Arctic Ecozone$^+$ status and trends assessment

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Canadian Biodiversity: Ecosystem Status and Trends 2010

Technical Ecozone$^+$ Report

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PREFACE

The Canadian Councils of Resource Ministers developed a Biodiversity Outcomes Framework (Environment Canada, 2006) in 2006 to focus conservation and restoration actions under the *Canadian Biodiversity Strategy* (Federal-Provincial-Territorial Biodiversity Working Group, 1995). *Canadian Biodiversity: Ecosystem Status and Trends 2010* (Federal, Provincial and Territorial Governments of Canada, 2010) was the first report under this framework. It presents 22 key findings that emerged from synthesis and analysis of background technical reports prepared on the status and trends for many cross-cutting national themes (the Technical Thematic Report Series) and for individual terrestrial and marine ecozones^{+} of Canada (the Technical Ecozone^{+} Reports). More than 500 experts participated in data analysis, writing, and review of these foundation documents. Summary reports were also prepared for each terrestrial ecozone^{+} to present the ecozone^{+}-specific evidence related to each of the 22 national key findings (the Evidence for Key Findings Summary Report Series). Together, the full complement of these products constitutes the 2010 Ecosystem Status and Trends Report (ESTR) (Figure 1).

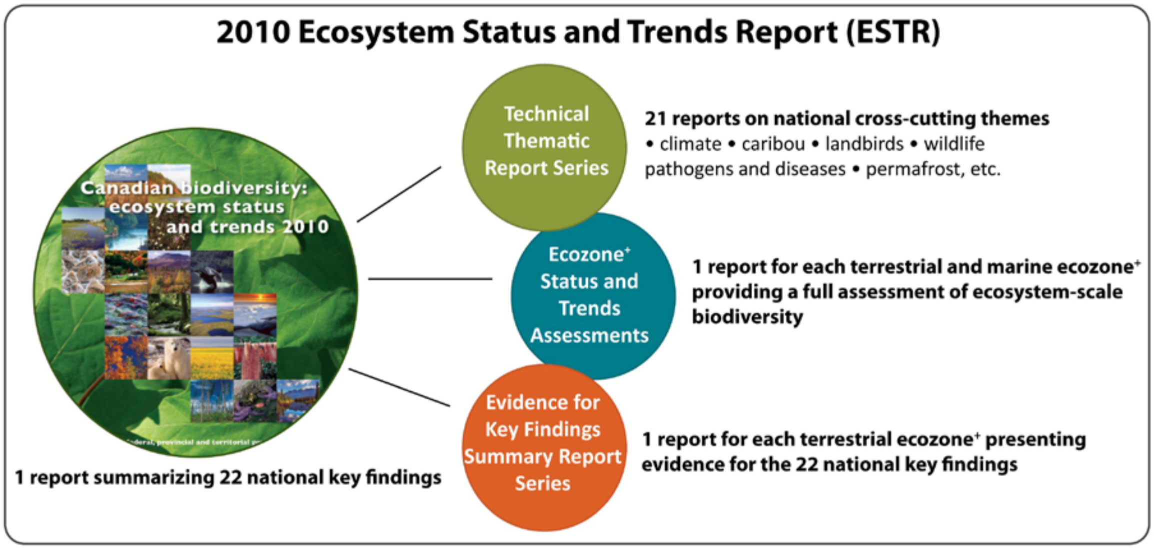

Figure 1. Ecosystem status and trends (ESTR) report series.

This report is the technical report for the Arctic Ecozone^{+} which includes all, or portions of, Yukon, Northwest Territories, Nunavut, Quebec, and Newfoundland and Labrador. A range of authors and reviewers contributed to the report from government, academia, non-governmental, and consulting sectors. No claim is made that the information presented is exhaustive. As in all ESTR products, the time frames over which trends are assessed vary – both because time frames that are meaningful for these diverse aspects of ecosystems vary and because the assessment is based on the best available information, which is over a range of time periods.

Information about the broader ESTR project and its other reporting products is available at: http://www.biodivcanada.ca/default.asp?lang=En&n=83A35E06-1.

Sources of information for this report include research results published in books, scientific journals, and government reports, and records from monitoring and management-related studies. Information from experts is also included (as personal communications) for some subjects when, for example, study results have not yet been published. An important source of information throughout this report is the series of thematic reports prepared for the ESTR. In most cases, these reports are national in scope, with status and trends synthesized both nationally and by ecozone^{+}. The reports include analyses of specific datasets, set in the context of the broader scientific literature. In this report, the thematic reports are summarized and discussed in terms of the Arctic Ecozone^{+}. Key trend analyses are reproduced. Note, however, that analyses in the thematic reports cover a range of dates and have not been updated. This report also draws on summaries and features prepared for the main ESTR report *Canadian biodiversity: Ecosystem status and trends 2010* (Federal, Provincial and Territorial Governments of Canada, 2010) and on Arctic Council's *Arctic Biodiversity Assessment* (CAFF, 2013), a source of information on Canadian Arctic biodiversity and on the circumpolar context of status and trends.

Ecological classification system – ecozones$^+$

A slightly modified version of the Terrestrial Ecozones of Canada, described in the *National Ecological Framework for Canada* (Ecological Stratification Working Group, 1995), provided the ecosystem-based units for all reports related to this project (Figure 2). Modifications from the original framework include: adjustments to terrestrial boundaries to reflect improvements from ground-truthing exercises; the combination of three Arctic ecozones into one; the use of two ecoprovinces – Western Interior Basin and Newfoundland Boreal; the addition of nine marine ecosystem-based units; and, the addition of the Great Lakes as a unit. This modified classification system is referred to as "ecozones$^+$" throughout these reports to avoid confusion with the more familiar "ecozones" of the original framework (Rankin et al., 2011).

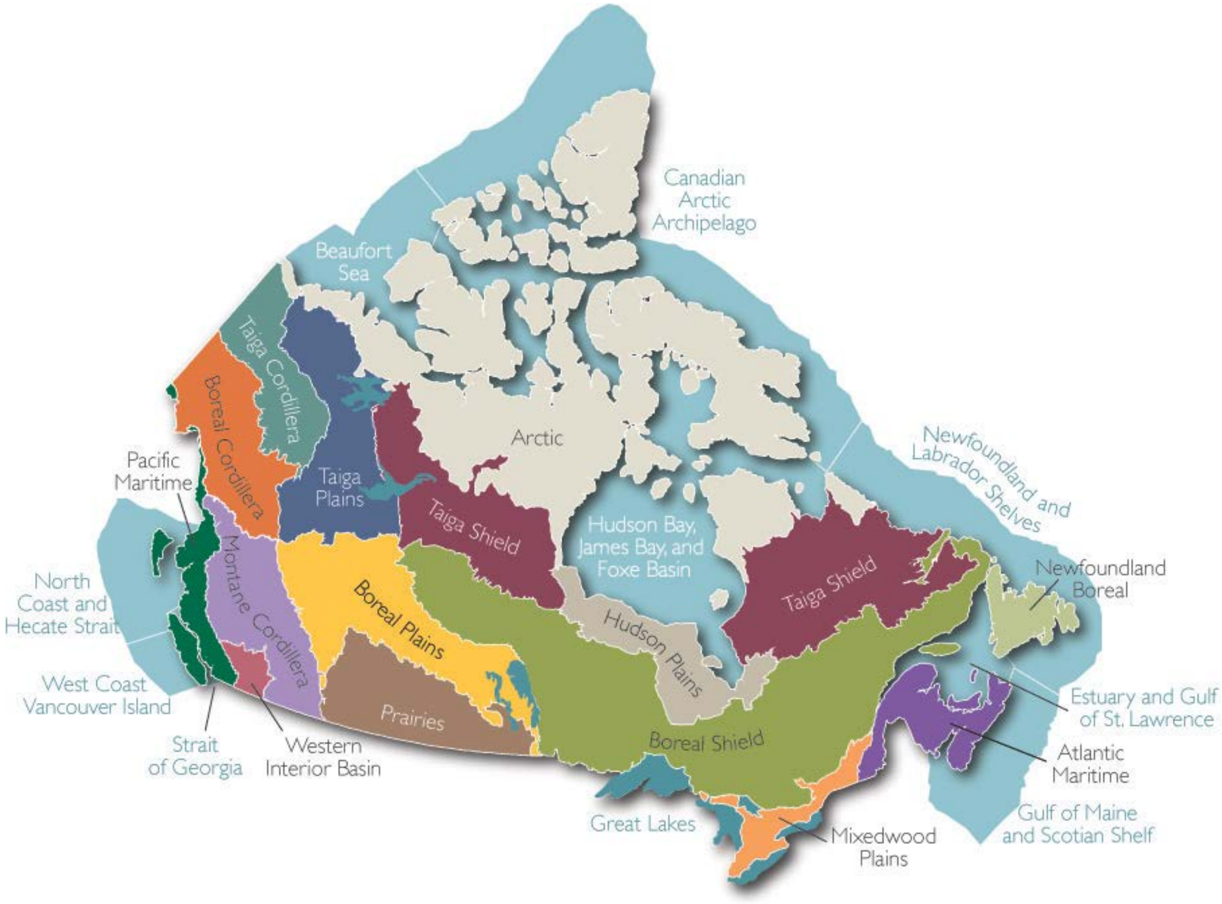

Figure 2. Canada's ecozones$^+$.

The southern boundary of the Arctic Ecozone$^+$ corresponds with the northern edge of the boreal forest. Adjustments to the Taiga Shield/Arctic boundary, based on ground-truthing, were made in the ecozone$^+$ classification. As noted above, three Arctic terrestrial ecozones were combined into the Arctic Ecozone$^+$. This is a vast and diverse ecozone$^+$ and the original three Arctic ecozones represent distinct ecological units. Where appropriate they are discussed separately and referred to as the Southern Arctic, Northern Arctic, and Arctic Cordillera regions of the Arctic Ecozone$^+$.

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Authors of ESTR Thematic Technical Reports from which material is drawn

Ecological classification system for the ecosystem status and trends report: R. Rankin, M. Austin, and J. Rice

Guidance for the preparation of ESTR products – classifying threats to biodiversity: C. Wong

Guidance for the preparation of ESTR products – land classification scheme: J. Frisk

Large-scale climate oscillations influencing Canada, 1900–2008: B. Bonsal and A. Shabbar

Canadian climate trends, 1950–2007: X. Zhang, R. Brown, L. Vincent, W. Skinner, Y. Feng, and E. Mekis

Wildlife pathogens and diseases in Canada: F.A. Leighton

Trends in permafrost conditions and ecology in northern Canada: S. Smith

Northern caribou population trends in Canada: A. Gunn, D. Russell, and J. Eamer

Landbird trends in Canada, 1968–2006: C. Downes, P. Blancher, and B. Collins

Trends in Canadian shorebirds: C. Gratto-Trevor, R.I.G. Morrison, B. Collin, J. Rausch, M. Drever, and V. Johnston

Monitoring biodiversity remotely: a selection of trends measured from satellite observations of Canada: F. Ahern, J. Frisk, R. Latifovic, and D. Pouliot

Biodiversity in Canadian lakes and rivers: W.A. Monk and D.J. Baird

Review conducted by scientists and renewable resource and wildlife managers from relevant territorial and federal government agencies through a review process recommended by the ESTR Steering Committee. Additional reviews of specific sections conducted by university researchers in their field of expertise at the request of the authors. Peer review of the final draft coordinated by the Canadian *Society* for *Ecology* and Evolution. The report was greatly improved by the reviews of D. Berteaux, P. Hale, and one anonymous reviewer.

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Aboriginal Traditional Knowledge compiled from publicly available sources by D. Hurlburt.

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Figure 3. Overview map of the Arctic Ecozone$^+$.

OVERVIEW OF THE ARCTIC ECOZONE^{+}

Long-term perspective: the last ice age and its retreat

At the height of the last glaciation of the Pleistocene, about 20,000 years before present, the Laurentide ice sheet covered the mainland and some of the southern Arctic islands, including Victoria and Baffin islands, while smaller ice caps covered Melville and Bathurst islands and the islands to the northeast (Pielou, 1991). Sea level was about 150 m lower than now. The area that became Banks Island (excluding the southwestern part of the island) and parts of the western Queen Elizabeth Islands were, as they are now, polar desert (Adams and Faure, 1997).

By 13,000 years before present, Banks Island had been separated from the mainland by rising seas. Melville Island, which was connected by land to Prince Patrick Island, as well as some smaller islands to the northeast, had become ice-free (Dyke, 2004). Rising water levels isolated caribou and muskox populations from those in Beringia and, to some extent, from each other, although the animals could still walk across winter ice.

Proxy records from the Arctic Ocean floor and from surrounding land indicate that sea ice has covered at least part of the Arctic Ocean for the past 13 to 14 million years and has been widespread over the past 2 to 3 million years (Polyak et al., 2010). During periods when Earth was warmer, ice cover was reduced in extent. The most recent of these warm periods (prior to the warming of recent years) started near the beginning of the Holocene, about 10,000 years before present (Polyak et al., 2010). Based on records from the Agassiz Ice Cap (Ellesmere Island), average temperatures reached about 3°C above those of the mid-20th century, resulting in loss of multiyear landfast ice (Polyak et al., 2010). Expanses of open water during much of the year isolated islands from the mainland, creating conditions for increasing diversity in wildlife and in other terrestrial ecosystem components, either by genetic drift or through adaptations to local environments. Caribou and muskoxen may have thrived during this period because the vegetation changed from polar desert to dry tundra (Adams and Faure, 1997). Thick, multiyear sea ice returned to the Canadian Arctic by about 5,500 years ago and has been present since, based on evidence from Ellesmere Island (England et al., 2008). Figure 4 shows the sequence of vegetation change from the last glacial maximum through the early Holocene.

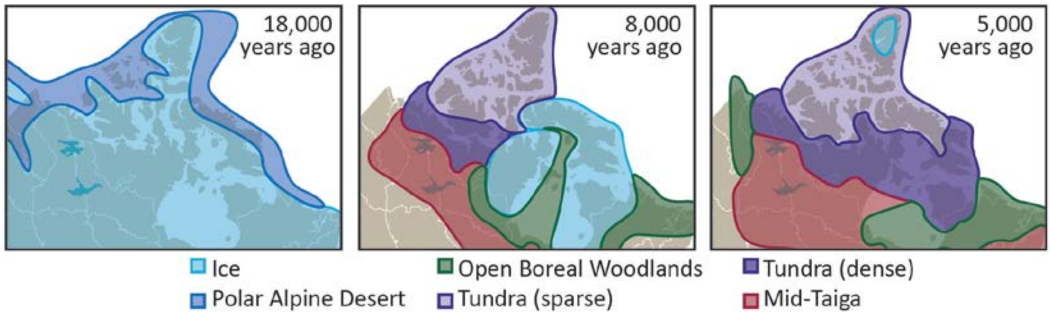


Figure 4. Reconstructed Canadian Arctic vegetation: 18,000, 8,000, and 5,000 years ago.
Years are radiocarbon years: because of different C14 abundance, 18,000 years ago in radiocarbon terms corresponds to about 20,000-21,000 years ago.
Source: Adams and Faure, 1997

Ecozone$^+$ characteristics

Overview by region

The Arctic Ecozone$^+$ is characterized by low air and soil temperatures, continuous permafrost, a short growing season, and limited vegetation productivity (Bolen, 1998). The area of land is approximately 3.148 million km^2 and the combined area of lakes, ponds, and rivers is about 80,000 km^2. The combined length of all the coastlines in the ecozone$^+$ is 179,950 km, almost three-quarters of Canada's total coastline (Natural Resources Canada, 2010). This extensive and complex interface between terrestrial and marine environments has a dominant influence on Arctic ecosystems and the wildlife and people who live there.

The Arctic Ecozone$^+$ includes three major regions (shown in Figure 5 and described in Table 1), each an ecozone in the National Ecological Classification System (see Preface: Ecological classification system – ecozones$^+$).

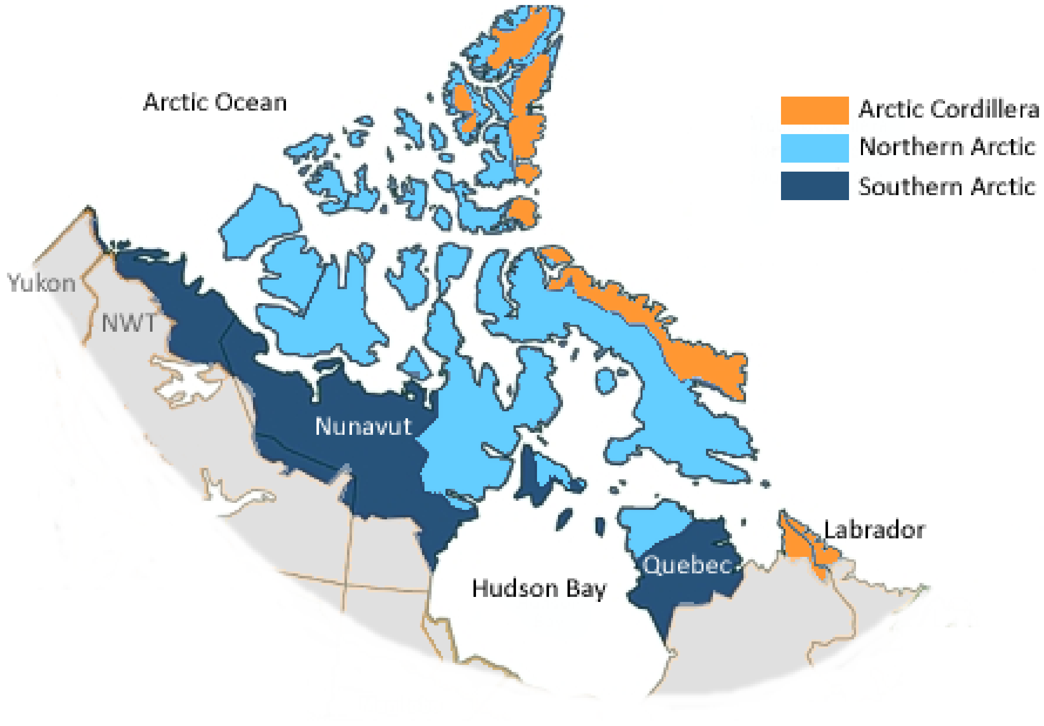


Figure 5. Regions of the Arctic Ecozone$^+$: the Arctic Cordillera, Northern Arctic, and Southern Arctic. Source: adapted from Ecological Stratification Working Group, 1995

Table 1. The three major regions of the Arctic Ecozone^{+}.

	Arctic Cordillera	**Northern Arctic**	**Southern Arctic**
Geographic description	Northeastern fringe of Nunavut and Labrador, including northeastern Baffin Island, eastern Devon Island,Ellesmere, Axel Heiberg, and Bylot islands, and the Torngat Mountains.	Non-mountainous areas of the Arctic islands plus parts of mainland Nunavut and northern Quebec.	Stretches across mainland Canada, from the Yukon coastal plain to Ungava Bay in northern Quebec.
Dominant Features	A vast mountain chain (the only major Canadian mountain ranges outside of the Western Cordillera); polar ice fields, and alpine glaciers.	Low, rolling plains covered in frost-patterned soils and broken rock debris left by glaciers; broad plains in coastal zones.	Dominant features are hills and plains, ponds and lakes.
Plant communities	Dominated by lichens and some mosses at high elevations and tundra at lower elevations.	Lichens and herbs dominate, with dry tundra interspersed with wetland species.	Low shrubs mixed with herbs, lichens, and sedges such as cotton-grass. Major river valleys support scattered clumps of stunted "krummholz" trees.
Human settlement	Sparse; main communities are Pond Inlet, Clyde River, and Qikiqtarjuaq.	Iqaluit, on Baffin Island, is the largest community. Other communities with populations over one thousand include Baker Lake, Cambridge Bay, Igloolik, and Pangnirtung.	Communities are found throughout the ecozone, primarily along the coast. Those with populations over one thousand include Rankin Inlet, Arviat, Puvimituq, and Salluit. Inuvik, in the Mackenzie Delta, is just south of the ecozone^{+}.

Sources: Environment Canada, 2005; Ecological Stratification Working Group, 1995

Rivers

Rivers draining the ecozone^{+} flow either to Hudson Bay, Ungava Bay, or directly to the Arctic Ocean (Figure 6). The eastern slope of northern Labrador drains to the Atlantic Ocean. Because of its position along the northern rim of the Canadian mainland, the ecozone^{+} includes the lower reaches and marine estuaries of rivers with drainage basins extending south of the ecozone^{+}. The vast Mackenzie River delta is partly in the Arctic Ecozone^{+} and partially in the Taiga Plains Ecozone^{+}.

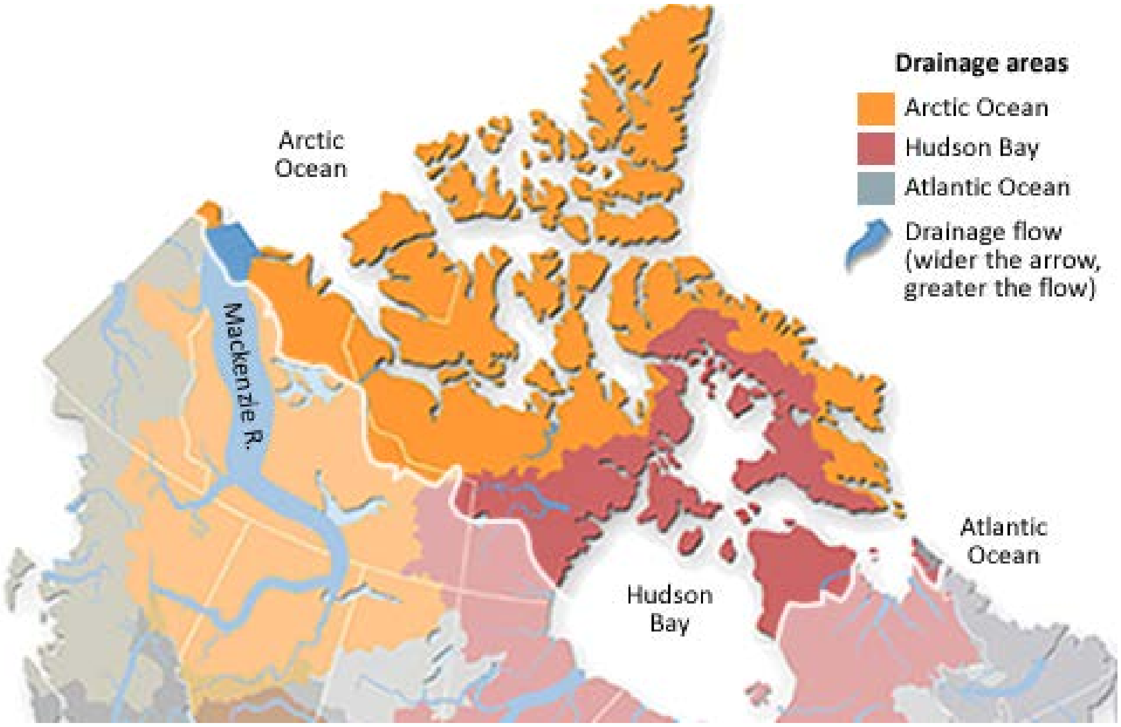


Figure 6. Major drainage areas of the Arctic Ecozone⁺.
Source: adapted from Environment Canada, 2012b

Topography and soils

Descriptions in this section are based on the report of the Ecological Stratification Working Group (1995).

The Arctic Cordillera has rugged mountains over 2,000 m high with massive ice fields and valley glaciers. Nunataks (mountain peaks bare of snow in summer and surrounded by valley glaciers) are common. U-shaped valleys and deep fjords extend far inland. Almost 75% of the landscape is ice or exposed bedrock. Soils are dominated by colluvial and morainal debris.

The western Northern Arctic is mainly lowland plains covered with glacial moraine, marine deposits and outcrops of Paleozoic and Mesozoic sedimentary bedrock. East of Prince of Wales and Somerset islands, granitoid Precambrian bedrock prevails. The permafrost may extend to depths of several hundred metres.

The Southern Arctic is mostly underlain by Precambrian granitic bedrock that occurs as outcrops, except in the west from Great Bear Lake to the Firth River, Yukon, where Cretaceous shales predominate. Soils are dominated by discontinuous glacial deposits except near the coasts where fine-textured marine sediments are more common. The undulating landscape has innumerable lakes, ponds and wetlands and sinuous eskers (raised ridges of gravel deposited by rivers running under the melting Laurentide ice sheet) up to 100 km long. Seasonal thaw layers are of variable thickness, and differences in moisture give rise to a variety of landforms and, consequently, plant and animal habitats.

Because of the low relief in the Southern and Northern Arctic, the high proportion of ice caps and bedrock in the Arctic Cordillera, the generally frozen soils throughout the ecozone+, and the long seasonal period during which streams are frozen and flow is low or non-existent, stream erosion has not been significant. The melting of ice lenses exposed by fluvial action along rivers and wave action along lake and sea shorelines, however, can cause large-scale slumping and sediment discharge.

Percentages (Figure 7) and distribution (Figure 8) of surficial materials in the Arctic Ecozone+ are presented below and classes of surficial materials are described in Appendix 1. See also the related analysis of land cover in the section on Ecosystem structure.

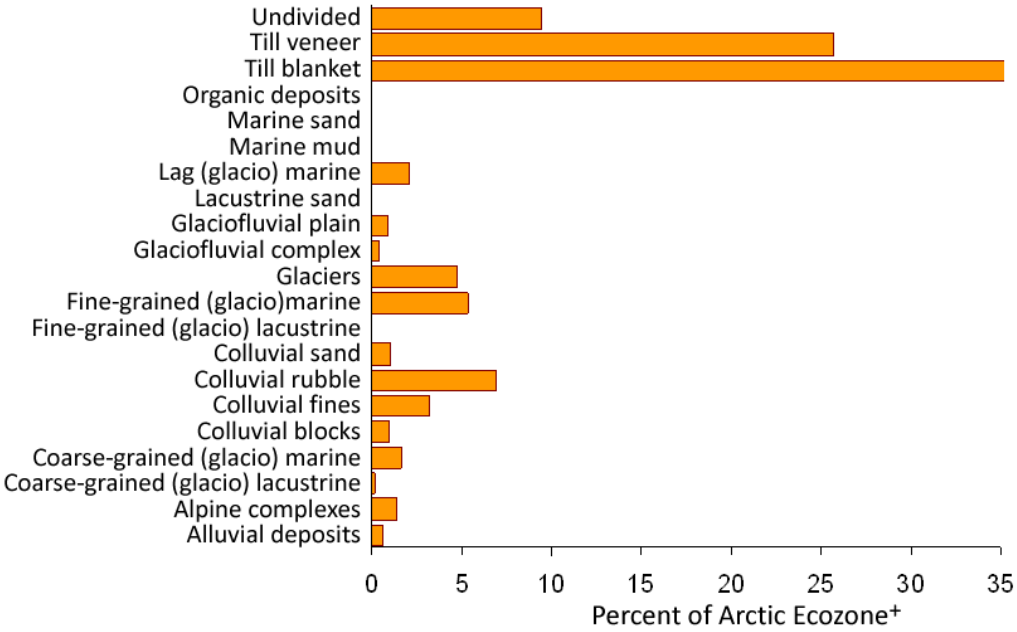


Figure 7. Surficial materials in the Arctic Ecozone+: percentage of total land area.
See Appendix 1 for descriptions of classes of surficial materials.
Source: Geological Survey of Canada, 1994

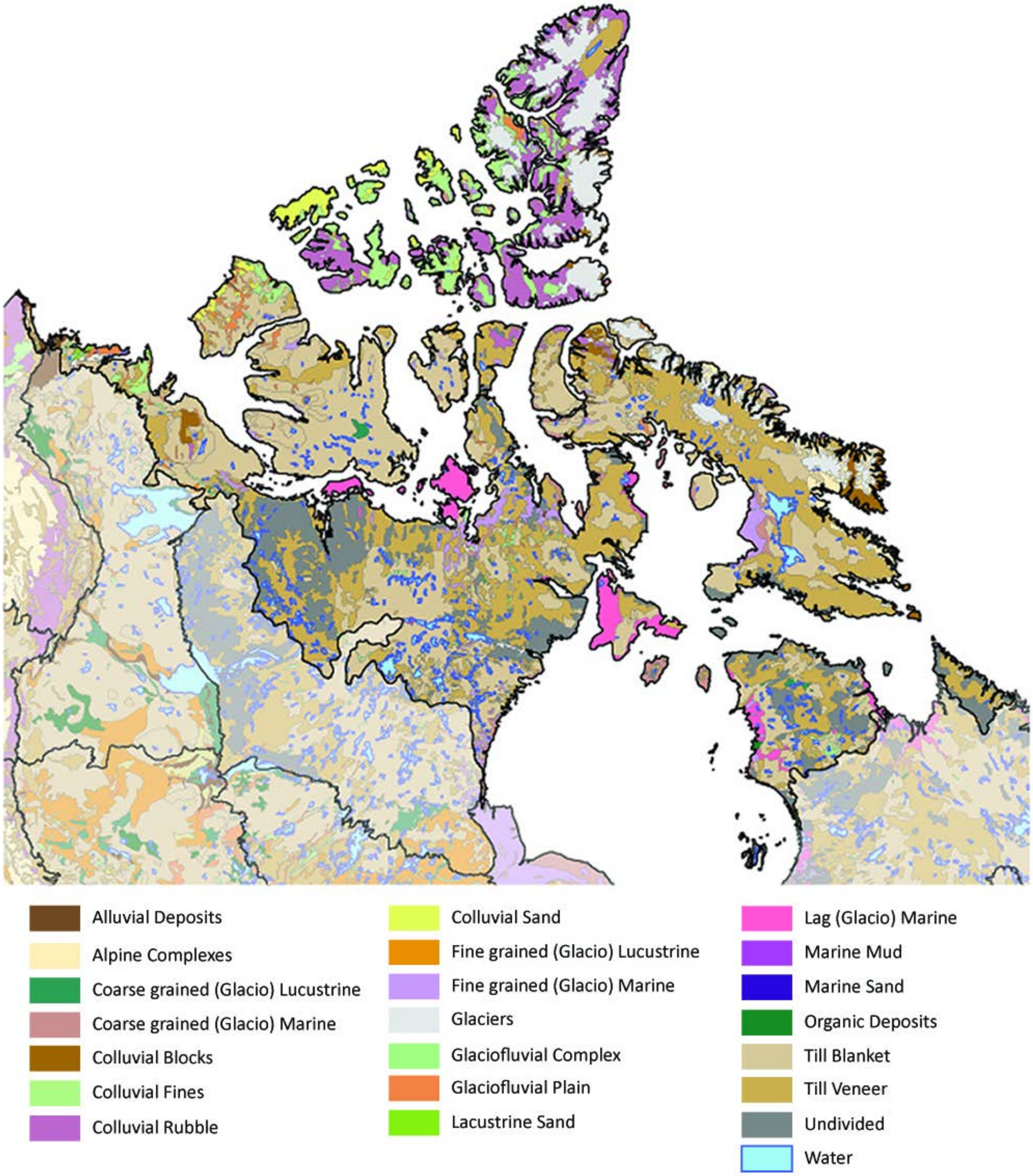


Figure 8. Map of surficial materials of the Arctic Ecozone^{+}.
Source: Geological Survey of Canada, 1994

Permafrost

The Arctic landscape is shaped by the complex relationships between climate, permafrost, and vegetation (Walker et al., 2003). Permafrost is defined as soil, rock, or sediment that remains at or below a temperature of 0°C for at least two consecutive years. It can contain ice as pore ice, ice wedges, or massive ice bodies. In the Arctic Ecozone$^+$ permafrost is continuous and may be several hundred metres thick and have temperatures colder than –5°C (Heginbottom et al., 1995; Smith et al., 2001a).

Above the permafrost, the soil profile has an active layer that is seasonally frozen. The thickness of the active layer is highly variable depending on aspect, vegetative cover, and the ambient temperature regime. Moisture and gas fluxes are generally confined to this seasonally thawed active layer. Thus, permafrost's presence and influence is through physical and chemical processes in the active layer which operate as thermal and hydrological gradients. Plant roots and nutrients accessible to plants are largely confined to the active layer, as are the burrowing activities of invertebrates and vertebrates.

Humans in the Arctic Ecozone$^+$

(This section is based on the following sources, with additional references as noted, McGhee, 1978; Bone, 1992; Black, 2002; Bonesteel, 2006; Nunatsiavut Government, 2009; Freeman, 2012; Natcher et al., 2012)

The Arctic Ecozone$^+$ is sparsely populated, though its human population approximately tripled between 1971 and 2006 (Figure 9). The majority of the people are Inuit and the ecozone$^+$ covers most of the four Inuit regions established through Canadian comprehensive land claim agreements: 1) the Inuvialuit Settlement Region (parts of the Yukon and the Northwest Territories); 2) Nunavut; 3) Nunavik (part of Quebec); and 4) Nunatsiavut (part of Newfoundland and Labrador). The land and marine areas covered by comprehensive land claim settlements in the ecozone$^+$ are shown in Figure 10. While the terms of these agreements differ, they all provide for wildlife and habitat management by boards or councils with input from Aboriginal regional and local governance bodies, as well as federal and territorial governments. These co-management regimes are a distinguishing element of the Canadian Arctic and central to all aspects of ecological management, monitoring, and research in the Arctic Ecozone$^+$.

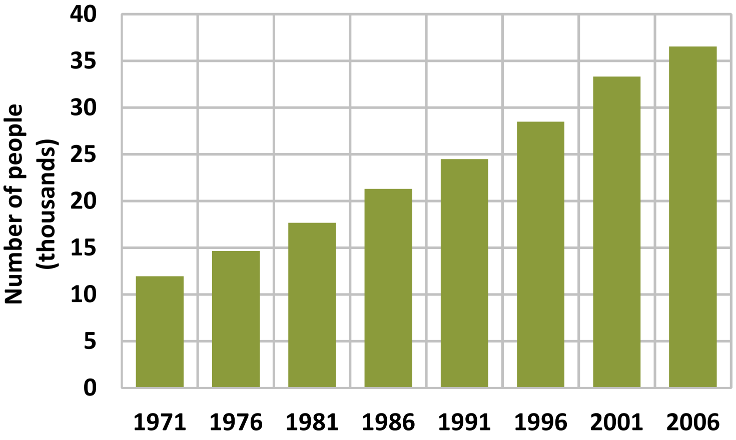


Figure 9. Human population trends for the Arctic Ecozone⁺, 1971–2006.
Source: based on data for the three Arctic ecozones compiled from Statistics Canada, 2000 and 2008b

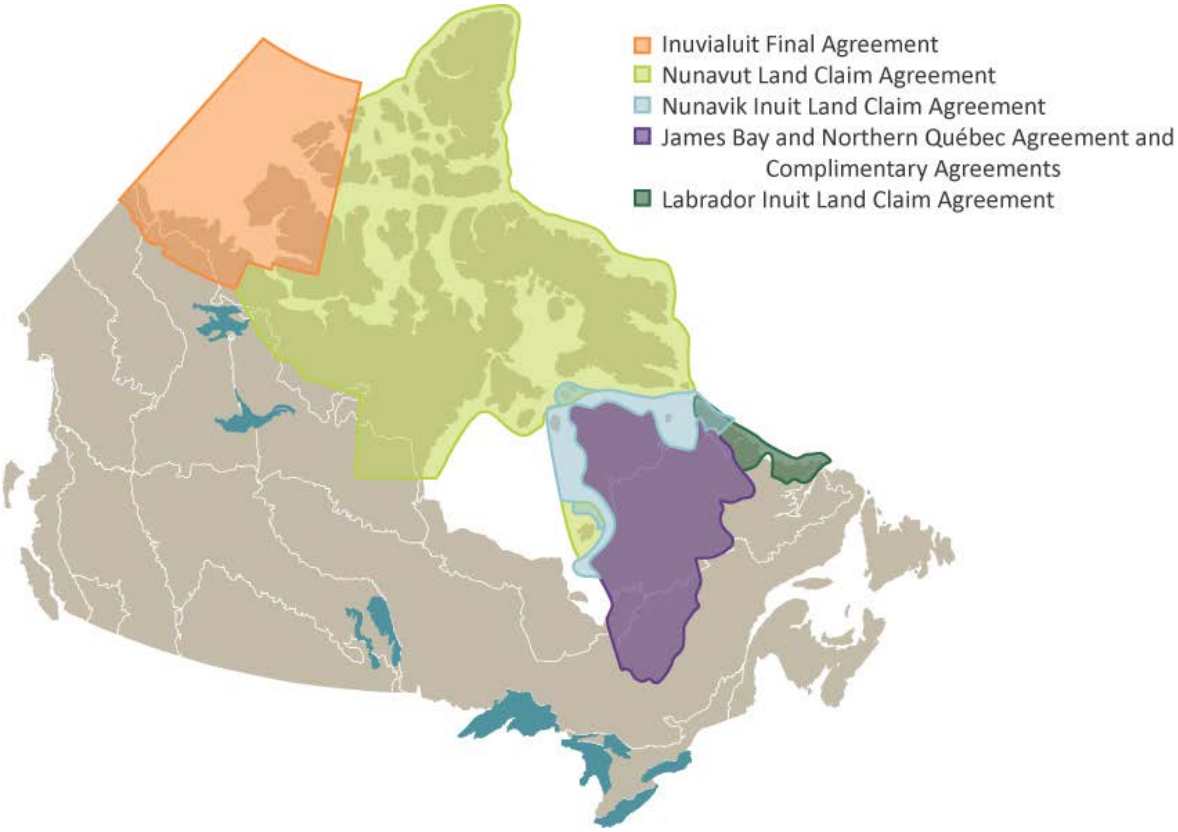


Figure 10. Regions covered by comprehensive land claim settlements in the Arctic Ecozone⁺.
Source: adapted from Government of Canada, 2011

Settlement and economic history

Humans have occupied the western side of Beringia for at least 30,000 years and are known to have crossed into North America by 14,000 years ago, either following the retreat of the Cordilleran and Laurentide ice sheets or routes along the coast of Alaska (Kitchen et al., 2008). About 4,500 years ago the first wave of human inhabitants is thought to have moved across the eastern Arctic from Alaska (Friesen, 2004). Based on distinctive material culture, archeologists have assigned these people several separate cultural phases, the last of which is the Dorset culture. The Pre-Dorset culture was defined by a highly mobile lifestyle, limited tool kits, and limited reliance on food storage (Milne et al., 2012). Between 1000 and 1200 AD the second wave of settlers, the Thule, entered the Arctic from the Bering Strait region. The Thule were the ancestors of the contemporary Inuit and are traditionally thought to have displaced the Dorset culture (Friesen and Arnold, 2008). There is some evidence of intermixing (Helgson et al., 2014). By 1200 to 1300 AD, the Thule had spread across the Arctic to Greenland (Kallreuth et al., 2012). With the Thule culture came a shift to a more sedentary way of life, the introduction of boats, and a high diversity of specialized tools (Anderson, 2004).

Europeans (the Norse) sailed to Baffin Island around 1000 AD, but did not settle. European exploration recommenced in the 16th century, with Martin Frobisher claiming Baffin Island for England in 1576. The 19th century commercial whaling boom began the widespread transition to a mixed economy for the Inuit, with guns and dry goods being received in exchange for food and guiding services. Whaling had a profound influence on the Inuit, introducing alien diseases including typhus, that spread rapidly and killed many people, and through impacts such as depletion of caribou for food for whaling crews, and an increasing reliance on European goods. Whaling was replaced in the early 20th century by the fur trade boom centred around the Hudson's Bay Company trading posts and accompanied by an expansion of the mixed economy to include trapping and trading for a wider selection of goods. Government posts, policing, and missions were also established. The Inuit altered their hunting practices and annual cycles to adapt to these changes and suffered economic and social hardship when the price of furs fell drastically in the 1930s and 1940s.

The Arctic attained a profile in North American defense during and after the Second World War (Figure 11). Mineral and energy resources began to be discovered and developed, creating a perception of the Arctic as a vast storehouse of resources. Starting in the 1950s, the Canadian government resettled many Inuit into permanent communities, providing social services and required education in English, with the aim of improving the well-being of the Inuit through assimilation into the mainstream culture and economy. Residential schools were established under this policy of assimilation, leading to many instances of physical and emotional abuse, with lifetime consequences for the children of this era and for their communities. In the 1960s, Inuit political movements rose in response to the family, social, cultural, and economic disruption that had resulted from decades of upheaval and hardship. Goals included restoring and reviving native languages and traditional lifestyles and regaining control over the management of lands and resources, the basis of Inuit culture and prosperity.

The Distant Early Warning (DEW) Line

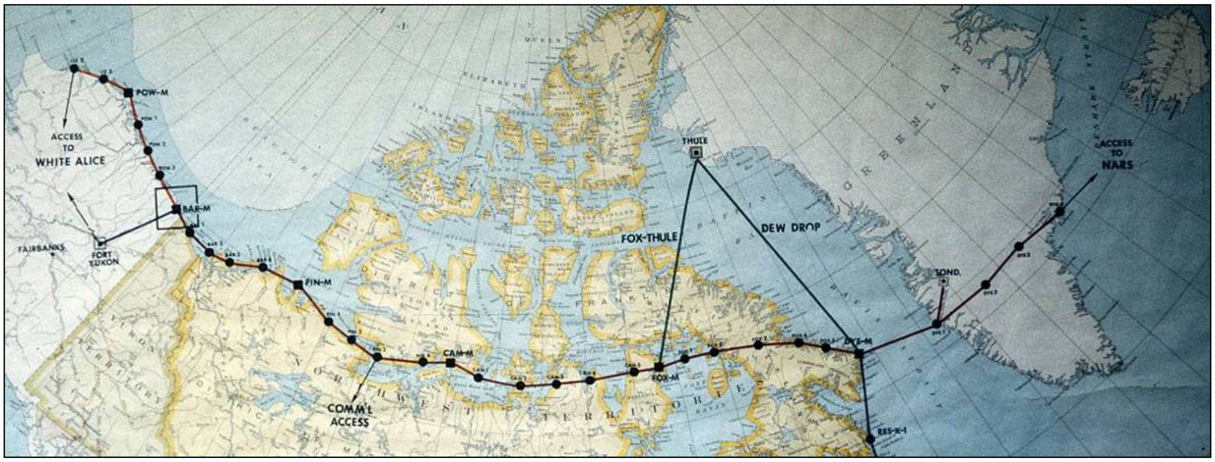


Figure 11. Wall map from a DEW Line station, showing the line and location of main stations. Photo by Tech. Sgt. D.L. Wetterman, U.S. Air Force, 1987 (Wikimedia commons)

Construction of the Distant Early Warning (DEW) line of 42 radar stations along the Canadian Arctic coast in the late 1950s provided some wage employment and introduced new technology (such as snowmobiles) to more communities. The rapid influx of shifting populations of people with very different cultural perspectives and the rapid introduction of new technologies, infrastructure, and goods also resulted in major social upheaval to the Inuit (Neufeld, 2002). The DEW Line was a joint project of the U.S. and Canada and a product of the Cold War: its purpose was to provide early warning of attacks on North America from the Soviet Union (Neufeld, 2002; Bonesteel, 2006). In the early 1960s, the smaller sites were decommissioned. The remaining 21 sites were operated by Canada to various dates up to 1993. The new North Warning System, which includes eight of the old radar stations, replaced the DEW line in the 1990s. Clean-up of the sites has been plagued by problems and escalating costs related to contamination, including from PCBs, DDT, hydrocarbons, and lead. Clean-up, nearing completion in 2013, has a goal of keeping contaminants out of the Arctic food chain and will be followed by a 25-year monitoring plan (DND, 2001; Aglukkaq, 2012); there have been local instances of contamination of biota and continuing concerns expressed by Arctic residents about impacts (e.g., Gamberg et al., 2005b).

The Inuvialuit Settlement Region

In the Western Arctic, commercial bowhead whaling stations in the latter part of the 19th century introduced infectious diseases that led to epidemics and the death of most indigenous Inuit (the “Mackenzie Eskimos”). Iñupiat from Alaska migrated eastward and integrated with the remaining residents in the Mackenzie Delta region. The Inuvialuit, as an Inuit group, are a newer group, formed by the Inuit from the Delta region uniting with those from Banks and Victoria islands in the 1970s to work together as a political and economic force, partly in response to the pressures imposed by Beaufort Sea oil and gas development. The Inuvialuit were successful in negotiating the Western Arctic agreement (Inuvialuit Final Agreement) in 1984.

The Inuvialuit Settlement Region (ISR) covers 435,000 km^2 of land and sea (Figure 10). Most of the land is part of the Northwest Territories, but it also includes the Yukon North Slope, designated under the agreement to fall under a special conservation regime. Known mineral deposits in the ISR include petroleum, natural gas, zinc, and nickel. The Mackenzie River is a commercial artery and there is an all-season road to Inuvik and seasonal ice roads from Inuvik to Aklavik and Tuktoyaktuk. Air access is important for all communities. National parks form 29% of the ISR.

Inuvik, the largest community of the ISR and its main administrative centre, and the town of Aklavik, are just south of the Arctic Ecozone$^+$. Within the ecozone$^+$ are the communities of Tuktoyaktuk, Sachs Harbour, Ulukhaktok (Holman), and Paulatuk. Tuktoyaktuk was an important centre for offshore oil drilling in the 1970s and 1980s. There are no year-round communities in the Yukon portion of the ISR. Wage employment is the main source of income throughout the ISR, but unemployment is high. The major wage employment is in the public sector. The Inuvialuit Development Corporation includes several firms involved in oil and gas activity and the Inuvialuit are involved in the development of a range of new economic initiatives.

Oil and gas activities were important in the region in the 1970s and 1980s. Renewed interest and activity in onshore and offshore exploration and in development of onshore fields in the Mackenzie Delta region began in the early 2000s. In addition to the conventional petroleum resources, there is good potential for unconventional petroleum resources in this region. Exploration has been undertaken on gas hydrate accumulations beneath the Mackenzie Delta, and, more recently, industry is beginning to evaluate the potential of extensive shale formations in the Mackenzie Valley.

Hunting remains important economically, socially, and culturally and provides the main source of meat. Caribou (*Rangifer tarandus*), muskoxen (*Ovibos moschatus*), polar bears (*Ursus maritimus*), beluga (*Delphinapterus leucas*), and seals are hunted, with variation in main species harvested from community to community. The introduction of gill nets in the 1830s led to the development of river fisheries, which remain important in Inuvialuit subsistence economies. Other economic activities include trapping, tourism, guided sport hunting (for polar bears and muskoxen), and the production of arts and crafts.

Nunavut

The Nunavut project, first proposed in 1976, eventually led to a final Nunavut Land Claims Agreement in 1993 and the creation of the Territory of Nunavut in 1999. Nunavut, at 2.12 million km^2, is the largest political unit in Canada, covering one-fifth of the country's landmass. The population is about 84% Inuit (Statistics Canada, 2008a). There are 29 communities, of which Baker Lake is the only one located away from the coast. Iqaluit is the only city [population 6,669 in 2011 (Statistics Canada, 2012)] and the territorial capital. Household income is from the wage economy, government transfers, and the land-based economy. Government is the largest employer, with about 50% of jobs being in the public sector. Tourism is the fastest growing economic sector but remains relatively small. There is no permanent road infrastructure linking Nunavut with the rest of Canada. The longest road, 21 km in length, connects Arctic Bay and Nanisivik.

Mineral resource exploration and extraction have historically dominated the private sector economy. Nanisivik, a lead-zinc mine on north Baffin Island, opened in 1976 and closed in 2002. The North Rankin Nickel Mine operated from 1957 to 1962 in Rankin Inlet. Polaris, a lead-zinc mine on Little Cornwallis Island, operated from 1982 to 2002. Lupin Gold Mine opened in 1982 and closed 2005. Nunavut's only diamond mine, Jericho Mine, operated from 2006 to early 2008 and is currently (2013) in care and maintenance mode. Lupin and Jericho mines are close to the border with the NWT and were served by a winter road from Yellowknife when they were operational. Meadowbank Gold Mine, near Baker Lake, started production in 2009 while Doris North Gold Mine, near Bathurst Inlet, obtained mine approval in 2006 but has yet to operate and has since gone into care and maintenance. The Mary River iron ore deposit on Baffin Island was approved for mine development in 2012 but is currently (2013) seeking an amendment to allow for changes to the project and has yet to go into production. Potential mines in the review process are the Meliadine gold project near Rankin Inlet and the Kiggavik uranium project near Baker Lake. A gold deposit at Back River, east of Lupin, is currently in the advanced exploration stage and other major exploration projects include base metal deposits at Izok Lake, High Lake, and Hackett River (west, north, and east of Lupin, respectively).

Nunavik

Nunavik covers 443,684 km^2, about one-third of the province of Quebec. The 2006 population was 11,627, 90% Inuit (Statistics Canada, 2008a). There are 15 communities along the coasts of Ungava Bay and the eastern shore of Hudson Bay. The major settlement is Kuujjuaq.

Pre-Dorset and Dorset artifacts are found along the coasts of both Hudson and Ungava bays, dating to around 4,000 years ago. These cultures were succeeded by the Thule culture. Commercial whaling started in the 18th century in Hudson and Ungava bays, and fur trade rose in the region. The hunter-gatherer economy was little changed for many years, until consumer goods became more widely traded. The distribution of villages today is influenced by locations of Hudson's Bay Company trading posts.

Major sources of energy and minerals attracted attention in the 1960s and 1970s. Hydro development proposals for James Bay and La Grande River (in the Taiga Shield Ecozone$^+$) led to widespread opposition by the Inuit and the Cree and the signing of the James Bay and Northern Quebec Agreement in 1975. The agreement addressed outstanding issues of indigenous rights and formed the Kativik Regional Government and Makivik Corporation, the political and fiscal arms of the Inuit of Nunavik. In 2006, the Nunavik Inuit Land Claims Agreement was signed. Applying to eligible beneficiaries of the James Bay and Northern Quebec Agreement, it provides protection for land and resource rights and mechanisms for involvement in management on areas traditionally used by the Inuit of Nunavik, including marine areas and lands in Nunavut and northern Labrador (Figure 10).

Nunatsiavut

The lineage of the Inuit of Nunatsiavut—the Sikumiut, 'people of the sea'—traces back to Thule Eskimos who migrated from Baffin Island. The long European settlement history (since the 17th century) in coastal Labrador influenced the Sikumiut through intermarriage and cultural change. Traditionally, the economy was based on sea mammal harvest and caribou hunting, supplemented with fishing and hunting waterfowl. When whales were depleted after 1800, seals became increasingly important. In the 1930s, government began to supplant the influence of the fur trade posts and missions. Confederation in 1949 brought profound social change but no recognition of the Sikumiut as Aboriginal peoples under the Canadian constitution. The Sikumiut were the last Inuit in Canada to achieve the major step of a land claim settlement.

The Labrador Inuit land Claims Agreement received Royal Assent in 2005. The agreement covers about 72,500 km^2 of land and 49,000 km^2 of sea (Figure 10). Intended to create a stable environment for land use and investment and contribute to development of the Labrador Inuit, it also led to formation of Torngat Mountains National Park and the establishment of mechanisms for involvement in environmental assessment and for co-management. A regional Inuit government, the Nunutsiavut government, was created by the agreement and provisions were established for sharing of revenue from mineral resources.

The population of Nunatsiavut was 2,427 in 2006, 89% Inuit (Statistics Canada, 2008a), mainly in five communities: Nain, Hopedale, Makkovik, Postville, and Rigolet, all located south of the Arctic Ecozone$^+$.

DESCRIPTION OF THE CONDITION OF THE ARCTIC ECOZONE$^+$

Abiotic drivers

Introduction to climate-related drivers

In the Arctic, as elsewhere, ecosystems are determined to a large extent by abiotic drivers such as mean and extreme seasonal temperatures and precipitation, number of degree-days and frost-free days, insolation (amount of sunlight reaching the surface), wind, and albedo (amount of sunlight reflected from the surface, rather than being absorbed). These factors, plus soil moisture, determine the extent of permafrost (perennially frozen ground), which limits root penetration and has other profound effects on plants and animals. Terrestrial habitats are further influenced by microclimates determined by such physical features as slope, aspect, soil type, and proximity to large water bodies.

The combined influence of these abiotic drivers creates an environment unfavourable to tree growth and generally defines Arctic ecosystems as “tundra,” a mosaic of wet and dry habitats dominated by varying proportions of mostly perennial grasses, forbs, and low-growing shrubs. Because of the variety and dominance of shrubs in the heather family (Ericaceae), shrublands are often called “shrub-heath” or “dwarf shrub-heath” habitats, although other families, such as willows (Salicaceae) and dwarf birches (Betulaceae), are also prominent. Spring growth, flowering, and seed-setting periods are brief, and animal life histories are closely linked to vegetation phenology. Tundra closely follows the limits of continuous permafrost, while discontinuous permafrost characterizes the transition between tundra and taiga.

Large-scale changes in abiotic drivers, principally temperature and precipitation, can have profound effects on distribution of tundra, and on the distribution and composition of habitats within it. For example, as noted in the introduction, a warming period beginning about 8,000 years ago changed the Northern Arctic from polar desert to dry tundra.

The Earth has warmed in recent decades, largely due to greenhouse gas emissions. The degree of warming has been unprecedented since earlier interglacial periods in the Pleistocene Epoch (IPCC, 2007). The Arctic Climate Impact Assessment (ACIA, 2005) and the IPCC Assessment Report 4 (IPCC, 2007) both reported that surface air temperatures in the circumpolar Arctic have increased over the past few decades at almost twice the rate of the global mean, with warming being particularly pronounced since the early 1990s. Figure 12 maps temperature anomalies for 2001–2005 relative to 1951–1980. This illustrates the greater warming in the circumpolar Arctic and also the regional trend of greater warming in the western part of the North American Arctic. Figure 13 shows the trends in mean annual global and Arctic temperatures from 1880 to 2012.

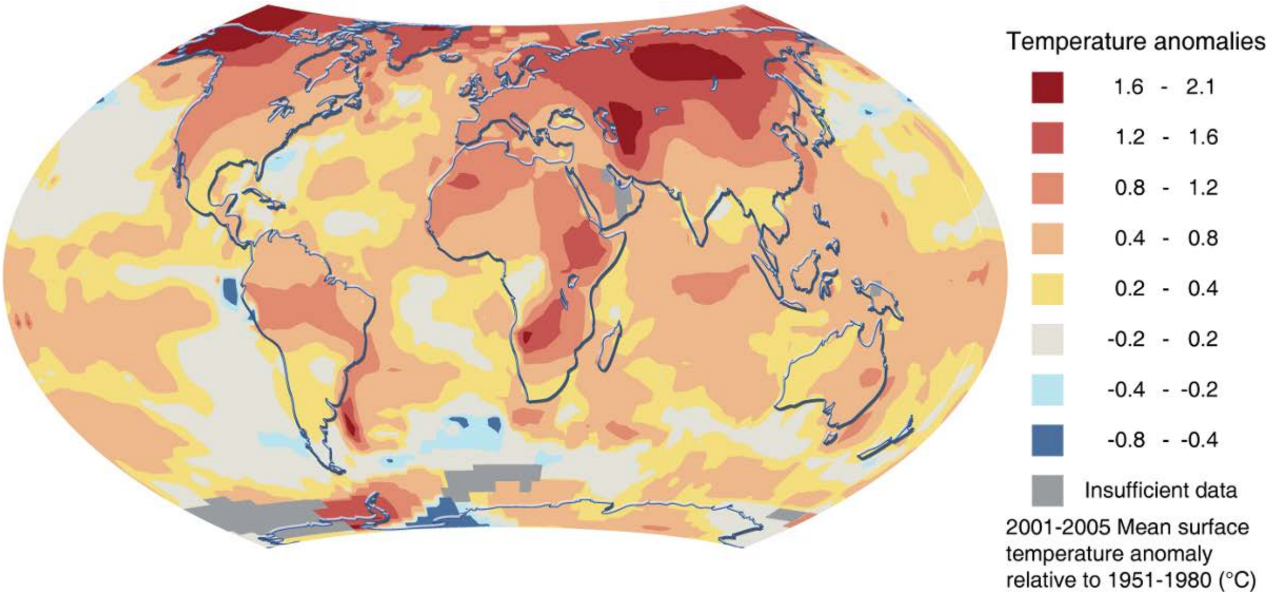


Figure 12. Global increases in annual temperatures for 2001–2005 relative to 1951–1980, showing the greatest warming over land and at high latitudes in the Northern Hemisphere.
Source: Overland et al., 2007 based on Hansen et al., 2006

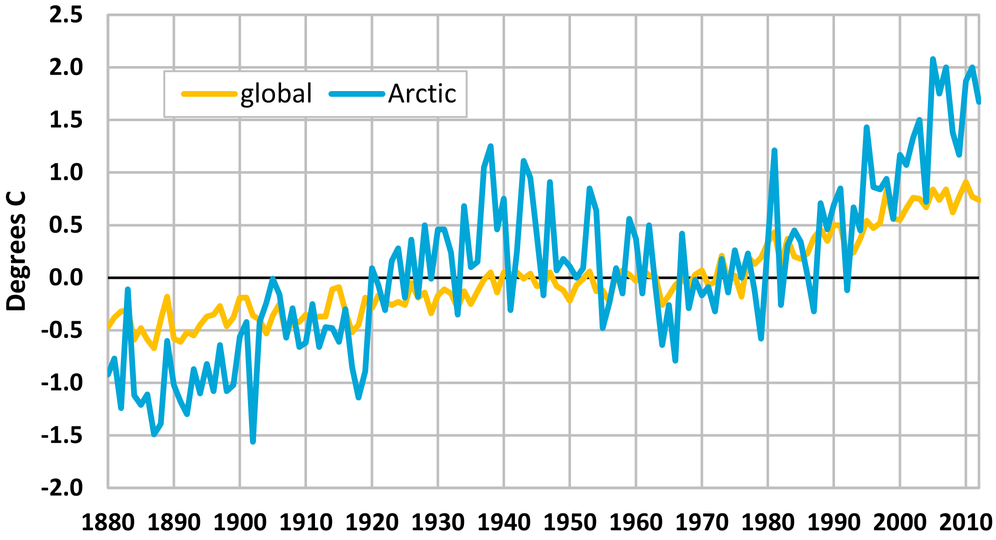


Figure 13. Mean annual global and circumpolar Arctic temperatures (64 to 90°N latitude), 1880-2012. Land-surface air temperature anomalies.
Based on meteorological station measurements, the annual means include the Canadian stations discussed below. Data are anomalies, calculated on the base period 1951-1980.
Source: data from NASA, 2013

A major consideration for the Arctic Ecozone$^+$ is the role of feedback mechanisms in amplification of climate change in the Arctic—particularly as these mechanisms involve changes in ecosystems. For example, reductions of surface albedo from the retreat of sea ice has led to enhanced warming of the lands peripheral to the Arctic Ocean (Walsh, 2008); shorter snow cover duration has resulted in greater surface warming and increased shrub growth, further reducing surface albedo and leading to additional regional warming (Sturm et al., 2001; Chapin III et al., 2005; Walsh, 2008). Increased shrub growth also increases the ability to retain a deeper snow cover which impacts the ground thermal and hydrological regimes, microbial activity, and nutrient and carbon cycling (Myers-Smith, 2007).

Observations by Canadian Inuit about climate change are documented in many sources, including the Arctic Climate Impact Assessment (ACIA, 2005) and studies based on interviews in Inuit communities (Kassi, 1993; McDonald et al., 1997; e.g., Fox, 1998; Fox, 2004; Huntington et al., 2005). A common theme is the increased variability of the climate. For example, people in Igloolik report that weather is more unpredictable and sudden weather changes have forced hunters to spend extra unplanned nights on the land (Ford et al., 2006). People in Nunavut report experiencing warmer temperatures year-round accompanied by changes in the length and timing of the seasons (Government of Nunavut, Department of Environment, 2005). The communities of the Inuvialuit Settlement Region have been observing changes associated with warming in their region for a longer period than those living in the Eastern Arctic communities and these changes appear more pronounced (Furgal et al., 2002; Nickels et al., 2002; Furgal and Seguin, 2006). Milder winters, warmer summers, a shorter fall, and a slower and later freeze-up were among the many weather-related changes observed by Banks Island Inuvialuit.

Fluctuations in the seasons were noted, particularly the earlier arrival of spring (Ashford and Castleden, 2001).

Climate variability and change is the unifying theme of this report and the material outlined above is expanded upon in the relevant sections throughout the report.

Climate trends since 1950

Overview

This section is based on records from Canada's network of climate stations, analysed for the country as a whole, and on an ecozone$^+$ basis for this report. Results of the analysis for Canada are presented in the ESTR thematic report *Canadian climate trends, 1950–2007* (Zhang et al., 2011). A base or reference period (1961–1990) was used to construct both a regionally averaged and station-by-station anomaly series of precipitation and temperature trends. The results of these trend analyses for the Arctic Ecozone$^+$ are summarized in Table 2 and presented and discussed below in relation to the Canadian trends from the same analysis and with reference to Canadian and circumpolar Arctic climate trends discussed by other authors. Zhang et al. (2011) also analysed trends related to snow, reported in the section on Ecosystem functions/processes (page 24).

Temperature and precipitation stations are widely distributed across the Arctic Ecozone$^+$, but station locations are biased to coastal areas and the station density is low. The ecozone$^+$ average values for temperature and precipitation cover at least three distinct climatic regions [Canadian Arctic Archipelago (also known as the Arctic Archipelago), Western Arctic, and Eastern Arctic] and long-term variations of climate in these regions may be different. This means that ecozone$^+$ averages are not necessarily representative of status and trends at different locations within the ecozone$^+$. Maps showing trends at individual stations are included to provide a better indication of regional trends.

See Zhang et al. (2011) for more information on methodology of this analysis.

Table 2. Overview of climate trends since 1950 across the Arctic Ecozone$^+$.

Climate variable	Trends from 1950 to 2007
Temperature	Significant increases in ecozone$^+$ seasonal average temperatures in summer and fall. Significant increases at many climate stations in all seasons (Figure 14). No significant cooling trends at any station in any season.
Precipitation	Increased significantly in all seasons across the ecozone$^+$, with the greatest increase being in winter. Significant increases at many climate stations in fall, winter and spring (Figure 16). Significant increase at only 1 station in summer.

Source: Zhang et al., 2011

Temperature

Annual mean temperature increased about 1.4°C between 1950 and 2007 across Canada, with warming trends evident in most parts of the country. Although for the country as a whole the greatest temperature increases occurred during winter and spring, warming in the Arctic Ecozone⁺ was most consistent in summer and fall, with significant temperature increases of 0.9°C in the summer and 1.7°C in the fall. These means mask the greater changes that have occurred at some locations, as can be seen on the maps showing magnitude and significance of temperature trends for each Arctic climate station included in the data set (Figure 14).

Growing season in this national analysis is defined as the period during which the mean daily temperature exceeds 5°C (based on a running average of 5 days). Significant increases in growing season length were observed in many places in Canada, largely due to earlier starts to the growing season. The selection of 5°C as the minimum temperature is arbitrary and the characteristics of the actual growing season vary among species, with cold-adapted tundra plants starting growth just above the freezing point. As this 5°C threshold for the start of the growing season was not met at many Arctic locations, trends in growing season in the Arctic were not detected through this analysis. The records, however, show significant increases in temperatures at many Arctic stations, across all seasons (Figure 14). Temperature measurements taken continuously since 1995 as part of the International Tundra Experiment (ITEX) site at Alexandra Fiord, Ellesmere Island, show an increase in the growing season length (measured as days above 0°C) of about 1.5 days per year over the 12 years of measurement (Figure 15). Plant productivity has increased across the ecozone⁺, indicating both an increase in peak productivity and in growing season length (Zhou et al., 2001; Goetz et al., 2007) – see the section on Primary productivity (page 69).

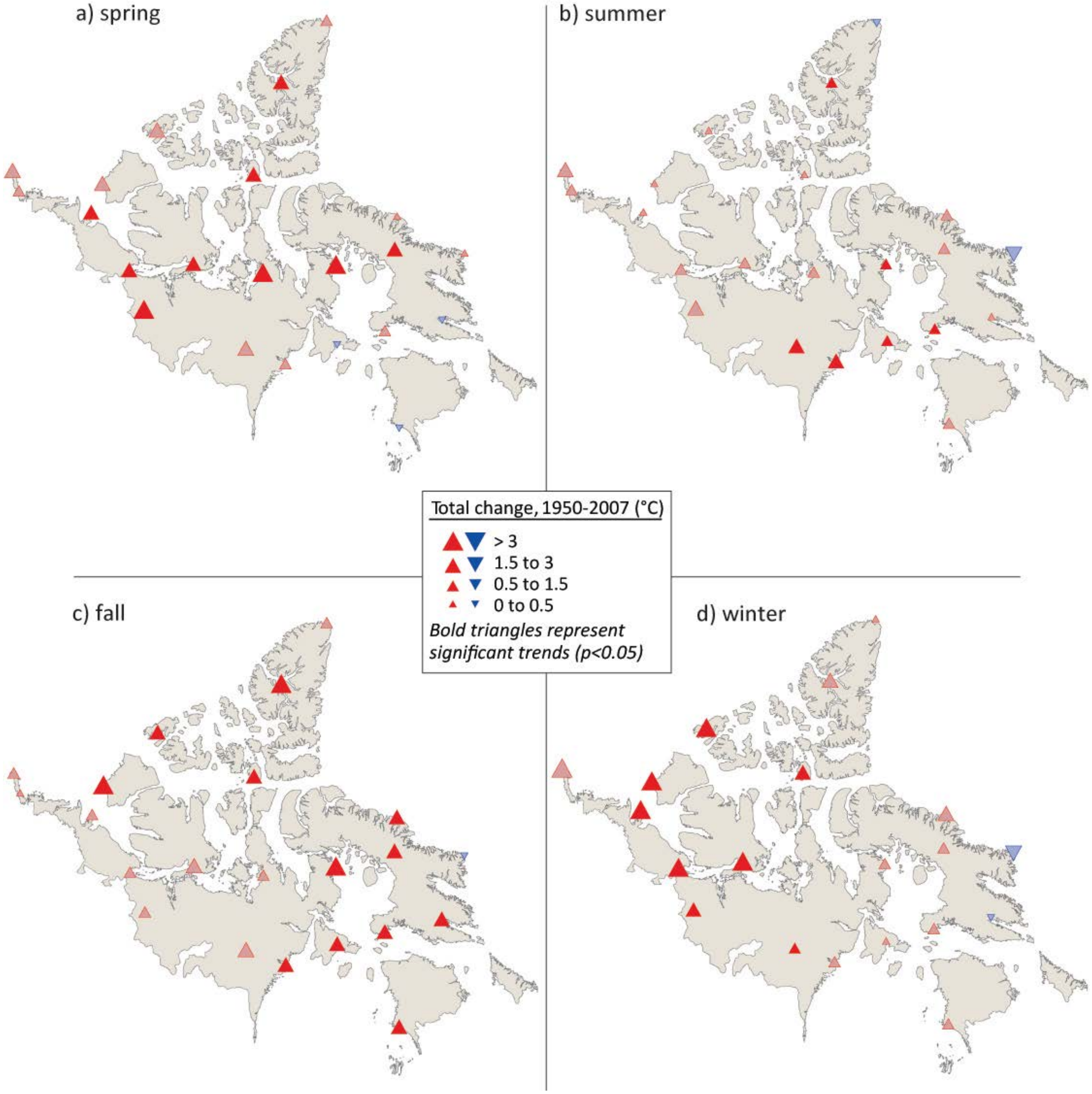


Figure 14. Change in mean temperature, 1950–2007, for: a) spring (March–May), b) summer (June–August), c) fall (September–November), and d) winter (December–February).
Source: ecozone$^+$ data provided by authors of Zhang et al., 2011

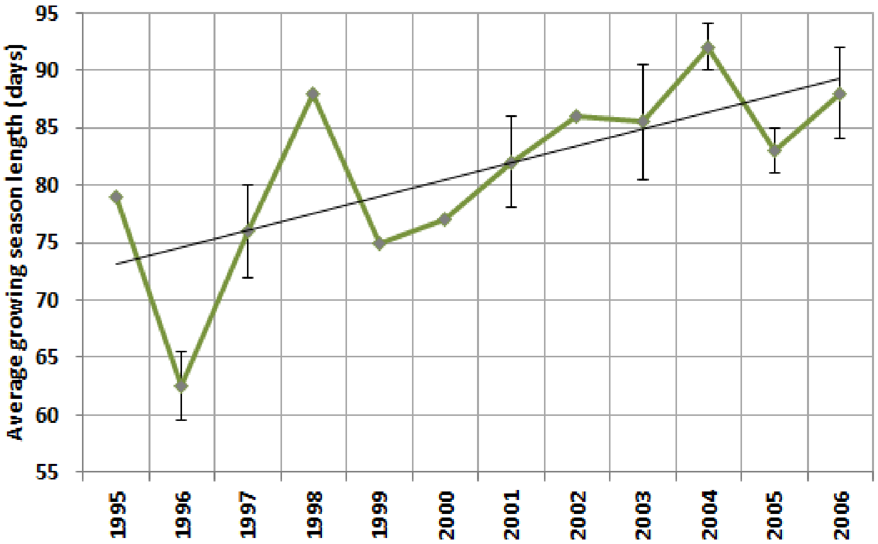


Figure 15. Growing season length in control plots of a long-term warming experiment in a Cassiope *heath tundra at Alexandra Fiord, Ellesmere Island, 1995-2006.*
Growing season is defined here as the number of days with mean daily air temperature greater than 0°C. Air temperature was measured at 10 cm above the ground surface. Data are means (n=2) with range bars. The trend line is a linear regression showing an increase of ca. 1.5 days per year over the 12 years.
Source: G. Henry, unpublished data

Precipitation

Precipitation increased at the majority of stations across Canada between 1950 and 2007, with stations in Arctic Canada experiencing the greatest relative increases. Precipitation increased significantly in all seasons in the Arctic Ecozone⁺, showing increases from 16 to 73% of the base period mean. Fall, winter, and spring precipitation showed signs of a regime shift in the mid-to-late 1970s, while summer appeared to be more stable—though still significantly increasing, by 16% of the base period mean. The abrupt change in precipitation during the fall, winter, and spring in the late 1970s may be related to a change in the atmospheric circulation, specifically a shift in the Pacific North America teleconnection pattern that occurred in 1976 (Brown and Braaten, 1998). Figure 16 shows the magnitude and significance of changes in precipitation for each season at each station.

While it is not yet clear what is responsible for the precipitation changes in Canada, a study by Min et al. (2008) suggests that precipitation increases over Northern Hemispheric high latitudes (north of 55°N) may have been a result of anthropogenic influences on climate. The observed trend towards more precipitation in Canada, especially in the North, is consistent with climate model projections of future changes in precipitation, and is thus likely to continue in the future (Zhang et al., 2011). As projected increases in precipitation are strongly correlated with projected warming, precipitation increases in the Arctic are likely to be much larger than global mean increases (Kattsov et al., 2007).

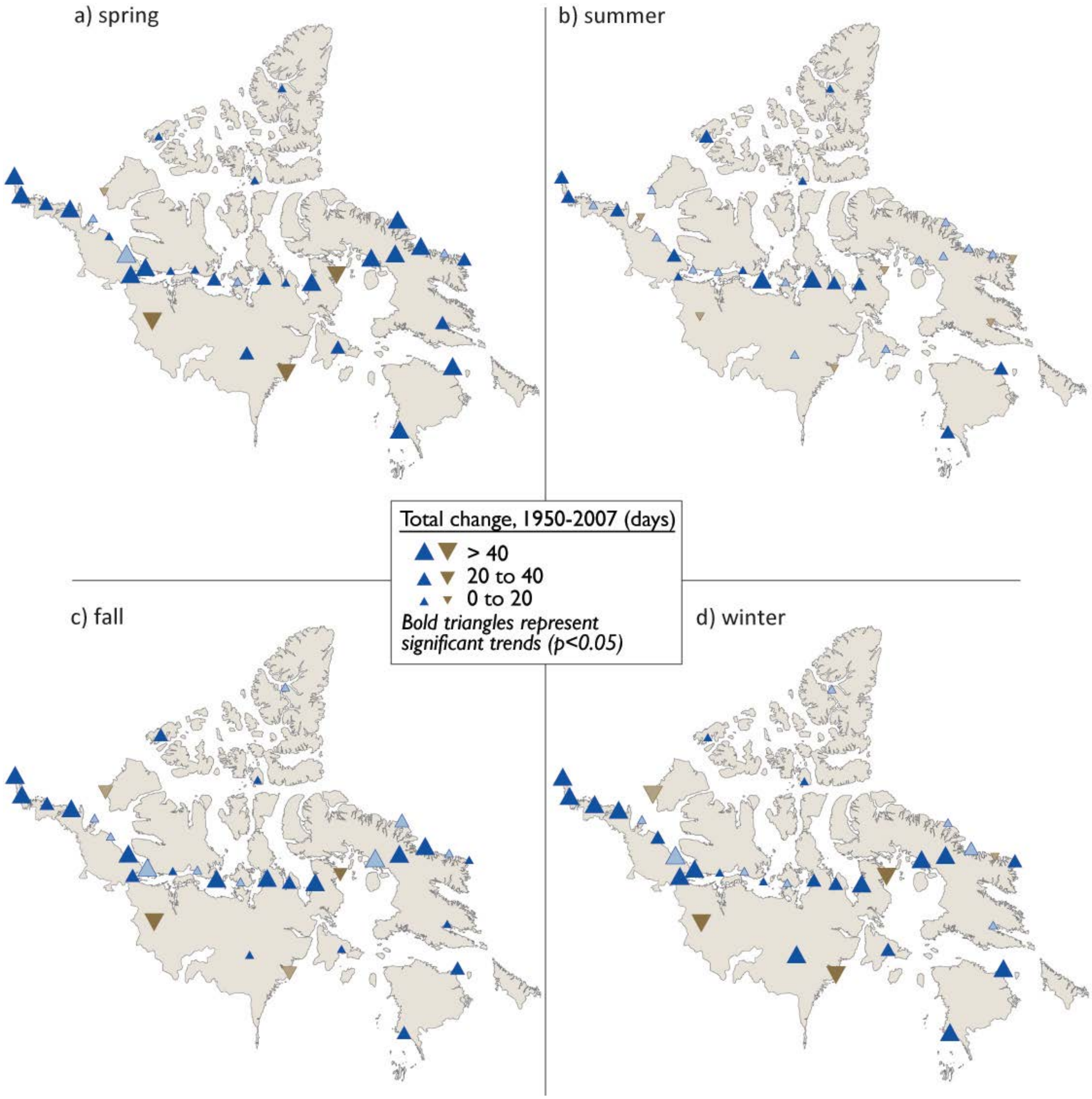


Figure 16. Change in the amount of precipitation, 1950–2007, for: a) spring (March–May), b) summer (June–August), c) fall (September–November), and d) winter (December–February).
Expressed as a percentage of the 1961–1990 mean.
Source: ecozone^{+} data provided by authors of Zhang et al., 2011

The Arctic Climate Impact Assessment (2005) reported an increase in precipitation of 1.4% per decade from 1900 to 2005 for the Arctic (60 to 90°N)—considerably greater magnitude of change than has been observed in this analysis for the Arctic Ecozone^{+}. The larger increases in winter precipitation observed in this analysis are consistent with both model projections and with observations for the Arctic as a whole (ACIA, 2005; Lemke et al., 2007).

Climate oscillations

The section is based on the ESTR thematic report *Large-scale climate oscillations influencing Canada, 1900–2008* (Bonsal and Shabbar, 2011) with the addition of information specific to the Arctic, as noted.

Observed trends and variability in Canadian climate are influenced by large-scale atmospheric and oceanic oscillations known as teleconnections. There are several identified teleconnection patterns used to describe various circulation features across the globe (Bonsal and Shabbar, 2011). Relationships between these teleconnections and Canadian climate are strongest during the cold season (late autumn through spring) although some connections with summer conditions have been identified. In addition, they have strongest and more consistent impacts on temperature variables and, to a lesser extent, on precipitation-related factors. The Arctic Ecozone$^+$ is particularly influenced by the Arctic Oscillation (AO) and by North Pacific patterns of oscillation, for example the Pacific Decadal Oscillation (PDO), though other climate oscillations such as the El Niño/Southern Oscillation (ENSO) also affect Arctic temperatures.

The impact on cold-season temperatures over various regions of Canada has led to significant relationships between teleconnection patterns and several spring climate-related variables. These include the duration of lake and river ice in western Canada (Bonsal et al., 2006), the timing of snowmelt and spring peak streamflow across western North America (Stewart et al., 2005), and the timing of spring runoff in the Mackenzie River (related to the PDO and other indices, but not the AO or ENSO) (Burn, 2008).

Indices that represent these large-scale oscillations are correlated with fluctuations and cycles in Arctic ecosystems. The linkages are not always straightforward, with ecosystem variability often being linked with more than one teleconnection, and with the relationship varying from region to region. Some examples of associations with Arctic biotic communities:

- High values of the AO are associated with periods of reduced plant growth and reduced reindeer populations on Svalbard (Norway) (Aanes et al., 2002).
- Large-scale climate patterns play a role in caribou population fluctuations, but the associations vary from region to region: in Alaska, the PDO is correlated with changes in western Alaska herds, while herds in northeastern Alaska are more affected by the AO (Joly et al., 2011).
- Mortality of greater snow goose young during summer on Bylot Island was shown to be lower at both extremes of the AO: in years when the index is very positive (low summer temperatures) and in years when it is very negative (high summer temperatures) (Dickey et al., 2008).

Arctic Oscillation Index

The Arctic Oscillation (AO) represents atmospheric circulation variability over the Northern Hemisphere north of the tropics where sea-level pressure over the polar regions varies in opposition with that over middle latitudes (about 45°N) (Thompson and Wallace, 1998). It is strongly correlated with the North Atlantic Oscillation (NAO) that influences climate in eastern North America and Europe. When the AO is negative, winter temperatures are lower, ocean waters are fresher, sea ice coverage is greater, and sea ice is thicker than when the AO is positive. During 1950–1980, the AO shifted frequently between positive and negative, and remained positive from 1989 to 1995 (Richter-Menge et al., 2006) (Figure 17). Features of the positive AO phase include warmer-than-average conditions over large regions in northern Europe and Asian Russia, with opposite anomalies in the eastern half of North America (Buermann et al., 2003). The AO has been linked with a variety of physical and ecological features in Arctic and subarctic latitudes. For example, positive AO levels are associated with less winter and spring season snow extent, especially in Eurasia (Bamzai, 2003), earlier melt onset, especially in the European Arctic (Tedesco et al., 2009), higher river discharge to Bering Strait, but lower river discharge to Hudson Bay (Déry et al., 2005), and declines in some Alaskan caribou herds, including the Porcupine Herd whose range extends into Canada (Stenseth et al., 2003).

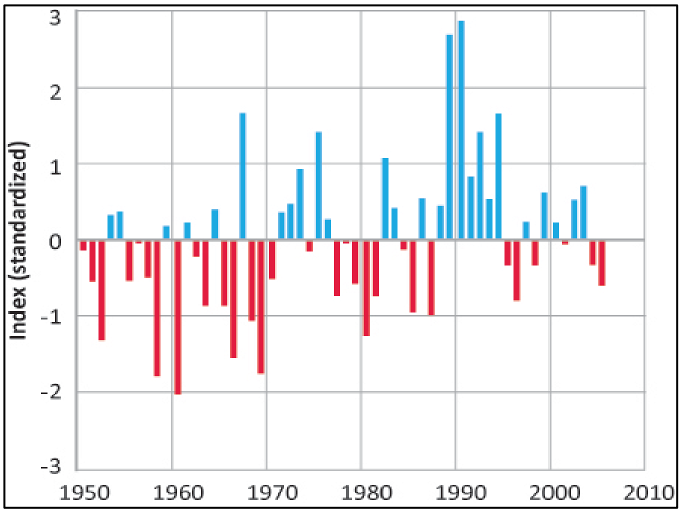


Figure 17. Time series of the annually averaged Arctic Oscillation Index (AO) for the period 1950–2005. Source: based on Figure 5 from Richter-Menge et al., 2006 (courtesy of I. Rigos)

The different phases in large-scale teleconnections often lead to larger-amplitude regional responses in observed climate. As a result, a number of the observed 20th century climate changes can be attributed, at least in part, to changes in these various teleconnection patterns (Solomon et al., 2007). For example, Hurrell (1996) found that the North Atlantic Oscillation, ENSO, and variations in North Pacific circulation collectively explained a significant portion of Northern Hemisphere winter temperature variability during the 20th century. Specifically for North America, the mid-1970s climate shift to positive Pacific Decadal Oscillation and more frequent El Niño events appear to have led to contrasting changes across the continent, with the west warming more than the east for a period thereafter (Trenberth et al., 2007). This shift to a positive PDO has therefore also been associated with the trend toward warmer winter and spring temperatures over western Canada, including in the Arctic.

It is not currently known how increases in greenhouse gas concentrations have affected the occurrence of these large-scale climate oscillations. Furthermore, the effects of projected future climate change on the major teleconnection patterns remain uncertain since there is a lack of agreement among the various climate models concerning the future frequency and structure of large-scale atmospheric and oceanic modes. Climate oscillations do not account for recent warming patterns in the Canadian and circumpolar Arctic (Figure 18).

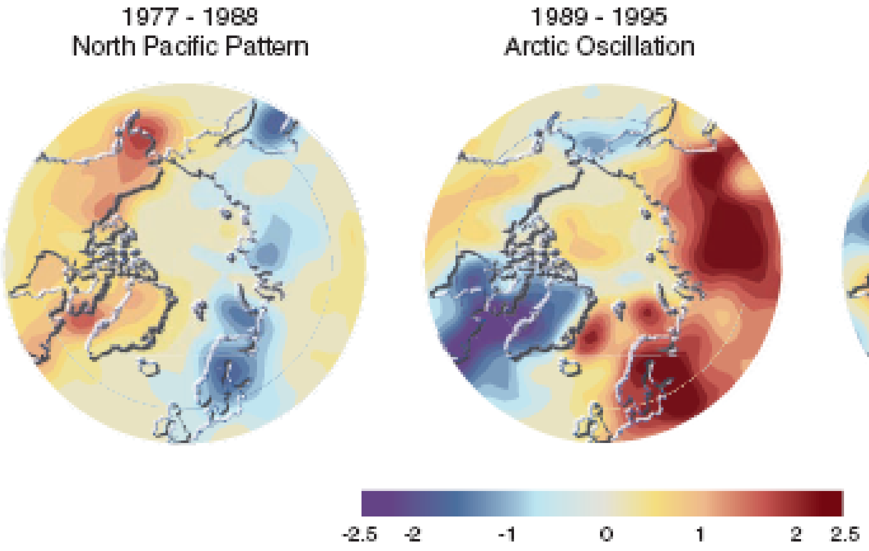


Figure 18. Northern Hemisphere surface temperature anomalies averaged over three periods with different types of dominating pattern of natural variability.
The North Pacific pattern was dominant in the Arctic from 1977 to 1988, while the Arctic Oscillation dominated the region from 1989 to 1995 (see Figure 17). In recent years (the period from 2000–2005), however, neither of these alternate states is evident and the pattern of warm temperature anomalies is circumpolar in distribution.
Source: Overland and Wang, 2007

Ecosystem functions/processes

Introduction

In 1973, Dunbar published a landmark paper, *Stability and fragility in Arctic ecosystems*. He examined the apparent dichotomy that Arctic ecosystems are susceptible to damage, even though species that live there all year are extraordinarily tough. Others migrate huge distances and habitats undergo extreme annual fluctuations in living conditions. Arctic terrestrial animals, such as lemmings , can crash to extremely low population levels or even be locally extirpated, but can recover as long as there is a vast area available in which to harbour recolonizing stocks. Species are protected from invasion by ecological equivalents, as well as their predators and pathogens, and by climatic limitations that maintain tundra habitats. He pointed out that “stability” has to be defined in terms of coherence and continuity of annual and multi-annual cycles, and that “fragility” relates not so much to low resistance to small-scale physical damage as to restricted growth and turnover rates that limit resilience. He gave this example of vulnerability due to slow turnover:

> *"If an Arctic lake full of char or lake trout is fished out in a season or two, which has happened many times, the damage is either permanent or will take many years to repair... Lakes, especially small lakes, in the Arctic, are probably the most vulnerable part of the landscape... Many of them harbour only one species of herbivore (a copepod crustacean), and one species of primary carnivore, usually a mysid or an amphipod. These are analogous to the lemming of the tundra, but without the protection of large geographic scale. They represent extreme cases of ecosystem simplicity, in which the removal of one species means ecological disaster."*

Since then, our understanding of resilience has improved, so that, for example, we know that, when populations drop below a minimum viable population level (a threshold), their recovery becomes dependent on conditions that enhance resilience for that species: for example, low mortality from predation or harvest, or sufficient range size and sufficient connectivity to allow other populations to act as sources for recolonization. Stochastic events, such as unusual winter storms, industrial disturbance, or introduction of a new predator or pathogen, may pose high probabilities of extinction or extirpation. Peary caribou (*Rangifer tarandus pearyi*) are a case in point, as discussed in the Ecosystem composition - Peary caribou section (page 105). After a decade of intensive research, scientists are beginning to understand how a changing climate may destabilize Arctic ecosystems—and, indeed, have already detected signs of destabilization—but we have much to learn. The science on cumulative impacts, including understanding of interactions between climate change and industrial disturbance, is in its infancy. The following sections focus on processes that may indicate or foretell ecosystem perturbations. Two important characteristics of the Arctic Ecozone⁺ strongly influence the resilience of its ecosystems to perturbation: 1) the huge scales (temporal and geographical) at which most Arctic ecosystems operate (at least for vertebrates) in the processes of annual movements, population cycles, and recolonization events; and 2) the dominant evolutionary role of winter in driving adaptive traits and habitat structure. Understanding the influence of scale the specific adaptations and habitat structures characteristic of Arctic wildlife, in the light of current change, is essential for conservation and enhancement of Arctic ecosystem resilience.

The Cryosphere

Changes to the cryosphere (permafrost, snow, sea ice, glaciers, and lake and river ice) could also be classified as abiotic drivers (secondary to the primary climatic drivers of temperature and precipitation). We include these trends in the Ecosystem processes section to highlight and discuss their linkages with ecosystem disturbance and other ecosystem processes such as hydrology and primary productivity. Rapid shrinking of all aspects of the cryosphere has been a dominant trend in Arctic ecosystems that began over 30 years ago and is particularly evident over the past decade. The implications of this loss of frozen water and frozen ground, and the interconnections with other ecosystem processes, are profound and, in many cases, poorly understood. A major source of the increasingly high rate of change cryosphere shrinkage is the positive feedback resulting from reduction in surface reflectivity (albedo) due to loss of ice and snow. This reduction in albedo increases the capacity to absorb and store heat in Arctic ecosystems, enabling more melting. In the face of projections for continued global warming, further losses and changes in timing and other characteristics of ice and snow in their various forms are expected (AMAP, 2011; Jeffries et al., 2012).

Permafrost

The sections below on permafrost trends are based on *Trends in permafrost conditions and ecology in northern Canada* (Smith, 2011), an ESTR national thematic report. Some information has been updated, as noted.

Information on recent trends in permafrost temperatures in the Arctic Ecozone$^+$ comes from a number of monitoring sites from the Western Arctic to the Eastern Arctic and the High Arctic. In general, changes in shallow permafrost temperatures over the last decade are greater in the Arctic Ecozone$^+$ compared to areas below the treeline (taiga and boreal ecozones$^+$). Tundra vegetation, with less of a vertical structure than forest, allows more direct light penetration to the soil and provides less insulation against re-radiation of absorbed heat. The presence of colder permafrost also means that phase change and the presence of unfrozen water do not obscure the climate signal. There is, therefore, a more direct link between changes in air temperature and changes in permafrost temperature.

Western Arctic

Permafrost temperature data collected since the 1980s from the northern Mackenzie Basin (Smith et al., 2005b) indicate that warming of permafrost has occurred since the early 1990s. On the Tuktoyaktuk Peninsula for example, at a depth of 28 m, permafrost temperatures increased between 1990 and 2002 at a rate between 0.02 and 0.06°C per year. Analysis by Burn and Kokelj (2009) indicates that near-surface ground temperatures in the tundra uplands of the Mackenzie Delta region increased by 1 to 2°C from the early 1970s to 2007. Modelling analysis conducted for a permafrost monitoring site at Herschel Island (in the Beaufort Sea off the Yukon coast) indicates that permafrost temperature at a depth of 20 m has increased by 1.9°C over the past 100 years (Burn and Zhang, 2009). Recent field observations at this monitoring site also indicate an increase in active layer thickness since 1985.

Central Southern Arctic

Permafrost temperatures to 3 m depth have been collected since 1997 at Baker Lake, Nunavut. Between 1997 and 2007 a general increase in thaw depth (Figure 19) has been observed, although there is some interannual variability within the short record (Smith et al., 2005b; Throop et al., 2008). The largest increase in thaw depth occurred between 1997 and 1998 and this was related to the longer thaw season in 1998 (Smith et al., 2001b).

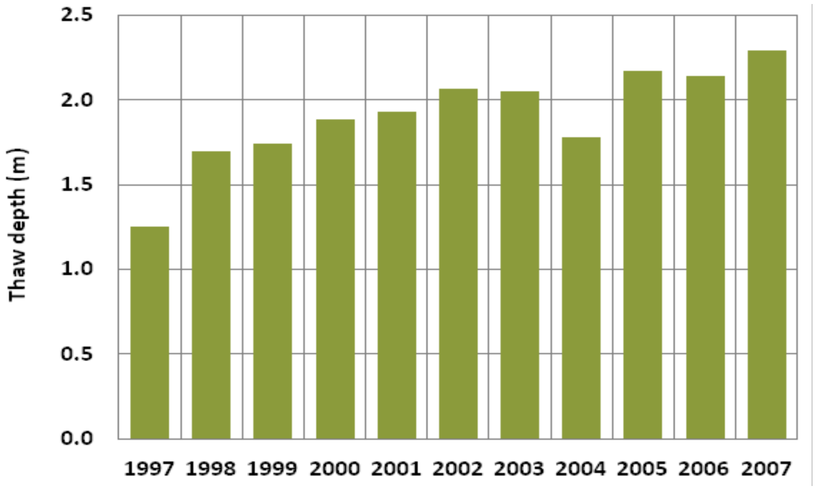


Figure 19. Maximum summer thaw depth for a site at Baker Lake, 1997-2007.
Source: Smith, 2011, adapted and updated from Smith et al., 2005b and Throop et al., 2008

Eastern Arctic

Permafrost temperature data collected since 1978 at CFS Alert Nunavut can be used to characterize trends in permafrost in the High Arctic. Although air temperatures have increased since the 1980s, distinct warming in shallow permafrost temperatures has only been observed since the mid-1990s. Between 1994 and 2001, an increase in permafrost temperatures of about 0.15°C per year occurred at a depth of 15 m (Smith et al., 2005b). Although some cooling of permafrost was observed between 2000 and 2002, more recent data indicates that warming of permafrost is continuing.

Figure 20 shows trends in annual mean temperatures at two depths and two boreholes up to 2011. Increases in winter temperatures appear to be the main factor in the recent ground temperature increases, especially at High Arctic sites with relatively low snow cover (and thus little insulation of the ground during winter) (Smith et al., 2012).

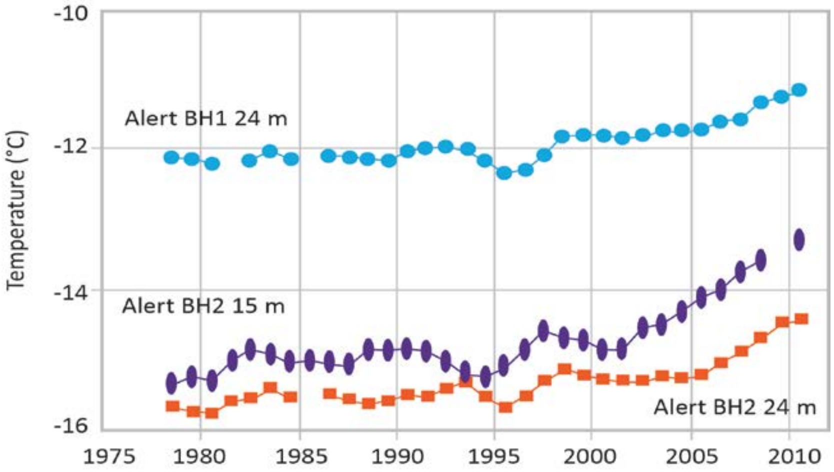


Figure 20. Mean annual ground temperature at two boreholes and depths at Alert, Nunavut, 1978-2011. Data standardized and data gaps addressed following methods in Smith et al.*, 2012.*
Source: adapted from Romanovsky et al.*, 2012*

Shallow permafrost temperatures at Iqaluit also began to warm in 1993, with warming continuing through the 1990s (Figure 21). Temperatures of shallow permafrost (measured at a depth of 5 m) increased at a rate of 0.4°C per year between 1993 and 2000. A similar trend has been observed in northern Quebec, where cooling of about 0.10°C per year was observed between the mid-1980s and mid-1990s at a depth of 10 m (Allard et al., 1995). An increase in air temperatures commencing in 1993 in northern Quebec has been associated with warming of permafrost since 1996 to depths of 20 m (Allard et al., 2002; Ouranos, 2004; Chouinard et al., 2007) and an increase in active layer thickness (Brown et al., 2000).

