Abstract

Establishing historical baselines of species' populations is important for contextualising present-day population trends, identifying significant anthropogenic threats, and preventing a cultural phenomenon known as 'shifting baseline syndrome'. However, our knowledge of historical baselines is limited by a lack of direct observation data on species abundance pre-1970. We present historical data of species-specific fur harvests from the Canadian government and Hudson's Bay Company as a proxy for estimating species abundance over multiple centuries. Using stochastic stock reduction analysis originally developed for marine species, we model historical population trends for eight mammals, and assess population trends based on two different baseline years: 1850 and 1970. Results show that population declines are significantly greater when using an 1850 baseline, as opposed to a 1970 baseline, and for four species, the population trend shifted from a population increase to a decrease. Overall, the median population change of the eight species changed from a 15% decline for 1850, to a 4% increase for 1970. This study shows the utility of harvest data for deriving population baselines for hunted terrestrial mammals which can be used in addition to other historical data such as local ecological knowledge. Results highlight the need for developing historically relevant population baselines in order to track abundances over time in threatened species and common species alike, to better inform species conservation programs, wildlife management plans and biodiversity indicators.

Keywords: Historical baselines; shifting baseline syndrome; population estimation; mammal conservation; Hudson's Bay Company; North America

1. Introduction

Species population declines and extinctions undermine the functioning and resilience of ecosystems on which humans and wildlife depend (Cardinale et al. 2012; Oliver et al. 2015). To monitor and respond to species losses, changes in population abundance are used as a sensitive metric of change (Collen et al. 2011; Shoemaker and Akçakaya 2015) and have been incorporated into globally adopted biodiversity indicators such as the Living Planet Index, which tracks changes in vertebrate population abundance from 1970 (Collen et al. 2009). However, data on population abundance typically become scarcer beyond a few decades from the present, prior to the implementation of species monitoring programmes (Willis et al. 2005; Bonebrake et al. 2010).

Knowledge of historical populations acts as an antidote to ‘shifting baseline syndrome’; a phenomenon in which with each new human generation comes a lowered expectation of a species population norm (Pauly 1995; Kahn and Friedman 1995; Soga and Gaston 2018). Historical population baselines have many practical policy implications, for example when defining population recovery and conservation legacy, deciding harvest quotas, and influencing the general public’s perception of a species (Papworth et al. 2009; Davies, Colombo, and Hanley 2014; Roman et al. 2015; Akcakaya et al. 2018; see Figure 1a). Additionally, estimating historical populations can help to differentiate between a population trend that is unidirectional or cyclical, such as the Atlantic Multidecadal Oscillation inducing bidirectional changes in fish abundance (Jackson et al. 2001; Willis et al. 2007; Sundby and Nakken 2005; see Figure 1b). Without long-term measurements, observers may misattribute downward phases of natural population cycles as human-caused population declines (Koslow and Couture 2013). Finally, historical population data can help to identify historic drivers of population change (see Figure 1c), which is important for quantifying the relative significance of each past and present threat in order to develop threat-specific management strategies and inform future scenario modelling (Baker and Clapham 2004; Pinnegar and Engelhard 2008).

Many techniques available to reconstruct historical population baselines emerged from the discipline of marine historical ecology (Lotze and Worm 2009). Faced with the need to sustainably manage fish stocks, fisheries researchers have used recorded history (e.g. ‘local ecological knowledge’) (Sáenz-Arroyo et al. 2005; Turvey et al. 2010), archaeogenomic data (e.g. analysis of relative stable isotope concentrations) (Finney et al. 2002), and fish stock assessments from historical catch data (Myers and Worm 2003; Baker and Clapham 2004) to extrapolate population size over time and capture stock collapses that pre-date direct monitoring.

Recorded history has also provided us with historical population estimates for terrestrial species, although not as frequently as in the marine realm. These studies are extremely valuable in painting a picture of past population condition (Cole and Woinarski 2000; Rowe and Terry 2014), but with each historical data source comes its own unique set of limitations. For instance, museum and fossil records are often patchy and taxonomically biased, and local ecological knowledge generally only covers a couple of generations spanning less than 100 years (Miller 2011). Here, we add to our growing knowledge on reconstructing population baselines by focussing on harvest data of terrestrial mammals as another data source which holds great potential in historical baseline reconstruction.

Reports from the Hudson's Bay Company (HBC), Canada, have been previously used to document lynx and muskrat population cycles (Elton and Nicholson 1942a; Elton and Nicholson 1942b), predator-prey dynamics of lynx and snowshoe hare (Krebs et al. 1995), and the potential roles of climate, productivity and disease in these cycles (Gamarra and Solé 2000; Yan et al. 2013; Row et al. 2014). Here, we (a) show the utility of these harvest data to reconstruct historic populations by applying a stochastic population model first developed for marine vertebrates (Christensen 2006), (b) use these population reconstructions to demonstrate that baselines differ when using over 100 years of data compared to less than 50 years of data and (c) show that choice of different baseline years results in different interpretation of estimated population trends.

2. Methods

2.1 Reconstructing historical abundance trends

To reconstruct historical trends in terrestrial mammal abundance, we used a stochastic stock reduction analysis (SSRA) originally developed by Walters *et al.* (2006) to analyse trends in fish populations. This method uses a simple growth model, and can be applied to species for which we have limited knowledge of life history parameters and catch-per-unit-effort (Kimura, Balsiger, and Ito 1984). The model and method outlined below was described in detail by Christensen (2006) for establishing historical baselines of marine mammals:

$$N_{t+1} = N_t + r_{max} N_t \left(1 - \frac{N_t}{K}\right) e^{w_t} - C_t \qquad (1)$$

where N_t is the number of individuals in a population at time t, r_{max} is the intrinsic rate of increase, K is the carrying capacity, w_t is the error term at time t, and C_t is the number of individuals harvested annually. The population is assumed to be at K at the start of hunting, and the model implies that there are no errors in our records of C_t.

Using a Bayesian approach, we randomly drew from a uniform prior distribution of K, which was bounded by a lower and an upper estimate, and normal prior distributions of r_{max} and w_t values. We used process error terms (τ_w) for the standard deviation of w_t values. This generated a N_t trajectory which was dependent on C_t (eq 1). We then calculated the likelihood (L) of deriving the known abundance estimate for the current population (y_t) based on the estimated N_t using eq. 2:

$$L(y_t \mid r_{max}, K, w_t) = n\left[\log(\sigma_y) + \frac{1}{2}\log(2\pi)\right] + \sum_{i=1}^{n} \frac{z_t^2}{2\sigma_y^2}$$

(2)

where n is the number of ~~current population~~ abundance estimates (y_t) available for each species, σ_y is the observation error (standard deviation of the abundance estimate, y_t), and z_t is the log-normal residual, calculated as:

$$z_t = \log(N_t) - \log(y_t)$$

(3)

Equation 1 to 3 were repeated 200,000 times. Generated N_t values were then resampled using the importance sampling procedure by Schnute (1994) and McAllister and Ianelli (1997), to estimate a posterior probability density function. Finally, we calculated the median and the 95% credible interval of the N_t trajectory, after computing the marginal posterior distributions of K.

2.2 Harvest data

We conducted a literature search for fur trade records of Canadian mammals from the Hudson's Bay Company (HBC). The HBC fur clothing business, originally London-based, became established in Canada in 1670 in response to the UK demand for fur and the abundance of furbearing mammal species in Canada (Simmons 2007). Expanding its area of influence over time, the company monopolised the North American fur trade from 1821 onward, and eventually operated over an area that spanned around one quarter of the extent of North America (Erickson 2015; Colpitts 2017; Company 2017).

Published annual HBC sales accounts for 25 mammals were acquired from Poland (1892) for the time period 1751 – 1847 and Jones (1914) for the time period 1848 – 1909. Additional HBC data was supplied for the American badger (*Taxidea taxus*), beaver (*Castor canadensis*) and Canadian lynx (*Lynx canadensis*) by the NERC Centre for Population Biology (NERC Centre for Population Biology 2010), Hewitt (1921) and Elton and Nicholson (1942) respectively. Canadian fur data for 1919 – 1970 were retrieved from Novak (1987) and data for 1970 – 2009 from the National Bureau of Statistics (NBS), Canada (2010). Using the individual number of furs across datasets as a proxy for the number of individuals taken for trade resulted in time series harvest data for 25 mammal species ranging from 1722 – 2009 (electronic supplementary material, Table S1). Of these 25 species, only thirteen had harvest data from two or more continuous datasets up to and including 2009 (the last year of data available from NBS, Canada). Harvest data for species modelled is supplied in the supplementary material (Table S2).

Jones' (1914) data represented the number of furs sent to HBC headquarters, and Poland (1892) represented the year the fur was sold at the London auctions; therefore, these data were backdated by two years and one year respectively following Elton and Nicholson (1942a). All time-series had a data gap for the years 1910 – 1918 and 2002. We estimated values for these years using a log-linear interpolation for all species except the lynx. Due to the 9-year cyclic nature of the lynx harvest data, a linear interpolation was applied around the peak year (1914) of the cycle for 1910 – 1914, and 1914 – 1918.

We can be fairly confident that the numbers reported from annual sales accounts is an underestimate of the actual number of individuals caught (e.g., Elton and Nicholson 1942a), due to escape from traps, illegal trade by trappers, lost or stolen carcasses in transit to London's auction house, bookkeeping errors, etc. Although statistical approaches such as employing a correction factor are available to counteract the undocumented loss, we select to estimate the *minimum* population decline based on reported numbers only.

2.3 Informative priors

In order to perform the Bayesian SSRA model, population growth rate (r_{max}) and ~~recent population~~ current abundance estimate(s) (y_t; 1979 – 2008) in eq. 1 and 2 were gathered from peer-reviewed publications for each species (electronic supplementary material, Table S3; Table S4). If countrywide ~~population~~ abundance estimates (y_t) for Canada were unavailable, province-wide abundance estimates were extrapolated to the entire current species range within Canada. Species range maps were downloaded from the IUCN Red List of Threatened Species (IUCN 2014). Of the 13 species with harvest data, muskrat (*Ondatra zibethicus*), marten (*Martes americana*), red fox (*Vulpes vulpes*), mink (*Neovison vison*), and coyote (*Canis latrans*) had no countrywide or province-wide estimates of ~~y_t~~ abundance and were therefore removed from the study.

The contribution of observation error (σ_y) to the total error term ($\kappa = 0.1$) for each species was determined by selection criteria adapted from Christensen (2006), based on the source and uncertainty (p) associated with the abundance estimate, y_t. Province-wide extrapolations of y_t were assigned the highest uncertainty and direct estimates were assigned the lowest uncertainty (electronic supplementary material, Table S5). The remaining proportion of the total error term was allocated to the independent process error (τ_w), which represented the ecologically mediated fluctuations within a true population size (Ahrestani, Hebblewhite, and Post 2013), such that $\sigma_y = \sqrt{p} * \sqrt{\kappa}$ and $\tau_w = \sqrt{1-p} * \sqrt{\kappa}$.

As no data exist on historical carrying capacity (K) of populations, we estimated priors for K from historical range maps from the 1900s for each species obtained from Seton (1909). We georeferenced each species' map in ArcGIS v9.3 (ESRI 2008) to obtain the historical range in km^2, assuming all of the historical range was viable habitat. We collected present day minimum and maximum density estimates (individuals/km^2) from the literature and produced a lower and upper bound for K by extrapolating the density estimates to the historical range. A uniform prior distribution was then drawn, which assumes that K is constant over time (electronic supplementary material, Table S6). We carried out elasticity analyses to assess the impact of variation in our estimates of informative priors on population change.

2.4 Analysis of historical baselines

We reconstructed historical population trajectories for eight species; the American badger, Arctic fox (*Vulpes lagopus*), beaver, black bear (*Ursus americanus*), bobcat (*Lynx rufus*), Canadian lynx, fisher (*Martes pennanti*) and polar bear (*Ursus maritimus*). We quantified population change for each species as follows:

$$Population\ Change = \frac{N_{2009} - N_b}{N_b} * 100 \tag{4}$$

where N_b represents the population estimated at the given baseline year b. Population change was estimated for two baseline years: 1) 1850, the earliest year that all species had available data; and 2) 1970, the baseline year of the Living Planet Index (McRae, Deinet, and Freeman 2017). We then conducted a t-test to assess the difference in population change across the two baselines. We also calculated the median population change for all eight species under the two baselines and recorded the degree and direction of change.

2.5 Single versus multiple ~~threats~~ causes of mortality

Modelling population estimates based on fur trade alone overlooks other reasons animals are removed by humans; fur trade, trophy hunting, self-defense, illegal kills, individuals found dead, and handling by scientists. We applied the SSRA to another dataset for the polar bear, substituting fur trade records from 1970 onwards with data published by the IUCN Species Survival Commission (SSC) Polar Bear Specialist Group, that documents number of bears killed between 1970-2008 from all causes listed above (Derocher and Commission. 1998; Lunn et al. 2002; electronic supplementary material, Table S7).

2.6 Elasticity analysis

We carried out elasticity analysis (de Kroon et al. 1986) to examine the relative effect of parameter selection on model prediction. We independently halved and doubled each model parameter in turn (intrinsic growth rate (r_{max}), number of individuals harvested (C_t), current ~~population size~~ abundance estimate (y_t), historical carrying capacity (K), and observation error (σ_y)), and calculated elasticities (e) of the 1850 – 2009 population change, where higher e means higher proportional change in the ~~parameter~~ population (de Kroon et al. 1986; Benton and Grant 1999; Hunter, Moller, and Fletcher 2000). Y represents the population change caused by the altered parameter, *X*:

$$e = \frac{\%\Delta Y}{\%\Delta X} \qquad (5)$$

All analyses were carried out in the statistical software RStudio version 1.0.143 (RStudio Team 2015).

3. Results

3.1 Analysis of historical baselines

The median population change across the eight species for 1850 – 2009 was a 15% decrease (-0.1%/yr), whereas populations between 1970 – 2009 showed a 4% increase (0.1%/yr) (paired t-test: t = -3.036,1 d.f. = 7, $p = 0.002$, $n = 8$; Table 1, Figure 3a). Choice of baseline year resulted in a switch from a downward population trend for the period 1850 – 2009 to an upward trend for 1970 – 2009 for four species (Arctic fox, bobcat, polar bear, beaver) (Figure 3b; Table S8; Figure S1). Six species exhibited a more severe annual rate of population change for the 1850 – 2009 period ~~as opposed to the~~ compared to trends derived from a 1970 baseline (Table 1; Figure 3b), with this pattern holding for five species when the annual rate of population change is considered (Table 1).

3.2 Single versus multiple ~~threats~~ causes of mortality

Analysis of the polar bear data that encompassed all recorded mortality events from 1970 onwards shows that an additional 294 (range = -57 to 538) individuals were killed in comparison to the number harvested for furs alone (electronic supplementary material, Table S7). This altered the polar bear population decline using the 1850 baseline from 22% based on fur harvest data alone to 41% using the IUCN/SSC data (Figure 4).

3.3 Elasticity analysis

Parameters ranked in order of lowest to highest elasticity (proportional change in the population) were observation error, carrying capacity, harvest data, current ~~population~~ abundance estimate and intrinsic growth rate (Figure 5, Electronic supplementary material, Table S9). In other words, small changes to intrinsic growth rate and abundance estimates cause larger alterations to the model output. Species ranked in order of lowest to highest average elasticity for all informative priors were fisher, black bear, American badger, arctic fox, bobcat, polar bear, Canadian Lynx and beaver.

4. Discussion

Our study demonstrates that for eight species of Canadian mammals, choice of baseline year greatly affects our understanding of historic population change. Collectively, using an 1850 baseline year rather than 1970 significantly altered the population trend. Analysis of individual species demonstrated that deriving population change from the 1850 baseline resulted in four species shifting from a population increase since 1970 to a population decrease of between 0 and -22% since 1850, and the magnitude of annual population change becoming more severe for six species (Figure 3). Interestingly, the polar bear, the only species to be listed as Vulnerable on the IUCN Red List (Wiig et al. 2015), was one of the four species to show a shift in population trend. From 1850, the population declined, reflecting the polar bear's IUCN conservation status, whereas the trend from 1970 showed the population as increasing.

Population time-series data allow for the detection of declines before species reach critical status (Balmford and Bond 2005; Di Fonzo et al. 2016; Collen et al. 2011). However, few long-term population studies used data more than 100 years old (e.g., 15%; Bonebrake et al. 2010). Biodiversity indicators that are reliant on such population time series data, such as the Living Planet Index (LPI), therefore often operate on baselines set to post-1950 (Bonebrake et al. 2010). At least for temperate systems, the LPI dataset may not date back far enough to detect often precipitous population declines which occurred prior to 1970 (Watson et al. 2005). For example, while some population recovery is observed in Europe since 1970 (Deinet et al. 2013), species are often recovering from declines which had reduced populations to a fraction of their historical population sizes by or prior to 1970. These short-term baselines are likely a result of the relatively recent advent of conservation biology in the 1970s and 1980s and corresponding emergence of long-term monitoring data for ecology and conservation (Collen et al. 2009).

In this study, we were able to model seven species currently listed as Least Concern on the IUCN Red List and not assessed under the Canadian Wildlife Species at Risk assessment (COSEWIC 2019). However, for five relatively common species: muskrat, marten, red fox, mink, and coyote that we collected harvest data for, a lack of recent population estimates at the regional or national scale prevented population modelling. Species are rarely subject to monitoring until declines are apparent and they may be of conservation concern (Bonebrake et al. 2010), while threatened species are often prioritised for monitoring, given limited financial resources and capacity (Gaston and Fuller 2008). As we are frequently witnessing population declines in common species, it is imperative that population monitoring also targets abundant or common species (Collen et al. 2011). This proactive approach enables early detection of declines, rather than a late reactive response when a species has already declined extensively, thus approaching extinction. This is of particular importance as abundant species play important parts in the functioning and maintenance of ecosystems (Winfree et al. 2015).

The strengths of using HBC data to estimate population change is the consistent unit over time (number of furs traded) and high temporal resolution (annual data), which overcome limitations oftentimes associated with historical records (Balmford and Bond 2005; Mcclenachan, Ferretti, and Baum 2012). While HBC data provides a very unique set of data, hunting statistics are available in many countries (e.g. Agetsuma 2018), and have a long tradition in regions such as Europe (Deinet et al. 2013; Hewitt and Hewitt 2015). Hunting bag records were used in a recent study to analyse long-term population trends across a number of European countries for several game bird, ungulate and carnivore species, although the study only reached back as far as 1970 (Reimoser and Reimoser 2017). In the UK, the voluntary National Gamebag Census was formally established in 1961 but resulting records on game birds and predatory species extend back to 1900 for many species (Whitlock, Aebischer, and Reynolds 2003; Aebischer and Baines 2008). Models such as the one utilised in this study play an important role not just in deriving generalised trends, but also population estimates using such data repositories. Hunting statistics can then complement, or be used in tandem with, data from different sources such as transgenerational local ecological knowledge to obtain historical population baselines.

Despite our robust methods, population estimates from historical information such as harvest data are of lower quality than those derived from direct counts and current population monitoring, and uncertainty surrounding the estimates is inevitably greater. While some losses are not reported, resulting in a potential underestimate of the population (see methods 2.2), the spatial pattern of hunting that occurred throughout Canada in the past is likely to lead to an overestimate of hunting offtake of species. Hunting efforts saw a large westward expansion between 1783 - 1821, driven in large part by local depletion of beaver in the 'Canadian Shield' (Hope 2016; Figure 2). New areas such as the Red River Settlement, which was established in 1811, would have allowed access to previously unharvested subpopulations of mammals, and a boost to total harvest numbers. On the other hand, sources suggest that HBC was attempting to conserve beaver and bison in the 1820s (Colpitts 2017), presumably by enacting policies to help local population recovery, thus lowering total harvest. These local variations in harvest effort are not visible in our dataset, and the impact of uncertainty on our population estimates is illustrated by the mid-range elasticity of our model to a halving and doubling of harvest data (*c*) (Figure 5; Table S7). To some degree, our baseline of 1850 may help circumvent some of the previous geographical shifts or expansions in hunting effort. However, next steps should endeavour to account for spatial variation in hunting records over time by adding a spatially explicit component to the stock reduction analysis.

In addition to hunting, other drivers of population change are likely to impact our species, and focus on only a single cause of population offtake is likely to underestimate population trends over time. Substituting IUCN/SSC data on polar bear mortality via direct contact with humans into the final 40 years of our model changed the estimated decline from 25% (based on fur hunting data alone) to 41% (Figure 4).

As well as providing hunting numbers, the IUCN/SSC polar bear data also includes 'individuals found dead'. This statistic includes mortality events which may be human-mediated or a result of natural causes, such as starvation from lack of sea ice. While hunting is a key human-mediated impact, it is but one of several sources of population decline facing mammals in North America (Brook, Sodhi, and Bradshaw 2008; Grooten and Almond 2018; Spooner, Pearson, and Freeman 2018).

Other anthropogenic drivers of population change such as habitat loss and the introduction of non-native species are not included in this analysis, again rendering our estimates conservative. For example, habitat loss affected the vast majority of threatened mammals in Canada, with direct human-caused mortality coming second (Imre and Derbowka 2009). Although modelling the impacts of all negative and positive anthropogenic effects on abundance is beyond the scope of this study, we can be fairly certain that these threats would play a role in determining the population size, intrinsic growth rates and carrying capacity within regions of Canada. As anthropogenic threats are dynamic (Wilcove et al. 1998), the degree of impact on the population will also alter over time. This highlights the need to quantify the impact of other threats which may influence these population parameters differentially over time (e.g. land use change altering carrying capacity) and incorporate these into baseline population models. One way of achieving this could be to allow carrying capacity K to vary across the time frame of our model in response to observed range contractions. In choosing a baseline for constructing historic population trends, we are ourselves guilty of ignoring what happened to our species prior to our chosen year of 1850. However, we recognize that the fur trade existed far back beyond this. Hunting for fur was carried out by Native Americans and European settlers alike; during the 17th century, the French owned the monopoly of the Canadian fur trade and by 1717, HBC had set up six trading posts (Voorhis 1930). By 1850, the fur trade had impacted wildlife on the east coast; between 1860 and 1920 the sea mink (*Neovison macrodon*) reportedly went extinct (Mowat 2012; Black, Reading, and Savage 1998). While we use 1850 to demonstrate the importance of a historical baseline for as many species as possible, the term 'baseline' is not used here to imply a 'start' date, nor do we claim that one date will suffice. Rather, historical baselines should be viewed as a sliding window, and in conservation, it is important to move that window as far back as possible through history, to provide better context for species' current population status.

