

Polar bears are iconic species of the Arctic, representing the fascination for wildlife in the cold northern region shared by people living in the Arctic as well as beyond. Photo: Wild Arctic Pictures/shutterstock.com



Mammals

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» The bears are more hungry. There is a problem with the ice. The rough ice makes it hard for them to find seals, but there is the same number of seals. [...] The only change I've noticed is when I was growing up the polar bears would scare easily and run away. Even when they were around shacks they didn't break windows or do damage but now they are not afraid. They used to avoid communities before and now they don't.

Dowsley 2007.

SUMMARY

There have been substantial changes during the past 50 years in the distribution and abundance of numerous Arctic mammals. The intensity and scope of these changes have been more pronounced in marine than terrestrial mammals. However, the lack of quantitative information for many species means that our assessment is biased towards the larger, more conspicuous and more economically useful species.

One set of changes is driven by a warming climate. Reductions in the duration, extent and quality of sea ice are forcing ice-dependent mammals (notably polar bears *Ursus maritimus*, seals and walrus *Odobenus rosmarus*) to change feeding behavior and areas, change habitats for reproduction and resting, and often travel further, with consequent reductions in population productivity and size. Increased frequency of winter rain and melting temperatures create ice cover on the ground or in the snowpack, making it more difficult for caribou/reindeer *Rangifer tarandus* and muskoxen *Ovibos moschatus* to reach food, and sometimes causing die-offs. Warming temperatures are driving greater growth and spread of primarily shrubs, but also trees, transforming the low Arctic tundra to sub-Arctic conditions with resultant influx of species (notably moose *Alces americanus*, Eurasian elk *Alces alces*, American beaver *Castor canadensis* and snowshoe hare *Lepus americanus*) that can use this new habitat. Later onset of snow in autumn and earlier spring melt shorten the duration and quality of the snow cover that is essential for lemming winter reproduction, and are implicated in reduced amplitude and longer periods in lemming cycles, and therefore reduced availability of lemming prey for numerous predators.

In addition to these patterns, other processes related to a warming climate include: changes in the onset, duration and amount of plant growth, changing distributions of ice-associated marine productivity, increased frequency of boreal and tundra wild fires, changes in the relative abundance of particular plant groups in tundra habitats, changing insect distribution and abundance, changing distributions of parasites and pests, together with more extreme weather events and storms. These are likely to have direct or indirect effects on the distribution, carrying capacity, productivity and ultimately population size of various mammals (notably migratory tundra caribou and voles). However, at present we still lack sufficient information to draw strong inferences about causal mechanisms between these acknowledged climate patterns and mammal distributions and demography.

Ecological changes related to a warming climate are happening so fast and are so pervasive that stabilization and major reductions in emissions of greenhouse gases, at the global scale, are the highest priority conservation action for the Arctic.

A second set of changes is driven by human activities. Harvesting of Arctic mammals has a long history. Com-

mercial interests have driven major declines in some populations of whales and reindeer, but intensive harvest management has demonstrated that many populations can recover, and that various species can sustain well-regulated harvests (e.g. whales, polar bears, seals, reindeer and caribou, Arctic fox *Vulpes lagopus*). Indigenous peoples have strong cultural and economic ties to the harvesting of mammals. These can be sustained with a combination of cultural tradition and better science-based monitoring of population sizes and harvest levels.

Humans have introduced or re-introduced populations of some species in the Arctic, considerably influencing their distributions and ecological roles. North American species such as muskrat *Ondatra zibethicus* and American mink *Neovison vison*, introduced to Eurasia, have spread into the low Arctic. Relocations of muskoxen have been successful in numerous circumpolar sites. We recommend against future introductions of mammals to previously unoccupied ranges, especially islands, because of uncertain and often disruptive ecological impacts.

The Arctic is experiencing more human activity and infrastructure developments at sea and on land in recent decades, as a result of hydrocarbon and mineral exploration and developments, new shipping routes, new roads and increased tourism. These bring risks of direct mortality (e.g. oiling from spills, ship collisions), of displacement from critical habitats (e.g. calving, pupping and feeding areas), of disturbance (e.g. aircraft, road or ship noise interfering with whale feeding or caribou suckling), and of increased human harvests.

The following are high priority actions to mitigate the risks of increasing human activities: (1) an expanded system of protected areas or more intensively managed zones, especially marine, with emphasis on coastlines, polynyas, deltas, the edge of the ice pack, and caribou calving grounds, (2) harmonized, cross-jurisdictional, regulatory and assessment regimes for ocean shipping, aircraft routing, seismic and drilling activities, hydrocarbon and mineral developments and tourism, and (3) a more complete mammal distribution and abundance monitoring program designed to test alternative hypotheses regarding mechanisms driving changes.

Arctic carnivorous mammals, especially marine, have increasing levels of contaminants, notably organochlorines and heavy metals, as a result of increased delivery of these substances to the Arctic food web as airborne pollutants or in runoff from freshwater Arctic drainages. There is little evidence of demographic consequences in wild mammals to date, but a growing need to better understand the origins of pollutants, with internationally coordinated efforts to reduce them at source.

The relative impact of current changes varies by species and biogeographic region. However, most changes have been, and will continue to be, in the low Arctic regions. This is where human activity is more intense, and where the most dramatic terrestrial and marine habitat changes

are taking place. Oceans pose an insurmountable barrier to any northward expansion of smaller-bodied terrestrial species currently confined to Arctic mainland, and these will experience the most significant range restrictions. Likewise many expanding boreal species within continental Eurasia and North America will be stopped by ocean barriers, and will be unable to reach the Arctic islands. This particular isolation of islands, such as the Canadian Arctic Archipelago, Novaya Zemlya and Severnaya Zemlya, to novel colonization by smaller mammals allows these islands to act as partial refuges for their existing mammal fauna in the face of climate-driven changes in distribution.

3.1. INTRODUCTION

Relatively few mammals occur in the Arctic. About 67 species of terrestrial mammals and 35 species of marine mammals occupy this biome, at least seasonally (Appendix 3.1), comprising about 2% of global mammalian diversity. This low percentage reflects the energetic constraints facing homeotherms in this environment, and the fact that large areas were covered in ice through various ice ages, and as recently as 7,000-14,000 years ago (Dyke 2004). As climates warmed in the late Pleistocene and the Holocene (i.e. the last c. 12,000 years), Arctic tundras changed in distribution and composition. Mammals redistributed themselves, evolved to the new conditions, or became extinct probably as a result of a complex combination of climate changes and hunting by humans (Lorenzen *et al.* 2011). The Arctic is now home to species belonging to the following mammalian orders: Rodentia (rodents), Lagomorpha (hares and pikas), Soricomorpha (shrews), Carnivora (dogs, bears, cats, weasels, walrus and seals), Artiodactyla (even-toed ungulates) and Cetacea (porpoises and whales). All of these are characteristic north temperate latitude groups, but representatives of two other such mammalian orders – Erinaceomorpha (hedgehogs) and Chiroptera (bats), both insectivorous – have not colonized Arctic latitudes in the Holocene.

The Arctic biome is generally defined in a terrestrial context, as tundra habitats where trees do not grow (see Section 2 in Meltøfte *et al.*, Introduction for this Assessment's delineations of low and high Arctic). Such a tree-line is imprecise in definition, and the sub-Arctic includes extensive shrub tundra interspersed with trees (northern taiga forest). We include terrestrial species with predominantly boreal, including sub-Arctic, distributions whose habitat affinities and documented distributions include some of the low Arctic. For marine ecosystems there is nothing equivalent to the treeline to allow a convenient ecological definition of 'Arctic'. We discuss in detail those species with a well-documented and consistent occupation of marine areas encompassed by low and high Arctic. We do not discuss species using sub-Arctic marine waters. We also acknowledge the occasional occurrence of other species within low Arctic waters (Appendix 3.2).

The taxonomy of Arctic mammals is fairly well studied, partly because there are relatively few species. However, there are still some uncertainties, especially among the rodents, shrews and hares. Pleistocene isolation in different refugia, and Holocene isolation following sea level rise, may or may not have led to sufficient genetic differentiation to warrant species status (Jarrell & Fredga 1993, Edingsaas *et al.* 2004, Wilson & Reeder 2005, Hope *et al.* 2011). For this assessment we follow the nomenclature in Wilson & Reeder (2005).

The broad distributions of Arctic mammal species are fairly well known, especially for conspicuous and recognizable larger-bodied species, although the amount of fine-scale information on distribution varies by species. Our confidence in the broad distributions of small-bodied species (all terrestrial) is high. These patterns are largely extrapolated from locations of well-documented presence and absence, and consider likely barriers to dispersal (mainly stretches of ocean and major rivers). However, the detailed distributions of these small-bodied species remain poorly documented, because the animals are inconspicuous and have not been surveyed in a widespread and repeated fashion through this very extensive and relatively inaccessible biome. We rely on various standard sources for broad distribution patterns (Wilson & Reeder 2005, Andreev *et al.* 2006, MacDonald & Cook 2009, IUCN 2011), and also on detailed data from species experts.

We present the diversity of Arctic mammals as species richness within various geographic regions (Appendix 3.1). For terrestrial mammals, regional boundaries are primarily water bodies (oceans and large rivers) that coincide with the boundaries of distributions of a number of species, leading to a strong inference that the water bodies played a role in geographic isolation and, sometimes, speciation (e.g. Ehrlich *et al.* 2000, Waltari *et al.* 2004). Occasionally, we also employ jurisdictional boundaries to define regions (e.g. Fennoscandia). For marine mammals, we present species richness within 12 marine regions defined generally by seas or archipelagos with some bathymetric or geographic separations (Appendix 3.2).

The quality of information on abundance varies a great deal among species and regions. Some mammals are central to the well-being of northern peoples as sources of spiritual meaning, food, income from hunting and trapping and as competitors. These relationships can be very old, and deeply embedded in northern cultures. Vyacheslav Shadrin, a Yughagir elder from Kolyma region of Siberia says: "...when there is an earthquake, we say that the mammoth are running. We even have a word for this, holgot" (Mustonen 2009). Some species attract scientific attention because they are key players in the food web or have particular conservation concerns. However, we have very little or no detailed information for numerous other terrestrial and marine species. In addition, there is a relative lack of accessible, published information for species occurring in Russia.

We present current knowledge on distributions, richness and abundance by species or population, depending on the detail available. We organize this information in four broad sections: (1) terrestrial herbivorous mammals, (2) terrestrial insectivorous mammals, (3) terrestrial carnivorous mammals, and (4) marine mammals.

3.2. BIOGEOGRAPHY

3.2.1. Terrestrial mammals

Much of the Arctic biome is relatively young in evolutionary and ecological time, having experienced numerous Pleistocene glaciations (ice ages), the most recent being the Last Glacial Maximum (LGM) or last ice age (Wisconsinan or Late Weichselian period), 12,000–18,000 years ago (Dyke 2004, Hjort *et al.* 2004). Various regions escaped glaciation as refugial tundra, including during the last ice age. Consequently, current patterns of terrestrial mammal distribution are prominently linked to these refugia during the LGM. In addition, these distributions reflect the patterns of colonization from refugia and from regions south of continental ice sheets into newly forming tundra habitats as the ice retreated in the Holocene (Macpherson 1965, Weider & Hobæk 2000, Waltari *et al.* 2004).

During the LGM, the great majority of N Asia and considerable parts of NW North America were ice free (Mangerud *et al.* 2002, Dyke 2004). A large ice sheet covered Fennoscandia, most of the Barents Sea including island complexes of Svalbard, Franz Josef Land and Novaya Zemlya, and portions of the Kara Sea from which it pushed onto land on the Taymyr Peninsula (Svendsen *et al.* 2004, Hjort *et al.* 2004, Mangerud 2004). There appears to have been a small refugium in the Andøya region of the present-day Lofoten Islands, Norway (Møller *et al.* 1992, Mangerud 2004, Parducci *et al.* 2012).

The eastern portion of the unglaciated region, including Asian Chukotka and much of Alaska and Yukon, was joined together as one land mass we now call Beringia. The west edge of Beringia may have been contiguous with the rest of unglaciated Asia and north Europe at the LGM, as far west as the Kanin Peninsula of Russia (Mangerud *et al.* 2002). Only small areas in the north Taymyr Peninsula and Putorana Plateau (east of Yenisey River) were covered in ice (Astakhov 2004, Hjort *et al.* 2004).

In the western hemisphere, there was another refugium, or a series of smaller refugia, along the northwest extremity of the present-day Canadian Arctic islands and continental shelf, comprising the Tuktoyaktuk Peninsula, most of Banks Island and parts of Prince Patrick, Eglinton and Melville Islands (Dyke 2004). In addition, tundra habitats existed south of the vast ice sheets to approximately 45° N (Dyke *et al.* 2002).

Of the 67 terrestrial species with distributions in Arctic regions, 49 (73%) are limited to the low Arctic, 15

(23%) occupy both low and high Arctic, and three (4%) are found only in the high Arctic (Appendix 3.1). These three, however, are somewhat anomalous. One (East European vole *Microtus levis*) was introduced (to Svalbard) from temperate regions. The other two (Wrangel Island collared lemming *Dicrostonyx vinogradovi* and Wrangel Island brown lemming *Lemmus portenkoi*) are restricted to Wrangel Island (Wilson & Reeder 2005), but their status as unique species remains unclear (see discussion in Section 3.3.1.1). Species richness falls dramatically from low to high Arctic, demonstrating that high Arctic regions are inhospitable and/or inaccessible for most mammals.

The distributions of most low Arctic species are predominantly outside the Arctic, in the boreal biome. These ‘boreal’ species can exist seasonally, or year-round, in Arctic regions, because their preferred habitats are grassland, sedge fen, shrub or alpine tundra habitats that spread quite seamlessly into low Arctic tundras.

The true Arctic terrestrial mammals are those whose distributions are almost entirely within the Arctic biome (18 species), and those with present-day, resident Arctic tundra populations that have paleo-historical links to a tundra refugium during the last ice age but also extensive boreal distributions (12 additional species) (Appendix 3.1). Species with distributions restricted almost entirely to the Arctic include the circumpolar Arctic fox *Vulpes lagopus* (found in all 20 possible zones), species that are less wide-ranging but still well dispersed (6–8 zones) such as Arctic hare *Lepus arcticus*, Nearctic collared lemming *Dicrostonyx groenlandicus*, Palearctic collared lemming *D. torquatus*, Siberian brown lemming *L. sibiricus* and muskox *Ovibos moschatus*, and also numerous species with very limited distributions (one or two zones) often on islands (four shrews *Sorex* spp., Alaska hare *Lepus othus*, Alaska marmot *Marmota broweri*, four collared lemmings *Dicrostonyx* spp., Wrangel Island brown lemming and insular vole *Microtus abbreviatus*). The 12 resident Arctic species with both refugial links and boreal affinities are: tundra shrew *Sorex tundrensis*, Arctic ground squirrel *Spermophilus parryii*, Norway lemming *Lemmus lemmus*, Nearctic brown lemming *Lemmus trimucronatus*, tundra vole *Microtus oeconomus*, singing vole *Microtus miurus*, caribou/reindeer *Rangifer tarandus*, gray wolf *Canis lupus*, brown bear *Ursus arctos*, weasel *Mustela nivalis*, stoat *M. erminea* and wolverine *Gulo gulo*. Most of these play prominent ecological roles in Arctic tundra ecosystems, and can be considered true Arctic species even though their distributions are not exclusively Arctic. One other species, the red fox *Vulpes vulpes*, appears to be a more recent Holocene colonizer of Arctic regions (Skrobov 1960, Macpherson 1964).

Arctic regions vary considerably in their composition of low and high Arctic species and in their species richness (Appendix 3.1, Fig. 3.1). Species richness is highest for regions that encompassed large refugia during the last ice age and also maintained land connections to boreal regions in the Holocene. These are Alaska/Yukon (37

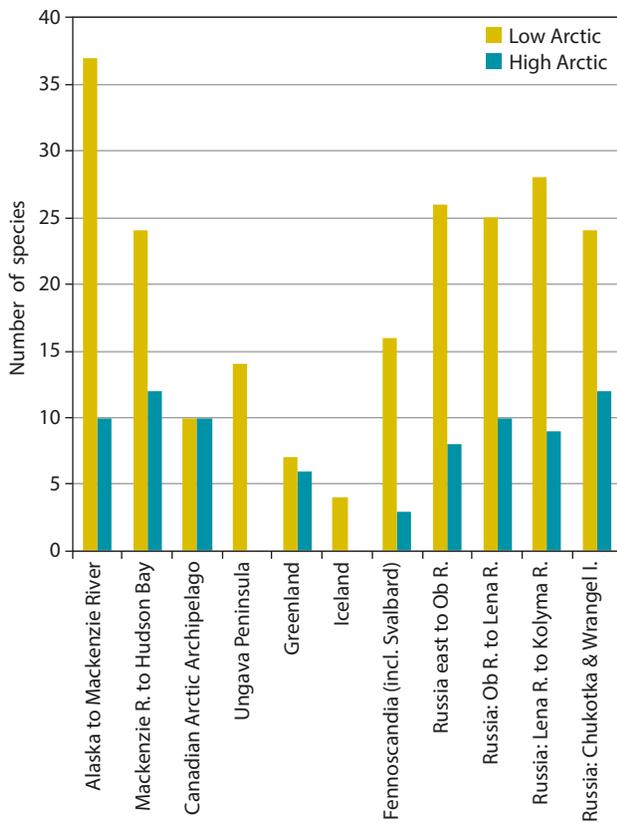


Figure 3.1. Number of terrestrial mammal species occupying low and high Arctic zones in each of the circumpolar Arctic regions. Data are summarized from Appendix 3.1.

species), Ob River to Lena River (26 species), Lena River to Kolyma River (28 species), and Kolyma River to Bering Strait (27 species). This richness is additionally enhanced in regions with extensive mountains (Alaska, Russia east of the Lena River) because of the additional habitat heterogeneity that diverse elevations provide.

Regions with smaller refugia in the last ice age (i.e. Fennoscandia and the Canadian Arctic Archipelago) have much lower species richness (Fig. 3.1; 17 and 10 species, respectively) probably because they sustained substantially fewer species through the last glacial advance. Their current diversity of low Arctic species strongly reflects their relative proximity to boreal habitats.

Regions completely covered in ice during the LGM, but well connected to refugia and to the boreal biome (i.e. Canadian mainland east of the Mackenzie, and western Russia), now have intermediate species richness (Fig. 3.1; 24 and 26 species respectively). This reflects colonization of newly formed habitats by tundra species from refugia and by boreal species from the south.

Greenland was likely completely covered by Pleistocene ice (Dyke 2004, Bennike *et al.* 2008) and remains largely covered to this day. Its sparse terrestrial mammal fauna (seven species) mostly originated from the refugium in the western Canadian Arctic Archipelago (Fedorov & Stenseth 2002, Waltari *et al.* 2004, Bennike *et al.* 2008).

Ungava Peninsula in Canada and Iceland only have low Arctic habitats, but also low species richness (Fig. 3.1; 14 and four species, respectively) because they have been isolated from other tundra regions by large water bodies for most of the Holocene. Some Beringian tundra species, such as Arctic ground squirrel and muskox, have not been able to colonize Ungava without human assistance. The retreating Laurentide ice sheet over Hudson Bay, and massive pro-glacial lakes to its south, collectively formed an ice and water barrier, now largely Hudson Bay, that blocked colonization from the west (Dyke 2004, Occhietti *et al.* 2004). Some true Arctic species (Arctic hare and Arctic fox) apparently colonized from the Canadian Arctic Archipelago, and others arrived from the south (e.g. the Ungava collared lemming *Dicrostonyx hudsonius*) (Macpherson 1965, Dyke 2004). Iceland was likely completely glaciated in the last ice age and also isolated from other Arctic lands by the North Atlantic (Dyke 2004, Geirsdóttir 2004). Only the Arctic fox, the most itinerant of all Arctic species, has colonized Iceland in the Holocene, the other three species being introductions.

Terrestrial mammals that persisted in refugia, especially Beringia, through the LGM and into the Holocene represent a subset of a more diverse Pleistocene fauna, often characterized by species with large body size (such as mammoths *Mammuthus* spp.), 35 genera of which went extinct in the late Pleistocene (Webb & Barnosky 1989, Grayson & Meltzer 2002). A number of grazing herbivores went extinct at the Pleistocene-Holocene transition (13,000-11,000 years BP) at the end of the LGM, coincident with a warming climate as refugia began to expand in size (Guthrie 2001). The most coherent explanation for such a widespread set of extinctions of grazers, including mammoths, horses *Equus* spp. and Beringian bison *Bison* spp. (Shapiro *et al.* 2004, Guthrie 2006), was a major shift in climate patterns (notably cloud and precipitation) driving changes in vegetation from a steppe-like graminoid tundra to a wetter regime supporting woody shrubs and mosses where slower decomposition resulted in peatlands and shrub tundras (Guthrie 2001, 2006). Humans are unlikely to have been the primary cause of these extinctions (Grayson & Meltzer 2002). The dominant low Arctic herbivores we observe today are species that prosper on the dominant plants in relatively waterlogged and peaty habitats. The fates of extant herbivores in a changing climate will likely depend on the particular trajectories that vegetation composition and structure follow in response to regional shifts in temperature, precipitation and solar insolation.

3.2.2. Marine mammals

Arctic marine mammals have changed their distribution with climate variation over time (Vibe 1967, Harington 2008), and a common theme for marine mammals during the Pleistocene was northerly range shifts during warm phases and southerly shifts during cold phases (Harington 2008). For the cetaceans, Dyke *et al.* (1996) used radiocarbon ages of subfossils to demonstrate that

distribution of bowhead whales *Balaena mysticetus* in the Canadian Arctic Archipelago expanded and contracted abruptly several times over the last 10,500 years. Those fluctuations allow for a reconstruction of the post-glacial sea ice history in the area, where bowheads were forced out of habitat due to year-round ice cover or allowed to expand their range into new habitat in the seasonal absence of ice cover. Evidence suggests Basque whalers harvested similar numbers of bowhead whales and right whales *Eubalaena glacialis* in the sixteenth century in the Strait of Belle Isle between Newfoundland and Labrador, a region far south of the present-day range of the bowhead whale, thus indicating a southward shift during the Little Ice Age (Cumbaa 1986, Rastogi *et al.* 2004, Weber *et al.* 2005). Furthermore, evidence that narwhals *Monodon monoceros* once occurred as far south as England during the Little Ice Age – observed in 1588 (Hay & Mansfield 1989) and post-Pliocene fossils in England and Germany (Owen 1846, Collings 1933) – indicate a substantial southerly shift of range with climate.

Polar bears *Ursus maritimus* evolved from brown bears but fossils are unfortunately rare (Harrington 2008). One of the oldest subfossils of a polar bear from the Palearctic is a left mandible found at Prins Karls Forland, Svalbard, and dated to Eemian-Early Weichselian (130,000–110,000 BP) (Ingólfsson & Wiig 2009). Based on a complete mitochondrial genome extracted from that bone, Lindqvist *et al.* (2010) suggested that polar bears evolved from brown bears about 150,000 BP. Analyses of mitochondrial DNA from another find dated to about 115,000 BP, from Kjøpsvik, Nordland, northern Norway revealed about the same age (160,000 BP) for the separation of polar bears from brown bears (Davison *et al.* 2011). A study using nuclear DNA indicated that polar bears evolved much earlier, in the mid-Pleistocene about 600,000 BP (Hailer *et al.* 2012). Edwards *et al.* (2011) suggest that there has been hybridization between polar bears and brown bears through time and that present day polar bears are closely related to earlier Irish brown bears. Miller *et al.* (2012) performed deep, high-throughput sequencing of the genomes of the polar bear mandible from Svalbard, two brown bears from the Alaskan archipelago, a non-archipelago brown bear, and an American black bear *Ursus americanus*. The comparative analyses demonstrated that these bear species evolved largely independently over a period of millions of years, which is in sharp contrast to the more recent estimates of polar bear origin mentioned above. Moreover, 5% to 10% of the nuclear genome of the archipelago brown bears was most closely related to polar bears, indicating ancient admixture between the species. Previously used gene-by-gene sequencing of single nuclear loci lacked sufficient power to detect such ancient admixture. These results are consistent with an ancient split between brown and polar bears approximately 4 to 5 million years BP, coinciding with the Miocene-Pliocene boundary, a period of environmental change that may have launched a radiation of bear species. This initial split was followed by occasional admixture until recently, leaving a clear polar-bear imprint on the nucle-

ar genomes of archipelago brown bears. Genome-based analysis of historical fluctuations in effective population size (i.e. number of interbreeding bear individuals) strongly indicates that polar bear evolution has tracked key climatic events since the Middle Pleistocene. Ten finds of sub-fossil polar bears are known from southern Scandinavia, of which six have been dated to the period between 12,500 BP and 10,500 BP (Aaris-Sørensen & Petersen 1984, Blystad *et al.* 1984, Berglund *et al.* 1992), evidence that strongly suggests that the distribution of polar bears was influenced by climate variation during late Pleistocene and early Holocene and that they had a more southerly distribution than today.

The walrus *Odobenus rosmarus* was a part of the fauna in the North Sea during the late Pleistocene and early Holocene. In the late 1500s they lived (and reproduced) at the Orkney Islands in Scotland (59° N) (Ray 1960). On the Atlantic coast of North America many records of walruses are available from late glacial and post-glacial time periods making it possible to track the northward expansion of walruses as the sea ice retracted (Dyke *et al.* 1999). The northern limit for walruses at the LGM was in the vicinity of present-day Long Island, New York, after which it advanced to the Bay of Fundy by 12,700 BP, to southern Labrador by 11,000 BP, and to the central Canadian Arctic by 9,700 BP. The southern distribution limit also retracted and was in the Bay of Fundy by 7,000 BP. There are very few records of Pacific walrus *O. r. divergens* from late glacial and early post-glacial time. The oldest find, from Vancouver Island, is about 70,000 years old. Another was found in San Francisco harbor and dated to 27,200 BP (Dyke *et al.* 1999).

Similar to contemporary terrestrial mammals, contemporary marine mammals in Arctic regions include a substan-

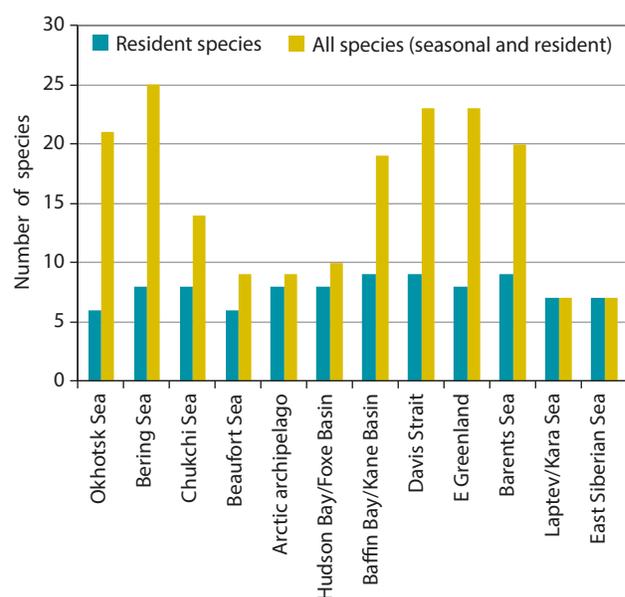


Figure 3.2. Number of marine mammal species in Arctic marine regions classified by resident species (n = 11 total) or all species (including seasonal visitors, n = 35 total).

tial number of low and high Arctic species. When all species of marine mammals that occur in low and high Arctic waters during some time of the year are considered ($n = 35$), species richness (total number of species) is highest in the Pacific low Arctic sectors (Sea of Okhotsk and Bering Sea, with 21 and 25 species, respectively) and in the Atlantic low Arctic (Davis Strait and E Greenland, each with 23 species) (Fig. 3.2). These areas are likely high in species richness because they are open to the large temperate ocean basins of the Pacific and Atlantic, from which many species seasonally migrate. When only resident Arctic marine mammals are considered ($n = 11$), species richness is lower and patterns are less variable. The highest species richness occurs in the Atlantic regions of Baffin Bay, Davis Strait and the Barents Sea ($n = 9$ species in each area). The lowest species richness occurs in the Sea of Okhotsk and the Beaufort Sea.

