Conservation of Arctic Flora and Fauna

CIRCUMPOLAR BIODIVERSITY MONITORING PROGRAM

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DEVELOPMENT OF A PAN-ARCTIC MONITORING PLAN FOR POLAR BEARS

BACKGROUND PAPER

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Foreword

Polar bears are a powerful Arctic icon – symbolizing a remote, often misunderstood region of the planet. However, the value of polar bears goes beyond that of an icon – they hold significant cultural and spiritual value for Inuit peoples, who have shared the Arctic with this species for millennia. As a top predator, they are a critical element of the Arctic marine environment, exerting a direct influence on a number of prey species and thereby, indirectly, having a cascading effect on the entire marine food chain. Loss of this species from the Arctic marine ecosystem could have dramatic and unpredictable impacts.

Despite the sustained attention given to polar bears and concerns raised regarding the impacts of climate change, we have only limited baseline information on most populations and a poor understanding of how polar bears have and will continue to respond to a rapidly changing climate. Effective conservation actions require not only an understanding of polar bear trends across the Arctic, but a clear understanding of the mechanisms driving those trends. To achieve this understanding, an efficient and coordinated pan-arctic research and monitoring effort is urgently needed. Indeed, such a plan was called for in the March, 2009 Meeting of the Parties to the 1973 Agreement on Polar Bears.

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1. Introduction

Polar bears (*Ursus maritimus*), by their very nature, and the extreme, remote environment in which they live, are inherently difficult to study and monitor. Monitoring polar bear populations is both arduous and costly and, to be effective, must be a long-term commitment. There are few jurisdictional governments and management boards with a mandate for polar bear research and management, and many have limited resources. Although population monitoring of polar bears has been a focus to some degree within most jurisdictions around the Arctic, of the 19 subpopulations recognised by the IUCN/ Species Survival Commission Polar Bear Specialist Group (PBSG), adequate scientific trend data exist for only three of the subpopulations, fair trend data for five and poor or no trend data for the remaining 11 subpopulations (PBSG 2010a). There are especially critical knowledge gaps for the subpopulations in East Greenland, in the Russian Kara and Laptev seas, and in the Chukchi Sea, which is shared between Russia and the United States. The range covered by these subpopulations represents a third of the total area (approx. 23 million km^2) of polar bears' current range, and more than half if the Arctic Basin is included. If we use popular terms, we know close to nothing about polar bears in this portion of their range.

As summer sea-ice extent, and to a lesser degree, spring-time extent, continues to retreat, outpacing model forecasts (Stroeve et al. 2007, Pedersen et al. 2009), polar bears face the challenge of adapting to rapidly changing habitats. There is a need to use current and synthesised information across the Arctic, and to develop new methods that will facilitate monitoring to generate new knowledge at a pan-Arctic scale. The circumpolar dimension can be lost when efforts are channelled into regional monitoring. Developing and implementing a plan that harmonises local, regional and global efforts will increase our power to detect and understand important trends for polar bears, with particular emphasis on how climate warming may differentially affect populations and habitats. Current knowledge is inadequate for a comprehensive understanding of the present and future impact of climate warming and its interaction with other stressors. The cumulative effects are unknown (Laidre et al. 2008). An integrated pan-Arctic research and monitoring plan will improve the ability to detect future trends, identify the most vulnerable subpopulations and guide effective conservation. There is a need to direct attention and resources where data are deficient to understand the mechanisms that drive trends, and to facilitate more effective and timely conservation response.

1. The term "subpopulation" is determined by IUCN terminology and criteria (IUCN 2010); this term is used interchangeably with management-units. We use the term 'population' more generally, or more specifically when referring to population-based methods, which assume demographic and geographic closure.
2. Review presented by the PBSG at the 2009 Meeting of the Parties to the 1973 Agreement on the conservation of polar bears, in Tromsø March 2009.


Elizabeth Peacock, US Geological Survey

1.1 Project objectives

Our objectives are to provide background science to support the development of:

1. a circumpolar polar bear monitoring plan, to be adopted across the Arctic that:
   - identifies the monitoring techniques and optimal sampling regimes that are likely to succeed in the 19 different subpopulations, given specific characteristics and logistics of the subpopulations themselves;
   - identifies suites of metrics that can provide parallel lines of evidence of the status of polar bear populations, where intensive research is not possible;
   - identifies standardized parameters for intensively researched subpopulations, with a specific focus on identifying factors responsible for determining mechanistic relationships and trends in population;
   - identifies new methods, including less-invasive approaches, for conducting directed research and monitoring, recognizing the need for more effective monitoring; and,
   - develops population projection models that incorporate response to environmental change.
2. a set of circumpolar indices and indicators to provide regular, consistent and credible reports on the status and trends of individual polar bear subpopulations.

1.2 Definition of monitoring

Monitoring of animal populations in its most strict sense implies precise, methodical and repetitive measurements of biological parameters, or metrics, which enable us to accurately describe any rate of change in the monitored parameter. Further, we must identify parameters that meaningfully represent the health or status of a population or its individuals. Although such monitoring is a laudable goal, it is incredibly demanding. For polar bears, even identifying appropriate biological parameters is difficult, as the relationships between fundamental ecological processes and biological parameters are largely unknown (Amstrup et al. 2008). Further, such a definition of monitoring describes an ideal situation. For polar bears, financial costs, international boundaries, ethical debates over methods, and the remote nature of the Arctic conspire to reduce the breadth and depth of conceivable monitoring. Monitoring at a level that provides adequate information for science-based management, which aims to mitigate the threats to polar bears, will likely be impossible. Consequently, we suggest ideal metrics and, in addition, a variety of other metrics and methods for monitoring, which can aid in the circumpolar monitoring of polar bear populations.

1.3 Adaptive management/implementation

A scientific monitoring plan should be followed by implementation under an adaptive management framework (Walters and Holling 1990). However, there are several federal and regional government jurisdictions, and aboriginal co-management boards that are responsible for the management of polar bears. These jurisdictions have different management goals and various protocols regarding the use of scientific data. For example, in Norway, polar bear harvest does not occur, but in Greenland, polar bear harvest is an important subsistence activity. As another example, in Nunavut, Canada, the use of traditional ecological knowledge (TEK) is required in making management decisions, but in the other jurisdictions scientific data are solely considered. In this document we do not consider the use of local knowledge or TEK as it is beyond our expertise. For similar reasons, we also do not consider human dimensions; as wildlife scientists, as opposed to wildlife managers (Enck et al. 2006), we do not propose a management plan.

A recommended adaptive management framework is therefore beyond the scope of this paper. We provide a perspective on which scientific parameters would best reflect the status of polar bear populations and, most importantly, how to measure these parameters using science. That said, considering the importance of human dimensions, especially as they relate to harvest, harvest monitoring and invasive research, we have included a section on the importance of community-based monitoring (of scientific parameters) and also the need to develop less invasive techniques. However, there is a great need to establish sound and practical links between monitoring actions/results and management decisions. Further, conservation management plans can only be successful with the integration of science with human dimensions (Servheen 1998), which, in the Arctic, will require collection of TEK. We encourage subsequent efforts to develop an effective adaptive management plan that incorporates not only scientific monitoring, but human dimensions (Henri et al. 2010) and TEK (Servheen 1998, Enck et al. 2006).

2. REVIEW OF BIOLOGY AND NATURAL HISTORY

2.1 Reproduction and vital rates

The polar bear is a circumpolar, ice-dependent apex predator. Polar bears are K-selected species with late sexual maturity, small litter size, high maternal investment and high adult survival. Polar bears' reproductive rate is among the lowest in all mammals (Bunnel and Tait 1981).

Reproduction in polar bears is similar to that in other ursids. Females generally mature at 4-5 years, and enter a prolonged estrus between late March and early June, although most mating occurs in April and early May. Ovulation is induced by mating (Derocher et al. 2010, Stirling 2009), and implantation is delayed until autumn. The total gestation period is 195-265 days (Uspenski 1977, Amstrup 2003). Whether or not the embryo implants and proceeds to develop is likely determined by the body condition of the female bear. Pregnant females enter their dens in snow drifts or slopes on land, close to the sea, or on the sea ice (in the Chukchi and Beaufort seas) in late autumn, as early as October (Lentfer and Hensel 1980). Females give birth inside the den, usually in early January in the Beaufort Sea (Amstrup 2003) or in December in western Hudson Bay (Derocher et al. 1992). Newborn polar bears are blind, sparsely haired and weigh approximately 0.6 kg (Blix and Lentfer 1979). Cubs grow rapidly, fed on rich milk from their mother (36% fat; Derocher et al. 1993), and when they emerge from the den sometime between early March and late April (Pedersen 1945, Wiig 1998), they weigh 10-12 kg (Amstrup 2003). In some regions, after emerging from the den, the female may not have fed for a period up to 8 months, which may be the longest period of food deprivation for any mammal (Watts and Hansen 1987). Although age at sexual maturity may vary between areas, depending on food availability and whether influenced by density dependence, females normally mature sexually at an age of 5, giving birth to their first cubs at age 6 (Stirling et al. 1976, Lentfer and Hensel 1980).

Polar bears most often give birth to twin cubs, singleton and triplet litters are less frequent. As is the case with most mammals, cub mortality is high in the first year (Larsen 1985; Amstrup and Durner 1995). Litter size increases with maternal age, until she is about 14 (Derocher and Stirling 1994). The pups usually follow the mother for two years (range 1.3-2.3 years; Lønø 1970; Stirling et al. 1976; Amstrup and Durner 1995), and consequently females on average don't enter a new reproductive cycle more often than every third year most places (Amstrup 2003). In contrast to their low reproductive rates, polar bears have high survival rates. Survival is the population parameter that is least sensitive to density dependence as polar bear populations increase towards carrying capacity (Eberhardt 1977).

Ian Stirling

2.2 Movement/migrations

Data from satellite telemetry of tagged polar bears have shown that polar bears do not wander aimlessly, but rather their movement and distribution are determined by the way they use the sea ice habitat as a platform between feeding, mating, denning, and in some populations, summer retreat areas. They tend to move on drifting ice to remain in productive habitats (e.g., over the productive continental shelf; Durner et al. 2009), which often implies moving against the direction of drift of the sea ice to remain in the same general location. In the Barents Sea, it has been shown that polar bears continuously walked northwards nine months of the year, though they remained largely in the same area (Mauritzen et al. 2003b).

Movement patterns of populations and individual bears within populations vary, but individual bears exhibit a high degree of seasonal fidelity to different areas within their home ranges. For example, many bears captured and tagged in Storfjorden in Svalbard are repeatedly recaptured in the same area over many years, while bears captured a few miles east, in the Barents Sea, never enter Storfjorden and have much larger home ranges (Mauritzen et al. 2002). Generally, polar bears inhabiting offshore drift ice have much larger home ranges than bears inhabiting consolidated ice (Born et al. 1997, Ferguson et al. 2001).

Some bear populations are more closely associated with pack ice than others, with individuals spending the majority of the year on the ice and with only a small proportion of individuals spending any amount of time on land (Schliebe et al. 2006). In other populations, such as those in the seasonal ice habitats of the eastern Canadian Arctic, the vast majority of individuals in the population come to land for an extended period in the late summer and early fall.

2.3 Diet

Polar bears are highly carnivorous in comparison to other ursids, which are omnivorous. Their primary prey species throughout their range are ringed seals *Phoca hispida*, preferably young of the year, and to a lesser extent bearded seals *Erignathus barbatus*. In some areas they are also known to prey on harp seals *Pagophilus groenlandicus*, hooded seals *Cystophora cristata*, and even larger prey species such as walrus *Odobenus rosmarus* and beluga *Delphinapterus leucas* (Thiemann et al. 2008b). Birds, fish, vegetation and kelp are also eaten to a certain extent (Pedersen 1945, Dyck et al. 2007, Born et al. 2010).

Polar bears digest fat more efficiently than protein (Best 1984). Polar bears are large compared to other ursid species which is a consequence of their energy-rich diet. Because of their large body size, it is unlikely they would be capable of gaining enough nutrition to survive from a purely terrestrial diet of berries and plants, occasional ungulates and the small number of fish that might be accessible to them (Rode et al. 2010).

2.4 Disease, parasites and pathogens

Polar bears are long-lived animals and, like other ursids, are not generally susceptible to disease (Schliebe et al. 2006).

Although polar bears primarily feed on fat, which is relatively free of parasites (Forbes 2000), larvae of the *Trichinella* parasite have been confirmed in polar bears throughout their range, and antibodies to the protozoan parasite *Toxoplasma gondii* has been found in polar bears from Alaska (Kirk et al. In press), Greenland and Svalbard (Oksanen et al. 2009). It is not yet clear how the presence of the *Toxoplasma* parasite might influence the health of polar bears.

Four morbilliviruses (canine distemper, dolphin morbillivirus, phocine distemper and porpoise morbillivirus) have been documented in polar bears from Alaska and Russia (Follmann et al. 1996, Garner et al. 2000). Jores et al. 2008 documented a 44% prevalence of *Clostridium* perfringens in polar bears from Svalbard, but the study concluded that the microbe was a normal inhabitant of the bacterial flora of the polar bear digestive system. One case of rabies in polar bear has been confirmed (Taylor et al. 1991).

The incidence of parasites and pathogens has been predicted to increase with a warming Arctic, which may be consequential for polar bears given a potentially reduced capacity for immune response. Some studies have related presence of disease, such as morbillivirus, with an immune response that suggest the potential for health effects (Kirk et al. 2010). In sum, however, to a large extent it is unknown how parasites and pathogens have or may have an impact on polar bears.

3. POLAR BEAR SUBPOPULATIONS

3.1 Distribution

The southern extent of the range of polar bears is the southern border of the subpopulation in Davis Strait at 47°N. The actual southernmost observation of a polar bear is likely the bear that was shot on the north shore of Lac St. Jean, Québec, in 1938, at 48°40'N (Jackson 1939). But at present time, in the Atlantic Arctic, polar bears can be occasionally found on the sea ice off of Newfoundland during the spring. In the Pacific Arctic in recent history polar bears summered as far south as St. Matthew's Island in the Bering Sea, although now are generally not observed south of Savoonga, Alaska. The farthest south present denning area is Akimiski Island in James Bay in Canada, at about 52°35'N (M.E. Obbard, pers. comm.). The northernmost documented observation of a polar bear is probably at 89°46,5'N, 25 km from the North Pole (van Meurs and Splettstoesser 2003).

3.2 Subpopulations/management units

At one time it was suggested that polar bears make large scale circumpolar movement. This movement was assumed to be both active and passive with the main direction of the ice drift. Hence polar bears were thought to constitute one large meta-population (Pedersen 1945).

In 1965, when the first meeting between polar bear nations and experts was held in Fairbanks, Alaska, USA (Flyger 1967) there was little information on the structure or discreteness of the world's polar bear populations. By the second meeting of the PBSG in 1970, more information was available: two distinct populations were thought to exist in Alaska, one to the north and one to the west, the one to the north possibly "originating" in Canada; a southern Hudson Bay population separate from the "main" Hudson Bay population; a shared population between Norway and Russia around Svalbard and Franz Josef Land; and two populations in Greenland, one in the northeast and one population on the west coast shared with Canada (PBSG 1970). When the first official status table was agreed upon by the PBSG in 1993, a total of 15 populations were acknowledged (PBSG 1993). At present, 19 population units of polar bears, called subpopulations, are recognized throughout the circumpolar Arctic by the PBSG (PBSG 2010a).

Genetic studies have shown that polar bears from the various subpopulations are genetically similar (Paetkau et al. 1999), and there is no evidence that any of the groups have been evolutionary separated for significant periods of time. Consequently, the rate of genetic exchange is such that these "units" cannot be considered as real populations in an evolutionary sense. According to IUCN criteria the polar bear population units have been called "subpopulations". Although demographic exchange can be limited between subpopulations (Mauritzen et al. 2002, PBSG 2010a), both demographic and genetic exchange clearly occurs, and consequently "management units" may be a more correct term. Taylor and Lee (1995) stated that: "For management purposes, populations should be sufficiently large that the effects of immigration and emigration on population dynamics can be ignored relative to rates of birth and death within the enclosed area", which is the "classical" definition of a population. Paetkau et al. (1999) found in an analysis of 16 microsatellite markers from 473 bears from 16 presumed subpopulations that the variation in the genetic material in general supported the existing borders of the subpopulations, except from the border between Kane Basin and Baffin Bay, and the border between Svalbard and Franz Josef Land. Satellite telemetry indicated the existence of two effectively separated management units or subpopulations in Kane Basin and Baffin Bay (Taylor et al. 2001). The border between Svalbard and Franz Josef Land was later removed, and the polar bears in these areas now constitute the Barents Sea subpopulation, which was also supported by satellite telemetry (Mauritzen et al. 2002). The study also found four genetic clusters in the Canadian Arctic (see ch. 3.4).

3.3 Presently delineated subpopulations

The following provides brief descriptions of the status of each of the presently 19 delineated subpopulations, based on the status reviews published in the proceedings from the 15th meeting of the PBSG in Copenhagen in 2009 (PBSG 2010a).

3.3.1 Arctic Basin (AB)

The large area surrounding the North Pole is a geographic catch-all subpopulation to account for the remainder of polar bears not accounted for in the delineated 18 subpopulations. Until 2001 the area consisted of another, but much smaller, geographic catch-all subpopulation, Queen Elizabeth Islands, which was intended to account for the remainder of northern Canada. In 2003, Canadian authorities decided that any bear in this area should be assigned to the unknown Arctic Basin subpopulation. Polar bears occur here at very low densities and it is known that bears from other subpopulations use the area (Durner and Amstrup 1995). As climate warming continues, it is anticipated that the

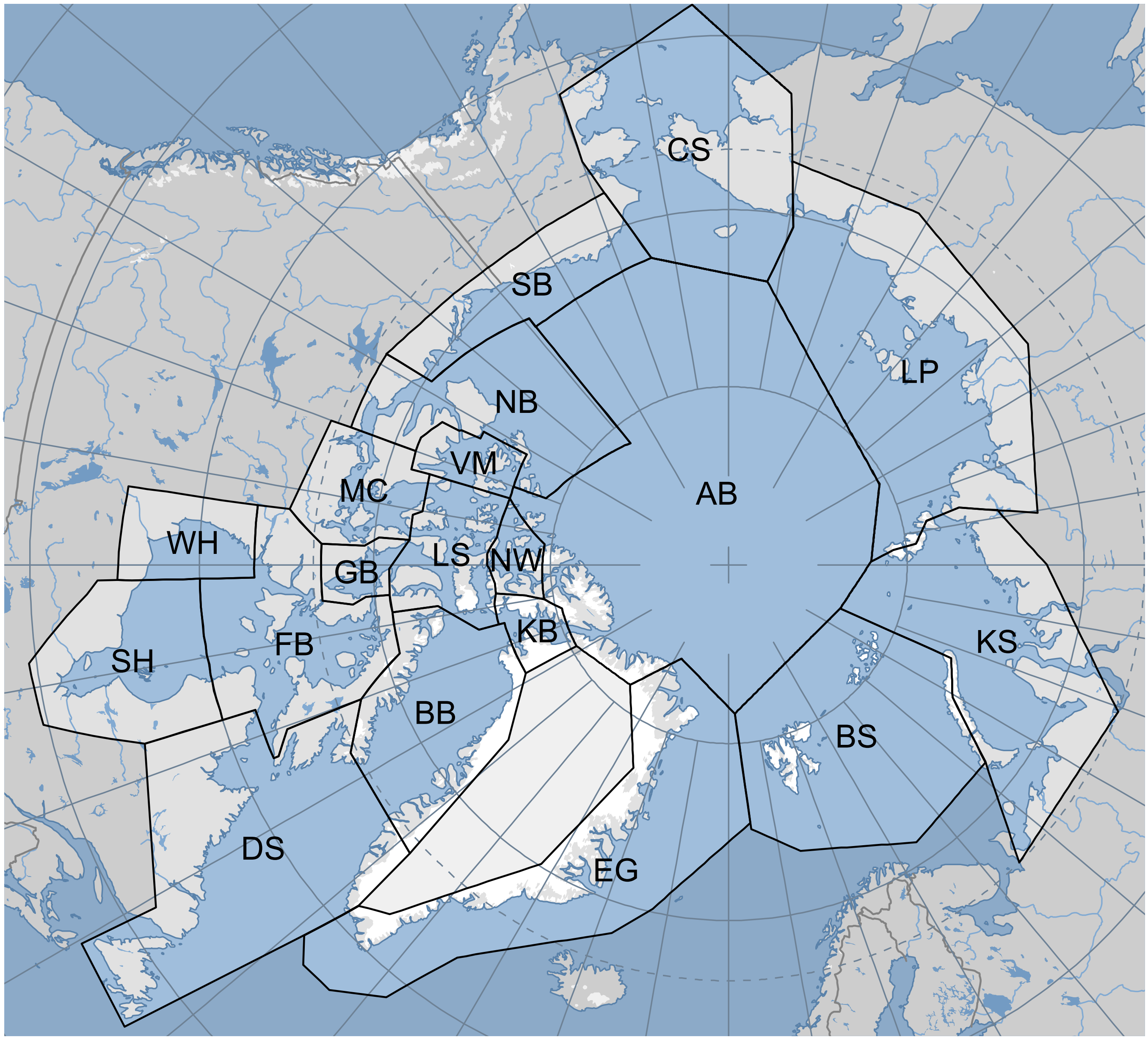

Fig. 1: Circumpolar map of all polar bear subpopulation as recognized by the IUCN/Polar Bear Specialist Group in 2009 (PBSG 2010a). Total area covered = 24 mill. km². Abbreviations are explained below.

