ALTERNATIVE METHODS FOR MONITORING POLAR BEARS IN THE NORTH AMERICAN ARCTIC

A DISSERTATION
SUBMITTED TO THE FACULTY OF
UNIVERSITY OF MINNESOTA
BY

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IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

ADVISOR: DR. DAVID L. GARSHELIS

DECEMBER 2013

Acknowledgements

Funding, logistical support, and in-kind support for this research were provided by: the Government of Nunavut, USGS-Changing Arctic Ecosystems Initiative (Ecosystems Mission Area-Wildlife Program), USGS-Alaska Science Center, BaffinLand Iron Mines Corporation, U.S. Bureau of Ocean Energy Management, Environment Canada, Greenland Institute of Natural Resources, Government of Yukon, Government of Northwest Territories, Indian and Northern Affairs Canada, Inuvialuit Implementation funds, Makivik Corporation, Manitoba Conservation, Nunavut Wildlife Research Trust, Ontario Ministry of Natural Resources, Parks Canada, Polar Continental Shelf Program, Polar Geospatial Center, and the University of Minnesota.

Foxe Basin research was conducted in cooperation with Hunting and Trapping Organizations in the Nunavut communities of Kimmirut, Cape Dorset, Igloolik, Hall Beach, Repulse Bay, Coral Harbour, and Chesterfield Inlet. I am grateful for the individuals who helped ensure the successful completion of the Foxe Basin work, including S. Atkinson, B. Ford, J. McConnell, S. Medill, and V. Sahanatien, as well as Department of Environment staff and members of local Hunting and Trapping Organizations who worked as survey observers.

I thank M. Campbell (Government of Nunavut, Department Of Environment), L. Muckpah, L. Ikakhik and P. Mikeeuneak (Arviat Hunting and Trapping Organization), N. Ford, T. Kabuitok and R. Nukitut (Rankin HTO), J. Savikataaq, J. Batstone (Manitoba Conservation), R. Redhead (Parks Canada) and J. Talon (Hudson Bay Helicopters) for their work as observers and data collectors during the Western Hudson Bay survey. Input

on survey design was provided by members of the Arviat, Rankin Inlet, and Whale Cove Hunting and Trapping Organizations. D. Lee and B. Dean (Nunavut Tunngavik Incorporated) also contributed to survey design, and I. Stirling and other reviewers provided useful comments on earlier drafts.

I thank the members of the Nattivak Hunting and Trapping Organization and Conservation Officers S. Ageeagok, B.J.Hainu and S. Levesque for their assistance with the Baffin Bay logistical preparations and field work. I appreciate the participation of the Inuvik, Aklavik and Tuktoyaktuk Hunters and Trappers Committees, including community observers L. Angasuk, L.J. Meyook and R. Cockney during the southern Beaufort Sea survey. I am thankful for the support of the Wildlife Management Advisory Councils of the Northwest Territories and the North Slope, the GY and the GNWT as well as the cooperation of the Inuvialuit Game Council. I appreciate the assistance of A. Pagano and S. Marshall.

I am very grateful for the guidance and support of my committee: David Garshelis, Elizabeth Peacock, Todd Arnold, and J.L. David Smith. Special thanks to my advisor Dave, for his steady support and direction, to both Dave and Lily for taking a chance on a guy who was spending his time studying sea turtles, and to Todd for his statistical wizardry. I also appreciate the assistance and support of Stephen Atkinson, who essentially functioned as an unofficial committee member. Thanks to the students, faculty, and staff of the Conservation Biology Graduate Program and the Department of Fisheries, Wildlife and Conservation Biology at the University of Minnesota. Special thanks also to Michelle LaRue for her assistance with the satellite imagery work.

The pilots and engineers – Universal Helicopters, Unaalik Aviation, Hudson Bay Helicopters, and Canadian Helicopters – obviously provided a critical function in the completion of this work, and I am very grateful for their safe-flying, judgment, and expertise.

Finally, I couldn't have completed this program without the unwavering support and patience of my family, especially my wife Carol. And my little boys - Jonah and baby yet-to-be-born – provided me with the necessary final jolt of energy and motivation.

Abstract

Because polar bears (*Ursus maritimus*) are dependent on sea ice, climate change poses a significant threat to their long-term existence. The forecasted impacts of sea ice loss are circumpolar, but to date, effects have been documented in only a few, well-studied populations. Data demonstrating the impacts of climate change are less conclusive or simply lacking elsewhere. In general, current inventory regimes do not enable monitoring with enough regularity to meet the information needs of decision-makers. This reality, combined with pressures from northern communities to reform invasive research techniques (i.e., capture and marking), provided the backdrop for my dissertation. My objective was to implement and evaluate novel, efficient and broadly applicable methods for monitoring polar bears. I first conducted comprehensive aerial (helicopter) surveys of the Foxe Basin population in Nunavut, Canada during the summer, ice-free season. This work demonstrated the utility of the method for estimating the abundance of polar bear populations on land and provided a model for applications in other seasonally ice-free populations. I applied this framework to a neighboring population (Western Hudson Bay) and compared the result to an estimate obtained from physical mark-recapture. This comparison suggested negative bias in the mark-recapture estimate due to spatially limited sampling and resultant capture heterogeneity. Next, I assessed the potential for employing aerial surveys on sea ice in springtime. Although results suggest that detection can be estimated with adequate precision, logistical constraints may hinder the ability to obtain a representative density estimate during springtime. Monitoring programs based on aerial surveys can be designed with sufficient power (>0.8) to detect declines of 40%

and 50% over 15- and 30-year periods, with costs comparable to mark-recapture. Costs may be significantly diminished and safety concerns alleviated, however, if bears could be monitored with satellite imagery. I evaluated this technique in a low topography, ice-free setting. Results indicate that bears were reliably identified on imagery, and an estimate of abundance was highly consistent with an independent aerial survey.

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INTRODUCTION

Climate change is one of the central challenges facing contemporary conservation biologists and wildlife managers. The Arctic, in particular, is contending with severe environmental, social and ecological change as a result of climate warming (Arctic Climate Impact Assessment 2004, Post et al. 2009). With forecasted warming expected to reach 5–8°C by the end of the 21st century (Parry et al. 2007) and recent Arctic sea ice loss outpacing projections (Stroeve et al. 2007, 2012, Comiso et al. 2008), biologists and managers tasked with understanding the impacts of climate change and conserving Arctic wildlife face a daunting task.

Perhaps more than any other species, the polar bear is an icon of the North, globally revered and invoked to symbolize climate change and promote broader climate change agendas (Slocum 2004, O'Neill et al. 2008, Clark et al. 2012). As sea ice obligates, their future is uncertain: by mid-century, substantial reductions in optimal sea ice habitat (Durner et al. 2009) may result in extirpation or significant declines of polar bears across much of their range (Amstrup et al. 2008). The impacts of climate change are projected to exhibit significant temporal and geographic variation, with the earliest effects anticipated in regions where sea ice melts completely during the summer or recedes northward into the polar basin (Derocher et al. 2004, Amstrup et al. 2008, Stirling and Derocher 2012). Initial changes – reduced adipose stores and deteriorating body condition resulting from diminished access to prey – will eventually lead to declining reproductive output, reduced survival rates, and ultimately decreased abundance (Derocher et al. 2004, Stirling and Derocher 2012). Two-thirds of the world's polar bears

may be lost by mid-century (Amstrup et al. 2008). These projections paint a somber picture for the future of polar bear conservation, but reductions in greenhouse gas emissions may improve their probability of persistence (Amstrup et al. 2010).

Polar bears occur in 19 recognized populations in 5 countries (Obbard et al. 2010). To date, documented effects of climate change are limited to a few, well-studied populations. In Western Hudson Bay, declines in abundance, survival and natality have been linked to increases in the duration of the ice-free season (Stirling et al. 1999, Regehr et al. 2007). Similarly, reductions in reproduction, body size and survival were associated with changing sea ice dynamics in the southern Beaufort Sea (Regehr et al. 2010, Rode et al. 2010), and body condition (Rode et al. 2012) and survival (Peacock et al. 2012) were positively related to summer sea ice conditions in Baffin Bay.

Elsewhere, despite climate change-related losses of sea ice, effects on polar bear demography are more ambiguous. For example, deteriorating body condition metrics were related to declining summer sea ice in Davis Strait (Rode et al. 2012), but abundance and survival there appear higher than historical levels (Peacock et al. 2013); density dependence may contribute to low recruitment and decreases in condition (Rode et al. 2012, Peacock et al. 2013). Likewise, data from Southern Hudson Bay, at the southern extent of polar bear range, are equivocal. Whereas declines in condition and survival rates have been reported (Obbard et al. 2006, 2007), abundance appears unchanged (Obbard et al. 2013). Conversely, recruitment and condition have been stable despite significant reductions in sea ice in the Chukchi Sea (Rode et al. 2013). Although

such variability is not unanticipated (e.g., Derocher et al. 2004), it complicates interpretation and highlights information and monitoring needs.

Polar bears are classified as Vulnerable by the International Union for the Conservation of Nature (Schliebe et al. 2008a), but reliable data are absent from large expanses of the circumpolar range. Indeed, more than half of the recognized populations are categorized as Data Deficient (Obbard et al. 2010). More geographically comprehensive data are required to accurately assess status and understand how polar bears are impacted by the changing environment (Vongraven et al. 2012).

Part of the challenge presently facing polar bear biologists and managers centers around research methodologies for population monitoring (Peacock et al. 2011). Physical capture-recapture has formed the basis for inventory programs in North America (e.g., Stirling et al. 1977, Derocher and Stirling 1995a, Taylor et al. 2005). Capture provides a wealth of information from which to assess status and facilitates a variety of other research initiatives, ranging from distribution and movement to diet and body condition. However, the technique also has drawbacks. Physical capture is resource intensive: in Nunavut, Canada, capture programs are generally completed over 3 years (e.g., Taylor et al. 2006), and costs of multi-year inventories can reach into the millions of dollars (e.g., Nielson et al. 2013). Given the resource and logistical requirements, populations generally are not revisited for 15 or more years under the present inventory cycle. This rotation lags well behind the speed at which the Arctic is changing and hence does not meet the information needs of decision-makers charged with polar bear management.

Additionally, some Inuit in northern communities strongly object to physical capture, largely because it conflicts with cultural values regarding relationships between people and wildlife (Nunavut Wildlife Management Board 2007, Peacock et al. 2011). Concerns about how capture programs have been implemented (i.e., incomplete sampling of study sites; see Nunavut Wildlife Management Board 2007) and the belief that immobilization and handling negatively impact bears and make them unsafe for consumption also contribute to this opposition. As a result, the implementation of capture initiatives has been prohibited in some areas (Government of Nunavut, unpublished data).

Given this ecological, social and management context, there is a clear need for the development and integration of alternative tools in polar bear inventory programs. Multiple techniques may satisfy the demands for more regular and geographically comprehensive information while better reflecting northern cultural values and fitting within the current resource construct. Importantly, however, these alternatives do not provide the same detailed information as capture-based studies.

Line transect aerial surveys offer one alternative. Aerial surveys are a well-established technique, widely implemented to estimate the abundance of wildlife populations. For decades, researchers have examined applications in large mammals (e.g., Siniff and Skoog 1964, Caughley 1974, Short and Bayliss 1985), including ursids (e.g., Barnes and Smith 1998, Quang and Becker 1999) and even polar bears (Belikov et al. 1991, Crete et al. 1991, McDonald et al. 1999, Wiig and Derocher 1999, Evans et al. 2003). With polar bears, however, logistical constraints and concerns over the ability to obtain a representative density (and thus abundance) estimate have limited

implementation. A recent study in the Barents Sea (Aars et al. 2009) represents the only population-wide survey of polar bears.

Recent technological advances in the field of remote sensing, specifically high resolution satellite imagery, may facilitate monitoring distribution and abundance at inaccessible locations. The utility of satellite imagery has been demonstrated for wildlife in the Antarctic (LaRue et al. 2011, Fretwell et al. 2012), yet rigorous assessments are absent for Arctic wildlife. Genetic mark-recapture via biopsy darting (Pagano et al. 2013, Government of Nunavut and Greenland Institute of Natural Resources, unpublished data) offers a third alternative, but lies beyond the scope of this dissertation.

The objective of my dissertation was to design, implement and evaluate alternatives to physical capture for inventorying and monitoring polar bears in the circumpolar Arctic. I sought to develop and examine tools with broad applicability that improved efficiency, thereby facilitating more regular and geographically comprehensive monitoring. In Chapter 1, I devised and implemented large-scale aerial surveys of the Foxe Basin population, a region in which polar bears are confined to land during the late summer following sea ice melt and break-up. The inventory provided the first new abundance estimate for the population since the early 1990s. In Chapter 2, I applied the framework developed in Foxe Basin and implemented a systematic and comprehensive aerial survey of the Western Hudson Bay population. Western Hudson Bay has been monitored intensively for more than four decades; this extensive dataset enabled me to evaluate aerial survey results alongside an independent estimate of abundance derived from mark-recapture population models. In Chapter 3, I examined the potential for using

aerial surveys in the springtime to monitor polar bears on sea ice, and I assessed trade-offs between statistical power, potential inventory programs, and projected costs to better inform the design of long-term monitoring initiatives. Finally, in Chapter 4, I evaluated the utility of high resolution satellite imagery as a tool to provide coarse information on abundance and distribution when polar bears are on land.

Each dissertation chapter is written in the form of a manuscript that has been or will be submitted to a peer-reviewed journal. Formatting varies commensurate with the requirements of the individual target journals. I used plural pronouns (e.g., ‘we’) throughout the text because all intended publications have multiple co-authors. However, I am the senior author on all manuscripts, and I am responsible for the content.

Chapter 1

ESTIMATING THE ABUNDANCE OF POLAR BEARS WITH AERIAL SURVEYS DURING THE ICE-FREE SEASON

INTRODUCTION

Polar bears (*Ursus maritimus*) are sea ice obligates and considered among the most highly sensitive marine mammals to the projected consequences of climate change (Laidre et al. 2008). With sea ice loss outpacing forecasts and declines accelerating in recent years (Stroeve et al. 2007, 2012, Comiso et al. 2008), the timeline for climate-induced impacts on polar bears has hastened. By mid-century, significant decreases in optimal sea ice habitat in the polar basin are projected (Durner et al. 2009), and polar bears may be extirpated from or substantially reduced across most of the circumpolar Arctic (Amstrup et al. 2008). In the near term, however, the impacts of climate change on polar bear demography will exhibit temporal and spatial variability (Stirling and Derocher 2012). Whereas declines in body condition and demographic parameters have been reported and linked to sea ice losses in some regions (e.g., Stirling et al. 1999, Regehr et al. 2007, 2010, Rode et al. 2010, 2012), elsewhere, effects have not been realized or are more ambiguous (e.g., Obbard et al. 2013, Peacock et al. 2013, Rode et al. 2013).

Understanding the impacts of climate change on polar bears requires regular information from across their range, and the rapidly changing environment has increased the data needs of jurisdictions charged with their management. However, significant gaps in basic information remain for large expanses of the Arctic (Vongraven et al. 2012).

Whereas broad and extensive datasets stretch back 30 to 40 years in 2 regions (i.e., the southern Beaufort Sea, Amstrup et al. 2001, and western Hudson Bay, Regehr et al. 2007), 11 of the 19 recognized populations are categorized as data deficient by the IUCN's Polar Bear Specialist Group (Obbard et al. 2010).

Historically, physical capture has been the primary tool used to study polar bear demography in the North American Arctic. Intensive mark-recapture studies are conducted periodically to generate estimates of abundance, survival, and recruitment metrics, thereby informing harvest management (e.g., Stirling and Kiliaan 1980, Taylor et al. 2002, Peacock et al. 2013). Capture programs also have facilitated addressing other research questions related to distribution, movements and habitat use (Durner et al. 2009, Towns et al. 2010, Cherry et al. 2013), diet (Thiemann et al. 2008), genetic structure (Paetkau et al. 1999), and body condition (Rode et al. 2012). The information obtained from this diversity of initiatives has formed the basis for our current understanding of polar bear ecology and population status.

However, capture programs have faced criticism as well. Capture-based inventories are time-, labor- and resource-intensive, and they require handling large numbers of bears (either within a small number of years or over a longer period) to obtain demographic estimates with desired levels of precision (e.g., Regehr et al. 2007, Peacock et al. 2013). In Nunavut, Canada, the inventory program is designed to rotate on a cycle, such that populations are intensively studied for 3 years and typically are not revisited for 15 years. This rotation is untenable given the speed at which the environment is changing and the public scrutiny that managers face. Moreover, immobilizing and handling bears

to attach ear tags and collect biometric data and tissue samples is inconsistent with traditional Inuit values; segments of the northern public are firmly opposed to wildlife capture and marking (Nunavut Wildlife Management Board 2007, Peacock et al. 2011). As such, there is a need for techniques that enable more regular and geographically comprehensive monitoring while better reflecting Inuit values.

Line transect aerial surveys (Aars et al. 2009), genetic mark-recapture via biopsy darting (Pagano et al. 2013) and tetracycline bio-marking (Taylor and Lee 1994, Taylor et al. 2006) are the primary alternatives that have been implemented or explored as inventory techniques for polar bears. Aerial surveys, in particular, have garnered significant attention as a monitoring tool. Small-scale studies on polar bears have been conducted under a range of conditions, in different seasons, from a variety of survey platforms, and across the Arctic (e.g., Belikov et al. 1991, Crete et al. 1991, McDonald et al. 1999, Wiig and Derocher 1999, Evans et al. 2003). To date, however, the only population-wide aerial survey was conducted on sea ice and land in the Barents Sea (Aars et al. 2009). The limited use of this technique is attributable to unpredictable weather and sea ice conditions, resource constraints, competing objectives, and skepticism about the ability to obtain representative and reliable density and abundance estimates.

Our objective was to develop an aerial survey methodological framework that can be broadly implemented for estimating polar bear abundance in the Canadian Arctic. We conducted this work in a seasonally ice-free population while polar bears were on land. The landscape provided greater contrast than sea ice for sighting bears, and because bears

were distributed over a smaller geographic area at higher densities, sampling was more efficient.

METHODS

Study Area

The Foxe Basin population spans ~1.1 million km^2 across the Nunavut territory and northern Quebec, Canada (Figure 1). The population's boundaries extend from Hudson Bay and Hudson Strait, northward to central and northern Baffin Island, and westward to the Melville Peninsula and the Fury and Hecla Strait. Topographic relief is highly variable, ranging from the flat tundra and sandy beaches of islands in northern Foxe Basin to rocky hills of Baffin Island and mainland Canada, to the mountainous east coast of Southampton Island. The region is sparsely vegetated. Foxe Basin is seasonally ice-free from July–August to November–December (Sahanatien and Derocher 2012), during which time bears come ashore.

Foxe Basin was last inventoried in the early 1990s (~2,200 bears; Taylor et al. 2006) and is classified as data deficient by the IUCN Polar Bear Specialist Group (Obbard et al. 2010). Harvest is regulated by a rigorous yet flexible quota system in Nunavut. Four communities in northern Quebec also harvest from the population.

Survey Design and Sampling Protocols

We implemented a systematic, stratified line-transect sampling design to comprehensively survey Foxe Basin during the 2009 and 2010 ice-free seasons. Polar bears in seasonal ice populations generally congregate near the shore during the ice-free season (Derocher and Stirling 1990, Towns et al. 2010). Hence, we divided the study area

into multiple strata based on proximity to the coastline. We delineated a very high density coastal zone; a high density stratum, including land within 5 km of the coast; a moderate density stratum, including land 5 – 15 km from the nearest coastline; and a low density stratum, including land 15 – 50 km from the nearest coastline (Figure 1). We defined strata and delineated the inland extent of the study area based on satellite telemetry data gathered in 2008–2009 (V. Sahanatien et al., University of Alberta, and Government of Nunavut, unpublished data) and local knowledge (Stapleton et al., unpublished data). During 2008, 2 of 15 satellite collared bears traveled beyond the inland extent of the defined study area, both moving westward into the Gulf of Boothia. Frequency and extent of inland movements were similar in 2009. We created an additional stratum for large islands in which transects extended across the width of the islands. Coats and Mansel islands in northern Hudson Bay were stratified based on coastal proximity in 2009, but we re-categorized them as 'large islands' in 2010. Stratification otherwise remained consistent between years.

We sampled the mainland and large islands using 2 types of transects. We surveyed the very high density coastal zone with contour transects flown along the shoreline. In 2009, these were flown ~200 m inland of the high water mark, whereas they were flown at or slightly below the high water mark in 2010. Secondly, we surveyed along transects oriented perpendicular to the shoreline and hence the coastal polar bear density gradient (hereafter perpendicular transects; see Buckland et al. 2001). Procedurally, we flew a perpendicular transect from the shoreline to a specified distance inland (which varied by strata; see below), then turned 90° and flew directly to the next

perpendicular transect, turned again and followed this transect to the coast. Upon reaching the water, we paused sampling while resetting our course, then surveyed along the coastal contour to the next perpendicular inland transect, and repeated the procedure. Since perpendicular transects extended to the shoreline, some bears could be sighted from both contour and perpendicular transects. As such, sightings from portions of transects in this region were not fully independent. However, field crews were instructed not to keep track of bears when we reset our course at the intersection of the perpendicular and contour transects so that previous knowledge of the presence of a bear did not impact search patterns. We did not announce sightings or investigate sightings until observers were provided a full opportunity to sight bears from both contour and perpendicular transects within the region of overlap.

Perpendicular transects were systematically spaced at 10-km intervals across Foxe Basin in both years. We maintained a ratio of 4:2:1 for transects extending 5, 15, and 50 km inland, respectively, in 2009. For large islands, we positioned transects across the island width and spaced at 10 km intervals. To increase sampling in the far inland stratum in 2010, we altered these ratios to 3:2:1. In a few areas, such as northern Hudson Strait, we were unable to fly contour transects due to the highly irregular coastline, so we sampled these areas only with perpendicular transects.

We sampled small islands with ad hoc transects to maximize coverage. Remnant ice floes were completely surveyed, and we recorded all bears sighted in the open water. In 2009, there were large offshore ice floes that could not be comprehensively surveyed. We flew randomized transects over ice in these regions and used weekly Canadian Ice

Service maps (regional sea ice charts; available: http://ice-glaces.ec.gc.ca) to delineate sea ice extent. These line transect data were incorporated with perpendicular transect data for analysis. If possible, ice floes and adjacent land were surveyed on the same day to minimize the potential for bears to move between them.

We surveyed the Quebec coastline and completed multiple inland transects in 2009 but not 2010. We sighted one bear during sampling of the entire coast (straight-line distance of coastline >550 km) in 2009, presumably due to the early recession of ice from this region. Similar sea ice dynamics in 2010 suggested that surveys were not warranted in this region.

We surveyed from a helicopter (Bell 206 LongRanger) at a groundspeed of about 150 km/hr and an above-ground altitude of ~120 m. We simultaneously collected double-observer (sight-resight; Pollock and Kendall 1987) and distance sampling (Buckland et al. 2001) data. Front (including the pilot) and rear observers comprised our first and second sampling periods, respectively. We constructed a partition between the seats to ensure that sightings by the front observers did not cue rear observers, and we did not communicate sightings until all observers had a full opportunity to sight a bear (i.e., each bear had been fully passed).

We recorded flight paths and flew off-transect to record each bear location (where it was first sighted) via GPS and measured perpendicular distances from sightings to the transect line in a GIS (Marques et al. 2006). We documented group (hereafter cluster) size, recorded activity at first sighting, remotely assessed age class [cub of the year (COY), yearling, subadult and adult], and approximated body condition using a

subjective condition index (Stirling et al. 2008). We defined clusters of bears as individuals whose detections were not independent (e.g., a family group comprised of an adult female and her offspring). For each sighting, we recorded environmental variables that may have influenced detection (all with a qualitative 1 to 3 scale). Covariates included visibility (VIS; excellent, fair, or poor), general habitat (2009: HAB; flat / minimal structure impeding detection; hilly / moderate structure; mountainous / high structure), topography (2010: TOPO; low, moderate, or high) and structure such as large rocks within a 30 m radius of the sighting (2010: STRUC; minimal, moderate, or extensive).

Data Analyses

Distance Sampling with Perpendicular Inland Transects

We used distance sampling (Program DISTANCE 6.0, Thomas et al. 2010) to estimate density and abundance with perpendicular transects. With distance sampling, a detection function $g(x)$ is fit to sighting distances from the transect line (i.e., helicopter flight path) to estimate $\hat{P}_a$, the probability of detecting a bear cluster located in surveyed region a (Buckland et al. 2001). Density is estimated as

$$\hat{D} = \frac{n}{2wL\hat{P}_a}$$

where n is the number of clusters detected and $2wL$ denotes the surveyed region (i.e., a; w is the right-truncation distance, or the maximum sighting distance included in analysis, and L is the total length of transects; Buckland et al. 2001). Density is inflated by mean cluster size $E(s)$ and extrapolated across the study area of size A to obtain an estimate of abundance.

A fundamental assumption of distance sampling is that all bears located on the transect line are detected (Buckland et al. 2001). We examined initial histograms of sightings distances and used double-observer data in a mark-recapture modeling framework to assess detection at distance 0. Double-observer (sight-resight) analysis is a variation of the mark-recapture method: independent teams of observers visually 'mark' (i.e., sight) and 'recapture' (resight) animals. We implemented the Huggins mark-recapture model (Huggins 1989, 1991) to estimate detection. Because rear observers had a blind spot directly beneath the helicopter, we used double-observer data in the areas that were available to both sets of observers (~75 m from the aircraft; i.e., we subtracted 75 m from all sightings, Borchers et al. 2006). We fit models including distance as a covariate and allowed detection probabilities and effects of distance to remain constant or vary between front and rear observers. We also evaluated the impact of other covariates on detection. We conducted these analyses in Program MARK (White and Burnham 1999) and used Akaike's Information Criteria, adjusted for small sample sizes (AIC_c) for model selection (Burnham and Anderson 2002).

