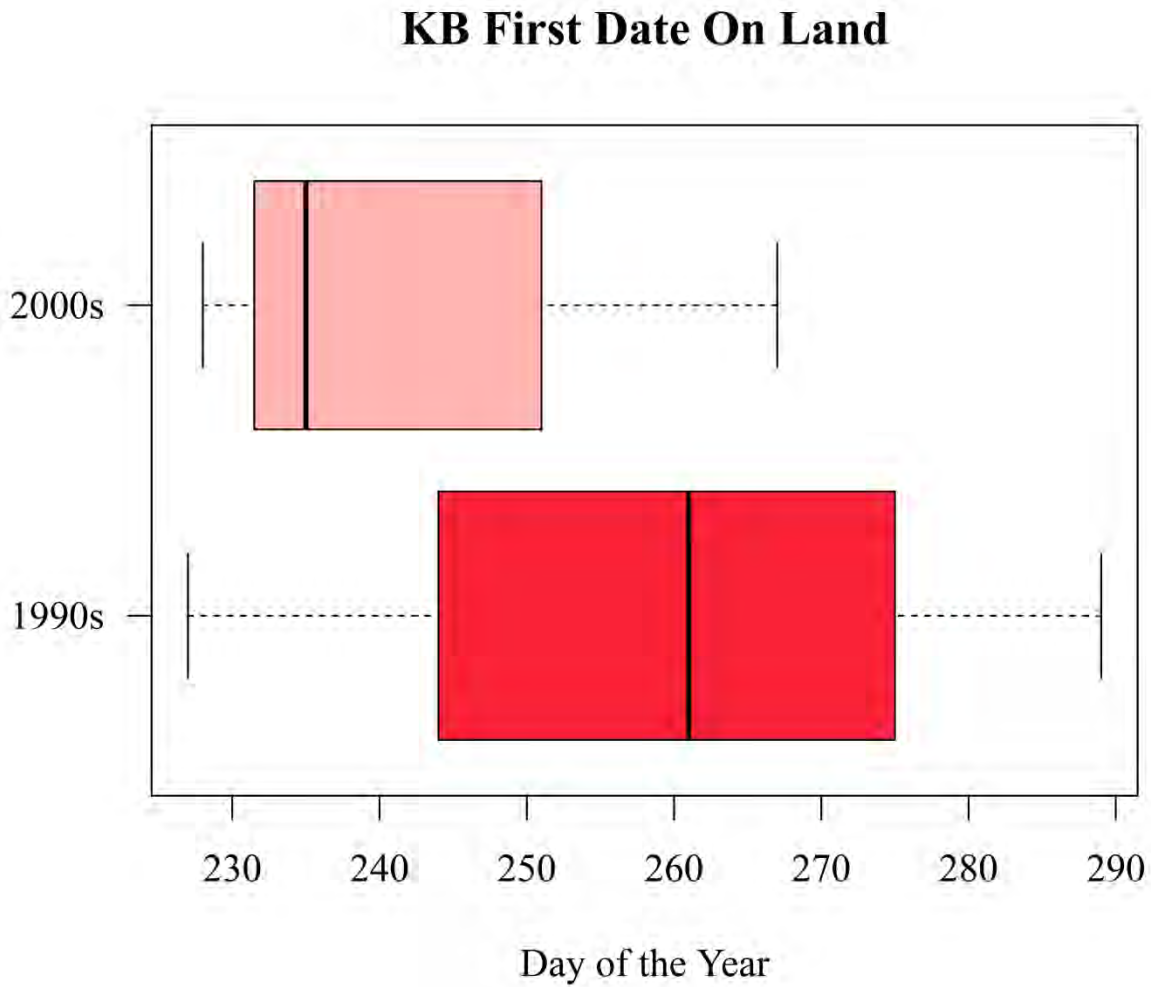
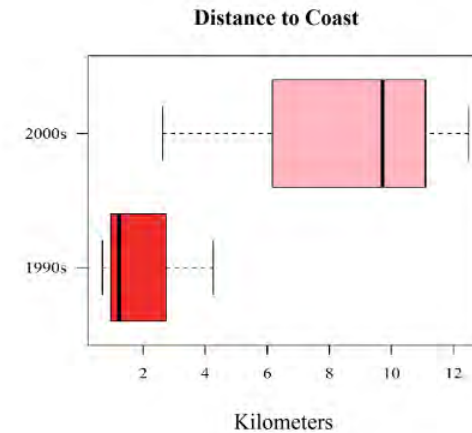
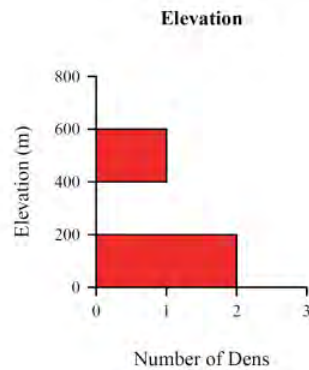
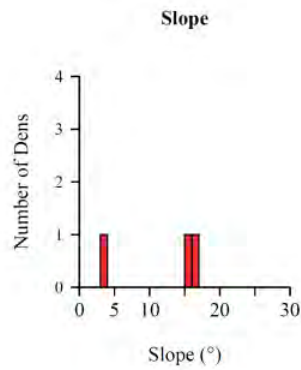
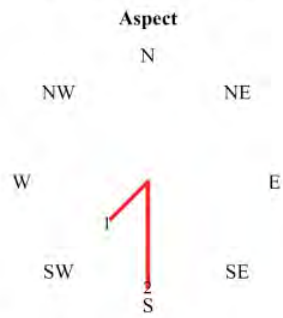
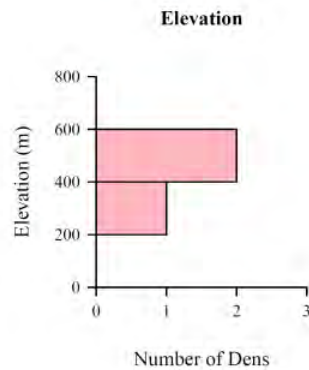
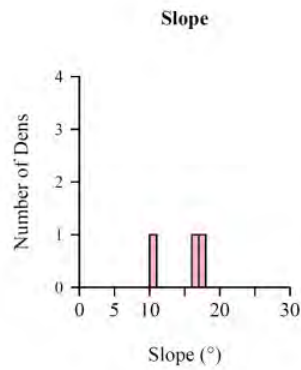
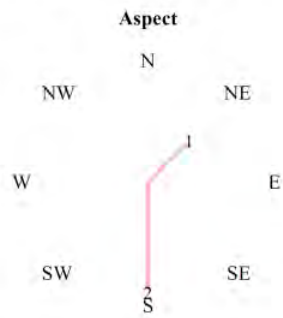


Figure 9.16. Plots comparing aspect, slope, elevation, and distance to coast for the 1990s ($n = 3$) and 2000s ($n = 3$) polar bear maternity dens in Kane Basin. The aspect plot consists of a compass face with lines marking the directions that dens faced. The lines are annotated with numbers noting how many dens were found at that aspect. None of the habitat variables significantly differed between the two time periods (elevation, aspect, distance to coast: $P = 0.2$; slope: $P = 0.4$).

1990s Maternity Dens



2000s Maternity Dens



CHAPTER 10

GENETIC MARK-RECAPTURE STUDY OF POLAR BEARS IN KANE BASIN

KEY FINDINGS

- We used joint live-recapture and dead-recovery mark-recapture models to analyze data for the Kane Basin (KB) polar bear subpopulation, with the goal of updating estimates of subpopulation size and survival. The dataset consisted of 277 initial live captures (1992-1997 = 150, 2012-2014 = 127), 89 live recaptures (1992-1997 = 53, 2012-2014 = 36), and 24 harvest returns of research-marked bears 1992-2014.
 - Mark-recapture research conducted in the Kane Basin subpopulation yielded an estimate of abundance of 357 polar bears (95% CI: 221 – 493) for 2013 – 2014. An estimate derived during 1995 – 1997 yielded 224 bears (95% CI: 145 – 303). Based on physical MR, the size of the KB subpopulation was previously estimated to be 164 polar bears (95% CI: 94-234) for 1994-1997 (noting that this estimate applies to different years than our re-analysis; Taylor et al. 2008).
 - We documented more bears in the eastern regions of the Kane Basin subpopulation during 2012 – 2014 than during the 1990s. Eastern Kane Basin was searched during the 1990s although with less effort than in the 2010s due to the low density of bears observed there. The difference in distribution between the 1990s and 2010s may reflect differences in spatial distribution of bears, possibly influenced by reduced hunting pressure by Greenland in eastern KB and thus an increased density of bears in KB, but also some differences in sampling protocols.
 - The 2013 – 2014 estimate of abundance suggests 357 (221 – 493) bears currently use KB in springtime (i.e., the Kane Basin super-population; Kendall et al. 1997), and the current point estimate is higher than the historical estimate. Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of KB abundance for 2013-2014 and the estimate for 1995-1997 was approximately 133 bears (standard deviation of the difference \approx 80 bears), with 95% of the sampling distribution suggesting that population change between the two time periods could have been positive. This suggests relatively strong evidence for a stable to increasing subpopulation, and is consistent with data on movements, condition and reproduction. We encourage some caution in interpretation of population growth due to potential expansion of the sampling frame and differences in sampling protocols between the 1990s versus the 2010s study periods.
 - Current estimates of total survival for age 3+ females (0.95; SE: 0.04) and dependent bears were consistent with previous research. Estimates of unharvested survival for 3+
-

females appear sufficiently high for positive population growth. Updated estimates of total survival are lower for age 3+ males (0.87; SE: 0.06). Our longer-term data set and several other ecological, sampling, and technical considerations may contribute to this result.

- We documented a reduction in mortality associated with harvest, likely attributable to implementation of Greenland's harvest quota in 2006.
 - Demographic modeling suggests Kane Basin bears exhibit relatively high fidelity to the springtime study area, with <5% of marked bears emigrating on an annual basis.
-

10.1. Introduction

Large-scale environmental changes are occurring across the circumpolar Arctic (Comiso et al. 2008, Stroeve et al. 2012, Laidre et al. 2015; see also Chapters 5 and 9), with general reductions in the temporal availability and spatial extent of sea ice. For sea ice obligate polar bears, which are among the most highly sensitive of marine mammals to the projected impacts of climate change (Laidre et al. 2008), long-term impacts are anticipated to be negative (Atwood et al. 2015). However, there likely will be significant temporal and spatial variability among subpopulations in the short-term (Stirling and Derocher 2012). Indeed, the effects of the changing Arctic environment on polar bears have been documented in some regions but are less clear or have not been realized elsewhere (e.g., Rode et al. 2012, 2014, Bromaghin et al. 2015, Obbard et al. 2015, Lunn et al. 2016).

The Kane Basin (KB) polar bear subpopulation, regarded as part of the Arctic archipelago region (Amstrup et al. 2008), covers a small region between Nunavut, Canada and NW Greenland. Abundance of KB was last estimated at ~164 (SE: 35) polar bears based on a physical mark-recapture study completed during 1992 – 1997 (Taylor et al. 2008). At the time of this estimate, the harvested population growth rate ($\lambda = 0.919$) indicated that the subpopulation was over-exploited. The unharvested growth rate also was low ($\lambda = 1.009$; Taylor et al. 2008), suggesting limited capacity for the KB subpopulation to increase even in the absence of human-

caused removals. In response, Greenland implemented a quota in 2006 that significantly reduced the total harvest from Kane Basin (see Methods below). The small subpopulation size, low growth rates, and long-term exploitation led Taylor et al. (2008) to suggest that Kane Basin may act as a sink for neighboring subpopulations such as Baffin Bay.

The KB subpopulation is currently considered to be declining (PBSG 2015): 100% of population viability analysis (PVA) simulations (using data on abundance and vital rates from Taylor et al. 2008 and reported Canadian and Greenlandic catches) resulted in a decline in abundance within 10 years. However, no new research to update estimates of abundance or vital rates has occurred since the 1990s study. Given the outdated demographic information, the substantial changes in Arctic sea-ice habitats over the past several decades (e.g., Stroeve et al. 2012, Chapters 4 and 9), and the reduction in harvest in 2006, there was uncertainty as to the current status of polar bears in Kane Basin. As such, there was a need for new information to inform status and harvest management (Chapter 1).

Although bears in KB are not genetically different from those in Baffin Bay (Paetkau et al. 1999, Chapter 2), satellite telemetry and capture records indicate that they move among KB and neighboring subpopulations but exhibit strong fidelity to specific regions (Taylor et al. 2001, Chapters 2, 5, and 9). These data have formed the basis for population delineation, and polar bears in Kane Basin are considered a distinct demographic unit for management purposes.

Our objective was to estimate the current abundance and vital rates, including survival, of polar bears in the KB subpopulation. We sought to compare new estimates of abundance with those derived from earlier research (Taylor et al. 2008). These results, in conjunction with information on sea-ice dynamics, spatial ecology, reproductive output, survival, and other metrics, will be used to inform subpopulation status. The data used in this project spanned a 23-

year period (1992-2014): an initial 6-year physical capture and dead recovery sampling period (1992 – 1997) was followed by a 14-year period with dead recoveries only (1998 – 2011) and a recent (2012 – 2014) live capture (physical and genetic) and dead recovery session. Jurisdictions across the Arctic have increasingly invested in non-physical capture based monitoring methods, largely to address social considerations, particularly in Nunavut, regarding wildlife handling (*cf.* Chapter 1) and to facilitate more rapid monitoring. Prior to this study and research in the Baffin Bay subpopulation (Chapter 5), however, the focus of such alternative methods has been aerial surveys (e.g., Aars et al. 2009, Obbard et al. 2015, Stapleton et al. 2016).

10.2. Materials and Methods

Study Area

The KB subpopulation covers ~150,000 km² and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (boundaries evaluated in Taylor et al. 2001; Figure 10.1). However, the boundaries of the KB subpopulation encompass a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* Figure 10.3 and 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area) and the southern part of Kennedy Channel. It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the Baffin Bay (BB) and Lancaster Sound (LS) subpopulations, and to the west by Norwegian Bay (NW; PBSG 2010). The KB subpopulation is regarded as a part of the Arctic archipelago ecoregion (Amstrup et al. 2008); historically sea ice remained present in the northern range (i.e., Nares Strait-Kane Basin)

throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, this pattern has changed markedly in recent decades (Chapter 9, Figure 9.2). KB is partially connected to neighboring subpopulations; particularly notable though limited interchange occurs with BB and LS (Chapter 4). The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a partial barrier between KB and BB – LS. The KB subpopulation is subjected to subsistence harvest by Inuit living in Jones Sound (Canada) and the Qaanaaq area (NW Greenland; PBSG 2010, Born et al. 2011).

Field Sampling

Initial surveying was conducted in Kane Basin during springs (April – May), 1992 – 1997 (described in Taylor et al. 2008). Additional sampling was completed during fall 1994, but we excluded these data from the present analyses to reduce temporal heterogeneity (e.g., sampling cubs-of-the-year in spring versus fall yields substantial differences in estimates of survival). All sighted bears, including dependent offspring, were chemically immobilized (Stirling et al. 1989) and uniquely marked with plastic ear tags and permanent lip tattoos (Taylor et al. 2008). Ages of independent bears were determined by extracting vestigial premolars and counting annular rings (Calvert and Ramsay 1998), whereas cubs-of-the-year and yearling bears were considered of known age. A sample of adult females was outfitted with satellite collars as part of a separate study quantifying movements and spatial ecology (Taylor et al. 2001, Chapter 9).

Although Taylor et al. (2008) reported that they conducted a uniform search of the study

site each year, records delineating their survey effort were unavailable. Subsequent examination of annual distributions of captures suggested incremental increases in the size of the study area, progressing northwards, with apparent expansions in the sampling frame between 1992 and 1993 (northward along eastern Ellesmere Island into the Nares Strait region). Between 1994 and 1995, survey efforts were expanded eastward into Kane Basin proper off the Humboldt Glacier in Northwest Greenland (E. Born, pers. obs.) which inferred from Taylor et al. (2001) was also the case in 1996 and 1997; Figure 10.2). No live-recapture sampling occurred during 1998 – 2011, but we obtained recoveries of harvested bears during this interval. Available information also suggests eastern Kane Basin was covered in the 1990s, though no captures were made there.

We surveyed KB during 25 April – 6 May, 2012; 27 April – 10 May, 2013; and 28 April – 19 May, 2014. Sampling windows were comparable to the 1990s, although surveying in 1992 and 1993 occurred earlier (mid-April) and for shorter windows of time. We sampled sea-ice habitats by helicopter (Bell 206 LongRanger) across the entirety of western and northern Kane Basin, including landfast ice in fjords and nearshore areas as well as offshore pack ice, but excluded more open water habitats of the North Water polynya. We also did not survey the sea ice in the fjords of the populated Qaanaaq area in NW Greenland (i.e., the eastern parts of the North Water polynya) because hunting pressure for marine mammals in these areas is generally high and consequently “resident” polar bears do not exist in the Qaanaaq area (E. Born, pers. obs.). Sampling was primarily completed via directed searching in 2012, with searches focused in areas believed to provide the most suitable polar bear habitat (“adaptive sampling”). In 2013, we completed directed searching and also flew ad hoc transects oriented approximately perpendicular to the coastline, particularly near Greenland, to ensure that effort was well-distributed across the landscape.

In 2012 and 2013, most bears were sampled via physical capture, including chemical immobilization and application of ear tags and lip tattoos as described above. We collected tissue samples from physically captured bears (for genotyping) and recorded additional information including sex, family status, field-estimated age class (cub-of-the-year, yearling, 2-year old, subadult, or adult) and standard morphometric measurements. We completed additional sampling in 2012 and 2013 via remote biopsy darting (Pagano et al. 2014) to collect genetic tissues for subsequent genotyping and analyses (e.g., Herreman and Peacock 2013). Cubs-of-the-year were too small in springtime to be biopsy darted and thus were not sampled when their mothers were biopsy darted (although COY were sampled during physical captures). For bears that were not physically immobilized, sex was confirmed upon genetic analyses (see below).

In connection with immobilization and handling, we deployed satellite transmitters on 36 polar bears in 2012 and 2013 (see Chapter 9; 2012: 6 satellite radio collars on adult females; 10 satellite ear-tags on adults and subadults of both sexes; 2013: 10 satellite radio collars on adult females; 10 satellite ear-tags on adults and subadults of both sexes). This work enabled us to evaluate distribution and habitat use during the genetic mark-recapture sampling and the aerial survey (Chapter 11) and to conduct a post hoc assessment of sampling representativeness during sampling.

We modified our sampling strategy during 2014. We stratified the study area into high- and low-density areas based on our observations of polar bears in 2012 and 2013 (i.e., presumed densities) and searched for bears from systematically spaced transects. This design enabled us to more efficiently allocate effort and reduced the potential for spatial heterogeneity in detection. Systematic sampling also facilitated the simultaneous completion of an aerial survey (Chapter

11) to derive an abundance estimate, based on different methodology, for comparison with the capture-based estimate.

Strata conformed to general landscape features and ice types: the high-density stratum included landfast ice within fjords as well as nearshore pack ice (within ~30 km of the nearest land mass); the low-density stratum included farther offshore pack ice (Figure 10.3). We delineated the landward extent of the study area using current GIS layers from Greenland and Nunavut. We used Moderate Resolution Imaging Spectroradiometer (MODIS; <http://modis.gsfc.nasa.gov/>) images with 1 km resolution to delineate the extent of available habitat by approximating the edge of the North Water polynya. Because the polynya's boundaries can change rapidly, we delineated the extent of the polynya adjacent to the section surveyed on a particular day using MODIS imagery from that day (when possible) or as close to that date as possible (when imagery was unclear on that date due to atmospheric conditions). We examined the delineated study area in relation to weekly regional sea-ice charts produced by the Canadian Ice Service (<https://www.ec.gc.ca/glaces-ice/>) for confirmation. During sampling, we also collected GPS waypoints at the edge of the polynya to verify delineation.

Transects were systematically spaced at 6-km and 18-km intervals in the high- and low-density strata, respectively, based on anticipated encounter rates and available resources. We also sampled during ferry flights (e.g., between survey transects). Survey protocols in 2014 (detailed in Chapter 11) were designed to facilitate the simultaneous collection of data for mark-recapture and the aerial survey. All mark-recapture sampling in 2014 was conducted via remote biopsy darting.

Harvest Recoveries

We used harvest records to compile dead recovery data for polar bears captured in KB and subsequently harvested there or in neighboring subpopulations during 1992 – 2013 (Burnham 1993). Harvest was monitored by the return of tags or lip tattoos during 1992 – 2010 and by genotyping during 2011 – 2014. Data including date and location of recovery and sex and estimated age were recorded for harvested bears and individuals killed in defense of life and property. Reported harvest rate in KB was relatively high during the 1990s (range: 6 – 17 bears / year) but significantly decreased by the mid-2000s (2 – 8 bears / year; Chapter 8), likely due to factors including changes in sea-ice conditions limiting hunter access by use of dog sleds to northeastern KB (E. Born, pers. obs.) and the implementation of a Greenlandic quota system in 2006. Greenland's reporting system also improved with the implementation of the quota (Chapter 8). Previous studies assumed that harvests of all marked bears were reported when natural survival was calculated and, therefore, the reporting rate r was interpreted as the proportion of mortality due to harvest (e.g., Taylor et al. 2005, 2008). However, more recently, genetic data suggested under-reporting of marked bears in the harvest, with decreases in reporting correlated with increasing marker age (Chapter 8).

Genetic Analyses

DNA Extraction – Dried biopsy samples, new and archived tissue 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 ~ 3mm² pieces of tissue with QIAGEN DNeasy Blood and Tissue Kits (<http://www.qiagen.com/>). Most biopsy darting samples consisted of a plug of a skin and subcutaneous tissue. This 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 these 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).

Marker Selection – 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 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.

Genotyping – 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 μ L of DNA per

reaction instead of the 3 μ L 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 which 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 one 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 also searched the dataset for genotype matches that seemed unlikely based on our field data. In each case, three extra markers were added to the genotypes to lower the probability of chance matches between individuals. The extra loci confirmed all of these matches. Once the genotyping and

error-checking was complete, we defined an individual for each unique 9-locus genotype.

Marker Power – 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 10.4). 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 Analyses

We analyzed joint live-capture and dead-recovery data from the KB subpopulation with the Burnham (1993) mark-recapture model, which combines the Cormack-Jolly-Seber (CJS) live-recapture model with the Brownie-Seber dead-recovery model to estimate survival (S), recapture (p), reporting (r), and fidelity (F) probabilities. With the Burnham model, live

recaptures are assumed to occur (relatively) instantaneously within the study area, whereas dead recoveries can occur year-round between live capture periods and may take place within or outside the live encounter study area. We assumed that harvests prior to April 15 occurred before the live encounter period in year t , (i.e., in year $t - 1$); post-April 15 harvests were considered to have occurred after the live encounter period (i.e., year t). This treatment of the data resulted in no instances in which a bear was recovered before being captured alive. We acknowledge that there was some temporal overlap of live recapture and dead recovery periods in KB, but for a long-lived species such as polar bears, the exact timing of harvest relative to the live capture sampling period is less important.

We analyzed data and constructed models in program MARK (White and Burnham 1999). We assembled capture histories from the live capture and dead recovery data and included harvest recoveries through 2013. Although ages were estimated with high resolution during the initial 1990s study period, there was uncertainty in field assessment of age during the 2012 – 2014 sampling frame, particularly with biopsy darting. Hence, we identified relatively coarse age classes (*cf.* Taylor et al. 2008, Peacock et al. 2013), including cubs-of-the-year (coy), yearlings (yrl), 2-year olds (2yr), and individuals age 3 and above (age 3+). Because KB is a small subpopulation, capture and recovery data were very sparse, and we identified a limited number of relatively simple candidate sub-model structures.

We hypothesized that survival would differ among age classes and included age structure in all candidate models; however, we constrained yearling survival equal to 2-year old survival due to sparse data. Because coy are fully dependent on their mothers for their survival, we assumed that survival would not vary between male and female coy. However, we expected that survival would differ between sexes for older age classes, largely due to the 2 : 1 male-to-female

sex ratio in the harvest, so we examined structures in which (1) S differed between sexes for age 3+ bears only and (2) S differed between sexes for yr1 / 2yr and age 3+ bears (additive effect of sex). Given the sparseness of the data, we did not examine year-to-year variability in S , or relationships between S and time-varying environmental covariates.

We examined seven sub-model structures for p (i.e., estimation is conditional on first capture). Estimates of p in the Burnham model reflect both the probability of an animal being located in the sampling area and thus available for recapture, and the probability of the animal being recaptured conditional on its presence in the sampling area (i.e., random temporary emigration is incorporated in p ; Burnham 1993). We hypothesized that female bears and dependent offspring (ages 0 and 1) may have a different p than independent male bears and evaluated models with this sex and age-class structure (family; sub-model structure 1). In addition, we suspected that search effort and sampling protocols may have differed between the two sampling epochs (1992-1997 vs. 2012-2014), so we considered structures with a temporal epoch effect (epoch; 2). Although the data were scant, we hypothesized that inter-annual variability in weather and sea-ice conditions may have resulted in p that varied significantly among years, so we also considered a fully time varying structure (time; 3). We considered structures with additive effects between (4) family and epoch and (5) family and time, as well as a structure including (6) an interactive effect between family and epoch. We also evaluated a null p sub-model (i.e., constant p ; 7).

Because some adult females in our 1990s sample were outfitted with satellite collars ($n = 12$) that may have assisted in locating them, we created a binary radio covariate indicating whether a bear was theoretically available for recapture with the assistance of radio telemetry. We applied the covariate for 2 years post-collaring during the 1990s sampling period, unless

there was evidence that the collar was physically removed from the bear. We included the radio covariate in all structures and coded dependent offspring such that they had the same covariate structure as their mother. Satellite collars were not used to locate bears during the 2012 – 2014 period. Some individuals were not successfully genotyped ($n = 25$) because either tissue samples were not located among the archives or the samples were inadequate to facilitate genotyping. These individuals had a reduced p during 2012 and 2013 (when physical capture and biopsy darting both occurred), but no probability of detection during 2014 (when bears only were sampled via biopsy darting). To reflect this, we created a binary ‘genotyped’ covariate (0 = successfully genotyped; 1 = not genotyped) and included it in all model structures for 2012 and 2013; for non-genotyped individuals, we fixed $p = 0$ in 2014. We also fixed $p = 0$ during 1998 – 2011, when there was no live recapture sampling.

The reporting (r) parameter represents the probability that a dead bear is identified and reported to authorities. Here, r reflected the proportion of mortality that can be attributed to reported harvest (including bears killed to protect life or property). We hypothesized that r would vary among age classes (yrl / 2yr and age 3+) and by sex for age 3+ individuals (sub-model structure 1) due to harvest regulations, including sex-selective harvest (2 males : 1 female). Because recovery data were sparse (≤ 5 total recoveries per year; typically 0 – 2 recoveries per year), we did not consider models with annual variation in r , but we created an alternative structure which included an additive effect for time period (pre-2006; 2006 - 2013) for age 3+ individuals to reflect the changes in harvest and improvements in the Greenlandic reporting system over the past decade (structure 2). There were no records of cubs-of-the-year marked in KB harvested during the first year post marking, so we fixed $r_{c_{oy}}$ to 0. Because only harvest data through 2013 were included in analyses, we fixed r to 0 for all age classes in 2014.

We hypothesized that polar bears may permanently emigrate from KB, based on the semi-discreteness of subpopulation boundaries (Taylor et al. 2001) and the spatial distribution of historical recapture and recovery data. Thus, we chose to estimate the F parameter, rather than assume that there was no permanent emigration and fix F to 1, as done in previous studies (e.g., Taylor et al. 2005, 2008, 2009). We considered structures in which (1) F was estimated as constant across all sex and age classes (constant) and (2) F was different for a combined class of coy, yr1 / 2yr, and age 3+ females vs. age 3+ males (3+ males).

We constructed the most generalized model (excluding individual covariates) and used the median \hat{c} method, as implemented in Program MARK, to estimate over-dispersion. Because results suggested the data were not significantly over-dispersed (i.e., \hat{c} was approximately 1), we proceeded with model selection via AIC_c . Given the relatively small set of candidate sub-model structures, we constructed all possible combinations of candidate sub-models.

We evaluated models via AIC_c and model-averaged parameters for models with $\Delta AIC_c < 4$ (Burnham and Anderson 2002), based on an initial sensitivity analysis. Our estimates of survival reflected harvest mortality, so we derived estimates of natural survival as $S + r * (1 - S)$ (following, e.g., Taylor et al. 2005, 2008, Peacock et al. 2013) and estimated variance via the delta method (following Taylor et al. 2008). This equation relies on several key 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), would lead to negative bias in estimates of natural survival. However, this derivation of natural survival also assumes that harvest mortality is completely additive, i.e., no bears that are harvested would otherwise die during a given interval. In contrast to under-reporting of marked bears in the harvest, a violation of the assumption of additive mortality would result in positive bias in

estimates of natural survival.

For highly supported models, we used a generalized Horvitz-Thompson estimator, $\hat{N} = \frac{n}{\hat{p}}$, where n is the number captured in group i and \hat{p} is the recapture probability for group i , to generate estimates of abundance by attribute group (e.g., family group status) for the yrl / 2yr and age 3+ classes. Because some coy were not marked during the 2012 – 2014 sampling period and estimates of n and p did not accurately reflect this age class, we incorporated coy by estimating the number of age 3+ females with coy litters via a Horvitz-Thompson estimator and multiplying by mean observed coy litter size. To obtain an overall estimate of abundance for KB by year, we summed individual estimates across groups. Following previous work (e.g., Taylor et al. 2005, 2008, Peacock et al. 2013), we estimated variances for total abundance estimates and incorporated variances and covariances (calculated in MARK) as well as variance of mean litter sizes via the delta method (Seber 1982, Powell 2007) using R (R Core Team 2015) package `emdbook` (Bolker 2016). We model-averaged estimates of total abundance using model weights for recapture probabilities and variances obtained with the delta method. We calculated mean overall estimates of abundance by sampling epoch and estimated variance using the delta method. We excluded 1993 – 1994 and 2012 from these mean estimates given the initial expansions of the sampling frame between 1992 and 1995 and the long interval without live recaptures preceding 2012, respectively (i.e., estimation of subpopulation size in 2012 was based on estimated recapture rates of bears marked during the 1990s applied to newly encountered bears in 2012).