3.3. TERRESTRIAL HERBIVOROUS MAMMALS

Herbivores comprise the majority of Arctic terrestrial mammal species, and can be divided into three groups based on body size. The small-bodied voles, lemmings and pikas (24 species; 25–250 g) are relatively inconspicuous, but they are often the most numerous mammals in tundra ecosystems, providing food for most carnivores and playing a key role in energy flow through the ecosystem (Krebs *et al.* 2003, Legagneux *et al.* 2012).

Medium-bodied herbivores (nine species; 0.5–35 kg) include the hares and the larger rodents (ground squirrel, marmots, muskrat *Ondatra zibethicus* and American beaver *Castor canadensis*). These are generally found at lower densities than small mammals, but can be locally more abundant depending on habitat patchiness. By diversifying the food supply for carnivores, they can have a stabilizing role in tundra food webs (e.g. Reid *et al.* 1997).

Large-bodied herbivores (six species; 40–600 kg) include caribou/reindeer (one species with English names used interchangeably here), Eurasian elk *Alces alces*, moose *Alces americanus*, sheep *Ovis* spp. and muskox. Caribou and reindeer are essential food species for Arctic peoples (Hummel & Ray 2008), and, along with muskox, are widespread. Domesticated reindeer are mostly discussed by Huntington (Chapter 18).

3.3.1. Species richness and distribution

3.3.1.1. Status

Rodents

Brown lemmings (*Lemmus* spp.) and collared lemmings (*Dicrostonyx* spp.) are the only small rodents with natural distributions in high Arctic regions. They are also found throughout the low Arctic, in conjunction with voles. In the Palearctic, there are four geographically separated species of brown lemmings: the Siberian brown lemming

L. sibiricus, the Norway lemming *L. lemmus*, the Wrangel Island brown lemming, and also portions of the range of the Nearctic brown lemming *L. trimucronatus*. The Wrangel Island brown lemming is recognized by some as a unique species (Wilson & Reeder 2005) though others consider it a genetically distinct clade (Fedorov *et al.* 1999a, 2003). The Palearctic collared lemming is widespread. The Wrangel Island collared lemming *D. vinogradovii* is often considered a distinct species (Wilson & Reeder 2005), but genetic evidence indicates a close relationship with the Nearctic collared lemming (Fedorov & Goropashnaya 1999, Fedorov *et al.* 1999b).

The Nearctic brown lemming is the sole species of this genus in North America. However, the Nearctic is inhabited by four species of collared lemmings: the Nearctic collared lemming, Nelson's collared lemming *Dicrostonyx nelsoni*, Richardson's collared lemming *Dicrostonyx richardsoni* and the Ungava collared lemming.

Various boreal voles occupy portions of the low Arctic. In both old and new worlds we find the tundra vole *Microtus oeconomus* with robust tundra populations (Pitelka & Batzli 1993, Linzey *et al.* 2008), and the northern red-backed vole *Myodes rutilus*, just marginally into the tundra (Pitelka & Batzli 1993). Voles inhabiting only the Palearctic include Middendorff's vole *Microtus middendorffii* (Tsytsulina *et al.* 2008), the narrow-headed vole *M. gregalis* (Batsaikhan *et al.* 2008a) and the gray red-backed vole (grey-sided vole) *Myodes rufocanus* (Sheftel & Henttonen 2008). Some authors treat the North Siberian vole *Microtus hyperboreus* as a distinct species (Andreev *et al.* 2006), but Wilson & Reeder (2005) include it within *M. middendorffii*. In addition, the northern range extent of boreal birch mouse *Sicista betulina*, water vole *Arvicola amphibius* and mountain vole *Alticola lemminus* all slightly overlap southern Palearctic tundra (Andreev *et al.* 2006, Batsaikhan *et al.* 2008b, Meinig *et al.* 2008). There are fewer vole species limited to the Nearctic, and the two recognized species – singing vole and insular vole – may be one species (MacDonald & Cook 2009, Weksler *et al.* 2010). The long-tailed field mouse *Apodemus sylvaticus* was introduced to Iceland centuries ago and is now a habituated species even in the low Arctic (Schlitter *et al.* 2008).

The Arctic ground squirrel is the most widespread large rodent, found in both E Asia and North America (Linzey 2008). The Alaska marmot and the black-capped marmot *Marmota camtschatica* are found in specific mountain ranges (Brooks Range of Alaska, and Orulgan and Kolymskiy Ranges of Siberia, respectively), which are largely boreal but extend somewhat into the low Arctic (Tsytsulina 2008a, Gunderson *et al.* 2009).

The muskrat and American beaver are primarily boreal and native to North America, where their Arctic distributions are marginal and patchy and often associated with deltas of large rivers (e.g. Yukon and Mackenzie Rivers) (Baker & Hill 2003, Erb & Perry 2003). Muskrats have been introduced to various low Arctic regions

of Eurasia (e.g. Yamalo-Nenetsky and Kolyma River) in the 20th century (Korytin *et al.* 1995, Andreev *et al.* 2006), and are now much more widespread (Lissovsky & Begletsov 2004).

Pikas and hares

Three species of pikas have distributions extending into the low Arctic. Two are in Russia – the northern pika *Ochotona hyperborea* and the Turuchan pika *O. turuchanensis*. One is in North America – the collared pika *O. collaris* (Hoffmann & Smith 2005). The Turuchan pika has been considered a subspecies of northern pika (Smith *et al.* 1990, Sokolov *et al.* 1994). However, recent molecular analyses indicate the Turuchan pika is most closely related to the non-Arctic alpine pika *O. alpina*, with controversy remaining as to whether it merits full species status (Formozov *et al.* 2006, Lissovsky *et al.* 2007).

Four species of hare occur in the Arctic: snowshoe hare *Lepus americanus*, Arctic hare, Alaskan hare, and mountain hare *L. timidus*. The first three live in the Nearctic, whereas the mountain hare occurs across the Palearctic. The snowshoe hare is genetically distinct and primarily boreal. The Arctic, Alaskan and mountain hares are closely related; earlier treatments combined all three within one species, the mountain hare (Waltari *et al.* 2004, Hoffmann & Smith 2005, Waltari & Cook 2005). Genetic evidence suggests that the Alaskan and Arctic hares persisted separately in two North American refugia (Beringia and Canadian Arctic islands, respectively) during the last ice age, whereas current populations across the broad range of the mountain hare originated from multiple Eurasian refugia (Waltari & Cook 2005).

Ungulates

Caribou or wild reindeer are classified by their behavior and ecology as ecotypes (Bergerud *et al.* 2008). In the Arctic we find two gregarious ecotypes: migratory tundra and Arctic island. Caribou and reindeer vary in the degree to which they are migratory and gregarious as their abundance changes. Gregarious behavior involves trade-offs between risks of predation and parasite infestation in relation to forage availability, which is the factor ultimately driving reproductive output (Bergerud *et al.* 2008, Hebblewhite & Merrill 2009, Gunn *et al.* 2011). The generally less gregarious populations of the Arctic island ecotype vary in their migratory behavior among years (Hansen *et al.* 2010), the causes of which are not fully understood.

Migratory tundra caribou and reindeer calve, summer and spend the fall on tundra ranges spread through the northern mainland of Eurasia and North America. Winter ranges for most herds extend into the sub-Arctic boreal forests (taiga), but some herds occasionally or usually winter on the tundra. The cows of any one herd migrate from their winter ranges to their calving grounds, which they tend to use repeatedly over many years.

In Russia, intensive reindeer husbandry, especially in western and far-eastern Siberia, has precluded wild rein-

deer from using the same ranges, and large wild reindeer populations are presently concentrated in central Siberia (Syroechkovskiy 2000, Klovov 2004). About 31 wild reindeer herds, of very variable population and range size, occupy Arctic tundra in Russia for at least part of the year, with the larger herds being Taymyr and Lena-Olenyok (Baskin & Miller 2007).

Semi-domesticated reindeer herds compete directly with wild reindeer for range. During the long history and wide geographic extent of semi-domesticated reindeer herding, during which semi-domesticated herds occupied ranges of wild herds, it is possible that some of the original wild herds have disappeared or inter-graded with semi-domesticated herds (Syroechkovskiy 2000, Baskin & Miller 2006).

In Alaska and Yukon, there are four wild caribou herds: Western Arctic, Teshekpuk, Central Arctic and Porcupine. East of the Mackenzie River, the Canadian mainland tundra is home to six large herds (Cape Bathurst, Bluenose West, Bluenose East, Bathurst, Beverly and Qamanirjuaq) that winter in the taiga forest. The Ahik and Dolphin & Union herds, along with several smaller herds on the northeast mainland in Nunavut, spend all seasons on the tundra. The northern islands in Hudson Bay, and also Baffin Island, are occupied by migratory herds of tundra caribou. On one of these, Southampton Island, caribou were extirpated around 1953 and later re-introduced (Heard & Ouellet 1994). The Leaf River and George River herds occupy Ungava Peninsula.

In SW Greenland, the larger Akia-Maniitsoq and Kangerlussuaq-Sisimiut herds live year-round on tundra and undergo relatively short migrations. Five small populations also occur farther north on Greenland's west coast. A population in the Thule district of NW Greenland was apparently extirpated in the late 20th century, but the region has been recolonised by caribou from Ellesmere Island (Roby *et al.* 1984). Wild reindeer disappeared from E Greenland in the late 19th century (Vibe 1967) and have not recolonised the region. In Iceland, one of several introductions of reindeer from Norway in the 1700s has led to a robust wild population (Sigurdarson & Haugerud 2004).

Across the circumpolar high Arctic islands, caribou inhabit a more extreme environment than that faced by migratory tundra herds, and have adapted with proportionally shorter limbs, smaller bodies and paler pelage. Isolation on archipelagos has led to subspeciation, including the Novozemel'sk reindeer *R. t. pearsoni* on Novaya Zemlya archipelago, Svalbard's reindeer *R.t. platyrhynchus*, and Peary caribou *R.t. pearyi* on the Canadian Arctic islands. The crossing of sea ice between seasonal ranges is typical of Arctic island caribou (Miller 2003).

The muskox had a circumpolar distribution in the Pleistocene. Holocene climate changes, including warmer conditions than at present, along with heavy hunting may have contributed to its disappearance in the Pale-

arctic and from Alaska and Yukon. The species currently occurs in most of the Canadian mainland tundra east of the Mackenzie River and west of Hudson Bay, and most of the well-vegetated tundra regions on the Canadian Arctic islands together with N and E Greenland (Gunn & Adamczewski 2003). In modern times, humans have reintroduced muskoxen to Alaska (Nunivak Island, Seward Peninsula, Cape Thompson, Nelson Island and the northeast). They were also successfully introduced to ranges unoccupied in the Holocene, in SW Greenland, Canada (Ungava Peninsula) and Norway, and to ranges in Russia (Taymyr Peninsula and Wrangel Island) that they may have occupied in the Holocene (Gunn & Adamczewski 2003).

The Eurasian elk is found in low Arctic wetlands and shrub-rich habitats from Norway through western Siberia (Henttonen *et al.* 2008). The moose occupies similar habitats in central and eastern Siberia and near the treeline of North America (Geist *et al.* 2008). Although considered here as two species (Wilson & Reeder 2005), differentiation at the species level may be tenuous (Hundertmark *et al.* 2002). Both have occurred in various low Arctic regions since the mid-20th century, using shrub tundra in summer and moving back to forest in winter (e.g. northern Norway (Fjellaksel 2010), Yamal (Korytin *et al.* 1995), eastern Siberia (Andreev *et al.* 2006) and N Yukon (Ruttan 1974)).

A subspecies of thinhorn sheep, Dall's sheep *Ovis dalli dalli*, and snow sheep *Ovis nivicola* are primarily found in northern boreal mountain ranges, but extend into the low Arctic in N Alaska and N Yukon, and in Chukotka and the Putorana Plateau of Siberia, respectively.

3.3.1.2. Trends

Arctic climates have changed sufficiently during the Holocene (last 10,000 years) that some landscapes which are currently tundra could have ranged from glacier to boreal forest cover, causing substantial shifts in species ranges. For example, low genetic diversity in most populations of Palearctic collared lemmings may be attributed to isolation of small populations on remnant tundra landscapes when the boreal forest expanded north during the warm period of the Holocene (Fedorov *et al.* 1999b). Palearctic brown lemming populations, by contrast, have generally high genetic diversity, indicating a relative lack of Holocene geographical separation, likely because their wet meadow habitats persisted through a warm period (Fedorov *et al.* 1999a, 2003, Ehrlich & Stenseth 2001). On Franz Josef Land, a Russian archipelago, wild reindeer no longer occur, but radio-carbon dates from antlers indicate their presence when the climate was warmer (Forman *et al.* 2000).

We have few data regarding historical patterns or changes in mammal species distributions, because there have been few repeated inventories across this vast and relatively inaccessible region (Callaghan *et al.* 2005). For example, there is little information about distributions of

pikas or hares. We know most about changes in species that are hunted or trapped, because these provide food and income and are the target of management actions.

Humans have driven the most dramatic recent changes in distributions by translocating species, sometimes to re-introduce them to previously occupied ranges and sometimes to introduce them in the hopes of economic returns. Muskoxen have spread out far from the numerous sites where they have been released, perhaps in search of new range as populations expanded (Reynolds 1998, Gunn & Adamczewski 2003). Reindeer on Svalbard have been released into areas where overharvest had occurred some 100 years previously (Hansen *et al.* 2010). Another example is the introduction of muskrats, formerly a Nearctic species, to numerous Palearctic locations (Erb & Perry 2003).

Various observers have witnessed changes in distribution, or inferred the changes through a series of observations. Yup'ik hunters and trappers report expansion of moose and American beaver distribution to the west in the shrub-rich habitats of the Yukon River delta in the past decade (Herman-Mercer *et al.* 2011). Eurasian elk and moose have expanded into new drainages and increased their use of upland tundra in various parts of Norway and Russia (Van Eerden 2000, Andreev *et al.* 2006, Lomanova 2007, Fjellaksel 2010). Similarly, there are more frequent sightings since the 1970s of moose in shrub-rich tundra regions north of treeline in the Northwest Territories and Nunavut (e.g. Thelon Game Sanctuary, Kazan River; NWT 2011). Russian researchers report that Siberian brown lemmings have almost disappeared over the past 20 years from the southern edge of their distribution on the southern Yamal Peninsula (Sokolov *et al.* in Reid *et al.* 2011a). Snowshoe hares have become well-established north of the Brooks Range in Alaska, occupying riparian shrub communities along several river drainages, and this expansion has coincided with a contraction of the range of Alaskan hares (D. Klein pers. com.). There may have been a general contraction of the southern boundary of the winter distribution for several caribou herds in the northern boreal forest since the 1800s and early 1900s, both in Canada and Russia (Banfield 1961, Syroechkovskiy 1995).

Animals do not occupy all parts of their general distribution every year. Some quite dramatic appearances and disappearances of species from fairly large Arctic landscapes do not represent a distribution change when viewed over a period of one or even many decades, because the animals often return to apparently abandoned ranges. Some species, such as the colonial Arctic ground squirrels, occupy sites intermittently in a meta-population process involving local extirpation and re-colonization. Caribou are particularly noted for shifting their seasonal ranges for periods of many years, with winter ranges shifting more frequently than calving and summer ranges (Syroechkovskiy 2000, Griffith *et al.* 2002, Schmelzer & Otto 2003, McNeill *et al.* 2005). When herds of migratory tundra caribou are at low abundance

their large winter range tends to contract; as abundance increases, winter range expands (Bergerud *et al.* 2008). In the late 20th century some of the Porcupine caribou herd stayed on portions of the summer range in north Yukon through the subsequent winter (Kofinas *et al.* 2002). Inuit elders on southern Baffin Island report a process of winter range expansion, followed by range drift (expansion on one side and contraction on another), and ultimately a complete change in winter range to a new region, all coupled to long-term population increase in the caribou herd from the 1940s to 1980s, which they believe to be cyclic (Ferguson *et al.* 1998). Dolgan hunters of the Taymyr herd in Siberia report major shifts in the numbers of animals being accessible to hunt from the town of Dudinka (Sillanpää 2008). Such changes, often cyclic, span a period lasting about a human lifetime (Ferguson *et al.* 1998), making the interpretation of change in the relatively short-term context of recent memory and climate warming much more difficult.

Long term monitoring has revealed occasional changes in calving grounds. During 42 years of monitoring the Bathurst herd in Canada, the average annual overlap was 43%, forming two geographically consistent clusters (1966-1984 & 1996-2011) broken by a brief period at peak caribou densities, when the calving ground shifted (Gunn *et al.* 2012). The location of Alaskan calving grounds is relatively predictable although with variation in the degree of annual overlap (Kelleyhouse 2001, Griffith *et al.* 2002). In eastern Canada, the Leaf River calving ground has also shifted as herd abundance has changed (Taillon *et al.* 2012).

The seasonal and annual distributions of Arctic island ecotype caribou also change through time. The use by Peary caribou of some islands expands and contracts with abundance (Miller *et al.* 1977, Gunn & Dragon 2002). Some such changes are long-lived and appear permanent in recent memory, such as the near disappearance of Peary caribou from Prince of Wales and Somerset Islands between 1985 and 1990, even though about 6,000 migrated between the two islands in the 1970s and early 1980s (Gunn *et al.* 2006).

3.3.1.3. Causes and prospects

Considering true Arctic herbivores, the lack of observed range expansion is probably best explained by the fact that these species already occupy most low and high Arctic regions, their expansion is blocked by insurmountable barriers, or their expansion may be limited by competition with closely related species. The Nearctic and Palearctic collared lemmings, Palearctic brown lemming and Arctic hare fit the first category. Oceanic, glacier and lowland habitats prevent the Arctic ground squirrel, the tundra vole, the Alaska marmot, and insular forms of lemmings and voles from any substantial expansion (Kerr & Packer 1998, Gilg *et al.* 2012). Richardson's and Nelson's collared lemmings would have to occupy habitats already occupied by Nearctic collared lemmings in any range expansion.

The remaining true Arctic herbivores could perhaps expand their distributions, and we mention these as hypotheses for future investigation. Alaskan hares could conceivably occupy the North Slope of Alaska and Yukon, a region they previously occupied (Klein 1995, MacDonald & Cook 2009). The Nearctic brown lemming might expand northwards across Lancaster Sound and Viscount Melville Channel to reach the northern Canadian Arctic Archipelago, though such a long distance ice crossing seems unlikely. By crossing substantial glaciers, caribou could recolonize E Greenland. Muskoxen could occupy substantial new ranges in Siberia and Alaska, mainly by expanding from regions of historical introduction. However, most true Arctic herbivores cannot readily expand their distributions, and we know of none that has done so in historical times without human assistance.

Low Arctic species with boreal affinities have greater opportunities for range expansion than the true Arctic herbivores, because low Arctic species are increasingly able to find suitable conditions for their survival as the southern tundra transforms to boreal shrubland and forest. In a time of changing climate and ecosystem conditions, factors that limit distributions are likely changing. Habitat changes are often the most noticeable. The most prominent of these are: an expansion of tree cover into the tundra (Hinzman *et al.* 2005), increases in primary production (Zhang *et al.* 2008), increases in cover of upright and prostrate woody shrubs (Tape *et al.* 2006, Forbes *et al.* 2009, Hudson & Henry 2009, Myers-Smith *et al.* 2011), increases in spatial extent of drier tundra plant communities (Hinzman *et al.* 2005), increases in cover of some graminoids and forbs (Kennedy *et al.* 2001, Walker *et al.* 2006) and decreases in moss and lichen cover (Cornelissen *et al.* 2001, Walker *et al.* 2006). However, these changes vary among sites, depending on local temperature and moisture regimes (Elmendorf *et al.* 2012). Herbivory, with associated nutrient additions, also alters the general patterns substantially (Gough *et al.* 2008, Post & Pedersen 2008, Ravolainen *et al.* 2011, Johnson *et al.* 2011) (see Ims & Ehrlich, Chapter 12 for more detail).

» Willows, in Russian **talnik**, grow much faster now on the banks of Kolyma. As well in the summer pasture areas along the Arctic Ocean tundra willows are more plentiful and more now. On River Suharnaya the willow bushes are much bigger.

(Reindeer herders of the Chukchi community of Nutendli, reported in Mustonen 2009).

Most of the recorded changes in distribution have been in sub-Arctic species apparently responding to these habitat changes, especially the expansion and/or increased height of shrubs. More extensive and taller growth of willows *Salix* spp. increases the spatial extent and carrying capacity of habitats for species that feed heavily on these shrubs (e.g. moose, hares and beaver). By providing increased cover from predators, as a result of increased structure and increased trapping of snow, shrub expansion may also enhance habitat quality for some vole

species. We may see new or continued expansions of the distributions of some boreal species such as the northern red-backed vole, snowshoe hare and perhaps American beaver into the expanding upright shrub communities, and the singing vole into the drying grass tundra. Much will depend on whether the resident Arctic species (such as brown lemmings and Arctic hares) are inferior competitors to the boreal species, and whether other limiting factors such as winter temperature regimes are also relaxed in a warming climate (e.g. the beaver, Jarema *et al.* 2009).

Most montane species such as Dall's sheep, snow sheep, black-capped marmot, Alaska marmot and northern pika currently range nearly to the northern limit of their mountainous habitats, so will not be able to expand appreciably. The collared pika may be an exception. Alpine tundra habitats exist in the Richardson and British Mountains well to the north of its present range limit, but we lack an understanding of what limits its northward colonization.

Distributions of many plants move slowly in response to warming, lagging behind the warmer conditions where they could potentially grow. Increased shrub growth alone will be insufficient to encourage substantive changes in animal distributions unless other necessary food and cover plants (e.g. berry-producing species, fungi, cone-bearing trees) are already present or have also expanded their distributions. For herbivores that depend specifically on certain slow-moving plants, distribution change may also have to lag behind. For example, moose and Eurasian elk mostly return to more sheltered forested valleys in the sub-Arctic for winter, and the extent of their summer movements onto tundra may become limited by the rate at which the treeline moves.

There are some herbivores, notably the graminoid-feeding voles, that inhabit both boreal and tundra biomes. Their distributions seem to be currently limited by the length of the snow-free growing season during which they need to produce sufficient litters for the population to survive the mortality of the subsequent winter (Ims & Fuglei 2005). The snow-free season on the tundra is definitely lengthening (Derksen & Brown 2012), which may increase the opportunities for voles to move north.

Herbivores can strongly influence the structure and composition of plant communities on which they feed (Post & Pedersen 2008, Ravolainen *et al.* 2011), and may have done so on a massive scale in the Pleistocene (Zimov *et al.* 1995). Such effects need more focused research as herbivore distributions continue to change.

The low Arctic zone, however, is narrow in some regions such as N Norway and N Yukon, and could effectively disappear as it transforms to boreal habitats. These are regions where some herbivores may disappear as their habitats change and key foods disappear, examples being the Norway lemming (Tast 1991) and Nearctic collared lemming. These are also regions where the

distributions of some species, such as Arctic ground squirrel and barren-ground shrew, may shrink because they cannot cross ocean channels to reach islands further north (Kerr & Packer 1998, Gilg *et al.* 2012).

» *Regarding the forest fires, some scientists say it's good for new growth. But do you know what the caribou eat? If the lichen burns, it will take over 100 years for the plants to grow back. Some scientists say these forest fires are good, but it's not like that for us. There never used to be so many forest fires.*

(Dene member Pierre Marlowe, quoted in Parlee *et al.* 2005).

The range shifts and contractions, often seasonal, observed in wide-ranging species such as the migratory tundra caribou appear to result from changing food availability which itself is driven by a complex mix of population abundance, wild fires, weather conditions and, increasingly, human activities. At the timescale of decades, changes in abundance appear to play a strong role, especially in the contraction and relocation of winter ranges. Terrestrial lichens are key winter foods, especially for the migratory tundra ecotype. These grow slowly so can be locally overgrazed forcing high-density caribou herds to relocate winter ranges (Kofinas *et al.* 2002, Miller 2003). The locations and extent of boreal forest fires correlate well with shifts in caribou winter ranges (Schmelzer & Otto 2003). Shrinking winter ranges will likely become food limiting for some herds, if fire frequency and average fire size increase as predicted by climate models (Miller 2003, Zinck *et al.* 2011) and as happens when more people occupy the land and access improves (Sillanpää 2008). This food limitation, and associated density dependent effects on fecundity and recruitment, is likely key to understanding the long-term dynamics of range use and population abundance (Messier *et al.* 1988, Ferguson 1996, Miller 2003).

Populations of high Arctic caribou (and probably other herbivores) occupying the more isolated island groups (e.g. Svalbard, Novaya Zemlya), are the ones most at risk of long-term range loss. Ongoing fluctuations in their inherently small population sizes, coupled with virtually no possibility of natural recolonization and no chance of emigration, increase the risk of extirpation. In the Canadian Arctic Archipelago, interlinked with winter ice, disappearance from one island may not represent extirpation, but simply emigration, though perhaps for a prolonged time.

Tracking the location and intensity of use of calving grounds is crucial for caribou conservation given that: (1) there is controversy over their locations over time, (2) barren cows frequently do not visit the calving grounds, and (3) cow-calf ratios on calving grounds have often been used as a measure of recruitment (Ruttan 2012). However, gaps in monitoring leave uncertainties which cloud our understanding (Gunn *et al.* 2011). For example, in central mainland Canada, the Beverly herd's use of its traditional calving ground markedly declined

between 1994 and 2010, perhaps reflecting a decline in herd size (Gunn *et al.* 2011) or an earlier, undocumented, shift to a more coastal calving ground (Nagy *et al.* 2011). We need to better understand how and why caribou shift calving grounds, and it is insightful that timing of snow melt correlates well with such shifts (Griffith *et al.* 2002, McNeill *et al.* 2005).

Human infrastructure and activities, including mineral exploration and development, roads and new settlements, are increasing rapidly on many caribou ranges, and caribou avoid many of these developments (Baskin 2005, Johnson *et al.* 2005, Joly *et al.* 2006). Caribou body condition and herd health need to be monitored to assess ongoing cumulative effects, and calving grounds should be protected from human activity to minimize any risk of reducing calf survival by interfering with suckling behavior (Hummel & Ray 2008).

» *Elders of the Kolymaskaya village, lower Kolyma region, Sakha-Yakutia, Russia, reported in 2006 that willows are moving to tundra and to river banks. They said: "It tells of the changes which are under way. You should graze cows and horses, not reindeer on these spots. All of the tundra is covered with willows and bushes. It grows very fast now. We do not know how we can herd reindeer in the middle of these changes."*

(Mustonen 2007).

Some true Arctic species are likely to lose some of their low Arctic distributions as these tundras change. Low Arctic ranges for reindeer and caribou will contract with the spread of erect shrub tundra. Continental collared lemming distributions may shrink because the dwarf shrub tundras they rely on are at risk of changing to erect shrub tundra or upland graminoid tundra (e.g. Kennedy *et al.* 2001, Myers-Smith *et al.* 2011), and they are poor competitors with at least some other rodents (Ale *et al.* 2011; see also Box 17.5 in Coole, Chapter 17). Where boreal herbivores are expanding their range into low Arctic tundras, they may provide a more abundant and diverse prey base for wide-ranging predators such as red fox and gray wolf. For example, the disappearance of Alaska hares from some regions may be related to the risk of sharing predators with expanding snowshoe hare populations, especially when snowshoe hare abundance drops (Klein 1995).

3.3.2. Population sizes and densities

3.3.2.1. Status

None of the Arctic terrestrial herbivores is classified as globally Threatened (i.e. Endangered or Vulnerable; IUCN 2011), though some are of conservation concern within regional jurisdictions. The two lemming species limited to Wrangel Island are listed as Data Deficient, meaning that we have insufficient information about likely population size and trend to confirm a listing

(IUCN 2001). All other herbivores are listed as Least Concern, meaning they are sufficiently widespread, abundant and stable that current threats do not warrant a Threatened classification. This generally encouraging conservation status of Arctic herbivores reflects the large distributions of most species, often encompassing portions of other biomes, and the relatively low levels of human development and activity in these regions. The latter historical fact is changing quickly, however, catalyzed by climate change.