5. Conclusions

By failing to estimate historical baselines, we may miss the historical demise of populations which have been exploited by humans since at least the 18th century in Europe and North America (Deinet et al. 2013), and adversely influence our perception of what constitutes species population norms. This may affect how scientists, decision makers and the general public perceive the growth of a population as a result of conservation action and species protection. While in many northern hemisphere regions, population recovery to an 1850 (or earlier) baseline may no longer be viable, due to the extensive anthropogenic land use change that has occurred in the interim, our current understanding of population recovery may focus on population sizes which are too small and could leave populations vulnerable to future threats or dependent on continued conservation action. We encourage the use of well-documented and quantifiable historical records in developing population baselines, thus establishing a broad temporal scale over which to analyse species population trends. Armed with such data, we can improve our communication about population declines and set ambitious goals for realistically achievable population recovery (Akcakaya et al. 2018), make better-informed conservation management decisions, and improve our historical ecological awareness.

Supporting Materials

All datasets supporting this article have been uploaded as part of the supplementary material.

Declaration of competing Interests

We have no competing interests to declare.

References

Aebischer, Nicholas J, and David Baines. 2008. "Monitoring Gamebird Abundance and Productivity in the UK : The GWCT Long-Term Datasets." *Revista Catalana d'Ornitologia* 24: 30–43.

Agetsuma, Naoki. 2018. "A Simple Method for Calculating Minimum Estimates of Previous Population Sizes of Wildlife from Hunting Records." *PLoS ONE* 13 (6): 1–10.

Ahrestani, Farshid S, Mark Hebblewhite, and Eric Post. 2013. "The Importance of Observation versus Process Error in Analyses of Global Ungulate Populations." *Nature Scientific Reports* 3 (3125): 1–10. https://doi.org/10.1038/srep03125.

Akcakaya, H. Resit, Elizabeth L Bennett, Thomas M Brooks, Molly K Grace, Anna Heath, Simon Hedges, Craig Hilton-taylor, et al. 2018. "Quantifying Species Recovery and Conservation Success to Develop an IUCN Green List of Species." *Conservation Biology* 32 (5): 1128–38. https://doi.org/10.1111/cobi.13112.

Baker, C. Scott, and Phillip J. Clapham. 2004. "Modelling the Past and Future of Whales and Whaling." *Trends in Ecology and Evolution* 19 (7): 365–71. https://doi.org/10.1016/j.tree.2004.05.005.

Balmford, Andrew, and William Bond. 2005. "Trends in the State of Nature and Their Implications for Human Well-Being." *Ecology Letters* 8 (11): 1218–34. https://doi.org/10.1111/j.1461-0248.2005.00814.x.

Benton, Tim G, and Alastair Grant. 1999. "In Evolutionary and Population Ecology." *Trends in Ecology and Evolution* 14 (12): 467–71.

Black, D W, J E Reading, and H G Savage. 1998. "Archaeological Records of the Extinct Sea Mink , Mustela Macrodon (Carnivora : Mustelidae), from Canada." *Canadian Field-Naturalist* 112 (1): 45–49.

Bonebrake, Timothy C., Jon Christensen, Carol L. Boggs, and Paul R. Ehrlich. 2010. "Population Decline Assessment, Historical Baselines, and Conservation." *Conservation Letters* 3 (6): 371–78. https://doi.org/10.1111/j.1755-

263X.2010.00139.x.

Brook, Barry W., Navjot S. Sodhi, and Corey J A Bradshaw. 2008. "Synergies among Extinction Drivers under Global Change." *Trends in Ecology and Evolution* 23 (8): 453–60. https://doi.org/10.1016/j.tree.2008.03.011.

Canada, Statistics. 2011. "Fur Statistics, Canada." Ottowa, Ontario. www.statcan.gc.ca.

Cardinale, Bradley J., J. Emmett Duffy, Andrew Gonzalez, David U. Hooper, Charles Perrings, Patrick Venail, Anita Narwani, et al. 2012. "Biodiversity Loss and Its Impact on Humanity." *Nature* 486 (7401): 59–67. https://doi.org/10.1038/nature11148.

Christensen, L B. 2006. "Reconstructing Historical Abundances of Exploited Marine Mammals at the Global Scale." University of British Columbia.

Cole, Jeffry R., and J. C.Z. Woinarski. 2000. "Rodents of the Arid Northern Territory: Conservation Status and Distribution." *Wildlife Research* 27 (4): 437–49. https://doi.org/10.1071/WR97053.

Collen, B., L. McRae, S. Deinet, A. De Palma, T. Carranza, N. Cooper, J. Loh, and J. E. M. Baillie. 2011. "Predicting How Populations Decline to Extinction." *Philosophical Transactions of the Royal Society B: Biological Sciences* 366 (1577): 2577–86. https://doi.org/10.1098/rstb.2011.0015.

Collen, Ben, Jonathan Loh, Sarah Whitmee, Louise McRae, Rajan Amin, and Jonathan E.M. Baillie. 2009. "Monitoring Change in Vertebrate Abundance: The Living Planet Index." *Conservation Biology* 23 (2): 317–27. https://doi.org/10.1111/j.1523-1739.2008.01117.x.

Colpitts, George. 2017. "Knowing Nature in the Business Records of the Hudson's Bay Company, 1670–1840." *Business History* 59 (7): 1054–80. https://doi.org/10.1080/00076791.2017.1304914.

Company, North West. 2017. "The North West Company History." 2017. http://www.northwest.ca/about-us/history.php#prettyPhoto.

Davies, Althea L., Sergio Colombo, and Nick Hanley. 2014. "Improving the Application of Long-Term Ecology in Conservation and Land Management." *Journal of Applied Ecology* 51 (1): 63–70. https://doi.org/10.1111/1365-2664.12163.

Deinet, S, C Ieronymidou, McRae L, Burfield IJ, Foppen RP, Collen B, Böhm M, L McRae, and IJ Burfield. 2013. "Wildlife Comeback in Europe:The Recovery of Selected Mammal and Bird Species." London. http://www.citeulike.org/group/15400/article/14089335.

Derocher, A.E., G.W. Garner, N.J Lunn, and O Wiig. 1998. "Polar Bears : Proceedings of the Twelfth Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 3-7 February 1997, Oslo, Norway." In , 159. Oslo, Norway: IUCN--the World Conservation Union.

Elton, C, and M Nicholson. 1942. "Fluctuations in Numbers of the Muskrat (Ondatra Zibethica) in Canada." *British Ecological Society* 11 (1): 96–126.

Elton, Charles, and Mary Nicholson. 1942. "The Ten-Year Cycle in Numbers of the Lynx in Canada." *Journal of Animal Ecology* 11 (2): 215–44. https://doi.org/10.2307/1358.

Erickson, Bruce. 2015. "Embodied Heritage on the French River: Canoe Routes and Colonial History." *The Canadian Geographer / Le Géographe Canadien* 59 (3): 317–27. https://doi.org/10.1111/cag.12171.

ESRI. 2008. "ArcGIS Desktop: Release 9.3." Redlands, CA: Environmental Systems Research Institute.

Finney, Bruce P., Irene Gregory-Eaves, Marianne S. V. Douglas, and John P. Smol. 2002. “Fisheries Productivity in the Northeastern Pacific Ocean over the Past 2,200 Years.” *Nature* 416 (6882): 729–33. https://doi.org/10.1038/416729a.

Fonzo, Martina M I Di, B Collen, A Chauvenet, Georgina M. Mace, and Claudia Bieber. 2016. “Patterns of Mammalian Population Decline Inform Conservation Action.” *Journal of Applied Ecology* 53 (4): 1046–54. https://doi.org/10.1111/1365-2664.12659.

Gamarra, Javier G P, and Ricard V. Solé. 2000. “Bifurcations and Chaos in Ecology: Lynx Returns Revisited.” *Ecology Letters* 3 (2): 114–21. https://doi.org/10.1046/j.1461-0248.2000.00128.x.

Gaston, Kevin J., and Richard A. Fuller. 2008. “Commonness, Population Depletion and Conservation Biology.” *Trends in Ecology and Evolution* 23 (1): 14–19. https://doi.org/10.1016/j.tree.2007.11.001.

Grooten, M, and REA Almond. 2018. “Living Planet Report 2018: Aiming Higher.” Gland, Switzerland. https://www.wwf.org.uk/sites/default/files/2018-10/wwfintl_livingplanet_full.pdf.

Hewitt, CG. 1921. *The Conservation of the Wild Life of Canada*. New York: C. Scribner.

Hewitt, David G, and David G Hewitt. 2015. “Hunters and the Conservation and Management of White-Tailed Deer (Odocoileus Virginianus).” *International Journal of Environmental Studies* 72 (5): 839–49. https://doi.org/10.1080/00207233.2015.1073473.

Hope, D. 2016. “Britain and the Fur Trade: Commerce and Consumers in the North-Atlantic World, 1783-1821.” Northumbria University. http://nrl.northumbria.ac.uk/31598/.

Hunter, C. M., H. Moller, and D. Fletcher. 2000. “Parameter Uncertainty and Elasticity Analyses of a Population Model: Setting Research Priorities for Shearwaters.” *Ecological Modelling* 134 (2–3): 299–324. https://doi.org/10.1016/S0304-3800(00)00344-6.

“Iucn Red List of Threatened Species.” n.d. Accessed January 1, 2014. https://www.google.com/search?q=iucn+red+list&oq=iucn+red+list&aqs=chrome..69i57j69i60j0l4.1621j0j1&sourceid=chrome&ie=UTF-8.

Jackson, Jeremy B C, Michael X Kirby, Wolfgang H Berger, Karen A Bjorndal, Louis W Botsford, Bruce J Bourque, Roger H Bradbury, et al. 2001. “Historical Collapse Overfishing of and the Recent Coastal Ecosystems.” *Science* 293 (5530): 629–38. https://doi.org/10.1126/science.1059199.

Jones, J W. 1914. *Fur Farming in Canada*. Edited by Game and Fur-Bearing Animals Commission of Conservation Canada, Committee on Fisheries. Ottawa: The Mortimer Co., Ltd.

Kahn, P. H, and B Friedman. 1995. “Environmental Views and Values of Children in an Inner-City Black Community.” *Child Development* 66: 1403–17.

Kimura, Daniel K., James W. Balsiger, and Daniel H. Ito. 1984. “Generalized Stock Reduction Analysis.” *Canadian Journal of Fisheries and Aquatic Sciences* 41 (9): 1325–33. https://doi.org/10.1139/f84-162.

Koslow, J. Anthony, and Jennifer Couture. 2013. “Follow the Fish.” *Nature* 502: 163–64. https://doi.org/10.1038/502163a.

Krebs, Charles J, S. Boutin, R Boonstra, Anthony R E Sinclair, J N M Smith, M R T Dale, K Martin, and R Turkington. 1995. “Impact of Food and Predation on the Snowshoe Hare Cycle.” *Science* 269 (5227): 1112–15.

Kroon, Hans de, Anton Plaisier, Jan van Groenendael, and Hal Caswell. 1986.

"Elasticity: The Relative Contribution of Demographic Parameters to Population Growth Rate." *Ecology* 67 (5): 1427–31.

Lotze, Heike K., and Boris Worm. 2009. "Historical Baselines for Large Marine Animals." *Trends in Ecology and Evolution* 24 (5): 254–62. https://doi.org/10.1016/j.tree.2008.12.004.

Lunn, N. J., Scott. Schliebe, Erik W. Born, and International Union for Conservation of Nature and Natural Resources. Species Survival Commission. 2002. "Polar Bears : Proceedings of the 13th Working Meeting of the IUCN/SSC Polar Bear Specialist Group." Nuuk, Greenland: IUCN--the World Conservation Union.

McAllister, Murdoch K, and James N Ianelli. 1997. "Bayesian Stock Assessment Using Catch-Age Data and the Sampling-Importance Resampling Algorithm." *Canadian Journal of Fisheries and Aquatic Sciences* 54: 284–300.

Mcclenachan, Loren, Francesco Ferretti, and Julia K. Baum. 2012. "From Archives to Conservation: Why Historical Data Are Needed to Set Baselines for Marine Animals and Ecosystems." *Conservation Letters* 5 (5): 349–59. https://doi.org/10.1111/j.1755-263X.2012.00253.x.

McRae, Louise, Stefanie Deinet, and Robin Freeman. 2017. "The Diversity-Weighted Living Planet Index: Controlling for Taxonomic Bias in a Global Biodiversity Indicator." Edited by Hideyuki Doi. *PLOS ONE* 12 (1): e0169156. https://doi.org/10.1371/journal.pone.0169156.

Miller, Joshua H. 2011. "Ghosts of Yellowstone: Multi-Decadal Histories of Wildlife Populations Captured by Bones on a Modern Landscape." *PLoS ONE* 6 (3). https://doi.org/10.1371/journal.pone.0018057.

Mowat, F. 2012. "Sea of Slaughter." In , 160–64. Vancouver, British Columbia: Douglas and McIntyre.

Myers, Ransom A., and Boris Worm. 2003. "Rapid Worldwide Depletion of Predatory Fish Communities." *Nature* 423 (6937): 280–83. https://doi.org/10.1038/nature01610.

NERC Centre for Population Biology, Imperial College. 2010. "The Global Population Dynamics Database Version 2." 2010. http://www.sw.ic.ac.uk/cpb/cpb/gpdd.html.

Novak, M. 1987. *Wild Furbearer Management and Conservation in North America*. Ontario: Association, Ontario Ministry of Natural Resources; Ontario Trappers. https://www.google.com/search?q=wild+furbearer+management+and+conservation&oq=wild+furbearer&aqs=chrome.0.0j69i57.2315j0j1&sourceid=chrome&ie=UTF-8.

Oliver, Tom H., Matthew S. Heard, Nick J B Isaac, David B. Roy, Deborah Procter, Felix Eigenbrod, Rob Freckleton, et al. 2015. "Biodiversity and Resilience of Ecosystem Functions." *Trends in Ecology & Evolution* 30 (11): 673–84. https://doi.org/10.1016/j.tree.2015.08.009.

Papworth, S.K., J. Rist, L. Coad, and E.J. Milner-Gulland. 2009. "Evidence for Shifting Baseline Syndrome in Conservation." *Conservation Letters* 2: 93–100. https://doi.org/10.1111/j.1755-263X.2009.00049.x.

Pauly, Daniel. 1995. "Anecdotes and the Shifting Baseline Syndrome of Fisheries." *Trends in Ecology & Evolution* 10 (10): 430. https://doi.org/10.1016/S0169-5347(00)89171-5.

Pinnegar, John K., and Georg H. Engelhard. 2008. "The 'shifting Baseline' Phenomenon: A Global Perspective." *Reviews in Fish Biology and Fisheries* 18 (1): 1–16. https://doi.org/10.1007/s11160-007-9058-6.

Poland, H. 1892. *Fur-Bearing Animals: In Nature and in Commerce*. Gurney &

Jackson.

Reimoser, Friedrich, and Susanne Reimoser. 2017. “Long-Term Trends of Hunting Bags and Wildlife Populations in Central Europe . Long-Term Trends of Hunting Bags and Wildlife Populations in Central Europe,” no. December 2016.

Roman, Joe, Meagan M. Dunphy-Daly, David W. Johnston, and Andrew J. Read. 2015. “Lifting Baselines to Address the Consequences of Conservation Success.” *Trends in Ecology and Evolution* 30 (6): 299–302. https://doi.org/10.1016/j.tree.2015.04.003.

Row, Jeffrey R., Paul J. Wilson, Celine Gomez, Erin L. Koen, Jeff Bowman, Daniel Thornton, and Dennis L. Murray. 2014. “The Subtle Role of Climate Change on Population Genetic Structure in Canada Lynx.” *Global Change Biology* 20 (7): 2076–86. https://doi.org/10.1111/gcb.12526.

Rowe, Rebecca J., and Rebecca C. Terry. 2014. “Small Mammal Responses to Environmental Change: Integrating Past and Present Dynamics.” *Journal of Mammalogy* 95 (6): 1157–74. https://doi.org/10.1644/13-MAMM-S-079.

RStudio Team. 2015. “RStudio: Integrated Development for R.” Boston, MA: RStudio, Inc. https://www.rstudio.com/.

Sáenz-Arroyo, a, C. M. Roberts, J Torres, and M Cariño-Olvera. 2005. “Using Fishers’ Anecdotes, Naturalists´observations and Grey Literature to Reassess Marine Species at Risk: The Case of the Gulf Grouper in the Gulf of California, Mexico.” *Fish and Fisheries* 6: 121–33. https://doi.org/10.1111/j.1467-2979.2005.00185.x.

Seton, E T. 1909. *Life Histories of Northern Animals: An Account of the Mammals of Manitoba*. New York City: Scribner. https://www.google.com/search?ei=0KJJWs6DAcLd0gLdtZK4Dw&q=seton+mammals+1909&oq=seton+mammals+1909&gs_l=psy-ab.3...71063.74022.0.74295.12.12.0.0.0.0.146.895.0j8.8.0....0...1c.1.64.psy-ab..4.4.475...0j0i22i30k1j33i160k1.0.W-vXm0nO9as.

Shoemaker, Kevin T., and H. Resit Akçakaya. 2015. “Inferring the Nature of Anthropogenic Threats from Long-Term Abundance Records.” *Conservation Biology* 29 (1): 238–49. https://doi.org/10.1111/cobi.12353.

Simmons, D. 2007. *Keepers of the Record: The History of the Hudson’s Bay Company Archives*. Quebec: McGill-Queen’s University Press.

Soga, Masashi, and Kevin J. Gaston. 2018. “Shifting Baseline Syndrome: Causes, Consequences, and Implications.” *Frontiers in Ecology and the Environment* 16 (4): 222–30. https://doi.org/10.1002/fee.1794.

Spooner, Fiona E.B., Richard G. Pearson, and Robin Freeman. 2018. “Rapid Warming Is Associated with Population Decline among Terrestrial Birds and Mammals Globally.” *Global Change Biology* 24 (10): 4521–31. https://doi.org/10.1111/gcb.14361.

Sundby, S, and O Nakken. 2005. “Spatial Shifts in Spawning Habitats of Arcto-Norwegian Cod Induced by Climate Change.” *ICES Journal of Marine Science* 65 (April): 953–62. http://icesjms.oxfordjournals.org/content/65/6/953.short.

Turvey, Samuel T., Leigh A. Barrett, Hao Yujiang, Zhang Lei, Zhang Xinqiao, Wang Xianyan, Huang Yadong, Zhou Kaiya, Tom Hart, and Wang Ding. 2010. “Rapidly Shifting Baselines in Yangtze Fishing Communities and Local Memory of Extinct Species.” *Conservation Biology* 24 (3): 778–87. https://doi.org/10.1111/j.1523-1739.2009.01395.x.

Voorhis, Ernest. 1930. “Historic Forts and Trading Posts of the French Regime and of the English Fur Trading Companies.” 1930.

Walters, Carl J, Steven J.D. Martell, and Josh Korman. 2006. "A Stochastic Approach to Stock Reduction Analysis." *Canadian Journal of Fisheries and Aquatic Sciences* 63 (1): 212–23. https://doi.org/10.1139/f05-213.

Watson, Robert T, Thomas Rosswall, Achim Steiner, Klaus Töpfer, Salvatore Arico, and Peter Bridgewater. 2005. *Ecosystems and Human Well-Being*. Edited by Robert T Watson, Thomas Rosswall, Achim Steiner, Klaus Töpfer, Salvatore Arico, and Peter Bridgewater. *Ecosystems*. Vol. 5. Millenium Assessments. World Resources Institute. https://doi.org/10.1196/annals.1439.003.

Whitlock, Rebecca E, Nicholas J Aebischer, and Jonathan C Reynolds. 2003. "The National Gamebag Census as a Tool for Monitoring Mammal Abundance in the UK A Report to JNCC," no. September: 1–59.

Wiig, O, S Amstrup, T Atwood, K Laidre, N Lunn, M Obbard, E Regehr, and G Thiemann. 2015. "Ursus Maritimus. The IUCN Red List of Threatened Species 2015." https://doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

Wilcove, David S., David Rothstein, Jason Dubow, Ali Phillips, and Elizabeth Losos. 1998. "Quantifying Threats to Imperiled Species in the United States." *BioScience* 48 (8): 607–15. https://doi.org/10.2307/1313420.

Willis, K. J, M. B Araujo, K. D Bennett, B. Figueroa-Rangel, C. A Froyd, and N. Myers. 2007. "How Can a Knowledge of the Past Help to Conserve the Future? Biodiversity Conservation and the Relevance of Long-Term Ecological Studies." *Philosophical Transactions of the Royal Society B: Biological Sciences* 362 (1478): 175–87. https://doi.org/10.1098/rstb.2006.1977.

Willis, K. J, L Gillson, T. M Brncic, and B. Figueroa-Rangel. 2005. "Providing Baselines for Biodiversity Measurement." *Trends in Ecology and Evolution* 20 (3): 107–8. https://doi.org/10.1016/j.tree.2005.01.007.

Winfree, Rachael, Jeremy W. Fox, Neal M. Williams, James R. Reilly, and Daniel P. Cariveau. 2015. "Abundance of Common Species, Not Species Richness, Drives Delivery of a Real-World Ecosystem Service." Edited by Katriona Shea. *Ecology Letters* 18 (7): 626–35. https://doi.org/10.1111/ele.12424.

Yan, Chuan, Nils Chr Stenseth, Charles J. Krebs, and Zhibin Zhang. 2013. "Linking Climate Change to Population Cycles of Hares and Lynx." *Global Change Biology* 19 (11): 3263–71. https://doi.org/10.1111/gcb.12321.

Tables and Figures

Table 1. Estimated percentage population change from 1850 and 1970 to 2009 for the eight modelled species. Mean population values are presented in bold and confidence intervals are presented in brackets.

