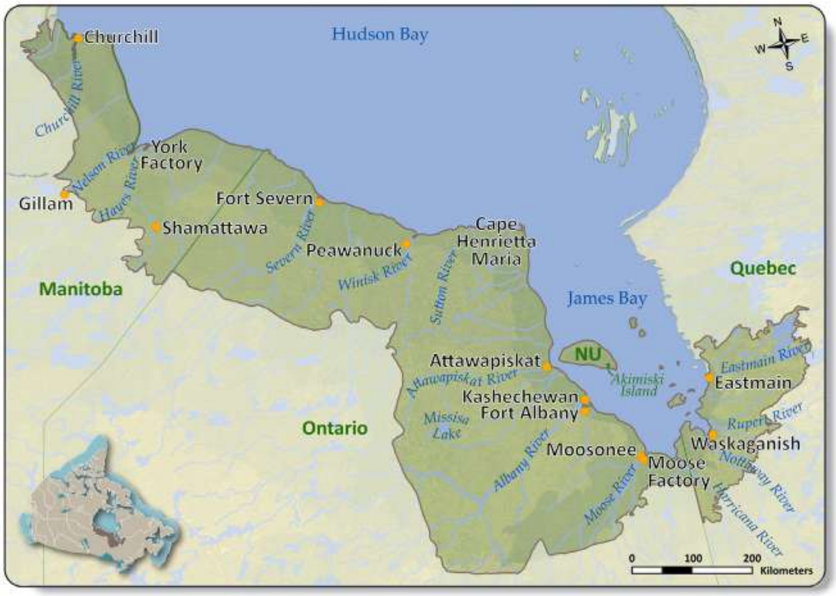
Hudson Plains Ecozone$^+$ status and trends assessment



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Ecosystem Status and Trends 2010

Technical Ecozone$^+$ Report

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PREFACE

The Canadian Councils of Resource Ministers developed a Biodiversity Outcomes Framework[*] in 2006 to focus conservation and restoration actions under the Canadian Biodiversity Strategy[†]. Canadian Biodiversity: Ecosystem Status and Trends 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).

This report is the technical report for the Hudson Plains Ecozone$^+$, which includes portions of northern Manitoba, Ontario, and Québec, as well as some islands in James Bay that are jurisdictionally part of Nunavut. A range of authors and reviewers contributed to the report from government, academia, non-governmental, and consulting sectors, with Ontario assuming the lead role in compilation under the guidance of an inter-jurisdictional steering committee and the ESTR Secretariat. For certain topics the information presented is, however, weighted somewhat more heavily towards the jurisdiction of the lead author of the topic. 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 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, including a summary report (available in both English and French) of key findings for the Hudson Plains Ecozone$^+$ (based on this technical report), is available at: http://www.biodivcanada.ca/default.asp?lang=En&n=83A35E06-1.

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[*] Environment Canada. 2006. Biodiversity Outcomes Framework for Canada. Canadian Councils of Resource Ministers, Ottawa, ON. 8 pp. Available at: http://www.biodivcanada.ca/default.asp?lang=En&n=F14D37B9-1

[†] Federal-Provincial-Territorial Biodiversity Working Group. 1995. Canadian Biodiversity Strategy: Canada's Response to the Convention on Biological Diversity. Environment Canada, Biodiversity Convention Office, Ottawa, ON. 86 pp. Available at: http://www.biodivcanada.ca/default.asp?lang=En&n=560ED58E-1

[‡] Federal, Provincial and Territorial Governments of Canada. 2010. Canadian Biodiversity: Ecosystem Status and Trends 2010. Canadian Councils of Resource Ministers, Ottawa, ON. vi + 142 pp. Available at: http://www.biodivcanada.ca/default.asp?lang=En&n=83A35E06-1

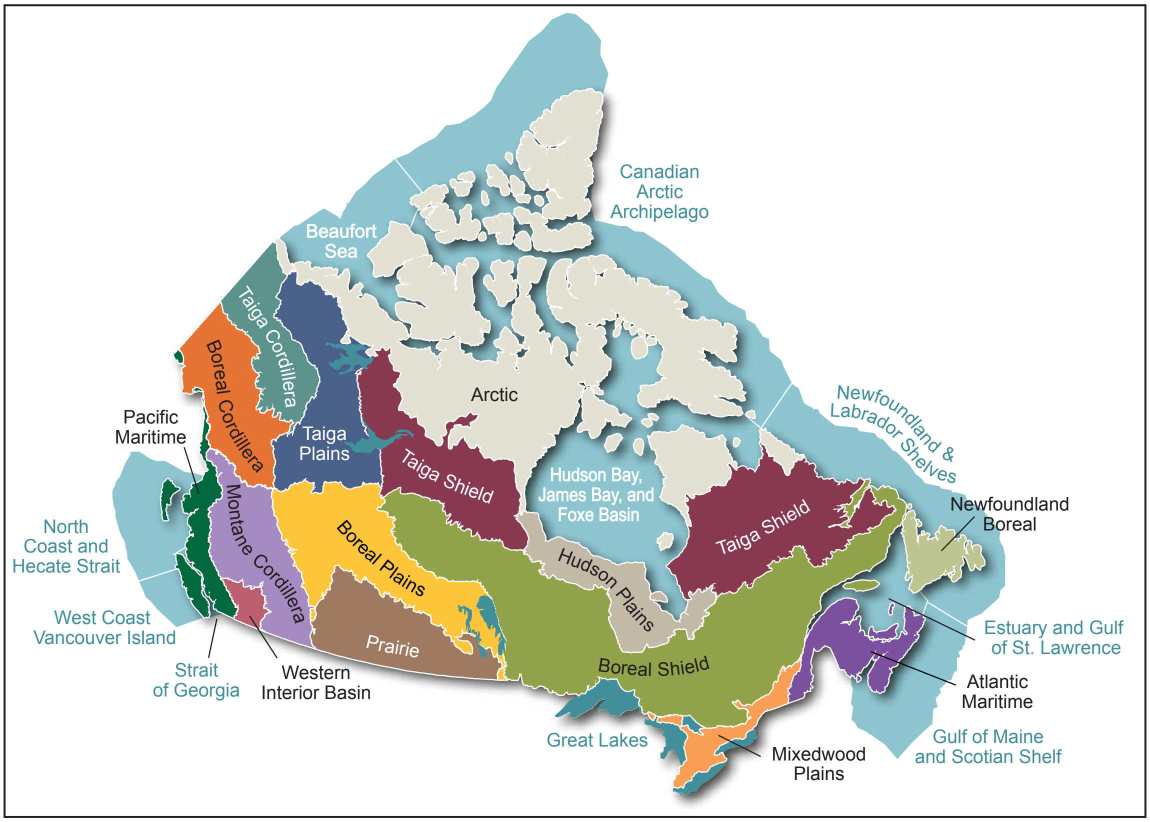
Ecological classification system – ecozones$^+$

The ecozone framework used in the ESTR project[§] is a slightly modified version of the Terrestrial Ecozones of Canada, which is described in the National Ecological Framework for Canada[**]. Modifications include (see diagrams below): adjustments to the boundaries of some terrestrial ecozones (including the Hudson Plains Ecozone) to reflect improvements from ground-truthing exercises; the combination of three arctic terrestrial ecozones into one; the use of two terrestrial ecoprovinces – Western Interior Basin and Newfoundland Boreal; the addition of nine marine ecosystem-based units; and the addition of one freshwater unit (Great Lakes). The modified classification system is referred to as *ecozones*$^+$ to avoid confusion with the more familiar *ecozones* of the original framework.

Relatively few analyses are, however, available for the Hudson Plains Ecozone$^+$ as a whole. These analyses (or re-analyses) were commissioned specifically for ESTR purposes, as the ecozones$^+$ framework was not in use in the published literature. Thus, the Hudson Plains Ecozone is generally referred to in the text of this report without a plus ($^+$) superscript. The few data analyses that apply specifically to the Hudson Plains Ecozone$^+$ (total area 352,980 km^2) are identified in relevant figure or table captions.

[§] Rankin, R., Austin, M. and Rice, J. *In Press*. Ecological Classification System for the Ecosystem Status and Trends Report. Ecosystem Status and Trends Report for Canada: Technical Thematic Report No. 1. Canadian Councils of Resource Ministers, Ottawa, ON.

[**] ESWG (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. 125 pp.



Ecozones⁺ framework used in the ESTR.

Comparison of ecozone boundaries between the ecozones⁺ framework used in the ESTR and the original National Ecological Framework for Canada.

ACKNOWLEDGEMENTS

This technical report is a contribution to the Ecosystem Status and Trends Report for Canada, which is comprised of a series of linked technical and summary reports about status and trends occurring in Canada's ecozones+ (see Preface). A joint federal-provincial-territorial project of this nature takes several years to complete. During this time, we were unfortunate to lose two contributors to this report and experts on Hudson Plains ecology: Drs. Robert (Bob) L. Jefferies (University of Toronto) and Rob Roughley (University of Manitoba). This report is dedicated to their memories.

The authors thank the many persons who supported this report with non-authored contributions. Special gratitude to all those who reviewed portions of this report (alphabetically): Peter Barnett, Yvonne Beaubien, Dean Berezanski, Warren Bernhardt, Jeff Bowman, Sam Brinker, Ryan Brook, Ross Brown, Chris Chenier, Steve Colombo, Peter Davis, Jim Duncan, Bruce Dunning, Cam Elliott, Louise Filion, Sylvie Gauthier, Cheri Gratto-Trevor, John Gunn, Jenny Harms, Colin Jones, Gordon Kayahara, Dana Kellett, Peter Kotanen, Hydro-Québec, Charles Latremouille, Frederick Leighton, Benoît Limoges, Nick Lunn, Don Macdonald, Lisa Mahon, Jim McLaughlin, John McLaughlin, Bruce Morrison, Guy Morrison, Martyn Obbard, Mike Oldham, Martin Ouellet, OMNR's remote sensing group, Cédric Paitre, Bill Parker, Richard Phillips, Irene Pines, Justina Ray, Rocky Rockwell, Nigel Roulet, Don Sutherland, John Thompson, Merritt Turetsky, Elizabeth Verghis, Larry Watkins, Richard Wilson, Xuebin Zhang, and anonymous reviewers. Much appreciation is likewise extended to those persons and organizations who contributed data, analysis, and/or graphics (alphabetically): Lise Aubry, Dean Berezanski, Sam Brinker, Rod Brook, Ryan Brook, Canadian Food Inspection Agency Centre of Expertise for Rabies, Canadian Wildlife Service, Tamara Cowie, Cree Regional Authority, Cree Trappers Association, Vince Crichton, Bill Crins, DIALOG-Aboriginal People Research and Knowledge Network, Environment Canada, Dave Etheridge, Bruce Ford, Stacy Gan, Linda Gormezano, Chris Heydon, Hudson Bay Project, Jean Iron, Laura Jackson, Andrew Jobes, Lori Martin, Manitoba Conservation, Manitoba Protected Areas Initiative, Mike Oldham, Ontario Ministry Natural Resources, Brent Patterson, Lisa Pollock, Dan Potvin, David Punter, Québec Ministère des Ressources Naturelles et de la Faune, Gilbert Racine, Rocky Rockwell, Don Sutherland, Stan Vasiliauskas, and Wapusk National Park and affiliates. Likewise, we thank the numerous other persons who provided helpful input through ESTR workshops in Toronto, Gatineau, and Québec City and/or through preparation of linked ESTR reports (see list below), as well as the various publishers who provided copyright permissions for reproducing published graphics.

Special thanks to the inter-jurisdictional ESTR Steering Committee for guidance and issues resolution (especially those representing jurisdictional interests in Hudson Plains: Bill Dalton, Mathieu Dumond, Veronika Kanya, and Benoît Limoges) and to the staff at the ESTR Secretariat for technical and graphics support, as well as their collaborative spirit, openness to discussion, stead-fast patience, and healthy sense of humour. Heartfelt thanks for the managerial support of Mary Ellen Stoll, Chris Davies, Ala Boyd, Bob Watt, David Hintz, Jane Toppozini, (Ontario Ministry of Natural Resources), Sheldon Kowalchuk (Wapusk National Park), and Jack Dubois and Jim Duncan (Manitoba Conservation), given to the lead authors.

Funding for report compilation and publication was provided by the Ontario Ministry of Natural Resources' Biodiversity Branch, Far North Branch, Science and Information Branch, and

Applied Research and Development Branch. In-kind contributions from Environment Canada and the affiliations of all contributing authors is likewise gratefully acknowledged. This report has been produced under the umbrella of the Canadian Councils of Resource Ministers by the Ontario Ministry of Natural Resources (Biodiversity Branch) in cooperation with the ESTR Secretariat. Special thanks to Carmen Misasi and Erika Luoma for publication services.

Sincere apologies to any one that may have been overlooked in these acknowledgements.

Authors of ESTR Thematic (National) Technical Reports from which information is drawn and/or cited (http://www.biodivcanada.ca/default.asp?lang=En&n=137E1147-1):

Technical Thematic Report No. 1. Ecological classification system for the ecosystem status and trends report: R. Rankin, M. Austin and J. Rice

Technical Thematic Report No. 2. Guidance for the preparation of ESTR products – classifying threats to biodiversity: C. Wong

Technical Thematic Report No. 3. Guidance for the preparation of ESTR products – land classification scheme: J. Frisk

Technical Thematic Report No. 4. Large-scale climate oscillations influencing Canada, 1900-2008: B. Bonsal and A. Shabbar

Technical Thematic Report No. 5. Canadian climate trends, 1950-2007: X. Zhang, R. Brown, L. Vincent, W. Skinner, Y. Feng and E. Mekis

Technical Thematic Report No. 6. Trends in large fires in Canada, 1959-2007: C.C. Krezek-Hanes, F. Ahern, A. Cantin and M.D. Flannigan

Technical Thematic Report No. 7. Wildlife pathogens and diseases in Canada: F.A. Leighton

Technical Thematic Report No. 9. Trends in permafrost conditions and ecology in northern Canada: S. Smith

Technical Thematic Report No. 10. Northern caribou population trends in Canada: A. Gunn, D. Russell and J. Eamer

Technical Thematic Report No. 11. Woodland caribou, Boreal Population, trends in Canada: C. Callaghan, S. Virc and J. Duffe

Technical Thematic Report No. 13. Trends in Canadian shorebirds: C. Gratto-Trevor, R.I.G. Morrison, B. Collins, J. Rausch, M. Drever and V. Johnston

Technical Thematic Report No. 16. Soil erosion on cropland: introduction and trends for Canada: B.G. McConkey, D.A. Lobb, S. Li, J.M.W. Black and P.M. Krug

Technical Thematic Report No. 17. Monitoring ecosystems remotely: a selection of trends measured from satellite observations of Canada: F. Ahern, J. Frisk, R. Latifovic and D. Pouliot

Technical Thematic Report No. 20. Biodiversity in Canadian lakes and rivers: W.A. Monk, D.J. Baird, R.A. Curry, N. Glozier and D.L. Peters

Other available ESTR Thematic (National) Technical Reports that are not cited here:

Technical Thematic Report No. 8. Trends in breeding waterfowl in Canada: M. Fast, B. Collins and M. Gendron

Technical Thematic Report No. 12. Landbird trends in Canada, 1968-2006: C. Downes, P. Blancher and B. Collins

Technical Thematic Report No. 14. Trends in wildlife habitat capacity on agricultural land in Canada, 1986-2006: S.K. Javorek and M.C. Grant

Technical Thematic Report No. 15. Trends in residual soil nitrogen for agricultural land in Canada, 1981-2006: C.F. Drury, J.Y. Yang and R. De Jong

Technical Thematic Report No. 18. Inland colonial waterbird and marsh bird trends for Canada: D.V.C. Weseloh

Technical Thematic Report No. 19. Climate-driven trends in Canadian streamflow, 1961-2003: A. Cannon, T. Lai and P. Whitfield

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1.0 OVERVIEW OF THE HUDSON PLAINS ECOZONE⁺ – PRE-HISTORY & HISTORIC CONTEXT

1.1 Geology, topography & climate

Lynda A. Dredge, Geological Survey of Canada

The Hudson Plains Ecozone⁺ [1,2] constitutes a 150-300 km wide swath of land that rises from the tidal flats surrounding Hudson and James bays (Figure 1). It also includes Akimiski Island and other small islands that lie within James Bay. This ecozone reaches a maximum elevation of 130 m near Nelson River and 240 m east of James Bay; it corresponds closely to the area covered by postglacial seas (Figure 2). Relief is generally low and, for the most part, the land rises gently inland from Hudson Bay at a rate of about 0.5 m/km. The plains are interrupted by incised valleys along major rivers, by a low bedrock ridge at Churchill, Manitoba, and by the Sutton Ridges, a 50 km long southwest-facing cuesta that protrudes 120 m above the surrounding plains (Figure 3). The Sutton Ridges lie 150 km southwest of Cape Henrietta Maria, Ontario.

The area has a maritime boreal climate, influenced significantly by Hudson Bay. Mean annual air temperatures vary from –7°C at Churchill to –1°C at Moosonee, and precipitation ranges from 430 mm to 680 mm correspondingly (Environment Canada 2010). The seasonal presence of sea ice in Hudson and James bays has an important affect on climate and contributes to the occurrence in the Hudson Plains Ecozone of the most southern continuous permafrost in North America (Gough and Leung 2002; Zhang et al. 2008). Permafrost is continuous around Churchill, where it reaches a thickness of 60 m, and along the Hudson Bay coast to Cape Henrietta Maria. It is discontinuous to the south and inland, and it becomes isolated in patches in the area inland from James Bay in the south (see also Figure 23, in Section 2.1, *Abiotic Drivers*).

The bedrock underlying most of the region is Paleozoic limestone and dolomite, although metasedimentary greywacke rocks of Precambrian age form the coastline at Churchill, and Precambrian granite, cherty dolomite, greywacke, iron formation, and gabbro form the Sutton Ridges of northern Ontario (Bostock 1971). Most of the region is covered by a sequence of glacial and postglacial deposits that is up to 80 m thick. Sediments in river bluffs reveal a long record of multiple glaciations and at least two interglaciations when climates were warmer than present, and sea level was somewhat higher (Dredge and Cowan 1989). During the last ice age, the area was affected by two ice flow centres within the Laurentide Ice Sheet: one in Keewatin, which

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[1] Relatively few analyses are available for the Hudson Plains Ecozone⁺ (newly adjusted ecozone boundaries) as a whole (see the Preface for an explanation of the ecozones⁺ framework). These analyses (or re-analyses) were commissioned specifically for ESTR purposes, given that the ecozones⁺ framework was not in use in the published literature. Thus, for simplicity, throughout this report the Hudson Plains Ecozone is generally referred to in the text without the ⁺ superscript. The limited data analyses specific to the Hudson Plains Ecozone⁺ as a whole are identified in relevant figure or table captions.

[2] Much of the literature uses the pre-existing term *Hudson Bay Lowland* to describe the physiographic region classified here as the Hudson Plains Ecozone. Unless otherwise noted, attributions to features or resources of the Hudson Bay Lowland by earlier authors are treated as attributions to the Hudson Plains Ecozone.

Figure 1. Overview map of the Hudson Plains Ecozone, as defined by the ecozones$^+$ framework. The Hudson Plains Ecozone$^+$ extends from 50° N to 59° N and occupies an area of 352,980 km^2 across portions of northern Manitoba (21%), Ontario (68%), and Québec (10%), as well as some islands in James Bay that are jurisdictionally part of Nunavut (1%).

maintained a southward ice flow near Churchill, and the other in Québec-Labrador. Westward and southwestward flow from the Labradorean ice sheet was responsible for the calcareous silty deposits of till that form the uppermost glacial deposits across most of this ecozone south of Churchill. During deglaciation, glacial lakes Agassiz and Ojibway abutted the northward-retreating ice margin, and there were several glacial re-advances of portions of the ice front into these glacial lakes. Most glacial lake deposits within this ecozone were later covered by postglacial marine sediments, but some of the re-advance deposits of clayey drumlinized till protrude above the marine sediments on relatively high terrain.

Following deglaciation about 8,000 years ago, when remnants of the Laurentide Ice Sheet broke-up in Hudson Bay, marine waters flooded the Hudson Plains Ecozone in areas where the earth's crust had been depressed below sea level by the weight of the ice. As a result, the surface sediments over much of the ecozone consist of fine-grained calcareous deposits from the marine transgression, overlain by thin blankets of marine sand or thicker sandy deposits forming beach ridges (Fulton 1995 and Figure 2). Major flights of closely spaced beach ridges occupy coastal terrain between the Hayes and Winisk rivers, where the regional land gradient is greater than for other parts of the lowlands. Despite rising global sea levels, the amount of crustal rebound has exceeded sea level rise, so that since deglaciation, sea water has been spilling off the land.

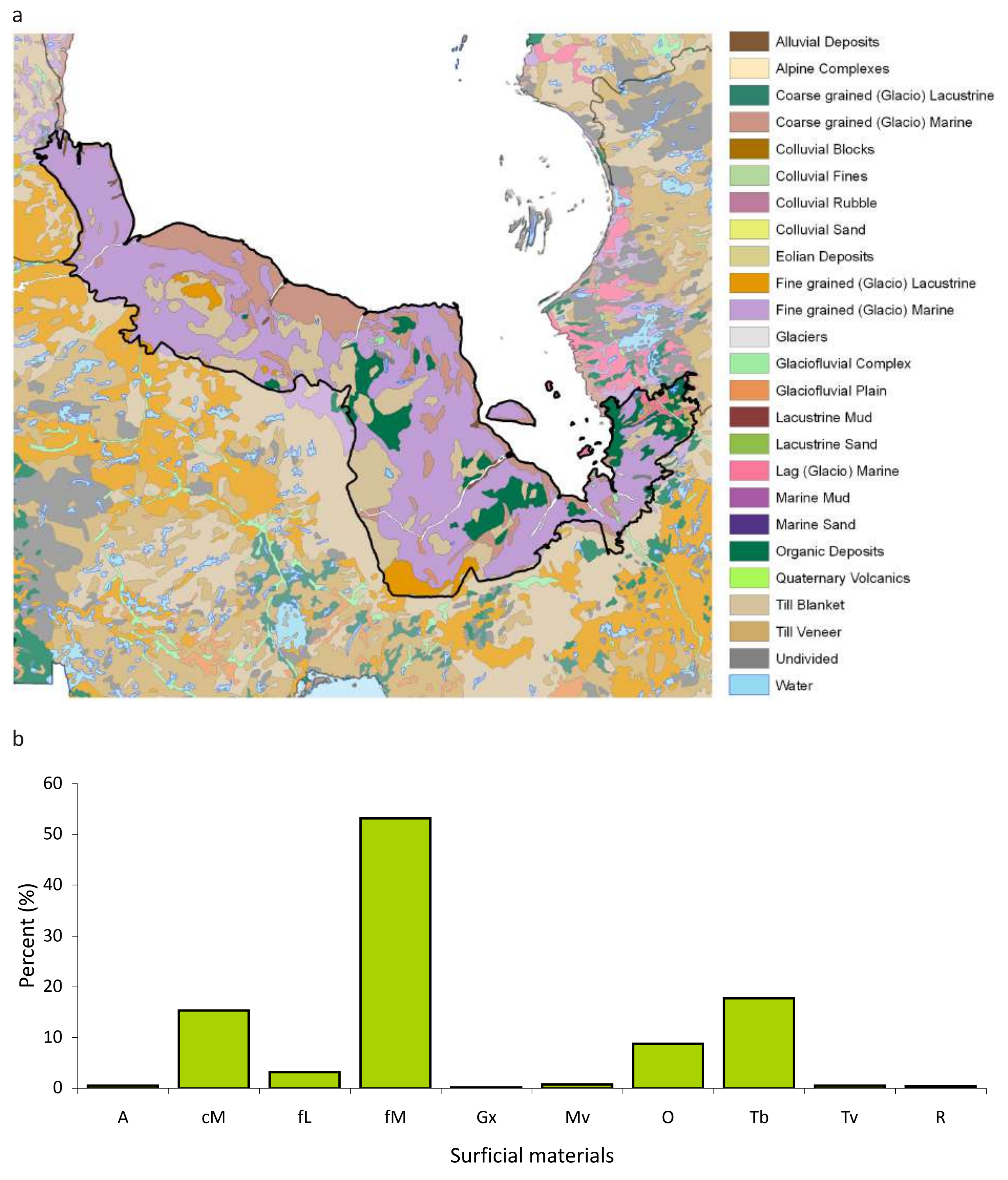

Figure 2. Surficial geology of the Hudson Plains Ecozone: a) spatial distribution of surficial materials and b) composition of surficial materials. Abbreviations: A, Alluvial Deposits; cM, Coarse grained (Glacio)Marine; fL, Fine grained (Glacio)Lacustrine; fM, Fine grained (Glacio)Marine; Gx, Glaciofluvial Complex; Mv, Lag (Glacio)Marine; O, Organic Deposits; Tb, Till Blanket; Tv, Till Veneer; and R, Undivided. This analysis is based on ecozone⁺ boundaries.

Source: Modified from Fulton (1995).

Figure 3. The Sutton Ridges, southwest of Cape Henrietta Maria, Ontario. Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

The land is rebounding at a present rate of between 0.90 m/century at Churchill and 1.1 m/century at Peawanuck (Sella et al. 2007), so new land is still being created along the coast, and the tidal flats are prograding into Hudson Bay.

Due to the low relief, impermeable clayey soils, and ground ice in permafrost, surface drainage is generally poor; as a result, bogs and fens cover extensive areas, making the Hudson Plains Ecozone the wettest ecozone in Canada (NWWG 1988) and the third largest wetland complex in the world (Fraser and Keddy 2005)[3]. Peat has accumulated on the geological substrate. It is thinnest in recently emerged areas along the coast, and it thickens inland to a depth of 3 to 5 metres (Dredge and Mott 2003).

The Hudson Plains Ecozone contains elements of arctic, boreal, and temperate biomes[4]. Persistence of the arctic elements of ecosystem diversity are made possible by the influence of Hudson Bay ice on thermal regimes and climate. Along the Hudson Bay coast from Churchill to Cape Henrietta Maria, in the area of continuous permafrost, tundra heath vegetation is common on frozen peat. Most local relief in this biome is created by large ice wedge polygons and

---

[3] Definitions of wetland vary. A high proportion of area considered to be wetlands in much of the published literature is instead classified in the ESTR framework (Frisk in press) as low-density forest (for further discussion, see Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*).

[4] The term *boreal* is commonly used when describing the range of species affinities in the Hudson Plains Ecozone (e.g., Crins et al. 2009), but it is not a favoured term in all disciplines. For example, the ecozone's vascular flora has been classified as being comprised of species with arctic, subarctic, and temperate geographic affinities (Riley 2003). In this case, plant species commonly referred to as being of *boreal* affinity would mostly straddle Riley's (2003) subarctic (taiga) and temperate (forested boreal) vascular plant classes, depending on their distribution (J. Riley, The Nature Conservancy of Canada, pers. comm.).

shallow thermokarst ponds in the peat (Dredge and Nixon 1992). Farther south and inland, in areas of discontinuous permafrost, boreal forests (most growing on peat), raised peat bogs, and fens containing palsa mounds dominate the landscape. Southernmost areas around James Bay have extensive fenlands but support boreal forest on peat islands or mineral substrate.

The ecozone includes parts of several major river drainage areas (Figure 4). The Nelson River, for instance, has its origins at the continental divide in the Rocky Mountains. Three major rivers – the Nelson, Abitibi (a tributary of the Moose River), and Eastmain – are affected by dams and diversions within the ecozone (Monk et al. in press; see also Section 2.2.2.4.2, *Rivers/Streams & Lakes*). As well, the Churchill, Nelson, Moose, Albany, Rupert, Nottaway, and Eastmain rivers (or tributaries) are affected by hydroelectric developments upstream, outside the ecozone. The remaining major rivers (Hayes, Severn, Winisk, Attawapiskat, Harricana, and Broadback) are unimpeded.

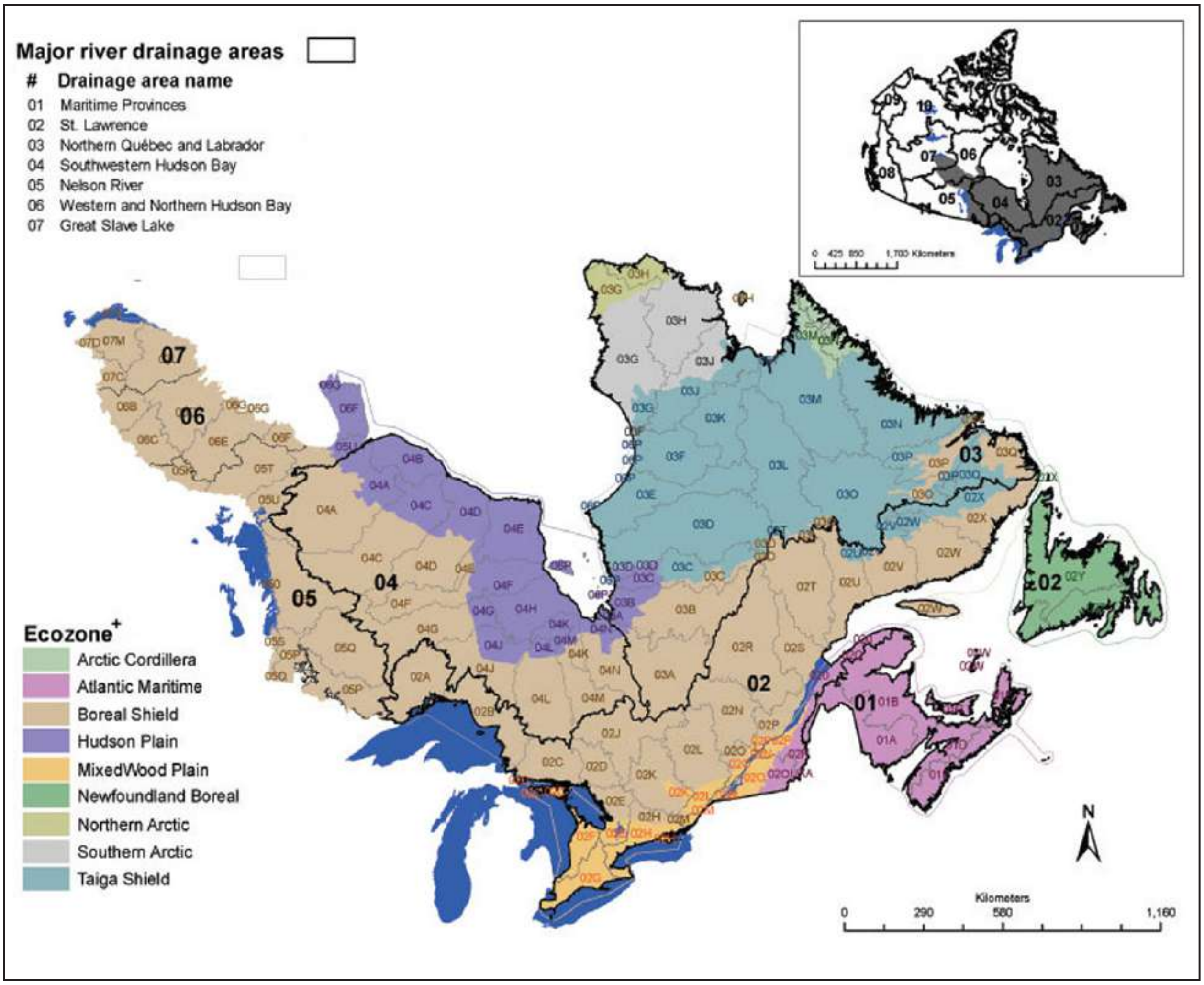

Figure 4. Major river drainage areas and sub-drainage areas in and around the Hudson Plains Ecozone. All ecozone boundaries in this image correspond to the ecozones+ framework.
Source: Walker (2008).

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1.2 **Human history**

Susan M. Tully, Ontario Ministry of Natural Resources

1.2.1 **Settlement history**

Prehistoric occupation of the Hudson Plains Ecozone began approximately 4,000 years ago (Stewart and Lockhart 2005). The earliest estimated date for occupation is 3,920 ± 180 years (Lister 1988). Artefacts also confirm that Aboriginal peoples lived there for over 1,000 years prior to European settlement (Stewart and Lockhart 2005).

Lowland Cree (also referred to as Muskekowuck Athinuwick, Omushkego, Swampy Cree, or West Main Cree) historically occupied Hudson Plains Ecozone and were among the first Aboriginal peoples to come into contact with European explorers, missionaries, and fur traders (Lytwyn 2002; Stewart and Lockhart 2005). Their population was not numerous. Early estimates (1700s) of the total Lowland Cree population were around 1,500-2,100. A small pox epidemic occurred throughout the area in 1782-1783, leading to high rates of mortality, probably 50% or more in some areas; their numbers recovered by the 1820s (Lytwyn 2002).

After an initial period of exploration for the Northwest Passage (1610-1632), contact with European fur traders and the establishment of trading post settlements along the coast marked the beginning of continuous interaction between Europeans and the Lowland Cree (Stewart and Lockhart 2005). From 1713 to 1782 (the latter year being when the small pox epidemic occurred), significant fur trade and associated provision trade occurred between the Lowland Cree and coastal trading posts. A network of inland trading posts rapidly expanded, such that few upland traders (from outside the lowlands) visited the coastal trading posts after 1782 (Lytwyn 2002). However, redundant trading posts in the interior later closed following the depletion of populations of American beaver (*Castor canadensis*) and other furbearers, which occurred over the period 1783-1821. Throughout the more modern settlement period (1903 to present), seasonal camps were gradually abandoned, and a sedentary lifestyle around the missions and trading posts was adopted. By the 1960s, most Aboriginal peoples lived in coastal settlements with schools, healthcare facilities, and some opportunities for wage employment (Stewart and Lockhart 2005).

Census data from Statistics Canada suggest that the ecozone's human population was relatively stable overall from 1971 to the latest census year of 2006 (Environment Canada 2009), albeit federal censuses missed some communities in some years (individual community sizes are also changing). Today, the Hudson Plains Ecozone is home to an estimated 14,000 residents, concentrated in 11 villages (Table 1) at a density of about one person per 24 km^2 (based on a total ecozone$^+$ area of 352,980 km^2). Most villages are coastal communities located near the mouths of major rivers (estuaries), and residents are primarily of Aboriginal descent, principally Cree[5] and Métis (Abraham and Keddy 2005). All of the ecozone's communities are equipped with amenities, such as general stores, nursing stations, and community roads (Stewart and Lockhart 2005). The traditional, subsistence way of life, although declining in some areas, remains socially and culturally important to the ecozone's Aboriginal peoples (Section 2.5.4, *Cultural Services*).

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[5] Includes Lowland Cree peoples (principally in Manitoba and Ontario) and Eastern Cree peoples (in Québec). The historical boundary between them is approximately the Nottaway River (Lytwyn 2002) or Harricana River (Stewart and Lockhart 2005).

Associated land use still occurs, with heavy use of the coast and waterways and to a lesser extent the interior (Berkes et al. 1995). Churchill and Moosonee are the only communities with non-native governance structures (Statistics Canada 2010).

Table 1. Hudson Plains Ecozone communities and populations in the federal census years of 2001 and 2006. The list of communities does not differ between newer (ecozone⁺) and older ecozone boundaries. Abbreviation: N/A, not applicable.
Sources: Statistics are from Statistics Canada (2010), unless otherwise noted (footnotes).

Province	Community[a]	Established	Census		Unemployment rate (%) in 2006
			2001	2006	
Manitoba	Churchill (town)	1929	963	923	14.5
	Churchill (reserve)	unknown	316	330	29.2
	Shamattawa	unknown	897	920	28.3
Ontario	Fort Severn	1689[b]	401	400[b]	N/A
	Peawanuck	1986[b]	193	221	14.3
	Attawapiskat	1929[b]	1,293	1,300[b]	N/A
	Fort Albany	1950 at present site, 1679 at original site[b]	441[c]	1,805	N/A
	Kashechewan	1950[b]	N/A	1,561[b]	N/A
	Mooosonee	1903[b]	1,916	2,006	N/A
	Moose Factory	1673[b]	N/A	2,700[b]	N/A
Québec	Waskaganish	1668	1,699	1,864	22.6
	Eastmain	1723	613	650	12.9

[a] Gillam, Manitoba (shown in Figure 1, Section 1.1, *Geology, Topography & Climate*) is not listed here, because it is outside ecozone⁺ boundaries.

[b] Corston and McComb (2008). Statistic not available from Statistics Canada (2010).

[c] Statistics Canada (2010) issued a data quality index for this census data, indicating a global non-response rate equal to or higher than 25%. This could help explain the difference in Fort Albany's population between 2001 and 2006.

1.2.2 **Economic history**

Economic development in the Hudson Plains Ecozone has been limited. Historically, the main driver of the economy was the fur trade, with transportation, hydroelectricity, and tourism more recent developments and mining increasing in importance.

Prior to the arrival of European fur traders, the Lowland Cree probably traded extensively with their upland neighbours, who occupied the upland shield region south and/or west of the Lowland Cree territory (Lytwyn 2002; Stewart and Lockhart 2005). The Lowland Cree participated directly in organized fur trade from the early 1700s. The role of Lowland Cree in supplying food (mainly waterfowl and caribou) to European trade settlements in exchange for European goods was slow to develop, but it became significant by the 1750s (Lytwyn 2002). With the establishment of inland trading posts, the participation of Lowland Cree in transport-related activities increased. After the small pox epidemic (1782-1783), involvement of the Lowland Cree in the fur trade was significantly impacted, resulting in a reduction of fur returns from these peoples for the next several years. The overall fur trade in the broader northern region peaked in activity from 1790 to 1810. European traders encouraged Lowland Cree and other subarctic peoples to harvest as many furbearing species as possible. By 1805, the stress on the fur resources was evident, and populations such as American beaver (*Castor canadensis*) (furbearer) as well as woodland caribou (*Rangifer tarandus caribou*) were in decline. Declines in these animal populations caused significant population movements of the Lowland Cree within and outside their region from 1783 to 1821 (Lytwyn 2002). Wildlife harvest has remained important in the mixed economies of the communities in this ecozone, withstanding the implementation of a modern wage economy (Berkes et al. 1994, 1995).

Commercial fishing has not been a large part of the economic history of the Hudson Plains Ecozone. Churchill was the focal point for the Hudson Bay Company's whaling activities in western Hudson Bay, which lasted until ~1968 (Lytwyn 2002). Commercial harvest of fish was important in the early 1900s (Lower 1915), and it has since varied with species and over time (e.g., Hutton and Black 1975). Historically in Ontario, regulated commercial fisheries for walleye (*Sander vitreus*) occurred in Missisa Lake and for lake sturgeon (*Acipenser fulvescens)* in the Moose River Basin (Thompson 1989). No regulated commercial sturgeon fishery exists in this area today (Seyler 1997; OMNR 2008, 2009). Likewise, regulated commercial fishing for lake sturgeon and other species apparently no longer occurs in the Manitoba (W. Bernhart, North/South Consultants Inc., pers. comm.) or Québec (C. Paitre, Environment Canada-Québec Region, pers. comm.) portions of the ecozone.

The Hudson Plains Ecozone accommodates rail, sea, ground, and air transportation. The two railways in the ecozone terminate respectively at the coastal communities of Churchill (completed 1929) and Moosonee (completed 1931) (Abraham and Keddy 2005; Stewart and Lockhart 2005). These railways are used primarily for transportation of goods and for access to the relatively roadless region (Stewart and Lockhart 2005). The Churchill harbour, which is the region's only deepwater port (and one of only three in the marine arctic), has been used since 1931 for the distribution of abundant grain products originating in the Canadian Prairies. The harbour is still used today to load prairie grain; to supply communities with food, dry goods, and fuel; and for occasional luxury liners (Lytwyn 2002). A saltwater port also exists at Moosonee, although it is not a deepwater port (Stewart and Lockhart 2005). Until relatively recently, this ecozone has been essentially roadless, with only winter roads connecting the coastal communities (Abraham and Keddy 2005; Stewart and Lockhart 2005; OMEI and OMNDMF 2009). An all-season road now connects the communities of Eastmain (completed 1995) and Waskaganish (completed 2001) in coastal Québec with the south (Hydro-Québec 2003; Stewart and Lockhart 2005), and additional permanent roads are likely in the future. Feasibility planning is in progress for an all-season road that would run along the western edge of the ecozone, from Gillam to Churchill, Manitoba and beyond to Rankin Inlet, Nunavut (Government of Nunavut and Government of Manitoba 2010; SNC Lavalin 2010), and a pre-feasibility study is in progress to assess possible routes for an all-season road that would connect coastal communities along James Bay in Ontario with the provincial highway system in the south (Government of Ontario 2009). All communities in the ecozone are accessible by air for transportation of both people and goods (Stewart and Lockhart 2005; OMEI and OMNDMF 2009).

Hydroelectric developments affecting areas of the Hudson Plains Ecozone were established beginning in the early 1900s (Abraham and Keddy 2005). However, the first hydroelectric facility located directly within ecozone boundaries was not established until 1961 (Otter Rapids generating station on the Abitibi River), with hydroelectric development in the ecozone peaking in the late 1970s to early 1980s but continuing today (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). These developments have significantly impacted the inhabitants of this area. Construction of power-generating stations produced improvements for some communities (Cragg and Schwartz 1996). Hydroelectric developments are, however, also seen as threats to habitat, natural resources, and the land-based economy that is a cornerstone of this area (e.g., Niezen 1993; McDonald et al. 1996; see also Section 2.6.1.2, *Cumulative Impacts*). All three provinces are considering or establishing new hydroelectric developments in the Hudson Plains Ecozone (OPA 2007; Hydro-Québec 2010; Manitoba Hydro 2010; MDDEP 2010). Potential also exists for wind energy farms to be established along the coasts of Hudson and James bays (Environment Canada 2008; OMNR 2010; see also Hélimax Énergie and AWS Truewind, LLC 2005).

Although relatively little resource extraction is occurring in the Hudson Plains Ecozone, mining and, to a lesser extent, forestry do exist, and have potential to increase in the future. One mine (an open-pit diamond mine) is currently operating in the Ontario portion of the ecozone, but a high potential exists there for more (OMEI and OMNDMF 2009; Golder Associates 2010; Micon International 2010). Construction began on the existing Victor mine near Attawapiskat in 2006, and it opened in 2008, providing a limited amount of new employment in the area (DeBeers Canada 2008). No mines are currently operating in the Manitoba or Québec portions of the ecozone. However, exploration has been ongoing in Manitoba (Manitoba Geological Survey 2003), and several former mining exploration sites occur in the Québec portion of the ecozone (CRA 2008), with one active gold mine (Casa Berardi Centre) located outside the southern ecozone boundary (MRNF 2010). Forestry has been limited in the Hudson Plains Ecozone, with most harvesting activity concentrated around select communities and to a greater degree around better-drained river levees (OMNR 1985). Timber harvest has typically been limited to fuel wood purposes, with no associated management plan (Berkes et al. 1994). Planning for potential commercial forestry has, however, been undertaken for the Moose Factory area by the Moose Cree First Nation (Forestry Futures Trust Committee 2007, 2009). Peat harvesting has not contributed to the economic development of this area, which has been considered too wet and too far from commercial centres to be suitable for peat exploitation (Martini 2006). Agriculture has not played an economic role in the Hudson Plains Ecozone in the past nor is it predicted to in the future (Statistics Canada 2009; McConkey et al. in press).

Tourism has been part of the economy of the Hudson Plains Ecozone for several decades. Most tourism activities take place in Churchill, where people come to see migratory birds, beluga (*Delphinapterus leucas*), and polar bear (*Ursus maritimus*). In Ontario, Moosonee and Moose Factory benefit economically from tourism associated with the Polar Bear Express passenger train that connects these coastal communities to the southern portion of the province. Tourism is also one of the main drivers of the local economy in the Québec portion of the ecozone (Abraham and Keddy 2005). The ecozone also supports recreational fishing and hunting, and a number of associated camps are or have been operated there by Aboriginal peoples (e.g., Wheeler 1985; McKnight and Hendry 1988; Scholten and Byers 1990; Scholten and Thompson 1992). With the exception of Wapusk National Park in Manitoba and Tidewater Provincial Park in Ontario, the ecozone's protected areas are not well used (e.g., Statistics Canada 1996), presumably owing to access and service constraints and the availability of wilderness areas outside of protected areas.

Overall, the Hudson Plains Ecozone now supports mixed economies (i.e., subsistence lifestyle; government transfer payments, grants, and programs; and wage employment). The wage economy is still relatively limited, with high unemployment rates (refer back to Table 1). Education, three levels of government, transportation, health, and the service sector are the primary employers. Many of the ecozone's Aboriginal peoples still depend largely on traditional wildlife and hunter-gatherer (subsistence) economies (Berkes et al. 1994, 1995; George et al. 1996; Fast and Berkes 1998; Abraham and Keddy 2005).

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

2.1 Abiotic drivers

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With inset (permafrost) by
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The dominant abiotic driver in the Hudson Plains Ecozone and surrounding region is the presence of the Hudson and James bays, which can be regarded as extensions of the Arctic Ocean. Hudson and James bays account for a plethora of direct and indirect effects on the ecozone's local climate, hydrology, and soil processes, including the maintenance of permafrost and the depth of the active layer at this extreme southerly latitude. As a result, the biological diversity associated with the Hudson Plains Ecozone is more characteristic of a northern cold climate than that expected at comparable latitudes elsewhere. Other important local drivers that affect landscape processes and diversity include the presence of numerous small freshwater lakes and ponds, permafrost, and isostatic rebound. From a historical perspective, however, these drivers are also the outcome of global climate change acting at the regional or local levels.

2.1.1 Present climate & trends to date

A number of important features characterize the general climate of the region. These features include: strong seasonality in incoming solar radiation; considerable local variations in climate brought about by the juxtaposition of different surface features in both marine and terrestrial environments (open sea, islands, sea ice, proximity to mainland coasts); complex inter-annual and decadal variations in climate; and polar amplification of climate change resulting from changes in albedo and feedback processes (Walsh 2008).

In summer, storms track across Hudson Bay along the elongated latitudinal gradients of low pressure associated with the Aleutian and Icelandic lows, where the pressure is less than about 96.0 kPa (Walsh 2008). In winter, the Arctic Ocean is dominated by a ridge of high pressure between the cold Asian air mass and that over North America, whereas the subarctic seas are under the influence of the subpolar lows. Generally, cloud prevails in the arctic marine environment, which affects solar and long-wave radiation at the surface. Spring is marked by an abrupt increase in cloud cover associated with the melting of sea ice, and a less abrupt decrease occurs in autumn associated with freeze-up. In the adjacent terrestrial Hudson Plains Ecozone these changes are less marked, but they are still evident. Stratiform clouds prevail in winter, but in summer mixed-stratiform convective clouds are more common (Walsh 2008). Annual global solar radiation decreases from about 140 W/m^2 to 125 W/m^2 between the southern and northern boundaries of the ecozone (Rouse et al. 1997), but annual net radiation decreases from 40 W/m^2 to 25 W/m^2 due to the persistence of snow cover at northern locations.

Long-term temperature data from the Hudson Plains Ecozone exist for only two stations, Churchill, Manitoba (58°44′N, 94°04′W, 29.3 m) and Moosonee, Ontario (51°16′N, 80°39′W, 10.0 m), which represent coastal conditions at the extreme northwest and southeast ends of the ecozone, respectively (Environment Canada 2010a). Limited additional weather data are available from other coastal sites in the ecozone (e.g., Winisk and Fort Albany in Ontario), including long-term precipitation data for a station at Eastmain, Québec (52°14′N, 78°31′W, 7.3 m). The data from this relatively sparse network of stations have been analyzed and reported on for different periods by various authors (e.g., Rouse 1991; Riley 2003).

Long-term data from the Churchill, Moosonee, and sometimes also Eastmain stations (Figure 5) were sufficient to support the 1950-2007 climate trends analysis for the ESTR (referenced below)[6]. The ecozone-level results from this analysis, however, represent average conditions that have limited interpretive value for the ecozone as a whole. High annual variability in the data record produced by this averaging method might also adversely influence the probability of detecting significant trends. Local modeling within this ecozone can be more informative and useful. Some of it has produced different results, and we note also that climatic trends are highly sensitive to the time period chosen to report (e.g., K. Middel, Ontario Ministry of Natural Resources, unpublished analysis of Moosonee climate data for the period 1930-2003). Thus, although climate trends are reported below for the ecozone as a whole, data from individual stations are also scrutinized for possible sub-ecozone trends. Climate trends for areas of the ecozone inland from the coast remain a knowledge gap, although spatial interpolation has been used to estimate temperature and precipitation conditions in inland areas for the period 1961-1990, albeit with high uncertainty (Environment Canada 2010b).

Average annual air temperatures in the ecozone vary from -1.1 °C at southern locations to -6.9 °C at northern sites, based on 30 year (1971-2000) climate normals (Environment Canada 2010a; see also Riley 2003). In January, daily average temperatures range from -20.7 °C at Moosonee (daily average minimum -27.0 °C, daily average maximum -14.2 °C) to -26.7 °C at Churchill (daily average minimum -30.7 °C, daily average maximum -22.7 °C). In July, daily average temperatures range from 15.4 °C at Moosonee (daily average minimum 8.5 °C, daily average maximum 22.2 °C) to 12.0 °C at Churchill (daily average minimum 6.8 °C, daily average maximum 17.3 °C) (Environment Canada 2010a).

At the ecozone scale, trend analysis indicates a significant 1.9 °C increase in mean summer air temperature (Figure 6b) but no significant change in mean spring, fall, or winter temperatures (Figure 6a,c,d, respectively). Partitioning out station-specific results shows similar trends in summer temperature at individual stations but also a significant 2.2 °C increase in mean winter temperature at Churchill (Figure 7). While neither Churchill nor Moosonee stations show significant increases in mean spring temperatures from this analysis (Figure 7a), increased spring temperatures in the broader region were reported by Gagnon and Gough (2002). K.

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[6] The climate trends analysis for the ESTR was done similarly for all ecozones$^+$ for the period 1950-2007, which provided the best possible spatial coverage across Canada for the longest period possible (relatively few long-term weather data exist for northern Canada prior to 1950). The homogenized data set was corrected for known sources of systematic errors, such as station shifts and changes in observing procedures (Zhang et al. in press).

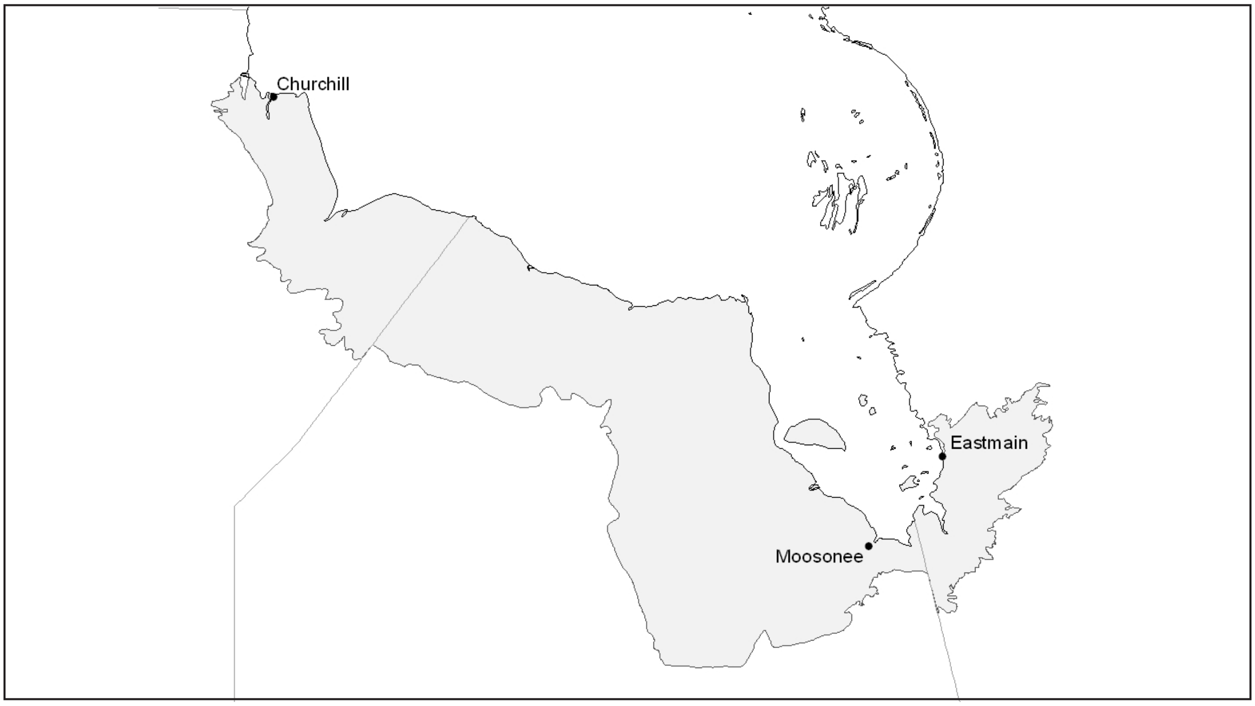

Figure 5. Locations of the three stations in the Hudson Plains Ecozone that have long-term data sufficient for the 1950-2007 climate trend analysis for the ESTR (as reported in this section). All three stations represent coastal conditions. The Eastmain station contributes to precipitation analyses only. Ecozone boundaries correspond to the ecozones+ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations. The nearest additional station is at Gillam, Manitoba, which is located in the Boreal Shield Ecozone near its junction with the western edge of the Hudson Plains Ecozone (Gillam is shown in Figure 1, Section 1.1, Geology, Topography & Climate*).*

Middel (Ontario Ministry of Natural Resources, unpublished) also found a significant ~2°C increase in mean daily May temperature at Moosonee over the somewhat longer period, 1930-2003. Overall, for the period 1950-2007, mean annual air temperature shows a significant trend only at the Churchill station, where it increased by 1.3 °C (Figure 8).

Total annual precipitation in the ecozone varies from about 430 mm in the northwest (Churchill) to about 680 mm in the southeast (Moosonee), based on 30 year (1971-2000) climate normals (Environment Canada 2010a; see also Riley 2003). At the ecozone scale, trends in seasonal precipitation are not evident from data averaged across the two to three stations with sufficient long-term data (Figure 9). Station-specific results, however, reveal a significant 28% reduction in the amount of spring precipitation at Moosonee (Figure 10). These precipitation trends for the Hudson Plains Ecozone contrast with those of the Arctic Ecozone, where the general trend is for greater precipitation in all seasons except summer (Zhang et al. in press). Overall, however, trends in precipitation in Hudson Plains Ecozone are not significant on an annual basis at either Churchill or Moosonee stations (data from the Eastmain station are insufficient to support annual analysis) (Figure 11).

Snowfall varies less across the ecozone than total precipitation, from 2.0 m in the northwest (Churchill) to 2.25 m in the southeast (Moosonee) (Environment Canada 2010a). The annual snow to total precipitation ratio shows no significant trend from 1950 to 2007 at the ecozone scale (for the combined data), although the pattern for the last two decades does suggest a decline in the contribution of snow (Figure 12). Partitioning out the station-specific results reveals a significant 7% reduction in this ratio at the Moosonee station (Figure 13).

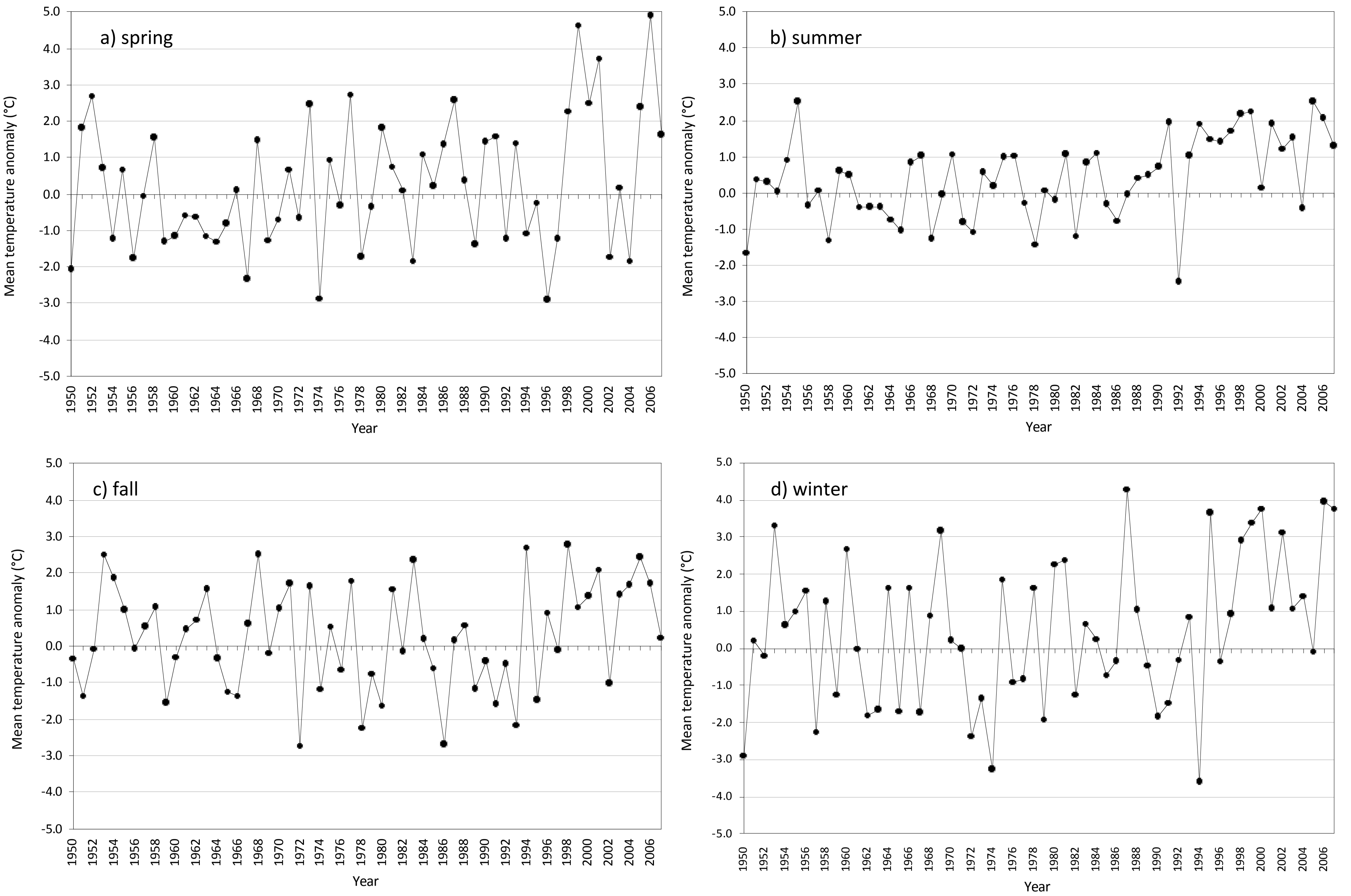

Figure 6. Ecozone-level mean air temperature anomalies for a) spring (March, April, May); b) summer (June, July, August); c) fall (September, October, November); and d) winter (December, January, February) for the period 1950-2007 relative to the base period (1961-1990) mean. The data indicate a significant ($p < 0.05$) increase in mean summer temperature of 1.9 °C from 1950 to 2007 but no significant change in mean spring, fall, or winter temperature during the same period. These analyses are based on data averaged from two stations in the ecozone (Churchill and Moosonee). Ecozone boundaries correspond to the ecozones^{+} framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.

Source: Data for ecozone provided by authors of Zhang et al. (in press).

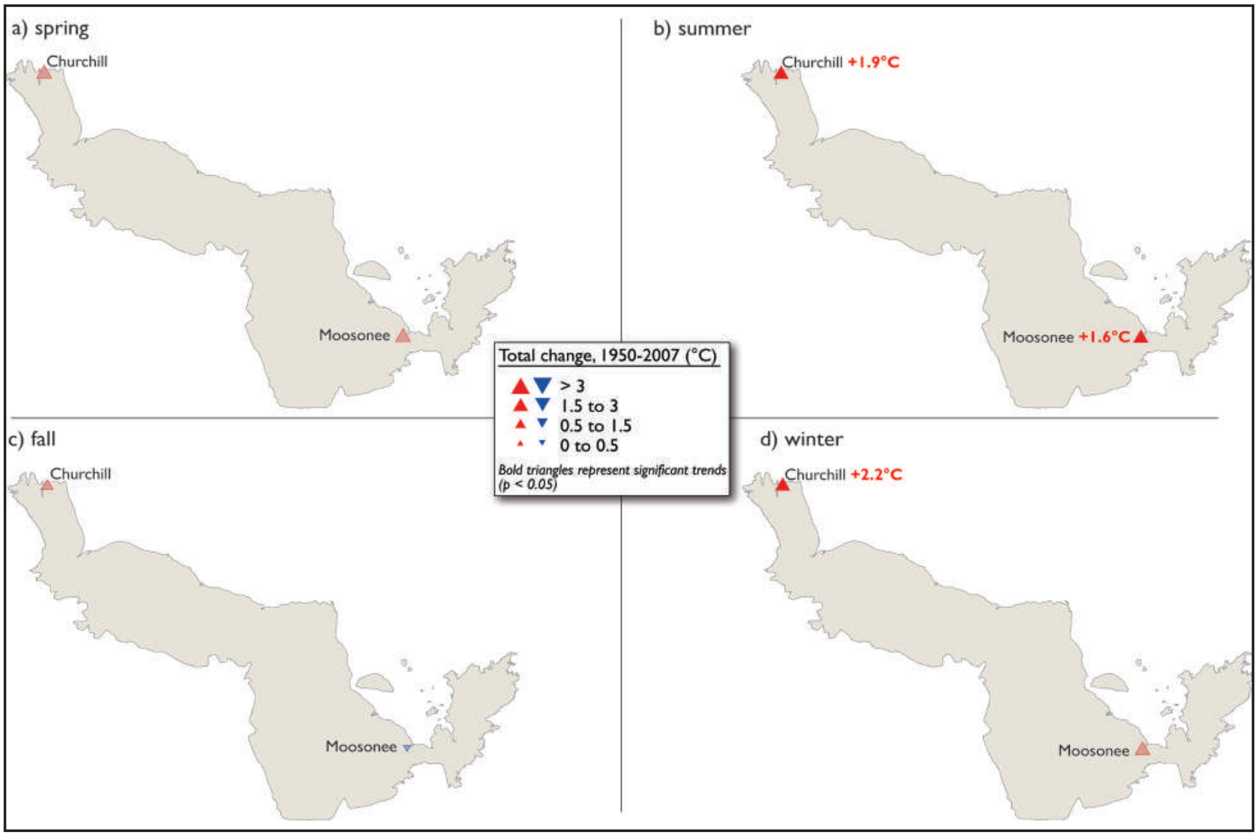

Figure 7. Station-specific trends in mean air temperature anomalies for a) spring (March, April, May); b) summer (June, July, August); c) fall (September, October, November); and d) winter (December, January, February) for the period 1950-2007 in the Hudson Plains Ecozone. Significant increases ($p < 0.05$) in temperature are evident in summer at both stations (Churchill and Moosonee) and in winter at Churchill. Ecozone boundaries correspond to the ecozones⁺ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.

Source: Data for ecozone provided by authors of Zhang et al. (in press).

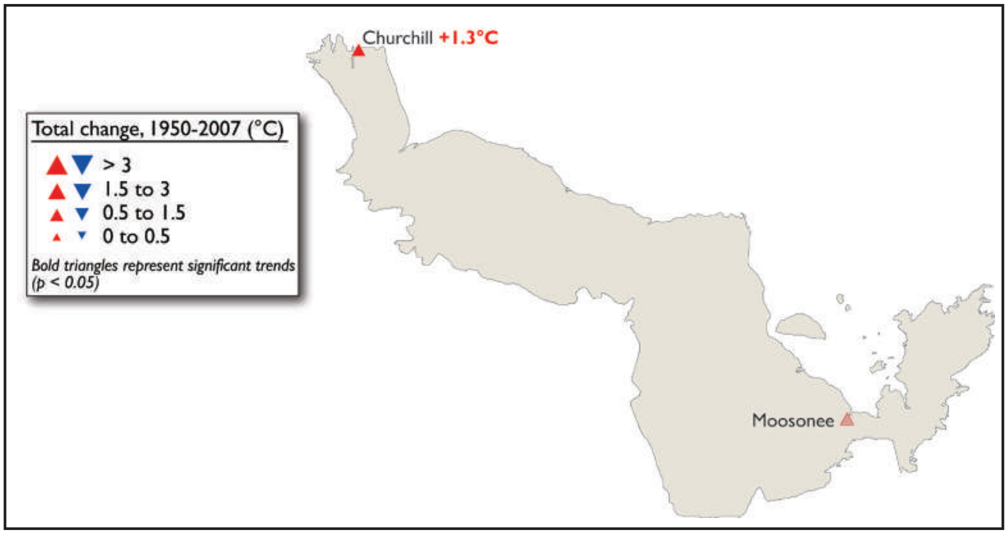

Figure 8. Station-specific trends in mean annual air temperature over the period 1950-2007 in the Hudson Plains Ecozone. A significant increase ($p < 0.05$) in mean annual air temperature is evident at the Churchill station but not at Moosonee. Ecozone boundaries correspond to the ecozones⁺ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.

Source: Data for ecozone provided by authors of Zhang et al. (in press).

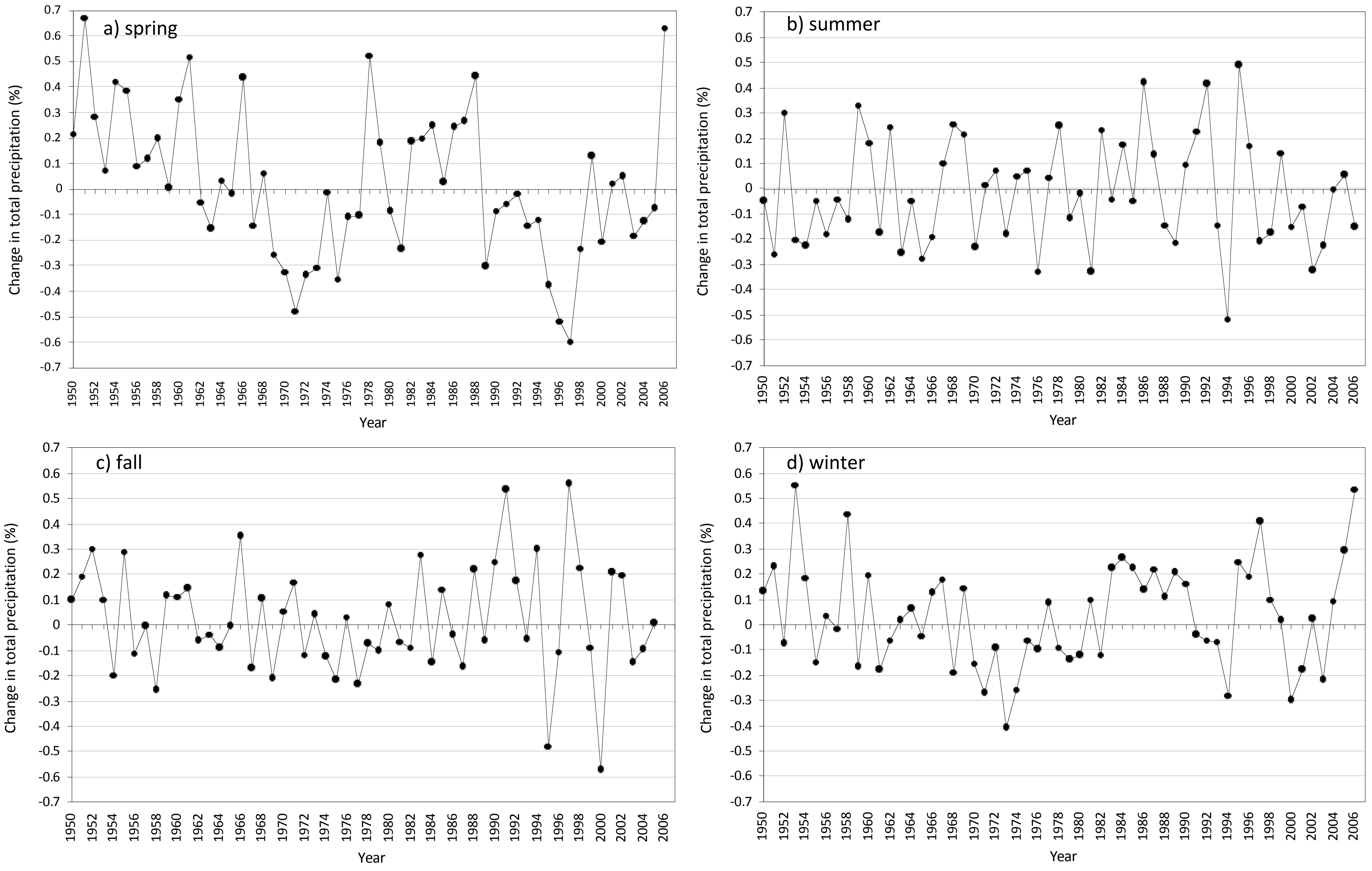

Figure 9. Ecozone-level annual total precipitation anomalies for a) spring (March, April, May); b) summer (June, July, August); c) fall (September, October, November); and d) winter (December, January, February) for the period 1950-2007 relative to the base period (1961-1990) mean. There is no significant (p >0.05) change in total spring, summer, fall, or winter precipitation from 1950 to 2007. The spring analysis is based on data averaged from two stations (Churchill and Moosonee). The summer, fall, and winter analyses are based on data averaged from three stations (Churchill, Moosonee, and Eastmain). Ecozone boundaries correspond to the ecozones$^+$ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

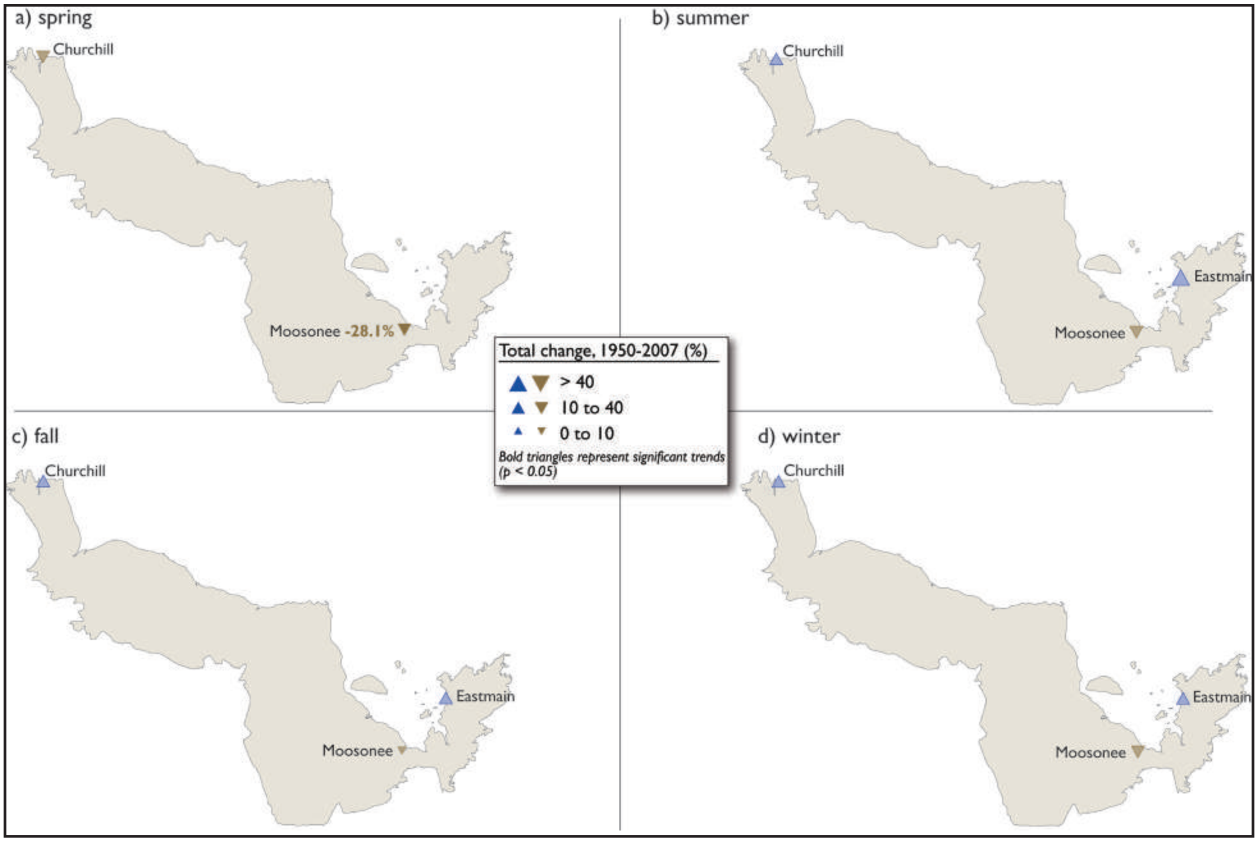

Figure 10. Station-specific trends in the amount of seasonal precipitation over the period 1950-2007 in the Hudson Plains Ecozone, expressed as a percentage of the 1961-1990 mean for a) spring (March, April, May); b) summer (June, July, August); c) fall (September, October, November); and d) winter (December, January, February). The spring analysis is for Churchill and Moosonee stations, while the summer, fall, and winter analyses also include the Eastmain station. A significant 28.1% reduction ($p < 0.05$) in spring precipitation is evident at the Moosonee station. Ecozone boundaries correspond to the ecozones$^+$ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

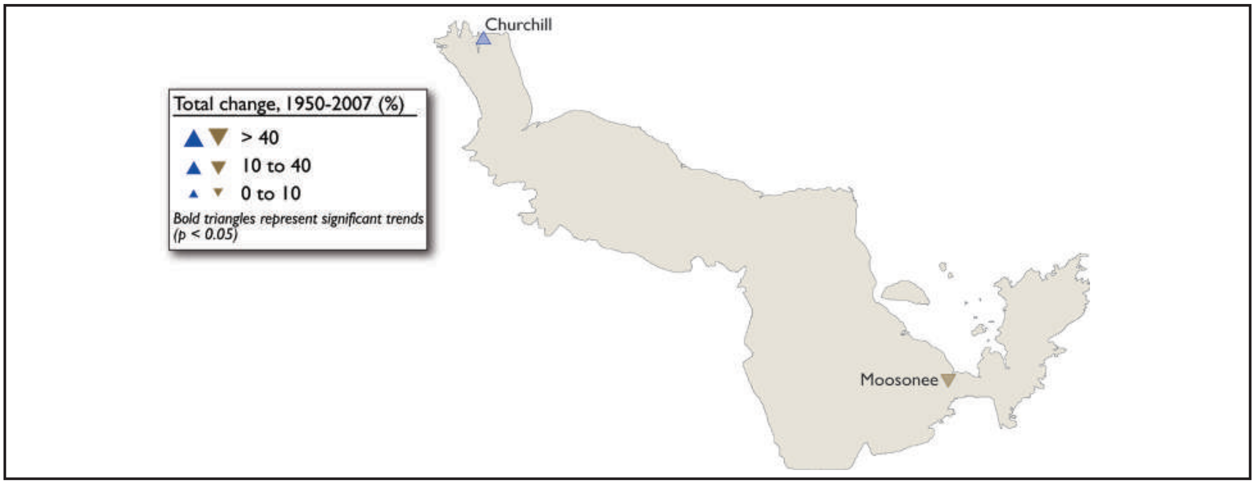

Figure 11. Station-specific trends in total annual precipitation over the period 1950-2007 in the Hudson Plains Ecozone, expressed as a percentage of the 1961-1990 mean. Trends are not significant ($p > 0.05$) for the two stations at Churchill and Moosonee. Ecozone boundaries correspond to the ecozones$^+$ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

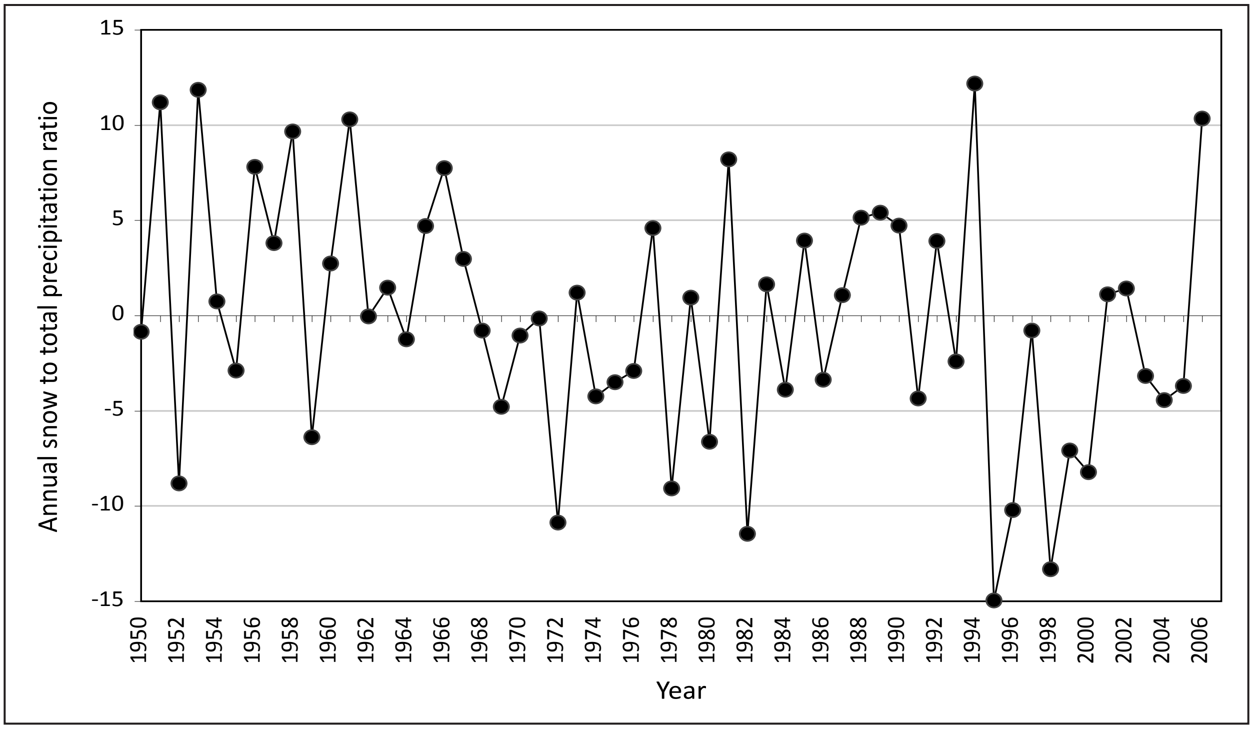

Figure 12. Mean annual snow to total precipitation ratio anomaly in the Hudson Plains Ecozone for 1950 to 2007 relative to the base period (1961-1990) mean. There is no significant ($p > 0.05$) change in the ratio of snow to total precipitation over the period 1950-2007. This analysis is based on data averaged from two stations in the ecozone (Churchill and Moosonee). Ecozone boundaries correspond to the ecozones^{+} framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations. Source: Data for ecozone provided by authors of Zhang et al. (in press).

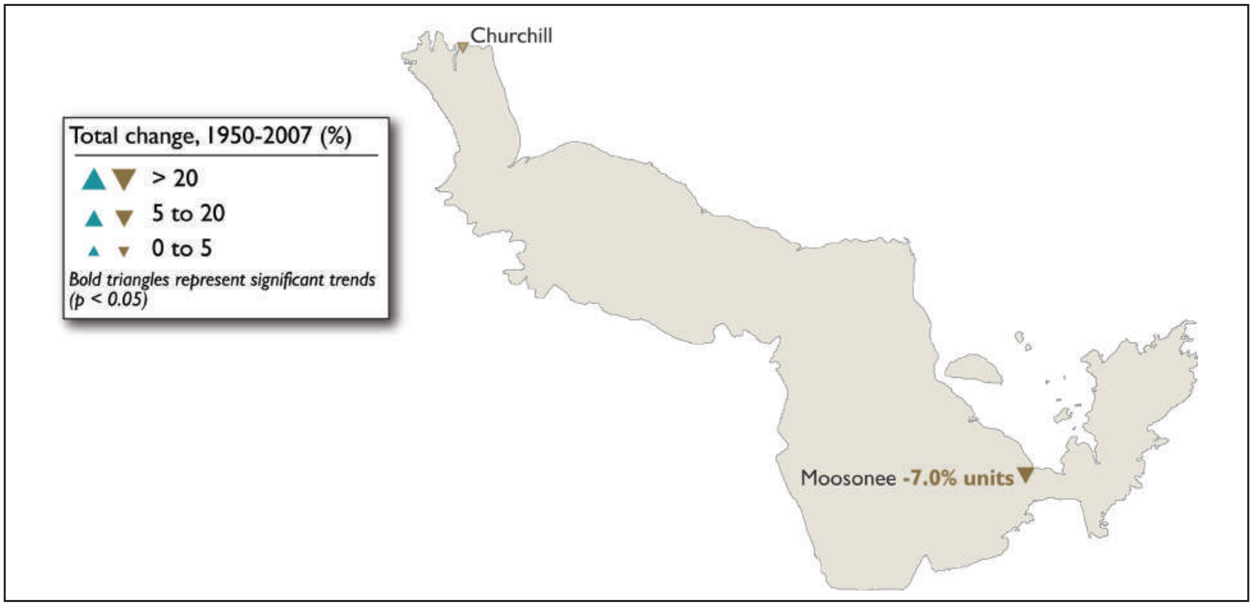

Figure 13. Station-specific trends in the snow to total precipitation ratio in the Hudson Plains Ecozone, 1950-2007. The significant ($p < 0.05$) downward trend at the Moosonee station indicates a decrease in the annual proportion of precipitation falling there as snow. Ecozone boundaries correspond to the ecozones^{+} framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations. Source: Data for ecozone provided by authors of Zhang et al. (in press).

The winter snow pack is typically present for about 6-9 months of the year (Findlay 1978). In the south, the snow pack generally exists until mid- or late-May, but in recent years it has disappeared earlier (late April to early May). Further north it can persist until the summer solstice (Rouse et al. 1997).

For the ecozone as a whole, the available station data do not show a significant change from 1950 to 2007 in the period of snow cover (Figure 14) or maximum annual snow depth (Figure 15). Station-specific analyses of these parameters likewise do not reveal significant trends at a sub-ecozone scale (Zhang et al. in press; not shown). However, a decline is suggested in both measures, when weighted by the last two decades (figures 14 and 15). Trends for decreasing snow cover are evident elsewhere in northern Canada (particularly in spring) (Zhang et al. in press) and the Northern Hemisphere more generally (Bates et al. 2008). Rivers discharging into Hudson, James, and Ungava bays also show trends for earlier mean annual date of peak discharge in spring, which are related to earlier snow melt (Déry et al. 2005, based on analysis of data from 1964-2000).

Although the total number of days per year with measurable precipitation has not changed at any analyzed station (Zhang et al. in press; not shown), seasonal changes in the distribution of precipitation are apparent. Specifically, the Churchill station shows a significant 14 day reduction in the number of spring days with measurable precipitation (Figure 16a), and the Eastmain station shows a significant 33 day reduction in the number of winter days with measurable precipitation (Figure 16d). These trends again contrast with the general trend in the Arctic Ecozone for a greater number of days with precipitation in all seasons (Zhang et al. in press).

The ecozone's growing season is short, being limited to about 65-75 days in the north. At the ecozone scale, no trends are evident in the start, end, and duration of the growing season for the period 1950-2007 (Figure 17), although the data set for one of the two stations used in this analysis is incomplete (see Figure 17 caption for explanation), and the large latitudinal span of the ecozone (50-59 °N) introduces large variability in the averaged measure. In fact, the number of growing degree-days >5 °C ranges from 595 at Churchill in the north to 1,110 at Moosonee in the south (Environment Canada 2010a). The Moosonee station also showed a significant increase in effective growing degree-days over the 1950-2007 period (Figure 18). This latter result for Moosonee is consistent with that of K. Middel for the period 1930-2003 (Ontario Ministry of Natural Resources, unpublished data), and it is related to increased average temperature there during the growing season rather than length of the growing period. In contrast, trends for increasing duration of growing-season are being observed nationally (Zhang et al. in press) and globally (Bates et al. 2008).

Net radiation is minimal in the mid-winter period (mid-November to March) in the Churchill area with small fluxes of conductive and convective heat and a cold atmosphere (Rouse et al. 1997). Although the daylight hours are long and the flux of solar radiation is high in the subsequent period from April to mid-May, net radiation is still low because of the presence of snow cover. The largest change in net radiation occurs with the melting of the snow in spring. At this time the heat flux into the ground is high, as is the latent heat flux associated with the evaporation of water. Although the albedo exceeds 0.8 when snow cover is present, in summer it ranges from 0.06 or 0.08 to 0.18, depending on terrain type and moisture (in general, wet sites have lower albedo than dry sites) (Lafleur et al. 1997; Rouse et al. 1997). In spite of these changes, surface air temperatures remain low in early summer, averaging about 7 °C until soil temperatures rise.

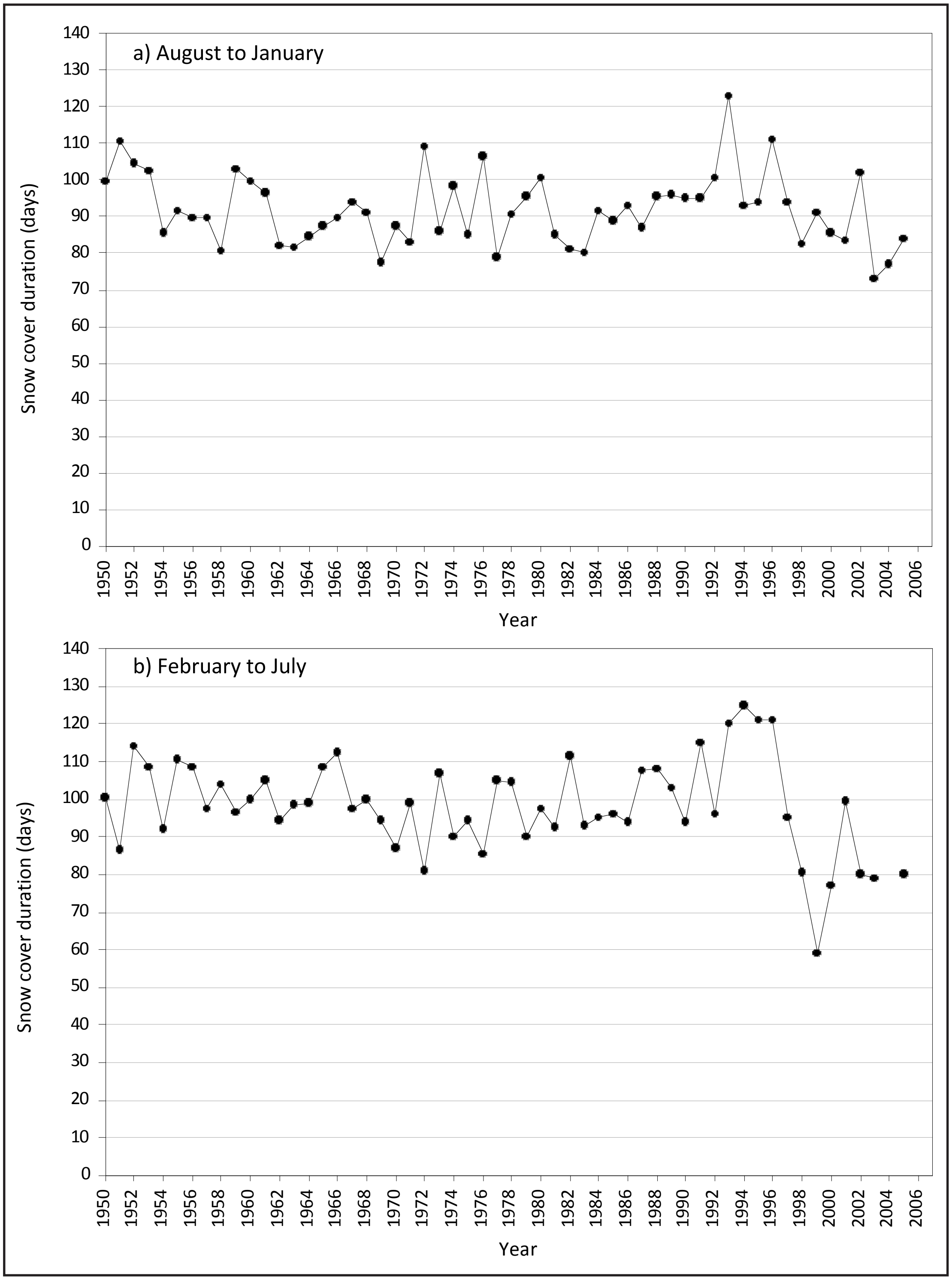

Figure 14. Number of days with ≥2 cm of snow on the ground (snow cover duration) from a) August to January (fall) and b) February to July (spring) over the period 1950-2007 in the Hudson Plains Ecozone. The number of days with snow cover shows no significant (p >0.05) change for either season. These analyses are based on data averaged from two stations in the ecozone (Churchill and Moosonee). Ecozone boundaries correspond to the ecozones⁺ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

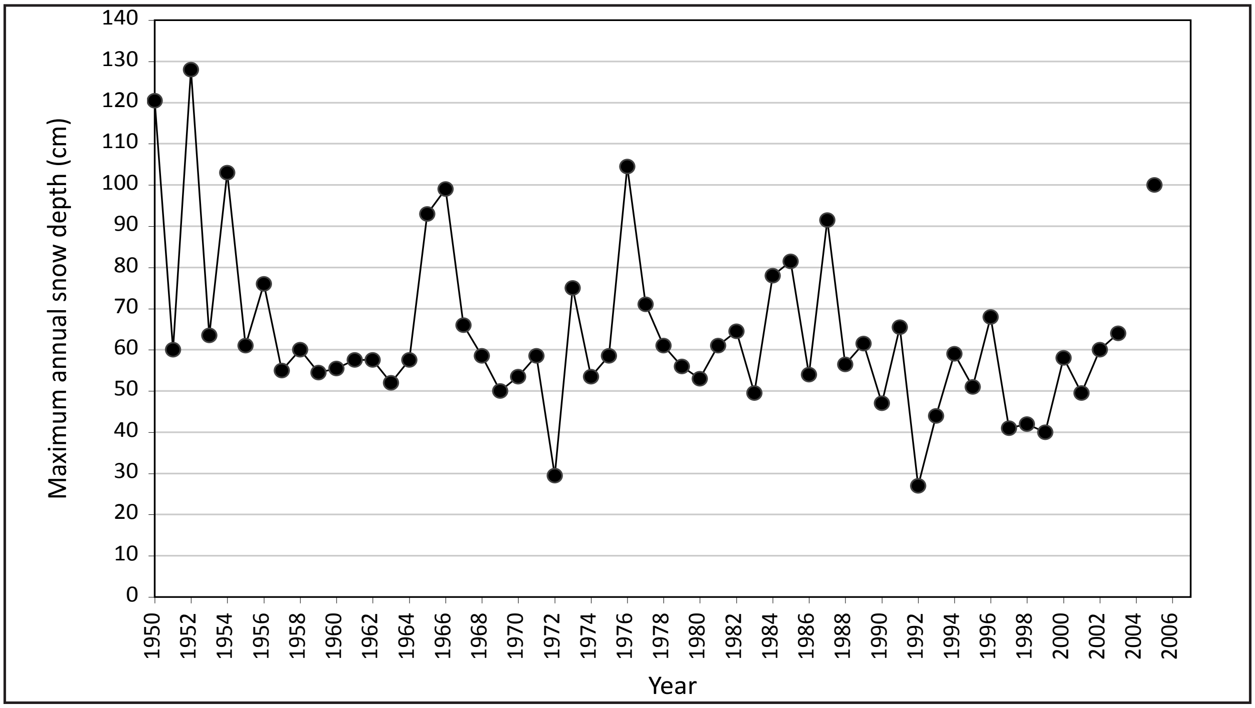

Figure 15. Maximum annual snow depth in the Hudson Plains Ecozone, 1950-2007. There is no significant ($p > 0.05$) change in the maximum annual snow depth over this period. This analysis is based on data from two stations in the ecozone (Churchill and Moosonee). Ecozone boundaries correspond to the ecozones^{+} framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations. Source: Data for ecozone provided by authors of Zhang et al. (in press).

In summer, net radiation falls due to the post-solstice decline in solar radiation, but higher air temperatures prevail in July. Evapotranspiration is a major component of the summer water-balance in community types such as wet sedge meadows, which are widespread in the ecozone, and the loss of water to the atmosphere usually exceeds the precipitation input during this period. The sensible heat flux is still large, because the drier strings and hummocks may be 20 °C warmer than the flarks or hollows that contain standing water and may be shaded (Halliwell et al. 1991).

In autumn, net radiation and convective heat fluxes return to low values. A large loss of ground heat occurs in winter, and this heat warms the lower atmosphere. Effectively, then, all of the significant energy fluxes in the ecozone occur in the four month period from June to October (Rouse et al. 1997).

Boudreau and Rouse (1995) measured the water balance of five terrain types (sedge-dominated wetland, upland lichen-heath, tundra lakes and ponds, willow-birch wetland, and open spruce forest). Runoff and stream flow were closely linked to the moisture status of the peatland and the depth of the active layer. The spring runoff event effectively recharged the system and, subsequently, ridges vegetated with lichen-heath or spruce forest shed a large proportion of stored water to the low-lying wetlands, which maintained stream flow and evapotranspiration. The drainage of water from the uplands to the lowlands maintained the soil moisture of the peatlands and contributed to stream flow throughout the summer, even when drought occurred (Boudreau and Rouse 1995).

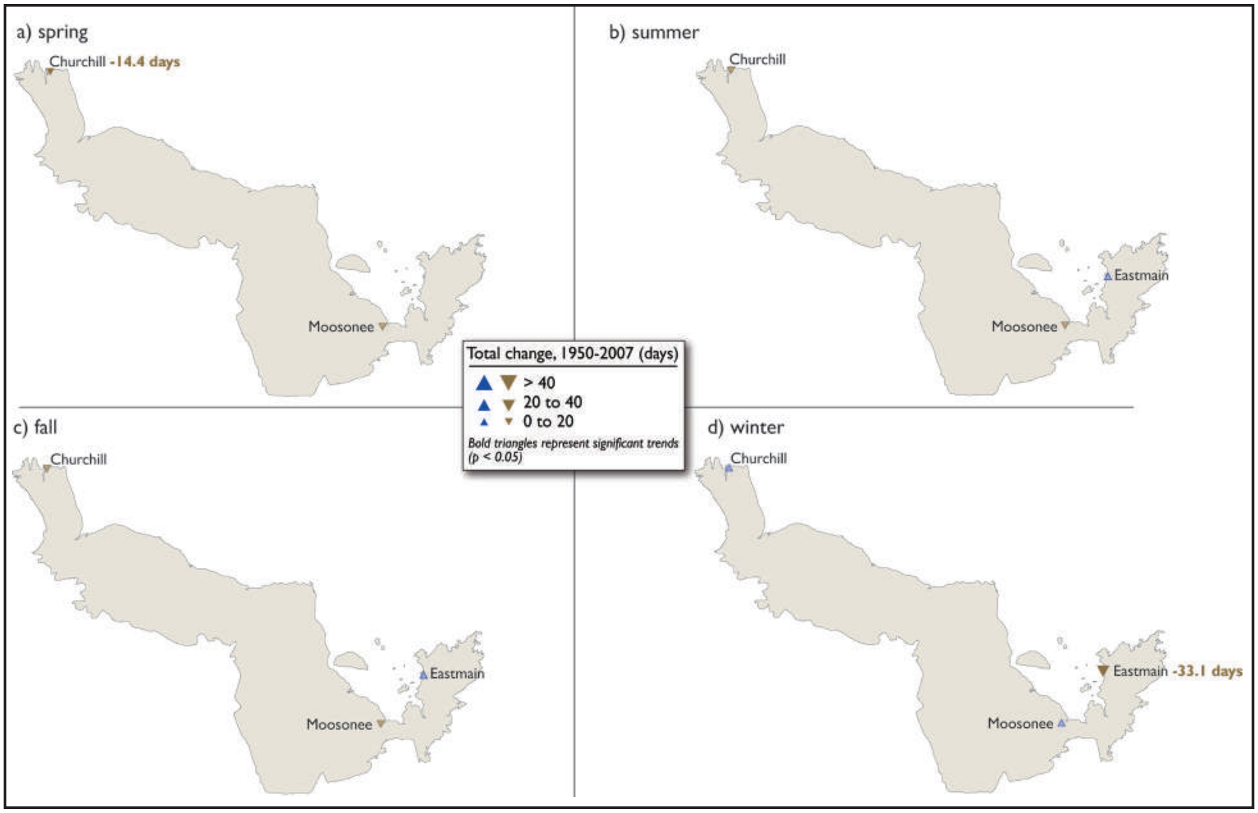

Figure 16. Station-specific changes in the total number of days with measurable precipitation over the period 1950-2007 in the Hudson Plains Ecozone for a) spring (March, April, May); b) summer (June, July, August); c) fall (September, October, November); and d) winter (December, January, February). The spring analysis uses data from two stations (Churchill, Moosonee). The summer, fall, and winter analyses are based on data from three stations (Churchill, Moosonee, and Eastmain). Churchill and Eastmain stations showed a significantly fewer total number of days with measurable precipitation in spring and winter, respectively. Ecozone boundaries correspond to the ecozones$^+$ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

Although peat surfaces dry out in summer, the peat remains moist below the surface due to its large moisture-holding capacity. A large vapour pressure gradient thus develops between the evaporative surfaces in the peat and the atmosphere immediately above. Turbulent atmospheric flow dissipates the latent heat flux and maintains the gradient (Rouse et al. 1997). Lichen mats present on the surface of the lichen-heathland hold considerable quantities of water at snow melt or after heavy rainfall, which then evaporate in summer without ever reaching the soil. This effect indirectly contributes to the drying of the peat (Bello and Arama 1989).

Trends in summer dryness are not apparent in the ecozone based on analysis of monthly drought code (1950 onward) as a moisture index (Girardin and Wotton 2009; Girardin et al. 2009). Likewise, an analysis of the summer Palmer Drought Severity Index (PDSI)[7] at the Churchill station shows no significant change there over the period 1950-2007 (Figure 19).

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[7] The Palmer Drought Severity Index (PDSI) is an index of water availability that reflects changes in long-term moisture, runoff, recharge, deep percolation, and evaporation over time spans of months or seasons. It is computed using concurrent and co-located temperature and precipitation data, but it is more sensitive to temperature than to precipitation.

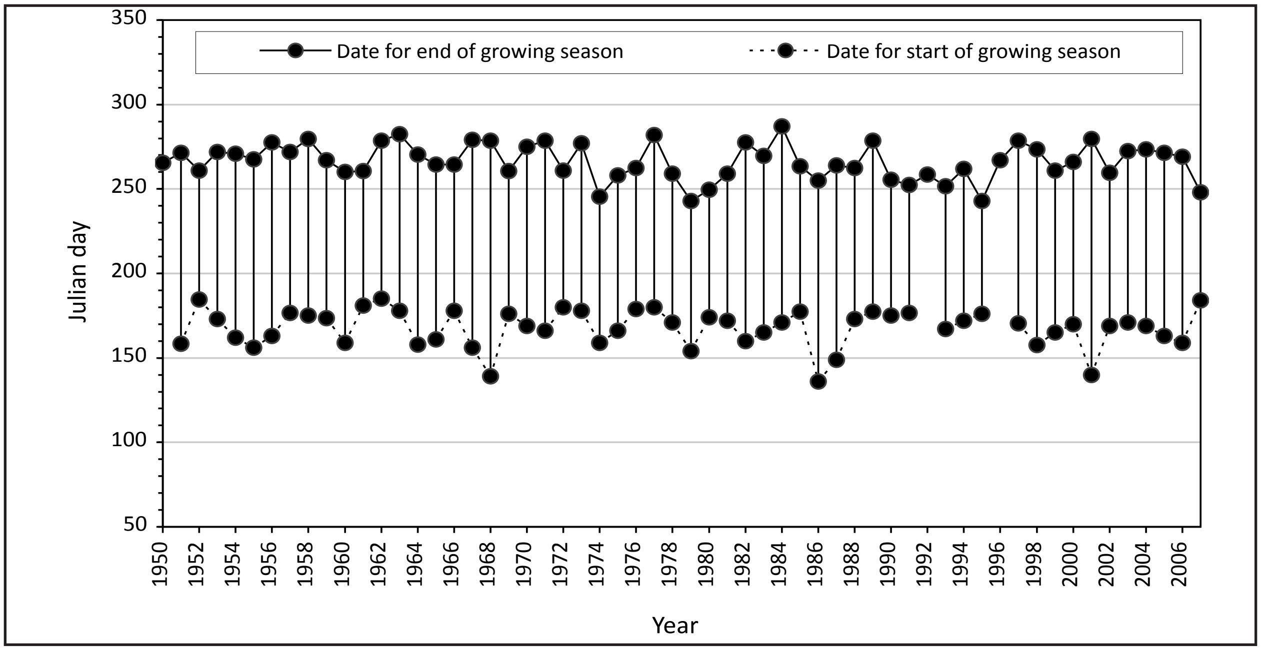


Figure 17. Start, end, and length of the growing season over the period 1950-2007 in the Hudson Plains Ecozone. The dotted line represents the start of the growing season (first day of the year when the 5 day moving average of Tmean has been >5 °C for 5 consecutive days). The solid line represents the end of the growing season (the last day of the year when the 5 day moving average of Tmean is >5 °C). The vertical lines represent the length of the growing season. The growing season ranges from 65 to 155 days, and no significant (p >0.05) changes are evident in the start, end, or length of the growing season over the period 1950-2007. These analyses are based on data averaged from two stations in the ecozone (Churchill and Moosonee), but the data for Churchill is incomplete (for Churchill it was not possible in some years to define the start of the growing season as the first day of the year when the 5 day moving average of Tmean was >5 °C). Ecozone boundaries correspond to the ecozones+ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

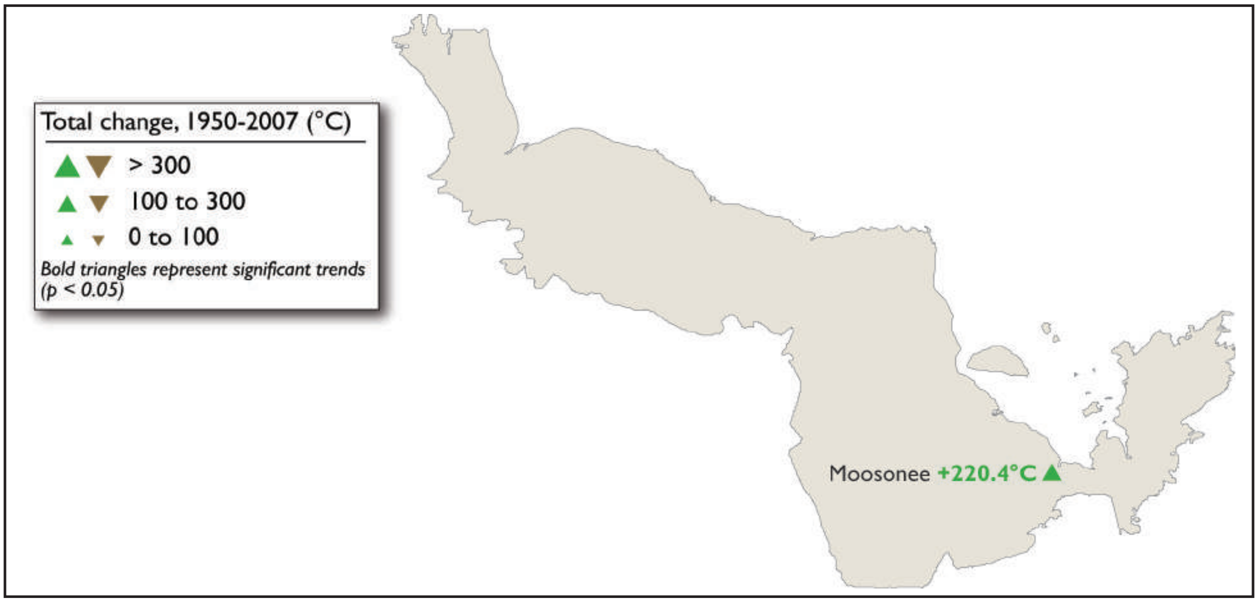
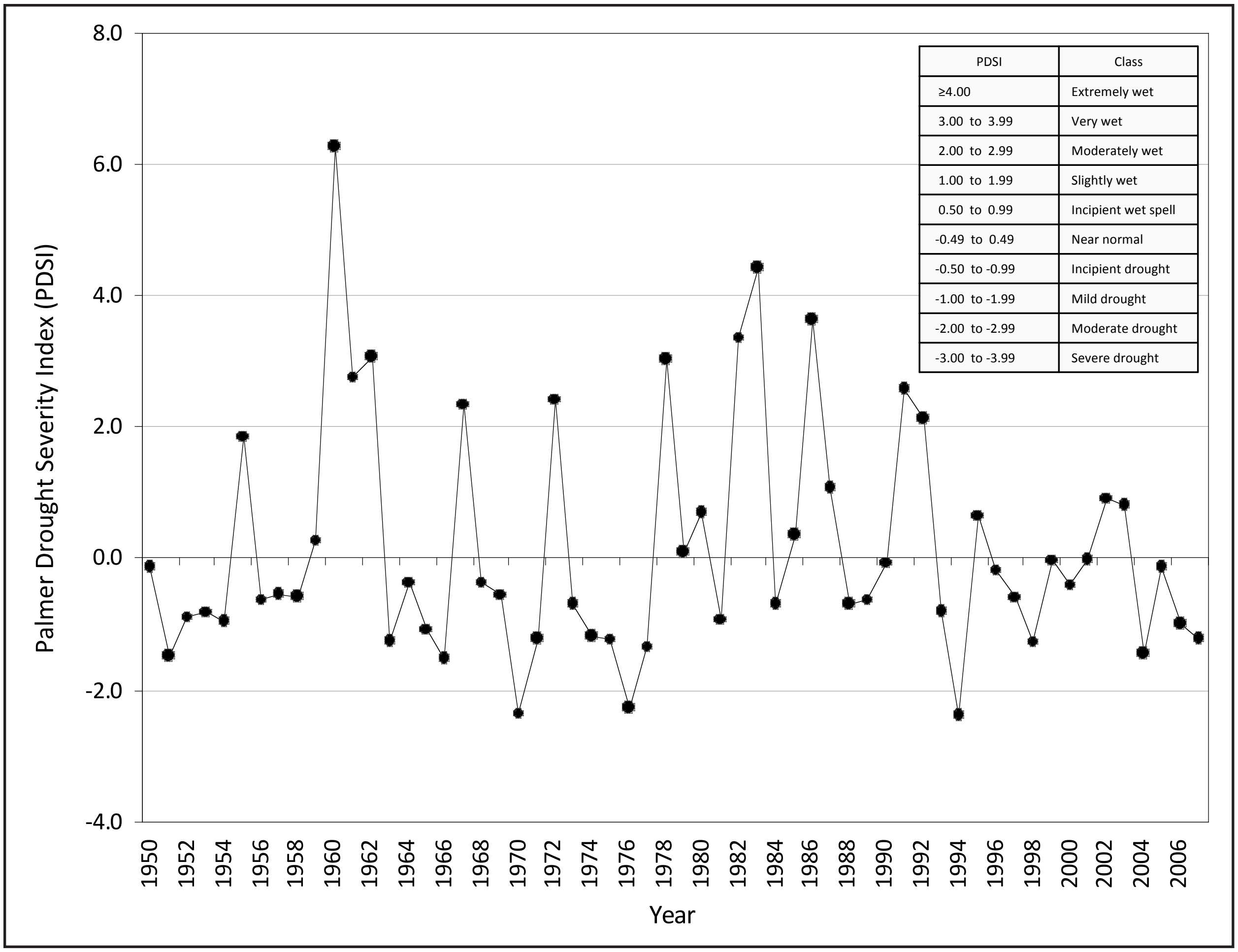


Figure 18. Station-specific trends in effective growing degree-days (a measure of accumulated heat during the growing season) over the period 1950-2007 in the Hudson Plains Ecozone. Trend analysis was not possible for the Churchill station, where growing season could not be determined in some years (for explanation, see Figure 17 caption). Ecozone boundaries correspond to the ecozones+ framework; use of ESWG (2005) boundaries would not result in the inclusion of additional stations.
Source: Data for ecozone provided by authors of Zhang et al. (in press).



PDSI	Class
≥4.00	Extremely wet
3.00 to 3.99	Very wet
2.00 to 2.99	Moderately wet
1.00 to 1.99	Slightly wet
0.50 to 0.99	Incipient wet spell
-0.49 to 0.49	Near normal
-0.50 to -0.99	Incipient drought
-1.00 to -1.99	Mild drought
-2.00 to -2.99	Moderate drought
-3.00 to -3.99	Severe drought

Figure 19. Trends in summer (June, July, August) Palmer Drought Severity Index (PDSI) at the Churchill station over the period 1950-2007. Positive PDSI values indicate wet conditions, while negative values indicate dry conditions. PDSI shows no significant change (p >0.05) over this period.
Source: Data for ecozone provided by authors of Zhang et al. (in press).

2.1.1.1 **Presence of Hudson & James bays**

Hudson Bay, combined with Foxe Basin to the north and James Bay to the south, is considered the world's largest inland sea, and it is largely cut off from advection of ice and currents from other ocean systems (Gagnon and Gough 2005a). Hudson Bay has a major advective effect on the Hudson Plains Ecozone, while comparatively little is known about the effect of James Bay on the southern part of the ecozone. Hudson Bay produces cold moist atmospheric conditions above its surface, and cold mesoscale winds impose cold air temperatures on the adjacent terrestrial environment, which suppresses evaporation there (Rouse 1991). In the intertidal zone of Hudson Bay, approximately two-thirds of the net radiation is used to heat the inshore water, while the remaining third is used for evaporation. As such, little energy is available to heat the atmosphere over the water. These overall effects along the coastal zone of the Hudson Plains Ecozone have been called the *winterization* of summer by Rouse (1991).

The onshore flow of air into the ecozone may be felt as much as 600 km inland. The temperature difference between the immediate coastline and 3 km inland can be as much as 8 °C in spring

(Rouse and Bello 1985), and temperature differentials continue to increase further inland at a slower rate (e.g., Dyke and Sladen 2010). Maximum rates of evapotranspiration are realized about 12 km inland; further inland rates of evaporation fall, as the vegetation (lichen-heath communities) is more xeric (Rouse 1991), and the peat is thicker (Dredge and Mott 2003; Glaser et al. 2004). The surface layers dry out quickly in summer due to rapid infiltration of water during rainfall events and evaporation from the soil surface. The result is a water deficit at the immediate surface and a strong surface resistance to evaporation (Rouse 1991).

The effects of ocean currents, tides, and sea ice in Hudson and James bays on the climate of the Hudson Plains Ecozone are highlighted below, while related coastal building processes are discussed later in Section 2.4.1. The effects of factors such as overland river flow and sedimentation on the adjoining marine environment are discussed principally in the Arctic Marine Ecozones report of the ESTR (Niemi et al. 2010).

2.1.1.1.1 **Ocean currents & tides**

During the summer, warm surface waters move in an anticlockwise direction in Hudson and James bays (Prinsenberg 1986b; see also Figure 101 in Section 2.4.1, *Coastal Building Processes*). The deeper waters are thought to travel in a similar direction. The Hudson Bay water is joined by cold arctic water that enters from Foxe Basin and flows in the general circulation pattern to the southern Hudson Bay coast. Some of this water from the Foxe Basin is deflected north and leaves the area via Hudson Strait (Prinsenberg 1986 a,b).

The circulation in Hudson Bay is stable, largely on account of the discharge of freshwater that results in strong density stratification in summer, lack of vertical mixing, and weak coastal currents (Wang et al. 1994a). The circulation is maintained by the various inflows and outflows and by wind movements. Residence times for water in Hudson Bay are uncertain. Estimates range from 1-2 years (Ingram and Prinsenberg 1998), to 3-4 years (Jones and Anderson 1994), to over 6 years (Prinsenberg 1984).

James Bay has a two-layer circulation system: a thin 20-30 cm upper layer that has a net outflow and a lower layer that has a net inflow (Prinsenberg 1982). The estimated outward flow of water from James Bay ranges from 65,000 m^3/s in August to 167,000 m^3/s in October (Prinsenberg 1982). The density-driven currents that result from the large inflow of freshwater into the bay are greatest in early summer. Notably, the anticlockwise movement of the warmer fluvial waters injected into the bay leads to a significant northward shift of permafrost zones on land on the east side of the bay (see the permafrost zones map, Figure 23, later in Inset 2). The residence time for the surface and bottom layers of water in James Bay is 3-7 months for summer, but during the winter it increases to 20 months due to the slower circulation (Prinsenberg 1982). Hydroelectric developments that increase runoff in winter (e.g., the 8-10 times higher winter flow of the La Grande River into James Bay; Therrien et al. 2004) might increase the winter circulation of water in James Bay.

Semi-diurnal tides enter Hudson Bay via Hudson Strait from the Atlantic Ocean, and a weak diurnal tide also exists within Hudson Bay (Freeman and Murty 1976). Tidal ranges are low compared to Hudson Strait, except where water is funneled in embayments or estuaries. At Churchill, for example, the range between high and low tide is 4 m, whereas along the eastern shore of James Bay it can be as little as 0.1 m (Godin 1972). The diurnal tide as distinct from the semi-diurnal tides is only 0.08 m and 0.03 m at these respective locations. Tidal currents in

Hudson Bay are generally less than 30 cm/s, but values as high 150-200 cm/s can occur around channel restrictions. The regularity of tides is affected by the spring freshet, ice cover, and weather disturbance. The spring break-up of ice in rivers limits tidal input into estuaries, and freshwater discharge extends into the bays (Godin 1972). The presence of sea ice cover dampens the tidal amplitude, and it advances the time of high water and the flood currents (Godin 1980, 1986). The mixing of tidal waters from marine and freshwater sources controls sensible heat transfer, which reduces sea ice formation and contributes to the maintenance of open water in polynyas close to islands and the surrounding terrestrial landscape.

Median wave height in Hudson Bay is 1-2 m, and the wave period is 5-6 seconds (Maxwell 1986). Storm surges can occur in autumn and early spring along the southern coast of Hudson Bay and in James Bay. When they occur the high water mark is extended well inland (~3 km) (Godin 1974). Murty (1972) predicted that a wind of 89 km in James Bay would result in a tidal surge of 6 m.

Much of the information available on the effects of ocean currents and tides on the Hudson Plains Ecozone is now quite dated. New data and new analyses are needed, especially in light of hydroelectric developments that might have altered conditions in Hudson and James bays in the last few decades (hydroelectric developments are discussed in Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

2.1.1.1.2 **Sea ice**

Sea ice has had a defining role in shaping and maintaining the Hudson Plains Ecozone. Hudson and James bays are unique among seas at this latitude in that they undergo a complete annual cryogenic cycle, completely freezing-over through the winter months and becoming fully ice-free for generally two months in late summer (Gough and Wolfe 2001; Parkinson and Cavalier 2008). On rare occasions, multi-year ice has been present, but it is limited to the northeast (Etkin and Ramseier 1993). The timing of freeze-up and break-up is closely related to ambient temperature but generally occurs in mid-November and late July, respectively. The timing of ice melt is generally more variable than that of ice formation (Wang et al. 1994b). The complete freeze-up of the bays essentially nullifies the moderating effect of warmer ocean waters along coastal areas, as would occur along other ocean shorelines that do not freeze-up (Gough and Wolfe 2001; Gagnon and Gough 2005a). This anti-moderating effect, combined with persistence of the sea ice late into mid-year, is believed to contribute to the presence of permafrost in the Hudson Plains Ecozone, including the most southern continuous permafrost in North America (Gough and Leung 2002; Zhang et al. 2008). Permafrost inhibits penetration of surface water generated by rain and snow melt from entering into the ground. Evaporation of this surface water in warmer months subsequently utilizes energy that would otherwise warm the surface (Maxwell 1986; Gagnon and Gough 2005a).

The presence of permafrost and its effective retention of surface water contribute greatly to the uniqueness of the Hudson Plains Ecozone as Canada's largest wetland complex and the third largest wetland complex in the world (Fraser and Keddy 2005). Lakes (mostly shallow lakes and ponds) can account for as much as 40% or more of the surface area as observed, for example, in the area south of Churchill (Bello and Smith 1990)[8]. The interconnectedness of this wetland

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[8] The value in Bello and Smith (1990) was obtained by digitizing a 1:25,000 black-and-white air photo. Smaller estimates (<10%) have been obtained in other areas or studies, in some cases using lower resolution maps or satellite imagery that did not resolve as well the vast number of smaller lakes and ponds (for details, see Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*).

complex with the presence of annual sea ice in Hudson Bay means that it, too, is vulnerable to changes in extent and longevity of the sea ice each year. Earlier melting of sea ice will likely result in an overall warming of the ecozone (Rouse 1991). With a shorter sea ice season, less cool air would be carried inland, resulting in less cooling there during the growing season. The implications of sea ice loss to this ecozone are profound (see Section 2.1.2, *Trends Assessment & Projected Changes*).

The winter maximum extent of sea ice in Hudson and James bays has not changed, and the bays continue to freeze completely over each year – albeit with considerable changes in phenology, including in areas adjacent to the Hudson Plains Ecozone (Stirling et al. 2004; Obbard et al. 2006; Stirling and Parkinson 2006). Using data from the Canadian Ice Service archives, Gagnon and Gough (2005b) found significant trends for longer ice-free periods each year, with later freeze-up dates, earlier break-up dates, or both (Inset 1). The pattern of sea ice formation and melt is predictable based on winds, tides, and prevailing currents in Hudson Bay, though the timing varies among years (Stirling et al. 1977; Etkin 1991; Stirling et al. 2004). Analysis of freeze-up and break-up dates indicates that Hudson Bay is not changing consistently across its extent, as ice formation and break-up dates are related to ambient air temperature, proximity to land, and active currents (Gagnon and Gough 2005b). The period of ice cover is substantially shorter along the western coast of Hudson Bay, with significant changes in the dates of both freeze-up and break-up. Southern Hudson Bay and James Bay show significant trends towards earlier break-up and non-significant trends for later freeze-up. On average, the period of ice cover in western Hudson Bay, southern Hudson Bay, and James Bay has decreased by ~3 weeks, since the mid-1970s (when Canadian Ice Service began collecting weekly or biweekly data for sea ice in this region) (Gagnon and Gough 2005b).

The broader Hudson Bay marine ecosystem has shown a significant negative trend in sea ice extent (minimum 15% concentration) over the period 1979-2006, on the order of –5.3 ± 1.1% per decade, with decreases evident in all seasons except winter (Figure 20); results for total cumulative area of ice coverage are qualitatively similar (Parkinson and Cavalieri 2008).

2.1.1.2 **Presence of numerous small lakes & ponds**

One of the characteristic features of the Hudson Plains Ecozone is the presence of numerous, mostly shallow thermokarst lakes and ponds. The area of such lakes as a percentage of the total area can be as high as 40% or more, as occurs in the region south of Churchill (Bello and Smith 1990) (see Footnote 8). These waterbodies have a strong effect on the local climate, as their ice-free period is very short. They also contribute (along with wetlands) to the presence of permafrost, as many of them freeze to the bottom such that no warmer water is in contact with the ground and transferring heat to it throughout the winter.

Across the ecozone, freshwater freeze-up generally occurs earlier in the north and later in the south, while the converse is true for break-up (Cohen et al. 1994; see also Déry et al. 2005). With the onset of winter, net radiation becomes increasingly negative and, as the water temperature drops below 4°C, the colder, least dense water stratifies just below the ice surface. With the formation of surface ice, turbulent mixing of the water in these lakes and ponds ceases. Ice can develop to a depth of 1.5 m to 1.75 m, with shallow waterbodies freezing to the bottom (Rouse et al. 1997; Duguay and Lafleur 2003). Snow accumulates on the ice surface, and 1 cm of snow has the same insulating effect as 25 cm of ice. In spring, ice melt usually occurs from the bottom of

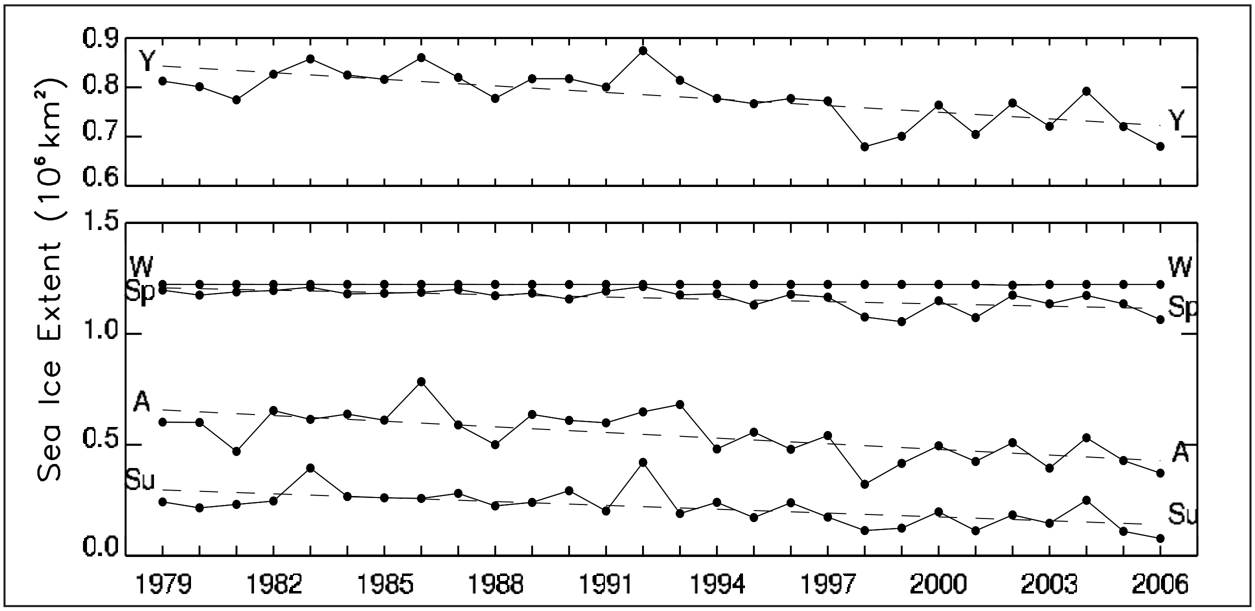

Figure 20. Yearly and seasonally averaged extent of sea ice in Hudson Bay over the period 1979-2006 from NASA satellite passive-microwave observations. Abbreviations: Y, yearly; W, winter (January-March); Sp, spring (April-June); A, autumn (October-December); and Su, summer (July-September). Source: Parkinson and Cavalieri (2008). Reprinted from Journal of Geophysical Research, Volume 113, Issue C07003, C.L. Parkinson and D.J. Cavalieri, Arctic sea ice variability and trends, 1979-2006, pp 1-28, Copyright 2008, with permission from American Geophysical Union.

the ice upwards in association with the slushing of ice at the boundary between ice and water. Ice that is bonded to the shoreline of lake sediments weakens, as a result of melt water runoff, which raises the water level by as much as 0.5 m in shallow lakes and causes the ice mass to lift and float. At this stage, thinning of the ice can occur from the top downwards due to convective heat transfer and the input of solar radiation on the ice surface. Hence, the surface albedo and the surface temperature control the net radiation in the different seasons. During the ice-free period, net radiation is approximately two-thirds of the incoming solar radiation for a medium-sized lake and probably for smaller lakes (Rouse et al. 1997).

Data are insufficient for examining long-term trends in freshwater (lake and river) ice in the Hudson Plains Ecozone (Zhang et al. 2001; Monk et al. in press; Zhang et al. in press). For example, trends in break-up dates for the lower Attawapiskat, Albany, and Moose rivers (near the James Bay coast) were inconclusive when examined from disparate community-based data sources (Ho et al. 2005). Changes in freshwater ice are, however, suspected. For example, Aboriginal peoples in western James Bay have noticed changes in the break-up and/or freeze-up of rivers (Ho et al. 2005), a reduction in ice thickness both in naturally flowing rivers and rivers with flows modified by hydroelectric development, and longer ice-free periods for some inland lakes (McDonald et al. 1996). At a national scale, analysis of available lake and river ice data shows a similar trend to that of sea ice, with freshwater ice typically breaking-up earlier in the year and in some cases also forming later (Bonsal et al. 2006; Duguay et al. 2006; Latifovic and Pouliot 2007). Although not consistently significant across the nation, Duguay et al. (2006) detail the relationship between changes in air temperature and related changes in freeze-up and break-up dates. Their work supports the findings of Magnuson et al. (1997) that, although other factors (e.g., solar radiation, wind, and snow depth) can influence ice formation and break-up, air

Inset 1. Changing sea ice conditions in Hudson & James bays

Kevin Middel, Ontario Ministry of Natural Resources

Hudson Bay, including James Bay to the south and Foxe Basin to the north, is considered one of the most unique seas in the world. It is the largest inland sea and the only sea at this latitude that goes through a complete cryogenic cycle each year (Gagnon and Gough 2005a; Parkinson and Cavalier 2008). This factor has been primary in shaping the ecosystem around it by creating much cooler temperatures than what is typical of this latitude (Rouse 1991), thereby providing the conditions necessary for maintaining the southernmost reach of continuous permafrost in North America (Gough and Leung 2002; Zhang et al. 2008) and supporting species of arctic affinity, such as polar bear (*Ursus maritimus*) (COSEWIC 2008), arctic fox (*Vulpes lagopus*) (Hersteinsson and Macdonald 1992), and some plants (Riley 2003) at their southernmost occurrence. If current trends of warming continue as projected (Skinner et al. 1998; Gagnon and Gough 2005a), much of this could change.

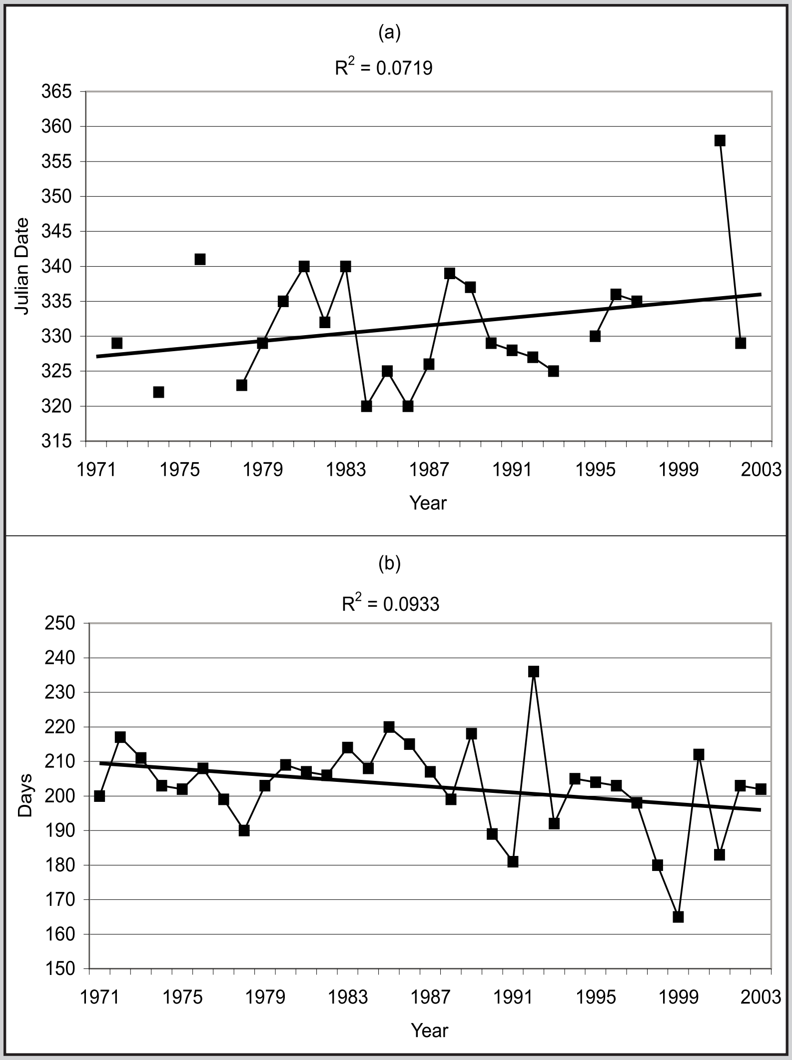

Figure 21. Trends in a) date of freeze-up and b) date of break-up of sea ice in southwestern Hudson Bay, based on data from the Canadian Ice Service archives.
Source: Gough et al. (2004). Reprinted from Arctic, Volume 57, Issue 3, W.A. Gough, A.R. Cornwell and L.J.S. Tsuji, Trends in seasonal sea ice duration in southwestern Hudson Bay, pp 299-305, Copyright 2004, with permission from Arctic Institute of North America.

Analysis of historical data on the concentration and coverage of ice in southern and western Hudson Bay reveals that it is becoming increasingly ice free (Stirling et al. 1999; Gough et al. 2004). Using historical ice-chart data for Hudson Bay for the period 1971-2003, Gough et al. (2004) found significant trends for earlier dates of sea ice break-up in southwestern Hudson Bay. Although they did not find a trend in later freeze-up dates, the trend for earlier break-up alone resulted in an approximate increase in ice-free conditions by approximately 0.49 days per year (Figure 21). Subsequent work by the same authors that expanded the study area to include the entire Hudson Bay region found a trend for earlier break-up in James Bay, southern Hudson Bay, and western Hudson Bay with magnitudes ranging from 0.49 to 1.25 days earlier per year, coincident with temperature trends in these areas (Gagnon and Gough 2005b). These trends in sea ice are correlated with significant impacts on polar bear, which is dependent on sea ice as habitat and as a platform for hunting and feeding on seals (Stirling and Archibald 1977; Smith 1980).
For more information on the ecozone's polar bears, refer to Section 2.3.3.1.1.

temperature alone is a reasonable predictor of freshwater ice phenology. As discussed earlier, air temperature is showing some significant upward trends in the Hudson Plains Ecozone (figures 6, 7, 8).

2.1.1.3 Other drivers

2.1.1.3.1 Isostatic rebound

Glooschenko and Martini (1978) estimate that in recent times 200 m of newly exposed shoreline appear every decade along the southern Hudson Bay and James Bay coastlines due to isostatic rebound. Much of this land is located in the intertidal zone, and it consists of glacial till, a rock-strewn shoreline, and the presence of marine clay. The emergence of this new land along the coast from isostatic rebound contributes to the distribution of permafrost in the ecozone. The present rate of this post-glacial rebound is between 0.90 m/century at Churchill and 1.1 m/century at Peawanuck (Sella et al. 2007). Earlier estimates were 0.7-1.2 m/century (Webber et al. 1970). The permafrost boundary between continuous and discontinuous permafrost is not a residue of the last glaciation, but rather it represents a progressive locational change over time as land emerges from the sea and permafrost develops in it. At Beech Bay near Churchill, a sill of permafrost is present under the beach, and it tapers within the current tidal zone. The sill has extended seawards since 1929, paralleling the shift of the tidal zone that results from isostatic rebound. This permafrost also cannot be considered fossil permafrost that developed before the Tyrell Sea inundation (Hansell et al. 1983).

2.1.1.3.2 River flow & sedimentation

A disproportionately large volume of water enters Hudson and James bays from overland runoff. The entire Hudson Bay Ocean Drainage Area (3.8 million km^2), which drains from several terrestrial ecozones, accounts for 30% of Canadian runoff – and includes several major river drainage areas that lie partly within the Hudson Plains Ecozone (refer back to Figure 4 in Section 1.1, *Geology, Topography & Climate*). The annual discharge rate of rivers into Hudson and James bays has been variously estimated at 20,700 m^2/s (Prinsenberg 1988) or 385 km^3/year for eastern Hudson Bay and 334 km^3/year for western Hudson Bay (Déry and Wood 2005)[9].

The large volume of freshwater that is discharged into Hudson and James bays affects the salinity of sea water there and, hence, density-driven currents. In fact, it dilutes the sea water to a salinity that is one-third that of normal oceanic water (Prinsenberg 1982). In turn, this lowered salinity contributes to the complete freeze-over of Hudson and James bays each year, which has profound effects on the climate of the Hudson Plains Ecozone (see earlier, Section 2.1.1.1.2, *Sea Ice*).

No clear trends in river flow regimes (e.g., magnitude, frequency, timing, duration, and flashiness of low and high flow events) are evident for undeveloped rivers in the Hudson Plains Ecozone over the period 1970-2005, based on only two reference gauging stations with data judged suitable for that analysis (Monk et al. in press). Trends for reduced total annual volume of freshwater naturally discharged from several of the ecozone's rivers are, however,

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[9] Mean annual river discharge rates into the broader Arctic Ocean and Hudson Bay have been estimated at 5,250 km^3/yr (Shiklomanov et al. 2000).

indicated in studies of the broader Hudson Bay ecosystem (Déry and Wood 2005; Déry et al. 2005; McClelland et al. 2006). Trends in total volume of freshwater discharged (1964 to 2000 or 2003)[10], disregarding rivers with hydroelectric development or correcting for them, are correlated with large-scale climate oscillations (Déry and Wood 2005; see also *Influence of Major Climate Oscillations*, below) and associated with a 4 day advance in annual peak discharge rate and a decline in peak intensity (Déry et al. 2005). Like elsewhere in Canada, changes in the hydrological regimes of some of the ecozone's rivers are also associated with hydroelectric developments, including water diversions (Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Large amounts of sediments enter Hudson and James bays from river discharge and are dispersed by current in an anticlockwise direction around the coastal shelf (Pelletier et al. 1968). For example, the sediment loading into James Bay from freshwater has been estimated at 4.12 X 107 tonnes annually (Kranck and Ruffman 1982), an amount that is disproportionately high for the inflow. Little of this sediment accumulates off river mouths; most of it is carried by tidal currents, long-shore drift, and ice-rafting to other locations, where it contributes to shaping coastal landforms (King and Martini 1983). An exception occurs where river flows have been substantially reduced by hydroelectric developments. For example, after 90% of the flow of the Eastmain River was diverted north to the La Grande River, the downstream current of the reduced flow segment of the Eastmain River no longer expelled fine sediments into James Bay, and its estuary became a sedimentary deposit zone (Hayeur 2001). Additional sediment was contributed by erosion of the exposed river bed. In recent decades there has also been slumping and collapse of river banks along the Hayes and Nelson rivers in the vicinity of York Factory, Manitoba (presumably associated with permafrost thaw). The collapse of the banks adds to the sediment load in these rivers that drain into Hudson Bay.

In addition to sediments being removed in river discharge, intense foraging in coastal salt marshes by a greatly increased population of lesser snow goose (*Chen caerulescens caerulescens*) has, over the last four decades, led to vegetation loss and exposure of underlying sediments along much of the ecozone's coast, from Manitoba to James Bay (Section 2.2.2.1, *Coastal*). These sediments are washed into Hudson and James bays during spring runoff, but the amount has not been quantified. Reduced surface albedo in exposed areas can accelerate permafrost thaw.

2.1.2 **Trends assessment & projected changes**

Most studies of trends in abiotic drivers and likely changes are at larger spatial scales than the Hudson Plains Ecozone, and climate projections are generally also for long temporal periods. A high degree of uncertainty therefore prevails as to the changes that will occur in this ecozone in the future. Nevertheless, it is possible to comment on trajectories of climatic and ecological change based on a growing number of studies.

2.1.2.1 **Influence of major climate oscillations**

The climate of the arctic marine environment and associated coastal lands and islands varies over a range of time scales (Walsh 2008). The annual mean warming of the arctic over the past 100 years is approximately 1°C (averaged over the polar cap north of 60 °N), but in parts

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[10] Rates have decreased 11% and 13% for eastern Hudson Bay and western Hudson Bay, respectively.

of northern North America, especially in western regions, the warming has been as much as 3 °C in the last 50 years (Walsh 2008; see also Zhang et al. in press). The recent warming, which occurs mostly in the winter, is partly attributable to systematic variations of large-scale atmospheric circulation, such as the Arctic Oscillation (AO) (Turner et al. 2006). However, it cannot be solely attributable to these oscillations and/or changes in oceanic circulation (ACIA 2004). Changes in precipitation are very difficult to assess in the north due to observational uncertainties and the absence of a well-defined network of monitoring stations.

Arctic sea level pressure has varied in recent decades due to shifts in the AO, from a generally negative phase in the 1970s to a generally positive phase in the late 1980s and early 1990s (Walsh 2008) – followed by a period of rapid transitions between positive and negative phases, including some unusually large negative amplitudes most recently (Figure 22; L'Heureux et al. 2010). Notably, the change to the generally positive phase was associated with a decrease in air pressure over the Arctic Ocean and, to some extent, the subarctic seas (Walsh 2008). The AO when positive (also the North Atlantic Oscillation, NAO, which is linked to the AO) leads to colder than normal winters over northeastern Canada, as a result of a northerly/northwesterly flow of cold air from the arctic that occurs when the Icelandic Low intensifies in winter associated with the positive phase of the NAO. The shift in atmospheric circulation over the Arctic Ocean has been linked to the redistribution of sea ice, especially a reduction in mean ice thickness in some sectors of the Arctic Ocean (Rigor et al. 2002). In subarctic regions this movement of cold air into the Hudson Bay and northeastern Canada delays the break-up of river and lake ice, peak stream flow in spring, and the onset of spring (Bonsal and Shabber in press). When the NAO is positive, these strong northwest winds move ice in summer into the southern sections of the bay, where it persists later than usual (Wang et al. 1994b; Gough et al. 2004). In 1973, 1983, and 1992, the decreased atmospheric temperature had an immediate effect on ice cover, with break-up occurring very late (Gough and Wolfe 2001) and with consequent effects on phenology in the terrestrial environment (e.g., poor reproduction in birds; Ganter and Boyd 2000). The effect of a positive AO and NAO on precipitation is associated with lower than normal precipitation in winter, but the effect is not strong. If cold, dry arctic air outbreaks occur, they result in reduced stream flows (Bonsal and Shabber in press). The most recent positive phase of the AO was associated with reduced precipitation and decreased discharge from rivers in northern Canada, including some rivers of the Hudson Plains Ecozone (Déry and Wood 2005; see also earlier, Section 2.1.1.3.2, *River Flow & Sedimentation*). ENSO might also influence ice cover in the arctic. During 12 ENSO events, considerable amounts of ice prevailed in the Arctic Ocean in summer (Wang et al. 1994b).

The recent, unusual extremes in the negative phase of the AO (Figure 22) were associated with above-average winter temperatures over much of the arctic, with the effect extending into Hudson and James bays (L'Heureux et al. 2010). Examination of the positive temperature anomalies associated with these extreme negative AO index values shows that, once the linear portion of the AO signal is accounted for, the residual temperature anomaly is still positive (indicating anomalous warmth) and most markedly so at higher latitudes. The result is interpreted as consistent with expected anthropogenic climate warming modulated by the ice-albedo feedback mechanism (L'Heureux et al. 2010). The meaning of the recent extremes in the AO is, however, not clear in context of the long-term trend in AO.

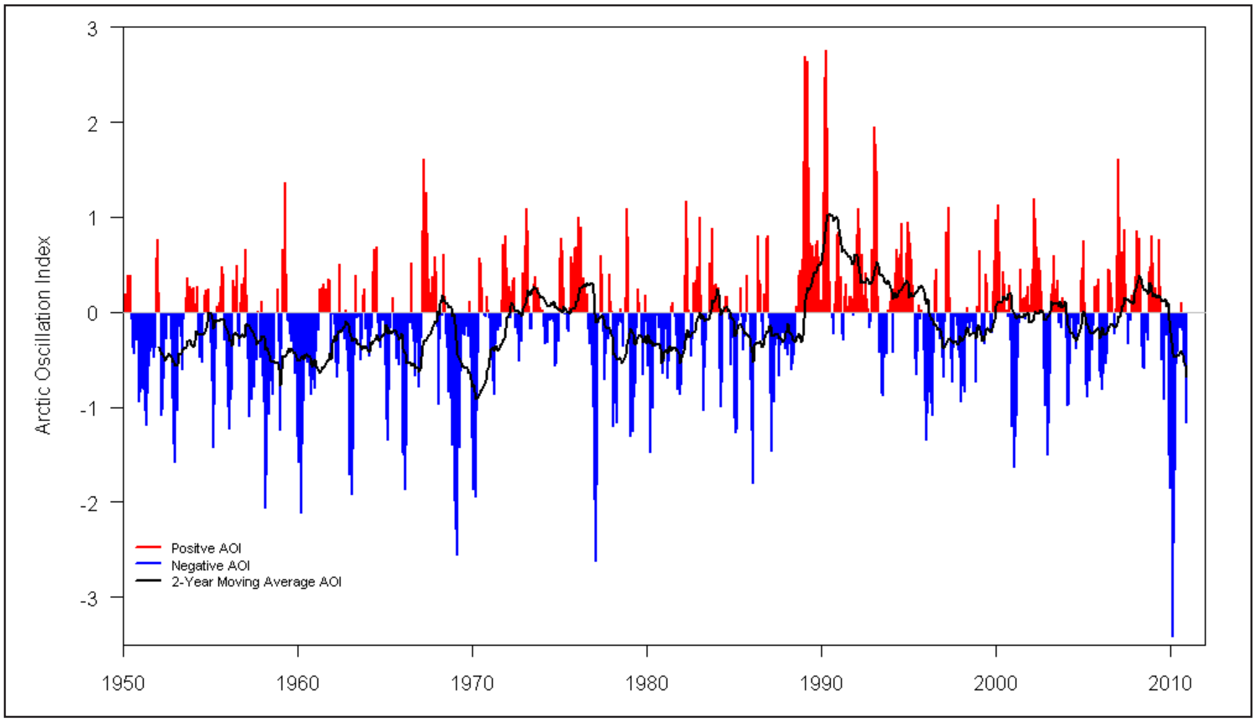

Figure 22. Arctic oscillation index (AOI), 1950-2010. The most recent negative values of the AOI are unprecedented in the 60 year record.
Source: Data from the National Weather Service, Climate Prediction Center (2010). Plot generated using a modified R-script based on original by D. Kelly O'Day (http://chartsgraphs.wordpress.com).

2.1.2.2 **Projected changes**

Climatic change is expected to be greater at higher latitudes due to the effect of changes in ice condition and longevity and feedback to the system through longer ice-free periods. Climate scenarios for the Hudson Bay region developed using various general circulation models (GCMs) and a regional climate model (RCM) project warmer temperatures over both sea and land in all seasons, with peak temperature differences generally occurring in winter (Gough and Wolfe 2001; Gagnon and Gough 2005a; Walsh 2008; Joly et al. 2010; McKenney et al. 2010). As the cover of sea ice disappears, an amplified warming of up to ~10 °C in winter (or 8 °C annual average) is projected through ice-albedo feedback effects (Gagnon and Gough 2005a), which would threaten permafrost throughout the ecozone (Gough and Wolfe 2001; Gough and Leung 2002; Gagnon and Gough 2005a).

Precipitation projections are more equivocal than those for temperature, but precipitation is also often forecast to increase. The six GCMs used by Gagnon and Gough (2005a) for the Hudson Bay region project an increase in precipitation over Hudson and James bays in all seasons owing to the poleward movement of water vapour that is associated with higher temperatures. Decreases in precipitation are, however, sometimes projected over areas of land depending on model, season, and/or specific location (e.g., Colombo et al. 2007; Bergeron et al. 2010). Regardless, conditions on land will likely be drier overall, because any increases in rainfall amount will tend to be offset by higher evaporation due to warmer temperatures (Gagnon and Gough 2005a).

Trends in sea ice are already evident (see earlier, Section 2.1.1.1.2, *Sea Ice*), and models project future scenarios of continuing loss of sea ice. The response of sea ice to climate change varies among GCMs, from a substantial lengthening of the sea ice-free season to complete loss of seasonal sea ice from James Bay and the southern portion of Hudson Bay, by 2100 (Gough and Wolfe 2001; Gagnon and Gough 2005a). Given that the Hudson Bay region is coarsely resolved in GCMs, a high resolution regional climate model of Hudson Bay and the surrounding land has been recommended to help clarify uncertainties associated with these projections (Gagnon and Gough 2005a) – particularly the impact of atmospheric forcing, which dominates in determining the response of sea ice to climate warming (Gough and Allakhverdova 1999).

Recently, a higher-resolution, regional climate model (RCM) was used to examine the sensitivity of sea ice and ocean climate in Hudson and James bays to warmer atmospheric forcing, albeit over a shorter forecasting period (Joly et al. 2010). Model projections for the period 2041-2070 indicate that downward trends in sea ice duration, volume, and cover will continue in the broader Hudson Bay marine system. The sea ice season is projected to be reduced 7-9 weeks by 2070, which is related to changes in the seasonal variability of water stratification (primarily controlled by salinity in the Hudson Bay system). Median dates of break-up are projected to occur 24 days (3.4 weeks) earlier for Hudson Bay and 39 days (5.6 weeks) earlier for James Bay, with freeze-up occurring 25-26 days (~3.7 weeks) later for both bays. Although the maximum area of sea ice cover in the Hudson Bay marine ecosystem is projected to be reduced 2.6% by 2070, maximum sea ice volume (thickness) will be reduced 31% (with the maximum being reached 2 weeks earlier). Thus, the extent of sea ice cover will be strongly reduced more specifically during freezing and melting periods. At the time of these projections (2070), the mean air temperature warming over Hudson Bay is expected to be 3.9 ºC. The greatest impacts on sea ice are forecast for James Bay and the southeast Hudson Bay coast, as well as Hudson Strait and Ungava Bay, when spatial variability in sea ice growth rates (expected to decline more than 40%) is taken into account. In that case, the cover of sea ice in James Bay might be reduced more than 50% during the winter and almost completely absent in June.

The amplified warming expected as sea ice is lost would threaten permafrost throughout the Hudson Plains Ecozone (Inset 2). The timing of any loss of continuous permafrost is unclear in response to climate warming in the Churchill region, but a lag is likely to occur as the surface air temperature rises, because of the latent heat of thawing (Gough and Leung 2002). A range of simulations shows a loss of ~50% or more of continuous permafrost and a virtual elimination of a climate that supports permafrost, by 2100 (Gough and Leung 2002). In the southern sectors of the Hudson Plains Ecozone, permafrost that is discontinuous or in isolated patches is projected to disappear altogether (Gagnon and Gough 2005a).

Because the ecozone's defining climatic and edaphic conditions are a result of sea ice and permafrost, cascading effects on the ecozone are expected as sea ice and permafrost are lost. Species dependent on sea ice are at most immediate risk, with current trends for deterioration in polar bear subpopulations expected to continue or accelerate (Stirling and Parkinson 2006; Obbard et al. 2007; Molnár et al. 2010; Section 2.3.3.1.1, *Polar Bear*). The presence of more open sea water increases the likelihood of increased wave action (coastal erosion) and storm surges, which could result in more frequent inundation events inland above the mean high watermark of spring tides (Martini et al. 2009). Conversely, sea level rise is less of a concern for this ecozone than for some other coastal areas of Canada (Tsuji et al. 2009), owing to an especially high rate of isostatic rebound (Sella et al. 2007). In the marine environment, polynyas are likely to be affected (Niemi et al. 2010), including the one located south of Akimiski Island (Martini and Protz 1981).

Inset 2. **Nature & fate of permafrost in the Hudson Plains Ecozone**

William A. Gough, University of Toronto

A full range of permafrost types are found in the Hudson Plains Ecozone (Figure 23). At the northern extent of the ecozone along the Hudson Bay coast, continuous permafrost can be found beneath coastal ridges and wetlands. Inland, in as little as 20 km in some areas, the terrain changes to palsas, localized geomorphic mounds indicative of a transition from continuous to discontinuous permafrost. Sporadic discontinuous permafrost and isolated patches of permafrost are found further south, while permafrost is absent in the most southerly reaches of the ecozone in areas away from the coast.

