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THE STATE OF NORTHWEST TERRITORIES FORESTS IN THE WAKE OF CLIMATE CHANGE: BASELINE CONDITIONS AND OBSERVED CHANGES TO FOREST ECOSYSTEMS

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The State of Northwest Territories Forests in the Wake of Climate Change: Baseline Conditions and Observed Changes to Forest Ecosystems

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ABSTRACT

The climate in Canada’s north has changed over the past 70 years. Indeed, mean annual temperature in the Northwest Territories has increased by 2°C to 4°C since record keeping began in 1950. Annual precipitation amounts and patterns have also changed during this period. These and other changes have already noticeably affected the appearance and function of northern forested ecosystems and will continue to do so into the future. As such, the Government of Northwest Territories and the Canadian Forest Service have collaborated to produce this baseline report summarizing the current ecological conditions of forested ecosystems in the Northwest Territories and identifying gaps in our understanding of northern forests and ecosystem processes. Key findings presented in this report include the following: 1) upland forests in western Northwest Territories, the most productive forests in the territory, experienced net negative growth trends between 1950 and 2002, which were correlated with reduced soil moisture; 2) permafrost has begun thawing across southern and central portions of the territory, which has been linked to localized flooding and will eventually lead to replacement of coniferous forests with shrub- and moss-dominated plant communities; 3) changes in wildfire frequency and intensity in recent decades will, over the long-term, alter the species composition of northern forests; 4) biotic and abiotic disturbances have become more common in the north and, during severe occurrences, will cause permanent damage to forests; 5) upland and peat plateau forests in the Northwest Territories have historically been roughly carbon neutral, but permafrost thaw and more intense wildfires are expected to increase carbon emissions from these forests; and 6) boreal caribou populations are currently considered self-sustaining in the Northwest Territories, but high-quality caribou habitat is at risk of degradation from wildfire, permafrost thaw, and increasing amounts of human development.

RÉSUMÉ

Le climat dans le nord du Canada a changé au cours des 70 dernières années. En effet, la température annuelle moyenne dans les Territoires du Nord-Ouest a augmenté de 2°C à 4°C depuis le début de la tenue de registres en 1950. Les quantités et les régimes de précipitations annuelles ont également changé au cours de cette période. Ces changements, et autres, ont déjà eu une incidence sur l'apparence et la fonction des écosystèmes forestiers du Nord et continueront de le faire dans l'avenir. Le gouvernement des Territoires du Nord-Ouest et le Service canadien des forêts ont donc collaboré pour préparer le présent rapport de base, lequel résume les conditions écologiques actuelles des écosystèmes forestiers des Territoires du Nord-Ouest et relève les lacunes dans nos connaissances sur les forêts septentrionales et les processus écosystémiques. Les principales conclusions présentées dans le rapport sont les suivantes : 1) entre 1950 et 2002, on a constaté une tendance de croissance négative nette des forêts les plus productives des Territoires du Nord-Ouest, à savoir les forêts des hautes terres de l'ouest, en raison de la réduction de l'humidité du sol; 2) le pergélisol a commencé à dégeler dans les parties sud et centrale du territoire, ce qui est causé par des inondations localisées et mènera éventuellement au remplacement des forêts de conifères par des communautés végétales dominées par des arbustes et des mousses; 3) les changements dans la fréquence et l'intensité des feux de forêt au cours des dernières décennies modifieront, à long terme, la composition des espèces des forêts septentrionales; 4) les perturbations biotiques et abiotiques sont devenues plus courantes dans le Nord et les perturbations graves causeront des dommages permanents aux forêts; 5) les forêts des hautes terres et des plateaux palsiques des Territoires du Nord-Ouest étaient par le passé relativement neutres en carbone, mais on s'attend à ce que la fonte du pergélisol et les feux de forêt plus intenses augmentent les émissions de carbone de ces forêts; 6) les populations de caribou boréal sont actuellement considérées comme stables dans les Territoires du Nord-Ouest, mais l'habitat de haute qualité du caribou risque d'être dégradé par les feux de forêt, le dégel du pergélisol et l'accroissement du développement humain.

CONTENTS

FIGURES

TABLES

EXECUTIVE SUMMARY

Climate change is having noticeable effects in northern Canada, which is warming at a rate up to four times the global average. As a result, the Canadian Forest Service (CFS), in collaboration with the Government of Northwest Territories, is undertaking a climate change adaptation study of the forests of the Northwest Territories to assess their resilience to climate change. The current report represents the first stage of this project, summarizing the baseline ecological conditions of all forested regions of the territory, including (but not limited to) the Dehcho and South Slave administrative regions (southern areas of the territory) and identifying the knowledge and research gaps. A literature review was conducted that focused on forest ecology in the Northwest Territories, with an emphasis on studies conducted by the CFS. Interviews were also conducted with researchers at the Northern Forestry Centre of the CFS regarding current and upcoming projects concerning the forests of the Northwest Territories. The research was then synthesized according to the following ecological themes: climate, forest types and productivity, permafrost, wildfire, forest insects and diseases, drought, flooding, carbon balance, and caribou habitat. It is understood that these topics overlap and influence one another, and links among them are discussed in the report wherever possible. The main purposes of this report are to describe recent climate-related changes in this region and to summarize current knowledge of baseline ecological conditions, against which subsequent changes can be compared.

The following paragraphs summarize the main findings presented in the report.

Climate

Between 1950 and 2010, mean annual temperatures increased by 2.1°C to 3.9°C throughout the region, with the greatest warming occurring during the winter months. Annual precipitation increased slightly in the southern half of the territory and decreased slightly in the northern half. The growing season (measured as growing degree days > 5°C) increased slightly, with the greatest increases recorded in the northern regions.

Forest Types and Productivity

The Northwest Territories contain 44 Mha (44×10^6 hectares) of forests, with spruce comprising more than 80% of tree cover. The most productive forests are found in the Taiga Plains Mid-Boreal ecoregion, in the western part of the territory. These stands average about 77 years in age, 9.9 m in height, 50 $m^3 \cdot ha^{-1}$ in volume, 52 $t \cdot ha^{-1}$ in aboveground biomass, and 41% crown closure. Forests in the territory are found on both mineral soil uplands and peat plateaus, but are most productive in the uplands. From 1950 to 2002, these forests experienced a net negative growth trend that correlated with a decrease in soil moisture, including average annual decreases of 0.1% for black spruce (*Picea mariana*) and 0.12% for white spruce (*Picea glauca*).

Permafrost

Permafrost is discontinuous in the southern and central portions of the Northwest Territories and continuous in the north. In the discontinuous zone, permafrost is more common and is generally found closer to the surface beneath peat plateaus, because the peat insulates the soil from summer heat. When permafrost thaws, the soil of the ice-rich peat plateaus can become saturated, causing a shift from a forest dominated by spruce and lichen to a collapse scar dominated by sedge and moss. Permafrost is currently experiencing net thaw in the Northwest Territories, and thaw rates are higher in the southern portions of the territory. For example, 9.5%–11.6% of the permafrost in the Scotty Creek watershed was lost between 1970 and 2010, and the thaw rate accelerated between 2000 and 2010.

Wildfire

The southern part of the Northwest Territories has a historical fire return interval of 50–100 years, but from the mid-19th century to the second half of the 20th century, the time between fires increased to 300 years, primarily because there were fewer anthropogenic fires. Between 1986 and 2018, 0.5% of the Taiga Plains ecoregion burned annually, for a fire return interval of 200 years; however, burning rates varied substantially across the territory. Most of the area burned has been affected by high-intensity crown fires, with surface fires being common only in jack pine (*Pinus banksiana*) stands. However, even during the severe fire season of 2014, 15% of the area within the perimeter of large fires completely escaped burning, and these areas can provide habitat and seed sources for fire-sensitive species. Many overstory trees that were killed during the fires did not experience full combustion, meaning that some serotinous cones remained viable after the fires. Black spruce stands had the highest level of canopy combustion because of the flammability of this species, and jack pine stands had the highest proportion of forest floor combustion because of the drier conditions typical of this forest type. Upland and peatland forests had similar fuel loads, but peatlands generally burned only under drought conditions. Forests in the Northwest Territories are adapted to stand-replacing fires, and they generally return to their pre-fire vegetation composition. Nonetheless, some evidence suggests that increasing frequency and severity of fires may lead to a decrease in spruce and an increase in jack pine and trembling aspen (*Populus tremuloides*) in the future.

Forest Insects and Diseases

Spruce budworm (*Choristoneura fumiferana*) is the most significant forest insect in the Northwest Territories. The most recent outbreak, in 2017, caused defoliation throughout more than 165 000 ha in the Dehcho administrative region. Spruce budworm outbreaks are spreading north, likely as a result of climate change. For example, spruce budworm damage was recorded for the first time in the Mackenzie Delta of the Inuvik region in 2015. Forest tent caterpillar (*Malacosoma disstria*) has been the most significant defoliator of aspen in the Northwest Territories, but a 20-year outbreak of aspen serpentine leafminer (*Phyllocnistis populiella*) is currently causing unknown permanent damage to trembling aspen forests.

Forest diseases have historically had less impact than insects on the forests of the Northwest Territories, but under a changing climate, fungal pathogens may become more prevalent in the future. If conditions become warmer and wetter, species like spruce-Labrador tea needle rust (*Chrysomyxa ledicola*) could thrive and cause significant defoliation of white and black spruce. If conditions become warmer and drier, trees could be weakened, which would make them more susceptible to root diseases such as *Armillaria sinapina* and *Armillaria ostoyae*, both of which can cause significant damage to trembling aspen and jack pine forests.

Flooding

Historically, little information on abiotic disturbances like flooding (meaning a persistent shift to saturated soils, not temporary river flooding) has been available in the Northwest Territories. This has made the detection of climate-induced changes difficult. For this reason, abiotic disturbances have been monitored as part of the annual forest health aerial surveys carried out by the government of the Northwest Territories since 2009. Extensive flooding was first recorded in the southern part of the territory starting in 2015, but has likely been occurring for up to 20 years. This flooding has caused significant mortality of trembling aspen and, to a lesser extent, coniferous trees in this region and is thought to be a result of hydrological changes, including melting permafrost, from forest fires that occurred in 1995 and again in 2014.

Drought

The Soil Moisture Index (a measure of the amount of soil water available to tree roots) declined in the Taiga Plains ecoregion from 1950 to 2002. In the Northwest Territories, the survival of all tree species (except jack pine) and the growth of all tree species are positively correlated with soil moisture. As such, under these increasingly dry conditions, the survival of tree species other than jack pine and the growth of all tree species are expected to decline. Among the boreal tree species, black spruce has above-average drought sensitivity, whereas white spruce has average sensitivity. Both have below-average migration capacity, making them more vulnerable to drought induced by climate change. Increasing drought also has the potential to increase the frequency and severity of wildfires.

Carbon Balance

Upland forests in the Taiga Plains ecoregion were estimated to have a net carbon sequestration rate of 73–147 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ (73–147 grams of carbon dioxide per square meter per year). However, because of the nearly constant amount of carbon in their soils, it is believed that upland forests in the Northwest Territories have historically been carbon neutral. Peat plateaus fluctuate between being small carbon sinks and being carbon sources, depending on the year. Collapse scars have a CO_2 sequestration rate of about 86 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ due to the growth of peat mosses (*Sphagnum* spp.), but they also have a methane (CH_4) emission rate of about 16 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$. The greenhouse gas potential of CH_4 is estimated to be 28 times that of CO_2. Thus, the conversion of a peat plateau to collapse scar through permafrost thaw would account for an estimated net increase in emissions of 362 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$. Wildfires in the Northwest Territories, particularly in black spruce stands, cause high carbon emissions. For example, fires in 2014 burned 2.9 Mha and emitted an estimated 346 Tg CO_2 (where 1 Tg [teragram] is equivalent to 10^6 t [tonnes or metric tons]).

Caribou Habitat

The boreal caribou (*Rangifer tarandus caribou*) population in the Northwest Territories is estimated at 6000–7000 individuals and is considered self-sustaining. However, undisturbed habitat in the southern portions of its range, where 53% of the population lives, is currently below self-sustaining levels. As a result, boreal caribou is listed as threatened under the *Species at Risk (NWT) Act*. Conservation measures currently focus on maintaining at least 65% of the landscape as undisturbed habitat. Boreal caribou require varied forest habitat, but mature spruce forest is particularly important, because lichen, a primary winter food source for caribou, is mainly found in this type of forest. Undisturbed caribou habitat is at risk from increasing frequency and severity of wildfires and anthropogenic disturbance. Lichen habitat can also be lost when peat plateaus convert to collapse scars after permafrost thaw, as these sites shift from spruce–lichen forests to sedge- and moss-dominated wetlands. Wolves are the greatest source of caribou mortality in the Northwest Territories, and the influence of an increase in alternative prey, such as deer and moose, on caribou bycatch is currently unknown.

INTRODUCTION

Canada's northern regions are experiencing the effects of climate change to a greater extent than the rest of the country, with warming two to four times the global average (Department of Environment and Natural Resources [N.W.T.] 2018a; Bush and Lemmen 2019). The effects on forests include permafrost thaw and subsequent tree loss due to soil saturation and instability, increasing numbers and extent of severe wildfires, the occurrence of insect and disease outbreaks farther north than ever previously recorded, and the northerly spread of tree, shrub, and wildlife species (Olesinski and Brett 2017; Department of Environment and Natural Resources [N.W.T.] 2018a). These effects are already discernible and are expected to strengthen as climate change progresses in the coming decades. Residents of the Northwest Territories, both Indigenous and non-Indigenous, depend on forests for various needs ranging from subsistence hunting to recreation (Department of Environment and Natural Resources [N.W.T.] 2018a), and most of the territory's communities are found within forested areas (Figure 1). Changes to the forests of the Northwest Territories due to climate change will therefore affect the lives of the people living there.

Figure 1. Map of the Northwest Territories. (Source: NWT Centre for Geomatics, https://www.geomatics.gov.nt.ca/en/resources/field_resource_category/maps-1?search_api_views_fulltext=treeline&sort_by=field_resource_publication_date&sort_order=DESC. [Accessed 4 June 2021].)

As a result of concerns about changes to northern forest ecosystems, and as part of its approach to climate change adaptation and resilience, the Government of Northwest Territories (GNWT) has called for an assessment of the vulnerability of the territory's forests to climate change (Department of Environment and Natural Resources [N.W.T.] 2018a). The climate change vulnerability of the forest ecosystem is the degree to which the ecosystem is susceptible to, and unable to cope with, environmental changes arising from climate change. It is a combination of exposure to changing conditions, the ecosystem's sensitivity to those changes, and its adaptive capacity, that is, its ability to respond to the effects of change (Williamson et al. 2012).

The GNWT has partnered with the Northern Forestry Centre (NoFC) of the Canadian Forest Service (CFS) to produce a vulnerability assessment, which is currently in development. The GNWT and the NoFC have a long history of partnerships in forest management, research, and monitoring. The vulnerability assessment will be an opportunity to address scientific research gaps relating to the state of Northwest Territories forests while also serving the needs of the territory's residents (Government of Northwest Territories 2017). The vulnerability assessment for the Northwest Territories is part of the Forest Change program, which was created by the CFS in 2011 to facilitate improved understanding of adaptation to the impacts of climate change on Canada's forests. One component of the Forest Change program consists of assessments of vulnerability to climate change that incorporate both biophysical and socioeconomic information. A Canada-wide integrated assessment has been carried out, and at the time of writing (early 2022) its results are still being analyzed (Ste-Marie et al. 2015; Natural Resources Canada 2017). Through the Forest Change program, the CFS is now conducting regional integrated assessments—of which this vulnerability assessment for the Northwest Territories is one—to interpret projected climate change impacts in a manner that is relevant for regional and local forest management.

