2021 Southern Hudson Bay polar bear subpopulation aerial survey

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INTRODUCTION

Climate warming is causing rapid and widespread impacts to Arctic ecosystems (Post et al. 2009) where temperatures are increasing at two to four times the global average (IPCC 2013, Rantanen et al. 2022). These impacts have had profound effects on a variety of Arctic wildlife species, causing population declines, reduced reproductive output, and shifts in the food web (Regehr et al. 2007, Post and Forchhammer 2008, Laidre et al. 2015, Descamps et al. 2017, Mallory and Boyce 2018). The impacts of climate change on Arctic ecosystems have had significant consequences for Indigenous peoples that rely on Arctic species for subsistence (Durkalec et al. 2015, Laidre et al. 2015, Ostapchuk et al. 2015, Kanatami 2019). As climate change continues to alter Arctic ecosystems (IPCC 2022), it is critical to monitor impacted species to provide information to local communities for use in decision-making and to assess general impacts to people and biodiversity from a warming climate.

Polar bears (*Ursus maritimus*) exemplify the challenges facing Arctic species under a changing climate. Polar bears are dependent on sea ice for nearly every stage of their life: they hunt their primary prey from the sea-ice platform, mate and, in some locations, even den on the sea ice (Amstrup and Gardner 1994). Thus, declines in sea ice have direct implications for nutrition, reproduction and the long-term population viability for polar bears. Although sea-ice extent and duration have declined in the last few decades over the circumpolar distribution of polar bears (Stern and Laidre 2016), the impacts to polar bear subpopulations have varied, with some experiencing declines in body condition, survival and abundance (Regehr et al. 2007, Lunn et al. 2016, Obbard et al. 2016, Obbard et al. 2018) and others experiencing limited effects or even near-term benefits as areas transition from multi-year ice to thinner, annual ice or areas in which access to shallow, highly productive ecoregions remains (Regehr et al. 2018, Laidre et al. 2020, Dyck et al. 2021, Dyck et al. 2022).

Polar bears are an important cultural, nutritional and financial species to Indigenous peoples that have coexisted with them for centuries (Wenzel 2004, Henri et al. 2010, Laforest et al. 2018).The harvest of polar bears is monitored through management frameworks in various jurisdictions across Canada (Taylor et al. 2008, Lunn et al. 2018), all aiming for sustainable harvest management and continued population viability. However, the logistical and analytical challenges involved with enumerating polar bear populations, as well as the often long intervals between surveys, adds uncertainty to the achievement of this goal. Compounding uncertainty of the responses of bears to climate warming increases the complexity of identifying the sustainability of harvest levels (Regehr et al. 2017, Regehr et al. 2021). Thus, monitoring polar bear populations in the face of ongoing climate warming is critical for providing local communities that rely on polar bears with additional information for harvest management decision-making.

Polar bears are divided into 19 relatively discrete subpopulations (Durner et al. 2018) delineated using a variety of methods, including capture and recapture data, genetics, and movement data from collared individuals (Paetkau et al. 1999, Taylor et al. 2001, Amstrup et al. 2004). The Southern Hudson Bay (SH) subpopulation represents the furthest south continuously occupied area of the globe for polar bears, and, as such, is a critical location for monitoring the impacts of climate warming. The marine portions of the SH subpopulation include the eastern and southern portions of Hudson Bay and all of James Bay (Fig. 1). The subpopulation also encompasses nearly the entirety of the coastline of Ontario, large areas of the western coastline of Québec, and areas of both provinces up to 120 km inland.

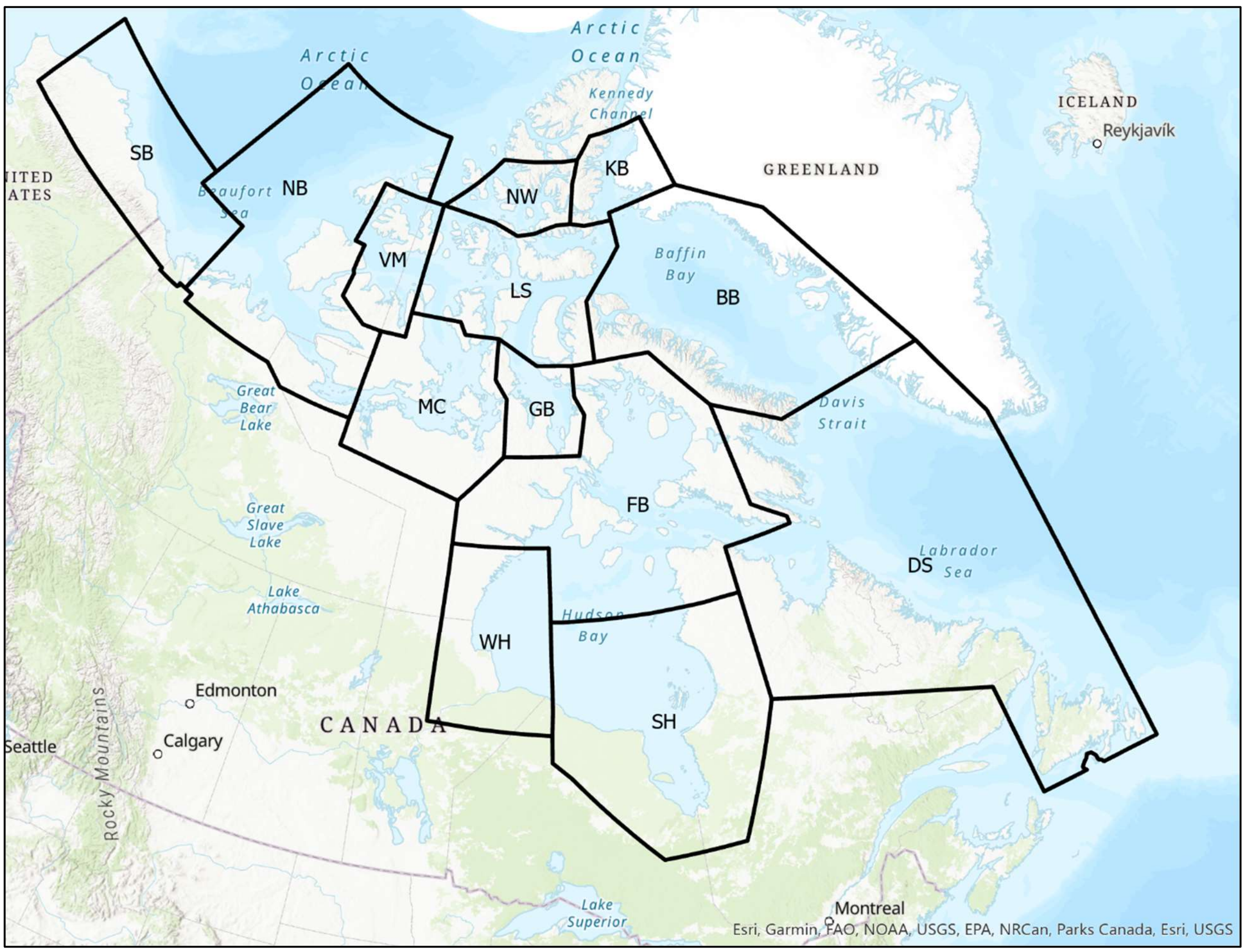


Figure 1. Boundaries of polar bear subpopulations that are partially or totally under management by Canadian jurisdictions. SB, Southern Beaufort Sea; NB, Northern Beaufort Sea; VM, Viscount Melville Sound; MC, M'Clintock Channel; LS, Lancaster Sound; NW, Norwegian Bay; KB, Kane Basin; BB, Baffin Bay; GB, Gulf of Boothia; FB, Foxe Basin; DS, Davis Strait; WH, Western Hudson Bay; and, SH, Southern Hudson Bay.

The first abundance estimate for SH was obtained between 1984 and 1986 by Kolenosky et al. (1992) using physical capture-mark-recapture conducted primarily along the Ontario coast of Hudson Bay and including some inland areas. This effort extended somewhat into the current limit of the Western Hudson Bay (WH) subpopulation and produced an estimate of 763 bears (± 323) but was later adjusted upwards to 1000 bears for management purposes because no sampling was conducted on the James Bay coast of Ontario, the Québec coast, or any of the offshore islands of James and Hudson bays (Lunn et al. 1998). During 1997 and 1998, a capture-mark-recapture effort was undertaken on Akimiski, North and South Twin Islands in James Bay. Although a formal estimate was never published for these efforts, Obbard et al. (2007) citing Obbard and Howe (unpublished data) report abundance estimates ranging from 70 to 110 bears, which were derived from several models (minimum lower confidence limit across models = 56 and maximum upper confidence limit across models = 195). Between 2003 and 2005, Obbard et al. (2007) conducted another physical capture-mark-recapture effort, covering the same area as assessed in the 1980s, but more thoroughly covering areas up to 40 km inland from the coast. Further, they reanalyzed the data from 1984-1986 excluding captures occurring outside of the current SH subpopulation boundary. This work estimated that there was an average of 641 bears (95% CI = 401-881) between 1984 and 1986 and 681 bears (95% CI = 401-961) between 2003 and 2005 in the study area, indicating the population in the surveyed area was likely very similar between the two survey periods. However, concurrent with these abundance estimates, declines in the point estimates of survival between the 1980s and 2000s were documented (Obbard et al. 2007) as well as significant declines in body condition of bears (Obbard et al. 2016). Further, the ice-free season in SH increased by approximately three weeks between the 1990s and 2010s (Hochheim and Barber 2014). Thus, while it appears that the population abundance along the Ontario coast of Hudson Bay and the areas inland was largely similar between the 1980s and mid-2000s, there was evidence that the population might be facing nutritional issues and attendant declines in survival and body condition related to declining sea ice. Concurrently, the adjacent WH subpopulation had seen similar declines in survival and body condition as well as abundance during the same period (Regehr et al. 2007, Lunn et al. 2016). Lastly, there remained areas of the subpopulation, including the Québec coast, large portions of the James Bay coast, and several James Bay and Hudson Bay islands, that had still not been surveyed rigorously enough to contribute to abundance estimates at that point (Leafloor 1990, Crête et al. 1991).

Although physical capture programs offer some of the best data for understanding polar bear vital rates and population dynamics and vital rates, while also enabling the collection of data on body condition, they are logistically challenging, expensive to undertake, and take several years to produce robust estimates. Further, Indigenous peoples that coexist with polar bears have raised concerns about the handling and chemical immobilization of polar bears for scientific and management purposes (Peacock et al. 2009, Service

Canadien de la Faune 2010, Henri et al. 2010, Wong et al. 2017, https://www.itk.ca/wp-content/uploads/2019/08/A09-06-11-Approval-of-Polar-Bear-Research-Methods.pdf accessed November 16, 2022). Starting in 2011, management authorities for SH and WH moved to an aerial survey-based approach for enumerating these subpopulations (Stapleton et al. 2014, Obbard et al. 2015, Dyck et al. 2017). Less information is gained through aerial surveys relative to mark-recapture efforts, so, after conducting power analyses, jurisdictions agreed that surveys would occur on a more regular basis and be repeated every five years. Thus, in 2011, Obbard et al. (2016) implemented a combined distance sampling and double-observer mark-recapture aerial survey of the Ontario coast and areas up to 60 km inland along with Akimiski Island. At the time, there was insufficient funding to also Survey the Québec coast and offshore islands of James and Eastern Hudson Bay (M. Obbard personal communication), but these areas were subsequently surveyed in 2012. This was the most comprehensive survey of the SH subpopulation to date and produced an estimate of 943 bears (95% CI = 658-1350). This survey was repeated in 2016, with all areas surveyed in a single season (Obbard et al, 2018). This effort produced an estimate of 780 bears (95% CI = 590-1029), suggesting the population may have declined between 2011 and 2016. Further, the age composition of observed bears in the 2016 survey was suggestive of a poor survival of cubs to yearling stage considering few yearling bears were seen. An additional double-observer mark-recapture survey of only the coastline of Ontario, where the greatest density of bears occurs, was conducted in 2018 to examine indices of recruitment and obtain an estimate of the coastal population. This survey was an exact replicate of a portion of the 2011 and 2016 double-observer mark-recapture surveys, which allowed for a direct comparison of this portion of the population across years. The results showed that the proportion of yearlings was slightly higher in this area in 2018 than in 2016, but the number of bears inhabiting the coast was slightly lower at 249 bears (95% CI = 230-270) compared to 2016 ($\bar{x}$ = 269, 95% CI = 214-297) and substantially lower than 2011 ($\bar{x}$ = 422, 95% CI = 381-467; Northrup and Howe 2019).

Similar to other subpopulations in Canada, the harvest of SH polar bears has long been targeted for a 4.5% removal rate at a sex ratio of 2 males per female. This rate has been considered sustainable for polar bears (Taylor et al. 1987), though there is evidence that it may have been conservative for bears in SH over the last 20 years (Regehr et al. 2021). Polar bears in the SH subpopulation are harvested by Inuit in Nunavut and Nunavik and by Cree in Québec and Ontario, though recorded Cree harvests in Ontario were much greater in the 1970s through 1990s than at the time of this report (OMNRF unpublished data). Management authority for the SH subpopulation is complex as it is the shared responsibility of the Governments of Ontario, Québec, Nunavut, and Canada, along with the Nunavut Wildlife Management Board, Nunavik Marine Region Wildlife Management Board, the Eeyou Marine Region Wildlife Board, Hunting, Fishing and Trapping Coordinating Committee, Land Claims Organizations representing Indigenous rights, specifically Nunavut Tunngavik Incorporated, Makivik Corporation and the Cree Nation Government in Québec, and several Cree First Nations in Ontario. The harvest of SH bears in Nunavut has been managed under a strict quota system since the 1970s, whereas harvest monitoring in Québec and Ontario remains incomplete as of this report. Total annual reported harvest within the subpopulation varies annually but averaged 48 bears between 2010-11 and 2020-21 (range 31-104; https://www.polarbearscanada.ca/en/polar-bears-canada/canadas-polar-bear-subpopulations; accessed July 22, 2022).

There are sixteen coastal communities in the SH subpopulation (Fig. 2). Between 1980 and 2019, the Inuit community of Sanikiluaq, Nunavut had a total allowable harvest (TAH) of 25 bears at a male to female ratio of 2:1. The Sanikiluaq harvest was reduced to 20 bears per year for two years following the 2011-12 aerial survey. The management framework allows for annual variation in the actual harvest depending on over- or under-harvest compared to the TAH (Government of Nunavut 2019). A revision of the Nunavut polar bear harvest management system in 2019 allows the sex ratio of the harvest to reach up to one female bear for every male bear (up to 1:1). With this management change, the TAH for Sanikiluaq remained at 25 bears, indicating the potential for a greater number of female bears to be harvested after this time. Harvest reporting in Nunavut is believed to approach 100% and the average annual reported harvest for the 2010-11 to 2020-21 period was 26.2 bears (range 20 to 47 bears).

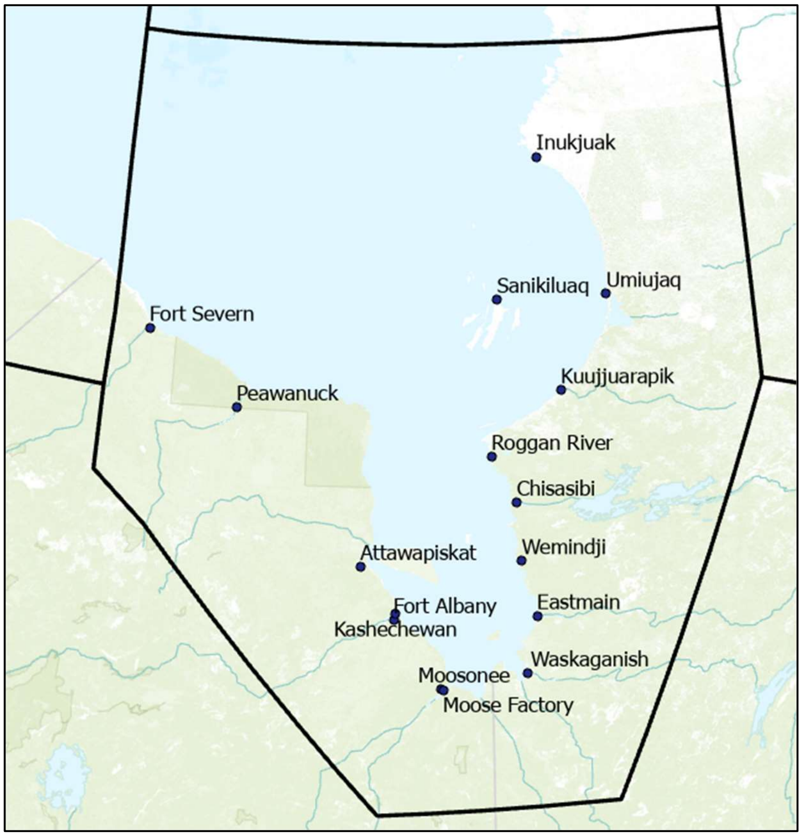


Figure 2. Coastal communities falling within the SH subpopulation boundary in Ontario, Québec and Nunavut.

In Québec, three Nunavik Inuit communities (Inukjuak, Umiujaq, and Kuujjuaraapik) and five coastal Cree communities (Whapmagoostui, Chisasibi, Wemindji, Eastmain and Waskaganish) potentially harvest from this subpopulation. There are currently no legal requirements for beneficiaries of the James Bay and Northern Québec Agreement (Québec Government 1976) to report human-caused polar bear mortalities but reporting and tagging of polar bear hides is necessary for hides to enter the domestic or international trade market. The proportion of the harvest reported to the Québec Government is currently unknown. Voluntary agreements were signed in 2011[1] and 2014[2] establishing harvest limits within the SH subpopulation for Nunavik Inuit and Cree of Eeyou Istchee and Ontario, and a total allowable take (TAT) was also established by the federal and Nunavut governments in 2016 for bears harvested within the Nunavik Marine

[1] A temporary voluntary limit of 26 bears for Nunavik Inuit, 25 for Inuit from Sanikiluaq, 4 for Cree of Eeyou Istchee, and 5 for Ontario Cree was established (including subsistence hunting and defense kills) for the 2011/12 harvest season.