10.3. Results

We recorded a total of 277 initial captures, 89 recaptures, and 24 dead recoveries over the

course of the 23-year study period (Table 10.1). Markedly more bears were captured in the eastern regions of KB (i.e., off Humboldt Glacier in Northwest Greenland) during 2012 – 2014 than during the 1990s (Figures 10.2 and 10.5). Capture data were particularly sparse during the 1990s, although sampling in 1995 yielded significantly more captures than other years in the 1990s (Table 10.1). Similarly, very few bears were recovered via the harvest during the 2000s (Table 10.1). Notably, no males initially marked in KB during the 1990s were recaptured during 2012 – 2014, and only one male marked in the 1990s was reported in the harvest after 2002. Although no COY were sampled in 2014 (all sampling was conducted via biopsy darting), we observed a total of 23 COY with their mothers that year. In addition, 3 COY with their mothers were not biopsy darted in both 2012 and 2013. Mean observed COY litter size during 2012 – 2014 was 1.60 (SD: 0.5).

The most highly supported models included an additive effect of sex for the yr1 / 2yr and 3+ age classes for S and a temporal effect (break at 2006) for r (Table 10.2). Although there was not clear support for specific structures for modeling p , complex (e.g., fully time-varying) structures for p were not supported in model selection, which was not surprising given the sparseness of the data. For model-averaging, we included 12 of 56 total models (cumulative model weight = 0.76).

Estimates of total survival of males were markedly lower than females for both the yr1 / 2yr and 3+ age classes, although we note that the additive effect in S was shared across age classes and not estimated separately for yr1/2yr vs. 3+ bears (Table 10.3). This pattern was also evident in estimates of unharvested survival (yr1 / 2yr females: 0.74, SE: 0.15; yr1 / 2yr males: 0.54, SE: 0.17; age 3+ females, 2006 – 2013: 0.96, SE: 0.04 and 3+ males, 2006 – 2013: 0.88, 0.05). As hypothesized, recent (2006 - 2013) estimates of r were less than 1992-2005 values,

although r did not significantly differ among age and sex classes (Table 10.3). Estimates of F suggest relatively strong fidelity to the springtime study area for females and dependent bears (F : 0.98, SE: 0.04) as well as age 3+ males (F : 0.96, SE: 0.07).

Annual estimates of abundance largely reflected the variability in sample sizes among years (e.g., 1995; Table 10.4, *cf.* Table 10.1). The estimated mean total abundance of the KB subpopulation during the 1995 – 1997 period was 224 (SE: 40; 95% CI: 145 – 303). The estimated mean total abundance for 2013 – 2014 was 357 (92; 221 – 493).

10.4. Discussion

We used a combination of physical and genetic mark-recapture techniques, including live recaptures and dead recoveries, to estimate demographic parameters of the Kane Basin polar bear subpopulation over a 23-year study period. Our estimate of abundance from the 1990s (224, 95% CI 145 – 303; averaged over 1995 – 1997) was consistent with previous analyses (164, averaged over 1994 - 1997; Taylor et al. 2008). Although the 2010s point estimate is ~36% greater than the 1990s estimate of Taylor et al. (2008), from the 1990s; this difference is largely attributable to our decision to derive a mean estimate of abundance from only 1995 – 1997. The sampling frame expanded during the 1990s, progressing northward and eastward in incremental steps such that, in the initial years, only a portion of KB was surveyed (Figure 10.2). Hence, we calculated mean abundance estimates by epoch only during periods when sampling was consistent and the sampling frames were generally comparable (1990s: 1995 – 1997). By contrast, Taylor et al.'s (2008) estimate was calculated as the mean estimated from 1994 – 1997; including 1994 in our estimate would reduce our point estimate from 224 to 198.

The 2013 – 2014 estimate of abundance suggests 357 (221 – 493) bears currently use KB

in springtime (i.e., the Kane Basin super-population; Kendall et al. 1997), and the current point estimate is higher than the historical estimate. Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of KB abundance for 2013-2014 and the estimate for 1995-1997 was approximately 133 bears (standard deviation of the difference \approx 80 bears), with 95% of the sampling distribution suggesting that population change between the two time periods could have been positive. This result suggests a stable to increasing subpopulation and is consistent with data on movements (Chapter 9), reproductive output (Chapter 12), and body condition (Chapter 13), suggesting that Kane Basin is currently a healthy subpopulation. However, we encourage caution in interpretation. We attempted to mitigate the impacts of apparent changes in sampling frames, particularly during the 1990s, by excluding 1993 and 1994 from our mean estimate of abundance during the 1990s. However, we were unable to address potential changes in survey effort between the 1995 – 1997 and 2012 – 2014 epochs.

Sampling occurred in the eastern regions of the KB subpopulation (i.e., near the Humboldt Glacier) during 1995 – 1997 and 2012 – 2014. When the eastern parts of Kane Basin (i.e., the areas east of the mid-sector line in the Nares Strait-Kane Basin area off the Humboldt Glacier) were surveyed in 1994 and 1995, only few signs of polar bear activity (i.e., tracks) were observed there and consequently only a few bears were tagged (0 in 1994 and 4 in 1995; E. Born, pers. obs., Taylor et al. 2001). Similarly, no bears were found and tagged there in 1996 and only 3 in 1997 (Taylor et al. 2001). The apparent very low densities of polar bears in eastern KB was assumed to reflect a long-term avoidance response because eastern KB has been hunted relatively intensively by hunters from the Qaanaaq region and in particular after it no longer became permitted for Greenland hunters to hunt polar bears in Canadian territory in the late

1960s. The presence of ringed seals in eastern KB was noted during the surveys in the 1990s and it was concluded that these areas were favourable habitat for polar bears (E. Born, pers. obs., Taylor et al. 2001).

However, it cannot be precluded that, this difference between the 1990s and the 2000s in the spatial distribution of bears, to a certain extent reflect some differences in sampling protocols, including increased survey intensity near the Humboldt Glacier and more uniform distribution of effort during 2012 – 2014. However, during both periods relative allocation of survey effort to a certain extent was decided based on assumptions of what was suitable polar bear habitat (i.e., areas with anticipated polar bear occurrence and/or areas with observed signs of polar bear habitat). In the 2000s large areas in central and southern KB with relatively open pack ice were not surveyed although satellite telemetry (Chapter 2) and aerial surveys (Heide-Jørgensen et al. 2013) indicate the presence of polar bears in this habitat albeit likely few. Whereas in the 1990s when the spring sea ice in KB was more consolidated (E. Born, pers. obs; Chapter 9) areas with little or no signs of polar bear activity in the eastern parts were surveyed less intensively.

It should be mentioned that sea-ice dynamics in Kane Basin also have changed since the 1990s (Chapter 9), limiting access of hunters from Greenland to the eastern parts of the region (Born et al. 2011) possibly resulting in an increased occurrence of polar bears in this area. Hence, we hypothesize that these differences in sea-ice dynamics and associated change in hunting pressure have led to the apparent shift in the distribution of bears toward the eastern parts of the Nares Strait-Kane Basin region.

We note that the Horvitz-Thompson estimator used to generate estimates of abundance in Kane Basin yielded biologically implausible rates of growth from 1994 to 1995 (119 – 318),

1995 to 1996 (318 – 189), and 2012 to 2013 (221 – 328; Table 10.4). Although the apparent changes in sampling frame (and thus the definition of the effective study population) from 1994 to 1995 may contribute to this finding in part, this result is primarily an artifact of the estimator itself. Horvitz-Thompson (H-T) estimators are calculated as $\hat{N} = \frac{n}{p}$ for each group (e.g., age class and sex), and the total abundance estimate is derived by summing estimates across all groups. As such, H-T estimators are sensitive to sample size, particularly if recapture probabilities are estimated as temporal constants. Because data for this analysis were very sparse and models specifying inter-annual variation in estimates of recapture probability were not supported, our Horvitz-Thompson estimates of abundance are influenced by variation in annual sample size of captured bears (n). We attempted to address this issue by integrating annual random effects for estimating recapture probability with complementary Markov Chain Monte Carlo (MCMC) analyses, but this approach did not resolve the issue. Given this limitation, we do not recommend interpreting inter-annual variation in estimates of abundance, and suggest that the larger estimate of abundance in 2013-2014 may be partially attributed to larger sample sizes in those years, compared to 1995-1997.

Our estimates of survival for dependent bears and age 3+ females are consistent with previous work in Kane Basin (Taylor et al. 2008), and estimates of unharvested survival rates of independent females appear capable of supporting positive subpopulation growth (Regehr et al. 2015). Our estimates of age 3+ male survival (present analysis: age 3+ males, 0.87, SE 0.06) are lower than previous work in KB (Taylor et al. 2008: age 5+ males, 0.96, SE: 0.05) but consistent with estimates of adult male survival derived in some other studies (e.g., Stirling et al. 2011, Peacock et al. 2012). We further note that data on males were particularly scant in this study (Table 10.1): no male bears initially marked in the 1990s were subsequently recaptured in the

2000s, and very few individuals were reported in the harvest over the past 15 years. We are uncertain if the disparity in estimates of male survival between our study and Taylor et al. (2008) reflects differences in modeling approaches, such as the broader age class designations in the present study and consideration of different model structures (unlike Taylor et al. [2008], we did not consider a structure in which S was estimated as constant between the sexes for age 3+ individuals); our inclusion of longer-term data; a disproportionate impact of the changing environment on males; reduced fidelity of males to the study area that was not effectively captured by our model-based estimates of the fidelity (F) parameter given the paucity of recovery data; under-reporting of male bears in the harvest; or some combination thereof.

Although we calculated estimates of natural survival following previous studies (e.g., Taylor et al. 2005), we note that the formula used to derive these estimates [$S + r * (1 - S)$] makes simplifying assumptions that can introduce bias into estimates of unharvested S under some conditions, although the impacts of using this vs. an alternative equation is likely minimal for KB data due to high estimates of S and low estimates of r . The documented under-reporting of marked bears in the harvest leads to an underestimation of natural survival, but this may be offset, to some extent, by a likely violation of the assumption that harvest mortality is completely additive. For example, Taylor et al. (2008) estimated natural survival for both adult females and males to be 0.997, meaning that virtually all mortality of adult (age 5+) bears in Kane Basin during 1992 - 1998 resulted from harvest, and <1 in 300 adult bears would die annually in the absence of harvest. Although Taylor et al. (2008) did not report their estimates of r , back calculating from survival rates in their Table 3 yields unrealistically high estimates of $r = 0.91$ for adult females and $r = 0.93$ for adult males, so their estimates of natural and harvest mortality should be treated cautiously.

Changing sea-ice conditions, a reduction in accessibility of Kane Basin to hunters from Canada and (in the Humboldt Glacier region) Greenland, and the implementation of a quota in Greenland have contributed to a net reduction in harvest since the 1990s (Chapter 8). This decline is reflected in estimates of the reporting parameter, as r is estimated lower for the period from 2006 – 2014 than 1992 – 2005 (Table 10.3). We note, however, that under-reporting of harvest, which anecdotally appears to increase with greater marker age (Chapter 8), also may contribute to lower estimates of r during 2006 – 2013, especially given the 14-year interval without live recaptures.

Despite the sparseness of the data and the unusual study design (6-year and 3-year live capture sessions connected by a 14-year period with dead recoveries only), we were able to generate estimates of F that seem biologically realistic (age 3+ males: 0.96; females and dependent bears: 0.98): polar bears show strong seasonal fidelity to the region in which they were captured, but a small proportion of individuals permanently emigrate to other subpopulations. These estimates appear consistent with findings from satellite telemetry data and capture records (Taylor et al. 2001, Chapters 2 and 9).

Our study indicates that there has been no decline in the size of the KB subpopulation. These findings are in accordance with TEK (Born et al. 2011).

Using estimates of subpopulation size in KB and vital parameters from the 1990s, York et al. (2016) modeled a decline in KB with an estimated subpopulation size of zero in 2013. According to York et al. (2016:9,18) the projected decline is consistent with TEK. It is mentioned that KB has been subject to chronic long-term overharvest and would not persist if it did not receive immigrants from adjacent subpopulations (Ibid.). As basis for the TEK information York et al. (2016) cite COSEWIC (2008) and M. Taylor (pers. comm. 1986-2008) in

the text, and COSEWIC (2008), CWS (2009) and PBTC (2014) in their table 2b but not a comprehensive TEK study in which NW Greenland polar bear hunters were interviewed (Born et al. 2011).

During this interview survey in Greenland experienced polar bear hunters who had been hunting in Kane Basin were of the opinion that polar bears in this region had expanded their range. Previously the hunters had to travel north to Washington Land (*ca.* 80° N) to find bears, whereas nowadays they only have to go as far as Inglefield Land (*ca.* 78° 30' N) to hunt polar bears in the eastern Nares Strait-Kane Basin region. Their reason for this was because “the bears have come closer” (Born et al. 2011:75,79). It was mentioned that previously polar bears were scarce in the eastern Kane Basin area (i.e., in front of the Humboldt Glacier) but now had expanded their range from Ellesmere Island eastward to the Nares Strait-Kane Basin region (ibid:80). Some of the interviewees were of the opinion that this change represented an increase in the number of polar bears. It was also mentioned that the hunting pressure in Kane Basin had decreased because poor sea-ice conditions (i.e., lack of dense sea ice) had made travels with dog sleds north more difficult (Born et al. 2011). Hence, information from experienced polar bear hunters in NW Greenland indicates that the KB-polar bear subpopulation has expanded its distribution area and increased in size which is in accordance with our study.

This study and concurrent research in the neighboring Baffin Bay subpopulation (Chapter 5) represent the first attempts to implement genetic mark-recapture for polar bears at a subpopulation-wide scale. The ability to successfully genotype bears from archived tissue and samples obtained via remote biopsy darting, combined with the ability to analyze data in well-established mark-recapture models, suggest that this approach is a promising tool for future polar bear inventories. However, there are some limitations of the technique including a generalized

age structure as bears are identified from the air. We opted for a conservative approach when designating age classes because there was inherent uncertainty in estimating age class remotely rather than aging via physical examination or with annular rings from an extracted tooth. However, our ability to accurately classify bears by age-class remotely (Chapter 5) suggests that future studies may be able to increase the resolution of age classification for obtaining estimates of survival (and other parameters), thus enhancing the utility of the technique.

10.5. Literature Cited

- Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. *Marine Mammal Science* 25:35–52.
- Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.
- Atwood, T., B. G. Marcot, D. C. Douglas, S. C. Amstrup, K. D. Rode, G. M. Durner, and J. E. Bromaghin. 2015. Evaluating and ranking threats to the long-term persistence of polar bears. U.S. Geological Survey Open-File Report 201-1254. U.S. Geological Survey, Reston, Virginia, USA.
- Bolker, B. 2016. Ecological Models and Data in R. R package version 1.3.9.
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusulanum Press, University of Copenhagen, Copenhagen, Denmark.
- Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. *Ecological*

- Applications 25:634–651.
- Burnham, K. P. 1993. A theory for combined analysis of ring recovery and recapture data. Pages 199–213 *in* J. D. Lebreton and P. M. North, editors. *Marked individuals in the study of bird populations*. Birkhauser-Verlag, Basel, Switzerland.
- Burnham, K. P., and D. R. Anderson. 2002. *Model selection and multimodel inference: a practical information–theoretic approach*. Springer, New York, New York, USA.
- Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. *Ursus* 10:449–453.
- Comiso, J. C., C. L. Parkinson, R. Gertsen and L. Stock. 2008. Accelerated decline in the Arctic sea ice cover. *Geophysical Research Letters* 35:L01703.
- COSEWIC [Committee on the Status of Endangered Wildlife in Canada]. 2008. COSEWIC assessment and update status report on the polar bear *Ursus maritimus* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, vii + 75 pp.
- CWS [Canadian Wildlife Service]. 2009. Nunavut Consultation Report. Consultations on the Proposed Listing of the Polar Bear as Special Concern under the Species at Risk Act. Conducted February–April 2009, Iqaluit, Nunavut, 247 pp.
- Heide-Jørgensen, M. P., L. M. Burt, R. G. Hansen, N. H. Nielsen, M. Rasmussen, S. Fossette, and H. Stern. 2013. The significance of the North Water Polynya to Arctic top predators. *Ambio* 42:596–610.
- Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. *Ursus* 24:148–163.
- Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. *Journal of Wildlife Management* 73:3–17.
- Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock’s robust design. *Ecology* 78:563–578.
- Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. *Ecological Applications* 18:S97–S125.
- Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D.

- Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology* 29:724–737.
- Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. *Ecological Applications* 26:1302–1320.
- Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. *Polar Biology* 38:1713–1725.
- Paetkau, D. 2003. An empirical exploration of data quality in DNA-based population inventories. *Molecular Ecology* 12:1375–1387.
- Paetkau, D., S. C. Amstrup, E. W. Born, W. Calvert, A. E. Derocher, G. W. Garner, F. Messier, I. Stirling, M. K. Taylor, Ø. Wiig, and C. Strobeck. 1999. Genetic structure of the world's polar bear populations. *Molecular Ecology* 8:1571–1584.
- Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. *Marine Mammal Science* 30:169–183.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2015. Summary of polar bear population status per 2014: Status table. <<http://pbsg.npolar.no/export/sites/pbsg/en/docs/status-table-2014.pdf>>. Accessed 4 May 2016.
- PBTC [Canadian Polar Bear Technical Committee]. 2014. 2014 Polar Bear Technical Committee Status Table, unpublished data.
- Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. *Journal of Wildlife Management* 77:463–476.
- Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. *Arctic* 65:391–400.
- Powell, L. 2007. Approximating variance of demographic parameters using the delta method: A

- reference for avian biologists. *Condor* 109:949–954.
- R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org/>.
- Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.
- Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. *Population Ecology* 54:3–18.
- Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. *Global Change Biology* 20:76–88.
- Sahanatien, V., and A. E. Derocher. 2012. Monitoring sea ice habitat fragmentation for polar bear conservation. *Animal Conservation* 15:397–406.
- Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Macmillan, New York, New York, USA.
- Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. *Marine Mammal Science* 32:181–201.
- Stirling, I., C. Spencer, and D. Andriashek. 1989. Immobilization of polar bears (*Ursus maritimus*) with Telazol[®] in the Canadian Arctic. *Journal of Wildlife Diseases* 25:159–168.
- Stirling, I., T. L. McDonald, E. S. Richardson, E. V. Regehr, and S. C. Amstrup. 2011. Polar bear population status in the northern Beaufort Sea, Canada, 1971-2006. *Ecological Applications* 21:859–876.
- Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. *Global Change Biology* 18:2694–2706.
- Stroeve, J. C., M. C. Serreze, M. M. Holland, J. E. Kay, J. Malanik, and A. P. Barrett. 2012. The Arctic’s rapidly shrinking sea ice cover: A research synthesis. *Climatic Change* 110:1005–1027.

- Taylor, M. K., S. Akeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. *Canadian Journal of Zoology* 79:690–709.
- Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. *Arctic* 58:203–214.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. *Bird Study* 46:S120–S139.
- York, J., M. Dowsley, A. Cornwell, M. Kuc, and M. Taylor. 2016. Demographic and traditional knowledge perspectives on the current status of Canadian polar bear subpopulations. *Ecology and Evolution* 6:2897-2924.

Table 10.1. Summary table of live captures and dead recoveries during the mark-recapture study of the Kane Basin polar bear subpopulation in Nunavut, Canada, and Greenland, 1992 – 2014. 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		
	Coy	Yrl / 2yr	3+	Coy	Yrl / 2yr	3+	Yrl / 2yr	3+	Yrl / 2yr	3+	Coy	Yrl / 2yr	3+	Coy	Yrl / 2yr	3+
1992	4	0	7	2	0	3					0	0	1	0	0	0
1993	1	3	6	3	1	8	0	2	0	0	0	0	0	0	0	0
1994	2	0	9	3	0	4	1	3	2	2	0	0	0	0	0	0
1995	12	3	21	5	2	13	0	7	0	6	0	0	1	0	0	1
1996	5	2	8	2	2	4	1	7	0	5	0	0	0	0	0	1
1997	0	4	4	3	1	3	1	8	0	8	0	1	1	0	0	2
1998												0	3		0	2
1999													2			2
2000													0			0
2001													1			0
2002													0			1
2003													2			0
2004													1			0
2005													0			0
2006													0			0
2007													0			0
2008													0			0
2009													0			0
2010													0			1
2011													0			0
2012	2	3	19	1	4	11		2		0	0	0	1	0	0	0
2013	6	4	20	2	2	19	0	9	1	3	0	0	0	0	0	0
2014	0	2	21	0	1	10	2	12	0	7	0	0	0	0	0	0
Totals	32	21	115	21	13	75	5	50	3	31	0	1	13	0	0	10

Table 10.2. Model selection results ($< \Delta AIC_c$ 4) from analysis of mark-recapture-recovery data from the Kane Basin polar bear subpopulation, 1992 – 2014. Coy = cubs of the year. Yrl = yearlings and 2-year olds. 3+ = bears aged 3 and older. For p , family = females / dependent bears and independent males (2 age / sex classes); and epoch = sampling period (1992 – 1997; 2012 – 2014). For r , time = 1992 – 2005 and 2006 – 2013.

Model Structures				Parameters	ΔAIC_c	AICc	
S	p	r	F			Weights	Deviance
coy, yrl, 3+(add sex with yrl)	Family	yrl, 3+(sex + time)	Constant	13	0	0.23	723.71
coy, yrl, 3+(add sex with yrl)	Family	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	14	1.22	0.13	722.76
coy, yrl, 3+(add sex with yrl)	family + epoch	yrl, 3+(sex + time)	Constant	14	1.23	0.13	722.77
coy, yrl, 3+(add sex with yrl)	Constant	yrl, 3+(sex + time)	Constant	12	1.88	0.09	727.74
coy, yrl, 3+(sex)	Family	yrl, 3+(sex + time)	Constant	13	2.19	0.08	725.90
coy, yrl, 3+(add sex with yrl)	family + epoch	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	15	2.57	0.06	721.94
coy, yrl, 3+(add sex with yrl)	family * epoch	yrl, 3+(sex + time)	Constant	15	2.62	0.06	721.99
coy, yrl, 3+(sex)	Family	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	14	2.83	0.06	724.38
coy, yrl, 3+(sex)	Constant	yrl, 3+(sex + time)	Constant	12	3.06	0.05	728.92
coy, yrl, 3+(add sex with yrl)	Epoch	yrl, 3+(sex + time)	Constant	13	3.37	0.04	727.08
coy, yrl, 3+(sex)	family + epoch	yrl, 3+(sex + time)	Constant	14	3.45	0.04	724.99
coy, yrl, 3+(add sex with yrl)	Constant	yrl, 3+(sex + time)	coy yrl 3+ F, 3+ M	13	3.90	0.03	727.61

Table 10.3. Model averaged ($<\Delta 4$ AICc) parameter estimates for the Kane Basin polar bear subpopulation obtained from mark-recapture study, 1992 – 2014.

Parameter	Class	Estimate (SE)
Total Survival (<i>S</i>)		
	Cubs of the year	0.45 (0.15)
	Yearlings / 2-year old females	0.73 (0.13)
	Yearlings / 2-year old males	0.52 (0.17)
	3+ females	0.95 (0.04)
	3+ males	0.87 (0.06)
Reporting (<i>r</i>)		
	Yearlings / 2-year olds	0.04 (0.04)
	3+ females, 1992 – 2005	0.42 (0.26)
	3+ females, 2006 - 2013	0.09 (0.08)
	3+ males, 1992 – 2005	0.32 (0.12)
	3+ males, 2006 – 2013	0.06 (0.05)
Fidelity (<i>F</i>)		
	Cubs of the year, yearlings, 2-year olds, and 3+ females	0.98 (0.04)
	3+ males	0.96 (0.07)

Table 10.4. Model averaged ($<\Delta 4$ AICc) estimates of abundance ($\hat{N} \pm SE$; [95% Confidence Interval]) of the Kane Basin polar bear subpopulation from mark-recapture study, 1992 – 2014.

1993	1994	1995	1996	1997	2012	2013	2014
120 ± 19 (83-156)	119 ± 21 (77-160)	318 ± 53 (214-429)	189 ± 36 (119-259)	164 ± 28 (110-218)	221 ± 41 (141-301)	328 ± 60 (211-445)	385 ± 78 (233-537)

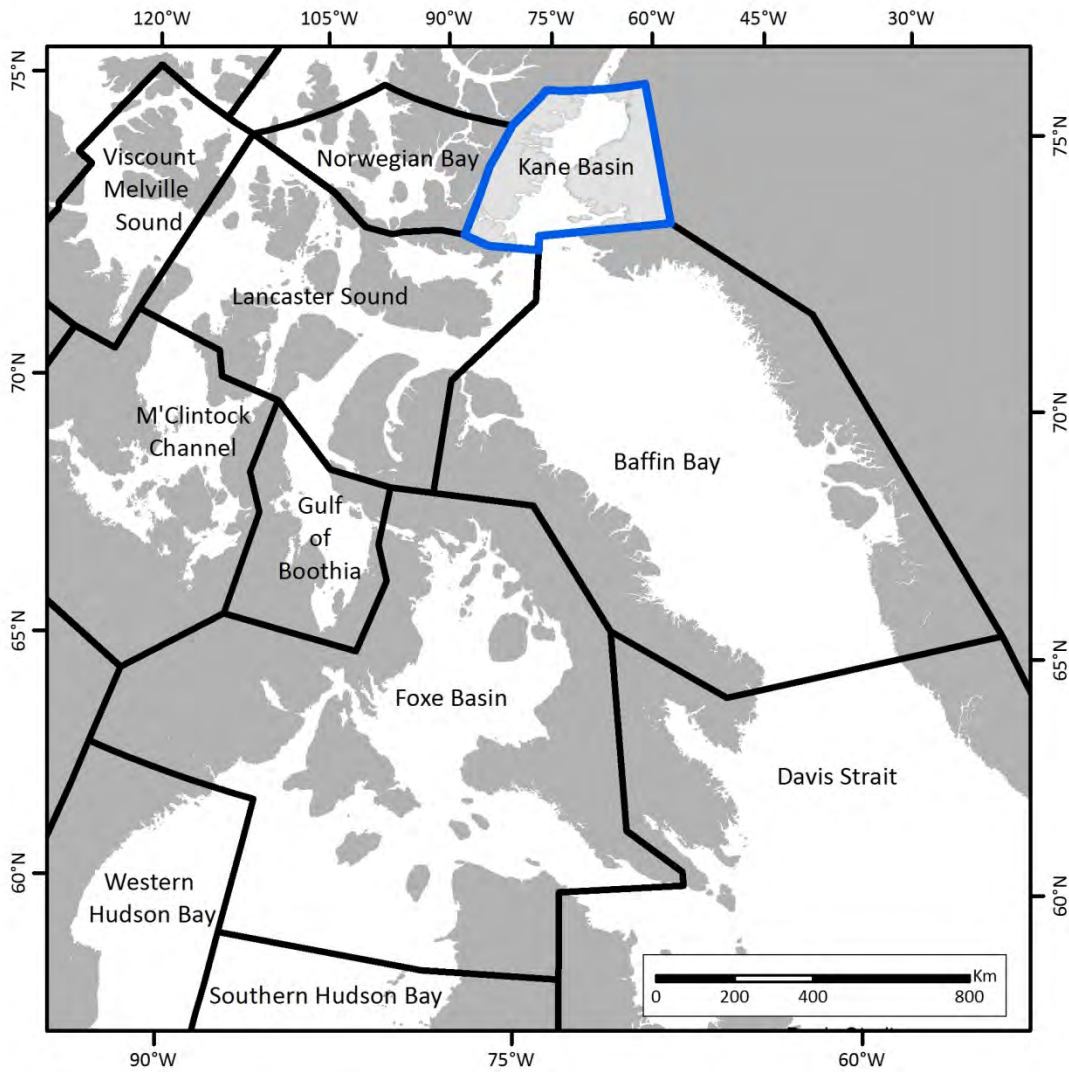


Figure 10.1. The Kane Basin polar bear subpopulation is located between Nunavut, Canada and Greenland and is regarded as belonging to the Arctic Archipelago region.