The vulnerability assessment for the Northwest Territories is based on a framework created by the Canadian Council of Forest Ministers (CCFM) and described in the Adapting Sustainable Forest Management to Climate Change series and the Climate Change and Sustainable Forest Management in Canada guidebook (Edwards et al. 2015). The CCFM vulnerability assessment framework entails several stages, one of which is a baseline report of past and present climate and forest conditions against which future changes can be compared (Williamson et al. 2012). For the Northwest Territories vulnerability assessment, the present document constitutes that baseline report. Each chapter of the report summarizes the current state of scientific knowledge about one particular aspect of the territory's forest ecosystems, specifically ecosystem classification, climate, forest inventory and productivity, permafrost, wildfire, insects and diseases, flooding, drought, the carbon cycle, and caribou habitat. The report is not meant to be comprehensive; rather, it focuses on research carried out by the CFS. This baseline report is therefore the first stage of the climate change vulnerability assessment of Northwest Territories forests and presents information for all forested areas of the territory, including the Taiga Plains, Taiga Shield, and the Boreal and Taiga Cordillera ecoregions whenever possible (see chapter 1 for ecoregion descriptions).

In contrast, the Northwest Territories vulnerability assessment, once completed, will focus on the more southerly Dehcho and South Slave administrative regions (Figure 2), between Fort Smith and Fort Liard, where most of the territory's productive forests are found. This area has also been identified as an important habitat for boreal caribou (*Rangifer tarandus caribou*), encompassing the southern portion of the NT1 caribou range (Environment Canada 2012b).

The CCFM vulnerability assessment framework defines six sustainable forest management criteria, which are consistent with the goals of the GNWT for the stewardship, protection, and sustainable management of forest resources. These criteria are biological diversity; ecosystem condition and productivity; soil and water conditions; role in global ecological cycles, including the carbon cycle; economic and social benefits; and society's responsibilities, including consultation with and the well-being of Indigenous and non-Indigenous forest-based communities (Canadian Council of Forest Ministers 2006). The GNWT is now drafting its own sustainable forest management objectives. The current vulnerability assessment is seen as an opportunity to further that work, with a view to building resilient and adaptable ecosystems, communities, infrastructure, and economies. The impacts of climate change are occurring concurrent with the ongoing transfer of authority from the federal government to the GNWT and autonomous

Indigenous governments and communities (Government of Canada et al. 2013). Thus, climate change presents an opportunity for governments and communities of the Northwest Territories to assess their own needs and plan their own future (Department of Environment and Natural Resources [N.W.T.] 2018a).

As noted above, this report is organized into chapters covering the various aspects of forest ecology that are likely to be affected by climate change. Although each topic has its own chapter, changes to these aspects of forest ecology will not occur independently. Rather, they will interact with and affect one another (Figure 3, Figure 4). For example, wildfires increase the rate of permafrost thaw, while permafrost thaw changes the hydrology and water table level of an area and therefore also affects wildfire dynamics. Both wildfires and permafrost thaw then affect carbon sequestration and caribou habitat, among other things. These and other interactions are complex and will lead to various types of feedback and unexpected outcomes (Figure 3, Figure 4). Tipping points, whereby small changes can shift entire systems to new stable states, are also a possibility (Price et al. 2013). Although the possible effects of climate change on each of these categories are described, along with some of the interactions between them, the authors are mindful of the large gaps in current knowledge regarding these effects and their interactions.

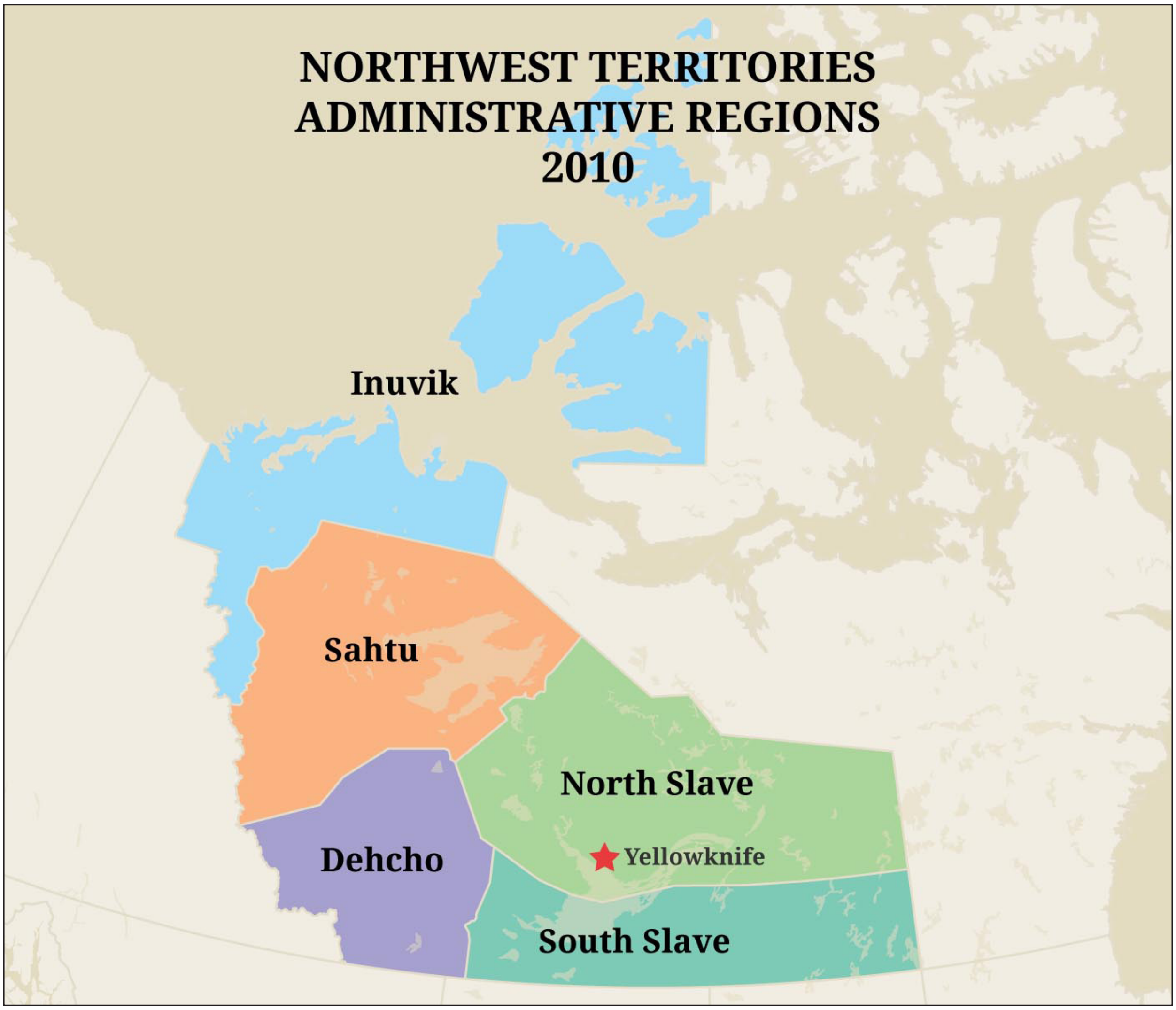


Figure 2. Administrative regions in the Northwest Territories. (Source: Prince of Wales Northern Heritage Centre, https://www.pwnhc.ca/wp-content/uploads/2014/10/map-regions-2010.png. [Accessed 22 Apr. 2020.])

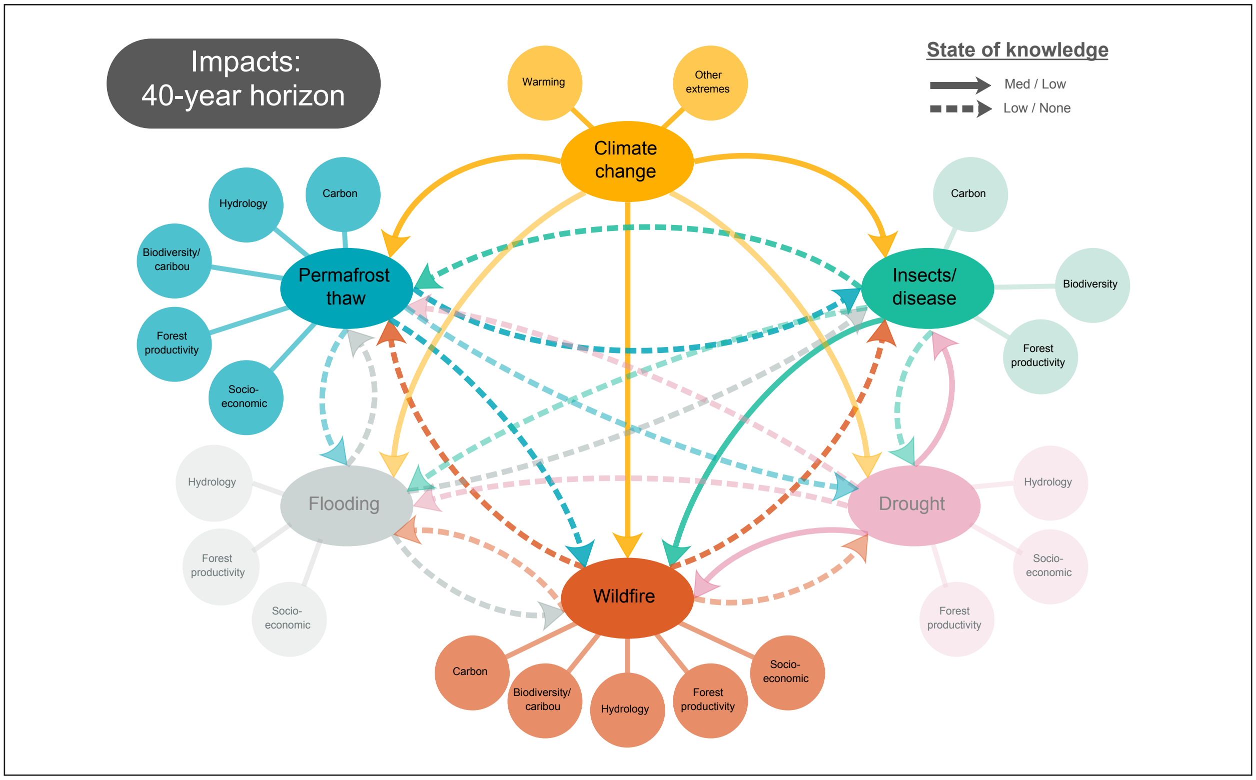


Figure 3. Climate change impacts, interactions, and state of knowledge of those relationships, according to themes in the baseline report. (Figure created by FUSE Consulting, Spruce Grove, Alberta.)

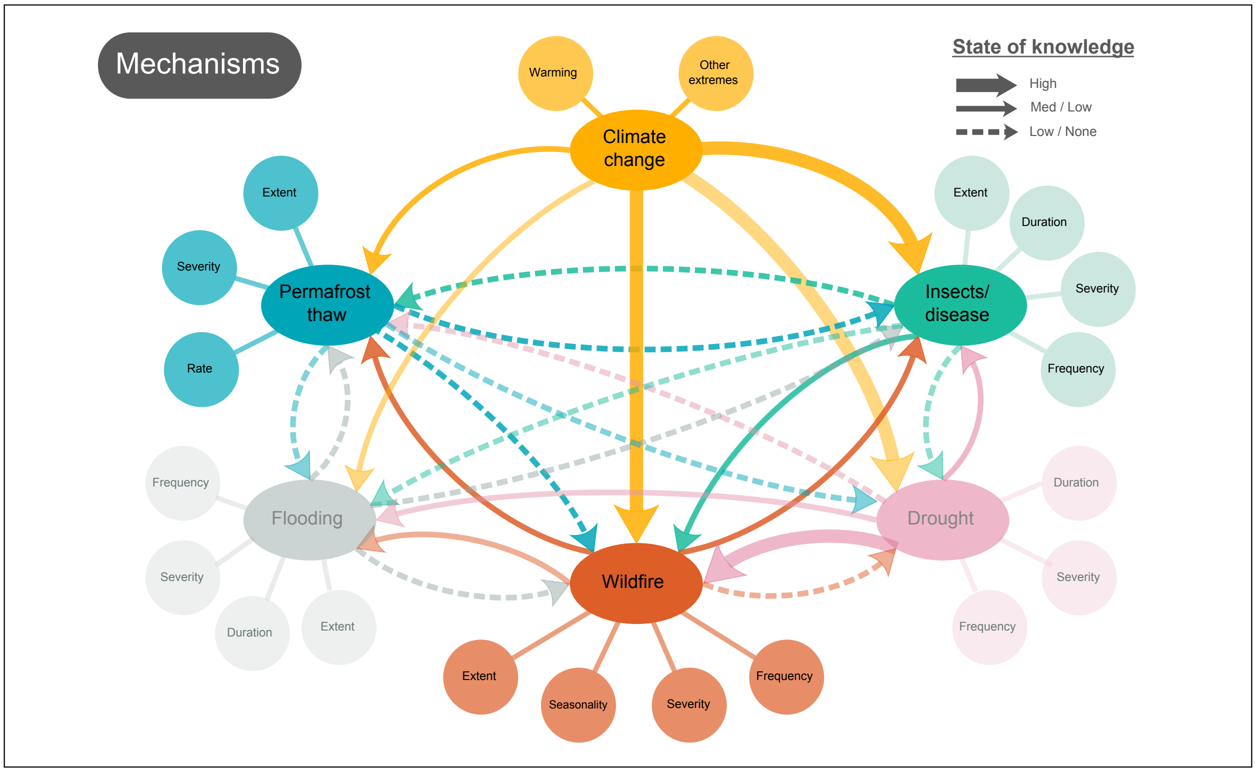


Figure 4. Climate change mechanisms, interactions, and state of knowledge of those relationships, according to themes in the baseline report. (Figure created by FUSE Consulting, Spruce Grove, Alberta.)

CHAPTER 1. NORTHWEST TERRITORIES ECOSYSTEM CLASSIFICATION

1.1 Introduction

The Northwest Territories ecosystem classification system divides the territory into four nested ecoregion levels (Ecosystem Classification Group 2007 [rev. 2009]). Level I ecoregions are global-scale biomes that highlight major ecological areas with broad similarities. There are 15 level I ecoregions in North America, three of which are found in the Northwest Territories: Tundra, Taiga, and Northwestern Forested Mountains. Level II ecoregions are subcontinental, with the Northwest Territories containing the level II Southern Arctic and Northern Arctic ecoregions within the level I Tundra ecoregion; the Taiga Plains (Figure 5), Taiga Shield (Figure 6), and Taiga Cordillera (Figure 7) ecoregions within the level I Taiga ecoregion; and the Boreal Cordillera ecoregion (Figure 7) within the level I Northwestern Forested Mountains ecoregion. Level III ecoregions are based on differences of regional climate. Level IV ecoregions are defined by local physiography and habitat.

The border between the Taiga Plains and the Taiga Shield ecoregions is defined by the abrupt shift from sedimentary plains bedrock in the west to granitic shield bedrock in the east. Most of the boundary between the mountainous Boreal and Taiga Cordillera ecoregions and the Taiga Plains ecoregion is defined by the 400-m elevation line. The boundary between the level I Tundra and Taiga ecoregions is the tree line, although it is not a clear-cut delineation. North of the tree line, trees are found in isolated patches along rivers, on deposits of coarse-textured soils, and on south-facing slopes, whereas northern portions of the Taiga Plains ecoregion may have extensive treeless areas, especially after wildfires, because of slow regeneration (Ecosystem Classification Group 2007 [rev. 2009]). The transition zone between forest and tundra in the Northwest Territories spans a latitudinal band that averages 112 km in width (Timoney et al. 1992).