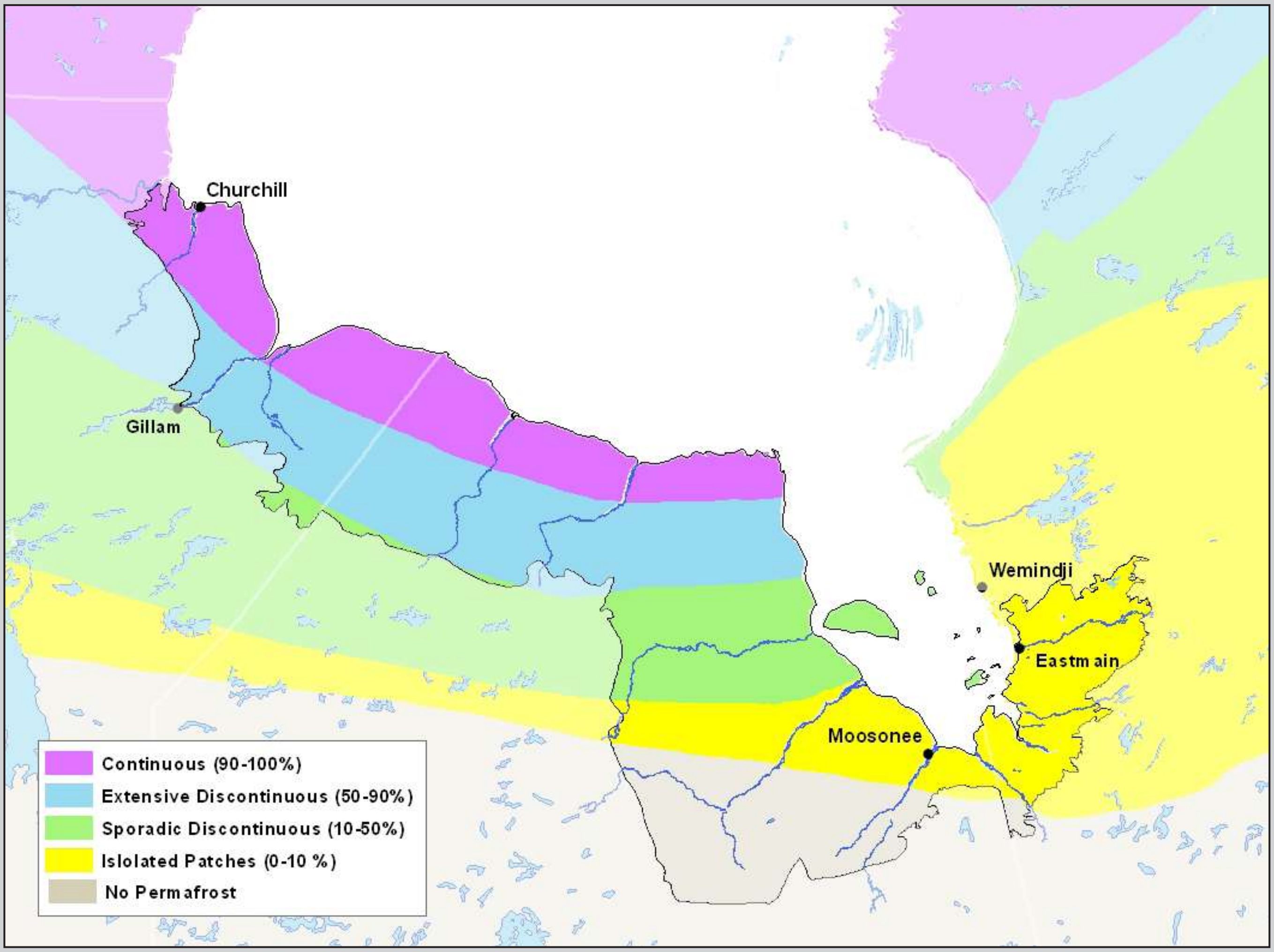

Figure 23. Permafrost zones in and around the Hudson Plains Ecozone. Ecozone boundaries correspond to the ecozones⁺ framework.
Source: Heginbottom et al. (1995). Copyright Department of Natural Resources Canada. All rights reserved.

The Hudson Plains Ecozone hosts the most southerly continuous permafrost in Northern America (Gough and Leung 2002; Zhang et al. 2008). The presence of permafrost in this ecozone is supported by a unique combination of local sea ice distribution and soil moisture characteristics (Gough and Leung 2002). Hudson and James bays experience seasonal sea ice, with long-lasting sea ice occurring along the Hudson Plains coast – a result of Hudson Bay surface flow and the prevailing wind (Gough et al. 2004). The presence of sea ice leads to what one researcher calls the *winterization of summer* along the southwestern Hudson Bay coast and to substantially cooler conditions in this ecozone compared to other locations at the same latitude (Rouse 1991). The mean annual temperature along the Hudson Bay coast (as measured in Peawanuck and Fort Severn, Ontario) is around -5°C, which is a little higher than the -7°C threshold typically found for continuous permafrost. The high soil moisture content there accounts for the differential. Soil conductivity increases with increasing water content, and it dramatically increases when soil moisture freezes. With the higher soil conductivity of frozen soil, the penetration of frost in winter is amplified compared to symmetric warming or thawing in the summer. Thus, permafrost conditions can be maintained there with a higher mean annual temperature (Gough and Leung 2002).

The future of permafrost in the Hudson Plains Ecozone is thus dependent on the impact of a changing climate on these processes. Climate model simulations of the Hudson Bay region show an amplified warming compared to other regions in Canada (Gough and Wolfe 2001; Gagnon and Gough 2005a). This amplified warming is closely linked to the loss of seasonal sea ice which, depending on the model, is projected to be greatly reduced or to disappear altogether by 2100 in James Bay and the southern portions of Hudson Bay (i.e., areas adjacent to the Hudson Plains Ecozone). Changes in sea ice conditions have already been detected in James Bay, southern Hudson Bay, and western Hudson Bay, with break-up occurring earlier by 0.49-1.25 days/year, depending on location (Stirling et al. 1999; Gough et al. 2004; Gagnon and Gough 2005b). As sea ice disappears, the winterization effect along the southwestern Hudson Bay coast also will disappear. An amplified warming over the bays of up to ~10°C in winter is projected (Gagnon and Gough 2005a), which will threaten permafrost throughout the ecozone. Frost penetration will abate, and the thawing depth will increase and soon exceed the freezing depth. It is possible that a remnant layer of permafrost might persist below this for some decades, as the deep soil slowly adjusts to the new climate conditions. Geomorphic features such as palsas will degrade and eventually disappear, which could substantially impact polar bear habitat (Stirling and Derocher 1993; Derocher et al. 2004; Obbard et al. 2006). Permafrost thaw also has other implications to the ecology of the Hudson Plains Ecozone, as described throughout this report. Modelling work that has specifically examined permafrost in the Hudson Bay region projects the loss of over 50% of the continuous permafrost (and complete loss of permafrost that is currently discontinuous and in isolated patches) and a virtual elimination of a climate that supports permafrost, by 2100 (Gough and Leung 2002; Gagnon and Gough 2005a).

Sufficient data are not currently available for evaluating trends in the extent and condition of permafrost in the Hudson Plains Ecozone or associated shifts in permafrost boundaries. Until relatively recently, no permafrost thermal monitoring sites were located and maintained there to help track changes, as is being done elsewhere in Canada's north (Smith et al. 2005; Smith in press). Ten-year data are now available for a permafrost site at Churchill, Manitoba (Kershaw 2010), a new permafrost monitoring site was established in 2007 at York Factory, Manitoba (Sladen et al. 2009), and two more sites were recently added in northern and southern areas (as coast-to-inland transects) of Wapusk National Park, Manitoba (H.M. Stewart, Wapusk National Park, pers. comm.). In Ontario, annual summer monitoring of permafrost began in 2007, and a permanent monitoring site (Brant River) is now in place.

Some changes in permafrost are, however, suspected in this ecozone. Both collapse and erosion features and aggrading features are visible in the ecozone's permafrost tension zone, and collapse features appear to have become more widespread over time, as in the Ekwan to Lake River areas of the northern James Bay coast (Riley 2003). Casual observations have also been made in recent decades of river bank slumping and collapse along the Hayes and Nelson rivers in the vicinity of York Factory, close to the boundary between continuous and discontinuous permafrost. Likewise, some evidence suggests a long-term (non-successional) change or trend for partial degradation and conversion of frozen peat plateaus to fens in the area from the Nelson River north to Churchill, as well as the enlargement of some associated lakes from eroding shorelines (Dyke and Sladen 2010). Moreover, while the relatively short 10 year permafrost record from Churchill shows no significant trend to date, comparison of this data with the much longer climate record at Churchill suggests that the air temperature warming there (see earlier, figures 7 and 8) might have resulted in permafrost warming of ~0.5 °C, since the mid-1970s (Kershaw 2010). Indeed, permafrost loss is known to be occurring just outside both western (Manitoba) and eastern (Québec) ecozone boundaries, in areas where permafrost is currently classified as being discontinuous and in isolated patches, respectively (Camill 2005; Thibault and Payette 2009).

In more northerly areas, thawing of permafrost is initially expected to collapse the peat, raise the water table, and form ponds (Gorham 1991). Conversely, vegetative communities in more southerly areas, where permafrost is currently more limited, might shift to shrub- and tree-dominated communities, as thawed peatlands become progressively more xeric as water tables decline (Camill 1999; Camill et al. 2001). Two of the coupled atmosphere-ocean models (CGCM1 and CSIRO) show a summer warming peak over land surrounding Hudson Bay that is associated with the loss of soil surface moisture and changes in permafrost distribution (Gagnon and Gough 2005a). At the time of a doubling of CO_2, soil moisture is projected to have decreased up to 2 cm from May to November. The loss of surface soil moisture is projected to peak in June, and south of Hudson Bay it is likely to reach 10% of the total surface soil moisture.

Increases in runoff are likely to affect water levels in rivers. River discharge rates into the Arctic Ocean are projected to increase by 2100 in response to climatic warming (Walsh et al. 2005). The seasonality of river flows are also likely to be affected, with lower spring flows from reduced or earlier snow melt and, conversely, higher winter flows (Bates et al. 2008; see also Déry et al. 2005). If the rise in water levels in rivers is associated with ice flow build-up, hydrological extremes might be anticipated with extensive flooding of the riparian hinterlands (Prowse and Beltanos 2002). Increases in runoff might also affect inshore salinities and nutrient fluxes in the marine environment, which could increase biological production there (Niemi et al. 2010).

As elsewhere, climate change is also expected to alter species' ranges and assemblages in both freshwater and terrestrial environments (e.g., Minns and Moore 1995; McKenney et al. 2007). Disturbances due to extreme weather, fire, and other agents are also likely to increase (Section 2.4.2, *Natural Disturbances).* Thresholds could be exceeded, and novel ecosystems are likely to develop (Juday et al. 2005).

Northern peatlands are actively involved in the cycling of carbon at the global scale (Moore et al. 1998; Roulet et al. 2007). The fate of the large carbon store in peatlands is of concern given the likely changes in climatic conditions and the potential for the release of this carbon to the atmosphere, which might lead to a positive feedback to atmospheric greenhouse gases (Schuur et al. 2009). Few measurements have been made of contemporary carbon exchange in northern peatlands, and considerable uncertainty exists in our ability to project the fate of carbon there in the coming decades (Roulet et al. 2007). Robust measurements of carbon fluxes are needed in northern peatlands, including the extensive peatlands of the Hudson Plains Ecozone. Carbon cycling in the ecozone is discussed further in Section 2.4.4.

Clearly, the Hudson Plains Ecozone is closely tied to the inland sea to its north. Future changes in the longevity and condition of sea ice due to climate warming will not only affect Hudson and James bays and species like polar bear that depend on sea ice as primary habitat, but it will also irreversibly change the land that surrounds it. Reductions in sea ice are already ahead of the projected rate of the summer retreat of the ice by nearly all of the GCMs used by the Arctic Climate Impact Assessment (ACIA 2005) and the fourth assessment of the Intergovernmental Panel on Climate Change (IPCC 2007) (e.g., Stroeve et al. 2007; Allison et al. 2009). To track changes adequately, more monitoring stations need to be established and maintained in the north to detect fluctuations and future trends in climate, ice conditions, permafrost, and water levels.

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2.2 Ecosystem structure

2.2.1 Overview of ecozone structure & land cover change

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2.2.1.1 Overview of ecozone structure

The Hudson Plains Ecozone is edaphically the wettest ecozone in Canada (NWWG 1988) and the third largest wetland complex in the world (Fraser and Keddy 2005). The unique structure of this ecozone is determined by: its flat-laying geological structure; poorly drained marine silts and clays, exacerbated by permafrost associated with organic soils; immature drainage development (lateral and downgrade surface flows); a relatively cold climate; and isostatic rebound. These conditions have resulted in an extensive flat plain characterized by a myriad of small shallow lakes, ponds, and creeks within a complex of different types of poorly drained peatlands (Coombs 1952, 1954; Brokx 1965; Wiken 1986; Wickware and Rubec 1989; Smith et al. 1998; Riley 2003; Crins et al. 2009) (Figure 24). A few deeper lakes are associated with the Sutton Ridges[11] southwest of Cape Henrietta Maria, Ontario (Crins et al. 2009). As well, a number of the large rivers have their headwaters in the Precambrian Shield, and they are fed by streams that generally have gentle gradients and slow-moving flows (Wiken et al. 1996).

Overall, the ecozone is comprised of four main biome types (sensu ESTR framework; Frisk in press): coastal, tundra, forests, and inland waters. The coastal biome (tidal flats, salt marshes, shallow waters) occurs along the coastal margins of Hudson and James bays, and the tundra

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[11] The Sutton Ridges (Figure 3 in Section 1.1, *Geology, Topography & Climate*) “comprise a distinctive landscape of intermittent Precambrian ridges, cuestas, and boulder pavements, with prominently developed cliffs, scarps, and columnar jointing” (Riley 2003).

biome occurs only in the northerly portion of the ecozone adjacent to the Hudson Bay coast. The remainder of the ecozone is a complex of slow-growing boreal forests (including transitional taiga forests) and inland waters (freshwater wetlands, lakes, and rivers/streams). Status and trends of individual biome types are reviewed later in Section 2.2.2.

Figure 24. Landscape photograph illustrating the nature of the Hudson Plains Ecozone as a wetland complex. Near Niskibi River, Ontario, ~40 km inland from Hudson Bay.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Peatlands (bogs and fens) formed through primary peatland formation (Kuhry and Turunen 2006) predominate in this ecozone, with organic matter accumulation generally being thinnest in recently emerged areas along the coast and progressively increasing in thickness inland (as peat accumulation exceeds decomposition), reflecting the length of time since deglaciation and emergence from the sea. A generalized peat thickness map for northeastern Manitoba illustrates this pattern in the northwest portion of the ecozone (Figure 25). Likewise, a gradient of peat depth was demonstrated in the Albany River area of Ontario, with mean peat depth increasing from 148 cm at a distance of 80 km from the coast (n=3) to 414 cm at a distance 250 km from the coast (n=3) ($r=0.80$, $p <0.001$; n=22; Glaser et al. 2004). These organic deposits overlay mineral materials, such as marine clay and silt deposited in the post-glacial Tyrell Sea, associated beach deposits of sand and gravel, tills of various texture, and bedrock (Riley 2003). Several subgroups of the soil great groups Mesisol and Fibrisol dominate the peatlands, with primary differences being organic decomposition rates (Cowell et al. 1991).

Upland features of this ecozone include: the Sutton Ridges in Ontario (Figure 3 in Section 1.1, *Geology, Topography & Climate*); a low bedrock ridge at Churchill, Manitoba (Figure 26 below); other areas of bedrock outcrop (Figure 27); drumlinized and fluted till plains; beach ridge complexes (Inset 3, figures 28 and 29); and levees or abandoned deltaic deposits along major rivers. Mineral soil profiles have developed on these upland features, some of which support tundra, lichen woodlands, and conifer forests (Riley 2003). Isolated, well-drained sands and gravels of uplands that are located inland from the coast are dominated by Regosols and Podzols. However, Brunisols are found on river levees, and Gleysols are present on imperfectly drained mineral sites (Riley 2003).

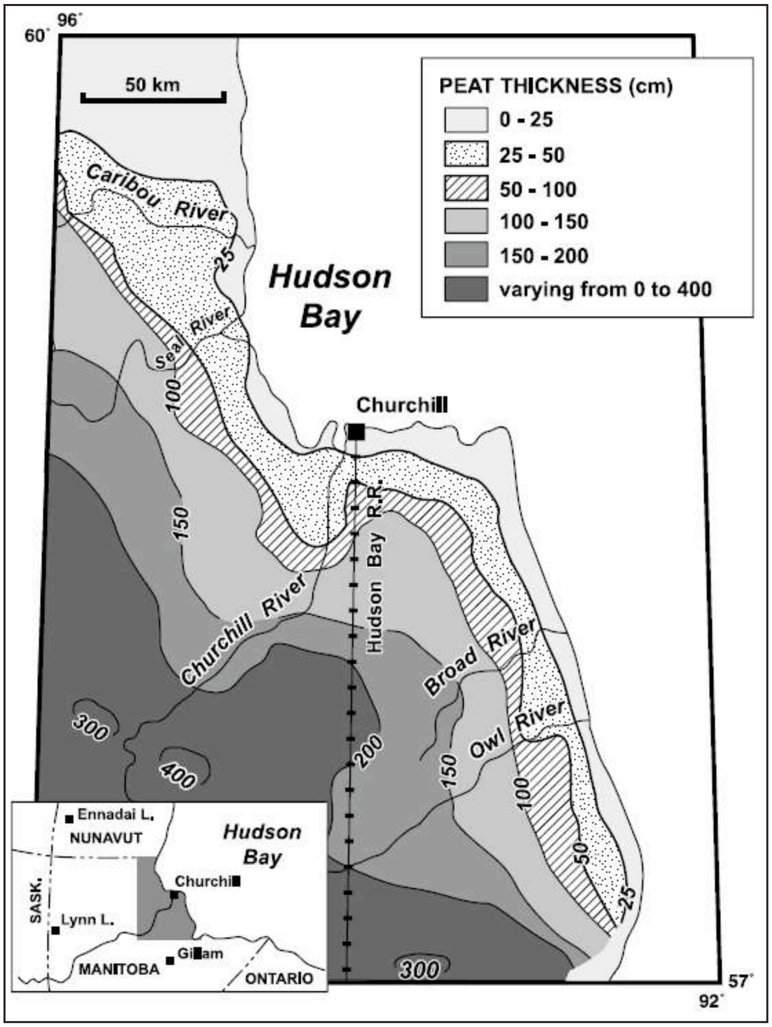

Figure 25. Generalized peat thickness map for northeastern Manitoba, illustrating increasing depth of peat with distance away from the Hudson Bay coast.
Source: Dredge and Mott (2003), based on data from 400 sites. Reprinted from Géographie physique et Quaternaire, Vol 57, L.A. Dredge and R.J. Mott, Holocene pollen records and peatland development, northeastern Manitoba, page 9, Copyright (2003). Reproduced with permission, under license from Copibec.

Figure 26. Exposed bedrock, ~6 km east of Churchill, along the coastal ridge leading east toward the Churchill Northern Studies Centre. Photo credit: R.F. Rockwell, American Museum of Natural History.

Figure 27. Bedrock outcrops in the southern portion of the ecozone, ~50 km southeast of Moosonee and 5 km east of Partridge River (escarpment transition area), Ontario.
Photo credit: S. Brinker, Ontario Ministry of Natural Resources.

Inset 3. **Beach ridges**

Sandy and gravely *beach ridges* are a defining feature of the Hudson Plains Ecozone, occurring parallel to the main coast throughout (Smith et al. 1998; Crins et al. 2009). The ridges start developing in the upper intertidal zone from cross-bedded sand and gravel with some capping plane beds (Martini 2006), and they continue to accumulate reworked sediments contributed by coastal building processes, such as wave and tidal regimes (see Section 2.4.1, *Coastal Building Processes*). Isostatic rebound leads to beach ridges varying in height and spacing (Martini et al. 2009) and becoming progressively colonized by vegetation (successionally, from halophytic dicots, to grasses, to shrubs, and then trees) as the distance of ridges from the coast increases (Martini 2006). Overall, beach ridges in the ecozone support a diversity of vegetation communities, including tundra, lichen woodlands, conifer forest, swamp, and treed bog, depending on latitude and distance from the coast (Riley 2003). The same community types occur on beach ridges along both Hudson Bay and James Bay coasts, but both the width and diversity of vegetation types are often truncated on the more exposed coasts of Hudson Bay (Riley 2003). Beach ridges are especially prominent along the Hudson Bay coast between the Hayes River in Manitoba and the Winisk River in Ontario, where the regional land gradient is greatest.

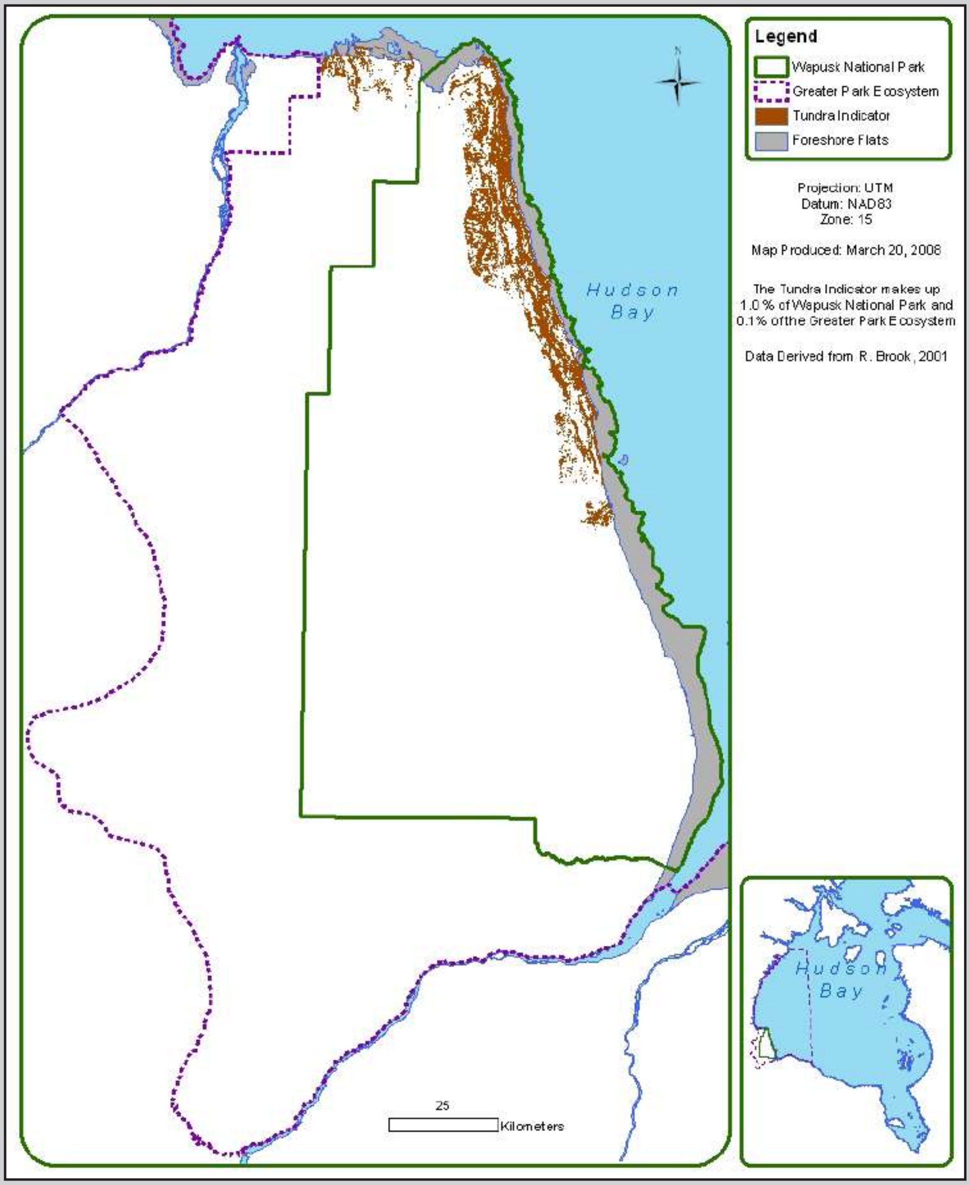

Figure 28. Coastal beach ridges vegetated by Dryas *Heath Upland (tundra) communities (illustrated with brown colourations) in and around Wapusk National Park, Manitoba. More mature, forested and wetland beach ridges are found further inland.*
Source: Map derived from Brook (2001).

The prominent beach ridges that parallel the Hudson Bay coast at Wapusk National Park, which are vegetated by *Dryas* Heath Upland (tundra) communities along the coast (Figure 28) and can be treed farther inland, are the subject of much current study. Work is ongoing to characterize the species present (Piercey-Normore 2005) and the effects of trampling on these unique and fragile vegetation communities (Elliott 2009). These beach ridges offered a north-south travel corridor for early people and wildlife along the coast, and this use continues today. Indeed, most cultural artefacts in Wapusk National Park are located on beach ridges, which now also offer dry places to set up visitor and research facilities. Although this type of beach ridge has homologues across the ecozone, it diminishes in extent east of the Winisk River and inland, while successionally vegetated, older types of beach ridge become more prominent, i.e., those covered with shrub but mostly spruce-lichen woodland.

Figure 29. Beach ridge complex, ~3 km inland from the Hudson Bay coast near the Blackcurrant River, Ontario.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

2.2.1.1.1 **Land cover**

A detailed, accurate, and precise inventory and map of land cover (including open freshwater) at a local scale (<1:10,000 scale) or regional scale (1:10,000 to 1:50,000 scale) is currently not available for the Hudson Plains Ecozone as a whole (i.e., using a common classification). Available land cover information is presented in this section as part of the account of the state of the knowledge for this ecozone, but only very general statements about ecosystem structure can be made based on these sources.

Most available information sources use fairly coarse measurements to suggest that open freshwater features account for <10% of the Hudson Plains Ecozone. Based on limited sampling with 2 km x 2 km photographic plots, the National Forest Inventory (NFI 2010) reports that, overall, large freshwater features (larger lakes and rivers) account for 7.6% of the total area of this ecozone, compared to 15.9% and 20.8% of the adjacent Boreal Shield and Taiga Shield ecozones, respectively (Table 2). Others have used different methods to estimate that the proportion of resolved freshwater features ranges from 3.0 to 8.2% (Table 3). However, areas of the Hudson Plains Ecozone might have much higher cover of freshwater than these various <10% estimates suggest, due to the myriad of small shallow lakes, ponds, and creeks that are not captured in either broad-level satellite imagery or small-scale maps. Indeed, when an area of treeless open bog (with peat plateau as a common permafrost terrain feature) in the vicinity of Fletcher Lake south of Churchill (inland) was digitized from aerial photographs at a finer 1:25,000 scale, open water comprised 41% of the ground surface (Bello and Smith 1990; Table 3). Another study in permafrost terrain near Churchill that used Landsat-5 TM data reported that shallow lakes and ponds occupied 32%, 24%, and 7% of tundra, forest-tundra, and open forest, respectively (Lafleur et al. 1997).

Table 2. Area of land and freshwater in the Hudson Plains Ecozone in 2006 relative to adjacent terrestrial ecozones and Canada as a whole. The freshwater category in this analysis represents larger freshwater features (larger lakes and rivers). This analysis is based on the ecozones⁺ framework.
Source: NFI (2010).

Terrestrial ecozones	Land		Freshwater		Not classified		Total area
	Area (million ha)	% of total area	Area (million ha)	% of total area	Area (million ha)	% of total area	
Hudson Plains	32.8	92.4	2.7	7.6	0.0	0.0	35.5
Boreal Shield	155.4	84.1	29.5	15.9	0.0	0.0	184.9
Taiga Shield	109.3	79.2	29.7	20.8	0.0	0.0	138.0
Canada	650.5	87.9	89.8	12.1	<0.1	<0.1	740.3

Table 3. Estimated proportion of freshwater in the Hudson Plains Ecozone using various methods.
Sources: As shown.

Area (%)	Source	Method
3.0	Rouse (1973)	Proportion of open water surfaces measured from a rough estimate using a planimeter on the 1:633,600 scale (1 inch: 10 miles) map produced by Brokx (1965) & modified by Bates and Simkin (1969) for the northern section of the Hudson Plains Ecozone in Ontario.
4.1	Cox (1978)	Proportion of lakes and ponds >10 ha measured from a census of lakes & ponds in the Ontario portion of the Hudson Plains Ecozone using a dot grid or planimeter technology on 1:250,000 national topographic maps. Only watershed units 90% within the ecozone were used to calculate the % area.
6.7	OMNR (2010a) (map & data presented later as Figure 32 & Table 6)	Proportion of open water in the entire Ontario portion of the Hudson Plains Ecozone estimated from Landsat images (30 m resolution satellite imagery) acquired between 1999 and 2002.
7.6	NFI (2010)	Proportion of the entire Hudson Plains Ecozone (ecozones⁺ framework) covered with large freshwater features (larger lakes and rivers) estimated using limited sampling with 2 km x 2 km photographic plots.
8.2	Roulet et al. (1994)	Proportion of cover classified as water measured using Landsat Thematic Mapper digital satellite image with pixel size of 30 m x 30 m for a 4,800 km² area in southern James Bay & a 900 km² area along the southwestern shore of Hudson Bay.
8.2	Brook (2005) (map & data are presented later as Figure 31 & Table 5)	Proportion of open water in the 23,420 km² Greater Wapusk Ecosystem (i.e., Wapusk National Park & surrounding watershed) in Manitoba estimated from Landsat satellite imagery, circa 2001.

Table 3, Cont.

Area (%)	Source	Method
32 tundra; 24 forest-tundra; 7 open forest	Lafleur et al. (1997)	Proportion of cover classified as lakes (generally small lakes and ponds) along a transect in permafrost terrain from tundra through forest-tundra transition to open forest about 15 km inland, near Churchill, Manitoba, as determined using late summer Landsat-5 Thematic Mapper (TM) imagery in relatively dry (1984) and wet (1991) years.
41	Bello & Smith (1990)	Proportion of ground surface in open water estimated for an area of treeless open bog (with peat plateau as a permafrost terrain feature) in the vicinity of Fletcher Lake (south of Churchill, Manitoba, inland from the coast) by digitizing a 1:25,000 scale aerial photograph.

Coarse (1 km) spatial resolution satellite remote-sensing, which does not differentiate water from the background class (and is thus not represented in Table 3), was used to analyze land cover similarly for all ecozones across Canada, in support of national ESTR assessment (Ahern et al. in press). The associated land cover analysis for the Hudson Plains Ecozone is shown below. Table 4 shows the area of the Hudson Plains Ecozone that occurs in different broad land cover classes in 2005, and Figure 30 shows the distribution of this cover at the same time.

Table 4. Area of the Hudson Plains Ecozone in each major land cover class in 2005. Data were generated from coarse (1 km) spatial resolution satellite remote-sensing using the advanced high-resolution radiometer (AVHRR) sensor. The original analysis was based on 31 land cover classes with a thematic resolution of 12 classes (Latifovic and Pouliot 2005); for national ESTR purposes these classes were further grouped into those shown here. This analysis is based on the ecozones⁺ framework.
Source: Data provided by authors of Ahern et al. (in press).

Land cover class	Description	Area, km^2	% of total area, excluding water
Forest[a]	Areas where tree crown density is >10%. Includes conifer forest (>80% needleleaf trees), broadleaf forest (>80% broadleaf trees), & mixed forests (20-80% of tree species either needleleaf or broadleaf).	229,518	67.9
Fire Scars[b]	Predominantly new (<5 years) & older (>5 years but not yet revegetated) burns that are expected to regenerate.	5,350	1.6
Shrubland	Areas with tree crown density <10% & shrub cover >40%. Includes most wetlands & other large areas of low forest productivity.	87,991	26.0
Grassland	Areas with tree or shrub cover <10% & herbaceous vegetation present.	0	0.0
Low Vegetation & Barren	Areas vegetated with lichens, heather, herbs, shrubs (<40%), bare soils, or rock outcrops.	14,917	4.4

Table 4, Cont.

Land cover class	Description	Area, km²	% of total area, excluding water
Agricultural Land	Includes cropland and cropland/woodland. Cropland is land covered with typically annual herbaceous crops & might contain <10% shrubs or trees (includes low, medium, & high biomass crops). Cropland/woodland is a mix of cropland, forest, shrubland, grassland, or built-up areas in which no one component comprises >70% area of the landscape.	7	2.1
Urban[c]	Land covered by buildings & other artificial structures (confusion with other non-vegetated classes might occur for small urban areas).	0	0.0
Snow / Ice / Glacier	Land covered with permanent ice or snow.	0	0.0
Total[d]		337,783	~100

[a] For the Hudson Plains Ecozone, needleleaf forest density was overestimated due to many small water bodies leading to low reflectance. The medium-density forest cover in this area was relabelled as low density forest. All forest types are, however, lumped together to form the forest category identified here.

[b] Although not necessarily relevant to the Hudson Plains Ecozone in particular, this land cover class could include larger areas of harvest, mining, or severe insect defoliation, if present. The resolution of the data was not fine enough to pick-up smaller disturbances, such as clearcuts or the spectral signature from insect disturbances with moderate defoliation, if present.

[c] The analysis was too coarse to resolve urban areas in the Hudson Plains Ecozone.

[d] The total land area of 337,783 km² does not represent the total area of the Hudson Plains Ecozone, because only a portion of inland waters (land covered with water in liquid form) is included in these land cover classes. Under the ESTR framework, the inland waters category is a composite of wetlands, rivers/streams, lakes, and reservoirs. Most wetlands are included in the shrubland class in this table, but large rivers/streams and lakes are not represented (the remote-sensing product did not include a distinct class for water, which was not differentiated from the background class; the water shown in the corresponding Figure 30 is instead from the National Atlas of Canada). The total area of the ecozone in this analysis is 352,980 km² (area based on the ecozones⁺ framework).

Unfortunately, the ecozone's land cover is not well represented by this coarse-scale analysis for the ESTR. The analysis is not associated with a high degree of accuracy in part due to the large pixel size used (Ahern et al. in press)[12]. The land cover classes are also very broad (e.g., there are no distinct wetland classes) and, for the Hudson Plains Ecozone, errors in represented classes are apparent, particularly for tundra and forest. The map does not show tundra in areas where it should be, such as in the Cape Henrietta Maria triangle, the Ontario-Manitoba border, and Cape Churchill. The map also erroneously suggests that there is relatively more forest cover in the northern portion of the ecozone than in the southern portion, e.g., compare the land cover map in Figure 30 with the EOSD moderate 30 m spatial resolution Landsat map of forested versus

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[12] Accuracy of the original land cover product (1985-2000) was 61.5% overall with a Kappa of 0.56 and 44.0% for reclassified areas identified in the change detection process (Latifovic and Pouliot 2005). These accuracy values are considered likely underestimates for the land cover analysis used in ESTR (1985-2005), because the original accuracy assessment mainly used areas with high land cover variability and was compared to the original 12-class product, not the simplified product used in ESTR (Ahern et al. in press).

non-forested cover in Figure 56a, Section 2.2.2.3, *Forests (Boreal)*, which similarly classifies forests as >10% tree crown cover but instead shows forest cover increasing from north to south (see also the description of forest density in the southerly James Bay Lowland Ecoregion later in Section 2.2.1.1.3, *Intra-Ecozone Variation: Ecozones*).

The ESTR-commissioned land cover analysis in Table 4 and Figure 30 is, therefore, not very useful for accurately describing variation in land cover within the Hudson Plains Ecozone, and it should not be used for that purpose. The source national land cover product (Latifovic and Pouliot 2005) that uses the same coarse (1 km) spatial resolution satellite remote-sensing but additional land cover classes (classes were rolled-up for ESTR purposes) necessarily also has limitations for describing land cover in this particular ecozone. From a national perspective,

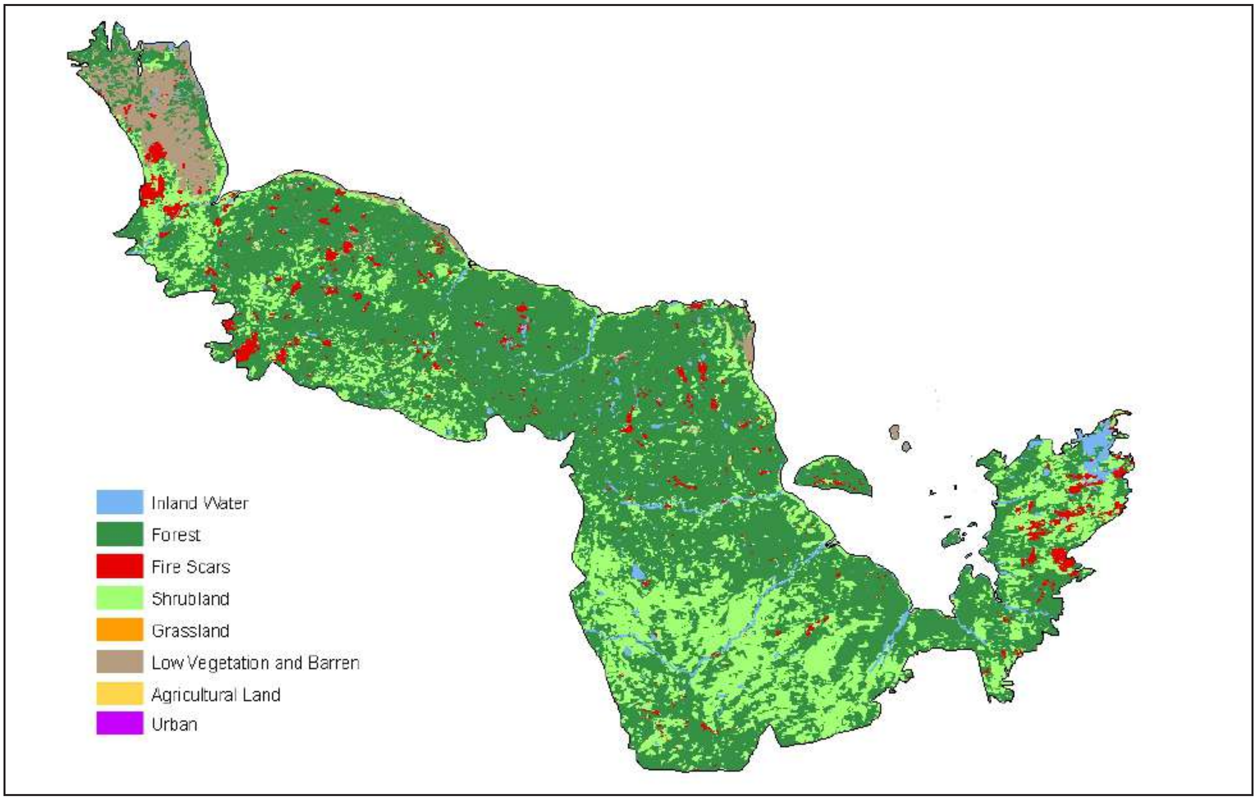

Figure 30. Land cover in the Hudson Plains Ecozone in 2005 based on coarse (1 km) spatial resolution satellite remote-sensing using the advanced high-resolution radiometer (AVHRR) sensor. The original analysis was based on 31 land cover classes with a thematic resolution of 12 classes (Latifovic and Pouliot 2005); for national ESTR purposes these classes were further grouped into those shown here. Water was not differentiated from the background class; the portion of inland water illustrated here is the 1996-1997 water status from the National Atlas of Canada (1995 satellite imagery from 1 km resolution AVHRR sensor). Forest is defined as areas where tree crown density is ~ >10% (definition does not include a minimum area and/or minimum height), and most of the area classified as forest is likely close to the lower tree crown density limit. Other land cover classes are defined in Table 4. Land areas in the grassland, agricultural land, and urban classes are very small (together <0.01%). Although this land cover analysis was done similarly for all of Canada's ecozones in support of ESTR assessment, it is not very accurate for describing variation in land cover within the Hudson Plains Ecozone and should not be used for that purpose (see text). This analysis is based on the ecozones⁺ framework.
Source: Data provided by authors of Ahern et al. (in press).

what this coarse resolution land cover analysis for ESTR (Table 4, Figure 30) does illustrate is that, compared to other ecozones in Canada (for which the same land cover analysis was done), four broad land cover classes are important in the Hudson Plains Ecozone: low vegetation and barren (4.4%), forest (67.9%), fire scars (1.6%), and shrubland (includes wetlands) (26.0%), whereas grassland, agricultural land, urban, and snow/ice/glacier classes (together representing <0.01% of the area) are not. Note that *forest* in this analysis is defined as areas where tree crown density is ~ >10% (definition does not include a minimum area and/or minimum height), and most of the area in the ecozone that is classified as forest is likely to be close to this lower limit (often characterized as wetlands in other work).

Assessment of ecozone-wide land cover is now possible via a moderate (250 m) resolution MODIS (MODerate Resolution Imaging Spectroradiometer) land cover product with additional land cover classes, but the error associated with it is also quite high (CEC 2010)[13], especially for a class such as wetlands in the Hudson Plains Ecozone (underestimated). As well, MODIS does not extend far enough back in time to allow for assessment of meaningful changes or trends in land cover (see Section 2.2.1.2.1, *Changes in Land Cover*). Clearly, a more accurate and precise inventory is required for the ecozone as a whole. Some finer land cover products do exist at sub-ecozone scales, and an improved land cover product is in progress for the large part of the ecozone that lies in Ontario (see below).

Finer analyses of land cover have been done for the Manitoba and Ontario portions of the ecozone but using different land cover classification frameworks. In Manitoba, Ritchie (1962) initiated regional mapping of the northern part of the province using black and white air photo interpretation. Brook (2001) later used 1996 Landsat-5 TM and extensive field data collected from 300 ground plots (Brook and Kenkel 2002) to map classes in Wapusk National Park. Based on a comparison of this data with 200 additional plots collected but kept out of the initial classification, the overall map accuracy was 97%, although accuracy varied from 88% to 100% among individual classes (Brook 2001). The dominant classes in the park are all wetland types: sedge bulrush poor fen (18.1%), lichen melt pond bog (17.4%), and lichen spruce bog (16.7%). Additional mapping work was completed within the Greater Wapusk Ecosystem, where the same classes were mapped, giving a finer resolution to the mapping for the national park as required for management activities. More recently, Brook (2005) updated the burn areas in the 2001 report (Brook 2001) by using Landsat satellite imagery obtained in summer 1996 and 2001. This latter work thus updates the vegetation map for the Wapusk National Park and Greater Wapusk Ecosystem using fires mapped for the years 1997 to 2001 (Figure 31, Table 5). Note that when this land cover map (Figure 31) is compared with the national mapping exercise (Figure 30), the map for the Wapusk National Park area better differentiates wetlands and better resolves tundra communities, i.e., it shows the tundra that is known to occur adjacent to the coast (Smith et al. 1998). Work is in progress to map the park's ecosystems using high resolution Quickbird satellite imagery along with EcognitionTM software (field work from this latter study has confirmed the composition and distribution of ecological units developed by Brook 2001; a description of the new project is available in Parks Canada 2008).

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[13] For the primary label only, overall accuracy was nearly 59% with a Kappa of 0.54 (CEC 2010). Overall accuracy increased to 69% and the Kappa increased to 0.65 when considering either the primary or secondary label as being correct. For primary labels interpreted with high confidence, representing 365 of 838 reference samples, overall accuracy increased again to 75% with a Kappa of 0.72. The increase in classification accuracy with high reference-data interpretation confidence was a function of greater land cover homogeneity within the reference data footprint and less ambiguity largely due to reference image quality.

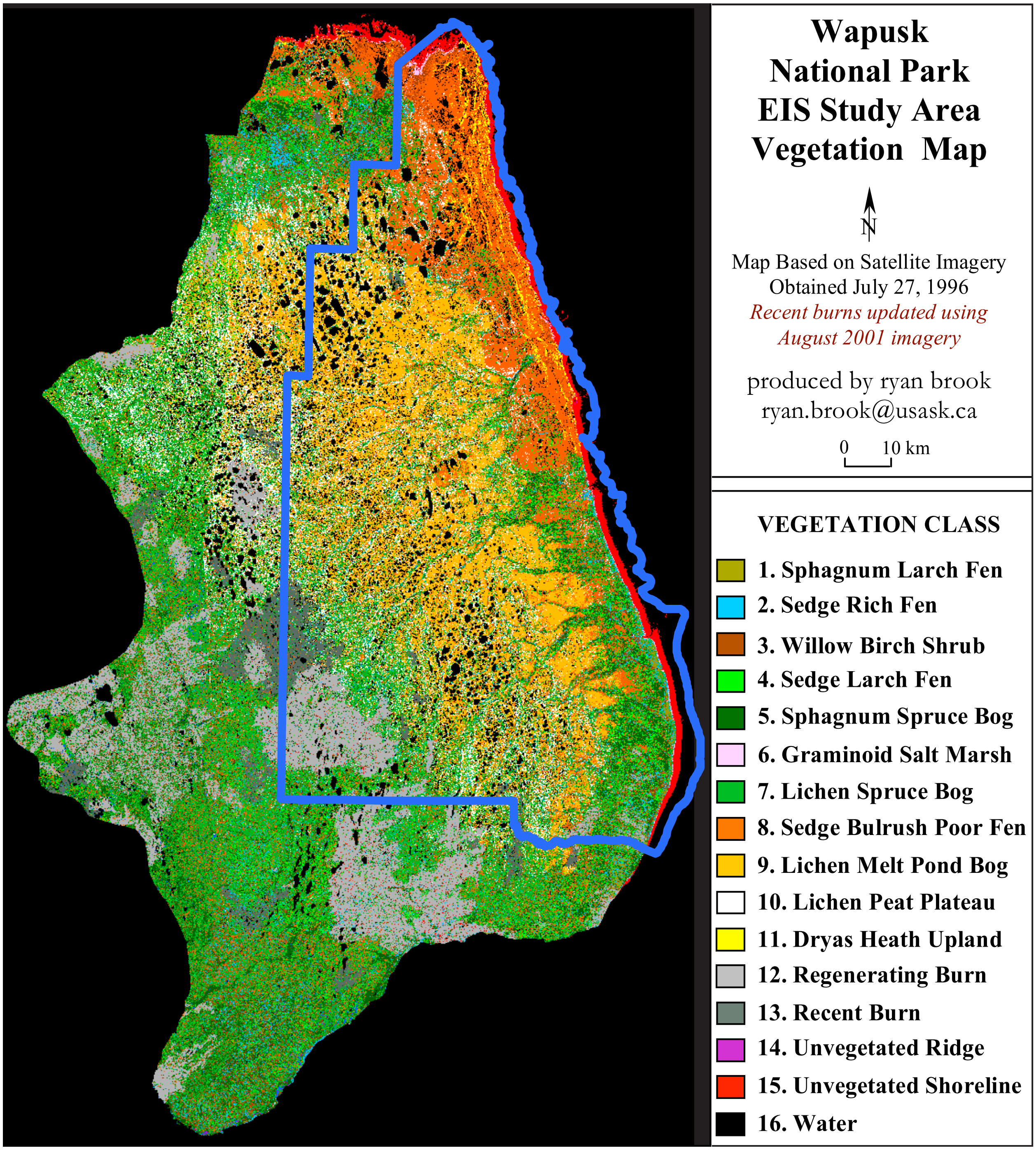

Figure 31. Land cover (circa 2001) for the Greater Wapusk Ecosytem, Manitoba, which includes Wapusk National Park (area within the blue boundary line) and the surrounding watershed, as shown. Source: Map is derived from Brook (2005). Prepared by, and used with the permission of, R. Brook.

Table 5. Area of the Greater Wapusk Ecosystem (Wapusk National Park and surrounding watershed) within major land cover classes, circa 2001.
Source: After Brook (2005). Prepared by, and used with the permission of, R. Brook.

Land cover class	Description	Area, km^2	% of total area
1. *Sphagnum* Larch Fen	Wet, nutrient-rich with an extensive ground cover of *Sphagnum* moss, low shrubs, & *Larix laricina*.	113	0.5
2. Sedge Rich Fen	Very wet, dominated by *Carex aquatilis* & *Scorpidium* moss. Small patches of *Betula glandulosa* and *Salix* sp. can be found throughout.	729	3.1
3. Willow Birch Shrub Fen	Composed primarily of shrubs, such as *Salix* sp., *Betula glandulosa,* & *Myrica gale,* ranging in height from 0.5 to 3.0 m tall.	859	3.7
4. Sedge Larch Fen	Mesic, dominated by *Carex aquatilis* in low wet areas & by *Larix laricina* & *Picea mariana* on raised hummocks.	1,098	4.7
5. *Sphagnum* Spruce Bog	Conifer tree cover mostly *Picea mariana* with extensive ground cover of *Sphagnum* sp. & shrubs, such as *Ledum groenlandicum*.	2,101	9.0
6. Graminoid Willow Salt Marsh	A complex mosaic of unvegetated clay & short graminoid vegetation, *Carex subspathacea* & *Puccinellia phyrangoides*, & *Salix* sp.	35	0.1
7. Lichen Spruce Bog	Tree cover primarily *Picea mariana* & *Larix laricina* with extensive ground cover of lichens, including *Cladina rangiferina*, *C. mitis*, & *C. stellaris*.	4,670	19.9
8. Sedge Bulrush Poor Fen	Wet, with moderate nutrients, dominated by *Carex aquatilis* & *Scirpus caespitosus*, along with *Salix* sp. and *Betula glandulosa* on hummocks.	3,578	15.3
9. Lichen Melt Pond Bog	Complex mix of *Cladina stellaris*, *C. mitis*, *Ledum decumbens*, peatlands, and melt ponds.	3,284	14.0
10. Lichen Peat Plateau Bog	Dominated by lichens, such as *Cladina stellaris*, *C. mitis*, & *C. rangiferina*, mixed with low shrubs, such as *Ledum decumbens*.	1,762	7.5
11. *Dryas* Heath	Sparse patches of *Dryas integrifolia* on relict coastal gravel beach ridges that parallel the Hudson Bay coastline.	128	0.5
12. Regenerating Burn	Primarily shrubs, such as *Ledum groenlandicum*, *Betula glandulosa*, & herbs, such as *Rubus chamomerus*.	2,181	9.3
13. Recent Burn	Charred black from fire in the last five years with <30% vegetation cover, *Epilobium angustifolium*, *Rubus chamomerus*, & *Salix* sp.	523	2.2
14. Unvegetated Ridge	Elevated, xeric gravel, sand, shattered limestone plates with <30% cover of vegetation, including *Salix* sp. & *Sheperdia canadensis*.	11	0.1
15. Unvegetated Shoreline	Low, wet areas with sand, rock, & clay with sparse cover & <30% cover of vegetation, *Hippurus vulgaris,* & *Honckenya peploides*.	427	1.8
16. Water	Vast number of very shallow & sometimes ephemeral lakes, ponds, & rivers that cover the ecozone.	1,921	8.2
Total		23,420	~100

Finer analyses of land cover for the Ontario portion of the Hudson Plains Ecozone consist of: 1) analysis of Landsat images in Ontario's Provincial Land Cover (PLC) 2000 database[14] (Figure 32, Table 6); and 2) a subjective synopsis of field surveys (Riley 1982; Roulet et al. 1994), for which an updated and detailed analysis of available field survey data for the wetland component will soon be available (Riley in press). Compared to the national land cover mapping (Figure 30, Table 4), these Ontario sources also better differentiate wetlands and resolve the tundra that is known to occur along the Hudson Bay coast (Crins et al. 2009). Ontario's satellite analysis (Figure 32, Table 6) is, however, still not associated with a high degree of accuracy, despite being produced from finer, 30 m resolution satellite data. Specifically, the classification accuracy was 51 ± 2.8% when compared to photo interpretation on available imagery, such as 10 m pansharpened-SPOT in combination with relevant data layers (e.g., field plot photos, shaded

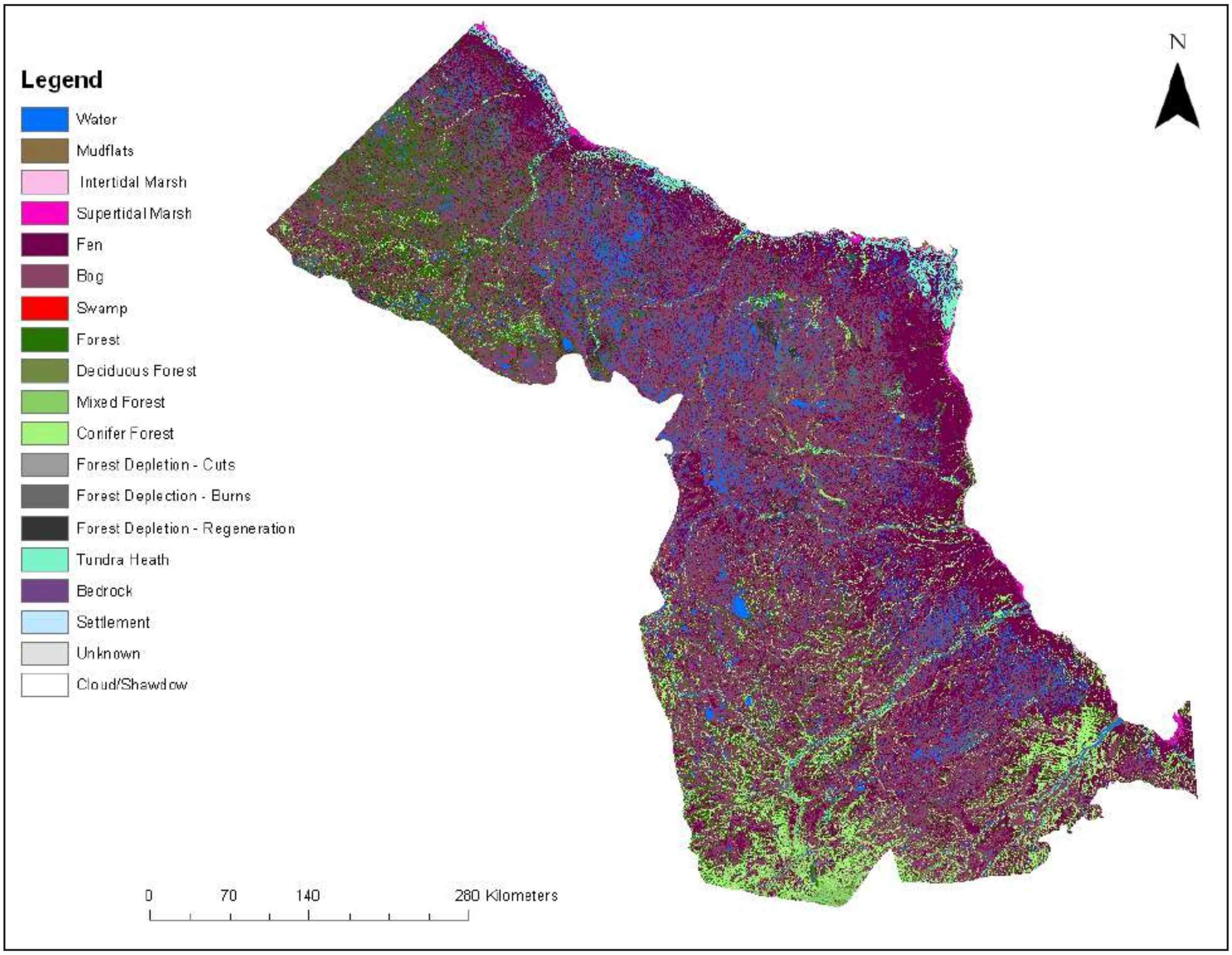

Figure 32. Land cover for the Ontario portion of the Hudson Plains Ecozone (circa 2000). Source: Ontario Ministry of Natural Resources (2010a). Map is derived from the Ontario Provincial Land Cover (PLC) 2000 database.

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[14] Ontario Ministry of Natural Resources' land cover database second edition, referred to as PLC 2000, was derived from Landsat images acquired between 1999 and 2002, with most from 2000 onward. The PLC database is available through the Ontario Land Information Warehouse.

relief, and surficial geology data) to aid in the interpretation (OMNR 2009)[15]. The production of a more rigorously developed land cover product is in progress for the bulk of the ecozone that lies in Ontario. It will include mapping disturbance (1990-2008) using time series data (30 m Landsat imagery) in conjunction with available ancillary data, such as existing fire-, insect-, and weather-related mapping (P. Sampson, Ontario Ministry of Natural Resources, pers. comm.).

Also notable when comparing the currently available Ontario land cover mapping with the ESTR-commissioned land cover mapping is a very large and mostly definitional discrepancy in the area classified as forest (e.g., much of the area classified as forest in the latter system is treed (conifer) bog). The ESTR analysis defines forest as polygons with >10% tree cover for a total area of 225,591 km^2 in the Hudson Plains Ecozone for the year 2000 (year 2000 data are shown later in Table 10), i.e., ~68% of the ecozone is classified as forest (Figure 30)[16]. In contrast, the Ontario analysis (circa 2000) defines forest as polygons with ≥30% tree cover for a total area of 51,662 km^2 (~21%) forest in the Ontario portion of the ecozone (Table 6). With this higher cut-off of tree cover defining forest classes, a much greater area (~69%) in the Ontario-based analysis is classified as wetlands instead.

Table 6. Area of the Ontario portion of the Hudson Plains Ecozone in each major land cover class in 2000. Data were generated from Landsat images acquired between 1999 and 2002.
Source: Ontario Ministry of Natural Resources (2010a). Data are derived from the Provincial Land Cover (PLC) 2000 database.

Land cover class	Area, km^2	% of total area	% of area excluding water & tidal influence
Water	16,536	6.7	
Mudflats	244	0.1	
Intertidal Marsh	218	0.1	
Supertidal Marsh	994	0.4	
Fen	69,415	28.0	30.2
Bog	100,608	40.6	43.7
Swamp	4	<0.1	0.0
Sparse Forest[a]	25,556	10.3	11.1
Deciduous Forest[b]	715	0.3	0.3
Mixed Forest[b]	7,138	2.9	3.1
Coniferous Forest[b]	18,253	7.3	7.9
Forest Depletion – Cuts	23	<0.1	0.0
Forest Depletion – Burns	3,606	1.5	1.6
Forest Depletion – Regeneration	1,324	0.5	0.6
Tundra Heath	2,836	1.1	1.2
Bedrock Outcrop & Quarry/Gravel Extraction	537	0.2	0.2
Settlement	20	<0.1	0.0
Unknown	11	<0.1	0.0
Cloud/Shadow	2	<0.1	0.0
Total	248,039	100.0	~100

[a] Sparse forest class has tree cover ≥30 and ≤60%.
[b] Deciduous, mixed, and coniferous forest classes have >60% tree cover.

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[15] No ground truthing was performed. An error matrix for the thematic (classification) accuracy of PLC 2000 for Ontario's Far North planning area (includes the Ontario portion of the Hudson Plains Ecozone but also additional land area) is available in OMNR (2009). The accuracy associated with swamps is particularly poor. The resolution of narrow, sinuous features is also generally poor.

[16] Assuming uniform forest distribution for purposes of illustration only, this area proportionately reduces to ~165,655 km^2 for the Ontario portion.

While neither of the two types of land cover analysis available for the Ontario part of the ecozone (i.e., analysis of Landsat images and subjective synopsis of field surveys) represent the ecozone as a whole, they do cover the bulk of the ecozone area (including all three ecoregions) and provide an indication of the relative proportioning of wet lowland (i.e., marshes, open fens, treed fens and swamps, open bogs, and treed bogs; Figure 33) and dry upland cover types within the ecozone. Overall, these analyses indicate that wet lowlands constitute 82% (Riley 1982) to 85.0% (Table 6) of the Ontario portion of the ecozone, with the most prevalent class being a combination of treed bog and treed fen (~ 42% cover; Riley 1982; Roulet et al. 1994) or just bogs (~40.6% cover; Table 6). The relatively small differences between the two Ontario sources are due in part to different methodologies and, in some cases, different definitions of land cover classes. For example, different definitions were used to separate bogs and fens from bog forests and fen forests. Dry upland vegetation communities cover a considerably smaller proportion of the area, on the order of 14.9% (Table 6) to 18% (Riley 1982; Roulet et al. 1994). Riley's (1982) further delineation of cover within wet lowland vegetation types is presented later (Table 9) in reference to ecoregions.

Figure 33. Some common wet lowland types in the Hudson Plains Ecozone: a) fen (near the Sutton River, Ontario) and b) treed bog (24 km southeast of Moosonee, just north of the Partridge River channel, Ontario).
Photo credits: a) K.F. Abraham, Ontario Ministry of Natural Resources and b) S. Brinker, Ontario Ministry of Natural Resources.

2.2.1.1.2 **Soils**

Pedogenesis within the Hudson Plains Ecozone is a reflection of several factors, which include cool humid climate, flat topography, isostatic rebound, coastal and fluvial parent material, and cryogenic and permafrost processes (Cowell et al. 1991; see also Section 2.1, *Abiotic Drivers*).

As a result of isostatic rebound, sites adjacent to Hudson and James bays are relatively young and recently emerged, while sites farther inland are older and tend to have much greater development of organic matter. Protz et al. (1983, 1987) determined from rebound initiation that organic matter accumulation in soils of southern James Bay peaked at 2,000 years, while in the Hudson Bay area accumulation peaked at about 3,000 years. After these peaks, the amount of organic matter decreased during the reduction stage, where decomposition became more prevalent as vegetation type changed. Thus, soil development is also influenced by growth and decay rates of organic material within this relatively flat, poorly drained topography (Cowell et al. 1991).

Although rates of organic matter development are slow, and climatic processes do not favour a long growing season, Protz et al. (1983) concluded that the podzolization process is rapid within this ecozone, and Podzols have developed on ridges in the Hudson Bay coast within the last 2,300 years. Protz et al. (1987) later determined that the podzolization process in the southern James Bay area is twice as rapid as near the Hudson Bay coast, due to warmer annual temperatures, higher precipitation, and a longer growing season. These factors contribute to a greater accumulation rate of organic carbon and vermiculite and to a greater rate of carbonate leaching within soils (i.e., Podzols) of southern James Bay.

Although soil descriptions associated with land cover classes are not available for the entire ecozone, soils have been described along the southwestern James Bay coast, between the Albany River and the Québec border (Cowell et al. 1991). Mineral and organic soils were sampled at 117 of 154 sites. Soil-vegetation relationships were described by first grouping sites into three major classes based on landform (i.e., peatlands, wetland mineral soils, and uplands) and then subdividing by physiognomic type. An amended summary of results is presented in Table 7, which includes soil subgroups found at each physiognomic type and corresponding drainage. Because vegetation and soils generally develop concomitantly, the relationships in Table 7 can likely be extended outside of the area sampled.

Table 7. Soil subgroups and physiognomic types of the southwestern James Bay coastal zone. Source: Cowell et al. (1991).

Class	Physiognomic type			Drainage	Soil subgroup
Peatlands	Fen	Open Fen	Graminoid Fen	Poor	Terric Mesisol, Typic Mesisol, Terric Fibrisol
			Low Shrub Fen	Poor	Terric Fibrisol, Terric Mesisol, Terric Fibric Mesisol, Fibric Mesisol
		Treed Fen	*Sphaghum*-Rich Treed Fen	Poor	Terric Fibric Mesisol, Terric Mesisol, Typic Mesisol
			Graminoid-Rich Treed Fen	Poor	Terric Fibric Mesisol, Terric Mesisol, Typic Mesisol, Mesic Fibrisol
	Bog	Open Bog	Graminoid Bog	Poor	Mesic Fibrisol
			Low Shrub Bog	Poor	Typic Mesisol
		Treed Bog	Graminoid Tree Bog	Poor	Terric (Mesic) Fibrisol
			Shrub-Rich Treed Bog	Poor	Terric Mesisol, Typic Mesisol, Terric Fibrisol, Mesic Fibrisol
		Conifer Swamp		Poor	Terric Mesisol, Terric Fibrisol, Rego Gleysol
Wetland Mineral Soils	Marsh	Brackish Meadow Marsh		Imperfect	Rego Gleysol
		Freshwater Meadow Marsh		Imperfect	Rego Gleysol, Terric Mesisol
	Swamp	Thicket Swamp		Imperfect	Rego Gleysol, Orthic Gleysol
Uplands	Beach Ridge	Forested		Rapid	Orthic Humic Podzol, Orthic Regosol
	Levee	Thicket		Well	Rego Humic Gleysol
		Forested		Well	Rego Gleysol, Rego Humic Gleysol, Gleyed Brunisol, Orthic Humo-Ferric Podzol

2.2.1.1.3 Intra-ecozone variation: ecoregions

The Hudson Plains Ecozone is not spatially homogenous. Vegetation and soils show zonations approximately parallel with the Hudson Bay and James Bay coasts (Ritchie 1957, 1962; Bates and Simkin 1969; OMNR 2010a and Figure 32). This variation in ecosystem structure from the coast to inland reflects time since emergence from the Tyrrell Sea superimposed on variation from north to south and proximity to the bays that reflects a climatic gradient. Consequently, pronounced soil and vegetation gradients occur in a direction perpendicular from the coast. Once land has emerged from the sea as a result of isostatic rebound, organic material begins to accumulate and mineral weathering is initiated within about 200 years (Protz 1982). After a short distance of tidal ecosystems and saltwater marsh, freshwater fens predominate along the coasts of Hudson and James bays, grading to bogs inland (Figure 34). In general, fens decrease in frequency from north to south (and inland), while bogs and swamps increase (Riley 1982). Forests predominate along the southern margin of the ecozone and along rivers (Figure 32). Corresponding soil subgroups are identified in Table 7.

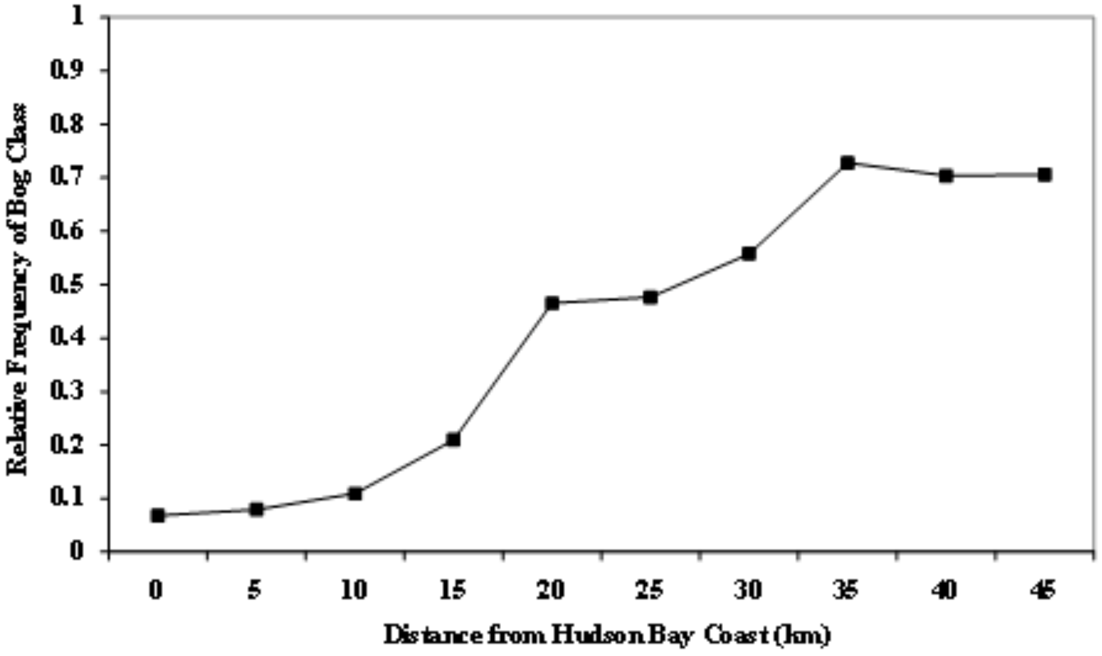

Figure 34. Relative frequency of bog vegetation along a gradient inland from the Hudson Bay coast in the Greater Wapusk Ecosystem, Manitoba (includes Wapusk National Park and the surrounding watershed, as depicted in Figure 31).
Source: After Brook (2001). Prepared by, and used with the permission of, R. Brook.

Consistent with the vegetation gradients described above, the Canadian Dynamic Habitat Index (DHI, sensu Coops et al. 2008) for the ecozone (Figure 35) shows cumulative annual greenness to be highest in the south and becoming medium along the shore of Hudson Bay, while seasonal variation in greenness (degree of vegetation seasonality) grades from high in the north to low in the south. Annual minimum vegetation cover (the minimum level of perennial cover) is relatively low throughout, but it becomes somewhat higher at the extreme southern end of the ecozone (not clearly evident in Figure 35, but see Ahern et al. in press). These 6 year average measures of greenness are, however, based on relatively coarse 1 km resolution data from MODIS sensors and limited validation.

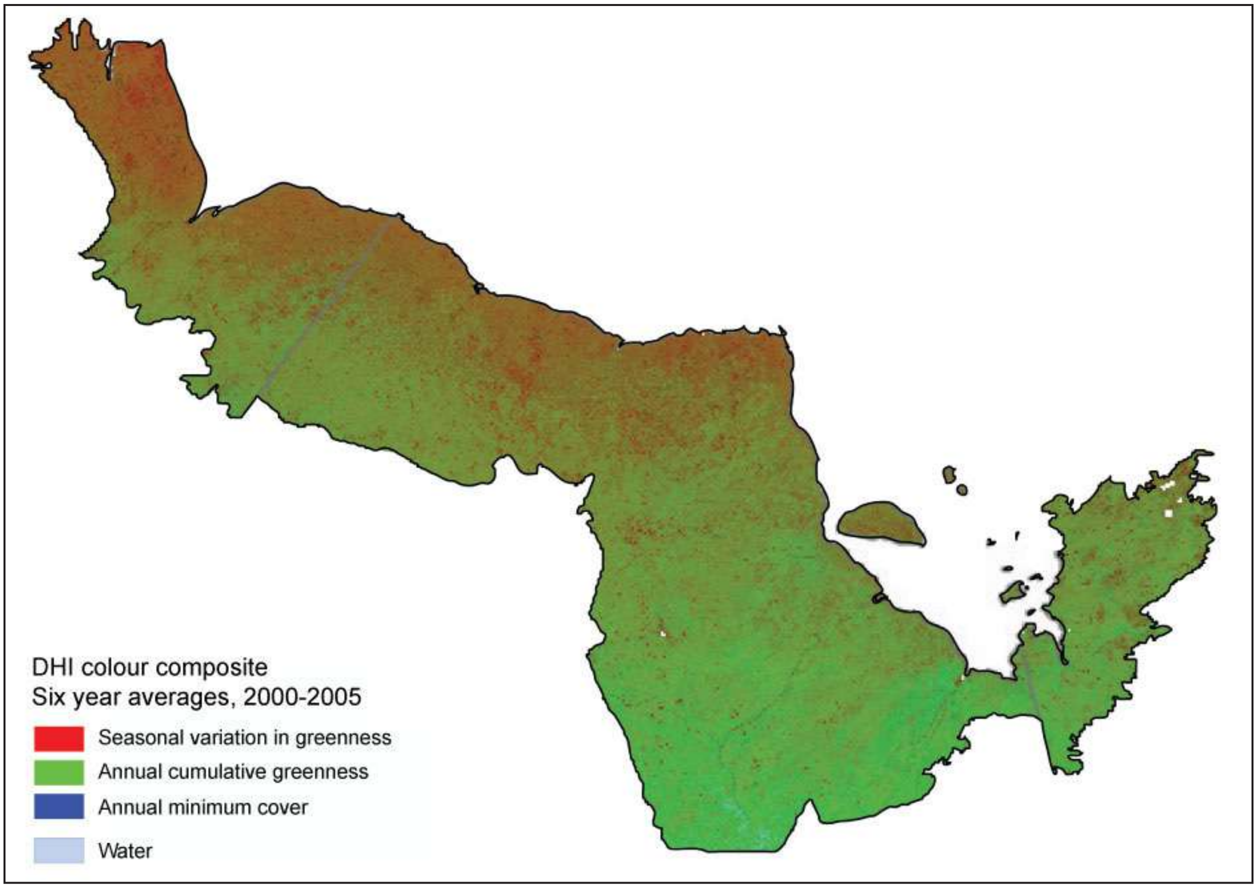

Figure 35. Canadian Dynamic Habitat Index (DHI) colour composite image of the Hudson Plains Ecozone based on a 6 year average of 1 km MODIS remote sensing data over the period 2000-2005. The DHI consists of three indicators of vegetation dynamics (equated with variation in habitat and food supply) compiled into this composite status image: 1) seasonal variation in greenness; 2) annual cumulative greenness; and 3) annual minimum (perennial) vegetation cover. The time series available for DHI is currently too short to analyze for trends. This analysis is based on the ecozones⁺ framework.
Source: Data provided by authors of Ahern et al. (in press); see also Coops et al. (2008).

The ecozone can be divided into three principal ecoregions based on physiography, climate, vegetation, and soil (ESWG 1995). From north to south, these ecoregions are: Coastal Hudson Bay Lowland (CHBL, Ecoregion Number 215); Hudson Bay Lowland (HBL, Ecoregion Number 216); and James Bay Lowland (JBL, Ecoregion Number 217). The three ecoregions are shown in Figure 36.

Defining features of each of the three principal ecoregions are summarized in Table 8, including characteristic zonal vegetation. The associated variation in ecosystem structure is outlined below. Information at the ecodistrict level of classification is limited (e.g., Smith et al. 1998) and, in Ontario, data for the Hudson Plains Ecozone are currently insufficient to allow well-described, further nested subdivisions to the ecodistrict level; only an approximation to the boundaries and characterization of ecodistricts is available (Crins et al. 2009). General descriptions of variation within ecoregions are, however, available (e.g., Coombs 1952, 1954).

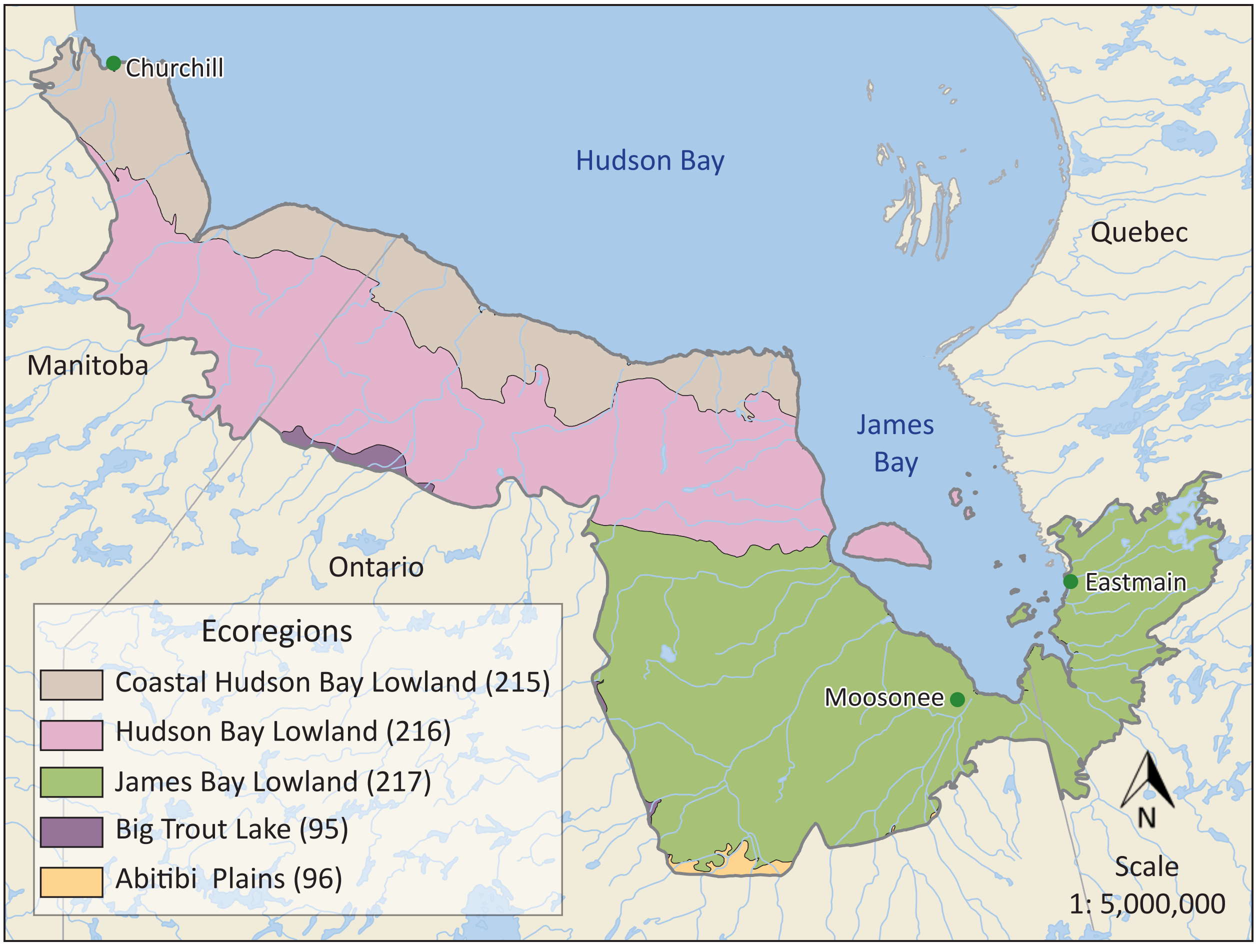

Figure 36. Ecoregions associated with the Hudson Plains Ecozone (ecozones⁺ framework). Ecoregion boundaries have not yet been modified in the new ecozones⁺ framework, which results in the revised ecozone currently containing minor portions of two additional ecoregions (95, 96), in addition to its three principal ecoregions (215, 216, 217), in certain locations where the ecozone boundary line changed.

Coastal Hudson Bay Lowland Ecoregion (215)

The Coastal Hudson Bay Lowland Ecoregion contains much of the ecozone's coastal biome and all of its tundra biome, north of the treeline. The tundra in this ecoregion represents the southernmost zone of continuous permafrost and tundra vegetation in North America (Rowe 1972; Bailey 1998; Bliss 2000; Gough and Leung 2002; Abraham and Keddy 2005; Stewart and Lockhart 2005; Zhang et al. 2008).

The overall vegetation community in this ecoregion grades from coastal vegetation to fens to bogs furthest inland (Ritchie 1957, 1962; Bates and Simkin 1969; Brook 2001; figures 31 and 32). Along coastal margins, marine deposits of clay and beach materials of sand and gravel continue to accumulate with little organic deposits, and coastal areas are dominated by salt marshes, shallow waters, and extensive tidal flats (Figure 37). Only a short distance inland from high tide, the influence of saltwater diminishes and a pattern of freshwater marshes, sedge meadows, and swamps alternates with parallel lines of exposed beach ridges with sparse vegetation and polar grassland or *Dryas*-heath (Coombs 1952, 1954; ESWG 1995; Brook 2001). Beach ridges are especially prominent in this ecoregion along the Hudson Bay coast between the Hayes River in Manitoba and the Winisk River in Ontario.

Table 8. Some defining features of the ecoregions of the Hudson Plains Ecozone.
Sources: a, ESWG (1995) and EWG (1989) with modifications for Ontario from Burger (1993); b, Brown (1967, 1968, 1973) and Dredge and Nixon (1992); c, Bailey (1998); d, Bliss (2000); e, Rowe (1972); f, Burger (1993); g, EWG (1989) and ESWG (1995); h, Scott (1995) and Elliot-Fisk (2000); i, EWG (1989) and Burger (1993); j, Elliot-Fisk (2000); and k, Burger (1993).

Ecoregion name & number	Formation	Climate[a]	Permafrost[b]	Characteristic zonal[1] vegetation	Characteristic zonal soils
Coastal Hudson Bay Lowland (215)	Part of the Tundra Division[c]; also Arctic Tundra Biome[d] or forest tundra ecotone[e] (north of the treeline)	High subarctic	Continuous climatic permafrost	Dwarf shrub-heath-lichen communities[f] with scattered stunted white spruce (*Picea glauca*) & eastern larch (*Larix laricina*)[g]	Perennially frozen Regosols
Hudson Bay Lowland (216)	Open lichen woodland[h]	Low subacrtic	Extensive to sporadic discontinuous edaphic permafrost associated with the insulating effects of a thick peat accumulation	Lichen woodland, open stands of black spruce (*Picea mariana*) with understories of Labrador tea (*Ledum groenlandicum*), lichen (*Cladina* spp.) & moss[i]	Eutric Brunisols
James Bay Lowland (217)	Closed boreal forest[j]	Humid high boreal	Sporadic discontinuous to isolated patches edaphic permafrost	Conifer forest of spruce-fir with early seral stage deciduous forest of trembling aspen (*Populus tremuloides*) & white birch (*Betula papyrifera*)[k]	Humo-Ferric Podzols

[1] *Zonal sites* are those sites typical for the climatic conditions of an area given relatively little influence from local edaphic conditions; thus, vegetation communities on zonal sites best represent climatic conditions (Bailey 1996, 1998). In the case of the Coastal Hudson Bay Lowland Ecoregion, the more characteristically observed wetland vegetation communities reflect local edaphic conditions rather than climate.

Peat accumulation on the meadows begins within only 1.5-3.5 km of the high-tide mark (Hustich 1957) and, combined with the base-rich parent material, the vegetation pattern changes to extensive areas of open fens with some treed fens alternating with treeless beach ridges of lichen *Dryas*-heath and treed ridges of lichen woodland (white spruce, *Picea glauca* and black spruce, *Picea mariana*) (Wickware and Rubec 1989; ESWG 1995; Smith et al. 1998). Open and treed bogs develop in situations where reduced drainage impedes the flow of mineral soil water (Brokx 1965; Bates and Simkin 1969) or where fens have succeeded to bogs (Riley 1982). Overall, complexes of wetlands (primarily open and treed fen, open bog, with palsa and peat plateau permafrost terrain features, such as ice-wedge polygon patterns in areas of shallow peat) and open water dominate the landscape (Figure 38), with tundra heath occurring on drier upland sites near the coast (Brook 2005; Crins et al. 2009; OMNR 2010a). Indeed, this ecoregion is considered to have the largest area of frozen peat plateau bog in Canada (Dyke and Sladen 2010). Along rivers and associated alluvial sediments (e.g., levees), combinations of white spruce, black spruce, eastern larch (*Larix laricina*), and balsam poplar (*Populus balsamifera*) develop, forming stands of low- to medium-density conifer, deciduous, and mixedwood forest (Smith et al. 1998; Crins et al. 2009) (Figure 39).

Soils within this area are dominated by perennially frozen Cryosols that have active layers of variable depths (Riley 2003). Along beach ridge systems closest to the coast, perennially frozen Regosols consist of well-drained sands, and fine gravel Regosolic soils transition

Figure 37. Coastal area south of Cape Churchill (Manitoba).
Photo credit: L. Aubry, the Hudson Bay Project.

Figure 38. Wetland and pond complex showing permafrost terrain features, i.e., lichen-covered palsas and peat plateaus. Due to their light colour, lichens reflect light, thereby helping to maintain the underlying permafrost. Near Peawanuck, Ontario.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

into young Podzols further inland, despite the existence of visible ice occurring within the profile. Podzols have been found only 6 to 10 km away from the central Hudson Bay coast (Riley 2003).

Figure 39. Treed area along the Sutton River, Ontario. Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Hudson Bay Lowland Ecoregion (216)

Within the Hudson Bay Lowland Ecoregion, similar zonation occurs with increasing distance from James Bay. Further inland (west) from the coastal-intertidal zone at James Bay, open fens and treed (eastern larch) fens dominate in a narrow band, followed by fen-bog complexes grading to treed bogs, which dominate the western boundary of the ecoregion. Similarly, south of the Coastal Hudson Bay Lowland Ecoregion boundary, fen-bog complexes grade into treed bogs, which dominate the southern boundary of the Hudson Bay Lowland Ecoregion (Brokx 1965; Bates and Simkin 1969). Beach ridges are prominent in this ecoregion along the James Bay coast south of Cape Henrietta Maria and north of Ekwan Point (Pala et al. 1991). The Sutton Ridges (Figure 3 in Section 1.1, *Geology, Topography & Climate*) are also located in this ecoregion, southwest of Cape Henrietta Maria.

Even though the Hudson Bay Lowland Ecoregion is delineated within the open lichen woodland formation (Scott 1995; Elliot-Fisk 2000) (Figure 40), this vegetation community is not dominant there due to poor drainage. Rather, dominating the landscape in this ecoregion are complexes of lichen woodland, bogs and fens, and open wetland forests of stunted black spruce and eastern larch with white spruce in the Ontario portion (Figure 41) (Coombs 1952, 1954; ESWG 1995) and bog-fen complexes in the Manitoba portion (Smith et al. 1998). Only 5.4% of the Ontario portion is mapped as lichen-woodland (likely an underestimate[17]), and it is divided between upland sites and peat plateaus (peat plateaus are most common in the northerly extent of this ecoregion and tend to be smaller than those in Ecoregion 215). Consequently, this area has been referred to as an *edaphic forest-tundra* (i.e., zonal tundra vegetation communities are common as azonal vegetation communities within the ecoregion), because it is not the climate that accounts for the paucity of forests but rather widespread wet edaphic conditions that promote peatland development (Hustich 1957).

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[17] Bates and Simkin (1969) define lichen woodlands as white and black spruce woodlands on higher marine and glacial landforms, with extensive dense stands of ground lichens and improved growth of conifers being characteristic of this type. The 5.4% value for lichen woodlands was determined from their map by dividing the area of lichen woodlands (5,086.42 km^2) indicated by the area of the Ontario portion of the Hudson Bay Lowland Ecoregion (94,823.19 km^2; Crins et al. 2009). The 5.4% value is likely an underestimate, because the scale of the map is 1:633,600; smaller stands of lichen woodlands would not be delineated at this scale (e.g., beach ridges, where not mapped separately).

Figure 40. a) Landscape view of the open lichen woodland formation, near the Sutton Ridges, Ontario and b) stand-level view of open lichen woodland, near the Winisk River southwest of Peawanuck, Ontario. Photo credits: K.F. Abraham, Ontario Ministry of Natural Resources.

The Hudson Bay Lowland Ecoregion can be divided into two subsections with the Severn River as the boundary (Coombs 1952, 1954)[18]. The immature drainage and wider presence of permafrost associated with a colder and drier climate (Brown 1973) separates the northwestern condition from the southeastern condition. In addition, much of the northwest section is covered with a complex network of dendritic streams with areas that are completely undrained (Coombs 1952, 1954) compared to the radial drainage pattern found in the south. In both sections, lichen woodland occurs on well-drained mineral soil uplands found primarily in the form of beach ridges and less commonly other gravel and sand formations, such as moraines, kames, and eskers. The main concentrations of lichen woodland communities and forested lichen-rock communities are generally associated with the better-drained uplands in the Sutton Ridges and with uplands on the Manitoba-Ontario border (Zoltai 1973). Beach ridges in the northern portions tend to be paludified treed bogs that are dominated by black spruce and lichen. Closed canopy stands of white spruce, black spruce, white birch, and trembling aspen occur along rivers and levees (Zoltai 1973; ESWG 1995; Smith et al. 1998; Crins et al. 2009), and open and treed fens and bogs occur everywhere else (Figure 42). Open canopy conifer forest occurs, but conifers are often not evenly dispersed in an area.

Figure 41. Complex of bog, fen, and clustered black spruce islands southwest of Peawanuck, Ontario. Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

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[18] The southeastern portion roughly corresponds to the Winisk River Ecodistrict (1E-3) of Ontario (Crins et al. 2009), and the northwestern portion roughly corresponds to the combined Dick River Ecodistrict (1E-2) of Ontario and the Winisk River Lowland, French Creek, and Sombart Lake Ecodistricts (1024, 1025, and 1026, respectively) of Manitoba (Smith et al. 1998).

Rather, conifers tend to be grouped along narrow strips of uplands between bogs, fens, and pools (Figure 43) and along rims of former pond beds that are now free of water (Coombs 1952, 1954).

Figure 42. a) Treed fen and b) treed bog at Akimiski Island, Nunavut.
Photo credits: K.F. Abraham, Ontario Ministry of Natural Resources.

Figure 43. Conifers grouped along thin beach ridges between fens and marsh, south of Lake River, Ontario.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

James Bay Lowland Ecoregion (217)

In the more southerly James Bay Lowland Ecoregion, extensive tidal flats occur along most of the coast from Akimiski Island, Nunavut to the Québec border. The distinctive narrow coastal band within this ecoregion is characterized by species associated with salt marshes and freshwater estuaries, and treed coastal beach ridges continue (Riley 2003). Even though the James Bay Lowland Ecoregion is considered part of the closed boreal forest (Elliot-Fisk 2000), the prevalence of wet edaphic conditions results in vegetation that continues to resemble a subarctic condition, especially in the north section. In addition, the ecoegion is associated with paludifcation of upland sites, as well as lowland sites (Klinger and Short 1996). This contrasts with the forests of the Boreal Shield Ecozone to the south, where paludification occurs only on lowland sites (Jeglum 1974; Simard et al. 2007).

The James Bay Lowland Ecoregion has been subdivided into north and south sections (Riley 1982), which roughly correspond to the Muskeg Zone and Dry Zone, respectively, of Coombs (1952, 1954)[19]. The distinguishing feature is the increased proportion of dry uplands in the Dry Zone, estimated at 23% (Riley 1982) to 40% (Coombs 1952, 1954). This increased prevalence of dry uplands in the south section is likely due to a more developed drainage system, a somewhat milder climate, and a slightly more undulating topography (Coombs 1952, 1954). Although the north section of the James Bay Lowland Ecoregion has a high boreal climate, the wet edaphic conditions result in similarities to the adjoining Hudson Bay Lowland Ecoregion, which Coombs (1952, 1954) places both as Muskeg and Small Lakes zone. This area is predominantly comprised of relatively stunted treed and open fens and treed and open bogs of black spruce and eastern larch (Wickware and Rubec 1989; ESWG 1995), with the presence of closed canopy upland forests demarcating the separation of high boreal climate from low subarctic climate. In the south section, fully developed stands of coniferous forests and mixedwood forests (Figure 44) that are similar in structure to mid-boreal forests develop on well-drained mineral soil uplands found primarily in the form of beach ridges, river banks and levees, and less commonly other gravel and sand formations, such as moraines, dunes, and eskers (Hills 1959; Wickware and Rubec 1989). Along the southwest corner (Figure 32 and see also Figure 56a in Section 2.2.2.3, *Forests (Boreal)*), stands of closed-canopy forests, which are characteristic of mid-boreal lowland forests, appear (Bates and Simkin 1969; ESWG 1995).

Table 9 shows the distribution and abundance of basic land cover types in the three ecoregions within the Ontario portion of the Hudson Plains Ecozone. A similar ecoregion-level analysis is not available for the ecozone as a whole.

Figure 44. Boreal mixedwood forest type, 80 km northwest of Hearst and 7 km southeast of Otasawian River, Ontario. Photo credit: S. Brinker, Ontario Ministry of Natural Resources.

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[19] This, in turn, roughly corresponds to the Albany Ecodistrict (2E-1) and the combined Moose River/Lower Kenogami Ecodistricts (2E-2/2E-4) of Ontario (Crins et al. 2009).

Table 9. Percent cover of wet lowland types and dry uplands in the Ontario portion of the Hudson Plains Ecozone (only): a) analyzed across the three ecoregions and b) analyzed within each of the three ecoregions. The James Bay Lowland Ecoregion is divided into north and south sections with the Albany River as the border.
Sources: Data are from Roulet et al. (1994), as derived from Riley (1982) and the National Wetland Working Group (NWWG 1988).

a) Ecoregion	Coastal Hudson Bay Lowland	Hudson Bay Lowland	James Bay Lowland		Totals
			North	South	
Area, km²	16,026	137,448	41,070	70,640	265,184
Marshes, %	0.5	1.5	0.6	1.1	3.7
Open fens, %	1.5	9.8	1.4	0.8	13.5
Treed fens & swamps, %	0.3	5.1	3.9	8.3	17.6
Open bogs, %	1.0	13.3	4.4	4.3	23.0
Treed bogs, %	1.5	12.8	3.4	6.1	23.9
Dry uplands, %	1.2	9.2	1.7	6.1	18.3
Totals	6.0	51.8	15.5	26.6	100.0

b) Ecoregion	Coastal Hudson Bay Lowland	Hudson Bay Lowland	James Bay Lowland		Total
			North	South	
Area, km²	16,026	137,448	41,070	70,640	265,184
Marshes, %	8	3	4	4	
Open fens, %	25	19	9	3	
Treed fens & swamps, %	5	10	25	31	
Open bogs, %	17	26	28	16	
Treed bogs, %	25	25	22	23	
Dry uplands, %	20	18	11	23	
Totals	100	100	100	100	

2.2.1.2 Land cover change

2.2.1.2.1 Changes in land cover

Coarse (1 km) resolution satellite remote-sensing analysis of changes in broad cover types provides no evidence that the Hudson Plains Ecozone as a whole has experienced important changes in land cover classification or use in recent times (1985-2005; Table 10, Figure 45). The analysis suggests that small losses in forest, shrubland, and low vegetation and barren classes were related to increases in the fire scar class (areas expected to regenerate; fire and other natural disturbances are discussed further in Section 2.4.2, *Natural Disturbances*), rather than to land cover conversion classes. This land cover change analysis is, however, derived from the same national land cover mapping for ESTR described earlier (Table 4, Figure 30), and it is associated with the same accuracy limitations. The error in mapping is, therefore, likely greater than the amount of change or disturbance being detected, i.e., variation in disturbance is potentially due to errors in mapping. The analysis also did not differentiate water from the background class, so it does not provide any indication of trends in water cover and conversion over time. Finer resolution land cover change assessments are not available for the ecozone as a whole at this time, nor for jurisdictionally specific portions of it. Although the MODIS global land cover product also employs time series at a regional scale, it does not extend far enough back in time to allow for meaningful analyses of change (Latifovic and Pouliot 2005).

Table 10. Area of the Hudson Plains Ecozone in each major land cover class from 1985 to 2005. Data were generated from coarse (1 km) spatial resolution satellite remote-sensing using the advanced high-resolution radiometer (AVHRR) sensor. Land cover was produced once in 1995. A methodology was developed to detect and map changes in land cover for each of the other dates in comparison with 1995. The original analysis was based on 31 land cover classes with a thematic resolution of 12 classes; these classes were further grouped for national ESTR purposes into the classes shown here (classes are defined in Table 4). This analysis is based on the ecozones$^+$ framework.
Source: Data for ecozone provided by authors of Ahern et al. (in press).

Land cover class	Area (km^2) in each land cover class, by year				
	1985	1990	1995	2000	2005
Forest	230,082	226,180	227,138	225,591	229,518
Fire Scars	3,655	9,441	7,363	11,896	5,350
Shrubland	88,540	86,546	87,965	85,513	87,991
Grassland	4	4	0	0	0
Low Vegetation & Barren	15,498	15,602	15,317	14,776	14,917
Agricultural Land	4	10	0	7	7
Urban	0	0	0	0	0
Snow / Ice / Glacier	0	0	0	0	0
Total	337,783	337,783	337,783	337,783	337,783

Future analyses of land cover change for this ecozone require an improved land cover product and should consider the dynamic nature of inter-annual changes found there (e.g., seasonality of swamps) (OMNR 2009). Indeed, the Hudson Plains Ecozone has one of the most highly inter-annually variable Dynamic Habitat Index (DHI) values among Canada's ecozones (based on major deviations from long-term, 2000-2006) (Coops et al. 2008). This variation in DHI (DHI is equated with variation in habitat and food supply) is thought to be attributable in large part to fluctuations in moisture conditions that might, for example, cause a forested area in this ecozone to be saturated or dry depending on seasonal or annual precipitation. In addition, given the high amount of wetland, especially fens with a very high water table combined with the inter-annual variation in water amount, differentiation of water from wetland is difficult. The scale-dependent nature of the analysis depends in large part on how one defines the boundary between open water and wetland from a remote-sensing perspective.

Even though available data are insufficient for adequately assessing land cover change in the Hudson Plains Ecozone, it is likely that relatively little change in land cover has occurred there owing to the limited amount of human alteration or impacts within the ecozone to date. The

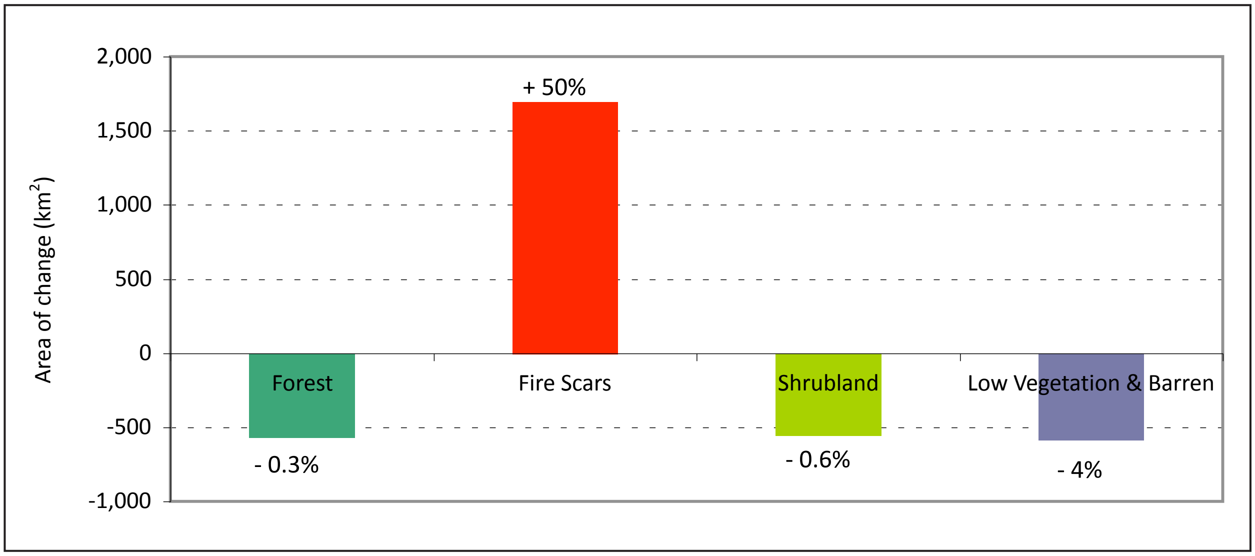

Figure 45. Summary of changes in land cover classes across the Hudson Plain Ecozone, 1985-2005. Data labels are the percentage change of each category, since 1985. In 2005, the % area of the ecozone in each class was: forest, 67.9%; fire scars, 1.6%; shrubland, 26.0%; and low vegetation and barren, 4.4% (per Table 10). Land cover in agricultural land and grassland classes was negligible for this ecozone in 2005 (<0.01% combined), and these classes are not shown here. The analysis was also too coarse to map changes in the urban class over time. The fire scars class is predominantly new (<5 years) and older (>5 years but not yet revegetated) burns, but it can include larger areas of harvest, mining, or severe insect defoliation, if present. The resolution of the data was not fine enough to pick up smaller disturbances, such as clearcuts, nor the spectral signature from insect disturbances, if present. This analysis is based on the ecozones⁺ framework.
Source: Data for ecozone provided by authors of Ahern et al. (in press).

low amount of land cover change suggested in the above analysis for 1985-2005, along with no reported conversion of natural land cover types to developed classes (Figure 45), is consistent with the ecozone's small human population (Section 1.2.1, *Settlement History*), effectively natural and apparently unchanged disturbance regime (Section 2.4.2, *Natural Disturbances*), near-absence of commercial forestry, as well as agriculture and peat harvesting (Section 1.2.2, *Economic History*), and low amount of resource development in the ecozone during the 1985-2005 analyzed period (the Limestone Dam was installed on the Nelson River in 1989; Monk et al. in press). Construction of the Victor mine near Attawapiskat began after the analyzed period, in 2006 (DeBeers Canada 2008). The area impacted by the mine may be considerably greater than the mine itself due to activities such as dewatering (AMEC 2008). What is unclear at this time is the extent of impacts, which might be as large as ~500 km² (due to dewatering), and the magnitude of impacts, which might result in some wetlands not being restorable (see sections 2.2.2.4.1, *Wetlands (Freshwater)* and 2.4.6, *Hydrological Processes*). No other major industrial development on the terrestrial landscape has occurred since, although pressure for new resource developments is mounting, particularly in mining (Manitoba Geological Survey 2003; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates 2010; Micon International 2010), hydroelectric (OPA 2007; Hydro-Québec 2010; MDDEP 2010; Manitoba Hydro 2010; OMNR 2010b), and, to a lesser extent, wind-farming (Environment Canada 2008; OMNR 2010b; see also Hélimax Énergie and AWS Truewind LLC 2005) sectors. Land use planning is in progress for at least part of the ecozone (Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*).

Several land cover changes have occurred in the ecozone over time that are not evident in the above coarse and temporally limited analysis of land cover change. These land cover changes are detailed in Section 2.2.2, *Changes in Extent & Quality of Important Biomes or Realms*. Included among these changes is an ~30% loss in coastal salt marsh vegetation, since the 1970s, due to ongoing excessive foraging principally by an overabundant migratory lesser snow goose (*Chen caerulescens caerulescens*) population. The geese are also damaging freshwater wetlands in the adjacent tundra to some extent. Some more localized land cover changes have occurred near the relatively new Victor mine, as well as in association with hydroelectric developments. The most notable land cover change associated with hydroelectric development was in 1980 (prior to the period of the above land cover change analysis), when 740 km^2 of land was flooded at the eastern edge of the ecozone to create the 1,040 km^2 Opinaca reservoir that is part of the La Grande hydroelectric complex (Therrien et al. 2004).

2.2.1.2.2 **Landscape fragmentation**

As one of Canada's ecozones least affected by human development, the Hudson Plains Ecozone has experienced very little anthropogenic fragmentation[20], i.e., is still highly intact. In a 2006 analysis (Lee et al. 2006[21]), this ecozone was determined to be covered by *intact forest landscape fragments*[22] (i.e., intact landscape patches or units) of more than 10,000 ha over 97% of its total area (Figure 46). In the same analysis, the Hudson Plains Ecozone evidenced as the least anthropogenically fragmented or most intact of all forested ecozones in Canada (i.e., of all ecozones with minimum 15% forest cover). The Hudson Plains Ecozone is, therefore, important habitat for species like woodland caribou (*Rangifer tarandus caribou*) and wolverine (*Gulo gulo*), which require large tracts of unfragmented and/or unroaded landscape and are especially vulnerable to human disturbance (see caribou and wolverine profiles in Section 2.3.3, *Trends in Species of Special Interest*).

Linear anthropogenic fragmentation of the landscape is currently limited to a relatively small number of transportation and hydroelectric transmission corridors. The western and eastern extremities of the ecozone are transected from the south by two railway lines (one each in Manitoba and Ontario) that terminate near the coast (Abraham and Keddy 2005; Stewart and Lockhart 2005), but the ecozone is still nearly roadless. One all-season road (James Bay Road) connects the coastal communities of Eastmain (1995) and Waskaganish (2001) in Québec with the highway system in the south (Hydro-Québec 2003; Stewart and Lockhart 2005). Winter roads seasonally connect the coastal communities, and a number of hydroelectric transmission corridors occur throughout (Hydro-Québec 2003; Abraham and Keddy 2005; Stewart and

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[20] In ESTR, fragmentation, or the process of fragmentation, is defined (sensu Fleishman and Mac Nally 2007) as the division of contiguous habitat into smaller patches. Both natural and anthropogenic processes can lead to fragmentation, which can decrease total habitat size and connectivity, decrease habitat density, and increase edge density (Ahern et al. in press).

[21] An updated version of the Lee et al. (2006) report is in preparation and preliminary results for Hudson Plains Ecozone are very similar to those shown here (Smith et al. 2009).

[22] An intact forest landscape is a contiguous mosaic of naturally occurring ecosystems (i.e., contiguous blocks of forest, bog, water, tundra, and rock outcrops) in a forested ecozone (i.e., an ecozone with at least 15% forest cover) that is essentially largely undisturbed by human influence. An intact forest landscape (Lee et al. 2006): 1) is free from substantial anthropogenic fragmentation; 2) is free from substantial human influence for periods that ensure that it is formed by naturally occurring ecological processes; 3) contains only naturally seeded indigenous plant species and supports viable populations of most native species associated with the ecosystem; and 4) is large enough to be resilient to edge effects and to survive most natural disturbance events.

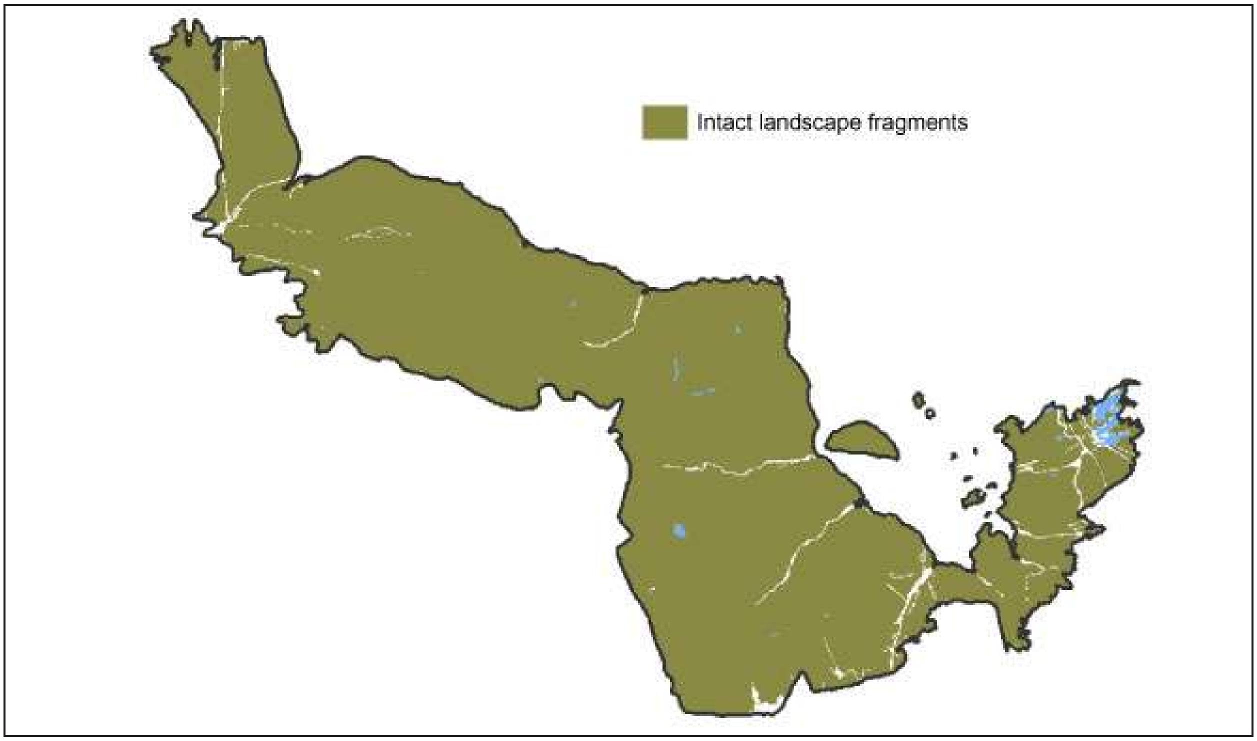

Figure 46. Intact landscape fragments larger than 10,000 ha in the Hudson Plains Ecozone, based on an analysis of anthropogenic disturbances in 2006. An intact landscape fragment is defined as a contiguous mosaic, naturally occurring, and essentially undisturbed by human influence. It is a mosaic of various natural ecosystems, including forest, bog, water, tundra, and rock outcrops (see Footnote 22). The Hudson Plains Ecozone is covered by intact landscape fragments over 97% of its total area as of 1996, based on the ecozones⁺ framework.
Source: Adapted from Lee et al. (2006) for ecozone⁺ boundaries.

Lockhart 2005; OMEI and OMNDMF 2009). The most recent fragmentation in the ecozone is from winter roads and a major transmission line[23] associated with the Victor mine near Attawapiskat (DeBeers Canada 2005), as well as the winter roads that now seasonally connect Peawanuck with Fort Severn and Fort Severn with Shammattawa.

Future fragmentation is likely from roads and adjoining developments. Currently, feasibility planning is in progress for an all-season road that would run along the western edge of the ecozone, from Gillam to Churchill, Manitoba and beyond to Rankin Inlet, Nunavut (Government of Nunavut and Government of Manitoba 2010; SNC Lavalin 2010). Likewise, a pre-feasibility study is in progress in Ontario to assess possible routes for an all-season road that would permanently connect the ecozone's communities of Attawapiskat, Fort Albany, Kashechewan, and Moosonee/Moose Factory to the provincial highway system to the south (via Highway 11) (Government of Ontario 2009). As well, the recent discovery of world-class chromite deposits inland, within the Ring of Fire mineral field west of Attawapiskat (Golder Associates 2010; Micon International 2010), suggests possible future north-south fragmentation of the landscape from the adjoining of east-west developments, from the Ring of Fire east to the Victor mine and on to Attawapiskat at the coast.

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[23] The Victor mine is serviced by a major electricity transmission line that runs along the James Bay coast from the Otter Rapids generating station (Abitibi River) at the southern boundary of the ecozone in Ontario (the Otter Rapids generating station is shown in Figure 65, Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Anthropogenic river fragmentation associated with hydroelectric development is discussed in Section 2.2.2.4.2, *Rivers/Streams & Lakes*.

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2.2.2 **Changes in extent & quality of important biomes or realms**

This sections reviews what is known about the status and trends occurring in each of the four major biome types[24] in the Hudson Plains Ecozone: coastal, polar-tundra, forests (boreal), and inland waters (wetlands, rivers/streams, and lakes).

2.2.2.1 **Coastal**

Susan M. Tully, Ontario Ministry of Natural Resources
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With inset (succession) by
Zaid Jumean, Ontario Ministry of Natural Resources

The coastal biome[25] of the Hudson Plains Ecozone is dominated by extensive tidal flats, salt marshes, and shallow waters (ESWG 1995; Figure 47). An intertidal zone occurs as a distinct vegetational, hydrological, and physiographic zone between the low tide mark along the Hudson Bay and James Bay coasts and the raised beach ridge and freshwater complexes of the interior, within all three ecoregions of the ecozone (Cowell et al. 1982). Subtidal eelgrass (*Zostera marina*) beds form an additionally important component of the coastal biome along the Québec coast in eastern James Bay and in isolated portions of the James Bay coast in Ontario (Curtis 1973; Dignard et al. 1991; Ettinger et al. 1995).

Figure 47. Coastal area along western James Bay (Ontario). Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Salt marshes, i.e., marshes that are inundated by saline or brackish water from tidal marine sources (Figure 48),

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[24] A biome is a large community of plants and animals that occupies a distinct type of environment. Major biomes recognized under the ESTR framework are defined by: Frisk, J. *In Press*. 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.

[25] Under the ESTR framework, the term coastal refers to intertidal zones, reefs, coastal dunes, and eelgrass communities (Frisk in press).

cover 85-90% of the coastal area (Glooschenko 1988), forming what are considered to be some of the largest and best-developed polar salt marshes in the world, i.e., those characterized by the presence of permafrost (Martini et al. 2009). These salt marshes are highly productive transitional intertidal ecosystems dominated by vegetation that is tolerant of wet and saline soils (Glooschenko 1988). They form both on open coasts and in swales on coasts with beach ridges (Martini et al. 2009), where excess sediment can be trapped without being moved about by large waves (Glooschenko 1988). Isostatic rebound aids in the creation of these marshes, as it creates sand spits that enhance silt sedimentation, improve drainage regime, and facilitate subsequent salt marsh development (Kershaw et al. 1975). Tides, sea ice, and wind also help shape these environments (see Section 2.4.1, *Coastal Building Processes*). Spatial variation in vegetative composition depends on salinity, elevation, water content, and soil texture (Glooschenko 1988; Stewart and Lockhart 2005), as well as the availability of propagules (McLaren and Jefferies 2004). Most salt marshes are, however, sedge- and/or grass-dominated (Cargill and Jefferies 1984; Glooschenko 1988). Plant communities and successional processes associated with both lower (seaward) and upper salt marsh communities have been well described (Inset 4).

Figure 48. Intact coastal salt marshes, east of the mouth of the Winisk River, Ontario. Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Inset 4. **Overview of salt marsh communities & their succession on the Hudson Bay coast**

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The plant communities characteristic of the ecozone's coastal salt marshes along Hudson Bay are shown in Table 11. The lower salt marsh consists of a considerably different dominant plant community (P-C) than the upper salt marsh community (C-F), especially on well-drained, frost-heave hummocks, which have lower soil salinity.

Table 11. Salt marsh communities along the Hudson Bay coast.
Sources: For lower salt marsh (P-C) community: Kershaw (1976); Hik et al. (1992); and Jefferies and Rockwell (2002). For upper salt marsh (C-F) community: Jefferies et al. (1979) and Hik et al. (1992).

Salt marsh type	Dominant plant species	Other frequent plant species
Lower salt marsh, P-C community	*Pucinellia phyganodes* (grass); *Carex subspathacea* (sedge)	*Triglochin palustris; Cochlearia officinalis; Plantago eriopoda; Potentilla egedii; Ranunculus cymbalaria; Stellaria humifusa*
Upper salt marsh, C-F community	*Calamagrostis deschampsioides* (grass); *Festuca rubra* (grass)	*Chrysanthemum arcticum;* *Potentilla egedii;* *Parnassia palustris*; *Stellaria longipes*

Successional processes in coastal salt marshes have been particularly well studied at La Pérouse Bay, Manitoba, but similar processes have been described along the Hudson Bay coast in Ontario as well (Glooschenko and Martini 1981). The continual emergence of new land along the coast from isostatic rebound results in a natural transition from poorly drained, high salinity soil in the intertidal zone to better drained, lower salinity soil higher on the elevational gradient inland. In the absence of geese foraging, the associated successional transition from a salt tolerant, lower marsh, *Pucinellia-Carex* (P-C) community to an upper marsh, *Calamagrostis-Festuca* (C-F) community occurs rapidly, within ~5 years of land being unveiled through rebound (Hik et al. 1992).

Both biotic and abiotic processes facilitate this transition. Ground cover accumulates in the lower marsh, increasing organic litter. The humus and plant litter insulates the ground below, thereby decreasing the surface temperature and evaporation from the soil-litter interface. The salinity of the surface sediment is lower as a result (Bazely and Jefferies 1986; Hik et al. 1992). Continued isostatic rebound and/or frost heave form raised hummocks that are less affected by environmental fluctuations along the coast (e.g., tidal inundation), and the sediment is stabilized by vegetation growth (Zoltai 1973; Jefferies et al. 1979). This action further modifies edaphic properties, allowing plant community transition seaward to landward. On these hummocks, plant communities tend more towards C-F dominated communities.

Geese foraging affects the rate of succession from P-C- to C-F communities in different ways, depending on the intensity of foraging. At low to moderate geese grazing intensities, the transition from the primary successional state (P-C) to the secondary successional state (C-F) occurs more gradually (i.e., *Puccinellia* and *Carex* can be present further inland than just at the seaward end of salt marshes) (Hik et al. 1992), because geese promote their preferred graminoid P-C species by removing the apical meristems of competing dicotyledonous species (Bazely and Jefferies 1986) and by enhancing the net above-ground primary productivity of the graminoids (Cargill and Jefferies 1984; Hik and Jefferies 1990; Hik et al. 1991; see also sections 2.4.3.2, *Herbivore-Plant Interactions* and 2.4.5, *Nutrient Cycling*). Succession proceeds, however, because at low to moderate foraging intensities the geese cannot arrest vegetative shifts associated with changes in edaphic conditions from isostatic rebound (Hik et al. 1992).

Conversely, and as detailed in the main text and Inset 5, excessive goose foraging of the ecozone's salt marsh vegetation has more dramatic effects on succession. Where vegetation damage is severe, an apparent trophic cascade leads to an alternate stable state of hypersaline bare sediment, from which recovery might take decades (succession must recommence from bare sediment). Reduced plant and animal diversity in damaged intertidal and supratidal areas (e.g., Bazely and Jefferies 1986; Milakovic et al. 2001), along with the presence of species characteristic of disturbed sites (Abraham et al. 2005b), are indications that natural succession has been interrupted.

The ecozone's extensive salt marsh ecosystems provide important breeding grounds and staging areas for a large number of migratory waterfowl and shorebirds (Morrison and Harrington 1979; Ross 1982; Thomas and Prevett 1982; Stewart and Lockhart 2005; Niles et al. 2010). Some of the most common waterfowl species found in this area are Canada goose (*Branta canadensis*), lesser snow goose (*Chen caerulescens caerulescens*), brant (*Branta bernicla*), pintail (*Anas acuta*), black duck (*A. rubripes*), green-winged teal (*A. crecca*), and mallard (*A. platyrhynchos*) (Ross 1982; Thomas and Prevett 1982; Glooschenko 1988). This coastal area is also an internationally recognized staging area for several shorebird species, including the semipalmated sandpiper (*Calidris pusilla*), greater yellowlegs (*Tringa melanoleuca*), lesser yellowlegs (*T. flavipes*), Hudsonian godwit (*Limosa haemastica*), and red knot (*Calidris canuta*) (Morrison and Harrington 1979; Glooschenko 1988; Niles et al. 2010). These shorebirds use well-defined portions of the salt marsh system, feeding largely on benthic invertebrates and occasionally on small fish trapped in pools (Glooschenko and Martini 1981).

2.2.2.1.1 **Trends & human influences**

The coastal zone of the Hudson Plains Ecozone, and in particular its salt marshes, has been under considerable stress over the past four decades, predominantly due to increased foraging pressure (grazing and grubbing) by a greatly expanded Mid-Continent population of lesser snow goose but also by increasing Canada goose breeding and moulting populations in the area. Within these grass- and sedge-dominated ecosystems, intensive destructive foraging by geese has led to vegetation loss, exposure and erosion of sediment, and the development of alternate stable states among plant species (Jefferies and Rockwell 2002; Jefferies et al. 2003, 2006). Approximately one-third of the coastal salt marsh vegetation in the Hudson Plains Ecozone has been destroyed by the geese, since the 1970s, and a far greater area will be severely damaged if this intense foraging pressure continues (Bertness et al. 2004).

The intense foraging by lesser snow goose in salt marshes of the Hudson Plains Ecozone is related to increases in the population size of this migratory waterfowl species, principally as a result of human influences outside the ecozone. The Mid-Continent population of lesser snow goose, to which individuals migrating through and nesting in both Manitoba and Ontario portions of the ecozone belong, has increased greatly over the past four decades, by as much as 7% per year (Abraham and Jefferies 1997; see also Section 2.3.3.3.2, *Waterfowl*). The adult portion of the Mid-Continent population reached as many as 7 million (Jefferies et al. 2006) or more. Relatively recent estimates of the La Pérouse Bay snow goose colony in Manitoba suggest that that colony has increased by approximately 5-7% per year, since the late 1960s (Jefferies et al. 2006), growing from 1,200 pairs to almost 30,000 pairs by 1997 (Kerbes et al. 2006). The increased size of the Mid-Continent population is attributable to a greater supply of agricultural food on wintering grounds (mostly in the southern United States) and along migration routes, declining harvest rate, and the development of migration and winter refuges (Abraham and Jefferies 1997; Jefferies et al. 2003). Some localized effects of declining harvest rates are attributable to changes in human activity within the ecozone. For example, coastal salt marshes near the mouth of the Winisk River experienced vegetative losses after snow goose numbers increased there in response to decreased hunting pressure, which was associated with relocation of the community of Winisk ~40 km further inland (as Peawanuck), following the flooding of Winisk in 1986 (Jefferies et al. 2006).

The rate of revegetation of intact swards (i.e., heavily grazed but not grubbed) depends on the intensity of grazing: if they are protected from grazing, vegetative change and biomass increase occurs rapidly, but, if grazing persists, change in vegetative composition can be more gradual (Hik et al. 1992; O et al. 2006). In some cases, transient plant species are able to colonize open areas. For example, annual halophyte species, such as *Salicornia borealis* and *Atriplex patula* var. *hastate*, have been documented colonizing mudflats (Handa and Jefferies 2000), and *Senecio congestus* is a common pioneer species in shallow grubbed depressions and organic soils on devegetated ridges (Jefferies et al. 2003).

If soils remain exposed in areas devoid of vegetation, erosion can occur; the top humus layer is lost, exposing marine clays and glacial tills (Figure 49). In these cases, revegetation could take decades, as unconsolidated organic matter needs to be deposited for plant colonization to take place, pending the cessation of foraging (Jefferies et al. 2006). Exposed soils are also susceptible to increases in soil surface temperature and increases in wind exposure. These increases can exacerbate evapotranspiration, ultimately leading to hypersaline conditions in soil water, which can limit the re-establishment of plants (Iacobelli and Jefferies 1991; Srivasta and Jefferies 1995; Handa and Jefferies 2000; Handa et al. 2002; Ngai and Jefferies 2004; Abraham et al. 2005a). Damaged areas eventually coalesce to create large expanses of barren landscape (McLaren and Jefferies 2004).

Figure 49. An example of the severe damage caused to coastal salt marsh ecosystems of the Hudson Plains Ecozone due to intense goose foraging, principally by a greatly increased Mid-Continent population of lesser snow goose. Geese were excluded from the area inside the fence. La Pérouse Bay, Manitoba.
Photo credit: Hudson Bay Project.

As vegetation composition changes in the salt marshes, so does the amount of forage available for geese. In some cases, a scarcity of preferred salt marsh forage (namely *Puccinellia phryganodes* and *Carex subspathacea*) has forced lesser snow geese to move inland to freshwater marshes and fens in the adjacent tundra to nest and feed (Jefferies et al. 2003; Ngai and Jefferies 2004; see also Section 2.2.2.2, *Polar-Tundra*). The herbivorous Canada goose has also been adversely affected by the vegetative changes. On Akimiski Island, for example, gosling Canada geese raised in areas with greater damage have significantly smaller body size and lower first year survival than those raised where lesser snow geese do not occur (Hill et al. 2003).

Not only does the destruction of salt marshes remove important food sources for species that feed directly on the vegetation (as above), the changes in vegetation, soil, and water also reduce the suitability of the zone for other components of the coastal food web. Terrestrial and aquatic insects and soil micro-organisms are affected by the indirect changes induced by the destructive foraging (Milakovic et al. 2001; Milakovic and Jefferies 2003). Sharp declines in soil invertebrate abundance, especially spiders (Araneae) and carabid beetles (Carabidae), have been documented (Milakovic and Jefferies 2003). Moreover, chironomid (Chironomidae) species richness was reduced from five species from five genera within undamaged saltwater marshes to only one halophytic chironomid species within damaged hypersaline saltwater marshes (Milakovic et al. 2001). Such changes to invertebrate communities affect foraging opportunities of passerine birds and shorebirds that are primarily insectivorous during the breeding season. Impacted bird species that depend on these habitats for nesting and invertebrates (for food) include savannah sparrow (*Passerculus sandwichensis*), semipalmated sandpiper, and yellow rail (*Coturnicops noveboracensis*) (Rockwell et al. 2003, 2009).

The impacts of lesser snow geese on the ecozone's salt marshes have been particularly well documented at La Pérouse Bay, Manitoba. Studies there show trends for increasing area damaged over time, and they otherwise exemplify the significant impact that intensive geese foraging is having on the ecozone's coastal zone (Inset 5).

Aside from indirect human influences on the ecozone's coastal biome via increased foraging pressure from a greatly expanded lesser snow goose population, other, more direct human influences are associated with changes to the coastal biome. Changes in overland river flow and associated sediment loads that result from hydroelectric developments in and around the ecozone have impacted salinity and other aspects of habitat quality in the interfacing estuarine and marine environments of Hudson and James bays. For example, the 90% reduction in flow at the mouth of the Eastmain River associated with its diversion north to the La Grande River led to greater sedimentation and intrusion of saltwater in the Eastmain River estuary, with associated impacts on the fish community (see Section 2.2.2.4.2, *Rivers/Streams & Lakes*). In the near-shore environment, hydroelectric development in the broader James Bay region, and particularly the increased flow output from the La Grande River (to which flows from the Eastmain and Opinaca rivers were diverted), has been implicated in a steep decline in subtidal eelgrass beds along the eastern James Bay coast (see the eelgrass discussion in Section 2.3.3.7, *Vascular Plants*). Eelgrass beds provide feeding grounds and nurseries for coastal fish species and invertebrates and forage for brant, Canada geese, and ducks.

Several factors also pose future threats to the ecozone's coastal biome. Notably, some of the major rivers that enter the western and southern shores of James Bay, including the Attawapiskat, Albany, Moose, and Harricana rivers, hold potential as sites for new hydroelectric projects (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). If these projects are carried out, the coastal biome could be further modified, as changes to the flow regime (and hence also salinity) and sediment load deposited in coastal areas occur (Glooschenko and Martini 1987). Climate change is also expected to impact coastal marshes directly by affecting vegetation growth and reproduction (productivity) through increased temperature and precipitation and through associated effects on sea ice, permafrost, and freshwater runoff and river discharge (Martini et al. 2009). The vegetation changes in coastal salt marshes caused by intensively foraging geese could also be indirectly exacerbated by climate change via prolonged staging periods of geese in years of late phenology induced by climate anomalies (Jefferies and Rockwell 2002). Conversely, greater use of some coastal areas might be promoted by relatively early spring thaw, as goose foraging can start earlier in the season (Abraham et al. 2005b). Indirect effects of climate change on the ecozone's low-lying coastal biome are also expected through increased wave action and storm surges under reduced sea ice condition (Martini et al. 2009). However, sea level rise is less of a concern for this ecozone than for some other coastal areas of Canada owing to an especially high rate of isostatic rebound, albeit the rate of successional development of coastal systems might be impacted (see also Section 2.4.1, *Coastal Building Processes*).

Inset 5. **Impacts of lesser snow goose on coastal vegetation at La Pérouse Bay, Manitoba**

Kenneth F. Abraham, Ontario Ministry of Natural Resources

Numerous studies have been conducted at La Pérouse Bay, Manitoba on the interactions of lesser snow goose with the major vegetative types in the area. An apparent trophic cascade, meaning *runaway consumption and downward dominance through a food chain* (Jefferies 1997), has been triggered by the exponential growth of the Mid-Continent population of lesser snow goose over the past four decades. This trophic cascade is sustained by positive feedbacks, which are self-amplifying. One such feedback involves grubbing in spring, whereby geese uproot large areas of *Puccinellia phryganodes* and *Carex subspathacea* and other species in the salt marshes, fragmenting swards and exposing the edges to secondary abiotic effects, such as erosion, drying, and hypersalinity. A large proportion of the Mid-Continent population migrates through the ecozone in spring, and six colonies of the species nest in the ecozone. In many years, especially those with late snow melt and thaw, millions of geese are held up on the northward journey, exacerbating the impact of their foraging. The combined effect of the grubbing and the secondary processes is to reduce the amount of above-ground vegetative matter produced by these plant species, which is the preferred forage for goslings and adults alike. The second feedback thus involves grazing during the nesting season and following hatch. The remaining sward area, both intact and fragmented, is grazed more intensively by ever larger numbers of geese, allowing for less compensatory growth and eventual exhaustion of the plants. The end result is an alternate stable state, wherein large areas of exposed sediments are resistant to recolonization, because few plants can germinate or establish in the saline sediments.

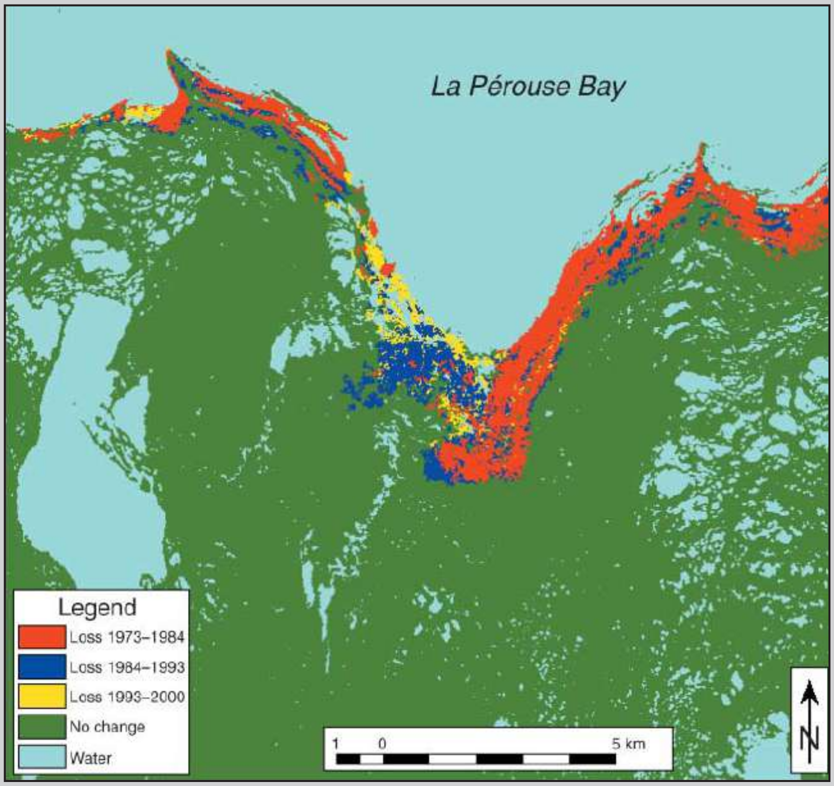

Figure 50. Normalized-difference vegetation index (NDVI) analysis of Landsat imagery showing areas with vegetation loss from goose foraging at La Pérouse Bay, Manitoba, for three successive periods between 1973 and 2000.
Source: Jefferies et al. (2006), reused by author from Journal of Ecology, Vol 94, No. 1, R.L. Jefferies, A.P. Jano and K.F. Abraham, A biotic agent promotes large scale catastrophic change in the coastal marshes of Hudson Bay, Copyright 2006, with permission from Blackwell Publishing Ltd.

Vegetation damage at La Pérouse Bay was extensive from 1973 to 2000, not only to intertidal and supratidal salt marshes but progressively also to adjacent brackish and fresh waters (Abraham et al. 2005b; Jefferies et al. 2006) (Figure 50). Between 1973 and 1984, most vegetation loss due to the geese occurred in the intertidal salt marsh (the preferred habitat). From 1984 to 1993, the majority of vegetation loss occurred in the supratidal salt marsh and inland saline areas, while from 1984 to 2000 damage was associated with riverine and brackish plant communities and in vegetation adjacent to ponds in inland freshwater sedge meadows. The successive waves of destruction of plant communities have transformed the entire coastal ecosystem.

As the Mid-Continent population migrates through the entire Hudson Plains Ecozone, similar processes, feeding pressure, and damage to coastal vegetation has occurred and been described from Manitoba to James Bay, including Akimiski Island, Nunavut.

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2.2.2.2 **Polar-tundra (including forest-tundra)**

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With insets (succession) by
Zaid Jumean, Ontario Ministry of Natural Resources
Kim Monson, University of Winnipeg

Under the ESTR framework, tundra is defined as treeless regions that contain nearly continuous (>50%) plant cover and occur within or close to the Arctic Ecozone in high latitude systems that are frozen for most of the year (Frisk in press). Tundra occurs in the northern extent of the Hudson Plains Ecozone, within the Coastal Hudson Bay Lowland Ecoregion (Abraham and Keddy 2005; Stewart and Lockhart 2005). More specifically, it forms a narrow band of land contiguous with the inland side of the ecozone's coastal-intertidal zone, from Churchill, Manitoba to near the Lakitusaki River, Ontario, in the area of continuous permafrost (Smith et al. 1998; Crins et al. 2009). This maritime tundra is typically 20-40 km wide along Hudson Bay, but it extends further inland in some areas, such as at Cape Churchill, Manitoba and at Cape Henrietta Maria, Ontario, the latter location being the widest area of open treeless tundra in Ontario (Riley 2003). The tree *line* itself has been described as erratic, extending farthest north on river levees and beach ridges, where drainage is better, and the active layer is deeper (Riley 2003). At more local scales, the treeline is also controlled by winter winds and snow (Scott et al. 1993; Scott and Rouse 1995).

The tundra in the Hudson Plains Ecozone represents the southernmost zone of continuous tundra vegetation and continuous permafrost in North America (Gough and Leung 2002; Abraham and Keddy 2005; Stewart and Lockhart 2005; Zhang et al. 2008), exhibiting true arctic climate characteristics at low latitudes (Rouse 1982). The ecozone's tundra landscape is comprised of a series of beach ridges created over the past ~1,500 years, in part by isostatic rebound, with extensive sedge meadows and shrub-dominated fens (underlain with continuous permafrost) forming the inter-ridge areas (Figure 51; Kershaw 1974; Rouse 1982). The soils of the beach ridges are dry, as a result of good drainage, minimum insulation from organic litter and peat, and greater exposure to harsh subarctic winds than recently emerged lowland sites along the coast or inter-ridge meadows (Larson and Kershaw 1974; Timoney et al. 1993). The most inland tundra sites comprise a forest-tundra landscape. Overall, then, the ecozone's tundra from coastline to forest is extremely heterogeneous (Larson and Kershaw 1974), consisting of heath-lichen tundra and shrub thicket on beach ridges, moss-hummock tundra on wet peat flats, sedge meadow tundra on flooded peat flats, and forest-tundra (Wrigley 1974; Crins et al. 2009).

Tundra vegetation is characteristically comprised of short-stature species, including dwarf shrubs, grasses, sedges, lichens, and mosses (Wrigley 1974; Riley 2003; Crins et al. 2009). White spruce (*Picea glauca*) is common on raised sites in forest-tundra transitional areas (Figure 52) and at the treeline (Hutton and Black 1975; Scott et al. 1987), but black spruce (*P. mariana*) and eastern larch (tamarack) (*Larix laricina*) also persist in stunted form on exposed sites (Crins et al. 2009); individual patches of *krummholz* could be very old (Riley 2003).

Figure 51. Tundra landscape, north of Thompson Point (Manitoba). Photo credit: L. Aubry, Hudson Bay Project.

Relatively little work has been done on plant communities in the tundra (cf. cataloguing flora), but generalized patterns of plant community succession have been described. Successional processes associated with beach ridges, inter-ridge areas, and forest-tundra are overviewed in Inset 6. Vegetative composition in the tundra changes along the gradient inland, and it is locally influenced by moisture (drainage), pH, nutrient availability, slope, aspect, wind, and snow accumulation. The general importance of substrate as a driver of secondary succession in the absence of fire has recently been demonstrated for the forest-tundra near Churchill (Inset 7). Fire can also be an important driver there (Brook 2001; Inset 6), helping to create the balance between tundra and forest cover (Timoney et al. 1993).

Figure 52. Tundra to forest-tundra transitional area northwest of Peawanuck, Ontario. Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Inset 6. Overview of tundra succession

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Plant community succession in the tundra of the Hudson Plains Ecozone is naturally controlled by a number of factors, and it differs for beach ridges, inter-ridge areas, and forest-tundra transitional areas, as overviewed below. Human influences on the tundra are discussed in the main text.

Beach ridges

Isostatic rebound contributes to distinct patterns of successional development in the ecozone. On young beach ridges (<300 years old) nearest the coast, dry, moderately saline soils facilitate the transition from intertidal salt marsh communities (i.e., *Puccinellia phyrganodes*–*Carex subspathacea*) to upland tundra freshwater communities. On these young ridges, white mountain-avens (*Dryas integrifolia*) and various *Saxifraga spp.* become the common woody species, and the grasses *Calamagrostis deschampsioides*, *Festuca rubra*, and *Leymus mollis* form the ground cover. Young beach ridges with low soil moisture are also colonized by lichen, including *Cetraria islandica and C. cucullata*, and by white mountain-avens (Larson and Kershaw 1974; Crins et al. 2009).

Older beach ridges (>800 years old) located further inland from the coast comprise very different vegetation communities (Kershaw 1974; Larson and Kershaw 1974). As beach ridges age, peat accumulation on the ridges, in addition to the presence of thick lichen-heath mats, serves to increase soil moisture and decrease soil pH. The resulting community consists of moisture tolerant and low pH tolerant vegetation, including *Vaccinium* spp. and lichens like *Cetraria nivalis* and *Cladina mitis* (Larson and Kershaw 1974). Furthermore, along the wet, acidic bases of these old ridges, black spruce (*Picea mariana*) is able to grow (Larson and Kershaw 1974; Payette et al. 2001).

Inter-ridge areas

Inter-ridge areas of the tundra consist mainly of sedge meadows largely underlain with permafrost (Kershaw 1974). Whereas sedge meadows closest to the coast support salt-tolerant marsh communities (Jefferies et al. 1979; Hik et al. 1992), sedge meadows increasingly become freshwater and accumulate peat along a gradient from the coast inland. Peat accumulation leads to elevation above the water table, a decrease in pH and nutrient supply, and insulation (permitting ice lens development), all of which influence the vegetation. Overall, successional processes with ageing of sedge meadows facilitate the transition of brackish environments to fen and finally to peat plateau bogs (Kershaw 1974; Brook 2001; Glaser et al. 2004). Peat plateau bogs in the tundra consist of lichen-spruce woodland, with *Ledum* spp. shrubs adding to the ground cover. Lichen species, such as *Cladonia mitis*, *C. stellaris*, *C. rangiferina*, and *C. amaurocrea*, are abundant on these plateaus. Overall, tundra sedge meadows range from permanently wet areas, at or below the water table, to areas of seasonal drying. They commonly support ericaceous shrubs, such as blueberry, cranberry, and other *Vaccinium* spp. and Labrador tea (*Ledum* spp.), while trees are generally absent (Crins et al. 2009). Dry areas are colonized by mosses (primarily *Dicranum spadiceum*), forming hummocks that can eventually rise above the water table (Kershaw 1974). Dry hummocks continue to grow and will eventually support other moss species, lichen, and even shrubs of *Salix* spp. and *Betula glandulosa* (Kershaw 1974).

Forest-tundra

More inland sites of the tundra comprise a transitional and forest-tundra landscape, with white spruce (*Picea glauca*), as well as black spruce and eastern larch (tamarack) (*Larix laricina*) (Brook 2001; Crins et al. 2009), contributing to broad community types that include white spruce forest, black spruce bogs and forest, and eastern larch treed fens (Monson 2003). Fire is infrequent, but it can still be an important driver in the forest-tundra (e.g., Brook 2001), including atop dry and exposed ridges. After a fire, treeless pockets generally tend towards lichen-shrub or lichen-spruce communities. Fires on ridges result in open ridge tops that are well-drained. Eventually, they are covered with a lichen-heath mat consisting mostly of *Salix* spp. shrubs. In depressions at the bases of ridges a few metres below, the lichen-heath habitat transitions to lichen-woodland and wet spruce-moss stands (Payette et al. 2001). Furthermore, along the edge of the forest-tundra and on tundra plains, the formation of raised hummocks through frost heave facilitates opportunistic successional development. Primary successional species such as lichen (e.g., *Peltigera rufescens*, *Physconia muscigena*, *Solorina saccata*) and moss (e.g., *Tetraplodon mnioides*) typically are the first colonizers of such hummocks, and they serve important roles in stabilizing the surface substrate, which allows the secondary colonization of vascular plants, such as *Salix* spp. (Piercey-Normore 2005).

Inset 7. **Drivers of secondary succession in the forest-tundra near Churchill, Manitoba**

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The forest-tundra of the Churchill area can be broadly divided into three vegetation communities: white spruce forest; black spruce bogs and forest; and eastern larch treed fens (Monson 2003). White spruce forest communities are usually limited in their distribution to coastal areas or inland beach and esker ridges. Black spruce communities are found inland from the coast on peat deposits underlain by permafrost, where black spruce reproduces vegetatively by layering to form multi-aged stands. Eastern larch occurs inland in pure stands, usually where there is a calcareous clay substrate, deep active layer, and a higher pH environment.

A study using dendrochronology and chrono-sequencing was conducted for the forest-tundra south of Churchill Manitoba (Monson 2003). Canonical correspondence analysis was used to examine the extent to which trends in vegetation reflected environmental variability. A number of factors were considered, including age of last disturbance and substrate.

The results indicate that, in the absence of disturbance by fire, secondary succession in the forest-tundra of the Churchill area tends to cycle on replacement, with variation in vegetative composition being principally related to substrate (Figure 53). Thus, the forest-tundra there appears to maintain its plant biodiversity in the absence of fire. At the landscape scale, changes in substrate, such as permafrost distribution and pauldification, will, over the long-term, likely play a greater role in determining vegetation patterns in the forest-tundra than fire.

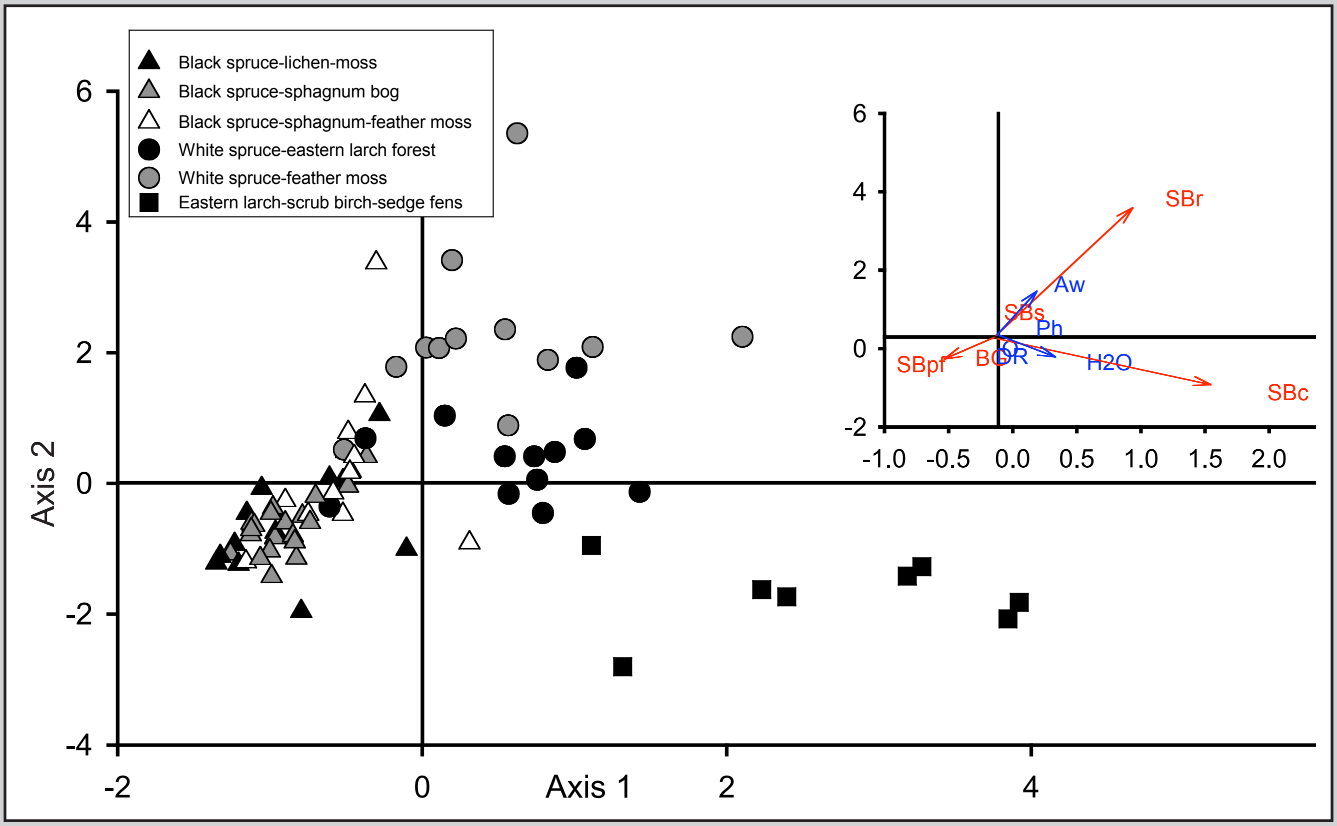

Figure 53. Results of the canonical correspondence analysis showing 81 sites along the first axes. Vectors with arrows at sharp angles are positively correlated, and the length of the vector represents the size of the coefficient. Obtuse angles between vectors indicate negative correlations. The significant variables retained at p = 0.05 are illustrated in red and are: Sbc, clay substrate; Sbs, sand substrate; Sbr, rock substrate; Sbpf, permafrost substrate; and BG, bare ground. Passive variables are shown in blue: pH; AW, west aspect; O, organic soil; DR, drainage; and H_2O, % surface water.
Source: After Monson (2003).

2.2.2.2.1 **Trends & human influences**

Data are insufficient to support quantitative analysis of trends in extent or condition of the ecozone's tundra biome as a whole (e.g., see remote sensing analysis of land cover in Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*). However, a portion of the ecozone's tundra, and in particular its freshwater marshes and fens, is affected by the excessive foraging of a greatly expanded Mid-Continent population of lesser snow goose (*Chen caerulescens careulescens*). Although the greatest impacts of lesser snow goose on the ecozone's vegetation are seen in salt marshes in the adjacent coastal biome (see Section 2.2.2.1, *Coastal*), in some cases the geese have so drastically depleted their preferred salt marsh graminoid food sources (*Puccinellia phyganodes and Carex subspathacea*) there that they have moved into the tundra to forage more in less desirable freshwater marshes and fens, with similarly devastating effects (Jefferies et al. 2003; Sammler et al. 2008). For example, local extirpation of *Leymus mollis* has sometimes occurred (Ganter et al. 1996). In response to the development of hypersaline soils in grubbed areas, *Salix* sp. shrubs have been reduced as much as 65% (Iacobelli and Jefferies 1991; Abraham et al. 2005) resulting, in turn, in declines of tundra-nesting bird populations that are located close to snow goose colonies (Jefferies et al. 2003; Rockwell et al. 2003). Sammler et al. (2008) have also shown that local nesting populations of semipalmated sandpiper (*Calidris pusilla*), dunlin (*C. alpine*), savannah sparrow (*Passerculus sandwichensis*), Lapland longspur (*Calcarius laponicus*), and other tundra-nesting passerines were more frequent in intact sedge meadow habitats than those altered by goose activity. Although no area-wide population effects were reported, it is likely that as degraded areas expand with continued goose foraging, area-wide effects will occur. High goose populations could also alter the dynamics of arctic fox (*Vulpes lagopus*) and its prey within the sedge meadow tundra, by encouraging high densities of this predator. The high predation pressure could negatively affect early ground-nesting bird species, such as shorebirds, long-tailed duck (*Clangula hyemalis*), willow ptarmigan (*Lagopus lagopus*), and arctic tern (*Sterna paradiseaea*) (Sammler et al. 2008).

Some damage to plant communities on both the drier beach ridges and wetter inter-ridge areas of the tundra is also being caused by the operation of wheeled vehicles (tundra buggies and/or ATVs) in both Manitoba (Smith et al. 1998) and Ontario (K.F. Abraham, Ontario Ministry of Natural Resources, pers. obs.) (Figure 54) (see also the *Dryas integrifolia* profile in Section 2.3.3.7, *Vascular Plants*). As described for the Manitoba portion of the ecozone (Smith et al. 1998), where wheels churn the peat surface a change in albedo leads to a deepening of the active layer. Subsidence through compression of the surface can then lead to the formation of pools and accelerated thermokarsting. Larger multi-passenger tundra buggies are used by the polar bear tourism industry near Churchill, which has grown from a few vehicles in the 1960s and 1970s to 18 vehicles that now collectively annually accommodate an estimated 8,000 visitors over 40-60 days in October and November (Dawson et al. 2010).

Currently, no strong evidence exists for climate-driven changes to the ecozone's tundra, even if some indications are apparent. Overall, there is little evidence that the ecozone's treeline is moving north (e.g., Scott et al. 1987), as has been documented in some other northerly locations in Canada and the world (Harsch et al. 2009). However, the treeline in this ecozone has also received relatively little direct study. Recently, Ballantyne (2009) documented increases of 12.6% and 6.9% of shrub and tree cover, respectively, in a 2.55 km^2 study area just north of the functional treeline at Churchill, Manitoba. As well, a long-term, non-successional change or trend involving partial degradation and conversion of frozen peat plateau bogs to fens is

Figure 54. An example of ATV damage to wet tundra, near Fort Severn, Ontario.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

suggested in an area from the Nelson River north to Churchill (Dyke and Sladen 2010). Permafrost thaw is suspected in this ecozone, but it cannot be confirmed due to insufficient trend data (see Section 2.1, *Abiotic Drivers*).

Canada's tundra, which reaches its most southerly extent in the Hudson Plains Ecozone, is especially vulnerable to climate change. For example, thawing permafrost will likely be associated with deepening of the active layer, thermokarst erosion, and increases in soil nutrients (Henry and Molau 1997). Productivity of tundra areas could, therefore, increase as the climate warms, as appears to be occurring in the Arctic Ecozone and some other northerly areas of Canada, likely owing to increases in vascular plant productivity in particular (cf. lichens might be drying) (Pouliot et al. 2009; Ahern et al. in press). As temperature increases and growing seasons lengthen, some plant species could also experience changes in vegetative (budburst) and/or reproductive phenology that contribute to changes in tundra plant community dynamics (Henry and Molau 1997; see also Section 2.4.3.4.2, *Plant Phenology*). The composition of tundra plant communities could likewise be affected by higher winter snowfalls (Scott and Rouse 1995).