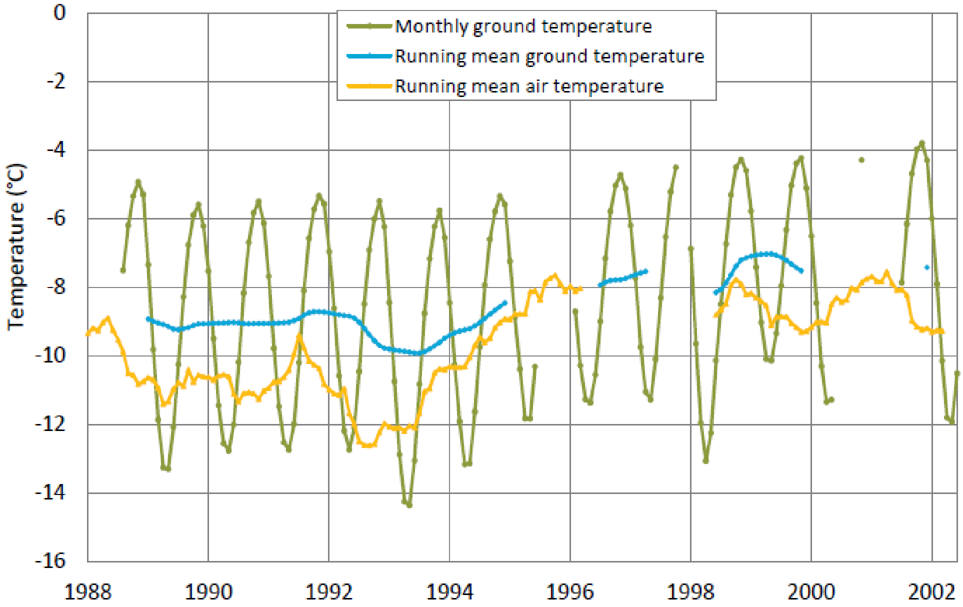


Figure 21. Monthly ground temperature at a depth of 5 m at Environment Canada's borehole at Iqaluit, 1988-2002.
The 12-month running means for ground and air temperatures are also shown.
Source: Smith, 2011, adapted from Smith et al., 2005b

Recent increases in shallow ground temperatures have also been observed in other Arctic regions, such as Scandinavia and Svalbard (Isaksen et al., 2007a and 2007b; Harris and Isaksen, 2008). Although snow cover is generally thin at these High Arctic sites, its variability can be an important factor affecting the response of permafrost temperatures to changes in air temperature (Smith et al., 2003). Changes in snow cover may counteract changes in air temperature occurring over the same period such that permafrost temperatures may increase in the High Arctic during periods of higher snow cover but lower air temperature (Taylor et al., 2006).

Ecological consequences of changes in permafrost

The observed trends of warming permafrost and increased thickness of the active layer are projected to continue and accelerate as the Arctic climate warms (ACIA, 2005; Intergovernmental Panel on Climate Change, 2007). The implications for ecosystem processes and ecosystem structure are huge. Melting of permafrost is also implicated in feedback loops that: 1) lead to increases in air temperatures in the region through changes in vegetation communities and albedo (Chapin III et al., 2005); and 2) lead to increases in warming at the global scale through changes in the carbon balance of the Arctic landscape. Research provides clear links between permafrost conditions and many ecosystem characteristics. Though it can be hard to demonstrate and measure trends in these interlinked impacts at the level of change to populations and ecosystem services, the observed trends in permafrost are likely now influencing Arctic ecosystem processes, structure, and composition. Warming and melting of permafrost projected for the decades to come will be accompanied by major ecosystem change. The discussion below outlines some of the ways in which permafrost drives and influences ecosystem processes. See also the section on Permafrost disturbance (page 52).

Patterned ground caused by permafrost provides diversity in microclimates for Arctic plants and arthropods (Strathdee and Bale, 1998; Vonlanthen et al., 2008; Walker et al., 2008a). The annual freeze-thaw cycles of the active layer lead to ground patterns such as frost boils and ice-wedge polygons which are a defining feature of the tundra. Frost boils are patches of barren or sparsely vegetated soil where differential frost heave forms distinct patterns of stones and sediments (Walker et al., 2004). Frost boils cycle organic matter and nutrients vertically within the boil (Walker et al., 2004) and are favourable sites for seed germination (Sutton et al., 2006), contributing to the diversity and amount of vegetation available to herbivores (Walker et al., 2001).

One of the most productive tundra ecosystems is non-acid tussock tundra which is a key to migratory tundra caribou herds and their associated predators (Walker et al., 2001). The productivity is driven by the freeze-thaw cycles of the active layer which induces the migration of cations essential to plants. The near-surface ice-rich zone of permafrost is a sink for cations such as calcium and magnesium which reach the active layer and become available for plant growth. When permafrost melts, more of these nutrients are contributed to the active layer (Kokelj and Burn, 2005).

The active layer is kept moist to water-logged by underlying permafrost. Correspondingly, microbial activity is reduced and the microbial release of nitrogen from decomposing organic matter is slow, limiting plant growth (Weintraub and Schimel, 2005). Net primary productivity is related to nutrient availability and is highest on deeply thawed soil, dominated by nitrogen fixers and disturbed by animals that release nutrients.

The depth of the active layer varies from tens of centimetres to 1 to 2 metres and constrains the activities of burrowing animals in summer. Frost boils and frost cracks are preferred sites for lemming burrows (Potter, 1972), possibly because they are warmer than burrows deeper in the active layer. Lemmings spend most of their time in their burrows in summer to reduce the risk from avian predators (Boonstra et al., 1996). The burrows and associated latrines add to the micro sites of disturbed ground and enhanced nutrient availability for plant growth. Other burrowing mammals, including Arctic ground squirrels (*Spermophilus parryii*), select sites without permafrost, such as river banks (Batzli and Sobaski, 1980).

Permafrost is the main factor influencing where Arctic foxes (*Vulpes lagopus*) den. The foxes select south-facing, well-drained sites with early snowmelt, conditions indicative of a deeper active layer (Szor et al., 2008). Pregnant polar bears denning on the Hudson Bay coast select thawed peat banks to dig dens. The re-use of the dens over hundreds of years as measured by annual tree ring chronologies suggest the importance of the thawed peat banks for the bears (Scott and Stirling, 2002).

Recent techniques have revealed bacteria and fungal spores within permafrost (Steven et al., 2008). Permafrost's frozen sediments and ice have carbon dioxide, methane, oxygen, and nitrogen in pore spaces, as well as a thin film of water, and evidence points to active microbial ecosystems within permafrost (Steven et al., 2006). This has implications for global warming. Steven et al. (2006) drew attention to "the considerable microbial biomass believed to inhabit the significant areas of terrestrial permafrost, [and] this may have important implications on global nutrient cycling and biogeochemical processes, such as the C, N, and S cycles".

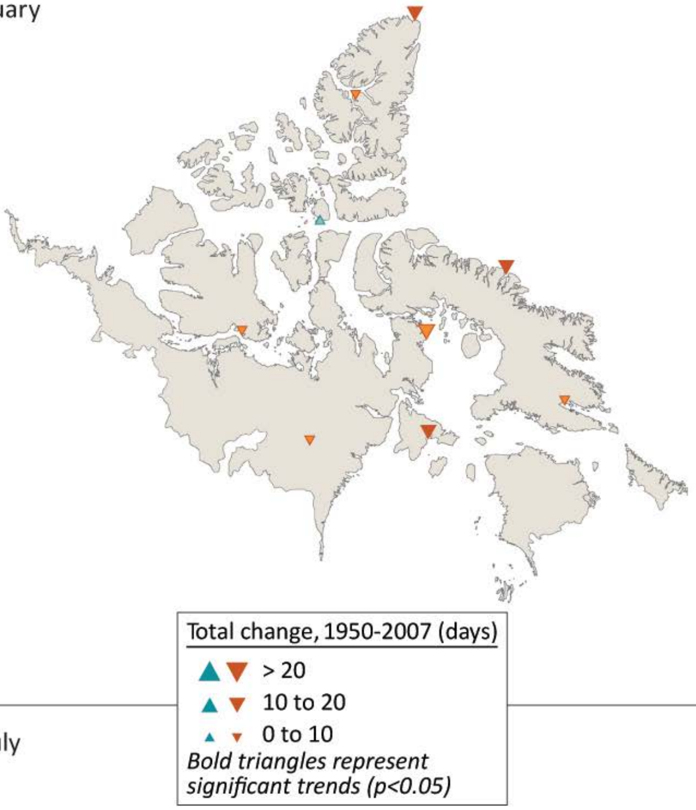
Snow

This section is based on analysis of ground-based snow cover measurements for Canada as a whole and for the Arctic Ecozone$^+$ as analysed for the ESTR report *Canadian climate trends, 1950–2007* (Zhang et al., 2011), with reference to additional studies, as noted.

Snow cover duration has significantly decreased almost everywhere in Canada, with the greatest changes occurring in spring over western and northern Canada. This is a consequence of warmer springs observed over these regions, and is part of a Northern Hemispheric trend toward spring warming and earlier melting of snow and ice (Lemke et al., 2007; Derksen and Brown, 2012).

Across the Arctic Ecozone$^+$, snow cover duration decreased by an average of 9 days in the fall and 8.5 days in the spring from the 1950/51 to the 2006/07 snow seasons. Figure 22 shows the fall and spring trends in snow cover duration at each station. Annual maximum snow depth also decreased by an average of 13.2 cm over the Arctic Ecozone$^+$ from 1950 to 2007 (Figure 23). This is part of a general tendency to lower maximum snow depths over Canada, but is less significant and less consistent from place to place than the decreases in snow cover duration. In southern Canada, the decrease in annual maximum snow depth is linked to a trend toward a smaller fraction of annual precipitation falling as snow. However, there were no significant trends in this ratio when averaged across the Arctic Ecozone$^+$—in fact, increases in the ratio were recorded at many stations (Figure 24), in line with the greater increase in precipitation during the snow season than during summer (Figure 16). The shorter snow season (later onset of snow cover and earlier melt) is likely offsetting the effects of this increase in cold-season precipitation. This is consistent with analysis by Brown and Mote (2009), which shows that, under a scenario of warming and increasing precipitation, maximum snow accumulation still decreases.

a) August to January



b) February to July

Figure 22. Change in snow cover duration (the number of days with greater than or equal to 2 cm of snow on the ground), 1950–2007.
Map a (the first half of the snow season) indicates changes in the start date of snow cover. Map b (the second half of the snow season) indicates changes in the end date of snow cover.
Source: ecozone$^+$ data provided by authors of Zhang et al., 2011

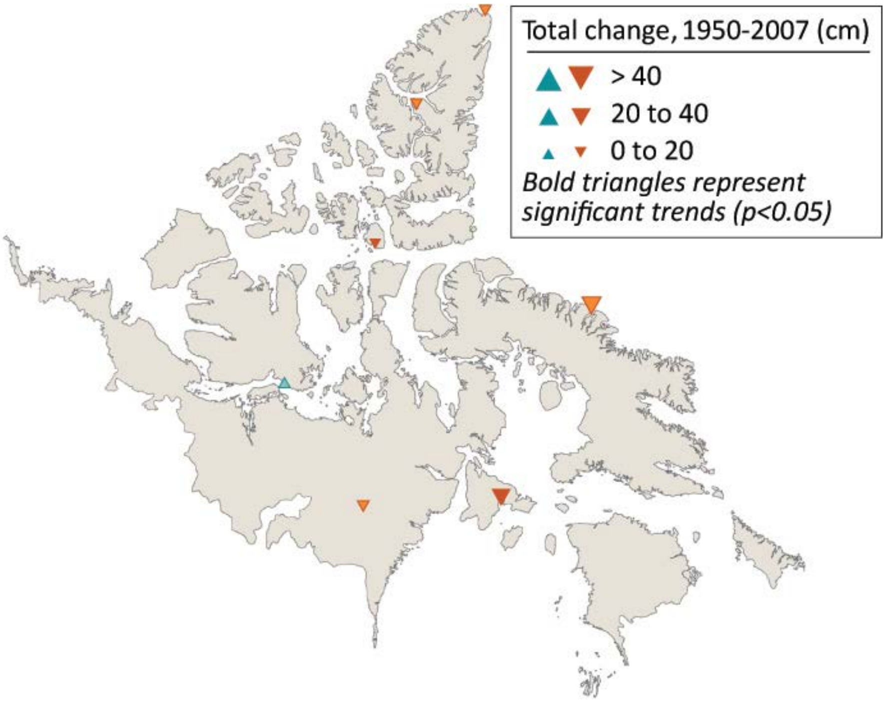


Figure 23. Change in the maximum annual snow depth, 1950–2007.
Source: ecozone$^+$ *data provided by authors of Zhang et al., 2011*

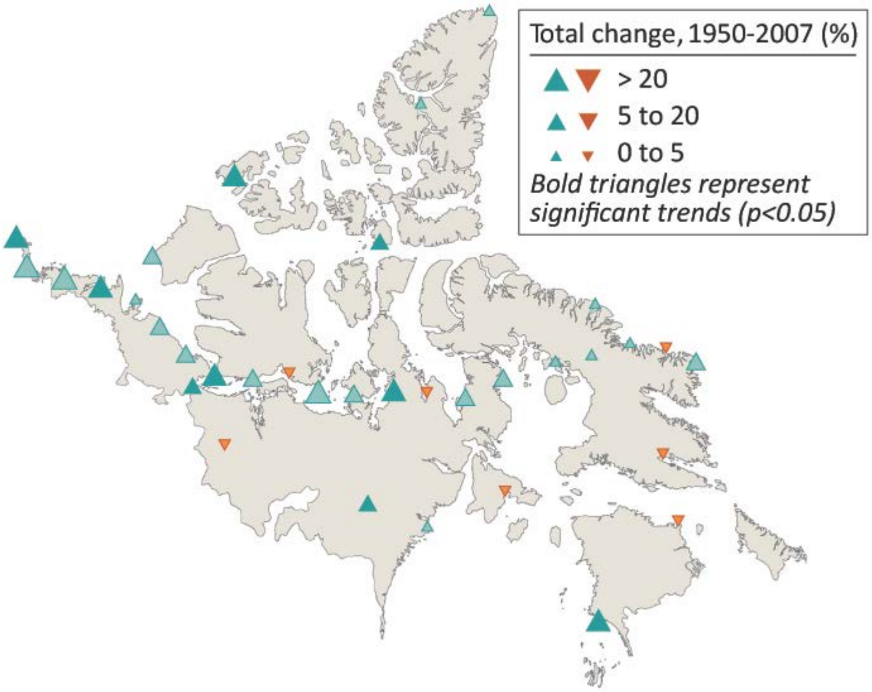


Figure 24. Absolute change in the ratio of snow to total precipitation in Canada, 1950–2007.
Increasing trends indicate an increase in the proportion of precipitation falling as snow.
Source: ecozone$^+$ *data provided by authors of Zhang et al., 2011*

The Arctic snow depth trend results include greater uncertainty than snow cover duration because: 1) snow depths exhibit much stronger local-scale variability than snow cover duration (Brown et al., 2007b); and 2) the snow depth observing network in the Canadian Arctic is sparse and biased to coastal locations. In addition, snow depth measurements are made at open sites near airports that may not be representative of surrounding terrain. With increasing shrubbiness in the Arctic, one would expect to see more snow being retained on the ground in areas of shrub expansion. Snow cover duration, however, is less affected by station conditions.

Circumpolar snow trends

The greatest and most rapid decreases in snow water equivalent and snow cover duration have occurred in high-precipitation maritime regions of the Arctic (Callaghan et al., 2011a). There are also differences between Eurasia and North America: declines in snow depth and snow cover tend to be more recent in Eurasia, mainly after about 1980, while declines are evident since the 1950s in North America. Snow depth, however, is increasing in parts of Eurasia (Callaghan et al., 2011a). Arctic spring snow cover in both Eurasia and North America has decreased greatly, especially over the past decade (Derksen and Brown, 2012) (Figure 25).

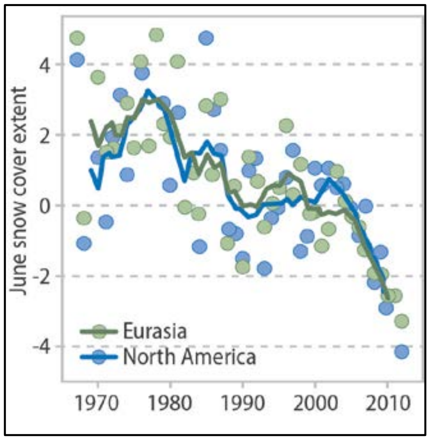


Figure 25. Trends in terrestrial spring snow cover for the Eurasian and the North American Arctic, 1967–2012. Data are June averages for all Northern Hemisphere snow cover, which is virtually all in the Arctic. Values are standardized anomalies with respect to the 1988–2007 mean. Solid lines are five-year running means. Source: CAFF, 2013, based on Derksen and Brown, 2012, data from NOAA CDR (satellite monitoring)

Ecosystem consequences of changes in snow

A reduction in duration of snow cover has been shown to have significant positive feedbacks to the Arctic climate system through reduced albedo (Chapin III et al., 2005; Screen and Simmonds, 2012). This mechanism is stronger over tundra than over taiga because of the shallow snowpack, high amounts of incoming solar radiation in the spring melt period, and the characteristics of the vegetation. Tundra vegetation has little vertical structure and therefore little capacity to shade the ground. Chapin et al. (2005) concluded that reductions in the length of the snow-cover period in Alaska over the past few decades have contributed substantially to local atmospheric heating in the summer—increasing heating by about 3 Watts per m^2 per decade. Across the entire Arctic region, feedback from changes in snow cover over the period 1970 to 2000 was simulated to have increased atmospheric heating by 0.9 Watts per m^2 per decade (Euskirchen et al., 2007). Natural and anthropogenic impurities in snow are also contributing to reduced albedo. Flanner et al. (2008) show that the surface deposition of black carbon (soot particles) reduces snow surface albedo (a process termed "darkening"), which contributes to melting of snow and ice and increases atmospheric heating.

Snow characteristics, including extent, depth, duration, and timing—but also the characteristics of the snow pack determined by events such as freeze-thaw cycles—strongly influence Arctic terrestrial and freshwater ecosystem processes including permafrost dynamics, hydrology, and primary production. Snow characteristics also directly affect biotic communities and many Arctic species. Observed and predicted effects from changes in snow are assessed in recent Arctic Council projects (AMAP, 2011; Callaghan et al., 2011b; CAFF, 2013) and discussed in several sections of this report.

Sea ice

Status and trends of sea ice are discussed more fully in the Arctic Marine report (Niemi et al., 2010). A summary is provided here because of the importance of sea ice to some of the animals discussed in this report (notably polar bears) and because of the implications to Arctic terrestrial ecosystems of the major changes now occurring in extent and quality of sea ice.

Sea ice extent throughout the year has decreased significantly over the period of record, as measured by remote sensing (Figure 26). The summer minimum ice cover (occurring in September) has declined particularly rapidly in the past few years (Figure 27). Extending the record further back, using ice observation networks and proxy data, indicates that the recent and current rate and extent of decline of summer ice extent is unprecedented at least over the past 1,450 years (Kinnard et al., 2011). Increased summer ice melting has led to a loss of multi-year ice. The percentage of the winter (March) ice cover composed of first-year ice increased from 58% in 1988 to 75% in 2012, while the percentage of ice aged four or more years decreased from 26% in 1988 to 7% in 2012 (Perovich et al., 2012).

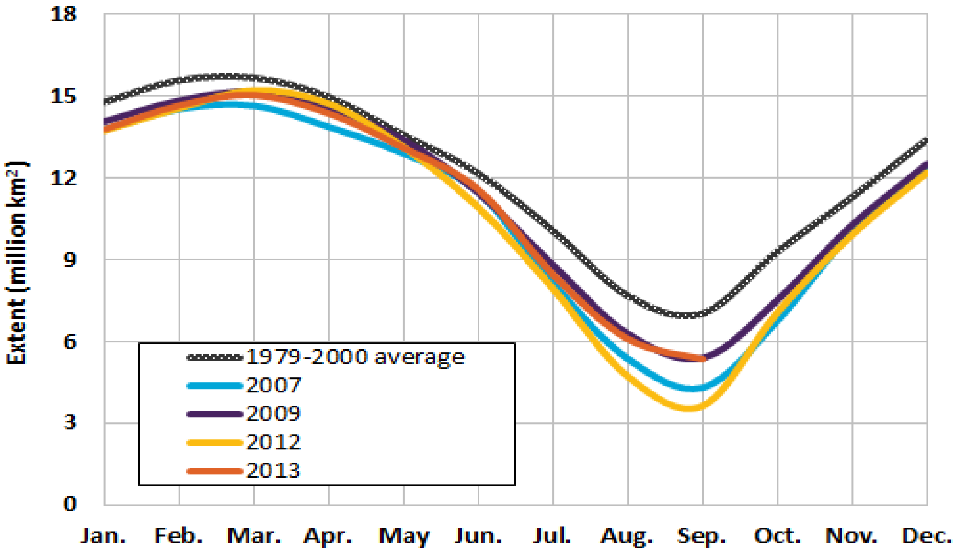


Figure 26. Extent of Arctic sea ice over the annual cycle, recent years compared to 1979–2000 average. Source: updated from Fetterer et al., 2010 with data from National Snow and Ice Data Center, 2013

Climate modellers project ice-free summers in the Arctic within 30 years (Wang and Overland, 2012). During an ice-free summer, there would still be remnants of land-fast ice and likely ice in sheltered waters between islands of the Canadian Arctic Archipelago (Wang and Overland, 2012). The spatial distribution of sea ice in winter and summer 2012 is shown in Figure 28.

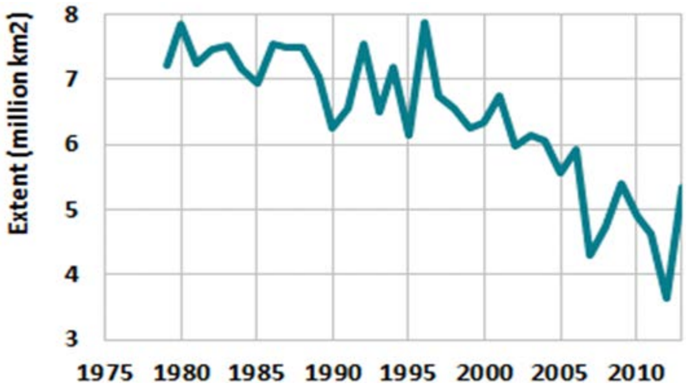
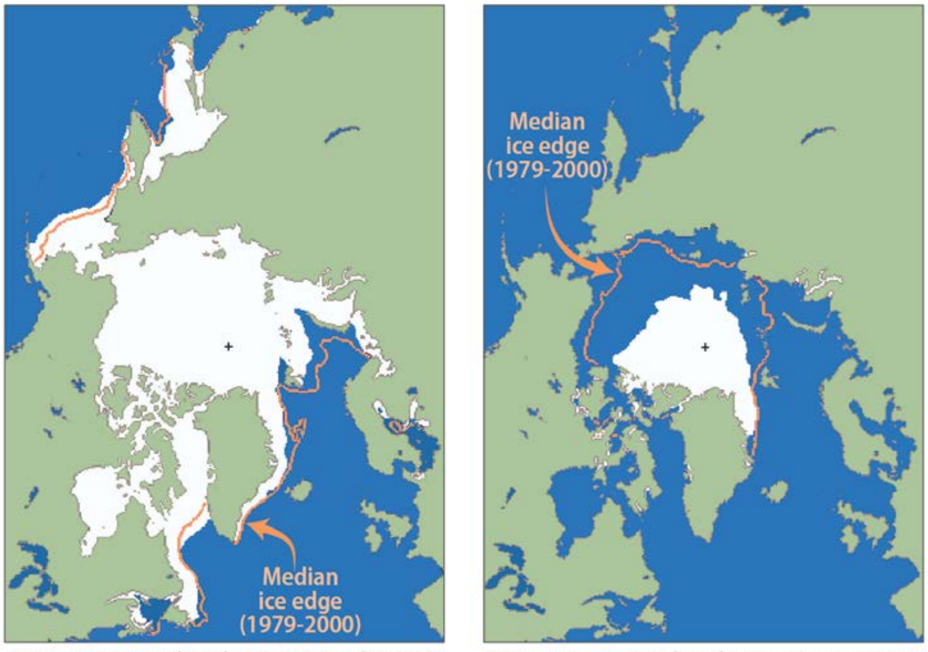


Figure 27. Trends in mean September sea ice cover, 1979–2013, showing the recent rapid decline. The September 2012 minimum ice extent was 48% below the 1979–2000 average. The cooler summer of 2013 resulted in a slightly greater ice minimum extent, similar to 2009 and 24% below the 1979–2000 average.
Source: updated from Fetterer et al., 2010 with data from National Snow and Ice Data Center, 2013



Winter sea ice distribution (March 2012)

Summer sea ice distribution (Sept. 2012)

Figure 28. Distribution of winter and summer sea ice, 2012, and median ice edge for both seasons.
Source: Eamer et al., 2013 based on data from National Snow and Ice Data Center, 2013

Ecological consequences of rapidly changing sea ice

Major consequences of this regime shift in Arctic sea ice for Canada's Arctic marine ecosystems are covered in the ESTR report *Ecosystem status and trends report: Arctic marine ecozones* (Niemi et al., 2010). Consequences at the circumpolar and global scale were assessed through recent Arctic Council projects (AMAP, 2011; CAFF, 2013; Eamer et al., 2013).

The extent and quality of sea ice cover has major implications for terrestrial ecosystems and wildlife, as well as for people. Open water has a much lower albedo (reflectivity) than ice, and consequently absorbs more sunlight. Reduction of sea ice has led to warming in adjacent coastal areas (Serreze and Barry, 2011) with consequences for tundra vegetation (Bhatt et al., 2010), as well as increased coastal erosion, affecting coastal habitats and coastal communities, especially along the Beaufort Sea coast (Lantuit and Pollard, 2008; AMAP, 2011). Sea ice is a major platform for travel and hunting for Inuit, and changes over the past decade have affected communities (Niemi et al., 2010; Eamer et al., 2013).

While the polar bear is the species (of those covered in this report) most clearly at risk from the decline of sea ice, terrestrial animals are also vulnerable. This discussion below on sea ice and tundra ecosystems is based partly on the Conservation of Arctic Flora and Fauna (CAFF) report *Life linked to ice: A guide to sea-ice-associated biodiversity in this time of rapid change* (Eamer et al., 2013).

Winter sea ice provides foxes and wolves with a means to access and colonize remote islands. A study of the genetics of Arctic foxes throughout their circumpolar range showed that the occurrence of sea ice is the main factor in determining how similar fox populations are to one another: the less sea ice, the more genetically distinct are the populations (Geffen et al., 2007). If island populations become isolated due to loss of winter ice, some populations will be at risk of decline or extinction due to loss of genetic diversity and inbreeding (Geffen et al., 2007; Noren et al., 2011). Wolf (*Canis lupis*) populations on Banks, Ellesmere, and Devon islands have declined and been reestablished in the past through colonization over the sea ice from other islands (Carmichael et al., 2008). As with foxes, wolf populations are at risk of reduction if winter ice conditions alter sufficiently to compromise movement over ice. As winter sea ice is expected to persist in the region of the Canadian Arctic Archipelago, this potential impact is more likely to occur in other circumpolar regions, including parts of the European and Russian Arctic.

Changes in sea ice also affect the movements of some caribou herds. Peary caribou have been observed moving from island to island over the sea ice (Jenkins et al., 2011). The Dolphin and Union Herd migrates annually across the Dolphin and Union Strait between Victoria Island and the mainland, one of the areas where the expansion of open water has been most extensive (COSEWIC, 2004). Late freeze-up of the strait will delay migration to the winter ranges.

Both Arctic foxes and wolves feed on marine resources during winter, travelling long distances to hunt and scavenge (Tarroux et al., 2010). Arctic foxes scavenge seals killed by polar bears (Roth, 2003; Geffen et al., 2007) and may also prey on ringed seal pups, as observed on sea ice off the coast of northern Labrador (Andriashek et al., 1985). They may rely heavily on marine foods during years of low lemming abundance (Tarroux et al., 2012). This suggests that the loss of sea ice could affect coastal populations of Arctic foxes during years of low terrestrial food abundance (Pamperin et al., 2008). Wolves are also known to forage on sea ice in winter, likely scavenging seal carcasses, based on an Alaskan study (Watts et al., 2010). The main changes in sea ice and their potential impacts on these primarily terrestrial carnivores are: 1) delayed onset of ice cover in autumn, leading to the carnivores having to spend longer on land in what could be a relatively food-poor situation (for example, when rodents are at low densities and birds have left); and 2) reduced summer ice cover leading to reduced polar bear populations and possibly reduced seal populations, and therefore reduced density of scavenging or hunting opportunities for the terrestrial carnivores when they are on the ice.

Recent studies using satellite tracking show that both gyrfalcons (Burnham and Newton, 2011) and snowy owls (Therrien et al., 2011) spend substantial periods of time far offshore in winter, presumably preying on marine birds and roosting on ice floes. Snowy owls have been observed hunting seabirds that congregate in patches of open water in Hudson Bay in winter (Gilchrist and Robertson, 2000). Satellite tracking over two winters (2007–2009) of nine adult female snowy owls that had been fitted with transmitters while on their summer range on Bylot Island showed that most spent several weeks on the sea ice between December and April. Analysis of high-resolution satellite images showed that they spent most of their time on the ice around open-water patches frequented by seabirds (Therrien et al., 2011). Changes in winter sea ice extent and polynya formation that affect seabirds could also affect the food intake of snowy owls. As owls are small-mammal predators during breeding season, changes in snowy owl populations would affect the tundra food web (Therrien et al., 2011).

Glaciers and ice caps

During the past 10,000 years, glaciers and ice caps—the remnants of the great ice sheets of the last ice age—have been melting, with intermittent periods of re-advance. The last major advance was during the Little Ice Age from the 14th to 19th centuries. Since then, glaciers around the globe have been shrinking, with increasing rates of ice loss since the mid-1980s. On a time-scale of decades, glaciers in some regions (including the Arctic) have shown intermittent advances, but overwhelmingly the trend is to increasingly rapid melt. The outlook, based on climate change scenarios, is for deglaciation of many mountain regions within the 21st century (Gerland et al., 2007).

About 300,000 km^2 of glaciers and ice caps (excluding Greenland) drain to the Arctic Ocean, most directly (Dyurgerov and Carter, 2004). Over half of this areal extent of land ice is in Canada's Arctic Archipelago (Table 3). These glaciers and ice caps fluctuate in mass and extent with temperature and snowfall (UNEP and WGMS, 2008; Colgan and Sharp, 2008), with the strongest linkage being between mass balance and air temperature (Koerner, 2005). A second important control on extent and volume changes of land ice has been iceberg calving that occurs as flux across the grounding line and retreat/advance of tidewater margins. Iceberg calving accounts for a large portion (up to 40%) of mass loss from ice caps in the Canadian Arctic (Burgess et al., 2005; Mair et al., 2009; Mair, 2012).

Glaciers in the Canadian Arctic Archipelago reached their last maximum extent relatively late: in the latter part of the 19th century. Time series of glacier fluctuations are sparse, with the only long-term mass balance series being for the White and Baby glaciers and the Devon, Meighen, and Melville ice caps and Agassiz Ice Field, all in the Queen Elizabeth Islands. Based on these time series and other evidence, there has been a general melting trend for Canadian Arctic Island glaciers since the late 19th century, with the trend slowing down for a period in the mid-20th century (UNEP and WGMS, 2008), and with accelerated loss in the past 25 years. Analysis of ice cores from the Devon Ice Cap (Devon Island), Penny Ice Cap (Baffin Island), and the Agassiz and Prince of Wales ice fields (Ellesmere Island) indicates that the melt rates since the mid-1990s are the highest in at least the last two millennia (Fisher et al., 2012). The longer record for the Agassiz Ice Field shows that this recent melt rate is the highest in 4,200 years and resembles the melt rate during the Holocene thermal maximum over 9,000 years ago (Fisher et al., 2012).

Table 3. Surface area of glaciers and ice caps, Canadian Arctic Archipelago, mid-1970s.

Island	**km^2**	**Island**	**km^2**
Ellesmere	80,000	Coburg	225
Ellesmere Ice Shelf	500	Meighen	85
Axel Heiberg	11,700	Melville	160
Devon	16,200	North Kent	152
Bylot	5,000	Others	736
Baffin	37,000		
Total surface glacier area for Arctic Archipelago: 151,800 km^2			

Source: Williams and Ferrigno, 2002

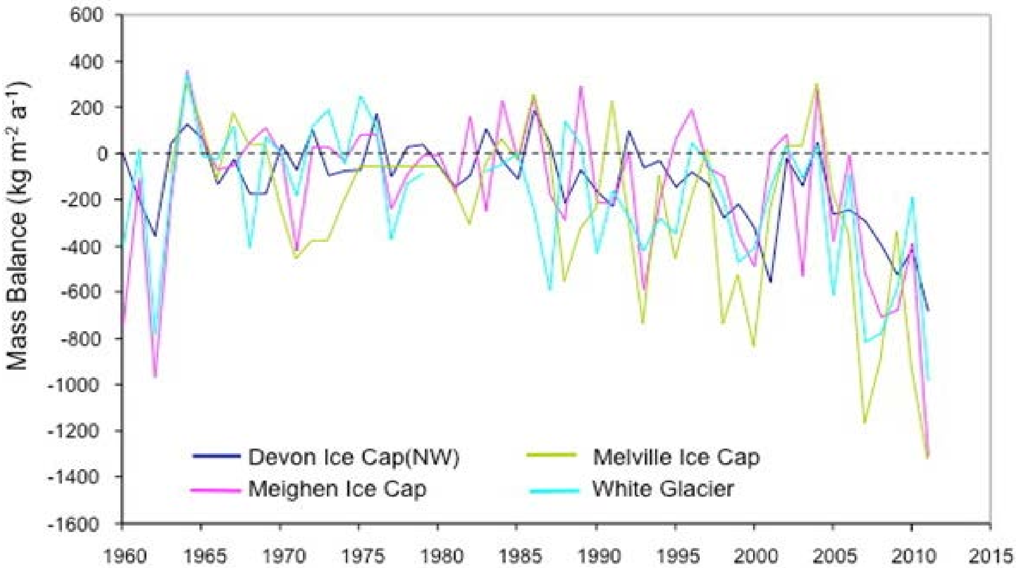


Figure 29. Annual net surface mass balance of four glaciers in the Queen Elizabeth Islands, 1960-2011. Source: Sharp et al., 2012 based on data from the World Glacier Monitoring Service

On Baffin Island, changes in the extent of a few small plateau ice caps (currently less than 50 km^2 in size) have been measured since 1975. This record has been extended back in time through the use of chemical and vegetation studies (Miller et al., 1975; Anderson et al., 2008). Some of these ice caps have disappeared already and researchers predict that all small ice caps will have disappeared completely by 2070. Figure 30 shows the measured and inferred rate of decrease in extent since 1940.

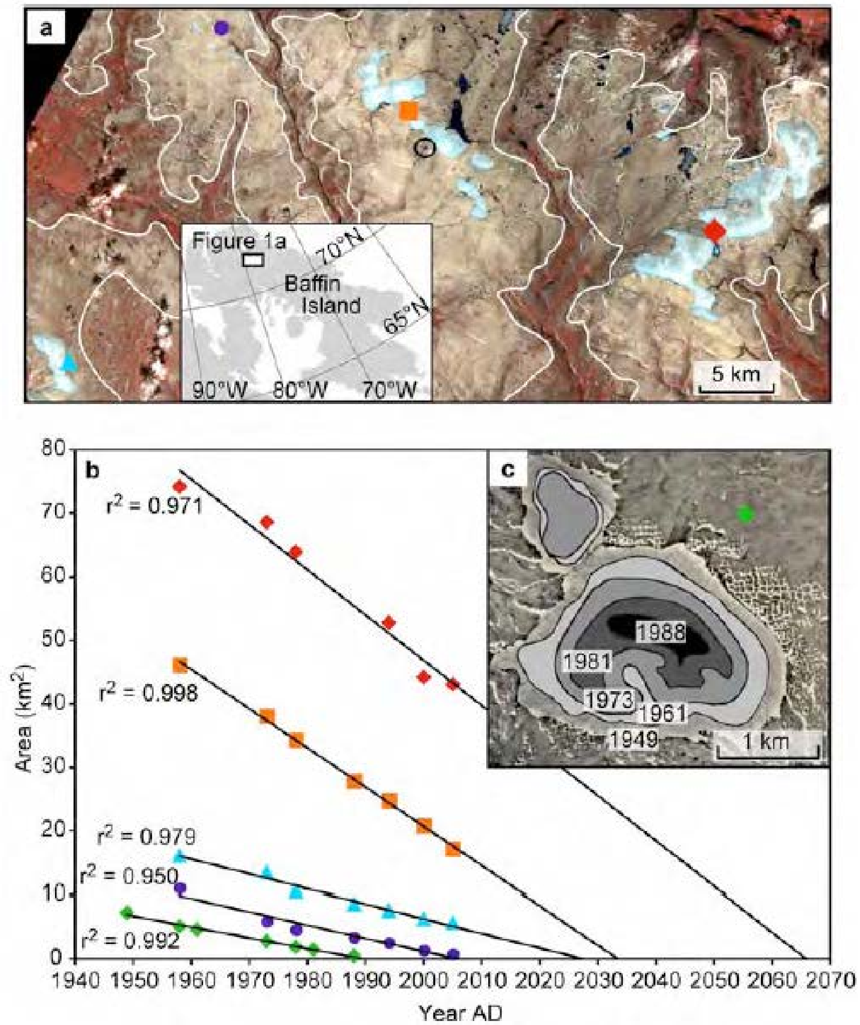


Figure 30. Location map and ice cap retreat on Baffin Island, 1949-2005.
(a) Landsat ETM+ satellite image, 10 August 2000, showing present-day ice caps. The white line delineates approximate extent of vegetation kill zone, corresponding to maximum ice-cap cover during the Little Ice Age. The black circle identifies the location of ^{14}C dates of 5.6 ka on aquatic moss in the basal sediment of a lake.
(b) Ice cap areas between 1949 and 2005. Corresponding ice caps are marked by symbols in (a). Linear regressions project approximate the dates of disappearance of the ice caps.
(c) Areal decrease of the Tiger Ice Cap, 1949–1988, which disappeared in summer 1992.
Source: Anderson et al., 2008

Mass loss of glaciers and ice caps due to surface melt and iceberg calving contributes significantly to sea level rise. As the Canadian Arctic Archipelago is the largest glaciated area in the world outside of Greenland and Antarctica, the changes in Arctic Ecozone^{+} glaciers and ice caps are of global significance.

A study published in 2009 estimates that glaciers and ice caps around the world will contribute 373 ± 21 mm of sea-level rise over the next 100 years, nearly half of the projected rise of 800 mm for the period (Bahr et al., 2009). An analysis of glacier mass balance based on both ground studies and remote sensing (Gardner et al., 2013) estimated that, between 2003 and 2009, glaciers and ice caps in northern Arctic Canada experienced a mass balance loss of a 33 ± 4 Gt/year and southern Arctic Canada glaciers lost 27 ± 4 Gt/year. The total for Arctic Canada is about 28% of the global glacier mass loss (excluding Antarctica and Greenland).

About half of sea-level rise is due to thermal expansion of the ocean. The next biggest source is meltwater from the Greenland and Antarctic ice sheets, followed by meltwater from the world's smaller glaciers and ice caps (IPCC, 2007). Although the contribution from the smaller glaciers and ice caps will, in the long run, be far less than from the Greenland and Antarctic ice sheets, they react faster to changes in climate and have more immediate effects (Bahr et al., 2009). From 2003 to 2009, global glacier mass loss, excluding Antarctica and Greenland, would have contributed 0.71 ± 0.08 mm/year to sea-level rise, which is about 29% of the observed sea level rise over this period (Gardner et al., 2013).

One of the major ecosystem changes will be the increase in land available for vegetation and the development of tundra ecosystems. This is a process that has been occurring since the late 19th century. The actual area of newly exposed land is not currently large, but could become increasingly important as ice masses disappear from the High Arctic islands (see Table 3 for areas covered). Areas exposed since the glacial maximum at the end of the Little Ice Age (about 1870) are visible as lichen-free zones near the glaciers. Satellite imagery of the Queen Elizabeth Islands (which includes the glaciers shown in Figure 29) shows that the area of terrestrial ice in the islands has decreased by 37% (62,387 km^2) since the last glacial maximum (Wolken et al., 2008). A study of ecosystem development in front of a glacier at Alexandra Fiord that has retreated over 300 m since 1980 described patterns of fairly rapid vegetation succession, with diverse plant communities being established, but with sparse plant cover after over 40 years (Jones and Henry, 2003). Areas achieving over 80% plant cover had been ice-free for over 300 years.

Lake and river ice

The duration of Arctic lake and river ice is determined by the dates of autumn freeze-up and spring break-up. In general, rivers experience much more dynamic events than those on lakes, as the formation, growth, and ablation of lake ice primarily occurs *in situ*, while that on rivers is controlled by the hydraulic effects of flowing water. Overall, however, the timing of such events on both systems is strongly controlled by climate (Prowse et al., 2011).

Analysis of changes in the remains of algae and invertebrates buried in lake sediments has been used by many researchers to identify warming trends and the historical presence/absence of ice cover on northern lakes (Douglas et al., 1994; Korhola et al., 2002; Sorvari et al., 2002; Michelutti et al., 2003; Ruhland et al., 2003; Smol et al., 2005), based on the relationships between ice-cover duration, growing season length, and annual primary production (Rouse et al., 1997; Douglas and Smol, 1999; Keatley, 2007). In general, around the circumpolar Arctic, this evidence points towards warming temperatures and shorter ice durations since the end of the Little Ice Age (around 1850), with greater changes observed in northernmost areas when compared to more temperate locations (Smol et al., 2005). (See section on Changes in algal and invertebrate species assemblages in lakes and ponds on page 94.)

In-situ monitoring datasets are not long enough to evaluate trends in lake and river ice in the Arctic Ecozone⁺ and remote sensing provides an alternative. Latifovic and Pouliot (2007) used ground-based monitoring datasets and remote sensing records to test comparability of the methodologies and examine trends in lake ice for 36 lakes across Canada. In doing so they developed records based just on remote sensing for six Arctic lakes. Break-up trends are shown in Figure 31A and freeze-up trends in Figure 31B. They concluded that, based on this limited set of lakes examined, the ice-free period increased significantly for lakes across all of Canada, but that the changes were of greater magnitude in the Arctic, at least in the past 20 years. Rates of change are shown in Figure 31C.

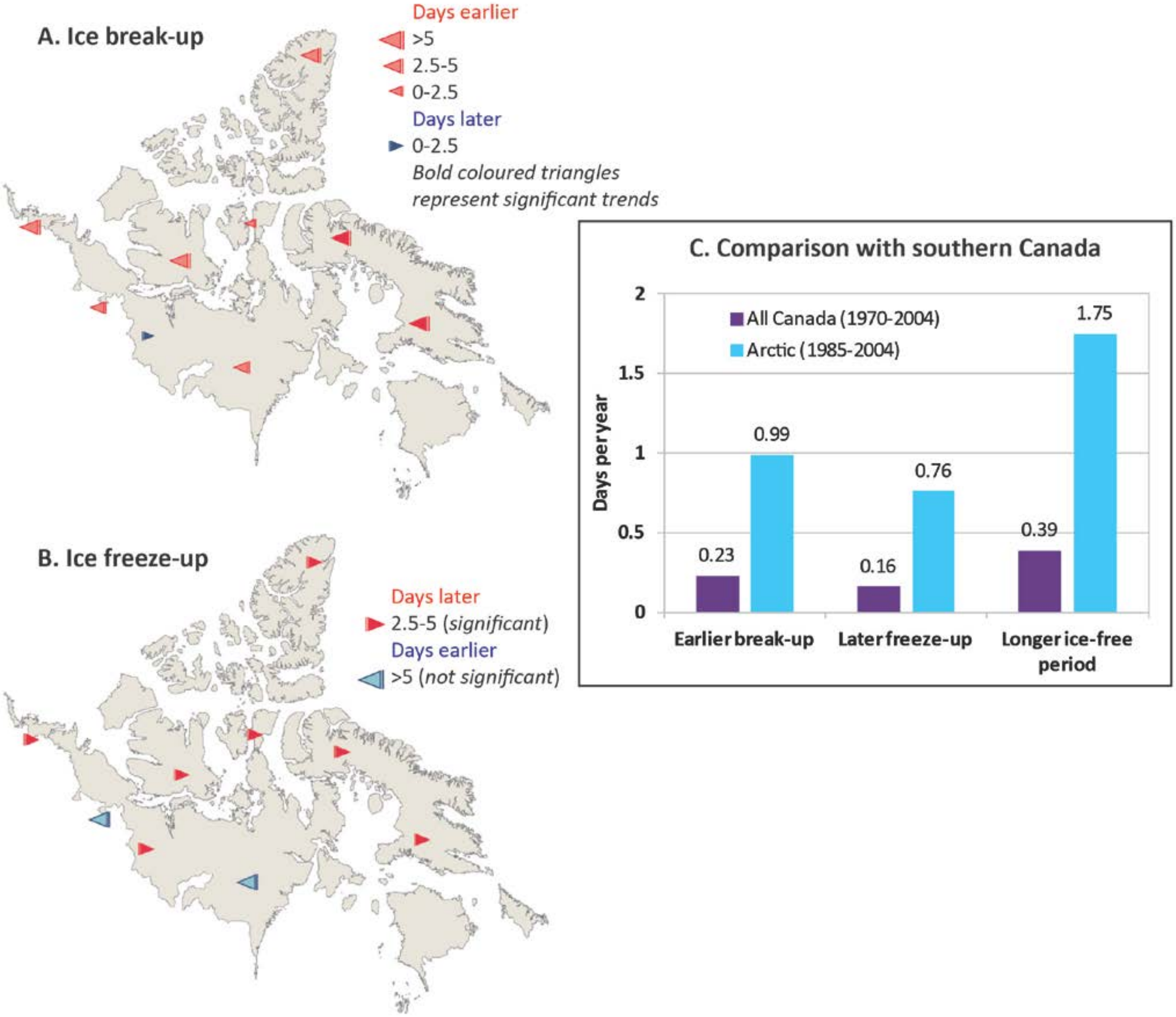


Figure 31. Ice break-up and freeze-up trends for six Arctic lakes, 1985–2004, and average trends compared with average trends for Canadian lakes south of the Arctic Ecozone^{+}.
Break-up and freeze-up dates are based on remote sensing as part of a larger study for all of Canada that found good agreement between ground-based records and the remote sensing methodology used. There were no ground-based records for the analysis for the Arctic lakes. Lakes are (west to east): Sitidgi, Washburn, Stanwell-Fletcher, Hazen, Angajurjualuk, and Amadjuak.
***A.** Break-up: only the trend for Lake Hazen was statistically significant (p<0.1)*
***B.** Freeze-up: trends for 3 of the 6 lakes were statistically significant (p<0.1)*
***C.** Comparison with lakes south of the ecozone*$^{+}$*: rates for the 30 lakes south of the Arctic were based on ground-based monitoring and remote sensing. Note that, because of the different time periods, the difference in rates of change reflects both the influence of greater warming in the past 2 decades and the effect of higher latitude.*
Source: adapted from Latifovic and Pouliot, 2007

Rivers and lakes

Trends in river discharge

Changing river discharge is a consequence of changing precipitation patterns, as well as changing temperature patterns that affect water storage features such as soil moisture, permafrost depth, and glacier melt. Arctic residents have noticed changes in river flows and water levels that may reflect climate change. For example, residents in Chesterfield Inlet reported a decline in water levels (rivers, streams, ponds, and lakes) by as much as a metre (Nunavut Research Institute, 2004). They observed that spawning char had more scratches and bruises caused by moving through shallow waters.

An assessment of Canada's lake and river systems undertaken for this report, ESTR thematic technical report *Biodiversity in Canadian lakes and rivers* (Monk and Baird, 2011), examines status and trends across the country and by ecozone$^+$ where there are sufficient long-term datasets. Only two hydrometric stations in the Northern Arctic and three in the Southern Arctic were of sufficient consistency and length to be included in the analysis of river flows. No stations were included from the Arctic Cordillera. Figure 32 shows the distribution of stations across all ecozones$^+$ and trends in one-day minimum runoff across Canada. Trends are generally grouped regionally and increases in minimum runoff are evident in two out of the five stations within the Arctic Ecozone$^+$. The same analysis was conducted for one-day maximum runoff, resulting in less clear regional trends across the country and no consistency in trends in the few Arctic stations (figure not shown).

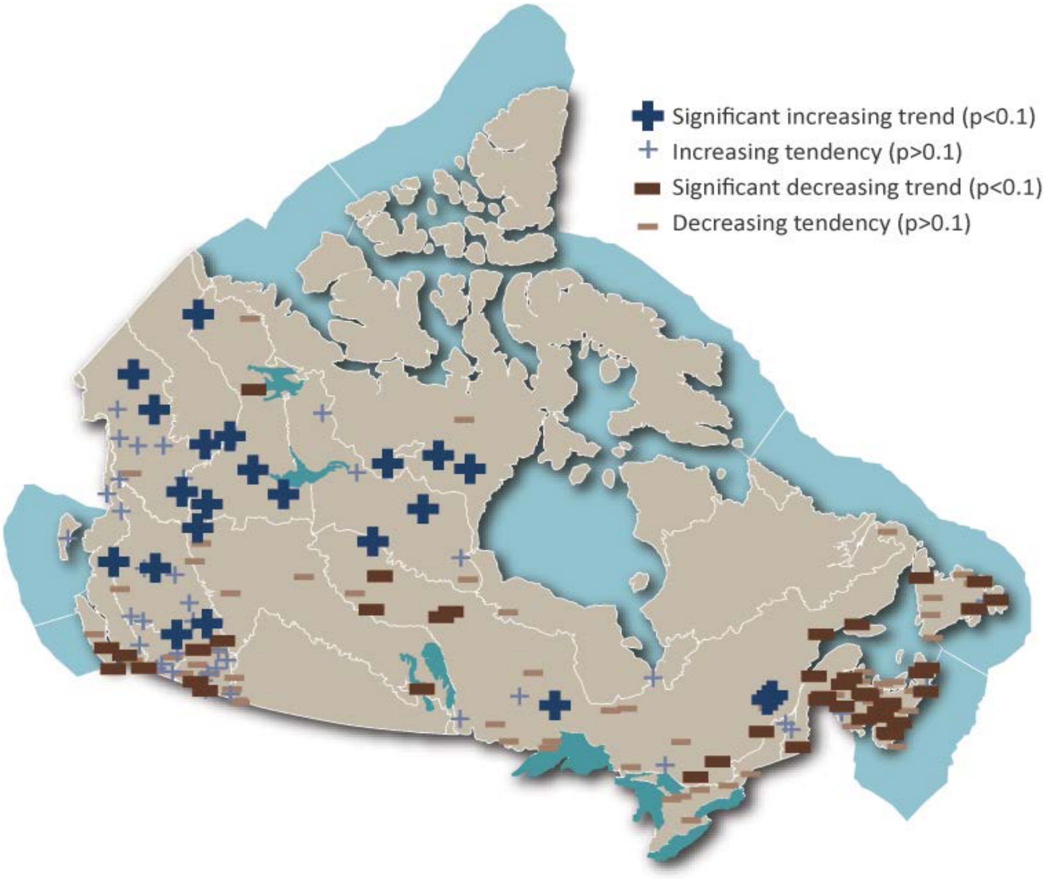


Figure 32. Map showing trends in 1- day minimum river flow in natural rivers across Canada, using data for hydrological years 1970–2005.
Source: Monk and Baird, 2011

With few long-term data sets for the smaller rivers that are wholly or mainly within the Arctic Ecozone$^+$, little is known about trends in discharge, especially in the Arctic Archipelago. An analysis conducted as part of International Polar Year research extended the sparse hydrometric records for the Arctic Archipelago through extrapolation of observed streamflow attributes to provide estimates of current discharge. The uncertainty associated with the estimates is high because of the sparse data coverage. The authors concluded that the current freshwater flux for the region, and if or how it is changing, are unknown (Spence and Burke, 2008). They highlight the importance of this information for domestic water management decisions and for meeting international obligations to quantify freshwater inputs to the oceans.

An evaluation of hydrometric monitoring for 76 river systems across the Canadian Arctic (1950 to 2008) was undertaken by Mlynowski et al. (2011). The peak in Arctic land area that was monitored through streamflow gauges was 64% in 1990, decreasing to 56% in 2008. Larger river systems typically had the most data available. The peak in record length was in 1998, when 22 river systems had more than 30 years of continuous records. The authors concluded that major improvements were needed in Arctic hydrometric monitoring, especially for the relatively small rivers.

About three-quarters of the landmass of Canada is drained by rivers discharging into the Arctic Ocean, Bering Strait, Hudson/James Bay, and the Labrador Sea (Déry and Wood, 2005). This flow involves almost half (47.9%) of the total discharge of Canadian rivers (Monk and Baird, 2011). Thus hydrological processes and aquatic ecosystem health of rivers that cross briefly through the Arctic Ecozone+ at the far northern end of their courses are influenced most strongly by climatic conditions, terrain, and stressors in ecozones+ to the south. A prime example is Canada's largest river basin, the Mackenzie River, which drains a total area of 1,787,000 km^2, or 20% of the nation's area (Culp et al., 2005). The river collects drainage from a number of other important rivers, including the Athabasca, Peace, Liard, Slave, Arctic Red, and Peel, before draining through the Mackenzie Delta in the Southern Arctic (The Atlas of Canada, 2008a and 2008b).

Results of trend analyses are very dependent on the subset of stations and the time period used. Long-term trends are further masked by decadal-scale variations related to climate oscillations. Déry and Wood (2005), for example, found significant links between the Arctic Oscillation, El Niño/Southern Oscillation, and Pacific Decadal Oscillation with total annual freshwater discharge of rivers in northern Canada. Results from recent analyses of trends in discharge of Canadian rivers draining to various combinations of high-latitude oceans are shown in Table 4. Three studies that examined trends to the early 2000s (Déry and Wood, 2005; Déry et al., 2005; McClelland et al., 2006a) showed significant decreases in streamflow. Examination of these trends by regional river basins, however, shows that the decreases were more pronounced in the Hudson Bay and Labrador Sea drainages and that there was no significant trend from the 1960s to the early 2000s for rivers draining directly to the Arctic Ocean (Déry and Wood, 2005). Analyses that include data into the 2000s show a reversal of earlier declines in discharge, including in Hudson Bay, and a significant increase in annual average flows since 1989 (Déry et al., 2009; Déry et al., 2011).

Table 4. Summary of recent research articles exploring statistical trends in streamflow and runoff in Canadian rivers draining to northern latitude oceans.

Study catchment	**Analysis period**	**Results / direction of trends**	**Reference**
64 hydrometric sites in Canada draining to high-latitude oceans	1964–2003	- Significant decreased trend in total annual freshwater discharge leading to a 10% decrease in the total annual discharge to the Arctic and North Atlantic Oceans. - Broken down by river basin, the declining trend was strongly influenced by reduced flows to Hudson Bay and the Labrador Sea.	Déry and Wood (2005)
45 rivers in northern Canada	1964–2007	- Last two decades (1989–2007) showed significant (15.5%) increase in annual average flows. -Increasing variability in streamflow in most drainage regions.	Déry *et al.* (2009)
56 rivers across North America (14 flow into Arctic Ocean and 42 flow into Hudson, Ungava and James bays)	1964–2000	- Discharge to Arctic Ocean decreased from sites in North America. - Discharge from sites draining Hudson, Ungava and James Bays decreased by about 2.5km^3/y/y during 1964-2000. - Suggest concomitant decreases in precipitation and river discharge.	McClelland *et al.* (2006a)
42 rivers draining into Hudson, James and Ungava bays	1964–2000	- Trends of decreasing annual discharge for 36 out of 42 rivers. - Total annual freshwater discharge in 2000 into Hudson, James and Ungava Bays decreased by 13% from its value in 1964. - Peak discharge rate associated with snowmelt advanced by 8 days and diminished in intensity. -Spring freshet varied by 5 days for each degree of latitude.	Déry et al. (2005)
23 rivers draining to Hudson and James bays	1964–2008	- No detectable overall trend in total discharge to Hudson Bay, but a decline up to the mid-1980s, followed by a period of relatively high flows, then an upward trend to the end of the study period. -A shift in seasonality, with a detectable increase in winter discharge and a detectable decrease in summer discharge from 1964 to 2008.	Déry et al. (2011)

All studies used Mann-Kendall statistical methods for trend analysis.

Source: updated from Monk and Baird, 2011

Circumpolar perspective

Freshwater discharge to these northern seas is significant not just in terms of the terrestrial/freshwater ecosystems in Canada's North but also to ocean processes, including sea ice dynamics and thermohaline circulation, which are strong influences on regional and global climates and of increasing interest as key components in understanding climate change. Piecing together the trends around the Arctic Ocean and subarctic seas puts both the Canadian trends and the importance of Canadian monitoring into perspective. An analysis of freshwater sources to all high latitude oceans (Peterson et al., 2006) concluded that there was a 5.3% increase in river discharge to the Arctic Ocean and an 8.0% decrease in river discharge to Hudson Bay in the 1990s relative to a 1936–1955 baseline. There was a shift in the late 1960s to early 1970s, marking the start of increasing pan-Arctic river discharge to the Arctic Ocean and decreasing river discharge to Hudson Bay. This increase in discharge to the Arctic Ocean is strongly influenced by annual discharge from the six largest Eurasian rivers, which increased by 7% from 1936 to 1999 (Peterson et al., 2002) and reached record highs in 2002 (Peterson et al., 2006) and 2008 (Figure 33).

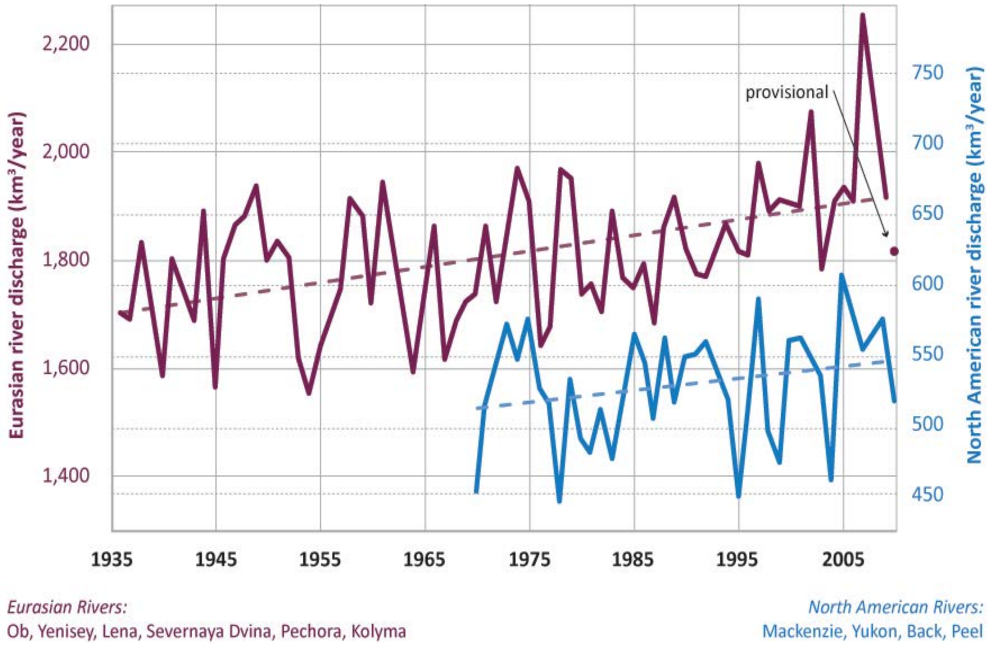


Figure 33. Overall trends in river discharge for six major Eurasian rivers (1936-2009) and four major North American rivers (1970-2010) rivers flowing to the Arctic Ocean.
North American rivers included in the analysis are the Mackenzie, Yukon, Back, and Peel rivers. The Peel River is a tributary of the Mackenzie River.
Source: Shiklomanov and Lammers, 2011

These analyses include rivers affected by hydroelectric projects, including rivers discharging to James and Hudson Bay. Hydroelectric dams and diversions have profound impacts on annual discharge patterns and water quality parameters like temperature and salinity, but do not have major effects on trends in total annual discharge. Changes in annual discharge related to periods when reservoirs were filling were accounted for in the analyses (Déry and Wood, 2005; McClelland et al., 2006b).

Trends in lake area

While remote sensing allows a broad view of change in lake areas, data for such analyses are limited to recent years. Carroll et al. (2011) examined the change in lake area across Canada from 60° to 70°N latitude from 2000 to 2009, based on analysis of MODIS satellite data. They concluded that, overall, lake area declined over the decade. They interpret this net loss of lake area to be related to longer ice-free periods and increased evapotranspiration, an effect that is particularly noticeable in the small, shallow lakes in Nunavut. Working counter to this trend, climate warming also can increase lake area as a result of thawing permafrost causing collapsing and flooding, as has been occurring, for example, in the Ungava Peninsula and James Bay area, south of the treeline (Taiga Shield Ecozone^{+}) (Payette et al., 2004b; Vallée and Payette, 2007; Thibault and Payette, 2009).

Ecological processes in lakes and rivers and linkages with climate change

The Arctic climate directly affects physical, chemical, and biological processes in aquatic systems. It also indirectly affects ecological processes through the control of terrestrial hydrologic systems and processes, particularly those associated with cryospheric components such as permafrost, freshwater ice, and snow accumulation and ablation (Prowse et al., 2006). In moist areas in the spring, thaw lakes and ponds have increased in amount and extent, evidently as a result of permafrost thawing and increased precipitation (Stow et al., 2004). However, in drier regions of the Northern Arctic and Arctic Cordillera, ponds have been reduced in extent and some have disappeared.

Ponds on Ellesmere Island that had been permanent water bodies for millennia (Douglas et al., 1994) dried up completely in the warm summers of 2005 and 2006. Surrounding moss and grass wetlands also dried, with the loss of seasonal standing water, and the vegetation became drier and more subject to fire. This drying of the landscape is attributed to increased evaporation related to high summer temperatures and decreased ice cover, based on analysis of specific conductance measurements taken in the ponds since 1983 (Smol and Douglas, 2007a). More permanent ponds and seasonal ponds and wetlands can be expected to be lost through desiccation as the climate warms further. Certainly many aspects of the physical, chemical, and biological characteristics of these sites will be affected.

Changes in hydrology and water temperature greatly affect the life cycles of aquatic organisms either directly, through temperature, day length, or photoperiod changes, or indirectly, through changes in water quality and available nutrients. Many studies have shown recent changes in aquatic ecosystems that are linked with, or portend, climate change. Effects include regime shifts, or widespread species changes and ecological reorganizations (Smol et al., 2005) (See section on Changes in algal and invertebrate species assemblages in lakes and ponds on page 94). Shifts projected for fish populations will range from positive to negative in overall effect, differ among species and also among populations within species depending upon their biology and tolerances, and will be integrated by the fish within their local habitats (Reist et al., 2006b).

Shorter periods of ice cover on lakes and rivers bring with them many ecosystem changes. For example, less ice leads to warmer water, changing lake mixing regimes, and the distribution of nutrients and oxygen. Changing river ice conditions alter hydrological events like the ice-jam driven spring floods that are a critical part of maintaining wetland and riparian habitat (Peters et al., 2006). Changes in ice duration have impacts at all trophic levels, as outlined in several assessments and reviews (Walsh et al., 2005; Wrona et al., 2005; Vincent et al., 2008). These changes are complex—for example, the increased abundance of food available for fish in river systems (from greater productivity), and the increased habitat availability with less ice may cause otherwise anadromous species to remain in rivers year round. Feeding at sea has been linked to larger sizes in fish and larger populations —thus the increased productivity from less lake ice may ultimately lead to decreased fish yields in lakes and rivers (Reist et al., 2006a).

Natural disturbance

Extreme weather events

In the Arctic Ecozone^{+}, severe weather events that influence the timing, amount, or quality of snow can have major ecological impacts on vegetation, small mammals, and ungulates. Deep snow and frozen snow layers block access to vegetation and prey or make food sources difficult to reach, leading to poor body condition and poor reproduction, as well as direct mortality (Miller and Barry, 2009; Hansen et al., 2013). While these conditions are a normal part of the Arctic winter, extreme weather events can mean that the conditions are more intense or last for longer than normal periods. In this section we have not attempted to define when an event is considered “extreme”, but rather we consider the continuum from normal variability to events that, due to their severity, pose risks of major mortality or reproductive failure to wildlife populations. For tundra ecosystem organisms, the major extreme events tend to occur in the shoulder seasons, autumn and spring, when the combination of temperatures and the timing of snow arrival and melt can result in major disturbances to their populations.

In the autumn, if the snow arrives too late or is too thin to insulate the ground, temperatures at the surface can be extreme for small mammals and invertebrates that depend on this insulation. On the other hand, rapid increases in snow depth in the autumn can make it difficult for large mammals to access forage plants or prey animals. It is for protection and insulation that small mammals seek deeper snow areas in tundra as overwintering habitat (Reid et al., 2012). Heavy snowfall events in August of 1997 and again in August 2000 in the Eureka region of Ellesmere Island covered vegetation, resulting in snow-free seasons of about half the normal length (Mech, 2004). Coincident population crashes of muskoxen, hares, and wolves were documented in the area and are attributed to increased energetic costs to the herbivores (Mech, 2004). In spring, a rapid snowmelt followed by a cold period can have strong effects on small mammals unable to get under the snow for insulation.

Extreme events falling within critical seasons are of particular ecological significance. In a classic paper on winter ecology, Pruitt (1957) described the temperature environment of a central boreal forest site in Alaska and related the changes to small mammal populations. He identified a fall critical period from the date that mean daily air temperatures went below the surface temperature to the date when a depth of about 20 cm of snow dampened daily surface temperature fluctuations. Similarly, the period from when snow depths become less than 20 cm to when mean daily air temperatures rise above surface temperatures is known as the spring critical period (Pruitt, 1957).

Another type of extreme weather event important in tundra systems is icing caused by rain on snow during the freeze period. A short period of temperatures above 0 °C with rain will result in ice layers either on the tundra surface or in the snow. These icing events can be devastating to wildlife populations if they seal the tundra surface making it very difficult for the animals to access forage. Miller and Barry (2009) noted that Peary caribou populations south-central Queen Elizabeth Islands were devastated (over 60% reduction) by exceptionally heavy snow and icing events in the four winters of 1973/74, 1994/95, 1995/96, and 1996/97. These events are projected to become more frequent and more widely distributed with climate change (Putkonen and Roe, 2003). The current distribution and frequency of icing events in the Canadian Arctic are not well-known and are difficult to reconstruct from data from the sparse network of climate stations (Grenfell and Putkonen, 2008). A rain-on-snow event in October 2003 on Banks Island contributed to a reduction of the muskoxen population by 20,000 over a three-year period. This event was analysed by satellite passive microwave imagery, developing a technique to map areas that have internal water and ice layers in the snowpack (Grenfell and Putkonen, 2008).

For tundra vegetation, icing events and extreme weather during the critical periods identified for small mammals are not as important—prolonged changes in temperature and snow regimes are of more significance. The distribution of vegetation along exposure gradients is based on the responses of plant species to the long-term site conditions, including snow depth and summer moisture availability (e.g., Walker, 2000). Sustained changes in soil moisture regimes as a result of flooding or permafrost disturbance (e.g., thermokarst; see section below) can greatly impact tundra vegetation (Vincent et al., 2011).

Additional types of extreme weather events with potential impacts on terrestrial ecosystems are likely increasing in frequency and/or intensity. Extreme heat events, for example, could result in increased fire frequency, loss of vegetation cover, and heat stress effects on tundra fauna, including outbreaks of disease. Intense storms in late summer and fall in the Arctic Ocean are known to alter ocean mixing and to play a role in reduction of sea ice (Jeffries et al., 2012), resulting in impacts on climate of the adjacent land, as well as increased coastal erosion.

Fire

Fire does not currently have a significant impact on Arctic ecosystems, as fires are rare and those that occur tend to be small, due to lack of fire-prone fuels and discontinuity of fuels (as patches of tundra are broken by barren lands, lakes, ponds, and rivers) (Krezek-Hanes et al., 2011). Records and research related to tundra fire ecology in Canada are sparse. Monitoring and analysis of satellite imagery indicates that there were only five large fires in the Arctic Ecozone$^+$ from 1960 to 2007 (Figure 34), though this low frequency may be partly a function of poor monitoring capacity in remote regions (Krezek, 2009, pers. comm.). A 1990 study of fire records, satellite imagery, and observations collected through a questionnaire concluded that fire was more frequent in the western Southern Arctic, rarer to the north and east, and rare to absent in the Arctic Archipelago, related to trends in biomass, human settlement, and climatologic conditions (Wein, 1990).

Most reported fires were small, with larger ones occurring nearer the forest-tundra zone, such as the 1968 fire north of Inuvik (the large fire shown in Figure 34), which spread 15 km from the forest into the tundra (Wein, 1990). The site of this fire was monitored for changes in active layer depth and vegetation until 1993, showing quite variable responses among monitoring locations (Mackay, 1995). At all locations where the active layer deepened (due to the burn or to increased summer air temperatures), the underlying ice-rich permafrost thawed to produce thaw settlement (see next section on Permafrost disturbance). In general, tundra fires remove the vegetation cover and result in a short-term deepening of the active layer (Mackay, 1995).

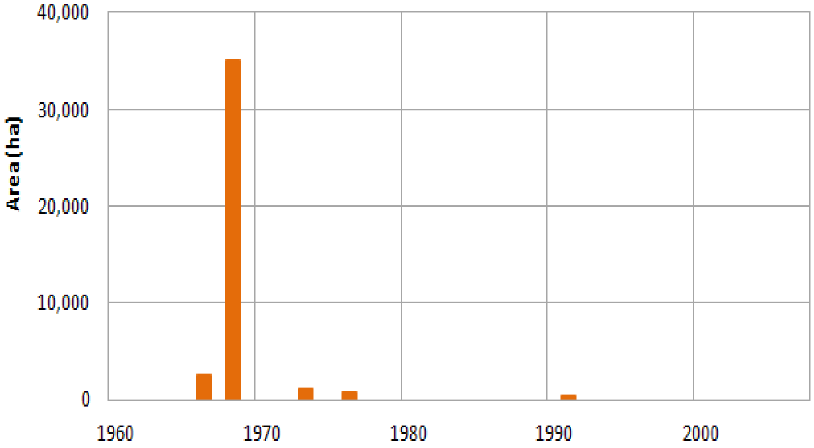


Figure 34. Area burned by large fires in the Arctic Ecozone^{+}, 1960–2007.
Based on Canadian Large Fire Database from 1960 to 1996 and on analysis of remote sensing imagery from 1997 to 2007.
Source: data provided by Canadian Forest Service; methodology presented in Stocks et al. (2003) and Krezek-Hanes et al., 2011

In northern Alaska a large tundra fire occurred in the exceptionally dry summer of 2007, burning 103,900 ha over a period of two months (Mack et al., 2011). This was the largest tundra fire recorded anywhere. Prior to this fire, researchers at the nearby Toolik Lake Long Term Ecological Research Network site had observed only two very small fires in the region over a 33-year period (LTER Network, 2008). The tundra fire was significant in terms of carbon balance: it was estimated to have released 2.1 teragrams of carbon to the atmosphere, an amount similar to the average annual carbon sink for the entire Alaska tundra biome (Mack et al., 2011).

Although tundra fires occur infrequently in modern times, analysis of charcoal and pollen in lake sediments in Arctic Alaska shows that the shrub tundra between 14,000 and 10,000 years ago burned with a comparable frequency to modern boreal forests (Higuera et al., 2008). These records, along with climate and vegetation reconstructions and analysis of tundra fires in Alaska since 1950 indicate that low effective moisture in summer (from a combination of higher temperatures and/or lower precipitation) and shrub-dominated vegetation cover are key factors in the fire regime. Several analyses now suggest that increases in tundra fires will likely accompany climate warming and the temperature-related increase of shrub vegetation cover that is now being observed in parts of the circumpolar Arctic, including Canada (Steven et al., 2006; Higuera et al., 2008; Hu et al., 2010; Myers-Smith et al., 2011a; Rocha et al., 2012; Elmendorf et al., 2012a).

Permafrost disturbance

Higher temperatures are thawing permafrost resulting in increased frequency and magnitude of slope failures and areas with thermokarst ponding (Lantz and Kokelj, 2008; Schuur et al., 2008; Lantz et al., 2009; Lamoureux and Lafreniere, 2009). These disturbances are restricted to areas with ice-rich permafrost in fine sediment. In the Mackenzie Delta region, retrogressive thaw slumps (Burn and Lewkowicz, 1990) have increased in numbers and size since the early 1970s (Lantz and Kokelj, 2008). Increases in these disturbances have also been observed in the High Arctic on the Fosheim Peninsula, Ellesmere Island (G. Henry, pers. observation, and Table 5) and on Melville Island (Lamoureux and Lafreniere, 2009) (Figure 35). The slumps result in areas with bare sediment, which can provide areas for different combinations of species to become established leading to greater landscape diversity. Lantz et al. (2009) found that green alder (*Alnus viridis*) had greater growth and reproductive effort on slumps than in undisturbed areas.

Table 5. Frequency of active-layer detachment failure at monitoring sites on the Fosheim Peninsula, Ellesmere Island.

Period	**Duration (years)**	**Number of detachment failures**	**Annual rate at which detachments are initiated**
Pre-1975	37–87*	230	2.6 to 6.2
1975–1987	13	164	12.6
1988–2000	13	190	14.6

*assumes slope failures had taken place in the past 50-100 yrs, based on the length of time they are estimated to remain visible on the landscape

Source: Lewkowicz and Harris, 2005

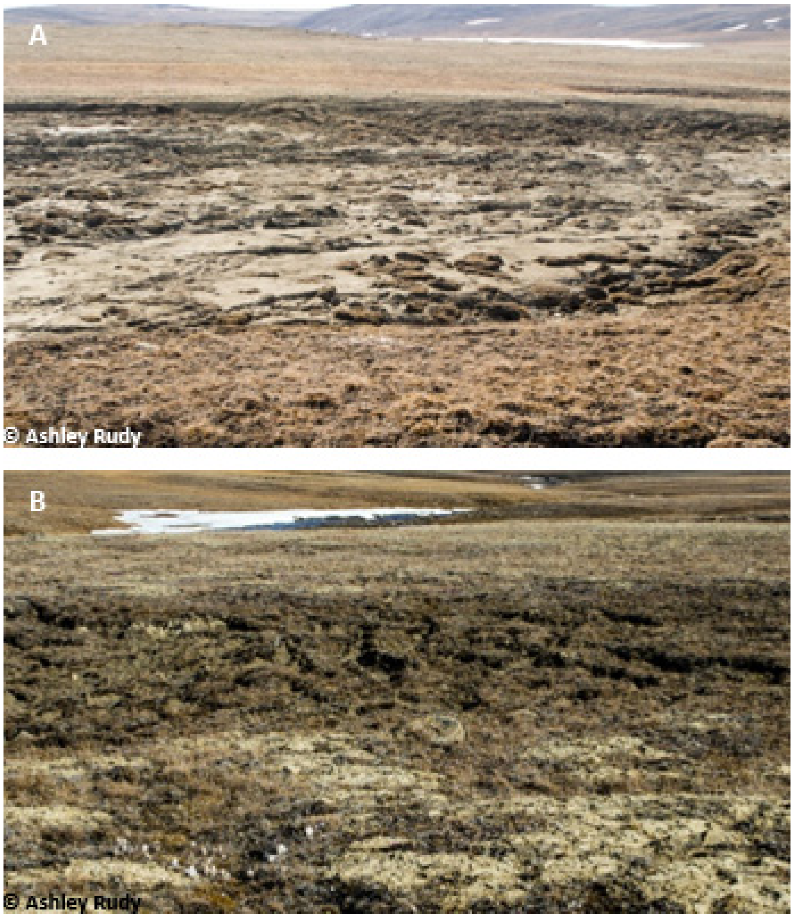


Figure 35. Active layer detachments on sloped tundra, Melville Island, Nunavut.
The photos show the upper zones of two types of active layer detachments: (A) an elongate detachment that is mainly stripped of its vegetative cover, and (B) a compact detachment that has kept more of its vegetation.
Source: Rudy et al., 2013

In the Low Arctic, these disturbances can expose previously frozen carbon to oxidation and change the tundra sites from carbon sinks to sources (Schuur et al., 2009). Many of the slumps in the Mackenzie Delta region occur on slopes above small lakes and the increased sediment load can significantly change the biogeochemistry of the aquatic environment and alter the biotic communities in the lakes (Kokelj et al., 2009; Thienpont et al., 2013). Lakes affected by slumping arising from permafrost thawing experience shifts in nutrient, light, and phytoplankton relationships, and slump-affected lakes generally have lower nutrients and lower productivity than lakes unaffected by slumps (Thompson et al., 2012).

In addition to the increases in slope failures, thawing of ice-rich permafrost can lead to subsidence and development of thermokarst ponds (Laurion et al., 2010). The conversion of terrestrial systems to ponds is a profound landscape change, with important implications for the carbon balance of these regions. Thermokarst ponds have been found to be important sources of methane and carbon dioxide (Laurion et al., 2010; Abnizova et al., 2012).

Deeper active layers also contribute to the drying of tundra soils, as water percolates further into the soil and may become unavailable for plants. Warming temperatures also increase evaporation from soils and water bodies, leading to further drying if the loss in moisture is not compensated for by increased precipitation and/or increased moisture from melting permafrost. One of the manifestations of this effect is seen in the drying of Arctic lakes and ponds (Smith et al., 2005a). Smol and Douglas (2007a) describe the drainage of ponds on eastern Ellesmere Island in 2006 which had been permanent water bodies through the Holocene, the ultimate threshold change for these systems. Loss of lakes can result both from increased evaporation and from drainage through the weakened melting permafrost (see also the section on Ecological processes in lakes and rivers and linkages with climate change on page 48). Drying tundra will have important effects on the system response to climate change and will depend on the initial conditions (Shaver et al., 2000).

Permafrost thaw has caused significant disturbances to Arctic landscapes and the combination of climate change and increasing industrial development has the potential to greatly increase rates and magnitudes of permafrost disturbance. These changes have important implications for the structure and function of Arctic terrestrial ecosystems, especially for carbon balance and feedbacks to the atmosphere.

Community and population dynamics

Arctic foods webs are characterized by short food chains, a few dominant animal species that often have cyclic population fluctuations, the central role of small mammals in driving populations at higher trophic levels, and a high degree of responsiveness to regional climatic oscillations and climatic variables such as precipitation, snow depth, and temperature (Elton, 1924; Norrdahl, 1995; Hudson and Bjornstad, 2003; Krebs et al., 2003; Gunn, 2003a; Hörnfeldt et al., 2005; Van Bogaert et al., 2007). They are influenced by a multitude of large-scale disturbances and are particularly vulnerable to global changes that may affect the sustained production of plants and animals on which Arctic cultures depend (Elmqvist et al., 2004). Arctic people are accustomed to large annual fluctuations in the wildlife on which they depend. However, a long-term decline in harvested species, such as caribou, could significantly affect their ability to maintain traditional lifestyles (see the section on Ecosystem goods and services on page 164).

Understanding trends in community and population dynamics requires examining ecosystem processes and integration of trends at all trophic levels, along with trends in drivers and stressors. There are few long-term monitoring programs in Arctic Canada that allow for this integration, though this was a focus of several Canadian-led International Polar Year research programs. For example, the International Tundra Experiment (ITEX) and the CircumArctic Rangifer Monitoring and Assessment (CARMA) networks work to consolidate existing research and monitoring and enhance ongoing observations for the Canadian and the circumpolar Arctic. At the broad-picture level, the Circumpolar Biodiversity Monitoring Program, a project of Conservation of Arctic Flora and Fauna (CAFF) (a working group of the Arctic Council), is working to improve coordination and integration of ecosystem monitoring.

Integration of monitoring and research through intensive study at research sites improves understanding of status and trends in community and population dynamics. These programs track and investigate trends and linkages to understand ecosystems. Examples for the circumpolar Arctic are the research and monitoring programs at Toolik Lake in Alaska, Zackenberg in Greenland, and Abisko in northern Sweden. Canada has not invested significantly in this approach to ecological monitoring and research in the Arctic and consequently understanding of Canadian trends in community and population dynamics is poor. Results from one Canadian site with long-term integrated research are summarized in the "Case study on ecosystem functions and processes: Bylot Island" (page 79). The general discussion on trophic dynamics below is developed further through specific examples in the case study. The bottom trophic level, primary producers, is addressed in the section on Primary productivity (page 69).

Predator-prey cycles

Dynamics of predators in Arctic tundra ecosystems are based mainly on lemmings and other small rodents (Callaghan et al., 2005b). Some predators, including snowy owls and weasels, specialize in rodents and their reproductive success is closely linked to small mammal cycles. Other predators, including foxes, wolverines (*Gulo gulo*), wolves, and grizzly bears (*Ursus arctos*), prey on a wider range of species and are able to switch to alternative prey in years of low abundance of small mammals. This means that alternative prey species, such as geese and shorebirds, become indirectly linked to small mammal cycles. Research in the eastern and central Canadian Arctic shows that fluctuations in lemming populations affect their main predators, snowy owls and Arctic foxes, which in turn indirectly affect snow geese, and possibly shorebirds, through shared-predator interactions (Kerbes et al., 2006).

Compared with boreal ecosystems, large predators are not abundant in the Arctic tundra and predation impacts on ungulates are usually low unless the ungulates are at low densities (Callaghan et al., 2005b; Legagneux et al., 2012). In a study across 12 sites in Arctic Canada, 79% of production of small to mid-sized herbivores, including voles, lemmings, and Arctic hares, was consumed by predators, while predators consumed only 9% of combined caribou and muskox production (Krebs et al., 2003).

Wolves and tundra grizzly bears depend on caribou, although the regulatory role of predation for caribou dynamics is uncertain. In the mid-1990s in the Southern Arctic, the Bathurst Herd of 350,000 caribou was estimated to support some 1,500 wolves (Cluff, 2004, pers. comm.) that likely annually killed 40,000 caribou. In addition, the herd was estimated to support 500 to 1,000 grizzly bears, with caribou making up 80% of their diet. A grizzly bear may eat between 9 and 18 adult caribou in a year (Gau et al., 2002). For the Northern Arctic Ecozone, Krebs et al. (2003) argue that the dominant ecosystem type is more one driven by variance in weather (bottom-up driven) rather than one controlled by predation (top-down driven). Lagagneux et al. (2012) contrast the predator-driven Bylot Island ecosystem, which lacks large ungulates, with the stronger controls exerted by bottom-up effects on systems in which caribou play a large functional role.

Herbivores and decomposers

Small and mid-sized herbivores are critical in Arctic food webs. The three- to five-year population cycles of small herbivorous mammals are well-known in the ecological literature. The cycles of the various species of lemmings [brown lemmings (*Lemmus trimucronatus*) and collared lemmings (*Dicrostonys spp*.)] (Krebs, 1996; Predavec et al., 2001; Gilg et al., 2003) and voles [tundra voles (*Microtus oeconomus*) and northern red-backed voles (*Myodes rutilus*)] (Krebs et al., 2002; Krebs, 2011) have been shown to have a high degree of synchrony across the Canadian Arctic (Krebs et al., 2002). Larger herbivores, such as caribou, may also cycle with a periodicity of 40 to 70 years (Gunn, 2003a) in the Arctic, but this is not well-documented because of the length of the cycles. Traditional knowledge does, however, provide ample evidence for the cycles in several regions. For example, one account of traditional knowledge passed along through generations of Chipewayan hunters indicates that the Bathurst Herd has been fluctuating in size with a cycle length about 30 years over the past 120 years (Nesbitt and Adamczewski, 2009).

Small mammals

Many small mammal populations throughout the circumpolar Arctic exhibit population cycles of more or less regularity. Some populations do not cycle but instead fluctuate without a pattern or remain at low population levels, with regular outbreaks of high densities being inhibited by conditions like patchy habitat, high predation rates, or particularly harsh winter conditions (for example, lemmings and collared voles showed no signs of cyclic abundance in a study on the Yukon North Slope (Krebs et al., 1995)). Scandinavian small mammal cycles have been well-studied. However, as most Norwegian lemmings (*Lemus lemus*) live in alpine tundra, the mechanisms driving their population cycles may not be broadly applicable to the Canadian Arctic. Data from northern Scandinavia indicate that lemming cycles may be dampened or lost for decades, while voles maintain regular cycles. Lemming populations in northern Scandinavia remained at low densities in recent decades, followed by recent population outbreaks, while vole cycles have weakened—observations attributed by researchers to the greater sensitivity of lemmings to variations in climate and predation (Ims et al., 2011). Of more relevance to the Canadian Arctic, 20-year studies of small mammals and predator-prey interactions in eastern Greenland indicate that climate change has the effect of increasing the length of the lemming cycle and decreasing maximum population densities, an impact that is detrimental to populations of predators reliant on lemmings (Gilg et al., 2009).

Using an indirect method of studying collared lemmings at 17 sites in the Canadian High Arctic and one site in Alaska, Predavec et al. (2001) were unable to detect a regular periodicity in fluctuations of lemming populations, although variability from year to year was high. Trends in small mammal densities from all monitored sites under the Northwest Territories/Nunavut Small Mammal Survey show, however, that population fluctuations can be synchronized over a large region, at least during the short period over which they have been recorded. Long-term monitoring covering approximately 10 cycles is necessary for a rigorous analysis of cyclic populations (Predavec et al., 2001).

There are no clear trends of long-term changes in the cycles in Canada. At Bylot Island, brown lemmings show cyclic, large-amplitude fluctuations in density with a periodicity of three to four years, though not collared lemmings. There is little evidence of a temporal trend in lemming density, but trend detection is made difficult by the large inter-annual variability. To smooth out those variations, Gauthier et al. (2013) looked at the three-year running mean of density. This analysis suggested a possible decreasing trend in summer lemming density. Lemming populations were especially low during the period 2002 to 2009, but the most recent peak (2011) was relatively high (Gauthier et al., 2013). Continuous, site-specific long-term monitoring of lemmings is still rare in the Canadian Arctic (Krebs et al., 2002), so it remains uncertain to what degree findings from Bylot Island are particular to the site. Only three peaks with similar abundance levels were detected between 1994 and 2012 at the only other site where long-term data on lemmings and voles exist, the Daring Lake Research Station in the central barrenlands, a site that is representative of the tundra-taiga ecotone (Figure 36).

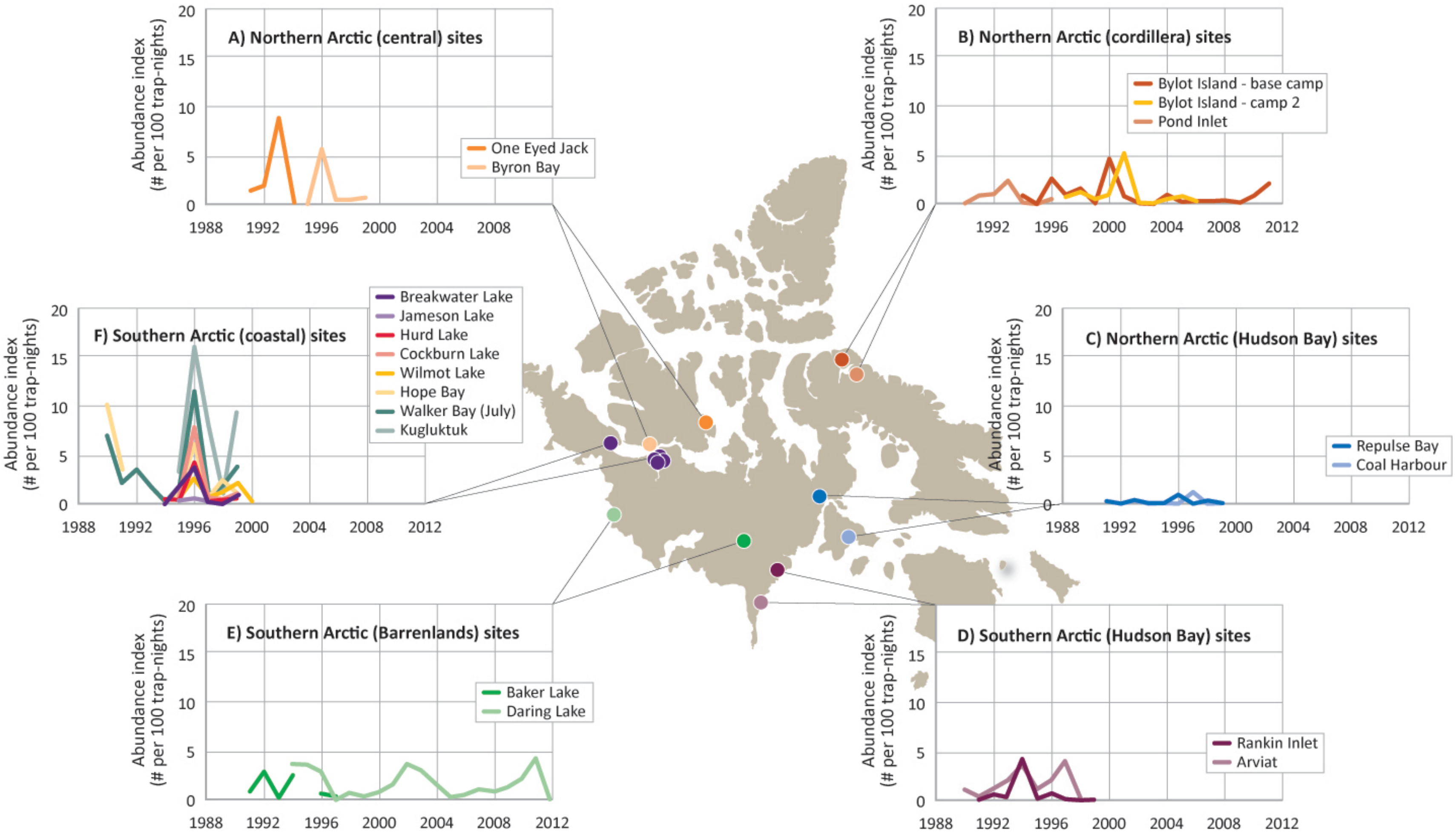


*Figure 36. Trends in small mammals (lemmings, voles, and shrews) at sites grouped according to apparent synchrony and ecological regions, 1990–2012. Abundance indices are numbers per 100 trap-nights corrected based on Beauvais and Buskirk (1999), modifying estimates of sampling effort to account for sprung traps. Species: northern red-backed vole (*Clethrionomys rutilus*), tundra vole (*Microtus oeconomus*), Meadow vole (*Microtus pennsylvanicus*), shrew (*Sorex sp.*), brown lemming (*Lemmus sibiricus*), and collared lemming (*Dicrostonyx groenlandicus*). Not all species are found at all sites.*

Sources: data coordinated by the NWT-Nunavut Small Mammal Survey, Government of the Northwest Territories (GNWT) and Government of Nunavut (GNU). A Byron Bay: C. Krebs (UBC, pers. comm.), One-eye Jack Lake: R. Bromley (GNWT, goose research, unpublished data). B Bylot Island: G. Gauthier (CEN, ULaval, updated from Gruyer (2007), Pond Inlet: GNU (unpublished data). C and D GNU (unpublished data). E Baker Lake: GNU (unpublished data), Daring Lake: GNWT (Arctic Research Station, S. Matthews, unpublished data). F Melville Sound Islands (Breakwater Lake, Jameson Lake, Hurd Lake, Wilmot Lake): C. Krebs (UBC, pers. comm.), Hope Bay: C. Shank (raptor research, GNWT, unpublished data), Walker Bay: B. Bromley (GNWT, goose research, unpublished data).

Caribou

Food webs in the boreal and Arctic ecozones$^+$ have relatively few links. The links, however, can be complex in terms of the dynamics of their inter-relationships. These northern ecosystems are nutrient-limited as so much carbon is inaccessible because only a shallow active layer of the soil thaws each year. Caribou, through their forage intake and output (faecal pellets), have complex and cascading effects, strongly patterned over time and space (Kielland et al., 2006). As well, caribou support a diverse group of other species, including external parasites such as blood-feeding mosquitoes. Mosquitoes, in turn, through the filter-feeding of their larvae, are a key element in nutrient cycling in aquatic systems. Further up the food webs, caribou support large-bodied and medium-sized predators and scavengers. Earlier debates about top-down (predator) or bottom-up (forage) regulation for caribou are now replaced by an appreciation of how nutrition and predation interact (Brown et al., 2007a).

Relationships between plants and caribou include the plants' responses to caribou's selective foraging. Caribou strongly select for individual plant species and forage for buds and unfolding leaves to maximize nutritional value (White and Trudell, 1980; Russell et al., 1993). The gregarious and migratory behaviours of migratory tundra caribou force their role in ecosystem structure and functioning to be strongly scale dependent (Griffith et al., 2002). Caribou convert plant tissue into body mass and faecal pellets. Through their local foraging movements and their seasonal migrations, they re-distribute nutrients within and across ecozones$^+$. In the taiga ecozones$^+$, the effects of caribou herbivory lag a season, as caribou are foraging during winter when most plant growth and nutrient cycling are quiescent due to sub-zero temperatures. Over the time scale of decades, caribou winter ranges expand and contract and the herds cycle from high to low abundance. Abundance can vary three-fold, with cascading effects on plants and nutrient cycling as the plant communities shift from one state to another. Succession of plant communities as a response to density of foraging may include, for example, lichen-dominated tundra shifting to greater dominance of moss, and then to increased dominance of grass (Van der Wal, 2006).

Nitrogen is a limiting factor for plant growth. Caribou summer browsing can increase the rate of soil nitrogen cycling through modifying the amount of plant litter, which changes the soil microclimate for decomposition and mineralization processes, and through adding soluble nitrogen from faecal pellets and urine (Olofsson et al., 2004). The changes vary with season and time, and with intensity of browsing (Kielland et al., 2006).

Decomposers

Decomposition rates are low in tundra ecosystems (see section on Permafrost on page 26 and section on Nutrient cycling on page 65). Most primary production enters the detrital food chain (Smith and Smith, 2001). A study on Devon Island showed that lemmings consumed about 3–4% of the standing crop most years, with most of the rest of the vegetation eventually being channelled to a variety of microbes and soil organisms, dominated by bacteria and fungi (Smith and Smith, 2001).

Wildlife diseases and parasites

Ungulates

This section, which draws material from the ESTR technical thematic report *Wildlife pathogens and diseases in Canada* (Leighton, 2011) and the ESTR technical thematic report *Northern caribou population trends in Canada* (Gunn et al., 2011c), covers disease and parasites related to Arctic ungulates and implications of climate change. Arctic ungulates, caribou in particular, are key components of Arctic terrestrial ecosystems and are vulnerable to changes in disease and parasite infestation. These changes could be related to disease type and frequency and also changes in the ranges of ungulates and of other parasite hosts. Warm weather can promote outbreaks of disease in cold-adapted animals and this may occur more frequently as the climate changes. For example, an introduced muskox population in Norway was struck by an outbreak of pasteurellosis in 2006, killing about 20% of the population. The outbreak was attributed to unusually hot and humid conditions (Ytrehus et al., 2008). An unusual outbreak of yersiniosis, a bacterial disease, in muskoxen on Banks Island in 1986 may also have been related to a period of unusually hot summer weather (Blake et al., 1991).