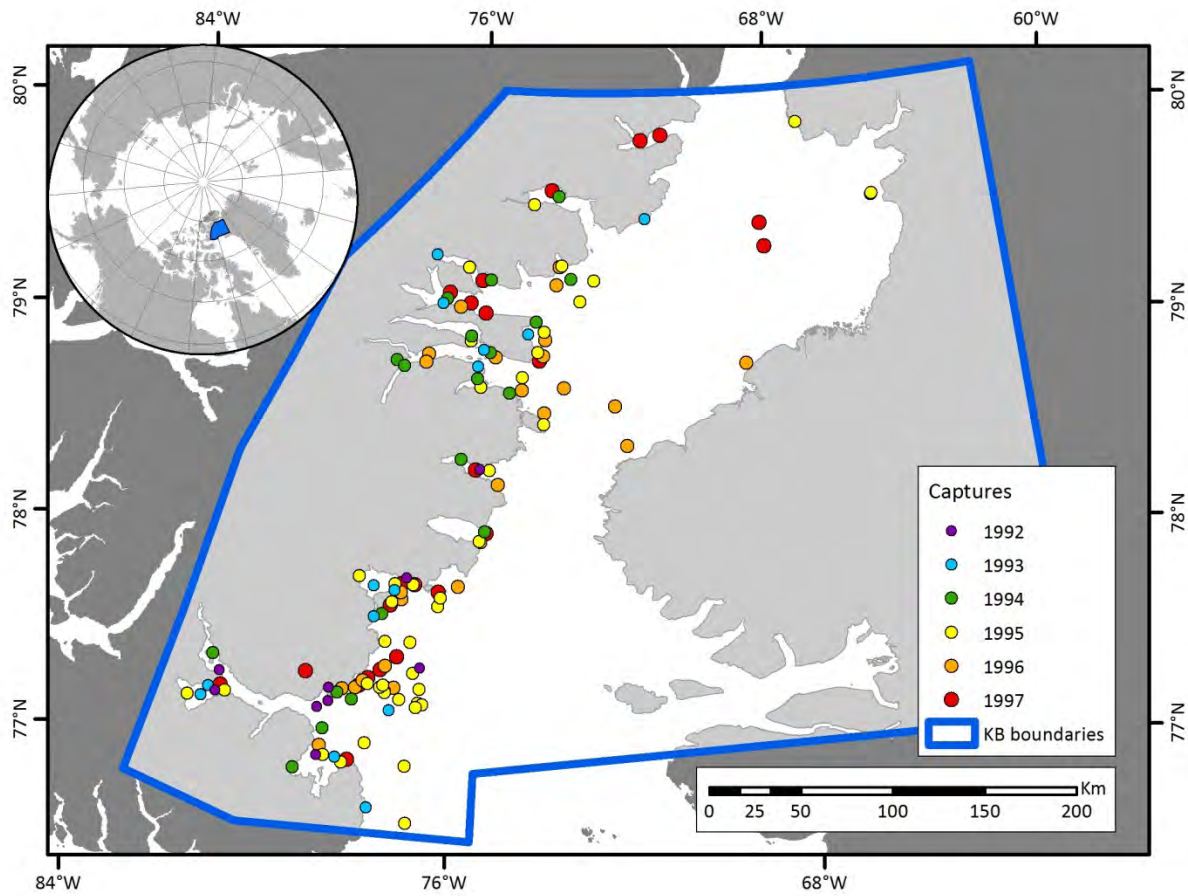


Figure 10.2. Locations of polar bears captured in the Kane Basin subpopulation during springtime, 1993 – 1995 and 1997. Kane Basin is highlighted in blue in the inset.

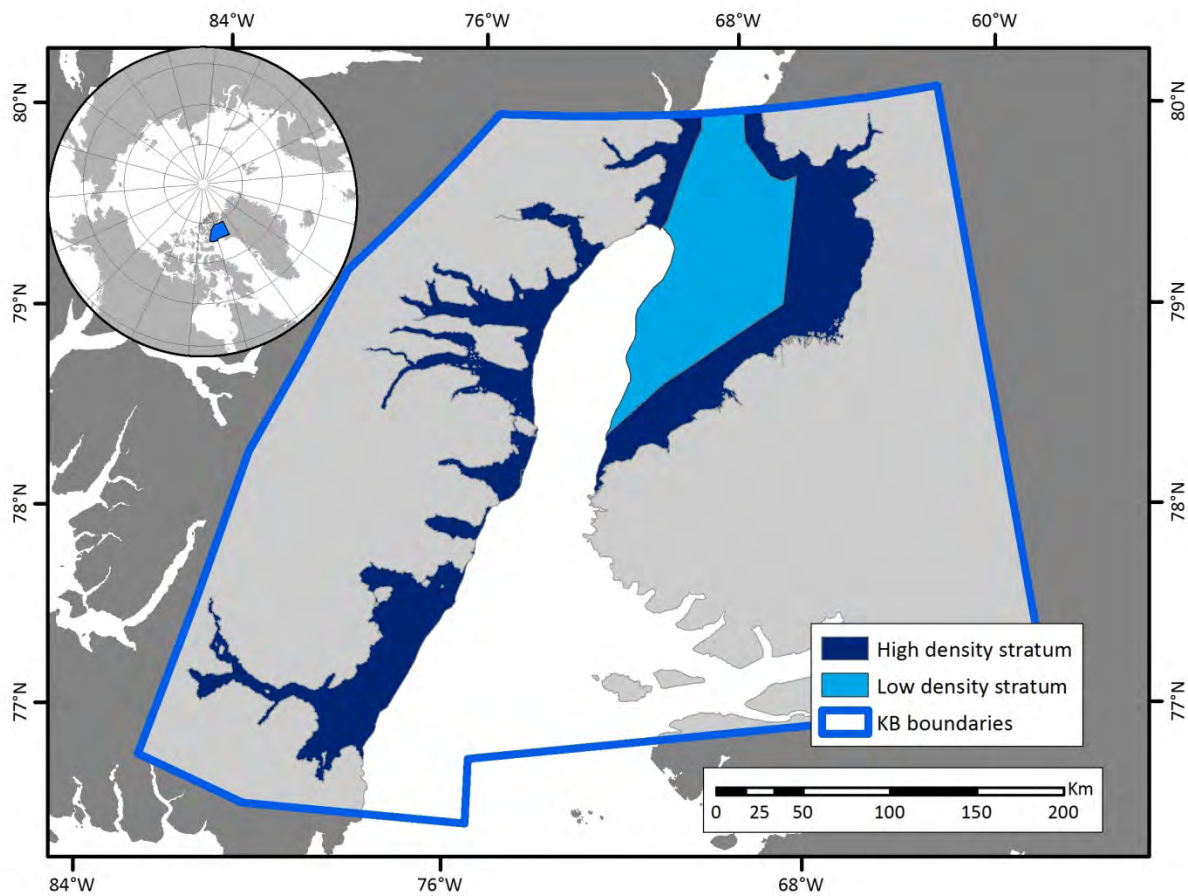


Figure 10.3. Sampling strata for genetic mark-recapture and aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

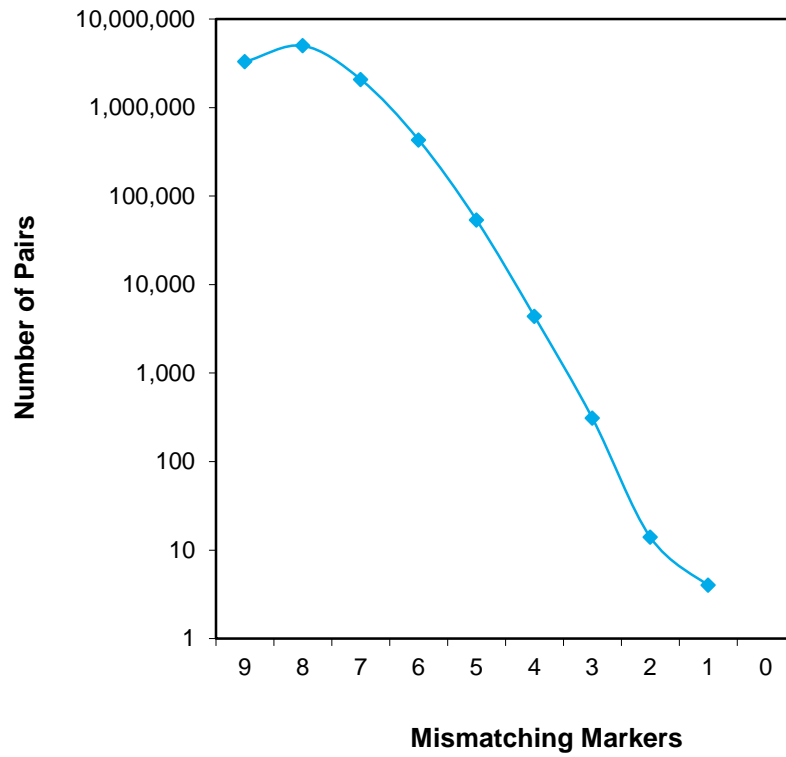


Figure 10.4. 9-locus mismatch distribution for 4,657 polar bears from Nunavut and the Greenland side of the Baffin Bay and Kane Basin polar bear subpopulations.

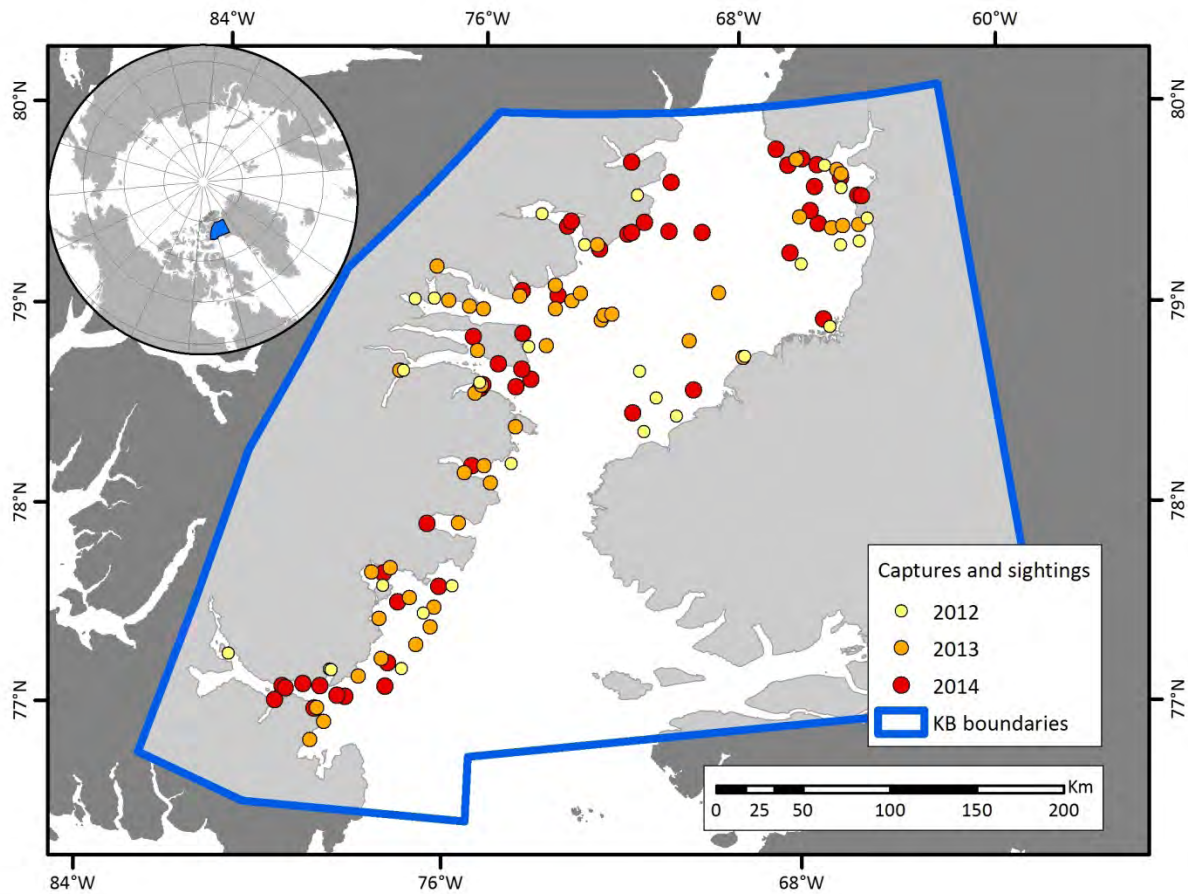


Figure 10.5. Locations of polar bears sighted in Kane Basin during research in April and May, 2012 – 2014. Kane Basin is highlighted in blue in the inset. The North Water polynya varied among years, but in general, included the south-central portion of the subpopulation in all years. We did not sample sea ice in southeastern Kane Basin due to logistical constraints presented by the polynya and anticipated low densities.

CHAPTER 11

AERIAL SURVEY OF POLAR BEARS IN KANE BASIN

KEY FINDINGS

- The estimate of abundance based on the springtime 2014 aerial survey in KB was 206 bears (95% lognormal CI: 83 - 510). However, due to insufficient coverage of offshore polar bear habitat this estimate is likely negatively biased.
 - Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of total abundance 2013-2014 from the MR study (357 bears, 95% CI = 221 – 493) and the aerial survey estimate was approximately 151 bears (standard deviation of the difference \approx 127 bears), with 88% of the sampling distribution suggesting that the difference was positive (i.e., that the MR estimate was at least one bear larger than the aerial survey estimate).
 - Differences between MR and aerial survey point estimates in KB require caution when comparing results from different techniques conducted during springtime. Aerial surveys yield a snapshot estimate of abundance, whereas MR generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period. We suggest that the MR estimate is appropriate for use in management.
 - As shown also in other areas of the Arctic aerial surveys provide a useful tool for inventorying polar bear subpopulations and the method has been used on even larger subpopulations than KB in remote areas (e.g. the Barents Sea).
 - The springtime aerial survey was successfully implemented due to the small geographic area and a period of good weather, but precision could be improved by increasing survey effort to better estimate the detection function and by ensuring that the entire range of the subpopulation is covered.
 - Aerial surveys of polar bears that also range in areas with offshore loose drift ice and open water, like Kane Basin, should be conducted from fixed-winged aircraft with a longer endurance than the single-engine helicopter used in the present study. This allows for offshore polar bear habitat to be monitored and will result in a more accurate estimate of abundance.
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11.1. Introduction

Physical mark-recapture has formed the basis for demographic studies of polar bears throughout the North American Arctic (e.g., Taylor et al. 2005, 2008, Peacock et al. 2013).

Capture-based research has generated information on abundance, vital rates, and harvest management, and facilitated a variety of other studies, including assessments of body condition (e.g., Rode et al. 2012), movements, habitat use and spatial ecology (e.g., Durner et al. 2009, Cherry et al. 2013), and diet (e.g., Thiemann et al. 2008). Over the past decade, however, jurisdictions have invested substantial resources in the development and implementation of less invasive monitoring techniques, in part to better address social concerns regarding wildlife handling and immobilization. Genetic mark-recapture, one such alternative method, has been used to estimate the number of polar bears using whale carcasses in Alaska (Herreman and Peacock 2013) and, more recently, to estimate the abundance and associated vital rates for the Baffin Bay and Kane Basin subpopulations (see Chapters 5 and 10, respectively). Aerial surveys also have been widely implemented, including studies conducted over land in seasonally ice-free subpopulations (Stapleton et al. 2014, 2016, Obbard et al. 2015) and over land and sea ice in the Barents Sea (Aars et al. 2009). Aerial surveys yield less detailed information on sex, age, body condition, and vital rates than both physical and genetic mark-recapture methods, but they can enable more frequent monitoring, an important consideration in the face of a rapidly changing Arctic.

Our objective was to evaluate the feasibility of estimating abundance with an aerial survey flown over springtime sea ice in the Kane Basin (KB) subpopulation. We designed and implemented the aerial survey to be conducted alongside a concurrent mark-recapture study in the KB subpopulation during 2014. This protocol ensured consistency in the sampling frames and study periods. It also allowed us to derive independent estimates of abundance from the two techniques, enabling us to directly compare and assess the results of the 2 methods. This important step is necessary to properly integrate population estimates derived from different

survey techniques. Although research elsewhere in has facilitated broad comparisons between mark-recapture and aerial survey methods (Western Hudson Bay – Stapleton et al. 2014, Lunn et al. 2016; Southern Hudson Bay – Obbard 2008, Obbard et al. 2015), the work in Kane Basin represents the first study in which an aerial survey was designed and implemented with a sampling frame identical to a simultaneous mark-recapture study. Aerial surveys yield snapshot estimates of abundance (i.e., the number of bears occupying the survey area during the study period; Buckland et al. 2001), whereas mark-recapture generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period (including individuals that are currently outside the survey area due to temporary emigration; Kendall et al. 1997). Because there is a lack of geographic closure among polar bear subpopulations such that they are only partially discrete (Taylor et al. 2001, Chapter 9), we hypothesized that our aerial survey-based estimate would be smaller than our mark-recapture-based estimate.

11.2. Materials and Methods

Study Area

The KB subpopulation covers ~150,000 km² and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (boundaries evaluated in Taylor et al. 2001; Figure 10.1). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* 11.2). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq areas). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the Baffin

Bay (BB) and Lancaster Sound (LS) subpopulations, and to the west by Norwegian Bay (NW; PBSG 2010). The KB subpopulation is regarded as belonging to the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, sea-ice conditions have changed markedly in the Kane Basin region in recent decades (Born et al. 2011; Figure 9.2). KB is partially connected to neighboring subpopulations; particularly notable interchange occurs with BB and LS. The North Water polynya, a large area of open water in northern Baffin Bay and southern Smith Sound, is a significant regional geographic feature that exhibits substantial intra- and inter-annual variability in spatial extent and is thought to form a barrier between KB and BB – LS.

Field Sampling

Using a helicopter (Bell 206 LongRanger), we implemented a line-transect aerial survey over springtime sea ice in the KB subpopulation during 28 April – 12 May 2014. To efficiently allocate effort and ensure that the study area was sampled as comprehensively as possible, we stratified the subpopulation into high- and low-density areas based on observations of polar bears during 2012 and 2013 mark-recapture surveys (i.e., presumed densities; see Chapter 10). Strata conformed to general landscape features and ice types: the high-density stratum included landfast ice along the coastline and within fjords as well as nearshore pack ice within ~30 km of the nearest land mass (~18,870 km²), whereas the low-density stratum included pack ice located farther offshore (~9,110 km²; Figure 11.1). Since the survey was conducted after adult females had left dens, we assumed that no bears were located on land during the study period. We used

GIS layers from Greenland and Nunavut to delineate the landward extent (i.e., coastline) of the study area. We delineated the extent of available habitat by approximating the edge of the North Water polynya with Moderate Resolution Imaging Spectroradiometer (MODIS; <http://modis.gsfc.nasa.gov/>) images (1 km resolution). The polynya's boundaries can change rapidly, so we delineated the extent of the polynya adjacent to the section surveyed on a particular day using MODIS imagery from that day, or from the closest date possible when same-day imagery was unclear due to atmospheric conditions. We also examined the delineated study area in relation to weekly regional sea-ice charts produced by the Canadian Ice Service (<https://www.ec.gc.ca/glaces-ice/>). During sampling, we collected GPS waypoints at the edge of the polynya to verify delineation. We did not sample in the polynya due to safety considerations. Polar bears occur in the polynya area (Heide-Jørgensen et al. 2013) but ice conditions in spring 2014 suggested that it was not suitable springtime habitat for polar bears due to its thin, forming (i.e., new and grey) ice and expansive open water (*cf.* Sahanatien and Derocher 2012). We also did not survey the sea ice in the fjords (~3,245 km²; Figure 11.1) of the populated Qaanaaq area in NW Greenland (i.e., in the eastern parts of the North Water polynya) because hunting pressure for marine mammals in these areas is generally high and consequently “resident” polar bears do not exist in the Qaanaaq area (Born et al. 2011, E. Born, pers. obs.).

Aerial transects were systematically spaced at 6-km and 18-km width intervals in the high- and low-density strata, respectively, based on anticipated encounter rates and available resources. We arranged transects in an east – west direction in open areas, but oriented them perpendicular to fjords (i.e., across the widths of fjords) to improve variance estimation (i.e., more numerous short transects) and reduce bias (i.e., sighting distances did not reflect potential density gradients, with highest densities along the sides of fjords; Figure 11.2).

During line-transect sampling, we surveyed at an altitude of ~120 m and groundspeed of ~150 km / hr. We sampled from most planned transects and included some ferry flights (during which we sampled) that were random with respect to the distribution of bears and presumed density gradients (i.e., highest densities near the polynya edge and along the sides of fjords) in analyses. Although many groups were observed during flights between consecutive transects, these typically occurred near the sides of fjords. As such, observations may have reflected a density gradient as well as the probability of detection and were thus inappropriate to include in distance sampling analyses (Stapleton et al. 2014).

We collected aerial survey data with mark-recapture distance sampling protocols (Laake and Borchers 2004, see also Stapleton et al. 2014, 2016). Two front (including the pilot) and two rear observers comprised the first and second capture periods, respectively, and teams of observers worked independently until both groups were afforded a full opportunity to observe a bear. After announcing a sighting, we flew off-transect to record the bear's initial location with a GPS, and we later estimated distance from transects in a GIS (Marques et al. 2006). During off-transect flights, we flew to within ~5 – 10 m of bears to obtain a tissue sample via biopsy darting for genetic analysis (see Chapter 10) and to estimate sex and age class of the bear. For each sighting, we recorded 3 covariates that potentially impacted detection probability: 1) habitat structure within a 30-m radius (smooth / low structure or moderate to high structure; i.e., smooth versus rough ice); 2) visibility (good or poor, due to fog, glare or precipitation); and 3) light conditions (i.e., cloud cover; clear: 0 – 25%; partly cloudy: 25 – 50%; mostly cloudy: 50 – 75%; or overcast: 75 – 100%).

Statistical Analyses

We analyzed line-transect data using distance sampling, which fits a function to observational data to describe how detection changes with increasing distance from the sampling transect (Buckland et al. 2001). We initially intended to use double observer (i.e., mark-recapture) distance sampling for analyses, but small sample sizes precluded this approach. We defined clusters as discrete groups of bears with non-independent detection probabilities (i.e., an adult female with 1 or more offspring or a breeding pair). We first examined a left-truncated data set (i.e., 75 m was subtracted from all observations; observations within 75 m were censored to account for blind spots directly beneath the helicopter; e.g., Borchers et al. 2006, Stapleton et al. 2014) to evaluate distance sampling's fundamental assumption of complete detection on the transect line (Buckland et al. 2001). Because these results indicated that the probability of detection by at least one observer was >96% at the adjusted transect line, we considered this assumption to be approximately met and proceeded with analyses including all observations (i.e., data were not left-truncated).

We completed analyses in the mark-recapture distance sampling (MRDS) engine of Program DISTANCE 6.2 (Thomas et al. 2010) and modeled the survey data as a single-observer study. We examined half-normal and hazard rate key functions and used multiple covariate distance sampling (Marques and Buckland 2003) to include a maximum of 1 covariate per model due to sample size constraints. We condensed light conditions into a binary covariate (0 – 25% cloud cover; >25% cloud cover) due to underrepresentation of some values. We considered each transect the sampling unit for variance estimation and used the Innes et al. (2002) method to estimate variance associated with global density and overall abundance.

11.3. Results

We surveyed 4,160 km of transects, including 3,850 km along 234 transects in the high-density stratum and 610 km along 14 transects in the low-density stratum. We observed 29 groups of polar bears (Figure 11.2), including 49 total bears (30 independent bears); cub-of-the-year and yearling litter sizes in this sample averaged 1.55 (SD: 0.5, $n = 11$) and 1.0 (SD: 0.0, $n = 2$), respectively. We right-truncated sightings data at 1,400 m to improve model fit and parsimony (Buckland et al. 2001), censoring one observation of an independent bear at >3,500 m, leaving 28 groups for estimating the detection function and abundance; 27 of these sightings occurred in the high-density stratum.

Sighting distance was not correlated with polar bear group size ($r = -0.10$, $P = 0.61$), so we used mean group size for abundance estimation. Histograms summarizing sightings distances indicated strong-support for a distance-based detection function (Figure 11.3), and all highly supported distance sampling models indicated adequate goodness-of-fit (chi-squared, Cramér-von Mises and Komolgorov-Smirnov tests: $P > 0.05$). The most highly supported model (half-normal key function) suggested that light conditions (cloud cover) affected detection probability (Figure 11.4). However, the small number of observations (see Buckland et al. 2001: at least 60 – 80 sightings are recommended for estimating the detection function) resulted in uncertainty in density and abundance estimation, and a model with a hazard rate key function had nearly equivalent support and estimated much higher densities (Figure 11.3, Table 11.1). Thus, we elected to model-average (Burnham and Anderson 2002) the 2 most highly supported models and obtained a subpopulation-wide estimate of 206 bears (SE: 101; 95% lognormal CI: 83 – 510; CV: 49%) in 2014.

11.4. Discussion

The estimate of abundance based on the springtime 2014 aerial survey in KB was 206 bears (95% lognormal CI: 83 - 510). However, due to insufficient coverage of offshore polar bear habitat (vast areas of offshore habitat in the North Water Polynya was not surveyed) this estimate is likely negatively biased. The estimate of abundance obtained from the aerial survey was negatively biased by about 30% or more (see below).

Based on a randomization procedure that assumed normal sampling distributions for abundance estimates, the mean difference between the estimate of total abundance 2013-2014 from the MR study (357 bears, 95% CI = 221 – 493) and the aerial survey estimate was approximately 151 bears (standard deviation of the difference \approx 127 bears), with 88% of the sampling distribution suggesting that the difference was positive (i.e., that the MR estimate was at least one bear larger than the aerial survey estimate).

Differences between MR and aerial survey point estimates in KB require caution when comparing results from different techniques conducted during springtime. Aerial surveys yield a snapshot estimate of abundance (i.e., the number of bears occupying the survey area during the study period), whereas MR generates a super-population estimate reflecting all bears with a non-zero probability of detection during the study period (including individuals that are currently outside the survey area due to temporary emigration) (Kendall et al. 1997). We suggest that the MR estimate is appropriate for use in management.

This finding reinforces that boundaries between subpopulations are not discrete, a result consistent with satellite telemetry (Taylor et al. 2001, Chapter 9) and capture and harvest records (e.g., Peacock et al. 2012, Chapter 10). Interchange among subpopulations is particularly prevalent during the springtime (Chapter 9), meaning that a large number of bears were likely exposed to sampling during the 3-year mark-recapture study period in the KB subpopulation.

These seasonal movement patterns contribute to the finding that the mark-recapture point estimate was ~85% greater than the aerial survey point estimate. We note that data used for the mark-recapture analysis were sparse and precluded implementing the Barker model (1997, 1999) to explicitly model temporary emigration. However, such an approach would enable a more direct comparison between methods by defining the mark-recapture estimate as pertaining to only those bears that were present in the study area and available for capture, rather than the entire super-population.

Although the aerial survey was not ideally designed (it relied up the use of a single-engine helicopter with limited range so that offshore habitat could not be surveyed), the KB subpopulation study provides the first opportunity to directly compare simultaneous mark-recapture and aerial survey studies. In Western Hudson Bay, estimates of abundance derived from mark-recapture and an aerial survey were similar (although the aerial survey snapshot estimate was somewhat greater than the mark-recapture super-population estimate), but differences in sampling frames limited inference (Stapleton et al. 2014, Lunn et al. 2016). Similarly, abundance estimates from an aerial survey and mark-recapture in Southern Hudson Bay were consistent, but several years elapsed between the inventories, and the mark-recapture estimate was adjusted upwards to reflect potential heterogeneity in capture probabilities and to account for un-sampled areas (Obbard 2008, Obbard et al. 2007, 2015). As jurisdictions incorporate alternative (non-capture based) methods for estimating abundance and monitoring populations, understanding the ability to compare results from different techniques will be critical to correctly interpreting status and trend (Stapleton et al. 2014). The differences in survey methods resulting in estimates of different “populations” (i.e., the MR estimate of the “super”-population versus the aerial survey’s real-time snapshot of abundance) suggest caution

when comparing results and assessing trends from different techniques implemented during the springtime, when polar bear movements among subpopulations are greatest.

We acknowledge that our estimate of abundance derived from the aerial survey is likely biased low. First, we did not sample the southeastern portion of the KB subpopulation and the large area of the North Water polynya because of logistical and safety considerations in a helicopter and the presumed relatively low densities of bears in these regions (Heide-Jørgensen et al. 2013). Satellite telemetry data indicated no collared bears ($n = 20$ adult females) were present in the un-surveyed areas during the aerial survey sampling period (see also Chapter 9). However, the un-sampled regions covered extensive areas (sea ice near Qaanaaq: 3,245 km², and the North Water polynya: 27,214 km²), such that even very low densities may significantly contribute to an overall estimate of abundance. This unsurveyed area in the North Water Polynya amounts to *ca.* 34% - 40% of the extension of the polynya (70,000-80,000 km²; Born et al. 2004 and references therein). Extrapolating our model-averaged estimate of density from the low-density stratum (3.39 bears / 1,000 km²) to the sea ice near Qaanaaq in southeastern KB yielded ~11 bears. For the North Water polynya, extrapolating a very low estimate of density (1.13 bears / 1,000 km², or roughly a third of the estimated density used for the sea ice near Qaanaaq) added 31 bears.

During May 2009 and 2010, Heide-Jørgensen et al. (2013) conducted an aerial survey over the North Water Polynya (NOW) between 76° N and 79° N (i.e., north to the southernmost part of the Nares Strait-Kane Basin region). Hence, they in effect covered a major part of the NOW with loose drift ice and open water which were not covered by us for safety reasons and because it was judged by us to be suboptimal or unsuitable polar bear habitat. Despite that their survey was a multi-species survey mainly targeting beluga (*Delphinapterus leucas*), narwhal

(*Monodon monoceros*), walrus (*Odobenus rosmarus*), and seals, they detected polar bears both in water and on ice. Heide-Jørgensen et al. (2013) derived an estimate of 60 polar bears (CV 0.96, range: 12-293 bears). Although their point estimate had a large uncertainty due to low sample size, it indicates that a substantial number of polar bears may occur “offshore” on loose drift ice in NOW (i.e., in habitat not covered during our 2014 survey). Given the inherent uncertainty in estimates of density for the unsampled regions, we hypothesize that negative bias arising from incomplete sampling of the Kane Basin polar bear subpopulation may have been roughly 10 – 30%.

Second, a fundamental assumption of distance sampling is perfect detection of target objects on the transect line (i.e., at distance 0; Buckland et al. 2001). Preliminary analyses with a left-truncated data set suggested that the probability of bears near the aircraft being sighted by at least one observer was >96%, so we considered this assumption to be approximately valid. Our data were too sparse to permit mark-recapture distance sampling analyses (Laake and Borchers 2004) to correct for less than perfect detection at distance 0, but our initial double-observer analyses suggest that any resultant negative bias was modest (<5%).