Arctic Basin, especially near the coastlines of the northern Canadian arctic islands and Greenland, may become more important for polar bears as a refuge. However, the biologically productive continental shelf is fairly narrow in this area and the majority of the area is over the deepest waters of the Arctic Ocean where productivity is thought to be low. Polar bears with cubs have recently been observed from icebreakers in late summer in the Arctic Basin, 440 miles north of Wrangel Island (Ovsyanikov 2010).

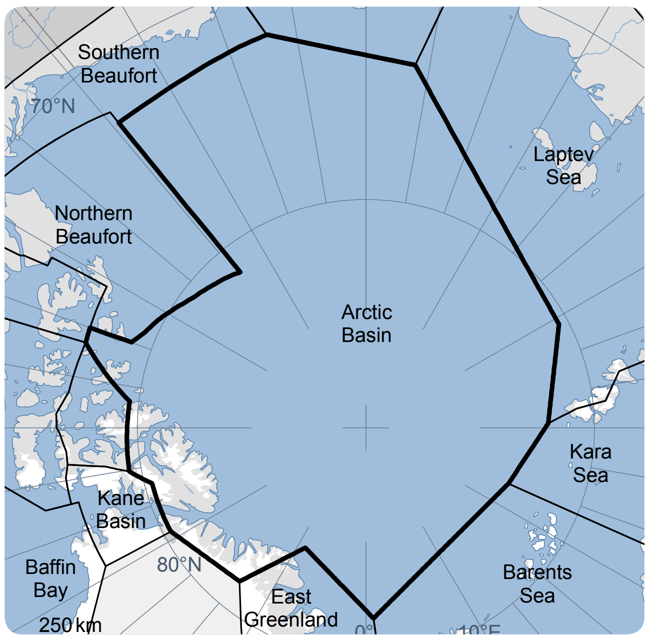

Fig.2 Arctic Basin subpopulation area = 4.23 mill. km²

3.3.2 Baffin Bay (BB)

The Baffin Bay subpopulation is shared between Greenland and Canada.

Based on the movements of adult females with satellite radio-collars and recaptures of tagged animals, the Baffin Bay (BB) subpopulation of polar bears is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island, Nunavut, Canada to the west (Taylor and Lee 1995, Taylor et al.

2001). A distinct southern boundary at Cape Dyer, Baffin Island, is evident from the movements of tagged bears (Stirling et al. 1980) and from polar bears monitored by satellite telemetry (Taylor et al. 2001). A study of microsatellite genetic variation did not reveal any significant differences between polar bears in BB and neighbouring Kane Basin, although there was significant genetic variation between polar bears in BB and those in Davis Strait (Paetkau et al. 1999). An initial subpopulation estimate of 300–600 bears was based on mark-recapture data collected in spring (1984–1989) in which the capture effort was restricted to shore-fast ice and the floe edge off northeast Baffin Island. However, work in the early 1990's showed that an unknown proportion of the subpopulation is typically offshore during the spring and, therefore, unavailable for capture. A second study (1993–1997) was carried out during September and October, when all polar bears were ashore in summer retreat areas on Bylot and Baffin islands (Taylor et al. 2005). Taylor et al. (2005) estimated the number of polar bears in BB at 2,074 ± 226 (SE). The current (2004) abundance estimate is less than 1,600 bears based on simulations using vital rates from the capture study (Taylor et al. 2005) and up-to-date Canadian and Greenland harvest records.

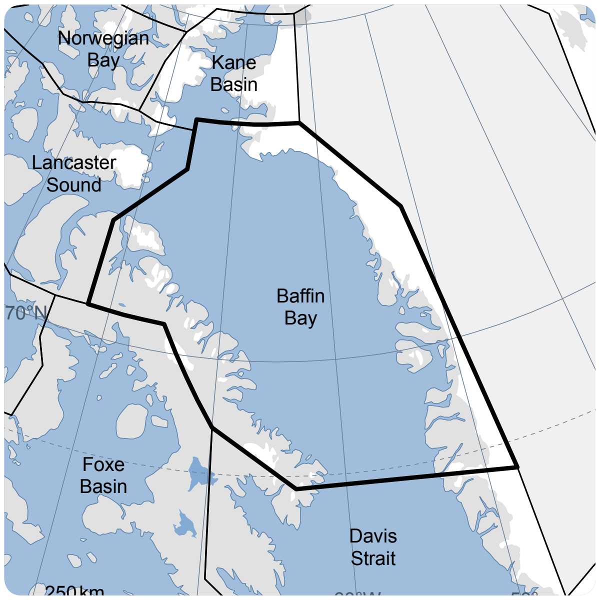

Fig.3 Baffin Bay subpopulation area = 1.08 mill. km²

3.3.3 Barents Sea (BS)

The size of the Barents Sea (BS) subpopulation was estimated, using aerial survey techniques, to be 2,650 (95% CI, approx. 1900–3600) in August 2004 (Aars et al. 2009). Ecological data indicate that the subpopulation grew steadily during the first decade after protection from hunting in 1973, and then either continued to grow or stabilized. Studies on individual movement using satellite telemetry and mark-recapture have been conducted in the Svalbard area since the early 1970s (Larsen 1972, 1985, Wiig 1995, Mauritzen et al. 2001, 2002). Studies show that some polar bears associated with Svalbard are very restricted in their movements, but some bears from BS range widely between Svalbard and Franz Josef Land (Wiig 1995, Mauritzen et al. 2001). Population boundaries based on satellite telemetry indicate that BS is a natural unit, albeit with some overlap to the east with the Kara Sea subpopulation (Mauritzen et al. 2002). Although overlap between BS and the East Greenland subpopulation may be limited (Born et al. 1997), low levels of genetic structure among all these subpopulations indicate substantial gene flow (Paetkau et al. 1999). There is also some preliminary evidence that home ranges of bears from the east Greenland subpopulation overlap with those from Svalbard in Fram Strait (Born et al. 2010).

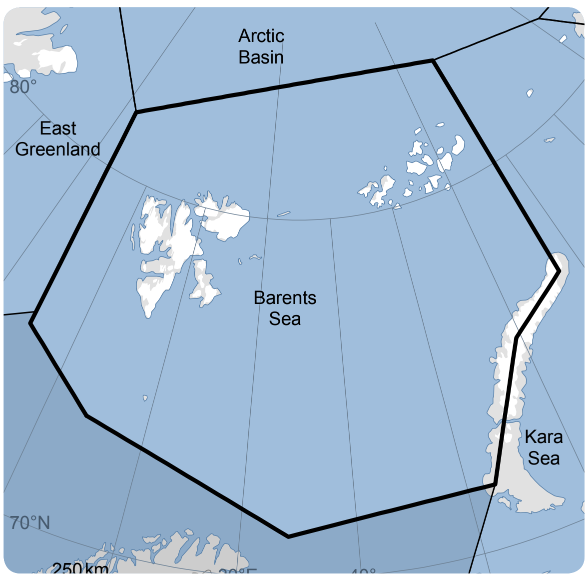

Fig.4 Barents Sea subpopulation area = 1.69 mill. km²

3.3.4 Chukchi Sea (CS)

Reliable estimates of population size or status based upon mark-recapture or other techniques (e.g., aerial survey) are not available for the CS. This subpopulation is believed to be declining based on reported high levels of illegal killing in Russia combined with continued legal harvest in the United States, and observed and projected losses in sea ice habitat (PBSG 2010a).

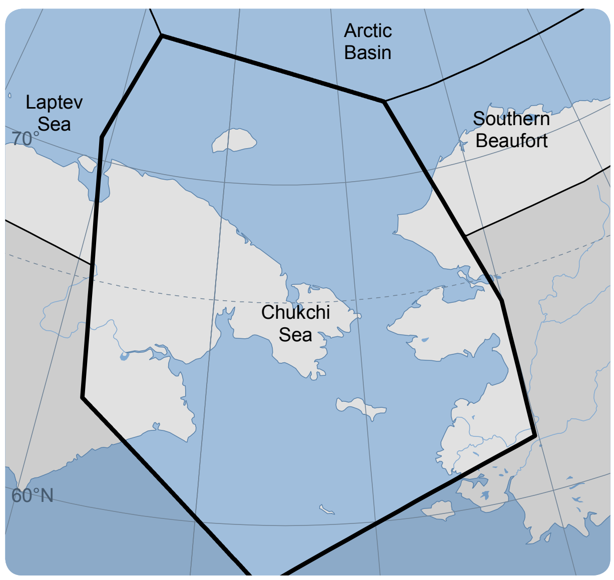

Fig.5 Chukchi Sea subpopulation area = 1.84 mill. km²

Cooperative studies between the US and Russia in the late 1980s and early 1990s revealed that polar bears in the CS, also known as the Alaska-Chukotka subpopulation, are widely distributed on the pack ice of the northern Bering, Chukchi, and eastern portions of the East Siberian seas (Garner et al. 1990, 1994, 1995). Based upon these telemetry studies, the western boundary of the subpopulation was set near Chaunskaya Bay in northeastern Russia. The eastern boundary with the southern Beaufort Sea population was set at Icy Cape, Alaska (Amstrup and DeMaster 1988, Garner et al. 1990, Amstrup et al. 1986, 2004a, 2005). Movement data have been used to determine probabilistic distributions and zones of overlap between CS and Southern Beaufort Sea subpopulations. This information can be used to more accurately assign the geographic distribution of identified harvest levels among communities that overlap the range of these populations (Amstrup et al. 2004a, 2005).

3.3.5 Davis Strait (DS)

The southernmost movements of some individuals within this subpopulation occur as far south as 47°N.

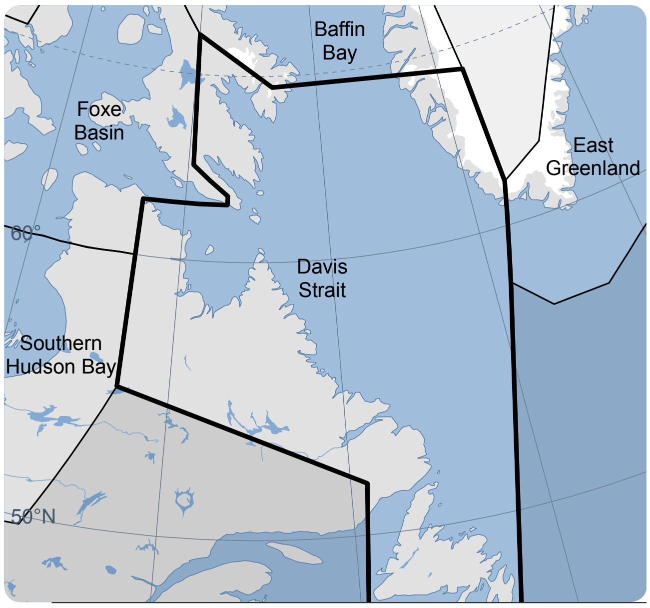

Fig.6 Davis Strait subpopulation area = 2.62 mill. km²

Based on the recapture or harvest of previously tagged animals and of adult females with satellite collars, the Davis Strait (DS) polar bear subpopulation occurs in the Labrador Sea, eastern Hudson Strait, Davis Strait south of Cape Dyer, and along an as yet undetermined portion of south-west Greenland (Stirling et al. 1980, Stirling & Kiliaan 1980, Taylor and Lee 1995, Taylor et al. 2001a). A genetic study of polar bears (Paetkau et al. 1999) indicated significant differences between bears from southern DS and both Baffin Bay and Foxe Basin; Crompton et al. (2008) found that individuals from northern portions of DS and those from bears in Hudson Strait share a high degree of ancestry. The initial subpopulation estimate of 900 bears for DS (Stirling et al. 1980, Stirling and Kiliaan 1980) was based on a correction from the original mark-recapture calculation of 726 bears, which was felt to be too low because of possible bias in the sampling. In 1993, the estimate was again increased to 1,400 bears and to 1,650 in 2005. These increases were to account for the bias resulting from springtime sampling and TEK that suggested that more bears were being seen over the last 20 years (Kotierk 2010). The most recent inventory of this subpopulation was completed in 2007; the new subpopulation estimate is 2,158 (95% log-normal CI, 1811 – 2534). Newly estimated survival rates are comparable to other populations, but recruitment (litter sizes and litter production rates) is the lowest recorded for any polar bear population (Peacock et al. in prep).

3.3.6 East Greenland (EG)

Although various studies have indicated that more or less resident groups of bears may occur within the range of polar bears in East Greenland (Born 1995, Dietz et al. 2000, Sandell et al. 2001), the EG polar bears are thought to constitute a single subpopulation with only limited exchange with other subpopulations (Wiig 1995, Born et al. 2009). Satellite-telemetry has indicated that polar bears range widely along the coast of eastern Greenland and in the pack ice in the Greenland Sea and Fram Strait (Born et al. 1997, 2009, Wiig et al. 2003; PBSG 2010a). Although there is little evidence of genetic difference between subpopulations in the eastern Greenland and Svalbard-Franz Josef Land regions (Paetkau et al. 1999), satellite telemetry and movement of marked animals indicate that the exchange between EG and the Barents Sea subpopulation is minimal (Wiig 1995, Born et al. 1997, 2009b, Wiig et al. 2003). No inventories have been conducted to determine the size of the polar bear subpopulation in eastern Greenland.

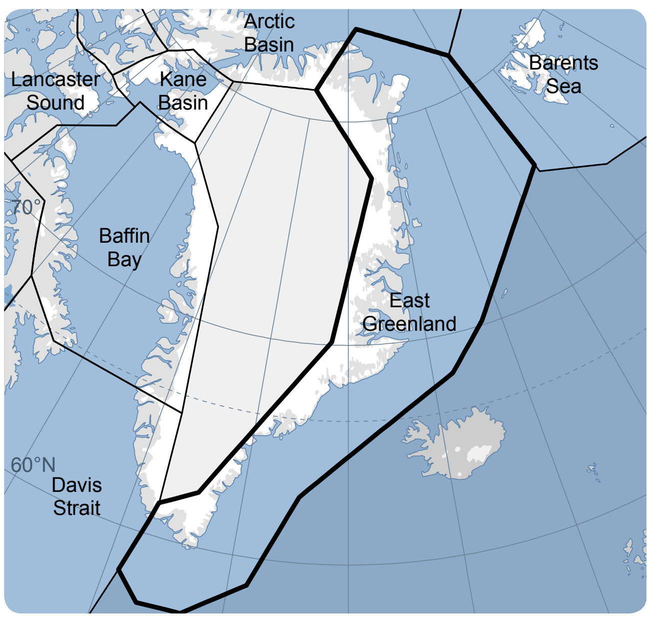

Fig.7 East Greenland subpopulation area = 2.05 mill. km²

3.3.7 Foxe Basin (FB)

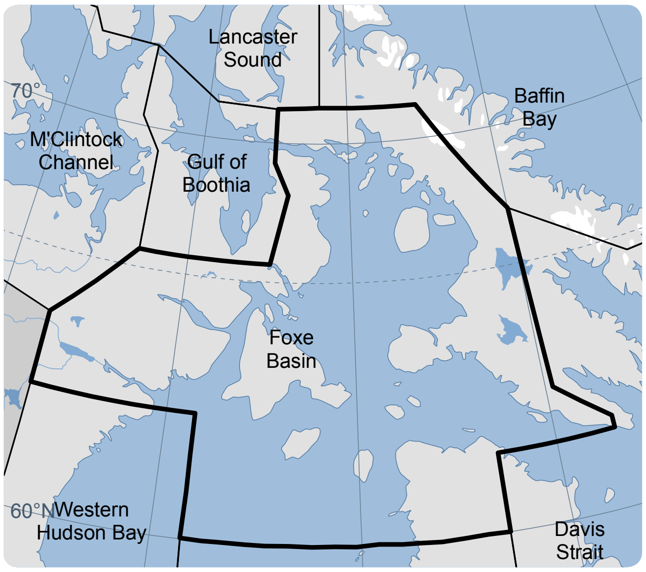

Fig.8 Foxe Basin subpopulation area = 1.18 mill. km^2

Based on 12 years of mark-recapture studies (primarily within Hudson Bay), tracking of female bears in western Hudson Bay and southern Hudson Bay, the Foxe Basin (FB) subpopulation appears to occur in Foxe Basin, northern Hudson Bay, and the western end of Hudson Strait (Taylor and Lee 1995). During the ice-free season, polar bears are concentrated on Southampton Island (the number of bears on the island was estimated at 240 independent bears in August 2008; S. Stapleton, unpublished data, Peacock et al. 2008) and along the Wager Bay coast; however, significant numbers of bears are also encountered on the islands and coastal regions throughout the Foxe Basin area (Peacock et al. 2008, 2009; Obbard et al. 2010). A total subpopulation estimate of 2,119 ± 349 was developed in 1996 (Taylor et al. 2006b) from a mark-recapture analysis based on tetracycline biomarkers. The marking effort was conducted during the ice-free season, and distributed throughout the entire area. TEK suggests the subpopulation of polar bears has increased (Government of Nunavut consultations in villages in Foxe Basin 2004–2009); the subpopulation estimate was increased to 2,300 bears in 2005. Survival and recruitment rates required for a population viability analysis are unavailable, and the rates observed from adjacent populations vary considerably. During a comprehensive summertime aerial survey in 2009 and 2010 covering 43,000 km, 816 and 1003 bears were observed in 2009 and 2010, respectively (S. Stapleton, unpublished data, University of Minnesota and Government of Nunavut; Peacock et al. 2009); abundance is currently being estimated.

3.3.8 Gulf of Boothia (GB)

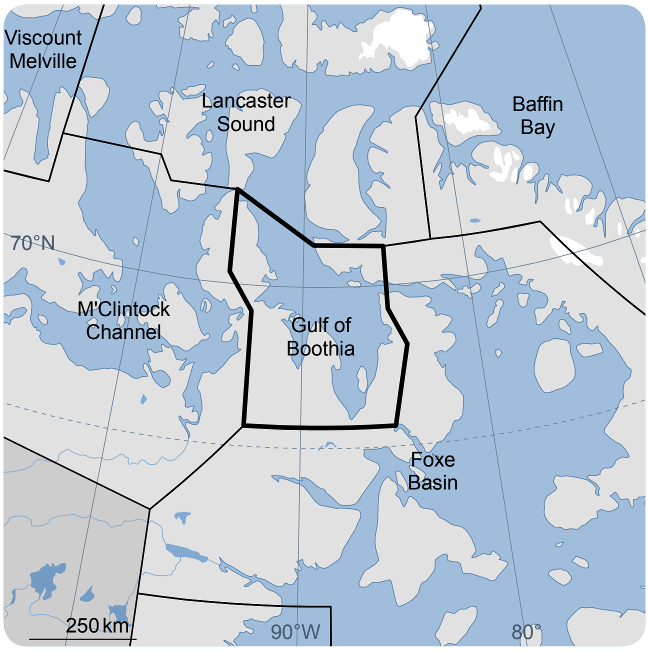

Fig.9 Gulf of Boothia subpopulation area = 170,000 km^2

The population boundaries of the Gulf of Boothia subpopulation are based on genetic studies (Paetkau et al. 1999), movements of tagged bears (Stirling et al. 1978, Taylor and Lee 1995), movements of adult females with satellite radio-collars in GB and adjacent areas (Taylor et al. 2001), and interpretations by local Inuit hunters of how local conditions influence the movements of polar bears in the area. An initial subpopulation estimate of 333 bears was derived from the data collected within the boundaries proposed for GB, as part of a study conducted over a larger area of the central Arctic (Furnell and Schweinsburg 1984). Although population data from this area were limited, local hunters reported that numbers remained constant or increased since the time of the central Arctic polar bear survey. Based on TEK, recognition of sampling deficiencies, and polar bear densities in other areas, an interim subpopulation estimate of 900 was established in the 1990s. Following the completion of a mark-recapture inventory in spring 2000, the subpopulation was estimated to number 1,523 ± 285 bears (Taylor et al. 2009). Natural survival and recruitment rates were estimated at values higher than the previous standardized estimates (Taylor et al. 1987a). Taylor et al. (2009) concluded that the subpopulation was increasing in 2000, as a result of high intrinsic rate of growth and low harvest. However, harvest rates have increased to a 5-year mean of 60 bears per year, from 40 bears per year reported in Taylor et al. (2009).