Brucellosis

Brucellosis is the name given to all diseases caused by infection with any of the several different species of the bacterial genus *Brucella*. The clinical manifestations of brucellosis are many, but the most common are infection and inflammation of the female and male reproductive tracts with resulting abortion and male infertility, and infection of joints and tendon sheaths resulting in progressive lameness. Infection persists, often for the lifetime of the animal. People are similarly susceptible to infection with *Brucella* sp., and brucellosis in animals with which people have contact is a public health risk (Chan et al., 1989; Forbes, 1991; Thorne, 2001).

In the Arctic Ecozone$^+$ infection with *Brucella* sp. is widespread and of potential ecological and public health significance in barren ground caribou populations (and one introduced herd of reindeer near Tuktoyaktuk, Northwest Territories). *Brucella suis* biotype 4 infects caribou across the Arctic, Taiga Cordillera, Taiga Plains, Taiga Shield, and the northern edges of the Boreal Plains, Boreal Shield, and Hudson Plains ecozones$^+$ (Forbes, 1991). Brucellosis is widespread in Arctic caribou, with 20–50% of animals in some herds infected (Leighton, unpublished data; Koller-Jones, 2006, pers. comm.). However, its ecological impact, if any, on infected populations is not known. Infection of northern people with this bacterium occurs and is associated with consumption of caribou (Chan et al., 1989; Forbes, 1991). Whether or not *B. suis* biotype 4 is a naturally-occurring pathogen in North America or a pathogen introduced from Europe in imported reindeer also is not known. There are no records of this infection in woodland caribou (*Rangifer tarandus caribou*), including in the George River herd of northern Quebec.

A serological survey of a large herd of reindeer in the western edge of the Arctic Ecozone$^+$ and of a barren-ground caribou (*Rangifer tarandus groenlandicus*) herd (Qamanurjuaq) in the Taiga Shield and adjacent Arctic regions of Manitoba and Nunavut in the 1960s found only 9% of reindeer and 4% of caribou infected (Broughton et al., 1970). The more recent infection rates of 20 to 50% may represent a trend of increasing prevalence and/or increasing surveillance. Any environmental changes that increase the overlap of barren-ground caribou with boreal caribou carry the risk that *Brucella suis* biotype 4 may become established in boreal caribou populations.

Brucellosis is considered to be responsible at least in part for the recent decline of the Southampton Island Caribou Herd. Since 1968, susceptibility to disease and parasites due to low genetic differences has been a likely catalyst for the widespread infection with brucellosis first detected in the Southampton Herd in 2000. Prevalence of the disease rose to 58.8% in 2011 and may be responsible for the drop in pregnancy rates in Southampton caribou since 2000, with the greatest decline in pregnancy rates being since 2008 (Department of Environment, 2013a).

Parasites

Host–parasite systems are particularly sensitive to climate change because many macroparasites have life cycles with free-living stages whose development and survival are strongly dependent on temperature and moisture conditions. Small changes in climate may have a large impact (Kutz et al., 2004) by influencing the development and survival rates of these free-living life stages as well as the abundance and activity of their arthropod vectors and mollusc intermediate hosts. Projected impacts and some empirical observations predict a longer season for development and transmission of parasites, increased infection levels in host populations, and increased outbreaks of disease (Kutz et al., 2005; Kutz et al., 2008).

Umingmakstrongylus pallikuukensis is a parasitic nematode worm prevalent in muskoxen in the western mainland parts of the Arctic Ecozone+. This lungworm requires a gastropod intermediate host to develop to the infective stage, which historically took two years. An empirically based model showed that temperatures have been warm enough since the early 1990s for the parasite to develop in one season. This could lead to potentially increased infection pressure on muskoxen, with possible impacts on fecundity and survival rates of muskoxen (Kutz et al., 2005). The extent to which this has affected muskoxen populations, however, is not known.

In Canada, *Besnoitia tarandi* infects caribou and reindeer, and probably muskoxen, across their ranges. It has been documented in caribou since 1922 in both Arctic and subarctic Canada (Ducrocq et al., 2013). Infection is common in barren-ground caribou and has been described in woodland caribou. Infection rates in muskoxen are not known (Choquette et al., 1967; Wobeser, 1976; Gunn et al., 1991a; Ayroud et al., 1995). Although occasional severe manifestations of infection on the skin have been seen, most infections appear to have little or no health consequences for these species. Hunter observations, confirmed by veterinary investigations, suggest that the protozoan *Besnoitia* has recently emerged as a disease-causing agent in caribou herds in Quebec and Labrador (Kutz et al., 2009; Ducrocq et al., 2013). In 2005, in focus group discussions in Inuvialuit, Gwich'in, and Dene communities in the Southern Arctic and taiga ecozones, some hunters reported increasing lesions (Kutz, 2007), possibly associated with Besnoitia cysts or even warble larvae (Kutz et al., 2009).

Climate change can also lead to changes in ranges of pathogens, both through increasing the area over which temperatures are warm enough for the pathogens to thrive, and through range changes in the host animals. Current knowledge regarding range extents of four ungulate protostrongylid nematodes is shown in Figure 37. Model projections indicate that, as temperatures rise, *U. pallikuuensis* will likely expand its range to the north and east (Kutz et al., 2005). Recent observations suggest that this parasite may have expanded its geographic range significantly in the last decade (Kutz et al., 2009) although interpretation of the data is complicated by increased surveillance for parasites. Protostrongylids have not been detected in ungulates from the Arctic islands and Greenland and may be excluded from high latitudes under current climate conditions but could invade these regions under warmed conditions if their host populations of moose, muskoxen, or caribou were to expand northward (Hoberg et al., 2008). *Parelaphostrongylus odocoilei*, a parasite with a range south of the Arctic Circle, may expand further north. In 1999, following a year with the warmest recorded annual temperature for the Mackenzie District, fatal pneumonia associated with *P. odocoilei* was first detected in Dall's sheep (*Ovis dalli*) in the Mackenzie Mountains (just south of the ecozone^{+}) (Hoberg et al., 2008).

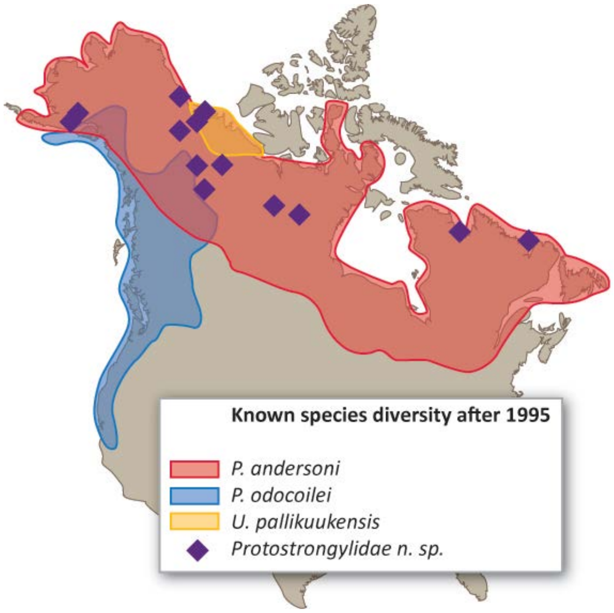


Figure 37. Geographic ranges for protostrongylid parasites in northern ungulates.
Distributions are depicted for Parelaphostrongylus andersoni *in caribou;* P. odocoilei *in wild thinhorn sheep, mountain goat, woodland caribou, black-tailed deer, and mule deer;* Umingmakstrongylus pallikuukensis *in muskoxen; and a putative new species of* Protostrongylidae *in moose, caribou, and muskoxen. The range for* P. andersoni *in the North is presumed to coincide with caribou, although records substantiated by survey are few.*
Source: Hoberg et al., 2008

Emergence of disease may follow climate change, but for macroparasites in particular, there are likely to be lag times determined by the life history of the parasites, including the period of development in the hosts (Hoberg et al., 2008). Cascading and cumulative long-term effects of climate change, including shifts in host–pathogen relationships, may be among the factors contributing to large-scale changes in abundance and distribution observed in keystone wildlife (e.g., woodland and barren-ground caribou) in northern North America.

Carnivores

Rabies persists in Arctic foxes and red foxes (*Vulpes vulpes*) in the circumpolar Arctic (Mork and Prestrud, 2004). It is difficult to assess trends in northern Canada, partly because samples are not routinely sent for testing once it has been established that rabies is present in the area during a particular winter (J. Kush, Rabies Lab, CFIA-Lethbridge, pers. comm.). Samples are submitted for rabies testing if a potentially infected animal is suspected of having been in contact with a person or a domestic dog. Although rare, rabies has also been documented in polar bears (P. Hale, Government of Nunavut, pers. comm.).

Little information exists on parasite communities of terrestrial Arctic carnivores. Without empirical baseline data on commonly occurring parasites in these animals, few studies will be able to assess impacts and changes of hosts and parasites as a result of climate change (Brooks and Hoberg, 2007).

Toxoplasma gondii

Using mathematical models to predict change, it has been postulated that the prevalence of *Toxoplasma gondii* infection in humans will increase due to environmental as well as anthropogenic factors related to climate change (Meerburg and Kijlstra, 2009). With increasing temperatures favouring survival of *T. gondii* in the environment, as well as changes in ecological distributions of vertebrate hosts, (more animals moving into Arctic areas), climate change will likely increase the prevalence of *T. gondii* in the Arctic (Hueffer et al., 2013).

Infection with *Toxoplasma gondii* is common in wildlife, domestic animals, and humans worldwide, including those in the Canadian Arctic (Dubey and Beattie, 1988; Mcdonald et al., 1990; Philippa et al., 2004). While some information exists on the occurrence of T. gondii in wild animals in the Arctic, the prevalence of the parasite in areas with a cold climate is low compared to humid, temperate, and tropical areas (Tenter et al., 2000). Infection of, or antibodies specific to, T. gondii have been found in some wild animals with ranges that include or are close to Arctic areas of North America, including moose (Kocan et al., 1986; Zarnke et al., 2001), wolves (Zarnke et al., 2001), wolverines (Reichard et al., 2008; Dubey et al., 2010), Canadian lynx (*Lynx canadensis*) (Labelle et al., 2001; Zarnke et al., 2001; Philippa et al., 2004), muskoxen (Kutz et al., 2000), and caribou (Kutz et al., 2001).

In the Arctic, Canadian lynx are the most likely wild definitive hosts for *T. gondii* and are responsible for contaminating the environment with oocysts. Studies conducted on Canadian lynx reported 1 of 5 (20%) from Nova Scotia, 47 of 106 (44%) from Quebec (Labelle et al., 2001), and 39 of 255 (15%) from Alaska (Zarnke et al., 2001) had antibodies to *T. gondii*. Canadian lynx generally prefer to stay below the treeline, but are occasionally observed and harvested north of treeline.

Close proximity to the treeline was shown to be an important variable for exposure to *T. gondii*. Surveys conducted by Kutz et al. (2000) to determine the seroprevalence of antibodies to *T. gondii* in muskox showed a higher proportion of *T. gondii* exposure in a mainland herd (closer to the treeline) compared to island populations (farther from the tree line) as well as a higher prevalence in adults and juveniles compared to calves. A significant difference was not found in the seroprevalence of *T. gondii* antibodies between male and female muskoxen (Kutz et al., 2000). In caribou, Kutz et al. (2001) showed that seroprevalence of *T. gondii* antibodies was significantly higher in mainland populations compared to island populations. However, there were no statistical differences in the seroprevalence of *T. gondii* antibodies among age classes or between sexes of caribou (Kutz et al., 2001). Reichard et al. (2008) showed that exposure to *T. gondii* was common in wolverines harvested in the Kitikmeot Region of Nunavut. Prevalence of *T. gondii* antibodies did not differ significantly between sex and age of wolverines, nor was exposure associated with proximate to the treeline at time of harvest (Reichard et al., 2008).

Phenology

The start and duration of the growing season in northern terrestrial systems are largely determined by snow cover. Changes in the depth and duration of snow cover over the past 50 years are discussed above in the section on Snow (page 30). The length of the snow-free season has increased significantly in most regions of the circumpolar Arctic, including Canada. In the Canadian Arctic, snowmelt dates shifted earlier by an average of about 8.5 days from 1950 to 2007 (Zhang et al., 2011).

Tundra plants can be divided into two groups based on their flowering phenology: early or late flowering (Molau, 1993). The flowering phenology is related to their reproductive strategies as either risking pollination (early-flowering species) or seed production (late-flowering species) (Molau, 1993). Early-flowering species dependent on insects for pollination may flower earlier or later than the peak of the insect populations, and any disconnect in timing may be exacerbated by climate change (Molau, 1997). Earlier snowmelt has resulted in earlier flowering by many species (Callaghan et al., 2011b; Oberbauer et al., 2013), and experimental warming has been shown to advance flowering (Arft et al., 1999; Aerts et al., 2006). Warmer temperatures in eastern Greenland have resulted in earlier flowering and a shorter flowering period, with a concomitant decrease in the number of potential insect pollinators (Høye et al., 2013). However, this is the only long-term monitoring study of the effects of warming on both flowering and insect phenology, and there is a need for more systematic studies on effects of earlier snow melt on pollinators in tundra systems. Late-flowering species risk lower seed production because of the unpredictability of late-season weather, especially in the High Arctic. In general, the end of the growing season and the onset of snow cover have not changed as much as snowmelt (Callaghan et al., 2011a), although experimental warming has been shown to delay fall senescence in some tundra species (Marchand et al., 2004). In many coastal locations, at least in the western Canadian Arctic, however, the predominant change in growing season length is in the autumn, with later onset of snow (due to the warming effect of the ocean) (Dye, 2002).

Leaf phenology also varies by species in deciduous shrubs and forbs, although leaf bud break occurs a few days after snowmelt in most species (Shaver and Kummerow, 1992). Experimental warming has shown that leaf bud break is occurring earlier in most species (Figure 38) (Henry and Molau, 1997; Arft et al., 1999; Oberbauer et al., 2013), and remote sensing has shown that the Arctic tundra regions are snow free and green-up earlier than 20 years ago (Zhou et al., 2001; Parmesan, 2007; Bhatt et al., 2010). A recent synthesis of phenological responses at the plot level over the past 30 yrs has shown that tundra plants are flowering and leafing-out earlier and this corresponds with the general warming over the same time period (Oberbauer et al., 2013).

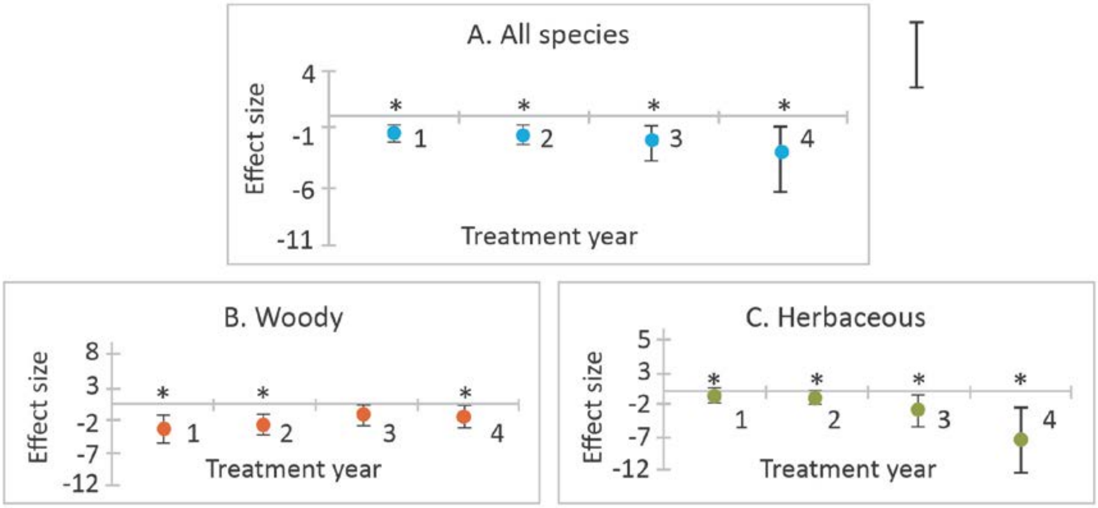


Figure 38. Meta-analysis of the effect of experimental warming on leaf bud break at all International Tundra Experiment (ITEX) sites, circumpolar Arctic.
*Significant effects are shown with * (P≤0.05). Mean effect size is shown with a coloured circle; the bars are 95% confidence intervals. The "effect size" (y-axis) is an index, with negative values indicating earlier leaf bud break. The experimental plots were warmed using open-top greenhouses.*
Source: adapted from Arft et al., 1999

Nutrient cycling

Tundra ecosystems are nutrient limited, as shown by relatively strong responses to low and moderate levels of fertilization (Henry et al., 1986; Chapin et al., 1995; Shaver and Chapin, 1995). This ultimately stems from the low temperature and low resource environment of the Arctic, with low rates of net primary production, decomposition, and mineralization (Hobbie, 1996; Nadelhoffer et al., 1997; Cornelissen et al., 2007). The uptake by plants of organic forms of nitrogen, for example amino acids, is important in tundra soils and is undoubtedly a response to the low rates of decomposition and mineralization (Kielland and Chapin III, 1992; Schimel and Chapin III, 1996). Research on the effects of climate change on nutrient cycling in tundra systems has shown that availability of inorganic and organic nitrogen can increase in experimentally warmed plots (Schmidt et al., 2002; Rolph, 2003; Aerts et al., 2006).

The soil microbial communities and their functioning are likely to be altered by climate change, both directly by changes in temperature and moisture, and indirectly through changes in vegetation, such as the increase in shrub abundance in the Arctic (Sturm et al., 2001; Bigelow et al., 2003; Stow et al., 2004; Deslippe et al., 2005; Tape et al., 2006; Myers-Smith et al., 2011a). The soil temperature regime has major consequences for Arctic ecosystems. In Alaska, winter biological processes are contributing to the conversion from tundra to shrub communities through a positive feedback that involves the snow-holding capacity of shrubs, the insulating properties of snow, a soil layer that has high water content because it overlies nearly impermeable permafrost, and hardy microbes that can maintain metabolic activity at low temperatures (Sturm et al., 2005; Myers-Smith et al., 2011a). Increasing shrub abundance leads to deeper snow, which promotes higher winter soil temperatures, greater microbial activity, and more plant-available nitrogen (Schimel et al., 2004). High levels of soil nitrogen favour shrub growth the following summer.

Experimental warming has been shown to alter microbial community structure (Walker et al., 2008b; Deslippe et al., 2012) and snow depth changes have been shown to alter microbial processes in some cases in the Low Arctic (e.g., Schimel et al., 2004). While experimental warming caused differences within sites in microbial composition, based on frequency and abundance of genotypes involved in nitrogen transformations, there were greater differences between sites. The greatest effects of warming were found to occur in wet sedge tundra communities (Walker et al., 2008b). However, resistance to change in microbial communities has been found in studies of warming and combined warming and fertilization of tundra communities in the High Arctic (Deslippe et al., 2005; Lamb et al., 2011).

In wet sedge meadow soils, microbial biomass and nutrient availability peaked early in the spring freeze-thaw phase, but then crashed after soil temperatures rose above 0°C, implying that competition for nutrients from roots results in the collapse of the microbial pool (Edwards et al., 2006). Earlier spring snowmelt and warmer temperatures would alter the timing of these changes, but the vascular plant growth would still depend on available nutrients and soil moisture (Nadelhoffer et al., 1997).

Nutrient uptake in many northern plant species depends on mycorrhizal associations. Deslippe et al. (2011) have shown that long-term experimental warming changes the diversity of ectomycorrhizal fungal families on root tips of dwarf birch (*Betula nana*), a deciduous shrub. Similarly, Fujimura *et al.* (2008) found a change in the density of genotypes of the fungal community associated with the roots of arctic willow (*Salix arctica*) in warmed plots of three High Arctic tundra communities, although there was no shift in relative abundance. The effects of these changes in mycorrhizal communities are unknown, but they will likely affect the ability of shrubs and other plant species to absorb nutrients in the warmer soils.

Arctic animals play a major role in nutrient cycling. Caribou, the main large herbivore in the Arctic, have the net effect of forage removal, production of greenhouse gas, and return of nutrients through faecal pellets. Based on energetics modelling (Russell et al., 2005; Gunn et al., 2011c), a caribou, annually, removes 900 kg of food (2.5 kg/day), produces 20 kg of methane (55 g/day) and returns to the ecosystem, nutrients in the form of faecal pellets, 270 kg (30 g x 25 times a day). At the herd scale, annually, 170,000–350,000 caribou remove 140–320 million kg of forage, produce 3–7 million kg of methane, and return 38–77 million kg of faecal pellets spread over the annual range (150–300 kg/km^2). As caribou travel and rest on frozen waterways, the nutrient return from faecal pellets is to aquatic as well as terrestrial ecosystems (Gunn et al., 2011c).

Geese, as well as cycling nutrients through their role as tundra grazers, import energy and nutrients into Arctic ecosystems when they return to their breeding grounds after a winter of feeding in fields and wetlands far to the south. These imported resources move through the food chain when the geese are preyed on, for example, by Arctic foxes (Giroux et al., 2012a). Migratory birds may also export nutrients and energy, for example through removing nitrogen from tundra ecosystems when young produced locally migrate south and die (Gauthier et al., 2011). While locally significant, the importance overall of these resource exchanges to tundra ecology is not known (Gauthier et al., 2011; Giroux et al., 2012a).

Carbon storage and release

Permafrost soils are estimated to contain 1,400 to 1,859 Pg of carbon in frozen and seasonally unfrozen surface layers, which amounts to 50% of the soil carbon in the world (McGuire et al., 2009; Tarnocai et al., 2009). In Canada, Arctic tundra systems were estimated in 2008 to contain about 76 Gt of soil organic carbon in the upper metre of the soil profile (Ping et al., 2008), but, given more recent research (Tarnocai et al., 2009; Grosse et al., 2011; Kuhry et al., 2013), this figure may be low. Soil carbon (C) might react to near-term climate change (Clein et al., 2000). Tundra ecosystems have been sinks for carbon over tens of thousands of years (Ping et al., 2008). This “service” was possible because of the cold soils, which restricted decomposition rates and the accretion of permafrost, which helped to freeze the carbon stored in the soil. There is concern that the warming climate will result in a switch in tundra ecosystems from sink to source of carbon for the atmosphere (Oechel and Vourlitis, 1997; Ping et al., 2008; Grosse et al., 2011; Lund et al., 2012).

Large amounts of C and nitrogen (N) could be released in inorganic forms as Arctic soils warm, the active layer deepens, decomposition rates increase, and the growing season lengthens (Nadelhoffer et al., 1997; Grosse et al., 2011). A large release of carbon dioxide (CO_2) from these soils would increase atmospheric CO_2, enhancing the rate and magnitude of climate change (a positive feedback). The release of other greenhouse gases, especially methane and nitrous oxide, could be increased from the permafrost soils as they warm, which would contribute to the positive feedback (Vincent et al., 2011).

A coordinated study of the net ecosystem production of Arctic tundra sites in the Canadian Arctic found that all sites were sinks for CO_2 during the summer, and the differences in rates followed expected patterns due to latitude and proximity to cold coastal conditions (Lafleur et al., 2012). Annual variability in net ecosystem production was considerable at both a wet and a mesic (moderately moist) tundra site in the Low Arctic, and was strongly related to differences in climate during the growing season (Humphreys and Lafleur, 2011).

Warming experiments to date have shown that wet tundra systems remain as sinks when warmed, as both photosynthesis and respiration are increased; however, warming greatly increased the loss of carbon from dry tundra ecosystems (Oberbauer et al., 2007) (Figure 39). In addition, there was a gradient in net carbon exchanges, with High Arctic systems remaining as sinks but with increasing loss of carbon from Low Arctic systems. There are very few studies of carbon flux in response to experimental warming and other environmental perturbations in Canadian tundra ecosystems.

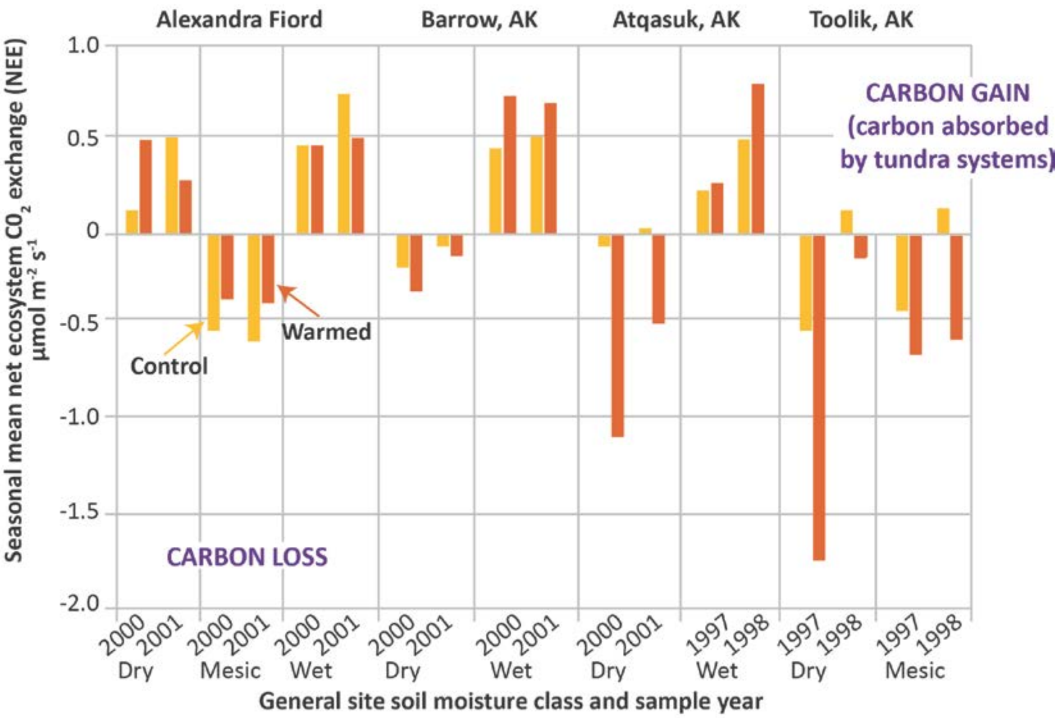


Figure 39. Effect of experimental warming on net ecosystem exchange of CO_2 at four ITEX sites in North America, including Alexandra Fiord, Ellesmere Island.
Source: modified from Oberbauer et al., 2007

Pollination

Phenological changes and changes in animals that act as pollinators could affect vegetation. This may not be as great a concern for the Arctic Ecozone⁺ as for other ecozones⁺, as many Arctic flowering plants can self-pollinate or are pollinated by the wind (Callaghan et al., 2005b). Nonetheless, most tundra flowering plants are dependent on insect pollination to set seeds (Kevan, 1972), and outcrossing through insect pollination is important to maintain genetic diversity. Insect pollination is an important part of the reproductive success of flowering Arctic plants because it provides more seed genetic variability than self-pollination or asexual reproduction. Insect pollination is more successful at pollen transfer than wind pollination or transfer by animals (Kevan, 1972; Crawford, 2008).

The main pollinators at the few Arctic locations where pollinators have been studied are flies (Diptera), with bumblebees and butterflies also being important (Ellesmere Island: Kevan, 1972; Zackenberg, Greenland: Klein et al., 2008; Høye et al., 2013). Studies in Greenland indicate that most insect pollinators are generalists, though some preferences for certain plants are exhibited (Klein et al., 2008). It is not known to what extent changes in plant and insect species distributions, arrival of new species to the Arctic, as well as changes in phenology, might affect pollination (Klein et al., 2008). However, Høye et al. (2013) have found that the flowering season has become shorter and insect flower visitors have declined as temperatures increased over the past 15 years at Zackenberg, Greenland.

There has been little study on pollination relationships in the Canadian Arctic and little is known about the status and current trends related to the changing climate. The synchronization between insects and flowering Arctic plants is complex and it is hard to predict how these relationships will be affected by climate change, but it is expected that impacts will vary with species (Danks, 1992 and 2004). Insects are highly dependent on the microclimates where they live and in the Arctic they are already at the limits of their ability to adapt in relation to cold hardiness, solar dependence, and sensitivity to desiccation (Danks, 2004). The northward movement of shrubs into new areas creates new microhabitats for insects, in particular pollinators that may be adapted to invade Arctic environments (Klein et al., 2008). Insects have the ability to adapt quite readily to environmental change by moving to areas with more favourable conditions and they have previously shown rapid movement into new habitats created by the retreat of glaciers at the end of the last ice age (Danks, 2004; Klein et al., 2008).

Primary productivity

Primary productivity is low in the Arctic compared with other ecosystems (Figure 40). Primary production ranged from a low of 3 kg dry mass/ha/yr to a high of 334 kg dry mass/ha/yr at 12 sites across the Canadian Arctic (Krebs et al., 2003). Within the Arctic, the standing crop of vascular plants was largest in the Western Arctic, while the standing crop of mosses was largest at High Arctic sites such as Melville Island and Ellef Ringnes Island (Krebs et al., 2003).

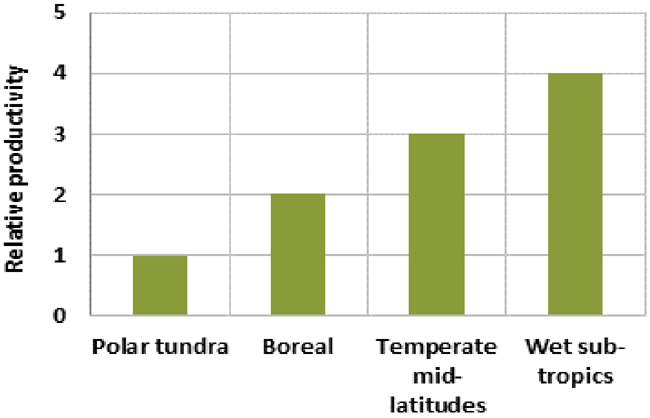


Figure 40. Relative net primary productivity of major world ecosystems.
Source: adapted from Schultz, 2005

Evidence is accumulating that the Arctic is getting greener and that the net productivity of Arctic ecosystems is increasing.

In lakes and ponds

In Arctic lakes and ponds, primary production has been shown in several studies to have increased, accompanied by changes in algal species assemblages (Smol et al., 2005; Antoniades et al., 2007). In a study designed to evaluate long-term trends in Arctic lake primary production (Michelutti et al., 2005), researchers used reflective spectroscopy to infer chlorophyll a concentrations in sediment cores. Results from the survey of six Baffin Island lakes indicate that there have been pronounced 20th century increases in primary production (see text box below and accompanying Figure 41 showing long-term changes in an Arctic lake). The changes appear to be synchronized with the record of recent climate change. See also discussion in section on Changes in algal and invertebrate species assemblages in lakes and ponds (page 94).

Primary production in Arctic lakes: Lost Pack Lake, Nunavut

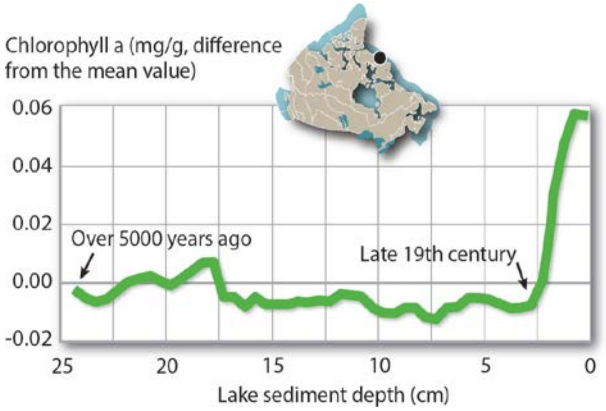


Figure 41. Primary production in Lost Pack Lake, as measured by variations in chlorophyll a concentration in lake sediment.
Note: chlorophyll a *is the main pigment found in plants and algae and is a measure of primary production. Values are inferred through spectral analysis of lake sediment cores. The mean chlorophyll* a *value is the mean of the full data set.*
Source: adapted from Michelutti et al., 2005

The figure shows chlorophyll *a* reconstructions from Lost Pack Lake, one of six Baffin Island lakes examined for long-term trends. All lakes show dramatic increases of inferred primary production within the most recently deposited sediment, following prolonged periods of comparatively low values (Michelutti et al., 2005). Dating of the sediment cores indicates that these rapid increases started in the late 19th century and continue to the present. The increases are a departure, in most lakes, from relatively stable levels of primary production that persisted for millennia. A widespread increase in freshwater production over much of northern Canada is also inferred from major shifts in species composition of algae in ponds and small lakes in many areas (also detected from studies of sediment cores) (Smol et al., 2005; Antoniades et al., 2007).

The best explanation for this change in algae is climatic warming leading to longer ice-free growing seasons and associated changes in lake ecosystems (Antoniades et al., 2005; Smol and Douglas, 2007b). The changes are most pronounced in the High Arctic, but similar shifts in algal species are found in many locations in the Northern Hemisphere—with changes being more recent in temperate latitudes (Rühland et al., 2008).

Source: Federal, Provincial and Territorial Governments of Canada (2010)

On land

On land, changing primary production and changing physical structure of biomass, represented by increases in distribution and height or vigour of shrubs (Elmendorf et al., 2012b), will affect herbivores and, consequently, predators. Increasing productivity and advancing tree and shrub cover have the potential to alter predator-prey relationships and to facilitate range extensions for some biota, such as certain perching birds that cannot otherwise occupy the tundra, as well as their predators, diseases, and parasites. Ecological changes such as these may be involved in some of the declining trends noted in the section on Ecosystem composition (page 100).

Several measures of primary productivity show marked and widespread increases, and ground observations and experimental studies provide confirmation and understanding of the nature of this change. This section examines trends in productivity from several studies, and the section on Ecosystem structure (page 86) discusses related changes in the tundra biome.

A study undertaken for this report (Ahern et al., 2011) used analysis of the Normalized-Difference Vegetation Index (NDVI) to measure trends in productivity for all of Canada's ecozones+. NDVI is a measure of the photosynthetic capacity of plant cover from space-based observation. Dense plant canopies have positive values of NDVI while snow and ice have negative NDVI values. The resulting map (Figure 42) shows significant trends in NDVI for the Arctic from 1985 to 2006. Only a small percentage of Canada showed a negative NDVI trend, while 22% of the nation's land area showed a positive trend, with the largest positive trends being in regions of Arctic tundra, alpine tundra, the Pacific coast, and the eastern prairies.

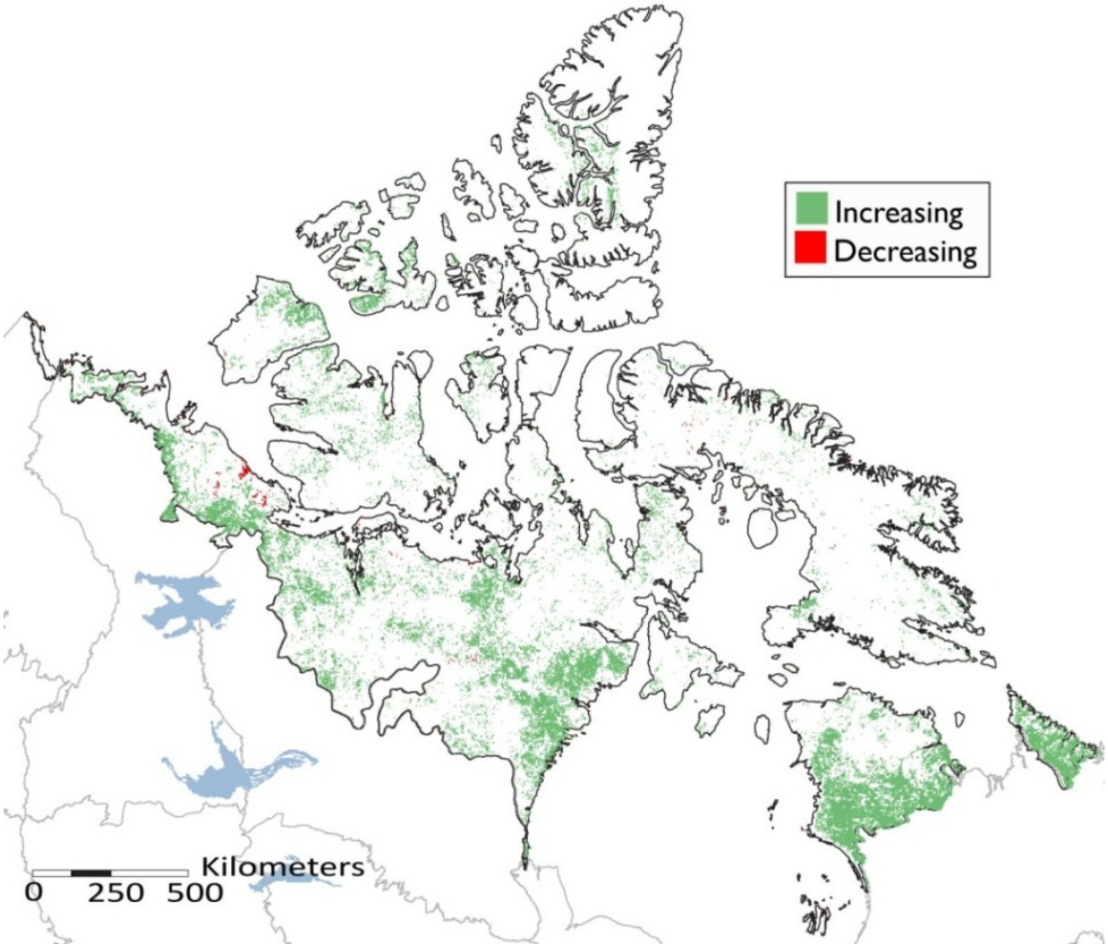


Figure 42. Trends in Normalized-Difference Vegetation Index (NDVI) for the Arctic Ecozone+, 1985–2006. Produced by TerraVista Earth Imaging from data supplied by CCRS.
Source: Ahern et al., 2011

Trends for the Arctic Ecozone$^+$

Arctic Cordillera: increasing trend in 9.2% of area, decreasing in 0%

The only area that has significant amounts of vegetation is the Labrador Peninsula. The entire peninsula exhibits a strong positive trend in NDVI, particularly the lower elevations bordering Ungava Bay. The vegetation in this area is low vegetation characteristic of tundra regions.

Northern Arctic: increasing trend in 6.8% of area, decreasing in 0.1%

Areas that are particularly notable for increases in NDVI are the northern portion of Banks Island, the Dundas and Sabine peninsulas of Melville Island, the south shore of Bowman Bay on Baffin Island, and the area along the northwestern shore of Hudson Bay. All of these areas are dominated by tundra vegetation.

Southern Arctic: increasing trend in 23.8% of area, decreasing in 0.3%

The Southern Arctic shows extensive areas of increasing NDVI, most notably along the northwestern shore of Hudson Bay, northeast of Great Bear Lake, and the southern portion of the Ungava Peninsula, where the trend is particularly pronounced. All of these increases are in areas of tundra vegetation.

These changes can be seen in a circumpolar perspective in Figure 43. This figure shows the regional trends in increases in tundra and taiga ecosystem NDVI—with major increases being in the Canadian Western Arctic, Alaska, and Siberia.

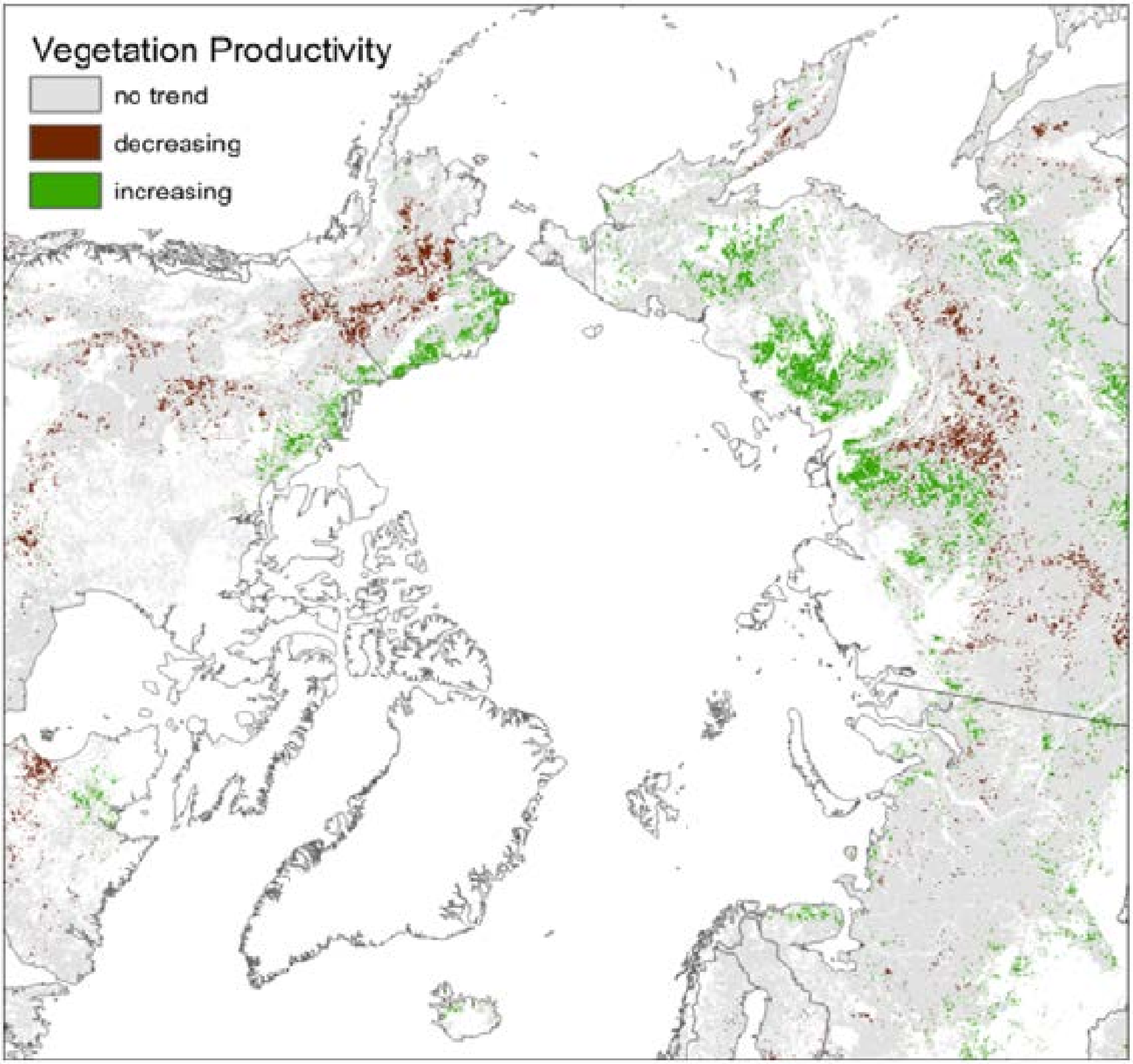


Figure 43. Trends in primary productivity, based on remote sensing, 1982–2008.
Grey areas showed no significant trend (P≤0.05); white areas were excluded from the analysis. Based on GIMMS (Global Inventory Modeling and Mapping Studies) NDVI (Normalized-Difference Vegetation Index). Note that most of the NDVI increases shown in Figure 42 above can also be seen on this presentation.
Source: Beck and Goetz, 2011 (corrected version, 2012)

Primary production (biomass) in tundra ecosystems

Primary production is typically low in tundra ecosystems and depends on latitude and local topographic position (Bliss and Matveyeva, 1992). Gould et al. (2003) mapped the vegetation of the Canadian Arctic (Figure 44) and provided estimates of the biomass (Figure 45) and net production of tundra ecosystems (Figure 46). They estimated that net primary production ranges from less than 20 g/m^2/yr in graminoid/forb barrens in the polar desert of the High Arctic, to as much as 1,000 g/m^2/yr in the lowland riparian areas of the Low Arctic.

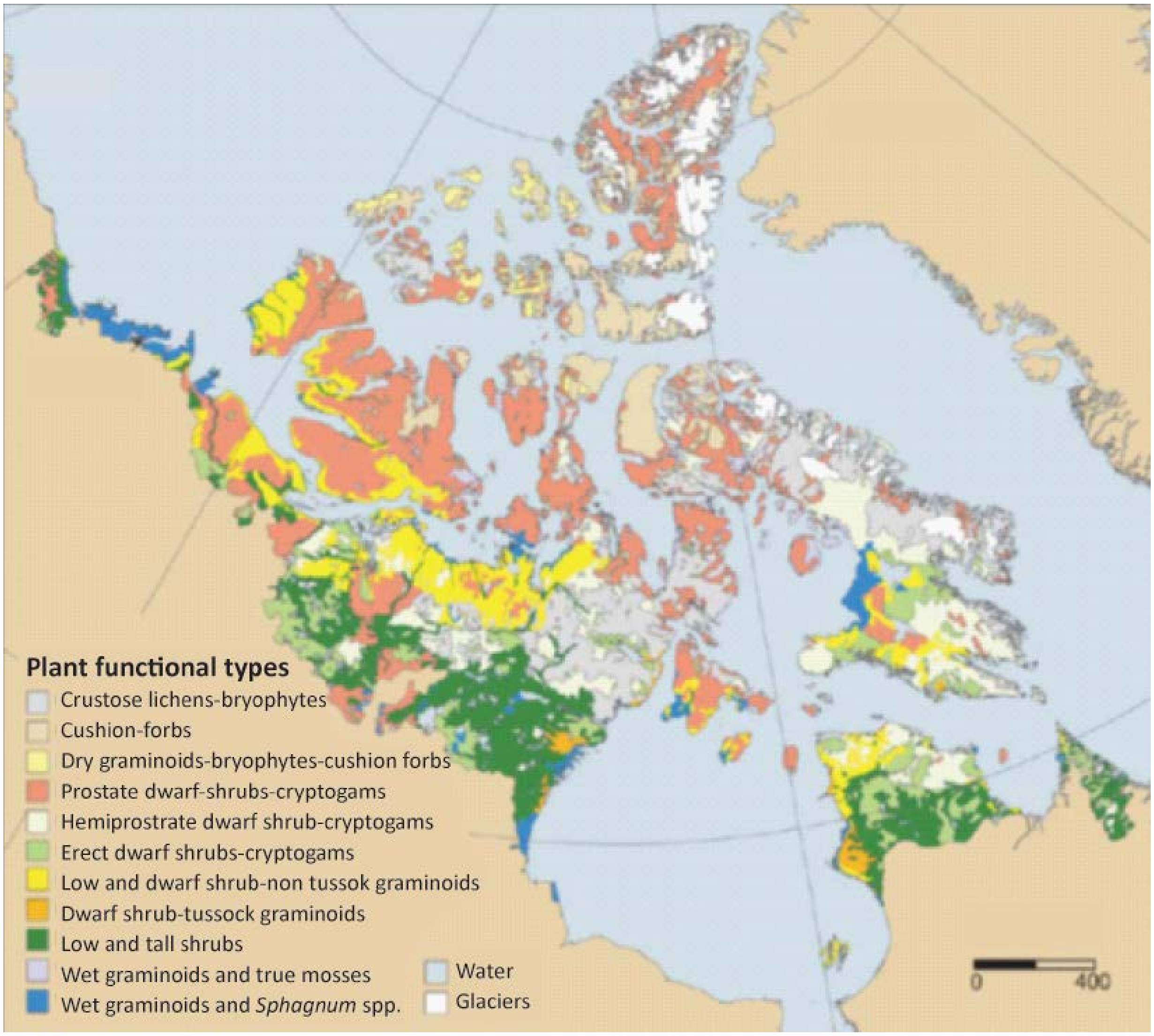


Figure 44. Vegetation map of dominant plant functional types of the Canadian Arctic.
Based on Advanced Very High Resolution Radiometer (AVHRR) data.
Source: Gould et al., 2003

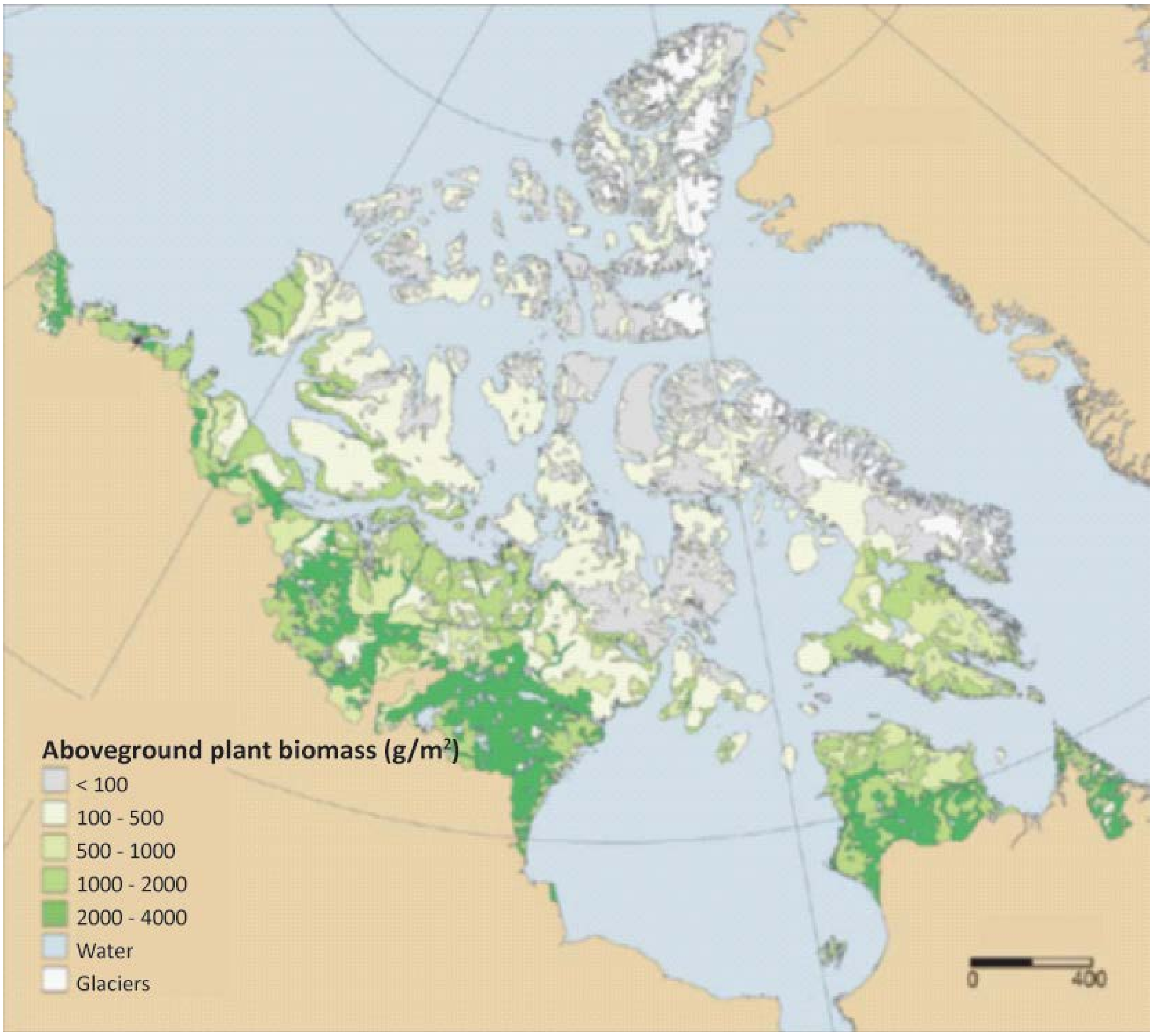


Figure 45. Above-ground plant biomass in the Canadian Arctic.
Biomass estimated by determining the range of published biomass values for the dominant vegetation type of each polygon in Figure 44.
Source: Gould et al., 2003

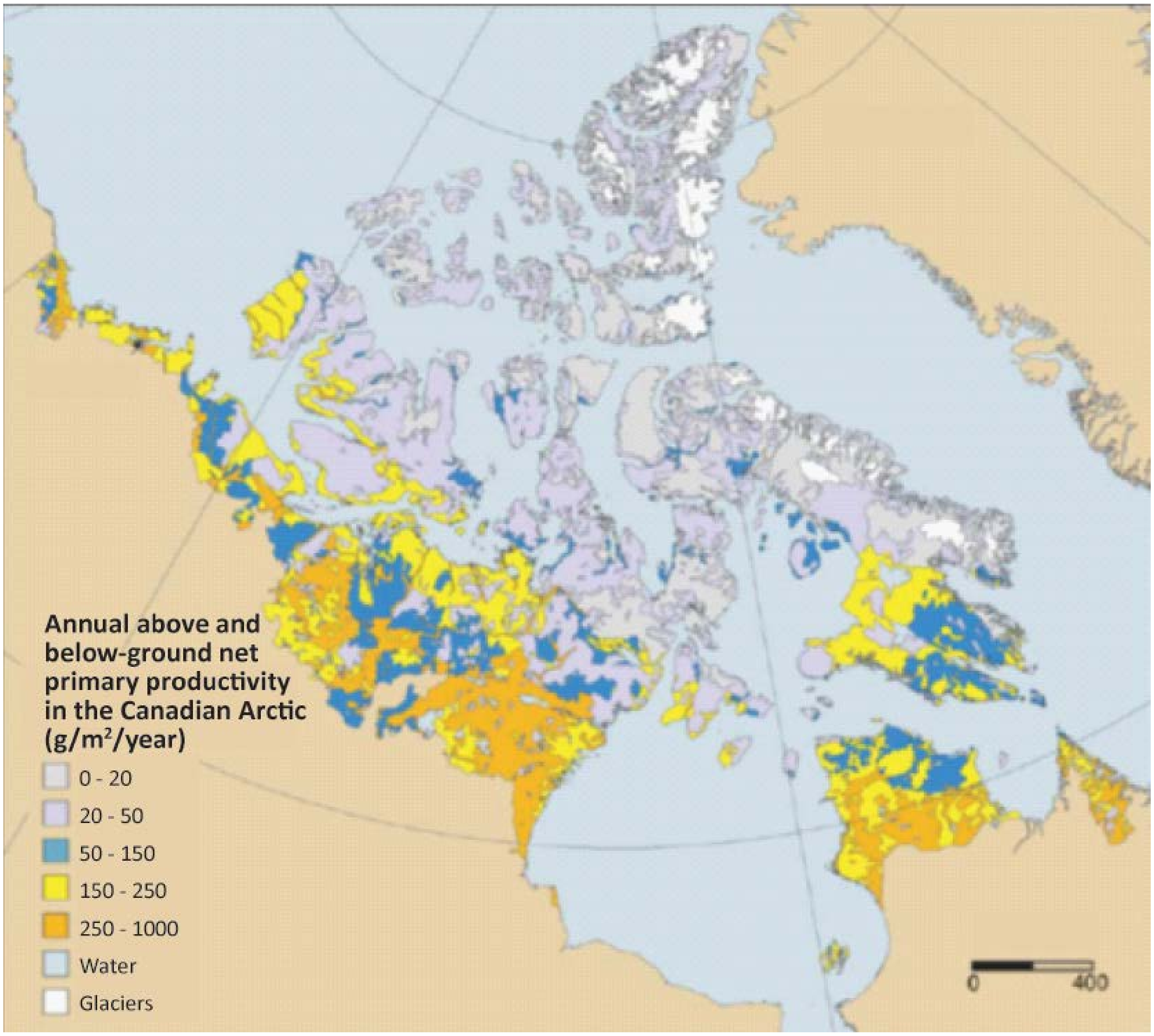


Figure 46. Net primary productivity in the Canadian Arctic.
Source: Gould et al., 2003

Experimental warming research at Alexandra Fiord, Ellesmere Island, has shown that a slight increase in summer temperature (about 1°C) results in significant increases in plant growth (Jones et al., 1997; Jones et al., 1999). Significant increases in growth were also noted in similar studies throughout the tundra biome as part of the International Tundra Experiment (ITEX) (Henry and Molau, 1997; Arft et al., 1999; Hollister et al., 2005; Jónsdóttir et al., 2005; Wahren et al., 2005; Walker et al., 2006; Elmendorf et al., 2012a). A meta-analysis of plant community change across the ITEX network showed that up to eight years of experimental warming caused significant increases in growth of shrubs and graminoids (Walker et al., 2006). After nearly 20 years of experimental warming at some sites, the same general patterns were found by Elmendorf et al. (2012a) (Figure 47).

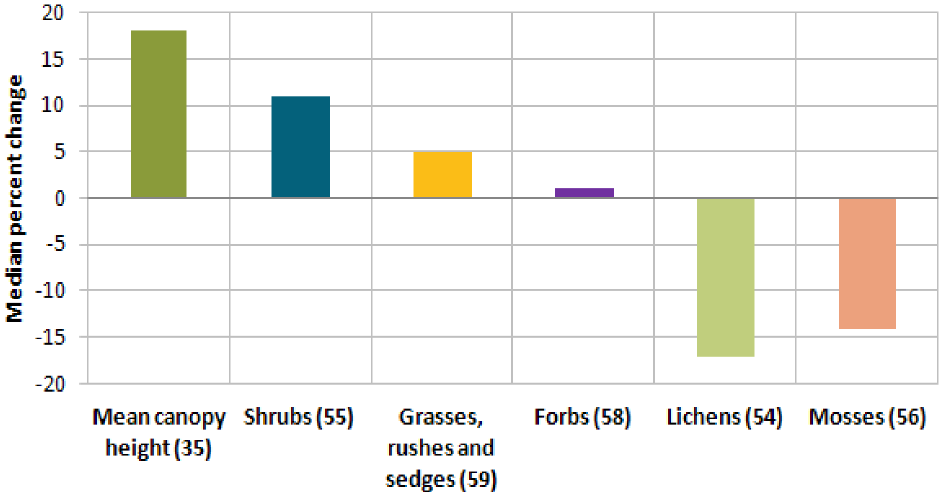


Figure 47. Results of meta-analysis of plant community changes at circumpolar ITEX sites after 1-6 years of experimental warming.

Species abundance and canopy height were measured using the standardized ITEX protocols (100 point counts of plants within a 1 m^2 plot). On average, daytime air temperature above ground in the experimental plots was raised by 1.5°C over the ambient temperatures in the control plots. The median percent change is for experimental plots in comparison with control plots. Numbers in parentheses are the number of studies included in calculations for each category. A total of 61 Arctic and alpine studies were used in the meta-analysis, five of which are in Canada. Experimental warming had been applied for fewer than six years for the majority of studies.

Source: adapted from Walker et al., 2006

Recent results from long-term studies at Alexandra Fiord, Ellesmere Island, show that there has been a significant increase in biomass (net production) in Canadian High Arctic tundra over the past 20-plus years in response to climate change (Hill, 2006; Hudson and Henry, 2009; Hill and Henry, 2011):

- Wet sedge tundra biomass increased both above and below ground, with increases from the early 1980s to 2005 of 158% in above-ground vegetation biomass, 67% in root biomass, and 139% in rhizome biomass (Figure 48) (Hill and Henry, 2011).
- Snow-bed heath community biomass (at the same site) increased in biomass by 160%, from 33 g/m^2 in 1981 to 87 g/m^2 in 2008 (Hudson and Henry, 2009). Figure 49 shows the changes in major plant groups in this heath community since 1995.

The increase in biomass in the heath community was in contrast to the effects of long-term experimental warming at the same site, where no increases in overall biomass were found in response to summer season warming of about 1°C (Hudson and Henry, 2010). The increase in average air temperature in the vicinity of the study area over the same time period has been nearly 2°C, which may indicate that certain tundra plant communities will be resistant to relatively small increases in temperature (Figure 50, and see also Figure 14 in the section on Climate trends since 1950 on page 17).

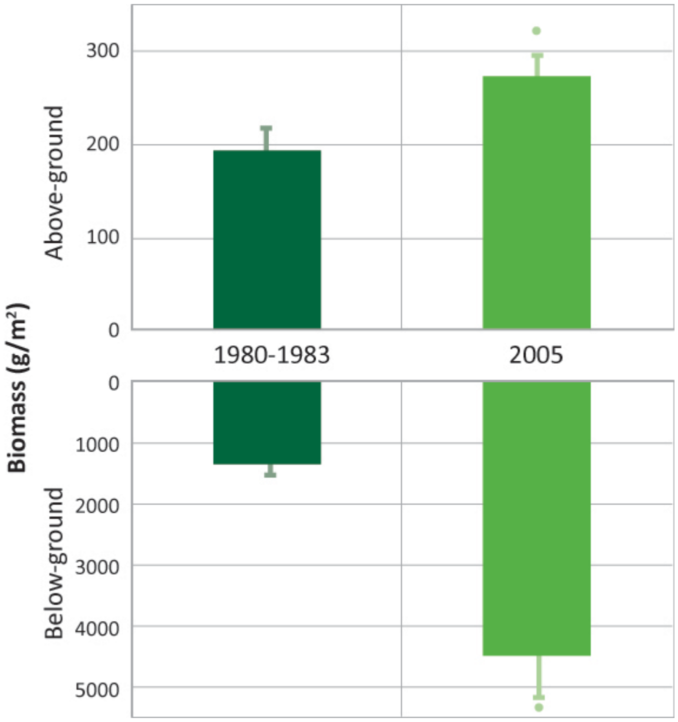


Figure 48. Change in above-ground and below-ground wet sedge meadow biomass over 25 years at Alexandra Fiord, Ellesmere Island.
Source: after Hill and Henry, 2011

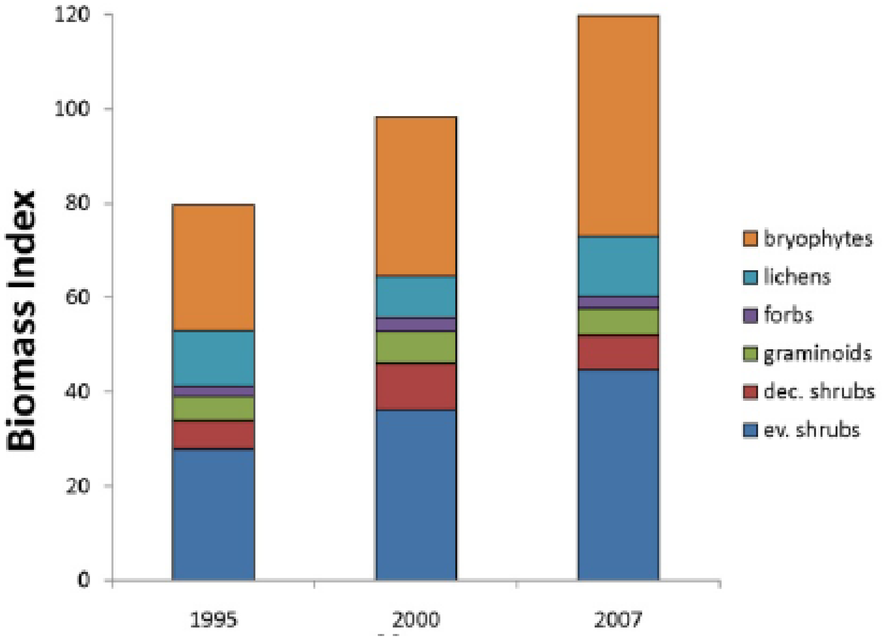


Figure 49. Changes in point cover (biomass index) of major plant functional groups in a snow bed heath community at Alexandra Fiord, Ellesmere Island, 1995-2007.
Cover was measured in permanent plots established in 1992 using the ITEX protocols for point frames.
Source: Hudson and Henry, 2009

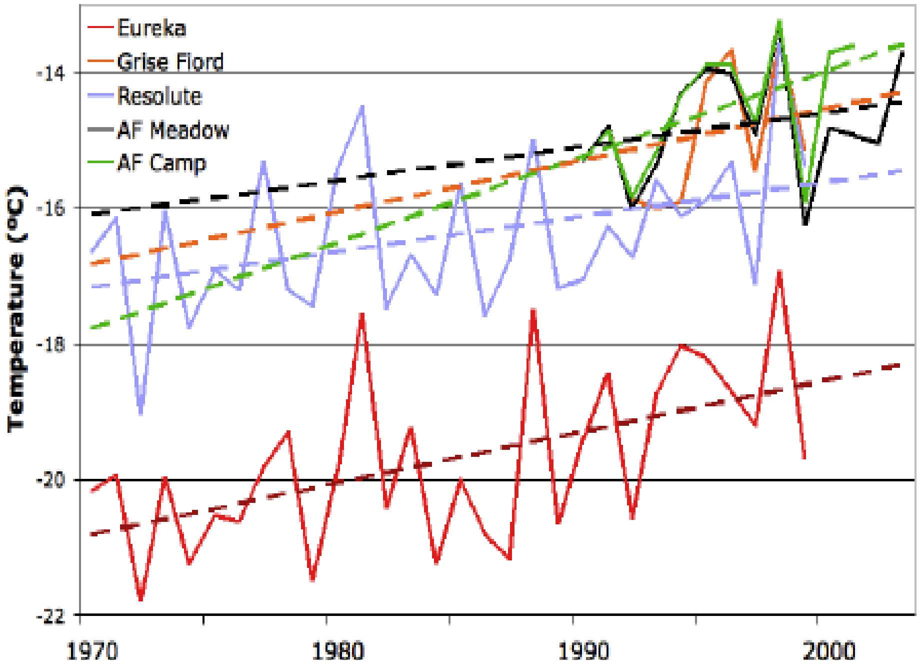


Figure 50. Mean annual air temperature at four sites in the Canadian High Arctic, 1970–2005. AF is Alexandra Fiord. Mean annual air temperature change is approaching 1°C per decade. Source: Hill, 2006

Case study on ecosystem functions and processes: Bylot Island

This case study was prepared by G. Gauthier and D. Berteaux for the Arctic biodiversity assessment (CAFF, 2013) and has been revised based on Gauthier et al. (2013).

Site description

Bylot Island is located in the Canadian Arctic Archipelago, at 73°N, 80°W. The 11,067-km^2 island is covered by mountains culminating at 1,905 m, an icecap, and several glaciers. The southern part of Bylot is a 1,600-km^2 plain gently sloping from about 400 m elevation near the mountains to sea level at the coast, and is bisected by several glacial rivers and creeks (Figure 51). Wet polygons abound in low-lying areas. This plain is covered by tundra vegetation that is rich for its latitude, owing to its southern exposure and protection from northerly winds by high mountains. It is dominated by prostrate dwarf-shrubs, graminoids, forb tundra, and, in polygonal areas, sedge/grass moss wetlands.

Small to intermediate body size species dominate the wildlife assemblage as caribou (*Rangifer tarandus*) and muskox (*Ovibos moschatus*) are absent. The bird fauna is especially rich (71 recorded species, 45 as breeders) and comprises a snow goose colony (*Chen caerulescens*) of about 20,000 pairs. Geese, brown lemmings (*Lemmus trimucronatus*), and collared lemmings (*Dicrostonyx groenlandicus*) are the most abundant herbivores. Several species of avian predators, Arctic fox (*Vulpes lagopus*), and ermine (*Mustela erminea*) are present.

Average annual temperature is –14.5°C: 4.5°C in summer and -32.8°C in winter (Gauthier et al., 2011). From 1976 to 2010, the area experienced a strong warming trend in the autumn (September to November), increasing 4.3°C over a 35-year period, and in spring and summer, increasing 2.8°C over the same period, but not in winter (December to February) (Gauthier et al., 2011). The Intergovernmental Panel on Climate Change (IPCC) (2007) projects a 3 to 6°C increase in annual average surface air temperatures from 1980–1999 to 2080–2099 for the area.

Figure 51. Bylot Island and research station location.
The southern part of Bylot Island, where continuous monitoring and intensive studies in the fields of climatology, animal and plant ecology, geomorphology, and limnology have taken place since 1989, is a gently sloping plain bisected by several glacial rivers and creeks. The arrow points to the main research station (Photo: D. Berteaux).

Monitoring and research

Two 100 km^2 study areas centered on glacial valleys have been the focus of continuous monitoring and intensive observational and experimental studies in the fields of climatology, animal and plant ecology, geomorphology, and limnology since 1989. Snow geese, lemmings, Arctic foxes, snowy owls (*Bubo scandiacus*), long-tailed jaegers (*Stercorarius longicaudus*), Lapland longspurs (*Calcarius laponicus*) and, more recently, several species of shorebirds, are the most intensively studied species. Annual plant production in wetlands, plant phenology, and seasonal insect abundance are also monitored. Traditional knowledge on foxes and geese has been collected from members of the community of Pond Inlet located south of Bylot (Gagnon and Berteaux 2006 and 2009).

Key results

The trophic dynamic on Bylot Island is dominated by regular, three- to four-year cycles in lemming abundance (Figure 52). Brown lemming populations show strong fluctuations (greater than forty-fold) but collared lemmings show relatively weak fluctuations (about four-fold) (Gruyer et al., 2008). Predators like foxes, ermines, owls, and jaegers quickly track these fluctuations and their number, reproductive activity, and lemming consumption rate increase dramatically in peak lemming years (Therrien, 2012; Tarroux et al., 2012) . This, in turn, has indirect effects on other species like snow geese due, in part, to prey switching by shared predators (Bety et al., 2002). Goose grazing has a significant impact on the wetland vegetation during the summer, but lemming grazing appears to have little impact on plant production, even in years of high abundance (Gauthier et al., 2004). A trophic balance model showed that less than 10% of the total annual primary production is consumed by herbivores, but 20 to 100% of the herbivore production is consumed by predators (Legagneux et al., 2012). This suggests that predation plays a key role in the functioning of this ecosystem.

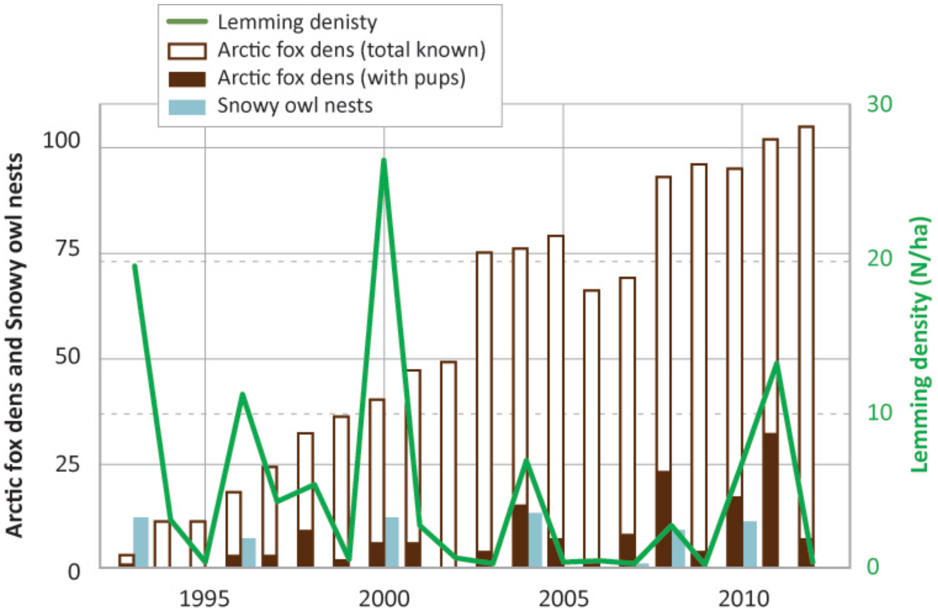


Figure 52. Temporal trends in lemming abundance and Arctic fox and snowy owl reproductive activity. Figure shows: lemming trends (combined brown and collared lemmings), Arctic fox reproductive activity, and snowy owl reproductive activity on Bylot Island.
Source: Provided by G. Gauthier and D. Berteaux

Allochthonous subsidies (energy and resources brought in from outside of the ecosystem) may be important to maintain high predator populations. For instance, high goose populations, which are in part fuelled by an agricultural food subsidy obtained in winter, may help to sustain fox populations, especially in low lemming years (Giroux et al., 2012a). In winter, predators like snowy owls and Arctic foxes use the sea ice for extensive periods, though this may be variable among years for foxes (Therrien et al., 2011; Tarroux et al., 2012), and thus they may depend upon the marine environment for their survival.

The Bylot Island environment has experienced significant changes in temperature and snow characteristics over the study period (Gauthier et al., 2013). Cumulative annual thawing degree-days increased by 37% (Figure 53) from 1989 to 2011 and average temperatures increased in spring and summer, including an increase in June of 3°C per decade, over the same time period. Snowmelt date advanced by 4 to 7 days from 1989 to 2012 and snow depth increased by 48% from 1994 to 2010. Mean annual ground temperature at 10 cm depth showed no trend.

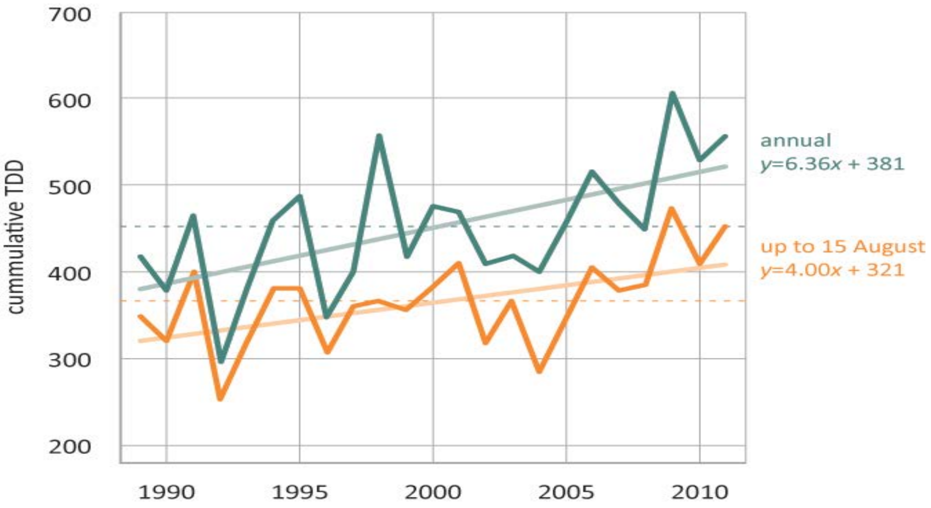


Figure 53. Cumulative thawing degree-days (above 0°C) in Qarlikturvik Valley, Bylot Island, 1989-2011. Dotted lines show the mean for the whole period.
Source: Gauthier et al., 2013

The strongest temporal trend detected in the Bylot Island ecosystem is an almost doubling (87% increase) of annual above-ground graminoid production (mostly *Dupontia fisheri* and *Eriophorum scheuchzeri*) in wetlands over a 20-year period (Figure 54). This is largely due to an increase in summer temperature—the sum of thawing degree-days explains a significant proportion of the annual variation in plant growth (Gauthier et al., 2011). The proportion of the primary production consumed by herbivores also showed a decreasing trend over time. Annual climatic variation is the most important driver of the annual production of young in several migratory birds, including snow geese, as warm spring temperatures increase their breeding effort and advance their phenology (Dickey et al., 2008; Morrissette et al., 2010).

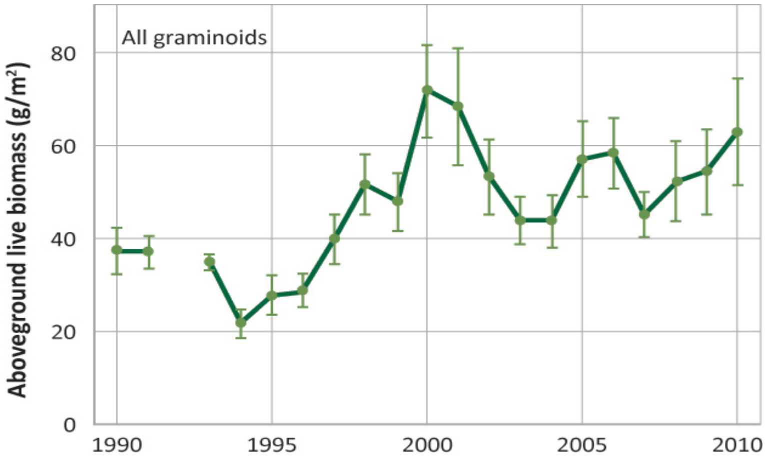


Figure 54. Temporal trend in above-ground live dry biomass of graminoid plants in wetland of Bylot Island, 1990–2010.
Mean ±SE (n = 12 samples per year; trend equation based on individual samples).
Source: Gauthier et al., 2011

Despite these strong links with temperature and the observed warming trend, we have yet to see climate-induced trends in most wildlife populations (Gauthier et al., 2013). There is no evidence that lemming cycles have dampened or disappeared in recent years, as has occurred at other sites. Recent analyses suggest, however, that snow depth and quality (density) can affect winter abundance of lemmings and the amplitude of cyclic fluctuations (Duchesne et al., 2011; Bilodeau et al. 2013a and 2013b). The snow goose population has increased significantly over the past 25 years, but this is due to events unrelated to what is happening in the Arctic, such as change in agricultural practices on their wintering ground (Gauthier et al., 2005).

Nonetheless, the more rapid response of lower trophic levels (plants) than higher levels (herbivores and predators) to climate warming may lead to a trophic mismatch. We have evidence that this is already occurring in geese. In years with an early spring, gosling growth is reduced because plants mature too rapidly and the young hatch too late, after the peak in plant nutritive quality (Dickey et al., 2008; Gauthier et al., 2013; M. Doiron, unpublished data). The median date that geese lay their eggs is related to snowmelt: the earlier the snow melts, the earlier the geese lay their eggs (Figure 55) (Gauthier et al., 2013). The geese undercompensate, however—for an advance in snowmelt date of 10 days, they advanced their laying date by only 3.8 days. Plant growth tracked changes in snowmelt much more closely, indicating a potential for mismatch in timing between hatching and the supply of high-nutrient food in years of early snowmelt.

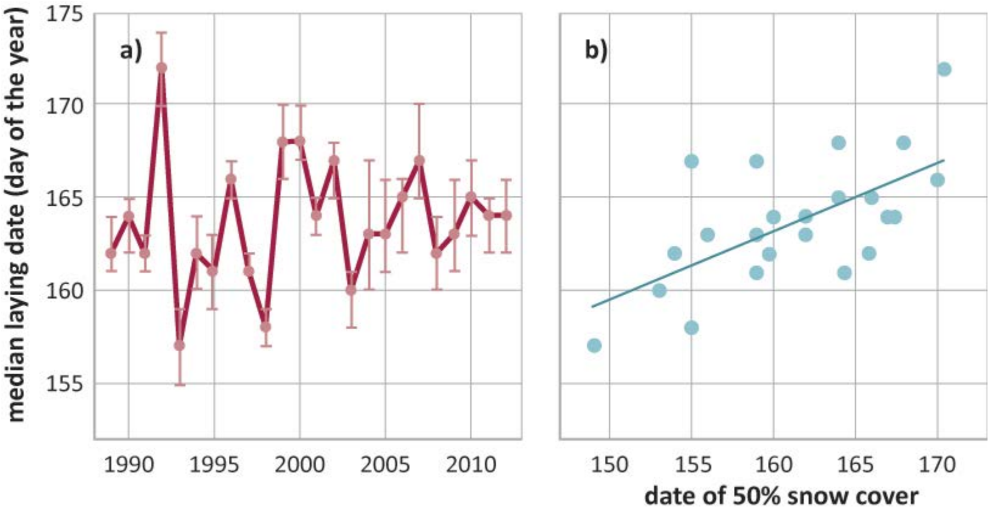


Figure 55. Annual mean goose laying date and relationship between goose laying date and date of 50% snow cover, Qalikturvik Valley, Bylot Island, 1989-2012.
In Figure (a), bars are 25% and 75% percentiles. The regression line in Figure (a) is not significant.
Source: Gauthier et al., 2013

Other aspects of ecosystem change include changes in species distribution and in wetlands. The red fox (*Vulpes vulpes*) invaded the Bylot area in the late 1940s (Gagnon and Berteaux, 2009) and is now reproducing regularly. Drainage and loss of some productive wetlands due to the rapid thermal erosion of ice wedges forming polygons during periods of high spring run-off is a source of concern in this area because such events are likely to increase with climate warming

(Fortier et al., 2007). Most of Bylot has been part of Sirmilik National Park since 2001, and this study was a main source of data for evaluating potential changes to the ecological integrity of the park (McLennan et al., 2012). However, the Nunavut Field Unit of Parks Canada has now switched to a much less detailed protocol of ecological integrity monitoring that does not include wildlife monitoring on Bylot Island (D. Berteaux, pers. comm.).

Human stressors on ecosystem functions and processes

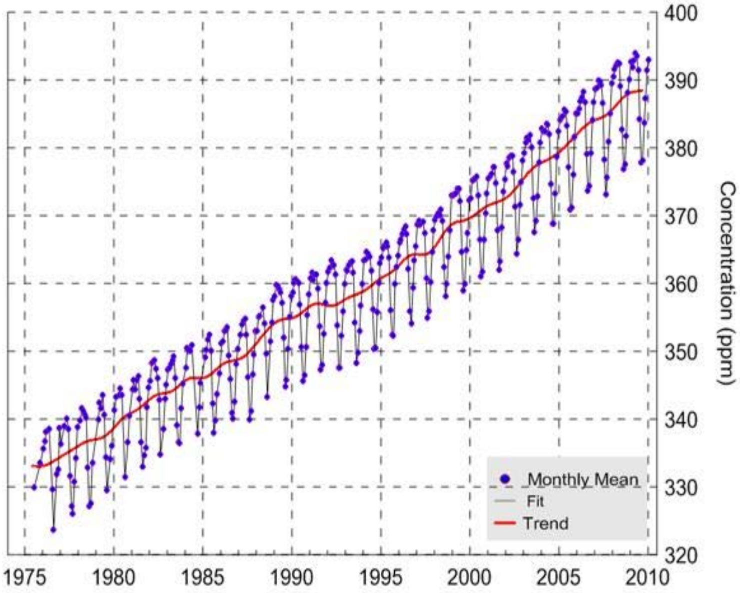
Climate change

Climate change is a major stressor for tundra ecosystems in general (ACIA, 2005). At local scales, effects of roads and other potential human disturbances are also important.

While climate change is expected to affect the entire globe, general circulation models consistently indicate that the most severe climate warming will occur in polar latitudes (ACIA, 2005; IPCC, 2007). The Arctic is currently warming at about double the rate of the global average, and some of the most marked changes have occurred in parts of the Canadian Arctic (IPCC, 2007; AMAP, 2011). Paleoecological data confirm that Arctic climate warming during the last century has been well outside the normal range of the previous 400 years and, though there is a natural component to climate change, including at regional and temporal scales related to oscillations, it is clear that the rise in greenhouse gases in the atmosphere due to human activities is a significant human stressor on Arctic ecosystem functions and processes. The impacts of this stressor are woven throughout the discussion in this and other sections of this report. See also the section on Climate trends since 1950 (page 17).

Climate change is a result of anthropogenic increases in greenhouse gases (IPCC, 2007). Trends in the main greenhouse gas, carbon dioxide, are shown in Figure 56. Air samples from Alert, Nunavut, distant from interference by local sources, are representative of global atmospheric carbon dioxide levels. Annual fluctuations are due to seasonal release of carbon dioxide by plant growth in the Northern Hemisphere. Ice core gas samples from Antarctica show the rapid increase of previously stable carbon dioxide concentrations starting in the late 19th century.

A. Atmospheric carbon dioxide measured at Alert, Nunavut, since 1975



B. Atmospheric carbon dioxide over the past millennium, based on Antarctic ice cores

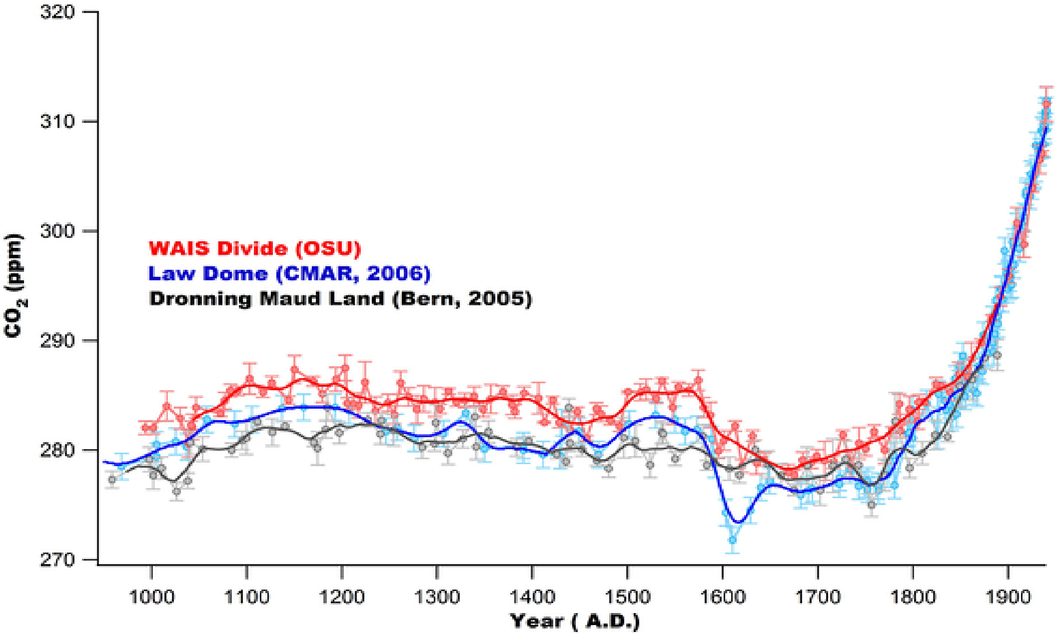


Figure 56. Atmospheric carbon dioxide trends, A) measured at Alert, Nunavut (1975–2010) and B) thousand-year trends based on Antarctic ice cores.

Alert measurements are based on weekly air samples. Long-term trends are based on analysis of gas in ice cores from three locations in Antarctica.

Source: (A) Environment Canada, 2012a; (B) Ahn et al., 2012

Other global-scale stressors

There are other human stressors related to large-scale change that have known or potential impacts on biodiversity in the Arctic, often by interacting with climate change. A full discussion of these is beyond the scope of this report. Changes to the atmosphere that potentially affect biodiversity include ozone depletion and consequent rise of UV-B levels (ACIA, 2005), and Arctic haze (see box below). Of recent concern is the impact of black carbon (soot) from natural and industrial sources. The net effect of black carbon (both in the atmosphere and settling on snow) on the Arctic climate is complex and not well understood, with most studies being based on modeling of interacting factors. Model-based analyses show that the effect of atmospheric black carbon varies considerably depending on the altitude at which it is present in the atmosphere (Flanner, 2013).

Arctic haze

By the 1990s, an unusual haze was noticed in the Arctic that was subsequently linked to emissions of sulphurous and nitrous aerosols. The principal source was air emissions from iron smelting, particularly in Russia. With improved international emission standards, sulphur sources have declined. Non-ferrous metal production remains the dominant source of emissions of acidifying gases to the atmosphere within the Arctic (AMAP, 2006). Other significant anthropogenic sources of sulphur emissions within or close to the Arctic include energy production plants and mining industries. Sources of nitrogen emissions within the Arctic include transportation, in particular shipping and oil and gas activities. Monitoring at Alert showed that, although sulphur aerosols have declined, levels of nitrate aerosol are increasing during the haze season. Forest fires produce soot, which also contributes to haze.

The causes and the effects of acidifying air pollutants and Arctic haze are closely linked to other environmental problems, though these relationships are not well understood. It is not clear, for example, how climate change will influence future acidification and Arctic haze pollution. The effects of haze-producing aerosols on the Arctic climate are complicated by feedbacks between aerosols, clouds, radiation, snow and ice cover, and vertical and horizontal transport processes. Whether the pollutant aerosols cause an overall warming or an overall cooling, or whether the haze itself may affect Arctic ecosystems, is not yet known (AMAP, 2006).

Ecosystem structure

Changes in the extent and quality of important biomes

Status: land cover classification and biomes

There are several classification systems that define biomes slightly differently. This report uses a generalized land cover classification system derived from the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005; Frisk, 2011). In the Arctic Ecozone$^+$, polar barrens, polar tundra, and wetlands cover the largest areas (Figure 57). Lakes and rivers, which comprise a small proportion of the total area of the ecozone$^+$, were excluded from the analysis. The five broadly defined biomes in the Arctic ecozone$^+$ are defined as:

1) Polar tundra – treeless regions with greater than 50% vegetation cover
2) Polar barren – treeless region with less than 50% vegetation cover
3) Wetlands – areas saturated for sufficient time to promote wetland or aquatic processes
4) Mountains – steep and high lands with specific criteria related to elevation and slope
5) Snow/ice/glaciers – masses of perennial snow and ice with definite lateral limits, typically flowing in a particular direction

Climatically, "Arctic" is defined as the region north of the 10°C summer isotherm; ecologically, it is the area north of the treeline.

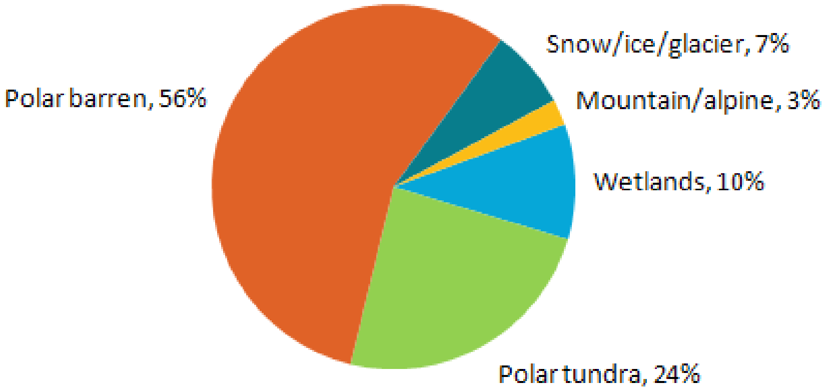
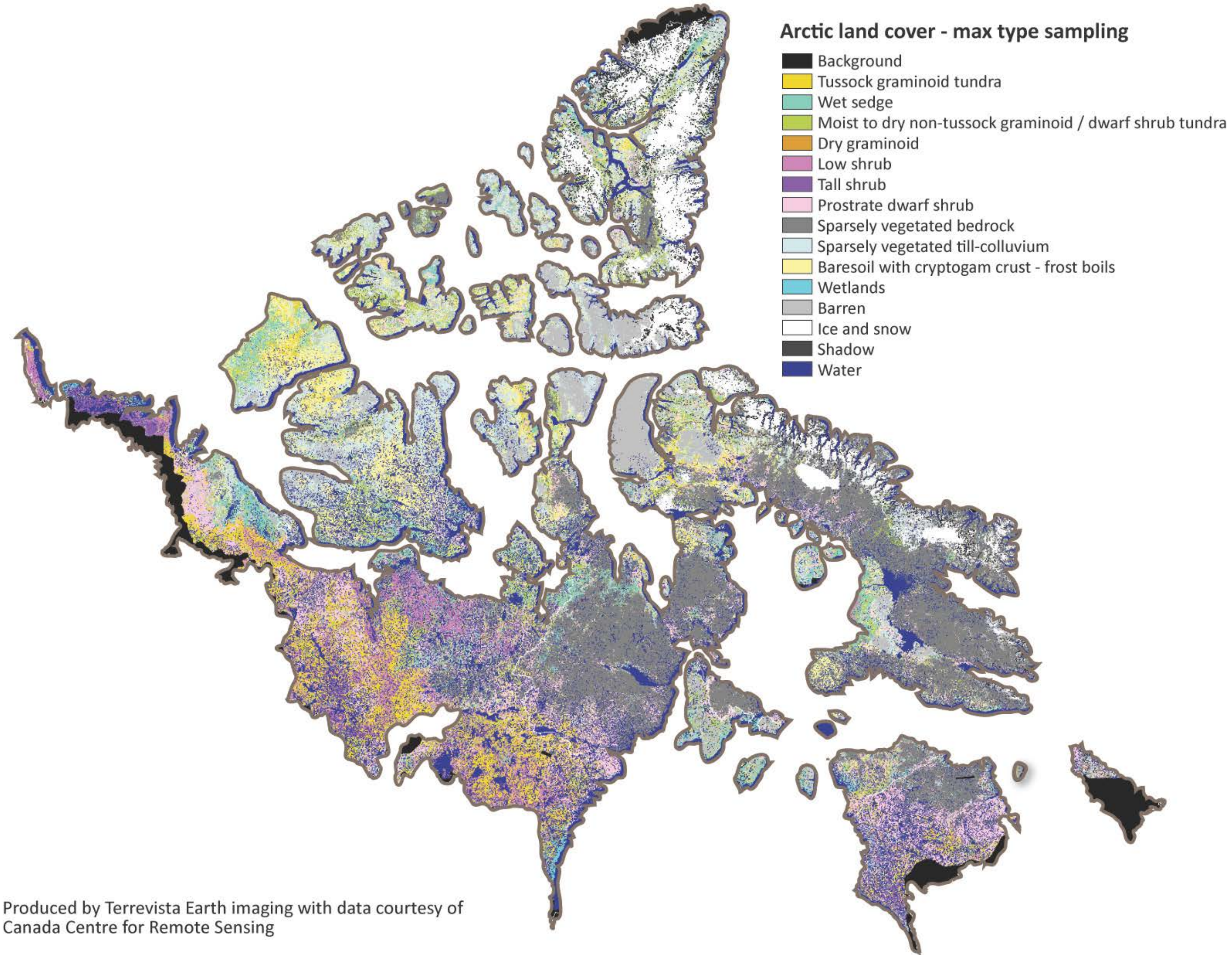


Figure 57. General land cover classes (biomes) in the Arctic Ecozone+.
See Appendix 2: Detailed land cover classes for more information.
Source: Ahern, 2010; Frisk, 2011; Ahern et al., 2011, using data from the Canada Centre for Remote Sensing

A more detailed analysis of land cover in the Arctic Ecozone+ (Olthof et al., 2005) has been developed by the Canadian Centre for Remote Sensing and was analysed for this report (Ahern, 2010) (Figure 58). This analysis uses 14 land cover classes, 12 of which are found in the Arctic Ecozone+ (see Appendix 2: Detailed land cover classes for more information) (Ahern, 2010; Frisk, 2011; Ahern et al., 2011). This analysis classifies the Arctic Cordillera as mostly snow and ice, mountains, and sparsely vegetated bedrock or till colluvium. The Northern Arctic is dominated by barren ground or sparsely vegetated bedrock, till colluviums, or barren ground. The Southern Arctic is dominated by low and dwarf shrubs.



