2021 AERIAL SURVEY OF THE WESTERN HUDSON BAY POLAR BEAR SUBPOPULATION

FINAL REPORT

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Disclaimer

The opinions in this report reflect those of the authors and not necessarily those of the Government of Nunavut, Department of Environment.

We dedicate this report to Markus Dyck; a dear friend and colleague who will be missed but of whom we have many fond memories. Your dedication to the future of polar bears and their place in the world will forever be remembered. We will do our best to make your contributions known to others. This is for you Markus. See you later mate.

(Photo credit: J. Ware, 2020)

SUMMARY

Relative to most polar bear subpopulations, the Western Hudson Bay (WH) subpopulation has been the subject of significant, long-term research, monitoring and management efforts. Concerns about the observed effects of long-term changes in sea-ice habitat, harvest pressure and increasing levels of human-bear conflict have made this subpopulation the focus of attention in discussions about the status of polar bears globally. Recently, part of the monitoring regimen for WH has involved a plan to conduct aerial surveys at 5-year intervals in-order to closely track abundance and distribution. Here, we report the third in this sequence of surveys.

Using double observer (sight-resight) and distance sampling methods from helicopter and fixed wing platforms, an aerial survey of the WH polar bear subpopulation was conducted in late August and early September 2021. Survey design, including study area stratification and transect spacing, was based on previous surveys in 2011 and 2016. Transects were oriented perpendicular to the coastline to align with bear density gradients.

We recorded a total of 194 bears in 125 groups. Like previous surveys, bears were concentrated along the coast and offshore islands, although both lone individuals and family groups were also regularly sighted inland, particularly within the Wapusk National Park region. The estimated abundance of WH in 2021 was 618 bears (SE=119.3, CI=425-899, CV=0.19). Comparison to aerial surveys estimates from 2011 and 2016 suggests that WH may be decreasing in abundance. Post-stratifying the results by sex and age classes revealed significant declines in the abundance of adult female and subadult bears between 2011 and 2021.

We were unable to definitively conclude whether the finding of declining abundance in WH over the last decade, specifically that of adult females and subadults, was the result of reduced survival and recruitment, movement of bears into neighbouring subpopulations (emigration), or harvest pressure. Based upon the multiple lines of evidence reviewed in

this report, it is plausible that all these factors have contributed to some degree. Of particular concern, however, is our finding that the observed declines are consistent with long-standing predictions regarding the demographic effects of climate change on polar bears. If these apparent trends continue, the progression of a reduced subadult cohort into the adult age class, combined with an already reduced adult female class, reproductive senescence, and mortality amongst older bears, may result in cascading effects on WH abundance and reproductive performance over the next decade. We therefore provide several recommendations for timely follow-up to these findings.

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1. INTRODUCTION

1.1. Status of Polar Bears

Across the Arctic, warming temperatures and changes in circulation patterns have led to profound changes in sea-ice, including declines in its extent and thickness, earlier spring melt (‘break-up’), later fall advance (‘freeze-up’) and longer seasonal ice-free periods (Stroeve et al. 2012; Overland and Wang 2013; Stern and Laidre 2016; Stroeve and Notz 2018). These climate-induced changes have been identified as the ultimate threat to the status of polar bears (*Ursus maritimus*); a species whose life history is reliant upon sea-ice as a primary habitat for foraging, movement, and reproduction (Wiig et al. 2015; Regehr et al. 2016). The sensitivity of polar bears to changes in sea-ice habitat quality is evident from the numerous field studies that have demonstrated associations between sea-ice habitat metrics and polar bear survival, body condition, growth, energy expenditure, movement patterns, distribution, reproductive performance and abundance (e.g. Stirling et al. 1999; Regehr et al. 2007; Rode et al. 2010; Cherry et al. 2013, 2016; Lunn et al. 2016; Obbard et al. 2016; Johnson et al. 2020; Laidre et al. 2020a and b; Pagano et al. 2021). Although most of these studies have documented negative consequences for polar bears from long-term changes in Arctic sea-ice, some have documented positive responses, in terms of body condition and abundance (Rode et al. 2014; 2018; Laidre et al. 2020b; Dyck et al. 2020a and b; Regehr et al. 2018; SWG 2016).

Observed variation in the responses of polar bears to a warming Arctic has largely been consistent with predictions (Stirling and Derocher 1993; Derocher et al. 2004; Stirling and Parkinson 2006; Stirling and Derocher 2012; Regehr et al. 2016). Subpopulations in southern portions of the species’ range, where the annual sea-ice melts completely during summer and autumn (e.g. polar bears in the seasonal sea-ice ecoregion *sensu* Amstrup et al. 2008) have been the first to experience negative effects (e.g. Striling et al. 1999; Rode et al. 2010; Obbard et al 2016; Lunn et al. 2016; Johnson et al. 2020). In contrast, some subpopulations in northern portions of the range are currently showing positive effects as these regions shift from multi-year sea-ice to thinner, annual sea-ice and a longer period of open water (Rode et al. 2014, 2018, 2021; Laidre et al. 2020b; Dyck et al. 2020a and b; Regehr et al. 2018). Thinner, annual ice provides better habitat for both polar bears and their ice-breeding seal prey. Combined with rising ocean temperatures, longer periods of open water, and greater penetration of sunlight into the water column, marine ecosystem productivity in these northerly areas appears to have increased thereby increasing the carrying capacity to support higher densities of bears (Stirling and Derocher 2012; Rode et al. 2014, 2018, 2021; Laidre et al. 2020; Dyck et al. 2020a ; Dyck et al. 2020b ; Häder et al. 2014, Frey et al. 2018). However, these benefits are predicted to be temporary and ultimately replaced by negative effects on polar bears if sea-ice conditions continue to deteriorate in the future. Overall, therefore, the predicted long-term consequence of continued sea-ice loss is that global polar bear abundance is likely to decline by greater than 30% over the next 4 decades (Regehr et al. 2016).

The polar bear is listed as a “Vulnerable” species by the International Union for the Conservation of Nature (IUCN) (Wiig et al. 2015) and a “Species of Special Concern” within Canada (ECCC 2020). Across its circumpolar range, the species is comprised of 19 subpopulations. Current assessments regarding the state of knowledge and demographic status of these subpopulations presents a varied picture reflecting the expected variation in subpopulation responses to climate change, as well as the immense financial and logistical challenges of monitoring this species in its remote, inhospitable range. As of 2021, ten subpopulations were assessed as ‘Data Deficient’, two as ‘Likely Increased’, four as ‘Likely Stable’ and three as ‘Likely Decreased’ by the IUCN’s Polar Bear Specialist Group (PBSG 2021).

1.2 Western Hudson Bay Subpopulation

The Western Hudson Bay (WH) subpopulation is one of three currently assessed as ‘Likely Decreased’ by the PBSG. This subpopulation ranks as one of the most intensively studied large mammal populations worldwide, with a history of scientific research and monitoring dating back five decades (e.g. Jonkel et al., 1972; Stirling et al., 1977;

Derocher and Stirling, 1995a; Regehr et al., 2007; Stapleton et al. 2014; Lunn et al. 2016; Dyck et al. 2016). Implemented predominately through mark-recapture sampling, findings from these studies suggest that WH abundance increased during the 1970s, remained somewhat stable, and then declined by an estimated 22% between 1987 and 2004 (Derocher and Stirling 1995; Lunn et al. 1997; Regehr et al. 2007). A more recent analysis suggests the population remained stable between 2001 and 2011 concurrent with a period of stability in sea-ice conditions (Lunn et al. 2016). In addition to these trends in abundance, linkages have been established between sea-ice conditions in WH and polar bear body condition, reproduction, movement patterns, distribution and survival (e.g. Stirling et al. 1999; Regehr et al. 2007; Cherry et al. 2013, 2016; Lunn et al. 2016; Johnson et al. 2020).

Long-term concern for WH polar bears, centers on the impact of a progressively earlier spring sea-ice break-up and delayed fall freeze-up that has, and will, result in bears spending longer periods on land where they have limited access to food (Stirling et al. 1999; Regehr et al. 2016; Stern and Laidre et a l. 2016; Johnson et al. 2020). Satellite-derived sea-ice data for the period 1979 to 2014, indicate that the length of the annual ice-free period in WH is increasing by approximately 9 days per decade (Stern and Laidre 2016). If this trend continues, further reductions in body condition, reproduction, survival and ultimately the abundance of polar bears are expected to occur (Molnar et al. 2010; Lunn et al. 2016). Harvest of WH bears is an additional concern that has the potential to exacerbate subpopulation decline if not managed appropriately.

Within Canada, the assessment of WH status based on scientific knowledge contrasts with that based on Indigenous Knowledge (IK); as summarized by several authors (McDonald et al. 1997; Dowsley and Taylor 2006; Tyrrell 2006; Nunavut Tunngavik Incorporated 2007; Committee on the Status of Endangered Wildlife in Canada 2008; Dowsley and Wenzel 2008; Canadian Wildlife Service 2009; York et al 2016; PBTC 2021). Canada’s Federal-Provincial-Territorial Polar Bear Technical Committee (PBTC) assesses WH as “likely declined” based on scientific information and as “increased” based on IK (PBTC 2021). There is consensus between science and IK that polar bear abundance in WH has increased since the 1970's (Tyrell 2006). There is also agreement that polar bear distribution has changed, that more bears are being sighted around communities, that sea-ice breakup is occurring earlier, and that climate change is negatively influencing seal populations (NWMB 2007). However, in contrast to scientific evidence, Inuit perceptions of WH do not support the notion that abundance has declined since the mid-1980's (Tyrell 2006). Reports of more bears summering on land in the Nunavut portion of Hudson Bay (i.e., the Kivalliq region) and increased incidences of problem bears around camps and communities have been attributed to factors such as higher abundance, habituation of bears to human activities such as ecotourism, changes in behavior due to capture and handling for scientific research, and increasing use of unmanaged garbage dumps in communities along the Hudson Bay coastline (Stirling and Parkinson 2006; Dyck et al. 2007; NWMB 2007; Stirling et al. 2008a; Smith et al. 2022).

With the large body of scientific knowledge about observed and predicted declines in status and the disparities between science- and IK- based assessments, monitoring of WH has become a priority amongst governments, co-management agencies and stakeholders. Accurate and timely information is essential for detecting sudden changes in the subpopulation's status, if and when they occur (Von Graven et al. 2012; Derocher 2013); assessing population viability (e.g. Lunn et al. 2016); supporting adaptive or 'state-dependent' management measures, such as adjustments in harvest levels (Regehr et al. 2017a, b); and resolving apparently diverging views.

In 2011, the GN adopted the use of aerial surveys for monitoring WH (Stapleton et al 2014). Aerial surveys have been used for decades as a tool in monitoring wildlife species worldwide. Although their use for studying polar bears is a recent development, they have proven to be an effective means of monitoring subpopulation abundance, distribution and reproductive performance (Aars et al. 2009, 2017; Stapleton et al. 2014, 2016; Obbard et al. 2015, 2018; Dyck et al. 2017; Conn et al. 2021; Wiig et al. 2021). Aerial surveys are relatively inexpensive and quick compared to the mark and recapture methods that have typically been used to study polar bears. As such, they are an ideal tool for providing the frequent information updates needed to support near-real time adjustments in the management and conservation of a subpopulation undergoing unprecedented change. For communities in Nunavut that have expressed concern about the handling of bears or potential sampling biases in mark-recapture studies, aerial surveys have also offered an acceptable alternative. Unlike mark-recapture studies, however, aerial surveys do not provide estimates of demographic parameters such as survival rates, needed for population modelling and trend projection, nor do they allow sampling of individual bears to support studies in other areas of priority such as body condition, growth, diet, disease status, movements, habitat use and contaminant burdens (Von Graven et al. 2012). In the context of WH, therefore, aerial surveys and mark-recapture are both considered key components of the monitoring scheme.

A subpopulation wide aerial survey of WH was designed and implemented in 2011 (Stapleton et al. 2014). Based on the results and subsequent simulations, it was determined this survey should be repeated at 5-year intervals in-order to provide sufficient power to detect changes in subpopulation abundance (GN unpublished data). The survey was repeated in 2016 using similar methods (Dyck et al. 2017). Results from both surveys were used to facilitate status assessment by the IUCN and PBTC, as well as inform local harvest management decisions (PBSG 2021; PBTC 2021). Here we report the results of the third scheduled aerial survey of the WH subpopulation, conducted in 2021.

2. OBJECTIVES

The study’s objectives were to:

a) Generate an accurate and precise estimate of polar bear abundance in WH via aerial survey.

b) Evaluate the distribution of polar bears in WH during the 2021 ice-free season.

c) Compare results from the 2021 survey with those of surveys conducted in 2011 and 2016 to examine trends in subpopulation abundance, composition and distribution.

3. METHODS

3.1. Study Area

The annual range of the Western Hudson Bay (WH) subpopulation, located at the southern extent of the global polar bear distribution, stretches across roughly 435,000 km^2 of Hudson Bay and the adjacent coastline including portions of Nunavut, Manitoba, and Ontario (Figure 1). WH is part of the Hudson Bay complex that includes the neighboring Foxe Basin (FB) and southern Hudson Bay (SH) subpopulations (Obbard et al. 2010; Thiemann et al. 2008, Peacock et al. 2010). Although there is marked spatial overlap of polar bear movements from these three subpopulations while on the sea-ice (e.g., Stirling et al. 1999; Obbard and Middel 2012; Sahanatien et al. 2015), past capture-mark-recapture studies (Stirling et al. 1977; Derocher and Stirling 1990; Ramsay and Stirling 1990; Kolenosky et al. 1992; Taylor and Lee 1995; Derocher et al. 1997; Lunn et al. 1997, 2016), genetic studies (Paetkau et al. 1995, 1999; Crompton et al. 2008; Malenfant et al. 2016; Viengkone et al. 2020), and analyses of satellite telemetry data (Stirling et al. 1999; Obbard and Middell 2012; Sahanatien et al. 2015; Viengkone et al. 2020) generally support the currently accepted WH subpopulation boundary (Obbard et al. 2010).

Our study area has been well-described by Brook (2001), Dredge and Nixon (1992), Ritchie (1962), Clark and Stirling (1998), Peacock et al. (2010) and Richardson et al. (2005) and includes the areas described by Stapleton et al. (2014) and Lunn et al. (2016). The terrestrial portion of the study area stretches for approximately 1,500 km from about 35 km southeast of the Manitoba-Ontario border and northwards into Nunavut (approximately 20 km south of Chesterfield) (Figure 1). In addition to Rankin Inlet, the communities of Whale Cove and Arviat (Nunavut) and Churchill (Manitoba) are located within the boundaries of WH. In general, the southern portion of the study area displays the characteristics of the Hudson Plains ecozone and the Coastal Hudson Bay and Hudson Bay Lowlands. As described by Dyck et al. (2017), the northern portion exhibits Taiga and the Southern Arctic ecozone (Ecological Framework of Canada 2016). Where trees (black spruce [*Picea mariana*], white spruce [*P. glauca*], and tamarack [*Larix laricina*]) are quite common in the southern extents, dwarf birch (Betula nana), willows (*Salix spp*.), and ericaceous shrubs (*Ericaceae spp*.) are the norm to the north. The near-coastal southern areas exhibit elevated beach ridges, marshes and extensive tidal flats. There is very little relief (<200 m) with underlying continuous and semi-continuous permafrost. Sea-ice is absent in this region generally from July to November (Stirling et al. 1999; Scott and Marshall 2010; Stern and Laidre, 2016), and biting insects are plentiful during the summer (Twinn 1950).

Spatial separation of WH polar bears from individuals in neighboring subpopulations of the Hudson Bay complex is most complete during the late summer and early fall ice-free season when bears are on land (Peacock et al. 2010). Polar bears of WH come ashore when sea-ice levels diminish to ≤ 50% (Stirling et al. 1999; Cherry et al. 2013, 2016), which generally occurs during July (Stern and Laidre, 2016). Once on land, the bears segregate by sex, age class, and reproductive status within the study area where they exhibit fidelity to their terrestrial summer retreat areas (Stirling et al. 1977; Derocher and Stirling 1990). In general, adult males are found along the coastline, pregnant females and females accompanied by offspring are found in the interior denning area, which is mostly included within Wapusk National Park, and subadults are distributed throughout the study area (Stirling et al. 1977; Derocher and Stirling, 1990; Ramsay and Stirling 1990; Clark and Stirling 1998; Clark et al. 1997; Richardson et al. 2005; Stapleton et al. 2014). When sea-ice reforms during November, all bears except pregnant females return to the ice. Pregnant females give birth in terrestrial dens during December and early January, and these family groups generally depart their dens in March and April to return to the sea-ice (Jonkel et al. 1972; Stirling et al. 1977; Ramsay and Stirling 1988).

3.2. Survey Design and Field Methods

3.2.1. Survey Timing

The polar bear aerial survey was conducted in late August to early September 2021. This survey window during the ice-free period was selected because (a) all polar bears of the WH population are forced to be on land during this time; (b) range overlap between subpopulations within the Hudson Bay complex reaches a minimum, since polar bears exhibit a high degree of fidelity to terrestrial habitats during this period (Derocher and Stirling 1990; Lunn et al. 1997; Stirling et al. 2004; Parks et al. 2006); (c) the absence of ice and snow means that polar bears are readily observable against the landscape; (d) pregnant females are less likely to have begun denning yet and can be detected while moving towards their inland denning area (Stapleton et al. 2014); (e) non-denning bears have not yet begun to make directional northerly movements as they are known to do in the late fall, prior to freeze-up (e.g., Stirling et al. 1977; Derocher & Stirling 1990; Stirling et al. 2004); and (f) the two previous surveys in 2011 and 2016 occurred during a similar window.

3.2.2. Stratification

Like the 2011 and 2016 surveys (Stapleton et al. 2014; Dyck et al. 2017), we implemented a systematic, stratified study design to allocate sampling effort and improve estimate precision. For consistency, we used the same strata and sampling transects as the 2016 survey; themselves a modification of those used in the 2011 survey (Figure 2). The survey strata included the following derived polar bear density distributions: 1) very low, 2) low, 3) moderate, and 4) high. Descriptions of these strata, as provided by Dyck et al. (2017) are presented in Table 1. 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 denoted as perpendicular transects) to improve precision and minimize potential biases (Figure 2; Buckland et al. 2001).

3.2.3. Aircraft

One de Haviland Twin Otter fixed-wing aircraft with radar altimeter and an Airbus AS 350B2 twin engine rotary-wing aircraft with radar altimeter were used to complete the survey. All aircraft throughout the survey maintained, as close as possible, an altitude of 400 feet above ground level (AGL) and an air speed of between 70 and 90 knots for the fixed-wing, and 70 to 80 knots for the rotary-wing aircraft while flying on transect. The Twin Otter fixed-wing aircraft was used to complete the low-density stratum within Nunavut and the very low and moderate density strata west and north of the high-density stratum bounded by the Churchill River, Manitoba, in the south. The twin engine fixed-wing and its ability to fly on one engine was chosen to increase safety while flying over extensive open water transects characteristic of the northern half of the survey study area within Nunavut.

3.2.4. Double Observers

The double observer pair (sight/resight) method is a variation of physical mark-recapture (Pollok and Kendall 1987). The aircraft's front and rear observers comprise two independent survey teams, visually 'marking' (i.e., front observers' sighting) and 'recapturing' (i.e., rear observers' resighting) polar bears. Observer teams must be independent to estimate detection probabilities. This resultant information provides an independent estimate of the number of bears present in the survey strip that were not observed by either team (Laake et al. 2008; Buckland et al. 2010).

The double observer pair method requires two pairs of observers on each of the left- and right-hand sides of the aircraft (Figure 3) (Buckland et al. 2001; Pollock and Kendall 1987). One "primary" observer sits in the front seat of the aircraft and a "secondary observer" is located behind the primary observer on the same side of the aircraft. To ensure visual isolation, a barrier was installed between same side observers to remove any visual cues that could modify an observer's ability to sight the animal. Observers waited until bear groups passed before calling out the observation to ensure independence of observations. The data recorder/recorders categorized and recorded counts of each bear (group) into “primary only”, “secondary only”, and “both”.