Population-wide aerial surveys of polar bears have been completed in the autumn over land in Foxe Basin, Western Hudson Bay, and Southern Hudson Bay (Stapleton et al. 2014, 2016, Obbard et al. 2015) and over both land and sea ice in the Barents Sea (Aars et al. 2009). Similarly, recent pilot aerial survey studies over springtime sea ice have been completed in the Baffin Bay and Southern Beaufort Sea subpopulations (Stapleton 2013). However, the aerial survey in the KB subpopulation represents the first attempt to complete a subpopulation-wide survey on springtime sea ice. Although most aerial surveys of polar bears have been conducted during the fall ice-free period, this study illustrates that, in small areas and under favorable

weather conditions, aerial surveys can provide a useful inventory technique on springtime sea ice as well. Aerial surveys may be a particularly valuable tool for monitoring small, remote subpopulations that are not subject to significant harvest pressure and where acquiring detailed demographic information through mark-recapture may be too costly to justify.

We note, however, that larger geographic areas and periods of inclement weather may require more time to complete a comprehensive aerial survey, thereby necessitating more complex study designs to accommodate potential changes in bear densities and the study area itself, especially if sea-ice dynamics are changing during the survey window. Aerial surveys of larger areas like Baffin Bay will require the use of more than one fixed-winged aircraft with long endurance to ensure that the entire area (including remote offshore habitat) is covered within a relatively narrow time frame (e.g., SWG 2011, Nielson et al. 2013). Nevertheless, based on the pilot aerial survey along SE Baffin Island in spring 2010, a group of survey experts concluded that it would be feasible to assess polar bear populations with a larger range (i.e., Baffin Bay) using aerial surveys (Chapter 1).

We recognize that large offshore areas with loose drift ice in the Kane Basin subpopulation's range could not be surveyed by us for safety reasons and because our helicopter had a relatively low range and endurance. Hence, future aerial surveys should consider using fixed-winged aircraft perhaps in combination with a helicopter (SWG 2011).

Our aerial survey estimate of abundance was based on a very small number of encounters ($n = 28$), resulting in some uncertainty in estimation of the detection function. Increasing the number of observations via greater sampling effort will likely improve precision; a minimum of 60 – 80 observations are recommended with distance sampling (Buckland et al. 2001), but even a marginal increase in sightings would improve estimation of the detection function. In addition, if

other sites adopt on-ice surveys using similar study designs and survey platforms, joint analysis in which observations are pooled might yield more reliable estimates of the detection function, thereby improving precision of abundance estimates.

11.5. Literature Cited

- Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. *Marine Mammal Science* 25:35–52.
- Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 *in* E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. *Arctic sea ice decline: Observations, projections, mechanisms and implications*. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.
- Barker, R. J. 1997. Joint modeling of live-recapture, tag-resight, and tag-recovery data. *Biometrics* 53:666–677.
- Barker, R. J. 1999. Joint analysis of mark-recapture, resighting and ring-recovery data with age-dependence and marking effect. *Bird Study* 46:S82–S91.
- Born, E. W., J. Teilmann, M. Acquarone, and F. Riget. 2004. Habitat use of ringed seal (*Phoca hispida*) in the North Water area (North Baffin Bay). *Arctic* 57:129–142.
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. *Monographs on Greenland, Man and Society* Volume 41. Museum Tusulanum Press, University of Copenhagen, Copenhagen, Denmark.
- Borchers, D. L., J. L. Laake, C. Southwell, and C. G. M. Paxton. 2006. Accommodating unmodeled heterogeneity in double-observer distance sampling surveys. *Biometrics* 62:372–378.
- Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. *Introduction to Distance sampling: Estimating abundance of biological populations*. Oxford University Press, Oxford, UK.

- Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information–theoretic approach. Springer, New York, New York, USA.
- Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. *Journal of Animal Ecology* 82:912–921.
- Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. *Ecological Monographs* 79:25–58.
- Heidi-Jørgensen, M. P., L. M. Burt, R. G. Hansen, N. H. Nielsen, M. Rasmussen, S. Fossette, and H. Stern. 2013. The significance of the North Water Polynya to Arctic top predators. *Ambio* 42:596–610.
- Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. *Ursus* 24:148–163.
- Innes, S., M. P. Heide-Jørgensen, J. L. Laake, K. L. Laidre, H. J. Cleator, P. Richard, and R. E. A. Stewart. 2002. Surveys of belugas and narwhals in the Canadian High Arctic in 1996. NAMMCO Scientific Publication 4:169–190.
- Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock’s robust design. *Ecology* 78:563–578.
- Laake, J. L., and D. Borchers. 2004. Methods for incomplete detection at distance zero. Pages 108–189 in S. Buckland, D. Anderson, K. Burnham, J. Laake, D. Borchers, and L. Thomas, editors. *Advanced Distance Sampling*. Oxford University Press, Oxford, UK.
- Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. *Ecological Applications* 26:1302–1320.
- Marques, F. F. C., and S. Buckland. 2003. Incorporating covariates into standard line transect analyses. *Biometrics* 59:924–935.
- Marques, T. A., M. Andersen, S. Christensen-Dalsgaard, S. Belikov, A. Boltunov, Ø. Wiig, S. T. Buckland, and J. Aars. 2006. The use of global positioning systems to record distances in a helicopter line-transect survey. *Wildlife Society Bulletin* 34:759–763.

- Nielson, R. M., T. J. Evans, and M. B. Stahl. 2013. Investigating the potential use of aerial line transect surveys for estimating polar bear abundance in sea ice habitats: a case study for the Chukchi Sea. *Marine Mammal Science* 29:389–406.
- Obbard, M. E. 2008. Southern Hudson Bay polar bear project 2003 – 2005: final report. Unpublished report, Wildlife Research and Development Section, Ontario Ministry of Natural Resources, Peterborough, Ontario, Canada.
- Obbard, M. E., T. L. McDonald, E. J. Howe, E. V. Regehr, and E. S. Richardson. 2007. Polar bear population status in Southern Hudson Bay, Canada. Administrative report, U.S. Geological Survey, Reston, Virginia, USA.
- Obbard, M., S. Stapleton, K. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. *Polar Biology* 38:1713–1725.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- Peacock, E., J. Laake, K.L. Laidre, E.W. Born, and S.N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. *Arctic* 65:391–400.
- Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre, and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. *Population Ecology* 54:3–18.
- Sahanatien, V., and A. E. Derocher. 2012. Monitoring sea ice habitat fragmentation for polar bear conservation. *Animal Conservation* 15:397–406.
- Stapleton, S. 2013. Alternative methods for monitoring polar bears in the North American Arctic. PhD Dissertation, University of Minnesota, St. Paul, Minnesota, USA.
- Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. *Biological Conservation* 170:38–47.
- Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. *Marine Mammal*

Science 32:181–201.

SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear].

2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.

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

Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. *Arctic* 58:203–214.

Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. B. Bishop, T. Marquest, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology* 47:5–14.

Table 11.1. Results from distance sampling analyses of an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014. The most highly supported models ($\Delta AIC_c < 2$) are shown. In the column Model, the key function is followed by the covariate (Light = light conditions). p = detection probability. High- and low- density refer to stratum-specific estimates.

Model	ΔAIC_c	p	Density (Bears / 1,000 km ²)			Abundance (SE)
			High-density	Low-density	Global	
Half-Normal / Light	0.00	0.60 (0.09)	7.5 (2.0)	3.1 (3.1)	6.1 (1.7)	170 (49)
Hazard / None	0.13	0.43 (0.20)	11.1 (6.0)	3.7 (4.0)	8.7 (4.5)	243 (125)
Half-Normal / None	0.37	0.62 (0.09)	7.6 (1.9)	2.5 (2.5)	5.9 (1.6)	166 (44)

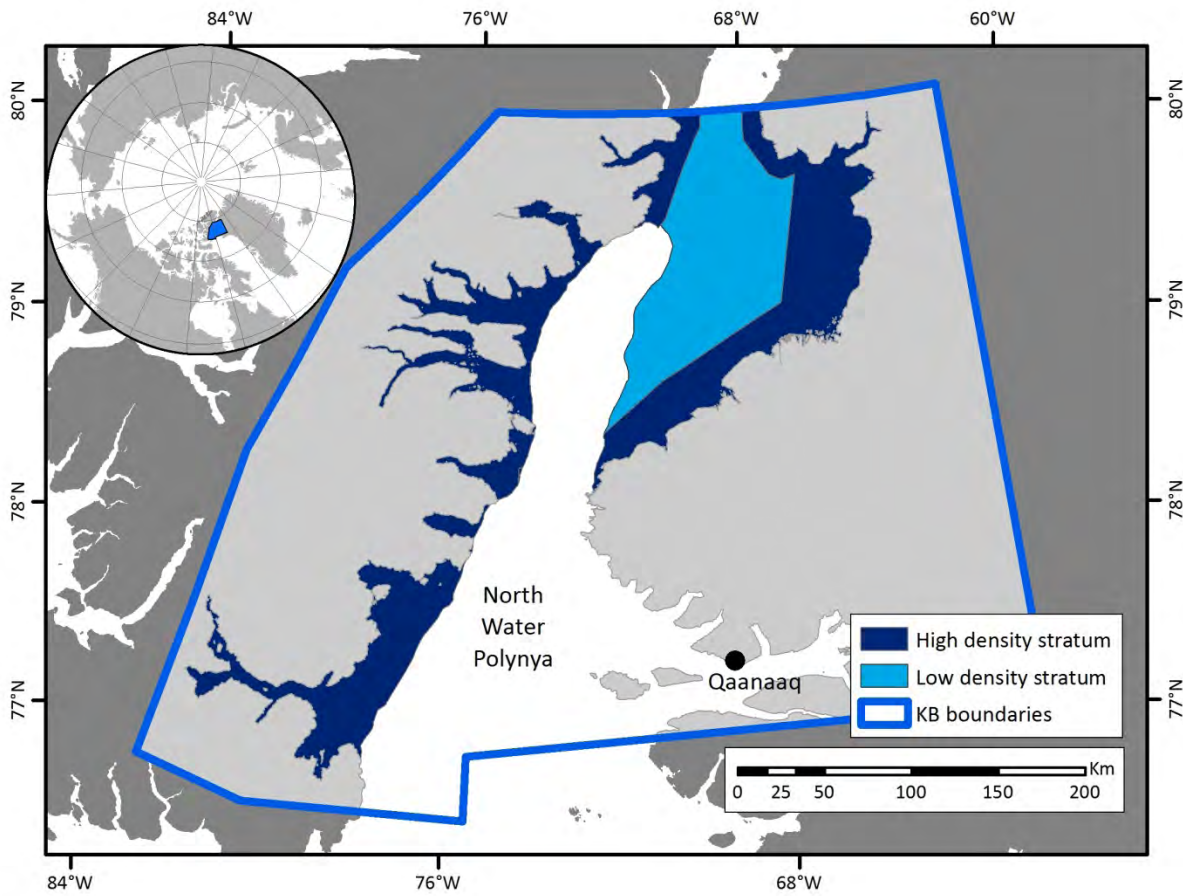


Figure 11.1. Sampling strata for genetic mark-recapture and aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

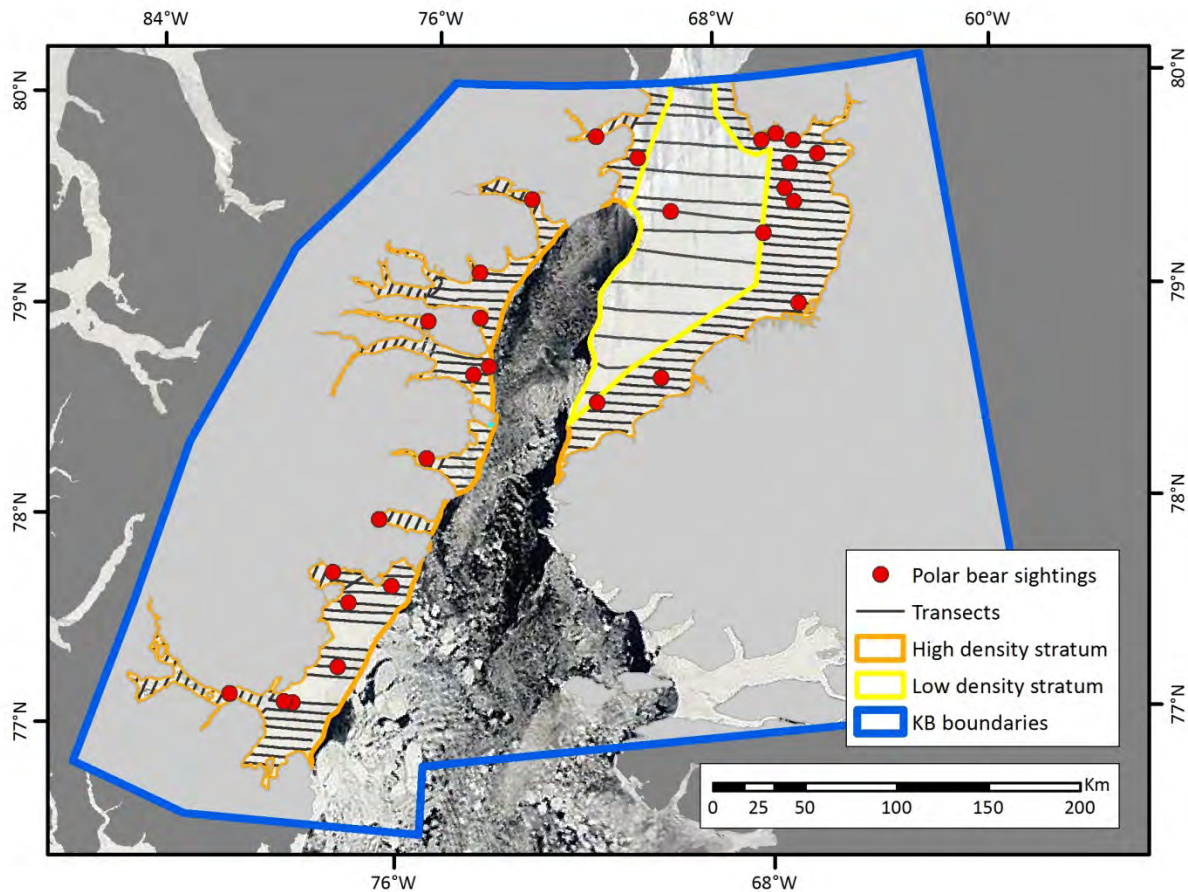


Figure 11.2. Transects surveyed and polar bear groups sighted during transect surveys of the Kane Basin subpopulation during April – May, 2014. Transects and sightings are overlaid on MODIS image (1 km resolution; available: <http://modis.gsfc.nasa.gov/>) collected on 5 May 2014. Sea ice in southeastern Kane Basin (i.e., to left of figure legend) was not sampled due to safety and logistical constraints presented by the North Water polynya and because we anticipated very low densities.

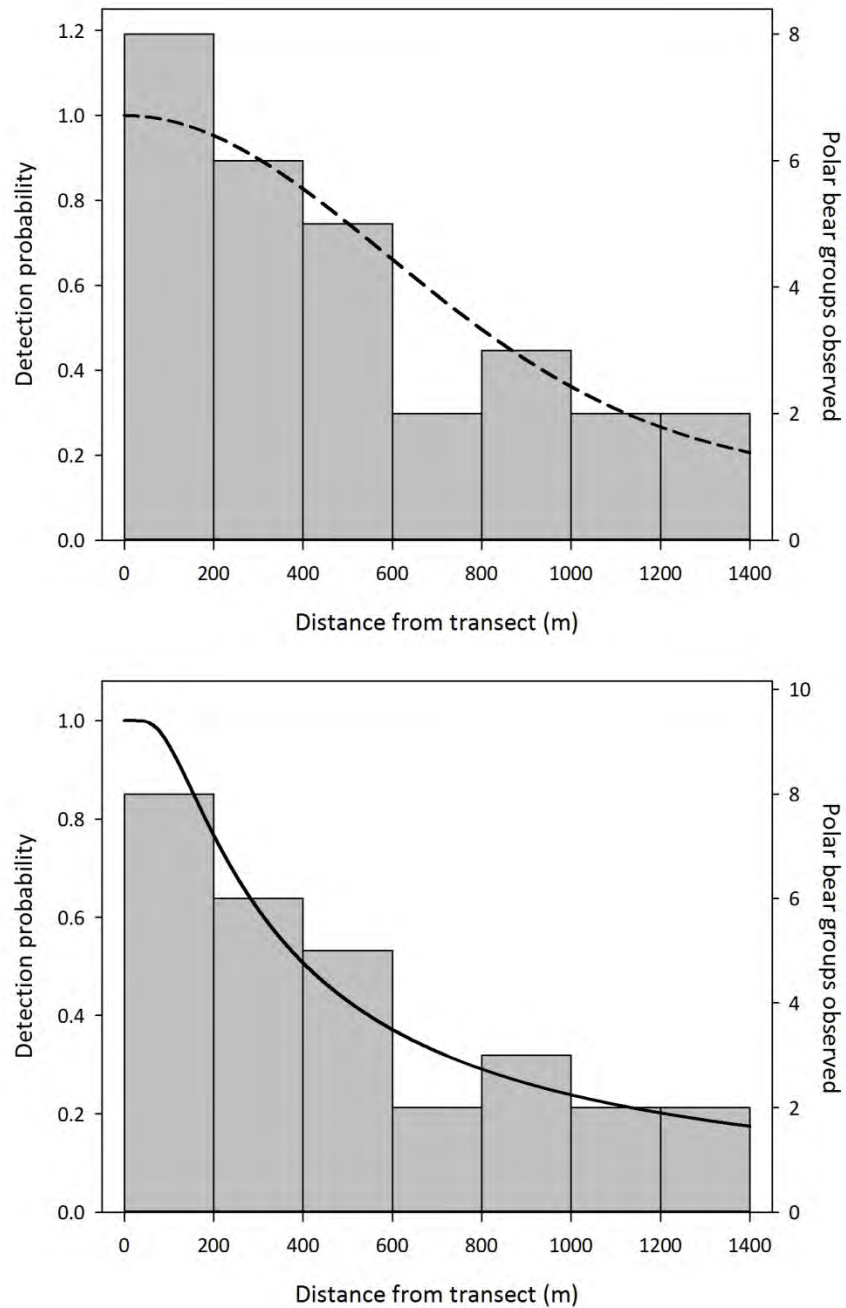


Figure 11.3. Histograms summarizing sighting distances and estimated detection functions from an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014. Top: Half-normal key function including a binary light conditions covariate. Bottom: Hazard rate key function with no adjustment terms or covariates.

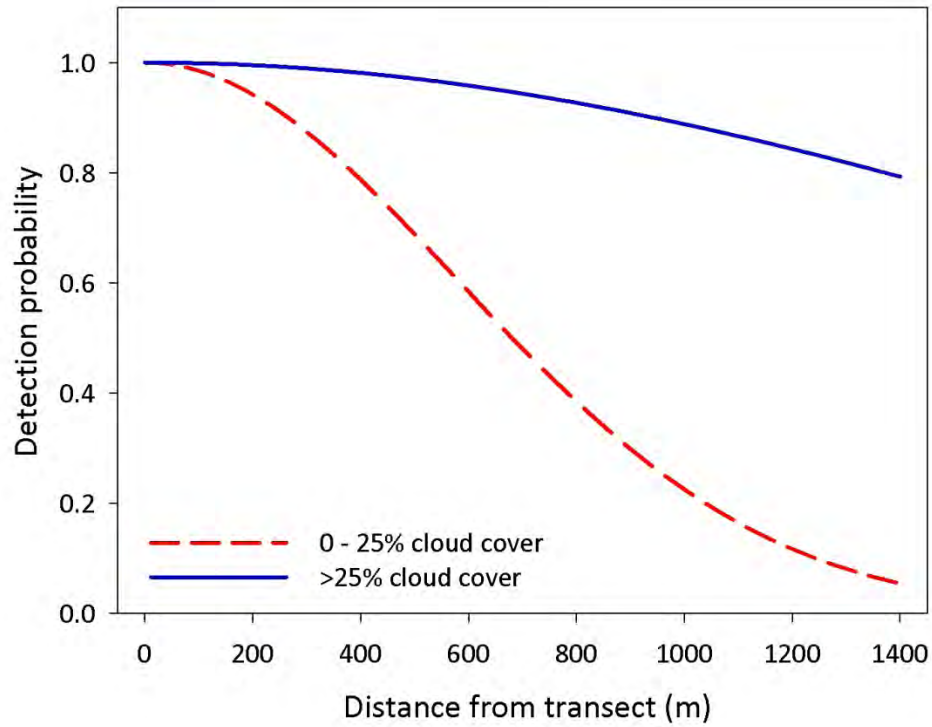


Figure 11.4. Distance sampling detection function (half-normal key function with binary light conditions covariate) estimated from data collected during an aerial survey of the Kane Basin polar bear subpopulation, April – May, 2014.

CHAPTER 12

REPRODUCTIVE METRICS FOR MARK-RECAPTURE SAMPLED POLAR BEARS IN KANE BASIN

KEY FINDINGS

- Data for the study were collected during two periods of MR sampling in KB. Sampling occurred on the sea ice in April and May. During the 1990s, bears were sampled by physical capture and examination using methods previously described. During the 2000s, sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals.
 - Reproductive metrics for KB, including mean litters sizes for cubs-of-the-year (COY) and yearlings, and an index of recruitment (calculated as the number of yearlings per adult female in the MR sample), were comparable between the 1990s and 2010s sampling periods. Mean litter sizes in KB (for COY: 1.67 in the 1990s and 1.60 in the 2010s) were similar to those observed in other polar bear subpopulations in the archipelago ecoregion (range 1.65 - 1.71). We found no evidence of lower reproductive performance in KB, but sparse data limited our conclusions.
 - During the years with the largest sample sizes (1995, 2013, and 2014), the total sample exceeded 50 bears. In these years, there was notable variation in the proportions of COY (15% - 30%). In contrast, proportions of yearlings and the recruitment index were relatively invariant among years.
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12.1. 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 therefore may provide a useful tool for the early detection of 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 periods of 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). Of particular concern, 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, Molnár 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 in association 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., fluctuating carrying capacity; Stirling and Øritsland 1995, Stirling and Lunn 1997, Stirling 2002, Rode et al. 2014). Information on reproduction therefore must be considered alongside other measures of subpopulation performance in-order to properly assess status.

The Kane Basin (KB) polar bear subpopulation is part of the archipelago ecoregion as defined by Amstrup et al. (2008), where sea ice does not melt entirely in the summer and some bears remain on the ice year-round. However, the sea ice situation in Kane Basin has changed markedly in recent decades (Chapter 9). Although currently designated as declining based on population viability modelling (PBSG 2010), a comparison of results from two mark recapture studies suggests the abundance of this small, low density subpopulation has not changed significantly over the two decades (Taylor et al. 2008a, Chapter 10). KB has experienced long term changes in sea-ice composition, and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapters 4 and 9). However, model projections predict that KB will be one of the last polar bear subpopulations to experience the negative consequences of climate change including reproductive failure (Amstrup et al. 2008, Hamilton et al. 2014). Here, we summarize reproductive metrics for KB using data collected during two periods of mark-recapture sampling from 1992 to 1997 (Taylor et al. 2008a) and 2012 to 2014 (Chapter 10). Our results provide additional context for interpreting the results of mark-recapture analyses and assessment of this subpopulation's present status.

12.2. Materials and Methods

Study Area

The KB polar bear subpopulation covers roughly 150,000 km² and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008a). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit; *cf.* Figure 11.2). The

subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq areas). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, the amount of sea ice during summer in Nares Strait-Kane Basin has dropped markedly in recent decades (e.g., Figure 9.2).

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in KB. Sampling occurred on the sea ice in April and May. During the 1990s, bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2008a). Data on the sex, 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 via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. Sampling occurred during 25 April-6 May in 2012 and during 27 April and 10 May in 2013. With biopsy darting, 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 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). Field 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 known age-class individuals (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., Derocher and Stirling 1995, Rode et al. 2010, Peacock et al. 2013, Stapleton et al. 2014, Regehr et al. 2015). For cubs-of-the-year (COY) and yearlings, mean litter sizes were calculated from observed litter sizes. Because we did not have estimated ages for adult females sampled in 2012-2014 and because sample sizes were small in most years, we calculated a pooled mean for each year rather than age-specific values. Numbers of COY and yearling were expressed as a proportion of the total bears sampled each year. An index of recruitment was calculated as the total number yearlings divided by the total number of adult females in the sample (Derocher and Stirling 1995, Regehr et al. 2015).

Some individuals were sampled more than once in a given year. These recaptures were excluded from analyses. Captures of the same individual over multiple years were included.

We examined annual variation in reproductive metrics and compared metrics between the two epochs (1992-1997 and 2012-2014). Statistical analyses were performed using the SPSS package (Version 24.0, IBM Corp. 2016).

12.3. Results

During 1992-1997 and 2012-2014, we sampled 53 family groups consisting of an adult female and 1-2 dependent COY (87 COY in total). We also sampled 24 family groups consisting of an adult female and 1-3 dependent yearlings (32 yearlings in total). The mean number of family groups sampled annually was 5.9 (range: 2-15) and 2.7 (range: 0-5) for COY and yearling families respectively. Annual reproductive metrics are presented in Table 12.1.

Annual variation in observed litters sizes was not significant amongst COY (Kruskal-Wallis, $H = 4.86$, $P = 0.772$) or yearlings (Kruskal-Wallis, $H = 9.49$, $P = 0.219$). COY comprised between 9 and 38% of the bears sampled annually. Yearlings comprised between 0 and 16% of annual observations. Recruitment ranged from zero to 0.43. Sample sizes were too small to permit further analyses of annual reproductive metrics. Pooling data within epochs there were no differences in mean litter sizes between the 1990s and 2000s (Mann-Whitney U test, $U = 369$, $P = 0.700$ for COY; $U = 79.5$, $P = 0.671$ for yearlings). Proportions of COY and yearlings were also similar between epochs (Table 12.2).

During sampling in 2012-2014, 9 (12%) of 78 adult females encountered were of known age (marked during the 1990s) and ranged in age from 18 to 35 years. Five were between 18 and 20 years old, three of which were observed with litters. None of the 4 (5%) bears > 20 years old

were observed with offspring. In comparison, during sampling in the 1990s, 77 adult females were sampled. Of these, 10 (13%) were 18 years old or greater (based on tooth aging) ranging in age from 18 to 21 years, and seven were accompanied by offspring. Two (2.6%) were greater than 20 years of age.

12.4. Discussion

Sample sizes were too small in most years to permit investigation of annual variation in reproductive metrics in KB. Observed variation likely was primarily the product of sample size rather than biological effect. During the years with the largest sample sizes (1995, 2013, and 2014), the total sample exceeded 50 bears, representing a sizeable portion of this small subpopulation. In these years, there was notable variation in the proportions of COY (15% - 30%; Table 12.1). In contrast, proportions of yearlings and the recruitment index were relatively invariant among years. We are unaware of any source of sampling bias that would account for this apparent variation in COY production and suggest that it may reflect pulsing or synchrony in reproduction. Anecdotally, dates of spring and fall sea-ice transition in the years prior to these 3 years were unremarkable in terms of variation; offering no explanation in terms of environmental conditions.

Our surveys were conducted in late April-early May. Den emergence date even at high latitudes (i.e., $> 76^\circ \text{N}$) late March (Ferguson et al. 2000; Chapter 9, Figure 9.14). Hence, we are confident that our observations of adult females with COYs were representative.

During recent sampling, 2012-2014, we recaptured a small number of older, known age adult females that were originally marked in the 1990s. Amongst this sample, none of the individuals >20 years of age were accompanied by offspring. Although the sample size is small,

this is consistent with the reproductive senescence reported in some other subpopulations (Ramsay and Stirling 1988, Derocher and Stirling 1994).

Reproductive metrics for KB in both time periods were comparable, and mean litter sizes in KB were within the range of observed variation for other polar bear subpopulations in the archipelago ecoregion (Table 12.2). In summary, we found no evidence of lower reproductive performance in KB, but sparse data limited our conclusions.

12.5. Literature Cited

- Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 in E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. Arctic sea ice decline: Observations, projections, mechanisms and implications. Geophysical Monograph Series 180. American Geophysical Union, Washington, DC, USA.
- Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. *Ursus* 10:449–453.
- Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. *Population Ecology* 47:267–275.
- Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. *Integrative and Comparative Biology* 44:163–176.
- Derocher, A. E., and I. Stirling. 1994. Age-specific reproductive performance of female polar bears (*Ursus maritimus*). *Journal of Zoology* 234:527–536.
- Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. *Canadian Journal of Zoology* 73:1657–1665.
- Ferguson, S. H., M. K. Taylor, A. Rosing-Asvid, E. W. Born, and F. Messier. 2000. Relationships between denning of polar bears and conditions of sea ice. *Journal of Mammalogy* 81:1118–1127.
- Fowler, C. W. 1981. Density dependence as related to life history strategy. *Ecology* 62:602–610.

- Fowler, C. W. 1987. A review of density dependence in populations of large mammals. *Current Mammalogy* 1:401–441.
- Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. *PLoS ONE* 9(11): e113746. doi:10.1371/journal.pone.0113746.
- Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology* 29:724–737.
- Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. *Nature Communications* 2:1–8.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 *in* M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. *Journal of Wildlife Management* 77:463–476.
- Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). *Journal of Zoology* 214:601–634.
- Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. *Journal of Wildlife Management* 71:2673–2683.
- Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.
- Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. *Ecological Applications* 20:768–782.
- Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat

- loss: feeding and reproductive ecology of two polar bear populations. *Global Change Biology* 20:76–88.
- Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. *Arctic* 55:59–76.
- Stirling, I., W. Calvert, and D. Andriashek. 1980. Population ecology studies of the polar bear in the area of southeastern Baffin Island. Canadian Wildlife Service Occasional Paper No. 44.
- Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. *Global Change Biology* 18:2694–2706.
- Stirling, I., and N. J. Lunn. 1997. Environmental fluctuations in arctic marine ecosystems as reflected by variability in reproduction of polar bears and ringed seals. Pages 167–181 in S. J. Woodin, and M. Marquiss, editors. *Ecology of arctic environments*. Special Publication No. 13 of the British Ecological Society, Blackwell Science Ltd., Oxford, UK.
- Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. *Arctic* 52:294–306.
- Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. *Canadian Journal of Fisheries and Aquatic Sciences* 52:2594–2612.
- Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. *Arctic* 59:261–275.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. *Journal of Wildlife Management* 70:1667–1673.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2008b. Mark-recapture and stochastic population models for polar bears of the high arctic. *Arctic* 61:143–152.

- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. *Marine Mammal Science* 25:778-796.
- Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. *Ursus* 5:1–66.

Table 12.1. Reproductive metrics derived from annual mark-recapture sampling data from Kane Basin. Captures and between season recaptures are included.

Year	Mean Litter Size (n, SD)		Proportion of Total Observations (n)		Recruitment Index ¹ (Yearlings/adult female)
	COY	Yearling	COY	Yearlings	
1992	2.00 (3, 0.00)	0.00 (0, 0.00)	0.38 (16)	0.00 (16)	0.00
1993	2.00 (2, 0.00)	1.00 (3, 0.00)	0.17 (23)	0.13 (23)	0.43
1994	1.60 (5, 0.55)	2.00 (2, 0.00)	0.26 (31)	0.13 (31)	0.40
1995	1.70 (10, 0.48)	1.50 (2, 0.71)	0.25 (67)	0.04 (67)	0.14
1996	1.40 (5, 0.55)	1.00 (3, 0.00)	0.19 (36)	0.08 (36)	0.23
1997	1.50 (2, 0.71)	2.00 (2, 1.41)	0.09 (32)	0.13 (32)	0.40
2012	1.50 (4, 0.58)	1.40 (5, 0.55)	0.14 (44)	0.16 (44)	0.37
2013	1.57 (7, 0.53)	1.33 (3, 0.58)	0.15 (71)	0.06 (71)	0.14
2014	1.65 (14, 0.50)	1.00 (4, 0.00)	0.30 (84)	0.05 (84)	0.13

¹ – *Sensu* Regehr et al. (2015)

Table 12.2. Comparison of reproductive metrics for some polar bear subpopulations in the Arctic archipelago ecoregion (Amstrup et al. 2008). Sampling occurred during ice-free periods.

Subpopulation	Mean Litter Size		Proportion of Total Observations		Source
	COY	Yearling	COY	Yearlings	
Kane Basin (1992-97)	1.67	1.42	0.22	0.08	Taylor et al. (2008a)
Kane Basin (2012-14)	1.60	1.25	0.21	0.08	This study
GB	1.65	-	-	-	Taylor et al. (2009)
LS	1.69	-	-	-	Taylor et al. (2008b)
MC	1.68	-	-	-	Taylor et al. (2006)
NW	1.71	-	-	-	Taylor et al. (2008b)

CHAPTER 13

OBSERVATIONS ON THE BODY CONDITION AND FORAGING HABITS OF POLAR BEARS IN KANE BASIN DURING THE SPRING

KEY FINDINGS

- Body condition was better amongst KB subadults and adult females with yearlings during the 2010s relative to the 1990s, though sample sizes were limited. In contrast, condition amongst adult males, adult females with COYs, and lone adult females was similar between these time periods.
 - Improved condition in the 2010s may reflect natural variation or a response to long-term changes in the sea-ice regimen in Kane Basin, largely turning into a system resembling a seasonal sea-ice ecoregion.
 - Seals and polar bears were similarly distributed in Kane Basin.
 - Relatively high densities of both seals and bears in northeastern Kane Basin, near the Humbolt Glacier, indicates that this region has high productivity and is important habitat for polar bears in the subpopulation.
 - A high proportion of KB bears were found to have extensive hair loss and skin ulcerations on their feet. The cause of these lesions is unknown to science. Traditional knowledge suggests this phenomenon is the result of abrasive injuries sustained by walking and digging in hard, icy, coarse snow cover on the spring sea ice combined with increased rates of movement during the peak mating and feeding periods.
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13.1. Introduction

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 limit 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 is characterized by large seasonal changes in body condition (Watts & 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 1990, 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. 2015a, Rode et al. 2015).

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, Molnár et al. 2010, Molnár et al. 2011). Tracking long-term trends in body condition has thus 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).

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 Kane Basin (KB) polar bear subpopulation is part of the archipelago ecoregion as defined by Amstrup et al. (2008), where sea ice does not melt entirely in the summer and some bears remain on the ice year-round. Although currently designated as declining based on population viability modelling (PBSG 2010), a comparison of results from two mark recapture

studies suggests the abundance of this small, low density subpopulation has not changed significantly over the two decades (Taylor et al. 2008a, Chapter 10). KB has experienced long term changes in sea-ice composition, and a trend towards earlier spring break-up and later fall freeze-up (Laidre et al. 2015, Chapter 4). However, model projections predict that KB will be one of the last polar bear subpopulations to experience the negative consequences of climate change including reproductive failure (Amstrup et al. 2008, Hamilton et al. 2014).

Here we summarize information on the body condition of polar bears in KB collected during two periods of mark-recapture sampling from 1992 to 1997 and 2012 to 2014. Using the Fatness Index (FI: Stirling et al. 2008b) as a qualitative metric we examine differences in condition between the two time periods. Our results provide supplementary information for interpreting the results of the genetic mark-recapture (Chapter 10) and other recent studies in KB (Chapters 2 and 9), and for understanding the present status of this subpopulation. We also report on incidental observations of prominent skin lesions that were found on some KB bears during the latter period of the study. Finally, we report incidental observations of spring time foraging by bears and the distribution of seals in KB.

13.2. Materials and Methods

Study Area

The KB polar bear subpopulation covers roughly 150,000 km² and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea-ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit (*cf.* Figure 13.1). The

subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, sea-ice conditions have changed markedly in the Kane Basin region in recent decades (Born et al. 2011; Figure 9.2).

Field Sampling

Data for the study were collected during two periods of mark-recapture sampling in KB. In both periods sampling occurred on the sea ice in April and May using a helicopter flying at 300-500 feet above sea-level to search for bears across the study area. During the 1990s (1992-97), bears were sampled by physical capture and examination using methods previously described (Taylor et al. 2008a). Data on the sex, 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 2010s (2012-14), sampling occurred via physical capture or biopsy darting and subsequent genetic analysis to determine genetic sex and identify individuals. With biopsy darting, 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 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). Field 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 known age-class individuals (Appendix B).

Body Condition Scoring

Because most of the bears sampled during the latter period of sampling (2012-2014) were biopsy darted rather than captured and handled, our ability to compare body condition between time periods was limited to visually assigned Fatness Index (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). 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. 2008b). During the 1990s, this score was based on physical examination of captured bears. For bears in the 2010s, FI scores for most

(67%) individuals were assigned based on examination from the air at a distance of 3-7 m above ground. The remaining portion was assigned FI scores based on physical examination after capture.

All bears were initially scored in the field according to the standard FI on a scale of 1 to 5 (Stirling et al. 2008b). This scoring system was subsequently simplified to a binary Body Condition Score (BCS) where individuals in poor (FI = 1, 2) and fair-good (FI = 3, 4 or 5) condition were assigned scores of 1 and 2 respectively. 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 necessary due to the small samples sizes overall in our study and the low frequencies of bears at the extremes of the 5 point FI scale (i.e., very few or no bears with FI scores of 1 or 5). This simplified scoring system was also a potential means to reduce bias in assigning condition scores. The assumption made was 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 condition even at distances of up to 7 m. All observers in our study had extensive experience studying polar bears including capture, handling and body condition scoring.

For analyses, we pooled BCS data collected in different years into two periods (epochs); the 1990s (1992-97) and the 2010 (2012-2014). Again this was necessary due to low samples sizes. 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 its response to annual variation in environmental conditions.

For different sex, age and reproductive classes of polar bears we compared BCS between the two epochs using contingency tables analyses (Cross Tabs procedure in SPSS Version 24.0, IBM Corp. 2016). We also considered the potential effect of the timing of sampling on the BCS of bears. Along with the binary categorical variable (Epoch: 1990s and 2010s), Julian Day of sampling was used as an independent variable in a logistic regression (Binary Logistic procedure in SPSS Version 24.0, IBM Corp. 2016) to examine variation in BCS. Both variables were entered into the regression model. We did not specify an interaction between Julian Day and Epoch. All tests were two-tailed with alpha at 0.05.

Other Observations

During sampling from 2012-2014, we also made several other types of observations either systematically or opportunistically. For each bear encountered we noted any evidence of recent feeding. The presence of a seal kill or bears with full pendulous stomachs constituted evidence of feeding. While searching for bears in 2013, we noted the locations of live seals using a GPS. Each group comprising 1 or more individuals was recorded as a single observation. Finally, during capture and physical examination of bears in 2011 and 2012 we noted the presence or absence of some prominent skin lesions that had not previously been described in the literature.

13.3. Results

Body Condition Scores

Body Condition Scores (BCS) were assigned to 129 and 135 subadult and adult polar bears encountered during sampling in KB in the 1990s and 2010s, respectively. BCS for adult males were similar between the two epochs (Table 13.1). In contrast, across all reproductive classes, adult females in the 2010s tended to be in better condition than those in the 1990s; although this was statistically significant for adult females with yearlings only. Similarly, subadults in the 2010s were in better condition at time of encounter.

Although sampling occurred in April and early May during both epochs, timing of sampling differed (Mann-Whitney $U = 1,557.00$, $P = 0.002$). Median Julian day of sampling was slightly earlier during the 1990s (121.45) relative to the 2010s (124.68) across all sex-age classes. Within sex-age classes, these slight differences in timing of sampling were maintained (e.g., adult females with yearlings, Mann-Whitney $U = 104.50$, $P = 0.060$; sub adults, Mann-Whitney $U = 275.0$, $P = 0.02$). Incorporating Julian day of sampling into a logistic regression did not explain variation in body condition amongst most classes of bears with the exception of adult females with cubs-of-the-year (COY) where bears sampled later tended to be in better condition (Appendix E). Amongst adult females with yearlings, Julian day was not a predictor of BCS and condition was better in the 2010s than in the 1990s. For subadults there was no effect of either timing of sampling or epoch on the probability of a bear being in poor versus fair-good condition.

Other Observations

Feeding – During sampling in 2012-2014, 14% of bears encountered showed evidence of recent feeding (excluding dependent offspring). Prevalence of feeding observations was highest amongst adult females with offspring and lowest in adult males and subadults (Table 13.2).

Seals – In 2013, 94 groups consisting of one or live ringed seals, *Phoca hispida*, were observed while searching for bears in KB. Notable concentrations of seals were encountered in north east Kane Basin in front of the Humbolt glacier and inside fiords along eastern Ellesmere Island (Figure 13.1).

Skin Lesions –In 2012 and 2013, 40% of the bears that were captured and physically examined were found to have unusual skin lesions. These were characterized as locally extensive alopecia (hair loss) over the feet, in most cases (75%) affecting all four feet (Figure 13.2). In addition, some of the affected individuals had multi-focal ulcerations on the plantar/palmar heel and digital foot pads and on the dorsal aspects of all 4 feet (Figure 13.3). Discharge from these lesions was purulent and sanguinous. Granulation tissue forming in some of these ulcers indicated they were chronic in nature. Even under anesthesia, some bears exhibited notable discomfort when these ulcers were gently palpated during examination, often reacting by moving the foot or lifting their head. Finally, two individuals (an adult male and a yearling) were found to have mild generalized alopecia over the dorsal neck, thorax and abdomen.

The prevalence of foot lesions was highest amongst adult males (75%) and lowest amongst cubs-of-the-years (0%) (Table 13.3). The prevalence of bleeding ulcerations on the feet, an indication of the severity and/or chronicity of the condition, was highest amongst adult males with 75% exhibiting some degree of ulceration. Also of note were two bears captured in 2012 without lesions that were recaptured in 2013 with lesions.

13.4. Discussion

Body Condition

With the limited data in this study we found evidence of differences in the spring time body condition of KB polar bears between the 1990s and 2010s. Condition amongst adult females with yearlings and subadults was better in 2010s. In contrast, condition amongst adult males, adult females with COY, and lone adult females was similar between these time periods. These findings may be attributable to several factors including bias in the data, natural variation in condition and long-term trends in environmental conditions.

Several sources of bias were possible in our study associated with use of a qualitative body condition score rather than a quantitative metric. BCS data were collected by several observers. In the 1990s, most data were collected by a single observer. In the 2010s all data were collected by a single but different observer. Differences in the assignment of condition scores by these two observers could therefore generate the apparent differences in condition between time periods. Since individual bears were not scored by more than one observer, teasing out potential observer effects is challenging. While we cannot exclude the possibility of observer bias in our study, several lines of evidence suggest that this potential bias is unlikely to account for our results. First, to reduce observer bias we employed (*post-hoc*) a simplified body condition scoring system that required observers to discriminate between bears in poor versus fair-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 accurately discriminating such bears. Second, 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). Finally, we found differences in condition between time periods amongst adult females with yearling and subadult only. If these differences in condition reflect observer bias we would expect this to be evident in all classes of bears.

Body condition scores in the 1990s were collected from bears captured and physically examined. In contrast scores in the 2000s were from bears either captured (33%) or observed from the air without handling (67%). 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 versus fair-good) in our study should have helped to reduce errors in scoring for bears observed from the air.

Another source of error in our study associated with differences in sampling between the 1990s (physical capture) and 2010s (physical capture or 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 (Chapter 5, Appendix B), especially amongst adult males and adult females with offspring. Misclassification was therefore unlikely to explain differences in condition of adult females with yearlings. 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.

Body condition amongst bears likely improves progressively during the spring and early summer as the availability of seals increases (Stirling and Archibald 1977, Smith 1980, Hammill and Smith 1991, Stirling and Øritsland 1995, Pilfold et al. 2012). Sampling in the 1990s tended to occur earlier in the spring than in the 2010s. This difference in the timing of sampling could therefore partially account for the better condition we observed amongst some classes of bears in the 2010s. However, we note that differences in timing although significant were relatively small (i.e., 3-4 day difference in median day of sampling) so the effect on condition data may be minor. Additionally, timing of sampling was not a significant predictor of body condition amongst adult females with yearling or subadults; the two classes of bear in which differences in condition scores were detected. Finally, if timing of sampling were a significant factor we would have expected similar bias in other classes. Interestingly, we found that condition amongst adult females with COY was a function of Julian day of sampling suggesting consistent with the hypothesis that females emerging from maternal dens begin to steadily recover lost body condition in the spring.

Several ecological explanations could explain our findings. Body condition amongst polar bears fluctuates on temporal and spatial scales in response to annual variation in environmental conditions regardless of any underlying long-term trends (Kingsley 1979, Stirling 2002). Our findings may simply reflect this normal variation in condition whereby sampling in the 2010s occurred at a higher point in condition than in the 1990s. However, this would not account for the fact that improved condition was only detected amongst adult females with

yearlings and subadults; since presumably all classes of bears can capitalize on improved environmental conditions. Failure to find differences in condition amongst other age classes may be due to the limited samples sizes, lack of precision in condition scoring and / or bias in our study. We note that condition tended to be better in 2010s amongst all classes of adult females; although only statistically significant for those with yearlings.

As an alternative explanation, differences in body condition between the 1990s and 2010s may reflect long-term changes in environmental conditions. At the southern extent of the polar bears' range, declining condition has been associated with reduced sea-ice cover resulting from climate change (e.g., Stirling et al. 1999, Rode et al. 2010, Rode et al. 2014, Obbard et al. 2016). However, High Arctic polar bears such as those in KB are predicted to be amongst the last members of the species negatively impacted by climate change (Derocher et al. 2004). An initial impact of climate change in KB has been an observed reduction in the extent of multi-year ice and replacement with thinner annual ice (Hamilton et al. 2014, Chapter 4). Such changes in ice regimen are predicted to have a positive effect on polar bears via increased primary productivity and access to prey (Derocher et al. 2004). Our finding of improved body condition amongst adult females and subadults is consistent with this prediction and may be a sign of improved environmental conditions (albeit temporarily). In some subpopulations where effects of climate change have been reported, body condition has been negatively affected to a greater and / or more easily detectable degree amongst the adult female and subadult classes (Obbard et al. 2006, Rode et al. 2010). This suggests that sensitivity to deteriorating environmental conditions varies by sex, age, and reproductive status; presumably as a result of differing nutritional and energetic requirements and / or rates of food intake. Conversely, it is reasonable to assume that the age classes most readily impacted by negative changes in the environment will be the first to respond

positively as conditions improved. This hypothesis is consistent with our finding that condition improved significantly amongst adult females and subadults but not adult males.

Given the limitations of our data set as discussed above, we are unable to confidently resolve between the differing explanations for our finding of improved body condition between the 1990s and 2010s. We therefore urge caution in interpreting these results. Never-the-less, it is reasonable to conclude that there has been no decline in condition in KB.

Feeding Observations

During the 2010s, the proportion of individuals showing signs of having recently fed was lowest amongst adult males. This is consistent with the hypothesis that adult males forgo foraging opportunities during the spring mating period while pursuing estrous females (Cherry et al. 2009, Stirling et al. 2015). The relatively low feeding rates we observed amongst subadults is consistent with the hypothesis that juvenile bears are less successful in hunting. We did not have data on feeding rates during the 1990s sampling period for comparison. Consequently, we are unable to assess potential changes in hunting success that could account for the improved condition observed amongst bears in the 2010s.

Seal Distribution

The distribution of seal observations made during mark-recapture sampling of polar bears was uncorrected for search effort. However, we note that the relative densities of seals along our search tracks was similar to the distribution of the polar bears we encountered (Chapters 10 and 11). Not surprisingly, where we found relatively high numbers of seals, we also found relatively high numbers of bears. The high densities of ringed seals found in north eastern KB at the front

of the Humbolt Glacier is consistent with other studies that have found sea ice in front of tide-water glaciers to be prime breeding habitat for seals (Lydersen et al. 2014). This area of Kane Basin appears to have high productivity and is likely an important feeding area for polar bears. This is consistent with traditional ecological knowledge indicating that the area was a preferred hunting location for Greenlandic Inuit (Born et al. 2011). According to PBSG (1998), Taylor et al. (2001), and Born (pers. obs. 1994 and 1995) both the Greenland and Canadian portions of Kane Basin were mostly mixed annual and multi-year sea ice that appeared to be favourable habitat for polar bears in the 1990s. Ringed seals were common in both eastern and western KB (ibid.). The relatively few polar bears encountered in this region during the 1990s was thus likely the result of hunting pressure rather than habitat suitability (Taylor et al. 2001).

Skin Lesions

We documented hair loss (alopecia) and ulcerations on the feet of polar bears in Kane Basin during capture sampling in 2012 and 2013. These types of lesions have not been reported previously in the literature. Atwood et al. (2015b) documented an alopecia syndrome of unknown etiology amongst polar bears from the Southern Beaufort Sea (SB). However, those lesions were largely confined to the head, neck and shoulders rather than the feet. Additionally, they were most prevalent amongst subadult bears whereas those in KB were most common amongst adult males. Overall prevalence of lesions in KB bears (40% in 2012 and 2013) was higher than peak prevalence observed in the SB (28%). In particular, we note that seventy-five percent of adult males sampled in KB in 2012 and 2013 were affected. Lesions on adult males were also more severe than on other age classes as indicated by the presence of skin ulcerations some of which were apparently very painful. Atwood et al. (2015b) found that bears with

alopecia were in poorer body condition. What impact, if any, the lesions found on KB bears may have had on condition is unknown since sample sizes were too small to support analyses.

Whether the lesions reported in SB and KB share a common etiology (cause) is unknown. Discussions with Inuit hunters from Grise Fiord (near Kane Basin) suggest this phenomenon of hair loss on the feet is well known to occur in spring time (M. Akeagok, J. Kiguktak, D. Akeagok pers. comm.). It is believed that increased rates of movement in spring, when bears are mating and hunting activity is high, result in abrasions to the feet. Dog teams in the Canadian Arctic are well known get similar lesions in the spring when travelling over coarse, icing snow formed by melting and refreezing as temperatures fluctuate throughout the day (D. Iqqaqrialu pers. comm.). The condition may become so severe that some dogs become lame and unable to pull sleds. Snow cover on the sea-ice in KB in 2012 and 2013 was noticeably icy and granular in composition (Figure 13.4) with a hard ice-covered crust on top. It is therefore plausible that the lesions we observed were the result of snow conditions. The finding that prevalence was highest in adult males may be due to their greater weight and the likelihood that they break the ice crust while walking, combined with potentially increased time allocated to travelling in the spring while searching for estrous females¹. Lesions similar to those observed in 2012 and 2013 were not seen on polar bears captured in KB between 1992 and 1997 (M. K. Taylor and E. W. Born, pers. comm.). Whether this is a new phenomenon in KB brought about by changing snow conditions and progressively warmer spring temperatures or an incidental observation in the years we were sampling bears is unknown.

13.5. Literature Cited

Amstrup, S. C., G. M. Durner, I. Stirling, N. J. Lunn, and F. Messier, F. 2000. Movements and

¹ Versus time spent stalking prey or resting

- distribution of polar bears in the Beaufort Sea. *Canadian Journal of Zoology* 78:948–966.
- Amstrup, S. C., I. Stirling, T. S. Smith, C. Perham, and G. W. Thiemann. 2006. Recent observations of intraspecific predation and cannibalism among polar bears in the southern Beaufort Sea. *Polar Biology* 29:997–1002.
- Amstrup, S. C., B. G. Marcot, and D. C. Douglas. 2008. A Bayesian network modeling approach to forecasting the 21st century worldwide status of polar bears. Pages 213–268 in E. T. DeWeaver, C. M. Bitz and L. B. Tremblay, editors. *Arctic sea ice decline: Observations, projections, mechanisms and implications*. Geophysical Monograph Series 180, American Geophysical Union, Washington, DC, USA.
- Atkinson, S. N., and M. A. Ramsay. 1995. The effects of prolonged fasting on the body composition and reproductive success of female polar bears (*Ursus maritimus*). *Functional Ecology* 9:559–567.
- Atkinson, S. N., R. A. Nelson, and M. A. Ramsay. 1996. Changes in body composition of fasting polar bears (*Ursus maritimus*): the effect of relative fatness on protein conservation. *Physiological Zoology* 69:304–316.
- Atwood, T. C., E. M. Peacock, M. A. McKinney, K. Lillie, R. R. Wilson, and S. Miler. 2015a. Demographic composition and behavior of polar bears summering on shore in Alaska: U.S. Geological Survey Administrative Report, 26 pp.
- Atwood, T., E. Peacock, K. Burek-Huntington, V. Shearn-Bochsler, B. Bodenstein, K. Beckmen, and G. Durner. 2015. Prevalence and spatio-temporal variation of an alopecia syndrome in polar bears (*Ursus maritimus*) of the southern Beaufort Sea. *Journal of Wildlife Diseases* 51:48-59.
- Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 in Ø. Wiig, E. W. Born, and G. Garner, editors. *Polar Bears*. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. *Monographs on Greenland, Man and Society* Volume 41. Museum Tusulanum Press, University of Copenhagen, Copenhagen, Denmark.
- Cattet, M. R. L., N. A. Caulkett, M. E. Obbard, and G. B. Stenhouse. 2002. A body condition index for ursids. *Canadian Journal of Zoology* 80:1156–1161.

- Cherry, S. G., A. E. Derocher, I. Stirling, and E. S. Richardson. 2009. Fasting physiology of polar bears in relation to environmental change and breeding behavior in the Beaufort Sea. *Polar Biology* 32:383–391.
- Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. *Integrative and Comparative Biology* 44:163–176.
- Derocher, A. E., and I. Stirling. 1990. Distribution of polar bears (*Ursus maritimus*) during the ice-free period in western Hudson Bay. *Canadian Journal of Zoology* 68:1395–1403.
- Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. *Canadian Journal of Zoology* 73:1657–1665.
- Derocher, A. E., and I. Stirling. 1996. Aspects of survival in juvenile polar bears. *Canadian Journal of Zoology* 74: 1246–1252.
- Fowler, C. W. 1987. A review of density dependence in populations of large mammals. *Current Mammalogy* 1:401–441.
- Fowler, C. W. 1990. Density dependence in northern fur seals (*Callorhinus ursinus*). *Marine Mammal Science* 6:171–196.
- Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. *Marine Mammal Science* 7:123–135.
- Hamilton, S. G., L. Castro de la Guardia, A. E. Derocher, V. Sahanatien, B. Tremblay, and D. Huard. 2014. Projected polar bear sea ice habitat in the Canadian Arctic Archipelago. *PLoS ONE* 9(11): e113746. doi:10.1371/journal.pone.0113746.
- Henriksen, E. O., Ø. Wiig, J. U. Skaare, G. W. Gabrielsen, and A. E. Derocher. 2001. Monitoring PCBs in polar bears: lessons learned from Svalbard. *Journal of Environmental Monitoring* 3:493–498.
- Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology* 29:724–737.
- Lydersen, C., P. Assmy, S. Falk-Petersen, J. Kohler, K. M. Kovacs, M. Reigstad, H. Steen, H. Strøm, A. Sundfjord, Ø. Varpe, W. Walczowski, J. M. Weslawski, and M. Zajackowski. 2014. The importance of tidewater glaciers for marine mammals and seabirds in

- Svalbard, Norway. *Journal of Marine Systems* 129:452–471.
- McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. *Ecology and Evolution* 4:516–527.
- Molnár, P. K., A. E. Derocher, G. W. Thiemann, and M. A. Lewis. 2010. Predicting survival, reproduction and abundance of polar bears under climate change. *Biological Conservation* 143:1612–1622.
- Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. *Nature Communications* 2:1–8.
- Obbard, M. E., M. R. L. Cattet, T. Moody, L. R. Walton, D. Potter, J. Inglis, and C. Chenier. 2006. Temporal trends in the body condition of Southern Hudson Bay polar bears. *Climate Change Research Information Note 3*, Ontario Ministry of Natural Resources, Applied Research and Development Branch, Sault Ste. Marie, Ontario, Canada.
- Obbard, M. E., M. R. L. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (*Ursus maritimus*) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. *Arctic Science* 2:15–32.
- Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. *Marine Mammal Science* 30:169–183.
- Patyk, K. A., C. Duncan, P. Nol, C. Sonne, K. Laidre, M. Obbard, Ø. Wiig, J. Aars, E. Regehr, L. L. Gustafson, and T. Atwood. 2015. Establishing a definition of polar bear (*Ursus maritimus*) health: A guide to research and management activities. *Science of the Total Environment* 514: 371-378.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 in A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. *Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.

- Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. PLoS ONE 7:e41429. doi:10.1371/journal.pone.0041429.
- Ramsay, M. A., R. A. Nelson, and I. Stirling. 1991. Seasonal changes in the ratio of serum urea to creatinine in feeding and fasting polar bears. Canadian Journal of Zoology 69:298–302.
- Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). Journal of Zoology 214:601–634.
- Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. Ecological Applications 20:768–782.
- Rode, K. D., E. Peacock, M. Taylor, I. Stirling, E. W. Born, K. L. Laidre and Ø Wiig. 2012. A tale of two polar bear populations: ice habitat, harvest and body condition. Population Ecology 54:3–18.
- Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. Global Change Biology 20:76–88.
- Rode, K. D., R. R. Wilson, E. V. Regehr, M. St Martin, D. C. Douglas, and J. Olson. 2015. Increased land use by Chukchi Sea polar bears in relation to changing sea ice conditions. PLoS ONE doi:10.1371/journal.pone.0142213.
- Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. Canadian Journal of Zoology 58:2201–2209.
- Stirling, I. 2002. Polar bears and seals in the eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic 55:59–76.
- Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. Journal of Fisheries Research Board of Canada 34:1126–1129.
- Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.

- Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: a review of the evidence. *Global Change Biology* 18:2694–2706.
- Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. *Arctic* 52:294–306.
- Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. *Canadian Journal of Fisheries and Aquatic Sciences* 52:2594–2612.
- Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. *Arctic* 59:261–275.
- Stirling, I., E. Richardson, G. W. Thiemann, and A. E. Derocher. 2008a. Unusual predation attempts of polar bears on ringed seals in the southern Beaufort Sea: Possible significance of changing spring ice conditions. *Arctic* 61:14–22.
- Stirling, I., G. W. Thiemann, and E. Richardson. 2008b. Quantitative support for a subjective fatness index for immobilized polar bears. *Journal of Wildlife Management* 72:568–574.
- Stirling, I., C. Spencer, and D. Andriashek. 2016. Behavior and activity budgets of wild breeding polar bears (*Ursus maritimus*). *Marine Mammal Science* 32:13–37.
- Taylor, M. K., S. Akeagok, D. Andriashek, W. Barbour, E. W. Born, W. Calvert, H. D. Cluff, S. Ferguson, J. Laake, A. Rosing-Asvid, I. Stirling, and F. Messier. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. *Canadian Journal of Zoology* 79:690–709.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. *Ursus* 5:1–66.
- Watts, P. D., and S. E. Hansen. 1987. Cyclic starvation as a reproductive strategy in the polar bear. *Symposia of the Zoological Society of London* 57:305–318.
- Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. *Science Progress* 91:151–173.

Zedrosser, A., B. Dahle, and J. E. Swenson. 2006. Population density and food conditions determine adult female body size in brown bears. *Journal of Mammalogy* 87:510–518.

Table 13.1. Body condition scores assigned to polar bears in Kane Basin during sampling in the 1990s (1992-1997) and 2010s (2012-2014). Within each epoch and sex-age class, the proportion of individuals in the two BCS categories is presented in parenthesis.