Arctic herbivore populations often exhibit dramatic population fluctuations through time, independent of human actions. These fluctuations appear cyclic with amplitude of one or two orders of magnitude, and a period of 3-6 years in lemmings, and 40-60 years in caribou (Stenseth & Ims 1993, Gunn 2003, Miller 2003, Bergerud *et al.* 2008).

The variability in period and amplitude of lemming cycles within and among sites indicates that a number of ecological factors influence the pattern. Trophic interactions play a dominant role in driving cyclic dynamics (Ims & Fuglei 2005, Legagneux *et al.* 2012), but the phenomenon requires further investigation (Krebs 2011). In some Nearctic regions (notably the north slope of Alaska's Brooks Range, N Yukon, and parts of the Northwest Territories east of Mackenzie River), lemmings remain at fairly low densities (Batzli & Jung 1980, Pitelka & Batzli 1993, Krebs *et al.* 1995, 2002). In regions where they irrupt cyclically, sympatric lemming and vole species tend to fluctuate synchronously, but not all Arctic regions fluctuate synchronously (Erlinge *et al.* 1999, Krebs *et al.* 2002).

We generally lack abundance estimates for ground squirrels and marmots in Arctic habitats. Similarly, we lack good estimates of population abundance for Arctic pikas. All species are talus-dwelling, and such pika species tend to be long-lived, persist at low densities and have a low reproductive rate (Smith 1988, Smith *et al.* 1990). The northern pika is different in two ways: it may, occasionally, be found at higher density and it may substitute banks of fallen trees or accumulations of driftwood for talus (Smith *et al.* 1990, Sokolov *et al.* 1994).

The population abundance of northern hares is also poorly documented. Hare populations fluctuate widely, the apparent cycles having different periods in different localities (Flux & Angermann 1990, Sokolov *et al.* 1994, Murray 2003). For example, the mountain hare may have

Table 3.1. Summary of historical population estimates for 22 circumpolar caribou and wild reindeer herds. Data courtesy of Circum-Arctic Rangifer Monitoring Assessment Network (CARMA) and D.E. Russell & A. Gunn; www.carmanetwork.com/display/public/home. Data vary substantially among herds and over time in accuracy and precision, and represent only general patterns of abundance.

Year	Western Arctic	Teshekpuk	Central Arctic	Porcupine	Cape Bathurst	Blue-nose West	Blue-nose East	Bathurst	Dolphin Union	Ahiak	Beverly	Oamanirjuaq	Southampton	George River	Leaf River	Aki-Maniitsoq	Kangerlussuaq-Sisimiut	Taymyr	Lena-Olenyok	Yana Indigurka	Sundrun	Chukotka
1974						92,000		251,000			180,000	50,000										3,750
1975														205,000	56,000			449,000	49,500	109,000		21,500
1976	75,000											44,000		263,000								
1977				105,000		42,000		160,000				44,000										
1978	107,000	4,000	5,000			27,000		127,000			130,000		1,181				475,000	52,600	86,600		27,600	
1979				110,000		35,000											470,000					
1980	138,000					65,000		140,000	3,424		110,000			390,000			485,000					
1981			9,000			46,000					164,338						510,000	61,000				
1982	172,000			137,000				174,000						360,000			525,000		113,000			27,000
1983			13,000	135,000		65,000						229,932			101,000		540,000					
1984								384,000			263,691			586,000			575,000					
1985		11,822										272,032					590,000			121,000		
1986	229,000					88,369		472,000		30,000							595,000					32,200
1987				165,000		106,887							4,033									
1988	343,000										189,561	220,999					570,000	73,000				
1989		16,686		178,000																115,900		
1990	416,000							351,683					9,319				625,000	80,900				
1991													13,676		276,000					101,400		
1992			23,000	160,000		112,360	15,544															
1993	450,000	27,686									87,728			780,000			670,000					
1994				152,000							276,000	496,000							77,800			
1995		25,076	18,000																			
1996	463,000							349,046		200,000												34,000
1997			20,000						38,000				30,381									
1998				129,000																		
1999		28,627																				
2000			27,000				119,584										1,000,000					
2001				123,000										385,000	628,000	46,000	52,000		90,000		29,500	
2002		45,166	31,800																			
2003	490,000							186,005					17,981									
2004																						
2005					2,434	20,801	70,081						20,582			36,000	90,500					
2006					1,821	18,050	66,754	128,047														
2007	377,000								27,000				15,452									
2008		64,107	66,772								5,000	345,000							80,000			
2009	348,000				1,934	17,897		31,900									700,000		95,000			
2010				169,000			98,600							74,131		31,000	98,300					70,000

a four-year cycle in Fennoscandia and at least a 10-year cycle in Russia (Flux & Angermann 1990, Prokopjev & Sedalichev 2009), and the Alaska hare a 10-year cycle (Buckley 1954). In the boreal forest of North America, the snowshoe hare cycles with period of 8-11 years (Keith 1981, Murray 2003), but its dynamics in shrub tundra have not been studied. In Sakha-Yakutia, prominent cycles in mountain hares in the mid-20th century have decreased markedly in amplitude, staying at lower densities in recent decades (Prokopjev & Sedalichev 2009). The Arctic hare may move about in large groups of 250-300, while vast areas may have no hares at all (Flux & Angermann 1990). Of these four species, the Alaska hare seems to be the rarest and least likely to reach high densities; it is also the least well-known species.

Caribou abundance is typically assessed at the scale of the herd. Herds are conventionally defined based on the repeated return of cows to the same calving grounds annually. Highs and lows in historical abundance since the 1800s have been reconstructed from the frequency of hoof scars on spruce roots, but only for the Bathurst and George River Herds despite the value of the technique (Morneau & Payette 2000, Zalatan *et al.* 2006). Herd size is often estimated from photographs of calving or summer aggregations. While herd size is fairly often tracked, biologists less frequently monitor rates of birth, recruitment and death, or indices of animal health. Progress needs to be made in relating these indicators to herd size to understand the mechanisms underlying changes in abundance (Boulanger *et al.* 2011).

Caribou herds can vary at least ten-fold through their population cycles. All herds do not fluctuate synchronously, but there can be a strong degree of synchrony among adjacent herds in large regions (e.g. Canadian mainland). The circumpolar caribou population has changed five-fold in historical times with a maximum of about 5.5 million. Currently, the surveyed herds total about 3 million (Tab. 3.1).

Muskox populations can also fluctuate dramatically over time, and appear limited mostly by forage availability as mediated by weather events such as icing and deep hard snow, with predation by gray wolf and brown bear being prominent and increasing in some populations (Reynolds *et al.* 2002, Gunn & Adamczewski 2003, Gunn & Forchhammer 2008, Nagy & Gunn 2009). Most muskoxen reside in Canada (c. 121,000 in 2008). On Greenland there are 9,500-12,500, and re-introduced populations in Alaska total about 3,700 (Gunn & Forchhammer 2008). A general estimate for Russia is 10,000 (Gruzdev 2011).

Sheep populations fluctuate in response to a variety of limiting factors such as winter severity, predation pressure, diseases and parasites. Their ability to access forage in winter is critical, and deep or crusted snow can reduce winter survival and subsequent reproductive output (Krausman & Bowyer 2003).

3.3.2.2. Trends

Rodents

Researchers have monitored Arctic lemming and vole population abundance at a variety of low and high Arctic sites (Tab. 3.2). Variability in amplitude of cycles is likely normal, so trends are inherently difficult to demonstrate. There are no consistent trends across all sites, and many time series are too short to derive clear trends. However, some fairly dramatic changes have occurred, especially during the period of recent Arctic climate warming since the early 1970s. Some prominent cyclic patterns have partly collapsed, with a much reduced amplitude and changed periodicity (Traill Island and Zackenberg, Greenland). A prominent cyclic pattern had declined but has recently recovered (north Norway). Some features of the cyclic pattern have changed: lengthening period between outbreaks (Lena River, Wrangel Island, Banks Island) and a less prominent decline phase (Banks Island). Further details are provided in Box 3.1.

Pikas and hares

We generally lack quantitative data to assess trend in Arctic pika and hare populations. Reductions of mountain hare populations in Sakha-Yakutia, Russia, are attributed to heavy harvesting by humans (Prokopjev & Sedalichev 2009). General observations indicate that snowshoe hares have increased in abundance north of the Brooks Range and in the Yukon River delta of Alaska, and there may have been a coincident decline in Alaska hares in the Yukon River delta (D. Klein pers. com.).

Ungulates

Trends in wild reindeer and caribou numbers must be assessed in the context of natural cycles or fluctuations and the inherent difficulties of counting large numbers of animals over vast areas. In northern Canada, indigenous elders recount stories and recall their own experiences of abundance and scarcity over periods of centuries (Ferguson *et al.* 1998, Legat *et al.* 2002). Methods for estimating population size have only become relatively standardized and rigorous in the past 30 or fewer years (Baskin 2005, Cuyler 2006, Russell & Gunn 2012). Many estimates, especially earlier than the 1980s, may be inaccurate, and gaining sufficient precision remains an issue even with current techniques.

In recent decades, the large majority of migratory tundra caribou herds had been declining at annual rates of 5-17% (Vors & Boyce 2009, Boulanger *et al.* 2011). Between 2000 and 2009, of the 22 migratory tundra herds with fairly substantial monitoring data, 17 herds declined, one was stable and four had increased (Tab. 3.1, some details in Box 3.2). Recent surveys indicate that some herds are now progressing to new phases of a population cycle, somewhat reversing the more general pattern of declines. Considering the herds in Tab. 3.1, 11 are now declining, four are stable, six are increasing and one is not reported by Russell & Gunn (2012).

Table 3.2. Summary of major features of small rodent population dynamics at circumpolar monitoring sites with rodent focus. In addition, reports of relative abundance of small rodents in association with breeding bird studies from approximately the last 15 years can be found at the Arctic Birds Breeding Conditions Survey: www.arcticbirds.net

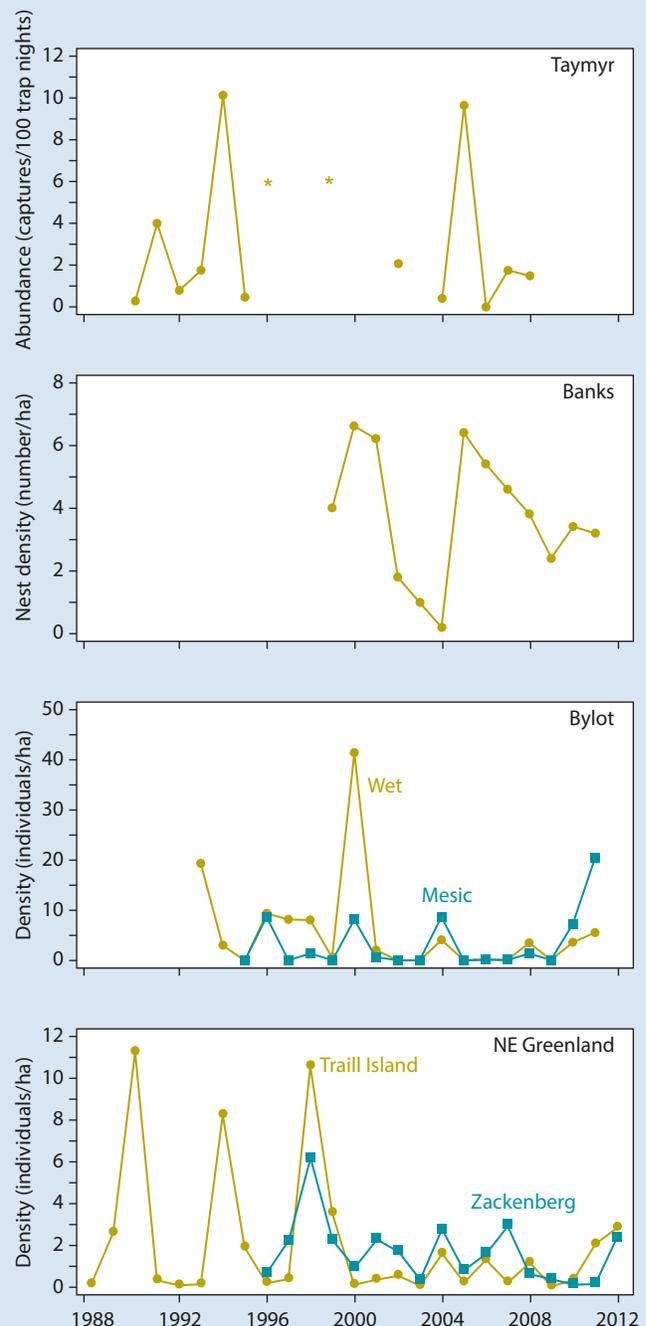
Site	Species	Features of population dynamics	Reference
Northern Norway	Norway lemming	1987-2006: Long period of low abundance with rare outbreak in 1988	Ims <i>et al.</i> 2011, Ims & Yoccoz unpubl.
		2007-2011: Moderate outbreaks every 4 years	
	Tundra & gray red-sided vole	1987-2011: Outbreaks every 5 years with moderate amplitude	
Nenetskaya Gryada, Russia	Tundra vole & Palearctic collared lemming	2004-2010: Outbreaks in 2004 & 2008 (4 yrs) with low amplitude	Ehrich <i>et al.</i> in Reid <i>et al.</i> 2011a
Southern Yamal Peninsula, Russia	Middendorff & narrow-headed voles; Siberian brown & Palearctic collared lemmings	1999 -present: Outbreaks in 1999, 2002, 2005 and 2010 (3 to 5 yr period), and low to moderate amplitude	Sokolov 2002, Sokolov in Reid <i>et al.</i> 2011a
Taymyr Peninsula, Russia	Siberian brown lemming	1960-2001: Outbreaks every 3 to 4 years; lower amplitude in 1990s	Kokorev & Kuksov 2002
		1993-2011: Fluctuations with more variable amplitude since 1990s; outbreaks in 2005 and 2008	Ebbinge & Masurov 2005, Popov 2009
Lena River Delta, Russia	Siberian brown lemming	1951-1967: Outbreaks every 2 to 4 years	Pozdnyakov 2004 and unpubl. data.
		1980-2011: Outbreaks every 3 to 5 years and longer period between recent outbreaks	
Kolyma River lowlands, Russia	Siberian brown lemming & Palearctic collared lemming	1980-1984 & 1991-1996: Synchronous outbreaks every 2 to 4 years	Chernyavsky 2002
Chaunskaya lowland, Russia	Siberian brown lemming & Palearctic collared lemming	1969-1989: Synchronous outbreaks every 2 to 4 years	Chernyavsky 2002
Wrangel Island, Russia	Wrangel Island brown lemming & Wrangel Island collared lemming	1970-2011: Synchronous, low amplitude, outbreaks with period lengthening from 4-5 years in 1970s to 7 to 8 years in 1990s and 2000s	Chernyavsky & Tkachev 1982, Menyushina <i>et al.</i> 2012
Point Barrow, Alaska, USA	Nearctic brown lemming & Nearctic collared lemming	1955-1973: Synchronous outbreaks every 4 to 6 years	Pitelka & Batzli 1993
North slope, Yukon, Canada	Nearctic brown lemming & tundra vole	1989-1998: No outbreaks. Persistent low densities	Krebs <i>et al.</i> 2002, 2011
		2006-2010: No outbreaks. Persistent low densities	
Banks Island, Northwest Territories, Canada	Nearctic brown lemming & Nearctic collared lemming	1993-1996: Outbreaks separated by 3 years, with different peak densities	Larter 1998
		1999-2011: Low amplitude outbreaks every 4 to 5 years. Less pronounced decline phase recently	Parks Canada 2009, Parks Canada unpubl. data
Pearce Point, Northwest Territories, Canada	Nearctic collared lemming & tundra vole	1987-1992: No outbreaks. Persistent low densities	Krebs <i>et al.</i> 1995, Reid <i>et al.</i> 1995
Kent Peninsula region (Hope Bay and Walker Bay), Nunavut, Canada	Nearctic brown lemming, Nearctic collared lemming, Tundra vole & northern red-backed vole	1984-2000: Synchronous outbreaks every 4 to 5 years	Krebs <i>et al.</i> 2002
Devon Island, Nunavut, Canada	Nearctic collared lemming	1967-1973: Outbreaks every 2 or 4 years	Fuller <i>et al.</i> 1975
Bylot Island, Nunavut, Canada	Nearctic brown lemming & Nearctic collared lemming	1994-2011: Variable amplitude outbreaks every 3 to 4 years. Synchrony variable.	Gruyer <i>et al.</i> 2008, G. Gauthier unpubl. data
Traill Island, NE Greenland	Nearctic collared lemming	1988-2000: High amplitude outbreaks every 4 years	Sittler 1995, Gilg <i>et al.</i> 2003, Sittler unpubl. data
		2000-2011: Low amplitude fluctuations, higher every 2 to 3 years	
Zackenbergl, NE Greenland	Nearctic collared lemming	1996-2000: High amplitude outbreak in phase with Traill Island	Schmidt <i>et al.</i> 2008, Schmidt unpubl. data
		2000-2007: Lower amplitude outbreaks every 3 years	
		2007-2011: No outbreaks. Persistent low densities	

Box 3.1. Lemming abundance trends

Lemming abundance is monitored at Arctic sites using density of winter nests, mark-recapture live trapping, or snap trapping. On the Taymyr Peninsula of Russia, Siberian brown lemming cycled with outbreaks every 3-4 years from the 1960s to 1990s (Kokorev & Kuksov 2002), and now appear to have a more variable period (Box 3.1 Fig. 1) (Ebbinge & Mazurov 2005, Popov 2009). Collared lemmings are less numerous but fluctuate in synchrony. On Wrangel Island, NE Russia, the period between years with peak densities has increased from five years in the 1970s to close to eight years in the 1990s and 2000s, perhaps because snow conditions conducive to winter reproduction are being interrupted more frequently with winter thaws and icing of the ground and snowpack (Menyushina *et al.* 2012).

On southern Banks Island, in the western Canadian Arctic Archipelago, outbreaks of Nearctic collared lemmings and Nearctic brown lemmings occurred every 3-4 years in the 1960s and 1990s (Maher 1967, Larter 1998). Further north on the Island, the cyclic period seems to have increased to five years since the late 1990s (Box 3.1 Fig. 1; Parks Canada 2009 and unpubl. data). On Bylot Island, in the eastern Canadian Arctic Archipelago, Nearctic collared lemmings and Nearctic brown lemmings fluctuate fairly synchronously, with much lower amplitude in the collared lemmings. The brown lemmings exhibit outbreaks with highly variable amplitude, every 3-4 years, in two different habitats (Box 3.1 Fig. 1; Gruyer *et al.* 2008, G. Gauthier unpubl.). However, there is no evidence of substantive shifts in the general pattern during the past two decades, and no trend towards poorer quality winter snow conditions (Bilodeau *et al.* 2012).

Only the Nearctic collared lemming is found on Greenland, and its abundance is tracked using winter nest counts at Traill Island (c. 72° N) and Zackenberg (c. 74° N), both in high Arctic NE Greenland. Until 2000, lemming dynamics on Traill Island were characterized by regular cycles of approximately four years (Box 3.1 Fig. 1; Gilg *et al.* 2003). Given the high degree of correlation in abundance between the two localities (Schmidt *et al.* 2008), the dynamics at Zackenberg were most likely similar to those on Traill Island prior to 1996. Around 2000, the population dynamics changed simultaneously at both localities, and regular cycles were replaced by irregular, lower amplitude fluctuations at low densities, especially at Traill Island (Box 3.1 Fig. 1). The observed decrease in amplitude of population fluctuations corresponds well with population dynamics modelled in climate change scenarios with longer snow-free periods (earlier melt and later onset) and more thaw-freeze events in winter (Gilg *et al.* 2009).



Box 3.1 Figure 1. Temporal changes in lemming abundance at various circumpolar sites: A) Taymyr Peninsula, Russia (Siberian brown lemming, stars are years with unquantified high densities; data courtesy of B. Ebbinge & I. Popov); B) Banks Island, Canada (Nearctic collared lemming and Nearctic brown lemming; data courtesy of Parks Canada and L. Nguyen); C) Bylot Island, Canada (Nearctic brown lemming in wet and mesic habitats; data courtesy G. Gauthier); D) NE Greenland (Nearctic collared lemming at Traill Island and Zackenberg; data courtesy of B. Sittler and N.M. Schmidt).

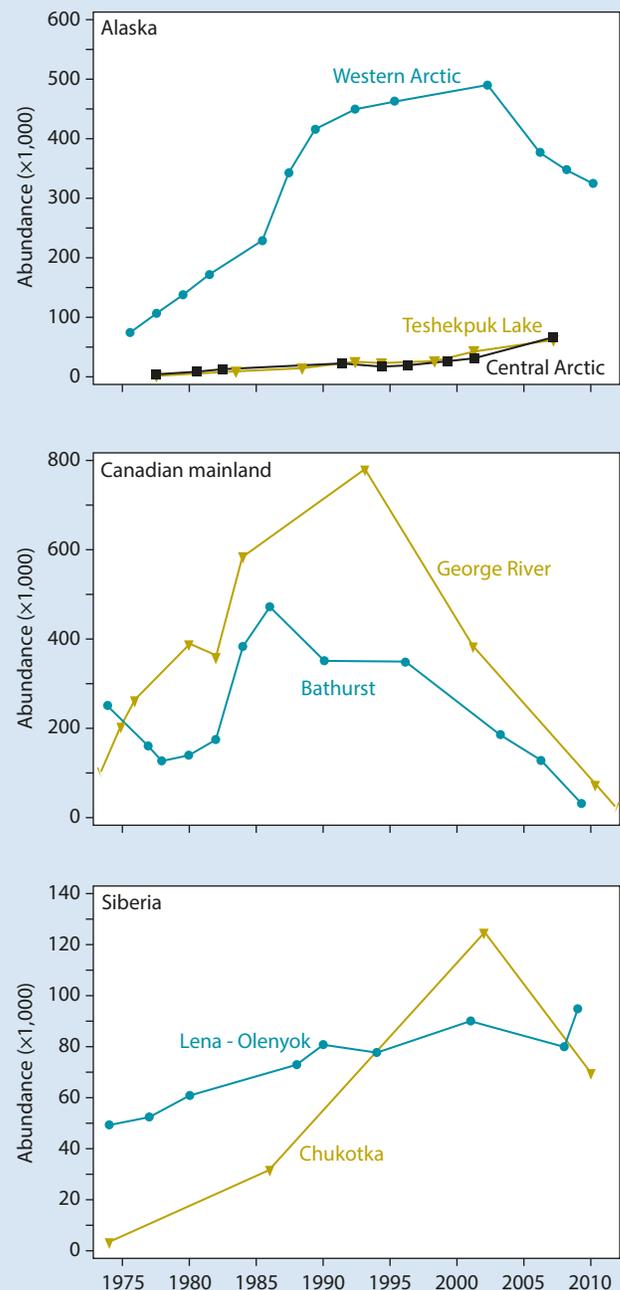
Box 3.2. Migratory tundra caribou trends

Trends in certain migratory tundra herds illustrate key features of caribou dynamics. Herds fluctuate with wide amplitude over periods of many decades, with a tendency towards synchrony in neighboring herds, but lack of global synchrony. Declines can be associated with reduced food availability on limiting ranges (e.g. reduced availability of winter lichen food after forest fires, icing events and competition with domesticated reindeer), high adult female mortality (not necessarily associated with predation or hunting), and reduced parturition rates. Increases can be associated with improved food availability (including reduced competition with domesticated reindeer) and tight controls on hunting of females.

In Alaska, the Western Arctic herd declined from 1970 to 1976, but, similar to the Canadian herds, increased during the 1980s and 1990s, and reached a peak of 490,000 in 2003 (Box 3.2 Fig. 1). The herd then declined somewhat with high adult female mortality (22-30%) and autumn icing in one year. Herd-wide mortality had increased since the 1980s, averaging 17% (Dau 2009). Harvest levels are quite heavily regulated in Alaska, and the decline of the Western Arctic herd is unlikely to have resulted from hunting and predation mortality alone, and is more likely a result of food limitation. In contrast, both the Teshekpuk and Central Arctic herds have shown a protracted increase since the 1970s. During the Teshekpuk Lake increase, recruitment has been in slow decline but adult survival fairly constant (Parrett 2009). The Central Arctic herd is managed to minimize the effects of the Prudhoe Bay oilfield on its calving and post-calving ranges. The herd has low mortality (1997-2007, 10.5%), with human harvest restricted to < 3%, and also high productivity (Lenart 2009).

The migratory tundra herds in mainland Canada, both west and east of Hudson Bay, have experienced dramatic recent fluctuations in fairly close synchrony. The prolonged decline of the Bathurst herd (Box 3.2 Fig. 1) may reflect delays in implementing harvest restrictions because of controversy over the cause of the decline (Wek'èezhii Renewable Resources Board 2010). East of Hudson Bay, the George River (Box 3.2 Fig. 1) and Leaf River herds have also increased and then declined dramatically over a period of about 40 years. Messier *et al.* (1988) provide evidence that the cyclic dynamic of the George River herd is driven by delayed density dependent food limitation at high abundances.

In Siberia, wild reindeer herds increased synchronously from the 1970s to approximately 2000 (Tab. 3.1, Box 3.2 Fig. 1). Population trends for Chukotka wild reindeer were inversely related to domesticated reindeer abundance, suggesting that competition for forage among herds may affect abundance. Wild reindeer were abundant in the 1890s but then declined, with only a few thousand surviving by the 1970s in small areas not used for reindeer herding (Syroechkovskiy 1995, Klovov 2004). The domestic reindeer industry collapsed from 587,000 in 1971 to about 92,000 by 2001 (Klovov 2004). Coincident with that decline, the wild reindeer recovered to 32,200 individuals by 1986 and 120,000-130,000 in 2002 (Box 3.2 Fig. 1).



Box 3.2 Figure 1. Recent time series abundance estimates (figures in thousands of animals) for some migratory tundra caribou and wild reindeer herds (data courtesy of CircumArctic Rangifer Monitoring and Assessment Network (CARMA), and Russell & Gunn 2012).

In Alaska and Yukon, all four migratory tundra caribou herds have been abundant in recent decades. The Porcupine herd has reversed a decline, and the Western Arctic herd is now declining (see Box 3.2). In mainland Canada, caribou numbers were low from the 1950s to the 1970s, when larger herds began to increase to peak sizes in the mid-1980s to late 1990s (Gunn *et al.* 2011). All these herds then declined (e.g. Bathurst herd, Box 3.2), often to historical minimums, though some may now be stable or increasing at low densities. In Ungava Peninsula, the George River and Leaf River herds have experienced dramatic fluctuations (Box 3.2) (Couturier *et al.* 2004).

The status of the 7-10 smaller herds on the northeast mainland, Baffin Island and smaller islands in Hudson Bay is currently unknown, as their abundance is rarely monitored. The exception is Southampton Island, where following the reintroduction in 1967, the herd grew to peak population in the 1990s, and has since declined to about 7,800 (Heard & Ouellet 1994, Russell & Gunn 2012).

W Greenland has a long history of cyclic fluctuations, with high numbers lasting 10-25 years and periods of low numbers of 35-70 years (Meldgaard 1986). Total abundance was about 100,000 in 1970, declining substantially but to uncertain levels by the late 1970s, and recovering to about 140,000 in 2001 (Cuyler 2006, Cuyler *et al.* 2007).