Our study design enabled 2 independent estimates of abundance in the coastal zone: (1) with data from perpendicular transects (via distance sampling) and (2) with data from contour transects (described below). Therefore, we created datasets including and excluding the perpendicular transect data from the coastal strip for both 2009 and 2010. Whereas bear density declined substantially within a few km of the coast, we assumed that this density gradient was negligible within our moderate and low density strata farther inland. Hence, we included sighting data from the legs connecting consecutive

perpendicular transects when they were located >1 km from inland boundaries in the moderate and low density strata.

We right-truncated (excluded) the most distant 5% of observations to improve model fit and smooth the tail of the detection function (Buckland et al. 2001). We estimated a global detection function and used strata-specific encounter rates and cluster sizes to generate density and abundance estimates by stratum. These figures were summed in DISTANCE to obtain abundance estimates for the regions surveyed with perpendicular transects.

We fit conventional distance sampling models using uniform, half-normal and hazard rate key functions and cosine, simple polynomial and hermite polynomial series expansion terms. We also fit distance sampling models including covariates to explain variability in detection (Marques and Buckland 2003) in place of series expansion terms (half-normal and hazard rate key functions). We used forward stepwise selection to evaluate potential covariates. Model fit was examined with q-q plots and Chi-squared, Kolmogorov-Smirnov, and Cramér-von Mises tests, and model selection was based on AIC_c.

We considered individual transects, partitioned by stratum, as sampling units for variance estimation. Transects that spanned a single stratum multiple times (e.g., due to crossing an inlet or other body of water) were pooled by stratum and categorized as a single unit (Aars et al. 2009). We used a bootstrapping procedure (1,000 iterations, implemented in DISTANCE) in which transects were resampled by stratum, a detection

function was fit to each dataset, and a unique abundance estimate was generated to estimate variance (Buckland et al. 2001).

Double-observer Analyses with Coastal Contour Transects and Small Islands

We flew coastal contour transects ~200 m inland of the high water mark in 2009 and near the high water mark in 2010. In both years, we sampled ~500 m inland on one side of the helicopter and seaward on the other side, meaning that we surveyed regions 700 m and 500 m inland of the high water mark in 2009 and 2010, respectively. For most parts of Foxe Basin we could see farther inland, but we chose these conservative widths to accommodate the most rugged portions of the population. For small island sampling, we used strip widths of 500 m and 750 m in 2009 and 2010, respectively, based on preliminary sightings distance histograms constructed from perpendicular transect data.

We used double-observer sighting data with the Huggins model (implemented in Program MARK) to estimate detection probabilities and derive abundance. Discrete clusters were treated as the sampling unit, and bears that were completely unavailable to the rear observers (i.e., in the blind spot) were coded as removals in capture histories. Covariates considered in these analyses included cluster size, VIS, HAB, TOPO, and STRUC. We allowed detection and VIS to vary between front and rear observers, and we modeled the effects of other covariates as constant between observer teams. We fit global models and selected covariates through backwards stepwise selection, based on AIC_c. A generalized Horvitz-Thompson estimator was implemented with detection probabilities to estimate the number of clusters present in the sampled areas. For both the coastal zone and the small island sampling, we extrapolated density estimates across the study area

and inflated variance via the delta method (Powell 2007). Estimates were multiplied by mean cluster size, and we conducted bootstrapping simulations (1,000 iterations) with estimated numbers of clusters and observed cluster sizes to estimate variance.

Total Abundance

We generated 2 abundance estimates in both 2009 and 2010. First, we estimated total abundance by summing estimates from perpendicular transect datasets (extending to the coast), small island sampling, and bears sighted on small ice floes or in the open water. The second estimate excluded the portions of the perpendicular transects that intersected the high density coastal zone (i.e., 700 m and 500 m from the coastline in 2009 and 2010, respectively) and instead used the abundance estimate from the coastal transects. Individual point estimates and their variances were added to obtain 4 separate abundance estimates. We assigned equal weights (i.e., $w = 0.25$) and used a model-averaging framework (Anderson et al. 2000) to generate an overall estimate that incorporated process and model uncertainty and reflected unconditional variance.

RESULTS

We conducted the aerial surveys during August – September, 2009 and August – October, 2010, flying >300 hours in each year. We began sampling in Quebec during 2009 and in northern Hudson Strait during 2010, and proceeded counter-clockwise around Foxe Basin. We successfully surveyed nearly all planned transects (Figure 2). In both years, however, we were unable to complete sampling in far southwestern Foxe Basin (portions of Chesterfield Inlet; Figure 2) due to logistical and resource constraints.

We observed 816 and 1,003 polar bears (616 and 790 independent bears) in 2009 and 2010, respectively. Observed litter sizes for cubs of the year (COY) and 1-2 year-olds (yearlings) were similar in 2009 (COY $\bar{x}$=1.57, SD=0.55, n=75; yearlings $\bar{x}$=1.55, SD=0.54, n=53) and 2010 (COY $\bar{x}$=1.53, SD=0.57, n=80; yearlings $\bar{x}$=1.4, SD=0.50, n=65).

The distribution of polar bears was consistent between years (Figure 3). High concentrations of bears were observed in west-central Foxe Basin near Southampton Island as well as the northern Foxe Basin islands. Conversely, relatively few bears were spotted along Hudson Strait and across much of western Baffin Island, and sightings were rare near communities. Bears were observed most frequently in the coastal strip, the nearshore inland stratum and on large and small islands, but sightings were documented in all strata.

Perpendicular Transects

Histograms of sighting distances provided strong support for a distance-based detection function (Figure 4). Double-observer analyses indicated that the probability of a bear located within 75 m from the aircraft being sighted by at least one observer was 97-98% in both years. This finding suggests that the assumption of complete detection at distance 0 was approximately valid, consistent with our perception in the field. We proceeded with conventional and multiple covariate distance sampling analyses.

Because density varied greatly across Foxe Basin, we post-stratified the study site into high and low density strata, using geographically discrete units (islands). This

process yielded a total of 8 inland strata (4 strata at 2 density levels), plus an ice stratum in 2009.

We found no relationship between cluster size and distance from transect. Hence, we used stratum mean cluster sizes in all models. In 2009, we observed a cluster of 11 bears congregating around a bowhead whale (*Balaena mysticetus*) carcass. We considered this an anomaly and replaced this with the stratum mean cluster size.

We surveyed ~12,400 km and >12,800 km along perpendicular transects during 2009 and 2010, respectively. After right truncation, distance sampling analyses including and excluding data from the coastal zone incorporated 183 and 114 clusters, respectively, in 2009 and 301 and 203 clusters in 2010. We condensed most covariates from a 1 to 3 scale into binary categories because of underrepresentation in some categories. Model selection was consistent between datasets within year but differed substantially between years (Table 1; Figure 4). Habitat-related covariates were included in top models in both years; a half-normal key function was best supported in 2009, whereas a hazard rate key function was most supported in 2010. Right-truncation distances differed by >300 m between years. These differences resulted in a wider effective strip half-width in 2010 versus 2009 (Table 1). However, density estimates were highly consistent within datasets and between similar datasets (i.e., including or excluding coastal zone data). All highly supported models ($\Delta AIC_c < 2$) indicated adequate model fit with Chi-squared, Kolmogorov-Smirnov and Cramér–von Mises goodness-of-fit tests ($P > 0.15$).

Densities estimated by stratum exhibited the expected coastal density gradient, supporting our stratification (Figure 5). Although densities in the far inland strata were

very low, these sites contributed to overall population size because the strata covered large geographic areas (Figure 6).

Global density estimates were consistent among highly supported models (Table 1), so we selected the top model in each dataset. Abundance estimates were consistent between years for the datasets including and excluding coastal zone sightings (Table 2).

Coastal Transects and Small Islands

We sampled 45% and 40% of the coastal zone and included 227 and 257 clusters in coastal analyses for 2009 and 2010, respectively. In 2009, the most supported model included HAB as a covariate and estimated equal detection for front and rear observers ($\bar{p}$=0.83, SE=0.022), producing $\widehat{N}$=241 (SE=5.5) clusters. In 2010, the best model included STRUC and estimated separate detection probabilities for front ($\bar{p}$=0.79, SE=0.028) and rear ($\bar{p}$=0.86, SE: 0.025) observers, yielding $\widehat{N}$=271 (SE=5.2) clusters. We extrapolated across unsampled regions and multiplied by mean annual cluster sizes to obtain estimates of $\widehat{N}$=727 (SE=22.6) and $\widehat{N}$=873 (SE=23.2) bears in coastal zones during 2009 and 2010.

We sampled 33% of small islands in 2009 and 42% in 2010, and we included 82 and 109 groups in analyses for the 2 years. We sampled proportionately fewer islands in those areas with high numbers of islands (e.g., northern Hudson Strait), so we delineated regions based on sampling intensity (high, moderate and low intensities), and calculated and extrapolated densities by sampling region. The most supported model in 2009 included no covariates but different detection probabilities for front ($\bar{p}$=0.73, SE=0.053) and rear ($\bar{p}$=0.85, SE=0.045) observer teams. In 2010, the top model included VIS and

constant detection between teams ($\bar{p}$=0.88, SE=0.026). Multiplying the total number of clusters by mean cluster size yielded estimates of $\widehat{N}$=281 (SE=16.6) bears in 2009 and $\widehat{N}$=238 (SE=9.1) in 2010 (Table 2).

In addition to the bear sighted in Quebec in 2009, we sighted 2 and 8 bears in 2009 and 2010, respectively, that were not in any strata (e.g., on small ice floes, in open water, or >50 km inland; Table 2).

Total Abundance

The abundance estimates generated in 2009 and 2010 were remarkably consistent (Table 2); in both years, estimates including coastal contour transects had higher precision. Averaging yielded an overall abundance estimate of $\widehat{N}$=2,585 bears (95% log-normal confidence interval=2,096 – 3189; CV=10.7%) for 2009-2010. This estimate includes COYs and yearlings with their mothers.

DISCUSSION

Aerial survey abundance estimation

Although line transect aerial surveys are widely implemented to monitor a variety of wildlife species, including ice-associated marine mammals (e.g., Borchers et al. 2006), their application to polar bears at large spatial scales has been limited due to perceived logistical and technical constraints. Prior to this research, only one population-wide aerial survey had been implemented (Aars et al. 2009). Our production of a reliable and relatively precise population estimate using land-based aerial surveys in Foxe Basin suggests that this technique can provide a useful tool to inventory polar bears.

We used double-observer and distance sampling protocols and surveyed from transects oriented perpendicular to and parallel to the coast to accommodate the coastal density gradient. Despite analytical differences, estimates were highly consistent with different combinations of transects and between years. Actual abundance was unlikely to have changed significantly from 2009 to 2010, given the slow growth rates of polar bear populations (e.g., Taylor et al. 2005, 2009) and the similarity in litter sizes between years. Thus, although we did not independently estimate abundance using a different method, completing the survey in consecutive years provided a measure of the reliability of our results. The consistency of our results and the relatively high precision of our final, model-averaged estimate (10.7% CV) demonstrate the utility of aerial surveys for longer-term monitoring.

Despite the consistency documented in our abundance estimates, we estimated markedly different detection functions with distance sampling in the 2 years (Figure 4). We hypothesize that observer skill and experience as well as inter-annual differences in sighting conditions contributed to this discrepancy. Regardless of the underlying causes, these results reinforce that detection functions are not transferrable between years or among populations. Our results encourage caution when pooling data across years, study sites, or observer teams.

Our findings were consistent between years and methods, and we elected to average results to obtain a final estimate of abundance. The 4 estimates, representing 2 datasets and 2 years, were equally weighted. This allowed us to incorporate process (i.e., which dataset best reflected true abundance) and model (e.g., among detection functions)

uncertainties. Averaging marginally reduced precision, but we believe that it resulted in a more reliable estimate (Anderson et al. 2000).

This study provides a framework for aerial surveys of populations in seasonally ice-free regions and other ecoregions. Surveys should be tailored to the unique conditions of a particular region and study season, and factors such as very low density or highly clumped distribution of bears may limit their utility in some regions. Key design and methodological considerations include:

1) *Stratification of the study area based on suspected density gradients* (e.g., Figure 5). Stratification facilitates efficient allocation of sampling effort while ensuring comprehensive coverage. With land-based studies, stratification should be based on the coastline, whereas surveys over sea ice may rely on proximity to the floe edge (where landfast ice meets open water and / or pack ice; Wiig and Derocher 1999). Satellite telemetry, capture records, and local knowledge can inform this element of study design.
2) *Orientation of transects perpendicular to the density gradient to improve precision and reduce bias* (Buckland et al 2001). For example, transects for land-based studies should be oriented perpendicular to the coastline since polar bears tend to congregate near the shore.
3) *Integration of both double-observer and distance sampling field protocols*. Collecting these data increases analytical flexibility and enables an explicit test of methodological assumptions, thereby enhancing estimate robustness.

4) *Intensified sampling of high-density areas*. In both 2009 and 2010, coastal contours improved precision, and they may reduce potential bias due to clumped polar bear distributions along the shoreline.
5) *Collection of covariates that may affect sighting probabilities.* We found that detection of polar bears on land was affected by topographic relief, habitat structure near the bear, and weather conditions (see also Aars et al. 2009).

Assumptions

In designing this study, we considered several fundamental assumptions underlying distance sampling and double-observer methods. With distance sampling, key requirements include: perfect detection on the transect line (or quantifying the proportion missed); sighting bears before they move in response to oncoming aircraft; accurately measuring distance from the flight path to bear clusters; and sampling randomly with respect to the distribution of bears (Buckland et al. 2001). Integrating a double-observer platform, recording flight paths and bear sightings at their initial locations, measuring sighting distances in a GIS (Marques et al. 2006), orienting transects across the coastal density gradient, and systematically spacing transects were core design elements that we implemented to fulfill these assumptions. As such, we believe our distance sampling results represent a relatively unbiased estimate of polar bear abundance in Foxe Basin.

Our double-observer protocols ensured that observers worked independently, yet we were certain of which bears were sighted because observers communicated detections immediately after passing bears. Nevertheless, sampling periods were nearly instantaneous: observations occurred from very similar vantage points at virtually the

same time. Therefore, some easy-to-spot bears were more likely observed by both teams, whereas other difficult-to-sight bears were more likely to be missed by both teams. Such heterogeneity in detection results in an underestimate of abundance if insufficiently modeled (Otis et al. 1978, Pollock et al. 1990). We examined potential sources of heterogeneity in our modeling, and we defined narrow strip widths for coastal contour transects and surveys of small islands to reduce heterogeneity. The consistency between estimates that included the contour transects analyzed with only double-observer models versus those based exclusively on distance sampling suggests that heterogeneity did not substantially affect our results.

We defined the study area and delineated strata based on available resources, logistical constraints, and information about polar bear distribution in seasonal ice populations. We acknowledge, however, that we did not survey all areas where bears could have occurred. We did not completely sample Quebec because densities were too low to warrant the expense. Additionally, sea ice breakup and polar bear distribution elsewhere in Foxe Basin were generally consistent between 2009 and 2010, suggesting that a significant distributional shift toward Quebec was unlikely to have impacted our results. We also saw some bears in unsurveyed areas (e.g., open water or far inland) while we ferried between transects. We believe that the number of bears located outside our defined study area likely comprised an insignificant proportion of the population. However, any bias in our study, either through incomplete delineation of the study area or from technical components (i.e., unmodeled heterogeneity with double-observer models,

detection at distance 0 < 1 with distance sampling models) would result in an underestimate of abundance.

Status of Foxe Basin

During the late 1980s and early 1990s, Taylor et al. (2006) estimated polar bear abundance in Foxe Basin by remotely marking bears with tetracycline (via darts) and using teeth from the harvest as the recovery sample (Taylor and Lee 1994). A Lincoln-Peterson estimator yielded 2,200 (SE: 260) bears in 1994, although this estimate is likely negatively biased due to heterogeneity in detection (Otis et al. 1978, Pollock et al. 1990). Our estimate ($\widehat{N}$=2,585, SE=277) suggests that the Foxe Basin population has remained relatively stable in the 16 years since that study, implying that harvests have been sustainable. Observed litter sizes were generally comparable to populations with robust annual growth rates (e.g., Baffin Bay; Taylor et al. 2005). Regionally, litter sizes observed in Foxe Basin were consistent neighboring Southern Hudson Bay (Obbard et al. 2013) and greater than Western Hudson Bay (this volume, chapter 2), populations which are classified as stable and declining, respectively (Obbard et al. 2010). Combined, the litter size metrics suggest that recruitment in Foxe Basin currently is consistent with a healthy population (but see Peacock et al. 2013: in Davis Strait, a positive growth rate was estimated from low recruitment and high adult survival rates). Anecdotally, polar bears observed during our aerial surveys generally appeared to be in good body condition (based on a qualitative fatness index; Stirling et al. 2008).

The impacts of climate change are anticipated to vary spatially, and Foxe Basin is located in the seasonal ice ecoregion, where the effects will be first exhibited (Amstrup et

al. 2008, Stirling and Derocher 2012). Because Foxe Basin generally has more persistent sea ice (i.e., a shorter ice-free season), it is presently less at-risk than other seasonal ice populations (Peacock et al. 2010).

Management Implications and Conclusions

The Foxe Basin aerial surveys, in addition to subsequent surveys in Western Hudson Bay (this volume, chapter 2) and Southern Hudson Bay (Obbard et al. 2013), demonstrate the utility of this technique as an inventory tool in populations of polar bears that come ashore during the ice-free season. Nevertheless, integrating aerial surveys in monitoring programs will require modifying assessment and quota setting practices. Aerial surveys yield reliable, snapshot abundance estimates, but they provide limited or no information on other demographic parameters (recruitment and survival) that have been used by managers for assessing status and setting harvest (via population viability analyses with RISKMAN, Taylor et al. 2001; e.g., Taylor et al. 2002, 2005). As such, managers should consider alternatives, such as ancillary information from aerial surveys (e.g., litter sizes, proportion of bears with cubs), to inform status. In accordance with the precautionary principle, the reduced information calls for more conservative management (i.e., lower harvest; see Peacock et al. 2011). In contrast to multi-year capture and tagging programs, however, aerial surveys generate an estimate of abundance within a single year and thus are relatively rapid and efficient. As such, aerial surveys may facilitate more frequent monitoring, thereby safeguarding against the potential consequences of overharvest and avoiding reliance on model projections based on dynamic vital rates. Ideally, aerial survey results should be rigorously compared with capture-based estimates

to assess potential biases in the methods, relative precision, and how these techniques can best complement one another in monitoring programs. Such comparisons will improve trend assessment, since estimates derived from multiple techniques must be evaluated to establish long-term trends.

Table 1. Results of distance sampling analyses from aerial surveys of the Foxe Basin polar bear population, late summer, 2009 and 2010. Highly supported models (ΔAIC_c <2) are presented. In the column Model, the key function (Hazard rate of half-normal) is followed by covariates. Global density reflects densities from only the regions surveyed with perpendicular transects (e.g., inclusion or exclusion of the coastal zone). Density confidence intervals are based on empirical estimation of variance. Hab: Habitat; Vis: Visibility; Topo: Topographic relief. ESW: Effective strip half-width. p: Detection probability.

			Estimate (95% Confidence Interval)		
Data Set	**Model**	**Δ AICc**	**ESW (m)**	**P**	**Global Density (bears / 1000 km^2**
2009: No coastal transects	Half-normal / Hab + Vis	0.00	764 (686-849)	0.55 (0.49-0.61)	8.1 (6.5-10.0)
	Half-normal / Hab	0.03	780 (704-864)	0.54 (0.51-0.62)	7.9 (6.4-9.8)
2009: Separate coastal transects	Half-normal / Hab	0.00	795 (691-914)	0.55 (0.48-0.63)	5.9 (4.5-7.7)
	Half-normal / Hab + Vis	0.45	786 (676-915)	0.54 (0.47-0.63)	5.9 (4.5-7.8)
2010: No coastal transects	Hazard rate / Topo	0.00	1193 (1119-1272)	0.69 (0.65-0.74)	6.9 (5.7-8.3)
	Hazard rate / Topo + Vis	1.91	1186 (1112-1265)	0.69 (0.65-0.74)	7.1 (5.9-8.5)
2010: Separate coastal transects	Hazard rate / Topo	0.00	1207 (1113-1309)	0.66 (0.61-0.72)	5.4 (4.3-6.7)
	Hazard rate	1.57	1196 (1099-1302)	0.66 (0.61-0.72)	5.5 (4.4-6.8)

Table 2. Summary of abundance components estimated from aerial surveys of the Foxe Basin polar bear population conducted during the late summer, ice-free seasons in 2009 and 2010. Standard error associated with perpendicular transect estimates was derived with bootstrapping simulations.

		Estimates (Standard Error)				
Year	**Data Set**	**Perpendicular transects**	**Coastal transects**	**Small Islands**	**Bears in open water, outside defined strata, or Quebec**	**Total Abundance (95% CI)**
2009	No coastal transects	2401 (276)	N/A	281 (17)	3	2685 (2196-3284)
	Separate coastal transects	1681 (230)	727 (23)	281 (17)	3	2692 (2274-3187)
2010	No coastal transects	2061 (205)	N/A	238 (9)	8	2307 (1939-2746)
	Separate coastal transects	1538 (176)	873 (23)	238 (9)	8	2657 (2331-3029)

Figure 1. Strata delineated for aerial surveys of the Foxe Basin polar bear population, conducted during the late summer, ice-free seasons in 2009 and 2010. The Quebec coastline was surveyed in 2009 but not in 2010.

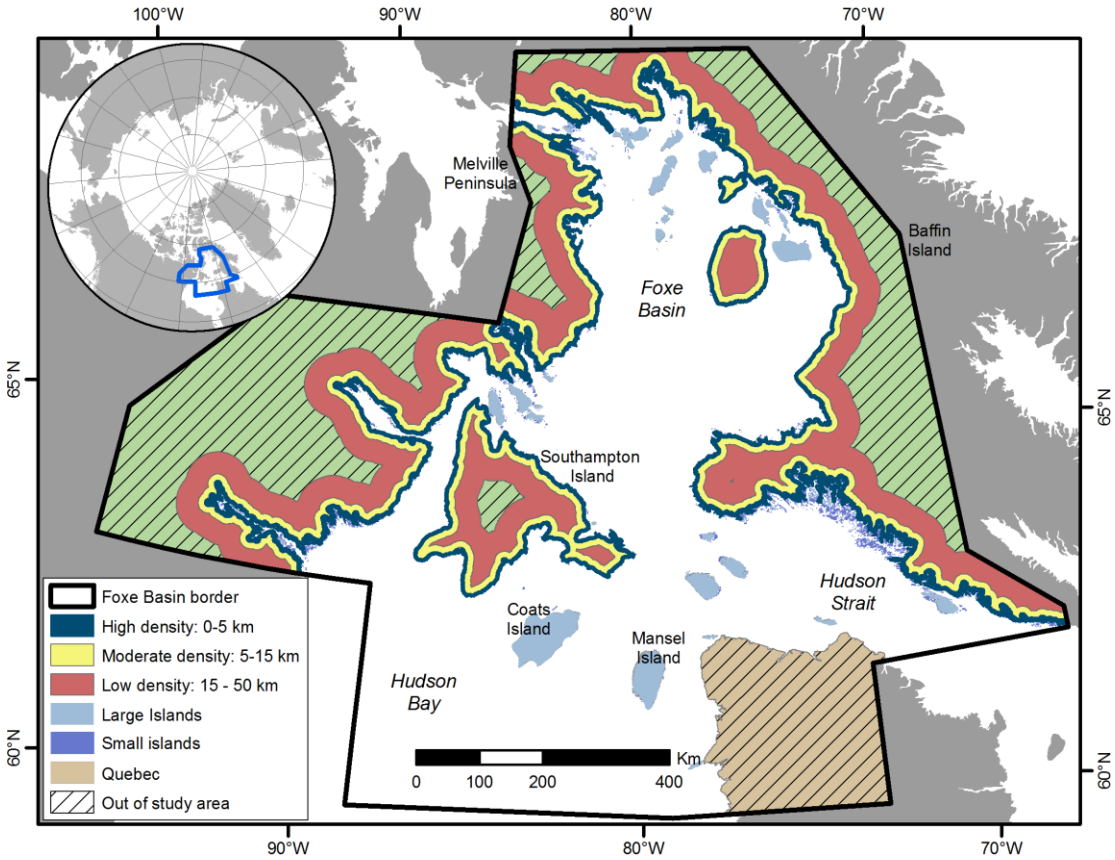


Figure 2 Transects flown during a helicopter-based aerial survey of the Foxe Basin polar bear population, August – October, 2010.