[2] A temporary voluntary limit of 22 bears for Nunavik Inuit, 20 for Inuit from Sanikiluaq, and 3 bears for Ontario and Québec Cree with alternating division per harvest season for Cree was established for the 2014/15 and 2015/16 harvest seasons.

Region[3]. However, enforcement of those harvest limits remains problematic, and no harvest limits have been established in most of the Eeyou Marine Region nor in onshore Québec. Average annual reported harvest in Québec for the 2010-11 to 2020-21 period was 19.7 bears (range 5 to 74 bears).

In Ontario, there are three coastal Cree communities that have traditionally harvested polar bears (Fort Severn, Winisk (Peawanuck) and Attawapiskat). There are three additional Cree communities (Moose Factory, Fort Albany, and Kashechewan), and one non-Indigenous community (Moosonee) that are outside the generally occupied range of bears but occasionally have defense of life and property kills. In 1976, an informal agreement between the Ontario government and the coastal Cree First Nation Communities established that a maximum of 30 bear hides could be sealed for trade annually. The 2011[4] and 2014[2] voluntary agreements also set maximum harvest limits on Ontario Cree but the proportion of the harvest that is reported to the Government of Ontario is currently unknown. Since polar bears were listed as threatened in Ontario in 2009, the sale of bear parts has been prohibited in the province.

A harvest risk assessment conducted by Regehr et al. (2021) indicated that under ongoing climate warming, harvest of polar bears in SH would likely need to decline in coming years to ensure harvest sustainability. Further, evidence outlined above suggests the SH subpopulation may be experiencing demographic challenges related to ongoing declines of sea ice. As such, there is a clear, continued need to assess the abundance of this subpopulation to monitor trend and support harvest management (Regehr et al. 2021). In keeping with management authority goals, a comprehensive aerial survey of SH was conducted in summer 2021 that maintained a nearly identical design as the previous surveys. Here we present the results of this third survey to provide a direct comparison across the three survey periods (2011/12, 2016 and 2021).

METHODS

Study area

The survey area was established according to the known distribution of SH bears during the ice-free season (Prevett and Kolenosky 1982, Obbard and Middel 2012). This area is large, topographically and vegetatively diverse, and has high variability in polar bear

[3] A harvest limit of 23 bears within the Nunavik Marine Region was established for Nunavik Inuit, with at least one tag allocated to the Cree of Eeyou Istchee for harvest within the Inuit-Cree overlap area.

[4] A temporary voluntary limit of 5 bears was established for the six coastal Cree Nations of Ontario (including subsistence hunting and defense kills) for the 2011/12 harvest season. Not all Ontario communities were included in discussion about this voluntary limit.

density. It spans large portions of the northern Ontario and northern Québec coasts and inland areas, with the islands of James Bay and Hudson Bay being part of the Territory of Nunavut (Fig. 1 and 2). The Ontario portions of the subpopulation are part of the Hudson Bay lowlands ecosystem, consisting of large wetland complexes, extensive treed areas and tundra along the coast of Hudson Bay (Fig. 3). This area has little topographic relief and the coastal portions include extensive tidal flats (Fig. 3). The Québec portion of the study area consists of a series of long and steep rocky nearshore islands forming the Nastapoka Island complex as well as a relatively flat and hilly shrub tundra shoreline. The subpopulation also includes a large number of islands in James and Hudson bays, including the large Akimiski Island, the Twin Islands and the Ottawa islands complex that are known to be used extensively by polar bears during the ice free season. Southeastern Hudson Bay also holds the Belcher islands archipelago spreading over almost 3000 km². There are numerous Cree and Inuit communities along the Ontario and Québec coast and one Inuit community on the Belcher Islands.

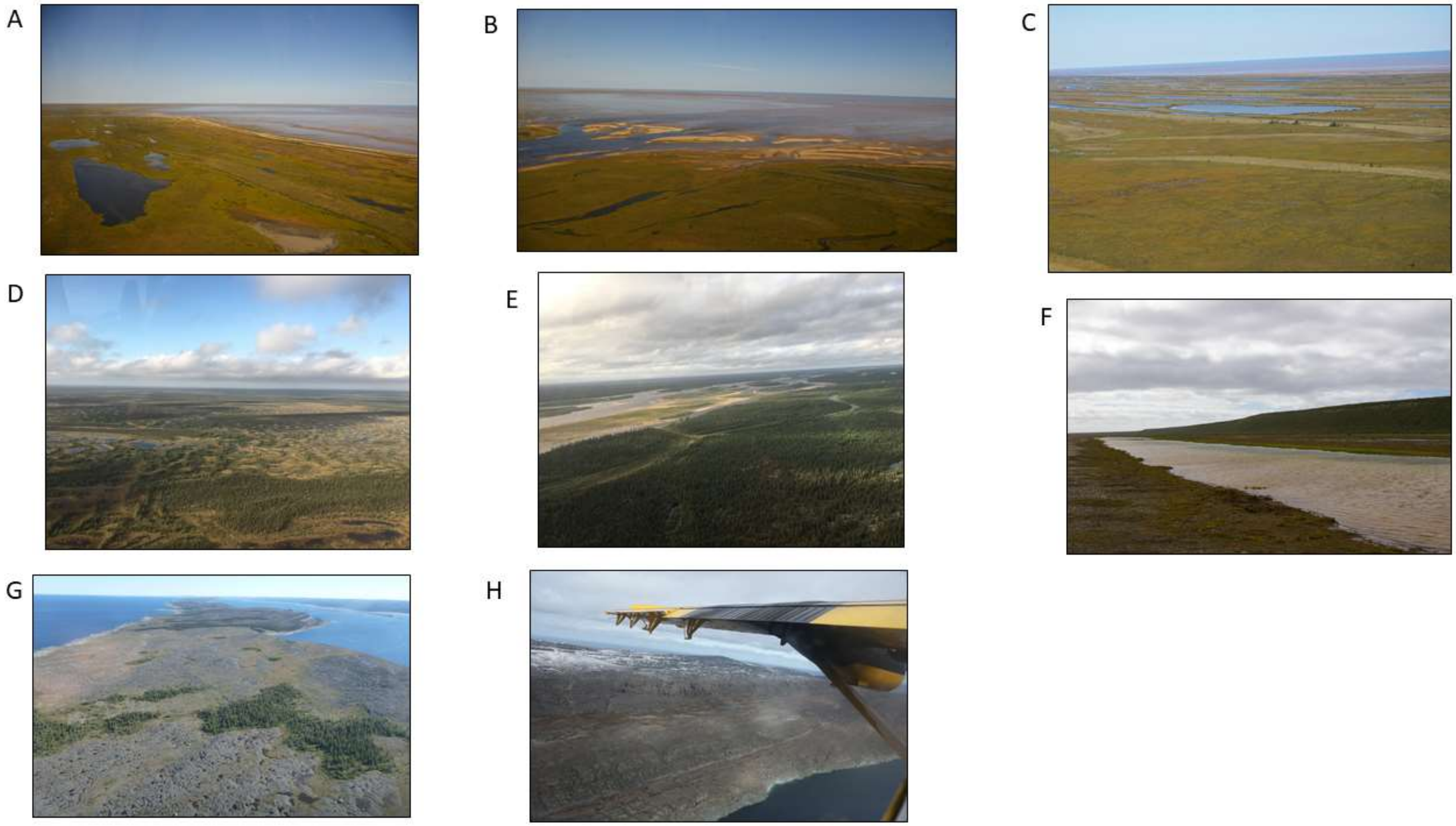


Figure 3. Representative photos of the vegetation and topography of the SH subpopulation. (A) The majority of the Hudson Bay coastline in Ontario consists of open tundra with interspersed wetlands and dry beach ridges. (B) There are extensive mudflats throughout the entirety of the Ontario coastal area. (C) Further inland from the Hudson Bay coast of Ontario is a mix of dry beach ridges, open tundra and wetlands. (D) Further inland from the Hudson Bay coast of Ontario and throughout most of the inland areas of James Bay there are interspersed treed areas, palsas and wetlands. (E) eventually, these areas give way to extensive treed areas and large riverine systems. (F) The islands of James Bay contain substantially more topography than the mainland Ontario portion of the study area. Shown here is North Twin Island. (G) The Québec coastline of James Bay is likewise more topographically diverse and consists of numerous small rocky islands. (H) Hudson Bay has numerous rocky islands where bears summer. Shown here is a portion of the Belcher Islands.

Survey design

We followed the survey design implemented in 2011/12 and 2016 (Obbard et al. 2015, Obbard et al. 2018) to provide a comparable population estimate. The 2011 and 2012 surveys were designed based on scientific information on the distribution of bears in SH during the ice-free season and information obtained from consultation with Indigenous communities in the region. Following the 2012 survey, a second round of consultation was conducted in Québec to address points raised by Inuit communities and Makivik Corporation. This resulted in the addition of a series of inland surveys perpendicular to the Québec coast along with a few additional islands in James Bay to the design of the 2016 survey to fully represent the scientific and Inuit knowledge of bear distribution in the area during the ice-free season. The surveys leverage the fact that Hudson Bay is entirely ice-free from approximately early August to late November each year during which time bears in SH are onshore. Further, females do not enter dens until October and November (Middel 2014), thus, between mid-August and the end of September, all bears are accessible (onshore) and available to be surveyed. We surveyed the subpopulation during this time and as close as possible to a similar survey being conducted in adjacent WH aimed to mirror the 2011 and 2016 WH surveys (Atkinson et al. 2022). As in past surveys (Obbard et al. 2015, Obbard et al. 2018), we subdivided the study area into regions based on expected bear density, aircraft type and survey design (Fig. 4). Past research has shown that the majority of bears in this subpopulation spend the ice-free season on the Ontario mainland, with a at least 10% of the population also inhabiting the islands of James Bay and eastern Hudson Bay (Obbard et al. 2015, Obbard et al. 2018). Although bears are regularly observed during winter along the Québec coast of Hudson Bay, bears are rare in that part of their range during the summer and are mostly sighted on Long Island and the Cape Jones area (Nunavik Marine Region Wildlife Board [NMRWB] 2018). This was also confirmed by the surveys in 2012 and 2016, which failed to observe any bears along the Québec coastline or inshore (Obbard et al. 2015, Obbard et al. 2018). Thus, we divided the study area into 1) the Ontario mainland, coastline, and Akimiski Island, located in James Bay, 2) the James Bay and Hudson Bay islands, excluding Akimiski Island, 3) nearshore islands off the Ontario coast and 4) the Québec coastline and nearshore islands (Fig. 4). Note that below, we aimed to refer to these areas exactly as they are listed above whenever mentioned to reduce confusion due to the complex nature of the study design.

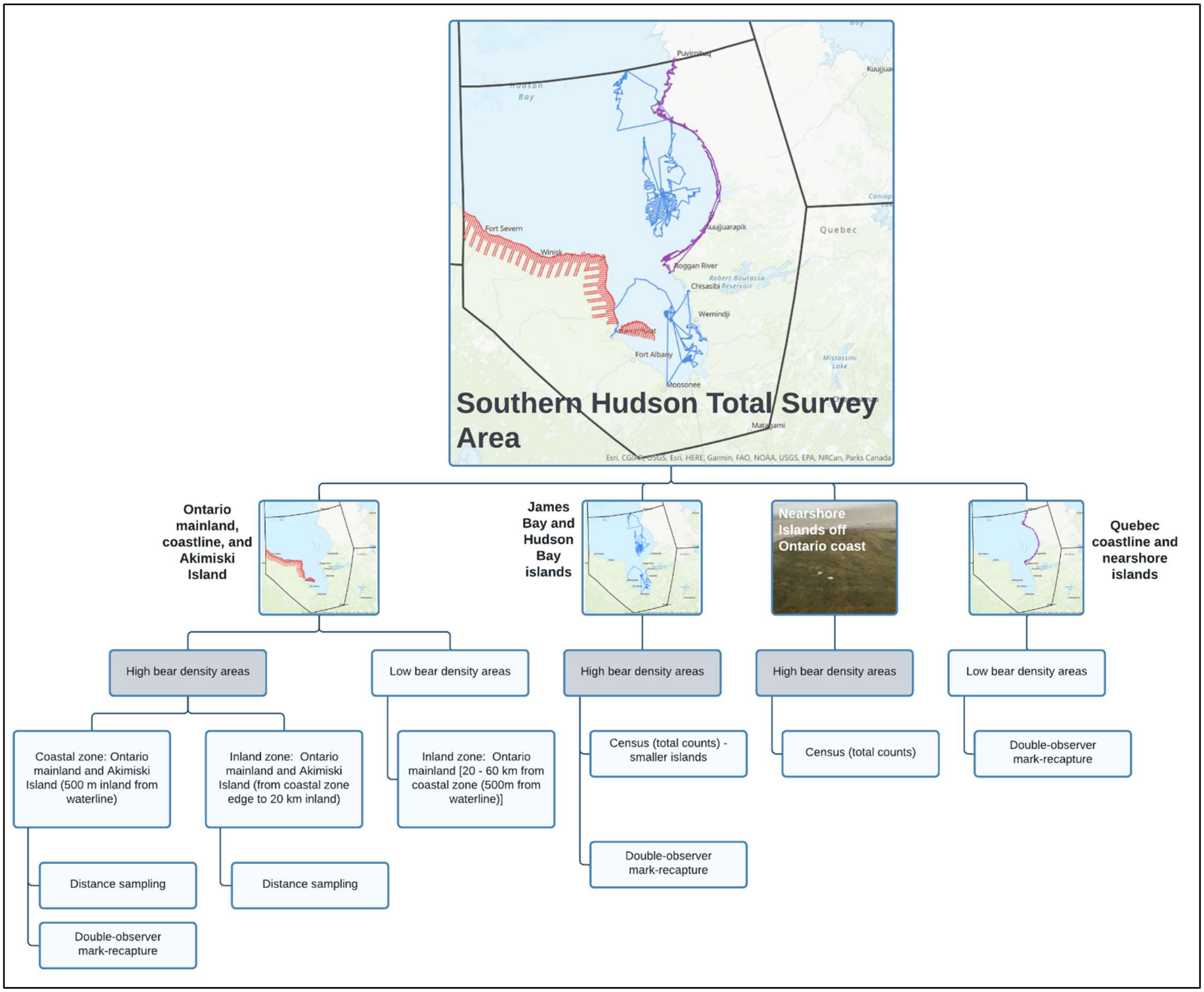


Figure 4. Schematic outlining the different survey areas, designs and analytical techniques used in SH polar bear survey in 2021.

Ontario mainland, coastline, and Akimiski Island

Most of the bears within the SH subpopulation summer on the Ontario mainland, with the majority of these bears concentrated along the coast (Kolenosky et al. 1992, Obbard and Middel 2012, Middel 2014, Obbard et al. 2015, Obbard et al. 2018). However, bears are also regularly documented far inland. Akimiski Island historically has held a high density of bears (Obbard et al. 2007), is only a short distance from mainland Ontario and is reachable via single-engine helicopter. Thus, it was surveyed in an identical manner to the Ontario mainland. We subdivided the Ontario mainland, coastline and Akimiski Island into 2 strata (Fig. 5). We designated areas from 20 km inland out to the waterline, including exposed mudflats, and the entirety of Akimiski Island as the high-density stratum. We designated all areas between 20 km and 60 km inland as the low-density stratum. Although bears have been documented further than 60 km inland (Kolenosky et al. 1992, Lemelin et al. 2010), such occurrences appear to be relatively rare, and the timing of the survey was such that pregnant females would not yet have entered their dens, which can occur far inland. Once the high-density stratum area was delineated, we further subdivided it into a coastal zone and inland zone (Figs. 4 and 6). The coastal zone consisted of all areas 500 m inland from the approximate high-tide line out to the waterline. Depending on when these areas were flown relative to high tide, this coastal zone could consist of large expanses of mud flats and numerous spits. The inland zone of the high-density stratum was all areas from 500 m inland from the approximate high-tide line to 20 km inland.

