Status Report on the World's Polar Bear Subpopulations

IUCN/SSC Polar Bear Specialist Group

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Introduction

Polar bears have a circumpolar distribution limited to portions of the northern hemisphere covered by sea ice for extended portions of time each year. They are not evenly distributed throughout the Arctic, nor do they comprise a single nomadic population. Rather, they occur in 19 relatively discrete subpopulations (Figure 1). The subpopulation summaries and statuses listed here are based on information provided and discussed by members of the International Union for Conservation of Nature (IUCN) / Species Survival Commission (SSC) Polar Bear Specialist Group (PBSG) by remote correspondence during autumn 2023. We present estimated subpopulation sizes and associated uncertainty in those estimates, subpopulation trends, changes in sea-ice habitat, recent human-caused mortality, and summaries of subpopulation-specific concerns and vulnerabilities.

Subpopulation delineation

Subpopulation boundaries were initially established in the 1960–1970s based on information on seasonal fidelity, reconnaissance surveys, local knowledge, natural barriers to movement (e.g., land, open water), management considerations, and as capture-recapture studies advanced, from recapture of marked bears and harvest tag returns. From the 1980s to date, telemetry data has augmented subpopulation delineation and been the primary means of establishing boundaries. Use of genetic analyses has also provided insights on population structure but has not been used to set boundaries between areas. Gene flow among adjacent subpopulations is common and thus, biologically, the 19 identified subpopulations have similarity with management units. Despite this similarity, the subpopulations are intended to reflect group membership and vital demographic processes internal to these groups, which are the primary drivers of population dynamics.

The boundaries of subpopulations are intended to identify discontinuities to movement and are based on the best available scientific data. Past studies identifying boundaries have used clustering or kernel density methods to identify units, although many earlier boundaries were based on limited data without rigorous scientific assessment. There is no single recognized approach for boundary delineation. To facilitate long-term monitoring and management, boundaries have been moved infrequently over time, although changes have been made if new analyses supported a biologically meaningful adjustment. New methods (e.g., dynamic movement models; Scharf *et al.* 2019) have been developed in recent years to delineate boundaries but there has not been a large-scale reassessment of existing subpopulation delineation for polar bears. Subpopulation boundaries are likely dynamic over long periods and are predicted to shift with climate warming and the resulting loss of sea-ice habitat.

Ecoregions

Polar bears rely on the sea ice to access their seal prey, but sea ice is not uniform across the Arctic, rather there are four distinct sea ice habitats—or ecoregions—that have been recognized as differing in sea ice freeze-up, break-up, and drift patterns (Amstrup *et al.* 2008). Also, the ice among and within ecoregions lies over ocean waters that vary in biological productivity. Sea ice will continue to be lost in all four ecoregions unless climate warming is addressed (IPCC 2019). However, differences in ice character, ocean productivity, and other factors mean that the impacts of warming on the 19 polar bear subpopulations living within the four ecoregions are, and will continue to be, different.

Seasonal Ice Ecoregion
The polar bear habitats in much of central and eastern Canada lie within the Seasonal Ice Ecoregion, where, unlike the rest of the Arctic, the sea ice always has melted entirely in summer, forcing bears ashore, where there is little to eat. During the ice-free periods, polar bears live largely off their fat reserves until the ice forms in the fall and they can hunt seals again. In the past, polar bears thrived in this seasonal environment because it almost entirely

encompasses shallow and productive waters over the continental shelf. The rich environment historically allowed polar bears to gain enough weight in spring to survive a long summer fast. Five polar bear subpopulations occur in the Seasonal Ice Ecoregion: Baffin Bay, Davis Strait, Foxe Basin, Southern Hudson Bay, and Western Hudson Bay. Throughout the Seasonal Ice Ecoregion, ice is melting earlier in the spring and freezing up later in the fall, and the bears are food deprived for longer periods. Much of the Seasonal Ice Ecoregion occurs at the southern extreme of the polar bear's range. The number of ice-free days faced by "southern bears" in the Hudson Bay region is impacting the survival of cubs, because of insufficient ability of adult females to maintain adequate body mass.

Polar Basin Divergent Ice Ecoregion

Across the rest of their range, polar bears used to be able to remain on perennial ice (i.e., ice that survives the summer melt) year-round. In the Divergent Ice Ecoregion, which extends around the Arctic from coastal Alaska to Svalbard, ocean currents continually carry the ice offshore as it forms. This movement of ice "diverging from the shoreline" is especially noticeable in summer. As the weather warms, new ice stops forming and the remaining ice drifts toward the center of the polar basin, leaving a gap of ocean water between land and the polar ice pack. Historically, the summer sea-ice retreat was small and these bears were able to hunt on the ice over productive shallow water all summer, reaching peak body weights by fall. With warmer temperatures, the sea ice is retreating farther from shore and these bears are faced with a choice of coming ashore, where there is little to eat, or following the sea ice over the deep polar basin where biological productivity is low. Because seals can live a pelagic lifestyle, they don't need to follow the ice and instead remain nearer shore where productivity is high. Thus, polar bears remaining on the ice, like those that come ashore, are largely food deprived and end up fasting until autumn freeze-up. Five polar bear subpopulations live in divergent ice areas: Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, and the Southern Beaufort Sea. These subpopulations differ greatly in the productivity of their waters. The Chukchi Sea, for example, may be the most productive of arctic seas and includes a vast continental shelf area. The neighboring Beaufort Sea, in contrast, is among the least productive of arctic waters.

Although sea-ice retreat in both neighboring subpopulations has been great, the productivity of the Chukchi Sea and the broad area of continental shelf means that polar bears there are likely to persist through longer fasting periods than are those in the Beaufort Sea. Greater persistence, even with dramatic sea ice decline, also may apply to the highly productive Barents Sea. Despite productivity differences, long fasts for bears of the Divergent Ice Ecoregion which, unlike those in the Seasonal Ice Ecoregion, are accustomed to feeding through the summer, make them among the most vulnerable of all polar bears to climate warming and loss of sea-ice habitats.

Polar Basin Convergent Ice Ecoregion

In the Convergent Ice Ecoregion, ice transported from the Divergent Ice Ecoregion, along with locally formed sea ice, collects along the shore, or moves parallel to shorelines, on its way out of the polar basin. The collection of ice along coastlines provides polar bears with access to seals over productive waters throughout the summer, and presently these bears can remain on the sea ice all year. Two polar bear subpopulations live in these areas: East Greenland and Northern Beaufort Sea. Amstrup *et al.* (2008) included the Queen Elizabeth Islands as a subpopulation in the Convergent Ice Ecoregion, although the Queen Elizabeth Islands is not considered by the PBSG to be one of the 19 recognized polar bear subpopulations inhabiting the circumpolar Arctic.

Archipelago Ecoregion

The ocean channels separating the islands of the far north Canadian Arctic have historically been covered by sea ice all summer, and polar bears living there have been able to remain on ice year-round. This ecoregion, along with the northernmost portions of the Convergent Ice Ecoregion, is likely to provide a last refuge for polar bears and their prey. Six polar bear populations live in the Archipelago Ecoregion: Gulf of Boothia, Kane Basin, Lancaster Sound, M'Clintock Channel, Norwegian Bay, and Viscount Melville Sound.

Assessing Global Status

Studying polar bears is expensive and logistically difficult (Vongraven *et al.* 2012), and subpopulations differ in how much information is available to understand their statuses and trends. There is high uncertainty about the discreteness and status of several subpopulations, particularly in the Russian Arctic. For example, until 2005, the PBSG Status Table included estimates for three subpopulations (Chukchi Sea, Kara Sea, and Laptev Sea) where accepted scientific methods had not been applied. These estimates were removed in 2005 to avoid the incorrect impression that reliable data were available. Similarly, the PBSG has never provided estimates of abundance for the East Greenland and Arctic Basin subpopulations.

The PBSG recognizes that there is public interest in the abundance of the global polar bear population. The group provided its first global population estimate in 1993 of 21,470–28,370 polar bears (PBSG 1995). Although this was based on the best available scientific information, confidence in estimates of subpopulation size varied due to different research methods and sampling intensity. Some estimates were based solely on knowledge of habitat quality or expert judgment. Recognizing that combining subpopulation estimates that differ greatly in quality could lead to a false sense of precision, the PBSG rounded the range of global abundance to 22,000–27,000 in 1997 (PBSG 1998). Reflecting additional discussion and data, the global range was adjusted to 21,500–25,000 in 2001 (PBSG 2002) and 20,000–25,000 in 2005 and 2009 (PBSG 2006, 2010).

Although better information is now available for several subpopulations, some estimates remain missing, outdated, or include large uncertainty. The most recent estimate of global abundance is 26,000 (95% Confidence Interval [CI] = 22,000–31,000; Regehr *et al.* 2016). Like previous ranges, these numbers must be interpreted with caution because they reflect the status of polar bears as well as the amount and quality of scientific information, both of which can change over time. Adjustments to the reported global estimate will continue as new information becomes available.

Recent demographic and ecological models have evaluated the relative influence of environmental and anthropogenic stressors on polar bears (Atwood *et al.* 2016), projected changes in the global population based on alternate assumptions about sea-ice conditions and how polar bears will respond to climate warming (Regehr *et al.* 2016), and provided estimates

of when polar bears in different parts of their range will reach reproductive and survival thresholds that precipitate declining numbers (Molnár *et al.* 2020). High-resolution estimates (e.g., predictions of specific years in which survival thresholds might be crossed) are not possible due to the absence of detailed data (e.g., on body condition) for many subpopulations, uncertainty about future greenhouse gas emissions and mitigation strategies, uncertainty about the mechanisms through which habitat change affects polar bear population dynamics, and other factors. Lack of detailed data on abundance for some subpopulations and uncertainty in estimates of future polar bear subpopulation status do not detract from the fundamental relationship between polar bear persistence and availability of sea ice over productive waters.
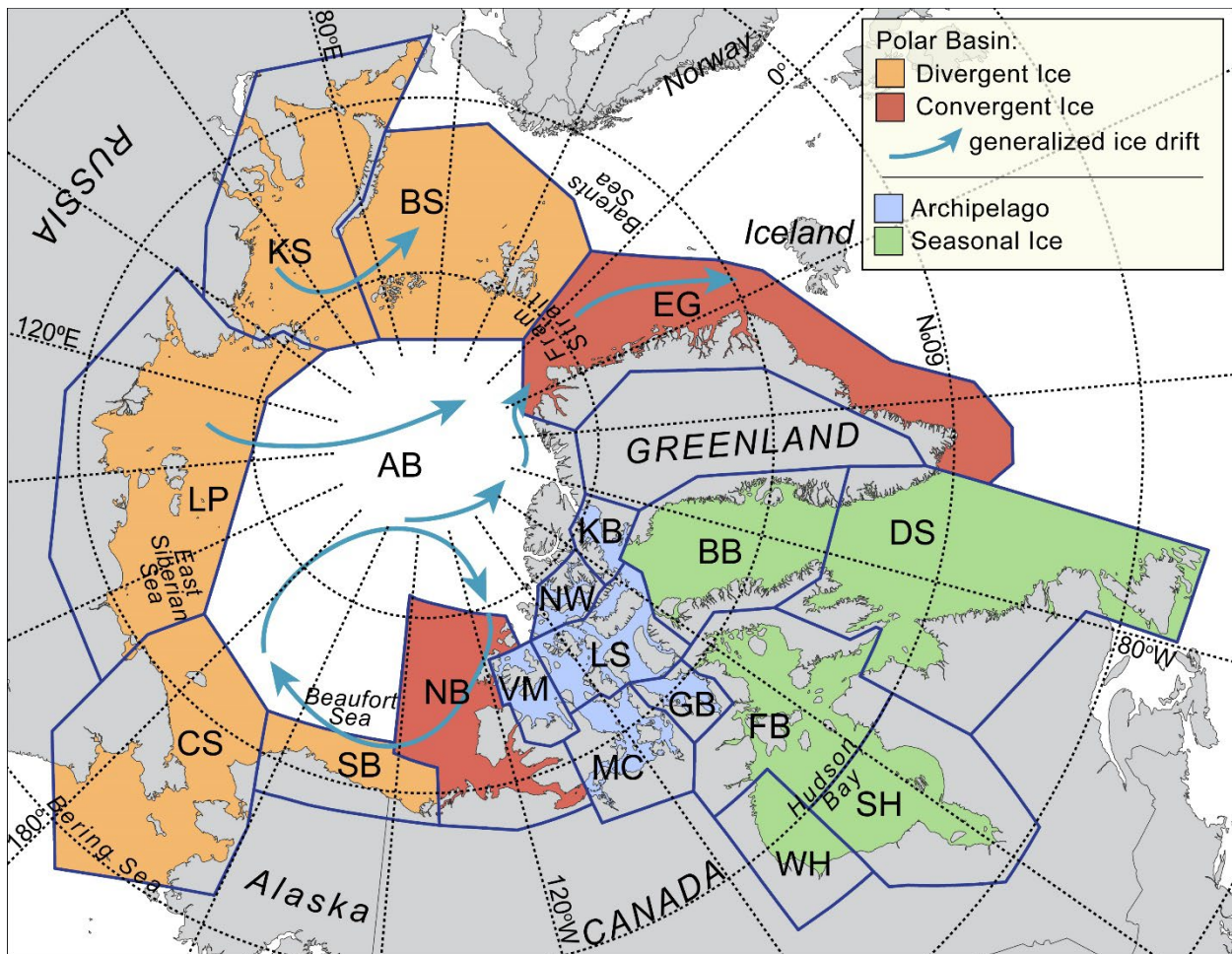
Figure 1. Map of four polar bear ecoregions defined by grouping recognized subpopulations which share seasonal patterns of ice motion (blue arrows) and distribution (excerpted and modified from Amstrup *et al.* 2008). The polar basin divergent ecoregion (PBDE) (orange) includes Southern Beaufort Sea (SB), Chukchi Sea (CS), Laptev Sea (LP), Kara Sea (KS), and the Barents Sea (BS). The polar basin convergent ecoregion (PBCE) (red) includes East Greenland (EG) and Northern Beaufort Sea (NB). The seasonal ice ecoregion (SIE) (green) includes southern Hudson Bay (SH), western Hudson Bay (WH), Foxe Basin (FB), Davis Strait (DS), and Baffin Bay (BB). The archipelago ecoregion (AE) (blue) includes Gulf of Boothia (GB), M'Clintock Channel (MC), Lancaster Sound (LS), Viscount-Melville Sound (VM), Norwegian Bay (NW), and Kane Basin (KB).