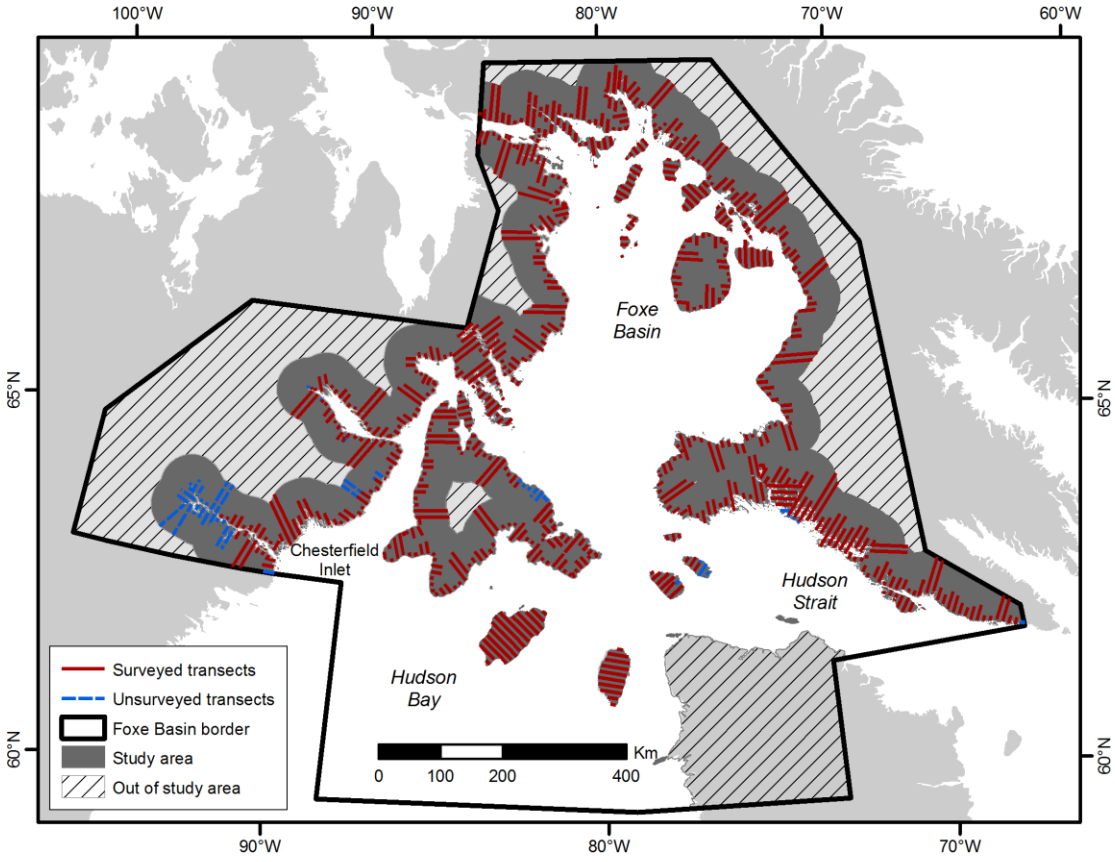


Figure 3 Locations of polar bears sighted from perpendicular transects during aerial surveys of the Foxe Basin population in late summer (a) 2009 and (b) 2010.

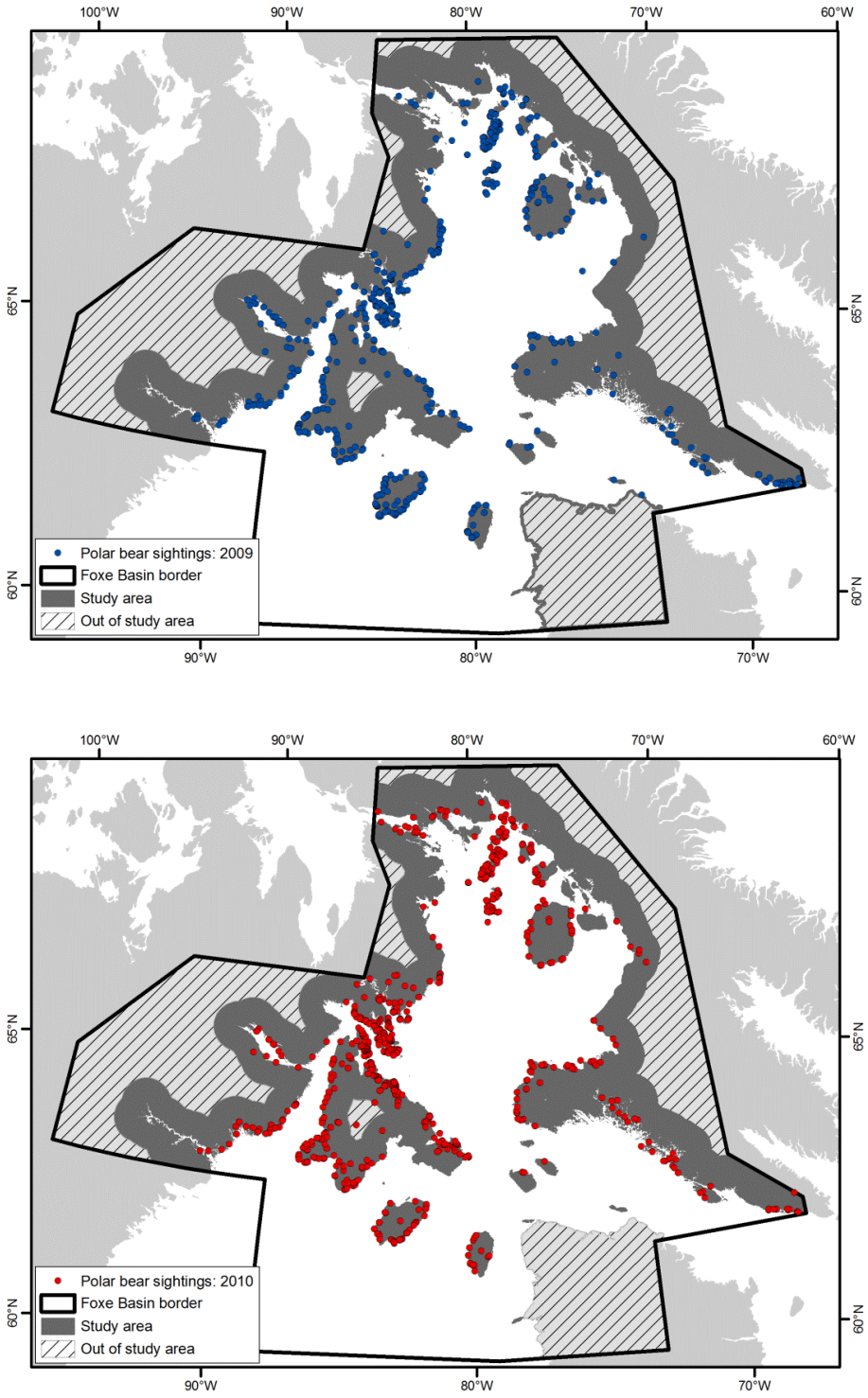
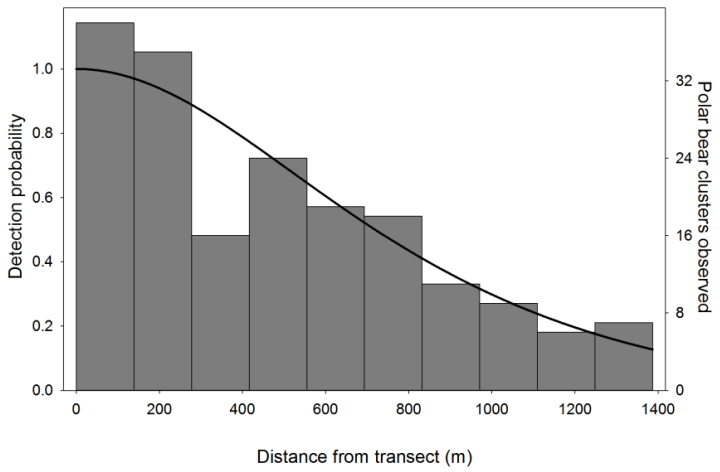


Figure 4. Distance sampling histograms and detection functions estimated with data collected from perpendicular transects during aerial surveys of the Foxe Basin polar bear population, late summer, (a) 2009 and (b) 2010. Detection functions are from the most highly supported models and are averaged over covariate values. The perpendicular transect data sets included sightings within the coastal zone.

(a)



(b)

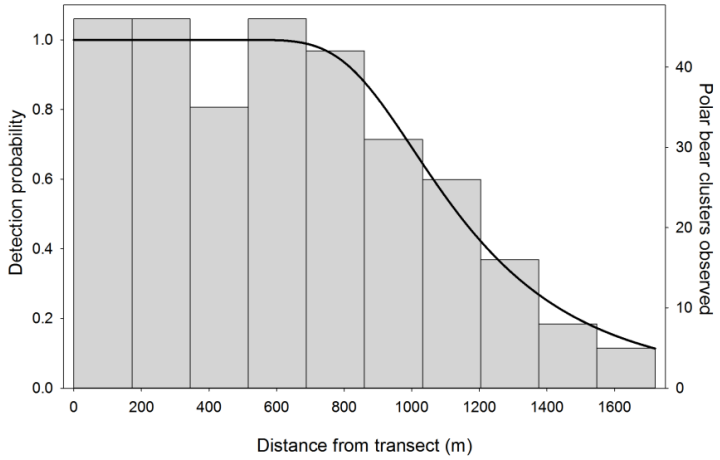


Figure 5. Polar bear densities (± 95% confidence intervals) estimated during aerial surveys of the Foxe Basin population, late summers, 2009 and 2010. Densities are from the most highly supported distance sampling models and are categorized by stratum. Analyses included the entire perpendicular transect datasets for each year. HD: High density. LD: Low density. Strata measurements refer to the distance from the coast. For example, HD: 0-5 km refers to the high density stratum extending from the coastline to 5 km inland. Ice was not as prevalent in 2010 and thus was not categorized as a stratum with distance sampling analyses.

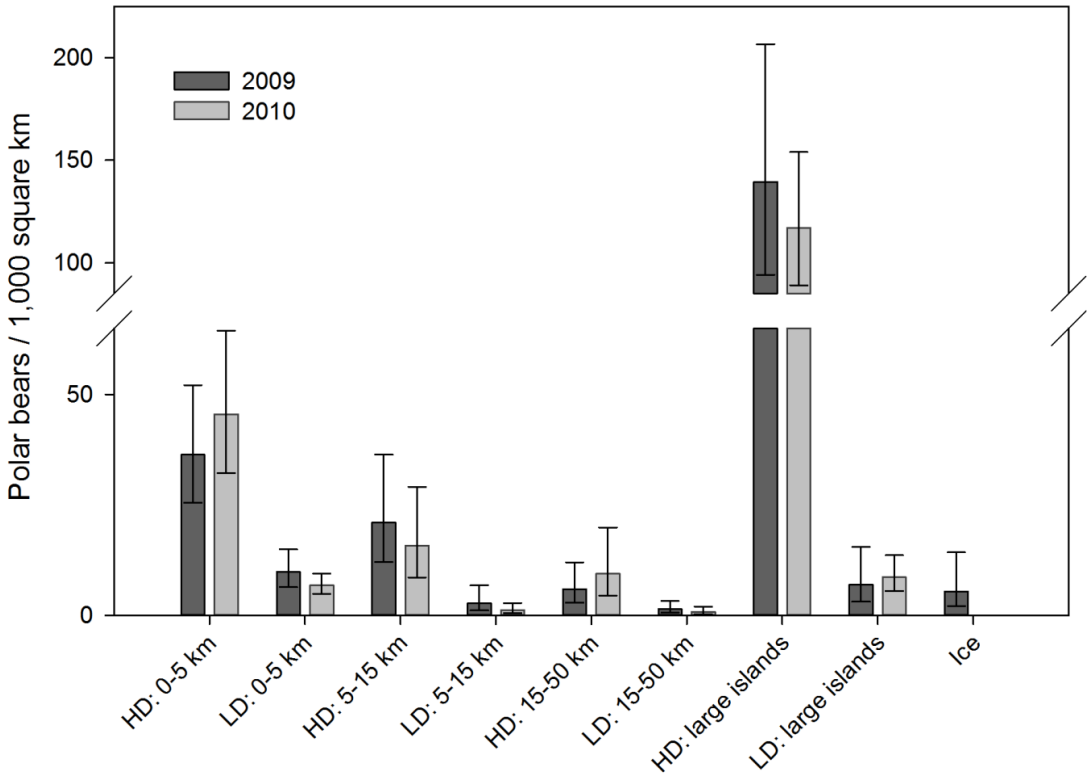
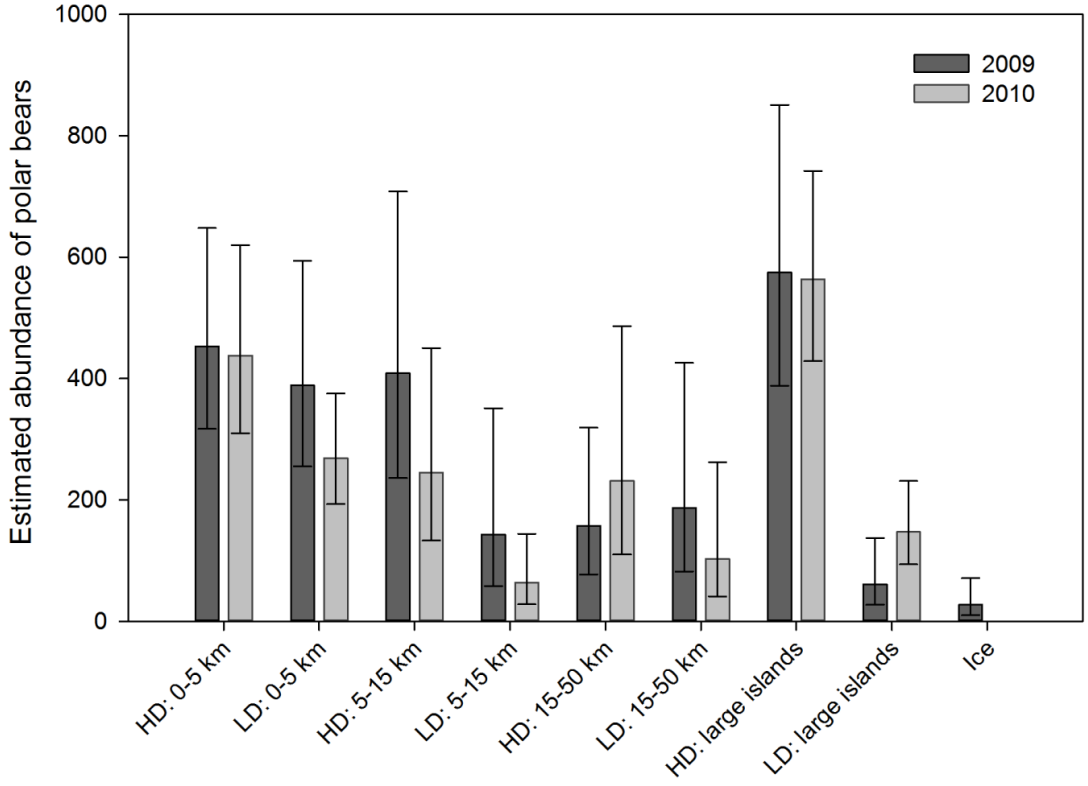


Figure 6**.** Abundances (± 95% confidence intervals) estimated during aerial surveys of the Foxe Basin polar bear population, late summers, 2009 and 2010. Densities are from the most highly supported distance sampling models and are categorized by stratum. Analyses included the entire perpendicular transect datasets for each year. HD: High density. LD: Low density. Strata measurements refer to the distance from the coast. For example, HD: 0-5 km refers to the high density stratum extending from the coastline to 5 km inland. Ice was not included as a stratum in 2010.



Chapter 2

REVISITING WESTERN HUDSON BAY:

USING AERIAL SURVEYS TO UPDATE POLAR BEAR ABUNDANCE IN A SENTINEL POPULATION

INTRODUCTION

Polar bears (*Ursus maritimus*) span the circumpolar Arctic, with an estimated 20,000 – 25,000 bears inhabiting 19 populations across 5 range states (Obbard et al. 2010). Although there are significant gaps in basic demographic information from portions of their range (Obbard et al. 2010), the Western Hudson Bay population (WH) in Canada ranks as one of the most intensively studied large mammal populations worldwide, with a research program dating back more than 4 decades (Jonkel et al. 1972, Stirling et al. 1977, Derocher and Stirling 1995a, Regehr et al. 2007).

Scientific evidence from the long-term capture and tagging program in WH suggests that the abundance of polar bears increased during the 1970s, remained stable for a period in the 1980s, and decreased by about 22% between 1984 and 2004 (Derocher and Stirling 1995a, Lunn et al. 1997, Regehr et al. 2007). The recent decline in abundance has been attributed to earlier sea ice breakup in Hudson Bay (Regehr et al. 2007). This trend in sea ice breakup and the resultant extension of the ice-free season have forced bears to spend longer periods on land without access to seals, their primary food source, leading to declines in survival, reproductive output, and body condition (Stirling et al. 1999, Regehr et al. 2007). Concurrently, an increase in incidences of human-polar bear conflicts in WH has been interpreted as a sign that the population is

undergoing significant change and has created public safety concerns (Stirling and Parkinson 2006, Towns et al. 2009, Peacock et al. 2010, Government of Nunavut, unpublished data). Bears in poor condition may exhibit an increased tendency to seek alternative food sources such as those around settlements and camps (Stirling and Parkinson 2006). Population viability analysis based on 2004 demographic data (Regehr et al. 2007) predicts that WH abundance has continued to decline over the past decade (Obbard et al. 2010). Additionally, climate models project that sea ice habitats in Hudson Bay will deteriorate, resulting in further impacts on polar bears (Amstrup et al. 2008).

There is general consensus between science and the traditional ecological knowledge (TEK) and observations of Inuit living along Hudson Bay that polar bear abundance in WH increased during the 1970's (Tyrell 2006). There is also agreement that polar bear distribution has changed, bear sightings have increased around communities, sea ice breakup is occurring earlier, and climate change is negatively influencing seal populations (Nunavut Wildlife Management Board [NWMB] 2007). However, in contrast to scientific evidence, Inuit perceptions do not support the notion that abundance in WH has declined since the 1980s (Tyrell 2006). Reports of more bears summering on land in northwestern Hudson Bay and increased incidences of problem bears around camps and communities instead have been attributed to several factors including increased abundance and an overpopulation of bears (Stirling and Parkinson 2006, NWMB 2007). This disparity between scientific findings and TEK has generated significant debate over the management and conservation of WH and led to calls for new

research to inform status assessment and resolve apparent differences between knowledge sources.

Physical capture forms the basis for our current understanding of polar bear ecology and facilitates a variety of research initiatives (e.g., habitat use and movements via satellite telemetry). However, among Inuit in the region, requests for new information also have included a desire to see alternative research methods employed. These concerns fall into 3 categories. First, although several studies have failed to detect impacts on body condition, survival, and reproduction resulting from polar bear capture and handling (Ramsay and Stirling 1986, Amstrup 1993, Derocher and Stirling 1995b, Messier 2000), concerns remain about the invasiveness of this method (e.g., Dyck et al. 2007). Second, the capture and marking of polar bears is viewed by many Inuit to be inconsistent with their cultural beliefs regarding human interactions with animals. Third, most polar bear research in WH has focused on the capture of bears in Manitoba, within a core study area that comprises the central and most densely occupied portion of the population's summer range (Figure 1). Multiple capture-based studies suggest that focusing research in this region has not significantly biased mark-recapture (MR) estimates of abundance and survival (e.g., Lunn et al. 1997, Regehr et al. 2007). Nevertheless, Inuit contend that a significant and increasing number of bears are spending the ice-free period outside the core study site; if true, failure to extend sampling across the entire WH would have negatively biased abundance and survival estimates (NWMB 2007).

To better inform status assessment, we conducted a comprehensive aerial survey of WH during the 2011, late summer ice-free period. Whereas aerial surveys are well-

established and widely used to estimate abundance of other species, their application to polar bears has been limited. Recent studies in the Barents Sea (Aars et al. 2009) and Foxe Basin (this volume, chapter 1) suggest that aerial surveys may be used to successfully estimate polar bear abundance in certain conditions. Because WH has been the site of an intensive capture program, it provides the opportunity to advance aerial survey development. Specifically, although capture research has focused in a core study area, MR estimates are considered to represent the entire WH population (e.g., Obbard et al. 2010). Direct comparison of aerial survey and capture-based estimates enables an assessment of the methods' potential biases and precision, promotes the acceptance of new techniques in the scientific community, and may suggest possible modifications in monitoring methods.

METHODS

Study Area

WH, located at the southern extent of the global polar bear distribution, stretches across roughly 435,000 km^2 of Hudson Bay and the adjacent coastal regions including portions of the Nunavut Territory and the provinces of Manitoba and Ontario (Figure 1). The region is seasonally free of sea ice, the primary habitat of polar bears, from about July through November.

WH shares borders with the Southern Hudson Bay and Foxe Basin populations. Boundary delineation was based on data derived from a variety of sources, including capture and recovery (Stirling et al. 1977, Derocher and Stirling 1990, Kolenosky et al. 1992, Taylor and Lee 1995, Derocher et al. 1997, Lunn et al. 1997), aerial surveys

(Stirling et al. 2004), satellite telemetry (Stirling et al. 1999, Peacock et al. 2010), and genetic analysis (Paetkau et al. 1995, 1999, Crompton et al. 2008). Although the boundaries are semi-discrete and interchange occurs among neighboring populations (Stirling et al. 1999, Crompton et al. 2008), their separation is most complete during the late summer and early fall ice-free period (Peacock et al. 2010).

Survey Design and Field Methods

We conducted an aerial survey during August, 2011, early in Hudson Bay's ice-free season. This period was selected for a number of reasons. First, bears are largely confined to land at this time, minimizing the survey area. Second, overlap with neighboring populations is at a minimum, since polar bears exhibit a high degree of site fidelity when ashore (Derocher and Stirling 1990, Lunn et al. 1997, Stirling et al. 2004, Parks et al. 2006). Third, the absence of ice and snow in late summer makes polar bears readily observable against a dark landscape. Finally, during August, few bears were likely to have started maternity denning (Clark et al. 1997, Clark and Stirling 1998, Lunn et al. 2004, Richardson et al. 2005) or making the seasonal, directional movements that typically occur prior to the formation of new sea ice (e.g., Stirling et al. 1977, Derocher and Stirling 1990, Stirling et al. 2004).

We implemented a systematic, stratified study design. We considered multiple sources of information to define the inland extent of the study area and delineate strata, including: (1) published information on the distribution of bears (e.g., Derocher and Stirling 1990, Lunn et al. 1997, Stirling et al. 2004, Richardson et al. 2005, Towns et al. 2010); (2) pilot aerial survey data collected during 2010 in northern WH, outside the

historical core study area (Stapleton et al., unpublished data); (3) local knowledge about bear distribution in northern WH provided by Inuit hunters from Nunavut; (4) capture records in Manitoba from 2003 – 2010 (n = ~700 records of independent bears; Environment Canada, unpublished data); (5) coastal and denning aerial surveys of portions of central and southern WH (Stirling et al. 2004, Manitoba Conservation, unpublished data); and (6) recent satellite telemetry data on the movements of collared polar bears (n = 12 bears in summer, 2010; A. Derocher et al., University of Alberta, and Environment Canada, unpublished data).

We defined 4 strata based on expected polar bear densities: (1) a high density zone corresponding to the historical core study area, including Wapusk National Park and extending up to 100 km inland; (2) a moderate density stratum, extending from the shoreline to 15 km inland elsewhere in Manitoba as well as Ontario; (3) a low density zone, from 15 km to 60 km inland in Manitoba and Ontario; and (4) a low density Nunavut stratum, extending from the coastline to 60 km inland from the Nunavut – Manitoba border to the community of Arviat, and from the shoreline to 50 km inland from Arviat to the northern boundary of WH (Figure 1). The Nunavut stratum also included 2 large islands.

We used a combination of overland transects, coastal contour transects, and small island sampling to survey WH. Polar bears tend to congregate along or near the shoreline during the ice-free season (Derocher and Stirling 1990, Towns et al. 2010), so overland transects were oriented roughly perpendicular to the coast (i.e., against the coastal density gradient; hereafter perpendicular transects) to improve precision and minimize potential

biases using distance sampling (Figure 1; Buckland et al. 2001). Transects also were extended over any exposed tidal flats. Because it was not possible to accurately delineate exposed flats in a GIS, polar bears sighted on tidal flats were considered to have occurred on the closest land for analysis. Sampling of tidal flats with perpendicular transects occurred at the same intensity as the nearshore inland strata, so any effect on the abundance estimate was negligible.

Perpendicular transects spanned from the shoreline up to 50 – 100 km inland. After reaching the most inland point, we flew roughly parallel to the shoreline to join the adjacent perpendicular transect and returned to the coast. Data collected during this cross-leg were generally not included in analyses. However, for 3 pairs of transects, we were unable to reach the far inland extent of the stratum due to logistical constraints. To incorporate sampling in the far inland portions of the strata in these instances, we included data collected along this cross leg.

Survey effort was allocated to maximize encounters while ensuring adequate coverage of all strata. Because polar bears are concentrated along the coast, we focused sampling in the nearshore inland zone. We also heavily sampled the high density stratum, which is a well-documented denning site (e.g., Richardson et al. 2005). Perpendicular transects were systematically spaced at 6, 7, and 10-km intervals in the high density, moderate density, and Nunavut low density strata, respectively (Figure 1). Every other pair of transects in the moderate density zone in Manitoba and Ontario was extended through the low density (far inland) stratum, such that transect spacing there averaged 14 km.