Produced by Terrevista Earth imaging with data courtesy of
Canada Centre for Remote Sensing

Figure 58. Detailed land cover in the Arctic Ecozone^{+}.
Source: Olthof et al., 2005 with additional analysis provided by Ahern, 2010

Circumpolar perspective

Analyses conducted for the production of the circumpolar Arctic vegetation map (Walker et al., 2005) produced the following summary results:

- Within the Arctic (total area = 7.11 x 10^6 km^2), about 5.05 x 10^6 km^2 is vegetated. The remainder is ice-covered.
- About 26% of the vegetated area is erect shrublands, 13% peaty graminoid tundra, 13% mountain complexes, 12% barrens, 11% mineral graminoid tundra, 11% prostrate-shrub tundra, and 7% wetlands.
- Canada, among Arctic nations, has the largest portion of Arctic terrain (36%) and has by far the most in the High Arctic ecosystem categories (63% of the global total), mostly associated with abundant barren types and prostrate dwarf-shrub tundra, whereas Russia has the largest area in the Low Arctic, predominantly low-shrub tundra.

Habitat diversity is also an important component of ecosystem structure. Some habitat types assume a particular importance because they are rare and are important, even critical, to the life cycle of particular species. The following examples illustrate the patchy distribution and importance of small habitat elements in a large landscape:

- Eskers occupy a small proportion of the Arctic but may be critically important as denning habitat for wolves, grizzly bears, and ground squirrels (Mueller, 1995; McLoughlin et al., 2004).
- Grizzly bears in the Mackenzie Delta can only den in river cut-banks in consolidated, upland soils of Pleistocene origin (or, rarely, pingos), not in more recent fluvial deposits of Holocene origin (Harding, 1976).
- Nesting shorebirds and geese tend to associate with wetland and riparian habitats that are unevenly distributed on the coastal plain (Brown et al., 2007c).

Trends in polar biomes

Reduction in tundra

There is evidence of approximately a 20% decrease over the past 25+ years in the extent of Tundra Climate, a measure of the cold temperature and low precipitation conditions that support the polar tundra, barrens, and ice and snow biomes (Figure 59) (Wang and Overland, 2004). The steady decline in the area of Tundra Climate is matched by reductions in the areas with a tundra signature in primary productivity as measured by NDVI and the strongest changes in Tundra Climate have occurred in Northwest Canada in the past 25+ years (Wang and Overland, 2004; Bhatt et al., 2010, and visible in Figure 63).

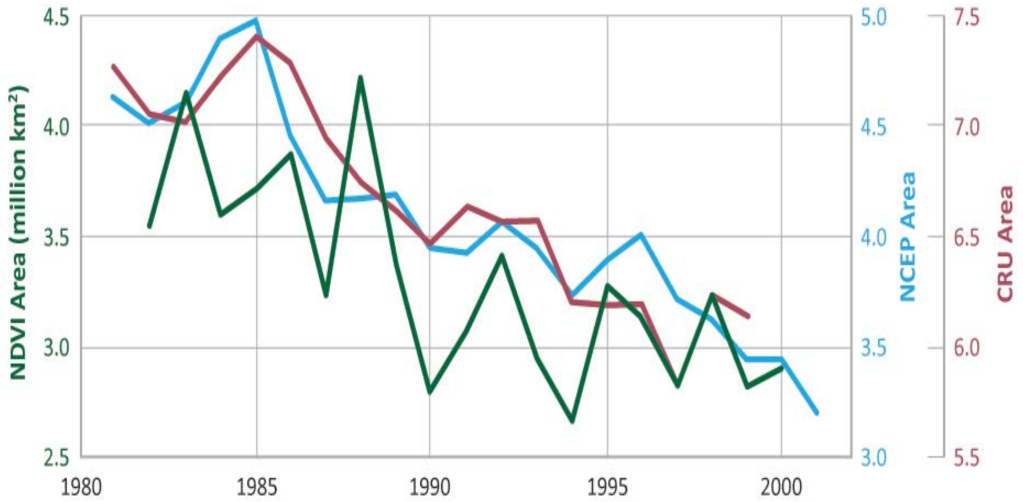
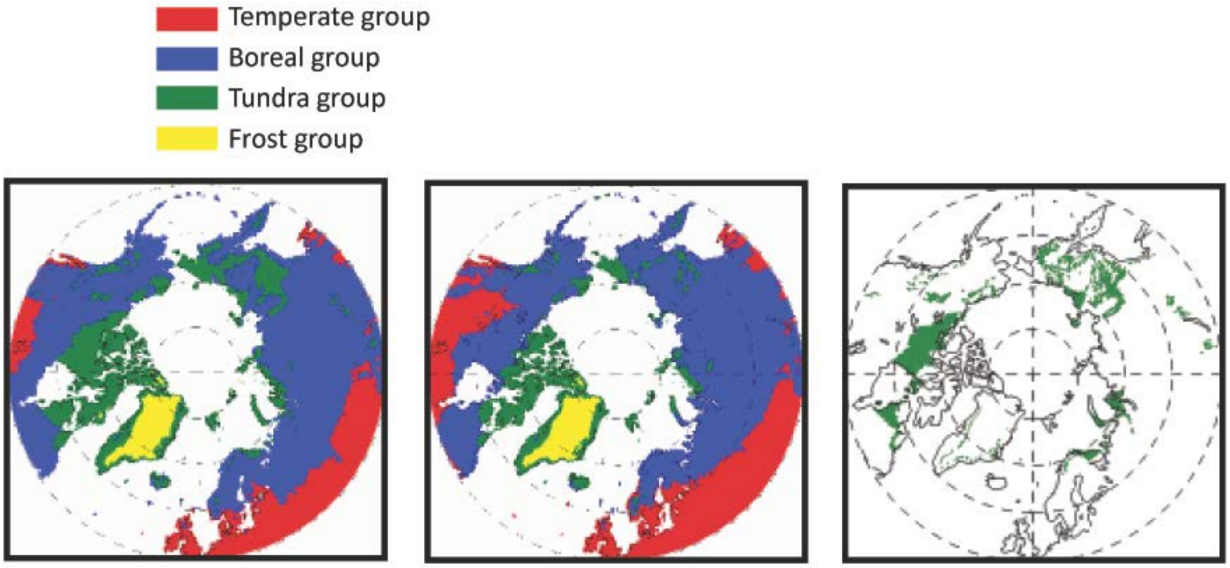


Figure 59. Changes in area of the tundra climatic zone around the circumpolar Arctic since 1982, based on NDVI and Köppen Climate Classification System.
"NCEP" and "CRU" are two datasets of monthly surface air temperatures. "NDVI" is a calculation based on light reflected by vegetation, as detected by satellite (see report section on Primary productivity).
Source: Wang and Overland, 2004



1978

1998

Lost area of tundra group
(areas that were in the tundra group in 1978 and in the boreal group in 1998)

Figure 60. Spatial distribution of difference in areas of tundra climate using the Köppen climate classifications between the two years 1978 and 1998.
Based on CRU climate dataset.
Source: Wang and Overland, 2004

The warming climate may have been the cause of increased biomass (inferred from increased NDVI) since 1986 on the Porcupine caribou range in northern Yukon and adjacent Alaska (Henry et al., 2012) (Figure 61). Structural changes south of the Arctic Ecozone+ also influence ecosystems within the ecozone+. For example, increased fire frequency is believed to be the reason for a decrease in the cover of forest tundra woodland in the winter range of the Bathurst caribou herd (Chen et al., 2013). As discussed in the section on Primary productivity (page 69), increases in biomass have been measured since 1980 in permanent plots at a High Arctic site (Figure 48 and Figure 49). These increases in biomass are likely responses to the warming climate (Figure 50).

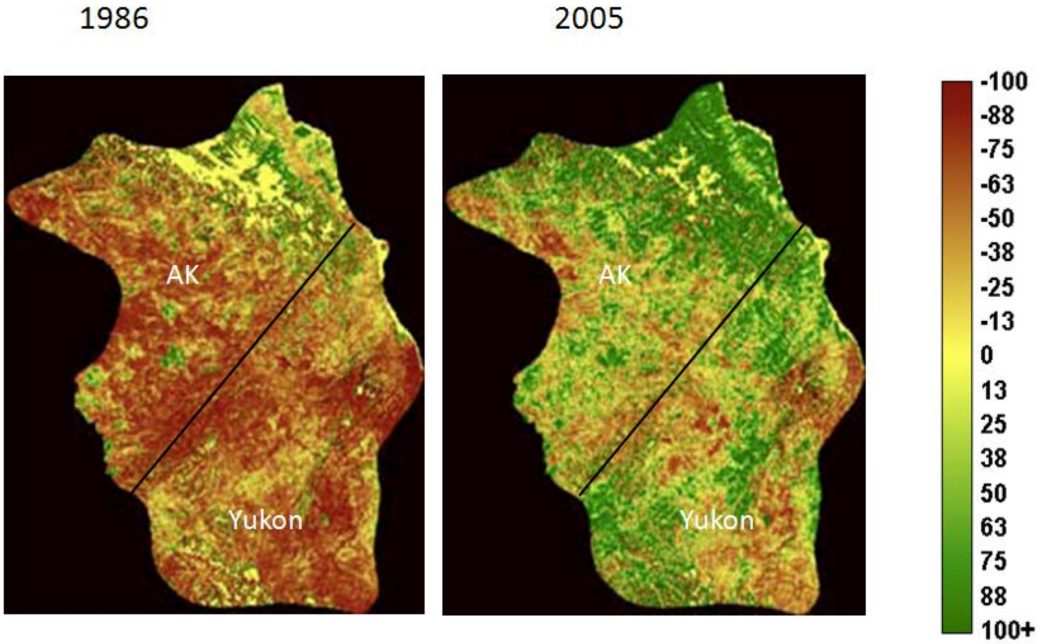


Figure 61. Changes in NDVI across the annual range of the Porcupine Caribou Herd in northern Yukon and adjacent Alaska, 1986 to 2005.
The Yukon portion of this analysis includes part of the Southern Arctic and part of the Taiga Cordillera immediately to the south. Change is relative to a 2000 baseline.
Source: Henry et al., 2012 (Supplementary data, Figure 3) based on Chen et al., unpublished data

Comparisons of historical and contemporary aerial photographs also provide evidence that Arctic vegetation has undergone significant shifts in recent decades. Increased shrub cover has been confirmed in two repeat photography studies in northern Alaska (Sturm et al., 2001; Tape et al., 2006) and one on Herschel Island in Canada (Myers-Smith et al., 2011b). A study in the Mackenzie Delta region of Canada's Southern Arctic showed that there has been an increase in deciduous shrubs in the region over the past 40 years (Lantz and Kokelj, 2008). A similar dramatic increase in shrub cover was documented by Tremblay et al. (2012) for the region around the community of Kangisuujuaq in Nunavik east of Ungava Bay (Figure 62).

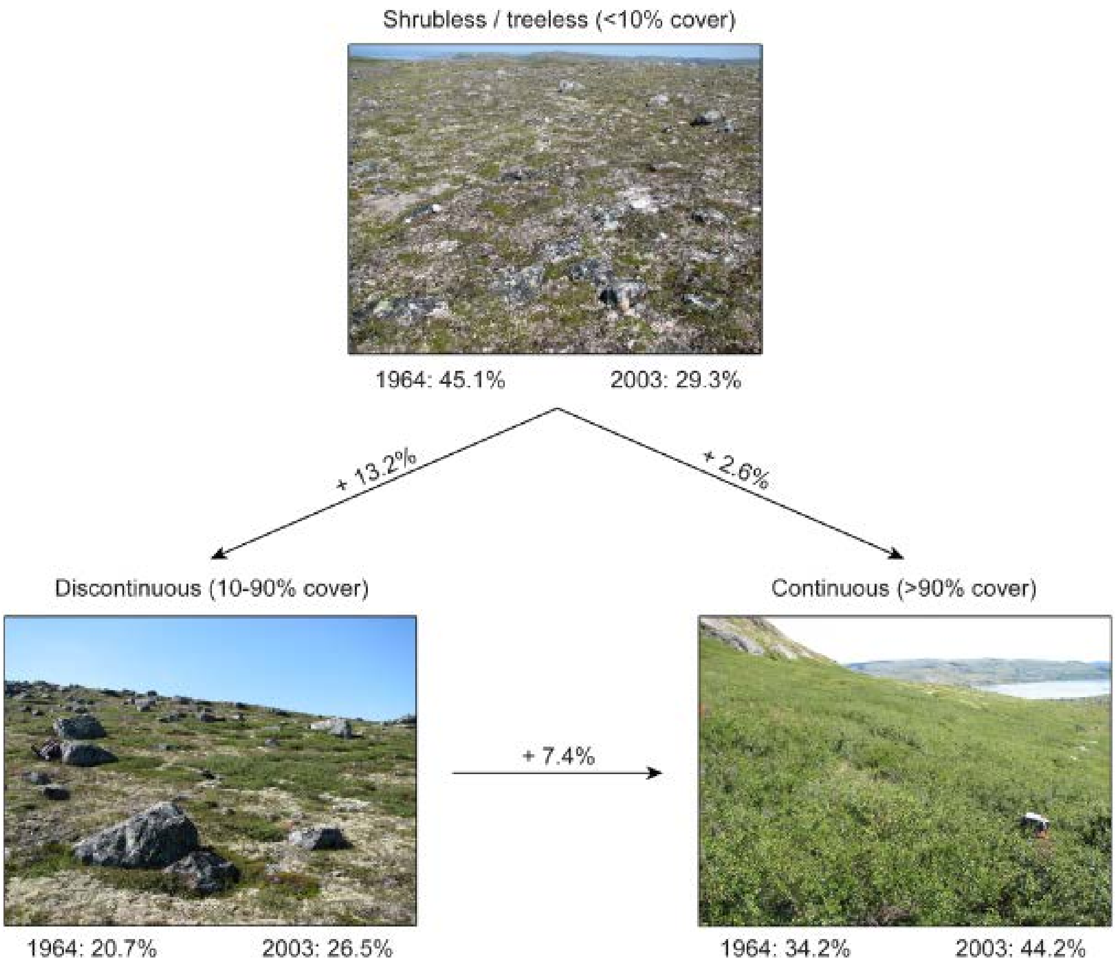


Figure 62. Changes in shrub cover in an area near Kangiqsualujjuaq, Nunavik, Quebec, 1964 and 2003, based on air photo analysis.
Values are percentages of the total study area (7.2 km^2). Arrows show direction of change from one vegetation cover type to another, accompanied by the amount of change.
Source: Tremblay et al., 2012

Plot-level data offer a more detailed view of vegetation changes than either remote sensing or aerial photography. A synthesis of resampled plots in tundra ecosystems around the world showed changes in vegetation over the past 10 to 30 years that are consistent with responses to warming, but with considerable variation among the sites (Elmendorf et al., 2012b). The responses were strongly controlled by the initial conditions of the site and the changes in soil moisture (Callaghan et al., 2011c; Elmendorf et al., 2012b). Resampling of plots established in the 1970s in Alaska yielded results consistent with a warming and drying trend, in which moist and wet community types tended to be replaced by dry community types through time (Hinzman et al., 2005). After resampling of control plots from a 13-year experiment at Toolik Lake, Alaska, Shaver et al. (2001) found that graminoids, mosses, and lichens were decreasing and evergreen shrubs were increasing in abundance. In a similar resampling study at a High Arctic site on Ellesmere Island, Hill and Henry (2011) found a significant increase in biomass of wet sedge tundra between 1980 and 2005 (see the section on Primary production (biomass) in tundra ecosystems on page 73).

Increased growth and cover by shrubs and infilling by tree species also threatens tundra at its southern margins in the forest-tundra ecotone. Correlations between temperature and treelines throughout the paleoecological record strongly support the notion that climate warming will advance treelines. For example, during the Little Ice Age, shrub tundra increased while treelines retreated (Tinner et al., 2008). As the northern boundary of tree distributions is often temperature limited, northern migration of trees might seem almost inevitable. However, caution in predicting the rate and locations of treeline expansion is warranted, as rates of treeline expansion may lag in some areas due to seed availability, disturbance frequency, permafrost changes, and moisture constraints (Hofgaard and Harper, 2011; Walker et al., 2012; Henry et al., 2012).

White spruce trees along the northern Quebec–Labrador treelines show different responses according to their position relative to the sea. Along the coast, invading spruce are taking hold above the current treeline, while in the interior recent warming has not been strong enough to change the regressive tree line trajectory (Payette, 2007). Treelines in the forest-tundra areas of Quebec have risen slightly either through establishment of white spruce from seed or through growth of stunted spruce already established on tundra hilltops (Gamache and Payette, 2005; Caccianiga and Payette, 2006).

Treeline is advancing in other regions of the circumpolar Arctic that are increasing in productivity (Ims et al., 2013). Expansion of white spruce forests into areas previously occupied by tundra has been documented in numerous locations in Alaska (Suarez et al., 1999; Lloyd and Bunn, 2007), while recent studies have also shown that moisture is an important limiting factor in treeline advance (Ohse et al., 2012). Treeline advance into the tundra has also been documented in mountainous areas of Siberia (Kharuk et al., 2010; Kirdyanov et al., 2012) and northwestern Siberia, where forests have expanded into tundra at the rate of 3 to 10 m per year (Kharuk et al., 2006).

Vegetation is a key component in surface energy balance, regulating climate at regional and global scales. The observed and projected increases in the cover and density of shrub species and the increasing density of trees in forest tundra areas (Sturm et al., 2001) will alter the surface energy balance of northern terrestrial systems by decreasing the albedo and increasing absorption of solar radiation (Chapin III et al., 2005; Sturm et al., 2005). The increased absorption of solar radiation will increase the heating of the atmosphere. Chapin et al. (2005) estimated that a change from tundra to boreal forest will increase the heating of the atmosphere by approximately 4 W/m^2, which is similar to the effect of doubling the CO_2 concentration in the atmosphere. This will be an increasingly important positive feedback.

Species assemblages and plant communities

Changes in algal and invertebrate species assemblages in lakes and ponds

Striking and often unprecedented changes in the biota of Arctic lakes since the1850s have been linked to ecological shifts consistent with climate warming (Smol et al., 2005). Many algal and invertebrate communities of Arctic lakes and ponds have crossed ecological thresholds and a plausible explanation is recent warming (Smol and Douglas, 2007b). Other potential causes including ultraviolet radiation, nutrient enrichment, and atmospheric transport of pollutants cannot account for the scale, the nature, and the timing of the observed changes in algal communities. The best explanation is climatic warming leading to longer ice-free growing seasons and associated limnological changes including thermal stratification, habitat availability, and lakewater chemistry (Antoniades et al., 2005; Smol and Douglas, 2007b).

Analysis of sediments to identify and measure types of fossil diatoms, as well as other indicators, has detected major changes in species assemblages starting in the mid-19th century. Sediments deposited before this time show that diatom communities were relatively stable over centuries, and in some cases millennia, compared to the recent changes. Analysis of cores from 55 circumpolar Arctic lakes (Smol et al., 2005) showed that lakes have become more productive with more diverse algal and invertebrate communities. Lakes in the Canadian High Arctic experienced abrupt shifts in favour of diatoms that thrive in littoral habitats and mossy substrates, consistent with longer growing seasons and habitat changes under warming conditions. In deeper lakes on Baffin and Ellesmere islands and in lakes throughout the subarctic, the earlier onset and longer duration of thermal stratification becomes increasingly important, with pronounced increases in planktonic diatoms that fare better under conditions of less ice cover and greater thermal stratification, and concurrent decreases in benthic diatom taxa associated with cold conditions and extensive ice cover. In contrast, no major shifts in diatom assemblages were observed over the last two centuries in lakes in Labrador and northern Quebec, areas that have experienced little warming until very recently.

Local topographic and geologic factors also influence the response to warming (Smol et al., 2005). Sediments from lakes on Ellef Ringnes Island showed a transition to different, more diverse diatom communities starting in about 1850 (Figure 63), typical of results reported from previous research (Smol et al., 2005). The same shift occurred in lake sediments near Alert, Ellesmere Island, but later, a difference that is attributed to varying sensitivity of the lakes to environmental change (Antoniades et al., 2005). For example, as reviewed by Smol and Douglas (2007b), lakes of different size would be expected to change at different time scales to climatic changes, with larger lakes typically responding more slowly than shallow ponds.

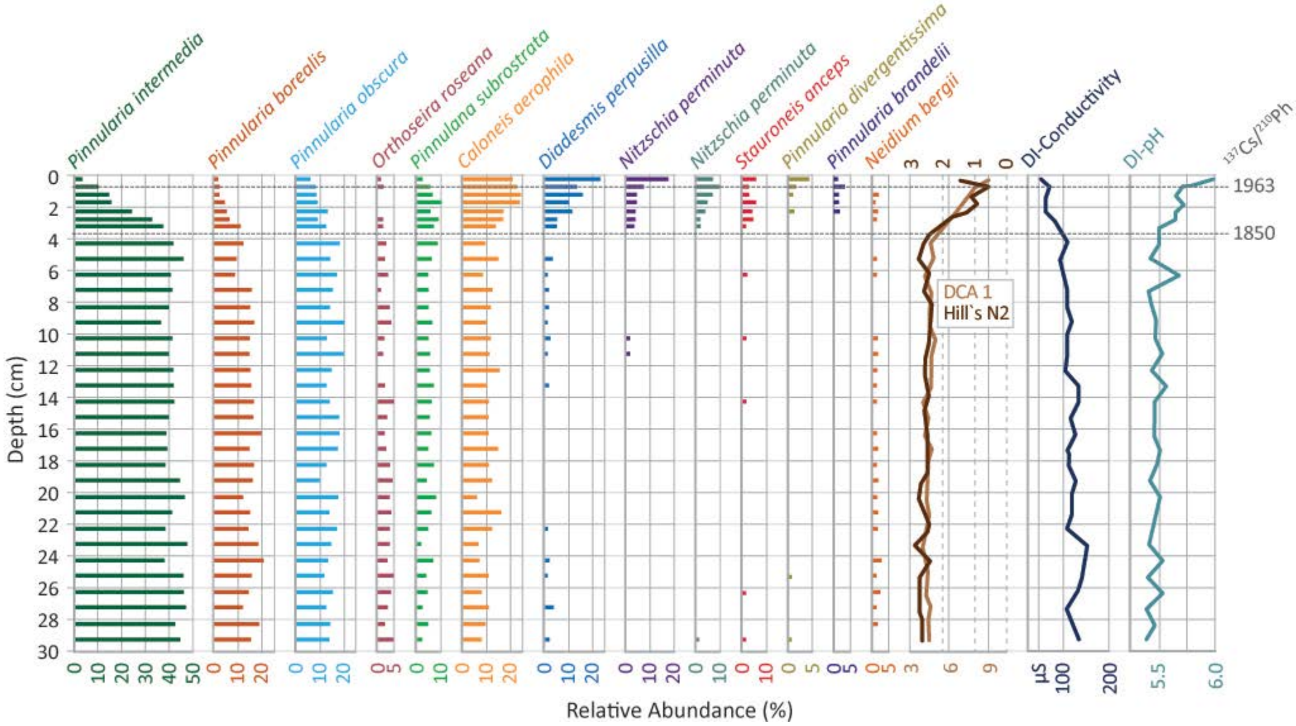


Figure 63. Diatom stratigraphy from a sediment core from a pond on Ellef Ringnes Island, showing the marked changes in species assemblages since 1850, especially in very recent years.
The core represents approximately 1,300 years of sediment deposition. The three columns on the right are characteristics inferred from the diatoms (right to left: pH, specific conductivity, and analysis of change in diatom assemblage structure by detrended correspondence analysis).
Source: Antoniades et al., 2005

In Canada's most northerly lake, Ward Hunt Lake, located at 83°N latitude and with perennial ice cover, diatoms appear only in the top 2.5 cm of sediment, corresponding to about the past 200 years of the 8,500 years of sediment records examined (Antoniades et al., 2007). This was accompanied by an increase in photosynthetic pigments (a measure of the standing crop of algae) of two orders of magnitude.

This pronounced shift in diatom assemblage composition is not a trend exclusive to the Canadian Arctic, or even just in northern latitudes. A synthesis of diatom-based paleolimnological studies of over 200 lake sediment records from around the Northern Hemisphere (Rühland et al., 2008) showed a general pattern of species-specific shifts in diatom assemblage composition since the mid-19th century that often followed millennia of relative stability in the diatom communities. Their analysis found that these changes occurred significantly earlier (by about 100 years) and often more dramatically in Arctic lakes than in lakes from temperate latitudes. They attributed this earlier and more rapid shift in Arctic lakes to the greater degree of warming that has occurred in the Arctic. The diatom species shifts observed at all latitudes were remarkably similar and occurred in conjunction with observed changes in freshwater habitat and structure that were linked to substantially warmer temperatures and changes in ice cover.

Remains of invertebrates in sediments show that the changes in lake conditions and algal communities have been transmitted to higher trophic levels. For example, aquatic insect (chironomid) populations in three Ellesmere Island ponds greatly expanded and diversified at the same time as diatom communities changed (Quinlan et al., 2005). Changes in both algae and invertebrate species assemblages are most pronounced in areas that have warmed the most, such as parts of the Canadian High Arctic (Smol et al., 2005).

Changes in tundra plant communities

Species diversity was found to decline in warming experiments across the tundra biome after up to six years of warming (Walker et al., 2006) and remained low after more than 10 years (Elmendorf et al., 2012a). The major factor in the decline in diversity was the loss of bryophyte and lichen species, which likely were shaded by the increasing cover and height of the vascular plants (see Figure 47).

As species composition changes along gradients, and as local gradients in soil moisture and exposure can be relatively short spatially, tundra plant communities can be relatively distinct (Bliss and Matveyeva, 1992). The local topography imposes exposure and moisture gradients by influencing snow depth. At the regional scale, the effects of latitude and distance from the ocean are the major gradients affecting plant community structure (Walker et al., 2005).

Snow bed tundra plant species have been identified as potentially at risk if snowfall becomes less and melt is more rapid, leading to longer growing seasons with drier soils (Bjork and Molau, 2007). Increased precipitation during the summer may also affect the species composition of tundra plant communities, especially in the dry polar semi-desert areas. The trajectory of precipitation regimes in the Arctic, however, is difficult to predict (ACIA, 2005) and there have been very few experimental studies altering both winter and summer precipitation in combination with warming. An experimental study in High Arctic tundra plant communities showed that watering with about 13 L every 10 days for three seasons did not affect net primary production (Henry et al., 1986).

Tundra plant communities are showing changes across the biome that are consistent with responses to warming (see also the section on Primary production (biomass) in tundra ecosystems on page 73). In a study of 158 plant communities across 46 sites that were resampled between 1980 and 2010, Elmendorf et al. (2012b) found increases in canopy height and height of most vascular growth forms, increased abundance of shrubs and litter, and decreased cover of bare ground (Figure 64). However, the responses varied greatly among climate zones and depended strongly on moisture and permafrost conditions. Overall, the responses were very similar to a biome-wide analysis of warming experiments (Elmendorf et al., 2012a) and provide plot-level support to the changes in reflectance measured by NDVI (Bhatt et al., 2010).

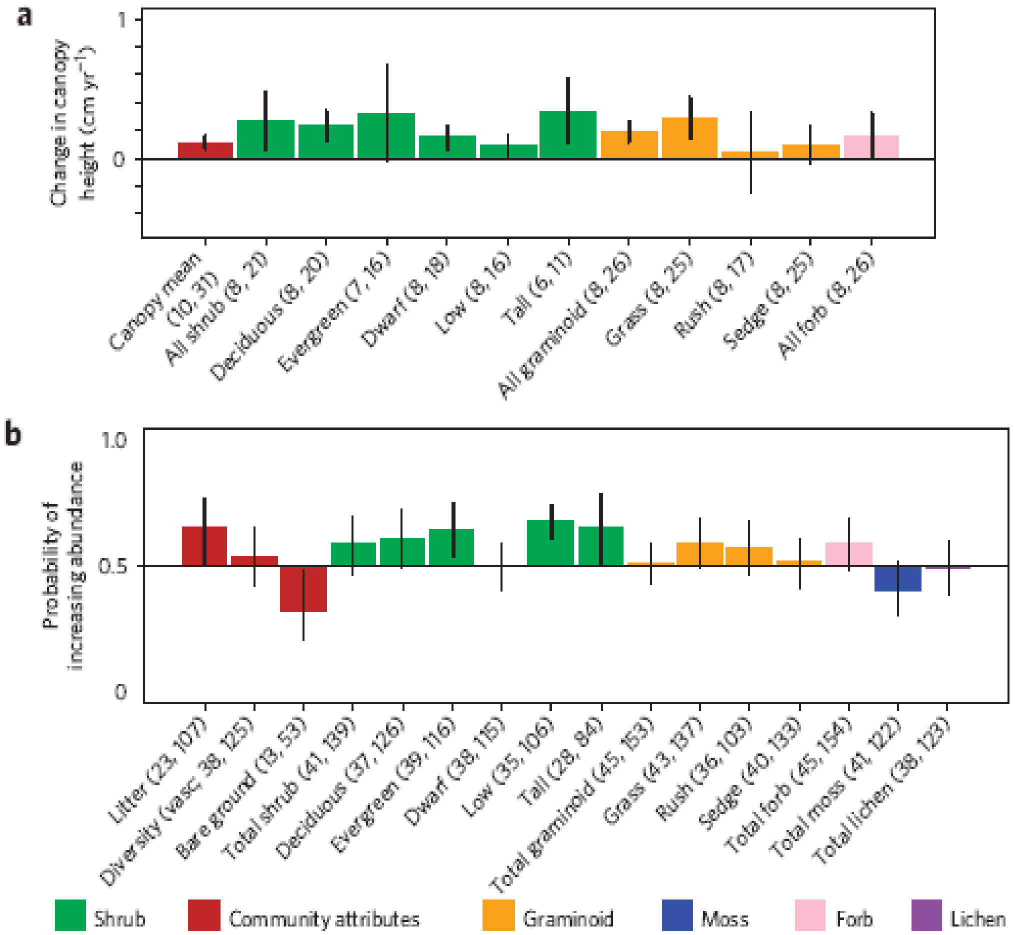


Figure 64. Biome-wide changes in vegetation height and abundance, circumpolar sites.
Height (a) is the mean height of the canopy for each category; abundance (b) is the probability of increase. Error bars show ± SE and are bold when mean change rates were significant (P<0.05). Sample sizes (number of studies, number of locations) are on the x-axis.
Source: Elmendorf et al., 2012b

Major human stressors on ecosystem structure

Climate change

As with the linked trends in ecosystem processes and functions, climate change is the main human-induced stressor on ecosystem structure in the Arctic Ecozone+.

Stressors related to industrial development, settlements, and human activity

Fragmentation and disturbance

Overall, the degree of human-induced fragmentation in the Arctic Ecozone+ is extremely low, as there are few roads and other linear features. Fragmentation, however, is a stressor of concern at the regional level, with the potential to become more widespread with increasing human population and industrial activity in the Arctic.

Disturbance from roads, pipelines, and other linear facilities, and from vehicle and aircraft traffic, has been widely hypothesized to affect wildlife. These hypotheses have been tested experimentally and by behavioural observations in many Arctic and subarctic settings (Slaney, 1975; Harding and Nagy, 1980; Gunn, 1984; James and Stuart-Smith, 2000; Gnieser, 2000; Dyer et al., 2002; Lunn et al., 2004). In general, the studies show modest impacts on animal energetics related to fleeing industrial activities, modest habitat losses from avoidance of areas with industrial facilities and activities, and little or no reluctance of migrating animals (for example, caribou) to cross linear facilities (Nellemann and Cameron, 1998; Wolfe et al., 2000b). In some cases, wild animals are attracted to such facilities. For example, caribou were attracted to the tailings pond and airstrip at Lupin Gold Mine, Contwyoto Lake (Gunn et al., 1998). It is, however, clear that some species, including caribou, grizzly bears, wolverines, and possibly muskoxen, do avoid areas of intensive activity and disturbance, at least for some life history requirements (Harding and Nagy, 1980; Dumond, 2006; Krebs et al., 2007).

A study in the Southern Arctic and Taiga Shield examined the cumulative impacts of human activities and associated infrastructure (mainly related to diamond mine development) on the distribution of caribou, wolves, grizzly bears, and wolverines over an area of 190,000 km^2, 400 km northeast of Yellowknife, from 1995 to 2000 (Johnson et al., 2005). The study was conducted as part of the West Kitikmeot/Slave Study (see Case study on environmental governance: Kitikmeot on page 179). Mines and other major developments had the greatest effect on species occurrence, followed by exploration activities and outfitter camps, but there was not a uniform response, with some carnivores being attracted to these facilities. Modeling techniques were used to assess the reduction in habitat effectiveness (i.e., to what degree the animals selected poorer quality habitats in response to proximity to disturbances). Grizzly bears and wolves showed the strongest negative response, followed by caribou and wolverines. Seasonally, however, the biggest impact was on post-calving caribou: models suggested a 37% reduction in the area of the highest quality habitats and an 84% increase in the area of lowest quality habitats.

Effects of human infrastructure on predator habitat

Human infrastructure (roads and airstrips, often with culverts, telecommunications poles and towers, navigation beacons, and buildings) greatly enhance habitat structure for some species by providing new denning and nesting opportunities (Liebezeit et al., 2009; Reid et al., 2011; Wilson et al., 2013). These new structures generally benefit predators (foxes and raptors) and thereby change the spatial distribution of the foraging areas of these predators. This can result in changes to the reproductive success of various prey species (notably shorebirds), (Liebezeit et al., 2009), and the nesting distribution of competing raptors (Reid et al., 2011). Mitigation measures include minimizing construction of such structures, designing structures to reduce raptor-nesting potential, and minimizing the extent of road and pipeline networks.

Effects of food supplementation on trophic interactions

Food supplementation results when garbage disposal is centralized, creating feeding opportunities in a few locations for scavengers such as foxes and gulls. The scavengers benefit, with potential spillover effects on other species. The expansion of the red fox into Arctic Canada coincided with centralized community development for Inuit in the 1930s through 1960s, and was likely related to this enhancement of food resources at certain limited sites (Marsh, 1938; Macpherson, 1964). Red fox outcompete Arctic fox, so the red fox expansion can result in loss of Arctic foxes from ecosystems. The two species co-exist on the Yukon North Slope, where there is currently no food supplementation (Gallant et al., 2012).

Case study: Cruise ship tourism, potential emerging stressor on ecosystem structure

Historically, ice conditions have prohibited most commercial shipping in the Canadian Arctic, and it was not until 1984 that cruise tourism got underway with the first cruise ship voyage through the Northwest Passage, by the MS Explorer (Jones, 1999). Increasing reduction of summer sea ice (see section on Sea ice on page 34) makes most of the Canadian Arctic increasingly accessible to cruise ship tourism (Stewart et al., 2007; Stewart et al., 2010).

From 1984 to 1991, activity was sporadic. From 1992 on, both number of cruises and diversity of cruise routes increased steadily. The number of planned cruises in Arctic Canada doubled between 2005 and 2006 (from 11 to 22) and continued to increase from 2007 to 2010 at a rate of approximately 10% per year (Dawson, 2012). Cruise ship activity has now begun to level off, likely due to the 2009 economic recession, limited number of appropriate ships available, and the fluctuating tourism market for Arctic cruises (Dawson, 2012). Lack of infrastructure (such as port facilities and ground transport), as well as few tourist sites and community activities for groups of visitors, also impose limitations on the growth of the cruise ship industry.

During the 2007 season, 23 separate cruises, run by six different companies, brought approximately 2,110 visitors to the Canadian Arctic (Stewart et al., 2010). Many of these ships made planned, and occasionally unplanned, stops in national parks such as Auyuittuq National Park and Sirmilik National Park on Baffin Island, and Quttinirpaaq National Park on Ellesmere Island, as well as to nearby locations (Table 6) (Stewart et al., 2008).

Table 6. Cruise ship visits to Arctic National Parks and nearby destinations.

Location	2006	2007	2008
Pond Inlet	12	9	9
Tanquary Fjord	1	1	1
Ellesmere Island	1	1	1
Pangnirtung	4	7	6
Qikiqtarjuaq	2	2	3
Auyuittuq National Park	4	3	2

Source: Stewart et al., 2008

Most cruising expeditions encourage tourists to hike in the parks. Without regulated access and trails, large number of tourists hiking in parks and in other accessible coastal sites could have enduring negative impacts on the surrounding ecosystems. During investigations of abandoned settlements on three islands in the eastern Canadian Arctic Archipelago, Forbes (1996) found that flora of the Arctic has a limited number of species able to respond to disturbance and anthropogenically disturbed patches may be extremely persistent. Even relatively low-intensity, small-scale disturbances have immediate and persistent effects on Arctic vegetation and soils. Where slope is minimal, such disturbances are capable of expanding over large areas in as short a time as four years. The result is an artificial mosaic of patches of highly variable quality and quantity, compromising feeding and nesting habitats for vertebrate herbivores (Forbes, 2001). Other potential impacts include wildlife disturbance, especially of nesting seabirds (Marquez and Eagles, 2007), increased risk of introduction of invasive plants, and large local increases in human waste that, if not disposed of properly, could result in habitat degradation.

Ecosystem composition

Composition refers to the species and species groups that make up the Arctic Ecozone^{+}.

Sources of knowledge about past distribution and abundance of Arctic species include Aboriginal Traditional Knowledge and records and narratives written by early explorers, who often recorded information from their Dene and Inuit guides along with their own observations and survey results. American and Canadian governments began sponsoring biological surveys in the late 19th century. These surveys brought to the attention of the world the iconic wildlife species of the Canadian Arctic: polar bears (*Ursus maritimus*), barren-ground grizzly bears (*Ursus arctos*), vast herds of migratory barren-ground caribou (*Rangifer tarandus groenlandicus*), small, nearly white Peary caribou (*Rangifer tarandus pearyi*) of the Northern Arctic, Arctic fox (*Vulpes lagopus*), Arctic hares (*Lepus arcticus*), lemmings (Arvicolinae) and muskox (*Ovibos moschatus*); marine mammals such ringed (*Pusa hispida*) and bearded (*Erignathus barbatus*) seals, bowhead (*Balaena mysticetus*) and beluga (*Delphinapterus leucas*) whales and narwhals (*Monodon monoceros*); walruses (*Odobenus rosmarus*); resident birds such as the snowy owl (*Bubo scandiacus*) and gyrfalcon (*Falco rusticolis*), and the Arctic breeding grounds of migratory birds such as the now presumably extinct Eskimo curlew (*Numenius borealis*).

Species diversity

Species diversity is relatively low in the Arctic Ecozone⁺, declining on a gradient from south to north and west to east. The northwestern region of the ecozone⁺ is part of the Beringia region and its diversity is enhanced by endemic or globally rare species that survived the last glaciations (Cannings et al., 2013).

Diversity of vascular plants in Canadian tundra ecosystems declines with decreasing temperature along the latitudinal gradient (Figure 65) (Rannie, 1986). While the number of bryophyte and lichen species also declines with latitude, the decrease in richness is not as great as for vascular species. Vertebrate species richness also declines from south to north, as shown in Figure 66.

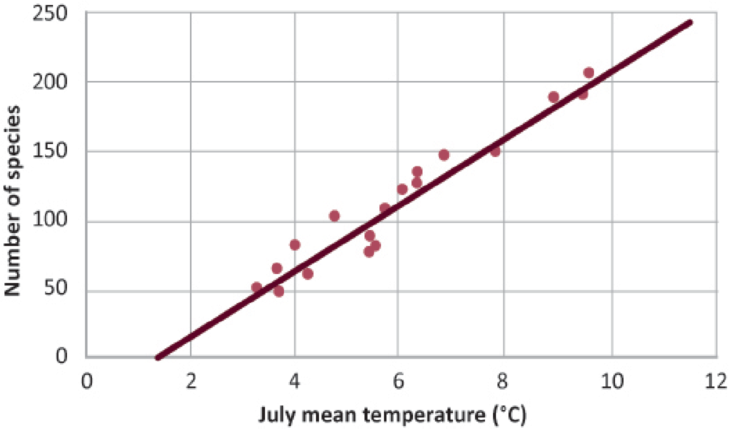


Figure 65. Relationship between vascular plant species richness and mean July temperature in the Canadian Arctic.
Source: Rannie, 1986

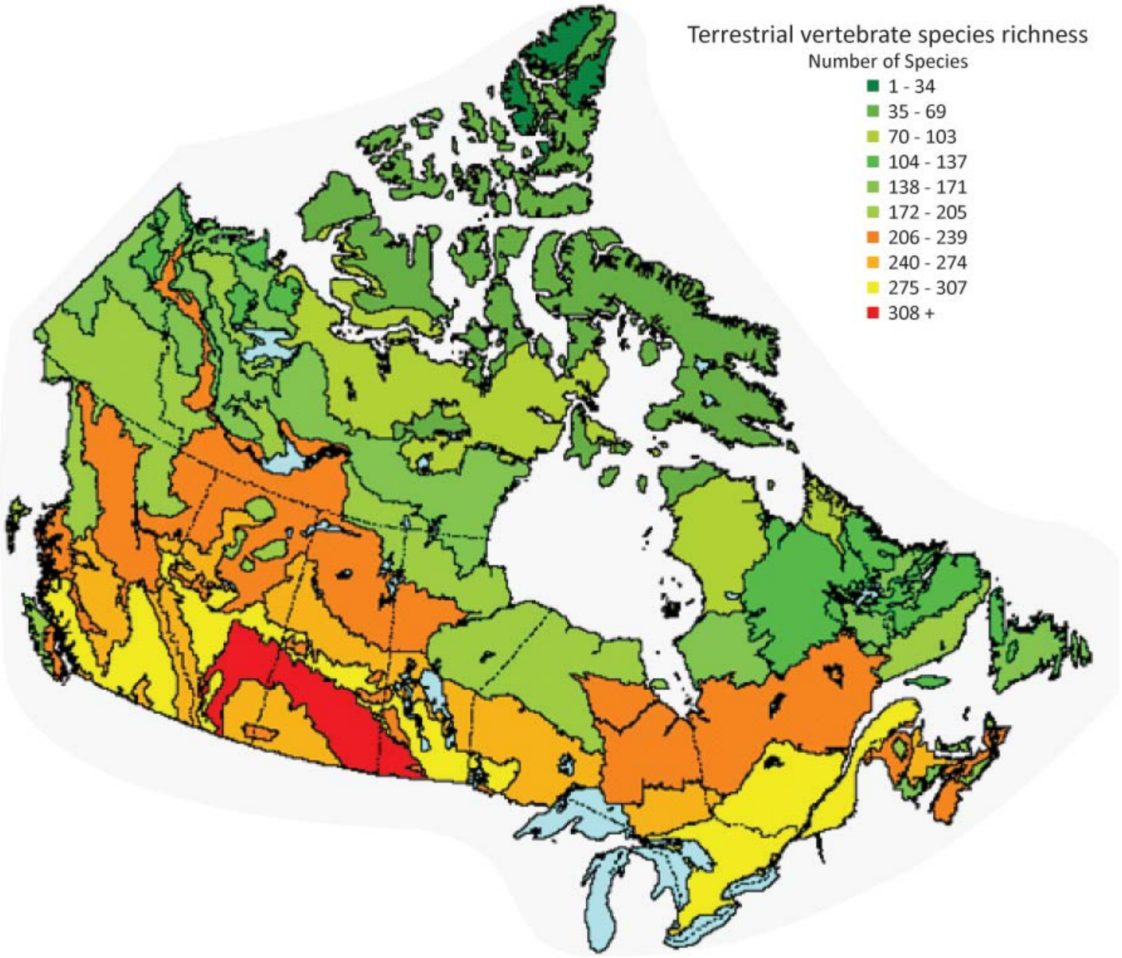


Figure 66. Terrestrial vertebrate species richness in Canada.
Source: GeoInsight Corporation, 1999

Arctic ecosystems have relatively simple food webs relative to more southern ecosystems (see section on Community and population dynamics on page 54). More complex ecosystems have multiple energy flow pathways and more species that occupy similar niches. The low species diversity and generally simple Arctic ecosystems may limit the ability of these ecosystems to resist perturbation in the first place, and to recover when damaged. However, the link between food web complexity and vulnerability to changes is not well-understood (see section on Community and population dynamics).

Nunavut, with a land area of 1.9 million km^2, almost all of which is within the Arctic Ecozone$^+$, has 38 species of terrestrial mammals, 151 species of birds breeding in the territory, 20 species of freshwater and anadromous fishes, 8 amphibian and 1 reptile species, 47 butterfly species, and 626 species of vascular plants (Department of Environment, 2013a). By contrast, British Columbia, with half the land area, has 47 regularly occurring mammal species, 528 species of birds breeding in the province, 120 of freshwater and anadromous fishes, 22 amphibians and 22 reptile species, 275 of butterfly species, and 3,097 species of vascular plants (BC Conservation Data Centre, 2007). This comparison illustrates the relative simplicity, in terms of species richness, of Arctic ecosystems.

Trends in species of conservation concern

Species assessment

The biological status of species refers to their risk of extinction as determined through standard criteria and assessed at the global, national, or provincial/territorial level. Subspecies and populations can be assessed and included in the term "species". Status at the global level is assessed by the International Union for the Conservation of Nature (IUCN). Species considered at risk in the world or of conservation concern for this overview are those assessed at a category of Near Threatened, Vulnerable, Endangered, or Critically Endangered.

The biological status of species at risk in Canada can be assessed as "Endangered" (facing imminent extirpation or extinction), "Threatened" (likely to become endangered if nothing is done to reverse threats), and "Special Concern" (at risk of becoming threatened or endangered). Species that are assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as being at risk, but are not yet legally protected under Canada's *Species at Risk Act* (SARA), are undergoing consultation for listing (Table 7). Although the Arctic Ecozone+, compared with other ecozones+, has the lowest number of species listed as at risk under SARA, several iconic or keystone species and species important to Inuit culture are under review.

Table 7. Species with a Canadian distribution restricted to the Arctic Ecozone+ and/or Arctic marine ecozones+ that are listed under Canada's Species at Risk Act (SARA) as being of conservation concern or are assessed as at risk by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

Species	Subspecies or Population	COSEWIC Status	SARA Status	SARA Schedule
Plants				
Hairy braya (*Braya pilosa*)		Endangered		
Porsild's bryum (*Haplodontium macrocarpum*)		Threatened	Threatened	Schedule 1
Spiked saxifrage (*Micranthes spicata*)		Threatened		
Fishes				
Atlantic cod (*Gadus morhua*)	Arctic Lakes population	Special Concern	No Status	No Schedule
Atlantic wolffish (*Anarhichas lupus)*		Special Concern	Special Concern	Schedule 1
Bering wolffish (*Anarhichas orientalis*)		Data Deficient	Special Concern	Schedule 3
Blackline prickleback (*Acantholumpenus mackayi*)		Data Deficient	Special Concern	Schedule 3
Bering cisco (*Coregonus laurettae*)		Special Concern	No Status	No Schedule
Bull trout (*Salvelinus confluentus*)	Western Arctic populations	Special Concern		
Dolly varden (*Salvelinus malma malma*)	Western Arctic population	Special Concern	No Status	No Schedule
Fourhorn sculpin (*Myoxocephalus quadricornis*)	Freshwater form	Data Deficient	Special Concern	Schedule 3
Northern wolffish (*Anarhichas denticulatus*)		Threatened	Threatened	Schedule 1
Roundnose grenadier (*Coryphaenoides rupestris*)		Endangered	No Status	No Schedule

Species	**Subspecies or Population**	**COSEWIC Status**	**SARA Status**	**SARA Schedule**
Spotted wolffish (*Anarhichas minor*)		Threatened	Threatened	Schedule 1
Thorny skate (*Amblyraja radiata*)		Special Concern	No Status	No Schedule
Birds				
Buff-breasted sandpiper (*Tryngites subruficollis*)		Special Concern	No Status	No Schedule
Eskimo curlew (*Numenius borealis*)		Endangered	Endangered	Schedule 1
Harlequin duck (*Histrionicus histrionicus*)	Eastern population	Special Concern	Special Concern	Schedule 1
Ivory gull (*Pagophila eburnea*)		Endangered	Endangered	Schedule 1
Red knot (*Calidris canutus*)	*rufa* subspecies (*C. canutus rufa*)	Endangered	Endangered	Schedule 1
	roselaari type (*C. canutus roselaari* type)	Threatened	Threatened	Schedule 1
	islandica subspecies (*C. canutus islandica*)	Special Concern	Special Concern	Schedule 1
Ross's gull (*Rhodostethia rosea*)		Threatened	Threatened	Schedule 1
Short-eared owl (*Asio flammeus*)		Special Concern	Special Concern	Schedule 1
Tundra peregrine falcon *(Falco peregrines tundrius*)		Non-active	Special Concern	Schedule 3
Land Mammals				
Barren-ground caribou (*Rangifer tarandus groenlandicus*)	Dolphin and Union population	Special Concern	Special Concern	Schedule 1
Grizzly bear (*Ursus arctos*)	Western population	Special Concern	No Status	No Schedule
Peary caribou (*Rangifer tarandus pearyi*)		Endangered	Endangered	Schedule 1
Polar bear (*Ursus maritimus*)		Special Concern	Special Concern	Schedule 1
Wolverine (*Gulo gulo*)	Western population	Special Concern	No Status	No Schedule
Marine Mammals				
Atlantic walrus (*Odobenus rosmarus rosmarus*)		Special Concern	No Status	No Schedule
Beluga whale (*Delphinapterus leucas*)	Southeast Baffin Island-Cumberland Sound population	Non-active	Endangered	Schedule 2
	Eastern Hudson Bay population	Endangered	No Status	No Schedule
	Ungava Bay population	Endangered	No Status	No Schedule
	Cumberland Sound population	Threatened	No Status	No Schedule
	Eastern High Arctic–Baffin Bay population	Special Concern	No Status	No Schedule
	Western Hudson Bay population	Special Concern	No Status	No Schedule
Bowhead whale (*Balaena mysticetus*)	Bering–Chukchi–Beaufort population	Special Concern	Special Concern	Schedule 1
	Eastern Arctic population	Non-active	Endangered	Schedule 2

Species	Subspecies or Population	COSEWIC Status	SARA Status	SARA Schedule
	Eastern Canada–West Greenland population	Special Concern	No Status	No Schedule
Grey whale (*Eschrichtius robustus*)	Eastern North Pacific population	Special Concern	Special Concern	Schedule 1
Humpback whale (*Megaptera novaengliae*)	Western North Atlantic population	Not at Risk	Special Concern	Schedule 3
Killer whale (*Orcinus orca*)	Northwest Atlantic/ Eastern Arctic population	Special Concern	No Status	No Schedule
Narwhal (*Monodon monoceros*)		Special Concern	No Status	No Schedule

Those species on SARA Schedules 2 and 3—or as yet on no schedule—are at some stage of assessment for consideration for listing on Schedule 1. Note that COSEWIC assessment and SARA listing are two separate processes. COSEWIC assessment as Endangered does not automatically mean SARA listing.

Source: SARA Registry, current to June 2013 (Government of Canada, 2013)

Below are discussions of status and trends of selected species of conservation concern.

Peary caribou

This section is based on the ESTR technical thematic report *Northern caribou population trends in Canada* (Gunn et al., 2011c) and includes updates to that report and expanded discussion based on recent assessments and research results. See also the Main threats to caribou section (page 167). The classification of caribou used in this report follows the current *Species at Risk Act* (SARA) classification system. In 2011, COSEWIC adopted 12 designatable units for caribou in Canada that will be used in caribou assessments and subsequent listing decisions under SARA beginning in 2014 (COSEWIC, 2011).

Over the last 50 years the number of Peary caribou have declined from about 44,000 to about 11,000 to 12,000 caribou (Species at Risk Committee, 2012a) and two geographic populations appear to have disappeared. The rate of decline has varied over time and between the different geographic populations, with both reversals of some declines and absence of recovery for other populations. Survey intervals are irregular: only two of the six geographic populations (Banks and Bathurst Island complex) have been surveyed at regular intervals. Overall trends differ between the northern High Arctic islands and the larger, southern (mid-Arctic) islands and Boothia Peninsula. Although these larger mid-Arctic islands had a relatively high abundance of Peary caribou until the 1990s, declines occurred. Populations on the Boothia Peninsula and on Prince of Wales and Somerset islands had almost disappeared by the 1990s. There is no evidence for recovery over a 20-year period on Banks and northwest Victoria islands, despite severely restricted harvesting since the early 1990s (Species at Risk Committee, 2012a).

Geographic populations of Peary caribou are identified based on knowledge of movements, seasonal distribution, and genetics (Jenkins et al., 2011; Species at Risk Committee, 2012a). The Arctic islands are linked for much of the year by sea ice, and Peary caribou seasonally migrate across the ice. Sea ice coverage, however, is minimal in fall, which likely influences the rut distribution and thus the maintenance of island geographic populations. Jenkins et al. (2011) described seven geographic populations on the more northern islands: Melville and Prince Patrick, the Prime Minister group, the Bathurst Island group, Devon Island, Axel Heiberg, Ellesmere, and the Ellef group. Two of four geographic populations on the southern islands (Prince of Wales and Somerset; Boothia Peninsula) have essentially disappeared since the 1990s, leaving the two southern island populations of Banks and northwest Victoria islands.

On the High Arctic islands, weather is an overwhelming influence as periodic severe winters trigger large-scale mortality and reduction in productivity (Miller and Gunn, 2003; Harding, 2004). Although the signals of climate warming are strong in the High Arctic (Zhang et al., 2011), relating those trends in weather to changes in Peary caribou abundance is uncertain, partly because of high annual variability in climate and infrequent monitoring for most Peary caribou. The other reason is that harvest and predation also affect Peary caribou abundance. Muskox trends in abundance tend to differ from Peary caribou, although this is area-specific. Muskox increases relative to Peary caribou decreases have raised the question of competition. The role of intra- or inter-specific competition for forage is conjectural, as diet and habitat selection differ considerably between caribou and muskoxen (Gunn and Dragon, 2002). On Banks Island, however, there was overlap in some plants eaten by the two species, with, for example, willow being eaten by both Peary caribou and muskoxen (Larter and Nagy, 2004). This suggests that a competitive relationship could occur between the two species. Less emphasis has been placed on determining whether the increasing muskox abundance supported increased wolf numbers which, in turn, could increase predation rates on Peary caribou (Gunn and Dragon, 2002). Even less attention has been given to studying the relationship between caribou and muskoxen and their parasites. Hughes et al. (2009), however, discussed levels of intestinal nematode worms and warble flies in muskoxen and caribou for the range of the Dolphin and Union caribou.

There are limitations to describing the trends for Peary caribou due to the infrequency of surveys and the relatively brief period over which surveys have occurred. It is uncertain whether the documented high numbers of Peary caribou in the early 1960s followed by a decline and prolonged low numbers are: 1) part of regular fluctuations; 2) a period of relative stability within an unusually prolonged decline; or 3) atypically high peak numbers. Possibly Peary caribou regularly fluctuate in abundance, driven by a relationship between amounts of forage and caribou abundance (Tews et al., 2007a and 2007b). Alternatively, Peary caribou are in a "non-equilibrium grazing system" where sporadic, unpredictable weather events affect vital rates and population trends (Caughley and Gunn, 1993; Behinke, 2000).

Trends in abundance

Melville and Prince Patrick islands: The infrequent surveys documented steep declines between 1961 (the first range-wide aerial survey) and 1997. On Prince Patrick Island, there was a 95% decline (from 1,797 to 84 year-plus old caribou). On Melville Island, Peary caribou declined by 92% (from 10,366 to 787 year-plus old caribou) (Tener, 1963; Gunn and Dragon, 2002). However, between 1997 and 2012, trends were reversed and a strong recovery was apparent for Melville, Prince Patrick, Byam Martin, Eglinton, and Emerald islands (Davison and

Williams, 2012). The 2012 estimate was about 6,000 Peary caribou which, while not a complete recovery to the abundance recorded in 1961 (16,000), is the highest abundance recorded since then. See Figures 67a and 67b. Lack of sea ice prevented the aerial survey of Mackenzie King, Brock, and Borden islands in 2012.

***Prince of Wales and Somerset islands*:** Peary caribou seasonally crossed the sea ice between the islands in this group and some caribou also wintered on the Boothia Peninsula. Between 1974 and 1980, caribou numbers were stable in the range 4,000 to 6,000, one of the largest Peary caribou populations in the 1970s and 1980s (Gunn et al., 2006). There was a 15-year hiatus in surveys until 1995, when only a few caribou were found (Gunn and Dragon, 1998). In 2004, no caribou were seen during an aerial survey of the islands (McRae et al., 2010). See Figure 67c.

***Bathurst Island (and its satellite islands)*:** Between 1961 and 1974, Peary caribou numbers declined by an order of magnitude (Miller, 1991a). Between 1974 and 1994, numbers recovered to the 1961 level (Miller, 1991a). An abrupt decline followed and, by 1997, fewer than 100 caribou remained (Gunn and Dragon, 2002). A survey in 2001 revealed the trend was for a recovering population (McRae et al., 2010). The Bathurst Island complex was resurveyed in May 2013, with preliminary estimates indicating that the herd has tripled in size since 2001 (Department of Environment, 2013a). The final population estimate based on this survey was not available at the time of report completion and is not included in the graph, Figure 67d.

***Banks Island*:** Peary caribou on Banks Island were one of the larger populations as they peaked at about 12,000 in the early 1970s (Gunn et al., 2000b) and remained relatively stable until 1982 (Nagy et al., 2009d). Numbers declined to about 1,000 caribou by 1992 (Nagy et al., 2009a) and an initial small recovery by 2001 was likely lost during an icing storm early winter 2003 (Nagy and Gunn, 2006). A 2010 survey led to an estimate of 1,097 ± 343 [95% confidence interval (CI)] non-calf caribou, which confirmed the persistence of low numbers on Banks Island (Davison et al., 2013). See Figure 67e.

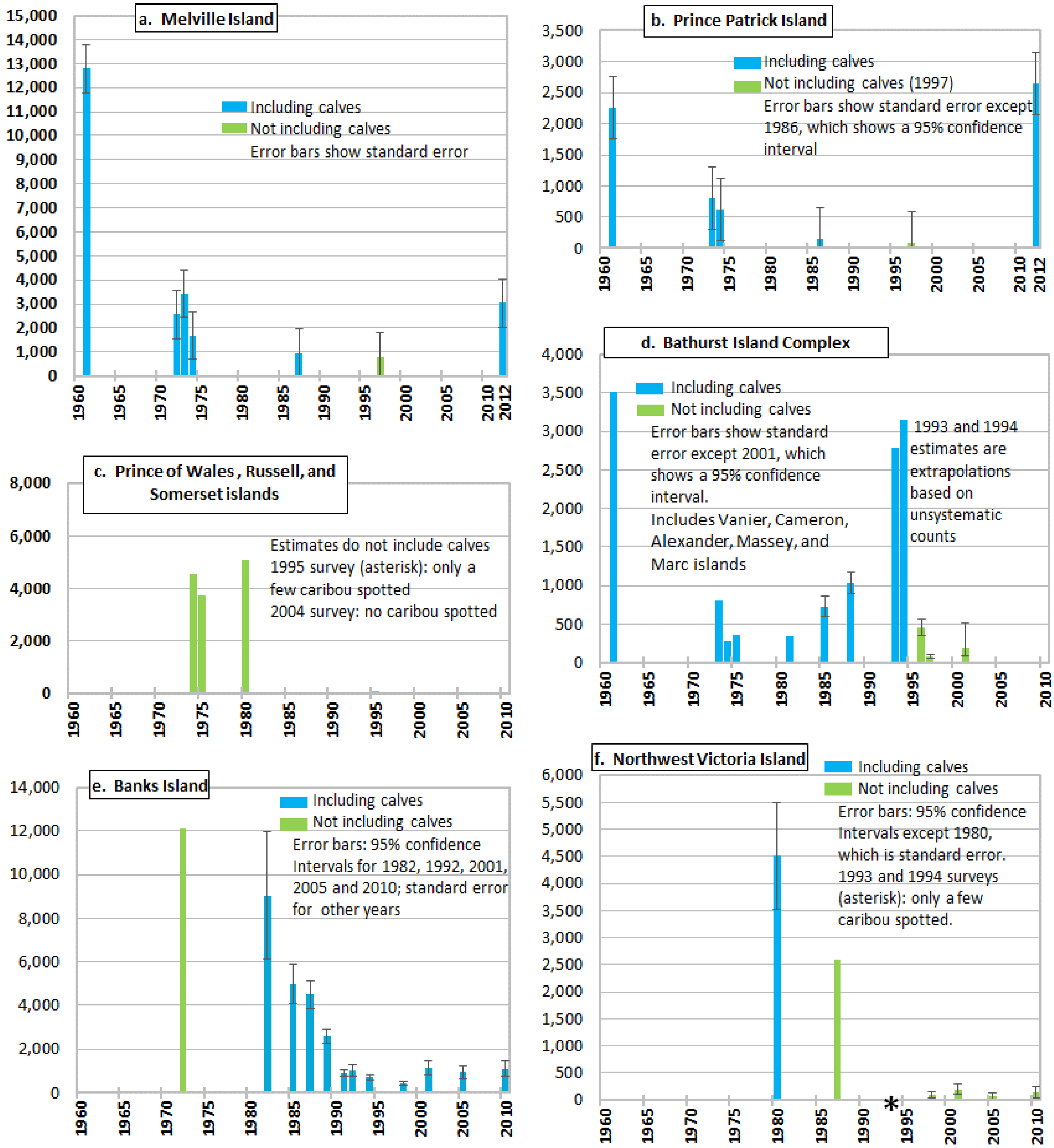
***Northwest Victoria Island*:** Trends in Peary caribou on northwest Victoria Island are less clear than on Banks Island as surveys have been less frequent. Numbers were high, about 2,600 in 1987 (Gunn et al., 2000b), and declined during the late 1980s until, in 1993, only a few caribou were seen during an aerial survey (Gunn, 2005). The population then slowly recovered, based on estimates of 95 ± 60 (95% CI) in 1998 (Nagy et al., 2009b) and 204 ± 103 in 2001 (Nagy et al., 2009c). However, in 2005, the estimate was 66 ± 61 non-calf caribou, which suggested that some recovery was lost during two winters (2002/03 and 2003/04) with icing events (Nagy and Gunn, 2006). A subsequent survey in 2010 returned an estimate of 150 ± 104 non-calf caribou, which confirmed the persistence of low numbers (Davison et al., In Prep.). See Figure 67f.

Boothia Peninsula: Peary caribou increased throughout the 1970s and early 1980s. A survey in 2006 in this region showed a decline from a 1985 estimate (Gunn and Dragon, 1998). Trends are difficult to distinguish as satellite telemetry has shown that both barren-ground caribou and Peary caribou calve and summer on the peninsula (Gunn et al., 2000a) and only three aerial surveys were flown between 1985 and 2006. During the most recent aerial survey in June 2006, only one caribou was identified as a Peary caribou and sightings of all caribou were few on the northern part of the peninsula which was used by Peary caribou for calving and summer in the mid-1980s and 1990s (Jenkins et al., 2011).

Eastern Queen Elizabeth Islands (Ellef Ringnes, Amund Ringnes, Devon, Ellesmere, Axel Heiberg Islands, Cornwall, King Christian, Graham): There is relatively little information available to assess trends as there have only been two extensive aerial surveys over a 50-year period. The islands were surveyed in 1961, although coverage was so low that the resulting figure of about 1,500 caribou was an approximation (Tener, 1963). Miller et al. (2005) re-analysed the 1961 data using updated areas for the islands and increased the estimate to 2,887± 642. The next extensive survey was between 2005 and 2008, when the Nunavut Department of Environment estimated 4,000 caribou 10 months and older based on aerial surveys (Jenkins et al., 2011):

- Ellesmere Island (including Graham Island), surveyed partly in 2005 and partly in 2006: estimate of 1,021 caribou;
- Axel Heiberg Islands, surveyed 2007: 2,291 (95% CI of 1,636 to 3,208) caribou
- Amund Ringnes, Ellef Ringnes, King Christian, Cornwall, and Meighen Islands, surveyed 2007: total of 282 (95% CI of 157 to 505) caribou
- Lougheed Island, surveyed 2007: 372 (95% CI of 205 to 672) caribou
- Devon Island, surveyed 2008: 17 caribou counted in an extensive survey

The information does not support any overall trend. The reworked 1961 estimate and the 2005–2008 estimate overlap in their confidence limits, but, with only two data points separated by almost 50 years, there is not sufficient information to interpret the trend as stable as the extent and nature of local declines and recoveries are not known.


a. Melville Island

Error bars show standard error

b. Prince Patrick Island

Error bars show standard error except 1986, which shows a 95% confidence interval

c. Prince of Wales, Russell, and Somerset islands

Estimates do not include calves
1995 survey (asterisk): only a few caribou spotted
2004 survey: no caribou spotted

d. Bathurst Island Complex

Error bars show standard error except 2001, which shows a 95% confidence interval.
Includes Vanier, Cameron, Alexander, Massey, and Marc islands

1993 and 1994 estimates are extrapolations based on unsystematic counts

e. Banks Island

Error bars: 95% confidence Intervals for 1982, 1992, 2001, 2005 and 2010; standard error for other years

f. Northwest Victoria Island

Error bars: 95% confidence Intervals except 1980, which is standard error.
1993 and 1994 surveys (asterisk): only a few caribou spotted.

Figure 67. Peary Caribou population estimates.
Source: Gunn et al., 2011c; Species at Risk Committee, 2012a; Jenkins et al, 2011. See also references in text.

Trends in distribution

Peary caribou only occur in Canada (except occasional sightings on the northwest Greenland coast) and are restricted to the High Arctic (Queen Elizabeth Islands -- 404 730 km^2) and the mid-Arctic islands -- 129 510 km^2, as well as the northern extension of the mainland (Boothia Peninsula -- 26 000 km^2) (Miller, 1991b) (Figure 68).

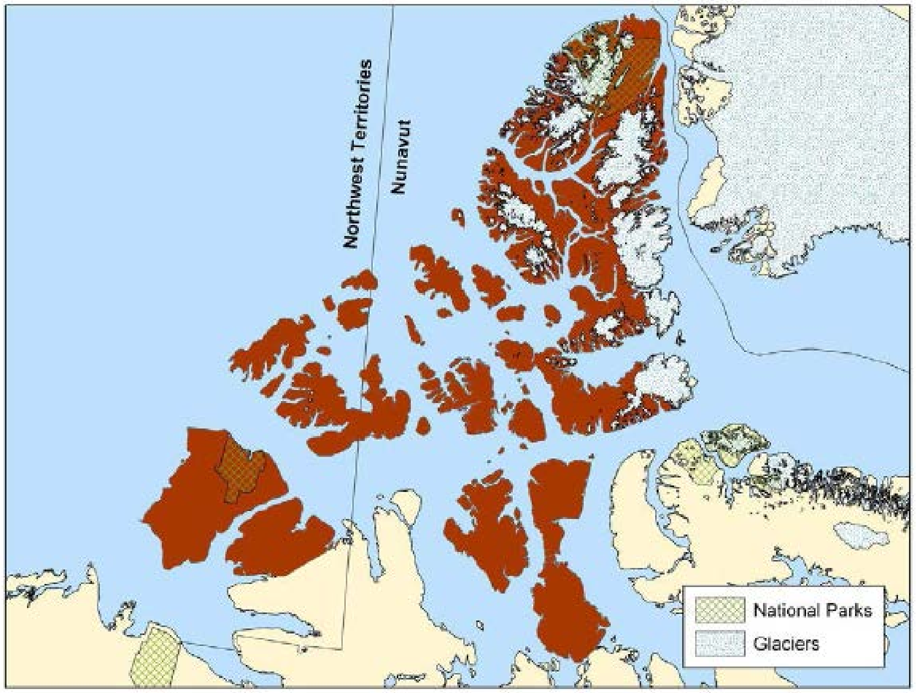


Figure 68. Global distribution of Peary caribou.
Source: Species at Risk Committee , 2012a

Much of the eastern Queen Elizabeth Islands is mountainous with ice fields and glaciers. Areas above 750 m above ground level, including permanent snow and ice fields, account for 20 to 43% of Ellesmere, Axel Heiberg, and Devon islands. Caribou densities are much lower and so, although the eastern Queen Elizabeth Islands are 78% of High Arctic landmass, in 1961 they held only 10% of the Peary caribou (Miller et al., 2005). With the higher rate of decline on the western islands, the proportionate distribution has changed and, based on the most recent surveys, about 40% of High Arctic Peary caribou are on the eastern Queen Elizabeth Islands.

Knowledge of trends in distribution is largely based on sightings during systematic aerial surveys, which vary in timing from island to island. On a few islands (for example, northwest Victoria Island and Banks Island), tracking satellite-collared caribou has increased understanding of seasonal and annual distribution (Gunn and Fournier, 2000b; Poole et al., 2010; Gunn et al., 2012). Caribou shift their distribution during unfavourable winters, as, for example, has been recorded on Bathurst Island (F.L. Miller, unpublished data in Gunn et al., 2012) (Figure 69).

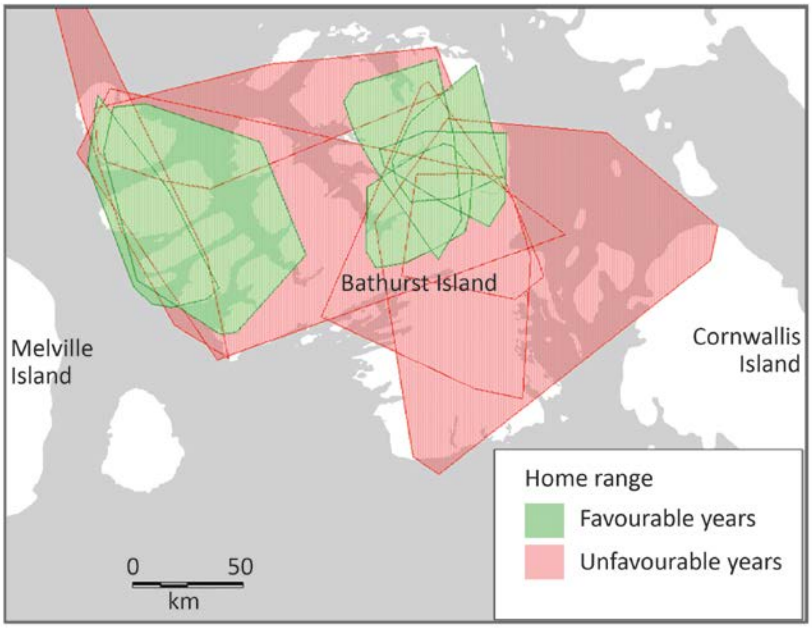


Figure 69. Home ranges of satellite-collared Peary caribou relative to favourable and unfavourable winters, Bathurst Island Complex, NU, 1993/94 and 1994-1997.
Three cow caribou were tracked for two years with favourable winters and two years with unfavourable winters. Each polygon encloses all tracked locations for one caribou for one year (the minimum home range for the individual for that year). The home ranges for the three females tracked were 5.5 times smaller during the favourable years than during the years with unusually deep snow.
Source: adapted from Gunn et al., 2012, based on Miller, 2013 and Miller and Barry, 2002

As abundance declines, distribution becomes more restricted with, for example, changes in the use of the smaller islands. In the western Queen Elizabeth Islands in 1997, Peary caribou were not seen on three islands (Brock, Eglinton, and Emerald islands) during aerial surveys (Gunn and Dragon, 2002). However, Peary caribou were consistently seen on those islands during 1961, 1972–1974, and 1987–1988 aerial surveys, and were seen there again in 2012 (Davison and Williams, 2012). Any trends in distribution within the larger islands are difficult to determine due to the variability in timing of surveys (Species at Risk Committee, 2012a).

Gaps in information due to the sporadic surveys, as well as the scale of annual variation, impede describing trends in distribution. This is especially true for the eastern Queen Elizabeth Islands, as the frequency of surveys is so low (Jenkins et al., 2011). Overall, the distribution of Peary caribou was reduced by 15% between 1980 and 2006, taking into account the decline of two populations to possibly only a few individuals (on Prince of Wales and Somerset islands and on the Boothia Peninsula).

Dolphin and Union Caribou Population

This section is based partly on *Northern caribou population trends in Canada* (Gunn *et al*., 2011c) and includes updates to trend data presented in that report.

Status and trends

Historical information and Inuit hunter reports indicate that there may have been as many as 100,000 caribou on Victoria Island in the early 1800s (Manning, 1960). By the early 1920s, numbers declined and migration across Dolphin and Union Strait halted. The causes are possibly a combination of icing storms and the introduction of rifles. The recovery was slow and caribou were rare until the 1970s. By the 1990s, numbers were increasing (Figure 70). In October 1997 and 2007, surveys of caribou staging along the south coast of Victoria Island led to estimates of 27,948 ± 3,367 [standard error (SE)] and 21,753 ± 2,343 (SE) caribou, respectively (uncorrected to account for caribou assumed to be outside of the census zone) (Nishi and Gunn, 2004; Dumond, 2011, pers. comm.). Those two estimates, together with variable annual pregnancy rates, relatively low cow survival (over the period 1999 to 2006), and high harvest rates, suggest an increased likelihood of a decline (Poole et al., 2010; Dumond, 2011, pers. comm.).

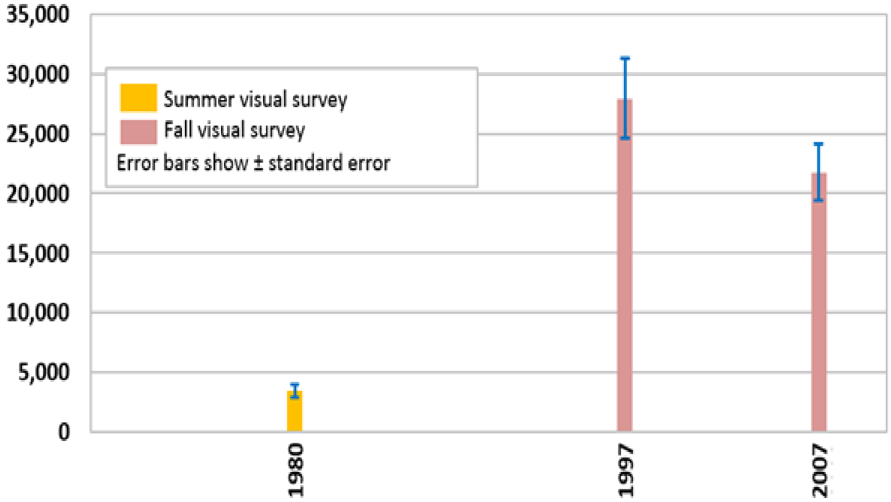


Figure 70. Dolphin and Union Caribou population estimates.
Source: Gunn et al., 2011c. See also references in text.

Calving is dispersed over about half of northern and central Victoria Island and, based on sightings and satellite telemetry, the summer, fall, and winter ranges have increased in size since the early 1980s into the late 1990s. A trend to an increasing size of winter range is due to fall migrations across the newly formed sea ice to the mainland coastal areas, which is a resumption of migrations observed up until the 1920s. The caribou return across the sea ice to Victoria Island in April to May. The date of freeze-up is increasingly delayed: 8 to 10 days later than in 1982, a trend which may lead to changes in the fall migration across the sea ice (Poole et al., 2010).

Polar bear

Approximately 55 to 65% of the world's 20,000 to 25,000 polar bears reside in Canada (COSEWIC, 2008). Figure 71 shows the global distribution and the status and trends of all polar bear subpopulations.

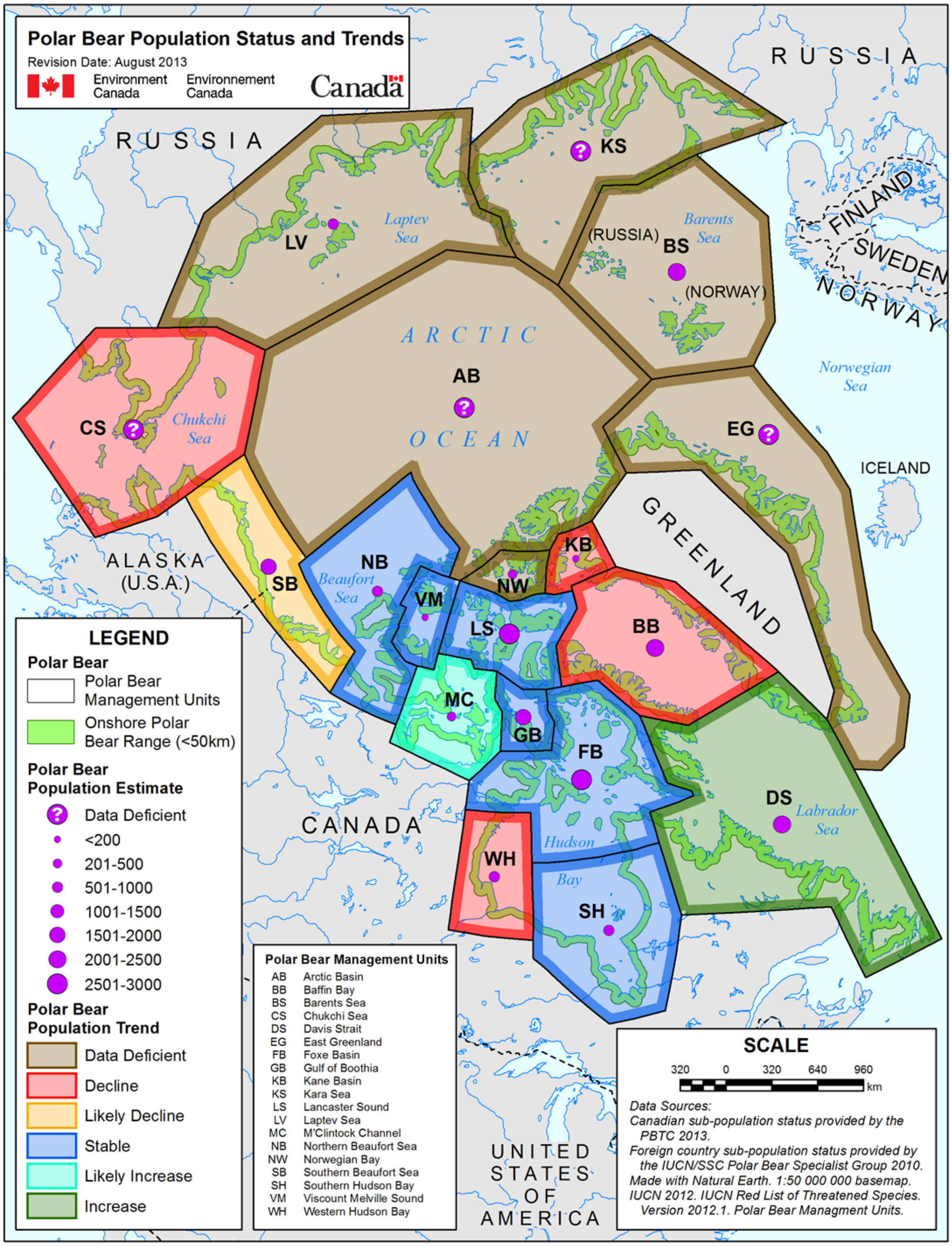


Figure 71. Circumpolar distribution of the polar bear, showing status and trends of subpopulations. Source: Environment Canada, 2013

Analysis of movement data from mark-recapture studies and tracking of adult female bears with satellite radio collars indicated that there are two subpopulations of polar bears in the Beaufort Sea: one that inhabits the west coast of Banks Island and Amundsen Gulf and a second that is resident along the mainland coast from about Baillie Islands in Canada to approximately Icy Cape in Alaska (Stirling, 2002). The central and Eastern Arctic has six population units: Viscount Melville Sound, Lancaster Sound, Norwegian Bay, Kane Basin, Baffin Bay, and Davis Strait (Taylor et al., 2001). The Hudson Bay complex has three subpopulations: Western Hudson Bay, Foxe Basin, and Southern Hudson Bay. Although these subpopulations are sufficiently distinct for management purposes, there is enough movement among them for adequate gene flow (Obbard et al., 2010). Abundance estimate data for Canadian polar bear subpopulations are presented in Table 8.

Table 8. Polar bear abundance estimates for Canadian subpopulations.

Subpopulation	Abundance estimate	95% confidence interval	Survey year
Baffin Bay	2,074	1542–2606	1997
Davis Strait	2,158	1798–2518	2007
Foxe Basin	2,580	2093–3180	2009/10
Gulf of Boothia	1,592	870–2314	2000
Kane Basin	164	94–234	1997
Lancaster Sound	2,541	1759–3323	1998
M'Clintock Channel	284	166–402	2000
Northern Beaufort Sea	980	670–1290	2006
Norwegian Bay	203	115–291	1998
Southern Beaufort Sea	1,526	1210–1842	2006
Southern Hudson Bay	969	688–1365	2011/12
Viscount Melville	161	93–229	1992
Western Hudson Bay	1,000	715–1398	2011

Source: Polar Bear Technical Committee, 2013

In addition to the science-based information, traditional and community knowledge forms part of decision-making about polar bear risk assessment and management. Aboriginal traditional knowledge (ATK) is considered alongside western science by wildlife management boards and government management agencies, especially in making decisions about harvest. Data brought to the management table from these different knowledge sources, reflecting different timeframes and scales as well as differences in experience, methodologies, and perspectives, are not always in agreement, though they are often complementary (e.g., see discussion in Peacock et al., 2011). As well, increasingly, shifts in distribution, timing, and habitat use make trends in overall abundance difficult to detect. Presentation of alternative information and perspectives on polar bear status and trends is beyond the scope of this primarily science-based report, but is covered elsewhere. For example, the recent Northwest Territories polar bear status assessment presents and synthesizes information from traditional and community knowledge and science studies (Species at Risk Committee, 2012b). Under the federal *Species at Risk Act* listing process, Environment Canada is obligated to consider ATK during assessments. Initiatives are underway to improve standardization of collection of ATK and to better integrate ATK and western science in assessments and management decisions.

Climate change and polar bears

Polar bears, adapted to hunting seals from the ice, cannot persist without seasonal sea ice (COSEWIC, 2008) and rapidly declining sea ice poses the most serious threat to polar bears (see section on Sea ice on page 34) (Peacock et al., 2011; Vongraven and Richardson, 2011; Reid et al., 2013). Polar bear subpopulations can be grouped according to ecoregions based on ice habitat (Vongraven et al., 2012). These ecoregions reflect differing types and levels of threat to polar bears from changes in ice timing and extent (Table 9). The divergent sea ice ecoregion has extensive annual ice that forms and then moves to the deep-water regions of the Arctic Ocean, while the convergent sea ice ecoregion has heavy multiyear ice. The convergent ice part of the range of the Norwegian Bay polar bear subpopulation is likely to remain the most viable for polar bear populations as seasonal ice shrinks. The Archipelago ecoregion has historically had a mixture of multiyear and seasonal ice filling the gaps between islands year-round, and polar bears in this ecoregion remain on ice all year. By contrast, in the seasonal ice ecoregion, bears are forced ashore and are deprived of food for periods when the ice melts in the summer.

Table 9. Relative imminent risk to polar bear subpopulations from climate change effects on ice habitat and known levels of toxic contaminants.

Ecoregion	Subpopulation	Current or imminent risk from climate change	Known levels of toxic contaminants
Divergent	Southern Beaufort Sea	High	Low
Convergent	Northern Beaufort Sea	Medium	Medium
Archipelago	Gulf of Boothia	Low	Low
	Kane Basin	?	Low
	Lancaster Sound	Medium	Low
	M’Clintock Channel	Low	Low
	Norwegian Bay	Low	Low
	Viscount Melville Sound	Low	?
Seasonal	Baffin Bay	High	Low
	Davis Strait	High	Low
	Foxe Basin	Medium	Low
	Southern Hudson Bay	High	Low
	Western Hudson Bay	High	Low

Source: Vongraven et al., 2012

Earlier sea-ice break-up around western Hudson Bay has led to poorer physical condition and poorer reproductive performance of polar bears (see next section on Declines in polar bear body condition). Sea-ice break-up has been linked with lower birth rates, lower survival in subadults and senescent bears, and lower body condition (Regehr et al., 2007). Satellite tracking has shown that polar bear movements have changed and productivity has declined in response to sea ice changes in both the Western and the Eastern Arctic (Stirling, 2002; Stirling et al., 2004), as has that of seals, their principal prey (Ferguson et al., 2005). Polar bear denning off Alaska has shifted landward and eastward in response to changing ice conditions (Fischbach et al., 2007).

These and other observations led Derocher et al. (2004) to suggest that polar bears will likely not survive as a species, should the predicted scenarios for total disappearance of summer sea ice in the Arctic come true. A small area of the High Arctic may remain as a refuge for polar bears as their habitat shrinks (Durner et al., 2009; Vongraven et al., 2012; Eamer et al., 2013).

Declines in polar bear body condition

This section is extracted from the Conservation of Arctic Flora and Fauna report *Life linked to ice* (Eamer et al., 2013), with an update for the Western Hudson Bay subpopulation as noted.

In areas where sea ice melts completely in the summer polar bears may be forced onto land. Earlier sea ice break-up in these areas reduces the amount of time bears have for hunting seals on the ice. In some areas where this is occurring, bears are becoming thinner, resulting in decreases in survival and reproduction.

Changes in sea ice over the past two decades have led to significant declines in physical condition of bears in the Western Hudson Bay (Stirling et al., 1999; Stirling and Parkinson, 2006), Southern Hudson Bay (Obbard et al., 2006), and Baffin Bay subpopulations (Rode et al., 2012). Regehr et al. (2007) showed that survival decreased in association with earlier sea ice break-up and that this contributed to a 22% decline in the size of the Western Hudson Bay polar bear subpopulation between 1987 and 2004. A recent aerial survey indicates that, while the population continues to decline, the rate of decline is less than anticipated based on the models used by Regehr et al. (2007). Mark-recapture studies are underway to obtain a new population estimate for the Western Hudson Bay subpopulation (R. Vallender, Environment Canada, pers. comm., 2013). Reduced survivorship in relation to sea ice conditions has also been demonstrated in the Southern Beaufort Sea polar bear population (Regehr et al., 2010).

For the western Hudson Bay population, the body condition of bears measured during the ice-free period declined from 1980 to 2007, as did the average weight of female polar bears in the fall (Figure 72). The female bears weighed were suspected to be pregnant.

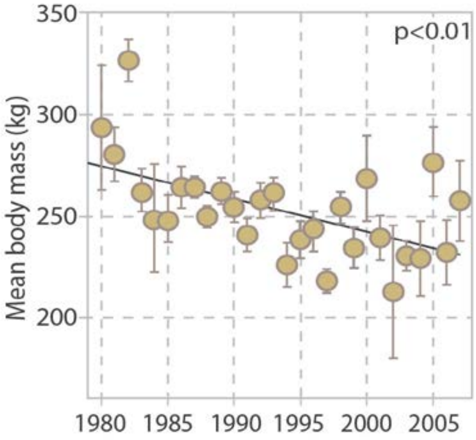
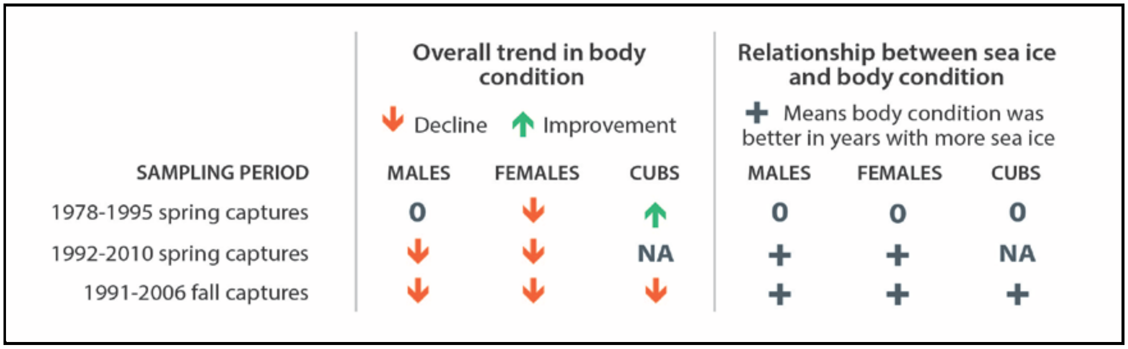


Figure 72. Declining fall weights of female polar bears, western Hudson Bay, 1980-2007. Body weights were estimated for females on their own, and thus likely to be pregnant. Source: Eamer et al., 2013, based on Stirling and Derocher, 2012

For the Baffin Bay population, the decline in body condition since the early 1990s is associated with deteriorating ice conditions (Rode et al., 2012). Polar bears were in significantly worse condition in years with less summer sea ice cover, starting in the 1990s when ice in these regions began its sharp decline (Table 10).

Table 10. Trends in body condition for the Baffin Bay polar bear subpopulation and relationship with sea ice conditions, 1977-2010.



	Overall trend in body condition ↓ Decline ↑ Improvement			Relationship between sea ice and body condition + Means body condition was better in years with more sea ice		
SAMPLING PERIOD	MALES	FEMALES	CUBS	MALES	FEMALES	CUBS
1978-1995 spring captures	0	↓	↑	0	0	0
1992-2010 spring captures	↓	↓	NA	+	+	NA
1991-2006 fall captures	↓	↓	↓	+	+	+

The sea ice measurement used represents sea ice habitat available to the bears from mid-May to mid-October. **“0”** means no trend. “**NA**” means not enough data to analyze.
Source: Eamer et al., 2013 based on Rode et al., 2012

Other threats: harvest and contaminants

Polar bears are also very sensitive to overharvest (COSEWIC, 2008) and currently regulated harvest is the main factor limiting the size of some subpopulations (Species at Risk Public Registry, 2012). Other threats include disturbance and loss of habitat due to increased activity and opening of new areas for shipping, oil and gas development, and mining. Polar bears, as long-lived animals at the top of marine food chains, are vulnerable to the accumulation of contaminants, including mercury and persistent organic pollutants (Table 9). Mercury in polar bears can exceed threshold values for toxicological effects and there is evidence that levels are increasing (Dietz et al., 2013). In contrast, levels of some, but not all, organic contaminants in polar bear tissues have declined in recent years (Mckinney et al., 2011). Levels of contaminants, globally, in polar bears, are highest in the East Greenland subpopulation (Vongraven et al., 2012).

Grizzly bear

Grizzly bears have been extirpated across much of their former North American range, losing habitat in the southern areas and expanding in parts of the Arctic (Figure 73). Grizzly bears are a useful indicator species for revealing landscape changes in ecosystems. Body size, vital rates, and food habits correlate with habitat and seasonality of food supplies, which led Ferguson and McLoughlin (2000) to distinguish three groups of grizzlies: Pacific-coastal, interior, and barrenground. The barrenground bears typically require large home ranges: those of adult males are 1,154 to 8,171 km^2 (Nagy et al., 1983; Clarkson and Liepins, 1994; McLoughlin et al., 1999), four times larger than the largest home ranges of 77 to 1,918 km^2 for grizzly bears in other areas of Canada (Ross, 2002). In a study of 41 collared grizzly bears on the coastal plain and mountainous terrain of the Yukon North Slope, ranges of male grizzlies averaged 1,020 km^2 with the most extensive range being over 3,000 km^2 (Wildlife Management Advisory Council (North Slope), 2008b).

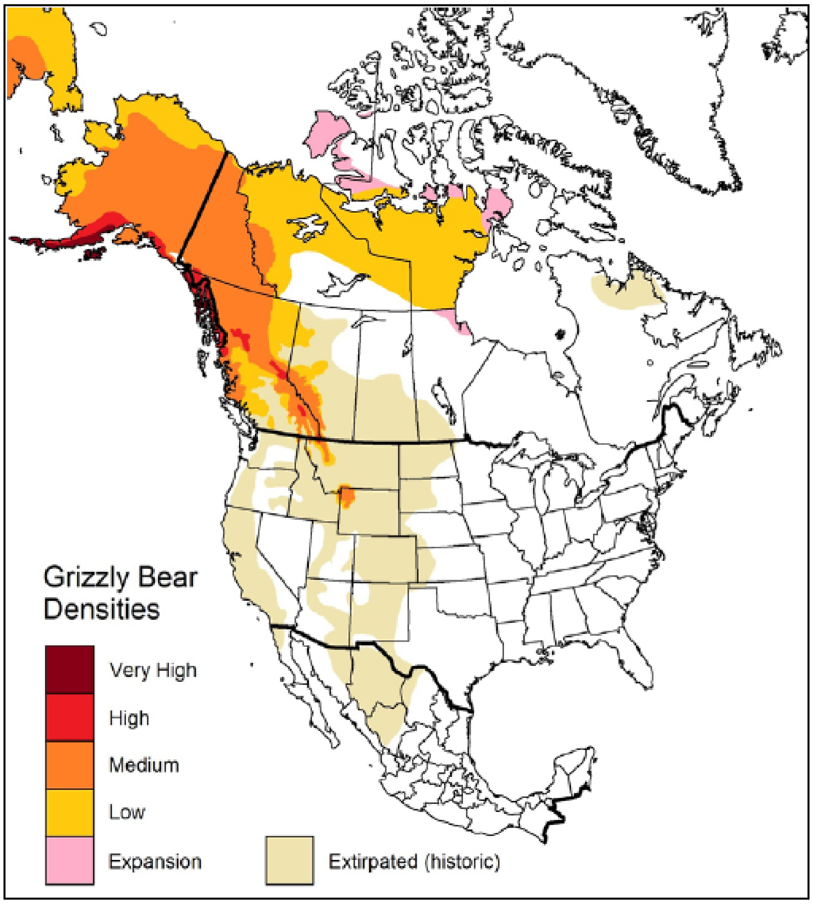


Figure 73. Approximate boundaries of current and historic (19th century) distribution of the grizzly bear in North America, showing current densities.
Worldwide, grizzly bear range has decreased by 50% since the mid-19th century.
Source: COSEWIC, 2012 and references therein

The most recent estimates are 3,500 to 4,000 grizzly bears in the Northwest Territories and between 1,500 and 2,000 in Nunavut, which is almost a quarter of the 26,000 grizzly bears estimated for all of Canada (COSEWIC, 2012). Trends in abundance are difficult to gauge in the Arctic, partly because there is little historic information and survey methods and jurisdictional boundaries have changed, and only few areas have been repeatably sampled over time. Estimating population sizes of grizzly bears is inherently difficult and costly, due to the low densities and difficulty in spotting bears. Information on status and trends will become more available through regional monitoring studies using mark-recapture techniques, including those based on DNA identification of hair that has been sampled by snagging on barbed wire. Studies using this technique are underway in the central barrens as a result of a collaborative effort between the mining industry and the Government of the Northwest Territories (for example NWT Department of Environment and Natural Resources, 2011) and on the Yukon

North Slope (Wildlife Management Advisory Council (North Slope), 2008b). A pilot project was carried out using hair snagging around Kugluktuk, Nunavut, from 2004 to 2006, followed by a larger study of grizzly bear populations in the West Kitikmeot area (Department of Environment, 2013a). Results from these studies were not available at the time of writing. Barrenground grizzly bear density estimates from earlier work are available for several areas of the Southern Arctic. The densities vary from 3.5 bears per 1,000 km^2 on the central barrenlands, in 1995–1999 (McLoughlin et al., 2003) to higher densities in the western NWT: the Anderson-Horton River area, for example, had 9.1 bears per 1,000 km^2 in 1994 (Clarkson and Liepins, 1994).