3.3.9 Kane Basin (KB)

Based on the movements of adult females with satellite collars and recaptures of tagged animals, the boundaries of the Kane Basin (KB) subpopulation include the North Water Polynya (to the south of KB), and Greenland and Ellesmere Island to the west, north, and east (Taylor et al. 2001). Polar bears in KB do not differ genetically from those in Baffin Bay (Paetkau et al. 1999). The size of the subpopulation was estimated to be 164 ± 35 (SE) for 1994 – 1997 (Taylor et al. 2008a). The intrinsic natural rate of growth for KB polar bears is low at 1.009 (SE, 0.010) (Taylor et al. 2008a), likely because of large expanses of multi-year ice and the low population density of seals (Born et al. 2004). Taylor et al. (2008a) suggested that KB might act as a demographic sink because of unsustainable rates of harvest, relatively unproductive habitat, and lack of genetic differentiation with BB.

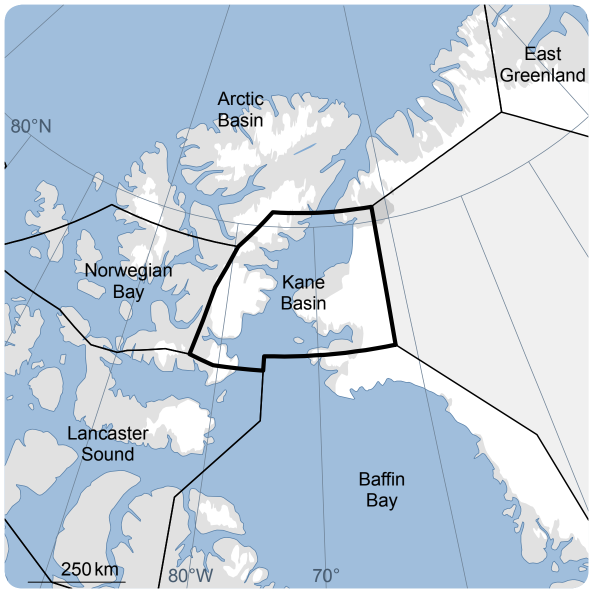

Fig.10 Kane Basin subpopulation area = 155,000 km²

3.3.10 Kara Sea (KS)

This subpopulation includes the Kara Sea and overlaps in the west with the Barents Sea subpopulation in the area of Franz Josef Land and Novaya Zemlya archipelagos. Data for KS and the Barents Sea are mainly based on aerial surveys and den counts (Parovshivkov 1965, Belikov and Matveev 1983, Uspenski 1989, Belikov et al. 1991, Belikov and Gorbunov 1991, Belikov 1993), though data are insufficient to estimate population size or trend. Telemetry studies of movements have been done throughout the area, but data to define the eastern boundary are incomplete (Belikov et al. 1998, Mauritzen et al. 2002).

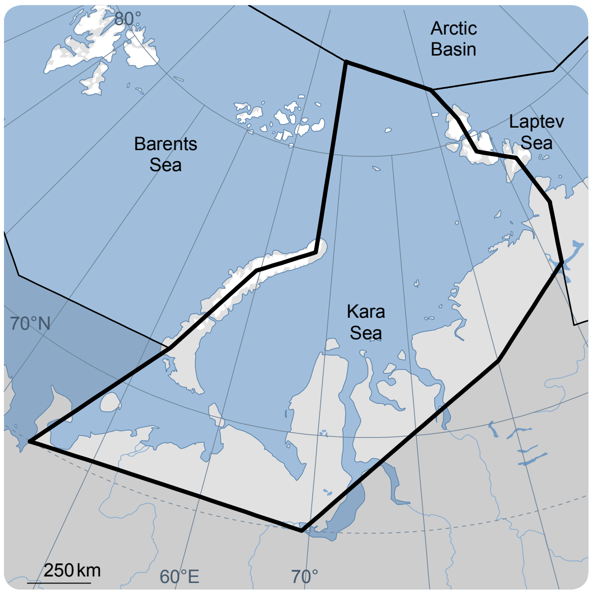

Fig.11 Kara Sea subpopulation area = 1.76 mill. km²

3.3.11 Lancaster Sound (LS)

Information on the movements of adult female polar bears monitored by satellite transmitters, and mark-recapture data, has shown that this subpopulation is distinct from the adjoining Viscount Melville Sound, M'Clintock Channel, Gulf of Boothia, Baffin Bay and Norwegian Bay subpopulations (Taylor et al. 2001). Survival rates of the pooled Norwegian Bay and LS populations were used in the population viability analysis to minimize sampling errors. The subpopulation estimate of 2,541 ± 391 is based on an analysis of both historical and current mark-recapture data collected through 1972 - 1997 (Taylor et al. 2008b). This estimate is considerably larger than a previous estimate of 1,675 that included Norwegian Bay (Stirling et al. 1984; the updated estimate for Norwegian Bay is 203 ± 44, see below, Taylor et al. 2008b). Taylor et al. (2008b) estimated survival and recruitment parameters that suggest this subpopulation has a lower renewal rate than previously estimated.

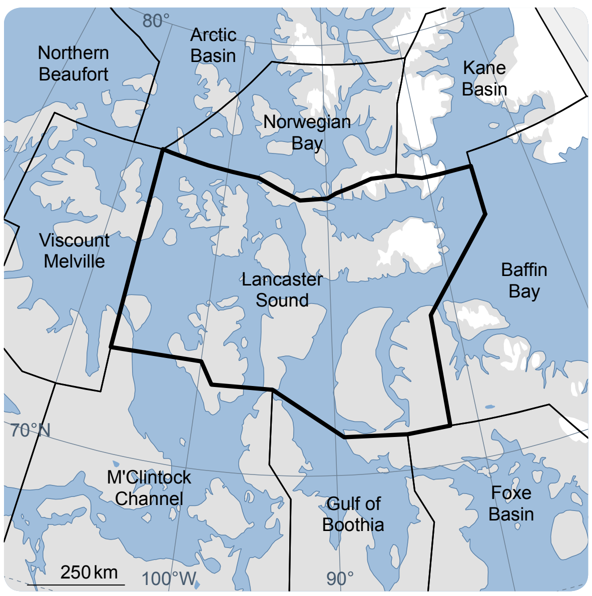

Fig.12 Lancaster Sound subpopulation area = 490,000 km²

3.3.12 Laptev Sea (LP)

The Laptev Sea subpopulation area includes the western half of the East Siberian Sea and most of the Laptev Sea, including the Novosibirsk and possibly Severnaya Zemlya islands (Belikov et al. 1998). The 1993 estimate of subpopulation size for LP (800 – 1,200) is based on aerial counts of dens on the Severnaya Zemlya in 1982 (Belikov and Randla 1987) and on anecdotal data collected in 1960–80s on the number of females coming to dens on Novosibirsk Islands and on the mainland coast (Kischinski 1969, Uspenski 1989).

3.3.13 M'Clintock Channel (MC)

The current population boundaries for the M'Clintock Channel subpopulation are based on recovery of tagged bears and movements of adult females with satellite radio-collars in adjacent areas (Taylor and Lee 1995, Taylor et al. 2001). These boundaries appear to be a consequence of large islands to the east and west, the mainland to the south, and the heavy multi-year ice in Viscount Melville Sound to the north. An estimate of 900 bears was derived from a 6-year study in the mid 1970s within the boundaries proposed for the MC subpopulation, as part of a study conducted over a larger area of the central Arctic (Furnell and Schweinsburg 1984). Following the completion of a mark-recapture inventory in spring 2000, the subpopulation was estimated to number 284 ± 59 (Taylor et al. 2006a). Natural survival and recruitment rates were estimated at values lower than previous standardized estimates (Taylor et al. 1987b).

3.3.14 Northern Beaufort Sea (NB)

Studies of movements and abundance estimates of polar bears in the eastern Beaufort Sea have been conducted using telemetry and mark-recapture at intervals since the early 1970s (Stirling et al. 1975; Demaster et al. 1980; Stirling et al. 1988, 2011; Lunn et al. 1995). From the earlier studies, it was recognized that there were separate populations in the North and South Beaufort Sea and not a single population as was suspected initially (Stirling et al. 1988, Amstrup et al. 1995, Taylor and Lee 1995, Bethke et al. 1996). The density of polar bears using the multi-year ice of the northernmost area was lower than it was further south. The subpopulation estimate of 1,200 (Stirling et al. 1988) for NB was believed to be relatively unbiased at the time but the northwestern coast of Banks Island was not completely surveyed in the 1980s because of perceived conflicts with guided sport hunters in the area. The northern region of the NB subpopulation was surveyed in 1990–92, but the densities encountered were low and the ratio of marked to unmarked polar bears was the same as for the southern portion of the subpopulation. There was no indication that the subpopulation estimate of 1,200 should be increased. A recently completed mark-recapture survey suggested that the size of the NB subpopulation has remained stable at approximately 1,200 bears, probably because ice conditions have remained stable and the harvest has been maintained at low levels (Stirling et al. 2011). The mean sea ice concentration at the September minimum has declined significantly between 1979 and 2010, as has the portion of the sea ice cover that remains over the continental shelf in summer (Stirling et al. 2011). If these trends continue, they are likely to have a negative effect on the population of polar bears in NB. Analyses using data from satellite tracking of female polar bears and spatial modelling techniques indicate the boundary between the NB and SB subpopulations may need to be adjusted, probably expanding the area occupied by bears from NB and retracting that of SB bears (Amstrup et al. 2004a, Amstrup et al. 2005). See summary of the southern Beaufort Sea subpopulation for more details regarding the NB-SB boundary.

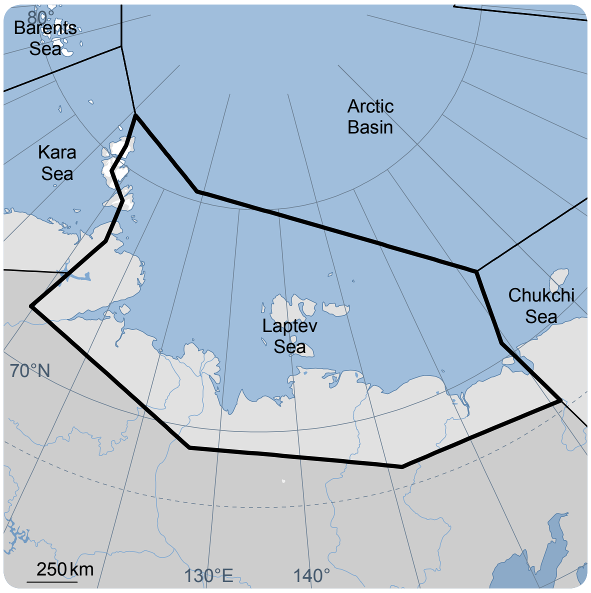

Fig.13: Laptev Sea subpopulation area = 2.46 mill. km^2

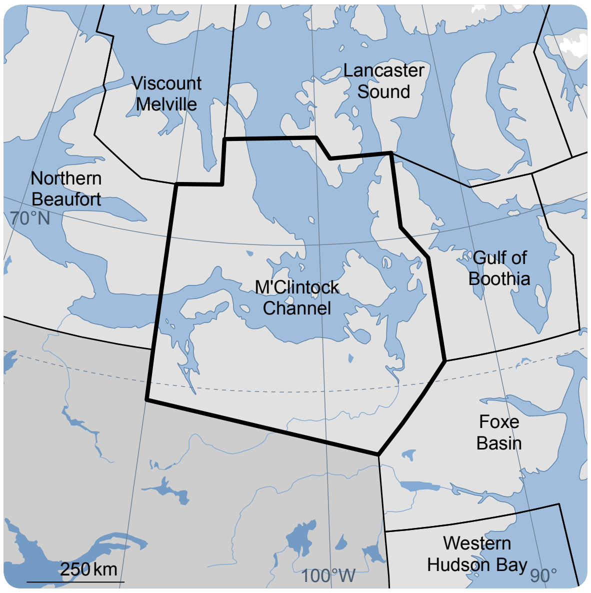

Fig.14: M'Clintock Channel subpopulation area = 500,000 km^2

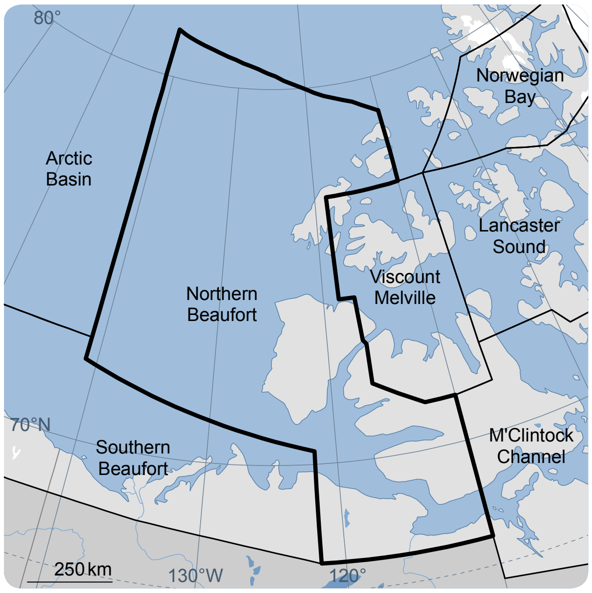

Fig.15: Northern Beaufort Sea subpopulation area = 940,000 km^2

3.3.15 Norwegian Bay (NW)

The Norwegian Bay subpopulation is bounded by heavy multi-year ice to the west, islands to the north, east, and west, and polynyas to the south (Stirling et al. 1993, Stirling 1997; Taylor et al. 2008b). From data collected during mark-recapture studies, and from satellite radio-tracking of adult female polar bears, it appears that most of the polar bears in this subpopulation are concentrated along the coastal tide cracks and ridges along the north, east, and southern boundaries (Taylor et al. 2001). The current (1993–97) estimate is 203 ± 44 (Taylor et al. 2008b). Survival rate estimates for the NW subpopulation were derived from pooled Lancaster Sound and NW data because the subpopulations are adjacent and the number of bears captured in NW alone was too small to generate reliable survival estimates.

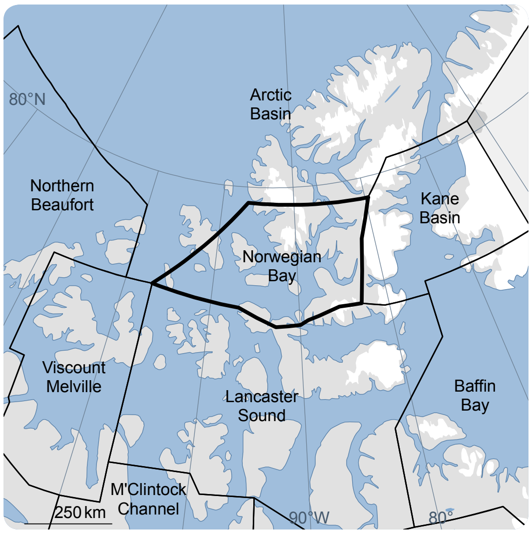

Fig.16 Norwegian Bay subpopulation area = 150,000 km²

3.3.16 Southern Beaufort Sea (SB)

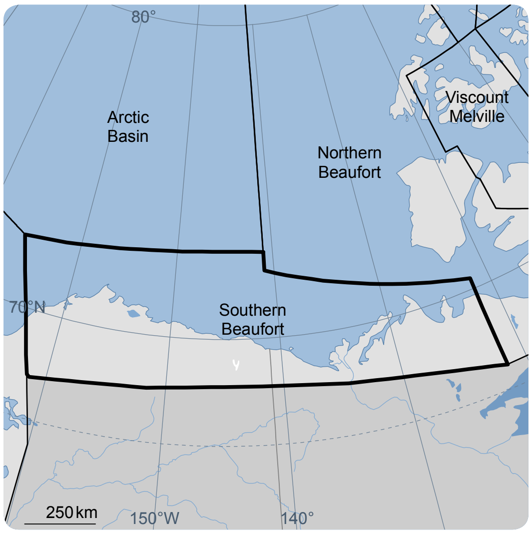

Fig.17 Southern Beaufort Sea subpopulation area = 715,000 km²

Research on polar bears in the SB has been ongoing since 1967 (Amstrup et al. 1986, Stirling 2002). Radio-telemetry and mark-recapture studies through the 1980s indicated that polar bears in the region comprised a single subpopulation, with an eastern boundary between Paulatuk and Baillie Island, Northwest Territories, Canada, and a western boundary near Icy Cape, Alaska (Amstrup et al. 1986; Amstrup and DeMaster 1988; Stirling et al. 1988). Analyses of more recent satellite relocations using probabilistic models indicate that, rather than exhibiting distinct boundaries, there are areas of overlap between the SB and adjacent subpopulations (Amstrup et al. 2004b, 2005). At Barrow, Alaska, USA in the west, 50% of polar bears are from the SB subpopulation and 50% are from the Chukchi Sea (CS) subpopulation. At Tuktoyaktuk, Northwest Territories, Canada, in the east, 50% of polar bears are from the SB subpopulation and 50% are from the northern Beaufort Sea (NB) subpopulation. Based on this analysis, polar bears in the vicinity of the current eastern boundary near Pearce Point, Northwest Territories, are rarely members of the SB subpopulation. To address this issue, user groups, scientists and resource managers are discussing a western shift of the SB-NB boundary. A decision on the potential boundary shift is expected soon, at which time the abundance and status of the SB and NB subpopulations will be re-evaluated. A similar boundary shift, or a change in the way harvest is allocated among subpopulations, may also be required on the western side of the SB subpopulation where it borders the CS subpopulation (Amstrup et al. 2005). Sound management requires that current scientific information be used to define biologically relevant polar bear subpopulations. This presents an increasing challenge, as sea ice loss and increased variability in sea ice extent have the potential to affect polar bear movements and distribution, including the breakdown of historic subpopulation boundaries (Derocher et al. 2004).

The size of the SB subpopulation was first estimated to be approximately 1,800 animals in 1986 (Amstrup et al. 1986). Survival rates of adult females and dependent young were estimated from radio-telemetry data collected from the early 1980s to the mid-1990s (Amstrup and Durner 1995). Through the 1980s and early 1990s, observations suggested that the SB subpopulation was increasing. Amstrup et al. (2001) found that the SB subpopulation may have reached as many as 2,500 polar bears in the late 1990s. However, that estimate was not considered reliable due to methodological difficulties, and management decisions continued to be based on a population size of 1,800. Results from an intensive mark-recapture study conducted from 2001-2006 in both the USA and Canada indicated that the SB subpopulation included 1,526 (95% CI = 1,211 – 1,841) polar bears in 2006 (Regehr et al. 2006). This suggests that the size of the SB subpopulation declined between the late 1990s and 2006, although low precision in the previous estimate of 1,800 precluded a statistical determination. Subsequent analyses of the 2001-2006 data using multi-state and demographic models indicated that the survival and breeding of polar bears during this period were affected by sea ice conditions, and that population growth rate was strongly negative in years with long ice-free seasons, such as 2005 when Arctic sea ice extent reached a record low (Hunter et al. 2010; Regehr et al. 2010). Thus, the SB subpopulation is currently considered to be declining due to sea ice loss. If the region continues to lose polar bear habitat as forecasted by global climate models (Durner et al. 2009), it is likely that the SB subpopulation will face extirpation in the next 100 years (Hunter et al. 2010).