We also conducted separate surveys following the contour of the entire WH coastline. These coastal contour transects were flown at or slightly below the high water line with one side of the aircraft dedicated to monitoring tidal flats and nearshore waters as the other side surveyed the strip of land along the shoreline. We flew coastal contours as close to high tide as possible to minimize tidal flat exposure. Because perpendicular transects were extended to the shoreline and over tidal flats, some bears along the shore could be sighted from both perpendicular and coastal transects. This design enabled us to estimate the abundance of bears in the coastal region with either perpendicular transects or coastal contour transects. Independent coastal and perpendicular transect data were treated separately to ensure that bears were not double-counted in abundance estimates (see Analyses). As with perpendicular transects, bears sighted on tidal flats or in nearshore waters were considered to be on land in order to calculate density and extrapolate to unsurveyed areas. We additionally sampled as many small islands as possible.

We surveyed the Nunavut and Manitoba – Ontario portions of the aerial survey from fixed wing (de Havilland DHC-2 MKIII Turbo Beaver) and helicopter (Bell 206L) platforms, respectively. Separate platforms were used to complete the survey within a narrow window of time and to enhance opportunities for participation by local people. The fixed wing survey crew consisted of 4 dedicated observers, with front and rear observer teams each comprised of 2 spotters, as well as a data recorder. With the helicopter, the pilot and observer in the co-pilot seat comprised the first team, and 2 individuals seated in the rear comprised the second team.

For each aircraft, we employed sampling protocols that facilitated the collection of data for both distance sampling (Buckland et al. 2001) and sight-resight (i.e., double observer; Pollock and Kendall 1987) analyses. Front and rear seat observers could not see each other, and their sightings were not announced until both teams were afforded a full opportunity to independently spot a bear. Transects were flown at an above-ground level altitude of about 120 m and groundspeed of roughly 160 km/hr with both platforms.

We recorded flight paths and bear locations at the time of first observation via GPS and measured perpendicular distances from sighted bears to the flight path in a GIS (adapted from Marques et al. 2006). We recorded group size and estimated sex and age class based on morphological characteristics. We defined a group as multiple individuals whose detections were non-independent (e.g., a family group including an adult female and her cubs or a fraternity of 2 or more adult males). For each sighting, we also recorded factors that may have influenced detection probability, including weather conditions, activity when first observed, and habitat characteristics (e.g., habitat structure within 30 m of an individual bear that may impede detection; qualitative 1 - 3 scale).

During the late summer and early fall, polar bears in WH, particularly pregnant females, may retreat to earthen dens (Jonkel et al. 1972, Lunn et al. 2004). Denning bears that are completely unavailable for sighting would be excluded from an aerial survey abundance estimate. We flew close to dens with recent digging or other signs of activity to determine if a bear was present.

Analyses

Perpendicular Transects

We used distance sampling (Buckland et al. 2001) to estimate abundance with data collected from perpendicular transects. We created 2 perpendicular transect datasets that 1) included sampling in the coastal zone and 2) excluded data from the coastal zone (bears were instead estimated with coastal contour transects). Histograms of sighting distances from the flight path suggested that detection declined predictably with increasing distance from the aircraft, indicating that distance sampling was an appropriate analytical method (Figure 2).

A key assumption of distance sampling is that sampling is random with respect to the distribution of bears (Buckland et al. 2001). Since polar bears concentrate along the shore during the ice-free season, we only used distance sampling to analyze data from perpendicular transects, which cut across this density gradient. Sighting distances from coastal contour transects may partially reflect the bears' density gradient, not just their sightability, and thus were inappropriate for distance sampling.

Detection of all objects on the transect line is another fundamental assumption of distance sampling (Buckland et al. 2001); violation of this assumption yields a negatively biased abundance estimate. Whereas conventional distance sampling and multiple covariate distance sampling (Marques and Buckland 2003) require perfect detection at distance 0 (i.e., from the flight path) to generate reliable abundance estimates, mark-recapture distance sampling (MRDS; Laake and Borchers, 2004) can correct for imperfect detection on the line using sight-resight data. In the helicopter, front observers could see the flight path, but rear observers had a 75-m blind spot on either side of the aircraft. Therefore, we initially examined a left-truncated dataset in which observations

within 75 m of the helicopter were censored and 75 m was subtracted from all other sighting distances (Borchers et al. 2006). This procedure established the transect line such that all bears were available to both teams of observers. Both teams of observers in the fixed wing had a blind spot of nearly 170 m on either side of the aircraft, so 170 m was subtracted from all observations to establish the transect line. Preliminary analyses of left-truncated double-observer data indicated that the probability of a bear near the transect line being detected by at least 1 observer was >97%. We thus considered the assumption of perfect detection at distance 0 to be approximately valid and proceeded with conventional and multiple covariate distance sampling analyses with the untruncated helicopter data. This approach enabled us to incorporate all sightings from perpendicular transects (i.e., including those within the rear observer blind spot), thereby increasing the number of observations and improving estimation of the detection function.

We initially fit detection functions using only data collected from the helicopter. We had insufficient data from the fixed wing to model a separate detection function, so we pooled sighting data from the helicopter with left-truncated data from the fixed wing (because none of the observers could sight bears within 170 m of the flight path). For the most highly supported models, pooling had a negligible impact on average detection probability and abundance estimates for the strata surveyed only from a helicopter. Hence, we proceeded with analyses incorporating untruncated data collected from the helicopter and the left-truncated data from the fixed wing. Additionally, we condensed the Nunavut stratum and the low density, far inland zone in Manitoba and Ontario into a single stratum due to limited encounters in these areas. Although sampling intensity was

greater in Nunavut, estimated densities were very low in these strata and individual encounter rates were similar.

We fit conventional distance sampling models in Program DISTANCE (Version 6.0, Release 2; Thomas et al. 2010) to evaluate detection functions and to assess whether group size influenced detection. Following this preliminary review, we fit all distance sampling models in the MRDS engine of Distance. Both datasets were modeled as single observer studies. Data were right-truncated at roughly 5% to smooth the tail of the detection function and improve model fit and parsimony (Buckland et al. 2001).

We fit distance sampling models with hazard and half-normal key functions, and we considered visibility (weather) and habitat structure within 30 m of a sighting as covariates in these models. We evaluated all combinations of key functions and covariates. Covariates were scored in the field on a 3-point scale, but we condensed these into binary categories because of underrepresentation of some values. We specified a global detection function and used stratum-specific encounter rates and group sizes to estimate density and abundance by stratum. Stratum abundance estimates were subsequently summed to obtain an overall abundance estimate.

We employed Akaike's Information Criteria for model selection (Burnham and Anderson 2002) and examined q-q plots and chi-square, Kolmogorov-Smirnov, and Cramér-von Mises tests to evaluate goodness of fit. Individual transects, within stratum, were considered sampling units for variance estimation. We used the Innes et al. (2002) method to estimate variance, since this technique does not require independence among variance components (i.e., stratum-specific abundance estimates were not fully

independent because we estimated a global detection function). We obtained model-averaged estimates for models within 2 ΔAIC for each dataset combination (Burnham and Anderson 2002). Model-averaging enabled us to account for variability in the estimation of the detection functions and associated densities.

Coastal Transects and Small Islands

We used mark-recapture models to obtain a separate estimate of coastal zone abundance from coastal contour transects with sight-resight data. We employed the Huggins model (Huggins 1989, 1991) to facilitate the inclusion of covariates to model variability in detection probabilities.

Front and rear observer teams comprised our first and second sampling periods, respectively, and we considered discrete groups of polar bears (as defined above) as the sampling unit. We sampled the coastal zone 500 m inland of the high-water line (since coastal contour transects were often flown below this line to improve coverage of the tidal flats) and censored sightings farther inland. We allowed detection probabilities to remain constant or vary between observers and used forward stepwise selection to evaluate covariates [habitat structure (binary); group size (1, 2, 3, ≥4); activity (binary)] potentially impacting detection. There was insufficient variability in other covariates to warrant their inclusion in modeling. Models were fit in Program MARK (White and Burnham 1999) and AIC adjusted for small sample sizes (AIC_c) was employed for model selection. We used detection probabilities from the most supported model and a generalized Horvitz-Thompson estimator to estimate the number of groups present in the sampled areas.

For small islands, the front team of observers spotted all groups that were sighted within the surveyed strip half-width of 750 m. Therefore, it was unnecessary to estimate individual detection probabilities via the Huggins model. For both the coastal contour transects and the small island sampling, we extrapolated group density estimates across the coastal zone and small islands and multiplied estimates by mean group sizes. We calculated group sampling variance following Buckland et al. (2001) and extrapolated and multiplied variances via the delta method (Powell 2007).

Total Abundance

Sampling and analytical protocols enabled us to generate 2 abundance estimates. One estimate was based on the complete set of perpendicular transects plus the small islands. The second was derived by summing estimates from perpendicular transects excluding coastal zone data, coastal contour transects, and small island sampling. We added point estimates from these components and summed their variances to obtain 2 population-wide abundance estimates. We assigned equal weights, averaged to obtain a final abundance estimate for WH, and estimated unconditional variance in a model averaging framework.

RESULTS

Sightings

During the 14 – 29 August, 2011, survey, we recorded 711 total polar bears, including 41 and 670 observations in the Nunavut and Manitoba – Ontario sections of WH, respectively (Figure 3). Sampling in Manitoba and Ontario, where >90% of sightings occurred, was completed within an 11-day period. Because the coastal contour

and perpendicular transects both covered the coastal zone, some bears were undoubtedly seen twice, but we were unable to calculate the number of unique bears that were sighted. However, sampling itineraries in Nunavut enabled us to estimate that no more than 31 unique bears were sighted there. Several aggregations of 4 or more bears, including 5 groups with 8 to 10 bears and a group with 21 individuals, were documented in southeastern WH (i.e., Area 2 in Figure 3) and near Cape Churchill (in Area 1). We calculated litter sizes and cub observations from all observations in Manitoba and Ontario and only unique bears in Nunavut; 50 cubs-of-the-year (coy) and 22 yearlings were observed. Mean litter sizes were 1.43 (SD: 0.50; n = 35) and 1.22 (SD: 0.43; n = 18) for coy and yearlings, respectively.

Distribution

Polar bear sightings were not uniformly distributed across WH (Figure 3). The greatest densities of bears occurred in the high density stratum (Area 1) and along the coast of southeastern WH. In general, observations were highly concentrated along or near the coast throughout the population (Figure 3). However, bears >10 km from the coastline were often recorded in the high density stratum (n = 49; 43% of bears sighted from perpendicular transects in the stratum) and less frequently observed in the southeastern portion of WH (n = 6; 8% of bears sighted from perpendicular transects in that region).

Abundance Estimation

Perpendicular Transects

We flew >7,800 km along perpendicular transects, including >2,750 km in the high-density stratum, nearly 1,100 km in the moderate density Manitoba – Ontario zone, and about 4,000 km in the Nunavut and low density (far inland) Manitoba – Ontario strata. After right truncation at about 5%, we included 139 and 62 polar bear groups in distance sampling analysis for the datasets that included and excluded the coastal zone, respectively.

Observed group sizes along perpendicular transects ranged from 1 to 8 ($\bar{x}$: 1.4; SD: 0.9). A group of 6 bears, including 2 family groups and 2 independent bears, was sighted in Nunavut, congregating around harvested whale carcasses. Because the low density Manitoba – Ontario and the Nunavut strata were pooled and we viewed this aggregation as an anomaly that would not reflect group sizes in interior Manitoba – Ontario, we instead calculated the stratum mean group size for this observation. Preliminary analyses did not indicate an effect of group size on detection probabilities, a finding that was consistent with our field observations. Goodness of fit metrics suggested adequate model fit for all highly supported models ($P > 0.05$ for all tests).

Model selection was similar among analyses and supported the inclusion of covariates to explain variability in detection probabilities (Table 1). Model-averaging yielded abundance estimates of $\widehat{N}$: 929 (SE: 186) and $\widehat{N}$: 561 (SE: 124) that included the coastal zone and excluded this region, respectively.

Coastal Transects and Small Islands

Nunavut's coastline is highly irregular in portions of northern WH, making it challenging to conduct and analyze a comprehensive coastal contour transect. This

reality, coupled with the low number of groups observed in the pooled low density stratum (n = 6), compelled us to rely exclusively on perpendicular transects to estimate coastal zone abundance in Nunavut. In Manitoba and Ontario, however, we sampled >95% of the coastline and included 190 polar bear groups in sight-resight analysis. Our highest ranked model incorporated separate detection probabilities for the front and rear observers and covariates for habitat structure and group size. Detection and abundance estimates were very consistent among the best supported models. Thus, we used detection probabilities from the most highly supported model ($\bar{p}_{\text{front}}$: 0.97, SE: 0.014; $\bar{p}_{\text{rear}}$: 0.86, SE: 0.027) to generate a group abundance estimate ($\hat{N}$: 192 groups; SE: 1.7) for the sampled areas and inflated across the entire coastal zone. Multiplying by mean group size ($\bar{x}$: 1.45; SD: 1.6) yielded a coastal zone abundance estimate of 291 (SE: 23.8) polar bears in Manitoba and Ontario.

We sampled about 85% and 60% of total island area in Manitoba – Ontario and Nunavut, respectively, observed 102 and 9 bears and obtained estimates of 120 (SE: 19.8) and 15 (SE: 1.6) bears on and near small islands in the 2 areas. Additionally, 2 groups totaling 4 bears were sighted beyond the maximum inland extent of the defined study site (>75 km and >60 km inland, respectively) during ferry flights between transects. Because we were unable to incorporate these individuals elsewhere in the analysis, these bears were added to final calculations. We observed no bears in dens during the survey.

Total Abundance

Summing estimates from the perpendicular transects including the coastal zone, small island sampling, and the bears observed beyond the extent of the inland strata

yielded an abundance estimate of 1,068 (SE: 187) bears. Estimates from coastal contour transects, perpendicular transects excluding the coastal zone data, and small island sampling, as well as the bears sighted beyond the extent of the inland strata, produced a total abundance of 991 (SE: 128). Averaging these estimates yielded an abundance of 1,030 (CV: 16.0; 95% lognormal CI: 754 – 1,406) for WH during the 2011 ice-free season.

DISCUSSION

Distribution

Because the aerial survey was systematic and comprehensive, our data provide unique insights into the distribution and densities of bears across the entirety of WH. Residents of communities along the Nunavut coastline of WH report that encounters with polar bears have been increasing since the 1970's (Tyrell 2006), resulting in a perception among local communities that abundance has increased and that a significant proportion of bears are now summering outside the core MR study area (Dowsley and Taylor 2006, NWMB 2007). However, aerial survey data did not indicate a large-scale range shift; only about 6% of sightings during the 2011 survey occurred in Nunavut. This finding is consistent with previous research and suggests that the vast majority of individuals within WH still summer in Manitoba.

Similar to previous studies (e.g. Stirling et al. 1977, Derocher and Stirling 1990, Lunn et al. 1997, Towns et al. 2010), we found marked differences in polar bear distribution comparing 2 broad geographic regions in Manitoba and Ontario (previously delineated by Stirling et al. 2004; Figure 3). In Area 1 (including the core study area of

the MR work), the highest densities of bears occurred along the coastline, but we also encountered a significant number of individuals >10 km inland, mostly within Wapusk National Park (Figure 3). In contrast, virtually all polar bears in Area 2 (i.e., southeastern WH) were highly concentrated in a relatively narrow strip along the coast. These differences in distribution have been well-documented previously and attributed to several factors, including variation in the availability of suitable inland habitats for activities such as denning, avoidance of conspecifics, and thermoregulation (Stirling et al. 1977, Derocher and Stirling 1990, Lunn et al. 1997, Clark and Stirling 1998, Richardson et al. 2005).

Because nearly half of the sightings in Manitoba – Ontario occurred in southeastern WH, outside the core MR study area, we reviewed multiple lines of evidence to examine the hypothesis that the high proportion of bears encountered there was an anomaly reflecting temporary immigration of bears from the adjacent Southern Hudson Bay (SH) population. There is not strong support for this hypothesis. First, historical data indicate that there is little overlap during the ice-free season and high fidelity to on-land areas (e.g., Lunn et al. 1997, Stirling et al. 2004). Second, although they represent a small (n=7) and sex-biased (i.e., all females) sample, bears outfitted with satellite collars in SH during 2011 did not exhibit unusual movements during the ice-free season and were well within SH's bounds during the late summer and fall (M. Obbard and K. Middel, Ontario Ministry of Natural Resources, unpublished data). Third, an aerial survey of SH was conducted in Ontario, where most bears in that population summer (Obbard et al. 2007), during September, 2011, and in Quebec and offshore

islands the following year (Obbard et al. 2013). The abundance estimate of SH (951, 95% CI: ~662 – 1,366) derived from this aerial survey is consistent with other recent abundance estimates (900 – 1,000; Obbard et al. 2007), suggesting that a large influx of bears from Southern Hudson Bay is unlikely to have significantly contributed to the high densities of bears observed in WH. Finally, the high number of bears sighted along the Area 2 coastline is consistent with long-term data from annual coastal surveys that show a trend of increasing use of this region (Stirling et al. 2004, Manitoba Conservation, unpublished data; Figure 4).

Abundance Estimation

Aerial Survey-Based Estimation

We generated an aerial survey estimate of abundance for WH using a combination of sampling and analytical techniques. While our results provide an estimate of current polar bear abundance, this figure alone does not indicate population status or trend. Multiple surveys repeated at regular intervals would be required to assess trend.

Our abundance estimate in which bears along the shore were estimated from perpendicular transects extending through the coastal zone (i.e., distance sampling) was consistent with our estimate in which the coastal zone abundance was based on separate contour transects (i.e., sight-resight). To incorporate uncertainty in model selection and estimated detection functions as well as variability between techniques, we used model-averaging in the analyses. This procedure slightly inflated precision. However, we believe that it resulted in an estimate that better reflected true abundance (Anderson et al. 2000).

We sampled from both a helicopter and a fixed wing aircraft during the aerial survey due to logistical constraints. Insufficient detections from the fixed wing compelled us to pool data from the 2 platforms in our analyses. Our preliminary analyses including and excluding the fixed wing data from modeling and our experiences with helicopter and fixed wing surveys elsewhere suggest that pooling data from the platforms had a negligible effect on our results. The consistency in the number and distribution of sightings in Nunavut between this study and previous, helicopter-based research (e.g., Peacock and Taylor 2007) also support this assertion. Moreover, distance sampling models are robust to pooling of data with different detection functions (Buckland et al. 2001).

Methodological Assumptions

Like all statistical methods, the ability of distance sampling to generate a reliable (unbiased) abundance estimate is contingent on meeting a set of assumptions. We attempted to minimize potential biases in the aerial survey through study design. Specifically, we surveyed with systematically spaced transects oriented against the coastal density gradient to ensure random sampling with respect to the distribution of bears. We also examined left-truncated data to account for blind spots and evaluated detection near the transect line with double-observer models, which we estimated at >97%. This result suggests that we detected virtually all animals on the transect line.

Detection of bears at their initial location (i.e., before responding to approaching observers) is another core distance sampling assumption (Buckland et al. 2001). Because polar bears in WH have been subject to an annual capture program over the past several

decades, we hypothesized that they may have moved in response to an approaching aircraft. However, >75% of bears sighted within 500 m of the aircraft along perpendicular transects were stationary when first detected, and sighting distance histograms (Figure 2) did not suggest significant responsive movement (sightings peaked in the distance bins closest to the transect). Additionally, the survey was flown at speeds much faster than a polar bear can travel, so the impact of movements prior to detection was likely minimal (Buckland et al. 2001).

Finally, accurate measurement of distances to sightings from the transect path is critical (Buckland et al. 2001). We used methods involving GPS and GIS technology adapted from Marques et al. (2006) that we previously implemented in large-scale polar bear aerial surveys in Foxe Basin (this volume, chapter 1). We are confident that our measures of perpendicular distance between the aircraft flight path and polar bears were accurate. Because our study met and evaluated fundamental distance sampling assumptions through proper study design and analysis, the aerial survey additionally fulfilled the implicit assumption that polar bear distances from the transects (i.e., observed and unobserved bears) followed a uniform statistical distribution (Fewster et al. 2008).

Abundance estimates derived from mark-recapture (sight-resight) models will be negatively biased if heterogeneity in detection probabilities is not sufficiently modeled (Otis et al. 1978, Pollock et al. 1990). Here, sight-resight results may have been particularly susceptible to underestimation since observations by front and rear observers were nearly instantaneous and from very similar vantage points. In other words, sightings

were not entirely independent (e.g., both observers may have been more likely to miss difficult-to-spot bears), potentially yielding an overestimate of detection. However, we adopted a conservative strip width and included multiple covariates to explain variability in detection. The habitat along the coast and on small islands also generally presented excellent sighting conditions, reducing the likelihood that a significant source of heterogeneity was not included in modeling.

Other Potential Biases

Several other factors may affect the accuracy of an aerial survey in WH. Available evidence suggests that our study area encompassed nearly all bears located within the bounds of the WH population during August, 2011. However, 2 groups were sighted beyond the inland extent of the study area, indicating that our delineation was not fully comprehensive. Polar bears located far from the coast during the ice-free season have been occasionally reported in the region, including a bear sighted in northeastern Saskatchewan, more than 400 km from the Hudson Bay coastline, during 1999 (Goodyear 2003). While it is impossible to quantify the extent of these occurrences, we believe such far inland bears are rare.

We also cannot discount the presence of some bears in far offshore waters during the survey period. Although we extended perpendicular transects over tidal flats and surveyed during ferry flights between small islands, safety concerns and logistical efficiency precluded systematically and intensively surveying offshore waters. Telemetry data indicate that bears predictably come ashore as sea ice melts and breaks up, rather than remaining in offshore, ice-free waters (Stirling et al. 1999, Parks et al. 2006, Cherry

et al. 2013). Arrival on land occurs 3 to 4 weeks after ice breakup, defined as the date at which total ice cover decreases to 50% (Stirling et al. 1999) or 30% (Cherry et al. 2013). Polar bears in WH remain onshore throughout the ice-free season (Stirling et al. 1977) and their movements are markedly reduced (Parks et al. 2006). In 2011, Hudson Bay was completely ice-free several weeks prior to the commencement of the aerial survey (Canadian Ice Service regional charts, available: http://ice-glaces.ec.gc.ca/). In Manitoba, our sampling itinerary began at the Nunavut border and continued southward, such that southeastern WH, where sea ice tends to persist longest, was surveyed about 4 weeks after the last remnant ice floes had melted. Thus, any bears swimming in WH's offshore waters likely represent a negligible portion of the total population.

Polar bears that are entirely hidden from observation are not incorporated in an aerial survey abundance estimate. Such availability bias could arise from 2 sources. Whereas much of the WH study area consists of open coastal plains or tundra, dense vegetation and small trees encountered in some inland regions may completely obscure some polar bears from view. While we are unable to quantify such availability bias, our impression in the field was that although trees and brush impeded detection (e.g., habitat structure was an important covariate in modeling detection functions), it is likely that few, if any, bears were completely concealed by vegetation. Second, polar bears in WH, particularly pregnant females, may use dens during the ice-free season, entering them as early as August (Stirling et al. 1977, Clark et al. 1997, Clark and Stirling 1998, Lunn et al. 2004, Richardson et al. 2005). We cannot correct for bears that were underground during the survey, but several pieces of evidence suggest that this issue was rare. We

observed numerous dens, some signs of recent digging, and sighted bears of various sex and age-classes in known denning areas (i.e., the high density stratum). However, we did not document any bears in dens or near mouths of dens, suggesting that overall denning activity was low during the survey. Additionally, more than 50% (28 of 54) of bears classified as adult females in Manitoba and Ontario were solitary. These presumably pregnant bears (i.e., the reproductive class most likely to enter dens in late summer and fall) were proportionately more abundant than females that had cubs the previous year (with coy) or the year before (with yearlings). These findings suggest that few bears were missed in dens. Nevertheless, any availability bias arising from bears being obscured in dens or by vegetation would cause our estimate to be negatively biased.

Comparison with Mark-Recapture Estimation

Our 2011 aerial survey results are consistent with a 2004 estimate of abundance based on MR (935; 95% CI: 794 – 1,076; Regehr et al. 2007). However, previous analyses based on vital rates estimated from capture-based studies suggested that abundance would continue to decline beyond 2004 (Regehr et al. 2007, Obbard et al. 2010). Thus, we expected that our abundance estimate would be substantially less than the 2004 estimate. A review of how MR has been implemented in WH is informative for evaluating potential differences between aerial survey and MR-derived estimates of abundance.

Equal probability of detection is a key assumption of capture-based methods. Unmodeled heterogeneity in capture probabilities produces a negatively biased abundance estimate and may impact survival estimates (Pollock et al. 1990). Thus,

obtaining a random sample of individuals that represents the entire population of interest (or completely modeling unequal capture probabilities to eliminate capture heterogeneity) is necessary to generate reliable results.

In WH, sampling effort for MR historically concentrated around Churchill and in Wapusk National Park (i.e., the high density stratum, Figure 1; e.g., Regehr et al. 2007), with limited and less frequent sampling elsewhere, such as southeastern WH (e.g., Area 2, Figure 3; Lunn et al. 1997). Despite this geographically limited sampling, MR analyses have been considered to reflect abundance and trends for the entire WH population (Regehr et al. 2007; Obbard et al. 2010), generally under the assumption that adequate 'mixing' ensures random sampling.

However, several lines of evidence suggest that this sampling strategy may have resulted in biased parameter estimates. Multiple studies have reported that polar bears in WH show a high degree of geographic fidelity within and between ice-free periods (Derocher and Stirling 1990, Stirling et al. 2004, Parks et al. 2006). Such site fidelity suggests that sampling a limited portion of WH may yield an estimate that includes only those bears that used the sampled area, not the entire population.