COSEWIC (2012) describes a trend of barren-ground grizzly bears expanding their range and becoming more common in the low to mid-Arctic tundra regions of northwest Canada. McLoughlin et al. (2003) reported the grizzly bear population in Nunavut was increasing in 1999 at an annual rate of 3%. There are increasingly reports of grizzly bears in areas where where they were seldom seen or not observed until recently on the Arctic islands, including Banks, Victoria, King William, and north to Melville Island (COSEWIC, 2012).

The trend toward expanding their northern distribution has had unexpected consequences. In April 2006, a sport hunter shot a hybrid polar-grizzly bear on Banks Island (Roach, 2006). His guide, Roger Kuptana of Sachs Harbor, identified it as a hybrid; DNA tests showed it had a polar bear mother and a grizzly bear father. Although polar-grizzly bear hybrids are known from zoos and are fertile (Stirling, 1999), this was the first confirmation of a wild hybrid.

Habitat trends include a large and widespread decline in caribou, which make up a large part of the diet of grizzly bears in the central barrens (Mowat and Heard, 2006). In a study in Nunavut and the NWT (Gau et al., 2002), caribou (both calves and adults) was the most prevalent item in the diet, although in early summer, when caribou were elsewhere, horsetails, sedges, and cotton grass dominated the diet, with berries also being prevalent in late summer. Some bears accompany the pre-calving migration of the caribou (Gau et al., 2002) to take advantage of the high densities of caribou on calving grounds. While caribou abundance has changed, most herds return to the same calving grounds each year, providing a predictable resource for the bears. Although the role of muskoxen in the diet of barren-ground grizzly bears has not been studied, grizzly bear predation is known to occur on muskoxen, especially on calves (Case and Stevenson, 1991; Gunn and Fournier, 2000a).

Barren-ground grizzly bears are vulnerable to the effects of industrial activities through possible displacement, but also through nuisance bear kills, and, possibly, through disturbance during denning (Harding and Nagy, 1980; Nagy et al., 1983; Clarkson and Liepins, 1994; McLoughlin et al., 1999; Edwards, 2006). Barren-ground grizzly bears have low productivity, probably as a consequence of the relatively short foraging season. In the Low Arctic tundra, average duration of den occupancy is 185 days (6.2 months) for males and 199 days (6.6 months) for females (McLoughlin et al., 2002). In the Western Arctic there may be an emerging trend toward bears entering into hibernation later in the year and emerging from hibernation earlier, based on observations of Inuvialuit hunters. This change is perhaps due to the longer growing seasons experienced in the Arctic in recent years (Wildlife Management Advisory Council (North Slope) and Aklavik Hunters and Trappers Committee, 2008; COSEWIC, 2012).

Wolverine

Wolverine abundance on the Arctic tundra is an indicator of both the availability of caribou for human harvest and the level of human activities. Wolverines are the largest and mostly solitary carnivore active in the winter on the tundra, as bears hibernate in winter and wolves mainly hunt in packs. Reproductive rates are relatively low and strongly influenced by food availability in winter. Typically, wolverine breed at two to three years of age and have litters averaging three kits (COSEWIC, 2003).

Wolverines scavenge carcasses from wolf and grizzly bear kills. In winter on the tundra, wolverines mainly eat caribou, most likely scavenged, as well as ptarmigan (*Lagopus* spp.) and small mammals. Wolverines also rely on food caches. In summer, their diet includes a wide range of small mammals, birds including geese and goose eggs, as well as berries (Mulders, 2000). Wolverines' dependence on caribou when in the tundra zone during the winter may be an important factor in habitat selection, as well explaining their large home ranges.

When caribou numbers decline, their fall and winter ranges contract and they spend more time on tundra rather than in the boreal forests. When caribou abundance is reduced, therefore, tundra-living wolverines likely still have access, at least initially, to caribou. However, if caribou abundance continues to decline, especially for herds that winter on the tundra (for example, the Dolphin and Union herd), wolverine abundance may also change. On the Bathurst caribou herd's summer-fall range, where caribou abundance declined 93% between 1986 and 2009 (Boulanger et al., 2011a), wolverine densities have also declined. For example, at Daring Lake, wolverine declined from an average density of 8 wolverine per 1,000 km^2 to 4 wolverine per 1,000 km^2 over a period of 7 years (2005 to 2011) (Boulanger et al., 2011b; Boulanger and Mulders, 2013).

Wolverine, with low reproductive potential, large ranges, and relatively low winter survival, are susceptible to reduction in numbers from trapping. Sustainable harvest relies on there being sufficient productive females surviving to recolonize areas following removal of animals through trapping (Slough, 2013). Monitoring, both of the harvest and of the distribution and density of wolverines, is key to ensuring that refugia are sufficient to sustain harvested populations, especially when other impacts, such as habitat loss due to resource extraction development, or decline of caribou populations, are present.

Wolverine population studies and programs to collect information from trappers are ongoing in the NWT (Slough, 2013), the Yukon North Slope (Wildllife Management Advisory Council (North Slope), 2012), and Nunavut (Department of Environment, 2013a). For example, in Nunavut, a harvest monitoring program, initiated in Kitikmeot in the 1980s, was expanded to other regions in 2009. The study covers geographic distribution, age, and sex of the harvested animals as well as feeding habits. A population monitoring study that involves local hunters in the collection of snagged hair samples to identify individual wolverine using DNA is planned for the Baker Lake area. This will establish baseline information: "natural" wolverine density in areas with limited or no harvest pressure (Department of Environment, 2013a).

Trends in abundance

Wolverine densities are at moderate levels on the western tundra in the NWT and Nunavut, and low on the Arctic islands and in eastern Nunavut: at the time of the 2003 COSEWIC assessment, the population estimates were 3,500 to 4,000 wolverine for the Northwest Territories and 2,000 to 2,500 for Nunavut (COSEWIC, 2003). The trends were considered stable for the NWT and Nunavut, but sensitive to changes in harvesting, as wolverine fur is highly regarded (COSEWIC, 2003). The estimate for numbers of adult wolverine resident in the Northwest Territories has since been revised to 5,100 (Slough, 2013).

Wolverine in northern Quebec and Labrador have been reduced to unconfirmed sightings since 1978 and the 1950s, respectively (COSEWIC, 2003). The declines are attributed to trapping and hunting and the extreme reduction in the caribou herds early in the 20th century, as well as human encroachment, reduced wolf numbers, and poison baiting (COSEWIC, 2003). Despite the increase in caribou populations up until the 1990s and restrictions on hunting and trapping of wolverine, populations of wolverine in these locations have not recovered (COSEWIC, 2003).

Further information on wolverine, based both on science studies and ATK, will soon be available. A new COSEWIC assessment of wolverine is due to be released early in 2014. The NWT's Species at Risk Committee is conducting an assessment of the western population, also to be released in 2014.

Trends in species of special interest

Species and groups of species of special interest for ecological, cultural, and economic reasons are discussed in this section. See the ESTR report *Ecosystem status and trends report: Arctic marine ecozones* (Niemi et al., 2010) for discussion of anadromous fish species of special interest (dolly varden and Arctic charr).

Migratory tundra caribou

This section is mainly excerpted and summarized from the ESTR national thematic report *Northern caribou population trends in Canada* (Gunn et al., 2011c) and includes updates as noted. Caribou groupings are based on that report while acknowledging that reviews of historical data and new information provide varying interpretations of geographical units relevent to caribou. Note that COSEWIC has published a report defining "designatable units" for caribou that is now in effect for the COSEWIC assessment process (COSEWIC, 2011).

Caribou, with their central role in tundra and taiga ecology and their inter-connection with the culture of many Aboriginal people, have parallels with the role of salmon on Canada's Pacific West Coast. Ecosystem aspects of caribou are discussed in previous sections and the importance of caribou to Aboriginal people is discussed in the section on Ecosystem goods and services (page 164).

Northern caribou include migratory caribou of three subspecies plus Peary caribou (Banfield, 1961; Rothfels and Russell, 2005). The three subspecies included in migratory tundra caribou are: 1) barren-ground caribou (*Rangifer tarandus groenlandicus*), ranging east of the Mackenzie

River; 2) Grant's caribou (*R. t. granti*), ranging west of the Mackenzie River,; and 3) the two large herds in Ungava of woodland caribou plus two smaller herds in the Hudson Plains Ecozone⁺ (*R. t. caribou*). Peary caribou (*R. t. pearyi)* range on the High and southern (mid-Arctic) Arctic islands and the Boothia Peninsula (Figure 74 and Figure 75).

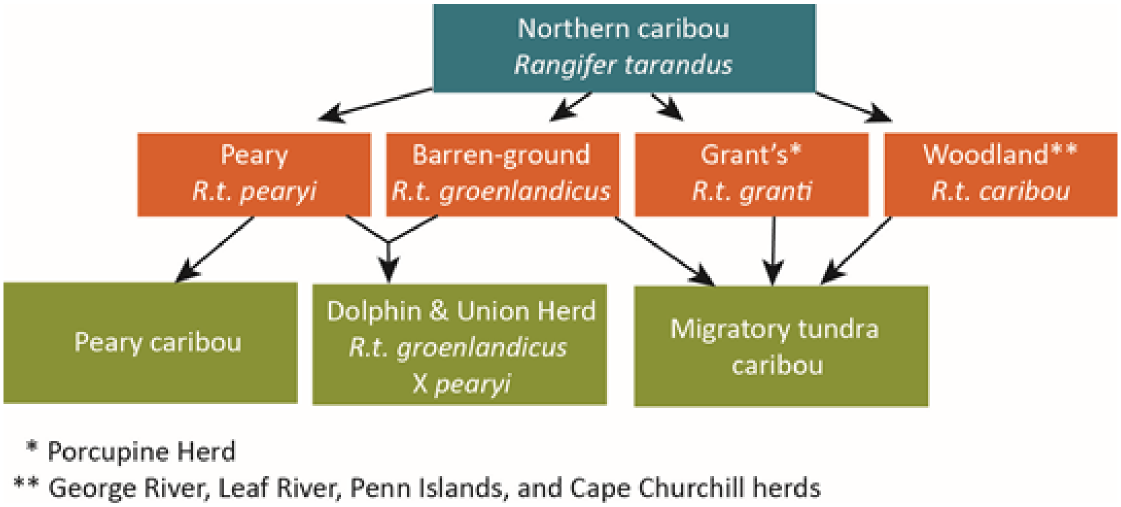


Figure 74. Northern caribou subspecies and groupings.
Source: Gunn et al., 2011c

Figure 75. Current distribution and status designations of Rangifer *in North America.*
The map depicts COSEWIC status designation. See Environment Canada, 2011 for an update of the current distribution of boreal caribou.
Source: adapted from Hummel and Ray, 2008. Map reprinted with permission from Dundurn Press Ltd.©2008

Trends in abundance

Aboriginal elders recall periods of abundance and scarcity. Other indicators of past caribou abundance and distribution include traditional place names (Legat et al., 2002). Highs and lows in historic abundance since the 1800s have been reconstructed from the frequency of hoof scars on spruce roots, at least for the Bathurst and George River herds (Payette et al., 2004a; Zalatan et al., 2006). Current ranges and recent trends are presented in Figure 76, based on information summarized in Gunn et al. (2011c). Note that the Beverly and Ahiak herds are shown here as they have been historically defined, consistent with Gunn et al. (2011c). Other authors using re-evaluation of satellite-collared caribou on the north-eastern mainland have reached different interpretations leading to uncertainty about the trends, distribution, and structure of subpopulations in that area (Nagy et al., 2011; Gunn et al., 2013b). Detailed discussion of this topic is beyond the scope of this report.

Figure 76. Ranges and recent trends of northern caribou populations in Canada.

The time spans used to assess the recent trends vary, depending on survey data available. This map has been updated from the version published in Canadian Biodiversity: Ecosystem Status and Trends 2010 *(Federal, Provincial and Territorial Governments of Canada, 2010). Further information on herd trends is available in the ESTR technical thematic report on caribou (Gunn et al., 2011c).*

Source: Gunn et al., 2011c. Bluenose East Herd information updated with 2013 census information (Environment and Natural Resources, 2014)

On the mainland, numbers were low from the 1950s to the 1970s, when the major herds began to increase (Kelsall, 1968; Gunn et al., 2011c). The increases continued into the 1980s. All eight major mainland caribou herds from the Western Arctic east to Hudson Bay declined following their peak abundance in the mid-1980s to mid-1990s (the exact timing depends on the herd). Two herds considered to be still in decline are the Leaf River and George River herds.

By 2012, the Cape Bathurst and Bluenose-West herds had stabilized at extremely low numbers following a period of sharp declines. A 2012 census for the Bathurst Herd revealed a slight decrease in the number of breeding females and a slight increase in the number of younger caribou—essentially the herd is stable at a very reduced abundance, despite the harvest being reduced from over 3,000 to 300 (B. Croft, pers. comm., 2012). At such low numbers it is difficult to detect whether "stability" is a slow decline, a slow recovery, or no trend. After a calving ground photographic census in 2008, which was the first census since 1994, the trend for the Qamanirjuaq Herd was determined to be a statistically insignificant decline.

The George River Herd declined following the mid-1980s, based on the census results for 2010. The neighboring Leaf River Herd, which increased from the mid-1980s at least until the most recent census (2001), is now considered to be declining based on information on demographic rates.

The status of the Ahiak and several herds on the northeast mainland (Wager Bay, Lorillard, Melville Peninsula, and other smaller herds on Boothia Peninsula and Simpson Peninsula), Baffin Island, and the smaller islands in Hudson Bay are currently unknown. The exception is the Southampton Island Herd whose abundance is tracked during aerial surveys at relatively regular intervals. By 2007, the herd had declined to half the peak size estimated in 1997 (30,000 caribou) (Gunn et al., 2011c) and the decline continued to about 7,900 caribou by 2011 (Greer, 2013). The 2013 survey resulted in an estimate of 7,000 caribou, with a higher proportion of calves considered to indicate that the population is stabilizing (based on interview with M. Campbell reported in Greer, 2013). On Baffin Island, Inuit observations and science-based studies indicate that caribou numbers are at a low in the cycle of abundance (Baffinland Iron Mines Corporation, 2012; Department of Environment, 2013b).

The trends in abundance are based on one indicator—the number of caribou in the herd, estimated either through calving-ground or-post calving counts (Gunn and Russell, 2008). In a few herds, such as the Bathurst and George River herds, the trends in total numbers are supported by measured trends in demographic indicators such as adult or calf survival. In other herds, especially the Beverly Herd, monitoring of herd size was infrequent and supporting data on demographic rates were not collected.

The rates of increase and decrease of individual herds vary greatly as can be seen when the rates of change for herds are plotted for periods when they were increasing (after 1970) and periods when they were decreasing (generally after the 1990s) (Figure 77). The herds with the greatest rates of increase were the Southampton and Bathurst, while the Bluenose-West and Porcupine herds showed the lowest rates of increase among herds for which there are sufficient data. During the decline phase, the Cape Bathurst Herd had the greatest rate of decline, although, with only a few breeding females on the Beverly traditional calving grounds in recent surveys, the rate of decline of the Beverly Herd may have been greater. Data are insufficient for the Beverly Herd to calculate this rate.

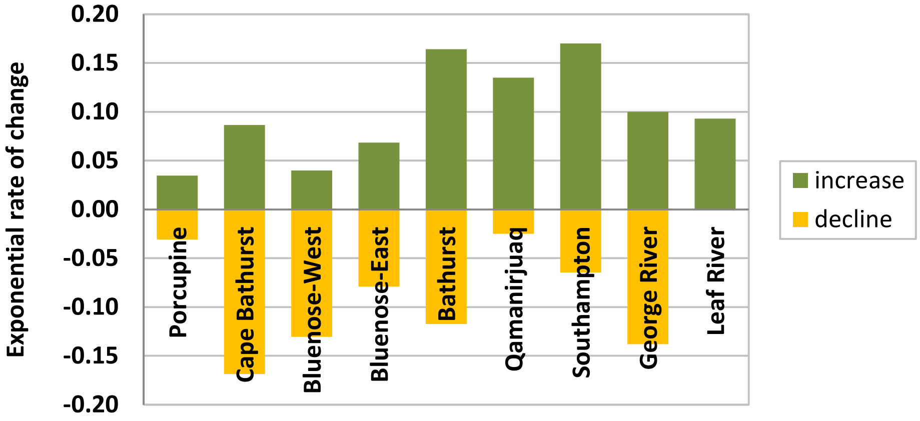


Figure 77. The exponential rate of increase and decline of major tundra dwelling caribou herds in Canada.
The chart shows the annual rate of change during increase and decline phases, based on conversion of the population estimates to natural logarithms. The years used vary among herds depending on when herds were increasing and decreasing and when population estimates were made. For the Porcupine Herd, where a change in trend direction was detected in the 2010 survey, the rate of increase is the average of 0.033 (1972–1989) and 0.035 (2001–2010).
Source: Gunn et al., 2011c

The current declining trends for some caribou herds, as well as the recent declining trends with current indications of stabilization or recovery for other herds, are likely a reflection of natural cycles in caribou abundance accentuated by the cumulative effects of increasing human presence on the caribou ranges (Gunn et al., 2013c). More conjectural is to what degree climate warming and attendant broad-scale habitat changes are factors in the natural cycles.

The causes of declines are complex, with the roles of the various contributing factors changing as the declines continue. Caribou are similar to other northern herbivorous mammals (voles, lemmings, and hares) in that their abundance is cyclic (Morneau and Payette, 2000; Gunn, 2003b; Zalatan et al., 2006) and, overall, the cycles are likely driven by climate interacting with forage availability, predation, and pathogens. Weather tends to have a decadal pattern, influenced by major patterns, such as the Arctic Oscillation, switching between negative and positive phases (Bonsal and Shabbar, 2011). Winter temperatures and snowfall patterns interact with forage growth and availability. Winter conditions and forage availability influence caribou condition, which determines birth rates and calf survival (Couturier et al., 2009a; Couturier et al., 2009b). Trends in annual calf survival and fecundity also play a role in changing herd abundance.

Weather also interacts with parasites such as warble flies, whose activity depends on summer weather. Weather affects the transmission of internal parasites, which in turn influences forage intake as caribou try to reduce their exposure to the parasites (Van der Wal et al., 2000). Predation and harvest by humans have a pivotal role in declines as even small annual reductions in adult female survival strongly influence population trends (Gaillard et al., 1998). Without mandatory harvest reporting, however, it is not possible to assess the impact of harvesting on the caribou populations.

Combining population estimate data on migratory herd numbers (including the Dolphin and Union population) since 1970 and scaling herd size relative to maximum estimates for each herd indicates that, on average, caribou numbers have increased from lows around 1975 to a peak around 1995, followed by a decline with some indication of a recent levelling off or reversal of the decline (Figure 78). The timing and magnitude of the changes vary.

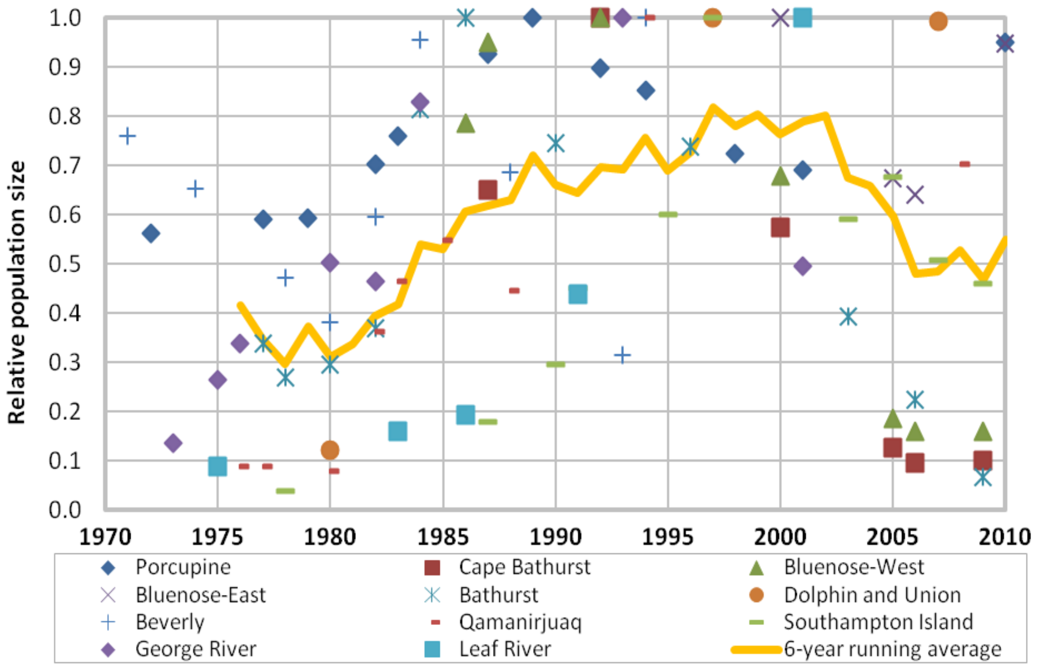


Figure 78. The relative size of tundra dwelling wild Rangifer *herds (Canada).*
The line represents the six-year running average. Other symbols represent individual herds. Relative population size is calculated as the population estimate for the year as a proportion of the maximum recorded estimate. Note that the maximum recorded estimate is not necessarily the peak population over this timeframe, as surveys usually did not cover the entire period and were not conducted every year.
Source: Gunn et al., 2011c

The cumulative effects of increasing human presence on caribou ranges (number of people as well as non-renewable resource exploration and extraction and infrastructure development) are largely unknown. However, tools are being developed to examine how responses of the individual caribou can be scaled up to measure population-level effects (Gunn et al., 2011b). Some recently constructed mine projects monitored effects on caribou and reported changes in caribou distribution and time spent foraging (Gartner Lee Limited, 2002). In response to large open-pit mines on the tundra summer range of the Bathurst Herd, caribou distribution was reduced in a 10 to 15 km zone of influence around the mines (Boulanger et al., 2004). Changes in the atmospheric transport of contaminants on individual caribou body burdens are monitored for some herds (Gamberg, 2009) and the results evaluated in relation to potential impacts on human health. These evaluations conclude that nutritional benefits of consuming caribou far outweigh any risks from the low levels of contaminants (Van Oostdam et al., 2005; Donaldson et al., 2010).

See Gunn et al. (2011c) for herd-specific assessments. These include summaries of status and trends and notes on each herd's ecology.

Trends in distribution

Herd distributions will change through time; shifts in distribution, however, are not well documented and are uncertain. Information from aerial surveys and satellite-collared individuals generally has not been analyzed to describe trends in distribution. Migratory tundra caribou characteristically shift their winter distribution among years and winter ranges often overlap between neighbouring herds (Schmelzer and Otto, 2003; Bergerud et al., 2008). Additionally, as herd abundance rises and falls, distribution—especially winter distribution—can shift (Bergerud et al., 2008). Maps of historical distribution (Banfield, 1961) and winter distribution since the 1970s, at least for the Beverly, Qamanirjuaq, and Bathurst herds (Gunn et al., 2001; BQCMB, 2004), hint at a contraction in the southern boundary of the winter distribution in northern Manitoba, Saskatchewan, and Alberta. During the 1996 to 2010 decline of the Bathurst Herd, the winter distribution of the satellite-collared cows showed a trend towards wintering further north of the 60th parallel (Gunn et al., 2011b).

Muskoxen

The muskox (*Ovibos moschatus*) is important as a symbol of the Arctic tundra and its long association with people, extending back to Paleolithic times (Lent, 1988). Its ecological role stems from its function as a large-bodied social herbivore, well-adapted for the pulse of summer productivity and the long winters that characterize tundra ecology. In many tundra areas, muskoxen are the only large-bodied prey available to wolves and humans during winter, as muskoxen are rarely long-distant migrants.

Muskoxen are distributed through northeast Canada and Greenland and have been introduced or re-introduced to Alaska, western Greenland, Scandinavia, and Russia (Figure 79). Muskoxen reintroduced to Alaska have spread, in small numbers, to the Yukon North Slope in the 1980s (Wildlife Management Advisory Council North Slope, 2008). By the early 1900s, muskox abundance in Canada had collapsed on the Arctic mainland and on some Arctic islands. Since then, numbers have built up, through natural recovery and range extension, aided by 30 to 50 years of almost no harvesting. Combining the most recent estimates for the islands and mainland, Canada has about 114,300 muskoxen, which is about three-quarters of the world's muskoxen (for references, see below). Most muskoxen occur on the Arctic islands (85%), and 77% of Northwest Territories and Nunavut muskoxen occur on the large islands (Banks and Victoria), even though these two islands form only 40% of the landmass of the Arctic islands' muskox range.

Circumpolar muskox status and trends

The world distribution of muskoxen, including introduced and re-introduced populations, is shown in Figure 79. Muskoxen have spread widely from their release sites, perhaps in search of new range as populations expanded (Reynolds, 1998; Gunn and Adamczewski, 2003). Alaska muskox numbers have stabilized and, in 2013, numbered over 4,200 in five regions (Harper, P., 2011; Gunn et al., 2013a). In Greenland, status and trends in muskoxen are unknown. The most recent survey, in northeast Greenland about 20 years ago, estimated 9,500 to 12,000 muskoxen (Boertmann et al., 1992). More is known about the introduced population at Kangerlussuaq, where the initial 27 muskoxen introduced in the 1960s increased rapidly. Rough estimates are for a current population of 10,000 to 25,000. Other populations in West Greenland have also shown good growth, and quota-based harvesting has been implemented. Based on the above, the total numbers of muskoxen in Greenland may be from 20,000 to 40,000 (Greenland Institute of Natural Resources, 2012). In Russia, re-introductions started in the mid-1970s in several locations. The total muskox population in Russia was estimated at over 10,000 in 2013, with the great majority (8,700) being on the Taimyr Peninsula (Gunn et al., 2013a).

Figure 79. Muskox world distribution.
Not all introduction sites represent current populations: for example, 17 muskoxen were introduced to Svalbard in 1929 and the small population that became established was extinct by 1982 (Long, 2003). The small areas marked in Nunavut are locations where populations were successfully introduced (Chubbs and Brazil, 2007). Muskoxen have since extended their range, but the current range is not sufficiently defined to map.
Source: Updated from Wikimedia Commons, based on Gunn and Forchhammer, 2008 and Large Herbivore Network, 2009

Since the Pleistocene, muskoxen have varied little genetically and in appearance. Skull and dental characteristics from contemporary and Pleistocene muskoxen are similar (Harington, 1970) and a comparison of ancient and modern DNA also suggests muskoxen have changed little over thousands of years and their variation has not been sufficient to recognize subspeciation (Tener, 1965; Groves, 1995). This is not to say that mainland and Arctic island muskoxen do not differ, but that the differences were not enough to warrant sub-specific designation. Muskoxen have notably low genetic variability, though mainland muskoxen are slightly more genetically variable than island muskoxen, based on microsatellite DNA analysis (Van Coeverden de Groot, 2001).

Information on changes in the number or distribution of distinct muskox populations is not available, as Canadian muskoxen are managed for harvesting based on management units (rather than natural populations). Often, for the Arctic Islands, the management units are islands or groups of islands. On the mainland, the units are based on hunting patterns and on changing information about distribution, and the managment unit boundaries are modified at intervals. Nunavut's management units were recently revised as part of the development of new muskox management plans (Dumond, 2006; Kivalliq Wildlife Board, 2010; Government of Nunavut, 2012). The NWT has seven muskox management units (Environment and Natural Resources, 2012a). The boundaries of aerial survey areas do not always coincide with management unit boundaries, or they may cover several units, which complicates analysis of historical trends (Fournier and Gunn, 1998; Dumond, 2006).

Trends in abundance

The record of European exploration and settlement explains why muskoxen on the mainland were reduced to isolated remnants by the early 1900s, as traders encouraged commercial hunting for sales of hides in European markets (Barr, 1991a). The muskox hides were marketed to replace the loss of bison robes as the bison numbers were collapsing, also in part due to unregulated commercial harvesting (Barr, 1991a). From 1862 to 1916, the Hudson's Bay Company purchased more than 15,000 muskox hides (Tener, 1965). The company continued to purchase hides as the number brought in fell from more than 1,000 per year in 1891 to only a few score in 1908, reflecting the declining population. In 1916, the last year before the Canadian government provided legislative protection, just one muskox skin was sold (Figure 80).

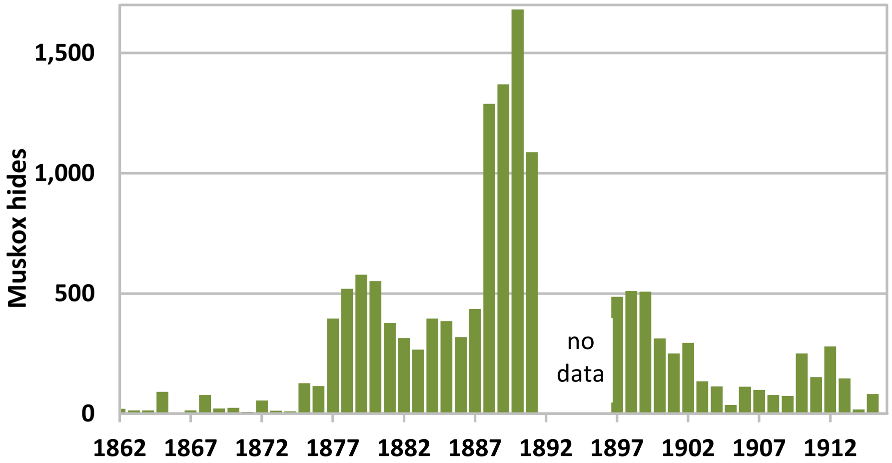


Figure 80. The rise and fall of the muskox fur trade in Canada: total numbers of muskox hides purchased annually by the Hudson's Bay Company, 1862-1916.
Source: Tener, 1965

On the Arctic islands, European explorers hired Inuit to provide meat for their expeditions, and Hone (1934) calculated that, between 1852 and 1916, British and Canadian expeditions to Melville Island killed over 600 muskoxen for meat and to capture calves for zoos. Over 1,000 muskoxen were killed on Ellesmere Island by Norwegian and American explorers between 1880 and 1917 (Barr, 1991b). However, how those harvests affected long-term trends is essentially unknown. On Banks and western Victoria islands, muskoxen had essentially disappeared by the late 1800s. This drastic decline was attributed to ice storms (Gunn, 1990).

Harvesting of muskoxen was suspended from 1924 to 1969, when a small quota was established for southern Ellesmere Island. The Thelon Game Sanctuary (later to become the Thelon Wildlife Sanctuary) was established in 1927, in large measure to protect muskoxen (Taylor, 2006). During the 1970s and 1980s, conservative quotas for harvesting became more widespread as hunters' reports and surveys revealed increasing muskox numbers (Barr, 1991b). Recovery appeared to be relatively slow over the decades, partly as a consequence of lack of surveys to track the recovery, and partly because the muskoxen were dispersing and recolonizing their previously occupied ranges (Barr, 1991b). Muskoxen have a potential doubling rate of three years, although that is relatively rare. They are similar to other large-bodied herbivores, having high adult survival, low fecundity, late onset of breeding, a single birth every one to three years, and females having a lifespan often exceeding 15 years (Gunn and Adamczewski, 2003). Muskox numbers reached an estimated 134,000 in 2001 (summarized in Dumond, 2006) but, since then, have declined, mostly because of the decline in muskoxen on Banks Island. Although the mid-Arctic islands of Banks and Victoria are only 9% of the landmass of the Arctic tundra, they are home to 72% of Canada's muskoxen. Only 14% of muskoxen are on the mainland. Estimates of Canadian muskox abundance from 1967 to 2012 are shown in Figure 81. The breakdown of the most recent estimates by location across the Arctic Ecozone$^+$ is shown in Table 11.

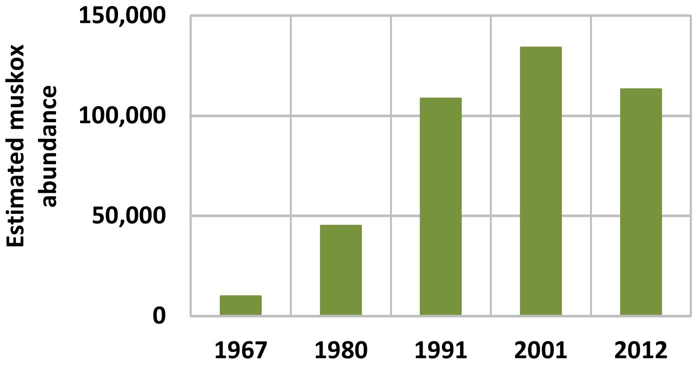


Figure 81. Estimated muskox abundance in Canada, 1967–2012.
Source: Dumond, 2006 and references therein; references in Table 11

Table 11. Muskox population estimates by region

Area	Muskox numbers	Error estimate (where available)	Survey year	Source
Nunavut				
Bathurst Island Complex	82		2001	Jenkins et al. (2011)
Cornwallis	18		2002	
Devon Island	513	302–864 (95% CI)	2008	
Prince of Wales	2,086	1,582–2,746 (95% CI)	2004	
Somerset	1,910	962–3,792 (95% CI)	2004	
North Ellesmere Island	8,115	6,632–9,930 (95% CI)	2006	
South Ellesmere Island	456	312–670 (95% CI)	2005	
Axel Heiberg Island	4,237	3,371–5,325 (95% CI)	2007	
Amund Ringnes, Ellef Ringnes, King Christen, Cornwall, Meighen, and Lougheed islands	21		2007	
King William Island	300			White (2002)
West of Bathurst Inlet (MX 19)*	2,141	±586 SE	2005	Dumond (2007a)
(MX 14, western part)	434	±168 SE	2005	
West of Coppermine R.	132	±71 SE	2007	Dumond (2007b)
Boothia Peninsula High density area	348	±62 SE	2006	Dumond (2007c)
Medium density area	645	±119 SE		
Low density area	104	±72 SE		
East Queen Maud Gulf	2,621	379 SE	2000	Campbell and Setterington (2001)
Northeast Kivalliq	165		2000	
North of Baker Lake	4,736	554 SE	2010	
Southeast Victoria Island (NU)	25,000-30,000		1992-1999	Dumond (2006)
Thelon Sanctury (NU and NT)	1,095	±281 SE	1994	Gunn et al. (2009)
Northwest Territories				
Northwest Victoria Island (NT)	11,442	1,637 (95% CI)	2010	Davison et al. (2013)
Banks Island	36,676	4,031 (95% CI)	2010	Davison et al. (2013)
Melville Island	3,033	852 (95% CI)	2012	Davison and Williams (2012)
Prince Patrick Island	507	320 (95% CI)	2012	
Eglinton Island	213	211 (95% CI)	2012	
South Paulatuk	1,215	±525 SE	2002	Community of Paulatuk et al. (2008)
North Great Bear	1,457	±919 SE	1997	Veitch (2013)
Alymer Lake	161	±39 SE	1991	Shank and Graf (2013)
Artillery Lake	1,606	±277 SE	1998	Bradley et al. (2001)
Beaverhill Lake	532	±149 SE	2000	Gunn et al. (Gunn et al., 2009)
Yukon				
Yukon North Slope	101		2011	Wildlife Management Advisory Council (North Slope) (2012)
Quebec (Nunavik)	1,400		2003	Chubbs (2007)

* MX indicates Nunavut muskox management units

Surveys are conducted in the spring and are for one-year-plus-old muskoxen. In general, when there is no range or error estimate provided, the number respresents a mimimum count (i.e., the actual number seen in the survey).

Trends in regional populations

Arctic mainland

By the 1930s, remnants of muskox populations (with a total of perhaps about 500 muskoxen) were scattered across the Arctic mainland, with a cluster of herds north of Great Bear Lake, south of Bathurst Inlet, in the Thelon Game Sanctuary, and south of Boothia Peninsula (Anderson, 1934; Barr, 1991b). As hunters reported sightings of muskoxen, aerials surveys were undertaken for several mainland regions. North of Great Bear Lake, muskoxen recolonized, and their numbers increased during the 1990s (Fournier and Gunn, 1998; Dumond, 2006).

While the muskoxen continued to spread west toward the Mackenzie River, their numbers declined behind the "colonizing front". For example, **north of Great Bear Lake and west of Coronation Gulf**, muskoxen increased from scattered remnants in the 1900s to 425 in 1967 (see review in Dumond (2007b)). In a small portion of this area, the Rae-Richardson watersheds, there were 869 ± 300 (SE) in 1980, 1,295 ± 279 (SE) in 1983, and 1,805 ± 289 (SE) in 1988 (Fournier and Gunn, 1998; Dumond, 2007b). However, between 1989 and 1994, the muskox numbers declined, possibly due to a parasitic lungworm (*Umingmakstrongylus pallikuukensis*) which may have increased muskox vulnerability to grizzly bear predation (see section on Wildlife diseases and parasites on page 60). Between 1994 and 2007, numbers appeared to stabilize. The 1994 estimate of 540 ± 139 (SE) is similar to a 2007 estimate of 509 adult muskoxen (Dumond, 2007b).

East of the Coppermine River to Bathurst Inlet, a 2005 survey found 2,141 ± 586 (SE) muskoxen in the western part of the study area, indicating an increase in muskox abundance since a 1991 survey. However, the abundance had declined to 434 ± 168 (SE) in the eastern part of the study area, compared to a 1986 survey (Dumond, 2007a).

Few muskoxen were reported in the **Queen Maud Gulf and Adelaide Peninsula area** until the 1960s. By August 1982, however, numbers were estimated at 8,494 ± 2,673 (SE) (Gunn et al., 1984). The trend between 1982 and 1996 was a 50% decline in numbers, as the 1996 estimate was 4,255 ± 680 (SE) (Fournier and Gunn, 1998). It is unclear to what extent the decline is a distributional shift, as hunters in Kivalliq and northeast Kitikmeot areas report increasing numbers and expanding distribution through the 1990s (Campbell and Setterington, 2001). South of the Queen Maud Gulf area, in the Kivalliq Region, muskoxen were surveyed in 1986, 1991, 1999, and 2010. Preliminary analyses of the 2010 findings indicate both population and range expansion. The 2010 estimate was 4,736 ± 554 (SE) muskoxen with a colonizing edge moving east toward Hudson Bay coast and a decline in densities in the survey area (Ford et al., 2012).

Further north and east of the Queen Maud Gulf, Gunn and Dragon (1998) estimated 554 muskoxen in 1995 on the **Boothia Peninsula** where they were scarce or absent through the 1980s (Gunn and Dragon, 1998). This number had approximately doubled to an estimated 1,097 by 2006 (Dumond, 2006).

South and west of the **Thelon Game Sanctuary**, muskoxen were re-colonizing their former ranges toward the treeline. The estimated number of adult muskoxen in the sanctuary in 1994 was 1,095 ± 281 (SE) (Gunn et al., 2009). Hunters in **Lutsel K'e** also reported increasing numbers of muskoxen east of Artillery Lake and surveys in 1989 and 1998 revealed that muskox numbers had doubled and that muskoxen had spread into the taiga to the west and southwest (Bradley et al., 2001). A survey documented the increased spread of muskoxen south toward the treeline when, in 2000, 1,320 ± 183 (SE) muskoxen were estimated (Gunn et al., 2009).

Muskoxen disappeared from the **Yukon North Slope** between 1858 and 1865 and were re-established in the Arctic National Wildlife Refuge in Alaska in 1969–1970 (Reynolds, 1998). From there a few animals moved into the Yukon in 1972 (L. Harding, pers. obs.) but it was not until the mid-1980s that repeated sightings of cow muskoxen were reported; mixed groups with calves were first sighted in 1987 (Johnson et al., 2005). The range continued to expand east to the Mackenzie River and south into the Taiga Cordillera Ecozone^{+}. Because the North Slope muskoxen move between Alaska and the Yukon, population estimates include animals on the Alaska and Yukon North Slope and areas to the south. The North Slope population was estimated in 2006 to be 400 animals, a decline from 700 animals estimated in the mid-1990s, based on extensive aerial surveys. A total of 291 muskoxen were spotted in the survey conducted in 2011—of these, 101 were in the Yukon (Wildlife Management Advisory Council (North Slope), 2012). Reports of sightings by Aboriginal people in the region indicate a decline since the 1990s (Arctic Borderlands Ecological Knowledge Co-op, 2006).

Muskoxen from Ellesmere Island were initially introduced to a farm in 1967 near Kuujjuaq and then released between 1973 and 1983 at three locations in Nunavik (Lehenaff and Crete, 1989a; Chubbs and Brazil, 2007). The muskoxen rapidly increased to 1,400 by 2003 and have been sighted in Labrador (Chubbs and Brazil, 2007).

Arctic islands

The most recent muskox numbers available for the Arctic Islands indicate a total of 94,130 (non-calves): these estimates span the period 1992 to 2012. This total includes 13,300 muskoxen on the **eastern Queen Elizabeth Islands**, based on aerial surveys between 2004 and 2008 (Jenkins et al., 2011). No trend for these islands is discernible as the previous estimate was in 1961 (Tener, 1963).

On the **western Queen Elizabeth Islands**, the more frequent surveys in the early 1970s, mid-1980s, and 1990s indicate that there have been two sharp declines and one or two periods of recovery. The total number of estimated muskoxen in 2012 on the western Queen Elizabeth Islands was 3,750, an increase of 48% over 15 years (Davison and Williams, 2012).

The trends for the western Queen Elizabeth Islands can be related to a pattern of increases and sharp decreases during winters with above-average snow depths and icing (while acknowledging that the weather data are sparse). For the islands in the Western Arctic with sufficient number of estimates to describe trends, there is very little regional consistency. This can be seen for Prince Patrick, Melville, and Bathurst islands, which are stretched over 600 km across a progressively decreasing Beaufort maritime influence (Maxwell, 1981). Muskox numbers on **Prince Patrick Island** increased from none to 507 ± 320 (95% confident interval) between 1961 and 2012, while, during the same period, muskox numbers on **Melville Island** peaked in 1986 and declined by the mid-1990s before recovering to 3,033 ± 852 (95% confident interval) (Davison and Williams, 2012). Based on aerial surveys, muskoxen on **Bathurst Island** have been through two declines and one recovery since 1961 (Figure 82). In 1961, muskox numbers were relatively high: Tener (1963) estimated 1,136 muskoxen. When the population was next estimated in 1973, it had declined 59% and, by August 1974, the continued decline had reduced the estimated number of muskoxen to 164 ± 70 (SE) after a severe winter (Miller et al., 1977). Muskox numbers recovered and increased between 1974 and 1993 to an estimated 1,200 (Ferguson, 1987; Miller, 1987 and 1998). A series of three winters, 1994 to 1997, with rain on snow and unusually deep snow reduced the muskox abundance and, in summer 1996, Miller (1998) estimated that there were 425 ± 136 (SE) muskoxen alive. By 1997, only an estimated 124 ± 45 (SE) one-plus-year-old muskoxen were alive, which indicated a 90% decline since 1994 (Gunn and Dragon, 2002). During an aerial survey in 2001, only 94 muskoxen were observed and no population estimate was derived (Jenkins et al., 2011).

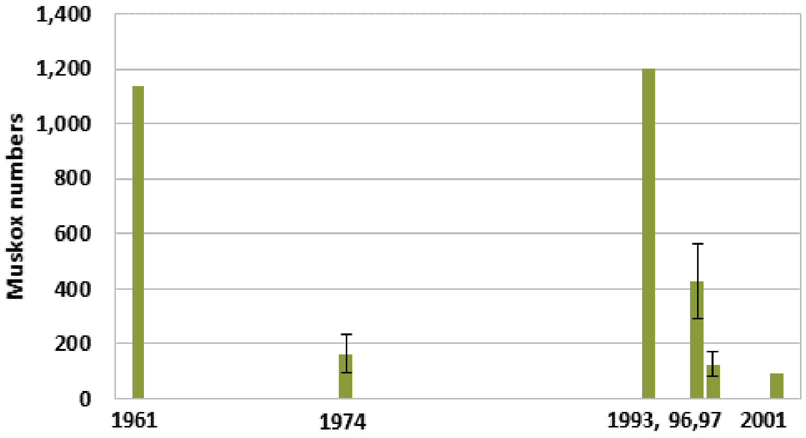


Figure 82. Muskox trends, Bathurst Island, 1961- 2001.
Error bars are ± standard error. See text for more information on trends.
Source: Tener, 1963; Miller et al., 1997; Miller, 1998; Gunn and Dragon, 2002; Jenkins et al., 2011

The muskox trends on the larger, more southern mid-Arctic islands are quite different from those on the High Arctic islands (Queen Elizabeth Islands). A relatively well-documented trend is that of Banks Island, where the population grew from hundreds in the 1960s, based on Inuvialuit accounts to a peak around 2001, followed by a decline to the most recent survey in 2010 (Table 11, Figure 83) (Gunn et al., 1991b; Inuvialuit Game Council, 2002; Lyberth, 2003; Davison et al., 2013).

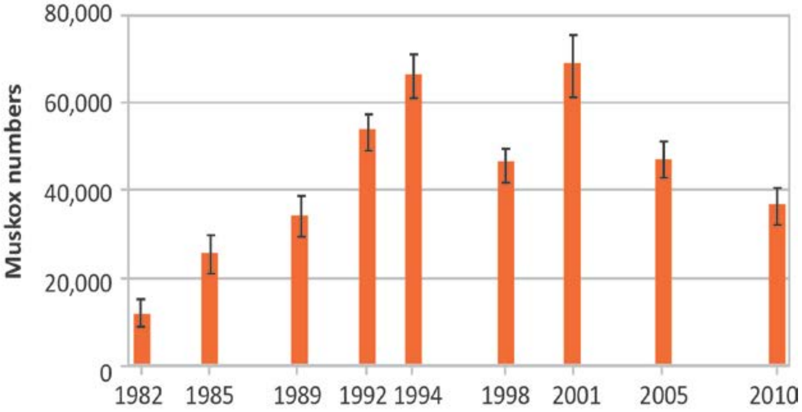


Figure 83. Muskox trends, Banks Island, 1982-2010.
Error bars are 95% confidence intervals.
Source: Davison et al., 2013

Similar to the trends on Banks Island, muskox numbers increased on Victoria Island during the 1970s to 1990s: the Nunavut part of Victoria Island had an estimated 25,000 to 30,000 muskoxen in 1992–1999 (Dumond, 2006; Gunn and Patterson, 2012) and the Northwest Territories portion had an estimated 11,442 in 2010 (Davison and Williams, 2013). In 2004, surveys of the large islands east of Victoria Island, 2,086 muskoxen were estimated on Prince of Wales, and 1,910 muskoxen were estimated on Somerset Island (Jenkins et al., 2011). The muskox populations on those two islands have been increasing since at least the 1970s (Nagy et al., 1996; Fournier and Gunn, 1998; Jenkins et al., 2011; Gunn and Patterson, 2012; Davison and Williams, 2013).

Trends in distribution

The Canadian population is almost entirely within the Arctic Ecozone$^+$ except for in the Thelon Wildlife Sanctuary and, recently, where muskoxen have spread along the treeline from northeast of Great Slave Lake toward the Saskatchewan border. They are distributed on the tundra in areas with either shallow snow (20 to 40 cm) or sufficient relief for the wind to maintain areas with shallow snow. Summer habitats include low-lying river valleys or coastal plains with sedge meadows, riparian willows, and gravel bars. Muskoxen are widely distributed and occur in the Low, Middle, and High Arctic tundra ecoregions (an area of 2,293,372 km^2).

There have been no overall analyses yet of the trends in distribution. Since the early 1900s, muskoxen have recolonized most of the mainland tundra except northeastern areas, including the Melville Peninsula. Likewise, muskoxen occur on most Arctic islands except the northeast islands (Baffin and the islands in Hudson Bay). Muskoxen disappeared from Baffin Island during the fifteenth century, with only ocassional records since then, such as a herd of eight muskoxen observed south of Clyde River in 1968 (Barr, 1991b). The recolonization of historical ranges has been natural, except for the Yukon North Slope and the successful introduction to northern Quebec. Muskoxen apparently did not naturally colonize Quebec after the glacial retreat (Lehenaff and Crete, 1989b). The rate of recolonization is slow—perhaps less than 10 km per year (Fournier and Gunn, 1998).

Muskoxen occasionally appear on and disappear from smaller and medium-sized islands. Tener (1963) did not find muskoxen on Prince Patrick, Eglinton, and Mackenzie King islands in 1961, although they were recorded there during the aerial surveys in 1972–1974, 1986, 1997, and 2012 (Miller et al., 1977; Gunn and Dragon, 2002; Jenkins et al., 2011; Davison and Williams, 2012). Within the islands, relatively little analysis has been undertaken for trends in distribution. Even so, information does suggest some changes. Between 1972 and 1980, exceptionally high densities of muskoxen were found on Bailey Point on southwest Melville Island (Thomas et al., 2013). The densities were from 0.6 to 1.1 muskoxen per km^2 over the entire peninsula, reaching 2.6 per km^2 below 100 m, which was comparable to other "hotspots" of muskox abundance, including the Thomsen River, Banks Island. Bailey Point was considered to have a particularly favourable microclimate to serve as a refugium for muskoxen. The high densities persisted at least to 1987 (Miller, 1988), but in 1997, when numbers had declined by about 50% since 1987, almost no muskoxen were seen at Bailey Point, a situation which persisted in 2005 and 2012 (Gunn and Dragon, 2002; Davison and Williams, 2012).

Trends in harvest

Muskoxen are harvested under a quota system introduced in 1969. Quotas have increased and, by 2011, in Nunavut had reached a Total Allowable Harvest of 2,303 muskoxen (Giroux et al., 2012b). This includes harvest for domestic use, commercial harvest, and sport hunting by non-residents (Wildlife Research Section, 2011). In the Northwest Territories, the total harvest quota is 1,112 in the Inuvialuit Settlement Region, which is mainly two quotas that cover Banks and northwest Victoria islands (NWT Environment and Natural Resources, 2011). These two quotas were assigned to influence the rate of increase in the 1990s and to encourage commercial harvesting. Annual use of the quotas varies, especially in the commercial harvesting for meat and qiviut (wool) sales.

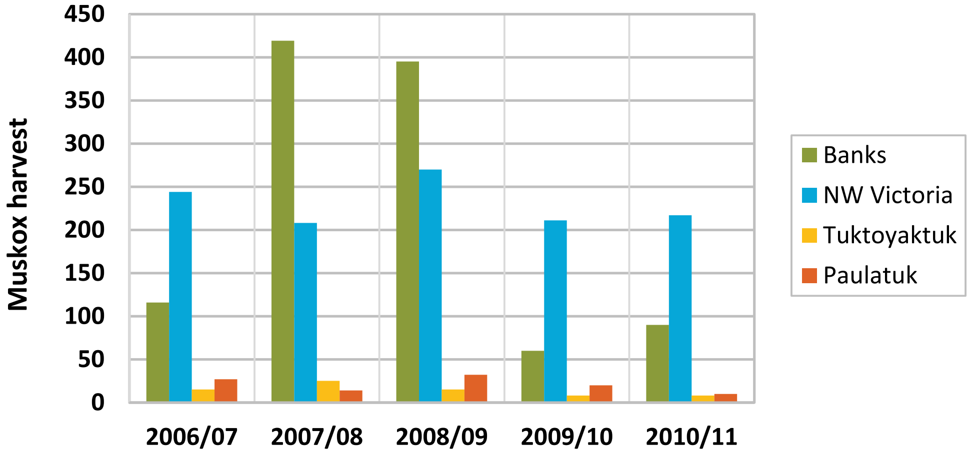


Figure 84. Annual Inuvialuit total muskox harvest, 2006-2011.
Source: NWT Environment and Natural Resources, 2011

Threats

Threats that cause or contribute to population declines may lie in the unexpected, which may be related to shortcomings in data collection or predictive models, or unanticipated environmental changes accumulating in ways not encompassed by traditional knowledge. The extent of effective management (including habitat and landscape management planning) is the deciding factor for anticipating and dealing with threats. The status of muskox management planning varies between Nunavut and Northwest Territories. Within Nunavut, regional management plans with a focus on harvest management have recently been completed (Dumond, 2006; Kivalliq Wildlife Board, 2010; Government of Nunavut, 2012) which include adaptive management to adjust harvest levels if popuation levels change. The Inuvialuit Settlement Area has community conservation plans and harvest reporting (Community of Paulatuk et al., 2008; NWT Environment and Natural Resources, 2011); governance, however, is more complex elsewhere in the Northwest Territories. The harvest, except on Banks and northwest Victoria islands, is relatively low and an overall approach to muskox management is absent.

Predators

Predation is a natural driving force in muskox evolutionary ecology. However, almost nothing is known about how muskoxen trade off the risks of predation (and parasitism) relative to foraging. Wolf predation on muskoxen is likely common, with packs or single wolves observed killing adult as well as younger muskoxen (Gray, 1987; Mech and Adams, 1999). On northern Ellesmere Island, and northern and eastern Greenland, muskox occurrence in wolf scats ranges from 65–98%, with lemmings and Arctic hares as the next most frequent item (Marquard-Petersen, 1998). On Banks Island, where Peary caribou are less numerous, muskox remains were found in 103 out of 115 stomachs and 34 out of 38 scat samples (1992 to 2001), with lemmings being the next most frequent item (Larter, 2013). Trends in wolf predation on muskoxen are unknown.

Although unmeasured, an increase in grizzly bear sightings on muskox ranges is being reported (Ford et al., 2012). Newborn calves that cannot keep up with a herd are easily killed by pursuing bears (Clarkson and Liepins, 1994). Deep, crusted snow increased the vulnerability of muskoxen in at least one instance of a grizzly bear killing a bull (Case and Stevenson, 1991).

Predation in some areas, such as west of Kugluktuk, may be accentuated by infection with a lungworm (see section on Wildlife diseases and parasites on page 60 and below). Muskox abundance increased and peaked at 1,800 in 1987, but then declined by about 50% in 1994 and remained stable in 2007 (Dumond, 2007b). The muskoxen were infected with the lungworm *Umingmakstronylus pallikuukensis*, which may increase muskox vulnerability to predation, especially from grizzly bears, as the lungworm forms cysts in the lungs, causing difficulties breathing.

Health and trends in health

It is difficult generally to detect trends in health, as knowledge of muskox parasites has been increasing since the 1990s. However, it likely that the spread of at least one parasitic lungworm may be associated with a warmer climate. In the 1990s the lungworm *Umingmakstrongylus pallikuukensis* was described (as a previously unknown species) with a range restricted to the western Canadian Arctic mainland, extending west of Kugluktuk (Hoberg et al., 1995). Climate models predicted range expansion of the parasite and it has now been found on southernVictoria Island (Kutz et al., 2009) (see section on Wildlife diseases and parasites on page 60).

Muskoxen may have a low resistance to parasites even when in good body condition (Alendal and Helle, 1983; Korsholm and Olesen, 1993). This characteristic may be related to their low genetic variability. Muskoxen in some areas, notably Banks Island, are subject to outbreaks of the bacterial disease *Yersinia pseudotuberculosis,* which may have an environmental trigger such as unusually warm weather. Exposure to *Yersinia* is high (Larter and Nagy, 1999) and periodic outbreaks include deaths. A previously unknown disease in muskoxen that led to deaths on Banks Island in 2012 was erysipelas, a bacterial disease that is common in livestock (M. Branigan. pers. comm., 2012).

Weather-related deaths

Most weather-related deaths are indirect, as they are caused by malnutrition. The general projections for a warmer climate include more incidents of rain on snow, which are associated with muskox deaths, at least on Banks Island (Rennert et al., 2009; Nagy and Gunn, 2009). Periods of rain when calves are young can also increase mortality (P. Hale, pers. comm., 2013).

Industrial disturbances

The trend is for increasing industrial exploration and development, including construction of new roads. Predicting how muskoxen and their habitat will be influenced is uncertain as relatively little is known about cumulative effects on muskoxen. Earlier studies focused on behavioural responses of muskoxen to seismic and aircraft disturbance (e.g., Miller and Gunn, 1980). Studies in Alaska related to the National Petroleum Reserve provide more information on potential disturbance and mitigation measures associated with development (Bureau of Land Management, 2012).

Large carnivores

Wolves are the major large carnivore in Arctic tundra systems, with grizzly bears being important in much of the Southern Arctic. Medium-sized predators include Arctic and red foxes and wolverines, and avian predators include snowy owls and jaegers. Grizzly bears and wolverines are discussed above in the section on Trends in species of conservation concern (page 117), and wolves are discussed below. See also the section on Community and population dynamics (page 54) and the Main threats to caribou (page 167) for discussion of predator-prey dynamics.

With changes in land-use patterns and the expansion of human populations in southern Canada, carnivores have lost much of their former North American ranges, making the northern regions of the continent increasingly important for species conservation (Figure 85).

Figure 85. Reduction in ranges for large carnivores in North America.
Source: Hummel and Ray, 2008

Wolves

A well-known species and top predator, the continued presence of wolves is an important indicator of Arctic ecosystem integrity. Wolf populations are not monitored regularly. Moreover, since some follow caribou and some do not, interpreting abundance and distribution data would be problematic for some areas. Wolves in highly productive environments can reproduce rapidly, making them resilient to various types of disturbance, including hunting.

Gray wolves, *Canis lupus*, in the Arctic Ecozone$^+$ were formerly considered to include up to eight subspecies (Mech, 1974). Although subspecies status is currently in doubt, genetic analyses suggest that wolves in the Northern Arctic (sometimes called “Arctic” wolves) are separable from Southern Arctic (“tundra”) wolves (Carmichael et al., 2007), and that these, in turn, are separable from boreal forest (“gray”) wolves (Musiani et al., 2007). Aside from their genetic structures, the three groups have different prey specialization patterns (Carmichael et al., 2001; Musiani et al., 2007).

Wolf populations of the Arctic Archipelago form a metapopulation, some populations of which have endured severe bottlenecks that probably occurred during catastrophic declines in their principal prey, Peary caribou and muskoxen. Arctic hares are also important prey species. From 2000 to 2006, Arctic wolves, Arctic hares, and muskoxen were counted in a study area on Ellesmere Island. Wolf numbers were strongly correlated with hare numbers, but not with muskox numbers (Mech, 2007).

Wolves on Banks, Ellesmere, and Devon islands show genetic signatures of recent population declines (Carmichael et al., 2008). Island wolf populations in general have significantly less genetic variability than mainland populations, suggesting that there is movement among the islands and that populations recover after a decline, mainly through recolonization from other islands (Carmichael et al., 2008). The wolves’ genetic diversity appears to have been augmented, however, by periodic migration from mainland populations, occurring primarily through two corridors: Baffin Island and Victoria Island. This gene flow could be compromised by the loss of sea ice due to climatic warming and increased human activity (Carmichael et al., 2008). In particular, year-round marine traffic would keep an open water route through certain marine passages, blocking movement of wolves in the winter.

Tundra wolves in the Southern Arctic make heavy use of eskers, which are also a source of gravel for industrial uses such as mine site and road construction, a resource often in short supply in Arctic infrastructure construction (McLoughlin et al., 2004). Some tundra wolves are caribou specialists and follow migrating caribou for long distances (Frame et al., 2004).

Trends in wolves

There is some information on trends in wolf abundance, based on community and traditional knowledge and on observations of wolves during systematic aerial surveys to count caribou and muskoxen. There is no direct information on predation rates.

On Banks Island, wolves were poisoned in the 1950s and recovered slowly, increasing during the 1980s and 1990s (Figure 86) and often spotted in areas with high muskoxen density.

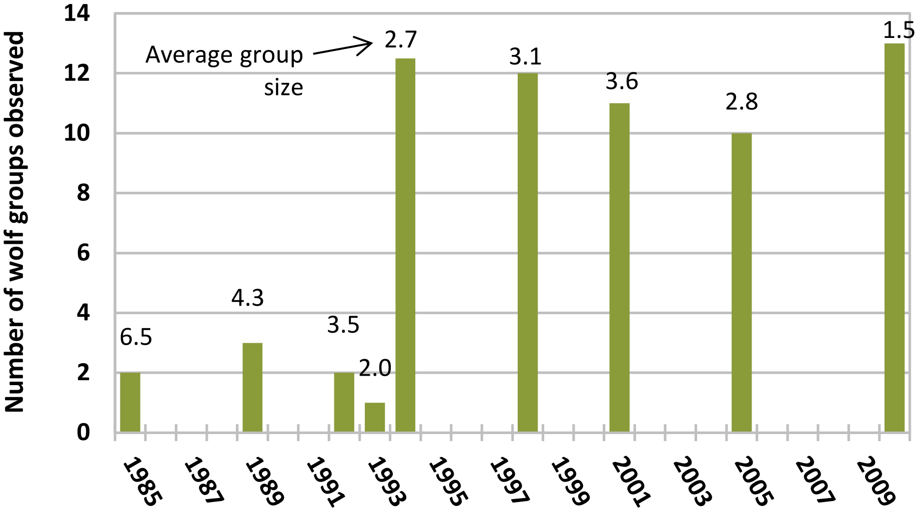


Figure 86. Wolves sighted during aerial surveys of Peary caribou and muskoxen, Banks Island, NWT. Group size includes adults and pups. Values for 1994 and 1998 are averages of two surveys. Source: data from Species at Risk Committee, 2012a

For the Bathurst Herd, wolf sightings between 1987 and 2008 during late winter aerial surveys to estimate caribou calf survival suggest no consistent trend in either wolf sightings or mean pack size (Williams and Fournier, 1996; Gunn, 2013; B. Croft, pers. comm., 2010). During that time, the numbers of caribou declined and the number of wolves seen at their dens, as well as the number of occupied dens, declined (D. Cluff, unpublished data). The rate of sightings in the vicinity of caribou, however, suggests that predation rates were maintained.

Birds

Arctic-nesting birds winter in many parts of the world where they may be vulnerable to stressors including loss of food supplies and habitat, pollution, disturbance, and overharvesting during winter and during migration. In the Arctic, they are vulnerable to changes in their habitat and food supplies and, in some cases, to overharvest. Data are lacking or sparse for many species, and it is often difficult to determine the causes of trends. Many Arctic-nesting shorebird and landbird species are known to be declining, as are some sea ducks. Other Arctic bird groups, such as geese and swans, have mainly stable populations or have increased over the past few decades. Trends in Arctic seabirds (not including sea ducks) are discussed in the ESTR report *Ecosystem status and trends report: Arctic marine ecozones* (Niemi et al., 2010). An overview of seabird trends for Canada is presented below (text box and Figure 87). For more information on the status of bird populations see *The State of Canada's Birds 2012* (NABCI, 2012).

Trends of seabirds in Canada

Seabirds nesting in the Arctic are covered in more detail in the ESTR report on Arctic marine ecosystems (Niemi et al., 2010). This text box, based on the ESTR review of status and trends of seabirds in Canada (Gaston et al., 2009a), is extracted from the report *Canadian biodiversity: ecosystem status and trends 2010* (Federal, Provincial and Territorial Governments of Canada, 2010).

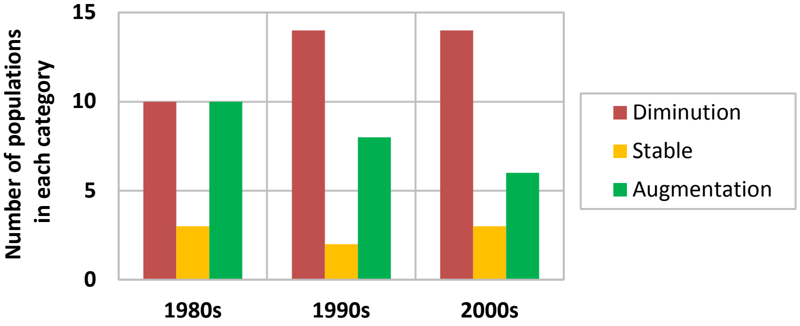


Figure 87. Trends in the status of breeding seabird populations in Canada, 1980s-2000s.
Note: only populations with significant breeding populations, long-term datasets, and those unaffected by terrestrial human activities are included.
Source: adapted from Gaston et al., 2009a

Worldwide, the status of seabirds is deteriorating faster than any other bird group (BirdLife International, 2010). In Canada, trends are regional in nature and result from a variety of factors, including climate change, fishing by-catch, resource extraction, transportation, and pollution (Coe and Rogers, 1997; Melvin and Parrish, 2001; Schreiber and Burger, 2002; Stenseth et al., 2004; ACIA, 2005; Gaston et al., 2009a). A trend to an earlier breeding date has been found in several populations (Bertram et al., 2001; Gjerdrum et al., 2003; Hipfner and Greenwood, 2008), as have changes in diet and condition (Parsons et al., 2008).

With the exception of ivory gulls (*Pagophila eburnea*), which are declining rapidly, change in Arctic seabird populations is slow and possibly the result of events on wintering grounds in the Northwest Atlantic (Gaston, 2002; Gaston et al., 2003). Changes in seabird diet and growth have been found to be related to reduction of Hudson Bay sea ice. This may have negative consequences for populations in the long term (Gaston et al., 2003). Conversely, in the High Arctic, less sea ice may benefit the birds (Gaston et al., 2005; Gaston et al., 2009b).

Waterfowl

Waterfowl (ducks, geese, and swans) are among the most important wildlife species for Inuit for food and ceremonial uses and large concentrations of migrating and nesting waterfowl are a prominent feature of the ecozone⁺. Status and trends of waterfowl are comparatively well known as federal agencies in the USA and Canada monitor populations annually in order to set hunting limits and guide conservation programs. There is limited monitoring on Arctic nesting grounds, but for many species, regular or periodic monitoring is conducted on wintering grounds or at points where the birds congregate during migration. This section describes trends for selected Arctic tundra nesting waterfowl species with substantive parts of their global populations nesting in the Canadian Arctic. In addition, the ecozone⁺ provides important habitat for waterfowl species, such as Canada geese (*Branta canadensis*) and scoters (*Melanitta* spp.) that also nest in more southerly ecozones⁺.

Eiders

Trends

King eiders (*Somateria spectabilis*) nesting in the Canadian Arctic overwinter both in the eastern and western part of the continent. There is growing evidence that the western population of King eiders has declined over past few decades. The total population of king eiders nesting in the western and central Canadian Arctic was estimated in 1960 to be 900,000; in the early 1990s the population had dropped to between 200,000 and 260,000 (Canadian Wildlife Service Waterfowl Committee, 2008). Counts are conducted during spring and fall migration at Point Barrow, Alaska, about every 10 years (Peacock et al., 2013; Quakenbush et al., 2013). The most recent estimate indicates that numbers remained stable between 1996 and 2003 (Figure 88). Surveys on nesting grounds in western Victoria Island conducted in the early 1990s were repeated in 2004 and 2005 (Raven and Dickson, 2006), showing a 56% decline in abundance (Figure 89).

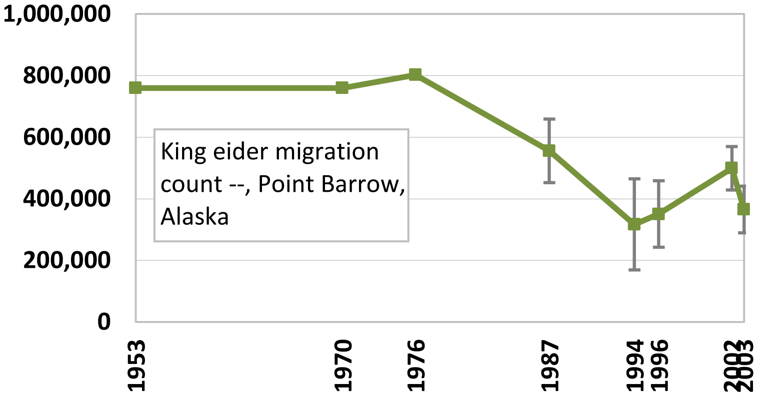


Figure 88. King eiders migration counts at Point Barrow, Alaska, 1953-2003.
The western population of king eiders migrate past Point Barrow, Alaska, each spring and fall to nesting areas in Alaska and western and central Canadian Arctic. Error bars are 95% confidence intervals.
Sources: Suydam et al., 2000 and Quakenbush et al., 2013

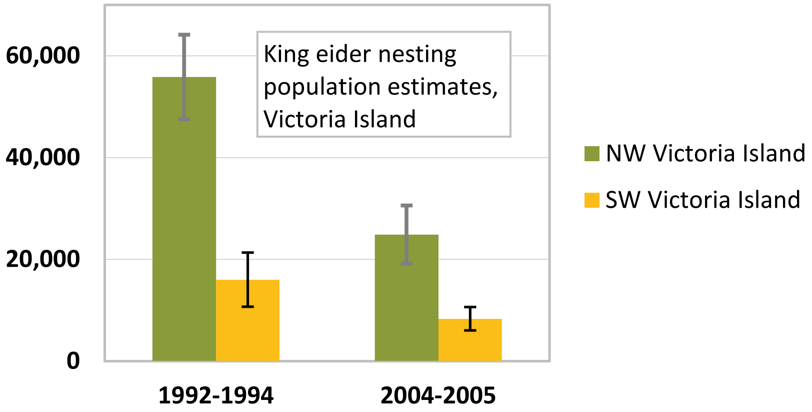


Figure 89. King eider nesting populations, western Victoria Island, 1990s and 2000s.
Error bars are 95% confidence intervals.
Source: Raven and Dickson, 2006

Data on wintering grounds in Greenland indicate that the eastern population of king eiders is also declining, though this decline may be related to a shift in distribution related to human disturbance (Canadian Wildlife Service Waterfowl Committee, 2008). Surveys in 2010 confirmed that large numbers of king eiders were wintering at the northern tip of Labrador and southern tip of Baffin Island (Canadian Wildlife Service Waterfowl Committee, 2012). King eiders nesting in the Rasmussen Lowlands, Nunavut, declined 86% between surveys conducted in 1974/75 and repeated in 1994/95 (Gratto-Trevor et al., 1998).

Three subspecies of **common eiders** (*S. mollissima*) nest in the Canadian Arctic: Pacific (nesting in the western and central Arctic), northern (nesting in the Eastern Arctic), and Hudson Bay.

Numbers of Pacific common eiders in migration counts at Point Barrow, Alaska, declined by 50% between 1976 and 1996, then increased again by 2002 (Figure 90). Areas of particular importance for nesting are the Dolphin and Union Strait, Coronation Gulf, and Queen Maud Gulf. The breeding population for the central Arctic was estimated at about 37,000 in surveys conducted during 1995 to 1998 (Canadian Wildlife Service Waterfowl Committee, 2012). Additional surveys, conducted at nesting areas in Bathurst Inlet to establish a baseline for detection of trends, indicate a decline of about 50% since 1995 (Canadian Wildlife Service Waterfowl Committee, 2012).

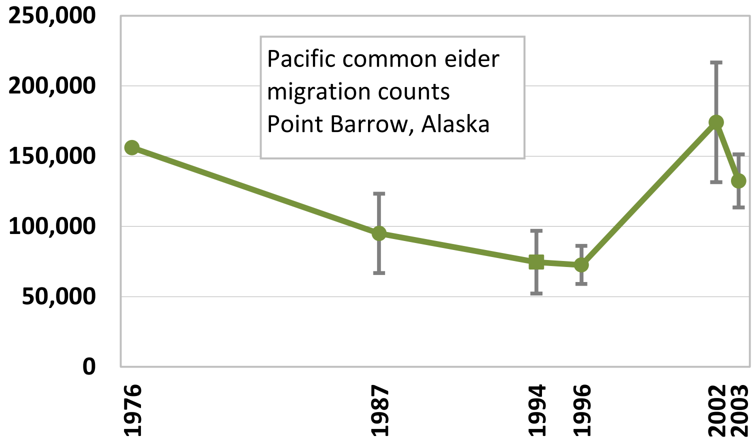


Figure 90. Pacific common eider migration counts at Point Barrow, Alaska, 1976-2003. 95% confidence intervals are shown where available.
Source: Suydam et al., 2000 *and Quakenbush et al.*, 2013

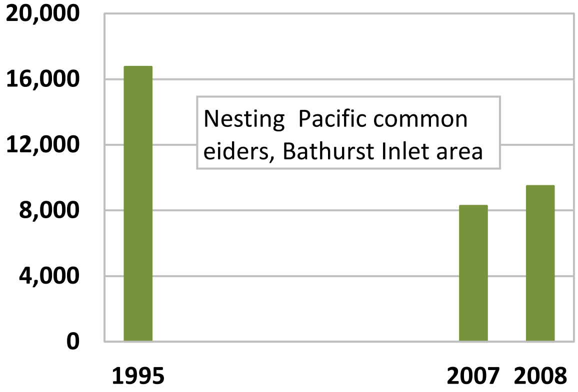


Figure 91. Pacific common eider counts on nesting grounds in Bathurst Inlet area, 1995, 2007, and 2008.
Source: Raven and Dickson, 2009

Northern common eiders winter along the southern coasts of Newfoundland and Labrador, Quebec, and Greenland. Surveys in Greenland, where most overwintering occurs, indicate severe declines since the 1970s (Canadian Wildlife Service Waterfowl Committee, 2012). No clear trends in nesting populations were shown in surveys in Ungava Bay (historical data and surveys in 2000), while surveys along the Labrador coast showed increases between the 1980s and early 2000s. Surveys have been conducted on the wintering range in eastern Canada every third year since 2003. Results to 2009 indicate a stable population of about 200,000 (Canadian Wildlife Service Waterfowl Committee, 2012).

Hudson Bay eiders, wintering entirely in Arctic waters near Belcher Islands and off the western coast of Quebec, are subject to periodic weather-related population crashes (Canadian Wildlife Service Waterfowl Committee, 2008). Belcher Islands breeding eiders declined 70% between 1985–1988 and 1997 (Erickson and Meegan, 2007), probably as a result of a large die-off during the winter of 1991/92 that occurred when areas of open water froze (Robertson and Gilchrist, 1998). There have been no significant winter kill events since then, and the population appears to be recovering (Canadian Wildlife Service Waterfowl Committee, 2012).

Threats: harvest and disease

Declines in some eider populations may be related to harvest. An estimated 115,000 eiders were harvested in 2001 in Chukotka, but the proportion of these that nest in Canada is not known (Canadian Wildlife Service Waterfowl Committee, 2012). The commercial and subsistence harvest of northern common eiders in southwest Greenland was estimated at over 100,000 birds, a large proportion of which are Canadian Arctic nesters (Canadian Wildlife Service Waterfowl Committee, 2008). Demographic modelling indicated that this was not sustainable, and more restrictive harvest regulations were put in place in Greenland from 2002 to 2004 (Canadian Wildlife Service Waterfowl Committee, 2012). Subsistence harvest in Canada is relatively low, but better information on harvest is needed, especially for the northern common eider.

Avian cholera may also be an emerging issue for northern common eiders—the first Arctic outbreak was recorded in 2004 in northern Quebec, and it has recurred there and in the vicinity of Southampton Island (Canadian Wildlife Service Waterfowl Committee, 2012). A small island off Southampton Island, the largest breeding colony of northern common eiders in the Canadian Eastern Arctic, has been studied since 1996. Colony size was stable at around 5,000 pairs between 2001 and 2005, then nearly doubled to an estimated 9,800 pairs in 2006. During an outbreak of avian cholera mortality, colony size dropped to 4,700 pairs in 2007 (Buttler, 2009). Further studies have demonstrated that outbreaks of avian cholera in common eider colonies at a frequency of more than one outbreak per decade could drive colonies to extinction (Descamps et al., 2012).

Tundra swans

The western population of tundra swans (*Cygnus columbianus*) breeds along the Alaska coast and winters in the western United States, while the eastern population breeds from Alaska to the northeast shore of Hudson Bay and Baffin Island, wintering in the eastern United States.

Estimates of the eastern population of tundra swans acquired through midwinter surveys on the U.S. wintering ground (Figure 92) show fluctuations around a mean of about 90,000 to 100,000 birds.

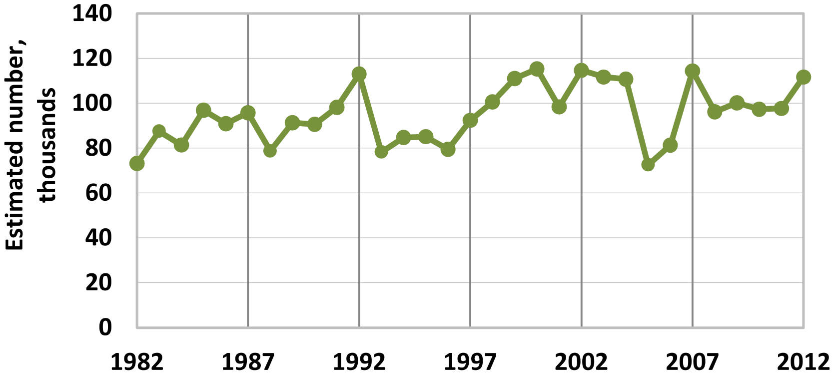


Figure 92. Tundra swan (eastern population) abundance, based on January surveys on wintering grounds, 1982–2012.
Source: U.S. Fish and Wildlife Service, 2012

Snow geese

Snow geese breed only in the Arctic, including along the Hudson Bay coast. Northernmost eastern breeding populations, greater snow geese (*Chen caerulescens atlanticus*), breed in the Canadian Eastern Arctic and northern Greenland and winter along the U.S. Atlantic coast, staging in southern Quebec during migration. More southerly and westerly breeding populations, lesser snow geese (*C. c. caerulescens*), nest in colonies in coastal and inland areas across the Canadian Arctic and winter in southern Canada and all North American flyways.

Greater snow geese increased dramatically from a few thousand birds in the 1930s to over 500,000 in the early 1990s, counted on the main staging grounds in southern Quebec (Canadian Wildlife Service Waterfowl Committee, 2012). The increase is related to changes in agricultural practices in the U.S. wintering grounds leading to a change of feeding habits from marshes to cultivated fields (Canadian Wildlife Service, 2003). The spring 2012 population estimate was about one million geese (Canadian Wildlife Service Waterfowl Committee, 2012). The population has remained fairly stable since 1999 when special conservation measures were implemented (Reed and Calvert, 2007). The dramatic increase in this population has had significant impacts on staging and breeding areas in Canada through overgrazing (Environment Canada, 2007a).

Lesser snow geese also show population growth, with a levelling off of the population, as estimated through the midwinter counts, after the late 1990s and a recent increase (Figure 93). This trend is also evident in inventories on major nesting grounds, with the exception of West Hudson Bay (Figure 93). Inventories provide only a sampling, and banding and harvest studies indicate that the population of lesser snow geese may be much higher than has been estimated—likely exceeding 15 million in 2010 (Canadian Wildlife Service Waterfowl Committee, 2012).

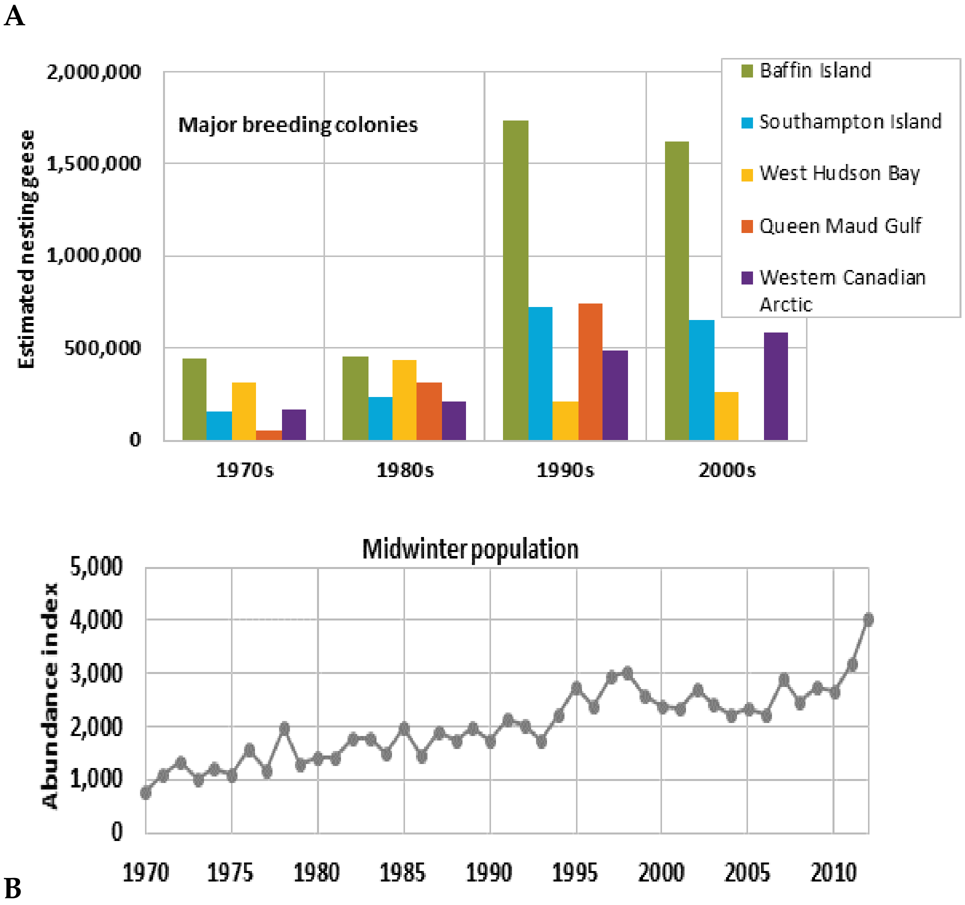


Figure 93. (A) Nesting lesser snow geese estimated through photo-inventories of major breeding colonies 1970s-2000s, and (B) midwinter abundance indices, 1970-2012.
Midwinter counts are primarily of lesser snow geese that breed in the central and Eastern Arctic.
Sources: Canadian Wildlife Service Waterfowl Committee, 2012 (breeding colony data) and U.S. Fish and Wildlife Servic, 2012 (midwinter population counts)

Shorebirds

This section is extracted from the ESTR technical thematic report prepared for this ecozone⁺, *Shorebird trends Arctic Ecozone⁺* (Gratto-Trevor et al., 2011) and has not been comprehensively updated.

The Arctic Ecozone⁺ is of great importance globally for shorebird production. Sixty percent of North American shorebirds breed in the Arctic. The Canadian Arctic alone provides 75% of the North American breeding range for 15 of the 49 species of shorebirds that are common to North America (Donaldson et al., 2000).

Globally, 44% of estimated population trends for Arctic-breeding shorebirds are declining (Figure 94) making the problem more widespread than was originally thought (Morrison et al., 2001). Overall, the Arctic breeders as a group are declining 1.9% per year (Bart et al., 2007).

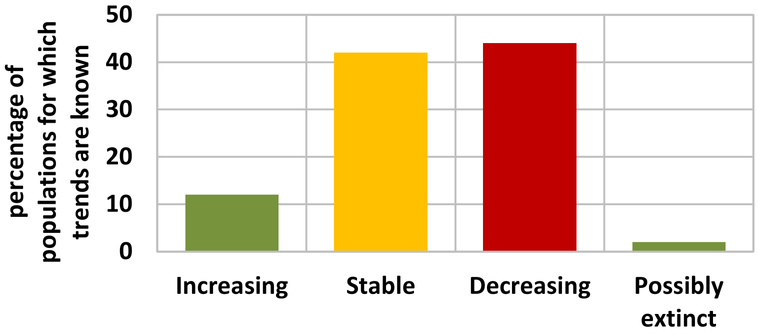


Figure 94. Summary of population trends for Arctic-breeding shorebirds, 2003.
Globally, population trends have been estimated for 52% of Arctic-breeding shorebirds (100 biogeographical populations of 37 species). Of these, 12% are increasing, 42% are stable, 44% are decreasing, and 2% are possibly extinct.
Source: Delany and Scott, 2006

An analysis of fall migration count data was undertaken to determine if the declining numbers of birds recorded on migration counts could be explained by changes in migration routes or timing or by changes in detection rates (Bart et al., 2007). The authors concluded that migration counts most likely reflected a true reduction in population size. They found no evidence of major shifts in the number of birds migrating along specific routes and no major changes in variables related to detection. Annual rates of change were calculated over the period 1974 to 1998 in this study—results are shown in Figure 95 for Arctic-breeding shorebirds with sufficient survey counts in fall migration surveys conducted in the Canadian-United States North Atlantic or United States Midwest regions.

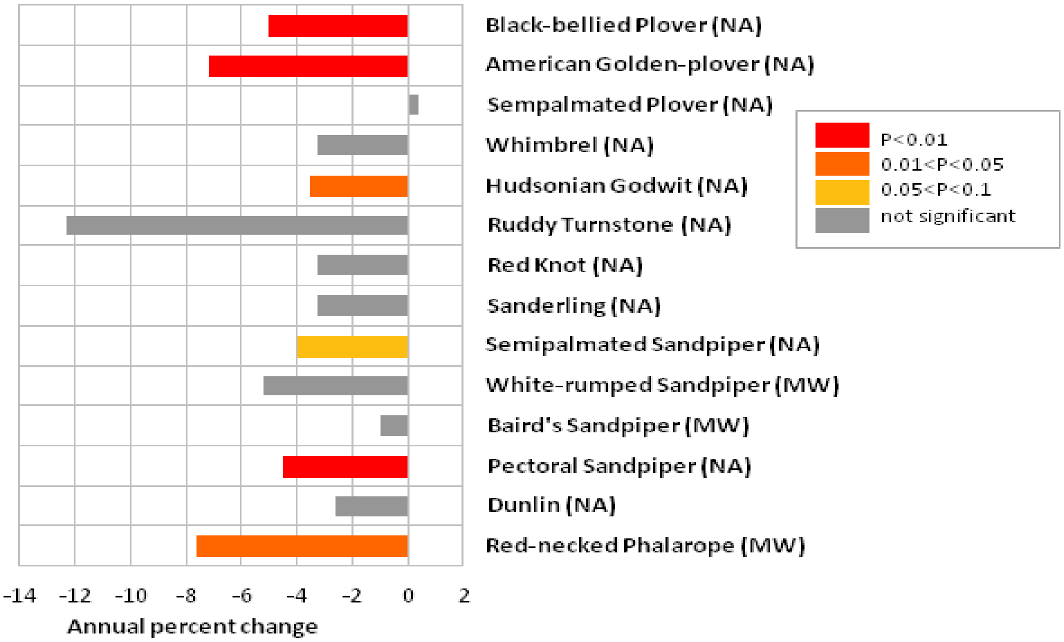


Figure 95. Estimated trends in Arctic-breeding shorebird fall migration counts, 1974–1998.
NA = North Atlantic migration survey; MW = Midwestern migration survey.
Source: data from Bart et al., 2007

Two major shorebird trend reviews by the U.S. Shorebird Conservation Plan Committee (in 2001 and 2004) and Canadian Wildlife Service Shorebird Committee (in 2001) assessed 18 species of Arctic-breeding shorebirds with very similar results (Table 12). Eight species were listed in both assessments as having significant population declines (Brown et al., 2001; Morrison et al., 2001; U.S. Shorebird Conservation Plan, 2004).

Table 12. Population trend assessments for Arctic-breeding shorebirds.

Species	**Trend Summary**[b]	**U.S. Shorebird Conservation Plan (2004)**	**CWS Shorebird Committee**
Black-bellied plover (*Pluvialis squatarola*)	↓↓	Significant decline	Significant decline
American golden-plover (*Pluvialis dominica*)	↓↓	Significant decline	Significant decline
Semipalmated plover (*Charadrius semipalmatus*)	↓?	Not enough information	Significant decline
Eskimo curlew (*Numenius borealis*)	↓↓	Significant decline	Likely extinct
Whimbrel (*Numenius phaeopus*)	↓?	Significant decline	Mixed trends
Hudsonian godwit (*Limosa haemastica*)	↓	Not enough information	Decline
Ruddy turnstone (*Arenaria interpres*)	↓↓	Decline	Significant decline
Red knot (*Calidrus canutus*)	↓↓	Significant decline	Significant decline
Sanderling (*Calidrus alba*)	↓↓	Significant decline	Significant decline
Semipalmated sandpiper (*Calidris pusilla*)	↓↓	Significant decline	Significant decline
White-rumped sandpiper (*Calidris fuscicollis*)	↔	Not enough information	Mixed trends
Baird's sandpiper (*Calidris bairdii*)	↓?	Not enough information	Decline
Pectoral sandpiper (*Calidris melanotos*)	↔	Not enough information	Mixed trend
Purple sandpiper (*Calidris maritima*)	↓?	Stable	Significant decline
Dunlin (*Calidris alpina*)	↓↓	Significant decline	Significant decline
Buff-breasted sandpiper (*Tryngites subruficollis*)	↓	Decline	Decline
Red-necked phalarope (*Phaloropus lobatus*)	↓↓	Decline	Significant decline
Red phalarope (*Phalaropus fulicarius*)	↓↓	Significant decline	Significant decline

[b] ↓↓ significant declining population trend; ↓probable or declining population trend, not statistically significant; ↔ not enough information to conclusively determine population trend (mixed trends); ↓? conflicting information

Trend data are based on many local datasets across North America spanning 1970s–2000s, as well as on expert opinion.

Source: extracted U.S. Shorebird Conservation Plan, 2004; Brown et al., 2001; and Morrison et al., 2001

What is of most concern is that over the past 30 years many species trends have changed from slightly declining to significantly declining, indicating that the decline is persistent and ongoing (Morrison et al., 2001; Delany and Scott, 2006). The declines are observed in species with a range of migration, habitat, and breeding strategies and needs. Preliminary investigations by Thomas et al. (2006a) and Bart et al. (2007) found no common factors among declining species.

In the U.S. Shorebird Conservation Plan (Brown et al., 2001), population trend information was combined with five other variables (relative abundance, threats during breeding season, threats during non-breeding season, breeding distribution, and non-breeding distribution) to create a conservation prioritization scheme. The scheme, adopted in the Canadian Shorebird Conservation Plan (Donaldson et al., 2000), is useful because species with stable or slightly downward-trending populations with threats on their wintering grounds and very specific breeding ground habitat requirements may be more at risk than species with significant population declines. The highest priority species were those designated 'highly imperiled'. Using this prioritization scheme, the only Arctic species listed in 2001 as 'highly imperiled', Eskimo curlew (*Numenius borealis*), is believed to be extinct (Environment Canada, 2007b).

In 2004, species were re-evaluated (U.S. Shorebird Conservation Plan, 2004) and the status of several species was upgraded (Table 13).

Table 13. Conservation status of tundra-nesting shorebirds as classified in the U.S. Shorebird Conservation Plan.

Highly imperiled (first priority)	Species of high concern (second priority)
• Eskimo curlew (*Numenius borealis*) (believed to be extinct) • *Buff-breasted sandpiper (*Tryngites subruficollis*) (globally) • *Red knot (*Calidrus canutus*) (Canadian Arctic-Atlantic Coast population)	• American golden-plover (*Pluvialis dominica*) (globally) • Whimbrel (*Numenius phaeopus*) (North American populations) • Hudsonian godwit (*Limosa haemastica*) (globally) • Ruddy turnstone (*Arenaria interpres*) (North American populations) • Red knot (*Calidrus canutus*) (populations other than the Canadian Arctic-Atlantic Coast population) • Sanderling (*Calidrus alba*) (North American populations) • *Dunlin (*Calidris alpina*) (Alaska-East Asian and Alaska-Pacific Coast populations)

Upgraded species are denoted with an asterisk (*)

Source: U.S. Shorebird Conservation Plan, 2004

Local studies have recorded population declines over a range of periods. Analysis of the Atlantic coastal migration stop-overs from 1972 to 1983 (Howe et al., 1989) found significant declines for black-bellied plover (*Pluvialis squatarola*) (decreasing by 5.4% per year), whimbrel (*Numenius phaeopus*) (–8.3% per year) and sanderling (*Calidrus alba*) (–13.7 per year). Breeding populations of red phalarope (*Phalaropus fulicarius*), black-bellied plover, and American golden-plover (*Pluvialis dominica*) decreased significantly, by 76, 87, and 79% respectively, in the Rasmussen Lowlands (Central Arctic) over a 20-year period (Gratto-Trevor et al., 1998). Given the long time interval between studies, natural fluctuation as a result of a series of poor breeding seasons rather than a persistent and continuous population decline could explain the differences between the two study periods, but it may represent a true decline in these species (Gratto-Trevor et al., 1998).

A study in the Foxe Basin (Prince Charles and Air Force islands) found significant population declines for white-rumped sandpiper (*Calidris fuscicollis*) (–61%) and red phalarope (–43%) over an eight-year time span (1989 to 1997) (Johnston and Pepper, 2009). For red phalarope the decline was even more pronounced at East Bay, Southampton Island, where there was a 93% decline over six years (1999 to 2005) (Pirie et al., 2012). All shorebird species (n=5) at East Bay declined by more than 90% over the same interval. In 2007, there was a small rebound to about 33% of the original 1999 values. This coincided with a high lemming (and therefore low predation) year (Pirie et al., 2012).

Near Churchill, Manitoba, a comparison of six qualitative bird abundance studies between 1930 and the 1990s found that semipalmated sandpiper (*Calidris pusilla*), stilt sandpiper *(Calidris himantopus*), and red-necked phalarope *(Phalaropus lobatus*) experienced a 'great decrease', and dunlin (*Calidris alpina*) a 'decrease' (Jehl and Lin, 2001). Huge declines were also noted at La Perouse Bay, Manitoba (40 km east of Churchill), for semipalmated sandpiper and red-necked phalarope (Gratto-Trevor, 1994).

One of the current major limitations to determining population trends for Arctic-breeding shorebird species is the lack of reliable population estimates. In many cases intensive surveys of shorebirds on the Arctic breeding grounds have led to increases to the world population estimate for a given species (Johnston et al., 2000; Latour et al., 2005; Johnston and Pepper, 2009). This does not reflect an increase in world population size but instead is an indication that initial population estimates were probably low (Brouwer et al., 2003; Morrison et al., 2006). The large-scale Program for Regional and International Shorebird Monitoring (PRISM), which has an Arctic component, is partway through a multi-year survey program that will produce continental population estimates for 19 species of shorebirds that breed in the North American Arctic. Once the first pass of surveys is complete, a second set is planned to assess species-specific as well as North American Arctic-wide population trends (Skagen et al., 2003; Bart and Earnst, 2004; Bart et al., 2005; Bart and Johnston, 2012).