Species	**1850 population**	**1970 population**	**2009 population**	**1850 – 2009 total (& annual) population loss or gain (%)**	**1970 -2009 total (& annual) population loss or gain (%)**
Arctic fox	133,000	132,000	133,000	**0** (0)	**1** (0.02)
Canadian lynx	262,000	258,000	290,000	**11** (0.06)	**12** (0.32)
Beaver	4,730,000	3,730,000	4,440,000	**-6** (-0.04)	**19** (0.49)
Bobcat	50,300	44,100	47,000	**-7** (-0.04)	**7** (0.17)
Polar bear	20,300	13,900	15,800	**-22** (-0.14)	**14** (0.35)
Black bear	831,000	414,000	390,000	**-53** (-0.33)	**-6** (-0.15)
Fisher	167,000	157,000	7,000	**-96** (-0.60)	**-96** (-2.45)
American badger	518,000	175,000	83,800	**-84** (-0.53)	**-52** (-1.34)

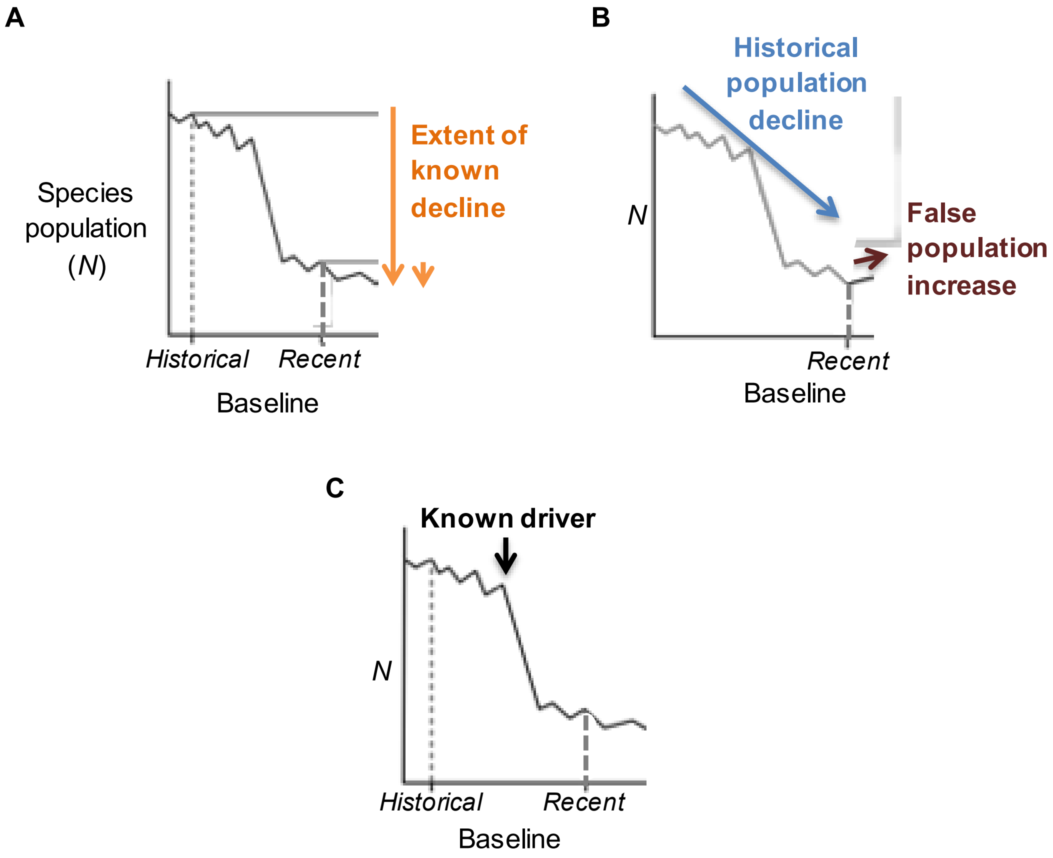


Figure 1. Schematic of hypothetical population trends highlighting the benefits of establishing a historical baseline through (A) providing better context on the extent of decline, thus allowing for better population recovery scenarios when advising on recovery targets (blue arrows signify extent of known decline), (B) identifying a trend through different time series lengths (represented by blue arrows) and discriminating between true decreasing trends and misattributed trends caused by population cycles, and (C) estimating the impact on populations from known drivers of decline captured through use of historical baselines.

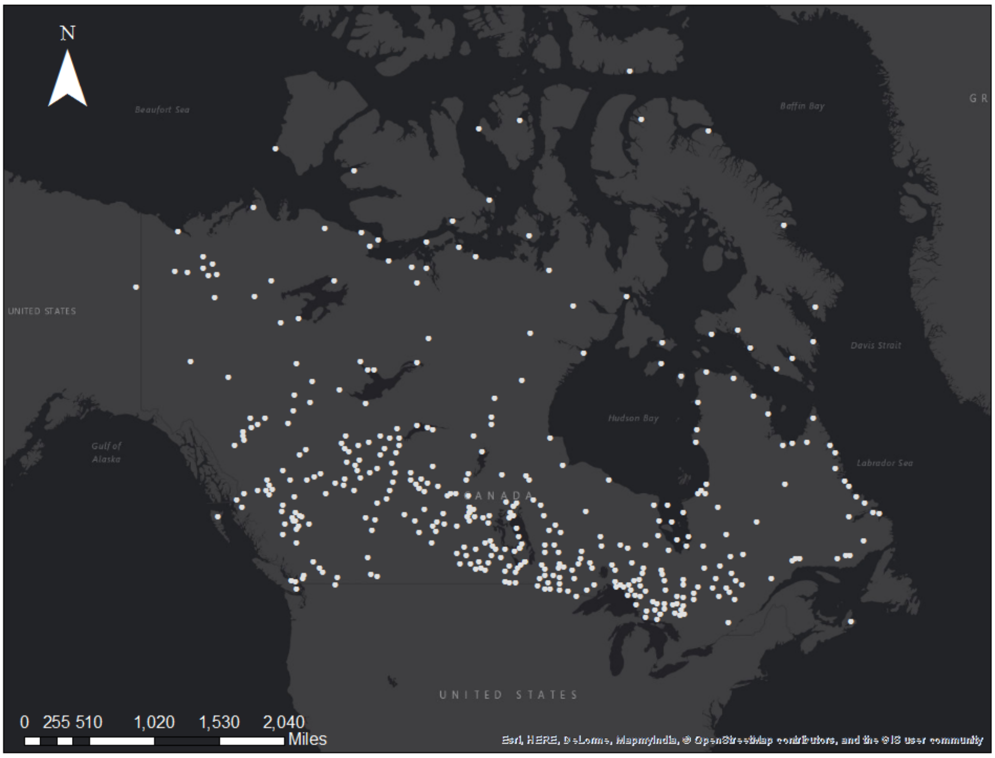


Figure 2. The approximate locations of all Hudson's Bay Company trading posts that were operating in 1927 in North America, adapted from Elton and Nicholson (1942a).

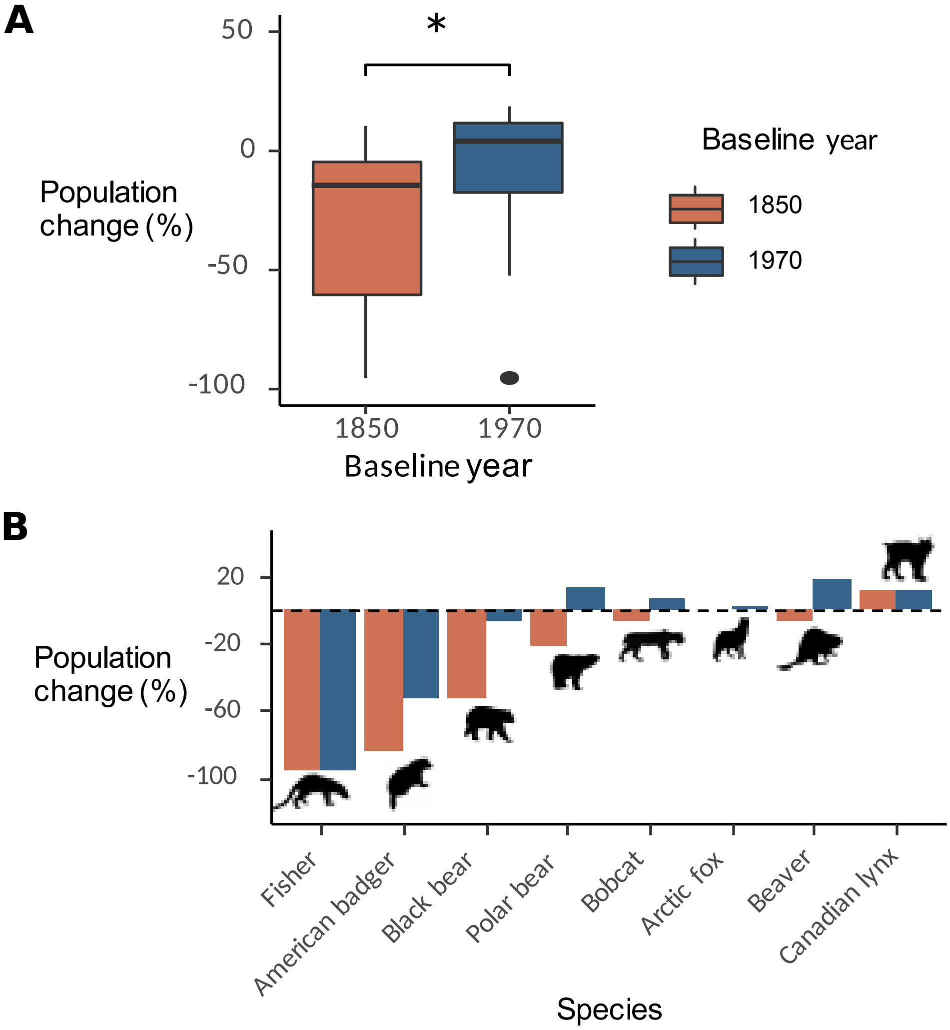


Figure 3. Percentage change in population estimates using two different baseline years: 1850-2009 and 1970-2009 for (A) species cumulatively (t-test: $p < 0.05$; 1970: 4%, 1850: -14.5% medians ± S.E, $n = 8$), and (B) species-specific population estimates.

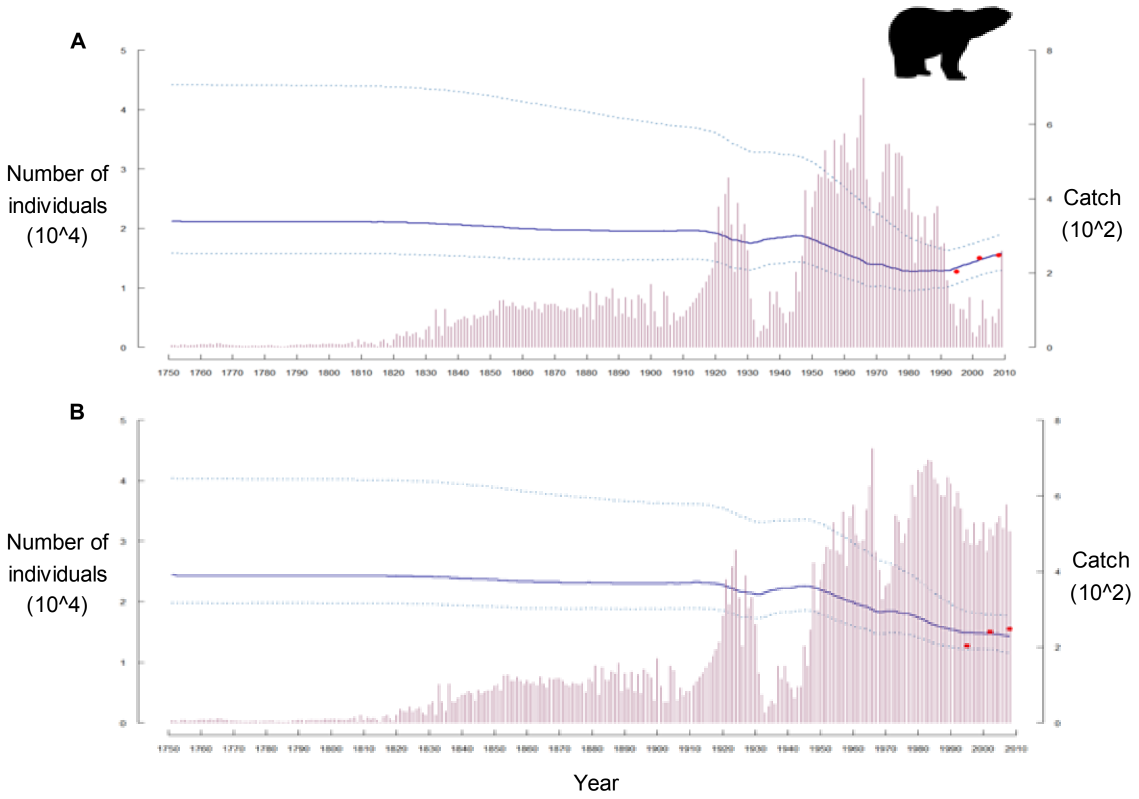


Figure 4. Population trajectories for polar bear, using, from 1970, (A) Canadian government annual fur harvest data and (B) IUCN/SSC statistics on number of polar bears annually killed in Canada. Pink bars are the annual fur harvest (right-hand axis), the blue solid line and blue dashed lines are the population projection and the 95% median confidence intervals respectively; the current Canada-wide population estimate is shown by the red dots (left-hand axis).

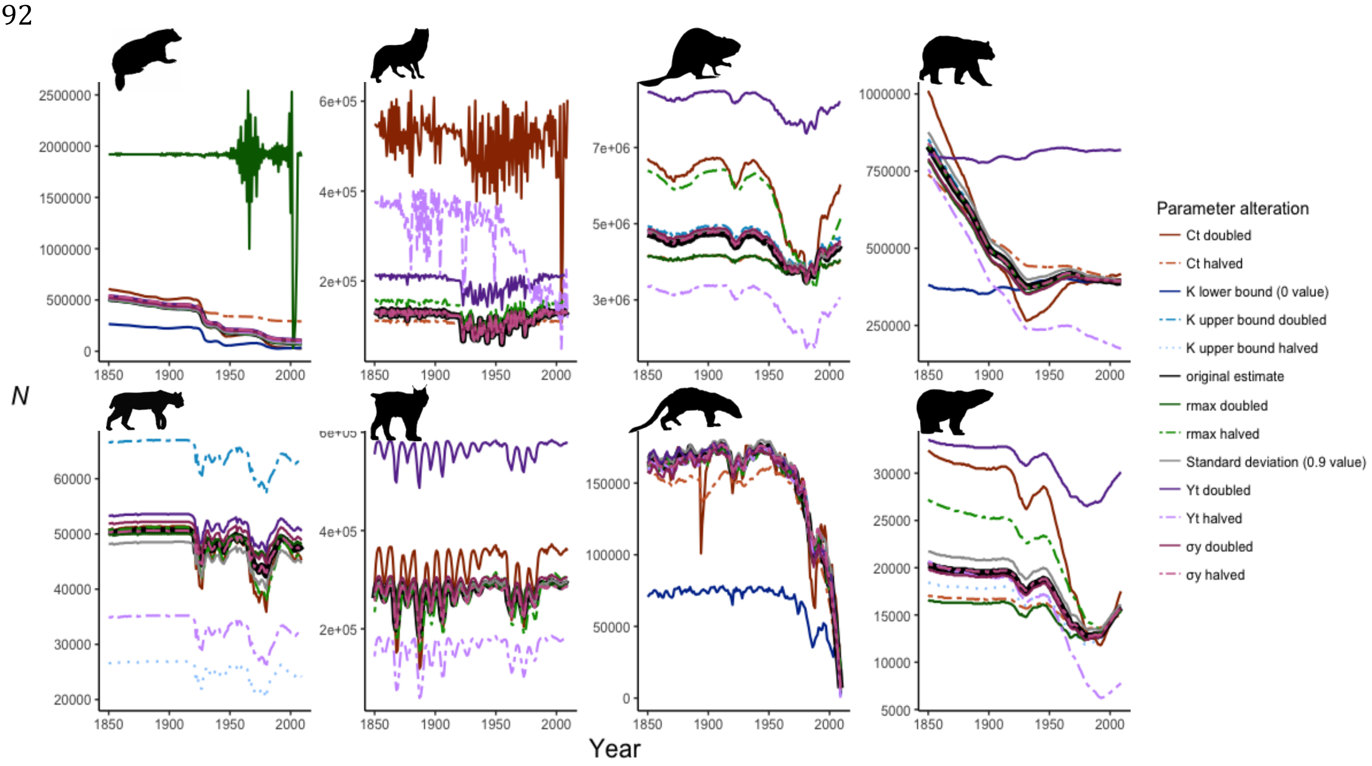


Figure 5. Elasticity analysis for each species, showing how population change between 1850-2009 altered when each parameter was doubled (+50) and halved (-50) for number of individuals caught (C_t, orange lines); carrying capacity (K) lower bound and carrying capacity upper bound (blue lines); original SSRA estimate (black line); intrinsic growth rate (r_{max}, green lines); standard deviation altered to 0.9 (grey line); ~~current population~~ abundance estimate(s) (y_t; purple lines); observation error (σ_y, pink lines).

Choice of baseline affects historical population trends in hunted mammals of North America

In memory of Dr. Ben Collen

Amy C. Collins[1]*, Monika Böhm[2], Ben Collen[3]

[1]Wildlife, Fish and Conservation Biology, University of California, Davis, Shields Avenue, CA, 95616, USA

[2]Institute of Zoology, Zoological Society of London, Regent's Park, London, NW1 4RY, UK

[3]Centre for Biodiversity & Environment Research, Department of Genetics, Evolution & Environment, University College London, London WC1E 6BT, UK

*Corresponding author (phone number: +1 530 760 9139; email: accollins@ucdavis.edu)

Abstract

Establishing historical baselines of species' populations is important for contextualising present-day population trends, identifying significant anthropogenic threats, and preventing a cultural phenomenon known as 'shifting baseline syndrome'. However, our knowledge of historical baselines is limited by a lack of direct observation data on species abundance pre-1970. We present historical data of species-specific fur harvests from the Canadian government and Hudson's Bay Company as a proxy for estimating species abundance over multiple centuries. Using stochastic stock reduction analysis originally developed for marine species, we model historical population trends for eight mammals, and assess population trends based on two different baseline years: 1850 and 1970. Results show that population declines are significantly greater when using an 1850 baseline, as opposed to a 1970 baseline, and for four species, the population trend shifted from a population increase to a decrease. Overall, the median population change of the eight species changed from a 15% decline for 1850, to a 4% increase for 1970. This study shows the utility of harvest data for deriving population baselines for hunted terrestrial mammals which can be used in addition to other historical data such as local ecological knowledge. Results highlight the need for developing historically relevant population baselines in order to track abundances over time in threatened species and common species alike, to better inform species conservation programs, wildlife management plans and biodiversity indicators.

Keywords: Historical baselines; shifting baseline syndrome; population estimation; mammal conservation; Hudson's Bay Company; North America

1. Introduction

Species population declines and extinctions undermine the functioning and resilience of ecosystems on which humans and wildlife depend (Cardinale et al. 2012; Oliver et al. 2015). To monitor and respond to species losses, changes in population abundance are used as a sensitive metric of change (Collen et al. 2011; Shoemaker and Akçakaya 2015) and have been incorporated into globally adopted biodiversity indicators such as the Living Planet Index, which tracks changes in vertebrate population abundance from 1970 (Collen et al. 2009). However, data on population abundance typically become scarcer beyond a few decades from the present, prior to the implementation of species monitoring programmes (Willis et al. 2005; Bonebrake et al. 2010).

Knowledge of historical populations acts as an antidote to 'shifting baseline syndrome'; a phenomenon in which with each new human generation comes a lowered expectation of a species population norm (Pauly 1995; Kahn and Friedman 1995; Soga and Gaston 2018). Historical population baselines have many practical policy implications, for example when defining population recovery and conservation legacy, deciding harvest quotas, and influencing the general public's perception of a species (Papworth et al. 2009; Davies, Colombo, and Hanley 2014; Roman et al. 2015; Akcakaya et al. 2018; see Figure 1a). Additionally, estimating historical populations can help to differentiate between a population trend that is unidirectional or cyclical, such as the Atlantic Multidecadal Oscillation inducing bidirectional changes in fish abundance (Jackson et al. 2001; Willis et al. 2007; Sundby and Nakken 2005; see Figure 1b). Without long-term measurements, observers may misattribute downward phases of natural population cycles as human-caused population declines (Koslow and Couture 2013). Finally, historical population data can help to identify historic drivers of population change (see Figure 1c), which is important for quantifying the relative significance of each past and present threat in order to develop threat-specific management strategies and inform future scenario modelling (Baker and Clapham 2004; Pinnegar and Engelhard 2008).

Many techniques available to reconstruct historical population baselines emerged from the discipline of marine historical ecology (Lotze and Worm 2009). Faced with the need to sustainably manage fish stocks, fisheries researchers have used recorded history (e.g. 'local ecological knowledge') (Sáenz-Arroyo et al. 2005; Turvey et al. 2010), archaeogenomic data (e.g. analysis of relative stable isotope concentrations) (Finney et al. 2002), and fish stock assessments from historical catch data (Myers and Worm 2003; Baker and Clapham 2004) to extrapolate population size over time and capture stock collapses that pre-date direct monitoring.

Recorded history has also provided us with historical population estimates for terrestrial species, although not as frequently as in the marine realm. These studies are extremely valuable in painting a picture of past population condition (Cole and Woinarski 2000; Rowe and Terry 2014), but with each historical data source comes its own unique set of limitations. For instance, museum and fossil records are often patchy and taxonomically biased, and local ecological knowledge generally only covers a couple of generations spanning less than 100 years (Miller 2011). Here, we add to our growing knowledge on reconstructing population baselines by focussing on harvest data of terrestrial mammals as another data source which holds great potential in historical baseline reconstruction.

Reports from the Hudson's Bay Company (HBC), Canada, have been previously used to document lynx and muskrat population cycles (Elton and Nicholson 1942a; Elton and Nicholson 1942b), predator-prey dynamics of lynx and snowshoe hare (Krebs et al. 1995), and the potential roles of climate, productivity and disease in these cycles (Gamarra and Solé 2000; Yan et al. 2013; Row et al. 2014). Here, we (a) show the utility of these harvest data to reconstruct historic populations by applying a stochastic population model first developed for marine vertebrates (Christensen 2006), (b) use these population reconstructions to demonstrate that baselines differ when using over 100 years of data compared to less than 50 years of data and (c) show that choice of different baseline years results in different interpretation of estimated population trends.

2. Methods

2.1 Reconstructing historical abundance trends

To reconstruct historical trends in terrestrial mammal abundance, we used a stochastic stock reduction analysis (SSRA) originally developed by Walters *et al.* (2006) to analyse trends in fish populations. This method uses a simple growth model, and can be applied to species for which we have limited knowledge of life history parameters and catch-per-unit-effort (Kimura, Balsiger, and Ito 1984). The model and method outlined below was described in detail by Christensen (2006) for establishing historical baselines of marine mammals:

$$N_{t+1} = N_t + r_{max} N_t \left(1 - \frac{N_t}{K}\right) e^{w_t} - C_t \qquad (1)$$

where N_t is the number of individuals in a population at time t, r_{max} is the intrinsic rate of increase, K is the carrying capacity, w_t is the error term at time t, and C_t is the number of individuals harvested annually. The population is assumed to be at K at the start of hunting, and the model implies that there are no errors in our records of C_t.

Using a Bayesian approach, we randomly drew from a uniform prior distribution of K, which was bounded by a lower and an upper estimate, and normal prior distributions of r_{max} and w_t values. We used process error terms (τ_w) for the standard deviation of w_t values. This generated a N_t trajectory which was dependent on C_t (eq 1). We then calculated the likelihood (L) of deriving the known abundance estimate for the current population (y_t) based on the estimated N_t using eq. 2:

$$L(y_t \mid r_{max}, K, w_t) = n\left[\log(\sigma_y) + \frac{1}{2}\log(2\pi)\right] + \sum_{i=1}^{n} \frac{z_t^2}{2\sigma_y^2} \tag{2}$$

where n is the number of abundance estimates (y_t) available for each species, σ_y is the observation error (standard deviation of the abundance estimate, y_t), and z_t is the log-normal residual, calculated as:

$$z_t = \log(N_t) - \log(y_t) \tag{3}$$

Equation 1 to 3 were repeated 200,000 times. Generated N_t values were then resampled using the importance sampling procedure by Schnute (1994) and McAllister and Ianelli (1997), to estimate a posterior probability density function. Finally, we calculated the median and the 95% credible interval of the N_t trajectory, after computing the marginal posterior distributions of K.

