POLAR BEARS: THE CONSERVATION OF AN ARCTIC ICON IN A WARMING CLIMATE

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A Thesis Submitted to the School of Graduate Studies in Partial Fulfilment of the Requirements for the Degree Master of Environmental Studies: Northern Environments and Cultures

Lakehead University

Thunder Bay, Ontario

August, 2014

ACKNOWLEDGEMENTS

I thank my thesis advisors, Dr. Martha Dowsley and Dr. Mitchell Taylor for their guidance and support. They provided me with the knowledge, skills, and encouragement needed to develop as a multidisciplinary research scientist. I thank Dr. Adam Cornwell for serving as a member of my thesis committee, for his critical input, and for sharing his knowledge of climate so generously. I thank my external examiner, Dr. Philip McLoughlin for his prompt review and useful comments. I thank Miroslaw Kuc for technical support with the RISKMAN program. Without his efforts to upgrade and verify the RISKMAN program, the completion of the PVA portion of the thesis would not have been possible. I acknowledge the remaining members of the Department of Geography for their insights and for equipping me with the skills and tools to participate in a post-graduate program. I thank the Torngat Secretariat for providing me with an opportunity to be involved in a traditional ecological knowledge (TEK) study. I thank my girlfriend Sarah for her patience, support and understanding during the last two years. Lastly, I thank my friends and family as this journey would not have been possible without their encouragement and support.

Funding for my studies was provided by Lakehead University in the form of a graduate fellowship, by Dr. Martha Dowsley in the form of a Research Agreement, and by the Social Sciences and Humanities Research Council (SSHRC) in the form of a scholarship.

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STATEMENT OF THESIS

This thesis considers the current status of polar bears in Canada where there are both scientific and traditional knowledge perspectives for all 13 subpopulations. All Canadian polar bear subpopulations are harvested under land claim co-management systems that require sound conservation practices. Some scientists claim that 70% of all polar bears will be lost due to climate warming caused sea ice reductions by 2050 (Amstrup et al., 2007), and also claim that several Canadian subpopulations are already in decline (Obbard et al., 2010). However, Inuit traditional knowledge holders disagree and call the present the "time of the most bears" (Dowsley and Wenzel, 2008). The only tool currently available to respond to indications of decline in polar bear numbers is to reduce the harvest quotas. Thus, any consideration of trend in polar bear numbers in Canada must consider and attempt to reconcile these two knowledge systems and to consider whether polar bears are experiencing a climate crisis.

This thesis is composed of two papers to examine this problem broadly (paper 1) and cover all subpopulations, and to provide a case study of TEK from one subpopulation and analyse it. In the first manuscript, I consider both demographic (scientific) and traditional knowledge perspectives on polar bear subpopulation status. In my second manuscript I consider Labrador Inuit perspectives on polar bear ecology and harvesting for the Davis Strait subpopulation, which was formerly listed as declining (Obbard et al., 2010) but is now recognized as increasing/stable (IUCN/PBSG, 2013). Both of these manuscripts identify points of agreement and differences between the two knowledge systems. I use logical argument (demographic simulation models and statistical analysis), new results (aerial surveys), and empirical observations (atmospheric, oceanographic, and sea ice trends) to understand why scientific and traditional perspectives might differ.

Science and traditional knowledge systems constitute different paths to knowledge, but both attempt to comprehend the same underlying reality. I accept the validity and benefit of both science and traditional ecological knowledge (TEK) in developing sound conservation programs. However, the two knowledge systems do not always agree (Freeman, 1992; Dowsley and Wenzel, 2008). My work demonstrates that TEK or Inuit knowledge of the long-term population trends of polar bears in their local area is reliable. Scientific perspectives depend on the assumptions of the various analysis models they employ being correct. When the assumptions (about the data) of the analysis models are not met, the results can be biased (incorrect). TEK as a knowledge system is inherently less biased because it is entirely empirical, is held collectively, and is constantly being updated. However, TEK can have difficulty in objectively ranking or rejecting competing explanations and views, especially when these views have social or cultural implications. TEK observations of population trends are reliable, but TEK explanations for the observed trends may vary and may be mistaken.

I follow the tradition of Sir Francis Bacon, who believed the best way to know the number of teeth in a horse's mouth is to count them. I show that TEK can be considered as a component of scientific inquiry when it is used to confirm the consistency of a scientific result with nature. I suggest that reliable empirical data are the drivers for both knowledge systems, and that TEK makes fewer assumptions about its observations and mitigates errors or misimpressions by considering information collectively rather than personally. Thus TEK would be expected to provide a better test of conformity for scientific results than the converse because TEK on trends is based on observations, and scientific perspectives of trends are based on methodological assumptions. Similarly I consider climate from an empirical perspective rather than accept a climate future projected by climate models.

I also follow the tradition of Alfred J. Lotka, who was one of the founders of modern population demography. Lotka advised that "the ideal definition is, undoubtedly, the quantitative definition, one that tells us how to measure the thing defined; or, at the least, one that furnishes a basis for the quantitative treatment of the subject to which it relates" (Lotka, 1925: 19). Sound science provides the quantification needed for management prescriptions (e.g., estimates of sustainable removal rates), while TEK is more qualitative. Both perspectives have their purposes, and I suggest that the best conservation measures will stem from scientific results that are consistent with TEK. My second manuscript provides an example of co-management in a jurisdiction where scientific and TEK perspectives have been harmonized (Labrador), and this example could perhaps identify an achievable goal for our Canadian wildlife management systems in general.

Scientists strive to gain reliable knowledge of natural systems, but science can only be validated by testing the predictions that stem from the knowledge that has been gained. Given such limitations, Lotka (1925: 164) also advised, "the best that can be done is to give crude estimates based in the most favourable instances, on counts or observations made with some degree of care, but without pretense of great precision." I follow this advice by utilizing the most recent and most reliable knowledge available from both science and social science perspectives to examine the status of Canadian polar bears. While we may never gain an absolute understanding of natural systems, scientists and wildlife co-managers are most successful at managing wildlife when management decisions are based on the best available information. With regards to polar bears, I suggest that the best available information includes both scientific results and TEK.

Neither manuscript finds support for the claims from some scientists and environmentalist non-government organizations (e.g., World Wildlife Fund, Polar Bears International) that polar bears are currently in a state of climate crisis. I hypothesize that the apparent declines are most likely due to errors in scientific methodology rather than mistaken TEK. I argue that in some circumstances (e.g., Labrador) polar bears may be able to mitigate the negative effects of climate warming and sea ice decline by adapting to the new ecological conditions. I conclude that TEK can and should be used as a form of correspondence with science in respect to polar bear subpopulation trend. Such use of both knowledge systems will reduce the likelihood of misrepresenting polar bear subpopulations and thus mitigate the effects of unnecessary policy or management decisions on resource users.

However, this is not a thesis about either science or TEK or climate change. This is essentially and intentionally a thesis about polar bears that considers these dimensions in an effort to provide reliable knowledge that could be used with confidence to further polar bear conservation.

The following two manuscripts provide several areas of investigation relating to the hypotheses of this thesis. Both manuscripts have been submitted to scholarly journals for publication. By convention, the collective "we" was used for the journal papers. However, as the title indicates, I was the primary and senior author on both manuscripts. Three of my co-authors on the status paper were my thesis supervisors and committee members. The other was Mr. Kuc, who is the programmer for the RISKMAN population viability analysis simulation model. For the Labrador TEK paper, two of my co-authors were committee members (Dr. Dowsley and Dr. Taylor) and the other three co-authors contributed the traditional ecological knowledge interviews that I archived and analyzed.

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Demographic, Traditional Knowledge and Environmentalist Perspectives on the Current Status of Canadian Polar Bear Subpopulations

2013 Canadian Polar Bear Status

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Key words: Canada, climate change, co-management, mark-recapture, polar bear, population viability analysis, RISKMAN, subpopulation status, traditional ecological knowledge, TEK, *Ursus maritimus*

Type: Primary Research

ABSTRACT

Subpopulation growth rates and the probability of decline at current harvest levels were determined for 13 subpopulations of polar bears (*Ursus maritimus*) that are within or shared with Canada based on the most recent demographic and harvest statistics using population viability analyses (PVA). Aboriginal traditional ecological knowledge (TEK) on subpopulation trend agreed with the seven stable/increasing results and one of the declining results, but disagreed with PVA status of five other declining subpopulations. The decline in the Baffin Bay subpopulation appeared to be due to over-reporting of harvested numbers from outside Canada. The remaining four disputed subpopulations (Southern Beaufort Sea, Northern Beaufort Sea, Southern Hudson Bay, and Western Hudson Bay) were all incompletely mark-recapture (M-R) sampled, which may have biased their survival and subpopulation estimates. Three of the four incompletely sampled subpopulations were PVA identified as non-viable (i.e., declining even with zero harvest mortality). TEK disagreement was non-random with respect to M-R sampling protocols. Cluster analysis also grouped subpopulations with ambiguous demographic and harvest rate estimates separately from those with apparently reliable demographic estimates based on PVA probability of decline and unharvested subpopulation growth rate criteria. We suggest that TEK can be used as a reliable correspondence test to evaluate scientific results independent of the assumptions of the scientific analysis models. Considering TEK as reliable for subpopulations where scientific information is suspect (unreliable), we suggest that the current status of Canadian polar bear subpopulations is 12 stable/increasing and one declining. We do not find support for the perspective that polar bears in Canada are currently in any sort of climate crisis. We suggest that climate warming, sea ice decline, and polar bear monitoring should be improved, and that adaptive management practices should be employed as warranted.

INTRODUCTION

Our message is not simple or conventional or consistent with the dire warnings present in much of the polar bear literature since 2006 (Schliebe et al., 2006). We will show that the scientific evidence that some polar bear subpopulations are declining due to climate change-mediated sea ice reductions is likely flawed by poor M-R sampling; and that the complex analysis models employed to overcome these capture issues fail to provide accurate estimates of the demographic parameters used to determine subpopulation status. Our evidence is partly scientific (comparison to subsequent surveys), partly logical (the demographic estimates suggest a dramatic decline that has not occurred) and partly taken from Inuit and Inuvialuit traditional ecological knowledge (TEK). We do not attempt to describe why M-R analysis appears to fail when the sampling does not cover the entire subpopulation area, only to document that the logical projections that use demographic estimates from these analyses are not supported by subsequent surveys or TEK. Our perspectives on climate warming and Arctic sea ice decline are developed from an empirical examination of the open-source data on various indicators of these phenomena. We see reason for concern, but no current climate crisis for polar bears; and suggest that the qualitative projections for dramatic reductions in population numbers and range are overly pessimistic given the response of polar bears, climate and sea ice to the present. We qualify our demographic projections by considering the effects of increasing uncertainty that is inherent to stochastic projections, and find that even projections based on sound estimates of vital rates eventually become too uncertain to provide accurate estimates of geometric mean population growth rate (λ). Our article considers M-R estimates of survival rates and population numbers, age structure estimates of recruitment, population viability analysis, aerial survey population estimates, and traditional ecological knowledge. We also look empirically at Greenland harvest data and at climate-related time series for sea ice, global temperature estimates, arctic temperature estimates, and ocean temperature estimates. We choose not to look at these things individually, nor to just accept what others have written about them because we are concerned that the polarizing influence from climate politics may have generated perspectives about polar bear conservation that are more argumentative than objective. We felt it was necessary to adopt a system approach that included the necessary components required for a comparative consideration of polar bear subpopulation status from a demographic, environmental (climate), and TEK perspective.

Polar bears have always been a symbol of the north and for many years were regarded as a conservation success story (Prestrud & Stirling, 1994; Lunn et al., 2002). Recently they have also become a poster species for “Second-Wave” Environmentalists (Dearden & Mitchell, 2009) seeking to convince policy makers and the public that anthropogenic global warming constitutes a climate crisis (Slocum, 2004). Climate warming is predicted and observed to affect higher latitudes first and most (IPCC, 2007, 2013), and Arctic sea ice during the open water season has been observed to be declining since satellite records began in 1978 (Parkinson et al., 1999; Comiso, 2006; National Snow and Ice Data Center (NSIDC)). Sea ice is required for polar bear movements to feeding areas (Stirling & Derocher, 1993; Ferguson et al., 2000, 2001; Amstrup et al., 2003), to summer retreat areas onshore and on the multi-year pack-ice (Stirling & Parkinson, 2006; Durner et al., 2009), and to locate mates during breeding season (Ramsay & Stirling, 1986; Stirling & Derocher, 1993). Several studies have documented nutritional and recruitment impacts from sea ice reductions on polar bear subpopulations (Stirling et al., 1999; Obbard et al., 2006; Rode et al., 2007; Stirling et al., 2008; Rode et al., 2010, 2014). Sea ice decline could negatively impact affected polar bear subpopulations.

Polar bears evolved from a common ancestor with the brown bear. The range of estimates for the age of polar bears as a species ranges from 4 million years based on deep nuclear genomic sequence data from both paternal and maternal linages (Miller et al., 2012) to 120 thousand years based on the mitochondrial genome (matrilineal) (Lindqvist et al., 2010). If polar bears have existed for the last 4 million years, they would have emerged during the mid-Pliocene approximately 1.25 million years before the onset of northern hemisphere glacial cycles (Bartoli et al., 2005). If polar bears emerged any time prior to or during the previous glacial cycle, they would have persisted through the Eemian interglacial period. During the Eemian interglacial mean annual temperatures were 4°C warmer than the current interglacial (Holocene) for northern latitudes (Müller, 2009), and some northern locations reached temperatures as high as ~7.5°C warmer than the mean temperature for the same area over the last thousand years (Dahl-Jensen et al., 2013). Both scenarios suggest that polar bears are able to mitigate impacts from sea ice decline to an extent not fully exhibited in modern times. Currently, the IPCC predicts globally averaged temperatures to warm ~2°C by 2100, and considers warming of ~4°C by 2100 to be possible although unlikely (IPCC, 2013). Reduction in the heavy multi-year ice and increased productivity from a longer open water season may even enhance polar bear habitat in some areas (Stirling & Derocher, 1993; Derocher et al., 2004; Stirling & Derocher, 2012; Rode et al., 2014). The majority of Canada's polar bears inhabit the Canadian Arctic archipelago (Obbard et al., 2010), where 5 of 13 subpopulations are currently and historically ice-free in late summer and early fall (Lunn et al., 2002; Aars et al., 2006; Obbard et al., 2010). Given the persistence of polar bears through the current and previous interglacial periods, and their ability to accommodate extended retreats onshore and based on the empirical observations of climate and sea ice change (S7); it seems unlikely that polar bears (as a species) are at risk from anthropogenic global warming. However, some subpopulations may experience diminished range, reduced productivity, and subsequent decline in numbers if sea ice declines occur as predicted (Stirling & Derocher, 1993; Derocher et al., 2004; Stirling & Derocher, 2012). While there are many projections of climate change that suggest a nearly ice free Arctic to occur in the warmer months (i.e., September) (IPCC, 2007; Durner et al., 2009; Amstrup et al., 2010; Mahlstein & Knutti, 2012; IPCC, 2013; Overland & Wang, 2013), there are currently no global climate model (GCM) projections of climate change that suggest a totally ice-free Arctic in any season or month.

The nutritional and recruitment impacts from sea ice reductions on polar bear subpopulations are based on direct measures of individuals that would be less likely to be affected by partial (local) sub-sampling. However, polar bear subpopulation status estimates are derived mainly from M-R estimates of subpopulation numbers and survival rates that are presumed to apply to the subpopulation as a whole (Aars et al., 2006; Obbard et al., 2010; Fig. 1). Nine of these subpopulation inventories (Baffin Bay "BB", Davis Strait "DS", Foxe Basin "FB", Gulf of Boothia "GB", Kane Basin "KB", Lancaster Sound "LS", M'Clintock Channel "MC", Norwegian Bay "NW", Viscount Melville Sound "VM") covered essentially all of the area used by the subpopulation during the season of capture (Taylor et al., 2002, 2005, 2006a, 2006b, 2008a, 2008b, 2009; Peacock et al., 2013). These inventories were conducted by capture teams including territorial biologists and aboriginal hunters. The remaining four subpopulation inventories (Northern Beaufort "NB", Southern Beaufort "SB", Southern Hudson "SH" and Western Hudson "WH") were conducted by provincial or federal agencies (i.e., Ontario Ministry of Natural Resources (MNR), United States Geological Survey (USGS) or Canadian Wildlife Services (CWS)), did not include aboriginal stakeholders as part of the regular capture teams, and did not capture polar bears throughout the entire subpopulation area (Regehr et al., 2006; Obbard et al., 2007; Regehr et al., 2007a, 2007b; Stirling et al., 2011).

In 2009, the International Union for Conservation of Nature/Species Survival Commission (IUCN/SSC) Polar Bear Specialists Group (PBSG) Status Report (Obbard et al., 2010) concluded that only 1 of 19 subpopulations is currently increasing, three are stable and eight are declining. For the remaining seven subpopulations, the 2009 PBSG concluded that the available data were insufficient to provide an assessment of trend (Obbard et al., 2010). Canada has or shares 13 of the 19 circumpolar subpopulations (Fig. 1), and the 2009 PBSG Status report lists Canada's subpopulations as: seven declining, four stable or increasing, and two data deficient. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) polar bear status report (COSEWIC, 2008) lists 7 of Canada's 13 subpopulations as stable/increasing, four as declining, and two as unknown. Vongraven & Richardson (2011) provide a status table "report card" that indicates that of 19 circumpolar subpopulations, seven are stable, five are increasing, and seven are data deficient. In December of 2013, the IUCN/SSC PBSG updated their status report listing 1 of 19 circumpolar subpopulations as increasing, five as stable, four as declining, and nine as data deficient (http://pbsg.npolar.no/en/status/status-table.html). For Canada's 13 subpopulations the 2013 PBSG Status report lists one as increasing, five as stable, three as declining, and four as data deficient.

Until 2012 the PBSG considered PVA and TEK in the creation of their status reports. However, the PBSG status report did not consider PVA or TEK perspectives for their most recent status report (IUCN/PBSG, 2012). Rather they employed qualitative judgements based on the expert opinions of their members. The FB listing was changed from "data deficient" to stable based on recent aerial survey results indicating a stable/increasing trend (Garshelis et al., 2012). However, WH continues to be listed as "declining" in spite of a recent aerial survey that indicates no difference (trend not significant at $p>0.05$) and actually indicated a numerical increase (Atkinson et al., 2012). The SH subpopulation was listed as "stable" in 2009 in spite of PVA projections for decline, and continues to be listed as stable, perhaps in response to the recent aerial survey that shows no change in numbers (Obbard & Stapleton, 2013). The DS subpopulation status was revised from "declining" to "stable" with no new research in DS to draw on. The LS and NW listings were also changed to "data deficient" based solely on the age of the subpopulation estimates of vital rates (IUCN/PBSG, 2013). Without a consistent rationale or consideration of all the information relevant to subpopulation status, the 2013 PBSG subpopulation status determinations are difficult to evaluate.

The main evidence for climate (reduced sea ice) effects on demography of subpopulations of polar bear is from the four subpopulations (NB, SB, SH, WH) that were M-R sub-sampled (Regehr et al., 2006; Obbard et al., 2007; Regehr et al., 2007a, 2007b; Stirling et al., 2011). The only subpopulation in Canada (or the world) where a decline was supposedly documented was the WH subpopulation (Regehr et al., 2007a, 2007b). However, recent aerial surveys (Atkinson et al., 2012; Obbard & Stapleton, 2013) indicate that the SH and WH subpopulations have not declined, suggesting that SH and WH demographic rates and subpopulation numbers were under-estimated by the previous M-R work. The time-series of scientific estimates of the circumpolar population and the Canadian subpopulations (Fig. 2) provide no support for a contemporary polar bear crisis (Wiig et al., 1993; Derocher et al., 1997a; Lunn et al., 2002; Aars et al., 2006; Obbard et al., 2010).

One measure of uncertainty in contemporary assessments of climate effects on polar bears is the divergence between scientific perspectives and aboriginal traditional ecological knowledge (TEK) on current subpopulation status (Dowsley, 2005; Tyrrell, 2006; Dowsley et al., 2007; Dowsley & Wenzel, 2008; Henri & Peacock, 2010; Lemelin et al., 2010). We summarize the demographic and TEK perspectives on the current status of all 13 Canadian subpopulations and explore reasons why the two perspectives differ for some subpopulations.

To Inuit and First Nations, polar bears and polar bear hunting are an integral part of their culture and an important part of their traditional economy. Polar bears have been an integral part of the northern traditional economy since the fur trade expanded to the Canadian North in the early 20th century (Honderich, 1991; Dowsley, 2009b; Wenzel, 2011). Aboriginal people (Inuit and First Nations) retain the right to harvest wildlife as both treaty and land claim rights, so long as their harvest is not a conservation concern. The Agreement on the Conservation of Polar Bears (ACPB) (ACPB, 1973) has been in effect since the mid-1970s. The ACPB recognizes the traditional right to hunt and use polar bears by the indigenous societies of the signatory states. Inappropriate and unnecessary harvest and trade restrictions on polar bears wrongfully reduce the benefits of this cultural and economic resource for northern indigenous peoples (Wenzel, 2011).

Historically the main conservation threat to polar bears was agreed to be hunting (Prestrud & Stirling, 1994). In 2005, the IUCN/SSC Polar Bear Specialists Group (PBSG) recommended up-listing the IUCN Red Book status to "threatened" based on a concern that declining sea ice might reduce polar bear stocks as much as 30% over three generations (Red Book definition of threatened). The Center for Biological Diversity (CBD) formally petitioned the US Fish and Wildlife Service (USFWS) to consider polar bears as a threatened species under the US Endangered Species Act (ESA) (Siegel & Cummings, 2005). The CBD did not suggest that polar bears were in jeopardy from hunting practices; rather they alleged that anthropogenic global warming and subsequent sea ice reduction was reducing polar bear habitat range wide.

The CBD petition led to a USFWS range-wide status review (Schliebe et al., 2006) which accepted uncritically the International Panel on Climate Change (IPCC) forecast for climate warming due to greenhouse gas emissions over the next century under expected emissions scenarios (IPCC, 2007). The USGS produced a series of reports in 2007 in support of up-listing polar bears to "threatened" status (Obbard et al., 2006; Amstrup et al., 2007; Hunter et al., 2007; Regehr et al., 2007a, 2007b; Stirling et al., 2007). The US identified polar bears throughout their range as a "threatened" ESA "species at risk" in May of 2008 (US Department of the Interior, 2008; Dowsley, 2009b). Up-listing had the effect of ending polar bear guided sport hunts from the US in Canada because of a provision of the US Marine Mammal Act (MMA) that automatically designates a US ESA "threatened" species as a MMA "depleted" species and importation is banned (Wenzel, 2011). The US sport hunt in Canada was a quota-based hunt, with quotas based on scientific estimates of sustainable yield (Freeman & Wenzel, 2005). This event resulted in the annual loss of about 1.5–2 million dollars into Inuit traditional economy in Nunavut and the Northwest Territories (NWT) (Dowsley, 2009b; Wenzel, 2011). Canada's COSEWIC assessed polar bears in 2008 as a species of "special concern" in Canada, which was no change from the previous three designations (COSEWIC, 2008) using the correct generation time of 12 years.

There is no trend evident from the summed subpopulation numbers from the PBSG status reports (Fig. 2). Other indications of individual subpopulation decline are in conflict with aerial survey results, TEK, or subject to sampling ambiguity, with the exception of the KB subpopulation. We hypothesize that when polar bear subpopulation trends are evaluated by both M-R sampling and TEK; notable differences are most likely due to errors in scientific methodology rather than mistaken TEK.

METHODS

TEK relating to Canadian polar bear subpopulations is summarized from previous status reports (COSEWIC, 2008; Taylor & Dowsley, 2008; Obbard et al., 2010), other internal and published reports and papers (e.g., McDonald et al., 1997; Keith et al., 2005; NTI, 2007; Dowsley & Wenzel, 2008; Kotierk, 2010), Canadian Federal/Provincial Polar Bear Technical (PBTC) Meetings, community and hunters and trappers' (HTO) consultations on polar bear research, management agreements and subpopulation Memoranda of Understanding (e.g., DS MOU) (Government of Nunavut archives), personal communications from: M. Dowsley, M. Taylor and M. Campbell, and various meeting participants. Comments were summarized using qualitative analysis guided by the written materials which include personal notes, management agreements and polar bear management MOU's. Subpopulation summaries were abstracted from the 2008 COSEWIC Polar Bear Status Report (COSEWIC, 2008) and 2009 PBSG Status Report (Obbard et al., 2010) and augmented/updated as required (S1). Subpopulation estimates, recruitment rate estimates, survival rate estimates, and mean annual anthropogenic removals of both males and females were taken from the most recent published and literature and internal reports, including other status reports (COSEWIC, 2008; Taylor & Dowsley, 2008; Obbard et al., 2010); specialist group minutes (Obbard et al., 2010; PBTC, 2011, 2012, 2013), academic presentations (IBA, 2011), and agency reports (S2, S3, S4). Sources for subpopulation-specific estimates of abundance, survival, and recruitment are summarized in Table 1. Subpopulation five-year mean annual anthropogenic removals were taken from Canadian Federal/Provincial Polar Bear Technical Meeting minutes (PBTC, 2008, 2009, 2010, 2011, 2012, 2013) and IUCN/SSC PBSG minutes (Aars et al., 2006; Obbard et al., 2010) that included harvest, defense, illegal, and accidental kills from all jurisdictions that shared a subpopulation (S4). The sex and age distribution of the harvest was estimated from long-term harvest records unless a change in the historic sex/age distribution was indicated for an extended (>5 year) period.

RISKMAN Population Viability Analysis

We used RISKMAN version 2.0 PVA software (Taylor et al., 2001b) to estimate subpopulation trajectories and the probability of decline for each of Canada's 13 polar bear subpopulations under both harvested and unharvested scenarios. RISKMAN is a stochastic, (based on White, 2000) demographic, individual-based, age structured simulation model that was written as an explicit description of the three-year reproduction cycle of polar bears (Taylor et al., 2002, 2005, 2006a, 2006b, 2007, 2008a, 2008b, 2009; Peacock et al., 2013). Taylor et al. (2009) found that the proportion of total variance in survival rates that was parameter (as opposed to environmental) variance was estimated at approximately 92% for adults, 80% for subadults, and 100% for cubs. The nonparametric estimates for recruitment parameters did not allow partitioning parameter and environmental variance, and the only M-R study to partition environmental and survival variance was the Gulf of Boothia (Taylor et al., 2009) study. Barber & Iacozza (2004) found no trends in Gulf of Boothia (GB) sea ice conditions or ringed seal habitat suitability indices in the interval 1980–2000, so the relative proportion of environmental variation may be reduced relative to other subpopulations or to contemporary conditions. We assumed that 75% of total uncertainty was due to parameter variance and 25% was due to environment variance for all subpopulations. We examined the sensitivity of subpopulation growth rate to how total variance was apportioned. Estimates of co-variance were not available for most of the survival and recruitment estimates, and RISKMAN does not have the capacity to incorporate co-variance estimates in stochastic simulations. However, RISKMAN does have a toggle that allows the user to assume independence (correlation coefficient (R) = 0) or complete positive correlation (correlation coefficient (R) = 1). We examined results for the bracketing cases (R=0 and R=1) to evaluate the effect of assuming independence in our simulations.

Estimate of 2013 Abundance and Standing Age Distribution

We ran 5000 Monte Carlo iterations to obtain a distribution of subpopulation trajectories that extended from the last published subpopulation estimate to the present (i.e., 2013). We used mean harvest levels for the pre-2013 simulation interval except for BB and NW where there was a qualitative change in harvest regime that identified two discrete intervals, and required a stepwise simulation with interval-specific mean harvest levels for the BB and NW pre-2013 simulations. The resulting subpopulation number was used as the current estimate of abundance (N_{2013}) and the resulting sex/age distribution was used as the 2013 standing age distribution for post-2013 simulations. The standard error (SE) of the N_{2013} estimate was the standard deviation (SD) of the 5000 Monte Carlo iteration results for 2013.

Subpopulation Growth Rates

We ran both deterministic and stochastic (5000 Monte Carlo iterations) simulations for a t= 20 year period initiated with the estimated 2013 standing age distribution, and using the 2013 estimate of abundance (N_{2013}) for the initial conditions for each subpopulation. For harvest simulations, we assumed the mean harvest level from the past five years (2007/2008 – 2011/2012) (S4 Table S1c) continued for that interval, and the harvested annual subpopulation geometric mean growth rates (λ_H) were determined using the stochastic model. The geometric mean subpopulation growth rate for all subpopulations can be estimated as both the geometric mean Monte Carlo λ_t and also the λ that satisfied: —, where N_t and N_0 were the Monte Carlo simulation mean values. We also monitored the number of Monte Carlo runs that were truncated for each subpopulation simulation. Our protocol for reporting geometric subpopulation growth rates when some Monte Carlo iterations were truncated is described below.

For some subpopulations (NB, SB, SH, WH), only total (includes harvest mortality) survival estimates were provided (Regehr et al., 2006; Obbard et al., 2007; Regehr et al., 2007a, 2007b; Stirling et al., 2011). For these subpopulations, simulations using "total survival" (includes harvest) rates were also conducted for a comparison to simulations using natural survival rates and annual harvest removals. Subpopulation-specific total and harvest mortality rates were provided by various status reports (COSEWIC, 2008; Obbard et al., 2010), allowing us to estimate natural mortality and thus natural survival rates for these total (TOT) survival subpopulations.

Simulations were run for a 20 year period because the long-term standing sex/age distribution implications inherent for sex-selective harvest of polar bears sometimes require 15+ years to become apparent (Taylor et al., 2008d). Our subpopulation status assessments are time-referenced to 2013, and are based on the most recent subpopulation survival (S2 Tables S1 and S2) and recruitment rates (S3 Table S1) and 5-year harvest rate averages (S4 Table S1c). This protocol does not imply that we believe that model projections are valid for 20 years or any specific time frame. The 2013 projection values provide an objective prediction of numbers and trend assuming that: (1) the initial or updated subpopulation estimate is unbiased (correct), (2) harvest numbers and demographic rate (SE) estimates are correct for the simulation interval to 2013, and (3) both harvest and demographic rates remain constant for the simulation interval beyond 2013.

Subpopulation Status and Probability of Decline

Our metric for subpopulation status was the unmodified (no expert correction) probability of decline over the simulation interval. The frequency of subpopulation simulations that declined over 20 years divided by the total number of Monte Carlo iterations was reported as the probability of decline. For subpopulations with total survival estimates (NB, SB, SH, WH), the probability of decline was also estimated with non-harvest simulations using “total” (includes harvest) rather than “natural” survival rate estimates.

Subpopulation Viability

To examine the current (2013) viability of each subpopulation under the most optimistic scenario (assuming no human removals) we ran both deterministic and stochastic (5000 Monte Carlo iterations) simulations initiated at stable age distribution using an initial total subpopulation of 10 000 (SE=0). We report the mean geometric subpopulation growth rate for both deterministic and stochastic simulations where there are no human removals (zero harvest).

”Truncated” Designation

We monitored the number and proportion of iteration runs that were truncated (N_t set to 0 when $N_t \leq 0$). Truncated runs caused estimates of subpopulation growth rate λ to be biased (S5). Truncations occurred for a number of reasons. One cause was the occurrence of the initial random deviate for $N_0 \leq 0$. Another cause for truncation was when all individuals were lost to mortality (individual based model), or when the subpopulation could no longer satisfy a set harvest number at the observed sex ratio (quota). Truncation during a run occurred most frequently in non-viable subpopulations (mean survival and recruitment rates were insufficient for subpopulation persistence even with zero harvest), when the harvest quota was unsustainable, and when the coefficient of variation for initial subpopulation numbers and vital rates was relatively high. All truncations were pooled as a single category regardless of the reason they occurred.

As mentioned above, we estimated the geometric mean subpopulation growth rate for all subpopulations as both the geometric mean Monte Carlo λ^t and also the λ that satisfied:

— ; and recorded the proportion of simulation iterations that were truncated. We report Monte Carlo and "N-based" estimates of λ only when there were no truncations in the subpopulation simulation interval because of concerns these estimates would be biased (S5). Our PVA status estimates are the proportion of runs that declined over the simulation interval which were not affected by truncations (S5).

Correspondence of PVA Trends to TEK Perspectives and Sampling Protocols

We used TEK estimates of status and recent subpopulation estimates from aerial surveys as a consistency check on the PVA subpopulation status determinations based on M-R data. A Fisher's Exact Test (Microsoft, N.D.) was used to compare PVA trends from M-R demographic studies of polar bear subpopulations that were entirely surveyed versus partially surveyed with TEK views on correspondence (to nature) versus non-correspondence. A non-parametric Mann-Whitney U Test (SPSS ©, 2011) was used to compare partially versus entirely surveyed subpopulation estimates of unharvested subpopulation growth rate (subpopulation viability) and subpopulation status (probability of decline) because the Mann-Whitney U Test has greater efficiency than the t-test on non-normal distributions and probability distributions are non-normal. We excluded KB from this analysis because abundance and survival estimates may have been under-estimated by source-sink dynamics and because of a known and long term over harvest (COSEWIC 2008; Taylor et al., 2008a; Obbard et al., 2010). We excluded BB from the subpopulation status (probability of decline) portion of this test because of the over-estimation of

Greenland harvest numbers (S1). A hierarchical cluster analysis using Ward's method applying squared Euclidean Distance as the distance or similarity measure (SPSS ©, 2011) was used to investigate the relationship of unharvested subpopulation growth rate to the probability of decline for harvested subpopulations, and the slopes of points within clusters were calculated by least squares regression. Recent subpopulation estimates from aerial surveys of the FB, and SH, and WH subpopulations (Garshelis et al., 2012; Atkinson et al., 2012; Obbard & Stapleton, 2013) were considered as an independent test of the validity of the trends indicated by the simulations using the mark recapture estimates. A two sample z-test was used to compare the simulation results (using natural survival and actual harvest estimates) to aerial survey estimates for the FB (entire area sampled) and SH and WH (partial area sampled) subpopulations.

RESULTS

Five year mean annual removals, proportion of removals that are female, subpopulation quotas, mean annual growth rates (and associated standard errors), the mean probabilities of decline (and associated standard errors), proportion of truncated runs, and TEK status summaries are listed for each subpopulation in Tables 2a and 2b. Seven of the 13 subpopulations (DS, FB, GB, LS, MC, NW, VM) were identified as approximately stable or increasing (Tables 2b and 3; Fig. 3), while the remaining six (BB, KB, NB, SB, SH, WH) were identified as declining (Tables 2b and 3; Fig. 4).

The trend estimate employing total survival estimates for each of these subpopulations was qualitatively the same as those using natural survival estimates and observed mean annual removal values (Table 2a). The range of deviance between the 2013 estimates of abundance based on natural survival rates to the 2013 estimates using total survival rates was 12.3-38.0 %.

Three (NB, SB, WH) of the six (BB, KB, NB, SB, SH, WH) subpopulations that appeared to be declining also had demographic rate estimates insufficient for long-term persistence (i.e., not viable subpopulations even with zero harvest) (Table 4). The SH subpopulation was projected as marginally viable but lacked sufficient productivity to sustain more than a miniscule fraction (one bear) of the historical annual kill (i.e., 48.625 bears per year) (Table 4). The projected decline in KB and projected stable/increase status for DS, FB, GB, LS, MC, NW, and VM were all consistent with TEK (COSEWIC, 2008; M. Taylor, pers. comm. 1986-2008). TEK perspectives on subpopulation trend were in general agreement with 8 of Canada's 13 subpopulations, but differed from five of six that were projected to be declining (Tables 2b and 3). The probability that TEK status perspectives would differ more often from declining subpopulations than from stable/increasing subpopulations by chance was $p<0.005$ (Table 5). Similarly, a PVA versus TEK consistency comparison suggested that scientific perspectives on trend from subpopulations that had been partially surveyed were less likely to be supported by TEK ($p<0.007$) (Table 6).

Mann Whitney U tests conducted using unharvested geometric subpopulation growth rates (Table 4) and the post-2013 harvested probability of decline (Table 2b) revealed that unharvested subpopulation growth rates were less for subpopulations that had been partially sampled than for subpopulations that were entirely sampled ($p\leq0.004$) and PVA status assessments were more likely to indicate decline for subpopulations that had been partially sampled than for subpopulations that were entirely sampled ($p\leq0.006$) (Table 7). A hierarchical cluster analysis, based on subpopulation unharvested subpopulation growth rate (intrinsic productivity) and harvested subpopulation probability of decline (status), identified two distinct subpopulation clusters (Fig. 5) (post hoc $p\leq 0.027$); Cluster one containing BB, KB, NB, SB, SH, WH, and Cluster two containing DS, FB, GB, LS, MC, NW, VM. The slopes within clusters were not significant, indicating no relationship between intrinsic productivity and probability of decline within clusters.

The sensitivity of simulation results to how total variance was partitioned and the effect of co-variance were relatively minor (Tables 8 and 9).

DISCUSSION

The diversity of perspectives on the status of polar bears has never been greater or more polarized (Treseder & Carpenter, 1989; Nageak et al., 1991; Prestrud & Stirling, 1994; Obbard et al., 2010; Stirling & Derocher, 2012). To some environmentalist Non-Government Organizations (NGO) (e.g., Polar Bears International, CBD, World Wildlife Fund and Greenpeace), polar bears have become both an icon and poster species (Slocum et al., 2004) for their efforts to influence governments and peoples to reduce carbon dioxide emissions, and thus limit or reduce the extent of anthropogenic global warming. To aboriginal people (Inuit and First Nations) polar bears and polar bear hunting remains an integral part of their culture; an important part of their traditional economy; and a constitutional, treaty and land claim right (Dowsley, 2009b; Wenzel, 2011). Although both groups agree that climate warming has caused a decline in sea ice, they disagree about what effects the changes in sea ice have had on polar bear numbers (Tables 2b and 3). Polar bear range states attempt to identify management policies that are responsive to both perspectives, but in practice, most polar bear management decisions are guided by agency researchers resulting in mainly science-based policies (Obbard et al., 2010). This approach fails to reconcile when TEK and science are qualitatively different, or when there is concern that scientific perspectives are influenced by external concerns or if aboriginal perspectives are overly influenced by a desire to harvest more polar bears.

We suggest that the difference between scientific and TEK in this case is partly caused by institutional (science establishment) reluctance to accept TEK as a valid test of correspondence between scientific predictions and observable reality (Aars et al., 2006, Resolution # 1-2005). We do not find evidence for systematic or local misrepresentation of polar bear subpopulation numbers or trends from TEK, aboriginal organizations, or co-management wildlife boards, but do document numerous occasions when TEK accurately identified polar bear subpopulation trends before scientific studies had been conducted that corroborated them (S6).