Additionally, the results of the aerial survey demonstrate that a significant proportion of bears are found outside the core MR study area in late summer, when most capture work historically has occurred (e.g., Regehr et al. 2007). Very low densities of polar bears in the Nunavut portion of WH during the early ice-free season suggest that any bias arising from limited sampling in this region is likely minimal. Conversely, high densities of bears along the coast in southeastern WH represent a large and seemingly

increasing proportion of the population (Figure 4); failure to adequately sample this region could negatively bias abundance estimates and obscure population trends. Lunn et al. (1997) reported that MR estimates of population size did not differ based on the inclusion or exclusion of capture data from southeastern WH. However, sampling was limited and inconsistent in this region compared to the extensive, long-term sampling within the core study area. In contrast, Regehr et al. (2007) noted disparities in comparing abundance estimates derived from 2 MR datasets and attributed this finding to under-sampling of subadults that tended to occupy areas closer to Churchill; the dataset that yielded a lower abundance estimate included less sampling around Churchill. Combined, these observations suggest that spatially limited sampling (i.e., concentrated within the densest region) may have resulted in an underestimate of abundance of WH.

The aerial survey represents the first systematic and geographically comprehensive survey of polar bears in WH. Thus, the results better represent the entire population than MR estimates based on sampling within a core study site. As such, WH appears to have more polar bears than previously thought. Our estimate, however, does not necessarily negate the declining trends in abundance, survival, reproductive output, and body condition reported from WH (Stirling et al. 1999, Regehr et al. 2007). Because the aerial survey reflects a larger effective area (the entirety of WH) than the MR studies (a smaller proportion of the population), results from the two methods cannot be directly compared to assess population trend. It would be erroneous to suggest that our estimate indicates that the population is not declining. Indeed, the aerial survey indicated that reproductive performance in WH during 2011 was relatively poor, consistent with

findings of capture-based studies (Regehr et al. 2007). Mean litter sizes and the proportions of cubs-of-the-year and yearlings recorded in WH were low relative to adjacent populations (Table 2), suggesting that WH is currently less productive than other populations in the Hudson Bay complex and nearby regions (Peacock et al. 2010). Nevertheless, the apparent increased use of southeastern WH, coupled with the lack of sampling there, could result in an inaccurate assessment of trends in abundance, survival, and other measures of population status. The differences in sampling frames and associated uncertainties must be clearly communicated to decision-makers, and southeastern WH should be fully integrated in future studies to evaluate demography and status of polar bears across the population. There are several fundamental questions about how this region functions within the broader Hudson Bay polar bear complex, including the suitability of current population delineation, the discreteness of bears that summer in southeastern WH from bears that summer elsewhere in the population, and the susceptibility of these bears to harvest by communities in Nunavut.

CONCLUSIONS

Recent changes in regional sea ice (Gough et al. 2004, Gagnon and Gough 2005, Scott and Marshall 2010) and reported deleterious impacts on body condition and vital rates (e.g., Stirling et al. 1999, Regehr et al. 2007) have placed Hudson Bay at the forefront of polar bear conservation and management. Indeed, polar bears in Western Hudson Bay are often cited as the most visible and dramatic example of the early impacts of climate change. The aerial survey results should not necessarily alter that impression, nor do they provide a more optimistic outlook for polar bears in the Hudson Bay region.

However, our findings do highlight the need to identify the limitations of both aerial survey and MR sampling programs and to reconcile potentially conflicting results in order to correctly assess population status and quantify the impacts of climate change. Our results further suggest that the systematic and comprehensive sampling of aerial surveys can effectively complement capture-based initiatives, providing a sound means to track abundance and distribution. Their application may become particularly important in WH and similar populations subject to changing environmental conditions, given the increasing need for rapid dissemination of information, the strong objections by Inuit to physical capture, and the possible effects of capture on bears in a declining state of health.

Table 1. Summary of most supported models (ΔAIC < 2) for distance sampling analyses of the WH polar bear aerial survey, conducted during August, 2011. In the column *Model*, the first term signifies the key function and subsequent terms represent covariates (Struc = Habitat structure within a 30 m radius of the polar bear; Vis = visibility).

Dataset	**Model**	**ΔAIC**	**Parameters**	**Global density[1] (bears per km^2)**	**Coefficient of variation (%)**
All Inland Sightings	Half-normal / Struc + Vis	0.000	3	0.011	17.6
	Hazard / Struc + Vis	0.106	4	0.013	20.7
Sightings excluding coastal zone	Half-normal / Struc + Vis	0.000	3	0.007	20.3
	Hazard / Vis	0.331	3	0.008	23.3
	Hazard / Struc + Vis	1.030	4	0.008	24.2
	HN / Vis	1.059	2	0.007	18.8
	HN / Struc	1.630	2	0.007	20.0

[1]Global density estimates refer to density within the region estimated by distance sampling. For example, datasets excluding sightings in the coastal strip do not incorporate those bears in the global density estimate.

Table 2. Polar bear litter sizes and number of dependent offspring observed (as proportion of total observations) during recent ice-free season studies in central and eastern Canada. Data are presented as mean (standard error).

Population	**Litter Size**		**Proportion of Total Observations**		**Source**
	Cubs of the year	*Yearlings*	*Cubs of the year*	*Yearlings*	
Western Hudson Bay (2011)	1.43 (0.08)	1.22 (0.10)	0.07	0.03	This study
Southern Hudson Bay (2011)	1.56 (0.06)	1.54 (0.08)	0.16	0.12	Obbard et al. 2013
Baffin Bay (2011)	1.57 (0.06)	1.51 (0.09)	0.19	0.10	Government of Nunavut (unpublished data)
Foxe Basin (2009-2010)	1.54 (0.04)	1.48 (0.05)	0.13	0.10	this volume, chapter 1
Davis Strait (2005-2007)	1.49 (0.15)	1.22 (0.28)	0.08	0.09	Peacock et al. (2013)

Figure 1. Strata and planned survey transects for the Western Hudson Bay polar bear aerial survey, August, 2011. The Western Hudson Bay population is highlighted in red in the inset. Hashed lines denote territorial and provincial borders, and red stars mark communities.

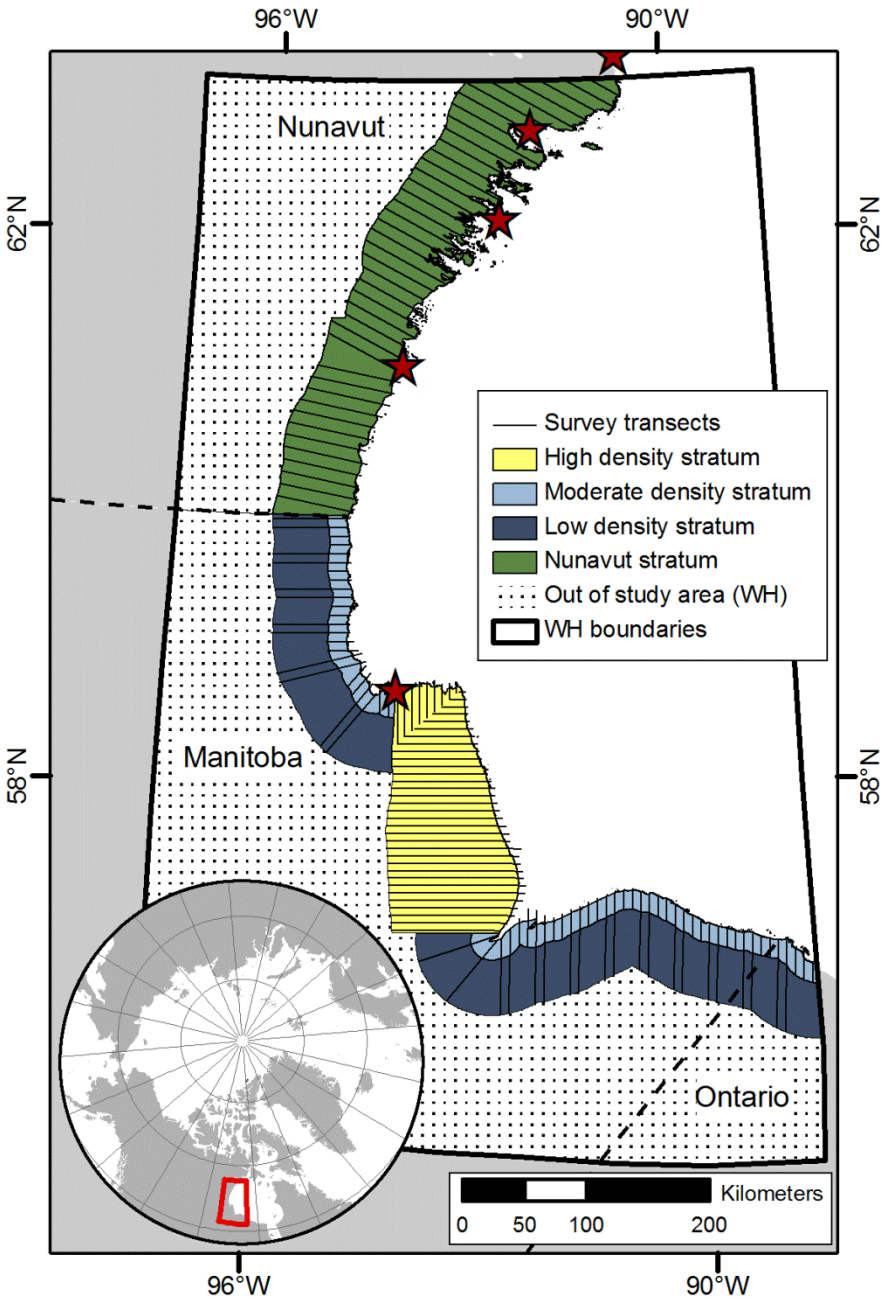


Figure 2. Distribution of polar bear sighting distances from the original transect line on perpendicular transects, Western Hudson Bay, August, 2011. All strata are pooled. The top histogram includes all sightings, including perpendicular transects extending through the coastal zone. The bottom only includes sightings inland of the coastal zone.

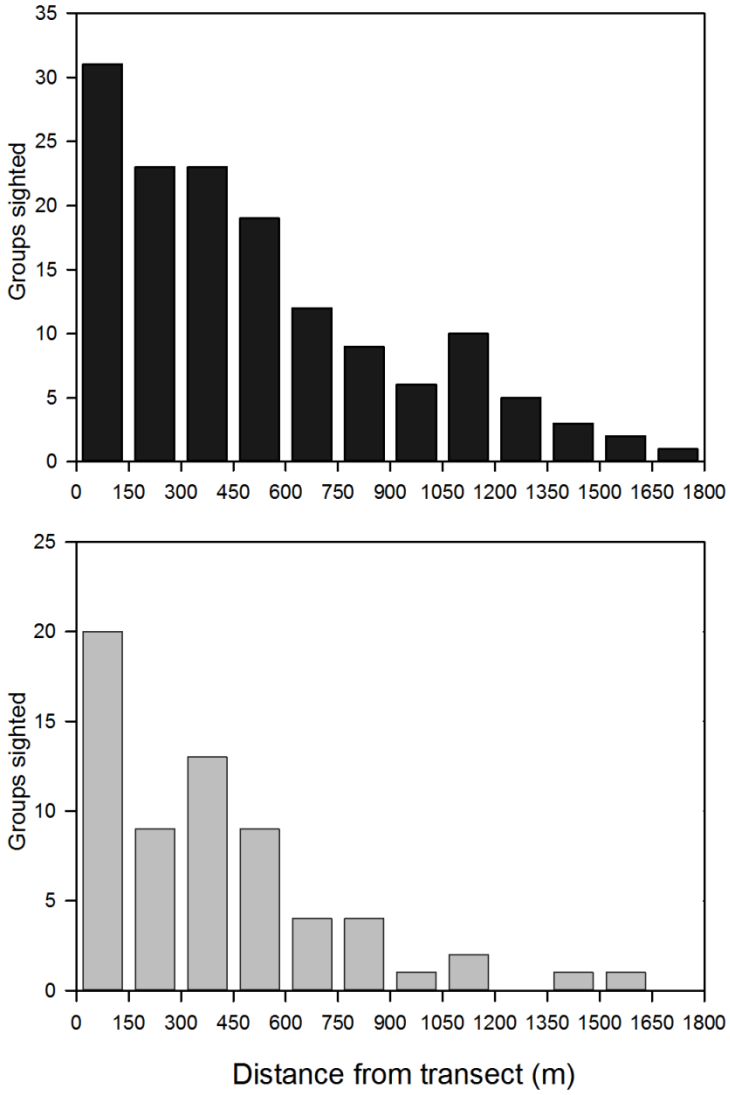


Figure 3. Polar bear sightings recorded during the Western Hudson Bay aerial survey, August, 2011.

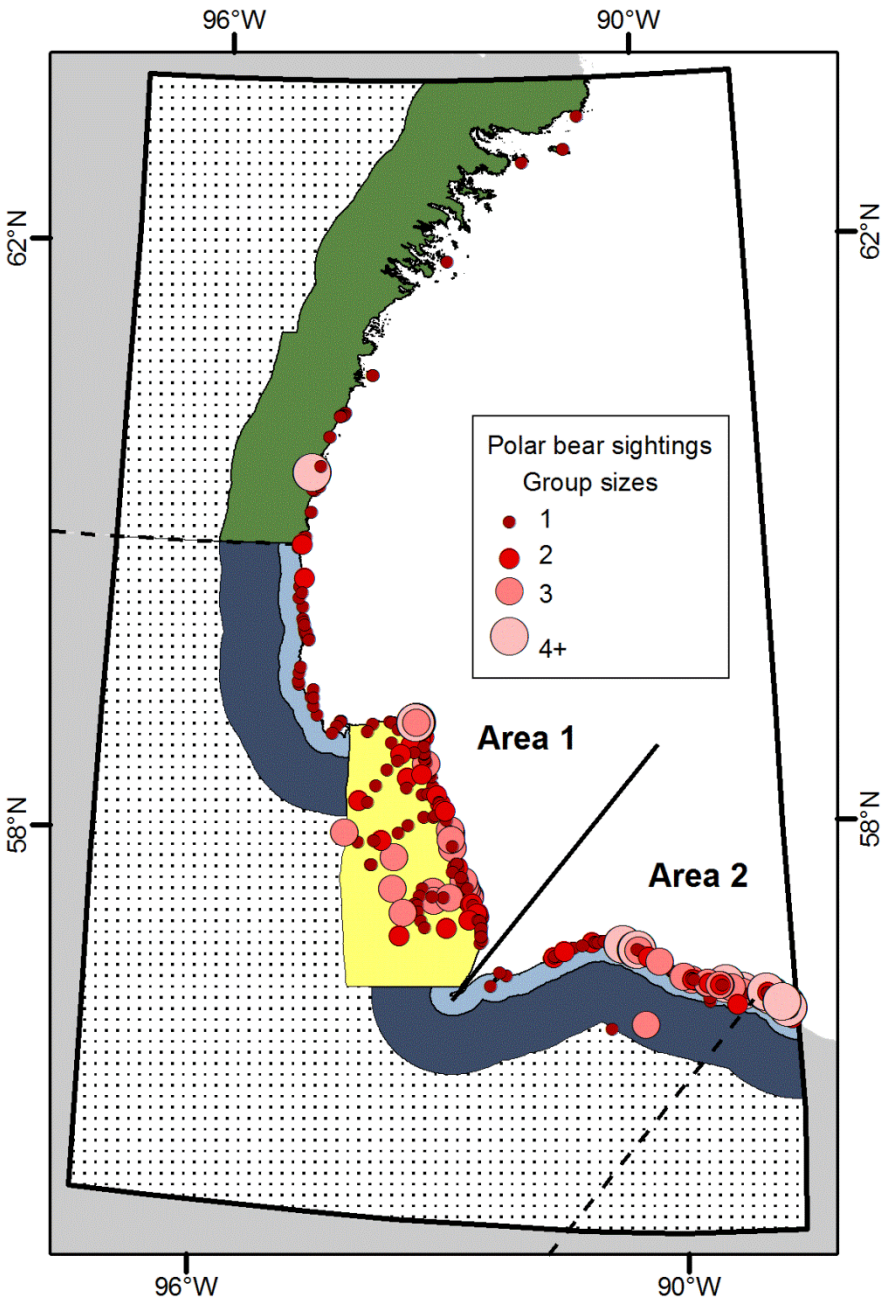
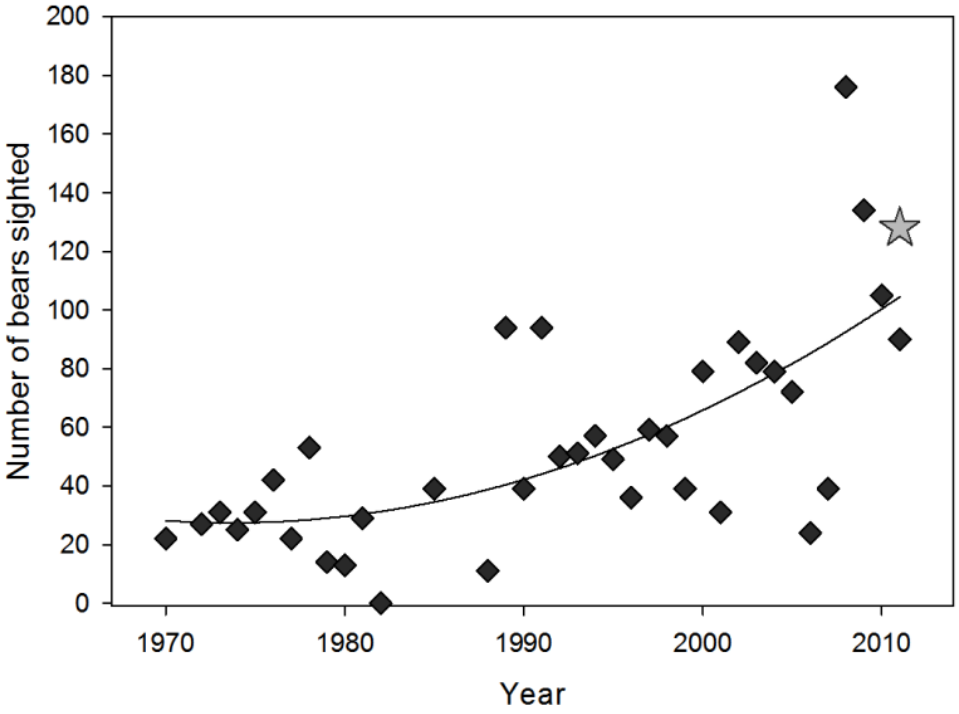


Figure 4. Polar bear counts from annual coastal surveys conducted between August 15 and September 15 from 1970 to 2011 in Western Hudson Bay in the region extending from the Nelson River to the Manitoba – Ontario border (i.e., Area 2; Stirling et al. 2004, Manitoba Conservation unpublished data). The number of bears observed during this survey, including those sighted along the coast and on small islands during the coastal contour transects, is denoted by a gray star.



Chapter 3

EFFICACY OF SPRINGTIME AERIAL SURVEYS ON SEA ICE FOR MONITORING POLAR BEAR ABUNDANCE

INTRODUCTION

Assessing the impacts of a rapidly changing environment on the sea ice-obligate polar bear (*Ursus maritimus*) and harvest management require information about demography and status from across their range (Vongraven et al. 2012). In North America and Greenland, estimates of polar bear abundance have been based almost exclusively on physical mark-recapture (e.g., Stirling and Kiliaan 1980; Taylor et al. 2005, 2006, 2008; Peacock et al. 2013). Management authorities have renewed their efforts to investigate other means of monitoring polar bears. These initiatives have stemmed from the recognition that methods not requiring wildlife immobilization and handling (Peacock et al. 2011), as well as rapid assessment techniques employable in regions with limited potential for capture-based research (Vongraven et al. 2012), are needed. Aerial surveys, widely implemented in a variety of other taxa to obtain 'snapshot' estimates of abundance, have shown promise as an alternative inventory tool for polar bears. Comprehensive land-based surveys during the ice-free season in Foxe Basin and Western Hudson Bay (this volume, chapters 1 and 2), as well as a combined overland – on-ice survey in the Barents Sea (Aars et al. 2009), have demonstrated the feasibility of aerial surveys for estimating and monitoring polar bear abundance.

Earlier pilot projects in the seas off Alaska (McDonald et al. 1999; Evans et al. 2003) and in the Barents Sea (Wiig and Derocher 1999) explored the utility of aerial

surveys for estimating polar bear abundance on sea ice. Although these studies varied by survey platform, geographic region, methodology, and study period, they generally concluded that the concentration of bears along the ice edge in late summer and fall provided the best opportunity for surveys. Despite promising results from this work, applications of large scale aerial surveys on sea ice have been limited (Aars et al. 2009). Lack of implementation can be attributed to a number of factors, including high variability in sea ice and bear distribution, the large effort required to generate useful results, unpredictable weather, and other logistical complications (S. Amstrup, U.S. Geological Survey, personal communication). For example, Aars et al. (2009) were compelled to use satellite telemetry to estimate bear densities over expanses of pack ice that were inaccessible due to inclement weather.

More recently, Nielson et al. (2013) used simulations that incorporated historical sea ice conditions, resource selection functions (RSFs) based on polar bear satellite telemetry data, and aerial survey data from Evans et al. (2003) to examine the utility and design considerations of fall aerial surveys in the Chukchi Sea. This work demonstrated that robust estimates may be generated from ship-based helicopter surveys over ice, though reliability was largely dependent on the duration of the study period and bear abundance (Nielson et al. 2013). Expenditures projected by Nielson et al. (2013), however, may render implementation of a large scale, ship-based aerial survey cost-prohibitive.

Whereas late summer and early fall were previously viewed as the optimal period for polar bear aerial surveys on sea ice, this recommendation may no longer be justified.

Climate and sea ice conditions have dramatically changed in recent years (Stroeve et al. 2007, 2012); timing of breakup is very inconsistent, and in regions such as the southern Beaufort Sea, pack ice retreats increasingly farther offshore during late summer, precluding consistent access by aircraft and necessitating the use of costly ice breakers (e.g., Evans et al. 2003). Springtime now may represent a more logistically and financially preferable alternative than the autumn. Specifically, ice is adjacent to the shore during spring, providing easier access by survey aircraft. Polar bear densities may be lower in spring due to more extensive ice coverage, but bears also may be easier to detect during this season because the ice is smoother and less fractured. The growing need to conduct inventories more regularly and to obtain baseline information from data-deficient subpopulations (Obbard et al. 2010) in remote regions underscores the need for continued development of aerial survey techniques.

We conducted pilot aerial surveys in the Baffin Bay and Southern Beaufort Sea polar bear populations to assess the current technical and logistical feasibility of spring aerial surveys on sea ice and to inform study designs for such surveys. We additionally sought to assess the utility of integrating on-ice aerial surveys in long-term monitoring programs.

METHODS

Study Areas

This study was conducted within 2 separate polar bear populations. The Baffin Bay population, located in the seasonal ice ecoregion (Amstrup et al. 2008), spans ca. 1 million km^2 across Nunavut, Canada and West Greenland. Ice annually recedes from

Greenland westward to Baffin Island in Canada during the spring and early summer. These ice dynamics reduce the extent of the sea ice to be surveyed and thus result in increased polar bear densities. Research was based from the community of Qikiqtarjuaq in southwestern Baffin Bay, Nunavut. Planned transects covered a variety of habitats, including landfast ice and offshore pack ice, and extended 125 – 300 km offshore. The total survey region encompassed about 75,000 km^2 (Figure 1).

The Southern Beaufort Sea population (hereafter Beaufort Sea) is located in the divergent ice ecoregion (Amstrup et al. 2008) and stretches over ca. 600,000 km^2, including portions of northern Alaska, Yukon, and Northwest Territories (NWT) in Canada. The survey area covered about 35,000 km^2, extending from the community of Tuktoyaktuk, NWT, westward about 310 km to the Alaska / Yukon border (Figure 2). We delineated the seaward extent of the study site (about 100 km offshore) by examining RSFs for female polar bears constructed from satellite collar data and annual ice conditions during March of the past decade (G. Durner et al., unpublished data). Although ice conditions during late winter and early spring, 2012 (the study period) suggested that moderate to optimal habitat comprised the vast majority of the study area, we included some less ideal habitats within the survey area to make results more generalizable to other habitats.

Study Design and Field Protocols

We employed line transect designs with mark-recapture distance sampling (MRDS) field protocols in both surveys. MRDS (Laake and Borchers 2004) combines distance sampling (Buckland et al. 2001) and mark-recapture (i.e., double observer or

sight-resight; Pollock and Kendall 1987) techniques. Density is estimated with a distance-based detection function, and the double observer data are used to evaluate detection on the transect line. MRDS thereby addresses the core distance sampling assumption of perfect detection on the transect line. Two teams within the same aircraft worked independently to sight bears, and sightings were not announced until both front and rear observers had an opportunity to detect each bear. Bears detected by observers in the front of the aircraft were considered marked (sighted), and those seen by aft observers were considered recaptured (re-sighted).

We conducted the aerial survey in Baffin Bay from a fixed wing (de Havilland DHC-6 Twin Otter) platform during late spring (27 May – 4 June), 2010. The Twin Otter provided access to far offshore pack ice and enabled us to survey with 4 trained, dedicated observers and a standby observer. We used the Canadian Ice Service's eastern Arctic sea ice map from 31 May to approximate sea ice conditions and extent during the study period. Planned transects were systematically spaced at 10 km intervals and arranged in a general east-west direction (Figure 1). The Twin Otter was outfitted with bubble windows to reduce blind spots directly beneath the aircraft. We surveyed at a speed of about 170 km/hr (ca. 90 knots) and an altitude of roughly 150 m (ca. 500 ft). Sighting angles were recorded with an inclinometer to estimate perpendicular distances of bears from the aircraft. Since small errors in low-angle inclinometer readings produce relatively large errors in sighting distances, we also flew off-transect and marked initial locations for bears located far from the flight path (generally >1 km) with a GPS

waypoint to better estimate perpendicular distances in a GIS (adapted from Marques et al. 2006).