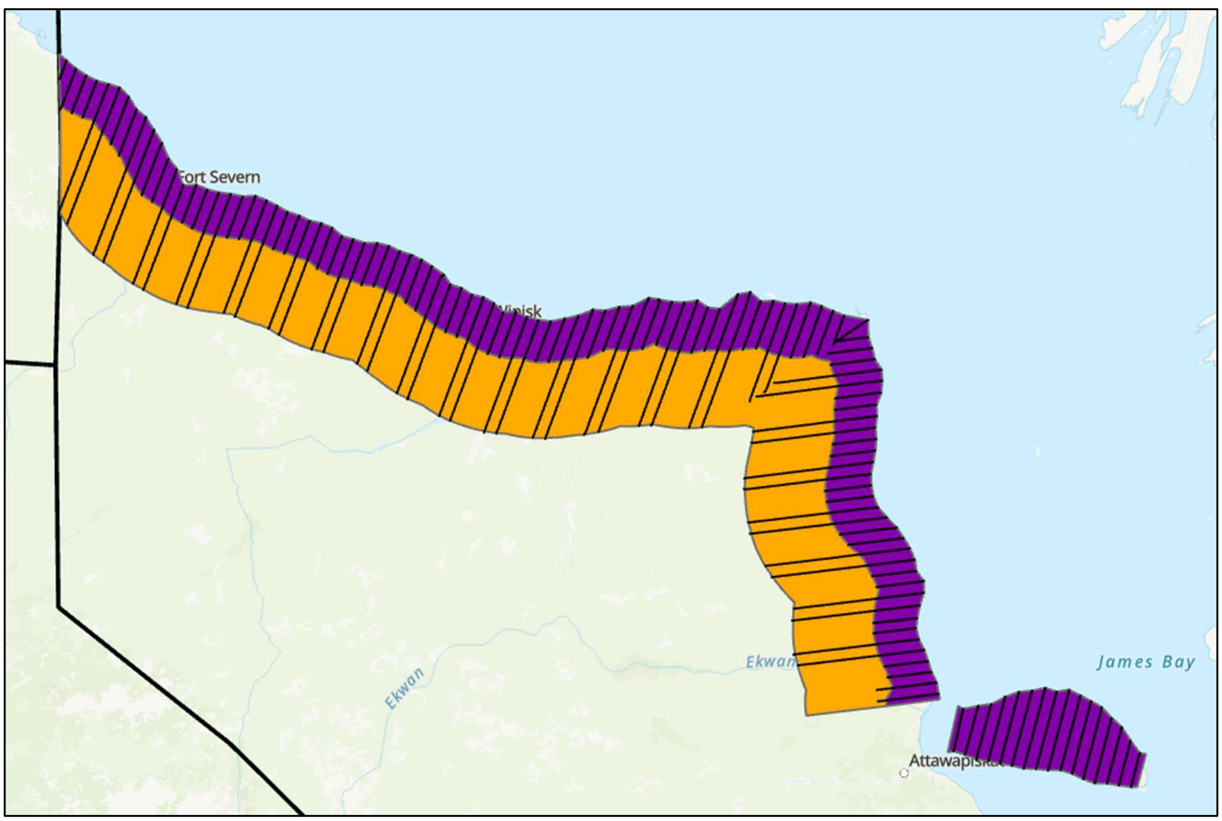


Figure 5. Flight lines (black lines) and stratum delineation for distance sampling survey of Ontario mainland, coastline and Akimiski Island. Purple shading represents the high-density stratum, consisting of all areas of mainland Ontario within 20 km of the waterline

as well as the entirety of Akimiski Island. Orange shading represents the low-density stratum, consisting of all areas between 20 and 60 km from the waterline.

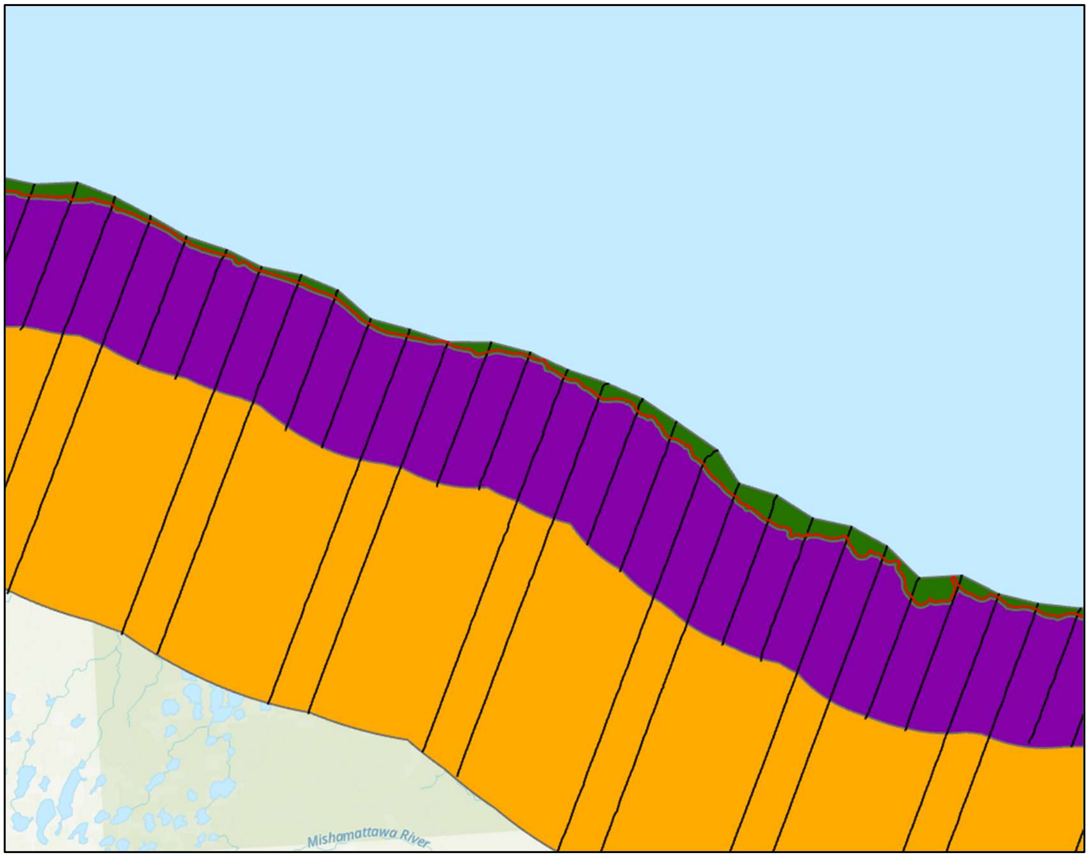


Figure 6. Close-up example of the delineation of the Ontario mainland, coastline and Akimiski Island area into different strata and survey approaches. Purple shading represents the inland zone of the high-density stratum, consisting of all areas of mainland Ontario between 20 km and 500 m from the approximate high-tide line, and the entirety of Akimiski Island further than 500 m from the approximate high-tide line. The green shading represents the coastal zone of the high-density stratum, consisting of all areas from 500 m inland from the high-tide line to the waterline. Orange shading represents the low-density stratum, consisting of all areas between 20 and 60 km from the approximate high-tide line. Red line represents the flight line for the double-observer mark-recapture portion of the survey.

Based on the above, the Ontario mainland, coastline and Akimiski Island area consisted of 3 sub-areas: 1) the coastal zone of the high-density stratum, 2) the inland zone of the high-density stratum, 3) the low-density stratum (Fig. 4 and 6). We employed two different survey techniques within these areas to address the strong variation in bear density among them. First, we employed a mark-recapture distance sampling survey covering the entirety of both the low and high-density stratum (i.e., both the inland and coastal zones in the high-density stratum). Following past surveys (Obbard et al. 2015, Obbard et al. 2018), transects were spaced 6 km apart across the entire high-density stratum including Akimiski Island (Fig. 5). Every other pair of transects was extended into the low-density stratum such that the low-density stratum was flown using pairs of transects spaced 6 km apart with the pairs separated by 18 km (Figs. 5 and 6). When present, these transects were extended out over exposed mudflats. If transects coincided with the small nearshore islands (see below) known to hold large numbers of bears, they were truncated at these islands to exclude the islands from our distance sampling estimate because these were surveyed separately as described below in section: *Nearshore islands off Ontario coast.*

For all three survey areas of the Ontario mainland, coastline and Akimiski Island, we employed distance sampling, flying transects in a Eurocopter EC-130 helicopter at an altitude of 120 m above ground level (AGL) and a speed of 160 km/h between August 22 and September 1, 2021. The crew consisted of a pilot, navigator (front right side of helicopter) and two rear observers positioned behind the pilot and navigator. All four, including the pilot, scanned for bears. Throughout the survey, the same pilot and observers participated, and all maintained the same position in the helicopter. We erected an opaque barrier between the front and rear of the helicopter to ensure rear observers were not alerted to the presence of a bear by the movements of the front observers. Further, observers allowed sufficient time from first detection of a bear for the other observers to have detected it. Once sufficient time had elapsed, it was determined whether the front observer, rear observer or both had detected the bear. We then flew to the approximate location of where the bear was first spotted and recorded a GPS location for calculating distance from the transect line. We recorded the position of who had observed the bear (pilot only, navigator only, back right only, back left only, both observers on the left or both observers on the right), the age class and sex of the bear (adult male, lone adult female, subadult, female with cubs of the year, female with yearlings), the group size, including all dependent offspring, the body condition on a 5 point scale (5 obese, 4 above average, 3 average, 2 below average and 1 emaciated), the activity of the bear when first spotted, the general habitat where the bear was first seen (e.g., mudflat or forest), a 3 point subjective scale for visibility, the general weather, vegetation height and density surrounding the bear, each on a 3 point scale, the degree to which glare from the sun was impacting visibility on a subjective 3 point scale and lastly, whether the bear was positioned relative to the helicopter such that it was unavailable to be observed by the rear observers (i.e., was in the rear observers' "blind-spot"). The availability of the bear to be observed by rear observers was reduced for bears near the transect line, but the exact distance varied depending on the orientation of the helicopter. In crosswind conditions, the helicopter often was "crabbing" and not oriented in the same direction as the transect line (Fig. 7). If another bear was observed while collecting covariate information off the transect line, it was not included in detections as it was assumed to have not been detected from the transect line.

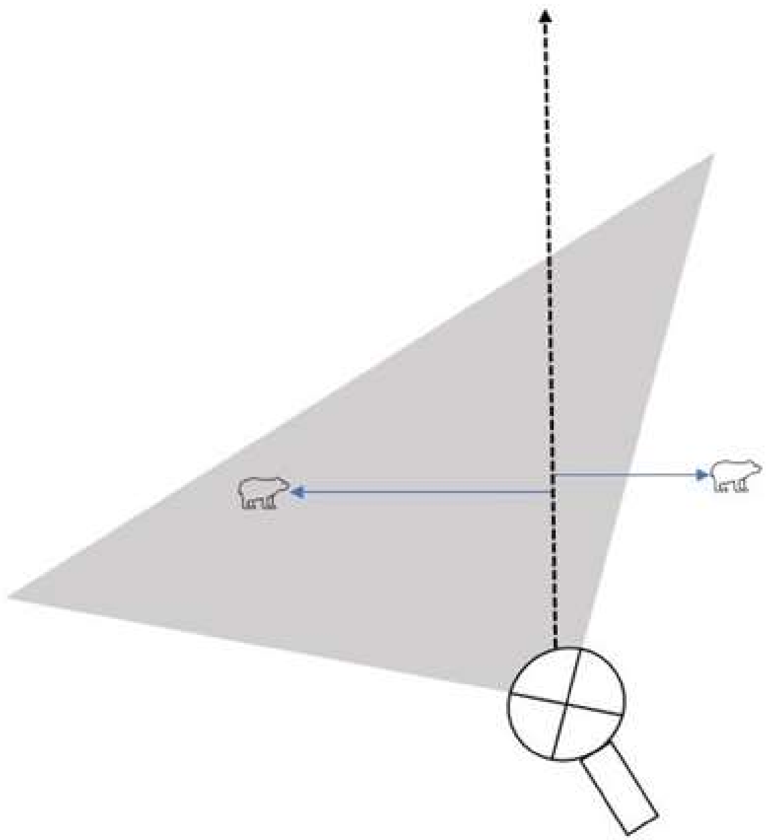


Figure 7. Schematic showing the influence of the orientation of the helicopter relative to the flight line on the ability of rear observers to observe bears on and close to the transect line. In this schematic, the dashed line represents the transect line and the gray polygon the blind-spot for rear observers. In this example, because the helicopter was oriented at an angle relative to the transect line, bears would be observable closer to the transect line for the right rear observer than the left rear observer.

In addition to the distance sampling survey, we also conducted a double-observer mark-recapture survey covering the coastal zone of the high-density stratum (i.e., the area within 500 m of the high-tide line extending out to the water line). We flew parallel to the coast at the approximate high-tide line and recorded detections of bears within 500 m inland and out to the waterline, including exposed mudflats. Observer setup within the helicopter, flight speeds, and recorded covariates were as described above. The use of both mark-recapture distance sampling and mark-recapture survey methodologies results in the coastal zone being sampled twice: once during the mark-recapture survey where we flew parallel to the coast and once during mark-recapture distance sampling where transects were flown perpendicular to the coast. Use of both surveys to obtain an averaged estimate (Obbard et al. 2015, Obbard et al. 2018) makes the assumption that bear position within the coastal zone is constant. Although movement of bears due to the helicopter generally appears only slight, the coastal zone is narrow and thus the estimate would be subject to fluctuation from bears moving into or out of the zone due to the helicopter. Thus, we attempted to fly the coastal zone mark-recapture survey on the same day, but prior to the overlapping distance sampling transects. Because the coastal zone is part of the high-density stratum, which extended an additional 19.5 km inland from the edge of the coastal zone, slight movements into or out of the coastal zone do not affect our distance sampling estimate. A large number of bears would need to move >20 km in a short period of time in response to the helicopter for bias to occur.

James Bay and Hudson Bay offshore islands

The James Bay and Hudson Bay Islands were considered high bear density areas and surveyed between September 2nd and September 10th, using double-observer mark-recapture from a de Havilland DHC-6 Twin Otter airplane. The coverage was identical to the area surveyed in the 2016 study. We flew at an average altitude of 150 m AGL and at a target speed of 150 km/h. The shape, size, and topography of the islands in James and Hudson Bays required variable flight patterns to ensure comprehensive coverage. We surveyed the Belcher Islands complex in Hudson Bay, which is the largest group of islands, using transects spaced 5 km apart and running perpendicular to the coast. All other islands in James and Hudson Bays were flown in a way to ensure complete coverage of the islands. The survey crew included one pilot and one data recorder in the front seats of the airplane and four active observers positioned in the rear of the airplane (two on the left and two on the right). We again erected an opaque barrier between the front and rear observers positioned in the rear of the airplane and conducted the survey identically to the mark-recapture protocol outlined above for the coastal zone of the high-density stratum, except that we did not fly over each individual animal to obtain a GPS location as the distance from the flight line was not of interest. In this survey, the pilot and data recorder only indicated that they had detected a bear if it was directly on the flight line and thus unavailable to the observers in the rear of the aircraft.

Nearshore islands off Ontario coast

Along the coast of Ontario, there are a few small islands that are known to have large numbers of bears. Survey methods of distance sampling or mark-recapture are not well suited due to the small area of the islands and high bear density. Thus, these islands were surveyed separately using a total count methodology. They were comprehensively flown with the observer setup outlined above and bears were censused on them.

Québec coastline and nearshore islands

The survey of the Québec coastline and nearshore islands was similar to the 2012 survey (Obbard et al. 2015) and was limited to the coastline and nearshore islands. Considering the absence of polar bears observed during the 2016 survey within the 20 km inland portion of the survey (Obbard et al. 2018), consultations were conducted with the three Nunavik communities (Fig. 2) to review important areas where polar bears might be observed during late summer. All communities agreed that very few bears were present inland during that time of the year but one additional coastal area, south of Cape Jones down to the mouth of Seal River, was recommended to be surveyed and was added to the survey plan (MFFP, Unpublished). The Québec coastline and nearshore islands were surveyed using an A-Star 350 B2, from August 23rd to 27th. A single transect was flown along the coastline, flying at an altitude of approximately 150 m AGL at a ground speed of 150 km/h. All nearshore islands were surveyed in a way to ensure total coverage. The crew consisted of a pilot and navigator in the front of the helicopter and two rear observers positioned behind the pilot and navigator, with an opaque divider between the front and back in order to apply the double-observer mark-recapture methodology as described above for the surveying of the coastal zone of the high-density stratum in the Ontario mainland, coastline and Akimiski Island area.

Statistical analysis of Ontario mainland, coastline and Akimiski Island distance sampling surveys

A schematic outlining how each survey and area was analyzed is shown in Figure 8. The Ontario mainland, coastline and Akimiski Island distance sampling survey was analyzed using both (1) conventional distance sampling models with covariates (multiple covariate distance sampling [MCDS]; Marques and Buckland 2003, Marques and Buckland 2004), following the analysis of Obbard et al. (2018) as closely as possible to facilitate comparisons, and (2) mark-recapture distance sampling models (MRDS; Borchers et al. 1998, Laake and Borchers 2004) to allow modelling of imperfect detection on the transect line. MCDS models assume perfect detection of bears on the transect line and underestimate abundance if this assumption is violated (Buckland et al. 2001). MRDS models include a mark-recapture sub-model to estimate probability of detection on the line thereby avoiding the assumption of perfect detection anywhere (Borchers et al. 1998, Laake and Borchers 2004). Groups of bears, rather than individuals, were treated as the unit of observation. Estimates of group abundance were multiplied by the mean group size to convert to estimates of animal abundance. We conducted replicate MCDS and MRDS analyses including and excluding data from the coastal zone. Both types of models were implemented in the ‘mrds’ R package version 2.2.6 (Miller et al. 2019, Laake et al. 2022).