Status Table Structure and Definitions

Column primary heading: Subpopulation

Column secondary heading: BLANK

Example in table: Gulf of Boothia

Subpopulations are defined as geographically or otherwise distinct subgroups of the total global population between which evidence suggested that demographic and genetic exchange historically had been low, following the IUCN Red List Categories and Criteria (IUCN 2017). Polar bear subpopulations may not meet the IUCN criteria of no more than one successful migrant individual per year on average. Furthermore, subpopulations are characterized by different levels of demographic, ecological, and genetic distinctness (Amstrup *et al.* 2004; Thiemann *et al.* 2008). Subpopulations are used for demographic assessments, harvest monitoring, and other research and management activities. The IUCN/SSC PBSG recognizes 19 subpopulations (Durner *et al.* 2018). Modifications to subpopulation boundaries are assessed based on the best available scientific information.

Column primary heading: Subpopulation size

Column secondary heading: Estimate and uncertainty

Example in table: 1,525 (95% CI = 949–2,101)

Subpopulation size is the total number of individuals (i.e., including dependent young) that are alive and rely primarily upon habitats within a subpopulation boundary for a significant part of the year, and are not enumerated as part of a different subpopulation. Subpopulation size may be observed, estimated, or projected (definitions in IUCN 2017, pages 19–20). The scientific bases for reported subpopulation sizes are publicly available (e.g., as a governmental report or journal publication). Estimates of subpopulation size are typically associated with uncertainty due to sampling error, natural variability, and other sources. When possible, uncertainty is characterized by an interval estimate derived using quantitative methods (e.g., 95% Confidence Interval) that is listed in parentheses. If an observed, estimated, or projected value of subpopulation size is not publicly available, it may be listed as Data Deficient.

Column primary heading: Subpopulation size

Column secondary heading: Method and type of evidence

Example in table: Physical capture-recapture (estimated)

A brief description of the quantitative or qualitative method of assessment used to determine
subpopulation size. The type of evidence on which the estimate is based (definitions in IUCN
2017, pages 19–20) is listed in parentheses. Example entries include but are not limited to:
Total count (observed), Capture-recapture (estimated), Distance sampling (estimated), and
Matrix-based projection (projected).

Column primary heading: Subpopulation size

Column secondary heading: Most recent year of estimate, and citation

Example in table: 2,000 (Taylor *et al.* 2009)

The most recent year to which the value of subpopulation size applies. For example, if an
assessment provided an average estimate of subpopulation size for the period 1998–2000, the
year would be listed as 2000. A citation for the value of subpopulation size is listed in
parentheses.

Column primary heading: Change in subpopulation size

Column secondary heading: Long term (>2 polar bear generations)

Example in table: Likely stable (1996 to 2020)

Change in subpopulation size over a period of approximately three polar bear generations (34.5
years; Regehr *et al.* 2016), ending with the most recent estimate of subpopulation size. If
necessary, a shorter period may be used if it is at least two polar bear generations (23 years).
The range of years to which the trend is referenced is listed in parentheses. Change in
subpopulation size may be observed or estimated (as defined by IUCN 2017, pages 19-20)
based on comparison of two, or more, point estimates of abundance. Possible values for
change in subpopulation size are: Increased, Stable, Decreased, and Data Deficient. The

reported change reflects the best current understanding and interpretation but may or may not meet standard criteria for statistical significance. The listed value is preceded by a modifier reflecting the likelihood of the change, as determined statistically, by weight of evidence, or by expert judgment, using likelihood terms established by the IPCC (Table 1.2 in Cubasch *et al.* 2013). Subpopulations for which a biologically meaningful change cannot be assessed with a likelihood of "Likely" (i.e., 66-100% probability of being true) or higher, are listed as Data Deficient. If available, information on historic changes in subpopulation size (i.e., extending more than three polar bear generations into the past) can be included in the column Comments, vulnerabilities, and concerns.

Column primary heading: Subpopulation trend

Column secondary heading: Short term (approx. 1 polar bear generation)

Example in table: Likely stable (2011 to 2022)

Subpopulation trend over a period of approximately one polar bear generation (11.5 years; Regehr *et al.* 2016), ending with the most recent year for which trend has been assessed. If necessary, a shorter period may be used. The range of years to which the trend is referenced is listed in parentheses. Short-term trend may be observed, estimated, projected, or inferred (as defined by IUCN 2017, pages 19–20). Possible values for recent trend are: Increased, Stable, Decreased, and Data Deficient. The reported trend reflects the best current scientific understanding and interpretation but may or may not meet standard criteria for statistical significance. The listed value is preceded by a modifier reflecting the likelihood of the trend, as determined either statistically or by expert opinion, using likelihood terms established by the IPCC (Table 1.2 in Cubasch *et al.* 2013). Subpopulations for which a biologically meaningful trend cannot be assessed with a likelihood of "Likely" (i.e., 66–100% probability of being true) or higher, are listed as Data Deficient. The timeframe for short-term trend may extend into the future if appropriate population projection methods have been used (e.g., quantitative Population Viability Analysis).

Column primary heading: Sea-ice metrics (1979 to 2022)

Column secondary heading: Change in date of spring sea-ice retreat (days per decade) /

change in date of fall sea-ice advance (days per decade)

Example in table: -9.8 / +14.3

Subpopulation-specific change in date of spring sea-ice retreat (days per decade) and change in date of fall sea-ice advance (days per decade) over the period 1979–2022, calculated using the methods of Stern and Laidre (2016) and sea-ice data from DiGirolamo *et al.* (2022). Every year, the area of sea ice reaches a maximum in March and a minimum in September. To measure the timing of the seasonal change in sea ice, we find the date each spring when the area of sea ice has dropped to a specific threshold, and the date each fall when the area has grown back to that same threshold. The region-specific threshold is halfway (50%) between the mean March sea-ice area and the mean September sea-ice area, where the means are calculated over the 30-year reference period 1991–2020.

Column primary heading: Sea-ice metrics (1979 to 2022)

Column secondary heading: Change in summer sea-ice area (percent change per decade)

Example in table: -12.2

Subpopulation-specific change in the summer (01 June to 31 October) sea-ice area (percent change per decade) over the period 1979–2022. Percent change is calculated relative to the average summer sea-ice area during the period 1991–2020, using the methods of Stern and Laidre (2016) and sea-ice data from DiGirolamo *et al.* (2022).

Column primary heading: Human-caused removals (2016/17 to 2020/21)

Column secondary heading: 5-year mean potential (bears per year)

Example in table: 72.4

The mean annual potential removals (e.g., harvest quota) over the five-year period, as determined and authorized by the jurisdictions and agencies with management authority for the subpopulation. The five-year period may be based on calendar years or harvest years, which are typically defined as 01 July in year t to 30 June in year t + 1. For subpopulations that are not subject to a legal harvest, the value "N/A" is used.

Column primary heading: Human-caused removals (2016/17 to 2020/21)

Column secondary heading: 5-year mean actual (bears per year) and what this represents as a percentage of the total population

Example in table: 61.8 (3.9%)

The mean annual number of bears that were actually removed from the subpopulation, including all forms of direct human-caused mortality, over the five-year period. The five-year period may be based on calendar years or harvest years, which are typically defined as 01 July in year t to 30 June in year t + 1.

Column primary heading: Comments, vulnerabilities, and concerns

Column secondary heading: BLANK

Example in table: "A new subpopulation assessment is underway."

The status of each polar bear subpopulation is assessed using the best available scientific data including subpopulation abundance, trend, vital rates, movements and habitat use, human-caused removals, and changes in habitat availability. The *Comments, vulnerabilities and concerns* column provides additional background information on subpopulation assessments, bear health, population modelling and ongoing research programs that contribute to the current scientific knowledge of the status of each subpopulation. This column may be used to highlight potential vulnerabilities and concerns that exist for the subpopulation.

Polar Bear Subpopulations

Arctic Basin (AB)

The AB subpopulation is a geographic designation to account for polar bears occurring in the most northern areas of the circumpolar Arctic that are not clearly part of other subpopulations. Historically, polar bears have occurred at low densities here, in part because of deep, cold, stratified, and less biologically productive waters and, formerly at least, extensive coverage by multiyear ice. Polar bears from several subpopulations are known to have traveled through the area (Durner and Amstrup 1993), and although there are multiple reports of bears in this area, it is uncertain whether they are residents or merely passing through to other more productive regions of the Arctic. The ice and ecological characteristics in the AB region have changed with climate warming, but the impact on polar bears is unknown. As climate warming continues, it is anticipated that areas where some ice remains over the continental shelf may become important for polar bears as a refuge, although a large part of the AB region is over the deepest waters of the Arctic Ocean and biological productivity will likely remain low. Further reduction in sea ice cover may also reduce connectivity between the AB region and several subpopulations in future. Polar bears with cubs have been observed from icebreakers in this region (Ovsyanikov 2010) and Rode *et al.* (2015) reported telemetry data indicating that three bears from the Chukchi Sea subpopulation denned in the AB region, although the fates of these bears and their cubs is not known. Laidre et al (2020*a*) identified maternity dens in the multi-year ice based on surveys along the northern coast of Greenland in 2022. The total numbers of bears that occupy the AB region seasonally or year-round is unknown. The northernmost documented observation was made at 89°46.5'N, which is 25 km from the North Pole (van Meurs and Splettstoesser 2003).