We conducted the survey in the Beaufort Sea during early spring (22 – 29 March), 2012, using a helicopter (Bell 206L). This study differed from the Baffin Bay survey in several respects, including time of year, sea ice habitats, survey platform, and observer team. Here, the pilot and front seat observer functioned as the first observer team, and the rear seat observers comprised the second team. We sampled at an altitude of about 120 m (400 ft) and speed of ca. 160 km/hr (85 knots). We anticipated a density gradient declining from the edge of the landfast ice (i.e., floe edge). Hence, we arranged transects in a north-south direction to facilitate sampling approximately perpendicular to this presumed gradient, thereby reducing bias and improving precision (Figure 2; Buckland et al. 2001). Transects were spaced at 8 km intervals. We recorded flight paths and the initial location of each bear with a GPS and measured distances from transects to polar bear observations in a GIS (Marques et al. 2006).

We flew off transect to view bears more closely, document cluster size, and estimate sex and age-class. Covariates potentially impacting detection, including weather (e.g., fog; VIS), light conditions (e.g., brightness; LIGHT), and surrounding sea ice surface structure (ICE) were qualitatively scored for each sighting to assess potential heterogeneity in sighting probabilities.

Data Analysis

Although the Twin Otter was outfitted with bubble windows, we estimated a blind spot of 90 m on each side of its flight path and thus left-truncated Baffin Bay sightings

accordingly (i.e., subtracted 90 m from all observations; Borchers et al. 2006) for analyses. In the helicopter, rear observers were also subject to a small blind spot, so we left-truncated the Beaufort Sea data at 75 m. Data also were right-truncated at about 5% (1,250 m) to remove outlying observations (Buckland et al. 2001).

We used the Huggins model (Huggins 1989, 1991) to first estimate detection at distance zero with double observer data. Due to limited sample sizes, we pooled data from the surveys for this analysis. Because bears were difficult to see on the ice, we used some sightings to help observers acquire a search image. In these cases (n = 3), we announced the sighting so all observers could see the bear instead of maintaining independence between observers. We coded these instances as removals in capture histories. In Baffin Bay, we also censored some pilot-announced sightings from the sight-resight analysis. We completed sight-resight modeling in Program MARK (White and Burnham 1999). We allowed detection probabilities to remain constant or vary between observer teams. Because we were interested in the effect of distance on detection and we hypothesized *a priori* that distance would be important to explain heterogeneity in detection, we considered it in all covariate models. We also examined SITE (Baffin Bay or Beaufort Sea), ICE, LIGHT, VIS, and cluster size as covariates. The data were sparse, so we considered a maximum of two covariates in candidate models (Giudice et al. 2012). Model selection was based on Akaike's Information Criteria, adjusted for small sample sizes (AIC_c; Burnham and Anderson 2002).

We fit distance sampling models to estimate detection functions and effective strip half-widths in Program DISTANCE (Version 6.0, Release 2; Thomas et al. 2010). We

included all observations, including pilot-announced sightings, to better estimate detection. The fundamental differences between the 2 studies compelled us to first consider data separately for modeling detection functions. We had sufficient observations to examine Baffin Bay data independently, but the small sample size in the Beaufort Sea did not facilitate estimating a separate detection function there. Hence, we also pooled distance data from the 2 surveys to derive a global detection function and better approach Buckland et al.'s (2001) recommendation of 60 – 80 observations to estimate detection. We stratified the analysis to calculate site-specific standardized encounter rates and cluster sizes.

We examined conventional distance sampling models with standard series expansion terms as well as detection functions that included a single covariate (LIGHT or ICE; SITE also was evaluated with pooled dataset) in each model (Marques and Buckland 2003) and employed AIC_c for model selection. We assessed model goodness of fit with Chi square, Kolmogorov-Smirnov, and Cramer-von Mises statistics.

To evaluate the feasibility and resources required for a large scale, on-ice survey, we calculated sampling intensity necessary to achieve a target precision, following Buckland et al. (2001)

$$L = \left[\frac{b}{(cv(\hat{D}))^2}\right]\left[\frac{L_0}{N_0}\right],$$

where L_0 and N_0 represent sampling effort and number of observations during the pilot study, *CV* the target coefficient of variation for density (and thus abundance), *b* a variance metric, and *L* the required effort to achieve that level of precision (e.g., total

linear km to survey). We incorporated all sightings and conservatively assumed $b = 3$ for these calculations (Burnham et al. 1980; Buckland et al. 2001).

We explored statistical power to detect changes in abundance by conducting a series of Monte Carlo simulations ($n = 10{,}000$ iterations) in Program MONITOR (Gibbs and Ene 2010). This enabled us to evaluate trade-offs between potential aerial survey monitoring programs, their respective abilities to detect change and associated costs. We used the Southern Beaufort Sea as a case study and defined a starting population of 1,500 bears (Regehr et al. 2006). We considered monitoring programs with coefficients of variation (CVs) for individual estimates ranging from 15% to 25% (based on our survey results) and inter-survey intervals of 3 to 6 years. We evaluated whether we would be able to detect total declines in abundance of 20 – 80% over 15- and 30- year periods. We assumed that the relationship of variance to the mean (i.e., CV) remained stable. We acknowledge that these simulations did not account for all sources of variability (e.g., environmental and demographic stochasticity). However, our intent was to implement a method that was easily repeatable for managers to inform the decision-making process. To estimate power, a simple linear regression was fit to the resultant points of each simulation iteration; detection of a significant negative trend in abundance was considered a ‘success’, and successes were tallied to estimate power. With increasing climatic uncertainty, failing to detect a decline (Type II error) may have more serious consequences, in terms of management and conservation, than reporting a nonexistent decline (Type I error; Toft and Shea 1983; Peterman 1990). Hence, we defined $\alpha = 0.1$ for the trend regression. We assumed that *a priori* information about the direction of

population change would be available from both ancillary scientific and local knowledge and thus specified a 1-tailed test. To evaluate the relationship between statistical power and projected expenses, we used the Beaufort Sea pilot survey expenditures (aircraft charter and fuel) to attach costs in present U.S. dollars to monitoring program alternatives.

RESULTS

Sampling Effort and Sightings

In Baffin Bay, we sampled about 35 hours and covered 4,801 km. Although we planned to survey at different times of day, highly variable conditions compelled us to survey whenever weather permitted; low cloud ceilings, fog, and poor visibility greatly hindered our ability to sample. Orientation and length of transects were frequently modified due to weather, and spacing was compressed on landfast ice and near the floe edge to increase encounter rates (Figure 1). We sighted 45 bears in 29 clusters, including 12 females with their offspring. Standardized encounter rates were greatest near the floe edge (floe edge ± 10 km: 11.9 clusters and 19.2 individuals / 1,000 km; landfast ice >10 km from floe edge: 3.5 clusters and 4.6 individuals / 1,000 km transects; pack ice >10 km from floe edge: 3.3 clusters and 4.9 individuals / 1,000 km; Figure 1). The maximum distance a bear was sighted from the floe edge was about 50 km, although we surveyed large areas of pack ice farther offshore.

In the Beaufort Sea, we surveyed landfast ice and heavily consolidated pack ice with no open water or discernible floe edge. We could not complete all planned transects, so we also surveyed during some ferry flights to increase efficiency. We surveyed 3,013

linear km and sighted 21 bears in 12 independent clusters, including 5 family groups, during about 20 hours of sampling (Figure 2).

Modeling Detection

After left-truncating by survey platform and censoring pilot-announced sightings, we included a total of 31 observations to evaluate detection with mark-recapture models. The most supported model specified constant detection between observer teams and included distance from transect and ICE (qualitative 1 to 3 scale) as covariates (logit [p] = 4.6 [SE: 1.6] – 1.8 [SE 1.6] x km – 2.4 [SE 1.0] x ice surface structure score). Inclusion of additional covariates in modeling was not highly supported. We estimated the mean, individual observer team detection probability in the strip as p = 0.49 (SE: 0.13) and detection at distance zero as p = 0.66 (SE: 0.17). These parameter estimates yielded inclusion probabilities of being detected by at least 1 observer (calculated as $1 - [1 - p]^2$) of 88% at distance zero and 74% across the entire strip. We did not fit full MRDS models due to sample size constraints.

Sighting distances were relatively consistent between Baffin Bay and the Beaufort Sea despite the limited number of observations and different survey platforms, and data suggested that detection declined predictably with increasing distance (Figure 3). For Baffin Bay, we post-stratified our effort into 2 zones – a landfast ice and floe edge (± 10 km) stratum and an offshore pack ice stratum – based on encounter rates and logistical considerations. Because inclement weather precluded flying planned transects in Baffin Bay, we treated daily flights rather than individual transects as the sampling unit for estimating variance of density.

Following truncation, we included 26 observations for fitting the BB detection function and 37 sightings for the pooled detection function. Goodness of fit statistics indicated adequate model fit (all $P > 0.2$) for strongly supported models. A Uniform key function with a single cosine adjustment term was the most highly supported model for Baffin Bay, whereas a half-normal key function with the covariate SITE was best supported in the pooled analysis (Figure 4). In both analyses, multiple alternative parameterizations were within 2 AIC_c (Tables 1 and 2). Detection probabilities, effective strip half-widths, and densities were consistent among models within each dataset. However, in comparison to estimates based only on data from Baffin Bay (Table 1), the inclusion of data from the Beaufort Sea marginally decreased detection and effective strip half-widths and increased estimates of abundance (Table 2). Estimated densities were similar in the Beaufort Sea and Baffin Bay pack ice, and about twice as high in the Baffin Bay landfast ice / floe edge stratum. Precision was poorest for the offshore stratum in Baffin Bay where encounter rates were highly variable.

Projected Precision and Power

Beaufort Sea encounter rates suggested that obtaining an abundance CV of about 20% is achievable in a population-wide survey by sampling nearly 20,000 km of transects; a 15% CV would require surveying >30,000 km (Figure 5; about 7x and 11x the effort of our pilot survey, respectively). In Baffin Bay, projections indicated that stratum-specific CVs of roughly 20% may be obtained through survey flights of <10,000 km and >20,000 km in the nearshore and offshore strata, respectively (Figure 5; about 4x and 9x the effort of our pilot study). Additionally, strata densities and estimated ice

coverage suggested that about two-thirds of polar bears were located in the extensive offshore, pack ice stratum in Baffin Bay during late spring.

Frequent (3 – 4 year intervals) and intense (achieving 15 – 20% CVs) aerial surveys would be able to detect population declines of 40 – 50% with at least 0.8 power over both 15 and 30-year periods (Figures 6 and 7). Using less frequent surveys and decreasing individual survey intensities resulted in marginal losses in statistical power. These reductions in power were greatest at rates of decline in abundance of 30 – 50%. Nearly all of the monitoring schemes we considered had >0.8 power to detect a decline if real decreases in abundance were 60% over 15 years and 50% over 30 years.

Based on the projected number of transects for a population-wide study with a target CV of 20%, we estimated that aircraft charter and fuel charges alone would cost about $550,000 U.S. for a one-year aerial survey. Total, projected monitoring program costs to attain 0.8 power to detect changes of 50% over 15 years would reach $2.5 to $3 million in present U.S. dollars (Figure 8). Obtaining the same power to detect a 40% decline over 30 years would require a roughly $4 million investment. Simulations suggested that a number of different combinations of inter-survey intervals and individual survey intensities would facilitate reaching these thresholds.

DISCUSSION

Results from our pilot studies suggest that an on-ice aerial survey for polar bears during springtime could yield results with acceptable precision. Sightings data facilitated robust estimation of detection functions, illustrated by the detection probability precision and consistency in density estimates among the most-supported models within each

analysis. Detection differed between distance sampling and sight-resight methods, presumably due to unmodeled heterogeneity in the sight-resight analyses. Although sight-resight analyses suggested that detection on and near the transect line was less than unity, we did not implement MRDS models due to small sample sizes. As such, our estimates of density underestimate true density by ~10%. The inclusion of a double-observer platform will ensure that larger scale studies fulfill this assumption.

We individually examined Baffin Bay sightings and then pooled them with Beaufort Sea data to model detection functions. The limited number of sightings in the Beaufort Sea necessitated this approach, but we would have preferred to estimate separate detection functions for each study. The sightings histograms and our modeling results (Tables 1 and 2) highlight that separate detection functions should be estimated for individual studies when possible.

Logistical and Design Considerations

Our results indicate that the importance of a number of design and logistical elements for springtime aerial surveys may vary by area. Although aerial surveys with MRDS sampling and analytical protocols may be feasible in many populations, individual units face unique challenges related to specific ice dynamics, geographic extent, and projected abundance.

Identification of a suitable temporal window is critical for any survey. In Baffin Bay, we chose late May and June based on several considerations, including the availability of all age and reproductive classes (e.g., emergence of females with cubs from maternity dens; Ferguson et al. 2000), improved light conditions, and reduced

extent of sea ice necessary to survey; nevertheless, we were unable to sample along planned transects due to weather conditions (Figure 1). Post-stratification allowed us to generate density estimates that accounted for variability in sampling intensity. However, because we were unable to survey extensively in the far offshore pack ice, our density estimates were derived from sampling that was not systematically distributed across the stratum and thus should be interpreted cautiously. The inclement and unpredictable weather associated with breakup and ice recession from Greenland, reduced detectability on highly fractured ice, and large-scale changes in the shape and extent of the study area that could occur during a survey suggest that late spring is probably not ideal for aerial surveys in Baffin Bay. In the Beaufort Sea, our March study period provided a more stable environment, but the survey likely occurred before all females with newborn cubs had emerged from dens (Smith et al. 2007) and thus was likely negatively biased.

The timing of a springtime survey must balance these considerations. In both regions, April would provide a better potential study period in terms of ice stability and availability of females with newborn cubs (Ferguson et al. 2000, Smith et al. 2007, U.S. Geological Survey (USGS), unpublished data). The survey period should be as narrow as possible (4 – 6 weeks) to minimize changes in habitat conditions that affect detection, study area extent, and density and distribution of bears. Achieving this may be difficult when adverse weather precludes surveying for extended periods: aircraft have been grounded an average of 28% of days during springtime work on the Alaskan side of the Beaufort Sea since 2001, with some years reaching nearly 50% (USGS, unpublished

data). These conditions may make it challenging to obtain density estimates representative of an entire subpopulation.

Encounter rates observed in this study suggest that at least 20,000 km of transects (ca. 125 flight hours, exclusive of ferrying, positioning, and off-transect flights) would need to be surveyed in springtime in both Baffin Bay and the Beaufort Sea to obtain CVs of ≤20%. As such, a comprehensive springtime survey represents a considerable logistical undertaking and would undoubtedly require sampling with at least 2 – 3 crews and aircraft. Stratification based on springtime RSFs (e.g., Durner et al. 2009) or regularly updated ice coverage maps will be essential to efficiently allocate sampling effort and improve precision. Although study design may be informed by RSFs, satellite telemetry data would also help delineate the study area by quantifying how far offshore bears range. In the Beaufort Sea, our sampling extended up to 100 km offshore, but this delineation did not encompass the entirety of the subpopulation. Bears collared in Alaska during 2004 – 2011 were, on average, about 65 km offshore during April, but the maximum individual average distance was >250 km from the coastline (USGS, unpublished data). These data provide some guidance and suggest that sampling should extend several hundred km offshore in springtime. Defining the extent of the study site and safely accessing such far offshore regions may be difficult to accomplish, particularly in divergent ice (Amstrup et al. 2008) subpopulations in which ice adjoins the polar basin.

Directional movements by bears during the survey period may bias density estimates if not accommodated by the study design. For example, if sampling transects

are oriented north-south, then uniform movement of bears from east to west (or vice versa) due to the pattern of ice melt and breakup may subject animals to double-counting or exclusion from estimates. Similarly, bears moving in a direction consistent with transect orientation may confound density estimation if they cross strata boundaries. These issues may be particularly problematic if inclement weather grounds aircraft for multiple days. Telemetry data from Beaufort Sea bears suggest that movements tend to be in a southerly direction during March to May (USGS, unpublished data), consistent with the planned transect orientation. Subdividing the study area based on sampling during short time intervals and estimating density and abundance within these smaller regions can be an effective means of addressing changes in densities (Nielson et al. 2013).

Monitoring with Aerial Surveys

Our simulations suggest that, while aerial survey-based monitoring programs would be able to document large decreases in abundance (ca. 40 – 50% declines over 15- and 30- year periods), they have limited power to detect more modest declines. Decreasing inter-survey intervals or increasing sampling effort per survey would improve power but require substantial increases in cost.

To address the ecological uncertainties associated with climate change and the need for sensitive monitoring programs, aerial surveys could be employed in combination with other monitoring methods to improve their utility. For example, aerial surveys and genetic mark-recapture via biopsy darting may be logistically compatible and technically complementary (e.g., spatially-explicit genetic mark-recapture)Biopsy darting involves collection of a small tissue sample from a remotely fired dart (e.g., from a helicopter;

Pagano et al. 2013, Government of Nunavut and Greenland Institute of Natural Resources, unpublished data). Genotyping tissue samples facilitates modeling of demographic parameters in a capture-recapture framework; Pagano et al. (2013) report that biopsy darting yielded a genetic identity for >99% of successfully retrieved samples. Aerial surveys generate relatively quick, 1-year abundance estimates, while the biopsy darting would provide estimates of vital rates that could be used to independently assess population trend. Moreover, biopsy darting conducted within the systematic sampling framework of an aerial survey may overcome some of the problems with spatial heterogeneity that may affect capture-based programs (Taylor and Lee 1995, Regehr et al. 2010, Peacock et al. 2013). We note that the additional time and costs to obtain biopsy samples during an aerial survey are likely negligible, given that survey protocols typically specify flying off-transect to each sighted bear to estimate age- and sex-class, collect GPS coordinates, and characterize habitat. Of course, the ability to combine methods is dependent on aircraft, as biopsy darting requires a helicopter.

The simulations provide a general overview of the statistical power of potential monitoring programs in the Beaufort Sea. We considered the two temporal windows (i.e., 15 and 30 years) to reflect the historical, long-term changes that have been documented in some regions (e.g., M'Clintock Channel, Taylor et al. 2006) as well as rapid, short-term declines that could occur with continued sea ice loss. Our adoption of $\alpha = 0.1$ and a 1-tailed test for the trend assessment regression is justified from management and conservation perspectives, where failing to detect a real decline in abundance is at least as risky for conservation as falsely attributing a decline to a stable subpopulation (Toft and

Shea 1983, Peterman 1990). Balancing α and β better reflects the actual consequences of Type I and II errors in wildlife management (Peterman 1990). We acknowledge that specifying a constant coefficient of variation in a declining population may be unrealistic in an actual monitoring scenario. In the absence of increased sampling intensity, fewer observations would be available to estimate the detection function in a reduced population, potentially leading to poorer precision. Hence, our simulations may overestimate power. Conversely, smoothing methods (e.g., bootstrapping) may reduce inter-annual variability in abundance estimation and improve our ability to detect trend from successive point estimates (e.g., Regehr et al. 2007).

Data from aerial surveys must be integrated with abundance estimates derived from previous and on-going capture-based methods to facilitate accurate assessment of status and trend. The Southern Beaufort Sea has been the focus of a decades-long capture-recapture program (Amstrup et al. 1986, Regehr et al. 2006), making it an ideal study site for the continued development of on-ice surveys (Vongraven et al. 2012). The existing dataset facilitates comparison of capture-based and aerial survey-derived abundance estimates to evaluate relative precision and potential biases of the methods. Consistency in these approaches would increase confidence in polar bear abundance estimation and in the reliability of the techniques themselves.

Financial expenditures associated with the aerial survey monitoring programs we examined are similar to or less than other inventory methods, given that an aerial survey requires only a single year of study (versus multi-year capture programs; e.g., Nielson et al. 2013). However, all costs and benefits of a prospective monitoring program, including

opportunity costs and potential externalities (e.g., wildlife handling), should be carefully evaluated to ensure the long-term success of the program (Caughlan and Oakley 2001). Explicit statements of management goals and acceptability of risks of making Type I and Type II errors will further inform the development and selection of methods to be used in monitoring programs.

Conclusions

Resurgent interest in polar bear aerial surveys, spurred by broad support from numerous jurisdictions and co-management boards, has led to renewed investigations of optimal timing, methods, and required effort. Aerial surveys on sea ice could provide a means to obtain baseline abundance estimates from data deficient subpopulations (Obbard et al. 2010) in more remote regions of the Arctic. Our results suggest that springtime on-ice surveys may be technically feasible, but logistical challenges are significant and design components warrant careful consideration. At the levels of precision investigated here, moderate but conservation-significant declines would not be detected. Aerial surveys, perhaps combined with genetic mark-recapture via biopsy darting, show some promise for long-term monitoring of abundance, depending on population-specific management objectives, acceptability of conservation risk, and desire to implement noninvasive techniques.

Table 1. Distance sampling model results derived with pooled data from springtime polar bear aerial surveys on the sea ice of Baffin Bay (2010). Models within 2 AIC of the best model are presented. ESW = effective strip half-width (meters); p = detection probability; CV = coefficient of variation for ESW and p. Model names are denoted as key function / adjustment term or covariate. Cluster density refers to clusters of bears / 1,000 km^2.

					Cluster Density (95% CI)	
Model	ΔAIC_c	ESW (95% CI)	p (95% CI)	CV, ESW/p	Baffin Bay, Fast ice / Floe edge	Baffin Bay, Pack ice
Uniform / Cosine	0.00	734 (580-928)	0.59 (0.46-0.74)	11.4	5.2 (2.8-9.4)	2.2 (0.36-14.1)
Half-Normal	0.48	722 (538-970)	0.58 (0.43-0.78)	14.4	5.3 (2.8-9.7)	2.3 (0.36-14.2)
Hazard	1.76	620 (324-1185)	0.50 (0.26-0.95)	32.2	6.1 (2.8-13.5)	2.7 (0.45-15.8)

Table 2. Distance sampling model results derived with pooled data from springtime polar bear aerial surveys on the sea ice of Baffin Bay (2010) and the southern Beaufort Sea (2012). Models within 2 AIC of the best model are presented. ESW = effective strip half-width (meters); p = detection probability; CV = coefficient of variation for ESW and p. Model names are denoted as key function / adjustment term or covariate. Cluster density refers to clusters of bears / 1,000 km^2.

					Cluster Density (95% CI)		
Model	ΔAIC_c	ESW (95% CI)	p (95% CI)	CV, ESW/p	Beaufort Sea	Baffin Bay, Fast ice / Floe edge	Baffin Bay, Pack ice
Half-Normal / SITE	0.00	611 (469-794)	0.49 (0.38-0.64)	13.0	3.1 (1.5-6.2)	6.2 (3.4-11.4)	2.7 (0.43-16.9)
Uniform / Cosine	0.38	687 (592-797)	0.55 (0.47-0.64)	7.3	2.7 (1.4-5.3)	5.5 (3.0-10.0)	2.4 (0.38-15.2)
Half-Normal	0.47	640 (508-806)	0.51 (0.41-0.65)	11.4	2.9 (1.5-5.8)	5.9 (3.3-10.8)	2.6 (0.41-16.2)
Half-Normal / Light	0.72	621 (483-799)	0.50 (0.39-0.64)	12.4	3.0 (1.5-6.0)	6.1 (3.3-11.2)	2.7 (0.42-16.7)
Hazard	1.30	619 (416-920)	0.49 (0.33-0.74)	19.7	3.0 (1.5-6.4)	6.1 (3.2-11.7)	2.7 (0.43-16.4)

Figure 1. Sampling transects and polar bears sighted during a pilot aerial survey in Baffin Bay, May – June, 2010. The study area is shaded dark gray in the inset

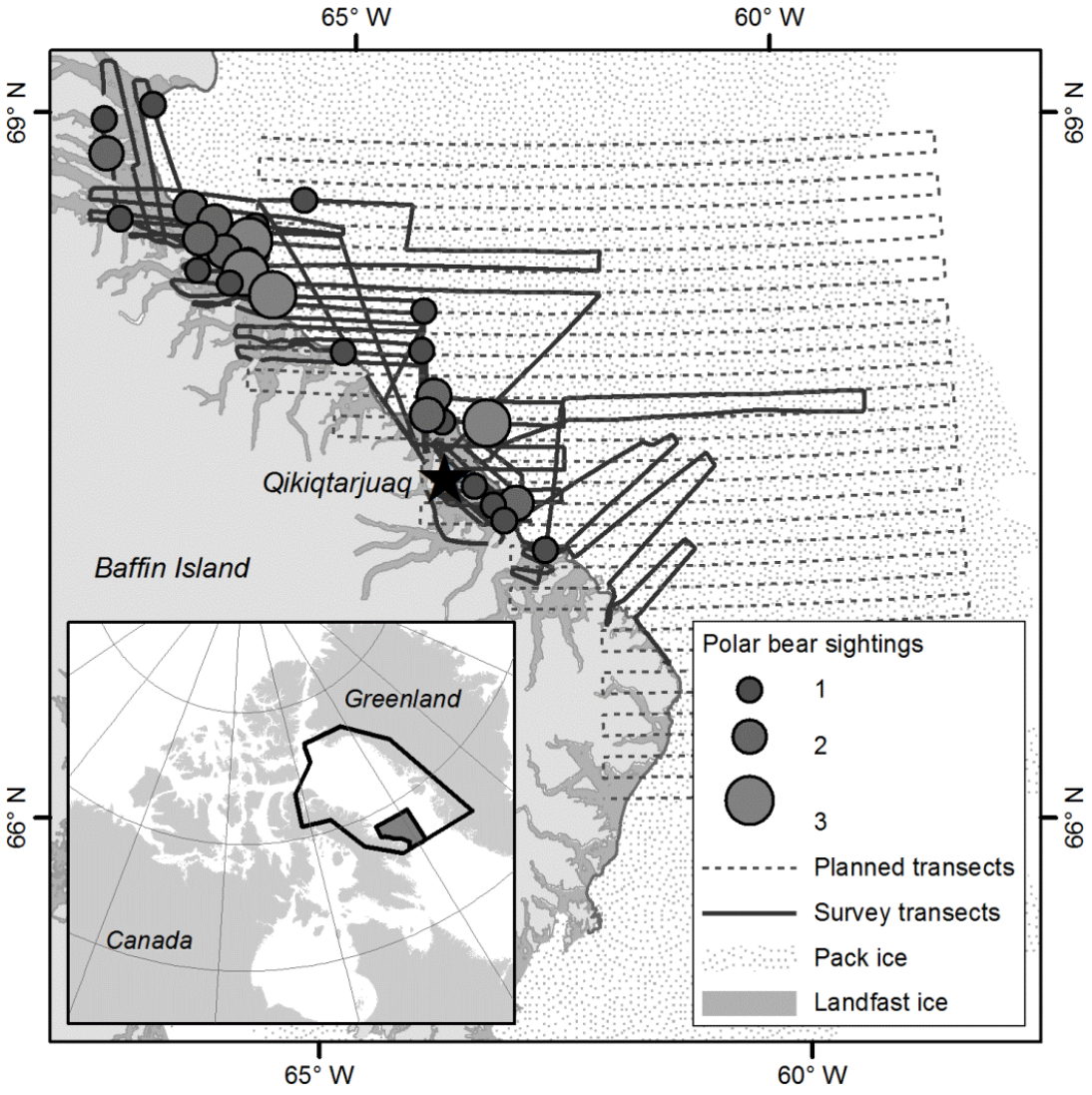


Figure 2. Sampling transects and polar bears sighted during a pilot aerial survey in the southern Beaufort Sea, March, 2012. The study area is shaded dark gray in the inset.