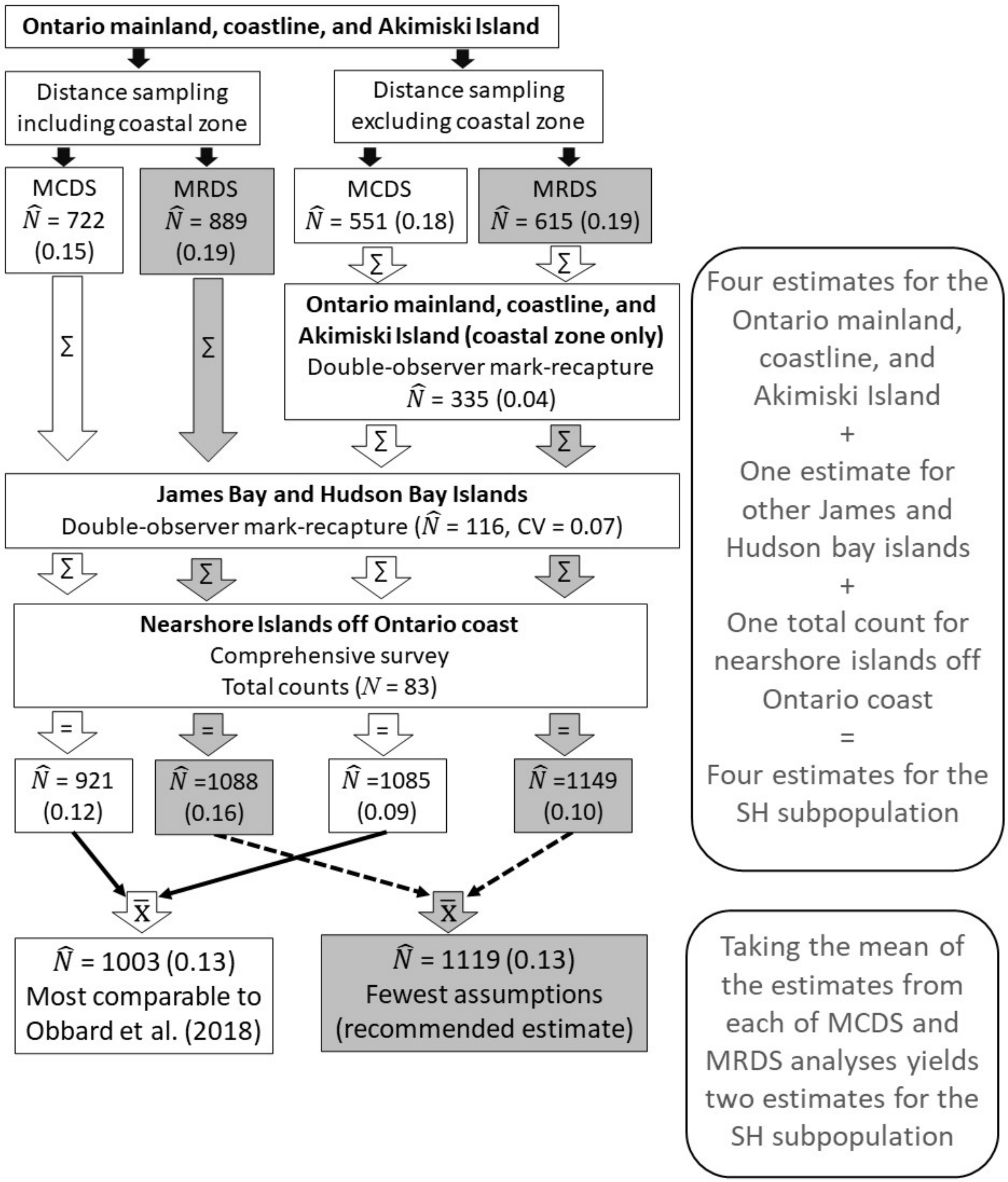


Figure 8: Schematic describing statistical analyses of data collected from different geographic areas and survey types. Geographic areas appear in bold and match those described under "survey design" above. ∑ indicates summation of estimates across different geographic areas, $\bar{x}$ indicates the mean across different estimates for the same geographic area. MCDS and MRDS refers to multiple covariate distance sampling and

mark-recapture distance sampling analyses, respectively. Gray boxes and arrows indicate estimates derived using MRDS for the Ontario mainland, coastline and Akimiski Island area, while white arrows and boxes indicate estimates derived using MCDS for the Ontario mainland, coastline and Akimiski Island area. Note that because no bears were observed in the Québec coastline and nearshore islands portion of the study, that geographic region is not shown in the schematic.

For the MCDS analyses we right-truncated the data at 1750 m following Obbard et al. (2018) after verifying that distance sampling models fit the truncated data adequately (tests described below) and that abundances estimated from simple models were not sensitive to right-truncation distance. We initially considered unadjusted half-normal and hazard rate forms of the detection function as well as a uniform model with a cosine adjustment of order 1. Uniform models fit the data poorly or failed to converge so were not considered further. Potential covariates of the detection function included visibility, vegetation height, and vegetation density to match the analysis of Obbard et al. (2018). Covariates were evaluated using forward stepwise model selection where only covariates that reduced Akaike's Information Criterion (AIC; Burnham and Anderson 2002) were retained; vegetation height and density covariates were correlated so were not included in the same model. We checked whether adjustment terms (cosine of order 1 for the half-normal model, and simple polynomial of order 4 for the hazard rate model) improved the fit of the AIC-minimizing covariate models. We tested for significant (α = 0.05) lack of fit using the X^2 goodness-of-fit test for binned distance data (Buckland et al. 2001, pp 69-71) and the distance sampling Cramér-von Mises test (Buckland et al. 2004, pp 388-389). The AIC-minimizing covariate model was selected for estimation (conditional on adequate fit), and final estimates were obtained by model averaging abundance estimates (as the AIC-weighted average abundance; Burnham and Anderson 2002) across hazard rate and half-normal models with the same covariate(s).

Data from the Ontario mainland and Akimiski Island distance sampling survey were also analyzed using MRDS models formulated for independent observers (Laake and Borchers 2004, Burt et al. 2014). Models with point independence rather than full independence were expected to be more appropriate for our data because the difference between front and rear observers' ability to see bears near the transects ensured that the correlation between detections from different observer positions increased with distance from the transect (Burt et al. 2014). We verified that simple point independence models reduced AIC relative to simple full independence models and used point independence models thereafter. We right-truncated at 2000 m because visibility was generally good in 2021 and exploratory analyses including goodness-of-fit testing indicated that this truncation distance provided a slightly better fit to simple DS models. We included distance as a covariate in all mark-recapture submodels (Buckland et al. 1993, Burt et al.

2014). We also considered a dummy covariate specific to the rear observers at short distances to account for their reduced probability of detecting groups of bears near the transect line (Wiig et al. 2022). The largest distance at which a group was recorded as unavailable to rear observers was 204 m, so all groups detected at this and shorter distances received a 1 for this "blind-spot" covariate. Other potential covariates of the MR submodel were group size, observer position (front or rear), side, the interaction between position and side, visibility, vegetation height, vegetation density, glare, and stratum (high or low density). Because vegetation height and density were correlated but describe potentially different effects of vegetation on observers' ability to detect bears, we also evaluated a combined vegetation covariate (Table 1); only one vegetation covariate was included in any submodel. Potential covariates of the distance sampling submodel included group size, side, visibility, vegetation height, vegetation density, the combined vegetation height and density covariate, glare, and stratum. After exploratory analyses we excluded the "activity" covariate because estimated effects were weak and indicated that stationary bears were more likely to be detected, including at longer distances, than moving bears.

Table 1. Definition of vegetation covariate representing the combination of vegetation height and density. The vegetation height covariate was recorded in the field on a 3-point scale with a height of 1 indicating vegetation was <1 m, 2 indicating 1-3m and 3 indicating >3 m. The vegetation density covariate was recorded in the field on a 3-point scale with a density of 1 indicating sparse vegetation, 2, indicating moderate and 3 dense.

Vegetation height	Vegetation density	Combined vegetation covariate
1 or 2	1	1
1 or 2	2	2
1 or 2	3	not present in data
3	1	2
3	2	3
3	3	4

We evaluated support for forms of the detection function (unadjusted half-normal or hazard rate) and covariates using a forward stepwise model selection procedure intended to avoid overfitting and the inclusion of uninformative covariates in estimating models. Covariates that increased AIC relative to a simpler model without that covariate were excluded, covariates that reduced AIC were retained but if the reduction was < 2.0 we also considered parameter-reduced models excluding those covariates. This approach differed slightly from the above analysis because here we considered more covariates and thus needed to evaluate more combinations of covariates. Thus, we required a larger reduction in AIC to avoid evaluating a cumbersome number of models. An exception to this procedure was that, following Northrup and Howe (2019), we considered a model with main effects of side and position and their two-way interaction in all mark-recapture submodels even if side and position were not supported as main effects alone. We conducted model selection in 3 steps. First, we held the distance sampling model constant as the unadjusted half-normal model with no covariates and evaluated covariates of the mark-recapture model. Next, we evaluated forms and covariates of the distance sampling model while holding the mark-recapture model constant at the AIC-minimizing model. Lastly, we created a set of models that was comprised of all combinations of the supported (ΔAIC < 2) mark-recapture and distance sampling submodels. We checked whether the adjustment terms described above for MCDS models improved the fit of the AIC-minimizing distance sampling submodels. Before estimating abundance we checked for significant (α = 0.05) lack of fit using X^2 tests across distance intervals for both the mark-recapture and distance sampling submodels, the total X^2 value across submodels, and the Cramér-von Mises test. Final MRDS estimates of abundance were obtained by model averaging across models with supported covariates and parameter-reduced models in the case of weakly-supported (ΔAIC < 2) covariates.

In both the MCDS and MRDS analyses, the variance of the abundance of individual bears combined three components of variance using the delta method (Buckland et al. 2001, Miller et al. 2019): the empirical variance of the encounter rate among transects (here estimated using Fewster et al. 's [2009] estimator "S2" for systematic designs), the variance of detection probability obtained from the fitted model estimated using standard maximum likelihood methods, and the variance of group size. Where estimates were calculated by model averaging, model selection uncertainty also contributed to the variance of bear abundance (Burnham and Anderson 2002).

We post-stratified estimates of abundance by age-sex category (adult females, adult males, subadults, yearlings, and cubs) to obtain age-sex class specific estimates of abundance. This was achieved by combining the estimated probability of detecting clusters of bears (and its variance) from the AIC-minimizing model fit to data from all clusters with age-sex class specific group sizes.

Statistical analysis of double-observer mark-recapture surveys

The Ontario mainland, coastline and Akimiski Island coastal zone mark-recapture helicopter survey and the James Bay and Hudson Bay islands fixed-wing mark-recapture surveys were analyzed using mark-recapture models for closed populations (Huggins 1989) implemented in the ‘RMark’ R package version 2.2.7 (Laake 2013, Laake et al. 2019). We conducted separate analyses of data obtained from the helicopter survey and the combined fixed-wing surveys (Fig. 8). Potential covariates of detection probability included observer position (front or rear, modelled as distinct temporal sampling occasions), group size, visibility, vegetation height, vegetation density, and position of the group relative to the aircraft (left, right, or under, coded as "under" where the group was recorded as unavailable to the rear observer). We fixed detection probability by the rear observers to 0 for groups that passed "under" the aircraft. We evaluated support for covariates using the same forward stepwise procedure described above for the mark-recapture and distance sampling submodels of MRDS models, except that we used the small sample bias-corrected version of AIC (AIC*c*; Burnham and Anderson 2002) rather than AIC. We obtained final estimates of the number of groups of bears and its unconditional variance by model-averaging abundance estimates across models with supported covariates, and parameter-reduced models in cases of weakly supported (ΔAIC < 2) covariates. We estimated the number of individual bears by multiplying by mean group size and included the variance of group size in the variance of the number of bears using the delta method.

We did not detect any bears during the mainland Québec coastal and nearshore island survey. As such, no statistical analyses were applied.

Total abundance estimates

The above analyses produced four separate estimates of bear abundance in the Ontario mainland, coastline and Akimiski Island area (see also Fig. 8): 1) an MCDS estimate for the entirety of the area (i.e., the areas overlain by the green, orange and purple polygons in Fig. 6), 2) an MCDS estimate for the low-density stratum and the inland zone of the high-density stratum (i.e., excluding the coastal zone, so the orange and purple polygons in Fig. 6) plus the estimate of the number of bears in the coastal zone (the area in green in Fig. 6) from the double-observer mark-recapture analysis, 3) an MRDS estimate for the entirety of the area (i.e., the areas overlain by the green, orange and purple polygons in Fig. 6), and 4) an MRDS estimate for the low-density stratum and the inland zone of the high-density stratum (i.e., excluding the coastal zone, so the orange and purple polygons in Fig. 6) plus the estimate of the number of bears in the coastal zone (the area in green in Fig. 6) from the mark-recapture analysis. We added the estimated number of bears on the James Bay and Hudson Bay Islands, and the census number of bears on small nearshore islands off the Ontario coast, to each of the four final estimates for the Ontario mainland, coastline and Akimiski Island area to generate estimates for the SH subpopulation. Finally, we produced two final estimates of the SH subpopulation as the mean of two subpopulation-level estimates: those calculated from estimates 1 and 2 above for the Ontario mainland, coastline and Akimiski Island area, and those calculated from estimates 3 and 4 above (see Fig. 8). Unconditional variances around these estimates were calculated in a model averaging framework assigning the two estimates equal weight. We present log-normal confidence intervals around all estimates of bear abundance. All analyses were performed using R software version 4.2.0 (R Core Development Team 2022).

RESULTS

We detected 138 groups of bears on distance sampling transects on the Ontario mainland, coastline and Akimiski Island area, 88 excluding the coastal zone. Right-truncating at 1750 m for the MCDS analysis removed 9% of observations from both data sets, leaving 125 and 80 groups in data including and excluding the coastal zone, respectively. Right-truncating at 2000 m for the MRDS analysis removed 8% of observations from the complete data set and 7% of observations from data excluding the coastal zone, leaving 127 and 82 groups in data including and excluding the coastal zone, respectively.

In the MCDS analysis of the dataset including the coastal zone, the half-normal model without covariates minimized AIC. However, half-normal and hazard rate models with the vegetation density covariate had similar support with ΔAIC of 0.61 and 0.76, respectively (Table S1), so, for the sake of consistency with Obbard et al. (2018), we estimated abundance by model averaging across these two models (Table 2). Visibility was the only supported covariate in data excluding the coastal zone; half-normal and hazard rate models with this covariate had similar support, and all other models had ΔAIC > 2 (Table S2), so we estimated abundance by model averaging across these two models (Table 2). All MCDS models considered for estimation provided adequate fits to the data (*P*-values associated with the X^2 test for binned distance data and the Cramér-von Mises tests were all > 0.30). Adjustment terms did not improve fit to either data set.

Table 2. Abundance estimates ($\widehat{N}$), standard errors (SE), coefficients of variation (CV) and 95% confidence intervals from multiple covariate distance sampling (MCDS) and mark-recapture distance sampling (MRDS) analyses of polar bear data including or excluding the coastal zone of the high-density stratum for the Ontario mainland, coastline and Akimiski island area only.

Analysis type	Coastal zone	$\widehat{N}$	SE	CV	95% CI
MCDS	Included	722	111	0.15	535 – 974
MCDS	Excluded	551	99	0.18	388 – 781
MRDS	Included	889	170	0.19	613 – 1288
MRDS	Excluded	615	119	0.19	422 – 897

In the MRDS analysis of the complete data set, the blind-spot covariate, observer position, side, and glare were supported covariates of the mark-recapture submodel and the interaction between position and side and visibility were weakly supported (ΔAIC < 2 relative to simpler models) so additional models including and excluding these latter covariates were considered. Three submodels with all supported covariates and different combinations of weakly supported covariates had ΔAIC < 2 and were crossed with supported distance sampling submodels. Glare was supported as a covariate of the distance sampling submodel (Fig. 9). The combined vegetation covariate was also supported (Fig. 9), but ΔAIC was < 2 in the case of half-normal models so we considered models excluding it. Adjustment terms did not improve fit. Three submodels had ΔAIC < 2 and were crossed with the three supported mark-recapture models. All nine supported MRDS models (Table S3) fit the data adequately (*P*-values associated with the total X^2 value across distance sampling and mark-recapture submodels and the Cramér-von Mises tests were all > 0.65) and were included in model-averaged estimates of abundance (Table 2).

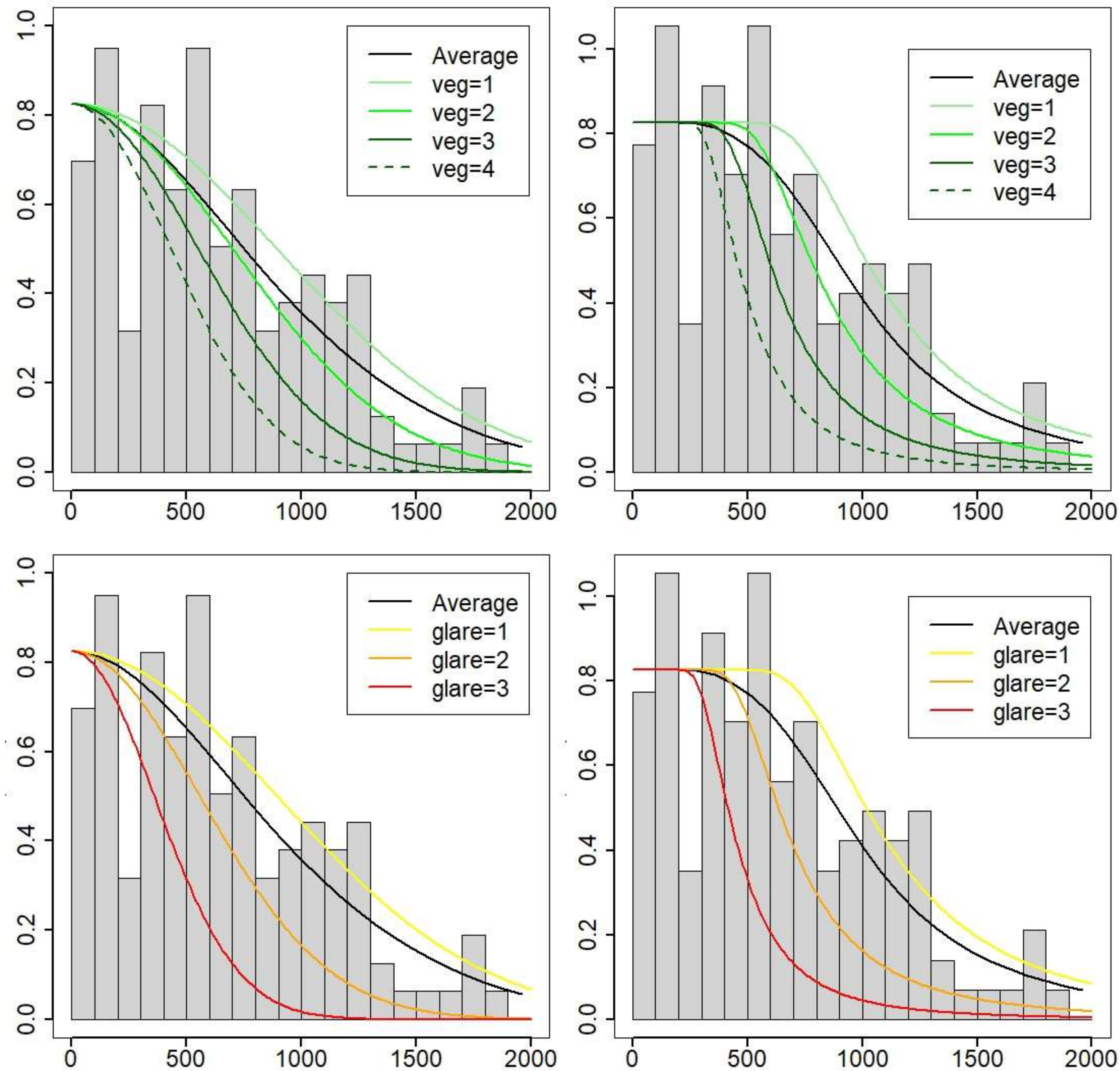


Figure 9. Half-normal (left column) and hazard rate (right column) detection functions estimated from the top two AIC-ranked mark-recapture distance sampling models fit to complete data from SH polar bears sighted from distance sampling transects in 2021 in the Ontario mainland, coastline and Akimiski island area, showing effects of supported covariates of the scale of the detection functions (the combined vegetation covariate and glare). Both models included the same covariates of both submodels; only key functions differed. The half-normal model ranked 1st and the hazard rate model had ΔAIC = 1.3. Top row shows the effect of the vegetation, bottom row shows the effect of glare. When plotting effects of one covariate, the other covariate was held constant at the mean value in the data. X-axes show distance from the transect in meters, y-axes show probability of detection.