Tundra ecosystems elsewhere have experienced compositional changes in the recent past, and they will likely continue to do so with climatic warming. Anticipated changes include shifts in dominant species and species ranges, such as the invasion of low arctic tundra by subarctic species (Henry and Molau 1997). Such shifts could ultimately lead to the conversion of tundra into forest, as local tundra sites gradually fill with colonizing trees (Payette et al. 2001).

Loss of tundra would likely affect many species in the Hudson Plains Ecozone. For example, at least 55 species of the ecozone's vascular flora are currently restricted to the tundra (Riley 2003). Affected bird species would include some waterfowl and waterbird species, such as lesser snow goose, red-throated loon (*Gavia stellata*), and Canada goose (*Branta canadensis*) – as well as many other birds not found elsewhere in Ontario, such as the arctic-affiliated shorebirds (e.g., whimbrel, *Numenius phaeopus*), which are largely dependent on tundra habitat for nesting (Ballantyne 2009). The Hudsonian godwit (*Limosa haemastica)* occurs mostly in the Ontario portion of this ecozone, with as much as 50% of the Canadian breeding population (Ross et al. 2003). Migratory caribou (*Rangifer tarandus*) use these areas to calve (Thompson and Abraham 1994; Abraham and Thompson 1998). Tundra also provides important habitat for arctic-affiliated mammals, such as polar bear (*Ursus maritimus*), arctic fox (*Vulpes lagopus*), and Richardson's collared lemming (*Dicrostonyx richardsoni*) (MDNR 1978).

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2.2.2.3 **Forests (boreal)**

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The boreal forests[26] (including transitional taiga forests) of the Hudson Plains Ecozone form an important part of the largest intact tract of forest in Canada, which is also considered one of the largest intact forests remaining in the world (World Resources Institute 2010). However, the Hudson Plains Ecozone has a naturally lower proportion and density of forest than many other, typically more southerly, forested ecozones in Canada, including the Boreal Shield (Wulder et al. 2008; Ahern et al. in press). The concept of a true boreal biome with generally continuous forest cover does not apply to the Hudson Plains Ecozone. Rather, owing to widespread wet edaphic conditions, forests there are primarily open and often poorly delineated from the many small bodies of open water and non-forested wetlands on the landscape (Figure 55). Forest density increases from north to south (Figure 56), being lowest in the most northerly Coastal Hudson Bay Ecoregion (215) and highest in the more southerly James Bay Lowland Ecoregion (217) (OMNR 2007). At the southern end of Ecoregion 217 (Figure 57), forest density becomes similar to that of the Claybelt area in the adjacent, more southerly Boreal Shield Ecozone. The naturally heterogenous landscape in this ecozone is reflected in the fact that a large portion of the ecozone shows forest edge densities (i.e., total edge

Figure 55. Landscape photograph illustrating how forests in the Hudson Plains Ecozone are primarily open and often poorly delineated from the many small bodies of open water and non-forested wetlands on the landscape (photograph taken ~180 km west-northwest of Moosonee, 30 km south of the Albany River, Ontario). Although open forests associated with poorly drained lowland conditions dominate, closed forest stands occur throughout on better-drained embankments, slopes, flats, and riverbank levees and are more common in the southern portion of the ecozone (see Figure 57). Photo credit: R. Brook, Ontario Ministry of Natural Resources.

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[26] *Forest* is defined in the ESTR framework as "land dominated by trees" or more specifically "areas where tree crown density is greater than ~10%" (Frisk in press). This particular definition of forest does not include a minimum area and/or minimum height, which introduces some classification challenges. For example, the definition does not help resolve whether forested clumps should be viewed as small groups of closed forest within unforested fens or the matrix of forest clumps and fens should be viewed as low density forest. Under the ESTR framework, many wetland complexes, where small areas of continuous tree cover are intermixed with fen and bog (e.g., ribbed fens, palsa bogs), are likely considered forested. As well, because a detailed inventory is not available for this ecozone, the ~10% proportion of forested area in the matrix is only a coarse estimate. Other available estimates of the extent and condition of the ecozone's forests are also presented in this section.

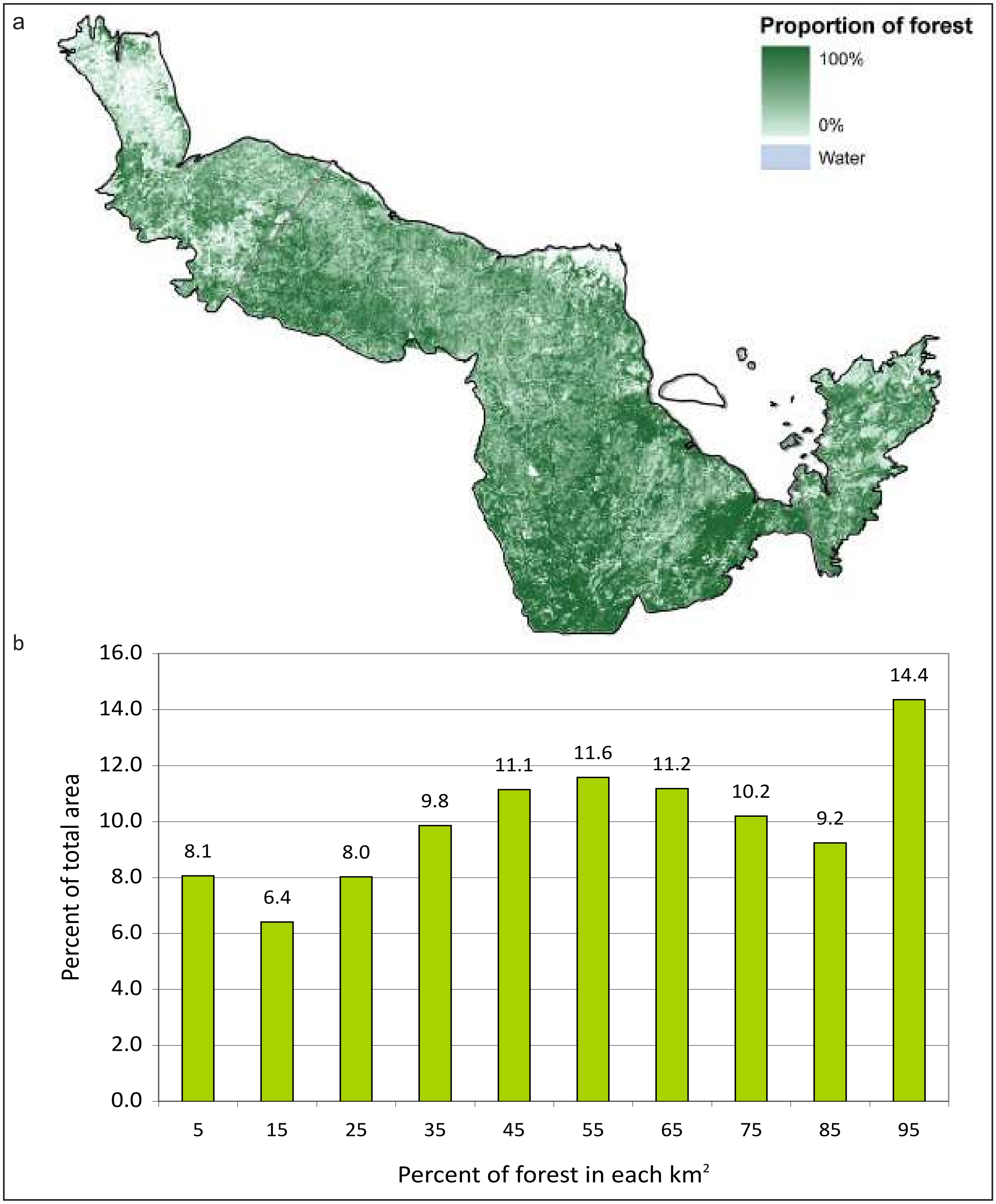

Figure 56. Distribution of forest density in the Hudson Plains Ecozone (as defined by the ecozones⁺ framework) circa 2000, calculated as the percentage of forested high spatial resolution (30 m) Landsat pixels in each 1 km² analysis unit: a) spatial distribution of forest density and b) quantitative analysis. The derivative Earth Observation for Sustainable Development (EOSD) dataset is for 1999-2002, with 90% of data for the year 2000 ± 1 year. Forest is defined as >10% tree cover; all other areas are classified as non-forest. No data are available for Akimiski Island, Nunavut.

Source: Data for ecozone provided by authors of Ahern et al. (in press); see also Wulder et al. (2008).

Figure 57. Higher density, coniferous-dominated forest in the southern portion of the ecozone (54 km northwest of Hearst and 16 km east of Kabinakagami River, Ontario).
Photo credit: S. Brinker, Ontario Ministry of Natural Resources.

density from both natural and anthropogenic processes) in the 250 m/km^2 range (Ahern et al. in press). Edge density is comparatively lower (0 to ~150 km^2) in the Boreal Shield Ecozone, where forests are more affected by fire suppression and commercial harvesting (Wulder et al. 2008; Ahern et al. in press). Forest productivity, as represented by volume per hectare, is low (42 m^3/ha) compared to the adjacent Boreal Shield Ecozone (128 m^3/ha) and even the Taiga Shield Ecozone (70 m^3/ha) (averages are calculated for ecozones$^+$ from the NFI database, and they exclude the unclassified forest type) (NFI 2010b)[27].

As of 2006, the area of land in the Hudson Plains Ecozone classified in the Canadian National Forest Inventory as *forest*[28] was 9.4 million ha (reported values are for ecozone$^+$ boundaries) (NFI 2010b). This area represents 2.7% of Canada's forest and 0.7% of Canada's total forest wood volume. Area-wise, the predominant forest types are coniferous and mixedwood; the broadleaf type represents a minor component (Table 12). Coniferous forest types also dominate on a volume and an above-ground biomass basis (Table 12). Spruce (*Picea*)-leading stands dominate on an area, volume, and above-ground biomass basis, whereas relatively few stands of other leading genera occur (Table 13).

The age class distribution of the ecozone's forests, which are essentially unaffected by either fire suppression (Section 2.4.2.2, *Fire*) or harvest (see later), is shown in Figure 58 based on ecozone$^+$ boundaries. The coniferous forest type falls primarily in the 81-100 (27.9%) and 101-120 (69.6%) year age classes, while the remaining 2.5% of this forest type is distributed in the surrounding age classes (from 1-20 to 141-160). The mixedwood forest type falls primarily in the 81-100 year age class (97.6%) and the remainder in younger age classes. The small amount of broadleaf forest type is in the 61-80 year age class (94.1%).

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[27] Only NFI photo plots (2 km x 2 km) have been established for the Hudson Plains Ecozone (https://nfi.nfis.org/plot_statistics.php?lang=en); the attributes from photo polygons are interpreted and are, therefore, only estimates (NFI 2008).

[28] For this analysis, *forest* refers to the FAO definition (FAO 2004): land spanning >0.5 ha with trees higher than 5 m and >10% canopy cover, or trees able to reach these thresholds in situ. It does not include land predominantly under agricultural or urban land use. The criterion for tree crown density in this definition of forest is similar to that in the ESTR classification framework (>~10% tree crown density), but the FAO definition also includes criteria for both minimum area and height.

Table 12. Percentage of forest stands in the Hudson Plains Ecozone (as defined by the ecozones⁺ framework) within different broad forest types. Definitions: coniferous, coniferous trees ≥75% of total; mixedwood, neither coniferous nor broadleaf trees ≥75% of total; broadleaf, broadleaf trees ≥75% of total; and unclassified, forest type missing or of an unknown age class.
Source: NFI (2010b). Analysis is based on sampling with 2 km x 2 km photo plots only.

Broad forest type	% of stands on an area basis	% of stands on a volume basis	% of stands on an above-ground biomass basis
Coniferous	54.9	75.4	70.7
Broadleaf	1.1	1.5	1.3
Mixedwood	34.6	23.1	26.3
Unclassified	9.5	—[a]	1.7

[a] A volume could not be determined for the unclassified category .

Table 13. Percentage of forest stands in the Hudson Plains Ecozone (as defined by the ecozones⁺ framework) with different leading genera.
Source: NFI (2010b).

Leading genus	% of stands on an area basis	% of stands on a volume basis	% of stands on an above ground biomass basis
Spruce (*Picea*)	87.7	92.9	91.4
Pine (*Pinus*)	0.2	0.4	0.5
Fir (*Abies*)	0.1	0.4	0.4
Larch (*Larix*)	0.1	0.1	0.1
Unspecified conifers	<0.1	<0.1	<0.1
Poplar (*Populus*)	2.4	6.1	5.9
Birch (*Betula*)	<0.1	<0.1	<0.1
Unclassified	9.5	—[a]	1.7

[a] A volume could not be determined for the unclassified category.

Notably, the concentration of the ecozone's forests in the 81-120 year age range gives the false impression that the ecozone's forests are fairly young, as is typical of areas with large and frequently recurring, stand-replacing fires. In fact, large and frequent fires are comparatively less important in this humid, cool, and wet ecozone than in the neighbouring Taiga Shield Ecozone and at least the western portion of the Boreal Shield Ecozone (Section 2.4.2.2, *Fire*). Conversely, successional self-replacement (in the absence of fire) could be relatively important (Inset 8). Indeed, in areas where fire cycles are longer, as they are in the Hudson Plains Ecozone, coniferous and mixedwood forest types tend to develop into an uneven-aged, gap-replacement condition (Bergeron and Dansereau 1993; Harper et al. 2003; Pham et al. 2004). In such cases, inventory polygon ages for tree species associated with late succession seral stage can be much younger than actual time-since-fire ages (Cyr et al. 2010).

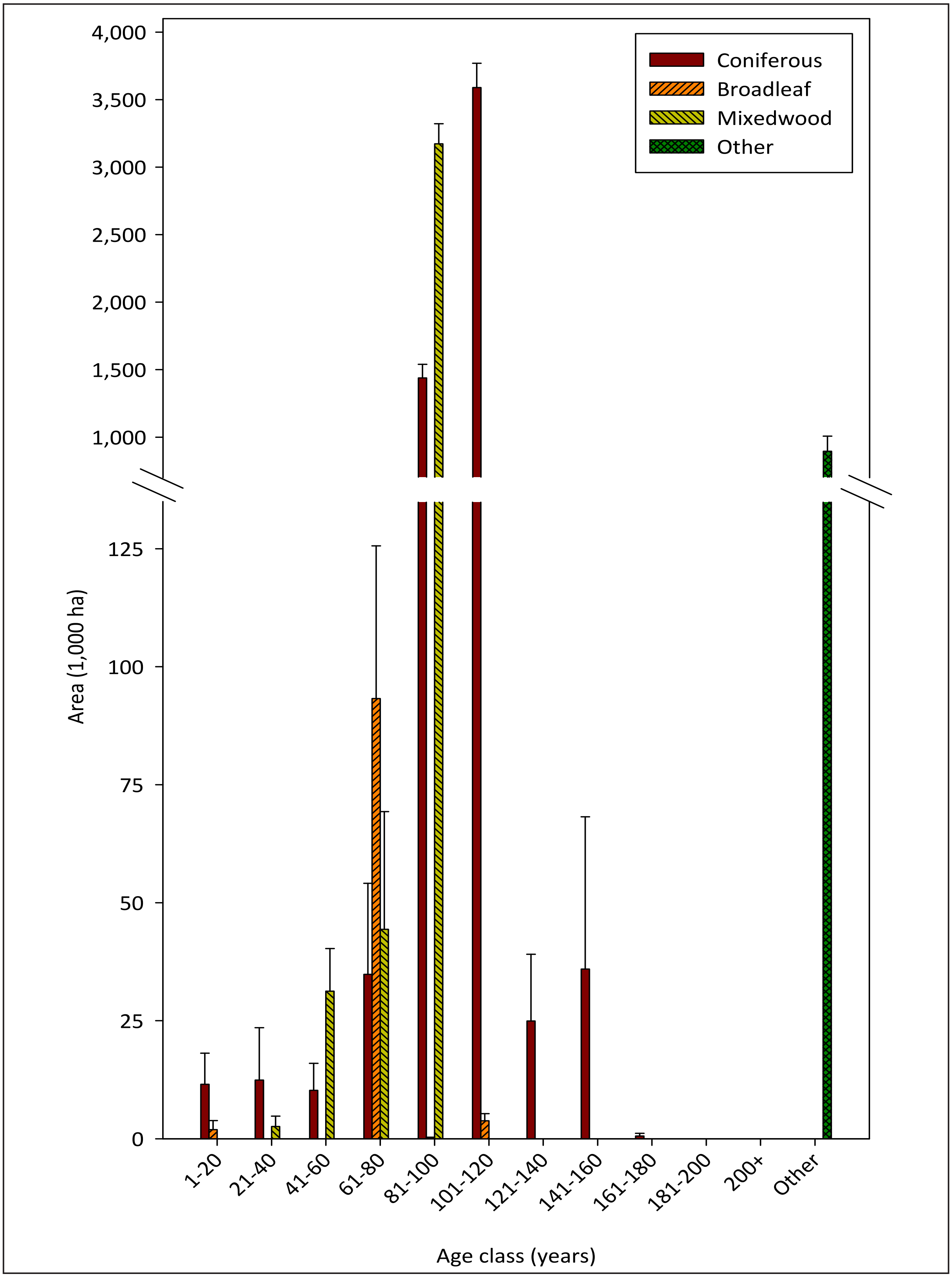

Figure 58. Distribution of the estimated total forested area (with standard error) of the Hudson Plains Ecozone (as defined by the ecozones⁺ framework) in 2006 by age class, for three broad forest types and a category (Other) with missing or unknown age classes (see text for other definitions). Total area is 9,403,074 ha. Age classes are estimated from photo interpretation.
Source: NFI (2010b).

Inset 8. **Overview of forest succession**

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Primary succession

As noted throughout this report, primary succession is expressed upon the landscape of the Hudson Plains Ecozone as a coast-to-inland chronosequence that captures a 7,000 year time frame of land shaped by glaciation and inundation of the Tyrrell Sea, followed by isostatic rebound and a general paucity of large disturbances (Glaser et al. 2004a). Along this chronosequence, the most recent terrestrial vegetation succession stage occurs at the coastal interface, and the latest succession stage occurs inland. Recognizing that succession is a complex process with various trajectories, the following description is a generalized account, with emphasis on forests, for the chronosequence given in Ritchie (1957), Riley (1982), Klinger and Short (1996), and Glaser et al. (2004a), with some description of the process by Kuhry and Turunen (2006). Forest types are described in the main text.

Poorly drained lowland sites

Original marsh communities on the coast succeed to fen communities, as a result of poor drainage and base-rich parent material. Through the autogenic process of primary peatland formation, peatland expansion takes place rapidly after glacio-lacustrine emergence (Kuhry 1998). Accumulating peatlands spread across the area coinciding with the emergence of new land from the sea, resulting in increased depth to the groundwater. The process of silvification (colonization by trees) begins with eastern larch (tamarack) (*Larix laricina*) filling-in, and the fen wetland succeeds to a treed fen community. Further peat accumulation gradually lifts the surface of the peatland above the influence of groundwater, which initiates a process of ombrotrophication, where vegetation is almost entirely reliant on precipitation for water and nutrient supply. *Sphagnum* mosses dominate, which further acidifies the forest floor. At this point black spruce (*Picea mariana*), which is tolerant of acidic conditions (Viereck and Johnston 1990), becomes the dominant tree species. In permafrost areas within the Hudson Bay Lowland Ecoregion, a peat plateau forms, and the drier forest floor results in the development of black spruce-feather moss communities or the peaty phase of spruce lichen woodland forests. Peat plateaus have aggrading and degrading stages, which might occur simultaneously on different parts of the same peat plateau (Zoltai and Tarnocai 1971). In the more southerly James Bay Lowland Ecoregion, where permafrost is classified as being sporadic-discontinuous to occurring in isolated patches (or is absent), lowland black spruce (*Sphagnum*-dominant) open forest develops and, through the process of paludification, it succeeds to an open bog (Klinger and Short 1996; Glaser et al. 2004a; Simard et al. 2007). Although peatland succession in general follows this pathway, when viewed on a regional basis these pathways are more complex and altered by local hydrogeological factors (Glaser et al. 2004b).

Well-drained upland sites

A primary succession sequence for well-drained upland sites can also be inferred from the beach ridge vegetation communities occurring from the coast to the inland portion of the ecozone. Coastal beach ridges, which are sparsely vegetated (e.g., with tundra vegetation), are replaced within a short distance of the coast by ridges of low-growing shrubs (e.g., *Salix* spp.). Further inland, a stunted and sparse growth of white spruce (*Picea glauc*a) and eastern larch is present, but in the northern Hudson Bay Lowlands Ecoregion it is soon replaced with inland beach ridges that are occupied by open stands of mature white and black spruce (Moir 1954). Still further inland, paludification processes dominate, and the beach ridges become overridden with *Sphagnum* mosses, succeeding first to black spruce lowlands and then to *Sphagnum* bog (Klinger and Short 1996; Simard et al. 2007).

Secondary succession

Secondary forest succession has received relatively little study in the Hudson Plains Ecozone, but generalized patterns can be inferred. For spruce lichen woodland forest, evidence from similar forests in the neighbouring

Taiga Shield Ecozone (eastern portion) suggests that these forests regenerate after stand-replacing fires, with various post-fire seral stages described, and they are self-perpetuating in the absence of fire (Morneau and Payette 1989; Payette and Morneau 1993; Thibault and Payette 2009). As well, successional self-replacement of forest stands has been demonstrated directly in the Hudson Plains Ecozone for the forest-tundra transition near Churchill, Manitoba (Monson 2003; see also Inset 7 in Section 2.2.2.2, *Polar-Tundra*). Similarly, if the lowland spruce forests are similar to those found on the Claybelt in the Boreal Shield Ecozone just south of the Hudson Plains Ecozone, then these forests regenerate after stand-replacing fires (Harper et al. 2002), and they are self-perpetuating at least in the short term (Harper et al. 2003), prior to the onset of paludification (Simard et al. 2007). The closed forests associated with rivers likely follow typical boreal succession patterns documented for the same forest type in the adjacent Boreal Shield Ecozone. Simplistically, after a stand-replacing disturbance from fire or flooding, balsam poplar (*Populus balsamifera*) or trembling aspen (*Populus tremuloides*) form the early seral stages, followed by deciduous-coniferous mixedwoods, and then end in self-replacing mixedwood or coniferous stands or brush fields (see Carleton and Maycock 1978; Bergeron 2000; Chen and Popadiouk 2002).

Little is known about the successional response of forests in the Hudson Plains Ecozone to natural disturbances other than stand-replacing fires, including insect disturbances (Section 2.4.2, *Natural Disturbances*).

More detailed information about the ecozone's forest types is limited, including their spatial extents, composition, condition (e.g., age class distribution and extent of insect and disease damage), and productivity. A very limited amount of additional inventory data exists for the Ontario portion of the ecozone (OMNR 2006; see also Ter-Mikaelian et al. 2009). This information, however, applies only to a very small area of productive forest within the portion of the Cochrane Area Forest Management Unit that lies within the extreme southern end of the ecozone, and it is based on very limited digital forest resource inventory (FRI)[29]. As such, it is not discussed here. Rather, aside from the inventory information described above (OMNR 2006; NFI 2010a,b) and some limited satellite-imaging results (reviewed in Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*), other information about the ecozone's forests consists of: 1) older vegetation descriptions that are akin to naturalist descriptions from field visits (Ritchie 1957); 2) field descriptions augmented by flights (Coombs 1952; Moir 1958); 3) 1950s aerial photographs (Ritchie 1960; Brokx 1965); and 4) various levels of ground sampling (Hustich 1957; Sjörs 1963; Riley 1982).

Based on this latter information, the ecozone's broadleaf, mixedwood, and coniferous forest types can be further categorically described as shown in Table 14. Considerable variation exists within forest types as a result of nuances in site variation and chance regeneration, but the descriptions are meant to portray typical conditions that capture the essence of forest stand structure in the ecozone. As already noted, the more dominant, open forest types are associated with poorly drained lowland conditions. Truly closed forest stands more typically associated with boreal forests are generally confined to better-drained embankments, slopes, flats, and riverbank levees (Coombs 1952; Sjörs 1959).

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[29] For the Ontario Hudson Bay Lowlands *forest region*, this area amounts to 286,200 ha of productive forest within 436,000 ha of the Cochrane Area Forest Management Unit, and it is based on 4.6% digital FRI.

Table 14. Description of the forested vegetation classes within each broad forest type and their general occurrence.
Source: Sources are as indicated.

Broad forest type	Forested vegetation class	Description & features	General occurrence
Broadleaf	Deciduous forest	Closed canopy forest, primarily comprised of trembling aspen, balsam poplar, and some white birch (Hills 1959; Rowe 1972; Smith et al. 1998). Balsam poplar is found on recent alluvium, particularly river islands (Ritchie 1960).	Better-drained embankments, slopes, flats, and riverbank levees associated with large riverways (Coombs 1952; Sjörs 1959) and surrounding lakes (Moir 1958). Deciduous stands are more common in the James Bay Lowland Ecoregion than in the north, where coniferous stands prevail (Hustich 1957; Sims et al. 1979).
Mixedwood	Mixedwood forest	Combination of grouped and intimate mixtures of broadleaf deciduous (primarily trembling aspen with white birch and balsam poplar on river floodplains) and coniferous trees (white spruce, black spruce, balsam fir) (Rowe 1972). Can be early- or mid-successional seral stages.	
Coniferous	Forested, closed conifer	Combination of grouped and intimate mixtures of white spruce, black spruce, and balsam fir. Balsam fir is associated with the late-successional seral stage. On river-associated landforms, black spruce with balsam fir and white spruce and a feather moss forest floor occur (Hustich 1955; Sjörs 1959). White spruce forests are found on recent alluvium, particularly river islands (Ritchie 1960). On rich riparian areas, mixed forests of balsam fir and white spruce occur (Hustich 1955).	
Coniferous	Forested, open conifer	On more northerly beach ridges that are physiographically younger and subject to a harsher climate, open stands of black and white spruce develop (Coombs 1952; Moir 1954; Hustich 1957). On the slopes of inland beach ridges, open black spruce stands develop with a moss-covered forest floor (Moir 1954). Some interior ridges that have experienced fires in the past support stands of jack pine, particularly south of the Attawapiskat River (Hustich 1955; Brokx 1965). Similarly, small isolated stands of jack pine with some white birch occur on moraine features (Moir 1958).	Inland beach ridges, moraine features.

Table 14, Cont.

Broad forest type	Forested vegetation class	Description & features	General occurrence
Coniferous	Spruce lichen woodland	On mineral soil sites, productive black spruce is the dominant tree species with associated forest floor vegetation of lichen and some mosses and dwarf shrubs (Hustich 1957; Sjörs 1961). On peat plateaus and palsas, the tree layer, ~3-4 m tall, is composed entirely of well-spaced black spruce with a ground cover dominated by *Ledum groenlandicum* and a continuous carpet of *Cladonia* species (Ritchie 1960; Zoltai and Tarnocai 1971).	Sutton Ridges. Widespread features associated with well-drained sandy soils on the crest of inland beach ridges and larger moraines in the interior. Well developed on river levees and on older, weathered beaches. Also found on peat plateaus and palsas within the Hudson Bay Lowlands Ecoregion with relatively deep peat (0.75-1.50 m) in areas raised slightly above the general level of the land (Moir 1954; Hustich 1957; Sjörs 1961; Zoltai and Tarnocai 1971; Zoltai 1973; Sims et al. 1979).
Coniferous	Black spruce feather moss	Open stands of black spruce ~4-5 m tall but with higher density than peaty lichen woodland sites (Ritchie 1960). *Ledum groenlandicum* is present, but feather mosses and some *Sphagnum* mosses dominate.	Characterized by deep peats (0.75-1.50 m), elevated up to 1.2 m above the water-saturated fen peat (Brown 1968). Widespread on peat plateaus and palsas in the Hudson Bay Lowland Ecoregion but wetter than peaty spruce lichen woodlands (Ritchie 1960; Zoltai and Tarnocai 1971).
Coniferous	Lowland black spruce, black spruce muskeg	Open forest of primarily black spruce with some eastern larch (tamarack) and a forest floor of *Sphagnum* mosses with or without mixtures of sedges and grasses (Coombs 1952; Hustich 1955). The surface tends to be hummocky due to varying depths of peat development. Tree density varies, the average height is 10 m, and trees can be grouped in clumps with numerous openings (Moir 1958).	Nutrient-poor sites with restricted drainage and thick organic soils (Coombs 1952; Moir 1958).

Table 14, Cont.

Broad forest type	Forested vegetation class	Description & features	General occurrence
Coniferous	Conifer swamps	Often dominated by black spruce with mature trees often over 4 m tall and cover usually much greater than 10%. Ground surface is uneven but sometimes with well-formed hummocks. Standing water is common in pools, channels, and depressions (Sims et al. 1982). In richer drainage ways, open forest of eastern larch and black spruce, singly or in mixtures with white spruce and eastern white cedar, occur (Hustich 1955; Sims et al. 1979).	Broad expanse of James Bay Lowland Ecoregion (Sims et al. 1979). Occurs near or along rivers and streams, often elongated features paralleling watercourses (Sims et al. 1982).
Coniferous	Treed fen	Often forming extensive uniform tracts, where eastern larch (≥10%) is dominant and regularly but widely spaced. Microtopography is flat or rolling with occasional hummocks, and standing water is in pools, where the water table is at or near the surface (Sims et al. 1982). Low density eastern larch can occur on fens associated with large shallow bays of lakes (Moir 1958). Clumped black spruce forming a pattern of raised bog islands and strings within a fen (Ritchie 1960; Sjörs 1963; Brokx 1965). Eastern larch is arranged on parallel strings of peat at right angles to the direction of drainage within fens and pools (Hustich 1957; Moir 1958; Brokx 1965).	Extensive uniform tracts not associated with river or stream drainage ways. Ribbed fens, spruce island fens.
Coniferous	Treed bog	Black spruce dominant with trees often under 3 m tall and widely spaced (≥10%). The surface is hummocky composed mainly of *Sphagnum* mosses. Standing water is occasionally found in deeper depressions among hummocks (Sims et al. 1982). Palsa bogs appear as a complex of small islands, elongate and sinuous ridges, sometimes discontinuous but more often continuous separated by open water (Hustich 1957; Moir 1958). Ombrotrophic vegetation varies from an open shrub-lichen *krummholtz* black spruce in the north to heavily wooded stands of black spruce in the south (Ritchie 1960; Sjörs 1961; Zoltai 1973; Sims et al. 1979).	Often associated with slightly drier and raised conditions; variable shape and often irregular in outline. Palsa bogs.

2.2.2.3.1 **Trends & human influences**

Information about forests in the Hudson Plains Ecozone is sufficiently limited that it supports trend analysis only in terms of forest cover. Even then, only temporally limited, coarse-scale satellite-imaging analysis of broad-scale forest changes over time is available for the ecozone as a whole (presented in Section 2.2.1.2, *Land Cover Change*). This satellite analysis suggests that only small reductions in the area of the ecozone classified as forest occurred from 1985 to 2005 (0.25%), most of which were attributable to fire, i.e., burned areas that have not yet regenerated (Ahern et al. in press, analysis based on ecozone$^+$ boundaries and the ESTR definition of forest as ~>10% tree crown cover). The resolution of data was, however, not fine enough to pick up smaller disturbances, if present, such as that due to harvest or insects. Given the coarse scale of the analysis, the errors in mapping may be greater than the small amount of change in forest cover being detected. The available forest inventory data for the ecozone (OMNR 2006; NFI 2010a,b) is not sufficient for assessing changes or trends in forest cover, nor tree species composition, age class or time-since-fire, and overall forest productivity. To obtain the information necessary to adequately assess ecological trends in the ecozone's forests, an inventory based on a systematic programme of field sampling combined with digital imagery at a resolution down to at least 1:1,000 scale is required. This level of resolution would capture events, such as collapsing palsas due to climate change, for example. In addition, an ecodistrict-edaphic framework is needed to classify forest vegetation types and otherwise replace the current simplistic vegetation community listing.

Despite the limited information available with which to assess trends in the ecozone's forests, their cover, composition, age class distribution, and overall productivity are assumed stable at this time. Disturbances to the ecozone's forests are almost all natural, and they are dominated by lightning-caused fire (see Section 2.4.2, *Natural Disturbances*). Few anthropogenic disturbances have occurred to date. Although commercial forestry is an important industry elsewhere in Canada's boreal forest (e.g., Anielski and Wilson 2009), it has not been important in this ecozone, presumably due to the low productivity of its forests, limited existing access to them, and insufficient markets. Some early harvesting occurred in the Hudson Bay Lowlands Ecoregion (216) in association with limited early mining (Moir 1958) and at the southern end of the James Bay Lowland Ecoregion (217) near the Moose and Abitibi rivers (Hustich 1957; Hutton and Black 1975). Currently, however, only a small portion of the southernmost end of the ecozone forms part of a forest management unit in Ontario, where commercial harvesting could be permitted (OMNR 2006). No Crown forest management currently occurs there, but planning for potential commercial forestry has been undertaken by the Moose Cree First Nation for the Moose Factory area (Forestry Futures Trust Committee 2007, 2009). A small amount of forest harvesting for fuel wood and building materials also occurs around communities, but harvest for these purposes has most often been confined to stands on better-drained river levees (OMNR 1985; DeBeers Canada 2005). Impacts of hydroelectric developments on forests have primarily been localized losses due to flooding (reservoir creation) (e.g., Hayeur 2001; McDonald et al. 1996). To date, the effects of mining on forested areas, including along riverbanks and creek margins, have also been localized and limited (DeBeers Canada 2005). Indeed, the Hudson Plains Ecozone is one of Canada's ecozones least affected by human development, with 97% of its area covered with *intact landscape fragments*[30] of more than 10,000 ha in 2006 (refer back to Figure 46 in Section 2.2.1.2.2, *Landscape Fragmentation*, or see Lee et al. 2006), i.e., anthropogenic forest fragmentation is also very minor.

Climate change is likely to exert an increasingly important influence on the ecozone's forests in the future. A northward shift in the treeline (and thus also a shift of forests into tundra) is possible, as discussed in Section 2.2.2.2, *Polar-Tundra*. Modeling work that examines potential changes in tree species' ranges with climate change, based on shifting climate envelopes (i.e., climatically suitable habitat), also projects that the future climate of the Hudson Plains Ecozone and other proximal northerly areas will support a greater diversity of tree species in the future (e.g., McKenney et al. 2007, 2010). In practice, however, edaphic and other constraints will also influence tree species migration with climate change (Iverson and Prasad 1998; Lafleur et al. 2010). Likely, the effects of climate change on the ecozone's forests will be somewhat constrained, because vegetation patterns there are principally controlled by local edaphic conditions rather than climate directly (see Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*). Thus, climate change might first need to cause major edaphic shifts and/or increased fire disturbance before forest communities are greatly affected (Lafleur et al. 2010; Lo et al. 2010). For example, the ecozone's large expanses of peatlands might prevent broadcast expansions of tree species like trembling aspen (*Populus tremuloides*) or white spruce (*Picea glauca*) which, unlike black spruce (*Picea mariana*), do not grow well on sites with wet, organic soils (Nienstaedt and Zasada 1990; Perala 1990; Viereck and Johnston 1990). Conversely, eastern white cedar (*Thuja occidentalis*) is more likely to migrate further north, because this species has broad tolerance to soil conditions (Johnston 1990).

Vegetation responses to climate change are also recognized as being more complex in landscapes with permafrost (Halsey et al. 1995; Camill and Clark 1998; Camill et al. 2010). If the discontinuous permafrost currently associated with the Hudson Bay Lowland Ecoregion and the mostly isolated patches of permafrost associated with the James Bay Lowland Ecoregion thaw from climate change as projected (Gough and Leung 2002), then it is anticipated that associated peat plateaus and palsas (principally in more northern sections) will collapse, the water table will rise, *Sphagnum* mosses will occupy the site, and succession will tend towards an open (non-treed) bog (Camill 1999). Under such conditions, regeneration challenges can be expected even with black spruce (primary constraints are shallow water table depth and growing moss surfaces) (Camill et al. 2010). With further climate warming, the peat could become progressively more xeric, and shifts to shrub- and tree-dominated communities might occur (Camill 1999). The James Bay Lowland Ecoregion might eventually experience increased productivity and density of lowland black spruce forests.

Because paludification is characteristic of lowlands in mid-boreal climates in the Claybelt area just south of the Hudson Plains Ecozone (Simard et al. 2007), it is also possible that the ecozone's bog communities will maintain themselves even with further climate warming. If climate

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[30] An intact forest landscape is a contiguous mosaic of naturally occurring ecosystems in a forest ecozone (i.e., contiguous blocks of forest, bog, water, tundra, and rock outcrops), essentially largely undisturbed by human influence. An intact forest landscape (Lee et al. 2006): 1) is free from substantial anthropogenic fragmentation; 2) is free from substantial human influence for periods that ensure that it is formed by naturally occurring ecological processes; 3) contains only naturally seeded indigenous plant species and supports viable populations of most native species associated with the ecosystem; and 4) is large enough to be resilient to edge effects and to survive most natural disturbance events.

change is accompanied by increasing fire disturbances, paludification might be forestalled, although on the Claybelt just south of the James Bay Lowland Ecoregion the fire cycle seems to have *increased* with the historical climate warming trend of the last century (Bergeron and Archambault 1993). Although clearly speculative, if trends in the Claybelt were repeated in the James Bay Lowlands with climate warming, then what few forests are associated with the peatlands in this ecozone might paludify and disappear. The current trend towards longer fire cycles in the Claybelt is, however, not also clearly evident for the Hudson Plains Ecozone, albeit there is some evidence for reduced dryness (decreased wildfire risk) at its most extreme southerly extent (Girardin and Wotton 2009). Although much uncertainty remains in the ecozone's future fire regime, a general increase in area burned is, however, projected (Flannigan et al. 2005; Bergeron et al. 2010). Current and future trends in the fire regime of the Hudson Plains Ecozone are discussed further in Section 2.4.2.2, *Fire*.

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2.2.2.4 **Inland waters: wetlands, rivers/streams & lakes**

Under the ESTR framework, inland waters are "permanent water bodies inland from the coastal zone and areas whose ecology and use are dominated by permanent, seasonal, or intermittent occurrence of flooded conditions"[31]. The significance, status, and trends of inland (freshwater) wetlands, rivers/streams, and lakes in the Hudson Plains Ecozone are discussed below, and the major stressors of these freshwater ecosystems are identified. Although trends in inland waters are necessarily discussed on an ecozone basis for ESTR purposes, it is recognized that drainage networks of connected systems can be better represented by drainage basin or watershed boundaries (refer back to Figure 4 in Section 1.1, *Geology, Topography & Climate* for delineation of the major river drainage areas and sub-drainage areas in and around the Hudson Plains Ecozone).

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[31] Frisk, J. *In Press*. 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.

2.2.2.4.1 **Wetlands (freshwater)**

Susan M. Tully, Ontario Ministry of Natural Resources
Kenneth F. Abraham, Ontario Ministry of Natural Resources

Under the ESTR framework (Frisk in press), wetlands are defined as "terrain affected by water table at, near, or above the land surface (maximum depth <2 m), which is saturated for sufficient time to promote wetland or aquatic processes" (also NWWG 1997). Promotion of wetland or aquatic processes is indicated by hydrolic soils, hydrophytic vegetation, and various kinds of biological activity that are adapted to the wet environment (Tarnocai 1980; NWWG 1988, 1997). Wetlands generally form in areas with relatively flat terrain, where precipitation exceeds evapotranspiration, and the underlying soil or bedrock has low permeability (Mortsch 1998).

Canada has approximately 24% of the world's wetlands (covering roughly 1.5 million km^2 or ~16% of the country's total land area), which is the greatest area of wetlands in the world (NWWG 1997). The Hudson Plains Ecozone is Canada's largest wetland complex and the third largest wetland in the world (Fraser and Keddy 2005). The extensive wetlands in this ecozone (Figure 59) are a function of poor surface drainage that is associated with low relief, impermeable clayey soils, and ground ice in permafrost. Surrounded by the Canadian Shield, the wetlands of the Hudson Plains Ecozone drain slowly northward into Hudson and James bays.

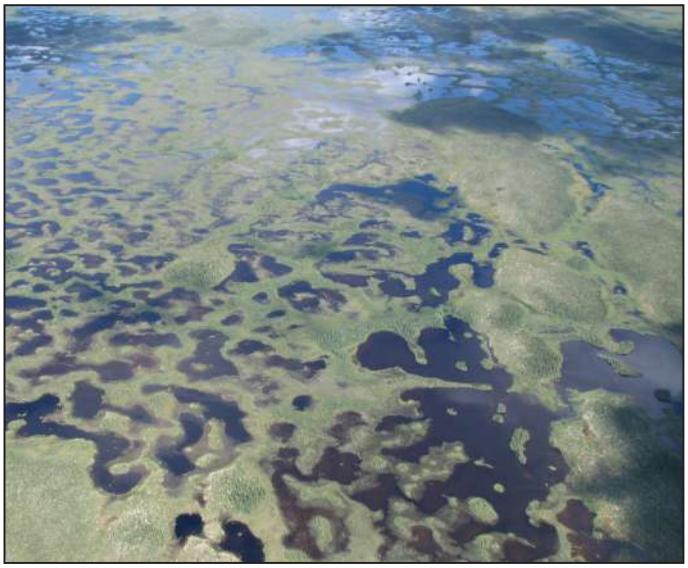

Figure 59. An example of the extensive inland wetlands of the Hudson Plains Ecozone, ~39 km northwest of Moosonee, Ontario.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Wetlands have important ecological roles that give them both global and local significance. Wetlands play an important role in the cycling of carbon through carbon sequestration in peat, methane production, atmospheric CO_2 exchange, and production and export of dissolved organic carbon (e.g., Gorham 1991; Moore et al. 1998; McLaughlin 2004). A large proportion of the wetlands in the Hudson Plains Ecozone are peat-forming wetlands (bogs and fens), making this ecozone North America's largest peatland complex and the second largest at northern latitudes (>40-50^o) (Tarnocai and Stolbovoy 2006). As such, the ecozone's wetlands have an important role in global carbon cycling and climate regulation (sections 2.4.4, *Carbon-Cycling* and 2.5.2.1, *Climate Regulating Services*), with high water levels and cold temperatures currently limiting oxidative processes and the associated release of carbon. At more regional and local scales, wetlands also play important roles in flood control (storing and slowing the flow of floodwaters) and water filtration (trapping pollutants, bacteria, sediment, and excess nutrients) (Zedler and Kercher 2005; Schindler and Lee 2010).

Globally and locally, wetlands also provide important habitat for a great diversity of flora and fauna (NWWG 1988; Abraham and Keddy 2005; Zedler and Kercher 2005). For example, the extensive wetlands of the Hudson Plains Ecozone provide critical habitat for many breeding bird populations (Gillespie et al. 1991). Two sites, Polar Bear Provincial Park and the Southern James Bay Migratory Bird Sanctuaries (Moose River and Hannah Bay), both with inland (freshwater) wetlands, have been designated Wetlands of International Importance (Ramsar sites) (Wetlands International 2010), because of the staging and breeding habitat these ecosystems respectively provide for geese, dabbling ducks, and tundra swans (Gillespie et al. 1991). Some bird species of national conservation concern (e.g., short-eared owl, *Asio flammeus* and yellow rail, *Coturnicops noveboracensis*) also use the inland wetlands of the Hudson Plains Ecozone (Abraham and Keddy 2005; COSEWIC 2010; see also sections 2.3.3.3, *Birds* and 2.3.2, *Trends in Species of National Conservation Concern*).

In Canada, wetlands are classified into seven regions based on similarities in topography, hydrology, and nutrient regimes, and these regions generally resemble broad climatic/vegetation zones (NWWG 1988). Wetland regions are further categorized into sub-regions based on characteristic wetland development. The Hudson Plains Ecozone includes portions of two of Canada's wetland regions (subarctic and boreal), and it is characterized more specifically by high subarctic, low subarctic, and high boreal wetland sub-regions (Figure 60).

Two general types of wetland occur in the Hudson Plains Ecozone, as elsewhere: 1) organic or peat-forming wetlands (*peatlands*, ≥40 cm peat); and 2) non-peat forming wetlands (NWWG 1988). Peat-forming wetland classes include bogs and fens, and non-peat forming wetland classes include swamp, marsh, and shallow open water (Table 15).

Within the Hudson Plains Ecozone, the relative abundance of wetland classes varies along a coast-to-inland successional gradient and from north to south. The coasts of Hudson and James bays are characterized by salt marshes (Section 2.2.2.1, *Coastal*). Progressing inland from the coast, fens develop where peat forms and accumulates. Bogs then form as greater amounts of peat accumulate, and nutrients are received only from the atmosphere (Glooschenko et al. 1994). In general, fens decrease from north to south, while bogs, swamps, and marshes increase (Riley 1982). Areas of shallow, open water occur throughout, depending on local topography. Overall, however, peatlands (bogs and fens) dominate (Riley 1982; Roulet et al. 1994), with permafrost features such as palsas and peat plateaus being important wetland forms (Abraham and Keddy 2005; Smith et al. 1998), including polygonal peat plateaus in the most northerly areas (Smith et al. 1998; Dyke and Sladen 2010). The spatial variability of wetland types currently found across the ecozone is illustrated within the provincial land cover mapping presented in Section 2.2.1 (*Overview of Ecozone Structure & Land Cover Change*). A more comprehensive inventory of the ecozone's wetlands is lacking.

Various pathways of wetland succession have been suggested in the Hudson Plains Ecozone (Sjörs 1963; Jeglum and Cowell 1982; Riley 1982; Tarnocai 1982; Glooschenko and Martini 1983; Klinger and Short 1996), but ultimately they tend to converge into bog communities (Klinger and Short 1996), with permafrost features (palsas and peat plateaus) contributing to a cyclical pattern of secondary succession between bogs and fens (Inset 9). Further characterization of wetland communities and their succession is beyond the scope of this review, where the focus is on overall status and trends. Notably, however, an in-depth analysis of wetlands (formations, subformations, physiognomic groups, dominance types, site types) and their more detailed

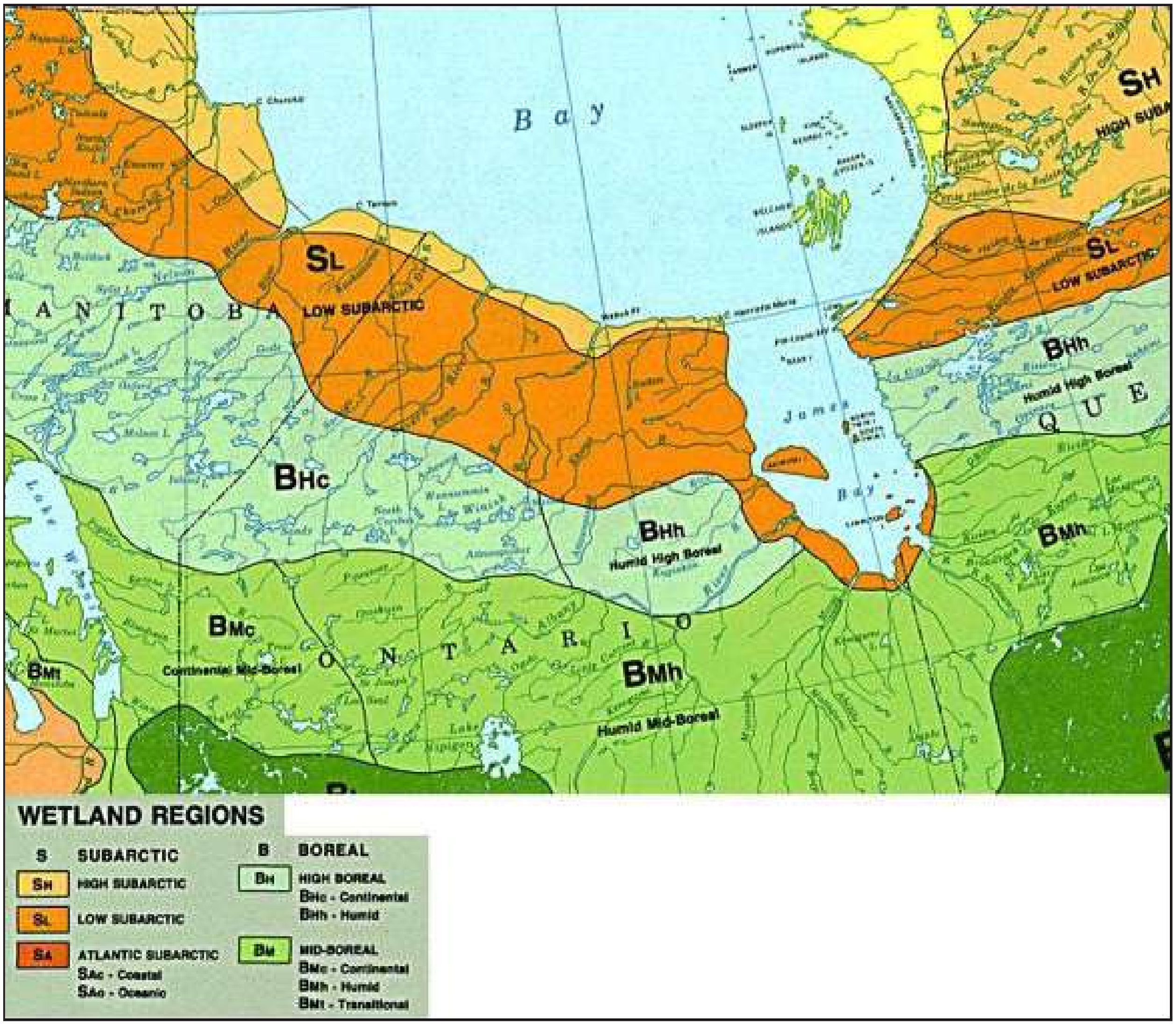

Figure 60. Wetland regions and sub-regions in and around the Hudson Plains Ecozone.
Source: Adapted from McLaughlin (2004). Reprinted from Ontario Ministry of Natural Resources Forest Research Information Paper No. 158, J. McLaughlin, under license with the Ontario Ministry of Natural Resources, copyright Queen's Printer for Ontario, 2004.

successional pathways will soon be available for the major part of the ecozone in Ontario (Riley in press; and see also Martini et al. 2009 for coastal wetlands).

The remainder of this section references only inland (freshwater) wetlands; trends in coastal salt marshes were discussed in Section 2.2.2.1, *Coastal*. Note, however, that although freshwater wetlands are considered together here (per the ESTR framework; Frisk in press), freshwater wetlands occur in other biome types, such as tundra. Further notable is that when the ESTR definition of *forest* (~>10% tree crown density, with no minimum area and/or minimum height criteria; Frisk in press) is applied in the Hudson Plains Ecozone, a high proportion of area in the ecozone that is considered wetlands in much of the published literature is classified instead as low density forest (for further discussion of this classification issue, see Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*).

Table 15. Defining characteristics of the five basic wetland classes.
Source: NWWG (1988).

Wetland class	Class description		
	Peat	Water source	Vegetative composition
Fen	Peat-forming	Ground water & overland runoff	Graminoids, nonericaceous shrubs, eastern white cedar (*Thuja occidentalis*), eastern larch (*Larix laricina*)
Bog	Peat-forming	Precipitation	*Sphagnum* mosses, ericaceous shrubs, black spruce (*Picea mariana*)
Swamp	Non-peat forming	Water table	Dense coniferous, dense deciduous forest, tall shrub thickets
Marsh	Non-peat forming	Water table	Emergent floating and submergent aquatic; reeds, rushes, and sedges
Shallow open water (<2 m maximum depth)	Non-peat forming	Precipitation	Floating, rooted, aquatic macrophytes

Trends & human influences

Although wetland loss continues nationally (Federal, Provincial and Territorial Governments of Canada 2010), few documented changes or trends are evident in the distribution, extent, or condition of inland (freshwater) wetlands in the Hudson Plains Ecozone. The ecozone's inland wetlands are, however, for the most part not being monitored[32]. Information is currently insufficient for analysis of contractions or expansions of inland wetlands at the ecozone scale (see satellite imaging-based analysis of land cover change in Section 2.2.1.2, *Land Cover Change*). Detection of any such changes by remote sensing is in any case complicated by the dynamic nature of the wetland cover in this ecozone (OMNR 2009), i.e., fluctuations in moisture conditions can cause an area to be saturated or dry, depending on seasonal or annual precipitation (Coops et al. 2008).

Nonetheless, the ecozone's freshwater wetlands are for the most part assumed healthy, with extensive peatlands largely intact (i.e., without major losses or alterations due to changed hydrology, salinization, eutrophication, sedimentation, filling, ground source contaminants, or invasive species; Zedler and Kercher 2005). There are, however, a few notable exceptions, where changes have occurred. One exception is the degradation and loss of freshwater marshes and fens that is occurring in the tundra, as a decrease in the preferred forage of overabundant lesser snow goose in the ecozone's coastal salt marshes forces the geese to move inland to nest and feed (for a discussion of this phenomenon, refer back to Section 2.2.2.2, *Polar-Tundra*). Again, the impacts that geese are having on the ecozone's wetlands are mostly attributable to human influences outside the ecozone that caused the Mid-Continent population of migratory lesser

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[32] A study is, however, underway to assess the utility of polarimetric mode aboard RADARSAT-2 for mapping and monitoring wetlands in Wapusk National Park, with a view to using these satellites operationally for monitoring and assessing the response of the park's wetlands to climate change (a project description is available in Parks Canada (2008)).

Inset 9. **Overview of wetland succession**

Susan M. Tully, Ontario Ministry of Natural Resources

In the Hudson Plains Ecozone, wetland succession tends toward climax bog communities (Klinger and Short 1996). The generalized successional trajectory from open water to fen and eventually to bog (Figure 61) can be described as follows (Tarnocai and Stolbovoy 2006). Initially, sedge- and grass-dominated peat accumulates at the edge of ponds, and it slowly moves inward until the entire surface is covered, creating an open fen. A slight slope results in patterns of ridges, creating a ribbed fen. At this stage, the peatland is minerotrophic, receiving water and minerals from groundwater sources. Over time, shrubs and trees establish, and peat continues to accumulate, becoming progressively more isolated from external water sources.

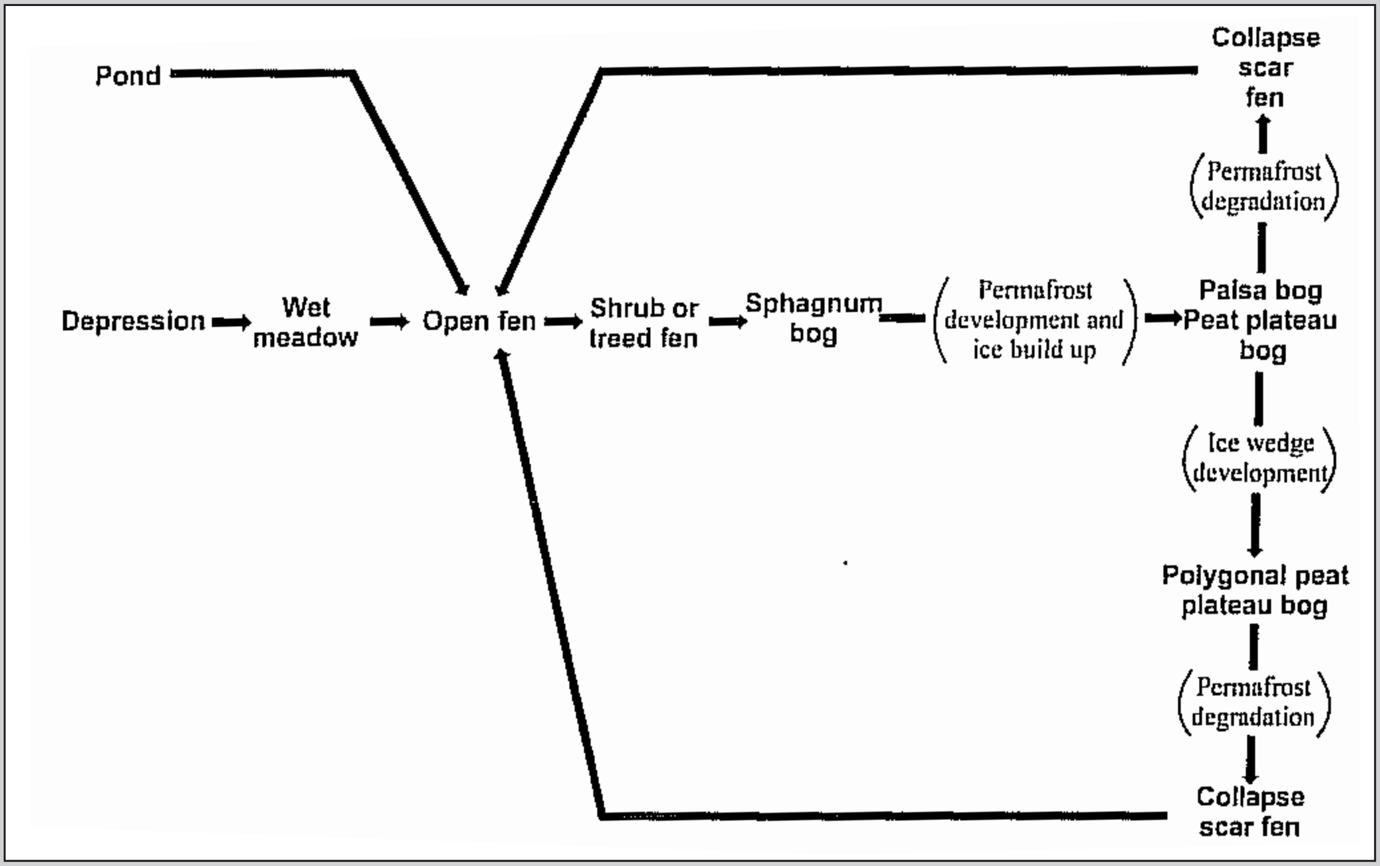

Figure 61. Schematic of wetland succession illustrating convergence on bogs and the cyclic nature of perennially frozen peatlands.

Source: Tarnocai and Stolbovoy (2006). Reprinted from Peatlands: Evolution and Records of Environmental and Climate Changes, C. Tarnocai and V. Stolbovoy, Northern peatlands: their characteristics, development and sensitivity to climate change (Chapter 2), p 34, Copyright 2006, with permission from Elsevier.

Eventually peat accumulation, now dominated by *Sphagnum* mosses, becomes sufficient to disconnect the peatland from groundwater. When water and mineral inputs occur only from precipitation, an ombrotrophic bog results. Bogs also form and expand through paludification, a process in which the accumulating or colonizing peat impedes water drainage and raises the water table.

Where permafrost occurs, frozen peat tends to develop in areas with an insulating cover of *Sphagnum* moss and black spruce. In areas of local perennial permafrost development (climate-dependent), ice lenses form that raise peat surfaces into palsas and peat plateaus. Peat plateaus can develop thermal cracks caused by ice wedges, creating a polygonal pattern (common in the northerly portion of the ecozone, e.g., Dyke and Sladen 2010). Where breaks occur in the insulating peat layer, thawing occurs and the peatland collapses, which forms a collapse scar and reinitiates the cycle of peat accumulation. Disturbances such as flooding (as with beaver activity) or fire can also set succession back to earlier seral stages.

snow goose to increase greatly over the last four decades (Abraham and Jefferies 1997; Jefferies et al. 2003; and see also Section 2.3.3.3.2, *Waterfowl*). Other exceptions include loss of wetlands or alteration of wetland classes in areas affected by a limited number of hydroelectric and mining projects, as described below. Although development pressure is increasing in this ecozone, climate change remains the greatest future threat to the ecozone's wetlands.

Hydroelectric developments

Wetlands have been lost or wetland classes have changed in some areas of the ecozone affected by hydroelectric developments (e.g., Hayeur 2001; Hydro-Québec 2004). For example, some wetlands were lost or altered by flooding in 1980 to create the 1,040 km^2 Opinaca reservoir at the eastern periphery of the ecozone, albeit when reservoir banks are gently sloping and covered with shrubs and wood debris, or when half-submerged, some small ponds and wetlands are apparently also created along the periphery (Hayeur 2001). Potential acceleration of the conversion of mercury to methylmercury (its more bioavailable form) and/or increased release of greenhouse gases (CO_2 and CH_4) are concerns when peatlands are inundated for reservoir formation (Duchemin et al. 1995; Rosenberg et al. 1997; and see Section 2.2.2.4.2, *Rivers/Streams & Lakes* for trends in fish mercury levels following creation of the Opinaca reservoir). Increased rates of methylmercury production in wetlands can also result from the conversion of air-borne mercury deposited from distant industrial sources (St. Louis et al. 1994), but the amount of such deposition in subarctic ecosystems is comparatively low (Muir et al. 2009). Conversely, in situations where hydroelectric development has reduced river flows (e.g., Eastmain and Opinaca rivers), wetlands have desiccated downstream with, for example, shrubby species expanding at the expense of pioneer wetland species (Hayeur 2001). The diversion in 2009 of 72% of the mean annual flow of the Rupert River north to the La Grande Complex is further changing wetland hydrology in the Québec portion of the ecozone (Hydro-Québec 2010).

New hydroelectric developments are also either in progress or being considered in the Manitoba (Manitoba Hydro 2010; W. Bernhardt, North-South Consultants, pers. comm.), Ontario (OPA 2007), and Québec (Hydro-Québec 2010; MDDEP 2010) portions of the ecozone (Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Mining

Mining is a relatively new human influence on wetlands of the Hudson Plains Ecozone. The ecozone's only active mine, the Victor mine, is an open-pit diamond mine ~90 km west of the mouth of the Attawapiskat River, Ontario. It was constructed beginning in 2006, opened in 2008, and is expected to operate for at least 12 years (DeBeers Canada 2005, 2008). Although it occupies a relatively small area of the ecozone (~28.8 km^2 in direct project-related developments; DeBeers Canada 2005), the potential area affected by it, like other mining operations, could be considerably larger than the mine itself. Dewatering, for example, is increasing annual recharge, possibly over an area as large as ~500 km^2 (AMEC 2008), potentially altering water balance over the life of the mine. Most of the affected area is fens and bogs (DeBeers Canada 2005).

Although a reclamation plan is in place for the mine (AMEC 2004; see also Section 2.6.3.3, *Overview of Other Restoration Initiatives*), some wetlands will not be restorable to their original classification (DeBeers Canada 2005). For example, some peatland ponds were infilled during the development of mine infrastructure. The project's mineral stockpiles and encroachment of the open pit also replaced some of the area's fens, and the fens surrounding the stockpiles (where drainage is permanently interrupted) might succeed into bog systems. Other peatland habitats have been altered by construction of winter roads and transmission lines from Attawapiskat (DeBeers Canada 2005). Some disturbed fens will be rehabilitated to forests (e.g., by revegetation of stockpiles and other now raised areas, using native seed mixes and/or seedlings) and other habitats that will support wildlife but not to their original classification. The open pit will be actively flooded at closure to create a small lake with closed drainage.

Mineral exploration is ongoing in the ecozone (Manitoba Geological Survey 2003; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates 2010; Micon International Limited 2010). The recent discovery of world-class chromite deposits inland, ~250 km west of Attawapiskat, within the Ring of Fire mineral field (Golder Associates 2010; Micon International 2010) especially portends more major mining-related infrastructure (Far North Science Advisory Panel 2010).

Climate change

Climate change is currently the most important threat to the ecozone's wetlands. Climate warming has already been documented for this area, and it is forecast to continue, along with generally drier conditions (Gagnon and Gough 2005; Section 2.1, *Abiotic Drivers*). Evaporation from wetlands is expected to increase (Rouse 1991) and, as permafrost thaws and the depth of the active layer increases, hydrology, geomorphic processes, and vegetative cover are likely to be affected (Anisimov and Nelson 1996). Much of the ecozone's wetlands are expected to be severely to extremely severely impacted (Figure 62). No climate-related changes in the extent of inland (freshwater) wetlands are yet reported in the Hudson Plains Ecozone. The possibility of such changes occurring might be suggested by a long-term (non-successional) change or trend involving partial degradation and conversion of frozen peat plateau bogs to fens in an area from the Nelson River north to Churchill (Dyke and Sladen 2010).

Predicting future changes in wetland community composition and function is challenging, particularly as permafrost thaw will likely lag behind climate warming and disturbance regimes are also likely to change (Gorham 1991; Camill 1999; Camill and Clark 2000). Changes in wetland classes imposed by climate warming are expected to vary due to differences in biotic and abiotic conditions within each class (Weltzin et al. 2003). The magnitude of response from peatland plants, for example, will depend on changes such as water table level, CO_2 levels, and nutrient availability from the soils (Moore et al. 1998). In general, however, in more northerly areas permafrost thawing is initially expected to collapse the peat, raise the water table, and form ponds (Gorham 1991). More southerly areas where permafrost is currently more limited might shift to shrub- and tree-dominated communities, as thawed peatlands become progressively more xeric as water tables decline (Gignac and Vitt 1994; Camill 1999; Camill et al. 2001). In the process, wetlands would become fragmented and be lost. If large areas of drier peatlands then burn, the increased fire activity could in turn lead to increased carbon emissions (Flannigan et al. 2009), as well as mercury emissions (Turetsky et al. 2006) (see also Section 2.4.2.2, *Fire*).

Overall, large changes are expected in the wetlands of the Hudson Plains Ecozone given continued climate warming (Tarnocai 2006), with potentially important implications for carbon cycling (Section 2.4.4) and associated climate regulating services (Section 2.5.2.1), as well as the flora and fauna that use these habitats. Changes to the distribution and structure of wetlands could, for example, affect the migration patterns; timing, distribution, breeding, and staging

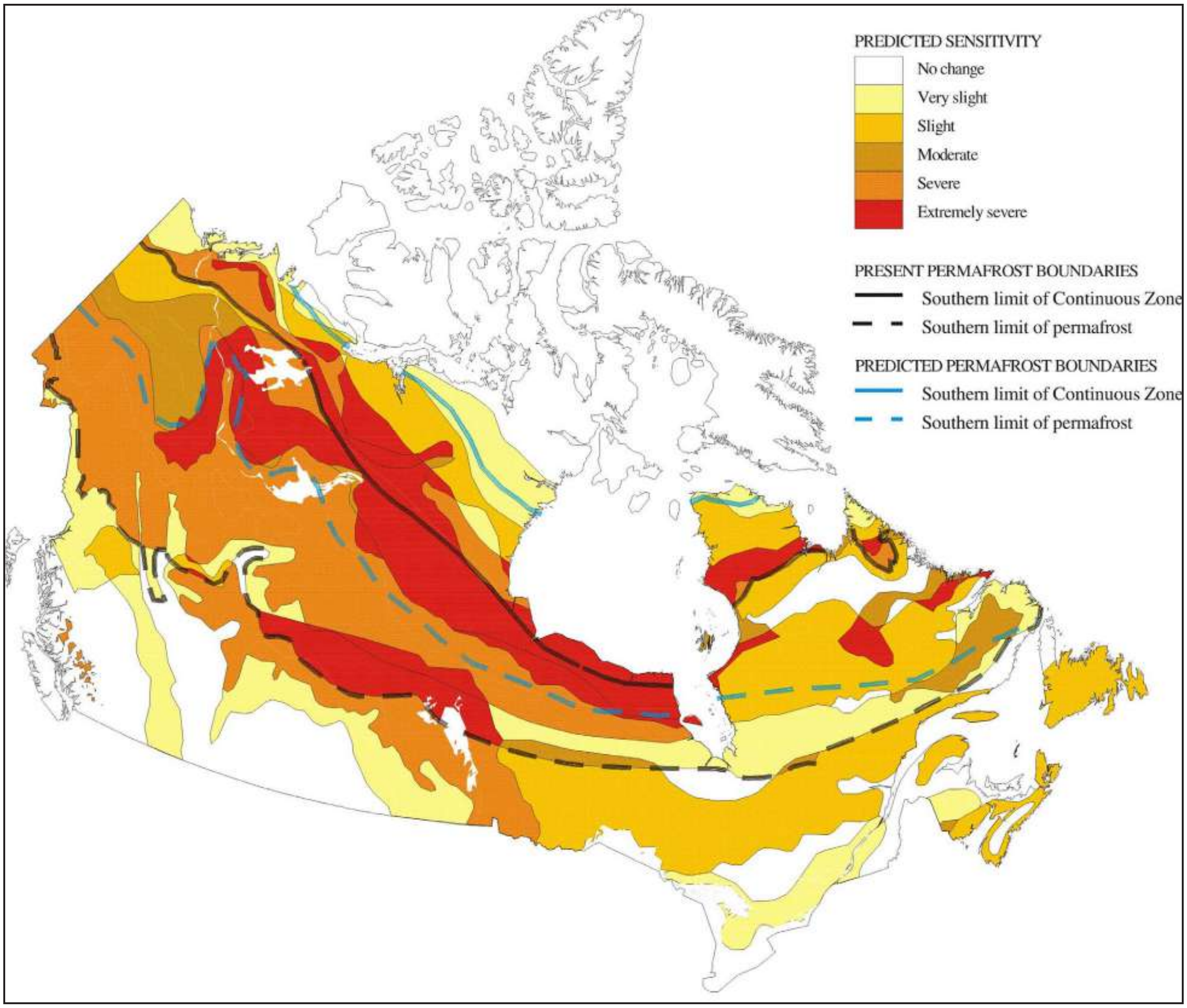

Figure 62. Peatland sensitivity map of Canada. Much of the Hudson Plains Ecozone is expected to be severely or extremely severely impacted by climate warming.
Source: Tarnocai (2006), after Kettles and Tarnocai (1999). Reprinted from Global and Planetary Change, Vol 53, C. Tarnocai, The effect of climate change on carbon in Canadian peatlands, page 224, Copyright (2006). Reproduced with permission from Elsevier BV and under license from Copibec.

resource availability; and overall abundance and diversity of inland breeding wetland-dependent birds in the ecozone (Abraham and Keddy 2005; Cadman et al. 2007).

As noted, to date no strong evidence exists for significant climate-related contractions or expansions of wetland area in the Hudson Plains Ecozone, although a long-term change or trend involving partial degradation and conversion of frozen peat plateau bogs to fens is suggested in an area from the Nelson River north to Churchill (Dyke and Sladen 2010). As climate change proceeds it will be important to track changes in the extent of the ecozone's wetlands, particularly given their importance for global carbon storage and biodiversity (Gorham 1991) and the climate-related trends and impacts being documented in other boreal and subarctic areas of Canada (Federal, Provincial and Territorial Governments of Canada (2010). Such tracking can be done with improved satellite imagery for remote sensing.

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2.2.2.4.2 **Rivers/streams & lakes**

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Rivers, streams, and lakes are an important part of the ecology of the Hudson Plains Ecozone and contribute to its relative uniqueness. This section addresses what is known about the status and trends of the ecozone's rivers/streams[33] and lakes. Under the ESTR framework, a lake is defined as a naturally occurring static body of inland water with a maximum depth ≥2 m, while static freshwater bodies with maximum depths <2 m are considered wetlands (Frisk in press). However, in the Hudson Plains Ecozone, a number of waterbodies (small lakes or ponds) with maximum depths <2 m support fish (including sport fish) populations. Several of these shallow systems also have quite large surface areas (>500 ha). Large, shallow lakes, with their wind- and temperature-affected properties and biota, could represent a relatively unique feature of the ecozone (J. Gunn, Laurentian University, pers. comm.). For these reasons, this section also makes some reference to waterbodies shallower than 2 m.

Description & status

Most of the Hudson Plains Ecozone lies within the Southwestern Hudson Bay drainage area, while smaller portions of it lie in the Western and Northern Hudson Bay, Nelson River, and Northern Québec and Labrador drainage areas (refer back to Figure 4 in Section 1.1, *Geology, Topography & Climate).* All rivers in the ecozone empty into the Hudson Bay ocean drainage basin, which is sometimes considered part of the Arctic ocean drainage basin.

Twelve major rivers drain the ecozone: the Churchill, Nelson, and Hayes rivers in Manitoba; the Moose, Albany, Attawapiskat, Winisk, and Severn rivers in Ontario; and the Harricana, Rupert, Eastmain, and Nottaway rivers in Québec. The Nelson and Churchill rivers have the largest drainage basins or systems. The smallest drainage systems are the Broadback and Harricana rivers. The Ontario portion of the ecozone alone has one primary, 13 secondary, and 30 tertiary watersheds – in which there are over 450 inland tributaries, over 305 coastal rivers and streams, and nearly 21,000 inland lakes, bogs, and ponds (OMNR 1985).

Rivers/streams

Rivers in the Hudson Plains Ecozone are shallow, slow moving, and have cut deeply into the clay and alluvial sediments; riverbanks commonly rise 5-15 m above the flood plain to form forested levees (Campbell et al. 1986) (Figure 63). Peak river flows occur in May to early June, and lows (ice-free season) historically occur in August (Déry et al. 2005). Spring floods can occur at elevations of up to 15 m above mid-summer levels (OMNR 1985; DeBeers Canada 2005). At low water, river channels usually contain pools and stretches of trapped water, and they are often blocked by shoals and bars.

River beds and banks are continually widened and deepened in many areas by the scouring action of ice and spring floods (Hutton and Black 1975). Large quantities of silt, sand, gravel, and cobbles are carried by the rivers and deposited in delta areas. These deposits, coupled with

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[33] A river/stream is defined under the ESTR framework (Frisk in press) as "a watercourse formed when water flows between continuous, definable banks. Flow may be intermittent or perennial, but does not include ephemeral flow where a channel with no definable banks is present. Gravel bars are part of a stream, while islands within a stream that have definable banks are not."

Figure 63. A section of the Albany River, near Albany Forks (Ontario).
Photo credit: D. Potvin, Ontario Ministry of Natural Resources.

the low-gradient coastal environment, have created numerous channel bars and other deposits that restrict navigation through the shallow waters (e.g., DeBeers Canada 2005). The large quantities of nutrients and organic material carried by the ecozone's many large rivers make the coastal zone (Section 2.2.2.1, *Coastal*), and especially river deltas, very productive for fish and wildlife. As well, the large volume of freshwater discharged by these rivers dilutes the saltwater in Hudson and James bays to a salinity one-third that of normal oceanic water (Prinsenberg 1982), which in turn allows this inland sea to freeze-over completely each year with profound effects on the ecozone's climate (Section 2.1, *Abiotic Drivers*).

Tidal influences in major rivers could occur 15-20 km upstream, and tidal fluctuations can average 2-3 m (OMNR 1985). The tidal rise observed in the rivers is due mainly to the *damming effect* of the tidal surge on the river, which causes the river to back up. Sea water generally does not move very far up the rivers (e.g., DeBeers Canada 2005).

Hydroelectric developments have affected the hydraulic and hydrological condition of several rivers in the Hudson Plains Ecozone. These developments are detailed later in this section (see *Trends & Human Influences*) but, overall, river channel fragmentation and/or flow regulation have strongly affected the Churchill and Nelson river systems in Manitoba, the Moose River system in Ontario, and the Eastmain and Rupert river systems in Québec (Dynesius and Nilsson 1994; Hydro-Québec 2010). Notwithstanding, some larger rivers that flow through the Hudson Plains Ecozone have remained unregulated. Specifically, the Hayes, Severn, Winisk, Attawapiskat, Harricana, and Broadback rivers are not affected.

Lakes & reservoirs

Across its expanse, the Hudson Plains Ecozone contains a multitude of shallow bog lakes and ponds (up to 41% of the surface area in some locations, such as south of Churchill; Bello and Smith 1990[34]), most of which tend to freeze to the bottom in winter (OMNR 1985; Dyke and Sladen 2010). Such lakes and ponds tend to be strongly tea-coloured and have high dissolved organic carbon, where they occur in shallow basins surrounded by large peat accumulations (Duguay and Lafleur 2003). A number of shallow coolwater lakes that support fish populations

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[34] Smaller estimates (~ <10%) have been obtained in other areas or studies, in some cases using lower resolution maps or satellite imagery that did not resolve as well the vast number of smaller lakes and ponds (for details, see Section 2.2.1.1, *Overview of Ecozone Structure*).

also occur (OMNR 1985; Schetagne et al. 2003). These shallow coolwater lakes tend to be more common where bedrock carbonates are exposed and drainage is more complete (OMNR 1985).

Lakes within the Manitoba portion of the ecozone are generally small and shallow (the largest lake there is unnamed and is ~1,900 ha in size), few support fish communities, and none support recreational fisheries (D. Macdonald, Manitoba Water Stewardship, pers. comm.). Bathymetry information, estimated from summer (1991) Landsat TM imagery (with some field verification), is available only for 16 lakes in the permafrost terrain near Churchill (Duguay and Lafleur 2003), where most lakes are small and shallow (<78 ha, <1 m depth) and likely freeze to the bottom in winter (Duguay and Lafleur 2003; Dyke and Sladen 2010). Mean depths of lakes in tundra, forest-tundra, and open forest areas from that study are 1 m, 1.4 m, and 1.8 m, respectively (Duguay and Lafleur 2003). Some larger lakes are, however, as deep as 2-3 m, and they likely do not freeze to the bottom (Duguay and Lafleur 2003; Dyke and Sladen 2010). The reservoir along the lower Churchill River is the result of a restoration effort that permanently rewatered 8.11 km^2 of river bed that had been dewatered during the partial diversion of that river (Section 2.6.3.1, *Restoration of the Lower Churchill River*).

In the Ontario portion of the ecozone, Missisa Lake is the largest lake at 19,212 ha, followed by 11 lakes at 2,000-6,000 ha, 26 lakes at 1,000-1,999 ha, 69 lakes at 500-999 ha, and 944 lakes at 100-499 ha. Ninety-five percent of the approximately 21,000 standing waterbodies there are <100 ha in size. Only 29 lakes have bathymetry information; these lakes contain sport fish populations, and they range in size from 10.1 to 19,212 ha. Two of these lakes are between 5,000 and 19,212 ha; seven lakes are between 2,500 and 4,999 ha; 14 lakes are between 500 and 4,999 ha; and six lakes are <499 ha. Maximum water depths in lakes that contain fish populations range from 1.2 to 82 m, with six of these lakes having maximum depths <2 m. Hawley and Sutton lakes are the deepest lakes, and they are located in the Sutton Ridges area.

The Québec portion of the ecozone likewise has lakes that support sport fish populations (Schetagne et al. 2003), as well as one large reservoir (Therrien et al. 2004). The Opinaca reservoir was formed in 1980 as part of the La Grande hydroelectric complex (which continues into the Taiga Shield Ecozone to the north), when the majority of flows from the Eastmain River and its tributary, the Opinaca River, were diverted to the more northerly La Grande River (Therrien et al. 2004). Physical attributes of the Opinaca reservoir are well characterized: maximum area 1,040 km^2; maximum 740 km^2 flooded land; mean depth 8.2 m; mean annual drawdown 3.6 m (max. 4.0 m); mean annual flow 845 m^3/s (to a reservoir north of the Hudson Plains Ecozone); and theoretical residence time 3.8 months.

Fish communities

The remoteness of the area's hundreds of rivers/streams and tens of thousands of small lakes and ponds has to date precluded a comprehensive survey of their component fish communities (Abraham and Keddy 2005; Browne 2007). For the bulk of the ecozone that lies in Ontario, the available fish species information is mostly dated and limited to only 41 lakes, 60 rivers, and 23 streams or approximately 6% of the water area there (Lower 1915; OMNR 1985; AMEC 2004; DeBeers Canada 2005). Information about fish populations in lakes in the Ontario portion of the ecozone is, however, expected to improve in the near future, given that this province initiated a broad-scale fisheries monitoring program in 2008[35]. The program monitors selected

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[35] Ecological Framework for Recreational Fisheries Management: http://www.mnr.gov.on.ca/en/Business/LetsFish/2ColumnSubPage/STEL02_166745.html

lakes that contain walleye (*Sander vitreus*), brook trout (*Salvelinus fontinalis*), and/or lake trout (*Salvelinus namaycush*) populations, within six Fisheries Management Zones (FMZs) of which the Hudson Plains Ecozone comprises parts of FMZs 1, 3, and 8. The intent is to conduct a fisheries population survey, as well as an aerial angler count survey and a water quality sampling survey, on each lake every 5 years (J. Amos, Ontario Ministry of Natural Resources, pers. comm.).

Fish communities have received some study in both Manitoba and Québec, mostly as part of monitoring for hydroelectric development impacts, including along the Nelson River (since the early 1980s) and Churchill River (since the 1990s) in Manitoba (W. Bernhardt, North/South Consultants, pers. comm.) and the Eastmain River, Opinaca River, Opinaca reservoir, and surrounding area (from 1973 or 1977/78 to 2000) in Québec (Verdon 2001; Therrien et al. 2004). Monitoring of fish communities in the Opinaca reservoir has been accompanied, for comparative purposes, with monitoring of fish communities in unaltered control lakes. Monitoring in the Rupert River began more recently (Environnement Illimité 2010a,b). None of the lakes in the Manitoba portion of the ecozone are large enough to merit survey or monitoring of their fish communities, although fish communities have been assessed for some small ponds near Churchill (D. Macdonald, Manitoba Water Stewardship, pers. comm.).

Overall, the available information indicates that northern pike (*Esox lucius*), walleye, lake whitefish (*Coregonus clupeaformis*), cisco (*Coregonus artedi*), and sucker (*Catostomus* and *Moxostoma species*) are widespread in larger lakes and rivers in the ecozone. Lake sturgeon (*Acipenser fulvescens*) is found in the largest rivers and some lakes. Most coastal streams and rivers contain sea run brook trout, lake whitefish, and cisco, which are anadromous species and unique in Ontario. Arctic grayling (*Thymallus arcticus*) is found at the mouth of the Churchill River in Manitoba. Sea run arctic char (*Salvelinus alpinus*) is also found in the estuary areas of coastal tributaries in the northern Manitoba portion of the ecozone, typically outside of the range of brook trout (D. Macdonald, Manitoba Water Stewardship, pers. comm.).

Lake trout is known to occur in only four lakes in the Ontario portion of the ecozone, which are found in the Sutton Ridges (OMNR 1985). The nearest other lake trout lake lies 350 km southwest on the Canadian Shield, making these four northern lake trout populations significant outliers. Lake trout also occurs marginally but naturally in small numbers in the eastern (Québec) portion of the ecozone (Therrien et al. 2004). In the Manitoba portion of the ecozone, lake trout are not known to occur, nor are they considered likely to occur, in any of the typically shallow lakes found there (D. Macdonald, Manitoba Water Stewardship, pers. comm.).

Portions of the Hudson Plains Ecozone support among the highest diversity of freshwater fish species in Canada (Chu et al. 2003; Abell et al. 2008). More information on fish species diversity and individual fish species of special interest in this ecozone is provided in Section 2.3.3.4, *Fish*.

Trends & human influences

The ability to track hydrometric trends in rivers/streams and lakes within the Hudson Plains Ecozone is very limited, owing to the paucity of reference hydrometric gauging stations located there (Monk et al. in press). In fact, the Hudson Bay watershed is recognized as having one of the most deficient stream flow networks in Canada (Mishra and Coulibaly 2010). No clear trends in overall river flow and lake-level regimes (e.g., magnitude, frequency, timing, duration, and flashiness of low- and high-flow events) are evident for undeveloped waters in the Hudson Plains Ecozone, albeit this result is based on data from only two reference hydrometric stations in the ecozone with data judged useful for the analysis (Monk et al. in press). Reduced total annual volume of freshwater naturally discharged from several of the ecozone's rivers is, however, indicated in studies of the broader Hudson Bay region (Déry & Wood 2005; Déry et al. 2005; McClelland et al. 2006). These trends for reduced total volume of freshwater discharged (1964 to 2000 or 2003), disregarding rivers with hydroelectric development or correcting for them, are correlated with large-scale climate oscillations (Déry & Wood 2005; trends in climate oscillations are discussed in Section 2.1.2.1) and associated with a 4 day advance in annual peak discharge rate and a decline in peak intensity (Déry et al. 2005). Like elsewhere in Canada, changes in the hydrological regimes of some rivers in the Hudson Plains Ecozone are also associated with hydroelectric developments, including water diversions (see later).

As noted, the remoteness of the hundreds of rivers/streams and tens of thousands of small lakes and ponds in the Hudson Plains Ecozone has to date precluded comprehensive survey and monitoring of their condition. In general, however, the ecozone's rivers/streams and lakes are assumed to be relatively healthy and undisturbed compared to ecozones in more developed areas of Canada (but see later for some specific documented impacts). In fact, the rivers and lakes of northern Ontario, in which a significant portion of the Hudson Plains Ecozone exists, make up the single largest area of high fish biodiversity currently exposed to few or no human stressors in Canada (Chu et al. 2003; Browne 2007).

Although present, most forms of aquatic pollution are not currently important in this ecozone. Acid deposition (i.e., acidification of surface waters from atmospheric transport and deposition of SO_2 and NO_x) is low compared to other, more southerly and developed regions in Canada (Moran et al. 2008), including the adjacent Boreal Shield Ecozone, where acidification appears to have reduced calcium levels in some lakes sufficiently to threaten *Daphnia* (keystone herbivores) (Jeziorski et al. 2008). The Hudson Plains Ecozone does, however, have some acid-sensitive terrain (Moran et al. 2008; Jeffries et al. 2010). Mercury deposited from distant industrial sources is also comparatively low in subarctic ecosystems, albeit it has been increasing, since pre-industrial times (Muir et al. 2009), and conversion of mercury to methylmercury (its more bioavailable form) is a concern when reservoirs are inundated (see later for observations specific to the ecozone). While agriculture and other point sources of anthropogenic nutrients are negligible within the Hudson Plains Ecozone, nutrients of anthropogenic origin can enter the ecozone through hydrological connections or atmospheric transport – although again atmospheric deposition of nutrients is low in the ecozone compared to more developed areas further south (e.g., Figure 2 in Galloway et al. 2004). Little is known about trends in nitrogen or phosphorus levels within the ecozone's rivers/streams and lakes, but trends in water nutrient levels are variable within the broader Hudson Bay ocean drainage basin (Monk et al. in press).

Although few overall quantitative trends are apparent in the condition of the ecozone's rivers/streams and lakes, existing or emerging pressures on these waterbodies and their component fish communities (as discussed below) include: hydroelectric and mining developments; introduced and potentially invasive species; harvest; climate change; and water export. Some of these human influences are associated with localized changes in the ecozone's aquatic environments.

Hydroelectric developments

Hydroelectric developments are currently the principal direct human influence on aquatic resources of the Hudson Plains Ecozone, albeit the amount of hydroelectric development there is still low relative to elsewhere in Canada (Figure 64).

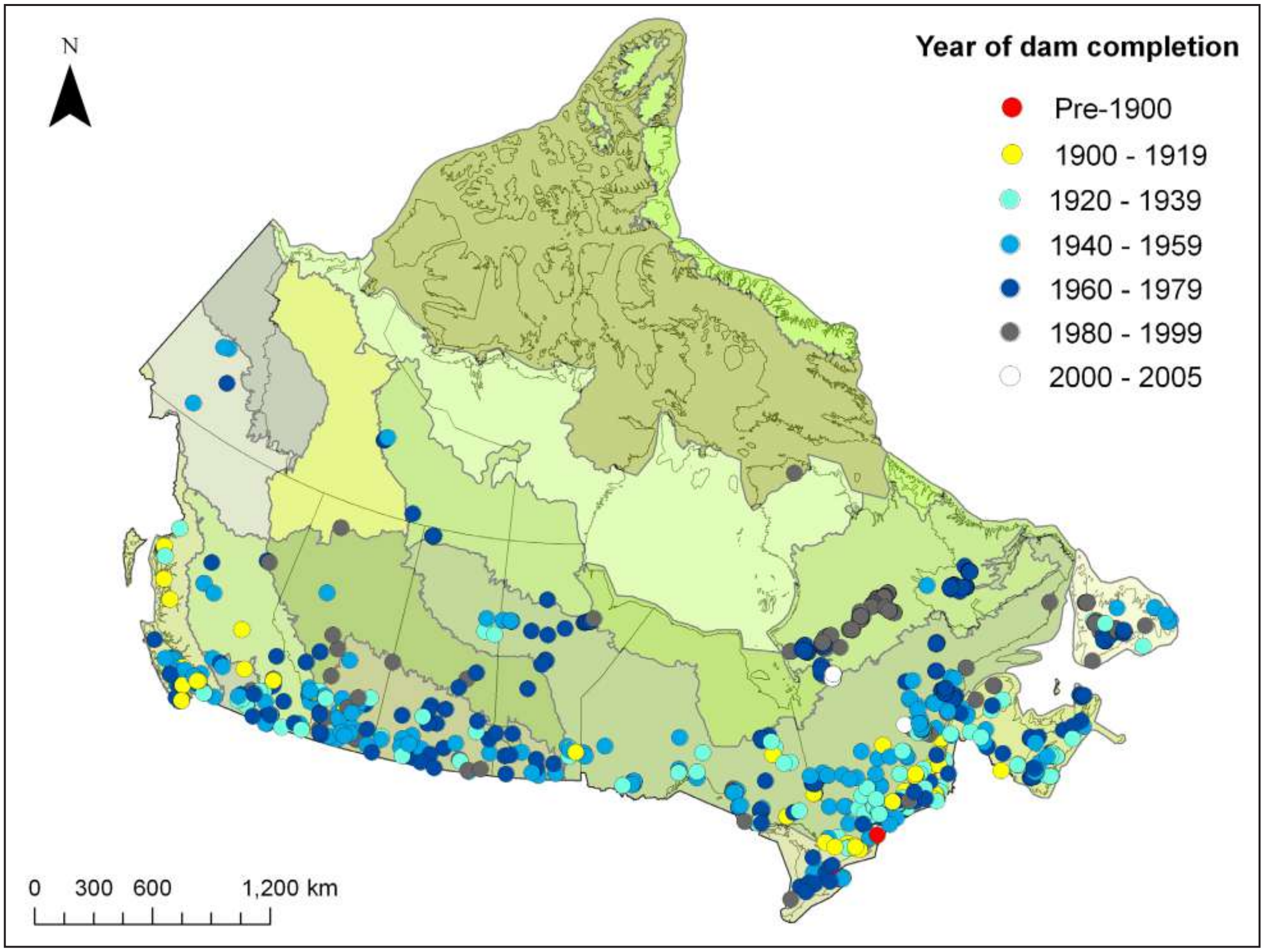

Figure 64. Spatial distribution of dams >10 m in Canada, grouped by year of completion between 1830 and 2005. Relatively few dams are located directly in the Hudson Plains Ecozone, but this ecozone is also influenced by dams located in the upstream portions of rivers (in adjacent ecozones) that flow through the Hudson Plains Ecozone before emptying into Hudson and James bays. This map does not include the new Rupert River diversion project in Québec (scheduled for completion in 2012), which has already impacted the flow of the Rupert River. All ecozone boundaries in this image correspond to the ecozones⁺ framework.
Source: Monk et al. (in press), using data from the Canadian Dam Association.

The few hydroelectric generating complexes or dam sites located directly in the Hudson Plains Ecozone are located near southern ecozone boundaries, close to the intersection with the Boreal Shield Ecozone, thereby concentrating downstream effects within the lowlands (Figure 65). Two hydroelectric generating complexes (Long Spruce, established 1976-77; and Limestone Rapids, established 1989) are located along the Nelson River in Manitoba[36]. In Ontario, one generating station (Otter Rapids, established 1961) is located on the Abitibi River (a tributary of the Moose River). Development within Québec's portion of the ecozone includes a complex of eight dam sites associated with the Eastmain River and Opinaca reservoir (established 1979-80), as part of the La Grande hydroelectric complex. The Opinaca reservoir was created by diverting the majority of the flow from the Eastmain River and its tributary, the Opinaca River, to the more

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[36] A third generating station, Kettle Rapids, is located 18 km upstream of the Long Spruce station, just outside ecozone boundaries. Daily discharge fluctuations at this site can be large (see Rosenberg et al. 1997).

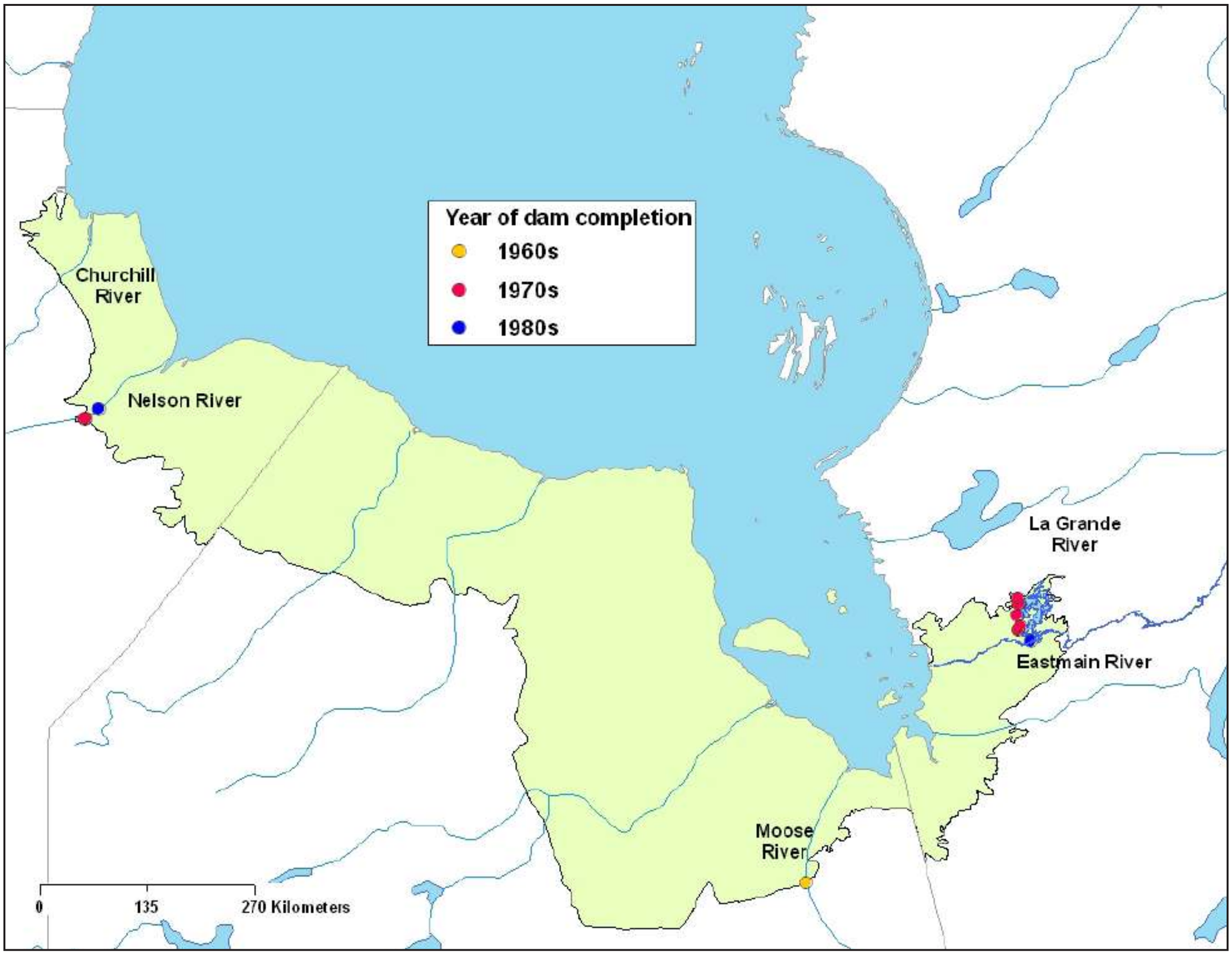

Figure 65. Spatial distribution of dam sites (>10 m height) in the Hudson Plains Ecozone to 2005, grouped by decade of completion. No dam sites were located directly in the ecozone prior to the 1960s. This analysis is based on ecozone⁺ boundaries.
Source: Data for ecozone provided by authors of Monk et al. (in press), based on data from the Canadian Dam Association.

northerly La Grande River – which resulted in flow reductions of 90% and 87% for the Eastmain River (at its mouth into James Bay) and the confluencing Opinaca River, respectively (Hayeur 2001; Therrien et al. 2004). The diversion changed otherwise turbulent water into a long series of calm lakes interspersed with short rapids and, in doing so, also increased residence time in the slowest sections from <1 day to about 20 days after the diversion – and subsequently to 40 days after five weirs were installed. Fish habitat and fish community composition changed in both the rivers and the reservoir following this development (Therrien et al. 2004).

Some hydroelectric developments that influence the Hudson Plains Ecozone are also located outside the ecozone, within the upstream portions of rivers that flow through the ecozone before emptying into Hudson and James bays (Figure 64). Particularly noteworthy is the partial diversion of waters from the Churchill River to the Nelson River[37]. The Churchill River diversion

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[37] Completed in 1977, the Churchill River diversion redirects most of the flow of the Churchill River at Southern Indian Lake into the Rat River, then the Burntwood River, and eventually through three generating stations on the lower Nelson River, two (Longspruce and Limestone) of which are located in the Hudson Plains Ecozone (Manitoba Hydro 2010a).

reduces the flow of the portion of the Churchill River in the Hudson Plains Ecozone by about 40%, from an average natural rate of discharge into Hudson Bay of 1,274 m^3/s to an average operative (with diversion) rate of 510 m^3/s (Manitoba Hydro 2010a; see also Déry et al. 2005). Another noteworthy development is the new partial diversion of the Rupert River in Québec. This project is scheduled for completion in 2012, but it began affecting the Rupert River in 2009, when 72% of its mean annual flow was diverted north to the La Grande hydroelectric complex (Hydro-Québec 2010) (but lateral flow from tributaries increases the flow at the river mouth to ~48%). The four dams associated with this project are located on the upstream portion of the Rupert River just outside of the Hudson Plains Ecozone, but some related dikes and weirs have been installed in the downstream, reduced flow portion of the river in the ecozone. Overall, river channel fragmentation and/or flow regulation have strongly affected the Churchill and Nelson river systems in Manitoba, the Moose River system in Ontario, and the Eastmain and Rupert river systems in Québec (Dynesius and Nilsson 1994; Hydro-Québec 2010). The Moose River system alone has 29 hydroelectric generating stations and water control structures on its tributaries (OPG 2010). The Albany River in Ontario and the Nottaway River in Québec are considered moderately affected (Dynesius and Nilsson 1994). The rivers in the Hudson Plains Ecozone are, thus, also affected by hydroelectric developments in adjacent ecozones via hydrological connectivity. Such developments tend to have widespread environmental effects within watersheds, well beyond the immediate vicinity of the development itself (Messier et al. 1986; Hayeur 2001).

Some larger rivers that flow through the Hudson Plains Ecozone have remained unaffected, viz., the Hayes, Severn, Winisk, Attawapiskat, Harricana, and Broadback rivers. These six river systems are among the few river systems in North America south of 55° that have remained without dams and unregulated (Dynesius and Nilsson 1994). As already noted, the Rupert River was also unregulated until very recently (Hydro-Québec 2010).

Although construction of hydroelectric facilities in the Hudson Plains Ecozone appears to have peaked in the late 1970s to early 1980s (Figure 66), the demand for power in more populated areas south of the ecozone is likely to result in further hydroelectric development of rivers in the Hudson Plains Ecozone. Two additional developments (Conawapa and Gillam Island generating stations) are being considered on the Nelson River (Manitoba Hydro 2010b; W. Bernhardt, North/South Consultants, pers. comm.)[38]. As well, seven of the 15 new hydroelectric development sites included in the Ontario Power Authority's supply mix plan for potential development by 2025 are in the ecozone – specifically on the Abitibi (4), Albany (2), and Moose (1) rivers (OPA 2007). The latter developments are, however, subject to the Northern Rivers and Moose River Basin commitments, which include a restriction on hydroelectric developments in the Severn, Winisk, Attawapiskat, and Albany (northern) rivers to ≤25 MW and require proposal by the local Aboriginal community or communities and/or their partner(s) (OPA 2007). Hydroelectric developments are also either in progress (Rupert River diversion) or being considered in Québec (Hydro-Québec 2010; MDDEP 2010).

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[38] If developed, the Conawapa generating station would be located on the Nelson River 28 km downstream of the existing Limestone Generating Station (Manitoba Hydro 2010b), and the Gillam Island generating station still further downstream on the same river. Information available for the proposed Conawapa generating station suggests it will require no significant upstream water storage and cause only limited flooding (~5 km^2) (Manitoba Hydro 2010b). Comparable details are not similarly posted for the Gillam Island station that is under consideration.

Dams have direct and indirect effects on river flow regimes, water quality, fish habitats, and fish populations and communities (Schetagne et al. 2003; Therrien et al. 2004). The impacts can occur at the site to watershed scales, and the effects can be single, multiple, or cumulative (Rosenberg et al. 1997). Dams can directly impact fish by entrainment and/or impingement (e.g., McKinley et al. 1998). They can create barriers to the movement of fish, restricting their access to habitats important to critical life stages, such as spawning (Therrien et al. 2004).

Downstream of dams, flow changes can impair fish spawning activities, lead to stranding of fish and invertebrates, change nutrient and sediment transport regimes, and cause abrupt changes in water temperature. Fish habitat can become less productive, as a result (Stull et al. 1987; Therrien et al. 2004). Indeed, species such as lake sturgeon and lake whitefish are impacted in the downstream, reduced flow portions of rivers in the ecozone that have been affected by hydroelectric developments in and around it (Therrien et al. 2004; and see Section 2.3.3.4, *Fish*). In addition to altering river flow rates, hydroelectric developments have altered the magnitude and timing of fluctuations in river flows. For example, post-development studies ~50 km downstream of the Otter Rapids generating station (near the southern boundary of the ecozone) on the Abitibi River reported diurnal water level fluctuations of 0.7-0.9 m in summer and dewatering of one-third to one-half of the river channel during low flows (Stanfield et al. 1972). More recently, Fiset (1998) observed that the effects of water level fluctuations due to the Otter Rapids generating station were apparent at least 75 km downstream of the dam. Fiset (1998) examined the lower Abitibi River based on the Serial Discontinuity concept proposed by Ward and Stanford (1983), which theorizes that hydroelectric dams reset environmental conditions

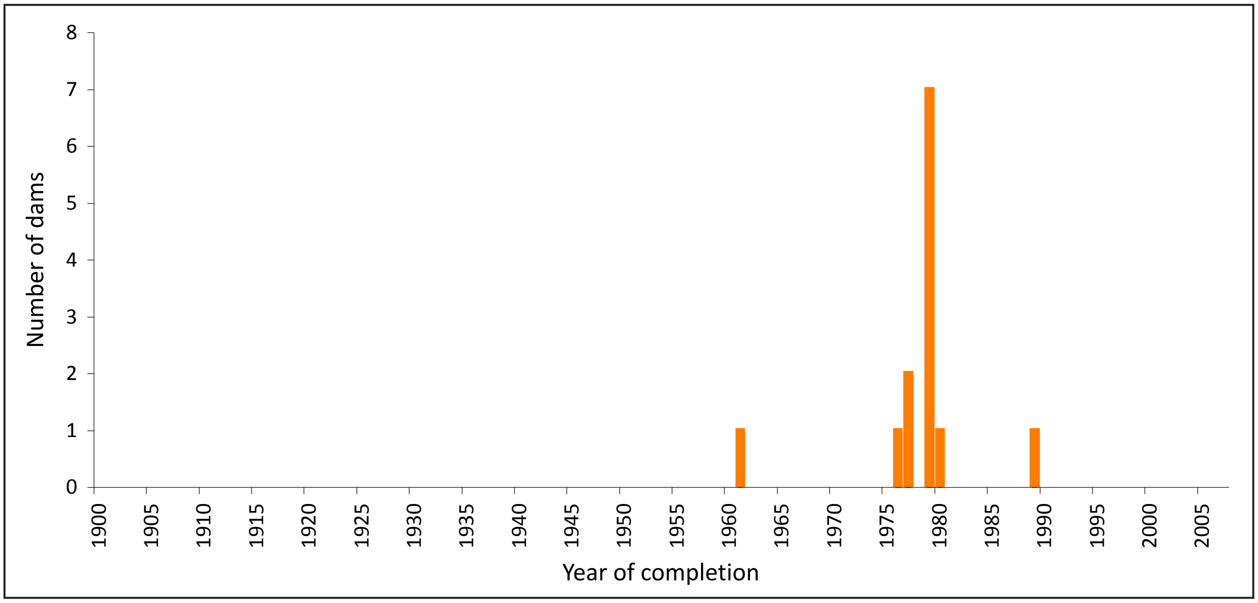

Figure 66. Trends in hydroelectric development in the Hudson Plains Ecozone, 1900-2005, based on dam structures >10 m. The first structure (Otter Rapids generating station) was completed in 1961 on the Abitibi River in Ontario. The cluster in 1976-1977 represents the Long Spruce generating complex on the Nelson River in Manitoba. The cluster in 1979-1980 represents the complex of eight dam sites associated with the Eastmain River and Opinaca reservoir. The most recent development shown (1989) is the Limestone generating complex on the Nelson River in Manitoba. This analysis is based on ecozone⁺ boundaries.
Source: Data for ecozone provided by authors of Monk et al. (in press), based on data from the Canadian Dam Association.

and aquatic communities downstream. Fiset (1998) estimated the Discontinuity Distance to be 30 km downstream of the Otter Rapids generating station, and he further noted that the greatest impacts of the generating station on benthic macroinvertebrate communities appear to occur within 9 km of the dam and the generating station.

Upstream of dams, flowing waters are typically transformed into reservoirs, resulting in loss of spawning areas (rapids), changes in water quality parameters (such as decreased pH, increased nutrient availability, reduced dissolved oxygen, and reduced turbidity), and increased erosion impacts associated with fluctuating water levels (McGovern 1983; Therrien et al. 2004). Within reservoirs, total fish yields typically decrease immediately after impoundment due to the dilution effect, but they recover gradually over time (~10-15 years) along with water quality parameters, albeit sometimes with shifts in relative species abundance. In the Opinaca reservoir, total fishing yield declined sharply after impoundment (1980), but it then increased in response to water enrichment, nearly doubling in 1983-1984, before stabilizing by 1996 at levels both near baseline and similar to a control lake (Therrien et al. 2004). The relative composition of fish species did not fully recover as, for example, both white sucker (*Catostomus commersoni*) and longnose sucker (*C. catostomus*) declined and remained low in relative abundance, while walleye, which was the predominant species prior to development at 42%, recovered from 3-8% after impoundment up to only 30%.

The prospect of increased methylmercury uptake within the aquatic food chain is one effect of particular concern when inundating new reservoirs over organic soils (flooding can promote bacterial conversion of inorganic mercury to methylmercury, the more bioavailable, organic form) (Rosenberg et al. 1997; Bodaly et al. 2007). Increased mercury concentration in fish flesh constitutes a potential human health risk, particularly to those dependant on those fish populations for subsistence food, and it could potentially impose economic impacts due to decreased marketability of the resource, e.g., mercury levels found in piscivorous fish in northern Québec often exceed the Canadian marketing standard, even in the absence of hydroelectric development (Schetagne et al. 2003). In the Hudson Plains Ecozone, mercury levels have been assessed for some fish species in some rivers and lakes (e.g., Seyler 1997; Schetagne et al. 2003; Therrien and Schetagne 2008) but mostly not monitored over time, an exception being the monitoring conducted in the Opinaca reservoir of the La Grande hydroelectric complex, since 1980 (Schetagne et al. 2003; Therrien and Schetagne 2008). Results for Opinaca show that, after inundation, methylmercury levels in reservoir waters increased and then declined to pre-impoundment values in about 8-10 years, while mercury levels in fish (bioaccumulated) have declined more gradually (Schetagne et al. 2003; Therrien and Schetagne 2005, 2008; see also Bodaly et al. 2007 for similar monitoring of fish mercury levels associated with hydroelectric developments upstream of the Manitoba portion of the ecozone), with the rate of recovery being species-dependent (Figure 67). Mercury levels were also elevated in the associated diversion but not in the reduced-flow segments of the lower Eastmain and Opinaca rivers. Any impacts of elevated mercury levels on these fish are not clear, but safe human consumption recommendations from public health institutions have been as low as two fish meals per month of piscivorous species (Therrien and Schetagne 2005) and four meals per month (occasional consumption) more recently (Therrien and Schetagne 2008). Fish mercury levels are projected to increase again in the Opinaca reservoir due to receipt of mercury exported from the recently impounded Eastmain-1 reservoir upstream, just outside of Hudson Plains Ecozone boundaries (see Therrien and Schetagne 2008).

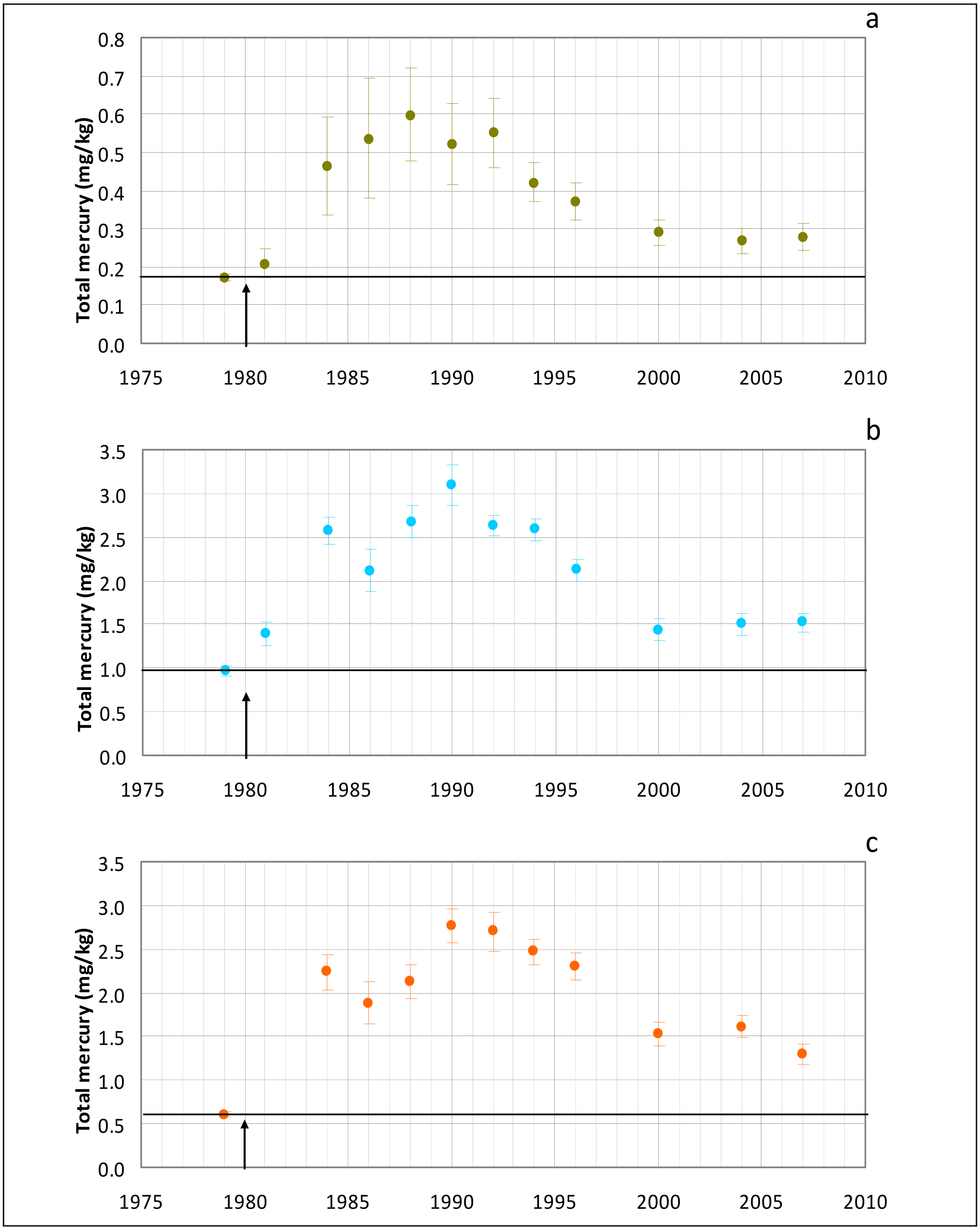

Figure 67. Trends in mercury levels (mg/kg) in the flesh of a) lake whitefish (non-piscivorous); b) walleye (piscivorous); and c) northern pike (piscivorous) in the Opinaca reservoir, 1981-2007 (note the differences in scale). Pre-inundation levels of mercury, shown as data points for 1979 (and as reference lines) represent the average natural level of mercury in these species in lakes in the area prior to reservoir creation in 1980 (arrows). All data points represent fish of standardized lengths, which are 500 mm for lake whitefish and walleye and 700 mm for northern pike.
Source: Data are from Therrien and Schetagne (2008).

Multiple dams within a river system can lead to river or watershed habitat, as well as fish community or fish population, fragmentation concerns (Rosenberg et al. 1997). Fragmentation at these larger scales, in turn, has impacts on the physical (hydraulic) diversity of these basins and can have effects on the genetic and biological diversity of aquatic ecosystems found within the watershed.

Changes in overland river flow and sedimentation that result from hydroelectric developments in and around the Hudson Plains Ecozone also have implications for salinity and water quality in the estuarine environments interfacing with Hudson and James bays. For example, the 90% reduction in flow at the mouth of the Eastmain River that is associated with La Grande hydroelectric complex led to more sedimentation (the current no longer expelled fine sediments into James Bay, and additional sediment was contributed by erosion of the exposed river bed) and a greater intrusion of saltwater into the Eastmain River estuary, with associated impacts on the fish community (Therrien et al. 2004). Marine species (sculpin, *Myoxocephalus* sp.; Greenland cod, *Gadus ogac*; sand lance, *Ammodytes* spp.) now inhabit the saltwater portion of the estuary; anadromous lake whitefish and cisco still migrate up the estuary in fall to spawn, but their over-wintering area is smaller due to the saltwater intrusion; and feeding grounds for walleye are now 5-10 km further upstream, and this species continually moves with the freshwater/saltwater interface.

Because all rivers that flow through the Hudson Plains Ecozone empty into Hudson and James bays, alteration of their natural rates of discharge or quality of freshwater by hydroelectric developments can also affect the salinity, turbidity, temperature, density stratification, ice cover, and circulation in Hudson and James bays (see the Arctic Marine Ecozones technical report of the ESTR, Niemi et al. 2010). Although Hudson and James bays are extensions of the Arctic Ocean, they function largely as a closed system (Section 2.1, *Abiotic Drivers*) and, under natural conditions, salinity in these bays is only one-third that of normal oceanic water due to the receipt of freshwater run-off from the land (Prinsenberg 1982). The impacts of changes in the discharge of individual rivers on the marine system as a whole is not straightforward as, for example, reduced discharge of the Eastmain River is accompanied by increased discharge from the La Grande River (to which flow from the Eastmain River was diverted) just outside northeastern ecozone boundaries (Therrien et al. 2004). Certainly, however, where major changes in freshwater inputs occur, at least localized effects on the marine environment are evident. For example, the 8-10 times higher winter flows of the La Grande River into James Bay is associated with a 3-5 times larger freshwater plume there under ice, for winter flows greater than 4,000 m^3/s (Therrien et al. 2004).

Concerns about cumulative impacts from hydroelectric developments are discussed further in Section 2.6.1.2, *Cumulative Impacts,* and the potential contribution of altered freshwater discharges from the La Grande hydroelectric complex to marked declines in subtidal eelgrass (*Zostera marina*) beds in the area is discussed in the eelgrass profile in Section 2.3.3.7, *Vascular Plants*.

Mining

Compared to hydroelectric developments, mining has the potential for much greater acute and chronic environmental impacts on aquatic environments, depending upon the mineral extracted and the manner in which it is processed. On the other hand, mining impacts tend to affect smaller areas than hydroelectric developments, unless multiple mines occur in the same watershed (Browne 2007). In general, mining operations can adversely affect aquatic environments by: 1) increasing human access to surrounding rivers and lakes through the construction of roads to the mining site; 2) physically altering or destroying aquatic habitats, as a result of mine construction and operation; 3) releasing polluting effluent into rivers and lakes from the mine and/or the ore processing facility; and 4) creating mine tailings that weather and leach metals and other contaminants into adjacent surface water and groundwater (Browne 2007).

Currently, one mine (Victor) is operating within the Hudson Plains Ecozone, ~90 km west of the mouth of the Attawapiskat River. Construction of this open-pit diamond mine with on-site ore processing began in 2006, and it opened in spring 2008 (DeBeers Canada 2008). The project involved the diversion of 2.6 km of South Granny Creek, as well as groundwater withdrawals from the Nayshkootayaow River for purposes of mine dewatering[39], which is being discharged to the Attawapiskat River (DeBeers Canada 2005). The mine is projected to operate for 12 years (approximate ore throughput of 2.5 million tonnes annually, or an average of about 7,000 tonnes per day) followed by a 3 year reclamation stage. A number of realized and potential impacts on aquatic environments were identified, including altered hydrology, disruption or loss of fish habitat, and changes in water quality from effluent discharge. Although quantitative data on specific ecological impacts and trends are not available at this time, it is notable that water is being added to maintain seasonal low flows in the Nayshkootayaow River. The longer-term plan following mine closure is to restore site drainage to the extent practicable, with the exception of Granny Creek (the new channel created after diversion will become the permanent creek channel) and the open pit, which will be actively flooded at closure to create a small lake with closed drainage (DeBeers Canada 2005).

A high potential exists for additional mining in the Hudson Plains Ecozone (Manitoba Geological Survey 2003; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates 2010; Micon International 2010). The intense exploration and planning for development that is currently underway at the periphery of the ecozone in the Ring of Fire mineral field especially portends additional mining-related activity (e.g., see Golder and Associates 2010 and Micon International 2010). Future mining developments can be expected to contribute additional impacts on aquatic habitat, as well as introduce infrastructure, roads, and transmission lines that inevitably lead to further human access, use, and development (McDonald et al. 1996; OMEI and OMNDMF 2009).

Introduced & potentially invasive species[40]

Introduced and potentially invasive fish species present in the Hudson Plains Ecozone include common carp (*Cyprinus carpio*), rainbow smelt *(Osmerus mordax*), and smallmouth bass (*Micropterus dolomieu*). Common carp (non-native) is a destructive bottom feeder present in the Nelson River that damages habitat for native fish by feeding heavily on vegetation and uprooting substrate (Badiou and Goldsborough 2006).

Rainbow smelt, which is a small, anadromous, and predatory non-native fish species, was illegally introduced into the Hudson Bay drainage at lakes in the Rainy and English/Wabigoon

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[39] The dewatering affects bogs, fens, ponds, creeks, smaller rivers, and riverbank and creek margin forests.

[40] Introduced species are considered to be species native or non-native to Canada that have been accidentally or deliberately introduced into habitats outside of their normal range. Invasive species are those harmful introduced species whose introduction or spread threatens the environment, the economy, and/or society, including human health (e.g., see OMNR 2005).

river systems of northwest Ontario in the 1970s or 1980s (Campbell et al. 1991; Franzin et al. 1994; Stewart et al. 2001). This species was eventually reported in Lake Winnipeg (outside the ecozone) in 1991, the lower Nelson River in 1998, and the lower Churchill River in 2002, where four smelt were documented in the stomach of a northern pike (*Esox lucius*) that was captured just upstream of tidal influence (Remnant et al. 1997; Zrum 1999). The species is now found in the Nelson River drainage and its estuary, but no smelt have been observed in the Churchill River, since being reported there in 2002, despite several attempts to capture more, i.e., the status of rainbow smelt in the Churchill River system is not currently clear (W. Bernhardt, North/South Consultants, pers. comm.). The spread of rainbow smelt is a concern for the health of native fish communities (Franzin et al. 1994; Stewart and Watkinson 2004). The species is a voracious predator of invertebrates and, therefore, it competes directly for food with many native fish species, especially lake whitefish and cisco, and preys upon their eggs and larvae. The potential impact of the spread of rainbow smelt along the coasts of Hudson and James bays and into other river systems has not been examined. Elsewhere, however, its introduction has reduced populations of pelagic and benthic planktivores and increased the growth and fat content of fish that begin eating it (Stewart and Watkinson 2004). The mercury content of predatory species, such as lake trout and walleye, might increase as a result.

Smallmouth bass is native to Canada, but it was introduced outside its natural range, including in Ontario (Brown et al. 2009). There, it was stocked into selected headwater lakes in the Boreal Shield Ecozone, since the 1920s, and it gradually spread to several river systems towards the north (Seyler 1997). This predatory warmwater species has invaded the upper Mattagami and Missinaibi river systems (Seyler 1997), and it was recently found in the Hudson Plains Ecozone in the Moose River (2008) and the lower Albany River (2009), for the first time (C. Chenier, pers. comm. and S. McGovern, pers. obs., Ontario Ministry of Natural Resources). Smallmouth bass is a concern for native fish community dynamics, because it is a strong competitor that typically has negative impacts on species such as brook trout, lake trout, and walleye (Laserby and Kerr 2000; Brown et al. 2009; Kaufman et al. 2009). Although the expansion of smallmouth bass in the Hudson Plains Ecozone is likely to be limited at present by harsh climatic and physical conditions (Seyler 1997), the species is expected to become more competitive there, as a result of climate change (see later).

Additional development in the Hudson Plains Ecozone might also facilitate introduction of other potentially invasive species as, for example, through increased use of live bait for angling and greater use of the deepwater shipping port at Churchill.

Harvest

The major form of fish harvest in the Hudson Plains Ecozone is fishing by local Aboriginal peoples. The fish harvest mainly occurs along major rivers, accessible inland tributaries, and coastal streams close to communities (Thompson and Hutchison 1987; Cummins 1991; Berkes et al. 1992; DeBeers Canada 2005). Cisco and lake whitefish have formed the major portion of the harvests, with brook trout, northern pike, walleye, sucker, burbot (*Lota lota*), and lake sturgeon being harvested as well (Thompson and Hutchison 1987; Berkes et al. 1992; DeBeers Canada 2005). Harvests are variable, partly because the availability of other animals, particularly snowshoe hare (*Lepus americanus*) and geese, appear to have a bearing on the quantity of fish harvested (Berkes 1979). Full- and part-time employment and season of employment also impact fish harvests.

Sport fishing by residents and non-residents has been less important overall than the Aboriginal food fishery (OMNR 1985; Thompson and Hutchison 1987; Berkes et al. 1992). Recreational angling is limited within the Manitoba portion of the ecozone, where it occurs almost exclusively on the major rivers and their tributaries. There, brook trout is the most vulnerable to over-harvest and special regulations are in place closing spawning rivers/streams, such as the Nelson River and its tributaries, in the fall (Government of Manitoba 2010). In Ontario, sport fishing pressure is likely high at Hawley Lake, where the annual lake trout harvest has been at the estimated sustained yield level (Scholten and Thompson 1992). It is also likely high on four of the most popular brook trout rivers, i.e., the Sutton, Brant, Shagamu, and Gorge rivers between Hawley and Sutton lakes (OMNR 1985). However, mortality rates from catch-and-release are not available, and impacts of sport fishing are otherwise unknown.

Estimates of angler fishing pressure and harvest have been very limited for the Hudson Plains Ecozone, with a few notable exceptions, including estimates for lake trout in Hawley and Sutton lakes (Scholten and Thompson 1992) and for brook trout in the Sutton River (McKnight and Hendry 1988; see also Section 2.3.3.4, *Fish*). In 1985, overall fishing pressure in the Ontario portion of the ecozone was estimated at 3,500 angler-days per year (OMNR 1985). A more recent recreational angling survey reported that about 5,000 licensed adult anglers (ages 18-65) actively fished for an estimated 77,500 angler-days in the Ontario portion of the ecozone in 2005, with additional contributions coming from an estimated 2,000 unlicensed anglers under the age of 18 and an unknown number of anglers over age 65 (Hogg et al. 2009). Such estimates should be viewed with caution, however, due to a low number of survey respondents and probable reporting bias for some areas. Nonetheless, fishing pressure in this ecozone remains very low relative to more southerly locales. If more roads are created to support additional development in the ecozone, it is expected that the provision of enhanced human access will increase fishing pressure, with negative impacts on populations of species such as lake trout, which is especially vulnerable to overexploitation (e.g., Kaufman et al. 2009).

Regulated commercial fishing for lake sturgeon once existed in each of the major rivers in the Manitoba portion of the ecozone. These commercial fisheries closed by 1994, albeit almost no harvest had occurred during the preceding three to four decades (D. Macdonald, Manitoba Water Stewardship, pers. comm.). In the Ontario portion of the ecozone, regulated commercial fishing for various species including lake sturgeon occurred until 1995, but no commercial licences have been issued since that time (Thompson 1989; Wilson 1996; Seyler 1997; OMNR 2008). Commercial harvesting had occurred on the North French, Moose, and Attawapiskat river systems and in Missisa Lake (Wilson 1996). Ontario recently issued a province-wide moratorium on the commercial, as well as recreational, harvest of lake sturgeon in response to international market pressures (see OMNR 2009). Likewise, regulated commercial fisheries apparently no longer occur in the Québec portion of the ecozone (C. Paitre, Environment Canada-Québec Region, pers. comm.).

Climate change & future range extension

Climate warming (Section 2.1, *Abiotic Drivers*) is expected to adversely affect aquatic ecosystems, including the quality of fish habitat in rivers and lakes (e.g., Schindler 2001; Prowse et al. 2006). The magnitude and timing of river flows and lake levels and water renewal times will change (e.g., timing and rate of spring melt, including ice break-up). Water quantity and quality will decline in many areas (Schindler 2001; Prowse et al. 2006), and permanent first-order streams could become ephemeral (Schindler et al. 1996). Habitats for cold stenothermic organisms will be reduced in small lakes. Warmer temperatures will affect fish migrations in some regions. Warmer temperatures could also enhance methylation of mercury (Bodaly et al. 1993), and climate change will in general interact with overexploitation, dams and diversions, habitat destruction, non-native species, and pollution to determine where fish can survive and thrive. Aquatic communities will be restructured due to changes in competition, changing life cycles of many organisms, and invasions by non-native species (Schindler 2001).

Climate warming may increase precipitation in areas of the Hudson Plains Ecozone (projections for precipitation are less unequivocal than those for temperature; see Section 2.1.2.2, *Projected Changes*), but it will also strongly enhance evaporation and evapotranspiration (Rouse 1991; Gagnon and Gough 2005). Such climatic changes, combined with more direct effects from the expected temperature increases, will impact fish communities. For example, Minns and Moore (1995) project, for temperature increases of 4.5-5.5 °C, that fish species richness in Ontario will increase from 11.7 to 60.4 species with a mean range of 31.8 for the 137 tertiary watersheds in their data set. Data analysis from Minns and Moore (1995) and Minns (1989) also suggests that for every 10 species increase in tertiary watershed richness, species richness per lake will increase by one. This implies a richness increase of one to six species per lake, which could have considerable impact given that fish species richness per lake is typically three to six throughout much of Ontario. Figure 68 shows that, for a climate warming of 4.5 to 5.5 °C, tertiary watersheds along the Hudson Bay coast and the north half of James Bay are projected to be invaded by 0-8 fish species, and watersheds in the south half of James Bay are projected to be invaded by up to 9-16 species.

As warmwater fish species move northward and their distributions overlap with native coldwater and coolwater fish species, food web interactions will change (Dove and Lewis in prep.). In many cases, these new arrivals might out-compete existing species, reducing their populations. Currently, smallmouth bass (a warmwater species) has invaded the Mattagami and Missinaibi river systems in Ontario (Seyler 1997). As already noted, the species also now appears to be advancing into the Hudson Plains Ecozone, given that one was caught by angling in the Moose River in 2008 (C. Chenier, Ontario Ministry of Natural Resources, pers. comm.) and in the Albany River in 2009 (S. McGovern, Ontario Ministry of Natural Resources, pers. obs.).

Magnuson et al. (1997) estimated that the boundaries of freshwater fish species could move north by 120 km for each 1 °C rise in air temperature. This trend could mean a 500–600 km northward expansion, by the 2050s. With mean global surface temperature increases projected at 2 °C to 4.5 °C by 2100 (IPCC 2007), large changes in the distribution of coldwater fish species in particular should be anticipated. Chu et al. (2005) project the demise of brook trout distribution in the Hudson Plains Ecozone by the year 2020, with ranges shifting northeast towards Québec-Labrador and west towards British Columbia. Other studies are more optimistic, suggesting that loss of permafrost might produce year-round groundwater flow that provides more stream habitat for brook trout (Meisner et al. 1988) or that lake trout might be able to adapt (Sellers et al. 1998) or adjust its behaviour (Snucins and Gunn 1995) to cope with increasing temperatures. Although most studies project potentially large shifts in fish species distributions, it is possible that some isolated populations of coldwater fish species could, at least for a time, remain in deep stratified lakes within their original range (Stefan et al. 1995), albeit most lakes in the Hudson Plains Ecozone are shallow (see earlier). In summary, projections based on global climate models remain to be verified. Still, it seems clear that warming will modify aquatic communities to

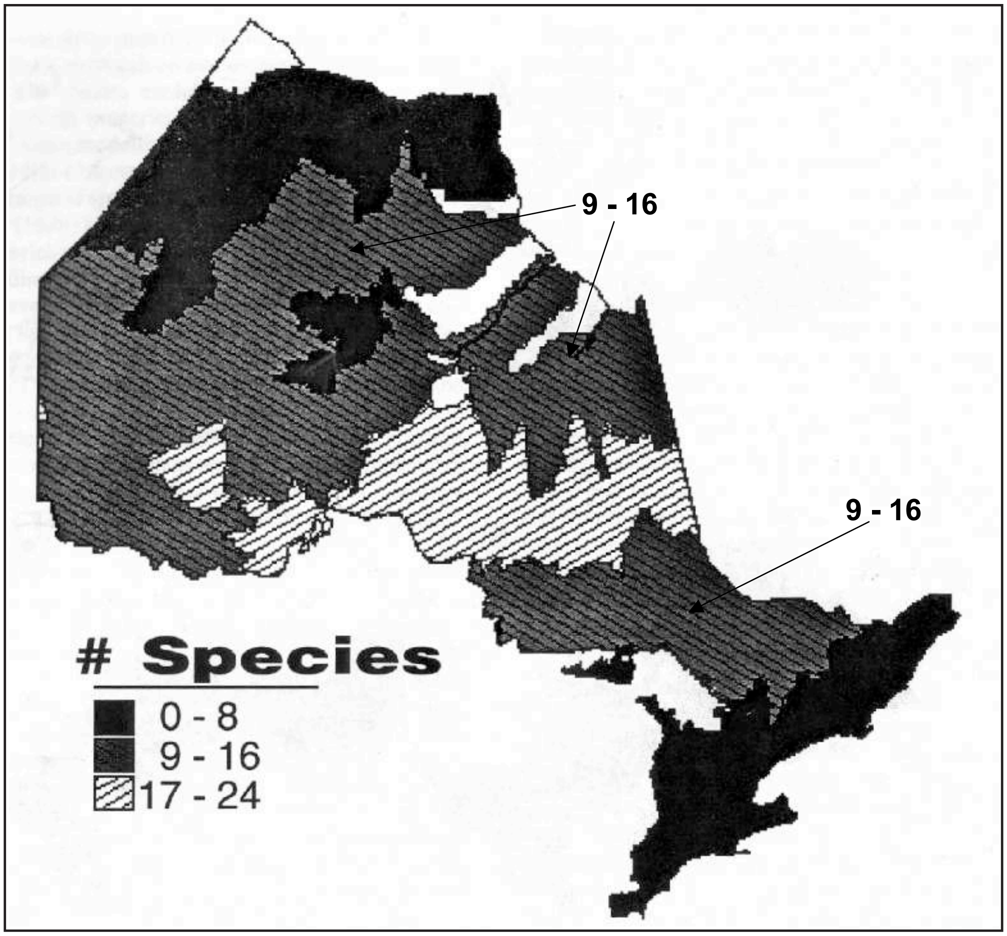

Figure 68. A map of Ontario showing those tertiary watersheds where 0-8, 9-16, and 17-24 of the 33 freshwater species with temperature-determined distribution boundaries are projected to be able to invade following climate warming of 4.5-5.5 °C.
Source: Minns and Moore (1995). Reprinted from Canadian Special Publication of Fisheries and Aquatic Sciences, Factors limiting the distributions of Ontario's freshwater fishes: the role of climate and other variables, and the potential impacts of climate change, Vol 121, 1995, p 153, C.K. Minns and J.E. Moore, Figure 6, with permission from Canadian Science Publishing, NRC Research Press.

affect interspecific interactions between predators and prey and warmwater and coldwater species (Schindler 2001). Increased temperature-dependent methylation of mercury is also a concern with climate change (Bodaly et al. 1993).

Although fish die-offs during warming events have rarely been recorded within arctic or subarctic watersheds, two such events were recently reported in the Hudson Plains Ecozone. One such major die-off of anadromous brook trout, as well as white sucker, occurred in 2001 in the lower Sutton River close to its intersection with the southern Hudson Bay coast (zone of continuous permafrost) (Gunn and Snucins 2010). Although warm air temperatures (daily maximums >30 °C) combined with unusual thermal stratification conditions in the headwater lake (Hawley Lake) (Figure 69) appeared to create the lethal conditions in the river (reduced river flows in the region, discussed earlier, might also have contributed to the warming), the trigger for the unusual warming itself was ultimately attributed (Gunn and Snucins 2010) to the shortening sea ice season in Hudson Bay (Section 2.1, *Abiotic Drivers*). It was suggested that

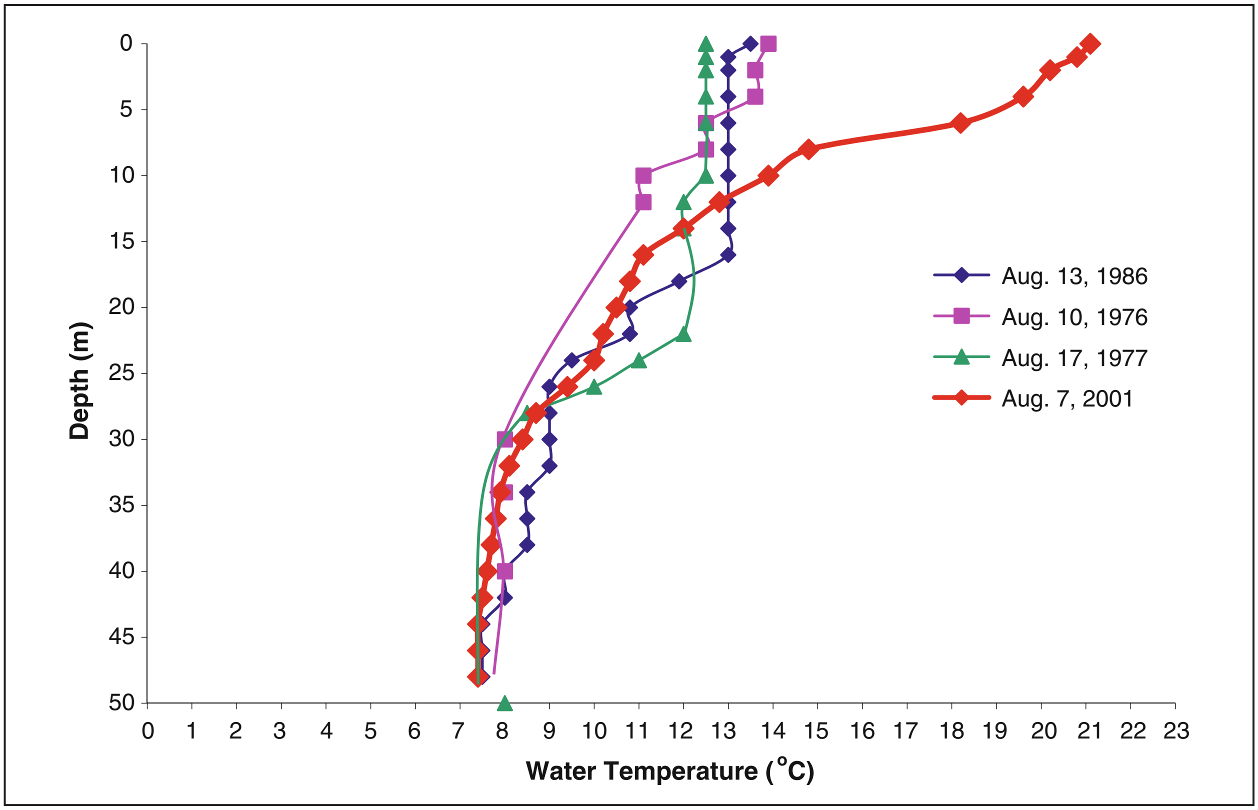

Figure 69. Temperature-depth profiles for Hawley Lake in the Hudson Plains Ecozone, 1976-2001. In 2001 the lake showed strong thermal stratification for one of the first times on record, with water temperatures exceeding 20 ˚C in the surface (discharge) layer.
Source: Gunn and Snucins (2010). Reprinted from Hydrobiologia, Brook charr mortalities during extreme temperature events in Sutton River, Hudson Bay Lowlands, Canada, Vol 650, 2010, p 82, J. Gunn and E. Snucins, Figure 2, with permission from Springer Science+Business Media.

the observed die-off of brook trout might be among the first of an increasing number of die-offs of vulnerable anadromous stocks that will occur as climate change proceeds (Gunn and Snucins 2010). Such stocks are dependent on seasonal sea ice cover in Hudson Bay to moderate the continental climate (brook trout return from the cold bay to spawn and overwinter in fresh water). Sutton River is considered by anglers to be one of the best anadromous brook trout rivers in the world.

The affected Hawley Lake (one of the deepest lakes in the ecozone) is one of the four geologically anomalous lake trout lakes in the Sutton Ridges area that is also highly prized by anglers. However, lake trout in that lake were not affected by the unusual warming in 2001, because ample coldwater habitat remained below the epilimnion and dissolved oxygen levels were high throughout the water column (Snucins 2003).

Elsewhere in the ecozone, Aboriginal knowledge indicates a die-off of whitefish and sucker also occurred in the Albany River along the James Bay coast during a heat wave and period of reduced precipitation in 2005 (Hori 2010).

At present, there is little evidence for overall trends (expansions or contractions) of lake and pond surface area in the ecozone, albeit there is also little monitoring of this nature, and it can be difficult to quantify the extent of water features in this ecozone (see Section 2.2.1, *Overview* *of Ecozone Structure & Land Cover Change*). In the area of Roberge Lake (Wapusk National Park), aerial photographs do, however, suggest a trend over the past ~58 years for lakes to develop enlarged fen-like margins, presumably in association with permafrost thaw (Dyke and Sladen 2010). Casual observations suggest that permafrost thaw might also be causing some recent slumping and collapse of river banks along the Hayes and Nelson rivers near York Factory, which adds to the sediment load in these rivers that drain into Hudson Bay (Section 2.1, *Abiotic Drivers*). Data are likewise insufficient for examining long-term trends in freshwater (river and lake) ice in the ecozone, but some changes in freshwater ice are suspected (see discussion in Section 2.1, *Abiotic Drivers*).

Future water export

The growth of human populations, industries, and climate warming will act in concert to increase the demand for water, both in Canada and elsewhere. Future demands that Canada share its abundant waters with water-poor regions are likely (Schindler 2001; Schindler and Lee 2010). Proponents of water export would have water exported from Canada to water-poor regions by tankers, pipelines, and rerouting of rivers. Some proposals are enormous in scale. The *Great Recycling and Northern Development* (GRAND) canal proposal popularized in the 1980s, for example, would dam James Bay, making it into a freshwater reservoir to store water entering it from the 20 or so rivers that surround it. A massive series of canals, locks, power plants, and dams would then divert the water to Georgian Bay, where it would be flushed through the Great Lakes to feed pipelines into water-sparse regions like the southwestern and southeastern United States (Bocking 1987; Barlow 2001; Schindler 2001). Related impacts on the health of aquatic ecosystems within the ecozone would likely be catastrophic.

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2.3 **Ecosystem composition**

2.3.1 **Overview of species diversity**

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With inset by
Sheldon Kowalchuk, Parks Canada – Wapusk National Park
R. Dean Phoenix, Ontario Ministry of Natural Resources
Zaid Jumean, Ontario Ministry of Natural Resources
Heather M. Stewart, Parks Canada – Wapusk National Park

The wide variety of habitats associated with the Hudson Plains Ecozone (coastal, tundra, forest, wetlands, lakes, and rivers; Section 2.2, *Ecosystem Structure*) support a diverse flora and fauna, which includes species of arctic, subarctic, and temperate (including *boreal*[41]) affinities (Riley 2003; Crins et al. 2009). The diversity of habitats and species in the ecozone is enhanced by the presence of sea ice, salt water, and tides associated with Hudson and James bays; rapid isostatic rebound; and the marked climatic gradient that occurs across the ecozone. For example, the most southerly presence of polar bear (*Ursus maritimus*) is linked to the presence of seasonal sea ice, which also occurs further south there than at comparable latitudes elsewhere. At the same time, however, the relatively harsh climate in this ecozone also restricts the total number of species that can occur (Currie 1991). Reptile and amphibian species are particularly poorly represented (e.g., McKenney et al. 1998; Oldham and Weller 2000). Even the diversity of mammal and tree species, which are largely dependent on annual temperature and evapotranspiration, respectively (Currie and Paquin 1987; Kerr and Packer 1998), are generally lower in the Hudson Plains Ecozone than at more southerly latitudes, and even some similar latitudes, within Canada and North America (Currie 1991; McKenney et al. 2007; Natural Resources Canada 2010). Similarly, although the coastal areas and extensive inland wetlands of the Hudson Plains Ecozone provide important summer habitat for many bird species, fewer bird species tend to

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[41] Floristic phytogeography often deals with the arctic-subarctic-temperate spread and generally does not include a *boreal* affinity descriptor, which can be vague in this context. In such cases, the closest synonymy to *boreal* would be high subarctic and low subarctic, but species of *boreal* affinity would straddle subarctic (taiga) and temperate (forested boreal) classes, depending on their distribution (J. Riley, The Nature Conservancy of Canada, pers. comm.).

occur there than in more southerly ecozones in Canada (e.g., Cadman et al. 2007) or at more southerly latitudes more generally (Currie 1991). Portions of the Hudson Plains Ecozone do, however, support among the highest diversity of freshwater fish species in Canada, even if these fish species are mostly common overall (Chu et al. 2003; Abell et al. 2008).

Species in the Hudson Plains Ecozone have not had much time to evolve *in situ* due to the relatively recent emergence of this land and the rapid migration that occurred into the area (Abraham and Keddy 2005). Consequently, endemic species are rare (Ricketts et al. 1999; Riley 2003; Abell et al. 2008), with one endemic species of vascular plant, *Linus lewisii* var. *lepagei*, occurring in coastal areas (Riley 2003; see also the Lepage wild flax profile in Section 2.3.3.7, *Vascular Plants*). However, many species populations are geographically disjunct from the principal range (Riley 2003; Cadman et al. 2007).

Comparatively few introduced[42] and potentially invasive species are also reported in this relatively remote and undisturbed ecozone of Canada (e.g., CFIA 2008). A number of species native and non-native (alien) to Canada have been introduced into the ecozone from outside of their normal range, but their impacts on the ecology of the ecozone are not well studied or monitored, i.e., their degree of invasiveness there is unknown. Most of the known introduced species are vascular plants (at least 98 species) (Riley 2003; see also Section 2.3.3.7, *Vascular Plants*), which are likewise the species group that contribute the highest proportion of introduced species at the national level (Environment Canada 2009). In the Hudson Plains Ecozone, most, if not all, introduced plants are, however, confined to the few villages and other areas with most human activity (Riley 2003). Introduced mammals include house mouse (*Mus musculus*; Wrigley 1974), and introduced birds include rock pigeon (*Columbia livia*), European starling (*Sturnus vulgaris*), and house sparrow (*Passer domesticus*) (Cadman et al. 2007; Rockwell et al. 2009; Appendix 1) – all also found in small numbers near villages. As previously noted (Section 2.2.2.4.2, *Rivers/Streams & Lakes*), a few introduced fish species, including common carp (*Cyprinus carpio*), rainbow smelt (*Osmerus mordax*), and smallmouth bass (*Micropterus dolomieu*), are also present, facilitated in part by the ecozone's hydrological connectivity to areas further south (in this area of Canada, rivers and wetlands drain north). The deepwater shipping port at Churchill, which is only one of three deepwater ports in the marine arctic, represents another route for the introduction of potentially invasive species (Lytwyn 2002), as does the smaller amount of marine traffic at the non-deepwater port of Moosonee. Other transportation routes into the ecozone are currently very limited, being comprised of air, as well as two railway lines (Manitoba and Ontario) and one all-season road (Québec) that connect the ecozone to land-based transportation systems in the south (Hydro-Québec 2003; Abraham and Keddy 2005; Stewart and Lockhart 2005; OMEI and OMNDMF 2009).

Broad changes in the complement of the ecozone's flora and fauna are evident along its climatic gradient from north to south and coast to inland, among the associated ecoregions. Along the described gradient, the species mix shifts from one with more representation of species with arctic affinity in the Coastal Hudson Bay Lowland Ecoregion (215) to one with decreasing representation of species with arctic affinity and more with temperate (including *boreal*) affinities (Wrigley 1974; Riley 2003; Crins et al. 2009). Indeed, many species of arctic affinity, such as polar bear, arctic fox (*Vulpes lagopus*), Richardson's collared lemming (*Dicrostonyx richardsoni*),

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[42] Introduced species are those species native or non-native (alien) to Canada that have been introduced into the ecozone from outside of their normal range.

Inset 10. Improving knowledge of species diversity

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Knowledge about species diversity in the Hudson Plains Ecozone is rapidly improving as, for example, through work in Wapusk National Park and Ontario.

Wapusk National Park is a relatively new (1996) and large park (11,475 km^2) located in the northwest extent of the Hudson Plains Ecozone. The transitional nature of this area, which is located at the southern extent of the continuous permafrost zone, provides habitat diversity for a variety of species due to the contrast between forest and subarctic tundra, as well as the terrestrial and marine interface. Based on four decades of work by members of the Hudson Bay Project, it is known that what is now Wapusk National Park is home to 198 bird species (Rockwell et al. 2009). Over the period 2002-2008 Parks Canada also engaged botanists from the University of Manitoba to conduct an inventory of both vascular and nonvascular plants and lichens there. The results indicate a total of 315 species of vascular plants, 270 species of lichens, and 105 species of bryophytes are found within the area (Punter et al. 2003; Piercey-Normore et al. 2004, 2006; Ford et al. 2005, 2007, 2008, 2009). As well, inventory work undertaken by the Manitoba Museum from 1998 to 2000 confirmed that Wapusk National Park is home to 40 species of terrestrial and marine mammals (Dubois 2001). An increase, since 2003, in the number of confirmed sightings of barren-ground grizzly bear (*Ursus arctos richardsoni*; Figure 70) in the transitional habitat of the park (Rockwell et al. 2008) is intriguing, because this species is a few hundred kilometres south of what is considered its normal range (COSEWIC 2002). Parks Canada will be working towards further improving the inventory of species in the northern portion of the Hudson Plains Ecozone over time.

Figure 70. Barren-ground grizzly in Wapusk National Park, Manitoba. Photo credit: L.J. Gormezano, American Museum of Natural History.

The Far North of Ontario includes its portion of the Hudson Plains Ecozone and the northern part of its Boreal Shield Ecozone. This region is the subject of the Government of Ontario's relatively new Far North initiative, which is intended to direct land use planning, protection, and resource development there (for more information on the Far North initiative, see Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*). The Far North initiative includes an Information and Knowledge Management Plan (IKMP) that is designed to inform land use planning activities. Among the information collection activities of the IKMP are the Terrestrial Biodiversity Study and the complementary Natural Heritage Program, which are both intended to provide a comprehensive suite of biodiversity information representative of Ontario's Far North community planning areas. The Terrestrial Biodiversity Study has adopted and modified the Multi Species Inventorying and Monitoring (MSIM) approach developed by the United States Forest Service (Manley 2006) to inventory a selection of terrestrial biodiversity components. MSIM is a modular approach to surveying multiple taxa on an unbiased sampling frame, which is intended to inventory species presence and monitor changes in occupancy (and abundance for common species) over time. Individual study areas are selected to encompass community homelands or planning areas. Within these study areas, a random selection of the National Forest Inventory 20 x 20 km grid points, stratified by ecodistrict, are sampled with variety of survey protocols. These protocols are intended to detect anurans, salamanders, snakes, birds, mammals (shrews, bats, rodents, medium-sized carnivores), insects and other invertebrates, and plants. The Terrestrial Biodiversity Study is planned for the period 2009-2011/13.

The Natural Heritage Program (NHP) aims to capture biodiversity data targeting sites previously identified as having significant natural heritage values in the Far North of Ontario including parts of the Hudson Plains Ecozone. Sites of interest in the ecozone include: candidate Areas of Natural and Scientific Interest, Important Bird Areas and Ramsar Sites, as well as Provincial Parks and Conservation Reserves. In addition, the Sutton Ridges, one of the few areas in the Ontario portion of the ecozone where granitic Precambrian bedrock outcrops occur, were surveyed. The NHP uses the plot-based Ontario Parks Inventorying and Monitoring Protocol (McCaul et al. 2010) to systematically and quantitatively assess biodiversity within selected plots. Additionally, targeted incidental surveys are employed to capture rare species and habitat specialists. Biodiversity components surveyed through either plot-based or targeted incidental work include vascular and non-vascular plants, birds, insects, mammals, amphibians, reptiles, and terrestrial molluscs.