3.3.17 Southern Hudson Bay (SH)

Boundaries of the Southern Hudson Bay polar bear subpopulation are based on observed movements of marked bears and telemetry studies (Jonkel et al. 1976, Kolenosky and Prevett 1983; Kolenosky et al. 1992; Taylor and Lee 1995). The initial estimate of population size came from a three-year (1984–1986) mark-recapture study, conducted mainly along the Ontario coastline (Kolenosky et al. 1992). This study and the more recent telemetry data have documented seasonal fidelity to the Ontario coast during the ice-free season, and some intermixing with the Western Hudson Bay and Foxe Basin subpopulations during winter and spring months. In 1988, a population-modeling workshop suggested an increase in the calculated subpopulation estimate from 900 to 1,000 bears because portions of the eastern and western coastal areas were not included in the area during original sampling. Additionally, the area away from the coast may have been under-sampled due to difficulties in locating polar bears inland (i.e., below tree line). In addition, there are some areas where bears are unsafe to capture. Thus, some classes of bears, especially pregnant females, were believed to be under-sampled. A new analysis of the 1984-1986 capture data produced an estimate for the study area of 634 (390 – 878 95% CI) and for 2003 – 2005, 673 (396 – 950, 95% CI) (Obbard et al. 2007). An additional analysis (Mh Chao implementation of a closed mark-recapture model) of bears in the Akimiski Island area, which is currently included in the geographic designation of the SH, resulted in 70 – 110 additional polar bears. As a result, the abundance estimate for the area currently defined for the SH subpopulation is approximately 900.

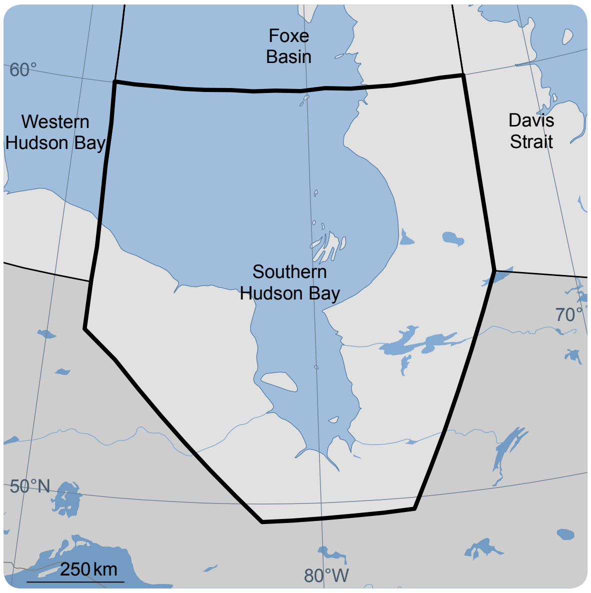

Fig.18 Southern Hudson Bay subpopulation area = 1.14 mill. km²

3.3.18 Viscount Melville Sound (VM)

A five-year study of movements and subpopulation size, using telemetry and mark-recapture, was completed for polar bears inhabiting Viscount Melville (VM) in 1992 (Messier et al. 1992, 1994, Taylor et al. 2002). Population boundaries were based on observed movements of female polar bears with satellite collars and movements of bears tagged in and out of the study area (Bethke et al. 1996, Taylor et al. 2001). The current subpopulation estimate of 215 ± 58 (1996) was based on simulations from parameters measured in 1993 (Taylor et al. 2002).

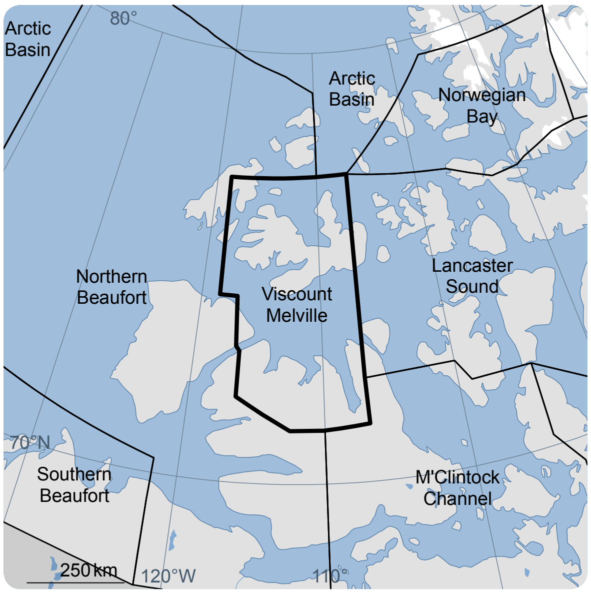

Fig.19 Viscount Melville subpopulation area = 210,000 km²

3.3.19 Western Hudson Bay (WH)

The distribution, abundance, and population boundaries of the Western Hudson Bay (WH) subpopulation have been the subject of research programs since the late 1960s (Stirling et al. 1977, 1999, Derocher et al. 1993, 1997, Derocher and Stirling 1995, Taylor and Lee 1995, Lunn et al. 1997, Regehr et al. 2007). At times, over 80% of the adult population has been marked, and there are extensive records from mark-recapture studies and the return of tags from bears killed by Inuit hunters. This subpopulation appears to be geographically segregated from southern Hudson Bay to the southeast and Foxe Basin to the north during the open-water season, although it mixes with both subpopulations on the Hudson Bay sea ice during the winter and spring (Stirling et al. 1977, Derocher and Stirling 1990, Stirling and Derocher 1993, Taylor and Lee 1995).

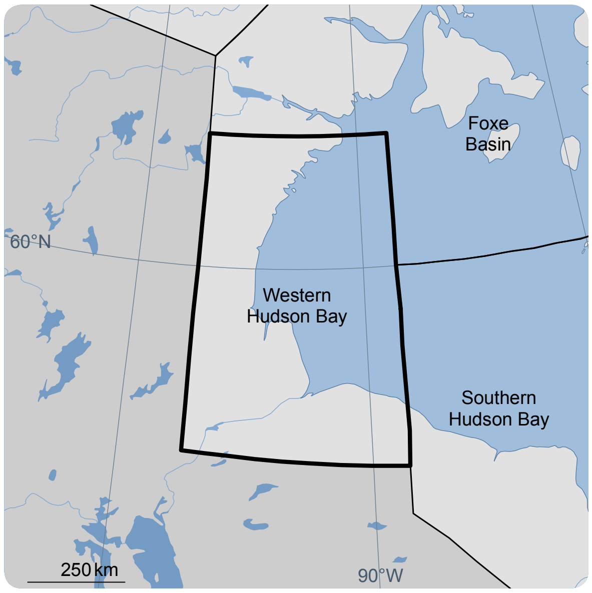

Fig.20 Western Hudson Bay area = 500,000 km²

Between 1987 and 2004, WH declined from 1194 (95% CI = 1020, 1368) in 1987 to 935 (95% CI = 794, 1076) in 2004, a reduction of about 22% (Regehr et al. 2007). In particular, the survival of cubs, sub-adults, and old bears were negatively correlated with the date of breakup. Before 1998 the subpopulation had apparently remained stable (Stirling et al. 1999), indicating that, prior to the onset of a decline brought about by the effects of climate warming on sea ice, the annual harvest of approximately 50 bears had been sustainable.

3.4 Alternative delineations

Animal monitoring programs should be conducted on animal groups that share common biological features in the sense that they would respond in a relatively uniform manner to outside pressures. It has been shown that polar bear subpopulations in a strict biological sense are not real populations, as they exchange genetic material to a degree that prevent any significant genetic differences between them to evolve (Paetkau et al. 1999).

Circumpolar monitoring of polar bears would be most effective by identifying regions occupied by multiple subpopulations that have similar population dynamics, habitat characteristics, and/or ecology. This approach would allow data collected in one part of the identified region to be extrapolated to other portions of the region (e.g. Amstrup et al. 2008; Thiemann et al. 2008b), and for resources available for research to be distributed and focused on representative populations for each identified region rather than across all 19 subpopulations. As there are obvious similarities based on genetics, habitat features, and demography, there is clearly merit in considering delineations other than those based on the currently identified subpopulation units (Peacock et al. In press).

Alternative delineations have been suggested for the Canadian subpopulations (Thiemann et al. 2008a) and regional classifications based on sea ice dynamics have been suggested (Amstrup et al. 2008). Canada is home to 13 of the 19 existing subpopulations either fully or partly under its jurisdiction. Based on genetic clusters in Canada found by Paetkau et al. (1999), and patterns in polar bear biodiversity, Thiemann and coworkers identified five "designatable units" that they felt would be better used for conservation purposes (see Figure 20). Similarly, Amstrup et al. (2008) identified four ecoregions based on similarities in seasonal sea ice dynamics.

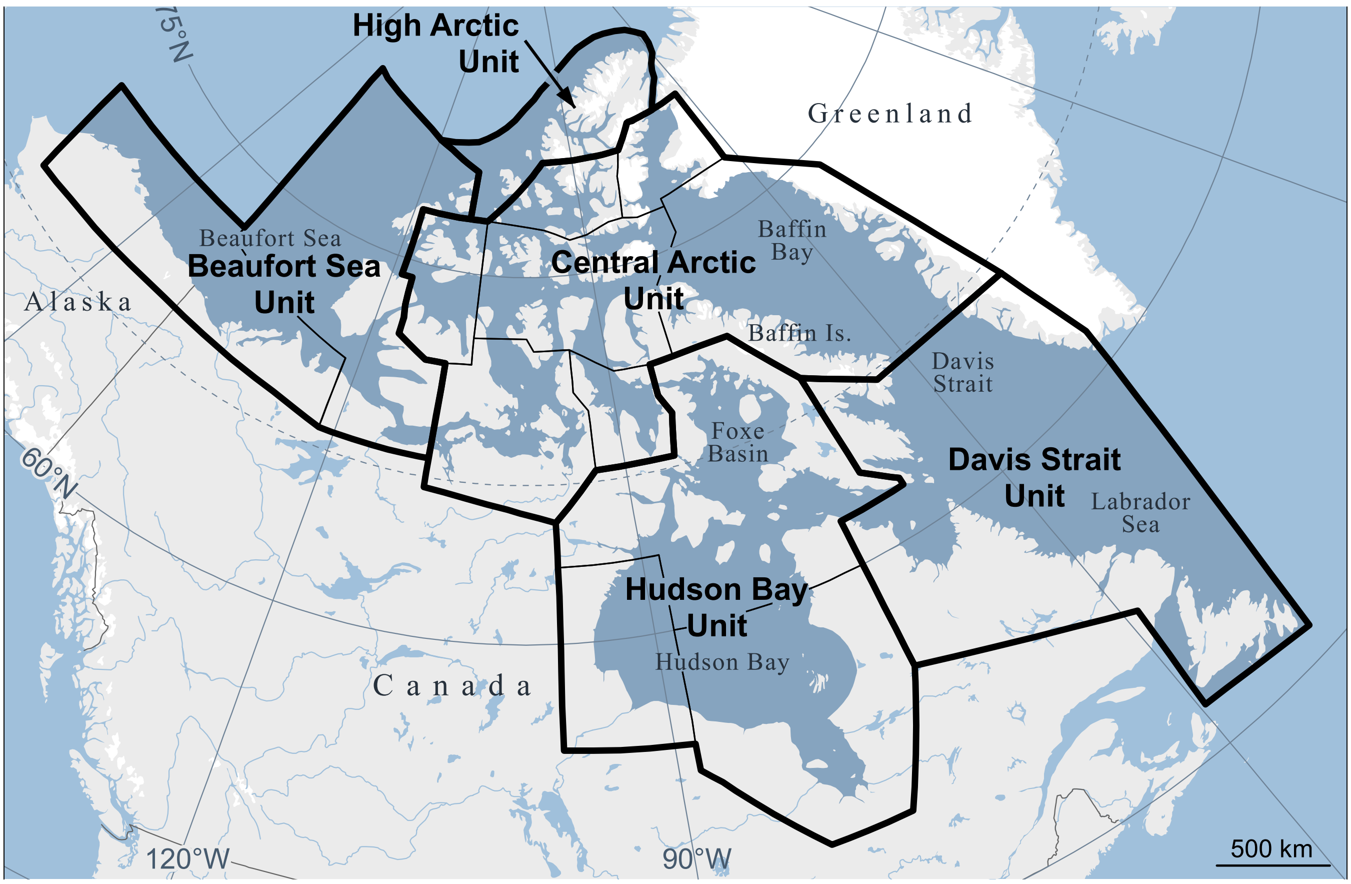

Fig.21: Alternative delineations between Canadian polar bear subpopulations, called designatable units, based on "broad patterns of polar bear biodiversity" (adapted from Thiemann *et al.* 2008a).

4. PRESSURES AND LIKELY IMPACTS ON POLAR BEARS

4.1 Climate warming

Since the third assessment report of the International Panel for Climate Change (IPCC) was made public in 2001, there has been an ever increasing focus on the level and speed of change in the extent of sea ice habitat in Arctic regions (IPCC 2001). This report concluded that spring and summer sea ice extent had decreased by 10 to 15% since the 1950s, and that the extent of snow cover had decreased by 10% since the late 1960s (IPCC 2001). Since then, the amount of modelling to document our climatic future has been immense (IPCC 2007), and there is ample evidence now that polar bears in most subpopulations will experience significant declines in their sea ice habitat over the next 50 to 100 years (ACIA 2005, Amstrup et al. 2008, Serreze and Rigor et al. 2006, Parkinson and Cavalieri 2008). However, new findings show the positive effect of cuts in greenhouse gas emissions on polar bear habitat and indirectly on polar bear numbers (Amstrup et al. 2010).

4.1.1 Observed change

Data provided by satellite passive microwave sensors have shown that the area covered by annual sea ice has decreased by approx. 4% per decade in the period from 1979 to 2006 (Comiso and Nishio 2008). The rate of change in sea ice habitat has outpaced the change as projected by many climate models (Stroeve et al. 2007; see Figure 22). The rate of decrease in sea ice differs markedly among regions (Perovich and Richter-Menge 2009) where polar bears live. During 1979-2006 the rate of decrease of sea ice has been as high as between -9.0 and – 11.3% per decade in the Baffin Bay, Greenland Sea and Barents Sea regions. During the same period the decrease has been -1.2% per decade in the Central Canadian High Arctic (ibid.). Snow cover extent, a critical habitat feature for ringed seals – the primary prey of most polar bear subpopulations – and for polar bear denning, has also diminished (IPCC 2007).

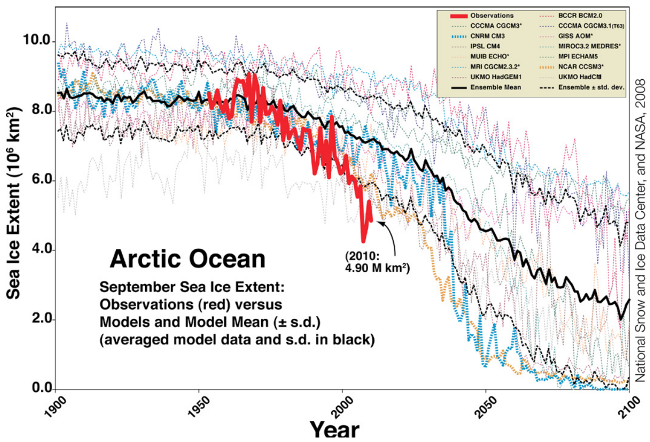

Fig.22 Arctic September sea ice extent from observations (red line) and 13 IPCC AR4 climate models (updated from Stroeve et al. 2007).

Not only have the warm temperatures during the last decades affected the extent of sea ice cover, but they have also had an impact on the timing of sea ice break up, most pronounced and documented in Western Hudson Bay, where polar bears have been forced to stay on land weeks longer than normal, effectively shortening the time polar bears can spend hunting seals on the ice (Stirling et al. 1999; Gough et al. 2004, Stirling and Parkinson 2006).

4.1.2 Projected change

The IPCC has projected an increase in global average surface temperature of 1.1- 6.4°C by 2100, depending on the greenhouse gas emission scenario (IPCC 2007). The panel also predicts that the warming will be greatest over high northern latitudes. Consequently, there will be further decrease in sea ice extent in the Arctic, especially in the summer. Arctic summers will become ice free in the last decades of this century (IPCC 2007), and the average annual decline in sea ice extent between 2000 and 2100 is projected to be in the range of 12-46% (ACIA 2005).

4.1.3 Effects on polar bears

Changes in sea-ice have been shown to alter polar bear survival and mortality and thereby negatively affect productivity and abundance. Furthermore, changes in distribution with bears spending more time on land have also been observed (Stirling et al. 1999; Fischbach et al. 2007; Laidre et al. 2008; Schliebe et al. 2008; Durner et al. 2009; Regehr et al. 2010; Gleason and Rode 2010; Rode et al. 2010). In other subpopulations, however, changes in sea ice have not been correlated with changes in survival, such as in southern Hudson Bay (Obbard et al. 2008) and Davis Strait (Peacock et al. in prep). Thus, the degree and dynamics of sea ice change, population density, and ecological factors may affect the response of subpopulations to changing sea ice conditions. Furthermore, sea ice loss is likely to affect bear body condition and reproduction prior to any observed effects on survival. Effects on reproduction and survival were not observed until more recently after prolonged and unidirectional changes in the ice breakup date (Regehr et al. 2007). Early indicators of effects of habitat loss will be an important aspect of monitoring programs, particularly for populations that are harvested. By the time changes in survival and population size are documented, recovering polar bear populations via management actions may be difficult if not impossible.

4.2 Human-caused mortality

A decade ago, human-caused mortality was perceived as being the most serious threat to polar bears (Taylor et al. 1987), and although climate warming will ultimately be the larger threat, regulated harvest, illegal harvest, and defence kills of polar bears still constitute a significant source of mortality in polar bears throughout the circumpolar Arctic (PBSG 2010a). Polar bears are harvested legally in Canada, Greenland and the United States, under provisions set by the International Agreement on the Conservation of Polar Bears (IUCN 1973). The numbers taken are regulated by quota in most regions; however, there are no legal limits to the number that may be taken in some jurisdictions (PBSG 2010a). In most jurisdictions harvest is based on quotas set by application of rigorous scientific procedures. However, in a few jurisdictions this is not the case, primarily due to lack of data on population status. Legal harvest activities are closely monitored (number, sex, and age of kill) in some areas to ensure that the subpopulations are harvested sustainably. In Nunavut and the Northwest Territories (Canada) the harvest of polar bears is managed to try to ensure a 2:1 skew towards harvesting males versus females. Hunters are encouraged to take males when possible because this has been suggested to be a way to increase the sustainable size of the total harvest without compromising the reproductive potential of the subpopulation. There is thought to be significant illegal harvest of polar bears in parts of eastern Russia (Chukotka) but, the extent of this activity is not well documented (Kochnev 2004).

Annual legal harvest of polar bears is between 700 and 800 or 3-4% of the estimated size of the total population of about 20-25,000 animals. Until recently, the harvest level has been thought to be sustainable in most subpopulations (PBSG 2010a).

Most polar bears are harvested by indigenous people for nutritional, cultural, and/or economic subsistence. The financial return from the sale of legally taken polar bear hides also provides some income for local people in Canada and Greenland. Sport hunting of polar bears, guided by local Inuit hunters, only occurs in Canada and these hunts form a part of the quota assigned to a community. In other words, bears taken by sport hunters are not additive to the quota. Sport hunting can be a source of income for remote settlements and the financial return from the hunt greatly exceeds that of the hide value (Foote and Wenzel 2009). This is important to the local economy where there is often a limited cash economy. Polar bears taken in hunts are used as food in some communities, especially in the eastern Arctic of Canada and Greenland. Hides and skulls are either sold commercially, converted to handicrafts, or used privately. All international trade in polar bear parts is surveyed and regulated by CITES[3].