We suggest that the evidence that polar bears are declining in the NB, SB, SH, and WH subpopulations may be unreliable because the M-R sampling that these studies are based on was conducted in a manner that was inconsistent with the analysis model (Fletcher et al., 2012; Abadi, 2013). The most direct empirical evidence to support this contention is the recent (Fall 2011) aerial surveys of the WH and SH subpopulations (Atkinson et al., 2012; Obbard & Stapleton, 2013) which documents an apparent increase ($p_{increaseWH} \geq 0.6767$ and $p_{increaseSH} \geq 0.7876$) for WH and SH polar bears in contrast to the M-R results (Obbard et al., 2007; Regehr et al., 2007a, 2007b) and various status reports (COSEWIC, 2008; PBTC, 2009; Obbard et al., 2010; Vongraven & Richardson, 2011). The WH subpopulation is often described as the "best known" or the "most thoroughly studied" (Derocher & Stirling, 1995b:215; Atkinson et al., 2012:5) polar bear subpopulation. Regehr et al. (2007a, 2007b) states that the WH polar bear subpopulation is in decline and that these (decline) results are reliable and require no qualification. For similar reasons, Ontario uplisted polar bears to "threatened" status under Ontario ESA (Ontario, 2007) in June 2009 (COSSARO, 2009). However, TEK maintains that both the SH and WH subpopulation numbers have not declined (Tyrrell, 2006; NTI, 2007; Lemelin et al., 2010). Recent aerial surveys (Atkinson et al., 2012; Obbard & Stapleton, 2013) support the local Inuit perspectives (i.e., no decline).

A qualitative difference was identified for the comparison of simulation results (natural survival) to aerial survey for the FB (entire area sampled) than for SH and WH (partial area sampled) subpopulations (Table 10). Both the simulation and the aerial survey resulted in a numerical increase for FB; but the simulation resulted in a numerical decline for SH and WH, while the aerial survey indicated a numerical increase for the partially sampled subpopulations (Table 10). The difference between FB and SH/WH was also evident from the percent difference between the simulation and aerial estimates [100* (higher-lower)/lower] values: FB (7.5%), SH (123%), WH (31%) (Table 10). However, the differences between the simulation and aerial survey estimates were not statistically significant for any the subpopulation comparisons (Table 10). A visual comparison of the M-R based PVA trajectory and the aerial survey estimates for SH and WH (Atkinson et al., 2012; Obbard & Stapleton, 2013) are provided in Figures 6 and 7.

We suggest that the lack of correspondence between PVA simulations results based on M-R studies, aerial survey results, and TEK causes trend estimates for subpopulations that were partially M-R sampled to be unreliable (Table 3). Given the correspondence between M-R based PVA simulations, aerial surveys, and TEK (Tables 2b, 3, and 10; Fig. 6, 7, and 8), we suggest that TEK perspectives on polar bear subpopulation status, given historical harvest levels, provide both a consistency check and an accurate and reliable alternative status measure when scientific results are in doubt.

Stirling & Parkinson (2006) assert that seasonal subpopulations (BB, DS, FB, SH, WH) where polar bears seek onshore retreats during the open water season are also (in addition to WH) likely to decline. However, four of these subpopulations (DS, FB, SH, WH) appear to have increased or remained at approximately historical levels since this paper was published (FB: Garshelis et al., 2012; WH: Atkinson et al., 2012; DS: Peacock et al., 2013; SH: Obbard & Stapleton, 2013). Four of five seasonal polar bear subpopulations appear to have increased or remained constant, not declined as Stirling & Parkinson (2006) suggest. The BB subpopulation status is disputed (Tables 2b and 3; S1). It appears that the perception of decline stems from over-reporting of the Greenland harvest (S1). In support of the TEK perspective, it seems unlikely that the BB subpopulation could have declined to less than half the number 15 years ago, without local hunters being aware of this decline (Table 2a; Fig. 4).

Concurrence with TEK and a low probability of decline (<0.5) suggests seven of the 13 subpopulations (DS, FB, GB, LS, MC, NW, VM) are being harvested sustainably (Table 2b) and are not declining due to climate or any other effects. Of the remaining six subpopulations (BB, KB, NB, SB, SH, WH), PVA simulations based on M-R sampling indicate that these subpopulations are more likely in decline than stable or increasing, but only two of these subpopulations (BB, KB) employ M-R estimates from subpopulations that were entirely sampled. Of the remaining four (NB, SB, SH, WH), three subpopulations (NB, SB, WH) have unharvested subpopulation growth rates that identify as non-viable ($\lambda_{H=0} < 1.00$), while the other one (SH) has an unharvested subpopulation growth rate estimated at less than 0.2% per year (Table 4).

Cluster analysis identifies one group of subpopulations (BB, KB, NB, SB, SH, WH) that had non-random/non-uniform capture sampling, or ambiguous harvest data, or was a source-sink (not closed) subpopulation (S1), and that also had a high (> 0.5) probability of decline at current harvest levels; and a second group of subpopulations (DS, FB, GB, LS, MC, NW, VM) that were sampled throughout their seasonal range, had unambiguous harvest data, and were demographically closed subpopulations and had a low (<0.5) probability of decline. Within clusters there was no relationship between productivity and probability of decline (Fig. 5). This suggests a qualitative difference between groups is methodological rather than ecological. We suggest that the difference is due to under-estimation of subpopulation numbers and survival rates for NB, SB, SH, and WH; over-estimation of Greenland harvest numbers for BB; and inappropriate application of a closed M-R model to a subpopulation that could only persist with immigration from adjacent subpopulations (KB is non-viable with current and historical harvest rates) and apparently also has low productivity. Taylor et al. (2009) found the habitat in KB favorable for polar bears, and cautioned that the KB abundance and survival estimates may have been affected (under-estimated) by the source-sink dynamics.

The expected number of individuals in KB at 2013 is zero (Table 2a) which is in agreement with TEK that KB has been subject to chronic long-term overharvest and would not persist if it did not receive immigrants from adjacent subpopulations (Taylor et al., 2009). TEK and recent survey observations (Dyck, pers.com) confirm that polar bears are currently present in KB. The harvest rate for KB may have changed due to Greenland quotas implemented in January 2006 (Nunavut Wildlife Research Section, 2007) and climate warming related difficulties for Greenland hunters to reach KB from Thule. TEK for the remaining five "declining" subpopulations (Table 2b) indicates that they are stable or increasing. Except for speculation about eventual climate change effects (e.g., Stirling & Parkinson, 2006; Amstrup et al., 2008; Stirling & Derocher, 2012), the scientific perspective that BB is declining is based solely on PVA simulations that show that the joint Greenland/Nunavut harvest could not be sustained by the subpopulation (Taylor et al., 2005; Table 2b). In open water season, the BB subpopulation summered onshore on Baffin and Bylott Islands in the late 1990's (Taylor et al., 2001a). However, most of the bears harvested from this subpopulation are taken in the spring when the bears are on the sea ice (Lee & Taylor, 1994). The Greenland harvest from BB was estimated from an unevaluated voluntary reporting system (Born, 2007). It appears that the portion of the kill reported for west Greenland and assigned to BB subpopulation was over-reported (S1). Over-reporting in Greenland is possible because of the tradition of dividing polar bear skins among all the hunters that participate in the hunt (Born, 2007), or possibly over-reporting occurred in anticipation of a Greenland polar bear quota system. The M-R estimates of the BB subpopulation numbers and productivity may have been under-estimated; however, unlike the SH and WH summer retreat M-R studies (Obbard et al., 2007; Regehr et al., 2007a, 2007b), the entire subpopulation summer retreat area was sampled (Taylor et al., 2005). Until there is independent confirmation, the status of the BB subpopulation is best regarded as disputed, but our prediction based on TEK and the accuracy of other subpopulation estimates where sampling was representative is that subpopulation numbers will have remained about the same.

Some studies (Stirling & Parkinson, 2006; Stirling & Derocher, 2012) suggest that TEK is unduly optimistic because aboriginal people have become confused about the true trends of subpopulations in their area by seeing increased numbers of hungry bears congregating near their communities, then falsely generalizing a positive subpopulation trend from these local concentration sightings. The 2005 PBSG passed the only non-unanimous resolution in its history stating that "(The IUCN Polar Bear Specialist Group) recommends that polar bear harvests can be increased on the basis of local and traditional knowledge only if supported by scientifically collected information" (Aars et al., 2006:57). In other words, using TEK is accepted, but only if it agrees with scientific results. The converse (e.g., scientific results only accepted if TEK concurs) was not proposed. Thus "precautionary" to the PBSG means using scientific results if science and TEK differ, but accepting TEK only if it supports a scientific perspective.

We suggest an alternative approach is to use TEK to confirm scientific perspectives through correspondence of predictions which can be directly observed, such as subpopulation trend (Popper, 1959). For resource users, requiring TEK confirmation prior to increased conservation measures would reduce the probability of undue restrictions (loss of hunting privileges and rights). Viewed this way, the precautionary principle becomes less of a tool for unaccountable environmental enthusiasts and more of a measured and fair protocol that accounts for both conservation concerns and impacts to resource users (Government of Canada, 2003).

Examples of TEK as a Successful Indicator of Trend

There are a number of previous incidences of TEK/science conflict in polar bear management where subsequent studies showed that TEK was correct and scientific results were incorrect (S6). There have also been instances where TEK proved to be conservative rather than exploitive prior to the availability of scientific information. In Baffin Bay, the 1993-1997 subpopulation study (Taylor et al., 2005) showed the 1974-1979 M-R study estimates were mathematically impossible because the known harvest would have extinguished the subpopulation (unpublished NWT file report, 1980; Davis, 1999). The 1974-1979 study estimated 350-600 bears for the whole subpopulation (unpublished NWT file report, 1980) which led to a quota reduction of 45/year and annual compensation payments of $1000.00 per bear until 1996 (Davis, 1999). Baffin Bay Inuit disagreed that polar bears were so few in number. In 1993, a polar bear came into Clyde River, Nunavut and became trapped in the school yard during community consultations on polar bear quotas (M. Taylor, pers. comm. 1986-2008). The bear was chased out of the community, and the Clyde River and Qikitarjuaq quotas were increased. The 1997 BB study (Taylor et al., 2005) was conducted jointly with local Inuit and estimated the subpopulation to number 2074 in 1997. The problem with the initial study was the failure to sample throughout the subpopulation area. The 1971-1976 capture crews were working in spring, and could not search and capture past the floe edge because the pack ice was too unstable to immobilize polar bears safely. Thus only a portion of the subpopulation was sampled (Schweinsburg et al., 1981; Taylor et al., 2005). We are aware of four instances (FB, KB, MC, VM) where TEK identified a subpopulation decline before corroborating scientific information was available to confirm it (Taylor et al., 2002, 2006a, 2006b, 2008a), and four instances where TEK identified stable or increasing subpopulations (DS, GB, LS, NW) before a study confirmed it (Taylor et al., 2008b, 2009; Peacock et al., 2011, 2013; M. Taylor, pers. comm. 1986-2008; M. Dowsley, pers. comm. 2003-2012). Short descriptions of these eight instances are provided in S6.

We observed only one conflict (BB) between PVA simulations and TEK when the M-R based demographic estimates were based on total area sampling (Table 3). In all cases, involving perceptions of trend in polar bear numbers that we are familiar with, when science and TEK did not agree, and subsequent research became available; the new results indicated that the TEK perspective on trend was correct (S6).

Scientific perceptions that polar bears are currently declining due to climate warming is based on observed declines in body condition (Stirling et al., 1999; Obbard et al., 2006; Rode et al., 2007; Stirling et al., 2008; Rode et al., 2012), survival and subpopulation estimates that are suspect because of M-R sampling problems (Table 6), and untested nutritional-ecological models (Molnár et al., 2007, 2010, 2011). TEK perspectives that polar bear subpopulations remain at or above historical levels appear to be supported by both PVA analysis where sampling is subpopulation wide and by recent aerial surveys of subpopulations where M-R estimates were based on partial sampling of the subpopulation area (Garshelis et al., 2012; Atkinson et al., 2012; Peacock et al., 2013; Obbard & Stapleton, 2013; Table 2b). Inconsistencies between our status determinations and those prepared by various polar bear specialists groups and others appear to be due to an inconsistent use of published subpopulation demographic estimates and use of subjective status categories (e.g., “data deficient” for subpopulations where there are data) that we cannot explain. Harvested subpopulations that either do not have sufficient productivity to sustain themselves without harvest (e.g., NB, SB, WH) or would decline with occasional removals (e.g., SH) are sometimes identified as stable (e.g., NB and SH; Obbard et al., 2010; Vongraven & Richardson, 2011), and subpopulations that are most probably increasing are identified as declining or data deficient (e.g., DS (decline) and FB (data deficient); Obbard et al., 2010). Determinations in other recent status reports contain a mixture of old subpopulation estimates and partially projected estimates with no explanation of why projection estimates were used for some subpopulations but not for others (COSEWIC, 2008; PBTC, 2009; Obbard et al., 2010; Vongraven & Richardson, 2011). Scientific information on declines in body condition associated with declines in sea ice was also based on a geographic subsampling of subpopulations; however, the body condition analysis assumptions did not require that every individual in the subpopulation was available for sampling, only that the individuals sampled were representative of the entire subpopulation (Rode et al., 2012). The scope of this review did not include a comparison of TEK versus scientific information on trends in polar bear body condition; but we would expect general agreement between both perspectives because the sampling for the scientific perspective seems appropriate.

Climate, Sea Ice Change, and Population Viability Analysis

A demographic approach to population viability typically assumes that the mean and variance of survival and recruitment rates remain constant for the simulation period. When demographic parameters change progressively as a result of density effects or progressive environmental effects, demographic effects can be modeled as functions of the controlling variables when both the functional relationships are known and the future values of the controlling variables can be estimated (e.g., sea ice decline as per Molnár et al., 2011). Amstrup et al. (2007, 2008) suggested that ~67% of all polar bears would lost by 2050 if CO_2 emissions were not curtailed due to sea ice loss. Stirling & Derocher (2012) review the evidence for climate warming and sea ice reduction effects on polar bear subpopulation numbers and vital rates. However, we found the evidence for sea ice mediated declines in subpopulation numbers and survival rates to be restricted to M-R studies where only a portion of the subpopulation seasonal range had been sampled. Evidence of reduced body condition and reduced recruitment rates associated with sea ice decline in the BB, DS, SB, SH, and WH subpopulations (Stirling et al., 1999; Obbard et al., 2006; Rode et al., 2007; Stirling et al., 2008; Rode et al., 2012) was unambiguous for SB, SH, and WH; however, evidence from BB was compromised because the body condition data that was compared was taken in different parts of the subpopulation area. Evidence for body condition decline as a function of sea ice reduction is ambiguous for DS because subpopulation density was increasing throughout the same period that sea ice was declining (Rode et al., 2012). Rode et al. (2014) found that adult females in the Chukchi Sea (CS) increased in body mass, had larger litters and heavier yearlings during a period of sea ice decline. RISKMAN has the capacity to model density effects, but the mechanism for density effects for polar bears has not been described or quantified for any subpopulation (Taylor, 1994). We did not find sufficient development of relationships between sea ice and demographic rates, or density effects and demographic rates to incorporate these dimensions into our analyses. For further discussion on climate and sea ice effects on polar bears refer to S7.

Management Considerations

We do not advocate polar bear management based on indefinite extrapolation of historical data. In addition to changing environmental conditions, the uncertainty associated with stochastic simulations increases with time. Monte Carlo estimates of geometric subpopulation growth rate are compromised (biased) when simulations must be truncated at zero. Large variances associated with subpopulation estimates (either simulation estimates or survey estimates) can result in Monte Carlo simulation truncations due to random variants ≤ 0 (S5). With few exceptions the demographic data for reliable population viability analysis for Canadian subpopulations are almost expired. There is a need to monitor all harvested subpopulations and periodically update the demographic information in order to estimate demographic performance, harvest sustainability and subpopulation status. Surveys that provide only subpopulation estimates (e.g., aerial surveys) or do not provide the full complement of age structured survival and recruitment estimates (e.g., DNA M-R) may not provide sufficient data to estimate current trends or project future subpopulation numbers. Environmental conditions change and adjustments to management are necessary for long-term sustainability, especially when subpopulations are harvested near maximum sustainable rates. We advocate a more moderate and inclusive approach to polar bear management, greater reliance on TEK to validate and augment scientific studies, and periodic estimation of the full demographic compliment required to estimate subpopulation status and guide harvest quotas.

Future Research

As discussed above, our PVA model software (RISKMAN) does not have a way of incorporating a progressive decline in survival or recruitment as might be expected from a continuing decline in environmental conditions due to climate change, industrial development, tourism, or other factors that could result in negative demographic effects on the polar bear subpopulations. We chose a simulation period of 20 years to estimate the likelihood of a decline to allow for a demographic (standing age distribution) response to sex-selective harvesting

(Taylor et al., 2008d), but we do not suggest that conditions are likely to remain constant for that interval of time. Another limitation to current PVA simulation models is the lack of parameter co-variance estimates and estimates of how total variance is partitioned into environmental and parameter components for survival and recruitment rate estimates (White, 2000). We investigated the effect of our variance partitioning convention (parameter variance = 75%, environmental variance = 25%, co-variance = 0) by exploring a range of partitioning assumptions for each subpopulation (Table 8). The effects of variance partitioning on PVA simulation results appeared to be minor, but may become more important if the environment becomes less stable (more variable) as predicted by climate models. We also examined the effect of co-variance by comparing the change probability of decline for a set of simulations using parameter variance = 75%/ environmental variance = 25% and covariance set to either R=0 (independent) or R=1 (100% correlated) (Table 9). No qualitative changes on PVA simulation results were found except for SH (decline) and KB (decline) when R=1. The SH subpopulation was unable to sustain even occasional removals, and the KB demographic estimates were exceptionally uncertain due to small sample size and the source-sink dynamics of this subpopulation. More accurate harvest reporting from shared subpopulations (especially those shared by Greenland and Quebec) and M-R sampling of entire subpopulation areas would improve the accuracy and reliability of PVA simulations. We recommend simultaneous and systematic collection of TEK to confirm correspondence of scientific results with nature, particularly when the information is used for management purposes.

ACKNOWLEDGEMENTS

The senior author was supported by Lakehead University graduate fellowship and a SSHRC scholarship. We thank Lakehead University Department of Geography for logistic support and discussions that improved this article. Support for Ontario's RISKMAN© upgrading was provided by Nunavut's Department of Environment. Thanks to Canada's Federal/Provincial Polar Bear Technical Committee (PBTC) for the polar bear harvest data to 2013 and for reviewing a preliminary draft of this paper. All comments and suggestions were considered, but not all of these resulted in modification of the manuscript.

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Table 1. Sources for Canadian polar bear subpopulation-specific estimates of abundance, survival, and recruitment.

Subpop.	Source	Year of Estimate	Recruitment Rate Estimate	Survival Rate Estimate	Estimate of Abundance
Baffin Bay	Taylor et al., 2005	1997	X	X	X
	Peacock et al., 2012	2009		X	
Davis Strait	Peacock et al., 2013	2007	X	X	X
Foxe Basin	Taylor et al., 2006b	1994			X
	Garshelis et al., 2012	2010			X
Gulf of Boothia	Taylor et al., 2009	2000	X	X	X
Kane Basin	Taylor et al., 2008a	1997	X	X	X
Lancaster Sound	Taylor et al., 2008b	1997	X	X	X
M'Clintock Channel	Taylor et al., 2006a	2000	X	X	X
Northern Beaufort Sea	Stirling et al., 2011	2006		X	X
	PBTC, 2007	N/A	X		
Norwegian Bay	Taylor et al., 2008b	1997	X	X	X
Southern Beaufort Sea	Regehr et al., 2006	2006	X	X	X
Southern Hudson Bay	Obbard et al., 2007	2005		X	X
	PBTC, 2007	N/A	X		
	Obbard & Stapleton, 2013	2012			X
Viscount Melville Sound	Taylor et al., 2002	1999	X	X	X
Western Hudson Bay	Regehr et al., 2007a, 2007b	2004		X	X
	Atkinson et al., 2012	2011			X
	PBTC, 2007	N/A	X		

Table 2a. Estimates of abundance for polar bear subpopulations within or shared by Canada (BB-Baffin Bay, DS-Davis Strait, FB-Foxe Basin, GB-Gulf of Boothia, LS-Lancaster Sound, MC-M'Clintock Channel, NB–Northern Beaufort, NW–Norwegian Bay, SB–Southern Beaufort, SH-Southern Hudson Bay, VM–Viscount Melville Sound, and WH-Western Hudson Bay). Current estimates were generated using survival and recruitment rate estimates (S2 Tables S1 and S2; S3 Table S1), and harvest data from the PBTC for the period of the most recent abundance estimate to the 2011/2012 harvest season (S4 Tables S1**a**, S1**b**, and S1**c**).

	Previous Abundance Estimate		Current Abundance Estimate			Human-Caused Mortality		
Subpop.	Year of Estimate	N1 (SE)	N2 NAT (2013) (SE)[1]	N2 TOT (2013) (SE)[2]	Prop. of Truncated Runs	Permitted Harvest (quota/year)[3]	5-year Mean harvest (bears/ year)	Prop. Female
BB[4,5,6]	1997	2074 (265)	610.6418 (946.2684)	N/A	0.0008/ 0.6648	178 + Greenland	164	0.36
DS[7,8]	2007	2158 (180)	2206.40 (342.8305)	N/A	0	54 + Quebec	81.2	0.36
FB[9,10,11]	1994	2200 (260)	2934.90 (1748.80)/ 2697.60 (374.3645)[12]	N/A	0.0158/ 0.0036/ 0[12]	106 + Quebec	108.8	0.40
GB[13]	2000	1592 (361)	2945.7 (1722.0)	N/A	0.0052	74	59.8	0.39
KB[14,15]	1997	164 (34.6)	0.6210 (14.4274)	N/A	0.9979	15	5	0.48
LS[16]	1997	2541 (391)	2963.5 (1316.8)	N/A	0.0076	85	84.6	0.31
MC[17]	2000	284 (59.3)	355.4872 (183.9414)	N/A	0	3	2.8	0.20
NB[18]	2006	1004 (275.5)	815.444 (616.8639)	555.104 (321.3018)	0.1088/ 0	65	32.4	0.41
NW[16,19]	1997	203 (44)	194.3868 (70.6449)	N/A	0/ 0.0012	4	1.8	0
SB[20,21,22]	2006	1526 (160.7)	1117.7 (396.8974)	1264.3 (404.8194)	0/ 0	80	36.8	0.33
SH[25,26]	2005	771 (143.3)	380.6094 (300.4066)/ 937.9704(216.6548)	509.02 (66.9076)/ 899.5848 (192.7927)	0.2682/ 0.1626/ 0 0 / 0 / 0	55 + Quebec	57.2	0.33
VM[27]	1999	215 (57.5)	487.4612 (322.5756)	N/A	0.0402	7	4.4	0.19
WH[28,29]	2004	935 (72)	625.6672 (121.2577)/ 965.3274 (160.2103)	509.02 (60.9076)/ 880.0106 (136.16)	0/ 0/ 0/ 0/ 0/ 0	8 + Manitoba	21.6	0.33
Total	N/A	15 667 (796.3)	15 638.5190 (3091.3733)/ 16 298.2402 (2570.2325)[30]	15 492.7538 (3035.8907)/ 16 060.7974 (2513.6153)[30]	N/A	734	660.4	0.314

[1] N2 NAT (2013) is the 2013 estimate calculated using natural survival rates (S2 Table S2).

[2] N2 TOT (2013) is the 2013 estimate calculated using total survival rates (S2 Table S1).

[3] Maximum harvest that is presently allowed by jurisdictions with an identified quota, plus what is taken by non-quota jurisdictions.

[4] Taylor et al. (2005).

[5] The BB simulations used to determine a 2013 estimate of abundance were split into two separate trajectories (1: 1997-2003; 2: 2003-2013) to address a significant increase in the number of bears being harvested (S4).

[6] Dowsley & Wenzel (2008).

[7] Peacock et al. (2013).

[8] Peacock E. unpublished data.

[9] Taylor et al. (2006b).

[10] Comments at community consultations throughout Foxe Basin.

[11] Survival and recruitment rates were established as BB survival and recruitment (Taylor et al., 2005) except that FB adult litter production was 0.85 (see FB comments for meta-analysis rationale).

[12] Simulations were also conducted using a recent aerial survey estimate from Garshelis et al. (2012).

[13] Taylor et al. (2009).

[14] Taylor et al. (2008a).

[15] Further simulations were not conducted because this subpopulation is clearly a harvest sink that can only persist from immigration from surrounding subpopulations.

[16] Taylor et al. (2008b).

[17] Taylor et al. (2006a).

[18] Stirling et al. (2011).

[19] The NW simulations used to determine a 2013 estimate of abundance were split into two separate trajectories (1: 1997-2004; 2: 2004-2013) to address the absence of females in the harvest after the 03/04 harvest season (S4).

[20] Regehr et al. (2006).

[21] Hunter et al. (2007).

[22] Rode et al. (2007).

[25] Obbard et al. (2007).

[26] Simulations were also conducted using a recent aerial survey estimate from Obbard & Stapleton (2013).

[27] Taylor et al. (2002).

[28] Regehr et al. (2007a, 2007b).

[29] Simulations were also conducted using a recent aerial survey estimate from Atkinson et al. (2012).

[30] The 2013 Canadian polar bear population estimate was corrected to account for the recent aerial survey estimates (13).

Table 2b. TEK and the PVA probability of decline for each Canadian polar bear subpopulation were examined to determine subpopulation status. We also included the proportion of runs that were truncated during post-2013 simulations for each Canadian subpopulation. Post-2013 harvested subpopulation growth rates were not reported because truncations are known to bias estimates of subpopulation growth rates (S5). Post-2013 simulations were run for a 20 year period using the 2007/2008-2011/2012 mean annual removals (Table E3) to determine the probability of decline.

	Post-2013 Simulation Results		
Subpop.	**PVA Probability of Decline (SE)**	**Prop. Of Truncated Runs**	**TEK[1]**
Baffin Bay	0.934 (0.0035)	0.9176	Abundant/ Stable/Increasing
Davis Strait	0.3894 (0.0069)	0.0056	Abundant/ Stable/Increasing
Foxe Basin	0.2892 (0.0064)/ 0.2224 (0.0059)[2]	0.180/ 0.0054[2]	Abundant/ Increasing
Gulf of Boothia	0.2016 (0.0057)	0.107	Abundant/ Stable/Increasing
Kane Basin	N/A	N/A	Overhunted/ Declining
Lancaster Sound	0.3632 (0.0068)	0.1312	Abundant/Stable
M'Clintock Channel	0.3178 (0.0066)	0.0458	Recovering/ Increasing
Northern Beaufort Sea	0.8348 (0.0053)/ 0.9344 (0.0035)[3]	0.7328/ 0[3]	Abundant/Stable
Norwegian Bay	0.4034 (0.0069)	0.0106	Low Density/Stable
Southern Beaufort Sea	0.889 (0.0044)/ 0.779 (0.0059)[3]	0.5008/ 0[3]	Abundant/Stable
Southern Hudson Bay	0.9816 (0.0019)/ 0.8772 (0.0046)[4]/ 0.0910 (0.0041)[3]/ 0.9586 (0.0028)[3,4]	0.9696/ 0.7586[4]/ 0[3]/ 0[3,4]	Abundant/Stable
Viscount Melville Sound	0.1884 (0.0055)	0.1106	Recovering/ Increasing
Western Hudson Bay	0.9954 (0.0010)/ 0.9766 (0.0021)[5]/ 1 (0)[3]/ 1 (0)[3,5]	0.747/ 0.1554[5]/ 0[3]/ 0[3,5]	Abundant/ Stable/Increasing

[1] TEK is summarized from previous status reports (COSEWIC, 2008; Taylor & Dowsley, 2008; Obbard et al., 2010), and various other reports, publications, and consultations (e.g., Dowsley & Wenzel, 2008; Kotierk, 2010; Keith et al., 2005; NTI, 2007; McDonald et al., 1997), and personal communications (Dowsley 2003-2012; Taylor 1986-2008; Campbell, 2003-2012; COSEWIC, 2008).

[2] Simulations were also conducted using a 2013 estimate simulated from a recent aerial survey estimate from Garshelis et al. (2012).

[3] Simulations were also conducted using total survival rates (S2 Table S1).

[4] Simulations were also conducted using a 2013 estimate simulated from a recent aerial survey estimate from Obbard & Stapleton (2013).

[5] Simulations were also conducted using a 2013 estimate simulated from a recent aerial survey estimate from Atkinson et al. (2012).

Table 3. The following table compares two methods for identifying subpopulation status. The first is based strictly on PVA and the second is based strictly on TEK. We also propose a third method which is based on a correspondence between both PVA and TEK, where when they do not agree the status is considered to be "uncertain". We also provide a summary of the primary evidence considered for each subpopulation.

Subpopulation	PVA Results	TEK	Trend	Primary Evidence
Baffin Bay	Declining	Stable/Increasing	Uncertain	M-R/TEK
Davis Strait	Stable/Increasing	Stable/Increasing	Stable/Increasing	M-R/TEK
Foxe Basin	Stable/Increasing	Stable/Increasing	Stable/Increasing	Aerial Survey/M-R/TEK
Gulf of Boothia	Stable/Increasing	Stable/Increasing	Stable/Increasing	M-R/TEK
Kane Basin	Declining	Declining	Declining	M-R/TEK
Lancaster Sound	Stable/Increasing	Stable/Increasing	Stable/Increasing	M-R/TEK
M'Clintock Channel	Stable/Increasing	Stable/Increasing	Stable/Increasing	M-R/TEK
Northern Beaufort Sea	Declining	Stable/Increasing	Uncertain	M-R/TEK
Norwegian Bay	Stable/Increasing	Stable/Increasing	Stable/Increasing	M-R/TEK
Southern Beaufort Sea	Declining	Stable/Increasing	Uncertain	M-R/TEK
Southern Hudson Bay	Declining	Stable/Increasing	Uncertain	Aerial Survey/M-R/TEK
Viscount Melville Sound	Stable/Increasing	Stable/Increasing	Stable/Increasing	M-R/TEK
Western Hudson Bay	Declining	Stable/Increasing	Uncertain	Aerial Survey/M-R/TEK

Table 4. Canadian polar bear subpopulation viability based on PVA results generated from natural survival and recruitment rate estimates (S2 Tables S2; S3 Table S1). Each subpopulation was simulated from a stable-age distribution from an initial subpopulation estimate of N = 10 000, SE = 0 for a 20 year period under a harvest moratorium. The unharvested geometric subpopulation growth rate ($\lambda_{H=0}$), PVA probability of decline ($p_{decline}$), and the number of truncations has been included.

Subpopulation	Deterministic $\lambda_{H=0}$	Stochastic $\lambda_{H=0}$ (SE)	$p_{decline}$ (SE)	TRUNC
Baffin Bay	1.0551	1.0547 (0.0274)	0.0026 (0.0007)	0
Davis Strait	1.0387	1.0385 (0.0175)	0.016 (0.0018)	0
Foxe Basin	1.0501	1.0491 (0.0196)	0.0076 (0.0012)	0
Gulf of Boothia	1.0646	1.0639 (0.0369)	0.0472 (0.0030)	0
Kane Basin	1.0064	1.0098 (0.0359)	0.4008 (0.0069)	0
Lancaster Sound	1.0247	1.0249 (0.0189)	0.0908 (0.0041)	0
M'Clintock Channel	1.0263	1.0245 (0.0345)	0.2054 (0.0057)	0
Northern Beaufort Sea	0.9947	0.9887 (0.0794)	0.5198 (0.0071)	0
Norwegian Bay	1.0077	1.0077 (0.0189)	0.3574 (0.0068)	0
Southern Beaufort Sea	0.9808	0.9795 (0.0415)	0.6734 (0.0066)	0
Southern Hudson Bay	1.0014	0.9999 (0.0397)	0.4876 (0.0071)	0
Viscount Melville Sound	1.0652	1.0621 (0.0426)	0.0732 (0.0037)	0
Western Hudson Bay	1.0004	0.9991 (0.0135)	0.5326 (0.0071)	0

Table 5. A Fisher's Exact Test comparison of Science versus TEK correspondence for PVA trends based on mark-recapture demographic studies of Canadian polar bear subpopulations declining suggested that scientific perspectives on trend from subpopulations that were declining were less likely to be supported by TEK ($p<0.005$).

Sample Protocol	TEK Supports	TEK Disputed
Stable/Increasing	7	0
Declining	1	5

Table 6. A Fisher's Exact Test comparison of Science versus TEK correspondence for PVA trends based on mark-recapture demographic studies of Canadian polar bear subpopulations suggested that scientific perspectives on trend from subpopulations that had been partially surveyed were less likely to be supported by TEK ($p<0.007$).

Sample Protocol	TEK Supports	TEK Disputed
Entire Subpopulation Area	8	1
Partial Subpopulation	0	4

Table 7. Mann Whitney U Tests were used to compare distributions for the unharvested geometric subpopulation growth rate ($\lambda_{H=0}$) (Table 4), the probability of decline ($p_{decline}$), and their associated rankings for partially and entirely mark-recapture sampled subpopulations (Table 2b). Estimates of unharvested subpopulation growth rate for $\lambda_{H=0}$ were lower ($p \leq 0.004$), and estimates of the probability of decline for harvested subpopulations were higher ($p \leq 0.006$) for partially sampled subpopulations.

Subpopulation	$\lambda_{H=0}$	$\lambda_{H=0}$ Rank	$p_{decline}$ (λ_H)	$p_{decline}$ Rank (λ_H)	Sample Protocol
Baffin Bay[1]	1.0547	3	0.9340	10	Entire
Davis Strait	1.0385	5	0.3894	6	Entire
Foxe Basin	1.0491	4	0.2892	3	Entire
Gulf of Boothia	1.0639	1	0.2016	2	Entire
Kane Basin[2,3]	1.0098	8	1	13	Entire
Lancaster Sound	1.0249	6	0.3632	5	Entire
M'Clintock Channel	1.0245	7	0.3178	4	Entire
Northern Beaufort Sea	0.9887	12	0.8348	8	Partial
Norwegian Bay	1.0077	9	0.4034	7	Entire
Southern Beaufort Sea	0.9795	13	0.8890	9	Partial
Southern Hudson Bay	0.9999	10	0.9816	11	Partial
Viscount Melville Sound	1.0621	2	0.1884	1	Entire
Western Hudson Bay	0.9991	11	0.9954	12	Partial

[1] BB was excluded from the probability of decline portion of the Mann Whitney U Test.

[2] KB was excluded from both Mann Whitney U Tests.

[3] Post 2013 simulations for the KB subpopulation were not conducted because it was depleted by the 2013 estimate (N=0; refer to Table 2a) and it appears to be a harvest sink that can only persist from immigration from surrounding subpopulations. Thus, a 1.0 probability of decline is assumed.

Table 8. The effect of total variance settings on the geometric mean subpopulation growth rate (λ G) and probability of decline ($P_{decline}$) was examined for each Canadian subpopulation. We examined the difference between total variance settings of 25% parameter variance / 75% environmental variance; 100% parameter variance; and 100% environmental variance. Each subpopulation was simulated from a stable-age distribution from an initial subpopulation estimate of N = 10 000, SE = 0 for a 20 year period under a harvest moratorium. Canadian polar bear subpopulations are defined as: Baffin Bay (BB), Davis Strait (DS), Foxe Basin (FB), Gulf of Boothia (GB), Kane Basin (KB), Lancaster Sound (LS), M'Clintock Channel (MC), Northern Beaufort Sea (NB), Norwegian Bay (NW), Southern Beaufort Sea (SB), Southern Hudson Bay (SH), Viscount Melville Sound (VM), and Western Hudson Bay (WH).

	Deterministic	100% Parameter			75% Parameter/ 25% Environmental			25% Parameter/ 75% Environmental			100% Environmental		
Subpop.	**λ G**	**λ G (SE)**	**$P_{decline}$**	**TRUNC**	**λ G (SE)**	**$P_{decline}$**	**TRUNC**	**λ G (SE)**	**$P_{decline}$**	**TRUNC**	**λ G (SE)**	**$P_{decline}$**	**TRUNC**
BB	1.0551	1.0549 (0.0224)	0.0088 (0.0013)	0	1.0547 (0.0183)	0.0026 (0.0007)	0	1.0544 (0.0125)	0 (0)	0	1.0548 (0.0051)	0 (0)	0
DS	1.0387	1.0384 (0.0120)	0.0304 (0.0024)	0	1.0385 (0.0175)	0.016 (0.0018)	0	1.0383 (0.0111)	0.0004 (0.0003)	0	1.0385 (0.0048)	0 (0)	0
FB	1.0501	1.0492 (0.0220)	0.0044 (0.0009)	0	1.0491 (0.0196)	0.0076 (0.0012)	0	1.0496 (0.0120)	0 (0)	0	1.0494 (0.0054)	0 (0)	0
GB	1.0646	1.0664 (0.0412)	0.0584 (0.0033)	0	1.0639 (0.0369)	0.0472 (0.0030)	0	1.0622 (0.0269)	0.0202 (0.0020)	0	1.0640 (0.0093)	0 (0)	0
KB	1.0064	1.0103 (0.0395)	0.4202 (0.0070)	0	1.0098 (0.0359)	0.4008 (0.0069)	0	1.0059 (0.0238)	0.3978 (0.0069)	0	1.0049 (0.0110)	0.317 (0.0066)	0
LS	1.0247	1.0249 (0.0219)	0.1292 (0.0047)	0	1.0249 (0.0189)	0.0908 (0.0041)	0	1.0244 (0.0119)	0.0212 (0.0020)	0	1.0240 (0.0053)	0 (0)	0
MC	1.0263	1.0258 (0.0363)	0.2142 (0.0058)	0	1.0245 (0.0345)	0.2054 (0.0057)	0	1.0252 (0.0197)	0.1008 (0.0043)	0	1.0258 (0.0082)	0.0016 (0.0006)	0
NB	0.9947	0.9871 (0.0969)	0.521 (0.0071)	0.0006	0.9887 (0.0794)	0.5198 (0.0071)	0	0.9819 (0.0613)	0.5672 (0.0070)	0	0.9922 (0.0224)	0.6248 (0.0068)	0
NW	1.0077	1.0078 (0.0209)	0.356 (0.0068)	0	1.0077 (0.0189)	0.3574 (0.0068)	0	1.0074 (0.0114)	0.2598 (0.0062)	0	1.0071 (0.0050)	0.0796 (0.0038)	0
SB	0.9808	0.9788 (0.0482)	0.6428 (0.0068)	0	0.9795 (0.0415)	0.6734 (0.0066)	0	0.9798 (0.0257)	0.7834 (0.0058)	0	0.9797 (0.0109)	0.9706 (0.0024)	0
SH	1.0014	1.0003 (0.0442)	0.4682 (0.0071)	0	0.9999 (0.0397)	0.4876 (0.0071)	0	1.0006 (0.0247)	0.4804 (0.0071)	0	1.0005 (0.011)	0.4784 (0.0071)	0
VM	1.0652	1.0641 (0.0429)	0.0712 (0.0036)	0	1.0621 (0.0426)	0.0732 (0.0037)	0	1.0550 (0.0448)	0.1106 (0.0044)	0	1.0635 (0.0103)	0 (0)	0
WH	1.0004	0.9987 (0.0159)	0.5312 (0.0071)	0	0.9991 (0.0135)	0.5326 (0.0071)	0	0.9999 (0.0082)	0.5224 (0.0071)	0	1.0001 (0.0039)	0.4954 (0.0071)	0

Table 9. The effect of co-variance R= 0 "independent" versus R=1 "100% correlated" on the geometric mean subpopulation growth rate (λ_G) and probability of decline ($P_{decline}$) was examined for each Canadian subpopulation using total variance settings of 75% parameter variance / 25% environmental variance. Each subpopulation was simulated from a stable-age distribution from an initial subpopulation estimate of N = 10 000, SE = 0 for a 20 year period under a harvest moratorium. Canadian polar bear subpopulations are defined as: Baffin Bay (BB), Davis Strait (DS), Foxe Basin (FB), Gulf of Boothia (GB), Kane Basin (KB), Lancaster Sound (LS), M'Clintock Channel (MC), Northern Beaufort Sea (NB), Norwegian Bay (NW), Southern Beaufort Sea (SB), Southern Hudson Bay (SH), Viscount Melville Sound (VM), and Western Hudson Bay (WH).