In Russia, wild reindeer have also been through declines and increases, most often in opposite trend to domesticated reindeer herds in the same regions, and sometimes strongly influenced by commercial hunting (Syroechkovskiy 2000, Baskin 2005) (Box 3.2). In 1999, wild reindeer were estimated at 1.3 million compared with the estimated 1.5 million domesticated reindeer, and an estimated carrying capacity for the entire range of about 5 million (Syroechkovskiy 2000). On the Yamal Peninsula, wild reindeer declined during the extensive development of the domesticated reindeer industry in the mid-20th century (Syroechkovskiy 1995), but have recently rebounded (Klokov 2004). The Taymyr herd, one of the largest in the world, increased in the mid-20th century, until commercial hunting held the herd at about 600,000. Following removal of subsidies to commercial hunters in the 1990s, the herd grew rapidly (Kolpashchikov *et al.* 2003), but is now assumed to be declining (Klokov 2004). In Sakha-Yakutia (central Siberia) wild reindeer recovered from the 1950s to 1980s, coincident with a steep decline in domesticated reindeer numbers, but then declined with illegal hunting being partly to blame (Klokov 2004). In Chukotka, wild reindeer numbers have also fluctuated in the opposite trend to numbers of domesticated reindeer (Syroechkovskiy 2000). Numerous small wild reindeer herds remain at low numbers and require particular protection from the risk of overharvest (Syroechkovskiy 2000).

Considering Arctic island caribou, recent trends show Palearctic herds increasing while Nearctic herds mostly

declined. On Svalbard, a decline was reversed when hunting was stopped in 1928. Numbers have since increased, with setbacks when icing restricted access to forage. Novozemel'sk reindeer declined in the early 20th century, but hunting was banned in 1934, and numbers recovered, despite setbacks from icing. Across the larger land mass of the Canadian Arctic islands, Peary caribou have declined dramatically in the last 50 years, largely because of severe winters (Miller & Gunn 2003).

Muskox populations grew in Canada in the late 20th century, concurrent with range expansion, and especially on the southern Canadian Arctic Archipelago, reaching about 121,000 by 2008 (Fournier & Gunn 1998, Gunn & Forchhammer 2008). Re-introduced populations in Alaska have generally grown quite rapidly since the 1970s (Reynolds 1998), then stabilized or fluctuated in the past decade (Alaska Department of Fish and Game 2012). Native populations in NE Greenland have experienced large fluctuations historically, recently rebounding from major declines following severe winters in the period 1940-1960 and in the early 1980s (Forchhammer & Boertmann 2006). They may fluctuate over fairly long time spans (> 5 years), and have recently increased in the Zackenberg area (Forchhammer *et al.* 2002, 2008). Introduced populations in Russia, particularly on the Taymyr Peninsula and Wrangel Island, have grown substantially (Gruzdev & Sipko 2007a, Sipko 2009), and introduced populations in W Greenland generally prospered in the first decades (Boertmann *et al.* 1991).

Some populations of Eurasian elk have grown in size, in conjunction with their increased summer use of low Arctic habitats. In Finnmark county, north Norway, only 15 Eurasian elk were hunted in 1961, but the harvest increased to over 800 by 2007, and the population now appears stable (Fjellaksel 2010). The same general situation appears true in western Russia (Lomanova 2007).

Populations of moose in the Yukon-Kuskokwim deltas of western Alaska have increased in recent years following a reduction in the number of illegal kills and reflecting the fact that moose are below carrying capacity in the region (Alaska Department of Fish and Game 2011, Herman-Mercer *et al.* 2011). Populations on the north slope of the Brooks Range have been gradually increasing from the 1990s to present, perhaps reflecting the relatively favorable winters and improving availability of foods, but here and on the Seward Peninsula moose numbers can be dramatically affected by hard winters and disease (Alaska Department of Fish and Game 2011). Similar increases in abundance have been reported by Inuvialuit harvesters on the north slope of the British Mountains in Yukon, Canada (Aklavik Hunters and Trappers Committee 2003).

Populations of snow sheep are quite disjunct and not well studied. The protected population on the Putorana Plateau increased through the 1980s and 1990s to approximately 5,500 individuals (Sipko & Larin 1999), but the populations in Chukotka are subject to hunting and

may total only 1,500 (Harris & Tsytsulina 2008). Dall's sheep populations in Arctic North America are generally believed to be stable, with adequate management of hunting (Alaska Department of Fish and Game 2012).

3.3.2.3. Causes and prospects

Arctic rodent population cycles, where they occur, seem to have become more variable in period and amplitude in recent decades. However, we lack long-term time series in most regions, so do not know whether changes are part of the general instability of these populations or a definite response to changing climate. Snow quality and quantity likely play a prominent role in population dynamics (Bilodeau *et al.* 2012), and are changing in a warming climate. Snow is tending to accumulate later and melt earlier (Dye 2002, Brown & Robinson 2011), and winter rain and thaws make it less insulative. A lemming outbreak depends, at least in part, on winter and spring reproduction under the snow (Stenseth & Ims 1993), so winter food availability and thermal conditions are crucial. In regions with low total snow fall but consistently cold temperatures (e.g. semi-desert of much of Arctic North America and Siberia), lemmings and voles select habitats with deeper snow (ideally > 60 cm deep), and their populations are more likely to grow in winter when snow comes early and deep in the autumn (Reid & Krebs 1996, Duchesne *et al.* 2011, Reid *et al.* 2011b). Using models, Gilg *et al.* (2009) found that the reduced amplitude of fluctuations in E Greenland could result from decreases in the duration of annual snow cover and increases in the frequency of thaw-refreeze events during winter. These factors are implicated in the lengthening of the cycle period on Wrangel Island (Menyushina *et al.* 2012). In regions with strong maritime influence, snow fall tends to be deeper but frequently influenced by melting temperatures that compact the snow pack and create ice layers within it and on the ground. Under such conditions, small rodents have difficulty creating and maintaining tunnels to access food, and experience reduced survival, reduced winter reproduction and a dampening of population fluctuations (Aars & Ims 2002, Korslund & Steen 2006, Kausrud *et al.* 2008). The recent recovery of high amplitude irruptions in Norway lemmings may result from more persistently cold winters with a snow pack more conducive to breeding (Ims *et al.* 2011).

Small rodent abundance, at least in summer, is heavily influenced by predation (Krebs *et al.* 2003, Legagneux *et al.* 2012), and removal of most of this predation pressure is a necessary condition for population growth (Reid *et al.* 1995, Wilson *et al.* 1999). The strength of an outbreak may increase when multiple small rodent species, with shared predators, are fluctuating synchronously (Ims *et al.* 2011). However, the presence of medium-sized herbivores in the food web allows predators to dampen cycles and keep rodents at low densities (Reid *et al.* 1997). Increasing diversity of herbivore species in the low Arctic food web could have divergent impacts on lemming and vole abundance, depending on which species are involved and how predators are able to respond.

Some Arctic rodent species have low pathogen and parasite loads (e.g. Norway lemming; Laakkonen *et al.* 2001), and parasites have not yet been implicated as a strong factor in their population dynamics. This is an understudied topic, and a warming climate may well influence the life-cycle dynamics, transmission rates and geographic distributions of pathogens that have free-living, intermediate or vector-borne hosts (Harvell *et al.* 2002).

The Arctic ground squirrel experiences temperatures in hibernacula far below freezing, and has evolved a particular mechanism of androgen-driven muscle accumulation in summer to fuel winter energy needs (Boonstra *et al.* 2011). Decreases in duration and depth of insulative snow may put this species at greater risk of lethal freezing in winter.

Populations of collared pika and American pika *O. princeps* living outside the Arctic respond demographically to changes in the quality and quantity of their foods, or in the temperature regime to which they are adapted, and patterns of snow accumulation and melt affect both their foods and sub-nivean temperatures (Li & Smith 2005, Morrison & Hik 2007, Beever *et al.* 2011). Arctic pika populations may respond in similar ways. Increased growth of foods and improved winter insulation with deeper snow could enhance populations. However, late snowmelt and winter icing could have the opposite effect.

The potential direct and indirect effects of a warming climate on hares include improved food quantity with increasing primary production and proliferation of willows, reduced access to winter foods with deeper and harder tundra snow packs, and increased predation pressure with an expanding diversity and abundance of other herbivores including other hare species and ungulates (Klein 1995, Murray 2003). Mech (2000) noted reduced reproduction and a summer decline in Arctic hares on Ellesmere Island, Canada, apparently because of energy deficit following an early snow fall the year before. Mech (2007) also found a strong correlation between gray wolf numbers and an index of Arctic hare density. We need standardized long-term censuses, coupled with hypothesis-driven measures of causal factors, to allow firm inferences about the relative effects of these factors in the future.

Considering caribou and wild reindeer, recent declines and current low numbers in many herds are likely part of long-term natural cycles. The demographic parameters most strongly correlated with abundance trends are adult female and calf survival (Boulanger *et al.* 2011). Survival is a complex outcome of the effects of various causes of death, forage availability and parasite load, perhaps integrated through stress levels. All these factors need to be considered and likely vary in strength at different times in the cycles. A widespread concern has been that the changing climate, with extreme weather events such as deeper or harder snow cover, was driving the synchronous declines in so many herds (Vors & Boyce 2009).

Recent reversals in some of the declines, and the inability to relate all declines to weather patterns, suggest that fluctuations may be part of longer term cycles and their underlying causes (Russell & Gunn 2012).

Population trends can be influenced by human harvest. Indigenous elders emphasize the need to show strong respect for caribou and limit harvest when numbers are low (Legat *et al.* 2002), and a mix of voluntary and mandatory harvest restrictions has been established within communities (e.g. PCMB 2010). Hunter behavior and lags in application of management actions (principally harvest restrictions) likely contribute to population fluctuations (Fryxell *et al.* 2010), especially by accelerating declines or prolonging the low phase. In some Canadian migratory herds, for which abundance is not monitored very frequently, declines were well underway before hunting levels were reduced to take a smaller proportion of the herd (Gunn *et al.* 2011).

In Russia, historical commercial hunting has taken large proportions of some herds and has even caused declines when population levels were quite high (Klovov 2004, Baskin 2005). In Greenland, failure to detect increasing numbers may have contributed to conservative harvest management at a time when the herds were likely increasing (Cuyler *et al.* 2007).

Maximum caribou abundance is likely limited by food availability, with some herds exceeding carrying capacity at high densities (Messier *et al.* 1988, Klein 1999, Miller 2003). In herds with relatively small Arctic tundra range as a proportion of the annual range (e.g. George and Leaf River herds), caribou may more readily exceed the carrying capacity of their tundra calving and summer ranges (Messier *et al.* 1988, Couturier *et al.* 1990). In many other herds, the tundra range is very extensive, and the forested winter range is a relatively small proportion of the annual range. These herds may more readily exceed the carrying capacity of these forested winter ranges, where ground lichen cover is the dominant winter food (Miller 2003). Declines from peak numbers in Russian wild reindeer often appear to result from herds exceeding the carrying capacity of their ranges (Syroechkovskiy 2000, Baskin 2005).

Weather patterns and events affect caribou in diverse ways. Warmer weather conditions in June can significantly enhance the survival of calves by increasing the green-up of vegetation with its associated pulse of nutrients coincident with peak lactation (Griffith *et al.* 2002). Deep snow reduces access to winter foods and reduces survival (Kumpula & Colpaert 2003). Icing of the ground or the snowpack, following winter rain or melting, is strongly correlated with starvation-induced die-offs of Peary caribou (Miller & Barry 2009) and population declines in Svalbard and Wrangel reindeer (Kohler & Aanes 2004, Gruzdev & Sipko 2007b). However, the adverse effects of reduced access to food in winter are more pronounced when caribou are already close to, or exceeding, the carrying capacity of their

range, and herds can often cope with difficult winters when food is still plentiful and therefore likely available in some habitats (Ferguson 1996, Tyler 2010). The negative effects of difficult snow conditions may be partially offset by projected increases in food abundance with warmer summers (Tews *et al.* 2007).

At a regional scale, long-term switches in climate regime, such as the North Atlantic or Arctic Oscillations (NAO or AO), may affect abundance of some herds through changes in productivity or quality of key foods and also weather events (snowfall, icing) affecting access to foods and ultimately survival (Griffith *et al.* 2002, Post & Forchhammer 2002, Forchhammer *et al.* 2005, Joly *et al.* 2011). These patterns may synchronize cycles across wide regions and influence the timing of declines through their influence on carrying capacity.

» Grigory Ivanovich Rynavryntyn was born in the village of Ilimej in the Bilibinsky district into a family of reindeer herders, and talks of the active mining industry in the Bering region of Siberia. "It had done a lot of harm to reindeer husbandry by destroying vast areas of their pasture lands."

(Bat'yanova 2008).

Human activity and infrastructure, most often resulting from exploration and development of mineral and hydrocarbon resources, can destroy tundra habitats if poorly managed, can facilitate heavy hunting and have significantly contributed to declines of some Russia herds (Baskin 2005). However, the demographic consequences are not necessarily detrimental and probably depend on how well the infrastructure is planned and developed to minimize its footprint, the availability of alternative range and the management of mortality factors. The Central Arctic herd in Alaska shifted its calving away from the vicinity of oilfield infrastructure, with a consequent reduction in nutrition for cows and reduced calf growth (Arthur & Del Vecchio 2009), but the herd has continued to increase since the 1970s. Pavlov *et al.* (1996) suggested that the combined effects of gas pipelines, railway roads and river traffic keeping the Yenisey River open may have reduced access for the Taymyr herd to its southwestern winter ranges in the 1970s and early 1980s, but the herd continued to increase while using other winter ranges to the east. Animals in the Bathurst herd avoided an area of 10-15 km around two open-pit diamond mines (Boulanger *et al.* 2012), the development of which coincided with, but cannot be directly implicated in, the herd's decline to a historic minimum in 2006.

The shift in Russian political economy from collective to private ownership of domesticated reindeer in the 1990s resulted in a decline in domesticated reindeer and an expansion of some wild reindeer herds, which gained access to more range (Syroechkovskiy 2000).

Human presence is increasing across most caribou and wild reindeer ranges. In Greenland, hunting of caribou

and muskoxen has increased with more people, stronger boats and a market economy for wild meat (Landa 2002). In Alaska, oil and coal reserves lie under the Western Arctic herd's calving ground: six roads and five mines are being planned (Dau 2009). Human activity on the calving grounds is probably the most risky, because calf survival depends heavily on a focused and strong bond with the mother (Miller 2003).

Trends in climate are now interacting with factors driving long-term caribou population fluctuations, with complex and uncertain outcomes. Warmer winter temperatures and a shorter snow season could reduce energetic costs of foraging and migrating, or increase them if warmer temperatures bring more extreme rain and icing events (Vors & Boyce 2009). Earlier snow melt makes the pulse of nutrient-rich new plant growth earlier, but pregnant cows risk falling behind and missing this generally advancing but crucial period of green-up (Post & Forchhammer 2008). They would have to migrate and give birth earlier to still benefit from this pulse, but it is unknown whether they can adapt by advancing the rut and changing the timing of migration. Although the trend is to earlier spring melt, variability is high, and no single set of behaviors will be adaptive in all situations. Migratory caribou will continue to face late melts and difficulties in traversing snow, or early and fast melts when some streams and rivers may become impassable. Warmer summers might increase levels of harassment by warble *Hypoderma tarandi* and nose-bot flies *Cephemyia trompe*, leading to less time spent feeding, but drier conditions might reduce mosquito populations (Vors & Boyce 2009). The frequency and severity of forest fires are predicted to increase (Zinck *et al.* 2011), potentially reducing the carrying capacity, and therefore peak herd size, of mature forest winter range for migratory tundra herds.

The cumulative effects of development and a warming climate increase the risks. The Dolphin and Union herd crosses 20-50 km of sea ice from summer range on Victoria Island to winter on the mainland (Poole *et al.* 2010). Rising November temperatures have delayed ice formation on average by 10 days from 1982 to 2008, delaying caribou migration and increasing the risk of deaths from falling through weak ice. These risks will be compounded by an increase in commercial shipping (Poole *et al.* 2010).

» *In recent years, all kinds of cruise ships are coming in to our area. Last year alone, there were maybe five or six cruise ships that came into town. More are coming every year. ... But hunters have been complaining about those ships because they go all over Cumberland Sound, even to the campsites. People are saying they are scaring away the animals, the mammals and whales.*

(Community member quoted in Inuit Circumpolar Council 2008).

Although there is little evidence of a strong effect of pathogens and parasites on Arctic ungulate populations,

these are emerging as a higher risk in a warming climate (Hoberg *et al.* 2003). Empirical prevalence and modelled dynamics of a protostrongyloid nematode *Umingmakstrongylus pallikuukensis* in muskoxen reveal broadened seasonal windows for transmission and reduced generation times in the parasite, likely leading to higher infection rates which predispose the hosts to predation (Kutz *et al.* 2001, 2005). A mosquito-borne filarioid nematode, *Setaria tundra*, is associated with die-offs of reindeer and Eurasian elk in Fennoscandia (Laaksonen *et al.* 2010).

Muskox populations are susceptible to starvation when ice encrusts the ground and prevents good access to food (Nagy & Gunn 2009), and deeper snow packs appear to inhibit population growth through starvation mortality or reduced subsequent productivity (Forchhammer *et al.* 2008). Some muskox populations are increasingly affected by predation from brown bears (Reynolds *et al.* 2002) and by extreme weather related accidents such as a storm surge trapping animals in ice (National Park Service 2011). Hunting is an important management tool in Alaska, especially for island populations without wild predators (Alaska Department of Fish and Game 2012).

3.4. TERRESTRIAL INSECTIVOROUS MAMMALS

» *When people lived in cabins made from logs, they saw shrews more often, as the shrews could get in more easily and run around. Unless they see the smaller size and pointy nose, they do not think 'shrew'.*

(Aklavik Hunters and Trappers Committee 2003).

One group of insectivorous mammals, the shrews (Soricidae), has colonized Arctic habitats. These small-bodied mammals (2-12 g) require snow cover as insulation from the winter cold and a steady ingestion of food to fuel their relatively high metabolic rates (Churchfield 1990, 2002). Shrews feed on a wide diversity of Arctic invertebrates and nutrient-rich seeds and also scavenge carcasses (Dokuchaev 1989, Churchfield 1990). In summer, the pulse of invertebrate reproduction and activity probably provides abundant food. In winter, most invertebrates are relatively inactive and hidden in soil or vegetation (Bale *et al.* 1997), so shrew survival decreases and appears strongly affected by food availability (Churchfield 1990, 2002). To deal with this winter shortage of energy and nutrients, individual shrews can increase the thickness of their fur, reduce their body size (Dehnel's phenomenon) and reduce their metabolic rate (Mezhzherin 1964, Merritt 1995, Churchfield 2002). Also, within a species, they are generally smaller at colder northern latitudes than further south, in contrast to Bergmann's rule (Ochocinska & Taylor 2003, Yom-Tov & Yom-Tov 2005).

3.4.1. Species richness and distribution

3.4.1.1. Status

In the Arctic we find representatives from only one genus (*Sorex*) of the large global diversity of shrews. The distributions of 14 *Sorex* species recognized by Wilson & Reeder (2005) overlap the Arctic biome, however, the exact number of species is still in question. Some consider St. Lawrence Island shrew *Sorex jacksoni*, Portenkoi's shrew *Sorex portenkoi* and the barren-ground shrew *Sorex ugyunak* to be conspecific (Dokuchaev 1999, Andreev *et al.* 2006). Many of the Siberian and North American species are closely related to the cinereus shrew *Sorex cinereus* and show little genetic differentiation from each other (Demboski & Cook 2003). Recent genetic evidence suggests that the Alaska tiny shrew *Sorex yukonicus* is the same species as at least the Siberian populations of the Eurasian tiny shrew *Sorex minutissimus* (Hope *et al.* 2010).

No shrew species inhabits the high Arctic, and nine species are primarily boreal in distribution, with small extensions into the low Arctic (Appendix 3.1). The tundra shrew and the tiny shrew species complex are the only shrews to claim a circumpolar distribution (Hope *et al.* 2010). Five shrew species can be considered truly Arctic, being distributed exclusively in the Arctic (four species) or having an extensive Arctic tundra distribution far from treeline (tundra shrew) (Appendix 3.1).

The Siberian and Alaska/Yukon regions have the highest diversity of shrews today, likely reflecting their ability to support some species during the last ice age, their direct connections to extensive boreal regions and isolation of the Pribilof and Saint Lawrence Island shrews with Holocene sea level rise. Shrews are absent today from land masses that were both largely ice-covered and subsequently isolated from mainland refugia by wide ocean passages (Canadian Arctic Archipelago, Greenland, Ungava, Iceland). This is despite the colonization of the Canadian Arctic mainland by the barren-ground shrew, likely from Beringia (Demboski & Cook 2003). These patterns suggest that the refugium on the Canadian Arctic Archipelago did not support shrews, and that shrews have been unable to traverse ocean passages more than a few kilometers wide even when ice covered.

3.4.1.2. Trends

We have no information on changes in shrew distributions in recent history. Genetic analyses of the circumpolar tundra shrew demonstrate population divisions coincident with late Pleistocene refugia and an ability of various lineages of this widespread species to persist through dramatic climate change in the Pleistocene probably because it occupied a variety of habitats (Bannikova *et al.* 2010, Hope *et al.* 2011).

3.4.1.3. Causes and prospects

Given relatively high metabolism and small body size, shrew survival is very likely affected by energy availability in winter, as determined by food and thermal cover of snow (Mezhzherin 1964, Churchfield 2002, Yom-Tov & Yom-Tov 2005). A warming climate may expand the niche for shrews by enhancing invertebrate production which is temperature dependent (Bale *et al.* 1997). The proliferation of erect shrub growth in some regions may increase local snow accumulations (Callaghan *et al.* 2005), expanding the geographic extent of their thermal niche. The relaxation of energetic constraints in a warming climate has a quick effect on body size, which in the cinereus shrew has increased in Alaska over the second half of the twentieth century (Yom-Tov & Yom-Tov 2005). Interference competition among shrew species appears to affect spacing behavior (Dokuchaev 1989) and may influence distribution patterns.

The Arctic Ocean and associated inter-island channels appear to form an absolute barrier to northward expansion of shrew distributions. As mainland habitat conditions change, the truly Arctic species may lose some of their distributions, but we do not know what habitat or competitive factors currently define those distributions, so any thoughts are speculative.

3.4.2. Population sizes and densities

3.4.2.1. Status

The global status ranking is Least Concern for 12 of 14 shrew species (IUCN 2011), meaning their population and distribution characteristics reveal no strong risks at present. Portenkoi's shrew is Data Deficient (Tsytulina 2008b). The Pribilof Island shrew *Sorex pribilofensis* is Endangered, because it is only found on one island (St. Paul) which is relatively small (< 500 km²) with limited known habitat and uncertain population abundance (Woodman *et al.* 2008).

3.4.2.2. Trends

» *Fish is a common bait in traps and shrews eat this bait to the bone. Trappers understood that this activity was related to the abundance of shrews.*

(Aklavik Hunters and Trappers Committee 2003).

Shrews are the least-studied terrestrial mammals in the Arctic, and we lack data on population abundance. Studies of north boreal shrews indicate that abundance follows an annual cycle with winter declines strongly influenced by food availability, and summer increases fuelled by reproduction (Henttonen 1985, Dokuchaev 1989). In boreal Siberia, with abundant snow, shrews follow a four-year cycle, coincident with the cycle in lemming and vole abundance (Sheftel 1989). Such cycles might be

fairly widespread (Henttonen 1985, Dokuchaev 1989), though less evident in regions of poor winter snow cover (Sheffel 1989). Shrews and rodents share the same suite of predators, but shrews are generally less palatable, so may experience heavier predation after the rodents have crashed (Henttonen 1985).

3.4.2.3. Causes and prospects

Arctic shrew populations may benefit from increasing invertebrate productivity and deeper snow packs. They might suffer from increased frequency of icing events and any increases in predation pressure (Aitchison 1987). Prospects are hypothetical, and point to the need for long-term monitoring of population abundance and demographic parameters in key regions.

3.5. TERRESTRIAL CARNIVOROUS MAMMALS

Plant growth and thus herbivore biomass are low in the tundra, therefore terrestrial carnivores are usually rare, highly mobile and mostly solitary. Yet they are present throughout the Arctic tundra, and most of them are active year-round. Their diet is highly diverse, varying from strictly carnivorous to largely vegetarian. They sometimes influence the size and distribution of other vertebrate populations by top-down effects through the Arctic food web (Berteaux 2005, Legagneux *et al.* 2012).

Carnivores can be prey, predators and competitors for humans (Fig. 3.3). People have thus evolved a long, varied and complex relationship with them, ranging from persecution to exploitation to veneration. For most of the 20th century, fur trading (mostly Arctic fox) was a critical economic activity for many Arctic communities, until the fur market largely collapsed in the 1980s (Robinson 2005). Veneration for some of the most charismatic terrestrial carnivore species has developed in many places. Today, many people give the largest carnivore species a high existence value rooted in their power, mystique and beauty (Dickman *et al.* 2011). The following stories show very well how humans and carnivores have built this rich and varied relationship.

» *Early 60s, they were poisoning wolf, at the same time I guess, wolverine and foxes and that go with it as well, and the population went really, really down to about nothing.*

(Participant #401 from Arviat, Nunavut; Cardinal 2004).

» *We know that the bear is a sacred animal. That is why we do not shoot the bear; we do not eat its meat.*

(Alexei Gavrilovich Tretyakov, a retired reindeer herder from Andreyushkino in the Lower Kolyma, Russia; Mustonen 2009).

» *After the war there were many wolves here, none really counted them though. But there were several packs plus some couples to add up with few lonely ones too. We had to herd the reindeer constantly because of these predators. If a pack of say, ten wolves would come hunting, with one single attack they could take ten reindeer. Another attack or another pack, and it would be another ten reindeer!. Since then few wolves have appeared in the region.*

(Late Saami Elder Niillas Vuolab, a reindeer herder from the Kaldoaivi region of Sápmi, Finland; Helander *et al.* 2004).

» *I was tracking a wolverine one time, and all of a sudden it turned towards the wind and it went for about a mile, and it dug into the snow and it retrieved a whole, you know weathered bone – caribou. Completely white, and yet a mile away.*

(Participant #401, Arviat, Nunavut; Cardinal 2004).

» *There used to be less wolverines. Now there is a massive number of them. They have increased in numbers and should be harvested. They kill a lot of reindeer. No difference to them, old and young alike are killed. Wolves tear and attack the reindeer as well. I think they are increasing as well. Before, when the price of the gasoline was lower, we used to shoot them from helicopters but no longer. Then we killed wolves from ski-doo's. That no longer happens either. I cannot say exactly how many, but the numbers were great. Mostly reindeer are killed by humans though. A human kills everything in front of him.*

(Saami reindeer herder Philippov from the community of Lovozero, Kola Peninsula, Russia; Snowchange Luujäü'rr (Lovozero) Oral History Archive 2002-06).

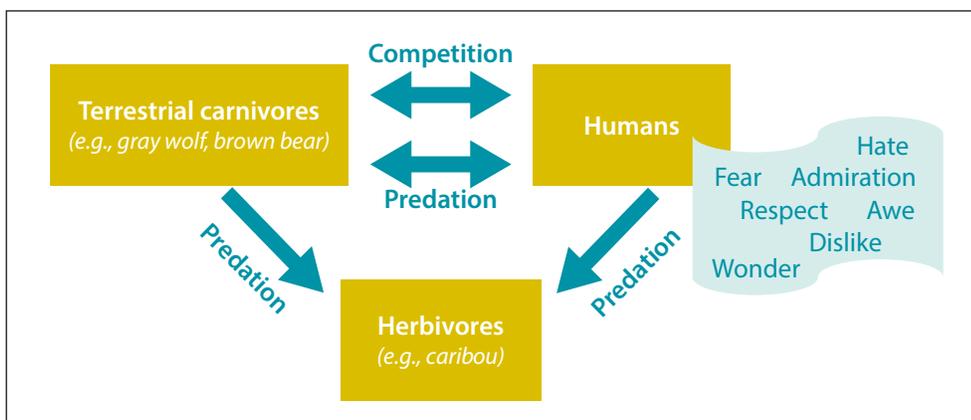


Figure 3.3. Main ecological relationships linking humans, terrestrial carnivorous mammals and their shared prey, as well as some of the emotions felt by humans towards carnivores.