Baffin Bay (BB)

Based on movements of adult females with satellite radiocollars and recaptures of tagged animals, the BB subpopulation is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island, Canada, to the west (Taylor and Lee 1995; Taylor *et al*. 2001;

Laidre *et al*. 2013). A distinct southern boundary at Cape Dyer on Baffin Island in Nunavut, Canada is evident from the movements of tagged bears (Stirling *et al*. 1980; Peacock *et al*. 2012) and from polar bears monitored by satellite telemetry (Taylor *et al*. 2001). This boundary overlaps with the northern boundary of the Davis Strait subpopulation. Studies of microsatellite genetic variation have not revealed significant differences between polar bears in BB and the neighboring Kane Basin subpopulation, although there was significant genetic variation between polar bears in BB and those in Davis Strait (Paetkau *et al*. 1999; Peacock *et al*. 2015; Malenfant *et al*. 2016; SWG 2016). However, polar bears in BB cluster with bears in northern Davis Strait (Peacock *et al*. 2015).

An initial subpopulation estimate of 300–600 bears in BB 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 1990s showed that an unknown proportion of the subpopulation was 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 thought to be ashore in summering 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). A three-year genetic mark-recapture survey (via biopsy darting) was completed in 2014 resulting in a new population estimate, survival rates, and habitat use analyses (SWG 2016). The mean estimate of total abundance of the BB subpopulation in 2012–2013 was 2,826 (95% CI = 2,284–3,367) polar bears (Atkinson *et al*. 2021). Due to statistical uncertainty and evidence that the sampling design and environmental conditions resulted in an underestimate of abundance in the 1990s, estimates from the 1990s and 2010s are not directly comparable and trend in abundance cannot be determined.

Satellite telemetry data and habitat selection studies in the 2000s identified ecological changes related to sea-ice loss in BB (SWG 2016; Laidre *et al*. 2020*b*). There has been a significant reduction in the range of the subpopulation in all months and seasons when compared to the 1990s. The most marked reduction is a 60% decline in the size of the subpopulation range in summer (Laidre *et al.* 2020b. Emigration from BB has declined since the 1990s, especially with a reduction of bears moving from BB into Davis Strait and Lancaster

Sound (Laidre *et al.* 2018a). The total number of bears marked during studies in 2011–2012 in BB was equivalent to approximately 34% of the estimated population size. Instances of emigration were ≤1% of the recaptures and recoveries of marks for the BB subpopulation.

Compared to the 1990s, adult female BB bears now use significantly lower sea-ice concentrations in winter and spring and spend 20–30 more days on land on Baffin Island in the ice-free season (Laidre *et al.* 2018a,b; Laidre et al. 2020c). Changes in maternity denning have been observed and entry dates into maternity dens are >1 month later in the 2000s than the 1990s. Furthermore, the first date of arrival on land by pregnant females is significantly earlier in the 2000s (Laidre *et al.* 2020b). Maternity dens in the 2000s occurred at higher elevations and steeper slopes than the 1990s, likely due to reduced snow cover (Escajeda *et al*. 2018).

Barents Sea (BS)

The size of the BS subpopulation was estimated to be 2,644 (95% CI = 1899–3592) in August 2004 using mark-recapture distance-sampling with data collected from aerial surveys (Aars *et al*. 2009). Earlier population estimates based on den counts and ship surveys (Larsen 1972) were higher. Ecological data supports that the BS subpopulation grew steadily during the first decade after hunting ceased in 1973, and then either continued to grow or stabilized. A new survey in the Norwegian extent of BS was conducted in August 2015 (Aars *et al.* 2017). During this survey, the ice edge was located beyond an ice-free gap north of the Svalbard Archipelago, thus local bears in Svalbard were separated from bears that occupied the areas along the sea ice. The number of bears encountered in Svalbard indicates that there is a local stock of approximately 200–300 bears, which did not differ much from the number detected in 2004. The results also indicate, in accordance with results from 2004, that most bears are offshore in the pack ice in autumn. The total estimate for the Norwegian Arctic (i.e., the Norwegian part of BS) was just under 1,000 bears, considerably higher than the total for the Norwegian side in 2004, but with a confidence interval overlapping the earlier estimate. During the 2015 survey, the distribution of bears was clumped along the ice edge with most bears close to the Russian border, but access to the Russian portion of BS for the aerial survey was not permitted, so no current reliable estimates exist for bear abundance in the Russian part of the BS subpopulation. Because of the overlapping confidence intervals, we cannot ascertain a trend in subpopulation

size. Recently, Popov and Davydova (2020) reported observations of polar bears from icebreakers in the Russian part of the BS region.

Subpopulation boundaries based on satellite telemetry data indicate that BS is a natural subpopulation unit, albeit with some overlap to the east with the Kara Sea (KS) subpopulation (Mauritzen *et al*. 2002). Overlap between BS and the East Greenland (EG) subpopulation may be limited (Born *et al*. 1997), although to some degree home ranges of bears from the EG overlap with those of bears from Svalbard in Fram Strait (Born *et al*. 2012). Genetically, polar bears from BS are similar to those in the EG, KS, and Laptev Sea (LP) subpopulations (Paetkau *et al*. 1999; Peacock *et al*. 2015). At a global level, polar bears in BS belong to the *Eastern Polar Basin* genetic cluster (one of four global genetic clusters); substantial directional gene flow occurs from the *Eastern Polar Basin* to the *Western Polar Basin* cluster (Peacock *et al*. 2015).

At a finer scale, there is evidence to support sub-structuring of polar bears within BS. 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). These data show that some bears associated with Svalbard are very restricted in their movements. Bears specifically from the Barents Sea, however, range widely between Svalbard and Franz Josef Land in the western Russian Arctic (i.e., a 'pelagic type;' Wiig 1995; Mauritzen *et al*. 2001; Blanchet *et al.* 2020). Within the BS subpopulation boundaries, substructure between local Svalbard bears and pelagic bears is likely increasing as sea ice around the islands disappears seasonally for longer durations. Fewer of the pelagic bears use maternity dens in the eastern part of Svalbard (Derocher *et al*. 2011; Aars 2013), in traditionally important denning areas, and it is likely that many of these bears now den more on Franz Josef Land. Some bears of the pelagic type from northern Svalbard move north to the Arctic Ocean in the summer, and return to northern Svalbard in the winter, whereas bears from southeast Svalbard follow retreating ice to the east (Blanchet *et al.* 2020). Capture-recapture data also show that movement between northwest and southeast Svalbard is rare (Lone *et al*. 2013, Brun *et al.* 2021).

Habitat loss for the BS subpopulation is projected to continue. Although pelagic bears in the BS have a slightly higher body condition than the local Svalbard bears (Blanchet *et al.* 2020),

consistent temporal trends in body condition (mainly coastal bears) have thus far been undetected (Lippold *et al.* 2019). In 2016, the Russian Federation expanded its Russian Arctic National Park with the inclusion of Franz Josef Land, which is an important summering area for polar bears.

Chukchi Sea (CS)

Studies in the late 1980s and early 1990s revealed that polar bears in the CS subpopulation (also known as the Alaska-Chukotka population, with slightly different boundaries) 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 was set at Icy Cape, Alaska, which is also the western boundary of the Southern Beaufort Sea (SB) subpopulation (Amstrup *et al*. 1986; Amstrup and DeMaster 1988; Garner *et al*. 1990; Amstrup *et al*. 2004, 2005). Support for the CS-SB boundary has been recently reaffirmed with an updated analysis of movement data (Scharf *et al.* 2019). However, similar movement data are not available to evaluate the western boundary between the CS and Laptev Sea (LS) subpopulations.

The first quantitative estimate of the size of the CS subpopulation (2,937 95% CI = 1,552–5,944) was obtained from capture-recapture research in the U.S. portion of the subpopulation's range, with density estimates extrapolated to other regions within the subpopulation boundary (Regehr *et al*. 2018*a*). An aerial survey in 2016 provided independent estimates of abundance that ranged from 3,435 (95% CI = 2,300–5,131) to 5,444 (95% CI = 3,636–8,152), where the range reflects uncertainty about how many bears were missed on the transect line in the Russian portion of the aerial survey (Conn et al. 2021). Subpopulation abundance was previously estimated to be between 2,000 and 5,000 animals based on the number of maternity dens observed on Wrangel and Herald islands and the Chukotkan coast, and the assumed proportion of females in the subpopulation (Belikov 1993). In recent years, sea ice has retreated farther north in the area occupied by the CS subpopulation resulting in more days in which the biologically productive waters of the continental shelf are ice free (Durner *et al*.

2009; Rode *et al*. 2014). Sea-ice loss is expected to continue (Wang *et al.* 2018). Rode *et al*. (2014, 2021) documented stable or improving body condition and reproduction for polar bears captured in the U.S. portion of the CS between 1986–1994 and 2008–2011, a period over which substantial sea-ice loss occurred. This suggests some resiliency of the CS subpopulation to summer habitat loss, likely associated with high biological productivity of the region. Autumn-based observations on Wrangel Island for the period 2004-2010, however, may indicate declining cub production and maternity denning (Ovsyanikov 2012). Systematic ground-based surveys on Wrangel Island were started in 2016 and continue annually, providing information on the critical importance of Wrangel Island as a denning and resting area for polar bears and helping to monitor the ecological and demographic status of the CS subpopulation (E. Regehr, *unpublished data*).

A quantitative harvest risk assessment has been completed using new estimates of abundance, vital rates, and human-caused removals (Regehr *et al*. 2018*b*, 2021*b*). Estimates of illegal take of polar bears in Russia are based on village interviews conducted 2010–2012. The current take level in Russia appears to be significantly lower than in the 1990s although up-to-date and accurate information is lacking (Kochnev and Zdor 2016). Uncertainty in previous estimates of abundance and other subpopulation parameters (e.g., levels of human-caused removals) result in a designation of "Data deficient" for the long-term change in subpopulation size. The designation of "Likely stable" for short-term subpopulation trend is based on estimates of population growth rate using vital rates for the period 2008–2016 (Regehr *et al*. 2018*b*).

New studies have found that CS polar bears have increased land use during the summer, primarily on Wrangel Island and the Chukotkan peninsula in Russia (Rode *et al*. 2015). Further, Wilson *et al*. (2014, 2016) found that habitat selection preferences of polar bears on the sea ice in the CS region have not changed over time despite declines in the availability of their preferred habitats. This suggests that CS bears are not changing habitat use, and that climate warming will continue to reduce the availability of preferred sea-ice habitat. The observed relationship between summer sea-ice availability and the duration of time and proportion of the CS subpopulation that comes to shore suggests that land use will increase as sea-ice loss

continues.

Davis Strait (DS)

Based on the recapture or harvest of previously tagged animals, and tracking data from adult female polar bears with satellite collars, the DS subpopulation occurs in Canada within the Labrador Sea, eastern Hudson Strait, Davis Strait south of Cape Dyer, and along a portion of southwest Greenland (Stirling and Kiliaan 1980; Stirling *et al*. 1980; Taylor and Lee 1995; Taylor *et al*. 2001). A genetic study of polar bears (Paetkau *et al*. 1999) indicated significant differences between bears from southern DS and both the Baffin Bay and Foxe Basin (FB) subpopulations. Crompton *et al*. (2008, 2014) found that individuals from northern portions of DS and those from FB share a high degree of ancestry. Peacock *et al*. (2015) used samples from both northern and southern DS in an updated circumpolar genetic analysis and found that the two regions belong to two different global genetic clusters (i.e., southern DS to the *Southern Canada* cluster and northern DS to the *Canadian Archipelago* cluster).