	75% Parameter/25% Environmental; R=0			75% Parameter/25% Environmental; R=1		
Subpop.	**λ_G (SE)**	**$P_{decline}$**	**TRUNC**	**λ_G (SE)**	**$P_{decline}$**	**TRUNC**
BB	1.0547 (0.0183)	0.0026 (0.0007)	0	1.0505 (0.0197)	0.0064 (0.0011)	0
DS	1.0385 (0.0175)	0.016 (0.0018)	0	1.0372 (0.0185)	0.0256 (0.0022)	0
FB	1.0491 (0.0196)	0.0076 (0.0012)	0	1.0503 (0.0187)	0.0046 (0.0010)	0
GB	1.0639 (0.0369)	0.0472 (0.0030)	0	1.0440 (0.0348)	0.0970 (0.0042)	0
KB	1.0098 (0.0359)	0.4008 (0.0069)	0	0.9685 (0.0365)	0.8074 (0.0056)	0
LS	1.0249 (0.0189)	0.0908 (0.0041)	0	1.0186 (0.0199)	0.1702 (0.053)	0
MC	1.0245 (0.0345)	0.2054 (0.0057)	0	1.0012 (0.0345)	0.4406 (0.0070)	0
NB	0.9887 (0.0794)	0.5198 (0.0071)	0	0.9292 (0.0699)	0.8626 (0.0049)	0
NW	1.0077 (0.0189)	0.3574 (0.0068)	0	1.0037 (0.0191)	0.4196 (0.0070)	0
SB	0.9795 (0.0415)	0.6734 (0.0066)	0	0.9751 (0.0404)	0.7168 (0.0064)	0
SH	0.9999 (0.0397)	0.4876 (0.0071)	0	0.9375 (0.0521)	0.9240 (0.0037)	0
VM	1.0621 (0.0426)	0.0732 (0.0037)	0	1.0495 (0.0415)	0.1018 (0.0043)	0
WH	0.9991 (0.0135)	0.5326 (0.0071)	0	0.9992 (0.0138)	0.5248 (0.0071)	0

Table 10. Mark-recapture estimates (N), simulation estimates (Sim) and aerial survey estimates (Survey) of abundance are available for the Foxe Basin (FB), Southern Hudson Bay (SH), and Western Hudson Bay (WH) subpopulations. A two sample z-test was used to compare the simulation results (natural survival) to aerial survey estimates for the FB (entire area sampled) and SH and WH (partial area sampled) subpopulations. While simulation results and aerial survey estimates appear numerically similar for FB (7.5% difference) and numerically different for SH (123% difference) and WH (31% difference), none of these differences were statistically significant (p>0.05).

		Foxe Basin		
	N_0[1]	**Sim_{2010}**[2]	**$Survey_{2010}$**[3]	
Year	1994	**2010**	**2010**	
N (SE)	2200 (260)	2772.7 (1307.4)	2580 (278)	$p \leq 0.8854$
		Southern Hudson Bay		
	N_0[1]	**Sim_{2012}**[2]	**$Survey_{2012}$**[3]	
Year	2005	**2012**	**2012**	
N (SE)	771 (143.3)	435.2 (276.8)	969 (202)	$p \leq 0.1193$
		Western Hudson Bay		
	N_0[1]	**Sim_{2011}**[2]	**$Survey_{2011}$**[3]	
Year	2004	**2011**	**2011**	
N (SE)	935 (72)	773.0 (110.6)	1013 (151)	$p \leq 0.1198$

[1] N_0 represents the most recent estimate of abundance from mark-recapture studies.

[2] Sim_t represents the results of simulation from N_0 to the year of the aerial survey.

[3] $Survey_t$ represents the estimate from the most recent aerial survey; FB (Garshelis et al., 2012), SH (Obbard & Stapleton, 2013), WH (Atkinson et al., 2012).

FIG. 1. 2013 Canadian polar bear subpopulation status, subpopulation boundaries, and minimum (September), maximum (April), and hyperphagic (June) sea ice extent. Boundaries of Canadian polar bear subpopulations are defined as: Canadian polar bear subpopulations are defined as: Baffin Bay (BB), Davis Strait (DS), Foxe Basin (FB), Gulf of Boothia (GB), Kane Basin (KB), Lancaster Sound (LS), M'Clintock Channel (MC), Northern Beaufort Sea (NB), Norwegian Bay (NW), Southern Beaufort Sea (SB), Southern Hudson Bay (SH), Viscount Melville Sound (VM), and Western Hudson Bay (WH). Data used for the production of this map was courtesy of NSIDC (http://nsidc.org/data/docs/noaa/g02135_seaice_index/) and Natural Earth (http://www.naturalearthdata.com/downloads/).

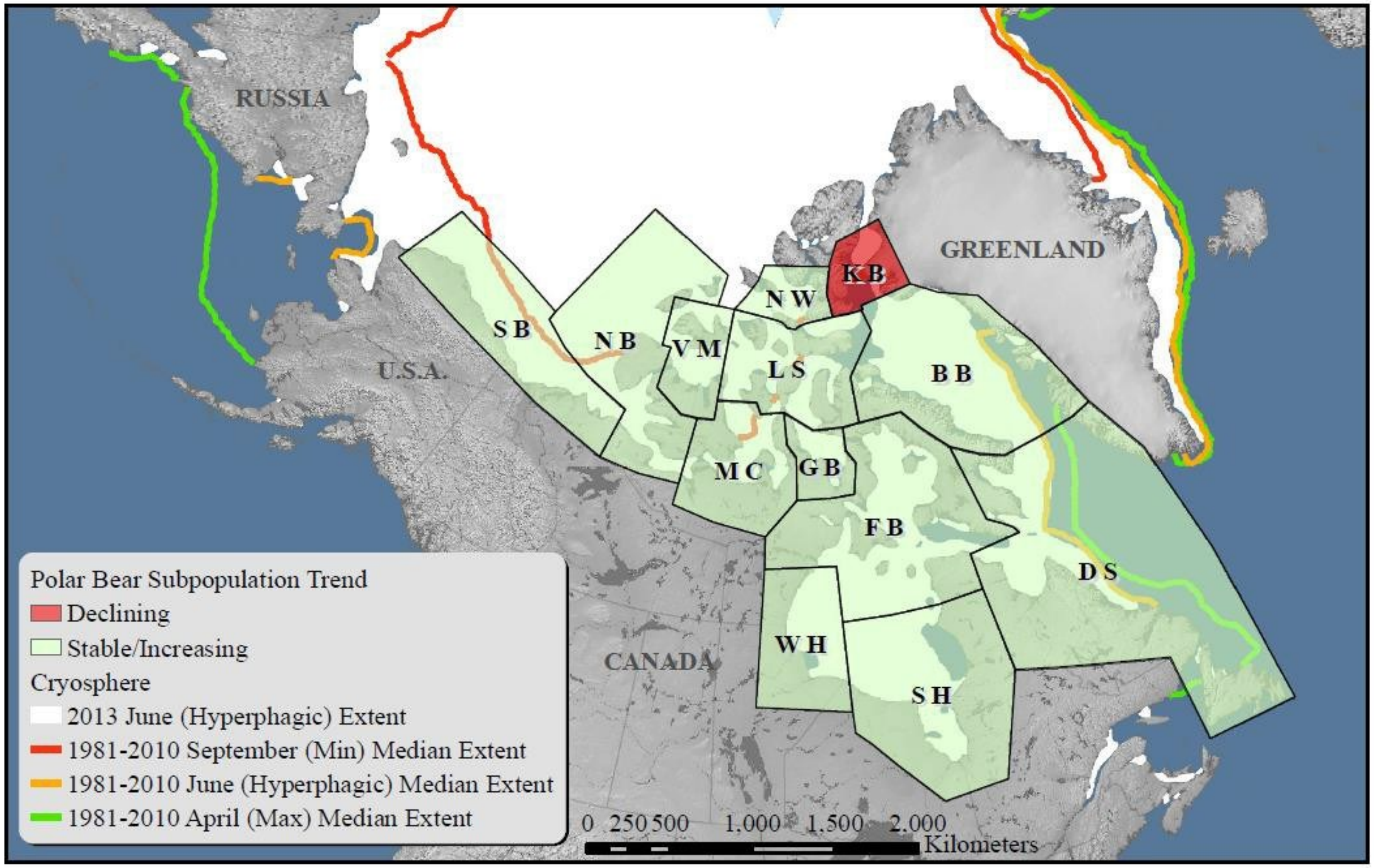


FIG. 2. World and Canadian polar bear subpopulation trends for the 1993-2013 period. Past estimates of abundance were taken from the International Union for Conservation of Nature Polar Bear Specialist Group (IUCN-PBSG) status reports (Wiig et al., 1993; Derocher et al., 1997a; Lunn et al., 2002; Aars et al., 2006; Obbard et al., 2010).

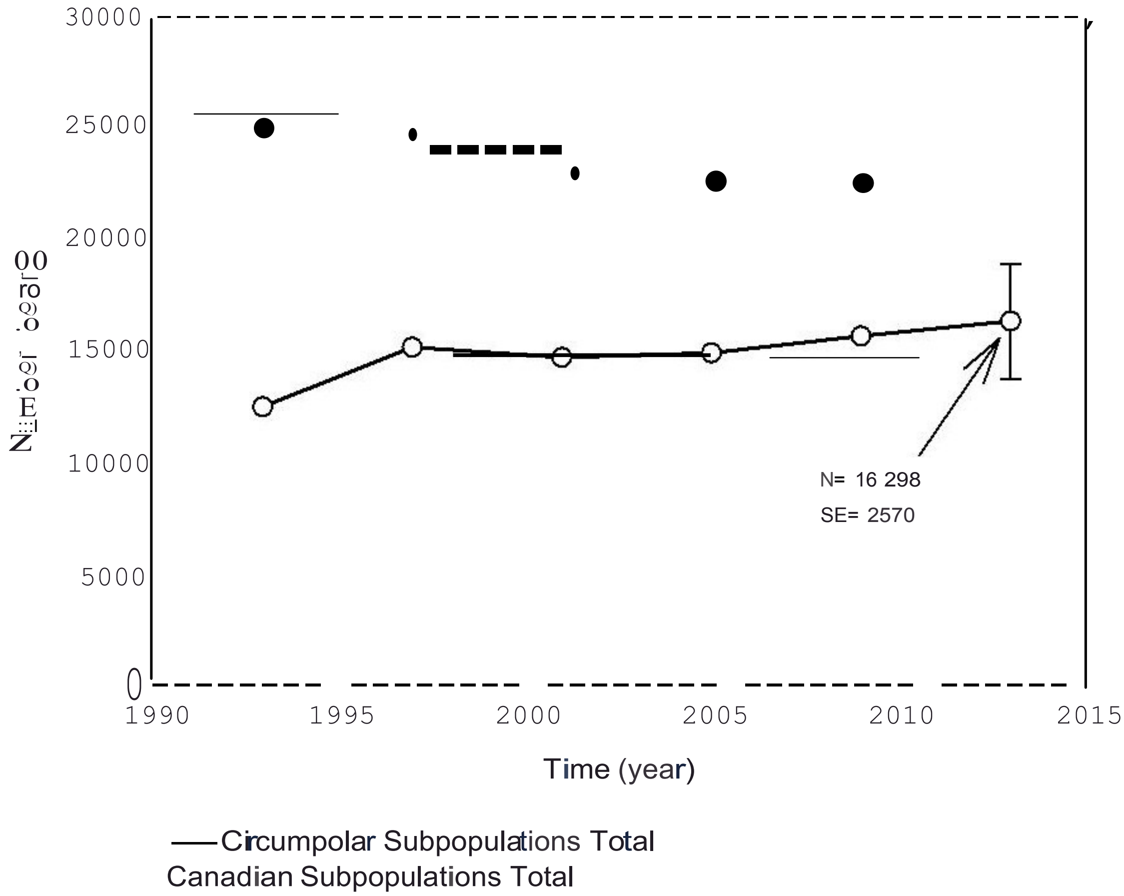


FIG. 3. Stable/Increasing subpopulation trajectories from the year of the most recent estimate of abundance to the present (2013). Davis Strait (DS), Foxe Basin (FB), Gulf of Boothia (GB), Lancaster Sound (LS), M'Clintock Channel (MC), and Viscount Melville Sound (VM) subpopulation trajectories (RISKMAN simulations) are time referenced to the year of the demographic estimate. Demographic estimates are from Peacock et al. (2013), Taylor et al. (2006b), Taylor et al. (2009), Taylor et al. (2008b), Taylor et al. (2006a), and Taylor et al. (2002). Harvest numbers and the proportion of females in the harvest are provided in S4 Tables S1**a**, S1**b**, and S1**c**.

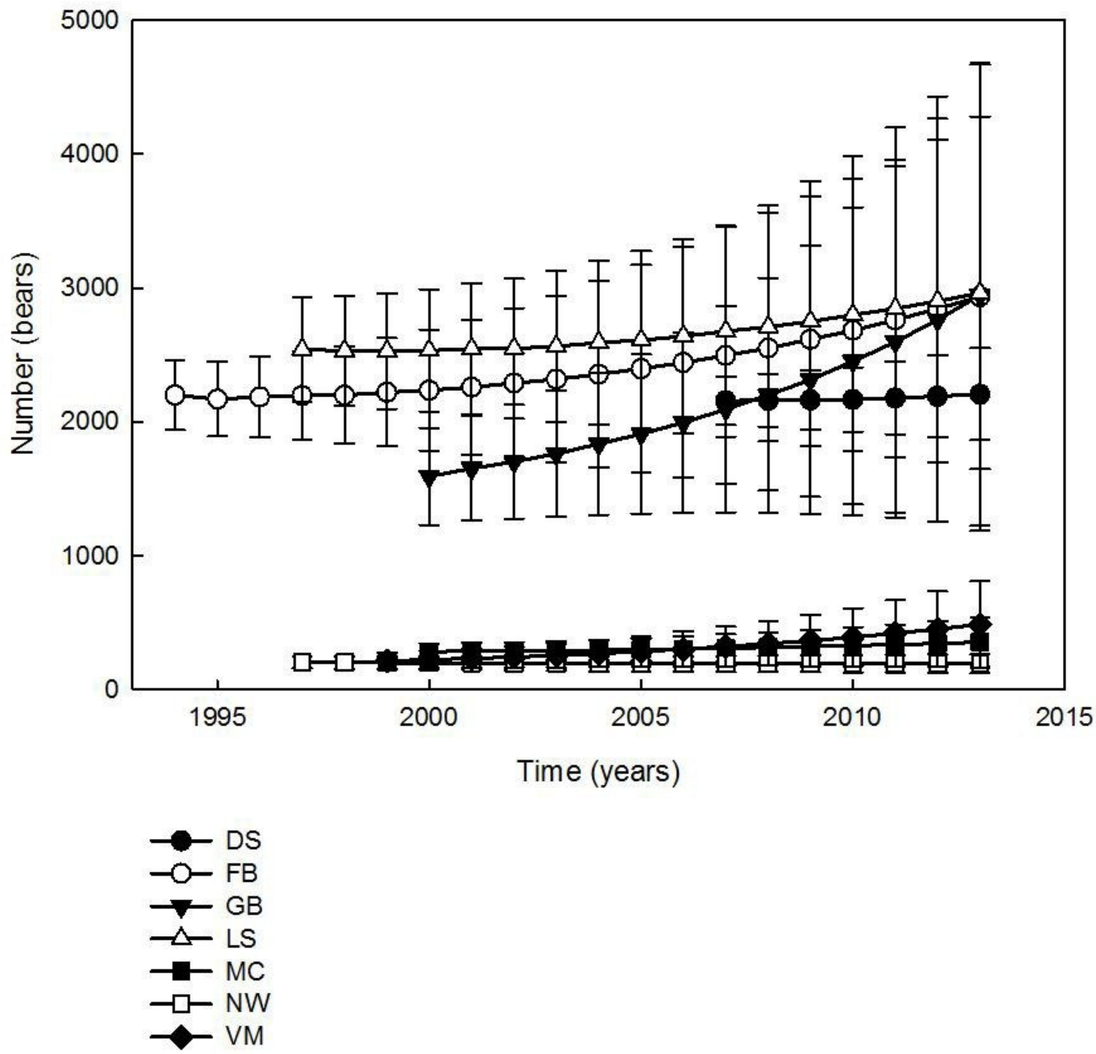


FIG. 4. Declining subpopulation trajectories from the year of the most recent estimate of abundance to the present (2013). Baffin Bay (BB), Kane Basin (KB), Northern Beaufort Sea (NB), Norwegian Bay (NW), Southern Beaufort Sea (SB), Southern Hudson Bay (SH), and Western Hudson Bay (WH) subpopulation trajectories (RISKMAN simulations) are time referenced to the year of the demographic estimate. Demographic estimates are from Taylor et al. (2005), Taylor et al. (2008a), Stirling et al. (2011), Taylor et al. (2008b), Regehr et al. (2006), Obbard et al. (2007), and Regehr et al. (2007a, 2007b). Harvest numbers and the proportion of females in the harvest are provided in S4 Tables S1**a**, S1**b**, and S1**c**.

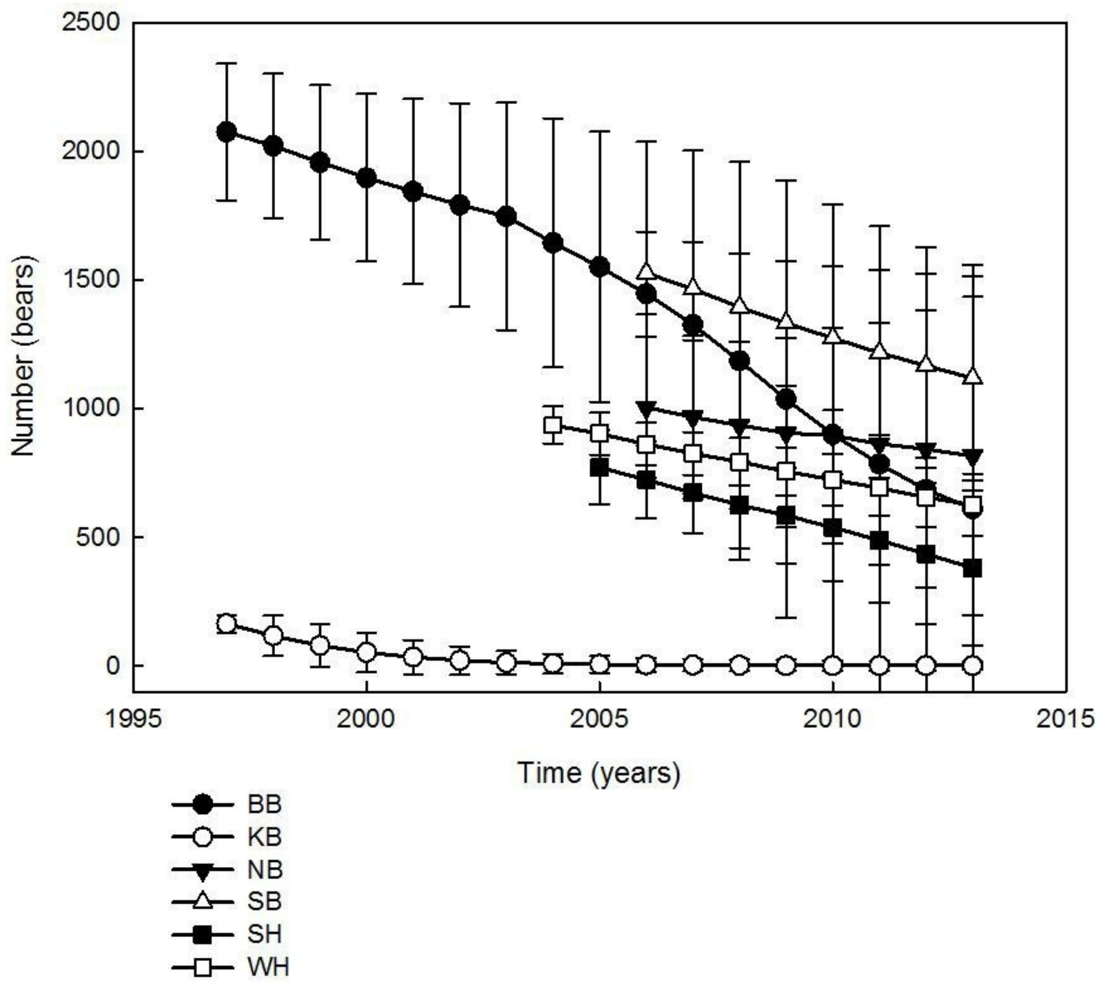


FIG. 5. A hierarchical cluster analysis was used to investigate the relationship between probability of decline as estimated by population viability analysis (PVA) for harvested Canadian polar bear subpopulations and unharvested subpopulation growth rates. Two distinct clusters were identified ($p \leq 0.027$): Cluster 1 containing Baffin Bay (BB), Kane Basin (KB), Northern Beaufort Sea (NB), Southern Beaufort Sea (SB), Southern Hudson (SH), and Western Hudson Bay (WH); and Cluster 2 containing Davis Strait (DS), Foxe Basin (FB), Gulf of Boothia (GB), Lancaster Sound (LS), M'Clintock Channel (MC), Norwegian Bay (NW) and Viscount Melville Sound (VM).

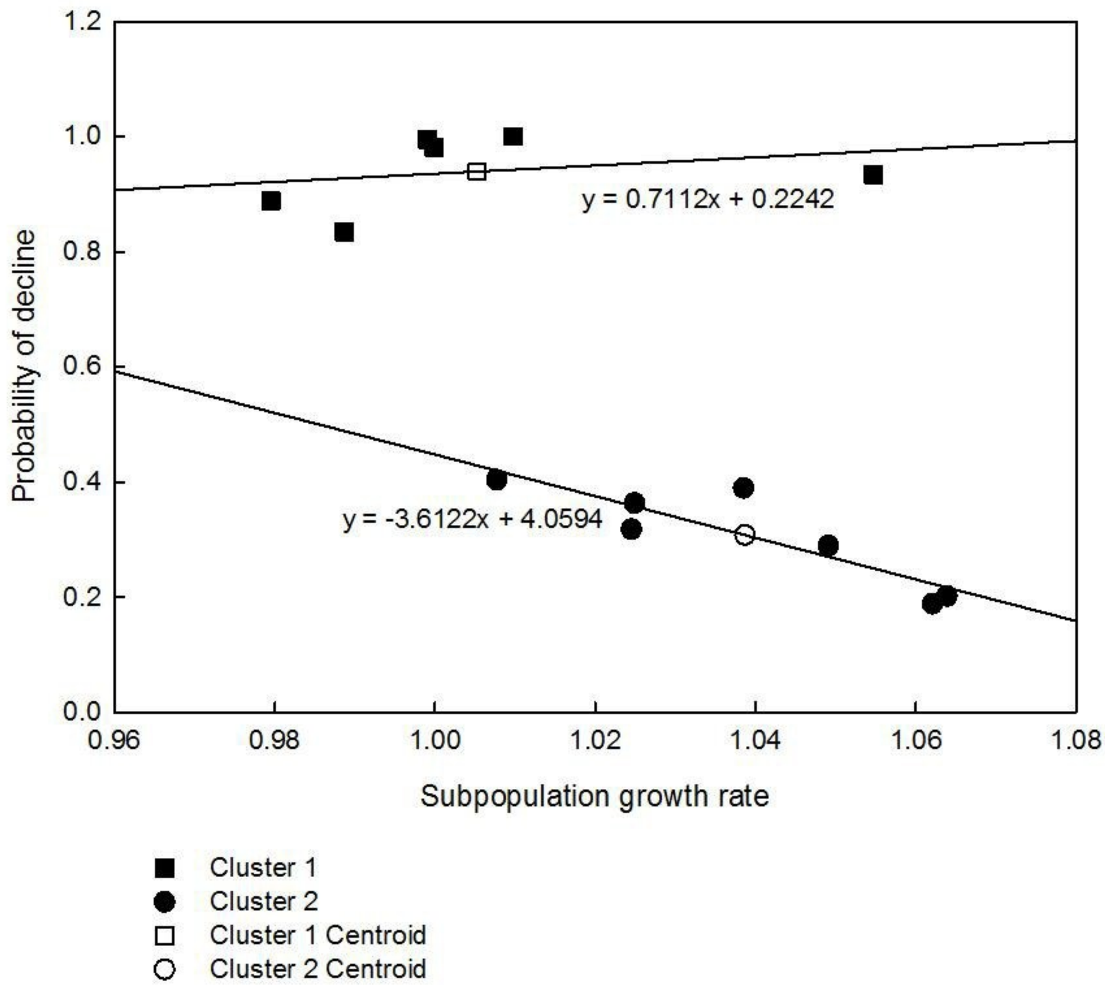


FIG. 6. Comparison of the Western Hudson Bay (WH) subpopulation trajectory (RISKMAN simulation) from 2004 (mark-recapture estimate) to 2011 (aerial survey) estimate.

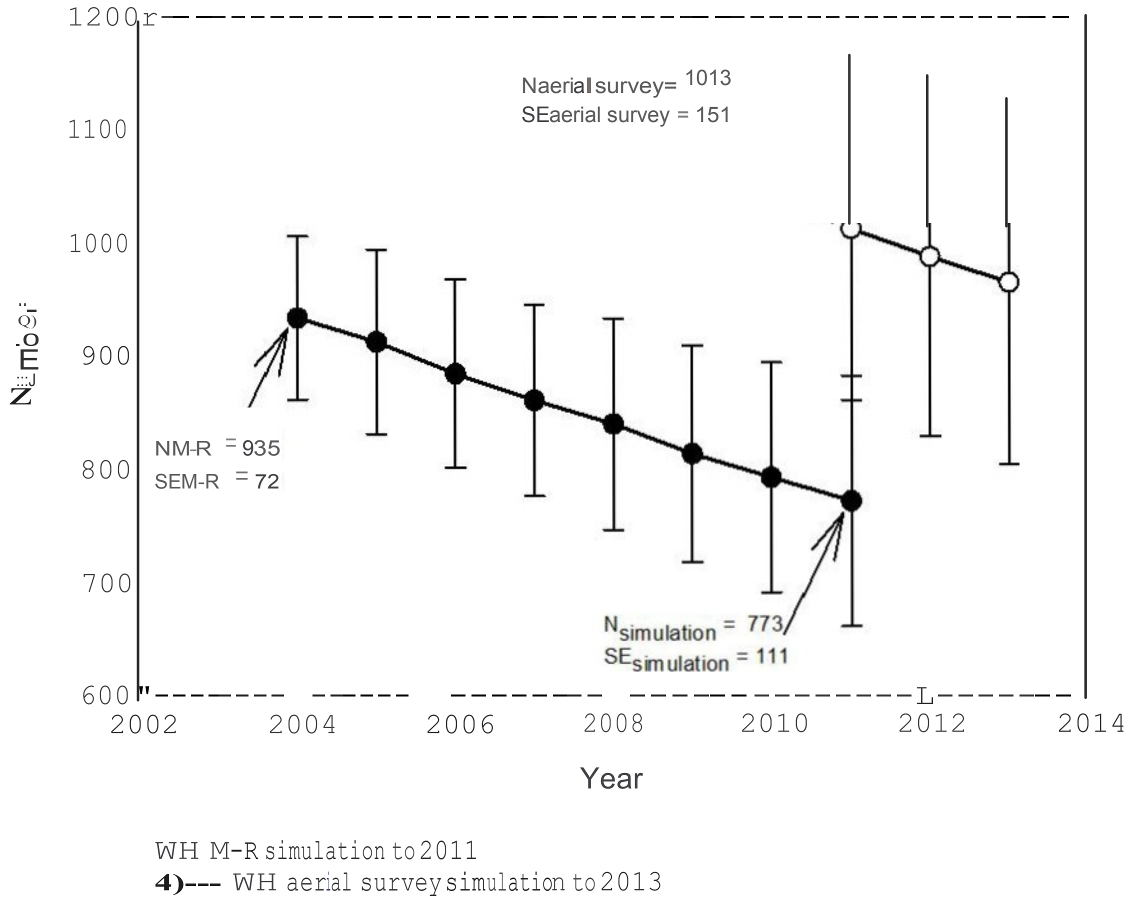


FIG. 7. Comparison of the Southern Hudson Bay (SH) subpopulation trajectory (RISKMAN simulation) from 2005 (mark-recapture estimate) to 2012 (aerial survey) estimate.

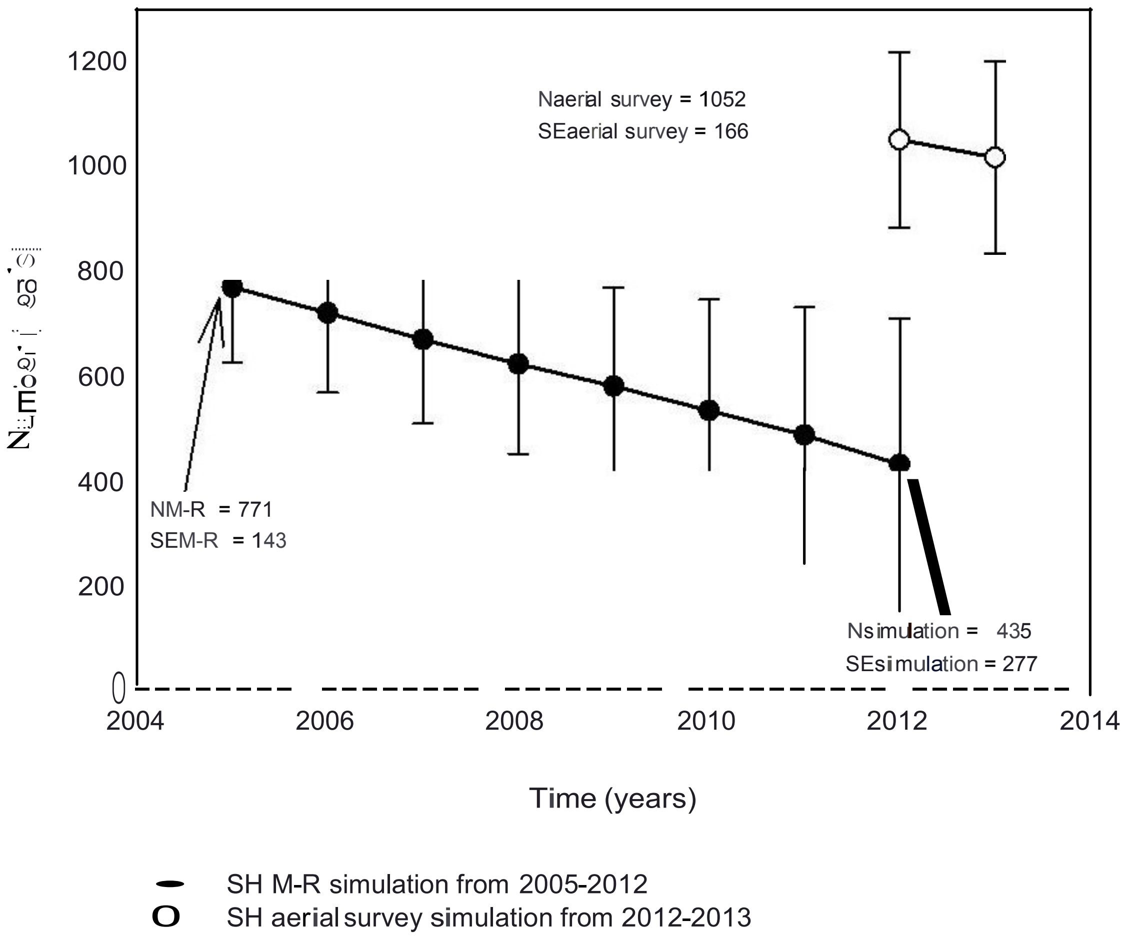
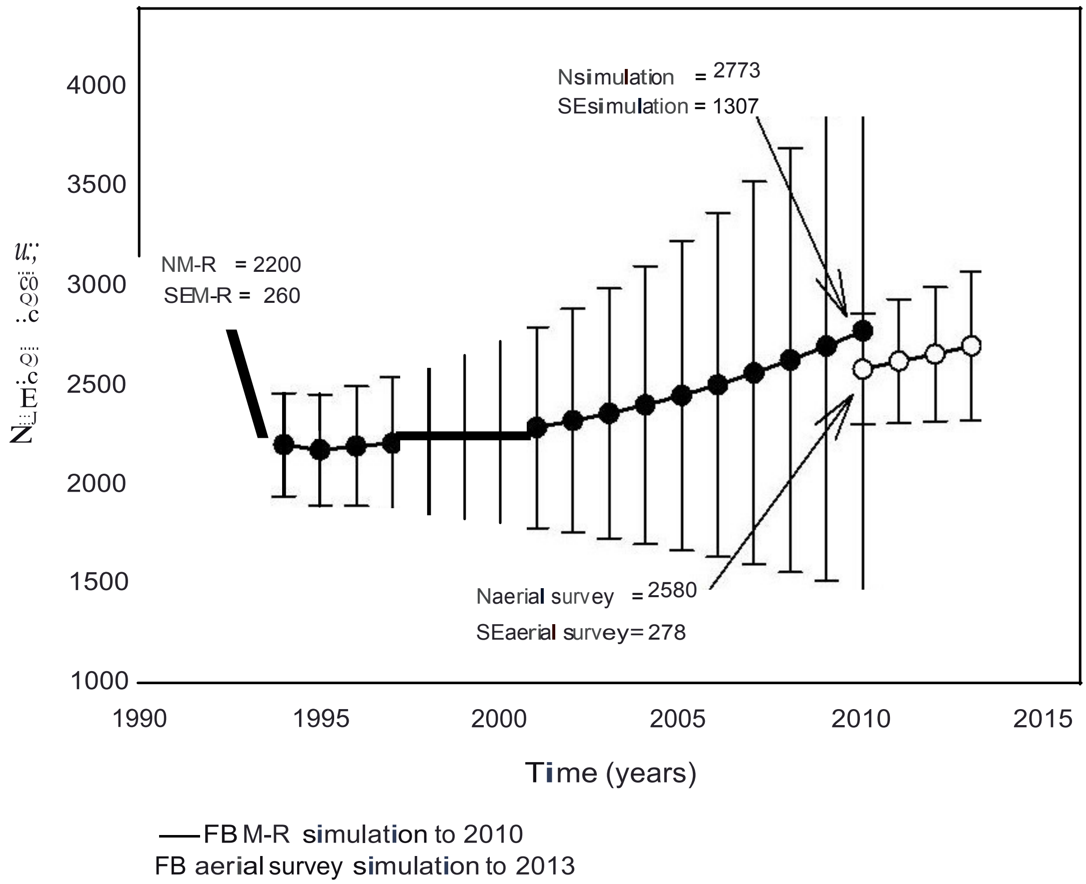


FIG. 8. Comparison of the Foxe Basin (FB) subpopulation trajectory (RISKMAN simulation) from 1994 (tetracycline M-R estimate) to 2010 (aerial survey) estimate using Baffin Bay (BB) birth and survival estimates (meta-analysis) (S2 Table S2; S3 Table S1), and the mean annual FB hanrest (S3 Tables S1a, S1b, and S1c).



Demographic, Traditional Knowledge and Environmentalist Perspectives on the Current Status of Canadian Polar Bear Subpopulations: Supporting Information

Supplementary 1. Ecological summaries of Canadian polar bear subpopulations derived from various status reports (COSEWIC, 2008; Taylor & Dowsley, 2008; Obbard et al., 2010) were updated to include information from recent studies.

Baffin Bay (BB)

Based on movements of adult females equipped with satellite radio-collars and recaptures of tagged animals, the area in which the BB subpopulation occurs is bounded by the North Water Polynya to the north, Greenland to the east and Baffin Island to the west (Taylor & Lee, 1995; Taylor et al., 2001a). A relatively distinct southern boundary at Cape Dyer (Baffin Island) is evident from the movements of tagged bears (Stirling et al., 1980) and recent movement data from polar bears monitored by satellite telemetry (Taylor et al., 2001a). A study of micro-satellite variation did not reveal any genetic differences between polar bears in BB and Kane Basin (KB), although bears of BB differed significantly from those of Davis Strait (DS) and Lancaster Sound (LS) (COSEWIC, 2008; Paetkau et al., 1999). An initial BB subpopulation estimate of 300–600 bears was made by the Government of the Northwest Territories from mark-recapture (M-R) data collected in spring of 1984–1989. However, both telemetry and hunter-kill returns have shown that an unknown proportion of the subpopulation was typically offshore during the spring and, therefore, unavailable for capture. A second study (1993–1997) was carried out annually during the months of September and October, when all polar bears were on shore in summer retreat areas on Bylot and Baffin islands (Taylor et al., 2005). Taylor et al. (2005) estimated the number of polar bears in BB (1998 estimate) at 2074 bears (SE = 266).