Through these surveys, provincially or federally listed species and provincially rare species and landforms have been recorded. Moreover, new species to the Ontario portion of the Hudson Plains Ecozone and to the province have been discovered (Figure 71). To date, the NHP has identified over 20 new species for the Ontario part of the ecozone, including six vascular plant species and two lichen species that are new records to Ontario. The six newly discovered vascular species for Ontario are: *Botrychium crenulatum* (crenulate moonwort), *Packera streptanthifolia* (cleftleaf ragwort), *Diapensia lapponica* (Lapland diapensia), *Kalmia (=Loiseleuria) procumbens* (alpine azalea)[1], *Saxifraga rivularis* (alpine brook saxifrage), and *Campanula uniflora* (arctic bellflower). The newly discovered lichen species are: *Ophioparma lapponica and Rinodina conradii.* Ongoing specimen identification and future field work will likely yield even more new records.

Figure 71. Some newly recorded species in the Ontario portion of the Hudson Plains Ecozone: a) Kalmia (=Loiseleuria) procumbens *(alpine azalea); b)* Campanula uniflora *(arctic bellflower); c)* Diapensia lapponica (*Lapland diapensia*)*; and d)* Ophioparma lapponica*.*
Photo credits: S. Brinker, Ontario Ministry of Natural Resources.

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[1] Authority is (L.) Gift, Kron, and Stevens for this species with a recent genus change.

a variety of birds, and more than half of the ecozone's arctic vascular flora, occur only in the Coastal Hudson Bay Lowland Ecoregion.

Overall, however, species diversity in the Hudson Plains Ecozone remains poorly inventoried. As such, it is not possible at this time to gauge whether or not overall species diversity in this ecozone might be changing. While knowledge about the ecozone's species diversity continues to improve (Inset 10), it will still be some time before species diversity, and the associated distribution and abundance of species, is well-described for the ecozone as a whole. Meantime, accelerated climate change remains probably the greatest threat to the ecozone's current complement of species (e.g., Kerr and Packer 1998; Chu et al. 2005) although, as evident throughout the rest of Section 2.3 (*Ecosystem Composition*), loss and alteration (including fragmentation) of aquatic and terrestrial habitats by current and proposed resource developments and associated infrastructure are also of concern. The migratory birds that use the ecozone on a seasonal basis are additionally threatened by human activities in more developed areas further south (COSEWIC 2010; Ontario Partners in Flight 2010).

More information about the diversity of individual taxa (mammals, birds, fish, herpetofauna, invertebrates, vascular plants, lichens) in the ecozone can be found under their respective headings in Section 2.3.3, *Trends in Species of Special Interest.*

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2.3.2 Trends in species of national conservation concern

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Like other northerly ecozones in Canada, the Hudson Plains Ecozone has relatively few species of national conservation concern compared to more southerly and populated ecozones (Natural Resources Canada 2010). Although some arctic and other species reach their range limits in this ecozone and might be uncommon there, they are not necessarily rare overall. Species present in the Hudson Plains Ecozone that are considered to be of conservation concern nationally under the Species at Risk Act (SARA) or recommended for listing under the act by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) are shown in Table 16. Some sensitive species that occur in the ecozone, such as the migratory forest-tundra ecotype of woodland caribou, have not had their status reviewed by COSEWIC.

Most of the ecozone's species that are of national conservation concern are migratory bird species, which can be strongly influenced by conditions outside the ecozone, in wintering areas and along migration routes (Table 16). All seven species added to the list of species of national conservation concern, since 2005, are also migratory birds (red knot, olive-sided flycatcher, rusty blackbird, Canada warbler, common nighthawk, bobolink, and band-tailed pigeon). All four Endangered bird species on the list, i.e., red knot, ivory gull, Eskimo curlew, and piping plover, are likewise migratory (Table 16). James Bay is a key migration area for red knot (COSEWIC 2007a; Niles et al. 2010), which is commonly found in the ecozone (Appendix 1). Although ivory gulls are still associated with the ecozone (e.g., MARC 2003; Jehl 2004; Appendix 1), Eskimo curlew has not been observed in the ecozone for a long time. The Committee on the Status of Species at Risk in Ontario (COSSARO) recently elevated the status of this species in that province from Endangered to Extirpated (COSSARO 2010; Government of Ontario 2010) and, in fact, Eskimo curlew might now be extinct (Environment Canada 2007; COSEWIC 2009). This species historically occurred in the Hudson Plains Ecozone during migration, particularly along its coasts. Piping plover, as well as several other bird species in Table 16 (Sprague's pipit, Barrow's goldeneye, and band-tailed pigeon), occur in the ecozone either as accidental or vagrant species (Appendix 1).

A mammal species of national conservation concern that has occasionally been observed in the Hudson Plains Ecozone (Rockwell et al. 2008) but not currently recognized by COSEWIC as occupying the ecozone as part of its regular range is the grizzly bear or, more specifically, the Northwestern (barren-ground) population of this species (COSEWIC 2002). The number of confirmed sightings of individuals from this population (COSEWIC Special Concern, SARA Schedule 3) has been increasing along the coast of Wapusk National Park in the Manitoba portion of the ecozone, since 2003 (Rockwell et al. 2008). These increased sightings, a few hundred kilometres south of the species' regular range, could be the result of increased observational effort or an expansion of the species' range into an area previously dominated by polar bears. The latter hypothesis is favoured, because research has been occurring in Wapusk National Park, since 1965 and at a consistent level, since 1993.

Table 16 also shows how the status listings for species repeatedly assessed by COSEWIC have changed over time. With the exception of Peregrine falcon and harlequin duck, species that have been repeatedly assessed have either remained stable in their classification or deteriorated, since the 1980s. Species that have deteriorated in status include polar bear, ivory gull (discussed above), Ross's gull, and lake sturgeon. Nationally, trends in species' status listings are considered difficult to interpret, because definitions and assessment criteria have changed, since SARA was introduced (Environment Canada 2009)[43]. Still, COSEWIC assessment reports suggest biologically based deterioration directly in the ecozone of at least the two polar bear subpopulations and some designatable units of lake sturgeon (COSEWIC 2006, 2008). Although Ross's gull appears to be in decline in Canada, the small number of individuals at any one site precludes analysis of population trends for this species. The Hudson Plains Ecozone, and more specifically Churchill, is only one of four known nesting locations for Ross's gull in Canada; the Churchill population has ranged from one to five pairs, since 1980 (COSEWIC 2007b). The number of individuals peaked in 1982, while more recent sightings are for one or two birds per year, and nesting success seems low (MARC 2003).

More information on the population status and trends of woodland caribou, wolverine, polar bear, lake sturgeon, and the major bird groupings (landbirds, waterfowl, shorebirds, and waterbirds) in the ecozone can be found below in Section 2.3.3, *Trends in Species of Special Interest*.

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[43] Globally, the conservation status and extinction risk of species on the International Union for Conservation of Nature (IUCN) Red List continues to deteriorate overall, with least effects evident in northern areas, including the Hudson Plains Ecozone (Hoffmann et al. 2010).

Table 16. Status of species associated with the Hudson Plains Ecozone that are recognized to be of conservation concern nationally under the Species at Risk Act (SARA) and/or by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). The national-level species conservation status listings in this table can differ from those assigned by individual provinces and territories.

Source: Government of Canada (2010), Species at Risk Public Registry (http://www.sararegistry.gc.ca) and COSEWIC (2010) species database (http://www.cosewic.gc.ca/eng/sct1/searchform_e.cfm).

Common name	Scientific name	Designatable unit (below species level, if applicable)	SARA listing[a]	COSEWIC listing	Major threats recognized by the COSEWIC listing	Listing trend between COSEWIC assessments
Mammals, marine interface[b]						
Polar bear[c]	*Ursus maritimus*	–	Special Concern, Schedule 3 (not listed 2010[c])	Special Concern	Climate change and over-harvest for some subpopulations.	Deteriorated from Not at Risk in 1986 to Special Concern in 1991; no change, 1991-2008.
Mammals, terrestrial						
Wolverine[d]	*Gulo gulo*	Western population	Special Concern, Schedule 3 (not listed)	Special Concern	Habitat and population fragmentation by industrial activity and increased motorized access that increases harvest pressure and other disturbances, particularly in the southern part of the species' distribution. The species has a low reproductive rate, and it requires vast secure areas to maintain viable populations.	No change, 1989-2003.
		Eastern population	Endangered, Schedule 1	Endangered	Population might be extirpated; only a few unconfirmed (not verified) reports in >25 years. Any remaining population would be extremely small and at high risk of extinction.	No change, 1989-2003.
Woodland caribou	*Rangifer tarandus caribou*	Boreal (forest-dwelling) population	Threatened, Schedule 1	Threatened	Habitat loss and increased predation, the latter possibly facilitated by human activities.	No change, 2000-2002.

Table 16, Cont.

Common name	Scientific name	Designatable unit (below species level, if applicable)	SARA listing[a]	COSEWIC listing	Major threats recognized by the COSEWIC listing	Listing trend between COSEWIC assessments
Grizzly bear[e]	*Ursus arctos*	Northwestern (barren-ground) population	Special Concern, Schedule 3 (not listed)	Special Concern	Habitat loss from expanding industrial, residential, and recreational developments.	No change, 1991-2002.
Birds						
Sprague's pipit	*Anthus spragueii*	–	Threatened, Schedule 1	Threatened	Ongoing habitat loss, degradation, and fragmentation on both breeding and wintering grounds (species requires large tracts of intact native grassland for breeding).	No change, 1999-2010.
Short-eared owl[f]	*Asio flammeus*	–	Special Concern, Schedule 3 (not listed)	Special Concern	Habitat loss and degradation on wintering grounds; secondary threats include continuing habitat loss and degradation on breeding grounds in southern Canada and pesticide use.	No change, 1994-2008.
Barrow's goldeneye	*Bucephala islandica*	Eastern	Special Concern, Schedule 1	Special Concern	Threatened by factors such as limited habitat availability and oil spill potential, but none currently at a scale that would negatively impact the population; population size is limited.	Single assessment only, 2002.
Red knot	*Calidris canutus rufa*	*rufa* subspecies	–	Endangered	Depletion of horseshoe crab eggs, a critical food source used during northern migration. The subspecies declined 70% in abundance over the past three generations (15 years), and no potential exists for rescue from other populations.	Single assessment only, 2007.

Table 16, Cont.

Common name	Scientific name	Designatable unit (below species level, if applicable)	SARA listing[a]	COSEWIC listing	Major threats recognized by the COSEWIC listing	Listing trend between COSEWIC assessments
Piping plover[g]	*Charadrius melodus circumcinctus*	*circumcinctus*	Endangered, Schedule 1	Endangered	Loss of quality nesting habitat in many places (e.g., increased human use of beaches and consequent disturbance of nesting sites). Low reproductive success, particularly in years of drought; nests also regularly lost to flooding.	Single assessment only, 2001.
Common nighthawk	*Chordeiles minor*	–	–	Threatened	Reduction of insect food sources and habitat availability in some regions, the latter caused by fire suppression, intensive agriculture, and declines in number of gravel rooftops.	Single assessment only, 2007.
Olive-sided flycatcher	*Contopus cooperi*	–	–	Threatened	Uncertain, but widespread and consistent population decline over the last 30 years, with no evidence that the decline has ceased. The Canadian population declined by ~79% from 1968 to 2006 and 29% from 1996-2006.	Single assessment only, 2007.
Yellow rail[h]	*Coturnicops noveboracensis*	–	Special Concern, Schedule 1	Special Concern	Ongoing threats to breeding and wintering wetland habitats.	No change, 1999-2009.
Bobolink	*Dolichonyx oryzivorus*	–	–	Threatened	Incidental mortality from agricultural operations, habitat loss and fragmentation, pesticide exposure, and bird control at wintering roosts.	Single assessment only, 2010.
Rusty blackbird	*Euphagus carolinus*	–	Special Concern, Schedule 1	Special Concern	Threats occur primarily on the winter range and include habitat conversion and blackbird control programs in the United States.	Single assessment only, 2006.

Table 16, Cont.

Common name	Scientific name	Designatable unit (below species level, if applicable)	SARA listing[a]	COSEWIC listing	Major threats recognized by the COSEWIC listing	Listing trend between COSEWIC assessments
Peregrine falcon[i]	*Falco peregrinus anatum*	*anatum*	Threatened, Schedule 1	Special Concern	Although the subspecies *anatum/ tundrius* (now combined) has increased, since the 1970s, up to near-historical numbers via re-introductions and a ban of organochlorine pesticides in Canada, continued use of pesticides on wintering grounds and known effects of new pesticides licensed for use in Canada are current concerns.	Improved from Endangered (*anatum*) or Threatened (*pealei*) in 1978 to Threatened (*anatum*) in 1999-2000 or Special Concern in 1992 (*pealei*); these two subspecies were combined into a single unit (*anatum/ tundrius*) in 2007 and assessed as Special Concern.
	Falco peregrinus tundrius	*tundrius*	Special Concern, Schedule 3 (not listed)			
Harlequin duck	*Histrionicus histrionicus*	Eastern	Special Concern, Schedule 1	Special Concern	Catastrophic events, such as oil spills, due to the tendency of this relatively small population to congregate in relatively large groups when moulting and wintering.	Improved from Endangered in 1990 to Special Concern in 2001.
Eskimo curlew	*Numenius borealis*	–	Endangered, Schedule 1	Endangered	Historically, uncontrolled market hunting and dramatic losses in amount and quality of spring stopover habitat (native grasslands); presently, very low population size, no known chance of rescue from outside populations, and historic and ongoing conversion of native grasslands on spring staging areas in Canada and the United States and on wintering grounds in Argentina.	No change, 1978-2009.

Table 16, Cont.

Common name	Scientific name	Designatable unit (below species level, if applicable)	SARA listing[a]	COSEWIC listing	Major threats recognized by the COSEWIC listing	Listing trend between COSEWIC assessments
Ivory gull	*Pagophila eburnea*	–	Endangered, Schedule 1	Endangered	Contaminants in food chain, continued hunting in Greenland, possible disturbance by mineral exploration at some breeding locations, and degradation of ice-related foraging habitats, as a result of climate change. The Canadian breeding population of this long-lived seabird has declined 80% over the last 20 years.	Deteriorated from Special Concern 1979-2001 to Endangered in 2006.
Band-tailed pigeon	*Patagioenas fasciata*	–	Special Concern, Schedule 1	Special Concern	Historical over-hunting. Forestry might negatively affect habitat in the long term, by creating dense second-growth forests with few berry-producing shrubs. Species is also susceptible to disturbance at isolated mineral sources needed for its nutrition.	Single assessment only, 2008.
Ross's gull[j]	*Rhodostethia rosea*	–	Threatened, Schedule 1	Threatened	Human disturbance in some breeding areas (including the Churchill area) and changes in ice and snow patterns associated with climate change.	Deteriorated from Special Concern in 1981-1996 to Threatened in 2001; no change, 2001-2007.
Canada warbler	*Wilsonia canadensis*	–	–	Threatened	Uncertain, but loss of primary forest on wintering grounds in South America is recognized as a potential cause.	Single assessment only, 2008.

Table 16, Cont.

Common name	Scientific name	Designatable unit (below species level, if applicable)	SARA listing[a]	COSEWIC listing	Major threats recognized by the COSEWIC listing	Listing trend between COSEWIC assessments
Fish						
Lake sturgeon[k]	*Acipenser fulvescens*	Western Hudson Bay populations (DU1)	–	Endangered	Historical overexploitation (commercial fishing); dams are probably the most important recent threat.	No change, 2005-2006, but species as a whole (prior to subdivision) has deteriorated from Not at Risk in 1986.
		Nelson River populations (DU3)	–			
		Southern Hudson Bay – James Bay populations (DU7)	–	Special Concern	Exploitation and declines in habitat from multiple dams.	No change, 2005-2006, but species as whole (prior to subdivision) has deteriorated from Not at Risk in 1986.
Reptiles & amphibians						
No species currently listed						
Invertebrates						
Monarch[l]	*Danaus plexippus*	–	Special Concern, Schedule 1	Special Concern	Species is highly restricted and vulnerable in its wintering range.	No change, 1997-2001.
Vascular plants, non-vascular plants, & lichens						
No species currently listed						

[a] Schedule 1 of the Species at Risk Act (SARA, Bill C-5) is the official list of wildlife species at risk. It includes Wildlife Species at Risk in the four risk categories (Extirpated, Endangered, Threatened, Special Concern) that have been re-assessed under the most current criteria by COSEWIC, since the bill's protections commenced upon proclamation of the law in October 1999. Schedule 2 contains additional species, in the three highest risk categories, which have yet to be re-assessed by COSEWIC using revised criteria. Schedule 3 contains those species of Special Concern, i.e., the lowest category of species at risk, which have not yet been re-assessed by COSEWIC. Under clause 27(1.1) of SARA, Cabinet can review an assessment received from COSEWIC over nine months and add the species, decide not to add it, or refer the matter back to COSEWIC for further consideration. As such, recent assessments by COSEWIC might not yet be reflected in the schedules of the act. All SARA information in this table is from the SARA public registry: http://www.sararegistry.gc.ca/default_e.cfm.

[b] Additional species of national conservation concern that occur in the adjoining marine ecosystem are listed in the companion Arctic Marine Ecozones report of the ESTR (Niemi et al. 2010).

[c] Movement and genetic data support a single designatable unit of polar bears in Canada, but trends differ by subpopulation. Notably, the Western Hudson Bay subpopulation is one of four subpopulations (out of a total of 13 found in Canada) that is considered at high risk of declining by 30% or more over the next three generations of bears (36 years). Declines in the Western Hudson Bay subpopulation are partly attributable to climate change. Although this report considers status and trends up to December 2010 only, the status of polar bear changed prior to the release of this document. Specifically, polar bear was listed as a species of Special Concern under the federal Species at Risk Act (Schedule 1) in November 2011 (see Canada Gazette Part II 145 (23): 2232-2384).

[d] Wolverine was originally considered a single unit and designated Special Concern in 1982. Two populations were defined in 1989, with the Western population (which is currently found in the Hudson Plains Ecozone) maintaining Special Concern status and the Eastern population assigned Endangered status.

[e] Grizzly bear was originally considered a single unit and designated Not at Risk in 1979. In 1991, the species was split into two populations. The Northwestern population was designated Special Concern at the time and again at its last assessment in 2002, while the Prairie population has remained in an Extirpated designation. The Hudson Plains Ecozone is not part of the grizzly bear's regular range, but the species has occasionally been observed in the northwestern portion of the ecozone, and the number of confirmed sightings there has been increasing, since 2003 (Rockwell et al. 2008).

[f] Short-eared owl has experienced continual population decline and nearly meets the criteria for Threatened status.

[g] Piping plover was considered a single unit and designated Threatened in 1978. Its status was re-examined in 1985 and designated Endangered. When the species was re-examined again in 2001, it was split into two subspecies, *circumcinctus* and *melodus*, which were both designated Endangered. Subspecies *circumcinctus* occurs in the Hudson Plains Ecozone as an accidental species.

[h] Yellow rail is close to meeting some criteria for Threatened status due to its relatively small population size, compressed wintering range, ongoing threats to breeding and wintering wetland habitats, and evidence for local declines in several parts of its breeding range.

[i] Of the two subspecies of peregrine falcon still recognized under SARA, the subspecies *anatum* occurs in the Hudson Plains Ecozone. COSEWIC combined the two subspecies in 2007.

[j] Ross's gull met criterion for Endangered, D1, but this species was designated Threatened, D1, because it has potential for rescue and more birds likely occur in unsurveyed areas.

[k] Lake sturgeon was considered a single unit and designated Not at Risk in 1986. The species was split into separate units in 2005, at which time units were designated a higher risk status. Further splits occurred in 2006 (e.g., Western population was split into five separate populations, including the Western Hudson Bay and Nelson River populations) but with no further changes in status for populations in the Hudson Plains Ecozone.

[l] Occurrences of monarch butterfly in the Hudson Plains Ecozone (near coastal villages along James Bay) are disjunct from the main population (Abraham and Keddy 2005).

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2.3.3 **Trends in species of special interest**

The principal intent of this section is to profile, for major life form groupings (mammals, birds, fish, herpetofauna, invertebrates, vascular plants, lichens) the status and trends of selected species, or assemblages of species, that are considered to be of *special interest* to the state of the Hudson Plains Ecozone. Under the ESTR framework, species (or species assemblages) can be of particular human relevance (special interest) to an ecozone due to their ecological, cultural, and/or economic importance, in addition to their biodiversity value. Such species are, therefore, generally: keystone species like top predators, umbrella species, species for which Canada has a significant amount of global responsibility, species of cultural significance, and/or species of particular economic importance. A theme that emerges when examining the species or life form groupings selected for profile in this section is that survey and monitoring data are largely insufficient for purposes of either current status assessment or trend analysis; with some exceptions, few population changes or trends are evident. Species diversity within some of the major life form groupings is also discussed in this section, building on the overview of species diversity in Section 2.3.1.

2.3.3.1 **Marine mammals**

The marine ecosystem that interfaces with the Hudson Plains Ecozone provides sea water and sea ice habitat for a number of marine mammals, including at least five species of whales, five species of seals, Atlantic walrus (*Obodenus rosmarus rosmarus*), and polar bear (*Ursus maritimus*) (Stewart and Lockhart 2005)[44]. Some such marine mammals also use the terrestrial environment of the Hudson Plains Ecozone (mainland or islands just off the coast) and/or its larger river estuaries, near-shore areas, and offshore leads and polynyas (OMNR 1985; Abraham and Keddy 2005; Stewart and Lockhart 2005). The status and trends of the whale, seal, and walrus species in this area are addressed by the complementary Arctic Marine Ecozones report of the ESTR (Niemi et al. 2010). The status and trends of the polar bear subpopulations that use sea ice in Hudson and James bays in winter but the terrestrial environment of the Hudson Plains Ecozone during the sea ice-free season are considered in the following section. Interactions between polar bear and seal species are also discussed later in this report, in Section 2.4.3.1, *Predator-Prey Relationships & Cycles*.

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[44] Arctic fox (*Vulpes lagopus*) also uses sea ice, but it is included with terrestrial mammals (furbearing mammals) in this report.

2.3.3.1.1 **Polar bear**

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In the Hudson Plains Ecozone, the polar bear (*Ursus maritimus)* is at the southern edge of its range, where the first effects of climate change on the species were predicted to occur (Stirling and Derocher 1993). Polar bears occupying southern areas of Hudson Bay are assigned to two subpopulations: Western Hudson Bay and Southern Hudson Bay (Aars et al. 2006; Figure 72). The harvest of polar bears in these two subpopulations is managed by the governments of Manitoba, Ontario, Québec, and/or Nunavut, Wildlife Management Boards, and Aboriginal communities. Though harvest is currently not the key factor affecting population trends, harvest is a recognized anthropogenic stressor on polar bear subpopulations, and it must be closely monitored in the future. Harvest will be particularly challenging to manage in the future, when these subpopulations are projected to decline in abundance in association with climate change (Peacock et al. 2010).

Bears of the Western Hudson Bay subpopulation summer on land in Manitoba (Derocher and Stirling 1990). Bears of the Southern Hudson Bay subpopulation summer on land in Ontario and on islands in James Bay that are under the jurisdiction of Nunavut (Kolenosky et al. 1992). Bears do not regularly summer on land along the Hudson Bay and James Bay shores of Québec, though they are found on offshore islands (Crête et al. 1991). When on the sea ice, polar bears cover vast areas and have home ranges exceeding 100,000 km^2 (Parks et al. 2006). Despite

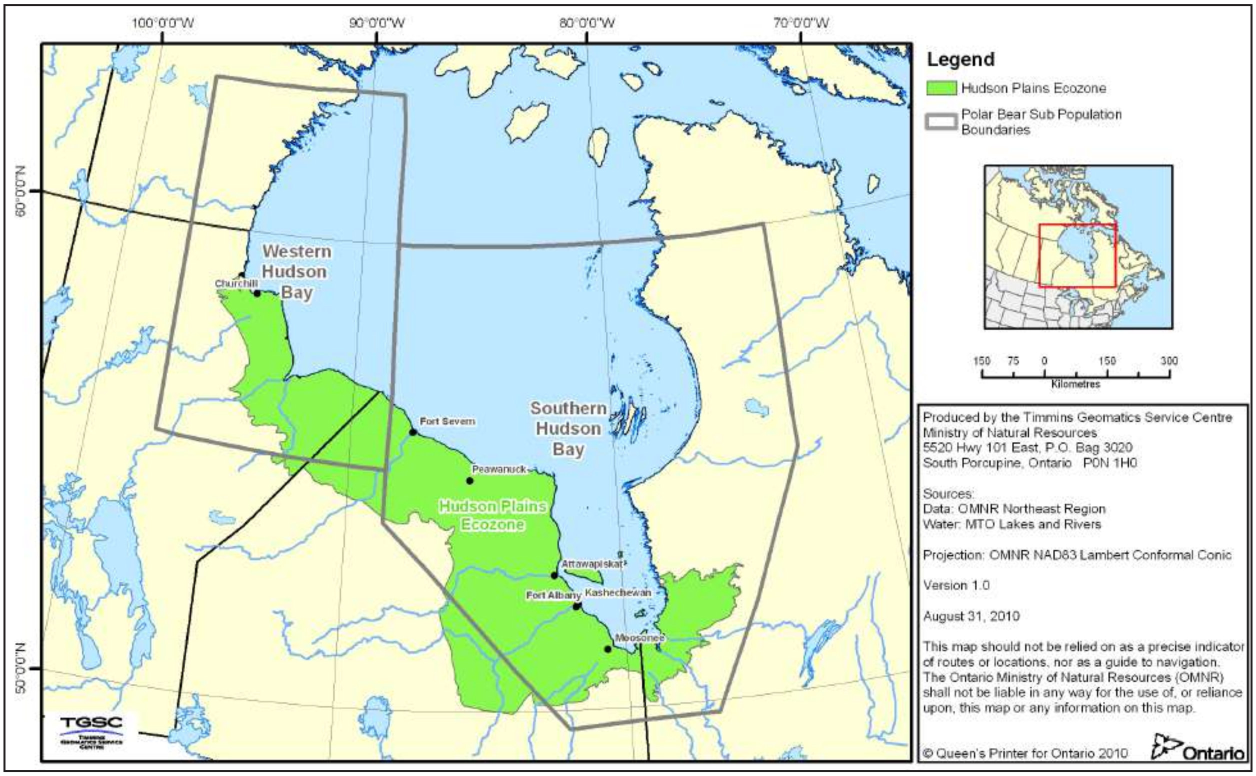

Figure 72. Map illustrating the management boundaries of Canada's Western Hudson Bay and Southern Hudson Bay subpopulations of polar bear relative to the Hudson Plains Ecozone.

such extensive movements on the ice, bears typically remain within the boundaries of the subpopulation (Stirling et al. 1999; Plante et al. 2001; Parks et al. 2006).

Perhaps more than any other animal, the polar bear is the iconic symbol of arctic and subarctic ecosystems; it is of major cultural significance to Aboriginal peoples in the region. As an apex predator in the marine system, it is a species that has already been affected negatively by climate change (Regehr et al. 2007). Declines in sea ice in Hudson Bay due to climate warming are projected to continue (Amstrup et al. 2008), and such trends will have negative impacts on polar bears (Molnár et al. 2010), so that continued monitoring of polar bear populations is critical. The polar bear was assessed as a species of Special Concern nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2008), but it is not currently listed under the Canadian Species at Risk Act (SARA; Government of Canada 2010); Environment Canada is currently conducting consultations on the proposed listing under SARA[45]. Under respective provincial legislation, Manitoba declared the Western Hudson Bay subpopulation to be Threatened in February 2008, and Ontario declared the Southern Hudson Bay subpopulation to be Threatened in September 2009. About 4,000 bears, or 20% of the total world population, occur in the entire Hudson Bay region, of which about 1,800 individuals are associated with the Hudson Plains Ecozone (Lunn et al. 2010).

Polar bears are adapted to the marine environment and spend much of the year on sea ice (Amstrup 2003), where they are closely linked to the distribution, density, and abundance of their primary prey, ringed seals (*Pusa hispida*) (Stirling and Øritsland 1995; Ferguson et al. 2000; Amstrup et al. 2001; Amstrup 2003). In Hudson Bay the sea ice melts completely each summer (Etkin 1991; Wang et al. 1994) forcing the bears ashore, so that the bears spend up to 5 months on shore waiting for ice to reform (Stirling et al. 2004). Pregnant bears spend up to 8 months ashore without feeding. A significant trend towards earlier break-up of the sea ice has been occurring; it now occurs as much as 3 weeks earlier than in the 1970s off the Manitoba coast (Stirling et al. 1999, 2004) and off the Ontario coast (Gough et al. 2004; Gagnon and Gough 2005) (see also Section 2.1, *Abiotic Drivers*).

Trends towards earlier break-up are associated with declines in body condition, survival, and abundance of the bears in Hudson Bay (Inset 11). For the Western Hudson Bay subpopulation, declines in body condition were first identified for the period 1981-1998 (Stirling et al. 1999). This trend continued and the subpopulation declined from about 1,194 in 1987 to 935 in 2004 (Regehr et al. 2007). Though survival rates of adult males and females were stable throughout the period, those of juvenile, subadult, and senescent bears were correlated with spring break-up date, decreasing by 2-5% for each week earlier than average that the sea ice broke-up (Figure 74). A causal relationship between earlier sea ice break-up due to climate warming, lost opportunities to hunt seals, and declines in polar bear survival was suggested (Stirling et al. 1999; Regehr et al. 2007).

Declines in body condition are also apparent for the Southern Hudson Bay subpopulation of polar bears (Obbard et al. 2006); these declines were most dramatic for pregnant female and subadult bears (Figure 75). Results of a 3 year capture-recapture study conducted from 2003 to 2005 (Obbard et al. 2007) indicated that the number of bears in the subpopulation had not declined, since the mid-1980s; however, there was evidence of declines in survival rates of all

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[45] Although this report considers status and trends up to December 2010 only, the status of polar bear changed prior to the release of this document. Specifically, polar bear was listed as a species of Special Concern under the federal Species at Risk Act (Schedule 1) in November 2011 (see Canada Gazette Part II 145 (23): 2232-2384).

Inset 11. Changes in polar bear subpopulations are correlated with changing sea ice conditions in Hudson & James bays

Lyle R. Walton, Ontario Ministry of Natural Resources

Climatic warming is considered to be the greatest long-term threat to polar bear (Figure 73). In both Hudson Bay and James Bay, trends for earlier break-up and later freeze-up of the sea ice are linked to climatic warming (Stirling et al. 1999; Gough et al. 2004; Stirling et al. 2004; Gagnon and Gough 2005; Obbard et al. 2006; Stirling and Parkinson 2006; see also Section 2.1, *Abiotic Drivers*). In consequence, the duration of the ice-free period is increasing, which reduces the time that polar bears have on the ice to hunt and feed on seals and, therefore, to put on fat stores for their seasonal period on land, where they eat only opportunistically.

For both the Western Hudson Bay (WHB) and the Southern Hudson Bay (SHB) polar bear subpopulations, a relationship exists between earlier break-up of sea ice and significant declines in their body condition (Stirling et al. 1999; Obbard et al. 2006; Stirling and Parkinson 2006; Regehr et al. 2007). Declines in body condition of bears in the WHB subpopulation likely contributed to observed declines in their reproductive success and survival and to a 22% reduction in their population numbers between 1987 and 2004 (Stirling et al. 1999; Regehr et al. 2007). Survival of juvenile, subadult, and senescent adult bears (≥ 20 years old) was also related to earlier break-up of sea ice, though survival of prime-aged adults remained stable. In the SHB subpopulation, evidence also exists for declines in survival rates of subadult and adult bears. However, the documented declines in body condition and survival have not so far led to a decline in the number of SHB bears (Obbard et al. 2007).

Figure 73. Male polar bear on the Southern Hudson Bay coast, Ontario.
Photo credit: L.R. Walton, Ontario Ministry of Natural Resources.

Changes in sea ice duration have been more dramatic in the western portions of Hudson and James bays, which might explain why the WHB subpopulation has already declined in abundance, yet there has been no detectable change in abundance for the SHB subpopulation. For example, by 2003 the break-up date of sea ice had advanced by about 10 days/decade in James Bay and western Hudson Bay but only by about 3 days/decade in Hudson Bay off the Ontario coast (Gough et al. 2004; Gagnon and Gough 2005). However, if current trends in sea ice distribution and duration continue, these trends might lead to a decline in abundance of the SHB subpopulation. This situation highlights the requirement for continued monitoring of both of these polar bear subpopulations.

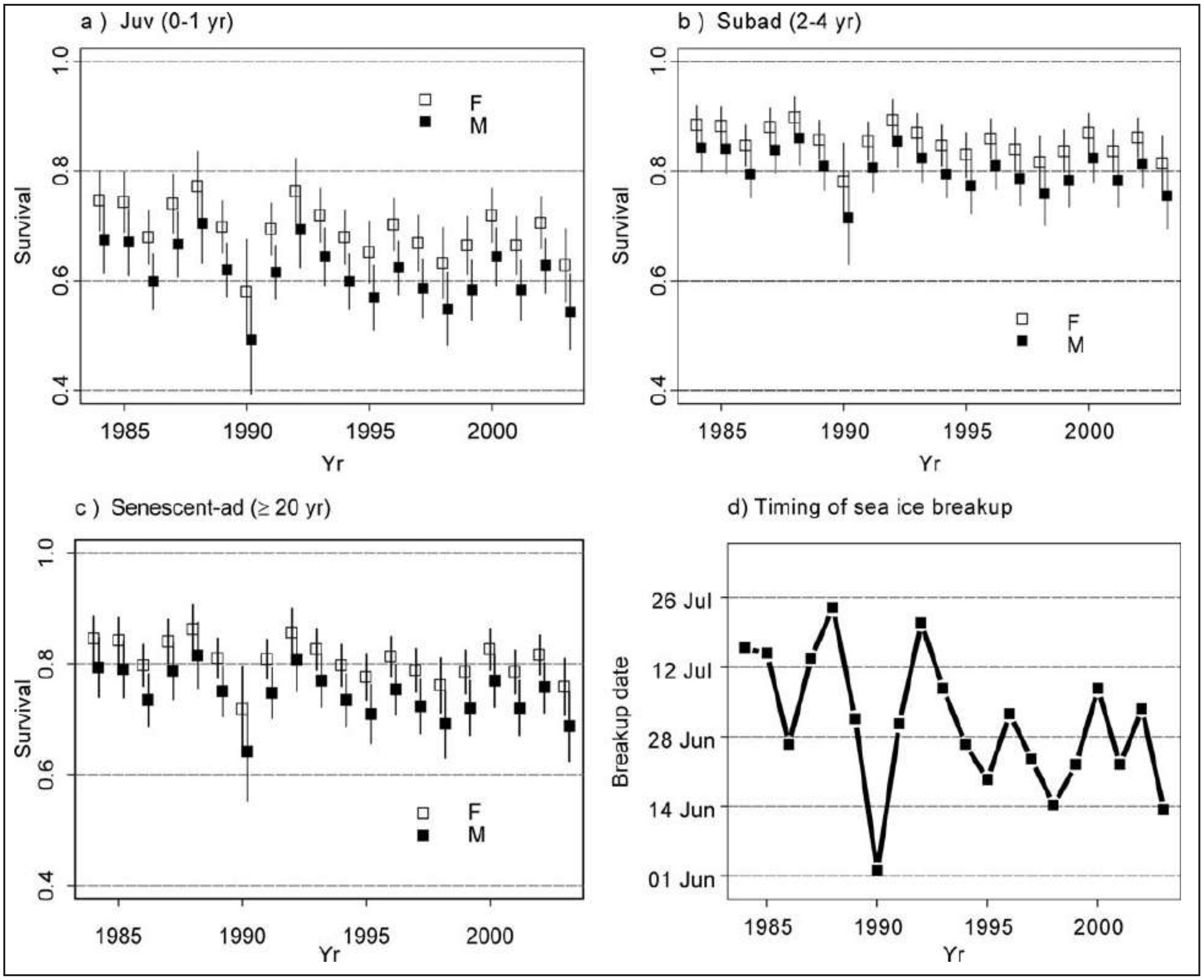

Figure 74. Total apparent survival and 95% confidence intervals for a) juvenile (Juv); b) subadult (Subad); and c) senescent-adult (Senescent-ad) polar bears in western Hudson Bay, 1984-2004, as estimated from the most supported model fit to capture-recapture data collected by the Canadian Wildlife Service and the Manitoba Department of Conservation (MDOC). Reduced survival rates for subadults and senescent-adults captured around Churchill, Manitoba by the MDOC are not shown. d) Timing of sea ice break-up in the Western Hudson Bay polar bear management area, which was the best predictor of survival for juvenile, subadult, and senescent-adult polar bears. Other abbreviations: F, female and M, male. Source: Regehr et al. (2007). Reprinted from The Journal of Wildlife Management, Vol 71, No. 8, E.V. Regehr, N.J. Lunn, S.C. Amstrup and I. Stirling, Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay, Copyright 2007, with permission from Allen Press Publishing Services.

age and sex classes. Obbard et al. (2007) suggested that the combination of significant declines in body condition and evidence of declines in survival rates indicate that the subpopulation might be at an ecological tipping point and likely to decline in abundance in the future. Climate warming and reductions in sea ice duration are expected to continue in the future in Hudson and James bays (Section 2.1.2.2, *Projected Changes*), so the negative trends in body condition, survival, and abundance identified for the Western Hudson Bay and Southern Hudson Bay subpopulations can be expected to continue (Stirling and Parkinson 2006).

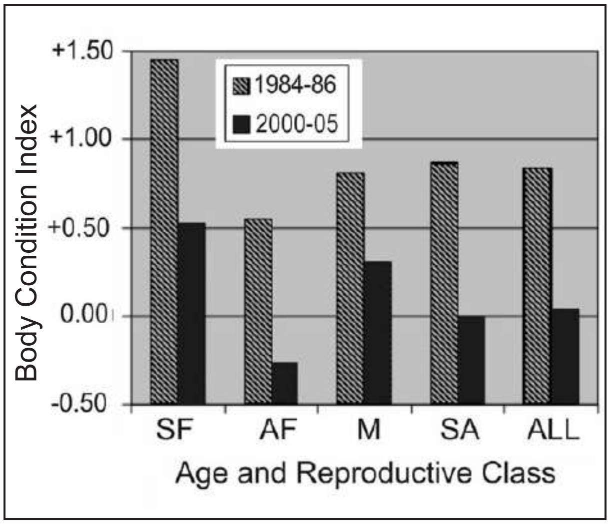

Figure 75. Mean Body Condition Index values (Cattet et al. 2002) for polar bears of the Southern Hudson Bay subpopulation, 1984-1986 and 2000-2005. Abbreviations: SF, solitary adult females; AF, adult females with young; M, adult males; SA, subadults; and ALL, all classes combined. A bear in average body condition has a Body Condition Index value of 0.0.
Source: Obbard et al. (2006). Reprinted from Ontario Ministry of Natural Resources Climate Change Research Information Note No. 3, M.E. Obbard, M.R.L. Cattet, T. Moody, L.R. Walton, D. Potter, J. Inglis and C. Chenier, under license with the Ontario Ministry of Natural Resources, copyright Queen's Printer for Ontario, 2006.

While on land during the ice-free season, polar bears require extensive undisturbed coastal areas for summer retreat habitat. In Manitoba, adult males tend to remain along the coast, whereas pregnant females and adult females accompanied by cubs and yearlings move inland (Stirling et al. 1977; Derocher and Stirling 1990; Clark and Stirling 1998). More recently, family groups are being found closer to the coast in Manitoba, perhaps due to changes in date of break-up (Towns et al. 2010). In Ontario, family groups are more common at the coast, though they generally avoid areas preferred by adult males, such as offshore spits, peninsulas, and small islands or areas with elevated sand banks (M. Obbard, Ontario Ministry of Natural Resources, unpublished data). Much of this coastal habitat is currently protected within Polar Bear Provincial Park in Ontario and within Wapusk National Park and the Kaskatamagan Wildlife Management Area in Manitoba. Tour operators near Churchill, Manitoba, offer viewing opportunities for polar bears, while the bears are on land waiting for the sea ice to reform. Many of the tours are offered from large tundra vehicles. Viewing of polar bears from tundra vehicles has been shown to increase vigilance behaviour in male bears, though the biological or physiological consequences of such changes in behaviour have not been quantified (Dyck and Baydack 2004, 2006), and only a small proportion of the subpopulation occurs in the regulated polar bear viewing area.

Additional critical terrestrial habitat for polar bear is that occupied by pregnant females for maternity denning. Pregnant females move inland in late summer or early fall to maternity denning areas, which might be over 100 km inland (Kolenosky and Prevett 1983; Lunn et al. 2004; Richardson et al. 2005). Female polar bears show a high degree of fidelity to the denning area but not to specific den sites (Ramsay and Stirling 1990; Scott and Stirling 2002). In Manitoba, pregnant females generally occupy deep earth dens dug into frozen peat banks in areas of discontinuous permafrost along the edges of lakes, streams, and hummocks (Clark et al. 1997; Richardson et al. 2005). In Ontario, pregnant females excavate shallow dens into elevated palsas, river banks, and gravel ridges, but they also den in the lee of clumps of spruces (Kolenosky and Prevett 1983; Obbard and Walton 2004). Implantation generally occurs in September or October, so that cubs are likely born in early December (Derocher et al. 1992). This is prior to major accumulations of snow in the area, so most pregnant females in the region likely give birth in shallow or deep earth dens and then expand these into overlaying snowdrifts later in the winter (Jonkel et al. 1972; Ramsay and Stirling 1990; Clark et al. 1997). Most maternity denning habitat in Manitoba is protected within Wapusk National Park and the Kaskatamagan Wildlife Management Area (Richardson et al. 2007). In Ontario, about 36% of maternity denning habitat is protected within Polar Bear Provincial Park (Obbard and Walton 2004).

Because earth dens are so frequently used, and pregnant females in this region have one of the longest fasting periods of any mammal (8 months; Ramsay and Stirling 1988), the availability and use of these features might be critical to reproduction and to the long-term viability of these subpopulations (Clark et al. 1997). The importance of these earth dens is likely to increase if projections of climatic models are correct, and the amount of available permafrost in the Hudson Bay region is greatly reduced or disappears altogether (Gough and Wolfe 2001; Gough and Leung 2002). Earth dens in the permafrost remain as persistent landscape features, and they can be re-used by bears over several decades (Scott and Stirling 2002). Richardson et al. (2007) showed that forest fires caused significant degradation of frozen peat in the Manitoba denning area by removing surface vegetation and enabling greater thawing of the active layer; this increased the likelihood of slumping and collapse of previously used dens. Annual mean temperature in Polar Bear Provincial Park is projected to increase from -4.5 °C to nearly 2 °C by 2100, and mean maximum temperature of the warmest month is projected to increase from 17.3 °C to 22.8 °C (McKenney et al. 2010). As forest fires are likely to increase under a warming climate (e.g., Price and Rind 1994; see also Section 2.4.2.2, *Fire*), it is a threat to critical maternity denning habitat in Manitoba and Ontario.

Polar bears in Hudson Bay feed primarily on ringed seals, with smaller amounts of other marine mammals, such as bearded seal (*Erignathus barbatus*), harbor seal (*Phoca vitulina*), and harp seal (*Phoca groenlandica*) (Thiemann et al. 2008). They are dependent on such prey species to meet their annual energy budget. When on land during the sea ice-free season, polar bears have been reported to eat various terrestrial food items, including berries, goose eggs, and flightless geese (Russell 1975; Smith and Hill 1996; Rockwell and Gormezano 2009; Rockwell et al. 2010; Smith et al. 2010), though these items are not considered to be significant contributions to their annual energy budget (Lunn and Stirling 1985; Ramsay and Hobson 1991; Hobson et al. 2009). Recently, Rockwell and colleagues (Rockwell and Gormezano 2009; Rockwell et al. 2010) suggested that, as the break-up of sea ice occurs progressively earlier in Hudson Bay, polar bears coming off the ice in Manitoba might overlap more and more with the nesting season of lesser snow goose (*Chen caerulescens caerulescens*), and this potential food source might supplement for lost seal-hunting opportunities (see also Section 2.4.3.4.1, *Animal Phenology*). It remains to be seen how important such supplemental terrestrial food sources will be to Hudson Plains Ecozone polar bear subpopulations in the future (see also Dyck and Kebreab 2009).

Observations of grizzly bears (*Ursus arctos*) have increased in the Manitoba portion of the Hudson Plains Ecozone, which is well south of the species' historic distribution (Clark 2000; Rockwell et al. 2008). The presence of grizzly bears there might lead to their increased interactions with polar bears, although the significance of such interactions is largely unknown (Doupé et al. 2007). Hybridization between the two species has occurred at least once in the wild (Schliebe et al. 2006).

As a top predator, polar bears tend to bioaccumulate fat-soluble environmental contaminants (Norstrom et al. 1988; Bentzen et al. 2008). Arctic distribution of environmentally persistent organic pollutants (POPs) occurs mainly through long-range transport via atmospheric and oceanic circulation from their source (Iwata et al. 1993). Many POPs have been detected in the tissues of polar bears throughout their global range, including bears of the Western Hudson Bay and Southern Hudson Bay subpopulations (Norstrom et al. 1988, 1998; Polischuk et al. 2002; McKinney et al. 2009). For example, contaminants found in the adipose tissue of adult bears (>4 years old) from these two subpopulations during 1989-1993 included polychlorinated biphenyl congeners (∑PCB), chlordane-related compounds and metabolites (∑CHL), 4,48-DDE (DDE), and dieldrin (DIEL). At the time, polar bears from the Southern Hudson Bay subpopulation showed particularly high concentrations of ∑CHL, DDE, and DIEL relative to other polar bear populations sampled from around the world (Norstrom et al. 1988, 1998).

Trends in contaminant levels in these subpopulations are variable (Braune et al. 2005; McKinney et al. 2009) with levels of some contaminants declining, including legacy contaminants, such as the pesticide DDT (dichlorodiphenyltrichloroethane) (Figure 76). However, new POPs like the brominated flame retardants (e.g., polybrominated diphenyl ethers, ∑PBDEs) and perfluoroalkyl contaminants are rapidly increasing in concentration in arctic and subarctic regions (Martin et al. 2004; Braune et al. 2005; Smithwick et al. 2005; Muir et al. 2006; McKinney et al. 2009), with polar bears from the Southern Hudson Bay subpopulation showing higher concentrations than those from other areas (Giesy and Kannan 2001; Martin et al. 2004; Smithwick et al. 2005; Letcher et al. 2009). In the Western Hudson Bay subpopulation, diet shifts by polar bears (1991-2007) have led some contaminant concentrations (∑PCB, ∑CHL, ∑PBDE, and β-HCH) to increase through time, with ∑PBDE increasing an estimated 28% faster than if diet had remained unchanged. It is suggested that these diet shifts are consistent with climate change predictions, and in years with earlier ice break-up there was a decrease in the consumption of ice-associated bearded seals (benthic feeders) and a subsequent increase in the consumption of one or more of the open water-associated seal species (harbour and/or harp seals) that are known to contain higher levels of some of these contaminants (Thiemann et al. 2008; McKinney et al. 2009). Interestingly, ∑DDT decreased an estimated 64% faster than if there had been no diet shift (McKinney et al. 2009). Based upon limited data (2001-2003), it was not clear if similar trends in diet might be occurring in the Southern Hudson Bay subpopulation (Thiemann et al. 2008). The impacts that these measured levels of contaminants might have on wild polar bears is unclear, but impaired endocrine and immune function and reproductive effects have been suggested (Haave et al. 2003; Lie et al. 2004; Fisk et al. 2005; Letcher et al. 2010), which emphasizes the importance of continued monitoring.

Many metals/elemental contaminants (i.e., mercury, lead, cadmium, etc.) occur naturally in the arctic and subarctic, although some have increased due to human activities and can accumulate to levels that might affect polar bears (see Braune et al. 1991; Dietz et al. 1998; Rush et al. 2008). Concentrations of 21 elements measured from Canadian polar bears (including the Southern Hudson Bay subpopulation) have not changed significantly, since the 1980s, and all levels within the Southern Hudson Bay subpopulation were below those associated with toxicity effects (Braune et al. 1991; Rush et al. 2008). Interestingly, levels of mercury and selenium in livers from Hudson Bay polar bears were much lower than in bears from other regions sampled in North America and Greenland, and levels were well below the thresholds thought to cause health effects (Rush et al. 2008).

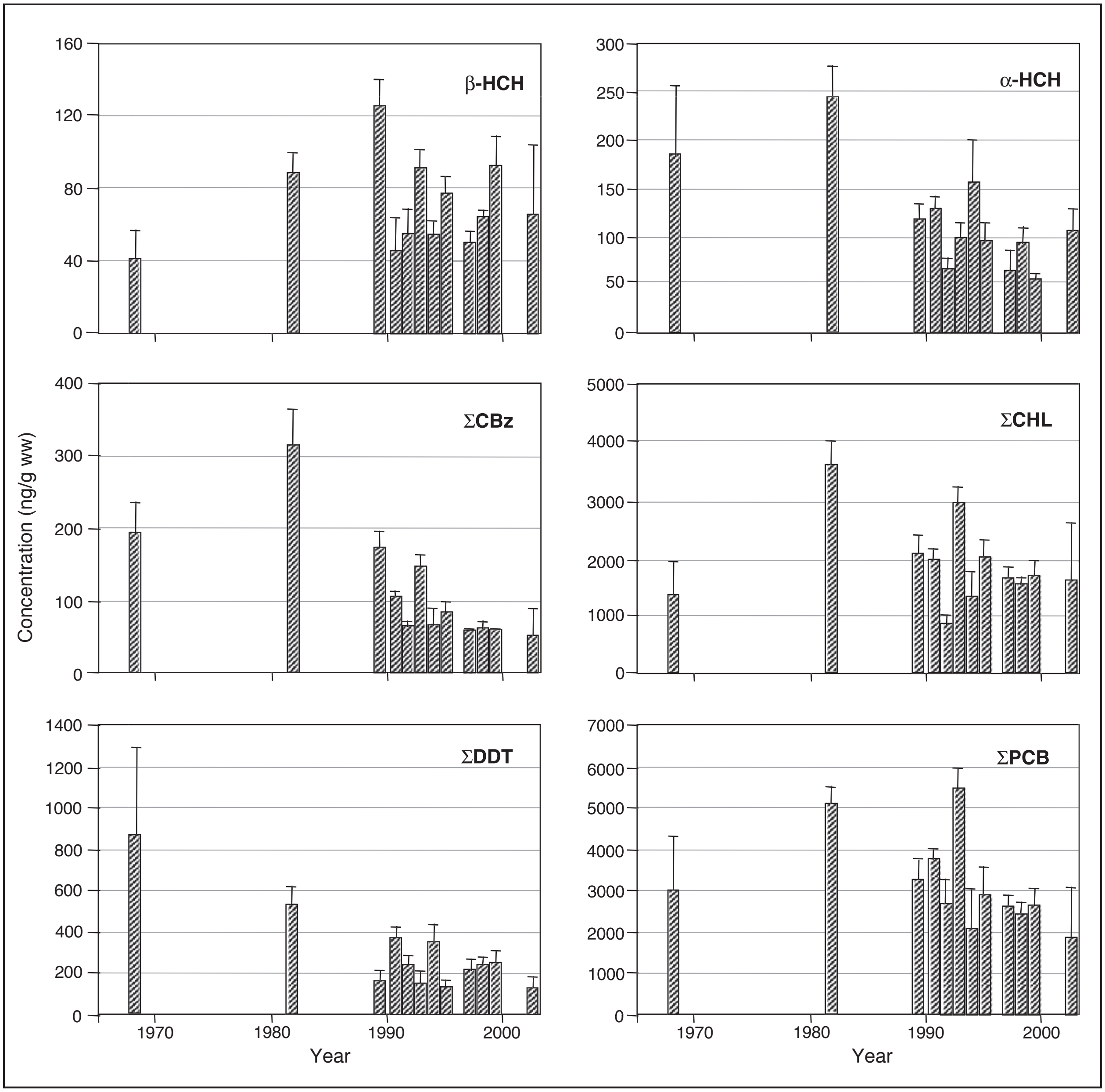

Figure 76. Temporal trends of major organochlorines in the adipose tissue of polar bears from the Western Hudson Bay subpopulation (Norstorm 2001; Letcher et al. 2003). Samples are from the Churchill area of western Hudson Bay from 1968 to 2002. Samples from 1991 to 2002 are fat biopsies, but earlier samples are adipose tissue. Abbreviations: β-HCH, beta-hexachlorocyclohexane; α-HCH, alpha-hexachlorocyclohexane; ∑CBz, chlorobenzenes; ∑CHL, chlordanes; ∑DDT, dichlorodiphenyltrichloroethane and its metabolites; and ∑PCB, polychlorinated biphenyl congeners.
Source: Braune et al. (2005). Reprinted from Science of the Total Environment, Vol 351-352, B.M. Braune, P.M. Outridge, A.T. Fisk, D.C.G. Muir, P.A. Helm, K. Hobbs, P.F. Hoekstra, Z.A. Kuzyk, M. Kwan, R.J. Letcher, W.L. Lockhart, R.J. Norstrom, G.A. Stern and I. Stirling, Persistent organic pollutants and mercury in marine biota of the Canadian arctic: an overview of spatial and temporal trends, p 42, Copyright 2005, with permission from Elsevier.

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2.3.3.2 **Terrestrial mammals**

Approximately 50 species of terrestrial (non-marine) mammals have been reported in the Hudson Plains Ecozone (Smith and Foster 1957; Wrigley 1974; Bider 1976; Dubois 2001; Dobbyn 1994; Prescott and Richard 2004; Redpath Museum 2010). Terrestrial mammals considered characteristic of the ecozone include caribou (*Rangifer tarandus*), moose (*Alces alces*), black bear (*Ursus americanus*), wolverine (*Gulo gulo*), arctic fox (*Vulpes lagopus*), and American marten (*Martes americana*) (Smith et al. 1998; Abraham and Keddy 2005; Crins et al. 2009). Other prominent species include grey wolf (*Canis lupus*), red fox (*Vulpes vulpes*), American beaver (*Castor canadensis*), snowshoe hare (*Lepus americanus*), arctic hare (*Lepus arcticus*), and common porcupine (*Erethizon dorsatum*). Additional carnivores (Carnivora) and rodents (Rodentia) also commonly occur, as do a variety of bats (Chiroptera) and shrews and moles (Insectivora). Other mammals are sighted only occasionally in the ecozone, such as the grizzly bear (*Ursus arctos richardsoni*) (Rockwell et al. 2008). No terrestrial mammal species native to the ecozone is currently reported as extirpated, i.e., a full complement of native species is present. The recovery of species that were historically extirpated or near-extirpated from this ecozone (e.g., American beaver during the fur trade era) was subsequently aided by restocking (see Section 2.3.3.2.3, *Furbearers*).

In general, the diversity of terrestrial mammal species, which is largely temperature-dependent in Canada (Kerr and Packer 1996), is lower in the Hudson Plains Ecozone than in areas further south (Currie 1991). Similarly, the diversity of mammals within the ecozone generally reflects climatic gradients – with mammals of arctic affinity (e.g., arctic fox and Richardson's collared lemming, *Dicrostonyx richardsoni*) found principally in more northerly tundra areas and a greater diversity of forest-dependent mammals further inland and south (e.g., Wrigley 1974; Hersteinsson and Macdonald 1992). Most of the ecozone's mammal species have received very limited or sporadic population monitoring, and existing observations of their distributions tend to be based largely on non-systematic sampling. Nonetheless, the ecozone's terrestrial mammal populations are generally assumed to be in comparatively good condition overall, given limited exposure to stressors (see Section 2.6.1, *Stressors & Cumulative Impacts*). Of the terrestrial mammal species that commonly occur in this ecozone, two are species of national conservation concern: wolverine and the boreal population of woodland caribou (forest-dwelling ecotype) (Section 2.3.2, *Trends in Species of National Conservation Concern*).

The profiles below detail what is known about status and trends of the following terrestrial mammal species or assemblages considered to be of *special interest*[46] to the Hudson Plains Ecozone: caribou, moose, and furbearing mammals (particularly wolverine, grey wolf, American marten, and American beaver). Although these and other mammal species native to the ecozone can be affected by other human influences (limited at present), accelerated climate change, which is expected to alter the distribution of much of Canada's mammal diversity in the future (Kerr and Packer 2006), is an important emerging threat. Sea ice-dependent mammals, such as arctic fox, and other species of arctic affinity are at most immediate risk, owing to expected habitat loss or greater interspecific competition at the southern extent of their distributions, combined with the Arctic Ocean acting as a barrier to their movement north (Kerr and Packer 2006).

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[46] The term *species of special interest* was defined in Section 2.3.3.

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2.3.3.2.1 **Caribou**

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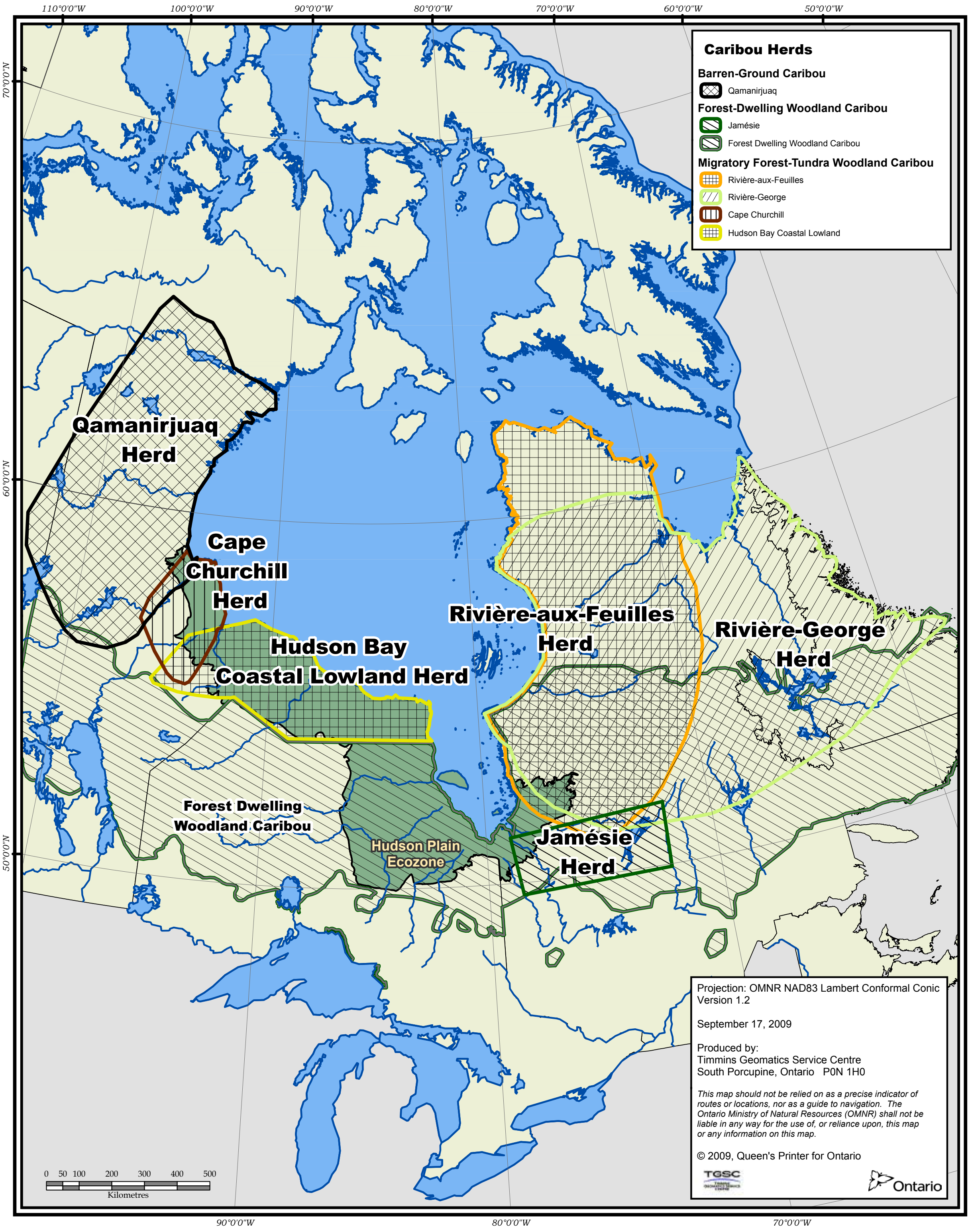
Woodland caribou (*Rangifer tarandus caribou*) is ecologically and culturally important in the Hudson Plains Ecozone. Ecologically, its status serves as an indicator of general ecosystem integrity. In general, this subspecies requires large areas of undisturbed mature coniferous forest and is sensitive to human disturbance (Rettie and Messier 2000; Vors et al. 2007; Bowman et al. 2010). Within the Hudson Plains Ecozone it also requires undisturbed coastal and tundra habitats, which are used from calving through rut (Abraham and Thompson 1998; Magoun et al. 2005). Woodland caribou is also culturally important to the ecozone's Aboriginal peoples, forming an important part of their traditional subsistence lifestyle (Berkes et al. 1994).

Currently, two ecotypes of woodland caribou inhabit the Hudson Plains Ecozone (Figure 77): the more southerly and sedentary, forest-dwelling ecotype and the more northerly and migratory, forest-tundra ecotype. Both ecotypes occur in the Ontario and Québec portions of the ecozone, where they overlap. Only the forest-tundra ecotype is thought to occur in the Manitoba portion of the ecozone (Manitoba Conservation 2005; Environment Canada 2008). During some winters, the barren-ground caribou subspecies (*Rangifer tarandus groenlandicus*) from the Qamanirjuaq herd occasionally migrate into the Manitoba portion of the ecozone (BQCMB 2009; see also Figure 77). Caribou rarely occur on Akimiski Island, Nunavut.

In 2002, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessed the Boreal population of the forest-dwelling ecotype of woodland caribou as Threatened due to population declines throughout most of the range and threats from habitat loss and increased predation, possibly facilitated by human activities (Thomas and Gray 2002). This population is legally protected under the federal Species at Risk Act (Government of Canada 2010). The status of the migratory forest-tundra ecotype of woodland caribou in Canada has not been reviewed by COSEWIC, however, its status in the Hudson Plains Ecozone was reviewed in Ontario by the Committee on the Status of Species at Risk in Ontario (COSSARO) (Harris 1999). It was not listed at the time, because of apparent increases of the Pen Islands herd, since the 1970s (the Pen Islands herd is represented in Figure 77 as the Hudson Bay Coastal Lowland herd). In Ontario, COSSARO has recently placed this ecotype on a priority list for reassessment. The status of barren-ground caribou has not been assessed by COSEWIC.

Monitoring efforts

Status and trends assessment of caribou in the Hudson Plains Ecozone is hindered by inconsistent and incomplete monitoring efforts. Manitoba does not have a basic inventory and monitoring program, but periodic surveys have been conducted in parts of the ecozone, including recent summer coastal surveys in 2008 and 2009. In Ontario, systematic winter caribou surveys of the entire ecozone have been conducted at various times, since 1959 but only sporadically and with changing methods (Magoun et al. 2005). Ontario is conducting a comprehensive aerial survey between 2008 and 2011 of most of its portion of the ecozone to determine current wintering areas and summer coastal distribution of woodland caribou. In 2009, Ontario radio-collared 100 woodland caribou, including both forest-dwelling and migratory forest-tundra ecotypes, more than half of which use the Hudson Plains Ecozone for at least part of their annual range (Ontario Ministry of Natural Resources, unpublished data). In 2010, an additional 29 forest-tundra caribou were collared within the Hudson Plains Ecozone, 20 and nine in Ontario and Manitoba, respectively. Manitoba is collaring additional caribou in this area in 2011 (V. Trim, Manitoba Conservation, pers. comm.). Combined with local information from Aboriginal communities, these programs should improve the current state of knowledge on caribou winter distribution and summer coastal distribution and better inform the delineation of forest-dwelling and forest-tundra ecotypes of woodland caribou. Additional collaring of woodland caribou was undertaken for impacts-monitoring purposes by AMEC in the vicinity of the Victor mine (90 km west of Attawapiskat) in Ontario (AMEC 2008) (see below). In Québec, some information is available for three populations of woodland caribou whose annual ranges touch the eastern extremes of the Hudson Plains Ecozone (Boulet et al. 2007).



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September 17, 2009

Produced by:
Timmins Geomatics Service Centre
South Porcupine, Ontario P0N 1H0

Figure 77. Approximate distribution of caribou herds in and around the Hudson Plains Ecozone (the ecozone is denoted with green shading). All herds shown are woodland caribou herds, except the Qamanirjuaq herd, which is a barren-ground herd that only occasionally migrates into the ecozone. The migratory Pen Islands herd of woodland caribou (forest-tundra ecotype) is represented on the map as the Hudson Bay Coastal Lowland herd. Caribou rarely occur on Akimiski Island.

Forest-dwelling ecotype of woodland caribou

The forest-dwelling ecotype of woodland caribou is more sedentary and solitary compared with the migratory forest-tundra ecotype of woodland caribou and barren-ground caribou. Forest-dwelling caribou do not tend to undertake long-distance migrations to open-country calving grounds nor do they generally occur in aggregations of hundreds, as do the forest-tundra animals. Instead, forest-dwelling caribou characteristically exist at low densities, and they spatially separate themselves from conspecifics at calving and for most of the year, as well as from predators and other ungulate species that serve as alternate prey for the major predators, to minimize predation risk (Bergerud 1985; Rettie and Messier 2000).

Related to the remoteness of the ecozone, there is currently no information to suggest range recession or population decline of forest-dwelling woodland caribou in the Hudson Plains Ecozone, as there is elsewhere in Ontario and Canada (Thomas and Gray 2002; Schaefer 2003; Vors et al. 2007; Ontario Woodland Caribou Recovery Team 2008). For purposes of management, the current distribution of forest-dwelling caribou in the ecozone (see Figure 77) is considered to be south of the southern boundary of Ontario's Wildlife Management Units (WMUs) 1A and 1B, though this line is not biologically based, and further information is required to better delineate a boundary or zone of overlap (Ontario Woodland Caribou Recovery Team 2008). For example, a recent caribou radio-collaring project near the Victor mine in Ontario, close to the current boundary between the two ecotypes of woodland caribou, has documented some extensive movements. Six of ten collared caribou wintered in the Hudson Plains Ecozone west of Peawanuck, Ontario, with two of them moving into Manitoba, notably near the village of Shamattawa (AMEC 2008). Preliminary results from 2009 and 2010 radio-collaring studies in Ontario suggest that there is considerable overlap during winter between the forest-dwelling ecotype and the forest-tundra ecotype, which move southward out of the Hudson Plains Ecozone. This overlap occurs at least along the ecotone of the Boreal Shield and Hudson Plains ecozones (Ontario Ministry of Natural Resources, unpublished data). However, detailed knowledge of the distribution and overlap between the ecotypes is lacking, because of a general paucity of data on movements and genetic relationships in areas and during seasons of overlap between them. Current studies under way on all of these aspects should help clarify these relationships.

Systematic winter caribou surveys undertaken periodically, since 1959, throughout the Ontario portion of the Hudson Plains Ecozone (summarized in Magoun et al. 2005) have shown that, from 1959 to 2003, winter distribution patterns of caribou remained similar, though some core wintering areas might have shifted. Winter densities of caribou calculated during these periodic surveys in Ontario varied between 0.015 caribou/km^2 and 0.141 caribou/km^2 (Figure 78; Magoun et al. 2005). Because study areas and methods varied among surveys, it was not possible to detect any trends or changes. However, preliminary data from a 2008 winter survey of forest-dwelling caribou in Ontario's WMU 1D (James Bay Lowlands), in which methods and study areas were similar to earlier work, suggests that caribou densities have increased in this WMU from 0.01 caribou/km^2 (Thompson 1986) to 0.04 caribou/km^2 (Chenier 2008), since 1983-1984.

In the Québec portion, the western range of the forest-dwelling Jamésie herd (~600 animals) occurs within the Hudson Plains Ecozone (Boulet et al. 2007; Figure 77). This herd is currently considered stable (Callaghan et al. in press).

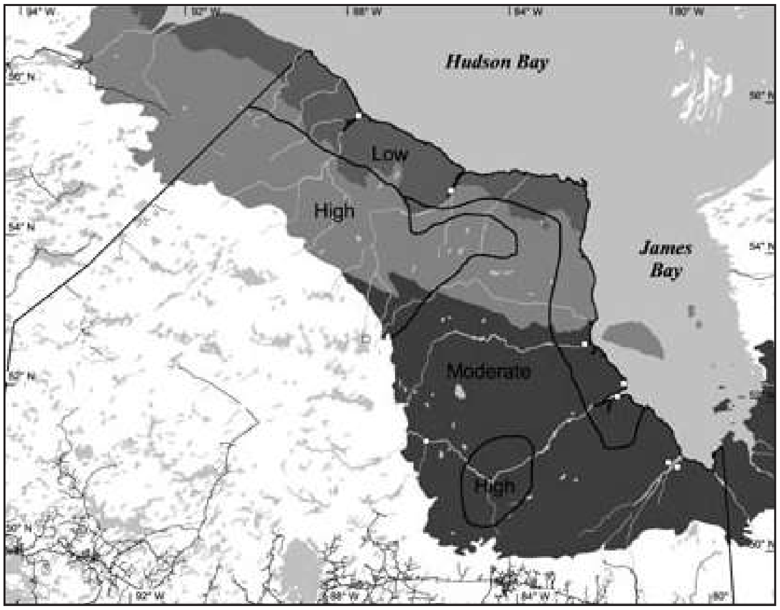

Figure 78. Relative abundance of woodland caribou in winter (January to March) in the Ontario portion of the Hudson Plains Ecozone, based on four systematic aerial surveys from 1959 to 2003. Shading denotes the three ecoregions. Areas of high, moderate, and low caribou abundance are delineated with solid lines and defined as: High, ≥0.04 caribou/km^2; Moderate, <0.02 caribou/km^2; and Low, areas nearly devoid of caribou during the survey period.
Source: Magoun et al. (2005). Figure is used with permission of the Nordic Council for Reindeer Husbandry Research, publisher of the journal Rangifer.

Figure 79. Migratory forest-tundra ecotype of woodland caribou near Cape Henrietta Maria, Ontario on July 11, 2008. Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Forest-tundra ecotype of woodland caribou

The forest-tundra ecotype of woodland caribou (Figure 79) has several features that distinguish it from the more sedentary forest-dwelling ecotype: it is migratory and more gregarious and, during calving, in summer, and during rut, this ecotype is found in the more open taiga and tundra habitats associated with coastal areas of the Hudson Plains Ecozone. A large proportion of females of this migratory ecotype are thought to give birth to calves in aggregations near the Hudson Bay coast in Manitoba and Ontario. Written descriptions of observations about this general behaviour go back as far as the 1770s (Thompson and Abraham 1994; Abraham and Thompson 1998).

The Cape Churchill migratory forest-tundra herd has been defined as ranging exclusively in the Cape Churchill Wildlife Management Area, though surveys of this herd have not been regularly undertaken. Aerial surveys of the herd in 1965 estimated the population at approximately 58 individuals, while another survey in 1988 estimated the population ranging between 1,800 and 2,200 individuals (Campbell 1994). The minimum population size was estimated in 1997-98 to be 3,013 adult caribou (Elliott 1998). Parks Canada conducted an aerial survey on May 28-29, 2005 and along flight lines over the known calving area and counted 644 animals (H. Stewart, Parks

Canada-Wapusk National Park, pers. obs.). Three counts of an opportunistic aerial photograph survey taken on July 20, 2007 averaged 2,937 adult animals, suggesting no change in the minimum population size from 1997-98. Observations indicate that Cape Churchill caribou and the barren-ground Qamanirjuaq herd occasionally share portions of the same winter range (BQCMB 2009). Some researchers suggest that the Cape Churchill herd is relatively isolated with some mixing with adjacent herds (e.g., Thomas and Gray 2002).

The Pen Islands migratory forest-tundra herd became the focus of attention in the mid-1980s after several years of observations of large numbers along the coast in summer. Periodic photographic counts of the Pen Islands herd during summer along the Hudson Bay coast of Manitoba and Ontario from 1979 to 1994 estimated that the herd increased from a minimum of 2,300 animals in 1979 to a high of 10,798 animals in 1994 (Abraham and Thompson 1998). This increase had implications for Aboriginal harvest, potential tourism opportunities, and inter-jurisdictional management (Thompson and Abraham 1994). An attempt in 1991 to form a Pen Islands Caribou Management Council comprised of Manitoba and Ontario governments and the Aboriginal communities of Shamattawa (Manitoba) and Fort Severn (Ontario) (see Appendix 1 in Thompson and Abraham 1994) failed. Since that time, the need for a management board has been raised by residents of Shamattawa, who have become increasingly concerned about the health and status of this herd (Manitoba Conservation, pers. comm.).

In recent years, major changes have occurred in the status of the Pen Islands herd, which traditionally calved and summered on the Hudson Bay coast in the area of the Pen Islands and wintered inland close to the boundary of the Boreal Shield and Hudson Plains ecozones near the Ontario/Manitoba border. Using photographic survey data (Thompson and Abraham 1994), non-systematic surveys, and incidental observations of caribou during summer along the Hudson Bay coast from 1965 to 2003, Magoun et al. (2005) documented an eastward shift in summer use of coastal areas by forest-tundra caribou, since the late 1990s, with caribou becoming more common east of the Severn River (Figure 80). During 2004-2007, opportunistic summer surveys of the Hudson Bay coastal area in Ontario revealed increasing numbers east of the Winisk River (Ontario Ministry of Natural Resources, unpublished data). Then, in 2008 and 2009, systematic spring and summer surveys of caribou were conducted over the entire southern Hudson Bay coastal area from the Hayes River in Manitoba to the Lakitusaki River in Ontario (Abraham et al. in press). These surveys documented further evidence of a major change in summer use of coastal areas, with most caribou being observed even farther east, near Cape Henrietta Maria. These surveys also indicate the possibility of a decline in the number of forest-tundra caribou that summer in the Coastal Hudson Bay Lowlands Ecoregion (Figure 78) (3,529 and 3,304 caribou were tallied in the 2008 and 2009 surveys, respectively). These results (proportionally larger numbers in the east and lower numbers in the coastal ecoregion overall) might represent: 1) a shift in range use and behaviour of the Pen Islands herd within the broader Hudson Plains Ecozone; 2) an independent decrease in numbers of caribou in the former Pen Islands range coupled with an independent increase in numbers in the east; or 3) some combination of those and other population or behavioural changes. Both groups of forest-tundra caribou appear to winter in the southern portion of the Hudson Plains Ecozone or on the adjacent ecotone with the Boreal Shield Ecozone. Some of the caribou in the new summer concentration area in the east spend some portion of the winter between the Severn River and James Bay and may also be associated with the former Pen Islands herd range (Ontario Ministry of Natural Resources, unpublished data).

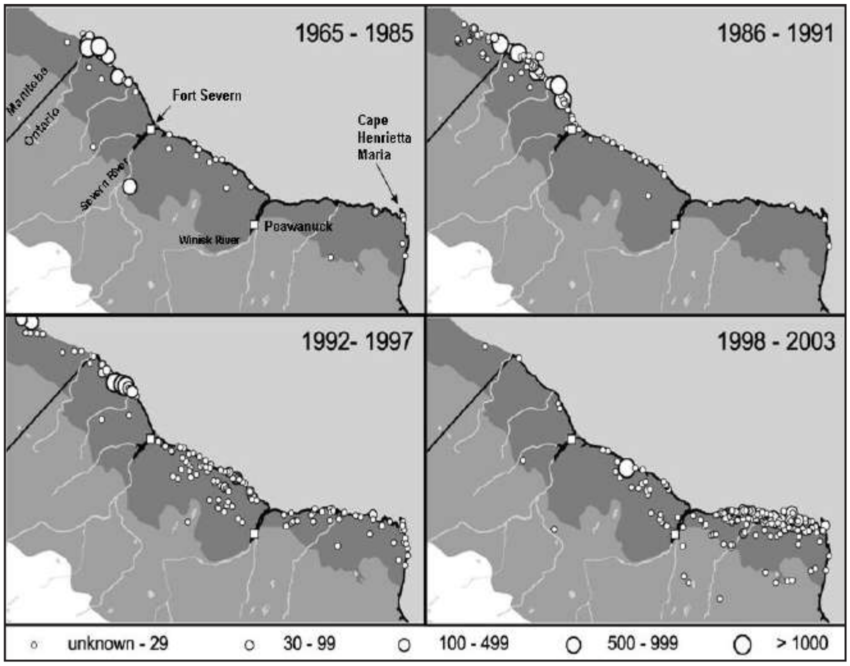

Figure 80. Summer locations of forest-tundra caribou along the Hudson Bay coast during four periods from 1965 to 2003, based on incidental observations and photographic surveys. Each circle represents an observation, with the relative size of the circle representing the size of the aggregation observed.
Source: Adapted from Magoun et al. (2005). Figure is used with permission of the Nordic Council for Reindeer Husbandry Research, publisher of the journal Rangifer.

Multiple factors might be responsible for the above two possibilities (i.e., distributional change or population decline). Among them are deterioration of range condition in the Pen Islands area, leading to decreased food availability; increased predator densities; disturbance; and harvest (non-Aboriginal hunting of caribou was banned in Ontario in 1929, but it continues on a limited and licensed scale in the Manitoba portion of the ecozone; OMNR 2008; Manitoba Conservation 2010; and for more information on caribou harvest, see Section 2.5.3, *Provisioning Services*). None of these factors has been examined. However, the learned avoidance of areas of high harvest pressure and/or high disturbance are suspected to be contributing, because of the pattern of change (Ontario Ministry of Natural Resources, unpublished data). For example, a winter road from Shamattawa to Fort Severn now bisects the winter range of the Pen Islands animals, and ATV travel on summer coastal ridge habitats in that area, where few caribou are now found, has greatly increased (Section 2.2.2.2, *Polar-Tundra*). Additionally, occasional unusually high Aboriginal harvests do occur. For example, on one occasion, an estimated 400-613 caribou were harvested in Peawanuck in the fall of 1996, while approximately 200 caribou were harvested from Fort Severn that same season (Ontario Ministry of Natural Resources, unpublished data). Similarly, the overall harvest by Aboriginal peoples in the Manitoba portion of the Hudson Plains Ecozone over the last two decades has been higher than previous harvest levels (V. Crichton and D. Hedman, Manitoba Conservation, pers. comm.). Nevertheless, no comprehensive estimate of forest-tundra caribou population size or true population trend currently exists for the ecozone. Therefore, the sustainable harvest level is unknown, and no clear cause-effect relationship can be ascribed.

The Québec portion of the Hudson Plains Ecozone is within the western periphery of the annual ranges of two herds of the migratory forest-tundra ecotype: the George River (Riviére-George) herd and the Leaf River (Rivière-aux-Feuilles) herd (Figure 77). Periodic surveys of the George River herd indicated that the herd began to grow in the 1960s, reaching a high of 776,000 (±104,000) (90% CI) in 1993 (Couturier et al. 1996) and decreasing to 385,000 (±108,000), by 2001 (Couturier et al. 2004, 2009b). Researchers suggested the decline likely started in 1989 due to deteriorating climatic and habitat conditions (Boudreau et al. 2003; Boudreau and Payette 2004a,b; Couturier et al. 2009a,b). A 2010 post-calving photocensus of the George River herd suggests that this herd has declined further, to 74,100 ± 12,600 animals (QRNF 2010). The Leaf River herd increased from 56,000 in 1975, and it continued to increase to 1,193,000 (±565,000) in 2001, though because of the large confidence interval in 2001, the lower confidence limit (628,000) was recommended as the basis for management (Couturier et al. 2004). Weather conditions prevented completion of the Leaf River herd survey in 2010, however, plans have been proposed to attempt this survey again in 2011 (QRNF 2010). Trends in these populations might be only minimally influenced by their limited use of the Hudson Plains Ecozone.

Barren-ground caribou

The barren-ground Qamanirjuaq herd numbers were low in the 1970s and grew to 496,000 animals by 1994 (Gunn et al. in press). In 2008, calving ground and fall composition surveys estimated 349,000 ± 44,900 caribou, which suggests a ~30% decline in the population from 1994. This decline is, however, not statistically significant. Herd composition surveys flown from 1992 to 2008 show a continuing trend for a decline in calf:cow ratios (Campbell et al. 2010). Concern exists that this herd might be declining and that overall harvest levels are unsustainable (BQCMB 2010). Trends in this population might be only minimally influenced by its limited use of the Hudson Plains Ecozone.

Threats

As noted, woodland caribou is a species that is sensitive to human disturbance; human activities tend to degrade caribou habitat (Schaefer 2003; Vors et al. 2007; Bowman et al. 2010). These animals require large patches of undisturbed, mature coniferous forest and, in the Hudson Plains Ecozone, undisturbed coastal and tundra habitats. Human developments in the Hudson Plains Ecozone are increasing, and they include existing and proposed hydroelectric and mining projects, electricity transportation corridors, and winter and all-season roads (see sections 1.2.2, *Economic History* and 2.6.1, *Stressors & Cumulative Impacts*). These developments allow hunters and predators greater access to caribou, and they can also create barriers to caribou movement and distribution (James and Stuart-Smith 2000; McLoughlin et al. 2003; Weir et al. 2007; Vistnes and Nellemann 2008). Additional threats include increased direct mortality through harvest (see Section 2.5.3, *Provisioning Services*). Another emerging issue is the effect of climate change on caribou habitat in the ecozone, with the potential consequence of changing caribou status (Sharma et al. 2009; Vors and Boyce 2009).

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2.3.3.2.2 **Moose**

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Moose (*Alces alces*) is a species of cultural and ecological importance in the Hudson Plains Ecozone, as an important food source for Aboriginal peoples, as well as predators. The species occurs throughout the ecozone (see below) but at generally lower densities than in the Boreal Shield Ecozone to the south (Ontario Ministry of Natural Resources, unpublished data). Within the Hudson Plains Ecozone, the most productive moose habitat is located along the better-drained riparian areas of rivers and where burns occurred in the last 20-30 years.

The current status and trends of moose populations is a knowledge gap for this ecozone. Manitoba currently does not have an established and active monitoring program for moose in its portion of the ecozone. Opportunistic data collected by Manitoba Conservation staff indicate that moose is present in the area, but the status and trends of this species there are unknown.

Ontario has not had a regularly scheduled moose population monitoring program across its part of the ecozone, though the southern half of its part of the ecozone has been surveyed on a more regular basis (range of 1 to 13 years between surveys from 1976 to 2008). Most moose surveys in the northern portion were conducted from 1982 to 1986. Overall, estimated moose densities range from 0.002 moose/km^2 to 0.08 moose/km^2 (Ontario Ministry of Natural Resources, unpublished data), with higher densities occurring in the southern portion. However, density estimates are low and variable throughout, with no clear trends or detectable changes in abundance. In part this is due to the fact that good moose habitat is highly non-random in this ecozone and not amenable to the transect survey method used in early surveys. Some additional ungulate survey work was completed in the Ontario portion of the ecozone during 2009 and 2010, as part of this province's Far North initiative (for more information on this initiative, see Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*).

Moose populations are surveyed periodically in Québec, but hunting Zone 22 (northerly zone, which includes, but is much larger than, Québec's portion of the Hudson Plains Ecozone) has not been aerially surveyed for moose since 1991, when moose density was estimated at 0.026 moose/km^2 (Annex 1 in Lefort and Huot 2008). This density of moose is the lowest among all hunting zones in Québec (Lefort and Huot 2008).

Compared with more human-populated ecozones, few anthropogenic stressors presently influence moose in the Hudson Plains Ecozone. Moose are highly valued as food by Aboriginal harvesters (Section 2.5.3, *Provisioning Services*). Current threats to moose in the ecozone are associated with resource development activities (i.e., hydroelectric development, mining, and associated infrastructure). These developments might result in increased human access and, thus, additional harvesting pressure (Mulrennan and Scott 1995) (for information on moose harvests in the ecozone, see Section 2.5.3, *Provisioning Services*).

The impacts that climate change might have on moose in the Hudson Plains Ecozone are poorly understood, but they might be both positive and negative depending on the scale of change. For example, an increase in forest fire frequency could be positive in generating younger forest stands and additional suitable moose habitat (e.g., Davis and Franzmann 1979; Rempel et al. 1997). However, at the southern edge of the ecozone, these habitat changes might enable wildlife species like white tailed deer (*Odocoileus virginianus*) to move north, bringing with them parasites and pathogens that could be serious for otherwise unexposed populations of moose in the Hudson Plains Ecozone (Murray et al. 2006).

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2.3.3.2.3 **Furbearers**

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Furbearers (species traditionally trapped or hunted for their fur) are a key group of mammalian species in the Hudson Plains Ecozone from an ecological, cultural, and economic point of view. Most North American furbearer species occur to some extent in the ecozone, although some are much more widespread and abundant than others (Table 17). Specifically, American beaver, American marten, northern river otter, American mink, and muskrat have been the most commonly harvested furbearers in this region, since the beginning of the fur trade in the 1600s (Lytwyn 2002) through to the present (Berkes et al. 1994; Manitoba Conservation, unpublished data; Ontario Ministry of Natural Resources, unpublished data; Québec Ministère des Ressources Naturelles et de la Faune, unpublished data). Historically, such species were valuable as subsistence resources for both pelts (in winter) and meat (year round), although residents of the Hudson Plains Ecozone focused more of their efforts on migratory bird hunting, fisheries, seasonal caribou hunts, and some marine mammal hunting (Lytwyn 2002; see also Section 2.5.3, *Provisioning Services*). In recent years, marten is the most harvested (and valuable) furbearer in this ecozone, followed by beaver and muskrat (Figure 81). The total number of animals harvested for each species is considerably smaller in Manitoba and Québec than in Ontario, with the exception of beaver, arctic fox, and wolverine (see Figure 83 for wolverine).

Table 17. North American furbearer species present in the Hudson Plains Ecozone.
Sources: Bider (1976); Dobbyn (1994); Manitoba Conservation (unpublished data); and Ontario Ministry of Natural Resources (unpublished data).

Species		Distribution in ecozone
Scientific name	Common name	
Canis latrans	Coyote	Localized at southern edge of ecozone
Canis lupus	Grey wolf	Widespread but scattered
Castor canadensis	American beaver	Widespread, more abundant inland
Gulo gulo	Wolverine	Western and northern part of ecozone
Lontra canadensis	Northern river otter	Widespread
Lynx canadensis	Canada lynx	Widespread but scattered
Martes americana	American marten	Widespread
Martes pennanti	Fisher	Restricted to southern edge of ecozone
Mephitis mephitis	Striped skunk	In human settlements
Mustela erminea	Short-tailed weasel	Widespread
Mustela nivalis	Least weasel	Likely widespread
Neovison vison	American mink	Widespread
Ondatra zibethicus	Muskrat	Widespread
Procyon lotor	Raccoon	At southern edge of ecozone and in human settlements
Tamiasciurus hudsonicus	Red squirrel	Widespread
Vulpes lagopus	Arctic fox	Hudson Bay and James Bay coasts
Vulpes vulpes	Red fox	Widespread

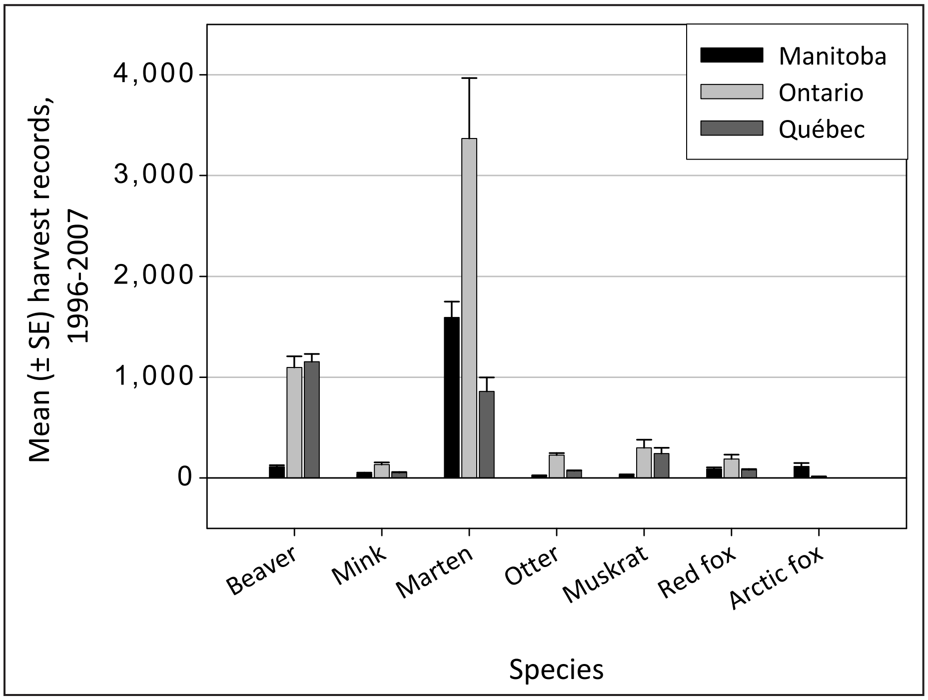

Figure 81. Mean (± SE) harvest records of most harvested furbearers in the Manitoba, Ontario, and Québec portions of the Hudson Plains Ecozone, 1996-2007. Arctic fox does not occur in James Bay south of Akimiski Island. Sources: Manitoba Conservation (unpublished data); Ontario Ministry of Natural Resources (unpublished data); and Québec Ministère des Ressources Naturelles et de la Faune (unpublished data).

This observation is at least partially explained by the combined portion of the ecozone in Manitoba and Québec being about one-third the area of Ontario's, although beaver harvest in Québec (10% of the area) was greater during this period than in Ontario (68% of the area).

Most furbearer species are mammalian predators and are generalized with respect to their habitat selection. Accordingly, their distribution and relative abundance tend to be more closely tied to prey abundance than vegetation characteristics. They are known to exhibit population cycles in relation to preferred prey species (e.g., lynx:snowshoe hare, marten:voles, mink:muskrat, arctic fox:lemmings), although the drivers and regularity of such cycles in the Hudson Plains Ecozone are largely unknown (see also Section 2.4.3.1, *Predator-Prey Relationships & Cycles*). Beaver, muskrat, and river otter are more tied to the inland aquatic components of the landscape, while other species, such as marten, are more widespread throughout the ecozone. Some species are quite restricted in their principal distribution, such as coastal tundra habitats (arctic fox) or the southern edge of the ecozone (fisher). Due to the lower productivity of this ecozone relative to the neighbouring Boreal Shield Ecozone, most species occur at relatively low densities.

Monitoring efforts

Because of the paucity of permanent road access (except the James Bay Road in Québec) and relatively low levels of mineral or energy resource exploitation and human settlement, the Hudson Plains Ecozone has received little research and monitoring attention compared to the more accessible and human-modified portions of southern Canada (Thompson 2000). In the Ontario portion of the ecozone, American beaver surveys were conducted in the 1980s (Gauthier and Threader 1981; Cree 1983; Phoenix and Haddow 1984). In addition, a wolverine study commenced in 2003 in Ontario, with aerial surveys for tracks undertaken in 2004-2010 over much of this species' range in the province, including the Hudson Plains Ecozone. Hence, the majority of information for most furbearer species is derived from historical harvest returns from Hudson Bay Company forts as far back as the 1600s and current provincial fur production records (usually, numbers of animals reported as sold to licensed fur dealers).

Production is influenced not just by species populations but also by trapper numbers and individual trapping effort. When these latter factors are thought to be constant (as in historical times), fur harvest records can be reliable indicators of population trends over time. However, concomitant with both the fall in commercial value of pelts, since the late 1980s, and dwindling harvest effort on the land, these records diminish in value for assessing furbearer population trends. They also do not include furbearers taken by trappers for domestic and/or cultural purposes. In this regard, data on grey wolf harvesting, while mandatory for non-Aboriginal harvesters to report, are often considered to be unreliable. Many harvested wolves are not reported if, for example, they are harvested for purposes other than their fur (e.g., nuisance individuals in and around communities, animals with mange) (Buss and de Almeida 1997). Aside from information from harvest returns and provincial fur production records, some information is also available from recorded local and indigenous knowledge (e.g., Berkes et al. 1994, 1995; Ray et al. 2005), although most knowledge of this nature that exists in the region has not been formally documented.

Species profiles

For this section, wolverine, grey wolf, American marten, and American beaver are profiled. Wolverine is the sole species of national conservation concern among Hudson Plains Ecozone furbearers, and grey wolf is generally a species of management interest wherever it occurs. Marten has consistently occupied a place as the most valuable furbearer in the fur trade, even more so than beaver, although the latter was particularly valued by European traders (Lytwyn 2002). Beaver merits additional attention not only as a valuable commercial and subsistence quarry for human hunters but for its ecological role as an ecosystem engineer and key prey species for large predators (Rosell et al. 2005).

Wolverine

Wolverine is a species of conservation concern in the Hudson Plains Ecozone. In Manitoba and Ontario, wolverines represent the eastern extension of the national Western population, which was designated by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as Special Concern (COSEWIC 2003), but it was never listed, and it receives no protection under the Canadian Species At Risk Act (SARA) (Government of Canada 2010). The Québec portion of the ecozone corresponds with the Eastern wolverine population, which was designated by COSEWIC as Endangered (COSEWIC 2003) and subsequently listed as such under SARA. However, there are no confirmed records of wolverine in Québec, since 1978 (Fortin et al. 2005).

Wolverines residing in this ecozone are strategically important for both the maintenance of the Western population and the recovery of the Eastern population. In Ontario, wolverine has been classified as Threatened under the Endangered Species Act, 2007 (Government of Ontario 2007), because: 1) the species' range within the province had declined by ≥50%; 2) human presence and resource development activity is increasing in areas where it presently occurs; and 3) the low reproductive rates and large home-range sizes of wolverines renders populations slow to recover from the loss of many individuals. The species is not designated as at risk under Manitoba legislation, but it has been recommended for Special Concern by the Manitoba Endangered Species Advisory Committee (Berezanski 2004). The Manitoba Conservation Data Centre has assigned it a ranking of S3 (demonstrably secure but can be vulnerable to large-scale disturbances), while Ontario's Natural Heritage Information Centre has assigned the higher status of S2.

Wolverine is physically adapted to northern boreal and tundra environments characterized by the Hudson Plains Ecozone (Banci 1994). While vulnerable to human disturbance, wolverine needs are not specific to any habitat type per se, although the species requires an adequate large ungulate prey base, adequate snow cover, and minimum human conflict (Banci 1994; Magoun and Copeland 1998). Wolverine acquires most large prey through scavenging. Such a lifestyle requires wolverine to cover home ranges much larger than similar-sized carnivores, and its population densities are therefore naturally low, even under optimal conditions (Banci 1994). In this region, direct beaver exploitation by wolverines is thought to be a common phenomenon (Ontario Wolverine Project, unpublished data), although the importance of beaver relative to large ungulates as a wolverine food source is unknown. The combination of factors such as naturally low densities, large home-range sizes, long-distance movements, and low reproductive rate render wolverine vulnerable to over-trapping – particularly when this activity occurs in concert with habitat change (Banci and Proulx 1999).

Wolverine is currently confined to undeveloped regions like the Hudson Plains Ecozone, having lost much of its southern range in Ontario, since the mid- to late-1800s. This range reduction is thought to have occurred through a combination of habitat conversion resulting from large increases in human settlement, logging, and railway construction and over-harvest of ungulate prey and beaver (Dawson 2000; Slough 2007). By 1950, most, if not all, of the province's wolverine population was found north of the Canadian National Railway line. Thereafter, wolverine distribution continued to retract westward, with most declines reported in the Hudson Plains Ecozone area. There have been no confirmed occurrences of this species in the Moosonee area, since the mid-1950s. The decline in wolverine numbers in the far north of Ontario was generally coincident with a decline in woodland caribou (*Rangifer tarandus caribou*) in that part of the province.

In Manitoba, wolverine numbers were initially depressed by the middle of the 20th century, likely through over-harvest. Once harvests became more regulated, incidental poison baits set for wolves kept numbers down up to 1970, when the practice was halted. Following this, the species' numbers (based on harvest figures) began to increase and wolverine re-occupied its range to as far south as the Winnipeg River, by the early 1990s (Berezanski 2004). Likewise in Ontario, after the late 1970s, wolverine began to spread north and eastward – most likely in response to the recovery of northern caribou populations. This reclamation of the historical range of wolverine in the Hudson Plains Ecozone appears to be continuing to this day. For example, aerial surveys, since 2003, have recorded wolverine tracks east of its peripheral range (Figure 82); in Peawanuck, interviews with trappers and elders have produced evidence of wolverine occurrence in the Weenusk traditional use area in the last 5 years, having been absent from the area, since the 1970s (Dawson 2000). There is no evidence of a similar recovery having taken place in Québec, since the 1970s.

Currently, the modeled core occupied range for wolverine in northern Ontario based on aerial surveys conducted in 2003-2004 (Magoun et al. 2005; Ontario Wolverine Project, in prep.; Figure 82) roughly corresponds to wolverine range based on wolverine harvest records from 1980 to 2007 (Ontario Wolverine Project, unpublished data). These and subsequent aerial surveys have found evidence of wolverine occurrence further eastward than indicated by harvest reports and particularly in the northeast to Cape Henrietta Maria (Ontario Wolverine Recovery Team,

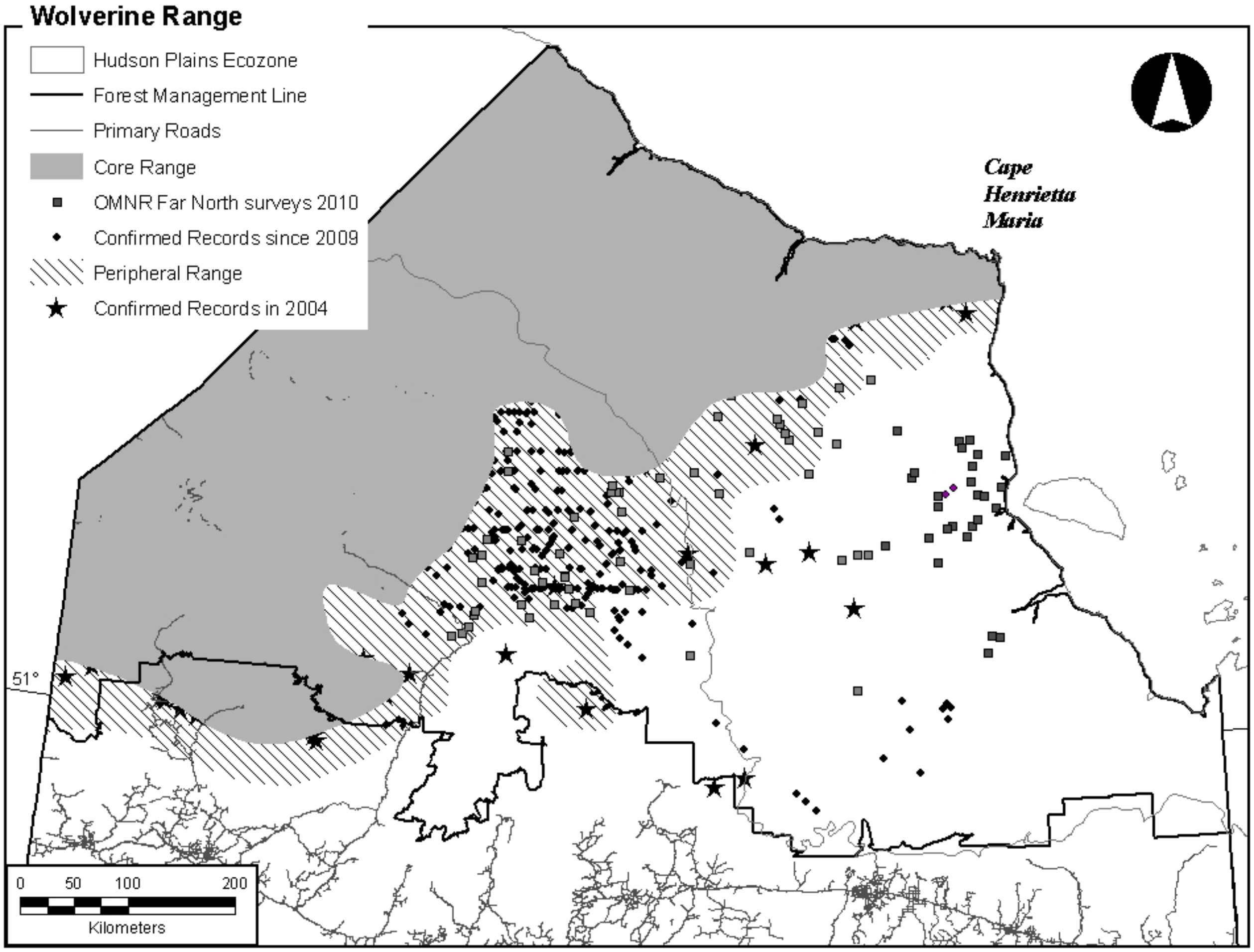

Figure 82. Core and peripheral wolverine range in northern Ontario based on mean detection probabilities derived from aerial surveys conducted in 2003-2004 with extra-limital observations from aerial surveys in 2004 and 2009-2010.
Source: Ontario Wolverine Project (unpublished data).

in review). Similar surveys have not been undertaken in Manitoba, however, harvest reports are concentrated in the north half of the province and, to a lesser degree, the east side of Lake Winnipeg (Berezanski 2004). In spite of Manitoba's smaller proportional area of the Hudson Plains Ecozone (21% vs. 68% for Ontario), Manitoba recorded a higher harvest rate of wolverines (total=129; mean±SD=6.4±3.6) than Ontario (total=82; mean±SD=4.3±2.4) during 1973-2007. This difference is also reflective of the fact that the Manitoba range was always fully occupied by wolverines, whereas the Ontario range was not. Overall, although wolverine harvest numbers have always been low in the Hudson Plains Ecozone (<20/yr; Figure 83), some evidence suggests that wolverine population numbers are increasing in the ecozone, and that the species is expanding eastward over the past two decades (Figure 82).

Wolverine populations in the Hudson Plains Ecozone are experiencing few threats or stressors at the present time, given relatively low harvest levels. However, increasing human presence and resource development activity associated with the Ring of Fire and continued mineral and gem exploration (Far North Science Advisory Panel 2010) will almost certainly increase opportunities for conflict situations with wolverines and incur increased mortalities. In southern parts of their range, roads appear to be an important driver of wolverine distribution (Bowman et al. 2010).

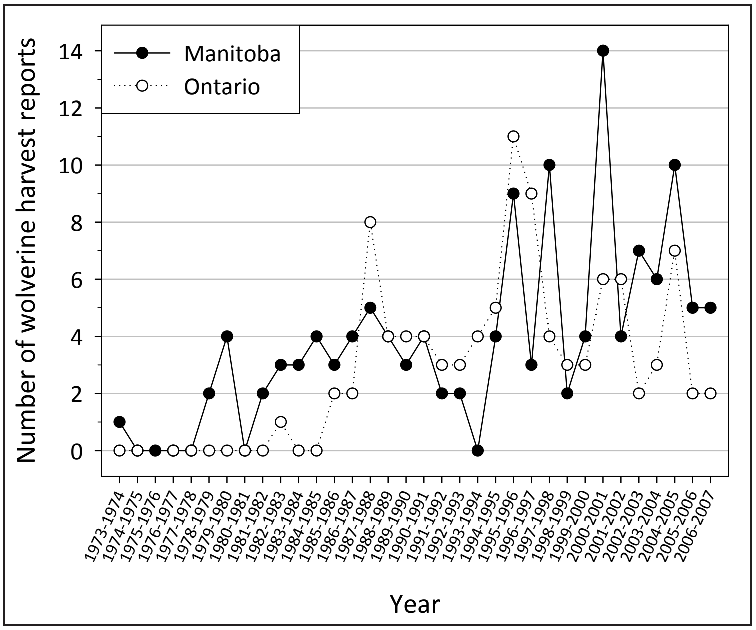

Figure 83. Wolverine harvest returns in the Manitoba and Ontario portions of the Hudson Plains Ecozone, 1973-2007 (wolverine has not been observed in Québec since 1978). Sources: Manitoba Conservation (unpublished data); Ontario Ministry of Natural Resources (unpublished data); and Québec Ministère des Ressources Naturelles et de la Faune (unpublished data).

Grey wolf

Throughout their North American range, regional wolf population densities are tied to the composition and abundance of large ungulates (Mech and Peterson 2003). While the grey wolf occurs throughout the Hudson Plains Ecozone, it is more prevalent in regions characterized by higher productivity and large prey densities. The Hudson Plains Ecozone lies beyond the range limits of white-tailed deer (*Odocoileus virginianus*); caribou are the dominant ungulate, and moose (*Alces alces*) are characterized by a more scattered distribution associated with recent fires.

Harvest data for wolves are less reliable than for other species, and no other consistent trend information is available for this species in the Hudson Plains Ecozone as a whole. However, in far northern Ontario, grey wolf densities are assumed to be low (5-6 wolves/1000 km^2) relative to southern parts of their range, with large (>1,000 km^2) territories (OMNR 2005). One survey specifically targeting wolves has been flown in the Ontario portion of the ecozone (Patterson 2009): In February 2009, two experienced wolf tracker/pilots systematically searched a 10,000 km^2 survey area for wolves or wolf tracks located approximately 40 km west of Attawapiskat. During the 3 day survey, the pilots covered 3,903 linear km spotting a total of 36 wolves in 12 independent groups. This included two single wolves, six pairs of wolves, one pack of three, one pack of five, and two packs of seven. The density was estimated at 3.1 wolves/1000 km^2. Although this estimate is higher than estimates obtained for some other areas north of the treeline in Ontario (B. Patterson, Ontario Ministry of Natural Resources, unpublished data), it is approximately an order of magnitude lower than observed in several areas along the southern edge of the wolf range in Ontario (e.g., Patterson et al. 2004), probably owing to an overall reduction in available prey biomass along a latitudinal gradient. This, coupled with the fact that most social groups of wolves observed during the present survey were pairs (as opposed to larger packs), suggests that low prey abundance is either limiting productivity or promoting first-year dispersal of wolf pups from their natal packs.

During 2003-2004 aerial surveys, approximately 5,800 km of flying in the Hudson Plains Ecozone yielded 25 grey wolf observations or tracks (Magoun et al. 2005). Pack sizes in winter are also low relative to those in southern productive habitats (Bowman et al. 2010), reflecting the low-density prey base. In the Hudson Plains Ecozone, grey wolves are often found searching for prey along river courses with occasional long-distance treks across frozen muskeg. They are also attracted to human settlements and are known to aggregate around garbage dumps in remote communities. Limited genetic sampling of wolves from Aboriginal harvesters in this ecozone (~10 samples) has demonstrated no evidence of hybridization – this is the only known region in Ontario with 100% pure grey wolves (B. Patterson, Ontario Ministry of Natural Resources, pers. comm.).

While current threats to wolves in the Hudson Plains Ecozone include mortality as a result of human conflict, because this is restricted to human settlements, it is most likely occurring at very low levels.

American marten & American beaver

American marten is another northern-adapted species that occurs through most habitats in the Hudson Plains Ecozone. Marten tracks are equally abundant in treed muskeg habitats as they are along coniferous shorelines of the many rivers in the region (Ontario Wolverine Project, unpublished data). This was not the case 60 years ago, when marten populations were extirpated from much of their Ontario range, including the Hudson Plains Ecozone (de Vos 1952). By 1950, the main strongholds for marten were in the areas around the Chapleau Crown Game Preserve and Algonquin Provincial Park (south of the Hudson Plains Ecozone) (de Vos 1952). The decline was attributed primarily to two factors: over-trapping and change in the forest due to excessive logging, forest fires, and settlement (de Vos 1952). During the 1950s, a number of live marten transfers were undertaken from the Chapleau Crown Game Preserve to areas across the far north, including portions of the Hudson Plains Ecozone. Whereas marten has been the subject of much research attention in more southern boreal and mixedwood regions, little is known of its current population status or quantitative trends or its ecology in northern taiga regions. Where this species has been studied, the key to its preferred habitat is complex physical structure, including available downed wood to provide thermal cover and access to subnivean spaces (Payer and Harrison 2003). This species has a well-documented susceptibility to habitat fragmentation through loss of interior forest conditions and/or habitat area, and provincially it is a Species of Concern in managed boreal forests of Ontario (Watt et al. 1996). Martens demonstrate a preference for older coniferous forests, and they do not tend to establish home ranges in areas with >25-40% early-successional forest (Potvin et al. 2000). Marten can be easily over-harvested, when such activities are being undertaken in concert with habitat disturbance (Strickland 1994). However, little human disturbance occurs in the Hudson Plains Ecozone, and trapping activities there tend to occur in pockets.

American beaver is widespread across the North American continent from just above the treeline to the Mexican desert, and it resides in a wide variety of habitats and climatic conditions. Although beaver has by now re-occupied much of its historical range following near-extirpation from the continent during the 17th and 18th centuries, its present numbers in some areas are thought to be a fraction of those prior to the commercial fur trade (Novak 1987). A large semi-aquatic rodent, beaver is known to have a unique role in creating and maintaining wetlands at landscape scales (Naiman et al. 1988). Beaver depends primarily on deciduous trees for food and necessary building supplies. Wetlands created by beaver enhance biodiversity by providing important habitat for multiple other elements of biodiversity. Although more is known about beaver ecology outside boreal forest habitats, beaver in the Hudson Plains Ecozone was the subject of some survey efforts in the early 1980s (Gauthier and Threader 1981; Cree 1983; Phoenix and

Haddow 1984). Beaver is thought to be generally more abundant in inland waters than along the coast, with the caveat that those dwelling in lakes and ponds are easier to count than river beaver.

By the early 1800s, when fur trading reached its peak, the impact on population numbers of the most commonly exploited furbearer species, including marten and beaver, was evident (Lytwyn 2002). Reports from fur trader posts in the Hudson Plains Ecozone indicated precipitous declines in beaver returns, which were exacerbated by unusual weather patterns (e.g., mild winters and flooding, drought-fueled fires), and marten were also adversely affected. Since the 1820s, the ecozone has been subject to periods of exploitation from intense competition during the fur trade, peaking in the early 19th century (Lytwyn 2002), and the infiltration of labourers onto traplines during hydroelectric and railway developments in the Moose River Basin in the early 1900s (George et al. 1995; Byers 1996). The ecozone has been in a state of recovery from overexploitation of its resident fauna ever since, aided by restocking of marten and beaver in the early 1950s (Byers 1996). Although harvesting has certainly continued since that time, it is not so prevalent across the entire landscape and not anywhere near the intensity. A further decline of trapping activities occurred with the fall of pelt prices after the 1980s. With uneven trapping effort, it is impossible to discern biological patterns from fur data (Figure 84).

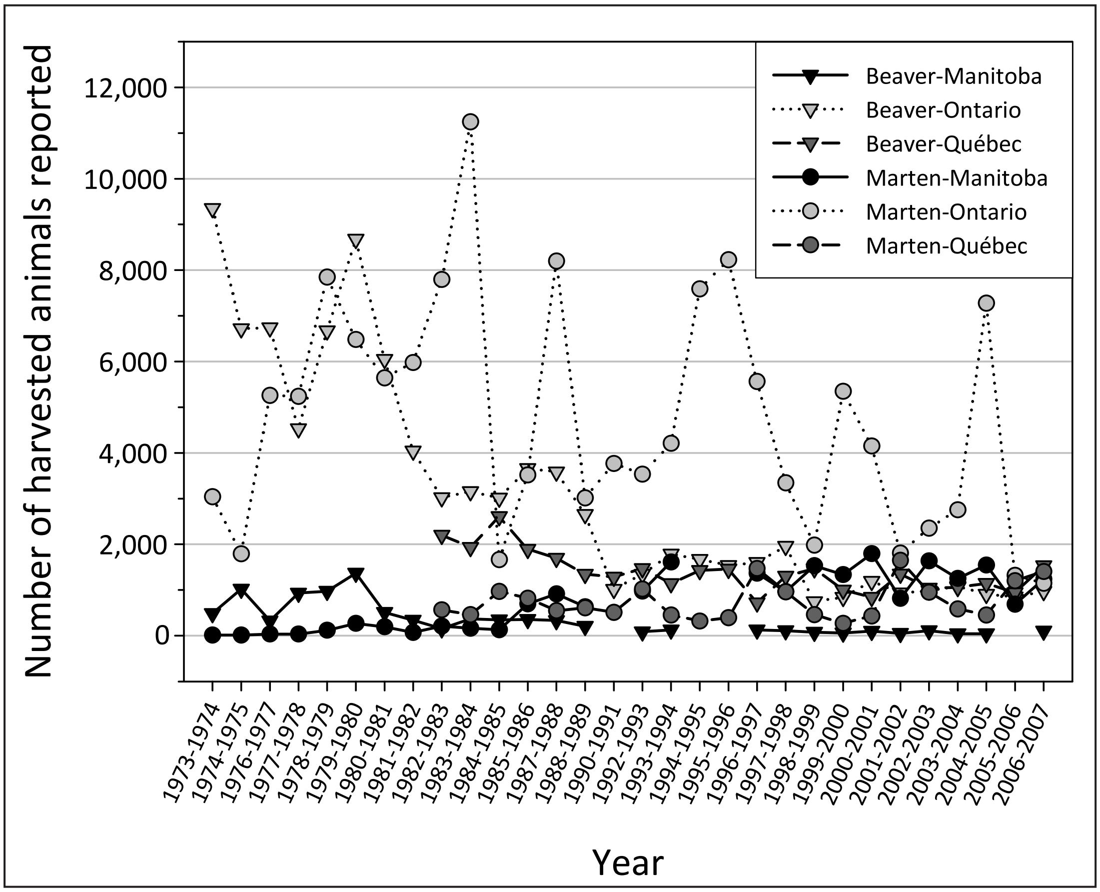

Figure 84. Fur returns of American marten and American beaver, the two most commonly harvested furbearer species, in the Manitoba, Ontario, and Québec portions of the Hudson Plains Ecozone, 1973-2007. Data are not available for all years.
Sources: Manitoba Conservation (unpublished data); Ontario Ministry of Natural Resources (unpublished data); and Ministère des Ressources Naturelles et de la Faune (unpublished data).

Future concerns

Changes in climatic conditions have already resulted in documented range shifts for several species in Ontario's north and are expected to do so under future climate warming (Thompson 2000). If snow depth and persistence is affected by climate change as expected, this could have implications for several species that are restricted to northern environments, such as wolverine (Copeland et al. 2010). Likewise, climate warming could promote the northward shift of favourable conditions for coyote, fishers, white-tailed deer, and even raccoon, with consequences for species interactions. As noted above, escalating plans for new development in the Hudson Plains Ecozone, including planned infrastructure associated with the Ring of Fire mineral exploration area, could result in increased harvesting rates and other mortality of species discussed here.

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2.3.3.3 **Birds**

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The Hudson Plains Ecozone supports a large (over 340 species) and diverse assemblage of birds, most of them migratory species in four basic groups: landbirds, waterfowl, shorebirds, and waterbirds (including seabirds) (MARC 2003; Abraham and Keddy 2005; Rockwell et al. 2009; Zeran et al. 2009; Ontario Partners in Flight 2010). No intensive inventory of the whole ecozone has taken place, mainly due to the difficulty in accessing much of the region. However, comprehensive studies of local avifaunas exist, particularly from the Churchill, Manitoba area (McLaren and McLaren 1981; Jehl 2004; Rockwell et al. 2009), where many of the vagrant species have been recorded. Breeding bird atlas work in Ontario has contributed much information on species abundance and distribution in that province's portion of the ecozone (Cadman et al. 1987, 2007). Similar breeding bird atlas projects are currently in progress in both Manitoba (Artuso et al. 2010; Bird Studies Canada 2010) and Québec (Atlas of the Breeding Birds of Québec 2010).

The diversity of breeding bird species in the Hudson Plains Ecozone is lower than it is in most other ecozones in North America. A total of 184 bird species have been recorded breeding in the ecozone, most of them regularly, and another 21 species are suspected to breed there or have bred there once (Appendix 1). Diversity increases towards the south; about 50 breeding species do so almost exclusively in the James Bay Lowlands Ecoregion (Cadman et al. 2007). Within the ecozone, higher diversity is found along river courses and in the wooded areas on eskers and other higher-elevation areas. An additional 25 or more species occur as regular migrants or winter visitors. The list of vagrants reported in the zone is over 85. Alien species include rock pigeon (*Columbia livia*), European starling (*Sturnus vulgaris*), and house sparrow (*Passer domesticus*) (Cadman et al. 2007; Rockwell et al. 2009; Appendix 1), all found around villages in small numbers. One formerly common species (Eskimo curlew, *Numenius borealis*) is assessed as Endangered by COSEWIC (2009), but it is now thought to be extinct by some (Jehl 2004). The passenger pigeon (*Ectopistes migratorius*), which also once occurred in this geography, is extinct.

The predominance of bird species that migrate long distances through greatly altered ecosystems on their way to wintering areas means that assessment of causes of trends is difficult; events on mid- and southern-latitude portions of the ranges of these species have high potential to affect population status. Climate change remains an emerging threat. Northward shifts in species breeding distributions are not yet apparent in this geography (Cadman et al. 2007), but some changes in goose phenology (advancing nesting dates) are evident in the ecozone (Rockwell and Gormezano 2009; Ontario Ministry of Natural Resources, unpublished data; see also Section 2.4.3.4.1, *Animal Phenology*). The status and trends of the area's landbird, waterfowl, shorebird, and waterbird populations in mainland areas are considered below. The status and trends of seabirds in the pelagic portion of the geographic area are addressed by the complementary Arctic Marine Ecozones report of the ESTR (Niemi et al. 2010).

The Hudson Plains Ecozone is part of the Taiga Shield and Hudson Plains Bird Conservation Region (BCR 7) designated by the North American Bird Conservation Initiative (NABCI). An Ontario landbird conservation plan for BCR 7 covers the Hudson Plains Ecozone or the provincial Hudson Bay Lowlands Ecozone (Ontario Partners in Flight 2010). This plan contains monitoring and trend data and assessment for this group. Conservation plans for other bird groups in BCR 7 have also been created or are in preparation at regional scales, and some of these are drawn upon for this summary (Gratto-Trevor et al. 2001; Ross et al. 2003; Aubry and Cotter 2007; Zeran et al. 2009; Eastern Habitat Joint Venture 2010).

Bird trend monitoring activity in the Hudson Plains Ecozone is a mixture of programs undertaken by government agencies, non-government conservation organizations, and academic institutions. There are no ecozone-wide monitoring programs in this ecozone. Most North American avian monitoring programs have poor coverage in remote areas, such as the Hudson Plains Ecozone. Of the four bird groups (waterfowl, waterbirds, shorebirds, and landbirds), the waterfowl group is the best monitored due to a history of federal and provincial agency and waterfowl flyway management of migratory game birds and some long-term academic studies. These programs include annual aerial inventory of Canada goose (*Branta canadensis*), periodic aerial inventory for ducks (for 19 regularly occurring species), periodic aerial inventory of lesser snow goose (*Chen caerulescens caerulescens*), and various ground-based programs in a small portion of the ecozone. Shorebird abundance and distribution have been documented periodically in parts of the ecozone but using different methods, including site-specific breeding bird surveys by academic institutions and more extensive aerial surveys by federal agencies and volunteer-based collaborative atlases. Landbird distribution has been documented by a variety of means but most comprehensively with the bird atlases in Ontario (Cadman et al. 1987, 2007), Québec (Atlas of the Breeding Birds of Québec 2010), and Manitoba (Bird Studies Canada 2010) and other compilations in Manitoba (MARC 2003). Limited long-term comparison using atlases is available only in Ontario, but it is in progress for Québec. Waterbirds are probably the least well monitored bird group in the Hudson Plains Ecozone, again relying primarily on the bird atlases for information but including some site-specific surveys and some casual aerial inventories. In Ontario, the Natural Heritage Information Centre of the Ministry of Natural Resources tracks the status of species at risk (those assessed by COSEWIC, Committee on the Status of Endangered Wildlife in Canada and COSSARO, Committee on the Status of Species at Risk in Ontario), while the Manitoba Conservation Data Centre does that in Manitoba and the Centre de données sur le patrimoine naturel du Québec does it in Québec.

2.3.3.3.1 **Landbirds**

A landbird conservation plan for the Taiga Shield and Hudson Plains Bird Conservation Region (BCR 7) has been prepared in Ontario (Ontario Partners in Flight 2010), but no comparable plans have been prepared for the Manitoba or Québec portions of the BCR. The Ontario plan lists 124 species that regularly breed or winter in the ecozone. Of these, 24 species (~20%) have been identified as priority conservation species, and 13 additional species have been identified as near-priority species. The Partners in Flight assessment process categorizes species based on their vulnerability, level of concern, and area importance, using data on six factors: distribution, population size, population trend, threats, relative density, and percent of species population. The species are also categorized as Concern or Stewardship species at the continental and regional scales, and species at risk nationally or provincially, and some may be priority species for regional management interest. Of the 24 priority species listed in BCR 7, most (17) are migratory and thus are affected by anthropogenic factors outside the ecozone. The remaining species are year-round residents (5) or partial migrants (2), whose priority is based on high stewardship responsibility, because of the high proportion of the breeding range within the

BCR. They are relatively evenly distributed among habitat guilds, except tundra: forest (8), shrub/successional (8), wetland/riparian (11), and tundra (3). Most of the 24 priority species are species of high stewardship responsibility at the national (12) or regional (4) scale. Eight are also species of concern at the continental level due to threats. Five are assessed as species at risk by COSEWIC (2010): olive-sided flycatcher (*Contopus cooperi*) (Threatened), Canada warbler (*Wilsonia canadensis*) (Threatened), common nighthawk (*Chordeiles minor*) (Threatened), rusty blackbird (*Euphagus carolinus*) (Special Concern), and short-eared owl (*Asio flammeus*) (Special Concern). Another two species are classed as species at risk by the Committee on the Status of Species at Risk in Ontario (COSSARO 2010): golden eagle (*Aquila chrysaetos*) (Endangered) and bald eagle (*Haliaeetus leucocephalus*) (Special Concern). In Québec, golden eagle, bald eagle, and peregrine falcon (*Falco peregrinus anatum*) are species at risk (QMRNF 2010). In Manitoba, the only landbird regularly occurring in the ecozone that is listed is the peregrine falcon (Endangered) (Government of Manitoba 2010). The bald eagle has increased in the ecozone, since the 1980s, both as a breeding species in the southern portion and as non-breeding birds along the coasts during summer (MARC 2003; Cadman et al. 2007). Aboriginal peoples report the same trend in community meetings. The increase is consistent with recovering populations across southern Canada due to reduced use of pesticides, since the 1970s. Local abundance in the ecozone may also be linked to increased goose populations (see below), which are a ready source of prey for young eagles.

2.3.3.3.2 **Waterfowl**

Waterfowl is the best monitored bird group, although not all such monitoring occurs within the ecozone. Most waterfowl species are hunted in the ecozone or outside, making it both necessary and important to understand their status for sustainability of population, subsistence uses, or economic activity related to hunting and viewing (waterfowl harvest in the ecozone is discussed in Section 2.5.3, *Provisioning Services*). Some species have annual aerial surveys that lead to extrapolated estimates of total breeding population, while most have some form of winter estimate or population index. Waterfowl surveys in the Hudson Plains Ecozone include the periodic United States Fish and Wildlife Service (USFWS)–Canadian Wildlife Service (CWS) Waterfowl Breeding Population and Habitat Survey (WBPHS) (CWSWC 2010; USFWS 2010), which provides long-term data on duck species in North America. Although the majority of the ecozone is now surveyed annually (Strata 57, 58, and 59 in Ontario and small parts of strata 68 and 69 in Québec), none of the Manitoba portion is covered. Additionally, these strata in this ecozone are part of the non-traditional survey area, and the Ontario-Québec portions have been included gradually only since 1990, so that long-term trends are unavailable. Finally, the data are not routinely analyzed on an ecozone basis, making discussion of ecozone trends general in nature.

In general, most species of waterfowl breeding in the Hudson Plains Ecozone have stable or increasing populations. Some increases are large (e.g., geese). Breeding geese are intensively surveyed within the ecozone on a population management unit basis; these results are reported annually in federal status reports and in flyway management plans. Canada geese of the Eastern Prairie Population (EPP) (Manitoba portion of ecozone), the Mississippi Valley Population (MVP) (Ontario portion of the ecozone south to Attawapiskat), and the Southern James Bay Populations (SJBP) (Ontario portion of the ecozone south of Attawapiskat and Nunavut portions), and Atlantic Population (AP) (Québec portion of the ecozone) are all annually monitored (EPPCGC 2006; Abraham et al. 2008; Brook and Luukkonen 2010). The EPP and MVP stocks have increased over the past four decades, but they have been stable (EPP) or declining (MVP) in recent years, while the SJBP and the AP declined from the 1970s to 1990s but have remained stable since then (CWSWC 2010; LePage and Bordage in prep.). The EPP, MVP, and SJBP have been affected locally in terms of reproductive success or nesting density by growth of the lesser snow goose population.

Lesser snow goose (Figure 85) nests in discrete colonies, of which there are six in the ecozone. These colonies are monitored by periodic photographic counts (e.g., Kerbes et al. 2006) or ground and/or aerial surveys (e.g., Abraham et al. 1998). Only three of the colonies existed in the 1970s: La Pérouse Bay (LPB), Cape Henrietta Maria (CHM), and Akimiski Island (AI) (Kerbes 1975; Abraham et al. 1999). However, with the quadrupling of the Mid-Continent population of lesser snow goose to which these colonies belong, there was expansion of those (LPB from 8,800 birds in 1979-1980 to 58,700 birds in 1997; CHM from 109,000 birds in 1979 to 320,000 birds in 1997; AI from 867 birds in 1973 to 3,450 birds in 1997), plus establishment of three more colonies (Knife-Seal River, West Pen Island, and Shell Brook), in the past four decades (Abraham et al. 1998, 1999; Kerbes et al. 2006). Increased foraging by this rapidly growing population of lesser snow goose has led to much damage to the ecozone's coastal marshes over the same period (Jefferies et al. 2006; and see Section 2.2.2.1, *Coastal*). The increased size of the Mid-Continent population is attributable to a greater supply of agricultural food on wintering grounds (mostly in the southern United States) and along migration routes, declining harvest rate, and the development of migration and winter refuges (Abraham and Jefferies 1997; Jefferies et al. 2003).


Figure 85. A brood flock in the lesser snow goose colony at West Pen Island, Nunavut, off the Hudson Bay coast near the Ontario-Manitoba border.
Photo credit: K.F. Abraham, Ontario Ministry of Natural Resources.

Ross's goose (*Chen rossii*) occurs, but it is minor as a breeding species, although it has increased since the 1970s (Kerbes et al. 2006; Abraham 2007c). Atlantic brant (*Branta bernicla hrota*) does not nest in the ecozone, but the entire population stages there and relies on the eelgrass (*Zostera marina*) beds and salt marshes of the coastal zone of James Bay during spring and fall migrations (Ward et al. 2005). Brant are known to follow the large rivers flowing into the south end of James Bay (K.F. Abraham, Ontario Ministry of Natural Resources, unpublished data). Their population is indexed in winter, and it appears to be broadly stable over the last decade (USFWS 2010); within the ecozone, there is uncertainty about the status of eelgrass (a primary forage species for brant) and the implications for the distribution of staging birds. Eelgrass has declined precipitously in the Québec coastal region (see eelgrass discussion in Section 2.3.3.7.3). Little is known about eelgrass in the Manitoba or Ontario portions of the ecozone; however, an increased use by brant of salt marshes on the western James Bay coast is suggested (Parliament of Canada 2008).

Species or groups of ducks that occur in the ecozone, and for which there is continental concern about declining populations, include greater scaup (*Aythya marila*), lesser scaup (*Aythya affinis*), northern pintail (*Anas acuta*), and sea ducks (especially eiders and scoters). Neither scaup species showed evidence of decline in the Ontario portion of the ecozone (Cadman et al. 2007) or the eastern strata of the Waterfowl Breeding Population and Habitat Survey (USFWS 2010). Northern pintail is widespread in the ecozone, and it is mostly associated with eastern North American migration and winter areas (Malecki et al. 2006). Atlas data from Ontario suggest little change or a moderate decline in northern pintail, since the 1980s (Gendron 2007). Data on scoters from breeding areas are too sparse to allow trend analysis; like brant, these species migrate along rivers into southern James Bay. Baseline counts of moulting male black scoters (*Melanitta nigra*) of the Atlantic subpopulation have been established for the western James Bay area, where they gather from throughout eastern North America; comparable surveys were undertaken in 1983, 1991 (Ross 1994), 2006, and 2009 (R.K. Ross, Canadian Wildlife Service and K.F. Abraham, Ontario Ministry of Natural Resources, unpublished data). There is evidence of broadly stable numbers between 1977 and 2009 during the late July to early August period, which is necessary to standardize annual estimates (Ross 1994). Common eider (*Somateria mollissima*) appears to be stable in the ecozone (Abraham 2007a; Rockwell et al. 2009), although king eider (*Somateria spectabilis*) is a rare breeding species in the ecozone, and it appears to have declined in the southern portion (Jehl 2004; Sutherland 2007).

2.3.3.3.3 **Shorebirds**

The Hudson Plains Ecozone supports a number of breeding species of shorebirds. Very little information, however, is available on population trends. Shorebirds have been studied extensively at Churchill, and nearly all studies have reported widespread declines in shorebirds (Jehl and Lin 2001; Jehl 2004). Declines were particularly notable in the semipalmated sandpiper (*Calidris pusilla*) (Figure 86), which used to be the most abundant breeding shorebird in the Churchill region up to the 1940s, but by 2004 it could no longer be found breeding in that area (Allen 1945; Gratto-Trevor 1994; Jehl 2007). At Cape Henrietta Maria at the north end of James Bay, the species was abundant in the 1970s, but it had become scarce by 2004-2005 (Peck and James 1983; Cadman et al. 1987; Jehl 2007; M. Peck in Jehl 2007). These results appear to be consistent with the declines reported for semipalmated sandpipers on migration in many other regions (e.g., Morrison et al. 1994, 2001; Bart et al. 2007) and other work summarized by Jehl (2007). However, Peck (2007b) reported that no trend of increase or decrease in the Ontario population could be detected using atlas methods between

Figure 86. Flock of semipalmated sandpipers and white-rumped sandpipers at North Point, Ontario, on the west coast of James Bay. Photo credit: J. Iron, Ontario.

two periods (1981-1985 vs. 2001-2005). Other somewhat anomalous results were reported by Sammler et al. (2008) at a study area 60 km east of Churchill, where results of line transect surveys indicated an increase in semipalmated sandpipers between 1984 and 1999, though many other larger ground-nesting species declined. While the precise reason(s) for the decline in semipalmated sandpipers remains unclear, it did not appear to be linked to the extensive damage to coastal habitats caused by the increasing population of lesser snow goose (Jehl 2007, but see Jehl 2004 and Rockwell et al. 2009 for an alternative scenario in a snow goose colony where the decline might be locally augmented). Overall, the widespread decline is more likely to be related to conditions outside of the breeding grounds (Jehl 2007).

The coastlines of Hudson and James bays are also extremely important as migration corridors for many shorebirds breeding in the eastern and central Canadian arctic en route to and from their nesting grounds (Morrison and Harrington 1979; Gratto-Trevor et al. 2001). The ecozone is of particular importance to breeding whimbrel (*Numenius phaeopus*) and Hudsonian godwit (*Limosa haemastica*) (Figure 87), and thus there is high national or hemispheric stewardship responsibility. Abundance of breeding whimbrels in the Churchill region of the ecozone is thought to have declined (Jehl and Lin 2001; Rockwell et al. 2009). Many Hudsonian godwits are thought to fly directly from the James Bay area to stopover areas in South America (Morrison 1984), and James Bay is also a key migration area for red knot (*Calidris canutus*) (Figure 87) (COSEWIC 2007; Niles et al. 2010), which is a species assessed as Endangered by COSEWIC (2010). No trend information is available for shorebird migrants passing through the area. Marbled godwits (*Limosa fedoa*) from the James Bay coast and islands are a small disjunct population with uncertain trends; recent research has shown that they make the longest distance migration of any North American population, a cross-continental migration to the Gulf of California in Sonora, Mexico, which makes the coastal habitats of the ecozone critical to their preparation for migration in fall and to their recovery of condition for breeding in spring (B. Olson, United States Fish and Wildlife Service, pers. comm.; Abraham 2007b).

Figure 87. Hudsonian godwits (right) and red knots (left) at Longridge Point, Ontario, on the west coast of James Bay. Photo credit: J. Iron, Ontario.

2.3.3.3.4 **Waterbirds**

Waterbirds are a mixed group, including loons, grebes, gulls, terns, jaegers, herons, pelicans and cormorants, rails, and cranes. Waterbird monitoring is limited and consequently so is trend information. Three species of loons (common loon, *Gavia immer*; Pacific loon, *Gavia pacifica*; and red-throated loon, *Gavia stellata*) are widespread in the ecozone and conspicuous due to calls and flight behaviour. As such, their status is relatively easier to assess than that of some secretive species, such as yellow rail (*Coturnicops noveboracensis*). Common loon experienced a modest increase between atlas periods in Ontario, a result that is consistent with regional and national trends (Jones and Timmermans 2007). The other two loon species were stable between atlas periods in Ontario (Abraham and Sutherland 2007; Peck and Sutherland 2007), which is consistent with limited information from Manitoba (Koes 2003a,b). Double-crested cormorant (*Phalacrocorax auritus*) is rare in the ecozone, but a breeding colony was recently established in Akimiski Strait, western James Bay (Weseloh 2007); this increase is in keeping with the exponential growth of the species in the Great Lakes and eastern North America, since the 1970s (Hatch and Weseloh 1999). American white pelican (*Pelecanus erythrorhynchos*) has also been increasingly reported in the ecozone in the last decade (Artuso et al. 2010) and has recently established breeding in Akimiski Strait (although the reasons are unclear), illustrating the easterly expansion of this species that is designated by COSSARO as Threatened (Peck 2007a). Colonial gulls and terns are conspicuous, but little effort has been made to document their abundance. In the Ontario portion, most such species are stable or increasing (herring gull, *Larus argentatus*; ring-billed gull, *Larus delawarensis*; Bonaparte's gull, *Larus philadelphia*; Caspian tern, *Sterna caspia*; common tern, *Sterna hirundo*; and arctic tern, *Sterna paradisaea*), while a few have declined (black tern, *Chlidonias niger*; parasitic jaeger, *Stercorarius parasiticus*) (Cadman et al. 2007). Ross's gull (*Rhodostethia rosea*, assessed as Threatened by COSEWIC) has declined in the Manitoba portion (MARC 2003). Yellow rail (assessed as Special Concern by COSEWIC), for which this ecozone is thought to contain about 90% of the North American breeding range, may have declined in the Ontario portion (Tozer 2007) and locally in the Manitoba portion (Rockwell et al. 2009); in both provinces, habitat degradation by lesser snow goose could be affecting local nesting densities via removal of the plant species that provide nesting cover in freshwater marshes. Sandhill crane (*Grus canadensis*) is thought to have increased in the ecozone (Pedlar and Ross 1997; Sutherland and Crins 2007). This species migrates into the Mid-Continent region, and it has likely benefited from agricultural practices. Overall, as a group, about two-thirds of the regularly occurring waterbird species are stable or increasing (Rockwell et al. 2009; Zeran et al. 2009).

The absence of a systematic, well-supported monitoring program that covers all birds groups in this ecozone is a significant impediment to status evaluation and trend monitoring. The recent emphasis on atlas projects in all jurisdictions, which incorporate quantitative abundance methods, as well as occurrence and breeding status assessment methods, will alleviate some, but not all, of that deficiency. Additional monitoring of migrants would be of great value, because of the importance of the coasts of this ecozone as areas of significant spring and particularly fall concentration of migrants accumulating body reserves for breeding (spring) or migration to distant staging and wintering areas (fall).

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2.3.3.4 **Fish**

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Knowledge of the status (i.e., distribution, abundance, condition) of fish species and trends in their populations is poor for the Hudson Plains Ecozone compared to many other ecozones in Canada, largely due to the expanse and remoteness of the area and the low level of large-scale industrial development there. Fish monitoring and assessment efforts have been limited and inconsistent over time throughout most of the ecozone. For example, fish species information is currently available for only about 6% of the water area in the majority of the ecozone that lies in Ontario and even that information is dated (OMNR 1985). Ontario's information is expected to improve, however, given that this province recently implemented a broad-scale fisheries monitoring program, which includes the Hudson Plains Ecozone (the monitoring program is described in Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Considerably more effort has been spent examining fish species in northern Manitoba and Québec, where large-scale hydroelectric developments have been constructed and additional hydroelectric developments are either in progress or proposed (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). In Manitoba, fisheries investigations and numerous long-term monitoring programs have been conducted along the Nelson River, since the early 1980s, and along the Churchill River, since the mid-1990s. In Québec, fish populations in the area of the ecozone affected by the La Grande hydroelectric complex (both natural and modified environments, including the Opinaca reservoir and the lower Eastmain and Opinaca rivers, from which flows were diverted) were monitored for their relative abundance and some biological characteristics periodically from 1978-1979 (just prior to development), until 2000, using permanent experimental fishing stations (Therrien et al. 2004). Mercury monitoring in these fish populations has been further extended, given that mercury levels are projected to increase again in the Opinaca reservoir, due to the receipt of mercury exported from the recently impounded Eastmain-1 reservoir upstream, just outside of ecozone boundaries (Therrien and Schetagne 2008; see also Figure 67 in Section 2.2.2.4.2, *Rivers/Streams & Lakes*). This latter impoundment is part of the Rupert River diversion project, which began reducing the flow of the Rupert River within the Hudson Plains Ecozone in 2009 (~72% of the mean annual flow was diverted, but lateral flow from tributaries increases the flow at the river mouth to ~48%). Some monitoring of fish populations is now also being carried out in the reduced flow segment of this river, albeit mostly upstream of ecozone boundaries (Environnement Illimité 2010a,b). Finally, some additional information on the distribution of fish species in the Québec portion of the ecozone has also been compiled, since 1973 (Verdon 2001).

Portions of the Hudson Plains Ecozone support among the highest diversity of freshwater fish species in Canada, albeit the ecozone's fish assemblage is comprised of species that are mostly common overall (Chu et al. 2003; Browne 2007; Abell et al. 2008). At least 52 freshwater fish species are known to occur in the ecozone (Table 18). Northern pike (*Esox lucius*), walleye (*Sander vitreus*), lake whitefish (*Coregonus clupeaformis*), cisco (*Coregonus artedi*), and sucker (*Catostomus* and *Moxostoma species*) are widespread in larger lakes and rivers. Lake sturgeon (*Acipenser fulvescens*) is found in the largest rivers and some lakes. Most coastal streams and rivers contain sea run (migratory or anadromous) brook trout (*Salvelinus fontinalis*), lake whitefish, and cisco

– forms of these species that are unique in Ontario. Lake trout (*Salvelinus namaycush*) is known to occur in only four lakes in the Ontario portion of the ecozone (OMNR 1985) and to occur naturally in low abundance or with "very marginal presence" in the eastern, Québec portion of the ecozone (Therrien et al. 2004; see also Verdon 2001 and Schetagne et al. 2003). In the Manitoba portion of the ecozone, lake trout is not known to occur, nor is it considered likely to occur, in any of the typically shallow lakes found there (D. Macdonald, Manitoba Water Stewardship, pers. comm.). Arctic grayling (*Thymallus arcticus*) is found at the mouth of the Churchill River, and sea run arctic char (*Salvelinus alpinus*) is also found in the estuarine areas of coastal tributaries in the Manitoba portion of the ecozone, typically outside the range of brook trout. Elsewhere in the ecozone, arctic char might conduct rare visitations, having been reported in the Severn and Winisk rivers in the early 1960s (Groot and Margolis 1991; Mandrak and Crossman 1992) but not in the Québec portion of the ecozone in more recent surveys (1978-2000) (Verdon 2001; Therrien et al. 2004)[47]. On the other hand, some marine species (sculpin, *Myoxocephalus* sp.; Greenland cod, *Gadus ogac*; sand lance, *Ammodytes spp.*) now inhabit at least one of the ecozone's estuaries after the intrusion of saltwater into the Eastmain River increased following hydroelectric diversion of the majority of that river's flow (Thierren et al. 2004 and see later). Pink salmon (*Oncorhynchus gorbuscha*) and chum salmon (*Oncorhynchus keta*) were introduced as eggs and fry in the Winisk, Attawapiskat, and Goose rivers in 1955-1956, but no adult individuals have ever been captured (Ryder et al. 1964).

Table 18. Freshwater fish species of the Hudson Plains Ecozone. Marine fish species that are incidental to the ecozone's estuaries are not listed (for information on marine fish species, see the Arctic Marine Ecozones technical report of the ESTR; Niemi et al. 2010).
Sources: Ryder et al. (1964); Mandrak and Crossman (1992); Seyler (1997); Verdon (2001); Therrien et al. (2004); and Ontario Ministry of Natural Resources (unpublished Far North digital library database). Also, for smallmouth bass only: C. Chenier (Ontario Ministry of Natural Resources, pers. comm.) and S. McGovern (Ontario Ministry of Natural Resources, pers. obs.).

Species		Species (con't)	
Scientific name	Common name	Scientific name	Common name
Acipenseridae		Gasterosteidae	
Acipenser fulvescens	Lake sturgeon	*Culaea inconstans*	Brook stickleback
Catostomidae		*Gasterosteus aculeatus*	Threespine stickleback
Catostomus catostomus	Longnose sucker	*Pungitius pungitius*	Ninespine stickleback
Catostomus commersoni	White sucker	Hiodontidae	
Moxostoma macrolepidotum	Shorthead redhorse	*Hiodon alosoides*	Goldeye
Moxostoma anisurum	Silver redhorse	*Hiodon tergisus*	Mooneye
Centrarchidae		Osmeridae	
Micropterus dolomieu[c]	Smallmouth bass[c]	*Osmerus mordax*[c]	Rainbow smelt[c]

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[47] Instead, the La Grande River, just north of the Hudson Plains Ecozone, was identified as the most southerly distribution of this species.

Table 18, Cont.

Species		Species (con't)	
Scientific name	Common name	Scientific name	Common name
Cottidae		Percidae	
Cottus bairdi	Mottled sculpin	*Perca flavescens*	Yellow perch
Cottus cognatus	Slimy sculpin	*Stizostedion canadense*	Sauger
Cottus ricei	Spoonhead sculpin	*Sander vitreus*	Walleye
Myoxocephalus quadricornis	Fourhorn sculpin	*Etheostoma exile*	Iowa darter
Cyprinidae		*Etheostoma nigrum*	Johnny darter
Carpiodes cyprinus	Quillback	*Percina caprodes*	Logperch
Phoxinus/ Chrosomus eos	Northern redbelly dace	Percopsidae	
Cyprinus carpio[c]	Common carp[c]	*Percopsis omiscomaycus*	Trout-perch
Phoxinus/Chrosomus neogaeus	Finescale dace	Petromyzontidae	
Couesius plumbeus	Lake chub	*Ichthyomyzon unicuspis*	Silver lamprey
Notemigonus crysoleucas	Golden shiner	Salmonidae	
Luxilus cornutus	Common shiner	*Coregonus clupeaformis*	Lake whitefish
Notropis atherinoides	Emerald shiner	*Coregonus artedi*	Cisco
Notropis heterolepis	Blacknose shiner	*Prosopium cylindraceum*	Round whitefish
Notropis hudsonius	Spottail shiner	*C. artedii X C. clupeaformis*	Cross whitefish/cisco
Pimephales promelas	Fathead minnow	*Thymallus arcticus*	Arctic grayling
Rhinichthys cataractae	Longnose dace	*Oncorhynchus gorbuscha*[b]	Pink salmon[b]
Semotilus atromaculatus	Creek chub	*Oncorhynchus keta*[b]	Chum salmon[b]
Semotilus corporalis	Fallfish	*Salvelinus alpinus*[a]	Arctic char[a]
Margariscus margarita	Pearl dace	*Salvelinus fontinalis*	Brook trout
Esocidae		*Salvelinus namaycush*	Lake trout
Esox lucius	Northern pike	Sciaenidae	
Gadidae		*Aplodinotus grunniens*	Freshwater drum
Lota lota	Burbot		

[a] Species might conduct rare visitations.

[b] Introduced species but no confirmed cases of captured adults, since the introduction.

[c] Introduced species known to be present (potential to be invasive).

Introduced and potentially invasive species that are present and might cause problems for native species include: common carp, extending northwards through the Nelson River system in Manitoba; rainbow smelt, found in the Nelson River drainage and its estuary; and smallmouth bass, recently found for the first time in both the Moose River (2008) (C. Chenier, Ontario Ministry of Natural Resources, pers. comm.) and the Albany River (2009) (S. McGovern, Ontario Ministry of Natural Resources, pers. obs.). These introduced species were discussed in Section 2.2.2.4.2, *Rivers/Streams & Lakes.* Smallmouth bass is considered further in Section 2.3.3.4.3, *Brook Trout*.

Among the native fish species known to occur in the ecozone, lake sturgeon is the only species considered to be of national conservation concern (Section 2.3.2, *Trends in Species of National Conservation Concern*; and see also the *Lake Sturgeon* heading below). Nationally, the number of native freshwater fish species that are of conservation concern continues to rise, with habitat loss, overexploitation, and competition from invasive non-native (alien) species as major contributing factors (Monk et al. in press). In contrast, the Hudson Plains Ecozone is an area of Canada where freshwater fishes are still exposed to few human influences (Chu et al. 2003; Browne 2007).

Lake sturgeon, lake trout, brook trout, lake whitefish, and cisco are profiled below as fish species that are of special interest to the ecozone.

2.3.3.4.1 **Lake sturgeon**

Lake sturgeon (Figure 88) is one of the largest freshwater fish species in Canada (Scott and Crossman 1973). It is unique to northeastern North America, reaching the northern limit of its range near the northernmost extent of the Hudson Plains Ecozone in Manitoba (Churchill and Seal rivers) and Ontario (Severn and Winisk rivers), and just north of the ecozone in Québec (Scott and Crossman 1973; Ferguson and Duckworth 1997; COSEWIC 2006). This species is the only representative of the sturgeon family that occurs in the ecozone.

Figure 88. Juvenile lake sturgeon from the Albany River (Hat Island), Ontario.
Photo credit: D. Potvin, Ontario Ministry of Natural Resources.

Lake sturgeon is a sensitive indicator of the health of aquatic environments, because it is a long-lived species thought to have strong site fidelity for spawning and other habitat requirements (COSEWIC 2006). Its slow rate of growth, late age to maturity (15-25 years), and infrequent spawning behavior (once every 4-6 years for females and every 2-3 years for males) make it very vulnerable to over-harvest and habitat change (Scott and Crossman 1973; Brousseau 1987). The species is also sensitive to river fragmentation, because it commonly migrates up to 100 km or more between summering, spawning, and wintering habitats (McKinley et al. 1998). Lake sturgeon is culturally important to local Aboriginal peoples, including as a valued food source (Thompson and Hutchison 1987; Berkes et al. 1992; DeBeers Canada 2005).

As already noted, lake sturgeon is the only fish species in the Hudson Plains Ecozone that is of national conservation concern (Section 2.3.2, *Trends in Species of National Conservation Concern*). It is currently assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as Special Concern over most of the Hudson Plains Ecozone (DU7 populations), while populations inhabiting the Churchill and Nelson rivers in Manitoba (DU1 and DU3 populations) are assessed as Endangered, because of reduced population sizes due to historic harvest activities and current hydroelectric development along those systems (COSEWIC 2006). These most recent COSEWIC designations for lake sturgeon are a deterioration from the Not at Risk designation last assigned by COSEWIC in 1986. Although lake sturgeon currently receives no legal protection under Canada's Species at Risk Act (Government of Canada 2010), any international trade in this species (e.g., for caviar) is governed under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 2010; listed under CITES' Appendix II). Provincially, lake sturgeon is considered a *heritage species* in Manitoba (MDNR 1991)[48], and the Hudson Bay-James Bay population of this species is listed as Special Concern in Ontario (COSSARO 2010).