3.2.5. Fixed-Wing

Within the fixed-wing aircraft, we utilized an 8-person platform; 4 dedicated observers, 2 data recorders (for each of the left and right primary and secondary observer pairs) and a pilot and co-pilot. Observers within the fixed-wing survey crew included members of Hunters and Trappers Organizations in the Kivalliq region of Nunavut. The observers were further divided into primary and secondary teams, each isolated from the other using visual barriers between the seats as well audio barriers using two independent intercom systems monitored by each of a primary data recorder/navigator and a secondary data recorder/navigator. The pilot’s responsibilities were to monitor air speed and altitude while following transects pre-programmed on a Garmin 650T Geographic positioning system (GPS). The data recorder/navigators were responsible for monitoring a second and third identically programmed GPS unit for the purposes of double-checking the position as well as to record the geographic position, body condition, composition and numbers of observed polar bear groups on data sheets.

The positioning of the four observers within the aircraft differed during and between survey days and was adjusted to account for any possible variability in sightability or the detection of polar bears associated with seat position. The front and rear observer exchanged spots mid-day, and the left and right sets of observers changed periodically. The data recorders and pilots did not vary their position within the aircraft and remained consistent during the survey.

3.2.6. Rotary-Wing

The AS 350B2 only allowed for a four-person configuration due to weight and balance issues while carrying full fuel as well as seating configuration. Using this configuration only the secondary observers were dedicated observers while the left primary observer seat was occupied by a data recorder/observer and the right primary position by a pilot/observer. Additionally, observers could not exchange primary and secondary positions using this configuration to determine sightability differences between seating positions. While the methods used during this study generally followed those used by Stapleton et al. (2014), it is important to note that no pooling of front and rear observers was made. All observations made during this study were independent.

3.2.7. Distance Sampling

In addition to the deployment of the double observer pair method within all aircraft, we also collected observations using distance sampling. The distance sampling method followed Buckland et al. (1993, 2004, 2010) and used the *mrds* (Laake et al. 2012) R package (R Development Core Team 2009) to model stratified line transect observation data and estimate density and abundance for polar bears. Using the conventional distance sampling approach (CDS), we modeled the probability of detecting a group of polar bears and their densities within five delineated strata as a function of distance where the detection function represents the probability of detecting a group of polar bears, given a known distance from the transect (Buckland et al. 2001). Recognizing that other variables may affect the detection probability, density estimates were also derived using multiple covariate distance sampling (MCDS), which allowed us to model probability of detection as a function of both distance and one or more additional covariates (Buckland et al. 2004). This approach was explored in-order to increase the reliability of density estimates made on subsets of the data based on terrain, vegetation, and environmental conditions, and to increase precision of the density estimates within each unique density-derived stratum (Marques et al. 2007).

For the fixed-wing portion of the survey only, and in addition to flying to the observed bears for position and data collection, we also used distance bins marked out with streamers and tape on the wing struts after Norton-Griffiths (1978) (Figure 4). In total, 6 distance bins were used including the following: 0-200 meters, 200-400 meters, 400-600 meters, 600-1,000 meters, 1,000-1,500 meters, and 1,500-2,000 meters. Though binned observations were not used during analysis, they did inform on the precision of binning for distance sampling platforms when compared to the actual observation waypoint recorded.

3.2.8. Observations

Polar bears observed while flying along a transect line were considered on-transect while those observed while ferrying to, from, or between transects, or to bear and/or wildlife sightings, were considered off-transect. Because polar bears are often found in groups, each observation (whether an individual or group) represented a group of polar bears. In this work a group of polar bears was defined as one or more individuals within a visually estimated 100-meter radius of one another. All observations were investigated by moving off the transect line to the center of the group as they were initially observed, to record the location, group size, sex/age classes, body condition.

We determined sex and body condition, to the extent possible, from approximately 30 meters altitude. Sex of bears was determined based on body size, the presence of morphometric characteristics (e.g., such as scars, large head, thick neck, long fur on front legs, vulva patch and urine stains) and behavior when encountered (SWG 2016). Age-class assessment from the air can be accomplished reliably for adult males, pregnant females, and members of family groups (Government of Nunavut, unpublished data; SWG 2016). Based on these methods, polar bears were classified as male or female, and as adult males (6+ years), adult females (5+ years), subadult males (2 to 5 years), subadult females (2 to 4 years), yearlings (>1 and < 2 years), and cubs-of-the-year (<1 year).

A general, relatively robust though subjective fat index has been successfully used in past studies to assess body condition of polar bears (Stirling et al. 2008; SWG 2016; Laidre et al. 2020a, b; Dyck et al. 2020a, b; Dyck et al. 2022). Standardized body condition indices [i.e., poor (1), fair (2), good (3), excellent (4) and obese (5)] were scored for each individual bear (Stirling et al. 2008). Each aircraft had at least one experienced biologist on board who could identify age classes and body conditions of observed bears with confidence.

Additional covariates that could affect detection probabilities were recorded for each group including activity when first sighted, height and density of surrounding vegetation, habitat class, visibility, cloud cover, glare and general habitat description (Table 2). Observation times were kept to a minimum to reduce disturbance and stress. All distances to the observations were measured perpendicularly from the transect line to the center of the observation and recorded along with the observation’s date and time of day. It was assumed that the bear location at initial observation was determined accurately with no effect of movement after detection on estimated distance from the transect line. The distance from line was then estimated using shapefiles of transect lines with GIS methods. All aircraft deployed the distance sampling methods and collection of covariate data consistently across the study.

3.2.9. Coastal Contour Transects

Like the 2011 WH survey (Stapleton et al. 2014), we additionally surveyed along comprehensive coastal contour transects covering the entire coastline of WH, independent of the transect flown perpendicular to the coast. Contour transects were flown at or slightly below the high-water line (HWL) with one side of the aircraft dedicated to monitoring tidal flats and near-shore waters (i.e. swimming bears). We surveyed along coastal contours as close to high tide as possible to minimize tidal flat exposure and reduce the need to double-back to ensure that the coastal zone was comprehensively covered. Observers looked for bears as far as they could reasonably see, not within a pre-defined strip width. Because perpendicular transects were extended to the shoreline and over tidal flats (where applicable), some bears along the shoreline could be sighted from both perpendicular and coastal transects. Bears sighted on tidal flats or in nearshore waters were considered within the coastal zone (i.e., on land, where area could be estimated with GIS) in order to calculate density. Although collected, coastal contour data are not analyzed and present in this report since such data were not collected during the 2016 survey thus preventing comparison of the three surveys.

3.3. Data Analyses

3.3.1. Data Screening and Truncation

Data was initially screened for outlier observations that occurred at far distances therefore creating a tail on the detection function that can be difficult to fit. A right truncation distance that eliminated the upper 5% of observations was considered to minimize the influence of these observations (Buckland et al. 1993, Stapleton et al. 2014). The blind spot under each aircraft was estimated using geometric formulas. From this, left truncation distances were estimated for the twin otter as 98.9 m and approximately 70 m for the AS350B helicopter. Adjusted distance from the transect line was then estimated as the distance from the transect line minus the left truncation distance for each aircraft.

Like the survey in 2016 (Dyck et al. 2017), but unlike that in 2011 (Stapleton et al. 2014), we left truncated both the front (pilot and data recorder) observations from the helicopter rather than only left truncating the rear observations. The rationale for this was that we wanted to keep the data sets as similar as possible for the double observer analysis. There were 3 observations of 7 bears that were only observed in the rear observer blind spot by the front observers in the helicopter. Therefore, the degree of reduction due to left truncation of the helicopter data was not large.

3.3.2. Distance Sampling Double Observer Analysis

Analysis methods

Mark-recapture distance sampling methods were applied to the survey data (Buckland et al. 2004, Laake et al. 2008a, Laake et al. 2008b, Buckland et al. 2010, Laake et al. 2012). A mark-recapture / distance sampling model assuming point independence was used which allows estimation of the detection probabilities at the transect line (or left truncation distance) using independent double observer methods with distance sampling methods used to model the decline in sighting probabilities as a function of distance from the survey line.

Covariates that affected bear sightability were considered that included environmental, observer and survey factors (Table 3). These covariates included group size, aircraft type, observer, and visibility. Like the 2016 survey, a remote sensing-based covariate (*RSveg*) based on LANDSAT 8 vegetation classification was also considered (Figure 5). The rationale behind this covariate was that it would systematically index dominant vegetation types in the proximity of observations therefore providing the best comparison of habitat and potential obstruction of observations across all observations. The main categories in Figure 5 that were present in the study area were gravel, shrub, trees, low vegetation, and water.

The twin otter fixed-wing aircraft had 2 dedicated observers per side of the aircraft. The A-star helicopter had 2 dedicated surveyors in the back seat of the helicopter and the pilot and data recorder/navigator as observers in the front. The pilot and data/recorder did not have the same view as the observers and were distracted by piloting the helicopter and navigating/data recording. Therefore, special covariates were formulated for the pilot and data recorder/observers in this aircraft.

Distance model fitting

A sequential process was used for model building. First, parsimonious distance sampling models were formulated using a mark recapture model with constant detection probabilities. Once the most supported distance model was determined, parsimonious mark-recapture models were formulated using the most supported distance model as a base model in the mark-recapture model analysis. As a final step, optimal distance and mark-recapture models were combined and assessed for goodness of fit and overall parsimony. Information theoretic methods (Burnham and Anderson 1992) methods were used to assess relative model fit. More exactly Akaike Information Criterion (AIC) were used as an index of model parsimony with lower scores indicating a model that explained the most variation in the data set with the least number of parameters. The difference between the most supported model and given model was evaluated (ΔAIC) to indicate relative support with models with ΔAIC values of less than 2 being of interest. Akaike weights were used to estimate proportional support of models. Models were averaged based on AICc weights using the *AICcmodavg* (Mazerolle 2016) package in program R (R Development Core Team 2009). The AIC score indexes relative fit but does not provide a test of overall goodness of fit. Goodness of fit tests incorporated the *mrds* package as well as graphical methods were used to further evaluate fit of the most supported models.

Abundance estimates were derived for the most supported models with variances being estimated assuming sequential systematic sampling (the S2 estimator in *mrds*). This estimator accounted for sequential lines sampling in the survey (Innes et al. 2002, Fewster 2011).

3.3.3. Trend Analyses

Given the previously observed declines in WH abundance and predictions regarding future decline, we examined estimates from the current series of 3 aerial surveys that have been conducted at 5-year intervals for evidence of decline. Data from the previous 2 projects were re-analyzed using the same methods (Dyck et al 2017). Trends in polar bear abundance estimates from the 2011, 2016 and 2021 WH distance sampling surveys were initially compared graphically. Estimates of trend were then derived using ratios of estimates. A simulation approach that assumed log-normal distributions of estimates was used to test for significance between successive estimates as well as confidence limits on overall (gross) change and yearly change in estimates. Log-normal distributions were assumed since they best describe the distribution of estimates from distance sampling and are also assumed when estimating confidence limits of estimates (Buckland et al. 2004). One thousand simulations of estimates were generated from a log-normal distribution for each year. The proportion of simulations where gross change (the ratio of successive estimates i.e. N_1:N_2) was greater than 1 was tallied. If this proportion was less than 0.05 then a significant decline was suggested. We note this test is equivalent to a one tailed test between two abundance estimates testing for evidence of decline (null hypothesis H_o: $N2 \geq N_1$ and alternative hypothesis H_a: $N_2 < N_1$). Confidence limits were then derived based on the 2.5th and 97.5th percentile of the resulting distributions of gross (GC) and annual change (with $\lambda = GC^{(1/\text{survey interval})}$). The 1-tailed hypothesis provides a more powerful test for decline than a 2-tailed test (which tests if the 2 estimates are equal). A 1-tailed test was justified given previously observed and predicted declines in the subpopulation; a question of key management interest.

To explore whether change was occurring within specific segments of the subpopulation, abundance estimates from 2011, 2016, and 2021 were post-stratified to derive estimates for adult males, adult females, and subadults. Trends within these segments were examined. Finally, there was interest in assessing whether change was occurring evenly across the study area or whether there were geographic patterns in change. To explore this, the 2011, 2016, and 2021 study areas were divided into Nunavut (Area 3), the area from Nunavut to the Nelson River (Area 2), and from the Nelson River to eastern boundary of WH in Ontario (Area 1). To do this required dividing the moderate strata used in 2011 and 2016 into a moderate north and south strata as was used in 2021. Estimates were derived based on strata in these areas for overall estimates as well as estimates of age/sex groups. The 2011 strata were redrawn and the areas of the 2 new strata double checked to make sure they were similar to the original single stratum.

Distance sampling analyses were conducted using the *mrds* (Laake et al. 2012) R package (R Development Core Team 2009). Data was explored graphically using the *ggplot* (Wickham 2009) R package with GIS analyses conducted using the simple features (sf) (Pebesma 2018) R package and QGIS program (QGIS Foundation 2020).

4. RESULTS

4.1. Survey Timing and Effort

Extensive forest fires across Canada in the summer of 2021 limited the availability of suitable aircraft and delayed the planned start of 2021 WH polar bear survey by approximately 1 week. The survey was flown between August 21st and September 6th. This compares to survey windows of August 13th to 29th and August 12th to 21st in 2011 and 2016, respectively. Using 50% sea-ice cover in WH as an index of ice break-up (e.g. Laidre and Stern 2016), the 2011, 2016 and 2021 surveys were conducted at 67 to 83, 55 to 64 and 65 to 81 days post-break-up, respectively.

Including weather-related delays, 2021 survey strata between Chesterfield Inlet and Churchill, flown using the Twin Otter, took 6 days to complete (August 21-27). The remainder of the study area, flown by helicopter, took 13 days (August 24-September 6). Approximately, 41 and 72 hours were flown with the Twin Otter and helicopter, respectively, for an estimated total distance of approximately 17,000 km, including ferry time. Weather during the survey was good and allowed for completion of all transects and coastal contour surveys.

4.2. Summary of Observations

Overall, 194 bears in 125 groups were observed during distance sampling with 176 observed on transect and 18 observed off transect (Table 4). Group sizes ranged from single individuals up to groups of 6. Eighteen cubs-of-the-year (COY) and 17 yearlings were observed. Mean litter sizes were 1.46 (SD: 0.50; n = 13) and 1.39 (SD: 0.63; n = 13) for COYs and yearlings, respectively[1].

Polar bear sightings were not uniformly distributed across WH (Figure 6; Table 4). Similar to surveys in 2011 and 2016, the highest concentrations of bears were documented in the high-density stratum, encompassing the historical Environment Canada study area, including Wapusk National Park and along the coast of southeastern WH. Eight percent

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[1] Includes off transect observations

of observations were in Nunavut. In general, observations were concentrated along or near the coast throughout the subpopulation (Figure 6). However, inland bears >10 km from the coastline were often recorded in the high-density stratum and less frequently observed in the southeastern portion of WH (the southern moderate density strata). Adult males were concentrated along the coast. In contrast, lone adult females or adult females with offspring, either COY or yearlings, were most often observed inland.

4.3. Abundance Estimation

4.3.1. Truncation of Observations

Observation data were left and right truncated (based on 97th percentile of distance from line after adjustment for left truncation) (Table 5 and Figure 7) yielding a dataset of 154 bears for analyses. Left truncation was based on measured blind spots for each aircraft and eliminated data not available to both observers. Using the right truncation distance (2100 m) eliminated excess observations at the tail end of the detection histogram that would exert influence on fitting of detection functions. A sensitivity analysis was conducted to validate the right truncation distance (Appendix 1).

4.3.2. Covariates

Habitat classes (as classified by observers) did not have sufficient sample sizes to allow modelling of detection functions for each class. These were pooled into similar categories (Figure 8). Overall, the detection histograms for each category were relatively similar (when considering differences in sample sizes). The remote sensing vegetation classification (RSveg) was also pooled into 3 categories (Figure 9). The detection histograms for each category were also relatively similar (when considering differences in sample sizes). When plotted on a map, both of these pooled vegetation classifications corresponded to shoreline and inland habitats (Figure 10). Other descriptors of vegetation, recorded for each bear observation, were vegetation height and density. Low density vegetation generally corresponded with vegetation of low height (Figure 11).

Vegetation density also corresponded to general habitat class with the shore/tidal areas mainly having low vegetation class ratings (Figure 12).

Group size can influence detectability. However, the relative range of group sizes was small with most observations being of single bears and few large groups over 3 bears (Figure 13). A bear’s activity when first observed was also considered. Observations were pooled into 4 main categories of activity (Figure 14). Visibility was recorded, with the majority of observations 101 of 102 being in clear to broken conditions and only one observation in fog. Glare, based on sun angle, was also recorded with 91 of 102 observations having no glare. Sun angle was overhead for most observations.

4.3.3. Summary of Double Observer Data

Sample sizes for observers were much higher for the helicopter that flew the higher density strata. Detection probabilities ranged from 0.6 to 1 (Table 6). Detection histograms amongst individuals were relatively similar for the helicopter-based observers (Figure 15). Low sample sizes precluded assessment of individual histograms for the Twin Otter-based observers.

A related issue occurred with the helicopter where the pilot and a potentially weaker observer were on the right side of the aircraft for the majority of the survey (Figure 16). This potentially resulted in a reduced number of observations on this side of the aircraft. To assess this issue, models were fit to exactly test the difference in detection function and double observer probabilities of sighting on the line for the right side of the helicopter. A HeliSide covariate was used which modelled detection functions (distance sampling) or detection probabilities at the line (MR analysis) for the right and left side of the helicopter and fixed-wing (sides pooled given limited sample sizes). Results of this analysis, presented in appendix 1, found little evidence of differences in detection function between the left and right side of the helicopter.

4.3.4. Model Selection

The first phase of model selection was assessment of covariates (Table 3) that describe the shape of the distance sampling detection function (Table 7). None of the covariates had substantial support with a constant hazard rate model being most supported. A variety of models had some support as indicated by ΔAICc scores of less than 2. The next step was assessment of variation in sightability near the aircraft from double observer models. The most supported distance model (hazard rate constant) was used for this analysis. Of covariates considered, vegetation density (VegDensity) had the highest support (Table 8).

The most supported distance and double observer models were combined (Table 9). Minimal variation was detected in the detection function based on covariates. A constant distance sampling model showed the highest support with double observer detection varying with vegetation density. Other models with helicopter navigator/data recorder and habitat class as covariates were supported, however, the sensitivity of abundance estimates (N) to model variation was low.

Goodness of fit of model 1 (Table 9) to the distance sampling (χ^2=4.7, d.f.=4, p=0.31), mark-recapture (χ^2=15.3, d.f.=12, p=0.23) components was adequate with adequate overall fit of (χ^2=20.1, df=12, p=0.23). A Von-Mises test also suggested adequate fit (Test statistic=0.028, p=0.98). Plots of fit to the model 1 suggest reasonable fit of predictions to front (observer=1) and rear (observer 2) observations as well as duplicate observations (seen by both observer) (Figure 17). Also suggested is minimal dependence between observer detections (as suggested by no directional trends in histograms by distance which indicate conditional probabilities). The mean single observer probability at the line for model 1 was 0.69 (SE=0.04, CV=0.07) with a combined (both observers) probability of 0.89 (SE=0.04, CV=0.46). Figure 18 shows predictions from model (DS (HR(.), MR(VegDensity) with predictions further delineated by VegDensity category and also group size observed, as represented by data point size

Estimates of abundance from the most supported model (Table 9, model 1) by strata are given in Table 10. A model averaged estimate of abundance using the models in Table 9 is 618 bears (SE=119.3, CI=425-899, CV=0.19). Of additional interest was a breakdown of estimates by adult males, adult females and subadults. Post-stratified estimates for these groups are presented in Table 11.

4.4. Abundance Trend Analyses

Data sets from WH aerial surveys conducted in 2011 and 2016 were analyzed and the results compared with those of the 2021 survey. A visual comparison of distance sampling total abundance estimates suggests a decrease in abundance from 2011 to 2021 (Figure 19). Additionally, post-stratified estimates suggest a decrease in the adult female and subadult classes with no apparent trend in adult males (Figure 19).