The fates of species such as the gray wolf, Arctic fox, brown bear and wolverine now get worldwide attention because these charismatic animals symbolize the last remaining wildernesses of the world. In addition, past and current exploitation for their valuable fur and competition with humans for some herbivores have generated strong interest in learning about these carnivores (Gagnon & Berteaux 2009). However, assessing the status and trends of carnivore populations is often difficult because of their secretive nature, large home ranges and the vast expanses of land that need to be covered during surveys. This results in numerous data gaps even for basic population sizes, densities and distributions.

3.5.1. Species richness and distribution

3.5.1.1. Status

There are 13 species of terrestrial carnivorous mammals in the Arctic (Appendix 3.1). This represents about 10% of the 128 species (Wilson & Mittermeier 2009) belonging to the order Carnivora worldwide. Among the Arctic species are four species of canids (red fox, Arctic fox, gray wolf and coyote *Canis latrans*). All have a circumpolar distribution and inhabit the high Arctic, except for the coyote which only occurs in the western North American low Arctic where it is very sporadic (Slough & Jung 2007). The two felids (Eurasian and Canadian lynx, *Lynx lynx* and *L. canadensis*) and two bears (brown or grizzly bear, and black bear) are mostly confined to the low Arctic. Lynx actually rarely go north of the treeline. The five mustelids (weasels and relatives) have various distributions, ranging from the circumpolar distributions of the wolverine, stoat (or short-tailed weasel or ermine) and weasel (or least weasel, *Mustela nivalis*) to the more restricted distributions of the American mink and North American river otter *Lontra canadensis*, which just reach the southern margin of the low Arctic.

Of the 13 species of terrestrial carnivorous mammals occurring in the Arctic, six occur in the high Arctic, but none is confined to the high Arctic. Overall diversity is highest in low Arctic areas such as the Alaska to Mackenzie River region, with 11 of the 13 species present (Fig. 3.4). In sharp contrast, the Arctic fox is the only terrestrial carnivore on some islands such as Svalbard. Most other Arctic regions have between four and nine species of terrestrial carnivores. Species of terrestrial carnivorous mammals that occur in the Arctic all have a distribution that is restricted to the northern hemisphere, except for the red fox which was introduced to Australia. No species is endemic to the Arctic, and the Arctic fox is the only species that can be considered an Arctic specialist, the few populations living south of the Arctic being restricted to tundra habitats. The Arctic fox may be the only terrestrial mammal to have been observed on the sea ice up to the North Pole (Angerbjörn *et al.* 2008a).

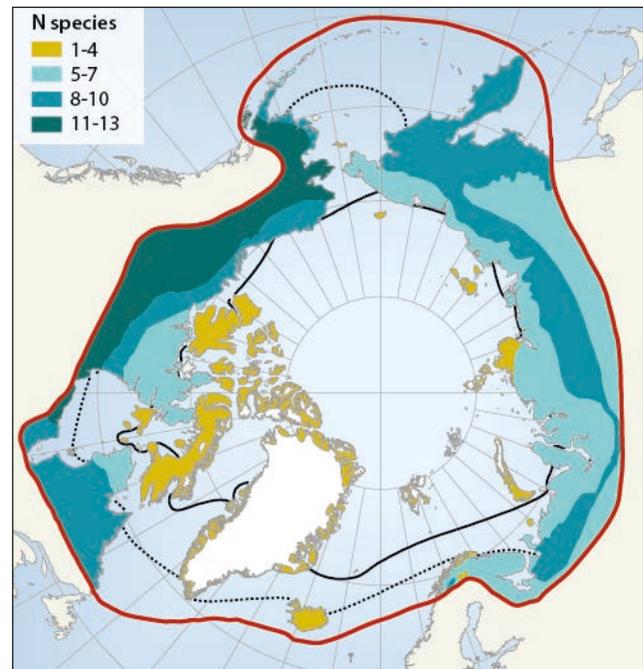


Figure 3.4. Diversity of terrestrial carnivorous mammals across the circumpolar Arctic and sub-Arctic (based on IUCN distribution maps www.iucnredlist.org/initiatives/mammals).

The species richness of terrestrial Arctic carnivores is relatively similar in the Palearctic and Nearctic, with about 10 species in each (Appendix 3.1, Fig. 3.4). However, there are twice as many species in the low Arctic as in the high Arctic, probably reflecting the higher diversity of prey and the higher productivity found at lower latitudes (Krebs *et al.* 2003).

The taxonomy of this group is now well established at the genus and species levels, although the status of the Arctic fox genus is still debated. Some place the species in the genus *Vulpes* (Wilson & Reeder 2005) and others in the genus *Alopex* (Wilson & Mittermeier 2009). There is more taxonomic debate at the level of the subspecies with concerns that the currently recognized subspecies do not always match the genetic data (Wilson & Mittermeier 2009). Wilson & Mittermeier (2009) recognized 106 subspecies in the 13 species of terrestrial carnivorous mammals found in the Arctic, but most subspecies do not belong to the region. The diversity found at the sub-species level is highly variable (the wolverine has only two subspecies whereas the stoat has 34) and may correlate with the dispersal abilities of the species.

3.5.1.2. Trends

» *I haven't seen the Arctic fox in a long time.*

(Much-respected Saami hunter Heikki Länsman spoke of the last observation of an Arctic fox in the Kaldoaivi region (Finland) and said that it was some 10 to 15 years ago; Helander *et al.* 2004).

» They [wolverine] seem to be moving north, northward a little bit. Starting to see them on Victoria Island, compared to the past there were not as much down there.

(A. Niptanatiak, Kugluktuk NU; Cardinal 2004).

There are many documented examples of changes in distribution of terrestrial mammalian carnivores, but trends vary widely among species, populations and regions, ranging from clear expansion to local extirpation. Generalizations are thus difficult. Lack of precise data at the scale of most populations also complicates a global analysis. In addition, carnivores often compete with each other for the same resources so that negative trends in one species often result in positive trends in another. For example, gray wolf and coyote densities have been found to be negatively correlated (Berger & Gese 2007), whereas red foxes may exclude Arctic foxes from newly colonized areas (Tannerfeldt *et al.* 2002). The following examples (summarized in Tab. 3.3 or described at length in the text below) illustrate some of the best documented changes in distributions of species or populations. However this is by no means an exhaustive list of observed changes throughout the circumpolar Arctic.

In 1966, the gray wolf was regarded as functionally extinct in Norway and Sweden (Wabakken *et al.* 2001). In 1978 the first confirmed reproduction on the peninsula in 14 years was recorded, and a small population now resides in Sweden and Norway (Vilà *et al.* 2003), although this is in the boreal region. Similarly, the wolf population in Finland was almost extirpated before the end of the 19th century (Aspi *et al.* 2006). Since then, the wolf population in Finland has increased and expanded its range as a result of conservation strategies and hunting control (Kojola *et al.* 2006), but also remains in the boreal. There are thus still no wolves in the Arctic or sub-Arctic areas of Fennoscandia. After being exterminated from E Greenland in the 1930s, the gray wolf has

recolonized and established a new population in this area during the last thirty years (Marquard-Petersen 2011). Interestingly, whereas humans had exterminated the local population, they may also have unintentionally favored their come-back as lone wolf immigrants may have followed military sled patrols from northern Greenland (Marquard-Petersen 2011).

The historical distribution of the coyote was restricted to the plains and deserts of central North America (Gier 1975, Bekoff & Wells 1986). However, its range has expanded to include Alaska and northwest Canada, coinciding with the removal of wolves. Some may now be found on the northern shores of Alaska and Yukon (Gese *et al.* 2008), although the evidence is controversial.

The Arctic fox has expanded its distribution in some parts of its range while retreating in other parts. The species was introduced to isolated islands in the Aleutian chain at the end of the 19th century by the fur industry (Bailey 1992), while the southern edge of the species' range may have moved northward during the 20th century resulting in a smaller total range (Hersteinsson & Macdonald 1992) both in North America and Eurasia. In parallel, the northern range of the red fox has shifted northward to include the tundra zones of Fennoscandia and Russia (Skrobov 1960 and Chirkova 1968 in Hersteinsson & Macdonald 1992, Killengreen *et al.* 2007, Rodnikova *et al.* 2011). Similarly, the distribution of the red fox has spread northwards into Canada's tundra during the last century (Macpherson 1964), and Pamperin *et al.* (2006) indicate that there is evidence for a similar range expansion of red foxes in Alaska. The timing of range expansion in the Canadian red fox population can be detected by a review of harvest and trading figures, and Macpherson (1964) reported the presence of red foxes on Baffin Island starting around 1918/1919 and north Baffin Island around 1947. Some residents of Pond Inlet, Nunavut, recall their first sightings of red

Table 3.3. Examples of historical and recent distribution changes observed in terrestrial carnivorous mammals in the Arctic.

Species	Observed change	Dates	Reference
Coyote	Northward range expansion in the Northwest Territories and Labrador, Canada	1990s	Chubbs & Phillips 2002, Cluff 2006
Arctic fox	Introduced to the Aleutian chain by the fur industry	Late 19 th century	Bailey 1992
Arctic fox	Disappearance from Finland	Late 20 th century	Henttonen <i>et al.</i> 2007
Red fox	Northward spread into eastern Canadian Arctic, Siberia and Fennoscandia	1920-1960	MacPherson 1964, Hersteinsson and Macdonald 1992
Brown bear	Northern range expansion onto King William Island	Late 20 th century	Keith & Arviq 2006
Brown bear	Northward expansion to the eastern bank of the Kolyma Delta, Russia	Late 20 th century	Mustonen 2009
Wolverine	Extirpation from the Cape Henrietta Maria region, James Bay, Canada	1970s	Dawson 2000
American mink	Introduced to Iceland in 1931 for fur-farming purposes; now present throughout most of the country	First half of 20 th century	Hersteinsson 1992

fox in 1947-1948 or during the 1950s near Pond Inlet (Gagnon & Berteaux 2009). A static relationship between Arctic fox and red fox in the western Canadian Arctic during the last four decades suggests that the red fox expansion may have met its limit in some places (Gallant *et al.* 2012).

The brown bear occupies most of Alaska (Miller *et al.* 1997) and mainland Nunavut (Ross 2002). Traditional ecological knowledge suggests that its range in Nunavut is expanding eastwards (McLoughlin & Messier 2001). Hunters and residents of Gjoa Haven (Nunavut) confirmed the northern range expansion of bears onto King William Island (Keith & Arqviq 2006). Chestin (1997) found no evidence of any changes in the historical northern distribution limit of brown bear in European Russia and W Siberia. Chestin *et al.* (1992) suggested that brown bears were never found on the Russian tundra, except in the Chukotka Peninsula. However, the situation appears to have changed. From Nenetsky district to Yamal Peninsula, brown bears are sometimes observed into the tundra by Nenets and scientists (N. Lecomte, unpubl.). Chukchi elders from the Kolyma delta region of Siberia now report many tundra brown bears that have come from the forest zone (Mustonen 2009).

Community fur returns and local knowledge suggest that there have been no change of the northern range of Canadian lynx in the Northwest Territories, Yukon or Alaska through the 1980s and 1990s (Poole 2003). The world's northernmost lynx population is found in Scandinavia (Odden *et al.* 2009). In this area, Eurasian lynx are becoming increasingly common (Hellborg *et al.* 2002).

There have been few studies of wolverines in North America, and the historical distribution remains unknown (COSEWIC 2003). However, Dawson (2000) reports that a small population on the coast of James Bay, in the Cape Henrietta Maria region, may have been extirpated during the 1970s, but that the species appears to be recolonizing some areas in NW Ontario. There have been no verified reports of wolverine in Quebec or Labrador for about 25 years (COSEWIC 2003). Wolverines were numerous and widely distributed in Fennoscandia up to the 19th century. Following intense persecution, their distribution and population size declined markedly during the 1900s (Landa *et al.* 2000, Flagstad *et al.* 2004). Landa *et al.* (2000) suggested that current distribution is restricted to the central and northern parts of Norway, Sweden, Finland and Russia.

The American mink was introduced to many parts of Europe in the 1920s and 1930s, mainly for fur-farming (Wildhagen 1956, Hersteinsson 1992, Bevanger & Henriksen 1995). Some introductions occurred in sub-Arctic areas such as near Murmansk in NW Russia (Lever 1985). Bonesi & Palazon (2007) report their presence in the low Arctic areas of Iceland, and they have been observed in the low Arctic of mainland Norway (N.G. Yoccoz, unpubl.).

3.5.1.3. Causes and prospects

Distributions of terrestrial carnivorous mammals have changed in the past century under both direct and indirect human influences. Direct human influences include overharvesting or persecution and introductions to previously unoccupied areas, both in direct relation to the economic incentives generated by the fur market (Robinson 2005). Indirect human influences probably include effects of climate change (Gilg *et al.* 2012), removal of competitors (usually other carnivores) and manipulation of herbivore population densities. Delivery of food subsidies, in the form of carcasses of semi-domesticated reindeer or simply human waste, has also influenced the distribution of terrestrial carnivorous mammals in the Arctic (Killengreen *et al.* 2011). Direct human influences may have been the most important drivers of change in the first half of the 20th century, whereas indirect human influences may have become more prevalent in the second half.

A rebirth of the fur market is possible if emerging economies resuscitate a high demand for fur products (Robinson 2005). However, climate change, industrialization of the Arctic and increased wildlife-human conflicts due to colonization by humans of new areas are more likely to influence populations of terrestrial carnivorous mammals in the decades to come.

These causal factors influencing past or future status and trends in species distributions are all mediated through changes in population abundance. We will therefore explore these causes and prospects in more detail in the next section.

3.5.2. Population sizes and densities

3.5.2.1. Status

Most of the 13 species of terrestrial carnivorous mammals covered here are socially solitary and maintain territories, which may limit their density and thus population size. Yet total population sizes of all species probably range at least in the tens of thousands given their wide geographical range. Global, and specifically Arctic, population size and density estimates exist for very few mammalian carnivore species (Tab. 3.4). In some species, most individuals live in the Arctic (e.g. Arctic fox) whereas in others only a very small (e.g. Eurasian lynx) or even insignificant (e.g. coyote) proportion of the global population is found there. It is often not possible to split populations between their Arctic and boreal components as individuals freely move across biome boundaries. For example, gray wolves can follow caribou during their seasonal migrations from tundra breeding grounds to boreal wintering areas (Walton *et al.* 2001).

None of the terrestrial carnivorous mammals included in this chapter is threatened at the global scale (Tab. 3.4). At the scale of individual jurisdictions, in many cases no

Table 3.4. Status and trends of Arctic terrestrial mammalian carnivores. Status and trends were assessed within political boundaries and thus refer to populations that often extend beyond the Arctic region.

	World IUCN	Alaska, USA	Canada					Green-land	Ice-land	Norway		Swe- den	Fin- land	Russia		
			All	Yukon	NWT	NU	Quebec, Labrador			Main- land	Sval- bard			All	East	West
Gray wolf	••		→	••	••		•			•		•	•	••	••	
Coyote	↑				73 ↑											
Arctic fox	•••	••• coast	•••		••		••	•• ↑		•	••	•	•	•• ↑	• Mednyi Island •• Bering Island	
Red fox	→				••											
Brown bear	•••	••	••	••	••	•••				•	•	•	••			
American black bear	↑	••	••		••											
Canadian lynx	→				••											
Eurasian lynx	→									•	•	•	••			
Least weasel	→									↑		••				
Stoat	••				••					→		••				
Wolverine	↓		w	••	••	••	Extir- pated?			•	•	•	••	••	••	
American mink	→				••											
American river otter	→				••											

Population Trends	
Dashed: based on expert opinion	
→	Stable trend
↓ ↑	Decreasing or increasing trend
Population Size (survey based)	
Grey: based on expert opinion	
•	< 100 individuals
••	101 to 1,000 individuals
•••	1,001 to 10,000 individuals
••••	10,001 to 50,000 individuals
•••••	50,000+ individuals

Population Status	
Red	Critically Endangered
Orange	Endangered
Yellow	Vulnerable / Threatened
Light Green	Near Threatened / Special Concern
Dark Green	Least Concern / Non-Endangered / Not at Risk
Grey	Data Deficient / No Information Not Evaluated / No Information on species status found
White	Not present in country/region

References:

Gray wolf: COSEWIC 1999a, Sillero-Zubiri *et al.* 2004, Cluff 2005, Gärdenfors 2005, Norwegian Red List 2006, Mech & Boitani 2008.
 Coyote: NWT 2004.
 Arctic fox: NWT 2004, Sillero-Zubiri *et al.* 2004, Gärdenfors 2005, Norwegian Red List 2006, Fuglei 2007, Angerbjörn *et al.* 2008a.
 Red fox: Voigt 1987, NWT 2004, Norwegian Red List 2006, Macdonald & Reynolds 2008.
 Brown bear: Miller *et al.* 1997, Gau & Veitch 1999, Rassi *et al.* 2001, COSEWIC 2002, Kindberg *et al.* 2004, Gau 2001, Norwegian Red List 2006, McLellan *et al.* 2008.
 American black bear: COSEWIC 1999b, NWT 2004, Garshelis *et al.* 2008.
 Canadian lynx: COSEWIC 2001, NWT 2004, Nowell 2008.
 Eurasian lynx: von Arx *et al.* 2004, Breitenmoser *et al.* 2008.
 Least weasel: NWT 2004, Hellstedt *et al.* 2006, Norwegian Red List 2006, Tikhonov *et al.* 2008.
 Stoat: NWT 2004, Hellstedt *et al.* 2006, Norwegian Red List 2006, Reid & Helgen 2008.
 Wolverine: Landa *et al.* 2001, Rassi *et al.* 2001, Gau and Mulders 2001, COSEWIC 2003, Novikov 2005, Abramov *et al.* 2009, Brøseth *et al.* 2009.
 American mink: NWT 2006.
 American river otter: NWT 2004, Serfass & Polechla 2008.

information exists on status of populations, as shown by the many gray cells in Tab. 3.4. For example, only the status of gray wolf and Arctic fox has been assessed in Alaska, whereas only the status of gray wolf has been assessed in Russia. At a regional scale, many populations are vulnerable, endangered and possibly extirpated (see Tab. 3.4 for individual species references). Several primarily boreal species are endangered or threatened in Fennoscandia, including gray wolf and wolverine in mainland Norway, Sweden and Finland, and brown bear in Norway. Arctic fox, the only truly Arctic carnivore in this region, is now one of the most endangered mammal species in Europe. The wolverine population in Quebec and Labrador may be locally extirpated (Fortin *et al.* 2005). The gray wolf subspecies *C. l. arctos*, found in the Northwest Territories and Nunavut was considered for protected status in 1999 but was not listed due to insufficient data (Van Zyll de Jong & Carbyn 1999).

3.5.2.2. Trends

The behavioral adaptations that make these species effective predators also make them elusive to researchers, so that detailed estimates of trends in population size are rare. In the smallest species (least weasel, stoat and American mink), such data are completely absent. Most species for which information exists are considered to be stable in terms of global population size (Tab. 3.4), with coyote and American black bear even increasing. One exception is the wolverine, which is considered as declining on a global scale.

This general stability of northern populations of terrestrial carnivores fits the trend of the Arctic Species Trends Index during the period 1970-2004, which was calculated from 306 species (965 populations) of vertebrates (McRae *et al.* 2010). Here we summarize some trends documented in a few species or populations to show the diversity of trends observed among populations and the nature of the evidence that exists regarding changes in size and density of populations. Some of the reported trends are for populations that largely spread south of the Arctic.

In NE Greenland, Dawes *et al.* (1985) report a decline in the wolf population during the 1930s, and by the early 1940s the species was most likely extirpated. However, the species came back after the cessation of fur trapping activities, and Marquard-Petersen (2009) found evidence that between 1978 and 1998 the wolf population of N and E Greenland consisted of up to 55 wolves in favorable times and maximum wolf density was estimated at 0.03 wolves/100 km² in this very alpine area. The gray wolf population in Scandinavia during the winter season of 2008-2009 was estimated between 213-252 individuals (Wabakken *et al.* 2009). Russia's total population is estimated to be about 70,000 and is fully viable (Mech & Boitani 2008). However, the population and density estimates specifically for Arctic Russia are not known. Indigenous communities of the Lower Kolyma region, in northeast Russia, report that the regional wolf population is stable and healthy (Mustonen 2007).

No information is available for coyote population estimates or densities within its northern range.

The Fennoscandian Arctic fox population declined between 1983 and 2000 (Angerbjörn *et al.* 1995) and was close to extinction around the year 2000. Numbers have increased since in response to intensive actions (Angerbjörn *et al.* 2008b), and today there are about 0-250 individuals distributed in four geographically separate areas (Dalén *et al.* 2006, Angerbjörn *et al.* 2008b). The number of Arctic foxes estimated in Norway (mainland) and Sweden is 150 and 80, respectively (Angerbjörn *et al.* 2008b). However, there have not been any confirmed litters born in Finland since 1996 (Kaikusalo *et al.* 2000, Dalén *et al.* 2006). The red fox has been reported to be increasing in numbers within the Fennoscandian mountain tundra (Østbye *et al.* 1978, Kaikusalo & Angerbjörn 1995, Tannerfeldt *et al.* 2002).

According to McLellan (1994), populations of brown bears in tundra habitat exist at the lowest recorded densities of all North American brown bears. Reynolds (1982) reported for Alaska North Slope populations that high bear densities in optimum habitat approached 2 bears/100 km², and densities in lower quality habitats were about 0.5 bears/100 km². Similarly, the density for bears of all ages in NW Alaska was estimated to be 2/100 km² (Ballard *et al.* 1990). Local hunters in NW Alaska believed brown bears were numerous and more abundant than observed historically (Loon & Georgette 1989). Brown bear population estimates for NE Europe suggest that there are about 37,500 bears (for the area including the Ural Mountains to the Finnish west coast, 53° to 69° N; Swenson 2000). Between 1998 and 2002 the number of adult female brown bears in Norway was estimated to be 6-12, with the highest density above the Arctic Circle (Swenson *et al.* 2003). This population does not meet the requirements to qualify as a viable population (Sæther *et al.* 1998) and is therefore dependent on the management of bear populations in Sweden and Finland (Norwegian Red List 2006). In Sweden, there were 1,635-2,840 bears in 2004, with a yearly growth rate of 4.7% (Gärdenfors 2005), rising to 3,000-3,700 in 2010 (Kindberg *et al.* 2011). Risk analysis of population demographic data from bear research in Scandinavia shows that today's Swedish bear population is viable (Gärdenfors 2005). The brown bear population size in Finland is estimated to be 810 (Kojola *et al.* 2006), and there are conflicting reports as to whether this population is increasing or decreasing (Kojola *et al.* 2006). Chestin *et al.* (1992) suggested a density of < 0.2 individuals/100 km² on the Russian tundra.

The total population of American black bears is believed to be increasing (Garshelis *et al.* 2008), but there are limited data to support this (Garshelis & Hristienko 2006). During the past two decades, most American black bear populations have grown both numerically and geographically (Williamson 2002). However, in northern latitudes few studies of black bears have been conducted to confirm this (Miller *et al.* 1997, Bertram & Vivion 2002).

The Eurasian lynx population (excluding Russia) is currently estimated to be 8,000 individuals, and the whole population (including Russia) is believed to be stable (Breitenmoser *et al.* 2008).

Wolverines are among the rarest and least studied mammalian carnivores in North America (Ruggiero *et al.* 1994, Copeland & Whitman 2003). Wolverine populations in Nunavut are believed to be stable, but sensitive to harvest pressures (COSEWIC 2003). In an indigenous traditional knowledge study of wolverines in N Canada, the majority of participants reported that wolverines were at low densities and rarely seen (Cardinal 2004). Most interviewees reported that populations were either stable or increasing, except near Yellowknife, Northwest Territories where the population might be decreasing. The wolverine population in N Yukon was thought to be increasing, due to less local trapping, and in the Kivalliq region, Nunavut, to have increased over the past 20-30 years (Cardinal 2004). The current population estimate of wolverines in the central to northern parts of Norway, Sweden, Finland and Russia is around 2,000 individuals (Landa *et al.* 2000), having increased since a period of persecution during the 1900s (Flagstad *et al.* 2004). The number of wolverines in the Murmansk, Kola Peninsula region of Russia is estimated to be 200-330 in 2004 (Makarova 2005), with Saami reporting increasing numbers (Mustonen & Mustonen 2011). Across Russia, an overall decrease in numbers seems to have taken place in the late 20th century (Landa *et al.* 2000).

American mink population densities and trends are unknown for most Arctic regions. Population numbers are reported as stable in the Northwest Territories (NWT 2006), but most data may come from non-Arctic areas. In Europe and Iceland, population estimates and trends are based on hunting records and largely show some stability in recent years. However, figures are again largely influenced by data coming from non-Arctic areas.

Limited information is available for North American river otter, although overall the population trend is believed to be stable (Serfass & Polechla 2008).

3.5.2.3. Causes and prospects

Many factors influence population numbers of Arctic carnivores and influencing factors rarely occur singly. Historically, population sizes were likely altered by hunting and trapping in some populations. While hunting and trapping have decreased for some of these species, they continue for others, some of which remain under high harvest pressure. Contaminants have been found in Arctic carnivores, but unfortunately, as with many other Arctic species, their sensitivities to contaminants and pollutants are largely unknown. Changes in climate, including temperature, snowfall and ice cover, have been shown to influence population densities. However, complex interactions between climate change and other factors can magnify impacts on biodiversity (CAFF 2010). Land-use changes have altered species distribu-

tions, migration routes and home ranges. Carnivorous species depend on other animals as targeted prey or via scavenging, so that changes in other species' populations often alter the demography of Arctic carnivores. There is limited information regarding effects of disease and parasites on Arctic terrestrial mammal carnivores. We now detail these causes and prospects.

Hunting and trapping

Current population sizes are commonly affected by hunting and trapping because fur-bearing mammals have experienced heavy harvest rates in some regions. For many mammalian carnivore populations, hunting and trapping pressure has decreased with declining fur prices (e.g. Arctic fox; Angerbjörn *et al.* 2004), yet for others it is still a concern. Increased access to northern regions via roads, all-terrain vehicles and snowmobiles has allowed hunters to cover more area, travel longer distances and visit more remote locations. This modernization of hunting has impacted all harvested species. In addition to legal harvesting, poaching is also an issue for some species.

In some regions of Canada's North, gray wolves were trapped and poisoned during the 1950s until they were extirpated. Poisoning, now illegal, has mostly been discontinued. Gray wolves are currently harvested for both commercial and subsistence use (Van Zyll de Jong & Carbyn 1999). On some Arctic islands, the annual harvest may be as high as 25% of the total population (Carmichael *et al.* 2001). In E Greenland, wolves were exterminated by commercial hunters in the 1930s, but recolonization has occurred due to migrating wolves from Canada (Marquard-Petersen 2009, 2011). Currently, wolverines are also harvested in some regions.

The Arctic fox in Scandinavia is classified as critically endangered after having experienced intensive hunting in the early 20th century, resulting in the population declining to a few hundred individuals (Lönnberg 1972 in Dalén *et al.* 2006); this population has failed to recover despite more than 65 years of protection (Dalén *et al.* 2006). However, Arctic foxes are sustainably hunted in Iceland, where red foxes are absent (Hersteinsson 2010).

Persson *et al.* (2009) suggest that poaching affects wolverine population dynamics in northern Scandinavia, causing up to 60% of adult mortality. In the forest and tundra areas of western Russia (specifically the Archangelsk Oblast region), the wolverine population is believed to be limited by motorized hunting (Landa *et al.* 2000). Wolverine populations in Canada's low Arctic tundra are likely to experience an increase in mortality due to increasing levels of resident and sport hunting, as well as resource development activity (Mulders *et al.* 2007), but possible effects on their population are not known. Although the Norwegian wolverine population is considered endangered, regular harvest or killing of litters has become an important tool in wolverine population management (Sæther *et al.* 2005). However, current management quotas may be too high to maintain a viable

wolverine population and, if continued, are likely to lead to extinction over large parts of Norway within a relatively short period (Sæther *et al.* 2005).