Status

Lake sturgeon occurs in all major rivers of the Hudson Plains Ecozone (Churchill, Nelson, Hayes, Severn, Winisk, Attawapiskat, Albany, Moose, Harricana, Nottaway, Rupert, and Eastmain rivers) and in their main tributaries and connecting large lakes (OMNR 1985; Verdon 2001; Kerr 2002; Schetagne et al. 2003; Therrien et al. 2004; COSEWIC 2006). Known occurrences of the species in rivers and lakes can be mapped (e.g., see Figure 89 for the Ontario portion of the ecozone or Verdon 2001 for the Québec portion), but without systematic sampling any apparent absence of the species on this type of distribution map does not necessarily mean that the species does not occur there. As well, the spatial extent of this species within rivers and its abundance, condition, and overall population and habitat ecology are largely unknown in most river systems in the ecozone, except some rivers directly affected by hydroelectric developments, including the Nelson, Eastmain, and Rupert rivers. The species has been the focus of considerable study along the Nelson River in Manitoba, with much effort spent monitoring the status of lake sturgeon stocks along the river, determining life history information, documenting movement patterns, and identifying and mapping critical habitats, such as spawning locations (e.g., see COSEWIC 2006). The status of lake sturgeon was also monitored from 1978-1979 to 2000 in the area affected by the La Grande hydroelectric complex in Québec, including the lower Eastmain River (and its tributary Opinaca River) and the Opinaca reservoir (Therrien et al. 2004). More recently, the species is being monitored in areas affected by the new Rupert River diversion project in Québec, including changes in its existing spawning grounds, albeit mostly upstream of ecozone boundaries (Environnement Illimité 2010a,b).

Limited data are available on lake sturgeon population sizes around Hudson and James bays. For the Manitoba portion of the ecozone, COSEWIC (2006) reports that about 500 fish were

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[48] Lake sturgeon is designated a heritage species in Manitoba due to its "unique life history characteristics, limited distribution, and economic, social, and historical significance" (MDNR 1991).

Data Source:
- Cochrane District - Aquatic Resource Areas
- Hearst District - Aquatic Resource Areas
- Atlas: Kerr, S.J. 2002 Atlas of Lake Sturgeon Water in Ontario, OMNR
- Milan Vukelich, OMNR NESI

Figure 89. Known occurrence of lake sturgeon in major rivers, main tributaries, and connecting large lakes in the Ontario portion of the Hudson Plains Ecozone, as of 2009.

recorded in the area below the Limestone Dam (lower Nelson River) in 1998. MacDonnell (1995) suggests that, because the remaining 100 km of the lower Nelson River below the Limestone Dam is barrier-free, it perhaps contains the last true riverine stock of lake sturgeon on that river. Studies indicate that the lake sturgeon that use the lower Nelson River are known to spawn only at its confluence with the Weir River (McDonnell 1997), where fish from the Hayes River might likewise spawn (Barth and MacDonnell 1999). In summarizing the observations from these studies on the lower Nelson River, the COSEWIC (2006) report suggests that the lack of older, larger fish and low numbers of larvae are, however, indicative of a stressed population with low recruitment. At least one additional hydroelectric development is proposed for that segment of the Nelson River (Manitoba Hydro 2010; W. Bernhardt, North/South Consultants, pers. comm.) (for details, see Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Three lake sturgeon population studies have been conducted in the Ontario portion of the ecozone. The Moose River study (Threader 1981; Threader and Weir 1981) provided a population estimate of 7,088 lake sturgeon, and it indicated no tolerance for increased fishing intensity under regulations at the time. The other studies were conducted on the Attawapiskat River (EAG 1988) and in the Mammamattawa area of the Kenogami River (Ecologistics 1987; Sandilands 1987). Unfortunately, due to a lack of recaptures in both studies, only a spring population estimate of 3,432 lake sturgeon was provided for the Mammamattawa population. At the time, it was also recommended that total annual harvest of lake sturgeon from all sources be limited to 200 kg. More recent population estimates are not available.

Lake sturgeon populations in the Moose and Attawapiskat rivers were also demonstrated to have below-average growth compared to those in the Groundhog River in the adjacent Boreal Shield Ecozone (Threader and Weir 1981). Marked differences in length-at-age were evident, and weight-length relationships appeared to follow a pattern of decreasing size for similar age classes the more northerly the habitat (Threader and Weir 1981; EAG 1988). It was suggested that the smaller size of lake sturgeon in the north could be attributed to shorter growing seasons, limited habitat selection, lower productivity of rivers, and/or overpopulation and competition. Although speculative, most such conditions are not uncommon to the Hudson Plains Ecozone, and they might interact to produce a *stunted* population (Threader and Weir 1981).

No population size estimates were located for lake sturgeon within the Québec portion of the ecozone, although the distribution of this species is being mapped there (Verdon 2001). As well, the impact of hydroelectric development on the relative abundance of this species has been monitored via fish station yields in areas affected by the La Grande hydroelectric complex, including the lower Eastmain River (Verdon 2001; Therrien et al. 2004; and see below for some associated trends). More recently, waters affected by the new Rupert River diversion project are being monitored for effects on lake sturgeon recruitment (albeit mostly upstream of the ecozone) (Environnement Illimité 2010a,b). Ferguson and Duckworth (1997) reported that lake sturgeon were relatively rare in the Harricana, Nottaway, and Eastmain rivers.

Trends & human influences

Lake sturgeon is currently susceptible to a number of human influences in the Hudson Plains Ecozone, including hydroelectric developments, contaminants, and harvest (see below). Still, the species is thought to be in comparatively good condition overall in the ecozone as a whole, owing largely to the remoteness and limited amount of human disturbance in most of this geography, particularly within the bulk of the ecozone that lies in Ontario. This situation contrasts with the decline or extirpation of lake sturgeon from many more developed areas of North America (LeBreton et al. 2004) and for which over-exploitation and habitat degradation and loss are identified as the main causes of the loss (Jelks et al. 2008). If further resource development (including additional hydroelectric development) and increased access occurs in the Hudson Plains Ecozone, it would be expected that additional stress on lake sturgeon populations would result.

Hydroelectric developments

Hydroelectric developments in and around the Hudson Plains Ecozone (detailed in Section 2.2.2.4.2, *Rivers/Streams & Lakes*) are currently the greatest stressor of lake sturgeon in the ecozone. Populations inhabiting the regulated Churchill River (mean annual flow reduced by ~40%) and Nelson River (to which flow from the Churchill River was diverted) in Manitoba are now classified as Endangered owing to reduced population sizes associated with both historic harvest activities (residual effects) and current hydroelectric development along those river systems (COSEWIC 2006). Towards the eastern edge of the Hudson Plains Ecozone in Québec, abundance of lake sturgeon has also strongly declined in the Eastmain River and its tributary Opinaca River after most of the flow (~90%) from these rivers was diverted north to the La Grande River (Hayeur 2001; Therrien et al. 2004). Declines in abundance in the Eastmain and Opinaca rivers are attributed to very low recruitment. Low recruitment is, in turn, thought to have resulted from reduced quality of, and/or access to, spawning grounds (due to the sharp reduction in flow and fragmentation or submersion of residual habitat by a series of weirs), along with increased harvest (increased ease of net fishing associated with the reduced flow and new road access) (Hayeur 2001; Therrien et al. 2004). Fish community composition has changed as well, with lake sturgeon no longer dominating the community (Therrien et al. 2004). Increasingly fewer small specimens of lake sturgeon were caught in fishing surveys in the reduced flow segments of the Eastmain and Opinaca rivers during the 1980-1996 post-development monitoring period for this species (Therrien et al. 2004), and the species was almost totally absent from catches in 1998 (Doyon and Belzile 2000). The effects of the new Rupert River diversion on the lake sturgeon population in the lower, reduced flow segment of the Rupert River are not yet known.

Dams directly and indirectly affect lake sturgeon by acting as barriers to their movement at certain times of the year (especially during spawning), causing seasonal disruptions in their habitat, and disrupting spawning triggers and timing (COSEWIC 2006). The fragmentation of lake sturgeon habitat by dams results in the separation or exclusion of particular life stages (adults upstream and yearling/juvenile downstream), as well as stranding downstream, in some cases, after peaking plant spillway flows (S. McGovern, Ontario Ministry of Natural Resources, pers. obs.). As such, due to their migratory life history, lake sturgeon require large unfragmented watersheds to thrive (Browne 2007). Any barriers, such as hydroelectric dams, could negatively impact their migratory behavior and access to areas that are required to maintain healthy populations.

It is not entirely clear whether lake sturgeon populations can persist over the long term following dam construction (Browne 2007). In southerly rivers, where dam construction occurred 70 to 100 years ago, many lake sturgeon populations have been lost entirely. Numerous studies have been conducted on northerly lake sturgeon populations in the upper Moose River Basin just south of the Hudson Plains Ecozone. However, because most of the dam construction in the lower Moose River Basin occurred in the mid-1960s, full effects on resident lake sturgeon populations might not yet be evident (Browne 2007).

Contaminants

Information on contaminants is very limited for lake sturgeon in the Hudson Plains Ecozone (OMOE 2009). It is, however, of interest, both because lake sturgeon is a long-lived, slow-growing species (bioaccumulated contaminants, including mercury, tend to occur at higher concentrations in older and larger fish; Seyler 1997) and because mercury levels are already relatively high in natural aquatic environments in this ecozone and proximal northerly areas, especially where organic content is high (Schetagne et al. 2003). Where sampled, levels of mercury in larger individuals of this species have been high enough to be a potential human health risk as, for example, in the lower Moose and Abitibi rivers, where levels also appeared somewhat higher than in the Claybelt portion of the adjacent Boreal Shield Ecozone (Seyler 1997). As such, the accelerated conversion of mercury to methylmercury that is associated with some hydroelectric developments (Schetagne et al. 2003; Therrien and Schetagne 2008) is a concern (Rosenberg et al. 1997). As previously noted, significant increases in mercury levels in a variety of other fish species were observed over an extended period of time following creation in 1980 of the Opinaca reservoir in the Quèbec portion of the ecozone (Figure 67, in Section 2.2.2.4.2, *Rivers/Streams & Lakes*). Although present in the area, lake sturgeon was not similarly a target species for the reporting of mercury monitoring efforts there (where it declined in abundance). However, the mean pre-development (baseline) level of mercury in lake sturgeon was fairly low at 0.25 (0.03-0.40) mg mercury/kg flesh, based on a sample of 186 lake sturgeon of standardized length (700 mm) from 11 lakes (Schetagne et al. 2003). For the same size class (relatively small fish), the level of mercury in lake sturgeon from the lower Moose and Abitibi rivers mentioned above was closer to 0.5 mg/kg (Seyler 2007).

Harvest

Lake sturgeon has been commercially and recreationally harvested in the Hudson Plains Ecozone. The contribution of historic harvest activities to reduced population sizes of this long-lived species is recognized as, for example, in the Churchill and Nelson rivers in Manitoba (COSEWIC 2006). Regulated commercial fishing for lake sturgeon once existed in each of the major rivers in the Manitoba portion of the ecozone, but these commercial fisheries closed by 1994. As well, regulated commercial and recreational harvesting of lake sturgeon ceased in the Ontario portion of the ecozone in 1995 (Wilson 1996) and 2008 (OMNR 2008), respectively. By 2009, Ontario also implemented a province-wide ban of any non-Aboriginal commercial or recreational harvest of lake sturgeon, so as to prevent overexploitation associated with an increasing demand for this species from the international market (OMNR 2009). In a major part of the lake sturgeon distribution area in northern Québec, lake sturgeon fishing has been limited to Aboriginal peoples, since 1975, according to the James Bay and Northern Québec Agreement (Governments of Canada and Québec 1976; Government of Québec 1998). South of this territory, recommendations have been made to close most of the lake sturgeon fisheries in the James Bay watershed (COSEWIC 2006). However, fishing pressure for lake sturgeon persists in the ecozone, mainly from the Aboriginal food fishery (Thompson and Hutchison 1987; Berkes et al. 1992; Wilson 1996; DeBeers Canada 2005), which can lead to localized over-harvest, particularly in the vicinity of population centres (OMNR 1985; Therrien et al. 2004). As already noted, the strong decline in lake sturgeon numbers in the downstream, reduced flow portion of the Eastmain River was attributed, in part, to increased harvest that was facilitated by the improved access and increased ease of net fishing associated with the hydroelectric development (Therrien et al. 2004).

2.3.3.4.2 **Lake trout**

Lake trout is a coldwater fish species widely distributed throughout Canada, from the arctic islands in the north to the Great Lakes in the south (Scott and Crossman 1973). Although Ontario alone has 20-25% of all lake trout lakes in the world (OMNR 2007), lake trout is confirmed in only four lakes in the Ontario portion of the Hudson Plains Ecozone (OMNR 1985): Hawley Lake (1,235 ha); Sutton Lake (3,764 ha); North Raft Lake (348 ha); and Aquatuk Lake (565 ha) (Figure 90). A small basin south of Aquatuk Lake is alleged to contain an additional naturally reproducing population (Scholten 1991), but this observation has never been verified. All four of the confirmed lake trout lakes in Ontario are associated with the Sutton Ridges in the central part of the ecozone (Figure 90). These lakes are apparent outliers for lake trout in Ontario, as the nearest other lake known to contain lake trout lies 350 km away on the Canadian Shield. Lake trout survived on the Sutton Ridges after the last glacier receded, because the volcanic geology there provided the deep waters and spawning habitat for trout not found in the sedimentary lowlands (Eason 1980).

Within Manitoba, lake trout occur mostly in larger, deeper lakes found outside of the Hudson Plains Ecozone (Stewart and Watkinson 2004). Some lake trout might occur in smaller lakes along the western periphery of the ecozone there (Stewart and Watkinson 2004; W. Bernhardt, North/South Consultants, pers. comm.). In the eastern, Québec portion of the ecozone, lake trout occur naturally in low abundance or with “very marginal presence” (Therrien et al. 2004; and for distribution see Verdon 2001).

Although limited in its distribution in the Hudson Plains Ecozone, lake trout is very important to the culture, nutrition, and economy of local Aboriginal peoples (Eason 1980; Thompson and Hutchinson 1987), and it offers high recreational value to sport anglers (McKnight and Hendry 1988). The species attracts a considerable number of fly-in sport fishermen, particularly at Hawley Lake and to a lesser extent at Sutton Lake (Scholten and Byers 1990; Scholten and Thompson 1992).

Status

Information about the lake trout populations in the Ontario portion of the ecozone (Hawley, Sutton, North Raft, and Aquatuk lakes) is dated. Numerous studies were conducted there over the period 1959-1991 (Hendry 1986; Scholten 1989; Thompson 1989; Wiechers et al. 1989; Scholten 1990a,b, 1991; Scholten and Thompson 1992). Studies focused on netting data to obtain abundance indices, fecundity information, genetic identification, and/or stomach analyses. Angling pressure and harvest data were obtained through on-lake creel surveys or an angler diary program for Hawley, Sutton, and Aquatuk lakes.

At that time, lake trout yields of 0.9 kg/ha/yr and 0.8 kg/ha/yr were predicted for Hawley Lake and Sutton Lake, respectively, which corresponded to total annual allowable harvests of approximately 1,100 and 3,000 kg (Scholten and Thompson 1992). Lake trout in Hawley Lake was being harvested at the maximum sustained level, but the population was not displaying signs of overexploitation. Lake trout in Sutton Lake was being harvested substantially below the sustained annual yield. At both Hawley and Sutton lakes, the rate of growth of lake trout was above the median for Ontario populations (Payne et al. 1990). Faster growth and lower harvest resulted in a larger average size of angled lake trout from Sutton Lake, although neither lake was producing trophy-sized lake trout. The high abundance of lake trout, which provided high angler success, appeared to be the main attraction for tourist anglers (Scholten and Thompson 1992).

Figure 90. Known occurrence of lake trout in the Ontario portion of the Hudson Plains Ecozone, as of 2009. Lake trout populations occur only in the four geologically anamalous lakes associated with the Sutton Ridges.

Lake trout populations in North Raft and Aquatuk lakes are comparatively smaller (Scholten and Thompson 1992). Catch-per-unit-effort (CUE) for lake trout by index gillnet for Aquatuk Lake in 1964 and 1988 was 0.35 and 0.43, respectively. North Raft Lake had a CUE of 0.26 and 3.72 for 1964 and 1988, respectively (Scholten 1989). In 1991, only a single lake trout was captured in Aquatuk Lake for a very low 0.03 lake trout per net set (Scholten 1991). Aquatuk Lake was dominated by northern pike and offered poor habitat for lake trout.

A lack of follow-up monitoring and assessment of the lake trout populations in the Ontario portion of the ecozone (1990s to present) has resulted in their current status being unclear. Updated information is, however, expected to become available in the near future due to the recent implementation of Ontario's new broad-scale fisheries monitoring program (described in Section 2.2.2.4.2, *Rivers/Streams & Lakes*), which includes sampling within the Hudson Plains Ecozone. As previously noted, the presence but not overall status of lake trout populations has been noted also in the Québec portion of the ecozone (Verdon 2001), where this species occurs marginally in low abundance (Therrien et al. 2004). This species did, however, become rarer in the area of the Opinaca reservoir after its impoundment (Hayeur 2001).

Trends & human influences

While the distribution of lake trout has not changed in the ecozone (lake trout still exists in all four lakes in Ontario and at the eastern edge of the ecozone in Québec), it is not clear whether any trends in the abundance and condition of its populations are occurring. What is clear is that lake trout in the ecozone is likely to continue to endure subsistence and recreational harvesting pressure. As well, climate change is anticipated to exert an important influence on its populations in the future.

Harvest

Both recreational angling by sport fishers and subsistence harvest by Aboriginal peoples are likely to continue in the lake trout lakes in the ecozone. Although fishing levels were not exceeding sustainable limits in the popular Hawley and Sutton lakes in the past (Scholten and Thompson 1992), it is unknown if current fishing levels are exceeding sustainable limits. The smaller lake trout populations in the North Raft and Aquatuk lakes (Scholten and Thompson 1992) are likely more vulnerable to high fishing pressure than the larger populations in Hawley and Sutton lakes. Any future road development in the ecozone that enhances human access to lakes with lake trout will likely increase harvest pressure considerably; lake trout populations tend to be highly vulnerable to road access (Gunn and Sein 2000; Kaufman et al. 2009).

Climate change

Climate change is anticipated to have major negative impacts on lake trout populations, as coldwater habitat warms and more competitive warmwater fishes move north, changing food web dynamics (e.g., Dove and Lewis in prep.). On the other hand, depending on the level of warming, lake trout could adapt to some extent (Sellers et al. 1998), by changing its behaviour in order to survive (Snucins and Gunn 1995) or, where in isolated populations, possibly persisting for a time in deeper stratified lakes within its original range (Stefan et al. 1995).

During the unusually warm summer of 2001, Hawley Lake showed strong thermal stratification for one of the first times on record, with temperatures exceeding 20 °C in the surface layer (Snucins 2003; Gunn and Snucins 2010; see also Figure 69 in Section 2.2.2.4.2, *Rivers/Streams & Lakes*). However, despite the unusual warming, ample habitat for lake trout persisted below the epilimnion, and dissolved oxygen levels were high (~9-10 mg/L) throughout the water column. Conversely, a major die-off of brook trout (*Salvelinus fontinalis*) and white sucker (*Catostomus commersoni*) was observed in the lower reaches of the Sutton River, downstream of this headwater lake (Gunn and Snucins 2010; and see also the *Brook Trout* heading below).

Ultimately, the effects of climate change on the ecozone's lake trout will depend largely on individual lake characteristics, owing to ecosystem complexity and the relationships among dissolved organic carbon, thermocline depth, habitat availability, and phosphorous concentrations. More knowledge of both the ecozone's lakes and lake trout populations is required to effectively manage these fish communities in the future (Dove and Lewis in prep.).

2.3.3.4.3 **Brook trout**

Brook trout is a coldwater fish species endemic to northeastern North America (Scott and Crossman 1973). In the Hudson Plains Ecozone, brook trout is found in most major rivers, tributary streams, and creeks, as well as smaller lakes (Ryder et al. 1964; Verdon 2001; North/South Consultants 2010). Most coastal waterbodies contain sea run brook trout, which are unique in Ontario. These migrant brook trout spend a portion of their life in rivers or streams and then migrate to the sea.

Brook trout is considered a *heritage species*[49] in Manitoba (MDNR 1991). In Ontario, one of North America's top quality brook trout fishing areas, the Sutton Ridges, is in the Hudson Plains Ecozone (Wheeler 1985; McKnight and Hendry 1988). The Sutton River itself has been considered a world-class brook trout fishery, having both abundant and large fish (Thompson 1989; Scholten and Byers 1990; Gunn and Snucins 2010). The growth rate of brook trout from this river has been considered a record for natural populations (Steele 1986).

Brook trout in the ecozone also serves as an important food source for resident Aboriginal peoples, although the contribution of brook trout to total Aboriginal fish harvest varies among communities (Melville et al. 1915; Thompson and Hutchison 1987; Berkes et al 1992; DeBeers Canada 2005). Aboriginal peoples have also benefited economically from the brook trout fishery, by providing camps used by recreational anglers (Wiechers et al. 1989).

Status

Very little specific information exists on the distribution, abundance, and condition of brook trout populations in the Hudson Plains Ecozone. Although brook trout is considered widely distributed and abundant along the coasts of Hudson and James bays (Scott and Crossman 1973), it has been neither extensively studied nor exploited, due largely to the relatively vast and generally inaccessible nature of most of this terrain (Wypkema 1975). Known occurrences of the species can be mapped (e.g., see Figure 91 for the Ontario portion of the ecozone or Verdon 2001 for the Québec portion), but without systematic sampling any apparent absence of the species does not categorically mean the species does not occur there. For example, in the Ontario portion of the ecozone, fish species occurrence is currently available for only 60 rivers, 23 streams, and 41 lakes. Of these, brook trout has been identified in 39 rivers, 20 streams, and 12 lakes (Lower 1915; Melville et al. 1915; Ryder et al. 1964; OMNR 1985; Seyler 1997; Kerr 2002; DeBeers Canada 2005). Most of these rivers and streams contain sea run brook trout. More

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[49] Brook trout is given heritage status in Manitoba, because of its limited natural distribution, socio-economic importance, and unique life history characteristics.

Data Source:
- Cochrane District - Aquatic Resource Areas
- Hearst District - Aquatic Resource Areas
- Milan Vukelich, OMNR NESI

Figure 91. Known occurrence of brook trout in rivers, streams, and lakes in the Ontario portion of the Hudson Plains Ecozone, as of 2009.

information about the distribution, abundance, and condition of brook trout in the ecozone is, however, expected to be generated by Ontario's new broad-scale fisheries monitoring program. In Québec, brook trout has been recorded in fish yield and condition monitoring studies associated with the La Grande hydroelectric complex, but this species has not been a focal point for reporting on the portion of the development that lies in the Hudson Plains Ecozone (Schetagne et al. 2003; Thierren et al. 2004).

The presence of a well-defined channel connecting a river to the sea appears to be a major difference that distinguishes rivers in the ecozone with notable sea runs of brook trout from those where sea runs are inconspicuous or lacking (Weir 1981). Although brook trout are able to leave rivers with poorly defined channels easily during spring run-off, it is difficult for them to re-enter these rivers from the sea during late summer, when water levels in the rivers are low. Thus, there might be a net export of fish from such rivers and a gain for those with well-defined channels (Weir 1981).

Brook trout population ecology, abundance, condition, harvest, and pressure assessments have been very limited in the Hudson Plains Ecozone to date. No studies have been conducted in Ontario, since the early 1990s. Life history studies and the role of anadromy were investigated on the Brant River during 1972-1974 (Wypkema 1975), the Wapiskau River during 1978-1979 (Weir 1981), and the Sutton River during 1983-1984 (Steele 1986) and 1990-1991 (Malette 1993). The importance of anadromy to the ecozone's brook trout was demonstrated. Mallette (1993) reported that Hudson Bay provided a good food source that allowed the energy demands of brook trout for reproduction and over-wintering to be replenished quickly in comparison to rivers; anadromy thus allowed more fish to reproduce in the Sutton River and to be more productive. In terms of population assessments, the Brant River population was reportedly being over-harvested (Wypkema 1975), while the Sutton River population was suggested to be stable (Malette 1993). Robinson and McDonald (1990) dispelled the assertion by Steele (1986) that brook trout in the Hudson Plains Ecozone had a blood clotting deficiency. This deficiency was previously believed to cause high hooking mortality rates in brook trout that had been angled and released.

Trends & human influences

No overall trends in the distribution, abundance, or condition of brook trout have been documented in the Hudson Plains Ecozone, where this species is thought to be wide-spread and abundant (as above). However, localized harvesting pressure exists and will continue. As well, the anadromous nature of this species renders it susceptible to hydroelectric development, and its coldwater nature renders it particularly susceptible to climate change.

Harvest

Recreational angling and the Aboriginal food fishery continue to be important influences on brook trout populations in parts of the Hudson Plains Ecozone. High fishing pressure was reported on four of the most popular brook trout rivers (Sutton, Brant, Shagamu, and Gorge rivers, between Hawley and Sutton lakes) in the 1970s and 1980s (Wypkema 1975; OMNR 1985; McKnight and Hendry 1988). Although fishing pressure is probably still relatively high in these locales, impacts are uncertain due to insufficient monitoring. The relationship between harvest and potential yield for specific waterbodies is currently unknown. Information on potential yield for the Ontario portion of the ecozone (OMNR 1985) does, however, suggest that the ecozone still supports healthy brook trout populations overall. Still, localized high harvest could potentially be occurring in the vicinity of population centres and other high recreational use areas.

Hydroelectric developments

Hydroelectric developments and other encumbrances that block fish migration in rivers are likely detrimental to anadromous brook trout populations in the Hudson Plains Ecozone (Weir 1981; Malette 1993). Access to a rich food supply in Hudson Bay is demonstrably important for the ecozone's brook trout to attain the higher energy levels required for reproduction in rivers and to otherwise maintain healthy populations (Mallette 1993). Dams can also impact upstream habitat by converting riverine habitat to lake habitat, executing winter drawdowns that affect fall spawners, and fragmenting habitat. Although quantitative data are sparse on the response of brook trout to such conditions specifically in the Hudson Plains Ecozone, monitoring has shown that this species declined in abundance in the Caniapiscau River just north of the ecozone (at its eastern extent) after the flow of this river was reduced by hydroelectric development (Therrien et al. 2004). Conversely, brook trout abundance increased in the La Grande River after the mean flow of this river more than doubled (from 1,700 m^3/s to 3,400 m^3/s), due to the receipt of water from diverted rivers to such an extent that summer water temperatures in the La Grande River substantially declined (from maximum 16 °C to maximum 8 °C) (Therrien et al. 2004). In the latter case, the abundance of brook trout and other coldwater species increased to the detriment of the warmwater species cisco and walleye. Within the ecozone, brook trout, which was marginal in the area of the Opinaca reservoir prior to its development, become rarer there following the impoundment (Hayeur 2001).

Climate change

Climate change is expected to have major implications for brook trout, due to direct effects of warming on suitable habitat and by promoting a northward shift in the ranges of competitive warmwater fish (Magnuson et al. 1997). Local observations in the Hudson Plains Ecozone do suggest that warming can be highly detrimental to brook trout habitat in some cases. For example, a major die-off of brook trout, as well as white sucker, occurred in the lower reaches of the Sutton River in 2001 after unusually warm summer temperatures greatly increased water temperatures in the headwater lake (Hawley Lake) (see Figure 69 and related discussion in Section 2.2.2.4.2, *Rivers/Streams & Lakes*). It was suggested that this observed die-off of brook trout could be among the first of an increasing number of die-offs of vulnerable anadromous stocks that will occur as climate change proceeds, given that such stocks are dependent on the seasonal sea ice cover in Hudson Bay to moderate the continental climate (brook trout return from the cold bay to spawn and over-winter in freshwater) (Gunn and Snucins 2010). Trends are already evident for a shortening sea ice season in Hudson Bay (Section 2.1, *Abiotic Drivers*).

A 49% decrease in brook trout distribution in Canada by the year 2050 is projected based on a CGCM2 climate model using an IS92a (business as usual) emissions scenario (Chu et al. 2005). This broad-scale analysis suggests major range changes for brook trout specifically in the area of the Hudson Plains Ecozone (Figure 92). On the other hand, it does not provide detailed information regarding the degree of vulnerability of brook trout populations in the ecozone. Thermal habitat availability has not been quantified for the ecozone's rivers and streams, nor has its importance to brook trout growth and production (Dove and Lewis in prep.).

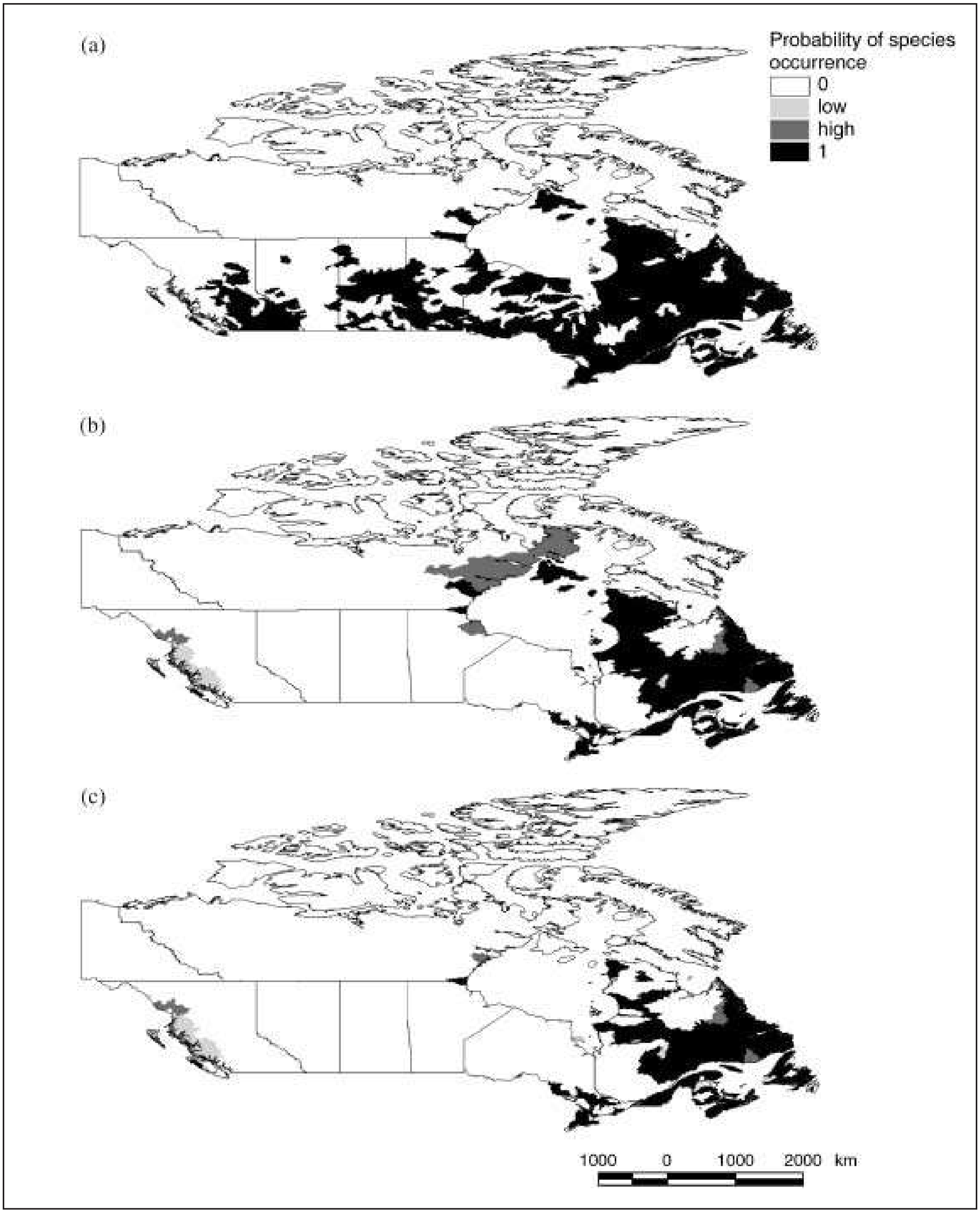

Figure 92. Projected changes in brook trout distribution in Canada: a) current distribution; b) potential distribution under CGCM2 (IS92a) climate scenario in 2020; and c) potential distribution under CGCM2 (IS92a) climate scenario in 2050.
Source: Chu et al. (2005). Reprinted from Diversity and Distributions, Vol 11, No. 4, C. Chu, N.E. Mandrak and C.K. Minns, Potential impacts of climate change on the distributions of several common and rare freshwater fishes in Canada, Copyright 2005, with permission from Blackwell Publishing Ltd.

Although most studies predict potentially large shifts in fish species distributions with climate warming, some isolated populations of coldwater fishes could potentially remain in deep stratified lakes and other suitable habitat within their original range (Stefan et al. 1995). The influence that permafrost thaw might have is also uncertain. Meisner et al. (1988) suggested that permafrost thaw might potentially produce year-round groundwater flow, providing more stream habitat for brook trout. However, no information is currently available about the effect of permafrost meltwater on groundwater and flow regimes in northern streams and rivers, nor any related effects on brook trout (Dove and Lewis in prep).

The projected northward shift in the ranges of warmwater fish (Magnuson et al. 1997)[50] is anticipated to increase competition for brook trout and other native coldwater fish species and otherwise change foodweb interactions (Minns and Moore 1995; Chu et al. 2005; Dove and Lewis in prep.). Brook trout is extremely sensitive to community changes, and it is typically out-competed by warmwater species, such as bass (Dove and Lewis in prep.). Climate warming over the next 50 to 100 years could allow smallmouth bass to establish in more northerly waters (Minns and Moore 1995; Chu et al. 2005). Smallmouth bass was recently (2008-2009) reported in the Hudson Plains Ecozone for the first time (see discussion of smallmouth bass under the *Introduced & Potentially Invasive Species* heading in Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

2.3.3.4.4 **Lake whitefish & cisco**

Lake whitefish and cisco are widely distributed throughout Canada (Scott and Crossman 1973). Within the Hudson Plains Ecozone, these fish species are found in all major rivers; in some smaller rivers, tributary streams, and creeks; and a number of inland lakes (OMNR 1985; Verdon 2001; Schetagne et al. 2003; Thierren et al. 2004). The rivers flowing directly into Hudson and James bays that contain these species host anadromous stocks, which move between freshwater for the spawning period (Lambert and Dodson 1990a,b) and estuarine waters (Ryder 1965), where feeding potential is high (Greendale and Hunter 1977).

Lake whitefish and cisco have been, and continue to be, highly valued aquatic resources in the Hudson Plains Ecozone. These species represented a large and important part of commercial harvests there in the early 1900s (Lower 1915), and currently they form an important part of the regulated commercial fishery in Manitoba (W. Bernhardt, North/South Consultants, Winnipeg, MB). Lake whitefish and cisco have also been the major fish species harvested by Aboriginal peoples in the ecozone (Thompson and Hutchison 1987; Berkes et al. 1992; Cummins 1992; Hughes et al. 1993; DeBeers Canada 2005). For example, the estimated total harvest of lake whitefish from eight communities in the Ontario portion of the ecozone averaged 71% of the total fish harvest in 1981-1983 and 59% in 1990 (Thompson and Hutchinson 1987; Berkes et al. 1992). Little recent harvest data exist, but lake whitefish and cisco very likely still provide a substantial subsistence food fishery for Aboriginal peoples throughout the ecozone.

Status

No comprehensive effort exists across the ecozone to monitor the status of lake whitefish or cisco populations. Again, known occurrences of these species can be mapped (e.g., see Figure 93 for the Ontario portion of the ecozone or Verdon 2001 for the Québec portion), but without systematic sampling any apparent absence of a species does not necessarily mean the species

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[50] Magnuson et al. (1997) estimated that the boundaries of freshwater fish could move north by 120 km for each 1 °C rise in air temperature, which could mean a 500-600 km northward expansion by the 2050s.

Figure 93. Known occurrence of lake whitefish and cisco in rivers, tributaries, and lakes in the Ontario portion of the Hudson Plains Ecozone, as of 2009.

does not occur there. In the Ontario portion of the ecozone, lake whitefish has been identified in 32 rivers, four streams, and 29 lakes out of the 60 rivers, 22 streams, and 41 lakes for which information on fish species occurrence is available. Cisco has been identified in 18 of these rivers, four of these streams, and 23 of these lakes (Lower 1915; Melville et al. 1915; OMNR 1985; Seyler 1997; Kerr 2002; DeBeers Canada 2005).

Even where lake whitefish and cisco are known to occur, their spatial distribution within rivers and their abundance, condition, and overall population and habitat ecology is for the most part not well understood. In the 1980s, when these species were being investigated in Ontario, anadromous stocks were believed to be in stable equilibrium with their physical and biotic environment along the coasts of Hudson and James bays, despite being subject to subsistence fisheries (Morin et al. 1982). More recent status assessments are not currently available. In Québec, changes in the relative abundance of both species has, however, been monitored in the area of the ecozone affected by the La Grande hydroelectric complex (see below).

Trends & human influences

The influence of commercial harvest on lake whitefish and cisco in the early 1900s was not well documented, nor was any subsequent recovery. Due to a general paucity of monitoring and assessment, it is also not clear whether any overall trends have been occurring in these populations in more recent times. Lake whitefish and cisco are, however, currently susceptible to a number of human influences, including hydroelectric developments and harvest and, as outlined below, some related localized declines in the relative abundance of these species are evident. Climate change is an emerging threat for both species, and the susceptibility of the ecozone’s lake whitefish to it is suggested by local Aboriginal knowledge, which indicates a die-off of lake whitefish (as well as sucker) occurred in the Albany River along the James Bay coast during a heat wave and period of reduced precipitation in 2005 (Hori 2010).

Hydroelectric developments

Hydroelectric developments within and adjacent to the Hudson Plains Ecozone can directly and indirectly affect lake whitefish and cisco. Aside from habitat alteration in the immediate vicinity of dams, hydroelectric developments block the long-distance movement of anadromous fish stocks (Rosenberg et al. 1997). Given that migration is an important component of the life cycle of lake whitefish and cisco (Lambert and Dodson 1990b), any encumbrances to their migration, such as hydroelectric developments, could be highly detrimental.

The impacts of hydroelectric development on these species was investigated in the area of the ecozone affected by the La Grande hydroelectric complex in Québec. Both lake whitefish and cisco populations recovered in the Opinaca reservoir after a period of adjustment, but lake whitefish recovered less well in the downstream, reduced flow portion of the Eastmain River (Therrien et al. 2004). The 90% reduction in flow at the mouth of the Eastmain River also led to a greater intrusion of saltwater. Although lake whitefish and cisco still migrate up the estuary in fall to spawn, their overwintering area is now smaller due to the saltwater intrusion (Therrien et al. 2004). Lake whitefish in the lower reaches of the Rupert River is currently a focus of monitoring activities associated with the new Rupert River diversion in Québec, although most such monitoring is occurring outside of ecozone boundaries (Environnement Illimité 2010a).

Hydroelectricity generation reservoirs can also affect contaminant load. As shown in Figure 67 in Section 2.2.2.4.2 (*Rivers/Streams & Lakes*), mercury levels in lake whitefish and other fish species were elevated for an extended period of time following impoundment of the Opinaca reservoir (Therrien and Schetagne 2008), a result similar to that of Bodaly et al. (2007) for hydroelectric developments upstream of the Manitoba portion of the ecozone. Another concern is that in some reservoirs (but not at the La Grande complex) the incidence of *Triaenophorus crassus* cysts has increased in the flesh of lake whitefish and affected its edibility (Hecky et al. 1984).

Harvest

Fishing pressure in the ecozone, and associated potential for harvest-driven declines in lake whitefish and cisco populations, has been mostly localized, e.g., close to communities and along major river systems (OMNR 1985). In the 1980s, when these species were being investigated in the Ontario portion of the ecozone, Aboriginal community harvests there were considerably less than the estimated potential food and sport fish yield (kg/yr) (OMNR 1985). More recent harvest data are, however, not available, so recent harvest trends are unknown. It is possible that sources of anadromous lake whitefish and cisco (as well as anadromous stocks of other species, like brook trout) might become favoured by members of the coastal communities, given that anadromous stocks, which presumably rely on more marine food sources than their freshwater counterparts, could carry lower levels of mercury, as has been observed elsewhere in northern Canada (Lockhart et al. 2005).

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2.3.3.5 **Reptiles & amphibians**

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The diversity of reptiles and amphibians (herpetofauna) in the Hudson Plains Ecozone is relatively low compared with that at more southerly latitudes (Currie 1991), as expected given that the distribution of herpetofauna is governed by their ecothermal nature and variable adaptations to tolerate freezing. The diversity and especially the distribution and abundance of the ecozone's herpetofauna is, however, not fully understood due to insufficient basic and systematic inventory effort in all component jurisdictions. For the most part, preliminary data (sight or specimen collection records) permit only simple speculation regarding their distribution.

Table 19 lists species or complexes of reptiles and amphibians documented in the ecozone. These species are all native and widespread in eastern Canada (CARCNET 2010), and none is currently classed by COSEWIC as being at risk (COSEWIC 2010 and see also Section 2.3.2, *Trends in Species of National Conservation Concern*). The lack of information for additional species does not categorically mean they are absent from the ecozone – particularly given the low amount of survey work done there to date. The only reported reptile is the common gartersnake (*Thamnophis sirtalis*) (Table 19), which is found in the James Bay Lowlands Ecoregion (217) in the Ontario and Québec portions of the ecozone. However, a single sight record in 1969 of a 4.5-6.8 kg snapping turtle (*Chelydra serpentina*) from Gillam, Manitoba (Schueler 1976) (just outside the northwestern ecozone boundary) suggests that this species might range further north into the Hudson Plains Ecozone (Preston 1982). More amphibian species than reptile species are known to occur in the ecozone (Table 19), although only toads and frogs have widespread sightings. A single record of the northern leopard frog (*Lithobates pipiens*) in the most northerly (Manitoba) portion of the ecozone (Preston 1982) might be intriguing, as this species is not recognized as freeze-tolerant. It has been reported only somewhat further south in the ecozone along the Hudson Bay coast and especially the James Bay coast of Ontario (Oldham and Weller 2000) and Québec (Bider 1976). More systematic evidence for species-specific spatial patterns of occurrence

Table 19. Species or complexes of reptiles and amphibians (herpetofauna) reported to occur in the Hudson Plains Ecozone by component jurisdiction. Herpetofaunal data specific to the ecozone's islands in James Bay (Nunavut) are sparse and mostly dated. The reports here for Nunavut are from Logier and Toner (1961) for American toad and boreal chorus frog on Akimiski Island and from Ouellet et al. (2009) for wood frog on Île Stag and mink frog and common gartersnake on Jacob Island. Species and common names follow Crother (2001).
Sources: Logier and Toner (1961); Preston (1982); Oldham and Weller (2000); Ouellet et al. (2009); CARCNET (2010); Ontario Nature (2010); Société d'Histoire Naturelle de la Vallée du Saint-Laurent et ministére des Ressources naturelles et de la Faune (2010); and J. Duncan, Manitoba Conservation Data Centre (2010).

Species		Jurisdiction(s) reported
Scientific name	Common name	
Reptiles[a]		
Snakes		
Thamnophis sirtalis	Common gartersnake	ON, QC, NU
Amphibians[b]		
Toads & frogs		
Anaxyrus americanus	American toad	ON, QC, NU
Lithobates pipiens	Northern leopard frog	MB, ON, QC
Lithobates septentrionalis	Mink frog	ON, QC, NU
Lithobates sylvaticus	Wood frog	MB, ON, QC, NU
Pseudacris crucifer	Spring peeper	ON, QC
Pseudacris maculata	Boreal chorus frog	MB, ON, QC, NU
Salamanders		
Ambystoma laterale	Blue-spotted salamander	ON, QC
Ambystoma jeffersonianum-laterale complex	Jefferson / blue-spotted salamander complex	ON
Eurycea bislineata	Northern two-lined salamander	ON, QC

[a] No species of lizards or turtles have been reported in the ecozone.
[b] No species of newts or mudpuppies have been reported in the ecozone.

is currently limited for this ecozone (Reiter et al. 2008; Ouellet et al. 2009). Overall, then, little can be concluded at this time, either about changes or trends in herpetofaunal diversity in the ecozone or how this diversity varies across the ecozone with climatic and edaphic gradients (but see McKenney et al. 1998 regarding the projection of species distributions based on climatic domains).

Herpetofauna, and especially amphibians, are considered good indicators of ecosystem health (Wake 1991; e.g., see Hecnar and M'Closkey 1996 for species relevant to the Hudson Plains Ecozone), because they have form and life history characteristics that make them particularly sensitive to environmental threats (Sparling et al. 2000). Unfortunately, the herpetofauna listed in Table 19 cannot currently be used as indicators of ecosystem health in the Hudson Plains Ecozone or otherwise profiled here in reference to their population trends, owing to a paucity of information on their distribution, abundance, and condition specifically in this ecozone. It is nonetheless noteworthy that populations of both reptiles and amphibians are declining globally (e.g., Gibbons et al. 2000; Houlahan et al. 2000; Hoffmann et al. 2010), including in areas of Canada where many species are at risk or in decline (Galois and Ouellet 2007; COSEWIC 2010). Changes in global amphibian populations are particularly well studied and show broad declines, since about 1960 (Houlahan et al. 2000).

Global declines in both reptiles and amphibians have been attributed to several main factors acting alone or in combination (Gibbons et al. 2000; Collins and Storfer 2003; Galois and Ouellet 2007): invasive species, unsustainable use, land use change (habitat loss and degradation), environmental pollution, global change (accelerated climate change and ultraviolet radiation), and disease. Herpetofaunal populations in the relatively remote and intact Hudson Plains Ecozone are unlikely to have changed much due to such factors originating directly in the ecozone (this report), with the exception of probable localized declines in amphibian abundance where tundra wetland (freshwater sedge meadow) habitat has been damaged by the excessive foraging of overabundant waterfowl (Mannan 2008; see also Section 2.2.2.2, *Polar-Tundra*). However, climate change and resource developments might be emerging or future threats.

A noteworthy disease of amphibians is the chytrid fungus, *Batrachochytrium dendrobatidis*, which causes the infectious skin disease chytridiomycosis that is now known to be responsible for some of the ongoing, rapid, and unprecedented decline, extirpation, and extinction of amphibians worldwide (e.g., Crawford et al. 2010; Dejean et al. 2010). Although this fungus is not currently a major factor affecting amphibian populations in Canada, the fungus has been present in southeastern Canada, since at least the early 1960s, and it apparently causes mild infections there in many of the same amphibian species that also occur in the Hudson Plains Ecozone (Ouellet et al. 2005). Although it is not known if *B. dendrobatidis* is present in the Hudson Plains Ecozone, this fungus is notably cold-tolerant, growing best at cooler temperatures under controlled conditions and causing lethal infections in association with cold temperatures in the field (see Ouellet et al. 2005 and references therein). Recent lethal outbreaks (increasing virulence) of this pathogen elsewhere in the world appear to be related to complex interactions between the pathogen and predisposing factors, including climate change (Rohr et al. 2008). Ranaviruses (family *Iridoviridae*), which are also present in southeastern Canada (Greer et al. 2005; Duffus et al. 2008), are another potentially lethal pathogen of amphibians that have been implicated in their decline (Chinchar 2002).

Standardized atlas survey efforts and monitoring and research across jurisdictions are needed to better understand the conservation status and population and range trends of reptiles and amphibians in the Hudson Plains Ecozone, including any stressors. New information for the Ontario portion of the ecozone is being gathered there through a new atlas project (Ontario Nature 2010) and a terrestrial biodiversity project (see Inset 10 in Section 2.3.1, *Overview of Species Diversity*). Monitoring protocols typically used in more southerly locales might not be easily applied in this ecozone, due to limited survey (calling) periods and access constraints (Mannan 2008; Reiter et al. 2008).

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2.3.3.6 **Invertebrates**

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Invertebrates perform critical ecological roles in northern climates. They serve as important food sources for many organisms, including other invertebrates (McClure 1943; Danks 1981), fish (Danks 1981), birds (Danks 1981; Milakovic et al. 2001; Milakovic and Jefferies 2003), and mammals. They are primary biotic pollinators of plants in subarctic climates (Danks 1981), and they facilitate the decomposition of organic matter and the cycling of nutrients in the environment (Danks 1981; Fleming et al. 2000). Knowledge of invertebrates within the Hudson Plains Ecozone is, however, poor. Due to sporadic and non-standardized sampling, any trends in the diversity, distribution, and abundance of invertebrates there are not apparent.

A single comprehensive study of biotic communities (including invertebrates) has been conducted in the Hudson Plains Ecozone near Churchill, Manitoba (McClure 1943). In addition, a small number of incidental surveys of the invertebrate fauna of the Ontario portion of the ecozone have been conducted over the past 50 years through the Northern Insect Survey (Freeman 1954; Riegert 1999), the Ontario Ministry of Natural Resources (Sutherland et al. 2005; Gan et al. 2009; Beresford et al. 2010; Forsyth 2010, in prep.; W.J. Crins, unpublished data; M. Oldham, unpublished data), and other researchers (e.g., Hammer 1955 – orabatid mites, Collembola in Churchill, MB; Herbert and Hahn 1986 – microcrustaceans in Churchill, MB; Keller and Pitblado 1989 – lacustrine microcrustaceans in northern ON; Shorthouse et al. 2002 – cynipid wasps, ground beetles, ground-dwelling spiders in Fort Severn, ON; Zhou et al. 2010 – Ephemeroptera, Plecoptera, Trichoptera in Churchill, MB). As well, specific invertebrate taxa were identified (e.g., spiders, beetles, and midges) as part of a broader ecological objective to relate goose foraging to species composition at La Pérouse Bay, Manitoba (Milakovic et al. 2001; Milakovic and Jefferies 2003). None of these surveys were comprehensive of the entire ecozone, and collections were limited to accessible locales near the Hudson Bay and James Bay coasts or around communities. There has been little effort to survey invertebrates in the remote interior of the ecozone. Invertebrate diversity from the Québec portion of the Hudson Plains Ecozone is likely similar to the remainder of the ecozone.

Invertebrate diversity in the ecozone is approximated in Table 20 from a cursory survey of literature and databases. McClure (1943) identified at least 436 of these species or morpho-species near Churchill, and the remainder were identified through some of the above-mentioned incidental, and mostly coastal, surveys.

Table 20. An approximation of invertebrate diversity of the Hudson Plains Ecozone. Information is non-exhaustive and derived from a cursory examination of available information.
Sources: Data compiled from McClure (1943); Shorthouse et al. (2002); Milakovic et al. (2001); Milakovic and Jefferies (2003); Sutherland et al. (2005); Gan et al. (2009); and M. Oldham (Ontario Ministry of Natural Resources, unpublished data).

Phylum	Number of families represented	Number of species represented
Platyhelminthes (flat worms)	1	1
Annelida (segmented worms)	2	2
Mollusca (including snails and bivalves)	7	9
Arthropoda (including spiders, mites, and insects)	165	570
Total	175	582

Within the arthropods, insects are the most abundant taxon, contributing ~75% of the diversity at both the family and species levels, while spiders contribute ~8% and ~15% of arthropod diversity at the family and species levels, respectively (Table 21). Within the insects, three orders (Diptera, Coleoptera, and Hymenoptera) comprise ~65% of the diversity at the species level (McClure 1943). The total number of insect species identified (417) is equivalent to one-half to one-third of the total number of insect species estimated in the Churchill area (R. Roughley, University of Manitoba, unpublished data). Clearly, more survey work is required to describe the diversity of insects and other invertebrates in the ecozone.

Table 21. An approximation of arthropod diversity of the Hudson Plains Ecozone. Information is non-exhaustive and derived from a cursory examination of available information.
Sources: Data compiled from McClure (1943); Shorthouse et al. (2002); Milakovic et al. (2001); Milakovic and Jefferies (2003); Sutherland et al. (2005); Gan et al. (2009); and M. Oldham (Ontario Ministry of Natural Resources, unpublished data).

Arthropod taxon	Number of families represented	Number of species represented
Crustaceans	9	14
Pseudoscorpions	1	1
Water mites	5	5
Mites	12	20
Spiders	14	89
Collembola	3	23
Insects	123	417
Total	167	569

Generally speaking, the invertebrate species described to date within the Hudson Plains Ecozone are not unique, as the majority of these species are of broader nearctic or holarctic distribution (Freeman 1954; Shorthouse et al. 2002); however, some species are at their northern or southern limit in the Hudson Plains Ecozone (McClure 1943; Freeman and Twinn 1954; Sutherland et al. 2005). Freeman (1954) describes two major entomofaunal divisions in Canada, the northern boreal forest and the arctic tundra, with little overlap in the transition zone (see also Danks and Foottit 1989). The former tends to be inhabited by a diverse suite of species, while the latter tends to be less diverse (Freeman 1954). A number of insect species meet the northern and southern limits of their range around Churchill. A number of Odonata (dragonflies and damselflies) have also been reported at their southern limits in the Ontario portion of the ecozone (Sutherland et al. 2005). Also found in the Ontario portion are a number of provincially rare to uncommon odonate species, including: *Aeshna septentrionalis*, *Aeshna subarctica*, *Leucorrhinia borealis,* and *Somatochlora hudsonica*, the latter two recently being recorded ~50 years since their last recorded observation (Sutherland et al. 2005). The Ontario distributions of two species (*Aeshna septentrionalis* and *Somatochlora whitehousei)* are apparently restricted to the Hudson Plains Ecozone, while several other species (*Aeshna juncea*, *Somatochlora hudsonica*, and *S. septentrionalis*) have the majority of their Ontario ranges within the ecozone.

The Hudson Plains Ecozone remained relatively undisturbed until recently. Presently, a number of realized or potential mining, hydroelectric, and wind power developments pose threats to the ecozone (see sections 2.2.1.2, *Land Cover Change* and 2.6.1, *Stressors & Cumulative Impacts*). Such abiotic disturbances, as well as biotic disturbances, invariably translate into hydrological disturbances, as this ecozone is considered up to 85% wet lowlands (Section 2.2.1.1, *Overview of Ecozone Structure*). This factor is important, because many of the invertebrates in the Hudson Plains Ecozone have aquatic, semi-aquatic, or riparian stages in their life history. Many species are, for example, associated with the ecozone's highly productive pools and ponds (e.g., aquatic midges and mosquitoes) (Danks and Foottit 1989). Human-facilitated biotic disturbances, such as excessive goose foraging along the Hudson Bay coast (Section 2.2.2.1, *Coastal*), have been shown to negatively impact spider and carabid beetle communities, with degraded salt marshes having lower abundances of both invertebrate groups than undamaged marshes (Milakovic and Jefferies 2003). Moreover, chironomid (midge) species richness decreased from seven species in an undamaged marsh to only one halophytic species in a damaged marsh (Milakovic et al. 2001).

Considering that the Hudson Plains Ecozone spans humid high boreal, low subarctic, and high subarctic climates (ESWG 1995), it might be expected that species composition and diversity will be altered if a shift from subarctic to boreal climates occurs with climate change (Payette et al. 2004). However, until thorough and standardized survey and monitoring programs are in place, the status and trends of invertebrates in the Hudson Plains Ecozone will remain largely speculative.

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2.3.3.7 Vascular plants

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2.3.3.7.1 State of knowledge (inventory)

Plants are more poorly inventoried in the Hudson Plains Ecozone than in most other ecozones in the southern parts of eastern Canada. The most comprehensive catalogue of plants available for this ecozone is that of Riley (2003), who integrated numerous existing published and unpublished data for the ecozone with additional data he collected during 1976-1980, as well as considerable data based on field work in the early 2000s (M.J. Oldham, Ontario Ministry of Natural Resources, pers. comm.). He visited as broad as possible a range of unique and representative habitat types in 11 predetermined collection areas in the Ontario portion of the ecozone. In his catalogue, Riley (2003) reported 915 species of vascular plants (816 native, 98 alien) in the ecozone, which represent about 25% of the vascular plant species occurring in Canada. Gaps in coverage in Riley's catalogue include the northern interior of the ecozone close to the boundary with the Canadian Shield in both Manitoba and Ontario and the Manitoba interior in general but especially between the Hayes River and the Ontario border.

For Manitoba, earliest collections at Churchill and York Factory date from the latter half of the 19th century. Collection increased in the 20th century after the railway from Winnipeg to Churchill was completed in 1931. The flora within a 20 km radius of Churchill has been reasonably well documented (Ritchie 1956; Scoggan 1957, 1959; Johnson 1987; Scott 1996). Plant studies undertaken at La Pérouse Bay in relation to goose foraging, from the mid-1970s to the present, have also provided details of flora and plant communities generally (e.g., Heagy and Cooke 1979; Jefferies et al. 1979; Wolff and Jefferies 1987; Chang et al. 2000; Handa et al. 2002; Srivastava and Jefferies 2002). York Factory has had some inventory work completed in about a 5 km radius by H.J. Scoggan in 1949 and D. Punter in 1969-1970. Detailed studies continue in the Manitoba portion of the ecozone today. Botanical investigation has been undertaken in limited areas on the coast at the mouth of Opoyastin Creek and 80 km east at the estuary of the Kaskattama River. Still, the flora remains poorly known along the coast east of York Factory to the Manitoba-Ontario border. The flora in the hinterland east of the Hayes River remains undocumented. Since 2002, botanical inventories for Wapusk National Park have been undertaken systematically by a research team from the University of Manitoba, for Parks Canada (Punter et al. 2003; Piercey-Normore 2004, 2006; Ford et al. 2005, 2007, 2008, 2009). These species lists and locations have increased knowledge of the flora along the coastline south of Cape Churchill and inland. To date, 315 vascular plant species have been identified in the park. Inventory work is also being completed there to determine biodiversity of each mapped vegetation class.

The history of botanical investigation elsewhere in the mainland portion of the ecozone is reviewed by Riley (2003). A notable series of vegetation investigations were conducted near the Manitoba-Ontario border. Plant associations and ecological processes influencing them were described, including for salt marshes, raised coastal beach ridges, and inter-ridge sedge meadows (e.g., Kershaw and Rouse 1973; Kershaw 1974, 1976). However, similar to the situation in Manitoba, more species tend to be added to the inventory of known plant species whenever new intensive surveys take place (for some recent discoveries, see Inset 10 in Section 2.3.1).

The same is true for the islands in James Bay that are part of the Hudson Plains Ecozone and jurisdictionally part of Nunavut. Riley (1981) provided an initial description of the flora at Akimiski Island (211 species), and Blaney and Kotanen (2000) built upon this work, adding many new species for a total of 273 native species and five alien species (the nearby mainland likely is more diverse). Thirty of these species are rare in the adjacent Ontario (mainland) portions of the ecozone and, in many cases, they represent species respectively near their southern or northern range limits. Blaney and Kotanen (2000) noted, however, that most of the interior of Akimiski Island has still not been investigated.

Thus, although some areas of the Hudson Plains Ecozone have been intensively surveyed for vascular plants, the state of investigation of vascular plants in the ecozone as a whole is, overall, still fairly scant. As such, all existing occurrence and distribution lists for flora in the ecozone must be considered in-progress works. This situation precludes a clear analysis of changes or trends in species gain and loss or distribution at the ecozone scale.

2.3.3.7.2 **Floristic composition**

The relative composition of the known flora of the Hudson Plains Ecozone has been described in terms of its geographical affinity with longitudinal and latitudinal floristic zones (Riley 2003). In reference to longitudinal floristic zones, the flora is 81% transcontinental, 4% western, and 15% eastern (Riley 2003). In reference to latitudinal zones, the flora is comprised of species with arctic, subarctic, and temperate geographic affinities[51], with a higher percentage of arctic and subarctic species than the flora of Canada as a whole (Table 22).

The Manitoba portion of the ecozone hosts a higher percentage of arctic species (22%) than the ecozone as a whole (17%) (Table 22), but the Manitoba portion also represents only the two most northerly of the ecozone's three ecoregions (i.e., Coastal Hudson Bay Lowland (215) and Hudson Bay Lowland (216); ecoregions are shown in Figure 36, Section 2.2.1.1.3). In a separate analysis of the approximately 400 species of the Churchill area (the mostly northerly part of the ecozone), Johnston (1987) reported an even higher proportion of high and low arctic species (36%) than high and low subarctic species (64%). Scott (1996) also described the composition of vascular

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[51] Floristic phytogeography deals with latitudinal and longitudinal zones. Floristic latitudinal zones represented in the ecozone are high and low arctic, high and low subarctic, and temperate. Although much of the ecozone is commonly thought of as *boreal*, that term is vague in this context. The closest synonymy to boreal would be high subarctic and low subarctic, but species of boreal affinity would straddle subarctic (taiga) and temperate (forested boreal) classes, depending on their distribution (J. Riley, The Nature Conservancy of Canada, pers. comm.). Longitudinal zones are western, eastern, and transcontinental (Riley 2003).

flora around Churchill but in three ecosystems (tundra, forest-tundra, and forest): vascular plant species were 67% arctic and 33% subarctic in the tundra, 33% arctic and 67% subarctic in forest-tundra, and 15% arctic and 85% subarctic in forest.

Table 22. Relative proportion of the flora of the Hudson Plains Ecozone by classes of geographic affinity (latitudinal zones). The composition of flora in the Manitoba portion of the ecozone, the province of Manitoba, and Canada are also shown for comparative purposes.
Sources: Riley (2003)[a] and Scoggan (1957)[b].

Geographic affinity	Proportion of flora (%)			
	Hudson Plains Ecozone[a]	Manitoba portion of the ecozone[a]	Province of Manitoba[b]	Canada[a]
High arctic	4	5	10	3
Low arctic	13	17		10
High subarctic	23	52	52	17
Low subarctic	28			23
Temperate	31	26	37	47

This shift to fewer arctic species from north to south across the ecoregions in Manitoba (i.e., gradient in phytogeographic variability) is also evident for the flora of the ecozone as a whole (Riley 2003). In fact, more than half of the ecozone's arctic taxa are restricted to within 20-40 km of Hudson Bay. Species of temperate affinity are more common in the more southerly James Bay Lowland Ecoregion (217) (Riley 2003), which is represented only in Ontario and Québec and the most southern islands in James Bay (Nunavut). Temperate species increase away from the north and coast, towards the southern interior of the ecozone. In contrast, representation from subarctic species varies little across the ecozone (Riley 2003).

Gradients in vascular plant diversity are also suggested at smaller scales, reflecting climatic conditions, coast-to-inland gradients, and changes in topography (Riley 2003). For example, in the Manitoba portion of the ecozone, the number of species appears to decline from the Churchill area (at least 400 species recorded) towards Cape Churchill and south along the east-facing coastline to the Owl River estuary. The coastal beach ridges south of Cape Churchill are exposed to the northeasterly storms, abrasion from ice crystals, and northwest winds due to the treeless terrain and height of ridge tops above the general topography. Inland, large sections of peat plateau complete with ice wedge polygons are dominated by lichen and have limited vascular plant diversity. South of the estuary, the treeline reaches the coast, and it continues to hug the coast to York Factory. About 300 species have been recorded in the vicinity of York Factory (D. Punter, University of Manitoba, pers. comm.). East of York Factory, 146 species have been recorded at Opoyastin Creek and 221 at Kaskattama River at the coast. At both of these locations the treeline is at the coast. Plant diversity appears to increase in the southwestern corner of Wapusk National Park, where a greater number of *boreal* species (see Footnote 51) are present. This increase in plant diversity might also reflect the time since land emergence from the Tyrrell Sea, as the beach ridges within 10 km of the coast are <2,500 years old. The larger rivers flowing to the coast generally have incised channels, often with banks over 9 m high. The plant species on the banks and channel edges are often quite different than on the land above the rivers. *Dryas integrifolia* and *Rhododendron lapponicum*, generally found on the coastal raised beaches, are found within some of the river channels upstream. *Elaeagnus commutata* and *Sisyrhinchium montanum*, two species recorded in the Owl River channel, are at the northern limit of their range in Manitoba. A number of species with arctic affinities are at the southern limit of their range along the coast throughout the ecozone, e.g., *Dryas integrifolia, Bartsia alpina, Arctagrostis latifolia, Cerastium alpinum, Loiseleuria procumbens,* and *Rhododendron lapponicum* (Riley 2003).

Thirty-five percent of the native vascular plant species in the ecozone are widespread and dominate volumetrically; they include five tree species (balsam poplar, *Populus balsamifera*; trembling aspen, *Populus tremuloides*; eastern larch, *Larix laricina*; black spruce, *Picea mariana*; and white spruce, *Picea glauca*) and over 40 shrub species (Riley 2003). The vascular flora contains no species of conservation concern nationally under the Species at Risk Act (SARA) (Government of Canada 2010) or by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2010; see also Section 2.3.2, *Trends in Species of National Conservation Concern*). A number of native species can, however, be considered regionally rare in this ecozone. For example, Riley (2003) considered almost 23% of the inventoried native vascular plant species to be rare (defined as being known from three or less sites), the bulk of which are found in the southern portion of the ecozone. As well, about 50 of the ecozone's taxa of vascular plants (30 of them coastal) have disjunct populations that are over 500 km from the nearest regional populations. Owing to its continentally anomalous habitats, the Hudson Plains Ecozone acts as a distributional bottleneck for the eastward and westward migration of many plant species and, as such, a disproportionate number of species reach their range limits there (Riley 2003).

The Hudson Plains Ecozone, like other northern ecozones in Canada, also has relatively few introduced vascular plant species compared to more southerly ecozones (e.g., CFIA 2008)[52]. The very minor non-native flora introduced into this ecozone has been attributed to the absence of modern large-scale development, a small human population (Section 1.2.1, *Settlement History*), and dependence on air and barge transportation (Riley 2003). Riley (2003) lists 71 species introduced widely into the Hudson Plains Ecozone and 98 alien species in total, all of which can be considered only locally invasive. The alien plant species that have persisted in the ecozone are still localized around the few villages and other areas with most human activity (Staniforth and Scott 1991; Riley 2003). Notably, 28% of the introduced taxa are known only from the Churchill area, and most of those species are attributable to the railhead and port-shipping of grain in the area (Riley 2003), but their establishment can be facilitated by human disturbances (Inset 12). Conversely, the relatively harsh climate of the Hudson Plains Ecozone might help limit the establishment, as well as the persistence, of some non-native flora.

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[52] Assessments of introduced plant species vary, but relative comparisons among ecozones (i.e., within individual studies) are valid.

Inset 12. **Introduced plants in the Churchill area**

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The area around Churchill has had high exposure to introduced and potentially invasive plant species, since the early 1930s, when a route for their introduction was established through completion of the Hudson Bay Railway (connecting Churchill to Winnipeg) and the Port of Churchill, and associated shipping of grain commenced in the area. The level of construction, disturbance, and soil movement around the Churchill town site, the Churchill Rocket Research Range, Fort Churchill, and the Goose Creek Weir site (see Section 2.6.3, *Restoration*) also created a great deal of opportunity for their establishment. In addition, relatively recent restoration of the Churchill open landfill site (closed in 2004), using spent grain tailings and sand and gravel extracted from nearby borrow pits, has the potential to move seed and vegetative material within the region. Such problems are likely to be exacerbated by the longer growing- and shipping-seasons expected with climate change.

Ecotourism is another activity that could facilitate establishment of introduced plants in the Churchill area. Since the advent of organized polar bear (*Ursus maritimus*) and bird viewing activities, increased traffic has occurred along the main roadway to both the Churchill Northern Studies Centre and the bear-viewing sites. Many smaller trails are used by wheeled vehicles in the Churchill Wildlife Management Area and, although most of the bear-viewing activity occurs in the fall when the ground is partially frozen, sometimes in the early shoulder season open peat scars increase or expand as traffic increases, and vehicles attempt to skirt wet patches around ponds and lakes (Smith et al. 1998). This disturbed soil creates potential habitat for the introduction of new species in the local region, more of which might become established with a warmer climate in the future (Government of Canada 2008).

That being said, not all alien plant species that become established are necessarily self-sustaining over time. Staniforth and Scott (1991) compared weed populations found along roads within an area defined by “Churchill River in the west, the Twin Lakes Road in the east, the coastline of Hudson Bay in the north, and the limitations of the road system in the south” in 1989 with those reported earlier for the area by Beckett (1959). While the number of alien plant species recorded increased over the 30 years, from 70 species in 1959 to 81 species in 1989 (attributed to an increased and diversified seed supply and possibly also a recent amelioration of the climate), only 52 species were common to both surveys (i.e., 18 species reported in 1959 were no longer found, while 29 new species were recorded). The authors observed that several older disturbed areas that had since been abandoned had regenerated back to native species to the extent that very few or no alien species were still present. As well, no alien species were found in natural plant communities that had no history of human disturbance (Staniforth and Scott 1991).

In total, 106 different alien plant species had been recorded for the Churchill study area by 1989 (81 species in the 1980 survey, plus 18 non-persistent species from the 1959 survey, plus seven additional species reported by others) (Staniforth and Scott 1991). In comparison, David Punter (University of Manitoba, pers. comm.) lists only seven alien species for York Factory, with several more reported but unconfirmed with specimens (2009). The York Factory tally might seem low given that there was livestock husbandry at the site until it closed, but again not unlike results from abandoned sites in Churchill.

2.3.3.7.3 **Species profiles**

Eelgrass (*Zostera marina*), Lepage wild flax (*Linum lewisii* var. *lepagei*), nard sedge (*Carex nardina*), and white mountain-avens (*Dryas integrifolia*) are some of the vascular plant species of special interest to the Hudson Plains Ecozone and are profiled below. Although eelgrass can also be considered part of the flora of the interfacing marine Hudson Bay, James Bay, and Foxe Basin Ecozone (Niemi et al. 2010), it is also an important component of the coastal biome of the Hudson Plains Ecozone (Section 2.2.2.1, *Coastal*), and it is formally recognized as part of this ecozone's flora (Riley 2003).

Eelgrass

James Bay is unusual in Canada's arctic marine regions in having rich eelgrass (*Zostera marina*) beds (Stewart and Lockhart 2005). Eelgrass is an aquatic plant that occurs in subtidal beds in warm shallow waters with fine sediments, low tidal range, and moderate to high salinity. It plays an important role in near-shore marine ecosystems by providing feeding grounds and nurseries for coastal fish species and invertebrates and forage for Atlantic brant (*Branta bernicla hrota*), Canada geese (*Branta canadensis*), and ducks. In the Hudson Plains Ecozone, eelgrass is an important component of the coastal biome along the Québec coast in eastern James Bay, where it has been found in large concentrations (Curtis 1973; Dignard et al. 1991; Ettinger et al. 1995). Recent evidence suggests a dramatic decline in eelgrass abundance there, warranting concern and special mention. Eelgrass presence is extremely limited and patchy elsewhere in the ecozone, along the Hudson Bay and James Bay coasts from Manitoba to Attawapiskat, Ontario (Stewart and Lockhart 2005). In Manitoba, Riley (2003) has eelgrass mapped in the Churchill area and Seal River based on historical records by Ritchie and Scoggan; Manitoba Museum records reference it as "washed ashore in intertidal pools" and "on tidal flats" (K. Johnson, Manitoba Museum, pers. comm.). No recent survey of this area has been completed to determine whether those beds are still present. Manitoba per se is not listed as a location for eelgrass in the Flora of North America (FNAEC 2000a), probably because the eelgrass meadows there are subtidal and, therefore, jurisdictionally part of Nunavut. There are no records of eelgrass at the mouths of the Churchill and Nelson rivers. It has been speculated that the absence of eelgrass there is due to high wave activity and ice scour and/or low salinity. In Ontario, eelgrass has been recorded at Moose River, Longridge Point, and North Point.

The best status and trends information for eelgrass in eastern James Bay comes from a monitoring study associated with hydroelectric development in the area. Hydro-Québec and Génivar Inc. (2001) documented a sudden decline in the abundance and biomass of eelgrass (measured in 1999 and 2000) following a period of relatively uniform abundance (measured 1988-1995). Although variable among six monitoring stations, dry leaf biomass at 1.5 m depth declined from 31-548 g/m^2 to 0-63 g/m^2 between 1995 and 1999, and shoots declined from 106-948/m^2 to 0-155/m^2. Short (2008) conducted more extensive sampling in eastern James Bay in 2006, and he found the decline to be of greater magnitude than previously thought. Coincident with the decline in eastern James Bay, in the spring of 1996 evidence of high mortality was noted on the north shore of Akimiski Island, Nunavut (K. Abraham, Ontario Ministry of Natural Resources, unpublished), as eelgrass litter was widespread along the north shore, festooned in willows at the upper end of the supratidal marshes. The same observation has not been made on Akimiski Island in other springs, despite annual field work of the same type over the same areas since then.

Hydro-Québec and Génivar Consulting Group (2001) suggested that the decline in eelgrass was likely attributable to a wasting disease caused by the protist *Labyrinthula zosterae* (Short et al. 1988). However, factors such as rising water temperature, increased currents (turbidity), and lower salinity associated with the hydroelectric development (increased freshwater outputs from the La Grande River) have been pointed to as contributing factors (Parliament of Canada 2008; Short 2008). Reduced salinity during the major growing period (June and July) and increased duration of ice cover related to reduced salinity were suggested as the major causes by Short (2008), who rejected wasting disease, climate change, and isostatic rebound as major causes.

On the James Bay coast, a decline in eelgrass equates to a loss of forage for Atlantic brant, Canada geese, and ducks, as well as reduced habitat quality for coastal fish species and invertebrates. Elsewhere in Canada, local jurisdictions are intent on halting declines through local stewardship, restoration, and mapping initiatives (e.g., Hanson 2004; SCWG 2004; Hessing-Lewis 2005).

Lepage wild flax

Lepage wild flax (*Linum lewisii* var. *lepagei*) (Figure 94) is considered endemic to the Hudson Bay and James Bay coasts and one of the few Canadian post-glacial endemics (Riley 2003). Lepage wild flax occurs along the Hudson Bay coast from the Churchill area in Manitoba east to Moosonee in Ontario. It is a perennial herb with white flowers up to 2.5 cm wide that last just one day, with petals falling in the afternoon. It grows in sand along with sea lyme grass (*Leymus mollis*) on beach ridges close to the high tide mark. Little is known about its population numbers or trends. It could be at risk where wheeled vehicles use shoreline beach ridges. Storm surges, as well as any sea level rise from climate change[53], might also disrupt the integrity of these beach ridges and, thus, reduce or eliminate the population.

Figure 94. Lepage wild flax, Wapusk National Park, Manitoba.
Photo credit: B.A. Ford, University of Manitoba.

Nard sedge

Nard sedge (*Carex nardina*) could be at great risk from climate change. This short-tufted sedge has strong arctic-alpine affinities; its range in Canada is generally north of 60 °N latitude and in mountains at high elevations (FNAEC 2000b). In the Hudson Plains Ecozone (slightly south of 60 °N), nard sedge is restricted to the ecozone's narrow band of tundra coast (Riley 2003), where it can form the dominant plant on upper calcareous sand and gravel surfaces of raised beach ridges. It

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[53] Sea level rise is less of a concern for this ecozone than for some other coastal areas, because it may not overtake the ecozone's especially high rate of isostatic rebound (see Section 2.4.1, *Coastal Building Processes*, and references therein). However, longer ice-free seasons in Hudson and James bays will likely render the coast susceptible to increased storm surges (inundation).

is found more specifically on the inland (western-most) treeless raised beach ridges in Manitoba and at Cape Henrietta Maria, Ontario (it is a provincially rare species in both Manitoba and Ontario) (Riley 2003). As suggested by the Arctic Climate Impact Assessment (ACIA 2005), such plant species of arctic affinity that occupy restricted local distributions could disappear at lower latitudes, wherever the tundra biome is narrow, as it is in the Hudson Plains Ecozone (see also Section 2.2.2.2, *Polar-Tundra*). The ecozone's scattered populations of nard sedge are not currently being monitored, and this species is often confused with other closely related species (FNAEC 2000b).

White mountain-avens

White mountain-avens (*Dryas integrifolia*) (Figure 95) is a species both sensitive to disturbance and for which changes in population and range might indicate a change in climate. This dwarf shrub is a dominant pioneer plant on the coastal beach ridges, where it often grows in a cushion-shaped form on calcareous gravel and sand substrate on top of the ridges. Its range in the Manitoba portion of the ecozone (where it is listed as S3) is about a 20 km band along the coastline, where beach ridges are present. It occurs similarly all along the coast in Ontario (where it is listed as S4), especially in the area of Cape Henrietta Maria and on Akimiski Island, Nunavut (Blaney and Kotanen 2000; Riley 2003). The cushions of white mountain-avens trap sand, organic matter, seeds, lichen, and moss fragments blown by the wind, forming a base and shelter in which other plants can become established on the exposed beach ridges. White mountain-avens becomes the dominant plant on inland raised beach ridges, until other plants become established with time.

Figure 95. White mountain-avens, Wapusk National Park, Manitoba.
Photo credit: D. Punter, University of Manitoba.

Similar to the situation with both LePage wild flax and nard sedge, little is known about the population numbers or trends of white mountain-avens. Although it is a pioneer species, white mountain-avens is sensitive to disturbance, and fragmentation of the cover of this species could result in bare ground. The deep imprints sometimes observed in mountain-avens are usually the result of one or more passes of wheeled vehicles, which also loosen the plant from its substrate and expose its roots to desiccation (see also the discussion of tundra buggy/ATV damage in Section 2.2.2.2, *Polar-Tundra*). Damage to the plant from desiccation can be severe if temperatures are high in summer or snow cover is thin in winter. Although caribou (*Rangifer tarandus*) trails also occur on the tops of raised beach ridges, most plants, including white mountain-avens, appear unable to colonize caribou tracks, perhaps due to the trails compacting, deepening, and starting to erode.

Given that white mountain-avens is another arctic species with a restricted local distribution in the ecozone (Riley 2003), changes in its population or range could also be indicative of a change in climate. For example, as shrubs (willow, *Salix* sp.) encroach and increasingly shade beach ridges, more snow is likely to be trapped. The increased moisture might further limit white mountain-avens, which is adapted to drier conditions. White mountain-avens is also one of a group of spring-flowering indicator plants now being used by PlantWatch North in the Churchill area to monitor changes in the onset of spring associated with climate change (Section 2.4.3.4.2, *Plant Phenology*).

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2.3.3.8 **Lichens**

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Lichens are well-adapted to northern ecosystems (Kershaw 1985; Longton 1988). Whereas few plants can take root in the permafrost, lichens are adapted to absorb nutrients from the atmosphere or the superficial layers of a substrate. Lichens also generally undergo photosynthesis at lower temperatures than many plants and are more metabolically active in the cooler parts of the day and the seasons. Lichens have a poikilohydric capacity to become dormant in the dry times of the day and season, and they become active again as moisture becomes available, allowing them to be highly successful in terms of coverage in the northern regions.

Much study of lichens has been done in northern ecosystems (Bird 1975; Bird et al. 1980, 1981; Robinson et al. 1989; Ahti and Oksanen 1990; Bliss and Gold 1993; Pristyazhnyuk 1996), the Ontario portion of the Hudson Plains Ecozone (Ahti 1964; Kershaw and Rouse 1973; Neal and Kershaw 1973a,b; Rouse and Kershaw 1973; Larson and Kershaw 1974; Kershaw 1975a,b; Larson and Kershaw 1975a,b), the Churchill region (Ritchie 1957; Thomson 1972; Scott 1996), and the forested portion of the Manitoba part of the Hudson Plains Ecozone (Ritchie 1960a,b). Still, knowledge about lichens in the Hudson Plains Ecozone remains fairly scant overall, and new intensive surveys typically reveal lichen species not previously recorded. For example, previous to a recent inventory project with Parks Canada, the only lichens reported for Wapusk National Park were 15 species identified by Brook (2001), as part of a ground-truthing survey for landscape classification. Since that time, Parks Canada invested in a botanical survey (2002-present), which has reported over 200 species of lichens for the park and new range extensions for Manitoba (Punter et al. 2003; Piercey-Normore et al. 2004, 2006; Ford et al. 2005, 2007, 2008, 2009; Piercey-Normore 2005, 2006, 2008, 2010). Additional ecological projects have been conducted in the area by Elliot (2008, 2009) and by Cassie and Piercey-Normore (2008), which included lichens and bryophytes (Cassie et al. 2008). Recent studies in Ontario are likewise revealing lichen species not previously reported in that portion of the ecozone (see Inset 10 in Section 2.3.1).

The diversity of lichen species native to the Hudson Plains Ecozone is not well known and cannot be compared to other regions, because the same extent of survey has not been done. However, many species recorded in the Hudson Plains Ecozone are also present in other arctic, subarctic, and temperate areas. Some species extend their range from eastern North America into the Manitoba portion of the Hudson Plains Ecozone, such as *Hypogymnia tubulosa, Platismatia glauca, Multiclavula vernalis,* and *Cladonia wainioi*. Although lichen diversity may not be particularly high in the Hudson Plains Ecozone, lichen coverage is high. Lichen coverage is particularly high in inland areas of Wapusk National Park.

Lichens occur across a range of ecosystem types in the Hudson Plains Ecozone, as illustrated by the relatively recent and detailed studies at Wapusk National Park (Punter et al. 2003; Piercey-Normore et al. 2004, 2006; Ford et al. 2005, 2007, 2008, 2009; Piercey-Normore 2005, 2006, 2008, 2010). Much of the park consists of limestone beach ridges that are exposed near the coast but are covered by more acidic peat plateaus and polygons in the interior regions. The substrate is an important determinant of colonization ability for many lichens; in the Hudson Plains Ecozone, the occurrence of glacially derived granite boulders provides a different source of nutrients than the limestone substrate of this predominantly calcareous environment, which allows the growth of acidiophilic lichens. The forest below the treeline in the interior and more southerly parts of the park also provides habitat for additional lichen species. A large number of species with arctic, subarctic, and temperate geographic affinities are present in these forests. The isolated white spruce patches scattered on the beach ridges also provide refuge for forest species and some species not common in the forests of the park. The fens in the low-lying areas between the beach ridges are dominated by mosses and sedges, with some lichens growing on the surface of raised *Sphagnum* hummocks along the edges of the fens. River banks have very low numbers of lichens, but the temporary pools on the surface of the peat plateaus have species of lichens specialized for growing along the edges of the pools and other wet areas.

The lichen studies at Wapusk National Park also illustrate how the specific complement of lichen species varies across habitat types in the ecozone (Piercey-Normore 2005, 2006, 2010). Raised beach ridges contain fragments and small individuals of fruticose lichens, such as *Thamnolia vermicularis* or *T. subuliformis*. Fragments represent an important form of dispersal, especially for sterile lichens. Fragments and small tufts of *Flavocetraria nivalis, Alectoria ochroleuca, A. nigricans,* and *Bryocaulon divergens* are also present. The ground-dwelling form of *Evernia divaricata* is present in small numbers. *Physconia muscigena*, a foliose lichen with frosted lobes, is common on the soil mixed with mosses. The calcareous pebbles of the ridges contain *Sarcogyne irregularis, Caloplaca* spp., an orange-appressed foliose lichen, *Xanthoria elegans,* and two endolithic crustose species, a black apothecial and a black perithecial lichen.

The coast in the southern portion of the park contains several jelly lichens, *Leptogium* spp., *Collema* spp., and free-living *Nostoc*. By contrast, the fens between beaches contain more mosses than lichens, except on the top of the dry hummocks and on shrubs. Lichens on the hummocks include *Cladonia pocillum, Lecanora epibryon, Ochrolechia androgyna,* and *Sphaerophorus globosus*. Lichens on dwarf birch and willows in the fen areas are *Caloplaca* spp.*, Evernia mesomorpha, Hypogymnia physodes, Lecanora circumborealis, Parmelia sulcata, Parmeliopsis ambigua, Physcia alnophila, Tuckermannopsis americana, T. sepincola,* and *Vulpicida pinastri.*

The isolated patches of white spruce on the coastal beach ridges provide habitat for some species such as *Alectoria nigricans, Protopannaria pezizoides, Ramalina farinacea, R. roesleri,* and *Tuckermannopsis chlorophylla* (Piercey-Normore 2008) that are absent from exposed coastal habitats. *R. farinacea* has not been recorded elsewhere in the park, suggesting that the *islands* may provide a specific microhabitat to support this species.

The interior peat plateaus have higher ground coverage of lichens but similar diversity to the coastal regions. The species present on the peat plateaus are different from those near the coast, but some of the interior species are represented by poorly growing specimens near the coast. These peat plateaus are dominated by the reindeer lichens (*Cladonia arbuscula, C. stellaris, C. stygia*), as well as the angel hair lichen (*Alectoria ochroleuca*) and dark coloured lichens (*Bryoria nitidula, Bryocaulon divergens, Cetraria islandica*). The dominant crustose lichen on the peat plateaus is the candy lichen (*Icmadophila ericetorum*), which is absent from the coastal beach ridges.

Lichens specialized to grow on granite are rare in the park and presumably elsewhere in the ecozone, because the substrate is rare. Some of these are *Arctoparmelia centrifuga, Lecanora* spp., *Melanelia disjuncta, M. commixa, M. hepatazon, Parmelia saxatilis*, *Physcia phaea, P. caesia, Pseudephebe minuscule, Rhizocarpon* spp., *Umbilicaria polyphylla,* and *U. hyperborea*.

Burned areas have a number of crustose lichens, such as *Cladonia acuminata, Placynthiella uliginosa,* and *Trapeliopsis granulosa,* on old wood and early successional fruticose lichens, such as members of the red-fruited section of *Cladonia* (*C. borealis, C. coccifera, C. pleurota, C. sulphurina*). These lichens are generally absent from unburned locations, except perhaps in disturbed patches that support early successional species. In the early part of the summer, the burned polygons contain abundant fruiting Basidiomycete lichens (*Lichenomphalia ericetorum, L. hudsoniana*). Temporary pools on the surface of the polygons have a conspicuous brown margin containing *Cladonia subfurcata, Cetrariella delisei*, and *Arctocetraria andrejevii*, which grow only on this habitat and other moist depressions on the interior plateaus and polygons.

The southern part of Wapusk National Park has forests that provide canopy cover, and it contains a larger number of lichen species with temperate geographic affinities than the northern exposed areas. Some of these species include *Cladonia phyllophora, C. cornuta, C. scabruiscula, C. subulata, C uncialis, Imshaugia aleurites, Stereocaulon tomentosum, Usnea lapponica*, and *U. cavernosa*.

The diversity of secondary metabolites in the genus *Cladonia* has been compared between Wapusk National Park and other areas within Manitoba (Piercey-Normore 2003, 2006, 2007), although it is difficult to make comparisons, because there are so few studies on secondary metabolite diversity. Although fewer samples were examined from the park, the diversity there was lower than that reported for other parts of Manitoba. However, further study might be warranted, because the secondary metabolites have implications for caribou feeding and digestion (Palo 1993), snail herbivory (Gauslaa 2005), the absorption of light on open polygons and beach ridges (Larson and Kershaw 1975a; Rikkinen 1995), and adaptation for climate change.

2.3.3.8.1 **Species profiles**

Reindeer or caribou lichens (*Cladonia rangiferina, C. stygia, C. arbuscula, C. stellaris, C. wainioi*) and white bone lichen (*Thamnolia subuliformis*) are highlighted below as species of special interest in the Hudson Plains Ecozone.

Reindeer/caribou lichens

In the Hudson Plains Ecozone, reindeer/caribou lichens (*Cladonia rangiferina, C. stygia, C. arbuscula, C. stellaris, C. wainioi*; Figure 96) form high cover in the open woodlands and on the peat plateaus (Figure 97). They are not rare within the ecozone, but they are highlighted because of the extent of coverage of these species on the peat plateaus, which is characteristic of this ecozone. Because of their light colour, they reflect light and therefore heat, which helps prevent thawing of permafrost; and they also intercept much of the moisture and nutrients from precipitation, hence out-competing many vascular plants that might be able to colonize these areas (Rikkinen 1995).

Reindeer/caribou lichens have high ecological significance in the ecozone, for both large herbivores and indirectly for top predators. More specifically, they form an integral part of the food chain by providing critical winter diet for caribou (*Rangifer tarandus* ssp.) (Scotter 1967; Campbell 1994); caribou are, in turn, ecologically important to grey wolves (*Canis lupus*) and

Figure 96. Some reindeer/caribou lichens of the Hudson Plains Ecozone: a) Cladonia stygia*; b)* C. stellaris*; and c)* C. wainioi*.*
Photo credits: M.D. Piercey-Normore, University of Manitoba.

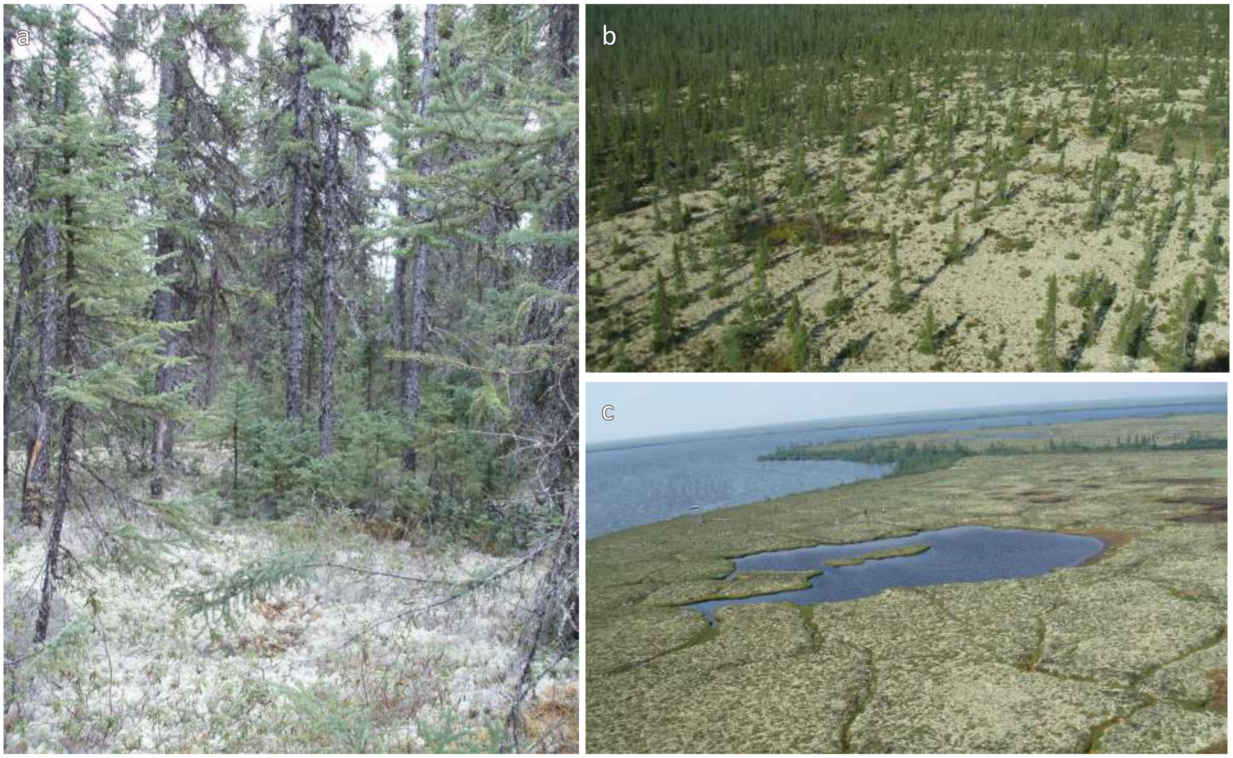

Figure 97. Examples of woodland or peatland habitats in the Hudson Plains Ecozone that are dominated by reindeer/caribou lichens: a) old woodland; b) open woodland; and c) open polygons.
Photo credits: M.D. Piercey-Normore, University of Manitoba.

wolverines (*Gulo gulo*) and culturally important to the ecozone's Aboriginal peoples. Reindeer/caribou lichens are also thought to be one of the indicators of den choice for polar bears (*Ursus maritimus*) (Richardson et al. 2005). As well, these lichens have some economic significance elsewhere as important sources of a well-studied medicinal compound (Usnic acid) and its induction by UV light (Nybakken and Julkunen-Tiitto 2006). They are also of some international significance, because of their rapid decline in Alaska from climate change, wildfires, and disturbance by caribou (Joly et al. 2009).

No information is available on changes or trends in the distribution and abundance of reindeer/caribou lichens within the Hudson Plains Ecozone. Overexploitation and trampling of ground lichens by caribou is, however, considered a possible contributing (but unexplored) explanation for the recent decline or distributional change in the Pen Islands caribou herd near the Manitoba-Ontario border (Section 2.4.3.2, *Herbivore-Plant Interactions*; and see also Section 2.3.3.2.1, *Caribou*). As noted above, climate change is also a potential threat (Joly et al. 2009).

The most common species of reindeer/caribou lichen in the Hudson Plains Ecozone is *C. arbuscula*. Information is beginning to accumulate on this species but mostly on its taxonomy (Ahti 2000; Myllys et al. 2003; Piercey-Normore et al. 2010) and on gene flow (Robertson and Piercey-Normore 2007; Kotelko et al. 2008).

White bone lichen

White bone lichen (*Thamnolia subuliformis*) is an arctic/alpine species that grows only on open beach ridges and surrounding areas. It is considered to be a sterile lichen, because no known sexual phase has been identified, yet it is highly successful and widely distributed in these habitats throughout the world. This lichen usually grows as upright white stalks on the ground, usually among rocks and small plants (see photo on p 678 of Brodo et al. 2001). In Wapusk National Park, it appears as three or four stalks or branches together (Figure 98), distributed among the pebbles and bryophytes on the surface of the beach ridges.

No information is available on changes or trends in the distribution and abundance of this species in the Hudson Plains Ecozone. The dry beach ridges where it occurs are, however, commonly used by people and animals as conduits between locations and receive high traffic. Increased use of these habitats could have a damaging effect on its survival and extent.

Figure 98. Branched fragment of white bone lichen. Photo credit: M.D. Piercey-Normore, University of Manitoba.

Also noteworthy with respect to white bone lichen is that it is one of only two species in the genus *Thamnolia*, both species of which are distributed world-wide but only in arctic/antarctic and alpine areas. *T. subuliformis* is more common in the northern hemisphere, and it is rare in the southern hemisphere. The other chemical species, *T. vermicularis* (also white bone lichen) has the opposite distribution pattern – it is common in the antarctic, and it is rare in the northern hemisphere. Both species are found in Hudson Plains Ecozone, but *T. vermicularis* is very rare. Some research has been conducted on the phylogenetics and clonality of the genus (Nelsen and Gargas 2009) and the population genetics of *T. subuliformis* in Wapusk National Park (Cassie and Piercey-Normore 2008). *T. vermicularis* was shown to have low levels of historic recombination and very little genetic variation (Nelsen and Gargas 2009). Similarly, *T. subuliformis* was shown to have low levels of variation and low levels of dispersal, with similar interpretation of recombination events (Cassie and Piercey-Normore 2008). Both are thought to be *chemical species*, suggesting that they are the same species but with two chemical variants. The question of whether environmental conditions influence production of the secondary compounds remains to be addressed.

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2.4 **Ecosystem functions/processes**

2.4.1 **Coastal building processes**

I. Peter Martini, University of Guelph

The modern and ancient uplifted coasts of the Hudson Plains Ecozone have developed along a cold marginal sea (Hudson and James bays) of a glaciated continent affected by uppermost Pleistocene-Holocene postglacial isostatic rebound. The processes involved in building these coasts and the resulting morphologies have remained the same throughout this time. However, their effectiveness and rate of development may have changed, owing to the presence of remnant glaciers that would have imparted a glacial climate during the early stages of the development of the inner part of the ecozone. Other varying influences include subsequent changes to climatic conditions and the progressive decrease in the rate of postglacial isostatic rebound (and hence emersion of land), which continues today. Although still not very damaging, direct human activities are increasingly affecting parts of the system, particularly the estuarine reaches and adjacent coasts.

Three principal types of modern coasts exist along the Canadian inland seas (James Bay, Hudson Bay and Foxe Basin), and they are all represented in the Hudson Plains Ecozone (figures 99, 100) (Martini et al. 1980; Martini 1981a, 1982, 1986). The three types are: 1) *rocky coasts* associated with low-lying promontories, where Precambrian metamorphic rocks or Paleozoic carbonates are exposed; 2) *estuarine coasts* associated with the major rivers; and 3) *depositional coasts* that range between two end members: a) coasts with beach ridges in slightly steeper shores exposed to the main bays and b) coasts with sandy and muddy tidal flats, which are generally in embayments or shores protected by partially or totally submerged low-lying ridges.

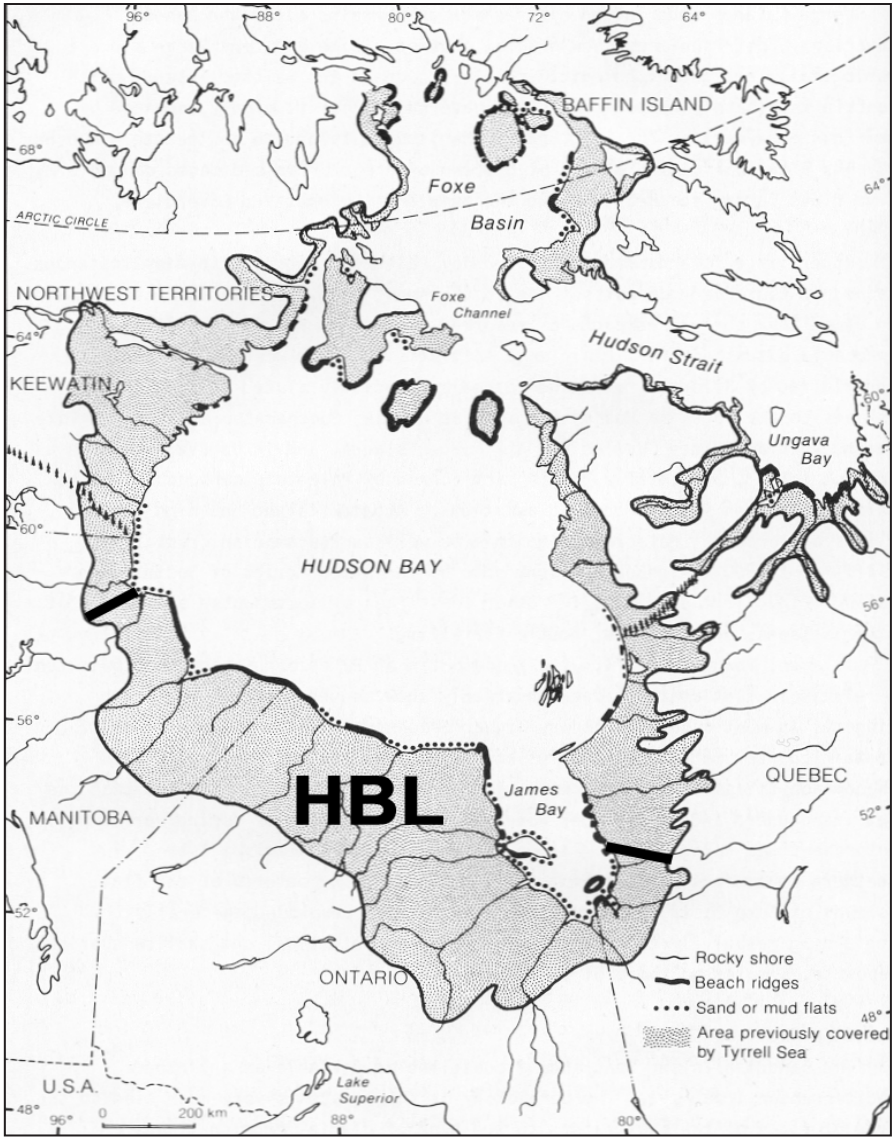

Figure 99. Generalized distribution of the principal types of coasts of Canadian inland seas, including those associated with the Hudson Plains Ecozone. Abbreviation: HBL, Hudson Bay Lowland, as defined by Bostock (1970), which is roughly equivalent to the Hudson Plains Ecozone.
Source: After Martini (1986). Reprinted from Canadian Inland Seas, Elsevier Oceanographic Series, Vol 44, I.P. Martini, Coastal features of Canadian inland seas, page 122, Copyright (1986), with permission from Elsevier.

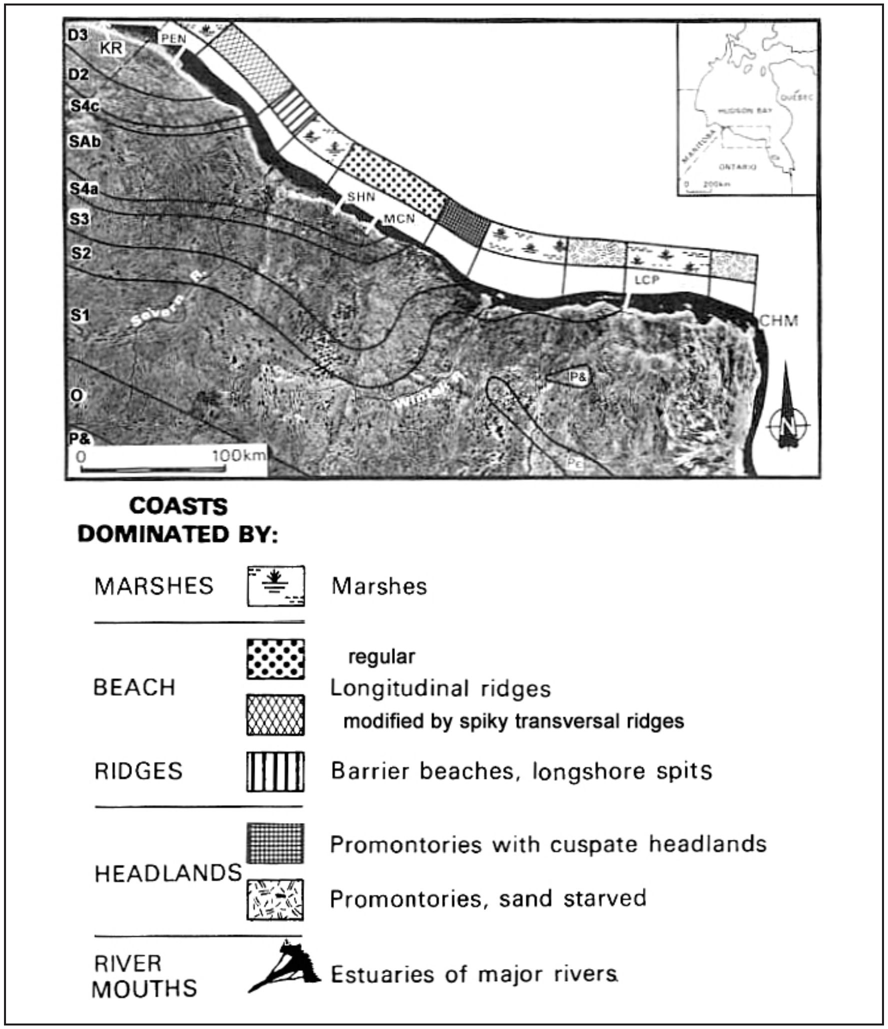

Figure 100. Details of an Ontario portion of the southwest coast of Hudson Bay, showing coastal types and geology. Geological sequence – Devonian: D3 Kwataboahegan Formation and D2 Stooping River Fm; Silurian: S4c Upper Kenogami River Fm, S4b Middle Kenogami Fm, S4a Lower Kenogami Fm, S3 Attawapiskat Fm, S2 Ekwan River Fm, and S1 Severn River F; Ordovician: O; Precambrian: Pε. Other abbreviations: KR, Kettle River; and CHM, Cape Henrietta Maria.
Source: Martini (1982). Reprinted from Le Naturaliste canadien, Vol 109, I.P. Martini, Geomorphological features of the Ontario coast of Hudson Bay, page 417, Copyright (1982), with permission.

2.4.1.1 Primary factors affecting the formation of coastal features

The primary factors affecting the formation of coastal features in the Hudson Plains Ecozone are: substrate materials that constitute a source of coastal sediments; stream water and sediment discharge, particularly from major rivers; wave and tidal regimes of Hudson and James bays; marine currents; ice regime and permafrost; coastal vegetation; and postglacial isostatic rebound. These factors were introduced in sections 2.1, *Abiotic Drivers* and 2.2, *Ecosystem Structure*, and they are further discussed briefly here in relation to coastal and estuarine settings. Major potential trends are noted.

2.4.1.1.1 Substrate materials as a source of coastal sediment

Fine-grained metamorphic rocks or thinly bedded carbonates exposed in rocky coasts provide some flat clasts that are used in forming shingle beach ridges, such as at the Cape Henrietta Maria promontory, which is located at the western transition between Hudson and James bays (Sanford et al. 1968; Morris 1986). However, the largest quantity of sediment reworked in the modern coasts derives from previously deposited glacial materials that range in size from boulders to large amounts of sand, silt, and clay.

2.4.1.1.2 Water & sediment discharge from major rivers

Coastal deposits also derive from river sediments that are transported to the sea, mostly during the spring freshet (floods), which is primarily associated with snow melting and ice break-up (Martini et al. 1993). Gravel and most sand is transported as bed load; silt and some sand are carried in suspension; and a relatively small amount of variously sized material, including gravel, is rafted by ice blocks.

Water and sediment discharge, and thus coastal deposits, can be notably altered by hydroelectric development. For example, after 90% of the flow of the Eastmain River was diverted north to the La Grande River, the downstream current of the reduced flow segment of the Eastmain River no longer expelled fine sediments into James Bay, and its estuary became a sedimentary deposit zone (Hayeur 2001). Additional sediment was contributed by erosion of the exposed river bed.

2.4.1.1.3 Wave & tidal regimes of Hudson and James bays

Wave and tidal regimes have important effects on coastal ecosystems (Grinham and Martini 1983; Prinsenberg and Freeman 1986). The nearshore areas of Hudson and James bays are shallow. During the ice-free season, storm waves affect the more exposed coasts by building coastal ridges in the upper part of the intertidal zone and eroding part of the glacial sedimentary substrate and by washing out finer particles and leaving behind a boulder (glacial erratics) strewn zone. Coastal marshes of various, mostly limited, extension develop inland from the coastal beach ridges. In more sheltered areas, the combined effect of tides (predominantly mesotidal, up to 3 m in excursion) and primarily constructive waves form locally extensive, sandy to muddy tidal flats (Martini 1991). It is inland from these flats that extensive coastal marshes develop (Martini 2006a,b).

2.4.1.1.4 Marine currents

Two types of marine currents influence the coastal environments of the Hudson Plains Ecozone. One is an anticlockwise geostrophic marine current (Figure 101) that redistributes material accordingly and, in doing so, affects the morphology of coastal features, such as spit-like beach ridges and much of the sedimentation of the coastal zone. The second type of marine current is driven by wave refraction. It may lead to erosion from local promontories and accumulation of sediments in adjacent embayments. The interaction of the marine currents with narrow, long promontories jutting out into the shallow shelf leads to characteristically wide, sandy tidal flats (Martini 1991) in the up-current side and erosion of the intertidal zone in the down-current side, where curved beach ridges link the promontory to the mainland coast (Figure 102). These ridges generate characteristic typical *chevron beach ridge* landscapes that can also be clearly recognized in the inland morphology of the ecozone, where they become progressively mantled by vegetation and eventually peat (Martini 2006a,b).

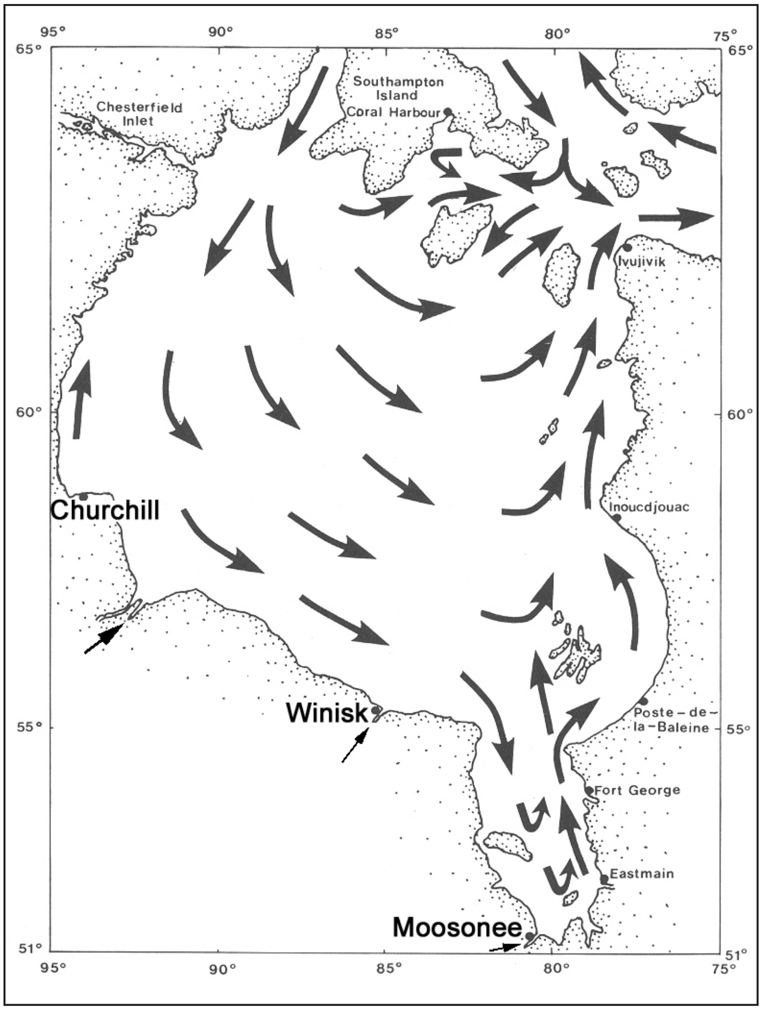

Figure 101. Anticlockwise flowing surficial, geostrophic current in Hudson and James bays. Note: A flood destroyed the community of Winisk in 1986; it was subsequently rebuilt ~40 km further inland and renamed Peawanuck.
Source: Prinsenberg (1986). Reprinted from Canadian Inland Seas, Elsevier Oceanographic Series, Vol 44, S.J. Prinsenberg, The circulation pattern and current structure of Hudson Bay, page 202, Copyright (1986), with permission from Elsevier.

2.4.1.1.5 **Ice regime & permafrost**

The coast and estuarine reaches of the Hudson Plains Ecozone front cold inland seas that are refrigerated by arctic waters and occur primarily in the subarctic climatic zone. They are, therefore, greatly affected by sea ice, river ice, and frozen ground conditions.

The coast is covered by sea ice for about 6 months of the year. The presence of sea ice completely changes the shore morphology and the effectiveness of the processes that act on it for part of the year. Only a few, limited open-water sites are retained during the winter, e.g., the polynya (quasi ice-free area) between the mainland and Akimiski Island (Martini 1981b). During winter, the cover of sea ice protects the coast from erosion and sedimentation. During break-up in late spring/early summer, however, ice blocks are lifted, pushed, and dragged by tides. These actions lead to intense erosion in various ways. When ice blocks that are frozen to the ground are lifted by flood tides, they remove chunks of substrate, most commonly from

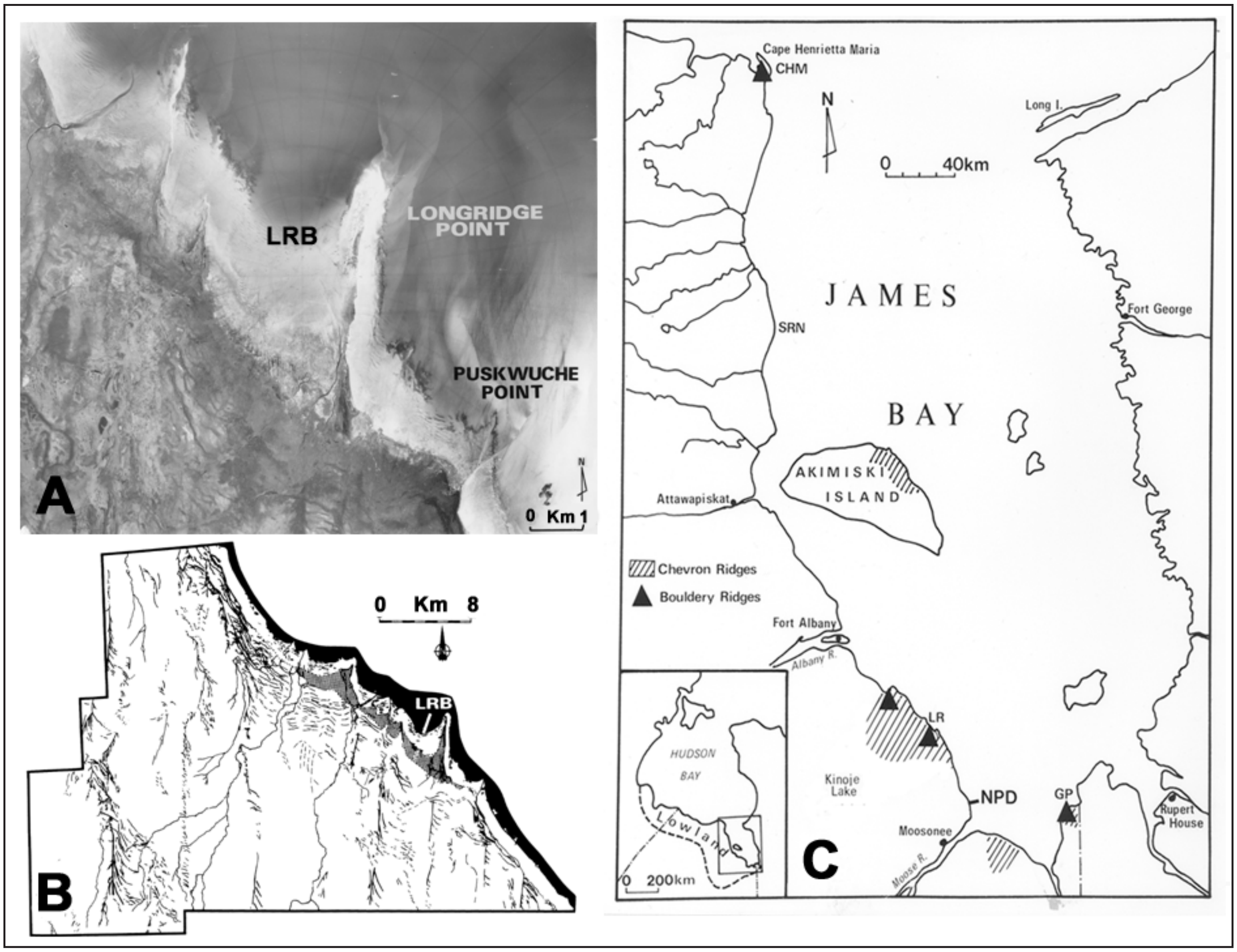

Figure 102. Chevron ridges: a characteristic coastal landscape generated through interaction of a geostrophic longshore current and onshore wave movements along the west coasts of James Bay.
a) Wide sandy tidal flats north of Long Ridge (LRB) promontory (southwest coast of James Bay) and erosional intertidal area and curved coastal beach ridges south of it, forming, together with the narrow promontory, chevron ridge features. b) Long Ridge (LRB) area showing inland chevron ridges that mimic present coastal characteristics. c) Areas along the western coast of James Bay with fairly well-developed chevron ridges, which are readily visible toward the present coast and are progressively covered by peat and forest inland.
Sources: a) Adapted from Martini et al. (1980), with permission of Natural Resources Canada, courtesy of the Geological Survey of Canada; and b) Martini (1981a), reprinted from Geographiska Annaler Series A, Physical Geography, Vol 63, I.P. Martini, Morphology and sediments of the emergent Ontario coast of James Bay, page 85, Copyright (1981), with permission from John Wiley & Sons Ltd.

the uppermost parts of the tidal flats and from the lower marshes. The result is a jigsaw pattern of ponds in the marshes, which is typical of these cold coasts. Waves and tides can push and move ice blocks around. Thick blocks can erode the substrate by furrowing, and thinner, wider, flatter blocks that are dragged along can peel off the top part of the middle and upper sandy tidal flats, thereby disturbing or destroying its infauna (via ice scour). Large ice blocks can also obstruct tidal flows and waves, which can lead to erosion of the substrate, with infauna being dumped on the blocks themselves. This material can eventually be rafted to other parts of the coast. Finally, ice formation can lead to (slight) changes in salinity in different parts of coastal marshes. Increases in salinity can occur due to expulsion of salt during freeze-up, and decreases can occur due to freshwater input during melting. All of this could have a significant impact on the infauna and, therefore, on the foodstuff and quality of staging sites of migratory animals, such as shorebirds.

River ice cover persists for many months, generally from the late fall to late spring/early summer. It varies in duration in rivers located at different latitudes and in different reaches within each of the northern flowing rivers. Freeze-up occurs later and break-up earlier in warmer southern reaches. During winter, shallower parts of the river are protected by freezing, whereas unfrozen deeper parts of the channels might instead experience some erosion and sediment movement. Given the natural reduction in water flow in winter, changes in sediment movement might not be of major importance. This sediment movement regime is, however, somewhat increased in rivers where water flow is regulated (augmented during winter) by hydroelectric stations or permanently reduced by river diversions.

During the period of river ice break-up, erosion and sediment transport and sedimentation are greatly increased. Erosion can occur due to the increased power of pipe-like stream flow, which can develop under the still solid ice cover at the start of significant annual snow melting. During the subsequent break-up, ice blocks frozen to the bottom sediments can be lifted; ice jams can develop locally, which create ice ridges that bulldoze banks; and eventually floating ice blocks (floes) can also impinge against some banks. Any sedimentation of ice-rafted materials in the upper estuarine reaches occurs within the channels, because over-bank escape of the ice floes is impeded by trees along the banks (Martini et al. 1993; Poehlman 1996). In the lower reaches of the estuary and nearby coasts, ice blocks can instead be strewn widely, escaping the lower, willow- or grass-covered banks. Most ice-rafted and water-suspended sediments are, however, transported into the marine bay and redistributed by waves, tides, and longshore currents.

Icy, frozen ground conditions also have an important influence on the coast and estuarine reaches of the Hudson Plains Ecozone. The ecozone encompasses seasonally frozen ground in the south, grading northward into sporadic, discontinuous, and finally continuous permafrost (permafrost zones are shown in Figure 23, Section 2.1, *Abiotic Drivers*). Frozen ground protects surficial sediments from erosion and organic matter from decomposing during the annual cold period. On the other hand, spring-summer thawing could enhance the rate of erosion, for instance by promoting slumping of river banks during break-up, hence producing a temporary increased sediment supply to the coast (Dumbrell 2000).

2.4.1.1.6 **Coastal vegetation**

Vegetation has several important effects on the coast and on the various organisms that use the coastal environments. One effect is to trap sediments transported by waves, currents, and tides, thus increasing the local rate of sedimentation both in the shallow subtidal areas and in the upper parts of the intertidal zone and in marshes (Martini 2006a,b; Martini et al. 2009). Furthermore, the vegetative cover (e.g., algae) and the intricate root maze of grassy and other plants stabilize the sediments, protecting them from erosion. Plants also contribute to the development and stabilization of the relatively small wind-blown dunes that form in certain areas on sandy-gravelly beach ridges. The other major effect of the vegetation is to provide ideal nursery and developmental habitats for invertebrates and other organisms. These organisms are valuable foodstuff for migratory animals, primarily migratory birds (Morrison and Gaston 1986). Where vegetation cover is severely reduced, such as in coastal areas of the ecozone affected by intense goose foraging (Jefferies et al. 2003; Section 2.2.2.1, *Coastal*) or locally by man along the banks of some streams, drastic changes can occur that leave the land amenable to increased erosion by wind and water flow (Poehlman 1996; Dumbrell 2000).

2.4.1.1.7 **Postglacial isostatic rebound**

The coasts of the Hudson Plains Ecozone are highly dynamic environments, not only because of strongly variable seasonal conditions but also, on a longer time span, because of continuous net emergence of the land, since retreat of the Laurentide Ice Sheet. The rate of emergence has changed through time, and it still varies along the modern shores from place to place. The rate of emergence is due first to the rate of isostatic rebound, which is relatable to the original thickness of the ice that depressed the earth's crust and to the subsequent relaxation of stresses within the crust, as the glacier melted. Second, it is related to the slope of the shores and to the change in sea level that is caused by the release of water from glaciers, as a result of globally variable climatic conditions. Early rates of rebound for the most inland ancient shores of the ecozone were estimated to be about 3 m/century. The rate of present day emergence owing to residual postglacial isostatic rebound has been variously estimated to average between 1.1 and 0.7 m/century in different parts of the coast (Webber et al. 1970; Gough and Robinson 2000; Sella et al. 2007). These rates of isostatic rebound are among the highest in North America.

Considering that the average slope of the Hudson Plains Ecozone ranges from 0.5 to 0.7 m/km, in flat areas the shoreline could experience maximum seaward progradation of up to 1 to 2 km/century. Such changes are not uniform along the coast. As an example, simple comparisons between air photographs taken in 1954 and 1976 indicate that, along parts of the Hudson Bay coast, some marshes prograded up to 400 m, with approximately 3.3 km^2 of new marsh formed over a 6 km long stretch of coast (figures 103, 104) (Martini 1982; 2006a). Furthermore, in a century the upper estuarine reaches, where most of the few villages of the Hudson Plains Ecozone are located, could become up to about 0.5 to 0.7 m shallower, because the rate of erosion of the rock substrate is not keeping pace with the rate of uplift. Additionally, the river mouths could be a few kilometres farther toward the centre of the bay than today. These changes will create increasing difficulties for local human populations, owing to reduced navigability and an increased risk of damaging floods, which is associated with an increased frequency of ice jams at break-up.

2.4.1.2 **Predicted changes**

Except for hydroelectric development on some river systems and a relatively new mining operation, the Hudson Plains Ecozone has been little affected by direct human activities (Section 2.6.1.1, *Summary of Stressors*). It is, however, a fragile ecosystem easily disturbed by any increase in activities, and it is particularly sensitive to global climatic change, because it straddles the boundaries of several climatic and permafrost zones (Kettles and Tarnocai 1999; Tarnocai 2006). The present global warming will shift these boundaries northward and result in significant changes, most importantly the partial melting of ground ice and a reduction in the annual persistence of surficial fluvial and marine ice (Section 2.1, *Abiotic Drivers*). Changes in temperature and ice regime will trigger other linked processes, which will lead to locally notable modifications of the ecosystems. Three main points are notable regarding the estuaries and coasts.

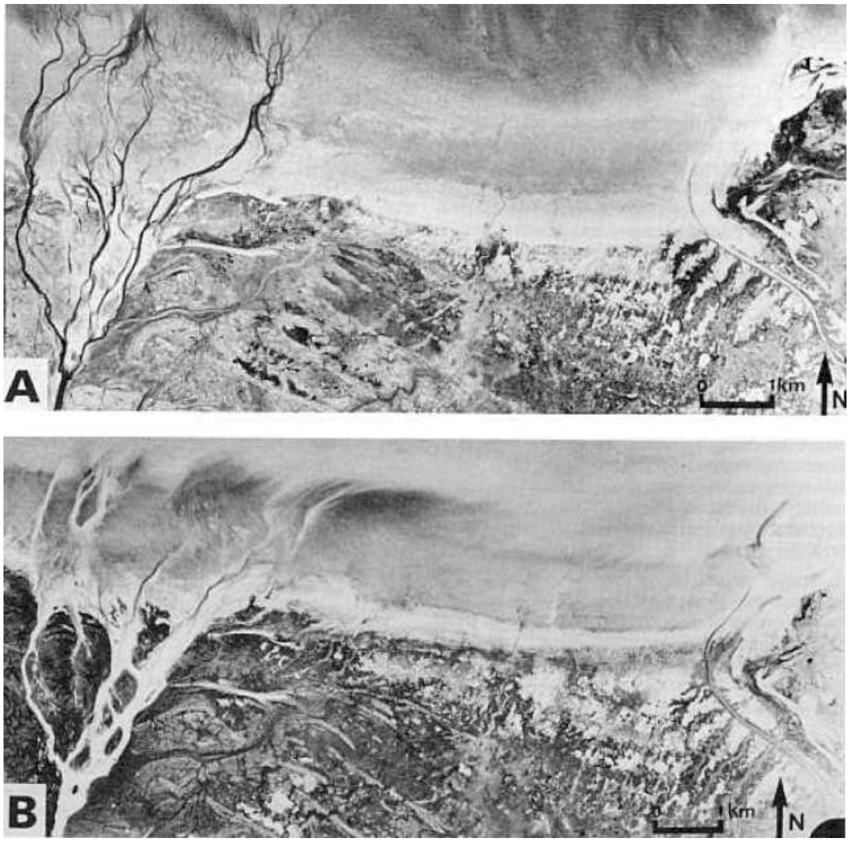

Figure 103. Coastal emergence and marsh development east of the mouth of the Black Duck River, near the Ontario-Manitoba border: a) 1954 air photograph and b) 1986 air photograph.
Source: Martini (1982). Reprinted from Le Naturaliste canadien, Vol 109, I.P. Martini, Geomorphological features of the Ontario coast of Hudson Bay, page 422, Copyright (1982), with permission.

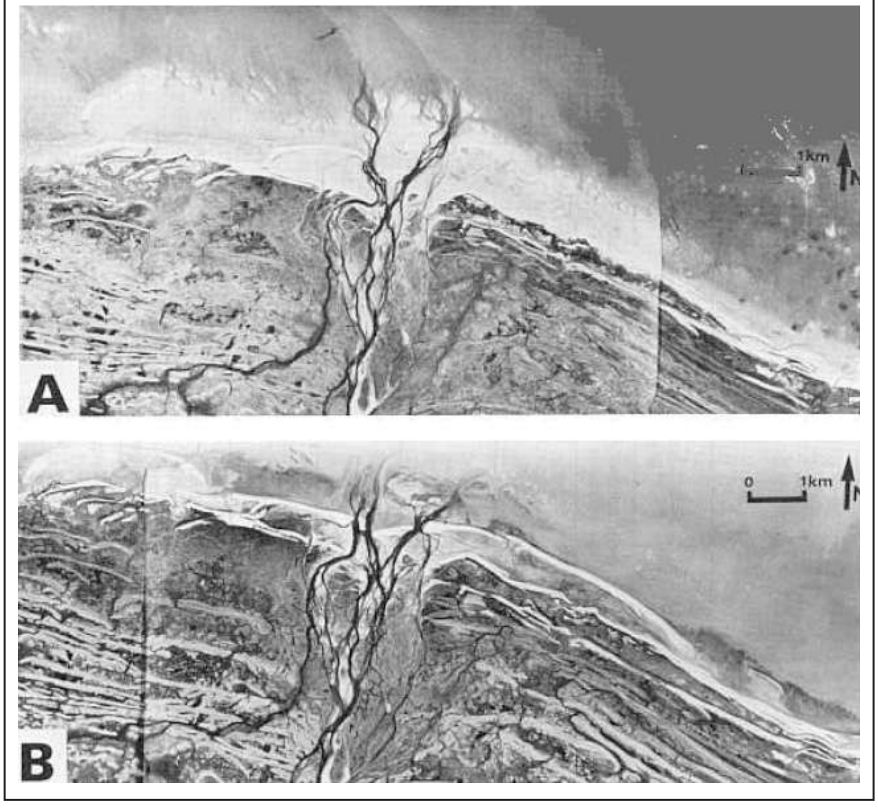

Figure 104. Emergence of lagoonal coast at the mouth of the Kettle River, 25 km west of the Ontario-Manitoba border: a) 1954 air photograph and b) 1986 air photograph.
Source: Martini (1982). Reprinted from Le Naturaliste canadien, Vol 109, I.P. Martini, Geomorphological features of the Ontario coast of Hudson Bay, page 424, Copyright (1982), with permission.

First, although it is difficult to predict specific changes for particular areas (Prowse and Beltaos 2002), the altered river flows and ice regimes (Section 2.1, *Abiotic Drivers*) might not drastically change the annual freshets and associated local ice jams and floods in the lower reaches of rivers and estuarine areas. Their adverse impacts will, however, persist on the communities of the estuarine areas. Furthermore, if the increase in temperature is large, the duration of a strong ice cover will be reduced, limiting the winter viability of the local people.

Second, reduced persistence of sea ice associated with a later annual freeze-up in the fall and/or an earlier melting in the spring/summer (associated with local climatic amelioration) (Section 2.1, *Abiotic Drivers*) has tremendous implications for ecosystems and for migratory fauna, such as the polar bears and birds that use them. The coastal ecosystems will also be affected by possible increases in wave intensity and more frequent recurrence of strong storm surges, which would lead to greater erosion of some shores and inundation of flatter near-shore lands (Martini et al. 2009).

Third, the coastal area of the Hudson Plains Ecozone has a dynamic equilibrium based on the balance between isostatic and eustatic movements. Postglacial isostatic rebound varies along the coast, and it tends to lessen over a long time, but its rate will not change significantly in the near future. The sea level (eustatism) could rise rapidly owing to global glacial melting, but it will not overtake the isostatic rebound. The combined effect along the coasts may be a reduction in rate of emergence. As such, the coastal systems could remain quasi stable for a longer time and achieve a greater degree of maturity. In estuarine reaches, the rate of shallowing of the channels could be reduced. This latter effect would facilitate navigability for local travel and tourism purposes, and potential harbours might become viable, due to the shorter annual period of sea ice cover in the bays. However, eustatic deepening of estuarine reaches could also be affected by an adverse increase in the rate and amount of sedimentation, which would require costly dredging.

Tsuji et al. (2009), taking into consideration the average rate of isostatic rebound and rise in sea level due to climate warming, have estimated approximate times (several centuries) that it would take for several islands of James Bay to become joined with the mainland of Ontario and Québec.

2.4.1.3 Knowledge gaps

The processes affecting the Hudson Plains Ecozone are not dissimilar from those of other cold coastal regions. They are, therefore, well known, although some are not completely understood. Regarding the coasts and estuaries, the most important knowledge gaps relate to systematic and accurate acquisition of key data and continuous monitoring of certain events. The determination of isostatic and eustatic movements is of prime importance for understanding changes that may affect the ecosystems and human settlements of the area. In the last decade or so, new work has been done on refining information on the rate of emersion of the coasts though Global Positioning Systems (GPS), revising the interpretations of the tidal-gauge data of Churchill, and on crustal models (see references in Tsuji et al. 2009). However, relative changes in sea level should be monitored directly and through remote sensing at specific localities, possibly at some distance from deltaic areas, to serve as ground truthing for the behavior of local critical coastal areas. Detailed studies, including leveling and ecosystem analyses of transects of the various types of coasts, were done starting in the 1970s along the Ontario coasts, and some detailed information is available in the published and unpublished documents of governments and others (e.g., Martini 2006a). Re-surveying and studying some of these transects could provide valuable information on changes over the past 30-40 years.

Similarly for the estuaries, some hydrologic mapping was done in the past, and it should be repeated and compared. Studies have been done on the channel morphologies and sediments of the lower reaches of some rivers, including: 1) those affected by hydroelectric works, either: a) permanently by partial river diversion (such as the Eastmain and Opinaca rivers in Québec and the Churchill and Nelson rivers in Manitoba) or b) temporarily by small dam construction in the headwaters on Precambrian terrains (such as the Moose River, Ontario); and 2) those rivers that were or are still pristine, including the Attawapiskat River (King and Martini 1983), which is now associated with large-scale mining along its middle course (DeBeers Canada 2005). These works and their derived concepts are worth considering and re-checking. More programs could be established for monitoring variations in water level and discharge, both natural and induced (by hydroelectric and other human activities), as well as sediment transport and the type and rate of sedimentation in various parts of the channels of the system. Repeated ground and high resolution remote-sensing image and/or air-photograph surveys of the morphology and stability of the channels and banks are needed, as well.

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2.4.2 **Natural disturbances**

Like other ecozones across Canada, the terrestrial Hudson Plains Ecozone is shaped by a variety of natural disturbances that occur at different scales, frequencies, and severities. However, comparatively little is yet known about the natural disturbance regime in this ecozone and its associated effects on the ecology of the area. It is also not clear at this time if the ecozone's overall disturbance regime has been changing. The satellite remote-sensing analysis of land cover change that is available for the ecozone as a whole (discussed earlier; see Section 2.2.1.2, *Land Cover Change*) might suggest that the ecozone's large disturbance regime remained relatively stable over the 1985-2005 analyzed period. However, the temporal scale of that analysis is very limited, and, given the coarse-scale (1 km) of resolution, the error in mapping is likely greater than the amount of change or disturbance being detected.

This section reviews what is known about the status and trends of four types of natural disturbance in the Hudson Plains Ecozone: extreme weather; fire; large-scale native insect outbreaks; and forest diseases. Although not similarly profiled here, herbivory (Section 2.4.3.2, *Herbivore-Plant Interactions*) is also an important disturbance agent in this ecozone, insofar as the excessive foraging by a greatly expanded population of lesser snow goose (*Chen caerulescens caerulescens*) has severely disturbed the ecozone's coastal salt marshes and, to a lesser extent, freshwater marshes in its tundra. This phenomenon was discussed in sections 2.2.2.1, *Coastal* and 2.2.2.2, *Polar-Tundra*. As such, it is not reitterated here. Flooding is likewise not profiled here (but see the section on extreme weather).

2.4.2.1 **Extreme weather**

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Extreme weather includes extreme temperature and precipitation events (i.e., daily extremes or longer periods of extreme weather, such as droughts), as well as severe local weather events like wind, snow, and ice storms (Trenberth et al. 2007). Because extreme weather can have more profound implications on biota than gradual changes in climate, it can be a driver of broad ecological responses to climatic trends (Parmesan et al. 2000). Natural disturbance regimes and associated population and trophic dynamics can both be altered by changes in the type, frequency, or severity of extreme weather (e.g., Parmesan et al. 2000; Rittenhouse et al. 2010).

It is unclear if there are significant trends to date in the frequency and/or severity of extreme weather events (and related disturbance) in the Hudson Plains Ecozone. Trends in extreme weather events are in general hard to evaluate, because assessment requires long-term observations; the rarer the event, the more difficult it is to identify long-term trends, because there are fewer cases to evaluate (Trenberth et al. 2007).

Reports of extreme weather events and their impacts are very limited for the Hudson Plains Ecozone. Historical reports from fur-trader posts in the ecozone suggest that extreme or unusual weather events during the period 1783-1821 (e.g., mild winters/low snow cover, spring flooding, prolonged drought, and related drought-fuelled fires) exacerbated the declines in furbearer populations that were occurring there due to excessive trapping at the time (Lytwyn 2002). A commonly cited catastrophic event in the ecozone in modern times is the flood that destroyed the coastal community of Winisk, Ontario in 1986[54], but this flood was related to the sensitivity of this low-lying community to annual river ice jams and spring flooding (see Newton et al. 2005). In recent decades, researchers working in the ecozone have observed ecosystem impacts from extreme weather events, such as the tree damage shown in Figure 105. Aboriginal and other peoples

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[54] The community was rebuilt ~40 km further inland and renamed Peawanuck.

in the area are likewise increasingly reporting unusual weather and associated impacts, including at least two occurrences in the past decade of fish die-offs in rivers (Sutton and Albany rivers) during periods of unusually warm summer weather and reduced precipitation and river flows (Gunn and Snuccins 2010; Hori 2010; see also Section 2.2.2.4.2, *Rivers/Streams & Lakes*). They also report more berry crop failures (Peloguin 2007; Sayles 2008)[55], and that weather is less predictable than before, with adverse effects on safety, hunting, and fishing (Ford et al. 2008; Hori 2010).

Figure 105. Bending and breakage of trees from ice or heavy snow in 2008 within 10 km of Burntpoint, on the Hudson Bay coast in Ontario.
Photo credit: R. Brook, Ontario Ministry of Natural Resources.

Several studies have, however, examined trends in indices or indicators of extreme weather, which are derived from daily temperature and precipitation data. At the ecozone scale, the occurrence rate of extreme drought years showed no change over the period 1901-2002, when estimated from July monthly drought code (MDC)[56] (Girardin et al. 2009). At the sub-ecozone scale, a trend towards decreasing dryness (lower MDC) was evident at the extreme southern end of the ecozone in Ontario and Québec over the same period (1901-2002) but not for the more recent period, since 1951 (Girardin and Wotton 2009)[57].

Also of interest at a sub-ecozone scale is that long-term weather data from stations at both Churchill and Moosonee were included in a national analysis of extreme weather indices (Bonsal et al. 2001; Zhang et al. 2001; Vincent and Mekis 2006; Qian et al. 2010; see also Peterson et al. 2008). Over the period 1950-2003, both of these stations showed a significant increase in diurnal temperature range (with Churchill additionally showing an increase in the standard deviation of temperature mean), and the Moosonee station showed significant trends for more warm days and more summer days (Table 23). Precipitation indices showed no significant trends over this period for either station (Table 23), albeit some increase in precipitation intensity is suggested for at least part of the ecozone, when station-level indices are area-averaged across

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[55] The cited references for berry crop failures apply near Wemindji, Québec. The community itself is just outside of Hudson Plains Ecozone boundaries.

[56] Monthly drought code (MDC) is a moisture index used to carry out seasonal drought characterization analyses, where daily weather data necessary for computation of the daily drought code (DC) are not available. Both DC and MDC represent the moisture content of organic matter that is, on average, 18 cm thick and 25 kg/m^2 dry weight, for a bulk density of 138.9 kg/m^3. July MDC also integrates the influence of the two previous months (May and June); conditions later in the season are not captured by this index. The MDC analysis for the Hudson Plains Ecozone was not biased by an excessively smoothed climate record associated with a paucity of climate data (MDC analyses for some other circumboreal ecozones, such as Canada's Taiga Shield Ecozone, were biased in this respect).

[57] The summer (June-August) Palmer Drought Severity Index (PDSI), analyzed only for the station at Churchill, also showed no significant increase in dryness over the period 1950-2007 (Zhang et al. in press).

the ecozone by grid-interpolation (Peterson et al. 2008). Hori (2010) also reported a significant trend for increasing homogenized-monthly mean-maximum temperature in summer for the Moosonee station for the period 1960-2006. Overall, however, these studies of indices of extreme weather suggest only limited change in extreme weather in this ecozone to date.

Table 23. Trends in extreme weather indices at two stations in the Hudson Plains Ecozone, 1950-2003. Abbreviations: Tmin, daily minimum temperature; Tmax, daily maximum temperature; Tmean, daily mean temperature; and NS, not statistically significant. Upward and downward arrows denote significant positive and negative trends, respectively.
Source: Vincent and Mekis (2006).

Index of extreme weather	Index definition	Station trends	
		Churchill, Manitoba	Moosonee, Ontario
Temperature indices			
Frost days	Number of days with Tmin <0 °C	NS	NS
Cold days	Number of days with Tmax <10th percentile	NS	NS
Cold nights	Number of days with Tmin <10th percentile	NS	NS
Summer days	Number of days with Tmax >25 °C	NS	↑
Warm days	Number of days with Tmax >90th percentile	NS	↑
Warm nights	Number of days with Tmin >90th percentile	NS	NS
Diurnal temperature range	Mean of the difference between Tmax and Tmin	↑	↑
Standard deviation of the temperature mean	Standard deviation of daily mean temperature from Tmean normal	↑	NS
Precipitation indices			
Simple day intensity index of precipitation	Annual total precipitation/number of days with precipitation >trace	NS	NS
Simple day intensity index of rain	Annual total rain/number of days with rain >trace	NS	NS
Maximum number of consecutive dry days	Maximum number of consecutive dry days (trace days excluded)	NS	NS
Highest 5 day precipitation amount	Maximum precipitation sum for 5 day interval (potential indicator of flood-producing events)	NS	NS
Very wet days (≥95th percentile)	Number of days with precipitation ≥95th percentile	NS	NS
Heavy precipitation days (≥10 mm)	Number of days with precipitation ≥10 mm	NS	NS

There is a clear evidence of anthropogenic influence on Canadian climate and on temperature in particular (Zhang et al. 2006). As extreme weather events are highly dependent on the state of the climate, extremes are expected to be influenced by anthropogenic activity, as well. The observed changes in indices of extremes in Canada (Vincent and Mekis 2006; Peterson et al. 2008; Qian et al. 2010) are consistent with what would be expected from a warming world. Additionally, anthropogenic influence has been detected in temperature extremes over Canada (Zwiers et al. in press). These observations are consistent with climate model projections that show an increased frequency of extreme weather events in the future (Christensen et al. 2007; Meehl et al. 2007).

Exactly how an increased frequency of extreme weather events will affect the Hudson Plains Ecozone in the future is difficult to predict. Northern environments are characterized by relatively high levels of spatial and temporal variability in abiotic conditions, such as temperature, precipitation, wind, and snow cover. While species inhabiting these environments tend to exhibit adaptations that provide some resilience to this variation, such adaptations have limits (e.g., Prestrud 1991; Martin and Wiebe 2004). For example, nesting failures of lesser snow goose (*Chen caerulescens caerulescens*) on the coast are observed in years with freezing rain, excessive rain, or cumulative snow fall in May (Skinner et al. 1998). Likewise, a high annual number of freeze-thaw cycles acts as a significant stressor on populations of Richardson's collared lemming (*Dicrostonyx richardsoni*), a species that reaches its southernmost range in the tundra of the Hudson Plains Ecozone near Churchill. In this area of transitional climate, lemming abundance is closely linked to extreme temperature and precipitation conditions (defined in that work as >1 SD from average), when these events occur during lows in the lemming population cycle (Scott 1993). This lemming and other species that reach the limits of their southern- or northern-most ranges of distribution (including physiological tolerances) in the Hudson Plains Ecozone might be among the most affected by changes in critical environmental conditions or extreme weather events (see Brown et al. 1996).

Some effects of extreme weather can be expected to interact with other components of the ecozone's disturbance regime. Extensive tree damage from storms or vegetation loss from drought could fuel fires, for example (fire is discussed below in Section 2.4.2.2). The increased oceanic mixing and wave generation (storminess) that will almost certainly be associated with the forecast reductions in sea ice cover in Hudson and James bays (Gagnon and Gough 2005; Joly et al. 2010; Section 2.1, *Abiotic Drivers*) might also lead to enhanced levels of coastal flooding and erosion (Walsh 2008; Martini et al. 2009). Although this latter effect is generally expected to be compounded by sea level rise (Walsh 2008), the Hudson Plains Ecozone is notably less susceptible to sea level rise than some other coastal areas of Canada (Tsuji et al. 2009), because of its especially high rate of isostatic rebound (Sella et al. 2007). Still, any inundation of near-shore lands could have relatively far-reaching effects, given that this ecozone exhibits only a very gentle grade (~0.5 m/km) inland from the coast and very little relief (Section 1.1, *Geology, Topography & Climate*). Most of the ecozone's human population is also located in low-lying coastal areas.

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2.4.2.2 **Fire**

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The Hudson Plains Ecozone is part of Canada's boreal forest, for which fire can be an important driver of ecosystem processes (such as succession, insect and disease regulation, and carbon and nutrient cycling) and, thus, overall diversity (Weber and Stocks 1998; Burton et al. 2008). Large fires in the Hudson Plains Ecozone tend, however, to be less frequent and of smaller maximum size than in the neighbouring Boreal Shield and Taiga Shield ecozones (Stocks et al. 2003; Parisien et al. 2006; Burton et al. 2008), presumably owing to an overall humid, cool climate and a predominance of wetlands and related effects on horizontal fuel discontinuity and overall fire resistance (Parisien et al. 2006). Fires in the Hudson Plains Ecozone continue to go largely unsuppressed (unless they threaten Aboriginal communities or other designated values), given the remoteness of this ecozone and efforts to accommodate the natural role of fire there (OMNR 2004; Burton et al. 2008). As such, the fire regime in this ecozone is essentially natural (e.g., Ter-Mikaelian et al. 2009).

More data are available on fire than for any other type of natural disturbance in the Hudson Plains Ecozone. These data come from all documented large fires ≥2 km^2 (≥200 ha) in the Canadian Large Fire Database (1959-1999)[58] (Stocks et al. 2003; Parisien et al. 2006) and, more recently, remote sensing (1995-2007+) (Krezek-Hanes et al. in press). Unfortunately, however, some of the older fire data have important limitations. Specifically, in the past, fire detection

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[58] Fires ≥2 km^2 (large stand-replacing fires) make up a small percentage of the total number of fires that occur each year in Canada, but these larger fires represent 97% of the area burned (Stocks et al. 2003). Such large fires are comparatively less common in the Hudson Plains Ecozone than areas like the western Boreal Shield (Stocks et al. 2003), but they are still ecologically important. Smaller fires are also important in the Hudson Plains Ecozone (Sims et al. 1979), but less is known about their status and trends.

was non-existent or limited in many northerly areas with limited or no intensive (active) fire suppression, thereby rendering records in the large fire database inaccurate for these areas until the mid-1970s (Stocks et al. 2003). Fire size statistics from the early records might also be of reduced quality due to the methods used to delineate them. Accordingly, all data on large fires for the 1960s and 1970s are screened-out in the tables and figures in this section[59], and 1959-2007 trend statistics are not reported given the probable bias towards increasing number of fires and area burned. The discussion focuses instead on the last three decades of large fire data, for which coverage and documentation have been reasonably complete (Stocks et al. 2003). Long-term, annually resolved fire statistics are needed in order to continue describing and monitoring changes in the ecozone's large fire regime. Information is also needed on smaller fires <2 km^2. Moreover, information and monitoring on fire severity is required.

Figure 106 shows the annual area burned by large fires (≥2 km^2) in the Hudson Plains Ecozone up to the year 2007. Considering the data from 1980 onward, large variability is evident in the area burned from one year to the next (see also Stocks et al. 2003; Brook 2006; Parisien et al. 2006). In some years, such as 1993 and 2004, the burned area is very low. In other years, fire activity is comparatively high, such as 1989 (4,572 km^2) and 2003 (3,455 km^2). No trend in the area burned is evident, including when the data are examined on a decadal basis (Figure 107, again disregarding 1960s and 1970s data). The reason for the apparent increase in the area burned from the 1980s to the 1990s is not clear, and it is likewise unclear if the slight decline suggested in the last decade is real or a result of pro-rating the data for the 2000s based on the first 7 years in that decade only. However, these same decadal trends from the 1980s onward are also apparent at the national scale, and at least some evidence suggests they might be linked to large-scale ocean circulation patterns (see discussion in Krezek-Hanes et al. in press). Overall, when area-corrected, the mean annual area burned in the Hudson Plains Ecozone appears lower than that of many fire-dominated ecozones in Canada, including the Taiga Shield Ecozone and at least the western portion of the Boreal Shield Ecozone (Parisien et al. 2006)[60]. However, inter-annual variation in the area burned is very high for most ecozones, with standard deviations typically greater than the means. Direct annual carbon emissions from large fires likewise appear lower overall in the Hudson Plains Ecozone than in neighbouring Taiga Shield and Boreal Shield Ecozones (Amiro et al. 2001).

Similar to other forested northern ecozones in Canada (Stocks et al. 2003; Parisien et al. 2006; Krezek-Hanes et al. in press), fire starts in the Hudson Plains Ecozone are dominated by lightning (Figure 108). Lightning ignitions accounted for about 93% of the large fires (≥2 km^2) in this ecozone in the 1980s and 1990s. There is no evidence from this temporally limited data set to suggest that the proportion of fires caused by humans or lightning is changing. In any case, lightning-caused fires tend to burn naturally in the Hudson Plains Ecozone because, as previously indicated, fire suppression is very limited there.

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[59] All tables and figures in this section on fire are derived from an analysis done for all ecozones$^+$ in Canada in support of national ESTR assessment (Krezek-Hanes et al. in press).

[60] Mean annual area burned 1980-1999, area-corrected for 500,000 km^2: Hudson Plains, 97,106 ± 151,526 ha; Taiga Shield (west), 311,222 ± 568,119 ha; Taiga Shield (east), 154,472 ± 306,296 ha; Boreal Shield (west), 514,715 ± 553,227 ha; and Boreal Shield (east), 77,189 ± 88,229. These values apply to the ecozones framework (ESWG 1995), rather than the ecozones$^+$ framework defined for the ESTR (Rankin et al. in press).