The initial subpopulation estimate of 900 bears for DS (Stirling and Kiliaan 1980; Stirling *et al*. 1980) was based on a subjective correction from the original mark-recapture estimate of 726 bears, which was thought to be too low because of possible bias in the sampling. In 1993, the estimate was subjectively increased to 1,400 bears and again to 1,650 in 2005. These increases were to account for bias resulting from springtime sampling, the fact that the existing harvest appeared to be sustainable and not have negative effects on the age structure, and traditional ecological knowledge (TEK) that suggested that more bears were being seen in the preceding 20 years. In addition, harp seals (*Pagophilus groenlandicus*), an important prey species for the subpopulation, had increased dramatically (due to reduction in their overharvest) over the same period, providing an enhanced potential prey base. Polar bears were seen and radio-tracked in the large pupping areas off the coast of southern Labrador in spring. A second inventory of DS was conducted from 2005 - 2007 using mark-recapture methods and the subpopulation estimate was 2,158 (95% CI = 1,833–2,542; Peacock *et al*. 2013). Polar bear survival in DS varied with time and geography (i.e., northern vs. southern DS) and was related to factors that included reductions in sea-ice habitat and increases of harp seal numbers (Peacock *et al*. 2013). It was suggested that the observed lowered reproductive rates and

declines in body condition of polar bears in DS were likely a result of habitat changes, changes in polar bear density, or both (Rode *et al*. 2012; Peacock *et al*. 2013).

During 2017 and 2018, DS was inventoried again using genetic biopsy capture-mark-recapture methodology. This most recent analysis capitalized on the large number of marked individuals, high recapture probabilities, number of dead-recoveries, and relatively short time period between surveys (9 years) to use a Bayesian approach to model temporal random effects to estimate a population level mean and annual survival rates for adult and subadult bears. Using these methods, estimated abundance for 2017 -2018 was 2,015 bears [SD] = 251; 95% Bayesian Credible Interval [CRI] 1,603 – 2,588) and 2,250 for 2006 -2007 [SD = 133; CRI 1,989 – 2,512] (Dyck *et al*., 2022). Mean number of cubs-of-the-year per adult female ranged from 0.23 to 0.45 and mean number of yearlings per adult female ranged from 0.23 to 0.41. Survival did not differ between cubs-of-the-year (COY) and yearlings or between sexes for any age class. Mean survival rates were 0.794 (95% CRI; 0.723, 0.861) for dependent young, 0.873 (95% CRI; 0.826, 0.914) for independent subadults, and 0.871 (95% CRI; 0.853, 0.892) for adults, which are lower than previously reported in Peacock *et al*. (2013) but fall within their confidence intervals. The geometric mean population growth rate between 2006 and 2018 indicates the subpopulation most likely declined (mean = 0.99; 95% CRI 0.97-1.01).

There was no support for a temporal trend or an effect of environmental factors on survival of subadult or adult bears (Dyck *et al*., 2022; Dunham *et al*., under review). In contrast to Peacock *et al*. (2013), no relationship between harp seal abundance or summer sea-ice concentration and survival over 13 years (2005-2018) was detectable, potentially because Peacock *et al*. (2013) analyzed data between 1974 and 2009, over which time harp seal abundance substantially increased (Hammill *et al*., 2021). Harp seal abundance also increased between 2005 and 2018 and numbers remained consistently high (> 5 million seals) (Hammill *et al*., 2021). While DS polar bear survival increased with harp seal abundance in previous decades during larges changes in the availability of this prey resource, it likely had less influence on polar bear survival once seals were abundant and no longer limiting. Peacock *et al*. (2013) identified support for a positive effect of mean summer sea-ice concentration on survival when the concentration was between 17% and 29%, however, mean summer sea-ice concentration did

not exceed 8% within the latest 2017-2018 study period.

Harvest in Davis Strait is shared among multiple jurisdictions; some of which have near 100% reporting and others which have no reporting requirement. Harvest appears correlated with hide prices (Cooper et al., 2022; Tomaselli et al., 2022) with high demand for hides driving increased hunting. The average reported harvest between 2005/06 and 2010/11 was 66 bears/year. This nearly doubled during times of high demand for hides to an average of 112 bears/year. In 2014, hide prices crashed and the average harvest per year from 2015/16 - 2017/18 was 63 bears. During this same period, in 2012/2013, based on the 2005-2007 abundance estimate and traditional knowledge, the jurisdiction of Nunavut implemented a management objective to reduce the subpopulation slightly by raising the allowable take. It is unclear if the short-term elevations in hunting preceding the last survey affected population abundance estimates for 2017/18.

East Greenland (EG)

Satellite telemetry data show that polar bears range widely in the annual and multi-year fast ice on the coast of northern and central east Greenland and on the pack ice in the Greenland Sea and Fram Strait (Born *et al*. 1997, 2009; Wiig *et al*. 2003; Laidre *et al*. 2013, 2015, Laidre *et al.* 2020*a*). Their range extends along the entire coastline as far as South Greenland. The EG subpopulation is thought to have limited exchange with other subpopulations (Born 1995; Wiig 1995; Dietz *et al*. 2000; Sandell *et al*. 2001; Born *et al*. 2009). There is little evidence of genetic difference between subpopulations in the northeastern Greenland and Svalbard-Franz Josef Land regions (Paetkau *et al*. 1999), and satellite telemetry and movement of marked animals have detected minimal exchange between polar bears in EG and the Barents Sea subpopulation (Wiig 1995; Born *et al*. 1997, 2009; Wiig *et al*. 2003; Laidre *et al*. 2013). The polar bears sampled from north and central in EG cluster with the *Eastern Polar Basin* genetic cluster, one of four global genetic clusters of polar bears (Peacock *et al*. 2015). Laidre *et al*. (2015) showed that due to multi-decadal sea-ice loss within EG, there have been changes in bears' habitat use in north and central east between the 1990s and 2000s. Adult females tracked in this area in the 2000s used areas with significantly lower sea-ice concentrations (10–15% lower) than adult females in

the 1990s during winter. They were also located significantly closer (100–150 km) to open water in all seasons and spent approximately two months longer in areas with <60% sea-ice concentration, compared to bears in the 1990s.

The first comprehensive scientific studies of polar bears in Southeast Greenland were conducted between 2015-2022 (Laidre *et al.* 2022). This work identified a previously undocumented and isolated group of polar bears living in Southeast Greenland between 60-64°N latitude. Southeast Greenland polar bears are the most genetically distinct polar bears in the Arctic, distinct from all other 19 subpopulations including bears in Northeast Greenland (north of 64°N latitude). Southeast Greenland polar bears are specially adapted to the physical geography, residing solely in the fjords in the area. The length of the sea ice season is short in Southeast Greenland and the bears partially use freshwater glacial mélange from the Greenland Ice Sheet as a platform to hunt seals (Laidre *et al.* 2022). In 2023, the Greenland Government issued an executive order determining that Southeast Greenland is a new and separate management unit and it is currently protected. An aerial survey was conducted in the inshore and offshore areas of north and central East Greenland in spring 2023 and an abundance estimate for the area is in progress.

Foxe Basin (FB)

Based on decades of mark-recapture studies and satellite tracking of female bears in Western Hudson Bay (WH) and Southern Hudson Bay (SH), the FB subpopulation appears to occur in Foxe Basin, northern Hudson Bay, and the western end of Hudson Strait (Taylor and Lee 1995; Sahanatien *et al*. 2015). The most recent mapping of satellite telemetry data indicates substantial overlap of the FB subpopulation with the WH and SH subpopulations and, to a lesser extent, with the Davis Strait subpopulation (Peacock *et al*. 2010; Sahanatien *et al*. 2015). During the ice-free season, polar bears are concentrated on Southampton Island and along the Wager Bay coast; however, significant numbers of bears are also encountered on the islands and coastal regions throughout the FB region (Stapleton *et al*. 2016). A total subpopulation estimate of 2,197 bears (standard error [SE] = 260) for 1994 was developed (Taylor *et al*. 2006*a*) from a mark-recapture analysis based on tetracycline biomarkers, where the marking effort was

conducted during the ice-free season and distributed throughout the entire area. Traditional ecological knowledge suggests that the FB subpopulation had increased (Government of Nunavut consultations in communities in Foxe Basin, 2004–2012). During a comprehensive summertime aerial survey in 2009 and 2010, based on distance sampling and double-observer estimation and covering about 40,000 km each year, 816 and 1,003 bears were observed, respectively (Stapleton *et al*. 2016). This most recent study yielded an abundance estimate of 2,585 (95% CI = 2,096–3,189) polar bears (Stapleton *et al*. 2016), which is not statistically different from the 1994 estimate, suggesting a stable population. Sea-ice habitat for polar bears has decreased substantially over the last several decades in FB (Sahanatien and Derocher 2012; Stern and Laidre 2016).

Gulf of Boothia (GB)

The boundaries of the GB subpopulation are based on genetic studies (Paetkau *et al*. 1999; Campagna *et al*. 2013; Peacock *et al*. 2015; Malenfant *et al*. 2016), movements of tagged bears (Furnell and Schweinsburg 1984; Taylor and Lee 1995), radiotelemetry in GB and adjacent areas (Taylor *et al*. 2001), and interpretations by Inuit hunters of how local conditions influence the movements of polar bears in the area. GB belongs in the *Canadian Archipelago* global genetic cluster (Peacock *et al*. 2015). 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 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 traditional ecological knowledge, 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 1,592 (SE = 361 bears; Taylor *et al*. 2009). Natural survival and recruitment rates were estimated at values higher than previous estimates (Taylor *et al*. 1987). Taylor *et al*. (2009) concluded that the subpopulation was increasing in 2000 because of a high intrinsic rate of growth and low harvest. Harvest rates were increased in 2005 based on the 2000 abundance estimate, and the

subpopulation was believed to be stable.

A three-year genetic mark-recapture study was conducted between 2015 and 2017. Results of live-capture dead-recovery models suggest a mean abundance estimate of 1,525 bears (SE = 294) for the period 2015–2017, which was similar to the previous mean abundance estimate during the period 1998–2000 (Dyck *et al.* 2020*b*). Mean cub-of-the-year and yearling litter sizes for the period 2015–2017 were 1.61 (95% CI = 1.51–1.70) and 1.53 (95% CI = 1.41–1.64), respectively, with no apparent trend compared to 1998–2000. The mean number of yearlings per adult female for the period 2015–2017 was 0.36 (95% CI = 0.26–0.47) which suggests that GB is currently a productive polar bear subpopulation despite sea-ice changes. This is consistent with the finding that polar bear body condition (i.e., fatness) in the spring improved between the periods 1998–2000 and 2015–2017. Sex- and age-specific variation in total survival rate (i.e., including harvest mortality) were detected with higher estimates for adult females (0.95; 95% CI = 0.81–0.99) than adult males (0.85; 95% CI = 0.74–0.92) for the period 2005–2017. A potentially related effect was detected as an increase in the proportional abundance of females from 0.57 in 1998–2000 to 0.61 in 2015–2017. The asymptotic, intrinsic population growth rate calculated using a matrix projection model with estimates of total survival was 0.06 (95% CI = -0.06–0.12) for the period between 2005 and 2017, suggesting strong potential for growth. However, the results for subpopulation size and trend should be interpreted with caution because the estimate of abundance reflects the "superpopulation" (i.e., it includes all bears that use the GB management area, some of which spend time in other subpopulations as well) and the estimate of population growth rate does not account for permanent emigration from the GB management area. Spatial and temporal changes in sea-ice extent and composition may have provided transient benefits to the GB subpopulation due to increased biological productivity associated with decreased sea-ice thickness and multiyear ice.

Kane Basin (KB)

Based on the movements of adult females with satellite collars and recaptures of tagged animals, boundaries of the KB subpopulation include the North Water Polynya to the south, the Kennedy Channel to the north, and Greenland and Ellesmere Island to the east and west (Taylor *et al*. 2001). Polar bears in KB do not differ genetically from those in Baffin Bay (Paetkau *et al*.