Gross and annual changes were estimated using the ratio of survey estimates with confidence limits calculated assuming a log-normal distribution of estimates for adult males, adult females, subadults, and all bears (Table 12). The annual rate of growth changed from 0.90 for subadults to 1.00 for adult males for the period 2011-21. The estimate of gross change was significantly lower than 1 for the 2011-21 interval for females, subadults, and all bears (at $\alpha=0.1$) suggesting declining abundance. This estimate was based on the proportion of log normal simulations where the estimate from 2021 was greater than the estimate of 2011. Similar results were derived from standard t-test comparisons of estimates (Appendix 2).

Of further interest was geographic trends in each sex and age class. Downward trends in adult females and subadults were suggested in Area 2 with less distinctive trends in other areas (Figure 20). Trends were significant for adult females and subadults between 2011 and 2021 for Area 2 but were not significant in other areas (Table 13).

4.5. Polar Bear Mortalities

Three polar bear carcasses were found while conducting the 2021 WH survey (Figure 21). Based on body size and dentition, one appeared to be an adult female and another a subadult. The third was identified as a 21-year-old adult male of known age, based on previous capture and tagging records. All three carcasses were in advanced stages of necrosis making cause of death difficult to determine. However, the adult male was noted to be in poor body condition making starvation a potential contributing factor in this case. A fourth polar bear carcass, that of an 18-year-old adult female, was found in WH in August 2021 during polar bear research unrelated to the aerial survey (S. Atkinson pers. obs). In 2 of the 4 cases, polar bears were observed feeding on the carcasses of their conspecifics.

5. DISCUSSION

5.1. Distribution

With some exceptions, discussed later in this report, the summertime distribution of polar bears within WH in 2021 (Figure 6) was broadly similar to previous surveys in 2011 and 2016(Stapleton et al. 2014; Dyck et al. 2017). The northern part of WH extending north from the Manitoba-Nunavut boundary, referred to as the low-density strata or Area 3 in this study, contained relatively few bears. Eight percent of bear observations recorded during the survey were found in this region, along the coast and offshore islands, compared to 6% and 5% in 2011 and 2016, respectively (Stapleton et al. 2014; Dyck et al. 2017). In the area extending Northwards from the Nelson River up to the Nunavut border, referred to as Area 2 in this study, the highest densities of bears occurred along the coastline. However, we also encountered a significant number of individuals far inland (>10 km), mostly within the bounds of Wapusk National Park. In contrast, virtually all polar bears in the region from the Nelson River eastward into Ontario (denoted as Area 1) were highly concentrated in a relatively narrow strip along the coast. In both areas, adult males were most often found on the coast while adult females tended to occupy areas further inland. These patterns of distribution and segregation have been well-documented in previous studies and attributed to several factors, including variation in the availability of suitable inland habitats for denning, the avoidance of conspecifics, thermoregulation, and insect avoidance (Stirling et al. 1977; Derocher and Stirling 1990; Lunn et al. 1997; Clark and Stirling 1998; Richardson et al. 2005).

Looking at estimates of abundance for the 3 areas of WH (Figure 20), we also found little evidence that the distribution of bears within WH has changed significantly over the last 10 years. Although, the estimated abundance of adult female and subadult bears in Area 2 decreased significantly between the 2011 and 2021 surveys, concurrent increases of these types of bears in Areas 1 (Cape Tatnum) or 3 (Nunavut) of WH were not found (Table 13). This suggests that the observed declines are not the result of more bears occupying regions to the north or southeast of what has historically been considered the core summer range of the WH subpopulation. Emigration, reduced survival and/or reduced reproductive performance could account for the observed declines. However, the finding that bear numbers (in absolute terms) were unchanged in areas bordering the northern (Area 3) and southeastern (Area 1) boundaries of the subpopulation between 2011 and 2021, makes emigration a somewhat less plausible explanation for the observed declines. Increased emigration from WH, if driven by changes in habitat availability, quality or phenology, such as sea-ice break-up patterns, would likely be concurrent with, or preceded by, apparent reductions in the abundance of bears in these ‘boundary’ areas. This is especially the case in Area 3, bordering the Southern Hudson Bay subpopulation, where there are high densities of bears and the distribution of remnant summertime sea-ice is known to affect the location bears come ashore (Stirling et al. 2004; Cherry et al. 2013).

Given our findings regarding distribution and the low number of bears observed and estimated in the Nunavut (Area 3) portion of the subpopulation, on-going mark-recapture studies in WH that focus sampling effort in a core study area centered around Churchill and Wapusk National Park (Area 2) are unlikely to contain significant bias in estimates of abundance or vital rates due to unsampled bears in Nunavut. However, relative to historical mark-recapture sampling (e.g. Regehr et al. 2007; Lunn et al. 2016), increased sampling effort is recommended in the area east of the Nelson River (Area 1), along the coast towards Ontario. As documented by the 2011, 2016, 2021 aerial surveys, and coastal surveys conducted by the Government of Manitoba, this area is typically occupied by several hundred bears during the on-land period. In generating estimates of WH abundance for conservation and management planning, it has been assumed by this and past aerial survey studies (Stapleton et al. 2014; Dyck et al. 2017) that these bears are part of the WH demographic unit. Studies are currently in progress to test this assumption (D. McGeachy per comm.).

5.2. Abundance

The estimated total abundance of WH polar bears in 2021 was 618 bears (SE=119.3, CI=385-852) based on model averaging. Overall, there was minimal change in estimates with different models suggesting that this estimate is robust to model selection uncertainty. The relative simplicity of the most supported distance sampling (constant) and double observer (vegetation density) models was surprising, given the range of covariates included in the analyses. However, inspection of histograms from the observed data does not indicate a large degree of variation in detection functions beyond differences in sample sizes. The covariate explaining variation in detectability in the vicinity of the aircraft was vegetation density. Vegetation density was also associated with vegetation height whereby low-density vegetation tended to be of low height (Figure 8). Support for this covariate suggests differences in detection between the inland versus coastal areas of WH. Bears occupying inland shrub or treed habitats tended to be harder to detect than those in open coastal areas.

Similar to the WH 2021 survey, previous aerial survey-based estimates of abundance in both WH and the neighbouring Southern Hudson Bay (SH) subpopulation (in 2011 and 2016) have all relied on relatively simple models with vegetation density, or a similar covariate reflecting vegetation density and height, included in top models (Stapleton et al. 2014; Dyck et al. 2017; Obbard et al. 2015, 2018). These earlier surveys have also included a covariate describing visibility, based on weather conditions, in some top models. This covariate was not among top models in the 2021 analysis potentially due to good survey conditions (i.e. clear, sunny skies) as well as the reduced sample sizes relative to 2011 and 2016 surveys. Overall, despite some differences in aircraft types and observers, the similarity of models amongst the 2011, 2016 and 2021 surveys suggests the sampling methods, maintained across these surveys, are robust and yield comparable datasets suitable for long-term trend monitoring.

The precision of the 2021 estimate was comparable to that of previous WH aerial surveys with a coefficient of variation (CV) of 19% compared to 17% for both the 2011 and 2016 surveys. Aerial surveys of polar bears conducted during the on-land or minimal sea-ice seasons have proven to be a cost-effective monitoring tool in subpopulations where flat terrain and high densities of bears that show interannual fidelity to on-shore regions facilitate detections (Stapleton et al. 2014, 2016; Dyck et al. 2017; Obbard et al. 2015, 2018). These types of surveys have yielded abundance estimates with CVs ranging from 11-19%. In contrast, surveys over sea-ice during the spring tend to be more expensive and have resulted in CVs ranging from 13-39% (Macdonald et al. 1999; Wiig and Derocher 1999; Evans et al. 2003; Aars et al. 2009, 2017; Conn et al. 2016; Wiig et al. 2021). The relatively poor precision of some ‘on-ice’ surveys is due to low bear densities and reduced detection probabilities on ice. On-ice surveying of WH is not recommended for several reasons including potential cost, expected poor precision and the extensive range overlap amongst individuals of the WH, Foxe Basin and Southern Hudson Bay subpopulations that is known to occur on the sea-ice (Peacock et al. 2010; Viengkone et al. 2018). Continued monitoring of the WH subpopulation via summertime aerial surveys is recommended.

5.3. Assumptions and Potential Biases

Generating unbiased (accurate) abundance estimates via the distance sampling method used in the survey is dependent on several assumptions (Buckland et al. 2001). To satisfy the assumption that bears were randomly distributed with respect to distance from the transect line, we surveyed with systematically spaced transects oriented perpendicular to the coastal density gradient; the same or similar transects flown in previous surveys (Stapleton et al. 2014; Dyck et al. 2017). A second assumption is that all bears present on the transect line or at the point of left truncation (i.e. distance zero) when the aircraft flies over are detected (Buckland et al. 2001). This assumption was evaluated in our analysis by estimating the combined double observer detection probability. Combined detection at the point of left truncation was 0.89 in 2021 compared to 0.97 and 0.90 for the 2011 and 2016 surveys respectively. This suggests that most, but not all, bears ‘on the line’ were detected. However, this bias was corrected by estimated detection probabilities on the line using the mark-recapture distance sampling approach.

A third assumption is that bears are not disturbed/displaced from their initial location, by the approaching survey aircraft, before being observed. This potential source of bias was minimized by maintaining rapid flight speeds of up to 150 km/hr thus reducing opportunities for bears to move great distances before detection. Ninety-three percent of bears were either sitting, laying, standing, walking or swimming and 7% were running when first observed during the survey. This suggests that most bears did not move or did not move significantly in response to aircraft. Accurate measurement of the distance between a bear sighting and the transect line is also essential (Buckland et al., 2001). We used GPS and GIS to measure distance from the line in accordance with accepted methods (Marques et al. 2006) that have been used extensively for polar bear aerial surveys (e.g. Aars et al. 2009, 2017; Obbard et al. 2016, 2018; Stapleton et al., 2014, 2016; Wiig et al. 2021). Assuming bears were recorded at their initial location, as discussed above, this method should have provided accurate distance data.

Another potential source of bias was the difference in number bear observations between the left and right sides of the helicopter either due to chance or weaker observers on the right side that were not accounted for by the mark-recapture distance sampling models. Although relatively small samples may have limited power, additional analyses to explore this issue did not provide evidence of bias in abundance estimate (Appendix 1). Additionally, any potential bias, while affecting the overall abundance estimate would not explain differences in abundance trends amongst adult males, adult females and subadults that we observed between 2011 and 2021.

Polar bears that were entirely hidden from both front and back observers during the survey would not have been incorporated into the abundance estimate. Two sources of this ‘availability bias’ were possible in our survey. Some WH polar bears, typically pregnant females, use earthen dens during the ice-free season, entering them as early as August (Jonkel et al. 1972; Stirling et al. 1977; Clark et al. 1997; Clark and Stirling 1998; Lunn et al. 2004; Richardson et al. 2005). Although we cannot correct for bears that were underground and entirely unavailable for observation during the survey, the number of such individuals was likely low. During the survey we observed numerous dens, including some that were freshly excavated. When observed, these dens were inspected from the air but none were found to be occupied. Bear occupying dense vegetation may also be harder to observe from the air than those in more open habitats. This reduced probability of observation was accounted for in the abundance estimate by incorporating a vegetation density covariate in models. Nevertheless, some individuals sheltering under dense vegetation, such as willows or trees, may be completely unavailable for observation. In particular, inland areas parts of WH, are tree covered creating the potential for bears to be unobservable from the air. However, densities of bears in these areas tend to be low since these treed areas are not preferred habitat for most bears. Thus, while we are unable to quantify this potential source of availability bias, our impression was that although trees and brush impeded detection and reduced sighting probabilities, it is likely that very few bears on or near the transect line were completely concealed by vegetation.

Directional, or migratory, movements of bears in, out or within the study area during a survey could lead to under and over counting of bears or observations of the same individuals more than once. Polar bears in WH make several directional movements during the on-land period. The first of these involves their migration from sea-ice onto land in the summer, the timing of which is known to be determined by sea-ice concentration (Cherry et al. 2013; Cherry et al. 2016; Pilfold et al. 2017). To minimize this potential bias, the survey was timed to occur after bears had migrated to land. Using 50% sea-ice cover as an index of break-up (Laidre and Stern 2016), the 2021 survey occurred 65 to 81 days post-break-up. Although the survey start was delayed due to availability of aircraft, the survey window was almost identical to that of the 2011 survey (67 to 83 days post-break-up) but approximately 2 weeks later than the 2016 survey. All three surveys occurred well after the 50% sea-ice threshold and when Canadian Ice Service maps indicate that WH was essentially ice-free. Thus, all WH bears should have been on-land at the start of these surveys. WH bears are also known the make directional movements northwards later in the fall (Stirling et al. 1977, Derocher and Stirling 1990; Bohart et al. 2020). Again, the survey was timed to avoid this period thus eliminating potential for this source of bias. Finally, field work was completed within a narrow temporal window and the aircraft were able to cover large expanses of land within a single day. Therefore, distributional shifts within WH during the study period did not impact our results.

In summary, because this study met analytical assumptions and potential sources of bias were minimized, we believe the aerial survey-based estimate of 618 bears (SE=119.3, CI=385-852) accurately reflects the number of polar bears within the bounds of WH during August 2021. Any biases in the aerial survey would likely result in an underestimate of the true polar bear abundance in WH. However, we note that such bias, if present, would not affect the ability to detect trends in abundance since the same methods (hence the same biases) were utilized in the 2011, 2016 and 2021 surveys.

5.4. Trends in Abundance

Estimates derived for the WH subpopulation indicated a possible decline in total bear abundance between 2011 and 2021. The 2011 survey produced an estimate of 949 bears (95%CI: 618–1280), the 2016 survey an estimate of 842 bears (95% CI: 562–1121) and this survey derived an estimate of 618 (SE=119.3, CI=385-852) for 2021. Although differences amongst these estimates were not statistically significant, total abundance has declined consistently between successive surveys. Significant reductions in the number of subadult bears and adult females appear to account for this observed change. Interestingly, the abundance of adult males remained unchanged during this period. These changes may be the result of internal demographic processes within WH (i.e. rates of birth and death) and/or changes in distribution leading to increased emigration of bears out of WH and into neighbouring subpopulations on a temporary or permanent basis. As discussed below, a review of multiple lines of evidence provides support for both hypotheses.

Several lines of evidence suggest that internal demographics have played a role in the observed decline in WH subpopulation abundance. The finding that abundance of subadult bears and adult females has declined whilst that of adult males has remained stable is particularly striking given its consistency with both long-standing hypotheses and field studies. Reduced recruitment and survival of subadults, hence reduced abundance, are typically among the first demographics effects to occur within large mammal populations experiencing density-dependent regulation (Fowler 1987). For polar bears, it was first predicted almost 30 years ago that negative impacts from things such as climate change would first appear amongst subadult and adult female bears (e.g. Stirling and Derocher 1993; Stirling et al. 1999; Derocher et al. 2004; Robbins et al. 2012; Klappenstein et al. 2020). These two classes of bears have nutritional ecologies that are likely to make them more vulnerable to deteriorating environmental conditions than other polar bears. Subadult bears must sustain the energetic costs of growth whilst also gaining experience in hunting. Adult females have the added costs of repeatedly raising litters of offspring over periods of up to 2 years, which is predicted to reduce their tolerance of fasting relative to that of adult males (Robbins et al. 2012; Stirling and Derocher 2012).

A series of field studies have validated the hypothesis that subadult and adult female polar bears are more sensitive environmental conditions than other classes of polar bears such as adult males. In some subpopulations, experiencing long-term declines in sea-ice, reductions in body condition (itself a precursor to reduced survival) have been greater and/or more readily detected amongst the adult female and subadult classes (Obbard et al. 2006, Rode et al. 2010; Laidre et al. 2020). In WH, Johnson et al. (2020) found that body condition and energy metrics have declined over time in relation to earlier sea-ice breakup with the most significant effects seen amongst solitary adult females and subadult males. Studies, including several in WH, have also demonstrated that links between sea-ice conditions and survival are stronger amongst subadults and adult females relative to other age and sex classes (Regehr et al. 2007; Bromaghin et al. 2015; Lunn et al. 2016). For subadults in WH, declining abundance between 2011 and 2021 may also be the result of reduced recruitment. Aerial surveys in 2011 and 2016, found low numbers of the yearlings compared to other subpopulations suggesting that recruitment into the subadult age class was poor in these years (Stapleton et al. 2014; Dyck et al. 2017). Additionally, mark-recapture sampling during this period also suggests recruitment has been low in multiple cohorts with the number of yearlings, expressed as a proportion of total captures, being less than 0.06 in 6 of the 10 years (ECCC unpublished data). A series of years with poor recruitment from the yearling age class, combined with potentially lowered survival amongst subadult cohorts, may thus have contributed to the observed decline in subadult abundance.

Changes in distribution leading to increased emigration of bears from WH into neighbouring subpopulations, such as SH, could also account for some of the variation in abundance observed across the 3 aerial surveys. Prevett and Kolenosky (1982) found significant interannual variation in aerial counts of bears along the southern coast of Hudson Bay in Manitoba and Ontario, around the WH and SH boundary, during the ice-free period. They attributed this to ice-dependent variation in on-shore arrival locations suggesting that in years when bear counts on the Manitoba (or WH) side of the boundary were high counts on the Ontario (or SH) side tended to be low and vice-versa. A series of subsequent studies utilizing mark-recapture, telemetry and coastal survey data did not find evidence to support this hypothesis instead finding that WH bears exhibited a high degree of fidelity to on-shore areas during the ice-free period (Derocher and Stirling, 1990; Kolenosky et al.,1992; Lunn et al., 1997; Stirling et al., 1999; Stirling et al. 2004). More recently, however, analyses of telemetry data for adult females in WH have found that the timing of bears' movements to shore and the locations where they arrive on-shore are primarily influenced by environmental variables including wind direction, ice concentration and ice distribution during break-up (Cherry et al. 2013, 2016; Pilfold et al. 2017; Bohart et al. 2020). In particular, Cherry et al. (2013) found that on average approximately 12% of WH adult females came ashore outside the boundaries of WH, typically further south and east, and within the boundaries of the SH subpopulation. Additionally, WH bears were more likely to come ashore outside WH in years when there was more remnant summer-time ice in SH relative to WH (Cherry et al. 2013). Based on these findings, a degree of year-to-year variation in aerial survey-based estimates of WH should be expected as a result of ice-dependent variation in the locations bear come ashore. In years when ice remains longer in SH relative to WH, a higher proportion of WH bears may come ashore within the bounds of SH where they would be included in a SH rather than WH estimate.

Several lines of evidence support the hypothesis that the observed decline in WH abundance between 2011 and 2021 was to some degree the result of interannual variation in the distribution of bears between WH and SH. Aerial surveys of the SH subpopulation have been conducted concurrently with the surveys in WH in 2011, 2016 and 2021. Pooling estimates from the WH and SH surveys show a decline in total 'Hudson Bay' abundance from 2011 to 2016 but no change from 2016 to 2021 (Figure 22). Notably, between 2016 and 2021, the estimated abundance of SH increased by 223 bears while that of WH decreased by 224 (Northrup et al. 2022). Changes in both subpopulations, at least between 2016 and 2021, could therefore be accounted for by movement of WH bears into SH. Preliminary results from genetic mark-recapture work conducted along the coast of WH and SH provide additional evidence to support this hypothesis. In 2021, biopsy darting conducted as part of a genetic mark-recapture program found that 22% of bears sampled in SH had been previously sampled in WH only (McGeachy et al. 2022). In contrast, sampling within WH did not detect any recaptures of bears previously sampled in SH only.