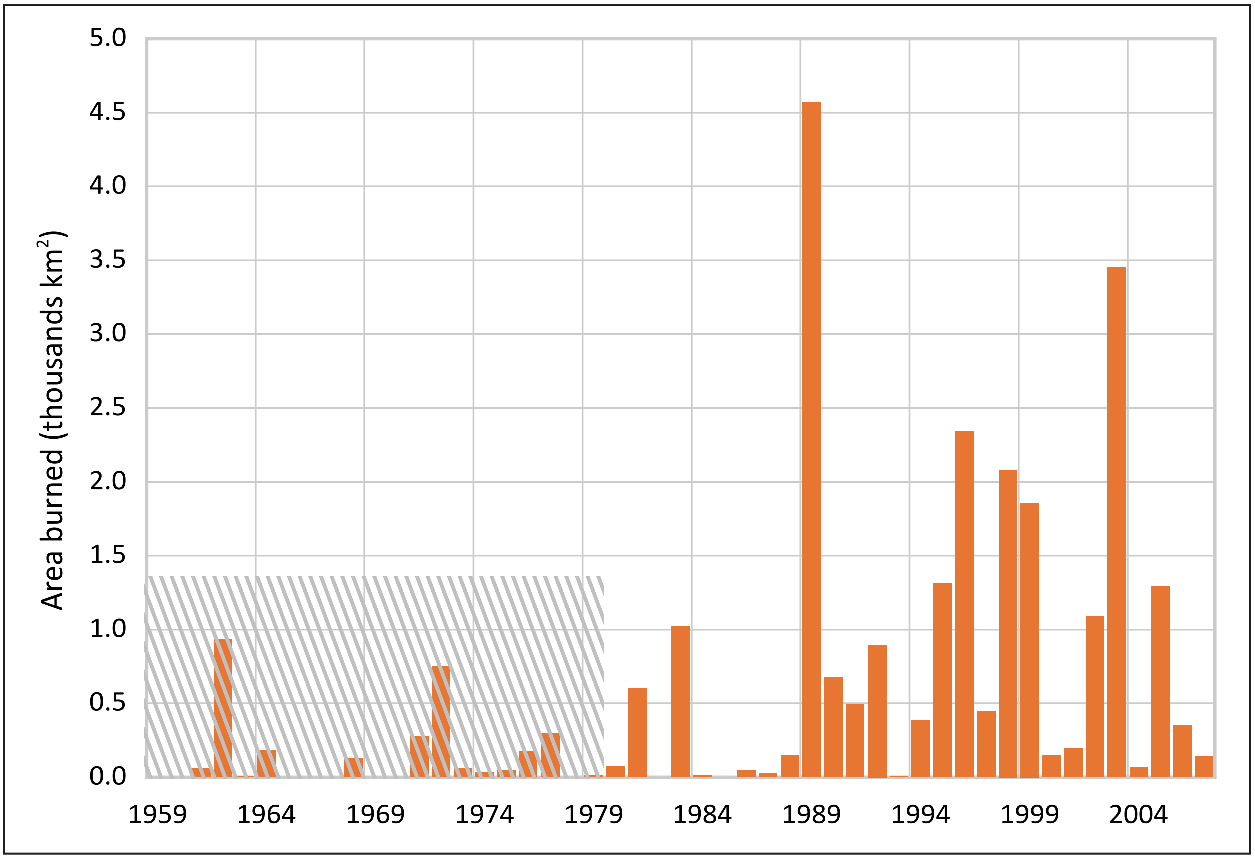

Figure 106. Annual area burned by large fires (≥2 km^2) in the Hudson Plains Ecozone, 1959-2007. Data up to 1979 are screened-out to denote probable inaccuracy; these data are from a time period when fire detection was non-existent or limited (see text). This analysis is based on the ecozones$^+$ framework. Source: Ecozone analysis provided by authors of Krezek-Hanes et al. (in press). Data from 1959-1994 are from the Canadian Large Fire Database. Data from 1995-2007 are derived from remote sensing using the HANDS (Hotspot and NDVI (Normalized Difference Vegetation Index) Differencing Synergy) processing method.

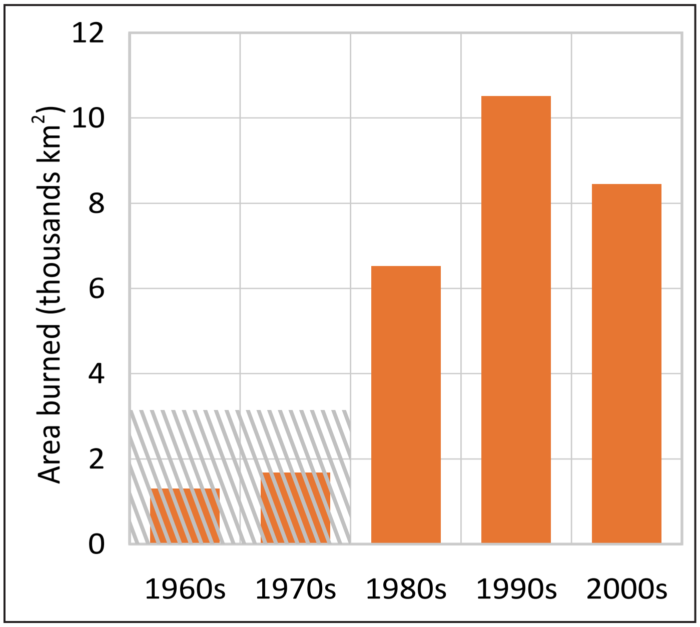

Figure 107. Trend in total area burned per decade (by large fires ≥2 km^2) in the Hudson Plains Ecozone. The value for the 2000s decade was pro-rated over 10 years based on the average from 2000-2007. Data from the 1960s and 1970s are screened-out to denote probable inaccuracy; these data are from a time period when fire detection was non-existent or limited (see text). This analysis is based on the ecozones$^+$ framework. Source: Data for ecozone provided by authors of Krezek-Hanes et al. (in press). Data from 1959-1994 are from the Canadian Large Fire Database. Data from 1995-2007 are derived from remote sensing using the HANDS (Hotspot and NDVI (Normalized Difference Vegetation Index) Differencing Synergy) processing method.

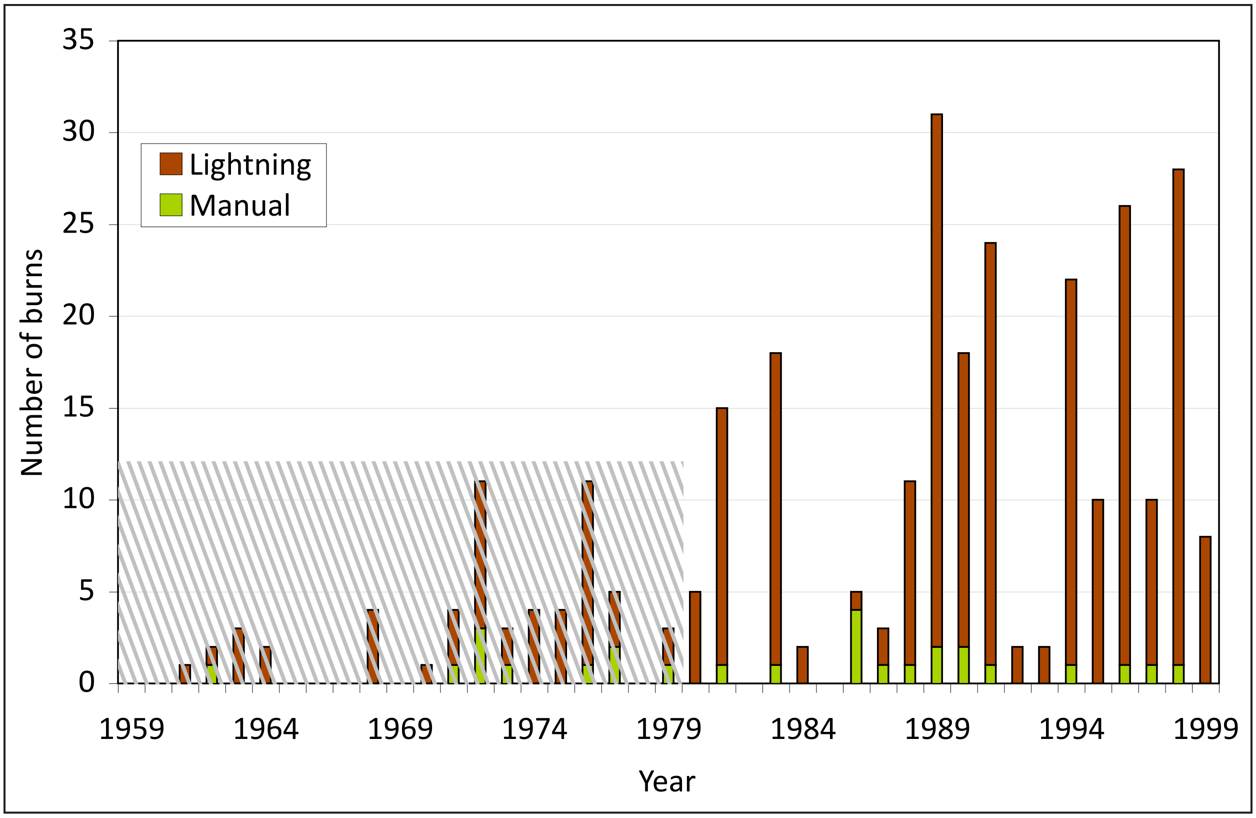

Figure 108. Percentage of lightning- and human-caused large fires (≥2 km²) in the Hudson Plains Ecozone, 1959-1999. Data are missing for 1965, 1966, 1967, 1969, 1978, 1982, and 1985. Data up to 1979 are screened-out to denote probable inaccuracy; these data are from a time period when fire detection was non-existent or limited (see text). This analysis is based on the ecozones⁺ framework.
Source: Ecozone analysis provided by authors of Krezek-Hanes et al. (in press), using data from the Canadian Large Fire Database, 1959-1999.

The seasonality and duration of the active fire season have not changed in the Hudson Plains Ecozone, since the 1980s (Table 24). Most large fires occur between May and August, fire activity peaks mid-period, and the average duration of the fire season (for large fires only) is about 55 days. The active fire season in the Hudson Plains Ecozone appears shorter than that in the neighbouring Boreal Shield and Taiga Shield ecozones (Krezek-Hanes et al. in press).

The above analysis of fire statistics from 1980 onward does not provide any evidence that fire disturbances are changing in the Hudson Plains Ecozone. Although the data and analysis are limited in terms of temporal scale, the results are consistent with recent studies of long-term trends in the July monthly drought code (MDC)[61], an indicator of wildfire risk across circumboreal forests (Girardin and Wotton 2009; Girardin et al. 2009). Overall, the Hudson Plains Ecozone has shown no linear change in wildfire risk or in the frequency of extreme fire-season conditions.

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[61] July MDC integrates the influence of the two previous months (May and June), but conditions later in the season are not captured by this index. Note that roughly 85% of the annual area burned in the Hudson Plains Ecozone occurs in these three months (see Table 24). The MDC analysis for the Hudson Plains Ecozone was not biased by an excessively smoothed climate record due to a paucity of climate data (MDC analyses for some other circumboreal ecozones, such as Canada's Taiga Shield, were biased in this respect).

Table 24. Trends in the proportion of large fires (≥2 km^2) that occur each month of the fire season, shown by decade. Monthly numbers are the percentage of the total number of fires that occurred during each month. Data from the 1960s and 1970s are screened-out to denote probable inaccuracy; these data are from a time period when fire detection was non-existent or limited (see text). This analysis is based on the ecozones$^+$ framework.
Sources: Ecozone analysis provided by authors of Krezek-Hanes et al. (in press), using data from the Canadian Large Fire Database, 1959-1999 inclusive.

Decade	Percent of the total number of large (≥2 km^2) fires occurring per month (%)					Duration of active fire season (days)
	May	June	July	August	September	
1960s	33.3	25.0	41.7	0.0	0.0	9
1970s	13.0	26.1	50.0	10.9	0.0	29
1980s	3.3	38.9	44.4	13.3	0.0	50
1990s	4.3	38.9	40.3	16.8	0.7	59

2.4.2.2.1 **Intra-ecozone variation in fire**

Visual inspection of data from the Canadian Large Fire Database suggests that notable spatial variation exists in the area burned within both the Hudson Plains Ecozone (Figure 109) and the Taiga Shield Ecozone, when compared to the other fire-dominated ecozones of Canada (Parisien et al. 2006). Few studies have, however, attempted to quantify how the fire regime varies across the Hudson Plains Ecozone. Notably, Bridge (2001) examined spatial and temporal variation in fire cycles and the area burned across Ontario, including parts of the Hudson Plains Ecozone. Fire cycles were estimated from time-since-fire distribution for all fires ≥2 km^2 for the period 1921-1995. No data were available for Ecoregion 215 (Coastal Hudson Bay Lowland or Ontario Ecoregion 0E) (ecoregions are shown in Figure 36, Section 2.2.1.1.3). However, as expected based on gradients in climate and thunderstorm frequency, Ecoregion 216 (Hudson Bay Lowland, or Ontario Ecoregion 1E)[62] was found to have a longer fire cycle than the more southerly Ecoregion 217 (James Bay Lowland, or Ontario Ecoregion 2E), and fire cycles in the James Bay Lowland Ecoregion were longer than those at comparable latitudes in the western part of the province's Boreal Shield Ecozone (Ontario's Ecoregion 2W) (Bridge 2001). Unfortunately, however, Bridge (2001) included the incomplete early fire data in his analysis. As such, little may be reliably concluded about the ecozone based on this work. A previous analysis for Ontario used only fire data from 1976 to 1998, but that study only confirmed that fire cycles in the James Bay Lowland Ecoregion (1,817 years) are longer than those at comparable latitudes in the western part of the province's Boreal Shield Ecozone (Frech et al. 1999). This latter study did not compare fire cycles among ecoregions within the Hudson Plains Ecozone over this relatively short period.

Although the studies of Bridge (2001) and Frech et al. (1999) did not include data for Ecoregion 215 (Coastal Hudson Bay Lowland), descriptive narratives for the Ontario portion of this ecoregion do suggest that major fires are infrequent there owing to the low amount of tree cover present; instead, fires are largely limited to smaller ones that occur occasionally in treed

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[62] Descriptive narratives additionally suggest that, within the Ontario portion of the Hudson Bay Lowland Ecoregion, fires tend to occur more frequently on or adjacent to the drier and better-drained Sutton Ridges (Crins et al. 2009).

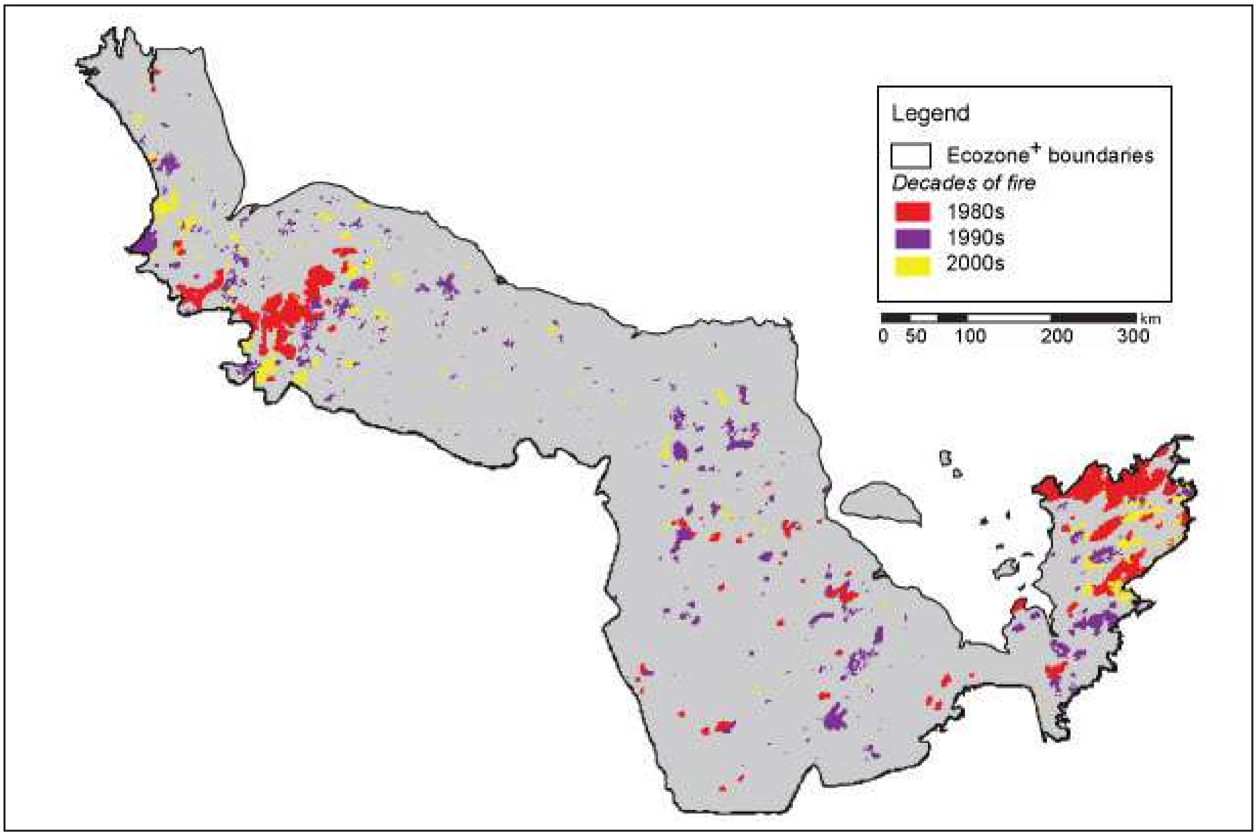

Figure 109. Spatial distribution of large fires (≥2 km^2) in the Hudson Plains Ecozone, 1980-2007. This analysis is based on the ecozones$^+$ framework.
Source: Ecozone analysis provided by authors of Krezek-Hanes et al. (in press), using data from the Canadian Large Fire Database.

patches (Crins et al. 2009). Fire activity within the climatically drier northwestern, Manitoba portion of Ecoregion 215 may be higher (Monson 2003; Riley 2003; Brook 2006) but, overall, fewer and smaller burns still occur close to the Hudson Bay coast, with larger burns becoming more common away from the coastline (Brook 2006). This spatial pattern of more fire occurring south of the north coast and tundra portions of the ecozone is consistent with fire mapping from the large fire database (Figure 109; see also Stocks et al. 2003; Burton et al. 2008) and Ontario's fire database (Figure 118 in Section 2.4.4, *Carbon Cycling*). Notably, large fires may occur more commonly in the southeastern, Québec portion of the ecozone (Stocks et al. 2003; Burton et al. 2008; see also Héon 2010 for a transect study that extends into this area from the Taiga Shield to the north). In fact, based on an analysis of data for 1980-1999, one of the most important fire clusters in eastern Canada occurs in Québec at the junction of the Hudson Plains Ecozone with the eastern portions of the Boreal Shield and Taiga Shield ecozones (Parisien et al. 2006). It is difficult to know if this cluster emerged randomly, was influenced by human activity, or is simply an area of persistently higher fire activity.

The susceptibility of different vegetation types to fire has received little study in the Hudson Plains Ecozone. Overall, lichen woodlands, inland beach ridges, forested riverbanks, and other upland forest communities tend to be particularly subject to fire (Sims et al. 1979; Riley 2003). Generally, surface fires are common across vegetation types, but bogs underlain by permafrost and elevated above the water table are the most likely peatland type to have moisture contents within ignitable and combustible levels for less frequent, shallow peat fires (Zoltai et al. 1998). Deep peat fires are expected to be rare, generally being restricted to disturbed areas with low water tables.

2.4.2.2.2 **Human influences & future trends**

Accelerated (human-induced) climate change might currently be one of the more important human influences on the fire regime of the Hudson Plains Ecozone, given that fires go largely unsuppressed there and logging and other anthropogenic activities remain very limited (although future development could increase the number of human-caused fires). That being said, the limitations associated with the early fire data, however, impose some challenges in determining the relative importance of climate change in driving changes in fire regime in this ecozone. Wildfire risk (or fire danger) indices, which offer a surrogate for fire statistics, provide little evidence for a changed fire regime in the Hudson Plains Ecozone, since the early 1900s (Girardin and Wotton 2009; Girardin et al. 2009; see also Ter-Mikaelian et al. 2009). This contrasts with some other areas in Canada, of which the southeastern and southwestern boreal show diminishing wildfire risk and area burned, and other areas show increases. Notably, in independent assessments, the Claybelt portion of the Boreal Shield Ecozone (an area of low productivity forested peatlands located just south of the Hudson Plains Ecozone's James Bay Lowland Ecoregion) shows an apparent decrease in fire activity, since the 1850s (Bergeron and Archambault 1993). The trend in the fire cycle there was attributed to a shift toward more advecting moist air masses over the area (Girardin et al. 2006). A trend towards decreasing dryness (reduced wildfire risk) is also evident at the extreme southern end of the Hudson Plains Ecozone in Ontario and Quèbec for the period 1901-2002 but not when only the more recent period from 1951 is considered (Girardin and Wotton 2009).

In terms of future projections with accelerated climate change, both the Canadian and Hadley General Circulation Models (GCMs), when used in tandem with historical relationships between weather/fire danger and area burned, suggest that ecoregions 216 and 217 of the Hudson Plains Ecozone will experience a 2-3 fold increase in area burned (see Figure 106 for an indication of the current area burned) under a 3 x CO_2 scenario (~2080-2100) (Flannigan et al. 2005)[63]. The projection is the same for the adjacent, northerly Taiga Shield Ecozone, but it contrasts with a smaller, 1.25-2 fold increase for the Boreal Shield Ecozone. These results should be viewed with caution for the Hudson Plains Ecozone, however, as this study combined this ecozone's ecoregions 216 and 217 with the Taiga Shield Ecozone based on similar historical fire activity (fire data for Ecoregion 215 were insufficient for analysis), i.e., modeling was not conducted separately for the Hudson Plains Ecozone. As well, the modeling included the early fire data (1959+), and it did not explicitly take into account changes in vegetation, ignitions, fire season length, and human activity that could influence the area burned (Flannigan et al. 2005).

Similar results (an approximate doubling of % area burned per year by 2100) have since been obtained, however, in a study that generated ensemble mean simulations from 19 GCM experiments with 7 GCMs (including the Canadian GCM)[64] driven by various scenarios of greenhouse gas emissions for a smaller geographic area in Ontario and Quèbec that includes

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[63] The increase in fire danger is less for this area under a 2 x CO_2 scenario (Stocks et al. 1998), and limited regional climate modelling for forests typical of the Québec portion of the ecozone suggests a more modest increase in area burned under both scenarios (Le Goff et al. 2009).

[64] The use of an ensemble mean allows the approximate climate change signal to be distinguished from the natural variability (cf. individual GCM experiments providing indications of the magnitude of natural variability in the system).

much of the southerly (Ecoregion 217) portion of the Hudson Plains Ecozone, as well as a portion of the adjacent Boreal Shield Ecozone (Bergeron et al. 2010). Like the study of Flannigan et al. (2005), this latter study similarly used the early (1959+) fire data, and it could not account for all factors that could influence projections. Notwithstanding those general modelling limitations, the study further determined that the projected increase in future fire risk in this area by 2100 is expected to move the burn rate towards the upper limit of its range of natural variability during most of the Holocene (over at least the last ~ 7,000 years)[65]. The system will, therefore, become particularly vulnerable to cumulative effects (e.g., from clearcut logging) that could push it beyond its long-term natural range of variability (Bergeron et al. 2010). Also notable from recent work is that some increase in the occurrence (number) of fires from lightning-caused ignitions is now suggested for the area evaluated by Flannigan et al. (2005), albeit the extent of the projected increase varies greatly with the model used (Wotton et al. 2010). Human-caused ignitions might also increase in this ecozone, given the expected increase in development and, hence, human activity in the area.

As accelerated climate change proceeds, warmer temperatures are also expected to extend the fire season later into the year in eastern Canada (Wotton and Flannigan 1993; Le Goff et al. 2009; see also Stocks et al. 1998). Warmer temperatures will not, however, universally increase wildfire risk (potential area burned). Rather, changes in wildfire risk in boreal forests are expected to continue to be heterogeneous, both within Canada and globally (Girardin et al. 2009). Additional work is required to understand and estimate future fire activity in the Hudson Plains Ecozone. Although this ecozone is currently relatively resistant to fire (due to an overall humid cool climate and predominance of wetlands), it is comprised of permafrost and peatland ecosystems that are particularly vulnerable to climate change (e.g., Tarnocai 2006), and climate change modelling projects both warmer temperatures and overall drier conditions there in the future (Section 2.1, *Abiotic Drivers*), including increased MDC (Bergeron et al. 2010).

Changes to the fire regime of the Hudson Plains Ecozone could have important implications for the ecology of the area, like elsewhere in Canada's boreal forest (Weber and Stocks 1998). Although fire behaviour in peatlands remains poorly understood, one concern is that extension of fires later into the growing season might render some peatlands more vulnerable to deep soil consumption (increased depth of burning), given that burning is more likely to occur during the period of maximum water table drawdown and fuel exposure (Kasischke and Turetsky 2006; Flannigan et al. 2009; Turetsky et al. 2010). Another concern is that *holdover fires* (fires that continue to smoulder under snow cover during the winter and reappear the following spring) could become more common (Flannigan et al. 2009). These and other changes in the fire regime could have important impacts on permafrost loss and ecosystem processes, such as succession (e.g., shifting stands to younger age classes), as well as carbon and nutrient cycling. Impacts on carbon dynamics are of global concern, given the potential for greatly exacerbated atmospheric carbon emissions from increased exposure of large areas of dry peatlands to fire (e.g., Weber and Stocks 1998; Flannigan et al. 2009). Carbon dynamics within the Hudson Plains Ecozone are, however, spatially and temporally complex and dynamic, and more work is needed in order to reliably predict the fate of carbon there (Section 2.4.4, *Carbon Cycling*). Increased fire activity could also lead to increased mercury emissions, which in boreal regions can be more than 10-fold greater from burning peatlands than from fires in non-peatland forests (Turetsky et

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[65] The increase in the ensemble mean burn rate predicted by 2100 for this area (0.45%/yr with 95% confidence interval 0.32, 0.59) falls within the long-term past variability range of 0.37-0.90%/yr.

al. 2006). The need to consider enhanced fire suppression efforts as climate change proceeds is recognized, even if increasing fire suppression will be logistically and economically challenging in this geography (e.g., Stocks and Ward in press).

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2.4.2.3 **Large-scale native insect outbreaks**

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In Canada, extensive aerial reconnaissance of forest insect disturbances has been conducted by provincial agencies and the Canadian Forest Service, since the early 1940s. This was a response to the extensive spruce budworm (*Choristoneura fumiferana*) outbreak sweeping eastern Canada at the time. The surveys were initially used to estimate forest damage, predict future damage, and help develop management plans, particularly about if, when, and where to apply insecticides (Sippell 1983; Howse 1995; Howse et al. 1995; Knowles and Westwood 1995). The surveys were expensive and thus understandably largely focused on the productive forests. An unfortunate consequence is that, while the surveys of insect disturbances were conducted in the northern boreal zone and in the James Bay Lowlands (the southern edges of the Hudson Plains Ecozone), they were almost never extended deep into the ecozone itself (Simpson and Coy 1999). As a result, there is little in the way of direct, extensive information on either the current, or recent historical, role and status of insects as disturbance agents in this ecozone. However, tree ring studies (e.g., Jardon et al. 1994; Girardin et al. 2005) provide useful information, especially in remote areas (or the distant past) where (and when) no insect surveys are available. Together, these sources of information suggest that the spruce budworm and the larch sawfly (*Pristiphora erichsonii*) are the two main defoliators in this ecozone. There, as in much of the boreal forest, white spruce (*Picea glauca*), black spruce (*P. mariana*), and eastern larch or tamarack (*Larix laricina*) are the dominant tree species.

The surveys report occasional episodes of defoliation by both the forest tent caterpillar (*Malacosoma disstria*) and the spruce budworm in the James Bay Lowlands (Simpson and Coy 1999). However, these episodes would be considered too short, too scattered, and too separated in time to expect substantial tree mortality or otherwise be considered an important disturbance, if they had occurred further south (Fleming et al. 2000; Volney and Fleming 2007). Thus, unless forest in the Hudson Plains Ecozone is exceedingly sensitive to mild levels of attack, one can assume that indigenous insects cause little direct disturbance to most tree species there.

Eastern larch is an exception: its sawfly is distributed widely over the Hudson Plains Ecozone (Turnock and McLeod 1966), and tree ring studies suggest that this insect sporadically produces substantial impacts on larch right up to the treeline (Cloutier and Filion 1991). The eastern larch beetle (*Dendroctonus simplex*) tends to follow outbreaks of the larch sawfly, further increasing the mortality of larch when it does. Langor and Raske (1989) reported widespread larch mortality in 1960 between Englehart and James Bay, Ontario, which would include the Hudson Plains Ecozone. Although not confirmed, eastern larch beetle might be the cause of recent larch mortality in the Churchill area, an observation made by Manitoba provincial staff in 2008 (I. Pines, Manitoba Conservation, pers. comm.).

Insects can also have major indirect impacts. It is becoming increasingly recognized that insect attack can leave stands vulnerable to other disturbances, such as fire (Fleming et al. 2002) or windthrow and ice storms (Lemieux and Filion 2004; Filion et al. 2006). These legacies of insect damage can be very long-lasting, even on the order of centuries (Bhiry and Filion 1996).

Due to the general lack of forest insect survey data for most of the Hudson Plains Ecozone and the sporadic occurrence of insect attacks from the surveyed areas of the southern parts of the ecozone, it is difficult to establish current trends directly from the surveys. Even if this were possible, it becomes dangerous to extrapolate such trends into the future, because they ignore climate change, and climate change will almost certainly be a major driver of future trends (e.g., Soja et al. 2007; Volney and Fleming 2007).

One practical alternative to simply extrapolating current trends into the future is to use information that applies to the southern Hudson Plains Ecozone and northern boreal forest, as a basis for inferring qualitative forecasts further north. This approach is based on the expectation that in a climatically warmed future, insect dynamics in more northerly areas could mimic those now occurring in 'otherwise ecologically corresponding' areas further south. This approach is complicated by the transient effects of climates, which are continuing to change during the forecast period. Using this approach but with very different statistical methods, Candau and Fleming (2008) and Gray (2008) forecast (for 2033 and 2100, respectively) increased spruce budworm defoliation in the southern Hudson Plains Ecozone and northern boreal forest.

As white spruce, a favored host of spruce budworm, is already established further north in the ecozone (Girardin et al. 2005), it seems likely that increased spruce budworm defoliation will also occur there. However, it will likely drop-off quickly as climatic limits are reached further north (Candau et al. 1998). In short, it seems likely that episodes of spruce budworm defoliation could become extensive and protracted enough to constitute disturbances in at least some of the southern parts of the Hudson Plains Ecozone. Exactly where and when will depend partly on local climate (including precipitation) and forest conditions. Severe budworm disturbances do not seem to be an immediate threat to forests further north.

Eastern larch is shade intolerant, and it is generally most abundant in wetlands. Some tree ring studies (Case and MacDonald 2003; Girardin et al. 2005) suggest that larch could suffer greater damage from sawfly attack in drier, warmer conditions. If so, one might also expect more damaging attacks from larch sawfly in the Hudson Plains Ecozone in the future.

As the last spruce budworm outbreak in eastern North America was winding down, Welsh (1983) summarized the state of research on the effects of spruce budworm disturbances on biodiversity. Budworm disturbances alter forest structure, foliage density, tree seed production, and the insects themselves are a food resource for many species of birds, small mammals, and other arthropods. Almost all direct data on the influences of these various effects pertains to birds that prey on budworm. Insectivorous birds, especially those closely linked to budworm, react quickly and strongly to budworm outbreaks (Venier and Holmes 2010). The authors suspect that the increased food supply presented by an outbreaking budworm population allows these birds to successfully raise more chicks. The broader bird community in general shows a weaker but also positive reaction to outbreaks, probably due to the longer-term change in forest structure caused by budworm outbreaks (Venier and Holmes 2010). Studies on small mammals suggest that, depending on their ecological needs, they can respond to changes in forest structure, seed production, or the abundance of budworm larvae that fall to the ground during budworm disturbances (Welsh 1983). Spruce budworm-caused changes in forest structure during disturbances can be expected to alter food resources, with complex consequences for moose populations. In general, it is difficult to say how relevant these observations and inferences are to the Hudson Plains Ecozone.

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2.4.2.4 **Forest diseases**

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Effectively no specific documentation exists regarding the composition of forest pathogens in the Hudson Plains Ecozone or their role there as natural disturbance agents. Pathogens capable of causing serious injury or death to host trees found in the ecozone have, however, been reported for north-central and north Ontario (sensu Davis and Myren 1990), which includes the most south and southwest portions of the Hudson Plains Ecozone (Table 25).

Table 25. Dominant tree species (host trees) and associated pathogens indexed in north-central and north Ontario (sensu Davis and Myren 1990). The pathogens listed have potential for causing serious injury or death to their host and have been collected from more than one specimen.
Sources: Davis and Myren (1990, 1991); Myren and Davis (1991); Myren et al. (1994); and J.A. McLaughlin, Ontario Ministry of Natural Resources (unpublished data from NEBIE study[a]).

Host tree	Pathogens
Picea mariana (black spruce)	*Armillaria ostoyae* (armillaria root disease) *Inonotus tomentosus* (tomentosus root rot) *Arceuthobium pusillum* (eastern dwarf mistletoe) *Chrysomyxa ledicola* (spruce-Labrador tea needle rust)
P. glauca (white spruce)	*A. ostoyae* *I. tomentosus* *C. ledicola*
Larix laricina (eastern larch)	*A. ostoyae*
Populus tremuloides (trembling aspen)	*A. sinapina* (butt rot) *A. ostoyae* *Hypoxylon mammatum* (hypoxylon canker) *Phellinus tremullae* (heart rot)
Salix spp. (willow)	*Venturia saliciperda* (willow scab)

[a] For information on the NEBIE study, see Bell et al. (2008).

A number of disease agents occur in the contiguous forest ~100 km south of the Hudson Plains Ecozone (in the Boreal Shield Ecozone), which attack tree species that are also present in the Hudson Plains Ecozone. Tomentosus root rot affects a large proportion of black and white spruce, while armillaria root disease (caused by *Armillaria ostoyae*) commonly kills balsam fir (*Abies balsamea*), in addition to black and white spruce (Whitney 1995). Butt rot (caused by *Armillaria sinapina*, a disease well adapted to wet sites in central Ontario; McLaughlin 2001) is a serious problem for trembling aspen, while hypoxylon canker (caused by *Hypoxylon mammatum*) and phellinus heart rot (caused by *Phellinus tremullae*) are ubiquitous disease agents that also affect trembling aspen in the boreal forests of Ontario. *Chrysomyxa ledicola*, a disease agent of black spruce and its alternate host Labrador tea (*Ledum groenlandicum*, a ubiquitous shrub in the Hudson Plains Ecozone; Riley 2003), is also present in Ontario's boreal forests. These same disease agents probably also occur in the same host species in the Hudson Plains Ecozone, but their importance as disturbance agents there is unknown.

Models predict a significant effect of climate change on the occurrence of forest diseases (e.g., for Ontario, see Griefenhagen and Noland 2003). With increased warming, precipitation patterns are expected to change (Section 2.1, *Abiotic Drivers*), with an increase in extreme weather events (Section 2.4.2.1, *Extreme Weather*), including drought, high and rapidly fluctuating temperatures, and wind, ice, and hail storms (Boland et al. 2004). Plant disease occurrence is expected to change along with these patterns. For example, tomentosus root rot is predicted to increase with warmer and drier growing seasons, while hypoxylon canker is expected to increase as increased wind and hail damage provides more infection sites. Abiotic diseases associated with environmental stresses (i.e., diseases caused by non-infectious factors, such as nutrient deficiencies, air pollutants, and temperature and moisture extremes) are expected to increase as the climate warms, and it is the interactions between biotic and abiotic diseases that might represent the most important effects of climate on plant diseases (Boland et al. 2004).

Baseline data relevant to forest diseases in the Hudson Plains Ecozone are critically needed. As the ecozone's permafrost thaws with continued climatic warming (Gough and Leung 2002; Payette et al. 2004; Gagnon and Gough 2005), it is likely that trends could be observed in the frequency and relative importance of forest diseases and other natural disturbance agents in the ecozone – along with associated implications for biodiversity.

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2.4.3 **Community & population dynamics**

The following aspects of community and population dynamics in the Hudson Plains Ecozone are considered in this section: predator-prey relationships and cycles; herbivore-plant interactions; wildlife diseases and parasites; and phenology.

2.4.3.1 **Predator-prey relationships & cycles**

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Although northern ecosystems have relatively simple food webs, population dynamics and resulting interactions among species can be quite complex. Fluctuations in the populations of one species can affect the populations of other species within the community and overall trophic dynamics (Angerbjorn et al. 1999; Bêty et al. 2002; Roth 2003; Gauthier et al. 2004; Gilg et al. 2006). Among the mechanisms for regulating population abundance, predator-prey interactions are important biotic mechanisms (Andersson and Erlinge 1977; Korpimäki and Krebs 1996; Sinclair and Krebs 2002). In northern regions, mammal and bird populations can exhibit cyclic fluctuations that may be synchronous over large geographical areas (Elton and Nicholson 1942; Bergerud 1970; Boutin et al. 1995; Korpimäki and Krebs 1996; Krebs 1996; Stenseth et al. 1999).

A paucity of detailed information exists on most predator-prey relationships and cycles in the Hudson Plains Ecozone (Table 26), where monitoring of predator-prey interactions has not been part of a regular wildlife management program. The general predator-prey relationships are recognized, but the drivers and regularity of cycles, where present, are largely unknown. As such, it is difficult to know whether any changes or trends are occurring in predator-prey relationships and cycles in the ecozone. Recent surveys of some mammals, such as polar bear and seals, likely contain the best data sets on predator-prey relationships in a predatory mammal that uses the Hudson Plains Ecozone, recognizing that this particular relationship also includes the adjacent marine ecozone (Hudson Bay, James Bay, and Foxe Basin Ecozone; see Niemi et al. 2010). The best evidence of cyclic behaviour in mammals occurring in the Hudson

Table 26. Information status of selected predator-prey relationships in the Hudson Plains Ecozone. Sources: As indicated.

Predator-prey relationship	Information status	Reference
Polar bear : seal	Fair	Stirling & Derocher (1993); Derocher et al. (2004); Thiemann et al. (2008); McKinney et al. (2009)
Wolf, wolverine : caribou	Poor	–
Arctic fox : lemming	Limited	Roth (2002); Roth (2003)[a]
Birds of prey : lemming	Poor	Shelford (1943)
Lynx : snowshoe hare	Poor	Bulmer (1974)[b]
Mink : muskrat	Limited	Erb et al. (2001)[c]; Viljugren et al. (2001)[c]
Marten : vole	Poor	–

[a] Demonstrated correlation of annual variation (1994-1997) in abundance of fox and lemming in the Hudson Plains Ecozone near Churchill, Manitoba.

[b] Hudson Bay Company harvest data (1948-1907) showed that snowshoe hare populations cycle relative to lynx populations in the Hudson Plains Ecozone.

[c] Hudson Bay Company harvest data (1925-1949) showed that mink population cycles generally lagged one year behind muskrat population cycles in the Hudson Plains Ecozone. Numerical dependencies between the two species were strong in Erb et al. (2001).

Plains Ecozone is for lemmings and their predators (Scott 1993; Roth 2002, 2003; Reiter and Andersen 2008). Potential changes in the relationships between polar bear and seals and between arctic fox and lemmings in the ecozone are profiled below. As well, some new or uncommonly reported interactions occurring on land between polar bear and other species are identified.

2.4.3.1.1 **Polar bear & ringed seal**

Polar bear (*Ursus maritimus*) is the top predator in the marine ecosystem of Hudson Bay (Lunn et al. 1997; Derocher et al. 2004). It is the most carnivorous of the ursids and an opportunistic feeder that tracks changes in its prey populations (Stirling and Øritsland 1995; Ferguson et al. 2000; Amstrup 2003). In Hudson and James bays, ringed seal (*Pusa hispida*) makes up the majority of the polar bear's diet, with smaller amounts of other marine mammals, such as bearded seal (*Erignathus barbatus*), harbor seal (*Phoca vitulina*), and harp seal (*Phoca groenlandica*) (Thiemann et al. 2008). The utilization and importance of alternative prey (other seal and whale species), including terrestrial prey, is not well understood, but it currently represents a small portion of the polar bear's diet (Ramsay and Hobson 1991; Derocher et al. 1993; Brook and Richardson 2002; Thiemann et al. 2008; Hobson et al. 2009; Rockwell and Gormezano 2009; Smith, P.A. et al. 2010). As such, fluctuations in population numbers and reproductive rates of ringed seals can cause variation in the productivity of polar bears (Stirling and Øritsland 1995; Stirling 2002). Parameters for this predator-prey relationship have, however, not been described for the polar bear subpopulations that use the Hudson Plains Ecozone.

Polar bear body condition, reproductive rates, and survival have been declining in the Hudson Bay area over the past 15-20 years. These trends in polar bear are correlated with the declining extent and duration of sea ice in Hudson and James bays, implying that the effect is due to polar bears having less total time available on the sea ice to hunt seals and a longer period on land during the ice-free season, when they eat only opportunistically (Stirling et al. 1999; Parks et al. 2006; Obbard et al. 2006, 2007; Regehr et al. 2007; Section 2.3.3.1.1, *Polar Bear*). Ringed seal is also dependent on the sea ice, and it might be affected by these same climatic patterns (Ferguson et al. 2005; Chambellant 2010). In Hudson Bay, the sea ice regime, snowfall patterns, and spring temperatures might be driving a decadal cycle in ringed seal abundance and reproductive performance, with lows in the 1990s and improvements again in the 2000s (Smith, T.G. 1975; Lunn et al. 1997; Ferguson et al. 2005; Stirling 2005; Chambellant 2010).

Little quantitative trend data exist on the abundance or demography of other marine prey species in Hudson Bay (Niemi et al. 2010), but changes in the relative use of seal species by polar bear has been observed. In western Hudson Bay from 1991 to 2007, a dietary shift of polar bear away from ice-associated bearded seal towards more open-water seal species (such as harp and harbour seals) could be related to changes in sea ice conditions associated with climatic warming (Thiemann et al. 2008; McKinney et al. 2009). This diet shift is also believed to have altered contaminant levels in polar bears (see Section 2.3.3.1.1, *Polar Bear*). Despite the annual variation in sea ice duration and seal reproductive rates, ice-associated ringed seals have, on the other hand, continued to make a relatively steady contribution to the diet of polar bears in western Hudson Bay, suggesting that the other seal species might not be sufficiently abundant or available to replace ringed seal in the diet of polar bear (Thiemann et al. 2008; McKinney et al. 2009). Therefore, if the abundance or reproductive rates of ringed seal decline over the long-term due to continued and projected climatic warming in the Hudson Bay region (Gough and Wolfe 2001; Ferguson et al. 2005; Gagnon and Gough 2005), additional declines in the body condition, reproductive success, and abundance of polar bear could be expected (Derocher et al. 2004).

Although quantitative data are available on the distribution, abundance, and life history of polar bear and ringed seal in Hudson Bay, the effect of climate change on this predator-prey relationship remains uncertain (Derocher et al. 2004; Stirling and Parkinson 2006; Thiemann et al. 2008; Chambellant 2010; Peacock et al. 2010). Long-term demographic studies will be important to monitor and model the dynamic interactions of a changing climate, polar bears, seals, and other prey.

2.4.3.1.2 **Arctic fox & lemming**

Deteriorating ice conditions in Hudson Bay could also affect interactions between arctic fox (*Vulpes lagopus*) and Richardson's collared lemming (*Dicrostonyx richardsoni*). Sea ice is used by arctic fox as a foraging surface to hunt for seal pups or scavenge carcasses from the ocean, when their primary terrestrial food sources are at their cyclical low point (Roth 2002, 2003). At low lemming population levels, marine-derived food becomes a critical component of arctic fox diet for maintaining body condition and survival over winter. Birds and bird eggs could also become important terrestrial foods for arctic fox at low points in the lemming cycle (Roth 2002; see also Bêty et al. 2001; Lecomte et al. 2008) and, in areas with large colonies of breeding migratory geese, cached goose eggs can supplement arctic fox diets over winter (Samelius et al. 2007). This pattern of using cached goose eggs during winter was, however, not detected in the Hudson Plains Ecozone, where foxes appeared to utilize marine resources instead (Roth 2002). Increasing temperatures, and therefore decreasing sea ice, could disrupt this marine-terrestrial ecosystem food web dynamic (Roth 2002).

2.4.3.1.3 **Changes in other predator-prey relationships**

New or uncommonly reported predator-prey relationships are being increasingly documented by the people who live in or frequent the Hudson Plains Ecozone (e.g., Brook and Richardson 2002; Arnold et al. 2006; Rockwell and Gormezano 2009). Polar bear is observed stalking and chasing woodland caribou (*Rangifer tarandus caribou*) in Wapusk National Park (Brook and Richardson 2002) and depredating lesser snow goose (*Chen caerulescens caerulescens*) and Canada goose (*Branta canadensis*) nests and moulting geese and flightless goslings along the coast (Smith, A.E. and Hill 1996; Rockwell and Gormezano 2009; see also Smith, P.A. et al. 2010 and Section 2.4.3.4.1, *Animal Phenology*). Most of these observations have been indirectly attributed to a changing climate and, as the climate continues to change and environmental components are altered, predator-prey dynamics are likely to change through all trophic levels of the subarctic food web (Parmesan 2006).

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2.4.3.2 **Herbivore-plant interactions**

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Some herbivore-plant interactions are changing in intensity in the Hudson Plains Ecozone. Most notable is the well-studied relationship between lesser snow goose (*Chen caerulescens caerulescens*), a migratory keystone herbivore, and its forage. Since the 1970s, the resource limitation on this herbivore has been reduced on its wintering grounds and along its migration routes (south of the ecozone), due principally to increased crop yields there from the application of fertilizers (Jefferies et al. 2003). Responding largely to this agricultural food subsidy, the Mid-Continent population of lesser snow goose increased geometrically over the same period (Section 2.3.3.3, *Birds*). Such large numbers of geese are, however, not well supported during their seasonal period many thousands of kilometres north in the Hudson Plains Ecozone, where similar increases in food supply have not occurred.

As a result, goose foraging pressure in the ecozone has been intense enough to alter what is otherwise a synergistic relationship between this herbivore and its preferred graminoid salt marsh forage species, *Puccinellia phryganodes* and *Carex subspathacea*. At low to moderate grazing intensities, geese promote these graminoid species by removing apical meristems from competing dicotyledonous species (Bazely and Jefferies 1986) and by enhancing the net above-ground primary productivity of the graminoids (Cargill and Jefferies 1984; Hik and Jefferies 1990; Hik et al. 1991; see also Section 2.4.5, *Nutrient Cycling*). In turn, the geese depend on these graminoids for nitrogen in this nitrogen-limited environment. Intensification of goose herbivory, however, has led to severe and cumulative damage to about one-third of the lesser snow goose's preferred salt marsh habitat in the ecozone for the foreseeable future and its replacement with an alternative stable state of hypersaline bare sediment. A range of other local and meso-scale effects on the ecozone's coastal and tundra biomes are also evident, as detailed elsewhere in this report (see sections 2.2.2.1, *Coastal*; 2.2.2.2, *Polar-Tundra*; 2.3.3.3, *Birds*; 2.3.3.5, *Reptiles & Amphibians*; and 2.4.5, *Nutrient Cycling*).

Note that the mismatch that has occurred in the ecozone between lesser snow goose and its forage has been prolonged in duration, because the major limitations to this herbivore (i.e., resource availability, predation) have not changed sufficiently there to compensate for the increased goose numbers. Although some reduction in clutch size and gosling survival and growth are associated with the loss of salt marsh forage, lesser snow goose has been largely escaping local density-dependent processes by dispersing to other coastal salt marsh sites and moving inland to alternatively forage on less nutritious freshwater marsh vegetation in the tundra during the post-hatch (brood-rearing) period (Cooch et al. 1991, 1993, 2001; see also Jefferies et al. 2003). A sustained increase in snow goose predation has also not occurred in the ecozone. Traditional predators of snow goose in the ecozone, such as arctic fox (*Vulpes lagopus*), herring gull (*Larus argentatus*), and parasitic jaeger (*Stercorarius parasiticus*) (Cook et al. 1995), are generally not able to sustain increased numbers in response to elevated goose numbers, because geese are only available to them seasonally (Jefferies et al. 2003). Polar bear (*Ursus maritimus*), however, appears to be increasing in importance as a predator of snow goose eggs. The earlier annual break-up of sea ice that is associated with climate warming (Section 2.1, *Abiotic Drivers*) is forcing the bears ashore earlier in the season (Section 2.3.3.1.1, *Polar Bear*), such that their arrival is increasingly overlapping with the period when lesser snow geese are still incubating eggs (Section 2.4.3.4.1, *Animal Phenology*). Modelling suggests that the increasing predation of snow goose eggs by polar bear will reduce the size and reproductive output of nesting colonies along portions of the coast (e.g., Cape Churchill Peninsula; Rockwell et al. 2010). As such, increased polar bear predation might aid management attempts to lower snow goose abundance (Section 2.6.3.3, *Overview of Other Restoration Initiatives*) and, therefore, reduce local coastal habitat damage (Rockwell et al. 2010).

Knowledge of direct effects that mammalian herbivores have on plants in the Hudson Plains Ecozone is minimal. Some prominent interactions and implications can be described, but no quantitative trends are available. Simkin (1965) and Ahti and Hepburn (1967) report on the distributions of woodland caribou (*Rangifer tarandus caribou*) as a function of food (mainly lichen) quality and quantity. Sites with high lichen cover have a corresponding high carrying capacity for caribou. In the Hudson Plains Ecozone, Ahti and Hepburn's (1967) highest quality sites correspond closely with the Hudson Bay Lowland and Coastal Hudson Bay Lowland ecoregions (ecoregions are shown in Figure 36, Section 2.2.1.1, *Overview of Ecozone Structure*). Lichens are susceptible to overexploitation, and, indeed, Couturier et al. (1990) reported that local overexploitation and trampling of lichen on calving grounds likely contributed to declines in the George River herd of woodland caribou in northern Québec. In the Hudson Plains Ecozone, the Pen Islands herd of woodland caribou increased in size from the 1970s to the late 1990s in a fairly limited range near the Manitoba-Ontario border, but numerous aerial surveys conducted since 2000, including 2009, suggest that this herd has since shifted east and is in possible decline (see Section 2.3.3.2.1, *Caribou*). One contributing explanation might be a change in habitat quality due to overexploitation and trampling of ground lichen. A comparison of lichen quantity and quality in these portions of the Hudson Plains Ecozone could potentially provide information to assess this hypothesis and interpret these distribution changes.

Interactions between American beaver (*Castor canadensis*) and vegetation have also been reported for the Hudson Plains Ecozone. In addition to the direct effects beavers have on the primarily woody deciduous species they harvest (e.g., *Populus*, *Alnus*, and *Salix* spp.), their damming activities influence other vegetation in the area. Primary effects of beaver damming include pond formation upstream of the dam as a result of wetland depression flooding and a change in the course of water flow (both surface- and ground-water), with the potential of new channels being created by the run-off (Woo and Waddington 1990). While beavers have an important ecological role in creating and maintaining wetlands at landscape scales (Naiman et al. 1988), the ponds created by beaver damming have been inferred to facilitate the degradation of palsa mounds (Lewkowicz and Coultish 2004). In the Hudson Plains Ecozone, palsa degradation will affect localized successional processes on the mounds and also decrease polar bear habitat, because bears use hollowed-out palsa mounds as denning substrates. The recession of ponds after degradation of relict dams results in palsa mound aggrading (Lewkowicz and Coultish 2004); however, mature palsa mounds require more than a century to form.

Specific effects of herbivory by cricetine rodents (e.g., lemmings, voles) have not been studied in the Hudson Plains Ecozone. However, studies in Greenland on the collared lemming (*Dicrostonyx groenlandicus*) have shown that lemmings can periodically have negative impacts on their preferred food plants. Following high winter populations of lemmings, summer populations of patchily distributed mountain avens (*Dryas octopetala*), a primary food plant of lemmings, are significantly reduced (Berg et al. 2008). A similar relationship is not seen with lemming-willow interactions, likely due to the even distribution of willow compared to mountain avens, which therefore experiences a high intensity of grazing per unit area (Berg et al. 2008). Similar interactions might occur in the Hudson Plains Ecozone, where Richardson's collared lemming (*Dicrostonyx richardsoni*) and white mountain-avens (*Dryas integrifolia*) co-occur.

Notable insect-plant interactions occur in the Hudson Plains Ecozone and have been discussed separately (see Section 2.4.2.3, *Large-Scale Native Insect Outbreaks*). Important insect herbivore-plant interactions include: larch sawfly (*Pristiphora erichsonii*)-eastern larch (*Larix laricina*); spruce budworm (*Choristoneura fumiferana*)-coniferous trees (primarily balsam fir, *Abies balsamea*; black spruce, *Picea mariana*; white spruce, *P. glauca*); and forest tent caterpillar (*Malacosoma disstria*)-*Populus* species (trembling aspen, *Populus tremuloides*; balsam poplar, *P. balsamifera*).

Human influences on herbivore-plant interactions are not apparent in the ecozone, again with the exception of excessive foraging by migratory lesser snow goose, the population of which increased greatly over the last four decades principally due to human influences outside the ecozone (see earlier). Climate change could affect herbivore-plant interactions in the ecozone through earlier onset of vegetation growth, increased plant primary productivity, and changes in temporal availability of nutrients in forage plants, leading to possible mismatches with herbivore reproduction and growth (Post and Forchhammer 2008).

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2.4.3.3 **Wildlife diseases & parasites**

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A paucity of data exist on which to assess the status and trends of wildlife diseases in the Hudson Plains Ecozone. Consequently, there is currently no strong evidence that the distribution or importance of wildlife diseases and parasites is changing there. Of the wildlife diseases present in the ecozone, rabies is profiled most extensively below, in part owing to its national significance (Leighton in press). Other wildlife diseases and parasites that are present or likely present are acknowledged more briefly, in some cases drawing inference from studies in similar subarctic and arctic habitats. It will be important to gather more data on wildlife diseases and parasites in the Hudson Plains Ecozone, particularly because climate change will lead to environmental conditions suitable for some disease agents and range shifts in vegetation and animals, which in turn will affect disease ranges and transmission (Ogden et al. 2008).

2.4.3.3.1 **Rabies**

Rabies is a highly infectious disease of the central nervous system that is caused by a lyssa virus (WHO 2005). The disease can affect all mammals, and it is almost always fatal if left untreated. Several strains of the rabies virus occur in the wild, but each seems to be maintained in nature by only one or a few mammalian species, although more species are susceptible. In the Hudson Plains Ecozone, the arctic fox strain of rabies is present, and it is maintained there in populations of red fox (*Vulpes vulpes*) and arctic fox (*Vulpes lagopus*) (Canadian Food Inspection Agency, Centre of Expertise for Rabies, unpublished data).

Rabies was first diagnosed in Canada's central arctic in 1947 (Plummer 1954), but anecdotal evidence suggests that the disease had been enzootic in arctic wildlife long before this. Rabies was confirmed in the Hudson Plains Ecozone by the early 1950s. In 1951, rabies was diagnosed in specimens from Churchill, Manitoba, as well as in two communities in northern Québec that are outside of the ecozone (Sugluk and Povungnituk) (Tabel et al. 1974). In northern Québec, rabies spread south through populations of red fox, and rabies eventually entered Ontario around the southern tip of James Bay in 1954. In 1955, rabies also spread through red fox populations in northern Manitoba into the Ontario portion of the ecozone, but it then abated (Tabel et al. 1974).

Rabies reporting has been limited in the remote and sparsely settled Hudson Plains Ecozone, and no rabies control programs are in place there. As a case study, the few known rabies case locations between 1985 and 2008 are shown in Figure 110 for the Ontario portion of the ecozone and the adjacent area of the Boreal Shield along Highway 11. The number of specimens submitted for laboratory testing and the number that tested positive for rabies each year are summarized in Table 27, for the same geography and time period. Unfortunately, only 52 of the 138 confirmed rabies cases in this northern area have associated geographic locations (compare the number of samples in Table 27 with the number of mapped points in Figure 110). Of those 52 cases, only eight are within the Hudson Plains Ecozone. The most recent rabies case reported in the Ontario portion of the ecozone (for this 1985-2008 data set) was a red fox from Fort Severn in 2008 (Figure 110a). The last group of rabies cases prior to that was located within 35 km of the Moosonee area. There, positive tests were obtained for two grey wolves (*Canis lupus*) in 1991, four red foxes in 1996, and one red fox in 2000 (Figure 110b). Between 1985 and 2008, 10 arctic foxes and five additional wolves also tested positive for rabies in northern Ontario, but specific location information for these records was unavailable. Based on distribution information for these species (Carbyn 1999; Garrott and Eberhardt 1999), the red foxes could also have been from the Hudson Plains Ecozone, and the arctic foxes must have been, as this latter species does not occur in any other ecozone in Ontario. The remaining 44 rabies cases in northern Ontario with known locations were south of the Hudson Plains Ecozone, within the Boreal Shield Ecozone, not far from the Québec border. Confirmation dates for these latter rabies cases ranged from 1994 to 2002, with almost all of these reported cases being located close to roads and towns along Highway 11 between Ramore and Hearst.

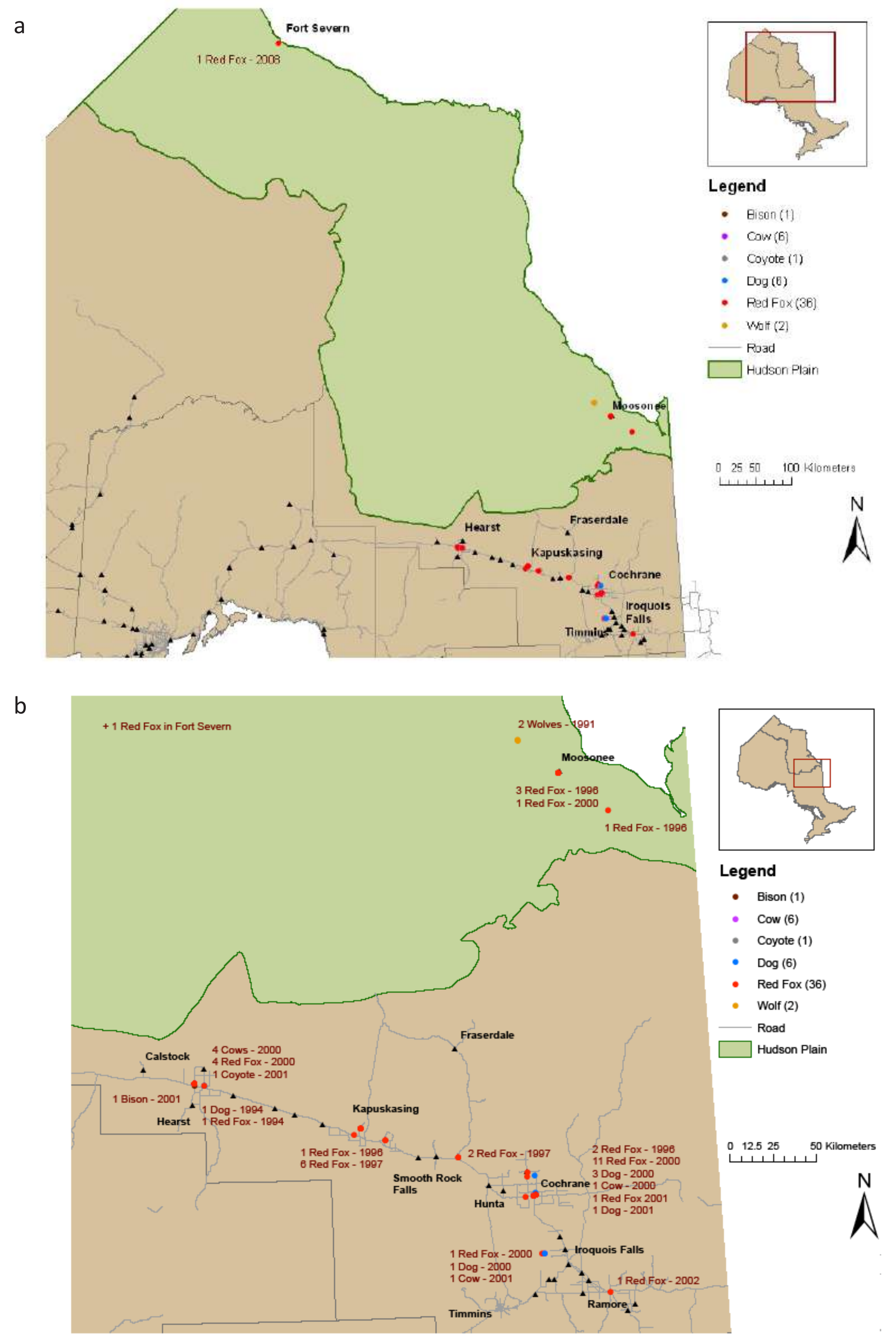

Figure 110. Rabies cases in the Ontario portion of the Hudson Plains Ecozone and the adjacent Boreal Shield Ecozone (ecozones⁺ boundaries) along the Highway 11 corridor, 1985-2008: a) positive case locations and b) number of species affected and confirmation dates for groups of rabies positive cases (map does not geographically show the red fox case in Fort Severn in 2008).
Source: Data are from the Canadian Food Inspection Agency, Centre of Expertise for Rabies.

Table 27. Number of animals submitted for rabies testing and the number of specimens that tested positive, from the Ontario portion of the Hudson Plains Ecozone and the adjacent Boreal Shield Ecozone (ecozones$^+$ boundaries), 1985-2008. Red fox (RFX) appeared to be the main wildlife vector of rabies during this time period. Shading denotes three apparent outbreak periods (see text). Abbreviations: AFX, arctic fox; BIS, bison; BOV, cow; CAT, cat; CYT, coyote; DOG, dog; FOX, fox (species unknown); GHG, groundhog; RFX, red fox; SSK, striped skunk; and WLF, wolf.
Source: Data are from the Canadian Food Inspection Agency, Centre of Expertise for Rabies.

Year	Total number submitted	Number testing positive for rabies	Species										
			AFX	BIS	BOV	CAT	CYT	DOG	FOX	GHG	RFX	SSK	WLF
1985	17	1										1	
1986	21	3						1				2	
1987	23	0											
1988	28	1									1		
1989	15	0											
1990	27	3	2								1		
1991	25	2											2
1992	75	13				1		3			7	1	1
1993	53	17									14	2	1
1994	30	8			1			1			6		
1995	22	2						1			1		
1996	34	11						1			10		
1997	26	14	1					1	1		11		
1998	15	2	1								1		
1999	16	5	2								2		1
2000	68	37			5			6		1	23	1	1
2001	41	9	1	1	2		1	1			2		1
2002	27	3	1								2		
2003	16	2	1								1		
2004	14	1									1		
2005	11	3	1					1			1		
2006	6	0											
2007	3	0											
2008	unknown	1									1		
Total Positive		138	10	1	8	1	1	16	1	1	85	7	7

Elton (1931) made one of the first observations that cycles of the *arctic dog disease* (rabies) coincided with population densities of foxes, and that the cycles varied between 3 and 5 years, but mostly lasted 4 years. More recent reports suggest that the cycle of rabies in wildlife populations might vary between 2 and 7 years (Steck and Wandeler 1980; Mørk and Prestrud 2004). Red fox appears to be the main wildlife rabies-maintenance host species, both in the Hudson Plains Ecozone and in the area immediately to the south, based on the number that tested positive for rabies between 1985 and 2008 (Table 27, Figure 110). Three recent rabies outbreak events appear to have occurred in northern Ontario at 3-4 year intervals: 1992-1993, 1996-1997, and 2000 (Table 27). There also appears to be 7 years of relatively low numbers of rabid foxes before and after the overall grouping of high numbers of fox rabies cases. Data prior to 1985, if available, would allow a more detailed interpretation of possible rabies cycles in this area.

Although epizootics of rabies appear widespread among arctic and red fox populations (Secord et al. 1980), the true incidence in the Hudson Plains Ecozone is likely to remain unknown for some time, given the sparse human population and the vastness and remoteness of the area. The causes and maintenance of rabies outbreaks in such isolated areas are not well known. Disease transmission is generally elevated when predator populations rise due to migration in search of food or when animals congregate around an abundant food source. Human influences, such as garbage dumps, litter, and handouts, provide artificial food sources for both arctic and red foxes and could result in large aggregations of animals, depending on the area, time of year, and availability of natural food items. Unfortunately, at high fox densities, the rabies virus is easily transmitted between wild and domestic animals through a bite from an infected individual, consumption of contaminated meat, or by the virus coming in contact with mucous membranes.

How the rabies virus is maintained in animal populations through periods of low population density is unknown. Although rodents and other small mammals have been proposed as reservoirs of rabies infection (Elton 1931; Rausch 1958), little is known of the natural occurrence of rabies in these species groups. Rabies antibodies have been found in arctic foxes that appeared healthy at the time of capture, suggesting that some foxes must have survived virus exposure or that the virus has a very long incubation phase in some animals (Ballard et al. 2001).

Little is also known about the incubation period of the rabies virus in wildlife in subarctic regions. Rabies can be acquired by eating carcasses of infected animals, especially the brain. Rates of virus survival under varying climate regimes is currently unknown, however, the cold arctic climate is said to favour virus survival (Tabel et al. 1974). Thus, a warming climate might affect the distribution and abundance of prey items, which in turn might cause changes in the distribution and abundance of the main wildlife rabies vector species and transmission of the rabies virus.

Red fox, the apparent major rabies vector species in the Hudson Plains Ecozone, has a high fecundity and dispersal potential, so it generally fares well in the face of hunting and trapping pressure, changes in local food supply, and/or disease (Voigt 1999). The red fox was a major rabies-maintenance species in southern Ontario in the late 1980s and early 1990s, spreading the disease to domestic livestock, family pets, and other wildlife. Rabies control measures were implemented there (large-scale oral rabies vaccination of red foxes), and the incidence of the disease was reduced to near zero (Bachmann et al. in preparation). In the absence of similar rabies control measures in the north, the Canadian arctic will most likely continue to be a source of rabies infection in the Hudson Plains Ecozone. Given the resilience of red fox populations, rabies will likely continue to cycle in both this ecozone and the adjacent Boreal Shield Ecozone, affecting both wildlife and domestic animals.

2.4.3.3.2 **Renal coccidiosis & parasitic nematodes**

The protozoan parasite, *Eimeria truncata*, causing renal coccidiosis in waterfowl, has been studied in the lesser snow goose (*Chen caerulescens caerulescens*) ~250 km north of Churchill (Gomis et al. 1996). The study showed that the incidence of renal coccidiosis and death from the disease are greater in geese along the coast of Hudson Bay than at inland breeding sites (Gomis et al. 1996). Similarly, Mellor and Rockwell (2006) showed that the parasite loads of two nematodes (*Trichstrongylus tenuis* and *Heterakis* sp.) were higher in coastal compared to inland colonies of lesser snow goose, although individual or population effects of infection were not examined. Both Gomis et al. (1996) and Mellor and Rockwell (2006) suggest that greater infection and/or mortality rates of geese along the coast could be a result of the up to ten-fold higher amount of goose faeces there (through which the parasite can be transmitted) (Gomis et al. 1996), which is associated with much higher goose densities compared to inland. A second reason could be overexploitation by geese of their food source. Excessive grazing and grubbing of coastal-intertidal vegetation by a greatly expanded lesser snow goose population is resulting in poor habitat and a reduction in food source (Section 2.2.2.1, *Coastal*), which could, in turn, lead to malnourished geese and goslings with less ability to resist infection (Gomis et al. 1996). Finally, Mellor and Rockwell (2006) further suggest that the infective stage of *T. tenuis* and *Heterakis* sp. could be more tolerant of, or more infective in, coastal salt marshes than inland, freshwater marshes.

2.4.3.3.3 **Avian cholera**

The bacterial parasite, *Pasteurella multocida*, which causes avian cholera, is common in Canada goose (*Branta canadensis*) and lesser snow goose (Brand 1984; Samuel et al. 1999). Brand (1984) reported that Canada geese of the Eastern Prairie and Mississippi Valley populations that were returning from their breeding grounds in northern North America, including the Hudson Plains Ecozone, were infected. This observation suggests that at least a portion of these populations that nest in the ecozone is likely afflicted with the disease. Moreover, Samuel et al. (1999) suggest that the disease is more prevalent at breeding sites where bird densities are high (such as in the Hudson Plains Ecozone), because disease transmission is likely to occur more readily via direct bird-to-bird contact or indirectly through shared feeding or drinking sources. Indeed, at their study site on Banks Island, Northwest Territories, mortality was 20% where lesser snow goose density reached 120 geese/ha, while mortality rarely exceeded 2% at goose densities of up to 45 geese/ha (Samuel et al. 1999).

2.4.3.3.4 **Brucellosis**

The bacterial pathogen, *Brucella suis* biotype 4, causing brucellosis, is known to affect barren-ground caribou (*Rangifer tarandus groenlandicus*). The range of the Beverly-Qamanirjuaq (Kaminuriak) barren-ground caribou herd (Broughton et al. 1970) extends peripherally into the northwestern most portion of the Hudson Plains Ecozone (see Figure 77 in Section 2.3.3.2.1, *Caribou*), and indeed, the presence of the disease in a single individual barren-ground caribou was reported there near Churchill (Forbes 1991). Surveys for the presence of the disease in woodland caribou *(Rangifer tarandus caribou)* have not been conducted in the Hudson Plains Ecozone. The disease is uncommon, and it has minimal effects on wild caribou populations (Broughton et al. 1970), but it could be a more significant threat to domestic game-ranched species (Forbes 1991). Effects of the disease on caribou include, with varying frequency: abortion, placental retention, metritis, sterility, orchitis, epididimytis, and burcitis. Transmission of the pathogen is optimal under cool climatic conditions (Forbes 1991), but it is unknown what effect climate change will have on the frequency of pathogen transmission.

Brucellosis has recently been found in arctic marine mammals (whales, walruses, seals), but the *Brucella* species causing the disease is different (Forbes et al. 2000). For more information, see the Arctic-Marine Ecozones report of the ESTR (Niemi et al. 2010).

2.4.3.3.5 **Trichinellosis**

The parasitic nematode, *Trichinella nativa*, which causes trichinellosis in a variety of arctic and marine mammals, has a circumpolar distribution (Forbes 2000) and is found in the Canadian subacrtic (Reichard et al. 2008). As such, the pathogen is likely found in mammals in the Hudson Plains Ecozone, even though no cases have been reported there to date. Within the ecozone, arctic fox and polar bear (*Ursus maritimus*) are candidate species for acquiring the pathogen. Arctic fox typically acquires the pathogen through scavenging on tainted meat, while polar bear is known to acquire the pathogen from feeding on infected ringed seals (*Phoca hispida*) and by cannibalising on cubs (Reichard et al. 2008). The disease is rarely fatal.

2.4.3.3.6 **Distemper**

Morbilliviruses, which cause distemper, have not been reported specifically in the Hudson Plains Ecozone, but they are likely present there, given that antibodies for the viruses have been found in animals in other arctic environments (Garner et al. 2000; Cattet et al. 2004). Distemper can afflict canids and bears (e.g., grey wolf, red fox, arctic fox, and polar bear) or phocine species (e.g., ringed seal), all of which are associated with the Hudson Plains Ecozone. Notably, there has been no evidence of outbreaks of distemper in polar bears or seals in arctic Canada (Cattet et al. 2004). The disease can cause mass morbidity and mortality, which would affect reproductive success at the population level (Garner et al. 2000). For more information on morbilliviruses in marine mammals, refer to the Arctic Marine Ecozones report of the ESTR (Niemi et al. 2010).

2.4.3.3.7 **External parasites: fleas & ticks**

External parasites might also affect wildlife in the Hudson Plains Ecozone now and in the future. For example, nest-inhabiting fleas (*Ceratophyllus vagabundus vagabundus*) are known to be present in great numbers in the nests of migratory lesser snow goose colonies in north-central Nunavut (Harriman et al. 2008). Harriman et al. (2008) speculate that climate change will facilitate range expansion of dwarf birch *(Betula nana exilis)*, which is a correlate for flea abundance. Therefore, *C. v. vagabundus* might become a more important parasite in northern latitudes, with warming temperatures and increasing goose populations.

Winter tick (*Dermacentor albipictus*) is common on moose (*Alces alces*) in North America, but it is seldom found on woodland caribou, which appears to be a suboptimal host (Welch et al. 1990). Effects of the tick on moose and caribou populations are normally minimal, but they typically involve emaciation and hair loss near the area of infection (Welch et al. 1990). The occurrence of the tick in the Hudson Plains Ecozone has not been studied, but Welch et al. (1990) suggest that the tick could emerge as a larger problem in northern latitudes as climate change brings earlier springs and later falls (conditions that enhance tick survival). Moreover, climate change models of *Ixodes scapularis*, the tick-vector of the Lyme disease bacterium (*Borrelia burgdorferi*), project that populations of this tick could be present in the Hudson Plains Ecozone via bird-borne transmission between 2000 and 2019, and that they might establish as low- and moderate-risk[66] populations as early as 2020-2049 and 2050+, respectively (Ogden et al. 2008).

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[66] Risk level is calculated as a function of modeled temperature suitability, tick immigration on migratory birds, and percent cover of forest habitat (Ogden et al. 2008).

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2.4.3.4 **Phenology**

Phenology refers to animal and plant life cycle events that are triggered by environmental changes, especially temperature (Schwartz 2003). Examples include bird migrations and other seasonal wildlife movements; budburst, flowering, and fruiting in plants; and hatching of larvae. Changes in phenology can be indicative of ecosystem response to global and local changes in climate and weather; advancing phenology is being increasingly reported worldwide, as the climate continues to change (e.g., for bird migration, see Murphy-Klassen et al. 2005 and Jonzén et al. 2006). However, differential phenological responses to climate change of species at different trophic levels can lead to changed interactions among species (food webs) (Both et al. 2009).

Limited but important changes in the ecozone's climate occurred over the 1950-2007 period analyzed for trends for the ESTR, along with associated changes in the timing of sea ice break-up and/or freeze-up (Section 2.1, *Abiotic Drivers*). Included among the significant trends were increased mean annual and/or mean seasonal temperature, depending on the location within the ecozone. Only one station (Moosonee) was analyzable for trends in growing season over the same period. Although trends in growing season were not significant at this station near the southern James Bay coast (cf. coarse-scale satellite analysis suggesting a general tendency for an earlier start to spring in the ecozone, at least in recent decades; Reed 2007), this station did show a significant increase in effective growing degree-days (>5 °C) over the same period[67]. The analysis of changes in tree phenology later in this section considers climatic trends over a

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[67] The number of effective growing degree-days is measure of accumulated heat during the growing season and available for plant growth.

somewhat longer period (1921-2008) than the ESTR climate analysis (1950-2007)[68], but in both analyses general trends for increasing temperatures and growing degree-days are similar. Although data are insufficient for examining trends in freshwater ice and permafrost in the Hudson Plains Ecozone (Section 2.1, *Abiotic Drivers*), this ecozone is an area where there is likely to be broad correspondence among cryospheric/hydrologic metrics and phenology (White et al. 2009).

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2.4.3.4.1 **Animal phenology**

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With few exceptions, animal phenology is not well monitored or otherwise studied in the Hudson Plains Ecozone. Trends are, however, evident in the reproductive (spring) phenologies of both lesser snow goose (*Chen caerulescens caerulescens*) and Canada goose (*Branta canadensis*) and in the timing of the seasonal movement of polar bear (*Ursus maritimus*) from sea ice onto land. These three species are among the best studied in the ecozone.

In terms of goose phenology, long-term monitoring data from the Cape Churchill Peninsula in Manitoba indicate that the mean hatching date (and therefore nesting period) of lesser snow goose has significantly advanced by 0.16 ± 0.07 d/yr over the period 1968-2007 (Figure 111). Canada goose nesting dates have likewise advanced in the ecozone by 0.50 ± 0.25 d/yr over the period 1993-2010 at Akimiski Island, Nunavut (Figure 112). Goose nest initiation is correlated

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[68] Climate trends analysis for the ESTR was completed similarly for all ecozones in Canada (ecozones⁺ framework) for the period 1950-2007, so as to provide the best possible spatial coverage across the nation for the longest period possible (relatively few weather data exist for some areas of northern Canada prior to 1950).

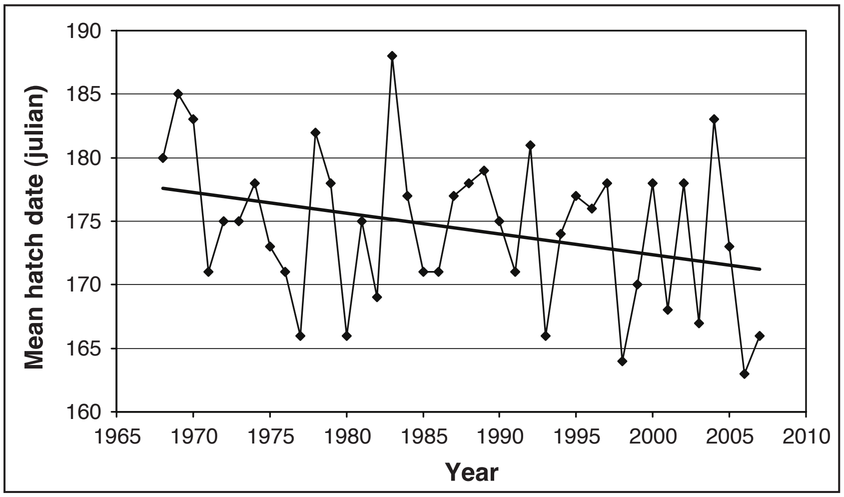

Figure 111. Trends in mean hatching dates of lesser snow geese nesting on the Cape Churchill Peninsula, 1968-2007. Mean hatching date has advanced 0.16 ± 0.07 d/yr in 40 years (p=0.048). Source: Rockwell and Gormezano (2009). Reprinted from Polar Biology, The early bear gets the goose: climate change, polar bears, and lesser snow geese in western Hudson Bay, Vol 32, 2009, p 543, R.F. Rockwell and L.J. Gormezano, figure 2, with permission from Springer Science+Business Media.

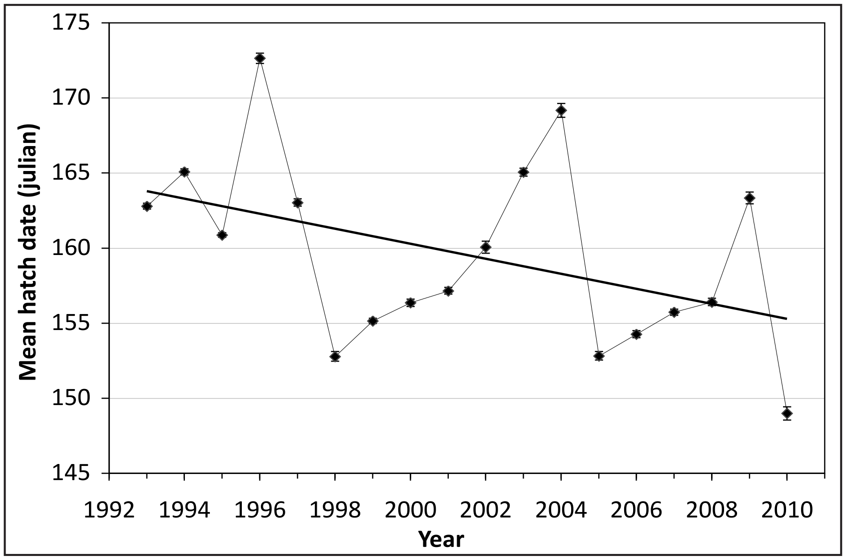

Figure 112. Trends in mean hatching dates of Canada geese nesting at Akimiski Island, 1993-2010. Mean hatching date has advanced 0.50 ± 0.25 d/yr in 17 years (p=0.04). Source: Ontario Ministry of Natural Resources (unpublished data).

with temperature and snow cover (Skinner et al. 1998) and, therefore, occurs earlier in years of early melt. However, although geese exhibit a general trend for advancing reproductive phenology, temperature anomalies associated with climate change could continue to result in years with late melt and thaw and, thus, halted migration and/or nesting (Jefferies and Rockwell 2002; see also Rockwell et al. 2010). In such years, millions of migrating geese (other than just local breeding individuals) can be held-up in the Hudson Plains Ecozone on their northward journey, which exacerbates the effects of goose foraging in the ecozone (Section 2.2.2.1, *Coastal*).

The seasonal movement of polar bears from sea ice to land is also occurring earlier, coincident with the earlier seasonal break-up of sea ice in Hudson and James bays (Section 2.3.3.1.1, *Polar Bear*). Polar bears now have less time on the sea ice to catch ringed seals (*Phoca hispida*), their primary prey, and, thus, to put on fat stores to support their seasonal period on land, when they eat only opportunistically. Given that trends in sea ice are projected to continue (Gagnon and Gough 2005; Joly et al. 2010), so too is the trend for earlier seasonal movements of polar bear from sea ice onto land. As such, the current trends for deterioration in polar bear subpopulations are also expected to continue or accelerate (Stirling and Parkinson 2006; Obbard et al. 2007; Molnár et al. 2010; Section 2.3.3.1.1, *Polar Bear*).

Interestingly, the above noted trends in the phenologies of polar bear and geese are affecting interactions between these species. Given that the break-up of sea ice and the movement of polar bears onto land is advancing (0.59 ± 0.05 d/yr for the western Hudson Bay subpopulation) 3.7 times faster than the nesting period of lesser snow goose (0.16 ± 0.07 d/yr for the Cape Churchill colony) (Rockwell et al. 2010), the earlier arrival of polar bears on shore in spring is increasingly coinciding with the period when lesser snow geese are still incubating eggs (Figure 113) (see also Smith et al. 2010 for areas further north). Goose eggs might, therefore, provide earlier arriving polar bears with an exploitable and abundant food source that was not available to them in the past. Indeed, polar bears have been increasingly observed consuming eggs from both lesser snow goose and Canada goose nests along the coast (Smith and Hill 1996; Rockwell and Gormezano 2009; Ontario Ministry of Natural Resources, unpublished data). However, although this newly exploitable and abundant food source might partly offset the energy shortfall of polar bears and stabilize their subpopulations temporarily, it is likely that other food sources, both terrestrial and marine, will need to play a role if polar bear is to persist (Rockwell and Gormezano 2009; Rockwell et al. 2010). Moreover, because goose nesting likely responds to

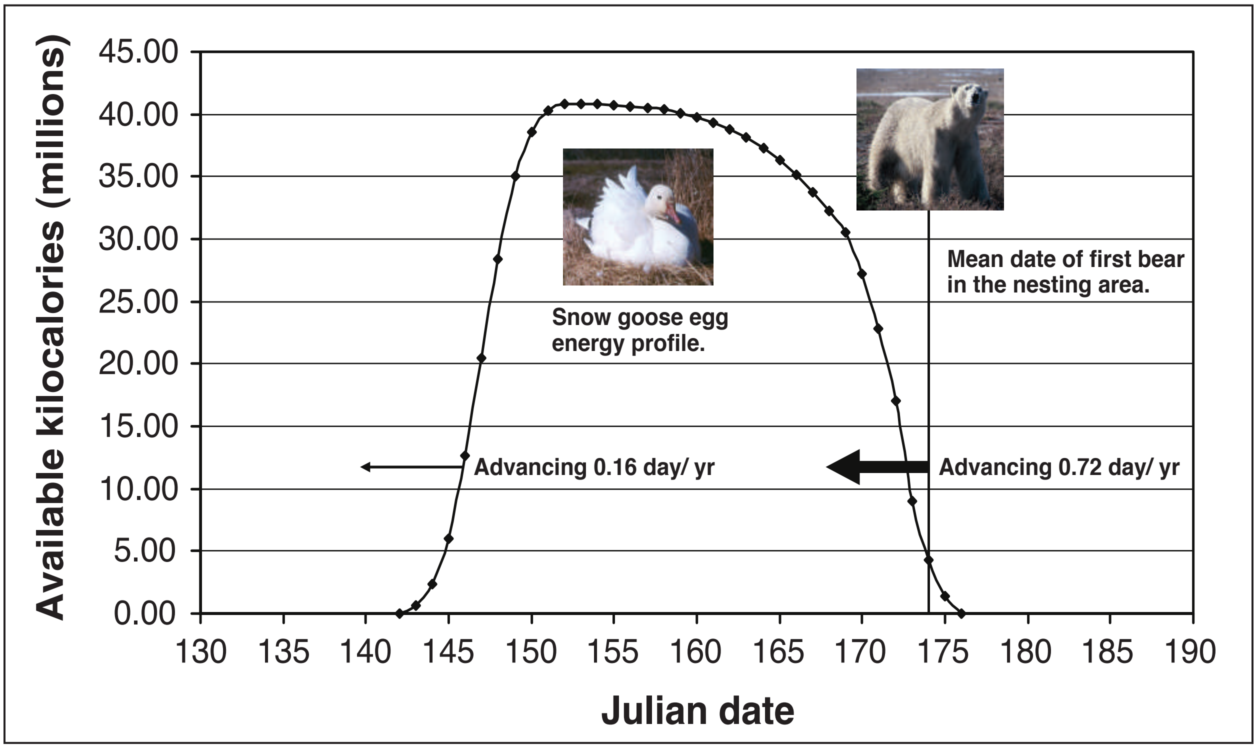

Figure 113. Diagrammatic representation of polar bears beginning to overlap the nesting period of lesser snow geese on the Cape Churchill Peninsula. As the advance of onshore arrival of polar bears is much faster than the advance in the nesting period of the geese (~4.5 times in this analysis or 3.7 times faster when assessed with a stochastic regression model in Rockwell et al. (2010)), the amount of energy available to the bears from snow goose eggs will increase as the overlap with the nesting period becomes earlier. The energy profile of eggs and the date on which the first polar bear was seen in the nesting area are averages for the period 2000 to 2007. The mean hatching date is June 21 and mean date for the first bear's arrival is June 23.

Source: Rockwell and Gormezano (2009). Reprinted from Polar Biology, The early bear gets the goose: climate change, polar bears, and lesser snow geese in western Hudson Bay, Vol 32, 2009, p 544, R.F. Rockwell and L.J. Gormezano, figure 4, with permission from Springer Science+Business Media.

variables other than just those affecting the timing of annual sea ice break-up, periodic annual mismatches are expected in the timing of goose nesting and polar bear arrival on shore, even if such mismatches will occur less frequently as climate change progresses (Rockwell and Gormezano 2009; Rockwell et al. 2010). Still, near-term forecasting (25 years into the future) with a stochastic model that takes into account changes in sea ice, polar bear subpopulations, and lesser snow goose phenology suggests that all but trivial rates of polar bear egg predation will reduce (but not eliminate) the size of the local nesting population of lesser snow goose in the Cape Churchill region of the ecozone (Rockwell et al. 2010).

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2.4.3.4.2 **Plant phenology**

Tree phenology

Rongzhou Man, Ontario Ministry of Natural Resources

In the northern hemisphere, activation of bud growth in spring is driven primarily by temperature, while the development of bud dormancy in the fall is induced by lengthening nights and decreasing temperatures (Perry 1971). Thus, trees will lose cold hardiness sooner and break buds earlier in spring and delay dormancy development in the fall in response to rising temperatures, as a result of climatic warming. Comparatively, flushing buds and growing shoots in the spring are much more sensitive to freezing temperatures than mature needles and dormant buds in the fall (Glerum 1973; Dang et al. 1992).

There have been no systematic, long-term direct observations of the phenology of tree species in the Hudson Plains Ecozone. The budbreak time of trees in spring can, however, be approximated from the thermal requirements of particular species in terms of growing degree-days and daily air temperature data (Colombo 1998). Table 28 provides the estimated budbreak time (EBD) for white spruce (*Picea glauca*), since 1920, for two locations in the Hudson Plains Ecozone and, for comparative purposes, three locations in the adjacent and more southerly Boreal Shield Ecozone.

Table 28. Monthly average air temperatures and estimated budbreak time in days (EBD) for white spruce, since early last century, for selected locations in the Hudson Plains Ecozone and the adjacent southern Boreal Shield Ecozone.
Source: Data are from Environment Canada.

Area	Period (years)	Monthly average air temperature (°C)				EBD
		March	April	May	June	
Hudson Plains Ecozone						
Churchill, Manitoba	1981-2008 (28)	-19.06	-9.78	-0.91	7.09	165
	1951-1980 (30)	-20.01	-10.08	-1.44	6.16	170
	1921-1950 (20)	-20.32	-11.13	-1.60	5.47	175
Moosonee, Ontario	1981-2008 (25)	-11.53	-1.90	6.38	12.08	148
	1951-1980 (30)	-12.26	-2.24	5.66	11.92	152
	1921-1950 (23)	-12.33	-3.29	4.78	11.45	157
Boreal Shield Ecozone						
Kapuskasing, Ontario	1981-2008 (28)	-8.74	0.72	8.91	14.52	141
	1951-1980 (30)	-9.53	0.43	8.30	14.11	144
	1921-1950 (30)	-9.75	-0.45	7.71	13.98	146
Timmins, Ontario	1981-2008 (27)	-7.63	1.67	9.52	14.90	139
	1951-1980 (30)	-8.35	1.19	9.10	14.75	142
	1921-1950 (27)	-8.22	0.14	8.14	14.82	143
North Bay, Ontario	1981-2008 (27)	-4.78	3.81	11.03	16.31	133
	1951-1980 (30)	-5.26	3.14	10.54	15.65	137
	1921-1950 (24)	-5.24	2.76	10.63	16.34	136

The long-term (1921-2008) weather data in Table 28 suggest that air temperatures are generally getting higher in spring (i.e., March-June in this analysis), especially since 1981. The average increase in March to June temperatures over the past 90 years in the Hudson Plains Ecozone was about 1.2 °C at Churchill and 1.1 °C at Moosonee (see also Section 2.1, *Abiotic Drivers*, for an analysis of climate trends for the period 1950-2007). Comparatively, the temperature increase in the Boreal Shield Ecozone was 1.0 °C at Kapuskasing, 0.9 °C at Timmins, and only 0.5 °C at North Bay, which is furthest south. The advance of EBD is therefore also greater in colder areas, ranging from 4-5 days every 30 years in Churchill and Moosonee (Hudson Plains Ecozone), to 2-3 days in Kapuskasing and further south, to about 1 day in Timmins and North Bay (Boreal Shield Ecozone). Other tree species, including woody shrubs, likely respond to rising temperatures in a similar way.

Given the increases in temperature projected with climate change for the Hudson Bay area (Gagnon and Gough 2005; Colombo et al. 2007; and see Section 2.1, *Abiotic Drivers*), budbreak of trees could further advance. Earlier budbreak could result in earlier loss of cold hardiness and increased risk of frost damage in spring, if the projection of increasing climatic variability and extreme weather events hold (Section 2.4.2.1, *Extreme Weather*). On the other hand, the potentially longer growing season, along with warmer temperatures and more precipitation (Gagnon and Gough 2005; Colombo et al. 2007), could favour tree growth.

One modelling exercise projects that earlier budbreak will not cause increased risk of freezing damage, but only after the climate has warmed substantially (Colombo 1998). In the nearer term, there is the possibility of increased risk of spring frost damage for white spruce in some boreal locations (Colombo 1998; Man et al. 2009). Increasing frost damage could restrict the capacity of trees to utilize longer and warmer growing seasons and even eliminate some early-flushing species from the ecosystem after several years of repeatedly severe frost events, thereby reducing productivity and biodiversity.

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Flowering plant phenology

Kim Monson, University of Winnipeg

Similar to the situation for trees, there has been no systematic, long-term monitoring of the phenology of the shorter-stature flowering plants in the Hudson Plains Ecozone. PlantWatch North is supporting casual surveys of first flowering dates for a number of indicator plant species in the Churchill area (Inset 13). The available data from this program are, however, currently too limited to support trend assessment.

Inset 13. **Phenological assessment of PlantWatch North data from Churchill, Manitoba**

Kim Monson, University of Winnipeg

Scientists are predicting that the some of the greatest increases in temperature due to climate change will occur in Northern Canada (e.g., Christensen et al. 2007). Increases in temperature will affect the onset of spring in the north and the timing of first bloom of many northern plants.

Plantwatch North was established to monitor the onset of spring by tracking the first and full bloom dates of a group of spring-flowering indicator plants. Bloom dates are currently being observed and recorded near the Churchill Northern Studies Centre, in Churchill, Manitoba, near the northwestern extent of the Hudson Plains Ecozone.

Some early data from this program are shown in Figure 114. The available data are limited to 6 years, and the results are preliminary. As such, caution should be exercised when using these early results to interpret climatic or environmental change. Stronger trend assessment will be enabled by longer-term observations from this program.

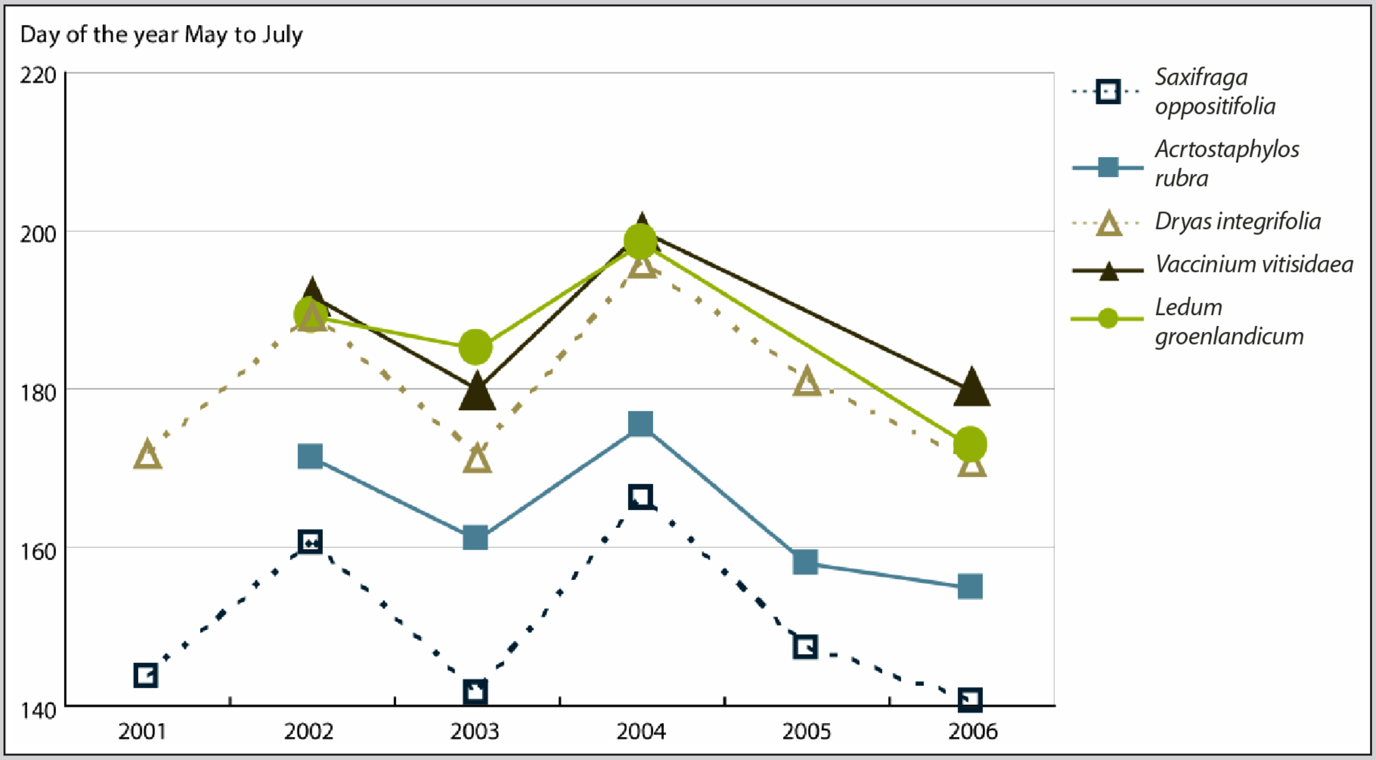

Figure 114. First bloom dates (Julian days) for five spring-flowering indicator plants from 2001 to 2006 near Churchill, Manitoba. Data were not available for three of the species in 2001 and two in 2005.

Reference

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2.4.4 Carbon cycling

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Jim W. McLaughlin, Ontario Ministry of Natural Resources

The Hudson Plains Ecozone represents Canada's largest peat basin and the second largest peat basin in the northern latitudes (>40-50°) and the world (Gorham 1991; Glooschenko et al. 1994). This massive expanse of peatland contains approximately 33 Gt of soil carbon [C] or 12% of the organic C stored in Canadian soils (Tarnocai et al. 2009). Understanding C-cycling (Figure 115) in this ecozone is critically important, because the implications could be large, with even a small change in this massive store of C, for both regional and global C budgets and climate change. Such an understanding requires knowledge of the diversity of peatland forms, their structure, and their interactions with hydrology. Peatland forms and their succession were described in Section 2.2.2.4.1, *Wetlands (Freshwater)*. This section focuses on the status and trends of key components of the C cycle in the Hudson Plains Ecozone, such as above-ground biomass production, net primary productivity, and decomposition.

2.4.4.1 Key peatland carbon cycle components & their status

2.4.4.1.1 Production

Aboveground biomass is variable across the ecozone's peatland chronosequence (from open water to marsh, then fen, and eventually bog). Biomass is lowest in sedge-dominated coastal marshes and open rich fens, moderate in shrub-dominated fens and bogs, and highest in shrub- and tree-dominated fen and bogs. Differences in the amount of biomass can be huge, ranging from 760 g/m^2 in open peatlands to 138,000 g/m^2 in forested peatlands (Grigal et al. 1985). Net primary production (NPP), or annual biomass produced, is less variable. Remote sensing analyses across the entire Hudson Plains Ecozone estimated NPP at 138 ± 84 $g\ C/m^2/yr$ (Liu et al. 2002). This estimate falls within the range of the ground-based measurements of NPP from the eastern part of the ecozone (range of ~50 to 100 $g\ C/m^2/yr$; Klinger et al. 1994) and northern Manitoba (range of ~125 to 275 $g\ C/m^2/yr$; Camill et al. 2001).

2.4.4.1.2 Decomposition

Peatland types differ in the magnitude of decomposition and the contributions of autotrophic versus heterotrophic respiration to carbon dioxide [CO_2] effluxes (Moore and Basiliko 2006). Depending on the overlying vegetation, belowground CO_2 production from root and rhizosphere processes can result in a similar amount of CO_2 as the microbial decomposition of peat (Crow and Wieder 2005; Johansson et al. 2006). The position of the water table, temperature, and litter quality are important drivers of decomposition (Moore et al. 2007; Turetsky et al. 2008). During early successional stages or during wetter periods of the year, the water table is high and decomposition proceeds slowly due to anaerobic conditions. The water progressively becomes more reducing, and a series of biogeochemical reactions change the chemistry of the peat and the porewater and produces gases, i.e., nitrous oxide [N_2O] from nitrate reduction (denitrification), hydrogen sulphide [H_2S] from sulphate reduction, and methane [CH_4] from carbonate reduction and acetate splitting. During later successional stages and drier periods of

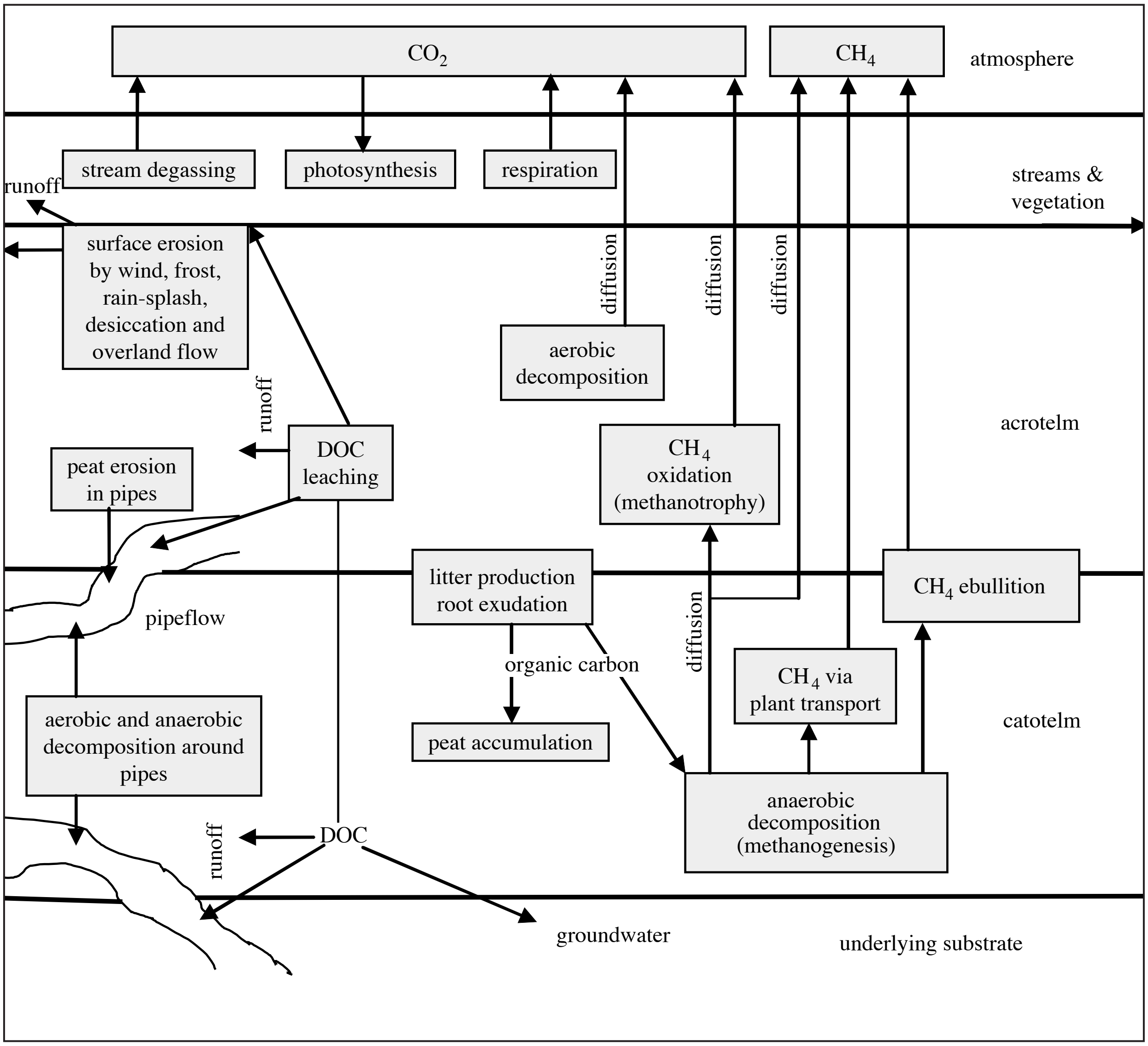

Figure 115. Components of the peatland carbon cycle.
Source: Holden (2005). Reprinted from Philosophical Transactions of the Royal Society A, Volume 363, J. Holden, Peatland hydrology and carbon release: why small-scale process matters, pp 2891-2913, Copyright 2005, with permission from the Royal Society.

the year, the water table drops and decomposition generally quickens with a return to aerobic conditions. Litter quality also affects decomposition, with vascular plant litter decomposing faster than non-vascular moss (Moore et al. 2007), although rates differ amongst different species of moss (e.g., *Sphagnum*; Turetsky et al. 2008).

2.4.4.1.3 **Dissolved organic carbon export**

As a result of its large store of organic matter and a porous structure, peat in the Hudson Plains Ecozone produces porewater that contains high C concentrations (Reeve et al. 1996). When the water table is high, further increases will raise this porewater to the surface, causing saturation excess and result in overland flows. Loss of C through export is a large, although often ignored, component of peatland C budgets. For example, losses of 13 g C/m^2/yr, primarily from dissolved organic carbon (DOC), accounting for 31 to 37% of net C uptake, were observed from a boreal fen in Sweden (Nilsson et al. 2008). In the Hudson Plains Ecozone, as well as the western boreal forest, DOC export is expected to be slightly lower, ranging from 3.6 to 11.4 g C/m^2/yr (Fraser et al. 2001), but it is still an important loss of ecosystem C. This dissolved form of C is an important part of the C budget (Roulet et al. 2007), and it also regulates stream acidity and the transport of other nutrients (e.g., nitrogen [N]) and metals (e.g., mercury) that affect the productivity of downstream communities (Kalbitz et al. 2000).

2.4.4.1.4 **Carbon accumulation**

Northern peatlands, with low production rates and even lower decomposition rates, have accumulated peat over thousands of years. Radiocarbon (^{14}C) dating of basal peat indicates that peatland development was initiated up to 6,000 years ago in the interior of the Hudson Plains Ecozone, with progressively younger peatlands created by isostatic rebound towards the coast (Klinger and Short 1996; Glaser et al. 2004). Estimates of apparent long-term C accumulation rates (LORCA) in the boreal wetland region range from 18.9 to 23 g C/m^2/yr (Zoltai and Johnson 1985; Gorham 1991). However, LORCA varied from 11 to 59 g C/m^2/yr during peatland development across 10 peatlands in the southern region of the Hudson Plains Ecozone (Garneau et al. 2007). These differences were predominantly the result of wetness, with C accumulation rates highest during periods of highest moisture (Garneau et al. 2007). In addition, surface accumulation rates are higher and ranged from 40 and 360 g C/m^2/yr along a high boreal to subarctic gradient in northern Manitoba (Bubier et al. 1999; Trumbore et al. 1999; Camill et al. 2001), with variability due to differences in productivity and environmental conditions. For the Hudson Plains Ecozone and especially the northern part, long-term C accumulation rates are likely on the low end of the range, because of permafrost occurrence.

2.4.4.1.5 **Net ecosystem & greenhouse gas exchange**

The stability of peatlands as atmospheric C sinks depends on the surface hydrology and the plant community composition and its phenology (Johannson et al. 2006; Lund et al. 2010). The timing of spring thaw and the growing season length influence the annual CO_2 sink strength of peatlands (Griffis et al. 2000; Aurela et al. 2004; Lund et al. 2010). In the mixedwood boreal region of Alberta, Glenn et al. (2006) found that a nutrient-poor fen with shrubs that were active earlier and later in the growing season was a stronger CO_2 sink (-90 ± 7 g C/m^2/season) than a nutrient-rich fen in which the grasses grew during a narrower period (-31 ± 18 g C/m^2/season). Early in the growing season, peatlands are net sinks for CO_2, but they can become a source later in the season, particularly in dry years (Moore et al. 2002). Although the average summer time net ecosystem exchange of CO_2 (NEE) is highly variable among northern peatlands, a literature synthesis indicated that all seven northern peatland sites where annual NEE measurements have been made were net CO_2 sinks (Lund et al. 2010).

Climate differences among years can also affect the net C exchange of a peatland. In warm and dry growing seasons, the strength of a peatland as a C sink is reduced, which in some cases could result in net CO_2 losses. Over a 6 year period, a fen in the Hudson Plains Ecozone near Churchill, Manitoba, ranged from a sink of -299 g C/m^2/season to a source of 76 g C/m^2/season (Griffis et al. 2000). Similarly, a fen slightly south in the adjacent Boreal Shield Ecozone near Thompson, Manitoba, switched from a sink (-120 g C/m^2/season) one year to a source the following year (+40 g C/m^2/season) (Joiner et al. 1999). Given that these estimates did not include the low rates of decomposition that continue between growing seasons (i.e., during winter), these peatlands were likely weaker sinks or larger sources of CO_2 to the atmosphere (Griffis et al. 2000).

Despite being CO_2 sinks during the summer months, wetlands can show positive radiative forcing due to emissions of N_2O and CH_4, which have 298 and 25 times the global warming potential of CO_2, respectively, over a 100 year timeframe (IPCC 2007). Nitrous oxide is a potent greenhouse gas (GHG) created through denitrification under mildly reducing conditions. The importance of N_2O produced from northern peatlands is thought to be low where N is limited. Low concentrations of nitrate will limit denitrification and N_2O production during much of the growing season. However, *where* (e.g., peatland perimeter) or *when* (e.g., early spring and late autumn) more nitrate occurs, rates of N_2O production could be higher (Wray and Bayley 2007).

Peatlands can be an important source of CH_4. Methane is generated from anaerobic respiration through two pathways. Acetate dissimilation (acetate pathway) dominates methane production in the upper layers of the peat, whereas carbonate reduction (hydrogen pathway) dominates production in deeper peat (Hornibrook et al. 1997). Gas diffuses to the surface either through the peat or through plant stems (Le Mer and Roger 2001). More rapid expulsion of CH_4 occurs through ebullition, when gas that is retained due to denser overburden (e.g., ice) is released, as a result of continued increases in internal pressure, decreases in atmospheric pressure, or removal of the overburden (Baird et al. 2004; Tokida et al. 2007). Methane fluxes from northern boreal peatlands in Scandinavia ranged from 8 to 13 g C/m^2/yr (Nilsson et al. 2008), but work in Hudson Plains Ecozone as part of the Northern Wetlands Study showed that this region has much lower rates of CH_4 emission (Roulet et al. 1994). On an area-weighted basis, 1.31 and 2.79 g C/m^2/yr from CH_4 were emitted from southern and northern portions of the Hudson Plains Ecozone, respectively. Low rates of CH_4 were attributed to pH and substrate quality limiting the fermentation step in CH_4 production (Valentine et al. 1994).

In addition to being emitters, peatlands can also be CH_4 consumers. During dry conditions, 60 to 90% can be reoxidized to CO_2 in the overlaying aerobic zones (Le Mer and Roger 2001). During these dry conditions, the microbes in the aerobic layer of the peat could actively draw CH_4 from the atmosphere (Le Mer and Roger 2001).

2.4.4.1.6 **Ponds**

Ponds can occupy as much as 40% or more of the landscape of the Hudson Plains Ecozone in some areas (Bello and Smith 1990). Formed from peat collapsing when permafrost thaws or in topographic depressions, the ponds accumulate sediment from internal processes, as well as receiving dissolved and particulate matter leached from neighbouring peatlands. Ponds contribute to the landscape C budget by accumulating C in sediments. They also have a role in gas exchange, but the amount of gases they produce and consume varies (Martini 2006). Hamilton et al. (1994) found that ponds at three sites in the Hudson Plains Ecozone were constant net sources of CH_4 (110-180 mg CH_4/m^2/day) and CO_2 (3,700-11,000 mg CO_2/m^2/day), and that these fluxes were much greater than surrounding vegetated surfaces. Carbon dioxide exchange in the ecozone's ponds has also been linked to sediment type, with organic sediments thought to be a CO_2 sink and mineral sediments a source (Macrae et al. 2004). Carbon dioxide and CH_4 exchange is also influenced by pond size (Moore et al. 1994; McEnroe et al. 2009), with shallower ponds having larger CO_2 (0.35 ± 0.47 g C/m^2/day) and CH_4 (0.31 ± 0.69 g C/m^2/day) fluxes (McEnroe et al. 2009).

2.4.4.2 **Trends & human influences**

2.4.4.2.1 **Current trends: landscape scale**

The heterogeneity in surface hydrology that affects C dynamics occurs at resolutions on the order of metres (Figure 116). Scaling this variability to resolutions acceptable for regional-level models is a major challenge (Sitch et al. 2007; Baird et al. 2009). Understanding the aggregated effect of C cycling from peatlands and ponds requires knowledge of their extent and how that changes through time.

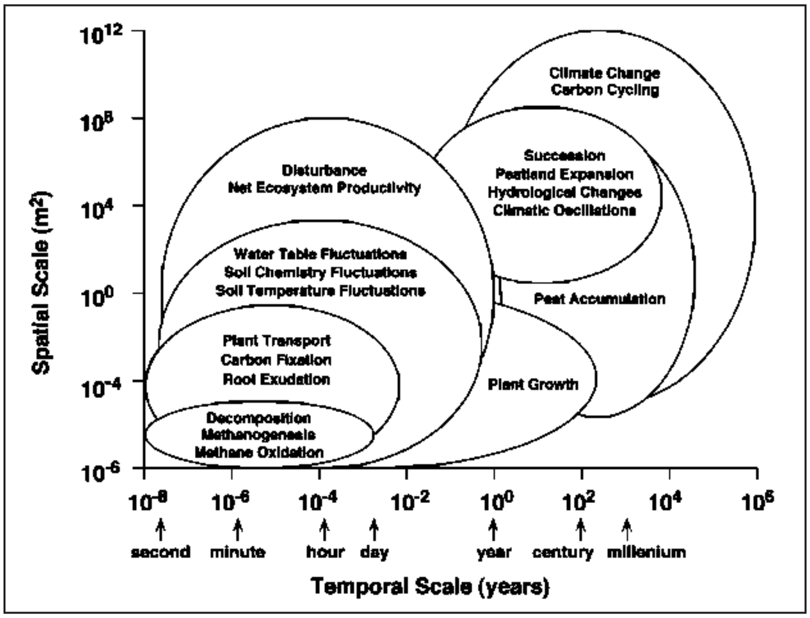

Figure 116. The temporal and spatial scales of carbon cycling processes in peatlands of the Hudson Plains Ecozone.
Source: Klinger et al. (1994). Reprinted from Journal of Geophysical Research, Volume 99, Issue D1, L.F. Klinger, P.R. Zimmerman, J.P. Greenberg, L.E. Heidt and A.B. Guenther, Carbon trace gas fluxes along a successional gradient in the Hudson Bay lowland, pp 1469-1494, Copyright 1994, with permission from American Geophysical Union.

Satellite remote sensing suggests that for the region during 1985 to 2006, aboveground NPP remained relatively stable, although the Normalized-Difference Vegetation Index (NDVI, a measure of primary productivity based on remote sensing) significantly increased over 4.9% of the land surface and decreased over 0.1% of the land surface of the ecozone (Figure 117; Pouliot et al. 2009; Ahern et al. in press). However, satellite observations cannot provide information on belowground processes affecting soil C and N dynamics (Sitch et al. 2007). Scaling production and decomposition processes over the large expanse of Hudson Plains Ecozone will require linkages between field studies that examine key biogeochemical processes controlling C cycling and process-based spatial models (Baird et al. 2009). Successful model implementation will require an extensive spatial database of climate, hydrology (e.g., soil moisture and areas of inundation), temperature, and productivity that will need to be obtained through a network of monitoring sites and/or remote sensing. Satellite remote sensing is becoming an important tool in regional mapping of biophysical variables needed for peatland research, providing information at spatial and temporal scales that are generally inaccessible or impractical for field observations (Sitch et al. 2007). Depending on the energy reflected and the type of sensor, information can be collected to: 1) classify vegetation and its greenness and phenology; 2) detect saturated and inundated water; and 3) measure permafrost and other environmental parameters (e.g., surface temperature, snow cover, topography) (Pietroniro and Leconte 2005).

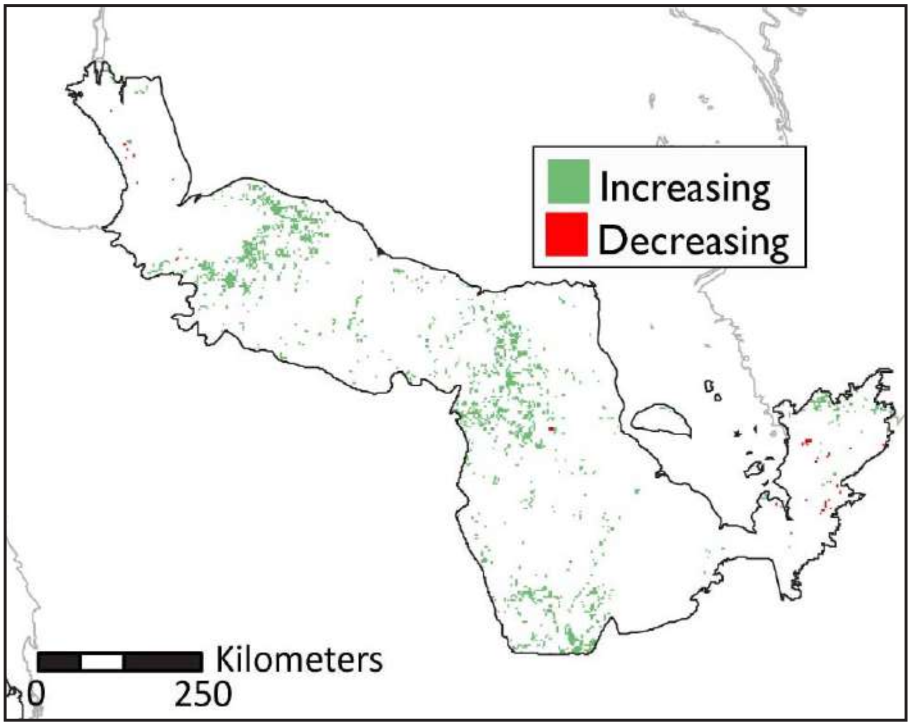

Figure 117. Trends in normalized-difference vegetation index (NDVI) between 1985 and 2006 in the Hudson Plains Ecozone (ecozone$^+$ boundaries). Areas coloured green increased significantly ($p < 0.05$) and areas coloured red decreased significantly.
Source: Data for ecozone provided by authors of Ahern et al. (in press).

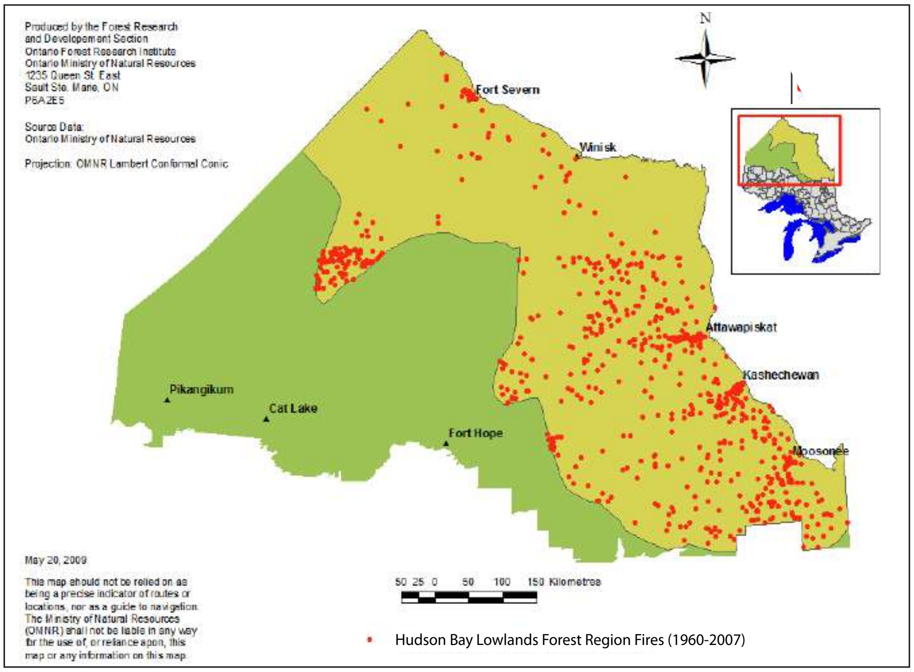

Figure 118. Fire occurrence in the general vicinity of the Hudson Plains Ecozone in Ontario, 1960-2007 (the shading does not represent ecozone boundaries).
Source: Data are from the Ontario Ministry of Natural Resources' fire database.

Fire

Wildfire occurs less frequently in peatland areas (return period of 80 to 1,100 years in temperate and boreal peatlands) than in upland areas (return period of 50 to 500 years in boreal forest) (per review by Flannigan et al. 2008). More fires occur in the southern part of the Hudson Plains Ecozone in Ontario (Figure 118), but further statistics are required before confident inferences about spatial and temporal patterns can be made (Section 2.4.2.2, *Fire*). Although fires are less frequent in peatlands, when they do burn, large amounts of organic matter are consumed. In the western continental peatlands, fires are responsible for emitting 22% of the annual fire-emitted C in Canada (Turetsky et al. 2004), with an average of 3.2 kg C/m^2 lost during an individual peatland fire (Turetsky and Wieder 2001). In the wetter Hudson Plains Ecozone, the impact of fire, while important, will likely result in less carbon loss.

2.4.4.2.2 **Human influences & future trends**

Climate change scenarios developed using various global circulation models project warmer temperatures, higher precipitation, but overall drier conditions in the Hudson Bay region (Gagnon and Gough 2005; see also Section 2.1, *Abiotic Drivers*). Peatland sensitivity models predict that the greatest effect of climate change will occur in the subarctic region, as a result of degradation of frozen peatlands (Tarnocai 2006; or see Figure 62 in Section 2.2.2.4.1, *Wetlands (Freshwater)*). Warming will occur following increases in air temperature, as well as changes in the timing and depth of snow cover (Osterkamp and Romanovsky 1999) and retreat of sea ice (Gagnon and Gough 2005). The effects of warming are complex, and understanding associated changes in hydrology is pivotal to a complete understanding of peatland C dynamics (Johansson et al. 2006) (Figure 119).

In the southern part of the Hudson Plains Ecozone, where permafrost is limited, warming will lower the water table and increase the depth of the layer active in aerobic decomposition (Gorham 1991). The response to warmer and drier conditions is complex. Higher sensitivities of soil respiration than NPP to changes in temperature (see Davidson and Janssens 2006) will initially increase C mineralization rates. However, increased N mineralization could stimulate NPP (Camill et al. 2001), as will CO_2 enrichment (Körner 2000). Shifts from herbaceous vegetation to shrubs and eventually to forest will also affect sequestration potential (Camill et al. 2001). Under a 2 x CO_2 scenario, northern fens are predicted to become a stronger CO_2 sink, while bogs could become a net source of atmospheric CO_2 (Waddington et al. 1998). The drier, southern peatlands will also be more vulnerable to burning. A higher frequency of large fires and burning during the period of maximum water table drawdown (Kasischke and Turetskey 2006) would produce large amounts of CO_2 and exacerbate atmospheric C emissions (Flannigan et al. 2008).

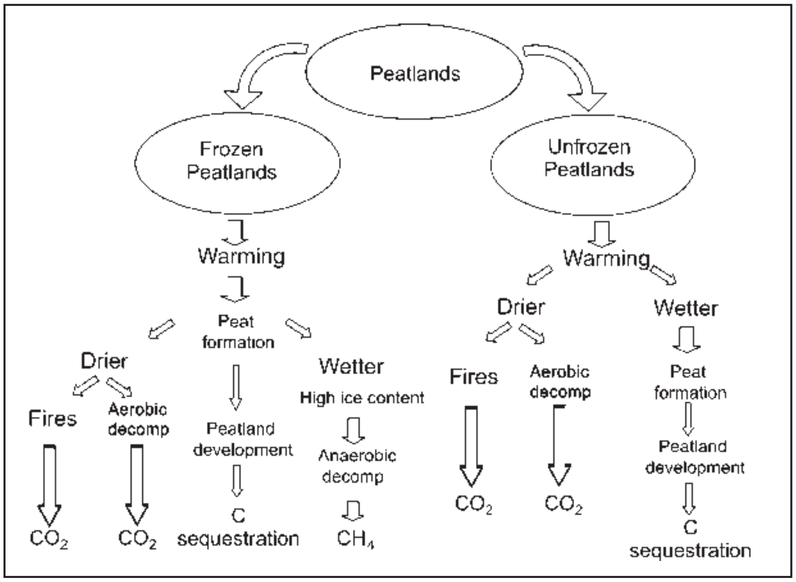

Figure 119. The effects of climate warming on carbon cycling processes and carbon fate for frozen (left) and unfrozen (right) peatlands.
Source: Tarnocai (2006). Reprinted from Global and Planetary Change, Volume 53, C. Tarnocai, The effect of climate change on carbon in Canadian peatlands, pp 222-232, Copyright 2006, with permission from Elsevier.

In the northern parts of the Hudson Plains Ecozone, where permafrost is more extensive, thawing will collapse the peat, raise the water table, and form ponds (Gorham 1991). Rates of permafrost thaw have accelerated, since the mid-20th century (e.g., Payette et al. 2004; Camill 2005), and modelling work in the Hudson Bay region projects the loss of over 50% of the continuous permafrost (and complete loss of permafrost that is currently discontinuous and in isolated patches) and a virtual elimination of a climate that supports permafrost there by 2100 (Gough and Leung 2002; Gagnon and Gough 2005). In these areas, CO_2 emissions will decrease, but CH_4 emissions, a stronger GHG, will increase, further increasing global warming potential (Turetsky et al. 2002). Similarly, if sea levels rise and coastal areas flood (Tarnocai and Stolbovoy 2006)[69], low-lying peatlands in these regions will become larger CH_4 sources. In contrast, peatlands in the continuous permafrost region could warm sufficiently to become larger C sinks, due to productivity increases (Tarnocai 2006). Understanding these processes and how they affect GHG production and fluxes is critical to predict current and future GHG sources and sinks for the region. The lack of long-term data in the Hudson Plains Ecozone limits opportunities for inferences about climate change impacts on C sequestration.

In addition to climate change, energy and resource developments are expected to expand within the region (see Section 2.2.1.2.1, *Changes in Land Cover*). Peat and resource mining and hydroelectric developments that disturb the surface of the peat or alter the hydrology can also affect the fate of peatland C. Activities that cause flooding (peat mining, surface mineral extraction, areas upstream of hydroelectric dams) decrease CO_2 emissions but will increase CH_4, while activities that lower the water table (e.g., areas downstream from hydroelectric dams) will initially stimulate decomposition processes and increase CO_2 emissions but in the long term could create net C sinks (Roulet 2000).

In conclusion, the peatland and pond complex within the Hudson Plains Ecozone is a dynamic ecosystem that has accumulated C over thousands of years. As climate conditions in the region change, the area's potential to continue accumulating C is uncertain. Loss of this important store of C could significantly affect the earth's climate system (Moore 2002; Chapman et al. 2003). Much more research is required to determine what controls the fate of the C in this spatially heterogeneous and temporally dynamic ecosystem to better predict how it will respond to changing conditions. Future research in the Hudson Plains Ecozone needs to be established using an experimental design that lends itself to scaling up using remote sensing. The data could then be used in a modelling framework to predict interactions of climate change and management activities on Hudson Plains Ecozone C dynamics and their effects on future global climate change.

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[69] Sea level rise is less of a concern for this ecozone than for some other coastal areas, because it may not overtake the ecozone's especially high rate of isostatic rebound (see Section 2.4.1, *Coastal Building Processes*, and references therein). However, longer ice-free seasons in Hudson and James bays will likely render the coast susceptible to increased storm surges (inundation events).

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2.4.5 **Nutrient cycling**

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Examples of efficient nutrient cycling scenarios are not common in the Hudson Plains Ecozone. Environmental factors such as isostatic rebound, poor drainage, permafrost, and a cold climate impede the classic nutrient cycling model, which requires the reuse of nutrients (particularly nitrogen and phosphorous) first harnessed through plant growth and then later released through mineralization. Conditions in fact reduce or exclude the oxygenation of soil profiles and impede mineralization to the extent that most of the nutrients used in ecosystems are not subsequently released again in the short term, but rather they remain in plant tissue build-up as fens and bogs develop. So dramatic is this build-up that at least 75% of the region is covered by soils from the Organic or Cryosolic (Organic Cyrosol great-group) orders (Scott 1995; Tarnocai 2000), making this ecozone one of the world's largest peatlands and Canada's largest peatland complex (Tarnocai and Stolbovoy 2006). Only to the south and southwest of James Bay, within the James Bay Lowland Ecoregion, are summer temperatures less severe, and conditions permit modest mineralization.

Exceptions are coastal salt marshes, as well as river estuaries and sheltered shorelines, where soils are so young that they have not yet had the opportunity to develop from mineral regolith (Regosols) – to become Static or Turbic Cyosols capped with sufficient surface organic residues to develop peaty phases or even eventually true Organic soils. The salt marshes that occur along the southern shores of Hudson Bay are limited in plant productivity first by available nitrogen and then, if nitrogen availability increases, by phosphorus (Cargill and Jefferies 1984a; Ngai and Jefferies 2004). In addition, available soil nitrogen and phosphorus amounts that are mineralized remain low in the soil, as the plant species inhabiting these sites sequester what becomes available. While coastal salt marshes are, therefore, low in available forms of nitrogen and phosphorus, the more palatable and nutritious grasses and sedges, such as *Puccinellia phryganodes* and *Carex subspathacea*, are the beneficiaries, as are the migratory waterfowl that forage on them (Cargill and Jefferies 1984a; Scott 1995).

Nutrient deposition is significant over most of the ecozone (e.g., see Galloway et al. 2004 for total inorganic nitrogen). As well, the ecozone's often carbonate-rich Paleozoic bedrock can, on weathering, sometimes release ample supplies of calcium and magnesium. Most of this soil-nutrient influencing weathering is, however, limited to situations where land has only recently emerged from the ocean due to isostatic rebound or where coarse-textured carbonate raised beach ridges have promoted xeric and, therefore, slow peat-growing environments.

With distance from the ocean, land has been above sea level for longer, will usually be at higher elevation, and therefore had time to undergo pedogenesis and profile development. This means that the younger, coastal soil profiles are generally mineral, will contain many bases such that the pH is slightly acidic to alkaline (minerotrophic), and, as organic material accumulates on lowland and level terrain, base-rich fens result. Although fens appear more nutrient rich, because bases such as calcium and magnesium raise the pH, mineralization rates for nitrogen-bearing residues remain slowed by cool soil temperatures. Still, fens support more nutritious sedges and lichens that provide suitable browse for migratory birds and ungulates.

With time paludification takes over, a process that sees fen peaty layers thicken, base supply within the upper soil profile reduced, soils become more acidic (ombrotrophic), mineralization rates sometimes reduced, organic layers thicken, and ultimately bogs with their different plant ecosystem covers develop. Bogs become almost totally dependent on nutrient inputs from precipitation, with only some internal nutrient cycling. In addition, bogs are poor environments for producing inputs via nitrogen fixation (Bridgham et al. 1998).

2.4.5.1 **Trends & human influences**

Atmospheric deposition and surface and subsurface runoff can be important sources of anthropogenic nitrogen and phosphorous inputs in Canada's boreal region (e.g., Pelster et al. 2008). Broad trends in long-distance transported sources are apparent. Atmospheric nitrogen transport increased globally between 1860 and the early 1990s, resulting in at least some increase in total inorganic nitrogen deposition rates in the Hudson Plains Ecozone (see Figure 2 in Galloway et al. 2004). Likewise, nitrogen transport through inland waters to the coast and ocean has increased somewhat, since pre-industrial times, although in the Hudson Plains

Ecozone most of this accelerated anthropogenic input appears to be in the southern portion of the ecozone around James Bay (see Appendix Figure A.9 in MEA 2005), and it likely originates outside the ecozone. Experiments within the ecozone using fertilizer in a graminoid-dominated intertidal salt marsh at La Pérouse Bay showed strong early growing season responses to both nitrogen and nitrogen+phosphorous treatments (Hargreaves et al. 2009) and, as such, they lend support to the possible consequences of long-distance atmospheric nutrient deposition.

Changes in nutrient inputs and cycling that are associated with resource extraction or land use activities directly in the ecozone are limited at present. Agriculture and associated fertilizer and livestock waste inputs are effectively non-existent (McConkey et al. in press), as are forest harvesting, peat harvesting, and biomass burning activities. On the other hand, hydroelectric and mining developments do exist in areas (although limited at present), and these developments have been associated with changes in nutrient availability. For example, nutrients such as phosphorus increased following impoundment of the Opinaca reservoir (part of the La Grande hydroelectric complex) in Québec, as the large area of flooded plants decomposed (Hayeur 2001).

The best documented human-facilitated change or trend in nutrient cycling in this ecozone is in coastal salt marshes affected by the foraging activities (grazing and grubbing) of staging and breeding migratory lesser snow geese (*Chen caerulescens caerulescens*). As discussed previously (see sections 2.2.2.1, *Coastal* and 2.3.3.3.2, *Waterfowl*), the Mid-Continent population of lesser snow goose increased greatly over the past four decades, largely due to an increase in the availability of high quality agricultural food (nitrogen-enriched pastures and crops) outside of the ecozone on their wintering grounds and along migration routes. During their seasonal period in the Hudson Plains Ecozone, geese from this population enhance the net above-ground primary productivity of coastal salt marshes at low to moderate grazing intensities (Cargill and Jefferies 1984b; Hik and Jefferies 1990; Hik et al. 1991). Increases in productivity are observed, because goose faeces act as a fertilizer (faeces contain high levels of soluble nitrogen compounds), increasing nitrogen availability at a time when plant nitrogen demand is greatest and prolonging the period of active growth. However, productivity has been lost at high goose foraging intensities (such as with intense grubbing in spring), owing to vegetation loss and the subsequent creation of barren patches, as hypersaline conditions develop (Srivastava and Jefferies 1995, 1996; McLaren and Jefferies 2004). In this latter case (i.e., formation of an alternative stable state of exposed saline sediment that supports little or no vegetation), the input of nitrogen from cyanobacterial fixation has become very limited (Walker et al. 2003), and any supplemental nitrogen in the form of goose faeces is blown away by wind, further decreasing available nitrogen in this already nitrogen-poor environment and limiting the capacity of coastal salt marshes to recover (McLaren and Jefferies 2004; Ngai and Jefferies 2004). A fully parameterized model was developed to simulate these altered nitrogen flows within the ecozone's coastal salt marshes and allow modeling of year-to-year trends for periods up to 50 years (Walker et al. 2003). Like field data suggest, catastrophic herbivory-linked disturbance leads to the loss of soil nitrogen and other plant nutrients (Buckeridge and Jefferies 2007), so recovery of degraded salt marshes could take decades (see Jefferies et al. 2003).

In terms of future projections, resource development activities, particularly in hydroelectric and mining sectors, are expected to increase in and around the Hudson Plains Ecozone in the near future, as emphasized throughout this report (Manitoba Geological Survey 2003; OPA

2007; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates Ltd 2010; Hydro-Québec 2010; Manitoba Hydro 2010; MDDEP 2010; Micon International 2010). Such large-scale resource developments might affect nutrient cycling in areas of this ecozone, as by altering hydrology (e.g., Pelster et al. 2008). In addition, it is anticipated that climate warming will force changes on nutrient cycles within this ecozone, as permafrost thaws and ecosystems adapt to greater warmth. The changes within slowly decomposing wetland soils are unlikely to limit ecosystem productivity as conditions warm (Turner et al. 2004). Permafrost degradation caused by warming will also affect infiltration and run-off. However, little is known about such complex responses and nonlinear impacts on nutrient cycles caused by hydro-climatic forcing (Manzoni et al. 2004) throughout this ecozone.

In closing, it is noteworthy that any future changes in nutrient cycling in the Hudson Plains Ecozone can be expected to contribute to changes in nutrient levels and productivity in the coastal marine environment, via river flow and overland runoff. For discussion of status and trends in the marine environment, see the Arctic Marine Ecozones technical report of the ESTR (Niemi et al. 2010).

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2.4.6 Hydrological processes

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Hydrological processes are a particularly important driver of the ecological communities in the Hudson Plains Ecozone, given the poor drainage arising from its low gradient, low permeability marine sediments, and permafrost, which have resulted in the development of a peatland-dominated landscape. Hydrological processes also exert an important influence on ecosystem services, such as climate regulation (peatland carbon storage) (Roulet et al. 1992), water quality (filtration) (Reeve et al. 1996), and disturbance moderation (flood abatement) (e.g., Schindler and Lee 2010). The seasonal drainage of water from uplands to lowlands, and its subsequent storage and slow flow through the ecozone's extensive wetlands, helps maintain evapotranspiration and stream flows in summer (Boudreau and Rouse 1995). Thus, the hydrological connections and processes in this ecozone also affect the freshwater, sediment, and nutrient discharges of rivers into Hudson and James bays, with associated influences on the ecology there (e.g., Hydro-Québec 2004; Déry et al. 2005). This section focuses on status and trends in terrestrial (predominantly peatland) hydrology of the Hudson Plains Ecozone; trends in river hydrology in the ecozone were discussed in Section 2.2.2.4.2, *Rivers/Streams & Lakes*.

While there has been considerable research on the hydrology of isolated peatland systems, the state of knowledge on the hydrology of peatland complexes is confounded by the wide array of forms and functions and their interaction with each other. Riley (in press) developed a classification system that reflected this complexity, because the peatland forms listed in the Canadian Wetland Classification System (NWWG 1997) are difficult to unambiguously identify (Di Febo in press). This difficulty occurs, in part because of the groundwater flow patterns that develop and the biogeochemical cycling this drives, and the ecological feedback (Siegel 1983; Reeve et al. 1996). Generally, water moves from raised bogs that often develop on the interfluves between incised channels to fen water tracks that are the conveyance network for water flow (Glaser et al. 2004). Water draining from fen water tracks is the primary water source for first order streams, and it has a predictable pattern of low-flow runoff, which is strongly correlated to drainage area (Richardson et al. submitted) except at higher flows, when connectivity switches from peatland morphology-dominated flow paths to one that is more influenced by larger-scale fluvial morphological feedbacks. Observed runoff patterns are more consistent with the variable source-area concept than the fill-and-spill generated flow regimes (Richardson et al. submitted). Nevertheless, the coupling of water flows within and between peatland types, and how this affects ecological processes, remains poorly understood.

2.4.6.1 Relationship between hydrology & predominant ecological communities

The Hudson Plains Ecozone rests on Silurian sedimentary bedrock overlain by blue silty clay deposited during the post-glacial Tyrell Sea episode (Lee 1960; McDonald 1989). Isostatic rebound of between 0.90 m/century at Churchill and 1.1 m/century at Peawanuck (Sella et al. 2007) (earlier estimates were 0.7-1.25 m/century; Webber et al. 1970) results in ~1 km extension of the coastline every century (see also Section 2.4.1, *Coastal Building Processes*). There, reworked sand and silt result in the development of beach ridges that eventually become isolated from the shoreline, as the coast aggrades (Martini 1981). Beach ridges are integral to the formation of wetlands as water becomes impounded behind them (Price and Woo 1988a), sometimes exacerbated by beaver activity (Woo and Waddington 1990), and marshes develop. These marshes are saline in character near the coast on account of fossil salt diffusing from the sediments (Price and Woo 1988c).

In the coastal zone, topographic expression resulting from beach ridges drives recharge/discharge processes in the near-surface sediments causing depressed salinity in raised beach sediments and elevated salinity in marshes that develop in the inter-ridge depressions (Price and Woo 1988b). This results in a distinct pattern of vegetation ranging from salt-intolerant sedge and woody plant communities on ridges and salt-tolerant species in depressions and along the coastal margin (Price et al. 1988).

Distance inland is a proxy for time (as a consequence of isostatic rebound), and thus the salinity effect diminishes about 6-14 km inland following centuries of flushing (Price et al. 1987). At these more inland locations salt-marsh species give way to alder and willow (on ridges) and tamarack fens (Price et al. 1988), where peat development obscures the topographic expression resulting from beach ridges. Tear-shaped black spruce islands develop in the eastern larch (tamarack) and sedge-dominated fens (Klinger and Short 1996), and they eventually expand and coalesce into forested bogs and ultimately into open *Sphagnum*-dominated bogs interspersed by fen water-track or more extensive horizontal fen (Glaser et al. 2004). Glaser et al. (2004) estimate that in the Albany River region, which is characteristic of most of the area, over 90% of the landscape is peatland, with about 65% fen and 35% bog.

The predominant ecological communities of the region are thus intricately tied to the poor drainage and wetland-dominated landscape. Interruption of wetland hydrological processes by human-induced disturbances will, therefore, impact the regional biodiversity. The sensitivity to hydrological change of the massive amount of carbon stored in the ecozone's peatlands is also of particular interest (see later).

2.4.6.2 **Trends & human influences**

The hydrological cycle is expected to be greatly altered by the future climate conditions forecast for this ecozone, which may result in permafrost thaw (Gough and Leung 2002; Gagnon and Gough 2005a), a shorter freshwater ice season, and overall drier conditions (Roulet et al. 1992) (Section 2.1, *Abiotic Drivers*). Carbon cycling is also likely to be strongly affected, given the role of permafrost in maintaining the cool and wet conditions (restricted drainage) that limit oxidation and stabilize carbon stores (Tarnocai 2006; see also Section 2.4.4, *Carbon Cycling*). Moreover, if large areas of dry peatlands are exposed to fire, greatly exacerbated atmospheric carbon emissions could result (e.g., Weber and Stocks 1998; Flannigan et al. 2009; see also Section 2.4.2.2, *Fire*).

The role of permafrost in the hydrological cycle and its potential loss due to climate warming are not well understood. Permafrost is sporadic in parts of the ecozone, for example occurring beneath palsas, which upon collapse become very wet (Tarnocai 1972). In the most northerly parts of the ecozone, where permafrost is (currently) continuous, the ecohydrological processes are adapted to it; its degradation will likely vary between bogs and fens and the ensuing changes to flow patterns are unknown.

Roulet et al. (1992) suggest a water table decline of ~14-22 cm would occur in peatlands in response to a 2 x CO_2 climate warming scenario, causing a shift in carbon exchange and vegetation succession (Hilbert et al. 2000). Seasonal evapotranspiration from bog peatlands could thus be affected (Lafleur et al. 2005). In a patterned peatland with an artificially lowered water table, Whittington and Price (2006) found the surface in *Sphagnum* lawns and pools subsided at a similar pace as the water table, but the consolidation altered the hydraulic properties, resulting in a shift in carbon exchange (greater respiration) and vegetation community structure (towards vascular plants) (Strack et al. 2006). Ridges were less susceptible to subsidence and experienced relatively more water table lowering, which favours colonization by more woody species (Minkkinen et al. 1999). Currently, however, there is no strong evidence of climate change affecting peatlands of the Hudson Plains Ecozone.

Large-scale impacts on hydrology are also caused by hydroelectric diversion. This is documented, for example, in the southeast portion of the Hudson Plains Ecozone in the Eastmain River system of the La Grande hydroelectric-complex project. In addition to strong changes in river flows (Section 2.2.2.4.2, *Rivers/Streams & Lakes*), the impacts on the hydrology are profound. This project flooded several hundreds of square kilometers of land (creating the Opinaca reservoir), when ~90% of the flows from the Eastmain and Opinaca rivers were diverted north (Hayeur 2001; Hydro-Québec 2004; Therrien et al. 2004; see also Section 2.2.2.4.2, *Rivers/Streams & Lakes*). The flooding promoted vegetation changes and led to the accelerated conversion of mercury to methylmercury and its subsequent bioaccumulation in fish within the reservoir, with fish mercury levels gradually declining again over time (Therrien and Schetagne 2008). As well, reservoir operations changed the natural flood cycle of the area (Therrien et al. 2004), and the river diversions led to some desiccation of wetlands and associated changes in vegetation downstream, along reduced flow river segments (Hayeur 2001). Diversion in 2009 of 72% of the mean annual flow of the Rupert River north to the La Grande Complex is further changing wetland hydrology in the Québec portion of the ecozone (Hydro-Québec 2004, 2010)[70]. Methane flux from these reservoirs is initially high, but it decays substantially over 10 years (Tremblay et al. 2010), probably because methane becomes oxidized in the water column before release (Duchemin et al. 1995).

More localized hydrological impacts are caused by mining activity in the ecozone. For example, dewatering of the Victor diamond mine near Attawapiskat, Ontario (constructed beginning in 2006, opened in 2008; DeBeers Canada 2005, 2008), is expected to increase annual recharge from +10 mm/yr at a radial distance of 15 km from the pit to +250 mm/yr at a distance 1-2 km from the mine (AMEC 2008). The affected area is thus predicted to be ~500 km^2, although the impact at the extremity of the drawdown zone may be negligible. The presence of bioherms (ancient coral reefs that are connected to the depressurized limestone aquifer and protrude near to or above the surface) form enhanced drainage nodes, which cause the peat within ~50 m of the bioherms to drain (Whittington and Price in press). Closer to the mine the increased recharge is likely to sufficiently alter the water balance over the life of the mine (~10-15 years) to initiate successional changes in vegetation, as described above. Over the medium-term (decades) the ecological and hydrological changes may persist, but over the long-term the system is likely to revert to its original form.

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[70] Lateral flow from tributaries increases the flow of the Rupert River at its mouth to ~48%.

The remoteness of the Hudson Plain Ecozone and its comparatively low level of known mineral resources have historically reduced the pressure of development on its biodiversity. However, development pressure is increasing, particularly in mining and hydroelectric sectors (e.g., OPA 2007; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates 2010; Hydro-Québec 2010; Manitoba Hydro 2010; Micon International 2010). While global effects (i.e., climate change) will impact the whole region, comparable peatland-dominated landscapes already exist in considerably drier regions (e.g., Lake Agassiz Peatlands) (Siegel 1983). Thus, climate change is unlikely to pose a severe threat to the ecozone, at least on the basis of hydrological change. However, hydrological change is likely to lead to important impacts on both wetland communities and carbon cycling (as above), although peatlands (especially fens) seem to have some resilience associated with their ability to subside as water is lost, thus maintaining wetter conditions than they otherwise would (Lafleur and Roulet 1992; Whittington and Price 2006).

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2.4.7 **Pollination**

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Aside from a rudimentary species inventory (e.g., see Section 2.3.3.6, *Invertebrates*), a paucity of baseline data exist concerning the status of pollinators and pollination in the Hudson Plains Ecozone. Likewise, no monitoring data exist with which to report on the population trends of pollinators.

The insect pollinator spectrum in subarctic and arctic environments is considerably different than that in more southern and temperate environments (Danks 1981). While flies (Diptera, especially Muscidae, Empididae, Syrphidae, and Calliphoridae) dominate in terms of numbers, bumblebees (Hymenoptera, especially *Bombus* spp.) are the most efficient pollinators (Kevan 1972; Danks 1981). Moths and butterflies (Lepidoptera) and beetles (Coleoptera) become less important (Kevan 1972; Danks 1981). Other insects contributing to pollination in subarctic and arctic environments include springtails (Collembola) and mosquitoes (Diptera: Culicidae) (Kevan 1972; Danks 1981).

The importance of insects to the pollination of subarctic and arctic plants has been reported by a number of authors and summarised in Danks (1981). The importance of insect pollinators (especially bumblebees) to a number of plants that occur in the Hudson Plains Ecozone (e.g., *Salix*, *Saxifraga*, *Dryas*, and *Pedicularis* spp.) has been detailed on Ellesmere Island in the arctic (Kevan 1972). For example, *Salix and Pedicularis* spp. appear entirely dependent on bumblebee-facilitated pollination, with the former relying heavily on pollination by *Bombus polaris*. *Dryas* and *Saxifraga* spp. require pollination by insects to maximize seed set (Kevan 1972).

Since the 1970s, bumblebee (especially *Bombus* spp.) populations have been declining at a continental scale, including in southwestern Ontario (Colla and Packer 2008). Proposed reasons for these declines include pathogen spillover from commercial colonies, pesticide use, and habitat loss due to agriculture and urbanization (Colla and Packer 2008). It is not known if similar trends in bumblebee populations are occurring in the Hudson Plains Ecozone, but commercial apiaries and large-scale agriculture contribute minimally there to pathogen spillover or off-target pesticide effects and habitat loss, respectively. Mining and hydroelectric developments in the ecozone (see summary in Section 2.6.1.1, *Summary of Stressors*) might be impacting pollinator populations at local scales (via habitat loss), but, until baseline data are collected, any effects of these and future developments on pollinator trends will remain unknown.

Ecosystems that experience declines in pollinator populations are susceptible to cascading effects and reduced biodiversity, as a result of extirpation of pollinator-specialist plants (Bond 1994; Vamosi et al. 2006). The same concept likely holds true in the transitional Hudson Plains Ecozone, which comprises species of arctic, subarctic, and temperate geographic affinities (Riley 2003)[71]. Plants in the ecozone that are heavily dependent on insect pollination would be susceptible to population declines in what is already a pollination-limited environment (e.g., short season of pollinator activity) (Carlson et al. 2008), and they could experience greater competition by wind-pollinated plants.

The importance of other animal taxa (e.g., mammals, birds) for pollination in the Hudson Plains Ecozone appears to be unstudied.

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[71] The ecozone's vascular flora has been classified as being comprised of species with arctic, subarctic, and temperate geographic affinities (Riley 2003). Plant species commonly referred to as being of *boreal* affinity would mostly straddle Riley's (2003) subarctic (taiga) and temperate (forested boreal) vascular plant classes, depending on their distribution (J. Riley, The Nature Conservancy of Canada, pers. comm.).

2.5 **Ecosystem services**

Ecosystem services are benefits people obtain from ecosystems (MEA 2003) or, more specifically, ecological phenomena used actively or passively to provide benefits that directly impact human welfare (Fisher et al. 2008)[72]. These services include provisioning, regulating, and cultural services that directly affect people. They also include supporting services that affect people more indirectly or occur over very long time scales but are needed to maintain all other services and, thus, the conditions for life (MEA 2003). The capacity of ecosystems to deliver services is notably dependent on the maintenance of biodiversity (both its quantity and quality attributes), which is the source of many ecosystem goods (e.g., food). The resilience or capacity of the Hudson Plains Ecozone to continue providing ecosystem services is inherently linked to the condition of the ecozone, as it was described in sections 2.2 (*Ecosystem Structure*), 2.3 (*Ecosystem Composition*), and 2.4 (*Ecosystem Functions/Processes*).