1999; Peacock *et al*. 2015). The size of the subpopulation was estimated to be 164 bears (SE = 35) for the period 1994–1997 by Taylor *et al*. (2008*a*). The intrinsic natural rate of growth for KB polar bears was estimated to be low at 1.009 (SE = 0.010; Taylor *et al*. 2008*a*), likely because of large expanses of multiyear ice and a low density of seals (Born *et al*. 2004). A genetic mark-recapture survey (via biopsy darting) and aerial survey were completed in 2014 resulting in a new subpopulation estimate, survival rates, and habitat use analyses (SWG 2016, Laidre *et al.* 2023). Using genetic mark-recapture, the estimated abundance of the KB subpopulation was 357 polar bears (95% CI = 221–493) for 2013–2014 (Laidre *et al.* 2023). More bears were documented in the eastern regions of the KB subpopulation during 2012–2014 than during 1994–1997. The difference in distribution between the 1990s and 2010s may reflect differences in spatial distribution of bears, possibly influenced by reduced hunting pressure by Greenland in eastern KB, but may also be due to differences in sampling protocols. An estimate of abundance based on a springtime 2014 aerial survey in KB was 190 bears (95% lognormal CI = 87–411; Wiig *et al.* 2022). However, due to insufficient coverage of offshore polar bear habitat, this estimate is likely negatively biased. The total number of bears marked during studies in 2012–2013 in KB was equivalent to approximately 25% of the estimated subpopulation size. Documented cases of emigration comprised <4% of recaptures and recoveries in KB.

Changing sea-ice conditions have resulted in broad movement and habitat-use patterns of KB bears that are similar to those of bears in seasonal sea-ice ecoregions (SWG 2016; Laidre *et al*. 2020*c*). The area is transitioning from multi-year ice to annual sea ice. The size of the subpopulation range has expanded since the 1990s in all seasons, especially in summer (June-September) when the subpopulation range doubled between the 1990s and the 2000s (Laidre *et al.* 2020c). Land use in KB during summer is intermittent because some sea ice remains inside fjords and coastal areas. Reproductive metrics for KB were comparable between the 1990s and 2010s sampling periods. Body condition in KB appeared to have slightly improved between sampling periods (see SWG 2016). Overall, the available data on abundance together with information on movements, body condition, and reproduction, suggest that the KB subpopulation has increased in size.

Kara Sea (KS)

The KS subpopulation overlaps in the west with the Barents Sea (BS) subpopulation in the area to the east of Franz Josef Land and includes the Novaya Zemlya archipelago. Data for KS and BS in the vicinity of Franz Josef Land and Novaya Zemlya are mainly based on dated aerial surveys and den counts (Parovshivkov 1965; Belikov and Matveev 1983; Uspenski 1989; Belikov and Gorbunov 1991; Belikov *et al*. 1991; Belikov 1993). 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). Using samples from the 1990s, at a global level, polar bears in KS belong to the *Eastern Polar Basin* genetic cluster (together with polar bears from the BS and Laptev Sea subpopulations); gene flow suggests substantial directionality (29-fold difference) from the *Eastern Polar Basin* cluster to the *Western Polar Basin* cluster (Peacock *et al*. 2015).

Lancaster Sound (LS)

Information on the movements of adult female polar bears monitored by satellite radiocollars and mark-recapture data has shown that the LS subpopulation is distinct from the adjoining Viscount Melville Sound, M'Clintock Channel, Gulf of Boothia, Baffin Bay, and Norwegian Bay (NW) subpopulations (Taylor *et al*. 2001). Survival rates of the pooled LS and NW subpopulations were used in a population viability analysis to minimize sampling errors, and the subpopulation estimate of 2,541 bears (SE = 391) is based on an analysis of mark-recapture data through 1997 (Taylor *et al*. 2008*b*). This abundance estimate is considerably larger than a previous estimate of 1,675 that included NW (Stirling *et al*. 1984). Taylor *et al*. (2008*b*) estimated survival and recruitment parameters that suggest the LS subpopulation has a lower renewal rate than previously estimated. However, what effect this may have on the status of the subpopulation is unknown, especially under changing sea-ice conditions. The available data for LS are dated, but the subpopulation is thought to be stable based on local traditional information. A comprehensive aerial abundance survey was conducted in spring 2023 and data analysis is currently underway.

Laptev Sea (LP)

The LP 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 Severnaya Zemlya in 1982 (Belikov and Randla 1987) and on anecdotal data collected in 1960s through 1980s on the number of females coming to dens on Novosibirsk Islands and the mainland coast (Kistchinski 1969; Uspenski 1989). At present, the subpopulation size is unknown.

M'Clintock Channel (MC)

The current boundaries for the MC subpopulation are based on recoveries of tagged bears, movements of adult females with satellite radiocollars in adjacent areas (Taylor and Lee 1995; Taylor *et al*. 2001), and genetics (Paetkau *et al*. 1999; Campagna *et al*. 2013; Peacock *et al*. 2015; Malenfant *et al*. 2016). These boundaries appear to be a consequence of large islands to the east and west, the mainland to the south, and the multiyear 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 bears (SE = 59; Taylor *et al*. 2006*b*). Natural survival and recruitment rates were estimated at values lower than previous standardized estimates (Taylor *et al*. 1987). Due to apparent declines in subpopulation abundance, and after an initial harvest moratorium, harvest levels for MC were reduced to levels that were expected to allow the subpopulation to recover and increase.

A three-year genetic mark-recapture study was conducted 2014–2016. Results of a closed capture-recapture model, implemented in a Bayesian framework and fitted to data for independent animals (i.e., >2 years), suggest a mean abundance of 716 bears (95% Credible Interval [CRI] = 545–955) for the period 2014–2016, indicating that the MC polar bear subpopulation increased since 1998–2000 (Dyck *et al.* 2020*a*) and likely demonstrating the effectiveness of local management. Both male and female segments of the subpopulation increased between study periods (i.e., 1998–2000 and 2014–2016), likely because of a combination of reduced harvest and improved habitat quality. Estimated apparent survival for bears aged 2 and older was 0.88 (SE = 0.02), although this is likely negatively biased due to

temporary or permanent movement of individual bears with respect to the study area and limited data on immigration and emigration. These scientific findings align with local observations that the subpopulation has recovered since the year 2000.

As with habitat in Gulf of Boothia, Barber and Iacozza (2004) found no trends in ringed seal (*Pusa hispida*) habitat or sea-ice conditions from 1980 to 2000 for MC. A general trend has been detected for earlier break-up and delayed freeze-up (Markus *et al*. 2009; Stern and Laidre 2016), but multiyear ice is predicted to persist into the near future (Howell *et al*. 2008; Sou and Flato 2009; Maslanik *et al*. 2011). Habitat quality for the MC subpopulation could be improved over the short-term as multiyear ice declines.

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 from the early 1970s to the present (e.g., Stirling *et al*. 1975; DeMaster *et al*. 1980; Stirling *et al*. 1988; Lunn *et al*. 1995; Stirling *et al*. 2011). From these studies, it became clear that there were separate polar bear subpopulations in the Northern and Southern Beaufort seas (i.e., the NB and SB subpopulations, respectively; Stirling *et al*. 1988; Amstrup *et al*. 1995; Taylor and Lee 1995). The density of polar bears using the offshore multiyear ice that predominated in much of the northernmost area of NB was lower than in the south where most polar bear habitat consisted of annual ice over the biologically productive continental shelf (Lunn *et al* 1995; Stirling *et al*. 2011). Abundance of the NB subpopulation in 1988 was estimated to be 1,200 (Stirling *et al*. 1988) although parts of the northwestern coast of Banks Island and M'Clure Strait were under-sampled because of local concern about disruption to guided polar bear sport hunters. The most recent mark-recapture assessment, completed in 2006, estimated the NB subpopulation to be 980 bears (SE = 155; Stirling *et al* 2011). The authors stated that, due to potential bias in the abundance estimate from 2006, the 2004–2005 estimates of approximately 1,200–1,300 bears were likely more reflective of the true abundance. The apparent stability of the NB subpopulation at the time was attributed to relatively stable ice conditions through 2006 and a harvest that was within sustainable limits (Stirling *et al*. 2011).

After the 2006 population reassessment and based on data from satellite tracking of female polar bears and spatial modeling techniques (Amstrup *et al*. 2005), the boundary between the NB and SB subpopulations was moved west to 133° W (at Tuktoyaktuk) from its previous eastern limit at Pearce Point (122° W). The revised boundary has been accepted and used by the responsible management authorities since 2014. The PBSG has adopted use of the revised boundary between the SB and NB, and it is used for reporting sea-ice conditions and harvest in the current Status Table. The Status Table will continue to report estimates of subpopulation size reflecting the previous boundary (e.g., Stirling *et al.* 2011; Bromaghin *et al.* 2015) until updated estimates pertaining to the revised boundary are available. Griswold *et al.* (2017) conducted an exploratory analysis to estimate NB and SB subpopulation abundance under the boundary at 133°W. Results indicated a mean of 311 bears should shift from SB to NB following the change in boundary. For management purposes, an abundance estimate of 1,710 bears is used, which reflects the boundary change and an adjustment based on potential negative bias due to lack of coverage of the entire area (ISR Management Plan – Joint Secretariat 2017). Since the early 2000s, sea-ice conditions in the eastern Beaufort Sea have declined (Stroeve *et al*. 2014; Stern and Laidre 2016) and have resulted in declines in body condition of adult ringed seals in the eastern Amundsen Gulf (Harwood *et al.* 2020) with predicted long-term declines in ringed seal pup survival in response to declining snow and sea-ice conditions (Reimer *et al.* 2019). Up-to-date scientific data are not available on how declines in the quality and availability of polar bear prey and habitat may be impacting the NB subpopulation. Harvest pressure in NB is low with levels consistently below a 4.5% total harvest rate, with ice conditions making travel for hunters difficult in some years (Stirling *et al*. 2011). A new demographic assessment of the NB subpopulation is currently underway.

Norwegian Bay (NW)

The NW subpopulation appears to be genetically unique (Malenfant *et al*. 2016). This subpopulation is bounded by heavy multiyear 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*. 2008*b*). Data collected during mark-recapture studies, and from satellite radiotracking of adult female polar

bears, suggest that most of the polar bears in the NW subpopulation are concentrated along the coastal tide cracks and ridges along the northern, eastern, and southern boundaries (Taylor *et al*. 2001). The most current (1993–1997) estimate of abundance is 203 bears (SE = 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 was too small to generate reliable survival estimates. The five-year mean harvest (0.8 bears per year between the harvest years 2016/17 and 2020/21) has been below the sustainable harvest level for the estimated subpopulation size. The available data for the NW subpopulation are dated.

Southern Beaufort Sea (SB)

Radiotelemetry and mark-recapture studies through the 1980s indicated that polar bears in the SB region comprised a single subpopulation, with an eastern boundary between Paulatuk and Baillie Island, Northwest Territories (NWT), Canada, and a western boundary near Icy Cape, Alaska, U.S. (Amstrup *et al*. 1986; Amstrup and DeMaster 1988; Stirling *et al*. 1988). More recently, data from female polar bears carrying satellite radiocollars suggested that at Utqiaġvik (formerly known as Barrow), Alaska, in the west, 50% of polar bears were from the SB subpopulation and 50% were from the adjacent Chukchi Sea (CS) subpopulation. Similarly, at Tuktoyaktuk, NWT, to the east, there was a 50% probability of polar bears being either from the SB or the adjacent Northern Beaufort Sea (NB) subpopulation (Amstrup *et al*. 2005). Based on these satellite tracking data and spatial modeling techniques (Amstrup *et al*. 2005), and recognizing changing patterns of breakup and freeze-up resulting from climate warming as well as traditional ecological knowledge (Joint Secretariat 2015), the original boundary between the NB and SB subpopulations was moved west from its previous eastern limit at Pearce Point (122° W). The revised boundary occurs at 133°W longitude and has been used by management authorities in the U.S., NWT, and Yukon since 2014.

The PBSG has adopted use of the revised boundary between the SB and NB, and it is used for reporting sea-ice conditions and harvest in the current Status Table. The Status Table will continue to report estimates of subpopulation size reflecting the previous boundary (e.g.,

Stirling *et al.* 2011; Bromaghin *et al.* 2015) until updated estimates pertaining to the revised boundary are available. Information that applies to the previous boundary is identified as such in the section "Comments, vulnerabilities, and concerns" of the Status Table. More recently, a boundary re-analysis using satellite tracking data was completed for the SB subpopulation where it borders the CS subpopulation, confirming support for the CS-SB boundary near Icy Cape, Alaska (Scharf *et al.* 2019).