2.2 Harvest data

We conducted a literature search for fur trade records of Canadian mammals from the Hudson's Bay Company (HBC). The HBC fur clothing business, originally London-based, became established in Canada in 1670 in response to the UK demand for fur and the abundance of furbearing mammal species in Canada (Simmons 2007). Expanding its area of influence over time, the company monopolised the North American fur trade from 1821 onward, and eventually operated over an area that spanned around one quarter of the extent of North America (Erickson 2015; Colpitts 2017; Company 2017).

Published annual HBC sales accounts for 25 mammals were acquired from Poland (1892) for the time period 1751 – 1847 and Jones (1914) for the time period 1848 – 1909. Additional HBC data was supplied for the American badger (*Taxidea taxus*), beaver (*Castor canadensis*) and Canadian lynx (*Lynx canadensis*) by the NERC Centre for Population Biology (NERC Centre for Population Biology 2010), Hewitt (1921) and Elton and Nicholson (1942) respectively. Canadian fur data for 1919 – 1970 were retrieved from Novak (1987) and data for 1970 – 2009 from the National Bureau of Statistics (NBS), Canada (2010). Using the individual number of furs across datasets as a proxy for the number of individuals taken for trade resulted in time series harvest data for 25 mammal species ranging from 1722 – 2009 (electronic supplementary material, Table S1). Of these 25 species, only thirteen had harvest data from two or more continuous datasets up to and including 2009 (the last year of data available from NBS, Canada). Harvest data for species modelled is supplied in the supplementary material (Table S2).

Jones' (1914) data represented the number of furs sent to HBC headquarters, and Poland (1892) represented the year the fur was sold at the London auctions; therefore, these data were backdated by two years and one year respectively following Elton and Nicholson (1942a). All time-series had a data gap for the years 1910 – 1918 and 2002. We estimated values for these years using a log-linear interpolation for all species except the lynx. Due to the 9-year cyclic nature of the lynx harvest data, a linear interpolation was applied around the peak year (1914) of the cycle for 1910 – 1914, and 1914 – 1918.

We can be fairly confident that the numbers reported from annual sales accounts is an underestimate of the actual number of individuals caught (e.g., Elton and Nicholson 1942a), due to escape from traps, illegal trade by trappers, lost or stolen carcasses in transit to London's auction house, bookkeeping errors, etc. Although statistical approaches such as employing a correction factor are available to counteract the undocumented loss, we select to estimate the *minimum* population decline based on reported numbers only.

2.3 Informative priors

In order to perform the Bayesian SSRA model, population growth rate (r_{max}) and current abundance estimate(s) (y_t; 1979 – 2008) in eq. 1 and 2 were gathered from peer-reviewed publications for each species (electronic supplementary material, Table S3; Table S4). If countrywide abundance estimates (y_t) for Canada were unavailable, province-wide abundance estimates were extrapolated to the entire current species range within Canada. Species range maps were downloaded from the IUCN Red List of Threatened Species (IUCN 2014). Of the 13 species with harvest data, muskrat (*Ondatra zibethicus*), marten (*Martes americana*), red fox (*Vulpes vulpes*), mink (*Neovison vison*), and coyote (*Canis latrans*) had no countrywide or province-wide estimates of abundance and were therefore removed from the study.

The contribution of observation error (σ_{y}) to the total error term ($\kappa = 0.1$) for each species was determined by selection criteria adapted from Christensen (2006), based on the source and uncertainty (p) associated with the abundance estimate, y_t. Province-wide extrapolations of y_t were assigned the highest uncertainty and direct estimates were assigned the lowest uncertainty (electronic supplementary material, Table S5). The remaining proportion of the total error term was allocated to the independent process error (τ_w), which represented the ecologically mediated fluctuations within a true population size (Ahrestani, Hebblewhite, and Post 2013), such that $\sigma_y = \sqrt{p} * \sqrt{\kappa}$ and $\tau_W = \sqrt{1-p} * \sqrt{\kappa}$.

As no data exist on historical carrying capacity (K) of populations, we estimated priors for K from historical range maps from the 1900s for each species obtained from Seton (1909). We georeferenced each species' map in ArcGIS v9.3 (ESRI 2008) to obtain the historical range in km^2, assuming all of the historical range was viable habitat. We collected present day minimum and maximum density estimates (individuals/km^2) from the literature and produced a lower and upper bound for K by extrapolating the density estimates to the historical range. A uniform prior distribution was then drawn, which assumes that K is constant over time (electronic supplementary material, Table S6). We carried out elasticity analyses to assess the impact of variation in our estimates of informative priors on population change.

2.4 Analysis of historical baselines

We reconstructed historical population trajectories for eight species; the American badger, Arctic fox (*Vulpes lagopus*), beaver, black bear (*Ursus americanus*), bobcat (*Lynx rufus*), Canadian lynx, fisher (*Martes pennanti*) and polar bear (*Ursus maritimus*). We quantified population change for each species as follows:

$$Population\ Change = \frac{N_{2009} - N_b}{N_b} * 100 \tag{4}$$

where N_b represents the population estimated at the given baseline year b. Population change was estimated for two baseline years: 1) 1850, the earliest year that all species had available data; and 2) 1970, the baseline year of the Living Planet Index (McRae, Deinet, and Freeman 2017). We then conducted a t-test to assess the difference in population change across the two baselines. We also calculated the median population change for all eight species under the two baselines and recorded the degree and direction of change.

2.5 Single versus multiple causes of mortality

Modelling population estimates based on fur trade alone overlooks other reasons animals are removed by humans; fur trade, trophy hunting, self-defence, illegal kills, individuals found dead, and handling by scientists. We applied the SSRA to another dataset for the polar bear, substituting fur trade records from 1970 onwards with data published by the IUCN Species Survival Commission (SSC) Polar Bear Specialist Group, that documents number of bears killed between 1970-2008 from all causes listed above (Derocher and Commission. 1998; Lunn et al. 2002; electronic supplementary material, Table S7).

2.6 Elasticity analysis

We carried out elasticity analysis (de Kroon et al. 1986) to examine the relative effect of parameter selection on model prediction. We independently halved and doubled each model parameter in turn (intrinsic growth rate (r_{max}), number of individuals harvested (C_t), current abundance estimate (y_t), historical carrying capacity (K), and observation error (σ_y)), and calculated elasticities (e) of the 1850 – 2009 population change, where higher e means higher proportional change in the population (de Kroon et al. 1986; Benton and Grant 1999; Hunter, Moller, and Fletcher 2000). Y represents the population change caused by the altered parameter, X:

$$e = \frac{\%\Delta Y}{\%\Delta X} \tag{5}$$

All analyses were carried out in the statistical software RStudio version 1.0.143 (RStudio Team 2015).

3. Results

3.1 Analysis of historical baselines

The median population change across the eight species for 1850 – 2009 was a 15% decrease (-0.1%/yr), whereas populations between 1970 – 2009 showed a 4% increase (0.1%/yr) (paired t-test: t = -3.036,1 d.f. = 7, *p* = 0.002, *n* = 8; Table 1, Figure 3a). Choice of baseline year resulted in a switch from a downward population trend for the period 1850 – 2009 to an upward trend for 1970 – 2009 for four species (Arctic fox, bobcat, polar bear, beaver) (Figure 3b; Table S8; Figure S1). Six species exhibited a more severe annual rate of population change for the 1850 – 2009 period compared to trends derived from a 1970 baseline (Table 1; Figure 3b), with this pattern holding for five species when the annual rate of population change is considered (Table 1).

3.2 Single versus multiple causes of mortality

Analysis of the polar bear data that encompassed all recorded mortality events from 1970 onwards shows that an additional 294 (range = -57 to 538) individuals were killed in comparison to the number harvested for furs alone (electronic supplementary material, Table S7). This altered the polar bear population decline using the 1850 baseline from 22% based on fur harvest data alone to 41% using the IUCN/SSC data (Figure 4).

3.3 Elasticity analysis

Parameters ranked in order of lowest to highest elasticity (proportional change in the population) were observation error, carrying capacity, harvest data, current abundance estimate and intrinsic growth rate (Figure 5, Electronic supplementary material, Table S9). In other words, small changes to intrinsic growth rate and abundance estimates cause larger alterations to the model output. Species ranked in order of lowest to highest average elasticity for all informative priors were fisher, black bear, American badger, arctic fox, bobcat, polar bear, Canadian Lynx and beaver.

4. Discussion

Our study demonstrates that for eight species of Canadian mammals, choice of baseline year greatly affects our understanding of historic population change. Collectively, using an 1850 baseline year rather than 1970 significantly altered the population trend. Analysis of individual species demonstrated that deriving population change from the 1850 baseline resulted in four species shifting from a population increase since 1970 to a population decrease of between 0 and -22% since 1850, and the magnitude of annual population change becoming more severe for six species (Figure 3). Interestingly, the polar bear, the only species to be listed as Vulnerable on the IUCN Red List (Wiig et al. 2015), was one of the four species to show a shift in population trend. From 1850, the population declined, reflecting the polar bear's IUCN conservation status, whereas the trend from 1970 showed the population as increasing.

Population time-series data allow for the detection of declines before species reach critical status (Balmford and Bond 2005; Di Fonzo et al. 2016; Collen et al. 2011). However, few long-term population studies used data more than 100 years old (e.g., 15%; Bonebrake et al. 2010). Biodiversity indicators that are reliant on such population time series data, such as the Living Planet Index (LPI), therefore often operate on baselines set to post-1950 (Bonebrake et al. 2010). At least for temperate systems, the LPI dataset may not date back far enough to detect often precipitous population declines which occurred prior to 1970 (Watson et al. 2005). For example, while some population recovery is observed in Europe since 1970 (Deinet et al. 2013), species are often recovering from declines which had reduced populations to a fraction of their historical population sizes by or prior to 1970. These short-term baselines are likely a result of the relatively recent advent of conservation biology in the 1970s and 1980s and corresponding emergence of long-term monitoring data for ecology and conservation (Collen et al. 2009).

In this study, we were able to model seven species currently listed as Least Concern on the IUCN Red List and not assessed under the Canadian Wildlife Species at Risk assessment (COSEWIC 2019). However, for five relatively common species: muskrat, marten, red fox, mink, and coyote that we collected harvest data for, a lack of recent population estimates at the regional or national scale prevented population modelling. Species are rarely subject to monitoring until declines are apparent and they may be of conservation concern (Bonebrake et al. 2010), while threatened species are often prioritised for monitoring, given limited financial resources and capacity (Gaston and Fuller 2008). As we are frequently witnessing population declines in common species, it is imperative that population monitoring also targets abundant or common species (Collen et al. 2011). This proactive approach enables early detection of declines, rather than a late reactive response when a species has already declined extensively, thus approaching extinction. This is of particular importance as abundant species play important parts in the functioning and maintenance of ecosystems (Winfree et al. 2015).

The strengths of using HBC data to estimate population change is the consistent unit over time (number of furs traded) and high temporal resolution (annual data), which overcome limitations oftentimes associated with historical records (Balmford and Bond 2005; Mcclenachan, Ferretti, and Baum 2012). While HBC data provides a very unique set of data, hunting statistics are available in many countries (e.g. Agetsuma 2018), and have a long tradition in regions such as Europe (Deinet et al. 2013; Hewitt and Hewitt 2015). Hunting bag records were used in a recent study to analyse long-term population trends across a number of European countries for several game bird, ungulate and carnivore species, although the study only reached back as far as 1970 (Reimoser and Reimoser 2017). In the UK, the voluntary National Gamebag Census was formally established in 1961 but resulting records on game birds and predatory species extend back to 1900 for many species (Whitlock, Aebischer, and Reynolds 2003; Aebischer and Baines 2008). Models such as the one utilised in this study play an important role not just in deriving generalised trends, but also population estimates using such data repositories. Hunting statistics can then complement, or be used in tandem with, data from different sources such as transgenerational local ecological knowledge to obtain historical population baselines.

Despite our robust methods, population estimates from historical information such as harvest data are of lower quality than those derived from direct counts and current population monitoring, and uncertainty surrounding the estimates is inevitably greater. While some losses are not reported, resulting in a potential underestimate of the population (see methods 2.2), the spatial pattern of hunting that occurred throughout Canada in the past is likely to lead to an overestimate of hunting offtake of species. Hunting efforts saw a large westward expansion between 1783 - 1821, driven in large part by local depletion of beaver in the ‘Canadian Shield’ (Hope 2016; Figure 2). New areas such as the Red River Settlement, which was established in 1811, would have allowed access to previously unharvested subpopulations of mammals, and a boost to total harvest numbers. On the other hand, sources suggest that HBC was attempting to conserve beaver and bison in the 1820s (Colpitts 2017), presumably by enacting policies to help local population recovery, thus lowering total harvest. These local variations in harvest effort are not visible in our dataset, and the impact of uncertainty on our population estimates is illustrated by the mid-range elasticity of our model to a halving and doubling of harvest data (c) (Figure 5; Table S7). To some degree, our baseline of 1850 may help circumvent some of the previous geographical shifts or expansions in hunting effort. However, next steps should endeavour to account for spatial variation in hunting records over time by adding a spatially explicit component to the stock reduction analysis.

In addition to hunting, other drivers of population change are likely to impact our species, and focus on only a single cause of population offtake is likely to underestimate population trends over time. Substituting IUCN/SSC data on polar bear mortality via direct contact with humans into the final 40 years of our model changed the estimated decline from 25% (based on fur hunting data alone) to 41% (Figure 4). As well as providing hunting numbers, the IUCN/SSC polar bear data also includes ‘individuals found dead’. This statistic includes mortality events which may be human-mediated or a result of natural causes, such as starvation from lack of sea ice. While hunting is a key human-mediated impact, it is but one of several sources of population decline facing mammals in North America (Brook, Sodhi, and Bradshaw 2008; Grooten and Almond 2018; Spooner, Pearson, and Freeman 2018).

Other anthropogenic drivers of population change such as habitat loss and the introduction of non-native species are not included in this analysis, again rendering our estimates conservative. For example, habitat loss affected the vast majority of threatened mammals in Canada, with direct human-caused mortality coming second (Imre and Derbowka 2009). Although modelling the impacts of all negative and positive anthropogenic effects on abundance is beyond the scope of this study, we can be fairly certain that these threats would play a role in determining the population size, intrinsic growth rates and carrying capacity within regions of Canada. As anthropogenic threats are dynamic (Wilcove et al. 1998), the degree of impact on the population will also alter over time. This highlights the need to quantify the impact of other threats which may influence these population parameters differentially over time (e.g. land use change altering carrying capacity) and incorporate these into baseline population models. One way of achieving this could be to allow carrying capacity K to vary across the time frame of our model in response to observed range contractions. In choosing a baseline for constructing historic population trends, we are ourselves guilty of ignoring what happened to our species prior to our chosen year of 1850. However, we recognize that the fur trade existed far back beyond this. Hunting for fur was carried out by Native Americans and European settlers alike; during the 17th century, the French owned the monopoly of the Canadian fur trade and by 1717, HBC had set up six trading posts (Voorhis 1930). By 1850, the fur trade had impacted wildlife on the east coast; between 1860 and 1920 the sea mink (*Neovison macrodon*) reportedly went extinct (Mowat 2012; Black, Reading, and Savage 1998). While we use 1850 to demonstrate the importance of a historical baseline for as many species as possible, the term ‘baseline’ is not used here to imply a ‘start’ date, nor do we claim that one date will suffice. Rather, historical baselines should be viewed as a sliding window, and in conservation, it is important to move that window as far back as possible through history, to provide better context for species’ current population status.

5. Conclusions

By failing to estimate historical baselines, we may miss the historical demise of populations which have been exploited by humans since at least the 18th century in Europe and North America (Deinet et al. 2013), and adversely influence our perception of what constitutes species population norms. This may affect how scientists, decision makers and the general public perceive the growth of a population as a result of conservation action and species protection. While in many northern hemisphere regions, population recovery to an 1850 (or earlier) baseline may no longer be viable, due to the extensive anthropogenic land use change that has occurred in the interim, our current understanding of population recovery may focus on population sizes which are too small and could leave populations vulnerable to future threats or dependent on continued conservation action. We encourage the use of well-documented and quantifiable historical records in developing population baselines, thus establishing a broad temporal scale over which to analyse species population trends. Armed with such data, we can improve our communication about population declines and set ambitious goals for realistically achievable population recovery (Akcakaya et al. 2018), make better-informed conservation management decisions, and improve our historical ecological awareness.

Supporting Materials

All datasets supporting this article have been uploaded as part of the supplementary material.

References

Aebischer, Nicholas J, and David Baines. 2008. "Monitoring Gamebird Abundance and Productivity in the UK : The GWCT Long-Term Datasets." *Revista Catalana d'Ornitologia* 24: 30–43.

Agetsuma, Naoki. 2018. "A Simple Method for Calculating Minimum Estimates of Previous Population Sizes of Wildlife from Hunting Records." *PLoS ONE* 13 (6): 1–10.

Ahrestani, Farshid S, Mark Hebblewhite, and Eric Post. 2013. "The Importance of Observation versus Process Error in Analyses of Global Ungulate Populations." *Nature Scientific Reports* 3 (3125): 1–10. https://doi.org/10.1038/srep03125.

Akcakaya, H. Resit, Elizabeth L Bennett, Thomas M Brooks, Molly K Grace, Anna Heath, Simon Hedges, Craig Hilton-taylor, et al. 2018. "Quantifying Species Recovery and Conservation Success to Develop an IUCN Green List of Species." *Conservation Biology* 32 (5): 1128–38. https://doi.org/10.1111/cobi.13112.

Baker, C. Scott, and Phillip J. Clapham. 2004. "Modelling the Past and Future of Whales and Whaling." *Trends in Ecology and Evolution* 19 (7): 365–71. https://doi.org/10.1016/j.tree.2004.05.005.

Balmford, Andrew, and William Bond. 2005. "Trends in the State of Nature and Their Implications for Human Well-Being." *Ecology Letters* 8 (11): 1218–34. https://doi.org/10.1111/j.1461-0248.2005.00814.x.

Benton, Tim G, and Alastair Grant. 1999. "In Evolutionary and Population Ecology." *Trends in Ecology and Evolution* 14 (12): 467–71.

Black, D W, J E Reading, and H G Savage. 1998. "Archaeological Records of the Extinct Sea Mink , Mustela Macrodon (Carnivora : Mustelidae), from Canada." *Canadian Field-Naturalist* 112 (1): 45–49.

Bonebrake, Timothy C., Jon Christensen, Carol L. Boggs, and Paul R. Ehrlich. 2010. "Population Decline Assessment, Historical Baselines, and Conservation." *Conservation Letters* 3 (6): 371–78. https://doi.org/10.1111/j.1755-263X.2010.00139.x.

Brook, Barry W., Navjot S. Sodhi, and Corey J A Bradshaw. 2008. "Synergies among Extinction Drivers under Global Change." *Trends in Ecology and Evolution* 23 (8): 453–60. https://doi.org/10.1016/j.tree.2008.03.011.

Canada, Statistics. 2011. "Fur Statistics, Canada." Ottowa, Ontario. www.statcan.gc.ca.

Cardinale, Bradley J., J. Emmett Duffy, Andrew Gonzalez, David U. Hooper, Charles Perrings, Patrick Venail, Anita Narwani, et al. 2012. "Biodiversity Loss and Its Impact on Humanity." *Nature* 486 (7401): 59–67. https://doi.org/10.1038/nature11148.

Christensen, L B. 2006. "Reconstructing Historical Abundances of Exploited Marine

Mammals at the Global Scale." University of British Columbia.

Cole, Jeffry R., and J. C.Z. Woinarski. 2000. "Rodents of the Arid Northern Territory: Conservation Status and Distribution." *Wildlife Research* 27 (4): 437–49. https://doi.org/10.1071/WR97053.

Collen, B., L. McRae, S. Deinet, A. De Palma, T. Carranza, N. Cooper, J. Loh, and J. E. M. Baillie. 2011. "Predicting How Populations Decline to Extinction." *Philosophical Transactions of the Royal Society B: Biological Sciences* 366 (1577): 2577–86. https://doi.org/10.1098/rstb.2011.0015.

Collen, Ben, Jonathan Loh, Sarah Whitmee, Louise McRae, Rajan Amin, and Jonathan E.M. Baillie. 2009. "Monitoring Change in Vertebrate Abundance: The Living Planet Index." *Conservation Biology* 23 (2): 317–27. https://doi.org/10.1111/j.1523-1739.2008.01117.x.

Colpitts, George. 2017. "Knowing Nature in the Business Records of the Hudson's Bay Company, 1670–1840." *Business History* 59 (7): 1054–80. https://doi.org/10.1080/00076791.2017.1304914.

Company, North West. 2017. "The North West Company History." 2017. http://www.northwest.ca/about-us/history.php#prettyPhoto.

Davies, Althea L., Sergio Colombo, and Nick Hanley. 2014. "Improving the Application of Long-Term Ecology in Conservation and Land Management." *Journal of Applied Ecology* 51 (1): 63–70. https://doi.org/10.1111/1365-2664.12163.

Deinet, S, C Ieronymidou, McRae L, Burfield IJ, Foppen RP, Collen B, Böhm M, L McRae, and IJ Burfield. 2013. "Wildlife Comeback in Europe:The Recovery of Selected Mammal and Bird Species." London. http://www.citeulike.org/group/15400/article/14089335.

Derocher, A.E., G.W. Garner, N.J Lunn, and O Wiig. 1998. "Polar Bears : Proceedings of the Twelfth Working Meeting of the IUCN/SSC Polar Bear Specialist Group, 3-7 February 1997, Oslo, Norway." In , 159. Oslo, Norway: IUCN--the World Conservation Union.

Elton, C, and M Nicholson. 1942. "Fluctuations in Numbers of the Muskrat (Ondatra Zibethica) in Canada." *British Ecological Society* 11 (1): 96–126.

Elton, Charles, and Mary Nicholson. 1942. "The Ten-Year Cycle in Numbers of the Lynx in Canada." *Journal of Animal Ecology* 11 (2): 215–44. https://doi.org/10.2307/1358.

Erickson, Bruce. 2015. "Embodied Heritage on the French River: Canoe Routes and Colonial History." *The Canadian Geographer / Le Géographe Canadien* 59 (3): 317–27. https://doi.org/10.1111/cag.12171.

ESRI. 2008. "ArcGIS Desktop: Release 9.3." Redlands, CA: Environmental Systems Research Institute.

Finney, Bruce P., Irene Gregory-Eaves, Marianne S. V. Douglas, and John P. Smol. 2002. "Fisheries Productivity in the Northeastern Pacific Ocean over the Past 2,200 Years." *Nature* 416 (6882): 729–33. https://doi.org/10.1038/416729a.