The concept of ecosystem services was developed to address the link between ecosystems and human welfare by providing policy- and decision-makers with an operational decision-support system (tool) for evaluating conservation-conversion trade-offs (cost-benefit assessments) in the pursuit of sustainability (Fisher et al. 2008; Sukhdev et al. 2010). Such assessments generally require that monetary value be assigned to individual ecosystem services, regardless of whether these services provide people with market or non-market benefits. Several key issues still need to be addressed before the concept can be fully used for operational decision-support in policy, resource management, and land use planning. For example, safe minimum standards for ecosystem service provision have, for the most part, not yet been defined (i.e., sustainable supply) nor has methodology for capturing the non-market benefits of ecosystem services (Fisher et al. 2008). Meantime, however, inadequate consideration of ecosystem services in decision-making is considered an important factor contributing to the continued loss and degradation of biodiversity globally (SCBD 2010). As such, significant changes in ecosystem services still need to be identified, even if it is not currently possible to monetarily evaluate all such changes (Sukhdev et al. 2010).

To date, very little work on valuation of ecosystem services has been completed for the Hudson Plains Ecozone. This ecozone was included in a preliminary, first approximation of a selected subset of market and non-market values for Canada's boreal region[73] for the year 2002 (Anielski and Wilson 2005, 2009). A recognized shortcoming of that analysis was insufficient data specific to the area in question, both in terms of natural capital resources and the condition of ecosystem functions/processes (Anielski and Wilson 2005, 2009). This situation has not changed for the Hudson Plains Ecozone (this report). At present it is likely that the ecozone's complement of ecosystem services are for the most part stable and functioning well, given the ecozone's high degree of intactness and minimal development to date (Section 2.2.1.2, *Land Cover Change*). The effects of climate change and further development on the resilience or capacity of this ecozone to continue supplying ecosystem services are, however, uncertain. What is clear is that the ecozone's Aboriginal peoples will be forced to adapt to climate change and associated changes

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[72] Ecosystem services exclude inanimate, subsoil mineral and energy resources.

[73] The referenced analysis of Canada's boreal region included the following ecozones: Hudson Plains, Taiga Plains, Taiga Shield, Boreal Shield, Boreal Plains, Taiga Cordillera, and Boreal Cordillera. The study represented a first attempt at developing a beta-model or framework (termed Boreal Ecosystem Wealth Accounting System, BEWAS) for natural capital accounting in Canada's boreal region.

in ecosystem services, because their culture and traditional and wage-based economies have been shaped by the local environment, and they are still tied very closely to the land (e.g., Laidler and Gough 2003).

The first version of the ESTR profiles only a few ecosystem services of particular importance to each ecozone, and it reports on how the capacity of each ecozone to provide these services might or might not be changing. The focus is on amounts of services (service delivery) rather than total economic value (market and non-market) of services. A more comprehensive assessment of ecosystem services might be possible in future ESTR-like assessments.

In the discussion that follows, supporting, regulating, provisioning, and cultural services are considered separately[74], following ESTR convention. This approach differs from much of the literature on ecosystem services, which considers all ecosystem services together in context of a specific ecosystem structure or composition component, such as wetlands, forests, or grasslands (Wilson 2008) or a species like birds (Whelan et al. 2008).

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[74] It is recognized that the division of ecosystem services into these four service categories can be problematic from the standpoint of valuation as, for example, different services could provide the same benefit, meaning that valuation of individual services can lead to erroneous accounting in cost-benefit analyses (Fisher et al. 2008). This shortcoming is not important for this first version of the ESTR, where the focus is on trends in amounts of services rather than trends in monetary gains from the services.

2.5.1 Supporting services (summary)

Supporting services are those services necessary for the production of all other ecosystem services and, therefore, the conditions for life (MEA 2003). They include ecosystem functions or processes like photosynthesis (production of atmospheric oxygen), decomposition, soil formation and retention, nutrient cycling, and pollination. Supporting services differ from provisioning, regulating, and cultural ecosystem services in that their impacts on people are either indirect (supporting the production of another service) or occur over very long time periods. What is known about changes in key supporting services in the Hudson Plains Ecozone is briefly summarized here, based on the review of these services in Section 2.4, *Ecosystem Functions/Processes*.

To date, little evidence exists for changes in the rate, frequency, or relative importance of the ecozone's supporting services. An important caveat, however, is that the supporting services of this ecozone are for the most part not being monitored, and major knowledge gaps about their condition exist. Still, for the most part, the supporting services of the Hudson Plains Ecozone are assumed to be functioning properly and to be in good condition overall, owing to the relatively low amount of ecosystem change (alteration and loss) and other human influences there to date (Section 2.6.1.1, *Summary of Stressors*).

One notable exception is the significant changes in supporting services that have been occurring in the ecozone's coastal salt marshes, since the 1970s. Specifically, the human-facilitated excessive foraging of salt marsh vegetation by lesser snow goose (*Chen caerulescens careulescens*), which is converting the ecozone's salt marshes into areas of bare, hypersaline sediment (Section 2.2.2.1, *Coastal*), has demonstratably reduced the capacity of these salt marshes to provide supporting services, such as soil retention, nutrient cycling, and above-ground primary productivity (see reviews by Jefferies et al. 2003, 2006, as well as sections 2.2.2.1, *Coastal*; 2.4.3.2, *Herbivore-Plant Interactions*; and 2.4.5, *Nutrient Cycling*).

Although feeding by low densities of geese can increase nitrogen availability and primary productivity in coastal salt marshes, nitrogen cycling (including cyanobacterial nitrogen fixation) and primary productivity become impaired at higher goose foraging intensities, as exposure and erosion of sediment occurs, and damage otherwise becomes severe. The damage to the ecozone's coastal salt marshes is an ongoing concern, particularly given that the recovery of severely damaged areas could take decades. The geese are similarly damaging some freshwater areas in the adjacent tundra (Section 2.2.2.2, *Polar-Tundra*).

Climate change can be expected to fundamentally change the ecozone's supporting services, including the temperature-dependent processes of photosynthesis, decomposition, and nutrient cycling. Although fundamental changes in primary productivity are already suggested for some areas of Canada, increases in primary productivity appear to be much less in the Hudson Plains Ecozone, based on 1985-2006 trends in Normalized-Difference Vegetation Index (NDVI, a measure of gross primary photosynthesis and a proxy for green leaf area, based on remote sensing) (Pouliot et al. 2009; Ahern et al. in press). Some increase in productivity might also be suggested by limited observations of increased tree and shrub cover above the treeline, as near Churchill (Ballantyne 2009). Changes in primary productivity in the north are likely to be climate-driven, given the few changes in land use (Pouliot et al. 2009).

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2.5.2 **Regulating services**

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Regulating services are benefits obtained from the regulation of ecosystem processes (MEA 2003). They generally include climate regulation (includes carbon sequestration and storage)[75], air and water quality regulation, disturbance moderation (protection from natural hazards, such as floods), and the regulation of human disease (MEA 2003; Fischlin et al. 2007).

Similar to the situation with supporting services, major knowledge gaps exist about the condition of regulating services in the Hudson Plains Ecozone (e.g., see Section 2.4.2 for information on disturbances). Climate regulation and the flood control (disturbance moderation) and water filtration (water quality regulation) services afforded by freshwater (including wetlands) are, however, sometimes considered the most important ecosystem services provided by Canada's boreal ecozones, with a value estimated to be many times that of current levels of resource extraction (Anielski and Wilson 2005, 2009; see also Schindler and Lee 2010). These same ecosystem services are also forecast to be strongly impacted by climate warming.

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[75] Climate regulation is considered a regulating ecosystem service, because local or global climate can be impacted by ecosystem changes over time scales relevant to human decision-making, i.e., decades or centuries (MEA 2003).

In this first version of the ESTR, climate regulation is profiled as a regulating service of particular importance to the Hudson Plains Ecozone, as well as globally as the earth's climate continues to change. Among Canada's boreal ecozones, the Hudson Plains Ecozone has an exceptionally high capacity for carbon storage, which is associated with its especially high proportion of peatlands (Tarnocai and Stolbovoy 2006)[76,77].

2.5.2.1 **Climate regulation services**

Climate regulation services can be affected at both local and global scales. At a local scale, changes in land cover, such as wetland drainage, flooding (e.g., reservoir creation), afforestation, and deforestation, can affect temperature and/or precipitation (MEA 2003). In the Hudson Plains Eoczone, the cumulative ~30% loss or severe damage to coastal salt marsh vegetation that has occurred over the past four decades due to excessive foraging by a greatly increased Mid-Continent population of migratory lesser snow goose (*Chen caerulescens careulescens*) is demonstratably associated with local changes in temperature and evapotranspiration, as described in Section 2.2.2.1, *Coastal*. Other longer-term or permanent changes in land cover that might contribute to changes in climate regulating services at a local scale have been fairly minimal in this ecozone – generally limited to areas altered by the Opinaca reservoir (Québec); water diversions affecting the lower Churchill, Eastmain, Opinaca, and Rupert rivers and their associated river beds and banks (Manitoba, Québec); and some area affected by the Victor mine (Ontario) (Section 2.6.1.1, *Summary of Stressors*).

At the global scale, ecosystems regulate climate in part through carbon storage and release, by either sequestering (as a sink) or emitting (as a source) greenhouse gases (MEA 2003). Canada accounts for 87% of the peatland area in North America, and the Hudson Plains Ecozone is its largest peatland complex and the second largest peatland globally at northern latitudes (>40-50°) (Tarnocai and Stolbovoy 2006). As such, this ecozone stores an exceptionally large amount of carbon, on both a national and global basis (Figure 120). One study estimates the carbon stored in the ecozone's peatlands at 6.483 trillion tonnes, which accounts for 33% of total peatland carbon in Canada's boreal region, even though the Hudson Plains Ecozone covers only 6% of this land area (Table 29). From the same analysis, another approximately 945 billion tonnes of carbon is stored this ecozone's forests (Anielski and Wilson 2005, 2009). In other assessments, the Hudson Plains Ecozone has been estimated to contain approximately 33 giga tonnes (Gt) of soil carbon or 12% of the organic carbon stored in Canadian soils (Tarnocai 2000). More recently, Tarnocai et al. (2009) discovered that permafrost-affected soils contain more carbon than

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[76] Peat, with its associated high organic content, has accumulated in the Hudson Plains Ecozone over thousands of years (Klinger and Short 1996; Glaser et al. 2004a) – with high water levels and cold temperatures restricting drainage and decomposition rates (Gorham 1991; Michel and van Everdingen 1994) and the associated release of carbon (see Section 2.4.4, *Carbon Cycling*). Peat increases in depth from recently emerged areas along the coast to greater depths inland at more successionally mature sites (Dredge and Mott 2003; Glaser et al. 2004b).

[77] Although Stechbart and Wilson (2010) estimated relatively low biocapacity for the portion of the Hudson Plains Ecozone that lies in Ontario (i.e., a relatively low supply of resources and waste assimilation capacity available for human use), their analysis excluded wetland, as well as tundra, land classes. These land classes are not traditionally included in biocapacity estimates, because they are considered too dispersed or not conventionally productive enough to provide direct products or waste assimilation services that can be directly accounted for in systems of national accounts.

previously thought, the implication being that absolute values of carbon storage might have been strongly underestimated in permafrost areas across the globe. Updated regional estimates of carbon storage are not fully available at this time, partly because the Tarnocai et al. (2009) study did not differentiate among various types of permafrost sites (see Schindler and Lee 2010). Nonetheless, on a relative basis, the Hudson Plains Ecozone still has some of the highest carbon densities globally (Tarnocai et al. 2009).

Figure 120. Carbon storage (tonnes per hectare) in terrestrial ecosystems of the world. The red ellipse denotes the general geographic area of the Hudson Plains Ecozone, which is an area with exceptionally high carbon storage capacity.
Source: UNEP-WCMC (2009), used with permission. Map is based on data from Ruesch and Gibbs (2008) and Scharlemann et al. (in prep).

The comparatively[78] large carbon store in the Hudson Plains Ecozone has high value to society. In their fourth report, the International Panel on Climate Change (IPCC 2007) reported an average 2005 value for carbon of $US43 (~ $CAN52) per tonne, based on the damage costs of climate change to society[79]. Using this value, a preliminary valuation exercise for Canada's boreal region (includes the Hudson Plains Ecozone) for the year 2002 suggested that the annualized economic value of carbon stored there (20 year amortization) was higher than the net economic value of a range of other market and non-market ecosystem services and subsoil assets, including resources traditionally extracted by forestry, oil and gas, mining, and

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[78] Estimates of carbon vary depending on factors such as definitions used for ecosystem types, methods of estimating ecosystem areas (e.g., resolution of satellite imaging), depth of component carbon measurements, and methods of carbon accounting.

[79] The IPCC (2007) derived this mean ($US43) based on available peer-reviewed values but noted a large range (~$US10-350 per tonne of carbon) around it due to differences in assumptions related to climate, damage, and other factors.

hydroelectricity industries (Anielski and Wilson 2009). Indeed, the carbon market has been one of the most rapidly growing markets related to ecosystem services (MEA 2005; Barrington et al. 2010).

Table 29. Comparative peatland carbon storage in Canada's boreal ecozones[a]. In this analysis, the Hudson Plains Ecozone accounts for ~6% of the area of Canada's boreal region but ~33% of the carbon stored in its boreal peatlands.
Source: Derived from Anielski and Wilson (2005); see also Anielski and Wilson (2009).

Ecozone	Total ecozone area (ha)[a]	Peatlands		Carbon storage in peatlands (millions of tonnes)
		Area (ha)	Percent of ecozone area (%)	
Taiga Cordillera	26,366,000	6,700	0.03	1.1
Taiga Plains	63,722,000	14,110,000	22.1	2,372
Taiga Shield	135,431,000	9,705,400	7.2	1,632
Hudson Plains	36,734,000	24,868,600	67.7	6,483
Boreal Shield	199,642,000	24,515,400	12.3	6,391
Boreal Plains	74,412,000	9,816,100	13.2	2,559
Boreal Cordillera	47,772,000	177,500	0.37	84
Total, Canada's Boreal Region	584,079,000	83,199,800	14.2	19,522

[a] Ecozone areas reported in this study are based on the Ecological Stratification Working Group (ESWG 2005) ecozone boundaries, which differ somewhat from the ecozones$^+$ boundaries defined in the ESTR framework (Rankin et al. in press; see also the Preface to this report).

Data are currently insufficient to examine trends in the amount of carbon stored in the ecozone's peatlands (see Section 2.4.4, *Carbon Cycling*). Changes in carbon sequestration or storage, however, result largely from changes in land cover or use and climate change, both of which can alter the rate of exchange among carbon cycle components (Section 2.4.4, *Carbon Cycling*). Flooding of peatlands to create hydroelectric reservoirs can, for example, promote some emission of the greenhouse gases CO_2 and CH_4 (Duchemin et al. 1995). As already noted, relatively few changes in land cover or use have occurred in the ecozone to date, although development interests are increasing (this report).

More notably, climate change modeling projects major climate and ecosystem changes in and around the Hudson Plains Ecozone by 2100, including large losses of both sea ice and permafrost (Section 2.1, *Abiotic Drivers*). Because the ecozone's defining climatic and edaphic conditions are a result of sea ice and permafrost, the ecozone's extensive peatlands are forecast to be strongly impacted by climate change (Tarnocai 2006; see also Section 2.2.2.4.1, *Wetlands* (*Freshwater*)). Some of the carbon stored in the ecozone's peatlands could be released to the atmosphere as permafrost thaws and peat warms and dries (Tarnocai 2006; Schuur et al. 2009). Such release could lead to a positive feedback to atmospheric greenhouse gases (Schuur et al.

2009), which might be further exacerbated if large areas of dry peatlands burn (Weber and Stocks 1998; Flannigan et al. 2009) as projected (Flannigan et al. 2005; Bergeron et al. 2010) (see also Section 2.4.2.2, *Fire*)[80]. As such, the fate of the ecozone's massive store of carbon and the capacity of its peatlands to continue sequestering or storing carbon are of global concern for biodiversity and human well-being. Monitoring and research are required to track and understand how climate change affects the ecozone's climate regulation services, because implications to the ecozone's carbon sequestration and storage potential are not currently clear (Section 2.4.4, *Carbon Cycling*).

The importance of maintaining the ecozone's large peatland carbon stores (to help mitigate climate change, with co-benefits for biodiversity) is being increasingly recognized by managing jurisdictions. The Government of Manitoba recently committed to develop a boreal peatlands stewardship strategy in co-operation with stakeholders and leading climate change non-governmental agencies (Government of Manitoba 2009). The commitment was made coincident with the establishment of two new protected areas in the ecozone that have significant carbon stores (Section 2.6.2.1, *Protected Areas*). In Ontario, the vision to maintain carbon sequestration and storage is now articulated in the province's Far North Act (Government of Ontario 2010). To support the intent of this act, a science advisory panel recommended to the Ontario government that some conservation areas be designated where the densest carbon pools exist, and that these carbon stores should be given economic value for the benefit of local communities (and, conversely, that a cost for the net loss of carbon function be included in assessments of land use change) (Far North Science Advisory Panel 2010). The need to consider enhanced fire suppression efforts as climate change proceeds is also recognized, even if increasing fire suppression will be logistically and economically challenging in this geography (e.g., Stocks and Ward in press).

2.5.2.1.1 Permafrost as an indicator of changes in climate regulation services

This first version of the ESTR uses permafrost as an indicator of changes in climate regulation services, and in particular carbon sequestration and storage, for those ecozones with permafrost. As already noted, the thawing and loss of permafrost with climate change can lead to the destabilization and release of carbon stores and, therefore, a positive feedback to atmospheric greenhouse gases (Schuur et al. 2009).

Temperatures are increasing in the Hudson Plains Ecozone (Section 2.1, *Abiotic Drivers*), but insufficient trend data are currently available with which to evaluate the extent of any associated change in permafrost. Until relatively recently, no permafrost thermal monitoring sites were located and maintained in the ecozone to help track changes in permafrost, as is being done elsewhere in Canada's north (Smith et al. 2005; Smith in press) (new permafrost sites in the ecozone are identified in Section 2.1). Changes in permafrost are, however, suspected in this ecozone. Both collapse and erosion features and aggrading features are visible in the ecozone's permafrost tension zone, and collapse features appear to have become more widespread over time, as in the Ekwan to Lake River areas of the northern James Bay coast (Riley 2003). In recent decades, casual observations have also been made of slumping and collapse of rivers banks in the northwestern portion, along the Hayes and Nelson rivers near

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[80] Fires result in an immediate pulse of carbon to the atmosphere, as well as post-fire enhancement of biogenic emissions (increased soil respiration and decomposition of remaining organic matter), which could be particularly significant in peatlands (Zoltai et al. 1998).

York Factory, which is an area geographically close to the boundary between discontinuous and continuous permafrost (permafrost class boundaries are shown in Figure 23 in Section 2.1, *Abiotic Drivers*). A long-term (non-successional) trend for partial degradation and conversion of frozen peat plateaus to fens, as well as the enlargement of some associated lakes from eroding shorelines, is also suggested in the area from the Nelson River north to Churchill (Dyke and Sladen 2010). Moreover, a permafrost warming of ~0.5 °C is suggested for Churchill, since the mid-1970s (Kershaw 2010).

Although permafrost thaw has not been quantitatively verified in the Hudson Plains Ecozone, it is known to be occurring in the southern extent of the permafrost zone in association with climate change (Smith in press). For example, analysis of tree rings and direct measurements suggested that permafrost decay rate significantly increased over the period 1950-2002 at Gillam, Manitoba, which is just outside the southwestern boundary of the ecozone, where permafrost is classified as discontinuous (Camill 2005); there, permafrost loss also accelerated during the last assessment period (1995-2002). Likewise, trend analysis using decadal field surveys (1973-2003, with 1957 baseline air photo) of permanent landforms showed that permafrost decay rate has also been increasing (and accelerating after 1993) within Québec's subarctic peatlands along the southeast coast of Hudson Bay, within the adjacent Taiga Shield Ecozone, where permafrost is also classified as discontinuous (Payette et al. 2004). A more recent study (2004-2005) further south in the James Bay area of Québec, i.e., very close to the northeastern boundary of the Hudson Plains Ecozone, where permafrost is classified as being in isolated patches, likewise reported evidence for accelerated permafrost thaw, and it further suggested that permafrost in the area might have already regressed 130 km north, likely over the last 50 years (Thibault and Payette 2009). The latter study was based on a comparison of current conditions in a post-fire chronosequence of bogs with historical air photos (including 1957 baseline) but with very limited ground sampling of permafrost (two bogs).

The Hudson Plains Ecozone is able to support the most southern continuous permafrost in North America (Zhang et al. 2008) due in large part to the presence of seasonal sea ice in Hudson and James bays (Gough and Leung 2002). The annual period of sea ice cover in western Hudson Bay, southern Hudson Bay, and James Bay (areas adjacent to the ecozone) is, however, decreasing; it has become an average of ~3 weeks shorter, since the mid-1970s (Gagnon and Gough 2005a; Section 2.1, *Abiotic Drivers*). Permafrost loss in the ecozone is expected to occur as a lagged dynamic of such sea ice loss, through feedback effects on the ecozone's climate. As discussed in Section 2.1.2.2, climate modeling specific to the Hudson Bay region (Gough and Wolfe 2001; Gough and Leung 2002; Gagnon and Gough 2005b) projects a substantial lengthening of the ice-free season to complete loss of seasonal sea ice from James Bay and the southern portion of Hudson Bay by 2100, along with an associated loss of at least 50% of the continuous permafrost (and complete loss of currently discontinuous and patchy permafrost) and a virtual elimination of a climate that supports permafrost in the Hudson Plains Ecozone (see also Joly et al. 2010 for sea ice projections to 2070 using a higher-resolution, regional climate model). Permafrost and associated carbon dynamics are, however, not expected to change uniformly across the ecozone (Section 2.4.4, *Carbon Cycling*), as this ecozone supports a full range of permafrost types and associated climatic and edaphic variation across its geography. Some priority would ideally be placed on permafrost monitoring in the Hudson Plains Ecozone, given both the currently suspected and projected loss of permafrost and associated implications to the ecozone's climate regulation services and biodiversity.

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2.5.3 **Provisioning services**

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The provisioning services of the Hudson Plains Ecozone (i.e., goods derived from the living portion of the ecosystem, such as food, furs, and plant fibre) are still very important for the majority of the principally Aboriginal peoples that live there (Berkes et al. 1994, 1995). The importance of provisioning services is suggested by several indicators, including wildlife harvesting effort, harvesting participation rates, magnitude of the harvest, frequency of bush food consumption, the degree of sharing, the replacement value of the harvest, and the contribution of the traditional sector to the overall regional economy. Moreover, ~75% of the land area in the Ontario portion of the ecozone alone is actively used by resident Aboriginal peoples for hunting, trapping, fishing, and gathering (Berkes et al. 1995).

In terms of provisioning services, the Hudson Plains Ecozone differs from many other ecozones in Canada in that it provides little opportunity for large-scale production of biotic goods from forest harvesting or agriculture, owing to its pedology and geomorphology (sections 1.1, *Geology, Topography & Climate* and 2.2.1, *Overview of Ecozone Structure & Land Cover Change)*. A small portion of the boreal forest in the southernmost part of the ecozone in Ontario is the only location where commercial harvesting of timber can occur (OMNR 2006). Agriculture remains unimportant (McConkey et al. in press), and food production occurs only at an extremely local scale, with some households tending gardens in their yards (OMNR 1985). Some opportunities for commercial exploitation of the Hudson Plains Ecozone do exist (e.g., hydroelectric and wind power generation, mineral and gem mining), and a limited number of hydroelectric and mining developments are present (see sections 1.2.2, *Economic History* and 2.6.1.1, *Summary of Stressors*). However, for the most part these activities do not centre on provisioning services provided by the living component of the ecozone. Exceptions include recreation- or tourism-based hunting and fishing.

The remainder of this section highlights wildlife harvest for food and fur as particularly important provisioning services in the Hudson Plains Ecozone. Current, standardized, and complete data on provisioning services are, however, lacking for the ecozone as a whole. As such, some of the trend information presented has spatial or temporal gaps in representation. Although trends are shown for the ecozone, or relatively large, jurisdictional portions of the ecozone, it is important to note that harvesting patterns and land use vary among individual

Aboriginal communities following spatial and temporal variations in resource abundance (Thompson and Hutchison 1987; Berkes et al. 1994). As well, flows of provisioning services do not always accurately reflect the capacity of the ecozone to provide these services. Flows could be affected by extraneous factors (e.g., market conditions, number of hunters, and individual trapping effort), and current flows might or might not be sustainable over the long term (MEA 2003). In the discussion that follows, extraneous factors that could be influencing the flow of provisioning services are identified, where known.

2.5.3.1 **Food from wildlife harvest**

Wildlife is very important as a food source for Aboriginal peoples in the Hudson Plains Ecozone. As illustrated for the Ontario and Québec portions of the ecozone in Figure 121, cervids (including caribou, *Rangifer tarandus* and moose, *Alces alces*) and waterfowl (including geese and ducks) form the major component of food harvested, with fish, furbearers, and small game contributing considerably less (NHR 1982; Berkes et al. 1994). Trends in cervid and waterfowl harvest are profiled below.

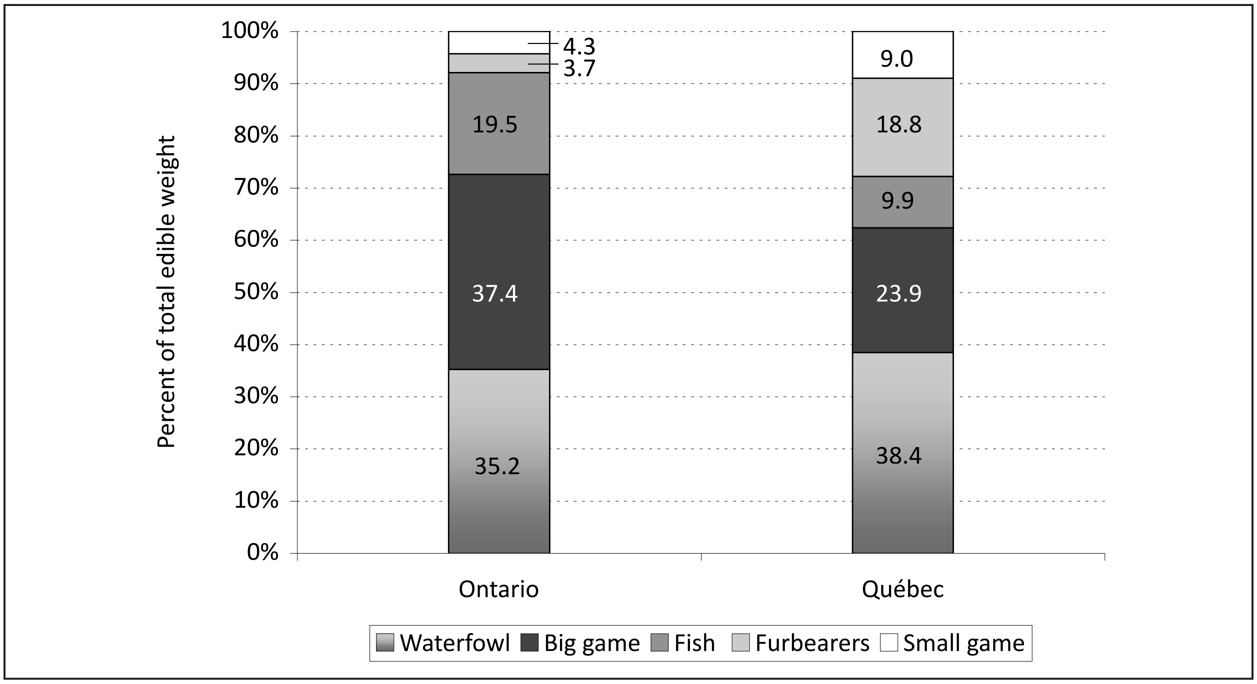

Figure 121. Relative importance of various animal groups harvested by Aboriginal peoples in the Ontario and Québec portions of the Hudson Plains Ecozone, as measured by proportion of total edible weight. The big game category is primarily comprised of woodland caribou and moose.
Sources: Data are from NHR (1982) and Berkes et al. (1994).

2.5.3.1.1 **Cervids: caribou & moose**

Caribou

Harvesting of caribou in the ecozone (herds are described in Section 2.3.3.2.1, *Caribou*) by Aboriginal peoples has most frequently occurred from March to May (OMNR 1985). Caribou harvesting by non-Aboriginal peoples was banned in Ontario in 1929 (OMNR 2008), but it continues on a limited and licensed scale in Manitoba's portion of the ecozone (Manitoba

Conservation 2010). In Québec, sport hunting for caribou is permitted in some northerly areas but not specifically within the geography of the Hudson Plains Ecozone (QRNF 2010).

In the Manitoba portion of the ecozone, licensed hunting of caribou occurs principally in the province's Game Hunting Areas (GHA) 2 and 3 (Manitoba Conservation 2010), e.g., for the 2009 hunting season, 120 and 75 licences were sold for these two GHAs, respectively (Manitoba Conservation, unpublished data). Harvest data from these areas are not available. Past experience has revealed that questionnaire returns are so small that it is impossible to obtain consistent data to ascertain trends. Harvest of woodland caribou (*Rangifer tarandus caribou*) by Aboriginal peoples has, however, been higher over the last two decades than previous levels (V. Crichton and D. Hedman, Manitoba Conservation, pers. comm.). In GHA 2 (Churchill), crude estimates of the harvest of barren-ground caribou (*Rangifer tarandus groenlandicus*, Qamanirjuaq herd), which only occasionally migrates into the ecozone (Section 2.3.3.2.1, *Caribou*), suggest a decrease in harvest, since the 2004-2005 season, with 150, 50, 20, 40, and 45 animals harvested annually over respective years to 2008-2009 (BQCMB 2005, 2006, 2007, 2008, 2009). Information was insufficient to estimate harvest levels for the 2009-2010 season, either in GHA 2 or elsewhere in this subspecies' range (BQCMB 2010). Some concern exists that the Qamanirjuaq herd might be declining, and that overall harvest levels could be unsustainable (BQCMB 2010).

Many other sources report on Aboriginal harvesting of caribou in the Hudson Plains Ecozone but years, methodology, units, and reporting of data are variable. For the bulk of the ecozone that lies in Ontario, these disjunctive sources might suggest an upward trend in the number of woodland caribou harvested annually, since the early 1980s. Thompson and Hutchison (1987) estimated that 461 and 559 caribou were harvested for the 1981-1982 and 1982-1983 hunts, respectively. OMNR (1985) estimated that ~700 caribou were harvested annually, and extrapolation of data from Berkes et al. (1994) gives an estimate of 728 caribou harvested for the 1989-1990 hunt. Notably, an unusually high harvest of 400-613 caribou was estimated for Peawanuck alone in the fall of 1996, while ~200 caribou were harvested from Fort Severn that same season (Ontario Ministry of Natural Resources, unpublished data). In the Ontario portion of the ecozone, a greater number of caribou appear to be harvested by northern communities than by communities south of Kashechewan (Thompson and Hutchison 1987; Berkes et al. 1994). The 1996 harvest values for Peawanuck and Fort Severn are, however, still unusually high for these northern communities.

In the Québec portion of the ecozone, harvest of woodland caribou was variable between 1972 and 1978, with annual harvest ranging from 15 to 42 individuals and a mean of 27.3 (NHR 1982). No recent reports of woodland caribou harvest were located for Aboriginal communities in this area of the ecozone.

Moose

Moose in the ecozone (Section 2.3.3.2.2, *Moose*) are harvested by both Aboriginal and non-Aboriginal peoples. There is no evidence for declining harvest. Over the past decade or so, no trends in licensed moose harvest have been evident in the Manitoba portion of the ecozone (Figure 122), and total moose harvest has been relatively stable in the Ontario portion (Figure 123). Moose hunting activity throughout the Manitoba portion of the ecozone is very low, with most activity occurring along the larger rivers in the province's GHA 3 and non-existent hunting pressure in GHA 2. In contrast to the situation in the Manitoba and Ontario portions of the ecozone, moose harvest in the Québec part of the ecozone appears to have increased, since the mid-1980s (Figure 124).

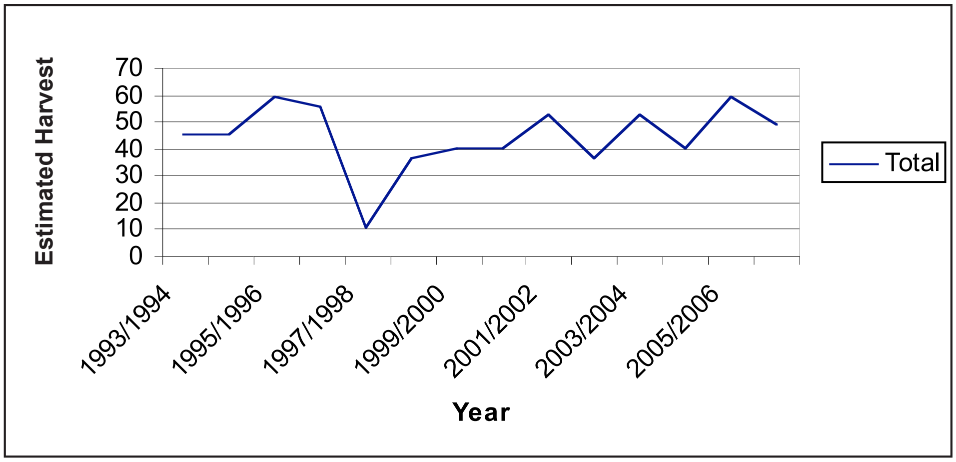

Figure 122. Moose harvest by licensed residents in Manitoba Game Hunting Areas (GHAs) 2 and 3 over the period 1993-1994 to 2005-2006. In addition to data for the resident moose hunt shown here, approximately 12 foreign licenses are issued annually with an associated harvest of 8-10 moose (not included). Source: Data are from Manitoba Conservation, Wildlife and Ecosystem Protection Branch.

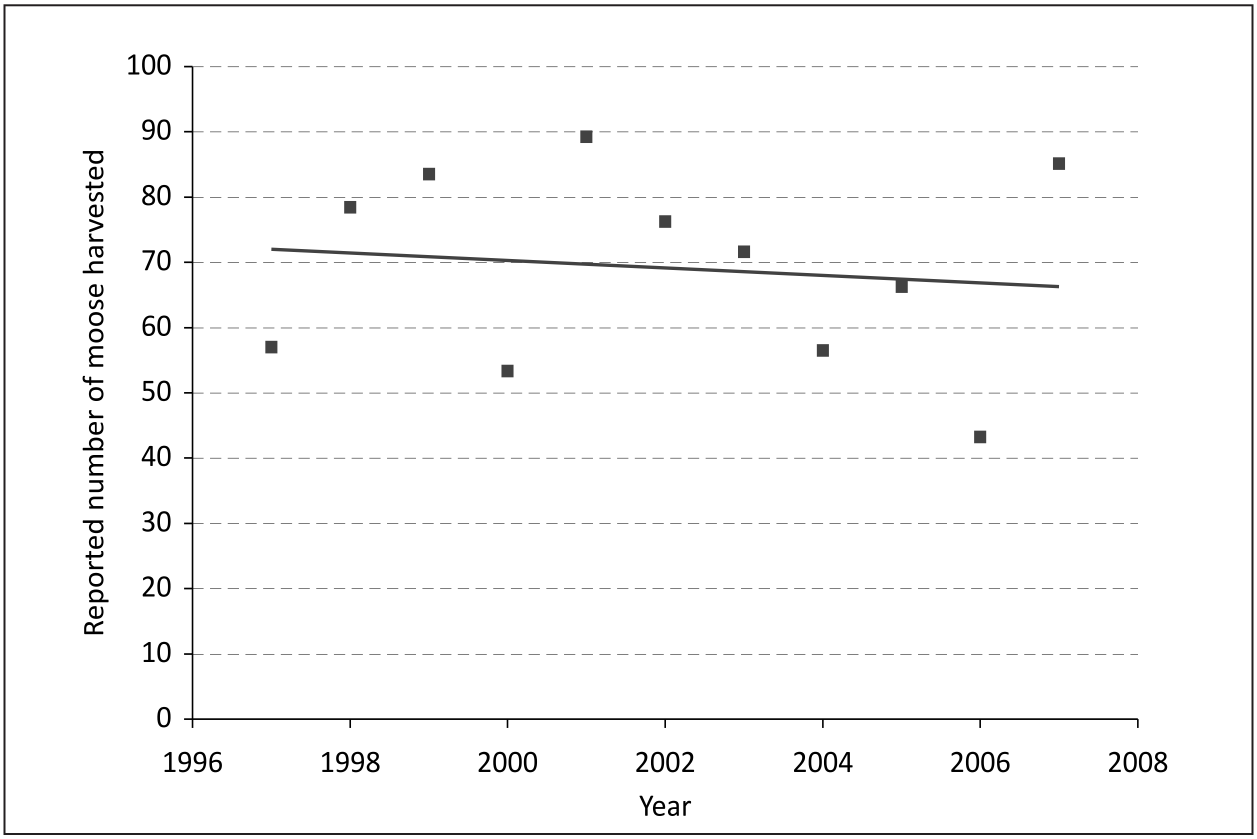

Figure 123. Moose harvest by Aboriginal and non-Aboriginal peoples in the Ontario portion of the Hudson Plains Ecozone, 1997-2007. Data are for Ontario Wildlife Management Units (WMUs) 01A, 01D, 18B, and 25 (WMUs 17, 24, and 26 not included).
Source: Data are from Ontario Ministry of Natural Resources, Ontario Terrestrial Assessment Program.

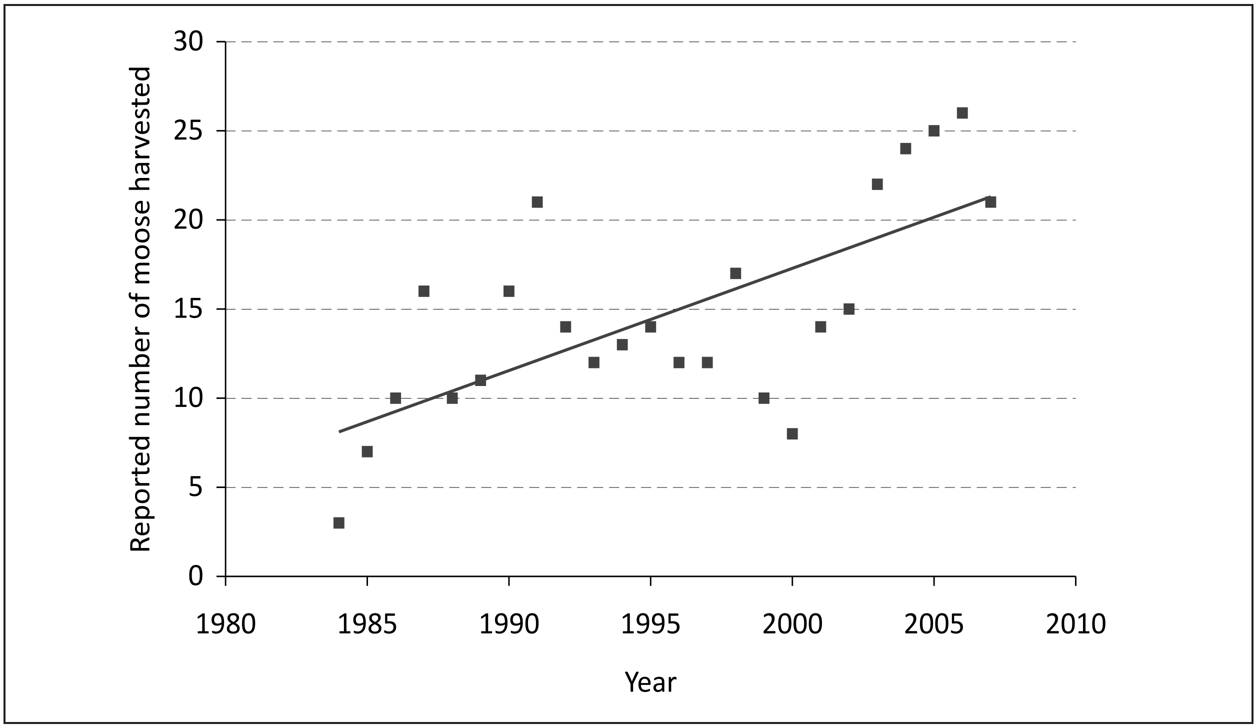

Figure 124. Moose harvest by Aboriginal and non-Aboriginal peoples in the Québec portion of the Hudson Plains Ecozone, 1984-2007.
Source: Data are from Pilote des systèmes grande faune et animaux à fourrure – Service de la Faune terrestre et de l'Avifaune, Ministère des Ressources Naturelles et de la Faune (data from Zone 22).

2.5.3.1.2 **Waterfowl: geese & ducks**

Waterfowl (Section 2.3.3.3.2) is another important food source for Aboriginal peoples in the Hudson Plains Ecozone, based on edible weight (refer back to Figure 121). The waterfowl hunt has also had the greatest percentage of participants for each community surveyed in the ecozone, ranging from a low of 55% to a high of 100% with a mean of 81% participation (Berkes et al. 1994[81]). For most Ontario communities in the ecozone, Canada goose (*Branta canadensis*) contributes most of the spring harvest (Hughes and Walton 2005), as it is the first species to arrive; harvesting these returning birds has high cultural significance, as part of the annual renewal of the relationship with the land. Lesser snow goose (*Chen caerulescens caerulescens*) migrates later in the spring, and it is a more coastal species, so that access to it is more difficult in parts of the ecozone (the more northerly communities on Hudson Bay harvest relatively more snow geese in spring) (Thompson and Hutchison 1987). Lesser snow goose was historically harvested more extensively in the fall (Hanson and Currie 1957; Prevett et al. 1983; Berkes et al. 1994). Although it is still more important overall than Canada goose to Aboriginal harvest in the ecozone, within the last decade or two the importance of the fall harvest of lesser snow goose has strongly declined in southern James Bay, which was once the major fall staging and harvesting area in the ecozone (Hughes and Walton 2005; Sayles 2008; see also the Québec harvest information later in this section). Duck (especially pintail, *Anas acuta*, and mallard, *A. platyrhynchos*) harvests become increasingly important for more inland communities in the ecozone (NHR 1982; Prevett et al. 1993).

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[81] New Port was removed from calculations, because it falls outside of the ecozone.

Recent and/or complete waterfowl harvest data for the Hudson Plains Ecozone are lacking in all three component provinces. No waterfowl harvest data (historical or current) are available for the Manitoba portion of the ecozone. As outlined below, thorough waterfowl harvest studies were reported for Ontario and Québec in the past but not after 1990 and 1979, respectively. Some Aboriginal communities in Ontario (e.g., Moose Cree, Attawapiskat) now gather their own waterfowl harvest data. In Québec, a pilot project was recently undertaken on subsistence harvest of waterfowl by the Cree of Waskaganish (see later).

Figure 125 shows the historical trends in reported waterfowl harvest per community in the Ontario portion of the ecozone. Prior to 1990, trends in lesser snow goose and duck harvest appear relatively stable, while Canada goose harvest might have increased up to that time period. More recent data are very limited. Some data are available for the 2003-2004 season for Moose Factory, Moosonee, Fort Albany, Kashechewan, and Attawapiskat (Hughes and Walton 2005). These data were excluded from Figure 125, because they are not readily comparable to the historical data; harvest per community can only be calculated based on an estimate of the number of potential hunters per community, rather than the number of actual hunters. As noted by Hughes and Walton (2005), mean harvest per hunter is instead the only metric on which to compare the 2003-2004 survey results to those of previous surveys (see below), and this metric does not address total harvest trend.

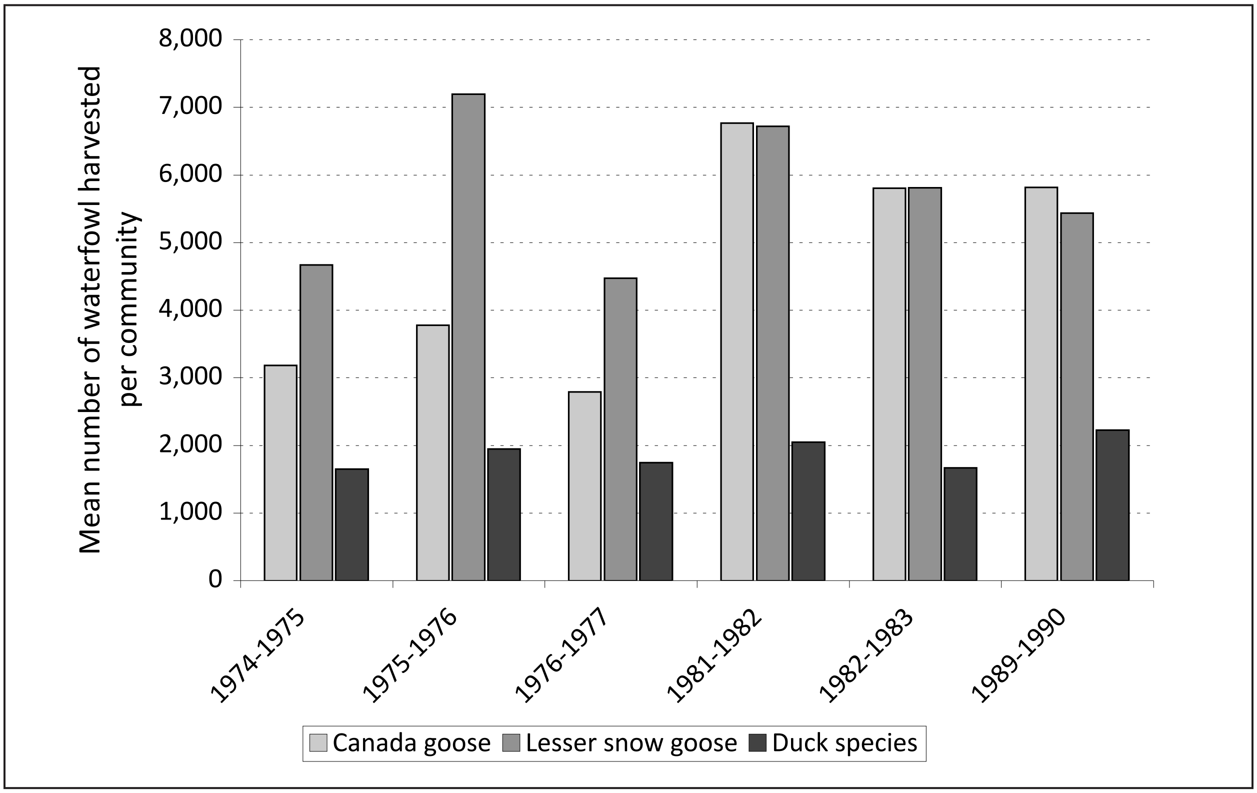

Figure 125. Mean number of reported Canada goose, lesser snow goose, and duck species (combined) harvested per community in the Ontario portion of the Hudson Plains Ecozone, 1974-1975 to 1989-1990. Means are calculated from seven communities (Moose Factory, Moosonee, Fort Albany, Kashechewan, Attawapiskat, Peawanuck, and Fort Severn) except for 1981-1983, which included Moose River Crossing and excluded Fort Severn.
Sources: Data are from OMNR (1985) and Berkes et al. (1994).

In the Ontario portion of the ecozone, average harvest of geese per hunter appears to have increased, since the 1950s. In the 1950s, ~30 geese were killed per Aboriginal hunter (Hanson and Currie 1957; Prevett et al. 1983). This value increased to an average of 94 geese/hunter by 1990, with a high of 176 geese/hunter in Fort Severn (Berkes et al. 1994). The most recent data from five Ontario communities on the west coast of James Bay (Moose Factory, Moosonee, Fort Albany, Kashechewan, Attawapiskat) suggests an average catch of ~73 geese/hunter, with a high of 166 geese/hunter in Attawapiskat (Hughes and Walton 2005).

These apparent increases in historical goose harvest and catch per hunter are also partly the result of Aboriginal peoples shifting to a more modern lifestyle. Many of the men (Prevett et al. 1983) and women are now wage earners, which results in families being increasingly tied to their community. This situation has resulted in a concomitant reduction in the number of families that hunt from interior traplines in the winter, which renders them available to hunt on the coast in late fall and early spring (when they would otherwise be in the interior). Moreover, firearms have become increasingly important to the hunt, while improvements in bush transportation (e.g., snowmobiles, outboard motorboats) and modern use of helicopters have made prime hunting locations quickly accessible from the communities, so that short weekend hunts become feasible (J. Robus, Ontario Ministry of Natural Resources, pers. comm.). Finally, community and personal freezers allow large quantities of geese to be quickly preserved (Prevett et al. 1983; OMNR 1985; Berkes et al. 1995).

Within the Québec portion of the ecozone, limited available data show historical waterfowl harvest to be variable but to follow a general pattern similar to that in Ontario, with the exception of a low harvest year in 1974-1975 (Figure 126). Mean harvest of both Canada goose and lesser snow goose per harvester showed annual variability between 1975 and 1979 (NHR 1982), similar to the pattern in Ontario. Historically, of the two major goose species, Canada goose was the more important of the two species harvested in Québec during pre-development studies, with greater relative proportions harvested in both the spring and fall (NHR 1982). A recent Québec Cree subsistence harvest pilot project conducted

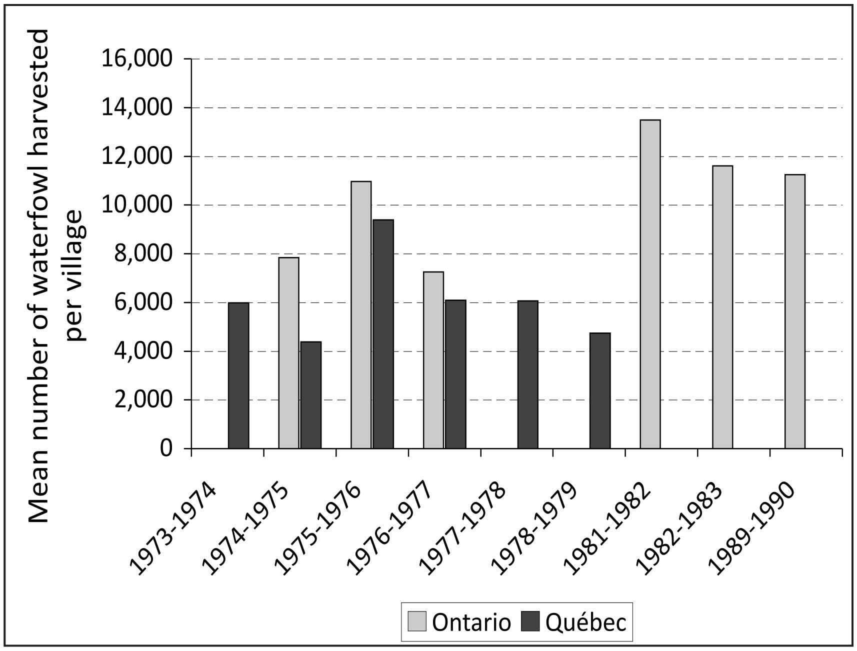

Figure 126. Mean number of waterfowl (pooled means for Canada goose and snow goose) harvested per community for the Ontario and Québec portions of the Hudson Plains Ecozone, 1973-1974 to 1989-1990. Waterfowl means for Ontario are calculated from seven communities (Moose Factory, Moosonee, Fort Albany, Kashechewan, Attawapiskat, Peawanuck, and Fort Severn) except for 1981-1983, which included Moose River Crossing and excluded Fort Severn. Waterfowl means for Québec are calculated from the communities of Eastmain and Waskaganish.
Sources: Data for Québec are from NHR (1982), and data for Ontario are from OMNR (1985) and Berkes et al. (1994).

as a partnership between the Cree Regional Authority, the Cree Trappers Association, and Environment Canada under the Northern Ecosystems Initiative (CRA 2009) indicates that Canada goose is still the most important waterfowl species harvested in Waskaganish (Figure 127). In 2005, 2006, and 2008, Canada goose harvest was over five times greater than the harvest of mallard duck (the next highest harvested species), with 49-60% of Waskaganish community members participating in the mallard hunt during this period (CRA 2009).

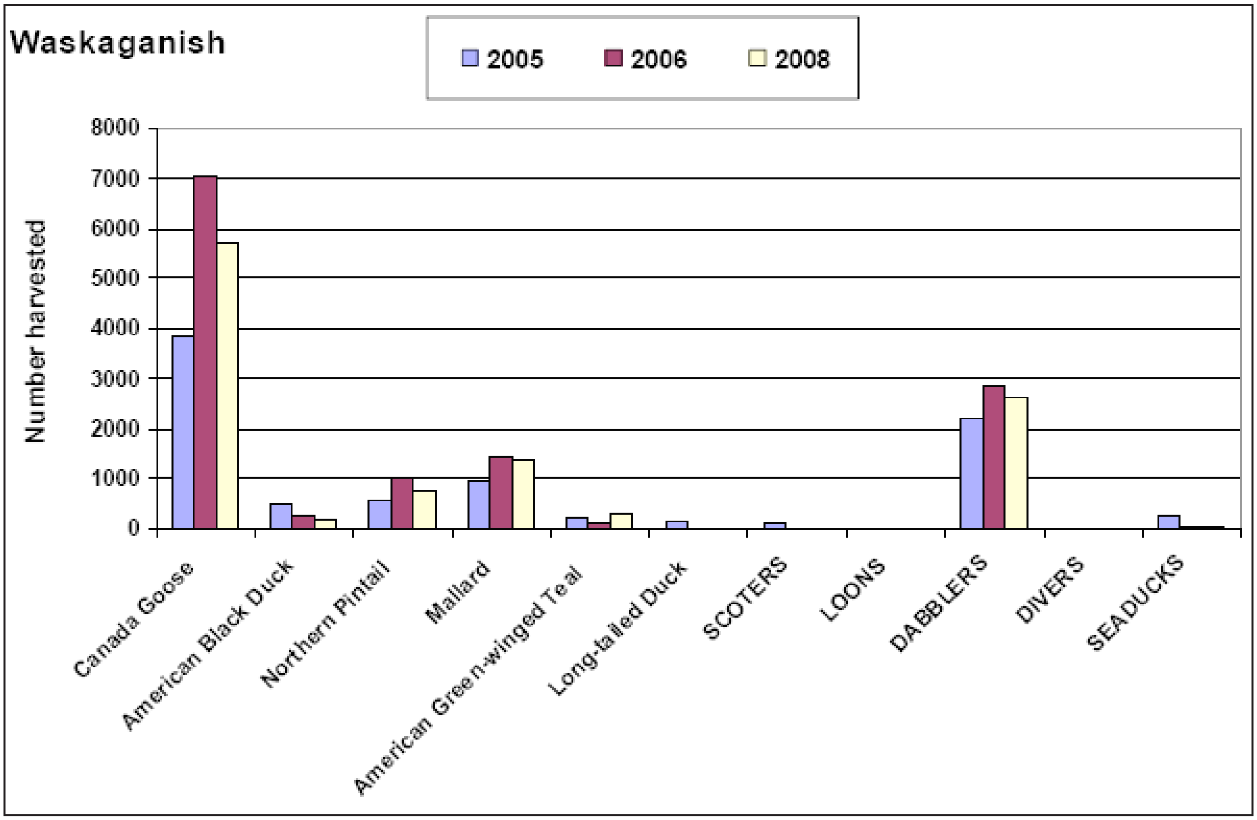

Figure 127. Subsistence harvest of the most common waterfowl species or groupings at Waskaganish, Québec during 2005, 2006, and 2008.
Source: Migratory bird data collection in Cree communities (2005-2008), Cree Regional Authority, Cree Trappers Association and Canadian Wildlife Service, 2009, used with permission.

Irrespective of limited harvest data (as above), Aboriginal peoples have reported a long-term trend (several decades) for a reduction in goose hunting success along the eastern James Bay coast (e.g., McDonald et al. 1996; Peloquin and Berkes 2009; Sayles and Mulrennan 2010). From a resource harvesting perspective, lesser snow goose has become largely unavailable there (note the lack of representation of snow goose in Figure 127), and the availability of Canada goose has been much reduced. Aboriginal peoples attribute this reduced hunting success to a number of behavioural changes in both geese (e.g., shifts in flight patterns inland from the coast) and hunters (various transgressions from customary practices). Although not necessarily causal, some changes in goose habitat have occurred there other than those related to natural isostatic rebound and traditional habitat alterations to improve hunting (such as the construction of dykes and tuuhiikaan (corridors) to retard coastal succession and influence goose flight paths). These other changes are associated with hydroelectric development, and they include deterioration in subtidal eelgrass (*Zostera marina*) beds (see eelgrass profile in Section 2.3.3.7, *Vascular Plants*) and creation of inland reservoirs, including the Opinaca reservoir (Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Overall, although recent data on waterfowl harvest are not available for most of the Hudson Plains Ecozone, waterfowl hunting is still a very important component of the Aboriginal lifestyle. Goose habitat in certain parts of the ecozone has changed (as above, and sections 2.2.2.1, *Coastal* and 2.2.2.2, *Polar-Tundra*), but overall goose population numbers are not similarly declining. The Mid-Continent population of lesser snow goose, as well as Canada goose populations (Eastern Prairie, Mississippi Valley, Southern James Bay, and Atlantic populations), are elevated above mid-twentieth century levels, primarily as a result of increased reliance on agricultural feeding resources further south (sections 2.2.2.1, *Coastal* and 2.3.3.3.2, *Waterfowl*). However, the capacity of some portions of the Hudson Plains Ecozone to provide snow geese for Aboriginal subsistence harvest has changed, including in the southernmost portion of the ecozone, where snow geese no longer migrate or stage in large numbers during either spring or fall hunting periods.

Waterfowl management needs to consider the shared nature of this resource, both by Aboriginal and non-Aboriginal peoples and at a broad geographic scale, because the waterfowl species are migratory within North America. In the Ontario part of the Hudson Plains Ecozone alone, Aboriginal peoples contributed ~10% of the total eastern North American goose harvest in the late 1970s (OMNR 1985). Overall, however, the proportion of the Mid-Continent population of lesser snow goose (which winters in the southern United States) harvested by these Aboriginal peoples was small, ~5% in the 1950s (Hanson and Currie 1957). Their harvest remains a very small fraction of the total lesser snow goose population today.

Clearly, improved waterfowl harvest monitoring is required in all component jurisdictions of the Hudson Plains Ecozone if trends in this important ecosystem good are to be adequately tracked over time.

2.5.3.2 Furs from wildlife harvest

In the Hudson Plains Ecozone, furbearers (Section 2.3.3.2.3) are trapped largely for their pelts (furs), which are typically sold outside of the ecozone (OMNR 1985; D. Berezanski, Manitoba Conservation, pers. comm.). In addition to export for sale, furs are used by Aboriginal peoples within the ecozone to make clothing for personal use and handicrafts for tourists. Importantly, most, if not all, records of fur harvest in the Hudson Plains Ecozone come from official sealing records and Manitoba mandatory dealer reports and, therefore, the absolute total harvest (i.e., including animals retained by Aboriginal peoples for personal use) is unknown. Still, revenue generated by the sale of pelts is important to many communities, where other sources of income might be few.

The most important furbearers trapped in the Ontario and Québec portions of the ecozone in terms of numbers harvested have been: American marten (*Martes americana*), American beaver (*Castor canadensis*), muskrat (*Ondatra zibethica*), American mink (*Mustela vison*), Northern river otter (*Lontra canadensis*), and red fox (*Vulpes vulpes*) (Figure 128). In Manitoba, the majority of furbearer harvest has been marten, with other species minimally harvested. Some other regional differences exist as, for example, beaver has been notably more important in Québec than Ontario, both in terms of proportion harvested (Figure 128) and revenue generated (NHR 1982). Most trapping in the ecozone has occurred adjacent to coastal communities, although some trappers move to the interior each fall.

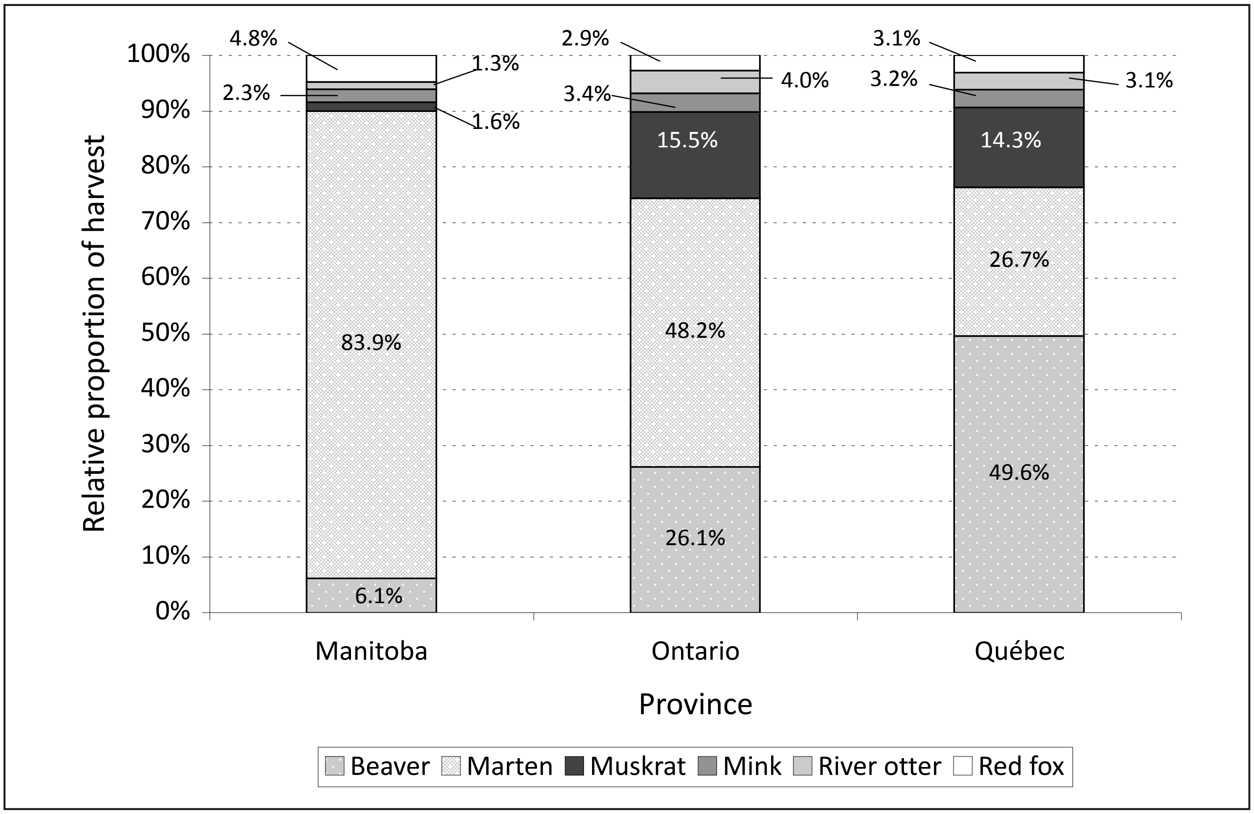

Figure 128. Relative proportion of furbearer harvest of select species measured by mean number reported or sealed per community for Manitoba (1996-1997 to 2006-2007), Ontario (1973-1974 to 2006-2007), and Québec (1983-1984 to 2006-2007) portions of the Hudson Plains Ecozone. For Manitoba, the average is based on mandatory fur dealer reports for traplines entirely or partially in the Hudson Plains Ecozone, i.e., those in the Churchill, Limestone, Shamattawa, Gods Lake, and Split Lake Registered Trapline sections. For Ontario, the number of communities participating varied from year to year; therefore, the average is based on seven main communities in the ecozone (Moose Factory, Moosonee, Fort Albany, Kashechewan, Attawapiskat, Peawanuck, and Fort Severn). For Québec, the average is based on sealing records from Eastmain and Waskaganish.
Sources: Data are from Manitoba Conservation; Ontario Ministry of Natural Resources; and Système Fourrure Ministère des Ressources Naturelles et de la Faune.

Since the early 1980s, the mean number of animals trapped for fur per community has decreased in the ecozone overall (Figure 129), largely as a result of trends in beaver and muskrat harvest (Figure 130). For the three major species trapped (marten, beaver, and muskrat), community averages by province show similar decreasing trends (Figure 130). Some regional differences are, however, apparent. In Manitoba, mean beaver and muskrat harvests per community have been considerably lower than both Ontario and Québec (Figure 130a), and a trend for decreasing marten harvest, since the mid-1980s, is apparent for Manitoba and Ontario but not Québec (Figure 130c). Mink, otter, and red fox harvests have also decreased but less markedly.

Population estimates for the furbearing species trapped within the Hudson Plains Ecozone are not available for any component jurisdiction or ecological subsection (Section 2.3.3.2.3, *Furbearers*). Thus, maximum sustainable harvest levels of furbearing mammals in the ecozone are also unknown. Nonetheless, the declines in fur harvest in the ecozone are probably not

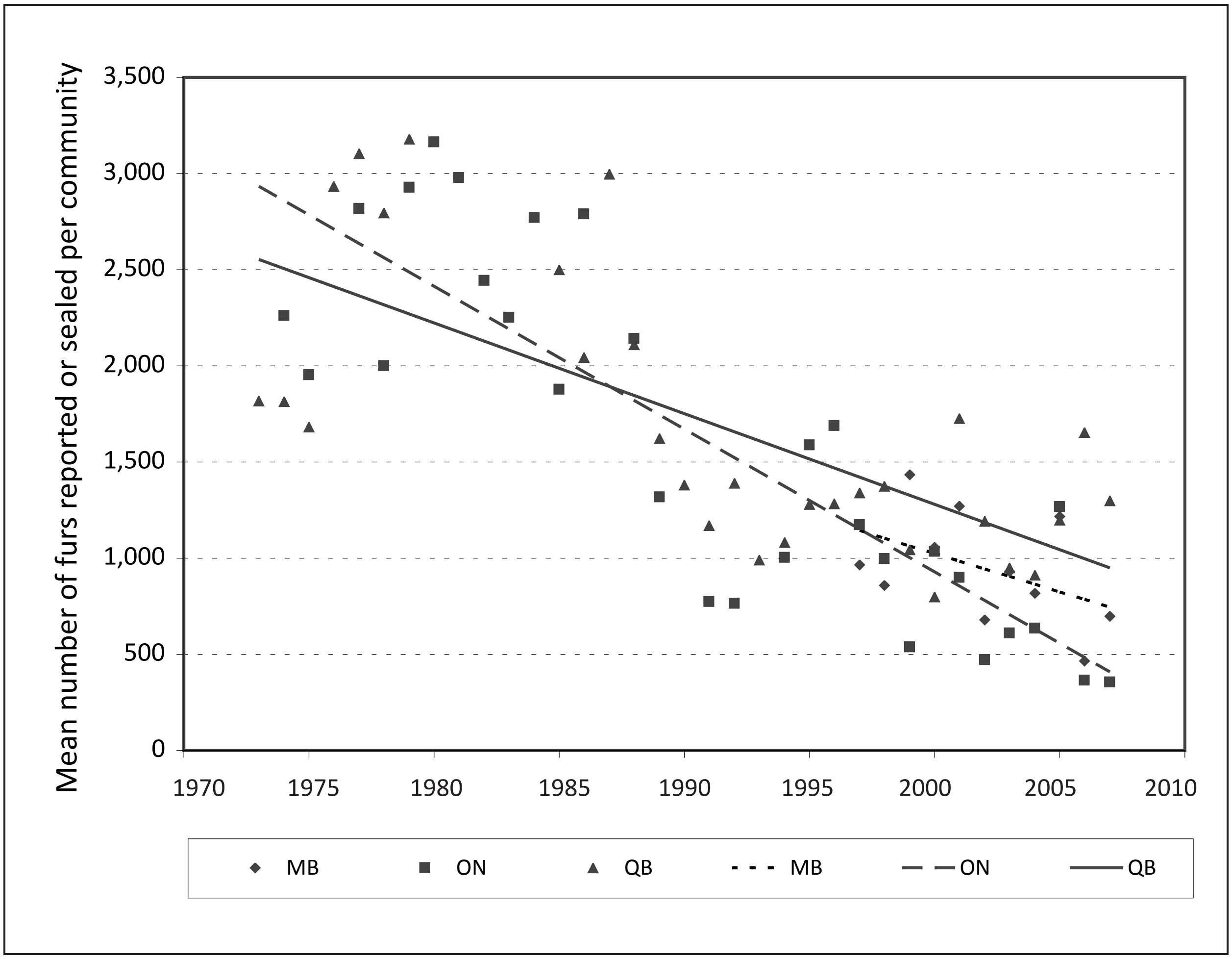

Figure 129. Harvest trends of furbearers, as measured by mean number reported or sealed per community for Manitoba (MB) (1996-1997 to 2006-2007), Ontario (ON) (1973-1974 to 2006-2007), and Québec (QC) (1983-1984 to 2006-2007) portions of the Hudson Plains Ecozone. For Manitoba, the average is based on mandatory fur dealer reports for traplines entirely or partially in the Hudson Plains Ecozone, i.e., those in the Churchill, Limestone, Shamattawa, Gods Lake, and Split Lake Registered Trapline sections. For Ontario, the number of communities participating varied from year to year; therefore, the average is based on seven main communities in the ecozone (Moose Factory, Moosonee, Fort Albany, Kashechewan, Attawapiskat, Peawanuck, and Fort Severn). For Québec, the average is based on sealing records from Eastmain and Waskaganish.
Sources: Data are from Manitoba Conservation; Ontario Ministry of Natural Resources; and Système Fourrure Ministère des Ressources Naturelles et de la Faune.

strongly related to decreasing furbearer populations, i.e., the capacity of the ecozone to continue supplying furs as an ecosystem good is not likely being compromised. Rather, declines in trapping effort (and resultant harvest) often coincide with changes in market conditions (OMNR 1985), including the fall in commercial value of pelts, since the late 1980s. Trapping effort has also depended on local economic conditions (e.g., job offers in other areas such as construction) and the desire of Aboriginal trappers to maintain a traditional trapping lifestyle (OMNR 1985).

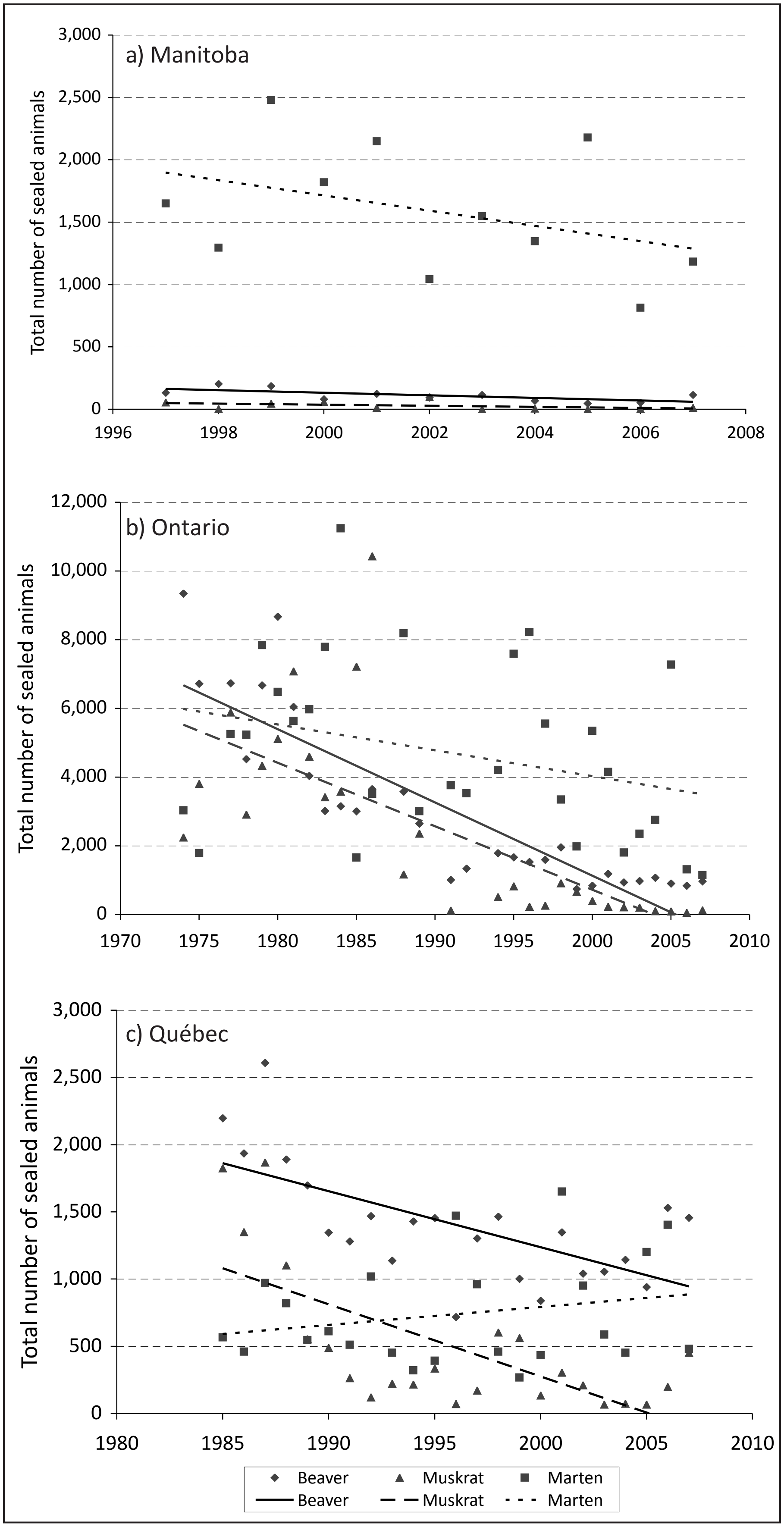

Figure 130. Total number of reported or sealed animals by province within the Hudson Plains Ecozone for the three most abundantly trapped furbearing species (beaver, muskrat, and marten): a) Manitoba (1996-1997 to 2006-2007); b) Ontario (1973-1974 to 2006-2007, excluding 1975-1976, 1986-1987, 1989-1990, and 1992-1993); and c) Québec (1984-1985 to 2006-2007). For Manitoba, the average is based on mandatory fur dealer reports for traplines entirely or partially in the Hudson Plains Ecozone, i.e., those in the Churchill, Limestone, Shamattawa, Gods Lake, and Split Lake Registered Trapline sections. For Ontario, the number of communities participating varied from year to year; therefore, the average is based on seven main communities in the ecozone (Moose Factory, Moosonee, Fort Albany, Kashechewan, Attawapiskat, Peawanuck, and Fort Severn). For Québec, the average is based on sealing records from Eastmain and Waskaganish. Note the differences in x-axis and y-axis scales.
Sources: Data are from Manitoba Conservation; Ontario Ministry of Natural Resources; and Système Fourrure Ministère des Ressources Naturelles et de la Faune.

2.5.3.3 **Other provisioning goods**

The above-described provisioning services (i.e., furs, as well as food in the form of meat from cervids and waterfowl) are perhaps the most economically and culturally important goods provided to the people of the Hudson Plains Ecozone. However, the ecozone provides numerous other goods to both resident Aboriginal peoples and, to a lesser extent, other peoples (Table 30). Of these additional goods, fish (Section 2.3.3.4) is notably important as, for example: 58% of the resident Aboriginal peoples in the Ontario portion of the ecozone participate in fishing; ~15% of the potential edible weight of wild food harvested by Aboriginal peoples in the Ontario and Québec portions of the ecozone consists of fish (including lake whitefish, *Coregonus clupeaformis*; sucker, *Catostomus* species; brook trout, *Salvelinus fontinalis*; walleye, *Sander vitreus*; northern pike, *Esox lucius*; lake sturgeon, *Acipenser fulvescens*; lake trout, *Salvelinus namaycush*; and cisco, *Coregonus artedi*); and ~20% of harvesters' time is spent fishing (NHR 1982; Berkes et al. 1994).

Table 30. Some provisioning services (goods) other than cervids and waterfowl provided by the Hudson Plains Ecozone and a selective list of their uses.
Sources: OMNR (1985); Marshall et al. (1989); and Berkes et al. (1994).

Provisioning service	Select use(s)
Fish	Food source
Small game (e.g., grouse, ptarmigan, hare)	Food source, hides for hand crafts
Marine mammals (e.g., polar bear, seal)	Food source, hides for hand crafts
Water	Drinking and cleansing source and medicinal uses
Animal by-products (e.g., fat, feathers, bones, skin/hides)	Clothing, medicinal products, tinctures, bedding products, building and crafting materials, insecticides
Wood	Fuel wood, building and crafting materials
Peat moss	Insulating material, soil additive
Other plants and plant parts (e.g., leaves, berries, resin, roots)	Food source, medicinal products, beverages and tinctures, crafting materials, colour dyes

Overall, there is no compelling evidence that the capacity of the Hudson Plains Ecozone to provide provisioning services (goods) to people has changed, based on limited information available for a select set of services examined. Current, standardized, and complete data on provisioning services is lacking for the ecozone as a whole. It is also difficult to detect trends in individual provisioning services from the data that are available, owing to the influence of extraneous factors. Existing trends in individual goods (as reported here) might not reflect changes in the capacity of the ecozone to provide/support the good so much as changes in the market demand of a resource (e.g., decrease in market demand for furs) or a general shift of Aboriginal peoples to a more modern lifestyle, which might increase (e.g., by more efficient hunting) or decrease (e.g., by lost skills) the use of some goods. Still, the capacity of the Hudson Plains Ecozone to provide provisioning services is considered for the most part stable, given the relative intactness of the ecozone and relatively low levels of development to date (Section 2.2.1.2, *Land Cover Change*).

Provisioning services continue to contribute a strong, yet monetarily undervalued, part of the economy of these northern Aboriginal communities and, likewise, a crucial cultural component to the traditional Aboriginal subsistence way of life. Recent estimates of the replacement value of the ecozone's provisioning services are not available, but a 1990 estimate for the Ontario portion of the ecozone was $7.8 million (or ~$7,000 per household) for bush food or $9.4 million (~$8,400 per household) including other products of the land (e.g., fur, fuelwood) – which comprised up to one-third of the total cash economy for the region (Berkes et al. 1994). Moreover, the hunt forms a core tradition for Aboriginal peoples on which morals, ethics, and values such as the importance of sharing and reciprocity are taught to the young (Berkes et al. 1994; see also Section 2.5.4, *Cultural Services*). Future land use and resource management plans in the ecozone must not compromise the capacity of the land to provide provisioning services (goods) and, therefore, the ability of future generations of Aboriginal peoples to retain their cultural identity, values, and social relationships to each other and the land.

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2.5.4 **Cultural services**

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Cultural services are non-material benefits that people obtain from ecosystems through spiritual enrichment, reflection, recreation, and aesthetic experience (MEA 2003). The Hudson Plains Ecozone has a majority of indigenous people: Lowland Cree (Mushkego) people on the Manitoba and Ontario sides, Eastern Cree people on the Québec side, and small numbers of Métis people throughout the area. Historically, this is the area where the fur trade started, and the first fur trade posts in Canada were established. Thus, the region has had European influence, since the early 1700s (Section 1.2, *Human History*). Indigenous language retention (predominantly Cree) is very high in most communities (AANDC 2010; Statistics Canada 2010). Churchill in Manitoba and Moosonee in Ontario are the only communities with non-native governance structures (Statistics Canada 2010). English is used comparatively more in both of these towns. English is also a significant linguistic component in the Ontario communities of Peawanuck (Weenusk) (AANDC 2010; Statistics Canada 2010) and Moose Factory, and it is being used more by younger families more generally (e.g., Ohmagari and Berkes 1997).

2.5.4.1 **Social & economic life**

Prior to World War II, many groups in the region led a relatively self-sufficient way of life based on extended families or kinship groups (George et al. 1995; Lytwyn 2002). Livelihoods were based mostly on hunting, trapping, fishing, gathering, and the trading of products from these pursuits. These activities provided a strong connection to the environment, which was important for survival and for maintaining social relationships and cultural identity.

After World War II, indigenous societies were impacted in a variety of ways. Mandatory education was introduced, and permanent villages were established, which made it difficult to pursue the seasonal round of activities on the land (Ohmagari and Berkes 1997). Transportation and communications changed at an increased pace. Motor boats, snowmobiles, and all terrain vehicles (ATVs) became common, reducing traveling time for hunting and fishing. Television became available in the 1970s, providing exposure to new lifestyles and role models. These trends are similar to those in most arctic and subarctic communities around the world (Csonka and Schweitzer 2004), and they continue today as, for example, the access of these communities to internet and other broadband technological services continues to improve (e.g., Government of Ontario 2010).

Given such pervasive trends in social and economic life, declines in the use of certain cultural services (identified below) are not necessarily related to changes in the health or capacity of the Hudson Plains Ecozone to supply people with cultural services. Rather, declines in Aboriginal use of these cultural services tend to be related more to changing values, declining bush skills, and less time spent in the bush, all of which are associated with the shift from traditional to modern lifestyles (e.g., George et al. 1995; Ohmagari and Berkes 1997). Indeed, little evidence exists to suggest a reduced capacity of the ecozone to continue supporting the traditional way of life in recent times (this report)[82]. Still, decreasing socio-cultural diversity among peoples is in itself a concern world-wide, and it often leads to a loss in biodiversity (given that biodiversity tends to be central to indigenous belief systems, worldviews, and identity) (SCBD 2010).

2.5.4.2 **Traditional knowledge, social relations & spiritual and religious values**

Using the environment and resources is important not only for obtaining food and maintaining Aboriginal culture but also as a way of building ecological knowledge and relationships (Ohmagari and Berkes 1997; Berkes 2008; Loutitt 2009). Indigenous knowledge includes empirical information, such as animal distributions. It is dynamic; some kinds of knowledge are lost, but others are added. The worldview of the Aboriginal peoples of the area is influenced by traditional religion (Berkes 2008). Although these people are mostly Anglican and Catholic, the adoption of Christianity did not result in the complete replacement of one religion with another. Rather, elements of old and new beliefs are mixed together, and the resulting system still retains many traditional values (Preston 2002).

These traditional Aboriginal values and knowledge, which have been declining in importance over time (e.g., George et al. 1995), are characterized by animism, belief in spiritual beings (Berkes 2008; Loutitt 2009). Animals and plants, in addition to humans, share a spirit that energizes them, and they are capable of independent action. For example, the Cree believe that animals gave themselves willingly to hunters and fishers, but for their part humans needed to maintain a relationship of respect and to observe certain rituals to take care of the spirits of the animals (Berkes 2008; Loutitt 2009).

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[82] Historically, the fur trade era led to some changes in the abundance and distribution of furbearers and other provisioning species (Lytwyn 2002). Recovery of American beaver (*Castor canadensis*) and American marten (*Martes americana*) was later aided by restocking, and reclamation of the historical range of some species, such as wolverine (*Gulo gulo*), continues today (Section 2.3.3.2.3, *Furbearers*).

The decline in traditional Aboriginal values and knowledge is associated with reduced transmission and mastery of certain traditional bush skills (e.g., Ohmagari and Berkes 1997). A decline in the number of Cree persons that speak their indigenous language (see earlier) may also be indicative of a decline in traditional knowledge (SCBD 2010).

2.5.4.3 **Cultural heritage values & sense of place**

Not only animals but certain areas of the land also receive respect in Aboriginal culture, because traditional beliefs and practices are often tied to the land (Loutitt 2009). Graves of ancestors, historically important locations, and parts of the land, where certain ceremonies take place, can be considered sacred. Sacred sites are found throughout the region, but they have not been systematically inventoried. The Aboriginal peoples of the area have a strong attachment to land; the traditional areas of family groups provide a *sense of place* that is encoded in place names.

There is cultural continuity through land use (George et al. 1995). A proxy for land use is the number of participants in the Income Security Program (ISP) under the James Bay and Northern Québec Agreement of 1975 (Governments of Canada and Québec 1976). Families who spend more than four months of the year on the land come under the ISP. According to the data collected, since the late 1970s, the number of families on ISP has been either steady or has declined by up to one-third in the various communities. Because of rapid population growth, however, the percentage of the families under ISP has declined sharply (see Figure 131, for the ecozone communities of Eastmain and Waskaganish). Similarly, studies in the Ontario portion of the ecozone show that the activity patterns of Cree harvesters have shifted from traditional long trips to numerous short trips of a few days in duration (Berkes et al. 1995). The fact that the majority of families in the area no longer spend a substantial part of the year on the land is consistent with the observation of many elders that the younger generations are not as well connected to the land, as in the past.

Still, traditional land use and associated pursuits have not disappeared, despite the influences of modernization. A substantial number of Aboriginal peoples in the Hudson Plains Ecozone still participate in traditional hunting and gathering, even if generally not for long periods at a time (Berkes et al. 1994, 1995; George et al. 1995). Transmission of many (not all) bush skills among women is also still relatively strong (Ohmagari and Berkes 1997). Significantly retained cultural aspects and key indicators of the traditional sector include: spatial extent of land use, wildlife harvesting effort, participation rates, magnitude of the harvest, frequency of bush food consumption, and degree of sharing (Berkes et al. 1994, 1995; George et al. 1995). Thus, although the overall trend in this ecozone has been towards a mixed economy (i.e., subsistence lifestyle; government transfer payments, grants, and programs; and wage employment), such a mixed economy does not appear to be a transition stage to an economy based on wage labour – but rather an arrangement that may persist in a culturally and environmentally sustainable manner (Berkes et al. 1994; George et al. 1995).

2.5.4.4 **Aesthetic values, recreation & ecotourism**

The land is also a source of aesthetic values, recreation, and ecotourism for non-Aboriginal peoples. The Hudson Plains Ecozone has a high capacity for providing these types of cultural services (Government of Québec 2009; Far North Science Advisory Panel 2010), even if such

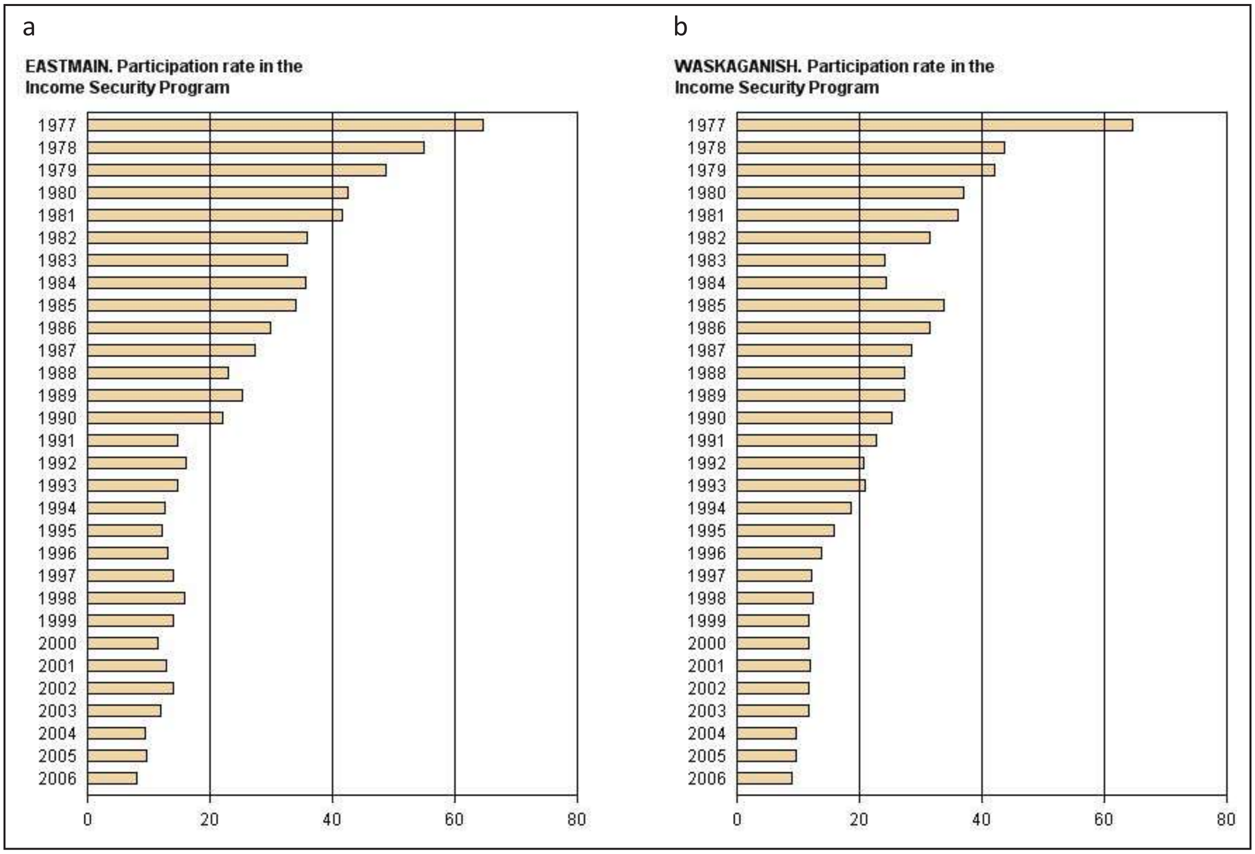

Figure 131. Percentage of the Cree members' population participating in the Cree Hunter and Trappers Income Security Program in a) Eastmain and b) Waskaganish, Québec.
Source: Data are from the Cree Hunter and Trappers Income Security Program, DIALOG-Research and Knowledge Network Relating to Aboriginal Peoples, used with permission.

activities are fairly limited there at present. The major tourist attractions are polar bear-viewing at Churchill and the Wapusk National Park in Manitoba (Berkes et al. 2005), while Moosonee and Moose Factory in Ontario benefit economically from a smaller tourism component associated with the Polar Bear Express passenger train (Abraham and Keddy 2005). Aside from Wapusk National Park, a number of other federal, provincial, and territorial protected areas and wildlife reserves are established throughout the ecozone (Section 2.6.2.1, *Protected Areas*), even if associated visitor facilities are few. Big game and goose hunting camps for non-Aboriginal hunters have been an important source of cash income for Aboriginal peoples. The trend is toward increasing ecotourism and recreational operations in areas, such as Churchill (Berkes et al. 2005; NDMF 2008)[83].

2.5.4.5 **Major trends & outlook**

People of the area are increasingly tied to national policies and to the global economy. Rapid economic, political, and demographic changes; transportation developments; hydroelectric, mining, and other resource developments; national defence installations; and animal rights

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[83] For example, the polar bear tourism industry near Churchill has grown from a few viewing-vehicles in the 1960s and 1970s to 18 vehicles that now collectively annually accommodate an estimated 8,000 visitors over 40-60 days in October and November (see Dawson et al. 2010).

campaigns (that resulted in the collapse of the fur trade in the 1980s) – have all produced shocks and stresses (e.g., George et al. 1995; McDonald et al. 1996; Rosenberg et al. 1997; Whitelaw et al. 2009). Ocean pollution in Hudson and James bays, mainly from long-range atmospheric transport of pollutants, is contaminating wild food sources (e.g., Section 2.3.3.1.1, *Polar Bear*). Climate change is affecting seasonal animal distributions (e.g., Section 2.4.3.4.1, *Animal Phenology*) and interfering with the ability of hunters to predict the weather and animals, with adverse affects on safety, hunting, and fishing (e.g., McDonald et al. 1996; Ford et al. 2008; Hori 2010). Against this backdrop, the people of the area have been adapting to change by pursuing land and resource rights and political autonomy through regional government powers (Adelson 2000; Scott 2001) and co-management agreements (Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*). They are among the world leaders in the struggle for indigenous rights, co-management, and recognition of indigenous knowledge.

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2.6 Human influences

This section directly addresses two basic types of human influences: those that create threats and stressors, which lead to individual and cumulative impacts; and more positive influences that are associated with stewardship and restoration activities.

2.6.1 Stressors & cumulative impacts

2.6.1.1 Summary of stressors

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Globally and nationally, the principal threats associated with continued biodiversity loss are habitat disturbance (loss or alteration), pollution (especially nitrogen), invasive alien species, overexploitation, and climate change (e.g., Federal, Provincial and Territorial Governments of Canada 2010; Hoffmann et al. 2010; Secretariat of the Convention on Biological Diversity 2010) – all of which are either generally constant or intensifying (Butchart et al. 2010; Secretariat of the Convention on Biological Diversity 2010). Economic and human population growth are important driving forces (Sukhdev et al. 2010).

Anthropogenic threats[84] and stressors of the relatively remote, sparsely populated, and little developed Hudson Plains Ecozone are summarized in Table 31, along with their associated sources and principal impacts. These threats, stressors, sources, and principal impacts are generally discussed where most relevant throughout this report. Overall, although stressors of the ecozone can be identified within each of ESTR's five basic anthropogenic threat classes, their impacts are mostly limited and localized in scale (Table 31). Presently, many stressors typical of more human-populated areas are comparatively unimportant in this ecozone, including nutrient loading, pesticides, acid precipitation, air pollution, light pollution, and invasive species. Moreover, some of the most important stressors and impacts in this ecozone are ultimately largely attributable to human influences outside of the ecozone. Specifically, severe damage to the ecozone's coastal salt marshes and tundra freshwater marshes from excessive goose foraging is principally related to land use changes outside of the ecozone, which have caused the Mid-Continent population of migratory lesser snow goose (*Chen caerulescens caerulescens*) to greatly increase over the past four decades (sections 2.2.2.1, *Coastal*; 2.2.2.2, *Polar-Tundra;* and 2.3.3.3.2, *Waterfowl*). Populations of other migratory bird species are also affected by anthropogenic factors outside of the ecozone, including habitat loss along their migration routes and in their wintering areas further south (sections 2.3.2, *Trends in Species of National Conservation Concern* and 2.3.3.3, *Birds*). Stressors associated with accelerated climate change, including increases in air temperature and decreases in the seasonal duration of sea ice in Hudson and James bays, are also mainly attributable to human influences outside of the ecozone, given that this ecozone currently has a very small human population (Section 1.2.1, *Settlement History*), few emitting industrial developments and roads (vehicles) (Section 1.2.2, *Economic History*), and a massive expanse of peatland that is considered a large and globally important carbon store (sections 2.4.4, *Carbon Cycling* and 2.5.2, *Regulating Services*). Furthermore, some stressors associated with hydroelectric development can be at least partly attributed to the hydrological connectivity this ecozone has with developments upstream, outside of ecozone boundaries (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). Likewise, although presently limited, many contaminants of the ecozone's ecosystems and biota arrive in the ecozone through long-distance air and water transport from industrial sources in more populated areas (Macdonald et al. 2000; e.g., for polar bears, see Section 2.3.3.1.1, *Polar Bear*). The most important stressors originating directly within the ecozone are those associated with hydroelectric and mining developments, though even these activities are presently limited, and they are driven largely by human demands from outside of the ecozone. Commercial forestry, peat harvesting, and agriculture are effectively absent, and subsoil asset extraction is limited to a single active mine.

As noted throughout this report, pressure for additional development of the ecozone is, however, mounting in mining, hydroelectricity, transportation, and, to a lesser extent, wind-farming sectors (Manitoba Geological Survey 2003; OPA 2007; Environment Canada 2008; Government of Ontario 2009; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates 2010; Government of Nunavut and Government of Manitoba 2010; Hydro-Québec 2010; Manitoba Hydro 2010; MDDEP 2010; Micon International 2010; OMNR 2010; SNC Lavalin 2010; see also Hélimax Énergie and AWS Truewind LLC 2005). At the same time, climate change is manifesting and, over time, it is expected to lead to major changes

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[84] In ESTR, a threat is defined as a force with an actual or potential negative impact on biodiversity (Wong et al. in press). Only anthropogenic threats are considered in this section on human influences, but the ESTR threat classification also recognizes a category of natural threats, which result from stochastic events or factors. In this report, extreme weather and other natural disturbance events are discussed in Section 2.4.2.

in the ecozone's defining climatic and edaphic conditions (Section 2.1, *Abiotic Drivers*) and therefore also its biodiversity. Comprehensive and integrated land use and conservation planning across the ecozone, including consideration of little-studied interactions among stressors and cumulative impacts (discussed in the next section), will be important for guiding successive development projects and maintaining the ecological integrity and resilience of this ecozone (Far North Science Advisory Panel 2010; see also Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*).

Table 31. Summary of stressors of the Hudson Plains Ecozone, along with their main sources and impacts, organized by five ecological threat classes. Threat and stressor classification and terminology follow ESTR convention, which is based largely on listings of the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and the International Union for Conservation of Nature (IUCN) classification of threats (Wong in press). Abbreviation: N/A, not applicable.
Sources: Summary is derived principally from this report; sources are as indicated for any information derived from elsewhere.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Threat class #1: Ecosystem change (loss & alteration)		
Substrate alteration (e.g., impervious soils) Alteration of vegetation cover, including long-term loss	Intense goose foraging, mostly by the Mid-Continent population of lesser snow goose, which increased greatly in size over the last four decades, principally due to land use changes (increased supply of agricultural food and development of refugia) south of the ecozone, on its wintering grounds and along migration routes; Canada goose (*Branta canadensis*) contributes to a lesser extent.	Intense goose foraging has resulted in an apparent trophic cascade, with an associated loss (severe damage), since the 1970s, of ~30% of coastal salt marsh habitat from Manitoba to James bay (including Akimiski Island, Nunavut). Loss of *Pucinellia-Carex* communities and exposure and erosion of sediment have resulted in an alternate state of bare, hypersaline sediment, from which recovery could take decades. Major changes to the coastal salt marsh food web are evident. Spider and carabid beetle populations and chironomid species richness are reduced in affected areas. Some bird species' colonies or populations have also deteriorated locally (e.g., Canada goose and some tundra-nesting passerines). Ecosystem functions/processes and/or services such as soil retention, nutrient cycling, and net above-ground primary productivity are impacted, in addition to vegetative succession and herbivore-plant interactions.
	Operation of wheeled vehicles (tundra buggies/ATVs) in both Manitoba and Ontario portions of the ecozone.	Direct damage to tundra plant communities and species, both in the wetter inter-ridge areas and on the drier beach ridges, in both Manitoba and Ontario portions of the ecozone. Where wheels churn the peat surface, a change in albedo leads to a deepening of the active layer; subsidence through surface compression then causes pool formation and accelerated thermokarsting. On the ridges, one or more passes of wheeled vehicles can leave deep imprints on plant species such as white mountain-avens (*Dryas integrifolia*), loosen the plants from their substrate, and expose their roots to desiccation. Desiccation damage can be severe if summer temperatures are high or winter snow cover is thin.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Complete ecosystem conversion – generally permanent land cover changes (outside of human settlements)	Reservoir creation in the ecozone and/or river diversions (reduced flows) from hydroelectric developments in and around the ecozone.	In Québec, purposive flooding (via diverted rivers flows) converted an area with wetlands and forests (terrestrial habitat) into a reservoir (Opinaca) in 1980. *Note: The reservoir along the lower Churchill River in Manitoba is a restoration effort that permanently rewatered ~8 km^2 of river bed that had been dewatered during partial river diversion (see Section 2.6.3.1,* Restoration of the Lower Churchill River*).* As well, partial diversion of the flows of the lower Churchill, Eastmain, and Opinaca rivers (and likely also the recently diverted Rupert River) caused their river beds and/or banks to dry and vegetate, coincident with reductions in the amount and quality of fish habitat in these rivers.
	Development and operation of a single mine within the ecozone.	A relatively small area of the ecozone (which was principally bogs and fens, but also ponds, creeks, smaller rivers, and riverbank and creek margin forests) is now occupied by the direct, project-related developments (~28.8 km^2) of an open-pit diamond mine near Attawapiskat (the potential area affected by the mine is, however, larger than the mine itself due to dewatering operations, as noted below). Although most of the disturbed area will be rehabilitated with native plant species (albeit not all to original ecological classifications), certain areas will not be restored after mine closure. The open pit will be actively flooded to create a small lake with closed drainage, and the new channel created after diversion of Granny Creek will become the permanent creek channel. The mine was established in 2006, opened in 2008, and has an expected lifetime of ~12 years.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Alteration of hydrology	Alteration of river hydrology (e.g., flow rates, magnitude and timing of peak flows) and terrestrial (mostly peatland) hydrology (flooding, drying) by hydroelectric developments and operations in and around the ecozone (trends for reduced river flows are also evident for undeveloped rivers, but such trends are correlated with large-scale climate oscillations).	Overall, river channel fragmentation and/or flow regulation from hydroelectric structures and operations in and around the ecozone have strongly affected the Churchill and Nelson River systems in Manitoba, the Moose River system in Ontario, and the Eastmain and Rupert River systems in Québec. The Albany River system in Ontario and Nottaway River system in Québec are considered moderately affected. Notably, flows from the Churchill, Eastmain, Opinaca, and Rupert rivers have been reduced ~40%, 90%, 87%, and 72%, respectively (albeit lateral flow from tributaries increases the flow of the Rupert River at its mouth to ~48%). Ecological effects are associated with these altered hydrologic regimes in some rivers (e.g., deteriorations in migratory lake sturgeon populations, shifts in fish communities, declines in benthic macroinvertebrates, and reduction of quality fish habitat) and associated estuaries (e.g., sediment loading, saltwater intrusion, shifts in fish communities) and near-shore environments (e.g., steep decline in subtidal eelgrass beds along the eastern James Bay coast, which also suggests a redistribution of Atlantic brant in the ecozone). As already noted, some terrestrial effects are also linked to changes in river flow regimes, due to either drying or flooding (e.g., loss or shifts in vegetative classes, such as from pioneer wetland species to shrubby species, following desiccation downstream). Cumulative impacts from existing and proposed or future (multiple) hydroelectric developments in the Hudson Bay watershed is an ongoing concern.
	Alteration of river and peatland hydrology, due to the dewatering operations of a single active mine within the ecozone.	Dewatering operations at the ecozone's Victor open-pit diamond mine are increasing annual recharge, possibly over an area as large as ~500 km^2, potentially altering water balance over the life of the mine. Dewatering affects bogs, fens, ponds, creeks, smaller rivers, and riverbank and creek margin forests. Groundwater withdrawals are being made from the Nayshkootayaow River for purposes of dewatering (this water is then discharged to the Attawapiskat River), albeit water is being added to maintain seasonal low flows in the Nayshkootayaow River. No published monitoring data (reported impacts) are available for this relatively new mine near Attawapiskat. As noted above, the mine was established in 2006, opened in 2008, and has an expected lifetime of 12 years. A reclamation plan is in place.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Fragmentation of the landscape (see *Alteration of Hydrology* for river fragmentation effects associated with hydroelectric developments)	Winter roads, all-season roads, and hydroelectricity transmission corridors related to resource (hydroelectric and mining) and community developments, all within the ecozone.	Fragmentation of the ecozone's landscape is very limited at present, being comprised of: two railway lines (MB, ON) and one all-season road (QC), which provide access into the ecozone from the south and terminate near the coast; winter roads that service projects (e.g., Victor mine) and seasonally connect coastal communities; and power transmission lines, including a major new transmission line that services the Victor mine. Linear fragmentation can potentially restrict wildlife movements, increase human and predator access to fish and wildlife (harvest and incidental kill), disconnect protected areas, and, over time, contribute to cumulative impacts. Maintaining large tracts of unfragmented and/or unroaded landscape is particularly important for sensitive species, such as woodland caribou (*Rangifer tarandus caribou*) and wolverine (*Gulo gulo*).
Threat class #2: Pollution		
Contamination of ecosystems and biota from long-distance transported contaminants	Environmentally persistent organic pollutants (POPs) and metals, such as mercury, which arrive in the ecozone via long-distance air and water transport from industrial sources in more human-populated areas[a] (stressor widespread).	Long-range transport contributes to chemical contaminants being bioaccumulated by polar bears, seals, and probably other wildlife (monitoring is very limited), with potential impacts on the biota (e.g., impaired endocrine and immune function and reproductive effects in polar bear) and the quality of provisioning goods for Aboriginal peoples. Perhaps the best contaminant trend information is for polar bears. Overall trends in persistent organic pollutants (POPs) in polar bears are variable, with some legacy contaminants declining (including the pesticide DDT, dichlorodiphenyltrichloroethane, and its metabolites) and some newer (emerging) contaminants increasing (including brominated flame retardants and perfluoroalkyl contaminants). In contrast, levels of many metals/elemental contaminants in polar bears, including mercury, have not changed, since the 1980s, and they are below levels associated with toxic effects. Mercury monitoring in fish has mostly been limited to areas affected by hydroelectric development (confounded), where inundation of land (organic matter) has accelerated the methylation of mercury to its more bioavailable form and led to elevated mercury levels in fish (see later).

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Contamination of ecosystems, habitats, and species from industrial by-products and by-products from renewable and non-renewable resource extraction activities	Former Mid-Canada Line radar (Doppler detection) sites and other former military sites within the ecozone are point sources of petroleum hydrocarbons, asbestos, heavy metals, pesticides, polychlorinated biphenyls (PCBs), and abandoned military waste and fuel barrels.	Ecosystems and biota (e.g., hares, leeches) are contaminated by industrial by-products, including PCBs, which might negatively affect the biota, as well as the traditional Aboriginal (subsistence) way of life – the latter through contamination of country foods. Impacts are localized around contaminated sites and associated waterways. Many such sites are currently the focus of restoration activities (e.g., Mid-Canada Line sites in Ontario and the Churchill Rocket Research Range and Wapusk National Park in Manitoba).
	A single active point-source major industrial facility, the Victor mine, is associated with some contaminant release via mine emissions, leachate, etc. (although acid generation is generally not expected to be an issue); also potential for promotion of methylmercury release, where mining activities disrupt hydrology (DeBeers Canada 2005).	Potentially some contamination of ecosystems from mining operations, but any impacts on biota may be relatively limited in scale. Published impacts-monitoring data are not yet available.
	Hydroelectric developments inside and outside of the ecozone. Although such developments do not contribute a new source of mercury, hydrological disturbances in the ecozone, including inundating new reservoirs over organic soils, can accelerate the bacterial conversion of inorganic mercury to methylmercury, its more bioavailable form.	The inundation of land during creation of the Opinaca reservoir in 1980 led to the accelerated release of methylmercury and its subsequent bioaccumulation in fish. Mercury levels in fish from this reservoir gradually declined over time, but they are forecast to increase again due to receipt of mercury exported from a recently impounded reservoir upstream, outside of the ecozone. Elevated mercury levels in fish can also reduce the quality of provisioning goods available to resident Aboriginal peoples, where levels exceed safe human consumption recommendations from public health institutions. Mercury levels are already relatively high in natural aquatic environments in the ecozone, where organic content is high.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Changes in sediment loads	Hydroelectric developments, which have substantially reduced river flows in the ecozone via flow diversion.	Diversion of 90% of the flow of the Eastmain River (and 87% of the flow of the confluencing Opinaca River) north to the La Grande River resulted in the downstream current of the reduced flow segment of the Eastmain River no longer expelling fine sediments into James Bay, such that its estuary became a sedimentary deposit zone; additional sediment was contributed by erosion of the exposed river bed. *Note: See also the* Accelerated Climate Change *category below, re: casual observations of additional sediment load in some rivers that drain into Hudson Bay (e.g., Hayes and Nelson rivers) being contributed by slumping and collapse of river banks, likely associated with permafrost thaw.*
Changes in nutrient loads (e.g., nitrogen & phosphorous additions)	Atmospheric transport and transport through inland waters (hydrological connections) from distance sources; also anthropogenically influenced disturbances within the ecozone. Direct agriculture and associated fertilizer and livestock waste inputs are effectively non-existent in the ecozone, as are forest harvesting, peat harvesting, and biomass burning activities. There have been no known algal blooms or related concerns from increased nutrient loads; trace ammonia from the pit sump at the Victor mine is being discharged to a linear fen system (after a settling pond), prior to release by natural drainage into the Nayshkootayaow River (DeBeers Canada 2005).	Most nutrient inputs come from long-distance transported sources (atmospheric deposition, transport through inland waters). Although such inputs are relatively minor compared to more developed areas further south, they have increased somewhat over time (e.g., nitrogen), although most of the accelerated nitrogen input via inland waters is in the southern portion of the ecozone around James Bay. Anthropogenic disturbances directly in the ecozone sometimes also increase nutrient levels. Phosphorus increased following impoundment of the Opinaca reservoir, as the large area of flooded plants decomposed. Nutrient losses are also evident from some anthropogenically influenced disturbances. Specifically, the catastrophic herbivory-linked disturbance of coastal salt marshes (see earlier, under the *Ecosystem Change* category) has led to loss of soil nitrogen and other nutrients, including retardation of cyanobacterial nitrogen fixation.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Contamination from pesticides	Relatively small inputs of pesticides from long-distance transport (point-sources of herbicides and insecticides from agriculture, horticulture, and forestry are negligible within the ecozone).	Long-distance transport is responsible for the contamination of polar bears with compounds such as the legacy herbicide DDT and its metabolites (see above, *Contamination of Ecosystems and Biota From Long-Distance Transported Contaminants*). No concerns with point sources originating in the ecozone.
Acid precipitation	Limited deposition from atmospheric transport of SO_2 and NO_x; minor local inputs of SO_2 and NO_x from the Victor mine (DeBeers Canada 2005).	No concerns at present. Atmospheric acid deposition, which could increase methylation of mercury to its more bioavailable form, is low compared to other, more southerly and developed regions in Canada, including the adjacent Boreal Shield Ecozone, where acidification appears to have reduced calcium levels in some lakes sufficiently to threaten *Daphnia* (keystone herbivores). The Hudson Plains Ecozone does, however, have some acid-sensitive terrain.
Air pollution (e.g., ground-level ozone, particulate matter), outside of human population centres	Very small, direct releases of air pollutants from the Victor mine, which is the principal emissions-based activity in the ecozone; emissions are related to dust generation (particulate matter), incineration, and fuel (diesel) combustion from heavy equipment and power plant operations (DeBeers Canada 2005). Emissions from personal vehicles are negligible.	No concerns at present. Relative to more southerly and populated areas of Canada, the Hudson Plains Ecozone has few active point-source industrial facilities (see above for long-distance sources). The ecozone's only major industrial facility, the Victor mine, is reporting to the National Pollutant Release Inventory only small, direct on-site releases of hexachlorobenzene to the air (Environment Canada 2010). Modelling during project development suggested that emission of other air-borne pollutants (particulate matter, SO_2, NO_x, CO, CO_2, HCl, Cd, Pb, Hg, dioxins, furans) would meet air quality standards (DeBeers Canada 2005).
Light and thermal pollution	N/A	No concerns at present, due to the very small human population and villages and a paucity of industrial discharges into aquatic environments.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Threat class #3: Invasive species (i.e., alien species and/or species native to Canada that have been introduced outside of their natural range)		
Competition	Routes of entry for the ecozone's existing, relatively small complement of introduced and potentially invasive species have not been definitively established, but some specific entry routes of concern are: shipping water ballast; railway shipments of grain; land rehabilitation projects that don't use native species; hydrological connections with areas further south; and use of live fishing bait.	A relatively small number of species both non-native and native to Canada have been introduced into the ecozone from outside their normal range, but their impacts on the ecology of the ecozone are not well studied or monitored (some of these species are, however, known to be invasive elsewhere). Most introduced species are vascular plants (at least 98 species), generally found near villages and other areas with most human activity and probably only locally invasive. A few introduced species of smaller mammals, birds, and fish are also known to be present, with smallmouth bass (a species native to Canada that was introduced outside its normal range) most recently discovered (2008-2009). The potential spread of this warmwater predatory species as the climate warms is a concern for fish community composition and dynamics.
Predation		
Hybridization		
Introduced pathogens		
Ecosystem modification		

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Threat class #4: Exploitation, harassment, or direct mortality of native species		
Regulated or subsistence mortality (harvesting, hunting, fishing)	Historic exploitation (e.g., fur trade), Aboriginal subsistence and recreational (regulated) hunting and fishing, all within the ecozone.	Populations of furbearing mammals, such as American beaver (*Castor canadensis*) and American marten (*Martes americana*), are still recovering from over-harvest during the fur trade era (aided restocking in the early 1950s). Lake sturgeon is a species at risk in this geography due in part to historic over-harvest (hydroelectric development is a more recent factor). With few exceptions, harvests (and population sizes) of most species are not currently being monitored adequately or at all. Some concern does, however, exist that high harvest pressure might have contributed to the reduced abundance of lake sturgeon in the lower Eastmain and Opinaca rivers post-development (harvest was facilitated by new road access and an increased ease of net fishing associated with the reduced river flows) – as well as an apparent range shift and/or lower numbers of the Pen Islands herd of migratory woodland caribou (forest-tundra ecotype) in the northern part of the ecozone. Harvest is also a recognized anthropogenic stressor on polar bear, even though harvest is not currently the key factor affecting the population trends of bears that use the ecozone. Harvest will, however, be challenging to manage in the future, when these polar bear subpopulations are projected to decline in abundance (due to continued loss of sea ice habitat associated with climate change). High fishing pressure for brook trout was reported in the 1970s and 1980s in popular brook trout rivers (Sutton, Brant, Shagamu, and Gorge rivers), but a lack of follow-up monitoring renders the present situation unclear. Provisioning species in general might be subject to localized over-harvest in the vicinity of population centres and other high recreational use areas.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Purposeful or incidental harm (e.g., harassment, persecution)	Human-wildlife interactions associated with landfill sites, communities, tourism, and more general use of tundra buggies/ATVs, all within the ecozone (minimal interactions are reported via linear disturbances, such as roads and hydroelectric transmission corridors).	The noted sources (landfill sites, communities, tourism, and more general use of tundra buggies/ATV) are associated with increased potential for purposeful or incidental mortality or harm of species (mostly localized), including: aggregation and/or persecution of *nuisance* wildlife around landfill sites and communities (e.g., polar bear and grey wolf, *Canis lupus*); destruction of tundra plant communities and species by wheeled tundra buggies/ATVs (e.g., white mountain-avens, *Dryas integrifolia*); and harassment from tourism-based use of large, wheeled vehicles for wildlife viewing (e.g., polar bear). Although generically reported, both sources and impacts are largely not quantified, and trends are unknown. Viewing of polar bears from tundra vehicles (near Churchill) has been shown to increase vigilance behaviour in male bears, though the biological or physiological consequences of such changes in behaviour have not been quantified, and only a small proportion of the subpopulation occurs in the regulated bear-viewing area. The polar bear tourism industry has grown from a few vehicles in the 1960s and 1970s to 18 multi-passenger vehicles that now accommodate ~8,000 visitors over 40-60 days annually in October and November.
Accidental mortality (e.g., road kills, bycatch)	Hydroelectric structures and operations within the ecozone.	Road kill and bycatch (not generally reported) are likely very low, due to the paucity of roads and commercial fishing, respectively. Hydroelectric structures and operations contribute to some direct mortality of fish by entrainment or impingement.
Disease	A paucity of information exists for most wildlife diseases and parasites in the ecozone, as well as forest tree diseases; no information exists to suggest that significant trends are occurring in diseases and parasites, with the possible exception of renal coccidiosis and parasitic nematodes in lesser snow goose.	An increased frequency of density-dependent renal coccidiosis (caused by the protozoan parasite, *Eimeria truncata*) and an increase in the load of two parasitic nematodes (*Trichstrongylus tenuis* and *Heterakis* sp.) are evident in the greatly expanded Mid-Continent population of lesser snow goose, within coastal versus inland environments. The geese are, however, still largely escaping density-dependent effects of habitat loss at the local scale, as they move to forage at other coastal and inland sites.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Threat class #5: Accelerated climate change[b]		
Changes in temperature	Anthropogenic-induced climate change, mostly from increased concentrations of greenhouse gases originating outside of the ecozone (global).	Depending on location, stations in the Hudson Plains Ecozone with data sufficient to support the 1950-2007 climate trend analysis period of the ESTR show significant trends for increased mean annual and/or mean seasonal temperature (winter and/or summer) and increased number of effective growing degree-days. Increases in temperature in the broader Hudson Bay ecosystem are associated with significant changes in sea ice regime (see below) which, given continued warming, will strongly affect the ecozone's defining climatic and edaphic conditions, with implications for all biomes and protected areas. In the ecozone, warming is already associated with: deterioration in the polar bear subpopulations that use the ecozone; advancing animal phenology (earlier hatching dates and therefore nesting period of lesser snow goose and Canada goose, and earlier transition of polar bear from sea ice to land); and changing predator-prey interactions involving polar bear (e.g., with seals and geese). Other early impacts might also be present but not detectable given a paucity of monitoring. Increases in primary productivity appear much less than for some other areas of Canada, based on 1985-2006 trends in Normalized-Difference Vegetation Index, but some increase in productivity might also be suggested by observations of increased tree and shrub cover above the treeline, as near Churchill. Modelling suggests that earlier budburst of conifers is also likely occurring (advancing plant phenology). Climate warming has also been implicated in recent die-offs of fish in the lower Sutton and Albany rivers (anadromous brook trout, white sucker, lake whitefish) in years with unusual summer heat waves (lower precipitation/reduced river flows might also have contributed).

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Changes in precipitation	Possibly climate change (mostly from increased concentrations of greenhouse gases originating outside of the ecozone (global)), but trends in the ecozone for lower precipitation are at least partly related to the most recent positive phase of the Arctic Oscillation.	Depending on location, stations in the Hudson Plains Ecozone with precipitation data sufficient to support the 1950-2007 climate trend analysis period of the ESTR show significant trends for: decreased total spring precipitation; decreased number of winter or spring days with measurable precipitation; and decreased annual snow to precipitation ratio. Although 1950-2007 trends in period of snow cover and maximum annual snow depth are not statistically significant, a decline is suggested in both measures, when weighted by the last two decades. As well, earlier snow melt in the region is suggested by trends (1964-2000) for earlier mean annual date of peak discharge in spring of rivers discharging into Hudson, James, and Ungava bays. Trends in summer dryness are not apparent at the ecozone scale, based on limited analysis of moisture indices, albeit a significant trend towards decreasing summer dryness (reduced wildfire risk) is evident at the extreme southern end of the ecozone (only) for the longer period 1901-2002 but not for the more recent period, since 1951.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Changes in ice and permafrost regimes	Anthropogenic-induced climate change, mostly from sources of greenhouse gases outside of the ecozone (global).	Sea ice in the broader marine arctic is changing more rapidly than expected, ahead of modelled climate change projections. In the marine areas directly adjacent to the Hudson Plains Ecozone (i.e., western Hudson Bay, southern Hudson Bay, and James Bay), the period of sea ice cover has shortened by ~3 weeks on average, since the mid-1970s. These trends in sea ice are correlated with declines in the body condition, survival, and abundance of the polar bear subpopulations that use sea ice as habitat in winter (for hunting and feeding on seals) and the terrestrial environment of the Hudson Plains Ecozone during their off-ice season. The sea ice trends are also linked to dietary shifts in the bears, which are feeding less on ice-associated bearded seal (which eats invertebrates) and more on open-water harbour and/or harp seal (which eats fish) (cf. ice-associated ringed seal, the primary prey, still forming a relatively stable component of the polar bear's diet). This dietary shift, in turn, appears to be affecting the rates at which some persistent organic pollutants (POPs) change in the bears: some legacy contaminants appear to be declining slower than if the bear's diet had not changed, and some newer (emerging) contaminants are increasing faster. Declines in freshwater (river and lake) ice are suspected. Aboriginal peoples have noticed changes in the break-up and/or freeze-up of rivers, a reduction in ice thickness both in naturally flowing rivers and rivers with flows modified by hydroelectric development, and longer ice-free periods for some inland lakes. Such changes cannot currently be confirmed by western science, due to insufficient monitoring data. Permafrost degradation, which is known to be occurring just outside of western and eastern ecozone boundaries, is also suspected, but it likewise cannot be confirmed due to insufficient monitoring data. Collapse and erosion features and aggrading features are visible in the permafrost tension zone, and collapse features appear to have become more widespread over time, such as in the Ekwan to Lake River areas of the northern James Bay coast. River banks along the Hayes and Nelson rivers near York Factory have been and slumping and collapsing (adding to the sediment load into Hudson Bay). Partial degradation and conversion of frozen peat plateaus to fens is suggested in an area from the Nelson River north to Churchill, as is the enlargement of some associated lakes from eroding shorelines. The air temperature warming at Churchill could have resulted in permafrost warming of ~0.5 °C there, since the mid-1970s.

Table 31, Cont.

Stressor	Observations for the Hudson Plains Ecozone	
	Contributing source(s)	Main impact(s) in the ecozone
Alteration of hydrological cycle (e.g., low stream flow, thawing permafrost, more variable stream flow, lake level changes)	Large-scale climate oscillations and possibly also anthropogenic-induced climate change, mostly from sources of greenhouse gases outside of the ecozone (global).	A robust assessment of climate-related hydrological trends is not presently possible for this ecozone, due to insufficient monitoring data. However, declines are evident in the total annual volume of freshwater naturally discharged by the area's rivers (11% and 13% for eastern Hudson Bay and western Hudson Bay, respectively, 1964 to 2000 or 2003), which are associated also with a 4 day advance in annual peak discharge rate and a decline in peak intensity. The role of climate change is not certain (the recent warming is partly but not wholly attributable to systematic variations in climate oscillations), but these changes in river hydrology are correlated with large-scale climate oscillations and, in particular, the most recent positive phase of the Arctic Oscillation.
Increased extreme weather patterns outside the natural range of variation	Anthropogenic-induced climate change, mostly from sources of greenhouse gases outside of the ecozone (global).	Trends in extreme weather have not been directly examined, but rather indirectly using indicators or indices derived from daily temperature and precipitation data. These indices suggest limited potential change in extreme weather in this ecozone to date: increased diurnal temperature range and variability; more warm days (days with daily maximum temperature >90th percentile); and more summer days (days with daily maximum temperature >25 °C) (but not cold or frost days), depending on location. In terms of more casual observations, in recent decades Aboriginal peoples and researchers have been noticing or reporting more potential extreme weather and impacts, such as unusual tree damage, more berry crop failures, and at least two occurrences of unusual fish die-offs in rivers (Sutton and Albany rivers) during periods of unusually warm summer weather. They also report that weather is less predictable than before, with adverse effects on safety, hunting, and fishing.

[a] Coastal areas will, however, be increasingly vulnerable to hydrocarbon development and transportation activities in the northern marine environment (see Niemi et al. 2010). Continued or potentially greater use of the deepwater shipping port at Churchill (and, to a lesser extent, the non-deepwater port at Moosonee) is more directly associated with the potential for oil spills and the introduction of other forms of pollution along the coast of the Hudson Plains Ecozone.

[b] Many impacts from climate change are linked to more than one of the associated stressor classes (i.e., changes in temperature, precipitation, ice and permafrost, hydrology, and extreme weather), but they are organized according to what is probably the most direct stressor.

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2.6.1.2 **Cumulative impacts**

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Cumulative impacts (or effects) are incremental and accumulating changes to the environment caused by an action in combination or interaction with other past, present, and reasonably foreseeable future actions (Hegmann et al. 1999). Cumulative impacts occur when individual stressors continue to produce environmental effects at rates greater than can be dissipated by the natural system (persistent stressor with additive effects) and when two or more different stressors produce effects that individually might not be cumulative but combine to produce compounding effects (Peterson et al. 1987). Cumulative impact assessments are intended to be carried out at regional scales (which may cross administrative boundaries), as part of an ecosystem approach, rather than at local or project-specific levels (Hegmann et al. 1999; Duinker and Greig 2006).

Cumulative impacts are an important but often inadequately addressed consideration in environmental impact assessments for proposed developments (Dubé 2003; Duinker and Greig 2006). Cumulative impacts assessment can theoretically help identify locations and strategies for projects and infrastructure, so as to minimize impacts and maintain sustainable use. Cumulative impacts are, however, difficult to analyze owing in part to challenges, such as dealing with residual historical effects of past stressors (e.g., historical over-fishing) and establishing cause-and-effect, particularly at large scales; challenges are more acute where data quality is poor or baseline data are largely lacking (Bunch and Reeves 1992; Rosenberg et al. 1997; Halpern et al. 2008; Anielski and Wilson 2009), as is generally the case in the Hudson Plains Ecozone (this report). In broad-scale analyses the distribution of summed anthropogenic drivers does tend to correlate with the distribution of cumulative impacts, but this ignores important small-scale spatial patterns associated with differential ecosystem vulnerability (Halpern et al. 2008). In addition to such analytical constraints, logistical challenges are encountered when coordinating cumulative impacts assessments among jurisdictions and/or managing agencies or cultures, as well as across the linked terrestrial and marine environments (Sallenave 1993).

There are no well-documented examples of regional cumulative impacts resulting from human activities occurring directly in the Hudson Plains Ecozone. However, cumulative impacts from multiple hydroelectric developments are likely in this ecozone and the adjoining marine environment, even if such impacts are difficult to predict and assess (see below). The cumulative damage to much of the ecozone's coastal salt marshes from the direct and indirect effects of persistent foraging by an overabundant migratory lesser snow goose (*Chen caerulescens caerulescens*) population is a notable and well-documented regional cumulative impact (Jefferies and Rockwell 2002), but it is attributable to human influences (largely land use changes) outside of the ecozone (for details, refer back to Section 2.2.2.1, *Coastal*). Cumulative impacts are more apparent at less relevant, smaller, and more project-specific scales in the ecozone, such as near coastal villages. For example, strong declines in the abundance of lake sturgeon (*Acipenser fulvescens*) in the lower reaches of the Eastmain and Opinaca rivers were attributed to recent hydroelectric development there in combination with increased harvest linked to coincidently improved human access to the fishery (increased ease of net fishing associated with the reduced flow and new road access) (Hayeur 2001; Therrien et al. 2004).

Clear potential exists for regional cumulative impacts from past and future (incremental) resource developments in the Hudson Plains Ecozone. A particular concern is that of cumulative impacts on wetland, river, coastal, and interfacing marine environments from multiple hydroelectric developments within the Hudson Bay watershed (e.g., Bunch and Reeves 1992; Rosenberg et al. 1995; McDonald et al. 1996; Arragutainaq et al. 2007). The existing hydroelectric developments in and around the Hudson Plains Ecozone are generally very large-scale developments associated with a multitude of potentially far-reaching effects (Rosenberg et al. 1995, 1997). Some concerns associated with these developments include: modification of the amount, timing, and location of freshwater discharge into the bays with impacts on sea ice, salinity, other aspects of oceanography, and related effects on regional climate and marine foodwebs, including subtidal eelgrass (*Zostera marina*) beds; mercury mobilization (as methylmercury) and subsequent bioaccumulation and contamination of fish and other biota; decreased productivity of anadromous fish populations; potential for increased greenhouse gas emissions from altered wetland hydrology; and related impacts on Aboriginal cultures and economies (Prinsenberg 1983; Bunch and Reeves 1992; McDonald et al. 1996; Arragutainaq et al. 2007; Short 2008). Each new hydroelectric development in and around the Hudson Plains Ecozone will be superimposed on a watershed and receiving bays already altered by past hydroelectric developments. Although cumulative impacts from these large-scale developments are not well defined, additional hydroelectric developments in the ecozone are either in progress or proposed (see Section 2.2.2.4.2, *Rivers/Streams & Lakes*), although in Ontario any new developments in the Severn, Winisk, Attawapiskat, and Albany rivers are currently restricted to ≤25 MW and require proposal by the local Aboriginal community or communities and/or their partner(s) (OPA 2007). A science advisory panel to the Ontario government recently recommended that this existing moratorium on large-scale hydroelectric developments be maintained until comprehensive cumulative impacts research is carried out (Far North Science Advisory Panel 2010). It further recommended that this moratorium be extended to include inter-basin water diversions.

Cumulative impacts from roads are another concern as development proceeds in the Hudson Plains Ecozone. Although only one all-season road currently exists (James Bay road, Québec) to facilitate land-based access into the ecozone from the south (Hydro-Québec 2003; Stewart and Lockhart 2005), feasibility planning is in progress for an all-season road that would run along the western edge of the ecozone from Gillam to Churchill, Manitoba and beyond to Rankin Inlet, Nunavut (Government of Nunavut and Government of Manitoba 2010; SNC Lavalin 2010). As well, studies are underway to assess possible routes for an additional all-season road proposed to permanently connect several coastal communities in the Ontario portion of the ecozone to the highway system in the south (Government of Ontario 2009). The numerous and varied resource development proposals, prospecting activities, and assessments of resource potentials in the ecozone, particularly in mining (Manitoba Geological Survey 2003; OMEI and OMNDMF 2009; Far North Science Advisory Panel 2010; Golder Associates 2010; Micon International 2010), hydroelectric (OPA 2007; Hydro-Québec 2010; MDDEP 2010; Manitoba Hydro 2010; OMNR 2010), and wind-farming (Environment Canada 2008; OMNR 2010; see also Hélimax Énergie and AWS Truewind LLC 2005) sectors, likewise suggest that development of additional roads and infrastructure is imminent. Further human access, use, and development inevitably follows (Hayeur 2001; McDonald et al. 2006; OMEI and OMNDMF 2009; Robinson et al. 2010), typically in association with habitat displacement, deterioration, and fragmentation; the introduction and spread of introduced and potentially invasive species; and declines in fish and wildlife populations – particularly sensitive species that occur at low densities, have large area requirements, are habitat specialists rather than generalists, and/or are particularly vulnerable to over-harvest (e.g., sturgeon, lake trout, wolverine, caribou, wolf) (Johnston et al. 2005; Kaufman et al. 2009; Robinson et al. 2010 and references therein). As such, plans to develop roads and other regional infrastructure to service projects and communities in the ecozone need to be coordinated to limit overall road densities and, thus, maintain a high degree of ecosystem connectivity, intactness, and resilience (Far North Science Advisory Panel 2010).

As elsewhere, accelerated (human-induced) climate change is also expected to become an increasingly important threat contributing to cumulative impacts in the Hudson Plains Ecozone (Far North Science Advisory Panel 2010). Climate change is forecast to manifest more strongly at northern latitudes, and it is already associated with a shorter sea ice season in Hudson and James bays that is, in turn, correlated with changes in higher trophic levels, particularly within portions of the food web involving the polar bear (*Ursus maritimus*) subpopulations that use the ecozone (see sections 2.3.3.1.1, *Polar Bear*; 2.4.3.1, *Predator-Prey Relationships & Cycles*; and 2.4.3.4.1, *Animal Phenology*). Climate change is interacting with other threats and stressors such as pollution, which is increasing the rate at which certain emerging (newer) contaminants are bioaccumulating in these bears (Section 2.3.3.1.1, *Polar Bear*). Climate change is in general expected to increasingly interact with and exacerbate threats and stressors, including ecosystem alteration and loss, overexploitation, invasive species, and pollution, resulting in a multitude of cumulative impacts on both terrestrial and aquatic ecosystems (Schindler et al. 2001; Schindler and Lee 2010). For example, the projected increase in future fire risk from climate change is expected to bring at least the southern portion of the area's fire cycle towards the upper limit of its range of natural variability during the Holocene (at least over the last ~7,000 years) and, as such, there is concern that in the future cumulative impacts, such as those potentially contributed from harvesting, may be sufficient to push the fire cycle there to a new condition outside its long-term natural range (Bergeron et al. 2010; see also Section 2.4.2.2, *Fire*). As such, the extent to which climate change affects this ecozone will depend, in part, on how it is developed, used, and serviced.

Research and monitoring are needed in order to understand and assess cumulative impacts in the Hudson Plains Ecozone. The specific ecosystem components that are most susceptible to cumulative impacts from anticipated increases in development and/or climate change need to be identified. Cumulative impacts on carbon cycling (especially peatland carbon stores) might be one such component of particular concern (e.g., see Turetsky et al. 2002), in addition to cumulative impacts on coastal and river ecosystems and sensitive fish and wildlife species, as above. From a cultural and social perspective, cumulative impacts on Aboriginal cultures and economies (which are dependent on local natural resources) are also important in this geography.

Institutional systems are also needed to manage and use information relevant to cumulative impacts to inform sustainable land use and environmental conservation planning at a regional scale (the Hudson Plains Ecozone is one of the few areas remaining in Canada where it is still possible to proactively consider mitigation of cumulative impacts in a regional land use planning framework) (Far North Science Advisory Panel 2010). Notable in this regard is that under Ontario's new Far North Act (Government of Ontario 2010), the joint First Nations-Government of Ontario body responsible for providing advice on a land use planning strategy for the Far North of Ontario (including the Hudson Plains Ecozone) has it within their mandate to recommend policy statements that include considerations for cumulative impacts. For more information on the Far North Act, see Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*.

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2.6.2 Stewardship/conservation

2.6.2.1 Protected areas

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The Hudson Plains Ecozone constitutes the third largest wetland in the world (Fraser and Keddy 2005), making it globally significant. Although there are numerous watersheds within this wetland, the extremely shallow grade, shallow active layer due to continuous and discontinuous permafrost, the clay substrates, and the extensive peat development contribute to lateral as well as down-grade surface flows. As a result, the watersheds are indistinct and, at least at certain times, non-mutually exclusive, resulting in functional lateral hydrological connectivity. The nature of the hydrology in this ecozone, as well as the biota and ecosystem structure and function, lead to the acknowledgement of this global significance and also provide a rationale for protection of these characteristics. At an international scale, portions of this ecozone have been designated as Wetlands of International Importance (Ramsar Convention sites) (Wetlands International 2010): Polar Bear Provincial Park (Ontario) and the Southern James Bay Migratory Bird Sanctuaries. The latter is comprised of Hannah Bay Migratory Bird Sanctuary (Ontario and Nunavut) and Moose River Migratory Bird Sanctuary (Ontario).

In Manitoba, one large national park (Wapusk National Park) exists at the northern limit of the ecozone. It was created from lands formerly contained within the Churchill Wildlife

Management Area, which was established to manage and protect the wildlife and its habitat in that area (Parks Canada 2007). A substantial portion of the Churchill Wildlife Management Area outside of Wapusk National Park is also proposed for protection. Furthermore, the Kaskatamagan Wildlife Management Area, which was originally designated as the Cape Tatnam Wildlife Management Area in 1973, provides a corridor along the Hudson Bay coast between the Wapusk–Churchill complex and the Ontario border. About one-third of the coastline in the Kaskatamagan Wildlife Management Area was recently legally protected in a central core that extends 50 km inland, and it includes the mouth of the Kaskattama River (Government of Manitoba 2009). The Kaskatamagan Sipi Wildlife Management Area also now protects an inland portion of the ecozone. Three Important Bird Areas (IBAs) have also been identified wholly within Manitoba's portion of the ecozone (IBA Canada 2010). The Kaskattama River Mouth IBA falls within the protected portion of Kaskatamagan Wildlife Management Area. Although the other two IBAs have no official protection status in their own right, parts of both are contained within one or more of the designated but not legally protected wildlife management areas noted above.

In the Ontario portion of the ecozone, the protected area system comprises one large wilderness class park (Polar Bear Provincial Park), several narrow linear corridors along segments of some of the major rivers (Winisk River, Otoskwin-Attawapiskat River, Nagagamisis, Little Current River, Missinaibi, Albany River, and Kesagami provincial parks), four small nature reserves (Adam Creek, Sextant Rapids, Coral Rapids, Williams Island), one natural environment class park (Tidewater Provincial Park), two conservation reserves (Jog Lake, Coral Rapids Wetland), and two tiny Wilderness Areas (Sutton Lake Gorge, Old Fort Albany) (Gray et al. 2009). As already noted, the federal Moose River Migratory Bird Sanctuary and most of the federal Hannah Bay Migratory Bird Sanctuary also lie in the Ontario portion of the ecozone. In addition to the above regulated protected areas, 17 Important Bird Areas are identified along the Hudson Bay and James Bay coasts within Ontario's borders (Gray et al. 2009; IBA Canada 2010). Although not protected, they clearly indicate where some of the most important areas exist, for shorebird and waterfowl breeding and staging in particular, and, as such, indicate where stewardship responsibilities lie. As in Manitoba, several of the IBAs overlap with provincial or federal regulated protected areas, so that stronger mandates exist to protect the natural heritage values contained within those.

In Nunavut, a large part of Akimiski Island is a federal Migratory Bird Sanctuary, and much of its coastal area has been included within an IBA, as well. Marine portions of the federal Hannah Bay and Boatswain Bay migratory bird sanctuaries also lie in Nunavut. The Twin Islands constitute a territorial Wildlife Sanctuary, as well as an IBA, but they have no legal protection at this time (Environment Canada 2009; IBA Canada 2010).

In Québec, several biodiversity reserves (Boatswain Bay, Ministikawatin Peninsula, Missisicabi Plain, Waskaganish, Niquet Stream, and Paakumshumwaau-Maatuskaau) and one aquatic reserve (North Harricana River) have been proposed and are undergoing review at the present time. The North Harricana River Aquatic Reserve is contiguous with the Upper Harricana River Aquatic Reserve and the Collines de Muskuchii Biodiversity Reserve to the south, both of which occur just south of the border of the Hudson Plains Ecozone (within the Boreal Shield Ecozone), and the Hannah Bay Migratory Bird Sanctuary to the north, thus further contributing to ecological connectivity from James Bay well inland. Most of the federal Boatswain Bay Migratory Bird Sanctuary also lies within Québec borders. A single

IBA (not legally protected) is identified in Québec's portion of the ecozone, which overlaps with the federal Boatswain Bay Migratory Bird Sanctuary, as well as the provincial Boatswain Bay Biodiversity Reserve.

Table 32 summarizes the protected areas (only) that lie wholly or partially within the Hudson Plains Ecozone, including their name, categorization, and area. Overall, 12.8% of the ecozone's landbase is now protected in 31 areas in IUCN categories I-III. IUCN (World Conservation Union – previously known as International Union for Conservation of Nature) categories of protected areas are based on primary management objectives. Categories I-III include nature reserves, wilderness areas, and other parks and reserves managed for conservation of ecosystems and natural and cultural features (IUCN 1994). No lands are currently protected as IUCN categories IV-VI.

Table 32. Summary of federal, provincial, and territorial protected areas (IUCN) currently associated with the Hudson Plains Ecozone, organized by date of establishment. This analysis is based on ecozone$^+$ boundaries.
Source: Environment Canada (2009), using Conservation Areas Reporting and Tracking System (CARTS) data from federal, provincial, and territorial jurisdictions.

Name (establishment date)	IUCN category	Adjusted size (km^2)[a]
Federal		
Hannah Bay Migratory Bird Sanctuary (1939)	Ib	239.83 (209.85 km^2 ON; 29.98 km^2 NU)
Akimiski Island Migratory Bird Sanctuary (1941)	Ib	2,023.34 (NU)
Boatswain Bay Migratory Bird Sanctuary (1941) (see also the provincial (QC) Boatswain Bay Biodiversity Reserve)	Ia	104.53 (90.40 km^2 QC; 14.13 km^2 NU)
Moose River Migratory Bird Sanctuary (1958)	Ib	4.96 (ON)
Wapusk National Park of Canada (1996) (established from lands originally part of the Cape Churchill Wildlife Management Area, below)	II	10,661.14 (MB)
Provincial – Manitoba		
Kaskatamagan Wildlife Management Area (2009) (originally established in 1973 as the Cape Tatnam WMA; was renamed in 2009, when a portion of it was legally protected)	II	2,595.30
Kaskatamagan Sipi Wildlife Management Area (2009)	Ib	1,338.20

Table 32, Cont.

Name (establishment date)	IUCN category	Adjusted size (km^2)^a
Provincial – Ontario		
Nagagamisis Provincial Park (1957)	II	34.64
Sutton Lake Gorge Wilderness Area (1960)	III	0.51
Old Fort Albany Wilderness Area (1960)	III	0.04
Winisk River Provincial Park (1969)	II	185.54
Polar Bear Provincial Park (1970)	Ib	23,328.26
Tidewater Provincial Park (1970)	II	9.80
Missinaibi Provincial Park (1970)	II	112.61
Kesagami Provincial Park (1983)	Ib	27.58
Williams Island Provincial Nature Reserve (1985)	Ia	0.08
Coral Rapids Provincial Nature Reserve (1985) (see also the provincial (ON) Coral Rapids Wetland Conservation Reserve)	Ia	0.12
Sextant Rapids Provincial Nature Reserve (1985)	Ia	0.04
Adam Creek Provincial Nature Reserve (1985)	Ia	0.50
Little Current River Provincial Park (1989)	II	32.11
Albany River Provincial Park (1989)	II	2.51
Otoskwin-Attawapiskat River Provincial Park (1989)	II	180.67
Jog Lake Conservation Reserve (1997)	II	484.82
Coral Rapids Wetland Conservation Reserve (2005) (see also the Coral Rapids Provincial Nature Reserve)	II	61.05
Provincial – Québec		
Boatswain Bay Biodiversity Reserve, proposed (2003) (see also the national Boatswain Bay Migratory Bird Sanctuary)	III	102.28
Ministikawatin Peninsula Biodiversity Reserve, proposed (2003)	III	873.52
Missisicabi Plain Biodiversity Reserve, proposed (2003)	III	631.10

Table 32, Cont.

Name (establishment date)	IUCN category	Adjusted size (km^2)[a]
North Harricana River Aquatic Reserve, proposed (2003)	III	203.67
Waskaganish Biodiversity Reserve, proposed (2004)	III	995.06
Niquet Stream Biodiversity Reserve, proposed (2005)	III	14.19
Paakumshumwaau-Maatuskaau Biodiversity Reserve, proposed (2008)	III	808.23
Territorial – Nunavut		
NA[c]		
Total IUCN area as of May 2009 (corresponds to figures 132 and 134)		41,122.73 (11.7%[b])
Total IUCN area as of December 2010 (includes two new legally protected areas in Manitoba announced in December 2009 and not shown in figures 132 and 134; see Figure 133)		45,056.23 (12.8%)

[a] Area values for terrestrial protected areas were adjusted to reflect only the portion of the protected area that falls within Hudson Plains Ecozone$^+$ boundaries. Protected areas with both terrestrial and marine components were considered a single unit and assigned to either the terrestrial ecozone (Hudson Plains Ecozone$^+$) or the adjacent marine ecozone (Hudson Bay, James Bay, and Foxe Basin Ecozone$^+$), based on centroid location.

[b] Percentage values are calculated based on the Hudson Plains Ecozone$^+$ size of 352,980 km^2.

[c] The Twin Islands Wildlife Sanctuary (1,425 km^2) is not currently legally protected (CCEA 2009).

The locations of protected areas and their overall distribution across the ecozone, as of May 2009, when they accounted for 11.7% of the land base, are shown in Figure 132. The two protected areas announced in Manitoba later in 2009, i.e., the Kaskatamagan Wildlife Management Area and Kaskatamagan Sipi Wildlife Management Area, are not included in this figure and are shown instead in Figure 133.

The growth of the ecozone's network of protected areas over time, from 1939 (when the first protected area, Hannah Bay Migratory Bird Sanctuary, was established) up to and including May 2009 (the two protected areas announced in December 2010 are again not shown) is shown in Figure 134. Area-wise, major gains were made with the addition of Polar Bear Provincial Park in 1970 and Wapusk National Park in 1996. Together, these two large protected areas currently account for 75% of the total area protected in this ecozone. Several small biodiversity reserves and other protected areas have been established, since 2003 (Table 32, Figure 134), including the Kaskatamagan Wildlife Management Area and Kaskatamagan Sipi Wildlife Management Area.

Although the protected area system is relatively well developed and extensive in the Hudson Plains Ecozone, representation gaps remain. Representation gaps in Manitoba are mostly in the Hudson Bay Lowland Ecoregion. Likewise, the Ontario portion of the Coastal Hudson Bay Lowland Ecoregion (Ontario Ecoregion 0E) has few gaps remaining, but substantial gaps remain in its portions of the Hudson Bay Lowland Ecoregion (Ontario Ecoregion 1E, no

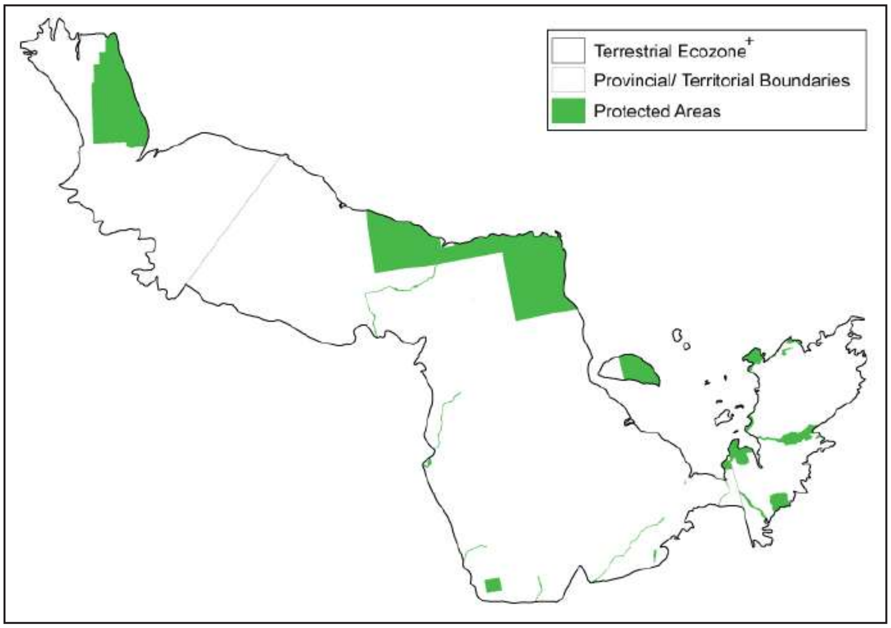

Figure 132. Map of protected areas (legally protected areas and, for Québec, also proposed and soon to be legally protected areas) in the Hudson Plains Ecozone (ecozone$^+$ boundaries), up to and including May 2009. The Kaskatamagan Sipi Wildlife Management Area and a portion of Kaskatamagan Wildlife Management Area that were announced in December 2009 are not shown in this illustration (see instead Figure 133). Note that some protected areas occur as narrow linear corridors along the segments of some of the major rivers. Source: Environment Canada (2009), using Conservation Areas Reporting and Tracking System (CARTS) data from federal, provincial, and territorial jurisdictions.

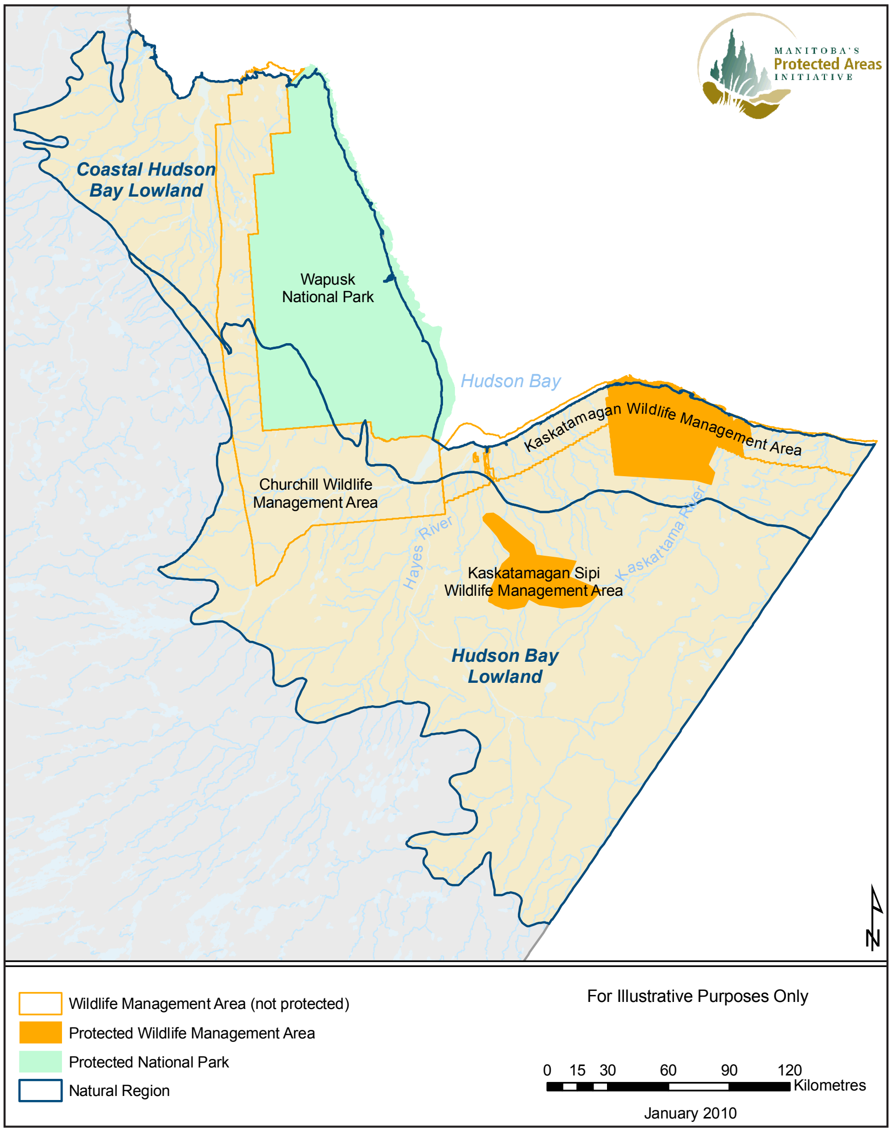

Figure 133. Map of legally protected areas, as well as designated but not legally protected Wildlife Management Areas, in the Manitoba portion of the Hudson Plains Ecozone, as of 2010. Note that the legally protected areas on the coast (both Wapusk National Park and one of the two provincially protected areas) are to some extent buffered and connected by the Churchill Wildlife Management Area (not legally protected) and the designated but not legally protected portion of the Kaskatamagan Wildlife Management Area. Source: Manitoba Conservation, Protected Areas Initiative.