Sex-Age Class	Epoch	Body Condition Score		Test Result ¹
		Poor	Fair-Good	
Adult Male	1990s	5 (0.10)	44 (0.90)	$P = 1.000$
	2010s	5 (0.11)	40 (0.89)	
Adult Female (Lone)	1990s	9 (0.28)	23 (0.72)	$\chi^2 = 2.100$
	2010s	5 (0.14)	31 (0.86)	
Adult Female (w/COY ²)	1990s	11 (0.42)	15 (0.58)	$\chi^2 = 1.922$
	2010s	6 (0.24)	19 (0.76)	
Adult Female (w/yearling)	1990s	8 (0.67)	4 (0.33)	$P = 0.015$
	2010s	2 (0.15)	11 (0.85)	
Subadults	1990s	4 (0.29)	10 (0.71)	$P = 0.037$
	2010s	0 (0.00)	16 (1.00)	

¹ Unless otherwise indicated all tests results report values of P for Fisher's exact test (two-tailed)

² Cub-of-the-year (COY)

Table 13.2. Observations of recent feeding amongst polar bears encountered in Kane Basin, 2012-2014. Evidence of feeding includes presence of seal kills and bears encountered with full, pendulous stomachs.

Sex-age class	Proportion Feeding (n)
Adult Male	0.07 (46)
Adult Female (Lone)	0.17 (36)
Adult Female (with offspring)	0.26 (43)
Subadult	0.07 (16)

Table 13.3. Frequency of hair-loss (alopecia) and skin ulcerations on the feet of polar bears captured in Kane Basin, Nunavut, 2012 and 2013.

Sex-Age Class	No. of Individuals Examined	Proportion with Alopecia	Proportion with Ulcerations
Adult Male	12	0.75	0.75
Adult Female (Lone)	9	0.22	0.00
Adult Female (with COY)	7	0.29	0.00
Adult Female (with Yearling)	6	0.50	0.17
Adult Female (with 2-year-old)	1	0.00	0.00
Subadult	4	0.50	0.25
2-year-old	1	0.00	0.00
Yearling	8	0.63	0.25
Cub-of-the-year (COY)	11	0.00	0.00
Total	59	0.40	0.22

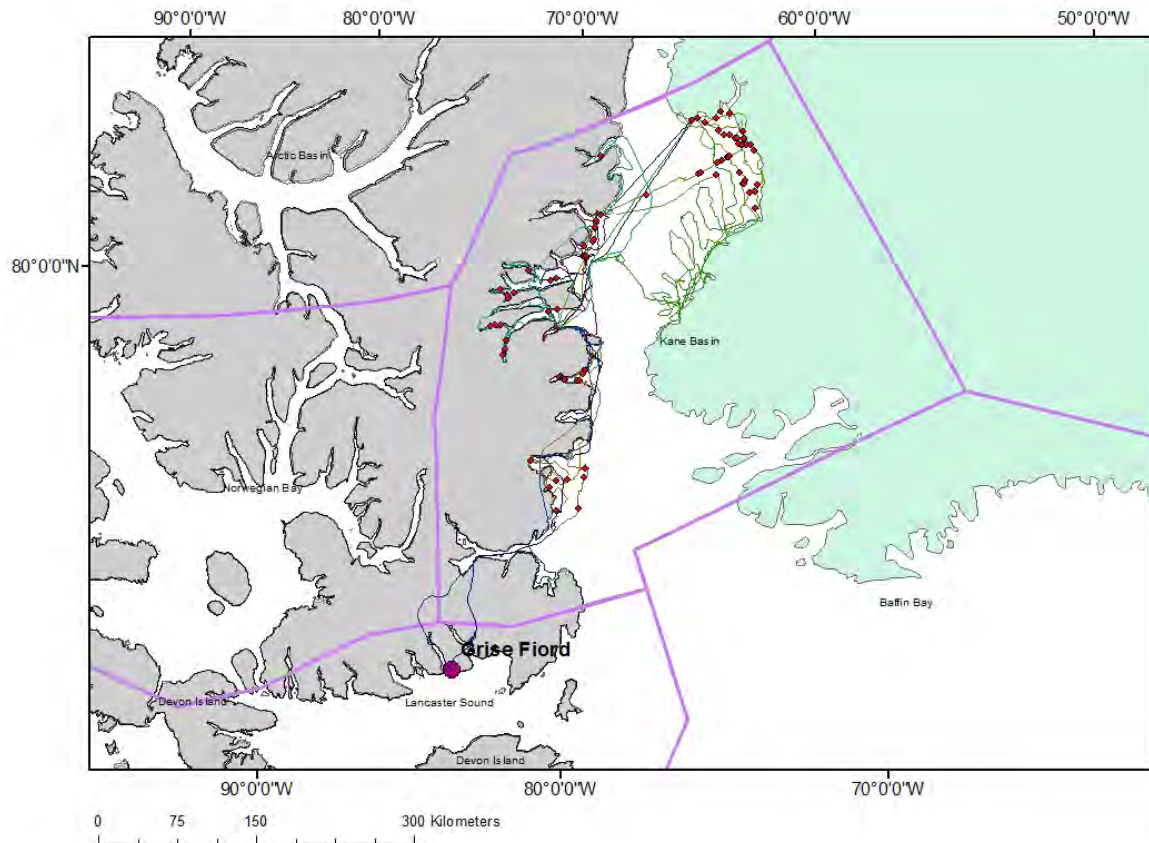


Figure 13.1. Distribution of seal observations during polar bear mark-recapture sampling in Kane Basin, 2013. Flight tracks are shown.



Figure 13.2. Examples of alopecia over the feet of polar bears handled in Kane Basin, April/May 2012 and 2013.

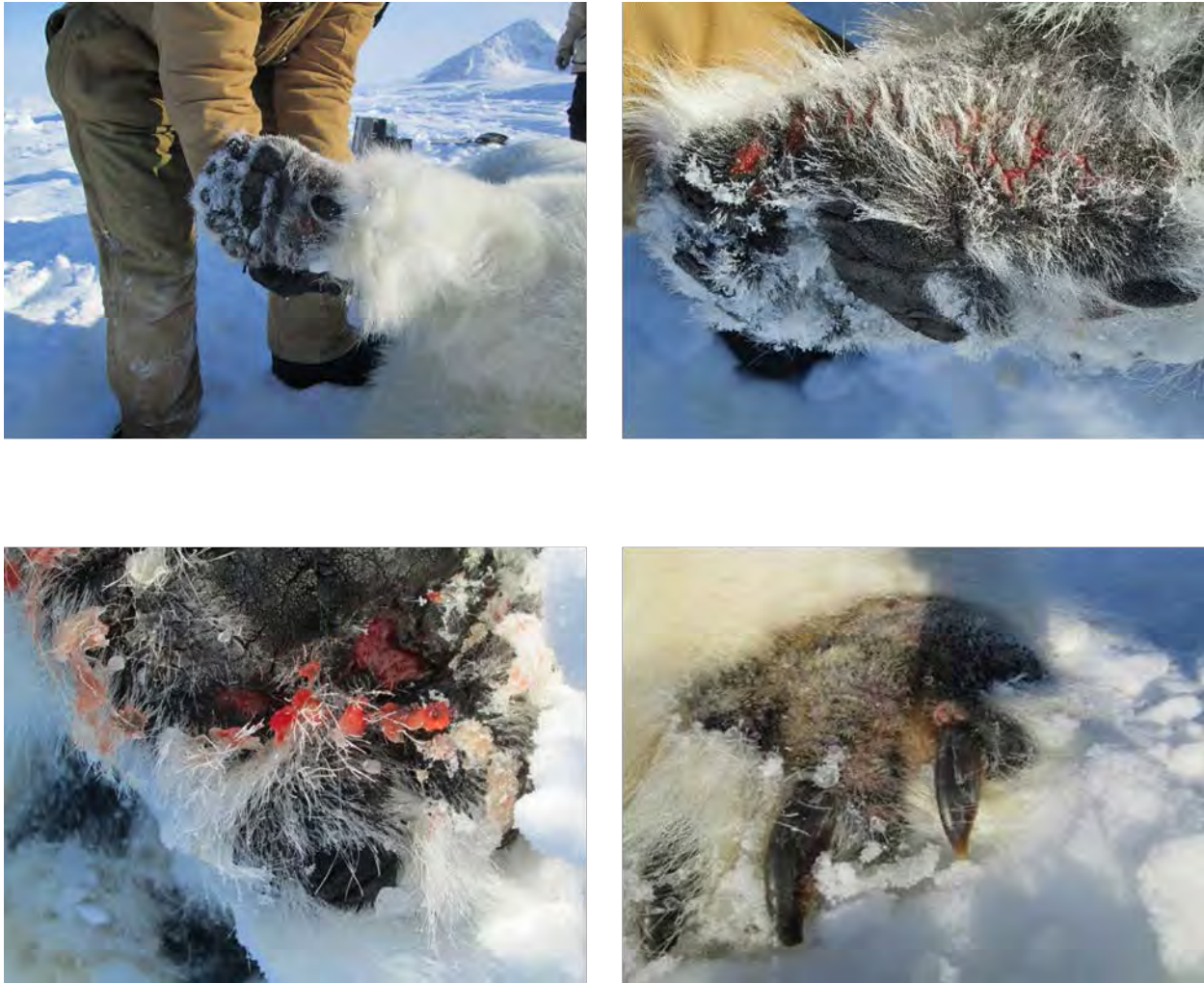


Figure 13.3. Examples of ulcerative lesions on the feet of polar bears handled in Kane Basin, April/May 2012 and 2013.



Figure 13.4. An example of the granular snow found in many parts of Kane basin during polar bear sampling in April/May 2012 and 2013.

CHAPTER 14

GENERAL DISCUSSION AND ADVICE TO THE JOINT COMMISSION

14.1. Conclusions

In 2010, the Canada-Greenland Joint Commission on Polar Bear (JC) tasked the Scientific Working Group with using the best available scientific information to (1) propose Total Allowable Harvest (TAH) levels for the Baffin Bay and Kane Basin subpopulations and provide the JC with a written report of its recommendations, and (2) provide science advice to the JC for monitoring the effects of habitat changes on polar bears. Given the age of the mark-recapture data on which abundance and vital rates for BB and KB were estimated combined with large-scale environmental changes in Baffin Bay during the last decades and suspected large-scale environmental changes in Kane Basin in recent time, the SWG strongly recommended that new estimates of subpopulation abundance, population delineation, and vital rates be given high priority.

Based on the decisions of the JC that physical MR should not be used in this study a multi-year programs began in 2011 (BB) and 2012 (KB) to re-assess the size of both subpopulations using genetic mark-recapture (MR) techniques that involved biopsy sampling from both live and harvested polar bears. In addition, satellite transmitters were deployed on male and female polar bears in NW Greenland during 2009-2013 to study polar bear movement and habitat choice and to gather data for planning and interpretation of the genetic MR study in Baffin Bay. With the same purpose satellite transmitters were deployed on male and female polar bears in both the Canadian (Nunavut) and Greenland parts of Kane Basin in 2012 and

2013. In addition, a helicopter-based aerial survey was flown in Kane Basin in spring 2014 concomitantly with the MR operation to evaluate the feasibility of estimating KB abundance with an aerial survey flown over springtime sea ice.

A total of 2,690 genetic samples were collected from live and harvested polar bears in BB and KB, 125 satellite transmitters were deployed, and 4,160 linear km of transects flown during the aerial survey in KB as part of the overall research program. In addition, many hours were subsequently spent processing samples; analyzing genetic, aerial survey, and sea-ice data; in discussion interpreting the results; and, writing this report.

The study has resulted in new estimates of abundance of polar bears in the Baffin Bay and Kane Basin subpopulations and provided significant and comprehensive information about polar bear ecology and sea-ice dynamics in Baffin Bay and Kane Basin. The results of this program are described in detail in Chapters 2 through 13. They are also summarized concisely in the Executive Summary document.

14.2. Lessons from Genetic Mark-Recapture

The SWG recommended that physical mark-recapture be used for assessing the size of the Baffin Bay and Kane Basin subpopulations. However, as described in Chapter 1 the Canada-Greenland Joint Commission decided to use genetic mark-recapture for assessment.

The Baffin Bay and Kane Basin studies represent the first time that genetic mark-recapture has been implemented at population-wide scales for estimating polar bear abundance and demographic rates. As such, these studies provide valuable information about the utility of genetic mark-recapture as a monitoring tool for assessing polar bear subpopulations ranging over

large areas like Baffin Bay with dynamic sea-ice conditions, and how the technique may be most successfully implemented in the future.

Sampling via biopsy darting was highly successful in yielding tissue samples suitable for genotyping in both Baffin Bay and Kane Basin (Chapters 5 and 10) essential if genetic mark-recapture is to be implemented in other subpopulations.

Moreover, genetic analyses identified that reporting of marked bears in the harvest is incomplete; specifically, decreases in reporting are related to greater marker age, suggesting that the loss of physical markers (loss of ear tags and fading of lip tattoos) over time makes it difficult for hunters to correctly identify marked bears. As such, we encourage the use of genetics for identifying marked bears in the harvest in the future; for those subpopulations not inventoried via genetic mark-recapture, this will require genotyping archived samples as well (see also [3] below).

Collecting samples for genotyping via biopsy darting is generally fast, efficient, and less invasive than physical mark-recapture, since bears are not immobilized (Chapter 5). Because biopsying for genetic MR estimation is less time consuming than handling individual bears during physical MR operations genetic MR has the potential of resulting in more “marks” and “recaptures” which theoretically improve precision of estimates of abundance in MR. In Baffin Bay success in sampling a large number of biopsies in the huge coastal distribution areas within a relatively short time was obtained by using three helicopters (2 along eastern Baffin Island and 1 in NW Greenland) during the same time in fall. Using three helicopters during fall biopsying for several years inevitably increased the costs of the surveys.

In physical mark-recapture information on a recapture is obtained from direct physical inspection of the presence of numbered ear tags and/or a number in the lips. Data on marking

and recapture is therefore readily at hand for analyses. In contrast, during genetic mark-recapture the information on whether a biopsied bear was a “recapture” or not is not obtained until after genetic analyses are conducted in a qualified laboratory. This adds a delay in the analyses. In our case the commitment of the contracted laboratory, one that is recognized worldwide as an expert and of high quality, to undertake other genetic analyses caused a serious delay in processing the polar bear samples. This resulted in a delay in data analyses for the abundance estimation.

However, there were trade-offs with the genetic mark-recapture method that resulted in lack of information that would have been available with a physical capture protocol. Physical mark-recapture provides a wider range of information including estimates of rates of birth and death, detailed age-structure and body condition information, and a suite of physical samples can further inform individual and population status. Because bears had to be identified from the air using genetic MR, there is uncertainty in the age structure especially for younger bears. Overall, physical mark-recapture permits a more comprehensive assessment of population status, as previously recommended by SWG (2010).

Furthermore, even if physical mark-recapture is not used for assessment, some physical capture is necessary for studies at this scale to provide data on movements of bears and habitat use. In this study, 139 bears were captured in West Greenland, and of these 38 adult females with collars informed both the mark-recapture assessment itself (e.g., temporary emigration analyses, range sizes) as well as provided key information on changes in sea-ice habitat use that set the mark-recapture results into context.

Research in the Kane Basin subpopulation during 2014 illustrated that aerial survey and genetic mark-recapture methods can be implemented simultaneously (Chapters 10 and 11) to

generate more comprehensive demographic information and to ensure efficient and representative allocation of sampling effort. In the present study we illustrate that combining the two methods simultaneously is feasible for subpopulations which are surveyed on sea ice during spring and which have a relatively small geographical distribution like the Kane Basin subpopulation. However, using a helicopter like we did with a relatively short range for both biopsy and aerial surveys at the same time may be suboptimal. Using only a single helicopter limits the ability to expand survey effort to offshore polar bear habitat with loose drift ice and open water as demonstrated in our study where a substantial portion of the KB subpopulation's range could not be surveyed (also for safety reasons) leading to an abundance estimate which was negatively biased to an unknown extent.

With genetic mark-recapture, the ability to leverage historical data to improve estimates of survival is limited by the availability of archived samples for genotyping. For both Baffin Bay and Kane Basin, tissues samples suitable for genotyping were available for most – but not all – bears initially marked during research in the 1990s. This lack of tissue samples for a small proportion of the sampled population necessitated identifying those individuals which could still be alive (based on harvest records and age at time of first capture; Chapters 5 and 10) and fixing their recapture probabilities during the 2010s sampling to zero. Although this solution complicated analyses, it enabled us to incorporate historical capture data. Moreover, the large sampling interval between the past and present BB and KB studies did not favor recoveries of old “marks” that also could have assisted in improving some vital rates. If successive population studies are envisaged within a 10-15 year time frame, then a single-year biopsy sampling session should be implemented 5-7 years after the study was completed in order to increase or maintain

marks in the population that can be recovered through either harvest or the subsequent new study.

During physical mark-recapture individual age is obtained from every single bear that is handled and marked. During immobilization a vestigial tooth is extracted and individual age is obtained from reading growth-layers in the cementum. This allow for implementing age-structured models for estimating abundance and vital parameters. The fact that individual ages are not obtained from bears that are biopsied during genetic mark-recapture represents a limitation on *post hoc* analyses as indicated in Chapter 5. Hence, we implemented only coarse age structures for demographic analyses, pooling individuals ≥ 2 years and ≥ 3 years for the Baffin Bay and Kane Basin studies, respectively (Chapters 5 and 10). Thus, survival could not be estimated for 2 year olds and subadults separately from adults. Ancillary data suggest that experienced biologists may be able to discriminate among finer age classes (i.e., 2 – 4 year old subadults versus adults ≥ 5 years) with a high degree of accuracy for adults and a lesser degree for subadults (Appendix B), particularly since the sex of individuals is confirmed via genetics (Appendix B). As such, genetic mark-recapture may have the potential to yield estimates of survival and reproductive output that are relevant to management and comparable to previous research however more work needs to be done (e.g., Taylor et al. 2005, 2008a, Peacock et al. 2013). However, it must be kept in mind that field-estimation of individual age during genetic mark-recapture relies heavily upon the individual researcher's experience in assigning polar bears to more specific age classes.

We completed a detailed review of historical records to assess previous research in Baffin Bay and Kane Basin. These reviews proved critical to our interpretation of results, as apparent changes in the sampling frames between epochs (i.e., incomplete spatial sampling during the

1990s, relative to the 2010s) limited our ability to assess trends in abundance. However, we were not able to locate original and detailed data files from the physical mark-recapture study and telemetry study conducted in Baffin Bay and Kane Basin in the 1990s. This finding underscores the need to archive data securely and for complete and consistent sampling of the study areas. We recommend that any polar bear study archives data securely at institutes that were central in the collaborative studies to allow future studies to thoroughly assess historical inventories to ensure proper interpretation of results. For the same reason, the Canada/Nunavut/Greenland institutes that conducted the recent study in Baffin Bay and Kane Basin have signed a contract where collaborating institutes have access to and must securely archive all data stemming from the joint study.

We defined and stratified the study areas for recent inventories based on recent telemetry data (and historical capture records). For this purpose, data obtained from satellite transmitters deployed in West and Northwest Greenland in 2009 and 2010 for other purposes proved to be an important tool. This process improved our allocation of effort and ensured that results reflected the entire subpopulations as best as possible. Satellite telemetry data (both historical from the 1990s and recent from 2009-2014) were also used to interpret *post-hoc* to what extent the biopsy samplings in the 1990s and 2000s were for the entire subpopulations in BB and KB. Hence, information on movement and area occupancy obtained from satellite telemetry is an essential tool in mark-recapture studies of polar bears.

Finally, the telemetry data identified that 18% of the adult females collared in West Greenland remained in Melville Bay for a least one summer season and in some cases for over a year. There would be value in future work investigating what fraction of the BB subpopulation uses Melville Bay year-round. This could make use of the existing genetic marks from this

study, with the addition of one or two sampling seasons in fall in Melville Bay only. Densities appeared low based on observations during data collection in 2012 and 2013 however there would be value in quantifying this using more detailed sampling of the area.

In conclusion, the present study has shown that a genetic mark-recapture assessment of a polar bear subpopulation at a large scale is possible. However, such a study would likely involve an absolute minimum of three years of sampling and, subsequently, additional time for laboratory analysis of samples, analysis of data, and report writing. Thus, these timelines must be considered in advance if timely information on abundance for management is paramount. In this study, our reporting of abundance of polar bears in Baffin Bay and Kane Basin is given 5+ years after initiation of the study in 2011.

14.3. Recommendations on Sustainable Harvest

Historically, the management of polar bears assumed that their sea-ice habitat was relatively stable and, once subpopulation size was known, conservation of polar bears could be achieved through harvest management (SWG 2011). The sustainable harvest of a subpopulation would, therefore, be largely dependent on an estimation of abundance, demographic rates (e.g., birth and death vital rates), and the magnitude, and sex and age composition of the harvest. Since the 1980s, management and conservation of polar bears, particularly in Canada, has been informed by predictive modelling, which has provided guidance to managers on the setting of sustainable harvest levels that have been based on a better understanding of the risk associated with different harvest scenarios (Taylor et al. 1987b, 2002, 2005). The primary predictive model used, RISKMAN (Taylor et al. 2001), focused on harvest management and did not account for

any future change in survival or birth rates during the period over which the simulations were run. Furthermore it did not include changes in carrying capacity (K).

Based on demographic rates derived from pooled subpopulation data collected across the Canadian High Arctic in the 1970s and 1980s, modeling indicated that under optimal conditions the sustainable yield of adult females is typically <1.6% of the total population (Taylor et al. 1987a). Hence, level of total sustainable catch (C) was inferred from this relationship (PBSG 1998): $C = N * 0.015 / \text{sex ratio in harvest}$; where N is subpopulation size, 0.015 is the sustainable rate of harvest of adult (independent) females from the population, and the denominator is the ratio of adult females in the harvest. Hence, the ratio of adult female bears in the harvest was important for the sustainable total yield. Using this relationship and a general 2:1 sex ratio of males to females in the harvest, the historical standard for the harvest rate of polar bear subpopulations has been 4.5%, which was based on a 2:1 sex ratio of males to females and, initially, on generalized demographic rates derived from pooled subpopulation data collected across the Canadian High Arctic in the 1970s and 1980s (Taylor et al. 1987a, b). More recently, as more studies were undertaken and additional analytical methods developed, it became clear that demographic rates were subpopulation specific (e.g., Amstrup et al. 2001, Taylor et al. 2002, 2005, PBSG 2006, Taylor et al. 2006, Regehr et al. 2007, PBSG 2010). In their recent development of a matrix-based demographic model for polar bears, Regehr et al. (2015) noted that harvest management based on this standard rate and the 2:1 male-biased sex ratio is reasonable under many biological and management conditions; although in some cases, lower or higher rates may be more appropriate.

The ultimate threat to polar bears throughout their range is the reduction in sea-ice habitat expanse, duration, and quality as a consequence of climate change (Derocher et al. 2004, Laidre

et al. 2008, 2015a, Wiig et al. 2015). At their 2009 Meeting in Tromsø, Norway, the Polar Bear Range States agreed that the impacts of climate change constitute the most important threat to polar bear conservation (Polar Bear Range States 2015). Hence, the effective and sound management of polar bears can no longer rely solely on estimates of abundance but must also incorporate impacts of a changing environment (e.g., loss of ice and reduction in carrying capacity). In addition, as other threats (i.e., pollution, resource exploration and development, tourism) become better understood, management of polar bear subpopulations will need to be modified, particularly if reproduction or survival rates are negatively affected (Vongraven and Peacock 2011, Vongraven et al. 2012). Thus, understanding the risks associated with a range of harvest management options is important for polar bear conservation.

One of the stated purposes and objectives of the 2009 Canada-Nunavut-Greenland Memorandum of Understanding is to manage polar bears within the KB and BB management units in order to ensure their conservation and sustainable management into the future (Anon. 2009). The SWG was subsequently tasked with proposing Total Allowable Harvest (TAH) levels for both the Baffin Bay and Kane Basin subpopulations (Chapter 1) but was not provided with any specific guidance on management goals.

To facilitate the ability of the SWG to provide the Joint Commission (JC) with useful recommendations on TAH, the SWG requested that the JC provide:

- 1) A statement of management objectives for each subpopulation,
- 2) Information on the expected frequency and intensity of future monitoring, and
- 3) A statement of risk tolerance with respect to the effects of human caused removals.

As a consequence of not receiving the necessary information from the JC on which to base recommendations on TAH despite repeated requests, the SWG is currently unable to

provide these recommendations. Specific objectives for the management and conservation of BB and KB polar bears are critical as these will largely influence recommendations on TAH, which in turn will ensure that decision makers have all the necessary information available to them. Risk tolerance and management objectives are not decisions to be made by scientific staff but rather by those with management authority for the resource and, ideally, made after consideration of not only subsistence harvest but also other sources of human-caused removals (e.g., human-bear interactions) and after consultation with all stakeholders.

Once the JC provides the requested information, the SWG strongly recommends that subsequent advice on TAH be based on population simulations using models that have the ability to incorporate changing sea-ice conditions (e.g., reductions in carrying capacity) and demographic data quality as part of the overall process to assess risk under different harvest management scenarios. For example, Regehr et al. (2015) developed a state-dependent management framework that linked the demographic model to simulated population assessments, which can be used to estimate the maximum sustainable rate of human-caused removals. It can also be used to calculate a recommended sustainable harvest rate, which Regehr et al. (2015) note is generally lower than the maximum sustainable rate because it is dependent on management objectives, the precision and frequency of population data, and risk tolerance.

14.4. Recommendations on Monitoring Habitat Change on Polar Bears

There have been numerous reports regarding the effects of climate change and in particular the loss of sea ice, on polar bears (Derocher et al. 1994, Laidre et al. 2008, Wiig et al. 2008). Changes in distribution of polar bears in several populations including in BB have been summarized by Stirling and Parkinson (2006). Furthermore, information on local observations

of the effects of climate change on polar bears in BB made by hunters and elders have been presented by Dowsley (2005), Dowsley and Taylor (2006), and Dowsley and Wenzel (2008) for eastern Baffin Island and by Born et al. (2008a, b, 2011) based on interviews with experienced polar bear hunters NW Greenland. These sources indicate that polar bears in BB (and likely also KB) currently are affected by large-scale environmental changes.

Monitoring habitat change will improve our understanding of the relationship between BB and KB polar bears and the environment. It provides insights into how factors such as sea ice and prey abundance and availability affect polar bear distribution and vital rates. The results of the habitat assessment work conducted in Baffin Bay and Kane Basin since 2009, largely based on satellite telemetry data from collared adult female polar bears, have provided a wealth of new information on habitat and habitat change, which directly inform the status of the BB and KB subpopulations (Chapters 2, 4, and 9). The movement information has also assisted in interpreting the BB and KB abundance and demographic data, which were associated with some biases that could be addressed through the use of the telemetry data (Chapter 3).

Vongraven et al (2012) developed a circumpolar monitoring framework for polar bears, which was focused on the sustained long-term monitoring necessary to understand ongoing effects of climate warming and other population-level stressors in order to inform management and policy responses to changing worldwide polar bear status and trends. The current scientific understanding of polar bears and their reliance on sea-ice habitats is the result of long-term monitoring that has been conducted in only a few subpopulations. There is variability in the response of each subpopulation to loss of sea ice, as manifested in this study through the differences in responses in KB and BB despite roughly the same rates of sea-ice loss (Chapters 4 and 9). Therefore it is critical that scientific studies be conducted within the subpopulation(s) of

interest to gain an in depth understanding of the complex ecological ramifications of climate change in that area (e.g., Rode et al. 2014), rather than making assumptions based on studies in other areas that may have different responses.

The BB and KB satellite telemetry studies allowed for comparison and quantification of range use across decades, seasons and months, changes in overlap of the population range over time, shifts in median latitude of bears across seasons, changes in immigration and emigration across subpopulation boundaries, changes in movement rates, shifts in sea ice and terrestrial habitat use and habitat selection, and changes in maternity denning timing, and changes in maternity denning areas and habitat. Though not included in the report, time series from captures in BB and KB provide information on causal links between factors that determine health, nutritional ecology and population-level processes. Analyses have been initiated (using samples from recent BB captures) and are expected to provide new information on feeding and nutritional ecology.

The SWG concludes that future physical capture and satellite tagging studies in BB and KB (collaring and tracking adult female bears for periods of years) will be critical to extending the current time series and informing managers of the impacts of sea-ice habitat loss. Given the large physical changes documented in this study, and clear responses of polar bears in both areas, continuation of the time series of satellite telemetry data will improve our understanding of the impacts of future biotic and abiotic changes on the two subpopulations. The satellite telemetry studies should be conducted on intervals of 10 years or less, with samples sizes roughly equivalent to those collected in the 1991-1997 and 2009-2013 (approximately 40 adult females tracked over a period of several years). Lower sample sizes will make assessments more difficult due to individual variability and lack of model convergence (as seen for KB).

By use of satellite telemetry, the present study documented that a group of adult female polar bears occur year round at glacier fronts in Melville Bay in NW Greenland (Chapter 2). To protect important polar bear habitat the Melville Bay Nature Reserve was established in 1980. All access and hunting within the central zone of the nature reserve is prohibited (Appendix D). The (re-)establishment of a local group of polar bears with affinity to Melville Bay appears to be relatively recent and is likely an effect of the protection places on this important polar bear habitat. Satellite telemetry in the 2000s has also shown females to be denning in the Melville Bay Nature Reserve. When the areas were surveyed during spring 1992 and 1993 very little signs of polar bear activity were observed in the nature reserve and of the 1990s satellite telemetry indicated that adult females did not use the Melville Bay (Chapter 2).

Hence, recent data indicate that the Melville Bay Nature Reserve represents an example of how important polar bear habitat can be protected resulting in polar bears reestablishing groups "locally" in prime habitat. Hence, the nature reserve exemplifies a means of protecting polar bears in the future. However, the broader effect of the nature reserve on polar bears should be followed. This can be done by regularly conducting a genetic mark-recapture estimation of trends in numbers in the local group of bears (baseline genetic data now exist from the present study) and by tracking individuals by use of satellite telemetry at 5-10 years intervals.