Fonzo, Martina M I Di, B Collen, A Chauvenet, Georgina M. Mace, and Claudia Bieber. 2016. "Patterns of Mammalian Population Decline Inform Conservation Action." *Journal of Applied Ecology* 53 (4): 1046–54. https://doi.org/10.1111/1365-2664.12659.

Gamarra, Javier G P, and Ricard V. Solé. 2000. "Bifurcations and Chaos in Ecology: Lynx Returns Revisited." *Ecology Letters* 3 (2): 114–21. https://doi.org/10.1046/j.1461-0248.2000.00128.x.

Gaston, Kevin J., and Richard A. Fuller. 2008. "Commonness, Population Depletion and Conservation Biology." *Trends in Ecology and Evolution* 23 (1): 14–19. https://doi.org/10.1016/j.tree.2007.11.001.

Grooten, M, and REA Almond. 2018. "Living Planet Report 2018: Aiming Higher." Gland, Switzerland. https://www.wwf.org.uk/sites/default/files/2018-10/wwfintl_livingplanet_full.pdf.

Hewitt, CG. 1921. *The Conservation of the Wild Life of Canada*. New York: C. Scribner.

Hewitt, David G, and David G Hewitt. 2015. "Hunters and the Conservation and Management of White-Tailed Deer (Odocoileus Virginianus)." *International Journal of Environmental Studies* 72 (5): 839–49. https://doi.org/10.1080/00207233.2015.1073473.

Hope, D. 2016. "Britain and the Fur Trade: Commerce and Consumers in the North-Atlantic World, 1783-1821." Northumbria University. http://nrl.northumbria.ac.uk/31598/.

Hunter, C. M., H. Moller, and D. Fletcher. 2000. "Parameter Uncertainty and Elasticity Analyses of a Population Model: Setting Research Priorities for Shearwaters." *Ecological Modelling* 134 (2–3): 299–324. https://doi.org/10.1016/S0304-3800(00)00344-6.

"Iucn Red List of Threatened Species." n.d. Accessed January 1, 2014. https://www.google.com/search?q=iucn+red+list&oq=iucn+red+list&aqs=chrome..69i57j69i60j0l4.1621j0j1&sourceid=chrome&ie=UTF-8.

Jackson, Jeremy B C, Michael X Kirby, Wolfgang H Berger, Karen A Bjorndal, Louis W Botsford, Bruce J Bourque, Roger H Bradbury, et al. 2001. "Historical Collapse Overfishing of and the Recent Coastal Ecosystems." *Science* 293 (5530): 629–38. https://doi.org/10.1126/science.1059199.

Jones, J W. 1914. *Fur Farming in Canada*. Edited by Game and Fur-Bearing Animals Commission of Conservation Canada, Committee on Fisheries. Ottawa: The Mortimer Co., Ltd.

Kahn, P. H, and B Friedman. 1995. "Environmental Views and Values of Children in an Inner-City Black Community." *Child Development* 66: 1403–17.

Kimura, Daniel K., James W. Balsiger, and Daniel H. Ito. 1984. "Generalized Stock Reduction Analysis." *Canadian Journal of Fisheries and Aquatic Sciences* 41 (9): 1325–33. https://doi.org/10.1139/f84-162.

Koslow, J. Anthony, and Jennifer Couture. 2013. "Follow the Fish." *Nature* 502: 163–64. https://doi.org/10.1038/502163a.

Krebs, Charles J, S. Boutin, R Boonstra, Anthony R E Sinclair, J N M Smith, M R T Dale, K Martin, and R Turkington. 1995. "Impact of Food and Predation on the Snowshoe Hare Cycle." *Science* 269 (5227): 1112–15.

Kroon, Hans de, Anton Plaisier, Jan van Groenendael, and Hal Caswell. 1986. "Elasticity: The Relative Contribution of Demographic Parameters to Population Growth Rate." *Ecology* 67 (5): 1427–31.

Lotze, Heike K., and Boris Worm. 2009. "Historical Baselines for Large Marine Animals." *Trends in Ecology and Evolution* 24 (5): 254–62. https://doi.org/10.1016/j.tree.2008.12.004.

Lunn, N. J., Scott. Schliebe, Erik W. Born, and International Union for Conservation of Nature and Natural Resources. Species Survival Commission. 2002. "Polar Bears : Proceedings of the 13th Working Meeting of the IUCN/SSC Polar Bear Specialist Group." Nuuk, Greenland: IUCN--the World Conservation Union.

McAllister, Murdoch K, and James N Ianelli. 1997. "Bayesian Stock Assessment Using Catch-Age Data and the Sampling-Importance Resampling Algorithm."

Canadian Journal of Fisheries and Aquatic Sciences 54: 284–300.

Mcclenachan, Loren, Francesco Ferretti, and Julia K. Baum. 2012. “From Archives to Conservation: Why Historical Data Are Needed to Set Baselines for Marine Animals and Ecosystems.” *Conservation Letters* 5 (5): 349–59. https://doi.org/10.1111/j.1755-263X.2012.00253.x.

McRae, Louise, Stefanie Deinet, and Robin Freeman. 2017. “The Diversity-Weighted Living Planet Index: Controlling for Taxonomic Bias in a Global Biodiversity Indicator.” Edited by Hideyuki Doi. *PLOS ONE* 12 (1): e0169156. https://doi.org/10.1371/journal.pone.0169156.

Miller, Joshua H. 2011. “Ghosts of Yellowstone: Multi-Decadal Histories of Wildlife Populations Captured by Bones on a Modern Landscape.” *PLoS ONE* 6 (3). https://doi.org/10.1371/journal.pone.0018057.

Mowat, F. 2012. “Sea of Slaughter.” In , 160–64. Vancouver, British Columbia: Douglas and McIntyre.

Myers, Ransom A., and Boris Worm. 2003. “Rapid Worldwide Depletion of Predatory Fish Communities.” *Nature* 423 (6937): 280–83. https://doi.org/10.1038/nature01610.

NERC Centre for Population Biology, Imperial College. 2010. “The Global Population Dynamics Database Version 2.” 2010. http://www.sw.ic.ac.uk/cpb/cpb/gpdd.html.

Novak, M. 1987. *Wild Furbearer Management and Conservation in North America*. Ontario: Association, Ontario Ministry of Natural Resources; Ontario Trappers. https://www.google.com/search?q=wild+furbearer+management+and+conservation&oq=wild+furbearer&aqs=chrome.0.0j69i57.2315j0j1&sourceid=chrome&ie=UTF-8.

Oliver, Tom H., Matthew S. Heard, Nick J B Isaac, David B. Roy, Deborah Procter, Felix Eigenbrod, Rob Freckleton, et al. 2015. “Biodiversity and Resilience of Ecosystem Functions.” *Trends in Ecology & Evolution* 30 (11): 673–84. https://doi.org/10.1016/j.tree.2015.08.009.

Papworth, S.K., J. Rist, L. Coad, and E.J. Milner-Gulland. 2009. “Evidence for Shifting Baseline Syndrome in Conservation.” *Conservation Letters* 2: 93–100. https://doi.org/10.1111/j.1755-263X.2009.00049.x.

Pauly, Daniel. 1995. “Anecdotes and the Shifting Baseline Syndrome of Fisheries.” *Trends in Ecology & Evolution* 10 (10): 430. https://doi.org/10.1016/S0169-5347(00)89171-5.

Pinnegar, John K., and Georg H. Engelhard. 2008. “The ‘shifting Baseline’ Phenomenon: A Global Perspective.” *Reviews in Fish Biology and Fisheries* 18 (1): 1–16. https://doi.org/10.1007/s11160-007-9058-6.

Poland, H. 1892. *Fur-Bearing Animals: In Nature and in Commerce*. Gurney & Jackson.

Reimoser, Friedrich, and Susanne Reimoser. 2017. “Long-Term Trends of Hunting Bags and Wildlife Populations in Central Europe . Long-Term Trends of Hunting Bags and Wildlife Populations in Central Europe,” no. December 2016.

Roman, Joe, Meagan M. Dunphy-Daly, David W. Johnston, and Andrew J. Read. 2015. “Lifting Baselines to Address the Consequences of Conservation Success.” *Trends in Ecology and Evolution* 30 (6): 299–302. https://doi.org/10.1016/j.tree.2015.04.003.

Row, Jeffrey R., Paul J. Wilson, Celine Gomez, Erin L. Koen, Jeff Bowman, Daniel Thornton, and Dennis L. Murray. 2014. “The Subtle Role of Climate Change on Population Genetic Structure in Canada Lynx.” *Global Change Biology* 20 (7):

2076–86. https://doi.org/10.1111/gcb.12526.

Rowe, Rebecca J., and Rebecca C. Terry. 2014. “Small Mammal Responses to Environmental Change: Integrating Past and Present Dynamics.” *Journal of Mammalogy* 95 (6): 1157–74. https://doi.org/10.1644/13-MAMM-S-079.

RStudio Team. 2015. “RStudio: Integrated Development for R.” Boston, MA: RStudio, Inc. https://www.rstudio.com/.

Sáenz-Arroyo, a, C. M. Roberts, J Torres, and M Cariño-Olvera. 2005. “Using Fishers’ Anecdotes, Naturalists´observations and Grey Literature to Reassess Marine Species at Risk: The Case of the Gulf Grouper in the Gulf of California, Mexico.” *Fish and Fisheries* 6: 121–33. https://doi.org/10.1111/j.1467-2979.2005.00185.x.

Seton, E T. 1909. *Life Histories of Northern Animals: An Account of the Mammals of Manitoba*. New York City: Scribner. https://www.google.com/search?ei=0KJJWs6DAcLd0gLdtZK4Dw&q=seton+mammals+1909&oq=seton+mammals+1909&gs_l=psy-ab.3...71063.74022.0.74295.12.12.0.0.0.0.146.895.0j8.8.0....0...1c.1.64.psy-ab..4.4.475...0j0i22i30k1j33i160k1.0.W-vXm0nO9as.

Shoemaker, Kevin T., and H. Resit Akçakaya. 2015. “Inferring the Nature of Anthropogenic Threats from Long-Term Abundance Records.” *Conservation Biology* 29 (1): 238–49. https://doi.org/10.1111/cobi.12353.

Simmons, D. 2007. *Keepers of the Record: The History of the Hudson’s Bay Company Archives*. Quebec: McGill-Queen’s University Press.

Soga, Masashi, and Kevin J. Gaston. 2018. “Shifting Baseline Syndrome: Causes, Consequences, and Implications.” *Frontiers in Ecology and the Environment* 16 (4): 222–30. https://doi.org/10.1002/fee.1794.

Spooner, Fiona E.B., Richard G. Pearson, and Robin Freeman. 2018. “Rapid Warming Is Associated with Population Decline among Terrestrial Birds and Mammals Globally.” *Global Change Biology* 24 (10): 4521–31. https://doi.org/10.1111/gcb.14361.

Sundby, S, and O Nakken. 2005. “Spatial Shifts in Spawning Habitats of Arcto-Norwegian Cod Induced by Climate Change.” *ICES Journal of Marine Science* 65 (April): 953–62. http://icesjms.oxfordjournals.org/content/65/6/953.short.

Turvey, Samuel T., Leigh A. Barrett, Hao Yujiang, Zhang Lei, Zhang Xinqiao, Wang Xianyan, Huang Yadong, Zhou Kaiya, Tom Hart, and Wang Ding. 2010. “Rapidly Shifting Baselines in Yangtze Fishing Communities and Local Memory of Extinct Species.” *Conservation Biology* 24 (3): 778–87. https://doi.org/10.1111/j.1523-1739.2009.01395.x.

Voorhis, Ernest. 1930. “Historic Forts and Trading Posts of the French Regime and of the English Fur Trading Companies.” 1930.

Walters, Carl J, Steven J.D. Martell, and Josh Korman. 2006. “A Stochastic Approach to Stock Reduction Analysis.” *Canadian Journal of Fisheries and Aquatic Sciences* 63 (1): 212–23. https://doi.org/10.1139/f05-213.

Watson, Robert T, Thomas Rosswall, Achim Steiner, Klaus Töpfer, Salvatore Arico, and Peter Bridgewater. 2005. *Ecosystems and Human Well-Being*. Edited by Robert T Watson, Thomas Rosswall, Achim Steiner, Klaus Töpfer, Salvatore Arico, and Peter Bridgewater. *Ecosystems*. Vol. 5. Millenium Assessments. World Resources Institute. https://doi.org/10.1196/annals.1439.003.

Whitlock, Rebecca E, Nicholas J Aebischer, and Jonathan C Reynolds. 2003. “The National Gamebag Census as a Tool for Monitoring Mammal Abundance in the UK A Report to JNCC,” no. September: 1–59.

Wiig, O, S Amstrup, T Atwood, K Laidre, N Lunn, M Obbard, E Regehr, and G Thiemann. 2015. “Ursus Maritimus. The IUCN Red List of Threatened Species 2015.” https://doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en.

Wilcove, David S., David Rothstein, Jason Dubow, Ali Phillips, and Elizabeth Losos. 1998. “Quantifying Threats to Imperiled Species in the United States.” *BioScience* 48 (8): 607–15. https://doi.org/10.2307/1313420.

Willis, K. J, M. B Araujo, K. D Bennett, B. Figueroa-Rangel, C. A Froyd, and N. Myers. 2007. “How Can a Knowledge of the Past Help to Conserve the Future? Biodiversity Conservation and the Relevance of Long-Term Ecological Studies.” *Philosophical Transactions of the Royal Society B: Biological Sciences* 362 (1478): 175–87. https://doi.org/10.1098/rstb.2006.1977.

Willis, K. J, L Gillson, T. M Brncic, and B. Figueroa-Rangel. 2005. “Providing Baselines for Biodiversity Measurement.” *Trends in Ecology and Evolution* 20 (3): 107–8. https://doi.org/10.1016/j.tree.2005.01.007.

Winfree, Rachael, Jeremy W. Fox, Neal M. Williams, James R. Reilly, and Daniel P. Cariveau. 2015. “Abundance of Common Species, Not Species Richness, Drives Delivery of a Real-World Ecosystem Service.” Edited by Katriona Shea. *Ecology Letters* 18 (7): 626–35. https://doi.org/10.1111/ele.12424.

Yan, Chuan, Nils Chr Stenseth, Charles J. Krebs, and Zhibin Zhang. 2013. “Linking Climate Change to Population Cycles of Hares and Lynx.” *Global Change Biology* 19 (11): 3263–71. https://doi.org/10.1111/gcb.12321.

Tables and Figures

Table 1. Estimated percentage population change from 1850 and 1970 to 2009 for the eight modelled species. Mean population values are presented in bold and confidence intervals are presented in brackets.

Species	**1850 population**	**1970 population**	**2009 population**	**1850 – 2009 total (& annual) population loss or gain (%)**	**1970 -2009 total (& annual) population loss or gain (%)**
Arctic fox	133,000	132,000	133,000	**0** (0)	**1** (0.02)
Canadian lynx	262,000	258,000	290,000	**11** (0.06)	**12** (0.32)
Beaver	4,730,000	3,730,000	4,440,000	**-6** (-0.04)	**19** (0.49)
Bobcat	50,300	44,100	47,000	**-7** (-0.04)	**7** (0.17)
Polar bear	20,300	13,900	15,800	**-22** (-0.14)	**14** (0.35)
Black bear	831,000	414,000	390,000	**-53** (-0.33)	**-6** (-0.15)
Fisher	167,000	157,000	7,000	**-96** (-0.60)	**-96** (-2.45)
American badger	518,000	175,000	83,800	**-84** (-0.53)	**-52** (-1.34)

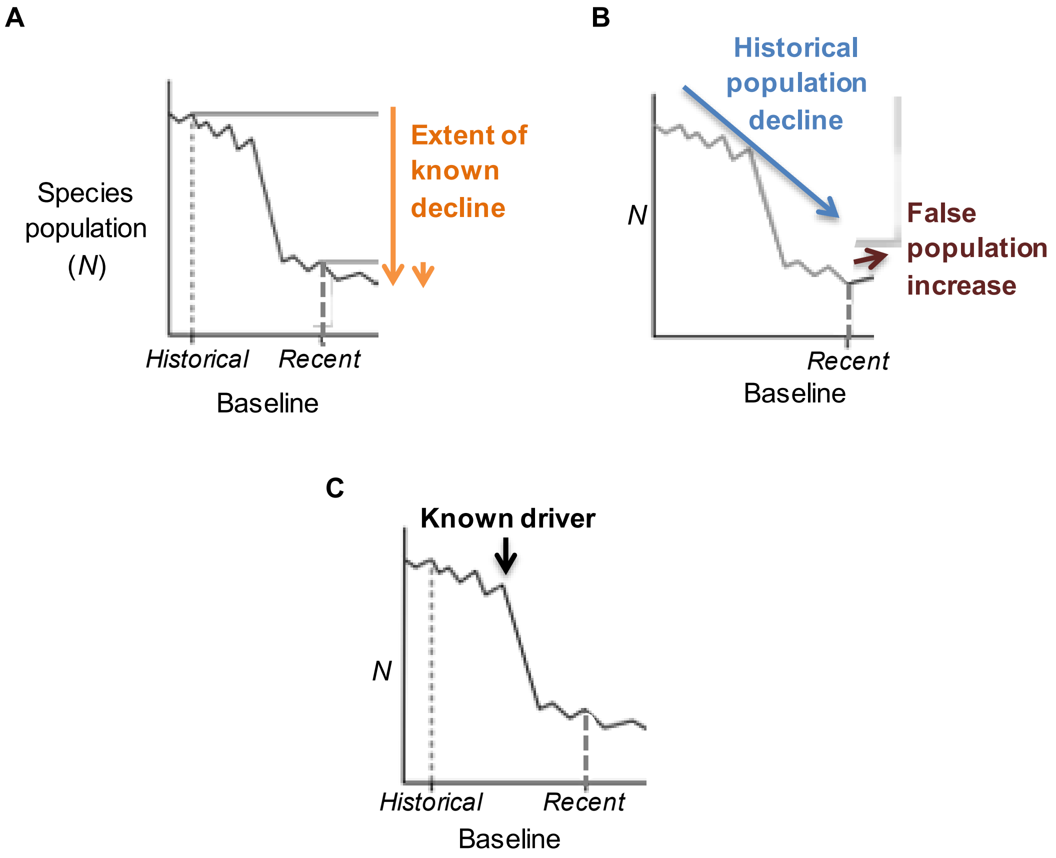


Figure 1. Schematic of hypothetical population trends highlighting the benefits of establishing a historical baseline through (A) providing better context on the extent of decline, thus allowing for better population recovery scenarios when advising on recovery targets (blue arrows signify extent of known decline), (B) identifying a trend through different time series lengths (represented by blue arrows) and discriminating between true decreasing trends and misattributed trends caused by population cycles, and (C) estimating the impact on populations from known drivers of decline captured through use of historical baselines.

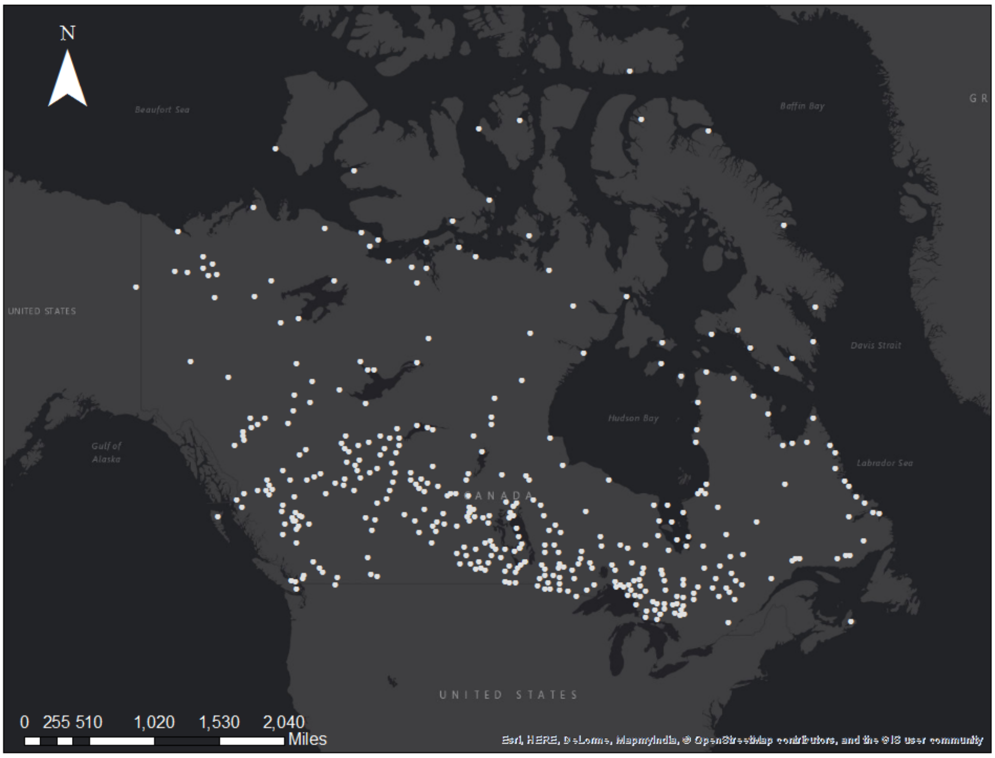


Figure 2. The approximate locations of all Hudson's Bay Company trading posts that were operating in 1927 in North America, adapted from Elton and Nicholson (1942a).

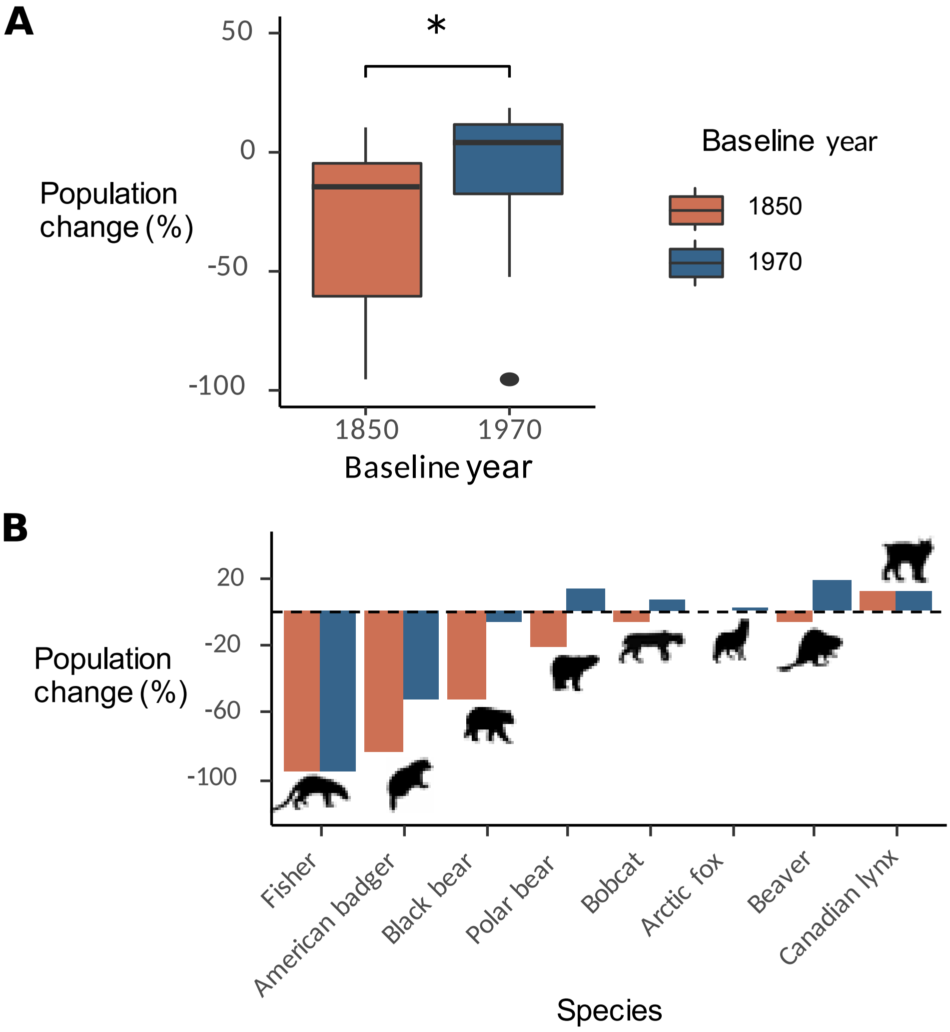


Figure 3. Percentage change in population estimates using two different baseline years: 1850-2009 and 1970-2009 for (A) species cumulatively (t-test: $p < 0.05$; 1970: 4%, 1850: -14.5% medians ± S.E, $n = 8$), and (B) species-specific population estimates.