In the Norwegian and western Russian Arctic, polar bears are protected from all forms of harvest except problem or

3. Convention on International Trade of Endangered Species.

Tor Ivan Karlsen, Norwegian Polar Institute defence kills. However, the United States and Russia signed an agreement in 2000 to jointly manage the Chukchi-Bering Seas (Alaska-Chukotka) polar bear population that recognizes the right of Native Chukotkans to harvest polar bears for subsistence purposes. Implementation of this agreement has resulted in the recent decision by a four member federal-native commission to share a quota of 58 bears between Alaska and Chukotkan Natives. This decision, made at a meeting in June 2010, was accompanied by the requirement that each country have an appropriate system in place to regulate and enforce the harvest limit. Both countries are currently working towards this objective and until then harvest remains illegal in Russia and a quota has not been implemented in the United States.

Defence and problem kills occur throughout the polar bears' range and are inevitable when polar bears and people occur together; these kills can be reduced with proper precautions and training (e.g., proper disposal of garbage in field camps). Poorly planned camps and improper garbage disposal can result in needless killing of bears.

Mortality from set-guns (a self-killing trap with bait attached to the trigger of a gun) and hunting from ships and aircraft have ceased as a result of the International Agreement (IUCN 1973).

Poaching, or illegal hunting of polar bears, is not thought to be a major concern generally although the extent of illegal hunting in eastern Russia has been reported to be as high as 100-200 bears per year (Kochnev 2004). At present, the PBSG is assessing the status of this problem in all jurisdictions.

Over-harvest is an ongoing concern for many polar bear populations (e.g. Baffin Bay and Kane Basin; PBSG 2010a, also in areas where there is no information on the population size (e.g. East Greenland, Sandell et al. 2001; Chukchi Sea, Obbard et al. 2010)). In some areas, the monitoring of polar bear harvest has been inconsistent (e.g., in Greenland, Québec and to some extent in the United States; Brower et al. 2002; PBSG 2010a). In some areas population inventory programs occur relatively infrequently such that if the harvest rate is above the sustainable level, the population may be severely reduced before the next inventory is made and a decline in population size may be detected. Recent development of co-management agreements and greater involvement of the aboriginal public is rendering management of polar bears more acceptable at the local level; in time this cooperation may improve efficacy of management and monitoring. Compared to the situation in the 1960s and 1970s, polar bear harvest management is vastly improved. Several populations have experienced demographic recovery due to harvest regulations (Amstrup et al. 1986; Stirling 2002; Derocher 2005).

Understanding the risk associated with a range of harvest management options is important for polar bear conservation. As threats such as pollution, climate change, tourism, and oil development are better understood, there will be changes in how polar bear harvest is managed. Clearly, if reproduction or survival rates are negatively affected by climate change or pollution, management of polar bear subpopulations will have to be altered accordingly.

4.3 Pollution

As an apex predator largely feeding on seal fat, polar bears have been shown to accumulate high levels of a range of fat-soluble environmental pollutants (McKinney et al. 2009, Verreault et al. 2008, Muir et al. 2006, Verreault et al. 2006, Smithwick et al. 2005, Sonne et al. 2005, Verreault et al. 2005a, Verreault et al. 2005b, Derocher et al. 2003, Lie et al. 2003, Muir et al. 1999, Norstrom et al. 1998).

Polar bears are exposed to high levels of pollutants because these pollutants are magnified in the food web. Many of the organochlorine pollutants are lipophilic, that is, they are deposited in the fat of the animals that consume them. Because animals in the Arctic marine ecosystem are highly dependent on fat for storing energy, growth, insulation and buoyancy, these pollutants are rapidly accumulated progressively up the food chain in a process known as biomagnification. As an apex predator, polar bears are particularly vulnerable to accumulation of organochlorines. Ringed, bearded, and harp seals comprise the primary food of polar bears, and the fat layer is preferentially eaten by the bears and subsequently, the intake of pollutants is high. Most pollution in the Arctic is transported northward by the large rivers, and on wind and ocean currents that bring pollutants from southern latitudes. The pollutants of most concern are organochlorines that are, or were, used in industry or as pesticides. A key characteristic of the pollutants is that they are persistent in the environment and resist degradation. Some pollutants such as polychlorinated biphenyls (PCBs) were used widely in industrial applications precisely because they were extremely stable. Other pollutants such as dieldrin, DDT, toxaphene, and chlordanes were used as pesticides, but they are also stable enough to be transported long distances to the Arctic. Many of the pollutants are now banned from use in most countries but they are so persistent that they will likely remain in the environment and the food chain for decades to come.

Previous studies of levels of PCBs in polar bears around the Arctic concluded that the most polluted polar bears lived in Northeast Greenland, the Barents Sea, and the Kara Sea (Norstrom et al. 1988; Verreault et al. 2005b) because of global transport and deposition patterns. Extrapolating the results of studies in other species, the pollutant load of polar bears in some areas could negatively affect the immune system, hormone regulation, growth, reproduction, and survival rates (Letcher et al. 2010). Recent studies have suggested that there is a relationship between the polar bear immune system response and higher levels of PCBs (Lie et al. 2004; Lie et al. 2005). A weakened immune system may result in these polar bears being more susceptible to disease. Additionally, there is evidence that hormone systems of polar bears are affected by pollution and this may interfere with reproduction and growth (Sonne et al. 2006). Species with delayed implantation may also be more vulnerable to the effects of pollution through endocrine disruption. Further, because female polar bears fast during gestation, pollution loads are mobilized from fat into metabolites and lactation. Because cubs are nursed on milk rich in fat, they can be exposed to very high pollution loads from their mother. Data are lacking to determine the effects of these pollutants on cubs, but there are suggestions that cubs of more polluted females have higher mortality rates (Norstrom et al. 1999).

Given that a polar bear likely contains several hundred chemicals that originated from humans, it is very difficult to determine the relative levels of impact caused by individual substances. Polar bears are efficient at metabolizing some pollutants, but the problem is that many of these metabolites are active in the body before they are excreted, and often metabolites are more toxic (Letcher et al. 2010).

New pollutants are also being found in polar bears. Recently, brominated flame retardants and perfluorinated alkyl substances have been detected (Verreault et al. 2005a; Dietz et al. 2008). It is also possible that many other compounds will be identified. Some pollutants like PCBs, now banned in most countries, are beginning to show signs of decrease in the Arctic and in polar bears (see Norwegian monitoring program for the Barents Sea at http://mosj.npolar.no/).

Although individual and/or population effects of pollutants on polar bears have been suggested (Derocher et al. 2003), there is no conclusive documentation of effects on polar bear body condition, reproductive rates, or survival. It is possible that pollutants may instead act as a cumulative factor, along with other stressors, that could compromise the health of polar bear populations.

The overall impact of chemical pollutants on polar bear health and fitness is unknown. However, there could be long term effects on survival, immune system response, and reproductive hormone action (Derocher et al. 2003; Sonne et al. 2006, 2010; Letcher et al. 2010). When combined with the stress from climate change impacts, the influence of pollutants can certainly be magnified. This is an area that needs further monitoring and research.

4.4 Oil, gas and mineral developments

Oil development in the Arctic poses a variety of threats to polar bears ranging from oil spills to increased human-bear interactions (Stirling 1988, 1990; Amstrup et al. 2006). As oil development increases in polar bear habitat, there is an increased risk that oil spills will occur. It is probable that an oil spill in sea ice habitat would result in oil being concentrated in leads and between ice floes resulting in both polar bears and seals being directly exposed to oil. Studies have shown that polar bears exposed to oil will absorb large quantities of oil in their fur (Øritsland et al. 1981; Hurst and Øritsland 1982). Following oil exposure, polar bears groom themselves and can digest sufficient oil to result in kidney failure, digestive system disorder, and brain damage that ultimately result in death. Other effects include loss of insulation from fur, hair loss, and skin and eye irritations (Øritsland et al. 1981).

Another concern for oil development is the potential for exploration methods such as seismic surveys and noise associated with increased vehicle use and other human activities to displace or disturb polar bears at denning sites. Disturbance could occur both when a pregnant female is selecting a den site and during the winter-spring after the cubs are born and the family emerges from their den. If exploration or development occurs sufficiently close to a den, the mother could abandon the den prematurely or abandon her offspring. In general, previous research suggests that females at den sites tolerate human activity within relatively close proximity to den sites (Amstrup 1993) and that polar bear dens are relatively well insulated from noise, including seismic activity (Blix and Lentfer 1992). It is thought that with careful planning and control of exploration activities, potential disturbance impacts can be reduced or avoided. The potential for development to preclude polar bears from using available denning habitat is less understood.

The effects of increased ship traffic, pollution from drilling compounds and noise on polar bears and their prey are unknown. Ice-breaking vessels and industrial noise have been shown to increase abandonment of subnivean seal structures on sea ice, and consequently may have negative impacts on ringed seal breeding (Kelly et al. 1988). Also, the cumulative impacts of extensive development and ship traffic on polar bears and their prey could pose an exacerbated threat.

Another area of concern is increased human-bear interactions. Polar bears are often attracted by the smells and sounds associated with human developments (Stirling 1990). This attraction to developments can increase the number of bears killed in an area. In some areas, special polar bear monitors are hired to deter bears from oil rigs and other developments. In areas where polar bears are responding to changing sea ice conditions by spending more time on land, there is the potential for increased human-bear interactions in villages and areas of development. Though the cause of this increase is unknown, these statistics support the importance of monitoring human-bear interactions and defense of life kills to allow mitigation of human-bear interactions (e.g., by providing support for village polar bear patrols) and thereby reduce bear mortalities.

4.5 Tourism/disturbance

As the Arctic becomes more accessible, both due to technical developments and less sea ice, human activities within polar bear habitat are likely to increase for a variety of reasons (PBSG 2010a). Tourism is increasing rapidly, and resource exploration introduces humans and infrastructure in the remote areas of the Arctic. This increase in human activity and the number of people in areas inhabited by polar bears increases the probability for disturbance and human-bear encounters.

Polar bears often investigate novel items, like snowmobiles, cabins, tents and humans. Inexperienced people perceive a curious bear as a threat and may shoot it. Poorly positioned or maintained camps also contribute to lethal encounters with polar bears. In many areas, polar bears have been killed at cabins or remote stations when they arrive to investigate food smells. Habituation of polar bears to humans through food conditioning results in polar bears that are more likely to seek out humans and as a result, be killed. In Churchill, Manitoba, several polar bears have died from eating items as diverse as sardine cans and lead acid batteries (Lunn and Stirling 1985). Improved management of bears near the garbage dump in Churchill has greatly reduced this problem. In Alaska, a polar bear died when it ingested anti-freeze mixed with dye to mark an aircraft runway on the ice (Amstrup et al. 1989). In another case, a polar bear drank several litres of hydraulic fluid with unknown consequences (Derocher and Stirling 1991), and there are additional reports of polar bears ingesting plastic, styrofoam, lead car batteries, tin cans and oil (Derocher and Stirling 1991, and references therein). Proper management of food and garbage is essential to reduce human-bear conflicts.

Polar bears appear particularly disturbed by snowmobiles and boats and often show avoidance behaviour of these motorized activities long before humans detect the bear (Andersen and Aars 2008). The impact of this type of harassment is unknown, though chronic harassment may result in polar bears abandoning preferred habitats or becoming habituated to human presence.

As human activities increase in the Arctic, polar bears will experience increased disturbance. Exactly how they will be affected is not known, and there are reasons to believe that some impacts can be controlled with good management. However, combined effects of several negative factors acting simultaneously (e.g., climatic stress, pollution and disturbance) can be difficult to predict and constitute a problem that needs increased attention from both scientists and managers.

The cumulative impact of chronic human disturbance, whether from industry or tourism, from infrastructure, direct poisoning or noise, is unknown, but could potentially be long-term and negative resulting in polar bears abandoning preferred habitats.

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5. EXISTING MONITORING

Both monitoring and research have been variable across the 19 individual polar bear subpopulations of the circumpolar Arctic. However, three subpopulations have been monitored fairly consistently for 30-40 years: the Northern and Southern Beaufort Sea of Canada and the USA, and the Western Hudson Bay population in Canada (Stirling et al. 1999, 2011; Amstrup et al. 2001; Stirling 2002). This continuous research has been important for the ability to detect changes in individuals' body condition and population productivity, and those changes that are correlated with changes in sea-ice habitat (Regehr et al. 2007; Durner et al. 2009; Hunter et al. 2010; Rode et al. 2010, Stirling et al. 2011) and harvest (Amstrup et al. 1986; Stirling 2002). Other populations have been studied less consistently, but some data on subpopulation size are available for temporal comparison: Southern Hudson Bay (Jonkel et al. 1976; Kolenosky et al. 1992; Obbard et al. 2006; Obbard 2008); Davis Strait (Stirling et al. 1980; Stirling and Kiliaan 1980; Peacock et al. in prep); and the Barents Sea (Larsen 1986; Derocher 2005; Aars et al. 2009). Remaining populations have had varying degrees of intensive study, but ultimately this work has not been consistent enough to provide information on population trends: Baffin Bay (Schweinsburg et al. 1982; Taylor et al. 2005; Born et al. 2011); Norwegian Bay (Taylor et al. 2008b), Lancaster Sound (Schweinsburg et al. 1982; Taylor et al. 2008b); M'Clintock Channel (Furnell and Schweinsburg 1984; Taylor et al. 2006a); Foxe Basin (Stenhouse and Lunn 1987; Taylor et al. 2006b; Peacock et al. 2009); Gulf of Boothia (Furnell and Schweinsburg 1984; Taylor et al. 2009); Viscount Melville (Taylor et al. 2002); and Kane Basin (Taylor et al. 2008a) and the Chukchi Sea (Garner et al. 1995; DeBruyn et al. 2010). Few, if any, population data are available for the Arctic Basin and the Kara, East Greenland and Laptev seas. For a review of the status of subpopulations and their levels of monitoring see Table 1.

Data and samples from most populations have been included in large-scale or circumpolar studies on genetics (Paetkau et al. 1999), on contaminants (Norstrom et al. 1998; Andersen et al. 2001; Smithwick et al. 2005; Verreault et al. 2005), on population delineation (e.g. Taylor et al. 2001; Mauritzen et al. 2002; Amstrup et al. 2005) and, for the divergent and convergent sea-ice eco-regions (Amstrup et al. 2008), on habitat selection (Durner et al. 2009).

Table 1: Status of polar bear subpopulations and level of general knowledge.

POPULATION	ESTIMATE OF POPULATION SIZE	YEAR OF ESTIMATE	STATUS	NOTABLE VULNERABILITIES	QUALITY OF INFORMATION	JURISDICTION
Arctic Basin	unknown		Data deficient	Unknown	Data deficient	Circumpolar
Baffin Bay	2074	1997	Declining	Habitat decline; high harvest	Fair; periodic capture studies; TEK studies; on-going movement study	Canada (Nunavut), Greenland
Barents Sea	2650	2004	Data deficient	Habitat decline	Good; aerial survey; consistent capture studies since 1960s	Norway, Russia
Chukchi Sea	unknown		Data deficient	Unknown levels of harvest; habitat decline	Poor; new ecological and TEK study on-going; periodic and regional studies since 1990s	Russia, USA
Davis Strait	2158	2007	Has increased, now stable; decline expected	Habitat decline	New mark-recapture abundance estimate and vital rates; TEK study; periodic capture studies since 1970s	Canada (Nunavut, Quebec, Newfoundland & Labrador), Greenland
East Greenland	unknown		Data deficient	Contaminants; possible long historic over-harvest	Data deficient; movement studies planned and on-going	Greenland
Foxe Basin	2197	1994	Data deficient	Habitat decline; proposed increased shipping for mineral extraction	Aerial surveys 2009, 2010; new estimate expected; on-going habitat, TEK and movement studies	Canada (Nunavut, Quebec)
Gulf of Boothia	1592	2000	Likely an increase		Fair; periodic capture studies; TEK study	Canada (Nunavut)
Kane Basin	164	1998	Declining due to harvest	Habitat decline; likely a sink population connected with Baffin Bay; small population	Fair; one capture study; TEK study; aerial survey planned	Canada (Nunavut), Greenland
Kara Sea	unknown		Data deficient	Unknown levels of harvest	Data deficient	Russia
Lancaster Sound	2541	1998	Likely a decline due to harvest	Habitat decline; sex-skewed harvest; shipping expected to increase	Fair; periodic capture studies since the 1980s	Canada (Nunavut)
Laptev Sea	unknown		Data deficient	Unknown levels of harvest	Data deficient	Russia
M'Clintock Channel	284	2000	Likely an increase	Shipping expected to increase	Fair	Canada (Nunavut)
Northern Beaufort	1202	2006	Stable	Increasing oil/gas development and shipping; habitat decline	Good; new mark-recapture abundance estimate; periodic capture studies since 1970s; TEK study on-going	Canada (Nunavut, Northwest Territories)
Norwegian Bay	190	1998	Declining	Small population	Fair; one capture study	Canada (Nunavut)
Southern Beaufort	1546	2006	Declining	Severe habitat decline	Good; consistent mark-recapture monitoring	Canada (Yukon, Northwest Territories), USA
Southern Hudson Bay	900-1000	2006	Stable; predicted to decline	Habitat decline; decline of permafrost-based denning habitat	Good; research has been periodic since 1960s	Canada (Nunavut, Quebec, Ontario)
Viscount Melville	161	1992	Data deficient	Shipping expected to increase	Data deficient; one capture study; mark-recapture study planned	Canada (Nunavut)
Western Hudson Bay	935	2004	Declining	Severe habitat decline	Good; annual mark-recapture monitoring	Canada (Manitoba, Nunavut)

A wide variety of local and regional ecological studies have been conducted on behaviour (e.g., Dyck and Baydack 2004; Ovsyanikov 2005; Smith et al. 2010), movement (e.g., Born et al. 1997; Belikov et al. 1998), relatedness (Lunn et al. 2000; Crompton et al. 2008; Zeyl et al. 2009), denning (e.g., Belikov and Matveev 1983; Schweinsburg et al. 1984; Stishov 1991; Durner et al. 2003), foraging (e.g., Russell 1975; Smith 1980; Derocher et al. 1993, Iverson et al. 2006; Thiemann et al. 2008b) and contaminants (e.g., Dietz et al. 1995; Sonne et al. 2005) throughout the circumpolar Arctic. There have also been studies recording the Traditional Ecological Knowledge (TEK) of polar bears (e.g., Kalxdorff 1997; McDonald et al. 1997; Kochnev et al. 2003; Van de Velde et al. 2003; Keith et al. 2005; Tyrrell 2007; Dowsley and Wenzel 2008; Shannon and Freeman 2009; Born et al. 2010). While these studies may be small-scale or short-duration they have substantially increased our understanding of polar bear ecology. As a result they are useful in understanding patterns of exploitation and mechanistic relationships that underlie population status; these mechanistic relationships are important for more accurate predictive modelling (Molnar et al. 2009; Amstrup et al. 2008).

5.1 Monitoring capacity

5.1.1 Canada

In Canada, where approximately 65% of the world's polar bears occur, much of the obligation to fund and conduct polar bear research and monitoring is at the regional level (provinces and territories). However, the federal government has also conducted long-term research of the western Hudson Bay subpopulation (the continuity of which was made possible by funding from non-governmental organizations (NGOs)) and more sporadically in Davis Strait and the Beaufort Sea subpopulations. Intensity of research depends on the consistency of funding and capacity for both regional, federal and university projects, which is provided by governments, co-management boards and NGOs. While intensity of study varies over time and across Canada, there are government guidelines and agreements with local communities that propose to conduct population inventories on a 10 – 15 year cycle. The primary objective and much of the funding for research is focused on population status (e.g., Regehr et al. 2007; Obbard 2008; Taylor et al. 2009; Stirling et al. 2011) to establish sustainable harvest (Taylor et al. 1987b), although some research relates to broader ecological and mechanistic inquiry (e.g., Stirling 1997, 2002; Stirling et al. 1999; Molnar et al. 2010). Habitat ecology, movement and population distribution research (e.g., Taylor and Lee 1995; Ferguson et al. 1999, 2000; Taylor et al. 2001; Crompton et al. 2008; Towns et al. 2010) have been additional of Canadian biologists. There have also been collaborative research efforts with Greenland in Baffin Bay and Kane Basin (Ferguson et al. 1999, 2001; Taylor et al. 2001, 2005, 2008). There is also a national programme, which has monitored contaminants in polar bears for several decades (Norstrom et al. 1988; Verreault et al. 2006). Granting of permissions to conduct research in Canada (variously in Nunavut, Northwest Territories and Quebéc) are becoming less predictable as a result of lack of local support for the capture and handling of polar bears. As a consequence, some jurisdictions in Canada are developing and implementing less-invasive methods for monitoring polar bears (Peacock et al. 2009; Obbard et al. 2010).