Continued assessment of changes in habitat in BB and KB via satellite-based observation of sea ice (passive microwave data, MODIS, or Radar images) provide useful context for changes in the physical environment. Arctic sea ice is the most critical habitat for the survival of polar bear subpopulations as distribution and timing of ice relative to critical phases of polar bear life history have been linked to subpopulation status and trend (Stirling et al. 1999, Hunter et al. 2010, Regehr et al. 2010). The SWG recommends continued monitoring of sea-ice habitat

change through these studies (described herein as well as Stern and Laidre 2016). Furthermore, contrasting changes in BB and KB with other polar bear subpopulations provides an important baseline for comparison.

Continued development and refinement of habitat models will be necessary in the future to identify habitat selection changes and better predict critical habitat in BB and KB.

Standardized methods of developing habitat models (resource selection functions, RSFs) for polar bears have been developed for several subpopulations (Mauritzen et al. 2003, Ferguson et al. 2000, Durner et al. 2004, 2006, Wilson et al. 2014, Laidre et al. 2015b, Chapters 4 and 9) and within a large part of polar bear range (Durner et al. 2009). RSFs are developed from satellite radio telemetry data of adult female bears and readily available sea-ice data in geographic information system (GIS) format. Habitat models are powerful tools for predicting the occurrence of terrestrial den habitat (Howlin et al. 2002, Richardson et al. 2005). Knowledge of the distribution of maternal den habitat has significant management potential to protect polar bears in dens. Trends in sea-ice den habitat may be estimated by monitoring sea-ice conditions as changes in the composition of sea ice has been linked to changes in den distribution (Fischbach et al. 2007).

Habitat availability and change have been linked to polar bear demography and/or condition in some subpopulations (Regehr et al. 2007, Bromaghin et al. 2015, Lunn et al. 2016). Quantitative links between habitat and demographic parameters are complex and need to be refined and specific to the subpopulation of interest. Continued habitat monitoring will improve the understanding between the links to demography and productivity for both BB and KB.

Stable isotope (Bentzen et al. 2007), fatty acid analysis (Iverson et al. 2006), and lipid content in adipose tissues (McKinney et al. 2014) conducted from blood, fat and hair collected

during captures can provide information on the polar bear prey base and help to identify shifts in food webs and body condition in BB and KB. This information can be used in concert with information on movements and habitat use from the telemetry to better inform how bears use the ecosystem. While some of this information can be collected from harvest sampling, physical capture of polar bears provides opportunities that are not available from harvest samples (e.g., collection of samples from the same individuals over time).

Continued monitoring of TEK and LEK in BB and KB will also be critical for providing information on how changes in sea ice are impacting the polar bear hunt (e.g., hunting practices), the overall harvest, and the condition of bears harvested. Local perspectives on changes both to the physical environment and the population are important inputs to managers. Repeated studies with a robust interview study design and data collection process (e.g., Born et al. 2011) are needed.

Finally continued subsistence harvest monitoring is needed in both Canada and Greenland, providing critical information on numbers, sex ratios and ages of bears taken in both areas. In this study, this information provided important content into changes in harvest patterns and composition of the harvest (Chapter 8). Genetic validation of the sex of individual bears (as reported by the hunters) showed that the gender was incorrectly reported in a significant number of cases. Improvement in gender reporting is needed; inaccuracies in gender reporting were greatest in Greenland.

14.5. Literature Cited

Amstrup, S. C., T. L. McDonald, and I. Stirling. 2001. Polar bears in the Beaufort Sea: A 30-year mark-recapture case history. *Journal of Agricultural, Biological, and Environmental Statistics* 6:221–234.

- Anon. 2009. *Memorandum of Understanding between the Government of Canada, the Government of Nunavut, and the Government of Greenland for the Conservation and Management of Polar Bear Populations*, 9 pp.
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2008a. Isbjørne i Nordvestgrønland En interviewundersøgelse om fangst og klima, 2008. Pinngortitaleriffik – Grønlands Naturinstitut Teknisk Rapport Nr. 70, 112 pp. (In Danish and Greenlandic with English abstract).
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2008b. Nannut Kalaallit Nunaata avannaata kitaani – Piniarneq silallu pissusaa pillugit apersuilluni misissuineq, 2006. – Pinngortitaleriffik – Nalunaarusiaq Teknikkimut tunngasoq No. 70, 116 pp. (In Greenlandic with English summary).
- Born, E. W., A. Heilmann, L. K. Holm, and K. L. Laidre. 2011. Polar bears in Northwest Greenland: an interview survey about the catch and the climate. Monographs on Greenland, Man and Society Volume 41. Museum Tusulanum Press, University of Copenhagen, Copenhagen, Denmark.
- Bentzen, T. W., E. H. Follmann, S. C. Amstrup, G. S. York, M. J. Wooller, and T. M. O’Hara. 2007. Variation in winter diet of southern Beaufort Sea polar bears inferred from stable isotope analysis. *Canadian Journal of Zoology* 85:596–608.
- Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. *Ecological Applications* 25:634–651.
- Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2007. *Advanced Distance Sampling: Estimating Abundance of Biological Populations*. Oxford University Press, Oxford, UK.
- Derocher, A. E. 2005. Population ecology of polar bears at Svalbard, Norway. *Population Ecology* 47:267–275.
- Derocher, A. E., and I. Stirling. 1994. Age-specific reproductive performance of female polar bears (*Ursus maritimus*). *Journal of Zoology* 234:527–536.
- Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. *Canadian Journal of Zoology* 73:1657–1665.

- Derocher, A. E., N. J. Lunn, and I. Stirling. 2004. Polar bears in a warming climate. *Integrative and Comparative Biology* 44:163–176.
- Dowsley, M. 2005. Inuit knowledge regarding climate change and the Baffin Bay polar bear population. Nunavut Wildlife Research Group Final Report No. 1, Government of Nunavut, 43 pp.
- Dowsley, M., and M. K. Taylor. 2006. Community consultations with Qikiqtarjuaq, Clyde River and Pond Inlet on management concerns for the Baffin Bay (BB) polar bear population: A summary of Inuit knowledge and community consultations. Nunavut Wildlife Research Group Final Report No. 2, Government of Nunavut, 83 pp.
- Dowsley, M., and G. W. Wenzel. 2008. “The time of the most polar bears”: A co-management conflict in Nunavut. *Arctic* 61:177–189.
- Durner, G. M., S. C. Amstrup, R. Nielson, and T. McDonald. 2004. Using discrete choice modeling to generate resource selection functions for polar bears in the Beaufort Sea. Pages 107–120 *in* S. Huzurbazar, editor. *Resource selection methods and applications*. Omnipress, Madison, Wisconsin, USA.
- Durner, G. M., D. C. Douglas, R. M. Nielson, and S. C. Amstrup. 2006. A model for autumn pelagic distribution of adult female polar bears in the Chukchi Sea, 1987–1994. Contract Completion Report 70181-5-N240. US Geological Survey Alaska Science Center, Anchorage, Alaska, USA.
- Durner, G. M., D. C. Douglas, R. M. Nielson, S. C. Amstrup, T. L. McDonald, I. Stirling, M. Mauritzen, E. W. Born, Ø. Wiig, E. DeWeaver, M. C. Serreze, S. E. Belikov, M. M. Holland, J. Maslanik, J. Aars, D. A. Bailey, and A. E. Derocher. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. *Ecological Monographs* 79:25–58.
- Ferguson, S. H., M. K. Taylor, and F. Messier. 2000. The influence of sea ice dynamics on habitat selection by polar bears. *Ecology* 81:761–772.
- Fischbach, A. S., S. C. Amstrup, and D. C. Douglas. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. *Polar Biology* 30:1395–1405.
- Hammill, M. O., and T. G. Smith. 1991. The role of predation in the ecology of the ringed seal in Barrow Strait, Northwest Territories, Canada. *Marine Mammal Science* 7:123–135.

- Hansen, K. W. 2010. Polar bear management in Greenland, 2005-2008. Pages 133-134 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Howlin, S., M. Stishov, L. L. McDonald, and S. L. Schliebe. 2002. Den site selection by polar bears on Wrangel Island. Pages 127–138 in N. J. Lunn, S. Schliebe, and E. W. Born, editors. Polar Bears: Proceedings of the 13th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Hunter, C. M., H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. *Ecology* 91:2883–2897.
- Iverson, S. J., I. Stirling, and S. L. C. Lang. 2006. Spatial and temporal variation in the diets of polar bears across the Canadian Arctic: indicators of changes in prey populations and environment. Pages 98–117 in I. L. Boyd, S. Wanless, and C. J. Camphuysen, editors. *Top Predators in Marine Ecosystems: Their Role in Monitoring and Management*. Cambridge University Press, Cambridge, UK.
- Laidre, K. L., E. W. Born, P. Heagerty, Ø. Wiig, H. Stern, R. Dietz, J. Aars, and M. Andersen. 2015b. Shifts in female polar bear (*Ursus maritimus*) habitat use in East Greenland. *Polar Biology* 38:879–893.
- Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015a. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology* 29:724–737.
- Laidre, K. L., I. Stirling, L. F. Lowry, Ø. Wiig, M. P. Heide-Jørgensen, and S. H. Ferguson. 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. *Ecological Applications* 18:S97–S125.
- Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. *Ecological Applications* 26:1302–1320.

- Mauritzen, M., S. E. Belikov, A. N. Boltunov, A. E. Derocher, E. Hansen, R. A. Ims, Ø. Wiig, and N. Yoccoz. 2003. Functional responses in polar bear habitat selection. *Oikos* 100:112–124.
- McKinney, M. A., T. Atwood, R. Dietz, C. Sonne, S. J. Iverson, and E. Peacock. 2014. Validation of adipose lipid content as a body condition index for polar bears. *Ecology and Evolution* 4:516–527.
- Molnár, P. K., A. E. Derocher, T. Klanjscek, and M. A. Lewis. 2011. Predicting climate change impacts on polar bear litter size. *Nature Communications* 2:1–8.
- Obbard, M. E., S. Stapleton, K. R. Middell, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. *Polar Biology* 38:1713–1725.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 1998. Status of the polar bear. Pages 23–44 in A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. *Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2006. Status of the polar bear. Pages 33–55 in J. Aars, N. J. Lunn, and A. E. Derocher, editors. *Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- PBSG [IUCN/SSC Polar Bear Specialist Group]. 2010. 2009 status report on the world's polar bear subpopulations. Pages 31–80 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. *Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group*. IUCN, Gland, Switzerland and Cambridge, UK.
- Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. *Journal of Wildlife Management* 77:463–476.
- Pilfold, N. W., A. E. Derocher, I. Stirling, E. Richardson, and D. Andriashek. 2012. Age and sex composition of seals killed by polar bears in the eastern Beaufort Sea. *PLoS ONE* 7:e41429. doi:10.1371/journal.pone.0041429.
- Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.

- Ramsay, M. A., and I. Stirling. 1988. Reproductive biology and ecology of female polar bears (*Ursus maritimus*). *Journal of Zoology* 214:601–634.
- Regehr, E. V., C. M. Hunter, H. Caswell, S. C. Amstrup, and I. Stirling. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. *Journal of Animal Ecology* 79:117–127.
- Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. *Journal of Wildlife Management* 71:2673–2683.
- Regehr, E. V., R. R. Wilson, K. D. Rode, and M. C. Runge. 2015. Resilience and risk – a demographic model to inform conservation planning for polar bears. U.S. Geological Survey Open-File Report 2015-1029. U.S. Geological Survey, Reston, Virginia, USA.
- Richardson, E., I. Stirling, and D. S. Hik. 2005. Polar bear (*Ursus maritimus*) maternity denning habitat in western Hudson Bay: a bottom-up approach to resource selection functions. *Canadian Journal of Zoology* 83:860–870.
- Rode, K. D., S. C. Amstrup, and E. V. Regehr. 2010. Reduced body size and cub recruitment in polar bears associated with sea ice decline. *Ecological Applications* 20:768–782.
- Rode, K. D., E. V. Regehr, D. Douglas, G. Durner, A. E. Derocher, G. W. Thiemann, and S. M. Budge. 2014. Variation in the response of an Arctic top predator experiencing habitat loss: feeding and reproductive ecology of two polar bear populations. *Global Change Biology* 20:76–88.
- Smith, T. G. 1980. Polar bear predation of ringed and bearded seals in the land-fast sea ice habitat. *Canadian Journal of Zoology* 58:2201–2209.
- Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. *Marine Mammal Science* 32:181–201.
- Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. *The Cryosphere Discussion Papers*, in review.
- Stirling, I., and W. R. Archibald. 1977. Aspects of predation of seals by polar bears in the eastern Beaufort Sea. *Journal of Fisheries Research Board of Canada* 34:1126–1129.
- Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. *Arctic* 52:294–306.

- Stirling, I., and N. A. Øritsland. 1995. Relationships between estimates of ringed seal and polar bear populations in the Canadian Arctic. *Canadian Journal of Fisheries and Aquatic Sciences* 52:2594–2612.
- Stirling, I., and C. L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. *Arctic* 59:261–275.
- SWG [Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear]. 2011. Survey options for assessment of the Baffin Bay (BB) and Kane Basin (KB) polar bear populations. Unpublished report submitted by the Scientific Working Group to the Canada-Greenland Joint Commission on Polar Bear to the Canada-Greenland Joint Commission on Polar Bear, 32 pp.
- Taylor, M., F. Bunnell, D. DeMaster, R. Schweinsburg, and J. Smith. 1987b. ANURSUS: a population analysis system for polar bears (*Ursus maritimus*). *International Conference on Bear Research and Management* 7:117-125.
- Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987a. Modelling the sustainable harvest of female polar bears. *Journal of Wildlife Management* 51:811–820.
- Taylor, M. K., J. Laake, H. D. Cluff, M. Ramsay, and F. Messier. 2002. Managing the risk from hunting for the Viscount Melville Sound polar bear population. *Ursus* 13:185–202.
- Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. *Arctic* 58:203–214.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2006. Demographic parameters and harvest-explicit population viability analysis for polar bears in M’Clintock Channel, Nunavut, Canada. *Journal of Wildlife Management* 70:1667-1673.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2008b. Mark-recapture and stochastic population models for polar bears of the high arctic. *Arctic* 61:143–152.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, and F. Messier. 2009. Demography and population viability of polar bears in the Gulf of Boothia, Nunavut. *Marine Mammal Science* 25:778-796.

- Taylor, M., M. Obbard, B. Pond, M. Kuc, and D. Abraham. 2001. RISKMAN: Stochastic and deterministic population modeling RISK MANagement decision tool for harvested and unharvested populations. Government of Nunavut, Iqaluit, Nunavut, Canada.
- Vongraven, D., and E. Peacock. 2011. Development of a pan-Arctic monitoring plan for polar bears: background paper. Circumpolar Biodiversity Monitoring Programme, CAFF Monitoring Series Report No. 1, CAFF International Secretariat, Akureyri, Iceland.
- Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. *Ursus* 5:1–66.
- Wiig, Ø., J. Aars, and E. W. Born. 2008. Effects of climate change on polar bears. *Science Progress* 91:151–173.
- Wiig, Ø., S. Amstrup, T. Atwood, K. Laidre, N. J. Lunn, M. Obbard, E. Regehr, and G. Thiemann. 2015. *Ursus maritimus*. The IUCN Red List of Threatened Species 2015: e.T22823A14871490. <http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en>.
- Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA.
- Wilson, R. R., J. S. Horne, K. D. Rode, E. V. Regehr, and G. M. Durner. 2014. Identifying polar bear resource selection patterns to inform offshore development in a dynamic and changing Arctic. *Ecosphere* 5(10):136. doi:10.1890/ES14-00193.1

APPENDICES

APPENDIX A. List of Scientific Working Group members and external experts involved in the re-assessment of the Baffin Bay and Kane Basin polar bear subpopulations.

Scientific Working Group Members	External Experts
Stephen N. Atkinson Government of Nunavut Canada	Todd Arnold University of Minnesota USA
Erik W. Born Greenland Institute of Natural Resources Greenland	Markus Dyck Government of Nunavut Canada
Kristin L. Laidre University of Washington and Greenland Institute of Natural Resources Greenland	Eric V. Regehr US Fish and Wildlife Service USA
Nicholas J. Lunn Environment and Climate Change Canada Canada	Seth Stapleton University of Minnesota USA
Øystein Wiig University of Oslo Norway	Harry Stern University of Washington USA

APPENDIX B. The accuracy of estimating polar bear age-class and sex from helicopter-based, aerial observations: Implications for the use of non-invasive survey methods in monitoring subpopulations.

INTRODUCTION

Although some subpopulations of polar bears (*Ursus maritimus*) have been studied and monitored for more than 4 decades (e.g., Stirling et al. 1977, Lunn et al. 2016), there is considerable variation in the depth and scope of knowledge across their circumpolar range (Vongraven et al. 2012). Large gaps in basic information exist. Growing concerns about the impacts of climate change, increasing industrial development, harvest and contaminants in the Arctic have prompted range state governments, researchers, environmental organizations and local communities to call for an enhanced and coordinated circumpolar monitoring effort (Vongraven et al. 2012, Range States 2015).

Polar bears typically have been monitored by means of physical mark-recapture. This method has yielded detailed demographic data, allowing researchers to assess the status of subpopulations and closely examine the impacts of climate change and other threats. Hence, physical mark recapture provides tissue samples and samples of e.g., blood and milk which have been used in a wealth of studies of health and pollution in polar bear populations (e.g., Regehr et al. 2007, Hunter et al. 2010, Sonne 2010, Peacock et al. 2013, Bromaghin et al. 2015, Lunn et al. 2016). However, gaps in knowledge and the demand for rapid dissemination of up-to-date information have generated interest in the use of alternative methods for monitoring polar bears. Aerial surveys have proven to be an effective and expedient way of assessing abundance in polar bear subpopulations even in subpopulations with a large range (Aars et al. 2009, Stapleton et al.

2014, 2016). Genetic mark-recapture (Herreman and Peacock 2013, Pagano et al. 2014, Chapters 5 and 10) and remote sensing (Stapleton et al. 2014) may also offer viable alternatives in situations where logistical and financial limitations preclude the use of physical mark-recapture or where concerns about the impacts of handling bears outweigh the benefits (Vongraven et al. 2012).

Aerial surveys are widely used for monitoring wildlife populations. Unlike physical mark-recapture studies where individuals are captured, marked and released, aerial surveys do not provide detailed demographic data such as estimates of birth rates and survival that can be used to project population growth. Nevertheless, aerial surveys, which only rely on one season are an effective means of yielding a snapshot of estimates of abundance/status. Inferences about population trends can be derived from repeated aerial surveys. In addition to estimating abundance, aerial surveys can also provide information on spatial distribution, sex and age composition, body condition and reproductive performance that can be used to facilitate population status assessment (e.g., Stapleton et al. 2014). In contrast to genetic and physical mark-recapture estimation, which relies on several years of sampling, aerial surveys can provide an estimate of abundance from only one season of study.

Genetic mark-recapture has been increasingly used for wildlife population monitoring (Palsbøll et al. 1997, Boersen et al. 2003, Boulanger et al. 2004, Kendall et al. 2009). Like aerial surveys, genetic mark-recapture does not require the capture and physical handling of individuals. Tissue samples are collected for genotyping and identification of individuals by methods such as biopsy darting or hair-snagging. With protocols such as biopsy darting, individuals can be observed from a distance, facilitating collection of additional information on

sex, age class and body condition. However, the reliability of this ancillary information depends on the ability to correctly classify individuals by sex and age class.

Using data collected during two recent genetic mark recapture studies of polar bears in the Baffin Bay (BB) and Kane Basin (KB) subpopulations, we assessed the accuracy of classifying polar bears into sex and age classes from the air without physical handling. From a sample of bears of known sex and age class, we examined variation in accuracy of classification dependent on the method of survey (aerial vs genetic mark-recapture) and amongst sex-age classes. We discuss the implications of the results for expanding the utility of aerial survey and genetic mark-recapture as less invasive methods for monitoring species status.

METHODS

Study Area

The boundaries of the BB polar bear subpopulation (BB) encompass an area ~1 million km² 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). A small number of bears remain on land in northwestern Greenland throughout the ice-retreat period.

The KB polar bear subpopulation covers roughly 150,000 km² and spans portions of Nunavut, Canada, including Ellesmere Island, as well as northwestern Greenland (Taylor et al. 2008). However, the boundaries of the KB subpopulation encompasses a substantial amount of land and glaciers so that the essential sea ice polar bear habitat only amounts to less than one half of the area enclosed by the borders of the management unit; *cf.* Figure 13.1). The subpopulation ranges over Kane Basin, Nares Strait, Smith Sound and adjacent fjords on eastern Ellesmere Island and Northwest Greenland (the Qaanaaq area). It is bounded to the north by the Arctic Basin subpopulation (via the Kennedy Channel), to the south by the BB and LS subpopulations, and to the west by Norwegian Bay (NW). Kane Basin forms part of the Arctic archipelago ecoregion (Amstrup et al. 2008); sea ice remains present in the northern range (i.e., Nares Strait-Kane Basin) throughout the year, largely due to the movement of polar pack ice from Arctic Basin, and reaches a minimum in late summer. However, in recent decades, sea ice conditions in KB have changed markedly (Chapter 9)

Genetic Mark-Recapture Study

Genetic mark-recapture studies were conducted in BB and KB between 2011-2013 and 2012-2014 respectively (Chapters 5 and 10 in this report). Sampling of bears in BB occurred from late August to mid- October along the east coast of Baffin Island and around Bylot Island, Canada. During this period, bears were on land in a variety of habitats ranging from flat coastal plains and beaches to steep rocky slopes and glaciers. Bears were observed against different backgrounds including sand, rocks, low lying vegetation, snow and water. Sampling in KB

occurred in April and May while bears were on the sea ice. Bears were observed on a range of sea-ice types including flat, shorefast ice, consolidated pack-ice and unconsolidated pack-ice.

Using a helicopter (Bell 206 LongRanger), we searched for and biopsy darted polar bears using methods described previously (Chapters 5 and 10). Upon encounter, the sex and age class (cub-of-the-year [COY], yearling, subadult [ages 2 – 4], and adult) of each bear was estimated from the air at a range of 3 – 7 m above ground. The individual identity and sex of each bear was later confirmed via genetic analysis (Chapter 5 and 11). 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 the tail of females). Field notes, and in some cases photographs, also assisted in *post-hoc* reassessment of age-sex class once genetic sex was known. All observations were made by experienced polar bear biologists who had previously participated in physical mark-recapture studies.

At the time of encounter, observers had no prior knowledge of the sex or age class of bears. However, amongst the individuals encountered, a proportion were of known sex and age class based on one or more lines of evidence (Table B1), including a number of bears whose genotypes matched those of bears handled during physical mark-recapture studies in BB (Taylor et al. 2005), KB (Taylor et al. 2008, Chapter 10 in this report) and Davis Strait (Peacock et al. 2013). We assessed the accuracy of remote classification using this sample of ‘known’ bears and examined two scenarios. The first scenario simulated the outcome of an aerial survey in which the sex of bears cannot be confirmed via genotyping. Sex and age classification under this

scenario therefore relies solely on field observation. The second scenario simulated a genetic mark-recapture, whereby inaccuracies in field sexing of bears can be corrected following genotyping and field notes made at the time of observation can be used to make *post-hoc* adjustments to age class once genetic sex is known¹.

We restricted our analyses to bears that were sub adults or adults at time of encounter due to small sample sizes for COYs and yearlings and because these dependent offspring can be easily identified when part of a family group (> 96% and 91% accurate for COY and yearlings respectively; GN unpublished data from Davis Strait). Our analysis was a simple comparison of the estimated and known frequencies of bears in each sex and age class under these two scenarios.

RESULTS

During genetic mark-recapture studies in BB and KB, 2011-2014, there were 309 encounters with individuals classified from the air as adult females based on the presence of accompanying offspring (either COY or yearlings), including 29 instances in which the sex and age class of the adult female was also known from capture and physical examination (n = 2) and tooth aging (n = 27) on a prior or future occasion. Twenty-five adult females were subject to aerial classification during fall when they were accompanied by COY (n = 12) or yearlings (n = 4) and 4 were classified during spring (3 with COY, 1 with yearlings). All adult females with dependent offspring were correctly classified from the air.

In addition to adult females with accompanying offspring, we recorded 128 unencumbered bears of known sex and age class (Table B2). Aerial classification of these bears without subsequent genotyping and reclassification based on genetic sex (i.e., the aerial survey scenario) resulted in an overall accuracy of 73%. For lone adults, 95% males and 74% of females were

¹ An example of a field note used for post-hoc adjustment of age class would be: “*If not female is a sub adult male*”.

correctly classified. Inaccuracies were greatest amongst subadult bears. Although 70% of subadults were correctly classified, only 23% were correctly classified as sub adults of a particular sex.

Aerial classification combined with subsequent reclassification based on genetic sex, field notes and photographs (i.e., the genetic mark-recapture scenario) resulted in an overall accuracy of 91% amongst the 128 known age, independent bears. Again, accuracy varied by sex and age class (Figure B1); accuracy was highest for adult males (97%) and lowest for sub adult females (79%) (Table B3).

DISCUSSION

One of the criteria used to classify adult females was the presence of dependent offspring (COY or yearling) at the time of aerial observation or during a prior encounter. Use of this criterion was based on the assumption that accuracy in identifying females with offspring of this age was at, or near 100%. Although the sample size was relatively small, our results support this assumption and the validity of this age classification criterion. All of the adult females with offspring whose age could also be confirmed by tooth aging or physical examination were correctly classified from the air. However, we did not have any adult females accompanied by 2-year-olds in our sample of known-aged bears so we were unable to test the accuracy of classifying adult females based on the presence of 2-year-old offspring nor were we able to test accuracy in classifying 2-year-olds themselves.

Our results suggest that experienced observers can estimate the sex and age class of bears from the air with high accuracy for most sex and age classes, particularly when aerial observations are combined with genetic sexing, field notes and photographs. These findings are consistent with results from a similar study using a larger dataset of known-age bears ($n = 445$

based on tooth aging) from Davis Strait in which aerial observers correctly classified 97%, 88%, 80% and 80% of adult males, adult females, subadult males and subadult females respectively (GN unpublished data).

Not surprisingly, the greatest inaccuracies occur in classifying subadult bears. While the ability to classify an individual as a subadult is reasonably good from the air, the ability to determine the sex of subadult bears based on aerial observation alone is poor. Another area of potential inaccuracy that we were unable to test was the identification of independent yearlings. In some polar bear subpopulations, a proportion of yearlings are found alone during the summer or fall; presumably having been weaned (Derocher and Stirling 1995, 2012, Stirling et al. 1999). In Baffin Bay during the 1990s, approximately 6% of yearlings encountered during mark-recapture sampling were independent (GN unpublished data). The extent to which weaning of yearlings is occurring in BB at present is unknown. Of 16 bears recaptured as yearlings during genetic mark-recapture sampling from 2011 to 2013, all were still with their mother, but this small sample size limits inferences. However, in Western Hudson Bay, the proportion of yearlings that are independent during the fall has declined dramatically from > 81% prior to 1980 to almost zero at present (Stirling and Derocher 2012). This decline in early weaning of offspring has occurred in association with changing sea-ice conditions leading to the suggestion that early weaning is associated with favorable environmental conditions. Given trends in sea ice in BB (Laidre et al. 2015), a reduction in the proportions of independent yearlings may also be occurring. The number of independent yearlings encountered during our genetic mark-recapture was likely negligible.

Based on the accuracy of classification documented in this study, we conclude that the sex and age class data derived from aerial surveys or genetic mark-recapture studies can provide

reliable data to support monitoring and assessment of population status. From aerial observations, adult males, adult females, COY, and yearlings can be identified with high accuracy. Subadult age classes also can be accurately determined, but classification by sex is poor. Combined with genetic sexing, field notes and photographs, accuracy is improved for all classes of bears. While we cannot derive specific ages for bears from aerial observations, such data do support monitoring the basic age structure of subpopulations. In addition, with genetic mark-recapture, there is an opportunity to model survival of specific age classes, albeit with a degree of uncertainty. For example, remote classification of sex and age classes does not permit modeling senescent age classes, nor can we model the transition from subadult to adult age classes with certainty. Finally, given the accuracy in identifying adult females and their dependent COY and yearling offspring, reproductive indices such as litters size and recruitment (yearlings per adult female) can be reliably obtained.

In our genetic mark-recapture studies in Baffin Bay and Kane Basin approximately one-third and two-thirds of sampling, respectively, occurred for bears of known age class based on our criteria (Table B1). The higher proportion of known age bears in Kane Basin was the result of physical captures completed to deploy satellite telemetry instruments. For the two thirds and one third of bears of ‘unknown’ age that were age classed based on aerial observations, genetic sex, field notes and photographs we can be confident in the accuracy of those classifications. For mark-recapture analyses, we adopted a coarser age class structure than was assessed in the present study due to concerns about the ability to remotely classify bears (Chapters 5 and 10). Our findings suggest that the accuracy of remote classification is sufficient to justify the use of finer scale age-class structures in the future.