Pollution

The sensitivities of most Arctic species to many contaminants are unknown (Brunström & Halldin 2000, Hoekstra *et al.* 2003), limiting the ability to determine the risk for future adverse health effects (Brunström & Halldin 2000). Top-level carnivores accumulate organochlorine contaminants (e.g. PCBs) to relatively high concentrations compared with other terrestrial mammals due to biomagnification (Hoekstra *et al.* 2003). Canadian data indicate that Arctic terrestrial mammals have lower organochlorine contaminant concentrations than marine mammals, but terrestrial mammals (e.g. Arctic fox, wolverine) that feed or scavenge on marine animals tend to have relatively high concentrations (Hoekstra *et al.* 2003, Fisk *et al.* 2005). Therefore, coastal populations could have higher organochlorine concentrations due to the increased contribution of marine biota to their overall diet. Conversely, heavy metal contaminant levels in Canadian terrestrial mammals are similar or greater than those seen in marine mammals (Fisk *et al.* 2005).

Climate change

Changes in climate may be providing new habitats for southern species to extend their northern range limits. The current number of brown bears may be increasing due to the appearance of new ecological niches because of rising air temperatures in sub-Arctic regions (Minnerv 2007). Similarly, North American river otters may prosper in previously marginal northern areas in Alaska (Feldhamer *et al.* 2003) and potentially other regions. More extensive research has been conducted on the range expansion of red fox. Hersteinsson & Macdonald (1992) analyzed factors limiting the distribution of red foxes at their northern limit and found that summer temperature limited fox distribution. However, variables related to winter conditions (minimum temperature, maximum snow depth and duration of snow cover) may also have a significant effect (Hersteinsson & Macdonald 1992, Barton & Zalewski 2007, Gallant *et al.* 2012). On Iceland red fox are absent, providing an important opportunity to study the Arctic fox's response to climate change.

Decreased amounts of snow may be considered as habitat loss in some species. Brodie & Post (2010) argue that wolverine population declines may in part result from declining snowpacks, with snow accumulations being reduced by warmer winters. Declining snowpack could affect demographics of carnivores by reducing reproductive success due to decreased availability of den sites that require snow for den structure and insulation (Magoun & Copeland 1998, Aubry *et al.* 2007), decreased juvenile survival due to altered thermal regimes (Pulliainen 1968 and Bjärvall *et al.* 1978 in Copeland *et al.* 2010), altered availability of food in winter and early spring (Persson 2005, Lofroth *et al.* 2007), and reduced density of ungulate carcasses through increased ungulate survival (Wilmers & Post 2006).

In some regions, prolonged snow cover during summer is also possible due to climate change (via increased precipitation and decreased summer temperatures). It has been suggested that snow-covered vegetation may increase herbivore mortality and as a result augment wolf reproduction on Ellesmere Island, Canada (Mech 2004).

Several Arctic mammals use sea ice for travel, and ongoing decreases in the extent or duration of sea ice may impact genetic diversity of these species. For example, sea ice is necessary for the dispersal of Arctic fox between island populations, and if seasonal ice connections between Arctic islands are lost, small isolated populations of foxes will lose genetic diversity and have higher levels of inbreeding and genetic divergence (Geffen *et al.* 2007). The two wolf populations on Banks Island and the high Arctic (Ellesmere and Devon Islands) show genetic signatures of recent population declines, suggesting that their recovery is a result of recolonization from other islands; therefore these wolves may function as a metapopulation, where migration is occurring primarily through two sea ice corridors from Baffin Island and Victoria Island, respectively (Carmichael *et al.* 2008).

Industrial development and land-use changes

Industrial developments and landscape alterations affect species by fragmenting habitat, altering migration corridors and increasing human access. An increase in barriers due to habitat loss between populations causes decreases in immigration and potentially reduced gene flow (see Linnell *et al.* 2005 and Aspi *et al.* 2009 for discussion on gray wolves).

Mineral exploration and developments in Canada's central Arctic has led to increased human presence in this region. Johnson *et al.* (2005) examined factors that influenced the distribution of gray wolves, brown bears, wolverines and barren-ground caribou. They found that mines and other major developments had the largest negative effect on species' occurrence, followed by exploration activities and outfitter camps. Wolves and bears had the strongest negative response to human disturbances (Johnson *et al.* 2005).

Migration of brown bears from Russia to Finland has provided a stable source of immigrants to repopulate Finland and maintain a high genetic diversity within the re-established region (Saarma & Kojola 2007), but increased fragmentation or barriers could limit this migration.

Human actions likely will be a controlling factor in the success and persistence of wolverine populations. Increasing levels of human development and harvesting pressures may cause further habitat loss and fragmentation for the wolverine populations across the Northwest Territories (COSEWIC 2003). Although the cumulative effects of habitat alteration, increased road building and traffic are not fully understood, developments in the Arctic tundra frequently attract wolverines, which may be killed as nuisance animals (COSEWIC 2003). Potential cumulative impacts of habitat loss, disturbance and

increasing mortality pressures may lead to a decline in wolverine abundance (Mulders *et al.* 2007). Increased road access usually results in greater hunting and trapping pressure, which is a primary mortality factor for wolverines (Hornocker & Hash 1981, Magoun 1985). Wolverines may be especially vulnerable on the Arctic tundra, where visibility and snowmobile access are good (COSEWIC 2003). A hunter from Inuvik, Northwest Territories, noted that seismic lines cut for oil and gas exploration are a threat to wolverine habitat (Cardinal 2004), and others noted that new roads would allow hunters greater access (Cardinal 2004).

In Canada's central Arctic, brown bears could be in danger of population decline if human activity proceeds at an increasing rate (McLoughlin *et al.* 2003). The barren-ground brown bear might be more susceptible to human activity due to their large spatial requirements compared with other brown bears in North America (McLoughlin *et al.* 1999). Similarly, disturbance from diamond mining and road construction near wolf dens has a negative effect on their reproductive success directly or, indirectly, by altering the distribution or timing of movements of caribou, their main prey (Walton *et al.* 2001).

Parasites and diseases

Due to the distances these species travel, diseases may be transmitted over long distances. However, the low population densities of these species also act to reduce disease transmission. An outbreak of sarcoptic mange *Sarcoptes scabiei* occurred in Scandinavian red foxes during the late 1970s and 1980s (Lindström *et al.* 1994). Within eight years, the sarcoptic mange had spread over mainland Sweden and killed approximately 50% of the red fox population (Lindström & Mörner 1985), with as much as 90% mortality reported in some regions (Mörner 1992). The Scandinavian red fox population began to recover in the late 1980s (Lindström *et al.* 1994). Encephalitozoonosis *Encephalitozoon cuniculi* is thought to influence Arctic fox population dynamics in Iceland (Hersteinsson *et al.* 1993). Currently, rabies is regarded as endemic among Arctic and red fox in northern and western regions of Alaska, with a 3-4 year cyclic epidemic occurrence (Ritter 1981 and Follmann 1990 in Mørk & Prestrud 2004), and in Svalbard and NW Russia (Mørk *et al.* 2011).

It is currently unknown what parasites and diseases may threaten Arctic carnivores in the future. A number of diseases and parasites of domestic species could be detrimental. Also, some previously recorded diseases, including those from temperate latitude populations, could be devastating if outbreaks were to occur. An example is sarcoptic mange in isolated fox populations such as those on islands (Henriksen *et al.* 1993).

3.6. MARINE MAMMALS

The world's marine mammals were recently classified as disproportionately threatened and data poor compared

with their terrestrial counterparts. Their status was noted of particular concern, and several reviews have outlined global conservation issues (Schipper *et al.* 2008, Kovacs *et al.* 2012). Several species inhabit the Arctic marine biome exclusively and are specially adapted to the dynamic and extreme environment (Laidre *et al.* 2008a). Arctic marine mammals are highly associated with sea ice or are sea ice obligates – meaning their life history events (reproduction, molting, resting) and feeding behavior are closely linked to sea ice dynamics. With some exceptions, these species range widely and undergo large seasonal migrations, covering thousands of kilometers in a single year. Most species of endemic Arctic marine mammals are also important cultural and food resources for indigenous peoples.

Assessing the status and trends of marine mammal populations in the Arctic is difficult because of the elusive nature of many species and their large ranges. Further, there are logistical challenges associated with surveying vast remote marine areas. Therefore, there are numerous data gaps even for basic information such as population sizes, trends and distributions. Here we summarize what is known about these parameters for marine mammals that inhabit low and high Arctic waters and we discuss implications of data gaps given predictions of sea ice loss and climate warming (e.g. IPCC 2007).

3.6.1. Species richness and distribution

3.6.1.1. Status

There are 35 species of marine mammals that inhabit or seasonally use Arctic waters. In this review, we discuss these species in the context of 12 regions in low or high Arctic waters (Appendix 3.2, Fig. 3.5). Of these species, seven are endemic to the Arctic and are dependent on, or highly associated with, sea ice for all or parts of the year. We refer to these as the core Arctic species, and they are the narwhal, beluga *Delphinapterus leucas*, bowhead whale, ringed seal *Pusa hispida*, bearded seal *Erigonathus barbatus*, walrus and polar bear. In addition, four other ice seal species are highly dependent on sea ice for pupping in the low Arctic in spring, but are generally pelagic or use sub-Arctic waters for the rest of the year. We include these in this assessment, and they are the spotted seal *Phoca largha*, ribbon seal *Phoca fasciata*, harp seal *Pagophilus groenlandicus* and hooded seal *Cystophora cristata*.

The three species of Arctic cetaceans have different patterns of distribution and population structure. The beluga has a circumpolar Arctic distribution and is found in discrete subpopulations in the high and low Arctic, generally defined by summering areas (O'Corry-Crowe *et al.* 1997, Richard *et al.* 2001, Innes *et al.* 2002, Palsbøll *et al.* 2002). The narwhal is confined to the Atlantic Arctic in the eastern Canadian high Arctic and in waters around W and E Greenland, Svalbard and Franz Joseph Land (Gjertz 1991, Koski & Davis 1994, Dietz *et al.*



Figure 3.5. Regions used to enumerate Arctic marine mammal species. High Arctic and low Arctic marine boundaries are shown with solid and dashed lines, respectively. These were used to define areas in Appendix 3.2. We do not include species that only use sub-Arctic waters, except for the seasonally ice covered Sea of Okhotsk, entirely within the sub-Arctic, where several populations of core Arctic marine mammals occur. Note that while populations and species are enumerated within regions, individuals frequently cross several regional boundaries.

2001, Heide-Jørgensen & Aquarone 2002, Innes *et al.* 2002). Narwhals sporadically move into Russian and occasionally Alaska Arctic waters from the Barents Sea through to the Chukchi Sea (Burdin *et al.* 2009, Allen & Angliss 2011). The bowhead whale, the only mysticete to inhabit the Arctic year round, has a circumpolar distribution with most populations moving between high Arctic waters in summer and low Arctic waters in winter (Braham *et al.* 1980, Ellison *et al.* 1987, George *et al.* 1989, Moore & Reeves 1993, Heide-Jørgensen *et al.* 2006, Citta *et al.* 2012). However, individuals of the Svalbard-Barents Sea stock overwinter in the high Arctic (Lydersen *et al.* 2012, Stafford *et al.* 2012).

Of the true Arctic pinnipeds, ringed seals have a circumpolar distribution, inhabiting permanently or seasonally ice-covered areas from the North Pole to the low Arctic with their distribution extending into some lake and river systems in northern Canada (Kovacs *et al.* 2008). Bearded seals also have a circumpolar distribution in the Arctic.

Polar bears have a circumpolar distribution. They occur throughout ice-covered Arctic regions, especially in areas of annual ice cover over the continental shelf and the inter-island channels of various archipelagos. Their distribution is not uniform, and the global population is divided into 19 recognized sub-populations (Paetkau *et al.* 1999, Obbard *et al.* 2010).

Walrus have a discontinuous circumpolar distribution with two recognized subspecies: the Atlantic walrus *Odobenus rosmarus rosmarus* distributed from the eastern Canadian Arctic to the Kara Sea, and the Pacific walrus distributed in the Pacific Arctic from Mys Shelagyskiy in Siberia to Barter Island in Alaska and in the Bering and Chukchi Seas. A third subspecies, the Laptev walrus *O. r. laptevi*, confined to the Laptev Sea region was suggested (Chapksii 1940) based on a limited sample (Fay 1985), but recent molecular genetics studies concluded that the Laptev walrus belongs with the Pacific subspecies (Lindqvist *et al.* 2009).

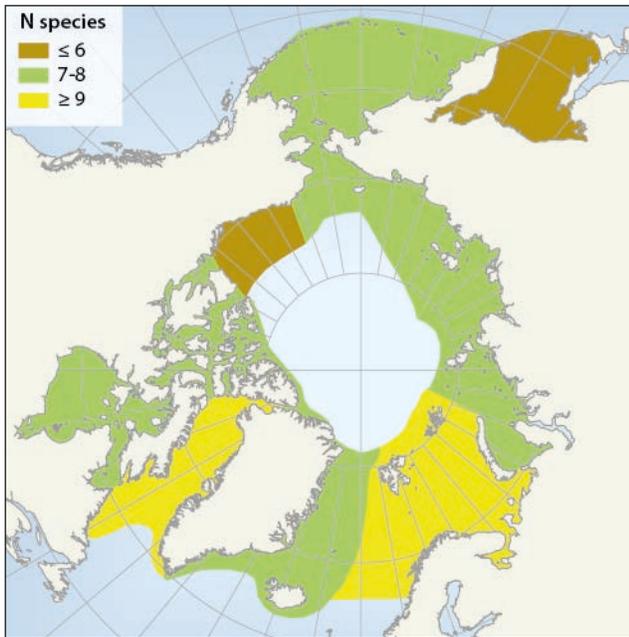


Figure 3.6. Species richness of marine mammals ($n = 11$ core Arctic marine mammals only) in high and low Arctic waters.

Of the four sub-Arctic ice seal species, the spotted seal and ribbon seal are found in the Pacific Arctic and peripheral seas (Burns 1981, Boveng *et al.* 2008, Boveng *et al.* 2009). Conversely, the harp seal and hooded seal are confined to the Atlantic Arctic, and are widely distributed throughout the North Atlantic and Arctic Ocean shelf and seas (Lavigne & Kovacs 1988, Kovacs 2008a, 2008b).

Species richness for the resident Arctic marine mammals ($n = 11$) is highest in three regions: Baffin Bay, Davis Strait and the Barents Sea, where nine of 11 species are present (Fig. 3.6). Most other regions have seven or eight Arctic species present. The Beaufort Sea and the Sea of Okhotsk regions have six species.

In addition to the 11 Arctic ice-dependent species discussed above, there are 24 other marine mammal species that occur in low Arctic waters or migrate to the high and low Arctic seasonally, from tropical and temperate waters, to feed (Appendix 3.2). These species do not depend on the Arctic ecosystem year-round. They include four species of pinnipeds: the northern fur seal *Callorhinus ursinus* and the Steller sea lion *Eumetopias jubatus* found in the Pacific low Arctic within the Okhotsk and Bering Seas (Pribilof Islands); the gray seal *Halichoerus grypus* found in the Atlantic Arctic, and the harbor seal *Phoca vitulina* which occurs in low Arctic waters of the Atlantic and Pacific, with one population living in the high Arctic in Svalbard.

Nineteen species of cetaceans seasonally use low or high Arctic waters. The North Pacific right whale *Eubalaena japonica* and the gray whale *Eschrichtius robustus* are confined to the Pacific low and high Arctic. The North Atlantic right whale uses low Arctic waters of E Greenland. The blue whale *Balaenoptera musculus*, fin whale *Balaenoptera*

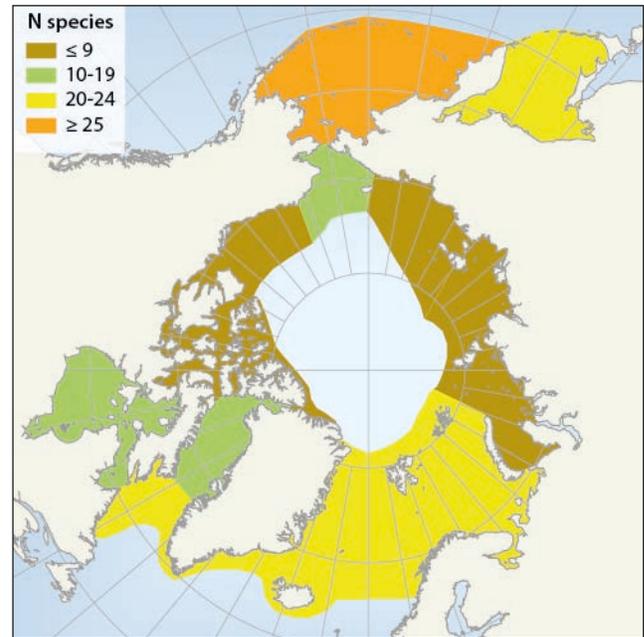


Figure 3.7. Species richness map of marine mammal that are present or seasonally occur in low and high Arctic waters at any time of the year ($n = 35$).

physalus, sei whale *Balaenoptera borealis*, minke whale *Balaenoptera acutorostrata* and humpback whale *Megaptera novaeangliae* are found in both low and high Arctic waters in summer in both the Atlantic and Pacific. The sperm whale *Physeter macrocephalus* can also be found in the low Arctic waters of both the Atlantic and Pacific. Baird's beaked whale *Berardius bairdii*, Stejneger's beaked whale *Berardius stejnegeri* and Cuvier's beaked whale *Ziphius cavirostris* are all found in the low Arctic waters of the Pacific Arctic, specifically in the Okhotsk and the Bering Seas. The northern bottlenose whale *Hyperoodon ampullatus* is found in the low Arctic waters of the Atlantic, including Davis Strait, Baffin Bay, E Greenland and the Barents Sea.

Among the delphinids and porpoises, the killer whale *Orcinus orca* visits circumpolar Arctic waters during the ice-free season in nearly all regions. The white-beaked dolphin *Lagenorhynchus albirostris*, long-finned pilot whale *Globicephala melas* and Atlantic white-sided dolphin *Lagenorhynchus acutus* are all found in the Atlantic low Arctic during summer. Dall's porpoise *Phocoenoides dalli* occurs in low Arctic waters of the Pacific, while harbor porpoise *Phocoena phocoena* are found in low Arctic waters of both the Atlantic and Pacific.

Finally, the sea otter *Enhydra lutris* extends peripherally into low Arctic waters in the southern Bering Sea. For this species, sea ice is a serious impediment, causing animals to die or abandon areas when coverage becomes too extensive (Schneider & Faro 1975). This population was nearly extirpated early in the Russian fur trade, but slowly re-colonized its range (Kenyon 1969).

When species richness is examined for all 35 marine mammal species that occur in the high and low Arctic combined, including seasonal visitors, diversity is highest

in the Bering Sea ($n = 25$ species present in some season of the year). Other regions with high species richness include Davis Strait ($n = 23$ species) and E Greenland ($n = 23$ species) (Fig. 3.7).

3.6.1.2. Trends

Many marine mammal populations were severely reduced due to extensive commercial whaling, commercial fur trade, or subsistence hunting that took place across the Arctic over the past several centuries. Heavy harvesting reduced many populations to very low numbers and contracted ranges, with most rendered close to extinct (e.g. E Greenland-Svalbard-Barents Sea bowhead whale, Wiig *et al.* 2010). Similarly, excessive harvest during the commercial whaling era extirpated the Atlantic gray whale and the Northeast Atlantic right whale *Eubalaena glacialis* (Krupnik 1993, Nowak 1999). Therefore, changes in the distributions of marine mammals in low and high Arctic areas have occurred as several populations have recovered over the past 50-100 years. In this review, we focus on recent changes (< 50 years) in distribution and abundance related to climate warming rather than changes in distribution related to recovery of populations after depletion.

» ... once the [commercial] whalers came they killed off lots of bowhead whales, thus the very evident decrease in population among the bowhead whales ...

(Mikitok Bruce, quoted in NWMB 2000).

Changes in the distribution of species or populations of marine mammals require investigations on long time-scales, far longer than the context of most present-day ecological studies or monitoring programs (Laidre *et al.* 2008a). Obtaining this trend information for Arctic marine mammals requires extensive and expensive surveys conducted over decades. Thus relatively few data are available from the past 50 years. Recent documented examples of changes in distribution of Arctic marine mammals are varied, ranging from expansion to distributional shifts to local extirpation, making broad generalizations difficult. Here we discuss some of the best documented changes in distribution that are thought to be climate-change related.

In W Greenland, a clear relationship between the extent of annual sea ice cover in Baffin Bay and the offshore distance of beluga whales was established based on 30 years of aerial survey data. Beluga whales have shifted their distribution westward (offshore) with the receding sea ice edge as the banks off W Greenland open up earlier in spring (Heide-Jørgensen *et al.* 2010a).

Recent data suggest that geographic barriers (such as heavy sea ice in narrow straits) that have separated bowhead whales in Alaska and Greenland may be disappearing with sea ice loss. Two satellite-tagged bowhead whales, one from W Greenland and one from N

Alaska, entered the Northwest Passage from opposite directions and spent approximately 10 days in the same area in 2010 (Heide-Jørgensen *et al.* 2011a). Other than ancillary evidence from harpoon remains in blubber, this is the first time geographic overlap between the Bering-Chukchi-Beaufort and the Davis Strait-Baffin Bay bowhead whale populations has been directly observed and documented. These populations were likely connected for periods during the Pleistocene, but have been assumed to be historically separated by sea ice in the Northwest Passage. Reductions in summer ice in the Northwest Passage may be facilitating exchanges between these and possibly other populations or species.

Polar bears have demonstrated shifts in summer and fall distribution in recent decades. As sea ice recedes or breaks up earlier, more polar bears are arriving on land earlier, staying for longer periods and appearing in areas not used previously (Stirling *et al.* 1999, Fischbach *et al.* 2007, Schliebe *et al.* 2008, Gleason & Rode 2009). This has been primarily documented in the Beaufort Sea and in western Hudson Bay and is attributed to sea ice loss. Polar bear denning locations have also shifted in some regions in response to changing ice conditions, with more dens appearing on land (Fischbach *et al.* 2007). Changes in access to traditional denning areas have also occurred with the disappearance of sea ice as a platform to allow movement to islands (Derocher *et al.* 2011).

Pacific walrus have recently been hauling out on land along the Alaska and Chukotka coasts of the Chukchi Sea in the summer (Garlich-Miller *et al.* 2011). This behavior is attributed to loss of annual Arctic sea ice and the retreat of the pack ice beyond the continental shelf of the Chukchi Sea in summer. While fall (October-November) migratory aggregations of Pacific walrus have been observed on the Alaska coast in the past, the summer haul outs are a new phenomenon and occur primarily north of Point Lay (Garlich-Miller *et al.* 2011).

Rosing-Asvid (2008) reported a change in distribution of whelping harp seals around W Greenland. This species is dependent on stable ice for the whelping and lactation period lasting 12 days in spring. In recent years, unprecedented high numbers of harp seals have been found concentrated along the ice edge of central W Greenland in late January and early February, including pregnant females with fetuses close to birth weight. This suggests that harp seals may be whelping in new areas. In addition, large anomalous herds of harp seals have been seen around Svalbard during winter where they previously occurred only in summer (Kovacs *et al.* 2011).

Recent studies suggest that changes in the distribution of sub-Arctic species have also been observed, although the increase in research effort and new techniques being used over the past 10 years may be responsible in part for the new information. In the Canadian high Arctic, killer whales have been reported to be expanding their range northward during the ice-free period (Higdon & Ferguson 2009). Opportunistic and anecdotal data also

suggest sub-Arctic baleen whales are occurring farther north, possibly expanding their range with longer ice-free seasons (Kovacs *et al.* 2011, Higdon & Ferguson 2011). Passive moored buoys indicate that fin whales are present in the Bering Sea and Davis Strait almost year-round, when the waters are free of sea ice (Simon *et al.* 2010, Stafford *et al.* 2010). In Fram Strait, calls from blue whales have been recorded from June to October while calls from fin whales were recorded from August to March (Moore *et al.* 2012a). Mellinger *et al.* (2011) recorded data from 2007-2008 showing evidence that North Atlantic right whales may now occupy SE Greenland, an important nineteenth century whaling area from which they were thought to have been extirpated.

3.6.2. Population sizes and densities

3.6.2.1. Status

Very few populations of marine mammals have been studied anywhere in the Arctic for long enough to allow a comprehensive assessment of the possible effects of long-term climate warming on population size and demographic parameters (Laidre *et al.* 2008a). The true population size of many species and subpopulations is unknown. For the 11 endemic Arctic marine mammal species, worldwide population sizes range widely, between ~ 20,000 to many millions (Tab. 3.5).

Beluga abundance worldwide is estimated to be at least 150,000 individuals (Harwood *et al.* 1996, Boltunov & Belikov 2002, Innes *et al.* 2002, Jefferson *et al.* 2008). This species is assessed according to 29 subpopulations defined by summering areas (IWC 2000). Large populations of beluga (20,000-40,000 individuals) occur in the eastern Bering Sea, the eastern Beaufort Sea and western Hudson Bay. However current population sizes are unavailable for some of these subpopulations because surveys have not been conducted in these areas for > 15 years (Allen & Angliss 2011). Population estimates are available for some subpopulations because they are monitored at regular intervals (e.g. Cook Inlet) (Hobbs *et al.* 2011). Lowry *et al.* (2008) documented a maximum uncorrected index count in Bristol Bay of approximately 1,000 whales. Almost nothing is known about population abundance of belugas in the Russian sector of the Arctic, in a continuum including the Kara, Laptev and East Siberian Seas. Similarly, no abundance determinations have been attempted for beluga in Norwegian waters.

» During winter, when the high tide comes, the beach ice will be saturated with water leaking through shoreline cracks. When the high tide and strong currents come, it is time to look for beluga whales because the winds will be strong and coming from the northwest. That is when you expect to see some beluga at the floe edge.

(Noah Isaac, quoted in McDonald *et al.* 1997).

The worldwide population of narwhals is ~ 100,000 animals (Koski & Davis 1994, Innes *et al.* 2002, Heide-Jørgensen *et al.* 2010b, Richard *et al.* 2010). Similar to belugas, narwhals are divided into several subpopulations based on summering location. The narwhals that summer in the Canadian high Arctic and eastern Baffin Island number at least 70,000 animals (Innes *et al.* 2002, NAMMCO 2005, Richard *et al.* 2010), and the primary subpopulations are located at Somerset Island, Admiralty Inlet, Eclipse Sound and E Baffin Island, and northern Hudson Bay. Some areas in Canada, such as near Ellesmere Island, contain other unsurveyed aggregations which are thought to contain small numbers of whales. In W Greenland, the primary subpopulations are centered at Inglefield Bredning and Melville Bay (Heide-Jørgensen 2004, Heide-Jørgensen *et al.* 2010b). The number of subpopulations in E Greenland is unknown, but approximately 6,000 summer between Scoresby Sound and Ammassalik (Heide-Jørgensen *et al.* 2010b).

Bowhead whales number fewer than 20,000 worldwide (George *et al.* 2004, Cosens *et al.* 2006, Heide-Jørgensen *et al.* 2007). There are five recognized subpopulations: Bering-Chukchi-Beaufort Seas, Hudson Bay-Foxe Basin, Davis Strait-Baffin Bay, Svalbard-Barents Sea and the Okhotsk Sea (Rugh *et al.* 2003). Recent data have called into question the distinction between the Hudson Bay-Foxe Basin and the Davis Strait-Baffin Bay stocks (Heide-Jørgensen *et al.* 2006, Ferguson *et al.* 2010, Givens *et al.* 2010), and herein we refer to these stocks as the combined eastern Canada-W Greenland stock. The largest fraction of the global population is located in the Bering-Chukchi-Beaufort Seas, with a population estimate from 2001 of 12,631 (95% CI: 7,900-19,000) (Koski *et al.* 2010). The eastern Canada-W Greenland stock is estimated to number about 6,500 individuals (Heide-Jørgensen *et al.* 2007, IWC 2008, Wiig *et al.* 2010). The population in Svalbard-Barents Sea has not been estimated due to low numbers, although up to 17 bowhead whales were sighted on summer surveys between 2006 and 2008 in NE Greenland and the Fram Strait, indicating that whales do persist in this area (Rugh *et al.* 2003, Boertmann *et al.* 2009, Wiig *et al.* 2010). There is a small population in the Sea of Okhotsk that likely numbers < 400 animals but no recent surveys have been conducted (Ivaschenko & Clapham 2009).