In the following sections, each level III ecoregion is briefly described, along with the level II ecoregions to which they belong. Most of the information is drawn from reports of the Ecosystem Classification Group. However, temperature and precipitation values are taken from modeling for the period 1961–1990, done at the NoFC of the CFS, as described in chapter 2.

1.2 Taiga Plains Ecoregion

The Taiga Plains ecoregion stretches from the northern borders of British Columbia and Alberta, between Fort Liard and Fort Smith, north to Inuvik, encompassing the area from northwest of Great Slave Lake to Great Bear Lake, as well as the area northwest of Great Bear Lake (Figure 5). Aside from small portions in northern Alberta, northeastern British Columbia, and western Yukon, most of the Taiga Plains ecoregion lies within the Northwest Territories, accounting for 48.05 Mha (48.05 x 10^6 hectares) within the territory (Ecosystem Classification Group 2007 [rev. 2009]). Within the Northwest Territories, peatlands cover about 45% of the Taiga Plains (21.8 Mha), mineral soil uplands about 36% (17.45 Mha), and open water about 18% (8.8 Mha), including most of Great Slave Lake and all of Great Bear Lake. Most of the region is drained by the Mackenzie River and its tributaries (Ecosystem Classification Group 2007 [rev. 2009]). The Taiga Plains ecoregion is subdivided into four level III ecoregions: Mid-Boreal, High Boreal, Low Subarctic, and High Subarctic (Figure 5). The first three of these ecoregions are located in the Dehcho and South Slave administrative regions, whereas the High Subarctic ecoregion is part of the Sahtú administrative region (Figure 2). The level III Boreal ecoregions should not to be conflated with the extent of the circumpolar boreal forest within the Northwest Territories; rather, all forests in the territory (including those in the Low Subarctic and High Subarctic ecoregions) are considered part of the boreal forest.

1.2.1 Taiga Plains Mid-Boreal Ecoregion

The level III Taiga Plains Mid-Boreal ecoregion is found in the southern third of the Taiga Plains, encompassing 10.69 Mha or 22% of the Taiga Plains area (Figure 5). It has the mildest climate of any level III ecoregion in the Northwest Territories and is characterized by warm, moist summers and very cold, snowy winters (Ecosystem Classification Group 2007 [rev. 2009]). Mean annual temperature is −3.5°C. Annual precipitation is 352 mm, with

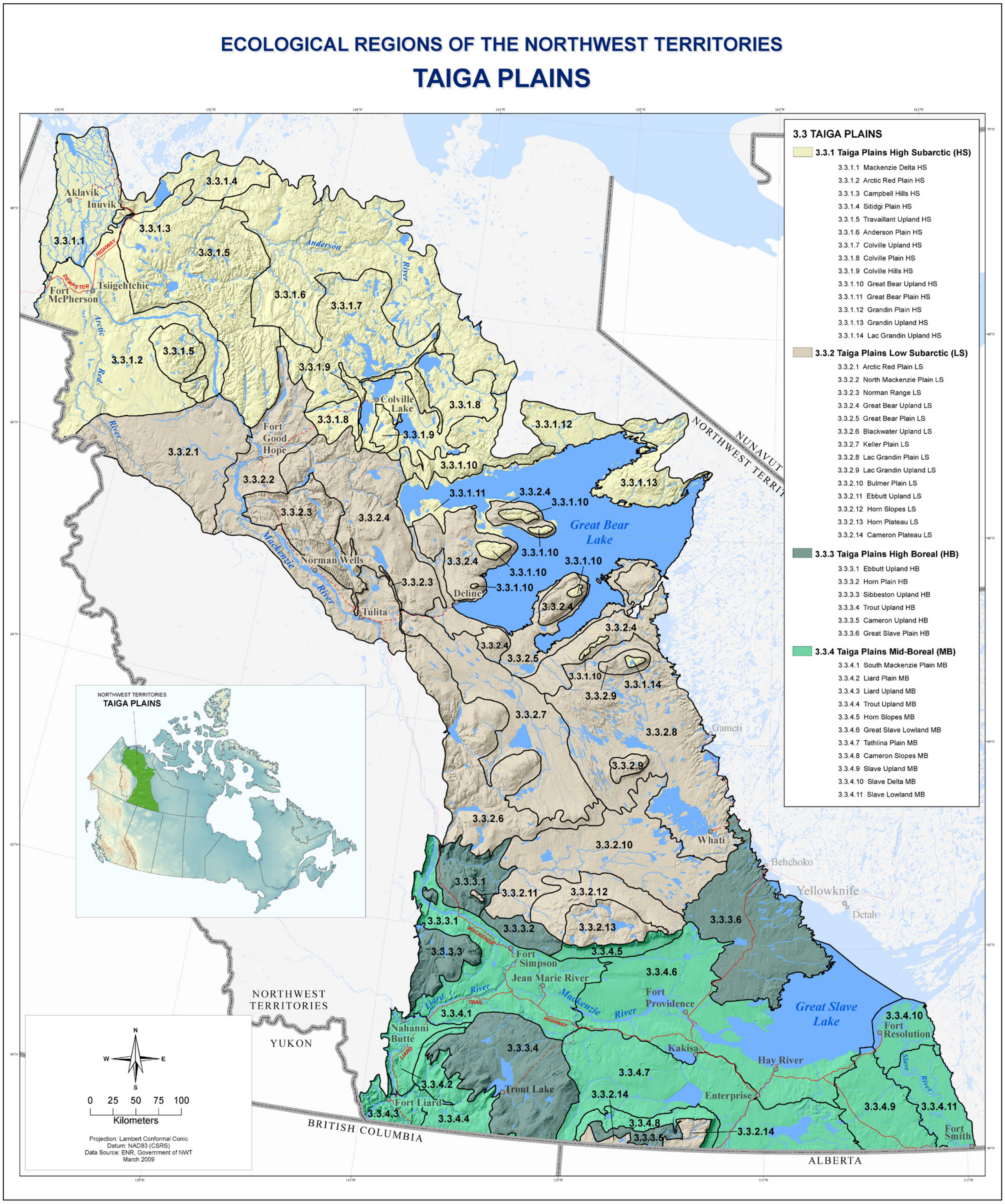


Figure 5. Taiga Plains ecoregion, Northwest Territories. (Source: Government of Northwest Territories, Department of Environment and Natural Resources. Ecosystem Classification Group 2007 [rev. 2009].)

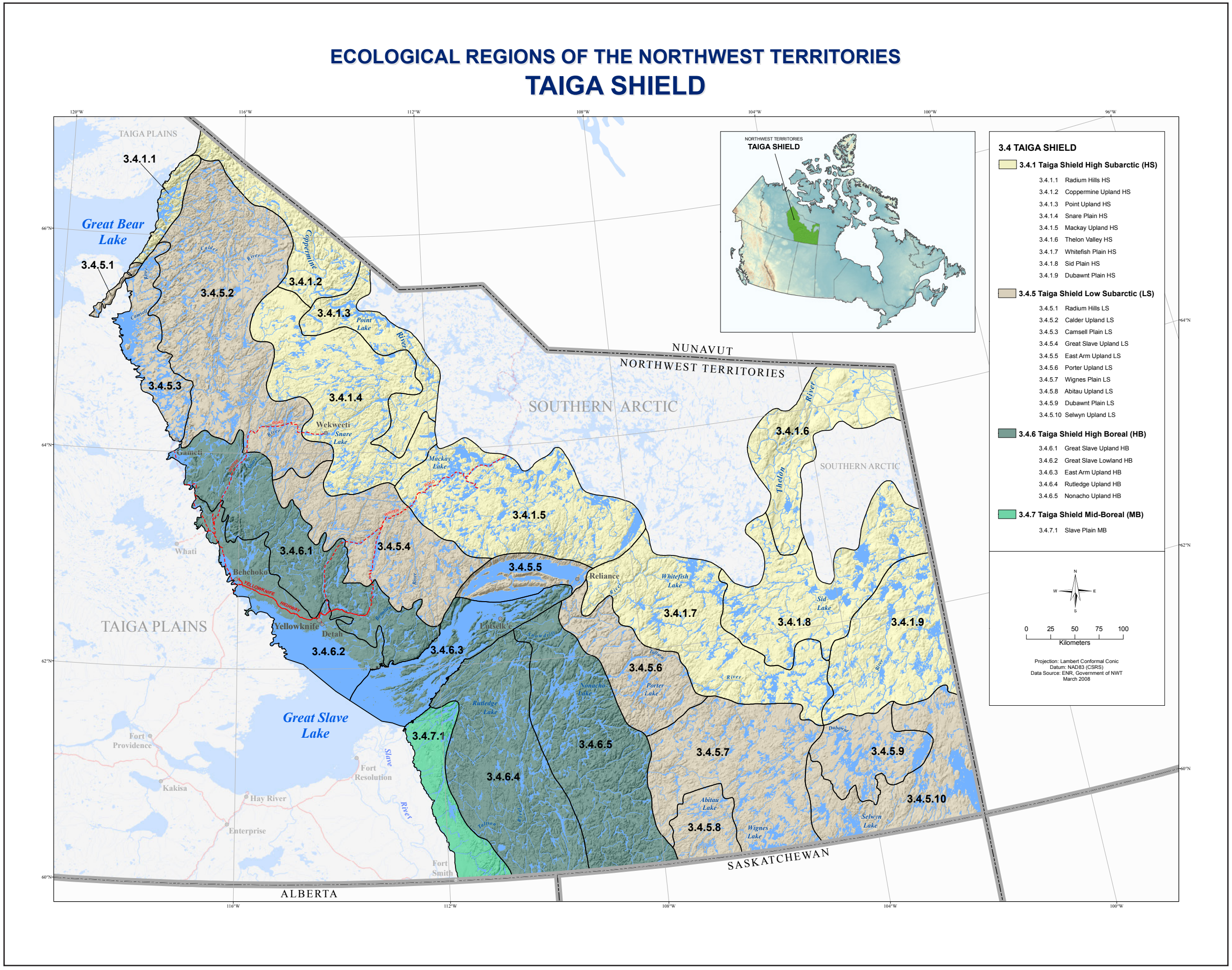


Figure 6. Taiga Shield ecoregion, Northwest Territories. (Source: Government of Northwest Territories, Department of Environment and Natural Resources. Ecosystem Classification Group 2008.)

55% of precipitation falling as rain and 45% as snow (Ecosystem Classification Group 2007 [rev. 2009]); June through August are the wettest months. Permafrost is categorized as sporadic discontinuous, meaning that it underlies 10%–50% of the area (Natural Resources Canada 2009) and is found 50–90 cm below the surface of peat plateaus and more than 1 m below the surface of collapse scars (Ecosystem Classification Group 2007 [rev. 2009]). Peatlands cover 40% of the Taiga Plains Mid-Boreal ecoregion, uplands about 42%, and open water about 18%. Peatlands include fens, bogs, and peat plateaus, which in this region contain large treeless collapse scars. Peat plateaus are underlain with permafrost and support open black spruce (*Picea mariana*) and lichen forests. Collapse scars are boggy and vegetated by sedges (*Carex* spp.), cottongrass (*Eriophorum* spp.), and mosses. Well-drained upland areas support closed-canopy mixed-wood forests of trembling aspen (*Populus tremuloides*) and white spruce (*Picea glauca*), with a diverse herb and shrub understory, while dense jack pine (*Pinus banksiana*) stands grow on drier sites (Ecosystem Classification Group 2007 [rev. 2009]). Communities within the Taiga Plains Mid-Boreal ecoregion include Fort Simpson, Fort Liard, Nahanni Butte, and Jean Marie River in the Dehcho administrative region and Fort Smith, Hay River, Fort Providence, Fort Resolution, Enterprise, and Kakisa in the South Slave administrative region. The Taiga Plains Mid-Boreal ecoregion is further divided into 11 level IV ecoregions (Ecosystem Classification Group 2007 [rev. 2009]).

1.2.2 Taiga Plains High Boreal Ecoregion

The Taiga Plains High Boreal ecoregion is found at higher elevations (300–700 m) within the same latitudes as the corresponding Mid-Boreal ecoregion or at slightly higher latitudes (Figure 5).

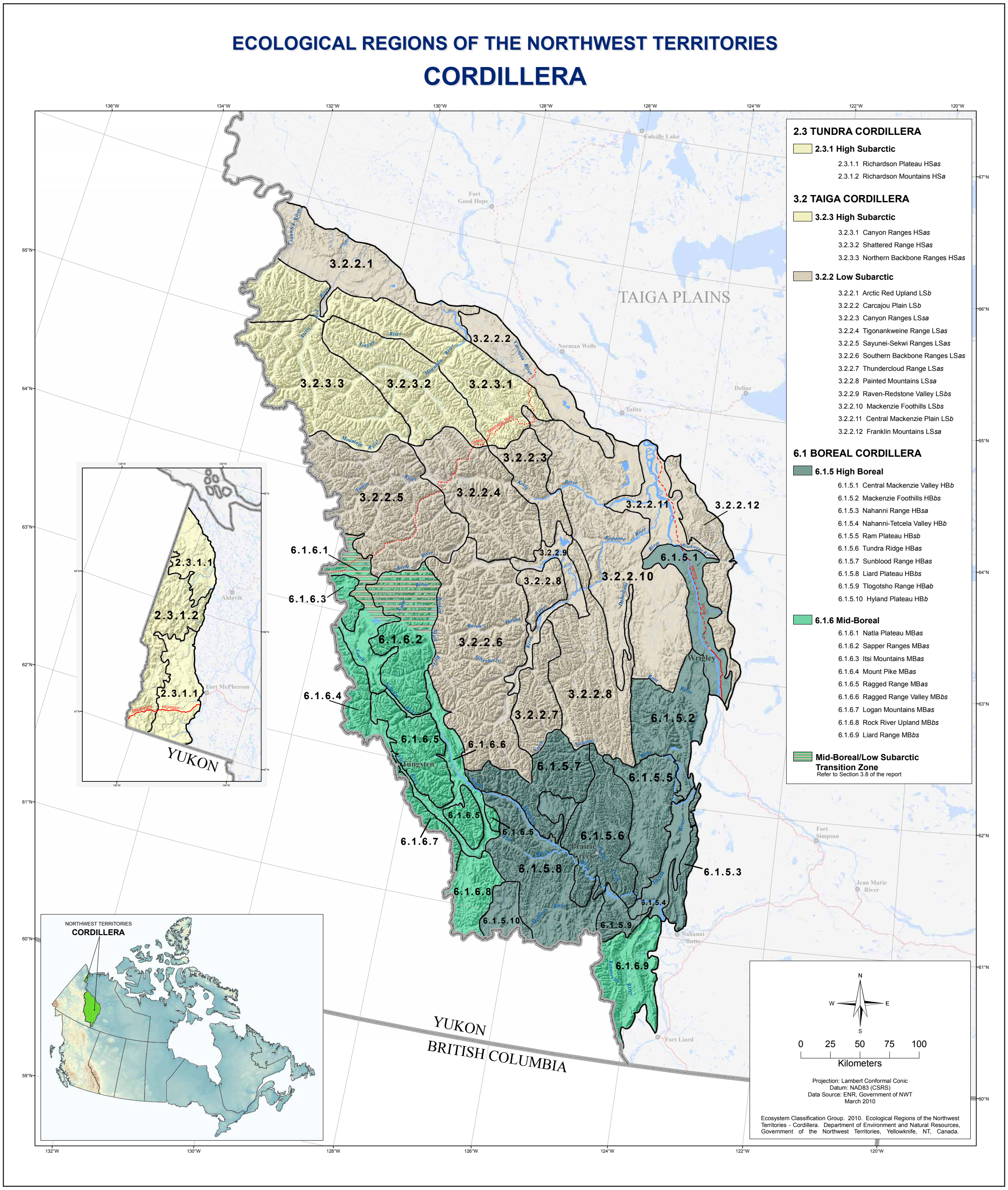


Figure 7. **Cordillera ecoregion, Northwest Territories.** (Source: Government of Northwest Territories, Department of Environment and Natural Resources. Ecosystem Classification Group 2010.)