The abundance of the SB subpopulation (using the previous SB-NB boundary at 122° W) was estimated to be approximately 1,800 (mean) animals from 1972–1983 (Amstrup *et al.* 1986). A subsequent analysis of capture data from 1967–1998 suggested that abundance had potentially reached 2,500 animals during the latter portion of this period (Amstrup *et al.* 2001). However, the estimate of 2,500 animals was not considered reliable due to concerns over heterogeneity within the data in the late 1990s, and managers were encouraged to continue using the abundance estimate of 1,800 individuals (Regehr *et al.* 2006). Results from a mark-recapture study conducted from 2001–2006 in both the U.S. and Canada indicated that the SB subpopulation included 1,526 (95% CI = 1,211–1,841) polar bears in 2006 (Regehr *et al*. 2006). That study and others found that the body condition, survival, and breeding success of polar bears were negatively affected by changing sea-ice conditions, and that population growth rate was negative in years with long ice-free seasons, such as 2005 when arctic sea ice extent reached a (then) record low (Hunter *et al*. 2010; Regehr *et al*. 2010, Rode *et al*. 2010). In 2009, Griswold *et al*. (2017) reanalyzed capture-recapture data for the SB subpopulation from 2001–2006 that were originally presented in Regehr *et al*. (2006) and determined that moving the boundary to 133°W longitude would decrease the estimated size of the SB subpopulation by approximately 311 polar bears. As a result, an estimated abundance of 1,215 polar bears in the SB subpopulation has been used in Canada for management purposes since the boundary shift in 2014.

Bromaghin *et al.* (2015) modeled survival and abundance (using the previous SB-NB boundary at 122° W) using data collected in the U.S. and Canada from 2001–2010 and found that survival estimates remained low through 2007 and increased through 2009, resulting in an abundance estimate of approximately 900 (90% CI = 606–1,212) polar bears in 2010. However,

they recognized the potential for un-modeled heterogeneity in mark-recapture data resulting from differences in sampling protocols over time, which could have biased both survival and abundance estimates. For example, the authors suspected negative bias in abundance estimates for the early years of the study associated with the distribution of search effort. They also noted the potential for negative bias in later years and concluded that the decline in abundance for the SB subpopulation was likely greater than 25% and may have approached 50% (see Bromaghin *et al*. 2015 for details). Although the authors expressed the need for caution in interpreting the estimated decline in abundance, other demographic evidence was consistent with a decline including a low proportion of yearlings in the subpopulation and failure of subadult survival rates to rebound in the latter years of the study. A new analysis of mark-recapture data collected in Alaska from 2001–2016, using a modeling approach that differed from Regehr *et al*. (2006) and Bromaghin *et al*. (2015), corroborated the decline in abundance in the Alaskan portion of the subpopulation from 2003–2006 and suggested stabilization through 2015 (Atwood *et al.* 2020). A Traditional Knowledge study from Canada concluded that the numbers of polar bears in regularly used hunting areas have remained relatively stable within living memory (Joint Secretariat 2015). A multiyear (2019-2023) biopsy-based mark-recapture abundance study was recently completed and analyses are underway.

Stirling *et al.* (2008) observed low hunting success of polar bears during the spring in 2003–2006, likely due to unusual sea-ice conditions and exacerbated by reproductive failure of ringed seals in 2005 (Harwood *et al*. 2020). Assessments of temporal patterns of feeding ecology found that the number of bears in a physiological fasting state in April and May increased from the mid-1980s to the mid-2010s (Cherry *et al*. 2009; Rode *et al*. 2018). These studies support the hypothesis that the energy balance of polar bears has changed in the SB (e.g., Pagano *et al.* 2020), which may explain declines in survival observed in the mid-2000s. The availability of sea-ice habitat for polar bears (Durner *et al*. 2009, 2019) is reduced with declining sea-ice extent (Stroeve *et al*. 2014), resulting from the continuing effects of climate warming. Atwood *et al*. (2016) found that polar bears in the SB are spending significantly more time on land, which is correlated with the extent of ice retreat. Further, while on land, many polar bears feed on the subsistence-harvested bowhead whale remains aggregated at Cross Island near the

Prudhoe Bay industrial infrastructure and Barter Island near the community of Kaktovik, Alaska (Herreman and Peacock 2013, Rogers *et al.* 2015). Increased polar bear activity near human settlements may increase exposure to terrestrial-based pathogens (Atwood *et al.* 2017) and the risk of human-bear interactions.

Southern Hudson Bay (SH)

Boundaries of the SH polar bear subpopulation are based on observed movements of marked and recaptured bears along with collared adult females (Jonkel *et al.* 1976; Kolenosky and Prevett 1983; Kolenosky *et al.* 1992; Obbard and Middel 2012; Middel 2014). The geographic range of the SH subpopulation includes much of eastern and southern Hudson Bay and James Bay and large expanses of the coastline of Ontario and Québec, as well as areas up to 120 km inland (Kolenosky and Prevett 1983; Obbard and Walton 2004; Obbard and Middel 2012).

An initial estimate of subpopulation size of 763 bears (SE = 323) was derived through a three-year (1984–1986) capture-recapture study conducted in mainland Ontario, primarily along the coast (Kolenosky *et al.* 1992). This estimate was subsequently adjusted to 1,000 for management purposes by the Canadian Polar Bear Technical Committee for numerous reasons: areas away from the coast may have been under-sampled due to the difficulty of locating polar bears in the boreal forest, an extensive fen area east of the Sutton River was under-sampled due to the difficulty of safely handling bears there, none of the offshore islands in James and Hudson bays were sampled, and some mainland areas around James Bay were not sampled (Lunn *et al.* 1998). A reanalysis of the 1984–1986 data produced an estimate for the study area of 641 (95% CI = 401–881 for those years; Obbard 2008; Obbard *et al.* 2007). A subsequent three-year (2003-2005) capture-recapture study conducted over the same geographical area and with similar capture effort, including the same biases due to likely under-sampling of islands, forested and very wet areas, produced an estimate of 673 (95% CI = 396–950; Obbard 2008). An analysis of bears captured on Akimiski Island in James Bay during 1997 and 1998 resulted in the addition of 70–110 bears to the total subpopulation estimate (Obbard 2008). Though the results of the two capture-recapture studies suggest that abundance was unchanged between 1984–1986 and 2003–2005, body condition declined and survival rates in

all age and sex categories tended to decline between the two capture periods, although point estimates were not significantly different because of the overlap in confidence intervals (Obbard *et al*. 2007; Obbard 2008).

Intensive aerial surveys were conducted during the fall ice-free season over mainland Ontario (same geographic area as for the capture-recapture studies, but with greater and more systematic coverage of inland areas) and Akimiski Island in 2011, and over the remaining islands in James Bay, the coastal areas of Québec from Long Island to the border between the SH and Foxe Basin subpopulations, and the off-shore islands in eastern Hudson Bay in 2012. Results of this survey developed using combined double-observer mark-recapture and mark-recapture-distance-sampling analysis provided an estimate of 860 bears (95% CI = 580–1,274) in the mainland Ontario, neighboring islands, and Akimiski Island portions of the SH management unit during the 2011 ice-free season. The estimate for the 2012 survey was 83 bears (SE = 4.5) in the 2012 study area. Thus, combining the aerial survey results from 2011 and 2012 to cover the entire SH subpopulation area yielded an estimate of 943 (95% CI = 658–1,350; Obbard *et al*. 2015). Overall, despite the difference in methodologies, assumptions, and biases between capture–recapture studies and aerial surveys, the evidence suggests it is likely that abundance of the SH subpopulation was broadly similar between 1986 and 2012.

The ice-free season within the SH subpopulation boundary increased by about 30 days from 1980 to 2012 (Obbard *et al*. 2016; Stern and Laidre 2016). Concurrently, body condition declined in all age and sex classes, though the decline was less for cubs than for other social classes (Obbard *et al*. 2016).

The intensive aerial survey was repeated in September 2016 to assess recent trend in abundance (Obbard et al., 2018). Two changes were made between the protocol for the 2011/2012 survey and that for the 2016 survey. First, all areas in Ontario, Nunavut and Québec were sampled within a three-week period to ensure complete coverage within the same year. Second, some inland transects were added along the Québec coast north of Long Island in response to feedback received during community consultations following the 2011/2012 survey and during planning for the 2016 survey. Data collection protocols and analyses were otherwise identical between the two surveys. Results of the 2016 survey suggest that abundance declined

by about 17% from 943 bears (95% CI = 658–1,350) in 2011/2012 to 780 (95% CI = 590–1,029) in 2016. The proportion of yearlings declined from 12% of the population in 2011 to 5% in 2016, whereas the proportion of cubs-of-the-year remained similar (16% in 2011 vs. 19% in 2016), suggesting low survival of the 2015 cohort (Obbard *et al*. 2018).

In 2018, a repeat of a portion of the aerial survey was conducted, flying a double-observer mark-recapture survey of the Ontario coastline and Akimiski Island coastline to assess whether the low proportion of yearlings seen in 2016 was a single event or indicative of declining recruitment. The results of this survey indicated a slightly lower abundance in the coastal area in 2018 (249 bears, 95% CI: 230 – 270) than in 2016 (269 bears, 95% CI: 244 – 297) and significantly lower abundance than in 2011 (422 bears, 95% CI: 381 – 467). The proportion of yearlings in the coastal area for the three surveys was variable (2011: 12%, 2016: 3%, 2018: 7%) as was the proportion of cubs (2011: 15%, 2016: 17%, 2018: 10%), but the proportion of adults in the coastal area increased in each survey (2011: 60%, 2016: 71%, 2018: 74%). The results of the 2018 survey should be used tentatively, as they are not a complete sample of the subpopulation.

In 2021, the comprehensive mark-recapture distance sampling and double-observer mark-recapture aerial survey was replicated following the 2016 and 2011/2012 designs (Northrup et al., 2022). Analyses yielded two separate estimates: one (N = 1003; 95% CI = 773-1302) assumed perfect detection on the transect line as in Obbard et al. (2018) and the other (N = 1119; 95% CI 860-1454) estimated detection probability on the transect line while accounting for the blind spot affecting rear observers (Wiig et al. 2022). The estimate of 1003 is most comparable to the 2016 estimate, whereas 1119 is a more robust estimate, reflecting advances in statistical approaches and the true abundance of bears in the study area in 2021. Both estimates indicate a substantially greater number of bears within SH than in 2016, with the 1003 estimate suggesting a 29% increase.

The increase in abundance relative to 2016 has two potential mechanisms based on the results of this survey and other available lines of evidence: 1) annual variation in the on-land distribution of bears in SH and the adjacent Western Hudson Bay (WH) subpopulation, and 2) an increase in population growth rate due to reduced mortality, increased birth rate or both.

Definitive evidence for either is not available, but there is evidence for each. First, distributional shifts of bears from the WH subpopulation into adjacent SH is supported through genetic identification of individuals biopsy-sampled along the Ontario coast, within the SH polar bear subpopulation management boundary, 2-7 days after the completion of the 2021 aerial surveys. Greater than 20% of the bears biopsied in SH, including both previously sampled and those never previously sampled, had previously only been sampled in WH (Environment and Climate Change Canada [ECCC] unpublished data). Although sampling effort had been higher in WH than SH in previous years, this suggests substantial movement of bears between the two subpopulations. A concurrent aerial survey of WH (Atkinson et al. 2022) revealed a decrease of 224 bears in WH from 2016 to 2021. This is numerically the same as the increase in between the estimate of SH abundance from Obbard et al. (2018) and the 2021 survey. The SH and WH subpopulations are not separated by any physical boundaries precluding movement between the two and there are large aggregations of bears on offshore islands and peninsulas at the boundary between the two subpopulations (61 bears were observed in SH within 50 km of the WH boundary and >60 bears have been observed on the Pen islands alone, which lay just inside the WH boundary). Minor variation in the distribution of these bears could greatly shift the number of individuals observed in WH or SH during aerial surveys. Variation in the distribution of bears within SH was also noted; specifically, there was a decline in the number of bears detected on Akimiski Island in 2021, with 50 bears detected in 2011, 46 in 2016 and only 14 in 2021.