Proposed causes of shorebird population declines include: loss of migration stop-over sites, loss of wintering habitat, and life history characteristics (migratory behaviour, life history, and biogeography) which may predispose shorebirds to population decline. Future population decline is expected to be accelerated by habitat changes on the Arctic breeding grounds.

Since many shorebirds are long-distance migrants that tend to gather in very large numbers at relatively few sites, loss of one or two major stop-over sites could have a huge effect on shorebird populations. Declining food availability at existing stop-over sites can also have a large impact on populations because birds may not be able to take in enough fuel to move to the next stop-over site, or may not be able to acquire the body stores essential for survival and successful reproduction (Senner and Howe, 1984; Donaldson et al., 2000; Morrison et al., 2001; Baker et al., 2004; Morrison et al., 2004; Morrison et al., 2007). Analysis of population trends of North American shorebirds found species that followed continental migration routes (as opposed to coastal or oceanic migration routes) were at higher risk of population decline because of ecosystem loss and alteration (Thomas et al., 2006a; Bart et al., 2007). Continental migrants use small, ephemeral ponds and wetlands that are scattered over a large area. These ponds and wetlands are difficult to delineate for conservation initiatives making it harder to preserve them as compared to larger stop-over sites (Thomas et al., 2006a). Little is known about Arctic stop-over sites because of their remoteness. Observations along a 200 km stretch of coast line in the Kivalliq Region (northwestern Hudson Bay) during the 2008 spring migration found hundreds of High Arctic nesting migrants feeding on insects in the wrack lines on their journey north to the breeding grounds (Johnston and Rausch, unpublished data). The importance of sites such as these to migration and subsequent breeding success is not known.

Loss or degradation of habitat on the non-breeding grounds from human activities such as oil pollution (Harrington and Morrison, 1980), mechanical dredging or fishing (Piersma et al., 2001), conversion of native grasslands and wetlands to agriculture (Isacch and Martinez, 2003; Shepherd et al., 2003), and tourism and development on marine beaches (Blanco et al., 2006) may be a cause of population decline (Thomas et al., 2006a). Complicating the assessment of the importance of wintering habitat is that little is known about food resources on the wintering grounds (Morrison et al., 2004). Threats on the wintering grounds, however, have been found to have a weak influence on the likelihood of a species being in population decline (Thomas et al., 2006a).

The intrinsic biology of shorebird species may make them more susceptible to population decline. Migratory behaviour (such as distance and routes) is suspected to be the most influential intrinsic factor, with more continental migrants in population decline than coastal or oceanic migrants (Thomas et al., 2006a). Phylogenetic characteristics such as body and clutch size, lifespan, and relatedness were found to be unimportant to population decline, but limited clutch sizes means that recovery following a decline is likely to be slow (Myers et al., 1987). Sexual selection may have an influence on declining populations since most socially polygamous species have declining populations while socially monogamous species have stable or increasing population trends—but the data are not conclusive. There are no clear intrinsic factors held in common by shorebird species with declining population trends and extrinsic factors are more likely to be the primary cause of decline (Thomas et al., 2006a; Thomas et al., 2006b; Bart et al., 2007).

Habitat changes in the Arctic caused by climate change are expected to have an exacerbating effect on the declining population trends of Arctic-breeding shorebirds (Bart et al., 2007). Arctic-breeding shorebirds are adapted to the annually variable weather conditions of the Arctic during the breeding season. However, their conservative life-history strategy (low reproduction and long lifespan) makes it difficult for them to adapt to climate change. This puts Arctic-breeding shorebirds more at risk of population decline than other groups (Donaldson et al., 2000; Meltofte et al., 2007). Effects of climate change on breeding habitat include: drying of tundra ponds (Walsh et al., 2005; Smol and Douglas, 2007a), shrub encroachment (Callaghan et al., 2005a), and asynchrony of insect-chick hatch (Tulp and Schekkerman, 2006).

The synchrony of shorebird chick hatch with the peak of insect emergence is not as critical as hatch occurring when there is sufficient food supply. The availability of the food supply is strongly influenced by weather and a sufficient supply is only available for 40% of the insect season (Tulp and Schekkerman, 2008). The peak date of insect emergence fell between 8 July and 23 July for 75% of the 33-year study period. These earliest and latest peak emergence dates were recorded in consecutive years, showing that the date of peak emergence is not advancing linearly with time. Overall, however, the date of peak insect emergence as well as the range of dates with sufficient food available for the normal growth of chicks is getting earlier in the season (Tulp and Schekkerman, 2008).

Since Arctic shorebirds time nest initiation to occur as soon as the snow melts, the advancement in the timing of insect emergence may not be as critical for the survival of chicks hatched from the earliest nests. It could be a serious problem for chicks from late nests, or from re-nests (clutches laid late to replace an earlier nest that was unsuccessful) because they will hatch too late in the season to obtain sufficient food resources (Meltofte et al., 2007). Studies indicate that, while shorebirds exhibit considerable flexibility in nest initiation, the accommodation in timing may not completely match the change in date of snowmelt, especially in years with exceptionally early springs (Smith et al., 2010; McKinnon et al., 2012; Grabowski et al., 2013). Further analysis is needed to determine if snowmelt is advancing at the same rate as the timing of insect emergence, which would permit birds to nest earlier. It is not known whether shorebirds will be able to adjust their migration strategies to arrive on the breeding grounds sooner in response to an earlier snow-free season. Species which make the final jump to the breeding ground from latitudes closer to the Arctic may be more successful than species that use internal length-of-day cues to initiate migration from very distant wintering grounds (Tulp and Schekkerman, 2008).

Landbirds

This report has been extracted from the ESTR thematic technical report *Landbird trends in Canada, 1968-*2006 (Downes et al., 2011) and has not been updated with new information.

The Arctic Ecozone$^+$ is relatively pristine and there are few immediate threats to landbirds from human activity, although species are affected by climate change, contaminants, and other wide-ranging factors. All birds listed in Table 14 overwinter in more populated areas of Canada and the United States where development pressures are more intense in both their wintering ranges and along their migration routes. Canada has a high stewardship responsibility for these species because large portions of their Western Hemisphere breeding populations are concentrated in the Arctic Ecozone$^+$.

There are relatively few landbird species in the Arctic Ecozone$^+$ and few data on their population trends. The lack of information on population status and trends has been highlighted as the most pressing conservation need in relation to landbirds for this region (Rich et al., 2004). Because of the remoteness and lack of roads, the Breeding Bird Survey (BBS) has not been carried out in the Arctic and there are few other surveys of breeding birds. However, many birds that breed in the Arctic spend their winters in the United States and more southerly parts of Canada, where their populations can be monitored by the Christmas Bird Count (CBC). The CBC, now over 100 years old, monitors the status and trends of winter bird populations through an all-day, annual census conducted by groups of volunteers throughout North America. Data from the CBC complement the BBS by providing results for some species that cannot be monitored on their breeding grounds. Results presented below are preliminary findings based on CBC data from Canada and the United States combined (Butcher and Niven, 2007).

Table 14. Trends in annual abundance of selected Arctic Ecozone$^+$ landbirds, 1966-2006, based on Christmas Bird Count results for North America.

Species	**Population Trend (%/yr)**	**P**	**CBC Abundance Index**				
			1970s	**1980s**	**1990s**	**2000s**	**Change**
Hoary redpoll (*Acanthis hornemanni*)	–4.97%	*	0.29	0.18	0.14	0.09	-68%
American tree sparrow (*Spizella arborea*)	–2.16%	*	62.8	56.3	42.4	34.4	-45%
Harris's sparrow (*Zonotrichia querula*)	–2.13%	*	9.6	7.5	6.2	5.3	-45%
Snowy owl[1] (*Bubo scandiacus*)	–2.12%	*	0.24	0.17	0.14	0.11	-53%
American pipit (*Anthus rubescens*)	–0.97%	*	5.9	4.8	4.4	4.7	-19%
Snow bunting (*Plectrophenax nivalis*)	–0.93%		15.8	14.4	11.6	9.3	-41%
Rough-legged hawk (*Buteo lagopus*)	–0.06%		1.8	1.6	1.6	1.7	-7%
Lapland longspur (*Calcarius lapponicus*)	0.40%		0.9	0.9	0.9	1.0	12%
Common redpoll (*Acanthis flammea*)	0.60%		19.0	17.8	18.1	17.9	-6%

[1] This reflects sightings of snowy owls in southern latitudes. Note, however, that many snowy owls remain the in the Arctic or northern taiga throughout the winter, far from human settlements.

Table shows the annual rate of change and the average CBC abundance index by decade

Asterisks (*) indicate significant trends ($P<0.05$)

"Change" is the percent change in the average index abundance between the first decade for which there are results (1970s) and the 2000s decade (2000–2006).

Source: Downes et al., 2011

CBC trends (Table 14 and Figure 96) suggest that several species such as Harris's sparrow (*Zonotrichia querula*) and snowy owl, have been undergoing long-term, statistically significant declines since the 1960s. Other species, such as the rough-legged hawk (*Buteo lagopus*) and Lapland longspur (*Calcarius lapponicus*), have shown relatively stable overall population trends.

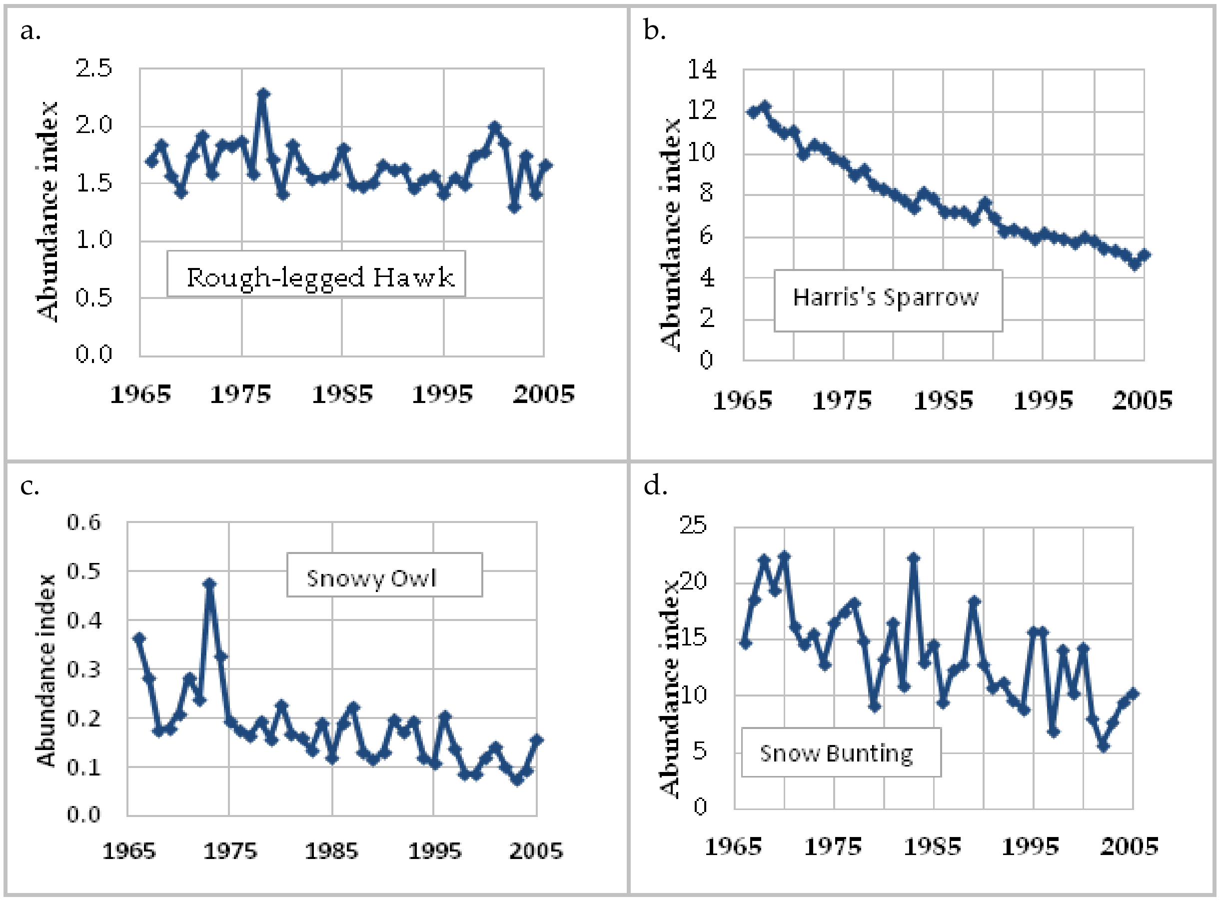


Figure 96. Annual indices of population change for the rough-legged hawk, snowy owl, Harris's sparrow, and snow bunting, 1966-2005, based on Christmas Bird Count results for North America.
The rough-legged hawk and snow bunting show no significant trends, though the latter may be in decline; the snowy owl and Harris's sparrow have declined significantly (P<0.05). See the note on snowy owl for Table 14.
Source: Downes et al., 2011

Harris's sparrow, a species with its entire breeding range in Canada, is classified by Partners in Flight as a Continental Watch List species (Rich et al., 2004). The species has apparently experienced a long-term decline over the last 40 years (Figure 96b). Because of its isolated breeding range, direct influence of human activity on the breeding range is unlikely to be a factor in its decline. Harris's sparrows, however, are susceptible to predation, especially by merlins (*Falco columbarius*), whose populations are increasing. The influence of factors such as climate change is unknown (Niven et al., 2004; Norment and Shackleton, 2008).

Population indices for the snow bunting (*Plectrophenax nivalis*) vary annually but this species has apparently experienced a large decline in its population over the long term (Audubon Society, 2007) (Figure 96d). The Arctic has a very high stewardship responsibility for snow buntings (Rich et al., 2004), which breed throughout the Arctic Cordillera and Northern Arctic and the northern portions of the Southern Arctic. Causes of the apparent snow bunting decline are not known and further demographic studies are needed. Potential factors affecting populations on the breeding grounds include availability of nest sites (rock talus or cavities, driftwood cavities, or sometimes buildings), predation on nests and incubating females, and availability of food (Lyon and Montgomerie, 2011). Reduction in snow bunting populations may be related to earlier thawing of the tundra, which allows more woody plants to grow, converting the open foraging habitats preferred by snow bunting to more shrub-dominated communities (Audubon Society, 2007).

Insects and pathogens

Warmer temperatures will benefit free-living bacteria and parasites whose survival and development is limited by temperature and there are indications that this is occurring in some Arctic muskox and caribou populations (see section on Wildlife diseases and parasites on page 60). Arthropods such as ticks that transmit disease agents may also benefit from climate change, and the diseases they spread may consequently become more prevalent or widespread. Other abundant flying insects in the Arctic, especially oestrids (warble flies) and mosquitoes, are a significant factor in caribou ecology and their abundance is closely related to climate conditions. Inuit from the Queen Elizabeth Islands report that mosquitoes and black flies have increased along with warmer weather in certain areas (e.g., Nunavut Tusaavut Inc., 1997).

Major range shifts of species native to Canada

Arctic residents have reported changes in animal behaviour and distribution. Indigenous people of the Arctic especially are familiar with the long-term trends, variability, and extent of species distribution, and habitat use and behaviour. As they travel and work in their lands, they observe changes that cannot be easily detected through other monitoring and research. These observations have been documented through several studies using interviews, questionnaires, and group discussions, but there is not a systematic methodology or repository for this information. See Table 15 for examples of reported observations of range shifts.

Table 15. Examples of documented observations by Inuit regarding range shifts of species native to Canada.

Location	**Observations that may indicate species range shifts**	**Reference**
Nunatsiavut	See new insects and strange kinds of birds: 2001 small yellow and red birds came in large groups that ate anything including seal skin and seal fat. Hummingbirds seen (new species).	Nickels et al. (2005) Nunatsiavut workshop
Nunavik	Deer were seen in the summer of 2002 for the first time around the community of Kangiqsujuaq.	Nickels et al. (2005) Nunavik workshop
Chesterfield Inlet	Insects, birds and even grizzly bear were reported to be appearing farther north than usual.	Nunavut Research Institute (2004)
Kitikmeot	New birds seen for the first time such as the robin and an unidentified yellow songbird. More abundant and new species of shrubs and lichens.	Thorpe (2000)
Baker Lake	At least ten new kinds of insects were reported to be seen in the area, all winged insects, some recognized from the treeline area.	Fox (2004)
Banks Island	New bird species observed include robins and barn swallows. Shorter winters, longer summers and more water were thought to have caused an increase in the number of insects and led to the arrival of new species of beetles and sand flies. Changes in behaviour include overwintering of birds that normally migrate.	Ashford and Castleden (2001)
Beaufort coast	Ravens and eagles more numerous further north including Tuktoyaktuk.	Gordon et al. (2008)
Western Inuvialuit Settlement Region	Cougars are new to the treeline areas.	Nickels et al. (2005) Inuvialuit workshop

Root et al. (2003) reviewed 143 journal articles related to changes in species distribution (globally) and found that 80% of species show changes that can be explained by physiological constraints. These included changes in density at a particular location, changes in range either poleward or up in elevation, changes in timing of migration, changes in phenology (plant growth and flowering), change in the timing of egg laying, changes in morphology (body size), and shifting frequencies of genetic markers. Many of the species with documented range changes were birds and butterflies, which can more easily adjust their distribution than other taxa (for example, small mammals) that may be constrained by geological features such as rivers.

Climate-related changes in tundra ecosystems—including reduction of lichens, increase in shrubs, and increased canopy height (see section on Changes in tundra plant communities on page 96)—underlie observed and anticipated changes in distribution of flora and fauna in the Arctic Ecozone^{+} (Gilg et al., 2012; Reid et al., 2013). More rapid changes may be caused by human introductions— either inadvertently or as deliberate moves to establish populations—for example, the introduction and re-introduction of muskoxen to some regions (see section on Muskoxen on page 128). Land-use change and disturbance from industrial activity and other human activities may also lead to changes in animal distribution.

Range extensions due to climate change could show up as previously vagrant/accidental species that appear more and more frequently and finally become established. Such extensions could involve boreal species moving into the Southern Arctic, or species from that zone moving into the Northern Arctic. Detecting such changes in status requires significant effort in well-designed monitoring programs.

Mammals that have experienced relatively recent range shifts include grizzly bears, which have moved northward in some areas (see the section on Trends in species of conservation concern, on page 103) and moose, which have also expanded their range northward (see text box below). Red foxes have expanded their distribution in parts of the Canadian Arctic and not in others. The current, best-supported hypothesis is that red fox expansion in the 20th century in Arctic Canada was promoted by human food supplementation, and that such food supplementation could lead to similar effects elsewhere if not checked (see the discussion on this topic under Major human stressors on ecosystem structure on page 99).

Expansion of moose northward

Moose (*Alces alces*) are found primarily in shrub habitats south of the Arctic Ecozone$^+$ but they have been increasing in recent decades in much of the Arctic Ecozone$^+$. Expansion of moose ranges can have an impact on populations of predators and of other ungulates. Moose can provide alternate prey for predators, and alternate food sources in Arctic subsistence economies, especially when caribou are scarce. Wolf populations in caribou winter ranges (in the taiga ecozones$^+$) can increase in response to higher levels of other prey such as moose. When caribou return to the winter range they are preyed on more heavily by the increased number of wolves (e.g., Basille et al., 2013).

Evidence of changes in moose distribution includes:

- Expansion of moose range northward into suitable habitat has been documented for the Northwest Territories and Nunavut, with more frequent sightings of moose in shrub-rich tundra regions, especially since the 1970s. In the NWT, moose have been sighted as far north as near Bathurst Inlet, Coronation Gulf, and the east side of Victoria Island (NWT Department of Environment and Natural Resources, 2012).
- Although not historically resident in Nunatsiavut, moose have increased in numbers and range over the past four decades. Moose hunting commenced there in 1977. The northward range expansion of moose reached the treeline in Nunatsiavut in the 1990s (Chubbs and Schaefer, 1997). Inuit report that moose have moved north of Nain and to the Voisey's Bay area, not far south of the Arctic Ecozone$^+$ (Nickels et al., 2005 Nunatsiavut workshop).
- Moose from the British–Richardson Mountains of the Yukon (in the Taiga Cordillera Ecozone$^+$) probably began frequenting the Yukon North Slope coastal plain within the last 100 years (Wildlife Management Advisory Council (North Slope), 2008a), utilizing mainly riparian shrub zones. Surveys in the 1980s showed that most of the moose summering along the coastal plain migrated about 100 km south during the winter—but a survey in 2000 (Smits, 2000) as well as observations by Inuvialuit (Arctic Borderlands Ecological Knowledge Co-op, 2003) indicate that increasing numbers are wintering in the Arctic Ecozone$^+$ section of their range . Moose in the northern part of their range in the Yukon are increasing: counts in the Richardson Mountains/coastal plains surveys increased by 67% from 1989 to 2000 (Smits, 2000).

Human stressors on ecosystem composition

Climate change

Climate change is currently a human-induced stressor for some species, such as polar bears, and is thought to contribute to changes in others, including shorebirds and caribou. Implications of climate change are dealt with throughout this report.

Human population increase

Despite the low density of human population in the Arctic Ecozone^{+}, human settlements and associated infrastructure affect ecosystem composition through habitat change, disturbance, and harvest. Increasing human populations place stress especially on large mammals. For example, disturbance to female polar bears during denning, or prior to denning if it alters their choice of dens, may affect growth of cubs (Lunn et al., 2004). Disturbance at den sites may reduce productivity and alter the distribution of tundra wolves (Walton et al., 2001).

The reverse is also true: decreasing intensity of human use may result in increasing wildlife populations. For example, muskox populations in the Queen Maud Gulf mainland and islands increased following withdrawal of Inuit families to more distant settlements in the 1950s. Barren-ground caribou, lesser snow goose, and Ross's goose (*Chen rossii*) populations have also rapidly increased in the area since the late 1960s (Gunn et al., 1984).

Contaminants and pollution

Sources of Arctic wildlife contamination include long-range transport of persistent organic pollutants (POPs) and mercury, mines, DEW line sites, transportation-related emissions, and at least one military base in addition to the DEW line sites. An emerging issue is potential future contamination related to the anticipated increase in marine traffic and oil and gas activity in the Arctic Ocean as the ice melts.

A local source of pollution, with the potential for transfer of contaminants to wildlife, is liquid and solid waste disposal in Arctic communities. For example, Nunavut dumps are seldom contained and waste is often burned in an uncontrolled manner (ARKTIS Solutions, 2011). Wastewater from small communities is stored in open lagoons that may lack liners. In summer, some communities decant the liquids from the lagoons onto the surrounding ground, into lakes, and into the sea (Jamieson and Krkosek, 2013).

Contaminants in wildlife

The concentration of toxic contaminants in wildlife has been a concern in Canada since the 1970s and monitoring has been ongoing since then for some species and some compounds. In the late 1980s concerns about elevated levels of contaminants in wildlife species that were important to the traditional diets of northern Aboriginal peoples led to increased monitoring and research in northern Canada. Early results found a wide variety of substances, many of which had no Arctic or Canadian sources, but which were, nevertheless, reaching unexpectedly high levels in Arctic ecosystems primarily through long range atmospheric transport (INAC, 2008).

Air currents transport several types of contaminants, for example, PCBs and mercury, to the Arctic from southern, more populated and industrial parts of the world. When the airborne pollutants arrive in the Arctic, cold temperatures cause the air, and the pollutants, to fall lower in the atmosphere and to be deposited on the land and into aquatic environments. The contaminants are transported through soil and water, incorporated into biota, and often concentrated as they move up the food chain. Many of these substances become concentrated in particular tissue types, especially fat, or in particular organs such as livers. In general, the highest contaminant levels tend to build up in marine fish-eating animals or their predators, especially those that are long-lived such as whales and polar bears.

Concentrations of all contaminants in wildlife vary depending on the individual, the species, and the location. Trends for some animal populations are shown in Table 16. Three classes of contaminants are of concern in the Arctic Ecozone$^+$ (Stow, 2008):

1. Concentrations of persistent organic pollutants (POPs), such as the pesticide dichlorodiphenyltrichloroethane (DDT), polychlorinated biphenyls (PCBs) and toxaphene, have generally declined (Riget et al., 2010). In long-lived species such as whales, decline of contaminant levels has been slower.
2. Brominated flame retardants [e.g., polybrominated diphenyl ethers (PBDEs) and fluorinated surfactants, perfluorooctane sulfonate (PFOS)] are examples of toxic contaminants that have been increasing since the mid-1980s in most locations and some species [e.g., ringed seals (*Pusa hispida*); see the ESTR report *Ecosystem status and trends report: Arctic marine ecozones* (Niemi et al., 2010)].
3. Increases in mercury have been observed since the mid-1970s in some Arctic marine mammals, seabirds, and fish around the circumpolar Arctic, with the highest proportion of increases being in Canada and Greenland (Riget et al., 2011). There were far fewer records for terrestrial mammals, and few are of adequate length to determine trends (Riget et al., 2011). While some of the mercury found in wildlife is from natural sources, much of the mercury found in marine and aquatic systems is from industrial sources.

Table 16. Trends in mercury and persistent organic pollutants from Northern Contaminants Program datasets.

Trend	Measurement taken from…	Contaminant	Time Frame	Reference
⇧	Caribou (Bluenose East Herd)	Mercury	1994–2002 increase, decline 2002-2006	Gamberg (2008)
⇧	Caribou (Porcupine Caribou Herd)	Mercury	1993 to 2007 (slight increasing trend)	Gamberg (2008)
⇧	Caribou (Bathurst Herd)	Mercury	1992–2006	Gamberg (2008)
↔	Charr muscle (Amituk Lake, Resolute)	Mercury	1989–2007	Muir (2008)
⇩	Charr muscle (Amituk Lake, Resolute)	PCBs, DDTs, toxaphene	1989–2007	Muir (2008)
↔	Charr muscle (Amituk Lake, Resolute)	Mercury	1993–2007	Muir (2008)
⇔	Charr muscle (Amituk Lake, Resolute)	PCBs	1993–2007	Muir (2008)
⇩	Charr muscle (Amituk Lake, Resolute)	PCBs, DDTs, toxaphene	1993–2007	Muir (2008)
⇩	Charr muscle (Lake Hazen, Quttinirpaaq National Park)	PCBs, DDTs, toxaphene	1990–2007	Muir (2008)
↔	Charr muscle (Lake Hazen, Quttinirpaaq National Park)	Mercury	1990 - 2007	Muir (2008)
⇔	Charr muscle (Amituk Lake, Resolute)	Mercury	1993–2007	Muir (2008)
⇔	Charr muscle (Amituk Lake, Resolute)	PCBs	1993–2007	Muir (2008)
⇩	Polar bear (Western Hudson Bay)	DDTs	1991–2008	Letcher (2008)
⇔	Polar bear (Western Hudson Bay)	PCBs	1991–2008	Letcher (2008)

Legend

⇧	Significant increasing trend
⇩	Significant decreasing trend
⇔	No significant trend
↔	No significant trend but indications of possible change – more data needed to draw conclusions

For trends in marine biota, see the Arctic Marine report.

Source: Adapted from an analysis for ESTR by Stow, 2008, based on Gamberg, 2008, Muir, 2008 and Letcher, 2008

Based on a review of research on ecological effects of POPs and metals, Fisk et al. (2005) found that concentrations in Arctic terrestrial wildlife, fish, and seabirds are generally below effects thresholds, with a few possible exceptions. These were PCBs in burbot (*Lota lota*) in some Yukon lakes (in the Boreal Cordillera Ecozone$^+$), Greenland shark (*Somniosus microcephalus*), glaucous and great black-backed gulls (*Larus hyperboreus* and *L. marinus*), and dioxin-like chemicals in seabird eggs. PCB and DDT concentrations in several Arctic marine mammal species exceed effects thresholds, although there was no evidence of stress in these populations. Some polar bears and beluga whales have high enough levels of contaminants that they may experience sublethal effects such as immune suppression. Papers reviewed found weak relationships between cadmium, mercury, and selenium burdens and health biomarkers in common eiders in Nunavut, although these metals were probably not influencing the health of these birds. The authors concluded that there is little evidence that contaminants are having widespread effects on the health of Canadian Arctic organisms, with the possible exception of polar bears. However, they recommended that further research and better understanding of contaminant exposure in Arctic biota is needed considering factors such as tissue levels that exceed effects thresholds, exposure to "new" organic contaminants of concern (such as brominated flame retardants), contaminated regions, and climate change (Fisk et al., 2005).

Sources of local and regional pollution

Current and past activities in the Arctic result in potential for both contamination of wildlife, affecting their safety as human food, and effects on the animals themselves. Overall, the Arctic has relatively few sources of local pollution, with little industry and a very low population density. Sources of pollutants include mining activity, especially abandoned mines, military activities, especially DEW line sites (see text box The Distant Early Warning (DEW) Line in the introductory section on page 11), and oil and gas activities.

An example of localized contamination from a still-active radar station is Saglek Bay, Nunatsiavut (Labrador), which had been the site of a military radar station since the late 1950s. PCB contamination in soils and leaching into the marine environment was discovered in 1986 and cleanup was initiated in 1997–1999. Sediments, fish, and seabirds (black guillemots, *Cepphus grylle*) have accumulated high levels of PCB concentrations due to marine sediment contamination (Kuzyk et al., 2003; Kuzyk et al., 2005).

In 2005, two sites in the Arctic Ecozone$^+$ had been designated as "contaminated" by the federal contaminated sites program: Kittigazuit Military Site and Atkinson Point Military Site (INAC, 2005), both within the Inuvialuit Settlement Region.

ECOSYSTEM GOODS AND SERVICES

Provisioning services

Living space

More so than perhaps any other group in Canada, modern Inuit and Inuvialuit occupy and use huge tracts of land as they travel to neighbouring communities to visit and conduct business and to remote (often traditional) camp sites to hunt, fish, and trap (Damas, 2002). "Travel" in this context means not only travel by dogsled, snow machine, and boat, but also the charter of aircraft to visit, confer with, and exchange goods with friends, relatives, and colleagues in remote communities. Many communities and individual families regularly occupy several traditional hunting, trapping, and fishing camps far from their primary homes for long periods at different seasons. These annual events are important sources of social cohesion. As with the capture and consumption of country foods (see below), use of space is integral to Inuit and Inuvialuit culture and tradition (Condon et al., 1995).

Ice and snow form an important part of this living space. Travel over river, lake, and sea ice provides access to the land for hunting and fishing and the main corridors for regular transport of goods and people over much of the year.

Permafrost is another important component of living space. It provides an unquantifiable service by supporting building structures; when it unexpectedly thaws, structures built on it collapse. Structures such as homes must be insulated to prevent them from thawing the permafrost beneath them, and this is an additional (quantifiable) cost; its presence, however, allows structures to be built on land that otherwise would be unusable for habitation, a definite benefit. The reduction in extent of permafrost and the increase in the depth of annual thawing that may accompany climate change are present risk factors and future threats.

Food

Detailed, quantitative studies since the early 1970s (Usher, 1976; Berger, 1977) through the 1980s (Gunn et al., 1986) and to the present (Wein et al., 1996; Helander-Renvall, 2005; Inuuvik Community Corporation et al., 2006) show that Inuit and Inuvialuit rely heavily on traditional foods (also called country foods): wild-caught fish and wildlife and collected plants such as berries. Harvest of country foods is not merely a matter of calories and nutrients in a subsistence economy, but a central feature of Inuit and Inuvialuit cultural identity.

Extent and patterns of subsistence activities were included in the Survey of Living Conditions in the Arctic (SLiCA), an interview-based study of Inuit around the Arctic, conducted in Canada in 2001 in cooperation with Statistics Canada, including all four major Inuit groups, and with a total sample size of 4,700 interviews (Kruse et al., 2009). Results were very consistent across Canadian Inuit groups and show the high rates of participation in fishing and hunting, with far fewer Inuit being active trappers (Figure 97). As shown in Figure 98, participation rates in subsistence activities are higher for people who live outside of major population centres of the Arctic, especially in Western Arctic.

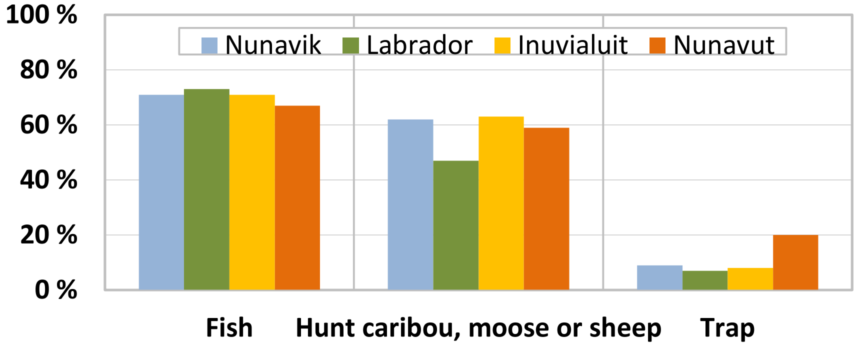
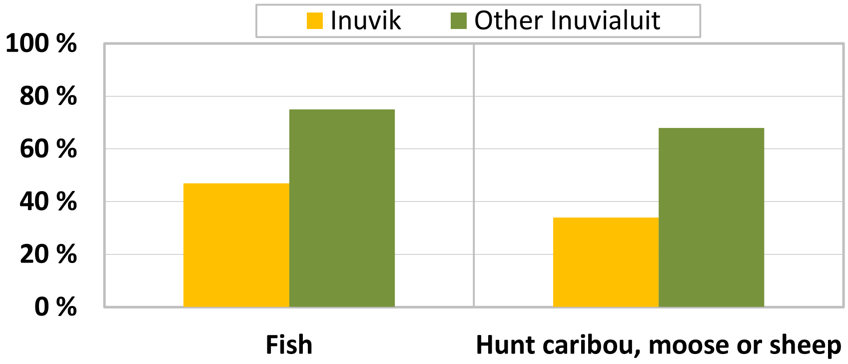


Figure 97. Participation in subsistence activities by region.
Based on the Survey of Living Conditions in the Arctic, a survey of circumpolar Inuit, a project of the Arctic Council. Percent is of respondents who engaged in the subsistence activity in the 12 month period before the interview (2001). Sample size is 4,700.
Source: Kruse et al., 2009

(a) Western Arctic



(b) Nunavut

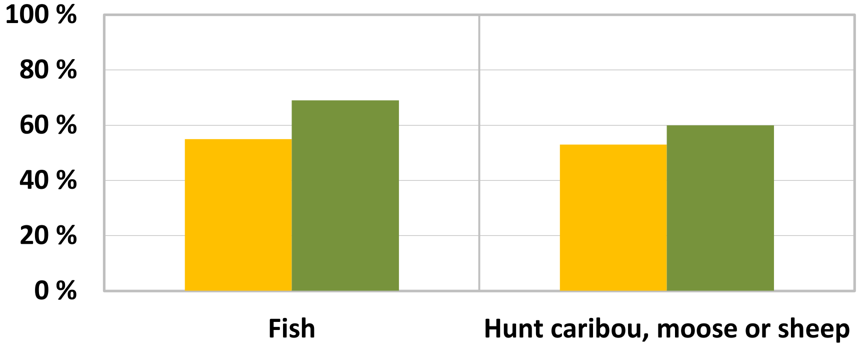


Figure 98. Participation in subsistence activities, comparing residents of major population centres with other residents of other areas: (a) Western Arctic and (b) Nunavut.
Based on the Survey of Living Conditions in the Arctic, a survey of circumpolar Inuit, a project of the Arctic Council. Percent is of respondents who engaged in the subsistence activity in the 12 month period before the interview (2001).
Source: Kruse et al., 2009

Caribou are an important part of Canadian Arctic cultures and still play a central role in people's lives. A measure of the importance is the annual harvest, which in Nunavut (1996 to 2001) averaged 24,522 caribou (Priest and Usher, 2004). In the Northwest Territories, Dene, Inuvialuit, Metis, and non-Aboriginal people from almost all communities hunt the migratory herds and the minimum annual harvest is 11,000 caribou, valued as at least $17 million dollars, including meat replacement and outfitting (Department of Environment and Natural Resources, 2006). Assuming an average carcass weight of 45 kg, the Northwest Territories and Nunavut harvest is about 1.6 million kg caribou. At a beef replacement value of $20 per kg, the annual average harvest of caribou just for meat replacement is $35 million. This excludes any commercial harvesting or any value for hides, let alone the intangible cultural strength and value of hunting. A study commissioned by the Beverly and Qamanirjuaq Caribou Management Board estimated the total net economic value of the caribou harvest of the Beverly and Qamanirjuaq herds at $20.0 million annually (InterGroup Consultants Ltd., 2013). This includes domestic and commercial harvest, less production costs.

Other wild meat, such as muskox (Campbell and Setterington, 2001), beluga whales, waterfowl, and, especially, fish, are important sources of protein and are essential connections to community heritage and traditions.

HUMAN INFLUENCES

Stressors / cumulative impacts

Stressors are discussed throughout this report, with a common theme of growing evidence of impacts of climate change. Some of the impacts, such as those from thawing permafrost, are only beginning to be observed and monitored, though there is a good body of research linking climate variability and change with ecological processes, structure, and composition. As all climate model projections show continued above global average increases in temperatures in the Arctic, climate change will continue to be the major large-scale stressor for the foreseeable future. Other stressors are also widespread, including contaminants from long-range atmospheric transport, but appear to be of lesser magnitude in terms of impacts on ecosystems. At the local and regional level, stressors that lead to habitat fragmentation and disturbance, overharvest, and localized contamination can also be significant for ecosystems.

These stressors interact, often in complex and unpredictable ways. Freshwater ecosystems in the Arctic Ecozone$^+$, for example, are undergoing changes related to climate change, but they also are affected by other human activities, including depletion of stratospheric ozone, deposition of persistent organic pollutants and mercury transported through the atmosphere, and land and watercourse disturbance associated with development activities (Schindler and Smol, 2006). Projected warming and changes in precipitation will result in higher contaminant loads and biomagnifications, while changes in ice cover are predicted to increase ultraviolet radiation levels, resulting in cumulative and/or synergistic effects on aquatic ecosystem structure and function (Wrona et al., 2006). The range of human stressors of concern for terrestrial systems is illustrated below in the section on Main threats to caribou. There are, of course, bound to be unexpected and poorly understood stressors affecting Arctic ecosystems—for example, long-term transport of nitrogen compounds (pollutants) may increase nitrogen availability in tundra soils, interacting with warming temperatures to alter habitat for Arctic species.

Main threats to caribou

This section is mainly summarized and updated from the ESTR technical thematic report *Northern caribou population trends in Canada* (Gunn et al., 2011c). Because migratory tundra caribou mainly winter to the south of the Arctic Ecozone+, some of this discussion also focuses on the taiga ecozones+.

In northern Canada, migratory tundra caribou typically have large annual ranges with annual migrations of hundreds to thousands of kilometres linking their seasonal ranges. While caribou are exposed over the spatial scale of their annual range, they are also accumulating responses to environmental influences over their lifetime (typically about 15 years for an adult female). While much attention has focused on the effects of industrial exploration and development on caribou, the extent and severity of effects depend on the vulnerability of caribou. Vulnerability changes during the cycles of caribou abundance as, during declines and the phase of low numbers, factors that influence caribou births and deaths can have stronger effects. The challenge for hunters, biologists, managers, and co-management boards, therefore, is to understand the vulnerability of their herds, keeping in mind all these factors and assessing not only what the impact might be on the herds, but also what the impact will be on those communities that depend on the abundance of caribou. In the following sections, we present some examples of how ecological influences such as predation, harvesting, and parasites change caribou vulnerability and how threats from development, contaminants, and climate change affect caribou.

Predation, parasites, disease

Although predation, diseases, and parasites are part of the ecology of migratory tundra caribou, they are listed here as threats because their role in influencing trends in caribou abundance interacts with stressors related to human activities. The interactions work in a number of ways, both at the individual and herd scales, and include variables such as whether predation is additive or compensatory to harvesting (meaning whether interactions cause the overall effect of predation plus harvesting to be more or less than the sum of the individual effects). The major predators of migratory tundra caribou are wolves and grizzly bears, but wolverines, lynx, and eagles all take caribou as well. As caribou abundance declines, the role of wolves and bears has greater effect on the caribou until the numbers of predators themselves decline (Bergerud et al., 2008).

Information on trends in predation rates or numbers of predators is patchy, but some comparisons over time are available using a relatively simple index. For example, Heard (1992) reported an average of eight wolves seen per 100 hours of flying surveys in the Queen Maud Gulf area during the 1980s -- lower than the 24 to 33 wolves per 100 hours observed during surveys of the Ahiak Herd in 2007 and 2008 (Poole et al., 2013). For the Bathurst Herd, wolf sightings observed for 16 years between 1987 and 2008 during late winter aerial surveys to estimate caribou calf survival suggest no consistent trend in either wolf sightings or mean pack size (Figure 99). During that time, the numbers of caribou declined 90% and the number of wolves observed at their dens and the number of occupied dens declined (D. Cluff, pers. comm., 2012). Thus, while caribou were declining and wolf populations were probably declining in the region, the sightings of wolves in the vicinity of caribou did not decline. This observation suggests that predation rates were maintained, which would have increased the vulnerability of the caribou herd.

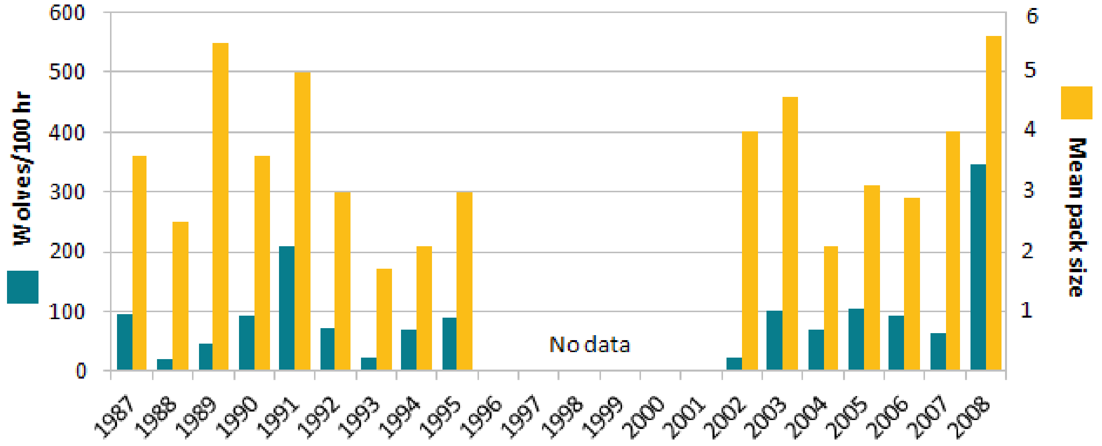


Figure 99. Lack of a trend in wolf sightings and pack size during peak abundance and the decline of the Bathurst Caribou Herd, 1987-2008.
Data are from observations during late winter surveys.
Source: data from Williams and Fournier, 1996; Gunn, 2013c; B. Croft, pers. comm., 2010

Information on the status and trends of diseases and parasites in migratory tundra caribou is fragmentary among herds. Knowledge about if and how disease rates change when caribou abundance changes is particularly lacking. It is known, however, that in at least one instance, a disease has played a role in increasing the vulnerability of caribou. On Southampton Island, caribou abundance and condition are monitored and a high incidence of brucellosis in both sexes is implicated in the herd's decline (see section on Wildlife diseases and parasites on page 60).

Harvest

A recurring theme in wildlife and fisheries management over the centuries is that numerical abundance is not always a hedge against declines to the point of local extinctions. One has only to think of salmon, northern cod, and plains bison to remember that numerical abundance carries the risk of over-confidence—"there's still lots". What determines persistence is rate of change, not the size of the starting population. Another contribution to confidence among users is that the caribou, being cyclic in their abundance, have been low in number before and have come back. Given changing environmental conditions, however, the past may not be a secure guide to the future.

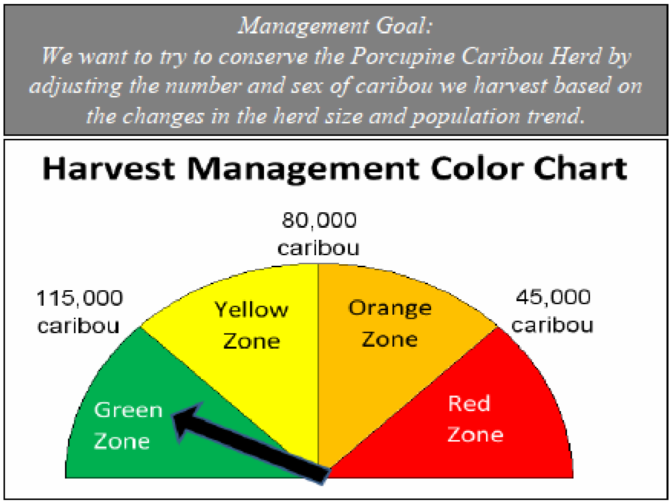
Changes that have occurred on caribou ranges since the 1970s that influence caribou vulnerability include an overall increase in the number of people and shifting socio-economic patterns (such as wage-earning), both of which may influence harvest levels. The human populations of the Arctic and the three taiga ecozones+ have all increased, the combined population almost doubling, from 59,390 in 1971 to 107,213 in 2006 (Environment Canada, 2009b). The increase in number of people is reflected in the increasing size of larger communities (centralization) and increased seasonal and all-year road developments, especially in the Northwest Territories and northern Saskatchewan, on the southern edges of the winter ranges in the taiga and boreal ecozones+ (BQCMB, 2011; Trottier, 2011, pers. comm.).

The increasing number of people, a shift to wage-earning, and changing technologies for hunting (snowmobiles, winter roads, and rapid communications) have likely altered hunting effort and made finding and harvesting caribou more efficient. The relationship between hunting effort and harvest levels, however, is largely unknown and this limits understanding of the effects of hunting. Most of the understanding about the importance of measuring hunting effort is from a range of exploited species other than caribou (Ludwig, 2001). Schooling fish have parallels with gregarious caribou in relation to harvest vulnerability. For pelagic fisheries, constant yield harvests can lead to population collapses if harvesting effort is not directly related to local abundance (Mullon et al., 2005).

Understanding how harvest can add or subtract from herd vulnerability is complicated as there are few measures of hunting effort, and data on harvest levels are sporadic over time. Levels of, or trends in, wounding loss are unknown, although increased effort in management planning includes education and help from Aboriginal elders about respectful hunting practices (e.g., PCMB, 2009; Tlicho Government and Department of Environment and Natural Resources, Government of the Northwest Territories, 2011). Harvest levels at the community scale vary; this is related to annual variability of the distribution of caribou on their seasonal ranges. In Canada's territories, most hunters are Aboriginal and, under Canada's constitution and land claims settlement acts, their hunting rights are respected. Generally, Aboriginal hunters can harvest unlimited numbers of caribou unless there is a conservation issue.

A second category of hunters is resident hunters. The trend is to fewer resident hunters in the Northwest Territories (Government of the Northwest Territories and NWT Biodiversity Team, 2010) as a result of restrictions following the decline of herds. The third category of hunters is under the heading "commercial", which can include harvesting for commercial meat sales and guided, outfitted hunts by non-residents. The level of commercial harvesting has varied among herds. Commercial use has been progressively reduced as the herds have declined and currently there is no commercial harvesting of any Northwest Territories barren-ground herd (Department of Environment and Natural Resources, 2006) and commercial harvesting has been sharply reduced in Nunavut (Coral Harbour, 2011; Dumond, 2011, pers. comm.).

Declines in caribou numbers have led to management measures that consider all user groups and that are tied to herd trend and herd abundance (see Figure 100, for example).



Management Goal:
We want to try to conserve the Porcupine Caribou Herd by adjusting the number and sex of caribou we harvest based on the changes in the herd size and population trend.

Figure 100. Adjusting management based on herd size and population trend: Harvest management plan for the Porcupine Caribou Herd in Canada.
Key elements of management actions, by colour code: Green "Take what you need"; Yellow "Voluntary bulls only" harvest; Orange "Mandatory bulls only and harvest limits"; Red "No hunting".
Source: Porcupine Caribou Management Board presentation from Eamer and Russel, 2013, and Porcupine Caribou Management Board, 2010

Development

Along with increasing numbers of people on the caribou ranges, there are trends toward more exploration and resource development activity. Most attention has focused on activities associated with mineral and hydrocarbon exploration and development, and caribou behavioural responses to these human activities are quite well known (Wolfe et al., 2000a; Stankowich, 2008). Impacts from power lines and shipping have received less attention. The lack of information of impacts from railway development became evident in the Mary River Project (Baffinland) Environmental Impact Assessment (Baffinland Iron Mines Corporation, 2012), leading to conditions of approval that included adaptive management measures to be enacted if unanticipated negative effects on caribou are observed.

As the rate of human activities increases, our lack of understanding about the cumulative effects on caribou at the individual and the herd level becomes a more worrisome gap (Cameron et al., 2005). Limited progress has been made in measuring and managing these cumulative effects (Festa-Bianchet et al., 2011; Gunn et al., 2011a). Examples of development activities and concerns for caribou populations are described below.

Mineral exploration activities boomed in the 1990s on the range of the Bathurst Herd and over the period 2003 to 2008 for the Beverly Herd's calving and summer ranges. The timing of the herd's decline coincided with an increase in mining exploration and development. While there had been mining exploration for decades, the discovery of diamonds in 1991 triggered a surge in exploration with an increase in camps, aircraft, and helicopters on the tundra ranges of the Bathurst herd (pre-calving to fall ranges). At the peak, 118,124 km^2 of new claims had been staked in 1993 in the NWT (G. Bouchard, Natural Resources Canada., pers. comm.). Since 1996, four diamond mines have been built and are operational on the Bathurst Herd's summer range. Three of the diamond mines are large open-pit mines and their activities have reduced the occurrence of caribou in their vicinity over a greater distance than expected (Johnson et al., 2005).

Elsewhere on caribou ranges, exploration and mining of uranium has been the greatest concern in the past among communities that harvest Beverly caribou. Uranium exploration and development has occurred for decades on the Beverly winter range in northern Saskatchewan. Mineral exploration has been increasing over the past 10 years on the ranges of both the Beverly and Qamanirjuaq herds in the Northwest Territories and Nunavut (BQCMB, 2010). As of May 2010, there were many active prospecting permits, mineral claims, and mineral leases on the Beverly and Qamanirjuaq traditional calving ground (BQCMB, 2010).

The possibility of opening the Arctic National Wildlife Refuge to oil and gas development (a proposal that has been under consideration in the United States since the late 1970s) has the potential to constitute a major threat to the Porcupine Caribou Herd. The "1002" area on the Alaskan coastal plain contains both high potential for hydrocarbon deposits and contains the most sensitive habitat for the herd during the calving and post-calving periods (Griffith et al., 2002).

Roads, mineral exploration, mines, and hydrocarbon production are land use activities whose effects on caribou may accumulate over time. Although there are concerns about those cumulative effects, progress on measuring them and managing them is barely underway. In particular, roads can increase human access and act as partial barriers to caribou movement. Table 17 indicates main caribou issues related to three Nunavut development prospects.

Table 17. Examples of industrial projects and caribou issues, 2012.

Project	**Location**	**Caribou issue**
Mary River Iron Mine	Baffin Island, eastern Nunavut	North Baffin caribou: low in 70-year cycle; railway effects on movement
Kiggavik Uranium Mine	Baker Lake, central Nunavut	Multiple herds: new road access through summer habitat; increased harvest pressure
Izok Corridor Zinc Mine	Western mainland Nunavut	Bathurst caribou: road at edge of traditional calving ground Dolphin and Union caribou: road in winter range

Source: Introduction by M. Setterington, in Eamer et al., 2013

Contaminants

It is uncertain as to how longer-distance and local sources of contaminants and pollutants influence the vulnerability of caribou. While levels are monitored and levels are mostly low, synergistic effects are uncertain and how contaminants may interact with health and condition is unknown. The Canadian Northern Contaminants Program has been active in monitoring Persistent Organic Pollutants (POPs), heavy metals, and radionucleotides for the last three decades. The following account is derived from the NCP summary report (Northern Contaminants Program, 2003), except where noted. Some 15 different caribou herds across Nunavut, the Northwest Territories, and the Yukon were monitored during the 1990s through two large monitoring programs; additional monitoring has been undertaken since then for some herds (Gamberg, 2009). Assessments of risk to human health from contaminants show that caribou is a safe and nutritious food choice across northern Canada (Donaldson et al., 2010).

Persistent organic pollutants such as DDT, PCBs, dioxins, and furans were found at only very low levels in caribou (often too low to be detected at all) and are not of concern for either caribou or human health (see also Gamberg et al., 2005a).

Some heavy metals, however, are found at elevated levels in caribou, though not to the same extent as in some marine mammals. There are wide variations in the levels of metals from herd to herd, probably due to the variation of levels in the underlying geology. Cadmium levels tend to be higher in the kidneys and livers of the Beverly caribou in the Northwest Territories and Nunavut, compared to the levels in other herds. Natural sources of cadmium in the underlying rocks in the area are likely responsible. This cadmium accumulates in lichen which is then eaten by the caribou. Mercury levels show no clear pattern (Gamberg et al., 2005a), with the highest levels found in the Beverly Herd and in Meta Incognita Peninsula caribou (part of the South Baffin population). In the central and northeastern parts of northern Canada, levels of mercury in caribou follow the same geographic pattern as levels found in sediments. Scientists consider that much of this mercury has been transported from human-made sources in other parts of the world. An exception is mercury in caribou from the Yukon, where local geology may be more important (Northern Contaminants Program, 2003).

Forest fires

The importance of trends in forest fires to caribou is related to winter forage, dominated by fruticose lichens which are most abundant in older successional forests. The effect of forest fires on caribou abundance has been controversial but most studies do document that caribou tend to avoid recently burned areas (Joly et al., 2003).

Climate change

Climate changes will affect the movements and distribution of the herd as well as the ability of hunters to get on the land to be able to continue their traditional hunting practices. Table 18 provides a general treatment of observed and potential climate impacts on caribou, their ranges, and the communities that depend on them. The table was developed for the Porcupine Caribou Herd and as such some of the impacts described (for example, changes to calving grounds on coastal plains) do not apply to all herds.

Table 18. Observed and potential impacts of climate change on migratory tundra caribou populations, based primarily on research conducted on the Porcupine Caribou Herd.

Climate change condition	**Impact on habitat**	**Impact on movement**	**Impact on body condition**	**Impact on productivity**	**Management implications**
Earlier snowmelt on coastal plain	· Higher plant growth rate	· Core calving grounds move further north · Less use of current calving grounds	· Cows replenish protein reserves faster · Higher calf growth rate	· Higher probability of pregnancy · Higher June calf survival	· Need for flexibility in calving ground protection (adaptive management)
Warmer, drier summer	· Earlier peak biomass · Plants harden earlier · Reduction in mosquito breeding sites · Increased parasitic (oestrid) fly harassment · Increased frequency of fires on winter range · Fewer "mushroom" years	· Movement off of calving grounds earlier · More use of insect relief habitat in July · Avoidance of recently burned winter habitat	· Increased harassment will lower fall body condition	· Reduced probability of pregnancy	· Protection of insect relief areas important
Warmer, wetter fall	· More frequent icing conditions	· Caribou abandon ranges with severe surface icing	· Higher winter mortality · Earlier weaning		
Warmer, wetter winters	· Deeper denser snow · Icing conditions, especially in tundra and Arctic islands	· Increased dependence on low snow regions · Stay on winter range longer	· Greater over winter weight loss · Higher incidence of extended lactation	· Lower over winter mortality on calves	· Need to consider protection of low snow regions (adaptive management)
Warmer springs	· More freeze/thaw cycles during spring migration · Faster spring melt	· Movement slowed and/or movement unto drier windswept ridges	· Accelerated weight loss in spring	· Higher wolf predation on cows and calves due to use of windswept ridges	· Concern over timing and location of spring migration in relation to traditional harvesting areas
Overall effect: In very general terms: the calving range improves but with movement and reliance on more northern portions of the calving range; animals leave calving range earlier; cows and calves suffer reduced summer and fall body reserves due to increase in oestrid fly harassment; mosquito harassment may be reduced if summers drier; more frequent icing in fall, winter, and spring ranges, which depend on the location of these ranges; may have moderate to severe implications to body condition and survival.					

Source: Gunn et al., 2011c; update of Chapter 10 of the Arctic Climate Impact Assessment, 2005, by the author (Russell)

It is difficult to attribute a single event such as an icing storm as being within the "normal" range or as an indication of a warming climate. Examples of this type of event occurred in the fall of 2003, when coastal areas from Alaska to Kugluktuk, Nunavut, experienced icing conditions that forced caribou to move in search of accessible forage. Ice on the land formed a barrier between the caribou and their food (Nagy, 2007).

Another factor that adds to the complexity of predicting impacts of climate change is that all herds have evolved and adapted to a unique suite of environmental factors within their ranges —some herds cope with winter ranges characterized by deep, persistent snow, while others enjoy mild winter conditions; some herds occupy excellent summer ranges with an abundance of fresh green vegetation; others have to replenish fat and protein reserves depleted over winter with vegetation that is limited by a brief, intense summer growing season. Changes that result in more severe winter range conditions, for example, would have a different effect on different herds—even neighbouring herds. For example, under a warmer climate, the annual range for the Leaf River Herd may increase while that of the George River Herd may contract (Sharma et al., 2009).

Further, at the population level, some herds have exhibited a high rate of increase, over 15% annually, while others have increased at rates of less than 5% annually, primarily reflecting higher adult female mortality rates (Figure 77). Environmental changes that result in an increase in adult female mortality would have a greater impact on herds that demonstrate a low rate of increase.

Within the range of the Porcupine Herd, for example, the trends of climate change are marked. Spring, in particular, has warmed over the last three decades. During late spring, after calving, this has resulted in early snowmelt and more food available for nursing mothers. As a consequence, early calf survival has improved (Griffith et al., 2002). In early spring, however, when the herd is on migration, warmer weather has resulted in more freeze-thaw cycles as temperatures get above freezing during the day and below freezing at night. Specifically, the number of days where the temperature has risen above zero during spring migration doubled during the population decrease phase (1989–2001) compared to the population increase phase (1975–1988) (Griffith et al., 2002). The greater difficulty in traveling and feeding through ice crusts would result in higher energetic costs, and moving onto wind-blown ridges during migration would result in potential increased mortality from wolves, as wolves are at an advantage in shallow snow (Griffith et al., 2002).

It is clear from this example that specific information on how changes in temperature and precipitation alter environmental conditions for caribou throughout during their annual cycle is needed to understand how each herd is adapted to its range and to be able to predict impacts of climate change on caribou abundance and distribution. Work is underway through the CircumArctic Rangifer Monitoring and Assessment Network (CARMA) to produce datasets of relevant climate metrics at the scale of the herd ranges (Russell et al., 2013). Climate data and other data (such as measures of biomass) from satellite monitoring are used to develop daily median values of "caribou-relevant" climate variables for defined regions within the range of each herd, for each of five seasons (calving, summer, fall, winter, and spring). Variables, aside from temperature and precipitation, include: accumulated growing degree days above 0°C and above 5°C, leaf area index and vegetation greenness fraction; measures related to mosquito and oestrid (warble and bot fly) activity; snow depth, density, cover, and melt rate; and, accumulated number of days with freeze/thaw events.

Cumulative energy costs to caribou

Cumulative effects assessment (CEA) is part of environmental impact assessment (EIA) and focuses on the combined impact of the individual effects of industrial development projects. However, the context is also set by multiple stresses from all human activities additional to natural environmental effects at both the individual and population scales. To scale up the individual animal's behavioural responses to the population requires being able to estimate the energy costs to the individual and whether those costs will affect its reproduction and survival. Estimating the costs of a behavioural response is not straightforward: as well as the energy costs of movement and interruption in foraging time, there may also be an effect on diet (energy protein intake) if a displacement puts the individual in a different habitat. Understanding and integrating the relationships between behaviour, habitat selection, energy, and protein intake relative to reproduction and survival is data intensive and requires interdisciplinary collaboration, as the understanding is based on ecology, nutritional ecology, and modeling. Russell (2012) describes modelling to integrate environmental influences to project caribou vulnerability to developments.

Stewardship / conservation

Addressing climate change

A major human stressor for the Arctic Ecozone^{+} is climate change as demonstrated throughout this report. This stressor cannot be overcome through stewardship and conservation measures within the ecozone^{+}, although measures can be taken to reduce greenhouse gas emissions within the ecozone^{+} and to adapt to coming changes. This is a global problem, and the most significant measure of actions taken to conserve Arctic ecosystems is the rate of reduction of releases of greenhouse gases worldwide. Globally, however, emissions continue to grow, though the pace of growth appears to be slowing (Environment Canada, 2013a).

Historical and projected trends in greenhouse gas emissions for Canada are graphed in Figure 101, showing that additional greenhouse gas reduction measures will be needed to achieve Canada's 2020 target for emissions. The decrease in emissions around 2009 due to the global recession reflects a global trend for that time period and shows that actual emissions are closely tied to economic factors (Environment Canada, 2013a).

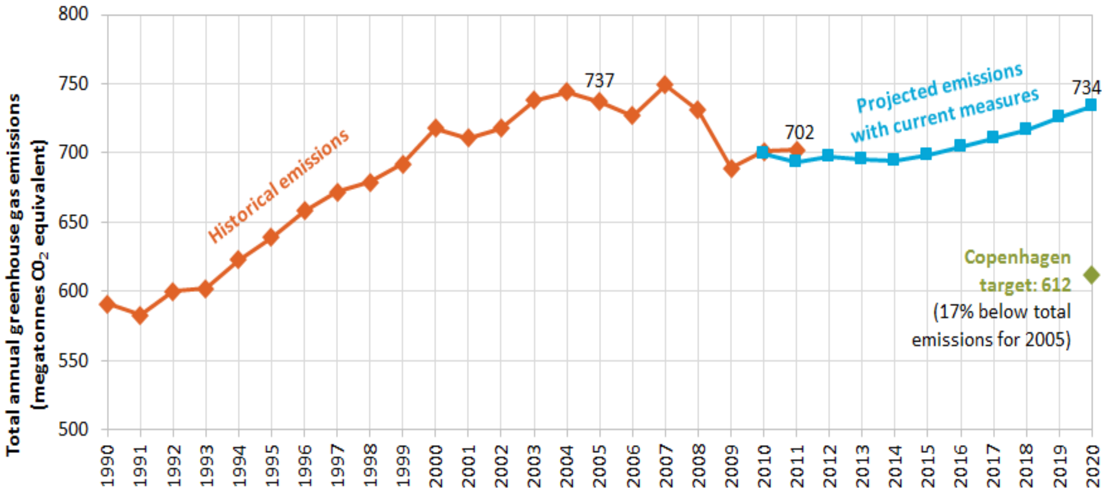


Figure 101. Canada's annual greenhouse gas emissions, 1990–2011, projected emissions with current mitigation measures, and Canada's international commitment for reduction by 2020.
Projections with current measures include the compliance contribution of the Land Use, Land-Use Change and Forestry (LULUCF) sector for each year; actual emission trends are projected to be 28 Mt higher than the projections shown in 2020.
Source: Environment Canada, 2013b and 2013c

Protected areas

The global scale of climate change does not mean that regional and local stewardship are not important in the Arctic. As plant and animal ranges shift and ecosystem function and processes are affected by climate change, it becomes even more important to focus on conservation of important lands and waters such as breeding areas for migratory birds and caribou, building into conservation plans the flexibility needed to adapt to the changing landscape. An important stewardship tool is the creation and management of protected areas. In addition, pressure is increasing in some areas and likely to increase further for development and expansion of human use of the sea, coastal zone, and land. Protected areas are an important management tool for conserving the ecosystems needed to support Arctic biodiversity and the ecosystem services they provide. Arctic ecosystems operate on huge scales, often crossing the terrestrial/marine divide. No one protected area is sufficiently large to maintain the integrity of ecosystems and stewardship of the entire land base is crucial, taking into consideration ecosystem elements and processes that routinely move in and out of protected areas, such as migratory birds and caribou.

Analyses for this section were conducted for ESTR based on data provided by federal, territorial, and provincial jurisdictions and have not been updated (see figures for references). Location of protected areas is shown for all of Canada in Figure 102 and the growth of protected areas in the Arctic Ecozone$^+$ is shown in Figure 103. In addition to federal parks and wildlife sanctuaries, the ecozone$^+$ has (generally smaller) territorial and Quebec parks and reserves, often protecting cultural heritage sites and areas such as estuaries with particularly high value for biodiversity.

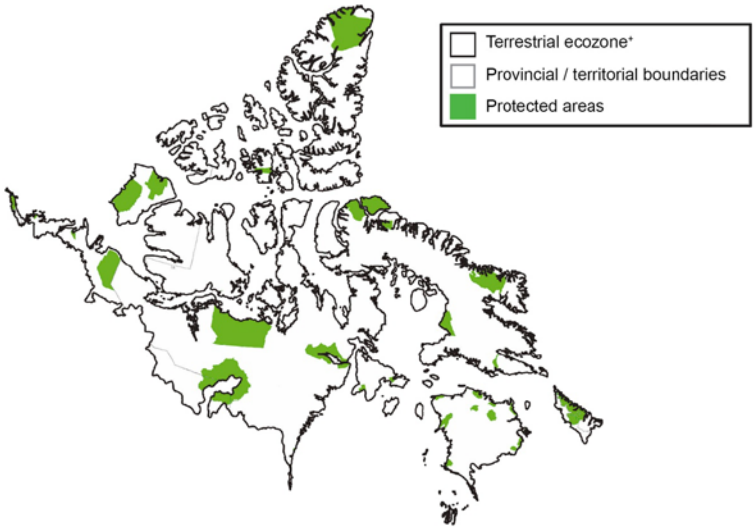


Figure 102. Protected areas in the Arctic Ecozone^{+}, 2009.
Note that smaller protected areas, such as the National Wildlife Areas described in the text box below, do not show up on this map due to the scale.
Source: Environment Canada (2009a) using data from the Conservation Areas Reporting and Tracking System (CARTS), v.2009.05, 2009 (CCEA, 2009); data provided by federal, provincial, and territorial jurisdictions

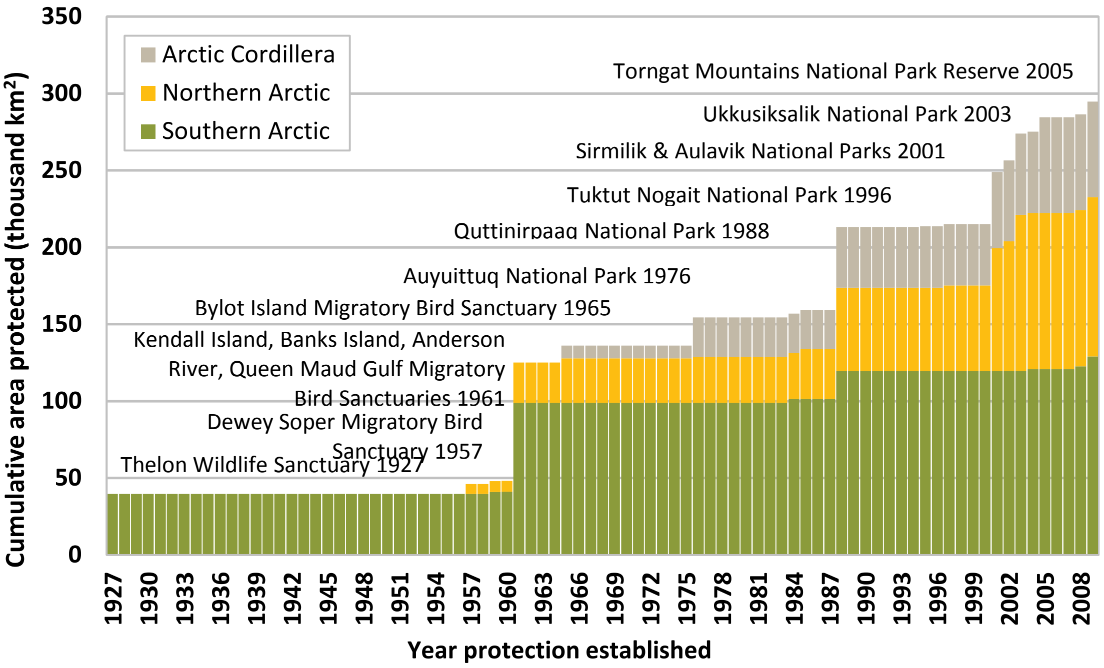


Figure 103. Growth of protected areas in the Arctic Ecozone^{+}, 1927–2009.
Data provided by federal, provincial, and territorial jurisdictions, updated to May 2009. Only legally protected areas are included. All areas shown are protected as IUCN (International Union for Conservation of Nature) categories I–III. IUCN categories are based on primary management objectives (see text for more information). Selected protected areas are shown with their dates of establishment.
Source: Source: Environment Canada (2009a) using data from the Conservation Areas Reporting and Tracking System (CARTS), v.2009.05, 2009 (CCEA, 2009); data provided by federal, provincial, and territorial jurisdictions

While the overall percentage of land protected is 11.3% for the ecozone$^+$, this is not evenly distributed. The Arctic Archipelago has (as of May 2009) the highest proportion of protected area at 24.0% in 10 protected areas, followed by the Southern Arctic with 15.9% protected through 21 protected areas. The Northern Arctic has 6.7% of its land protected through 22 protected areas.

Much of the growth of protected areas in the Arctic has been related to the settlement of land claims. Many Arctic parks are managed through regimes created through land claim settlements, protecting the harvesting rights of Inuit within the parks, while working to conserve species and areas of special cultural and ecological significance.

National Wildlife Areas in Nunavut

National Wildlife Areas protect nationally significant habitat for migratory birds, support species or ecosystems at risk, or protect rare or unusual habitat. Critical natural features are conserved and activities considered harmful to species or habitats are prohibited. Three new National Wildlife Areas were created in Nunavut in June 2010 to protect critical habitat for Arctic seabirds, bowhead whales, and other species. They will be co-managed by local and federal governments, and were chosen based on advocacy and involvement from the communities of Qikiqtarjuak and Clyde River.

Akpait National Wildlife Area (774 km^2) is an important area for migratory birds. It provides breeding habitat for one of Canada's largest thick-billed murre (*Uria lomvia*) colonies, black-legged kittiwakes (*Rissa tridactyla*), glaucous gulls (*Larus hyperboreus*), and black guillemots. It is also home to polar bears, walruses, and several species of seals.

Qaqulluit National Wildlife Area (398 km^2) is home to Canada's largest colony of northern fulmars (*Fulmarus glacialis*), representing an estimated 22% of the total Canadian population. Marine animals, including walrus and ringed seals, also use the waters of this National Wildlife Area.

Ninginganiq National Wildlife Area (Isabella Bay) (336 km^2) protects critical summer habitat for the Eastern Arctic population of bowhead whales, a Threatened species.

Source: Latour et al., 2008

Environmental governance

Beginning in the 1970s and accelerating since the close of the Cold War, "the North" is emerging as a new circumpolar geopolitical entity (Heininen and Nicol, 2007). Arctic countries and their indigenous peoples' organizations are entering new discourses concerning the scale and nature of circumpolar regionalism and critical new environmental, human security, and economic development challenges. These northern discourses stress regional co-operation, human security, and sustainable development (Heininen and Nicol, 2007).

Several features set Arctic governance apart from the rest of Canada:

- There is only one Aboriginal language group and culture (with important regional differences) across the entire region—a quarter of Canada—and its members remain numerically dominant.
- The Inuit have forged alliances with related cultures/linguistic groups around the circumpolar region, for example, through the Inuit Circumpolar Council (ICC), an international non-government organization representing approximately 150,000 Inuit of Alaska, Canada, Greenland, and Chukotka in Russia.

- The ICC and other Arctic indigenous organizations are Permanent Participants at the Arctic Council, an intergovernmental organization of the eight circumpolar countries that provides a forum for collaboration and oversight of many international initiatives related to ecological science and sustainable development.
- Resource management boards established pursuant to the land claims settlements have become the dominant forces in management of land and natural resources. The networks of resource management boards, councils, and local hunters' and trappers' associations function as "bottom-up", co-operative decision-making systems that are mandated to make use of science and Aboriginal traditional knowledge in their decision-making.

Wildlife in the Arctic Ecozone^{+} portion of the Yukon, NWT, and Nunavut is co-managed by governments and Inuvialuit pursuant to the Inuvialuit Final Agreement Indian Affairs and Northern Development Canada, 1984) and by governments and Inuit pursuant to the Nunavut Land Claims Agreement (Indian Affairs and Northern Development Canada, 1993). While the terms of these agreements differ, in general they recognize the Aboriginal rights of Inuit and Inuvialuit to manage the harvest of wildlife, subject only to the need for conservation and public safety. The primary management bodies are two Wildlife Management Advisory Councils (WMAC) for the Inuvialuit Settlement Region in the Northwest Territories and the Yukon North Slope, and the Nunavut Wildlife Management Board (NWMB) for Nunavut. Although the mandates of these boards differ, both organizations bring together Inuit and government representatives.

In Nunavut, the Nunavut Department of Environment and Environment Canada (Canadian Wildlife Service) both appoint members to sit on the NWMB, along with Inuit appointed by their regional organizations. NWMB membership also includes other federal departments and Nunavut Tunngavik Incorporated. However, as an institution of public government, all NWMB members represent the public interest and not necessarily the interests or opinions of their appointing bodies. These boards are supported by hunters' and trappers' associations from each community and other community committees.

A comprehensive review of how these new governance structures influence environmental status and trends, and knowledge thereof, is beyond the scope of this report. A case study will, however, serve as an example of evolving Arctic environmental governance.

Case study on environmental governance: Kitikmeot and the West Kitikmeot/Slave Study

In addition to the Nunavut Wildlife Management Board, three regional Inuit associations were established under the Nunavut Land Claims Agreement to manage Inuit-owned lands in the regions. The Kitikmeot Inuit Association (KIA) has a mandate "To represent the interests of Kitikmeot Inuit by protecting and promoting our social, cultural, political, environmental and economic well-being" (Kitikmeot Inuit Association, 2013). Under the KIA, the Kitikmeot Corporation and the Kitikmeot Economic Development Commission are delegated the responsibility of promoting economic development in the region. The KIA is responsible for administering surface lands in the Kitikmeot region. There is also a Kitikmeot Heritage Society that promotes Kitikmeot culture and history.

Even before its formal establishment, Kitikmeot communities were cooperating to organize the generation of information needed to inform environmental management decisions. For example, more than 20 years ago they completed a major study that quantified the country foods that they harvested from the land (Gunn et al., 1986).

The discovery of diamonds in 1991 at Lac de Gras initiated one of the largest mineral staking rushes in the history of the world (Environment and Natural Resources, 2012b). Out of concern about the impacts of rapid development on the environment, Kitikmeot communities formed a partnership with other Aboriginal (Dene) organizations, environmental organizations, government, and industry. The West Kitikmeot/Slave Study (WKSS) Society was founded in 1996 by nine partner organizations. The purpose of the Society was to collect environmental and socioeconomic information to enable better-informed planning and contribute to a baseline for assessing and mitigating cumulative effects of development (Environment and Natural Resources, 2012b). Community capacity and traditional knowledge were priorities.

The WKSS region encompassed the western part of Kitikmeot (Arctic Ecozone$^+$, in Nunavut) plus the area between Great Slave Lake and Contwoyto Lake (Taiga Shield Ecozone$^+$, in the NWT), with the treeline dividing the area approximately in half (Figure 104) (West Kitikmeot Slave Study Society, 2001). The calving, migrating, and wintering ranges of the Bathurst Caribou Herd were largely within the WKSS region. The Inuit communities of Kugluktuk, Bathurst Inlet, Umingmaktok, and Cambridge Bay, and the Dene communities of Gameti, Wha Ti, Rae Edzo, Wekweti, Yellowknife, Dettah, and Lutselk'e were included in the study area. The partners developed an initial five-year research program to provide information necessary to examine the long- and short-term effects of development in the WKSS area. The initial WKSS research program ended in 2001, but the society continued to fund projects until 2009, when it was dissolved (Environment and Natural Resources, 2012b).

The WKSS sponsored a number of wildlife-related studies, including studies of traditional ecological knowledge, specific habitats (such as the use of eskers by wildlife), place names as indicators of biogeographical knowledge, use of native plants for mine reclamation and revegetation, water quality, and ecology of important wildlife species, including caribou, moose, muskoxen, grizzly bears, wolves, and wolverines. Results of scientific studies were often presented at workshops attended by people from villages throughout the region so that results could be integrated with traditional ecological knowledge and priorities for action could be developed (Dumond et al., 2009). Conclusions were fed into the management process, to inform decisions, for example, on harvest quotas and on monitoring and research needs.

Collaborative programs and projects such as those undertaken through the WKSS can be effective because they provide a structure for making effective use of both scientific and traditional ecological knowledge (Gunn et al., 1988; Freeman, 1992; Stevenson, 1996; Zamparo, 1996; Berkes, 1998; Duerden and Kuhn, 1998; Usher, 2000). A side benefit is the promotion of mutual understanding, mutual respect, and mutual learning among local communities, regional management authorities, researchers, and other stakeholders, including industry.

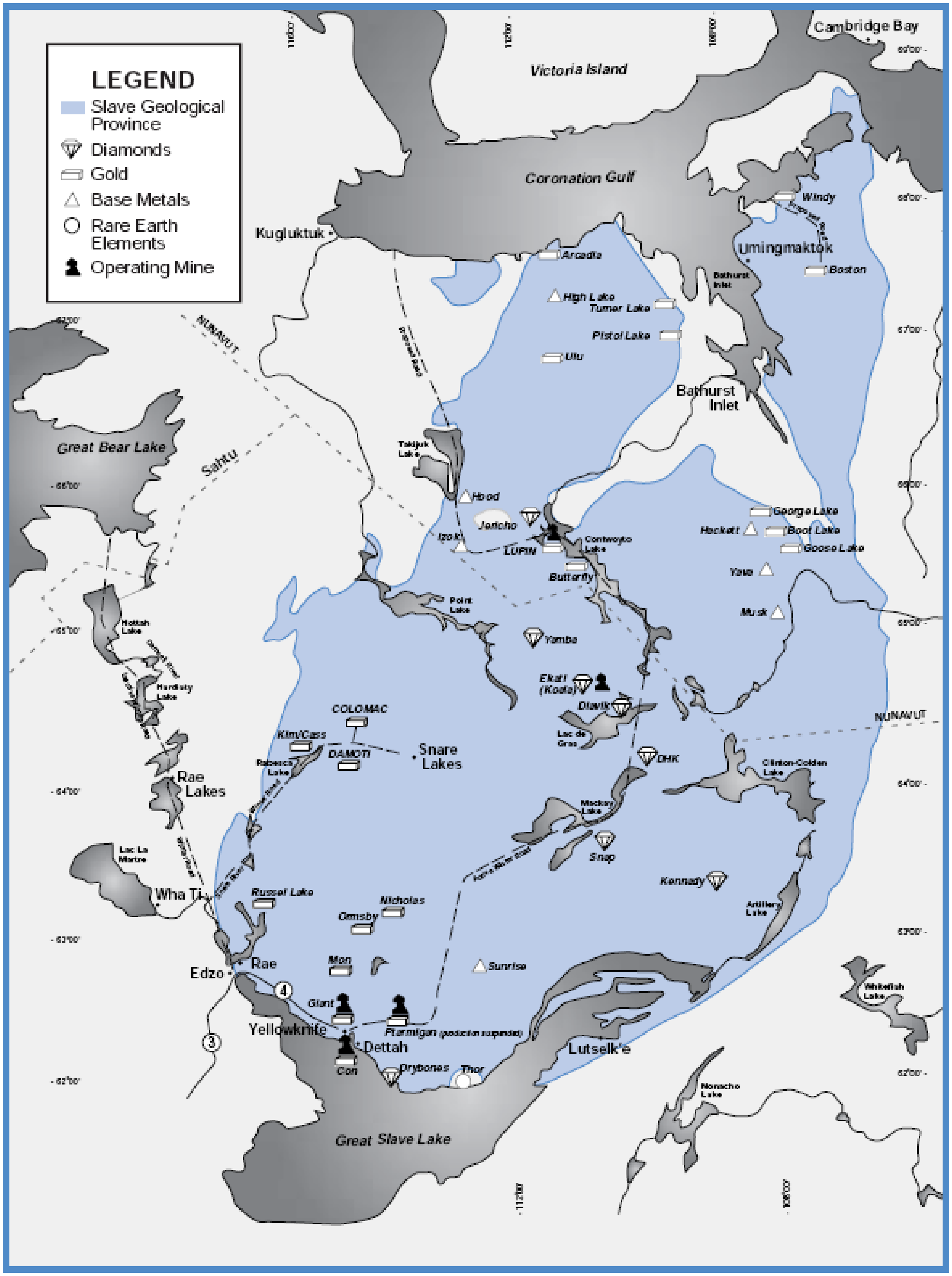


Figure 104. West Kitikmeot/Slave Study area.
Source: West Kitikmeot Slave Study Society, 2001

APPENDIX 1: DESCRIPTIONS OF SURFICIAL MATERIALS

Alluvial Deposits: stratified silt, sand, clay, and gravel; floodplain, delta, and fan deposits; in places overlies and includes glaciofluvial deposits

Alpine Complexes: rock, colluvium, and till; rock and Quaternary deposits complex in an area, characterized by alpine and glacial landforms

Coarse grained (Glacio)Lacustrine: sand, silt, and gravel; deposited as deltas, sheet sands, and lag deposits

Coarse grained (Glacio)Marine: sand and gravel; deposited as sheet sands, deltas, and extensive flights of beaches

Colluvial Blocks: Blocks, and rubble with sand and silt; derived from crystalline bedrock, medium grade metamorphic substrate, and cemented sandstone

Colluvial Fines: silt, clay, and fine sand; derived from substrate weakly consolidated shale and siltstone substrate

Colluvial Rubble: rubble and silt; derived from carbonate and consolidated fine clastic sedimentary rock substrate

Colluvial Sand: Sand and gravel; derived from poorly lithified sandstone and conglomerate substrate

Eolian Deposits: sand and minor silt; dunes, blowouts, and undulating plains; in most places overlies deltaic sediments, coarse lacustrine sediments, or glaciofluvial deposits

Fine grained (Glacio)Lacustrine: silt, and clay, locally containing stones; deposited as quiet water sediments

Fine grained (Glacio)Marine: dominantly silt and clay, locally containing stones; deposited as quiet water sediments

Glaciers: ice and minor morainal debris

Glaciofluvial Complex: sand and gravel and locally diamicton; undifferentiated ice contact stratified drift, and outwash; locally includes till and rock

Glaciofluvial Plain: sand and gravel; deposited as outwash sheets, valley trains, and terrace deposits

Lacustrine Mud: fluid silty clay and clayey silt; deposited as quiet water sediments'

Lacustrine Sand: sand and locally gravel; deposited as sheet sands, lags, and beaches

Lag (Glacio)Marine: sand, gravel, and pockets of finer sediment; thin to discontinuous sediment veneer and residual lag developed during marine submergence; includes areas of washed till and rock

Marine Mud: fluid silty clay and clayey silt; deposited as quiet water sediments

Marine Sand: sand and locally gravel; deposited as sheet sands, lags, and beaches

Organic Deposits: peat, muck and minor inorganic sediments; large bog, fen, and swamp areas where organic fill masks underlying surficial materials; geneally >2 m thick

Quaternary Volcanics: consolidated lava, breccia and tephra; dominantly basaltic and andesitic in composition; includes flows, volcanic piles, and cinder cones

Till Blanket: thick and continuous till

Till Veneer: thin and discontinuous till; may include extensive areas of rock outcrop

Undivided: rock with minor Quaternary deposits

Source: Geological Survey of Canada, 1994

APPENDIX 2: DETAILED LAND COVER CLASSES

Table 19. Comparison of detailed land cover classes in the Arctic Ecozone+ with generalized land cover classifications. The Southern Arctic is dominated by graminoid and shrub tundra; the Northern Arctic is dominated by sparsely vegetated bedrock, till colluvium, and barren lands; the Arctic Cordillera is dominated by snow and ice and mountains.

Class	**Land Cover Type**	**Abbrev.**	**ESTR Land Cover Class**	**Southern Arctic**		**Northern Arctic**		**Arctic Cordillera**		**Total Arctic Ecozone$^+$**	
				km^2	%	km^2	%	km^2	%	km^2	%
1	Tussock graminoid tundra	Tgt	Barren	107,242	18.62%	65,659	5.0%	1,461	0.7%	174,362	8.3%
2	Wet sedge	Sed/w	Wetland	50,585	8.8%	94,135	7.1%	3,418	1.6%	148,138	7.0%
3	Moist to dry non-tussock graminoid/dwarf shrub tundra	Tntg/ds	polar tundra	51,247	8.9%	81236	6.2%	3,660	1.7%	136,143	6.5%
4	Dry graminoid prostrate dwarf shrub tundra	Tg/pds	polar tundra	9,821	1.7%	18,959	1.4%	146	0.1%	28,926	1.4%
5	Low shrub	Sl	polar tundra	87,461	15.2%	4,315	0.3%	203	0.1%	91,979	4.4%
6	Tall shrub	St	polar tundra	24,699	4.3%	167	0.0%	32	0.0%	24,898	1.2%
7	Prostrate dwarf shrub	Spd	polar tundra	109,871	19.1%	110,034	8.4%	4,785	2.2%	224,690	10.7%
8	Sparsely vegetated bedrock	Rsv	Barren	52,243	9.1%	323,284	24.5%	18,575	8.7%	394,102	18.7%
9	Sparsely vegetated till colluvium	TCsv	Barren	11,539	2.0%	177,724	13.5%	14,853	7.0%	204,116	9.7%
10	Bare soil with cryptogam crust - frost boils	TScc	Barren	18,823	3.3%	155,281	11.8%	2,935	1.4%	177,039	8.4%
11	Wetlands	Wet	Wetland	32,528	5.6%	33,763	2.6%	965	0.5%	67,256	3.2%
12	Barren	Bscalc	Barren	19,470	3.4%	195,134	14.8%	20,946	9.8%	235,550	11.2%
13	Snow and/or Ice	S/I	snow/ice/glacier	303	0.1%	33,633	2.6%	115,736	54.4%	149,672	7.1%
14	topographic shadows	Shad	mountain/alpine	8	0.0%	24,006	1.8%	25006	11.8%	49020	2.3%
	SUMS			575,840	100.0%	1,317,330	100.0%	212,721	100.0%	2,105,891	100.0%

Source: Ahern, 2010; Frisk, 2011; Ahern et al., 2011

REFERENCES

Aanes, R., Saether, B.E., Smith, F.M., Cooper, E.J., Wookey, P.A. and Oritsland, N.A. 2002. The Arctic Oscillation predicts effects of climate change in two trophic levels in a high Arctic ecosystem. Ecology Letters 5:445-453.

Abnizova, A., Siemens, J., Langer, M. and Boike, J. 2012. Small ponds with major impact: the relevance of ponds and lakes in permafrost landscapes to carbon dioxide emissions. Global Biogeochemical Cycles 26, GB2041.

ACIA. 2005. Arctic Climate Impact Assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. 1042 p.

Adams, J.M. and Faure, H. 1997. Preliminary vegetation maps of the world since the last glacial maximum: an aid to archaeological understanding. Journal of Archaeological Science 24:623-647.

Aerts, R., Cornelissen, J.H.C. and Dorrepaal, E. 2006. Plant performance in a warmer world: general responses of plants from cold, northern biomes and the importance of winter and spring events. Plant Ecology 182:65-77.

Aglukkaq, L. 2012. Minister Aglukkaq announces $14.7 million contract for Distant Early Warning (DEW) Line clean-up project (news release, April 18, 2012) [online]. http://www.leonaaglukkaq.ca/media_/riding-news/minister-aglukkaq-announces--14.7-million-contract-for-distant-early-warning--dew--line-clean-up-project (accessed 1 December, 2012).

Ahern, F. 2010. Analysis of the Landsat ETM+ mosaic of northern Canada for the Ecoystem Status and Trends Report (ESTR). TerreVista Earth Imaging. Data courtesy of the Canada Centre for Remote Sensing. See Olthof et al., 2005 for more information.

Ahern, F., Frisk, J., Latifovic, R. and Pouliot, D. 2011. Monitoring ecosystems remotely: a selection of trends measured from satellite observations of Canada. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 17. Canadian Councils of Resource Ministers. Ottawa, ON. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Ahn, J., Brook, E.J., Mitchell, L., Rosen, J., McConnell, J.R., Taylor, K., Etheridge, D. and Rubino, M. 2012. Atmospheric CO_2 over the last 1000 years: A high-resolution record from the West Antarctic Ice Sheet (WAIS) Divide ice core. Global Biogeochemical Cycles 26, GB2027.

Alendal, E. and Helle, O. 1983. Helmith parasites of muskoxen *Ovibos moschatus* in Norway incl. Spitsbergen and in Sweden, with a synopsis of parasites reported from this host. Fauna Norv. A 4:41-52.

Allard, M., Fortier, R., Duguay, C. and Barrette, N. 2002. A trend of fast climate warming in northern Quebec since 1993. Impacts on permafrost and man-made infrastructures. *In* American Geophysical Union, Fall Meeting 2002. Abstract #B11E-03. American Geophysical Union. San Francisco, CA. Vol. 83(47).

Allard, M., Wang, B.L. and Pilon, J.A. 1995. Recent cooling along the southern shore of Hudson Strait, Quebec, Canada, documented from permafrost temperature-measurements. Arctic and Alpine Research 27:157-166.

AMAP. 2006. AMAP assessment 2006: acidifying pollutants, arctic haze, and acidification in the Arctic. Arctic Monitoring and Assessment Programme. Oslo, Norway. xii + 112 p.

AMAP. 2011. Snow, water, ice and permafrost in the Arctic (SWIPA): climate change and the cryosphere. Arctic Monitoring and Assessment Programme. Oslo, Norway. xii + 538 pp. p.

Anderson, J. 2004. From Alaska to Greenland: A comparison of the Arctic Small Tool and Thule Traditions. Totem 12:1-13.

Anderson, R.K., Miller, G.H., Briner, J.P., Lifton, N.A. and DeVogel, S.B. 2008. A millennial perspective on Arctic warming from ^{14}C in quartz and plants emerging from beneath ice caps. Geophysical Research Letters 35 (L01502):1-5.

Anderson, R.M. 1934. The distribution, abundance, and economic importance of the game and fur-bearing mammals of western North America. Proceedings of the fifth Pacific Science Congress. University of Toronto Press. Toronto, ON. Vol. 5, pp. 4055-4075.

Andriashek, D., Kiliaan, H.P.L. and Taylor, M.K. 1985. Observations on foxes, *Alopex-lagopus* and Vulpes vulpes, and Wolves, *Canis lupus*, on the off-shore sea ice of Northern Labrador. Canadian Field Naturalist 99:86-89.

Antoniades, D., Crawley, C., Douglas, M.S.V., Pienitz, R., Andersen, D., Doran, P.T., Hawes, I., Pollard, W. and Vincent, W.F. 2007. Abrupt environmental change in Canada's northernmost lake inferred from fossil diatom and pigment stratigraphy. Geophysical Research Letters 34 (L18708):1-5.

Antoniades, D., Douglas, M.S.V. and Smol, J.P. 2005. Quantitative estimates of recent environmental changes in the Canadian high Arctic inferred from diatoms in lake and pond sediments. Journal of Paleolimnology 33:349-360.

Arctic Borderlands Ecological Knowledge Co-op. 2003. Proceedings of the Arctic Borderlands Ecological Knowledge Co-op seventh annual gathering. Fort McPherson, NT. 28 February - 2 March 2002. Arctic Borderlands Ecological Knowledge Society. Whitehorse, YT. 63 p. + appendices.

Arctic Borderlands Ecological Knowledge Co-op. 2006. Proceedings of the tenth annual gathering. Old Crow, YT. March 18-19, 2006. Arctic Borderlands Ecological Knowledge Society. Whitehorse, YT. 40 p.

Arctic Climate Impact Assessment. 2005. Arctic Climate Impact Assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. 1042 p.

Arft, A.M., Walker, M.D., Turner, P.L., Gurevitch, J., Alatalo, J.M., Molau, U., Nordenhäll, U., Stenström, A., Stenström, M., Bret-Harte, M.S., Dale, M., Diemer, M., Gugerli, F. and Henry, G.H.R. 1999. Responses of tundra plants to experimental warming: meta-analysis of the International Tundra Experiment. Ecological Monographs 69:491-511.

ARKTIS Solutions. 2011. Report on current state of solid waste management and facilities in Nunavut and cost-benefit analysis of selected solid waste management approaches. Submitted to the Government of Nunavut, Community and Government Services. 83 p.

Ashford, G. and Castleden, J. 2001. Inuit observations on climate change: final report. International Institute for Sustainable Development. 27 p.

Audubon Society. 2007. State of the birds: common species in decline [online]. Christmas Bird Count (CBC) data for 1996-2006. http://stateofthebirds.audubon.org/cbid/profile.php?id=3 (accessed 14 February, 2008).

Ayroud, M., Leighton, F.A. and Tessaro, S.V. 1995. The morphology and pathology of *Besnoitia* sp. in reindeer (*Rangifer tarandus tarandus*). Journal of Wildlife Diseases 31:319-326.

Baffinland Iron Mines Corporation. 2012. Mary River Project: final environmental impact statement.

Bahr, D.B., Dyurgerov, M. and Meier, M.F. 2009. Sea-level rise from glaciers and ice caps: a lower bound. Geophysical Research Letters 36 (L03501):1-4.

Baker, A.J., González, P.M., Piersma, T., Niles, L.J., de Lima Serrano do Nascimento, I., Atkinson, P.W., Clark, N.A., Minton, C.D.T., Peck, M.K. and Aarts, G. 2004. Rapid population decline in red knots: fitness consequences of decreased refuelling rates and late arrival in Delaware Bay. Proceedings of the Royal Society of London, Series B: Biological Sciences 271:875-882.

Bamzai, A.S. 2003. Relationship between snow cover variability and Arctic Oscillation index on a hierarchy of time scales. International Journal of Climatology 23:131-142.

Banfield, A.W.F. 1961. A revision of the reindeer and caribou, genus *Rangifer*. National Museum of Canada Bulletin No. 177. Queen's Printer. Ottawa, ON. 137 p.

Barr, W. 1991a. The commercial trade in muskox hides in the Northwest Territories 1860-1916. Rangifer 11:81-82.

Barr, W. 1991b. Back from the brink: the road to muskox conservation in the Northwest Territories. Komatik Series 3:1-127.