The BB subpopulation of polar bears is shared with Greenland, which until January 2006, did not limit the number of bears killed in a year. Based on M-R sampling and harvest recoveries of marked bears, Taylor et al. (2005) estimated the Greenland annual removal at 18–35 bears for the period 1993–1997. However, Born (2002) reported that the estimated the Greenland average annual catch of polar bears from BB was 73 bears per year over the period 1993–1998. Greenland also reported that the average kill by Greenland hunters in BB for the period 2002–2007 was 147 bears per year (range: 75–206 bears per year; COSEWIC, 2008; PBTC, 2008; Obbard et al., 2010). The current (2007-2012) combined five year average reported kill for BB is 167.4 (Table 2a).

The 2004 estimate of <1600 bears is based on subpopulation simulations that employ the pooled Canadian and Greenland harvest records since 1998 (PBTC, 2006, 2007; Obbard et al., 2010). Obviously, if the subpopulation was declining in 2004, and the number of removals was not reduced, there would be still fewer individuals in 2013. Greenland adopted a quota system effective 1 January 2006. However, the 100-bear West Greenland quota will likely include 75–85 bears taken per year in BB, with the remainder being taken from KB (5-25 annually) and DS (<5 annually). In response to community suggestions that polar bears increased in abundance in recent years, the Government of Nunavut increased its quota in BB from 64 to 105 bears in December of 2004. However, when the increased Greenland kill was reported in 2004, and in response to national and international pressure of over-hunting, a phased (10 per year) reduction of the Nunavut BB quota to 65 was initiated. The current 167.4 five-year average annual removal rate in 2012 (Table 2a; S4 Table S1c) is 21% less than the 2009 five-year average annual removal rate of 212 (Obbard et al., 2010). Greenland has thus far not reduced its West Greenland quota, or instituted a harvest program that validates which subpopulation West Greenland kills are assigned to. For many Greenland kills, the location of the kill is recorded as the location of the community that hunter lives in (Obbard et al., 2010).

Baffin Bay Inuit have reported higher or constant abundance of BB polar bears in recent years. TEK from three Baffin Bay communities (Pond Inlet, Clyde River and Qikiqtarjuaq) indicates that hunters and residents have been seeing more polar bears on the land and around communities in the past few years compared to 10–15 years ago (Dowsley, 2005). Significantly more people in the two northern communities experienced this increase compared to people in Qikiqtarjuaq (Dowsley, 2005). Bear encounters have increased, especially among Pond Inlet and Clyde River outpost camp residents, and safety concerns have grown for people on the land, as well as concerns about damaged property (Dowsley and Taylor, 2006a).

From the perspective of population viability analysis (PVA), the BB subpopulation is substantially over-harvested. The discrepancy between TEK and scientific data regarding the trajectory of the BB subpopulation of polar bears is a matter of concern. It has been suggested (Stirling and Parkinson, 2006) that local observations of increased abundance may be due to higher levels of bear activity in response to increased time spent on-shore by polar bears in response to climate warming in the region. Movements inland during summer have apparently increased in places in recent years, but numbers seen near communities have not increased (Dowsley, 2005). All three Baffin Bay communities have reported climate change impacts on the sea ice, such as less shore-fast ice, fewer icebergs and thinner ice, which some people (5/12 people who discussed the idea) thought might contribute to changes in polar bear distribution (Dowsley, 2005; Dowsley and Taylor, 2006a). However, no Inuit respondents expressed confusion about the reason for increased sightings of polar bears or felt that densities around communities were greater than densities elsewhere. The consensus among Baffin Bay Inuit is that they are seeing more bears because there are more bears (Dowsley, 2005; Dowsley &

Taylor, 2006a), in spite of simulation results suggesting that the subpopulation should have been reduced to less than half of its number in 1997.

Peacock et al. (2012) use the harvest recoveries from the previous M-R studies to estimate natural and total survival rates for various time intervals from 1979 to 2009. Although all estimates of total survival have overlapping 95% confidence intervals, the most recent (2002-2009) estimates of the mean for both natural and total survival are numerically lower. Peacock et al. (2012) argues that the decline in the point estimates is real because it is consistent with simulation projections suggesting over-harvest and recent (1999-2008) sea ice decline in Baffin Bay.

Peacock et al. (2012) provide estimates of both natural and total survival, so it was possible to compare simulations from 1997-2013 using survival rates from Taylor et al. (2005) from 1997 to 2013 versus Taylor et al. (2005) survival estimates from 1997-2002, and Peacock et al. (2012) survival estimates from 2003 to 2013 (S1 Fig. S1). The PVA simulation using Peacock et al. (2012) survival rates for the interval 2003-2013 declines to essentially zero (99% of the runs were truncated by the end of the simulation). The Peacock et al. (2012) total survival rate PVA simulation provided a more believable trajectory (S1 Fig. S1), but when Peacock et al. harvest mortality rate (total survival rate-natural survival rate) was used to calculate actual harvest removals in the total survival simulation (S1 Fig. S1), the simulation harvest removals were substantially less than the reported harvest (S1 Tables S1 and S2). We examined the effect that the projected subpopulation decline from 1997 to 2003 had on this inconsistency by also conducting a simulation from 2003 to 2013 from the same 1997 stable age starting conditions (N=2074, SE=266). The discrepancy between annual harvest records and the harvest mortality implied by the Peacock et al. (2012) harvest mortality rate estimate was less for the higher initial subpopulation, however it was still substantial (S1 Table S2).

We explored the hypothesis that recoveries had been under-reported from the Greenland harvest data. We do not consider that harvest recoveries could have been over-reported from the Nunavut harvest data because harvest reporting to a wildlife officer is mandatory, and there is a payment for lip tattoos and ear tags. In Nunavut a vestigial premolar tooth is extracted from harvest and all killed polar bears to allow for aging. This tooth is extracted by the wildlife officer who also inspects for the lip tattoo and usually takes a tissue sample. Harvest reporting in Greenland was the voluntary *Piniarneq* system (filling in a form) until recently (Aars et al., 2006; Obbard et al., 2010). Greenland hunters have a tradition of sharing the hide and meat from harvested polar bears so it is possible that some kills were reported more than once. The *Piniarneq* system involves reporting the annual catch from all species, and reporting is linked to the issuing of hunting licenses for the subsequent year. The issues with this system were summarized in the Greenland Management report to the 2005 Polar Bear Specialist Group (Aars et al., 2006: 141) as follows: "Whether this leads to an under-reporting, over-reporting, or just arbitrary reporting in order to meet requirements when renewing licenses is not clear. An example of sources of error is the report in 2004 of 24 and 10 polar bears reported for Sisimiut and Maniitsoq, respectively (Table 22). Some of these (10 and 5) were reported by hunters with a "part-time" hunting license and are suspected to be of muskoxen. This is currently being investigated by the Greenlandic Ministry of Fisheries (O. Heinrich in litt. 2005)."

The Greenland kill increased dramatically in the early 2000's. This dramatic increase in reported polar bear kills coincided with public discussions of shared quotas for Canadian and Greenland shared subpopulations of marine mammals. Concurrently the extent of Baffin Bay spring (April-June) sea ice declined (Peacock et al., 2012). Peacock et al.'s (2012) apparent over-estimation of harvest mortality based on recoveries could occur from non-reporting of marked animals in the harvest or from over-reporting of the harvest (which would necessarily be unmarked individuals). We used Fisher's Exact Test (Faul et al., 2007) to compare the proportion marked in the Greenland BB reported kill to the proportion marked in the Nunavut BB reported kill for the time bin 2003-2009 identified in Peacock et al. (2012) to determine if these samples were drawn from the same subpopulation (S1 Tables S3 and S4). The proportion marked was greater in the Nunavut kill ($p<0.0041$). We also made the same Nunavut versus Greenland BB recoveries Fisher's Exact Test comparison for the 1998-2002, 1998-2001, and 1998-2000 time bins. The null hypothesis that recoveries were drawn from the same subpopulation was rejected for the intervals 1998-2002 ($p< 0.0103$) and 1998-2001 $p< 0.0424$; but not rejected for the interval 1998-2000 ($p<0.3579$, power=$(1-\beta)=0.67$).

The Burnham (1993) recovery-recapture model used by Peacock et al. (2012) requires that all recoveries are reported and (obviously) that recovery data are correct. We were not able to determine if the divergence in 2003-2009 was due to a failure to report marked recoveries, or an over-reporting the Greenland BB polar bear harvest. However, we believe that over-reporting is the most likely cause of the differential in marked proportion of recoveries between Greenland and Nunavut because that explanation would also explain the inconsistency between simulation results suggesting a precipitous subpopulation decline versus the TEK and anecdotal scientific observations that polar bear numbers have not collapsed in BB.

For these reasons we retained the survival estimates time-referenced to 1997 (Taylor et al., 2005) for our simulations, and suggest that the Peacock et al. (2012) suggestion that survival rates had declined since 2003 is not demonstrated and may be an artifact of sampling issues. We predict that subsequent surveys of BB will confirm that the subpopulation has not declined as the PVA based on the reported kill data suggest it should be (Table 2b; S1 Fig. S1) because the Greenland kill for the interval 2001-2009 (at least) was somehow over-reported.

Davis Strait (DS)

Based on movements of tagged animals and, more recently, of adult females with satellite radio-collars, the DS subpopulation is comprised of bears from the Labrador Sea, eastern Hudson Strait, Davis Strait south of Cape Dyer, and along the eastern edge of the Davis Strait-southern Baffin Bay pack ice (Taylor et al., 2001a). When bears occur in the latter area they are subject to hunting by Greenlanders (Stirling & Kiliaan, 1980; Stirling et al., 1980; Taylor & Lee, 1995; Taylor et al., 2001a).

The initial subpopulation estimate of 900 bears for DS (Stirling et al., 1980) was based on a subjective correction from a M-R estimate of 726 bears, which was felt to be too low. Densities of bears were substantially higher in eastern DS than in the Foxe Basin (FB) subpopulation in the survey of the Quebec coast by Crête et al. (1991). In 1993, the Federal/Provincial/Territorial Polar Bear Technical Committee (PBTC) viewed the DS subpopulation estimate to be 1400 bears to account for bias in sampling created by the inability of researchers to survey the extensive area of offshore pack ice (PBTC, 1993). The first subpopulation study to cover the entire area was Peacock et al. (2013). Peacock et al. (2013) do not report the stochastic subpopulation growth rate but provide the estimates required to show that the DS subpopulation is increasing or stable at current harvest levels.

Stirling and Parkinson (2006) and Peacock et al. (2013) note that sea ice in Davis Strait does not increase to greater than 50% (which has been used to indicate break-up and freeze-up for other polar bear subpopulations) in some years. Stirling and Parkinson (2006) suggest that (like other seasonal subpopulations) the DS subpopulation is thus likely to experience negative effects from sea ice decline. The current DS subpopulation (N = 2158, SE = 180) has relatively low recruitment, but survival rate estimates are comparable to other polar bear subpopulations in Canada (Peacock et al., 2013; S2 and S3). The DS subpopulation appears to have increased from the mid-1970s (a period of climate warming) until the present, mostly due to reduced hunting, but also because of increased harp seal numbers. The data are insufficient to determine the trajectory of the historical subpopulation increase or identify when recruitment rates began to decline. Sea ice records show a recent decline in summer sea ice and earlier and more extended "open water" beginning in the mid-1990s. The data are insufficient to determine if density effects from the subpopulation increase, or declining sea ice, or both are responsible for the decline in recruitment. A decline in body condition is also suggested (Rode et al., 2012), but the data are insufficient to establish whether the decline is due to reductions in sea ice or increases in subpopulation density. At historical harvest levels (~67 per year) the subpopulation is projected to increase at about 0.8% per year (λ_H=1.008). With no harvest the subpopulation would increase at about 3.6% per year (λ_N=1.036). An increase in the harvest to ~82.4 per year would be approximately the maximum sustainable yield (λ_{MSY}=1.000).

Genetic, demographic and space-use strata were detected in DS (Peacock et al., 2013). The southern strata (Labrador) appears to contain bears that are in relatively better condition (Rode et al., 2007), perhaps because these bears are closer to harp seal areas during the ice covered season when most polar bear feeding occurs. However, these indications of spatial stratification are individually and collectively insufficient to support identification of a distinct Labrador subpopulation. DS is best considered a single demographic unit for harvest management purposes based on available scientific data.

The change in status from the 2009 PBSG status report (Obbard et al., 2010) is due to an error in the 2009 PBSG status report PVA simulations.

Foxe Basin (FB)

Based on 12 years of M-R studies, tracking of female bears with conventional radios, and satellite tracking of adult females in Western and Southern Hudson Bay, the Foxe Basin (FB) subpopulation is thought to comprise a demographic unit in Foxe Basin, northern Hudson Bay, and the western end of Hudson Strait (Taylor & Lee, 1995; Taylor et al., 2001a). During the ice-free season, polar bears concentrate on Southampton Island and along the Wager Bay coast; however, significant numbers of bears also occur on the islands and coastal regions throughout the Foxe Basin area. Crête et al. (1991) found relatively few bears of the FB subpopulation along the Quebec shore during the ice-free season. A 1996 total abundance estimate of 2200 (SE = 260) was developed from a M-R analysis based on tetracycline biomarkers (Taylor & Lee, 1994; Taylor et al., 2006b). The marking effort was conducted during the ice-free season, and distributed throughout the entire area. The abundance estimate is believed to have been accurate, and was supported by TEK that the subpopulation had been reduced by harvest but was still abundant. Simulation studies suggest that harvest quotas prior to 1996 reduced the subpopulation from approximately 3000 in the early 1970s to 2100 bears in 1996. Harvest levels were reduced in 1996 to permit recovery of this subpopulation, provided that harvest in Quebec did not increase. Simulation studies using demographic rate estimates from BB (Taylor et al., 2005) and the observed mean harvest rate predict an increase in numbers to 2780 (SE = 806) by 2010.

TEK suggested that the FB subpopulation had increased since 1996 (McDonald et al., 1997). For example, on Southampton Island hunters often fill their quota in a matter of days

(McDonald et al., 1997). However, TEK from the Ivujivik, Quebec area suggested a decrease in polar bear numbers. One hypothesis proposed to explain this observation is that ocean currents in the region are now weaker, allowing bears to become distributed more evenly on the ice during mid-winter rather than congregating at the mouth of Hudson Strait (McDonald et al., 1997). After consultations with Foxe Basin Inuit, Nunavut increased the harvest quota in 2004 to a level consistent with a subpopulation size of 2300 bears (109 bears per year). An aerial survey conducted in 2010 estimated the number of FB polar bears had increased to 2580, SE = 278 (Garshelis et al., 2012), but did not estimate current subpopulation growth rate.

Recruitment and survival rates have not been estimated for FB. Meta-analysis employing the adjacent BB demographic rates appeared to over-estimate FB subpopulation performance by a small and insignificant margin. We reduced BB adult litter production rates from 1.0 to 0.85 (i.e., 85% of available-to-reproduce females produced litters) to produce a simulation that resulted in ~2700 individuals in 2010, and used this empirically corrected (to match the Garshelis et al. (2012) aerial survey estimate) Baffin Bay demographic rates for the FB PVA (Tables 2a and 2b). While this meta-analysis approach is supported by the consistency of the simulation outcome with the aerial survey result, other combinations of recruitment and survival rates could have been identified that provided the same result. The qualitative result of subpopulation increase is not in doubt; however, PVA estimates of the associated uncertainty associated with simulation results should be viewed with caution.

Effects of climate change on the FB subpopulation of polar bears have not been evaluated scientifically. As Foxe Basin is immediately north of Western Hudson Bay and has experienced earlier timing of break-up of sea ice in similar fashion as the rest of Hudson Bay, Stirling and Parkinson (2006) predict a decline in FB polar bears similar to the report for the Western Hudson

Bay (WH) subpopulation (Regehr et al., 2007a, 200b), and various previous polar bear status reports (Obbard et al., 2010; PBTC, 2010). The 2009 PBSG status report (Obbard et al., 2010) lists FB as data deficient. However, the observed increase in number from 2200 in 1996, to 2580 in 2010; suggests that the subpopulation has increased as per the Nunavut harvest management goals during a period of climate warming. Unfortunately, aerial survey does not provide any estimate of current demographic performance.

Gulf of Boothia (GB)

Boundaries of the subpopulation of polar bears inhabiting the Gulf of Boothia were largely based on movements of tagged bears (Taylor & Lee, 1995), movements of collared females in the Gulf of Boothia and adjacent areas (Taylor et al., 2001a), and information from Inuit hunters about how local conditions influence the movements of polar bears. Hunting in the Gulf of Boothia increased from historic levels through the 1970s (Brice-Bennett, 1976); however, the initial quota established by the Government of the Northwest Territories in the Gulf of Boothia was less than the maximum sustainable yield. Local hunters reported that the subpopulation had increased during the 1980s after results of Furnell and Schweinsburg (1984) suggested GB abundance was about 300 bears (based on limited sampling of a small portion of the GB area). Based on Inuit knowledge, recognition of past sampling deficiencies, and an increased understanding of polar bear densities in other areas, the interim subpopulation estimate in the 1990s for the GB subpopulation was 900 bears (M. Taylor, pers. comm.1986-2008). Following completion of a M-R inventory in spring of 2000, the subpopulation was estimated to number 1528 bears (SE = 285; Taylor et al., 2008c). Recruitment and survival rates (S2 and S3) were estimated to be relatively high. In 2004, harvest quotas were increased by the Government of Nunavut to 74 bears per year.

Kane Basin (KB)

Based on movements of adult females equipped with satellite radio-collars and recaptures of tagged animals, the boundaries of the KB subpopulation include the North Water Polynya (to the south), and Greenland and Ellesmere Island to the west, north, and east (Taylor et al., 2001a). Polar bears in KB do not differ genetically from those in BB (Paetkau et al., 1999, their Tables 1 and 2). Prior to 1997, this subpopulation was essentially unharvested in Canadian territory because of its distance from Grise Fiord, the closest Canadian community, and because conditions for travel in the region are typically difficult. However, bears from this subpopulation have occasionally been harvested by hunters from Grise Fiord (since 1997) and harvest continues on the Greenland side of KB. In some years, Greenland hunters also harvest polar bears in the Canadian portion of Kane Basin (i.e., western Kane Basin and Smith Sound) (Rosing-Asvid and Born, 1990; Taylor et al., 2008a).

Few polar bears were encountered along the Greenland coast during M-R surveys between 1994 and 1997, presumably because of harvest pressure by Greenland hunters. The current and only estimate of the KB subpopulation is 164 bears (SE = 35; Taylor et al., 2008a). The best estimate of the Greenland kill is 10 bears per year during 1999–2003 (Born, 2005; Born and Sonne. 2005). However, the actual number being taken by Greenland hunters is uncertain (Rosing-Asvid, 2002; Born and Sonne, 2005) and must be validated before a reliable estimate of KB removals can be developed. The Canadian quota for this subpopulation is five bears per year. The annual combined Canadian and Greenlandic take of 10–15 bears from this subpopulation is unsustainable (Table 2a). Although the habitat appears suitable for polar bears on both the Greenland and Canadian sides of Kane Basin, the density of bears on the Greenland side is much lower than on the Canadian side. The estimates of recruitment and natural (no harvest) survival rates indicate that the subpopulation would most likely continue to decline to extinction even without hunting. The long-term persistence of the KB subpopulation could be understood as a source-sink situation with BB providing immigrants to keep the subpopulation from being exterminated. Another non-exclusive explanation is that the small sample size, influx of non-resident bears, and spatially selective over-hunting resulted in demographic rate estimates that may not be representative of a stable, well-managed subpopulation (Taylor et al., 2008a).

Co-management discussions regarding the hunting of polar bears have been ongoing between Greenland and Canada. Greenland enacted a quota system on 1 January 2006 (West Greenland harvest is not to exceed 100 bears per year, PBTC, 2006); however, because KB, BB, and DS are treated as a single unit for management purposes by Greenland, it is unclear whether reductions in the harvest of bears in KB will result from the establishment of this quota. The mean kill of polar bears in KB has been 10 bears per year for hunters of Greenland in recent years, and <1 for hunters of Nunavut (PBTC, 2006, 2010).

Lancaster Sound (LS)

The central and eastern portion of the LS subpopulation is characterized by high productivity and thus high densities of ringed seals and polar bears (Schweinsburg et al., 1982; Kingsley et al., 1985; Welch et al., 1992; Taylor et al., 2008b). Inuit hunters of Resolute, Grise Fiord, and Arctic Bay have all historically hunted polar bears in Lancaster Sound (Brody, 1976; Riewe, 1976). The western third of this region (eastern Viscount Melville Sound) is dominated by multi-year ice and apparently low biological productivity, leading to low densities of ringed seals (Kingsley et al., 1985). In the spring and summer, densities of polar bears in the western third of the area are relatively low; however, as break-up occurs, polar bears move west to summer on the multi-year pack (Taylor et al., 2001a, 2008b).

M-R data and data on movements of adult females fitted with satellite radio-collars have been collected for bears of LS (Taylor et al., 2001a, 2008b). The current abundance estimate of 2541 bears (SE = 391) is based on an analysis of M-R data current to 1997 (Taylor et al., 2008b). This estimate is considerably larger than the 1979 estimate of 1031 ± 236 bears (mean ± 95% CI) published by Schweinsburg et al. (1982); however, Schweinsburg et al. (1982) sampled from only a portion of the subpopulation area (study area) and Taylor et al. (2008b) covered the entire area. It is not possible to unambiguously determine if the difference in estimates is due to an under-estimate from the 1979 study caused by sampling only a portion of the area, or a subpopulation increase. However, harvest rates in this area have remained approximately constant, therefore the most likely explanation is that the 1979 estimate was biased low as other partial coverage estimates have been (e.g., BB, DS, GB). Schweinsburg et al. (1982) focused on a much smaller area that extended into northern Baffin Bay, compared to the entire LS area (Fig. 1) surveyed by Taylor et al. (2008b).

M'Clintock Channel (MC)

The current boundaries for the MC subpopulation are based on recoveries of tagged bears and movements of adult females with satellite radio-collars in adjacent areas (Taylor & Lee, 1995; Taylor et al., 2001a). These boundaries appear to be a consequence of large islands to the east and west, the mainland to the south, and the multi-year ice in Viscount Melville Sound to the north. A 6-year M-R study in the mid-1970s (Furnell and Schweinsburg, 1984) reported ~1100 animals for an area that overlapped both M'Clintock Channel and the Gulf of Boothia (Fig. 1). The PBTC "corrected" the MC estimate to 900 bears (COSEWIC, 2008) possibly based on approximate extrapolation of apparent densities from the Furnell and Schweinsburg (1984) study. During community consultations in 1993, local hunters suggested reducing the MC estimate to ~700 animals because of a perceived decline in abundance in the southern and western porting of the subpopulation at current harvest rates. The revised TEK estimate of 700 was accepted by the PBTC as an interim value until a new study could be completed, and subsequent MC polar bear quotas and status determinations were based on the TEK estimate. No confidence intervals were identified for either estimate.

Following completion of a M-R inventory in spring of 2000, the subpopulation was estimated to number only 284 bears (SE = 59.3; Taylor et al., 2006a). The legal harvest (averaging 34.0 bears per year from 1979–1999) for MC, which was based on the 700 estimate was not sustainable. The Government of Nunavut implemented a moratorium on hunting for the 2001/2002 and 2002/2003 hunting seasons. The current annual quota for MC was identified as a number that would permit some harvesting and still allow subpopulation growth (i.e., three bears per year). The subpopulation is presumed to be increasing (Tables 2b and 3).

Scientific data, which suggests low abundance of polar bears in MC due to over-harvest, is supported by TEK. Recently, hunters of Gjoa Haven reported that the number of bears near their community had declined over the past 30 years (Keith et al., 2005). Other areas where decreased numbers of polar bears have been reported include the Royal Geographical Society Islands, Pasley Bay, northern King William Island, Gateshead Island, Larsen Sound, and the M'Clintock Channel itself (Atatahak & Banci, 2001). Inuit suggest that polar bears are no longer present in the Queen Maud Gulf area (Keith et al., 2005). Inuit hunters also report a decline in the number of adult male bears in MC but that large males can be found further to the north (Atatahak & Banci, 2001; Keith et al., 2005). This finding is consistent with what one could expect from a male-selective over-harvest (Taylor et al., 2008d).

In addition to unsustainable harvesting, recent changes in habitat and disturbance by humans have been identified by Inuit as potential reasons for the reduced abundance of bears in MC (Keith et al., 2005). One noted habitat change has been the recent absence of multi-year ice and icebergs, which may reduce the quality of habitat because of tide crack (breathing hole) formation at the edges of these bergs and multi-year ice floes. Human disturbances such as the construction of Distant Early Warning (DEW) line sites, construction of Inuksuit, and noise from aircraft and snowmobiles are also thought to have contributed to the low density of bears around Gjoa Haven (Keith et al., 2005).

Northern Beaufort Sea (NB)

Studies of polar bears in the Northern Beaufort Sea have used telemetry and mark and recapture programs at regular intervals since the early 1970s (Stirling et al., 1975, 1988; DeMaster et al., 1980; Lunn et al., 1995). Results suggested that there were separate subpopulations in the Northern and Southern Beaufort Sea areas and not a single subpopulation as was initially thought (Stirling et al., 1988; Amstrup, 1995; Taylor & Lee, 1995; Bethke et al., 1996; Taylor et al., 2001a). An abundance estimate of 1200 polar bears in the late 1980s (Stirling et al., 1988) was believed to be unbiased, but Stirling et al. (2011) now estimate that the number throughout the 2000s was about 980 (SE = 77.5) in the NB subpopulation. The Stirling et al. (2011) constancy of NB subpopulation estimates is inconsistent with the trend from simulations using the survival estimates from the same paper and recruitment estimates from COSEWIC (2008). The historical permitted harvest (65) exceeds a generic estimated sustainable harvest (Taylor et al., 1987c) by 65-44= 21. However, the historical harvest has been less than 44 per year (COSEWIC, 2008; Table 2a) suggesting that the actual kill has remained within sustainable limits. The 2008 COSEWIC status report (COSEWIC, 2008) and the 2009 PBSG status report

(Obbard et al., 2010) both list the NB subpopulation as constant, but also estimate the number at 1200. Using the same PVA criteria used for all other subpopulations, we identify the NB subpopulation as declining, but acknowledge that current estimates of survival and subpopulation numbers are ambiguous. The historic Inuvialuit harvest quota from NB was 59 and Nunavut harvest quota from NB was 6. NB and Southern Beaufort Sea (SB) boundary changes have recently been implemented and resulted in new quotas based on new subpopulation estimates that recognize the incomplete sampling effort.

Hunting of polar bears of the NB subpopulation has historically focused on the Amundsen Gulf (Usher, 1976; Farquharson, 1976), although the western coast and associated sea ice of Banks Island are also important for Inuit hunters (Usher, 1976). In a 2001 interview for the Paulatuuq Oral History project, an elder hunter suggested that the subpopulation in the area had been stable over the past 30 years (Parks Canada, 2004). One explanation for the inconsistency between simulation subpopulation trajectories and Stirling et al. (2011) subpopulation estimates is that Stirling et al. (2011) survival rates were biased low due to un-modeled capture heterogeneity due to incomplete sampling of the subpopulation area. Capture bias would also have causes the subpopulation estimates to be biased low. For this reason, the relationship between scientific perspectives and TEK is ambiguous for the NB.

Analyses, using data from satellite tracking of female polar bears and spatial modelling techniques, suggested that the boundary between the NB and the SB subpopulations needed to be adjusted by expanding the area occupied by bears from NB and reducing that of SB (Amstrup et al., 2004). After consultation between the Inuvialuit, the Inupiat, affected Inuvialuit communities and affected Nunavut Kitkmeot Inuit, the SB-NB boundary was shifted west to 133°. Stirling et al. (2011) estimated the NB subpopulation was about 980. However, the

Inuvialuit Wildlife Management Advisory Committee (WMAC), guided by the suggestion in Stirling et al. (2011) that 1200-1300 (point estimates when 2006 data excluded) could more accurately reflected the current number of polar bears in the NB subpopulation, corrected the Stirling et al. (2011) estimate to 1400 to account for the suspected bias and for areas that were not surveyed during the study. The addition of the 310 bears due to the boundary change raised the final NB estimate to 1710. A new NB quota of 77 (Inuvialuit 71, Nunavut 6) was identified based on the historical estimate of maximum sustained yield (MSY) of 4.5% per year at a 33% female and 67% male sex ratio. This change indicates that NB TEK agrees with Stirling et al. (2011) that numbers have remained stable or increased. The 2013 PBTC status table (PBTC, 2013) designates the NB subpopulation as stable based on TEK. Additionally, the use of the historical MSY index suggests that TEK does not support a decline in subpopulation productivity in the NB. This change in quota occurred after the 2012/2013 harvest season so our analyses used the Stirling et al. (2011) estimates.

Norwegian Bay (NW)

The polar bear subpopulation of Norwegian Bay is bounded by multi-year ice to the west, islands to the north, east, and west, and polynyas to the south (Taylor et al., 2001a; Taylor et al., 2008b). Based on data from M-R studies and satellite radio-tracking of adult females, it appears that most bears concentrate along coastal tide cracks and ridges in the northern, eastern, and southern regions of Norwegian Bay (Taylor et al., 2001a). The preponderance of multi-year ice through most of the central and western areas contributes to low densities of ringed seals (Kingsley et al., 1985) and, consequently, low polar bear density. Grise Fiord hunters reported high concentrations of polar bears in Norwegian Bay during the early 1970s (Riewe, 1976). Taylor et al. (2008b) estimate the number in 1997 in the NW subpopulation was 190 bears (SE =

48.1). Estimates of survival rates (S2) for NW are derived from pooled LS and NW data because these two subpopulations are adjacent and because the number of bears captured in Norwegian Bay was too small for reliable survival estimates (Taylor et al., 2008b). Polar bears in Norwegian Bay are likely to benefit from a warming climate (at least over the short term), which may increase abundance of and accessibility to seals (Derocher et al., 2004).

The harvest quota for the NW subpopulation was reduced to four bears (3M: 1F) in 1996 and remains at this level today. The harvest has been all male for the last five years (Tables 2a; S4 Table S3) and mainly male since 1996 because most harvesting in Norwegian Bay is guided sport hunts. The PBSG status report (Obbard et al., 2010) identified the NW subpopulation as declining with high certainty; however, their PVA short-term simulations (five years) assumed that a portion of the harvest was female based on historical data and did not allow the subpopulation to increase as the females and thus subpopulation productivity increased (Taylor et al., 2008d). The COSEWIC (2008) status report listed the subpopulation as "stable" based on a consideration of the sex ratio of the harvest and the length of time required for the subpopulation trend from an all-male harvest to become clear. TEK from NW suggests no change in polar bear subpopulation numbers there (M. Taylor, pers. comm. 1986-2008).

Southern Beaufort Sea (SB)

The subpopulation of polar bears inhabiting the Southern Beaufort Sea is shared between Canada and the United States (Alaska). On the Canadian side of the border, the historical harvest of bears has been relatively light. The subpopulation experienced an increase in hunting activity in the late 1950s due to an increase in fur prices (Usher 1976); however, by the mid-1970s polar bears were only killed opportunistically during hunts for other species by Aklavik and Inuvik hunters (Usher 1976). Hunters of Tuktoyaktuk recall people from their community also hunting polar bears during this time (Frank Pokiak, Chair, Inuvialuit Game Council, letter to COSEWIC Terrestrial Mammals Specialist Subcommittee, 19 January 2007; COSEWIC, 2008). The Cape Bathurst area was reported to be an important area for hunting polar bears (Usher 1976).

During the early 1980s, radio-collared individuals were tracked from the Canadian portion of the Southern Beaufort Sea into the eastern Chukchi Sea of Alaska (Taylor, 1987c; Amstrup et al., 1986; Amstrup & DeMaster, 1988). Telemetry data combined with re-sightings of tagged individuals suggested that bears of the Southern Beaufort Sea comprised a single subpopulation with an eastern boundary between Paulatuk and Baillie Island, NWT, Canada, and a western boundary near Icy Cape, Alaska (Taylor, 1987c; Amstrup et al., 1986; Amstrup and DeMaster, 1988; Stirling et al., 1988). Recognition that bears were shared by Canada and Alaska prompted the *Polar Bear Management Agreement for the Southern Beaufort Sea* (the *Agreement*) (Treseder and Carpenter, 1989; Nageak et al., 1991). The *Agreement*, between the Inupiat hunters of Alaska and the Inuvialuit hunters of Canada, was ratified by both parties in 1988. The *Agreement* included provisions to protect bears in dens and females with cubs, and stated that the annual sustainable harvest from the Southern Beaufort Sea would be shared between the two jurisdictions. Harvest levels were to be reviewed annually in light of the best scientific information available (Treseder & Carpenter, 1989; Nageak et al., 1991). Brower et al. (2002) evaluated the effectiveness of the *Agreement* after the first 10 years and concluded that; overall, it had been successful in ensuring that the total harvest and the harvest of adult females remained within what were thought to be sustainable limits. The Inuvialuit and Inupiat harvest quota from SB was 80 bears but the Joint Commissioner recommended a reduction to 70 in 2013 while NB/SB boundary changes were under discussion between co-management authorities.

Taylor et al. (1987c) and Amstrup et al. (1986) estimated the size of the SB subpopulation to be approximately 1800 bears, with a minimum and maximum of 1300 and 2500 bears, respectively. Amstrup et al. (2001), claiming the previous estimates were unreliable, estimates the total SB subpopulation to number to be >2500 as of 1998. However, Regehr et al. (2006) recalculates that the Amstrup et al. (2001) estimate was actually 2185, and provides a current (2006) SB estimate of 1526 (SE=164) polar bears, and concludes that no trend could be determined. The no-trend finding stands in contrast to PVA analyses that suggest the SB is declining (Hunter et al., 2010; Table 3).

Rates of survival and recruitment have recently been developed for bears of the Southern Beaufort Sea (Regehr et al., 2006, 2010; PBTC, 2007; COSEWIC, 2008; Obbard et al., 2010; S2 and S3). Based on published recruitment and survival rates, the combined U.S.-Canadian harvest of bears in the Southern Beaufort Sea would cause the subpopulation to decline (Hunter et al., 2010; Tables 2a and 2b). Our simulations indicate that if the harvest was reduced to zero, the SB subpopulation would continue to decline to extirpation (Table 4). This perspective is not consistent with current TEK or management perspectives for this subpopulation.

After consultation with the Inupiat, affected Inuvialuit communities and affected Nunavut Kitkmeot Inuit, the NB-SB boundary was shifted west to 133^{o} and the Regehr et al. (2006) SB subpopulation estimate was reduced by 310 individuals to about 1200. A new SB quota of 56 was identified based on the historical estimate of maximum sustained yield (MSY) of 4.5% per year at a 33% female and 67% male sex ratio. This change brings the SB quota into agreement with the Regehr et al. (2006) subpopulation estimate that was less than (but not significantly different from) the previous (Amstrup et al. 1986) estimate of 1800 by 274 individuals. The 2013 PBTC status table (PBTC, 2013) designates the SB subpopulation as stable based on TEK.

Additionally, the use of the historical MSY index suggests that user groups feel there has been no decline in SB subpopulation productivity. The boundary and quota change occurred after the 2012/2013 harvest season so our analyses used the 2006 subpopulation estimate (Regehr et al., 2006).

Southern Hudson Bay (SH)

Both coastal surveys (Stirling et al., 2004) and Inuit hunting in the SH subpopulation of polar bears reported an increase in the number of bears that have historically occurred in the area (McDonald et al., 1997). The offshore islands of eastern Hudson Bay apparently did not have any bears 50 years ago, and the species was rare around Inukjuak, only appearing "recently" (McDonald et al., 1997). Similarly, in Sanikiluaq, it was rare to kill a polar bear in the 1960s but now the community's annual quota is filled in approximately three weeks, with increased observations of bears coming into the community (L. Arragutainaq, pers. comm. 2006). In 1986, Crête et al. (1991) found relatively high numbers of bears on the Twin Islands in James Bay during the ice-free season. Cree in western James Bay report increased aggressiveness among bears and an increase in litter size (McDonald et al., 1997). Communities along the Hudson Bay and James Bay coasts in Ontario report an increase in bear encounters and property damage caused by polar bears (M. Carpenter, pers. comm. 2006; A. Solomon, pers. comm. 2006; P. Kapashesit, pers. comm. 2006; COSEWIC, 2008). In the past five years, polar bears have also been observed to travel more frequently during the open water season all the way to the Moosonee area of southern James Bay (approximately one sighting per year). Previously, bears were observed around Moosonee roughly once in five or six years (P. Kapashesit, pers. comm. 2006; A. Solomon, pers. comm. 2006; COSEWIC, 2008). Explanations offered for observations of higher numbers of bears include potential immigration of bears in response to increased ringed seals in the region, extended ice coverage in the area, and under-harvesting by the four aboriginal groups (Nunavut Inuit, Ontario Cree, and Quebec Inuit and Quebec Cree) of the SH subpopulation.

Boundaries of the SH subpopulation of polar bears are currently based on data from movements of marked bears of all sexes and telemetry studies of females (Jonkel et al., 1976; Kolenosky & Prevett, 1983; Kolenosky et al., 1992; Taylor & Lee, 1995). Crompton (2004) suggests that the current boundaries that define the SH subpopulation may need to be revisited, as she observed at least three breeding groups in the southern portion of Hudson Bay (including James Bay). However, the notion of three distinct breeding groups in such a small area devoid of any barriers to movements is inconsistent with other genetic information (Paetkau et al., 1999) and the great mobility of polar bears (COSEWIC, 2008).