The worldwide abundance of ringed seals is likely in the low millions (Frost & Lowry 1981, Reeves 1998), estimated by Kelly *et al.* (2010) as 4-7 million. There are few data available on regional population sizes. Five subspecies of ringed seals are recognized: *P. h. hispida* (Arctic ringed seal) is thought to number about 2.5 million animals, while *P. h. ochotensis* (Sea of Okhotsk ringed seal) numbers > 800,000 animals (Miyazaki 2002). The three other sub-species *P. h. botnica* (Baltic Sea ringed seal), *P. h. ladogensis* (Lake Ladoga ringed seal), and *P. h. saimensis* (Lake Saimaa ringed seal) are not assessed here. Few regional estimates exist for *P. h. hispida*. In Hudson Bay, ringed seal abundance has been estimated at over 500,000 individuals (Stewart & Lockhart 2005, Hoover

Table 3.5. Subpopulations or stocks for each true Arctic marine mammal together with abundance estimate (abundance may be from dedicated survey with 95% CI, ballpark/rough estimate, or simulated from Population Viability Analysis). Year for estimate is given together with known trend in abundance (increasing, decreasing, stable or unknown). See text for rates.

Species	Subpopulation/Stock	Abundance	Year	Trend	Citation
Beluga	E Siberian & W Chukchi Seas	Unknown		Unknown	
	Eastern Chukchi Sea	3,700	1992	Unknown	Frost <i>et al.</i> 1993
	Eastern Beaufort Sea	41,800	1999	Unknown	Duval 1993, Kingsley & Gauthier 2002, Allen & Angliss 2011
	Eastern Bering Sea	18,000	1989-1991	Unknown	Allen & Angliss 2011
	Bristol Bay	1,600	2000	Increasing	Lowry <i>et al.</i> 2008
	Cook Inlet	284 (95% CI: 207-389)	2010	Declining	Allen & Angliss 2011, Hobbs <i>et al.</i> 2011
	Western Hudson Bay	57,300 (95% CI: 37,700-87,100)	1978, 1987, 2004	Unknown	Richard <i>et al.</i> 1990, Richard 1993, Richard 2005
	Southern Hudson Bay	7,000	1987	Unknown	Ognetov 1987, Richard 2005
	James Bay	9,292 (95% CI: 2,828-30,530)	2008	Unknown	Ognetov 1987, Gosselin <i>et al.</i> 2009
	Eastern Hudson Bay	2,646 (SE = 1,959)	2008	Declining	Gosselin <i>et al.</i> 2009, Bourdages <i>et al.</i> 2002
	St. Lawrence Estuary	1,100	1997	Stable	Gosselin <i>et al.</i> 2007, Hammill <i>et al.</i> 2007
	Ungava Bay	<50	2007	Unknown	Gosselin <i>et al.</i> 2009, Hammill <i>et al.</i> 2004
	Cumberland Sound	1,500	2001	Unknown	COSEWIC 2004
	E high Arctic-Baffin Bay	21,200, ± 25% CV	1996	Unknown	Innes <i>et al.</i> 2002
	W Greenland winter	10,595 (95% CI: 4,904-24,650)	2006	Unknown	Heide-Jørgensen & Aquarone 2002, Heide-Jørgensen <i>et al.</i> 2003, Heide-Jørgensen <i>et al.</i> 2010a
	White Sea	8,000	2005	Declining	Burdin <i>et al.</i> 2009
	Svalbard	Unknown		Unknown	Kovacs & Lydersen 2006, Gjertz & Wiig 1994
	Kara & Laptev Seas	Unknown		Unknown	
	Gulf of Anadyr	Unknown		Unknown	
	Okhotsk Sea	18,000-20,000	1987	Unknown	Ognetov 1987
Narwhal	Eclipse Sound stock	20,225 (95% CI: 9,471-37,096)	2004	Unknown	Richard <i>et al.</i> 2010
	Admiralty Inlet stock	18,049 (95% CI: 11,613-28,053)	2010	Unknown	Richard <i>et al.</i> 2010, Asselin & Richard 2011
	Somerset Island stock	45,358 (95% CI: 23,397-87,932)	2002	Unknown	Innes <i>et al.</i> 2002, Richard <i>et al.</i> 2010
	E Baffin fiords stocks	10,073 (95% CI: 5,333-17,474)	2003	Unknown	Richard <i>et al.</i> 2010
	W Greenland Inglefield Bredning stock	8,368 (95% CI: 5,209-13,442)	2007	Unknown	Heide-Jørgensen <i>et al.</i> 2010b
	W Greenland Melville Bay stock	6,024 (95% CI: 1,403-25,860)	2007	Unknown	Heide-Jørgensen <i>et al.</i> 2010b
	Northern Hudson Bay	5,053 ± 40% CV	2000	Unknown	COSEWIC 2004, Richard 2008
	W Greenland winter aggregations	7,819 (95% 4,358-14,029)	2006	Unknown	Heide-Jørgensen <i>et al.</i> 2010b
	E Greenland	6,444 (95% 2,505-16,575)	2008	Unknown	Heide-Jørgensen <i>et al.</i> 2010b
Bowhead	Bering-Chukchi-Beaufort Seas (BCB)	12,631 (95% CI: 7,900-19,000)	2001	Increasing	George <i>et al.</i> 2004, Koski <i>et al.</i> 2010
	E Canada-W Greenland (BBDS and FBHB)	6,500	2002-2009	Increasing	Heide-Jørgensen <i>et al.</i> 2007, IWC 2008, Wiig <i>et al.</i> 2011
	Svalbard-Barents Sea	Unknown	–	Unknown	Rugh <i>et al.</i> 2003, Boertmann <i>et al.</i> 2009, Wiig <i>et al.</i> 2010
	Okhotsk Sea	<400	1979	Unknown	Rugh <i>et al.</i> 2003, Ivaschenko & Clapham 2009

(continues >)

(continued)

Species	Subpopulation/Stock	Abundance	Year	Trend	Citation
Ringed seal	Arctic subspecies	~2.5 million	1970s	Unknown	Miyazaki 2002
	Okhotsk Sea subspecies	>800,000	1971	Unknown	Miyazaki 2002
	Hudson Bay	516,000	1995	Unknown	Stewart & Lockhart 2005
Bearded seal	Bering-Chukchi Seas	250,000-300,000	1970s	Unknown	Fedoseev 2000
	Canadian waters	190,000	1958-1979	Unknown	Cleator 1996
	Atlantic and Russian Arctic	Unknown	–	Unknown	–
	Okhotsk Sea	200,000-250,000	1968-1969	Unknown	Fedoseev 2000
Walrus	Bering-Chukchi Seas	~129,000	2006	Unknown	Speckman <i>et al.</i> 2011
	Atlantic subspecies	~20,000	1995-2009	Mixed	Born <i>et al.</i> 1995, Witting & Born 2005, COSEWIC 2006, Lydersen <i>et al.</i> 2008, NAMMCO 2009
	Laptev Sea	3,000-5,000	1992	Unknown	Belikov & Boltunov 2005, Burdin <i>et al.</i> 2009
Polar bear	Arctic Basin	Unknown	–	Unknown	Obbard <i>et al.</i> 2010
	Baffin Bay	1,546 (690-2,402)	2004	Declining	Obbard <i>et al.</i> 2010
	Barents Sea	2,650 (1,900-3,600)	2004	Unknown	Obbard <i>et al.</i> 2010
	Chukchi Sea	Unknown	–	Declining	Obbard <i>et al.</i> 2010
	Davis Strait	2,158 (95% CI: 1,833-2,542)	2007	Stable	Peacock <i>et al.</i> 2013
	E Greenland	Unknown	–	Unknown	Obbard <i>et al.</i> 2010
	Foxe Basin	2,578 (2,088-3,182)	2007	Unknown	Obbard <i>et al.</i> 2010
	Gulf of Boothia	1,592 (870-2,314)	2000	Stable	Obbard <i>et al.</i> 2010
	Kane Basin	164 (94-234)	1998	Declining	Obbard <i>et al.</i> 2010
	Kara Sea	Unknown	–	Unknown	Obbard <i>et al.</i> 2010
	Lancaster Sound	2,541 (1,759-3,323)	1998	Declining	Obbard <i>et al.</i> 2010
	Laptev Sea	Unknown	1993	Unknown	Obbard <i>et al.</i> 2010
	M'Clintock Channel	284	2000	Increasing	Obbard <i>et al.</i> 2010
	Northern Beaufort Sea	1,202 (686-1,718)	2006	Stable	Obbard <i>et al.</i> 2010
	Norwegian Bay	190 (102-278)	1998	Declining	Obbard <i>et al.</i> 2010
	Southern Beaufort Sea	1,526 (1,210-1,842)	2006	Declining	Obbard <i>et al.</i> 2010
	Southern Hudson Bay	900-1,000 (496-1,050)	2005	Stable	Obbard <i>et al.</i> 2010
	Viscount Melville	215 (99-331)	1992	Unknown	Obbard <i>et al.</i> 2010
	Western Hudson Bay	935 (794-1,076)	2004	Declining	Obbard <i>et al.</i> 2010

2010). In this area, densities are greater on landfast ice (1.3-3.4 seals/km²) compared with pack ice (0.2-1.8 seals/km²) (Chambellant 2010), and density estimates vary considerably from year-to-year (0.5-1.6 seals/km²) (Smith & Stirling 1975, Breton-Provencher 1979, Lunn *et al.* 1997, Chambellant 2010). An abundance estimate that included Svalbard's west and north coast suggested a population of 7,585 seals (95% CI: 6,332-9,085) (Krafft *et al.* 2006). In Svalbard, densities of ringed seals in the fjords range from 0.2 to 8.0 seals/km² (Krafft *et al.* 2006, Krafft *et al.* 2007) with large year to year variability due to sea ice cover. Overall ringed seals occur at lower densities in multi-year ice of the high Arctic compared with their preferred habitat in annual ice areas (Kingsley *et al.* 1985) probably because productivity is

lower in the thicker ice and it is more difficult to maintain breathing holes in or between multi-year ice floes.

The global population size of bearded seals is unknown, but it has been estimated to be conservatively 438,000 (Cameron *et al.* 2010), at least 500,000 individuals (Kovacs & Lowry 2008), or even up to 750,000 (Chapskii 1966, Potelov 1975, Burns 1981, Cleator 1996). There are two putative subspecies of bearded seal, *Erignathus barbatus barbatus* and *E. b. nauticus* (Kovacs 2009), with the Atlantic subspecies *barbatus* occurring from the central Canadian Arctic east to the central Eurasian Arctic (Laptev Sea) and the Pacific subspecies *nauticus* occurring from the Laptev Sea east to the central Canadian Arctic, including the Sea of Okhotsk (Rice 1998). Rough

estimates for bearded seal population size in all regions are over 50 years out of date, but range from ~ 300,000 animals in the Bering-Chukchi Seas, to about 200,000 animals in Canadian waters, to 250,000 in the Sea of Okhotsk. Numbers in the Atlantic and Russian regions are unknown (Cleator 1996, Fedoseev 2000). Lunn *et al.* (1997) estimated approximately 12,290 (SE = 2,520) bearded seals (or 0.122 seals/km² of sea ice) in western Hudson Bay.

Nineteen subpopulations of polar bears occur throughout the circumpolar Arctic (Obbard *et al.* 2010, Vongraven *et al.* 2012). The global population size of polar bears is 20,000 to 25,000 animals (Obbard *et al.* 2010). Genetic analysis indicates that there is considerable gene flow between some subpopulations, though others are relatively discrete (Paetkau *et al.* 1999). The largest polar bear subpopulation estimates are in Davis Strait with 2,158 (95% CI: 1,833-2,542) (Peacock *et al.* 2013), the Barents Sea with 2,650 animals (95% CI: 1,900-3,600) (Aars *et al.* 2009) and Foxe Basin with 2,578 animals (95% CI: 2,088-3,182) (Obbard *et al.* 2010). There are critical knowledge gaps about subpopulation sizes in E Greenland, the Russian Kara and Laptev seas, the Arctic Basin and the Chukchi Sea (Vongraven *et al.* 2012).

The size of the Pacific walrus population, which is managed as a single panmictic population (USFWS 2010), has never been known with much precision. Based on large sustained harvests in the 18th and 19th centuries, Fay (1982) speculated that the pre-exploitation population was a minimum of 200,000 animals. A recent survey conducted in the Bering Sea in 2006, estimated a population size of 129,000 individuals (95% CI: 55,000 to 507,000) in a portion of the range, or about half the potential walrus habitat (Speckman *et al.* 2011). The total population size of Atlantic walrus is thought to be about 20,000 animals (Born *et al.* 1995, Witting & Born 2005, COSEWIC 2006, Lydersen *et al.* 2008, NAMMCO 2009), comprising at least nine separate stocks: Foxe Basin, SE Hudson Bay, N Hudson Bay-Hudson Strait-N Labrador-SE Baffin Island-Central W Greenland, N Baffin Bay, W Jones Sound, Penny Strait-Lancaster Sound, E Greenland, Svalbard-Franz Josef Land, and Kara Sea-S Barents Sea-Novaya Zemlya. Walrus in the Laptev Sea are considered a separate stock based on geographical separation (Belikov & Boltunov 2005) and are estimated to number 3,000-5,000 animals.

Spotted seal range-wide abundance is poorly known. Boveng *et al.* (2009) concluded that there are likely to be at least 100,000 spotted seals in the Bering Sea (including the seasonal inhabitants of the Chukchi Sea), 100,000 in the Sea of Okhotsk and about 3,300 in the Yellow Sea and Sea of Japan. An aerial survey of a large portion of the breeding area of the eastern and central Bering Sea resulted in an estimate of 145,700 (95% CI: 96,893-331,700) (P. Boveng, pers. com.). Mizuno *et al.* (2002) flew aerial line-transect surveys over pack ice in parts of the southern Okhotsk Sea in March 2000 and estimated 13,653 spotted seals in a 25,000 km² region.

Other estimates, reviewed by Boveng *et al.* (2009), are mostly outdated or unreliable due to weak or undocumented methodology or insufficient coverage.

Ribbon seal range-wide population size is poorly known. An aerial survey in 2007 of a breeding area in the eastern and central Bering Sea resulted in an estimate of about 62,478 (95% CI: 31,000 – 218,970) (P. Boveng, pers. com.). Based on that survey and historical estimates of relative numbers of ribbon seals in the eastern and western Bering Sea and the Sea of Okhotsk, Boveng *et al.* (2008) concluded that there are likely to be at least 200,000 ribbon seals. Other estimates include that from Burns (1981) who estimated the worldwide population of ribbon seals at 240,000 in the mid-1970s, with 90,000-100,000 animals in the Bering Sea and 140,000 animals in the Sea of Okhotsk. Fedoseev (2002) also reported an estimate of 120,000 to 140,000 animals for the Bering Sea in 1987 and estimated between 200,000 (1968-1974) and 630,000 (1988-1990) for the Sea of Okhotsk. Most of these historical estimates are of doubtful reliability because the methods were weak or undocumented. There are two main breeding areas for ribbon seals, one in the Sea of Okhotsk and one in the Bering Sea, but thus far there is no evidence on which to base a separation into distinct populations.

Harp seals are the most abundant pinniped species in the Northern Hemisphere (Kovacs 2008a) and worldwide number approximately 8 million. Three subpopulations of harp seals are recognized associated with the three whelping areas: (1) Labrador and Newfoundland coasts and in the Gulf of St Lawrence, (2) E Greenland (north of Jan Mayen), and (3) the White Sea (Lavigne & Kovacs 1988). Pup production at all breeding sites combined is at least 1.4 million pups per year (Potelov *et al.* 2003, Stenson *et al.* 2003, Haug *et al.* 2006). The NW Atlantic stock of harp seal numbers approximately 5.9 million animals (DFO 2005). The breeding group in E Greenland was estimated at 750,000 animals in 2008 (ICES 2008). The White Sea breeding group was estimated to be 1.8 million animals in 2000 (Potelov *et al.* 2003).

Hooded seal population size in the NW Atlantic has been estimated to be 592,000 individuals based on pup counts in 2005 (116,900 pups born) (Waring *et al.* 2005). In the NE Atlantic, hooded seal pup production in the Greenland Sea stock (West ice, near Jan Mayen) was 15,250 pups in 2005, and the stock size was estimated to be 82,000 animals in 2007 (Øigård & Haug 2007, ICES 2008, Salberg *et al.* 2008).

Few data are available on Arctic-specific population abundance for the 24 other marine mammal species that seasonally occur in the low or high Arctic. This is because the Arctic comprises only part of these species total seasonal range (e.g. humpback whales migrate to northern areas but not all enter the Arctic in summer), and worldwide or subpopulation abundance is generally assessed outside the Arctic.

Data are available for population sizes of large baleen whales in Baffin Bay due to a local subsistence harvest for these species in Greenland. In 2007, a fully corrected estimate of 3,272 (95% CI: 1,300-8,233) humpback whales was obtained for the coast of W Greenland in summer (Heide-Jørgensen *et al.* 2012). Furthermore, that same year fully corrected estimates of 16,609 (95% CI: 7,172-38,461) common minke whales and 4,468 fin whales (95% CI: 1,343-14,871) were obtained for the same area (Heide-Jørgensen *et al.* 2010c, 2010d).

In the northeastern Bering Sea (primarily within low Arctic waters) Friday *et al.* (2012) estimated cetacean abundance in 1999 and 2002 as follows: 2,729 (95% CI: 1,348-5,527) and 257 (95% CI: 84-789) fin whales, 1,048 (95% CI: 474-2,319) and 47 (95% CI: 10-215) minke whales, 12,486 (95% CI: 5,511-28,289) and 14,597 (95% CI: 8,387-25,403) Dall's porpoise, and 540 (95% CI: 185-1,580) and 87 (95% CI: 16-482) harbor porpoise. Friday *et al.* (2012) estimated 28 (95% CI: 6-130) humpback whales in the same area in 2002. There was considerable variability in estimates across years. Gray whales are commonly seen in the northern Bering and Chukchi Sea (Moore *et al.* 2000), but the portion of the overall Eastern North Pacific population (numbering ~ 21,000; Punt & Wade 2010) using Arctic waters is unknown. Bradford (2011) estimated that 140 gray whales were associated with the Sakhalin feeding ground in the Sea of Okhotsk between 1997 and 2007.

Northern fur seals in the Bering Sea constitute at least 30% of the worldwide population. The most recent estimate for the number of fur seals in this area, based on pup counts from 2008 on Sea Lion Rock, St. Paul and St. George Islands, and from 2007 on Bogoslof Island, is 653,171 seals (Allen & Angliss 2011). The Bering Sea stock of harbor seals in the low Arctic, specifically animals hauling out on the Pribilof Islands, numbers about 232 animals (Allen & Angliss 2011), and numbers in Greenland are low. There are occasional sightings of sea otters on St. George Island but no established population exists (Riedman & Estes 1990). Population size of Steller sea lions in the sub-Arctic Sea of Okhotsk is approximately 5,000 individuals (Burkanov *et al.* 2011), while northern fur seals in the Sea of Okhotsk (rookery on Tuleny Island/Robben Island) number about 100,000 animals (Ream & Burkanov 2006).

3.6.2.2. Trends

Detailed estimates of trends in population size for Arctic marine mammals are rare, and in several cases data are completely absent. Some populations are assumed to be stable or increasing. However, for other populations, given known harvest rates and/or associated population viability analyses, it is suspected that populations are declining, but the rate of decline is unknown. Here we present available data on population trends documented for the 11 ice-associated Arctic marine mammal species.

Trends in abundance are unavailable for most beluga subpopulations due to a lack of data or outdated surveys. Of the beluga subpopulations that have been assessed, the Bristol Bay subpopulation increased at 4.8%/year (95% CI: 2.1%-7.5%) between 1993 and 2005 (Lowry *et al.* 2008). Three subpopulations of belugas are known to be declining: Cook Inlet at -1.1% per year (SE 1.1) (Hobbs *et al.* 2011), the eastern Hudson Bay subpopulation which has declined by almost 50% since 1985 (Bourdages *et al.* 2002, Gosselin *et al.* 2009), and the White Sea subpopulation (Burdin *et al.* 2009). Although the abundance of St. Lawrence Estuary beluga has decreased from 7,800 (SE = 600) in 1866 to approximately 1,000 animals in 1985 (recent estimate 1,100 in 2006) due to overhunting, they have remained stable during the 30 years of protection from hunting (Hammill *et al.* 2007). No belugas were sighted on the latest survey of the endangered Ungava Bay subpopulation, which numbers < 50 whales (Gosselin *et al.* 2009). In some areas, specific winter aggregations of whales are surveyed to provide management advice for subsistence harvests (Heide-Jørgensen & Aquarone 2002, Heide-Jørgensen *et al.* 2010a). In W Greenland, numbers of wintering belugas have increased during the 21st century (Heide-Jørgensen *et al.* 2010a), but these aggregations constitute an unknown fraction of subpopulations (Heide-Jørgensen *et al.* 2003).

Although good population estimates are available for most narwhal stocks (Heide-Jørgensen *et al.* 2010b, Richard *et al.* 2010), they cannot be used for trends in abundance because of a lack of long-term monitoring or changes in survey methods making estimates incomparable. Surveys in central W Greenland in late winter are considered important for estimating trends in narwhals, but those surveys cover unknown proportions of whales from different summering subpopulations from W Greenland and Canada (Heide-Jørgensen *et al.* 2010b).

The Bering-Chukchi-Beaufort population of bowhead whales has increased at a rate of 3% per year since the late 1970s (George *et al.* 2004). The bowhead whales in Disko Bay, W Greenland, have increased at a rate of approximately 5% per year since 2000 (Heide-Jørgensen *et al.* 2007, Wiig *et al.* 2011) and comprise a spring aggregation which is part of the eastern Canada-W Greenland population. Trends in the subpopulations inhabiting the Svalbard-Barents Sea and the Sea of Okhotsk are unknown.

» *It seemed that from 1964 onward the bowhead whales seemed to be increasing annually in numbers in our waters. ... when you look at the year 1964 and compare it to today there are so many bowhead whales close by. We even on occasion see bowhead whales at the floe edge during the months of May and June between Baffin Island and Igloodik. ... before that time you wouldn't dream of ever seeing a bowhead whale at the floe edge.*

(Simon Iyyiriaq of Igloodik quoted in NWMB 2000).

The trends in global population abundance and regional population abundance for ringed seals and bearded seals are unknown. Ringed seal density estimates in western Hudson Bay from nine aerial surveys over the past 16 years showed an approximate 10-year cycle with a maximum density of 1.22 seals/km² of ice in 1995, to 0.45 in 1999, to 0.92 in 2007, to a minimum of 0.28 in 2009, followed by an increase to 0.73 in 2010 (Ferguson & Young 2011).

Among the 19 polar bear subpopulations, seven are assessed as declining (Baffin Bay, Chukchi Sea, Kane Basin, Lancaster Sound, Norwegian Bay, S Beaufort Sea, W Hudson Bay), four are considered stable (Davis Strait, Gulf of Boothia, N Beaufort Sea, S Hudson Bay) and one is considered to be increasing (M'Clintock Channel). There are not enough data to determine trend for the other seven subpopulations (Arctic Basin, Barents Sea, E Greenland, Foxe Basin, Kara Sea, Laptev Sea, Viscount Melville Sound) (Obbard *et al.* 2010). Trends are assessed by the IUCN/SSC Polar Bear Specialist Group, however the Canadian Polar Bear Technical Committee (PBTC) also considers the status of polar bear populations in Canada separately. Only a few polar bear populations are studied frequently enough for assessing trends (Stirling *et al.* 1977, Amstrup *et al.* 1986, Aars *et al.* 2009, Regehr *et al.* 2010). The W Hudson Bay and S Beaufort Sea populations have the best time series, and both of these populations have been determined to be declining. Regehr *et al.* (2007) documented a decline in W Hudson Bay of about 22% from 1,200 bears in 1987 to about 935 in 2004. This change was linked to the decline in reproduction and survival of young and very old bears due to climate warming (Stirling *et al.* 1999, Stirling & Parkinson 2006) in combination with harvesting at unsustainable levels. In the S Beaufort Sea, an intensive mark-recapture study conducted from 2001 to 2006 indicated that the subpopulation was 1,526 (95% CI: 1,211-1,841) polar bears in 2006 (Regehr *et al.* 2006). Further analyses indicated that survival and breeding during this period were affected by sea ice conditions, and that population growth rate was strongly negative in years with long ice-free seasons (Hunter *et al.* 2010, Regehr *et al.* 2010). Thus, the S Beaufort Sea population is currently considered to be declining due to sea ice loss.

» *I think the reason why the bears come closer and closer is that the sea does not freeze over any more during the winter time of year. Therefore, the bears come closer and closer. Since our sea has begun to freeze late, some bears have become very thin. When the ice forms early, the bears we catch are usually fat and taste good. Since the late 1990s, due to the fact that the sea freezes late, almost all of the bears that we have caught have not had any fat on them. The bears we caught this year were like that too. They have no blubber on them and they are not fat. If you disregard the pingajoqqat [mothers with two cubs; author's note] that we caught, which were a little bit plump. ... The rest of the bears that we catch are thin. That is because the sea out there does not freeze over – The difference is noticeable.*

(A hunter from Savissivik quoted in Born *et al.* 2011).

The trend in global population abundance for walrus is unknown. Regionally, where walrus population size estimates exist there are no or few previous reliable estimates from which a trend can be calculated. Modeling and simulation studies indicate that populations in W Greenland and the North Water have been declining due to over-exploitation, while the population in E Greenland has perhaps been increasing (Witting & Born 2005, NAMMCO 2009). The number of walruses summering in Svalbard increased from a few hundred animals to 2,629 (CI: 2,318-2,998) between the 1980s and 2006 (Lydersen *et al.* 2008). The population size and trends in Franz Josef Land are unknown, however, the population has been protected from hunting since the 1950s (Born *et al.* 1995). Large-scale commercial harvests of Pacific walrus reduced the population to 50,000-100,000 animals in the mid-1950s (Fay *et al.* 1997). This population increased rapidly during the 1960s and 1970s in response to harvest regulations limiting the take of females (Fay *et al.* 1997). Between 1975 and 1990, aerial surveys produced population estimates ranging from 201,039 to 290,000 (Udevitz *et al.* 2001); the most recent estimate from 2006 covering a portion of the range is 129,000 (95% CI: 55,000 to 507,000) (Speckman *et al.* 2011).

There are no available data for estimating trends for spotted seal and ribbon seals. The few data available lack precision.

A population model was used to examine changes in the size of the NW Atlantic harp seal population between 1952 and 2010 and resulted in an estimated exponential population growth to a total population in 2008 of 8.0 million (95% CI: 6.8-9.3 million) animals (Hammill & Stenson 2011). Harp seal pup production estimates in the White Sea stock have experienced significant declines since 2004, dropping from over 300,000 pups to approximately 150,000 pups. The reasons for the declines are not known, but changes in sea ice cover, industrial activity and hunting have been suggested (Chernook & Boltnev 2008, Chernook *et al.* 2008, Vorontsova *et al.* 2008, Zabavnikov *et al.* 2008). Recent model runs by ICES (2008) have confirmed that the population of harp seals in E Greenland may have increased in size from its earlier depleted state since ca. 1970, and it has been predicted that the population could continue to increase under the current harvest regime, which involves very small annual removals (Øigård *et al.* 2010). There are concerns that both female fecundity and neonatal survival have been reduced in recent decades in the Canadian harp seal subpopulations both in the Gulf and on the Front due to declines in sea ice stability and thickness (Bajzak *et al.* 2011, Johnston *et al.* 2012).