Bart, J., Andres, B., Brown, S., Donaldson, G., Harrington, B.A., Johnston, V., Jones, S., Morrison, R.I.G. and Skagen, S. 2005. The Program for Regional and International Shorebird Monitoring (PRISM). USDA Forest Service General Technical Report No. PSW-GTR-191. 901 p.

Bart, J., Brown, S., Harrington, B. and Morrison, R.I.G. 2007. Survey trends of North American shorebirds: population declines or shifting distributions? Journal of Avian Biology 38:73-82.

Bart, J. and Earnst, S.L. 2004. Methods for shorebird surveys in the Arctic. *In* Proceedings of the third annual Partners in Flight International Symposium. Edited by Ralph, C.J. and Rich, T. United States Department of Agriculture Forest Service General Technical Report PSW-GTR-191. Asilomar, CA. pp. 907-917.

Bart, J. and Johnston, V. (eds). 2012. Arctic shorebirds in North America: a decade of monitoring. Studies in Avian Biology Monograph Series No. 44. University of California Press. Berkeley, CA. In press.

Basille, M., Fortin, D., Dussault, C., Ouellet, J.P. and Courtois, R. 2013. Ecologically based definition of seasons clarifies predator prey interactions. Ecography 36:220-229.

Batzli, G.O. and Sobaski, S.T. 1980. Distribution, abundance, and foraging patterns of ground-squirrels near Atkasook, Alaska. Arctic and Alpine Research 12:501-510.

BC Conservation Data Centre. 2007. BC species and ecosystems explorer [online]. BC Ministry of Environment. http://srmapps.gov.bc.ca/apps/eswp/ (accessed 19 February, 2008).

Beauvais, G.P. and Buskirk, S. 1999. Modifying estimates of sampling effort to account for sprung traps. Wildl. Soc. Bull. 27:39-43.

Beck, P.S.A. and Goetz, S.J. 2011. Satellite observations of high northern latitude vegetation productivity changes between 1982 and 2008: ecological variability and regional differences. Environmental Research Letters 6.

Behinke, R.H. 2000. Equilibrium and non-equilibrium models of livestock population dynamics in pastoral Africa: their relevance to arctic grazing systems. Rangifer Special Issue 20:141-152.

Berger, T.R. 1977. Northern frontier, northern homeland: the report of the Mackenzie Valley pipeline inquiry (volume 2, terms and conditions). Supply and Services Canada. Ottawa, ON. 268 p.

Bergerud, A.T., Luttich, S.N. and Camps, L. 2008. The return of the caribou to Ungava. McGill-Queen's University Press. Montréal, QC. 586 p.

Berkes, F. 1998. Indigenous knowledge and resource management systems in the Canadian Subarctic. *In* Linking social and ecological management systems: management practices and social mechanisms. Edited by Folke, C. Cambridge University. Cambridge, UK. pp. 98-128.

Bertram, D.F., Mackas, D.L. and McKinnell, S.M. 2001. The seasonal cycle revisited: interannual variation and ecosystem consequences. Progress in Oceanography 49:283-307.

Bety, J., Gauthier, G., Korpimaki, E. and Giroux, J.F. 2002. Shared predators and indirect trophic interactions: lemming cycles and arctic-nesting geese. Journal of Animal Ecology 71:88-98.

Bhatt, U.S., Walker, D.A., Raynolds, M.K., Comiso, J.C., Epstein, H.E., Jia, G.S., Gens, R., Pinzon, J.E., Tucker, C.J., Tweedie, C.E. and Webber, P.J. 2010. Circumpolar arctic tundra vegetation change is linked to sea ice decline. Earth Interactions 14:1-20.

Bigelow, N.H., Brubaker, L.B., Edwards, M.E., Harrison, S.P., Prentice, I.C., Anderson, P.M., Andreev, A.A., Bartlein, P.J., Christensen, T.R., Cramer, W., Kaplan, J.O., Lozhkin, A.V., Matveyeva, N.V., Murray, D.F., McGuire, A.D., Razzhivin, V.Y., Ritchie, J.C., Smith, B., Walker, D.A., Gajewski, K., Wolf, V., Holmqvist, B.H., Igarashi, Y., Kremenetskii, K., Paus, A., Pisaric, M.F.J. and Volkova, V.S. 2003. Climate change and Arctic ecosystems: 1. vegetation changes north of 55°N between the last glacial maximum, mid-Holocene, and present. Journal of Geophysical Research 108 (D19):(11-1)-(11-25).

Bilodeau, F., Reid, D., Gauthier, G., Krebs, C.J., Berteaux, D. and Kenny, A. 2013a. Demographic response of tundra small mammals to a snow fencing experiment. Oikos . In press.

Bilodeau, F., Gauthier, G. and Berteaux, D. 2013b. The effect of snow cover on lemming population cycles in the Canadian high Arctic. Oecologia . In press.

BirdLife International. 2010. State of the world's birds. [online]. BirdLife International. http://www.biodiversityinfo.org (accessed 28 April, 2010).

Bjork, R.G. and Molau, U. 2007. Ecology of alpine snowbeds and the impact of global change. Arctic Antarctic and Alpine Research 39:34-43.

Black, S. (ed.). 2002. Canada's Western Arctic. Western Arctic Handbook Committee. Inuvik.

Blake, J.E., McLean, B.D. and Gunn, A. 1991. Yersiniosis in free-ranging muskoxen on Banks Island, Northwest Territories, Canada. Journal of Wildlife Diseases 27:527-533.

Blanco, D.E., Yorio, P., Petracci, P.F. and Pugnali, G. 2006. Distribution and abundance of non-breeding shorebirds along the coasts of the Buenos Aires Province, Argentina. Waterbirds 29:381-390.

Bliss, L.C. and Matveyeva, N. 1992. Circumpolar Arctic vegetation. *In* Arctic ecosystems in a changing climate: an ecophysiological perspective. Edited by Chapin III, F.S., Jefferies, R.L., Reynolds, J.F., Shaver, G.R., Svoboda, J. and Chu, E.W. Academic Press. San Diego, CA. pp. 59-89.

Boertmann, D., Forchhammer, M., Olesen, C.R., Aastrup, P. and Thing, H. 1992. The Greenland muskox population status 1990. Rangifer 12:5-12.

Bolen, E. 1998. Tundra. *In* Ecology of North America. John Wiley & Sons Inc. New York, NY. Chapter 2. pp. 19-64.

Bone, R.M. 1992. The geography of the Canadian North. Oxford University Press. Toronto. 284 p.

Bonesteel, S. 2006. Canada's relationship with Inuit: A history of policy and program development. Edited by Anderson, E . Prepared by Pulblic History Inc. for Indian and Northern Affairs Canada. 237 p.

Bonsal, B. and Shabbar, A. 2011. Large-scale climate oscillations influencing Canada, 1900-2008. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 4. Canadian Councils of Resource Ministers. Ottawa, ON. iii + 15 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Bonsal, B.R., Prowse, T.D., Duguay, C.R. and Lacroix, M.P. 2006. Impacts of large-scale teleconnections on freshwater-ice break/freeze-up dates over Canada. Journal of Hydrology 330:340-353.

Boonstra, R., Krebs, C.J. and Kenney, A. 1996. Why lemmings have indoor plumbing in summer. Canadian Journal of Zoology-Revue Canadienne de Zoologie 74:1947-1949.

Boulanger, J., Gunn, A., Adamczewski, J. and Croft, B. 2011a. A data-driven demographic model to explore the decline of Bathurst Caribou Herd. Journal of Wildlife Management 75:883-896.

Boulanger, J., Gunn, A., Adamczewski, J. and Croft, B. 2011b. Estimation of population size for wolverines at Daring Lake, Northwest Territories, using DNA based mark-recapture methods. Journal of Wildlife Management (Suppl. 2) 75:883-896.

Boulanger, J. and Mulders, R. 2013. Analysis of wolverine DNA mark-recapture sampling at Daring Lake, Diavik, and Ekati, Northwest Territories, from 2005 to 2011. Integrated Ecological Research, Nelson, BC, and Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 28 p.

Boulanger, J., Poole, K., Fournier, B., Wierzchoswski, J., Gaines, T. and Gunn, A. 2004. Assessment of Bathurst caribou movements and distribution in the Slave Geological Province. Manuscript Report No. 158. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 108 p.

BQCMB. 2004. Protecting calving grounds, post-calving areas and other important habitats for Beverly and Qamanirjuaq caribou: a position paper. BQCMB annual report, 2007-2008. Beverly and Qamanirjuaq Caribou Management Board. Stonewall, MB. 26 p.

BQCMB. 2010. Mining interests on the caribou ranges [online]. Beverly and Qamanirjuaq Caribou Management Board. http://www.arctic-caribou.com/mining.html (accessed 11 April, 2011).

BQCMB. 2011. BQCMB caribou workshop February 2010 overview report: challenges facing the Beverly and Qamanirjuaq caribou herds and some possible solutions. Beverly and Qamanirjuaq Caribou Management Board. Stonewall, MB. 26 p.

Bradley, M., Gunn, A. and Dragon, J. 2001. Numbers and abundance of muskoxen east of Artillery Lake, NWT. July 1998. Manuscript Report No. 141. Department of Environment and Natural Resources, Northwest Territories.

Brooks, D.R. and Hoberg, E.P. 2007. How will global climate change affect parasite-host assemblages? Trends in Parasitology 23:571-574.

Broughton, E., Choquette, L.P.E., Cousineau, J.G. and Miller, F.L. 1970. Brucellosis in reindeer, *Rangifer tarandus* L., and the migratory barren-ground caribou, *Rangifer tarandus groenlandicus* (L.), in Canada. Canadian Journal of Zoology 48:1023-1027.

Brouwer, J., Baker, N.E. and Trolliet, B. 2003. Estimating bird population sizes and trends: what are the hard data, what are the unavoidable assumptions? A plea for good documentation. Wader Study Group Bulletin 100:197-201.

Brown, G.S., Landriault, L., Sleep, D.J.H. and Mallory, F.F. 2007a. Comment arising from a paper by Wittmer et al.: hypothesis testing for top-down and bottom-up effects in woodland caribou population dynamics. Oecologia 154:485-492.

Brown, J., Hinkel, K.M. and Nelson, F.E. 2000. The Circumpolar Active Layer Monitoring (CALM) program: research designs and initial results. Polar Geography 24:166-258.

Brown, R., Derksen, C. and Wang, L. 2007b. Assessment of spring snow cover duration variability over Northern Canada from satellite datasets. Remote Sensing of Environment 111:367-381.

Brown, R.D. and Braaten, R.O. 1998. Spatial and temporal variability of Canadian monthly snow depths, 1946-1995. Atmosphere-Ocean 36:37-54.

Brown, R.D. and Mote, P.W. 2009. The response of Northern Hemisphere snow cover to a changing climate. Journal of Climate 22:2124-2145.

Brown, S., Bart, J., Lanctot, R.B., Johnson, J.A., Kendall, S., Payer, D. and Johnson, J. 2007c. Shorebird abundance and distribution on the coastal plain of the Arctic National Wildlife Refuge. Condor 109:1-14.

Brown, S.C., Hickey, C., Harrington, B.A. and Gill, R.E. 2001. United States shorebird conservation plan. 2nd Edition. Manomet Centre for Conservation Sciences. Manomet, MA. 64 p.

Buermann, W., Anderson, B., Tucker, C.J., Dickinson, R.E., Lucht, W., Potter, C.S. and Myneni, R.B. 2003. Interannual covariability in Northern Hemisphere air temperatures and greenness associated with El Niño Southern Oscillation and the Arctic Oscillation. Journal of Geophysical Research Atmospheres 108.

Bureau of Land Management. 2012. National Petroleum Reserve-Alaska: Final integrated activity plan/environmental impact statement. US Department of the Interior. Anchorage, AK.

Burgess, D.O., Sharp, M.J., Mair, D.W.F., Dowdeswell, J.A. and Benham, T.J. 2005. Flow dynamics and iceberg calving rates of Devon Ice Cap, Nunavut, Canada. Journal of Glaciology 51:219-230.

Burn, C.R. and Kokelj, S.V. 2009. The environment and permafrost of the Mackenzie Delta area. Permafrost and Periglacial Processes 20:83-105.

Burn, C.R. and Lewkowicz, A.G. 1990. Canadian Landform Examples .17. Retrogressive thaw slumps. Canadian Geographer-Geographe Canadien 34:273-276.

Burn, C.R. and Zhang, Y. 2009. Permafrost and climate change at Herschel Island (Qikiqtaruq), Yukon Territory, Canada. Journal of Geophysical Research-Earth Surface 114.

Burn, D.H. 2008. Climatic influences on streamflow timing in the headwaters of the Mackenzie River Basin. Journal of Hydrology 352:225-238.

Burnham, K.K. and Newton, I. 2011. Seasonal movements of *Gyrfalcons falco* rusticolus include extensive periods at sea. Ibis 153:468-484.

Butcher, G.S. and Niven, D.K. 2007. Combining data from the Christmas Bird Count and the Breeding Bird Survey to determine the continental status and trends of North American birds. National Audubon Society. Ivyland, PA. 34 p.

Buttler, I. 2009. Avian cholera among Arctic breeding common eiders, *Somateria mollissima,* temporal dynamics and the role of handling stress on reproduction and survival. Thesis (M.Sc.). Carleton University. Ottawa, ON. 83 p.

Caccianiga, M. and Payette, S. 2006. Recent advance of white spruce (*Picea glauca*) in the coastal tundra of the eastern shore of Hudson Bay (Quebec, Canada). Journal of Biogeography 33:2120-2135.

CAFF. 2013. Arctic biodiversity assessment: status and trends in Arctic biodiversity. Conservation of Arctic Flora and Fauna. Akureyri, Iceland. S95 p.

Callaghan, T., Bjorn, L.O., Chapin III, F.S., Chernov, Y., Christensen, T.R., Huntley, B., Ims, R., Johansson, M., Riedlinger, D.J., Jonasson, S., Matveyeva, N., Oechel, W.C., Panikov, N. and Shaver, G.R. 2005a. Arctic tundra and polar desert ecosystems. *In* Arctic climate impact assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. Chapter 7. pp. 243-352.

Callaghan, T.V., Bjorn, L.O., Chapin III, F.S., Chernov, Y., Christensen, T.R., Huntley, B., Ims, R.A., Johansson, M., Riedlinger, D.J., Jonasson, S., Matveyeva, N.V., Oechel, W., Panikov, N., Shaver, G.R., Elster, J., Henttonen, H., Jonsdottir, I.S., Laine, K., Schaphoff, S., Taulavuori, E., Taulavuori, K. and Zockler, C. 2005b. Arctic tundra and polar desert ecosystems. *In* Arctic climate impact assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. Chapter 7. pp. 243-352.

Callaghan, T.V., Johansson, M., Brown, R.D., Groisman, P.Y., Labba, N., Radionov, V., Barry, R.G., Bulygina, O.N., Essery, R.L.H., Frolov, D.M., Golubev, V.N., Grenfell, T.C., Petrushina, M.N., Razuvaev, V.N., Robinson, D.A., Romanov, P., Shindell, D., Shmakin, A.B., Sokratov, S.A., Warren, S. and Yang, D.Q. 2011a. The changing face of arctic snow cover: a synthesis of observed and projected changes. Ambio 40:17-31.

Callaghan, T.V., Johansson, M., Brown, R.D., Groisman, P.Y., Labba, N., Radionov, V., Bradley, R.S., Blangy, S., Bulygina, O.N., Christensen, T.R., Colman, J.E., Essery, R.L.H., Forbes, B.C., Forchhammer, M.C., Golubev, V.N., Honrath, R.E., Juday, G.P., Meshcherskaya, A.V., Phoenix, G.K., Pomeroy, J., Rautio, A., Robinson, D.A., Schmidt, N.M., Serreze, M.C., Shevchenko, V.P., Shiklomanov, A.I., Shmakin, A.B., Skold, P., Sturm, M., Woo, M.K. and Wood, E.F. 2011b. Multiple effects of changes in arctic snow cover. Ambio 40:32-45.

Callaghan, T.V., Tweedie, C.E., Akerman, J., Andrews, C., Bergstedt, J., Butler, M.G., Christensen, T.R., Cooley, D., Dahlberg, U., Danby, R.K., Daniels, F.J.A., de Molenaar, J.G., Dick, J., Mortensen, C.E., Ebert-May, D., Emanuelsson, U., Eriksson, H., Hedenas, H., Henry, G.H.R., Hik, D.S., Hobbie, J.E., Jantze, E.J., Jaspers, C., Johansson, C., Johansson, M., Johnson, D.R., Johnstone, J.F., Jonasson, C., Kennedy, C., Kenney, A.J., Keuper, F., Koh, S., Krebs, C.J., Lantuit, H., Lara, M.J., Lin, D., Lougheed, V.L., Madsen, J., Matveyeva, N., Mcewen, D.C., Myers-Smith, I.H., Narozhniy, Y.K., Olsson, H., Pohjola, V.A., Price, L.W., Riget, F., Rundqvist, S., Sandstrom, A., Tamstorf, M., Van Bogaert, R., Villarreal, S., Webber, P.J. and Zemtsov, V.A. 2011c. Multi-decadal changes in tundra environments and ecosystems: synthesis of the international polar year-back to the future project (IPY-BTF). Ambio 40:705-716.

Cameron, R.D., Smith, W.T., White, R.G. and Griffith, B. 2005. Central Arctic caribou and petroleum development: distributional, nutritional, and reproductive implications. Arctic 58:1-9.

Campbell, M. and Setterington, M. 2001. The re-evaluation of Kivalliq and Northeast Kitikmeot muskox (*Ovibos moschatus*) populations, management zones, and quotas. Technical Report Series 2001 No. 1-02. Department of Sustainable Development, Nunavut Wildlife Service. Arviat, NU. 97 p.

Canadian Wildlife Service. 2003. Greater snow goose [Hinterland who's who fact sheet]. Canadian Wildlife Service. 7 p.

Canadian Wildlife Service Waterfowl Committee. 2008. Population status of migratory game birds in Canada, November 2008. CWS Migratory Birds Regulatory Report No. 25. Environment Canada. Ottawa, ON. 92 p.

Canadian Wildlife Service Waterfowl Committee. 2012. Population status of migratory game birds in Canada - November 2012. CWS Migratory Birds Regulatory Report No. 37. Environment Canada. Gatineau QC.

Cannings, S., Anions, M., Rainier, R. and Stein, B. 2013. Our home and native land: Canadian species of global conservation concern. NatureServe Canada. Ottawa, ON. 39 p.

Carmichael, L.E., Krizan, J., Nagy, J.A., Dumond, M., Johnson, D., Veitch, A. and Strobeck, C. 2008. Northwest passages: conservation genetics of arctic island wolves. Conservation Genetics 9:879-892.

Carmichael, L.E., Krizan, J., Nagy, J.A., Fuglei, E., Dumond, M., Johnson, D., Veitch, A., Berteaux, D. and Strobeck, C. 2007. Historical and ecological determinants of genetic structure in arctic canids. Molecular Ecology 16:3466-3483.

Carmichael, L.E., Nagy, J.A., Larter, N.C. and Strobeck, C. 2001. Prey specialization may influence patterns of gene flow in wolves of the Canadian Northwest. Molecular Ecology 10:2787-2798.

Carroll, M.L., Townshend, J.R.G., DiMiceli, C.M., Loboda, T. and Sohlberg, R.A. 2011. Shrinking lakes of the Arctic: spatial relationships and trajectory of change. Geophysical Research Letters 38.

Case, R. and Stevenson, J. 1991. Observation of barren-ground grizzly bear, *Ursus arctos*, Predation on Muskoxen, *Ovibos moschatus*, in the Northwest-Territories. Canadian Field-Naturalist 105:105-106.

Caughley, G. and Gunn, A. 1993. Dynamics of large herbivores in deserts - kangaroos and caribou. Oikos 67:47-55.

CCEA. 2009. Conservation Areas Reporting and Tracking System (CARTS), v.2009.05 [online]. Canadian Council on Ecological Areas. http://ccea.org/en_carts.html (accessed 5 November, 2009).

Chan, J., Baxter, C. and Wenman, W.M. 1989. Brucellosis in an Inuit child, probably related to caribou meat consumption. Scandinavian Journal of Infectious Diseases 21:337-338.

Chapin III, F.S., Sturm, M., Serreze, M.C., McFadden, J.P., Key, J.R., Lloyd, A.H., McGuire, A.D., Rupp, T.S., Lynch, A.H., Schimel, J.P., Beringer, J., Chapman, W.L., Epstein, H.E., Euskirchen, E.S., Hinzman, L.D., Jia, G., Ping, C.L., Tape, K.D., Thompson, C.D.C., Walker, D.A. and Welker, J.M. 2005. Role of land-surface changes in Arctic summer warming. Science 310:657-660.

Chapin, F.S., Shaver, G.R., Giblin, A.E., Nadelhoffer, K.J. and Laundre, J.A. 1995. Responses of arctic tundra to experimental and observed changes in climate. Ecology 76:694-711.

Chen, W., Russell, D., Gunn, A., Croft, B., Chen, W.R., Fernandes, R., Zhao, H., Li, J., Zhang, Y., Koehler, K., Ithof, I., Fraser, R.H., Leblanc, S.G., Henry, G.R., White, R.G. and Finstad, G.L. 2013. Monitoring habitat condition changes during winter and pre-calving migration for Bathurst caribou in northern Canada. Biodiversity 14:36-44.

Choquette, L.P.E., Broughton, E., Miller, F.L., Gibbs, H.C. and Cousineau, J.G. 1967. Besnoitiosis in barren-ground caribou in Northern Canada. Canadian Veterinary Journal 8:282-287.

Chouinard, C., Fortier, R. and Mareschal, J.C. 2007. Recent climate variations in the subarctic inferred from three borehole temperature profiles in northern Quebec, Canada. Earth and Planetary Science Letters 263:355-369.

Chubbs, T.E. and Brazil, J. 2007. The occurrence of muskoxen, *Ovibos moshatus*, in Labrador. Canadian Field-Naturalist, Vol. 121, pp. 81- 84.

Chubbs, T.E. and Schaefer, J.A. 1997. Population growth of moose (*Alces alces*) in Labrador. Canadian Field Naturalist 111:238-242.

Clarkson, P.L. and Liepins, I.S. 1994. Grizzly bear population estimate and characteristics in the Anderson and Horton rivers area, Northwest Territories, 1987-89. *In* Part 1: A Selection of Papers from the Ninth International Conference on Bear Research and Management. Bears: their biology and management. Missoula, MT. 23-28 February 1992. pp. 213-221.

Clein, J.S., Kwiatkowski, B.L., McGuire, A.D., Hobbie, J.E., Rastetter, E.B., Melillo, J.M. and Kicklighter, D.W. 2000. Modelling carbon responses of tundra ecosystems to historical and projected climate: a comparison of a plot and a global-scale ecosystem model to identify process-based uncertainties. Global Change Biology 6:127-140.

Cluff, D. 2004. Department of Environment and Natural Resources, Government of the Northwest Territories. Personal communication.

Coe, J.M. and Rogers, D.B. 1997. Marine debris: sources, impacts and solutions. Springer Publishers. New York, NY. 432 p.

Colgan, W. and Sharp, M. 2008. Combined oceanic and atmospheric influences on net accumulation on Devon Ice Cap, Nunavut, Canada. Journal of Glaciology 54:28-40.

Community of Paulatuk, Wildlife Management Advisory Council (NWT) and Joint Secretariat. 2008. Paulatuk community conservation plan. 142 p.

Condon, R.G., Collings, P. and Wenzel, G. 1995. The best part of life: subsistence hunting, ethnicity, and economic adaptation among young adult Inuit males. Arctic 48:31-46.

Coral Harbour. 2011. Coral Harbour, the legendary Canadian North [online]. Coral Harbour. http://www.coralharbour.ca/coralharbour_wildlife.html (accessed 21 May, 2011).

Cornelissen, J.H.C., van Bodegom, P.M., Aerts, R., Callaghan, T.V., Van Logtestijn, R.S., Alatalo, J.M., Chapin, F.S., III, Gerdol, R., Gudmundsson, J., Gwynn-Jones, D., Hartley, A.E., Hik, D.S., Hofgaard, A., Jonsdottir, I., Karlsson, S., Klein, J.A., Laundre, J., Magnusson, B., Michelsen, A., Molau, A., Onipchenko, V.G., Quested, H.M., Sandvik, S.M., Schmidt, I.K., Shaver, G.R., Solheim, B., Soudzilovskaia, N.A., Stenstrom, A., Tolvanen, A., Totland, O., Wada, N., Welker, J.M. and Zhao, X. 2007. Global negative vegetation feedback to climate warming responses of leaf litter decomposition rates in cold biomes. Ecology Letters 10:619-627.

COSEWIC. 2003. COSEWIC assessment and update status report on the wolverine *Gulo gulo* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. 41 p.

COSEWIC. 2004. COSEWIC assessment and update status report on the Peary caribou *Rangifer tarandus pearyi* and the barren-ground caribou *Rangifer tarandus groenlandicus* (Dolphin and Union population) in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. 91 p.

COSEWIC. 2008. COSEWIC assessment and update status report on the polar bear *Ursus maritimus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. vii + 75 p.

COSEWIC. 2011. Designatable units for caribou (*Rangifer tarandus*) in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. 88 p.

COSEWIC. 2012. COSEWIC assessment and status report on the grizzly bear *Ursus arctos* in Canada. Committtee on the Status of Endangered Wildlife in Canada. Ottawa. xiv + 84 p. (www.registrelep-sararegistry.gc.ca/default_e.cfm).

Couturier, S., Côté, S.D., Huot, J. and Otto, R. 2009a. Body-condition dynamics in a northern ungulate gaining fat in winter. Canadian Journal of Zoology 87:367-378.

Couturier, S., Côté, S.D., Otto, R., Weladji, R.B. and Huot, J. 2009b. Variation in calf body mass in migratory caribou: the role of habitat, climate, and movements. Journal of Mammalogy 90:442-452.

Crawford, R.M.M. 2008. Plants at the margin: ecological limits and climate change. Cambridge University Press. Cambridge, UK. 478 p.

Culp, J.M., Prowse, T.D. and Luiker, E.A. 2005. Mackenzie River Basin. *In* Rivers of North America. Edited by Benke, A.C. and Cushing, C.E. Elsevier Academic Press. London, UK. Chapter 18. pp. 805-849.

Damas, D. 2002. Arctic migrants/Arctic villagers: the transformation of Inuit settlement in the central Arctic. McGill-Queen's University Press. Montreal, QC and Kingston, ON. McGill-Queen's native and northern series: 32. 336 p.

Danks, H.V. 1992. Arctic insects as indicators of environmental change. Arctic 45:159-166.

Danks, H.V. 2004. Seasonal adaptations in arctic insects. Integrative and Comparative Biology 44:85-94.

Davison, T., Pongracz, J. and Williams, J. In Prep. Caribou and muskox survey on Banks Island and northwest Victoria Island, 2010 summary. Department of Environment and Natural Resources, Government of the Northwest Territories. Inuvik, NT.

Davison, T., Pongracz, J. and Williams, J. 2013. Population survey of Peary caribou (*Rangifer tarandus pearyi*) and muskoxen (*Ovibos moschatus*) on Banks Island, Northwest Territories, July 2010. Rangifer 33:135-140.

Davison, T. and Williams, J. 2012. Caribou and muskoxen survey on Melville and Prince Patrick Island, 2012 summary. Community summary. Environment and Natural Resources, Government of the Northwest Territories. Inuvik, NT. 6 p. Provided with comments on SARC report (draft 3) on Peary Caribou. October, 2012.

Davison, T. and Williams, J. 2013. Peary caribou (*Rangifer tarandus pearyi*) and muskoxen (*Ovibos moschatus*) on northwest Victoria Island, Northwest Territories. Rangifer 33:129-134.

Dawson, J. 2012. Cruise tourism in Arctic Canada [online]. University of Ottawa. http://artsites.uottawa.ca/ctac/ (accessed 7 July, 2013).

Delany, S. and Scott, D. 2006. Waterbird population estimates. 4th Edition. Wetlands International. Wageningen, Netherlands. viii + 23 p.

Department of Environment. 2013a. Statutory report on wildlife to the Nunavut Legislative Assembly, Section 176 of the Wildlife Act. Government of Nunavut. Iqaluit, NU. 75 p.

Department of Environment. 2013b. Working together for Baffin Island caribou: workshop report. Government of Nunavut.

Department of Environment and Natural Resources. 2006. Caribou forever - our heritage, our responsibility: a barren-ground caribou management strategy for the Northwest Territories 2006-2010. Department of Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 38 p.

Derksen, C. and Brown, R. 2012. Spring snow cover extent reductions in the 2008-2012 period exceeding climate model projections. Geophysical Research Letters 39:1-6.

Derocher, A.E., Lunn, N.J. and Stirling, I. 2004. Polar bears in a warming climate. Integrative and Comparative Biology 44:163-176.

Déry, S.J., Hernandez-Henriquez, M.A., Burford, J.E. and Wood, E.F. 2009. Observational evidence of an intensifying hydrological cycle in northern Canada. Geophysical Research Letters 36 L13402.

Déry, S.J., Mlynowski, T.J., Hernandez-Henriquez, M.A. and Straneo, F. 2011. Interannual variability and interdecadal trends in Hudson Bay streamflow. Journal of Marine Systems 88:341-351.

Déry, S.J., Stieglitz, M., McKenna, E.C. and Wood, E.F. 2005. Characteristics and trends of river discharge into Hudson, James, and Ungava bays, 1964-2000. Journal of Climate 18:2540-2557.

Déry, S.J. and Wood, E.F. 2005. Decreasing river discharge in northern Canada. Geophysical Research Letters 32, L10401, 4 p.

Descamps, S., Jenouvrier, S., Gilchrist, H.G. and Forbes, M.R. 2012. Avian cholera, a threat to the viability of an Arctic seabird colony? PLoS One 7.

Deslippe, J.R., Egger, K.N. and Henry, G.H.R. 2005. Impacts of warming and fertilization on nitrogen-fixing microbial communities in the Canadian High Arctic. FEMS microbiology ecology 53:41-50.

Deslippe, J.R., Hartmann, M., Mohn, W.W. and Simard, S.W. 2011. Long-term experimental manipulation of climate alters the ectomycorrhizal community of *Betula nana* in Arctic tundra. Global Change Biology 17:1625-1636.

Deslippe, J.R., Hartmann, M., Simard, S.W. and Mohn, W.W. 2012. Long-term warming alters the composition of Arctic soil microbial communities. FEMS microbiology ecology 82:303-315.

Dickey, M.H., Gauthier, G. and Cadieux, M.C. 2008. Climatic effects on the breeding phenology and reproductive success of an arctic-nesting goose species. Global Change Biology 14:1973-1985.

Dietz, R., Sonne, C., Basu, N., Braune, B., O'Hara, T., Letcher, R.J., Scheuhammer, T., Andersen, M., Andreasen, C., Andriashek, D., Asmund, G., Aubail, A., Baagoe, H., Born, E.W., Chan, H.M., Derocher, A.E., Grandjean, P., Knott, K., Kirkegaard, M., Krey, A., Lunn, N., Messier, F., Obbard, M., Olsen, M.T., Ostertag, S., Peacock, E., Renzoni, A., Riget, F.F., Skaare, J.U., Stern, G., Stirling, I., Taylor, M., Wiig, O., Wilson, S. and Aars, J. 2013. What are the toxicological effects of mercury in Arctic biota? Science of the Total Environment 443:775-790.

DND. 2001. The distant early warning line clean up project [online]. Department of National Defence. http://www.forces.gc.ca/site/news-nouvelles/view-news-afficher-nouvelles-eng.asp?id=205 (accessed 17 March, 2009).

Donaldson, G., Hyslop, C., Morrison, R.I.G., Dickson, D.L. and Davidson, I. 2000. Canadian shorebird conservation plan. Canadian Wildlife Service, Environment Canada. Ottawa, ON. ii + 28 p.

Donaldson, S.G., Van Oostdam, J., Tikhonov, C., Feeley, M., Armstrong, B., Ayotte, P., Boucher, O., Bowers, W., Chan, L., Dallaire, F., Dallaire, R., Dewailly, E., Edwards, J., Egeland, G.M., Fontaine, J., Furgal, C., Leech, T., Loring, E., Muckle, G., Nancarrow, T., Pereg, D., Plusquellec, P., Potyrala, M., Receveur, O. and Shearer, R.G. 2010. Environmental contaminants and human health in the Canadian Arctic. Science of the Total Environment 408:5165-5234.

Douglas, M.S.V. and Smol, J.P. 1999. Freshwater diatoms as indicators of environmental change in the high Arctic. *In* The diatoms: Applications for the environmental and earth sciences. Edited by Stoermer, E.F. and Smol, J.P. Cambridge University Press. Cambridge, UK. pp. 227-244.

Douglas, M.S.V., Smol, J.P. and Blake, W. 1994. Marked post-18th century environmental change in High Arctic ecosystems. Science 266:416-419.

Downes, C., Blancher, P. and Collins, B. 2011. Landbird trends in Canada, 1968-2006. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 12. Canadian Councils of Resource Ministers. Ottawa, ON. x + 94 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Dubey, J.P. and Beattie, C.P. 1988. Toxoplasmosis of animals and man. CRC Press. Boca Raton, Florida. 220 p.

Dubey, J.P., Reichard, M.V., Torretti, L., Garvon, J.M., Sundar, N. and Grigg, M.E. 2010. Two new species of Sarcocystis (Apicomplexa: Sarcocystidae) infecting the wolverine (*Gulo gulo*) from Nunavut, Canada. Journal of Parasitology 96:972-976.

Duchesne, D., Gauthier, G. and Berteaux, D. 2011. Habitat selection, reproduction and predation of wintering lemmings in the Arctic. Oecologia 167:967-980.

Ducrocq, J., Beauchamp, G., Kutz, S., Simard, M., Taillon, J., Cote, S.D., Brodeur, V. and Lair, S. 2013. Variables associated with besnoitia tarandi prevalence and cyst density in barren-ground caribou (*Rangifer tarandus*) populations. Journal of Wildlife Diseases 49:29-38.

Duerden, F. and Kuhn, R.G. 1998. Scale, context and application of traditional knowledge of the Canadian North. Polar Record 34:31-38.

Dumond, M. 2006. Review of muskox status in the Kitikmeot Region of Nunavut. Interim Wildlife Report No. 6. Government of Nunavut, Department of Environment. Iqaluit. 29 p. Working draft.

Dumond, M. 2007a. Muskox distribution and abundance in the area between Bathurst Inlet and the Coppermine River, Kitikmeot Region, Nunavut (mx19 and west of mx14). Status Report No. 35. Government of Nunavut, Department of Environment. Iqaluit. 26 p.

Dumond, M. 2007b. Muskox distribution and abundance in the area west of the Coppermine River, Kitikmeot Region, Nunavut. Status Report No. 33. Government of Nunavut, Department of Environment. Iqaluit. 31 p.

Dumond, M. 2007c. Muskox distribution and abundance on the Boothia Peninsula, Kitikmeot Region. Status Report No. 34. Government of Nunavut, Department of Environment. Iqaluit. 25 p.

Dumond, M. 2011. Information on Nunavut caribou surveys and harvest levels. Department of Environment, Government of Nunavut. Kugluktuk, NU. Personal communication.

Dumond, M., Algona, B., Bolt, C., Bolt, M., Ivarluk, J., Klengenberg, S., Taktogon, P., Adjun, C., Atatahak, G., Niptanatiak, A., Campbell, M., Torretti, L., Himiak, J. and Taptuna, P. 2009. Western Kitikmeot caribou workshop. Technical Report. Department of Environment, Government of Nunavut. 47 p.

Dunbar, M. 1973. Stability and fragility in arctic ecosystems. Arctic 26:178-185.

Durner, G.M., Douglas, D.C., Nielson, R.M., Amstrup, S.C., McDonald, T.L., Stirling, I., Mauritzen, M., Born, E.W., Wiig, O., DeWeaver, E., Serreze, M.C., Belikov, S.E., Holland, M.M., Maslanik, J., Aars, J., Bailey, D.A. and Derocher, A.E. 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs 79:25-58.

Dye, D.G. 2002. Variability and trends in the annual snow-cover cycle in Northern Hemisphere land areas, 1972-2000. Hydrological Processes 16:3065-3077.

Dyer, S.J., O'Neill, J.P., Wasel, S.M. and Boutin, S. 2002. Quantifying barrier effects of roads and seismic lines on movements of female woodland caribou in Northeastern Alberta. Canadian Journal of Zoology 80:839-845.

Dyke, A.S. 2004. An outline of North American deglaciation with emphasis on central and northern Canada. *In* Quaternary glaciations, extent and chronology. Part II. North America. Edited by Ehlers, J. and Gibbard, P.L. Elsevier. Amsterdam. pp. 373-424.

Dyurgerov, M.B. and Carter, C.L. 2004. Observational evidence of increases in freshwater inflow to the Arctic Ocean. Arctic, Antarctic, and Alpine Research 36:117-122.

Eamer, J., Donaldson, G.M., Gaston, A.J., Kosobokova.K.N., Larusson, K.F., Melnikov, I.A., Reist, J.D., Richardson, E., Staples, L. and von Quillfeldt, C.H. 2013. Life linked to ice: A guide to sea-ice-associated biodiversity in this time of rapid change. CAFF Assessment Series No. 10, October 2013. Conservation of Arctic Flora and Fauna. Iceland. 115 p.

Eamer, J. and Russell, D. (eds.). 2013. CARMA 8, Moving forward: knowledge to action (conference proceedings). CircumArctic Rangifer Monitoring and Assessment Network, December 4-6, 2012. Vancouver, BC.

Ecological Stratification Working Group. 1995. A national ecological framework for Canada. Agriculture and Agri-Food Canada, Research Branch, Centre for Land and Biological Resources Research and Environment Canada, State of the Environment Directorate, Ecozone Analysis Branch. Ottawa, ON/Hull, QC. vii + 125 p.

Edwards, K.A., McCulloch, J., Kershaw, G.P. and Jefferies, R.L. 2006. Soil microbial and nutrient dynamics in a wet arctic sedge meadow in late winter and early spring. Soil Biology and Biochemistry 38:2843-2851.

Edwards, M.A. 2006. Habitat and movement ecology of grizzly bears in the Mackenzie Delta, NWT. Infonorth 453-456.

Elmendorf, S.C., Henry, G.H.R., Hollister, R.D., Bjork, R.G., Bjorkman, A.D., Callaghan, T.V., Collier, L.S., Cooper, E.J., Cornelissen, J.H.C., Day, T.A., Fosaa, A.M., Gould, W.A., Gretarsdottir, J., Harte, J., Hermanutz, L., Hik, D.S., Hofgaard, A., Jarrad, F., Jonsdottir, I.S., Keuper, F., Klanderud, K., Klein, J.A., Koh, S., Kudo, G., Lang, S.I., Loewen, V., May, J.L., Mercado, J., Michelsen, A., Molau, U., Myers-Smith, I.H., Oberbauer, S.F., Pieper, S., Post, E., Rixen, C., Robinson, C.H., Schmidt, N.M., Shaver, G.R., Stenstrom, A., Tolvanen, A., Totland, O., Troxler, T., Wahren, C.H., Webber, P.J., Welker, J.M. and Wookey, P.A. 2012a. Global assessment of experimental climate warming on tundra vegetation: heterogeneity over space and time. Ecology Letters 15:164-175.

Elmendorf, S.C., Henry, G.H., Hollister, R.D., Bjork, R.G., Boulanger-Lapointe, N., Cooper, E.J., Cornelissen, J.H., Day, T.A., Dorrepaal, E., Elumeeva, T.G., Gill, M., Gould, W.A., Harte, J., Hik, D.S., Hofgaard, A., Johnson, D.R., Johnstone, J.F., Jonsdottir, I.S., Jorgenson, J.C., Klanderud, K., Klein, J.A., Koh, S., Kudo, G., Lara, M., Levesque, E., Magnusson, B., May, J.L., Mercado-Diaz, J.A., Michelsen, A., Molau, U., Myers-Smith, I.H., Oberbauer, S.F., Onipchenko, V.G., Rixen, C., Schmidt, N.M., Shaver, G.R., Spasojevic, M.J., Porhallsdottir, P.E., Tolvanen, A., Troxler, T., Tweedie, C.E., Villareal, S., Wahren, C.H., Walker, X., Webber, P.J., Welker, J.M. and Wipf, S. 2012b. Plot-scale evidence of tundra vegetation change and links to recent summer warming. Nature Climate Change 2:453-457.

Elmqvist, T., Berkes, F., Folke, C., Angelstam, P., Crépin, A.S. and Niemelä, J. 2004. The dynamics of ecosystems, biodiversity management and social institutions at high northern latitudes. AMBIO: a Journal of the Human Environment 33:350-355.

Elton, C.S. 1924. Periodic fluctuations in the numbers of animals: their causes and effects. Journal of Experimental Biology 2:119-163.

England, J.H., Lakeman, T.R., Lemmen, D.S., Bednarski, J.M., Stewart, T.G. and Evans, D.J.A. 2008. A millennial-scale record of Arctic Ocean sea ice variability and the demise of the Ellesmere Island ice shelves. Geophysical Research Letters 35.

Environment and Natural Resources. 2012a. Muskox in the NWT [online]. Government of the Northwest Territories. http://www.enr.gov.nt.ca/_live/pages/wpPages/Muskox.aspx (accessed 7 July, 2013a).

Environment and Natural Resources. 2012b. The West Kitikmeot/Slave Study Society [online]. Government of the Northwest Territories. http://www.enr.gov.nt.ca/_live/pages/wpPages/West_Kitikmeot_Slave_Study.aspx (accessed 10 December, 20130b).

Environment and Natural Resources. 2014. Bluenose East Herd [online]. Government of Northwest Territories. http://www.enr.gov.nt.ca/_live/pages/wpPages/Bluenose-East_herd.aspx (accessed 5 January, 2014).

Environment Canada. 2005. Narrative descriptions of terrestrial ecozones and ecoregions of Canada [online]. Environment Canada. http://www.ec.gc.ca/soer-ree/English/framework/NarDesc/canada_e.cfm (accessed 15 May, 2008).

Environment Canada. 2006. Biodiversity outcomes framework for Canada. Canadian Councils of Resource Ministers. Ottawa, ON. 8 p. http://www.biodivcanada.ca/default.asp?lang=En&n=F14D37B9-1.

Environment Canada. 2007a. Arctic habitats threatened by snow geese: Overpopulation problem [online]. http://www.pnr-rpn.ec.gc.ca/nature/migratorybirds/dc00s10.en.html

Environment Canada. 2007b. Recovery strategy for the eskimo curlew (*Numenius borealis*) in Canada. Species at Risk Act Recovery Strategy Series. Environment Canada. Ottawa, ON. v + 10 p.

Environment Canada. 2009a. Unpublished analysis of data by ecozone$^+$ from: Conservation Areas Reporting and Tracking System (CARTS), v.2009.05 [online]. Canadian Council on Ecological Areas. http://ccea.org/en_carts.html (accessed 5 November, 2009a).

Environment Canada. 2009b. Unpublished analysis of population data by ecozone$^+$ from: Statistics Canada Human Activity and the Environment Series, 1971-2006. Community profile data was used to make adjustments due to differences in the ecozone/ecozone+ boundary.

Environment Canada. 2012a. Highlights of recent research: increasing levels of global carbon dioxide [online]. http://www.ec.gc.ca/sc-cs/default.asp?lang=En&n=0EC06FB9-1#2 (accessed 6 June, 2013a).

Environment Canada. 2012b. Rivers [online]. http://www.ec.gc.ca/eau-water/default.asp?lang=En&n=45BBB7B8-1

Environment Canada. 2013a. Canada's emissions trends. Government of Canada. 80 p.

Environment Canada. 2013b. Greenhouse gas emissions data [online]. http://www.ec.gc.ca/indicateurs-indicators/default.asp?n=BFB1B398 (accessed 7 July, 2013b).

Environment Canada. 2013c. National inventory report: Greenhouse gas sources and sinks in Canada 1990-2011. The Canadian Government's submission to the UN Framework Convention on Climate Change. Executive summary. Government of Canada. 16 p.

Erickson, K. and Meegan, S. 2007. Results of the historical waterfowl population and distribution data collection effort: a technical report in support of the Supplemental Environmental Impact Statement (SEIS) for introduction of the FA/18 E/F Hornet aircraft to the east coast of the United States. Prepared for the U.S. Department of the Navy by Ecology and Environment Inc. Norfolk, VA. 94 p.

Euskirchen, E.S., McGuire, A.D. and Chapin, F.S. 2007. Energy feedbacks of northern high-latitude ecosystems to the climate system due to reduced snow cover during 20th century warming. Global Change Biology 13:2425-2438.

Federal, Provincial and Territorial Governments of Canada. 2010. Canadian biodiversity: ecosystem status and trends 2010. Canadian Councils of Resource Ministers. Ottawa, ON. vi + 142 p. http://www.biodivcanada.ca/default.asp?lang=En&n=83A35E06-1.

Federal-Provincial-Territorial Biodiversity Working Group. 1995. Canadian biodiversity strategy: Canada's response to the Convention on Biological Diversity. Environment Canada, Biodiversity Convention Office. Hull, QC. 86 p. http://www.biodivcanada.ca/default.asp?lang=En&n=560ED58E-1.

Ferguson, M.A.D. 1987. Status of Peary caribou and muskox populations on Bathurst Island, N.W.T., August 1981. Arctic 40:131-137.

Ferguson, S.H. and McLoughlin, P.D. 2000. Effect of energy availability, seasonality, and geographic range on brown bear life history. Ecography 23:193-200.

Ferguson, S.H., Stirling, I. and McLoughlin, P. 2005. Climate change and ringed seal *(Phoca hispida)* recruitment in Western Hudson Bay. Marine Mammal Science 21:121-135.

Festa-Bianchet, M., Ray, J.C., Boutin, S., Cote, S. and Gunn, A. 2011. Conservation of caribou (*Rangifer tarandus*) in Canada: an uncertain future. Canadian Journal of Zoology 89:419-434.

Fetterer, F., Knowles, K., Meier, W. and Savoie, M. 2010. Sea ice index [online]. National Snow and Ice Data Center, Boulder, CO. http://nsidc.org/data/g02135.html (accessed 4 October, 2010).

Fischbach, A.S., Amstrup, S.C. and Douglas, D.C. 2007. Landward and eastward shift of Alaskan polar bear denning associated with recent sea ice changes. Polar Biology 30:1395-1405.

Fisher, D., Zheng, J., Burgess, D., Zdanowicz, C., Kinnard, C., Sharp, M. and Bourgeois, J. 2012. Recent melt rates of Canadian arctic ice caps are the highest in four millennia. Global and Planetary Change 84-85:3-7.

Fisk, A.T., de Wit, C.A., Wayland, M., Kuzyk, Z.Z., Burgess, N., Letcher, R., Braune, B., Norstrom, R., Blum, S.P. and Sandau, C. 2005. An assessment of the toxicological significance of anthropogenic contaminants in Canadian arctic wildlife. Science of the Total Environment 57-93. doi:10.1016/j.scitotenv.2005.01.051.

Flanner, M.G. 2013. Arctic climate sensitivity to local black carbon. Journal of Geophysical Research-Atmospheres 118:1840-1851.

Flanner, M.G., Zender, C.S., Hess, P.G., Mahowald, N.M., Painter, T. and Ramanathan, V.R.P.J. 2008. Springtime warming and reduced snow cover from carbonaceous particles. Atmospheric Chemistry and Physics Discussions 8:19819-19859.

Forbes, B.C. 1996. Plant communities of archaeological sites, abandoned dwellings, and trampled tundra in the eastern Canadian arctic: A multivariate analysis. Arctic 49:141-154.

Forbes, B.C. 2001. Anthropogenic disturbance and patch dynamics in circumpolar arctic ecosystems (vol 15, pg 958, 2001). Conservation Biology 15:954-969.

Forbes, L.B. 1991. Isolates of *Brucella suis* biovar 4 from animals and humans in Canada, 1982-1990. The Canadian Veterinary Journal 32:686-688.

Ford, J.D., Smit, B., Wandel, J. and MacDonald, J. 2006. Vulnerability to climate change in Igloolik, Nunavut: what we can learn from the past and present. Polar Record 42:127-138.

Ford, N., Campbell, M. and Lee, D. 2012. A final report to the Nunavut Wildlife Management Board on the re-evaluation of muskoxen (*Ovibos moschatus*) populations in the Kivaliq Region No. 2-10-08. Prepared by: The Rankin Inlet HTO, Nunavut Department of Environment Kivalliq Region, and Nunavut Tunngavik Inc. Rankin Inlet, NU. 21 p.

Fortier, D., Allard, M. and Shur, Y. 2007. Observation of rapid drainage system development by thermal erosion of ice wedges on Bylot Island, Canadian Arctic Archipelago. Permafrost and Periglacial Processes 18:229-243.

Fournier, B. and Gunn, A. 1998. Muskox numbers and distribution in the Northwest Territories, 1997. File Report No. 121. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NWT.

Fox, S. 1998. Inuit knowledge of climate and climate change. Thesis (M.A.). University of Waterloo. Waterloo, ON.

Fox, S. 2004. When the weather is uggianaqtuq: linking inuit and scientific observations of recent environmental change in Nunavut, Canada. Thesis (Ph.D.). University of Colorado. Boulder, CO.

Frame, P.F., Hik, D.S., Cluff, H.D. and Paquet, P.C. 2004. Long foraging movement of a denning tundra wolf. Arctic 57:196-203.

Freeman, M. 1992. The nature and variability of Traditional Ecological Knowledge. Northern Perspectives 20:9-12.

Freeman, M.M.R. 2012. Aboriginal people: Arctic [online]. The Canadian Encyclopedia, Historica-Dominion. http://www.thecanadianencyclopedia.com/articles/native-people-arctic (accessed 1 December, 2012).

Friesen, T.M. and Arnold, C.M. 2008. The timing of the Thule migration: new dates from the Western Canadian Arctic. American Antiquity 73:527-538.

Friesen, T.M. 2004. Contemporaneity of Dorset and Thule cultures in the North American Arctic: new radiocarbon dates from Victoria Island, Nunavut. Current Anthropology 45:685-691.

Frisk, J. 2011. Guidance for the preparation of ESTR products - land classification scheme. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 3. Canadian Councils of Resource Ministers. Ottawa, ON. iv + 34 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Fujimura, K.E., Egger, K.N. and Henry, G.H. 2008. The effect of experimental warming on the root-associated fungal community of *Salix arctica*. Isme Journal 2:105-114.

Furgal, C., Martin, D. and Gosselin, P. 2002. Climate change and health in Nunavik and Labrador: lessons from Inuit knowledge. *In* The earth is faster now: Indigenous observations of Arctic environmental change. Edited by Krupnik, I. and Jolly, D. Arctic Research Consortium of the United States. Fairbanks, AK.

Furgal, C. and Seguin, J. 2006. Climate change, health, and vulnerability in Canadian northern Aboriginal communities. Environmental Health Perspectives 114:1964-1970.

Gagnon, C.A. and Berteaux, D. 2006. Integrating traditional and scientific knowledge: management of Canada's national parks. *In* Climate change: linking traditional and scientific knowledge. Edited by Riewe, R. and Oakes, J. Aboriginal Issues Press, University of Manitoba. Winnipeg, Manitoba. pp. 209-221.

Gagnon, C.A. and Berteaux, D. 2009. Integrating traditional ecological knowledge and ecological science: a question of scale. Ecology and Society 14:1-19.

Gaillard, J.M., Festa-Bianchet, M. and Yoccoz, N.G. 1998. Population dynamics of large herbivores: variable recruitment with constant adult survival. Trends in Ecology & Evolution 13:58-63.

Gallant, D., Slough, B.G., Reid, D.G. and Berteaux, D. 2012. Arctic fox versus red fox in the warming Arctic: four decades of den surveys in north Yukon. Polar Biology 35:1421-1431.

Gamache, I. and Payette, S. 2005. Latitudinal response of Subarctic tree lines to recent climate change in eastern Canada. Journal of Biogeography 32:849-862.

Gamberg, M. 2008. Arctic caribou and moose contaminant monitoring program. Technical Background Reports: Ecosystem Status and Trends Report for Canada. In preparation.

Gamberg, M. 2009. Arctic caribou and moose contaminant monitoring program. *In* Synopsis of research conducted under the 2008-2009 Northern Contaminants Program. Edited by Smith, S., Stow, J. and Edwards, J. Indian and Northern Affairs Canada. Ottawa, ON. pp. 179-184.

Gamberg, M., Braune, B., Davey, E., Elkin, B., Hoekstra, P.F., Kennedy, D., Macdonald, C., Muir, D., Nirwal, A., Wayland, M. and Zeeb, B. 2005a. Spatial and temporal trends of contaminants in terrestrial biota from the Canadian Arctic. Science of the Total Environment 351:148-164.

Gamberg, M., Braune, B., Davey, E., Elkin, B., Hoekstra, P.F., Kennedy, F., MacDonald, C., Muir, D., Nirwal, A., Wayland, M. and Zeeb, B. 2005b. Spatial and temporal trends of contaminants in terrestrial biota from the Canadian Arctic. Science of the Total Environment 351:148-164.

Gardner, A.S., Moholdt, G., Cogley, J.G., Wouters, B., Arendt, A.A., Wahr, J., Berthier, E., Hock, R., Pfeffer, W.T., Kaser, G., Ligtenberg, S.R.M., Bolch, T., Sharp, M.J., hagen, J.O., van den Broeke, M.R. and Paul, F. 2013. A Reconciled Estimate of Glacier Contributions to Sea Level Rise: 2003 to 2009. Science 340:852-857.

Gartner Lee Limited. 2002. The Bathurst Caribou Herd: a compilation of research and monitoring. Prepared for the North Slave Métis Alliance. Yellowknife, YT. 17 p.

Gaston, A.J. 2002. Results of monitoring thick-billed murre populations in the eastern Canadian Arctic, 1976-2000. Occasional Paper No. 106. Canadian Wildlife Service, Environment Canada. Ottawa, ON. 50 p.

Gaston, A.J., Bertram, D.F., Boyne, A.W., Chardine, J.W., Davoren, G., Diamond, A.W., Hedd, A., Montevecchi, W.A., Hipfner, J.M., Lemon, M.J.F., Mallory, M.L., Rail, J.-F. and Robertson, G.J. 2009a. Changes in Canadian seabird populations and ecology since 1970 in relation to changes in oceanography and food webs. Environmental Reviews 17:267-286.

Gaston, A.J., Gilchrist, H.G. and Hipfner, J.M. 2005. Climate change, ice conditions and reproduction in an Arctic nesting marine bird: Brunnich's guillemot *(Uria lomvia* L.*)*. Journal of Animal Ecology 74:832-841.

Gaston, A.J., Glichrist, H.G., Mallory, M.L. and Smith, P.A. 2009b. Changes in seasonal events, peak food availability and consequent breeding adjustment in a marine bird: a case of progressive mis-matching. The Condor 111:111-119.

Gaston, A.J., Woo, K. and Hipfner, J.M. 2003. Trends in forage fish populations in northern Hudson Bay since 1981, as determined from the diet of nestling thick-billed murres *Uria lomvia*. Arctic 56:227-233.

Gates, C.C., Mitchell, J., Wierzchowski, J. and Giles, L. 2001. A landscape evaluation of bison movements and distribution in northern Canada. Axys Environmental Consulting Ltd. Calgary, AB. 113 p.

Gau, R.J., Case, R., Penner, D.F. and McLoughlin, P.D. 2002. Feeding patterns barren-ground grizzly bears in the central Canadian Arctic. Arctic 55:339-344.

Gauthier, G., Berteaux, D., Bety, J., Tarroux, A., Therrien, J.F., McKinnon, L., Legagneux, P. and Cadieux, M.C. 2011. The tundra food web of Bylot Island in a changing climate and the role of exchanges between ecosystems. Ecoscience 18:223-235.

Gauthier, G., Bêty, J., Cadieux, M.-C., Legagneux, P., Doiron, M., Chevallier, C., Lai, S., Tarroux, A. and Berteaux, D. 2013. Long-term monitoring at multiple trophic levels suggests heterogeneity in responses to climate change in the Canadian Arctic tundra. Phil. Trans. R. Soc. B. 368:20120482-.

Gauthier, G., Bêty, J., Giroux, J.-F. and Rochefort, L. 2004. Trophic interactions in a high Arctic snow goose colony. Integrative and Comparative Biology 44:119-129.

Gauthier, G., Giroux, J.F., Reed, A., Bechet, A. and Belanger, L. 2005. Interactions between land use, habitat use, and population increase in greater snow geese: what are the consequences for natural wetlands? Global Change Biology 11:856-868.

Geffen, E., Waidyaratne, S., Dalen, L., Angerbjorn, A., Vila, C., Hersteinsson, P., Fuglei, E., White, P.A., Goltsman, M., Kapel, C.M.O. and Wayne, R.K. 2007. Sea ice occurrence predicts genetic isolation in the Arctic fox. Molecular Ecology 16:4241-4255.

GeoInsight Corporation. 1999. An approach to assessing risk to terrestrial biodiversity in Canada. Prepared for Indicators and Assessment Office, Environment Canada. State of the Environment Reporting Occasional Paper Series No. 11. Ottawa, ON. 50 p.

Geological Survey of Canada. 1994. Surficial materials of Canada, map 1880A [online]. Natural Resources Canada. http://geoscan.ess.nrcan.gc.ca/starweb/geoscan/servlet.starweb (accessed 23 October, 2009).

Gerland, S., Aars, J., Bracegirdle, T., Carmack, E., Hop, H., Hovelsrud, G.K., Kovacs, K.M., Lydersen, C., Perovich, D.K., Richter-Menge, J., Rybråten, S., Strøm, H. and Turner, J. 2007. Ice in the sea. *In* Global outlook for ice and snow. United Nations Environment Programme. Nairobi, Kenya. Chapter 5. pp. 63-96.

Gilchrist, H.G. and Robertson, G.J. 2000. Observations of marine birds and mammals wintering at polynyas and ice edges in the Belcher Islands, Nunavut, Canada. Arctic 53:61-68.

Gilg, O., Hanski, I. and Sittler, B. 2003. Cyclic dynamics in a simple vertebrate predator-prey community. Science 302:866-868.

Gilg, O., Kovacs, K.M., Aars, J., Fort, J., Gauthier, G., Gremillet, D., Ims, R.A., Meltofte, H., Moreau, J., Post, E., Schmidt, N.M., Yannic, G. and Bollache, L. 2012. Climate change and the ecology and evolution of Arctic vertebrates. Year in Ecology and Conservation Biology 1249:166-190.

Gilg, O., Sittler, B. and Hanski, I. 2009. Climate change and cyclic predator-prey population dynamics in the high Arctic. Global Change Biology 15:2634-2652.

Giroux, M.A., Berteaux, D., Lecomte, N., Gauthier, G., Szor, G. and Bety, J. 2012a. Benefiting from a migratory prey: spatio-temporal patterns in allochthonous subsidization of an arctic predator. Journal of Animal Ecology 81:533-542.

Giroux, M.-A., Campbell, M., Dumond, M. and Jenkins, D. 2012b. Availability of caribou and muskoxen for local human consumption across Nunavut. Version 1.8. Report presented to the Nunavut Anti-Poverty Secretariat.

Gjerdrum, C., Vallée, A.M.J., St.Clair, C.C., Bertram, D.F., Ryder, J.L. and Blackburn, G.S. 2003. Tufted puffin reproduction reveals ocean climate variability. Proceedings of the National Academy of Sciences of the United States of America 100:9377-9382.

Gnieser, C.H. 2000. Ecological consequences of recreation on subarctic-alpine tundra: experimental assessment and predictive modeling as planning tools for sustainable visitor management in protected areas. Thesis (Ph.D.). University of Calgary, Department of Geography. Calgary, AB. 435 p.

Goetz, S.J., Mack, M.C., Gurney, K.R., Randerson, J.T. and Houghton, R.A. 2007. Ecosystem responses to recent climate change and fire disturbance at northern high latitudes: observations and model results contrasting Northern Eurasia and North America. Environmental Research Letters 2, 04531:1-9.

Gordon, A.B., Andre, M., Kaglik, B., Cockney, S., Allen, M., Tetlichi, R., Buckle, R., Firth, A., Andre, J., Gilbert, M., Iglangasak, B. and Rexford, F. 2008. Arctic Borderlands Ecological Knowledge Co-op community reports 2006-2007. Arctic Borderlands Ecological Knowledge Society. Whitehorse, YT. 56 p.

Gould, W.A., Raynolds, M. and Walker, D.A. 2003. Vegetation, plant biomass, and net primary productivity patterns in the Canadian Arctic. Journal of Geophysical Research 108:1-8.

Government of Canada. 2011. Guide for federal implementers of comprehensive land claims and self-government agreements. Department of Indian Affairs and Northern Development. 58 p.

Government of Canada. 2013. Species at risk public registry [online]. http://www.sararegistry.gc.ca/default_e.cfm (accessed 6 June, 2013).

Government of Nunavut. 2012. Draft management plan for High Arctic muskoxen of the Qikiqtaaluk Region. Submitted to the Nunavut Wildlife Management Board. Iqaluit, NU. 20 p.

Government of Nunavut, Department of Environment. 2005. Inuit Qaujimajatuqangit of climate change in Nunavut 2005 - a sample of Inuit experiences of recent climate and environmental changes in Clyde River, Pond Inlet, Resolute Bay and Grise Fiord, Nunavut. Environmental Protection Division, Government of Nunavut Department of Environment. 37 p.

Government of the Northwest Territories and NWT Biodiversity Team. 2010. Northwest Territories state of the environment - 2010 biodiversity special edition. Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 36 p.

Grabowski, M.M., Doyle, F.I., Reid, D.G., Mossop, D. and Talarico, D. 2013. Do Arctic-nesting birds respond to earlier snowmelt? A multi-species study in north Yukon, Canada. Polar Biology 36:1097-1105.

Gratto-Trevor, C., Morrison, R.I.G., Collins, B., Rausch, J., Drever, M. and Johnston, V. 2011. Trends in Canadian shorebirds. Canadian biodiversity: ecosystem status and trends 2010: technical background report series. Technical Thematic Report No. 13. Canadian Councils of Resources Ministers. Ottawa, ON.

Gratto-Trevor, C.L. 1994. Monitoring shorebird populations in the Arctic. Bird Trends 3:10-12.

Gratto-Trevor, C.L., Johnston, V.H. and Pepper, S.T. 1998. Changes in shorebird and eider abundance in the Rasmussen Lowlands, NWT. Wilson Bulletin 110:316-325.

Gray, D.R. 1987. The muskoxen of Polar Bear Pass. National Museum of Natural Sciences/Fitzhenry and Whiteside. Markham, ON. 191 p.

Greenland Institute of Natural Resources. 2012. Muskoxen [online]. http://www.natur.gl/en/birds-and-mammals/terrestrial-mammals/muskoxen/ (accessed 7 July, 2013).

Greer, D. 2013. Hanging on for survival. Kivalliq news, June 12, 2013 [online]. Northern News Services. http://nnsl.com/archive/pdf-archives/ki061213.pdf (accessed 7 July, 2013).

Grenfell, T.C. and Putkonen, J. 2008. A method for the detection of the severe rain-on-snow event on Banks Island, October 2003, using passive microwave remote sensing. Water Resources Research 44:1-9.

Griffith, B., Douglas, D.C., Walsh, N.E., Young, D.D., McCabe, T.R., Russell, D.E., White, R.G., Cameron, R.D. and Whitten, K.R. 2002. The Porcupine Caribou Herd. *In* Arctic refuge coastal plain terrestrial wildlife research summaries. Edited by Douglas, D.C., Reynolds, P.E. and Rhode, E.B. U.S. Geological Survey, Biological Resources Division, Biological Science Report USGS/BRD/BSR-2002-0001. pp. 8-37.

Grosse, G., Harden, J., Turetsky, M., McGuire, A.D., Camill, P., Tarnocai, C., Frolking, S., Schuur, E.A.G., Jorgenson, T., Marchenko, S., Romanovsky, V., Wickland, K.P., French, N., Waldrop, M., Bourgeau-Chavez, L. and Striegl, R.G. 2011. Vulnerability of high-latitude soil organic carbon in North America to disturbance. Journal of Geophysical Research-Biogeosciences 116.

Groves, P. 1995. Muskox husbandry, a guide for the care, feeding and breeding of captive muskoxen. Biiological papers, special report No. 5. University of Alaska. Fairbanks.

Gruyer, N. 2007. Étude comparative de la démographie de deux espèces de lemmings (*Lemmus sibericus* et *Dicrostonyx groenlandicus*), à l'Île Bylot, Nunavut, Canada. Thesis (MSc). Université Laval. Quebec, Canada.

Gruyer, N., Gauthier, G. and Berteaux, D. 2008. Cyclic dynamics of sympatric lemming populations on Bylot Island, Nunavut, Canada. Canadian Journal of Zoology-Revue Canadienne de Zoologie 86:910-917.

Gunn, A. 2003a. Voles, lemmings and caribou - population cycles revisited? Rangifer, Special Issue 14:105-111.

Gunn, A. and Fournier, B. 2000a. Calf survival and seasonal migrations of a mainland muskox population. File Rep. No. 124. Northwest Territories, Department of Resources, Wildlife and Economic Development. 113 p.

Gunn, A., Poole, K. and Wierzchowski, J. 2012. Peary caribou distribution within the Bathurst Island Complex relative to the Federal Government boundary proposed for Qausuittuq National Park, northern Bathurst Island, Nunavut. Unpublished report for Parks Canada. Ottawa. 137 p.

Gunn, A. 1984. A review of research on the effects of human activities on barren-ground caribou of the Beverly and Kaminuriak herds, Northwest Territories. File Report No. 43. Government of the Northwest Territories. Yellowknife, NT. 66 p.

Gunn, A. 1990. The decline and recovery of caribou and muskoxen on Victoria Island. *In* The Canadian arctic islands: Canada's missing dimension. Edited by Harington, R. National Museum of Natural Science. Ottawa, ON. pp. 188-195.

Gunn, A. 2003b. Voles, lemmings and caribou - population cycles revisited? Rangifer Special Issue 14:105-111.

Gunn, A. 2005. The decline of caribou on northwest Victoria Island: a review. File Report No. 133. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 57 p.

Gunn, A. 2013. Satellite collaring and calf survival in the Bathurst herd of barren-ground caribou 2003-2005. Manuscript Report No. 228. Department of Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 62 p.

Gunn, A. and Adamczewski, J. 2003. Muskox. *In* Wild mammals of North America. Edited by Feldhamer, G., Chapman, B.A. and Chapman, J.A. The John Hopkins University Press. Baltimore. Chapter 50. 1232.

Gunn, A., Arlooktoo, G. and Kaomayok, D. 1988. The contribution of ecological knowledge of Inuit to wildlife management in the Northwest Territories. *In* Traditional knowledge and renewable resources management in northern regions. Edition 23. Edited by Carbyn, L. Boreal Institute for Northern Studies. pp. 22-30.

Gunn, A., Decker, R. and Barry, T.W. 1984. Possible causes and consequences of an expanding muskox population, Queen Maud Gulf area, Northwest Territories. *In* Proceedings of the first international muskox symposium. Edited by Klein, D.R., White, R.G. and Keller, S. University of Alaska. Fairbanks, AK. pp. 41-46.

Gunn, A. and Dragon, J. 1998. Status of caribou and muskox populations within the Prince of Wales Island-Somerset Island-Boothia Peninsula complex, NWT, July-August 1995. File Report No. 122. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 45 p.

Gunn, A. and Dragon, J. 2002. Peary caribou and muskox abundance and distribution on the Western Queen Elizabeth Islands, Northwest Territories and Nunavut June-July 1997. File Report No. 130. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 93 p.

Gunn, A., Dragon, J. and Boulanger, J. 2001. Seasonal movements of satellite-collared caribou from the Bathurst Herd. Final report to the West Kitikmeot Slave Study Society. Yellowknife, NT. 72 p.

Gunn, A., Dragon, J., Papik, S., Panayi, D., Svoboda, M., Sutherland, M. and D'Entremont, M. 1998. Final Report to the West Kitikmeot/Slave Study. Summer behaviour of Bathurst caribou at mine sites and response of caribou to fencing and plastic deflector. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 11 p. + appendices.

Gunn, A., Eamer, J., Reynolds, P., Sipko, T.P. and Gruzdev, A.R. 2013a. Muskoxen. In Arctic Report Card: Update for 2013 [online]. NOAA. http://www.arctic.noaa.gov/reportcard/muskox.html (accessed 13 December, 20130a).

Gunn, A. and Forchhammer, M.C. 2008. *Ovibos moschatus*. In: IUCSN 2013 IUCN Red List of threatened species. Version 2013.1. [online]. www.iucnredlist.org (accessed 7 July, 2013).

Gunn, A. and Fournier, B. 2000b. Identification and substantiation of caribou calving grounds on the NWT mainland and islands. File Report No. 123. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 177 p.

Gunn, A., Fournier, B. and Morrison, R. 2000a. Seasonal movements and distribution of satellite-collared caribou cows on the Boothia and Simpson Peninsula areas, Northwest Territories 1991-93. Manuscript Report No. 126. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 77 p.

Gunn, A., Jingfors, K. and Evalik, P. 1986. The Kitikmeot harvest study: a successful example for the collection of wildlife harvest statistics in the Northwest Territories. *In* Symposium: Native People and renewable resource management, 29 April - 1 May 1986. Alberta Society of Professional Biologists. Edmonton, AB. pp. 49-65.

Gunn, A., Johnson, C.J., Nishi, J.S., Daniel, C.J., Carlson, M., Russell, D.E. and Adamczewski, J.Z. 2011a. Addressing cumulative effects in the Canadian central Arctic - understanding the impacts of human activities on barren-ground caribou. *In* Cumulative effects in wildlife management: impact mitigation. Edited by Harris, L.K. CRC Press. Chapter 8. pp. 113-133.

Gunn, A., Leighton, T. and Wobeser, G. 1991a. Wildlife diseases and parasites in the Kitikmeot region, 1984-90. Final Report No. 104. Department of Renewable Resources, Government of the Northwest Territories. Coppermine, NT. 52 p.

Gunn, A., Miller, F. and Nishi, J. 2000b. Status of endangered and threatened caribou on Canada's Arctic islands. Rangifer Special Issue 12:39-50.

Gunn, A., Miller, F.L., Barry, S.J. and Buchan, A. 2006. A near-total decline in caribou on Prince of Wales, Somerset, and Russell islands, Canadian Arctic. Arctic 59:1-13.

Gunn, A. and Patterson, B.R. 2012. Distribution and abundance of muskoxen on southeastern Victoria Island, nunavut, 1988-1999. Manuscript Report No. 222. Department of Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 41 p.

Gunn, A., Poole, K., Wierzchowski, J., Nishi, J., Adamczewski, J., Russell, D. and D'Hont, A. 2013b. Have geographical influences and changing abundance led to sub-population structure in the Ahiak caribou herd, Nunavut, Canada? Rangifer Special Issue 33:35-58.

Gunn, A., Poole, K.G. and Wierzchowski, J. 2011b. Migratory tundra caribou seasonal and annual distribution relative to Thaidene Nene, a national park reserve proposal in the East Arm of Great Slave Lake and Artillery Lake area, Northwest Territories. Unpublished report for Parks Canada. Fort Smith, NT. 46 p.

Gunn, A. and Russell, D. (eds.). 2008. Monitoring *Rangifer* herds (population dynamics). CircumArctic Rangifer Monitoring and Assessment Network. 52 p.

Gunn, A., Russell, D. and Eamer, J. 2011c. Northern caribou population trends in Canada. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 10. Canadian Councils of Resource Ministers. Ottawa, ON. iv + 71 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Gunn, A., Russell, D. and Greig, L. 2013c. Insights into integrating cumulative effects and collaborative co-management for migratory tundra caribou herds, NWT, Canada. Ecology and Society . In Press.

Gunn, A., Shank, C. and McLean, B. 1991b. The history, status and management of muskoxen on Banks Island. Arctic 44:188-195.

Gunn, A., Williams, J. and Shank, C. 2009. Distribution and abundance of muskoxen in the Beaverhill Lake area (2000) and Thelon Wildlife Sanctuary (1994), Northwest Territories. Manuscript Report No. 190. Department of Environment and Natural Resources, Government of the Northwest Territories. 76 p.

Hansen, B.B., Grotan, V., Aanes, R., Saether, B.E., Stien, A., Fuglei, E., Ims, R.A., Yoccoz, N.G. and Pedersen, A.O. 2013. Climate events synchronize the dynamics of a resident vertebrate community in the High Arctic. Science 339:313-315.

Hansen, J., Sato, M., Ruedy, R., Lo, K., Lea, D.W. and Medina-Elizade, M. 2006. Global temperature change. Proceedings of the National Academy of Sciences of the United States of America 103:14288-14293.

Harding, L.E. 1976. Den site characteristics of arctic coastal grizzly bears (*Ursus arctos* L.) on Richards Islands, Northwest Territories, Canada. Canadian Journal of Zoology 54:1357-1363.

Harding, L.E. 2004. The future of Peary caribou (*Rangifer tarandus pearyi*) in a changing climate. Edited by Hooper, T.D. Victoria, BC.9.

Harding, L.E. and Nagy, J.A. 1980. Responses of grizzly bears (*Ursus arctos* L.) to hydrocarbon exploration on Richards Islands, Northwest Territories, Canada. *In* Selected papers of the fourth international conference on bear research and management. Kalispell, MT. pp. 277-280.

Harington, C.R. 1970. A postglacial muskox, *Ovibos moschatus*, from Grandview, Manitoba, and comments on the zoogeography of *Ovibos*. National Museums Canada Publication, Palaeontology 2:1-13.

Harper, P. (ed.). 2011. Muskox management report of survey and inventory activities, 1 July 2008 - 30 June 2010. Alaska Department of Fish and Game, Project 16.0. Juneau, AK. 85 p.

Harrington, B.A. and Morrison, R.I.G. 1980. Notes on the wintering areas of red knot *Calidris canutus rufa* in Argentina, South America. Wader Study Group Bulletin 28:40-42.

Harris, C. and Isaksen, K. 2008. Recent warming of European permafrost: evidence from borehole monitoring. *In* Proceedings of the ninth international conference on permafrost. Fairbanks, AK, 29 June-3 July, 2008. Edited by Kane, D.L. and Hinkel, K.M. Institute of Northern Engineering, University of Alaska Fairbanks. Vol. 1, pp. 655-661.

Heard, D.C. 1992. The effect of wolf predation and snow cover on musk-ox group size. American Naturalist 139:190-204.

Heginbottom, J.A., Dubreuil, M.A. and Harker, P.A.C. 1995. Permafrost, 1995. *In* The National Atlas of Canada. Edition 5. National Atlas Information Service, Geomatics Canada and Geological Survey of Canada. Ottawa, ON. Map.

Heininen, L. and Nicol, H.N. 2007. The importance of northern dimension foreign policies in the geopolitics of the circumpolar North. Geopolitics 12:133-165.

Helander-Renvall, E. 2005. Composite report on status and trends regarding the knowledge, innovations and practices of Indigenous and local communities, region: Arctic. United Nations Environmental Programme, Convention on Biological Diversity. Granada, Spain. 105 p.

Helgson, A., Pálsson, G., Angulailik, E., Gunnarsdóttir, E.D., Yngvadóttir, B. and Stefánsson, K. 2014. mtDNA variation in Inuit populations of Greenland and Canada: Migration history and population structure. Am. J. Phys. Anthropol 130:123-134.

Henry, G.H.R., Freedman, B. and Svoboda, J. 1986. Effects of fertilization on three tundra plant communities of a polar desert oasis. Canadian Journal of Botany 64:2502-2507.

Henry, G.H.R., Harper, K.A., Chen, W.J., Deslippe, J.R., Grant, R.F., Lafleur, P.M., Levesque, E., Siciliano, S.D. and Simard, S.W. 2012. Effects of observed and experimental climate change on terrestrial ecosystems in northern Canada: results from the Canadian IPY program. Climatic Change 115:207-234.

Henry, G.H.R. and Molau, U. 1997. Tundra plants and climate change: The International Tundra Experiment (ITEX). Global Change Biology 3 (Suppl. 1):1-9.

Higuera, P.E., Brubaker, L.B., Anderson, P.M., Brown, T.A., Kennedy, A.T. and Hu, F.S. 2008. Frequent fires in ancient shrub tundra: implications of paleorecords for Arctic environmental change. PLoS One 3(3):1744.

Hill, G. 2006. Responses of High Arctic sedge meadows to climate warming at Alexandra Fiord, Ellesmere Island, since 1980. Thesis (M.Sc.). University of British Columbia. Vancouver, BC. viii + 65 p.

Hill, G.B. and Henry, G.H.R. 2011. Responses of high Arctic wet sedge tundra to climate warming since 1980. Global Change Biology 17:276-287.

Hinzman, L.D., Bettez, N.D., Bolton, W.R., Chapin, F.S., Dyurgerov, M.B., Fastie, C.L., Griffith, B., Hollister, R.D., Hope, A., Huntington, H.P., Jensen, A.M., Jia, G.J., Jorgenson, T., Kane, D.L., Klein, D.R., Kofinas, G., Lynch, A.H., Lloyd, A.H., McGuire, A.D., Nelson, F.E., Oechel, W.C., Osterkamp, T.E., Racine, C.H., Romanovsky, V.E., Stone, R.S., Stow, D.A., Sturm, M., Tweedie, C.E., Vourlitis, G.L., Walker, M.D., Walker, D.A., Webber, P.J., Welker, J.M., Winker, K. and Yoshikawa, K. 2005. Evidence and implications of recent climate change in Northern Alaska and other arctic regions. Climatic Change 72:251-298.

Hipfner, J.M. and Greenwood, J.L. 2008. Breeding biology of the common murre at Triangle Island, British Columbia, 2002-2007. Northwestern Naturalist 89:76-84.

Hobbie, S.E. 1996. Temperature and plant species control over litter decomposition in Alaskan tundra. Ecological Monographs 66:503-522.

Hoberg, E.P., Polley, L., Gunn, A. and Nishi, J.S. 1995. *Umingmakstrongylus pallikuukensis* gen nov et sp nov (Nematoda: Protostrongylidae) from muskoxen, Ovibos moschatus, in the central Canadian Arctic, with comments on biology and biogeography. Canadian Journal of Zoology-Revue Canadienne de Zoologie 73:2266-2282.

Hoberg, E.P., Polley, L., Jenkins, E.J., Kutz, S.J., Veitch, A.M. and Elkin, B.T. 2008. Integrated approaches and empirical models for investigation of parasitic diseases in northern wildlife. Emerging Infectious Diseases 14:10-17.

Hofgaard, A. and Harper, K.A. 2011. Tree recruitment, growth, and distribution at the circumpolar forest-tundra transition: introduction. Canadian Journal of Forest Research-Revue Canadienne de Recherche Forestiere 41:435-436.

Hollister, R.D., Webber, P.J. and Tweedie, C.E. 2005. The response of Alaskan Arctic tundra to experimental warming: differences between short- and long-term responses. Global Change Biology 11:525-536.

Hone, E. 1934. The present status of the muskox in Arctic North America and Greenland. Spec. Publ. Amer. Comm. Internat. Wildl. Protection 5:1-87.

Hörnfeldt, B., Hipkiss, T. and Eklund, U. 2005. Fading out of vole and predator cycles? Proceedings: Biological Sciences 272:2045-2049.

Howe, M.A., Geissler, P.H. and Harrington, B.A. 1989. Population trends of North American shorebirds based on the International Shorebird Survey. Biological Conservation 49:185-199.

Høye, T.T., Post, E., Schmidt, N.M., Trøjelsgaard, K. and Forchammer, M.C. 2013. Shorter flowering seasons and declining abundance of flower visitors in a warmer Arctic. Nature Climate Change 3:759-763.

Hu, F.S., Higuera, P.E., Walsh, J.E., Chapman, W.L., Duffy, P.A., Brubaker, L.B. and Chipman, M.L. 2010. Tundra burning in Alaska: Linkages to climatic change and sea ice retreat. Journal of Geophysical Research-Biogeosciences 115.

Hudson, J.M.G. and Henry, G.H.R. 2009. Increased plant biomass in a high Arctic heath community from 1981 to 2008. Ecology 90:2657-2663.

Hudson, J.M.G. and Henry, G.H.R. 2010. high Arctic plant community resists 15 years of experimental warming. Journal of Ecology 98:1035-1041.

Hudson, P.J. and Bjornstad, O.N. 2003. Vole stranglers and lemming cycles. Science 302:797-798.

Hueffer, K., Parkinson, A.J., Gerlach, R. and Berner, J. 2013. Zoonotic infections in Alaska: disease prevalence, potential impact of climate change and recommended actions for earlier disease detection, research, prevention and control. International Journal of Circumpolar Health 72 19562.

Hughes, J., Albon, S.D., Irvine, R.J. and Woodin, S. 2009. Is there a cost of parasites to caribou? Parasitology 136:253-265.

Hummel, M. and Ray, J.C. 2008. Caribou and the North: a shared future. Dundurn Press. Toronto, ON. 320 p.

Humphreys, E.R. and Lafleur, P.M. 2011. Does earlier snowmelt lead to greater CO2 sequestration in two low Arctic tundra ecosystems? Geophysical Research Letters 38 L09703.

Huntington, H.P., Fox, S., Berkes, F. and Krupnik, I. 2005. The changing Arctic: indigenous perspectives. *In* Arctic Climate Impact Assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. Chapter 3. pp. 61-98.

Hurrell, J.W. 1996. Influence of variations in extratropical wintertime teleconnections on Northern Hemisphere temperature. Geophysical Research Letters 23:665-668.

Ims, R., Ehrich, D., Forbes, B., Huntley, B., Walker, S., Wookey, P., Bhatt, U., Berteaux, D., Brathen, K., Gautier, G., Gilbert, S., Edwards, M.E., Epstein, H.E., Forchhammer, M.C., Fuglei, E., Menyushina, I.E., Ovsyanikov, N., Post, E., Reid, D.G., Raynolds, M.K., Schmidt, N.M., Stien, A., Sumina, O.I. and Van der Wal, R. 2013. Terrestrial ecosystems. *In* Arctic biodiversity assessment: status and trends in Arctic biodiversity. Edited by Meltofte, H. Conservation of Arctic Flora and Fauna (CAFF). Akureyri, Iceland. Chapter 12. In Prep.

Ims, R.A., Yoccoz, N.G. and Killengreen, S.T. 2011. Determinants of lemming outbreaks. Proceedings of the National Academy of Sciences of the United States of America 108:1970-1974.

INAC. 2005. Northern contaminated sites program [online]. Indian and Northern Affairs Canada. http://www.ainc-inac.gc.ca/nth/ct/ncsp/index-eng.asp (accessed 30 November, 2009).

INAC. 2008. Northern contaminants program [online]. Indian and Northern Affairs Canada. http://www.science.gc.ca/default.asp?lang=En&n=7A463DBA-1 (accessed 13 May, 2008).

Indian Affairs and Northern Development Canada. 1984. The Western Arctic Claim - The Inuvialuit Final Agreement. Gouvernnement du Canada. Ottawa, ON. 115 p.

Indian Affairs and Northern Development Canada. 1993. Agreement between the Inuit of the Nunavut Settlement Area and Her Majesty the Queen in Right of Canada. Government of Canada and the Tunngavik Federation of Nunavut. Ottawa, ON. 282 p.

Intergovernmental Panel on Climate Change. 2007. The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Edited by Solomon, S., Qin, S., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L. Cambridge University Press. Cambridge, UK and New York, NY. 996 p.

InterGroup Consultants Ltd. 2013. Economic valuation and socio-cultural perspectives of the estimated harvest of the Beverly and Qamanirjuaq caribou herds. Beverly and Qamanijuaq Caribou Management Board (update from 2008 report). Stonewall, MB. 32 p. + 3 appendices.

Inuuvik Community Corporation, Tuktuuyaqtuuq Community Corporation and Aklavik Community Corporation. 2006. Inuvialuit settlement region traditional knowledge report. Mackenzie Project Environmental Group. Calgary, AB. 199 p.

Inuvialuit Game Council. 2002. Draft co-management plan for caribou, muskox, arctic wolves, small herbivores, king eiders and common eiders on Banks Island. Wildlife Management Advisory Council (NWT). Inuvik, NT. 63 p.

IPCC. 2007. Climate change 2007: synthesis report, summary for policymakers. Intergovernmental Panel on Climate Change. Valencia, Spain. 22 p.

Isacch, J.P. and Martinez, M.M. 2003. Temporal variation in abundance and the population status of non-breeding Nearctic and Patagonian shorebirds in the flooding pampa grasslands of Argentina. Journal of Field Ornithology 74:233-242.

Isaksen, K., Benestad, R.E., Harris, C. and Sollid, J.L. 2007a. Recent extreme near-surface permafrost temperatures on Svalbard in relation to future climate scenarios. Geophysical Research Letters 34, L17502:1-5.

Isaksen, K., Sollid, J.L., Holmlund, P. and Harris, C. 2007b. Recent warming of mountain permafrost in Svalbard and Scandinavia. Journal of Geophysical Research-Earth Surface 112 (F02S04):1-11.

James, A.R.C. and Stuart-Smith, A.K. 2000. Distribution of caribou and wolves in relation to linear corridors. Journal of Wildlife Management 64:154-159.

Jamieson, R. and Krkosek, W. 2013. Breaking the ice. Water Canada (January-February 2013 online edition) [online]. http://watercanada.net/2013/breaking-the-ice/ (accessed 10 December, 20130).

Jeffries, D., Richter-Menge, J.A. and Overland, J.E. (eds.). 2012. Arctic report card 2012, http://www.arctic.noaa.gov/reportcard.

Jehl, J.R. and Lin, W.L. 2001. Population status of shorebirds nesting at Churchill, Manitoba. Canadian Field Naturalist 115:487-494.

Jenkins, D.A., Campbell, M., Hope, G., Goorts, J. and McLoughlin, P. 2011. Recent trends in abundance of Peary caribou (*Rangifer tarandus pearyi*) and muskoxen (*Ovibos moschatus*) in the Canadian Arctic Archipelago, Nunavut. Wildlife Report No. 1. Department of Environment, Government of Nunavut. Pond Inlet, NU. 184 p.

Johnson, C.J., Boyce, M.S., Case, R.L., Cluff, H.D., Gau, R.J., Gunn, A. and Mulders, R. 2005. Cumulative effects of human developments on arctic wildlife. Wildlife Monographs 160:1-36.

Johnston, V. and Rausch, J. 2008. Preliminary results from Kivalliq Region shorebird surveys. Canadian Wildlife Service, Environment Canada. Yellowknife, NT. Unpublished data.

Johnston, V.H., Gratto-Trevor, C.L. and Pepper, S.T. 2000. Assessment of bird populations in the Rasmussen Lowlands, Nunavut. Occassional Paper No. 101. Canadian Wildlife Service, Environment Canada. Ottawa, ON. 56 p.

Johnston, V.H. and Pepper, S.T. 2009. The birds of Prince Charles Island and Air Force Island, Foxe Basin, Nunavut. Occasional Paper No. 117. Canadian Wildlife Service, Environment Canada. Ottawa, ON. 54 p.

Joly, K., Dale, B.W., Collins, W.B. and Adams, L.G. 2003. Winter habitat use by female caribou in relation to wildland fires in interior Alaska. Canadian Journal of Zoology 81:1192-1201.

Joly, K., Klein, D.R., Verbyla, D.L., Rupp, T.S. and Chapin, F.S. 2011. Linkages between large-scale climate patterns and the dynamics of Arctic caribou populations. Ecography 34:345-352.

Jones, C.S. 1999. Arctic ship tourism: an industry in adolescence. The Northern Raven 13:28-31.

Jones, G.A. and Henry, G.H.R. 2003. Primary plant succession on recently deglaciated terrain in the Canadian high Arctic. Journal of Biogeography 30:277-296.

Jones, M.H., Bay, C. and Nordenhäll, U. 1997. Effects of experimental warming on arctic willows (*Salix spp*.): a comparison of responses from the Canadian High Arctic, Alaskan Arctic, and Swedish Subarctic. Global Change Biology 3:55-60.

Jones, M.H., Macdonald, S.E. and Henry, G.H.R. 1999. Sex- and habitat-specific responses of a high arctic willow, *Salix arctica*, to experimental climate change. Oikos 87:129-138.

Jónsdóttir, I.S., Magnússon, B., Gudmundsson, J., Elmarsdóttir, A. and Hjartarson, H. 2005. Variable sensitivity of plant communities in Iceland to experimental warming. Global Change Biology 11:553-563.

Kallreuth, W., Andreasen, C., Petersen, H.I. and Stemmerik, L. 2012. The petrology and provenance of coal artifacts from Thule settlements in north-eastern Greenland . Bulletin of the Gelogical Society of Denmark 60:1-13.

Kassi, N. 1993. Native perspective on climate change. *In* Impacts of climate change on resource management in the North. Edited by Wall, G. University of Waterloo, Department of Geography Publication Series, Occasional Paper No. 16. Waterloo, ON. pp. 43-49.

Kattsov, V.M., Walsh, J.E., Chapman, W.L., Govorkova, V.A., Pavlova, T.V. and Zhang, X.D. 2007. Simulation and projection of arctic freshwater budget components by the IPCC AR4 global climate models. Journal of Hydrometeorology 8:571-589.

Keatley, B.E. 2007. Limnological and paleolimnological investigations of environmental change in three distinct ecosystem types, Canadian high Arctic. Thesis (Ph.D). Queen's University. 336 p. Kingston, On.

Kelsall, J.P. 1968. The migratory barren-ground caribou of Canada. Canadian Wildlife Service Monograph Series No. 3. Department of Indian Affairs and Northern Development. Ottawa, ON. 340 p.

Kerbes, R.H., Meeres, K.M., Alisaukas, R.T., Caswell, F.D., Abraham, K.F. and Ross, R.K. 2006. Surveys of nesting mid-continent lesser snow geese and ross's geese in Eastern and Central Canada (1997-98). Technical Report Series No. 447. Canadian Wildlife Service, Prairie and Northern Region. Saskatoon, SK. 65 p.

Kevan, P.G. 1972. Insect pollination of High Arctic flowers. Journal of Ecology 60:831-847.

Kharuk, V.I., Im, S.T. and Dvinskaya, M.L. 2010. Forest-tundra ecotone response to climate change in the Western Sayan Mountains, Siberia. Scandinavian Journal of Forest Research 25:224-233.

Kharuk, V.I., Ranson, K.J., Im, S.T. and Naurzbaev, M.M. 2006. Forest-tundra larch forests and climatic trends. Russian Journal of Ecology 37:291-298.

Kielland, K., Bryant, J.P. and Ruess, R.W. 2006. Mammalian herbivory, ecosystem engineering and ecological cascades in Alaskan boreal forests. *In* Alaska's changing boreal forest. Edited by Chapin III, F.S., Osowood, M.W., Van Cleve, K., Viereck, L.A. and Verbyla, D.L. Oxford University Press. New York, NY. pp. 211-226.

Kielland, K. and Chapin III, F.S. 1992. Nutrient absorption and accumulation in arctic plants. *In* Arctic ecosystems in a changing climate: an ecophysiological perspective. Edited by Chapin III, F.S., Jefferies, R.L., Reynolds, J.F., Shaver, G.R. and Svoboda, J. Academic Press Inc. San Diego, CA. pp. 321-335.

Kinnard, C., Zdanowicz, C.M., Fisher, D.A., Isaksson, E., de Vernal, A. and Thompson, L.G. 2011. Reconstructed changes in Arctic sea ice over the past 1,450 years. Nature 479:509-512.

Kirdyanov, A.V., Hagedorn, F., Knorre, A.A., Fedotova, E.V., Vaganov, E.A., Naurzbaev, M.M., Moiseev, P.A. and Rigling, A. 2012. 20th century tree-line advance and vegetation changes along an altitudinal transect in the Putorana Mountains, northern Siberia. Boreas 41:56-67.

Kitchen, A., Miyamoto, M.M. and Mulligan, C.J. 2008. A three-stage colonization model for the peopling of the Americas. PLoS One 3: e1596.

Kitikmeot Inuit Association. 2013. About KIA [online]. http://kitia.ca/en/about-kia (accessed 10 December, 20130).

Kivalliq Wildlife Board. 2010. Kivalliq musk ox management plan 2010-2015. Submitted to teh Nunavut Wildlife Management Board. Iqaluit, NU. 8 p.

Klein, D.R., Bruun, H.H., Lundgren, R. and Philipp, M. 2008. Climate change influences on species interrelationships and distributions in High Arctic Greenland. Advances in Ecological Research 40:81-100.

Kocan, A.A., Barron, S.J., Fox, J.C. and Franzmann, A.W. 1986. Antibodies to *Toxoplasma gondii* in Moose (*Alces alces L*) from Alaska. Journal of Wildlife Diseases 22:432-432.

Koerner, R.M. 2005. Mass balance of glaciers in the Queen Elizabeth Islands, Nunavut, Canada. Annals of Glaciology 42:417-423.

Kokelj, S.V. and Burn, C.R. 2005. Geochemistry of the active layer and near-surface permafrost, Mackenzie Delta region, Northwest Territories, Canada. Canadian Journal of Earth Sciences 42:37-48.

Kokelj, S.V., Zajdlik, B. and Thompson, M.S. 2009. The impacts of thawing permafrost on the chemistry of lakes across the subarctic boreal-tundra transition, Mackenzie Delta region, Canada. Permafrost and Periglacial Processes 20:185-199.

Koller-Jones, M. 14 August, 2006. Canadian Food Inspection Agency. Personal communication.

Korhola, A., Vasko, K., Toivonen, H.T.T. and Olander, H. 2002. Holocene temperature changes in northern Fennoscandia reconstructed from chironomids using Bayesian modelling. Quaternary Science Reviews 21:1841-1860.

Korsholm, H. and Olesen, C.R. 1993. Preliminary investigations on the parasite burden and distribution of endoparasite species of muskox (*Ovios moschatus*) and caribou (*Rangifer tarandus groenlandicus*) in West Greenland. Rangifer 13:185-189.

Krebs, C.J. 1996. Population cycles revisited. Journal of Mammalogy 77:8-24.

Krebs, C.J. 2011. Of lemmings and snowshoe hares: the ecology of northern Canada. Proceedings of the Royal Society B-Biological Sciences 278:481-489.

Krebs, C.J., Boonstra, R. and Kenney, A.J. 1995. Population dynamics of the collared lemming and the tundra vole at Pearce-Point, Northwest-Territories, Canada. Oecologia 103:481-489.