Although they provide evidence of a potential distribution shift, comparison of abundance estimates for WH and SH, as well as preliminary findings of the genetic mark-recapture program should be interpreted with caution for several reasons. First, recent physical and genetic mark-recapture sampling effort has been markedly greater in WH relative to SH. For example, for genetic mark-recapture alone, more than 350 bears were marked within WH between 2017 and 2020, while none were marked in SH. This disparity in marking effort would have increased the likelihood of recapturing bears, originally marked in WH, within SH in 2021 and reduced the likelihood of SH marks being recaptured in WH. Second, the vast majority of bears marked in WH and recaptured in SH in 2021 were adult males yet the observed decline in WH appears to be due to declining numbers of subadults and females rather than adult males. Third, adult male bears are likely to exhibit greater flexibility in where they come ashore to spend the summer. Unlike adult females they do not require access to suitable denning habitat or in-land areas in which to avoid infanticide of dependent off-spring by conspecifics (Derocher and Striling 1990; Stirling et al. 2004). Consequently, trends in abundance resulting from sea-ice related shifts in summer range are more likely to be observed in WH adult males. Our findings that adult female and subadult abundance has declined while adult male abundance has remained unchanged are thus inconsistent with a range-shift hypothesis. Fourth, as noted above, WH bears exhibit a greater likelihood of coming ashore in SH in years when there is more remnant summer-time ice in SH relative to WH (Cherry et al. 2013). If an ice-dependent shift in summertime range were responsible for the observed decline in WH, sea-ice data for 2011, 2016 and 2021 should show that greater amounts of late break-up sea-ice were present in SH, relative to WH, in 2021 versus 2011 or 2016. However, the data suggest sea-ice that remnant sea-ice conditions in July, for example, were very similar in 2021 and 2011 (Figure 23). Finally, if trends in WH abundance were due to a distributional shift with bears moving out of WH, this shift would likely also be apparent within the boundaries of the subpopulation itself. We examined trends in abundance within 3 zones of WH and found declining abundance within the central or core zone of the WH summer range (zone 2). However, similar trends were not apparent in the areas of WH bordering the neighbouring SH and FB subpopulations. Again, these findings do not support a distribution shift hypothesis to explain the changes in WH abundance.

Harvest mortality has also likely contributed to the observed changes in WH abundance. Dependent on sea-ice conditions, the most recent demographic assessment of WH projected a long-term population growth rate ranging from 0.97 to 1.02 from 2011 onwards. Between 2011-2021, annual harvest in WH increased in accordance with the regulated Total Allowable Harvest (GN unpublished data; Figure 24) from a rate of approximately 2.5% of estimated abundance in 2011/12 up to 6.6% in 2021[2]. Since the rate of harvest has exceeded projected population growth, a decline in total abundance between 2011 and 2021 would be expected due to harvest pressure. However, the apparent decline in adult females and subadults but lack of trend in adult males is harder to rectify with a harvest-induced effect. Harvest in WH has been sex-selective with an annual average of 66% males between 2011 and 2021. Subadults have compromised approximately 26% of annual harvest [3], a level comparable with other subpopulations (GN unpublished data). Lunn et al. (2016) found that probability of harvest (*H*) was highest amongst young adult males (5-9 yrs) and lowest amongst adult females at 0.73 and 0.05, respectively. For subadults, H was 0.44 and 0.28 for males and females, respectively. Given the higher harvest pressure on adult males compared to other classes, a harvest-induced change in subpopulation composition would be expected to appear first amongst adult males rather than subadults and adult females, a pattern not consistent with our results.

5.5. Reproductive Performance

As discussed earlier in this report, aerial surveys in 2011 and 2016 suggested that reproductive performance in WH was poor. Relative to neighbouring subpopulations in the seasonal ice ecoregion, the number of yearlings, expressed as a proportion of all individuals observed, was low suggesting poor recruitment into the subadult age classes (Table 14). Similarly, mark-recapture sampling yielded relatively low numbers of yearlings in 6 of 10 years from 2011 to 2020 (ECCC unpublished data). These findings

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[2] Based on 2011 and 2021 aerial survey abundance estimates.

[3] Data available for the period 2010-2019 only.

suggest that over the previous decade WH has experienced a series of years with low recruitment into the subadult age class.

Results from the 2021 survey indicate that yearling numbers were higher and more comparable with levels seen in other subpopulations (Table 14). A similar improvement in yearling numbers was also seen in SH in 2021, suggesting conditions in Hudson Bay over the last few years have been generally favorable for raising offspring (Northrup et al. 2022). Whether this improvement in reproductive performance will continue is unknown. Regehr et al. (2015) estimated conditions necessary for polar bear population persistence which included recruitment levels of 0.1-0.3 yearling per adult female and adult female survival rates between 0.93-0.96. Raw aerial survey observation data indicate that recruitment was approximately 0.14, 0.15 and 0.31 in 2011, 2016 and 2021, respectively suggesting that reproductive performance although variable has likely been sufficient. Of concern with respect to future reproductive capacity in WH, is the apparent decline in subadult abundance. A reduced subadult cohort will eventually result in cascading negative effects on reproduction as these individuals enter the adult age classes and current adult bears begin to either enter reproductive senescence or die (Regehr et al. 2021).

5.6. Mortalities

What appears to be an unusually high number of polar bear carcasses were found in WH during 2021. Three were located during the aerial survey and 1 during other polar bear research activities. Based on flying times for both projects, this equates to approximately 3 carcasses per 100 hours of search effort. By comparison, while flying more than 1200 hours conducting aerial sampling of polar bears during the ice-free period in 4 other subpopulations in the seasonal ice ecoregion, over the last decade, 3 polar bear carcasses (0.25 per 100 hours) were found (S. Atkinson pers. obs.).

Cause and timing of mortality was unknown in all cases, although starvation was likely involved in one case. Interestingly, three of the 4 were adults which is somewhat unexpected since adult survival rates are higher than other age classes. The observation of other bears feeding on the carcasses is consistent with previous reports of cannibalism amongst polar bears (Lunn and Stenhouse 1985; Taylor et al. 1985; Amstrup et al. 2006; Dyck and Daley 2002; Derocher and Wiig 1999; Stone and Derocher 2007). Amongst bear species, polar bears display the highest reported levels of cannibalism (Allen et al. 2022). Amstrup et al. (2006) suggested incidences of predation and cannibalism amongst polar bears may reflect increased nutritional stress within a subpopulation. However, for the cannibalism observed in WH in 2021, it is unknown whether the mortalities were the result of conspecific predation or opportunistic scavenging on available carcasses.

Given the high number of carcasses found in 2021 and the fact that at least 3 were adults, careful documentation of future carcass observations is recommended to assess whether deeper investigation is warranted.

6. CONCLUSIONS

Ultimately, it is important to stress that we are unable to definitively conclude whether the finding of declining abundance in WH over the last decade, specifically that of adult females and subadults, is the result of reduced survival and recruitment, movement of bears into neighbouring subpopulations (emigration), or harvest pressure. Based upon the multiple lines of evidence reviewed in this report, it is plausible that all these factors have contributed to some degree. Of particular concern, however, is our finding that the observed declines in subadults and adult females are consistent with long-standing predictions regarding the order in which negative demographic effects will emerge amongst the different sex and age classes of polar bears due to climate related environmental change. If these apparent trends continue, the progression of a reduced subadult cohort into the adult age class, combined with an already reduced adult female class, reproductive senescence, and mortality amongst older bears, may result in cascading effects on WH abundance and reproductive performance over the next decade. We therefore recommend follow-up on these findings in several forms as follows:

1. Analyses of on-going physical and genetic mark-recapture programs are best suited to address the uncertainties arising from our aerial survey findings and are a strongly recommended next step in the assessment of WH. An analysis of physical and genetic mark-recapture data for 2011 to 2021 is needed to determine whether demographic effects such as reduced recruitment and survival of subadults have occurred in WH.

2. Management agencies should consider increasing monitoring efforts in WH by changing the frequency of future aerial surveys from every 5 years to every 3 years, in the near term. The purpose of this increased frequency would be to determine whether the subpopulation is entering a period rapid change in abundance requiring more frequent adjustments in harvest management strategy than at present.

3. Questions remain about the interannual movements of bears between WH and SH and the effect these movements have on aerial survey abundance estimates. It is also unclear whether the significant number of bears, mostly adult males, that occupy the southeastern coast of WH during the ice-free season are available for harvest by WH communities or whether these bears function as either an unharvested segment of the subpopulation or are in fact harvested in SH. Results from current genetic sampling and telemetry studies in this region are expected to address these questions and inform WH management. Priority should be placed on completing these studies.

4. A harvest risk assessment should be conducted to determine the impact of recent and future harvest under current conditions.

5. The high number of polar bear carcasses found during the WH survey is concerning. We recommend closer monitoring and reporting of polar bear carcasses found in WH during future aerial surveys and on-going mark-recapture studies.

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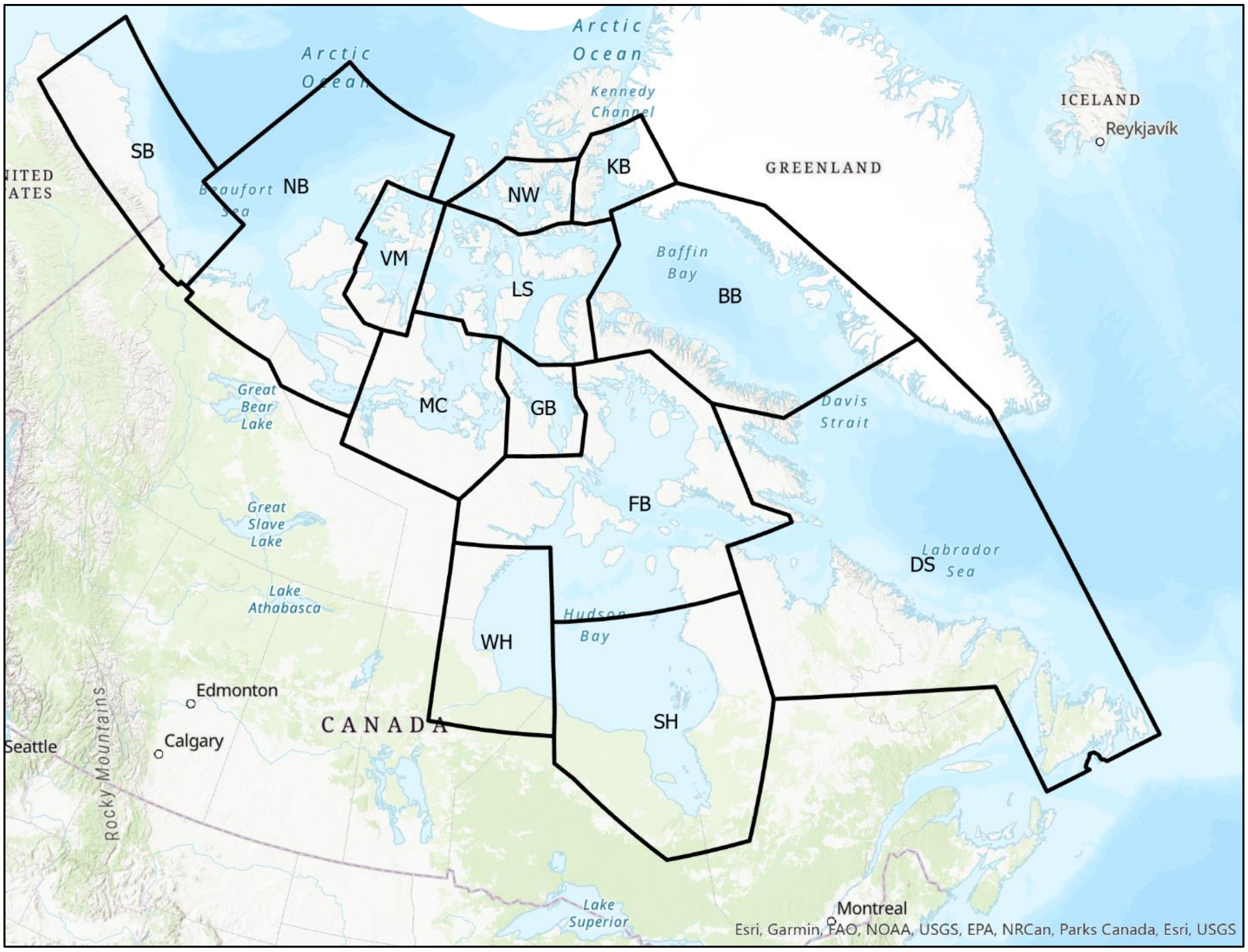

Figure 1. Boundaries of polar bear subpopulations that are partially or totally under management from Canadian jurisdictions. These include Southern Beaufort Sea (SB), Northern Beaufort Sea (NB), Viscount Melville Sound (VM), McClintock Channel (MC), Lancaster Sound (LS), Norwegian Bay (NW), Kane Basin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

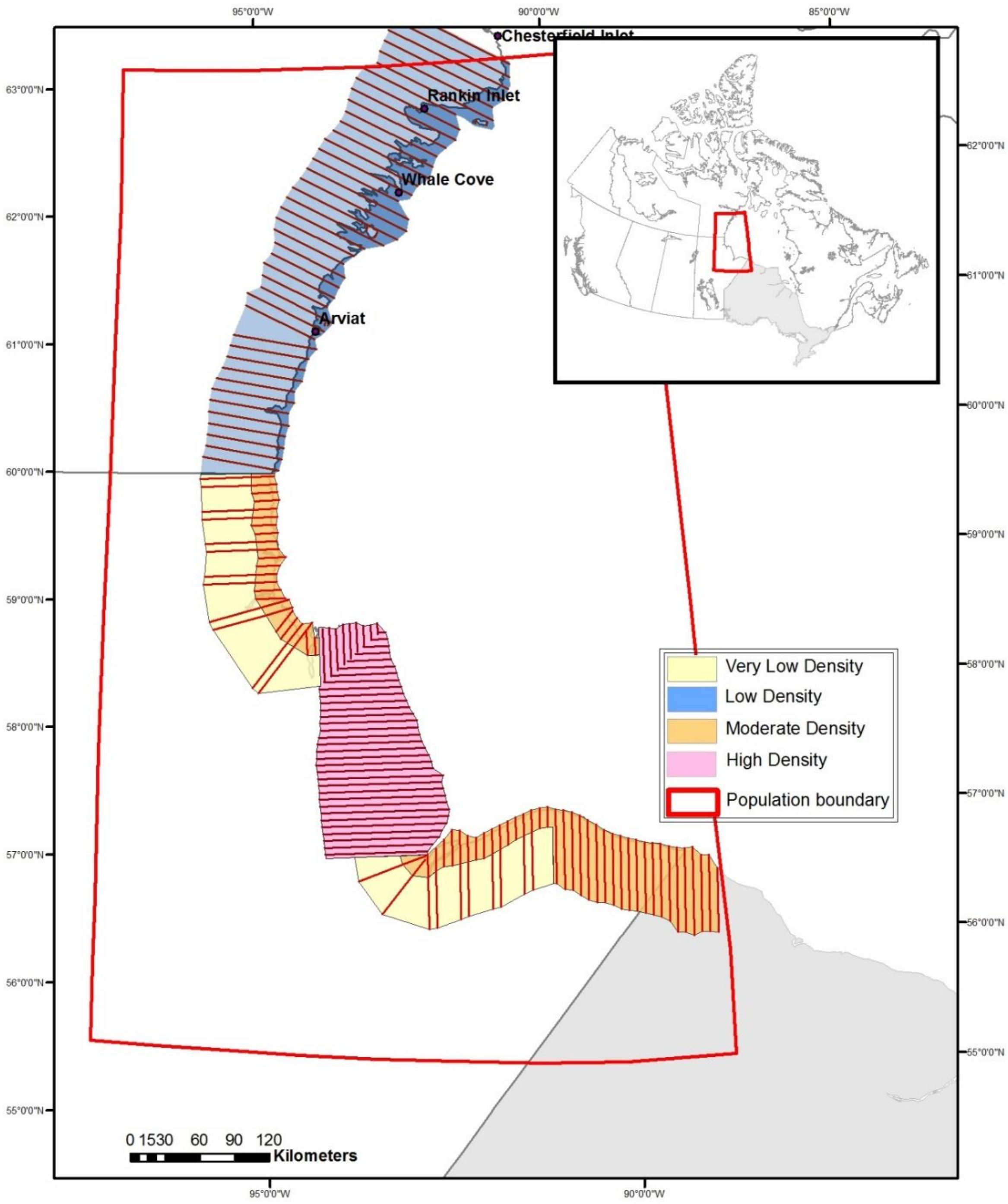

Figure 2. Survey strata and transects for the 2021 aerial survey of the Western Hudson Bay polar bear subpopulation.

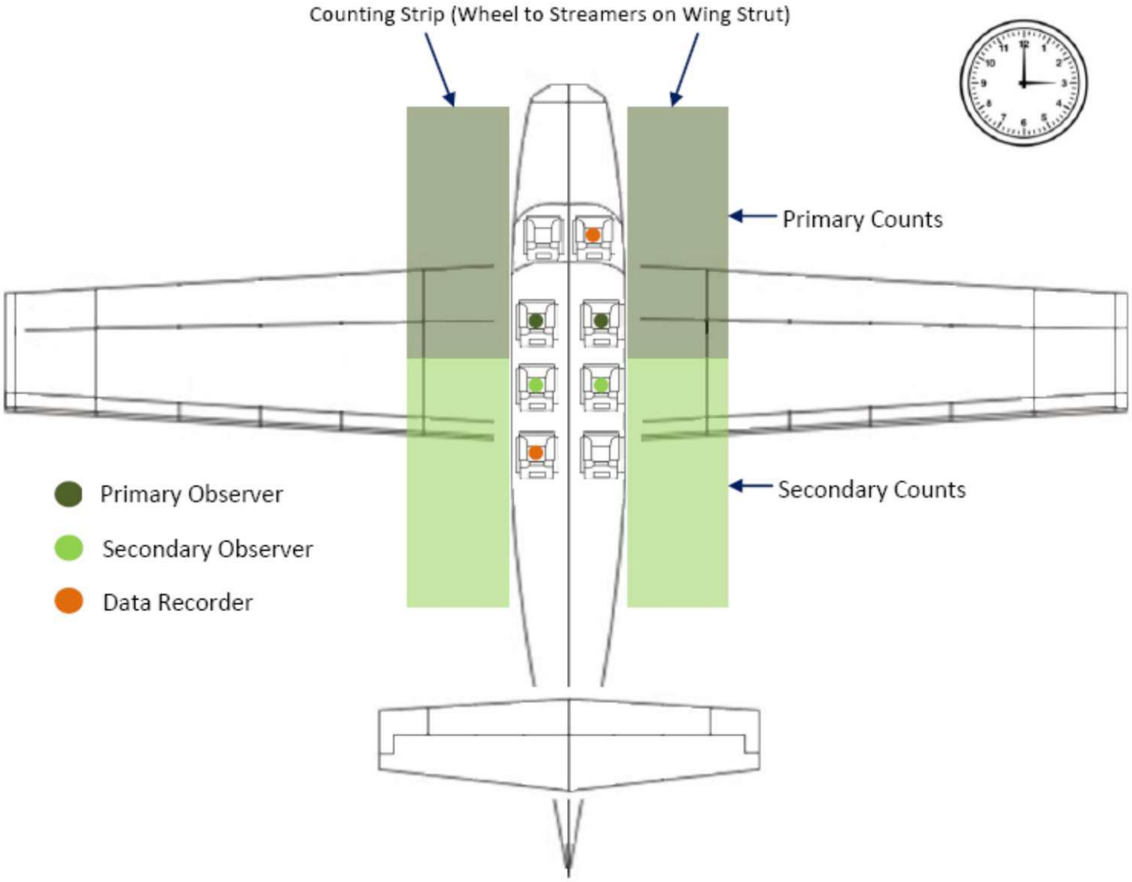

Figure 3. Observer position for the double observer method employed on this survey. The secondary observer calls polar bears not seen by the primary observer after the polar bear/bears have passed the main field of vision of the primary observer at a point halfway between same side primary and secondary observers. The small hand on a clock is used to reference relative locations of polar bear groups (e.g. “Polar bear group at 3 o’clock” would suggest a polar bear group 90o to the right of the aircrafts longitudinal axis).