LITERATURE CITED

- Aars, J., T. A. Marques, S. T. Buckland, M. Andersen, S. Belikov, A. Boltunov, and Ø. Wiig. 2009. Estimating the Barents Sea polar bear subpopulation size. *Marine Mammal Science* 25:35–52.
- Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. *Wildlife Society Bulletin* 31:197–207.
- Boulanger, J., S. Himmer, and C. Swan. 2004. Monitoring of grizzly bear population trends and demography using DNA mark-recapture methods in the Owikeno Lake area of British Columbia. *Canadian Journal of Zoology* 82:1267–1277.
- Bromaghin, J. F., T. L. McDonald, I. Stirling, A. E. Derocher, E. S. Richardson, E. V. Regehr, D. C. Douglas, G. M. Durner, T. Atwood, and S. C. Amstrup. 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. *Ecological Applications* 25:634–651.
- Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. *Ursus* 10:449–453.
- Derocher, A. E., and I. Stirling. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. *Canadian Journal of Zoology* 73:1657–1665.
- Herreman, J., and E. Peacock. 2013. Polar bear use of a persistent food subsidy: insights from non-invasive genetic sampling in Alaska. *Ursus* 24:148–163.
- Hunter, C. M., H. Caswell, M. C. Runge, E. V. Regehr, S. C. Amstrup, and I. Stirling. 2010. Climate change threatens polar bear populations: a stochastic demographic analysis. *Ecology* 91:2883–2897.
- Kendall, K. C., J. B. Stetz, J. Boulanger, A. C. Macleod, D. Paetkau, and G. C. White. 2009. Demography and genetic structure of a recovering grizzly bear population. *Journal of Wildlife Management* 73:3–17.
- Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology* 29:724–737.
- Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016.

- Demography of an apex predator at the edge of its range – impacts of changing sea ice on polar bears in Hudson Bay. *Ecological Applications* 26:1302–1320.
- Pagano, A. M., E. Peacock, and M. A. McKinney. 2014. Remote biopsy darting and marking of polar bears. *Marine Mammal Science* 30:169–183.
- Palsbøll, P. J., J. Allen, M. Bérube, P.J. Clapham, T. P. Feddersen, P. S. Hammond, R. R. Hudson, H. Jørgensen, S. Katona, A. H. Larsen, F. Larsen, J. Lien, D. K. Mattila, J. Sigurjónsson, R. Sears, T. Smith, R. Sponer, P. Stevick, and N. Øien. 1997. Genetic tagging of humpback whales. *Nature* 388:767–769.
- Peacock, E., M. K. Taylor, J. Laake, and I. Stirling. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. *Journal of Wildlife Management* 77:463–476.
- Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.
- Regehr, E. V., N. J. Lunn, S. C. Amstrup, I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. *Journal of Wildlife Management* 71:2673–2683.
- Sonne, C. 2010. Health effects from long-range transported contaminants in Arctic top predators: An integrated review based on studies of polar bears and relevant model species. *Environment International* 36:461–491.
- Stapleton, S., S. A. Atkinson, D. Hedman, and D. Garshelis. 2014a. Revisiting Western Hudson Bay: Using aerial surveys to update polar bear abundance in a sentinel population. *Biological Conservation* 170:38–47.
- Stapleton, S., M. A. LaRue, N. Lecomte, S. N. Atkinson, D. Garshelis, C. Porter, and T. Atwood. 2014b. Polar bears from space: Assessing satellite imagery as a tool to monitor *Ursus maritimus*. *PLoS ONE* doi:10.1371/journal.pone.0101513.
- Stapleton, S., E. Peacock, and D. Garshelis. 2016. Aerial surveys suggest long-term stability in the seasonally ice-free Foxe Basin (Nunavut) polar bear population. *Marine Mammal Science* 32:181–201.
- Stirling, I., and A. E. Derocher. 2012. Effects of climate warming on polar bears: A review of the evidence. *Global Change Biology* 18:2694–2706.

- Stirling, I., N. J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. *Arctic* 52:294–306.
- Stirling, I., C. Jonkel, P. Smith, R. Robertson, and D. Cross. 1977. The ecology of the polar bear (*Ursus maritimus*) along the western coast of Hudson Bay. Canadian Wildlife Service Occasional Paper 33.
- Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. *Arctic* 58:203–214.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- Vongraven, D., J. Aars, S. Amstrup, S. N. Atkinson, S. Belikov, E. W. Born, T. D. DeBruyn, A. E. Derocher, G. Durner, M. Gill, N. Lunn, M. E. Obbard, J. Omelak, N. Ovsyanikov, E. Peacock, E. Richardson, V. Sahanatien, I. Stirling, and Ø. Wiig. 2012. A circumpolar monitoring framework for polar bears. *Ursus* 5:1–66.

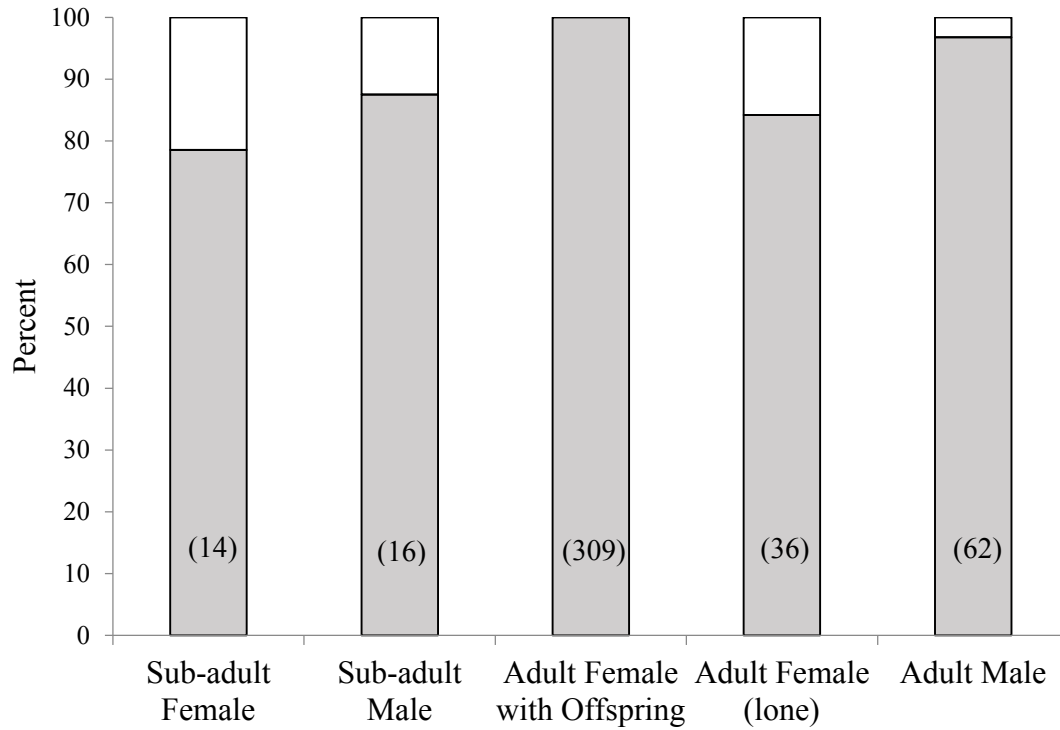


Figure B1. Accuracy of estimating the age class of bears during genetic mark-recapture studies in Baffin Bay and Kane Basin (2011-2014). Data are for bears of known age-class and sex. Sex is based on genotyping and age class is based on one or more of the criteria listed in Table A1. Data are presented as percentages correctly (grey) and incorrectly (white) classified with sample sizes in parentheses.

Table B1. Lines of evidence (criteria) used to determine the ‘known’ sex and age class of polar bears.

Sex-Age Class	Evidence Used to Determine Class
Subadults	<ul style="list-style-type: none"> • Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior or future occasion. Age at capture determined by tooth¹. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be subadult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth¹.
Adult Females with dependent offspring (COY or yearling)	<ul style="list-style-type: none"> • Accompanied by dependent offspring at time of encounter or during previous encounter and / or capture. • Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior or future occasion. Age at capture determined by tooth¹. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth¹.
Lone Adult Females	<ul style="list-style-type: none"> • Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior occasion or future occasion. Age at capture determined by tooth¹. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to an adult female previously accompanied by dependent offspring. • Genotype matched to a bear subsequently harvested for which age was determined by tooth¹.
Adult Males	<ul style="list-style-type: none"> • Genotype matched to an individual previously encountered and / or captured as a COY or yearling. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by tooth¹. • Genotype matched to an individual captured on a prior occasion. Age at capture determined by physical examination and the interval between capture and biopsy confirms the individual to be adult. • Genotype matched to a bear subsequently harvested for which age was determined by tooth¹.

¹ Age estimated by counting annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998).

Table B2. Comparison of sex and age classes as estimated from the air versus known sex and age for bears observed in Baffin Bay and Kane Basin, Canada, 2011-2014. Estimated sex and age based on aerial observation only (aerial survey scenario). Data are frequencies. Percentage of bears correctly classified are presented in parentheses.

		<i>Known</i>					
		<u>Sub adult (<5yrs)</u>		<u>Adult</u>			
		Female	Male	Female (with offspring)	Female (Lone)	Male	
<i>Estimated</i>	Sub adult (<5yrs)	Female	3 (21.4)				
		Male	3	4 (25.0)		1	
		Unknown/Not recorded	5	6			
	Sub adult or adult	Female			1		
		Male			2	1	
		Unknown/Not recorded			1		
	Adult	Female (with offspring)			309 (100)		
		Female (Lone)	3	4		28 (73.7)	1
		Male		2		4	59 (95.2)
		Total Individuals	14	16	309	36	62

Table B3. Comparison of sex and age classes as estimated from the air versus known sex and age for bears observed in Baffin Bay and Kane Basin, Canada, 2011-2014. Estimated sex and age based on aerial observation and post-hoc correction for genetic sex (genetic mark-recapture scenario). Data are frequencies. Percentage of bears correctly classified are presented in parentheses.

		<i>Known</i>					
		Sub adult (<5yrs)		Adult			
		Female	Male	Female (with offspring)	Female (Lone)	Male	
<i>Estimated</i>	Sub adult (<5yrs)	Female	11 (78.6)				
		Male		14 (87.5)		1	
		Unknown/Not recorded					
	Sub adult or adult	Female				4	
		Male					1
		Unknown/Not recorded					
	Adult	Female (with offspring)			309 (100)		
		Female (Lone)	3			32 (88.9)	
		Male		2			60 (96.8)
	Total Individuals		14	16	309	36	62

APPENDIX C. Regression results for an annual body condition metric for polar bears in Baffin Bay (BB). 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%. Data for all years were collected within a standardized sampling area (see chapter 3). 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	59.89	0.97	0.001	Quadratic (2 nd order)
Adult Male	Year	18.90	0.79	0.007	Linear
Adult Female (alone)	Spring Ice Transition	0.25	0.05	0.635	Linear
Adult Female (alone)	Year	2.09	0.29	0.208	Linear
Adult Female (with offspring)	Spring Ice Transition	51.77	0.91	0.001	Exponential
Adult Female (with offspring)	Year	13.24	0.73	0.015	Exponential
Yearling	Spring Ice Transition	9.75	0.83	0.029	Quadratic (2 nd order)
Yearling	Year	4.71	0.49	0.082	Linear

LITERATURE CITED

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

APPENDIX D. Description of the polar bear harvest management and monitoring systems in Canada and Greenland.

HARVEST MANAGEMENT AND MONITORING IN CANADA

Within Canada, harvesting of polar bears has been managed on a quota system since 1967 (Lee and Taylor 1994). This system has undergone several revisions over time both in terms of the size of quotas and the methods of management. Since 1996, the quota system for BB, KB and other neighboring subpopulations managed by the Government of the Northwest Territories (and now the Government of Nunavut) has had several notable features. The term quota has been replaced by the term Total Allowable Harvest (TAH) to reflect language in the Nunavut Land Claims Agreement (NLCA). However, TAH and quota are often used interchangeably in reports. The TAH is managed on a flexible system that allows the annual level of harvest to vary somewhat from year-to-year to some degree. Each community hunting from a subpopulation is allocated a portion of the TAH. Exceedance of that allocation in a given year is compensated for by a reduction in the community's allowable harvest the following year. Exceedances occur when the total number of bears harvested is greater than the available limit or when too many females are harvested. This sex selective harvest management system is based on a target sex ratio of 2 or more males for every female harvested which allows a higher sustainable harvest than a 1:1 sex ratio (Taylor et al. 2008b). All human-caused mortalities are counted against the available TAH. Adult females with dependent offspring (cubs-of-the-year, yearling or two-year-olds) and those in or constructing dens are protected from hunting. The hunting season runs from July 1st to June 30th the following year. Most hunting is for subsistence purposes by Inuit. However, a portion of each community's TAH may be allocated to guided

sport hunting by non-Inuit, at the discretion of Inuit (Tyrell 2009; Wenzel 2008, 2011). Methods of hunting are regulated under the *Nunavut Wildlife Act*. With the exception of sport hunters who must travel by dog-team only, hunting may be facilitated by use of various forms of transportation including snow machine, All Terrain Vehicle or boat and with firearms of authorized calibers.

In Canada (Nunavut), the reporting of all harvested bears is mandatory. Hunters are required to provide evidence of the sex in the form of a baculum from harvested males. In the absence of a baculum, sex may be determined by DNA (Prior to 2005, proof of sex could also be established by sworn affidavit). When proof of sex is unavailable a harvested bear is counted as a female for quota management purposes. In addition to proof of sex, hunters are required to submit a set of standard specimens from each harvested bear including the lower jaw (or skull) for extraction of a tooth for aging, lip tattoos and ear tags if present. Payment is provided by the Government of Nunavut (GN) for these specimens.

Following harvest of a bear, hunters submit the required specimens and other information to local GN Conservation Officers. For each bear, details are recorded including location of harvest, date, hide length, estimated age, sex, type of hunt (e.g., regular subsistence, sport hunt, defense-of-life-and-property), ear tag number (and tags) if present and lip tattoo number (and tattoo) if present. The hide from each harvested bear is then marked with a uniquely numbered tag (hide seal) that is permanently affixed. This hide seal is required for export of hides from Nunavut and sale.

The information collected on each harvested bear is recorded on Hunter Kill Return (HKR) forms completed by Conservation Officers. HKR forms are submitted to the GN's polar bear management program where they were checked for omissions and errors before entry into

the GN's polar bear harvest database. The sex of harvested bears is again verified by submission of a baculum or by DNA submitted for analysis to Wildlife Genetics International (Nelson, BC, Canada). The age of harvested bears is determined from counts of annular rings in an extracted vestigial premolar tooth (Calvert and Ramsay 1998).

HARVEST MANAGEMENT AND MONITORING IN GREENLAND

Harvest management

Following the signing of the *Agreement on Conservation of Polar Bears* in 1973 (Anon. 1973), regulations for the harvest and the protection of polar bears for all areas of Greenland were introduced and were enforced by 1 January 1975 (Anon. 1976, Vibe 1985). Since then, several amendments have been made to the regulations (*cf.* Born 1995). The latest amendment was made in 2005 (Anon. 2005).

When Greenland Home Rule was established in 1979, Greenland took over the legal responsibility for management of its renewable resources, including polar bears. In October 2005, a new Executive Order (Anon. 2005) came into force. Some important protective measures in this executive order are (Anon. 2005, Lønstrup 2006, Hansen 2010):

- year round protection of all cubs (regardless of age) and females accompanied by cubs. The executive order also introduces a prohibition of the export of polar bear cubs;
- protection of all polar bears from 1 July to 31 August; in the local authority districts of Ittoqqortoormiit og Ammassalik from 1 August to 30 September;
- prohibition to disturb or dig out polar bears in dens;
- introduction of quotas from 1 January 2006 and the possibility that part of the quota may be used for trophy hunting. There has never been and currently is no trophy

hunting of polar bears in Greenland. Special provisions on trophy hunting will be laid down in a separate executive order;

- only Greenland residents who hunt as a full-time occupation are allowed to hunt polar bears;
- it is mandatory to report to the Greenland management authorities all catches including struck-and-lost polar bears;
- aircraft, helicopters, motorized vehicles, including snow scooters and boats larger than 20 GRT/15GT are not allowed in the hunt or for transportation to and from the hunting grounds;
- poison, traps, foot snares or self-shooting guns are not allowed;
- rim-fire rifles, shot guns or semi- or fully automatic weapons are not allowed. Polar bears may only be hunted using a rifle with a minimum caliber of 30.06 (7.62 mm);
- all meat, skin and other useable parts of the bear must be brought back (or cached in the field for later use); and,
- no parts of the polar bear must be sold until the catch has been officially registered and the license has received an official stamp.

The Melville Bay Nature Reserve offers protection of polar bears in the Baffin Bay subpopulation. This reserve (10 500 km²) was established in 1980 to protect important polar bear habitat. All hunting within the central (coastal) zone I of nature reserve is prohibited (Vibe 1985, Anon. 1989).

The Greenland Home Rule Act. No. 12 of 29 October 1999 provides the legal framework for wildlife management. Various laws on environmental protection and animal welfare also apply to the management of polar bears (Polar Bear Range States 2015).

Responsibility for the management of polar bears resides with the Department of Fisheries, Hunting and Agriculture, DFHA (Aalisarnermut, Piniarnermut Nunalerinermullu Naalakkersuisoqarfik, APNN) of the Greenland Government.

Quotas for the take of polar bears in Greenland were introduced in 2005 taking effect 1 January 2006 (Lønstrup 2006). The Minister of Fisheries, Hunting and Agriculture sets an annual polar bear quota. The minister drafts a preliminary regional allocation of the quota based on the latest scientific advice and harvest results for the preceding harvest season, and then sends the draft to the Hunters' National Association, the municipalities, the Ministry of Nature and Environment and the Greenland Institute of Natural Resources for consultation for a period of not less than five weeks. Based on the resulting consultation, the Ministry of Fisheries, Hunting and Agriculture prepares a final presentation of the total annual quota to the Minister (Polar Bear Range States 2015). Licenses to hunt polar bears are issued by the municipalities, within annual quotas set by DFHA and the National Government.

The Greenland Fisheries License Control Authority is tasked with enforcing the regulations set by the government and the municipalities (Polar Bear Range States 2015).

When polar bear studies conducted by the Greenland Fisheries Research Institute (predecessor of Greenland Institute of Natural Resources, Nuuk) were initiated in 1991, Greenland de facto took over the responsibility providing scientific data for the management of its polar bear subpopulations as outlined in the 1973 *Agreement on Conservation of Polar Bears*.

During the fall of 2000, the Greenland Home Rule Government signed a Memorandum of Understanding (MOU) with the Government of Nunavut (Canada). An appendix to this MOU contains a prioritized list of items, including that there should be cooperation between both regarding shared polar bear subpopulations (Lønstrup 2006).

Harvest Monitoring

Home Rule was established in Greenland 1979. Since then the Department of Fisheries, Hunting and Agriculture (DFHA, Nuuk) has been responsible for organizing the collection of catch statistics in Greenland and for summarizing and publishing the data. Until 1987 information about the number of polar bears taken in Greenland was available through the Hunters' Lists of Game (HLG) where hunters reported their catch of various wildlife including polar bears voluntarily. The HLG (Anon. 1954-83 and unpublished 1984-87) was based upon the principle that an appointed person from each settlement kept count of the catch of various hunting animals by all the hunters in his settlement and reported the numbers to the authorities (Rosing-Asvid 2002). When such reports for some reason were missing, the central authority added an estimate to account for unreported catch based upon "other information" (i.e., for example notices of catches in newspapers or records of trade of skin etc.). The HLG-summaries of the catch, including estimates of unreported catch, were published annually by the Ministry for Greenland in Copenhagen (until 1983). After the Greenland Home Rule Government took over the HLG-system, unpublished summaries of the catch in 1985, 1986 and 1987 became available from the Department of Fisheries, Hunting and Agriculture, DFHA (Nuuk). However, the reliability of the reporting in the HLG deteriorated since about the mid-1970s (Born 1995, Rosing 1998) and this way of monitoring the catch in Greenland stopped in 1987.

Some information (HLG and trade) was available during the decade prior to the introduction of a new system of reporting catches in 1993 (see the following). However, none of these sources gave the total picture and the size of the annual catch of polar bears from the BB and KB subpopulations in the 1980s and the information of annual catches was largely based on estimates (Born 1995). Born (1995) and Rosing-Asvid (2002) estimated that during 1980-1992 a

total of between 25(30) and 70(80) polar bears were taken each year by Greenlanders from the BB and KB subpopulations (the estimates of the total catch was not separated to BB and KB).

On January 1993 a new system of reporting catches – the "Piniarneq" (Greenlandic word for "catch") – was introduced in Greenland on 1 January 1996. The "Piniarneq" relies upon each hunter voluntarily reporting his annual catch (between 1 October and 30 September) of various species including polar bears. The Piniarneq system is linked to the issuing of hunting licenses, of which two categories exist: one for full-time hunters and another for part-time hunters.

Hunters in both categories have to pay a small fee for renewal of the license, at which time they are obliged to report their catches during the previous 12 months. Only full-time hunters can get a license to hunt polar bears. The information on catches is compiled by the Department of Fishery, Hunting and Agriculture (Nuuk) which publishes summaries of the annual catch by area. In the summaries, the catch is reported by municipality, meaning that in "Piniarneq" there are no records of the exact site of kill. There is also no information on sex and age of the caught bears or whether the bear(s) was (were) killed during a hunt involving more hunters than the one reporting the catch(es).

In recognition of the fact that a potential problem of reporting catches of polar bears via Piniarneq might be that more than one hunter reports the catch of a bear because several hunters participated in the hunt leading to multiple-reporting of a kill (see Discussion) it became mandatory from 1994 to report every kill of a polar bear on specific forms ("Special Reporting Forms"; "Særmeldingsskemaer"). On these forms the hunter's name, civil registration number, settlement/town, place and date of the kill, sex and approximate age (young, adult, old) of the bear has to be given. To be able to reduce the problem of potential multi-reporting of a single kill the hunter who finished off the bear was supposed to fill in the form and also give the name etc.

of the other participants in the catch. During the years this system worked with variable success and not all bears that were caught were reported (Born 1998).

Quotas on polar hunting were first introduced in Greenland in 2006. After the introduction of quotas taking effect 1 January 2006 the Ministry of Fisheries, Hunting and Agriculture has improved the hunting statistics by developing a new database and a double reporting system. This means that a hunter must be issued a license before the hunt and immediately following the hunt the hunter must report the catch to the local authority using a standardized form. This standardized form includes information on the name of the hunter(s), place of residence, date, license number, location of kill, and the sex and age category, and whether the bear was marked. As an additional control, all hunters must report their annual harvest of all species (including polar bears) in Piniarneq (Hansen 2010).

In connection with studies of movement and subpopulation assessment more than 1500 polar bears have been physically marked in the Kane Basin and Baffin Bay subpopulations since the 1970s (Born 1995, Taylor et al. 2005, 2008, Peacock et al. 2012). Before the quotas the hunters received a token payment for returning marks (and transmitters) to the authorities with information on the bear (date, site, sex etc. etc). By the introduction of quotas it became mandatory to report whether a bear was marked or not and return tags (and transmitters) (Anon. 2005).

Since the 1980s biological samples from the polar bear catch (various tissues, sexual organs, teeth for ageing etc.) have been collected during various specific programs in connection with studies of pollution and the demography of the catch (e.g., Rosing-Asvid 2002, Sonne et al. 2012). However, these programs which relied upon the hunters collecting the samples with an

economical compensation served specific purposes and in case of monitoring the catch have been intermittent (Rosing-Asvid 2002).

In order to continuously monitor the Greenland catch of polar bears demographically and provide information on sex and age composition of the catch it became mandatory in 2012 for the polar bear hunters to deliver a tissue sample (for genetic analyses) and a small vestigial tooth (for age determination) from each bear killed. The samples shall be sent to the Greenland Institute of Natural Resources in Nuuk which is responsible for arranging the sampling program practically and for processing the samples. Each sample is accompanied by a filled form where with details about the catch (date, site, name of hunter, sex of the bear and its approximate age etc. etc.). The hunters are required to also send the same information to the Greenland management authorities (i.e., the Department of Fisheries Hunting and Agriculture in Nuuk).

LITERATURE CITED

- Anon. 1954-87. Sammendrag af Grønlands Fangstlister. Ministeriet for Grønland [Summaries of “Hunters’ Lists of Game”]. Summarized during 1954-1984 by the Danish Ministry of Greenland, Copenhagen, and during 1985-1987 by the Ministry for Fisheries, Hunting and Agriculture, Nuuk. [In Danish].
- Anon. 1973. *Agreement on the Conservation of Polar Bears*. Available at: <http://pbsg.npolar.no/en/agreements/agreement1973.html>.
- Anon. 1976. Bekendtgørelse om ændring af bekendtgørelse om fredning af isbjørne i Grønland [Executive order of preservation of polar bears in Greenland]. Nalunaerutit-Grønlandsk Lovsamling, Serie A, Nr. I, MJ.G.j. nr. 1701-D8: 109. [In Danish].
- Anon. 1989. Hjemmestyrets bekendtgørelse nr. 21 af 17. maj 1989 om naturreservatet i Melville Bugt [Greenland Home Rule Executive order No. 21 on the Melville Bay Nature Reserve]. http://www.lovgivning.gl/gh.gl/love/dk/1989/bkg/bkg_nr_21-1989_dk/bkg_nr_21-19,05-01-2010. [In Danish].

- Anon. 2005. Hjemmestyrets bekendtgørelse nr. 21 af 22. september 2005 om beskyttelse og fangst af isbjørne [Greenland Home Rule Executive order No. 21 of 22 September 2005 on protection and harvest of polar bears], 5 pp. [In Danish].
- Born, E. W. 1995. Status of the polar bear in Greenland. Pages 81–103 in Ø. Wiig, E. W. Born, and G. Garner, editors. Polar Bears. Proceedings of the 11th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Born, E. W. 1998. Research on polar bears in Greenland, primo 1993 to primo 1997. Pages 23–44 in A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Calvert, W., and M. A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. *Ursus* 10:449–453.
- Hansen, K. W. 2010. Polar bear management in Greenland, 2005-2008. Pages 133-134 in M. E. Obbard, G. W. Thiemann, E. Peacock, and T. D. DeBruyn, editors. Polar Bears: Proceedings of the 15th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Lee, J., and M. Taylor. 1994. Aspects of the polar bear harvest in the Northwest Territories, Canada. *International Conference on Bear Research and Management* 9:237–243.
- Lønstrup, J. 2006. Polar bear management in Greenland. Pages 133–134 in J. Aars, N. J. Lunn, and A. E. Derocher, editors. Polar Bears: Proceedings of the 14th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Peacock, E., J. Laake, K. L. Laidre, E. W. Born, and S. N. Atkinson. 2012. The utility of harvest recoveries of marked individuals to assess polar bear (*Ursus maritimus*) survival. *Arctic* 65:391–400.
- Polar Bear Range States. 2015. Circumpolar Action Plan: Conservation Strategy for Polar Bears. A product of the representatives of the parties to the 1973 *Agreement on the Conservation of Polar Bears*.
- Rosing, B. 1998. Issues pertaining to polar bear management in Greenland. Pages 93-95 in A. E. Derocher, G. W. Garner, N. J. Lunn, and Ø. Wiig, editors. Polar Bears: Proceedings of

- the 12th Working Meeting of the IUCN/SCC Polar Bear Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
- Rosing-Asvid, A. 2002. The polar bear hunt in Greenland. Greenland Institute of Natural Resources Technical Report 45, 25 pp.
- Sonne, C., R. J. Letcher, T. Ø. Bechshøft, F. F. Rigét, D. C. G. Muir, P. S. Leifsson, E. W. Born, L. Hyldstrup, N. Basu, M. Kirkegaard, and R. Dietz. 2012. Two decades of biomonitoring polar bear health in Greenland: A review. *Acta Veterinaria Scandinavica* 54 (Supplement 1):S15 doi:10.1186/1751-0147-54-S1-S15.
- Taylor, M. K., J. Laake, P. D. McLoughlin, E. W. Born, H. D. Cluff, S. H. Ferguson, A. Rosing-Asvid, R. Schweinsburg, and F. Messier. 2005. Demography and viability of a hunted population of polar bears. *Arctic* 58:203–214.
- Taylor, M. K., J. Laake, P. D. McLoughlin, H. D. Cluff, E. W. Born, A. Rosing-Asvid, and F. Messier. 2008a. Population parameters and harvest risks for polar bears (*Ursus maritimus*) of Kane Basin, Canada and Greenland. *Polar Biology* 31:491–499.
- Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008b. Sex-selective harvesting of polar bears *Ursus maritimus*. *Wildlife Biology* 14:52–60.
- Tyrrell, M. 2009. Guiding, opportunity, identity: the multiple roles of the Arviat polar bear conservation hunt. Pages 25–38 in M. M. R. Freeman, and L. Foote, editors. *Inuit, Polar Bears and Sustainable Use: Local, National and International Perspectives*. Canadian Circumpolar Institute Press, Edmonton, Alberta, Canada.
- Vibe, C. 1985. Miscellaneous information relating to polar bears in Greenland. Pages 44–57 in *Polar Bears: Proceedings of the 8th Working Meeting of the IUCN/SCC Polar Bear Specialist Group*. IUCN, Gland, Switzerland.
- Wenzel, G. 2008. Sometimes hunting can seem like business: Polar bear sport hunting in Nunavut. Canadian Circumpolar Institute Press, Edmonton, Alberta, Canada.
- Wenzel, G. W. 2011. Polar bear management, sport hunting and Inuit subsistence at Clyde River, Nunavut. *Marine Policy* 35:457–465.

APPENDIX E. Results of binary logistic regressions for body condition scores (poor, fair-good) for polar bears in Kane Basin.

Sex-Age Class	Independent Variables	<i>B</i>	Wald Statistic	<i>P</i>
Adult male	Julian Day	-0.095	1.617	0.204
	Epoch (1990s-2010s)	0.230	0.082	0.774
Adult female (lone)	Julian Day	0.104	2.413	0.120
	Epoch (1990s-2010s)	0.775	1.152	0.283
Adult female (with COY)	Julian Day	0.251	9.210	0.002
	Epoch (1990s-2010s)	0.327	0.194	0.660
Adult female (with yearling)	Julian Day	0.082	0.617	0.432
	Epoch (1990s-2010s)	2.064	4.081	0.043
Subadult	Julian Day	0.111	0.741	0.389
	Epoch (1990s-2010s)	20.056	0.000	0.998