Seventy percent of the world's legal polar bear harvest occurs in Canada (Peacock et al. in press) and harvest sampling and monitoring of polar bears is comprehensive throughout much of the seven regional jurisdictions that have polar bears. The Federal and Provincial/Territorial Polar Bear Technical Committee summarises annual harvest reporting. As a result of harvest monitoring, long-term harvest information has supplemented ecological research (Taylor and Lee 1995; Dyck et al. 2004), and harvest samples are provided for studies on foraging ecology (Thiemann et al. 2008b), contaminants (e.g., Verreault et al. 2006) and genetics (Paetkau et al. 1995, 1999). The monitoring and study of deterrence and defence kills have been on-going (Stenhouse et al. 1988; Dyck 2006; Towns et al. 2009).

5.1.2 United States

In the United States, the research, monitoring, and management of polar bears are conducted largely by the federal government. Additional research is conducted by the North Slope Borough, and there are auxiliary studies on disease, contaminants and foraging ecology by universities in conjunction with federal research (e.g., Bentzen et al. 2007; O'Hara et al. 2010). Habitat and demographic studies in the southern Beaufort Sea have been long term (Amstrup et al. 2001; Durner et al. 2009) and often performed in cooperation with Canadian scientists (Regehr et al. 2010). There have also been collaborative research efforts with Russia in the Chukchi Sea (Garner et al. 1990; Evans et al. 2003). The funding and human capacity for the study and management of polar bears in the United States is extensive. New infusions of capacity and funding as a result of the 2008 listing of the polar bear under the U.S. Endangered Species Act and the signing of the Agreement between the government of the United States of America and the government of the Russian Federation on the Conservation and Management of the Alaska-Chukotka polar bear population have increased analysis, research and monitoring of polar bears in the southern Beaufort Sea, and also has resulted in a new research and monitoring program focused on the Chukchi Sea subpopulation (DeBruyn et al. 2010). Notably, recent research efforts have focused on predicting future habitat, distribution and abundance of polar bears world-wide (Amstrup et al. 2008; Durner et al. 2009; Amstrup et al. 2010). The new research program in the Chukchi Sea (2008-2013) is expected to provide information on habitat use, movement patterns, reproduction, body condition, health, local and traditional knowledge, and foraging ecology as well as potential effects of sea ice loss. The ultimate goal of this program is to estimate population size, trends, and survival rates. The appropriate methods to accurately estimate these parameters for this wide-ranging and disperse population are still being investigated, including a genetics-based mark-recapture approach.

The U.S. Marine Mammal Protection Act requires hunters to report harvests within 30 days and the U.S. Fish and Wildlife Service maintains a marking, tagging, and reporting program. Harvest sampling is not required and few of requested samples are obtained from hunters. Reporting compliance has been estimated to be high (>90% of harvested bears are reported), but sample collection, including teeth for aging, has been low (<25%). Harvest samples have been contributed to circumpolar monitoring of contaminants and genetics.

5.1.3 Russia

There are four subpopulations of polar bears in the Russian Arctic: the Barents, Kara, Laptev and Chukchi seas. Since 1990, the Russian Academy of Sciences has collaborated with US Geological Survey scientists to study the sea ice habitats of shared polar bear populations. The research has pioneered new analytical methods for monitoring the decline of sea ice and investigating underlying mechanisms. Main findings have included: 1) duration of the summer melt season in the Arctic began to lengthen after the Arctic Oscillation shifted to a positive phase, especially in areas of the Chukchi Sea (Belchansky et al. 2004ab); 2) temporal patterns of perennial ice decline during the 1990s were related to the Arctic Oscillation while spatial patterns were related to melt dynamics (Belchansky et al. 2004c); 3) longer melt seasons and pronounced losses of perennial ice north of the Bering Strait were related to anomalous northward advection of warm southern air masses during spring (Stone et al. 2005); 4) since the mid-1990s, losses of very old ice (>10 years) were not being compensated by recruitment due to a prior depletion of most mature ice-age classes (Belchansky et al. 2005); and 5) average ice thickness and volume fluctuated during 1982–2003, peaking in the late 1980s, thinning until the mid-1990s and slightly thickening thereafter – but without a corresponding volume increase (Belchansky et al. 2008).

Russian scientists have developed collaborations with Norwegian biologists in research on the population status of the Barents Sea subpopulation (Aars et al. 2009) and with American researchers to study the Chukchi Sea subpopulation (DeBruyn et al. 2010). Coastal monitoring programs (http://www.belyemedvedi.ru/) have been developed across the Russian Arctic by local residents, in collaboration with non-governmental organizations, to increase deterrence activities, monitor poaching activities and to collect monitoring data (Belikov et al. 2010). Since 1990, there has been a research and monitoring project on polar bear behaviour, condition, demography and denning on Wrangel Island in the Chukchi Sea (Ovsyanikov 2005; Ovsyanikov 2010). There is a need for information on polar bears from the Kara and Laptev seas.

5.1.4 Norway

Long term research conducted by the Norwegian Polar Institute (federal government) has resulted in extensive information on population ecology (Lønø 1970; Larsen 1972, 1986; Wiig 1998; Derocher 2005; Aars et al. 2009), movement (Mauritzen et al. 2001, 2002), denning (Larsen 1985), behaviour (Derocher and Wiig 1999a, 1999b; Zeyl et al. 2009) and contaminant load (Skaare et al. 2000; Derocher et al. 2003) of polar bears on Svalbard and in the Barents Sea (Aars et al. 2010). One main focus of Norwegian work has been contributing to the monitoring of contaminants in polar bears (Bernhoft 1997). There has been no legal harvest of polar bears in Norway since the early 1970s, but numbers and characteristics of defence kills are monitored, as well as human-bear interactions, especially in relation to tourism on Svalbard (Gjertz and Persen 1987; Gjertz and Scheie 1998; Vongraven et al. 2010).

There is at present considerable effort being put into investigating cost-efficient monitoring schemes for polar bears in the Barents Sea. This work is lead by the Norwegian Polar Institute, and by 2011 a long term monitoring scheme of polar bear reproductive parameters as it relates to sea ice change will be established. It has been shown that the extent of sea ice cover in the areas around Svalbard have decreased the last three decades (O. Pavlova, pers. comm.), that sea ice thickness around Hopen has been reduced in the period 1966-2007 (Gerland et al. 2008), and that this affects polar bear den ecology (Derocher et al. submitted). Since 1988, there has been an annual springtime capture effort in Svalbard, and there are data on litter size, cub production, and age distribution (Derocher 2005) to be used for monitoring purposes.

5.1.5 Greenland

The monitoring and research of polar bears in Greenland is primarily conducted by the Greenland Institute of Natural Resources and has been on-going since the 1980s (Born 1991). The Danish Environmental Research Institute also has monitoring programs in collaboration with the Greenland Institute of Natural Resources and international research agencies specifically on the contaminants of polar bears (e.g., Dietz et al. 2006, 2008; Sonne et al. 2007a, 2007b). Since 1991, Greenland has collaborated on population inventories with the territorial government of Nunavut in Canada and other Canadian research agencies, in the shared populations of Kane Basin and Baffin Bay (Taylor et al. 2001; Taylor et al. 2005; Taylor et al. 2008a). Greenland has also focused on telemetry studies of movement and habitat use in Baffin Bay and Kane Basin with Canadian scientists (Taylor et al. 2001; Ferguson et al. 1999, 2001), and in east Greenland with Norwegian research institutes (Born et al. 1997; Wiig et al. 2003; Born et al. 2010). Recent studies include research on TEK of climate change and polar bear harvest (Rosing-Asvid and Born 1990, Sandell et al. 2001, Born et al. 2010). Regular monitoring of the substantial polar bear harvest in Greenland has occurred since 1955 (Born 1991). The collection of consistent hunter reports has been difficult and variable, but with the establishment of a system of reporting (Piniarneq) in 2003, and new reporting requirements and harvest quotas in 2006 (Born et al. 2010), harvest monitoring will become increasingly reliable. Harvest sampling is being planned.


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6. MONITORING METRICS AND METHODS

A science-based component of a circumpolar adaptive management plan for polar bears requires monitoring of the following crucial parameters: abundance, trend, distribution, harvest, human-bear interactions, individual body condition, and habitat.

Here we outline the metrics and describe available methods that are necessary for monitoring of these parameters in polar bears. At the end of each section, we highlight the recommended metrics.

6.1 Abundance

6.1.1 Metrics

The numerical size of polar bear subpopulations has been used as a primary indicator, or as a constituent, in a metric, for population trend and to determine sustainability of harvest. The PBSG reports a range of 20,000 – 25,000 bears worldwide, based on abundance estimates of some of the world's 19 subpopulations (PBSG 2010a); these estimates are variously dated, often imprecise and may have known and unknown biases. Nevertheless, this combined estimate is arguably more robust than the first global estimates of abundance of polar bears (Flyger 1967; Uspenski 1979; Larsen and Stirling 2009). Temporal comparisons of estimates of abundance, globally or in particular subpopulations, are fraught with caveats because of variation in methodology. There will likely be a continued call for estimates of abundance, for both the assessment of sustainable harvest and as an indicator of status. Abundance can be further used for an evaluation of population density when related to an accurate assessment of sea-ice habitat.

6.1.2 Methods

Estimating the abundance of polar bear subpopulations requires the infusion of money, effort, time, and expertise. The simple numerical goal defies the complex field and analytical requirements for estimation. We have no information on the abundance of polar bears in 5 of the 19 subpopulations. Data deficiencies are primarily a function of lack of financial support and logistical difficulties of operating in remote areas. Further, useful population estimates require accuracy and precision. Yet, even with considerable effort, bias and the lack of precision of estimates remain. The precision of abundance estimates vary based, essentially, on sample size, and variance of these estimates is an easily expressed and understood parameter. On the other hand, bias, the accuracy of an estimate, is more difficult to address and express, and is largely a result of non-representative sampling. Representative sampling of polar bears is difficult due to the remote nature of polar bear habitats. In addition, research access to their habitats can vary significantly. Often, there are no ancillary data (e.g., telemetry) with which to address or correct bias in a dataset.

Physical mark-recapture has been the cornerstone method for abundance estimation, especially in North America and western Greenland (Stirling et al. 1980; Stirling and Kiliaan 1980; Schweinsburg et al. 1982; Furnell and Schweinsburg 1984; Amstrup et al. 1986; Kolenosky et al. 1992; Derocher and Stirling 1995; Lunn et al. 1997; Amstrup et al. 2001; Taylor et al. 2002; Taylor et al. 2005; Regehr et al. 2006; Taylor et al. 2006a; Regehr et al. 2007; Stirling et al. 2007; Obbard 2008; Taylor et al. 2008a; Taylor et al. 2008b; Taylor et al. 2009). Field methods (Stirling et al. 1989; Cattet et al. 1997b; Cattet et al. 2003; Cattet and Obbard 2010) and analytical models (e.g., Seber 1982; Burnham 1993; McDonald and Amstrup 2001; Pledger and Phillpot 2008) have improved and expanded over time. Improvements in the data are mainly due to longer time series, increased effort in the field, inclusion of hunter recoveries, and auxiliary data on distribution obtained from satellite telemetry. A general sentiment is that abundance estimates for polar bears have become more precise and less biased. That said, capture and/or harvest recovery heterogeneity can still act to negatively bias estimates of abundance through unequal probability of capture/recovery due to behaviour, innate characteristics of individuals, and/or incomplete sampling. Lack of geographic closure, which is ubiquitous in polar bear subpopulations, also compromises the accuracy of abundance estimates.

Remote mark-recapture methods have also been used to obtain abundance estimates of terrestrial bears (e.g., Woods et al. 1999). In general, remote marking allows for the benefits of mark-recapture analysis, when physical capture is not possible, dangerous, or too expensive. For example, Taylor et al. (2006b) used darts to inject polar bears with a biomarker (tetracycline), which marked bones and teeth that were then recovered in the harvest, and provided the first size estimate for the Foxe Basin subpopulation in Canada. Mark-recapture methods using genetic identity marks have promise for polar bears, particularly the use of biopsy darts (Peacock et al. 2009) from helicopters, which would maintain broad geographic access to a population. Genetic identities can substitute for or supplement physical identities in a mark-recapture framework; this can reduce bias and increase precision of abundance estimates. Further, in regions where sufficient research access is not possible, collection of genetic material from multiple local sites (e.g., polar bear hair in dens and day beds, Umky Polar Bear Patrol, unpublished data; hair sampling in areas of high concentrations, Herreman and Peacock, unpublished data) can be used to supplement a broader data set of individual-identity data, to be used in multiple-source mark-recapture analyses (Cooch and White 2007).

Aerial surveys have been used to estimate abundance of portions of populations, entire populations or concentrations of polar bears (Crete et al. 1991; Wiig and Derocher 1999; Evans et al. 2003; Stirling et al. 2004; Schliebe et al. 2008; Aars et al. 2009; Peacock et al. 2009). In some areas, aerial surveys may be used to supply a single snap-shot abundance and density estimate. As with mark-recapture methods, analytical advances in the collection of these data for population estimation have improved (e.g., Buckland et al. 1993, 2001; Borchers et al. 1998, 2006; Laake et al. 2008). Specifically, double-observer and line-transect (i.e., distance-sampling) methods can be combined to decrease visibility bias, thereby reducing an historical tendency for surveys to produce negatively biased abundance estimates (e.g., Laake 1999 and see references therein). Bias in aerial survey estimates can be reduced if careful adjustments are made to the study design and analysis (Buckland et al. 2001). For example, perception bias (on the transect line) of observers can introduce sighting heterogeneity, such as the tendency for both observers to more likely see moving bears or larger groups; with sufficient data, perception bias can be addressed using co-variates associated with observations. Availability bias (e.g., bears in dens) can be accounted for by using outside information (e.g., as diving behaviour has been accounted for by other marine mammals; Barlow 1999). The entire geographic area to which line-transect data will be extrapolated needs to be surveyed. Alternatively, data can be extrapolated to un-surveyed areas based on relative area occupancy obtained from satellite telemetry, or on the assumption that the density of polar bears is the same. Low sighting probability of the white bears on ice or snow, and low densities of polar bears also requires a large number of transects for sufficient precision. Yet, adequate precision and standard effort is essential for detecting a trend between multiple estimates, because single aerial survey efforts cannot, notably, provide adequate information to measure population growth rate.

Ultimately, the decision to use genetic or physical mark-recapture or aerial survey for abundance estimation will depend on polar bear distribution and habitat specific to each subpopulation, and also the management requirements. As a result, background information on habitat use and seasonal distribution patterns is necessary before an appropriate methodological approach can be determined. Mark-recapture methodology has been a preferred method, because of the ability of the method to produce precise and accurate results, but also because of the supplementary data collected (especially for recruitment and age-specific survival), samples obtained, and deployment of satellite tags during polar bear capture. Further, physical (or genetic) marks become lasting marks in the subpopulation, which can be tracked to obtain subsequent, updated information on survival and delineation using data on the harvest recovery of marked bears (Taylor and Lee 1995). If capture efforts are made in subsequent years, updated abundance and survival estimates can be calculated. However, marks will attrite from the subpopulation as bears die, and estimates will beome increasingly biased and less precise. It should be noted that for genetic mark-recapture, because biological samples that provide genetic-IDs from earlier capture efforts may not exist, it will take time to develop genetic mark-recapture databases on par with some of the existing physical mark-recapture databases. Mark-recapture should only be used when marks can be applied systematically or randomly within the entire subpopulation (e.g., when the entire population is concentrated on shore, e.g., Derocher et al. 2005, Taylor et al. 2005, or are all accessible on sea ice, such as in the case of the Canadian Arctic Archipelago in the spring, Taylor et al. 2009). Alternatively, the method is appropriate when it can be correctly assumed, and supported with empirical data that bears marked from a smaller, accessible study area, randomly mix with the actual, larger population. Aerial surveys, which must have systematic transects across the area to which the data are inferred, may avoid the pitfalls of capture heterogeneity and geographic closure, especially when a non-representative portion of the population is available for capture (e.g., in the Barents Sea; Aars et al. 2009)

Because of local opposition to the capture and handling of bears in some regions of the Arctic (Peacock et al. 2010), concerns on the sub-lethal affects of capture and handling (Cattet et al. 2008) and difficulties associated with capturing a representative sample of bears (e.g., as noted in Aars et al. 2009), there has been renewed interest in the use of aerial surveys for polar bears. Furthermore, systematic data on polar bear habitat use can be simultaneously collected during aerial surveys; these data can then be used in various analyses involving occupancy-modelling (Gardner et al. 2010) or resource selection functions (RSF; Manly 2002). Aircraft platforms can also use belly-cameras to provide information on prey abundance and distribution (Estep et al. 1994) across the surveyed area. Further, aerial surveys can provide a relatively quick abundance estimate, whereas mark-recapture studies take at minimum two years (for abundance). Finally, while adequate training is certainly necessary, less specialised teams can be deployed to conduct aerial surveys.

Specific recommendations or metrics:

- Abundance of demographically independent units that are tractable for research.
- Abundance of multiple neighbouring subpopulations, which are related through demographic exchange, through analysis of combined datasets.

6.2 Trend: Recruitment, Mortality and Growth

6.2.1 Metrics

The trend of a population of polar bears is likely the most valuable metric for describing the status of the population. Population trend is equivalent to the population growth parameter, λ, and can be modelled with accurate assessments of abundance, natural survival, harvest, sex and age-structure and recruitment. Alternatively, trend can be assessed by comparisons of population estimates over time, or with temporal indices of abundance, provided they can be reliably calibrated. While in the past, human-caused mortality was considered the most important influence on trend, it is now apparent that the changing sea-ice habitat also impacts populations. Thus, there is a need for more effective and rapid monitoring of trends of polar bear subpopulations throughout the circumpolar Arctic. This will likely require developing a broader suite of methods for monitoring.

6.2.2 Methods

The primary method for assessing the growth rate or trend for polar bear populations has been the use of population viability analysis (PVA), which uses population-specific sex/age structures, vital rates, and selectivity/vulnerability-to-harvest matrices in life-table projections (Taylor et al. 2003; Taylor et al. 2005; Taylor et al. 2006a; Taylor et al. 2008a; Taylor et al. 2008b; Taylor et al. 2009). The stochastic PVA, RISKMAN (Taylor et al. 2003), has been formulated to allow annual population projection for a species with a three-year reproduction cycle (e.g., polar bears, walrus). This method of assessing population trend is only appropriate where good data on survival, recruitment, harvest and abundance are available for a population.

Analysis of long-term datasets of marked (physical or genetic) individuals can provide estimates of total and natural survival (e.g., Regehr et al. 2007; Taylor et al. 2008a; Taylor et al. 2009; Regehr et al. 2010). Survival estimates derived from mark-recapture data sets, however, are subject to bias, because they are sensitive to how data are collected and violations of model assumptions (in a similar manner to bias in abundance estimates, see above). Furthermore, datasets, which are short compared to the life of a polar bear, are less likely to provide precise and accurate estimates of survival, because relatively few bears actually die over the course of a short study. Increasingly, survival rates will vary with changing habitat, and therefore empirical estimates of survival and reproductive parameters (see below) will be valid for a shorter duration. Attempts to employ these methods should be undertaken when quantitative ecologists work in close association with field biologists, when financing is adequate to conduct extensive sampling, and when empirical data suggest that the study area is representative of the demographic population. Further, as with the estimation of abundance, the estimation of survival rates requires a good understanding of, and adjustment for, the geographic closure of a subpopulation. Because of the difficulties in obtaining accurate and precise estimates of survival, in some regions, it may be preferential to employ alternative methods to assess population trend.

Reproductive parameters often measured for polar bears include: litter size; litter production rates; mating interval; age of first reproduction; age of weaning; sex ratio at birth; natality; and whole litter loss. For some comparative analyses, the PBSG have used standardized methods for these metrics provided by Taylor et al. (1987b) which incorporate litter production rates that account for a 3-year birth interval (Taylor et al. 1987a); these rates are estimated from the standing age distribution based on captures, which are thought to be representative of the population (Taylor et al. 1987a).

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There are other formulations of litter production rates that have been used for polar bears (Stirling et al. 1980; Furnell and Schweinsburg 1984; Ramsay and Stirling 1988). There is a need to develop estimates of reproductive parameters for polar bears that can be gathered during aerial surveys, or potentially from harvest samples such as teeth (e.g., Medill et al. 2010).