When data from the coastal zone were excluded, the blind spot covariate, observer position, side, the interaction between position and side, visibility, and glare were supported covariates of the mark-recapture submodel in the MRDS analysis. However, models with the visibility or glare covariates exhibited lack of fit that was sometimes significant at α = 0.05 and always significant at α = 0.10 (P-values associated with the total X^2 value ranged from 0.03 – 0.08); furthermore, these models yielded unrealistically high estimates of abundance, suggesting data were insufficient to support this level of model complexity. We therefore combined only the mark-recapture submodel with the blind spot covariate, position, side, and the interaction between position and side with supported distance sampling models. All other submodels that fit well and yielded reasonable abundance estimates had ΔAIC > 2 relative to this submodel. Only visibility was supported as a covariate of the distance sampling submodel; it reduced AIC of the hazard rate model by < 2 so we retained models excluding it and combined four distance sampling submodels (half-normal and hazard rate with and without the visibility covariate) with the selected mark-recapture submodel (Table S4). Adjustment terms did not improve fit. All four of these models fit the data adequately and were included in model averaged estimates of abundance (Table 2). MCDS and MRDS estimates of abundance were sensitive to the form of the detection function (half-normal or hazard rate) and less sensitive to covariates.

Post-stratification by age-sex class suggests an adult sex ratio strongly skewed towards females (Table 3). Raw observations from the distance sampling survey showed a similar pattern, but raw observations from the coastal mark-recapture survey showed a strongly male biased sex ratio (Table 4). In total, we saw 148 family groups during the survey, including those seen while off transect or transiting. 75 of these were females with cubs of the year and 73 with yearlings. The average cub of the year litter size was 1.57 and the average yearling litter size was 1.47.

Table 3. Estimates of abundance ($\widehat{N}$), standard errors (SE), coefficients of variation (CV), lower 95% confidence limit (LCL), upper 95% confidence limit (UCL) and the mean proportion (Prop.) of the total estimate comprised of that sex and age class, obtained from post-stratification of MRDS model fit to distance sampling observation of polar bears in SH in 2021.

Age-sex class	$\widehat{N}$	SE	CV	LCL	UCL	Prop.
Adult female	366	70	0.19	251	533	0.40
Adult male	173	71	0.41	79	378	0.19
Subadult	59	21	0.36	30	118	0.06
Yearling	156	38	0.24	98	250	0.17
COY	167	52	0.31	91	305	0.18

Table 4. Proportions of observed animals falling into different sex and age classes for distance sampling and coastal mark-recapture surveys of polar bears in SH between 2011 and 2021.

	Year	Adult female	Adult male	Subadult	Yearling	COY
Distance sampling						
	2011	0.36	0.20	0.08	0.15	0.19
	2016	0.34	0.19	0.06	0.05	0.30
	2021	0.38	0.20	0.07	0.18	0.18
Coastal mark-recapture						
	2011	0.20	0.40	0.13	0.12	0.15
	2016	0.19	0.52	0.08	0.03	0.17
	2018	0.19	0.55	0.09	0.07	0.10
	2021	0.22	0.42	0.09	0.10	0.12

No covariates of detection probability were supported in mark-recapture analyses of data from the helicopter survey of the coastal zone. Probabilities of detection were high (0.87 from the null model) and estimates of abundance were similar across all models. Multiplying the estimated number of groups from the null model by mean group size (1.567; SE 0.063) yielded an estimate of 335 bears (SE 13.9, CV 0.04, 95% CI = 309 – 363). Side and group size were weakly supported covariates in the mark-recapture analysis of data from the fixed wing survey of the James and Hudson Bay Islands. Estimated probabilities of detection were again high (0.841 from the null model) and estimates of abundance were similar across models. Model averaging and multiplying by mean group size (1.455; SE 0.090) yielded an estimate of 116 bears (SE 7.93, CV 0.07, 95 % CI = 102 – 133).

Estimates of total abundance at the subpopulation level ranged from 921 to 1149 and were lower where we assumed perfect detection on the line during distance sampling surveys (Table 5).

Table 5. Estimates of subpopulation-wide abundance ($\widehat{N}$), standard errors (SE), coefficients of variation (CV), lower 95% confidence limit (LCL) and upper 95% confidence limit (UCL) for polar bears in the Southern Hudson Bay subpopulation. 6 estimates are presented representing either multiple covariate distance sampling (MCDS) or mark-recapture distance sampling (MRDS), excluding the coastal zone, including the coastal zone or averaging across these two approaches.

Estimate	Method and areas included	$\widehat{N}$	SE	CV	LCL	UCL
1	MCDS including coastal zone	921	111	0.121	727	1166
2	MCDS excluding coastal + coastal zone MR	1085	100	0.092	905	1300
3	Mean of 1 & 2	1003	134	0.134	773	1301
4	MRDS including coastal zone	1087	170	0.156	802	1474
5	MRDS excluding coastal + coastal zone MR	1149	120	0.105	937	1410
6	Mean of 4 & 5	1119	150	0.134	860	1454

Discussion

The number of polar bears present in the SH subpopulation at the time of the 2021 survey was substantially higher compared to the last comprehensive survey conducted in 2016. In 2016, the subpopulation estimate was 780 (95% confidence interval 590-1029; Obbard et al. 2018), which represented a 17% decline from 2011/12 when the subpopulation was estimated at 943 (95% confidence interval 658-1350; Obbard et al. 2015). In our current work, we produced two separate estimates, one (N = 1003 95% CI = 773-1302) that assumed perfect detection on the transect line as Obbard et al. (2018) did to allow for direct comparison and one (N = 1119 95% CI 860-1454) that took advantage of a novel approach to estimating the probability of detection on the transect line while accounting for the blind spot affecting rear observers (Wiig et al. 2022). The former estimate is most comparable to the 2016 estimate, but the latter is a more robust estimate of the true subpopulation size in 2021. Both estimates indicate a greater number of bears within this subpopulation than in 2016, with the former estimate suggesting a 29% increase in the number of bears found within the subpopulation in 2021 compared to 2016.

The greater number of bears in SH in 2021 compared to 2016 has two plausible biological drivers based on the results of this survey and other available lines of evidence, both of which may be at play to varying degrees: 1) annual variation in the on-land distribution of bears in SH and WH, and 2) an increase in population growth rate due to reduced mortality, increased birth rate or both. At the writing of this report, we do not have definitive evidence for either driver, but discuss the existing evidence for each of these in turn. First, it seems likely that there was some movement of bears into SH from the adjacent WH subpopulation in 2021. An increase of nearly 30% in 5 years seems highly implausible for a species such as polar bears that has a slow life history strategy. Further, the 2016 survey showed very few yearlings, and a survey of only the coastal area in 2018 found even fewer bears than in 2016 in this portion of the subpopulation. These findings suggest that an even greater rate of increase would have to have occurred between 2018 and 2021, making it highly unlikely that all of the increase from 2016 to 2021 was from greater reproductive output or reduced mortality alone. A simultaneous survey of WH (Atkinson et al. 2022) indicated a decline of 224 bears in WH from 2016 to 2021, which numerically is the same as the increase in the estimate of SH abundance from Obbard et al. (2018) and our 2021 survey. Further, genetic identification of individuals sampled through capture-recapture surveys conducted along the coast of SH and WH indicated that > 20% of the bears sampled in SH in 2021 had previously been sampled exclusively in WH (Environment and Climate Change Canada [ECCC] unpublished data). These joint lines of evidence suggest that there is variation in the annual on-land distribution of bears between SH and WH, with more of these bears in SH in 2021. Although the boundary between WH and SH, in northwestern Ontario, was based in part on movement and mark-recapture data, there is no major physiographic feature present and there are large aggregations of bears on offshore islands and peninsulas near the boundary. Thus, minor variation in the distribution of these bears could greatly shift the number of individuals present in WH or SH. Prevett and Kolenosky (1982) suggested that movements of large numbers of bears occurred between the southern Manitoba coast of Hudson Bay and Ontario, though this finding was not corroborated by Stirling et al. (2004) using surveys conducted earlier in the ice-free season. Derocher and Stirling (1990), focusing on the area of WH directly south of Churchill, MB likewise did not document movements between the two subpopulations, but did not cover the area of WH closest to SH where relatively minor annual variation in distribution could lead to large shifts in the number of bears present in each subpopulation. Further, collaring data from female bears shows generally high fidelity to onshore areas (Stirling et al. 2004, Obbard and Middel 2012). However, more recently, Cherry et al. (2013) showed that ice conditions were an important predictor of annual fidelity to onshore areas in WH. Specifically, they found that when there was greater ice later in the season in SH relative to WH, bears collared in WH tended to come ashore further from their collaring location. Further, they predicted greater declines in seasonal fidelity to onshore areas with continued sea-ice decline. The biopsy darting work (ECCC unpublished data), in combination with ongoing physical capture (ECCC, unpublished data) covered the coast of WH from the border between Manitoba and Nunavut to the WH-SH border, along with much of the SH coast and is the most comprehensive data available to date on individual movements; these data are more comprehensive in coverage than either Derocher and Stirling (1990) or Prevett and Kolenosky (1982) and use more effective methods for documenting annual movement of individuals of all sex and ages classes than does telemetry or aerial surveys (e.g., Stirling et al. 2004, Obbard and Middel 2012).

In contrast to the above evidence for annual variation in distribution of bears leading to the increase in SH, it is possible that this increase was influenced in part by improved demographic rates in SH. Several lines of evidence support that the decline in WH from 2016 to 2021 was at least partially driven by reduced reproduction. If this is the case, then the increase in SH could not be solely driven by distribution shift. First, reproduction and recruitment in WH appear to have been low throughout the last decade relative to SH and other polar bear subpopulations (Atkinson et al. 2022). Specifically, cubs of the year comprised 7%, 11% and 9% of observations in 2011, 2016 and 2021 in WH, while yearlings comprised 3%, 3% and 9% (Stapleton et al. 2014, Dyck et al. 2017, Atkinson et al. 2022). In comparison, cubs of the year comprised 16%, 19% and 18% of bears in SH in 2011, 2016 and 2021 and yearlings comprised 12%, 5% and 18% of observed bears (Obbard et al. 2015, Obbard et al. 2018). Further, physical mark-recapture in part of WH indicates there have been few yearlings during many of the last 10 years (ECCC unpublished data). These numbers alone suggest reproduction is substantially greater in SH than WH. WH also has seen strong evidence of changes in sex and age class ratios across the three surveys, with declines in adult females and sub-adults (Atkinson et al. 2022). Although we were unable to compare post-stratified sex and age class ratios as done in WH because these estimates were not produced in 2016 and 2011, our raw observations indicate quite consistent sex and age structure. Further, the proportion of the population in different sex and age classes estimated through post stratification was very similar to the proportions calculated from the observed data and, as such, we assume the observed proportions from the 2011 and 2016 surveys provide adequate comparisons. However, the number of yearlings in 2021 was high and indicates a rebound from the particularly low numbers seen in 2016 (Obbard et al. 2018). Annual variability in survival of COYs to yearlings is not surprising as autumn yearling litter sizes are highly variable (Derocher and Stirling 1995). We also note that the two years preceding 2021 were two of the three years with the longest duration of sea-ice since 2011 (Figs. 10 & 11). These conditions would have been favorable for high reproductive output and survival of cubs in the previous two years. Importantly, with continued warming, these conditions are unlikely to persist and we expect low recruitment in the coming years.

The above numbers suggest that in recent years, demography is different in WH and SH, with what appears to be lower reproduction and recruitment in WH. If this is the case, then the decline seen in WH by Atkinson et al. (2022) may not be all attributable to distribution shifts of bears to SH. Following, the increase in SH would have to be at least partially due to increased population growth rate. This potential is supported by the fact that ice conditions have generally been good over the last 5 years relative to the time period between 2011 and 2016 (Fig. 10) and that SH appears to have a high capacity for growth (Regehr et al. 2021). Further, polar bear harvest in SH was lower between 2016 and 2021 than between 2010 and 2015 (37.8 bears per year compared to 58.8 bears per year; https://www.polarbearscanada.ca/en/polar-bears-canada/canadas-polar-bear-subpopulations; accessed July 22, 2022). This decrease was in part driven by the exceptionally large harvest of 104 bears in the 2010/2011 harvest season, of which many were female. Such a large increase in annual harvest must have had downstream negative demographic effects due to the increased harvest of adult females, subsequently potentially depressing growth for a few years. Thus, it seems plausible that the high harvest in 2010/11 and higher average harvest early in the last decade, along with relatively poor ice years, could have driven a decline between 2011 and 2016. In contrast, a subsequent rebound to 2021 levels could be due to lower annual harvests with the resulting downstream positive demographic effects combined with better ice conditions that resulted in higher juvenile survival. However, we note again that a 29% increase over 5 years is highly unlikely for polar bears without distribution shift playing some role. Lastly, it is possible that the apparent increase in SH between 2016 and 2021 was simply sampling variance in one or both years, whereby the true difference in numbers between the surveys was exaggerated. We note that it is equally likely that the difference was underestimated, however.