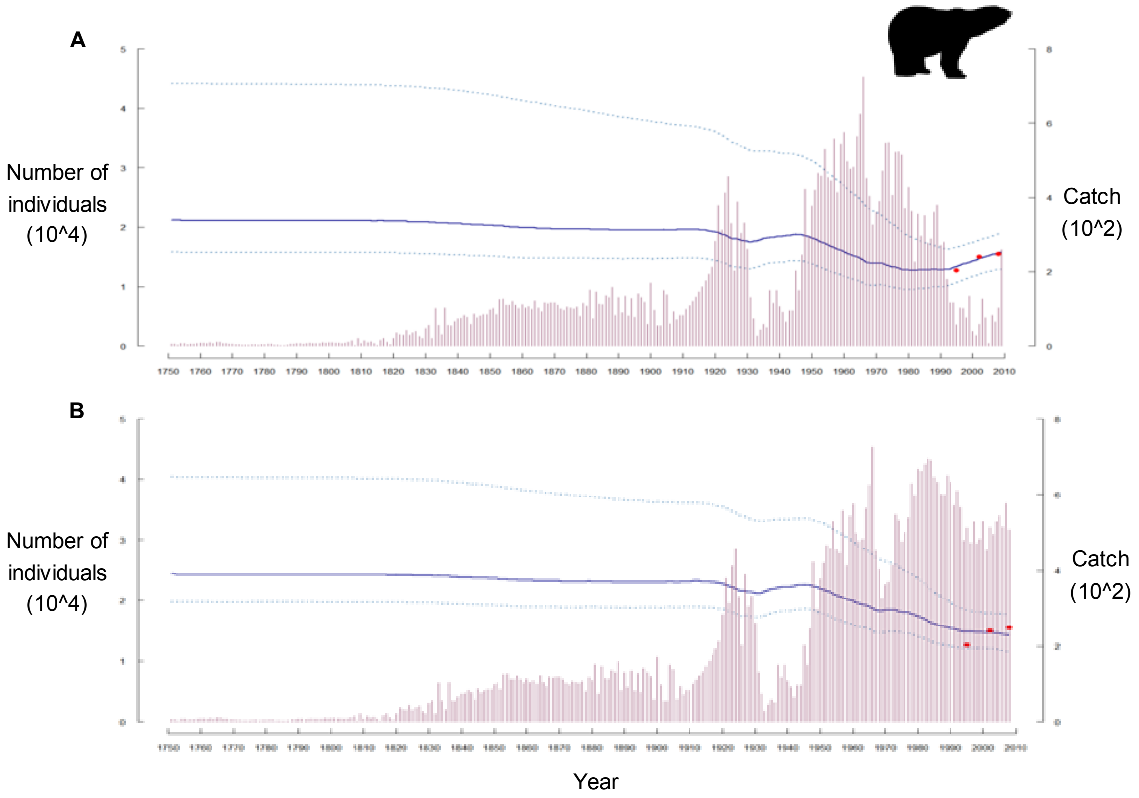


Figure 4. Population trajectories for polar bear, using, from 1970, (A) Canadian government annual fur harvest data and (B) IUCN/SSC statistics on number of polar bears annually killed in Canada. Pink bars are the annual fur harvest (right-hand axis), the blue solid line and blue dashed lines are the population projection and the 95% median confidence intervals respectively; the current Canada-wide population estimate is shown by the red dots (left-hand axis).

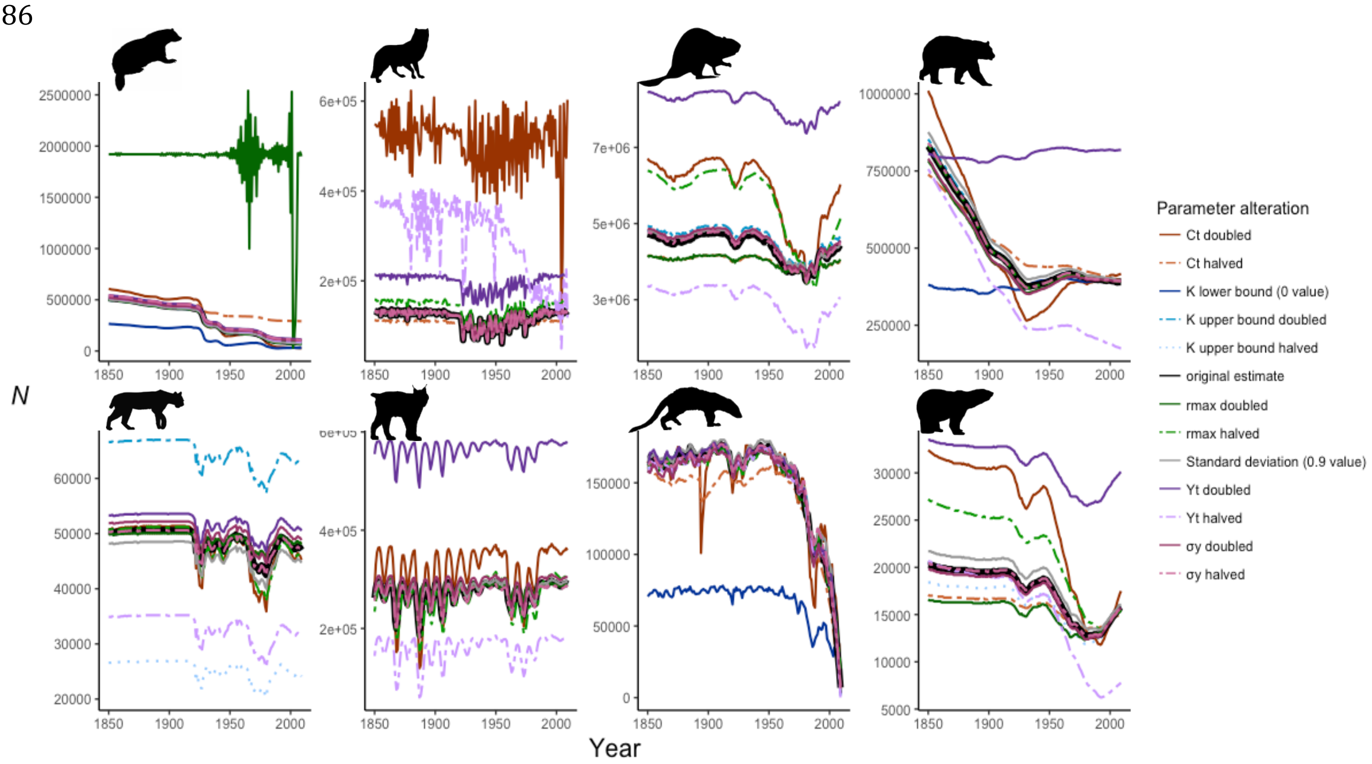


Figure 5. Elasticity analysis for each species, showing how population change between 1850-2009 altered when each parameter was doubled (+50) and halved (-50) for number of individuals caught (C_t, orange lines); carrying capacity (K) lower bound and carrying capacity upper bound (blue lines); original SSRA estimate (black line); intrinsic growth rate (r_{max}, green lines); standard deviation altered to 0.9 (grey line); abundance estimate(s) (y_t; purple lines); observation error (σ_{y}, pink lines).

Acknowledgements

Dedicated to the memory of Ben Collen, a truly inspiring mentor and friend.

The authors would like to thank Christensen (2006) for original R code to the SSRA model, and Louise McRae, John Mola and Kate Tiedeman for comments on the manuscript.

Funding sources

Funding for this project was awarded by the Vodafone’s World of Difference and a generous grant from the Rufford Foundation.

Declaration of Interest

The authors have no competing interests to declare.

Authors’ contributions

Amy Collins: Conceptualization, Methodology, Software, Formal analysis, Writing- Original Draft, Visualization. **Monika Böhm:** Resources, Writing – Review & Editing, Supervision. **Ben Collen:** Conceptualization, Validation, Writing – Review & Editing, Supervision.

Supplementary Material for

Reconstructing historical population baselines for hunted mammals

Tables

Table S1. Data availability for 25 mammal species for which Hudson's Bay Company data exists. The year highlighted in red represents the first year that harvest data is available (BC = British Columbia).

Species		Harvest datasets						Current estimate yt
	Common name	**Poland (1892)** 1751-1889	**Jones (1914)** 1890-1909	**Novak (1987)** 1919-1969	**Fur Statistics Canada (2010)** 1970-2009	**Other harvest Data**	**Extra interpolated years**[1]	
Ursus americanus	Black (and brown) bear	✓ 1751	✓	✓	✓			✓
Lynx canadensis	Canadian Lynx	✓	✓	✓	✓	Elton and Nicholson (1942) 1735-1750	1942	✓
Martes pennanti	Fisher	✓ 1766	✓	✓	✓			Extrapolated from BC
Ursus maritimus	Polar bear	✓ 1751	✓	✓	✓			✓
Vulpes lagopus	Arctic fox	✓ 1751	✓	✓	✓			✓
Taxidea taxus	American Badger	✓ 1842	✓	✓	✓	Gobal Population Dynamics Database (2010) 1891-1908	1908-1910	✓
Castor canadensis	Beaver	✓	✓	✓	✓	Carlos &	1815, 1827-	Extrapolated

						Lewis (1983) 1722-1750 GPDD (2010) 1891-1908	1831 1908-1910	from BC
Lynx rufus	Bobcat	☒	✓ (wild cat) 1848	✓	✓		1885, 1888, 1903, 1906, 1908, 1919	Extrapolated from US states which share a border with Canada
Ondatra zibethicus	Muskrat	✓	✓	✓	✓	GPDD (2010) 1919-1956		
Martes americana	Marten	✓	✓	✓	✓	GPDD (2010) 1934-1970		
Vulpes vulpes	Red fox	✓	✓	✓	✓		1919	
Neovison vison	Mink	✓	✓	✓	✓		1763	
Ursus arctos	Grizzly bear	☒	☒	☒	✓			
Canis latrans	Coyote	☒	☒	✓	✓			
Puma concolor	Cougar	☒	☒	☒	✓			
Canis lupus	Wolf	✓	✓	☒	✓			
Lontra canadensis	Otter	✓	✓	☒	✓			
procyon lotor	Raccoon	✓	✓	☒	✓			
Gulo gulo	Wolverine	✓	✓	☒	✓			

	Deer	✓	☒	☒	✓
Ovibos moschatus	Muskox	☒	✓	☒	✓
	Rabbit	✓	☒	☒	✓
Mephitis mephitis	Skunk	☒	✓	☒	✓
	Swan	✓	☒	☒	✓
Mustela erminea	Stoat	☒	✓	☒	✓

[1] **in addition to 1910-1918 and 2002, see methods in the main manuscript**

Table S2. Hudson's Bay Company harvest data for the eight species modelled in our analysis. HBC fur harvest data may report the year the furs were caught (y); the year when the furs were sent to HBC headquarters ($y + 1$); or the year the furs were sold at the London auctions ($y + 2$) (Powell *et. al.* 2003). To ensure consistency across both datasets, we backdated the data to reflect the number of furs caught in year y, by backdating the data from Poland (1892) by one year and that of Jones (1914) by two years, as described in Elton & Nicholson (1942).

Year	Species Harvest Data							
	Arctic fox	American Badger	Beaver	Black Bear	Bobcat	Canadian Lynx	Fisher	Polar Bear
1850	854	1471	62277	7106	243	8519	5967	97
1851	4012	1710	52491	7381	222	5361	5861	101
1852	4104	956	60691	6244	135	4552	4933	85
1853	1406	900	62914	9139	381	5682	4901	125
1854	10413	1240	72425	9218	330	11358	5210	126
1855	5014	999	76825	8070	214	23362	5563	110
1856	2123	1185	86414	8019	208	31642	5957	110
1857	1592	1369	94058	8800	189	33757	6950	120
1858	3398	1773	106797	8032	143	23226	7197	110
1859	5111	1213	107745	7372	134	15178	5853	101
1860	2828	1091	105562	8101	115	7272	5980	111
1861	3394	1642	109636	7467	164	4448	6053	102
1862	12324	1295	127674	7770	75	4926	5424	106
1863	4854	1562	118118	7236	63	5437	4953	99
1864	5955	1235	155880	8809	117	16498	4605	121
1865	5446	618	150192	7499	83	35971	4804	103
1866	2554	1644	145654	6825	94	76556	6311	93
1867	12212	1781	158110	8542	89	68392	7477	117
1868	4677	2288	129039	8305	68	37447	7959	114
1869	1820	1986	173181	8471	82	15686	6743	116
1870	2842	2274	174461	8452	46	7942	7072	116

1871	7415	2786	157764	8060	24	5123	3639	110
1872	5375	1512	105369	7329	28	7106	3539	100
1873	6127	1870	139393	7022	189	11250	3578	96
1874	4381	2280	129976	7697	83	18774	3263	105
1875	5347	2273	126959	7440	40	30508	3338	102
1876	24641	2474	145706	7313	10	42834	5461	100
1877	6018	2031	147263	7689	10	27345	6132	105
1878	2335	2866	132099	5869	2	17834	4216	80
1879	4412	1867	120836	8414	24	15386	5059	115
1880	5777	1428	119698	7911	6	9443	5143	108
1881	5923	1499	118728	11035	19	7599	4640	151
1882	6537	746	104459	5439	10	8061	3820	74
1883	2819	1330	119549	10618	24	27187	4200	145
1884	3298	1473	102589	8271	10	51511	1041	113
1885	4187	749	83589	8166	18	74050	4510	112
1886	13243	1109	102745	9942	33	78773	6165	136
1887	9628	777	33061	9474	18	33899	5408	130
1888	2918	1301	73355	11558	16	18886	6557	158
1889	3763	2445	64246	10371	14	11520	5683	83
1890	9709	4000	55000	13289	13	8352	5208	130
1891	4759	2600	56000	11073	5	8660	4828	90
1892	3265	3000	45000	8834	7	12902	4044	134
1893	5017	2000	45000	9810	29	20331	38631	81
1894	6748	2200	50000	9557	15	36853	4169	128
1895	3542	2000	50000	10348	50	56407	4805	77
1896	3274	1100	42000	10138	32	39437	5247	141
1897	6742	1200	34000	9903	27	26761	4964	130
1898	3642	1300	44000	10034	67	15185	5042	118

1899	2953	600	41000	8607	41	4473	3454	58
1900	8583	1100	43000	7875	5	5781	3716	170
1901	10841	700	45000	7171	4	9117	3235	96
1902	5622	500	35000	6725	5	19267	2590	55
1903	4707	500	30000	5077	3	36116	2095	54
1904	6438	500	55000	5536	2	58850	3020	149
1905	11539	300	35000	4612	2	61478	4022	138
1906	6849	100	32000	4488	1	36300	4701	60
1907	2082	100	35000	4439	1	9704	3600	93
1908	4831	138	40651	5032	1	3410	2525	71
1909	14805	190	47213	5348	2	3774	2310	82
1910	15094	262	54836	5251	3	6149	2679	82
1911	15388	362	63689	5555	6	10020	3107	99
1912	15689	499	73971	5876	10	16327	3604	109
1913	15995	688	85913	6216	18	26604	4180	120
1914	16307	948	99783	6576	31	43349	4848	132
1915	16625	1308	115893	6956	55	31998	5623	145
1916	16949	1804	134603	7359	95	23619	6522	160
1917	17280	2487	156334	7784	164	17434	7565	176
1918	17617	3430	181574	8235	285	12869	8774	194
1919	17961	4730	210888	8771	495	9499	10176	213
1920	21537	930	164656	7786	859	6509	4866	282
1921	41082	1627	232134	9511	3583	11673	5689	378
1922	77648	2773	175275	7368	1129	17317	3976	313
1923	34973	5147	169172	6999	3279	26437	4158	412
1924	44535	9298	151913	8429	3941	29608	4230	456
1925	35397	13373	111707	6173	3324	33054	5899	330
1926	51552	22073	100364	6374	3403	28706	7893	203

1927	34099	33516	74338	6857	1493	21369	8641	388
1928	18687	34761	67043	6828	1728	11604	6606	304
1929	38028	14793	47715	6588	827	7621	4274	331
1930	72176	10039	51313	4634	464	7976	3282	261
1931	67924	4964	65276	2754	800	8454	2739	130
1932	33582	3159	71699	2241	1654	11932	2530	73
1933	61782	3047	59199	1416	1917	16799	3171	27
1934	69033	1225	50175	1123	2218	22014	3728	43
1935	46151	157	44600	1025	1884	22456	4624	59
1936	22625	141	55759	1885	1565	17539	5237	49
1937	56879	203	54148	1642	1093	10538	3505	150
1938	57007	3237	64086	1626	1125	8109	3399	115
1939	33111	4663	78659	1058	1184	7473	2886	150
1940	48980	8240	90123	1056	1138	6684	2212	106
1941	63654	11478	106176	1218	2124	7109	3408	91
1942	74637	7575	102241	1032	2117	8512	2165	65
1943	30716	11212	130764	1448	2214	10191	3303	95
1944	18122	5708	128999	2344	2200	12329	3631	95
1945	27585	6373	153899	1145	1585	9338	4141	202
1946	68363	2090	127622	1150	1365	8147	4064	150
1947	55960	1034	135629	827	1265	6582	2788	246
1948	33460	685	161926	579	476	4110	4390	422
1949	20019	1125	157416	760	781	3734	2698	297
1950	53141	702	180817	410	649	9662	3690	377
1951	54393	508	222932	330	662	7324	5274	420
1952	41149	215	224606	598	501	13099	5531	465
1953	36853	268	242452	437	490	11325	5794	458
1954	82626	249	320389	324	345	14427	6790	530

1955	31999	271	282036	319	1404	9988	6324	454
1956	28565	395	280671	578	512	8748	5076	445
1957	32170	663	341674	775	564	9506	5720	558
1958	26752	348	328584	493	778	14165	4850	413
1959	14632	1446	344766	411	1336	40408	6462	544
1960	52245	827	399459	404	1326	42016	6206	575
1961	45769	658	386823	2256	855	47625	5863	497
1962	9934	388	436780	715	1070	51376	6254	477
1963	32618	409	463837	683	1133	36197	8364	482
1964	41038	395	415261	1211	1533	24534	7950	564
1965	11726	792	372635	2482	3305	14583	8216	624
1966	34315	1163	371533	1938	3658	13038	6856	724
1967	29791	1251	420437	1777	2483	15848	5535	452
1968	20299	1215	437875	3224	3266	20677	7627	404
1969	7477	4084	433408	3200	4118	37477	8146	326
1970	26301	2014	355379	2007	3408	42365	6637	361
1971	33788	2121	375213	2522	3837	53589	8278	389
1972	10265	5170	452275	3008	3682	53400	13798	472
1973	53623	5134	431071	4261	4129	35372	12566	546
1974	32120	3626	357732	3585	3425	20648	10163	548
1975	26913	5124	334924	3531	3103	13162	8698	406
1976	36840	6834	404625	3402	3459	15132	9664	522
1977	33262	5279	397125	3170	3790	21131	8798	523
1978	25853	7800	446416	4109	4499	29987	9771	515
1979	35697	7736	602044	4710	4494	34366	14725	371
1980	40828	4013	522966	3872	2499	34502	14935	427
1981	18291	3965	382893	2175	2773	39274	16413	363
1982	14532	4333	335711	2464	2300	29533	21005	227

1983	16096	3028	323877	2340	2128	13445	17931	355
1984	22952	3723	371685	2411	2128	8625	14483	296
1985	6493	2036	424086	2385	1861	6853	15537	351
1986	8394	2280	505996	2879	1749	6953	15020	277
1987	18663	2107	464992	3330	1343	6574	12664	288
1988	2524	1068	267427	1820	952	8265	9183	360
1989	2266	648	222605	3169	537	9977	7759	380
1990	1112	574	179962	2095	583	7579	8831	271
1991	3190	933	219764	1901	436	11542	15381	279
1992	3946	686	185965	2647	628	7180	13377	180
1993	10020	758	244561	2507	727	4713	13105	117
1994	11434	863	327229	2992	1066	4907	14578	116
1995	1999	884	243724	2807	976	2819	11806	57
1996	6106	646	320358	2224	1320	5171	15351	102
1997	6572	731	314804	2140	1457	6873	20253	101
1998	4358	397	266146	1984	1452	6148	15223	76
1999	3674	353	215246	2015	1801	8573	16638	134
2000	5897	496	221118	2918	1769	9361	16109	39
2001	7398	490	260421	3406	2070	11328	23456	28
2002	5347	849	224658	3250	2004	12341	21687	59
2003	3864	1471	193807	3102	1941	11896	20052	126
2004	1311	486	190930	2674	1632	9788	19345	76
2005	1430	774	198304	2289	1517	8445	19771	7
2006	3505	1063	210423	2183	1827	8455	19675	83
2007	2610	982	147685	2213	1866	6630	17151	65
2008	2520	624	152782	2008	2192	9216	17575	102
2009	1003	512	139220	2055	1770	7490	16373	259

Table S3. Median population growth rate r_{max}, standard deviation, and original referenced r_{max} values used for the eight species' prior probability in the Stochastic Stock Reduction Analysis.