Currently, the RISKMAN PVA does not allow for vital rates to vary, specifically according to some relationship to the environment, over the course of a simulation of population growth. While Hunter et al. (2010) combined a demographic matrix population model with habitat models for the polar bear population in the Southern Beaufort Sea, such sophisticated data are not available for most subpopulations. Population projection models, using vital rates and empirical relationships with projected changes in habitat, are ideal, but given logistical constraints, may not be useful for most subpopulations of polar bears.

Table 2: Harvest data and samples recommended for circumpolar monitoring program.

METRIC OR SAMPLE	DESCRIPTION
Number	Annual total human-caused mortality for each management unit
Sex	Sex of harvested bear; verification required
Field class	Adult, subadult, cub, family status
Location of harvest	Latitude and longitude, and written description
Hours/days actively searching	Only relevant in directed, controlled studies
Distance travelled to harvest	Kilometres or 'at camp or village'; can be gathered by directed study with GPS
Mode of conveyance	Boat, ATV, dog sled, snow machine, on foot
Type of human-caused mortality	Regulated, illegal, defence, research, sport
Lip-tattoo and/or ear tags	Research number for use in population modelling and distribution analysis
Tissue sample	Genetic individual identification; genetic sex identification; stable-isotope analysis
Fat sample	Fatty-acid diet analysis; contaminant analysis
Hair sample	Stable-isotope diet analysis; contaminant analysis
Reproductive tracts and organs	To understand implications of contaminant burden
Skull morphometrics	Skull length, zygometric breadth, etc.
Body condition	1-5 index; axillary girth measured by rope
Lower premolar tooth	Analysis of cementum growth layers for age
Age structure of harvest	To understand selectivity of harvest
Sex ratio of harvest	To understand selectivity of harvest
Polar bear parts traded commercially	Number and sources of hides, skull, claws; international and domestic sales

Trend of a population can also be inferred from the comparison of abundance estimates, given sufficient precision of the estimates, and attention to changing methodology that can bias the estimates differentially. Yet, even in one of the most intensively studied subpopulations of polar bears, because of uncertainty of the earlier estimates, no statistical change in population size of the southern Beaufort Sea was detected (Regehr et al. 2006), despite declines in habitat (Durner et al. 2009), survival (Regehr et al. 2010), and body condition (Rode et al. 2010). Whether or not the abundance of polar bears in the southern Beaufort Sea has in fact declined, the difficulty of detecting trend from subsequent abundance estimates is noted. Further, the abundance estimates, used to assess trend, can be ascertained through a variety of methods (see above). However, it is important to note that if mark-recapture is used, care should be taken to space intensive efforts relative to longevity of marks in the population. If subsequent mark-recapture efforts are undertaken, while sufficient numbers of marks are still present in the population, a new abundance estimate (for trend assessment) and updated survival estimates can be generated with only one additional sampling period.

In other species of wildlife, trend has also been determined by a known-fates method that uses vital rates derived from a sample of satellite-tagged female bears (Cherry et al. 2002); this method has potential to be employed with polar bears. However, the assumption is that mortality can be ascertained from satellite-tracking data. Further, this method could only occur in subpopulations with long-term satellite-tracking programs.

Given financial and logistical difficulties of obtaining accurate and sufficiently precise trend assessments (Garshelis and Hristienko 2006), in some cases, it may be advisable to focus on alternative, but multiple, lines of evidence. If the ultimate goal is to understand trend, short-term intensive work, to provide abundance and vital rates information, may fail due to the difficulties in obtaining adequate precision. Using parallel lines of evidence such as changes in condition or health (Sonne et al. 2005, 2006), an index of numbers observed per unit effort (Stirling et al. 2004; Schliebe et al. 2008), or cubs per adult female (Schwartz et al. 2008) may be cheaper, and more feasible to replicate across the circumpolar Arctic, and thus ultimately more informative. Indices need not be post hoc assessments of trend based on available data, but metrics can be carefully selected that are appropriate for each region, given various local constraints (Table 3). Suites of indices should be a priori designed and can come from harvest (Table 2, 3), capture (Table 4), or survey data (Table 3) collected by scientists or local people, trained in scientific-data collection. As an example of parallel lines of evidence, in a subpopulation where a non-representative portion of the subpopulation can be captured, a suite of indices could be designed: 1) cub: female ratios obtained from aerial and ground-survey data in various parts of the range; 2) condition metrics (1-5 index (Stirling et al. 2008) of harvested bears; 3) diet/food-web change from fatty-acid analysis of fat samples from the harvest (Thiemann et al. 2008b); 4) age of harvested bears; 5) catch-per-unit effort; 6) genetic samples from a combination of capture, biopsy, day beds, dens, biopsy darts, harvest analysed in a mark-recapture-recovery framework; and 7) aerial surveys for counts (Larsen 1972; Stirling et al. 2004; Schliebe et al. 2008; Gleason and Rode 2009). To the degree possible, the accuracy of any index used needs to be evaluated against quantitatively reliable population-based results, regardless of what parameter is being monitored.

To establish a circumpolar monitoring plan for assessment of population trends, care should be taken to determine the appropriate approach, because it is likely that the same methodology may not be appropriate for all populations. Options for monitoring trends include: population viability analyses; known-fates analyses; comparisons of periodic abundance estimates; or, parallel lines of evidence of multiple indices. After an approach is determined, appropriate intervals and intensities of study need to be determined, based on a priori power analyses of potential effect sizes.

Specific recommendations or metrics:

- Trend in abundance of demographically independent units by repeated population assessment.
- Trend in abundance by modelling vital parameters.
- Trend in indices developed from capture samples (Table 4), harvest samples (Tables 2 and 3) or remote observation (Table 3).

Table 3: Potential indices that could be researched, developed and implemented to provide parallel lines of scientific evidence of polar bear population status, in regions where more sophisticated mark-recapture, distance-sampling and capture methods cannot be used. Indices suggested here can be collected through remote observation or harvest by scientists and/or trained-local users. Annual indices must be conducted as the same time, in the same defined area and controlled for effort. Indices that can be determined through capture data are listed in Table 4.

INDEX	METHOD	LEVEL OF INFORMATION (i.e. representing the population or the harvest)
Minimum counts per unit area	• Aerial survey • Snow machine, ATV, boat or dog-team (community-based monitoring) • At concentrations (e.g. consistent bow-head whale piles or summer concentration areas) • Visual or genetic (material collected actively at corrals or with biopsy darts, or passively at day beds or dens)	Population
Cubs of the year per female; Yearlings per female	• Aerial survey • Snow machine, ATV, boat or dog-team (community-based monitoring) • At concentrations (e.g. consistent bow-head whale piles or summer concentration areas) • Visual or genetic (material collected actively at corrals or with biopsy darts, or passively at day beds or dens)	Population
Age structure	Ageing of growth layers in teeth	Harvest
Condition	• Tape/rope used by hunters to measure axillary girth of newly harvested bears (done with sternally recumbent carcass when freshly killed) • Condition index assessed by hunters (1-5) • Fat thickness at predetermined point(s) similar to what is often used for seals	Harvest
Catch-per-unit-effort	Hunter effort and geographic extent recorded by GPS	Population
Food-web/diet change	Fatty acid or stable isotope analysis of tissue samples	Harvest
Body metrics	Measurements of body metrics of harvested bears, for example: • skull metrics • straight line body length • axillary girth	Harvest
Litter production rates	Aerial or ground survey, using annual observation data on adult females, COY and yearlings	Population
Survival	• Relative age structure and availability: selectivity matrix (availability can be corrected by catch-per-unit) • Mark-recapture samples	Harvest and population

Table 4: Recommended data and samples (* minimum recommended) to be collected during scientific capture operations. In some cases capture programs are not intended to produce mark-recapture estimates of status. Therefore, indices of population status (population and individual health/condition) generated from these data could be researched and developed. Indices can also be determined through remote data collection (Table 3) and/or harvest (Table 2 and 3). Many of these data and samples are already collected by some or all scientists, however all metrics should be standardized at a pan-Arctic level.

INDEX	SAMPLE OR METHOD
Minimum counts per unit area	Record geographic tracks flown and numbers/types of bears observed (whether ultimately captured or not)*
Cubs of the year per female; Yearlings per female	Record reproductive status of each bear (whether ultimately captured or not)*
Age structure	Age class* or ageing of growth layers in teeth
Sex*	Male or female
Body condition	• Axillary girth* • Condition index assessed by scientists (1-5) • Mass • Bio-electrical impedance analysis (BIA)
Food-web/diet change	• Fatty acid analysis on fat plug or blood sample • Stable-isotope analysis on plucked hair* (least invasive), ear plug or blood
Body metrics	• Zygomatic breadth of skull* • Straight line body weight (to final tail vertebra)*
Contaminants	Hair* or fat plug

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6.3 Distribution

6.3.1 Metrics

Monitoring the seasonal and annual distribution of polar bears and their dens, and the delineation of their populations, allows us to document one way in which climate warming affects polar bears. Model assumptions for the estimation of abundance and survival using mark-recapture rely on defining a demographically-independent group of animals (i.e., a population), which necessitates geographic delineation. Further, shifts in the distributions of polar bears have been correlated with sea-ice conditions (Fischbach et al. 2007; Gleason and Rode 2009; Towns et al. 2010). Predicted reductions in polar bear optimal habitat (Durner et al. 2009) based on climate and ice projection models, have been used to predict reductions in polar bear range (Amstrup et al. 2008). Because the distribution of polar bears, and therefore delineation of populations, is not static, continued monitoring of distribution is essential in a rapidly changing sea ice habitat. Depending on research questions, biologists will need to subsequently adapt study areas over time as a result of changing distribution. Data on distribution and movements patterns can also assist in identifying potential conflict areas between humans and bears to inform management activities.

6.3.2 Methods

The distribution of polar bears and delineation of populations can be determined with satellite-tagging data, following fates of marked animals, and aerial surveys. Utilization distributions (Amstrup et al. 2004) and cluster analyses (Taylor et al. 2001, Mauritzen et al. 2002) have been the most common analytical approaches. Resource Selection Functions (RSFs) have also been used to assess distribution based on habitat selection of tagged animals and distribution of specific habitat attributes (see below; Ferguson et al. 2000; Durner et al. 2009) of satellite-tagged animals. Employing local knowledge or TEK, and using RSF and occupancy-modelling of aerial survey data may also be useful for describing polar bear distribution. The delineation of populations can further be informed by genetics (Crompton et al. 2008, Paetkau et al. 1995, 1999), and the capture, recapture and harvest recovery of physically (Taylor and Lee 1995) or genetically marked animals.

Specific recommendations or metrics:

- Delineated independent demographic units (populations).
- Delineated genetic units.
- Delineated ecological units.
- Extent of essential habitat.

6.4 Human-caused mortality

6.4.1 Metrics

Legal and illegal human-caused mortality of polar bears worldwide constitutes a significant source of mortality for polar bears (PBSG 2010a). The majority of human-caused mortality is through the regulated subsistence harvest of polar bears by aboriginal peoples. The interaction with climate warming will likely exacerbate direct interactions of polar bears by humans (Stirling and Derocher 1993; Stirling and Parkinson 2006; Amstrup et al. 2008) unless deterrence programs are successful (Towns et al. 2009). Therefore, despite the expected continued sea-ice habitat reduction, or in the event of mitigation of habitat decline through reduction of greenhouse gases (Amstrup et al. 2010), human-caused mortality of polar bears is critical to continue to monitor. Monitoring human-caused mortality of polar bears is variable across the Arctic, and needs to be improved, whether the harvest is regulated by government or by communities, or is illegal. Harvest monitoring can provide information on direct polar bear mortality, but there is also a broad wealth of ecological information that can be garnered from harvest samples and data, given attention to biases of a harvest sample (e.g., Peacock and Garshelis 2006; Table 2). Importantly, monitoring mortalities can work most effectively with community monitoring programs (see ch. 7). The collaboration of regional/federal governments with local communities can in turn garner support for research and conservation. Poaching should be monitored to the extent possible, through community programs and through monitoring of internet black-market sales. International and inter-jurisdictional trade on polar bear parts should continue to be monitored, including the relationship between commercial trade and harvest levels.

6.4.2 Methods

Monitoring of the regulated harvest should minimally include collection of the following parameters: numbers; location of kill; age-class; sex; individual identification (ear tag, tattoo, collars; Table 2). The full use of data from the harvest of marked individuals can provide additional information for age-structure, survival, abundance and population delineation. Harvest totals, age-at-harvest analyses (Conn et al. 2008) and catch-per-unit effort, with proper caveats, can augment ecological data on population abundance and trend (Tables 2, 3). Harvest sampling can contribute to ecological monitoring on population age-structure (teeth), contaminants (hair, fat, liver), genetics (any tissue sample), body condition (subjective index, axillary girth) and measurements of individuals (skull size), movement of marked individuals (tag recovery), diet (stable-isotope or fatty-acid analysis of samples) and health (pathology of samples, necropsy, reproductive tracts). Local hunters are keen observers and can be employed to properly document catch-per-unit and provide sampling.

Specific recommendations or metrics:

See Table 2.

6.5 Human/bear conflicts and problem bears

6.5.1 Metrics

Conflicts between humans and bears are inevitable in areas where both reside, but the level and number of conflicts is increasing (e.g., in Foxe Basin, Peacock et al. 2010), and in some areas are correlated with a change in distribution of polar bears due to climate warming (e.g., in western Hudson Bay; Stirling and Parkinson 2006, Towns et al. 2009). Ultimately, as an increased number of people (increasing local populations, tourists and employees in resource extraction) inhabit the Arctic, there will be increased interactions across the range of polar bears. As human-bear interactions have the potential to increase direct human-caused mortality, it is important to monitor and mitigate (in an adaptive management framework) the trend in human-bear conflicts, defence kills and distribution of bears in areas of human activity.

6.5.2 Methods

Monitoring of the frequency and temporal trends of human-polar bear interactions requires that the definitions of human-bear interactions and "problem" bears be standardized. Due to the interest in understanding and mitigating the increasing level of conflicts between humans and bears and the need to standardize data collected on bear-human interactions, recent discussions between the Parties to the 1973 Agreement on the Conservation of Polar Bears focused on addressing this issue. As a result, USA and Norway have in unison started implementing a Polar Bear Human Information Management System (PBHIMS) to understand when and why interactions between polar bears and humans occur (DeBruyn et al. 2010).

Specific recommendations or metrics:

Continuous monitoring of human-polar bear conflicts and mortalities.

6.6 Individual condition and health

6.6.1 Metrics

It has been predicted that changes in body condition precede demographic change in polar bear populations (Stirling and Derocher 1993; Obbard 2008). Indeed this appears to have been the case for polar bears in western Hudson Bay (Stirling et al. 1999; Regehr et al. 2007), the southern Beaufort Sea (Regehr et al. 2010; Rode et al. 2010), and potentially in southern Hudson Bay (Stirling and Parkinson 2006; Obbard 2008). While various pathologies of polar bears have been opportunistically studied (Born and Henriksen 1990; Taylor et al. 1991; Asbakk et al. 2010; Jensen et al. 2010; O'Hara et al. 2010), there are no consistent circumpolar monitoring programs on polar bear condition or disease, likely because of the relatively low incidence of polar bear disease (see ch. 2.4). A wide variety of condition, health and contaminant metrics and samples can be collected from captured and harvested bears. To some extent, condition metrics can be determined through ground and aerial surveys. Currently, measures employed to assess body condition and health vary greatly across local and regional research programs. A critical aspect of a pan-arctic polar bear monitoring program will be to standardize the specific measurements that are collected and used to assess body condition and health across subpopulations and regions. Table 4 lists minimal body condition and health metrics that should be collected during capture operations; Tables 2 and 3 lists metrics that can be collected during harvest and remote observation.

6.6.2 Methods

Individual body condition of polar bears has been quantified for captured bears using a wide variety of measures, including body mass (Rode et al. 2010), morphometrics (e.g., skull width; Rode et al. 2010), bioelectrical impedance analyses to estimate body fat (Farley and Robbins 1994), body composition modelling (Molnar et al. 2009), and several condition indices (Cattet et al. 1997a; Cattet and Obbard 2005; Stirling et al. 2008). There is a need to identify which of these measures best represent reproductive output and survival and thereby, are appropriate indicators of population health and status. For example, body mass, measured directly or via equations incorporating axillary girth and calibrated for specific populations at specific times, has been correlated with reproduction and cub survival (Derocher and Stirling 1996; Rode et al. 2010) and other ursids (Noyce and Garshelis 1994; Hilderbrand et al. 1998).

Efforts have been made to identify condition indices that will calibrate body mass across ages and bears of different physical stature (Cattet et al. 2002; Stirling et al. 2008), but these efforts may not be necessary and have not always been verified to correlate with reproduction and/or survival. In general, studies of ursids support that larger-bodied individuals equate to populations that have more high quality and/or abundant food resources and occur at higher population densities (Noyce and Garshelis 1994; Hilderbrand et al. 1998; Zedrosser et al. 2006). Thus, the overall size of bears of similar ages may be the best metric of individual condition. Identifying the simplest and easiest measurement (most precise and accurate) that can be ascertained from capture or harvest would be the best metric for a circumpolar monitoring program. Axillary girth, in particular, holds significant promise as it is does not require specialized tools and can be accurately measured on both harvested and captured polar bears. While skull width has been shown to be a useful measure of bear condition (Zedrosser et al . 2006; Rode et al. 2010), measures would have to be consistently taken on unskinned skulls (see Tables 2 - 4 for recommended condition metrics).

Specific recommendations or metrics:

Systematic collection of measurements and samples on body condition during capture (Table 4) and harvest (Tables 2, 3).

6.7 Habitat

6.7.1 Metrics

Sea-ice habitat provides for foraging, mating, sheltering/denning and migration, and constitutes an important metric to monitor. To date, there has been no concerted effort to standardize annual monitoring of habitat for polar bears. Properties of sea-ice are collected daily by a variety of radar and satellite imagery tools (e.g., SSMRI, MODIS, RADAR SAT). These images can be used, variously, to measure the availability of important polar bear sea ice habitat. Polar bears have been shown to select certain features of sea ice habitats that can be monitored with radar and satellite imagery tools including the timing of ice formation and melt, ice thickness, ice age, ice extent, floe size, and also the ocean depth over which ice forms. Micro-scale sea ice features such as pressure ridges and polynyas are also important for polar bear foraging (Stirling 1997). Durner et al. (2009) defined optimal habitat for adult female polar bears in the divergent and convergent-ice eco-regions as annual ice over the continental shelf. Thus there are a wide variety of potential ice metrics that may prove useful for the monitoring of habitat use, selection and decline (Ferguson et al. 2000, 2001; Mauritzen et al. 2003a; Durner et al. 2009). Dedicated study on both polar bears and their habitat and habitat selection (see below) will be necessary to identify the most pertinent habitat metrics for annual monitoring. This approach will be most effective if key habitat features have demonstrated empirical relationships with population health, density and/or trend.