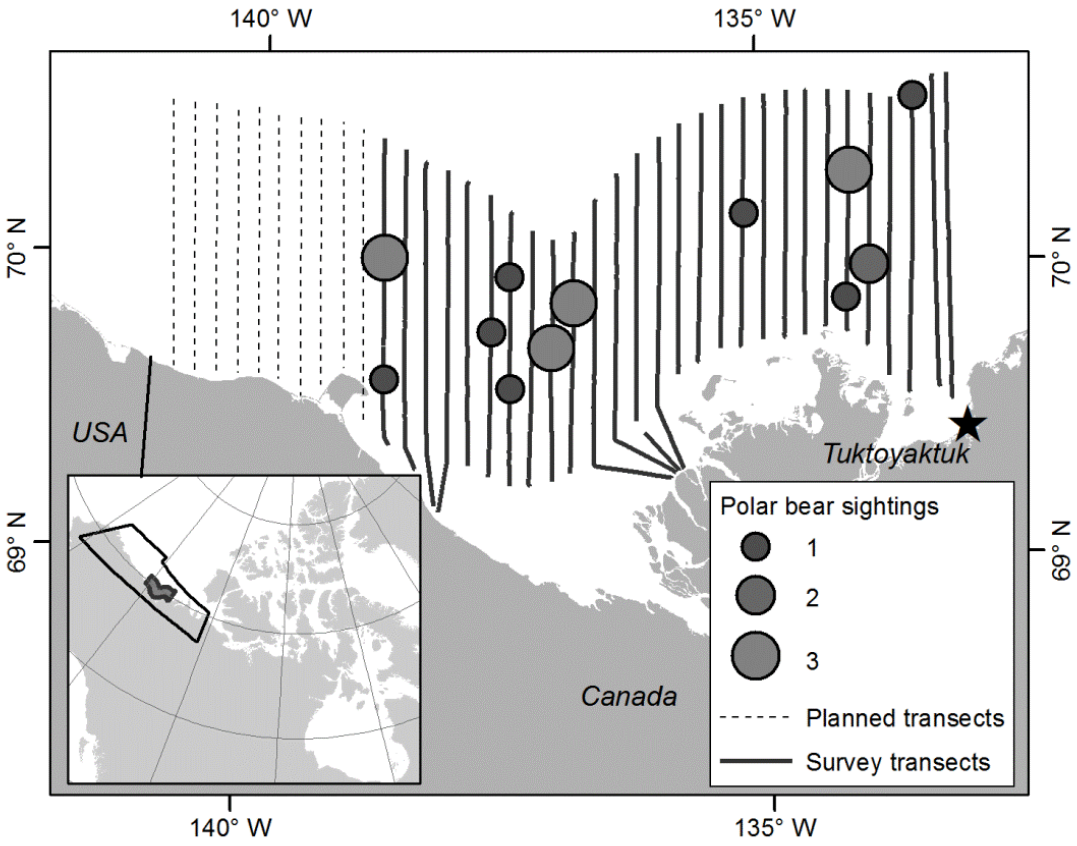


Figure 3. Histogram of sightings from springtime polar bear aerial surveys in Baffin Bay and the southern Beaufort Sea. Data were left-truncated at 90 m for Baffin Bay and 75 m for the Southern Beaufort Sea to accommodate blind spots beneath the aircraft.

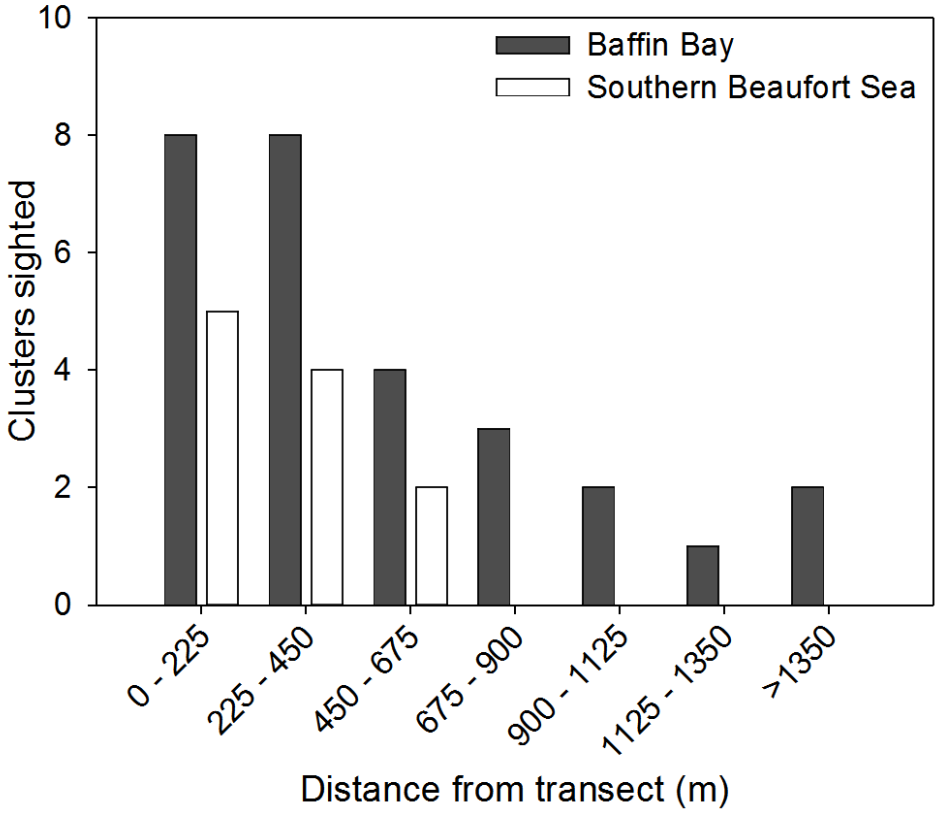


Figure 4. Distance sampling detection function (half-normal key function with a covariate for site) derived from pooled data from polar bear aerial surveys in Baffin Bay and the southern Beaufort Sea. Both covariate levels (Baffin Bay and Beaufort Sea) are plotted.

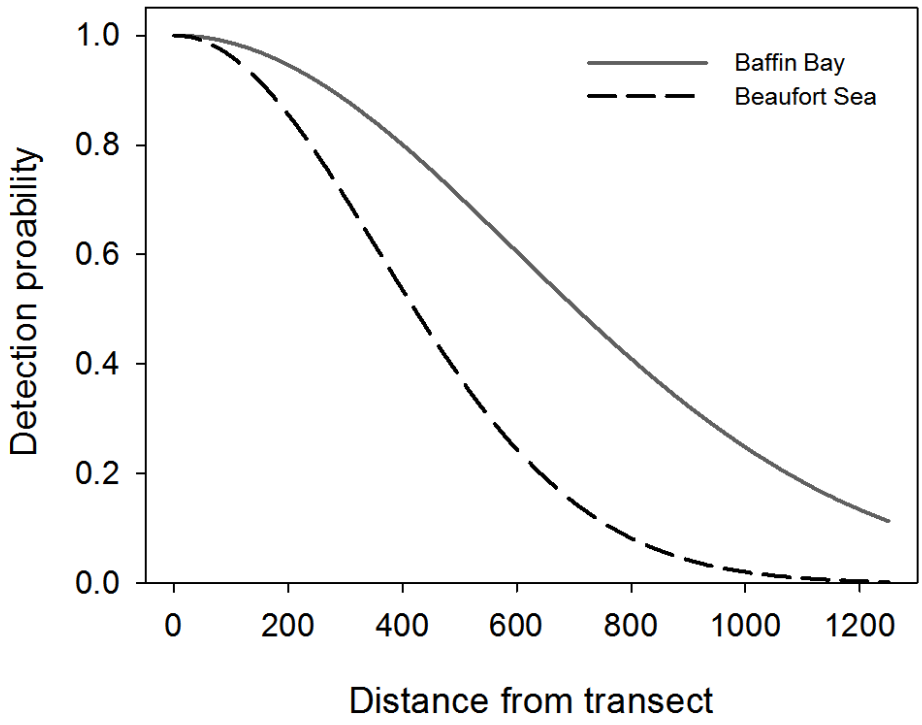


Figure 5. Projected survey effort required to obtain target abundance coefficients of variation in Baffin Bay and the southern Beaufort Sea. Coefficients of variation (CVs) calculated for Baffin Bay reflect stratum-level precision. Projections are based on pilot polar bear aerial survey data.

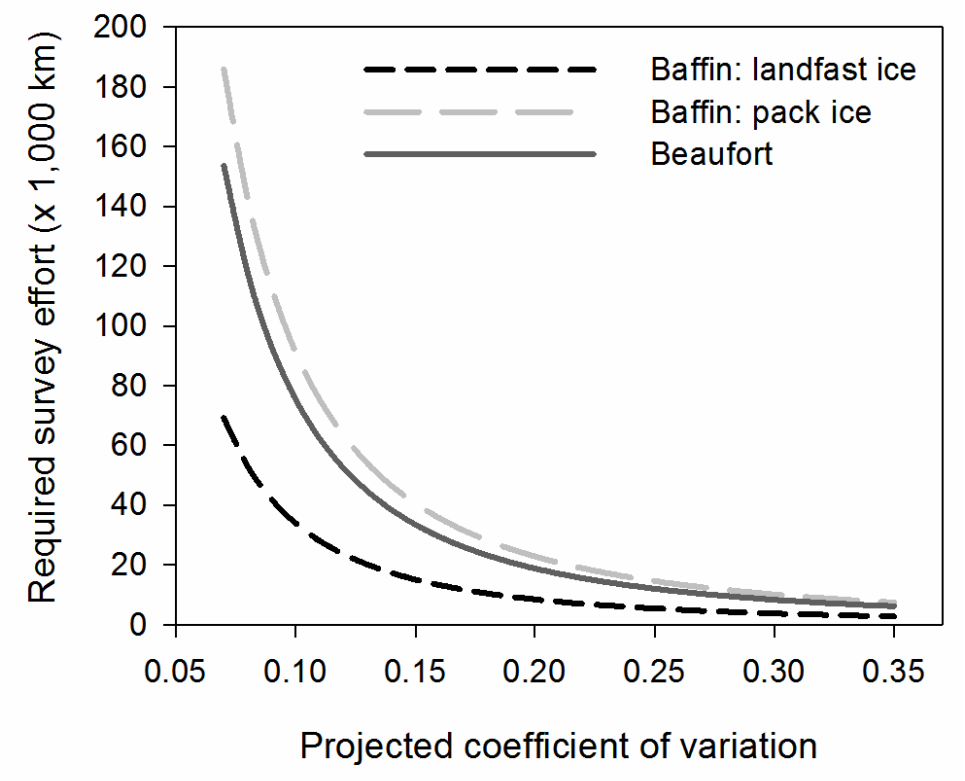


Figure 6. Estimated statistical power of different springtime, aerial survey monitoring programs to detect changes in polar bear abundance over a 15 year period. Simulations assumed an initial survey at time t = 0 and regular surveys thereafter based on the monitoring program. Four-year interval programs include an additional survey in year 15 to facilitate comparison among programs.

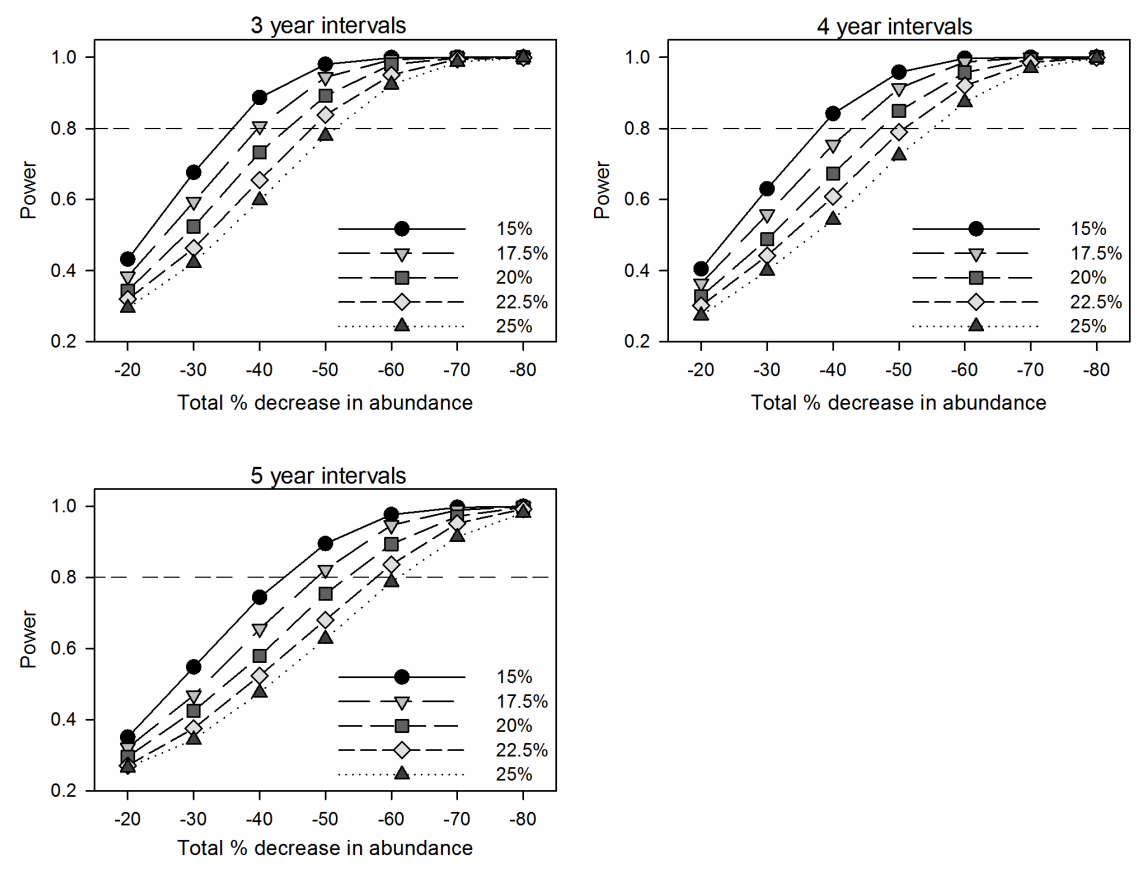


Figure 7. Estimated statistical power of different springtime, aerial survey monitoring programs to detect changes in polar bear abundance over a 30 year period. Simulations assumed an initial survey at time t = 0 and regular surveys thereafter based on the monitoring program. Four-year interval programs include an additional survey in year 30 to facilitate comparison among programs.

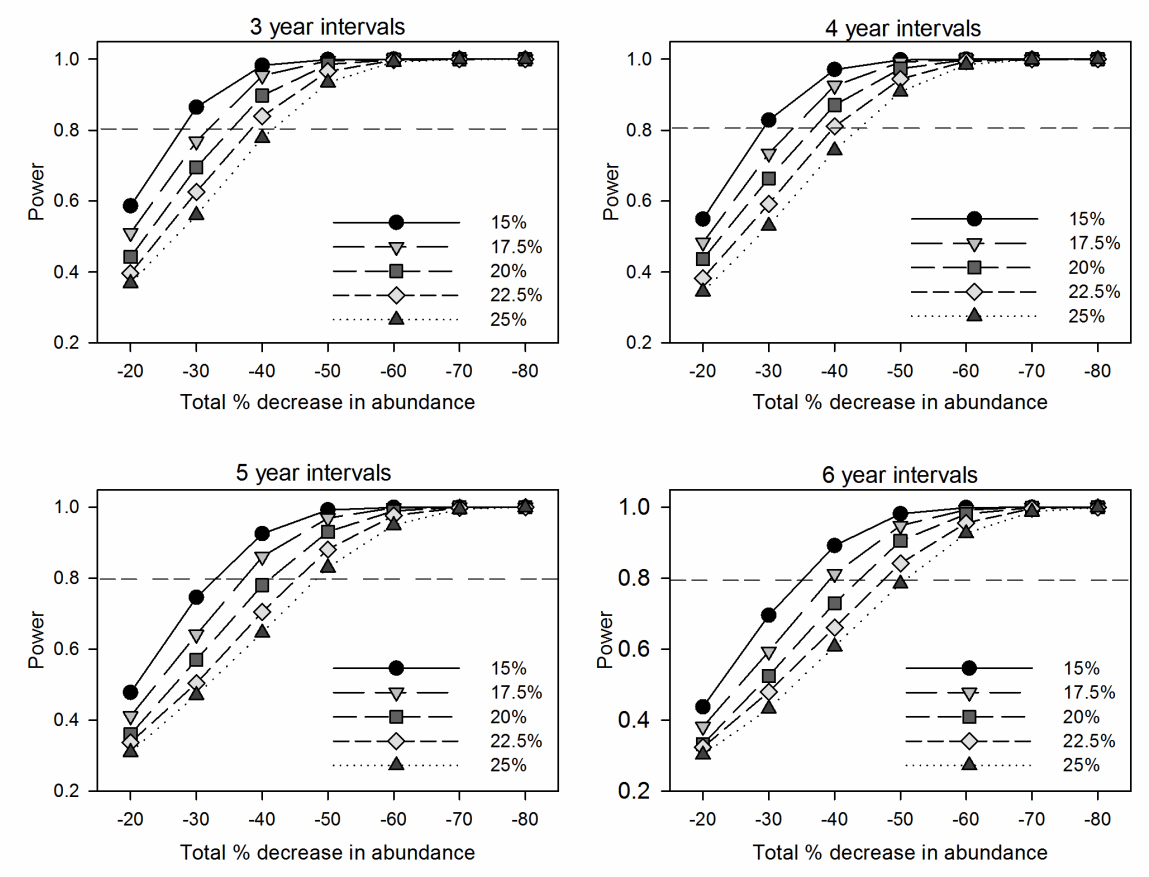
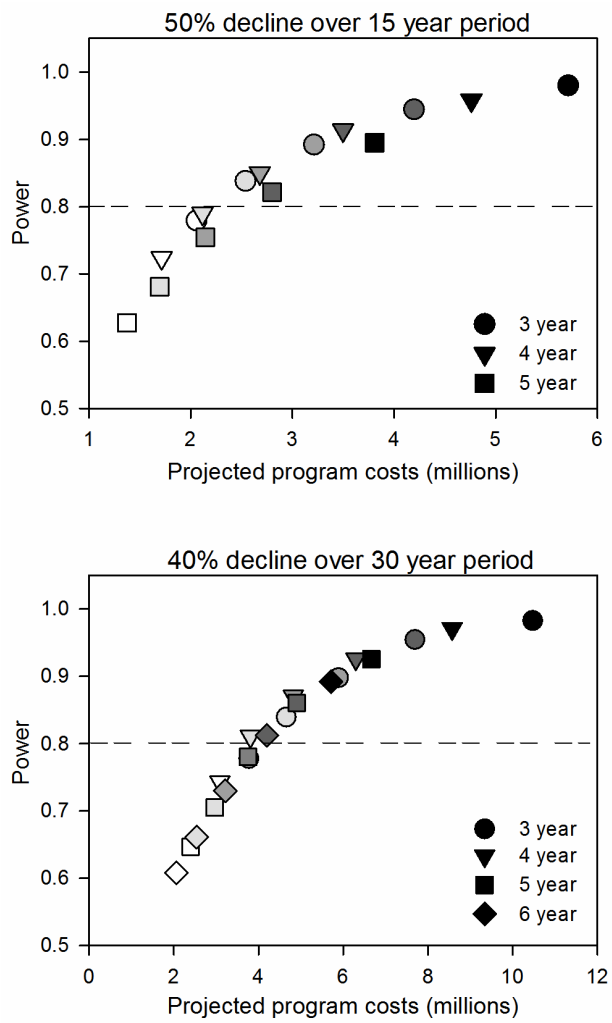


Figure 8. Projected costs of polar bear aerial survey monitoring programs in present US dollars relative to statistical power derived from different monitoring programs for: a) a 50% decline in abundance over 15 years (4.5% net annual decline) and b) a 40% decline in abundance over 30 years (1.7% net annual decline). Gradient shading reflects monitoring program precision, from white = 25% CV to black = 15% CV. Estimated costs are based on pilot aerial survey from the southern Beaufort Sea and include only aircraft charters and fuel. Four-year interval programs include additional surveys in years 15 and 30 to facilitate comparison among programs.



Chapter 4

POLAR BEARS FROM SPACE: ASSESSING SATELLITE IMAGERY AS A TOOL TO TRACK ARCTIC WILDLIFE

INTRODUCTION

The loss of Arctic sea ice has accelerated during recent years (Stroeve et al. 2007, 2012, Comiso et al. 2008), with minimum sea ice extent reaching a record low during September, 2012. A nearly ice-free summer is now forecasted to occur as early as 2016 (Maslowski et al. 2012, Overland and Wang 2013). Such large-scale, precipitous environmental changes will be detrimental for many species dependent on sea ice habitats (Laidre et al. 2008).

Despite potentially massive ecological impacts, regimes for monitoring wildlife remain deficient across large portions of the Arctic. For example, marine mammal assessment programs traditionally have used some combination of costly aircraft- or ship-based surveys and / or mark recapture programs (e.g., Garner et al. 1999), but the precision of resulting demographic estimates is often inadequate to detect trends in abundance (Taylor et al. 2006). Moreover, some areas are simply too inaccessible for routine monitoring. As such, baseline or long-term data are lacking for numerous species, precluding status and trend assessment and hindering management efforts. Walrus (*Odobenus rosmarus*; Lowry et al. 2008) and ribbon seals (*Histriophoca fasciata*; Burkanov and Lowry 2008) are among the Arctic marine mammals currently classified as data deficient by the International Union for Conservation of Nature. Likewise, data are insufficient to assess polar bear (*Ursus maritimus*) status across large portions of their

range (Obbard et al. 2010); even in surveyed areas, monitoring intervals are often inadequate (Vongraven et al. 2012). More frequent, systematic and efficient population surveys are needed to match the data needs of resource managers faced with a rapidly changing environment.

Recent advancements in satellite technology (resolutions of 0.5–5 m) have provided new tools for monitoring wildlife. Previous studies used satellite imagery to estimate abundance at Weddell seal (*Leptonychotes weddellii*) haul-outs (LaRue et al. 2011) and emperor penguin (*Aptenodytes forsteri*) colonies in Antarctica (Fretwell et al. 2012). Similarly, Platonov et al. (2013) reported that polar bears, walrus, and other marine mammals are visible on imagery, but their findings are limited by an absence of ground-truthed data. Remote sensing affords access to vast expanses of otherwise inaccessible sites without concerns over human safety and disturbance to wildlife.

Here, our goal was to evaluate the utility of high resolution satellite imagery to monitor Arctic wildlife, using polar bears as a case study. Whereas polar bears rank among the most studied large mammals globally, with capture datasets in some regions extending >30 years (e.g., Regehr et al. 2007), most of that research has focused on a few easily accessible subpopulations. Polar bears are categorized as Vulnerable by the IUCN, largely owing to projected sea ice losses (Schliebe et al. 2008a), but a paucity of population-level data exists across several broad regions. There is a need for efficient methods that overcome logistical challenges, facilitate regular monitoring, and are consistent with the values of northern communities concerned about disturbance to wildlife. Additionally, the changing sea ice dynamics have led to shifts in the onshore

distribution and abundance of polar bears (e.g., Schliebe et al. 2008b). Mitigating potential increases in human – bear conflicts requires an understanding of and ability to predict these shifts.