Species Common Name	Species	Median R_{max}	SD	R_{max}	Source
Arctic Fox	*Alopex lagopus*	1.43	1.077821878	2.64	Hutchings *et al.* (2012)
				1.43	Hennemann (1983)
				0.49	The Global Population Dynamics Database (GPDD) in Fagan *et al.* (2001)
American Badger	*Taxidea taxus*	0.678	0.405862253	0.38	Hutchings *et al.* (2012)
				0.976	Fagan et al (2013)
				1.01	Thompson (1987)
				0.22	GPDD in Fagan *et al.* (2001)
Beaver	*Castor canadensis*	0.47	0.056047599	0.47	GPDD in Fagan *et al.* (2001)
				0.442	Lancia and Bishir (1985)
				0.55	Runge (1999)
Black Bear	*Ursus americanus*	0.23	0.125989998	0.23	Hutchings *et al.* (2012)
				0.197	Fagan et al (2013)
				0.43	GPDD in Fagan *et al.* (2001)
Bobcat	*Lynx rufus*	0.62	0.410121933	0.91	Hutchings *et al.* (2012)
				0.33	GPDD in Fagan *et al.* (2001)
Canadian Lynx	*Lynx canadensis*	0.855	0.111015765	0.855	Tanner (1975)
				1.012	Tanner (1975)
				0.26	GPDD in Fagan *et al.* (2001)
Fisher	*Martes pennanti*	0.428	0.152879315	0.536	Fagan et al 2013 [11]
				0.32	GPDD in Fagan *et al.* (2001)
Polar Bear	*Ursus maritimus*	0.06	0.202262775	0.0603	http://polarbearfeed.etiennebenson.com/detail/50/None/4/
				0.0055	Taylor *et al.* (2005)
				0.38	GPDD in Fagan *et al.* (2001)

Table S4. Current population estimates (y_t; 1979 – 2013) and data source.

Species Common Name	Species	Population estimate(s)	Year of estimate	Source
Arctic Fox	*Alopex lagopus*	100,000	2004	Sillero-Zubiri, Hoffman & Macdonald (2004)
American Badger	*Taxidea taxus*	31,225	2002	Scobie (2002)
Beaver	*Castor canadensis*	3,550,000	1979	
Black Bear	*Ursus americanus*	337,200	1989	Williamson (2002)
		369,500	1992	
		436,000	1996	
		434,400	2001	Hristienko & McDonald (2007)
Bobcat	*Lynx rufus*	59,832	2008	
Canadian Lynx	*Lynx canadensis*	275,000	2003	Poole (2003)
Fisher	*Martes pennanti*	8,131	1999	Weir (2000)
Polar Bear	*Ursus maritimus*	12,700	1995	Taylor & Lee (1995)
		15,000	2002	
		15,500	2008	

Table S5. Selection criteria used for allocating the proportion of the total error term, *k*, to the observation error, σ_y, based on the source and uncertainty associated with the data, with province-wide extrapolation of population size assigned the highest proportion. The remaining total error term is attributed to the process error, w_t.

Proportion of the total error term	Selection criteria
0.3	Single or multiple population estimates taken from a scientific journal/published article. Associated error or uncertainty surrounding the estimate(s) is recorded and small.
0.4	Single population estimate taken from a published article or published report and/or large associated error surrounding the estimate.
0.5	State-wide extrapolations to the entire current range required to obtain the current abundance.

Table S6. Upper and lower bound calculated for carrying capacity (*K*) priors.

Species Common Name	Species	Lower bound K	Upper bound K	Density estimate per km^2	Historical range estimate km^2 (Seton Vol II 1929 **[17]** unless stated otherwise)
Arctic Fox	*Alopex lagopus*	94,239	1,366,458	0.02 - 0.2 (Angerbjörn *et al.* 1999)	4,711,926
American Badger	*Taxidea taxus*	504,089	3,780,666	0.8 (Goodrich & Buskirk 1998) – 6 Messick & Hornocker 1981)	630,111
Beaver	*Castor canadensis*	2,554,444	352,513,327	0.4 (Feldhamer, Thompson & Chapman 2003) – 55.2 (Naiman, Johnston & Kelley 1988)	6,386,110
Black Bear	*Ursus americanus*	773,845	3,009,398	0.09 - 0.35 (McLean & Pelton 1994)	8,598,280 Pelton & Coley 1999)
Bobcat	*Lynx rufus*	9,454	55,568	0.09 - 0.529 (Roberts & Crimmins 2010)	105,042.7
Canadian Lynx	*Lynx canadensis*	115,500	2,598,750	0.02 - 0.45 (Sunquist & Sunquist 2002)	5,775,000
Fisher	*Martes pennanti*	182,786	1,389,175	0.05 - 0.38 Powell *et al.* 2003)	3,655,724
Polar Bear	*Ursus maritimus*	2,700	49,500	0.001 (Taylor & Lee 1995) - 0.018 (Derocher 1998)	2,701,800

Table S7. Number of polar bears killed for fur versus number killed from all major pressures from 1970 to 2008.

year	IUCN data - all anthropogenic pressures	Canadian government data – single threat, fur harvest	Extra number of individuals killed from all pressures
1970	362	361	1
1971	368	389	-21
1972	415	472	-57
1973	548	546	2
1974	532	548	-16
1975	476	406	70
1976	499	522	-23
1977	540	523	17
1978	629	515	114
1979	598	371	227
1980	668	427	241
1981	663	363	300
1982	680	227	453
1983	694	355	339
1984	690	296	394
1985	645	351	294
1986	632	277	355
1987	600	288	312
1988	599	360	239
1989	648	380	268
1990	632	271	361
1991	572	279	293
1992	609	180	429

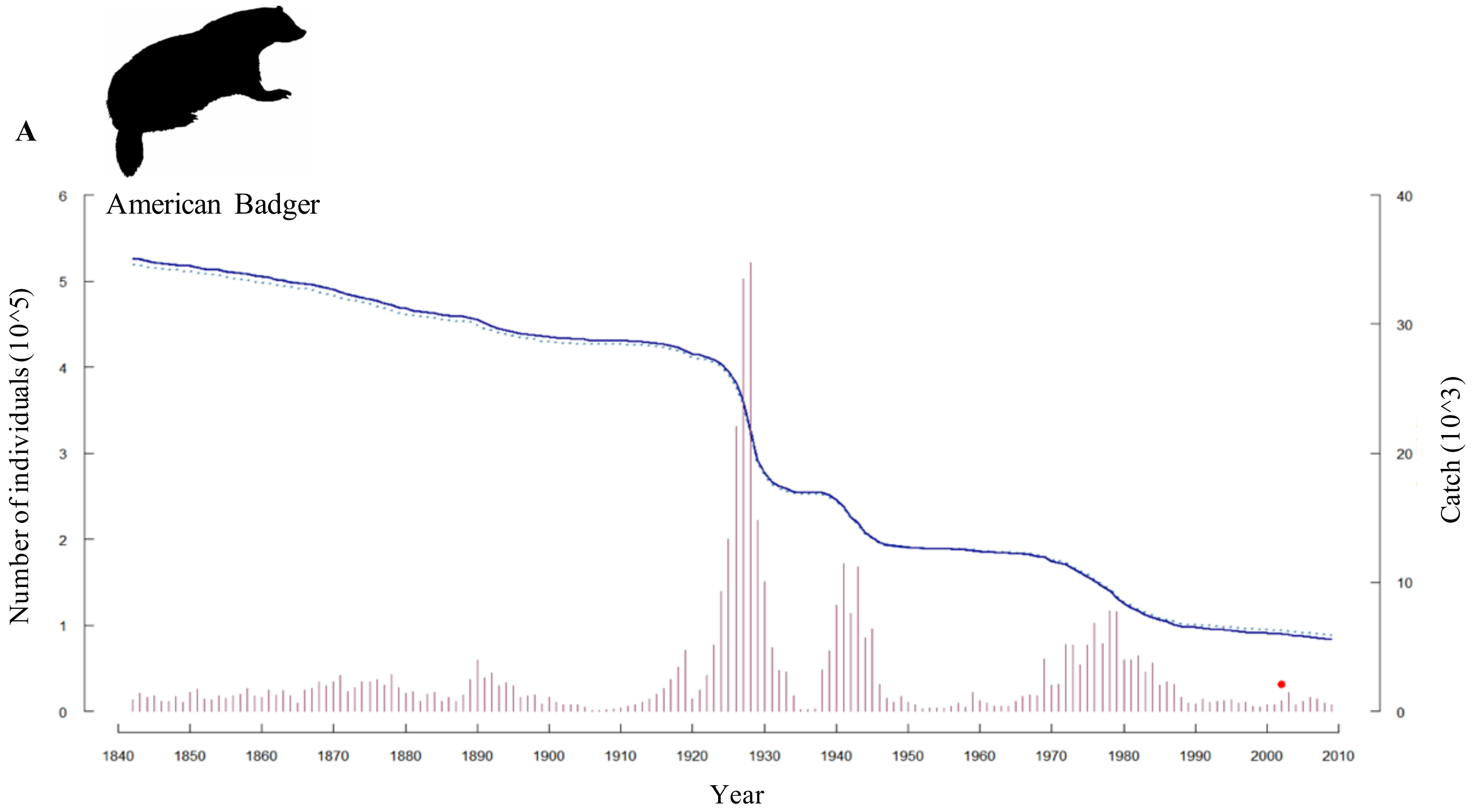
1993	567	117	450
1994	478	116	362
1995	511	57	454
1996	472	102	370
1997	529	101	428
1998	471	76	395
1999	482	134	348
2000	530	39	491
2001	453	28	425
2002	512	59	453
2003	495	126	369
2004	529	76	453
2005	545	7	538
2006	514	83	431
2007	577	65	512
2008	506	102	404
average/year	551	256	294

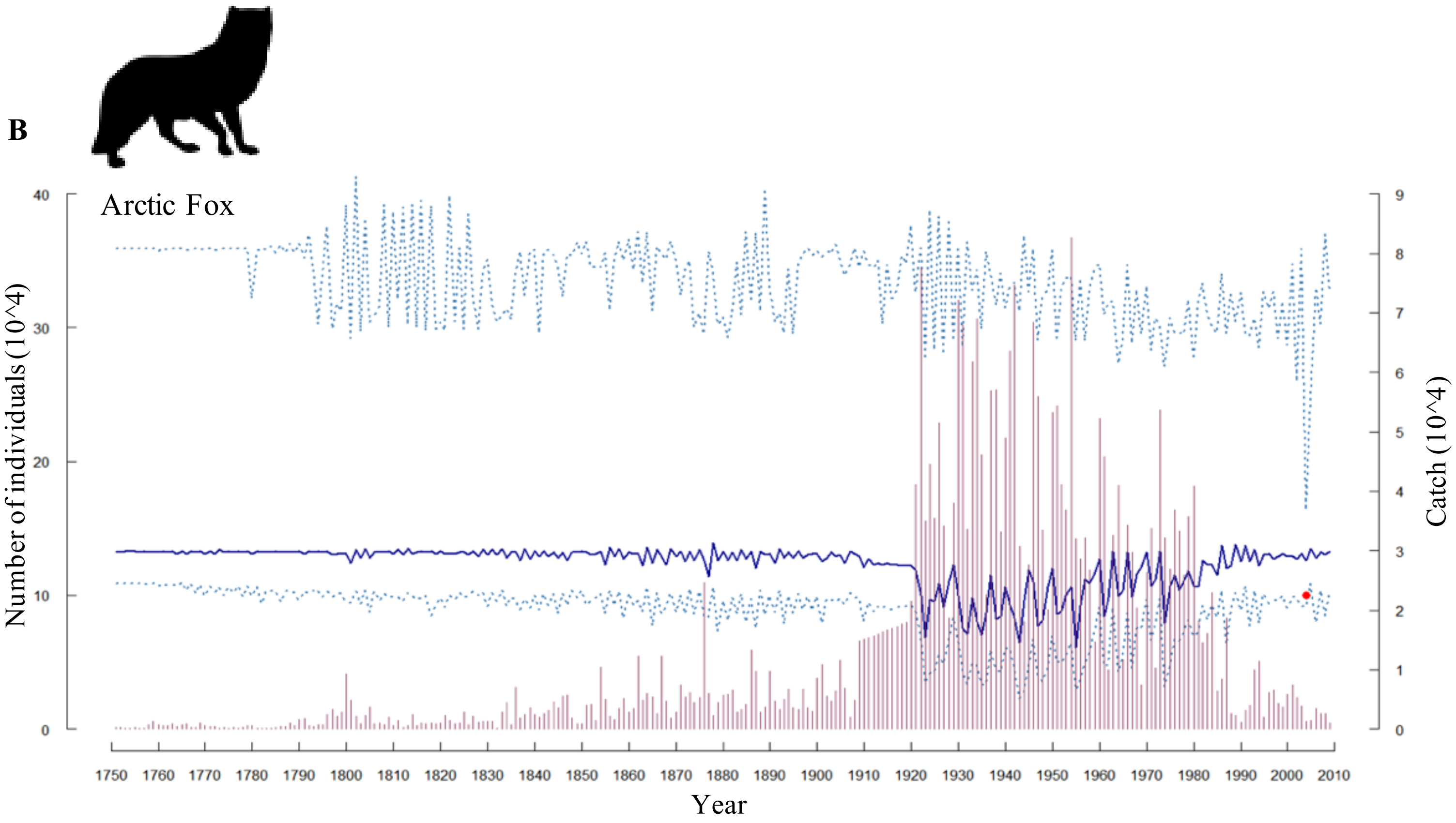
Table S8. Population change estimated from the SSRA model for each baseline scenario.

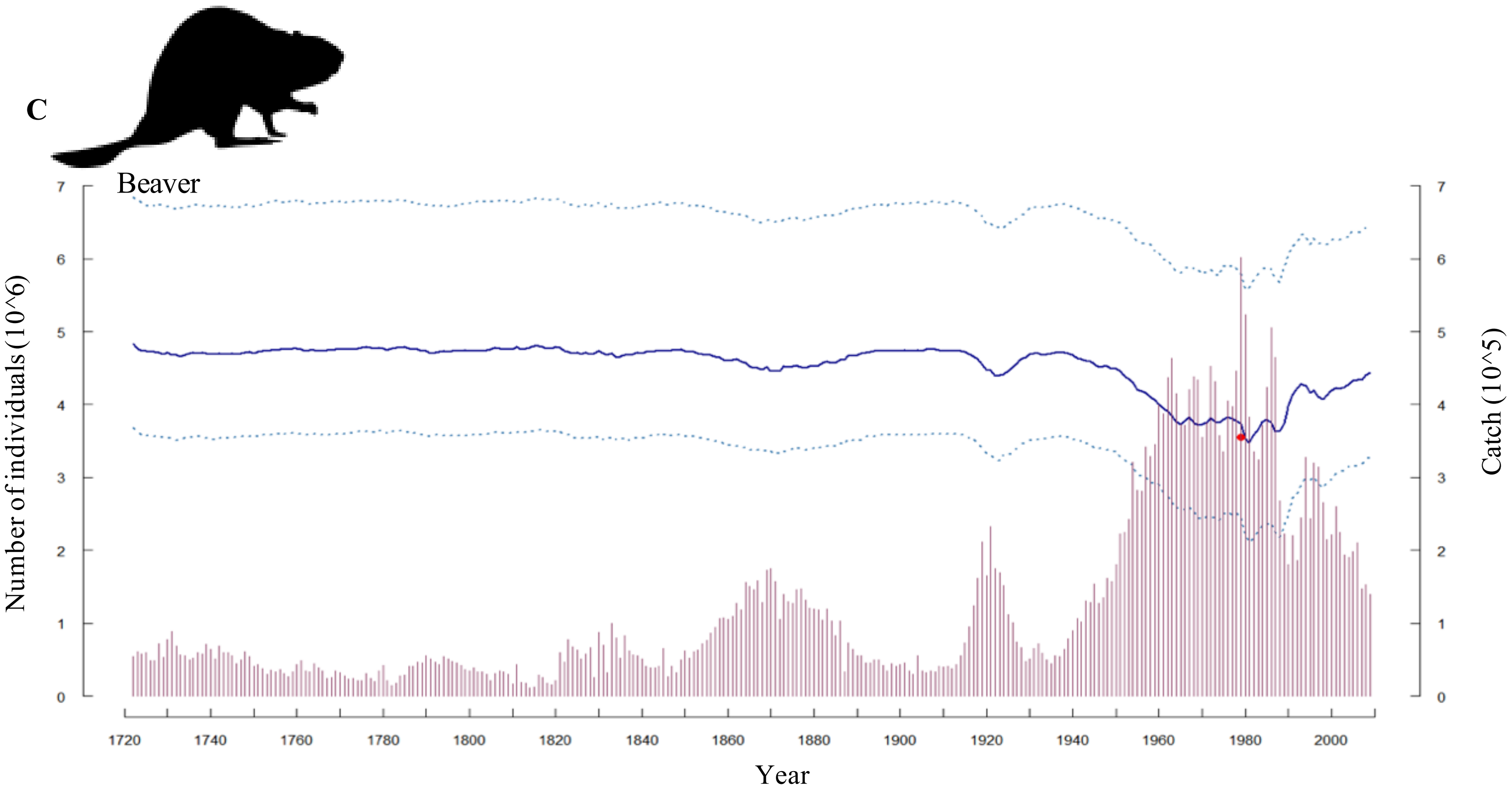
Species	Population change (%)	
Baseline	**1850**	**1970**
Arctic fox	0	1
American Badger	-84	-52
Beaver	-6	19
Black bear	-53	-6
Bobcat	-7	7
Fisher	-96	-96
Canadian Lynx	11	12
Polar bear	-22	14

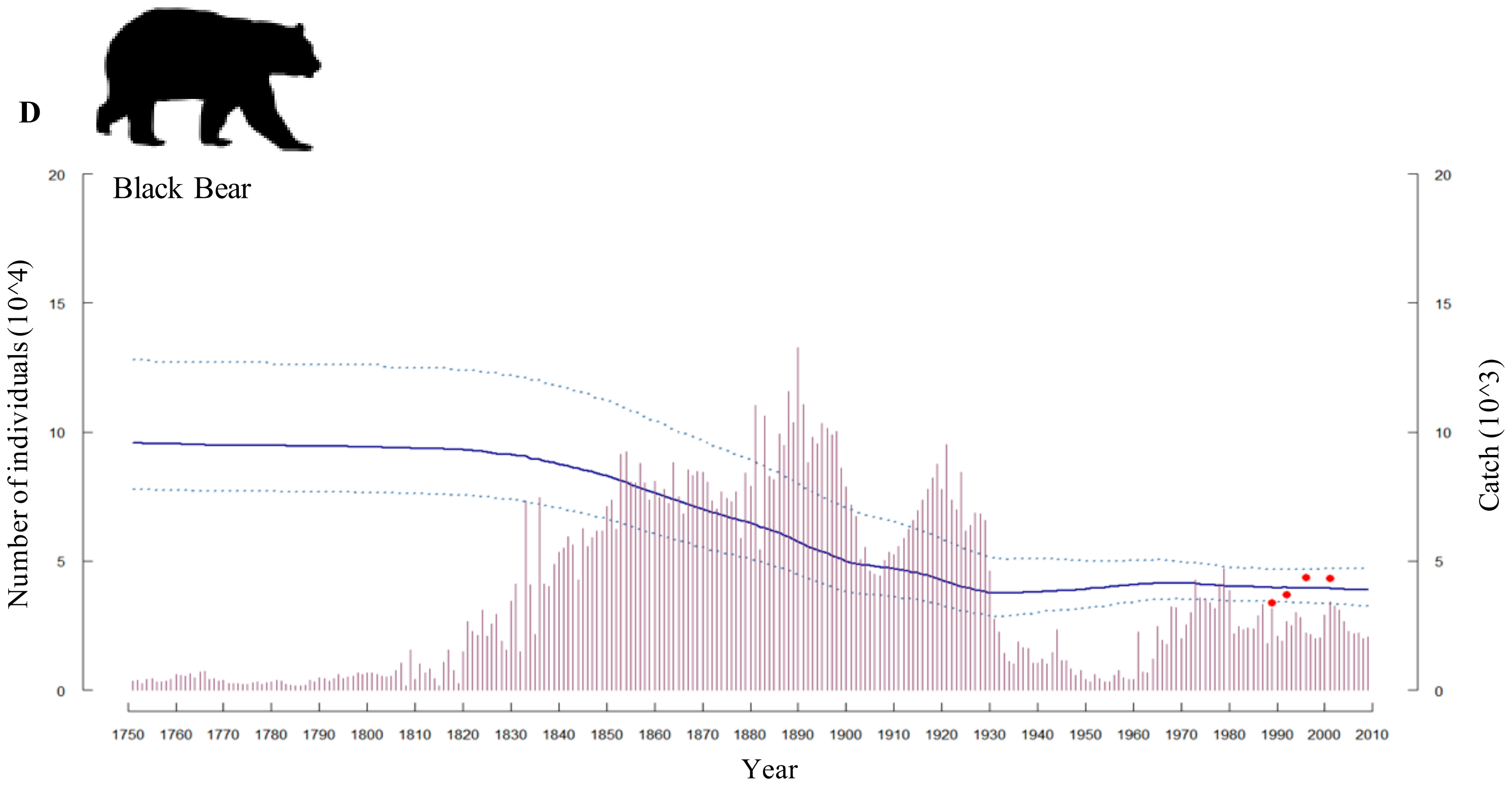
Table S9. Elasticities of species-specific population change (1850 – 2009) with respect to parameter alterations.