Krebs, C.J., Danell, K., Angerbjörn, A., Agrell, J., Berteaux, D., Brathen, K.A., Danell, O., Erlinge, S., Fedorov, V., Fredga, K., Hjältén, J., Högstedt, G., Jonsdotttir, I.S., Kenney, A.J., Kjellén, N., Nordin, T., Roininen, H., Svensson, M., Tannerfeldt, M. and Wiklund, C. 2003. Terrestrial trophic dynamics in the Canadian Arctic. Canadian Journal of Zoology 81:827-843.

Krebs, C.J., Kenney, A.J., Gilbert, S., Danell, K., Angerbjörn, A., Erlinge, S., Bromley, R.G., Shank, C. and Carriere, S. 2002. Synchrony in lemming and vole populations in the Canadian Arctic. Canadian Journal of Zoology 8:1323-1333.

Krebs, J., Lofroth, E.C. and Parfitt, E. 2007. Multiscale habitat use by wolverines in British Columbia, Canada. Journal of Wildlife Management 71:2180-2192.

Krezek, C.C. 13 March, 2009. Personal communication.

Krezek-Hanes, C.C., Ahern, F., Cantin, A. and Flannigan, M.D. 2011. Trends in large fires in Canada, 1959-2007. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 6. Canadian Councils of Resource Ministers. Ottawa, ON. v + 48 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Kruse, J., Poppel, B., Abryutina, L., Duhaime, G., Martin, S., Poppel, M., Kruse, M., Ward, E., Cochran, P. and Hanna, V. 2009. Survey of Living Conditions in the Arctic (SLiCA). Springer. Dordrecht. Barometers of qualtiy of life around the globe: how we are doing. Social Indicators Research Series. 134 p.

Kuhry, P., Grosse, G., Harden, J.W., Hugelius, G., Koven, C.D., Ping, C.L., Schirrmeister, L. and Tarnocai, C. 2013. Characterisation of the permafrost carbon pool. Permafrost and Periglacial Processes 24:146-155.

Kutz, S. 2007. An evaluation of the role of climate change in the emergence of pathogens and diseases in Arctic and Subarctic caribou populations. Climate Change Action Fund Project A760. Government of Canada. 132 p.

Kutz, S., Dumond, M., Seegers, K., Torretti, L., Ducrocq, J., deBruyn, N., Veitch, A., Curry, P., Hoberg, E.P., Elkin, B. and Spans, R. 2009. Climate change and parasitism in Arctic and Subarctic terrestrial mammals. Wildlife Society Arctic Chapter Conference. April, 2009.

Kutz, S., Schock, D., Brook, R. and Hoberg, E.P. 2008. Impending ills. The Wildlife Society. The Wildlife Professional, pp. 42- 46.

Kutz, S.J., Elkin, B., Gunn, A. and Dubey, J.P. 2000. Prevalence of *Toxoplasma gondii* antibodies in muskox (*Ovibos moschatus*) sera from northern Canada. Journal of Parasitology 86:879-882.

Kutz, S.J., Elkin, B.T., Panay, D. and Dubey, J.P. 2001. Prevalence of *Toxoplasma gondii* antibodies in barren-ground caribou (*Rangifer tarandus groenlandicus*) from the Canadian Arctic. Journal of Parasitology 87:439-442.

Kutz, S.J., Hoberg, E.P., Nagy, J., Polley, L. and Elkin, B. 2004. "Emerging" parasitic infections in arctic ungulates. Integrative and Comparative Biology 44:109-118.

Kutz, S.J., Hoberg, E.P., Polley, L. and Jenkins, E.J. 2005. Global warming is changing the dynamics of Arctic host-parasite systems. Proceedings of the Royal Society B: Biological Sciences 272:2571-2576.

Kuzyk, Z.A., P.V., H., Solomon, S.M. and Reimer, K.J. 2005. Biological responses to PCB exposure in shorthorn sculpin from Saglek Bay, Labrador. Science of the Total Environment 351-352:285-300.

Kuzyk, Z.Z.A., N.M.Burgess., J.P.Stow and G.A.Fox. 2003. Biological effects of marine PCB contamination on black guillemot nestlings at Saglek, Labrador: liver biomarkers. Journal Ecotoxicology 12:183-197.

Labelle, P., Dubey, J.P., Mikaelian, I., Blanchette, N., Lafond, R., St-Onge, S. and Martineau, D. 2001. Seroprevalence of antibodies to *Toxoplasma gondii* in lynx (*Lynx canadensis*) and bobcats (*Lynx rufus*) from Quebec, Canada. Journal of Parasitology 87:1194-1196.

Lafleur, P.M., Humphreys, E.R., St Louis, V.L., Myklebust, M.C., Papakyriakou, T., Poissant, L., Barker, J.D., Pilote, M. and Swystun, K.A. 2012. Variation in peak growing season net ecosystem production across the Canadian Arctic. Environmental Science & Technology 46:7971-7977.

Lamb, E.G., Han, S., Lanoil, B.D., Henry, G.H.R., Brummell, M.E., Banerjee, S. and Siciliano, S.D. 2011. A high Arctic soil ecosystem resists long-term environmental manipulations. Global Change Biology 17:3187-3194.

Lamoureux, S.F. and Lafreniere, M.J. 2009. Fluvial impact of extensive active layer detachments, Cape Bounty, Melville Island, Canada. Arctic, Antarctic, and Alpine Research 41:59-68.

Lantuit, H. and Pollard, W.H. 2008. Fifty years of coastal erosion and retrogressive thaw slump activity on Herschel Island, southern Beaufort Sea, Yukon Territory, Canada. Geomorphology 95:84-102.

Lantz, T.C. and Kokelj, S.V. 2008. Increasing rates of retrogressive thaw slump activity in the Mackenzie Delta region, NWT, Canada. Geophysical Research Letters 35, L06502:1-5.

Lantz, T.C., Kokelj, S.V., Gergel, S.V. and Henry, G.H.R. 2009. Relative impacts of disturbance and temperature: persistent changes in microenvironment and vegetation in retrogressive thaw slumps. Global Change Biology 15:1664-1675.

Large Herbivore Network. 2009. Musk Ox - *Ovibos moschatus* - distribution maps, historical and current [online]. http://www.lhnet.org/musk-ox/#Maps (accessed 7 July, 2013).

Larter, N.C. and Nagy, J.A. 1999. Muskox mortality survey, Banks Island, August 1996. Manuscript Report No. 117. Northwest Territories Department of Resources, Wildlife and Economic Development. 13 p.

Larter, N.C. and Nagy, J.A. 2004. Seasonal changes in the composition of the diets of Peary caribou and muskoxen on Banks Island. Polar Research 23:131-140.

Larter, N.C. 2013. Diet of Arctic wolves on Banks and Northwest Victoria islands. Manuscript Report No. 230. Department of Environment and Natural Resources, Government of the Northwest Territories. iii + 11 p.

Latifovic, R. and Pouliot, D. 2007. Analysis of climate change impacts on lake ice phenology in Canada using the historical satellite data record. Remote Sensing of Environment 106:492-507.

Latour, P.B., Leger, J., Hines, J.E., Mallory, M.L., Mulders, D.L., Gilchrist, H.G., Smith, P.A. and Dickson, D.L. 2008. Key migratory bird terrestrial habitat sites in the Northwest Territories and Nunavut, third edition. Occasional Paper No. 114. Edited by Gaston, A.J. Canadian Wildlife Service, Environment Canada. 18 p.

Latour, P.B., Machtans, C.S. and Beyersbergen, G.W. 2005. Shorebird and passerine abundance and habitat use at a high Arctic breeding site: Creswell Bay, Nunavut. Arctic 58:55-65.

Laurion, I., Vincent, W.F., MacIntyre, S., Retamal, L., Dupont, C., Francus, P. and Pienitz, R. 2010. Variability in greenhouse gas emissions from permafrost thaw ponds. Limnology and Oceanography 55:115-133.

Legagneux, P., Gauthier, G., Berteaux, D., Bety, J., Cadieux, M., Bilodeau, F., Bolduc, E., McKinnon, L., Tarroux, A., Therrien, J., Morissette, L. and Krebs, C. 2012. Disentangling trophic relationships in a high Arctic tundra ecosystem through food web modeling. Ecology 93:1707-1716.

Legat, A., Chocolate, G., Chocolate, M., Williah, P. and Zoe, S.A. 2002. Habitat of Dogrib traditional territory: placenames as indicators of biogeographical knowledge - final report. Dogrib Treaty 11 Council and West Kitikmeot Slave Study Society. Yellowknife, NT. 92 p.

Lehenaff, D. and Crete, M. 1989a. Introduction of muskoxen in Northern Quebec - the demographic explosion of a colonizing herbivore. Canadian Journal of Zoology-Revue Canadienne de Zoologie 67:1102-1105.

Lehenaff, D. and Crete, M. 1989b. Introduction of Muskoxen in Northern Quebec - the Demographic Explosion of A Colonizing Herbivore. Canadian Journal of Zoology-Revue Canadienne de Zoologie 67:1102-1105.

Leighton, F.A. Brucellosis in arctic caribou. Unpublished data.

Leighton, F.A. 2011. Wildlife pathogens and diseases in Canada. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 7. Canadian Councils of Resource Ministers. Ottawa, ON. iv + 53 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Lemke, P., Ren, J., Alley, R.B., Allison, I., Carrasco, J., Flato, G., Fujii, Y., Kaser, G., Mote, P., Thomas, R.H. and Zhang, T. 2007. Observations: changes in snow, ice and frozen ground. *In* Intergovernmental Panel on Climate Change fourth assessment report: climate change 2007: working group I: the physical science basis. Edited by Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L. Cambridge University Press. Cambridge, UK and New York, NY. Chapter 4.

Lent, P.C. 1988. *Ovibos moschatus*. Mammalian Species 302:1-9.

Letcher, R. 2008. Temporal and spatial trends of organic and metal contaminants in Canadian polar bears: 2007-2008 NCP project summary report. Technical Background Reports: Ecosystem Status and Trends Report for Canada. In preparation.

Lewkowicz, A.G. and Harris, C. 2005. Frequency and magnitude of active-layer detachment failures in discontinuous and continuous permafrost, Northern Canada. Permafrost and Periglacial Processes 16:115-130.

Liebezeit, J.R., Kendall, S.J., Brown, S., Johnson, C.B., Martin, P., McDonald, T.L., Payer, D.C., Rea, C.L., Streever, B., Wildman, A.M. and Zack, S. 2009. Influence of human development and predators on nest survival of tundra birds, Arctic Coastal Plain, Alaska. Ecological Applications 19:1628-1644.

Lloyd, A.H. and Bunn, A.G. 2007. Responses of the circumpolar boreal forest to 20th century climate variability. Environmental Research Letters 2 045013 .

Long, J.L. 2003. Introduced mammals of the world: their history, distribution and influence. CSIRO Publishing. Collingwood, Australia. 612 p.

LTER Network. 2008. Alaska fire sparks new research at Arctic LTER [online]. Long Term Ecological Research Network. http://www.lternet.edu/news/article197.html (accessed 17 March, 2009).

Ludwig, D. 2001. The era of management is over. Ecosystems 4:758-764.

Lund, M., Falk, J.M., Friborg, T., Mbufong, H.N., Sigsgaard, C., Soegaard, H. and Tamstorf, M.P. 2012. Trends in CO2 exchange in a High Arctic tundra heath, 2000-2010. Journal of Geophysical Research-Biogeosciences 117.

Lunn, N.J., Stirling, I., Andriashek, D. and Richardson, E. 2004. Selection of maternity dens by female polar bears in Western Hudson Bay, Canada and the effects of human disturbance. Polar Biology 27:350-356.

Lyberth, B. 2003. Muskox in the high Arctic, life history and spatiotemporal differences. Thesis (M.Sc.). University of Copenhagen, Department of Population Ecology. Copenhagen, Denmark.

Lyon, B. and Montgomerie, R. 2011. Snow bunting (*Plectrophenax nivalis*). *In* The birds of North America online. Edited by Poole, A. Cornell Lab of Ornithology. Ithaca, NY. http://bna.birds.cornell.edu/bna/species/198.

Mack, M.C., Bret-Harte, M.S., Hollingsworth, T.N., Jandt, R.R., Schuur, E.A.G., Shaver, G.R. and Verbyla, D.L. 2011. Carbon loss from an unprecedented Arctic tundra wildfire. Nature 475:489-492.

Mackay, J.R. 1995. Active layer changes (1968 to 1993) following the forest-tundra fire near Inuvik, NWT, Canada. Arctic and Alpine Research 27:323-336.

Macpherson, A.H. 1964. A northward range extension of the red fox in the eastern Canadian Arctic. Journal of Mammalogy 45:138-140.

Mair, D. 2012. Glaciology: Research update I. Progress in Physical Geography 36:813-832.

Mair, D., Burgess, D., Sharp, M., Dowdeswell, J.A., Benham, T., Marshall, S. and Cawkwell, F. 2009. Mass balance of the Prince of Wales Icefield, Ellesmere Island, Nunavut, Canada. Journal of Geophysical Research-Earth Surface 114, F02011:1-15.

Manning, T.H. 1960. The relationship of the Peary caribou and barren-ground caribou. Technical Paper No. 4. Arctic Institute of North America. Montréal, QC. 42 p.

Marchand, F.L., Nijs, I., Heuer, M., Mertens, S., Kockelbergh, F., Pontailler, J., Impens, I. and Beyens, L. 2004. Climate warming postpones senescence in High Arctic tundra. Arctic, Antarctic, and Alpine Research 36:390-394.

Marquard-Petersen, U. 1998. Food habits of arctic wolves in Greenland. Journal of Mammalogy 79:236-244.

Marquez, J. and Eagles, P.F.J. 2007. Working towards policy and guideline creation for cruise ship tourism in parks and protected areas in the eastern Canadian Arctic. Tourism in Marine Environments 4:85-96.

Marsh, D.B. 1938. The influx of the red fox and its colour phases into the barren lands. Canadian Field Naturalist 52:60-61.

Maxwell, J.B. 1981. Climatic regions of the Canadian Arctic islands. Arctic 34:225-240.

McClelland, J.W., Déry, S.J., Peterson, B.J., Holmes, R.M. and Wood, E.F. 2006a. A pan-arctic evaluation of changes in river discharge during the latter half of the 20th century. Geophysical Research Letters 33, L06715:1-4.

McClelland, J.W., Dery, S.J., Peterson, B.J., Holmes, R.M. and Wood, E.F. 2006b. A pan-arctic evaluation of changes in river discharge during the latter half of the 20th century. Geophysical Research Letters 33.

Mcdonald, J.C., Gyorkos, T.W., Alberton, B., Maclean, J.D., Richer, G. and Juranek, D. 1990. An outbreak of toxoplasmosis in pregnant women in Northern Quebec. Journal of Infectious Diseases 161:769-774.

McDonald, M., Arragutainaq, L. and Novalinga, Z. (compilers). 1997. Voices from the bay: Traditional Ecological Knowledge of Inuit and Cree in the Hudson Bay bioregion. Canadian Arctic Resources Committee and Environmental Committee of the Municipality of Sanikiluaq. Ottawa, ON. xiii + 98 p.

McGhee, R. 1978. Canadian Arctic prehistory. Van Nostrand Reinhold. Toronto and New York. 128 p.

McGuire, A.D., Anderson, L.G., Christensen, T.R., Dallimore, S., Guo, L.D., Hayes, D.J., Heimann, M., Lorenson, T.D., MacDonald, R.W. and Roulet, N. 2009. Sensitivity of the carbon cycle in the Arctic to climate change. Ecological Monographs 79:523-555.

Mckinney, M.A., Letcher, R.J., Aars, J., Born, E.W., Branigan, M., Dietz, R., Evans, T.J., Gabrielsen, G.W., Peacock, E. and Sonne, C. 2011. Flame retardants and legacy contaminants in polar bears from Alaska, Canada, East Greenland and Svalbard, 2005-2008. Environment International 37:365-374.

McKinnon, L., Picotin, M., Bolduc, E., Juillet, C. and Bety, J. 2012. Timing of breeding, peak food availability, and effects of mismatch on chick growth in birds nesting in the high Arctic. Canadian Journal of Zoology-Revue Canadienne de Zoologie 90:961-971.

McLennan, D., Bell, T., Berteaux, D., Chen, W., Copland, L., Fraser, R., Gallant, D., Gauthier, G., Hik, D., Krebs, C., Myers-Smith, I., Olthof, I., Reid, D., Sladen, W., Tarnocai, C., Vincent, W. and Zhang, Y. 2012. Recent climate-related terrestrial biodiversity research in Canada's Arctic national parks: review, summary, and management implications. Biodiversity 13:157-173.

McLoughlin, P.D., Cluff, H.D. and Messier, F. 2002. Denning ecology of barren-ground grizzly bears in the central Arctic. Journal of Mammalogy 83:188-198.

McLoughlin, P.D., R.L.Case, R.J.Gau, S.H.Ferguson and F.Messier. 1999. Annual and seasonal movement patterns of barren-ground grizzly bears in the Central Northwest Territories. *Ursus*, A selection of papers from the eleventh international conference on bear research and management, Graz, Austria, September 1997, and Gatlinburg, Tennessee, April 1998. International Association for Bear Research and Management. pp. 79-86.

McLoughlin, P.D., Taylor, M.K., Cluff, H.D., Robert, J.G., Mulders, R., Case, R.L., Stan, B. and Messier, F. 2003. Demography of barren-ground grizzly bears. Canadian Journal of Zoology 81:294-301.

McLoughlin, P.D., Walton, L.R., Cluff, H.D., Paquet, P.C. and Ramsay, M.A. 2004. Hierarchical habitat selection by tundra wolves. Journal of Mammalogy 85:576-580.

McRae, L., Zöckler, C., Gill, M., Loh, J., Latham, J., Harrison, N., Martin, J. and Collen, B. 2010. Arctic Species Trend Index 2010. Tracking trends in arctic wildlife. Circumpolar Biodiversity Monitoring Program. 39 p.

Mech, D. 1974. *Canis lupus*. Mammalian Species 37:1-6.

Mech, L.D. 2004. Is climate change affecting wolf populations in the High Arctic? Climatic Change 67:87-93.

Mech, L.D. 2007. Annual arctic wolf pack size related to arctic hare numbers. Arctic 60:309-311.

Mech, L.D. and Adams, L.G. 1999. Killing of a muskox, *Ovibos moschatus*, by two wolves, *Canis lupus*, and subsequent caching. Canadian Field-Naturalist 113:673-675.

Meerburg, B.G. and Kijlstra, A. 2009. Changing climate-changing pathogens: *Toxoplasma gondii* in North-Western Europe. Parasitology Research 105:17-24.

Meltofte, H., Piersma, T., Boyd, H., McCaffery, B., Ganter, B., Golovnyuk, V.V., Graham, K., Gratto-Trevor, C.L., Morrison, R.I.G., Nol, E., Rösner, H.-U., Schamel, D., Schekkerman, H., Soloviev, M.Y., Tomkovich, P.S., Tracy, D.M., Tulp, I. and Wennerberg, L. 2007. Effects of climate variation on the breeding ecology of arctic shorebirds. Meddelelser om Grønland Bioscience 59. Danish Polar Center. Copenhagen, Denmark. 48 p.

Melvin, E.F. and Parrish, J.K. (eds.). 2001. Seabird bycatch: trends, roadblocks and solutions. Proceedings of the symposium. Blaine, WA, 26-27 February, 1999. University of Alaska Sea Grant, AK-SG-01-01. Fairbanks, AK. 206 p.

Michelutti, N., Douglas, M.S.V. and Smol, J.P. 2003. Diatom response to recent climatic change in a high Arctic lake (Char Lake, Cornwallis Island, Nunavut). Global and Planetary Change 38:257-271.

Michelutti, N., Wolfe, A.P., Vinebrooke, R.D., Rivard, B. and Briner, J.P. 2005. Recent primary production increases in arctic lakes. Geophysical Research Letters 32, L19715:1-4.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: biodiversity synthesis. Millennium Ecosystem Assessment Series. World Resources Institute. Washington, DC. 100 p.

Miller, F.L. 1987. Peary caribou and muskoxen on Bathurst, Alexander, Marc, Massey, Vanier, Cameron, Helena, Lougheed, and Edmund Walker Islands, Northwest Territories, July 1985. Technicial Report Series No. 20. Canadian Wildlife Service, Prairie and Northern Region. Edmonton, AB. 45 p.

Miller, F.L. 1988. Peary caribou and muskoxen on Melville and Byam Martin islands, Northwest Territories, July 1987. Technical Report Series No. 37. Canadian Wildlife Service, Prairie and Northern Region. Edmonton, AB. 58 p.

Miller, F.L. 1991a. Estimating Bathurst Island Peary caribou and muskox populations. Arctic 44:57-62.

Miller, F.L. 1991b. Updated status report on the Peary caribou, *Rangifer tarandus pearyi*, in Canada. Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Ottawa, ON. 116 p.

Miller, F.L. 1998. Status of Peary caribou and muskox populations within the Bathurst Island complex, south-central Queen Elizabeth Islands, Northwest Territories, July 1996. Technical Report Series No. 317. Canadian Wildlife Service. Edmonton, AB. 147 p.

Miller, F.L. and Barry, S.J. 2009. Long-term control of Peary caribou numbers by unpredictable, exceptionally severe snow or ice conditions in a non-equilibrium grazing system. Arctic 62:175-189.

Miller, F.L., Barry, S.J. and Calvert, W.A. 2005. Conservation of Peary caribou based on a recalculation of the 1961 aerial survey on the Queen Elizabeth Islands, Arctic Canada. Rangifer Special Issue 16:65-75.

Miller, F.L. and Gunn, A. 1980. Behavioral responses of muskox herds to simulation of cargo slinging by helicopter, Northwest Territories. Canadian Field-Naturalist 94:52-60.

Miller, F.L. and Gunn, A. 2003. Catastrophic die-off of Peary caribou on the western Queen Elizabeth Islands, Canadian High Arctic. Arctic 56:381-390.

Miller, F.L., Russell, R.H. and Gunn, A. 1977. Distributions, movements and numbers of Peary caribou and muskoxen on western Queen Elizabeth Islands, Northwest Territories, 1972-74. Technical Report Series No. 40. Canadian Wildlife Service. Edmonton, AB. 55 p.

Miller, G.H., Bradley, R.S. and Andrews, J.T. 1975. The glaciation level and lowest equilibrium line altitude in the High Canadian Arctic: maps and climatic interpretation. Arctic and Alpine Research 7:155-168.

Milne, S.B., Park, R.W. and Stenton, D.R. 2012. Dorset culture land use strategies and the case of inland Southern Baffin Island. Canadian Journal of Archaeology 36:267-288.

Min, S.K., Zhang, X.B. and Zwiers, F. 2008. Human-induced Arctic moistening. Science 320:518-520.

Mlynowski, T.J., Hernandez-Henriquez, M.A. and Dery, S.J. 2011. An evaluation of hydrometric monitoring across the Canadian pan-Arctic region, 1950-2008. Hydrology Research 42:479-490.

Molau, U. 1993. Relationships between flowering phenology and life history strategies in tundra plants. Arctic and Alpine Research 25:391-402.

Molau, U. 1997. Responses to natural climatic variation and experimental warming in two tundra plant species with contrasting life forms: *Cassiope tetragona* and *Ranunculus nivalis*. Global Change Biology 3:97-107.

Monk, W.A. and Baird, D.J. 2011. Biodiversity in Canadian lakes and rivers. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 19. Canadian Councils of Resource Ministers. Ottawa, ON. Draft report.

Monk, W.A., Peters, D.L., Curry, R.A. and Baird, D.J. 2011. Quantifying trends in indicator hydroecological variables for regime-based groups of Canadian rivers. Hydrological Processes 25:3086-3100.

Mork, T. and Prestrud, A. 2004. Arctic rabies - A review. Acta Veterinaria Scandinavica 45:1-9.

Morneau, C. and Payette, S. 2000. Long-term fluctuations of a caribou population revealed by tree-ring data. Canadian Journal of Zoology 78:1784-1790.

Morrison, R.I.G., Aubry, Y., Butler, R.W., Beyersbergen, G.W., Downes, C., Donaldson, G.M., Gratto-Trevor, C.L., Hicklin, P.W., Johnston, V.H. and Ross, R.K. 2001. Declines in North American shorebird populations. Wader Study Group Bulletin 94:34-38.

Morrison, R.I.G., Davidson, N.C. and Wilson, J.R. 2007. Survival of the fattest: body stores on migration and survival in red knots *Calidris canutus islandica*. Journal of Avian Biology 38:479-487.

Morrison, R.I.G., McCaffery, B.J., Gill, R.E., Skagen, S.K., Jones, S.L., Page, G.W., Gratto-Trevor, C.L. and Andres, B.A. 2006. Population estimates of North American shorebirds, 2006. Wader Study Group Bulletin 111:67-85.

Morrison, R.I.G., Ross, R.K. and Niles, L.J. 2004. Declines in wintering populations of red knots in southern South America. Condor 106:60-70.

Morrissette, M., Bety, J., Gauthier, G., Reed, A. and Lefebvre, J. 2010. Climate, trophic interactions, density dependence and carry-over effects on the population productivity of a migratory Arctic herbivorous bird. Oikos 119:1181-1191.

Mowat, G. and Heard, D.C. 2006. Major components of grizzly bear diet across North America. Canadian Journal of Zoology-Revue Canadienne de Zoologie 84:473-489.

Mueller, F.P. 1995. Tundra esker systems and denning by grizzly bears, wolves, foxes, and ground squirrels in the central Arctic, Northwest Territories. File Report No. 115. Department of Renewable Resources, Government of the Northwest Territories. Yellowknife, NT. 84 p.

Muir, D. 2008. Temporal trends of peresistent organic pollutants and mercury in landlocked char in the High Arctic. Technical Background Reports: Ecosystem Status and Trends Report for Canada. In preparation.

Mulders, R.L. 2000. Wolverine ecology, distribution, and productivity in the Slave Geological Province. West Kitikmeot/Slave Study Society. 92 p.

Mullon, C., Freon, P. and Cury, P. 2005. The dynamics of collapse in world fisheries. Fish and Fisheries 6:111-120.

Musiani, M., Leonard, J.A., Cluff, H.D., Gates, C.C., Mariani, S., Paquet, P.C., Vila, C. and Wayne, R.K. 2007. Differentiation of tundra/taiga and boreal coniferous forest wolves: genetics, coat colour and association with migratory caribou. Molecular Ecology 16:4149-4170.

Myers, J.P., Morrison, R.I.G., Antas, P.Z., Harrington, B.A., Lovejoy, T.E., Sallaberry, M., Senner, S.E. and Tarak, A. 1987. Conservation strategy for migratory species. American Scientist 75:19-26.

Myers-Smith, I. 2007. Shrub line advance in alpine tundra of the Kluane Region: mechanisms of expansion and ecosystem impacts. Arctic 60:447-451.

Myers-Smith, I.H., Forbes, B.C., Wilmking, M., Hallinger, M., Lantz, T., Blok, D., Tape, K.D., Macias-Fauria, M., Sass-Klaassen, U., Levesque, E., Boudreau, S., Ropars, P., Hermanutz, L., Trant, A., Collier, L.S., Weijers, S., Rozema, J., Rayback, S.A., Schmidt, N.M., Schaepman-Strub, G., Wipf, S., Rixen, C., Menard, C.B., Venn, S., Goetz, S., Andreu-Hayles, L., Elmendorf, S., Ravolainen, V., Welker, J., Grogan, P., Epstein, H.E. and Hik, D.S. 2011a. Shrub expansion in tundra ecosystems: dynamics, impacts and research priorities. Environmental Research Letters 6 045509.

Myers-Smith, I.H., Hik, D.S., Kennedy, C., Cooley, D., Johnstone, J.F., Kenney, A.J. and Krebs, C.J. 2011b. Expansion of canopy-forming willows over the twentieth century on Herschel Island, Yukon Territory, Canada. Ambio 40:610-623.

NABCI. 2012. The state of Canada's birds. Environment Canada, on behalf of the North American Bird Conservation Initiative Canada. Ottawa, ON. 36 p.

Nadelhoffer, K.J., Shaver, G.R., Giblin, A.E. and Rastetter, E.B. 1997. Potential impacts of climate change on nutrient cycling, decomposition, and productivity in Arctic ecosystems. *In* Global change and Arctic terrestrial ecosystems. Edited by Oechel, W.C. Springer-Verlag. New York, NY. pp. 349-364.

Nagy, J. and Gunn, A. 2006. Productivity of Peary caribou on Banks, Northwest Victoria and Melville islands. Department of Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. Unpublished report.

Nagy, J. and Gunn, A. 2009. Productivity of Peary caribou and muskoxen on Banks and Melville Islands, NT, July 2004. Manuscript Report No. 204. Department of Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 25 p.

Nagy, J.A. 2007. Plummeting numbers: the mysterious decline of the Arctic coastal herds. Northern Perspectives 31:36-37.

Nagy, J.A., Gunn, A. and Wright, W.H. 2009a. Population estimates for Peary caribou and muskox on Banks Island, NT, August 1992. Manuscript Report No. 198. Department of Environment and Natural Resources, Government of the Northwest Territories. Inuvik, NT and Yellowknife, NT. 31 p.

Nagy, J.A., Johnson, D.L., Larter, N.C., Campbell, M.W., Derocher, A.E., Kelly, A., Dumond, M., Allaire, D. and Croft, B. 2011. Subpopulation structure of caribou (*Rangifer tarandus* L.) in Arctic and Subarctic Canada. Ecological Applications 21:2334-2348.

Nagy, J.A., Larter, N. and Wright, W.H. 2009b. Population estimates for Peary caribou (Minto Inlet Herd), Dolphin and Union caribou, and muskox on northwest Victoria Island, NT, July 1998. Manuscript Report No. 201. Department of Environment and Natural Resources, Government of the Northwest Territories. Inuvik, NT and Fort Simpson, NT. 29 p.

Nagy, J.A., Larter, N. and Wright, W.H. 2009c. Population estimates for Peary caribou (Minto Inlet Herd), Dolphin and Union caribou, and muskox on northwest Victoria Island, NT, July 2001. Manuscript Report No. 202. Department of Environment and Natural Resources, Government of the Northwest Territories. Inuvik, NT and Fort Simpson, NT. 36 p.

Nagy, J.A., Larter, N.C. and Fraser, V.P. 1996. Population demography of Peary caribou and muskox on Banks Island, N.W.T. 1982-1992. Rangifer Special Issue 9:213-222.

Nagy, J.A., Latour, P. and Wright, W.H. 2009d. Population estimates for Peary caribou and muskox on Banks Island, NT, July 1982: a retrospective analysis. Manuscript Report No. 197. Department of Environment and Natural Resources, Government of the Northwest Territories. Inuvik, NT. 42 p.

Nagy, J.A., Russell, R.H., Pearson, A.M., Kingsley, M.C.S. and Larsen, C.B. 1983. A study of grizzly bears on the barren grounds of Tuktoyaktuk Peninsula and Richards Island, Northwest Territories, 1974 to 1978. Canadian Wildlife Service. Edmonton, AB. 136 p.

NASA. 2013. GISS surface temperature anaysis (GISTEMP) [online]. http://data.giss.nasa.gov/gistemp/ (accessed 7 March, 2013).

Natcher, D.C., Felt, L. and Procter, A. (eds.). 2012. The Nunatsiavummiut experience: Settlement, subsistence, and change among the Labrador Inuit. University of Manitoba Press. Winnipeg.

National Snow and Ice Data Center. 2013. Sea ice index [online]. http://nsidc.org/data/seaice_index/

Natural Resources Canada. 2010. Coastline and shoreline (the atlas of Canada) [online]. http://atlas.nrcan.gc.ca/site/english/learningresources/facts/coastline.html (accessed 23 May, 2010).

Nellemann, C. and Cameron, R.D. 1998. Cumulative impacts of an evolving oil-field complex on the distribution of calving caribou. Canadian Journal of Zoology 76:1425-1430.

Nesbitt, L. and Adamczewski, J. 2009. Decline and recovery of the Bathurst Caribou Herd: summary report of workshops Oct. 1-2 and 5-6 in Yellowknife. 35 p.

Neufeld, D. 2002. The Distant Early Warning (DEW) Line: a preliminary assessment of its role and effects upon northern Canada, revised for the Arctic Institute of North America, May, 2002. Parks Canada. 21 p.

Nickels, S., Furgal, C., Buell, M. and Moquin, H. 2005. Unikkaaqatigiit - putting the human face on climate change: Perspectives from Inuit in Canada. Joint publication of Inuit Tapiriit Kanatami Nasivvik Centre for Inuit Health and Changing Environments at Université Laval and Ajunnginiq Centre at the National Aboriginal Health Organization. Ottawa. 124 p.

Nickels, S., Furgal, C., Castleden, J., Moss-Davies, P., Buell, M., Armstrong, B., Dillion, D. and Fonger, R. 2002. Putting the human face on climate change through community workshops: Inuit knowledge, partnerships and research. *In* The earth is faster now: Indigenous observations of arctic environmental change. Edited by Krupnik, I. and Jolly, D. Arctic Research Consortium of the United States. Fairbanks, AK.

Niemi, A., Paulic, J. and Cobb, D. 2010. Ecosystem status and trends report: Arctic marine ecozones. Canadian Science Advisory Secretariat Research Document No. 2010/066. Fisheries and Oceans Canada. Winnipeg MB. 66 p.

Nishi, J.S. and Gunn, A. 2004. An estimate of the herd size for the migratory Dolphin and Union Caribou Herd during the rut (17-22 October 1997). File Report No. 131. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife, NT. 52 p.

Niven, D.K., Sauer, J.R., Butcher, G.S. and Link, W.A. 2004. Population change in boreal birds from the Christmas Bird Count. American Birds 58:10-20.

Noren, K., Carmichael, L., Dalen, L., Hersteinsson, P., Samelius, G., Fuglei, E., Kapel, C.M.O., Menyushina, I., Strobeck, C. and Angerbjorn, A. 2011. Arctic fox *Vulpes lagopus* population structure: circumpolar patterns and processes. Oikos 120:873-885.

Norment, C.J. and Shackleton, S.A. 2008. Harris's sparrow (*Zonotrichia querula*). *In* The birds of North America [online]. Edited by Poole, A. Cornell Lab of Ornithology. Ithaca, NY. http://bna.birds.cornell.edu/bna/species/064.

Norrdahl, K. 1995. Population cycles in northern small mammals. Biological Reviews 70:621-637.

Northern Contaminants Program. 2003. Canadian Arctic Contaminants Assessment Report II: highlights. Indian and Northern Affairs and Northern Development. Ottawa, ON. 118 p.

Nunatsiavut Government. 2009. Labrador Inuit Land Claim [online]. http://www.nunatsiavut.com/index.php/lilca (accessed 1 December, 2012).

Nunavut Research Institute. 2004. A summary of results from climate change research undertaken in Nunavut during 2004 [online]. Compiled by Nunavut Research Institute and the Canadian Cimate Impacts and Adaptation Research Network (C-CAIRN) North. http://www.nri.nu.ca/publications.html (accessed 30 November, 2009).

Nunavut Tusaavut Inc. 1997. Travelling to Bathurst Island: interviews from Resolute Bay. Parks Canada. Ottawa, ON. 91 p.

NWT Department of Environment and Natural Resources. 2011. 2011 Annual report of NWT wildlife research permits. Department of Environment and Natural Resources, Government of the Northwest Territories. 29 p.

NWT Department of Environment and Natural Resources. 2012. Moose in the NWT [online]. Government of the Northwest Territories. http://www.enr.gov.nt.ca/_live/pages/wpPages/Moose.aspx (accessed 20 July, 2013).

NWT Environment and Natural Resources. 2011. Summary of harvest data for species under quota in the inuvialuit Settlement Region July 2006 to June 2011. Government of the Northwest Territories. Inuvik, NT.

Obbard, M.E., Cattet, M.R.L., Moody, T., Walton, L.R., Potter, D., Inglis, J. and Chenier, C. 2006. Temporal trends in the body condition of southern Hudson Bay polar bears. Climate Change Research Information Note No. 3. Applied Research and Development Branch, Ministry of Natural Resources, Government of Ontario. 8 p.

Obbard, M.E., Thiemann, G.W., Peacock, E. and DeBruyn, T.D. (eds.). 2010. Polar bears: Proceedings of the 15th Working Meeting of the IUCN/SSC Polar Bear Specialist Group, Copenhagen, Denmark, 29 June-3 July 2009. IUCN. Gland and Cambridge. 235 p.

Oberbauer, S.F., Elmendorf, S.C., Troxler, T.G., Hollister, R.D., Rocha, A.V., Bret-Harte, M.S., Dawes, M.A., Fosaa, A.M., Henry, G.H., Høve, T.T., Jarrad, F.C., Jónsdóttir, I.S., Klanderud, K., Klein, J.A., Molau, U., Rixen, C., Schmidt, N.M., Shaver, G.R., Slider, R.T., Totland, O., Wahren, C.H. and Welker, J.M. 2013. Phenological responses of tundra plants to background climate variation tested using the International Tundra Experiment (ITEX). Phil Trans Roy Soc B 368(1624).

Oberbauer, S.F., Tweedie, C.E., Welker, J.M., Fahnestock, J.T., Henry, G.H.R., Webber, P.J., Hollister, R.D., Walker, M.D., Kuchy, A., Elmore, E. and Starr, G. 2007. Tundra CO_2 fluxes in response to experimental warming across latitudinal and moisture gradients. Ecological Monographs 77:221-238.

Oechel, W.C. and Vourlitis, G.L. 1997. Climate change in northern latitudes: alterations in ecosystem structure and function and effects on carbon sequestration. *In* Global change and Arctic terrestrial ecosystems. Edited by Oechel, W.C. Springer-Verlag. New York, NY. pp. 381-401.

Ohse, B., Jansen, F. and Wilmking, M. 2012. Do limiting factors at Alaskan treelines shift with climatic regimes? Environmental Research Letters 7 015505 .

Olofsson, J., Stark, S. and Oksanen, L. 2004. Reindeer influence on ecosystem processes in the tundra. Oikos 105:386-396.

Olthof, I., C.Butson, R.Fernandes, R.Fraser, R.Latifovic and J.Orazietti. 2005. Landsat ETM+ mosaic of Northern Canada. Canadian Journal of Remote Sensing 31:412-419.

Ouranos. 2004. Adapting to climate change. Montreal, QC. 83 p.

Overland, J. and Wang, M. 2007. Future regional arctic sea ice declines. Geophysical Research Letters 34:1-7.

Overland, J.E., J.E.Walsh and M.Wang. 2007. Why are Ice and Snow Changing? *In* Global Outlook for Ice and Snow. Edited by Eamer, J. United Nations Environment Programme. Nairobi. Chapter 3. pp. 29-38.

Pamperin, N.J., Follmann, E.H. and Person, B.T. 2008. Sea-ice use by arctic foxes in Northern Alaska. Polar Biology 31:1421-1426.

Parmesan, C. 2007. Influences of species, latitudes and methodologies on estimates of phenological response to global warming. Global Change Biology 13:1860-1872.

Parsons, M., Mitchell, I., Butler, A., Ratchliffe, N., Frederiksen, M., Foster, S. and Reid, J.B. 2008. Seabirds as indicators of the marine environment. ICES Journal of Marine Science 65:1520-1526.

Payette, S. 2007. Contrasted dynamics of northern Labrador tree lines caused by climate change and migrational lag. Ecology 88:770-780.

Payette, S., Boudreau, S., Morneau, C. and Pitre, N. 2004a. Long-term interactions between migratory caribou, wildfires and Nunavik hunters inferred from tree rings. Ambio 33:482-486.

Payette, S., Delwaide, A., Caccianiga, M. and Beauchemin, M. 2004b. Accelerated thawing of Subarctic peatland permafrost over the last 50 years. Geophysical Research Letters 31, L18208, 4 p.

PCMB. 2009. Harvest management plan for the Porcupine Caribou Herd in Canada. Porcupine Caribou Management Board. 41 p.

Peacock, E., Derocher, A.E., Thiemann, G.W. and Stirling, I. 2011. Conservation and management of Canada's polar bears (*Ursus maritimus*) in a changing Arctic. Canadian Journal of Zoology-Revue Canadienne de Zoologie 89:371-385.

Peacock, E., Taylor, M.K., Laake, J. and Stirling, I. 2013. Population ecology of polar bears in Davis Strait, Canada and Greenland. Journal of Wildlife Management 77:463-476.

Perovich, D., Meier, W., Tshudi, M., Gerland, S. and Richter-Menge, J. 2012. Sea ice [in Arctic report card 2012] [online]. http://www.arctic.noaa.gov/reportcard/sea_ice.html (accessed 6 June, 2013).

Peters, D.L., Prowse, T.D., Pietroniro, A. and Leconte, R. 2006. Flood hydrology of the Peace-Athabasca Delta, Northern Canada. Hydrological Processes 20:4073-4096.

Peterson, B.J., Holmes, R.M., McClelland, J.W., Volosmarty, C.J. and Lammers, R.B. 2002. Increasing river discharge to the Arctic Ocean. Science 298:2171-2173.

Peterson, B.J., McClelland, J., Curry, R., Holmes, R.M., Walsh, J.E. and Aargaard, K. 2006. Trajectory shifts in the Arctic and Subarctic freshwater cycle. Science 313:1061-1066.

Philippa, J.D.W., Leighton, F.A., Daoust, P.Y., Nielsen, O., Pagliarulo, M., Schwantje, H., Shury, T., Van Herwijnen, R., Martina, B.E.E., Kuiken, I., Van De Bildt, M.W.G. and Osterhaus, A.D.M.E. 2004. Antibodies to selected pathogens in free-ranging terrestrial carnivores and marine mammals in Canada. Veterinary Record 155:135-140.

Pielou, E.C. 1991. After the ice age. University of Chicago Press. Chicago, IL. 366 p.

Piersma, T., Koolhaas, A., Dekinga, A., Beukema, J.J., Dekker, R. and Essink, K. 2001. Long-term indirect effects of mechanical cockle-dredging on intertidal bivalve stocks in the Wadden Sea. Journal of Applied Ecology 38:976-990.

Ping, C.L., Michaelson, G.J., Jorgenson, M.T., Kimble, J.M., Epstein, H., Romanovsky, V.E. and Walker, D.A. 2008. High stocks of soil organic carbon in the North American Arctic region. Nature Geoscience 1:615-619.

Pirie, L., Johnston, B.H. and Smith, P.A. 2012. Tier 2 surveys. *In* Arctic shorebirds in North America: a decade of monitoring. Studies in Avian Biology Monograph Series No. 44. Edited by Bart, J. and Johnston, B.H. University of California Press. Berkeley, CA. 320. In Press.

Polar Bear Technical Committee. 2013. Polar bear abundance estimates by population for Canada. Compilation of published and unpublished data from various sources.

Polyak, L., Alley, R.B., Andrews, J.T., Brigham-Grette, J., Cronin, T.M., Darby, D.A., Dyke, A.S., Fitzpatrick, J.J., Funder, S., Holland, M., Jennings, A.E., Miller, G.H., O'Regan, M., Savelle, J., Serreze, M., St John, K., White, J.W.C. and Wolff, E. 2010. History of sea ice in the Arctic. Quaternary Science Reviews 29:1757-1778.

Poole, K.G., Gunn, A., Patterson, B.R. and Dumond, M. 2010. Sea ice and migration of the Dolphin and Union Caribou Herd in the Canadian Arctic: an uncertain future. Arctic 62:414-428.

Poole, K.G., Gunn, A. and Wierzchowski, J. 2013. An operations guide to barren-ground caribou calving ground density, dispersion and distribution surveys, based on an assessment of the June 2007 and 2008 surveys, Northwest Territories and Nunavut. Manuscript Report No. 226. Department of Environment and Natural Resources, Government of the Northwest Territories. Yellowknife, NT. 89 p.

Porcupine Caribou Management Board. 2010. Harvest management plan for the Porcupine Caribou Herd in Canada. First Nation of NaCho Nyäk Dun, Gwich'in Tribal Council, inuvialuit Game Council, Tr'ondëk Hwëch'in, Vuntut Gwitchin Government, Government of the Northwest Territories, Government of Yukon, and Government of Canada. 41 p.

Potter, L.D. 1972. Plant Ecology of Walakpa Bay Area, Alaska. Arctic 25:115-130.

Predavec, M., Krebs, C.J., Dannell, K. and Hyndman, R.J. 2001. Cycles and synchrony in the collared lemming *(Dicrostonyx groenlandicus)* in Arctic North America. Oecologia 126:216-224.

Priest, H. and Usher, P.J. 2004. The Nunavut wildlife harvest study. Nunavut Wildlife Management Board. Iqaluit, NU. 816 p.

Prowse, T., Alfredsen, K., Beltaos, S., Bonsal, B., Duguay, C., Korhola, A., McNamara, J., Vincent, W.F., Vuglinsky, V. and Weyhenmeyer, G.A. 2011. Arctic freshwater ice and its climatic role. Ambio 40:46-52.

Prowse, T.D., Wrona, F.J., Reist, J.D., Hobbie, J.E., Lévesque, L.M.J. and Warwick, W.F. 2006. General features of the Arctic relevant to climate change in freshwater ecosystems. Ambio 35:330-338.

Pruitt, W.O.J. 1957. Observations on the bioclimate of some taiga mammals. Arctic 10:130-138.

Putkonen, J. and Roe, G. 2003. Rain-on-snow events impact soil temperatures and affect ungulate survival. Geophysical Research Letters 30:1-37.

Quakenbush, L.T., Suydam, R.S., Acker, R., Knoche, M. and Citta, J. 2013. Migration of king and common eiders past Point Barrow, Alaska, during summer/fall 2002 through spring 2004: Population trends and effects of wind. OCS Study MMS No. 2009-036. Coastal Marine Institute, University of Alaska.

Quinlan, R., Douglas, M.S.V. and Smol, J.P. 2005. Food web changes in arctic ecosystems related to climate warming. Global Change Biology 11:1381-1386.

Rankin, R., Austin, M. and Rice, J. 2011. Ecological classification system for the ecosystem status and trends report. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 1. Canadian Councils of Resource Ministers. Ottawa, ON. ii + 14 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-1.

Rannie, W.F. 1986. Summer air-temperature and number of vascular species in Arctic Canada. Arctic 39:133-137.

Raven, G.H. and Dickson, D.L. 2006. Changes in distribution and abundance of birds on Western Victoria Island from 1992-1994 to 2004-2005. Technical Report Series No. 456. Canadian Wildlife Service, Environment Canada. Edmonton, AB. 60 p.

Raven, G.H. and Dickson, D.L. 2009. Surveys of Pacific common eiders (*Somateria mollissima v-nigra*) in the Bathurst Inlet area of Nunavut, 2006-2008. Technical Report Series No. 503. Canadian Wildlife Service. Edmonton, Alberta. 58 p.

Reed, E.T. and Calvert, A.M. 2007. Evaluation of the special conservation measures for greater snow geese : report of the Greater Snow Goose Working Group. Greater Snow Goose Working Group. Edited by Reed, E.T. and Calvert, A.M. Canadian Wildlife Service. Sainte-Foy, QC. 88 p.

Regehr, E.V., Hunter, C.M., Caswell, H., Amstrup, S.C. and Stirling, I. 2010. Survival and breeding of polar bears in the southern Beaufort Sea in relation to sea ice. Journal of Animal Ecology 79:117-127.

Regehr, E.V., Lunn, N.J., Amstrup, S.C. and Stirling, L. 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management 71:2673-2683.

Reichard, M.V., Torretti, L., Garvon, J.M. and Dubey, J.P. 2008. Prevalence of antibodies to *Toxoplasma gondii* in wolverines from Nunavut, Canada. Journal of Parasitology 94:764-765.

Reid, D.G., Berteaux, D. and Laidre, K. 2013. Mammals. *In* Arctic biodiversity assessment. Edited by Meltofte, H. Conservation of Arctic Flora and Fauna. Akureyri, Iceland. Chapter 3. pp. 79-141.

Reid, D.G., Doyle, F.I., Kenney, A.J. and Krebs, C.J. 2011. Some observations of short-eared owl, *Asio flammeus*, ecology on arctic tundra, Yukon, Canada. Canadian Field-Naturalist 125:307-315.

Reid, D.G., Bilodeau, F., Krebs, C.J., Gauthier, G., Kenney, A.J., Gilbert, B., Leung, M.C., Duchesne, D. and Hofer, E. 2012. Lemming winter habitat choice: a snow-fencing experiment. Oecologia 168:935-946.

Reist, J.D., Wrona, F.J., Prowse, T.D., Dempson, J.B., Power, M., Köck, G., Carmichael, T.J., Sawatzky, C.D., Lehtonen, H. and Tallman, R.F. 2006a. Effects of climate change and UV radiation on fisheries for arctic freshwater and anadromous species. Ambio 35:402-410.

Reist, J.D., Wrona, F.J., Prowse, T.D., Power, M., Dempson, J.B., Beamish, R.J., King, J.R., Carmichael, T.J. and Sawatzky, C.D. 2006b. General effects of climate change on arctic fishes and fish populations. Ambio 35:370-380.

Rennert, K.J., Roe, G., Putkonen, J. and Bitz, C.M. 2009. Soil thermal and ecological impacts of rain on snow events in the circumpolar Arctic. Journal of Climate 22:2302-2315.

Reynolds, P.E. 1998. Dynamics and range expansion of a reestablished muskox population. The Journal of Wildlife Management 62:734-744.

Rich, T.D., Beardmore, C.J., Berlanga, H., Blancher, P.J., Bradstreet, M.S.W., Butcher, G.S., Demarest, D.W., Dunn, E.H., Hunter, W.C., Iñigo-Elias, E.E., Kennedy, J.A., Martell, A.M., Panjabe, A.O., Pashley, D.N., Rosenberg, K.V., Rustay, C.M., Wendt, J.S. and Will, T.C. 2004. Partners in flight North American landbird conservation plan. Cornell Lab of Ornithology. Ithaca, NY. 84 p.

Richter-Menge, J., Overland, J., Proshutinsky, A., Romanovsky, V., Bengtsson, L., Brigham, L., Dyurgerov, M., Gascard, J.C., Gerland, S., Graversen, R., Haas, C., Karcher, M., Kuhry, P., Maslanik, J., Melling, H., Maslowski, W., Morison, J., Perovich, D., Przybylak, R., Rachold, V., Rigor, I., Shiklomanov, A., Stroeve, J., Walker, D. and Walsh, J. 2006. State of the Arctic report. NOAA/OAR Special Report. NOAA/OAR/PMEL. Seattle, WA. 36 p.

Riget, F., Bignert, A., Braune, B., Stow, J. and Wilson, S. 2010. Temporal trends of legacy POPs in Arctic biota, an update. Science of the Total Environment 408:2874-2884.

Riget, F., Braune, B., Bignert, A., Wilson, S., Aars, J., Born, E., Dam, M., Dietz, R., Evans, M., Evans, T., Gamberg, M., Gantner, N., Green, N., Gunnlaugsdottir, H., Kannan, K., Letcher, R., Muir, D., Roach, P., Sonne, C., Stern, G. and Wiig, O. 2011. Temporal trends of Hg in Arctic biota, an update. Science of the Total Environment 409:3520-3526.

Roach, J. 16 May 2006. Grizzly-polar bear hybrid found - but what does it mean? National Geographic News.

Robertson, G.J. and Gilchrist, H.G. 1998. Evidence of population declines among common eiders breeding in the Belcher Islands, Northwest Territories. Arctic 51:378-385.

Rocha, A.V., Loranty, M.M., Higuera, P.E., Mack, M.C., Hu, F.S., Jones, B.M., Breen, A.L., Rastetter, E.B., Goetz, S.J. and Shaver, G.R. 2012. The footprint of Alaskan tundra fires during the past half-century: implications for surface properties and radiative forcing. Environmental Research Letters 7 044039.

Rode, K.D., Peacock, E., Taylor, M., Stirling, I., Born, E.W., Laidre, K.L. and Wiig, O. 2012. A tale of two polar bear populations: ice habitat, harvest, and body condition. Population Ecology 54:3-18.

Rolph, S.G. 2003. Effects of a ten-year climate warming experiment on nitrogen cycling in high Arctic tundra. Thesis (M.Sc.). University of British Columbia. Vancouver. 61 p.

Romanovsky, V.E., Smith, S.L., Christiansen, H.H., Shiklomanov, N.I., Streletskiy, D.A., Drozdov, D.S., Oberman, N.G., Knolodov, A.L. and Marchenko, S.S. 2012. Permafrost [in Arctic report 2012] [online]. http://www.arctic.noaa.gov/reportcard/permafrost.html (accessed 6 June, 2013).

Root, T.L., Price, J.T., Hall, K.R., Schneider, S.H., Rosenzweig, C. and Pounds, J.A. 2003. Fingerprints of global warming on wild animals and plants. Nature 421:57-60.

Ross, P.I. 2002. Update COSEWIC status report on the grizzly bear *Ursus arctos* in Canada. *In* COSEWIC Assessment and update status report on the grizzly bear *Ursus arctos* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, ON. 91.

Roth, J.D. 2003. Variability in marine resources affects arctic fox population dynamics. Journal of Animal Ecology 72:668-676.

Rothfels, M. and Russell, D. 2005. Mammal fact sheets: caribou. Hinterland Who's Who. Canadian Wildlife Federation and Environment Canada. http://www.hww.ca/en/species/mammals/caribou.html.

Rouse, W.R., Douglas, M.S.V., Hecky, R.E., Hershey, A.E., Kling, G.W., Lesack, L., Marsh, P., McDonald, M., Nicholson, B.J., Roulet, N.T. and Smol, J.P. 1997. Effects of climate change on the freshwaters of arctic and subarctic North America. Hydrological Processes 11:873-902.

Rudy, A.C.A., Lamoureux, S.F., Treitz, P. and Collingwood, A. 2013. Identifying permafrost slope disturbance using multi-temporal optical satellite images and change detection techniques. Cold Regions Science and Technology 88:37-49.

Rühland, K., Paterson, A.M. and Smol, J.P. 2008. Hemispheric-scale patterns of climate-related shifts in planktonic diatoms from North American and European lakes. Global Change Biology 14:2740-2754.

Ruhland, K., Priesnitz, A. and Smol, J.P. 2003. Paleolimnological evidence from diatoms for recent environmental changes in 50 lakes across Canadian arctic treeline. Arctic Antarctic and Alpine Research 35:110-123.

Russell, D. 2012. Appendix 6H: Energy-protein modeling of north Baffin caribou in relatin to the Mary River mine project. *In* Mary River Project final environmental impact statement, Volume 6: Terrestrial environment. Baffinland Iron mines Corp. Toronto. 41.

Russell, D., Whitfield, P.H., Cai, J., Gunn, A., White, R.G. and Poole, K. 2013. CARMA's MERRA-based caribou range climate database. Rangifer 33:145-152.

Russell, D.E., Martell, A.M. and Nixon, W.A. 1993. The range ecology of the Porcupine Caribou Herd in Canada. Rangifer Special Issue 8:1-168.

Russell, D.E., White, R.G. and Daniel, C.J. 2005. Energetics of the Porcupine Caribou Herd: a computer simulation model. Technical Report Series No. 431. Canadian Wildlife Service. Ottawa, ON. 64 p.

Schimel, J.P., Bilbrough, C. and Welker, J.A. 2004. Increased snow depth affects microbial activity and nitrogen mineralization in two Arctic tundra communities. Soil Biology & Biochemistry 36:217-227.

Schimel, J.P. and Chapin III, F.S. 1996. Tundra plant uptake of amino acid and NH_4^+ nitrogen in situ: plants compete well for amino acid N. Ecology 77:2142-2147.

Schindler, D.W. and Smol, J.P. 2006. Cumulative effects of climate warming and other human activities on freshwaters of Arctic and Subarctic North America. Ambio 35:160-168.

Schmelzer, I. and Otto, R. 2003. Winter range drift in the George River Caribou Herd: a response to summer forage limitation? Rangifer Special Issue 14:113-122.

Schmidt, I.K., Jonasson, S., Shaver, G.R., Michelsen, A. and Nordin, A. 2002. Mineralization and distribution of nutrients in plants and microbes in four arctic ecosystems: responses to warming. Plant and Soil 242:93-106.

Schreiber, E.A. and Burger, J. 2002. Biology of marine birds. CRC Press. New York, NY. 722 p.

Schultz J. 2005. The ecozones of the world. Springer-Verlag. Berlin. 252 p.

Schuur, E.A.G., Bockheim, J., Canadell, J.G., Euskirchen, E., Field, C.B., Goryachkin, S.V., Hagemann, S., Kuhry, P., Lafleur, P.M., Lee, H., Mazhitova, G., Nelson, F.E., Rinke, A., Romanovsky, V.E., Shiklomanov, N., Tarnocai, C., Venevsky, S., Vogel, J.G. and Zimov, S.A. 2008. Vulnerability of permafrost carbon to climate change: Implications for the global carbon cycle. Bioscience 58:701-714.

Schuur, E.A.G., Vogel, J.G., Crummer, K.G., Lee, H., Sickman, J.O. and Osterkamp, T.E. 2009. The effect of permafrost thaw on old carbon release and net carbon exchange from tundra. Nature 459:556-559.

Scott, P.A. and Stirling, I. 2002. Chronology of terrestrial den use by polar bears in western Hudson Bay as indicated by tree growth anomalies. Arctic 55:151-166.

Screen, J.A. and Simmonds, I. 2012. Declining summer snowfall in the Arctic: causes, impacts and feedbacks. Climate Dynamics 38:2243-2256.

Senner, S.E. and Howe, M.A. 1984. Conservation of nearctic shorebirds. *In* Shorebirds: breeding behaviour and populations. Edited by Burger, J. and Olla, B.L. Plenum Press. New York, NY. pp. 379-421.

Serreze, M.C. and Barry, R.G. 2011. Processes and impacts of Arctic amplification: A research synthesis. Global and Planetary Change 77:85-96.

Shank, C.C. and Graf, R. 2013. Abundance and distribution of muskoxen near Aylmer Lake, NWT, July 1991. Manuscript Report No. 56. Department of Environment and Natural Resources, Northwest Territories. 16 p.

Sharma, S., Couturier, S. and Côté, S.D. 2009. Impacts of climate change on the seasonal distribution of migratory caribou. Global Change Biology 15:2549-2562.

Sharp, M., Wolken, G., Geai, M.-L. and Burgess, D. 2012. Mountain glaciers and ice caps (outside Greenland) [in Arctic report card 2012]. http://www.arctic.noaa.gov/report12/glaciers_ice_caps.html.

Shaver, G.R., Bret-Harte, S.M., Jones, M.H., Johnstone, J., Gough, L., Laundre, J. and Chapin, F.S. 2001. Species composition interacts with fertilizer to control long-term change in tundra productivity. Ecology 82:3163-3181.

Shaver, G.R., Canadell, J., Chapin, F.S., III, Gurevitch, J., Harte, J., Henry, G.H.R., Ineson, P., Jonasson, S., Melillo, J.M., Pitelka, L. and Rustad, L. 2000. Global warming and terrestrial ecosystems: A conceptual framework for analysis. Bioscience 50:871-882.

Shaver, G.R. and Chapin, F.S. 1995. Long-term responses to factorial, NPK fertilizer treatment by Alaskan wet and moist tundra sedge species. Ecography 18:259-275.

Shaver, G.R. and Kummerow, J. 1992. Phenology, resource allocation, and growth of arctic vascular plants. *In* Arctic ecosystems in a changing climate: an ecophysiological perspective. Edited by Chapin III, F.S., Jefferies, R.L., Reynolds, J.F., Shaver, G.R. and Svoboda, J. Academic Press Inc. San Diego, CA. pp. 193-211.

Shepherd, P.C.F., Evans Ogden, L.J. and Lank, D.B. 2003. Integrating marine and terrestrial habitats in shorebird conservation planning. Wader Study Group Bulletin 100:40-42.

Shiklomanov, A.I. and Lammers, R.B. 2011. River discharge [in Arctic report card 2011] [online]. http://www.arctic.noaa.gov/report11/river_discharge.html (accessed 28 June, 2013).

Skagen, S., Bart, J., Andres, B., Brown, S., Donaldson, G., Harrington, B.A., Johnston, V., Jones, S. and Morrison, R.I.G. 2003. Monitoring shorebirds of North America: towards a unified approach. Wader Study Group Bulletin 100:102-104.

Slaney, F.F. 1975. Peary caribou and muskoxen and Panarctic's seismic operations on Bathurst Island, N.W.T., 1975. Panarctic Oils Ltd. Calgary, AB. 11 p.

Slough, B.G. 2013. Species status report (scientific knowledge component) for wolverine (*Gulo gulo*) in the Northwest Territories, draft for expert review (26 April 2013). Prepared for the Northwest Territories Species at Risk Committee. **"Do not cite without permission of the SARC Chairperson" - check for update prior to publication**.

Smith, L.C., Sheng, Y., MacDonald, G.M. and Hinzman, L.D. 2005a. Disappearing arctic lakes. Science 308:1429-1429.

Smith, P.A., Gilchrist, H.G., Forbes, M.R., Martin, J.L. and Allard, K. 2010. Inter-annual variation in the breeding chronology of arctic shorebirds: effects of weather, snow melt and predators. Journal of Avian Biology 41:292-304.

Smith, R.L. and Smith, T.M. 2001. Ecology and field biology. Sixth Edition. Benjamin Cummings. 771 p.

Smith, S. 2011. Trends in permafrost conditions and ecology in Northern Canada. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 9. Canadian Councils of Resource Ministers. Ottawa, ON. iii + 22 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Smith, S.L., Burgess, M.M. and Heginbottom, J.A. 2001a. Permafrost in Canada, a challenge to northern development. *In* A synthesis of geological hazards in Canada. Edited by Brooks, G.R. Geological Survey of Canada, Bulletin 548. pp. 241-264.

Smith, S.L., Burgess, M.M. and Nixon, F.M. 2001b. Response of active-layer and permafrost temperatures to warming during 1998 in the Mackenzie Delta, Northwest Territories and at Canadian Forces station Alert and Baker Lake, Nunavut. Current Research 2001-E5. Geological Survey of Canada. 12 p.

Smith, S.L., Burgess, M.M., Riseborough, D. and Nixon, F.M. 2005b. Recent trends from Canadian permafrost thermal monitoring network sites. Permafrost and Periglacial Processes 16:19-30.

Smith, S.L., Burgess, M.M. and Taylor, A.E. 2003. High Arctic permafrost observatory at Alert, Nunavut - analysis of a 23 year data set. *In* Proceedings of the Eighth International Conference on Permafrost. Zurich, Switzerland, 21-25 July, 2003. Edited by Phillips, M., Springman, S.M. and Arenson, L.U. Swets & Zeitlinger. Lisse, Netherlands. pp. 1073-1078.

Smith, S.L., Throop, J. and Lewkowicz, A.G. 2012. Recent changes in climate and permafrost temperatures at forested and polar desert sites in northern Canada. Canadian Journal of Earth Sciences 49:914-924.

Smits, C.M.M. 2000. 2000 North Richardson Mountains moose survey results. Fish and Wildlife Branch, Yukon Department of Renewable Resources. Whitehorse, YT. 6 p. Unpublished manuscript.

Smol, J.P. and Douglas, M.S. 2007a. Crossing the final ecological threshold in High Arctic ponds. Proceedings of the National Academy of Sciences 104:12395-12397.

Smol, J.P. and Douglas, M.S.V. 2007b. From controversy to consensus: making the case for recent climate change using lake sediments. Frontiers in Ecology and the Environment 5:466-474.

Smol, J.P., Wolfe, A.P., Birkds, H.J.B., Douglas, M.S.V., Jones, V.J., Korhola, A., Pienitz, R., Ruhland, K., Sorvari, S., Antoniades, D., Brooks, S.J., Fullu, M.A., Huges, M., Keatley, B.E., Laing, T.E., Michelutti, N., Nazarova, L., Nyman, J., Paterson, A.M., Perren, B., Quinlan, R., Ruatio, M., Saulnier-Talbot, E., Sutonen, S., Solvieva, N. and Weckstrom, J. 2005. Climate-driven regime shifts in the biological communities of arctic lakes. Proceedings of the National Academy of Sciences 102:4397-4402.

Solomon, S., Qin, D., Manning, M., Alley, R.B., Bernsten, T., Bindoff, N.L., Chen, Z., Chidthaisong, A., Gregory, J.M., Hergerl, G.C., Heimann, M., Hewitson, B., Hoskins, B.J., Joos, F., Jouzel, J., Kattsov, V., Lohmann, U., Matsuno, T., Molina, M., Nicholls, N., Overpeck, J., Raga, G., Ramaswamy, V., Ren, J., Rusticucci, M., Somervilles, R., Stocker, T.F., Whetton, P., Wood, R.A. and Wratt, D. 2007. Technical summary. *In* Climate change 2007: the physical science basis. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Edited by Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L. Cambridge University Press. Cambridge, UK and New York, NY. pp. 19-91.

Sorvari, S., Korhola, A. and Thompson, R. 2002. Lake diatom response to recent Arctic warming in Finnish Lapland. Global Change Biology 8:171-181.

Species at Risk Committee. 2012a. Species status report for Peary caribou (*Rangifer tarandus pearyi*) in the Northwest Territories. Species at Risk Committee. Yellowknife, NT. 137 p.

Species at Risk Committee. 2012b. Species status report for polar bear (*Ursus maritimus*) in the Northwest Territories. Species at Risk Committee. Yellowknife, NT.

Species at Risk Public Registry. 2012. Polar bear (species profile) [online]. http://www.sararegistry.gc.ca/species/speciesDetails_e.cfm?sid=167 (accessed 7 July, 2013).

Spence, C. and Burke, A. 2008. Estimates of Canadian Arctic Archipelago runoff from observed hydrometric data. Journal of Hydrology 362:247-259.

Stankowich, T. 2008. Ungulate flight responses to human disturbance: a review and meta-analysis. Biological Conservation 141:2159-2173.

Statistics Canada. 2000. Human activity and the environment 2000. Human Activity and the Environment, Catalogue No. 11-509-XPE. Statistics Canada. Ottawa, ON. 332 p.

Statistics Canada. 2008a. Aboriginal peoples in Canada in 2006: Inuit, Métis and First Nations, 2006 census: findings [online]. Statistics Canada. http://www12.statcan.ca/english/census06/analysis/aboriginal/surpass.cfm

Statistics Canada. 2008b. Human activity and the environment: annual statistics 2007 and 2008. Human Activity and the Environment, Catalogue No. 16-201-X. Statistics Canada. Ottawa, ON. 159 p.

Statistics Canada. 2012. Focus on geography series, 2011 census. Statistics Canada Catalogue no. 98-310-XWE2011004. Analytical products, 2011 census. [online]. http://www12.statcan.gc.ca/census-recensement/2011/as-sa/fogs-spg/Facts-csd-eng.cfm?LANG=Eng&GK=CSD&GC=6204003 (accessed 16 October, 2013).

Stenseth, N.C., Ottersen, G., Hurrell, J.W. and Belgrano, A. 2004. Marine ecosystem and climate variation. Oxford University Press. Oxford, UK. 264 p.

Stenseth, N.C., Ottersen, G., Hurrell, J.W., Mysterud, A., Lima, M., Chan, K.S., Yoccoz, N.G. and Ådlandsvik, B. 2003. Review article. Studying climate effects on ecology through the use of climate indices: the North Atlantic Oscillation, El Niño Southern Oscillation and beyond. Proceedings of the Royal Society B: Biological Sciences 270:2087-2096.

Steven, B., Leveille, R., Pollard, W.H. and Whyte, L.G. 2006. Microbial ecology and biodiversity in permafrost. Extremophiles 10:259-267.

Steven, B., Pollard, W.H., Greer, C.W. and Whyte, L.G. 2008. Microbial diversity and activity through a permafrost/ground ice core profile from the Canadian High Arctic. Environmental Microbiology 10:3388-3403.

Stevenson, M.G. 1996. Indigenous knowledge in environmental assessment. Arctic 49:278-291.

Stewart, E., Howell, S., Draper, D., Yackel, J. and Tivy, A. 2007. Sea ice in Canada's Arctic: Implications for cruise tourism. Arctic 60:370-380.

Stewart, E., Tivy, A., Howell, S., Dawson, J. and Draper, D. 2010. Cruise tourism and sea ice in Canada's Hudson Bay region. Arctic 63:57-66.

Stewart, E.J., Howell, S.E., Draper, D., Yackel, J. and Tivy, A. 2008. Cruise ship tourism in a warming Arctic: implications for northern National parks [online]. http://dspace.ucalgary.ca/bitstream/1880/46913/1/Stewart.pdf (accessed 7 July, 2013).

Stewart, I.T., Cayan, D.R. and Dettinger, M.D. 2005. Changes toward earlier streamflow timing across Western North America. Journal of Climate 18:1136-1155.

Stirling, I. 1999. Polar bears. University of Michigan Press. Ann Arbor, MI. 232 p.

Stirling, I. 2002. Polar bears and seals in the Eastern Beaufort Sea and Amundsen Gulf: a synthesis of population trends and ecological relationships over three decades. Arctic 55:59-76.

Stirling, I. and Derocher, A.E. 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology 18:2694-2706.

Stirling, I., Lunn, N.J. and Iacozza, J. 1999. Long-term trends in the population ecology of polar bears in Western Hudson Bay in relation to climatic change. Arctic 52:294-306.

Stirling, I., Lunn, N.J., Iacozza, J., Elliott, C. and Obard, M. 2004. Polar bear distribution and abundance on the southwestern Hudson Bay coast during open water season. Arctic 15:15-24.

Stirling, I. and Parkinson, C.L. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. Arctic 59:261-275.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D., Flannigan, M.D., Hirsch, K.G., Logan, K.A., Martell, D.L. and Skinner, W.R. 2003. Large forest fires in Canada, 1959-1997. Journal of Geophysical Research Atmospheres 108, 8149 FFR 5:1-12.

Stow, D.A., Hope, A., McGuire, D., Verbyla, D., Gamon, J., Huemmrich, F., Houston, S., Racine, C., Sturm, M., Tape, K., Hinzman, L., Yoshikawa, K., Tweedie, C., Noyle, B., Silapaswan, C., Douglas, D., Griffith, B., Jia, G., Epstein, H., Walker, D., Daeschner, S., Petersen, A., Zhou, L. and Myneni, R. 2004. Remote sensing of vegetation and land-cover change in arctic tundra ecosystems. Remote Sensing of Environment 89:281-308.

Stow, J. 2008. Northern contaminants overview. Technical Background Reports: Ecosystem Status and Trends Report for Canada. In preparation.

Strathdee, A.T. and Bale, J.S. 1998. Life on the edge:Insect ecology in arctic environments. Annual Review of Entomology 43:85-106.

Sturm, M., Racine, C. and Tape, K. 2001. Climate change: increasing shrub abundance in the Arctic. Nature 411:546-547.

Sturm, M., Schimel, J., Michaelson, G., Welker, J.M., Oberbauer, S.F., Liston, G.E., Fahnestock, J. and Romanovsky, V.E. 2005. Winter biological processes could help convert Arctic tundra to shrubland. Bioscience 55:17-26.

Suarez, F., Binkley, D., Kaye, M.W. and Stottlemyer, R. 1999. Expansion of forest stands into tundra in the Noatak National Preserve, northwest Alaska. Ecoscience 6:465-470.

Sutton, J.T., Hermanutz, L. and Jacobs, J.D. 2006. Are frost boils important for the recruitment of arctic-alpine plants? Arctic Antarctic and Alpine Research 38:273-275.

Suydam, R.S., Dickson, D.L., Fadely, J.B. and Quakenbush, L.T. 2000. Population declines of king and common eiders of the Beaufort Sea. The Condor 102:219-222.

Szor, G., Berteaux, D. and Gauthier, G. 2008. Finding the right home: distribution of food resources and terrain characteristics influence selection of denning sites and reproductive dens in arctic foxes. Polar Biology 31:351-362.

Tape, K., Sturm, M. and Racine, C. 2006. The evidence for shrub expansion in northern Alaska and the Pan-arctic. Global Change Biology 12:686-702.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G. and Zimov, S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. Global Biogeochemical Cycles 23 GB2023.

Tarroux, A., Berteaux, D. and Bety, J. 2010. Northern nomads: ability for extensive movements in adult arctic foxes. Polar Biology 33:1021-1026.

Tarroux, A., Bety, J., Gauthier, G. and Berteaux, D. 2012. The marine side of a terrestrial carnivore: intra-population variation in use of allochthonous resources by arctic foxes. PLoS One 7 e42427.

Taylor, A.E., Wang, K.L., Smith, S.L., Burgess, M.M. and Judge, A.S. 2006. Canadian Arctic permafrost observatories: detecting contemporary climate change through inversion of subsurface temperature time series. Journal of Geophysical Research-Solid Earth 111, B02411:1-14.

Taylor, M. 2006. Thelon Wildlife Sanctuary. Final Wildlife Report No. 10. Wildlife Research Section, Department of Environment, Government of Nunavut. 5 p.

Taylor, M.K., Akeeagok, S., Andriashek, D., Barbour, W., Born, E.W., Calvert, W., Cluff, H.D., Ferguson, S., Laake, J., Rosing-Asvid, A., Stirling, I. and Messier, F. 2001. Delineating Canadian and Greenland polar bear (*Ursus maritimus*) populations by cluster analysis of movements. Canadian Journal of Zoology 79:690-709.

Tedesco, M., Brodzik, M., Armstrong, R., Savoie, M. and Ramage, J. 2009. Pan Arctic terrestrial snowmelt trends (1979-2008) from spaceborne passive microwave data and correlation with the Arctic Oscillation. Geophysical Research Letters 36 L21402.

Tener, J.S. 1957. A review of past and present distribution of musk-oxen (*Ovibus moschatus* subsp.) in Canada. CWSC No. 130. Canadian Wildlife Service, Department of Northern Affairs and Natural Resources. Ottawa, ON. 28 p.

Tener, J.S. 1963. Queen Elizabeth Islands game survey, 1961. Canadian Wildlife Service Occasional Paper No. 4. National Parks Branch, Department of Northern Affairs and National Resources. Ottawa, ON. 50 p.

Tener, J.S. 1965. Muskoxen in Canada: a biological and taxonomic review. Department of Northern Affairs and National Resources, Canadian Wildlife Service. Ottawa, ON. Monograph series - CWS. 166 p.

Tenter, A.M., Heckeroth, A.R. and Weiss, L.M. 2000. *Toxoplasma gondii*: from animals to humans. International Journal for Parasitology 30:1217-1258.

Tews, J., Ferguson, M. and Fahrig, L. 2007a. Modeling density dependence and climatic disturbances in caribou: a case study from the Bathurst Island complex, Canadian high Arctic. Journal of Zoology 272:209-217.

Tews, J., Ferguson, M. and Fahrig, L. 2007b. Potential net effects of climate change on high Arctic Peary caribou: Lessons from a spatially explicit simulation model. Ecological Modelling 207:85-98.

The Atlas of Canada. 2008a. Lakes [online]. Natural Resources Canada. http://atlas.nrcan.gc.ca/site/english/learningresources/facts/lakes.html (accessed 1 March, 2009a).

The Atlas of Canada. 2008b. Rivers [online]. Natural Resources Canada. http://atlas.nrcan.gc.ca/site/english/learningresources/facts/rivers.html (accessed 1 March, 2009b).

Therrien, J.F. 2012. Réponses des prédateurs aviaires aux fluctuations d'abondance de proies dans la toundra. Thesis (PhD). dépt. biologie, Université Laval, Québec.

Therrien, J.F., Gauthier, G. and Bety, J. 2011. An avian terrestrial predator of the Arctic relies on the marine ecosystem during winter. Journal of Avian Biology 42:363-369.

Thibault, S. and Payette, S. 2009. Recent permafrost degradation in bogs of the James Bay area, northern Quebec, Canada. Permafrost and Periglacial Processes 20:383-389.

Thienpont, J.R., Ruhland, K.M., Pisaric, M.F.J., Kokelj, S.V., Kimpe, L.E., Blais, J.M. and Smol, J.P. 2013. Biological responses to permafrost thaw slumping in Canadian Arctic lakes. Freshwater Biology 58:337-353.

Thomas, D.C., Miller, F.L., Russell, R.H. and Parker, G.R. 2013. The Bailey Point region and other muskox refugia in the Canadian Arctic: A short review. Arctic 34:34-36.

Thomas, G.H., Lanctot, R.B. and Szekely, T. 2006a. Can intrinsic factors explain population declines in North American breeding shorebirds? A comparative analysis. Animal Conservation 9:252-258.

Thomas, G.H., Lanctot, R.B. and Szekely, T. 2006b. Population declines in North American shorebirds: ecology, life-history and sexual selection. *In* Waterbirds around the world. Edited by Boere, G.C., Galbraith, C.A. and Stroud, D.A. The Stationary Office. Edinburgh, UK. pp. 207-208.

Thompson, D.W.J. and Wallace, J.M. 1998. The Arctic Oscillation signature in the wintertime geopotential height and temperature fields. Geophysical Research Letters 25:1297-1300.

Thompson, M.S., Wrona, F.J. and Prowse, T.D. 2012. Shifts in plankton, nutrient and light relationships in small tundra lakes caused by localized permafrost thaw. Arctic 65:367-376.

Thorne, E.T. 2001. Brucellosis. *In* Infectious diseases of wild mammals. Edition 3. Edited by Williams, E.S. and Barker, I.K. Iowa State University Press. Ames, IA. Chapter 22. pp. 372-395.

Thorpe, N.L. 2000. Contributions of Inuit ecological knowledge to understanding the impacts of climate change on the bathurst caribou herd in the Kitikmeot Region, Nunavut. Thesis (Master of Resource Management). Simon Fraser University, School of Resource and Environmental Management. 124 p.

Throop, J.L., Smith, S.L. and Lewkowicz, A.G. 2008. Effect of a snow fence on the shallow ground thermal regime, Baker Lake, Nunavut, Canada. *In* Proceedings of the ninth international conference on permafrost, extended abstracts. Fairbanks, AK. 29 June, 2008-3 July, 2008. Edited by Kane, D.L. and Hinkel, K.M. Institute of Northern Engineering, University of Alaska. Fairbanks, AK. pp. 313-314.

Tinner, W., Bigler, C., Gedye, S., Gregory-Eaves, I., Jones, R.T., Kaltenrieder, P., Krahenbuhl, U. and Hu, F.S. 2008. A 700-year paleoecological record of boreal ecosystem responses to climatic variation from Alaska. Ecology 89:729-743.

Tlicho Government and Department of Environment and Natural Resources, Government of the Northwest Territories. 2011. Revised joint proposal on caribou management actions in Wek'èezhìi: implementation plan. Behchoko, NT and Yellowknife, NT. 61 p. Report submitted to the Wek'èezhìi Renewable Resource Board.

Tremblay, B., Levesque, E. and Boudreau, S. 2012. Recent expansion of erect shrubs in the Low Arctic: evidence from Eastern Nunavik. Environmental Research Letters 7 035501.

Trenberth, K.E., Jones, P.D., Ambenje, P., Bojariu, R., Easterling, D., Klein Tank, A., Parker, D., Rahimzadeh, F., Renwick, J.A., Rusticucci, M., Soden, B. and Zhai, P. 2007. Observations: surface and atmospheric climate change. *In* Climate change 2007: the physical science basis. Edited by Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L. Cambridge University Press. Cambridge, UK and New York, NY. pp. 235-336.

Trottier, T. 2011. Unpublished information on population growth and development in northern Saskatchewan. Government of Saskatchewan and Beverly and Qamanirjuaq Caribou Management Board. Personal communication.

Tulp, I. and Schekkerman, H. 2006. Time allocation between feeding and incubation in uniparental Arctic-breeding shorebirds: energy reserves provide leeway in a tight schedule. Journal of Avian Biology 37:207-218.

Tulp, I. and Schekkerman, H. 2008. Has prey availability for Arctic birds advanced with climate change? Hindcasting the abundance of tundra arthropods using weather and seasonal variations. Arctic 61:48-60.

U.S. Shorebird Conservation Plan. 2004. High priority shorebirds - 2004. U.S. Fish and Wildlife Service. Arlington, VA. 5 p.

UNEP and WGMS. 2008. Global glacier changes: facts and figures. United Nations Environment Programme and World Glacier Monitoring Service. Nairobi, Kenya and Zurich, Switzerland. 88 p.

US Fish and Wildlife Service. 2012. Waterfowl population status, 2012. U.S. Department of the Interior. Washington, DC. 78 p.

Usher, P.J. 1976. Evaluating country food in the northern native economy. Arctic 29:105-120.

Usher, P.J. 2000. Traditional Ecological Knowledge in environmental assessment and management. Arctic 53:183-193.

Vallée, S. and Payette, S. 2007. Collapse of permafrost mounds along a Subarctic river over the last 100 years (northern Québec). Geomorphology 90:162-170.

Van Bogaert, R., Walker, D., Jia, G.J., Hallinger, M., De Dapper, M., Jonasson, C. and Callaghan, T.V. 2007. Recent changes in vegetation. *In* Arctic report card 2007: tracking recent environmental changes. National Oceanographic and Atmospheric Administration. pp. 32-39.

Van Coeverden de Groot, P.J. 2001. Conservation genetic implications of microsatellite variation in the muskox *Ovibos moschatus*: the effect of refugial isolation and the Arctic Ocean on genetic structure. PhD thesis. Queen's University. Kingston, ON. 198 p.

Van der Wal, R. 2006. Do herbivores cause habitat degradation or vegetation state transition? Evidence from the tundra. Oikos 114:177-186.

Van der Wal, R., Irvine, J., Stien, A., Shepherd, N. and Albon, S.D. 2000. Faecal avoidance and the risk of infection by nematodes in a natural population of reindeer. Oecologia 124:19-25.

Van Oostdam, J., Donaldson, S.G., Feeley, M., Arnold, D., Ayotte, P., Bondy, G., Chan, L., Dewaily, E., Furgal, C.M., Kuhnlein, H., Loring, E., Muckle, G., Myles, E., Receveur, O., Tracy, B., Gill, U. and Kalhok, S. 2005. Human health implications of environmental contaminants in Arctic Canada: a review. Science of the Total Environment 351:165-246.

Veitch, A.M. 2013. An aerial survey for muskoxen In the northern Sahtu Settlement Area, March 1997. Manuscript Report No. 103. Department of Environment and Natural Resources, Northwest Territories. 42 p.

Vincent, W.F., Callaghan, T.V., Dahl-Jensen, D., Johansson, M., Kovacs, K.M., Michel, C., Prowse, T., Reist, J.D. and Sharp, M. 2011. Ecological Implications of Changes in the Arctic Cryosphere. Ambio 40:87-99.

Vincent, W.F., Hobbie, J.E. and Laybourn-Parry, J. 2008. Introduction to the limnology of high-latitude lake and river ecosystems. *In* Polar lakes and rivers. Edited by Vincent, W.F. and Laybourn-Parry, J. Oxford University Press. Oxford, UK. pp. 1-23.

Vongraven, D., Aars, J., Amstrup, S., Atkinson, S., Belikov, S., Born, E.W., DeBruyn, T.D., Derocher, A.E., Durner, G., Gill, M., Lunn, N., Obbard, M.E., Omelak, J., Ovsyanikov, N., Peacock, E., Richardson, E., Sahanatien, V., Stirling, I. and Wiig, O. 2012. A circumpolar monitoring framework for polar bears. Ursus Monograph Series 5:1-66.

Vongraven, D. and Richardson, E. 2011. Diversity - Status and trends of polar bears [Arctic report card 2011] [online]. www.arctic.noaa.gov/report11/biodiv_polar_bears.html. http://www.arctic.noaa.gov/report11/biodiv_polar_bears.html (accessed 2 August, 2013).

Vonlanthen, C.M., Walker, D.A., Raynolds, M.K., Kade, A., Kuss, P., Daniels, F.J.A. and Matveyeva, N.V. 2008. Patterned-ground plant communities along a bioclimate gradient in the high Arctic, Canada. Phytocoenologia 38:23-63.

Wahren, C.H., Walker, M.D. and Bret-Harte, M.S. 2005. Vegetation responses in Alaskan arctic tundra after eight years of a summer warming and winter snow manipulation experiment. Global Change Biology 11:537-552.

Walker, D.A. 2000. Hierarchical subdivision of Arctic tundra based on vegetation response to climate, parent material and topography. Global Change Biology 6:19-34.

Walker, D.A., Bockheim, J.G., Chapin, F.S., Eugster, W., Nelson, F.E. and Ping, C.L. 2001. Calcium-rich tundra, wildlife, and the "Mammoth Steppe". Quaternary Science Reviews 20:149-163.

Walker, D.A., Epstein, H.E., Gould, W.A., Kelley, A.M., Kade, A.N., Knudson, J.A., Krantz, W.B., Michaelson, G., Peterson, R.A., Ping, C.L., Raynolds, M.K., Romanovsky, V.E. and Shur, Y. 2004. Frost-boil ecosystems: complex interactions between landforms, soils, vegetation and climate. Permafrost and Periglacial Processes 15:171-188.

Walker, D.A., Epstein, H.E., Romanovsky, V.E., Ping, C.L., Michaelson, G.J., Daanen, R.P., Shur, Y., Peterson, R.A., Krantz, W.B., Raynolds, M.K., Gould, W.A., Gonzalez, G., Nicolsky, D.J., Vonlanthen, C.M., Kade, A.N., Kuss, P., Kelley, A.M., Munger, C.A., Tamocai, C.T., Matveyeva, N.V. and Daniels, F.J.A. 2008a. Arctic patterned-ground ecosystems: A synthesis of field studies and models along a North American Arctic transect. Journal of Geophysical Research-Biogeosciences 113 G03501:1-17.

Walker, D.A., Jia, G.J., Epstein, H.E., Raynolds, M.K., Chapin III, F.S., Copass, C., Hinzman, L.D., Knudson, J.A., Maier, H.A., Michaelson, G.J., Nelson, F., Ping, C.L., Romanovsky, V.E. and Shiklomanov, N. 2003. Vegetation-soil-thaw-depth relationships along a low-arctic bioclimate gradient, Alaska: synthesis of information from the ATLAS studies. Permafrost and Periglacial Processes 14:103-123.

Walker, D.A., Raynolds, M.K., Daniels, F.J.A., Einarsson, E., Elvebakk, A., Gould, W.A., Katenin, A.E., Kholod, S.S., Markon, C.J., Melnikov, E.S., Moskalenko, N.G., Talbot, S.S. and Yurtsev, B.A. 2005. The Circumpolar Arctic vegetation map. Journal of Vegetation Science 16:267-282.

Walker, J.K.M., Egger, K.N. and Henry, G.H.R. 2008b. Long-term experimental warming alters nitrogen-cycling communities but site factors remain the primary drivers of community structure in High Arctic tundra soils. Isme Journal 2:982-995.

Walker, M.D., Wahren, C.H., Hollister, R.D., Henry, G.H.R., Ahlquist, L.E., Alatalo, J.M., Bret-Harte, M.S., Calef, M.P., Callaghan, T.V., Carroll, A.B., Epstein, H.E., Jónsdóttir, I.S., Klein, J.A., Magnússom, B., Molau, U., Oberbauer, S.F., Rewa, S.P., Robinson, C.H., Shaver, G.R., Suding, K.N., Thompson, C.C., Tolvanen, A., Totland, O., Turner, P.L., Tweedie, C.E., Webber, P.J. and Wookey, P.A. 2006. Plant community responses to experimental warming across the tundra biome. Proceedings of the National Academy of Sciences 103:1342-1346.

Walker, X., Henry, G.H.R., McLeod, K. and Hofgaard, A. 2012. Reproduction and seedling establishment of *Picea glauca* across the northernmost forest-tundra region in Canada. Global Change Biology 18:3202-3211.

Walsh, J.E. 2008. Climate of the Arctic marine environment. Ecological Applications 18:S3-S22.

Walsh, J.E., Anisimov, O., Hagen, J.O.M., Jakobsson, T., Oerlemans, J., Prowse, T.D., Romanovsky, V., Savelieva, N., Serreze, M., Shiklomanov, I. and Solomon, S. 2005. Cryosphere and hydrology. *In* Arctic climate impact assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. Chapter 6. pp. 183-242.

Walton, L.R., Cluff, H.D., Paquet, P.C. and Ramsay, M.A. 2001. Movement patterns of barren-ground wolves in the central Canadian Arctic. Journal of Mammalogy 82:867-876.

Wang, M.Y. and Overland, J.E. 2004. Detecting Arctic climate change using Köppen climate classification. Climatic Change 67:43-62.

Wang, M.Y. and Overland, J.E. 2012. A sea ice free summer Arctic within 30 years: An update from CMIP5 models. Geophysical Research Letters 39 L18501.

Watts, D.E., Butler, L.G., Dale, B.W. and Cox, R.D. 2010. The Ilnik wolf *Canis lupus* pack: use of marine mammals and offshore sea ice. Wildlife Biology 16:144-149.

Weijers, S., Wagner-Cremer, F., Sass-Klaassen, U., Broekman, R. and Rozema, J. 2013. Reconstructing high Arctic growing season intensity from shoot length growth of a dwarf shrub. Holocene 23:721-731.

Wein, E.E., Freeman, M.M.R. and Makus, J.C. 1996. Use of and preference for traditional foods among the Belcher Island Inuit. Arctic 49:256-264.

Wein, R. 1990. Frequency and characteristics of Arctic tundra fires. Arctic 29:213-222.

Weintraub, M.N. and Schimel, J.P. 2005. The seasonal dynamics of amino acids and other nutrients in Alaskan Arctic tundra soils. Biogeochemistry 73:359-380.

West Kitikmeot Slave Study Society. 2001. West Kitikmeot Slave study final report. Yellowknife, NT. 62 p.

White, D.R. 2002. Distribution and abundance of muskoxen on King William Island, Nunavut. Nunauvut Wildlife Division, Government of Nunavut. Igloolik, NU. Unpublished report.

White, R. and Trudell, J. 1980. Patterns of herbivory and nutrient intake of reindeer grazing tundra vegetation. *In* Proceedings of the Second International Reindeer/Caribou Symposium, 17-21 September 1979, Roros, Norway. pp. 180-195.

Wildlife Management Advisory Council (North Slope). 2008a. Moose (*Alces alces*) - Tuttuvak [online]. http://www.wmacns.ca/north_slope/wildlife/moose/ (accessed 12 March, 2009a).

Wildlife Management Advisory Council (North Slope). 2008b. Yukon North Slope grizzly bear population study mid-term project report. 18 p. (report date acquired from authors).

Wildlife Management Advisory Council (North Slope). 2012. Wildlife watch, April 2012 No. 20.

Wildlife Management Advisory Council (North Slope) and Aklavik Hunters and Trappers Committee. 2008. Alkavik local and traditional knowledge about grizzly bears of the Yukon North Slope: Final report. Wildlife Management Advisory Council (North Slope). Whitehorse, YT. 44 p.

Wildlife Management Advisory Council North Slope. 2008. Muskox [online]. http://www.wmacns.ca/north_slope/wildlife/muskox/ (accessed 7 July, 2013).

Wildlife Research Section. 2011. Muskoxen and Peary caribou harvest database summary report. Version 2.0. Department of Environment, Baffin Region, Government of Nunavut. Pond Inlet, NU. 23 p.

Wildllife Management Advisory Council (North Slope). 2012. Species status reports for the Yukon North Slope. 142 p.

Williams, R.S. and Ferrigno, J.G. 2002. Introduction. *In* Satellite image atlas of glaciers of the world, volume J, Glaciers of North America - Glaciers of Canada. Edited by Williams, R.S.J. and Ferrigno, J.G. pp. J1-J26.

Williams, T.M. and Fournier, B. 1996. Summary of spring classification surveys of the Bathurst Caribou Herd, 1985-1995. Manuscript Report No. 92. Department of Resources, Wildlife and Economic Development, Government of the Northwest Territories. Yellowknife NT. 50 p.

Wilson, R.R., Liebezeit, J.R. and Loya, W.M. 2013. Accounting for uncertainty in oil and gas development impacts to wildlife in Alaska. Conservation Letters 6:350-358.

Wobeser, G. 1976. Besnoitiosis in a woodland caribou. Journal of Wildlife Diseases 12:566-571.

Wolfe, S.A., Griffith, B. and Gray, C.A.G. 2000a. Response of reindeer and caribou to human activities. Polar Research 19:63-73.

Wolfe, S.A., Griffith, B. and Wolfe, C.A.G. 2000b. Response of reindeer and caribou to human activities. Polar Research 19:63-73.

Wolken, G.J., England, J.H. and Dyke, A.S. 2008. Changes in late-neoglacial perennial snow/ice extent and equilibrium-line altitudes in the Queen Elizabeth Islands, Arctic Canada. Holocene 18:615-627.

Wrona, F.J., Prowse, T.D. and Reist, J.D. 2005. Freshwater ecosystems and fisheries. *In* Arctic climate impact assessment. Edited by Symon, C., Arris, L. and Heal, B. Cambridge University Press. New York, NY. Chapter 8. pp. 353-452.

Wrona, F.J., Prowse, T.D., Reist, J.D., Hobbie, J.E., Lévesque, L.M.J. and Vincent, W.F. 2006. Climate change effects on aquatic biota, ecostructure and function. Ambio 35:359-369.

Ytrehus, B., Bretten, T., Bergsjo, B. and Isaksen, K. 2008. Fatal pneumonia epizootic in musk ox (*Ovibos moschatus*) in a period of extraordinary weather conditions. Ecohealth 5:213-223.

Zalatan, R., Gunn, A. and Henry, G.H.R. 2006. Long-term abundance patterns of barren-ground caribou using trampling scars on roots of *Picea mariana* in the Northwest Territories, Canada. Arctic, Antarctic, and Alpine Research 38:624-630.

Zamparo, J. 1996. Informing the fact: Inuit traditional knowledge contributes another perspective. Geoscience 23:263-266.

Zarnke, R.L., Dubey, J.P., Ver Hoef, J.M., Mcnay, M.E. and Kwok, O.C.H. 2001. Serologic survey for *Toxoplasma gondii* in lynx from interior Alaska. Journal of Wildlife Diseases 37:36-38.

Zhang, X., Brown, R., Vincent, L., Skinner, W., Feng, Y. and Mekis, E. 2011. Canadian climate trends, 1950-2007. Canadian Biodiversity: Ecosystem Status and Trends 2010, Technical Thematic Report No. 5. Canadian Councils of Resource Ministers. Ottawa, ON. iv + 21 p. http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-0.

Zhou, L., Tucker, C.J., Kaufmann, R.K., Slayback, D., Shabanov, N.V. and Myneni, R.B. 2001. Variations in northern vegetation activity inferred from satellite data of vegetation index during 1981 to 1999. Journal of Geophysical Research-Atmospheres 106:20,069-20,083.