It has an area of 4.21 Mha, or 9% of the Taiga Plains ecoregion. It is characterized by short, cool summers and very cold winters with persistent snow cover (Ecosystem Classification Group 2007 [rev. 2009]). Mean annual temperature is −4.5°C. Annual precipitation is 376 mm, with 55% of precipitation falling as rain and 45% as snow (Ecosystem Classification Group 2007 [rev. 2009]); June through August are the wettest months. Permafrost is sporadic discontinuous (Natural Resources Canada 2009), but this ecoregion contains a higher proportion of peat plateau than the Mid-Boreal ecoregion. Peatlands cover about 39% of the area, uplands about 54%, and open water about 7%. Upland forests are typically closed to semiclosed black spruce and jack pine forests, with understories of Labrador tea (*Rhododendron groenlandicum*) and feather moss (order Hypnales). Drier sites contain open forest, whereas mixed-wood forests of white spruce and trembling aspen are found on warmer southerly and westerly aspects. Bogs and fens are common on poorly drained sites (Ecosystem Classification Group 2007 [rev. 2009]). Communities within the Taiga Plains High Boreal ecoregion include Sambaa K'e (formerly known as Trout Lake). This ecoregion is further divided into six level IV ecoregions (Ecosystem Classification Group 2007 [rev. 2009]).

Along with the rest of the southern Northwest Territories, the Taiga Plains Mid-Boreal and High Boreal ecoregions have experienced an increase in mean annual air temperature of more than 2°C since 1958 and are expected to experience continuing warming at a rate twice the global average (Department of Environment and Natural Resources [N.W.T.] 2018a).

1.2.3 Taiga Plains Low Subarctic Ecoregion

The Taiga Plains Low Subarctic ecoregion is located in the central portion of the Taiga Plains from north of Fort Simpson to northwest of Great Bear Lake, with a small outlying portion farther south in the Cameron Plateau (Figure 5). It has an extent of 16.14 Mha, or 34% of the Taiga Plains ecoregion. It is characterized by short, cool summers and very cold winters (Ecosystem Classification Group 2007 [rev. 2009]). Mean annual temperature is −6.8°C. Mean annual precipitation is 310 mm, mostly falling in late summer and early fall and divided about evenly between rain and snow (Ecosystem Classification Group 2007 [rev. 2009]). Permafrost is extensive discontinuous (50%–90%) (Natural Resources Canada 2009) and lies within 30 cm of the surface. About 30% of the area is covered by peatland, 60% by uplands, and 10% by open water. Large areas are covered by peat plateaus, with polygonal peat plateaus locally in wetter, colder locations. The ecoregion is a mosaic of uplands and wetlands. Open stands of black and white spruce with understories of dwarf birch (*Betula glandulosa*), Labrador tea, peat mosses (*Sphagnum* spp.), and reindeer lichen (*Cladina* spp.) dominate unburned uplands, with stands of dwarf birch and mixed spruce on recently burned sites. Peat plateaus consist of nearly treeless communities of lichen, Labrador tea, and peat moss. Semiclosed to open black spruce stands dominate uplands farther north, with white spruce, birch, and aspen stands growing on moist upland sites and jack pine on drier, coarser sites farther south. Tree growth throughout the ecoregion is better alongside streams. The elevation ranges from 200 to 1000 m, and the topography is characterized by extensive low-elevation plains, uplands, and hill systems within the uplands (Ecosystem Classification Group 2007 [rev. 2009]). Communities within the Taiga Plains Low Subarctic ecoregion include Norman Wells, Tulita, Déline, Fort Good Hope, and Whatì. This level III ecoregion is further divided into 14 level IV ecoregions (Ecosystem Classification Group 2007 [rev. 2009]).

1.2.4 Taiga Plains High Subarctic Ecoregion

The Taiga Plains High Subarctic ecoregion is located in the northernmost portion of the Taiga Plains ecoregion, extending from the Mackenzie Delta southeast to the north shore of Great Bear Lake and covering 17.01 Mha, or 35% of the Taiga Plains (Figure 5). It is characterized by very short, cool summers and very cold winters (Ecosystem Classification Group 2007 [rev. 2009]). Mean annual temperature is −9.1°C. Mean annual precipitation is 249 mm, mostly falling in late summer and early fall and divided about evenly between rain and snow (Ecosystem Classification Group 2007 [rev. 2009]). Permafrost is continuous (90%–100%), except for an extensive discontinuous section in the Mackenzie Delta (Natural Resources Canada 2009), and lies within 30 cm of the surface. Polygonal peat plateaus are the most common form of permafrost and wetland. Peatlands cover about 12% of the area, uplands about 58%, and open water about 30%. Typical upland sites have open, stunted stands of white and black spruce and larch (*Larix laricina*), with understories of dwarf birch, willow (*Salix* spp.), Labrador tea, cottongrass, mosses, and lichens. Low shrub tundra is common on higher-elevation sites. White spruce is the dominant tree species, with stand density and height declining with increasing latitude (Ecosystem Classification Group 2007 [rev. 2009]). Communities include Inuvik, Aklavik, Fort McPherson, Tsiigehtchic, and Colville Lake. This level III ecoregion is further divided into 14 level IV ecoregions (Ecosystem Classification Group 2007 [rev. 2009]).

1.3 Taiga Shield Ecoregion

Within Canada, the Taiga Shield is divided into two discontinuous sections: an eastern section in northern Labrador and Quebec and a western section in northern Manitoba, northern Saskatchewan, eastern Northwest Territories, and southwestern Nunavut. In the Northwest Territories, the Taiga Shield lies to the east of the Taiga Plains from the Saskatchewan border north to Great Bear Lake and east to the Nunavut border (Figure 6). The Northwest Territories contains the largest proportion of the western Taiga Shield, with 33 Mha of the total 62.8 Mha (Ecosystem Classification Group 2008). Nearly 200 000 lakes of all sizes dot the surface, and the ecoregion is drained by Great Bear and Great Slave Lakes to the Arctic Ocean and by the Thelon and Dubawnt Rivers to Hudson Bay. Roughly 6% of the Taiga Shield is covered by peatland and 26% by open water, with mineral soil uplands and exposed bedrock accounting for the remainder (Ecosystem Classification Group 2008). This level II ecoregion is subdivided into four level III ecoregions: Mid-Boreal, High Boreal, Low Subarctic, and High Subarctic (Figure 6).

1.3.1 Taiga Shield Mid-Boreal Ecoregion

The Taiga Shield Mid-Boreal is the smallest of the four level III ecoregions in the Taiga Shield, occupying 663 900 ha in the southwest portion, or 2% of the Northwest Territories Taiga Shield (Figure 6). It has warm, moist summers and very cold, snowy winters (Ecosystem Classification Group 2008). Mean annual temperature is −4.0°C. Mean annual precipitation is 305 mm, with 60% falling as rain and 40% as snow; summer is the wettest season (Ecosystem Classification Group 2008). Permafrost is sporadic–extensive discontinuous (Natural Resources Canada 2009) and lies at depths greater than 60 cm. Peat plateaus occupy hollows between exposed bedrock, and these plateaus have large collapse scars. About 30% of the Taiga Shield Mid-Boreal ecoregion is peatland, 59% uplands, and 11% open water. Bedrock exposure is dominant in the northern half of the region and more scattered in the southern half. Moist-to-wet, fine-textured soils surround knobby outcrops of granitic bedrock upon which dense mixed-wood and coniferous stands of white and black spruce, jack pine, larch, aspen, balsam poplar (*Populus balsamifera*), and paper birch (*Betula papyrifera, Betula neoalaskana*) grow, especially in the southern half of the region. Poorly drained sites have closed-to-open forests of black spruce, larch, and paper birch. Shrub and sedge fens and marshes are common as well. Exposed rock is usually covered with lichen, and scattered, small stands of jack pine and black spruce may be found growing on shallow soil or in fractures (Ecosystem Classification Group 2008).

1.3.2 Taiga Shield High Boreal Ecoregion

The Taiga Shield High Boreal ecoregion stretches from the Saskatchewan border to the east arm of Great Slave Lake in a band 150 to 200 km wide, and then north along the east shore of Great Slave Lake to Gamètì in a band 80 to 100 km wide (Figure 6). It covers 8.48 Mha, or 26% of the Northwest Territories Taiga Shield, and has short, cool summers and very cold winters with persistent snow cover. Mean annual temperature is −5.6°C. Mean annual precipitation is 306 mm, about evenly divided between rain and snow, with summer as the wettest season (Ecosystem Classification Group 2008). Permafrost is extensive discontinuous (Natural Resources Canada 2009). Because of the predominance of bedrock or boulder till, peatlands are uncommon, making up only about 4% of the region, whereas open water covers about 30%. Lichen woodland with jack pine, black spruce, and paper birch are found on thin soils over bedrock, in bedrock fractures, and in coarse-textured glacial outwash. Moderately dense black spruce forests with feather moss understories are found at sites where deeper till deposits have created deeper, moister soils. White spruce stands, with or without paper birch and aspen, are found on similar sites in warmer areas within the lower-elevation western part of the ecoregion. Jack pine dominates recently burned areas, whereas black spruce dominates areas with longer fire return intervals. Peat plateaus, shore, and floating fens are the main peatland types (Ecosystem Classification Group 2008). Communities include Yellowknife, Dettah, Behchokǫ̀, Gamètì, and Łutselk'e. This level III ecoregion is subdivided into five level IV ecoregions.

1.3.3 Taiga Shield Low Subarctic Ecoregion

The Taiga Shield Low Subarctic ecoregion stretches along the southern third of the Northwest Territories from the Taiga Shield High Boreal ecoregion to the Nunavut border and in a band northwest to Great Bear Lake (Figure 6). It covers 14.4 Mha, or 44% of the Northwest Territories Taiga Shield. It has short, cool summers and very cold winters (Ecosystem Classification Group 2008). Mean annual temperature is −7.4°C. Mean annual precipitation is 312 mm, with 60% falling as rain and 40% as snow; summer is the wettest season (Ecosystem Classification Group 2008). Permafrost is mostly extensive discontinuous, except for a small section of continuous permafrost east of Great Bear Lake (Natural Resources Canada 2009). Peatlands account for about 8% of the area and consist mostly of nearly treeless peat plateaus, as well as shore fens. About 25% of the area is open water. South of Great Slave Lake, the glacial till is deeper, and open black spruce forest with understories of dwarf birch, Labrador tea, and lichen is the most common vegetation community in unburned uplands, whereas communities of dwarf birch and spruce dominate burned areas, and jack pine is found on coarse soils. North of Great Slave Lake, black spruce–lichen stands occur in patches between areas of exposed bedrock (Ecosystem Classification Group 2008). Communities in the Taiga Shield Low Subarctic ecoregion include Reliance. This level III ecoregion is subdivided into 10 level IV ecoregions (Ecosystem Classification Group 2008).

1.3.4 Taiga Shield High Subarctic Ecoregion

The Taiga Shield High Subarctic ecoregion stretches across the northern third of the Northwest Territories Taiga Shield and covers 12.4 Mha or 38% of the Northwest Territories Taiga Shield (Figure 6). It has very short, cool summers and very cold winters (Ecosystem Classification Group 2008). Mean annual temperature is −9.1°C. Mean annual precipitation is 291 mm, with about 60% falling as rain and 40% as snow; summer is the wettest season (Ecosystem Classification Group 2008). Permafrost is extensive discontinuous in the southern half and continuous in the northern half (Natural Resources Canada 2009). Peatlands account for about 3% and open water about 23% of the ecoregion. The southern third of the ecoregion has extensive burned areas with regenerating dwarf birch. South of the tree line, open, stunted spruce–lichen forests are found at lower elevations and shrub tundra at higher elevations, which range from 200 to 500 m. Because black spruce is more tolerant of poor nutrient conditions and white spruce is more tolerant of colder temperatures, the tree line plunges southward at locations where poor, sandy sites exist (Ecosystem Classification Group 2008). Communities in this ecoregion include Wekweètì. This level III ecoregion is subdivided into nine level IV ecoregions (Ecosystem Classification Group 2008).

1.4 Boreal and Taiga Cordillera Ecoregions

The Cordillera encompasses the south-central and western portion of the Northwest Territories where mountain foothills reach their easternmost extent (Figure 7). It is a system of mountains, foothills, and plateaus between the Yukon border and the Taiga Plains, covering 16.4 Mha. It spans three level I ecoregions: Boreal, Taiga, and Tundra. The Tundra Cordillera is a discontinuous unit along the Yukon border west of the Mackenzie Delta, occupying 782 800 ha or 5% of the Northwest Territories Cordillera. Because only a few trees grow in the valley bottoms and on the south-facing slopes in this area, no greater detail is provided for the Tundra Cordillera ecoregion. The Boreal Cordillera is found in the southern third of the Northwest Territories Cordillera and occupies 5.7 Mha or 35% of total area. The Taiga Cordillera is found in the northern two-thirds of the Northwest Territories Cordillera and occupies 9.95 Mha or 61% of the total area. The Boreal Cordillera is subdivided into the Mid-Boreal and High Boreal ecoregions, whereas the Taiga Cordillera is divided into the Low Subarctic and High Subarctic ecoregions (Ecosystem Classification Group 2010) (Figure 7). It should be noted that the National Ecological Framework of Canada places most of the Northwest Territories Boreal Cordillera within the Taiga Plains (Ecosystem Classification Group 2010). However, this report follows the classification of the Northwest Territories Ecosystem Classification Group, except where it uses data that relies on other classification systems, as noted. In addition to latitudinal gradients, elevational gradients play a large role in determining the vegetation of the Cordillera ecoregions.

1.4.1 Boreal Cordillera Mid-Boreal Ecoregion

The Boreal Cordillera Mid-Boreal ecoregion occupies a small north–south strip along the Yukon border north of Fort Liard and a larger north–south strip along the border for about half the Cordillera length (Figure 7). It covers 2.4 Mha or 42% of the Boreal Cordillera ecoregion. It has short, wet summers and very cold, snowy winters because of moist Pacific air that is pushed upward by the mountains (Ecosystem Classification Group 2010). Mean annual temperature is –6.9°C. Mean annual precipitation is 506 mm, about evenly divided between rain and snow (Ecosystem Classification Group 2010). Permafrost is mostly extensive discontinuous, excluding glaciers (Natural Resources Canada 2009; Ecosystem Classification Group 2010). Dense aspen, white spruce, and mixed-wood forests, as well as sedge-dominated wetlands, occur on valley bottoms and lower slopes. Lodgepole pine (*Pinus contorta*) is found in the southern section, within the Liard range and along the valleys of the South Nahanni and Flat Rivers. Subalpine fir (*Abies lasiocarpa*) and a herbaceous understory characteristic of Pacific Cordilleran climates are found in the subalpine area. The tree line is located at about 1650 m, the highest in the entire Cordillera (Ecosystem Classification Group 2010). Communities in the Boreal Cordillera Mid-Boreal ecoregion include Tungsten. This level III ecoregion is subdivided into nine level IV ecoregions (Ecosystem Classification Group 2010).

1.4.2 Boreal Cordillera High Boreal Ecoregion

The Boreal Cordillera High Boreal ecoregion consists of 3.3 Mha or 58% of the Boreal Cordillera, northeast of the Mid-Boreal ecoregion and along a portion of the central Mackenzie River (Figure 7).