There was a moderate increase in hooded seal pup production and population size in the NW Atlantic between the mid-1980s and 2005 (Stenson *et al.* 1997, Hammill & Stenson 2007). In contrast, the NE Atlantic population of hooded seals has declined by 85-90 % over the last 40-60 years (Kovacs 2008b, Øigård *et al.* 2010). The cause of the decline is unknown, but it is likely related to

changing ice conditions and perhaps also overharvesting. Hunting has recently been prohibited (Kovacs 2008b). Øigård *et al.* (2010) suggest that the population may now be reaching a stable lower level.

Few population trends are available for marine mammals that visit low Arctic regions seasonally. Those that are available, however, are mostly positive. Humpback whales off the coast of W Greenland have been increasing 9.4% per year (SE = 0.01) since 1984 (Larsen & Hammond 2004, Heide-Jørgensen *et al.* 2012). The population of fin whales that occurs off the west coast of Greenland has also likely been increasing, but the rate is unknown due to differences in survey methodology and correction factors (Heide-Jørgensen *et al.* 2010d). The aggregation of gray whales using the summer feeding ground in the Sea of Okhotsk has been growing at about 3% per year since 1997 (Bradford *et al.* 2008), and the population of gray whales summering in the Bering and Chukchi Seas was growing in the 1990s but may have reached carrying capacity (Rugh *et al.* 2005).

During 1998–2006, Northern fur seal pup production on St. Paul Island declined by 6.1% per year (SE = 0.45%) and by 3.4% per year (SE = 0.60%) on St. George Island (Pribilofs) (Allen & Angliss 2011). Steller sea lions in Sea of Okhotsk have been slowly increasing since the 1970s (Burkanov & Loughlin 2005, Burkanov *et al.* 2011), and northern fur seals in Sea of Okhotsk have also been increasing (Ream & Burkanov 2006).

3.6.2.3. Causes and prospects

Loss of sea ice

The loss of Arctic sea ice is the greatest threat to Arctic marine mammals, particularly to ice-associated pinnipeds and polar bears. Springtime is an especially important period for several species because it coincides with critical reproduction periods or important feeding opportunities. Sea ice declines over the past several decades have been clearly documented (IPCC 2007, Perovich & Richter-Menge 2009, Stroeve *et al.* 2012), and the latest projections indicate an ice-free high Arctic in summer within three decades (AMAP 2011, Wang & Overland 2012). Thus, further habitat loss and habitat degradation for all ice-associated species can be expected.

For pinnipeds, reduced sea ice cover and snow cover will have negative impacts on pupping, molting and resting platform availability in many areas (IPCC 2007, Hezel *et al.* 2012). Declines in reproduction and survival of ringed seals have been linked to variations in their sea ice habitat including responses to early or late ice break-up in spring, and relatively heavy or light ice conditions (Smith 1987, Kingsley & Byers 1998, Harwood *et al.* 2000). Years with low snow cover (Ferguson *et al.* 2005) and unusually warm weather or rain events in the spring (Stirling & Smith 2004) have also resulted in reduced survival. In late winter and early spring, ringed seals give birth and nurse their pups in subnivean (under snow) lairs that they excavate above the breathing holes

(Smith & Stirling 1975). While some pups are born on pack ice (Wiig *et al.* 1999), landfast ice with sufficient snow cover is required to build lairs, which provide relative safety from polar bear predation (McLaren 1958, Burns 1970, Hammill & Smith 1991). Warmer conditions or rain events can destroy lairs or make them impossible to build (Kelly *et al.* 2010).

The primary threat to walrus is also considered to be the reduction of summer sea ice, especially for the Pacific population. In spring, ice is important for breeding (Fay *et al.* 1984), giving birth and nursing and care of young (Fay 1982). In the Bering and Chukchi Seas, floating pack ice serves as a substrate for resting between shallow feeding bouts (Fay 1982, Ray *et al.* 2006). In the Chukchi Sea, reduction of summer sea ice as a resting and feeding platform poses the greatest risk to females and calves. Sea ice provides access to offshore feeding areas, isolation from terrestrial predators and hunting pressure (Fay 1982, Kochnev 2004, Ovsyanikov *et al.* 2007), and protection from stormy seas (Fay 1982). On the east coast of Greenland, reduced sea ice cover may actually increase feeding opportunities for walrus (Born 2005), but this is not the situation for Atlantic walrus in other areas such as the Barents Sea-Svalbard subpopulation (Kovacs *et al.* 2011). More open water tends to support more pelagic and less benthic biomass, which could affect walrus prey (Moline *et al.* 2008).

» *Most bears are thin. They are getting thin. ... Yes...well, some of the ones that I have caught, they have been like that. They have started to be like that in recent years. ... They weren't like that in the past. They used to be very fat in the past. Of course it is not each and every one that is like that. But for most of them, their layer of blubber has got thinner. Some of them have almost no blubber. This winter I caught yet another one without very much blubber.*

(Hunter from Nuussuaq, Greenland, quoted in Born *et al.* 2011).

Like the true Arctic seals, the primary threat to Pacific and Atlantic low-Arctic ice seals is also seasonal sea ice loss in the late winter and spring. In the Pacific, ribbon and spotted seals rely on pack ice at the southern limit of the ice extent in the Bering Sea and Sea of Okhotsk for whelping and rearing pups (Boveng *et al.* 2008). Decreased availability of stable platforms for adults to complete their molt out of the water may also lower survival. In the Atlantic, where sea ice is declining rapidly, harp seals and hooded seals require seasonal sea ice cover for whelping, lactation, resting and molting, for short but specific periods in spring (Johnston *et al.* 2005, Kovacs & Lydersen 2008, Laidre *et al.* 2008a). Johnston *et al.* (2012) revealed negative correlations between both ice cover and the NAO index and harp seal mortality in the Gulf of St. Lawrence, indicating that lighter ice cover and lower NAO values result in higher mortality. They also assessed the long-term negative trends in sea ice cover in the breeding regions of harp seals across the entire North Atlantic during 1979 through 2011 and found that sea ice cover in all harp seal breeding regions has

been declining by as much as 6% per decade. For all ice-associated pinnipeds, the quality of the ice habitat (i.e. thickness) and the age and the duration of sea ice cover also play an important role during the pupping season (Friedlaender *et al.* 2010, Bajzak *et al.* 2011). Location of traditional sites might also be an important factor, though Rosing-Asvid's (2008) finding suggests that some shifting of whelping locales might be possible for these seasonally dispersed populations.

Declining sea ice habitat has been broadly recognized as the most significant threat to polar bears (Derocher *et al.* 2004, Aars *et al.* 2006, Amstrup *et al.* 2006, Wiig *et al.* 2008, Durner *et al.* 2009, Obbard *et al.* 2010, Stirling & Derocher 2012) and has been associated with declines in population abundance (Regehr *et al.* 2007), declines in survival (Regehr *et al.* 2010, Peacock *et al.* 2012), declines in body condition (Stirling *et al.* 1999, Rode *et al.* 2010, Rode *et al.* 2012), declines in recruitment (Rode *et al.* 2010) and increased swimming (Pagano *et al.* 2012).

It is less clear what the impacts of sea ice loss will be on Arctic cetacean populations. Sea ice loss opens up new habitat (Heide-Jørgensen *et al.* 2011a) and may increase the duration of the production season allowing for increased foraging opportunities for baleen whales (Moore & Laidre 2006, Laidre *et al.* 2010). Annual sea ice cover exerts broad-scale control on energy flux, levels of biological production (Laidre *et al.* 2008a) and ultimately survival and reproduction of predators at the top of the food chain. The primary production bloom is the main food source for zooplankton (secondary production), which play a critical role in the transfer of energy between primary producers and secondary consumers like forage fish. Indirect changes in the ecosystem that occur with the loss of sea ice may have negative impacts at the population level (Laidre *et al.* 2008a, Kovacs *et al.* 2011), including increased predation (Higdon *et al.* 2012, Ferguson *et al.* 2012a, 2012b), disease (Burek *et al.* 2008) and competition from temperate species extending their range into high latitudes (Higdon & Ferguson 2011). Additionally, irregular freeze-up patterns may have negative effects for ice-associated Arctic cetaceans like belugas and narwhals, which are susceptible to sea ice entrapments if ice conditions change rapidly (Kleinenberg *et al.* 1964, Laidre & Heide-Jørgensen 2005, Laidre *et al.* 2011). Increasing frequency and intensity of storm events might also have greater impacts on Arctic cetaceans if they no longer have ice available to them; this is likely particularly important for juvenile animals.

The impacts of sea ice loss on marine mammals that seasonally use the Arctic have generally not been documented but are likely positive given that sea ice loss will allow southern species to extend their northern range limits and/or occupy previously ice-covered areas (e.g. Moore & Huntington 2008, Heide-Jørgensen *et al.* 2011b). Expected shifts in species distribution could also bring new predators such as killer whales (Higdon & Ferguson 2009) or competition with sub-Arctic species (Higdon & Ferguson 2011).

Harvest

Arctic marine mammals are harvested by indigenous peoples for nutritional, cultural and economic reasons. While some subsistence harvests are closely monitored by local, national, governmental and international organizations (e.g. Alaska Beluga Whale Committee, Nunavut Wildlife Management Board, U.S. Fish and Wildlife Service, North Atlantic Marine Mammal Commission, International Whaling Commission) and many hunts are sustainable, overharvest is a problem for some populations, especially where population abundance and harvest rates are unknown or harvests are not well regulated. This is still the case for some subpopulations of polar bears even when both harvest rates and population sizes are known (Obbard *et al.* 2010). Overharvest has been demonstrated to cause declines for several local or small isolated populations of Arctic marine mammals (e.g. West Greenland walrus) (Taylor *et al.* 2002, Witting & Born 2005, Taylor *et al.* 2008, Hobbs *et al.* 2011, Peacock *et al.* 2011). Sport hunting for polar bears only occurs in Canada; numbers of sport-hunted bears are set by the communities from the overall quota assigned to each community (Vongraven & Peacock 2011).

Anthropogenic activities

Besides hunting, known or potential anthropogenic threats include industrial activities such as oil and gas exploration and development (seismic exploration, drilling), commercial shipping and increased tourism, northward expansion of fisheries (with possible implications for bycatch, competition and resource depletion; e.g. narwhal and Greenland halibut), incidental mortality and serious injury caused by entanglement in fishing gear and ship strikes (e.g. bowhead whales), hydroelectric development (e.g. beluga whales in Hudson Bay), concomitant increases in underwater noise (Moore *et al.* 2012b), and industrial and urban pollution (Laidre *et al.* 2008a). The ringed seal subspecies that live in freshwater lakes are also vulnerable to manipulation of water levels, recreational snow machine operation, net-fishing and poaching, bycatch, boating, tourism and development, predation by terrestrial mammals, and in a few cases industrial pollution (Kovacs *et al.* 2012).

Pollution and disease

Pollution, emerging parasites and disease are also issues for Arctic marine mammals. Top-level carnivores accumulate heavy metals, such as mercury, and organochlorine contaminants at relatively high concentrations due to bio-accumulation. After two decades of monitoring, E Greenland polar bears have been determined to be among the most polluted animals on the planet (Sonne *et al.* 2012). In general, organochlorine contaminant concentrations are highest in marine organisms (Noyes *et al.* 2009). However, population level effects are difficult to quantify given uncertainty in population trends and multiple sources of variability. Due to the geographic distances many of these species travel, and the influx of southern species, diseases and parasites may be an important issue in the future, but specific risks are uncertain at present.

Prospects

Ideally, insight on the effects of climate change on Arctic marine mammal species and populations is collected through long-term monitoring of basic population metrics, life history or behavior in combination with environmental time series and quantitative modeling (e.g. Regehr *et al.* 2010, Rode *et al.* 2010, Jay *et al.* 2011). Unfortunately there are very few long-term studies. The few studies that have been conducted over a span of several decades (e.g. W Hudson Bay and S Beaufort Sea polar bears) have been critical for determining the impacts of climate change. To date, no studies have been designed or implemented that attempt to distinguish between the effects of climate change and the effects of increasing anthropogenic activity in the Arctic for marine mammals.

Unusual or rare phenological or behavioral observations that can be documented in concert with altered environmental conditions are insightful. Examples include extremely long polar bear swims (Durner *et al.* 2009, Pagano *et al.* 2012), drowning polar bears (Monnett & Gleason 2006), polar bear cannibalism (Amstrup *et al.* 2006, Stirling & Ross 2011), abandoned walrus calves (Cooper *et al.* 2006) and ice entrapments of narwhals in unusual areas (Laidre *et al.* 2011). However, because these are rarely observed events, it is difficult to quantitatively link them to the population level.

Several studies have made predictions about the possible impacts on population abundance, survival and reproduction of Arctic marine mammals from sea ice loss. Predictive modeling of the future global distribution and abundance of polar bears forecasts declines in abundance, survival and reproduction (Durner *et al.* 2009, Amstrup *et al.* 2010, Hunter *et al.* 2010, Molnár *et al.* 2010). In general, large future reductions in most subpopulations of polar bears are expected. Predictive Bayesian network models for Pacific walrus, integrating potential effects of changing environmental conditions and anthropogenic stressors, demonstrate a clear future trend of worsening conditions for the subspecies (Jay *et al.* 2011). Few predictive models exist for Arctic cetaceans, but in general it is believed impacts will be species-specific depending on how well species adapt to changing food webs and sea ice regimes (Laidre *et al.* 2008a, 2011). Increased efforts combining population and habitat modeling are needed to predict population persistence in the face of climate change (Peacock *et al.* 2011).

Conflicts between conclusions reached by scientific methods compared to those by traditional ecological knowledge have been increasing in polar bear management in Canada (Peacock *et al.* 2011). In recent years, some permits for management-oriented research on polar bears were denied by the Government of Nunavut, local hunting and trapping organizations in the Northwest Territories, and by Makivik Corporation in Quebec. Furthermore, in a 2009 resolution, Inuit Tapiriit Kanatami opposed the capture of polar bears throughout

Canada. This prevented scientists from applying physical markings to polar bears in order to estimate population sizes (Stirling *et al.* 1999, Peacock *et al.* 2011). Less-invasive and non-invasive protocols for biological sampling and monitoring are increasing, such as remote biopsy darting to collect fat and tissue samples and aerial surveys to estimate abundance.

Overall, increased monitoring is needed to fill large gaps in knowledge about population sizes and trends for Arctic marine mammals. Several circumpolar Arctic marine mammal monitoring plans have been drafted by groups of experts, but these plans have not yet been implemented largely due to lack of dedicated funding (Kovacs 2008c, Laidre *et al.* 2008b, Simpkins *et al.* 2009, Vongraven *et al.* 2012). Dedicated monitoring efforts across several spatial and temporal scales, although costly and difficult, are critical for quantifying future impacts.

3.7. CONCLUSIONS AND RECOMMENDATIONS

3.7.1. Valuable areas and productivity hotspots

Three types of habitat are particularly valuable due to their unique biological richness and large-scale influence on Arctic ecosystems: caribou calving grounds, coastal zones and margins of the sea ice-pack.

Migratory tundra caribou calving grounds require special attention. Caribou choose these fairly restricted areas because of high food quality and relatively low predation risk, and thereby maximize the survival and vigor of calves. Human activities and infrastructure (e.g. aircraft flight paths, roads, off-road vehicle use, pipelines) should be prohibited or strongly regulated in these landscapes during the calving seasons when the activities can readily disrupt the optimum bonding and behavior of cows and calves with negative consequences for calf recruitment. Calving grounds are site-specific by herd, though they do shift somewhat over time. Many are currently undergoing some mineral and hydrocarbon exploration and road development (e.g. Beverly) or are under such threat (e.g. Bathurst, Porcupine).

Coastal zones, especially over the relatively shallow continental shelf and banks, are particularly productive marine areas. Along coastlines, the mixing of marine water with nutrient-rich fresh water, from land-based drainages and melting sea ice, enhances productivity and attracts large concentrations of marine mammals. Migratory marine mammals rely on this spatially-concentrated ocean productivity for foraging opportunities. Deltas and offshore plumes from the major rivers (notably the Mackenzie and Lena) are heavily used feeding areas. Coastlines and near-shore ice and barrier islands are particularly important for polar bears, combining high-value habitats for reproduction and resting with relatively high marine productivity

especially in spring and summer. Coastal zones are particularly at risk because expanding human activities (e.g. shipping, fishing, oil and gas developments, transportation infrastructure and settlements) are and will be concentrated in and beside these zones of high ecological productivity and easier access to resources.

Sea ice margins are also particularly productive marine areas that attract numerous marine and some terrestrial mammals. They include the geographically widespread ice margins of the Bering/Chukchi Seas, Baffin Bay, Davis Strait, E Greenland and the Barents Sea. These zones change position somewhat between years as patterns of ice melt change, and are likely to shift systematically in response to changing climate. Nevertheless, they require particular attention because of their importance to many marine mammals.

In winter, a particular set of sea ice margins is found at polynyas or flaw leads, where substantial areas of water remain open or only occasionally frozen due to particular combinations of wind and currents. These are important habitats for winter-resident Arctic marine and terrestrial mammals as well as seabirds. They are seasonally delimited habitats, requiring particular conservation attention in winter. Key examples of polynyas include North Water (N Baffin Bay), St. Lawrence Island (Bering Sea) and North East Water (NE coast Greenland); and of flaw leads include NE Chukchi Sea, Cape Bathurst (Beaufort Sea) and Laptev Sea (see Box 14.2 in Michel, Chapter 14).

While managers need to pay attention to habitats of high ecological value, conservation attention also needs to be focused on biological 'hotspots' that overlap areas of particular interest to oil, gas and mining industries, because of the increased disturbance that is likely to occur in those areas. These tend to be geographically large areas in the exploration phase, leading to site-specific developments.

Four regions appear to be of particular interest to the oil and gas industry at present: Barents Sea, Beaufort-Chukchi Seas, Baffin Bay and E Greenland. These regions deserve particular attention because the exploration, development and production phases of this industry may cause displacement of species from important feeding or breeding habitats, changes in the underwater acoustic environment, impacts to calving and migratory habitats, and potentially direct mortality or changes in vital rates due to collisions, oil spills or contamination. The risks of population declines for both marine and terrestrial mammals can only be addressed, and perhaps mitigated, through environmental assessments (including collection of new data not already available to resource managers); controls on the intensity, timing and structure of exploration and development activities; and dedicated work with local communities to ensure the implementation of cautious management and harvest plans for mammals that might be affected. Given the paucity of data on many Arctic mammal populations, it is difficult to detect population

changes and attribute their cause to either human-induced or natural factors. Therefore, strengthened research and monitoring programs must precede and accompany proposed development activities in Arctic regions.

The global rush for minerals is resulting in many new mine developments in the Arctic. Each potential new mine site requires focused attention to determine its potential direct and indirect impact on terrestrial mammals. Marine mammals may also be impacted by increased shipping and activity in coastal zones, and various other factors resulting from industrial development and its infrastructure. Concerted efforts must be made to forecast the impact of any one development project, as well as the cumulative impacts in a particular region. Environmental impact assessments are a necessary component of our management, but the ability of these assessments to consider multiple scales of potential impacts over both time and space is limited and must be improved. Special attention should be given to the use of new technologies that reduce the extent of infrastructure required (e.g. air ships), and to operational measures that reduce the potential for changing mammal behavior (e.g. proper garbage management, controls on human harvesting of wildlife). Monitoring of Arctic mammals and potential impacts on them must be an integral and funded portion of any developments.

3.7.2. Key knowledge gaps

One major conclusion of this review is that detailed, long-term data on population trends for Arctic mammals are rare. There are no abundance or trend estimates for many key populations and species of marine (e.g. all of the ice-dependent pinniped populations and several polar bear populations) and terrestrial (e.g. Arctic wolf, many lemming populations) mammals. Demographic data are also absent for many species, and if available they are rarely of high quality. This is largely explained by high costs and logistical hurdles of monitoring populations in large and remote areas. Information on population trends is important for natural resource managers to take management actions when populations face single or cumulative impacts, and to measure recovery from any perturbation. Good population monitoring is the first requirement for biodiversity assessment, and our knowledge of the status and trends of Arctic species will remain relatively poor unless we invest more resources into monitoring their numbers and understanding their ecology.

Weather patterns and extreme weather events are prominent limiting factors for Arctic herbivores. Global climate patterns, such as the North Atlantic and Arctic Oscillations, affect seasonal weather patterns and therefore timing and productivity of plant growth over multi-annual and decadal periods. These relationships deserve increased attention including investigations of patterns in a greater diversity of weather-related phenomena that impact mammals (e.g. freezing rain and icing events, thaw-freeze cycles in winter, timing of

snowmelt, timing of snow onset, taiga and tundra wild-fire frequency). Such investigations need to be coupled with long-term studies of how such weather phenomena are affecting demographic parameters in mammals (e.g. over-winter survival and reproductive output in rodents and lagomorphs, conception and calf survival in caribou and muskoxen). Northern community members who are frequently on the land can be employed in recording patterns of weather, especially unusual events, and animal responses (see Huntington, Chapter 19, for discussion of community-based monitoring).

Caribou herd viability, and the ability to monitor herds, depend on a good understanding of locations and temporal use of calving grounds by reproductive and barren cows. For some herds, this information is still unclear, but is crucial when population monitoring depends on calving ground counts. Improved mapping and tracking of calving grounds and the landscapes used by barren cows in the same season will allow more robust population estimation, and improved application of land management guidelines.

Cumulative impacts assessments of multiple direct and indirect anthropogenic activities over space and time need improvement. Given a general lack of predictive models for cumulative impacts assessment, we need new approaches to both detecting negative effects as quickly as possible, and combining effects in decision-making. For caribou, one approach lies in monitoring herd status by sampling individual health status (pregnancy rates, body condition, parasite load and survival) integrated in energy allocation models (Russell *et al.* 2005), coupled with research on relationships between herd status and environmental factors such as weather, snow and fire.

3.7.3. Recommended conservation actions

The most urgent conservation need is a stabilization and reduction of greenhouse gases at the global scale, so that climate change can be slowed and limited in intensity world-wide. Continued increases in greenhouse gas production, mostly outside the Arctic, will exacerbate the ongoing disruption of Arctic ecosystem processes. Climate warming in the Arctic has had the most dramatic effects on snow, ice and water (the cryosphere) (AMAP 2011). These are prominent components of Arctic habitats, and consequently some Arctic mammal populations that are economically and culturally important will be significantly reduced in distribution and abundance. Ice-associated mammals, especially polar bear and pinnipeds, are highly threatened by reductions in duration of the sea ice season and in spatial extent of summer ice. Some populations are at high risk of extirpation within decades. The probability of global extinction of an Arctic mammal species has not been estimated, but appears to be growing with the increasing pace of habitat and ecosystem change.

The variety of legislation, regulations and policies across the circumpolar Arctic needs to be harmonized, ideally

with the assistance of the Arctic Council. Environmental legislation and regulations vary in strength and intensity across jurisdictions. These include: (1) environmental impact assessment for major industrial projects, (2) endangered species protection, (3) harvest management, (4) marine transportation safety, pollution and routing regulations, (5) offshore oil and gas drilling and extraction standards, and (6) identification of responsibility for providing resources for necessary studies before new anthropogenic activities occur. Without such harmonization, the level of environmental risk and consequent negative impact on a population will vary from jurisdiction to jurisdiction and negative impacts in one region will affect other regions. For example, some jurisdictions require substantial environmental impact assessments where the risks to impacted mammal populations are minimized with mitigation measures imposed; other jurisdictions lack a robust assessment process. Trans-boundary populations may experience relatively heavy negative impacts in a jurisdiction with weaker legislation and regulations, despite strong conservation efforts in a jurisdiction with higher environmental standards. The chances of one jurisdiction suffering the consequences of poorer environmental standards in another jurisdiction will continue to increase as development proceeds.

A coordinated mammal population abundance monitoring plan needs to be developed and implemented in the field, with the support of jurisdictions. Strategic attention should be focused on specific combinations of species and region from which most inferences can be drawn. Such a plan needs to build on long-term data sets and requires integration with existing local or national monitoring through the circumpolar Arctic. Particular attention to monitoring in Eurasia is warranted. Such monitoring plans have already been discussed for marine mammals such as belugas, ringed seals and polar bears, but none has actually been fully developed or implemented. Migratory tundra caribou are the subject of an international monitoring effort (CircumArctic Rangifer Monitoring and Assessment Network (CARMA)), but many other species are currently overlooked.

The Circumpolar Biodiversity Monitoring Program (CBMP) is a valuable start to the large task of archiving, reporting and making accessible data on population distribution and abundance for Arctic species. This program needs to be maintained and supported in its goal of better integration with field-based monitoring programs. However, merely tracking population size and demographic parameters is not enough. Monitoring must be designed to test alternative hypotheses about the role of limiting factors (e.g. weather, primary production, disturbance, harvest) on distribution and abundance. Hypotheses explaining past, present and future changes must be set and tested as integral parts of monitoring activities. Maximizing the number of counted populations is not as important as investigating limiting factors in conjunction with following a suite of strategically chosen populations.

In conjunction with abundance monitoring, all user groups need to collaborate in improved monitoring and record keeping of animal harvest levels across jurisdictions, so the sustainability of the total harvest can be assessed for biological populations. Harvest of wildlife is a critical component of human subsistence in the Arctic. Harvest can be a factor in population declines, and science-based harvest management can reduce the risk of population collapse and ensure that subsistence resources are available for future generations. Some components of these harvests are monitored by scientific or co-management committees. However, some are not monitored at all, and many of them involve transboundary populations. Harmonization of harvest reporting and documentation across jurisdictions would improve conservation and management regimes.

Previously depleted populations of harvested Arctic mammal species, and of species currently well below historical levels, need to be recovered wherever possible, especially where there is high likelihood that excessive human harvesting was (e.g. SW Greenland beluga), or still is (e.g. W Greenland walrus), a major factor in reducing abundance. The international moratorium on commercial whaling appears to have facilitated the recovery of some bowhead whale sub-populations (George *et al.* 2004, Heide-Jørgensen *et al.* 2007). Harvest restrictions also can assist caribou population recovery at low density, but the inherently cyclic nature of caribou population abundance confounds the definition of a targeted abundance for recovery and complicates the suite of management actions to facilitate recovery.

There is an urgent need for the establishment of a comprehensive set of protected areas, based on eco-regional representation, biodiversity hotspot analyses, the subsistence economy of northern peoples, and climate change risk assessment. Protected areas with minimal human activity are valuable as ecological benchmarks for understanding ecological processes and as refuge areas during key seasonal periods in the life cycle. If chosen well they can also be relative refuges from the effects of climate change. Northern peoples often harvest mammals in traditional areas related to animal concentrations and accessibility, and precluding other developments to maintain harvests in these areas is a strong rationale for protection. There are a considerable number of land-based protected areas, but relatively few marine protected areas in the Arctic. As climate change is known to be causing environmental changes throughout Arctic ecosystems, some administrative flexibility is needed to ensure that protected areas can be modified or adaptively managed to continue to cover the necessary areas, both now and in the future. Protected areas have spatial but also potentially temporal dimensions. For example, calving grounds of migratory tundra caribou need strong protection during the calving season, but could conceivably sustain some human activities and functioning infrastructure in other seasons.

3.7.4. Other key messages

Many Arctic mammal populations are co-managed between national or sub-national government agencies and indigenous government or community agencies. Knowledge derived from both community experience and scientific studies are expected to contribute to decision making. Smooth decision making has been thwarted in some cases by breakdowns in communication and trust. Solutions are not always clear, but do depend on open-mindedness, honest communication and joint realization that the sustainability of the population is a shared goal of all involved.

Scientific understanding of the direct and indirect effects of climate change and other stressors on Arctic ecosystems is still in its infancy. Society's ability to manage change and implement a valid conservation agenda depends on increased funding for both hypothesis-driven monitoring and basic research into factors driving the distribution and abundance of Arctic mammals.

The Arctic encompasses many of the last wilderness regions on the planet, with species that are marvels of adaptation to difficult conditions, and ingenious human cultures that are intimately linked to harvesting mammals. Conserving the biological and cultural diversity of the Arctic deserves society's utmost efforts and attention in these changing times.

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Appendix 3: www.abds.is/aba-2013-appendix-3