Obbard et al. (2007) indicates there has been no significant decline in abundance of polar bears in SH since the 1980s. Obbard et al. (2007) estimated abundance of polar bears in SH as 641 (95% CI: 401–881) in 1986 and 681 (95% CI: 401–961) in 2005. These estimates are lower than previously stated for the SH (e.g., 1000 bears). Obbard et al. (2007:6) states that "the goodness of fit analysis did not detect any over-dispersion in the SH data nor any heterogeneity that could not be explained with covariates". However, the 2009 PBSG status report (Obbard et al., 2010) states that the 1986 estimate was 634 (95% CI 390-878) and the 2005 estimate was 673 (95% CI: 396-950), and cites Obbard et al. (2007) as the reference. In contrast to the Obbard et al. (2007) determination that the estimates were accurate, the PBSG status report (Obbard et al., 2010) states that 2007 estimates are likely an underestimate because of lack of complete coverage of the subpopulation (e.g., only coastal coverage on Ontario's north coast and no coverage in James Bay). Obbard et al. (2007) and the PBSG 2009 status report (Obbard et al.,

2010) lists the current SH estimate as 900 individuals. We find these "corrections" to be unsupported and arbitrary. However, taking the middle of the range given for James Bay as: 90 (SE = 10), and adding this value to the "accurate" Obbard et al. (2007) estimate for north coastal Ontario (681, SE = 280) yields 771 (SE = 300). The SH subpopulation is listed as stable in both the COSEWIC (2008) status report and the PBSG 2009 status report (Obbard et al., 2010) essentially on the strength of no trend between the 1985 and 2005 north coastal M-R estimates. When considered in a PVA context neither the subpopulation estimates nor the survival rate estimates from either of these studies are consistent with "stable" status for SH polar bears at existing harvest levels, suggesting that any perspective on trend drawn from these data is speculative.

Stirling et al. (2004), in their recent analysis of coastal survey data, suggested that the abundance of polar bears in SH has been increasing in recent years, although numbers in WH have remained about constant. A decline in body condition in the WH polar bear subpopulation has been correlated to a decline in sea ice, survival, and recruitment (Stirling et al., 1999; Regehr et al., 2007a, 2007b). A decline in body condition was also documented for the SH subpopulation when comparing bears captured in 1984–1986 with those captured in 2000–2004 (Obbard et al., 2006; PBTC, 2006; Obbard et al., 2007); however, unlike the WH subpopulation; there has been no concurrent decline in sea ice, survival or recruitment estimates (Obbard et al., 2007). Increased density may be the cause of the decline in condition; however, that notion is not supported by the subpopulation estimates which, in contrast to Stirling et al. (2004), show no trend. Although overall abundance in SH appears to have been stable in at least north coastal Ontario since the 1980s, both the 1986 and 2005 subpopulation estimates are ambiguous because only portions of the SH subpopulation summer retreat area were sampled. Thus both survival and subpopulation estimates may be biased and low, including the summed value for subpopulation numbers listed above.

A 2012 aerial survey of the SH subpopulation (Obbard & Stapleton, 2013) estimated the subpopulation to number 969 (SE=202), suggesting that mark recapture results under-estimate subpopulation number and survival rates, and thus under-estimate subpopulation trend.

Viscount Melville Sound (VM)

Only in the past 30 years have polar bears of the Viscount Melville Sound experienced modern levels of hunting pressure. Farquharson (1976) noted that by the mid-1970s, hunters from the Holman area had expanded their traditional hunting range to kill polar bears along the western and northern coasts of Victoria Island to Wyanniatt Bay. At the same time, Inuit from Cambridge Bay began travelling by land or air to reach northern Victoria Island to hunt polar bears in Hadley Bay. In response to increased interest in hunting bears of the VM subpopulation, the Government of the Northwest Territories established quotas. When quotas were originally allocated in the 1970s, the size and productivity of the VM subpopulation was overestimated. Polar bear density is lower in VM compared to other regions because of large expanses of multi-year ice and low densities of ringed seals (Kingsley et al., 1985). The consequence of overestimating abundance when quotas were first established was substantial over-harvest of bears in the region during the 1980s and early 1990s (e.g., 1985–1990 mean of 19.6 bears per year; Taylor et al., 2002).

A five-year moratorium on hunting was enacted in 1994/1995. Hunting resumed in 1999/2000 with an annual quota of four bears. In 2004/2005 the annual quota was increased to seven bears per year (Northwest Territories 4, Nunavut 3) to accommodate hunters on both sides of the new territorial border. Polar bear numbers in the VM are anticipated to still be increasing with this increase in quotas. The current (increased quota) kill is less that the sustainable yield using the 1996 subpopulation estimates with for the PVA (Table 2a).

A five-year study of movements and size of the VM subpopulation of polar bears using satellite telemetry and mark and recapture sampling was completed in 1992 (Messier et al., 1992, 1994; Taylor et al., 2002). Current boundaries are based on observed movements of females with satellite radio-collars and movements of bears tagged inside and outside of the study area (Bethke et al., 1996; Taylor et al., 2001a). The published 1996 abundance estimate of 215 bears (SE = 58) in Taylor et al. (2002) was based on the 1993 estimate plus three years of simulated subpopulation growth. Polar bears in VM are likely to benefit from a warming climate (at least over the short-term), which may increase the abundance and accessibility of seals by reducing amounts of multi-year ice.

Western Hudson Bay (WH)

The distribution, abundance, and boundaries of the WH subpopulation of polar bears have been studied since the late 1960s (e.g., Stirling et al., 1977; Derocher & Stirling, 1990, 1992, 1995a, 1995b; Taylor & Lee, 1995; Lunn et al., 1997, 2006). Between 60–80% of adults have been marked at any given time and there are extensive records from mark recapture studies and the return of tags from bears killed by Inuit hunters, and from the ongoing and long-term Polar Bear Alert Program of the Government of Manitoba. This subpopulation appears to be geographically segregated during the open-water season, although it mixes with those of SH and FB on the Hudson Bay sea ice during the winter and spring (Stirling et al., 1977; Derocher & Stirling, 1990; Stirling & Derocher, 1993; Taylor & Lee, 1995; Stirling et al., 2004).

Nirlungayuk (G. Nirlungayuk, pers. comm., 2008; COSEWIC, 2008) summarizes Nunavut Inuit TEK that polar bear numbers in the areas of Western Hudson Bay are considerably higher at the present time than any time previously. One explanation for the increase is supplemental feeding by garbage (around Churchill). For example, prior to the increase in the population of humans in Churchill in the 1940s, polar bears were best hunted nearer Wager Bay, Southampton Island, and Coates Island. After polar bear hunting regulations were initiated in 1968, if people wanted to be guaranteed a polar bear, they would travel down to south of Arviat. Observing polar bears by boat in Western Hudson Bay was once a rare event; now "lots of bears are there." Concurrently but contrary to the scientific M-R re-assessment of abundance (below), Inuit along the western coast of Hudson Bay recently reported seeing greater numbers of polar bears, which they interpreted as evidence of an increasing subpopulation (McDonald et al., 1997; Dowsley & Taylor, 2006b). Polar bears have been reported as "numerous" at Chesterfield Inlet in September and have been increasing in that area since 1988. Bears have been present for several years near Arviat, from September to December, but have recently increased in number according to TEK, especially in September.

The dangers posed by on-shore polar bears in the region are a concern to Inuit, and numbers are believed to be "too high" by some people (G. Nirlungayuk, pers. comm. 2008; COSEWIC, 2008). Encounters in the region have increased through the 1970s and 1980s. Since the 1980s, Arviat has been warning hunters to not to go out alone to ensure safety from bears. Nunavut Tunngavik Incorporated (NTI) recently collaborated with five experienced hunters from communities in Western Hudson Bay to complete a series of interviews and a workshop (NTI, 2005). TEK indicates an increasing number of bears in the Arviat area since the 1970s and around Whale Cove and Rankin Inlet since the 1980s. This increase in numbers has also been noted by Inuit of Chesterfield Inlet. In Arviat, the recent increase has been noted in all seasons except winter, while Inuit of other areas report an increase in all seasons. In the Chesterfield Inlet area, groups (gatherings) of polar bears have been observed recently, something that was apparently rare in the past.

Over the past 30 years, the condition of adults and the proportion of independent yearlings captured during the open-water season have declined significantly in the WH subpopulation (Derocher & Stirling, 1992,1995b; Stirling & Lunn, 1997; Stirling et al., 1999; COSEWIC, 2008). Over the same period, the average date of break-up of the sea ice has advanced by three weeks (Stirling et al., 1999, 2004; Ferguson et al., 2005). Stirling et al. (1999) documented that the earlier the timing of break-up, the poorer the condition of adult females. Inuit confirm the scientific information on changes in sea ice in Western Hudson Bay (G. Nirlungayuk, pers. comm. 2008; COSEWIC, 2008).

The number of polar bears in the WH subpopulation was recently estimated by Regehr et al. (2007a, 2007b). Regehr et al. (2007a, 2007b) estimated that abundance has declined from 1194 (95% CI = 1020–1368) to 935 (95% CI = 794-1076) between 1987 and 2004, a reduction of approximately 22%. Progressive declines in the condition and survival of cubs, subadults, and bears 20 years of age and older contributed to the decline in the size of the subpopulation. Once the subpopulation productivity began to decline, the existing harvest was no longer sustainable, and thus contributed to the reduction in the size of the WH subpopulation. The harvest sex ratio of 2M:1F in WH has resulted in a sex ratio that is 58% female and 42% male (Derocher et al., 1997b, Taylor et al., 2008d).

In summer 2007, the Government of Nunavut conducted a M-R survey of bears from Churchill to Chesterfield Inlet to determine whether or not there were large numbers of bears along the Kivalliq coast during the summer as suggested by TEK (Peacock & Taylor, 2007). The survey included those areas identified by TEK as being areas where polar bears were becoming more common. A total of 25 bears were captured during the three-day survey that ended north of the CWS study area. The proportion of marked individuals in the capture sample (p = 0.46, SE = 0.11) was lower but not statistically different from the proportion of marked animals in the Canadian Wildlife Service (CWS) capture sample (p = 0.59, SE = 0.01). However, the mean time since marked bears were last handled was significantly greater for marked polar bears captured outside the CWS "study area". Statistical power was low for this analysis due to the small sample size and because the capture effort only extended to the mouth of the Seal River (~59 degrees latitude). These results suggest that actual numbers of bears in WH and annual survival rates could thus be higher (due to un-modelled heterogeneity) than estimated by Regehr et al. (2007a, 2007b). Peacock and Taylor (2007) recommend that in future years, CWS capture teams work north to Arviat to capture polar bears in the entire area where polar bears summer.

Climate change reductions to sea ice with consequent reductions in body condition, survival and recruitment and resulting over-hunting is considered to be the major threat to the WH subpopulation. The subpopulation is believed to be declining at a substantial rate (Regehr et al., 2007a, 2007b; Table 2b), and the quota for hunting polar bears in WH was reduced to eight animals in 2008–2009; however, PVA analysis using demographic rates from Regehr et al. (2007a, 2007b) indicate that the WH subpopulation will decline with no removals for control or harvest.

TEK perspectives and scientific perspectives are qualitatively different for WH. Currently the WH subpopulation is the only subpopulation of 19 circumpolar subpopulations where a decline due to climate warming can be demonstrated (Regehr et al., 2007a, 2007b; Obbard et al., 2010). An aerial survey covering the entire WH summer retreat area was undertaken in fall of 2011. The result of the 2011 aerial survey (N=1013, SE=151) confirms

Inuit knowledge that this subpopulation has not declined and may have increased (Atkinson et al., 2012). Independent of the aerial survey results, it does appear that body condition, survival rate, and recruitment rates have declined. An alternative hypothesis to climate mediated decline in sea ice as the main cause for reduced subpopulation productivity is density effects as suggested by the asymptotic behaviour of the WH abundance index provided by annual beach transects (Stirling et al., 2004).

S1 Table S1. Baffin Bay (BB) mortality rates were applied to this RISKMAN stable-age distribution as a consistency check between the Peacock et al., 2012 estimated survival rates and the BB reported harvest (S1 Table S2). The BB RISKMAN stable age-distribution uses the 1997 estimate of abundance (N=2074, SE=265) from Taylor et al., 2005.

Age	Males	Females Unencumbered	Females Encumbered					
0	161.9169	166.5149						
1	100.9874	104.4790						
2	63.04853	65.6201						
3	52.61324	52.8113						
4	43.90512	38.4224	1.6851	2.3951				
5	36.63829	4.3424	11.2512	15.9915	1.7609	0.8602		
6	35.5692	5.9151	1.7174	2.4410	13.9899	6.8340	1.4919	0.3676
7	34.5313	5.9722	3.0748	4.3703	2.1355	1.0432	11.8523	2.9206
8	33.5237	2.2277	8.2047	11.6614	3.8233	1.8677	1.8092	0.4458
9	32.5455	5.2515	1.7729	2.5198	10.2017	4.9835	3.2391	0.7982
10	31.5959	4.5986	3.6737	5.2215	2.2044	1.0769	8.6430	2.1298
11	30.6739	2.5334	6.0794	8.6407	4.5679	2.2314	1.8676	0.4602
12	29.7789	4.5322	1.9226	2.7326	7.5591	3.6926	3.8700	0.9536
13	28.9100	3.6923	3.7002	5.2591	2.3906	1.1678	6.4042	1.5781
14	28.0664	2.6146	4.6173	6.5626	4.6009	2.2475	2.0253	0.4991
15	27.2474	3.8603	2.0324	2.8887	5.7412	2.8045	3.8979	0.9605
16	26.4524	3.0721	3.4482	4.9010	2.5272	1.2345	4.8640	1.1985
17	25.6805	2.5471	3.6127	5.1348	4.2875	2.0944	2.1410	0.5276
18	24.9312	3.2745	2.0628	2.9319	4.4921	2.1944	3.6324	0.8951
19	24.2037	2.6248	3.0857	4.3857	2.5649	1.2530	3.8057	0.9378
20	23.4974	2.3914	2.9142	4.1419	3.8368	1.8742	2.1730	0.5355
21	22.8118	2.7811	2.0170	2.8668	3.6235	1.7701	3.2505	0.8010
22	22.1462	2.2825	2.7023	3.8408	2.5080	1.2251	3.0699	0.7565
23	21.5000	2.1910	2.4160	3.4339	3.3600	1.6414	2.1248	0.5236
24	20.8726	2.3718	1.9140	2.7203	3.0041	1.4675	2.8467	0.7015
25	20.2636	2.0054	2.3413	3.3278	2.3798	1.1625	2.5451	0.6271
26	19.6723	1.9748	2.0478	2.9105	2.9112	1.4221	2.0162	0.4968
27	19.0983	2.0337	1.7749	2.5227	2.5462	1.2438	2.4664	0.6078
28	18.5410	1.7712	2.0202	2.8713	2.2069	1.0781	2.1572	0.5316
29	18.0000	1.7606	1.7639	2.5070	2.5119	1.2271	1.8697	0.4607
30	17.4747	1.7534	1.6180	2.2997	2.1932	1.0714	2.1281	0.5244

S1 Table S2. Baffin Bay (BB) mortality rates (natural, total, and harvest) were calculated based on the natural and total survival rates reported in Peacock et al., 2012. The application of BB mortality rates to a RISKMAN stable-age distribution (S1 Table S1) suggested an inconsistency between the Peacock et al., 2012 estimated survival rates and the BB reported harvest (S1 Tables S3; S4 Tables S1a, S1b, and S1c). For comparison, the average BB harvest for the 2003-2009 period was ~215 bears (S1 Table S3), while the estimated annual harvest removal from Peacock et al., 2012 was only ~25 bears per year.

Age Class	**Total Bears**	**Natural Mortality Rate**	**Natural Mortality**	**Total Mortality Rate**	**Total Mortality**	**Harvest Mortality Rate**	**Harvest Mortality**
Juvenile Male	262.9043	0.1170	30.7598	0.1440	37.8582	0.0270	7.0984
Juvenile Female	270.9939	0.1290	34.9582	0.1410	38.2101	0.0120	3.2519
Subadult Male	159.5669	0.2410	38.4556	0.2980	47.5509	0.0570	9.0953
Subadult Female	160.9340	0.2680	43.1303	0.2940	47.3146	0.0260	4.1843
Adult Male	674.2259	0.3340	225.1915	0.3350	225.8657	0.0010	0.6742
Adult Female	545.3742	0.3300	179.9735	0.3310	180.5189	0.0010	0.5454
Total	2073.9990	N/A	552.4689	N/A	577.3184	N/A	24.8495

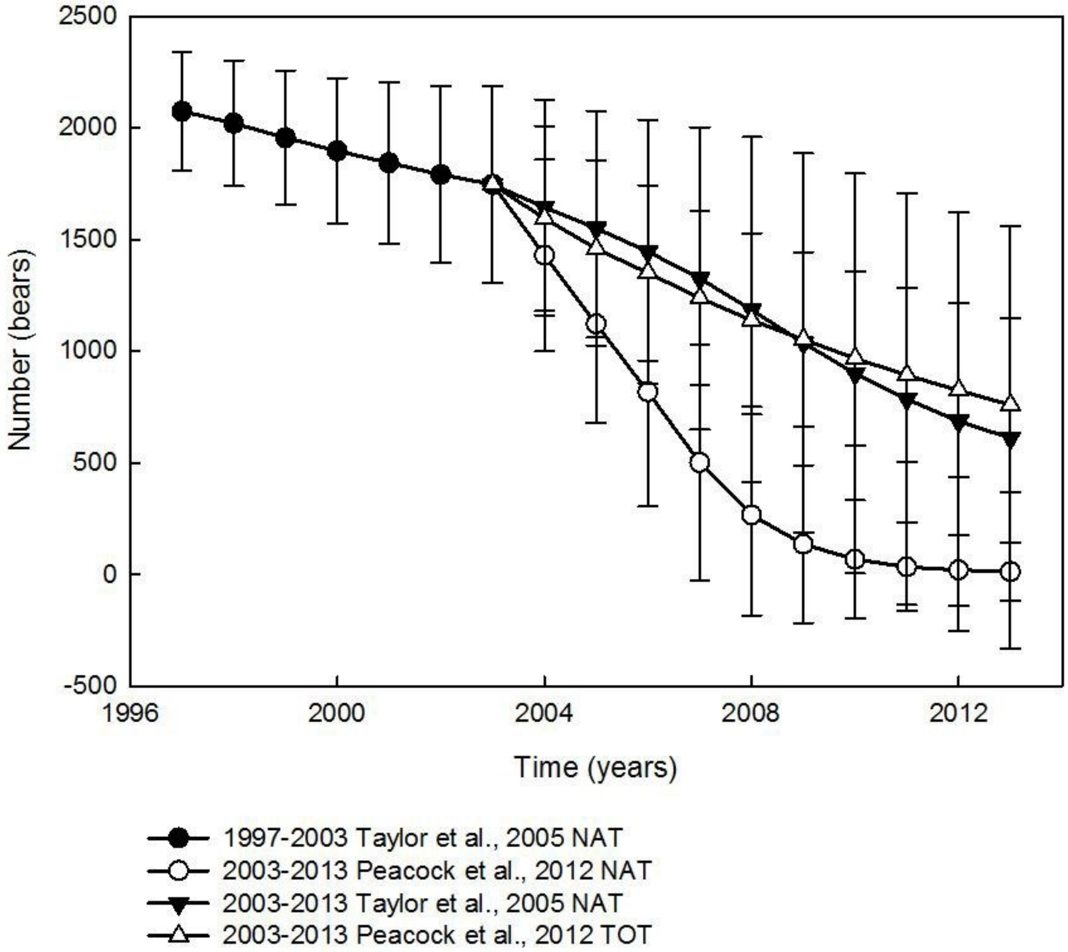
S1 Table S3. Baffin Bay (BB) marked/unmarked bears by jurisdiction (Nunavut and Greenland) for the 1998-2009 period. BB harvest statistics were taken from the PBTC harvest records and PBSG status reports (Lunn et al., 2002; Aars et al., 2006; Obbard et al., 2010).

Year	Nunavut Marked	Nunavut Unmarked	Nunavut Total	Greenland Marked	Greenland Unmarked	Greenland Total
1998	14	83	97	7	89	96
1999	5	59	64	14	83	97
2000	7	44	51	4	64	68
2001	8	54	62	2	95	97
2002	5	58	63	3	115	118
2003	5	56	61	6	200	206
2004	3	69	72	5	159	164
2005	7	90	97	2	153	155
2006	6	92	98	2	133	135
2007	8	91	99	4	75	79
2008	6	93	99	3	63	66
2009	6	97	103	7	66	73

S1 Table S4. A Fisher's Exact Test comparison of Nunavut versus Greenland Baffin Bay recoveries for the 1998-2000, 1998-2001, 1998-2002, and 2003-2009 time bins. The results suggested that recoveries were not drawn from the same subpopulation for the intervals 1998-2002 ($p < 0.0103$) and 1998-2001 $p < 0.0424$; but were drawn from the same subpopulation for the 1998-2000 interval ($p<0.3579$, power=$(1-\beta)=0.67$).

1998-2000: p<0.3579, power=(1-β)=0.67		
Jurisdiction	Marked	Unmarked
Nunavut	26	186
Greenland	25	236
1998-2001: p<0.0424		
Jurisdiction	Marked	Unmarked
Nunavut	34	240
Greenland	27	331
1998-2002: p<0.0103		
Jurisdiction	Marked	Unmarked
Nunavut	39	298
Greenland	30	446
2003-2009 p<0.0041		
Jurisdiction	Marked	Unmarked
Nunavut	41	588
Greenland	29	849

S1 FIG. S1. Baffin Bay (BB) subpopulation trajectories from 1997-2013. A comparison of the effect of different BB survival rates (Taylor et al., 2005 [Natural]; Peacock et al., 2012 [Natural and Total]) for producing a subpopulation trajectory for the 2003-2013 period.



Supplementary 2. Canadian Polar Bear Subpopulation Survival Rates.

Age specific survival rates vary among subpopulations. Polar bear survival rates have been estimated using the following age strata definitions: cubs-of-the-year (COYs), yearlings and subadults (ages 1–4), prime-age adults (ages 5–20), and senescent adults (ages 21-30); except for (Western Hudson Bay) where Regehr et al. (2007a, 2007b) defined prime-age adults as ages 5-19 and senescent adults as ages 20-30. Survival rates by age strata class for each Canadian subpopulation are provided in S2 Tables S1 and S2. Total survival rates (S2 Table S1) include harvest mortality and are calculated without correction for harvest levels in some estimates. Natural survival rates (S1 Table S2) are corrected for harvest mortality and are the values used to determine the effect of a given harvest rate on subpopulation growth rate in a PVA simulation.

S2 Table S1. Mean (standard error [SE] in parentheses) of total (i.e., harvested) annual survival rates for age and sex classes of subpopulations of Canadian polar bears.

	Males					**Females**				
Subpopulation	**Total Survival / SE**					**Total Survival / SE**				
(primary data source)	**0**	**1**	**2 - 4**	**5 – 20**	**>20**	**0**	**1**	**2 - 4**	**5 - 20**	**>20**
Baffin Bay (Taylor et al., 2005)	0.538	0.879	0.879	0.923	0.874	0.600	0.901	0.901	0.940	0.913
	(0.094)	(0.049)	(0.049)	(0.024)	(0.062)	(0.096)	(0.045)	(0.045)	(0.021)	(0.047)
Davis Strait (Peacock et al., 2013)[1]	0.538	0.879	0.879	0.923	0.874	0.600	0.901	0.901	0.940	0.913
	(0.094)	(0.049)	(0.049)	(0.024)	(0.062)	(0.096)	(0.045)	(0.045)	(0.021)	(0.047)
Foxe Basin[2]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Gulf of Boothia (Taylor et al., 2008c)	0.8889	0.883	0.883	0.917	0.917	0.889	0.883	0.883	0.919	0.919
	(0.179)	(0.087)	(0.087)	(0.041)	(0.041)	(0.179)	(0.087)	(0.087)	(0.044)	(0.044)
Kane Basin (Taylor et al., 2008a)	0.308	0.617	0.617	0.957	0.957	0.374	0.686	0.686	0.967	0.967
	(0.172)	(0.180)	(0.180)	(0.046)	(0.046)	(0.180)	(0.157)	(0.157)	(0.043)	(0.043)
Lancaster Sound (Taylor et al., 2008b)[3]	0.633	0.790	0.790	0.892	0.653	0.749	0.879	0.879	0.936	0.758
	(0.123)	(0.073)	(0.073)	(0.030)	(0.085)	(0.105)	(0.050)	(0.050)	(0.019)	(0.054)
M'Clintock Channel (Taylor et al., 2006a)	0.620	0.900	0.900	0.880	0.880	0.620	0.900	0.900	0.900	0.900
	(0.150)	(0.040)	(0.040)	(0.040)	(0.040)	(0.150)	(0.040)	(0.040)	(0.040)	(0.040)
Northern Beaufort Sea (Stirling et al., 2011)[4]	0.516	0.328	0.823	0.825	0.401	0.537	0.333	0.905	0.906	0.575
	(0.349)	(0.311)	(0.148)	(0.145)	(0.304)	(0.285)	(0.314)	(0.094)	(0.092)	(0.283)
Norwegian Bay (Taylor et al., 2008b)[3]	0.633	0.790	0.790	0.892	0.653	0.749	0.879	0.879	0.936	0.758
	(0.123)	(0.073)	(0.073)	(0.030)	(0.085)	(0.105)	(0.050)	(0.050)	(0.019)	(0.054)
Southern Beaufort Sea (Regehr et al., 2006; 2010)	0.430	0.920	0.920	0.920	0.920	0.430	0.920	0.920	0.920	0.920
	(0.110)	(0.040)	(0.040)	(0.040)	(0.040)	(0.110)	(0.040)	(0.040)	(0.040)	(0.040)
Southern Hudson Bay (Obbard et al., 2007)[5]	0.492	0.644	0.811	0.811	0.293	0.485	0.645	0.892	0.892	0.444
	(0.141)	(0.141)	(0.075)	(0.075)	(0.132)	(0.133)	(0.133)	(0.051)	(0.051)	(0.146)
Viscount Melville Sound (Taylor et al., 2002)	0.448	0.774	0.774	0.774	0.774	0.693	0.905	0.905	0.905	0.905
	(0.216)	(0.081)	(0.081)	(0.081)	(0.081)	(0.183)	(0.026)	(0.026)	(0.026)	(0.026)
Western Hudson Bay (Regehr et al., 2007a, 2007b)[6,7]	0.620	0.620	0.810	0.900	0.750	0.700	0.700	0.860	0.930	0.810
	(0.020)	(0.020)	(0.015)	(0.005)	(0.020)	(0.020)	(0.020)	(0.015)	(0.005)	(0.015)

[1] Davis Strait uses Baffin Bay total survival values due to meta-analysis. See comments.

[2] No survival rates are available for Foxe Basin.

[3] Survival rates pooled for Lancaster Sound and Norwegian Bay (see Taylor et al., 2008b).

[4] 2003–2005 means. Estimated SE is the difference between the mean estimate and mean upper confidence limit, divided by 1.96.

[5] 2003-2004 means. Estimated SE is the confidence interval, divided by 3.92. Results may differ from COSEWIC (2008) due to errors found in correspondence with Obbard et al. (2007).

[6] Regehr et al. (2007a, 2007b) present total apparent survival rates for Western Hudson Bay polar bears as 95% CI. Estimated SE is the difference between the estimate and upper CL, divided by 1.96. Survival rates presented for 2-4 and 20+ adults are those that are not reduced from capture events around Churchill (see Regehr et al. [2007a, 2007b]). Survival rates for 2-4 and ≥20 age categories in Western Hudson Bay may be as low as 0.72 and 0.65 for males and 0.78 and 0.72 for females, respectively. The true survival rates for subadult and senescent bears in Western Hudson Bay likely lie somewhere between the rates in the table and those stated in the previous sentence (pers. comm..Regehr, E. 2007; COSEWIC, 2008).

[7] Age strata classes from Regehr et al. (2007a, 2007b) differ from the other papers, adhering to the following groupings: 0-1, 2-4, 5-19, and 20-30.

S2 Table S2. Mean (standard error [SE] in parentheses) of natural (i.e., unharvested) annual survival rates for age and sex classes of subpopulations of Canadian polar bears.

	Males					**Females**				
Subpopulation	**Natural Survival / SE**					**Natural Survival / SE**				
(primary data source)	**0**	**1**	**2 - 4**	**5 – 20**	**>20**	**0**	**1**	**2 - 4**	**5 - 20**	**>20**
Baffin Bay (Taylor et al., 2005)	0.570	0.938	0.938	0.947	0.887	0.620	0.938	0.938	0.953	0.919
	(0.094)	(0.045)	(0.045)	(0.022)	(0.060)	(0.095)	(0.042)	(0.042)	(0.020)	(0.046)
Davis Strait (Peacock et al., 2013)	0.916	0.934	0.923	0.955	0.897	0.916	0.934	0.931	0.962	0.911
	(0.057)	(0.032)	(0.034)	(0.020)	(0.073)	(0.057)	(0.032)	(0.033)	(0.019)	(0.070)
Foxe Basin (Taylor et al., 2005)[1]	0.570	0.938	0.938	0.947	0.887	0.620	0.938	0.938	0.953	0.919
	(0.094)	(0.045)	(0.045)	(0.022)	(0.060)	(0.095)	(0.042)	(0.042)	(0.020)	(0.046)
Gulf of Boothia (Taylor et al., 2009)	0.889	0.897	0.897	0.955	0.955	0.889	0.897	0.897	0.955	0.955
	(0.179)	(0.078)	(0.078)	(0.036)	(0.036)	(0.179)	(0.078)	(0.078)	(0.035)	(0.035)
Kane Basin (Taylor et al., 2008a)	0.345	0.663	0.663	0.997	0.997	0.410	0.756	0.756	0.997	0.997
	(0.200)	(0.197)	(0.197)	(0.026)	(0.026)	(0.200)	(0.159)	(0.159)	(0.026)	(0.026)
Lancaster Sound (Taylor et al., 2008b)[3]	0.634	0.838	0.838	0.974	0.715	0.750	0.898	0.898	0.946	0.771
	(0.123)	(0.075)	(0.075)	(0.030)	(0.095)	(0.104)	(0.005)	(0.005)	(0.018)	(0.054)
M'Clintock Channel (Taylor et al., 2006a)	0.620	0.983	0.983	0.977	0.977	0.619	0.983	0.983	0.921	0.921
	(0.150)	(0.034)	(0.034)	(0.033)	(0.033)	(0.151)	(0.034)	(0.034)	(0.046)	(0.046)
Northern Beaufort Sea (Stirling et al., 2011)[4]	0.457	0.930	0.892	0.872	0.676	0.443	0.930	0.958	0.932	0.584
	(0.481)	(0.040)	(0.156)	(0.152)	(0.351)	(0.344)	(0.040)	(0.098)	(0.094)	(0.323)
Norwegian Bay (Taylor et al., 2008b)[3]	0.634	0.838	0.838	0.974	0.715	0.750	0.898	0.898	0.946	0.771
	(0.123)	(0.075)	(0.075)	(0.030)	(0.095)	(0.104)	(0.005)	(0.005)	(0.018)	(0.054)
Southern Beaufort Sea (Regehr et al., 2006; 2010)[5]	0.430	0.930	0.930	0.930	0.930	0.430	0.930	0.930	0.930	0.930
	(0.11)	(0.040)	(0.040)	(0.040)	(0.040)	(0.11)	(0.040)	(0.040)	(0.040)	(0.040)
Southern Hudson Bay (Obbard et al., 2007)[6]	0.492	0.672	0.928	0.892	0.556	0.485	0.650	0.972	0.951	0.523
	(0.141)	(0.141)	(0.075)	(0.075)	(0.132)	(0.133)	(0.133)	(0.051)	(0.051)	(0.146)
Viscount Melville Sound (Taylor et al., 2002)	0.448	0.924	0.924	0.924	0.924	0.693	0.957	0.957	0.957	0.957
	(0.216)	(0.109)	(0.109)	(0.109)	(0.109)	(0.183)	(0.028)	(0.028)	(0.028)	(0.028)
Western Hudson Bay (Regehr et al., 2007a, 2007b)[7,8]	0.710	0.710	0.940	0.940	0.820	0.730	0.730	0.930	0.930	0.820
	(0.020)	(0.020)	(0.015)	(0.005)	(0.020)	(0.020)	(0.020)	(0.015)	(0.005)	(0.015)

[1] No survival rates are available for Foxe Basin. Natural survival rates from a nearby subpopulation (BB) have been substituted in order to perform the simulations.

[3] Survival estimates pooled for Lancaster Sound and Norwegian Bay (see Taylor et al., 2008b).

[4] Natural survival estimates for NB were estimated by: (1) subtracting the difference between NB total survival and NB natural survival from the 2008 COSEWIC status report (COSEWIC, 2008), then (2) subtracting this difference from the NB total survival estimates in Stirling et al. (2011).

[5] Natural survival estimates for SB were estimated by: (1) subtracting the difference between SB total survival and SB natural survival from the 2008 COSEWIC status report (COSEWIC, 2008), then (2) subtracting this difference from the SB total survival estimates in Regehr et al. (2006).

[6] Natural survival estimates for SH were estimated by: (1) subtracting the difference between SH total survival and SH natural survival from the 2008 COSEWIC status report, then (2) subtracting this difference from the SH 2003-2004 pooled total survival estimates in Obbard et al. (2007).

[7] Natural survival rates provided in Regehr et al. (2007a, 2007b) include two estimates for the 2-4 and 20+ age categories. One estimate is for the mark-recapture model that excludes a capture effect on mortality of handling bears in Churchill; the other estimate is rates that are reduced to reflect heterogeneity in the data associated with captures around Churchill by the Manitoba Department of Conservation. We used the former of these two estimates. Regehr et al. (2007a, 2007b) present no error estimates with these rates; for simulations the errors associated with total survival rates were used.

[8] Age classifications from Regehr et al. (2007a, 2007b) differ from the other papers, adhering to the following groupings: 0-1, 2-4, 5-19, and 20-30.

Supplementary 3. Canadian Polar Bear Recruitment Rates.

The three year reproduction cycle of polar bears requires a different parameterization than the usual annual birth-pulse lifetable m_x values. The time of census is the same as for survival estimates (S2). The mean and standard error of the proportion of males in a litter, litter size (includes males and females), and age-specific probabilities of litter production for available females are provided in S3 Table S1. Available females are females with no cubs or females with 2-year olds. Encumbered females with cubs of the year (COYs) or yearlings do not engage in breeding and do not produce litters the following year.

S3 Table S1. Estimated means (and standard errors [SE] in parentheses) of post-den emergence litter size and age-specific probabilities of litter production (LPR) for lone females or females with dispersing (2-year-old) cubs (because of the 3-year reproductive cycle of polar bears, females with cubs-of-the-year or yearlings are not available to mate and are not included in the LPR computation).

Subpopulation (primary data source)	Cub (age 0) litter size/ SE	Age 4 LPR/ SE	Age 5 LPR/ SE	Age 6 LPR/ SE	Age 7+ LPR/ SE	Proportion of Males at Birth/ SE
Baffin Bay (Taylor et al., 2005)	1.587	0.096	0.880	1.000	1.000	0.493
	(0.073)	(0.120)	(0.400)	(0.167)	(0.167)	(0.029)
Davis Strait (Peacock et al., 2013)[1]	1.487	0.069	0.543	0.338	0.441	0.550
	(0.140)	(0.064)	(0.203)	(0.976)	(0.107)	(0.040)
Foxe Basin (Taylor et al., 2005)[2]	1.587	0.096	0.880	0.850[3]	0.850[3]	0.493
	(0.073)	(0.120)	(0.400)	(0.167)	(0.167)	(0.029)
Gulf of Boothia (Taylor et al., 2008c)	1.648	0.000	0.194	0.467	0.970	0.460
	(0.098)	(0)	(0.178)	(0.168)	(0.300)	(0.090)
Kane Basin (Taylor et al., 2008a)	1.667	0.000	0.000	0.357	0.978	0.426
	(0.083)	(0)	(0)	(0.731)	(0.083)	(0.029)
Lancaster Sound (Taylor et al., 2008b)	1.688	0.000	0.107	0.312	0.954	0.531
	(0.012)	(0)	(0.050)	(0.210)	(0.083)	(0.048)
M'Clintock Channel (Taylor et al., 2006a)[4]	1.680	0.000	0.11	0.29	0.93	0.550
	(0.147)	(0)	(0.11)	(0.47)	(0.33)	(0.060)
Northern Beaufort Sea (PBTC, 2007)	1.756	0.118	0.283	0.883	0.883	0.502
	(0.166)	(0.183)	(0.515)	(0.622)	(0.622)	(0.035)
Norwegian Bay (Taylor et al., 2008b)	1.714	0.000	0.000	0.000	0.689	0.544
	(0.080)	(0)	(0)	(0)	(0.053)	(0.066)
Southern Beaufort Sea (Regehr et al., 2006)[5]	1.750	0.000	0.470	0.470	0.470	0.520
	(0.170)	(0)	(0.090)	(0.090)	(0.090)	(0.040)
Southern Hudson Bay (PBTC, 2007)[6]	1.575	0.087	0.966	0.967	0.967	0.467
	(0.116)	(0.202)	(0.821)	(0.022)	(0.022)	(0.086)
Viscount Melville Sound (Taylor et al., 2002)	1.640	0.000	0.623	0.872	0.872	0.535
	(0.125)	(0)	(0.414)	(0.712)	(0.712)	(0.118)
Western Hudson Bay (Aars et al., 2006; PBTC, 2007)[7]	1.540	0.000	0.257	0.790	0.790	0.480
	(0.110)	(0)	(0.442)	(0.180)	(0.180)	(0.110)

[1] DS reproductive rates were also provided through correspondence with Dr. Lily Peacock.

[2] No reproductive rates are available for Foxe Basin. Reproductive rates from a nearby subpopulation (BB) have been substituted in order to perform the simulations.

[3] Baffin Bay adult litter production rates were reduced from 1.0 to 0.85 to produce a simulation that resulted in ~2700 individuals in 2010 (to match the Garshelis et al. (2012) aerial survey estimate), and used these empirically corrected Baffin Bay demographic rates for the Foxe Basin PVA.

[4] Results may differ from COSEWIC (2008) due to errors found in correspondence with Taylor et al. (2006a).

[5] No mean LPR for an age category is presented in Regehr et al. (2007a, 2007b). Selected values provided by E. Regehr (USGS, Alaska Science Centre, Anchorage, AK) for the 2007 meeting of the PBTC.

[6] Also presented in Aars et al. (2006).

[7] Data presented in Table 3 of Aars et al. (2006), updated online version only.

Supplementary 4. Total Human-Caused Mortality Rates for Canadian Polar Bear Subpopulations.

Total anthropogenic mortalities (harvest, defense, accidental, and illegal) are monitored and reported for all subpopulations within or shared by Canada. The total anthropogenic mortality and the sex and age distribution of the harvest are available by harvest season for each Canadian polar bear subpopulation from the year of most recent estimate of abundance to the 2011/2012 harvest season. These are provided in S4 Tables S1**a**, S1**b**, and S1**c** (updated from York, 2012 Tables 4.5 and 4.6).