It has short, cool summers and very cold winters. It is drier than the Mid-Boreal ecoregion because of a rain-shadow effect (Ecosystem Classification Group 2010). Mean annual temperature is –5.9°C. Mean annual precipitation is 470 mm, falling mostly as rain in the spring and summer (Ecosystem Classification Group 2010). Permafrost is extensive discontinuous (Natural Resources Canada 2009), although peat plateaus are uncommon and lie mainly along the border with the Taiga Cordillera ecoregion. Lodgepole and jack pine are extensive at mid-slope levels. Aspen, white spruce, and mixed-wood stands, as well as sedge-dominated wetlands, are found on lower slopes and in valley bottoms. Scattered stands of subalpine fir occur in subalpine locations. High-elevation forested plateaus are extensively burned, and unburned stands are usually open black spruce–shrub–moss forests. In the north of the region, such forests also dominate valley bottoms and lower slopes (Ecosystem Classification Group 2010). Communities include Wrigley. The Boreal Cordillera High Boreal ecoregion is subdivided into 10 level IV ecoregions (Ecosystem Classification Group 2010).

1.4.3 Taiga Cordillera Low Subarctic Ecoregion

The Taiga Cordillera Low Subarctic ecoregion consists of 7.5 Mha or 75% of the Taiga Cordillera in the central and northern portions of the Cordillera massif (Figure 7). It has short, cool summers and very cold winters (Ecosystem Classification Group 2010). Mean annual temperature is –8.0°C. Mean annual precipitation is 441 mm, most of which falls as rain between May and September (Ecosystem Classification Group 2010). Permafrost is mostly continuous, except in the vicinity of the Mackenzie Valley, where it is extensive discontinuous (Natural Resources Canada 2009). In the mountainous western portion of the region, spruce–shrub–lichen woodlands dominate valley bottoms and are widespread up to elevations greater than 1400 m, with shrubland also occurring on lower slopes that receive moisture. Alluvial fans and terraces support stands of pure white spruce and mixed white spruce, paper birch, and balsam poplar. The foothills and low plateaus in the eastern portion of the region burn extensively and have regenerating shrub and spruce stands. Jack pine is more common in the Mackenzie Valley. Unburned uplands support black spruce–shrub–lichen woodland, while dense stands of spruce, paper birch, and balsam poplar occur on terraces and lower slopes (Ecosystem Classification Group 2010). The Taiga Cordillera Low Subarctic ecoregion is subdivided into 12 level IV ecoregions (Ecosystem Classification Group 2010).

1.4.4 Taiga Cordillera High Subarctic Ecoregion

The Taiga Cordillera High Subarctic ecoregion is found in the northeast of the Cordillera massif and occupies 2.4 Mha or 24% of the Taiga Cordillera (Figure 7). This ecoregion is located in the Inuvik administrative region (Figure 2). It has very short, cold summers and extremely cold winters

(Ecosystem Classification Group 2010). Mean annual temperature is –9.6°C. Mean annual precipitation is 444 mm, most of which falls as rain between May and September (Ecosystem Classification Group 2010). Permafrost is continuous. Open, stunted stands of spruce, shrubs, and lichens occur in the valley bottoms at elevations below 1200 m. Krummholz (stunted, wind blown growth form) spruce occurs at elevations up to 1500 m on southern slopes in the south of the region, but only to elevations below 1000 m farther north and east. Lower-elevation valleys in the east of the region support mixed forests of spruce, balsam poplar, and paper birch, as well as thickets of willow and alder (Alnus spp.) on alluvial terraces (Ecosystem Classification Group 2010). The Taiga Cordillera High Subarctic ecoregion is subdivided into three level IV ecoregions (Ecosystem Classification Group 2010).

Key Points

- The Northwest Territories ecosystem classification system describes four nested levels of ecoregions.
- Level I ecoregions are global-scale biomes and highlight major ecological areas with broad similarities. There are 15 level I ecoregions in North America, three of which are found in the Northwest Territories: Tundra, Taiga, and Northwestern Forested Mountains.
- Level II ecoregions describe subcontinental-scale regions, based on broad physiological (such as the bedrock form) and climatic features. Within the forested area of the Northwest Territories, the level II ecoregions are the Taiga Plains, the Taiga Shield, the Boreal Cordillera, and the Taiga Cordillera.
- Level III ecoregions are based on regional climates; within the above-listed level II ecoregions, these are the Mid-Boreal, High Boreal, Low Subarctic, and High Subarctic ecoregions.
- Level IV ecoregions are based on a suite of regional ecological characteristics and describe defined habitats; as such, they are not described in this report, but may be useful to land and forest managers.
- The Taiga Plains is the most densely forested ecoregion in the Northwest Territories. Within this ecoregion, tree growth generally increases from the High Subarctic to the Low Subarctic to the High Boreal to the Mid-Boreal.

Knowledge Gaps

- It is unclear how the changing climate and associated changes to the extent of permafrost and location of the tree line may affect ecosystem classification, particularly for level IV ecoregions, which are mostly defined according to local landform features.

CHAPTER 2. CLIMATE NORMALS

2.1 Methods

According to the World Meteorological Organization (2017), climate normals should be computed for a period of 30 consecutive years. In the past, one commonly used benchmark period was 1961–1990, but data are now available to calculate climate normals for more recent periods, such as 1981–2010. The World Meteorological Organization recommends that multiple climate normals be calculated when possible. The 1981–2010 data provide the most up-to-date trends for the purposes of predicting changes in the near future, whereas the 1961–1990 data set provides a historical benchmark against which to compare future changes (World Meteorological Organization 2017).

Researchers at the NoFC of the CFS have created a suite of climate normals for the Northwest Territories for the periods 1961–1990 and 1981–2010, as well as a linear temperature trend from 1951 to 2010, using the climate modeling method described by McKenney et al. (2011) and Price et al. (2011). Climate station coverage in the Northwest Territories is sparse and has been declining over the past few decades. Nevertheless, CFS researchers were able to use ANUSPLIN thin-plate spline software to transform the climate data from these few stations to a continuous cover of values along a grid for the entire forested area of the Northwest Territories. These values were then averaged for each level III ecoregion (Table 1, Table 2, Table 3, Table 4, Table 5, Table 6, Table 7, and Table 8). Historical mean monthly temperatures were calculated as the average of the mean daily minimum and maximum temperatures for each month. Mean annual temperatures were then calculated from these mean monthly temperatures, as well as from the minimum and maximum monthly temperatures (Table 5 and Table 6). Seasonal temperatures were calculated as the average of 3-month groupings, with December as the first winter month (Table 3 and Table 4).

Climate normals were calculated for level III ecoregions because this is the ecoregion level most clearly defined by climatic variables (Ecosystem Classification Group 2007 [rev. 2009]). The two most important growth variables for boreal tree species are summer temperature and summer soil moisture (Girardin et al. 2016b). Therefore, in determining the effects of climate change on forests, special attention should be paid to summer climate.

2.2 Temperature

The predominant spatial temperature gradient in all level II ecoregions was a general decline across the component level III ecoregions, from the Mid-Boreal through the High Boreal through the Low Subarctic to the High Subarctic (Table 1 and Table 2). The warmest month in all ecoregions was July, with mean temperatures in the Taiga Plains ecoregion ranging from 12.6°C in the High Subarctic to 16.1°C in the Mid-Boreal during the period 1961–1990 and from 13.4°C in the High Subarctic to 16.3°C in the Mid-Boreal during the period 1981–2010. The coldest month in all ecoregions was January, with mean temperatures in the Taiga Plains ecoregion ranging from −25.4°C in the Mid-Boreal to −29.0°C in the High Subarctic during the period 1961–1990 and from −23.0°C in the Mid-Boreal to −26.8°C in the High Subarctic during the period 1981–2010 (Table 1 and Table 2). Mean annual temperatures in the Taiga Plains ecoregion ranged from −3.5°C in the Mid-Boreal to −9.1°C in the High Subarctic during the period 1961–1990 and from −2.8°C in the Mid-Boreal to −7.8°C in the High Subarctic during the period 1981–2010 (Table 5 and Table 6).

Between the climate normal periods of 1961–1990 and 1981–2010, temperatures increased for all months and all ecoregions, with the greatest warming in the winter and the least warming in either summer or fall. In the Taiga Plains ecoregion, mean winter temperature increased by about 2°C in all component ecoregions, whereas mean summer temperature increased by only 0.2°C in the Mid-Boreal and 0.04°C in the High Boreal, and mean fall temperature increased by only 0.4°C in the Low Subarctic and 0.7°C in the High Subarctic (Figure 8). Mean annual temperature in the Taiga Plains ecoregion increased by 0.8°C in the Mid and High Boreal ecoregions, 1.0°C in the Low Subarctic, and 1.2°C in the High Subarctic (Figure 8).

Table 1. Mean monthly temperature and precipitation in level III ecoregions of the Northwest Territories, 1981–2010.
(Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level III ecoregion	Dec[a]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Mean Overall Monthly Temperature (°C)													
Taiga Plains	Mid-Boreal	−20.3	−23.0	−19.8	−13.2	−1.5	6.8	13.5	16.3	14.1	7.9	−0.8	−13.0
	High Boreal	−21.4	−24.0	−20.7	−14.4	−2.6	6.1	13.0	15.7	13.3	7.0	−2.0	−14.5
	Low Subarctic	−23.6	−26.3	−23.8	−18.1	−5.8	4.3	12.5	15.4	12.3	5.6	−4.3	−17.4
	High Subarctic	−24.0	−26.8	−25.4	−21.4	−10.1	0.5	10.3	13.4	10.5	4.0	−6.1	−18.9
Taiga Shield	Mid-Boreal	−19.6	−23.0	−20.1	−13.5	−2.4	5.8	13.3	16.4	14.1	7.8	−0.1	−11.1
	High Boreal	−21.6	−25.1	−22.3	−15.9	−4.9	4.0	12.1	15.6	13.3	6.7	−1.5	−12.8
	Low Subarctic	−23.7	−27.2	−24.6	−18.7	−7.8	1.4	10.5	14.6	12.1	5.4	−3.3	−15.3
	High Subarctic	−24.6	−28.3	−26.4	−21.2	−11.1	−1.4	8.8	13.5	11.2	4.4	−4.8	−17.0
Boreal Cordillera	Mid-Boreal	−20.8	−22.4	−19.1	−14.3	−4.8	2.9	9.1	11.0	8.7	2.9	−5.3	−16.9
	High Boreal	−23.1	−24.5	−20.6	−14.5	−3.1	5.3	11.6	13.9	11.4	5.1	−4.3	−17.6
Taiga Cordillera	Low Subarctic	−23.2	−24.9	−22.4	−17.5	−6.4	3.2	10.5	12.6	9.7	3.2	−6.5	−18.9
	High Subarctic	−22.5	−24.2	−22.8	−19.1	−8.6	1.2	9.2	10.9	7.9	1.4	−8.3	−19.5
Mean Minimum Monthly Temperature (°C)													
Taiga Plains	Mid-Boreal	−24.7	−27.7	−25.4	−19.8	−7.9	0.4	7.1	10.2	8.3	2.8	−8.4	−17.0
	High Boreal	−25.9	−28.6	−26.3	−21.1	−9.1	−0.3	6.7	9.7	7.7	2.0	−8.9	−18.7
	Low Subarctic	−27.6	−30.3	−28.6	−24.1	−11.9	−1.5	6.8	9.7	7.1	1.2	−11.2	−21.2
	High Subarctic	−27.8	−30.6	−29.7	−26.6	−15.5	−4.3	5.1	8.4	6.0	0.5	−9.7	−22.6
Taiga Shield	Mid-Boreal	−23.3	−27.0	−24.8	−19.2	−8.1	0.3	7.5	10.9	9.2	3.7	−5.8	−14.4
	High Boreal	−25.4	−29.1	−26.9	−21.7	−10.7	−1.4	6.8	10.6	8.9	3.1	−8.9	−16.3
	Low Subarctic	−27.7	−31.3	−29.4	−24.4	−13.5	−3.8	5.2	9.4	7.6	1.9	−7.5	−19.1
	High Subarctic	−28.6	−32.3	−30.8	−26.3	−16.4	−6.0	3.8	8.6	6.9	1.1	−4.7	−20.8

Table 1. Continued.

Level II ecoregion	Level III ecoregion	Dec[a]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
							Mean Minimum Monthly Temperature (°C)						
Boreal Cordillera	Mid-Boreal	−24.9	−26.4	−24.0	−20.1	−10.5	−2.7	3.2	5.3	3.2	−1.7	−4.3	−20.8
	High Boreal	−27.5	−29.0	−26.1	−21.3	−9.6	−1.2	5.2	7.6	5.2	−0.4	−7.6	−21.7
Taiga Cordillera	Low Subarctic	−27.0	−28.7	−26.9	−23.1	−12.1	−2.4	4.7	6.9	4.3	−1.3	−6.1	−22.5
	High Subarctic	−26.2	−27.6	−26.6	−24.0	−13.9	−3.8	3.9	5.6	2.9	−2.6	−3.1	−22.7
							Mean Maximum Monthly Temperature (°C)						
Taiga Plains	Mid-Boreal	−15.8	−18.30	−14.21	−6.68	4.86	13.12	19.97	22.40	19.78	13.1	3.15	−9.03
	High Boreal	−16.9	−19.3	−15.2	−7.7	3.9	12.4	19.2	21.6	18.8	12.0	1.8	−10.39
	Low Subarctic	−19.5	−22.2	−19.1	−12.1	0.3	10.1	18.3	21.0	17.6	9.9	−1.1	−13.56
	High Subarctic	−20.2	−23.0	−21.2	−16.3	−4.7	5.4	15.4	18.4	15.0	7.6	−3.3	−15.31
Taiga Shield	Mid-Boreal	−15.9	−19.1	−15.5	−7.7	3.2	11.4	19.1	21.9	19.1	11.9	2.9	−7.80
	High Boreal	−17.7	−21.1	−17.7	−10.1	0.9	9.3	17.4	20.6	17.7	10.4	1.3	−9.35
	Low Subarctic	−19.6	−23.0	−19.8	−12.9	−2.1	6.7	15.8	19.7	16.6	9.0	−0.4	−11.53
	High Subarctic	−20.6	−24.3	−22.0	−16.0	−5.8	3.3	13.8	18.5	15.4	7.7	−2.0	−13.16
Boreal Cordillera	Mid-Boreal	−16.8	−18.3	−14.2	−8.5	0.9	8.6	14.9	16.8	14.3	7.6	−1.7	−13.05
	High Boreal	−18.7	−20.1	−15.0	−7.8	3.5	11.8	18.1	20.2	17.6	10.6	−0.2	−13.50
Taiga Cordillera	Low Subarctic	−19.4	−21.2	−18.0	−12.0	−0.8	8.8	16.2	18.3	15.1	7.7	−3.2	−15.37
	High Subarctic	−18.9	−20.8	−19.0	−14.3	−3.4	6.1	14.4	16.2	12.8	5.4	−5.4	−16.16
							Mean Monthly Precipitation (mm)						
Taiga Plains	Mid-Boreal	19.3	19.5	16.7	16.5	16.1	29.0	43.5	53.9	54.2	39.3	35.1	26.4
	High Boreal	20.5	20.7	18.6	18.1	18.2	31.5	48.2	59.2	55.4	39.6	36.1	27.5
	Low Subarctic	17.8	17.0	15.8	14.0	13.4	21.2	34.4	41.4	42.0	32.7	29.0	22.2
	High Subarctic	14.9	14.1	13.1	12.0	11.1	15.6	22.3	31.7	34.0	27.4	23.3	17.3

Table 1. Concluded.