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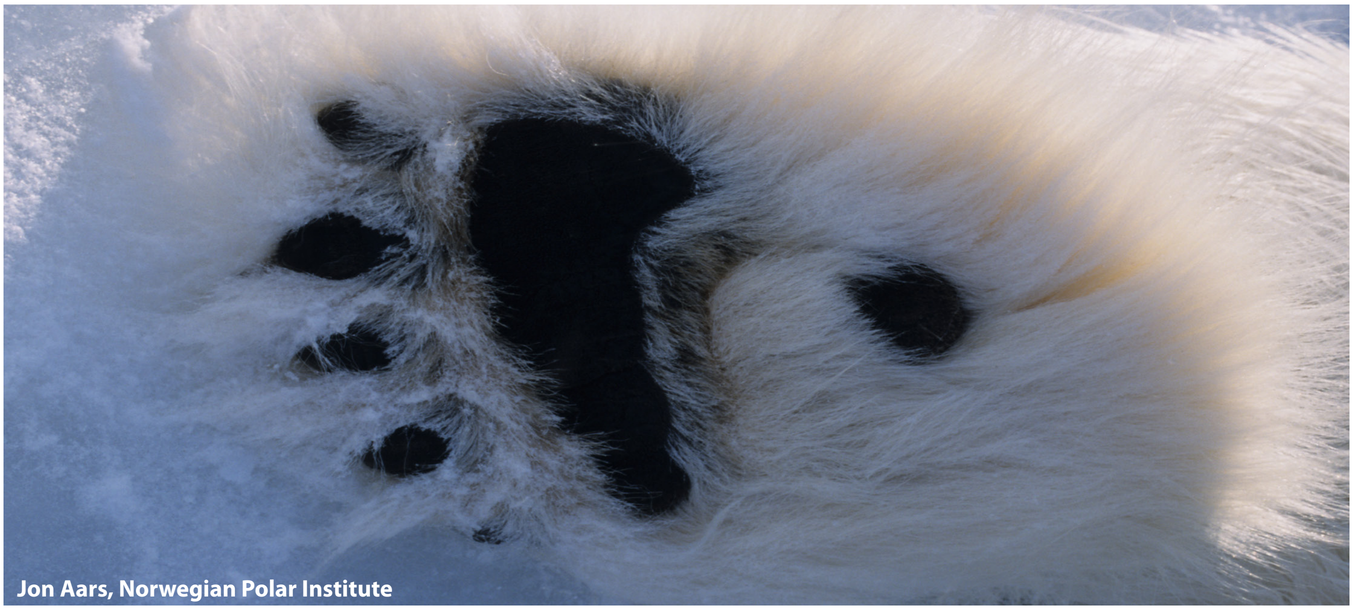
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6.7.2 Methods

Monitoring habitat will require collaboration between polar bear specialists and experts in sea-ice imagery. First, defining optimal polar bear habitat (e.g., Durner et al. 2009) is still a developing field of study, given the ecological variation across the circumpolar Arctic; thus we first must define a common metric of optimal habitat or other important habitat elements (e.g., combination of factors such as ice-free days, water depth, breakup date). This metric should not only relate to population productivity, but should be easy to measure on an annual basis, across the Arctic; the metric may be different for different eco-regions (Amstrup et al. 2008). There are a wide variety of satellite-imagery products available, and research (see below) is necessary to determine which metric(s) provide best indications of crucial habitat elements (e.g., freeze-up or annual ice over the continental shelf) that may indicate potential change in the status of polar bears. The most appropriate metric, or combination of metrics, may vary with different populations.

Specific recommendations or metrics:

Determination via satellite telemetry and TEK the extent and change in polar bear optimal habitat.

6.8 Monitoring changing foodwebs

Other Arctic marine mammals, which constitute the prey base of polar bears, are also undergoing stress as a result of climate warming (Laidre et al. 2008). The consequence of food-web change on polar bear population status is unknown, although some studies indicate that large-scale changes in reproductive rates, which were thought to be related to abundance of marine mammal prey, can have a quick and significant impact on polar bear populations (Stirling 2002). There has been some suggestion that food-web change as a result of climate warming may also increase contaminant burdens for polar bears (McKinney et al. 2009). Comprehensive monitoring can provide annual changes in diets of polar bears via techniques such as fatty-acid (FA) or stable-isotopes (SI) analysis of polar bear tissues by annual capture (fat, blood, hair), harvest (fat, hair, muscle) and/or community monitoring programs (hair from day beds, dens, barbed-wire corrals). The variation of polar bear food-webs is an active area of research (e.g., Bentzen et al. 2007; Thiemann et al. 2008b) as polar bear food-webs are changing (Ferguson et al. 2010) and laboratory techniques are being verified and improved. Combining fatty acid and stable isotope approaches will likely be the most effective for monitoring polar bears diets. Verification of coefficients used to model diets via fatty acids is still needed, but this approach allows for many potential dietary items to be considered. A new model for applying stable isotopes to quantify polar bears diets (Cherry et al. 2011) will significantly improve the accuracy of diet estimation based on stable isotopes. Before a monitoring program of food-webs can be designed, more research is necessary to coalesce on the most appropriate metrics to monitor and which techniques to use.

Specific recommendations or metrics:

Collect appropriate samples from capture and harvest samples for fatty-acid and stable-isotope analysis.

7. COMMUNITY-BASED MONITORING

Community-based monitoring can be an effective method for collecting data and engaging the public that lives in polar bear habitat. Across the globe, human dimensions of wildlife management are incorporated in conservation planning, and in some cases, may prove to be the single most important element of a successful government conservation (and research/ monitoring) program (e.g., Servheen 1998). Ignoring public opinion regarding wildlife status and its conservation can result in failure of on-the-ground conservation efforts. As an example, the Committee on the Status of Endangered Wildlife in Canada re-assessed the polar bear in 2008 as a species of special concern (COSEWIC 2008). Some scientific experts believe that this assessment failed in recognizing the more vulnerable predicament of polar bears, as a result of habitat decline in the face of climate warming (PBSG 2010a; Vongraven 2009). In contrast, the plurality of local people who live in polar bear habitat vigorously rejected COSEWIC's assessment as too extreme on the basis that they viewed polar bear populations to be healthy (Government of Nunavut 2010; PBSG 2010b). The result is that 4 years after the assessment, the federal government of Canada has yet to make a decision on how to officially classify the status of the polar bear; conservation action has been stalled. How do scientists and local people come to diametrically opposite conclusions? The issue may be a result of different observation platforms, geographic and temporal perspectives and ways of interpreting observations. The problems may also stem from the difficulty for government managers to communicate (presently or historically), failure to address local concerns of safety and property damage from polar bears, and failure to demonstrate the incorporation of TEK and local values in decision-making (Henri et al. 2010, Peacock et al. 2010). Further, the communication of results of scientific studies to local communities often does not occur (at least in an effective way). One clear step towards facilitating conservation efforts is to re-double efforts to engage local people in decision-making, but moreover, as stakeholders in the research and monitoring of polar bears (e.g., see Kindberg et al. 2009).

The local monitoring of harvest is a natural fit for community-based monitoring. Governments can require and design the collection of harvest data and samples (see harvest section above), but after training, individuals in communities can often be relied on to carry out monitoring, if local government representatives are not available. Further, token financial compensation for provision of harvest samples, likely is not on its own a significant incentive to hunt a polar bear. Data on catch-per-unit effort and geographic distribution of hunting effort can also provide useful metrics that, over time, might be useable as an index of population size or trend. It is essential however that the validity of such indices be confirmed independently with some other quantitative methodology.

There are several examples of community-based scientific data collection that have been successful, specifically in the Arctic. For example, the annual spring bowhead whale (*Balaena mysticetus*) census conducted by the Inupiat and the local North Slope Borough government (Alaska, USA) since the late 1970s has been successfully used alongside government studies (George et al. 2004). The North Slope Borough has begun collaboration with U.S. federal scientists to monitor polar bear use of bowhead whale bone piles using genetic tagging (Herreman and Peacock, unpublished data). Laidler (2006) and Laidler et al. (2008, 2009) use community-based monitoring of ice conditions for sea-ice modelling. In Arctic Russia, the Umky Bear Patrol, which originally grew out of the need to mount deterrence patrols, is now participating in polar bear den monitoring and genetic sample collection. Lastly, training in local communities provides jobs and education associated with science and western-style wildlife management. As an example, some satellite-tagging of bearded seals on the North Slope of Alaska and of ringed-seals in Hudson Bay and the Beaufort Sea (S.H. Ferguson, pers. comm., T.G. Smith, pers. comm.) is now conducted by trained local residents, without the presence of professional scientists.

Community-based monitoring or research does not suggest using TEK in preference to science, but rather aims at obtaining a synergy between systematically collected TEK and scientific data, while simultaneously increasing the participation of local users in the management of their resource and facilitating dialog between local people and government. Further, the collection of actual TEK can generate scientific hypotheses on causation and mechanisms of ecological change for polar bears (Peacock et al. In press).

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8. KNOWLEDGE GAPS AND RESEARCH NEEDS

Some information needs for the conservation and management of polar bears supersede what can be ascertained from monitoring efforts. Although much of the information gathered through monitoring (e.g., samples, vital rates, and abundance) can also be used to understand underlying ecological mechanisms, there are some knowledge gaps that will require either baseline or more sophisticated ecological research. These knowledge gaps can be categorized as: 1) content gaps (e.g., unknown empirical links between habitat and population demographics); 2) geographic gaps (noted in the Introduction); 3) methodological development (e.g., predictive modelling, bioinformatics, laboratory techniques); and 4) development of management policies in need of scientific input and/or human-dimensions research.

Polar bear habitat and its rate of change

Prior to creation of a monitoring plan for polar bear habitat, we must conduct research to determine which attributes of polar bear habitat are related to demographic change. A basic component of a necessary habitat research program, therefore, is to more fundamentally understand the relationship between sea ice habitat and polar bear population health and productivity. Inherent in identifying a metric for monitoring, sea-ice modelling (e.g., Belchansky et al. 2004; Gough et al. 2004; Douglas 2010), polar bear habitat selection (Ferguson et al. 2000; Mauritzen et al. 2003a) and predictive modelling of habitat change (Durner et al. 2009) are necessary research foci.

Properties of sea ice habitat are measured by a variety of radar and satellite imagery tools (e.g., SSMRI, MODIS, RADAR SAT). The imagery products differ in their resolution, frequency, and scope. More research is needed to: 1) develop bio-informatics capabilities to process satellite imagery important for polar bear habitat; 2) use satellite imagery to define both more broadly and more specifically the variation in sea-ice habitat selection; and 3) develop less-invasive, more reliable satellite tracking devices (glue on tags and ear tags) that can be deployed on various sex and age classes of polar bears.

Researching polar bear habitat largely requires continued deployment of satellite tags on polar bears, especially on bears of other sex and age classes besides adult females. There is therefore a need for extensive bio-informatics (managing of telemetry databases) and intensive field operations, which require human and financial capacity, and coordination across scientific disciplines and jurisdictions. Available TEK represents a very specific understanding of sea-ice (Laidler 2006) and its use by polar bears (Obbard et al. 2010); the use of TEK in understanding polar bear habitat ecology should not be underestimated (Laidler 2006; Sahanatien et al. In prep).

Durner et al. (2009) used RSFs (Arthur et al. 1996) to determine the characteristics and availability of sea ice and other habitat features selected by satellite tagged polar bears in the convergent and divergent ice eco-regions. Projected future availability of sea ice habitat based on climate models was subsequently used to predict future global distribution and abundance of polar bears (Amstrup et al. 2008). Knowledge gaps include similar habitat projection models in the two other geographic eco-regions identified by Amstrup et al. (2008) where ice forms and melts differently. In addition, habitat selection models are based only on satellite tracked adult females; other sex and age classes of polar bears likely use habitat differentially.

Combining habitat and demographic projections (Hunter et al. 2010) should also be a focus of new research. The interaction between polar bear population and habitat ecology is not well understood as polar bear habitat is variable throughout the species range (habitat can be categorized into different eco-regions, following the nature of how ice forms and melts (Amstrup et al. 2008; Durner et al. 2009).

Much of the information needed to understand polar bear habitat will require dedicated research, above and beyond the monitoring of sea ice habitat. Further, deployment of satellite tags can occur simultaneous to capture efforts aimed at population monitoring (see ch. 5).

Food-webs and food-web change

Decades of behavioural research (e.g., Smith and Stirling 1975, 1978; Stirling and Latour 1978; Stirling and Øritsland 1995) and analysis of polar bear fat (Thiemann et al. 2008b) have largely confirmed the ringed seal as polar bears' primary prey, with other food as secondary prey items, notably including the bearded seal, harp seal, walrus, beluga, and the scavenging of bowhead whale. Only recently, through the use of stable isotope and fatty-acid analysis, have the assimilated diets of populations and individuals been reconstructed (e.g., Derocher et al. 2002, Thiemann et al. 2008b). These studies have demonstrated geographic and sex-age class variation in polar bear diets. More research is needed to document the geographic variability in food webs, individual variation and the changes in food web structure throughout polar bear range. More fundamentally, we need to further understand basic empirical links between: 1) habitat quality/quantity and polar bear prey and diet; and 2) diet and population health and productivity. Much of this mechanistic research on polar bear diet can be undertaken simultaneously with capture and harvest monitoring programs, during which tissue samples can be collected. More fundamental physiological questions will require dedicated research and collaboration between polar bear specialists and physiologists, nutritional ecologists and those who work with captive polar bears. For example, the fractionation of stable-isotopes in the variety of polar bear tissues and the physiological calibration between stable-isotope and fatty-acid profiles and actual diet, remain obstacles to a more complete understanding of polar bear diet. Lastly, coordination with seal research and monitoring is highly advised to understand potential changes in polar bear distribution and population health.

Impact of natural resource development

Currently, polar bear habitat and industrial development overlap most significantly in the Beaufort Sea and the Arctic coastal plain of Alaska, USA and the Yukon and Northwest Territories, Canada. Researchers have documented little effect of the current industrial foot-print on polar bear abundance, habitat-use or denning (Amstrup 1993, Durner et al. 2000). Empirical research on den distribution, the modelling of preferred den habitat (Durner et al. 2001) and the development of den-detection tools (Amstrup et al. 2004a) can help industry design development to avoid polar bear dens. With climate warming, Arctic waterways will become more accessible to resource development. We do not know if expanding industrial footprints and most importantly, their cumulative effects, including increased ice-breaking, will begin to impact polar bears and their prey. At the very least, we must better describe den distribution where it has not been well-documented (e.g., Foxe Basin, Davis Strait, Laptev Sea, Kara Sea, East and West Greenland). Further, research on polar bear movements and habitat selection near ice-breaking channels and industrial off-shore and on-shore developments is needed.

Crude oil is fatal to polar bears due to disruption of their thermoregulation (Øritsland et al. 1981, Hurst et al. 1991). Little is known about the response of polar bears to oil spills (Stirling et al. 1990). The modelling of polar bears' response to the hypothetical trajectory of an oil spill (Amstrup et al. 2006), using location-specific currents and locations of drilling rigs and polar bear habitat-use can likely aid in planning for an oil spill event. Yet, detailed knowledge of polar bear habitat use is, for the most part, poorly understood at a holarctic level.

At a management level, there has been little input by polar bear specialists on action plans for response to oil spills in the Arctic. Concerns include lack of: 1) infrastructure (boats, ice breakers, deep water ports, airstrips, logistic bases) for response access in remote regions; 2) methods to contain and collect oil from ice-covered seas; 3) basic knowledge of polar bear distribution and abundance in areas of industrial development, and finally, 4) protocols, knowledge and infrastructure to rehabilitate and relocate oil-soaked polar bears.

Illegal harvest and un-monitored harvest

There are populations of polar bears, where part or all of the harvest is not monitored (illegal harvest in Russia; legal harvest in Quebec, Canada), and where harvest monitoring and sampling can be improved (Greenland; USA). Dedicated research projects (e.g. Born et al. 2011) can be used to retrospectively understand patterns in polar bear harvest. Further, human-dimensions research can be aimed at developing the best methods of community-based monitoring of harvest. Investigative research can also be conducted on the internet, or by community-based researchers to ascertain the level of illegal harvest (Kochnev 2004).

Ecological correlates of population change

Population vital rates (recruitment, mortality, abundance, density) are important metrics for monitoring polar bears. To understand how these vital parameters are related to habitat characteristics and prey base will require devoted research. Amstrup et al. (2008) used a Bayesian Network (BN) to predict future polar bear abundance and distribution, as an attempt to include more ecological information than a simplistic model based on known densities and future predicted ice area concentrations. The BN relied on both empirical data, but also expert knowledge, where empirical data were lacking. A substantial gap in knowledge was an understanding of the relationships between ecological attributes such as habitat, food webs, or levels of pathogens to demographic productivity. Perhaps more than the creation of new modelling techniques (see below) is the importance of understanding the basic and mechanistic ecological relationships. This information will allow us to better predict future polar bear harvest levels, abundance and distribution (Amstrup et al. 2008; Molnar et al. 2009).

Modelling capabilities

Since the 1980s, much of polar bear management and conservation (especially in Canada) has relied on predictive modelling (Taylor et al. 1987). Such modelling has provided managers with information to provide for the maximum sustained harvest of populations (Taylor et al. 1987b) and population recovery from overharvest (Taylor et al. 2002). More recently, modelling efforts have also focused on RSFs for the understanding of polar bear habitat (Durner et al. 2004, 2009; Ferguson et al. 2000; Mauritzen et al. 2003) and to forecast changes in the availability of polar bear habitat (Durner et al. 2009). Combining empirical vital rates and harvest data have been used to model potential for allee effects in harvested polar bear populations with skewed sex ratios (Molnar et al. 2009). Further, physiological metrics, vital rates and habitat information have been used to predict changes for polar bears (Molnar et al. 2010). As mentioned above, Amstrup et al. (2008) have used BNs to combine a large variety of empirical information with expert opinion to broadly predict future polar bear abundance and distribution with climate warming. Finally, Amstrup et al. (2010) have explored various greenhouse gas mitigation scenarios in habitat models for polar bears, to conclude that curtailing emissions sufficiently could mitigate the decline in polar bear habitat.

While there has been much progress in the modelling capabilities for polar bears over the last decade, some specific research needs remain. First, more empirical data to inform models is needed, particularly with respect to predator-prey relationships and the potential effects of sea ice change on these relationships. Second, there is a need to develop analytic techniques that allow different types of data and their linkages to be combined for abundance and distribution forecasts. For example, there is a need to combine predictive habitat change models with population viability analyses (e.g., Hunter et al. 2010). Third, population viability models that are built on survival and recruitment data that can be obtained from studies that do not require polar bear capture (aerial surveys, genetics and/or from harvest) would be very useful. Finally, there is also a need for database design and bio-informatics development to better collect, filter, combine and process large data sets (e.g., marked bear data, contaminants data and satellite telemetry locations, habitat imagery).

Human Dimensions Research: Lack of management mechanisms for catastrophic events

One predicted consequence of climate change is the increased variability in weather patterns. In the Arctic, in terms of polar bears, this prediction can be manifested by very late freeze-up dates, early-break up dates, increased impact of strong winds on thinning ice packs, or rain on snow events, which could impact seal recruitment and affect snow dens of polar bears. Sudden events may cause unexpected increased density of nutritionally-stressed polar bears on land and/or close to human settlements. There are no management plans in place to cope with potentially catastrophic consequences for polar bears. However, the public, both southern and northern, will expect action from wildlife managers. Managers will request guidance from scientists and local users. It is incumbent upon polar bear and northern experts to contribute to the formulation of these plans. For example, in an emergency scenario (e.g., stranding, starvation, disease, oil spill), would culling of polar bears be recommended, feasible or ethical? Would there be sufficient zoo infrastructure to house some polar bears and, if so, how many? Can orphaned cubs be re-released into the wild after rehabilitation? Can polar bears be re-located or, should they be, and if so, under what circumstances? These questions need to be addressed, which will require research to best understand the ecological and biological underpinnings of the event, public perception of the ethics of responding in different ways, and polar bear husbandry.

Plasticity

Many questions from the public and wildlife managers focus on whether the polar bear will "adapt" to its changing circumstance. Every biological population undergoes evolution by natural selection, which occurs via the differential survival and reproduction of individuals with various heritable traits. Plasticity can be defined as the intrinsic capacity for variation in life-history of polar bears (physiology, behaviour and phenology), and therefore is the raw material required for adaptation. There is, in fact, geographic and phenotypic variation in polar bear life-history (e.g., Charmantier et al. 2008; Derocher 1999; Thiemann et al. 2008a; Amstrup et al. 2008) and there have been documented changes in polar bear behaviour, which have been linked to changes in climate (Fischbach et al. 2007, Gleason and Rode 2009; Towns et al. 2009). However, the ability of polar bears to cope with environmental change via adaptation by natural selection is unlikely to occur rapidly enough to facilitate their continued survival in regions that are predicted to become ice-free (Derocher et al. 2004; e.g., Hudson Bay, Davis Strait and Baffin Bay, Amstrup et al. 2008). The physiological changes that would be necessary for a population to adapt (the polar bear has a 15-year generation time) to foraging in a substantially altered ice-free environment would unlikely occur at the rate that ice habitat is projected to change. Of course, the global polar bear population will evolve, as do all biological populations, though this change will more likely manifest itself via a severely curtailed distribution (Amstrup et al. 2008), as opposed to a physiologically- or morphologically-altered bear. Nonetheless, a more thorough understanding of existing polar bear life history will render predictive modelling of future polar bear abundance and distribution more accurate.

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