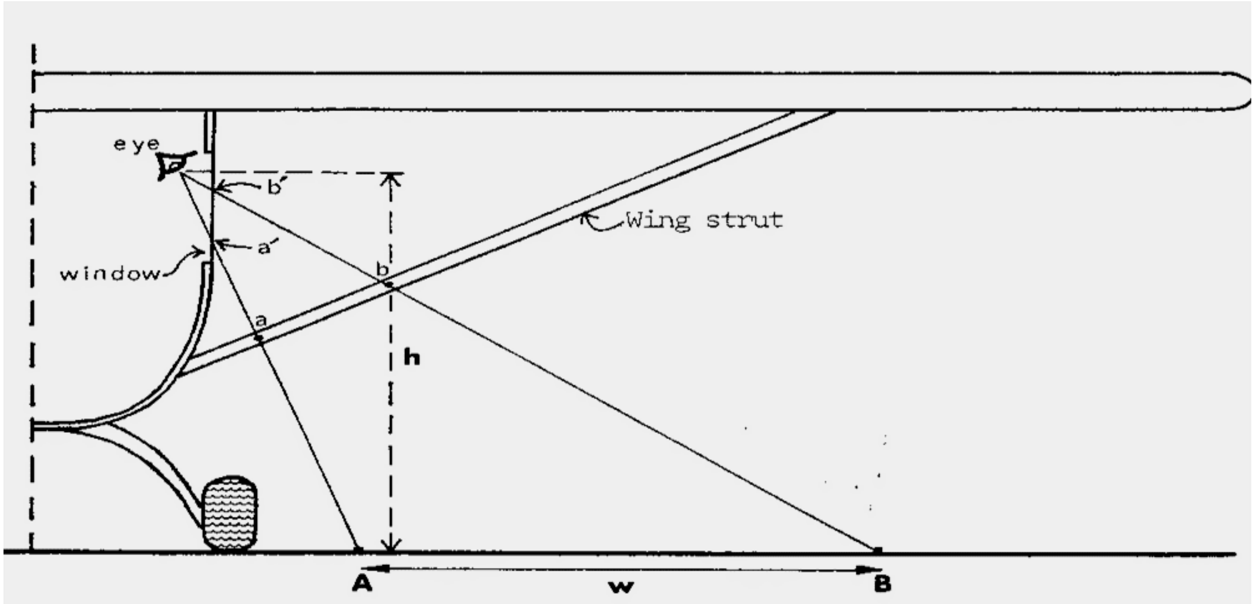

Figure 4. Schematic diagram of aircraft configuration for strip width sampling (Norton-Griffiths, 1978). W is marked out on the tarmac, and the two lines of sight a’ – a – A and b’ – b – B established. The streamers are attached to the struts at a and b, whereas a’ and b’ are the window marks. (After Jolly 1969)

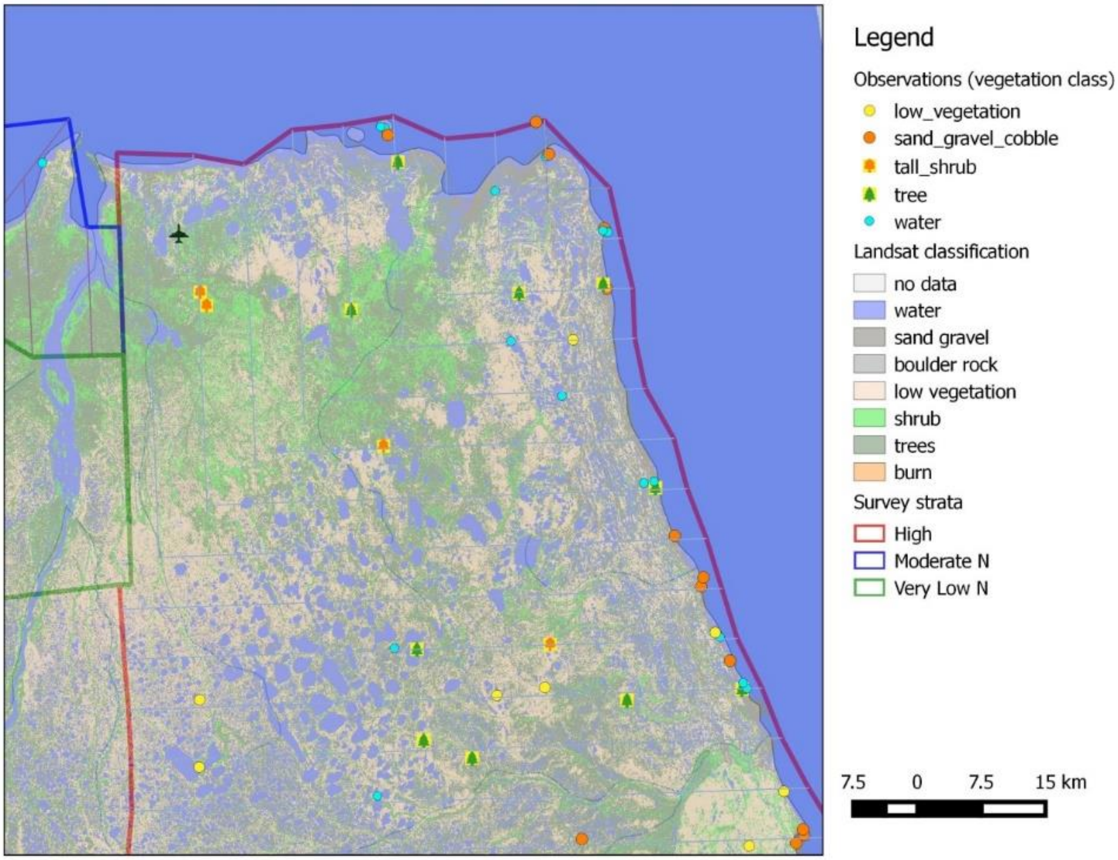

Figure 5. Landsat habitat classification and observations for a section of the 2016 high stratum (Dyck et al. 2017)

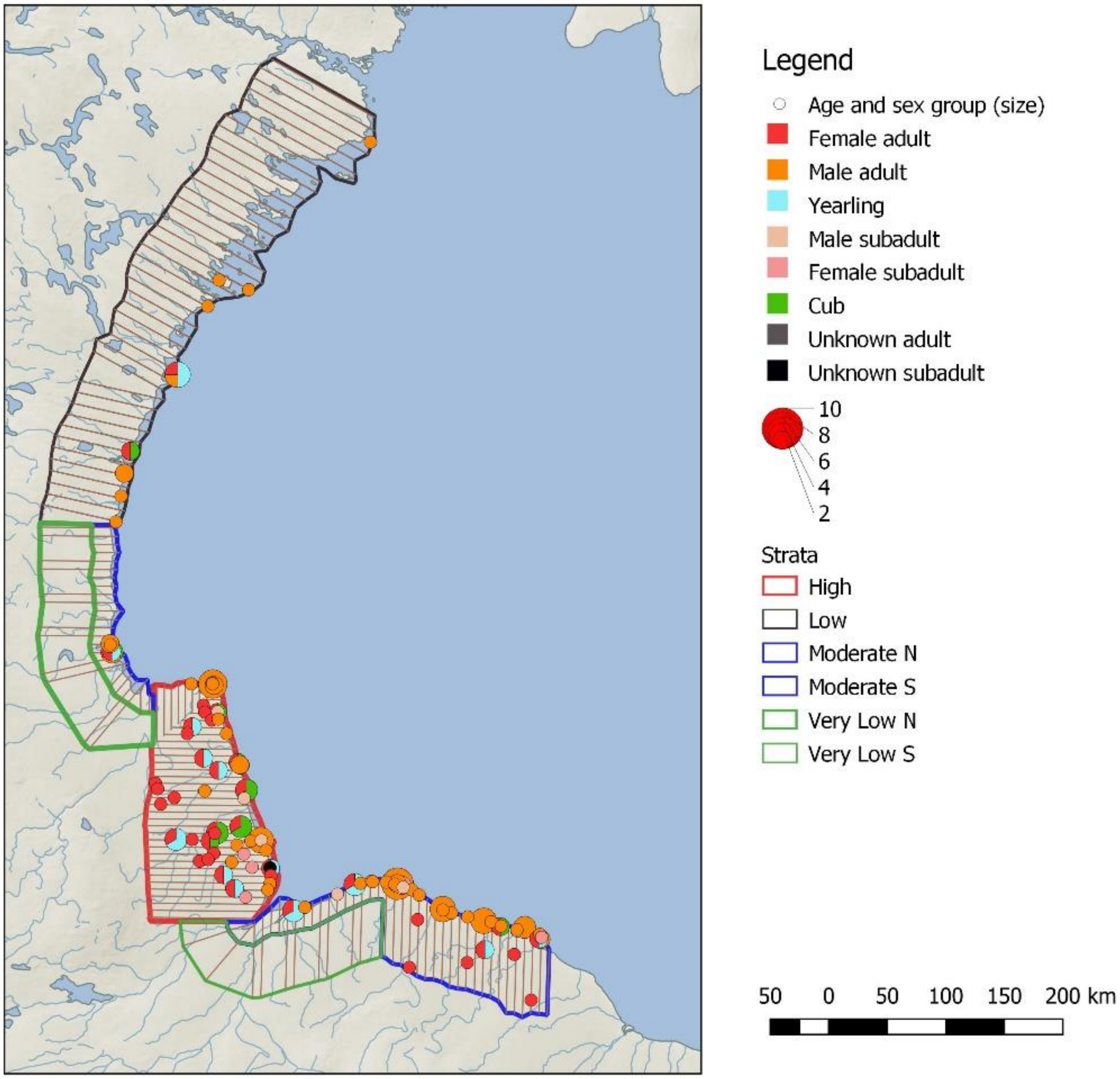

Figure 6: Distributions of bears observed on transect during the 2021 WH aerial survey; with group composition and size noted.

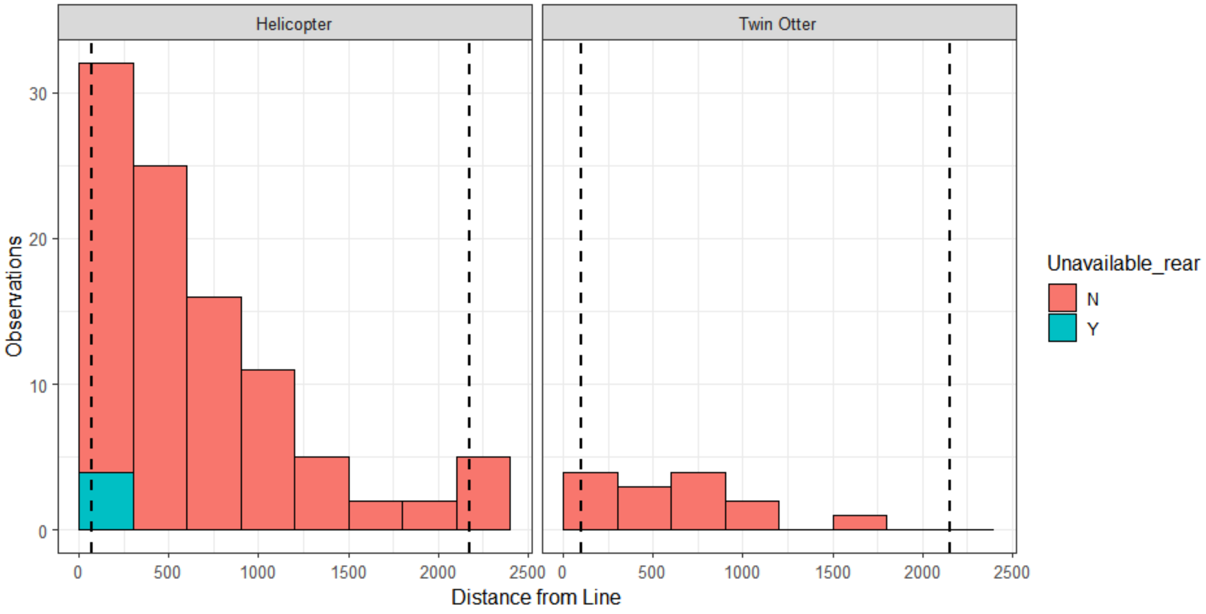

Figure 7: Detection histogram of full data set before right and left truncation by aircraft type. Left and right truncation distances are shown as hashed vertical line

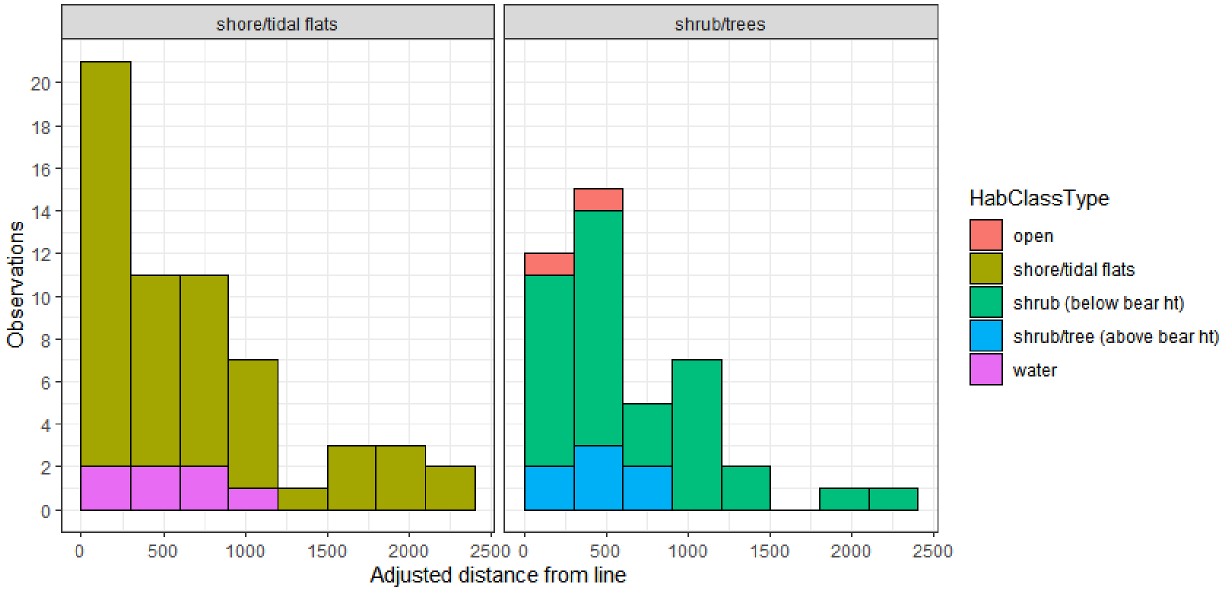

Figure 8: Detection histograms for pooled habitat class (HabClassP) categories with original categories shown as sub-bars. Adjusted distance from line (left truncation distance subtracted) is displayed on the x-axis.

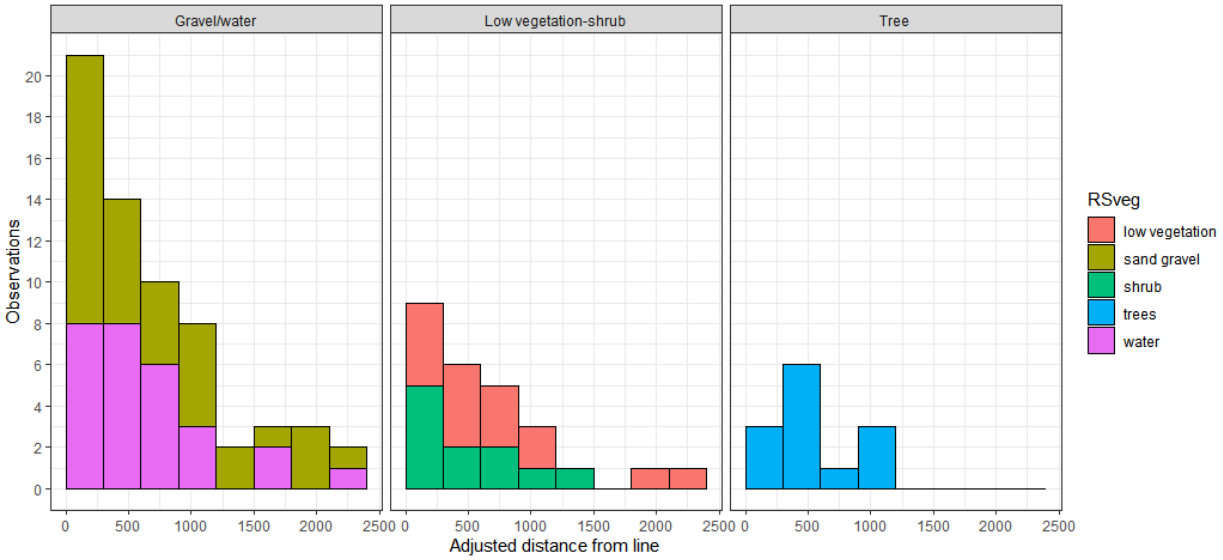

Figure 9: Detection histograms for pooled RSveg habitat categories with original categories shown as sub-bars. Adjusted distance from line (left truncation distance subtracted) is displayed on the x-axis.

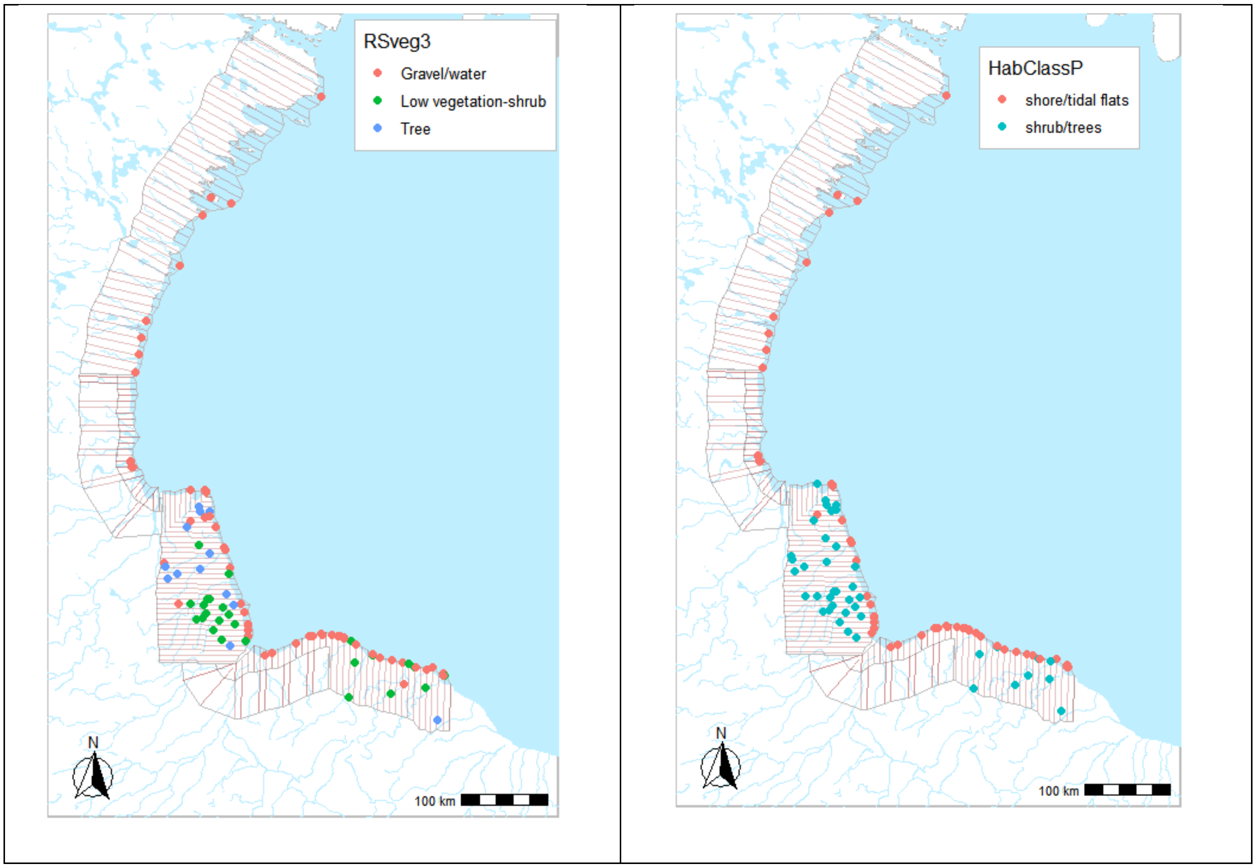

Figure 10: Distribution of observations by RSVeg and HabClass classifications.