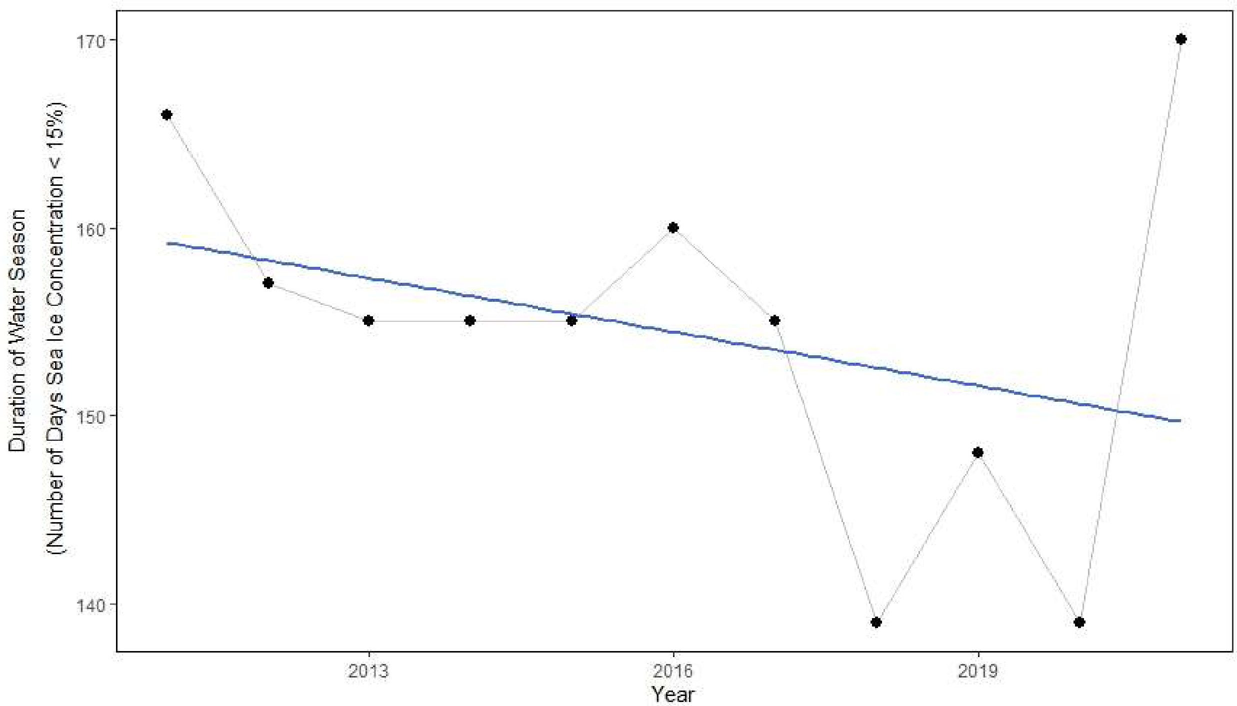
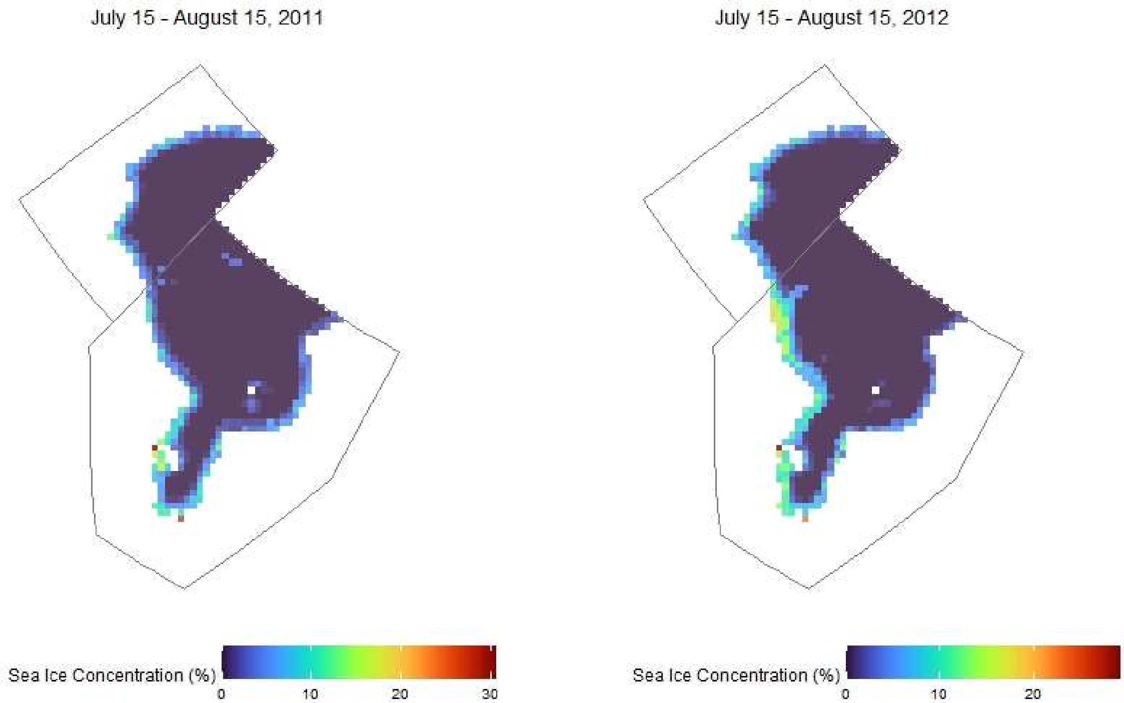
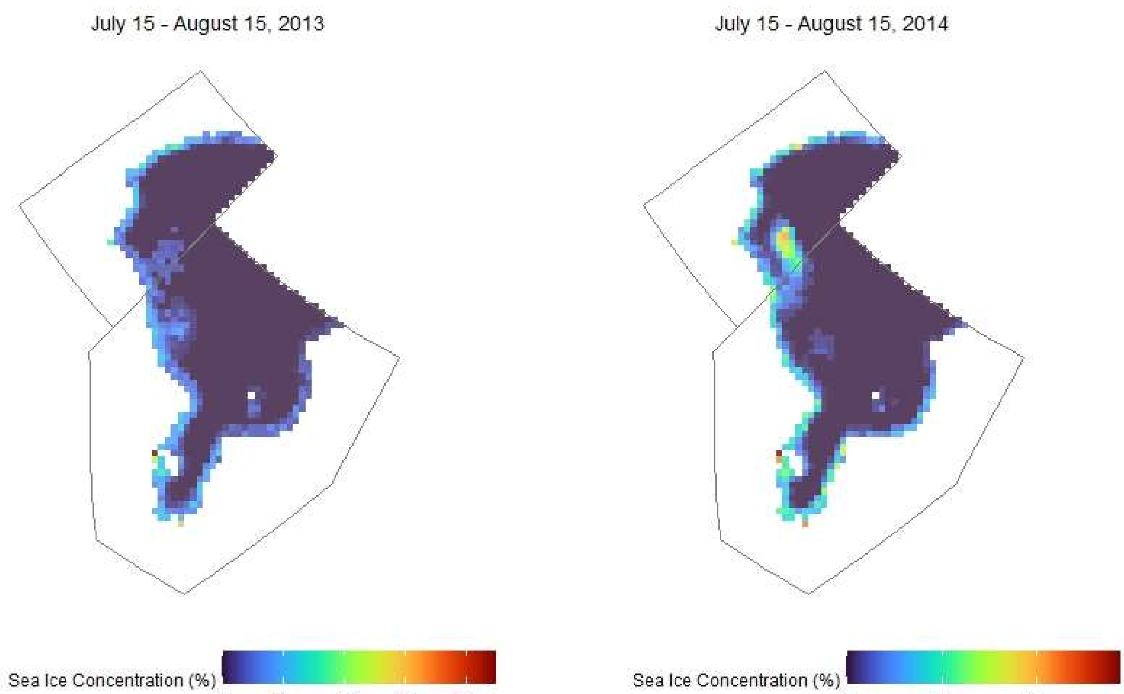
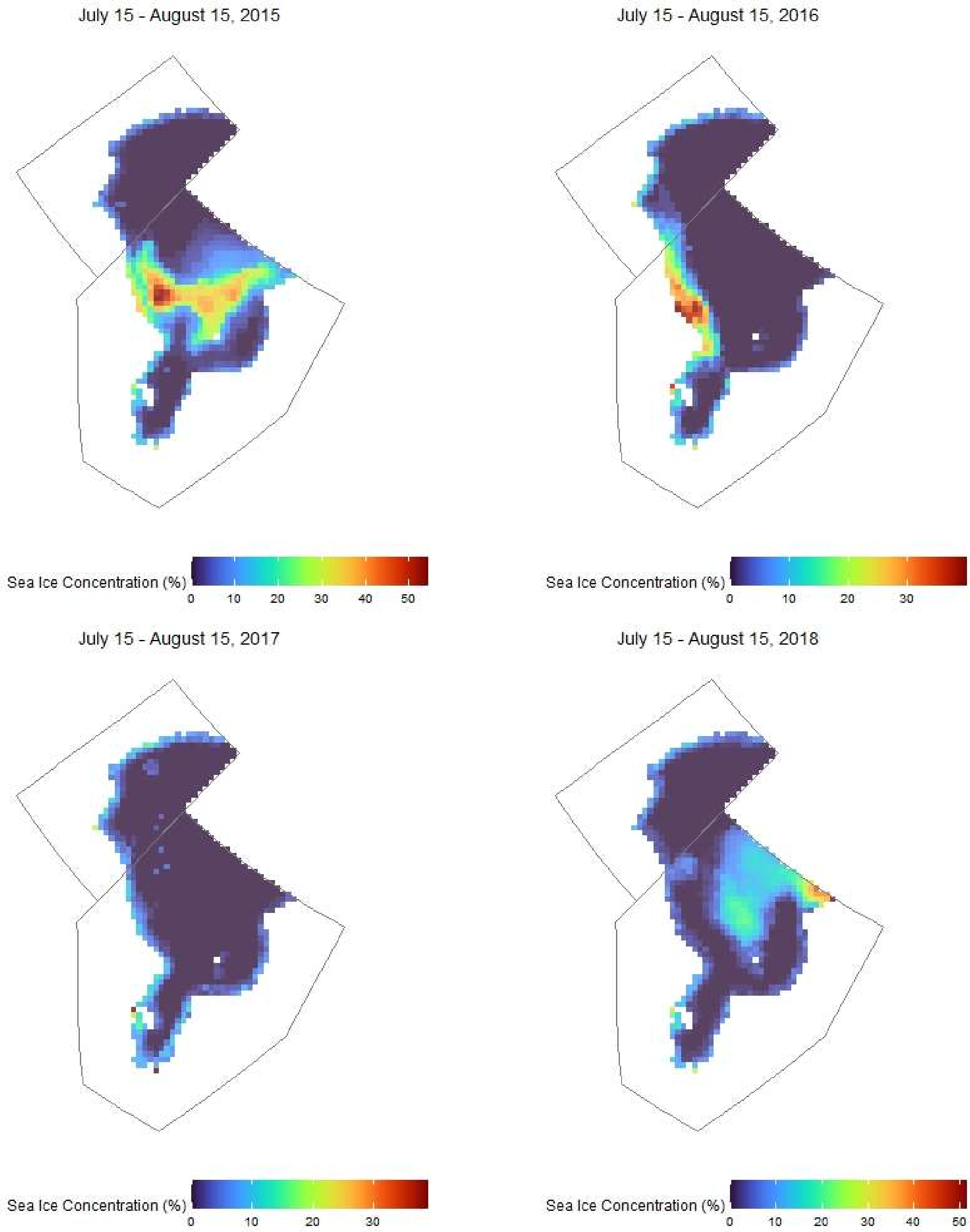


Figure 10. Duration of ice-free season in the combined Western and Southern Hudson Bay polar bear subpopulations, calculated as the number of days in which the combined area had less than 15% sea-ice concentration. The blue line represents a trend fit to the ice-free days.


July 15 - August 15, 2011
Sea Ice Concentration (%)
0 10 20 30
July 15 - August 15, 2012
Sea Ice Concentration (%)
0 10 20


July 15 - August 15, 2013
Sea Ice Concentration (%)
0 10 20 30 40
July 15 - August 15, 2014
Sea Ice Concentration (%)
0 10 20


July 15 - August 15, 2015
Sea Ice Concentration (%)
0 10 20 30 40 50
July 15 - August 15, 2016
Sea Ice Concentration (%)
0 10 20 30
July 15 - August 15, 2017
Sea Ice Concentration (%)
0 10 20 30
July 15 - August 15, 2018
Sea Ice Concentration (%)
0 10 20 30 40 50

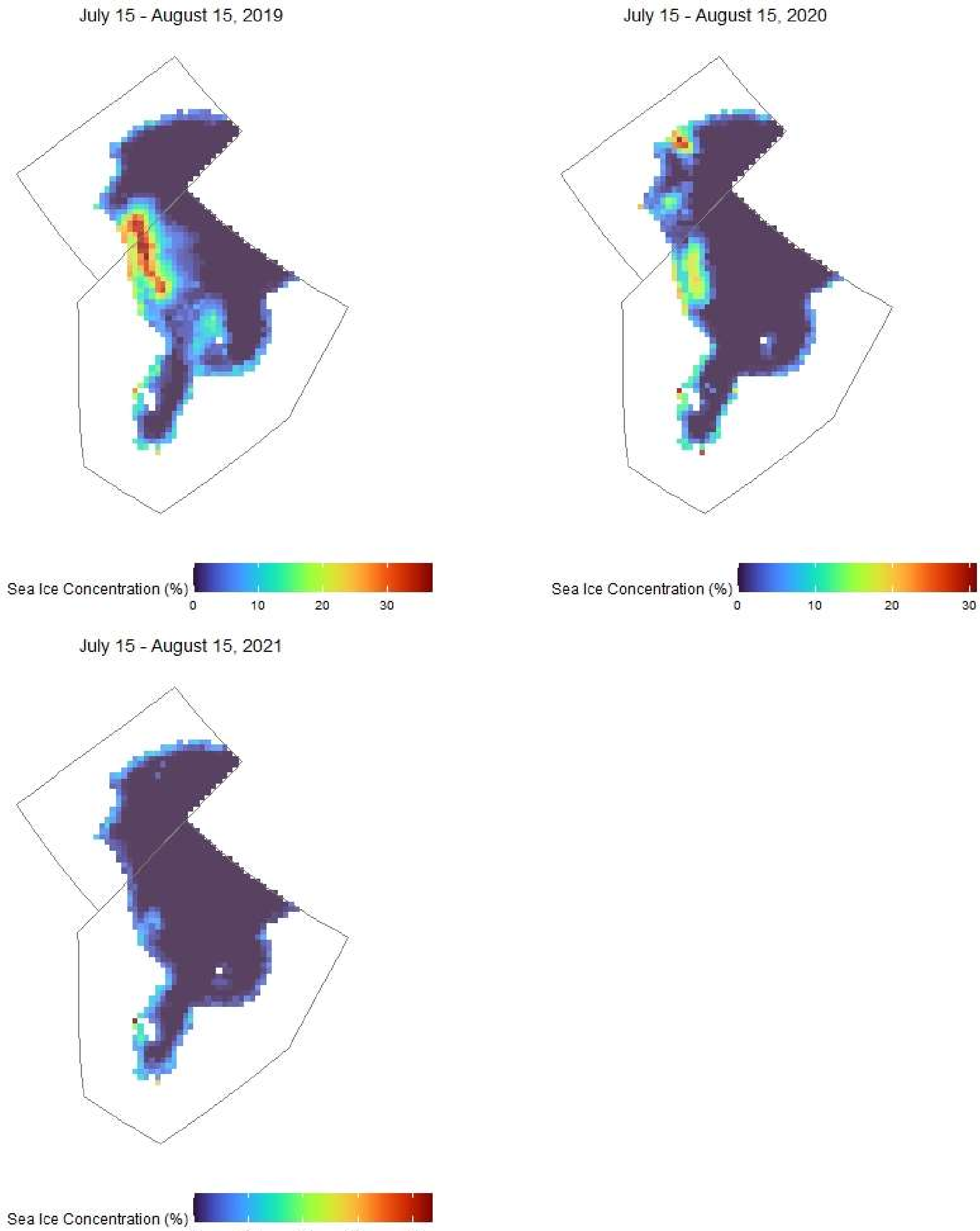
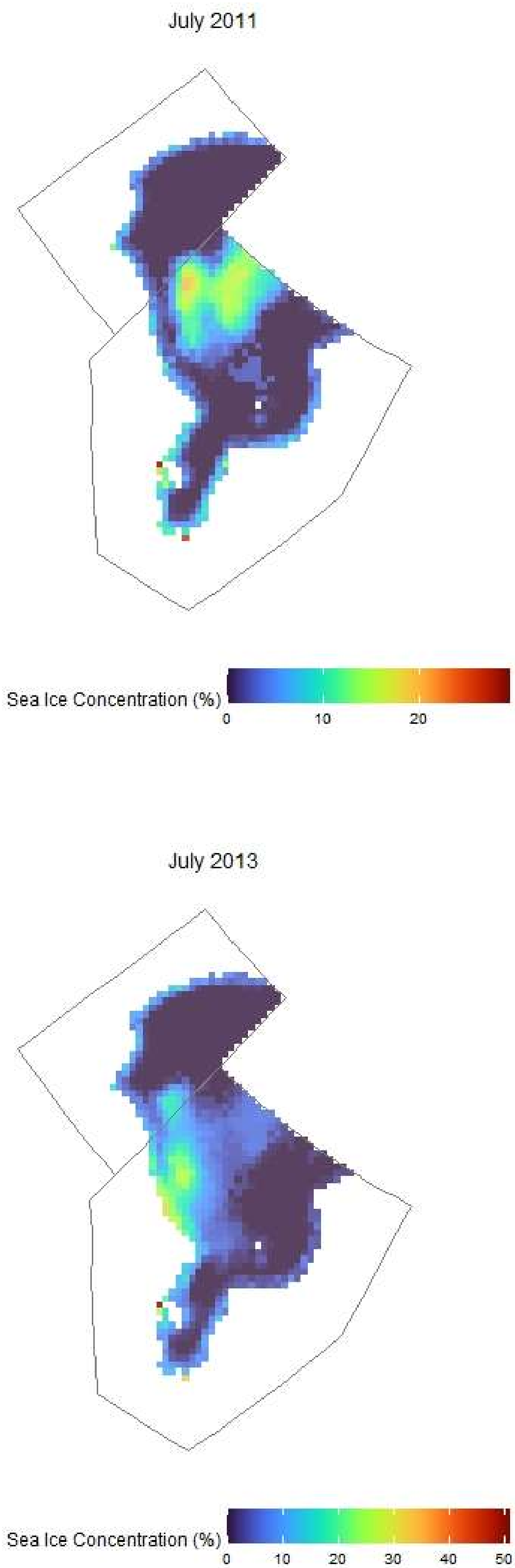
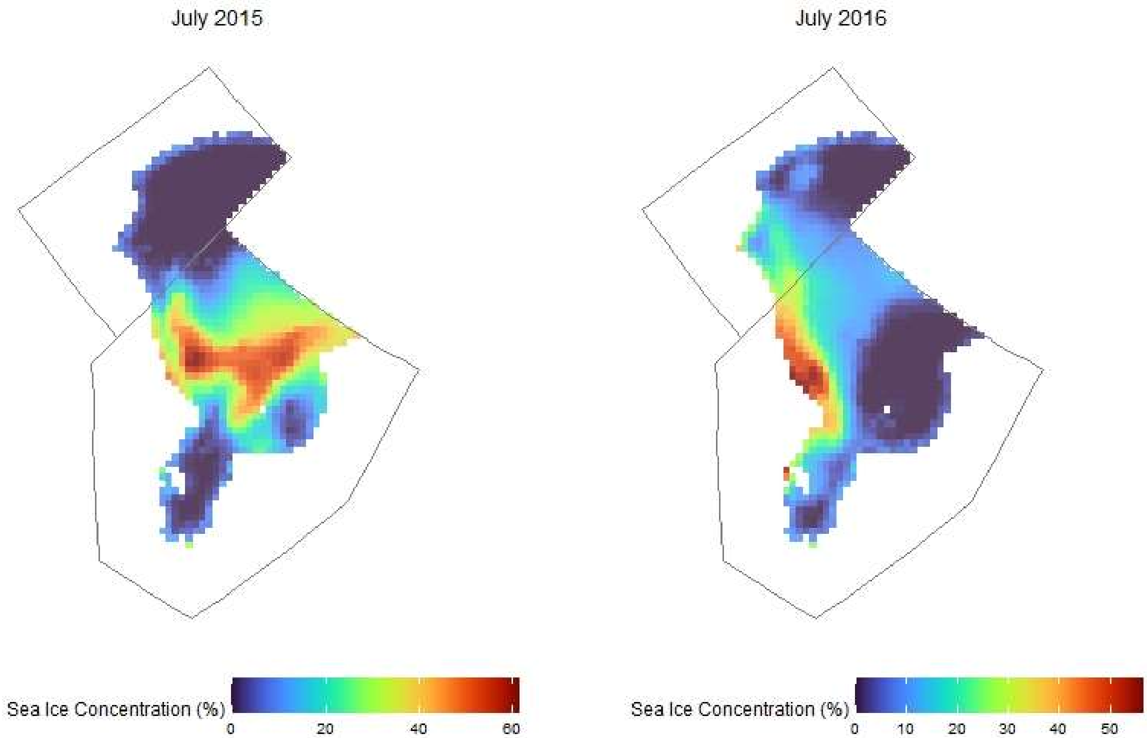
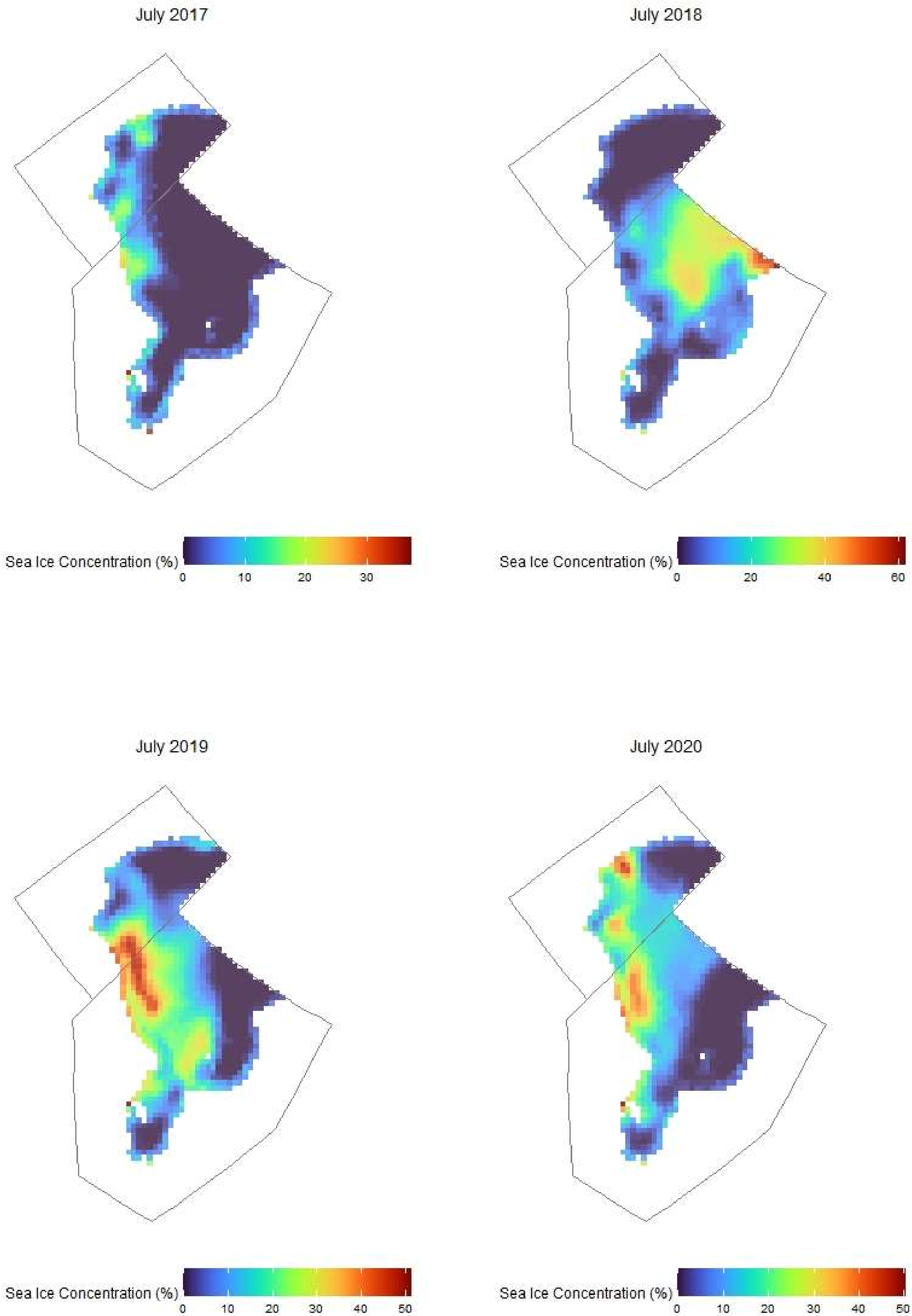