Level II ecoregion	Level III ecoregion	Dec[a]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
							Mean Monthly Precipitation (mm)						
Taiga Shield	Mid-Boreal	17.6	17.6	13.9	14.6	13.1	22.4	36.5	43.1	44.7	38.3	31.5	25.7
	High Boreal	18.4	18.3	15.8	16.4	14.4	22.1	33.5	41.3	43.0	38.0	30.4	27.1
	Low Subarctic	17.8	17.8	15.5	17.8	15.9	24.1	34.0	44.7	48.2	41.0	29.8	25.0
	High Subarctic	16.9	15.3	13.9	16.7	15.1	23.2	31.7	41.4	48.7	39.1	27.7	22.0
Boreal Cordillera	Mid-Boreal	35.3	40.4	34.9	33.1	28.3	45.8	65.0	76.5	67.0	53.8	45.4	44.6
	High Boreal	27.6	28.9	24.8	22.8	19.1	35.4	54.0	62.6	52.3	38.9	34.0	31.5
Taiga Cordillera	Low Subarctic	27.7	29.2	27.0	23.9	20.8	32.9	53.3	60.6	55.0	42.8	36.6	34.1
	High Subarctic	28.3	28.9	28.2	25.3	23.5	35.3	54.5	64.5	58.8	47.5	39.2	35.2

[a] *Winter is December–February, spring is March–May, summer is June–August, fall is September–November.*

Table 2. Mean monthly temperature and precipitation in level III ecoregions of the Northwest Territories, 1961–1990. (Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level III ecoregion	Dec[a]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
							Mean Overall Monthly Temperature (°C)						
Taiga Plains	Mid-Boreal	−22.1	−25.4	−21.7	−14.9	−2.3	7.2	13.5	16.1	13.9	7.6	−0.4	−13.9
	High Boreal	−23.2	−26.6	−22.7	−16.0	−3.6	6.2	13.0	15.6	13.2	6.6	−1.7	−15.4
	Low Subarctic	−25.1	−29.0	−26.0	−19.8	−7.7	3.9	12.3	14.8	12.0	5.1	−4.1	−18.2
	High Subarctic	−25.4	−29.0	−27.7	−23.2	−12.5	−0.3	9.5	12.6	10.0	3.5	−6.3	−20.0
Taiga Shield	Mid-Boreal	−22.2	−26.3	−22.8	−16.1	−3.7	6.3	12.9	15.8	14.0	7.2	−0.1	−12.7
	High Boreal	−23.7	−28.2	−24.9	−18.8	−6.5	4.0	11.7	15.1	13.2	6.1	−1.4	−14.3
	Low Subarctic	−24.4	−29.4	−26.7	−21.1	−9.5	1.3	9.9	13.6	11.6	4.7	−3.0	−16.0
	High Subarctic	−25.7	−30.5	−28.4	−23.6	−12.3	−1.3	7.9	12.5	10.7	3.8	−4.6	−17.6
Boreal Cordillera	Mid-Boreal	−22.5	−24.6	−21.4	−16.4	−7.3	1.6	8.0	10.6	8.3	2.7	−5.3	−17.2

Table 2. Continued.

Level II ecoregion	Level III ecoregion	Dec[a]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Mean Overall Monthly Temperature (°C)													
Taiga Cordillera	High Boreal	−24.1	−26.0	−22.4	−16.0	−4.7	4.3	10.9	13.1	10.6	4.8	−4.1	−17.5
	Low Subarctic	−25.0	−27.1	−24.7	−19.5	−8.9	1.8	9.4	11.9	9.2	2.8	−6.4	−19.3
	High Subarctic	−25.3	−27.3	−25.7	−21.7	−12.2	−0.9	7.5	10.3	7.5	0.9	−8.4	−20.2
Mean Minimum Monthly Temperature (°C)													
Taiga Plains	Mid-Boreal	−26.7	−30.2	−27.6	−21.8	−8.7	0.8	7.0	9.9	7.8	2.2	−4.4	−18.0
	High Boreal	−27.7	−31.1	−28.4	−22.9	−10.0	−0.1	6.5	9.4	7.1	1.4	−5.5	−19.6
	Low Subarctic	−29.1	−33.1	−30.8	−25.9	−13.8	−1.9	6.3	8.9	6.5	0.6	−7.2	−21.9
	High Subarctic	−29.2	−32.9	−32.0	−28.3	−17.9	−5.1	4.2	7.4	5.3	−0.2	−9.2	−23.5
Taiga Shield	Mid-Boreal	−26.6	−31.3	−28.4	−22.9	−9.9	0.2	6.8	10.1	8.5	2.7	−3.6	−16.7
	High Boreal	−27.9	−32.9	−30.3	−25.4	−12.8	−1.7	5.9	9.7	8.3	2.2	−4.5	−18.2
	Low Subarctic	−28.5	−33.7	−31.6	−27.0	−15.3	−4.0	4.4	8.5	7.1	1.2	−5.9	−19.8
	High Subarctic	−29.6	−34.7	−33.1	−29.2	−18.0	−6.2	2.8	7.6	6.5	0.7	−7.4	−21.4
Boreal Cordillera	Mid-Boreal	−27.4	−29.3	−27.1	−22.9	−13.7	−4.0	1.6	4.5	2.3	−2.4	−9.2	−21.9
Taiga Cordillera	High Boreal	−28.7	−30.6	−27.9	−22.7	−11.1	−1.8	4.4	6.8	4.4	−0.6	−8.1	−22.0
	Low Subarctic	−29.4	−31.5	−29.8	−25.7	−15.3	−3.9	3.2	5.8	3.2	−2.0	−9.9	−23.5
	High Subarctic	−29.8	−31.6	−30.5	−27.5	−18.4	−6.1	1.5	4.5	1.7	−3.6	−11.7	−24.4
Mean Maximum Monthly Temperature (°C)													
Taiga Plains	Mid-Boreal	−17.6	−20.6	−15.8	−7.9	4.2	13.5	20.0	22.3	20.0	13.0	3.6	−9.7
	High Boreal	−18.8	−22.0	−17.0	−9.2	2.9	12.5	19.5	21.8	19.3	11.9	2.2	−11.2
	Low Subarctic	−21.2	−25.0	−21.2	−13.7	−1.5	9.7	18.4	20.7	17.6	9.6	−0.9	−14.4
	High Subarctic	−21.6	−25.1	−23.4	−18.0	−7.1	4.6	14.7	17.8	14.7	7.1	−3.5	−16.4
Taiga Shield	Mid-Boreal	−17.9	−21.3	−17.2	−9.3	2.6	12.4	19.0	21.6	19.5	11.7	3.4	−8.7

Table 2. Concluded.

Level II ecoregion	Level III ecoregion	Dec[a]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Mean Maximum Monthly Temperature (°C)													
	High Boreal	–19.5	–23.4	–19.5	–12.2	–0.3	9.8	17.5	20.5	18.1	10.0	1.7	–10.4
	Low Subarctic	–20.4	–25.0	–21.7	–15.2	–3.6	6.5	15.4	18.8	16.2	8.2	–0.1	–12.3
	High Subarctic	–21.8	–26.2	–23.8	–18.1	–6.7	3.6	13.1	17.5	14.9	7.0	–1.7	–13.8
Boreal Cordillera	Mid-Boreal	–17.6	–19.8	–15.7	–9.9	–0.8	7.3	14.4	16.7	14.4	7.7	–1.3	–12.6
	High Boreal	–19.4	–21.4	–16.9	–9.3	1.7	10.4	17.4	19.3	16.7	10.2	–0.2	–12.9
Taiga Cordillera	Low Subarctic	–20.5	–22.7	–19.7	–13.3	–2.6	7.4	15.6	17.9	15.1	7.6	–3.0	–15.0
	High Subarctic	–20.9	–22.9	–21.0	–15.9	–6.0	4.4	13.5	16.1	13.3	5.4	–5.2	–15.9
Mean Monthly Precipitation (mm)													
Taiga Plains	Mid-Boreal	20.8	21.3	17.0	17.2	16.5	26.9	39.9	52.7	48.1	32.7	30.3	28.5
	High Boreal	21.6	21.2	18.4	19.5	18.8	28.5	44.7	58.7	51.9	32.0	32.8	27.9
	Low Subarctic	18.9	18.1	14.9	14.0	14.5	19.3	36.0	46.2	47.3	27.9	30.5	22.2
	High Subarctic	15.9	14.0	11.6	11.1	11.7	13.6	24.4	36.0	41.6	24.6	27.4	17.2
Taiga Shield	Mid-Boreal	17.1	18.9	13.6	12.4	12.1	21.7	31.9	45.3	42.6	35.1	27.9	26.1
	High Boreal	17.9	17.7	14.2	13.4	13.9	18.7	30.9	42.7	45.7	33.0	31.4	26.4
	Low Subarctic	17.7	16.3	12.7	14.6	15.3	18.3	32.2	45.0	49.7	35.0	31.4	24.1
	High Subarctic	14.7	12.9	10.4	12.7	14.7	17.4	30.2	42.7	46.9	36.1	31.2	20.8
Boreal Cordillera	Mid-Boreal	28.4	27.8	26.0	25.4	26.9	32.5	56.8	80.2	72.1	43.5	51.6	35.1
	High Boreal	24.3	26.0	22.9	24.8	24.4	35.5	59.8	78.3	65.1	35.3	43.1	30.2
Taiga Cordillera	Low Subarctic	24.5	23.5	20.9	21.7	22.9	26.3	53.1	69.6	64.8	37.5	45.5	30.3
	High Subarctic	29.1	24.9	21.1	24.6	23.3	21.7	48.3	66.7	63.2	39.2	47.3	34.1

[a] *Winter is December–February, spring is March–May, summer is June–August, fall is September–November.*

Table 3. Mean seasonal temperature and precipitation in level III ecoregions of the Northwest Territories, 1981–2010[a].
(Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level III ecoregion	Winter	Spring	Summer	Fall
		Mean Overall Seasonal Temperature (°C)			
Taiga Plains	Mid-Boreal	−21.0	−2.7	14.6	−2.0
	High Boreal	−22.0	−3.7	14.0	−3.2
	Low Subarctic	−24.6	−6.5	13.4	−5.4
	High Subarctic	−25.4	−10.3	11.4	−7.0
Taiga Shield	Mid-Boreal	−20.9	−3.3	14.6	−1.1
	High Boreal	−23.0	−5.6	13.7	−2.5
	Low Subarctic	−25.1	−8.3	12.4	−4.4
	High Subarctic	−26.4	−11.2	11.2	−5.8
Boreal Cordillera	Mid-Boreal	−20.8	−5.4	9.6	−6.4
	High Boreal	−22.7	−4.1	12.3	−5.6
Taiga Cordillera	Low Subarctic	−23.5	−6.9	10.9	−7.4
	High Subarctic	−23.2	−8.9	9.3	−8.8
		Mean Minimum Seasonal Temperature (°C)			
Taiga Plains	Mid-Boreal	−25.9	−9.1	8.5	−7.5
	High Boreal	−27.0	−10.1	8.1	−8.6
	Low Subarctic	−28.8	−12.5	7.9	−10.4
	High Subarctic	−29.4	−15.5	6.5	−10.6
Taiga Shield	Mid-Boreal	−25.0	−9.0	9.2	−5.5
	High Boreal	−27.2	−11.3	8.8	−7.4
	Low Subarctic	−29.5	−13.9	7.4	−8.3
	High Subarctic	−30.6	−16.2	6.4	−8.1
Boreal Cordillera	Mid-Boreal	−25.1	−11.1	3.9	−8.9
	High Boreal	−27.5	−10.7	6.0	−9.9
Taiga Cordillera	Low Subarctic	−27.5	−12.5	5.3	−10.0
	High Subarctic	−26.8	−13.9	4.2	−9.5
		Mean Maximum Seasonal Temperature (°C)			
Taiga Plains	Mid-Boreal	−16.1	3.8	20.7	2.4
	High Boreal	−17.1	2.9	19.9	1.1
	Low Subarctic	−20.3	−0.6	19.0	−1.6
	High Subarctic	−21.5	−5.2	16.2	−3.7
Taiga Shield	Mid-Boreal	−16.8	2.3	20.0	2.4
	High Boreal	−18.8	0.0	18.6	0.8
	Low Subarctic	−20.8	−2.8	17.3	−1.0
	High Subarctic	−22.3	−6.2	15.9	−2.5
Boreal Cordillera	Mid-Boreal	−16.5	0.3	15.3	−2.4
	High Boreal	−18.0	2.5	18.6	−1.0
Taiga Cordillera	Low Subarctic	−19.5	−1.3	16.5	−3.6
	High Subarctic	−19.5	−3.9	14.4	−5.4

Table 3. Concluded.

Level II ecoregion	Level III ecoregion	Winter	Spring	Summer	Fall
		Mean Seasonal Precipitation (mm)			
Taiga Plains	Mid-Boreal	55.5	61.6	151.7	100.8
	High Boreal	59.8	67.7	162.8	103.1
	Low Subarctic	50.6	48.6	117.8	83.9
	High Subarctic	42.1	38.7	88.0	68.0
Taiga Shield	Mid-Boreal	49.1	50.1	124.2	95.5
	High Boreal	52.5	53.0	117.8	95.5
	Low Subarctic	51.1	57.8	126.8	95.8
	High Subarctic	46.2	54.9	121.8	88.7
Boreal Cordillera	Mid-Boreal	110.6	107.2	208.4	143.8
	High Boreal	81.3	77.3	169.6	104.4
Taiga Cordillera	Low Subarctic	83.9	77.6	169.0	113.5
	High Subarctic	85.3	84.1	177.8	121.9

[a] *Winter is December–February, spring is March–May, summer is June–August, fall is September–November.*

Table 4. Mean seasonal temperature and precipitation in level III ecoregions of the Northwest Territories, 1961–1990[a]. (Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level III ecoregion	Winter	Spring	Summer	Fall
		Mean Overall Seasonal Temperature (°C)			
Taiga Plains	Mid-Boreal	−23.1	−3.3	14.5	−2.2
	High Boreal	−24.2	−4.5	13.9	−3.5
	Low Subarctic	−26.7	−7.9	13.1	−5.7
	High Subarctic	−27.4	−12.0	10.7	−7.6
Taiga Shield	Mid-Boreal	−23.8	−4.5	14.3	−1.9
	High Boreal	−25.6	−7.1	13.3	−3.2
	Low Subarctic	−26.8	−9.8	11.7	−4.8
	High Subarctic	−28.2	−12.4	10.4	−6.1
Boreal Cordillera	Mid-Boreal	−22.8	−7.3	9.0	−6.6
	High Boreal	−24.1	−5.5	11.5	−5.6
Taiga Cordillera	Low Subarctic	−25.6	−8.9	10.2	−7.6
	High Subarctic	−26.1	−11.6	8.4	−9.2
		Mean Minimum Seasonal Temperature (°C)			
Taiga Plains	Mid-Boreal	−28.2	−9.9	8.2	−6.8
	High Boreal	−29.1	−11.0	7.7	−7.9
	Low Subarctic	−31.0	−13.9	7.2	−9.5
	High Subarctic	−31.3	−17.1	5.6	−11.0
Taiga Shield	Mid-Boreal	−28.8	−10.9	8.4	−5.9
	High Boreal	−30.4	−13.3	8.0	−6.8
	Low Subarctic	−31.3	−15.5	6.7	−8.2

Table 4. Concluded.