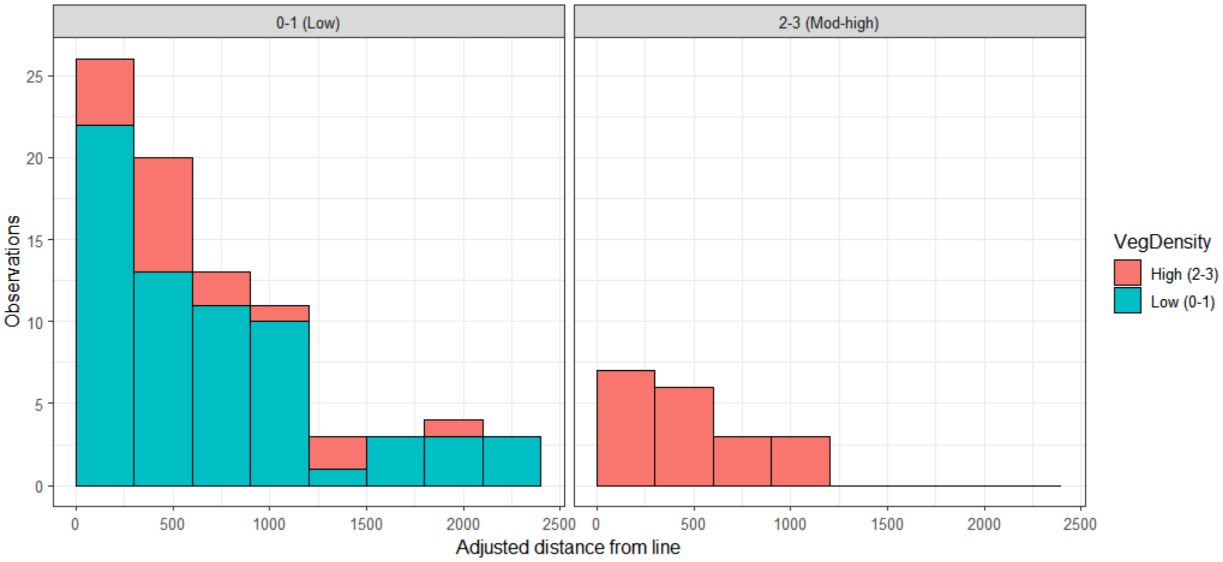

Figure 11: Detection of vegetation height and density (sub-bars).

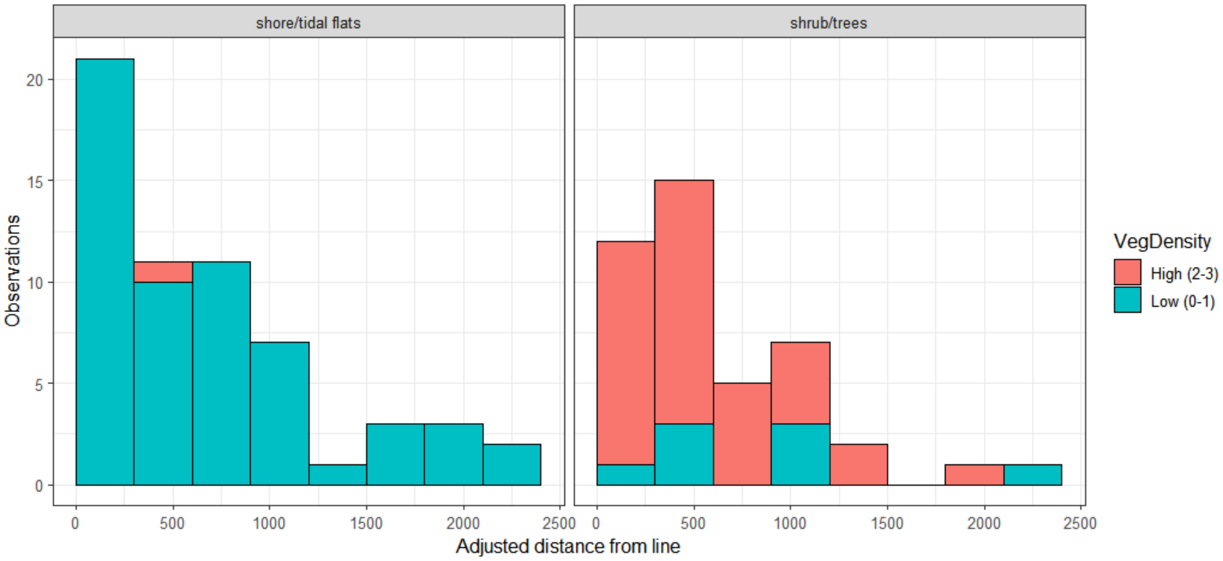

Figure 12: Detection of vegetation height and density (sub-bars).

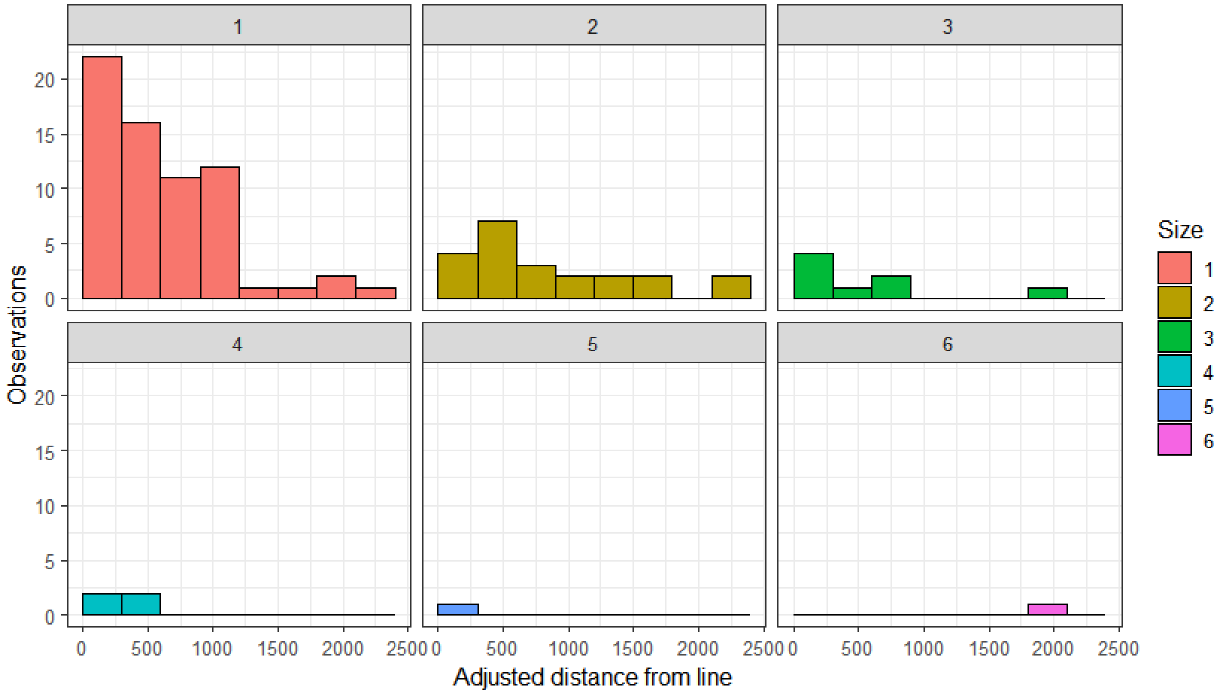

Figure 13: Detection histograms as a function of group size observed.

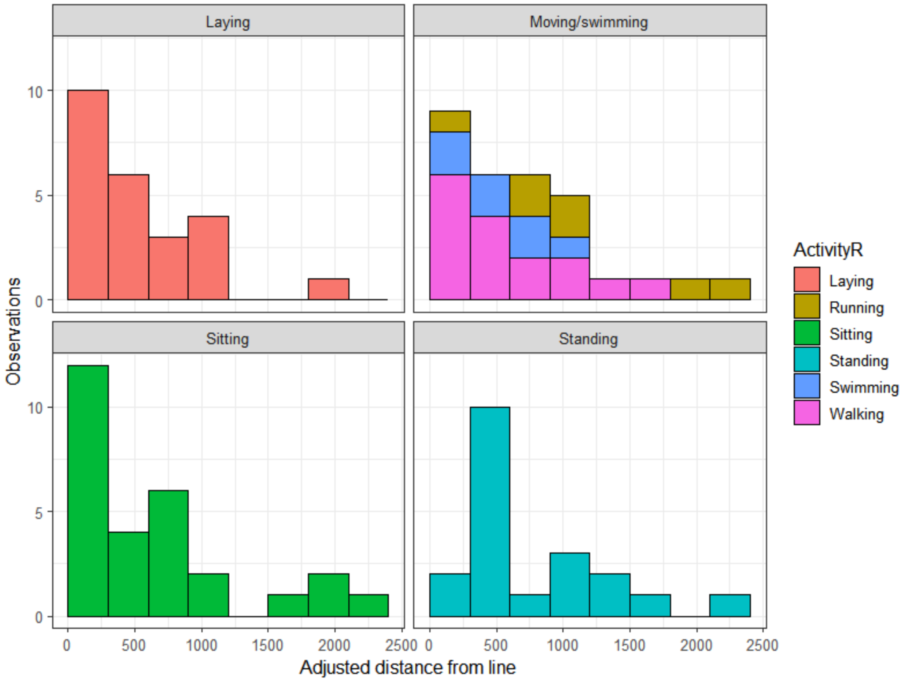

Figure 14: Detection histograms by activity type.

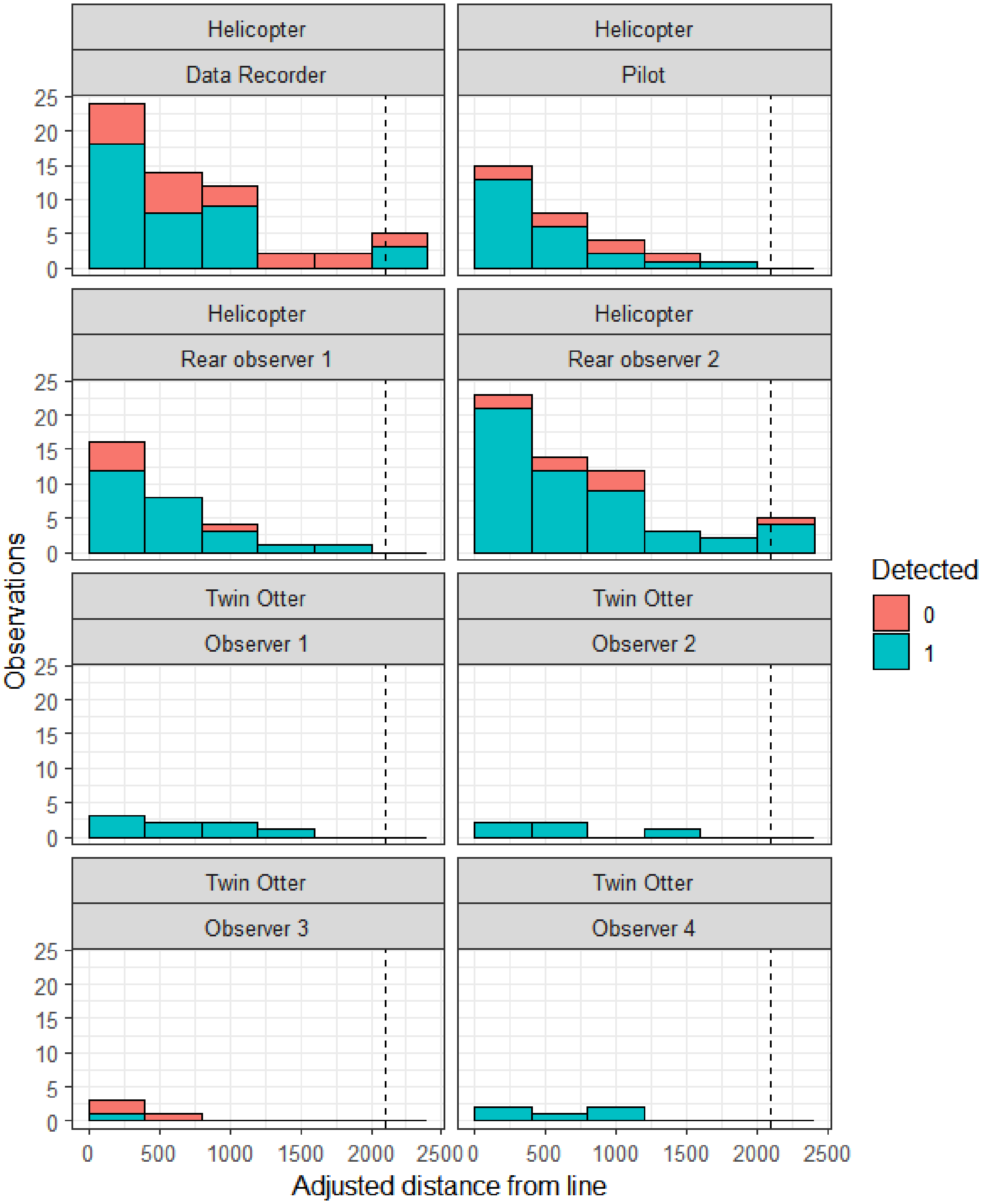

Figure 15: Observer detection histograms for the observers with detections noted (0-not detected, 1-detected)

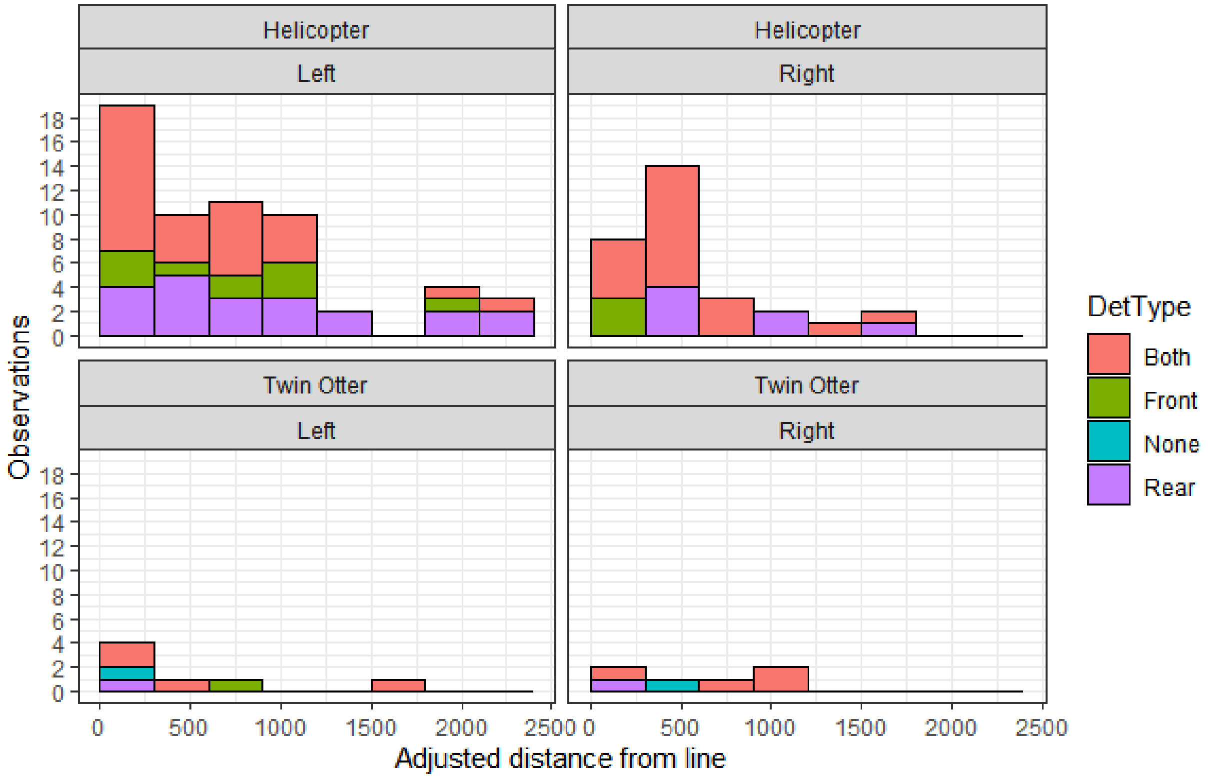

Figure 16: Observations by side of aircraft with detection type noted.

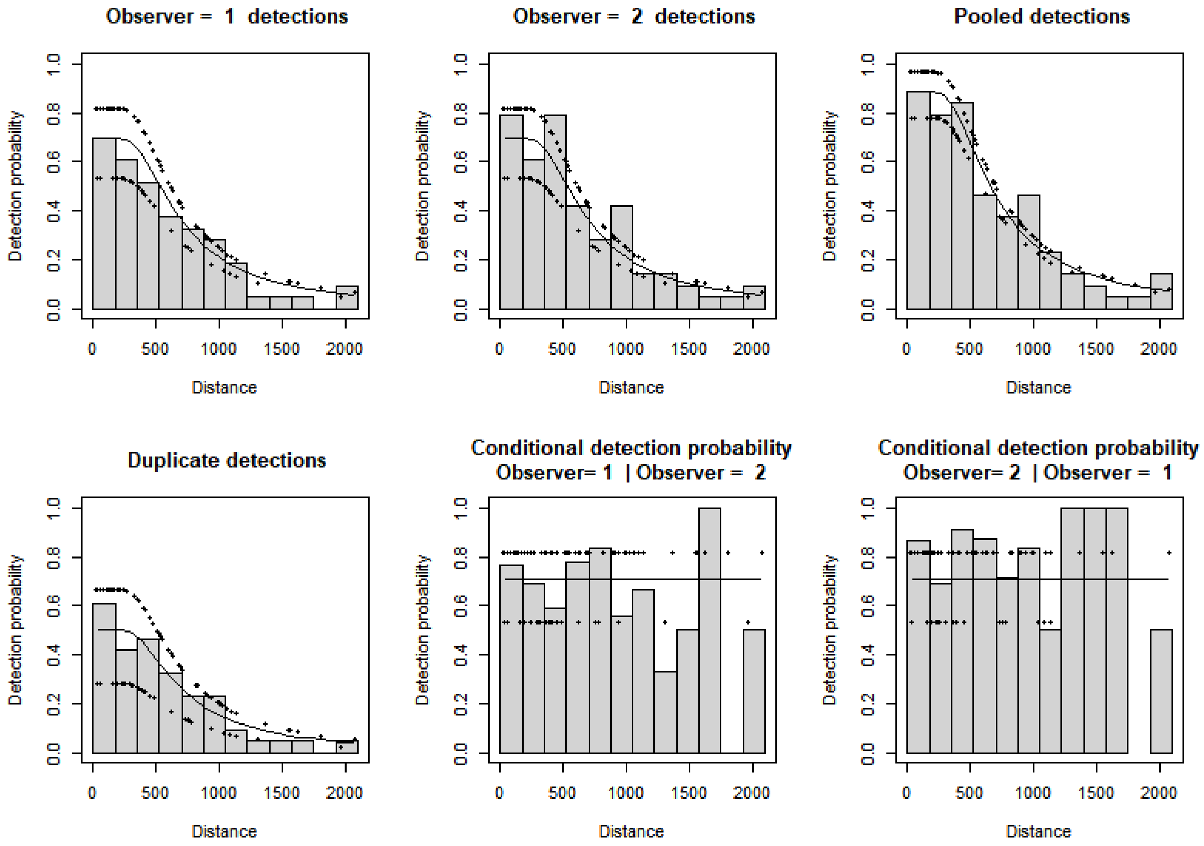

Figure 17: Fit of the most supported distance sampling/double observer model (DS HR(.), MR(VegDensity)) by observers (front and rear). Predictions are given as points for 2 levels of VegDensity.

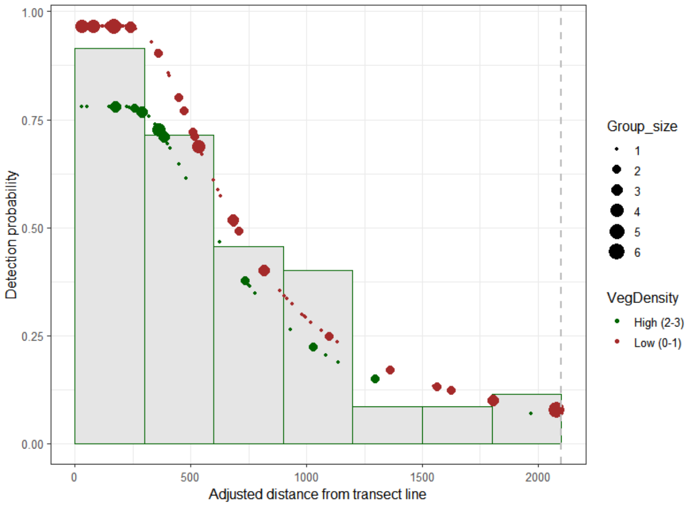

Figure 182: Fit of the most supported distance sampling/double observer model (DS HR(.), MR(VegDensity)) with group sizes and Veg Density delineated.