Figure 11. Average sea-ice concentration from July 15 through August 15 for each year from 2011 through 2021 for the Western and Southern Hudson Bay polar bear subpopulations.


July 2011
Sea Ice Concentration (%)
0
10
20
July 2012
Sea Ice Concentration (%)
0
10
20
30
July 2013
Sea Ice Concentration (%)
0
10
20
30
40
50
July 2014
Sea Ice Concentration (%)
0
10
20
30


July 2015
July 2016
Sea Ice Concentration (%)
0
20
40
60
Sea Ice Concentration (%)
0
10
20
30
40
50


July 2017
Sea Ice Concentration (%)
0
10
20
30
July 2018
Sea Ice Concentration (%)
0
20
40
60
July 2019
Sea Ice Concentration (%)
0
10
20
30
40
50
July 2020
Sea Ice Concentration (%)
0
10
20
30
40
50

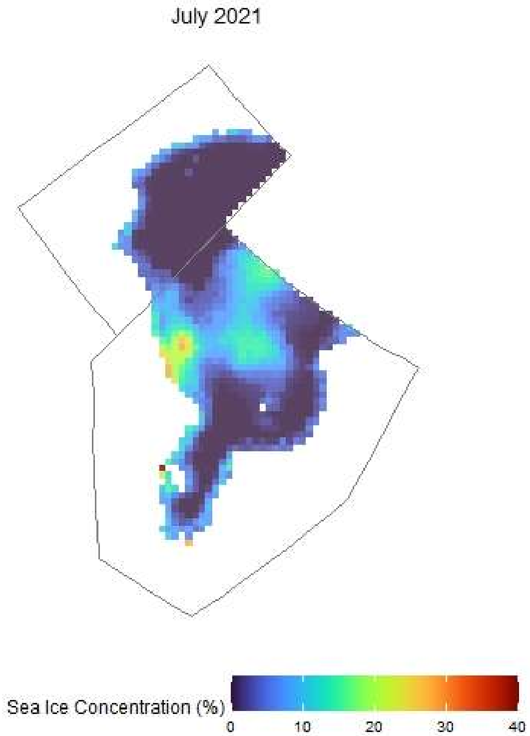


Figure 12. Average sea-ice concentration from July 1 through July 31 for each year from 2011 through 2021 for the Western and Southern Hudson Bay polar bear subpopulations.

These results have complex implications for harvest management. It is our opinion that the increase in SH is due to a combination of reduced harvest mortality during 2016-2021 relative to the 2010-2015 period and improved reproductive output due to both lower harvest levels and improved ice conditions along with annual variation in the distribution of bears between SH and WH. Resolving the degree to which each of these factors is at play is critical for harvest management. Harvest levels are set based, in part, on the number of bears within these subpopulations at the time of surveys. If there are large shifts of the broader distribution, abundances can appear higher or lower than the true number of bears available to be harvested in the respective, current subpopulation boundaries. It remains unclear however, whether such shifts in bears during the ice-free season persists through the ice season or if WH bears shift out of SH and closer to their original marking location in WH once they arrive on land the following year. Ongoing genetic biopsy work along the coastal areas of Manitoba and Ontario along with genetic identification of harvested individuals in WH and SH may help provide insight into the seasonal distribution and movements of bears under dynamic sea-ice changes.

Despite the apparent increase in bears in SH from 2016 to 2021, overall, the combined estimate of WH and SH has declined from 2011 through 2016 and appeared to remain stable between 2016 and 2021. Bears in WH and SH have experienced declines in survival and body condition at least partially related to changes in sea ice (Lunn et al. 1997, Obbard et al. 2007, Regehr et al. 2007, Lunn et al. 2016, Obbard et al. 2016, Sciullo et al. 2016) over the last several decades. Further, both subpopulations are experiencing longer ice-free periods than in the 1980s (Stern and Laidre 2016) providing less access for bears to hunt their preferred prey. This research, in conjunction with harvest data showing high relative harvest rates between 2010 and 2015 plus the results of the 2016 surveys showing declines in abundance and low numbers of yearlings in both subpopulations (Dyck et al. 2017, Obbard et al. 2018) appeared to suggest that a decline in abundance was perhaps underway. However, between 2016 and 2021, ice conditions were more favorable for bears, on average, than between 2011 and 2016, with bears often able to remain on the ice into August (Figs. 10, 11 & 12, OMNRF and ECCC unpublished data). These years of relatively good ice conditions, combined with reduced harvest, may have buffered the population against decline. Indeed, in this current survey, reproduction appeared healthy with a high proportion of yearlings and cubs. However, 2021 was one of the shortest ice seasons of the past decade and survival of yearlings and cubs could be impacted. Our post-stratification estimates indicated that 35% of the SH subpopulation consisted of yearlings and cubs of the year. If the short ice season in 2021 equates to low survival of these bears, the current estimate could immediately become overly optimistic. Continued monitoring of reproduction, survival and inter-annual movements within and between both WH and SH will be critical to continue to inform management during the intervals between aerial surveys.

Limitations and caveats

This survey and analyses were designed and completed to allow for direct comparison to the 2016 aerial survey while taking advantage of recent conceptual advances in mark-recapture distance sampling of polar bears to avoid the underestimation of abundance that results from incorrectly assuming perfect detection of bears on or very close to the transect line. These dual estimates could cause confusion, so we provide rationale for the modelling differences and suggest the most appropriate uses for the different estimates here. In all three years of the SH survey (2011, 2016 and 2021), there were challenges in fitting MRDS models. Specifically, models with distance as a covariate of the mark-recapture submodel counterintuitively did not fit the data well and were not supported by AIC in any of the 3 surveys. Our analysis of data from 2021 suggests that the rear observers' reduced probability of detecting bears near the transect line, such that the overall probability of detecting bears apparently increased with distance near the transect, at least partially explains this lack of fit. Obbard et al. (2018) and our MCDS analyses assumed perfect detection on the transect line. However, these MCDS estimates are negatively biased if bears on the transect line went undetected during the surveys. Modelling imperfect detection on the line (MRDS analyses) yields more accurate estimates if detection probability on the line was < 1.0, and so the best available estimate of SHB polar bear abundance in 2021 is the MRDS estimate of 1119 (95% CI 860-1454) bears. Future research should analyze data from all three surveys together using a consistent analytical approach to more formally assess change in bear numbers over time.

In addition to the above caveat, the three SH surveys show that there is likely some underestimation in our distance sampling estimate. In each of the three surveys, the estimate of abundance that combines the distance sampling estimate excluding the coastal zone with the double-observer mark-recapture estimate for the Ontario mainland, coastline and Akimiski island area produced a larger abundance estimate than that of the distance sampling estimate alone. In theory, these estimates should be identical because the total area included in each estimate is the same, only the method used to sample and estimate bear numbers within the coastal zone are different. However, in the 2011 survey, the estimate combining the distance sampling and coastal mark-recapture surveys was 189 bears higher (20% of the final averaged estimate), in 2016 it was 33 bears higher (4% of the final averaged estimate) and in 2021 was > 171 bears higher in the MCDS estimate (17% of the estimate) and 274 bears higher in the MRDS estimate (24% of the estimate). We attribute these differences to the highly clustered nature of bear distribution along the coast, which lends itself to high sampling variability. This proposition is supported by our sex and age class results; we estimated through post stratification that there were 173 adult male bears in the Ontario mainland, coastline and Akimiski Island area (95% CI 79-378) when using the distance sampling survey including the coastal zone but saw 184 adult male bears during the coastal mark-recapture survey. These numbers indicate that our point-estimate of adult male bears from the distance sampling portion of the survey was an underestimate, and because adult males concentrate along the coast in large aggregations, we believe the spatial heterogeneity of this class of bears along the coast is the driving cause. This logic would also suggest that our averaged estimate is likely an underestimate of the total number of bears in the subpopulation and was likewise an underestimate in 2011 and a smaller underestimate in 2016. The differences across years also matches well with the evidence that bears are displaying substantial variation in their distribution from year to year. Male bears are likely the least philopatric to their summering areas because they do not need to access known inland areas for denning. Thus, if as theorized, the ice conditions in 2011 and 2021 were conducive to greater numbers of bears in SH, with fewer bears in 2016, we assume that most of these bears would be adult males, concentrating along the coast and leading to the larger differences in the estimates in 2011 and 2021 relative to 2016.

Abundance estimate and trend

In light of the above discussion of limitations, the best available evidence indicates that using the most up-to-date modeling approach, the best estimate of abundance of the SH subpopulation in fall 2021 was 1119 (95% CI 860-1454) bears.

Conclusion

Management of polar bears in Canada makes an implicit assumption that subpopulations are discrete units. Surveys are conducted within the boundaries of subpopulations, and quotas are subsequently developed based on those results, with bears only counted against a quota if they are harvested within the bounds of a subpopulation. Although this assumption is almost certainly violated to some degree in every subpopulation, the implications for sustainable harvest of polar bears likely varies greatly depending on the degree of interchange between subpopulations that occurs when surveys to update estimates of abundance are undertaken. As first proposed by Prevett and Kolenosky (1982), our results, combined with those of Atkinson et al. (2022) and ECCC unpublished data suggest that, at least in some years, there is the potential for significant distributional shifts across the boundary between WH and SH. Therefore, these subpopulations are not acting as discrete units, which raises significant challenges for developing quotas based on management boundaries. Further complicating this issue is that much of the WH harvest occurs during the ice-free season when bears are onshore, whereas the majority of SH harvest is on the sea ice (Government of Nunavut, unpublished data) when bears from Foxe Basin, SH, and WH are free to mix (Peacock et al. 2010). In addition, there may be strong demographic differences between these subpopulations. We suggest further research aimed at assessing interannual shifts in distribution, particularly with ongoing climate warming, examining the proportion of bears harvested in subpopulations different from the one they are present in during the survey period and continued monitoring of vital rates in both subpopulations will be key for future management decisions in WH and SH.

Literature cited

Amstrup, S. C., and C. Gardner. 1994. Polar bear maternity denning in the Beaufort Sea. The Journal of Wildlife Management:1-10.

Amstrup, S. C., T. L. McDonald, and G. M. Durner. 2004. Using satellite radiotelemetry data to delineate and manage wildlife populations. Wildlife Society Bulletin 32:661-679.

Atkinson, S. N., J. Boulanger, M. Campbell, V. Trim, J. Ware, and A. Roberto-Charron. 2022. 2021 Aerial survey of the Western Hudson Bay polar Bear Subpopulation.

Final Report. Government of Nunavut, Department of Environment, Wildlife Research Section, Status Report 2022-xx.

Borchers, D. L., W. Zucchini, and R. M. Fewster. 1998. Mark-recapture models for line transect surveys. Biometrics:1207-1220.

Buckland, S., J. Breiwick, K. Cattanach, and J. Laake. 1993. Estimated population size of the California gray whale. Marine Mammal Science 9:235-249.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling: estimating abundance of biological populations.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2004. Advanced distance sampling: estimating abundance of biological populations. OUP Oxford.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer Science & Business Media.

Burt, M. L., D. L. Borchers, K. J. Jenkins, and T. A. Marques. 2014. Using mark–recapture distance sampling methods on line transect surveys. Methods in Ecology and Evolution 5:1180-1191.

Cherry, S. G., A. E. Derocher, G. W. Thiemann, and N. J. Lunn. 2013. Migration phenology and seasonal fidelity of an Arctic marine predator in relation to sea ice dynamics. Journal of Animal Ecology 82:912-921.

Crête, M., D. Vandal, L.-P. Rivest, and F. Potvin. 1991. Double counts in aerial surveys to estimate polar bear numbers during the ice-free period. Arctic:275-278.

Derocher, A. E., and I. Stirling. 1990. Distribution of polar bears (Ursus maritimus) during the ice-free period in western Hudson Bay. Canadian Journal of Zoology 68:1395-1403.

_____. 1995. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Canadian Journal of Zoology 73:1657-1665.

Descamps, S., J. Aars, E. Fuglei, K. M. Kovacs, C. Lydersen, O. Pavlova, Å. Ø. Pedersen, V. Ravolainen, and H. Strøm. 2017. Climate change impacts on wildlife in a High Arctic archipelago – Svalbard, Norway. Global Change Biology 23:490-502.

Durkalec, A., C. Furgal, M. W. Skinner, and T. Sheldon. 2015. Climate change influences on environment as a determinant of Indigenous health: Relationships to place, sea ice, and health in an Inuit community. Social science & medicine 136:17-26.

Durner, G., K. Laidre, and G. York. 2018. Polar bears: Proceedings of the 18th working meeting of the IUCN/SSC polar bear specialist group, 7–11 June 2016, Anchorage, Alaska. Gland, Switzerland and Cambridge, UK: IUCN.

Dyck, M., M. Campbell, D. S. Lee, J. Doulanger, and D. Hedman. 2017. Aerial survey of the Western Hudson Bay polar bear subpopulation: final report. . Status report 2017-xx. Nunavut Department of Environment, Wildlife Research Section, Igloolik, NU. 82p.

Dyck, M., P. M. Lukacs, and J. Ware. 2021. Recovery From Reduction: The M'Clintock Channel Polar Bear Subpopulation, Nunavut, Canada. Arctic 74:509-524.

Dyck, M., E. V. Regehr, and J. V. Ware. 2022. Demographic assessment using physical and genetic sampling finds stable polar bear subpopulation in Gulf of Boothia, Canada. Marine Mammal Science.

Faune, S. C. d. l. 2010. Consultations sur la proposition d'inscrire l'ours blanc comme espèce préoccupante selon la Loi sur les espèces en péril. Service Canadien de la Faune.