Level II ecoregion	Level III ecoregion	Winter	Spring	Summer	Fall
		Mean Minimum Seasonal Temperature (°C)			
	High Subarctic	−32.4	−17.8	5.6	−9.4
Boreal Cordillera	Mid-Boreal	−27.9	−13.5	2.8	−11.2
	High Boreal	−29.1	−11.9	5.2	−10.3
Taiga Cordillera	Low Subarctic	−30.2	−15.0	4.1	−11.8
	High Subarctic	−30.6	−17.3	2.6	−13.2
		Mean Maximum Seasonal Temperature (°C)			
Taiga Plains	Mid-Boreal	−18.0	3.3	20.8	2.3
	High Boreal	−19.2	2.1	20.2	1.0
	Low Subarctic	−22.4	−1.9	18.9	−1.9
	High Subarctic	−23.4	−6.9	15.7	−4.3
Taiga Shield	Mid-Boreal	−18.8	1.9	20.1	2.1
	High Boreal	−20.8	−0.9	18.7	0.5
	Low Subarctic	−22.4	−4.1	16.8	−1.4
	High Subarctic	−24.0	−7.1	15.2	−2.9
Boreal Cordillera	Mid-Boreal	−17.7	−1.1	15.2	−2.1
	High Boreal	−19.2	0.9	17.8	−1.0
Taiga Cordillera	Low Subarctic	−21.0	−2.8	16.2	−3.5
	High Subarctic	−21.6	−5.9	14.3	−5.2
		Mean Seasonal Precipitation (mm)			
Taiga Plains	Mid-Boreal	59.1	60.5	140.7	91.5
	High Boreal	61.3	66.8	155.2	92.7
	Low Subarctic	51.9	47.7	129.5	80.6
	High Subarctic	41.4	36.5	102.0	69.2
Taiga Shield	Mid-Boreal	49.6	46.2	119.8	89.1
	High Boreal	49.8	46.0	119.2	90.7
	Low Subarctic	46.7	48.2	126.9	90.4
	High Subarctic	38.0	44.7	119.9	88.1
Boreal Cordillera	Mid-Boreal	82.2	84.8	209.1	130.1
	High Boreal	73.2	84.7	203.3	108.6
Taiga Cordillera	Low Subarctic	68.9	70.9	187.5	113.3
	High Subarctic	75.2	69.6	178.2	120.7

[a] *Winter is December–February, spring is March–May, summer is June–August, fall is September–November.*

Table 5 Mean annual[a] temperature and precipitation in level III ecoregions of the Northwest Territories, 1981–2010.
(Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level III ecoregion	Mean annual temperature (°C)	Mean annual minimum temperature (°C)	Mean annual maximum temperature (°C)	Mean annual precipitation (mm)
Taiga Plains	Mid-Boreal	−2.8	−8.2	2.7	369.6
	High Boreal	−3.7	−9.1	1.7	393.4
	Low Subarctic	−5.8	−10.7	−0.9	300.8
	High Subarctic	−7.8	−12.2	−3.5	236.8
Taiga Shield	Mid-Boreal	−2.7	−7.4	2.0	318.9
	High Boreal	−4.4	−8.9	0.1	318.8
	Low Subarctic	−6.4	−10.9	−1.8	331.6
	High Subarctic	−8.1	−12.4	−3.8	311.6
Boreal Cordillera	Mid-Boreal	−5.7	−10.7	−0.8	570.1
	High Boreal	−5.0	−10.6	0.5	432.6
Taiga Cordillera	Low Subarctic	−6.7	−11.5	−2.0	444.0
	High Subarctic	−7.9	−12.2	−3.6	469.1

[a] *Mean annual values are calculated from mean monthly values.*

Table 6. Mean annual[a] temperature and precipitation in level III ecoregions of the Northwest Territories, 1961–1990.
(Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level III ecoregion	Mean annual temperature (°C)	Mean annual minimum temperature (°C)	Mean annual maximum temperature (°C)	Mean annual precipitation (mm)
Taiga Plains	Mid-Boreal	−3.5	−9.2	2.1	351.9
	High Boreal	−4.5	−10.1	1.0	376.0
	Low Subarctic	−6.8	−11.8	−1.8	309.8
	High Subarctic	−9.1	−13.5	−4.7	249.1
Taiga Shield	Mid-Boreal	−4.0	−9.3	1.3	304.7
	High Boreal	−5.6	−10.6	−0.6	305.7
	Low Subarctic	−7.4	−12.1	−2.8	312.2
	High Subarctic	−9.1	−13.5	−4.7	290.7
Boreal Cordillera	Mid-Boreal	−6.9	−12.5	−1.4	506.1
	High Boreal	−5.9	−11.5	−0.4	469.8
Taiga Cordillera	Low Subarctic	−8.0	−13.2	−2.8	440.6
	High Subarctic	−9.6	−14.7	−4.6	443.6

[a] *Mean annual values are calculated from mean monthly values.*

Table 7. Growing degree days (GDD) and climate moisture index in level III ecoregions of the Northwest Territories, 1981–2010. (Data supplied by J. Beckers, Natural Resources Canada.)

			Climate moisture index[b]				
Level II ecoregion	**Level III ecoregion**	**GDD (5°C base)[a]**	**Annual**	**Winter**	**Spring**	**Summer**	**Fall**
Taiga Plains	Mid-Boreal	1029	3.1	5.6	1.3	–9.0	5.3
	High Boreal	921	8.0	6.0	2.5	–6.8	6.3
	Low Subarctic	802	3.4	5.1	2.1	–9.4	5.6
	High Subarctic	589	3.8	4.2	2.7	–8.1	5.0
Taiga Shield	Mid-Boreal	996	2.0	4.9	1.3	–9.6	5.4
	High Boreal	852	6.5	5.3	2.9	–7.9	6.3
	Low Subarctic	700	10.6	5.1	4.3	–5.9	7.1
	High Subarctic	572	12.1	4.6	4.8	–4.3	7.0
Boreal Cordillera	Mid-Boreal	442	34.0	11.1	8.1	2.6	12.2
	High Boreal	719	13.0	8.1	3.6	–6.0	7.2
Taiga Cordillera	Low Subarctic	559	20.7	8.4	5.4	–2.4	9.3
	High Subarctic	396	27.7	8.5	6.9	1.4	10.9

[a] *GDD was calculated using mean monthly temperature rather than mean daily temperature.*

[b] *Winter is December–February, spring is March–May, summer is June–August, fall is September–November.*

Table 8. Growing degree days (GDD) and climate moisture index in level III ecoregions of the Northwest Territories, 1961–1990. (Data supplied by J. Beckers, Natural Resources Canada.)

			Climate moisture index[b]				
Level II ecoregion	**Level III ecoregion**	**GDD (5°C base)[a]**	**Annual**	**Winter**	**Spring**	**Summer**	**Fall**
Taiga Plains	Mid-Boreal	1018	1.0	5.9	1.2	–10.4	4.3
	High Boreal	909	5.5	6.1	2.5	–8.4	5.2
	Low Subarctic	756	4.3	5.2	2.3	–8.6	5.4
	High Subarctic	523	5.9	4.1	2.7	–6.2	5.2
Taiga Shield	Mid-Boreal	957	–0.3	5.0	0.7	–10.7	4.8
	High Boreal	805	4.4	5.0	2.1	–8.6	5.9
	Low Subarctic	625	9.5	4.7	3.4	–5.3	6.8
	High Subarctic	499	11.2	3.8	3.8	–3.5	7.1
Boreal Cordillera	Mid-Boreal	380	28.5	8.2	6.5	3.0	10.9
	High Boreal	630	19.1	7.3	5.3	–1.4	7.8
Taiga Cordillera	Low Subarctic	481	21.4	6.9	5.3	–0.2	9.4
	High Subarctic	320	25.9	7.5	6.0	1.5	10.8

[a] *GDD was calculated using mean monthly temperature rather than mean daily temperature.*

[b] *Winter is December–February, spring is March–May, summer is June–August, fall is September–November.*

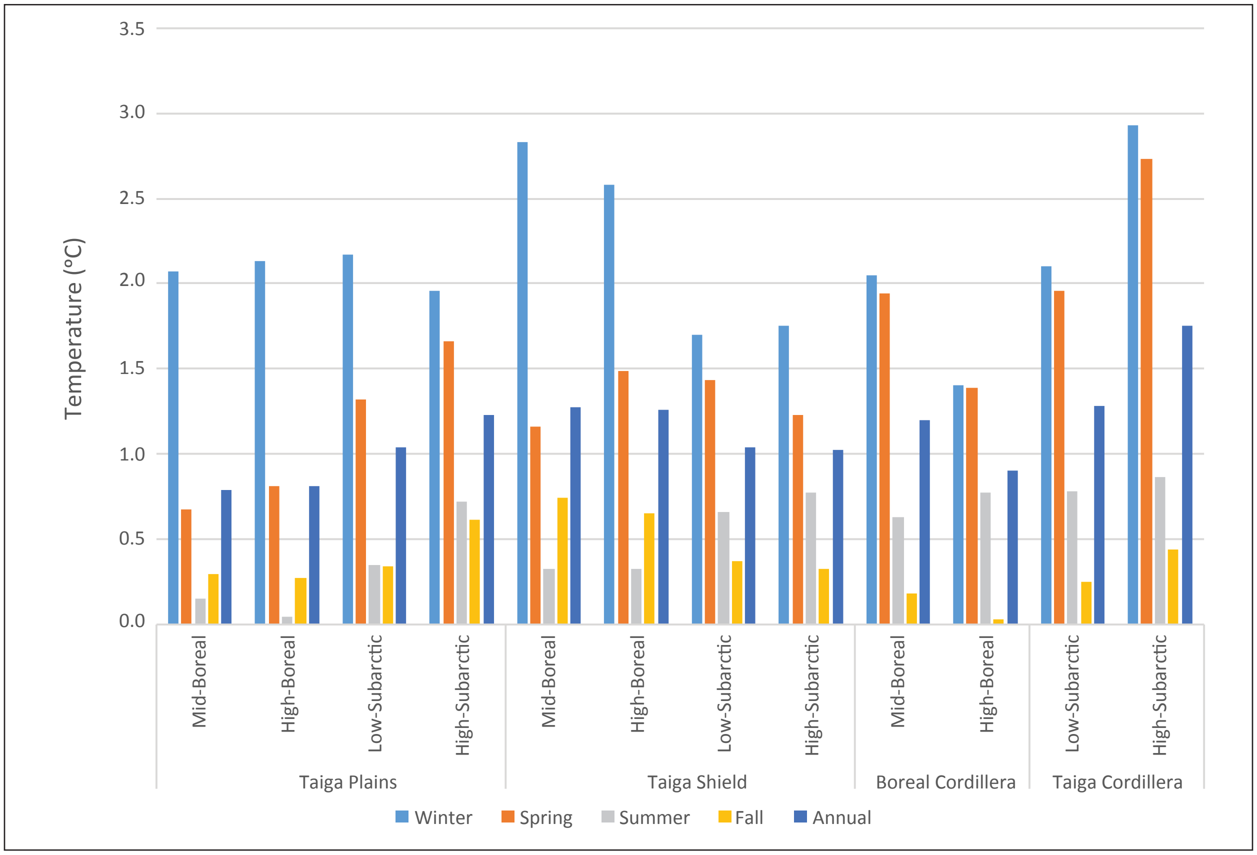


Figure 8. Seasonal and annual temperature increases (based on climate normals) between 1961–1990 and 1981–2010 for level III ecoregions of the Northwest Territories. (Data supplied by J. Beckers, Natural Resources Canada.)

Mean annual temperature followed a linear increasing trend in all ecoregions between 1950 and 2010, with the increase ranging from 2.1°C in the Taiga Plains Mid and High Boreal ecoregions to 3.9°C in the Taiga Cordillera High Subarctic ecoregion (Figure 9, Table 9).

The northern tree line of the boreal forest in western Canada appears to run along the 12°C warm-month mean isotherm (Brandt 2009). The mean of the warmest month in the Taiga Shield High Subarctic ecoregion (July) was just above this level, at 12.5°C during the period 1961–1990 (Table 2). The July mean for the High Subarctic ecoregion rose to 13.5°C for the period 1981–2010 (Table 1). The July mean for the Taiga Shield Low Subarctic ecoregion north of the tree line during the period 1961–1990 was 13.6°C but rose to 14.6°C during the period 1981–2010. The increase in the warm-month mean averaged over the ecoregion implies a likely northward shift of the tree line.

On average, annual temperatures increased across all ecoregions by 1.1°C between the 1961—1990 and 1981—2010 climate normal periods. Winter was the season with the greatest average increase at 2.1°C (Table 1 and Table 2).

2.3 Precipitation

For all ecoregions, precipitation was highest in the summer, with highs for the three summer months combined in the Taiga Plains ecoregion ranging from 88.0 mm in the High Subarctic to 162.8 mm in the High Boreal during the period 1981–2010 and from 102.0 mm in the High Subarctic to 155.2 mm in the High Boreal during the period 1961–1990. Precipitation was lowest in the spring months, with values in the Taiga Plains ecoregion ranging from 36.5 mm in the High Subarctic to 66.8 mm in the High Boreal during the period 1961–1990 and from 38.7 mm in the High Subarctic to 67.7 mm in the High Boreal during the period 1981–2010 (Table 3 and Table 4). Annual precipitation in the Taiga Plains ecoregion ranged from 249.1 mm in the High Subarctic to 376.0 mm in the High Boreal during the period 1961–1990 and from 236.8 mm in the High Subarctic to 393.4 mm in the High Boreal during the period 1981–2010 (Table 5 and Table 6).

Figure 9. Linear trend of mean annual temperature from 1950 to 2010 for level III ecoregions of the Northwest Territories. (Data supplied by J. Beckers, Natural Resources Canada.)

Table 9. Linear trend of mean annual temperature from 1951 to 2010, by level III ecoregion, Northwest Territories. (Data supplied by J. Beckers, Natural Resources Canada.)

Level II ecoregion	Level II ecoregion	Linear trend, 1951–2010 (°C)
Taiga Plains	Mid-Boreal	2.1
	High Boreal	2.1
	Low Subarctic	2.3
	High Subarctic	2.8
Taiga Shield	Mid-Boreal	2.5
	High Boreal	2.4
	Low Subarctic	2.2
	High Subarctic	2.2
Boreal Cordillera	Mid-Boreal	3.1
	High Boreal	2.4
Taiga Cordillera	Low Subarctic	3.1
	High Subarctic	3.9

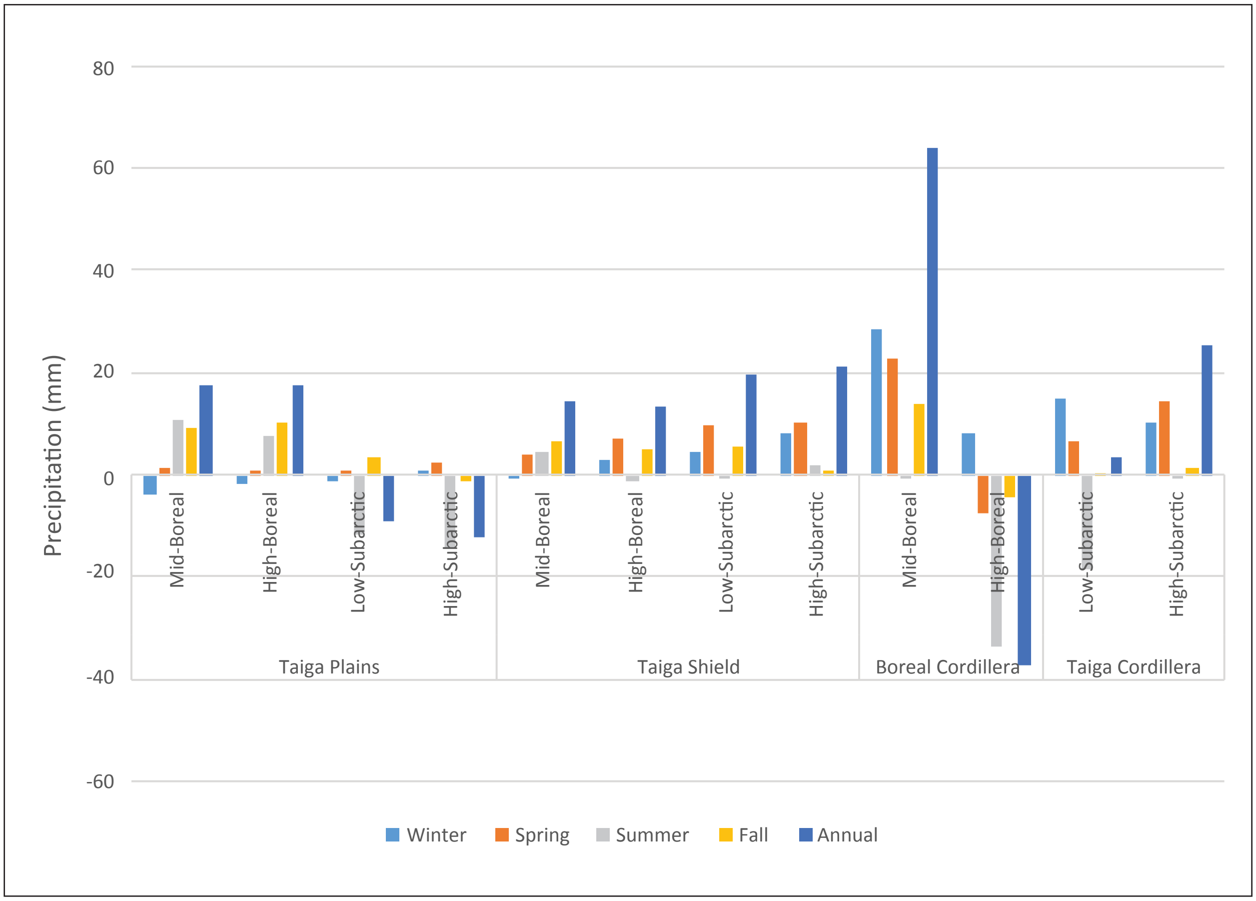


Figure 10. Seasonal and annual changes in precipitation (based on climate normals) between 1961–1990 and 1981–2010 for level III ecoregions of the Northwest Territories. (Data supplied by J. Beckers, Natural Resources Canada.)