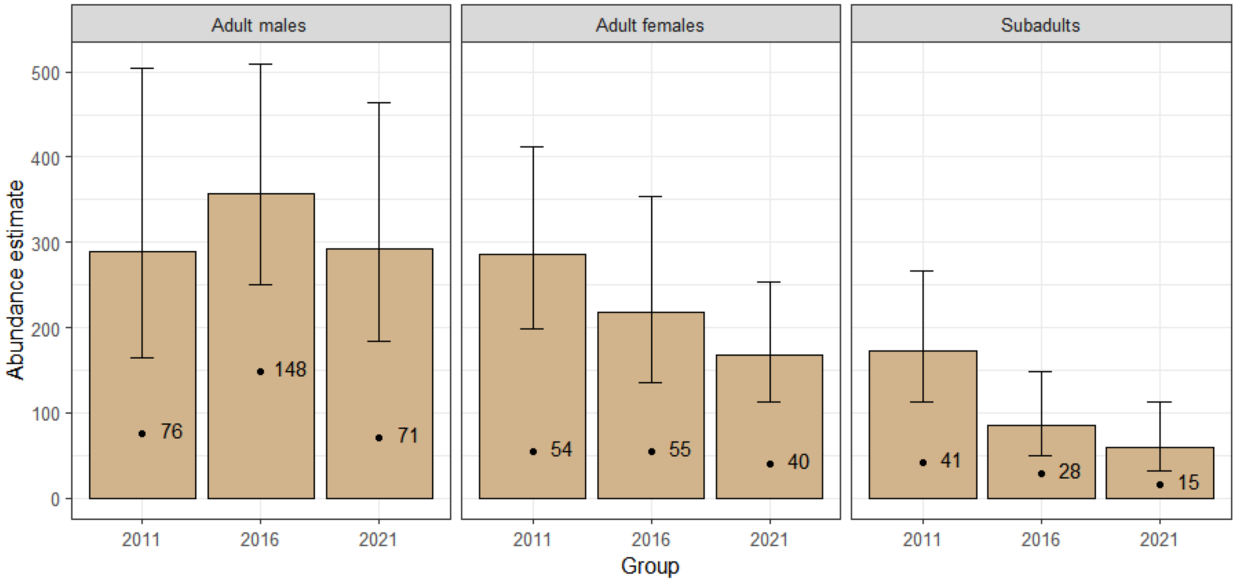

Figure 3: Post-stratified estimates of each age and sex group for 2011, 2016 and 2021. The number of bears observed on transect is also given as a data point.

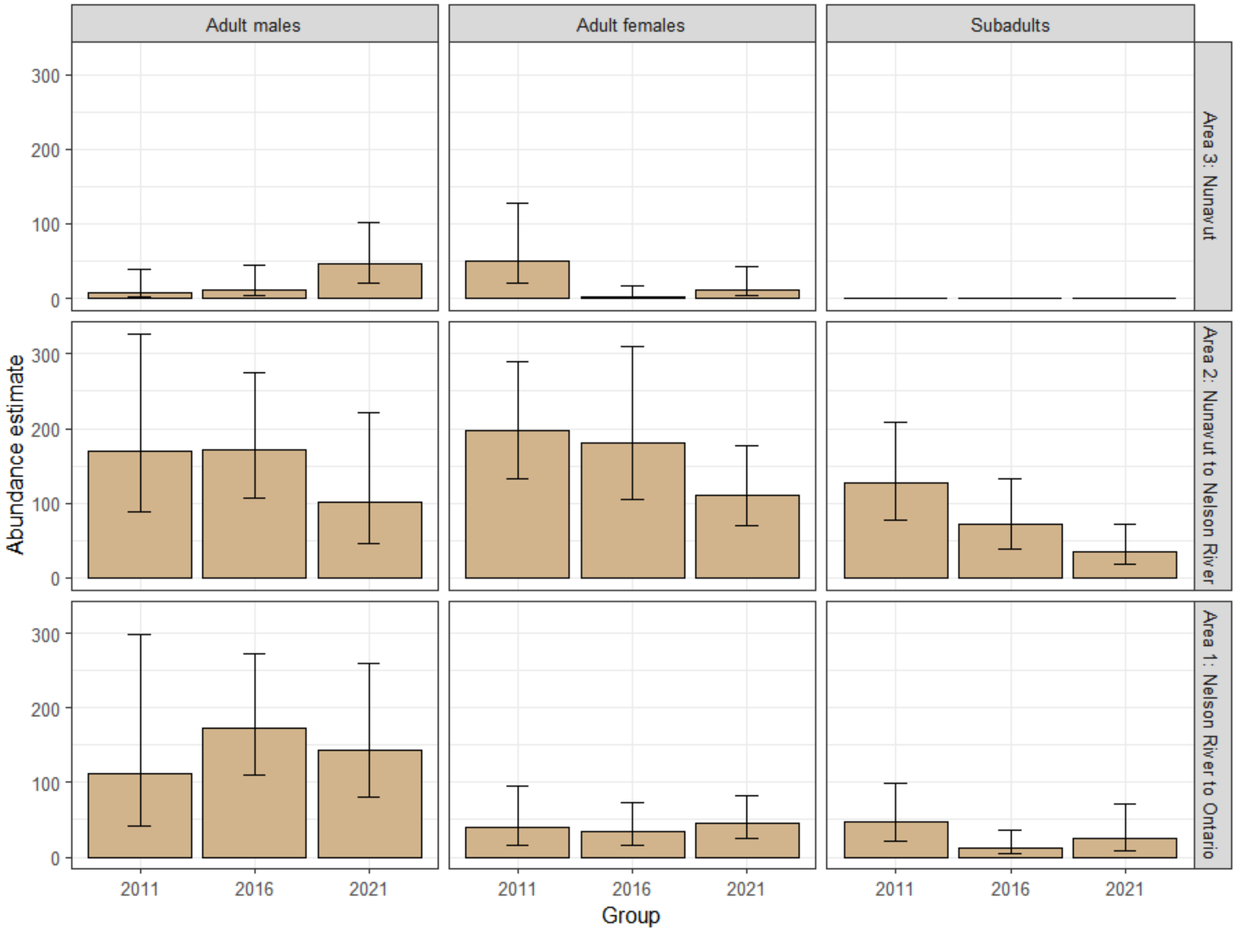

Figure 4: Estimate of abundance by geographic region for adult males, adult female, and subadults.

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Figure 21. Two of three polar bear carcasses encountered during the 2021 WH survey.
(Photo credit: S. Atkinson)

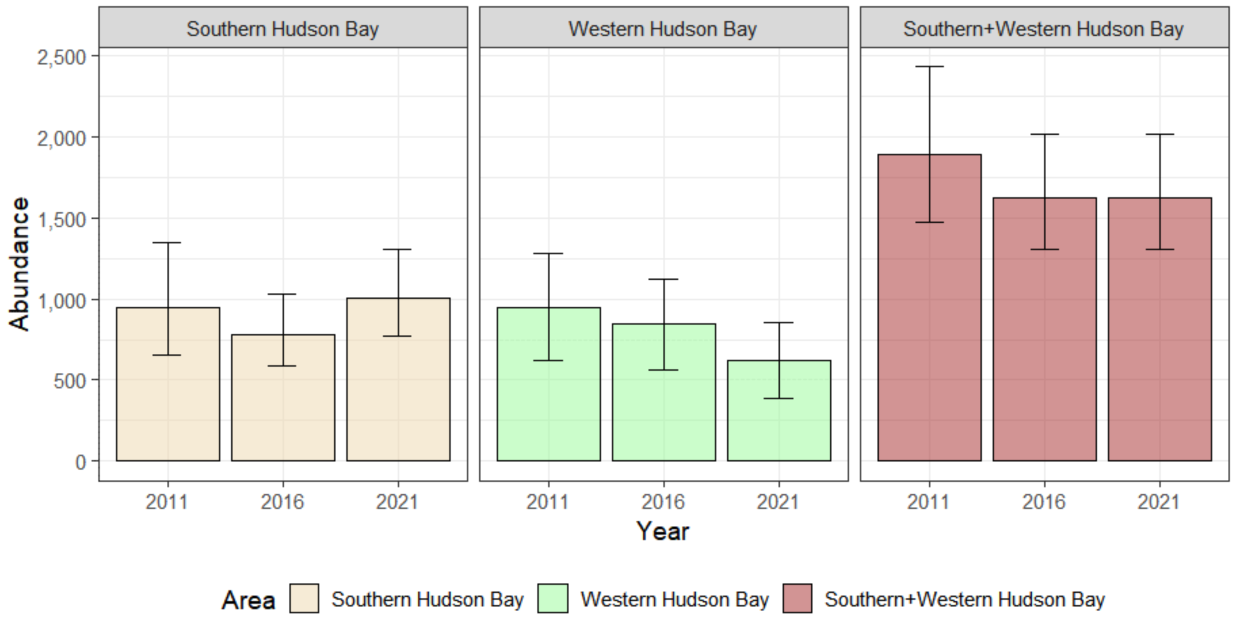

Figure 22. Individual and pooled aerial survey abundance estimates for the Western Hudson Bay (WH) and Southern Hudson Bay (SH), polar bear subpopulations.

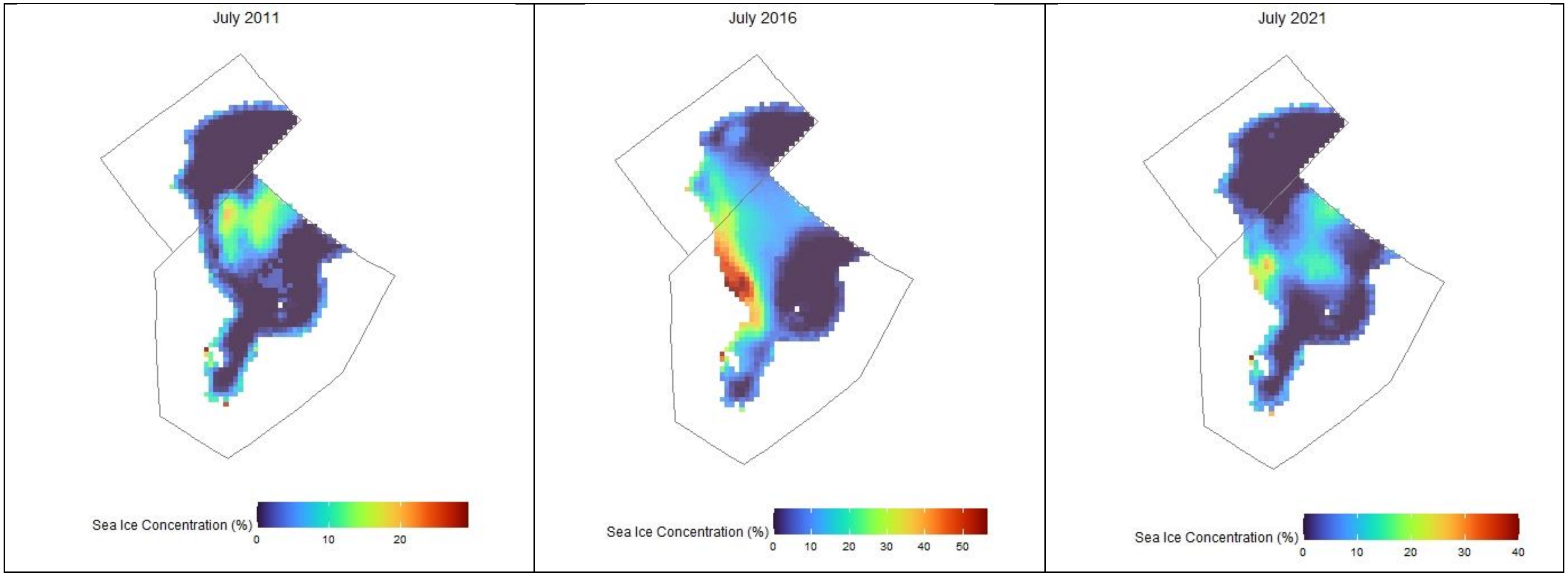

Figure 23. Mean daily sea ice concentration in WH and SH for the month of July from 2011-2021. We used sea ice concentration data from the Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave data set available from the National Snow and Ice Data Center (NSIDC). The monthly average was calculated for each 25 x 25 km grid cell over the eleven-year period.

Table 1. Description of survey strata used in the 2016 and 2021 WH polar bear aerial surveys. (Source: Dyck et al. 2017).

Strata Name	Description
Very Low Density	These strata and transects represented the inland portions of the survey area outside of the Wapusk National Park high density stratum boundaries (Figure 2). These strata were divided further into two main areas, one north and west of the Churchill River up to the Nunavut/Manitoba boundary in the north, and the second south and east of the Nelson River bounded to the east by Cape Tatnam. The very low-density strata covered only inland transects generally ending within 20 to 30 km of the Hudson Bay coastline. Transect spacing was irregular but averaged 17 km across the strata.
Low Density	The stratum and transects occupied the northern extents of the WH polar bear population boundary (approximately 20 km south of Chesterfield Inlet) to the Nunavut/Manitoba border (Figure 2). Modifications from Stapleton et al. (2014) included IQ-based transect extensions both over water and inland within the northern extent of this stratum. Overwater extensions within the remaining extents including 2 transects bi-secting Sentry Island were derived solely from Inuit Qaujimajatuqangit (IQ) reports and recommendations. Transect lines in this stratum were spaced 10 km apart, and extended up to 90 km inland, and up to 30 km into Hudson Bay beyond the coast to incorporate the many offshore islands characterizing this coastline. The development of this stratum was largely based on local knowledge which strongly recommended the extension of coastal transects inland and across open water and coastal islands.
Moderate Density	These strata and transects were divided into two areas, one north and west of the Churchill River up to the Nunavut/Manitoba boundary in the north, and the second south and east of the Nelson River, approximately 60 km east into Ontario to the eastern extent of the WH polar bear population boundary. These strata primarily covered a Hudson Bay coastal strip that was approximately 20 to 30 km wide. Transect spacing within this stratum was 7 km with transects extended beyond the tidal flats into open water. Recent information collected by the Manitoba Department of Sustainable Development on summer and spring polar bear habitat including denning sites, spring emergence habitat, and coastal summer retreat, led this survey effort to modify Stapleton et al. (2014) survey design to define a moderate-density stratum from Cape Tatnam east toward East Penn Island with transects extending beyond the coastal strip up to 70 km inland into known denning habitat (Figure 2).
High Density	The stratum and transects followed those described by Stapleton et al. (2014). The stratum boundary ran between the Churchill River in the west to the coast of Hudson Bay in the east with Churchill forming the northern boundary and the Nelson River approximating the southern boundary. The core of the high-density stratum included Wapusk National Park which is known to be a high density summering area, and further inland, a heavily used denning area (Lunn et al. 2016). Transects in this stratum extended up to 100 km inland and were spaced 6 km apart. As with all other survey strata, all transects were extended 5-30 km beyond the coast into Hudson Bay which enabled the survey design to include bears either in water or on the extensive tidal flats known to be occupied by bears during summer and fall periods (Dyck, 2001; Clark and Stirling 1997).

Table 2. Covariates collected for each polar bear observation.

Covariate	Description
Activity when sighted	Sitting, lying, running, walking, swimming, other
Vegetation Height	1 = <1m, 2 = 1-3m, 3 = >3m
Vegetation Density	1 = sparse/tundra, 2 = moderate, 3 = dense
Habitat Class	1 = open, 2 = water, 3 =shore/tidal flats, 4 = shrub (below bear head height), 5 = shrub (above bear head height) or trees
Habitat Description	Specify general habitat type (e.g. Coastal plain, tundra, beach, rocky coast, island)
Visibility	1= poor, 2 = fair, 3 = excellent
Cloud Cover	Clear, broken, overcast
Glare	Effect on observer ability to see bears. 1 = no effect, 2 = moderate, 3 = strong.

Table 3: Covariates considered in the mark-recapture/distance sampling analysis. The primary use of the covariate for distance sampling analysis (DS) and mark-recapture analysis (MR) is denoted.

Covariate	Type	DS	MR	description
size	continuous	x	x	group size
aircraft	binary	x	x	helicopter or airplane
helip	binary	x	x	Pilot of helicopter
helir	binary	x	x	Recorder/Navigator of helicopter.
hab	categorical	x	x	habitat within 30m of observation as classified by observers (Open, Water, Shore, low shrub, tall shrub, and Tree)
RSveg	categorical	x	x	Landsat habitat (Gravel, Low vegetation, Shrub, Tree, and water) at pixel (625 m^2) scale
Veg_height	continuous	x	x	Relative height of vegetation (0-3)
Veg_density	continuous	x	x	Relative density of vegetation (0-3)
vis	binary	x	x	Visibility based on weather
observer	categorical		x	Observers (12)
side	categorical	x	x	Side of plane
Heliside	Categorical	x	x	Fixed-wing, heli right and heli left categories
glare	continuous	x	x	Sun altitude; only in equation if sun was facing observer
Activity	categorical	x	x	Activity of bear when first observed

Table 4. Summary of bears counted on and off transect during distance sampling for the Western Hudson Bay survey 2021.

Strata	On/off transect	Bears observed	Groups observed	Mean group size	SD group size	min	max
High	On	92	60	1.53	0.87	1	5
Low	On	14	9	1.56	1.01	1	4
Moderate N	On	9	5	1.80	0.84	1	3
Moderate S	On	61	38	1.61	1.14	1	6
Very Low S	On	1	1	1.00	0.00	1	1
total		176	112	1.57	0.97	1	6
High	Off	10	6	1.67	1.63	1	5
Low	Off	3	3	1.00	0.00	1	1
Moderate N	Off	2	2	1.00	0.00	1	1
Moderate S	Off	3	2	1.50	0.71	1	2
Total		18	13	1.38	1.12	1	5
Total (on+off)		194	125	1.55	0.98	1	6

Table 5: Summary of bears included and excluded from the distance analysis based on left truncation (measured blind spot for twin otter=99 m, ASTAR-73.5m on each side of aircraft), right truncated (2100 meters after left truncation subtracted). Also, 2 bears were only observed by data recorders and were not included in the analysis.

Strata	Left Truncated	Included in analysis	Only observed by data recorder	Right truncated
High	13	76	0	3
Low	0	13	1	0
Moderate N	1	7	1	0
Moderate S	1	58	0	2
Total	15	154	2	5

Table 6: Summary of observer detection frequencies and naïve detection probabilities based on frequencies of detections. Note that sample sizes pertain to an event (bears seen on a side of the aircraft) and therefore will be double the actual number of observations (given that 2 observers were involved in each observation).

Aircraft	Observer	Observed	missed	total	naïve p
Heli	pilot	23	7	30	0.77
Heli	Navigator/Data Recorder	38	21	59	0.64
Heli	1	25	5	30	0.83
Heli	2	51	8	59	0.86
Otter	1	7	1	8	0.88
Otter	2	5	1	6	0.83
Otter	3	5	1	6	0.83
Otter	4	1	4	5	0.20
Otter	5	1	0	1	Constant

Table 7: Model selection results for distance sampling analysis. The mark-recapture component of the MRDS model was set at constant for this analysis step. Covariates are listed in Table 1. The detection function (hr=hazard rate, hn=half normal) is shown along with covariates. Constant models are shaded. Akaike information criterion (AIC), the differences between AIC of the given model and most supported model ∆AIC, Akaike weight (w_i), and Log-likelihood of each model is also shown.