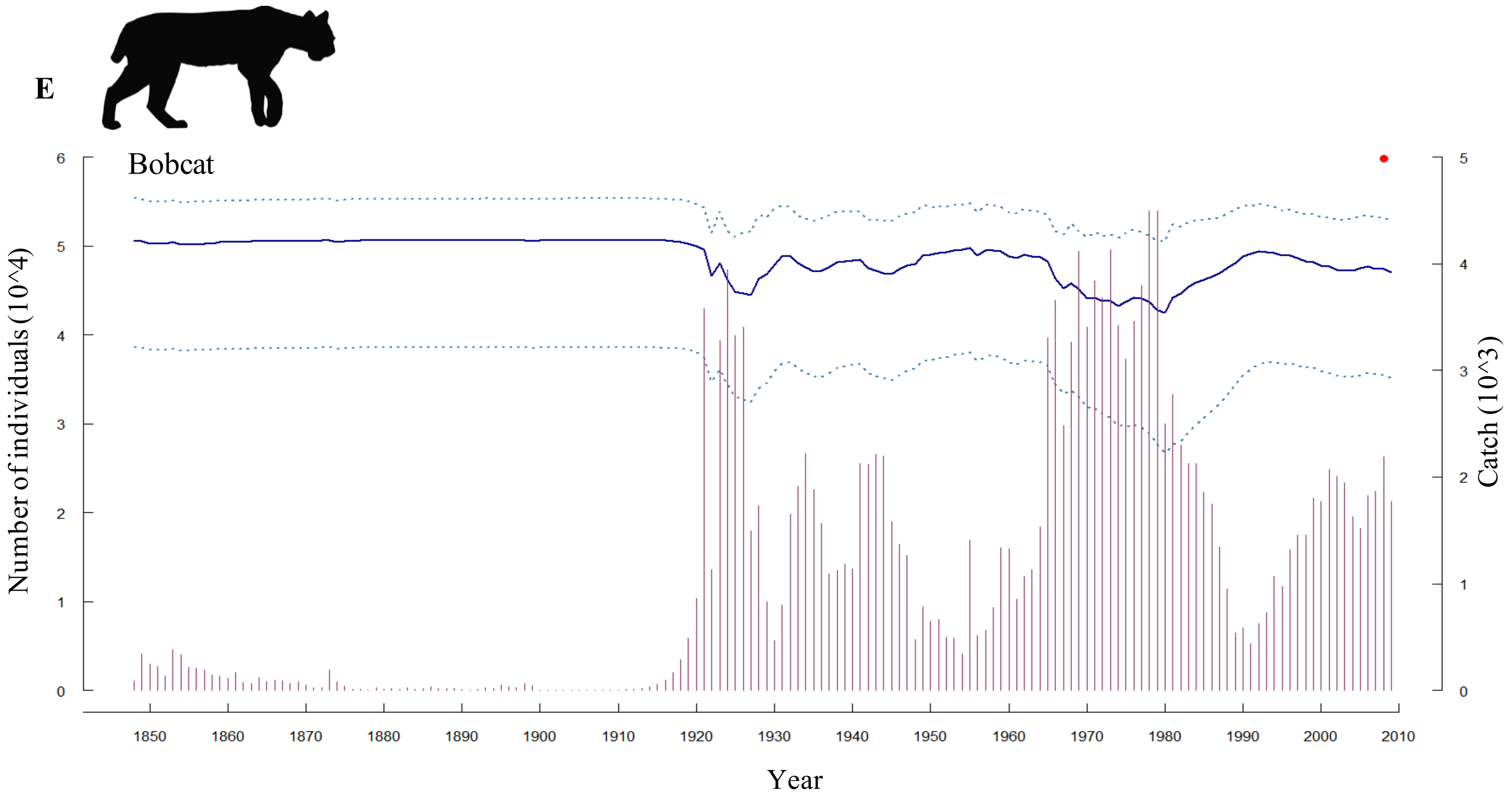
Parameter	Magnitude of altered value	Arctic fox	American Badger	Beaver	Black bear	Bobcat	Fisher	Canadian Lynx	Polar bear
Harvest data	**Double**	0.05	0.07	0.34	0.05	0.27	-0.03	0.68	0.55
	Half	0.04	1.00	0.57	0.25	0.97	-0.05	1.29	0.94
Upper bound carrying capacity	**Double**	-0.01	0.02	-0.03	0.01	-0.06	-0.04	0.00	0.02
	Half	0.02	-0.06	-0.07	0.01	-0.69	-0.07	0.05	0.76
Lower bound carrying capacity	**Zero**	0.01	-0.05	0.03	1.07	0.06	-0.01	-0.01	-0.02
Observation error	**Double**	0.00	-0.02	0.05	-0.03	-0.07	-0.01	-0.02	-0.03
	Half	-0.02	0.08	0.10	-0.01	0.11	0.00	0.02	-0.09
Current population estimate	**Double**	0.00	-0.03	-0.23	-0.52	-0.12	0.01	-0.32	-0.27
	Half	1.37	0.03	-1.17	-0.90	-0.86	-0.08	-2.42	-3.68
Intrinsic growth rate	**Double**	-0.01	-0.49	-0.28	-0.03	-0.24	0.01	-0.51	-0.34
	Half	0.03	-0.06	-4.57	0.00	-0.97	0.18	-3.60	-1.91


A
American Badger
Number of individuals (10^5)
0
1
2
3
4
5
6
Catch (10^3)
0
10
20
30
40
1840
1850
1860
1870
1880
1890
1900
1910
1920
1930
1940
1950
1960
1970
1980
1990
2000
2010
Year

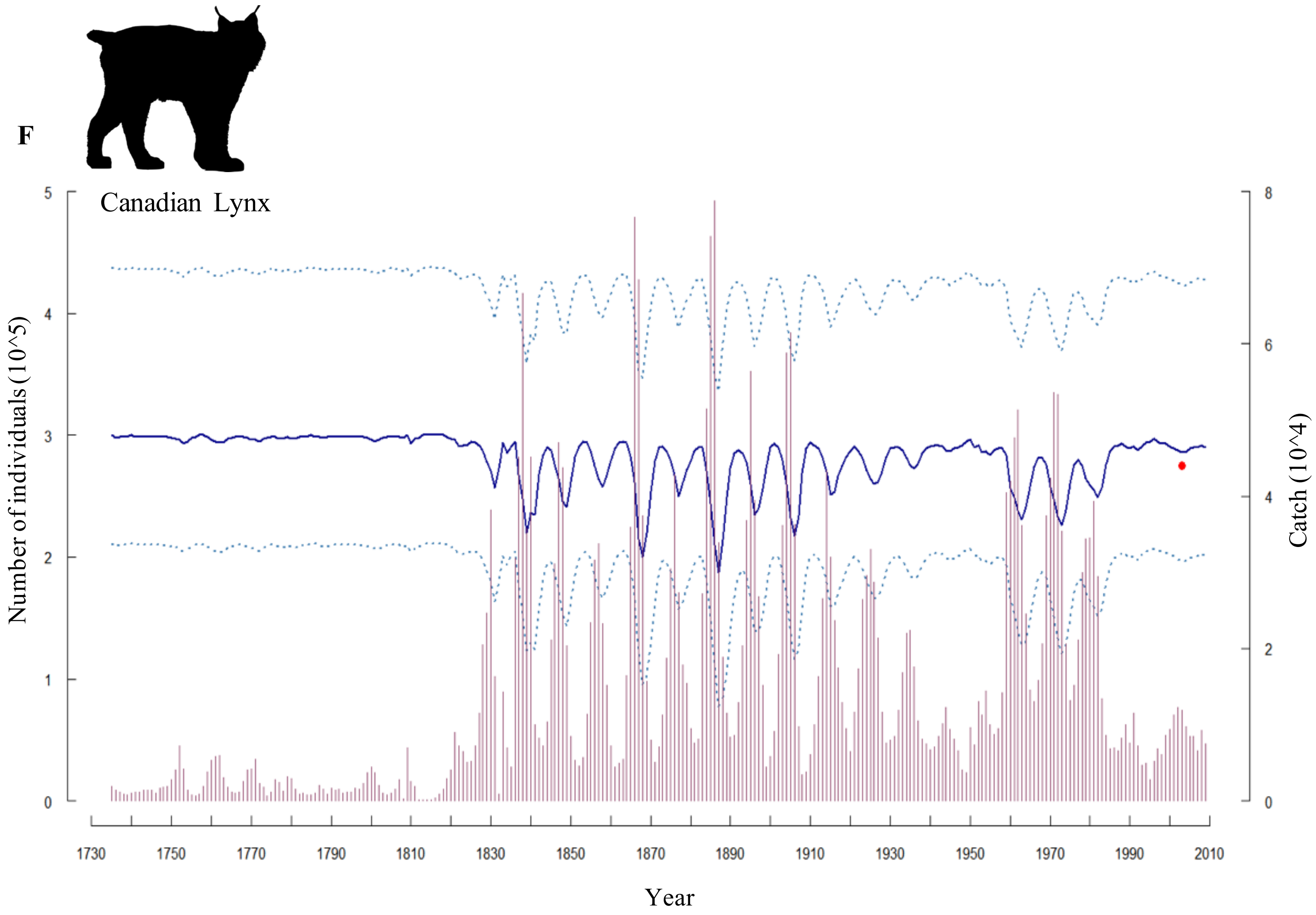

B
Arctic Fox
Number of individuals (10^4)
0
10
20
30
40
Catch (10^4)
0
1
2
3
4
5
6
7
8
9
1750
1760
1770
1780
1790
1800
1810
1820
1830
1840
1850
1860
1870
1880
1890
1900
1910
1920
1930
1940
1950
1960
1970
1980
1990
2000
2010
Year

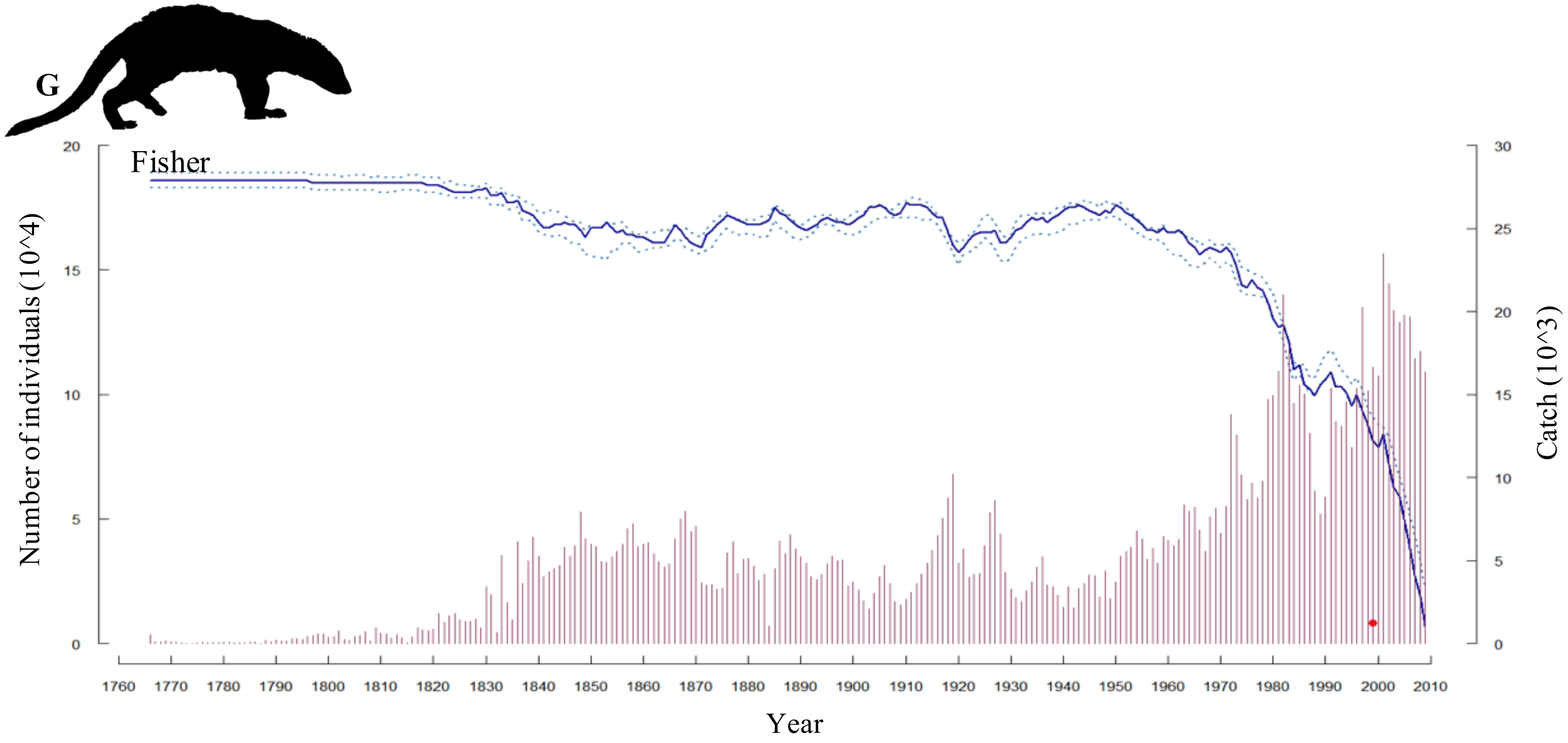

C
Beaver
Number of individuals (10^6)
Catch (10^5)
Year
0
1
2
3
4
5
6
7
1720
1740
1760
1780
1800
1820
1840
1860
1880
1900
1920
1940
1960
1980
2000


D
Black Bear
Number of individuals (10^4)
Catch (10^3)
Year
0
5
10
15
20
1750
1760
1770
1780
1790
1800
1810
1820
1830
1840
1850
1860
1870
1880
1890
1900
1910
1920
1930
1940
1950
1960
1970
1980
1990
2000
2010


E
Bobcat
Number of individuals (10^4)
0
1
2
3
4
5
6
Catch (10^3)
0
1
2
3
4
5
1850
1860
1870
1880
1890
1900
1910
1920
1930
1940
1950
1960
1970
1980
1990
2000
2010
Year

F


Canadian Lynx
Number of individuals (10^5)
0
1
2
3
4
5
Catch (10^4)
0
2
4
6
8
Year
1730
1750
1770
1790
1810
1830
1850
1870
1890
1910
1930
1950
1970
1990
2010


G
Fisher
Number of individuals (10^4)
0
5
10
15
20
Catch (10^3)
0
5
10
15
20
25
30
Year
1760
1770
1780
1790
1800
1810
1820
1830
1840
1850
1860
1870
1880
1890
1900
1910
1920
1930
1940
1950
1960
1970
1980
1990
2000
2010

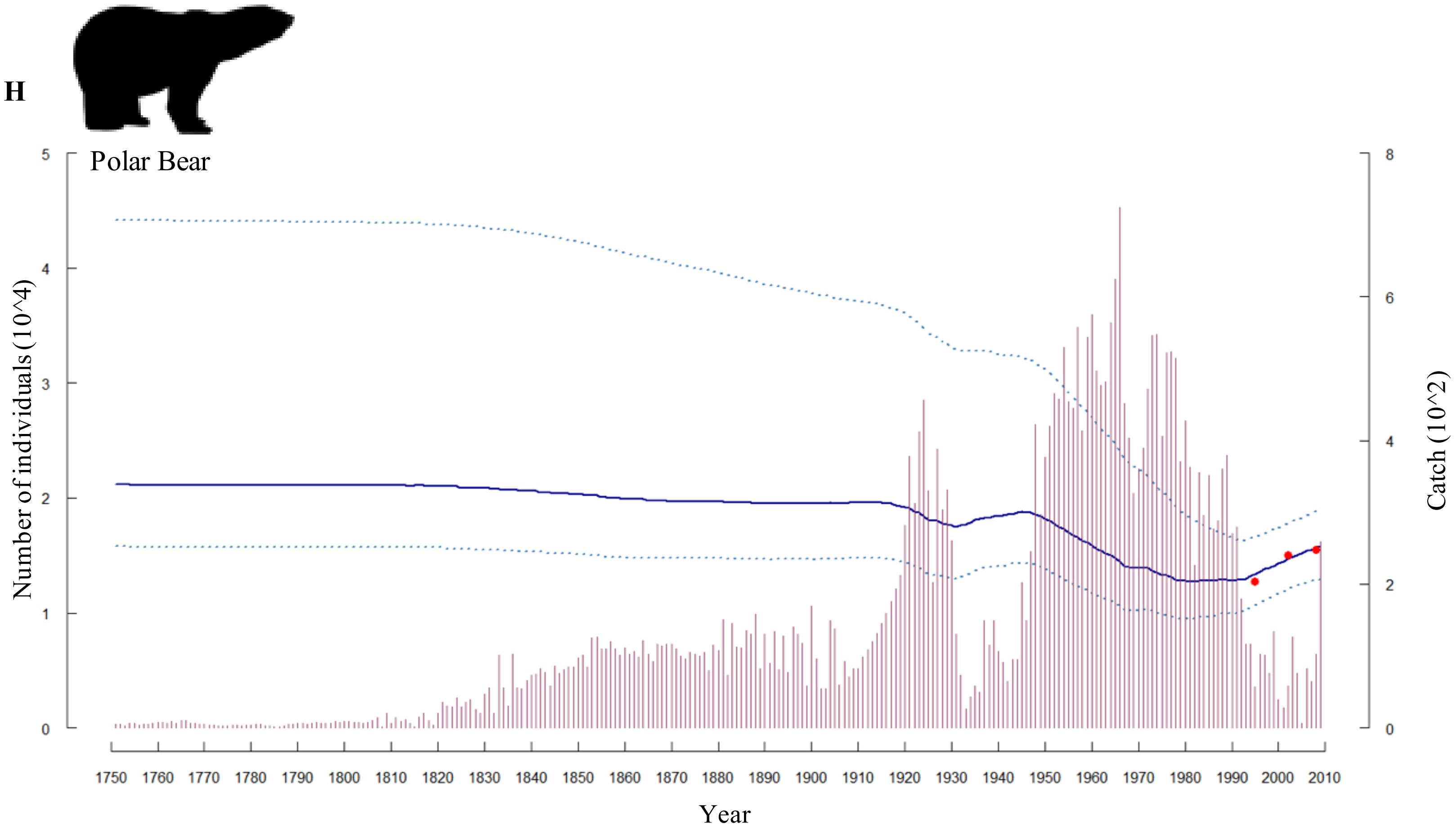


Figure S1. Modelled historical population projections using the pre-1850 baseline for A) American badger, B) arctic fox, C) beaver, D) black bear, E) bobcat, F) Canadian lynx, G) marten, H) polar bear. Pink bars are the annual fur harvest (right-hand axis), the blue solid line and blue dashed lines are the population projection and the 95% median confidence intervals respectively, and the current Canada-wide population estimate is shown by the red dots (left-hand axis).

References

Angerbjörn A, Tannerfeldt M, Erlinge S. 1999 Predator-prey relationships: Arctic foxes and lemmings. *J. Anim. Ecol.* **68**, 34–49. (doi:10.1046/j.1365-2656.1999.00258.x)

Canada S. 2010 Fur Statistics, Canada.

Carlos AM, Lewis FD. 1993 Indians, the Beaver, and the Bay: The Economics of Depletion in the Lands of the Hudson's Bay Company, 1700—1765. *J. Econ. Hist.* **53**, 465–494. (doi:10.1017/S0022050700013450)

Derocher AE, Commission. IU for C of N and NRSS. 1998 Polar bears : proceedings of the Twelfth Working Meeting of the IUCN/SSC Polar Bear Specialist Group.

Elton C, Nicholson M. 1942 The ten-year cycle in numbers of the lynx in Canada. *J. Anim. Ecol.* **11**, 215–244. (doi:10.2307/1358)

Fagan WF, Meir E, Prendergast JR, Folarin A, Karieva P. 2001 Characterizing population vulnerability for 758 species. *Ecol. Lett.* **4**, 132–138.

Fagan WF, Pearson YE, Larsen EA, Lynch HJ, Turner JB, Staver H, Noble AE, Bewick S, Goldberg EE. 2013 Phylogenetic prediction of the maximum per capita rate of population growth. *Proc. R. Soc. B Biol. Sci.* **280**, 20130523–20130523. (doi:10.1098/rspb.2013.0523)

Feldhamer GA, Thompson BC, Chapman JA. 2003 *Wild mammals of North America: biology, management, and conservation*. JHU Press.

Goodrich JM, Buskirk SW. 1998 Spacing and Ecology of North American Badgers (Taxidea taxus) in a Prairie-Dog (Cynomys leucurus) Complex. *J. Mammal.* **79**, 171–179. (doi:10.2307/1382852)

Hennemann III WW. 1983 Relationship among Body Mass , Metabolic Rate and the Intrinsic Rate of Natural Increase in Mammals. *Oecologia* **56**, 104–108.

Hristienko H, Jr, Mcdonald JE. 2007 Going into the 21 St century : a perspective on trends and controversies in the management of the American black bear. *Ursus* **18**, 72–88.

Hutchings AJ, Myers RA, Garcia VB, Lucifora LO, Kuparinen A. 2012 Life-history correlates of extinction risk and recovery potential. *Communications* **22**, 1061–1067.

Jones JW. 1914 Fur Farming in Canada.

Lancia A, Bishir JW. 1985 Rates of Beaver in Newfoundland: A Comment. *J. Wildl. Manage.* **49**, 879–881.

McLean PK, Pelton MR. 1994 Estimates of population density and growth of black bears in the Smoky Mountains. *Int. Conf. Bear Res. Manag.* **9**, 253–261.

Messick, J P, Hornocker, M G. 1981 Ecology of the Badger in Southwestern Idaho. *Wildl. Monogr.* **76**, 3–53.

Naiman RJ, Johnston CA, Kelley JC. 1988 Alteration of North American Streams by Beaver. *Bioscience* **38**, 753–762. (doi:10.2307/1310784)

Novak M. 1987 *wild furbearer management and conservation in north America*. See https://www.google.com/search?q=wild+furbearer+management+and+conservation&oq=wild+furbearer&aqs=chrome.0.0j69i57.2315j0j1&sourceid=chrome&ie=UTF-8.

NERC Centre for Population Biology IC. 2010 The Global Population Dynamics Database Version 2. See http://www.sw.ic.ac.uk/cpb/cpb/gpdd.html (accessed on 1 January 2013).

Pelton M, Coley A. 1999 American black bear conservation action plan. *Bears. Status Surv. Conserv. Action Plan* , 144–156.

Poland H. 1892 *Fur-bearing animals: in nature and in commerce*. Gurney & Jackson.

Poole KG. 2003 A review of the Canada lynx, Lynx canadensis, in Canada. *Can. Field-Naturalist* **117**, 360–376. (doi:10.22621/cfn.v117i3.738)

Powell R a, Buskirk SW, Zielinski WJ. 2003 Fisher and Marten. *Wild Mamm. North Am. Biol. Manag. Conserv.* , 635–649.

Roberts NM, Crimmins SM. 2010 Bobcat Population Status and Management in North America: Evidence of Large-Scale Population Increase. *J. Fish Wildl. Manag.* **1**, 169–174. (doi:10.3996/122009-JFWM-026)

Runge MC. 1999 Design and Analysis of a Population Model for Beaver (Castor canadensis).

Scobie D. 2002 Status of the American Badger (Taxidea taxus) in Alberta.

Seton ET. 1929 *Lives of Game Animals*. Garden City N.Y.: Doubleday, Doran & Co.

Sillero-Zubiri C, Hoffmann M, Macdonald DW. 2004 *Canids: Foxes, Wolves, Jackals and Dogs*.

Sunquist M, Sunquist F. 2002 *Wild Cats of the World*. Chicago and London: University of Chicago Press.

Tanner JT. 1975 The Stability and the Intrinsic Growth Rates of Prey and Predator Populations. *Ecol. Soc. Am.* **56**, 855–867.

Taylor MK, Laake J, McLoughlin PD, Born EW, Cluff HD, Ferguson SH, Rosing-Asvid A, Schweinsburg R, Messier F. 2005 Demography and viability of a hunted population of polar bears. *Arctic* **58**, 203–214.

Taylor M, Lee J. 1995 Distribution and abundance of Canadian polar bear populations: A management perspective. *Arctic* **48**, 147–154

Thompson SD. 1987 Body size, duration of parental care, and the intrinsic rate of natural increase in eutherian and metatherian mammals. *Oecologia* **71**, 201–209. (doi:10.1007/BF00377285)

Weir RD. 2000 Ecology of Fishers in the Sub-boreal Forests of North-central British Columbia Year IV (1999-2000): Radiotelemetry Monitoring And Habitat Sampling.

Williamson DF. 2002 In the Black: Status, management, and trade of the American Black Bear (Ursus americanus) in North America.