S4 Table S1a. Total anthropogenic (harvest, defense, accidental, and illegal) mortality rates (Kill) and the proportion that were females (Prop F) for each Canadian subpopulation are summarized by harvest season for the 1993/1994 to 1999/2000 interval (York, 2012; PBTC, 2013). Anthropogenic mortality rates are presented starting from the last published subpopulation estimate to the present; intervals relevant to our pre-2013 PVA simulations (see Methods).

Season	*93/94*		*94/95*		*95/96*		*96/97*		*97/98*		*98/99*		*99/00*	
Subpop.	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*
Baffin Bay									193	0.31	161	0.37	119	0.34
Davis Strait														
Foxe Basin	100	0.485	118	0.31	95	0.35	97	0.33	76	0.49	82	0.24	95	0.37
Gulf of Boothia													33	0.39
Kane Basin									11	0.27	11	0.27	10	0.3
Lancaster Sound									76	0.28	74	0.20	75	0.25
M'Clintock Channel													22	0.27
Northern Beaufort Sea														
Norwegian Bay									3	0.33	4	0.25	4	0.25
Southern Beaufort Sea														
Southern Hudson Bay														
Viscount Melville Sound											0	0	4	0.25
Western Hudson Bay														

S4 Table S1b. Total anthropogenic (harvest, defense, accidental, and illegal) mortality rates (Kill) and the proportion that were females (Prop F) for each Canadian subpopulation are summarized by harvest season for the 2000/2001 to 2006/2007 interval (York, 2012; PBTC, 2013). Anthropogenic mortality rates are presented starting from the last published subpopulation estimate to the present; intervals relevant to our pre-2013 PVA simulations (see Methods).

Season	*00/01*		*01/02*		*02/03*		*03/04*		*04/05*		*05/06*		*06/07*	
Subpop.	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*
Baffin Bay	159	0.43	181	0.29	267	0.33	236	0.33	252	0.35	233	0.32	178	0.35
Davis Strait														0.24
Foxe Basin	99	0.27	98	0.35	96	0.38	95	0.26	94	0.27	103	0.38	102	0.41
Gulf of Boothia	41	0.34	43	0.32	38	0.47	41	0.39	66	0.32	65	0.31	72	0.36
Kane Basin	11	0.36	10	0.3	10	0.3	8	0.38	11	0.36	25	0.2	9	0.22
Lancaster Sound	62	0.16	71	0.25	71	0.21	79	0.28	87	0.23	81	0.22	94	0.21
M'Clintock Channel	12	0.33	0	0	1	0	0	0	2	0.5	3	0	3	0.00
Northern Beaufort Sea														0.35
Norwegian Bay	4	0.25	1	0	1	0	3	0.33	4	0	3	0	4	0.00
Southern Beaufort Sea														0.34
Southern Hudson Bay											38	0.36	38	0.28
Viscount Melville Sound	4	0.5	4	0	4	0	5	0.2	5	0.4	4	0.25	6	0.33
Western Hudson Bay									43	0.30	37	0.44	58	0.31

S4 Table S1c. Total anthropogenic (harvest, defense, accidental, and illegal) mortality rates (Kill) and the proportion that were females (Prop F) for each Canadian subpopulation are summarized by harvest season for the 2007/2008 to the 2011/2012 interval (York, 2012; PBTC, 2013). Anthropogenic mortality rates are presented starting from the last published subpopulation estimate to the present; intervals relevant to our pre-2013 PVA simulations (see Methods). Five-year mean human-caused mortality rates are also presented for the post-2013 PVA simulations (see Methods).

Season	*07/08*		*08/09*		*09/10*		*10/11*		*11/12*		5-year mean	
Subpopulation	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*	*Kill*	*Prop F*
Baffin Bay	165	0.448	176	0.341	155	0.303	163	0.337	161	0.354	164.0	0.357
Davis Strait	68	0.235	76	0.368	69	0.377	86	0.395	107	0.402	81.2	0.356
Foxe Basin	107	0.626	109	0.284	109	0.367	107	0.374	112	0.366	108.8	0.403
Gulf of Boothia	56	0.411	72	0.403	57	0.439	45	0.400	69	0.275	59.8	0.385
Kane Basin	5	0.800	7	0.143	3	0.667	5	0.400	5	0.4	5.0	0.482
Lancaster Sound	74	0.257	94	0.383	73	0.370	84	0.238	98	0.306	84.6	0.311
M'Clintock Channel	3	0.333	2	0.000	3	0.000	3	0.333	3	0.333	2.8	0.200
Northern Beaufort Sea	18	0.389	34	0.471	13	0.538	45	0.311	52	0.346	32.4	0.411
Norwegian Bay	4	0.000	0	0.000	1	0.000	3	0.000	1	0	1.8	0.000
Southern Beaufort Sea	28	0.429	31	0.032	24	0.500	51	0.412	50	0.28	36.8	0.331
Southern Hudson Bay	34	0.294	37	0.351	62	0.387	104	0.346	49	0.265	57.2	0.329
Viscount Melville Sound	3	0.000	5	0.000	3	0.000	7	0.429	4	0.5	4.4	0.186
Western Hudson Bay	32	0.438	14	0.357	18	0.167	15	0.400	29	0.31	21.6	0.334

Supplementary 5. Effect of Truncated Iterations on Monte Carlo Estimates of Subpopulation Growth Rate (λ).

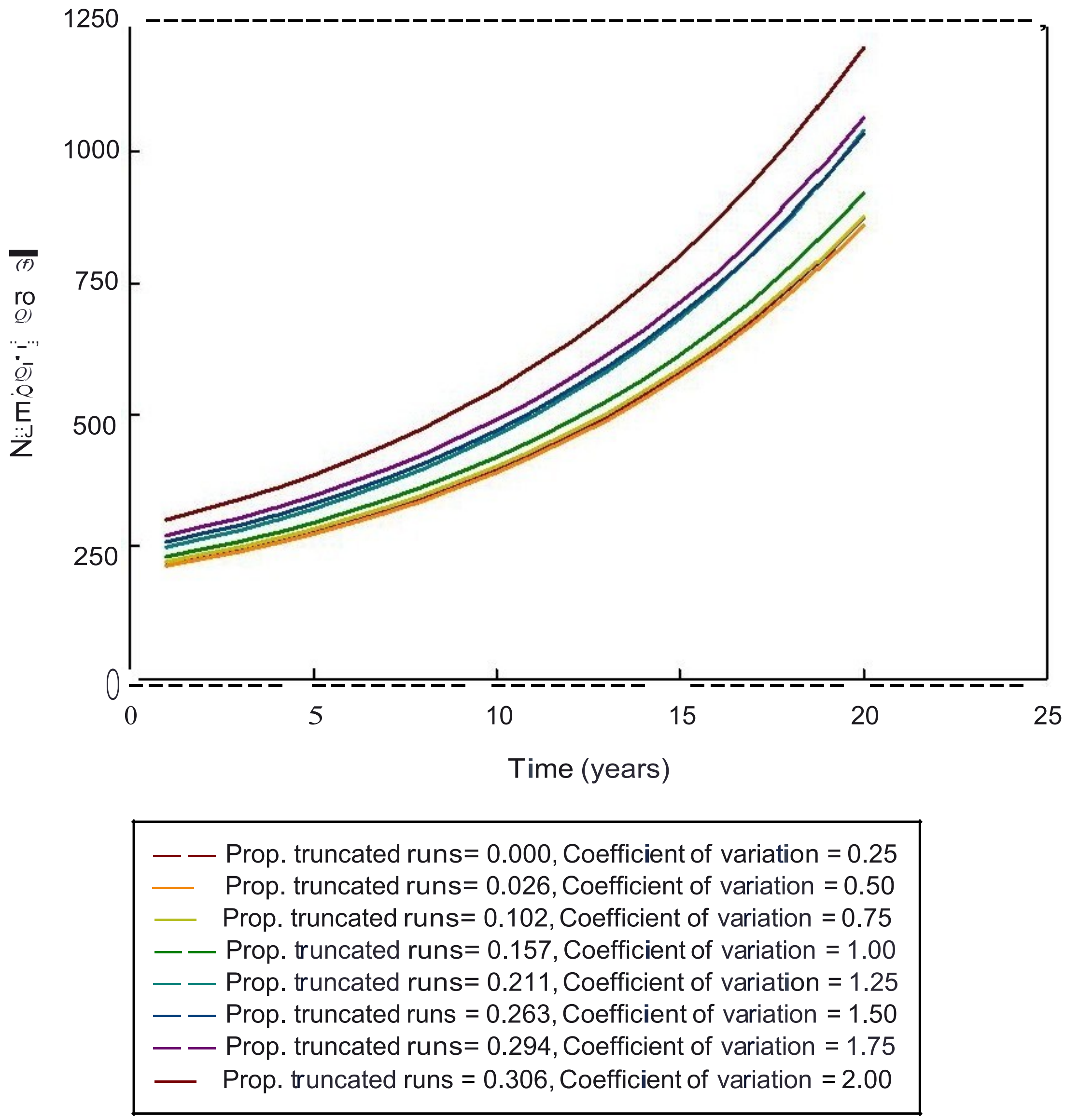
The simulation protocol for occasions when the initial subpopulation random deviate was ≤ 0, or the harvest could not be satisfied by the subpopulation, or all individuals remaining in the subpopulation did not survive (individual based model) was to set the simulation $N_t = 0$ for the remaining years (t). Subsequent estimates of subpopulation growth rate (λ) were undefined because the denominator of N_{t+1}/N_t was zero, so post-truncation λ was set to zero. This protocol caused Monte Carlo mean N_t to be over-estimated (positive bias) because the upper values were not bounded, but the lower values were bounded at 0 (S5 Fig. S1). Additionally, direct Monte Carlo estimates (mean of iteration values) of both arithmetic and geometric λ^t were biased (under-estimated) by either including truncated runs as λ=0 (S5 Table S1; S5 Fig. S2). The effect of a constant harvest (quota) accelerates when the subpopulation declines and decelerates when the subpopulation increases. The abrupt decline to λ=0 when the harvest cannot be satisfied or the projected number is less than 0 has a large negative effect on Monte Carlo mean λ_t relative to simulation with no truncations. The estimate of mean λ^t as N_t/N_0 also had a positive bias because of the positive bias in N_t estimates from truncated runs mentioned above.

The Monte Carlo estimate of N_t could be argued to be unbiased when truncations occur because subpopulations cannot be less than 0. However, in a PVA context, estimates of λ and N_t are viewed as summary parameters of a given survival, recruitment, and harvest schedule. PVA implicitly assumes that these summary parameters estimate not only subpopulation performance over some interval of time, but also provide an expectation of future performance providing all things remain equal. Although an argument could be made that the arithmetic mean λ for the last time interval is the best indication of subpopulation performance for that interval, that estimate is the λ associated with the extant (non-truncated) runs only, which are by definition the most optimistic of the simulation set. The Monte Carlo geometric λ estimate and the "N-based" (mean N_t/mean N_0) estimate are only unbiased when there are no truncated Monte Carlo runs. For consistency (all subpopulations considered with the same criteria) we report subpopulation status only as the probability of decline from 2013 (Table 2b) because this metric is not affected by truncations.

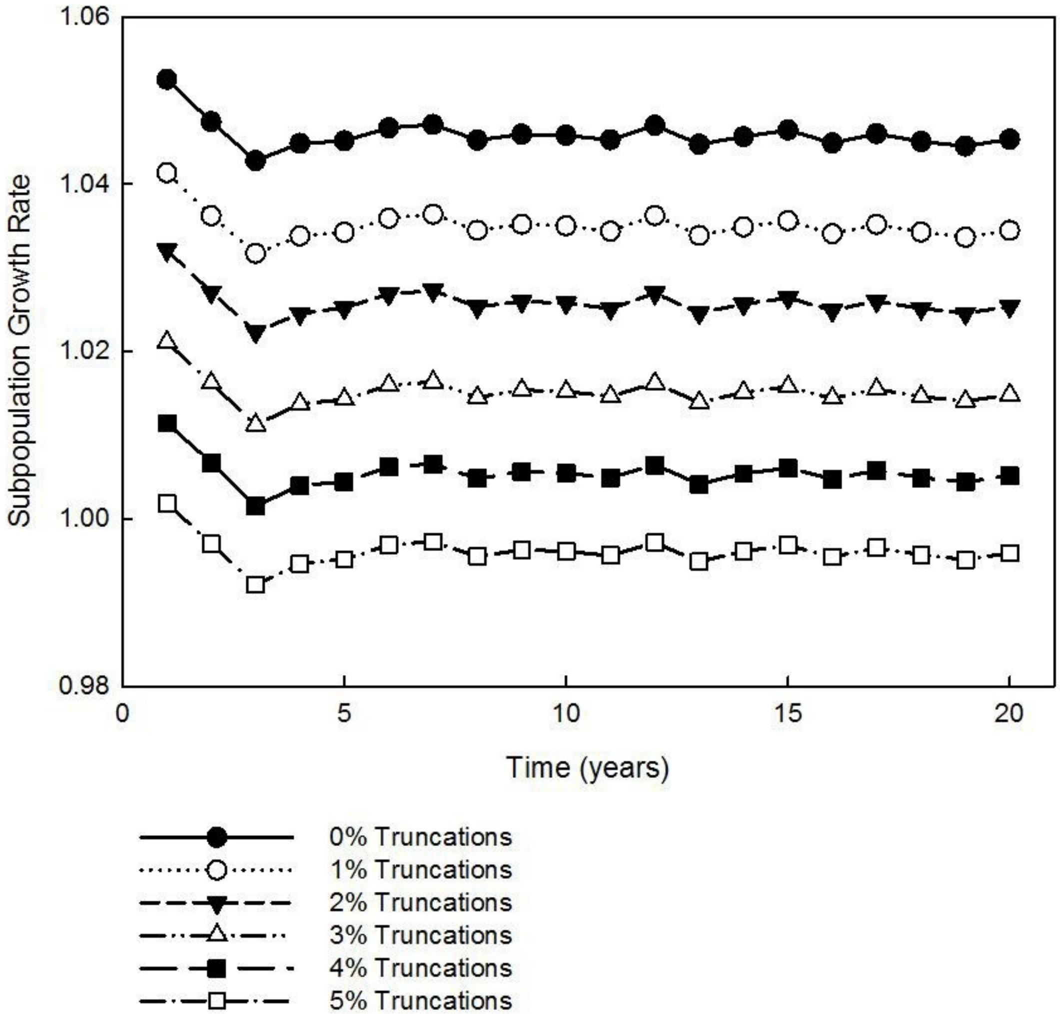
S5 Table S1. The effect of including/excluding truncated runs in the calculations of geometric mean subpopulation growth rates (λ). Each subpopulation was simulated from a current estimate of abundance (2013) for a 20 year period, using natural survival estimates and the 2007-2012 mean annual removals (S4 Table S3). The "Monte Carlo" geometric λ estimate and the "N-Based" (mean N_t/mean N_0) estimate are provided for each subpopulation are provided for both scenarios.

	Truncated Runs Included		Truncated Runs Excluded			
	Monte Carlo	N-based	Monte Carlo	N-based		
Subpop.	**λ**	**Λ**	**λ**	**λ**	**Probability of Decline**	**Proportion Of Truncated Runs**
Baffin Bay	0.2594	1.0330	1.0312	1.1073	0.788	0.7484
Davis Strait	1.0001	1.0137	1.0058	1.0135	0.3894	0.0056
Foxe Basin	0.8464	1.0420	1.0322	1.0522	0.2892	0.180
Gulf of Boothia	0.9400	1.0694	1.0540	1.0766	0.2016	0.107
Kane Basin	N/A	N/A	N/A	N/A	N/A	N/A
Lancaster Sound	0.8815	1.0204	1.0141	1.0272	0.3632	0.1312
M'Clintock Channel	0.9700	1.0322	1.0168	1.0345	0.3178	0.0458
Northern Beaufort Sea	0.2707	0.9944	1.0104	1.0574	0.8348	0.7328
Norwegian Bay	0.9943	1.0097	1.0050	1.0105	0.4034	0.0106
Southern Beaufort Sea	0.4827	0.9564	0.9659	0.9886	0.889	0.5008
Southern Hudson Bay	0.0306	0.8904	1.0036	1.0554	0.9816	0.9696
Viscount Melville Sound	0.9412	1.0745	1.0599	1.0820	0.1884	0.1106
Western Hudson Bay	0.2395	0.8984	0.9469	0.9629	0.9954	0.747

SS FIG. St. The potential effect of truncated runs on subpopulation abundance was estimated fi:om a series ofRISKMAN simulations using increasing initial subpopulation variance (CV) for the Viscount Melville Sound (VM) subpopulation. The simulations were run for 5000 iterations over a 20 year period under a harvest moratorium.



S5 FIG. S2. The potential effect of truncated runs on geometric subpopulation growth rate estimated from a set of 100 Monte Carlo iterations for the Viscount Melville Sound (VM) subpopulation for 20 year period under a harvest moratorium.



Supplementary 6. Eight Instances Where TEK Identified a Polar Bear Subpopulation Trend or Biological Feature before Science Could Identify or Confirm It.

1. Cambridge and Gjoa Haven requested quota reductions in the 1993 polar bear M'Clintock Channel (MC) Memoranda of Understanding (MOU) (M. Taylor, pers. comm. 1986-2008) because they felt numbers were declining in the eastern portion of MC. Taloyoak hunts from the MC subpopulation core had not seen reductions in their hunting area; and did not request a quota reduction for their community.
2. The Foxe Basin (FB) elders advised (through Hunters and Trappers' Organization reps) that FB numbers had declined, after caribou were lost from Southampton Island and polar bear hunting increased. The input from the elders allowed the 1993 FB MOUs completed, even though FB communities had their polar bear quotas reduced. The recent FB aerial survey confirms that the FB subpopulation did increase slowly (as planned) and that the quota increase identified in 2004 FB MOUs (by TEK) were sustainable. A comparison of the PVA trajectory and the aerial survey estimate for FB is displayed in Figure 8.
3. Grise Fiord was aware of unsustainable (and sometimes illegal) hunting by Greenlanders in Kane Basin (KB) and requested that NWT, Greenland, and Canada try to do something about it. The KB study in the mid-1990's (Taylor et al., 2009) confirmed their impressions.
4. Hunters in Davis Strait (DS) were aware of the subpopulation increase in DS well before the recent study was done. The DS subpopulation was reported as declining by the Polar Bear Specialist Group (PBSG) in their 2009 status report (Obbard et al., 2010), but the PBSG status report and subsequent status report that draw from it are in error (Peacock et

al., 2013). The subpopulation was projected to increase at about 0.6 % at current harvest levels (Peacock et al., 2013), while we estimate a 38.94% probability of decline (Table 2b).

5. Gulf of Boothia (GB) communities advised in 1993 and 1996 that their subpopulation had increased and was being under-utilized. This was confirmed by the GB study (Taylor et al., 2009) that finished in 2000.
6. Inuvialuit advised that Viscount Melville Sound (VM) had been over-hunted, and supported (1993 VM MOUs) which specified a five-year moratorium to allow recovery then reduced harvesting to allow continued recovery afterwards.
7. Lancaster Sound (LS) communities agreed that historical quotas had been sustained, and did not report a subpopulation increase or request a quota increase. This perspective was supported by the LS inventory in the mid-1990's (Taylor et al., 2008b).
8. Inuit advised that Norwegian Bay (NW) bears were distinct from those in surrounding areas (Taylor et al., 2008b), and genetic analysis (Paetkau et al., 1999) confirmed that NW bears were the most distinct of any extant subpopulation of polar bears.

Supplementary 7. Evaluation of Global Temperature, Arctic temperature, Global Ocean Heat Content (0-700 m), Arctic Sea Ice Extent Trends, and the Effects of Climate Change on Canadian Polar Bears.

The Arctic is expected to warm more rapidly and to a greater extent than the rest of the globe (Manabe and Stouffer, 1980; Screen and Simmonds, 2010). During the late 1990s and the 2000s, climate warming has coincided with a decline in seasonal and perennial Arctic sea ice cover (Kwok et al., 2009). Several authors have recently suggested that climate warming with consequent sea ice reduction poses the most significant threat to polar bears as a species (Amstrup et al., 2007, 2008, 2010; Hunter et al., 2010; Stirling, 2011; Stirling & Derocher, 2012; Derocher et al., 2013). Management activities aimed towards other threats (e.g., overhunting) have been viewed as secondary, claiming that they are unlikely to make a difference in the prognosis for polar bears as a species in a warming climate due to increased greenhouse gas emissions (Amstrup et al., 2011). A number of recent papers on polar bear distribution, subpopulation status, nutrition, and even genetics assume explicitly that current General Circulation Models (GCM) can accurately forecast future climate (IPCC, 2007; IPCC, 2013), and thus allow reliable predictions about how climate impacts will affect polar bear subpopulations (Amstrup et al., 2007, 2008; Hunter et al., 2010; Obbard et al., 2010; Stirling, 2011; Stirling & Derocher, 2012; Derocher et al., 2013). The recent polar bear literature is replete with phrases stating that the Arctic is rapidly warming and sea ice is rapidly declining (e.g., "The sea ice habitat upon which polar bears depend for successful foraging is rapidly declining in response to greenhouse gas (GHG)-driven global warming", Derocher et al., 2013). In considering the relevance of our demographic perspectives on polar bear subpopulations in this time of "rapid climate warming", we looked empirically at the recent trends in global atmospheric temperature, Arctic atmospheric temperature, ocean heat content, and Arctic sea ice extent.

Global temperatures (HadCRUT4), upper ocean (0-700 meters) temperatures (National Oceanic Data Center), Arctic temperatures (Remote Sensing Systems, Inc.), and Arctic sea ice extent (National Snow and Ice Data Center) have been estimated and are available for the 1980-2013 period (S7 Tables S1 and S2); where reliable data is available for each of the attributes. We used break-point linear regression to investigate trends for each dataset. Null hypotheses of no trends or no correlations were rejected at $p < 0.05$. We also estimated the power (1-β) of the test using G-Power software (Faul et al., 2007).

Both the atmosphere and the oceans have warmed (IPCC, 2007, 2013; Levitus et al., 2012; S7 Tables S1, S2, and S3; S7 Fig. S1, S2, and S3) and Arctic sea ice has declined since satellite records began in 1978 (Parkinson et al., 1999; Comiso, 2008; S7 Fig. S4). The 1980-2013 trends in global atmospheric temperature ($p \leq 0.001$), Arctic atmospheric temperature ($p \leq 0.001$), and ocean heat content ($p \leq 0.001$) were all statistically significant. Global atmospheric temperatures have been increasing at a rate of approximately 0.16°C per decade since 1980 (S7 Table S3). Arctic atmospheric temperatures have also experienced a statistically significant ($p \leq 0.001$) long-term warming at a rate of 0.32°C per decade since 1980 (S7 Table S4). The 1980-2013 trend in ocean heat content indicates that the amount of heat stored in the upper ocean has increased at a rate of 0.445 10^22 Joules per decade (S7 Table S3; S7 Fig. S3). As expected during a period of climate warming, the trend in Arctic sea ice extent (as an annual average) for this interval is declining (0.55 10^6 km^2 per year ($p<0.001$) (S7 Table S3). However, in recent years all of these trends are reduced except for upper ocean warming; and many are no longer significant over time spans longer than a decade (e.g., global atmospheric temperature trend has no significant trend since 1997 ($p = 0.179$) (S7 Table S3; S7 Fig. S1).

Sea ice has declined more in the warmer months than the colder months (Parkinson et al., 1999; S7 Tables S1, S2, and S4; S7 Fig. S5). Trends in sea ice extent for the warmer months (i.e., June-October) and the colder months (i.e., November-May) were all numerically negative and all statistically significant since 1980 (S7 Table S4). However, breakpoint regression from 2013 identified no significant trend in sea ice extent by month for periods ranging from 8 to 19 years (S7 Tables S5 and S6) with the November-June average no-trend length being 12.5 years and the July-September average no-tend length being 8.5 years ($p<0.028$). Failure to reject Ho: no trend could be because there is no trend, or because the power of the test was insufficient to detect an existing trend. Our protocol also examined the power of the test to detect a trend as large as or larger than the trend identified from least squares regression for the "no-trend" interval (S7 Table S5).

The correlation between annual sea ice extent and global temperature for the interval 1980-2013 was highly significant for all months of the year, while the same comparison for the 1997-2013 interval was not significant for any months of the year (S7 Tables S7 and S8; S7 Fig. S6). The lack of correlation could occur because the two measures were uncoupled, or because they were out of phase (lag effect).

Breakpoint regression of Arctic temperatures since 1980 (S7 Table S3; S7 Fig. S2) reveals that the interval from 1980 to 2013 can be divided into two intervals with no significant trend (1980-1998 and 1998-2013, with a highly significant difference of -0.616°C ($p<0.001$) between the mean temperatures of these two intervals (S7 Table S9). The perspective of climate change occurring as a series of shifting climate states rather than a single (constant) incremental long-term climate trends has also been identified for global atmospheric temperatures (Swanson and Tsonis, 2009) and ocean temperatures (Douglas & Knox, 2012).

Does the evidence for reduced warming rates in recent decades indicate that the rapid Arctic sea ice decline over the first decade of the 21st century could be driven by factors other than CO_2 driven global climate change? Or is this apparent "pause" in global surface warming, Arctic sea ice decline, and Arctic warming due to global surface temperatures being dampened by other climate factors such as heat sequestering in the oceans (e.g., Levitus et al., 2012; Meehl et al., 2011)? These questions are relevant to the long-term prognosis for polar bear subpopulations, but beyond the scope of this paper. The empirical evidence we have summarized indicates that warming has not "paused" because ocean temperatures have continued to increase while no trends were apparent in the recent global atmospheric mean annual temperature, Arctic mean annual temperature, and Arctic mean annual and monthly sea ice extent (S7 Table S3). We were surprised at the low power associated with the highly publicized no-trend findings for recent global atmospheric temperature. We contrast the relatively high power (β=0.933) associated with the 1980-1997 positive trend for global atmospheric temperature (17 years) with the low power (β=0.282) associated with the recent (1997-2013, 17 years), no-trend interval (S7 Table S3; S7 Fig. S1). Our interpretation for this time series would be no *detectable* trend for the recent interval because of the reduced slope and underlying environmental variance. We interpret these finding as evidence that global atmospheric warming has slowed, but the evidence is insufficient to conclude that atmospheric warming has stopped or paused. The same pattern is not evident for Arctic atmospheric warming. We see two distinct intervals (1980-1998 and 1998-2013), with the more recent interval being warmer by about 0.616°C ($p<0.001$) (S7 Table S3; S7 Fig. S2). As detailed above, recent Arctic sea ice trends appear to have two distinct seasonal strata; with July-Oct ice declining and no trend (approximately medium power, Cohen, 1988) for Nov-June sea ice.

Recent changes in Arctic sea ice were not predicted by GCMs, and we are not aware of a comprehensive explanation for this behaviour even in the context of trends in global atmospheric temperature, Arctic atmospheric temperature and upper (0-700 m) ocean warming. Combined with recent unexpected behaviour in Antarctic sea ice (Parkinson and Cavalieri, 2012) we suggest that the science is not "settled" with regards to the sea ice response to transient climate change. In this context, we question the certainty with which GCM projections have been used to predict polar bear response by 2050 (e.g., Amstrup et al., 2007) given the complex climate response observed just 6 years later. In our opinion, the use of GCM simulations of sea ice cover to predict polar bear subpopulation trend and persistence over the next several decades is essentially hypothetical.

Although GCM predictions have been steadily improving over the past two decades (Maslowski et al., 2012), discrepancies between observed trends and GCM forecasts of sea ice extent suggest that the treatment of sea ice in these models remains problematic. A comparison of individual GCM forecasts for sea ice and global temperature to current sea ice extent and current global temperature (Kirtman et al., 2013; S7 Fig. S7) shows that while global warming has been on the low end of GCM projections for this period, predictions of summer Arctic sea ice decline have been too conservative (Stroeve et al., 2007; S7 Fig. S8 and S9). One reason could be that ocean currents have caused greater heat transfer from the tropics to the poles than the 2007 GCMs anticipated (Stroeve et al., 2012). The oceans hold about 2100 times more heat than the atmosphere, and at least 70% of stored heat occurs in the upper 700 m of the oceans

(Levitus et al., 2012). Current GCMs may have underestimated the contribution of ocean heat to convective Arctic sea ice reduction.

With respect to climate effects on polar bears, the lack of Arctic sea ice decline in the late fall to early summer months in the last 10 years demonstrates that seasonal Arctic sea ice regeneration in cold months can occur even as Arctic sea ice extent in warmer seasons declines (S7 Fig. S5); even when multiyear ice has been reduced (Maslanik et al., 2011). Polar bears are hyperphagic in the late spring and early summer (Stirling, 1998, 2011; Cherry et al., 2013) so this recent slowing of sea ice decline in these months may reduce the impact of sea ice in the "open-water" months in areas where sea ice did decline during the "open water" season.

While warming and sea ice decline have occurred throughout Arctic regions, the magnitude of these effects differs regionally (Amstrup et al., 2008; Thiemann et al., 2008). Even adjacent regions can be qualitatively different. For example, the Canadian Archipelago, where approximately one third of Canada's polar bears reside (COSEWIC, 2008; Obbard et al., 2010; Table 2a), continues to have about the same multi-year and annual sea ice in all seasons (Sou & Flato, 2009), as opposed to the adjacent Arctic Basin region which annual sea ice has declined in the summer months and Arctic Basin sea ice mass has declined due to loss of both annual and multi-year ice (Maslanik et al., 2011). As a result, some researchers have hypothesized that the effects of climate warming and sea ice decline on polar bears will not be uniform across the Arctic based on differential climate and sea ice dynamics (Amstrup et al., 2008; Thiemann et al., 2008).

GCM modeling of Arctic sea ice tends to focus on the Arctic Basin beyond the Canadian Archipelago (Sou & Flato, 2009). This large basin of distinct land, sea and multiyear ice surfaces is easier to resolve (model) in GCMs. Polar bears use habitats differently depending on both physical and biological factors. Annual sea ice is preferred for hunting, but heavy multi-year sea ice is also used for denning in some areas (Lentfer, 1975; Amstrup & Gardner, 1994). Annual sea ice located over the continental shelf is preferred habitat for polar bears because of higher biological productivity compared to deep-water regions and greater accessibility to prey species (Derocher et al., 2004; Durner et al., 2004; Durner et al., 2009; Harwood et al., 2012). Over half of the world's polar bear subpopulations and more than half of the world's polar bears occur within or adjacent to the Canadian Archipelago, thus almost all Canadian polar bears (and most polar bears throughout the circumpolar basin) occur in in preferred habitat regions that are not well-resolved by GCMs and that have received less research attention. How climate warming is experienced by polar bears within the Canadian Archipelago will depend on the onset and duration of seasonal open water there (Derocher et al., 2004; Stirling & Derocher, 2012); and especially on sea ice state during the late spring and early summer hyperphagic hunting season (Stirling, 1998; Stirling, 2011; Cherry et al., 2013). Changes in critical aspects of sea ice characteristics in polar bear subpopulations are difficult to anticipate because the ecological circumstances for each subpopulation are distinct (Thiemann et al., 2008), and because predictive sea ice models for the Archipelago are still developing (Sou & Flato, 2009). Given the uncertainty of future sea ice dynamics, especially in the Canadian Archipelago, and the uncertainty of demographic response of polar bear subpopulations to these changes; we suggest that predictions of declines in polar bear numbers and productivity based on current GCM forecasts are premature and unreliable.

The current view of the IPCC and most climate scientists is that long term climatic changes are dependent on the future path of greenhouse gas emissions (IPCC, 2007; IPCC; 2013; Moss et al., 2010). If emissions are limited – whether through concerted mitigation action (c.f.

van Vuuren et al., 2011), exhaustion of fossil fuel reserves (c.f. Nel & Cooper, 2009), or some combination of the two – along the lines of the B1 (or RCP2.6) scenario then severe impacts on most subpopulations will be avoided (Amstrup et al., 2010). Given the apparent lack of numerical impact on polar bears after 130 years of anthropogenic global warming, and the ability of polar bears to survive warmer temperatures during the last interglacial (Ingólfsson & Wiig 2009; Lindqvist et al., 2010; Edwards et al., 2011); it seems unlikely that polar bears are at risk as a species (Fig. 2). There is good evidence for condition and recruitment impacts from reduced sea ice in the Baffin Bay, Southern Beaufort Sea, Southern Hudson Bay, and Western Hudson Bay subpopulations (Obbard et al., 2006; Regehr et al., 2006; 2007a, 2007b; Obbard et al., 2007; Peacock et al., 2012), however we show that M-R evidence for subpopulation decline in these areas is suspect, and also contradicted by other evidence (Tables 5, 6, 7, and 9; Fig. 5, 6, 7, and 8). We agree that these subpopulations should be monitored preferentially because they appear to be the most vulnerable to climatological changes in Arctic sea ice state (Stirling et al., 1999; 2004; Stirling & Parkinson, 2006; Regehr et al., 2007a, 2007b).

It is generally accepted that polar bears have increased in numbers as they recovered from over-hunting prior to the International Agreement on the Conservation of Polar Bears (Prestrud & Stirling, 1994; Lunn et al., 2002). Although contemporary management approaches did not explicitly recognize and accommodate progressive environmental effects until recently, the efforts to manage the harvest have resulted in secure and productive subpopulations of polar bears throughout most of Canada. Increased monitoring and adaptive management are warranted; however, trade restrictions and ultra-precautionary status designations that precede any actual decline in numbers or loss of range works against conservation because such practices reduce the credibility of polar bear management practices to aboriginal people who continue to harvest at sustainable levels for nutritional, cultural, and economic purposes.

S7 Table S1. HadCRUT4 global temperature, REMSS Arctic temperature, NSIDC sea ice extent, and NOAA-NODC ocean heat content data for the 1980-1996 period.

	HADCRUT Annual Global Temp.	*REMSS Annual Arctic Temp.*	*Sea Ice Extent 10^6 km sq.*												*Ocean Heat Content*
YEAR	**TEMP (°C)**		**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**	**10^{22} Joules**
1980	0.093	0.259	14.96	15.98	16.13	15.49	14.04	12.31	10.39	8.04	7.85	9.46	11.69	13.72	1.091
1981	0.1415	0.561	15.03	15.65	15.61	15.12	13.9	12.57	10.62	7.86	7.25	9.19	11.17	13.74	0.122
1982	0.0115	-0.404	15.26	16.06	16.15	15.57	14.17	12.69	10.75	8.26	7.45	9.98	11.91	13.83	-2.306
1983	0.190	0.084	15.1	16.02	16.1	15.3	13.54	12.36	10.91	8.36	7.52	9.64	11.64	13.44	-2.763
1984	-0.0145	-0.079	14.61	15.32	15.62	15.15	13.68	12.2	10.15	7.87	7.17	8.84	11.29	13.18	-0.459
1985	-0.0285	-0.14	14.86	15.67	16.06	15.34	14.23	12.4	10.09	7.46	6.93	8.88	11.39	13.19	0.11
1986	0.0465	-0.223	15.02	15.89	16.08	15.15	13.52	12.1	10.47	8.01	7.54	9.89	11.78	13.4	-1.037
1987	0.1855	-0.442	15.2	16.11	15.95	15.33	13.81	12.57	9.98	7.69	7.48	9.29	11.52	N/A	-0.893
1988	0.201	0.125	N/A	15.61	16.13	15.21	13.69	12.02	10.04	7.9	7.49	9.47	11.69	13.78	1.088
1989	0.121	-0.143	15.12	15.56	15.52	14.44	12.98	12.31	10.38	7.92	7.04	9.52	11.5	13.47	0.903
1990	0.2925	-0.195	14.95	15.56	15.88	14.68	13.3	11.68	9.62	6.82	6.24	9.35	11.31	13.27	0.177
1991	0.253	0.208	14.46	15.26	15.5	14.93	13.51	12.23	9.68	7.4	6.55	9.16	11.12	13.17	2.646
1992	0.102	-0.542	14.72	15.5	15.47	14.7	13.25	12.13	10.61	7.86	7.55	9.6	11.87	13.46	0.572
1993	0.1435	-0.186	15.08	15.73	15.88	15.18	13.54	11.99	9.66	7.29	6.5	9.18	11.73	13.52	0.684
1994	0.2045	-0.060	14.82	15.61	15.58	14.95	13.73	12.1	10.22	7.61	7.18	9.48	11.3	13.53	1.51
1995	0.323	0.581	14.62	15.24	15.32	14.59	13.04	11.55	9.15	6.68	6.13	8.94	10.97	12.98	2.264
1996	0.1775	0.233	14.21	15.17	15.13	14.22	13.06	12.1	10.36	8.17	7.88	9.39	10.56	13.14	4.544

S7 Table S2. HADCRUT4 global temperature, REMSS Arctic temperature, NSIDC sea ice extent, and NOAA-NODC ocean heat content data for the 1997-2013 period.