To achieve the abundance estimated in 2021 from 2016, population growth would need to be nearly 30%. This seems highly implausible for a species such as the polar bear that has a slow life history strategy and would indicate that realized growth was near the upper confidence limit estimated by Regehr et al. (2021) for this subpopulation. However, improved demographic rates were likely a contributing factor to an increase in bears in SH. The years 2019 and 2020 were two of the three years with the longest duration of sea-ice since 2011 and ice conditions in SH were generally better between 2016 and 2021 than between 2011 and 2016 providing favorable conditions for higher reproductive output. Thirty-six percent of the bears observed in SH in 2021 were COYs and yearlings. The average COY litter size was 1.57 (σ =

0.52) and the average yearling litter size was 1.47 (σ = 0.50). The proportion of yearlings at 18% were markedly higher than 2016 (5%) and cubs-of-the-year comprised 18% which is similar to both the 2011/12 and 2016 surveys.

Decreased harvest mortality in this subpopulation was reported between 2017 and 2021 compared to 2011 and 2016 (37.8 bears per year compared to 45 bears per year). Further, in 2010, harvest in SH was substantially higher than average and above a generally considered sustainable level, with over 100 bears removed. This harvest would have reduced the population size just prior to the 2011 survey, but also, because it included a large number of females (>30 for a female harvest rate that may have approached 6%) may have had additional negative effects on population growth. The continued higher harvest of bears between the 2011 and 2016 survey compared to that between the 2016 and 2021 survey also is potentially significant. Thus, years of relatively good ice conditions, combined with comparatively reduced harvest from 2016 - 2021 may have buffered the population against further decline or allowed for recovery.

Recently, a quantitative harvest risk assessment was completed using data from harvested bears, capture-recapture studies, and aerial surveys conducted between 1984 and 2016 (Regehr *et al.* 2021*a*). This assessment considered alternative scenarios for how habitat loss may affect the demographic status of the SH subpopulation in the future and indicated that current harvest rates, under expected climate change scenarios, would have to be reduced to ensure a high probability of keeping the population above the maximum net productivity level (also referred to as maximum sustainable yield or the population size at which the population adds the greatest number of individuals each year).

Viscount Melville Sound (VM)

Initial research on the VM polar bear subpopulation was conducted in the early 1970s but only included field work in the southern portions of the subpopulation's range (Schweinsburg *et al.* 1981). A five-year study of movements and subpopulation size, using telemetry and mark-recapture, was completed for polar bears inhabiting VM in 1992 (Messier *et al*. 1992, 1994; Taylor *et al*. 2002). Subpopulation boundaries were based on observed movements of female

polar bears with satellite radiocollars and movements of bears tagged in and out of the study area (Bethke *et al*. 1996; Taylor *et al*. 2001). The most recent subpopulation estimate of 161 bears (SE = 40) dates from 1992 (Taylor *et al*. 2002), and the PBSG regards VM as a data-deficient subpopulation. However, in spring 2014, the field component of a mark-recapture study (2012–2014) to reassess abundance and status of the VM subpopulation was completed. This study included applying radiocollars to female polar bears to evaluate movements and population delineation. Analysis of the resulting capture-recapture and movement data are underway, and a revised subpopulation estimate is expected in the near future. Harvest for the VM subpopulation has been typically below the quota.

Western Hudson Bay (WH)

Hudson Bay is a relatively shallow inland sea that is ice covered in winter and ice free in summer (Hochheim *et al*. 2010). Although three subpopulations of polar bears (Foxe Basin [FB], Southern Hudson Bay [SH], and WH) occur on the sea ice of Hudson Bay in winter and spring, they appear to be largely segregated during the open-water season (Derocher and Stirling 1990; Peacock *et al*. 2010; Viengkone *et al*. 2016). During the ice-free period, WH polar bears exhibit strong fidelity to terrestrial summering areas in northeastern Manitoba (Stirling *et al*. 1977; Derocher and Stirling 1990; Cherry *et al*. 2013; Stapleton *et al*. 2014; Lunn *et al*. 2016). The current WH subpopulation boundary is based largely on capture-recapture data and harvest data for tagged animals (Stirling *et al*. 1977; Derocher and Stirling 1990, 1995*a*; Taylor and Lee 1995; Lunn *et al*. 1997).

Although the size of the WH subpopulation was unknown until the 1990s (Derocher and Stirling 1995*a*), sightings of polar bears by Inuit have increased in recent decades relative to historical levels of the early 1900s to 1970s (McDonald *et al*. 1997; Tyrrell 2006, 2009; Nirlungayuk and Lee 2009; Henri *et al*. 2010; Kotierk 2012). A factor likely contributing to this observed increase was a population-level response to decreased hunting pressure that occurred in the 1950s and 1960s resulting from the closure of the fur trading post at York Factory, withdrawal of military personnel from Churchill, and the closure of hunting in Manitoba (Stirling *et al*. 1977; Derocher and Stirling 1995*a*).

Derocher and Stirling (1995*a*) estimated the mean subpopulation size for 1978–1992 to be 1,000 bears (SE = 51). However, this estimate was considered conservative because the study had not covered the southern portion of the range east of the Nelson River (Calvert *et al*. 1995; PBSG 1995). Therefore, for management purposes, the subpopulation size was adjusted to 1,200 (Calvert *et al*. 1998). In 1994 and 1995, Lunn *et al*. (1997) expanded the capture program to sample animals to the management boundary between the WH and SH subpopulations, and estimated abundance to be 1,233 (SE = 209) in 1995. Regehr *et al*. (2007) reported a decline in abundance from 1,194 (95% CI = 1,020–1,368) in 1987 to 935 (95% CI = 794–1,076) in 2004, and documented that the survival rates of cubs, subadults, and old bears (>20 years) were negatively correlated with the date of sea-ice breakup.

A mark-recapture distance-sampling study resulted in an abundance estimate of 1,030 bears (95% CI = 754–1,406) in 2011 (Stapleton *et al*. 2014). During this survey, 711 total bears were observed and more bears, particularly adult males, were observed in the coastal areas east of the Nelson River towards the WH-SH boundary than were documented during the late 1990s (Stirling *et al*. 2004). Stapleton *et al*. (2014) suggested that a distributional shift may have negatively biased abundance estimates derived from capture samples. Mean litter size (cubs-of-the-year, 1.43 [SE = 0.08]; yearlings, 1.22 [SE = 0.10]) and numbers of cubs as a proportion of total observations (cubs-of-the-year, 0.07; yearlings, 0.03) were lower than those recorded for the neighboring subpopulations of FB and SH, which is consistent with the WH subpopulation having low reproduction (Regehr *et al*. 2007; Peacock *et al*. 2010; Stapleton *et al*. 2014). The body mass of solitary adult female polar bears has declined over the past 40 years, which has likely contributed to declining reproductive success (Derocher and Stirling 1995*b*; Stirling *et al*. 1999; Sciullo *et al*. 2016; Lunn and McGeachy 2020; Molnár *et al*. 2020).

Lunn *et al*. (2016) evaluated the demography and status of the WH subpopulation for the period 1984–2011 using a Bayesian implementation of multistate capture-recapture models coupled with a matrix-based demographic projection model to integrate several types of data and to incorporate sampling uncertainty and demographic and environmental stochasticity across the polar bear life cycle. Their analysis resulted in an estimate of 806 bears (95% CI = 653–984) in the core area of study north of the Nelson River in 2011. Although the abundance

estimates from the aerial survey and capture-recapture model are broadly similar with overlapping confidence intervals, it is difficult to make direct comparisons because the studies differed with respect to spatial and temporal perspectives and the assumptions of each method (Lunn *et al*. 2016). The aerial survey provides a snapshot estimate of the total number of polar bears in the WH management unit at the time of the survey, whereas the point estimate of abundance from the capture-recapture model is based on the number of bears that moved through the smaller capture-recapture sampling area over multiple years.

A second aerial survey was flown in 2016 using a mark-recapture distance-sampling methodology (Dyck *et al*. 2017). Pre-survey consultations with Nunavut Hunters' and Trappers' Organizations, Kivalliq communities, and with the Manitoba Department of Sustainable Development were conducted to include local and traditional knowledge in the study design. Dyck *et al*. (2017) estimated there to be 842 bears (95% CI = 562–1121) that, although not significantly different from the previous aerial survey estimate, represented an 18% decline in abundance between 2011 and 2016. Over the same period and using similar methods, Obbard *et al*. (2018) estimated a 17% decline in abundance for the neighboring SH subpopulation. Similar to observations from the 2011 survey, cubs-of-the-year and yearling cubs comprised a small proportion of the sample size (Dyck *et al*. 2017), suggesting that low reproductive performance of the WH subpopulation has continued.

A third aerial survey was conducted in 2021 using double observer and distance sampling methods and based on the previous surveys completed in 2011 and 2016 (Atkinson et al., 2022). This provided three replicates of aerial surveys for the WH subpopulation over a 10-year period. The overall distribution of bears mirrored previous surveys with high concentrations of bears along the coast and offshore islands.

The estimated abundance of WH in 2021 was 618 bears (SE = 119.3, CI = 425-899, CV=0.19). Comparison to aerial survey estimates from 2011 and 2016 suggested a decreasing trend in abundance in WH. Specifically, the number of adult female and subadult bears declined between 2011 and 2021 while adult males showed no trend. The mechanism behind this significant reduction of these cohorts is unclear but may reflect multiple factors, including

reduced recruitment and survival of subadults, distributional shifts on land, and harvest pressure.

Few yearlings were seen in the 2011 and 2016 surveys supporting potential low recruitment compared to other subpopulations (Stapleton et al., 2014; Dyck et al., 2017) over the past 10 years. In contrast, in 2021, mean litter sizes were 1.46 (SD: 0.50; n = 13) and 1.39 (SD: 0.63; n = 13) for cubs-of-the-year and yearlings, respectively, which are similar to other subpopulations. Neighboring SH also saw increased yearlings in 2021 and proposed that the favorable ice conditions in the previous two years to the survey were contributing factors to improved survival and recruitment of this cohort (Northrup et al., 2022).

Distributional shifts of bears across the WH-SH management boundary could exert large effects on abundance estimates calculated from aerial surveys. Evidence suggests that there is some movement between the subpopulations, related to sea ice conditions (Cherry et al., 2013). However, preliminary evidence from ongoing genetic mark-recapture work near the subpopulations' boundaries shows marked WH bears recaptured inside the SH boundary to be predominantly adult males (McGeachy et al., 2022). Given that the declines in WH are predominately adult females and subadults, return of these males back into WH may increase abundance but does not address the observed declines of females and subadults.

The decline in WH abundance may also be related to increased harvest pressure. Public pressure to increase harvest quotas to address safety concerns and IK stating an increase in abundance of bears in WH over the last 10 years have resulted in harvest going from 2.5% of estimated abundance to ~6.5% in 2021. Harvest management originally established 2:1 male-biased quotas but changes in harvest management policy in 2020 now allow up to 50% of the harvest to be female.

Although Atkinson *et al*. (2022) were unable to definitively conclude whether the finding of declining abundance in WH was the result of reduced survival and recruitment, movement of bears into neighbouring subpopulations (emigration), harvest pressure, or a combination thereof, they concluded that it was plausible that all these factors have contributed to some degree and that the observed declines are consistent with long-standing predictions regarding the demographic effects of climate change on polar bears.

From the 1930s through the 1960s, encounters with polar bears in the interior of the Kivalliq mainland and along the Kivalliq coast of Hudson Bay were rare (Nirlungayuk and Lee 2009; Tyrrell 2009). Within the last few decades, encounters with polar bears in the Kivalliq region have increased, resulting in more bear-human encounters and increased concerns for human safety and property damage (Tyrrell 2006, 2009; Henri *et al*. 2010).