No	DF	Detection function model	AICc	∆AICc	w_i	K	LogL
1	hr	constant	1621.73	0.00	0.13	3	-807.7
2	hr	VegDensity	1622.14	0.41	0.10	4	-806.9
4	hr	Helirecorder	1622.72	0.98	0.08	4	-807.1
5	hr	HabClassP	1622.91	1.17	0.07	4	-807.2
6	hr	Side	1623.11	1.38	0.06	4	-807.3
7	hr	VegDensity + Helirecorder	1623.15	1.41	0.06	5	-806.3
8	hn	constant	1623.19	1.45	0.06	2	-809.5
10	hn	size	1623.60	1.86	0.05	3	-808.7
11	hr	VegDensity + Side	1623.68	1.95	0.05	5	-806.5
12	hr	Glare	1623.70	1.96	0.05	4	-807.6
13	hr	size	1623.71	1.97	0.05	4	-807.6
14	hr	Aircraft	1623.90	2.17	0.04	4	-807.7
15	hr	VegDensity + VegHeight	1624.06	2.32	0.04	5	-806.7
16	hr	VegDensity + HabClassP	1624.36	2.63	0.03	5	-806.9
17	hr	HeliSide	1624.60	2.87	0.03	5	-807.0
18	hr	RSveg3	1625.02	3.29	0.02	5	-807.2
19	hr	VegDensity + HabClassP + Helir	1625.38	3.65	0.02	6	-806.2
20	hr	Strata	1626.05	4.32	0.01	6	-806.6
	hr	ObName	1626.51	4.77	0.01	8	-804.5
21	hr	ActivityP	1627.04	5.31	0.01	6	-807.1
	hr	HabClassType	1628.01	6.28	0.01	7	-806.4

Table 8: Model selection results for double observer analysis. The most supported distance sampling model (HR constant) was used for the distance sampling component. Covariates are listed in Table 1. The detection function (hr=hazard rate, hn=half normal) is shown along with covariates. Constant models are shaded. Akaike information criterion (AIC), the differences between AIC of the given model and most supported model ∆AIC, Akaike weight (*w*_i), and Log-likelihood of each model is also shown.

No	Double observer model	AICc	∆AICc	w_i	K	LogL
1	VegDensity	1613.95	0.00	0.30	4	-802.8
2	VegDensity + HeliSide	1615.05	1.10	0.17	6	-801.1
3	VegHeight + VegDensity	1615.81	1.86	0.12	5	-802.6
4	HabClassP	1615.89	1.95	0.11	4	-803.7
5	RSveg3	1616.84	2.89	0.07	5	-803.1
6	VegHeight	1617.82	3.88	0.04	4	-804.7
7	Heli recorder	1617.99	4.04	0.04	4	-804.8
8	Heli pilot	1618.41	4.47	0.03	4	-805.0
9	Position	1619.15	5.20	0.02	4	-805.4
10	Activity	1619.27	5.33	0.02	6	-803.2
11	Helip + Helir	1619.56	5.62	0.02	5	-804.5
12	Side	1620.71	6.76	0.01	4	-806.1
13	ObName	1621.09	7.14	0.01	11	-798.0
14	size	1621.22	7.27	0.01	4	-806.4
15	Constant	1621.73	7.79	0.01	3	-807.7
16	Glare	1621.98	8.03	0.01	4	-806.8
17	HeliRight+HeliLeft+Fixedwing	1622.04	8.09	0.01	5	-805.7
18	HeliRight (only)	1622.27	8.33	0.00	6	-804.7
19	AirPosition	1622.55	8.60	0.00	4	-807.1
20	Aircraft	1623.85	9.91	0.00	7	-804.3
21	Helip + Helir +jb +vt	1624.06	10.11	0.00	6	-805.6
22	Strata	1624.35	10.40	0.00	6	-805.7

Table 9: Model selection results for composite distance sampling and double observer analysis. The most supported distance sampling model (HR constant) was used for the distance sampling component. Covariates are listed in Table 1. The detection function (hr=hazard rate, hn=half normal) is shown along with covariates. Constant models are shaded. Akaike information criterion (AIC), the differences between AIC of the given model and most supported model ∆AIC, Akaike weight (w_i), and Log-likelihood of each model is also shown. Abundance estimates across all strata are given for reference.

Model selection									**Abundance**	
No	**DSDF**	**DS model**	**Double ob model**	**AICc**	**∆AICc**	**w_i**	**K**	**LogL**	**N**	**CV**
1	hr	constant	VegDensity	1613.94	0.00	0.13	4	-802.8	639	19.2%
2	hr	VegDensity	VegDensity	1614.39	0.45	0.10	5	-801.9	608	18.6%
3	hr	Hellir	VegDensity	1614.96	1.02	0.08	5	-802.2	619	19.2%
4	hr	constant	VegDensity + HeliSide	1615.03	1.09	0.07	6	-801.1	648	19.2%
5	hr	HabClassP	VegDensity	1615.15	1.21	0.07	5	-802.3	605	18.5%
6	hn	constant	VegDensity	1615.35	1.41	0.06	3	-804.6	578	15.7%
7	hr	VegDensity + Bellr	VegDensity	1615.44	1.50	0.06	6	-801.3	604	18.8%
8	hr	VegDensity	VegDensity + HeliSide	1615.57	1.64	0.06	7	-800.2	619	18.8%
9	hr	constant	VegDensity + Ob_jb	1615.63	1.69	0.05	5	-802.5	638	19.1%
10	hr	constant	Veg_Height+ VegDensity	1615.79	1.85	0.05	5	-802.6	639	19.1%
11	hr	HeliRight	VegDensity + HeliSide	1615.87	1.93	0.05	7	-800.3	598	18.7%
12	hr	size	VegDensity	1615.95	2.01	0.05	5	-802.7	671	21.8%
13	hr	VegDensity + Side	VegDensity	1615.97	2.03	0.05	6	-801.5	603	18.8%
14	hr	HeliSide	VegDensity + HeliRight	1616.27	2.33	0.04	7	-800.5	599	18.8%
15	hr	VegDensity	HabClass	1616.34	2.40	0.04	5	-802.9	596	18.3%
16	hr	Veg_HeightP+ VegDensity	VegDensity	1616.34	2.41	0.04	6	-801.7	607	18.5%
17	hr	VegDensity+ HeliSide	VegDensity	1617.69	3.75	0.02	7	-801.2	599	18.7%
18	hr	constant	constant	1621.73	7.79	0.00	3	-807.7	622	19.1%
19	hr	VegDensity + HeliSide	HeliSide	1625.92	11.98	0.00	8	-804.2	578	18.1%

Table 10: Strata estimates from the most supported DS/MR model (Table 9, model 1)

Strata	individuals	N	SE	Conf. Limit		CV
High	76	290	75.01	175	482	0.26
Low	13	76	32.77	33	174	0.43
Moderate N	7	28	21.39	7	113	0.76
Moderate S	58	244	57.08	154	387	0.23

Table 11. Post-stratified estimates of sex and age groups from model 1 (Table 9).

Group	Strata	Individuals	N	SE	CIL	CIU	CV
Males	High	25	89.3	39.7	38	210	0.44
Males	Low	8	46.5	18.9	21	102	0.41
Males	Moderate N	3	12.1	8.9	3	47	0.74
Males	Moderate S	35	144.0	43.1	80	259	0.30
Males	Very Low S	0	0.0	0.0	0	0	0.00
Males	Total	71	292.0	69.2	184	463	0.24
Females	High	26	102.7	24.5	64	164	0.24
Females	Low	2	11.6	8.5	3	43	0.73
Females	Moderate N	2	8.0	8.0	1	45	1.00
Females	Moderate S	10	46.1	13.9	26	83	0.30
Females	Very Low S	0	0.0	0.0	0	0	0.00
Females	Total	40	168.5	35.2	112	253	0.21
Cubs	High	9	35.9	15.3	16	82	0.43
Cubs	Low	1	5.8	5.9	1	31	1.01
Cubs	Moderate N	1	4.0	4.0	1	22	1.00
Cubs	Moderate S	1	4.9	4.9	1	27	1.01
Cubs	Very Low S	0	0.0	0.0	0	0	0.00
Cubs	Total	12	50.6	18.2	25	101	0.36
Yearlings	High	7	27.4	11.6	12	62	0.42
Yearlings	Low	2	11.6	12.0	2	64	1.03
Yearlings	Moderate N	1	4.0	4.0	1	22	1.00
Yearlings	Moderate S	6	24.7	13.2	9	68	0.53
Yearlings	Very Low S	0	0.0	0.0	0	0	0.00
Yearlings	Total	16	67.7	22.8	35	129	0.34
Subadults	High	9	35.1	12.7	17	71	0.36
Subadults	Low	0	0.0	0.0	0	0	0.00
Subadults	Moderate N	0	0.0	0.0	0	0	0.00
Subadults	Moderate S	6	24.7	13.7	9	70	0.56
Subadults	Very Low S	0	0.0	0.0	0	0	0.00
Subadults	Total	15	59.8	19.5	32	112	0.33

Table12. Estimates of gross change (GC), probability that gross change is greater than 1 $p(GC<1)$, and annual rate of change (λ) at different survey intervals for distance sampling estimates.

Interval Group	**GC**	**SE**	**Conf. Limit**		**p(GC>1)**	**λ**	**SE**	**Conf. limit**	
<u>2011 to 2016</u>									
Males	1.24	0.46	0.66	2.43	0.759	1.04	0.07	0.92	1.19
Females	0.76	0.25	0.41	1.37	0.177	0.95	0.06	0.84	1.06
Subadults	0.49	0.19	0.24	0.97	**0.021**	0.87	0.06	0.75	0.99
All	0.87	0.22	0.54	1.39	0.282	0.97	0.05	0.89	1.07
<u>2016 to 2021</u>									
Males	0.82	0.26	0.45	1.44	0.237	0.96	0.06	0.85	1.08
Females	0.77	0.27	0.42	1.45	0.214	0.95	0.06	0.84	1.08
Subadults	0.70	0.34	0.30	1.59	0.193	0.93	0.08	0.79	1.10
All	0.77	0.20	0.47	1.25	0.144	0.95	0.05	0.86	1.05
<u>2011 to 2021</u>									
Males	1.01	0.41	0.50	2.10	0.525	1.00	0.04	0.93	1.08
Females	0.59	0.17	0.34	1.01	**0.027**	0.95	0.03	0.90	1.00
Subadults	0.35	0.15	0.16	0.72	**0.002**	0.90	0.03	0.83	0.97
All	0.67	0.18	0.40	1.10	**0.056**	0.96	0.02	0.91	1.01

Table 13. Estimates of gross change (GC), probability that gross change is greater than 1 and annual rate of change (λ) from 2011 to 2021 for distance sampling estimates for age/sex groups by geographic region. Estimates are given for adult females, adult males and subadults.

Area Group	**GC**	**SE**	**Conf. Limit**		**p(GC>1)**	**λ**	**SE**	**Conf. Limit**	
<u>Area 3: Nunavut</u>									
Males	6.60	16.34	1.40	55.55	0.990	1.21	0.12	1.03	1.49
Females	0.23	0.27	0.04	1.00	**0.025**	0.86	0.07	0.73	1.00
Subadults		None observed							
<u>Area 2 Nunavut to Nelson River</u>									
Males	0.60	0.36	0.22	1.58	0.145	0.95	0.05	0.86	1.05
Females	0.56	0.18	0.31	1.02	**0.028**	0.94	0.03	0.89	1.00
Subadults	0.28	0.13	0.12	0.63	**0.001**	0.88	0.04	0.81	0.95
<u>Area 1: Nelson River to Ontario</u>									
Males	1.29	1.00	0.46	4.21	0.720	1.03	0.06	0.93	1.15
Females	1.18	0.81	0.44	3.49	0.661	1.02	0.05	0.92	1.13
Subadults	0.53	0.44	0.14	1.76	0.140	0.94	0.06	0.82	1.06

Table 14. 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).

Subpopulation	**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	Stapleton et al. (2014)
Western Hudson Bay (2016)	1.63 (0.10)	1.25 (0.16)	0.11	0.03	Dyck et al. (2017)
Western Hudson Bay (2021)	1.46 (0.13)	1.39 (0.18)	0.09	0.09	This report
Southern Hudson Bay (2011)	1.56 (0.06)	1.54 (0.08)	0.16	0.12	Obbard et al. (2015)
Southern Hudson Bay (2016)	1.46 (0.06)	1.32 (0.10)	0.19	0.05	Obbard et al. (2018)
Southern Hudson Bay (2021)	1.57	1.47	0.18	0.18	Northrup et al. (2022)
Baffin Bay (2011-13)	1.57 (0.06)	1.51 (0.09)	0.19	0.10	SWG (2016)
Foxe Basin (2009-2010)	1.54 (0.04)	1.48 (0.05)	0.13	0.10	Stapleton et al. (2015)
Davis Strait (2017-2018)	1.42 (0.15)	1.54 (na)	0.12	0.09	Dyck et al. (2022)

Appendix 1 - Sensitivity Analyses to Truncation and Observer Issues

a) Right Truncation

Model 1 was also run with distances up to the maximum distance (2210 included) to test sensitivity of model fit and estimates to right truncation. Model fit was reduced (overall χ^2=24.1, df=16, p=0.0.088) with an overall estimate of 650 bears (SE=128.1, CI=442-966, CV=19.7%). The reduction of model fit suggested that right truncation was justified with minimal overall change in estimates.

b) Left Truncation

Data was left truncated at intervals further from the predefined distances up to 100 meters with minimal change in estimates (Figure A2-1).

c) Sensitivity to Lower Number of Observations on the Right Side of the Helicopter

The right side of the helicopter had the pilot as the front observer and the same individual as second observer for most of the survey. The number of observations from this side was reduced with 30 groups observed compared to 59 to the left side of the helicopter. The difference in detections was modelled using the HeliSide term that allowed for distinct detection function or detection probabilities for the fixed wing (sides pooled), right side and left side of the helicopter. A model that allowed distinct detection probabilities for each side of the helicopter (Table 7, model 4) showed moderate support from the data (ΔAICc=1.1) with an estimate that was 10 bears higher than the most supported model. The main challenge with this analysis is that the actual detection probabilities for the pilot and right observer as indicated by double observer data suggests reasonable detection probabilities (Table 4) with the pilot and right observer showing naïve detection probabilities of 0.77 and 0.83. Therefore, missing of observations is not evident from these probabilities, which could be due to low sample sizes. Detection plots for each observer (Figure 12) also show observations at the full range of distances and therefore there is little evidence of differences in detection functions.

One approach to assess potential bias is to post-stratify the data by side of aircraft. For this the full data set is used to model detection, and then the right and left side observations are used for estimates. Estimates from the 2 sides then add up to the total estimate (Table A2-1). If this is done approximately 40% of the estimate comes from the right side of aircraft and 60% from the left side. Obviously, it would not be expected for the 2 estimates to be equal, however, this gives a sense of the difference in estimates by side of aircraft.

The main potential bias with the right side of the helicopter would be unmodelled heterogeneity of detection probabilities which are exacerbated when detection probabilities become lower. Having 2 weaker observers paired together makes it harder to assess if this is occurring. The amount of data available to model this potential effect is limiting.

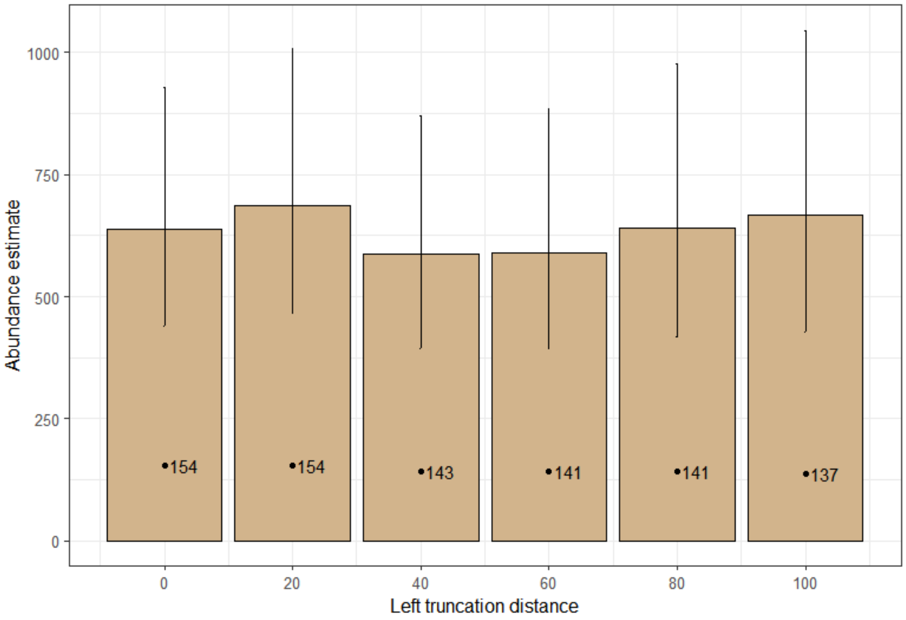

Figure A2-1. Model 1 sensitivity to additional left truncation distances. Points are the sample size of bears used in the analysis.

Table A2-1: Post-stratified estimates by side of aircraft.

Strata	**Individuals**	**N**	**SE**	**CIL**	**CIU**	**CV**
right side						
High	31	115	49.9	50	265	43.4%
Low	9	52	30.2	18	154	57.7%
Moderate N	2	8	8.1	1	45	100.6%
Moderate S	18	79	29.0	38	161	36.9%
Very Low S	0	0	0.0	0	0	0.0%
Total	60	254	71.1	148	437	28.0%
left side						
High	45	175	41.5	110	279	23.7%
Low	4	23	11.8	9	61	50.5%
Moderate N	5	20	20.1	4	112	100.0%
Moderate S	40	166	40.3	103	267	24.3%
Very Low S	0	0	0.0	0	0	0.0%
Total	94	385	73.9	264	560	19.2%
Combined Total (left +right)	154.00	638.56				

Appendix 2 – Further estimation of trend for age-sex post stratified estimates

We used a simulation methodology to estimate confidence limits on gross and annual change in post-stratified estimates of males, adult females (excluding cubs and yearlings), subadults, and all bears. This simulation test is equivalent to a one-sided hypothesis test for decline where the null hypothesis is that the estimate in the previous year is equal or greater than the current year H_o: $N_2 \geq N_1$ and the alternative hypothesis is that the current year estimate is lower (Ha: $N_2 < N_1$). We cross-validated these results using a standard t-test (Satterthwaite 1946, Zar 1996). Comparison of p-values and pGC reveals that these two test methods yielded very similar results.

We note that the 1-tailed hypothesis provides a more powerful test for decline than a 2-tailed test (which tests if the 2 estimates are equal). We felt the 1 tailed test was justified given that the question of management interest was whether a decline was occurring as opposed to whether the two estimates were equal. We also note that overlap of confidence intervals (Figure 20) is not a valid test for statistical significance especially if the hypothesis being tested is one tailed.

Table 1: Results of t-tests for decline in yearly estimates of age/sex groups for WHB (null hypothesis H_o: N2≥N_1 and alternative hypothesis H_a: $N_2 < N_1$). Degrees of freedom (df) for each survey and combined degrees of freedom are also given.

Year	Estimates							t-test		simulation
group	N_1	SE_N_1	df_N_2	N_2	SE_N_2	df_N_2	t-test	df (N_1&N_2)	p-value (1-tailed)	p(GC<1)
<u>2011-6</u>										
Males	289	82.9	91.4	357	65.0	224.6	0.65	206.4	0.741	0.759
Females	286	53.1	204.5	219	54.0	154.3	-0.89	350.1	0.186	0.177
Subadults	173	38.3	124.0	85	24.2	75.3	-1.94	192.4	0.027	0.021
All	956	166.3	244.9	831	138.6	217.5	-0.58	455.7	0.282	0.281
<u>2016-21</u>										
Males	357	65.0	224.6	292	69.2	170.0	-0.68	378.9	0.248	0.236
Females	219	54.0	154.3	168	35.2	192.8	-0.78	273.9	0.219	0.216
Subadults	85	24.2	75.3	60	19.5	108.9	-0.82	158.7	0.207	0.194
All	831	138.6	217.5	639	122.3	198.9	-1.04	413.7	0.149	0.144

2011-21										
Males	289	82.9	91.4	292	69.2	170.0	0.03	208.7	0.512	0.525
Females	286	53.1	204.5	168	35.2	192.8	-1.85	351.6	0.033	0.027
Subadults	173	38.3	124.0	60	19.5	108.9	-2.63	182.8	0.005	0.002
All	956	166.3	244.9	639	122.3	198.9	-1.54	427.5	0.062	0.056