Fewster, R. M., S. T. Buckland, K. P. Burnham, D. L. Borchers, P. E. Jupp, J. L. Laake, and L. Thomas. 2009. Estimating the encounter rate variance in distance sampling. Biometrics 65:225-236.

Henri, D., H. Gilchrist, and E. Peacock. 2010. Understanding and managing wildlife in Hudson Bay under a changing climate: Some recent contributions from Inuit and Cree ecological knowledge. Pages 267-289 *in* A little less arctic. Springer.

Hochheim, K. P., and D. G. Barber. 2014. An update on the ice climatology of the Hudson Bay system. Arctic, antarctic, and alpine research 46:66-83.

Huggins, R. 1989. On the statistical analysis of capture experiments. Biometrika 76:133-140.

IPCC. 2013. Evaluation of climate models. In: climate change 2013: the physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Report 110766182X.

Kanatami, I. T. 2019. National Inuit Climate Change Strategy. Inuit Tapiriit Kanatami.

Kolenosky, G., K. F. Abraham, and C. Greenwood. 1992. Polar bears of Southern Hudson Bay. Polar Bear Project, 1984-1988, Final Report. . Wildlife Research Section, Ontario Ministry of Natural Resources,.

Laake, J., and D. L. Borchers. 2004. Methods for incomplete detection at distance zero. *in* S. T. Buckland, D. E. Andersen, K. P. Burnham, J. Laake, D. L. Borchers, andL. Thomas, editors. Advanced distance sampling. Oxford University Press, Oxford.

Laake, J., D. L. Borchers, L. Thomas, D. Miller, and J. Bishop. 2022. mrds: Mark-Recapture Distance Sampling. R package version 2.2.6m.

Laake, J., B. A. Rakhimberdiev, D. Turek, B. T. McClintock, P. J.I., and C. Wood. 2019. Package RMARK: R code for mark analysis. *in*.

Laake, J. L. 2013. RMark: an R interface for analysis of capture-recapture data with MARK.

Laforest, B. J., J. S. Hébert, M. E. Obbard, and G. W. Thiemann. 2018. Traditional ecological knowledge of polar bears in the Northern Eeyou Marine Region, Québec, Canada. Arctic 71:40-58.

Laidre, K. L., S. N. Atkinson, E. V. Regehr, H. L. Stern, E. W. Born, Ø. Wiig, N. J. Lunn, M. Dyck, P. Heagerty, and B. R. Cohen. 2020. Transient benefits of climate change for a high-Arctic polar bear (Ursus maritimus) subpopulation. Global Change Biology 26:6251-6265.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, Ø. Wiig, P. Boveng, and R. P. Angliss. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724-737.

Leafloor, J. O. 1990. Summary of polar bear surveys in Ontario, 1963-1990. Unpublished report. Ontario Ministry of Natural Resources.

Lemelin, R. H., M. Dowsley, B. Walmark, F. Siebel, L. Bird, G. Hunter, T. Myles, M. Mack, M. Gull, and M. Kakekaspan. 2010. Wabusk of the Omushkegouk: Cree-polar bear (Ursus maritimus) interactions in northern Ontario. Human Ecology 38:803-815.

Lunn, N., I. Stirling, D. Andriashek, and G. Kolenosky. 1997. Re-estimating the size of the polar bear population in western Hudson Bay. Arctic:234-240.

Lunn, N., M. Taylor, W. Calvert, I. Stirling, M. Obbard, C. Elliott, G. Lamontagne, J. Schaeffer, S. Atkinson, and D. Clark. Polar bear management in Canada 1993–1996. by: IUCN, Gland, Switzerland and Cambridge, UK, 1998.

Lunn, N. J., M. Branigan, K. Breton-Honeyman, L. H. Carpenter, M. Dyck, G. Gilbert, J. Goudie, D. Hedman, E. Keenan, D. Lee, A. Maher, R. Maraj, M. E. Obbard, J. Pisapio, F. Pokiak, L. Staples, and G. Szor. 2018. Management of polar bears in Canada, 2009-2016. Pages 33-67 *in* G. M. Durner, K. Laidre, andG. York, editors. Polar Bears: Proceedings of the Eighteenth Working Meeting of the IUCN/SSC Polar Bear Specialist Group. SSC Occassional Paper No. 63. . IUCN, Gland Switzerland, and Cambrige UK.

Lunn, N. J., S. Servanty, E. V. Regehr, S. J. Converse, E. Richardson, and I. Stirling. 2016. Demography of an apex predator at the edge of its range: impacts of changing sea ice on polar bears in Hudson Bay. Ecological Applications 26:1302-1320.

Mallory, C. D., and M. S. Boyce. 2018. Observed and predicted effects of climate change on Arctic caribou and reindeer. Environmental Reviews 26:13-25.

Marques, F., and S. T. Buckland. 2004. Covariate models for the detection function. Pages 31-47 *in* S. T. Buckland, D. R. Anderson, K. P. Burnham, J. Laake, D. L. Borchers, andL. Thomas, editors. Advanced distance sampling. Oxford University Press, Oxford.

Marques, F. F., and S. T. Buckland. 2003. Incorporating covariates into standard line transect analyses. Biometrics 59:924-935.

Middel, K. R. 2014. Movement parameters and space use for the Southern Hudson Bay polar bear subpopulation in the face of a changing climate. Unpublished M. Sc thesis, Trent University, Peterborough, Ontario.

Miller, D. L., E. Rexstad, L. Thomas, L. Marshall, and J. L. Laake. 2019. Distance Sampling in R. Journal of Statistical Software 89:1-28.

Northrup, J. M., and E. J. Howe. 2019. Polar bear research activity in Ontario 2018. Report to The Canadian Federal-Provincial-Territorial Polar Bear Technical Committee, Edmonton, AB.

Nunavik Marine Region Wildlife Board (NMRWB) 2018. Nunavik Inuit Knowledge and Observations of Polar Bears: Polar bears of the Southern Hudson Bay sub-population. Project conducted and report prepared for the NMRWB by Basterfield, M., Breton-Honeyman, K., Furgal, C., Rae, J. and M. O'Connor.

Nunavut, G. o. 2019. Nunavut polar bear co-management plan. *in*.

Obbard, M. E., M. R. Cattet, E. J. Howe, K. R. Middel, E. J. Newton, G. B. Kolenosky, K. F. Abraham, and C. J. Greenwood. 2016. Trends in body condition in polar bears (Ursus maritimus) from the Southern Hudson Bay subpopulation in relation to changes in sea ice. Arctic Science 2:15-32.

Obbard, M. E., T. L. McDonald, E. J. Howe, E. V. Regehr, and E. S. Richardson. 2007. Polar bear population status in southern Hudson Bay, Canada. US Geological Survey Administrative Report. US Department of the Interior, Reston, VA.

Obbard, M. E., and K. R. Middel. 2012. Bounding the Southern Hudson Bay polar bear subpopulation. Ursus 23:134-144.

Obbard, M. E., S. Stapleton, K. R. Middel, I. Thibault, V. Brodeur, and C. Jutras. 2015. Estimating the abundance of the Southern Hudson Bay polar bear subpopulation with aerial surveys. Polar Biology 38:1713-1725.

Obbard, M. E., S. Stapleton, G. Szor, K. R. Middel, C. Jutras, and M. Dyck. 2018. Re-assessing abundance of Southern Hudson Bay polar bears by aerial survey: effects of climate change at the southern edge of the range. Arctic Science.

Ostapchuk, J., S. Harper, A. C. Willox, and V. L. Edge. 2015. Exploring Elders' and Seniors' Perceptions of How Climate Change is Impacting Health and Well-being in Rigolet, Nunatsiavut/ᕿᒥᕐᕈᓂᖅ ᐃᓐᓇᐃᑦ ᐊᒻᒪᓗ ᐃᓄᑐᖃᐃᑦ ᐃᓱᒪᔾᔪᓯᖕᒋᓐᓂᒃ ᓇᒍᓚᑦ, ᓄᓇᑦᓯᐊᕗᒻᒥ ᓯᓚᐅᑉ ᐊᓯᔾᔨᐸᓪᓕᐊᓂᖓᓂᒃ ᐊᑦᑐᐃᓂᖃᖅᑎᓪᓗᒍ ᐃᓗᓯᕐᒥᒃ ᐊᒻᒪᓗ ᖃᓄᐃᓐᖏᓂᖕᒋᓐᓂᒃ. International Journal of Indigenous Health 9:6-24.

Paetkau, D., S. C. Amstrup, E. Born, W. Calvert, A. Derocher, G. Garner, F. Messier, I. Stirling, M. Taylor, and Ø. Wiig. 1999. Genetic structure of the world's polar bear populations. Molecular Ecology 8:1571-1584.

Peacock, E., A. Derocher, N. Lunn, and M. Obbard. 2010. Polar bear ecology and management in Hudson Bay in the face of climate change. Pages 93-116 *in* A little less Arctic. Springer.

Peacock, E., V. Sahanatien, S. Stapleton, A. Derocher, and D. Garshelis. 2009. Foxe Basin Polar Bear Project: 2009 Interim Report. Department of Environment, Government of Nunavut, Igloolik, NU.

Pörtner, H.-O., D. C. Roberts, H. Adams, C. Adler, P. Aldunce, E. Ali, R. A. Begum, R. Betts, R. B. Kerr, and R. Biesbroek. 2022. Climate change 2022: impacts, adaptation, and vulnerability. Contribution of working group II to the sixth assessment report of the intergovernmental panel on climate change.

Post, E., and M. C. Forchhammer. 2008. Climate change reduces reproductive success of an Arctic herbivore through trophic mismatch. Philosophical Transactions of the Royal Society B: Biological Sciences 363:2367-2373.

Post, E., M. C. Forchhammer, M. S. Bret-Harte, T. V. Callaghan, T. R. Christensen, B. Elberling, A. D. Fox, O. Gilg, D. S. Hik, and T. T. Høye. 2009. Ecological dynamics across the Arctic associated with recent climate change. Science 325:1355-1358.

Prevett, J., and G. Kolenosky. 1982. The status of polar bears in Ontario. Naturaliste Canadien 109:933-939.

R Core Development Team. 2022. R: a language and environment for statistical computing. R Development Core Team.

Rantanen, M., A. Y. Karpechko, A. Lipponen, K. Nordling, O. Hyvärinen, K. Ruosteenoja, T. Vihma, and A. Laaksonen. 2022. The Arctic has warmed nearly four times faster than the globe since 1979. Communications Earth & Environment 3:1-10.

Regehr, E. V., M. Dyck, S. Iverson, D. S. Lee, N. J. Lunn, J. M. Northrup, M.-C. Richer, G. Szor, and M. C. Runge. 2021. Incorporating climate change in a harvest risk assessment for polar bears Ursus maritimus in Southern Hudson Bay. Biological Conservation 258:109128.

Regehr, E. V., N. J. Hostetter, R. R. Wilson, K. D. Rode, M. S. Martin, and S. J. Converse. 2018. Integrated population modeling provides the first empirical estimates of vital rates and abundance for polar bears in the Chukchi Sea. Scientific Reports 8:1-12.

Regehr, E. V., N. J. Lunn, S. C. Amstrup, and I. Stirling. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673-2683.

Regehr, E. V., R. R. Wilson, K. D. Rode, M. C. Runge, and H. L. Stern. 2017. Harvesting wildlife affected by climate change: a modelling and management approach for polar bears. Journal of Applied Ecology 54:1534-1543.

Sciullo, L., G. Thiemann, and N. Lunn. 2016. Comparative assessment of metrics for monitoring the body condition of polar bears in western Hudson Bay. Journal of Zoology 300:45-58.

Stapleton, S., S. Atkinson, D. Hedman, and D. Garshelis. 2014. Revisiting Western Hudson Bay: using aerial surveys to update polar bear abundance in a sentinel population. Biological Conservation 170:38-47.

Stern, H. L., and K. L. Laidre. 2016. Sea-ice indicators of polar bear habitat. The Cryosphere 10:2027-2041.

Stirling, I., N. Lunn, J. Iacozza, C. Elliott, and M. Obbard. 2004. Polar bear distribution and abundance on the southwestern Hudson Bay coast during open water season, in relation to population trends and annual ice patterns. Arctic:15-26.

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

Taylor, M. K., D. P. DeMaster, F. L. Bunnell, and R. E. Schweinsburg. 1987. Modeling the sustainable harvest of female polar bears. The Journal of Wildlife Management:811-820.

Taylor, M. K., P. D. McLoughlin, and F. Messier. 2008. Sex-selective harvesting of polar bears Ursus maritimus. Wildlife Biology 14:52-60.

Wenzel, G. W. Polar bear as a resource: an overview. Northern Research Forum, 2004.

Wiig, Ø., S. N. Atkinson, E. W. Born, S. Stapleton, T. Arnold, M. Dyck, K. L. Laidre, N. J. Lunn, and E. V. Regehr. 2022. An on-ice aerial survey of the Kane Basin polar bear (Ursus maritimus) subpopulation. Polar Biology 45:89-100.

Wong, P. B., M. Dyck, A. Hunters, I. Hunters, M. Hunters, and R. Murphy. 2017. Inuit perspectives of polar bear research: lessons for community-based collaborations. Polar Record 53:257-270.

Supplemental material

Table S1. Multiple-covariate distance sampling (MCDS) models, degrees of freedom, Akaike's information criterion (AIC) values and change in AIC from the top model (ΔAIC) for models fit to polar bear distance sampling data collected across the entirety of the Ontario mainland, coastline and Akimiski island area in 2021. Abundance was estimated by model averaging across models marked with asterisks. See main text for description of model structure.

MCDS model	df	AIC	ΔAIC
Half-normal	1	1831.83	0.00
Half-normal + vegetation density*	2	1832.45	0.61
Hazard rate + vegetation density*	3	1832.60	0.76
Half-normal + vegetation height	2	1833.51	1.67
Hazard rate	2	1833.54	1.70
Half-normal + visibility	2	1833.64	1.80
Hazard rate + vegetation height	3	1833.78	1.95
Hazard rate + visibility	3	1835.34	3.50

Table S2. Multiple-covariate distance sampling (MCDS) models, degrees of freedom, Akaike's information criterion (AIC) values and change in AIC from the top model (ΔAIC) for models fit to polar bear distance sampling data collected across the Ontario mainland, coastline and Akimiski island area excluding the coastal zone in 2021. Abundance was estimated by model averaging across models marked with asterisks. See main text for description of model structure.

MCDS model	df	AIC	ΔAIC
Half-normal + visibility*	2	1161.75	0.00
Hazard rate + visibility*	3	1162.84	1.09
Half-normal	1	1164.05	2.30
Hazard rate	2	1164.48	2.72
Hazard rate + vegetation density	3	1165.00	3.25
Half-normal + vegetation density	2	1165.51	3.75
Hazard rate + vegetation height	3	1165.89	4.14
Half-normal + vegetation height	2	1166.05	4.30

Table S3. Mark-recapture distance sampling (MRDS) models, degrees of freedom (df), Akaike's information criterion (AIC) values, difference in AIC from the top model (ΔAIC) and model weights (w_i) used in model averaging for models fit to polar bear distance sampling data collected across the entirety of the Ontario mainland, coastline and Akimiski island area in 2021. All models were included when model-averaging to estimate abundance. We use the top model to estimate the number of bears of different ages by post-stratification. See main text for description of model structure.

Mark-recapture submodel	Distance sampling submodel					
Covariates	Key function	Covariates	df	AIC	ΔAIC	w_i
Blind spot + observer × side + visibility + glare	Half-normal	Vegetation + glare	11	2113.29	0.00	0.30
Blind spot + observer × side + visibility + glare	Hazard rate	Vegetation + glare	12	2114.60	1.32	0.16
Blind spot + observer + side + visibility + glare	Half-normal	Vegetation + glare	10	2114.68	1.39	0.15
Blind spot + observer × side + visibility + glare	Half-normal	Glare	10	2114.87	1.59	0.14
Blind spot + observer + side + visibility + glare	Hazard rate	Vegetation + glare	11	2115.99	2.71	0.08
Blind spot + observer + side + visibility + glare	Half-normal	Glare	9	2116.26	2.98	0.07
Blind spot + observer × side + glare	Half-normal	Vegetation + glare	9	2116.55	3.26	0.06
Blind spot + observer × side + glare	Hazard rate	Vegetation + glare	10	2117.86	4.57	0.03
Blind spot + observer × side + glare	Half-normal	Glare	8	2118.13	4.84	0.03

Table S4. Mark-recapture distance sampling (MRDS) models, degrees of freedom (df), Akaike's information criterion (AIC) values, difference in AIC from the top model (ΔAIC) and model weights (w_i) used in model averaging for models fit to polar bear distance sampling data collected across the Ontario mainland, coastline and Akimiski island area excluding the coastal zone in 2021. All models were included when model-averaging to estimate abundance. See main text for description of model structure.

Mark-recapture submodel	Distance sampling submodel					
Covariates	Key function	Covariates	df	AIC	ΔAIC	(w_i)
Blind spot + observer × side	Half-normal	Visibility	8	1359.18	0.00	0.51
Blind spot + observer × side	Hazard rate	Visibility	9	1360.53	1.36	0.26
Blind spot + observer × side	Half-normal	None	7	1362.06	2.88	0.12
Blind spot + observer × side	Hazard rate	None	8	1362.41	3.23	0.10

Table S5. Estimates of polar bear abundance within the coastal zone, obtained using double-observer mark-recapture methods, proportion of cubs, yearlings and adults for 4 years of surveys.

Year	Abundance estimate (95% CI) coastal transect	Proportion cubs observed coastal transect	Proportion yearlings observed coastal transect	Proportion adults observed coastal transect
2011	422 (381 – 467)	0.15	0.12	0.60
2016	269 (244 – 297)	0.17	0.03	0.71
2018	249 (230 – 270)	0.10	0.07	0.74
2021	335 (309 – 363)	0.12	0.10	0.64