METHODS

Study Area

We obtained high-resolution satellite imagery from Foxe Basin, Nunavut, located in a seasonal ice region of the eastern Canadian Arctic. Recent comprehensive aerial surveys documented high densities of polar bears on relatively small islands (totaling <3,000 km^2) in northern Foxe Basin with low topographic relief and no snow cover during the late summer, ice-free season (this volume, chapter 1). As the ice melts across Foxe Basin, bears become stranded on small ice floes and eventually retreat to nearby land masses where they wait for ice to return. Hence, high densities of bears tend to accrue on land adjacent to late-melting ice, especially islands where dispersion is limited. We selected Rowley Island as our study site: its high density of bears during the ice-free season, contrasting dark landscape, and flat terrain provided an ideal setting to evaluate the utility of satellite imagery.

Satellite Imagery

We procured target images of Rowley Island (~1,100 km^2) from DigitalGlobe, Inc. (WorldView-2 satellite; ~0.5m resolution at nadir; Quickbird, 0.65 m resolution), during early September, 2012. We compared these images to reference images to discriminate bears (~2-m white objects on the target image but not the reference image) from non-target objects. Reference imagery was acquired during August, 2009 and 2010

(WorldView-1, 0.5m resolution) and August, 2012 (Quickbird). We corrected all images for terrain (i.e., orthorectification). To account for any differences in sensor exposure settings and sun irradiance based on time of year and day, we calculated top-of-atmosphere reflectance (following Updike and Comp 2010) using relevant metadata from the imagery (per band), earth-sun distance at time of acquisition, and sun elevation angle. We applied an additional histogram stretch to brighten darker, non-ice areas (identical for all images) in order to facilitate image comparison by human analysts. We used a python script that leverages the open-source Geospatial Data Abstraction Library package for image manipulation and ArcGIS 10.1 (Environmental Systems Research Institute; Redlands, California, USA) to overlay target images on reference imagery. Two independent observers visually identified potential polar bears on the September, 2012 image and recorded latitude and longitude. Observers initially reviewed imagery at a fixed scale of 1:2,000 to 1:3,000 and subsequently examined potential polar bears at multiple scales (up to ~1:250) and in comparison to reference images to help distinguish likely bears.

Following this independent review, the two observers jointly examined imagery to resolve uncertainties in identification of potential bears. We did not categorize an object as a "presumed bear" unless observers were in agreement and confident in that classification. We thus deleted some points from each observer's initial list, but observers did not add points to their respective sightings during this process.

We treated each observer's review as an independent sampling period, enabling us to generate capture histories for mark-recapture analysis. We employed a full

likelihood-based, closed population model (Otis et al. 1978), facilitating direct estimation of abundance and detection. We allowed detection probabilities to vary between observers and conducted modeling in Program MARK (White and Burnham 1999).

Ground-truthing

We used a helicopter (Bell 206L) survey to assess how well we distinguished polar bears from objects of similar size and color on imagery. We categorized 26 points on imagery as either polar bears or non-target, light-colored control points (e.g., rocks, foam on water surface), and we flew to these sites to confirm identity. We assumed that a bear had been present when the site was photographed if 1) there was no rock or other feature that could be confused with a bear and 2) the site was not prone to ephemeral landscape features (e.g., not downwind of a pond that could have had foam when the image was collected).

. We also used a helicopter survey to obtain a second population estimate of bears on Rowley Island via mark-recapture distance sampling (Laake and Borchers 2004). We could not directly compare polar bear sightings during this aerial survey (August 31 – September 1, 2012) with points on the target image (September 3) because bears moved in the 2−3 days that elapsed between events. However, we assumed that Rowley Island was a closed population during this short time frame, enabling us to compare abundance estimates derived by the two techniques.

We implemented mark-recapture distance sampling (MRDS; Laake and Borchers 2004) protocols for abundance estimation. MRDS combines distance sampling with a double-observer platform; the double-observer data are incorporated in a mark-recapture

modeling framework to explicitly test distance sampling's assumption of perfect detection at distance 0 (Buckland et al. 2001) and inflate density estimates, if necessary. Here, bears observed by the pilot and front seat observer were considered marked, while those observed by rear seat observers were considered recaptured. We surveyed Rowley and nearby islands to obtain a sufficient sample for estimating the detection function. We oriented sampling transects perpendicular to each island's primary axis and extended them across island widths (Figure 1). Transects were spaced at 7 km intervals, and we sampled at an above ground level altitude of 120 m (400 feet) and target airspeed of 160 km/h (85 knots). Flight parameters were based on previous overland aerial surveys of polar bears in the region (this volume, chapter 1). Bears observed by the pilot and front seat observer were considered marked, while those observed by rear seat observers were considered recaptured. We recorded flight paths and locations of polar bear sightings with a GPS, and measured distances from transects to observations in a GIS (modified from Marques et al. 2006). We documented group size and recorded conditions that may have impacted detection (weather, lighting).

We conducted preliminary double-observer analyses with the Huggins model (Huggins 1989, 1991), which suggested that detection on and near the transect line was nearly perfect. Hence, we analyzed data in the conventional and multiple covariate distance sampling engines of Program DISTANCE 6.0 (Thomas et al. 2010). We pooled sightings data from all islands to estimate a common detection function and used encounter rates and group sizes from Rowley Island to obtain an island-specific abundance estimate. We considered models with standard key functions and series

expansion terms as well as covariate-based models. Because we could not reliably differentiate family groups on satellite imagery, our aerial survey estimate included only independent bears (i.e., excluded cubs or yearlings with their mother). We used Akaike's Information Criteria, adjusted for small sample sizes (AIC_c) (Burnham and Anderson 2002), for model selection.

Preliminary analyses suggested that detection on and near the transect line was nearly perfect, a fundamental assumption of distance sampling (Buckland et al. 2001). Hence, we analyzed data in the conventional and multiple covariate distance sampling engines of program Distance 6.0 (Thomas et al. 2010). We pooled sightings data from all islands to estimate a common detection function and used encounter rates and group sizes from Rowley Island to obtain an island-specific abundance estimate. We considered models with standard key functions and series expansion terms as well as covariate-based models. Because we could not reliably differentiate family groups on satellite imagery, our aerial survey estimate included only independent bears (i.e., excluded cubs or yearlings with their mother). We used Akaike's Information Criteria, adjusted for small sample sizes (AIC_c), for model selection.

RESULTS

Satellite Imagery

We detected 92 presumed bears on Rowley Island (Figure 1) and documented likely family groups (adult females with cubs) on five occasions. The most highly supported model included separate detection probabilities for the two observers and yielded an abundance estimate of 94 (95% confidence interval: 92–105) independent

bears. Individual detection probabilities varied greatly between the two observers (96% [95% CI: 83%–99%] and 42% [95% CI: 32%–52%]). Although it was generally straightforward to distinguish bears from other objects (Figure 2), landscape features and environmental characteristics sometimes complicated detection (Figure 3). About 12% of the reference imagery was obscured by clouds, and strong winds on the date of imagery collection created large expanses of foam along the banks of ponds that initially appeared to be bears and were absent from reference imagery. Additionally, some rocks reflected light differently between successive photos, requiring careful scrutiny to differentiate them from bears. However, joint review of imagery enabled us to correctly categorize all points that we ground-truthed via helicopter as presumed bears (n = 13) or inanimate objects (n = 13).

Aerial Survey Abundance Estimation

During the helicopter aerial survey, we sighted 56 polar bear groups totaling 77 individuals along ca. 400 km of transects across all study islands; this included 33 groups (34 independent bears) during ca. 160 km of sampling on Rowley Island (Figure 1). Despite a small number of detections, our data facilitated estimation of a robust detection function, and abundance estimates were consistent among the most highly supported models (Table 1). Our model-averaged estimate of abundance (including models $\Delta AIC_c < 3$) yielded 102 independent bears on Rowley Island (95% CI: 69 – 152).

DISCUSSION

Satellite imagery shows promise as a means to quickly and safely monitor the abundance and distribution of polar bears using onshore habitats. We were able to

discriminate among presumed bears and non-targets by comparing high resolution images collected at different points in time. The remarkable consistency between our estimates of abundance derived from imagery and established aerial survey techniques suggests that bear identification using imagery was quite accurate. We believe that the methods employed here (use of reference images, review by multiple observers to build consensus and generate capture histories, and estimation of abundance and detection probabilities via population models) provide a framework for other small-scale studies. However, applications at broader geographic scales may necessitate the development of automated image classification processes to expedite review and analysis; our initial, independent review of imagery required a combined 100 hours.

Observers differed substantially in their abilities to detect bears with imagery. Although this finding was unexpected, it does not diminish the robustness of our results or the potential utility of satellite imagery in other monitoring applications. In our study, the two observers had vastly different levels of experience: one had several seasons of experience studying polar bears in this landscape during the ice-free season, whereas the other had extensive experience interpreting remote sensing imagery but no direct experience with polar bears. The observer with field experience had better detection of bears on the images, suggesting that familiarity with the study landscape and first-hand knowledge of bear biology and behavior (e.g., variation in color and body outline based on posture) greatly improved detection. Moreover, the observers searched imagery somewhat differently. We found that detection was higher when the target and reference images were regularly compared, rather than using the reference image to simply verify

the presence of bears; one's eye was attracted to white spots on the target image not present on the reference image. These experiences suggest that a rigorous training program, explicit search protocols, and integration of individuals with relevant, on-the-ground experience with the target species will be essential to successful implementation of the technique.

The two techniques provided significantly different estimates of precision (coefficients of variation for line transect aerial survey: 20.4% versus satellite imagery: 2.5%). Distance sampling incorporated multiple variance components, including detection and encounter rates. Conversely, the satellite imagery modeling only included a variance component for detection, since we reviewed imagery from the entire island. The very high detection probability of one imagery observer also contributed to this difference. Variance estimated from manual review of imagery would increase in applications in which observers have lower detection probabilities or if procured imagery only 'samples' the study site.

Synchronizing collection of satellite imagery with visual surveys is not currently possible, since there is no assurance as to when the satellite's orbit will pass above the study area and if weather will be conducive to shooting imagery or conducting an aerial survey. This reality prohibits directly matching bears identified on photos with bears observed during an aerial survey. As such, absolute confirmation of presumed bears is impossible, and thus some false positives (i.e., inanimate objects classified as bears) or negatives are likely to occur.

Because one observer had a very high detection probability (96%), we deemed it unnecessary to model potential sources of heterogeneity. However, future studies may be compelled to quantify variables potentially impacting detection. We hypothesize that environmental conditions including wind, light, and the presence of clouds and small onshore ice floes may affect detection (Figure 3). Other prospective covariates may include bear reflectance values, bear size (i.e., pixels), reflectance values and complexity metrics for the surrounding landscape at multiple spatial scales, image exposure, and off-nadir angle at image collection (Boltunov et al. 2012).

We presumed that cubs were not consistently identifiable on imagery, given the resolution constraints. Their presence was suggested by multiple white spots of notably different sizes in a cluster (ca. <20 m). We detected only five likely family groups with imagery, whereas the nine family groups sighted on Rowley Island during aerial survey sampling suggest that there were ~28 family groups present island-wide. The inability to reliably discern family groups poses some limits on the utility of imagery for demographic studies. However, the advent of higher resolution imagery (e.g., WorldView-3 platform, set to launch in 2014, will shoot at 0.3 m resolution at nadir) may permit differentiation of cubs, as well as improve detection of smaller species, in the future.

With minimal topographic relief and high densities of polar bears during late summer, our study islands provided a model setting to test satellite imagery as a monitoring tool. Conditions elsewhere in the Arctic, however, are less ideal, and further technique development will be required to more broadly apply the technology. Priority

research and development areas for polar bears should include assessing onshore sites with lower densities and more variable landscapes (e.g., higher topographic relief) and evaluating sampling intensities necessary to obtain reliable density estimates and distributional information. Additionally, multi-spectral imagery may better capture unique spectral signatures of the target species, thereby improving manual and automated detection in more challenging onshore environments. Multi-spectral imagery also may facilitate the detection of polar bears on sea ice, given the apparent spectral differences between bears and snow at short wavelengths (G. LeBlanc, National Research Council Canada and C. Francis, Environment Canada, unpublished data).

The success of this technique with polar bears suggests that satellite imagery would likely provide a useful means to inventory other megafauna as well. In the Arctic, darker species such as musk oxen (*Ovibos moschatus*) and caribou (*Rangifer tarandus*) may be readily detected against a snow-covered, springtime landscape. Whereas satellite imagery does not yield the same detail of information as traditional capture programs and aerial surveys, it has tremendous potential to provide coarse abundance and distribution data from sites otherwise too logistically challenging or costly to routinely access. The technology can open vast, remote regions to regular monitoring, facilitating the collection of data across species' ranges and at global scales. Understanding and predicting shifts in abundance and distribution of wildlife is critical to evaluating ecological impacts of a rapidly changing climate. With archives dating back nearly a decade, imagery provides the opportunity to establish short-term longitudinal data.

Table 1. Results of distance sampling analyses of a polar bear aerial survey conducted in northern Foxe Basin, Nunavut, Canada during August – September, 2012. Highly supported models ($\Delta AIC_c < 3$) are presented. In the column Model, the key function is followed by adjustment terms or covariates (VIS = visibility; poor / fair (e.g., glare, light fog or rain) or excellent; LIGHT = light conditions; overcast, mostly cloudy, or partly cloudy / clear). *w* = Model weight. ESW = Effective strip width (meters). p = Detection probability. $\widehat{N}$ = Abundance estimate. Goodness of Fit metrics: C-S = Chi-squared; K-S = Kolmogorov-Smirnov; C-vM = Cramér-von Mises.

			Estimate (SE)			**Goodness of Fit**		
Model	**ΔAIC_c**	***W***	**ESW**	**P**	**$\widehat{N}$**	**C-S**	**K-S**	**C-vM**
Uniform / Cosine	0.00	0.306	1234 (65)	0.53 (0.028)	97 (17.8)	0.75	0.79	0.6
Half-normal / None	0.15	0.284	1151 (114)	0.50 (0.049)	104 (21.0)	0.75	0.94	0.8
Half-normal / VIS	1.12	0.175	1136 (114)	0.49 (0.049)	105 (21.3)	0.66	0.87	0.7
Half-normal / LIGHT	1.38	0.153	1112 (115)	0.48 (0.050)	108 (22.1)	0.55	0.91	0.8
Hazard / None	2.62	0.082	1201 (168.4)	0.52 (0.073)	100 (22.5)	0.53	0.89	0.8

Figure 1. Polar bears detected with high resolution satellite imagery and during the helicopter-based aerial survey in northern Foxe Basin, Nunavut. Target imagery was acquired from Rowley Island (dark shade) with the WorldView-2 satellite on September 3, 2012. Transects were spaced at 7 km intervals during the aerial survey. The Foxe Basin polar bear subpopulation is outlined in black and the study area shaded red in the inset.

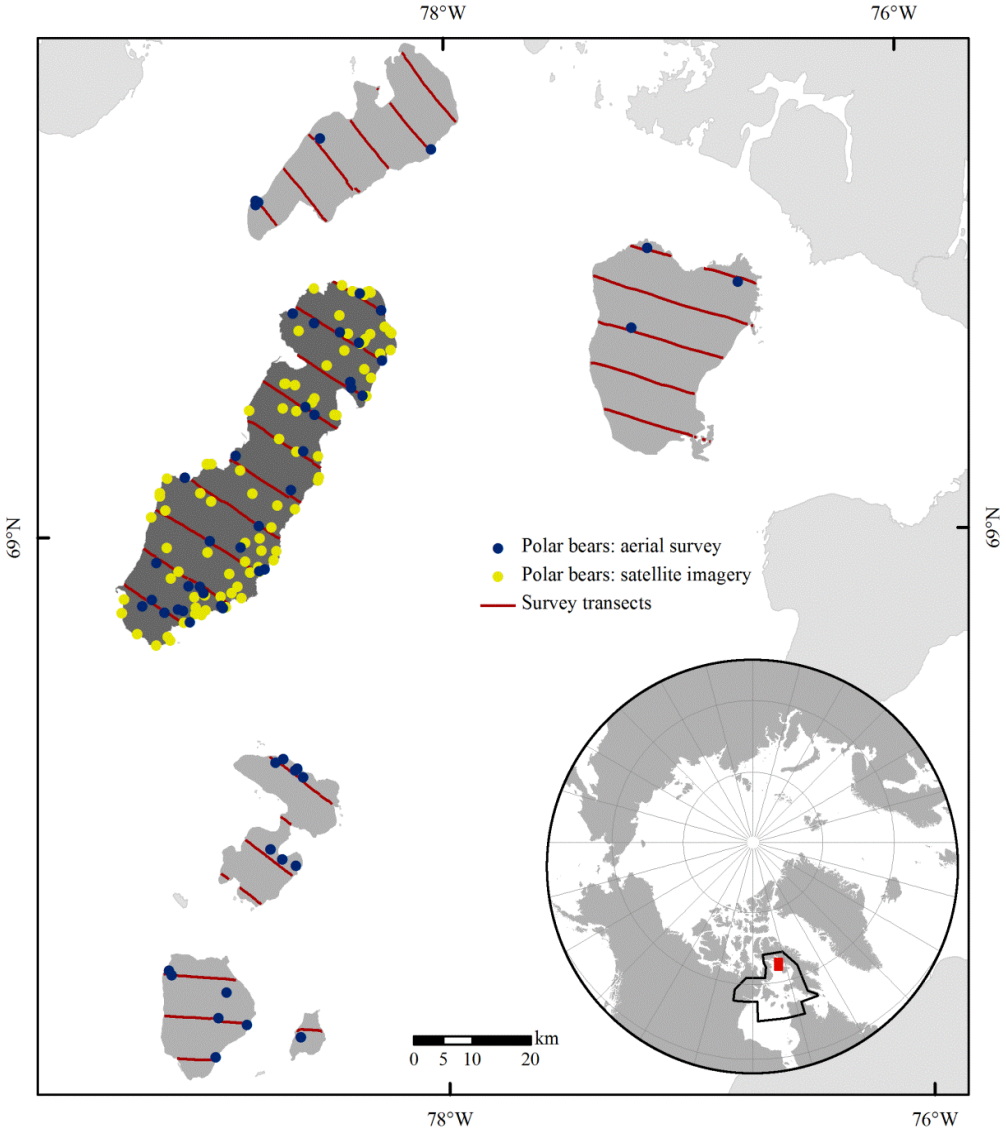


Figure 2. Example of high resolution satellite imagery used to detect polar bears on Rowley Island in Foxe Basin, Nunavut during late summer, 2012. The target imagery (top) was searched for polar bears, and the reference imagery (bottom) was used for comparison. Polar bears are present in the example target image but absent in the reference image (yellow circles). Landscape features that remain consistent between images, including rocks and substrate, are denoted with red arrows. Imagery courtesy of Digital Globe, Inc.

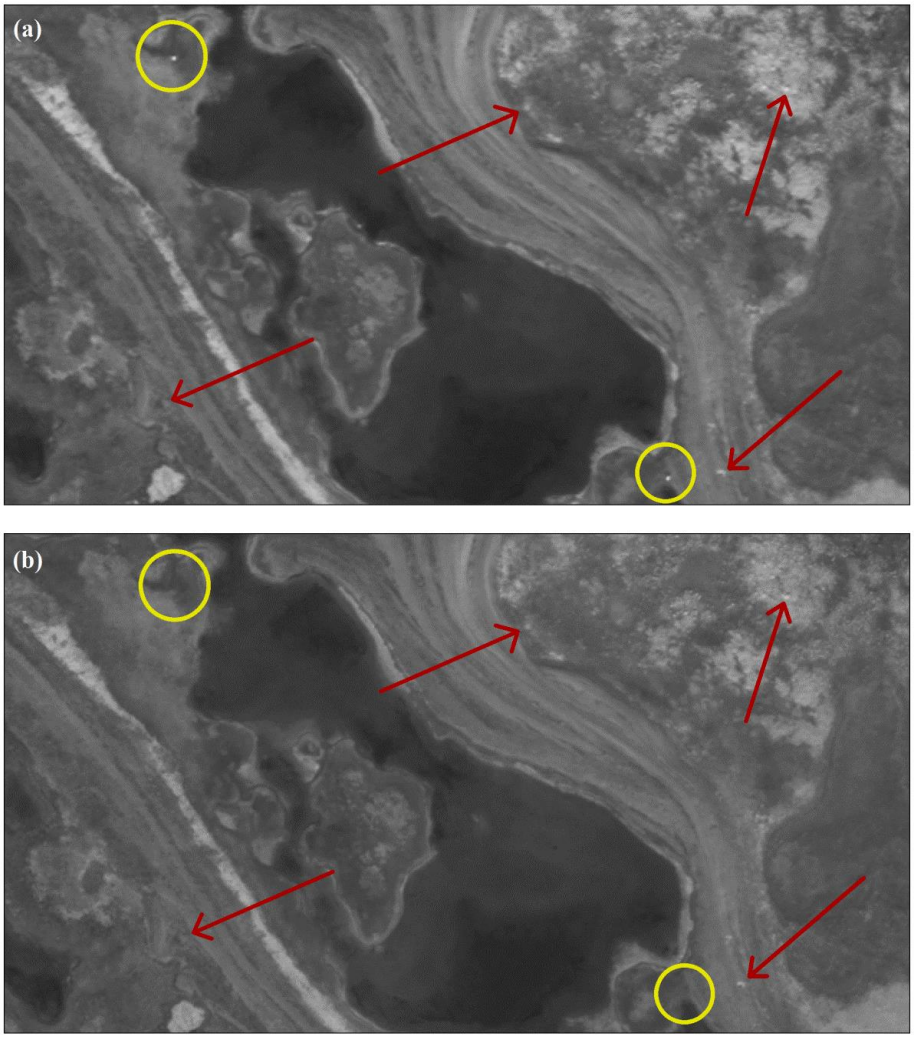
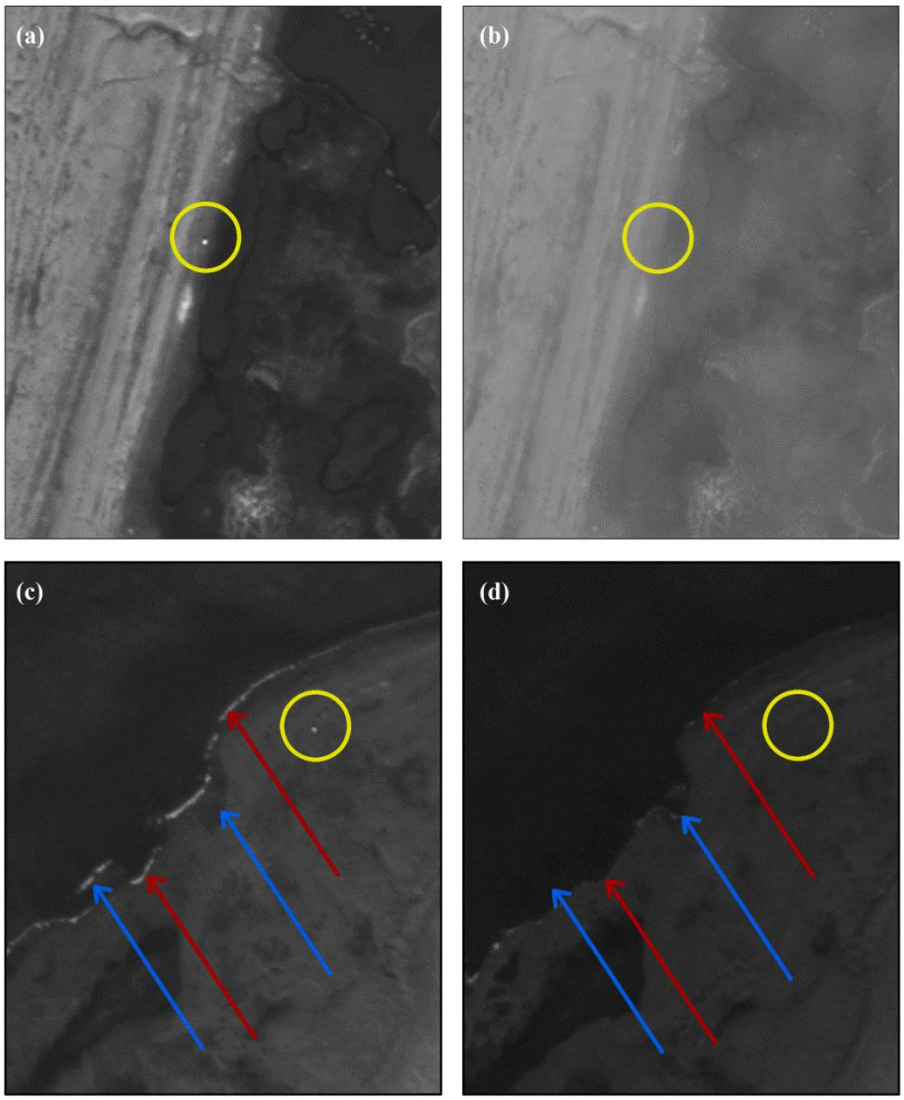


Figure 3. Clouds (top) and variable water conditions (bottom) are among the factors that may hamper detection of bears. Bear locations are indicated in target (left) and reference (right) images with yellow circles. Foam accumulating along the edges of water bodies and changes in water levels between target and reference images are indicated in the bottom pair of shots by red and blue arrows, respectively. Imagery courtesy of DigitalGlobe.



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