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Status Table for the World's Polar Bear Subpopulations

IUCN/SSC Polar Bear Specialist Group

Version: 20231017

Subpopulation	Subpopulation size — Estimate and uncertainty	Subpopulation size — Method and type of evidence	Subpopulation size — Most recent year of estimate, and citation	Change in subpopulation size — Long term (≥2 polar bear generations [≥23 yr])	Subpopulation trend — Short term (approx. 1 polar bear generation [11.5 yr])	Sea-ice metrics (1979 to 2022) — Change in date of spring sea-ice retreat (days per decade) / change in date of fall sea-ice advance (days per decade)	Sea-ice metrics — Change in summer sea ice area (percent change per decade)	Human-caused removals (2016/17 to 2020/21) — 5-year mean potential (bears per year)	Human-caused removals — 5-year mean actual (bears per year) and what this represents as a percentage of the total population	Comments, vulnerabilities, and concerns
Arctic Basin	Unknown	N/A	N/A	Data deficient	Data deficient	-14.9 / +11.9	-7.7	N/A	N/A	The AB subpopulation is a geographic designation to account for polar bears occurring in the most northern areas of the circumpolar Arctic that are not clearly part of other subpopulations. The total number of bears that use the AB region, and whether these bears are residents or transients, is unknown.
Baffin Bay	2826 (95% CI = 2284–3367)	Genetic capture-recapture (estimated)	2013 (Atkinson et al. 2021)	Data deficient	Data deficient	-5.6 / +4.1	-21.5	160 (NU:80+GL:80)	144.2 (5.1%)	Due to evidence that the sampling design and environmental conditions likely resulted in an underestimate of abundance in the 1990s, the estimates of abundance for the 1990s and 2010s are not directly comparable and trend cannot be determined. Satellite telemetry analyses comparing movements of adult females in 1990s to 2000s indicate reduced seasonal ranges, increased isolation, 30+ days more on land on Baffin Island in summer, reduced body condition, reduced cub recruitment with early sea-ice breakup, and increased swimming.
Barents Sea	2644 (95% CI = 1899–3592)	Distance sampling (estimated)	2004 (Aars et al. 2009)	Very likely increased (1973-2015)	Likely stable (2004 to 2015)	-13.5 / +18.7	-23.5	N/A	N/A	There has been no legal hunting of polar bears in Russia since 1957 and in Norway since 1973. Different sources of demographic studies suggest the population grew steadily after 1973, and at least until recently (ca. 2015; Aars et al. 2017). Recent habitat decline has led to late sea-ice formation in autumn around some important denning habitat, and in such years few females den in these areas. Den distribution may have shifted from east Svalbard to Franz Josef Land in most years. In 2015, the Norwegian portion of the subpopulation was surveyed. It was indicated that the number of local bears in Svalbard was similar to in 2004, and that more bears were on the pack ice. Possibly, bears could have shifted westward from Russian to Norwegian areas in the pack ice, thus an increase is not conclusive over the last generation. There is no evidence of large scale reduction in body condition.
Chukchi Sea	2937 (95% CI = 1552–5944)	Physical capture-recapture with density extrapolation (estimated/projected)	2016 (Regehr et al. 2018a)	Data Deficient	Likely stable (2008 to 2016)	-6.2 / +6.9	-31.5	85 (changed from 58 in July 2018)	12.4 (0.4%) in U.S. + approx. 32 (1.1%) in Russia	Estimate of subpopulation trend is from Regehr et al. (2018b). Independent estimates of abundance from a 2016 aerial survey (Conn et al. 2021) are larger than, but of similar magnitude to, the estimate of 2,937 from Regehr et al. (2018a). Indices of of body condition and recruitment from springtime research have been good, although autumn observations from 2004–2010 may suggest declining cub survival. Longer ice-free periods are increasing land use. Subsistence harvest is legal and monitored in the US. Harvest is illegal and un-monitored in Russia.
Davis Strait	2015 (95% CRI = 1603–2588)	Genetic capture-recapture (estimated)	2018 (Dyck et al. 2022)	Data deficient	Likely decreased (2007 - 2018)	-5.8 / +7.3	-21.4	QC + 76 (NU:61+NL:12+GL:3)	66.8 (3.3%)	Management objective to slightly decrease the subpopulation from the Nunavut jurisdiction resulted in increases to harvest quotas and an increase in harvesting in Quebec, where reporting is not mandatory, which may have contributed to the slight decline. Most recent analysis did not detect links between survival and various ice conditions in the study period (2005-2018). Longer ice-free periods may be increasing land use and driving concerns of increases in human-wildlife conflict.
East Greenland	Unknown	N/A	N/A	Data deficient	Data deficient	-5.7 / +7.0	-8.3	65.0	68.0 (N/A)	Reduction in sea-ice habitat quality has led to changes in habitat use based on telemetry analyses. A new assessment of the subpopulation began in 2014.
Foxe Basin	2585 (95% CI = 2096–3189)	Mark-recapture distance-sampling (estimated)	2010 (Stapleton et al. 2016)	Data deficient	Likely stable (1994 to 2010)	-4.5 / +4.5	-16.2	QC + 123 (NU:123)	109.8 (4.2%)	There are no estimates of vital rates. Harvest appears to be sustainable.
Gulf of Boothia	1525 (95% CI = 949–2101)	Genetic capture-recapture (estimated)	2017 (Dyck et al. 2020b)	Data deficient	Likely stable (2000 to 2017)	-7.2 / +7.3	-16.2	74 (NU:74)	65.8 (4.3%)	Change from multiyear to annual ice provides likely improved habitat productivity at present. Increased shipping could become a concern.
Kane Basin	357 (95% CI = 221–493)	Genetic capture-recapture (estimated)	2014 (Laidre et al. 2023)	Data deficient	Likely increased (1997 to 2014)	-6.0 / +4.9	-11.2	11 (NU:5+GL:6) through 2017, changed to 14 (NU:5+GL:9) in 2018	7.6 (2.1%)	More bears were documented in the eastern regions of the KB subpopulation area during 2012 – 2014 than during 1990s surveys which may reflect differences in spatial distribution of bears, possibly influenced by reduced hunting pressure by Greenland in eastern KB, but also some differences in sampling protocols between decades. Some caution should be taken in the interpretation of population growth. An additional estimate of abundance based on a springtime 2014 aerial survey in KB was 190 bears (95% lognormal CI: 87 - 411; Wiig et al. 2022).
Kara Sea	Unknown	N/A	N/A	Data deficient	Data deficient	-9.7 / +8.7	-29.1	N/A	N/A	There has been no legal harvest in the KS subpopulation since 1957. Amount of illegal hunting unknown.
Lancaster Sound	2541 (95% CI = 1759–3323)	Physical capture-recapture (estimated)	1997 (Taylor et al. 2008b)	Data deficient	Data deficient	-5.5 / +4.7	-7.7	85 (NU:85)	70.8 (2.8%)	Demographic data are >20 years old. Selective hunting for males in the harvest decreased due to the U.S. import ban and listing under the U.S. Endangered Species Act in 2009, but as of 2021 are nearly back to pre-ESA listing levels. Increase in shipping activities.
Laptev Sea	Unknown	N/A	N/A	Data deficient	Data deficient	-7.4 / +6.8	-22.6	N/A	N/A	There has been no hunting in the LS subpopulation since 1957. In 2018, a federal sanctuary (zakaznik) on the archipelago of the Novosibirsk Islands was created by a decree of the Government of the Russian Federation.
M'Clintock Channel	716 (95% CRI = 545–955)	Genetic capture-recapture (estimated)	2016 (Dyck et al. 2020a)	Data deficient	Likely increased (2000 to 2016)	-3.7 / +5.2	-9.5	12 (NU:12)	9.4 (1.3%)	Potential for shipping activities. The subpopulation was managed for recovery with harvest below sustainable rates. Change from multiyear to annual ice provides likely improved habitat productivity at present. Harvest quota was increased to 21 bears per year for 2021/22 harvest season based on most recent abundance estimate.
Northern Beaufort Sea	980 (95% CI = 825–1135)	Physical capture-recapture (estimated)	2006 (Stirling et al. 2011)	Data deficient	Data deficient	-7.2 / +3.3	-6.3	77 (NWT+GN)	31.6 (3.2%)	Declines in sea-ice habitat and observed and predicted declines in the health and abundance of ringed seals are of concern. Harvest is currently managed using the updated (ca. 2020) subpopulation boundary at 133° W between the Southern Beaufort Sea and Northern Beaufort Sea subpopulations. Sea ice metrics used in the status table reflect the updated boundary from 1979-2022. Current harvest rates are based on a managed subpopulation size of 1,710 bears reflecting (1) the revised subpopulation boundary, which increased the geographic area of the NB subpopulation (Griswold et al. 2017); and (2) potential negative bias in the current abundance resulting from incomplete sampling of the subpopulation (Stirling et al. 2011). A new subpopulation assessment is underway.
Norwegian Bay	203 (95% CI = 115–291)	Physical capture-recapture (estimated)	1997 (Taylor et al. 2008b)	Data deficient	Data deficient	-1.5 / +4.7	-1.9	4 (NU:4)	0.8 (0.4%)	Initial population viability analysis simulations resulted in population decline after 10 years, although vital rates from the NW and LS subpopulations were pooled for the analyses. Projections of decline were also high because of small sample size. Current data are >20 years old; small population.
Southern Beaufort Sea	~900 (90% CI = 606–1212)	Physical capture-recapture (estimated)	2010 (Bromaghin et al. 2015)	Likely decreased (1983 to 2015)	Likely decreased (2001 to 2015)	-9.5 / +9.0	-31.2	56 (21:NWT+35US)	18.6 (2.1%)	The population-wide estimate of abundance from Bromaghin et al. (2015) reflects the previous SB-NB boundary at 122° W. The subpopulation trend reflects information presented in Bromaghin et al. (2021), which assessed survival and abundance in the U.S. portion of the SB subpopulation. Other data in the Status Table reflect the revised boundary at 133° W (updated ca. 2020), including sea ice metrics for 1979-2022. Concerns include declining body condition and increased frequency of fasting, periods of low survival, and growing reliance on land during summer, as identified in studies based on the previous eastern boundary delineation (Rode et al. 2018, Atwood et al. 2021, Bromaghin et al. 2021, Rode et al. 2022). An additional concern in Alaska is the growing potential for human-polar bear conflict arising from increased industrial development of the coastal plain (Atwood et al. 2020, Wilson and Durner 2020). A new subpopulation assessment is underway.
Southern Hudson Bay	1119 (95% CI 860-1454)	Mark-recapture distance-sampling (estimated)	2021 (Northrup et al. 2022)	Likely stable (1986 to 2021)	Likely stable (2012 to 2021)	-1.6 / +2.9	-7.7	ON + QC + 48 (NU:25+NMR:23)	37.8 (3.4%)	Increased time onshore due to changes in breakup and freeze-up; declining body condition; declining survival rates, especially for cubs-of-the-year. The 2021 point estimate was at least 29% greater than the 2016 estimate. Distributional shifts from adjacent WH and population growth from decreased harvest mortality and improved recruitment are likely drivers of this increase. The years 2019 and 2020 were good ice years relative to the years prior to the 2016 survey.
Viscount Melville Sound	161 (95% CI = 93–229)	Physical capture-recapture (estimated)	1992 (Taylor et al. 2002)	Data deficient	Data deficient	-1.8 / +7.2	-4.6	7 (NWT+GN)	2.0 (1.2%)	Low densities of ringed seals and polar bears were observed during capture-recapture sampling conducted 2012–2014. Field sampling to estimate abundance was completed 2014. Analyses are underway but the final report is not yet available.

| Western Hudson Bay | 618 (SE = 119.3, CI = 425-899) | Mark-recapture distance-sampling (estimated) | 2021 (Atkinson et al. 2022) | Very likely decreased (1995 to 2021) | Very likely decreased (2011 to 2021) | -4.2 / +2.8 | -19.9 | MB + 38 (NU:38) | 30.8 (5.0%) | Concerns include harvest, increased time onshore due to changing dates of breakup and freeze-up, declines in body condition, and lower productivity. Earlier declines in size of subpopulation linked to reduced survival due to timing of sea-ice breakup. The 2021 point estimate of abundance was 26.7% lower than the 2016 estimate and partially due to temporary movement of some bears to SH as well as diminished vital rates. |