	HADCRUT Annual Global Temp.	*REMSS Annual Arctic Temp.*	*Sea Ice Extent 10^6 km sq.*												*Ocean Heat Content*
YEAR	**TEMP (°C)**		**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**	**10^{22} Joules**
1997	0.392	0.313	14.47	15.52	15.58	14.59	13.32	11.91	9.59	7.3	6.74	8.76	10.91	13.29	3.245
1998	0.530	0.397	14.81	15.77	15.66	14.89	13.8	11.85	9.62	7.49	6.56	8.85	10.75	13.26	4.304
1999	0.301	0.186	14.47	15.37	15.4	15.13	13.86	12.1	9.59	7.38	6.24	9.1	10.99	12.88	5.943
2000	0.295	0.511	14.41	15.18	15.27	14.63	13.18	11.71	9.75	7.21	6.32	8.92	10.54	12.81	5.857
2001	0.4385	0.443	14.31	15.27	15.61	14.86	13.72	11.69	9.22	7.47	6.75	8.59	10.92	12.84	4.117
2002	0.494	0.508	14.45	15.36	15.44	14.37	13.12	11.69	9.49	6.53	5.96	8.81	10.78	12.82	6.789
2003	0.505	0.841	14.46	15.25	15.49	14.57	13	11.77	9.46	6.85	6.15	8.65	10.29	12.82	9.952
2004	0.4455	0.307	14.03	14.93	15.05	14.11	12.58	11.51	9.6	6.83	6.05	8.48	10.65	12.72	10.24
2005	0.5415	1.093	13.66	14.36	14.74	14.07	12.99	11.29	8.93	6.3	5.57	8.45	10.47	12.47	8.412
2006	0.496	0.705	13.6	14.42	14.43	13.97	12.62	11.06	8.67	6.52	5.92	8.33	9.84	12.27	10.43
2007	0.4845	0.821	13.77	14.53	14.65	13.87	12.89	11.49	8.13	5.36	4.3	6.77	10.05	12.39	9.478
2008	0.3885	0.502	14.05	15.01	15.22	14.42	13.19	11.36	8.99	6.05	4.73	8.42	10.62	12.52	10.052
2009	0.495	0.427	14.08	14.85	15.14	14.57	13.4	11.46	8.8	6.28	5.39	7.52	10.27	12.51	10.126
2010	0.5475	1.175	13.8	14.59	15.11	14.7	13.11	10.82	8.36	6.01	4.93	7.71	9.88	12.02	10.367
2011	0.4085	0.791	13.57	14.38	14.58	14.16	12.81	10.99	7.91	5.55	4.63	7.14	10	12.4	10.869
2012	0.451	0.949	13.77	14.59	15.24	14.72	13.12	10.92	7.93	4.71	3.63	7.07	9.92	12.2	10.941
2013	0.488	0.486	13.66	14.66	14.99	14.3	13.04	11.43	8.23	6.05	5.24	7.4	9.95	12.18	12.601

S7 Table S3. Examining Ho: slope = 0 for world ocean heat content, global atmospheric temperature, Arctic (60-82.5 N) atmospheric temperature, and annual Arctic sea ice extent vs. time. The probability of a Type II error (false negative rate) is β, and the power is equal to (1−β).

			Bivariate Correlation		Linear Regression				Power
Variable	**Period**	**Sample Size**	**Pearson's R**	**p ≤**	**R Square**	**Intercept**	**Slope**	**p ≤**	**(1- β)**
Ocean Heat	1980 to 2013	34	0.951	0.001	0.904	-884.227	0.445	0.001	1.0
Ocean Heat	2010 to 2013	4	0.90	0.10	0.810	-1351.396	0.677	0.10	0.583
Ocean Heat	1980 to 2010	31	0.936	0.001	0.877	-879.417	0.493	0.001	1.0
Global Temp	1980 to 2013	34	0.870	0.001	0.757	-31.185	0.016	0.001	1.0
Global Temp	1997 to 2013	17	0.342	0.179	0.117	-9.767	0.005	0.179	0.282
Global Temp	1980 to 1997	18	0.646	0.004	0.418	-27.178	0.014	0.004	0.933
Arctic Temp	1980 to 2013	34	0.723	0.001	0.523	-63.506	0.032	0.001	1.0
Arctic Temp	1998 to 2013	16	0.488	0.055	0.367	-78.339	0.029	0.055	0.548
Arctic Temp	1980 to 1998	19	0.20	0.412	0.040	-22.978	0.012	0.412	0.141
Sea Ice Extent	1980 to 2013	34	-0.930	0.001	0.865	120.710	0.055	0.001	1.0
Sea Ice Extent	2004 to 2013	10	-0.587	0.074	0.345	108.851	-0.049	0.074	0.531
Sea Ice Extent	1980 to 2004	25	-0.885	0.001	0.782	90.842	-0.040	0.001	1.0

S7 Table S4. Examining Ho: slope = 0 for monthly Arctic sea ice extent vs. time for the January 1980 to December 2013 period. The probability of a Type II error (false negative rate) is β, and the power is equal to (1−β).

		January 1980 to December 2013						
		Bivariate Correlation		Linear Regression				Power
Month	**Sample Size**	**Pearson's R**	**p ≤**	**R Square**	**Intercept**	**Slope**	**p ≤**	**(1- β)**
January	33/34	-0.893	0.001	0.797	108.258	-0.047	0.001	1.0
February	34	-0.863	0.001	0.745	103.805	-0.044	0.001	1.0
March	34	-0.818	0.001	0.669	91.854	-0.038	0.001	1.0
April	34	-0.764	0.001	0.583	84.690	-0.035	0.001	1.0
May	34	-0.698	0.001	0.487	73.218	-0.030	0.001	1.0
June	34	-0.895	0.001	0.802	100.107	-0.044	0.001	1.0
July	34	-0.892	0.001	0.796	158.457	-0.075	0.001	1.0
August	34	-0.860	0.001	0.740	163.214	-0.078	0.001	1.0
September	34	-0.861	0.001	0.741	192.759	-0.093	0.001	1.0
October	34	-0.838	0.001	0.702	147.599	-0.070	0.001	1.0
November	34	-0.895	0.001	0.802	126.766	-0.058	0.001	1.0
December	33/34	-0.915	0.001	0.837	107.764	-0.047	0.001	1.0

S7 Table S5. Examining Ho: slope = 0 for monthly Arctic sea ice extent vs. time for the period at which the relationship between the two variables was no longer significant. The probability of a Type II error (false negative rate) is β, and the power is equal to (1−β).

			Bivariate Correlation		Linear Regression				Power
Period	**Month**	**Sample Size**	**Pearson's R**	**p ≤**	**R Square**	**Intercept**	**Slope**	**p ≤**	**(1- β)**
2003 to 2013	*January*	12	-0.515	0.105	0.265	97.830	-0.042	0.105	0.434
2002 to 2013	*February*	13	-0.548	0.065	0.300	117.014	-0.051	0.065	0.538
1999 to 2013	*March*	15	-0.467	0.080	0.218	89.026	-0.037	0.080	0.477
1994 to 2013	*April*	20	-0.394	0.086	0.155	60.655	-0.023	0.086	0.441
1995 to 2013	*May*	19	-0.365	0.124	0.133	58.820	-0.023	0.124	0.371
2003 to 2013	*June*	11	-0.558	0.074	0.312	111.134	-0.050	0.074	0.517
2005 to 2013	*July*	9	-0.661	0.052	0.438	212.352	-0.101	0.052	0.629
2004 to 2013	*August*	10	-0.622	0.055	0.387	259.402	-0.126	0.055	0.596
2004 to 2013	*September*	10	-0.607	0.063	0.368	306.557	-0.150	0.063	0.563
2005 to 2013	*October*	9	-0.578	0.103	0.335	273.838	-0.132	0.103	0.454
2003 to 2013	*November*	11	-0.578	0.063	0.334	114.770	-0.052	0.063	0.552
2005 to 2013	*December*	9	-0.492	0.179	0.242	74.273	-0.031	0.179	0.314

S7 Table S6. Examining Ho: slope = 0 for monthly Arctic sea ice extent vs. time for the period prior to the breakpoint, at which the relationship between the two variables was no longer significant. The probability of a Type II error (false negative rate) is β, and the power is equal to (1−β).

			Bivariate Correlation		Linear Regression				Power
Period	**Month**	**Sample Size**	**Pearson's R**	**p ≤**	**R Square**	**Intercept**	**Slope**	**p ≤**	**(1- β)**
1980 to 2003	*January*	23/24	-0.740	0.001	0.547	78.659	-0.032	0.001	0.999
1980 to 2002	*February*	23	-0.654	0.001	0.428	71.241	-0.028	0.001	0.990
1980 to 1999	*March*	20	-0.675	0.001	0.455	86.238	-0.035	0.001	0.965
1980 to 1994	*April*	15	-0.633	0.011	0.401	103.098	-0.044	0.011	0.825
1980 to 1995	*May*	16	-0.657	0.006	0.432	113.960	-0.050	0.006	0.889
1980 to 2003	*June*	24	-0.781	0.001	0.609	80.861	-0.035	0.001	1.0
1980 to 2005	*July*	26	-0.780	0.001	0.608	116.345	-0.053	0.001	1.0
1980 to 2004	*August*	25	-0.701	0.001	0.491	104.096	-0.048	0.001	0.996
1980 to 2004	*September*	25	-0.712	0.001	0.507	123.479	-0.059	0.001	0.999
1980 to 2005	*October*	26	-0.706	0.001	0.498	86.022	-0.039	0.001	0.999
1980 to 2003	*November*	24	-0.761	0.001	0.579	109.043	-0.049	0.001	1.0
1980 to 2005	*December*	25/26	-0.828	0.001	0.686	91.649	-0.039	0.001	1.0

S7 Table S7. Examining: correlations for annual global temperatures vs. monthly Arctic sea ice extent for the 1980-2013 period.

		January 1980 to December 2013					
		Bivariate Correlation		Linear Regression			
Month	**Sample Size**	**Pearson's R**	**p ≤**	**R Square**	**Intercept**	**Slope**	**p ≤**
January	33/34	-0.749	0.001	0.562	15.118	-2.161	0.001
February	34	-0.710	0.001	0.503	15.894	-2.010	0.001
March	34	-0.691	0.001	0.477	15.994	-1.783	0.001
April	34	-0.697	0.001	0.486	15.270	-1.763	0.001
May	34	-0.621	0.001	0.386	13.814	-1.472	0.001
June	34	-0.808	0.001	0.653	12.491	-7.752	0.001
July	34	-0.786	0.001	0.617	10.650	-3.622	0.001
August	34	-0.738	0.001	0.738	8.195	-3.698	0.001
September	34	-0.734	0.001	0.734	7.688	-4.389	0.001
October	34	-0.710	0.001	0.503	9.741	-3.247	0.001
November	34	-0.832	0.001	0.693	11.808	-2.976	0.001
December	33/34	-0.782	0.001	0.612	13.676	-2.218	0.001

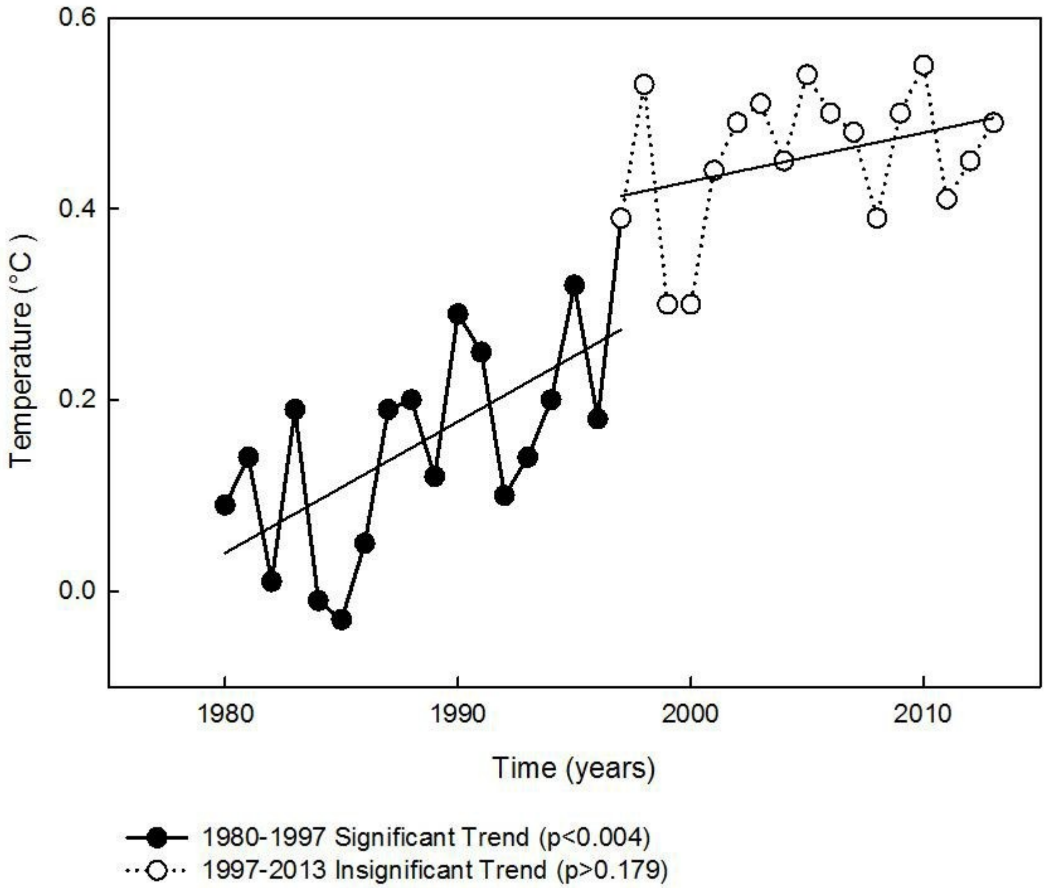
S7 Table S8. Examining correlations for annual global temperatures vs. monthly Arctic sea ice extent for the period at which the relationship between the two variables was no longer significant. The probability of a Type II error (false negative rate) is β, and the power is equal to (1−β).

			Bivariate Correlation		Linear Regression			
Period	**Month**	**Sample Size**	**Pearson's R**	**p ≤**	**R Square**	**Intercept**	**Slope**	**p ≤**
1995 to 2013	*January*	19	-0.328	0.170	0.108	14.671	-1.284	0.170
1994 to 2013	*February*	20	-0.435	0.055	0.189	15.731	-1.729	0.055
1993 to 2013	*March*	21	-0.413	0.063	0.170	15.736	-1.28-	0.063
1992 to 2013	*April*	22	-0.422	0.050	0.178	14.968	-1.123	0.050
1987 to 2013	*May*	27	-0.362	0.063	0.131	13.570	-0.900	0.063
1997 to 2013	*June*	17	-0.406	0.106	0.165	12.371	-1.977	0.106
1997 to 2013	*July*	17	-0.318	0.214	0.101	10.199	-2.734	0.214
1997 to 2013	*August*	17	-0.267	0.301	0.071	7.747	-2.824	0.301
1996 to 2013	*September*	18	-0.455	0.058	0.207	7.831	-4.806	0.058
1996 to 2013	*October*	18	-0.466	0.055	0.211	9.842	-3.645	0.055
1995 to 2013	*November*	19	-0.445	0.056	0.198	11.227	-1.819	0.056
1996 to 2013	*December*	18	-0.455	0.058	0.207	13.404	-1.737	0.058

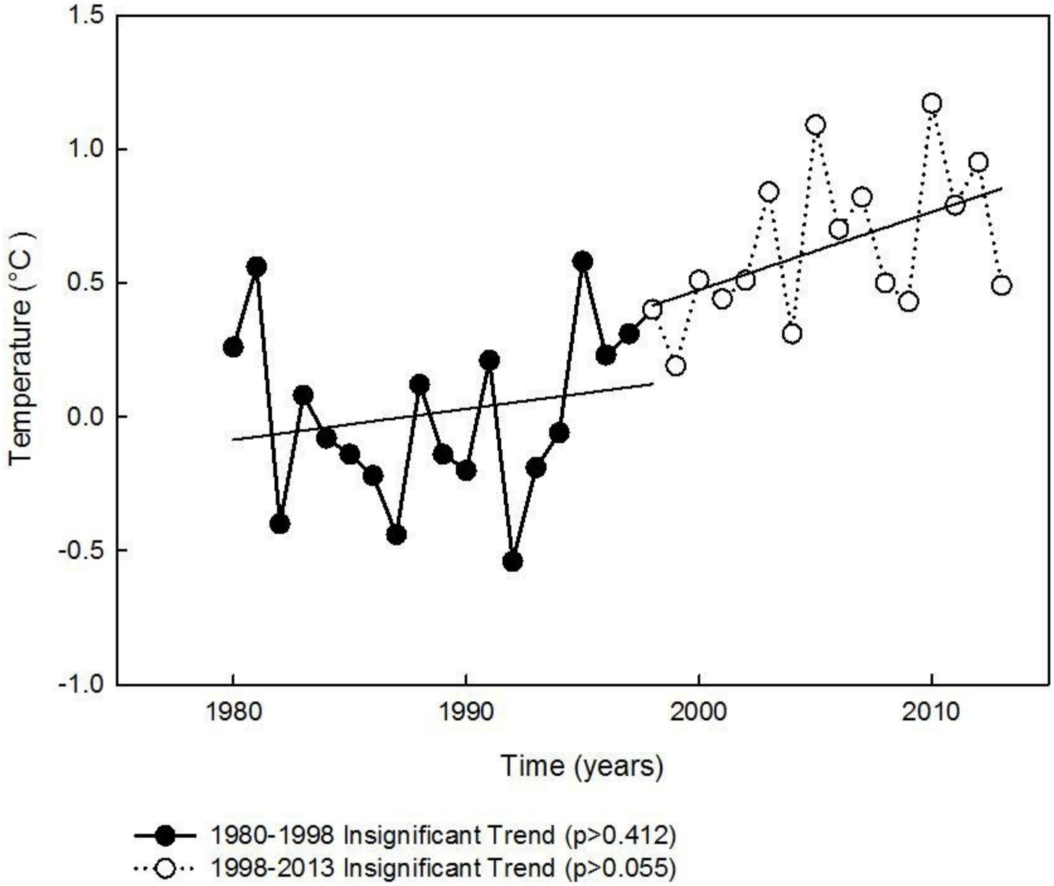
S7 Table S9. Comparing Arctic atmospheric temperature means for the time periods identified by a breakpoint regression of Arctic temperatures since 1980 (Table H3).

Period	N	Mean (SE)	t-test; Ho: difference = 0			Levene's Test for Equality of Variances	
			Mean Difference	SE Difference	p ≤	F	p ≤
1980-1998	19	0.018 (0.075)	-0.616	0.104	0.001	0.380	0.542
1998-2013	16	0.634 (0.071)					

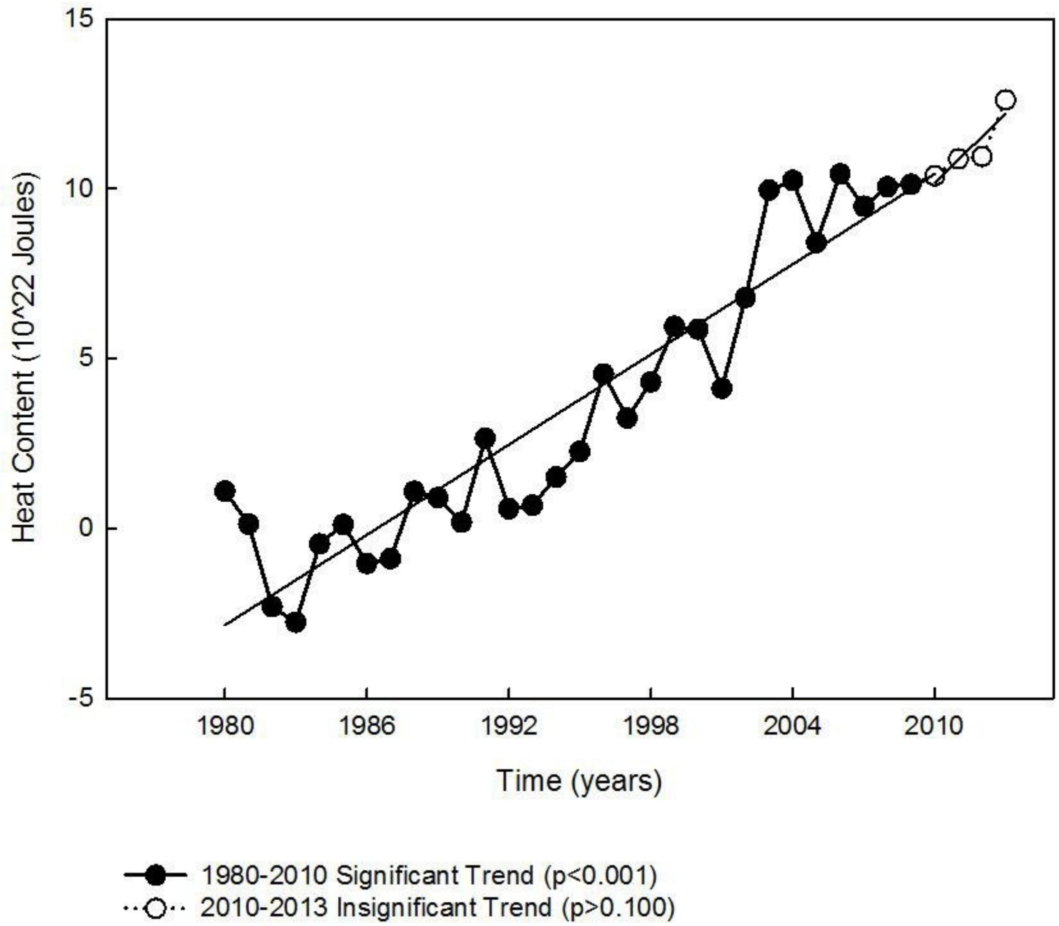
S7 FIG. S1. HadCRUT4 annual global temperature for the 1980-2013 period. A breakpoint regression was used to divide the period into two distinct intervals (Table G3).



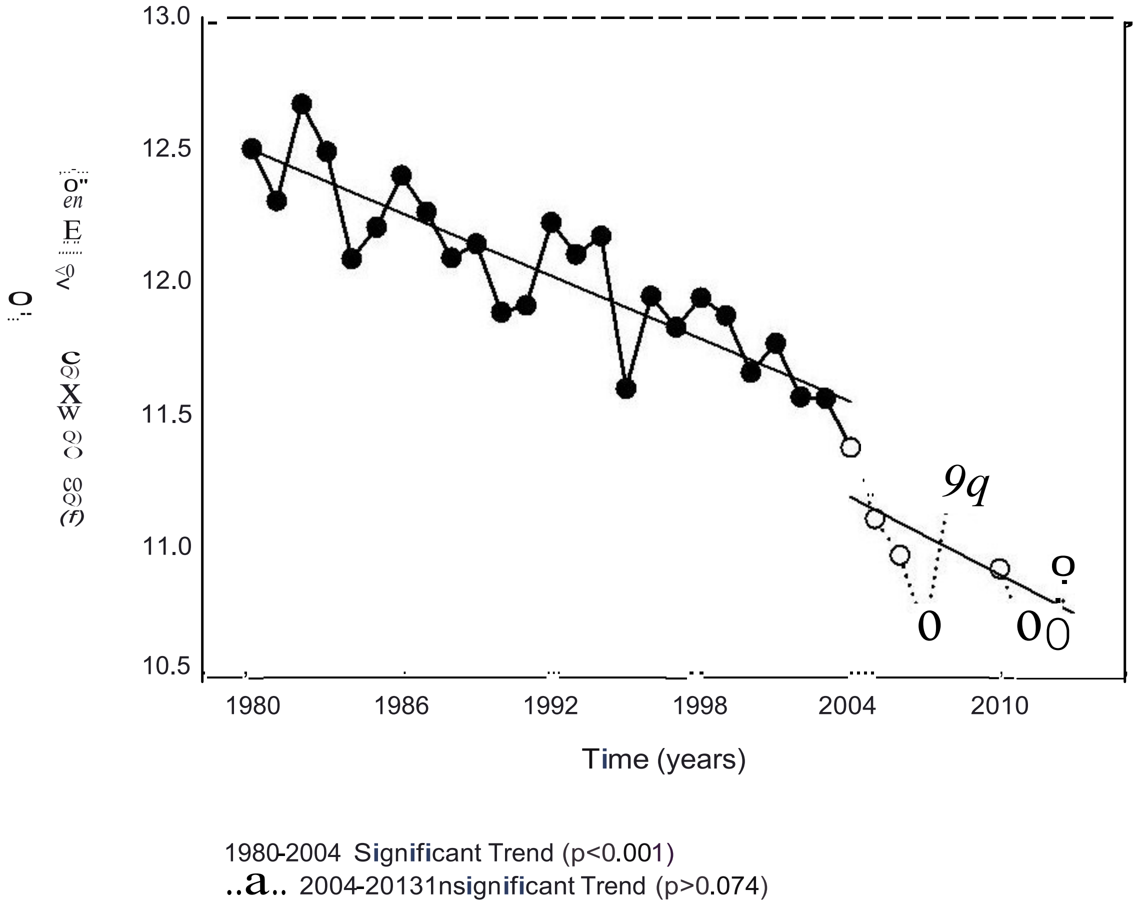
S7 FIG. S2. REMSS annual Arctic temperature for the 1980-2013 period. A breakpoint regression was used to divide the period into two distinct intervals (Table G3). The difference between the two interval means was -0.616 (0.104) (Table G9).



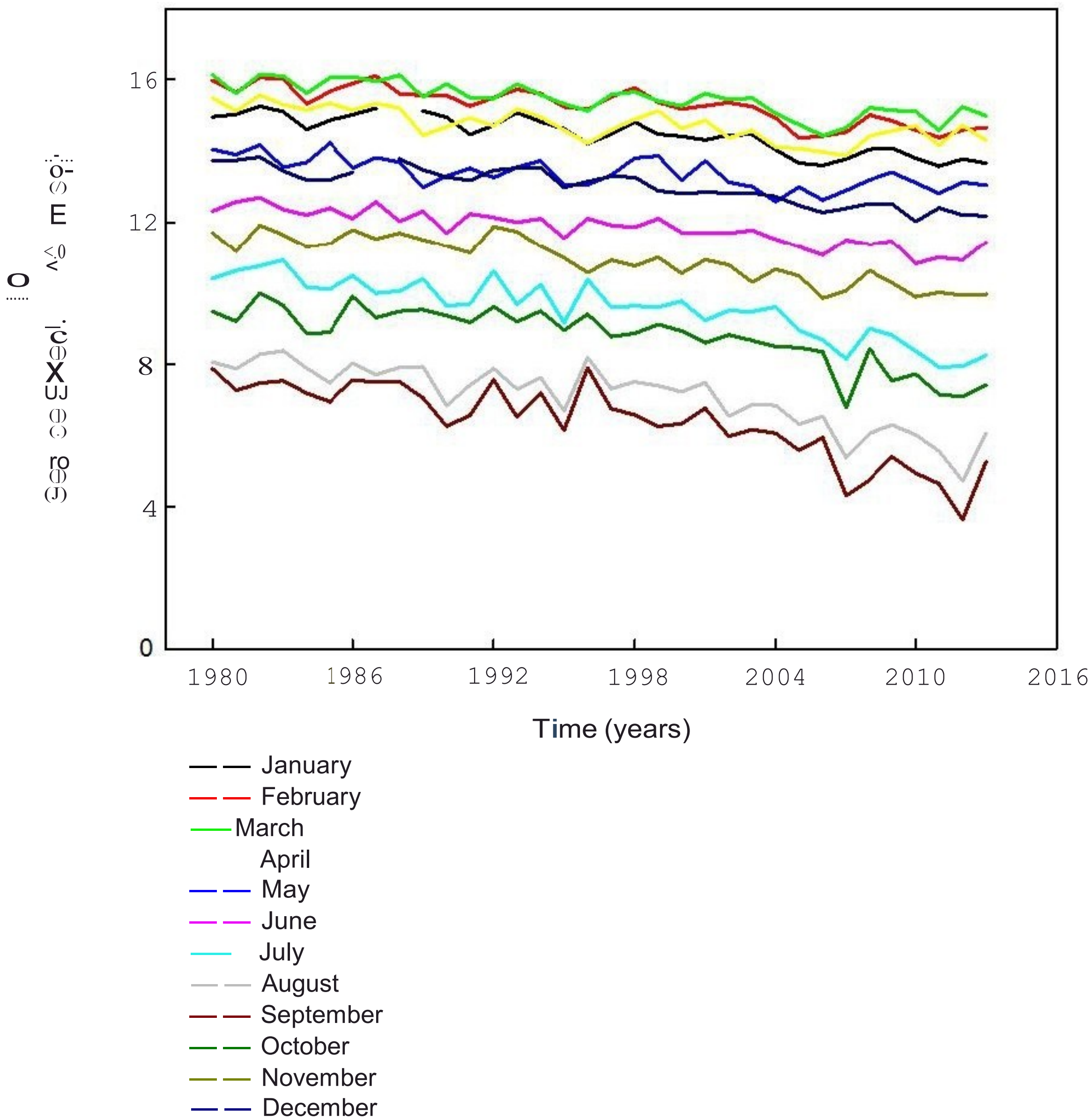
S7 FIG. S3. NOAA-NODC global ocean heat content (0-700 meters) for the 1980-2013 period. A breakpoint regression was used to divide the period into two distinct intervals (Table G3).



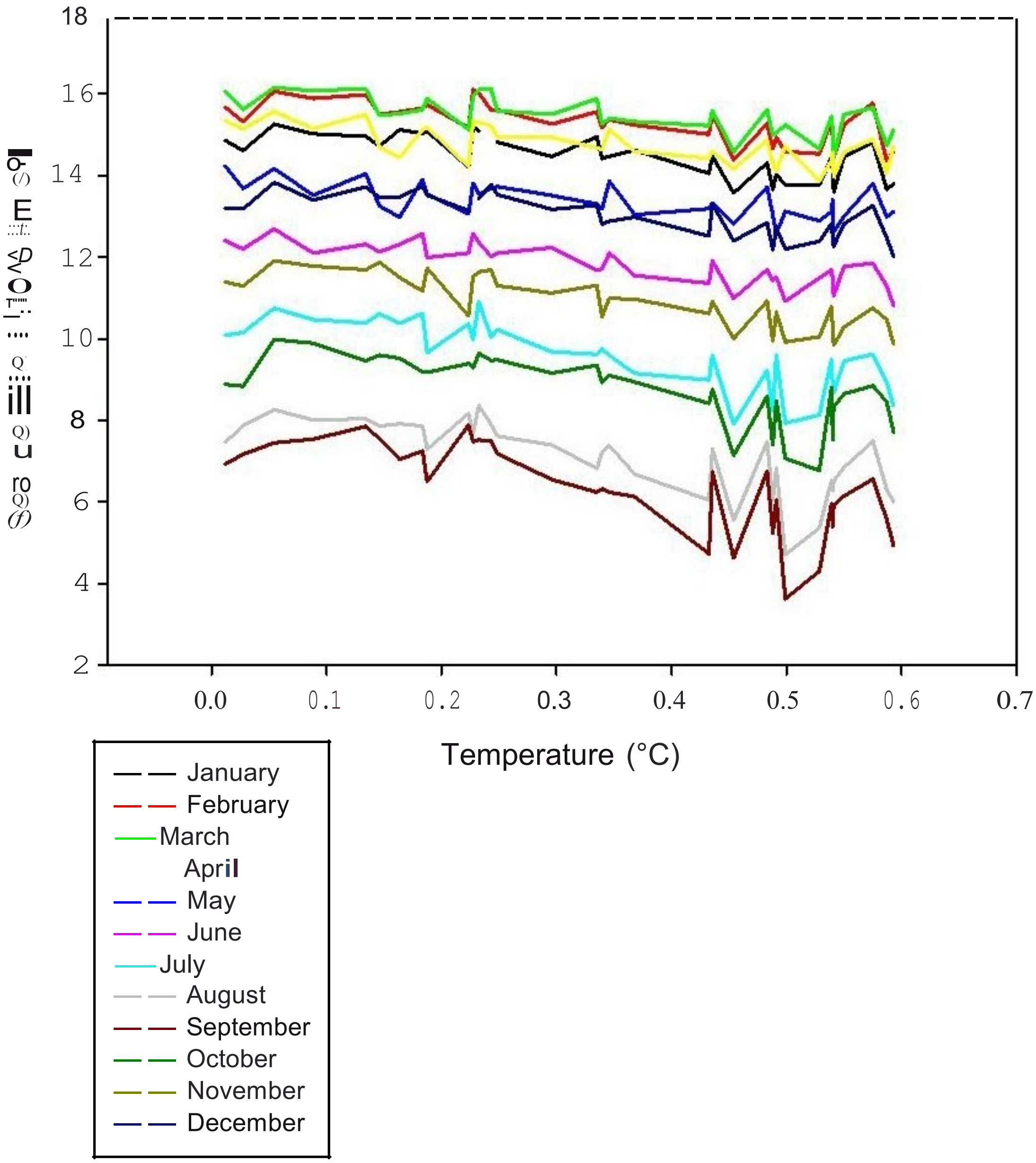
S7 FIG. S4. NSIDC armual Arctic sea ice extent for the 1980-2013 period. A breakpoint regression was used to divide the period into two distinct intervals (Table G3).



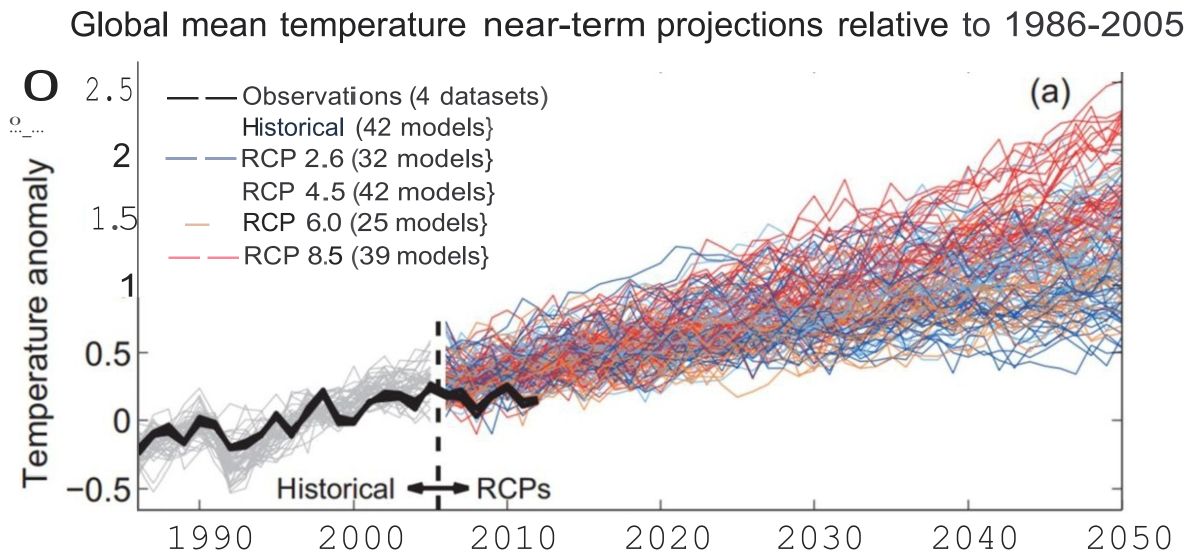
S7 FIG. SS. Observed monthly sea ice extent (NSIDC) for the Arctic from January 1980 to December 2013.



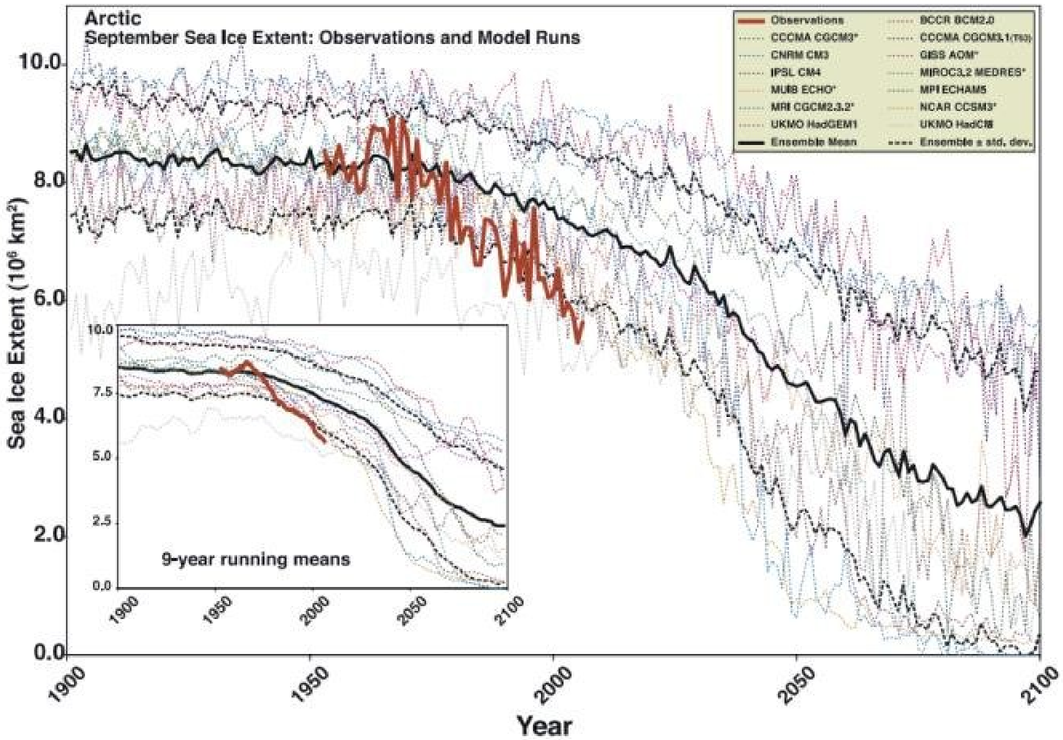
S7 FIG. S6. Annual global temperature (HadCRUT4) and monthly sea ice extent (NSIDC) for the Arctic during the January 1980 to December 2013 period.



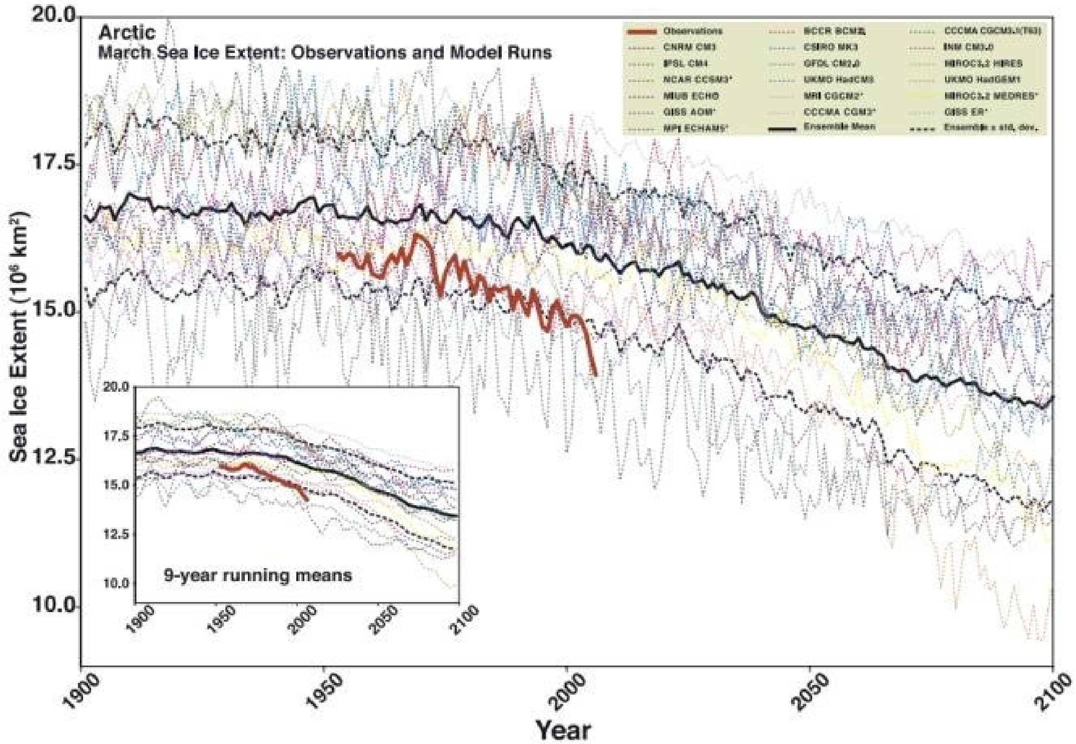
S7 FIG. S7. Global mean temperature near-term projections relative to 1986-2005 (From: Kirtman et al., (2013) Figure 11.25(a)).