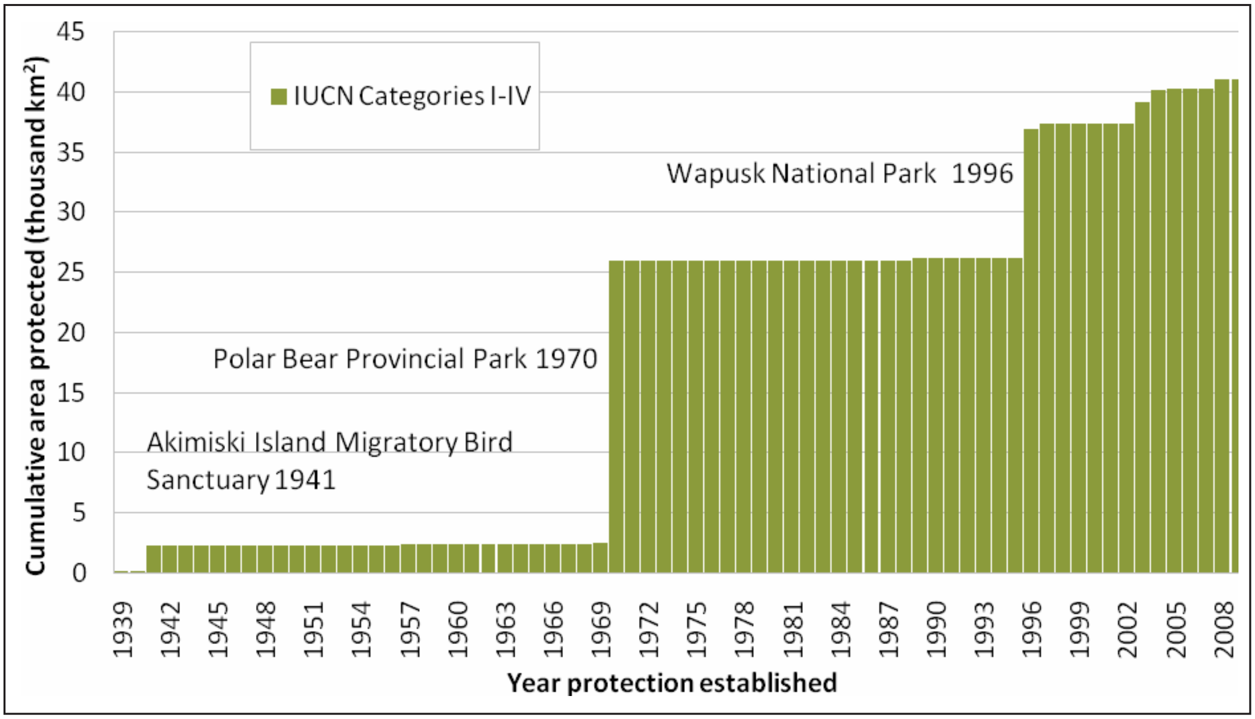

Figure 134. Growth of protected areas in the Hudson Plains Ecozone from 1939 up to and including May 2009 (data correspond with Figure 132 and include legally protected areas, as well as some proposed and soon to be legally protected areas in Québec). The three largest protected areas, are noted, along with their dates of establishment. Several small biodiversity reserves and other protected areas have been established, since 2003. As the year 2009 represents the period up to and including May 2009, it does not include the two newest protected areas announced in Manitoba in December 2009. The Kaskatamagan Wildlife Management Area (protected portion) and Kaskatamagan Sipi Wildlife Management Area contribute an additional 2,595 km^2 and 1,338 km^2 of legally protected area to the ecozone, respectively. This analysis is based on ecozone$^+$ boundaries.
Source: Environment Canada (2009), using Conservation Areas Reporting and Tracking System (CARTS) data from federal, provincial, and territorial jurisdictions.

protected areas) and the James Bay Lowland Ecoregion (Ontario Ecoregion 2E, contains only small protected areas in each of its three ecodistricts). Although protected areas have increased greatly in the Québec portion of the Hudson Plains Ecozone, since 2002, representation gaps remain there as well (Brassard et al. 2010).

Despite the relatively extensive protected area system that exists in the ecozone, the degree of connectivity between protected areas is also low in some areas. A large coastal gap of about 150 km exists between Polar Bear Provincial Park in Ontario and the Kaskatamagan Wildlife Management Area in Manitoba. Portions of this unprotected coast have been identified as Important Bird Areas (IBA Canada 2010), but these IBAs are not regulated, and they have no legal standing. Further notable in regards to coastal connectivity, however, is that nearly the entire coastline of the Coastal Hudson Bay Lowland Ecoregion in Manitoba is either designated or protected lands. Two protected areas, a portion of Kaskatamagan Wildlife Management Area and Wapusk National Park, are buffered by designated wildlife management areas (Figure 133).

Ontario recently began a major multi-year community-based land use planning initiative with a protection component (Far North Land Use Planning Initiative) that includes its portion of the Hudson Plains Ecozone and is now supported by legislation in the form of the Far North Act (Government of Ontario 2010; and see also Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*). Associated with this initiative is the protection of at least 50% of the area of the Far North of Ontario (includes the Ontario portion of the Hudson Plains Ecozone) in an interconnected network of protected areas. The configuration and location of the new protected areas is yet to be determined, but the existing approaches to earth and life science representation and protected area design used in the province suggest that there is a high probability that additional protected areas will be established in the Ontario portion of the ecozone in the future. Manitoba (Manitoba Conservation 2010) and Québec (Government of Québec 2009) also identified new initiatives that include the potential for new protected areas in their portions of the broader boreal forest and could include additional areas in this ecozone. Manitoba has identified *Areas of Special Interest* within the ecozone that are being considered for possible enhanced protection (Manitoba Conservation 2010).

Wapusk National Park is currently mandated to report on its ecological integrity. The park submitted its first 5 year ecological integrity monitoring plan in March 2008 (Wapusk National Park 2008). The plan includes both condition monitoring and management effectiveness monitoring. Results will be reported through a State of the Park Report. Ecological integrity also is the guiding principle in Ontario's Provincial Parks and Conservation Reserves Act (Government of Ontario 2006). Although individual park reporting is not required, a system-wide State of Ontario's Protected Area Report is required every 5 years. Ontario's first report will be published in 2011. There is no periodic re-assessment of the ecological integrity of the other protected areas in the ecozone at the present time. The anthropogenic pressures on the existing protected areas are relatively limited, although they may increase in the near future as proposals increase for mining, hydroelectric, wind power, oil, and gas developments in this ecozone (Section 1.2, *Human History*). Such developments, including associated roads, have the potential to endanger the globally significant natural heritage values found in the ecozone. Climate change is also an important emerging threat to the ecozone's protected areas (McKenney et al. 2010). On the positive side, within the Ontario portion of the ecozone, the former Mid-Canada Line radar sites that exist within Polar Bear Provincial Park are being cleaned-up (contamination, building materials, etc.) over the next number of years (see Section 2.6.3.2, *Restoration of Former Mid-Canada Line Radar Sites*).

Finally, it is noteworthy in closing that the protected area system in the adjoining marine environment is currently less well developed than in the terrestrial Hudson Plains Ecozone. Only a very small area of the marine Hudson Bay, James Bay, and Foxe Basin Ecozone is legally protected in IUCN categories (Niemi et al. 2010). As well, no Ecologically and Biologically Sensitive Areas (EBSAs) nor Marine Protected Areas (MPAs) have yet been officially designated within Hudson Bay, although much important habitat for marine mammals and birds is recognized along the Hudson Bay and James Bay coasts and the associated near-shore marine environment. For more information, see the Arctic Marine Ecozones report of the ESTR (Niemi et al. 2010).

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2.6.2.2 **Land use & ecosystem management initiatives**

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Several land use planning and ecosystem management initiatives are in place in the Hudson Plains Ecozone that involve federal, provincial (Manitoba, Ontario, Québec) or territorial (Nunavut), and/or Aboriginal partners (Table 33). However, the degree of engagement among jurisdictions is not uniform nor do component jurisdictions have similar initiatives in place. Moreover, the effectiveness of these activities for conserving and improving biodiversity and ecosystem health has generally not been assessed. Co-management agreements among First Nations and other levels of government are, however, a notably important type of stewardship initiative in this ecozone, with some important new initiatives introduced in recent years.

Table 33. Select land use planning and ecosystem management initiatives in the Hudson Plains Ecozone. Sources: As indicated.

Participant	Description	Main objectives	Source
Federal, Québec, Aboriginal communities	James Bay and Northern Québec Agreement (JBNQA)	Provide environmental and social protection; coordinate hunting, trapping, and fishing in a sustainable way.	Government of Québec (1998)
Federal, Nunavut, Aboriginal communities	Eeyou Marine Region Land Claims Agreement (EMRLCA)	For islands in James Bay and Hudson Bay that are offshore of Québec and not covered by the JBNQA, address issues of access and responsibility for management related to the environment – especially contaminated sites, protected areas, and wildlife harvesting and management.	Government of Canada, Government of Nunavut, and Grand Council of the Crees (2010); Grand Council of the Crees (2010)
Federal, Québec, Aboriginal communities, Hydro-Québec, research institutions	Northern Québec Regional Steering Committee (this committee was initially associated with the now complete Northern Ecosystem Initiative, but it still operates more informally)	Encourage consensus among stakeholders, including Aboriginal, on environmental issues in the north; work together to address Aboriginal concerns and effectively encourage and support community-based projects.	
Federal, Manitoba, Municipal (Town of Churchill), Aboriginal communities	Wapusk Management Board	Deal with decisions related to the planning, development, and management of Wapusk National Park; reflect Parks Canada's mandate through its philosophy, rooted in Aboriginal culture, that people are Keepers of the Land.	Parks Canada (2010)

Table 33, Cont.

Participant	Description	Main objectives	Source
Manitoba	Protected Areas Initiative	Permanently protect a representative sample of all 18 natural regions and sub-regions in the province, including the Hudson Plains Ecozone.	Manitoba Conservation (2010a)
Manitoba	Wildlife Management Areas	Manage, conserve, and enhance wildlife resources, including in the Churchill, Kaskatamagan, and Kaskatamagan Sipi Wildlife Management Areas of the Hudson Plains Ecozone.	Government of Manitoba (2010); Manitoba Conservation (2010b)
Manitoba, Aboriginal communities	Resource Management Boards	Provide integrated land use planning in Resource Management Areas, including the York Factory, Fox Lake, and Spit Lake Resource Management Areas, which occur wholly or partially in the Hudson Plains Ecozone.	
Ontario, Aboriginal communities	Far North Land Use Planning Initiative, supported by the Far North Act	1) Set out a process for community-based land use planning with a leadership role by First Nations, with such land use plans being in place in advance of major developments; 2) support protection for at least half of the area of the Far North of Ontario in an interconnected network of protected areas designated in community-based land use plans; 3) maintain biological diversity and ecological processes/functions, including carbon storage and sequestration; and 4) enable sustainable economic development of natural resources that benefits First Nations, while recognizing the environmental, social, and economic interests of all Ontarians.	Government of Ontario (2010)
Ontario, Aboriginal communities	Far North Forestry Development Initiative[a]	Provide funding to First Nations communities for projects related to land use planning and forest-based economic development, including commercial forestry, in the Northern Boreal Initiative Area of the Far North of Ontario.	Forestry Futures Trust Committee (2010)
Ontario	Waterpower Site Release and Development Review policy	Contribute to the environmental, social, and economic well being of the peoples of Ontario, including Aboriginal communities, through the provision of opportunities for waterpower development and the sustainable development of Ontario's crown land, while recognizing the Ontario Ministry of Natural Resources' mission of ecological sustainability. Two key policy areas are Northern Rivers and the Moose River Basin north of Highway 11.	OPA (2005); OMNR (2010b)

Table 33, Cont.

Participant	Description	Main objectives	Source
Ontario	Ontario Biodiversity Strategy	Aim to protect the genetic, species, and ecosystem diversity in Ontario (many features of the Hudson Plains Ecozone are not adequately represented in existing protected areas in Ontario).	OMNR (2005)
Ontario	Ontario's Woodland Caribou Conservation Plan	Provide broad policy direction regarding conservation and recovery of the forest-dwelling ecotype of woodland caribou (*Rangifer tarandus caribou*). The plan summarizes actions and initiatives the Government of Ontario intends to undertake to conserve and recover this ecotype of woodland caribou (ecotypes are described in Section 2.3.3.2.1, *Caribou*).	OMNR (2009a)
Ontario	Cervid Ecological Framework	Provide policy advice to address cervid management at the broad landscape level. Consolidates and integrates Ontario's approach to managing cervid species in relation to each other, with consideration of the broader ecosystem(s) they share. Within the Hudson Plains Ecozone, this is addressed through land and resource planning processes, whereby habitat management for caribou is emphasized. Species-specific policy and program direction are contained in other policy documents (e.g., Ontario's Woodland Caribou Conservation Plan).	OMNR (2009b)

[a] This initiative is being superceded by Ontario's Far North Land Use Planning Initiative, but the Forestry Futures Trust Committee continues to work with its applicants to ensure successful completion of projects funded under this initiative.

Some initiatives, such as the James Bay and Northern Québec Agreement (JBNQA), have the capability to have a direct influence at a broad level on the ecozone's biodiversity and integrity. The JBNQA was signed in 1975 (Government of Canada and Québec 1976) after plans were announced to build a system of hydroelectric dams in northern Québec (Peters 1999). These areas were still used for Aboriginal hunting pursuits, so as part of the agreement an environmental protection regime was outlined. This mandated that consideration be given to such aspects as the protection of hunting, fishing, and trapping rights; protection of wildlife resources, physical and biotic environments, and ecological systems; and minimizing negative environmental and social impacts – all with respect to development activities (Government of Canada and Québec 1976; Government of Québec 1998). Under this agreement, protection bodies, which include Aboriginal, federal, and Québec provincial government representatives, are appointed for the review and formulation of laws and regulations for environmental protection, to set guidelines for environmental and social impact assessment, and to evaluate and review impact assessments (Government of Canada and Québec 1976; Government of Québec 1998; Peters 1999). This strategy has potential to address biodiversity in the Québec portion of the Hudson Plains Ecozone, as established working groups address species-specific conservation issues. Although the JBNQA does affect coastal development, it does not, however, address land use planning or include offshore waters.

The Eeyou Marine Region Land Claims Agreement (EMRLCA) (Government of Canada, Government of Nunavut, and Grand Council of the Crees 2010) was recently concluded for some islands located offshore of Québec in Hudson and James bays (Nunavut), which are not covered by the JBNQA (Figure 135). This new federal-territorial (Nunavut)-Aboriginal agreement is to address issues of access and responsibility for management related to the environment, especially contaminated sites and protected areas, as well as wildlife harvesting and management. The Government of Canada and the Eeyou Istchee Cree agreed to base the EMRLCA on the Nunavik Inuit Land Claims Agreement, which received Royal Assent in 2008. The negotiation process for the Agreement-in-Principle took less than 5 years. The EMRLCA was approved by referendum of the Eeyou Istchee Cree in March 2010 (Grand Council of the Crees 2010), and it was signed by all parties in July 2010 (Government of Canada, Government of Nunavut, and Grand Council of the Crees 2010). Like the Nunavik Inuit Land Claims Agreement, the EMRLCA has a unique jurisdictional aspect: its beneficiaries are in Québec, while the claim is located in Nunavut.

Other initiatives with federal engagement include the Northern Québec Regional Steering Committee and the Wapusk Management Board in Manitoba (Table 33). The Northern Québec Regional Steering Committee evolved out of an initiative by Environment Canada's Québec Region to encourage potential partners, especially Aboriginal, to reach consensus on environmental issues in the north and to develop a cooperative plan based on the Northern Ecosystem Initiative's (NEI)[85] priorities. Although the NEI is now complete (2008), the committee assembled to construct and implement the Northern Québec Action Plan (representatives from Aboriginal communities; federal, provincial, and regional governments; research institutions; and Hydro-Québec) continues to operate more informally.

The Wapusk Management Board was formed to co-manage Wapusk National Park after its creation in 1996 from part of the former Cape Churchill Wildlife Management Area[86]. This co-management arrangement was articulated in the Federal-Provincial Memorandum of Agreement for Wapusk National Park, through two major intents: 1) the park is to be managed in the context of its adjoining lands, and 2) the residents of the area are to continue to have access to the park lands (Parks Canada 2007). The Wapusk Management Board consists of 10 members appointed by associated Aboriginal communities and federal, provincial, and municipal governments (Parks Canada 2007, 2010). The board makes recommendations to the federal Minister of Environment on matters related to the planning, management, and

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[85] The Northern Ecosystem Initiative (NEI) (now complete) was established by Environment Canada in 1998 to enhance the future health and sustainability of northern communities and the ecosystems they depend upon. It brought partners together to identify shared priorities and work cooperatively to address them (see https://www.ec.gc.ca/Publications/default.asp?lang=En&xml=8EFA7A77-C3BE-40E8-8C89-50E7C4D48947).

[86] In Manitoba, the earliest land use planning efforts to conserve the invaluable ecosystems found along the Hudson Bay coast resulted in the establishment and designation in 1973 of the Cape Churchill Wildlife Management Area (WMA) and the Cape Tatnam WMA. The remainder of the Cape Churchill WMA (that which is not in Wapusk National Park) is now called the Churchill WMA. The Cape Tatnam WMA was renamed the Kaskatamagan WMA after a portion of it was legally protected in 2009 (see Section 2.6.2.1, *Protected Areas*).

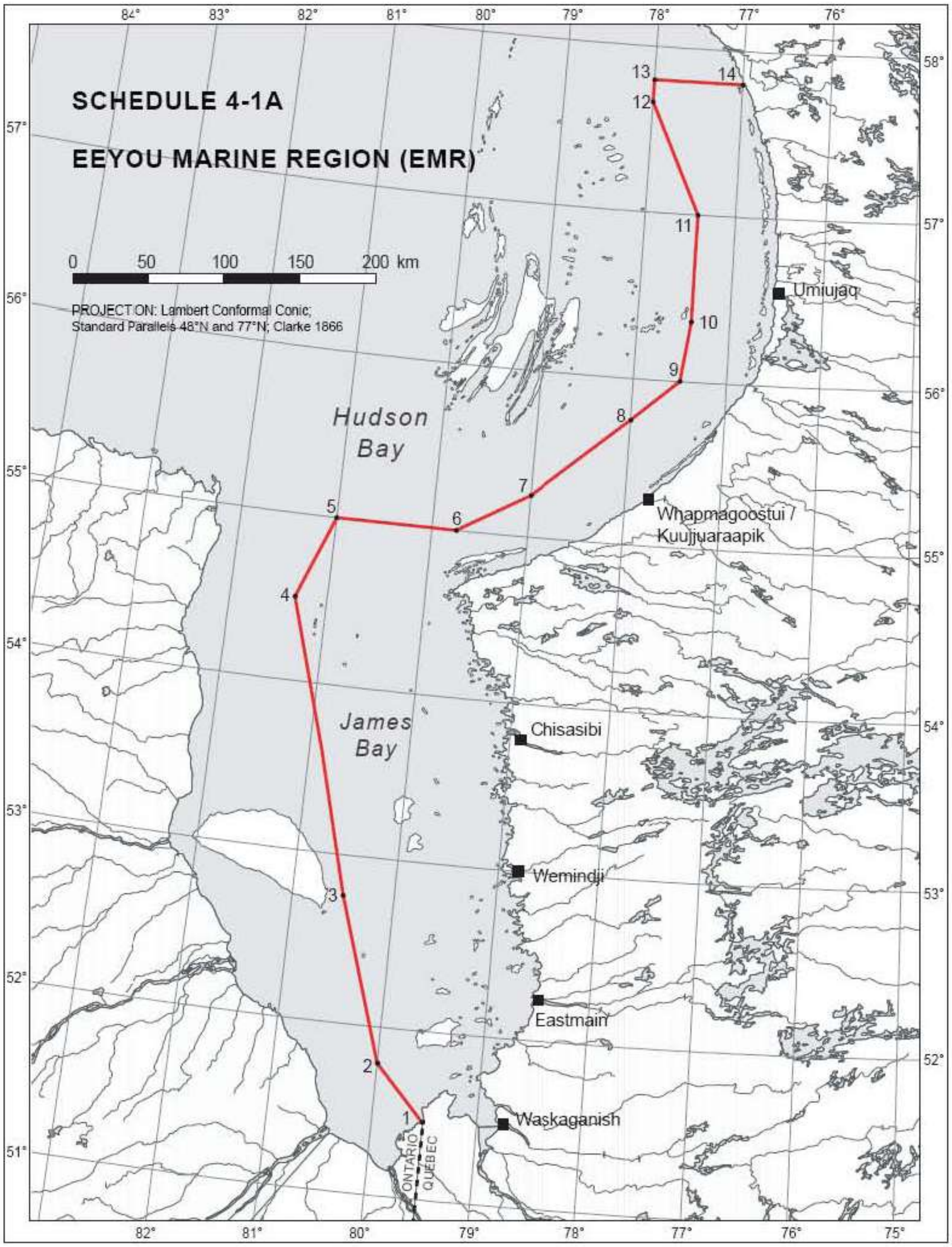

Figure 135. Map showing the area of offshore islands in Hudson and James bays covered by the Eeyou Marine Region Land Claims Agreement. The agreement applies to islands in Nunavut that are offshore of Québec's portion of the Hudson Plains Ecozone, as well some islands offshore of the more northerly Taiga Shield Ecozone. The numerical points reference specific geographic coordinates, as defined in Government of Canada, Government of Nunavut, and Grand Council of the Crees (2010).
Source: Government of Canada, Government of Nunavut, and Grand Council of the Crees (2010). Figure used with permission of Her Majesty the Queen Right of Canada.

development of the park, while Parks Canada administers day-to-day operations (Parks Canada 2007)[87].

Provincial management and planning efforts (Table 33) demonstrate various targeted management levels (i.e., landscape level or ecosystem level), and they are inconsistent among provinces. Manitoba's main land management strategy with respect to the Hudson Plains Ecozone is its Protected Areas Initiative. This initiative, established in 1990, is a provincial commitment to the environment and its natural diversity, with primary emphasis on conserving biodiversity. The intent is to protect a representative sample of all 18 of Manitoba's natural regions and sub-regions within the Hudson Plains Ecozone. Increasing protection in the Hudson Plains Ecozone has been a priority (for a description of new protected areas in the Manitoba portion of the ecozone, see Section 2.6.2.1, *Protected Areas*). As well, Manitoba has three Resource Management Areas (RMAs) that occur at least partially within ecozone boundaries: York Factory Resource Management Area (entirely within), Fox Lake Resource Management Area (mostly within), and Spit Lake Resource Management Area (eastern edge within). All of these RMAs have Resource Management Boards comprised of members of the associated First Nation and provincial government departments. The mandate of the RMA boards is to promote land use planning, resource management, and environmental monitoring in their respective Resource Management Area.

Ontario announced a comprehensive land use planning and protection initiative (Far North Land Use Planning Initiative) in the summer of 2008 that is now supported by legislation in the form of a Far North Act (Government of Ontario 2010). The objectives, which apply to public land in the Far North of Ontario (including the Ontario portion of the Hudson Plains Ecozone), are to: 1) set out a process for community-based land use planning that includes a significant role for First Nations in the planning (community-based land use plans are to be developed by First Nations in advance of major developments[88]); 2) support protection for at least half of the area of the Far North of Ontario in an interconnected network of protected areas designated in community-based land use plans; 3) maintain biological diversity and ecological processes/functions, including carbon storage and sequestration; and 4) enable sustainable economic development of natural resources that benefits the First Nations, while recognizing the environmental, social, and economic interests of all Ontarians. This initiative has substantial potential for protection of biodiversity and ecosystem integrity for the majority of the Hudson Plains Ecozone that lies in Ontario, particularly in the face of increasing pressure there for further resource developments (e.g., OPA 2007; Environment Canada 2008; OMEI and OMNDMF 2009; Golder Associates 2010; Micon International 2010; OMNR 2010a). The initiative was further supported by a Far North Science Advisory Panel (2010) to the Ontario Ministry of Natural Resources. This advisory panel recommended a regional scale *conservation-matrix*

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[87] Although this conservation measure for Wapusk National Park, as well as certain other conservation measures, are described in this section as they relate to ecosystem management, protected areas are more thoroughly discussed in the preceding section (*Protected Areas*).

[88] As specified under Section 12(1) of the act, if no community-based land use plan is in place, with few exemptions, no person shall undertake any of the following developments except if required authorization was obtained prior to the act coming into force (Government of Ontario 2010): opening a mine; commercial timber harvest; oil and gas exploration or production; constructing or expanding wind power or water power (hydroelectricity) generation facilities; constructing or expanding electrical transmission and distribution systems and lines; constructing or expanding all-season transportation infrastructure, including roads; and any other land use or activity that is prescribed.

model for the aforementioned land use planning, supported by adaptive management, and an associated, sustained commitment to the collection and sharing of both scientific and Aboriginal information (Far North Science Advisory Panel 2010). The proposed conservation matrix allows for a variety of land uses and would consist of: 1) large core protected areas that serve as landscape-level benchmarks; 2) site-specific protected areas that are geared to protecting specific values, which might not be well represented within benchmark areas; 3) active management areas, where development and settlement are actively occurring; and 4) the surrounding landscape or remaining conservation matrix, within which the other three elements are embedded, and within which roads and other linear corridors that connect development areas are carefully planned to maintain a high degree of connectivity or overall intactness.

This type of comprehensive land use planning has not yet been developed in either Manitoba or Québec to similarly help guide development in the Hudson Plains Ecozone. That said, both jurisdictions recently put forth intents for further stewardship of their respective far north lands (including their portions of the Hudson Plains Ecozone). Specifically, the Government of Manitoba (2009) committed to developing a boreal peatlands stewardship strategy in co-operation with stakeholders and leading climate change non-governmental agencies, and the Government of Québec (2009) committed to protect from industrial development at least 50% of the area covered by its Plan Nord, i.e., lands north of the 49th parallel.

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2.6.3 Restoration

In the Hudson Plains Ecozone, restoration efforts have been concentrated in large projects, such as aquatic habitat in the Lower Churchill River of Manitoba and clean-up of the former Mid-Canada Line radar sites in Ontario. These two projects are profiled in this section, followed by a brief overview of some of the other restoration projects initiated in this ecozone.

2.6.3.1 Restoration of the lower Churchill River

Warren J. Bernhardt, North/South Consultants Inc.

In 1976, the Churchill River was impounded at Southern Indian Lake, Manitoba (outside of the Hudson Plains Ecozone), and most of its flow was diverted by means of the Churchill River Diversion (CRD) to hydroelectric generating stations on the Nelson River. Consequently, post-CRD discharge along the lower Churchill River (in the Hudson Plains Ecozone) has been considerably lower than historical levels (Manitoba Hydro and Town of Churchill 1997). The diversion reduced the average natural rate of discharge into Hudson Bay by about 40%, from 1,274 m^3/s to an average operative (with diversion) rate of 510 m^3/s (Manitoba Hydro 2010).

In 1994, Manitoba Hydro and the Town of Churchill initiated the Lower Churchill River Water Level Enhancement Weir Project to address concerns raised by the town with respect to adverse effects of the CRD on the water regime in the lower Churchill River. The primary objectives of the project were to improve boat access along the lower Churchill River and to increase the amount and productivity of fish habitat in the lower Churchill River. To meet these objectives, a low-head rock weir was constructed across the Churchill River in 1998 at a location immediately upstream of tidal influence (Figure 136). The weir raised the water level immediately upstream of it by ~2 m and created a reservoir that permanently rewatered 8.11 km^2 of pre-CRD river bed along a 10 km reach of river (Bernhardt and Posthumous 2003). The resultant reservoir provided more lacustrine habitat along that reach of river.

Immediately prior to construction of the project, the most abundant larger fish species occurring along the lower Churchill River included lake whitefish (*Coregonus clupeaformis*), white sucker (*Catostomus commersoni*), longnose sucker (*Catostomus catostomus*), and northern pike (*Esox lucius*). Small numbers of lake sturgeon (*Acipenser fulvescens*), cisco (*Coregonus artedi*), round whitefish (*Prosopium cylindraceum*), brook trout (*Salvelinus fontinalis*), arctic grayling (*Thymallus arcticus*), and walleye (*Sander vitreus*) were also found. Lake whitefish and northern pike were the primary species of concern to a small domestic fishery on the lower Churchill River. It was anticipated that additional wetted area and water depth resulting from construction of the weir would result in increased overwintering habitat and a larger benthic invertebrate food base for fish (Manitoba Hydro and Town of Churchill 1997). Shifts in fish community structure towards species that favour lacustrine conditions, such as white sucker, were anticipated. Species that favour riverine conditions, such as round whitefish and longnose sucker, were expected to decrease in abundance within the reservoir. Permanent increases in the availability of overwintering habitat and an enhanced food base, in combination with long-term increases in the amount of spawning habitat, were expected to have a positive effect on the northern pike population. Although the availability of spawning habitat was not believed to be limiting lake whitefish abundance, the increased suitability and availability of feeding and overwintering habitat in the reservoir were expected to have a positive effect on lake whitefish population size.

Studies conducted prior to construction of the weir indicated that cisco, juvenile lake whitefish, and small numbers of brook trout moved between the lower Churchill River and the Churchill River estuary (Manitoba Hydro and Town of Churchill 1997). To accommodate upstream fish movement past the weir, two boulder-garden style fishways were constructed, as integral components to the weir. One was positioned in the middle of the river channel (Main Stem fishway; Figure 137), and the second was located on the east side of the river, where a small tributary enters the reservoir (Goose Creek fishway; Figure 137). The fishways were designed to allow a 200 mm cisco or whitefish to ascend the weir (Manitoba Hydro and Town of Churchill 1997).

A 7 year post-project monitoring program was developed and initiated in 1999 to monitor aquatic ecosystem responses to operation of the project. Monitoring program components included documenting changes to water quality; the distribution, abundance, and composition of invertebrate communities; fish community structure and population size within the reservoir; fish movement upstream past the weir; and fish utilization in tributaries entering into the reservoir.

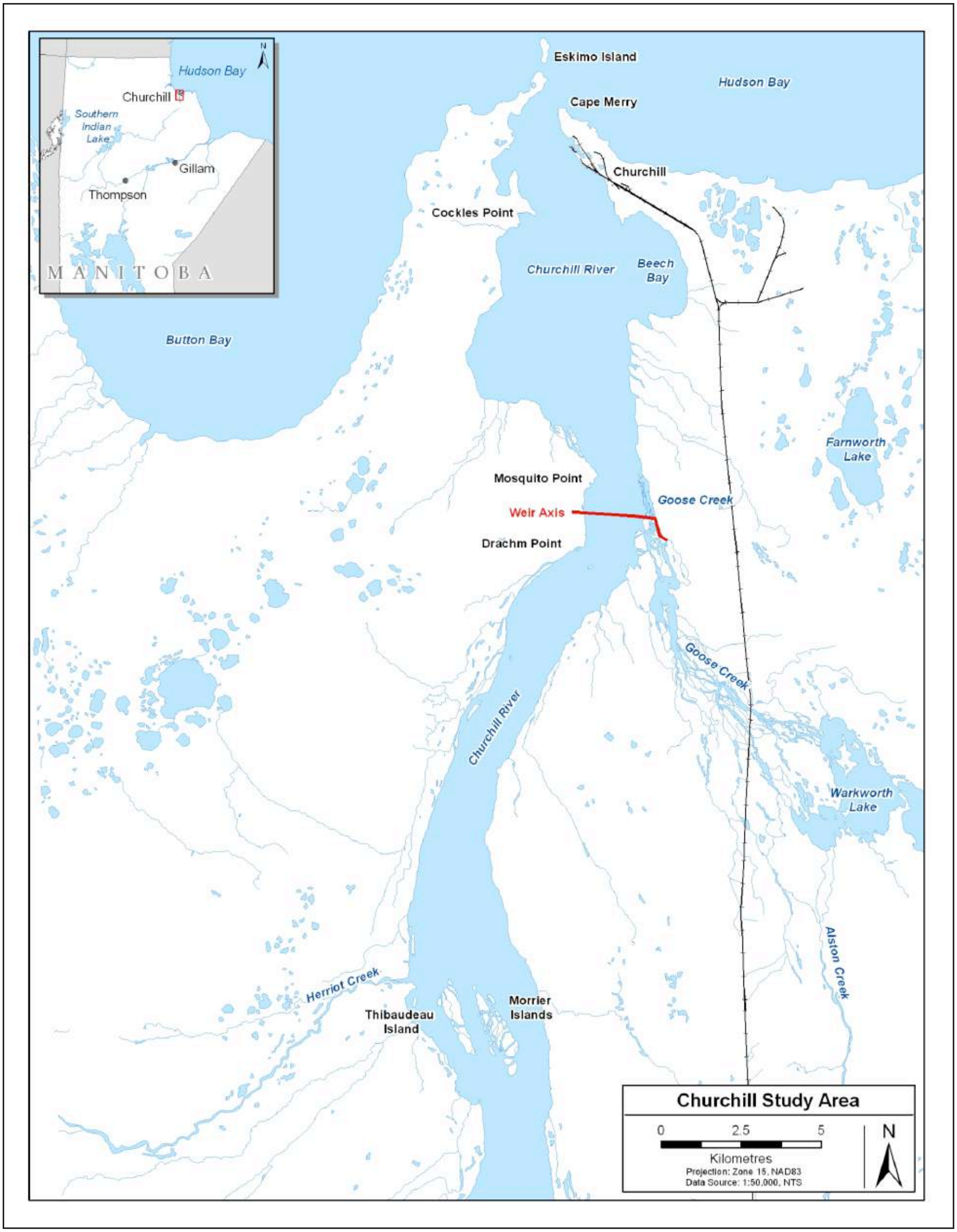

Figure 136. The lower Churchill River study area, illustrating the location of the weir axis and other key sites along the lower Churchill River in the Hudson Plains Ecozone.
Source: North/South Consultants Inc.

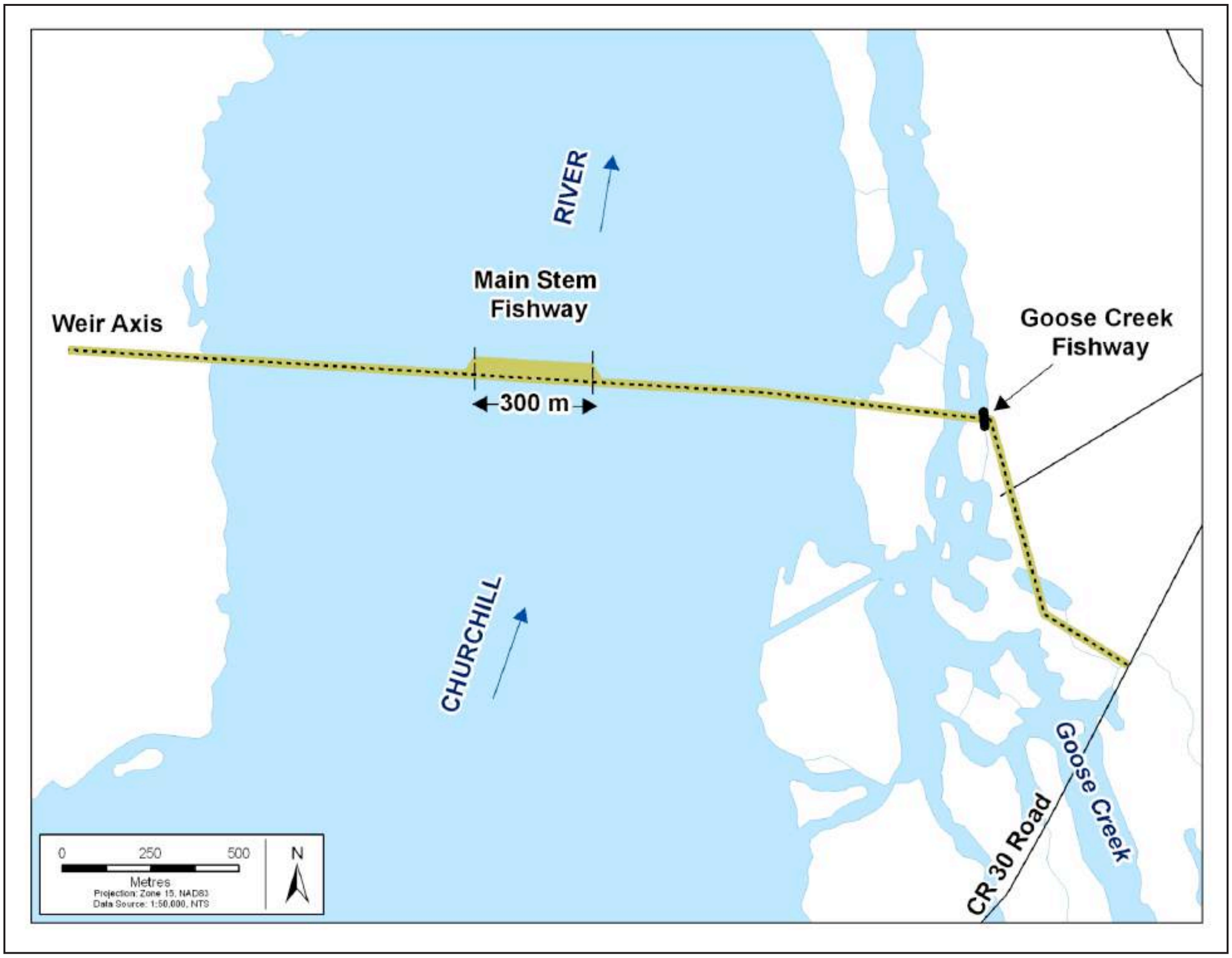

Figure 137. Locations of the Main Stem and Goose Creek fishways along the Churchill River weir. Source: North/South Consultants Inc.

Results indicated no change to water quality (Bezte 2006), and that benthic invertebrates had colonized rewatered habitat within the reservoir, as expected (Capar et al. 2006). Benthic invertebrate density in those areas was at least as high as in the original river channel. Monitoring of fish communities within the reservoir indicated a decline in abundance of riverine species. In particular, round whitefish and arctic grayling became less abundant. Walleye, which is a species more frequently associated with lacustrine environments, increased slightly in abundance within the reservoir. However, at the end of the monitoring program in 2005, the abundance of northern pike and lake whitefish had not increased as expected. Further, monitoring of fish utilization in local tributaries revealed an abrupt decline in the numbers of longnose and white suckers using the creeks (Bernhardt 2005; Bernhardt and Holm 2005). This result was unexpected, and reasons for it remain unclear.

Fish movement upstream past the weir was monitored using a variety of techniques, including radio and acoustic telemetry, radio frequency identification technology (RFIT), and by examining otolith microchemistry to determine anadromous movements. Results indicated that northern pike and arctic grayling were able to successfully ascend the Goose Creek fishway during spring, and that some pike also ascended via the Main Stem fishway (Peake 2001; Peake and Bernhardt 2002). Monitoring at the Main Stem fishway was hindered by the size of the fishway (300 m wide and 50 m long), difficult working conditions, and changes to the fishway between years caused by ice during spring break-up. Results have differed between studies and methodologies, showing varying proportions of tagged cisco and lake whitefish successfully ascending the weir during fall. Examination of strontium concentrations in otoliths taken from cisco and lake whitefish captured upstream of the weir indicated that all cisco and 50% of whitefish sampled had been in marine waters the previous summer, indicating successful movement over the weir (Bernhardt 2003). In contrast, radio, acoustic, and RFIT studies showed considerably fewer tagged fish ascending the weir (Peake 2001; Peake and Bernhardt 2002; Peake 2003, 2004). Most studies at the Main Stem fishway were hampered by technological difficulties associated with the harsh working conditions, and it is believed that results underestimate the ability of fish to successfully ascend the weir.

As a result of unexpected fish community changes upstream of the weir and the inconclusive results from the fish passage monitoring programs, studies have been continued to document the status of northern pike and lake whitefish populations within the reservoir, to monitor the status of sucker populations and understand the reasons for the decline in use of local tributaries, and to further assess fish passage through the fishways.

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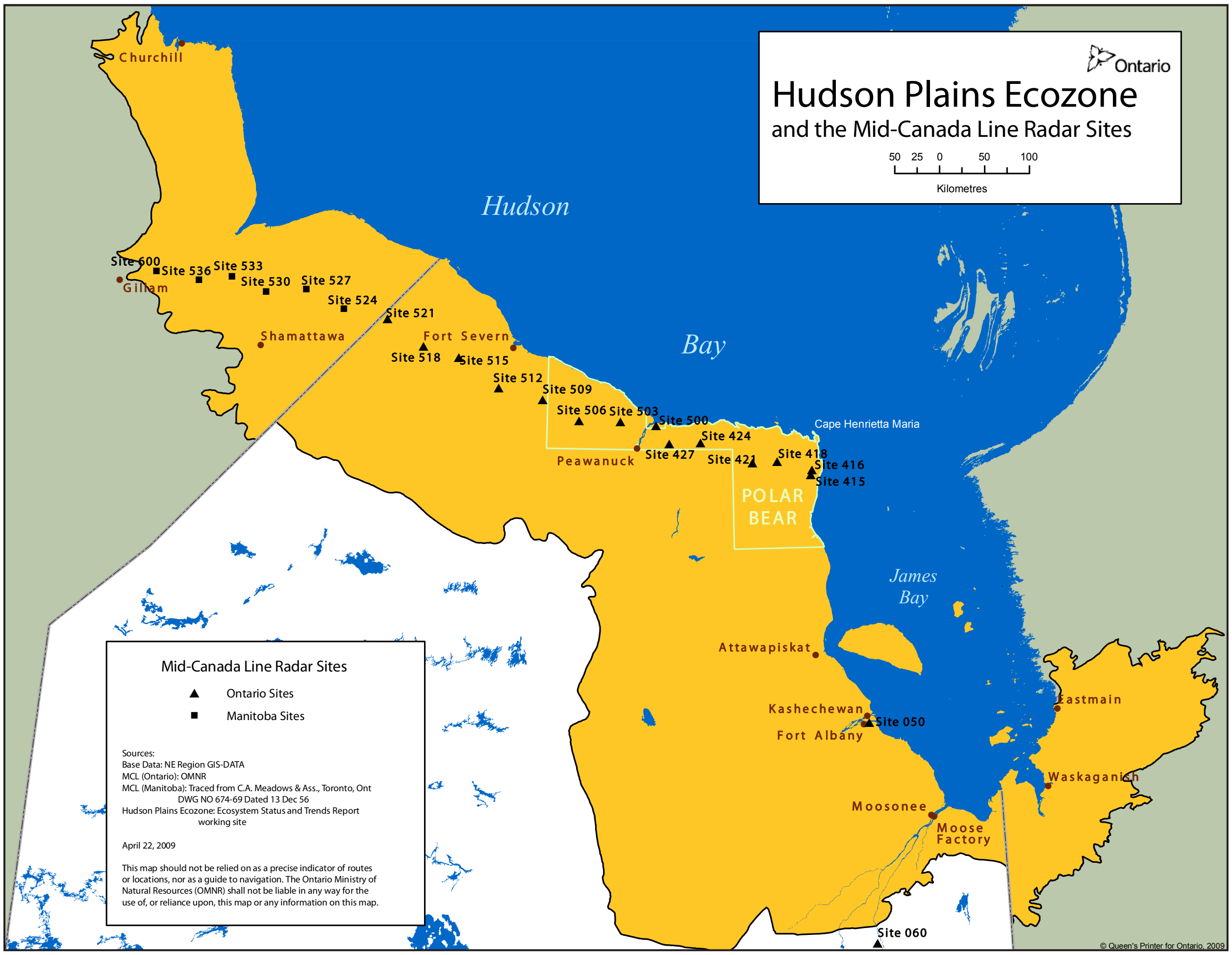
2.6.3.2 **Restoration of former Mid-Canada Line radar sites**

Bruce Mighton, Ontario Ministry of Natural Resources

The Mid-Canada Line (MCL) was designed, constructed, and operated by the Canadian Department of National Defence as an early warning *Doppler* detection system during the cold war era. The radar line became fully operational in 1958 and closed in 1965, when it was determined no longer economically feasible or strategically required, due to new technologies. Located along the 55th parallel between the DEW Line and the Pinetree Line detection systems, the MCL consisted of 90 unmanned Doppler Detection Stations (DDS) and eight sector control stations connected through a series of relay stations in Ontario, to North Bay. Twenty-one of these former MCL sites are located within the Hudson Plains Ecozone: 15 in Ontario and six in Manitoba (Figure 138). Some former MCL sites are also found on Bear Island, Nunavut (near the mouth of James Bay) and in northern Québec, but those MCL sites are located north of Hudson Plains Ecozone boundaries.



Sources:
Base Data: NE Region GIS-DATA
MCL (Ontario): OMNR
MCL (Manitoba): Traced from C.A. Meadows & Ass., Toronto, Ont DWG NO 674-69 Dated 13 Dec 56
Hudson Plains Ecozone: Ecosystem Status and Trends Report working site

April 22, 2009

This map should not be relied on as a precise indicator of routes or locations, nor as a guide to navigation. The Ontario Ministry of Natural Resources (OMNR) shall not be liable in any way for the use of, or reliance upon, this map or any information on this map.

Figure 138. Geographic locations of former Mid-Canada Line radar sites in the Hudson Plains Ecozone (ecozone+ boundaries). A number of the radar sites are located in Polar Bear Provincial Park (the park is delineated by the green line).

In 1996, First Nation leadership formally raised the issue of the need to clean-up the abandoned Mid-Canada Line radar sites in Ontario to the Indian Commissioner of Ontario. Further assessment and delineation were conducted by the governments of Canada and Ontario to determine the nature and extent of contamination and waste at each MCL site in the Ontario portion of the Hudson Plains Ecozone, plus two other sites in the province (Proctor and Redfern Ltd 1992; Henderson Paddon and Associates 1996; ESG 1998, 1999a,b; SNC Lavalin 2001, 2002, 2004, 2005; OMNR 2006; MERC 2007a; DST Consulting Engineers 2008). The studies confirmed the presence of environmental contamination, including petroleum hydrocarbons, asbestos, heavy metals, pesticides, and polychlorinated biphenyls (PCBs). Limited evaluation of hare tissue samples at Site 060 (Relay/Foxville) resulted in a health advisory for country foods at that site, because PCB levels exceeded Health Canada guidelines for safe food consumption (OMOE 2000)[89]. Studies by the Mushkegowuk Environmental Research Centre document continued First Nation concern about contamination and impacts to local ecosystems and human health (MERC 2006, 2007b). These concerns are based on the Health Canada advisory, as well as local Aboriginal perspectives and knowledge about impacts in the area.

Negotiations for a cost-sharing agreement between the governments of Canada and Ontario to clean-up the former Mid-Canada Line radar sites in Ontario broke down early in 2000. At that time, the government of Ontario acted to fund and clean-up the most contaminated site (Site 050, Anderson Island), which is located adjacent to Fort Albany. Based on the clean-up of the Fort Albany site, the Ontario Ministry of Natural Resources developed an Integrated Project Plan and Business Case (February 2008) for the remediation of the remaining 16 MCL sites in the province. The governments of Canada and Ontario then negotiated an agreement to fund the 6 year remediation project, which began in the summer of 2009 (Table 34). The Ontario Ministry of Natural Resources has worked directly with First Nations throughout the project planning process, through a chiefs' steering committee and a technical advisory committee.

Leeches (*Haemopis* spp.) were used to monitor PCB levels in the Albany River following remediation of the first site in 2001 (Site 050, Anderson Island) (Tsuji and Martin 2009). Although PCB levels were still elevated 4 years after remediation, the study results suggested that PCB levels were declining. Thus, removal of the terrestrial source of PCBs at Site 050 appears to have removed the main source of PCBs in the river (Tsuji and Martin 2009).

Approved remediation plans are not similarly in place to clean-up the six former MCL sites located in the Manitoba portion of the Hudson Plains Ecozone. It is notable, however, that clean-up of the two former MCL radar sites located on Bear Island, Nunavut (just outside the northern boundary of the Hudson Plains Ecozone) is now complete (INAC 2010).

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[89] There is no specific guideline for PCBs in hares. *Level of concern* was established by comparison with the federal maximum residue limit (MRL) for PCBs in beef and other meat under the Food and Drugs Act and Regulations (OMOE 2000). The whole weight MRL is 20 ppb PCBs, whereas hares from the Relay site averaged 45 ppb and 23 ppb in liver and muscle tissue, respectively. PCBs were non-detectable in the liver tissues of off-site hares (muscle tissue was not sampled in off-site hares). Sampling was limited to three specimens both on-site and off-site. Insufficient information was available on how much hare and other traditional foods are eaten locally, to assess the actual risk.

Table 34. Six year clean-up schedule for the former Mid-Canada Line radar sites in the Ontario portion of the Hudson Plains Ecozone. Shading denotes the anticipated clean-up period for each site. Abbreviation: DDS, Doppler Detection Station (small site).

Former Mid-Canada Line radar site	Year					
	2009	2010	2011	2012	2013	2014
Site 050	N/A, clean-up complete (2001)					
Site 060, Relay[a]	▨					
Site 070, Ramore[a]	▨					
Site 500, Winisk		▨	▨	▨		
Site 424, DDS Polar Bear Provincial Park			▨			
Site 427, DDS Polar Bear Provincial Park			▨			
Site 506, DDS Polar Bear Provincial Park				▨		
Site 503, DDS Polar Bear Provincial Park				▨		
Site 415, Cape Henrietta Maria				▨	▨	▨
Site 421, DDS Polar Bear Provincial Park					▨	
Site 418, DDS Polar Bear Provincial Park					▨	
Site 416, DDS Polar Bear Provincial Park					▨	
Site 521, DDS					▨	▨
Site 518, DDS					▨	▨
Site 515, DDS					▨	▨
Site 512, DDS					▨	▨
Site 509, DDS					▨	▨

[a] Sites located outside of the Hudson Plains Ecozone.

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2.6.3.3 **Overview of other restoration initiatives**

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2.6.3.3.1 **Coastal areas damaged by geese**

Habitat degradation in coastal areas of this ecozone, caused by intense goose foraging (Section 2.2.2.1, *Coastal*), has prompted discussion and actions towards the management and restoration of this habitat, which has been described as being in peril (Batt 1997). Dramatic increases in the Mid-Continent population of lesser snow goose (*Chen caerulescens caerulescens*) during the last four decades have led to major loss of vegetation in coastal salt marshes (damaged areas extend from Manitoba to James Bay, including Akimiski Island, Nunavut) (Section 2.2.2.1, *Coastal*), as well as progressive vegetation loss in adjacent tundra wetlands (Section 2.2.2.2, *Polar-Tundra*). The capacity and potential for severely degraded coastal habitat (i.e., areas transformed into hypersaline bare sediment) to recover is not well known, but recovery (defined as a return to the previous plant community) is not likely to be attained by simply lowering levels of goose herbivory (Handa and Jefferies 2000; Handa et al. 2002). Small-scale studies that evaluate the use of native species for the revegetation of severely damaged areas have yielded good results (Handa and Jefferies 2000), but they also show that in the short-term recovery likely cannot occur unassisted at some sites (Handa and Jefferies 2000). Rather, manipulations of various attributes (water, propagules, nitrogen) are needed to move the system into a more stable vegetated state (Handa et al. 2002). Unassisted revegetation (defined as development of a new vegetation community from an unvegetated state caused by the geese) requires erosion of consolidated sediments and the establishment of plant assemblages in fresh unconsolidated sediments, which might occur on an unknown time scale, likely decades.

The Arctic Goose Habitat Working Group was formed with the purpose of evaluating the impacts of overabundant snow geese and determining whether and how snow goose populations could be reduced. The goal was to limit further damage by the geese and aid the recovery of degraded arctic and subarctic marshes (Batt et al. 1997). This working group recommended reducing snow goose populations by lengthening hunting seasons, liberalizing bag and possession limits and means of hunting, and increasing egg collecting by northern Aboriginal peoples, in order to bring population growth below replacement levels. Pursuant to the working group's recommendations, extraordinary changes were made to migratory bird treaty and regulations in the United States and Canada, which took effect in 1999 (USFWS 1999). The effectiveness of this effort to reduce snow goose populations with the aid of hunters has been reviewed (Alisauskas et al. in press; J. Leafloor, Candian Wildlife Service, in prep.). The harvest rate increased and survival rate decreased for a small portion (~10%) of the Mid-Continent population of lesser snow goose (the southern-nesting stratum, south of 60° N latitude), but no change occurred in the other 90% of this population that nests north of 60° N (northern-nesting stratum) and thus not for the overall population (Alisauskas et al. in press). It is clear that snow goose overpopulation and the associated consequences will not be corrected with one single management strategy. It is not clear whether the goal of population reduction can be met or how. The population has continued to increase toward carrying capacity, although more slowly than before. Recovery of arctic and subarctic marshes cannot occur without reduction of foraging pressure from overabundant geese. The recent increase in the importance of polar bear (*Ursus maritimus*) as a predator of snow goose eggs (associated with climate change) might aid management attempts to reduce the size and reproductive output of some local nesting colonies of lesser snow goose along portions of the coast (sections 2.4.3.2, *Herbivore-Plant Interactions* and 2.4.3.4.1, *Animal Phenology*).

2.6.3.3.2 **Gravel & landfill sites in the Churchill area**

In the Churchill area, considerable environmental disturbance was caused by the excavation and extraction of gravel for development purposes (Firlotte and Staniforth 1995). The reclamation potential of some of these gravel extraction sites has been evaluated, however, no restoration projects have been initiated (Firlotte and Staniforth 1995; Firlotte 1998; Rausch and Kershaw 2007).

The Churchill open landfill site attracted polar bear for many years, some individuals of which became dependent or problem bears (Stirling 1998). This site was covered with gravel and grain dust from the Port of Churchill and closed in 2004 (A. Meijering, Town of Churchill, pers. comm.). The grain dust was used to rehabilitate the site and accelerate vegetative growth. The town now transports all garbage to Thompson, Manitoba for disposal, but, in the future, it plans to use an exhausted gravel pit as landfill for the disposal of used construction material.

2.6.3.3.3 Churchill Rocket Research Range site

The Churchill Rocket Research Range, site of the Aerobee rocket launcher, was designated a National Historic Site by Parks Canada in 1988, and it was plaqued in 2001. The general property around the rocket launcher and centre was transferred to provincial jurisdiction in 1990 under the responsibility of the Department of Government Services and Infrastructure (S. Kearney, Manitoba Conservation, pers. comm.). The site was assessed with consultants in 2008, and soil samples were taken to determine possible contaminant levels. A remediation action plan is to be developed based on final results received under the Final Environmental Assessment Report. The Churchill Northern Studies Centre, now located in the former rocket range building on the site, has also been working with Parks Canada to clean-up fuel barrels at the site (M. Goodyear, Churchill Northern Studies Centre, pers. comm.)

2.6.3.3.4 Wapusk National Park

In 2002, Parks Canada retained a consultant to conduct a combined Phase 1 and 2 Environmental Site Assessment (ESA) of Wapusk National Park. The ESA was limited to fuel barrels, landfills, and military rockets. The ESA study identified 128 empty and 26 partially full fuel barrels at inactive fuel-cashing sites and 122 empty and 160 partially full fuel barrels at active fuel-cashing sites (EBA Engineering Consultants 2002). In 2008, a clean-up program was initiated, which consists of removal of old fuel barrels during the winter months by skidoo and komotic (sled). Fuel cashing is only done at designated sites, and each active site has a marked spill remediation kit. In the park there are also a number of military rockets, observation towers, and camp sites, which are being assessed as cultural artefacts to determine whether they should be removed or remain in situ.

2.6.3.3.5 Victor mine

The Victor open-pit diamond mine near Attawapiskat, Ontario was established in an area that was principally fens and bogs but also contained ponds, creeks, smaller rivers, and riverbank and creek margin forests (DeBeers Canada 2005; see also sections 2.2.2.4.1, *Wetlands (Freshwater)*; 2.2.2.4.2, *Rivers/Streams & Lakes*; and 2.4.6, *Hydrological Processes*). The mine was constructed beginning in 2006, began operation in 2008, and has an estimated lifespan of 12 years (DeBeers Canada 2005, 2008). After the production phase of the project, environmental monitoring and aggressive rehabilitation programs will be initiated to address the primary environmental considerations: water, fisheries, and wildlife and habitat displacement. Studies began in conjunction with Laurentian University in 2003 to determine methods and native species (only) for use in the re-establishment of original peatland vegetation (Stevens et al. 2004). Although most of the disturbed area will be rehabilitated after mine closure (albeit not all to original ecological classifications), certain areas will not be restored, e.g., the new channel created after diversion of Granny Creek will become the permanent creek channel, and the open pit will be actively flooded to create a small lake with closed drainage (DeBeers Canada 2005). A reclamation closure plan is in place (AMEC 2004).

2.6.3.3.6 Areas affected by hydroelectric developments in Québec

In Québec, measures aimed at mitigating impacts on fish populations were implemented in areas of the ecozone being affected by impoundment of the Opinaca reservoir, which is part of the La Grande hydroelectric complex (Therrien et al. 2004). Specifically, shore cleaning was done to promote spawning (especially of lake trout, *Salvelinus namaycush*) (1978-1981), tributaries were also cleaned to promote spawning of the main species (1979-1983), and weirs were constructed to raise water levels in the reduced-flow portions of the Eastmain and Opinaca rivers (1981-1984). On balance these restoration attempts were not very successful. Cleanings did not meet their objective to promote spawning and, although weirs in the Eastmain and Opinaca rivers benefited species like walleye (*Sander vitreus*), cisco (*Coregonus artedi*), and white sucker (*Catostomus commersoni*) by enhancing plankton production, they also created blockages implicated in the failure of lake sturgeon (*Acipenser fulvescens*) recruitment (Therrien et al. 2004). The new Rupert River diversion project in Québec was designed to include several weirs that are intended to maintain water levels within the reduced flow section of the lower Rupert River, which lies in the Hudson Plains Ecozone (Hydro-Quèbec 2010; see also Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

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3.0 INTEGRATED ANALYSIS OF HUDSON PLAINS ECOZONE$^+$ STATUS & TRENDS

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This assessment report for Canada's Hudson Plains Ecozone has systematically examined the status, changes, and trends occurring in a range of individual ecosystem attributes relevant to ecosystem health, viz., abiotic drivers of the ecozone; elements of ecozone structure, composition, and function; ecosystem services; and human influences. In this final section of the report, the information hereto presented mostly by reporting framework component (i.e., in context of individual ecosystem attributes) is integrated to address the following topics for this ecozone: 1) national and international significance and implied stewardship responsibility; 2) information status; 3) overall ecozone health; and 4) emerging issues and related information needs. Statements are supported by the appropriately cross-referenced section(s) of the report, where more details are provided; any citations not available in the main report are identified with footnotes.

3.1 Uniqueness & implied stewardship responsibility

A number of attributes render the Hudson Plains Ecozone of national and international significance and implied stewardship responsibility. Ultimately, many of these attributes are linked to the ecozone's close proximity to, and interconnectedness with, Hudson and James bays – extensions of the Arctic Ocean and part of the largest inland sea in the world (Section 2.1, *Abiotic Drivers*). The presence of seasonal sea ice in these bays leads to cooler temperatures in the adjacent terrestrial environment than what is otherwise typical of this latitude. As such, the Hudson Plains Ecozone, which flanks both bays, supports the southernmost continuous permafrost and tundra vegetation in North America. Some species of arctic affinity, including polar bear (*Ursus maritimus*), arctic fox (*Vulpes lagopus*), and some plants, also reach their southernmost occurrence in this ecozone.

The ecozone is Canada's largest wetland complex and the third largest in the world (Section 2.2.1.1, *Overview of Ecozone Structure*). Its coasts represent some of the largest and best-developed polar salt marshes in the world (Section 2.2.2.1, *Coastal*), while its inland areas are characterized by extensive freshwater wetlands (Section 2.2.2.4.1, *Wetlands (Freshwater)*). The wetlands that predominate in this ecozone are a function of its poor surface drainage, which is associated with low relief, impermeable clayey soils, and permafrost.

The ecozone's extensive wetlands are of hemispheric importance to migratory birds (Section 2.3.3.3, *Birds*). At an international scale, parts of the ecozone are designated Wetlands of International Importance (Ramsar sites) (Section 2.6.2.1, *Protected Areas*): Polar Bear Provincial Park and Southern James Bay Migratory Bird Sanctuaries (Moose River Bird Sanctuary and Hannah Bay Bird Sanctuary). As well, an extensive network of Important Bird Areas (IBAs) is identified along the ecozone's coasts in all three component provinces (Manitoba, Ontario, Québec) and on Nunavut's islands. The ecozone supports a number of bird species of national conservation concern (sections 2.3.2, *Trends in Species of National Conservation Concern* and 2.3.3.3, *Birds*). Such species include the Endangered red knot (*Calidris canutus rufa*), for which James Bay is a key migration (staging) area and the Special Concern yellow rail (*Coturnicops noveboracensis*), for which the ecozone is thought to represent about 90% of its North American breeding range.

A large proportion of the ecozone's wetlands are peat-forming wetlands (bogs and fens), which also makes this ecozone Canada's largest peatland complex and the second largest at northern latitudes (>40-50°) (Section 2.2.2.4.1, *Wetlands (Freshwater)*). The massive amount of carbon contained in these peatlands is stabilized by cold temperatures and high water levels, and peat has accumulated in the ecozone over thousands of years (Section 2.4.4, *Carbon Cycling*). These exceptional peatlands are significant for carbon storage and cycling at the global scale and, thus, for regulating the effects of accelerated (human-induced) climate change (sections 2.4.4, *Carbon Cycling* and 2.5.2.1, *Climate Regulation Services*).

Another feature that contributes to the national and international significance of the Hudson Plains Ecozone is its exceptionally high degree of intactness or low level of anthropogenic fragmentation (Section 2.2.1.2.2, *Landscape Fragmentation*). The ecozone's largely undisturbed boreal (including transitional taiga) forests form an important part of the largest intact tract of forest in Canada, which is also considered one of the largest intact forests remaining in the world (Section 2.2.2.3, *Forests (Boreal)*). Its intact landscapes are important for species of national conservation concern like polar bear, woodland caribou (*Rangifer tarandus caribou*), and wolverine (*Gulo gulo*), which require large tracts of unfragmented and/or unroaded landscape and are especially vulnerable to human disturbance (sections 2.3.3.1.1, *Polar Bear*; 2.3.3.2.1, *Caribou;* and 2.3.3.2.3, *Furbearers*). Presently, the ecozone's intact landscapes are interrupted by only a single all-season road (QC), two railway lines (MB, ON), winter roads, and hydroelectric transmission corridors (Section 2.2.1.2.2, *Landscape Fragmentation*). Development is limited to some hydroelectric projects within (as well as upstream of) the ecozone and one active mine; commercial forestry, peat harvesting, and agriculture are effectively absent (sections 2.2.1.2.1, *Changes in Land Cover* and 2.2.2.4.2, *Rivers/Streams & Lakes*).

The ecozone also hosts many of Canada's large rivers, and portions of the ecozone are also recognized as supporting among the highest diversity of freshwater fish species in Canada (sections 2.2.2.4.2, *Rivers/Streams & Lakes* and 2.3.3.4, *Fish*). Many rivers remain unregulated or still have areas of natural flow, particularly in the majority of the ecozone that lies in Ontario. The natural state of many rivers and streams is important for anadromous fish species, including brook trout, lake whitefish, and cisco, as well as the migratory lake sturgeon, which is another species of national conservation concern.

The Hudson Plains Ecozone continues to support the traditional Aboriginal subsistence way of life, including mixed economies (sections 1.2, *Human History*; 2.5.3, *Provisioning Services;* and 2.5.4, *Cultural Services*). As such, this ecozone also has significance for the socio-cultural diversity of peoples.

3.2 Information status

The Hudson Plains Ecozone has received comparatively little inventory, monitoring, and research owing to its remoteness, limited access, harsh climate, wet edaphic conditions, and associated low amount of resource development to date. Much of the information available for this ecozone is now dated, because it was generated during a hydroelectric development phase in the 1970s to early 1980s. Inventory, monitoring, and related work has tended to be episodic (typically associated with development cycles), without continuity over the long term – given certain exceptions, including climate station monitoring and some studies of waterfowl, polar bear (and sea ice in the broader geographic area), and fish mercury levels in areas affected by hydroelectric development in Québec. A geographical bias to the available information is also evident, with more information pertaining to coastal than inland areas. Moreover, the available scientific information is contained in disparate sources of variable accessibility (some unpublished). Aboriginal traditional knowledge (ATK) is even less accessible in a form suitable for incorporation into this type of reporting framework.

In short, then, a relative paucity of status and trends information was available for this assessment of the Hudson Plains Ecozone. As such, extensive temporal and spatial gaps in knowledge and general understanding are evident among most of the environmental and ecological parameters examined. There was, therefore, inherent difficulty in detecting changes, trends, and thresholds; describing natural ranges of variability; and, in many cases, even providing quantitative, baseline or other point-in-time measures of ecosystem attributes. As well, the valuation of ecosystem services is in its infancy (Section 2.5, *Ecosystem Services*).

Similarly, it was often difficult to report definitively and quantitatively on causes of ecosystem change. Environmental data are comparatively limited for this ecozone. For example, no trend data are yet available from a limited number of recently installed permafrost-monitoring stations (Section 2.1, *Abiotic Drivers*), and comparatively few hydrometric gauging stations are located within the ecozone (the Hudson Bay watershed more generally is recognized as having one of the most deficient streamflow networks in Canada) (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). Even basic potential stressors important to fish and wildlife management, particularly harvest, are insufficiently spatially and temporally tracked (sections 2.3.3, *Trends in Species of Special Interest* and 2.5.3, *Provisioning Services*). Little effort has also been spent monitoring contaminants (except persistent organic pollutants and metals in polar bear and mercury in some fish populations affected by hydroelectric development), interactions between and among stressors, or examining cumulative impacts (Section 2.6.1, *Stressors & Cumulative Impacts*). As well, the efficacy of most ecosystem management initiatives (Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*) and restoration initiatives (Section 2.6.3, *Restoration*) has not been assessed.

Of the data and information that is available for the Hudson Plains Ecozone, broad-ranging data sets or analyses (which are often managed or co-managed by national organizations, including federal government departments) are potentially most suited to informing this scale of assessment – due to their enhanced spatial coverage and associated standardization of data across component jurisdictions. The Hudson Plains Ecozone is, however, included in only a relatively small number of environmental and ecological data sets of this nature (this report), viz., climate; sea ice; hydrometric parameters; surficial geology; remotely sensed land cover (and related assessments of net primary productivity, Normalized-Difference Vegetation Index, Dynamic Habitat Index, and intactness or fragmentation); forest inventory (photo plots only); disturbances from large fires (≥ 2 km^2); forest insect disturbances; protected areas; and dams.

Challenges are still encountered using some of these broad-ranging analyses. Although such challenges are not necessarily unique to the Hudson Plains Ecozone, they are often more acute in this geography. Insufficient spatial and/or temporal coverage or sampling (and thus poor change detection) is common. For example, although long-term climate data from Environment Canada represents probably one of the best spatial and temporal environmental data sets in Canada, only 2-3 climate stations in the ecozone had data sufficient to support the 1950-2007 climate trends analysis of the ESTR (Section 2.1, *Abiotic Drivers*). These few stations are also all coastal; inland conditions are not represented. The forest insect disturbance surveys conducted cooperatively by the Canadian Forest Service and provincial agencies also generally provide long-term spatial coverage across Canada, but they have not extended very far into the Hudson Plains Ecozone, owing to its low priority for management intervention (Section 2.4.2.3, *Large-Scale Native Insect Outbreaks*). Sampling difficulties led to insufficient spatial coverage of the ecozone in other broad-scale data sets. For example, federal censuses by Statistics Canada lack data from some of the ecozone's small and mostly Aboriginal communities in some years, such that historical census data cannot be reliably used to report on human population trends in this ecozone (as a whole) (Section 1.2.1, *Settlement History*).

Issues with data quality or resolution are also commonly encountered, irrespective of spatial and temporal coverage. For example, the coarse (1 km) spatial resolution remote sensing land cover analysis used nationally for all ecozones in the ESTR was not very useful for the Hudson Plains Ecozone (Section 2.2.1, *Overview of Ecozone Structure & Land Cover Change*). Even though it provided complete spatial coverage of the ecozone and was extended over a period long enough to support land cover change analysis (1985-2005), its land cover classes are very broad (exclude distinct wetland classes), and the land cover product also poorly resolves or erroneously represents the ecozone's tundra and forests. Furthermore, the usefulness of remote sensing is in general limited in this geography by the dynamic nature of the inter-annual changes found there (e.g., seasonality of swamps), difficulty in differentiating open water from wetlands, and the paucity of ground truthing in this largely inaccessible terrain (Section 2.2.1.2.1, *Changes in Land Cover*). Similarly, the National Forest Inventory information for the Hudson Plains Ecozone is currently based only on 2 km x 2 km photographic plots (no ground-based plots), meaning that its inventory attributes are estimated only via photo-interpretation (cf. ground-based inventory being available for other forested ecozones) (sections 2.2.1.1.1, *Land Cover* and 2.2.2.3, *Forests (Boreal)*). As well, no trend data are available, and inventory polygon ages do not accurately represent stand age.

Changing methodologies also renders it difficult to ascertain trends in some cases. Insufficient fire detection makes data from the Canadian Large Fire Database inaccurate for this area, until about the mid-1970s (Section 2.4.2.2, *Fire*). A more general issue, which applies to all ecozones, is that trends in COSEWIC status listings are difficult to interpret, because definitions and assessment criteria have changed, since the Species at Risk Act (SARA) was introduced (Section 2.3.2, *Trends in Species of National Conservation Concern*).

Much more of the data and information that is available for the Hudson Plains Ecozone comes from smaller-scale data collection efforts that are not more broadly coordinated. Most of this type of information is associated with individual jurisdictional governments or independent bodies (e.g., academia, industry). These types of studies provide most of the population information available for individual species, and they contribute information about the types of impacts occurring and for which at least part of the ecozone is vulnerable. In some cases, they are also able to contribute more comprehensive data (e.g., local avifauna studies in the Churchill area and lichen surveys in Wapusk National Park; sections 2.3.3.3, *Birds* and 2.3.3.8, *Lichens*) or higher resolution data (e.g., Landsat imagery of damaged coastal areas; Section 2.2.2.1, *Coastal*).

On the other hand, smaller-scale studies of this type are often not repeated or carried out comparably across component areas or jurisdictions of the ecozone (e.g., regional land cover classifications differ; Section 2.2.1.1.1, *Land Cover*). Exceptions are certain cooperative studies, such as the 2008-2009 Manitoba-Ontario surveys of the forest-tundra ecotype of woodland caribou, which were coordinated across those provincial governments (Section 2.3.3.2.1, *Caribou*) and the Ontario-Québec waterfowl surveys that have been included, since 1990, in the joint United States Fish and Wildlife Service-Canadian Wildlife Service Waterfowl Breeding Population and Habitat Survey (Section 2.3.3.3, *Birds*). Many smaller-scale studies also concentrate on affected areas (e.g., most study of the ecozone's aquatic ecosystems has been done in relation to hydroelectric development; sections 2.2.2.4.2, *Rivers/Streams & Lakes* and 2.3.3.4, *Fish*). In other cases, such as for ecosystem processes/functions like carbon cycling, scaling local ecosystem variability up to resolutions acceptable for regional-level models is inherently difficult and requires much more work (Section 2.4.4, *Carbon Cycling*). For all of these reasons, smaller-scale studies that are not coordinated (for ease of roll-up) across the broader landscape often cannot be easily extrapolated to the ecozone scale. Notwithstanding, some smaller-scale data collection efforts that are not more broadly coordinated *are* relatively comparable and, where data from them are not already being combined centrally (e.g., protected areas via the Conservation Areas Reporting and Tracking System of the Canadian Council on Ecological Areas; Section 2.6.2.1), they can be readily compiled for assessment purposes. Examples include the provincial furbearer harvests (sections 2.3.3.2.3, *Furbearers* and 2.5.3.2, *Furs from Wildlife Harvest*) and some of the bird data that represent a mixture of programs undertaken by various agencies (Section 2.3.3.3, *Birds*). Still, jurisdictional or other smaller-scale data sets that are not part of more broadly coordinated programs can sometimes be difficult to access, and the same types of spatial and temporal sampling and data quality issues can apply, as described above for broader-scale studies. For example, the jurisdictional furbearer harvest records for the ecozone are based on mandatory fur dealer reports or official sealing records and do not take into account harvests for domestic and/or cultural uses; as such, they do not provide estimates of total harvest. Another example is the recreational angling surveys in the Ontario portion of the ecozone that provide unreliable estimates due to low survey response and probable bias for some areas (Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Overall, although major knowledge gaps exist for most topics examined in this assessment, perhaps the most pronounced gaps evidenced for permafrost, freshwater ice, carbon cycling, peatland hydrology, insect disturbances, forest diseases, pollination, and populations of reptiles, amphibians, and invertebrates. Commentary on the prioritization of the fuller suite of knowledge gaps (in a management context) is provided in Section 3.4, *Emerging Issues & Related Information Needs*.

Although not outwardly evident in this assessment, the state of knowledge about the Hudson Plains Ecozone is currently in a state of flux. Collection of new information is being driven there by increasing interests in both climate change and major economic development. Results from the last phases of some major research programs with components relevant to the Hudson Plains Ecozone (e.g., Arcticnet[90], International Polar Year[91]) are becoming available and more should be available within a few years. In addition, much new inventory, monitoring, and research is being generated for jurisdictionally specific portions of the ecozone in association with, for example, Manitoba's Biodiversity Conservation program[92], Wapusk National Park[93], and Ontario's Far North Land Use Planning Initiative[94]. New or updated information is also being generated from non-government initiatives, such as the in-progress breeding bird atlases in Manitoba and Québec, which are similar to atlas work in Ontario (see Section 2.3.3.3, *Birds*). Monitoring also continues in areas affected by hydroelectric development in Québec, particularly those affected by the recent Rupert River diversion (Section 2.3.3.4, *Fish*).

Notwithstanding, much work remains to be done, and monitoring systems in particular are still inadequate for supporting future trends assessment. Future needs for status and trends assessment, including identification of thresholds and rapid or unexpected changes, centre around long-term (sustained) monitoring at ecologically relevant scales (ATK can be an important component of long-term monitoring), standardized across jurisdictions, where needed, and with consideration to interactions between and among stressors and cumulative impacts. Research remains critical for identifying mechanistic causes of change (cause and effect) and, thus, for informing adaptive management.

3.3 Assessment of ecozone health

A review of the available environmental and ecological information for the Hudson Plains Ecozone revealed a number of important changes or trends occurring there, despite the ecozone's small human population, comparatively little development, and poor information base. Many of the changes and trends that were identified through the review of individual ecosystem attributes are interrelated or associated with similar human influences. As such, they are summarized below in an integrated way, in context of their associated or probable principal underlying causes (recognizing that association does not necessarily equate with causation): 1) climate change; 2) excessive goose foraging; 3) resource developments; and 4) other causes (includes changes associated with additional factors, and those less clearly associated with any specific factor). Commentary on overall ecozone health is then provided.

3.3.1 Contextual analysis – interrelated changes & trends

3.3.1.1 Early climate change effects

A climate change signal is evident in the Hudson Plains Ecozone, despite its limited network of climate stations (Section 2.1, *Abiotic Drivers*), and some related ecological effects are apparent (this report). Some climatic trends over the period 1950-2007 are statistically significant:

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[90] http://www.arcticnet.ulaval.ca

[91] http://classic.ipy.org/about/index.htm

[92] http://www.gov.mb.ca/conservation/wildlife/biodiv/biodiversity.html

[93] Parks Canada. 2007. Annual Report of Research and Monitoring in Wapusk National Park 2007-2008. Wapusk National Park, Churchill, MB. 39 pp.

[94] http://www.mnr.gov.on.ca/en/Business/FarNorth/index.html?CSB_ic-name=specialInitiatives&CSB_ic-info=farNorthOntario_Eng

increased mean annual and mean seasonal temperature (winter and/or summer); increased number of effective growing degree-days; decreased total spring precipitation; decreased number of winter or spring days with measurable precipitation; and decreased annual snow to total precipitation ratio, depending on location. Climatic trends are also evident in the broader Hudson Bay region and, importantly, the recent warming cannot be solely attributed to large-scale climate oscillations or changes in oceanic circulation. Large-scale climate oscillations and, in particular, a positive phase of the Arctic Oscillation, were correlated over the period 1964 to 2000 or 2003 with trends for reduced precipitation and an 11-13% decline in the total annual volume of water naturally discharged by rivers in eastern and western Hudson Bay – also associated with a 4 day advance in annual peak discharge rate and a decline in peak intensity.

First-order effects of climate change are manifest in the broader marine arctic as rapidly deteriorating sea ice conditions, ahead of the projected rate of the summer retreat of the ice by nearly all of the GCMs used by the Arctic Climate Impact Assessment and the fourth assessment of the Intergovernmental Panel on Climate Change (Section 2.1, *Abiotic Drivers*). On average, the period of ice cover in western Hudson Bay, southern Hudson Bay, and James Bay (areas directly adjacent to the Hudson Plains Ecozone) has shortened by ~3 weeks, since the mid-1970s.

The shortening sea ice season in Hudson and James bays is, in turn, correlated with deteriorations in the polar bear subpopulations that use this sea ice as habitat in winter (for hunting and feeding on seals) and the terrestrial environment of the Hudson Plains Ecozone during their off-ice season (Section 2.3.3.1.1, *Polar Bear*). The Western Hudson Bay subpopulation declined 22% from 1987 to 2004, along with indications of declining body condition and reduced survival rates in some age classes. The Southern Hudson Bay subpopulation was stable in abundance from the mid-1980s until last assessed in 2003-2005 but is showing significant declines in body condition, as well as evidence of declines in survival rates of all age and sex classes. The latter observations suggest that the Southern Hudson Bay subpopulation might be at an ecological tipping point or threshold and likely to decline in abundance in the future.

As the sea ice season shortens, predator-prey relationships are changing between polar bear and species such as seals and geese (sections 2.4.3.1, *Predator-Prey Relationships & Cycles* and 2.4.3.4.1, *Animal Phenology*), and unexpected interactions are being observed, such as dietary shifts in polar bears amplifying changes in their contaminant levels (Section 2.3.3.1.1, *Polar Bear*). Polar bears of the Western Hudson Bay subpopulation are feeding less on ice-associated bearded seal (which eats invertebrates) and more on open-water harbour and/or harp seals (which eat fish), although ice-associated ringed seals (primary prey) continue to form a relatively stable component of the polar bear's diet[95]. This dietary shift (documented for 1991-2007), in turn, appears to be affecting the rates at which some persistent organic pollutants (POPs) change in the bears: some legacy contaminants (e.g., the pesticide DDT, dichlorodiphenyltrichloroethane) appear to be declining slower than if the bear's diet had not changed, and some newer (emerging) contaminants (e.g., brominated flame retardants) are increasing at a faster rate. It is not clear from limited data (2001-2003) if similar dietary trends are occurring in the Southern Hudson Bay subpopulation (Section 2.3.3.1.1, *Polar Bear*).

Polar bear is also increasingly observed consuming goose eggs, moulting geese, and flightless goslings along the coast (Section 2.4.3.1, *Predator-Prey Relationships & Cycles)*. The earlier annual

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[95] This latter observation suggests that other seal species may not be sufficiently abundant or available to replace ringed seals in the polar bear diet.

transition of polar bear from sea ice onto land means that its arrival on shore is increasingly overlapping with the period when lesser snow goose is still incubating eggs – thereby providing early arriving polar bears with an exploitable and abundant food source not available to them in the past (Section 2.4.3.4.1, *Animal Phenology*)[96]. Although the early season phenology of lesser snow goose is also advancing, its mean hatching date is advancing at a rate slower than the break-up of sea ice and associated transition of polar bear onto land. Earlier hatching dates are also being observed for Canada goose (Section 2.4.3.4.1, *Animal Phenology*), which is likewise vulnerable to increased polar bear predation.

Climate warming has also been implicated in recent die-offs of fish in the lower Sutton and Albany rivers (brook trout, *Salvelinus fontinalis*; sucker, *Catostomus spp.*; whitefish, *Coregonus spp.*) in years with unusual summer heat waves (lower precipitation/reduced river flows, which are more closely linked to large-scale climate oscillations, might also have contributed) (sections 2.2.2.4.2, *Rivers/Streams & Lakes* and 2.3.3.4, *Fish*). The 2001 die-off of brook trout in the Sutton River was suggested to represent the first of an increasing number of die-offs of vulnerable anadromous stocks that will likely occur in the north, as climate change proceeds and rivers warm.

Other potentially early effects of climate change might be present in the ecozone but not detectable given its relatively poor information status (Section 3.2). It is not known if permafrost is thawing and wetlands are changing, or if the freshwater ice season for lakes and rivers is shortening, as is occurring elsewhere in Canada's north – but such changes are suspected[97] (Section 2.1, *Abiotic Drivers*). Suggested increases in primary productivity (base of food webs) are small compared to some other areas of northern Canada, based on remotely sensed trends (1985-2006) in Normalized-Difference Vegetation Index (a measure of gross primary photosynthesis and a proxy for green leaf area) (Section 2.4.4, *Carbon Cycling*). There is also no strong evidence of movement of the treeline north (Section 2.2.2.2, *Polar-Tundra*). However, the treeline is little studied, and some increase in shrub and tree cover is apparent just north of the functional treeline at Churchill. Modelling based on air temperature trends suggests that earlier budbreak (advancing spring phenology) of conifers is also likely occurring (Section 2.4.3.4.2, *Plant Phenology*). Any other climate-related changes in ecosystem processes, including carbon cycling, are not known, but they are also insufficiently studied (Section 2.4, *Ecosystem Functions/Processes*). Little direct evidence suggests that the natural disturbance regime is changing, albeit an analysis of indicators of extreme weather over the period 1950-2003 suggests some limited potential increases in temperature variability, as well as the number of days exceeding temperature maxima (Section 2.4.2, *Natural Disturbances*).

3.3.1.2 **Goose damage to coasts & tundra**

A somewhat anomalous finding for this relatively remote and undisturbed ecozone is that a large portion (~30%) of its coastal salt marsh habitat has been destroyed and additional areas damaged, since the 1970s (Section 2.2.2.1, *Coastal*). The damage is severe, cumulative, continuing, and widespread, extending from Manitoba to James Bay, including Akimiski Island. Ultimately,

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[96] This food source might partly offset the energy shortfall of polar bears and stabilize their subpopulations temporarily, but it is likely that other food sources, both terrestrial and marine, will need to play a role if polar bear is to meet its energy requirements.

[97] These changes are suspected based on scientific studies just outside of western and eastern ecozone boundaries, as well as Aboriginal knowledge and casual observations within the ecozone.

it is attributable to human influences outside of the ecozone that have caused the Mid-Continent population of the migratory and herbivorous lesser snow goose to quadruple over the last four decades (specifically, an increased supply of agricultural food on its wintering grounds and along migration routes, declining harvest rate, and the development of migration and wintering refuges). It is principally excessive foraging (grazing and grubbing) by these overabundant lesser snow geese, but also increasing Canada goose (*Branta canadensis*) breeding and moulting populations in the area, that is causing an apparent trophic cascade leading to an alternate stable state of bare, hypersaline sediment along much of the coast of the Hudson Plains Ecozone, from which recovery might take decades. Further, in some cases the geese have so drastically depleted their preferred salt marsh forage that they have moved farther inland to nest and feed in less desirable areas within freshwater marshes and fens in the tundra, with similarly devastating effects (Section 2.2.2.2, *Polar-Tundra*).

The effects of goose damage on the ecozone's coastal and tundra ecosystems are relatively well studied (sections 2.2.2.1, *Coastal*; 2.2.2.2, *Polar-Tundra*; 2.3.3.3, *Birds*; 2.3.3.5, *Reptiles & Amphibians*; 2.4.3.2, *Herbivore-Plant Interactions*; and 2.4.5, *Nutrient Cycling*). A range of ecosystem processes and/or services are severely negatively altered, including herbivore-plant interactions, vegetative succession, net above-ground primary productivity, soil retention, and nutrient cycling. Major changes in the affected coastal and tundra food webs are evident, including localized losses or serious reductions in invertebrates, plants, and local colonies or populations of birds, and probably also amphibians. Three of the four Canada goose populations (also herbivorous) that use the ecozone (Eastern Prairie, Mississippi Valley, and Southern James Bay) are affected locally in terms of reproductive success or nesting density by the growth of the lesser snow goose population. Insectivorous nesting birds are also affected, including savannah sparrow (*Passerculus sandwichensis*), semipalmated sandpiper, and yellow rail. The declines in local nesting densities of yellow rail (classified as Special Concern by COSEWIC) that are likely occurring in the Manitoba and Ontario portions of the ecozone are thought to be related to this habitat degradation.

3.3.1.3 Changes associated with resource developments

More localized ecological changes and trends are associated with a comparatively small number of hydroelectric and mining projects in the ecozone. The first hydroelectric development was established in the ecozone in 1961 (Abitibi River, Ontario) (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). Most other hydroelectric facilities in the ecozone, which are located in Manitoba and Québec), were constructed in the 1970s to early 1980s. The ecozone is now also affected by the 2009 Rupert River diversion in Québec. Overall, river channel fragmentation and/or flow regulation from hydroelectric structures and operations in and around the Hudson Plains Ecozone have strongly affected the Churchill and Nelson river systems in Manitoba, the Moose River system in Ontario, and the Eastmain and Rupert river systems in Québec (some other rivers are moderately affected) (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). As well, a large reservoir (Opinaca) was created within ecozone boundaries (Québec). The single mine (Victor) operating in the ecozone (west of Attawapiskat, Ontario) is a relatively recent development; it was constructed beginning in 2006, opened in 2008, and is expected to operate for at least 12 years (Section 1.2.2, *Economic History*).

Hydroelectric development is associated with many documented impacts on ecosystems and biota in the Hudson Plains Ecozone, including: strong alterations in river flow rates (reduced

~40%, 90%, 87%, and 48%[98], respectively, for the lower Churchill, Eastmain, Opinaca, and Rupert rivers) and other physical parameters of rivers (e.g., magnitude and timing of fluctuations); drying of river bed and bank areas and proximal wetlands, with associated changes in vegetation; shifts in fish and benthic macro-invertebrate community compositions, including strong declines in abundance and loss of dominance by lake sturgeon (accompanied by low recruitment)[99]; sedimentation and/or saltwater intrusion in estuaries; sharp deterioration of subtidal eelgrass beds in the near-shore coastal environment in eastern James Bay[100]; and flooding of terrestrial areas and mobilization and bioaccumulation of mercury in fish, following impoundment of the Opinaca reservoir in 1980 (sections 2.2.2.1, *Coastal*; 2.2.2.4.2, *Rivers/Streams & Lakes*; 2.3.3.4, *Fish*; and 2.3.3.7, *Vascular Plants (Eelgrass)*)[101]. Restoration of at least some habitat affected by hydroelectric development has been attempted (e.g., permanent rewatering of an 8 km^2 section of river bed along a 10 km reach of the Churchill River via weir installation), but mostly with mixed success (Section 2.6.3, *Restoration*). Also notable is that although the elevated fish mercury levels in the Opinaca reservoir gradually declined over time, they are forecast to increase again due to receipt of mercury exported from the recently impounded Eastmain-1 reservoir upstream, outside of ecozone boundaries (Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Mining impacts are limited to those associated with the single open-pit diamond mine (e.g., sections 2.2.2.4.1, *Wetlands (Freshwater)* and 2.2.2.4.2, *Rivers/Streams & Lakes*). There, an area that was principally fens and bogs, but also contained ponds, creeks, smaller rivers, and riverbank and creek margin forests, is now occupied by the direct, project-related developments (~28.8 km^2) of the mine. The potential area affected by the mine is, however, considerably larger than the mine itself. Dewatering operations are increasing annual recharge, possibly over an area as large as ~500 km^2, potentially altering water balance over the life of the mine (sections 2.2.2.4.1, *Wetlands (Freshwater)* and 2.4.6, *Hydrological Processes*). Groundwater withdrawals are being made from the Nayshkootayaow River for purposes of dewatering (then discharged to the Attawapiskat River), albeit water is being added to maintain seasonal low flows in the Nayshkootayaow River (Section 2.2.2.4.2, *Rivers/Streams & Lakes*). No published monitoring data are available, but a reclamation plan is in place. Although most of the disturbed area will be rehabilitated after mine closure (albeit not all to original ecological classifications), certain areas will not be restored. The new channel created after diversion of Granny Creek will become the permanent creek channel, and the open pit will be actively flooded to create a small lake with closed drainage.

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[98] 72% of the flow of the Rupert River was diverted, but lateral flow from tributaries increases the flow at the river mouth to ~48%.

[99] The declines in abundance of lake sturgeon in the lower Eastmain and Opinaca rivers (Québec) were also attributed, in part, to high harvest pressure promoted by new road access and increased ease of net fishing associated with the reduced river flow (Section 2.3.3.4.1, *Lake Sturgeon*).

[100] Reduced salinity during the major growing period (June and July) and increased duration of ice cover related to reduced salinity (hydroelectric development) were suggested as the major causes near the La Grande River, while wasting disease, climate change, and isostatic rebound were rejected as major causes.

[101] Hydroelectric developments do not constitute a new source of mercury, but hydrological disturbances like reservoir impoundment (flooding, particularly over organic soils) can accelerate the bacterial conversion of mercury to methylmercury, its more bioavailable form. Some new mercury is deposited from long-distance transport, but the amount of such deposition in subarctic ecosystems is comparatively low. Atmospheric acid deposition, which could increase the methylation of mercury, is not currently an issue in this geography, but the ecozone does have some acid-sensitive terrain.

3.3.1.4 **Changes associated with other factors**

Most ecological changes or trends reported for the Hudson Plains Ecozone that are not associated in some obvious way with climate change, goose damage, or resource developments (as above) relate to the distribution and abundance of populations of individual species. An exception is some of the habitat alteration that is occurring in the tundra in both Manitoba and Ontario. The damage, which affects both the wetter inter-ridge areas and drier beach ridges of the tundra, is associated with human access and more specifically the operation of wheeled vehicles (tundra buggies and ATVs) (sections 2.2.2.2, *Polar-Tundra*; 2.3.3.2.1, *Caribou*; and 2.3.3.7, *Vascular Plants*). The changes and trends in species populations that are less clearly associated with climate change or either goose- or development-related alterations of habitat are summarized below. In some cases, these observations have been attributed to other causes (specified, where available); in other cases, the cause(s) is(are) not known or reported.

Several mammals show some evidence of improved abundance and/or expansions in range or use within the ecozone. Wolverine, which was reduced by overexploitation during the fur trade era, has been reclaiming its historical range in the ecozone, since the 1970s, and 2003-2010 surveys suggest further expansion of this species east, along with some likely increase in its abundance (Section 2.3.3.2.3, *Furbearers*). A recent (2008) winter survey in the southern part of the ecozone in Ontario (Ontario Wildlife Management Unit 1D, James Bay Lowlands) also suggests that the forest-dwelling ecotype of woodland caribou increased there (from about 0.01 caribou/km^2 to 0.04 caribou/km^2), since a comparable survey was last done in the area, circa 1983-1984 (Section 2.3.3.2.1, *Caribou*). Since 2003, barren-ground grizzly bear (*Ursus arctos richardsoni*; COSEWIC Special Concern) is being increasingly sighted in the ecozone along the coast of Wapusk National Park, which is a few hundred kilometres south of its normal range (Section 2.3.2, *Trends in Species of National Conservation Concern*). The increased number of confirmed sightings could be the result of increased observational effort, but it is considered more likely to represent an expansion of the species' range into the ecozone.

Some migratory bird species also show evidence of increased abundance or expansions of range or use specifically within the ecozone, although changes in the broader populations of these migratory species are generally tied to conditions outside of the ecozone (Section 2.3.3.3, *Birds*). Since the 1970s, the number of discrete lesser snow goose colonies nesting in the ecozone has increased from three to six, along with much expansion of the original three colonies. As previously noted, these observations in the ecozone are related to the quadrupling, during the same period, of the broader Mid-Continent population to which these geese belong, which is in turn predominantly due to human influences outside of the ecozone. Also notable is that, within the last decade, both American white pelican (*Pelecanus erythrorhynchos*) and double-crested cormorant (*Phalacrocorax auritus*) established breeding in Akimiski Strait. The increase of double-crested cormorant in the ecozone is consistent with the exponential growth of the species in eastern North America, since the 1970s, but the reason for the easterly expansion of American white pelican is less clear. The bald eagle has increased, since the 1980s, both as a breeding species in the southern portion and as non-breeding birds along the coasts during summer, which is consistent with reduced use of pesticides outside the ecozone, since the 1970s, and increases in local abundance of goose prey populations (Section 2.3.3.3, *Birds*).

Two introduced and potentially invasive fish species are relatively new to the ecozone (Section 2.2.2.4.2, *Rivers/Streams & Lakes*): rainbow smelt (*Osmerus mordax*) was found in the lower Nelson and Churchill rivers in 1998-2002 (but its current status in the Churchill River is unclear), and smallmouth bass (*Micropterus dolomieu*) was found in the Moose and Albany rivers in 2008-2009.

In terms of species showing evidence of possible deterioration in the ecozone, the Pen Islands (aka Hudson Bay Coastal Lowland) herd of migratory forest-tundra ecotype of woodland caribou shows an eastern shift in its summer use of coastal areas and possible population decline, following a substantial increase in its numbers between 1979 and 1994 (Section 2.3.3.2.1, *Caribou*)[102]. Surveys, since 2000, and as recently as 2008-2009, show these animals have been shifting east. They are no longer found in large aggregations in their traditional area near the Manitoba-Ontario border, but rather most (>80%) are now found near Cape Henrietta Maria. No cause-and-effect relationship has been ascribed, but multiple factors could be responsible, including: deterioration of range condition in the Pen Islands area, leading to decreased food availability; increased predator densities; disturbance; and harvest. However, the learned avoidance by caribou of areas of high harvest pressure and high disturbance and/or harvest are suspected to be contributing. In the traditional summer use area near the Manitoba-Ontario border, ATV travel on coastal ridge habitats has increased and unusually high harvests occasionally occur; as well, a winter road between Shamattawa and Fort Severn now also bisects their winter range (Section 2.3.3.2.1, *Caribou*).

Some migratory bird species also show deteriorations in local, seasonal population or colony abundance or distribution specifically within the Hudson Plains Ecozone, although again changes in the broader populations of these migratory species are generally tied to conditions outside of the ecozone (Section 2.3.3.3, *Birds*). Notable are the widespread declines in semipalmated sandpiper (*Calidris pusilla*), a species that used to be abundant in the ecozone (ca. 1940-1970) but by 2004-2005 could no longer be found breeding (Churchill) or was scarce (Cape Henrietta Maria). Although these local declines might in some cases be augmented by goose habitat damage, the widespread decline is more likely related to conditions outside of the breeding grounds. Ross's gull (*Rhodostethia rosea*) (classified as Threatened by COSEWIC) and breeding whimbrel (*Numenius phaeopus*) are also both thought to have declined in the Manitoba portion of the ecozone. Some local declines in other bird species, including yellow rail (Special Concern), appear to be linked to excessive goose habitat damage, as noted above (Section 3.3.1.2, *Goose Damage to Coasts & Tundra*). Finally, the sharp decline in eelgrass beds along the eastern James Bay coast (sections 2.3.3.7, *Vascular Plants (Eelgrass)* and 3.3.1.3, *Changes Associated with Resource Developments*) might suggest a redistribution of Atlantic brant within the ecozone over the past two decades, given that its entire population stages there and relies on the eelgrass beds and salt marshes during spring and fall migrations (Section 2.3.3.3, *Birds*).

In closing this section, it is perhaps noteworthy that harvest is not often identified as a known or potential cause of changes and trends in the ecozone's fish and wildlife populations (disregarding residual effects from the historical over-harvest of several mammal species during the fur trade era and the commercial fishing era for lake sturgeon). The ecozone's resident population of Aboriginal peoples is small, and comparatively little regulated or licensed harvest by non-Aboriginal peoples occurs. However, harvest of any kind also tends to be incompletely or episodically tracked in this ecozone (sections 2.3.3, *Trends in Species of Special Interest* and 2.5.3, *Provisioning Services*), and, as noted, some concerns do exist that high harvest pressure

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[102] Some migratory caribou herds that only peripherally extend into the Hudson Plains Ecozone (and are probably only minimally influenced by it) have been declining in recent years: the barren-ground Qamanirjuaq herd at the western periphery and the George River herd at the eastern periphery.

might have contributed to population declines in lake sturgeon in the lower Eastmain and Opinaca rivers post-development (sections 2.3.3.4.1, *Lake Sturgeon* and 3.3.1.3, *Changes Associated with Resource Developments*) and the apparent range shift of the Pen Islands herd of migratory woodland caribou in the northern part of the ecozone (as above and see also Section 2.3.3.2.1, *Caribou*). Localized high fishing pressure (i.e., near or exceeding sustained levels) was also reported in the past for lake sturgeon in some rivers, as well as other fish species in popular angling areas, including Hawley Lake (lake trout) and the Sutton, Brant, Shagamu, and Gorge rivers (brook trout); however, insufficient follow-up monitoring renders the present situation unclear (sections 2.2.2.4.2, *Rivers/Streams & Lakes* and 2.3.3.4, *Fish*). Provisioning species in general might be subject to localized high harvest pressure in the vicinity of population centres and high recreational use areas.

3.3.2 **Ecosystem health**

The ecosystem focus of the Ecosystems Status and Trends Report (ESTR) project complements Canada's historical focus on species diversity, and it is intended to help assess the goal of the Biodiversity Outcomes Framework for "healthy and diverse ecosystems"[103,104]. This objective is addressed at a national level in other, linked reports[105,106], while the condition of individual ecozones is assessed more directly through supporting technical reports, including this report for the Hudson Plains Ecozone.

Ecosystem health has generally been difficult to measure quantitatively[107], with few indicator frameworks developed for this purpose[108]. It requires extensive biophysical information, as well as related information on economics and human health, and its integration. This challenge is compounded in ESTR, because extensive information gaps for biophysical factors exist for many ecozones across Canada, and because it was not possible to consider economics and human health other than through a relatively cursory examination of selected ecosystem services (see the introduction to Section 2.5, *Ecosystem Services*). As such, while all factors examined are relevant, assessment of ecosystem health in ESTR is weighted heavily toward biophysical components, and it is not based on an indicator framework that allows quantification as a specific measure(s). Rather, ecosystem health is necessarily examined more subjectively, including the link between biodiversity and human well-being, which is largely implied.

Overall, despite many knowledge gaps, the information compiled for this assessment suggests that, at present, the Hudson Plains Ecozone is comparatively very healthy overall[109], which is in

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[103] Environment Canada. 2006. Biodiversity Outcomes Framework for Canada. Canadian Councils of Resource Ministers, Ottawa, ON. 8 pp. Available at: http://www.biodivcanada.ca/default.asp?lang=En&n=F14D37B9-1

[104] See also the Preface to this report.

[105] Environment Canada. 2009. Canada's 4th National Report to the United Nations Convention on Biological Diversity. 185 pp.

[106] Federal, Provincial and Territorial Governments of Canada. 2010. Canadian Biodiversity: Ecosystem Status and Trends 2010. Canadian Councils of Resource Ministers, Ottawa, ON. vi + 142 pp.

[107] A healthy ecosystem is one that is stable and sustainable and maintains its organization and autonomy over time, as well as its resilience to stress (see Costanza, R. 1992. Toward an operational definition of ecosystem health. pp 239-256 *in* Ecosystem Health: New Goals for Environmental Management. *Edited by* R. Costanza, B.G. Norton and B.D. Haskell. Island Press, Washington DC. 269 pp.).

[108] Rapport, D.J., Costanza, R. and McMichael, A.J. 1998. Assessing ecosystem health. Trends in Ecology and Evolution: 13: 397-402.

[109] cf. Federal, Provincial and Territorial Governments of Canada. 2010. Canadian Biodiversity: Ecosystem Status and Trends 2010. Canadian Councils of Resource Ministers, Ottawa, ON. vi + 142 pp.

contrast to the decline in biodiversity that is continuing nationally[110] and globally[111,112]. This still sparsely populated ecozone has experienced a very limited amount of ecosystem conversion, anthropogenic fragmentation, and other human influences (sections 2.2, *Ecozone Structure* and 2.6.1.1, *Summary of Stressors*). It is the most intact or least anthropogenically fragmented of all of the forested ecozones in Canada, based on a 2006 analysis that reported 97% of its area in intact landscape patches of more than 10,000 ha (see Section 2.2.1.2.2, *Landscape Fragmentation*). It still has many intact waterways (Section 2.2.2.4.2, *Rivers/Streams & Lakes*) and expansive tracts of intact landscapes (Section 2.2.1.2, *Land Cover Change*) that include internationally renowned wetlands (sections 2.2.2.1, *Coastal* and 2.2.2.4.1, *Wetlands* (*Freshwater*)). Such habitat provides important refuge areas for a number of species of national conservation concern (Section 2.3, *Ecosystem Composition*). No loss of large predators has occurred (top of food webs), and no evidence otherwise exists in this ecozone for a decline in species diversity[113] or reductions in species' ranges (disregarding the historical fur trade era) (Section 2.3, *Ecosystem Composition*). The ecozone's disturbance regime (outside of human settlements) remains effectively natural and apparently stable (Section 2.4.2, *Natural Disturbances*), and little evidence exists for broad-scale changes in primary production (base of food webs) (Section 2.4.4, *Carbon Cycling*). Introduced (potentially invasive) species and point sources of pollution are few (sections 2.6.1.1, *Summary of Stressors* and 2.3, *Ecosystem Composition*). Rather, most pollution (though still limited) arrives in the ecozone from long-range transported sources.

The proportion of the ecozone's landbase in protected areas is comparatively high (12.8% in 2010, or 11.7% in 2009, when Canada's ecozones were compared[114]), even if some representation and connectivity gaps remain (Section 2.6.2.1, *Protected Areas*). A number of stewardship/conservation initiatives are also in place, many in the form of co-management agreements among resident First Nations and other levels of government (Section 2.6.2, *Stewardship/Conservation*). As well, some disturbed or contaminated areas are or have been the focus of restoration efforts (Section 2.6.3, *Restoration*). No compelling evidence exists at this time to suggest broad-scale reductions in the capacity of the ecozone to provide people with ecosystem services (except in severely goose-damaged coastal areas) (Section 2.5, *Ecosystem Services*); the ecozone still well supports the Aboriginal subsistence way of life, even if the percentage of families practicing the traditional subsistence lifestyle on the land is declining due to influences associated with modernization (sections 2.5.3, *Provisioning Services* and 2.5.4, *Cultural Services*). Tourism and recreational potential also remain high, even if such activities are limited at present (sections 1.2.2, *Economic History* and 2.5.4, *Cultural Services*).

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[110] Environment Canada. 2009. Canada's 4th National Report to the United Nations Convention on Biological Diversity. 185 pp.

[111] Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N. et al. 2010. Global biodiversity: indicators of recent declines. Science 328: 1164-1168.

[112] SCBD (Secretariat of the Convention on Biological Diversity). 2010. Global Biodiversity Outlook 3. Secretariat of the Convention on Biological Diversity, Montréal, QC. 94 pp.

[113] The loss or suspected loss of two migratory bird species that formerly used the Hudson Plains Ecozone is not attributable to conditions in the ecozone (Section 2.3.3.3, *Birds*). The passenger pigeon (*Ectopistes migratorius*) has been extinct since 1914. The Eskimo curlew (*Numenius borealis*), which is currently assessed by COSEWIC as Endangered, is thought to be extinct by some.

[114] Federal, Provincial and Territorial Governments of Canada. 2010. Canadian Biodiversity: Ecosystem Status and Trends 2010. Canadian Councils of Resource Ministers, Ottawa, ON. vi + 142 pp. The percent protected areas in the Hudson Plains Ecozone was exceeded only by Pacific Maritime, Boreal Cordillera, and Montane Cordillera ecozones, all in western Canada.

Notwithstanding all of the above indications of healthy landscapes, habitats, and species, changes and trends are evident in certain biomes, biotic populations and communities, and ecological processes/functions of the ecozone (see summaries in sections 2.6.1.1, *Summary of Stressors* and 3.3.1, *Contextual Analysis–Interrelated Changes & Trends*). Not all such changes and trends are necessarily either permanent or long-term (little is known of natural ranges of spatial and temporal variability and the capacity to recover), but they do suggest that the health of this still highly intact ecozone has some vulnerabilities to the major threats identified, i.e., climate change and threats related to development and to excessive goose damage. Such vulnerability is of management interest, particularly given that climate change and development are forecast not only to continue but to accelerate and interact (see below, Section 3.4, *Emerging Issues & Related Information Needs*).

To better gauge which, if any, of the changes and trends known to be occurring in the ecozone are most likely to lead to a decline in ecosystem health (versus being more innocuous fluxes within the natural range of variability), it may be helpful to consider ecosystem health in a narrower biophysical context. Three components are usually of principal interest, because they relate to the most essential functions and key attributes of stable and sustainable ecosystems[115,116]: 1) vigour (activity, metabolism, or primary productivity); 2) organization (diversity and number of interactions among ecosystem components); and 3) resilience (a system's capacity to maintain structure and function in the presence of stress, rather than converting to an alternate state). These components are also embedded in the first of the two desired conservation and use outcomes of the healthy and diverse ecosystems goal of the Biodiversity Outcomes Framework[117]: "productive, resilient, diverse ecosystems with the capacity to recover and adapt".

An assessment of the known[118] changes and trends in Hudson Plains Ecozone in context of these three principal biophysical components leads to the same basic conclusion (i.e., that the ecozone appears comparatively very healthy overall) – except that the changes and trends that are most likely leading to a decline in ecozone health are more clearly evident. The ecozone shows little evidence of a change in vigour at a broad scale, with only small potential increases suggested in primary productivity (Section 2.4.4, *Carbon Cycling*) and tree and shrub cover above the treeline (Section 2.2.2.2, *Polar-Tundra*). The ecozone has also mostly retained its organization (no loss of top predators or other evidence of a broad-scale reduction in diversity or incomplete food webs, i.e., the full complement of native species is present[119]; Section 2.3, *Ecosystem Composition*), as well as its resilience – with a few exceptions. That organization is not being fully maintained is suggested by changed predator-prey interactions involving polar bear (sections 2.4.3.1,

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[115] Costanza, R. 1992. Toward an operational definition of ecosystem health. pp 239-256 *in* Ecosystem Health: New Goals for Environmental Management. *Edited by* R. Costanza, B.G. Norton and B.D. Haskell. Island Press, Washington DC. 269 pp.

[116] Rapport, D.J., Costanza, R. and McMichael, A.J. 1998. Assessing ecosystem health. Trends in Ecology and Evolution: 13: 397-402.

[117] Environment Canada. 2006. Biodiversity Outcomes Framework for Canada. Canadian Councils of Resource Ministers, Ottawa, ON. 8 pp.

[118] The absence of scientific information about additional changes (whether from poor detection or insufficient study) does not mean that other changes are not also occurring.

[119] Again, exceptions are two migratory bird species that formerly seasonally used the ecozone, as part of much broader ranges (Section 2.3.3.3, *Birds*). Passenger pigeon (*Ectopistes migratorius*) became extinct in the early 20th century. Eskimo curlew (*Numenius borealis*) is still classified as Endangered by COSEWIC, but it, too, might now be extinct. The demise of both species is linked to anthropogenic influences outside of the Hudson Plains Ecozone.

Predator-Prey Relationships & Cycles and 2.4.3.4.1, *Animal Phenology*). These fundamental changes in relationships among species, and associated changes in animal phenology, are likely not reversible (given that climate change itself is not reversible), although they may continue to change. Loss of resilience is not evident at an ecozone scale, but it is evident across the ecozone's coastal biome, where salt marshes have been so severely damaged by excessive goose foraging that they have flipped to an alternative stable state of hypersaline bare sediment (Section 2.2.2.1, *Coastal*). This loss of resilience in coastal salt marshes is associated with documented losses in their vigour (declines in net above-ground primary productivity and nitrogen fixation) and changes in their organization (loss or serious reduction of important components of the food web, including invertebrates, plants, and local colonies or populations of birds) (sections 2.2.2.1, *Coastal*; 2.4.3.2, *Herbivore-Plant Interactions;* and 2.4.5, *Nutrient Cycling*).

The suite of changes or trends in land cover and rivers that are associated with hydroelectric development and mining (sections 2.2.2, *Changes in Extent & Quality of Important Biomes or Realms*; 2.6.1.1, *Summary of Stressor*s; and 3.3.1.3, *Changes Associated with Resource Developments*) falls less completely or conclusively within the three biophysical components that relate to the most essential functions and key attributes of stable and sustainable ecosystems. Exceptions are cases where complete ecosystem conversion of areas that were formerly mostly wetlands and forests has occurred, e.g., creation of the Opinaca reservoir in 1980 and establishment of the open pit at the Victor mine, which will be actively flooded after mine closure to create a small lake with closed drainage[120]. Otherwise, neither hydroelectric development nor mining is generally reported to have changed ecosystem organization to the extent that either the number of species or the number of interactions among ecosystem components has declined (Section 2.3, *Ecosystem Composition*) – with the caveat that strong declines and loss of dominance of lake sturgeon have been reported among fish communities in some rivers affected by hydroelectric development, and the ultimate fate of these populations is not entirely clear (sections 2.2.2.4.2, *Rivers/Streams & Lakes* and 2.3.3.4.1, *Lake Sturgeon*).

Finally, it is noteworthy that the second desired conservation and use outcome of the healthy and diverse ecosystems goal of the Biodiversity Outcomes Framework[121] is that damaged ecosystems are restored. Again, it can in general be difficult to evaluate to what extent all changes observed in ecosystems constitute damage. That said, two ecosystems associated with the Hudson Plains Ecozone are clearly damaged and without effective restoration: 1) sea ice habitat in the interfacing marine ecozone (Hudson Bay, James Bay, and Foxe Basin), which co-supports species of the Hudson Plains Ecozone (e.g., polar bear, arctic fox) and strongly influences its climate; and 2) coastal salt marshes in the Hudson Plains Ecozone that are so severely damaged that they have flipped to an alternate stable state of hypersaline bare sediment. As already noted, reversal of the climate-driven trend for loss of sea ice is not likely, either naturally or through restoration efforts (no restoration attempts made), given that the source of stressor (global climate change) is not being removed. Similarly, no organized management attempt is being made to restore severely goose damaged areas, although experimental trials show success in areas protected from goose foraging (Section 2.6.3, *Restoration*). Meantime, inter-jurisdictional (United States-Canada) wildlife management

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[120] Although development and operation of the mine has converted other localized areas into biologically unproductive zones (e.g., mineral stockpiles), a reclamation plan is in place that will see such areas restored back into production following the mine's ~12 year lifespan, even if not all to original ecological classification.

[121] Environment Canada. 2006. Biodiversity Outcomes Framework for Canada. Canadian Councils of Resource Ministers, Ottawa, ON. 8 pp.

efforts are attempting to reduce the stressor (the size of the Mid-Continent population of lesser snow goose), but with limited success to date. Both goose population control (reduced foraging pressure) and assisted revegetation (restoration) are required for the short-term recovery of these coastal salt marshes. Even in the absence of continued foraging pressure, unassisted revegetation requires consolidated sediments and the establishment of plant assemblages in fresh unconsolidated sediments, which might occur on an unknown time scale, likely decades (Section 2.6.3, *Restoration*). Interestingly, near-term model forecasting (25 years into the future) suggests that the recent increase in the importance of polar bear as a predator of snow goose eggs (associated with climate change, as above) will reduce (but not eliminate) the size and reproductive output of some local nesting colonies of lesser snow goose along portions of the coast (e.g., Cape Churchill Peninsula) and, thus, aid management attempts to lower snow goose abundance there (sections 2.4.3.2, *Herbivore-Plant Interactions* and 2.4.3.4.1, *Animal Phenology*)[122].

A variety of restoration efforts have been undertaken, are in progress, or are planned, in other areas of the ecozone, where anthropogenic influences have led to some manner of ecosystem change (Section 2.6.3, *Restoration*). Most such restoration work is or has been concentrated in large projects, namely clean-up of the former Mid-Canada Line radar sites in Ontario (in-progress, and includes sites in Polar Bear Provincial Park) and restoration of areas affected by hydroelectric development (mixed success), including the reduced flow portions of rivers and tributaries (Churchill, Eastmain, Opinaca). As already noted, a reclamation plan is in place for future restoration of the Victor mine site, even if not all affected areas will be restored to their original ecological classification. Smaller-scale restoration projects also exist.

In summary, the Hudson Plains Ecozone can be considered comparatively very healthy overall, with the exceptions noted above (particularly in coastal salt marshes), and some of the changes or trends that have occurred in the ecozone, since the 1970s (especially those linked to climatic warming), may signal an imminent overall decline in ecosystem health.

3.4 **Emerging issues & related information needs**

The greatest emerging issues facing the Hudson Plains Ecozone are accelerated (human-induced) climate change and increasing pressure for additional resource- and transportation-based developments. Slowing the rate of climate change is essential, but it is a global issue, requiring major adjustments in the way people live and work, i.e., it can be difficult to address with only local management actions. At a more local level (and to the extent possible), peatland carbon stores can be maintained, climate change otherwise mitigated, and connectivity and biodiversity preserved, as defences against the complexities and uncertainties about how this ecozone's ecosystems and species will respond. Development (including that which affects peatland carbon stores) is under stronger local management control than climate change, though its effects will interact with climate change and not always in predictable ways. The extent to

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[122] As discussed in Section 2.4.3.2 (*Herbivore-Plant Interactions*), lesser snow goose has been largely escaping local density-dependent processes, because 1) it is dispersing to other coastal salt marsh sites and moving inland to forage on less nutritious vegetation in the tundra during the post-hatch (brood-rearing) period; and 2) a sustained increase in snow goose predation by traditional predators has not occurred, because geese are only available to them seasonally. There is, however, some indication of an increased frequency of renal coccidiosis and in loads of two parasitic nematodes in geese in coastal versus inland areas (Section 2.4.3.3.2, *Renal Coccidiosis & Parasitic Nematodes*).

which climate change affects this ecozone will, therefore, depend in part on how the ecozone is developed, used, and serviced.

Inventory and other baseline data and tracking are now critically needed to support and monitor the land use and environmental conservation planning and related policy and management decisions that need to be made, given the pressures of climate change and likely future development.

3.4.1 **Future climate change**

The Hudson Plains Ecozone is one of Canada's climatically sensitive northern latitude ecozones that is expected to experience among the strongest impacts of climate change; amplified warming is expected based on ice-albedo feedback effects, which are associated with the loss of sea ice in the broader region (Section 2.1, *Abiotic Drivers*).

Climate projections forecast, by 2100, a substantial or complete loss of seasonal sea ice in marine areas adjacent to the ecozone and a ~50% or more reduction in continuous permafrost (and complete loss of permafrost that is currently discontinuous or in isolated patches) in the ecozone (Section 2.1.2.2, *Projected Changes*). Because the ecozone's defining climatic and edaphic conditions are a function of sea ice and permafrost, cascading effects on the ecozone's ecosystems and biota are expected. The specific changes that might come are highly uncertain, but they could include some rapid or unexpected changes. Already notable is the rapid shortening of the sea ice season (Section 2.1, *Abiotic Drivers*) and its link to deteriorating polar bear subpopulations (Section 2.3.3.1.1, *Polar Bear*) and unexpected dietary shifts and their outcomes (Section 3.3.1.1, *Early Climate Change Effects*). Future changes could also lead to conditions outside of the natural range of variability. For example, by 2100, the projected increase in future fire risk is expected to bring at least the southern portion of the area's burn rate towards the upper limit of its range of natural variability during the last ~7,000 years (Section 2.4.2.2, *Fire*).

Although species in the ecozone that are dependent on sea ice are at most immediate risk (e.g., Section 2.3.3.1.1, *Polar Bear*), habitat is also particularly at risk in other climate-sensitive areas of the ecozone, i.e., the coast, tundra, and freshwater wetlands (including peatlands) – ecological communities that are intricately tied to the permafrost and poor drainage that are characteristic of this ecozone (Section 2.2.2, *Changes in Extent & Quality of Important Biomes or Realms*). As well, longer sea ice-free seasons in Hudson and James bays would likely render the coast susceptible to increased storm surges (inundation) and wave action (coastal erosion), and possibly promote greater use of the deepwater shipping port at Churchill and improve the viability of other coastal access points (sections 2.2.2.1, *Coastal* and 2.4.1, *Coastal Building Processes*). Sea level rise is less of a concern, because it is not forecast to overcome the ecozone's especially high rate of isostatic rebound. As elsewhere, species' ranges and assemblages are in general expected to change (general movement north) in both freshwater and terrestrial environments (Section 2.1.2.2, *Projected Changes*), with consequences for species and food web interactions (e.g., see the *Climate Change & Future Range Extension* subheading in Section 2.2.2.4.2, *Rivers/Streams & Lakes*).

Many ecosystem functions/processes are also likely be impacted (Section 2.4). Notably, it is not clear how well the ecozone's massive expanse of peatlands will be able to continue storing carbon, but potential changes in carbon balance are of global concern for biodiversity and human well-being (sections 2.4.4, *Carbon Cycling* and 2.5.2.1, *Climate Regulation Services*). If carbon stored in the ecozone's peatlands is released to the atmosphere, the release could lead to a positive feedback to atmospheric greenhouse gases, which could be further exacerbated if large areas of dry peatlands burn. Disturbances are in general expected to increase, including an increase in the area burned (Section 2.4.2, *Natural Disturbances*).

The effects of climate change on the resilience of this ecozone and its capacity to continue supplying ecosystem services are uncertain – and, thus, so are the impacts on economic opportunities and human health. What is certain is that local Aboriginal peoples will be forced to adapt to the changed climate and altered ecosystem, because their culture and traditional and wage-based economies have been shaped by the local environment and are still tied very closely to the land (Section 2.5, *Ecosystem Services*). These peoples also mostly live near the coast, where increased storm surges and wave action are anticipated (as above).

Clearly, it is important to understand and model the climate-driven changes in this ecozone and monitor their ecological effects. To accomplish this, higher resolution climate modeling of Hudson Bay and the surrounding landscape is needed to clarify uncertainties in climate change projections. Such modeling depends on reliable inputs from a sufficient set of environmental monitoring stations, ideally established and maintained over time to provide data to detect, track, and model fluctuations and future trends in climate, freshwater ice conditions, permafrost, hydrology, and carbon flux (these factors are particularly important among knowledge gaps). Mapping biophysical variables and monitoring the changes in ecosystem productivity and land cover that are projected to occur (e.g., tundra and wetlands) are also dependent on improved inputs and usage of technologies such as satellite imagery.

Finally, climate change is expected to increasingly interact with and exacerbate other anthropogenic threats and stressors, including those associated with development, in both terrestrial and aquatic ecosystems (Section 2.6.1.2, *Cumulative Impacts*).

3.4.2 **Future development**

Although development is currently very limited in the Hudson Plains Ecozone, pressure for additional resource development there is mounting in mining, hydroelectric, transportation and, to a lesser extent, wind-farming sectors (sections 1.2.2, *Economic History*; 2.2.1.2, *Land Cover Change*; and 2.6.1, *Stressors & Cumulative Impacts*). Recent discovery of world-class chromite deposits inland, within the Ring of Fire mineral field of Ontario, especially portends major mining-related infrastructure in the area. Although likely to bring additional jobs to the ecozone's wage economy, the high potential for additional resource developments is of ecological concern, not only due to potential direct effects, but also because it drives the establishment of roads and other infrastructure (e.g., a winter road and a major transmission line were recently added to service the new Victor mine). Roads and other infrastructure contribute fragmentation to the landscape and facilitate further human access, use, and development, along with associated influences on ecosystems and biota (Section 2.6.1.2, *Cumulative Impacts*). For example, given likely development in the Ring of Fire area, future north-south fragmentation of the ecozone could conceivably occur from the adjoining of east-west developments – from the Ring of Fire east to the Victor mine and on to Attawapiskat at the coast. Additional transportation developments might also occur in the near future irrespective of future resource developments. Feasibility planning is in progress for an all-season road that would run along the western edge of the ecozone from Gillam to Churchill, Manitoba and beyond to Rankin Inlet, Nunavut. Likewise, a pre-feasibility study is in progress in Ontario to assess possible routes for an all-season road that would connect communities along the coast of James Bay with the provincial highway system in the south.

Cumulative impacts, especially those from roads and multiple hydroelectric developments in the Hudson Bay watershed, are a concern and need to be addressed effectively to help guide decisions about future development (Section 2.6.1.2, *Cumulative Impacts*). Plans to develop roads and other regional infrastructure to service projects and communities would ideally be coordinated so as to limit overall road densities and, thus, maintain a high degree of ecosystem connectivity, intactness, and resilience. Ample opportunity still exists to do this type of land use planning, as well as conservation planning, over much of this ecozone, in advance of major development.

All three provinces (MB, ON, QC) recently announced new initiatives that further address their stewardship responsibilities for this unique ecozone in Canada (Section 2.6.2.2, *Land Use & Ecosystem Management Initiatives*). In 2008, Ontario announced a comprehensive land use planning initiative for its northern lands (including the Hudson Plains Ecozone), which is being done in partnership with its Aboriginal peoples and is supported by legislation in the form of a Far North Act. The objective is to conserve the province's far north environment (including carbon storage and sequestration, and support for an expanded and interconnected network of protected areas), while allowing for some resource development that can provide an environmentally stable economic future and greater community prosperity. Both Manitoba and Québec also recently (2009) put forth intents for further stewardship of their respective far north lands (including the Hudson Plains Ecozone). Specifically, in 2009 the Government of Manitoba committed to develop a boreal peatlands stewardship strategy in co-operation with stakeholders and leading climate change non-governmental agencies (co-incident with its announcement of two new legally protected areas in the ecozone with significant carbon stores). Also in 2009, the Government of Québec committed to protect from industrial development at least 50% of the area covered by its Plan Nord, i.e., lands north of the 49th parallel. Also notable is the new (2010) federal-territorial (Nunavut)-Aboriginal Eeyou Marine Region Land Claims Agreement (EMRLCA), which applies to some islands located offshore of Québec in Hudson and James bays (Nunavut) that are not covered by the James Bay and Northern Québec Agreement. For these islands (including those that are part of the Hudson Plains Ecozone), EMRLCA addresses issues related to contaminated sites and protected areas, as well as wildlife harvesting and management.

Despite such enhanced management interest, new developments that do proceed in the Hudson Plains Ecozone could still potentially be doing so in the face of much uncertainty – given the present insufficiency of environmental and ecological information (Section 3.2, *Information Status*), as well as uncertainties due to climate change and how it might interact with anthropogenic stressors related to development.

While it is out of scope to attempt to prioritize inventory, monitoring, research, and ATK collection needs for the Hudson Plains Ecozone as part of this status and trends assessment, such prioritization would ideally be done within and across jurisdictions with a view to providing decision-support for sustainable use, i.e., informing policy, land use planning, and resource management needs. Such an exercise would identify critical knowledge gaps associated with defined management needs, giving due consideration to if and at what scale knowledge gaps identified in this assessment might currently be being filled, i.e., areas where investments have been made but for which results are not yet available. The latter should consider relevant programs that are in their final phases, such as ArcticNet and International Polar Year, as well as the substantive amount of work currently in progress through Manitoba's biodiversity program, Wapusk National Park, Ontario's Far North Land Use Planning Initiative, and other initiatives. In general, however, ecosystem components in the Hudson Plains Ecozone that are of particular management interest due to their susceptibility to impacts from anticipated changes in human imprint (climate change and industrial development) include: permafrost; hydrology; carbon cycling; coastal and tundra ecosystems; river and lake ecosystems; wetlands and bird populations; plant communities; and sensitive fish and wildlife species (e.g., lake sturgeon, lake trout, polar bear, woodland caribou, wolverine). The valuation of ecosystem services also requires advancement, so that non-market services (e.g., climate regulating services via peatland carbon storage) can be adequately considered in policy and management decisions.

3.5 **Conclusion**

The Hudson Plains Ecozone is one of Canada's ecozones with the least amount of human influence to date – it is relatively intact and has a full complement of native flora and fauna, which for the most part also have viable populations and ranges. As such, ample opportunity still exists there to conduct resource, land use, and environmental conservation planning with resident Aboriginal peoples in advance of major development, including the careful planning of roads and other infrastructure from which more human access, use, and development inevitably follows. On the other hand, the Hudson Plains Ecozone is also one of Canada's climatically sensitive northern latitude ecozones that is expected to experience among the strongest impacts of accelerated (human-induced) climate change. Furthermore, climate change is expected to increasingly interact with and exacerbate other anthropogenic threats and stressors, including those associated with resource development.

To meet stewardship responsibilities for this unique ecozone in Canada, future management approaches need to ensure that nationally and internationally significant features of the ecozone are maintained and that the ecosystem services it provides are not compromised (Section 3.1, *Uniqueness & Implied Stewardship Responsibility*). These ecosystem services include the ability of resident Aboriginal peoples to retain their cultural identity, values, and social relationships to each other and the land, as well as the climate regulation services afforded by the ecozone's large carbon stores that benefit all life on earth. A responsive, adaptive management framework, supported by a sustained commitment to the collection, management, and sharing of both scientific and Aboriginal information (e.g., to detect early warning signals and rapid changes before thresholds are crossed), will be key to the effective future management of this ecozone. This baseline report for 2010, which is in many ways as much about the state of knowledge as it is about status and trends, indicates that the environmental and ecological information that exists for this ecozone is strongly insufficient.

Appendix 1 – Provisional list of the bird species of the Hudson Plains Ecozone[123], their breeding status & their relative abundance

Source: Abraham, K.F. and Brady C. 2010. Provisional Checklist of Birds of the Hudson Bay Lowlands. Unpublished compendium of information from published checklists and unpublished databases and checklists. Ontario Ministry of Natural Resources, Peterborough, ON.

Taxonomic order follows the American Ornithologist's Union checklist 2008 (http://www.aou.org/checklist/north/).

Abbreviations for breeding status: **BR**, known breeding; **br**, suspected breeding; and **no evidence**, no evidence of breeding.

Abbreviations and definitions for relative abundance: **C**, common – found in moderate to large numbers and easily found in appropriate habitat at the right time of year; **F**, fairly common – found in small to moderate numbers and usually easy to find in appropriate habitat at the right time of year; **U**, uncommon — found in small numbers and usually, but not always, found with some effort in appropriate habitat at the right time of year; **R**, rare – occurs annually in very small numbers; not expected on any given day but may be found with extended effort over the course of the appropriate season(s); **A**, accidental – has occurred, but less than annually, in very small numbers or at great intervals; **H**, hypothetical – has been reported at least once but not substantiated; and **E**, extinct – extinct or probably extinct but formerly common.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Anatidae			
Anser albifrons	Greater white-fronted goose	No evidence	R
Chen caerulescens[a]	Snow goose	BR	C
Chen rossii	Ross's goose	BR	F
Branta bernicla	Atlantic brant	No evidence	C
Branta leucopsis	Barnacle goose	No evidence	A
Branta hutchinsii	Cackling goose	BR	R
Branta canadensis	Canada goose	BR	C
Cygnus olor	Mute swan	No evidence	A
Cygnus buccinator	Trumpeter swan	br	R
Cygnus columbianus	Tundra swan	BR	C
Aix sponsa	Wood duck	br	R
Anas strepera	Gadwall	BR	R
Anas penelope	Eurasian wigeon	No evidence	A
Anas americana	American wigeon	BR	F
Anas rubripes	American black duck	BR	C
Anas platyrhynchos	Mallard	BR	C

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[123] The Hudson Plains Ecozone is considered roughly the equivalent of the Hudson Bay Lowlands at the scale that the bird inventories have been done.

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Anas discors	Blue-winged teal	BR	F
Anas clypeata	Northern shoveler	BR	F
Anas acuta	Northern pintail	BR	C
Anas crecca	Green-winged teal	BR	C
Aythya valisineria	Canvasback	No evidence	R
Aythya americana	Redhead	br	R
Aythya collaris	Ring-necked duck	BR	U
Aythya marila	Greater scaup	BR	C
Aythya affinis	Lesser scaup	BR	C
Somateria spectabilis	King eider	BR	R
Somateria mollissima	Common eider	BR	C
Histrionicus histrionicus	Harlequin duck	No evidence	U
Melanitta perspicillata	Surf scoter	BR	F
Melanitta fusca	White-winged scoter	BR	U
Melanitta americana	Black scoter	BR	F
Clangula hyemalis	Long-tailed duck	BR	F
Bucephala albeola	Bufflehead	BR	F
Bucephala clangula	Common goldeneye	BR	C
Bucephala islandica	Barrow's goldeneye	No evidence	A
Mergellus albellus	Smew	No evidence	H
Lophodytes cucullatus	Hooded merganser	BR	F
Mergus merganser	Common merganser	BR	F
Mergus serrator	Red-breasted merganser	BR	F
Oxyura jamaicensis	Ruddy duck	br	R
Phasianidae			
Bonasa umbellus	Ruffed grouse	BR	F
Falcipennis canadensis	Spruce grouse	BR	F
Lagopus lagopus	Willow ptarmigan	BR	C
Lagopus mutus	Rock ptarmigan	No evidence	F
Tympanuchus phasianellus	Sharp-tailed grouse	BR	F
Gaviidae			
Gavia stellata	Red-throated loon	BR	F
Gavia arctica	Arctic loon	BR	R
Gavia pacifica	Pacific loon	BR	C
Gavia immer	Common loon	BR	F

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Gavia adamsii	Yellow-billed loon	No evidence	A
Podicipedidae			
Podilymbus podiceps	Pied-billed grebe	BR	U
Podiceps auritus	Horned grebe	BR	U
Podiceps grisegena	Red-necked grebe	No evidence	U
Podiceps nigricollis	Eared grebe	No evidence	U
Procellariidae			
Fulmarus glacialis	Northern fulmar	No evidence	R
Puffinus sp.	Shearwater sp.	No evidence	A
Hydrobatidae			
Oceanodroma leucorhoa	Leach's storm-petrel	No evidence	A
Sulidae			
Morus bassanus	Northern gannet	No evidence	A
Phalacrocoracidae			
Phalacrocorax auritus	Double-crested cormorant	BR	U
Pelecanidae			
Pelecanus erythrorynchos	American white pelican	No evidence	R
Ardeidae			
Botaurus lentiginosus	American bittern	BR	F
Ixobrychus exilis	Least bittern	No evidence	A
Ardea herodias	Great blue heron	BR	F
Ardea alba	Great egret	No evidence	A
Egretta caerulea	Little blue heron	No evidence	A
Egretta tricolor	Tricolored heron	No evidence	H
Bubulcus ibis	Cattle egret	No evidence	A
Butorides virescens	Green heron	No evidence	R
Nycticorax nycticorax	Black-crowned night-heron	No evidence	A
Cathartidae			
Cathartes aura	Turkey vulture	No evidence	A
Pandionidae			
Pandion haliaetus	Osprey	BR	F
Accipitridae			
Haliaeetus leucocephalus	Bald eagle	BR	F
Circus cyaneus	Northern harrier	BR	C
Accipiter striatus	Sharp-shinned hawk	BR	U
Accipiter cooperii	Cooper's hawk	No evidence	A

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Accipiter gentilis	Northern goshawk	BR	U
Buteo lineatus	Red-shouldered hawk	No evidence	A
Buteo platypterus	Broad-winged hawk	No evidence	A
Buteo swainsoni	Swainson's hawk	No evidence	A
Buteo jamaicensis	Red-tailed hawk	BR	F
Buteo lagopus	Rough-legged hawk	BR	C
Aquila chrysaetos	Golden eagle	BR	R
Falconidae			
Falco sparverius	American kestrel	BR	F
Falco columbarius	Merlin	BR	F
Falco rusticolus	Gyrfalcon	No evidence	U
Falco peregrinus	Peregrine falcon	No evidence	U
Falco mexicanus	Prairie falcon	No evidence	A
Rallidae			
Coturnicops noveboracensis	Yellow rail	BR	F
Rallus limicola	Virginia rail	br	R
Porzana carolina	Sora	BR	F
Galllinula galeata	Common gallinule	No evidence	A
Fulica americana	American coot	BR	U
Gruidae			
Grus canadensis	Sandhill crane	BR	F
Grus americana	Whooping crane	No evidence	A
Charadriidae			
Vanellus vanellus	Northern lapwing	No evidence	H
Pluvialis squatarola	Black-bellied plover	No evidence	C
Pluvialis dominica	American golden-plover	BR	C
Pluvialis fulva	Pacific golden-plover	BR	R
Charadrius semipalmatus	Semipalmated plover	BR	C
Charadrius melodus	Piping plover	No evidence	A
Charadrius vociferus	Killdeer	BR	C
Recurvirostridae			
Recurvirostra americana	American avocet	No evidence	A
Scolopacidae			
Xenus cinereus	Terek sandpiper	No evidence	H

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Actitis macularius	Spotted sandpiper	BR	C
Tringa solitaria	Solitary sandpiper	BR	F
Tringa incana	Wandering tattler	No evidence	A
Tringa melanoleuca	Greater yellowlegs	BR	C
Tringa semipalmata	Willet	No evidence	A
Tringa flavipes	Lesser yellowlegs	BR	C
Bartramia longicauda	Upland sandpiper	No evidence	A
Numenius borealis	Eskimo curlew	No evidence	E
Numenius phaeopus	Whimbrel	BR	C
Numenius americanus	Long-billed curlew	No evidence	A
Limosa haemastica	Hudsonian godwit	BR	C
Limosa lapponica	Bar-tailed godwit	No evidence	H
Limosa fedoa	Marbled godwit	BR	F
Arenaria interpres	Ruddy turnstone	No evidence	C
Calidris canutus	Red knot	No evidence	C
Calidris alba	Sanderling	No evidence	C
Calidris pusilla	Semipalmated sandpiper	BR	C
Calidris mauri	Western sandpiper	No evidence	R
Calidris minuta	Little stint	No evidence	A
Calidris minutilla	Least sandpiper	BR	C
Calidris fuscicollis	White-rumped sandpiper	No evidence	F
Calidris bairdii	Baird’s sandpiper	No evidence	U
Calidris melanotos	Pectoral sandpiper	BR	C
Calidris maritima	Purple sandpiper	br	R
Calidris alpina	Dunlin	BR	C
Calidris ferruginea	Curlew sandpiper	No evidence	A
Calidris himantopus	Stilt sandpiper	BR	F
Tryngites subruficollis	Buff-breasted sandpiper	No evidence	U
Philomachus pugnax	Ruff	No evidence	A
Limnodromus griseus	Short-billed dowitcher	BR	C
Limnodromus scolopaceus	Long-billed dowitcher	No evidence	F
Gallinago delicata	Wilson's snipe	BR	F
Scolopax minor	American woodcock	br	U
Phalaropus tricolor	Wilson’s phalarope	BR	F
Phalaropus lobatus	Red-necked phalarope	BR	F
Phalaropus fulicarius	Red phalarope	No evidence	U

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Laridae			
Rissa tridactyla	Black-legged kittiwake	No evidence	A
Pagophila eburnea	Ivory gull	No evidence	R
Xema sabini	Sabine’s gull	No evidence	R
Chroicocephalus philadelphia	Bonaparte’s gull	BR	C
Chroicocephalus ridibundus	Common black-headed gull	No evidence	A
Hydrocoloeus minutus	Little gull	BR	U
Rhodostethia rosea	Ross’s gull	BR	R
Leucophaeus articilla	Laughing gull	No evidence	A
Leucophaeus pipixcan	Franklin’s gull	No evidence	A
Larus canus	Mew gull	No evidence	A
Larus delawarensis	Ring-billed gull	BR	C
Larus californicus	California gull	No evidence	A
Larus argentatus	Herring gull	BR	C
Larus thayeri	Thayer’s gull	No evidence	R
Larus glaucoides	Iceland gull	No evidence	R
Larus fuscus	Lesser black-backed gull	No evidence	A
Larus glaucescens	Glaucous-winged gull	No evidence	A
Larus hyperboreus	Glaucous gull	BR	U
Larus marinus	Great black-backed gull	No evidence	R
Hydroprogne caspia	Caspian tern	BR	U
Childonias niger	Black tern	BR	F
Childonias leucopterus	White-winged tern	No evidence	A
Sterna hirundo	Common tern	BR	C
Sterna paradisaea	Artic tern	BR	C
Sterna forsteri	Forester’s tern	No evidence	A
Stercorariidae			
Stercorarius pomarinus	Pomarine jaeger	No evidence	U
Stercorarius parasiticus	Parasitic jaeger	BR	F
Stercorarius longicaudus	Long-tailed jaeger	No evidence	U
Alcidae			
Alle alle	Dovekie	No evidence	A
Uria lomvia	Thick-billed murre	No evidence	R
Cepphus grylle	Black guillemot	BR	U
Columbidae			
Columba livia	Rock pigeon	No evidence	A

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Patagioenas fasciata	Band-tailed pigeon	No evidence	A
Zenaida asiatica	White-winged dove	No evidence	A
Zenaida macroura	Mourning dove	No evidence	U
Ectopistes migratorius	Passenger pigeon	No evidence	E
Cuculidae			
Coccyzus americanus	Yellow-billed cuckoo	No evidence	A
Coccyzus erythropthalmus	Black-billed cuckoo	br	U
Strigidae			
Bubo virginianus	Great horned owl	BR	F
Bubo scandiacus	Snowy owl	br	F
Surnia ulula	Northern hawk owl	BR	U
Strix varia	Barred owl	No evidence	R
Strix nebulosa	Great gray owl	BR	U
Asio otus	Long-eared owl	BR	U
Asio flammeus	Short-eared owl	BR	F
Aegolius funereus	Boreal owl	BR	U
Aegolius acadicus	Northern saw-whet owl	br	U
Caprimulgidae			
Chordeiles minor	Common nighthawk	BR	F
Phalaenoptilus nuttallii	Common poorwill	No evidence	A
Caprimulgus vociferus	Eastern whip-poor-will	No evidence	A
Apodidae			
Chaetura pelagica	Chimney swift	No evidence	A
Trochilidae			
Archilochus colubris	Ruby-throated hummingbird	No evidence	U
Selasphorus rufus	Rufous hummingbird	No evidence	A
Alcedinidae			
Megaceryle alcyon	Belted kingfisher	BR	C
Picidae			
Melanerpes lewis	Lewis's woodpecker	No evidence	A
Melanerpes erythrocephalus	Red-headed woodpecker	No evidence	A
Melanerpes carolinus	Red-bellied woodpecker	No evidence	R
Sphyrapicus varius	Yellow-bellied sapsucker	BR	F
Picoides pubescens	Downy woodpecker	BR	F
Picoides villosus	Hairy woodpecker	BR	F

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Picoides dorsalis	American three-toed woodpecker	BR	F
Picoides arcticus	Black-backed woodpecker	BR	F
Colaptes auratus	Northern flicker	BR	F
Dryocopus pileatus	Pileated woodpecker	BR	R
Tyrannidae			
Contopus cooperi	Olive-sided flycatcher	BR	U
Contopus sordidulus	Western wood-pewee	No evidence	A
Contopus virens	Eastern wood-pewee	No evidence	U
Empidonax flaviventris	Yellow-bellied flycatcher	BR	F
Empidonax alnorum	Alder flycatcher	BR	C
Empidonax minimus	Least flycatcher	BR	C
Sayornis phoebe	Eastern phoebe	No evidence	A
Sayornis saya	Say's phoebe	No evidence	A
Myiarchus crinitus	Great crested flycatcher	No evidence	A
Tyrannus verticalis	Western kingbird	No evidence	A
Tyrannus tyrannus	Eastern kingbird	BR	U
Tyrannus forficatus	Scissor-tailed flycatcher	No evidence	A
Tyrannus savana	Fork-tailed flycatcher	No evidence	A
Laniidae			
Lanius ludovicianus	Loggerhead shrike	No evidence	A
Lanius excubitor	Northern shrike	BR	U
Vireonidae			
Vireo solitarius	Blue-headed vireo	BR	F
Vireo gilvus	Warbling vireo	br	R
Vireo philadelphicus	Philidelphia vireo	BR	C
Vireo olivaceus	Red-eyed vireo	BR	C
Corvidae			
Perisoreus canadensis	Gray jay	BR	F
Cyanocitta cristata	Blue jay	No evidence	A
Pica hudsonia	Black-billed magpie	No evidence	A
Corvus brachyrhynchos	American crow	BR	C
Corvus corax	Common raven	BR	C
Alaudidae			
Eremophila alpestris	Horned lark	BR	C
Hirundinidae			
Progne subis	Purple martin	No evidence	A

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Tachycineta bicolor	Tree swallow	BR	C
Tachycineta thalassina	Violet-green swallow	No evidence	A
Stelgidopteryx serripennis	Northern rough-winged swallow	No evidence	A
Riparia riparia	Bank swallow	BR	C
Petrochelidon pyrrhonota	Cliff swallow	BR	C
Hirundo rustica	Barn swallow	BR	C
Paridae			
Poecile atricapillus	Black-capped chickadee	BR	F
Poecile hudsonicus	Boreal chickadee	BR	C
Sittidae			
Sitta canadensis	Red-breasted nuthatch	br	F
Sitta carolinensis	White-breasted nuthatch	No evidence	R
Certhiidae			
Certhia americana	Brown creeper	BR	F
Troglodytidae			
Salpinctes obsoletus	Rock wren	BR	R
Troglodytes aedon	House wren	br	A
Troglodytes hiemalis	Winter wren	BR	F
Cistothorus platensis	Sedge wren	No evidence	R
Cistothorus palustris	Marsh wren	BR	A
Polioptilidae			
Polioptila caerulea	Blue-gray gnatcatcher	No evidence	A
Regulidae			
Regulus satrapa	Golden-crowned kinglet	BR	F
Regulus calendula	Ruby-crowned kinglet	BR	C
Muscicapidae			
Oenanthe oenanthe	Northern wheatear	No evidence	R
Turdidae			
Sialia sialis	Eastern bluebird	No evidence	U
Sialia currucoides	Mountain bluebird	No evidence	A
Myadestes townsendi	Townsend's solitaire	No evidence	A
Catharus fuscescens	Veery	No evidence	A
Catharus minimus	Gray-cheeked thrush	BR	F
Catharus ustulatus	Swainson's thrush	BR	C
Catharus guttatus	Hermit thrush	BR	F
Hylocichla mustelina	Wood thrush	No evidence	A

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Turdus migratorius	American robin	BR	C
Ixoreus naevius	Varied thrush	No evidence	A
Mimidae			
Dumetella carolinensis	Gray catbird	BR	U
Mimus polyglottos	Northern mockingbird	br	R
Toxostoma rufum	Brown thrasher	br	R
Sturnidae			
Sturnus vulgaris	European starling	BR	C
Motacillidae			
Motacilla tschutschensis	Eastern yellow wagtail	No evidence	H
Anthus rubescens	American pipit	BR	C
Anthus spragueii	Sprague’s pipit	No evidence	A
Bombycillidae			
Bombycilla garrulus	Bohemian waxwing	BR	U
Bombycilla cedrorum	Cedar waxwing	BR	C
Calcariidae			
Calcarius lapponicus	Lapland longspur	BR	C
Calcarius pictus	Smith’s longspur	BR	C
Plectrophenax nivalis	Snow bunting	BR	C
Parulidae			
Seiurus aurocapilla	Ovenbird	BR	C
Parkrsia noveboracensis	Northern waterthrush	BR	C
Vermivora cyanoptera	Blue-winged warbler	No evidence	A
Mniotilta varia	Black-and-white warbler	BR	C
Protonotaria citrea	Prothonotary warbler	No evidence	A
Oreothlypis peregrina	Tennessee warbler	BR	C
Oreothlypis celata	Orange-crowned warbler	BR	F
Oreothlypis ruficapilla	Nashville warbler	BR	F
Oporonis agilis	Connecticut warbler	br	C
Geothlypis formosa	Kentucky warbler	No evidence	A
Geothlypis philidelphia	Mourning warbler	BR	F
Geothlypis trichas	Common yellowthroat	BR	C
Setophaga citrina	Hooded warbler	No evidence	A
Setophaga ruticilla	American redstart	BR	C
Setophaga tigrina	Cape May warbler	br	U
Setophaga americana	Northern parula	br	U

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Setophaga magnolia	Magnolia warbler	BR	C
Setophaga castanea	Bay-breasted warbler	BR	F
Setophaga fusca	Blackburnian warbler	br	U
Setophaga petechia	Yellow warbler	BR	C
Setophaga pensylvanica	Chestnut-sided warbler	BR	U
Setophaga striata	Blackpoll warbler	BR	C
Setophaga caerulescens	Black-throated blue warbler	No evidence	A
Setophaga palmarum	Palm warbler	BR	F
Setophaga pinus	Pine warbler	No evidence	A
Setophaga coronata	Yellow-rumped warbler	BR	C
Setophaga dominica	Yellow-throated warbler	No evidence	A
Setophaga virens	Black-throated green warbler	BR	U
Cardellina canadensis	Canada warbler	BR	U
Cardellina pusilla	Wilson's warbler	BR	C
Icteria virens	Yellow-breasted chat	No evidence	A
Emberizidae			
Pipilo chlorurus	Green-tailed towhee	No evidence	H
Pipilo maculatus	Spotted towhee	No evidence	A
Pipilo erythrophthalmus	Eastern towhee	No evidence	R
Spizella arborea	American tree sparrow	BR	C
Spizella passerina	Chipping sparrow	BR	C
Spizella pallida	Clay-colored sparrow	BR	U
Spizella breweri	Brewer's sparrow	No evidence	H
Spizella pusilla	Field sparrow	No evidence	A
Pooecetes gramineus	Vesper sparrow	BR	U
Chondestes grammacus	Lark sparrow	No evidence	A
Calamospiza melanocorys	Lark bunting	No evidence	A
Passerculus sandwichensis	Savannah sparrow	BR	C
Ammodramus henslowii	Henslow's sparrow	No evidence	A
Ammodramus leconteii	Le Conte's sparrow	BR	U
Ammodramus nelsoni	Nelson's sparrow	BR	F
Passerella iliaca	Fox sparrow	BR	C
Melospiza melodia	Song sparrow	BR	C
Melospiza lincolnii	Lincoln's sparrow	BR	C
Melospiza georgiana	Swamp sparrow	BR	C
Zonotrichia albicollis	White-throated sparrow	BR	C

Appendix 1, Cont.

Species		Status in the Hudson Plains Ecozone	
Scientific name	Common name	Breeding	Relative abundance
Zonotrichia querula	Harris's sparrow	BR	U
Zonotrichia leucophrys	White-crowned sparrow	BR	C
Junco hyemalis	Dark-eyed junco	BR	C
Cardinalidae			
Piranga olivacea	Scarlet tanager	No evidence	U
Piranga ludoviciana	Western tanager	No evidence	A
Cardinalis cardinalis	Northern cardinal	No evidence	R
Pheucticus ludovicianus	Rose-breasted grosbeak	br	U
Passerina cyanea	Indigo bunting	No evidence	R
Passerina ciris	Painted bunting	No evidence	A
Spiza americana	Dickcissel	No evidence	A
Icteridae			
Dolichonyx oryzivorus	Bobolink	No evidence	U
Agelaius phoeniceus	Red-winged blackbird	BR	C
Sturnella magna	Eastern meadowlark	No evidence	A
Sturnella neglecta	Western meadowlark	No evidence	R
Xanthocephalus xanthocephalus	Yellow-headed blackbird	No evidence	R
Euphagus carolinus	Rusty blackbird	BR	C
Euphagus cyanocephalus	Brewer's blackbird	No evidence	A
Quiscalus quiscula	Common grackle	BR	C
Molothrus ater	Brown-headed cowbird	BR	F
Icterus galbula	Baltimore oriole	No evidence	A
Fringillidae			
Pinicola enucleator	Pine grosbeak	BR	F
Carpodacus purpureus	Purple finch	BR	F
Loxia curvirostra	Red crossbill	No evidence	A
Loxia leucoptera	White-winged crossbill	BR	F
Acanthis flammea	Common redpoll	BR	F
Acanthis hornemanni	Hoary redpoll	BR	U
Spinus pinus	Pine siskin	BR	U
Spinus tristis	American goldfinch	br	F
Coccothraustes vespertinus	Evening grosbeak	BR	F
Passeridae			
Passer domesticus	House sparrow	BR	R

[a] In the text of this report, this species is referred to as lesser snow goose, *C. caerulescens caerulescens*, the dominant breeding and migrating form of the species found in the ecozone; the greater snow goose (*C. caerulescens atlantica*) occurs in the ecozone as an *accidental* species.