S7 FIG. S8. Arctic September sea ice extent (x10^6 km^2) from observations (thick red line) and 13 IPCC AR4 climate models, together with the multi-model ensemble mean (solid black line) and standard deviation (dotted black line). Models with more than one ensemble member are indicated with an asterisk. Inset shows 9-year running means (From Stroeve et al., 2007, Fig. 1).



S7 FIG. S9. Arctic March sea ice extent (x10^6 km^2) from observations (thick red line) and 18 IPCC AR4 climate models together with the multi-model ensemble mean (solid black line) and standard deviation (dotted black line). Models with more than one ensemble member are indicated with an asterisk. Inset shows 9-year running means (From Stroeve et al., 2007, Fig. 2).



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Labrador Polar Bear Traditional Ecological Knowledge: A Time of Change

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ABSTRACT

Traditional Ecological Knowledge (TEK) on Labrador polar bears was collected from Inuit inhabiting four coastal communities in Labrador. Polar bear harvesting is a part of the culture and tradition of Labrador Inuit and contemporary hunters harvest polar bears for food and for the monetary value of the fur. Labrador polar bears are part of the Davis Strait (DS) subpopulation which has experienced a decline in body condition due to density effects and/or sea ice decline. Respondents confirmed that sea ice had declined, agreed that polar bears had increased in number; but indicated that Labrador polar bears had retained their body condition. TEK explanations for the apparent lack of effect from increased density or sea ice decline included a dramatic increase in harp seals (prey), continued seal hunting success in spite of sea ice decline, increased feeding on alternative (non-seal) food sources, and increased feeding during the open-water season when most polar bears are on-shore. Labrador Inuit hunters indicated a general satisfaction for the relatively new co-management system for polar bears, and did not identify economic constraints or loss of traditional lifestyles as a threat to cultural retention of polar bear harvesting. Both Labrador Inuit and Labrador polar bears appear to have adjusted to the changing conditions in this area in a manner that suggests a strong inherent capacity for adaptation. This was the first systematic effort to collect polar bear TEK in Labrador.

INTRODUCTION

Today's Inuit are the descendants of the Thule whaling culture (McCartney 1980) who migrated into the Canadian Arctic Archipelago from Alaska in the 11th century during the warmer conditions of the Medieval Climate Optimum. The Thule expanded south to Labrador from the Canadian Arctic around 1500 AD (Fitzhugh 1972). A cooling climate and subsequent increase in sea ice (Little Ice Age) occurred during 1650 – 1800 AD, resulting in rapid changes to Thule culture and their subsistence lifestyle (Fitzhugh 1972). Adaptation to the new conditions resulted in the emergence of a new Inuit culture and lifestyle that relied on a more diverse combination of marine and land animals. For Labrador Inuit, these changes included mastering their distinctive contemporary boreal-maritime environment (Fitzhugh 1972).

Currently ~ 2,300 Inuit reside in the Labrador Inuit Settlement Area (Nunatsiavut) in northern Labrador, representing 89.1% of the total population (Statistics Canada 2011). Labrador Inuit are distinct from other Canadian Inuit groups due to geographic and social factors (Brice-Bennett 1977). Initial European contact occurred in the early 1600s and there were intermittent trade relations by the mid-1700's (Fitzhugh 1972, Brice-Bennett 1977, Taylor 1979). Increasing participation in the European cash economy caused transition from subsistence to a traditional economy lifestyle based in settlements (Brice-Bennett 1977). Moravian missionaries and the Newfoundland and Labrador government encouraged Labrador Inuit to harvest species that produced the greatest cash return, increasing their dependence on the fur economy. As the traditional economy declined, increased wage-labour opportunities and the establishment of health care and educational services encouraged the Labrador Inuit to remain in settlements (Proctor 2012). The shift in Labrador Inuit from subsistence hunting economy to a modern economy characterized by a mix of hunting, wage labour and transfer payments requires more advanced technology to harvest wildlife, including motorized boats and snow machines (Montevecchi 2007, Dombrowski et al. 2013).

In the 1950s, the Newfoundland and Labrador provincial government relocated Inuit from the two most northern communities (Nutak and Hebron), into five remaining coastal communities (Nain, Hopedale, Postville, Makkovik and Rigolet) (Figure 1) (Evans 2012, ITK 2012). Labrador Inuit initiated a land claim proposal in 1977, which was finalized in 2005. The Labrador Inuit Lands Claims Agreement (Canada 2005) provides the legislative framework for polar bear management in Labrador. Within the Labrador Inuit Settlement Area, Inuit have the exclusive right to harvest some species of wildlife, including polar bears (*Ursus maritimus*).

Concurrent with this period of political development, the collapse of markets for seal fur (especially harp seals) (Wenzel 1978) resulted in a dramatic increase in the prey base for Labrador polar bears (Iverson et al. 2006). A recent population study of the Davis Strait (DS) polar bear subpopulation that contains Labrador polar bears found that DS polar bears were also increasing during this period in spite of climate mediated sea ice declines that began in the mid-1990's (Peacock et al. 2013; IPCC 2013).

During this period of social, economic, political, and environmental change, coastal Labrador Inuit have adapted their lifestyles while retaining traditional harvest activities and access to country foods (Felt et al. 2012; Dombrowski et al. 2013). Greater participation of Inuit in wildlife management decisions has led to increased awareness of resource managers for the value of traditional ecological knowledge (TEK) (Freeman and Wenzel 2005, Metcalf and Robards 2008, Dowsley 2009, Henri et al. 2010). We asked Labrador polar bear hunters to comment on the changes they have observed, and how these changes affected polar bears and polar bear hunting. Our study was the first systematic effort to collect TEK for Labrador polar bears.

Labrador has a seasonal-ice ecology (Amstrup et al. 2008, Thiemann et al. 2008) where the ice melts completely in the warmer months (July-October), and refreezes as a thin band of shore-fast ice and extensive offshore pack ice in the colder months. The distribution of Labrador polar bears is determined by these seasonal changes in sea ice (Taylor et al. 2001, Peacock et al. 2013). Labrador polar bears migrate seasonally as the sea ice forms and recedes; typically moving south in the early spring and returning north as sea ice recedes in southern latitudes during breakup (Harrington 1994). The open-water period causes polar bears to move on-shore and remain on land until freeze up allows them a marine platform for seal hunting. Steep terrain along the coastline appears to discourage polar bears from moving far on-shore during the open-water season (Peacock et al. 2013).

Labrador Inuit inhabit five coastal communities in Nunatsiavut (Fig. 1). Contemporary hunters mainly employ modern technology such as snow machines and motorized boats to participate in traditional hunting activities. The hunting season for polar bears runs from February to June each year. The Torngat Wildlife and Plants Co-Management Board (TWPCB) may establish, when necessary, total allowable harvests (TAH) for Non-Migratory Species of Wildlife (including polar bears). The TWPCB decisions are subject to a review by the Newfoundland and Labrador Minister of Environment and Conservation. Once the TAH is established by this co-management system, it is implemented by the Nunatsiavut Government

(NG) as a quota system. The quota is distributed among the five communities. In 2009 the NG reduced the five year waiting period between harvests per individual to three years. In January 2012, the Minister of Environment and Conservation confirmed the TWPCB decision to increase the quota from 6 bears to 12 bears, and the NG increased the period harvesters can hold the licence for from 72 hours to 7 days.

METHODS

We considered traditional knowledge to be "the knowledge, understanding and values held by Inuit based on personal observation, collective experience and oral transmission over generations" (Canada, 2005: Section 8.1.1, p. 117). The TEK provided was mainly experiential, but also included the experiences of others and explanations of what had been observed.

This study was reviewed and approved by the Nunatsiavut Research Advisory Committee and the Memorial University's Interdisciplinary Committee on Ethics in Human Research. The interview questions were developed by the Torngat Secretariat in consultation with Dr. Larry Felt of Memorial University of Newfoundland and later modified based on the initial interviews. Survey participants were identified from a list of hunters who had successfully harvested bears in Labrador and from consultation with Inuit elders, community leaders, and wildlife officials.

Interviews with 15 Labrador Inuit hunters were conducted from July to November of 2012 in the communities of Nain, Postville, and Hopedale. One hunter from Rigolet was also flown to Hopedale to participate in the study. 14 of the 15 interview participants were men who have been, or are currently, engaged in harvesting activities. One of the participants was a woman who was also active in hunting and other traditional activities.

We used semi-directed interviews to ensure that changes in abundance, distribution, behavior, and harvest practices were considered. Respondents were also encouraged to contribute any observations, explanations or stories that they wished to mention. No limits were put on the timeframes for responses. Interviews lasted approximately one hour and focused on polar bear abundance, distribution, condition, nutrition, denning, adaptation, Inuit hunting, and climate/sea ice trends. Interview responses were summarized according to these eight general topic areas. Participants were also asked to identify specific features on a physical map. Participants were sometimes asked ad-hoc supplementary questions to encourage the participant to elaborate on their answer to a given question. Participant responses were recorded on audio/video.

Interviews were translated and a database (NVivo version 10 software) was created by coding participant responses to individual interview questions. Thematic classification was employed to identify response categories and facilitate summaries of the responses. ESRI ArcGIS software was used to catalogue and produce geo-referenced summaries of the participant-identified features and to produce maps to represent each of them.

The relationships between the responses and the age, gender, or community of the interview participants were not examined due insufficient sample size. Not all respondents were asked, or provided answers to, every interview question. Results were summarized based on actual responses provided. The number of respondents who were asked a given question but did not respond was also reported. A detailed report summarizing all responses (York et al. 2014), the NVivo database, the translated interviews, the voice and video recordings, and the digital and hard copy maps were archived with the Torngat Secretariat (Jamie Snook, pers.com). Quotes provided in Results and Discussion can be cross-referenced to respondent using NVivo search options.

RESULTS AND DISCUSSION

Our summary of Labrador Inuit polar bear TEK should be regarded as descriptive rather than definitive. A larger sample of Labrador Inuit, especially women, and better representation from the two southern Labrador coastal communities would have facilitated quantitative comparisons between communities, gender, and age of category of respondents. Contemporary hunters mainly employ modern technology such as snow machines and motorized boats to participate in hunting activities (Table 1). Labrador hunters are not able to travel on or through the pack ice, so their access to polar bears is restricted to shore-fast ice during winter and spring, or on-shore retreats during the open-water season. Unless Labrador Inuit are hunting for other species or fishing, or travelling to northern Labrador for other reasons (e.g., leisure or employment), it is unlikely that they will observe polar bears while they are on-shore or in the pack ice. Thus, Labrador Inuit TEK is mainly concerned with polar bears on shore-fast ice during ice-covered season and on-shore in northern Labrador during the open-water season. Table 1 summarizes specific responses and themes that emerged from the respondents' answers. Table 2 summarizes the non-anthropogenic foods that Labrador polar bears were reported to have consumed.

"What's changed now is our method of transportation I guess. Most people hunt bears now by ski-doo. Basically you come across a new polar bear track and within a half an hour you can probably catch up with it, you know, if you don't see it right away because with the snowmobiles now, they're dependable and fast and you can go long distances, without costing any amount of money really."

The conservation governance system along the Labrador coast has historically been "top-down" with hunting regulations coming from the provincial capital, St. John's, Newfoundland and Labrador, and the research information that the regulations were based on coming from external federal, provincial and territorial initiatives (Proctor 2012). The interview protocol was deliberately structured to investigate both polar bear TEK and to determine how Labrador Inuit were experiencing current and past polar bear governance. Canada is rife with examples of tensions between aboriginal peoples and governmental regulators regarding natural resource management (Notzke 1994, Suluk and Blakney 2008), but this study contained few responses to suggest tension between resource users and government. Respondents acknowledged concurrence between polar bear hunting TEK and the current hunting regulations (e.g., protecting family groups with dependent cubs) (Table 1):

"My thing to teach young people would be that you don't kill a polar bear with cubs. We've always been taught that, you never ever kill a bear that has young ones with them."

Respondents expressed qualified support for current management strategies, including hunting seasons and the quota system (Table 1). The main criticisms of the management system concerned regulation issues that have been recently addressed. Two respondents noted that the 72-hour time limits allotted to polar bear licences was too restrictive, especially during periods of poor weather that would restrict their ability to hunt. Three respondents expressed concerns over having to wait five years to obtain a license to hunt polar bear again. One respondent stated their community was having difficulty fully utilizing its quota. The issue was that smaller communities (Rigolet, Makkovik, and Postville) might have too few active hunters to utilize the tags. If some of the hunters are restricted from obtaining a license for an extended period of time; a portion of the quota may go unused. The respondent suggested removing the waiting period or allowing unused licenses to be transferrable to larger communities where they could be used. Criticisms of the polar bear system were relatively limited to few respondents (Table 1).

Except for minor implementation issues, there was no suggestion that past or current management systems had been misguided or unfair. Most respondents did not view that their transition to co-management governance was either onerous or invasive, but rather a progression that flowed naturally from both political development and socio-economic conditions. It does appear that there has been a lag in communication of some management changes to some polar bear hunters.

"When I went polar bear hunting, I had three days. If I didn't get any, I had to take back the licence. It was not good to have only three days, especially when the weather was bad. The last time I had the licence, it was very bad weather, and where you only had a certain amount of time with the licence, and it was not good. It was bad weather for two days and I couldn't get the bear. That is something that has changed, that is something I did not like. I think things had changed though."

"I guess I will go back in time a bit, just use the example of the first polar bear that was killed here. At the time there was no quota. Even of speaking of in the 60's when that bear was killed. I'm not sure, by under provincial legislation I'm not even sure that it was legal or not. People did harvest them for whatever reason. We have right now in Nunatsiavut, we have an open. I guess this coming year again harvest of 12. So, people do respect this. As a hunter I think I could use some examples of even when it was six bears. There were times due to weather conditions and ice conditions; there were Spring's where not all six were taken. Because we have a quota of 12 that doesn't necessarily mean that hunters are going to harvest 12. In saying that I think we have to remember that defence kills especially in Nunatsiavut which might have to come off the quota."

Similarly, we found general agreement between published scientific observations and TEK for Labrador polar bears. Labrador Inuit TEK indicated that polar bears had been increasing in numbers and expanding their range, especially over the last two decades (Table 1). One of their prey species (harp seals) was reported to have increased in numbers since the harp seal harvest was curtailed in the late 1970's (Table 1). Most respondents stated that northern Labrador was a polar bear denning area, but few had traveled inland to actually observe the over-winter maternity dens (Table 1). Respondents reported temporary denning to avoid inclement weather in southern areas, which was distinguished from the birth lairs located further north.

"Bears, I've heard that they den near cliff areas, around the capes. I heard that they would be around Napartuk Bay, but now I've heard that they moved down towards the Nain area. Those kinds of areas, the snow is usually hard, it's usually hit a lot by the wind. I've seen where the polar bear actually sleeps, but I've never seen a den with the cubs."

Climate warmed and sea ice has declined seasonally across the Canadian north in recent decades (IPCC 2013). The respondents stated that climate warming had increased the duration of the open-water season and restricted the duration and sometimes the safety of the coastal shore-fast ice travel corridor for winter and spring snow machine travel (Table 1).

"We are starting to see some changes, yes. We could be on skidoo and on the sea ice tagiuk (salt). Probably in October and November. Sometimes it seems…some years it seems like it's later in December or January even before we can use siku (sea ice) in Labrador imak (water)…tagiuk (salt). Tasiks (ponds) they freeze fairly fast. Inland water freezes fairly quick. At the end of the seasons, sometimes we can be on skidoo even in June, even in July in some areas. Sometimes in these later years we have to put up our skidoo by the end of April because our ice is melting."

Labrador hunters (N=6) mentioned a significant harp seal mortality in 2010 and 2011 that coincided with poor sea ice conditions (Stenson and Hammill 2011). They noted numerous seal carcasses washing up on the shores of Labrador throughout those years, and polar bears feeding on them. However, the Canadian Department of Fisheries and Oceans reported that Northwest Atlantic harp seal numbers are near the highest levels observed since monitoring began in the 1950's (Hammill et al. 2013). Respondents noted that polar bear numbers had increased and hunting success has remained high (Table 1). Respondents also indicated that polar bears were now found close to communities more often in winter (Table 1).

"Yes they (harp seals) are more, and I really believe the reason why the bear population has gone up. You have to be very, very careful in everything that we do and everything we live on the land and I told my children and my inoKatiks (inuuqatiks) now that it doesn't make any difference. You always have to respect nanuk (polar bear) for their abilities and their strength. We're starting to find that many of our cabins are being broke into; many of our tents are being destroyed. I even had nanuk try to come inside of my boat and I believe it's a result that the population of nanuks are going up. There's more now than what we've seen before."

Rode et al. (2012) and Peacock et al. (2013) reported a decline in body condition for polar bears in the DS subpopulation and observed that this decline could be caused by decreased sea ice, density effects from increased numbers, or both. However, this decline in body condition was not observed by most hunters in Labrador ($p=0.071$, $SE=0.069$) (Table 1). Labrador Inuit hunters found polar bears to be generally healthy (in good condition), except for infrequent sightings of old, injured, or possibly sick bears (Table 1). One explanation for this apparent inconsistency could be the increase in harp seal numbers that produce young on Labrador waters in the late spring sea ice. Both Rode et al. (2012) and Peacock et al. (2013) speculate that because harp seal abundance has been increasing in Labrador and DS generally, the negative effects of sea ice decline and increased numbers may have been mitigated in the southern portion of the DS range. We suggest that an analysis of the trends in body condition determined from morphometric measurements restricted to samples from Labrador could show a different trend from the DS subpopulation as a whole.

Another explanation for the retention of body condition in Labrador throughout this period of declining sea ice and increasing population numbers is that Labrador bears are exploiting a range of alternative (not seal captured as live prey) food sources (Table 1). Although only 15 individuals were interviewed, these individuals identified 19 different food types that polar bears utilize (Table 2). They described how polar bears were continuing to hunt seals successfully in open-water, from on-shore ambush sites where land forms extended into the water, and from tide pools where seals were sometimes stranded during low tides (Table 1). Although not all respondents agreed, the majority did not indicate that increased sightings near communities were because starving polar bears were being attracted there (Table 1). While bears were reported to have been attracted to garbage dump sites, few attributed this to the bears being hungrier (p=0.2, SE=178). Instead, most respondents report that increased sightings near communities were due to an increase in abundance and an expanded range (Table 1).

"It could be a surprise sometimes of what they might eat. When they come ashore on those certain outside islands in the spring of the year, they could eat eggs, there's a lot of ducks and certain kind of birds lays eggs on the outside islands, even well up north, puffins and that, I'm sure they gets a little diet of that in the summer but seal is their primary food for sure."

"Nanuk (polar bear) can adapt. If they find dead puijik (seal) or pamiuligaks (minke whale) or anything dead they'll eat them. They'll eat grass or berries or fish. They'll eat that. Ikaluk (char) I've seen them having ikaluk before. They're very adaptable."

The majority opinion offered was that while polar bears had increased (p=0.909, SE= 0.086), and sea ice had declined (p=0.933, SE= 0.064), this had not affected Inuit hunters or polar bears appreciably; and that polar bears seemed to be adapting to the new conditions without apparent negative impacts (Table 1). Explanations for the bears' ability to adapt to the new conditions fell into three categories: 1) a sufficient number of seals continued to be caught; 2) increased consumption of alternative food sources was substituting for any declines in seal consumption; and 3) bears were working harder to obtain seals throughout the open-water period. Some respondents (p=0.428, SE=0.187) stated that they were concerned about the bears continuing to obtain a sufficient amount of food to maintain a healthy body condition. Respondents who expressed concerns over the availability of seals also mentioned the 2010 and 2011 seal mortality events. Most respondents spoke of alternative foods as a part of the long term polar bear ecology along the Labrador coast.

"From what we know it is the majority of their food is seals. Where you hunt bears they're always looking for seal holes or sees evidence of dead seals on the ice. Other than that from the last few years bears eating eggs. They can't get fat on eggs. It is just things they are coming across now, maybe they always done it. Berries and that but for the most part it's seals, or dead whales and that kind of thing that drive ashore."

Most respondents noted that modern hunting technology (e.g., snow machines and high caliber rifles) provided a distinct advantage compared to historical methods (Table 1). Although not a focus of this study, respondents often (p=0.5, SE=0.158) explained that social values and the economy of traditional harvesting activities had changed. Hunting polar bears by dog team as far as the northern tip of Labrador was a common practice until the 1970's (Wenzel 1978); however, few hunters have retained dog teams and there was no mention of polar bear hunting by dog team in modern times. The typical explanation given was that it was not practical or efficient to hunt with a dog team given the increased distance that could be covered with a snow machine. Therefore, Inuit subsistence hunting activities appears to have adapted to utilization of modern technology, however this equipment can only be obtained by participating in the modern mixed economy (Wenzel 1991).

Dombrowski et al. (2013) make similar claims with regards to Labrador Inuit reliance on modern technology to access wildlife resources. Dombrowski et al. (2013) reported that when Labrador Inuit were unable to get out on the land due to economic or employment constraints Inuit cultural identity was receding. Contrary to Dombrowski et al. (2013), the respondents in our study did not indicate that reliance on modern hunting technologies and participation in wage employment were threats to culture, or to the continuation of TEK as reliable knowledge. Comments about changes to harvesting methods were usually offered as a qualifier to the information provided by a respondent.

"Big difference 25 years ago and now. Not much more than 25 years ago, you'd have to be on a dog team down here, ski-doo's wasn't down here for many years. That was before me, I don't have memory of it or anything like that, I wasn't born, well, that's a lie, my grandfather had a dog team and I got a memory of that and that's all he used but there were ski-doo's around then, I seen it like that but It's a big difference, if they wanted to go now on a hunt they could be gone for a month sometimes and I can go in a day or a day and a half where it use to take them weeks. I sits on a ski-doo and I can drive a 140km an hour if I wanted to on this machine and travel fast if the going is good versus them with 6 and 7 dogs, a month's supply of food and most typically it used to be grandfather and his brothers, just 2 people on the Kamutik (sled) and all this gear plus your dog's food, you could almost walk faster than they can go, you know what I mean? But they had to go with dogs and carry all of these things so It use to take them a long time versus us, we can almost do it in overnight, big difference in the way we travel."

"We got a ski-doo now versus their dogs, just the technology with how we live now everything's so different, eh? Ski-doo is a big thing versus dog team, that's the big change is there now in regards of the hunt but they would never go for a polar bear hunt like that now like we do now. I love to do it because I loves to hunt. I grew up hunting all my life with my grandfather, father and it's just something that I want to keep doing but for them they hunted because they had to, they sure loved what they did but they had to do it, we don't have to do it now so much, even though we feel like we do but we're taking way more bears then there was ever took, like our elders and stuff like that. Even though I hunted, as long as they tells me I can get a licence I can go and get one but I think they should lower them quotas and put a little more thought into that."

With respect to polar bears, Labrador Inuit appear to have adapted to the shift in community distribution (elimination of northern communities). Although Evans (2012) documents the harm that occurred to Inuit that were relocated from Hebron and Nutak, and the subsequent apology and compensation that was provided by the Government of Newfoundland and Labrador; respondents did not identify community re-location as an issue for polar bear hunters or disruption in the continuity of polar bear TEK. Significant changes in polar bear distribution (increased numbers in the south in the ice covered months and in the north in all seasons), new hunting technologies (Dombrowski et al. 2013), a shift from subsistence harvesting and traditional economy to the modern mixed economy (Wenzel 2000, 2013, Montevecchi 2007), climate warming and sea ice decline, and implementation of Land Claim co-management were apparently experienced without any significant discontinuity in polar bear harvest practices or interruption in polar bear TEK.

One could view this response as a transition away from an Inuit traditional lifestyle. However, we argue that this response would be better considered as cultural adaptation. Our study indicates social and cultural resilience was sufficient to retain traditional practices and ecological knowledge through a period of extensive environmental, social, economic, and governance change, at least with respect to polar bears. Similarly, other Canadian Inuit groups have been reported to maintain aspects of distinctive traditional lifestyles (e.g., subsistence hunting and food sharing networks) even under the severe influence of political and socio-economic changes (Kishigami 2000, Wenzel 2000, Chabot 2003). We suggest that the capacity for adaptation may be an evolved and under-estimated characteristic of northern cultures, northern species and northern ecosystems. It is self-evident that Thule ancestors of the Inuit and polar bears as a species have been tested by changing conditions in the past. It is only reasonable to consider that the capacity for adaptation could be ingrained in both biological and cultural capacities. In this context, we see that TEK can provide broader information and perhaps even guidance to other cultures, in addition to accurate historical and real time observations of nature.

ACKNOWLEDGEMENTS

We thank the Torngat Secretariat and Lakehead University for their collaboration towards the development of this paper. We thank Dr. Larry Felt of Memorial University for his contribution to the design of this study. We thank the hunters who participated in this study for the generosity that they showed by sharing their knowledge and allowing it to be included in this paper. The senior author was supported by a Lakehead University graduate fellowship and a SSHRC scholarship.

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Table 1. Not all of those interviewed commented on every topic and comments from those who did respond (respondents) were not always consistent. However, within a topic area certain themes were sometimes evident. The proportion (p) of respondents who provided comments that were consistent with a thematic category and the number of those interviewed who were respondents for a given thematic category (N) is listed. The total number of those interviewed and asked to comment who did not respond (no response) is also listed.

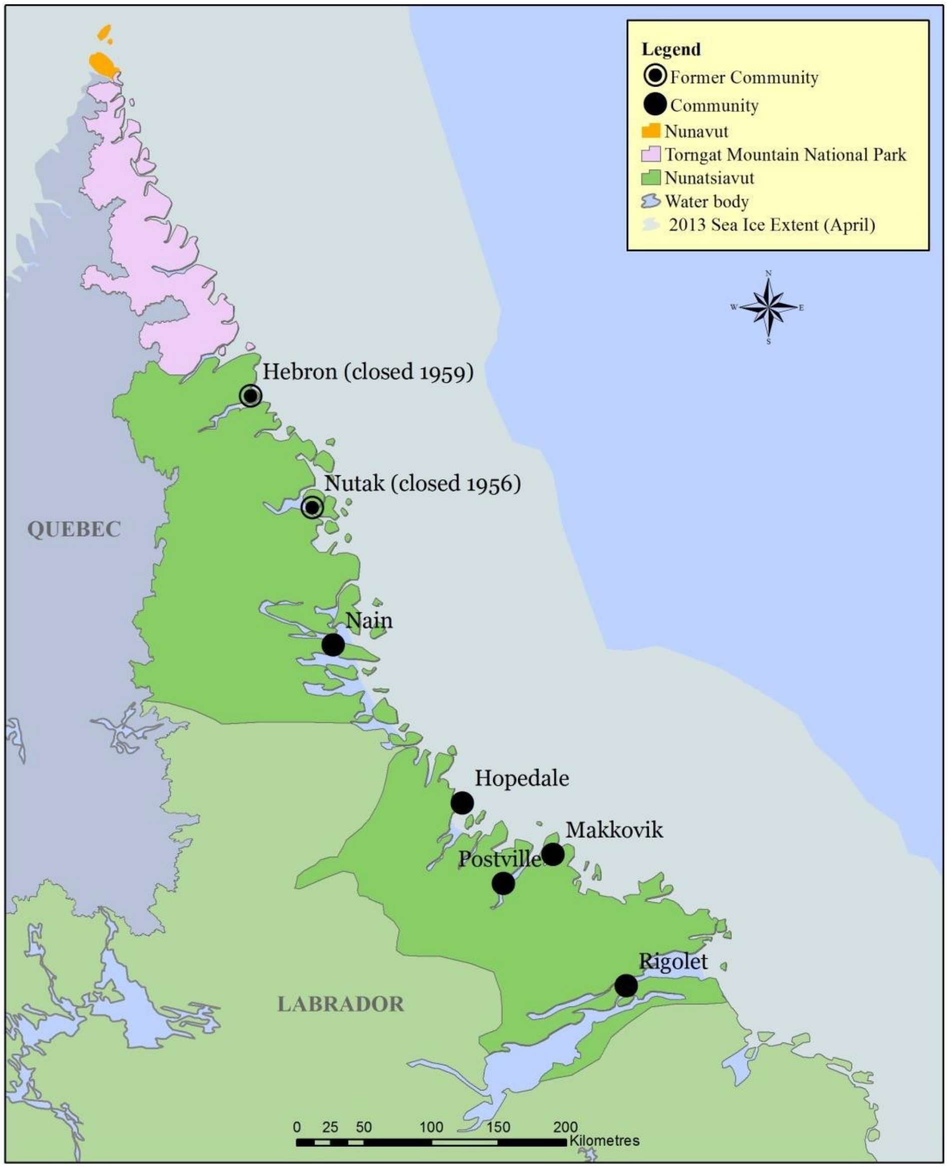
Topic	Thematic Category	p (SE)	N	No response
Abundance	Polar bears have increased in number	0.909 (0.086)	11	1
Distribution	Polar bears have expanded their range	1 (0)	3	4
	Polar bears have been encountered near communities	0.786 (0.109)	14	0
	Polar bears have been encountered near cabin areas	0.929 (0.068)	14	0
	Polar bears were attracted to communities and cabins because of garbage	0.6 (0.219)	5	0
	Polar bears were attracted to communities and cabins because of hunger or starvation	0.2 (0.178)	5	0
	Encounters near communities and cabins have increased because there are more bears	0.3 (0.144)	10	0
Nutrition	Harp seal numbers have increased in recent decades	0.571 (0.187)	7	8
	Harp seal numbers have declined in recent decades	0.428 (0.187)	7	8
	Polar bears are catching as many or more seals than in the past	1 (0)	1	0
	Polar bears are catching seal in open-water or on land	0.333 (0.272)	3	4
	Polar bears are consuming foods other than seal (alternative foods)	0.714 (0.120)	14	0
	No changes were displayed in the diets of polar bears	0.643 (0.144)	11	1
Condition	Polar bears have displayed changes in body condition	0.071 (0.069)	14	1
	Polar bears are considered to be healthy	0.286 (0.120)	14	1
	There have been changes to the sightings of sick bears	0 (0)	14	1
	There have been changes to the sightings of starving bears	0 (0)	14	1
Denning	General or specific denning areas were identified in northern Labrador	0.3 (0.144)	10	4

	Temporary (shelter) or denning areas have been identified	0.667 (0.157)	9	1
Climate/ Sea Ice Trends	Open-water season has extended in duration	0.933 (0.064)	15	0
	Changes in sea ice conditions have restricted travel	0.467 (0.128)	15	0
Adaptation	Polar bears are adapting to weather changes	0.875 (0.116)	8	2
	Polar bears are adapting to sea ice changes	0.5 (0.353)	2	1
	Catching seals on land and in water is adaptive behaviour	0.333 (0.272)	3	4
	Consuming alternative (not seals captured as live prey) food sources is adaptive behaviour	0.667 (0.272)	3	4
Hunting	Modern technology is used to hunt polar bears	0.5 (0.158)	10	1
	Hunting TEK prohibited hunting during certain seasons	0.5 (0.353)	2	5
	Hunting TEK prohibiting the hunting of mothers and cubs	0.857 (0.093)	14	0
	The quota system should be utilized for the best protection of the bears	0.5 (0.158)	10	1

Table 2. The following is a list of 19 non-anthropogenic, non-seal, alternative food sources that Labrador polar bears were noted to consume.

Food Item	**Specified Type**
Plants	Berries (unspecified), Grass, Kelp/Seaweed
Eggs	Duck, Puffin, Ambiguous (unspecified)
Birds	Duck
Fish and other aquatic species	Sculpin, Char, Porpoise, Mussels
Marine mammals	Minke whale, Beluga whale, Walrus
Terrestrial mammals	Caribou, Moose, Rodents
Carcasses/carrion	Marine and Terrestrial mammals, Unspecified

Figure 1. Labrador political boundaries and settlements map. The 2013 maximum (April) sea ice extent was also included. Data used for the production of this map was developed from NSIDC (http://nsidc.org/data/docs/noaa/g02135_seaice_index/) and Natural Earth (http://www.naturalearthdata.com/downloads/).



CONCLUSION

In my first manuscript, I provide an objective perspective on the status of Canadian polar bear subpopulations by critically considering both science (population viability analysis from published population demography studies) and traditional ecological knowledge (TEK) perspectives. My second manuscript provides a more focused and regional perspective by considering the ecology of polar bears and the polar bear co-management system from the perspective of Inuit hunters of Labrador.

I demonstrate that TEK has been more reliable than science at identifying the trends of Canadian polar bear subpopulations, particularly when scientific estimates of polar bear subpopulation trends are based on incomplete mark-recapture sampling and inaccurate harvest reporting. I reference cases in which the Canadian and international management systems (including Land Claim Wildlife Co-Management Boards and CITES) have uncritically accepted scientific results that were inconsistent with TEK, essentially favoring non-validated science over TEK (Aars et al., 2006). Subsequent scientific results (aerial surveys) and a re-examination of the Greenland harvest data for Baffin Bay have demonstrated that the scientific perspective was in error. Unfortunately, these actions resulted in conflict between managers and resource users because unnecessary harvest restrictions and economic sanctions were imposed. These negative outcomes were unnecessary and could be avoided with sound science and/or TEK validation before scientific results were used for management purposes. The Davis Strait (DS) subpopulation was designated as “declining” in 2009 by the International Union for Conservation of Nature Polar Bear Specialist Group (IUCN PBSG) while the TEK suggested the subpopulation was stable or increasing (Obbard et al., 2010). No additional studies have been conducted on this subpopulation since 2007, yet in 2013 the IUCN PBSG revised the status of polar bears from declining to stable/increase (IUCN PBSG, 2013). The reason for the change was not a problem with the mark-recapture data; it was an error with how these data were analyzed. When the error was corrected the scientific perspective corresponded to the TEK perspective. The DS mark-recapture survey results (Peacock et al., 2013) supported the increase in population numbers reported by Nunavut, Quebec, and Labrador Inuit at current quota levels. The respective co-management boards identified a shared quota increase that was implemented in 2012 and that satisfied Labrador Inuit. I conclude that sound scientific estimates of subpopulation trends will agree with TEK perceptions of trend, perhaps because TEK is held collectively, and because it is updated and validated continuously.

This thesis demonstrates the importance of accurate and timely estimates of population numbers and rates of survival and recruitment for Canadian polar bear subpopulations. The current removal rates are close to the maximum that can be sustained, and Arctic sea ice is declining. It is prudent to ensure that estimates of demographic parameters are both accurate and current so they can be used effectively for revising estimates of the sustainable removal rate if a subpopulation’s carrying capacity declines. Without reasonably current data and accurate (sound sampling) estimates of vital rates and population numbers, population viability analysis (PVA) can be biased and unreliable. However, TEK by itself is not able to make recommendations for sustainable removal rates except through a process of trial and error. Both manuscripts suggest that management is enhanced by scientific perspectives validated by TEK. Thus sound management of polar bears also requires a concurrent requirement for accurate and recent TEK on polar bear trends. When both perspectives agree, management practices can be modified to ensure good conservation practices with harvester support. The Labrador TEK study demonstrates that an adaptive and responsive co-management system based on sound information results in user support for the conservation regulations.

The Labrador TEK manuscript also demonstrates the utility of TEK and the capacity for adaptation that seems inherent to northern cultures and northern species. Labrador TEK not only provides insight into polar bear ecology, but also provides an understanding of the social, economic, and political changes that have occurred in coastal Labrador. Both Labrador Inuit and polar bears appear to have adapted to a variety of recent changes without suffering negative consequences. Labrador TEK disputes the generalized declines in body condition for the DS subpopulation as a whole (Rode et al., 2012), and concurs with scientific conjecture that the dramatic increase in harp seals in southern DS may have mitigated the effects of sea ice decline there (Peacock et al., 2013). Labrador Inuit also identify a number of alternative prey species and suggest that adaptive behaviours may also have contributed to retention of polar bear body condition in this area. Labrador Inuit have also demonstrated the ability to adapt as they have maintained aspects of their distinctive traditional lifestyle in spite of the severe influence of political and socio-economic changes. Northern cultures have traditionally been regarded as pristine and lacking the ability to change without compromising traditional values and/or cultural identity (Wenzel, 2013). The TEK collected regarding Labrador Inuit polar bear hunters suggests otherwise.

This thesis contributes to our understanding of polar bear population ecology, Canadian co-management systems, and the politics that can affect a high profile “species at risk”. Methodological errors and extended monitoring intervals have contributed to polarized perspectives between managers and resource users in some cases. Additionally the use of polar bears as a poster species for Environmentalist Non-Government Organizations (e.g., World Wildlife Fund (WWF) and Polar Bears International (PBI)) concerned about climate warming has shifted the focus of polar bear conservation from harvest management to climate crisis control. The result has been negative for Canadian polar bear co-management in spite of the increased profile of polar bears. Climate is warming, Arctic sea ice is declining, and declines in body condition have been reported for some subpopulations (Stirling et al., 1999; Obbard et al., 2006; Rode et al., 2007; Stirling et al., 2008; Rode et al., 2012). However, at least one other subpopulation has experienced an increase in body condition during a period of sea ice decline (Rode et al., 2014), and in most cases Canadian subpopulations are exhibiting numbers that exceed historical population sizes. Increased monitoring is warranted and recommended. This thesis suggests that Canadian polar bears do not currently appear to be in a state of climate crisis. Concurrently TEK studies should be conducted to provide a measure of concordance to insure that scientific results are consistent with the real-time range wide observations that comprise TEK.

Future research could include incorporation of environmental dynamics to polar bear demographic models (e.g., sea ice changes, industrial development impacts, and tourism effects).

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