Between the climate normal periods of 1961–1990 and 1981–2010, precipitation increased in some seasons and some ecoregions and decreased in others. In the Taiga Plains ecoregion, summer precipitation increased by 11 mm (+7.8%) in the Mid-Boreal and 7.6 mm (+5.0%) in the High Boreal, and decreased by 11.7 mm (−9.0%) in the Low Subarctic and 14.0 mm (−13.7%) in the High Subarctic (Figure 10). Also in the Taiga Plains ecoregion, annual precipitation increased by 17.7 mm (+5.0%) in the Mid-Boreal and 17.4 mm (+4.6%) in the High Boreal, and decreased by 9.0 mm (−2.9%) in the Low Subarctic and 12.3 mm (−4.9%) in the High Subarctic (Figure 10).

On average, annual precipitation increased across all ecoregions by 11 mm between the 1961—1990 and 1981—2010 climate normal periods. The greatest increase in annual precipitation was measured in the Boreal Cordillera Mid Boreal ecoregion (64 mm), while the greatest decrease occurred in the Boreal Cordillera High Boreal ecoregion (37.2 mm) (Table 5 and Table 6).

2.4 Climate Moisture Index

The climate moisture index (CMI) represents the moisture available to plants and is calculated on a monthly basis by subtracting potential evapotranspiration from mean precipitation, where potential evapotranspiration is calculated from monthly means of daily minimum and maximum temperatures, as well as elevation (Hogg 1997). The annual CMI is calculated as the sum of monthly values. The southern limit of the boreal forest closely follows the zero isoline of the CMI. South of that line, annual evapotranspiration is greater than mean annual precipitation, and the forest gives way to open parkland (Hogg 1994). Wang et al. (2014) calculated CMI trends for the boreal forest from 1951 to 2010 according to level II ecoregions. Within the Northwest Territories, they found a significant drying trend in the northern Taiga Plains ecoregion and neighboring Taiga Cordillera ecoregion, but no significant trends elsewhere in the territory (Wang et al. 2014). For this report,

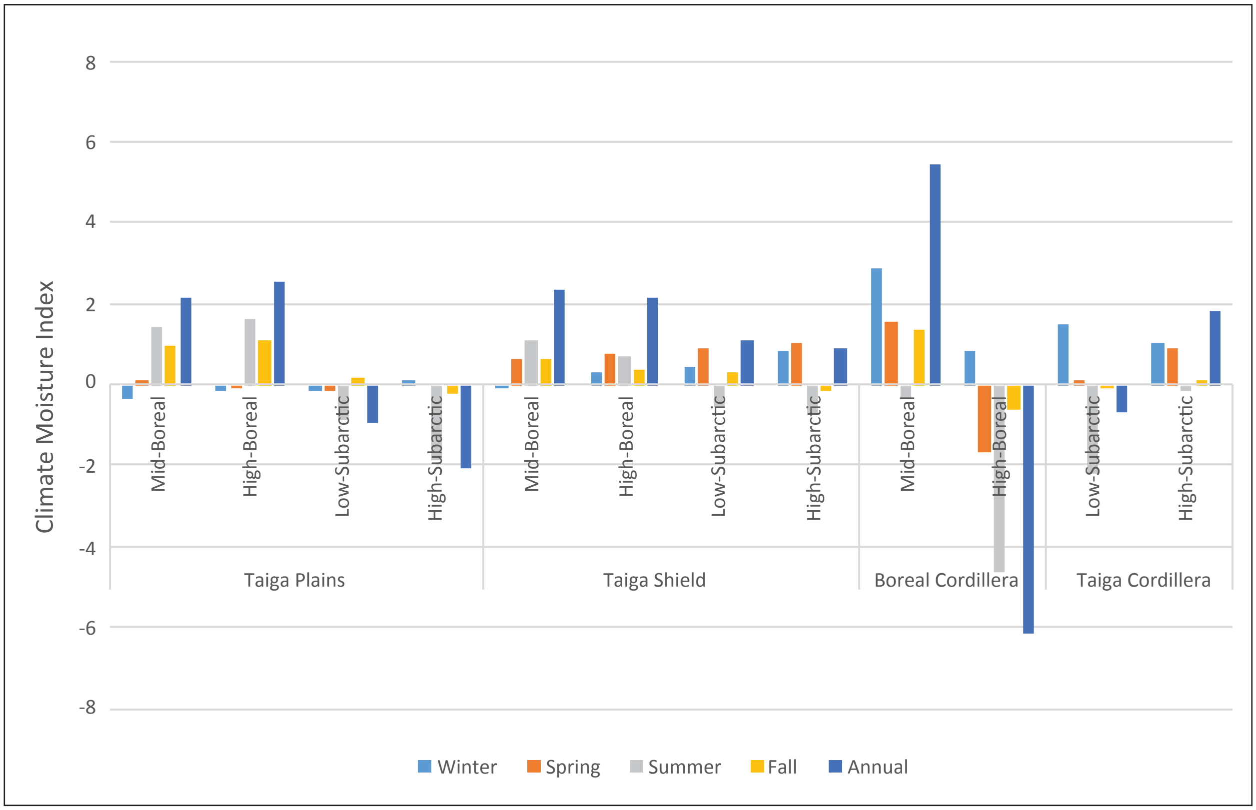


Figure 11. Seasonal and annual changes in climate moisture index (CMI) (based on climate normals) between 1961–1990 and 1981–2010 for level III ecoregions of the Northwest Territories. (Data supplied by J. Beckers, Natural Resources Canada.)

CMI normals for the periods 1961–1990 and 1981–2010 were calculated for the smaller-scale level III ecoregions using the method presented by Hogg (1997).

From 1981 to 2010, the mean annual CMI ranged from 2.0 in the Taiga Shield Mid-Boreal ecoregion to 34.0 in the Boreal Cordillera Mid-Boreal ecoregion (Table 7). Soil moisture in the Northwest Territories is limited only in the summer, and the summer values of CMI are therefore the most important. Summer CMI in the period 1961–1990 ranged from −10.7 in the Taiga Shield Mid-Boreal ecoregion to 3.0 in the Boreal Cordillera Mid-Boreal ecoregion (Table 8). Changes in CMI between the climate normal periods 1961–1990 and 1981–2010 generally tracked changes to precipitation, although increases were dampened and decreases enhanced by the increases in temperature (Figure 11). In the Taiga Plains, summer CMI increased between the two climate normal periods, by 1.4 in the Mid-Boreal and 1.6 in the High Boreal, but decreased by 0.8 in the Low Subarctic and 1.9 in the High Subarctic (Figure 11).

2.5 Growing Degree Days

Growing degree days (GDD) is the annual sum of positive deviations in daily mean temperature from a base temperature, typically 5°C (Hogg 1994). Values of GDD are calculated as $\Sigma(T_{max} + T_{min})/2\text{-}T_{Base}$ where $(T_{max} + T_{min})/2\text{-}T_{Base} \geq 0$ (McMaster and Wilhelm 1997). For this analysis, GDD values were calculated from mean monthly temperatures, rather than directly from daily averages; therefore, only months with mean temperature above 5°C contributed toward the annual GDD. This approach gives significantly lower values of GDD than if daily averages are used. The months of May–September contributed data for the Mid-Boreal ecoregion, June–September for the High Boreal ecoregion, and June–August for the Low and High Subarctic ecoregions. As such, these periods can serve as rough estimates of the growing season in these regions. From 1981 to 2010, the average annual GDD in the Taiga Plains was 1029 in the Mid-Boreal ecoregion (the highest value of all ecoregions), 921 in the High Boreal ecoregion, 802 in the Low Subarctic ecoregion, and 589 in the High Subarctic ecoregion (Table 7). The lowest GDD was 396, in the Taiga Cordillera High Subarctic ecoregion. According to data sets for 1961–1990 and 1981–2010, GDD in the Taiga Plains ecoregion grew by 11 (1.1%) in the Mid-Boreal, 12 (1.3%) in the High Boreal, 46 (6.1%) in the Low Subarctic, and 66 (12.6%) in the High Subarctic (Figure 12).

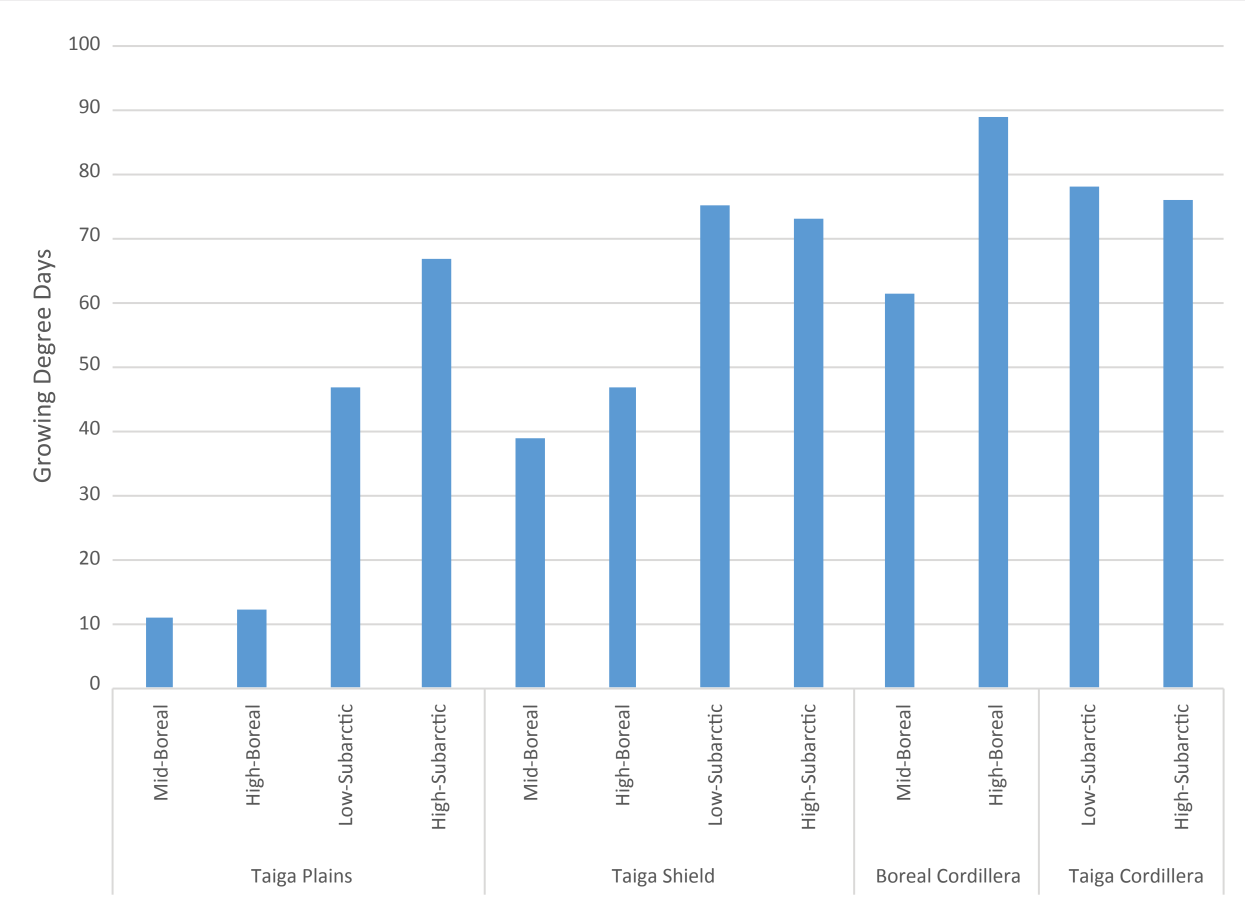


Figure 12. Increases in growing degree days (GDD) using a 5°C base temperature (based on climate normals) between 1961–1990 and 1981–2010 for level III ecoregions of the Northwest Territories. (Data supplied by J. Beckers, Natural Resources Canada.)

Key Points

- Gridded climate normals were created from weather station data using thin-plate spline modeling.
- Climate normals for 1961–1990 and 1981–2010 and a linear trend of annual temperature from 1951 to 2010 were created for level III ecoregions.
- Temperatures increased from the period 1961–1990 to the period 1981–2010 in all ecoregions and all seasons. The average increase in annual temperature across ecoregions between the climate normal periods was 1.1°C. Winter was the season with the greatest average increase (2.1°C).
- Mean annual temperature increased linearly between 1950 and 2010, by 2.1°C to 3.9°C.
- Precipitation trends were more varied, with increases in some ecoregions and some seasons and decreases in others. Average annual precipitation across all ecoregions increased by 11 mm between the climate normal periods, with the greatest increase (64 mm) occurring in the Boreal Cordillera Mid-Boreal ecoregion and the greatest decrease (37.2 mm) occurring in the Boreal Cordillera High Boreal ecoregion.
- The climate moisture index is influenced by changes in precipitation and temperature and showed variable trends between the two climate normal periods across months and ecoregions, with a small average annual increase of 0.7 overall.
- The growing season, measured in terms of growing degree days (GDD) above 5°C, increased in all ecoregions, with the greatest increases in the Cordillera ecoregions.

Knowledge Gaps

- The number of climate stations in the Northwest Territories is small, relative to the large territorial area, which increases the uncertainty of the climate modeling used to produce these data.

CHAPTER 3. FOREST TYPES AND PRODUCTIVITY

3.1 Introduction

In the Northwest Territories, forests are found in the Taiga Plains, Taiga Shield, and Cordillera ecoregions, with the tree line roughly following the border between those ecoregions and the Southern Arctic ecoregion (Department of Environment and Natural Resources [N.W.T.] 2015). All of these forests are part of the boreal forest, the vast forest that circles the northern hemisphere. In addition to Canada, the boreal forest can be found within the borders of the United States, Norway, Sweden, Finland, Russia, China, and Japan, with approximately 30% of the 1.9 Gha (1.9 × 10^9 hectares) total area being located in Canada (Natural Resources Canada 2020). The boreal forest of the Northwest Territories stretches from the southern border of the territory north to the tree line over an area of 80 Mha (80 × 10^6 ha), of which an area of 44 Mha is considered forested (EOSD 2017). The vast extent of the forested area, the remote location, and the absence of a large forestry industry have resulted in fewer forest inventories than in the rest of Canada (Van der Sluijs et al. 2016).

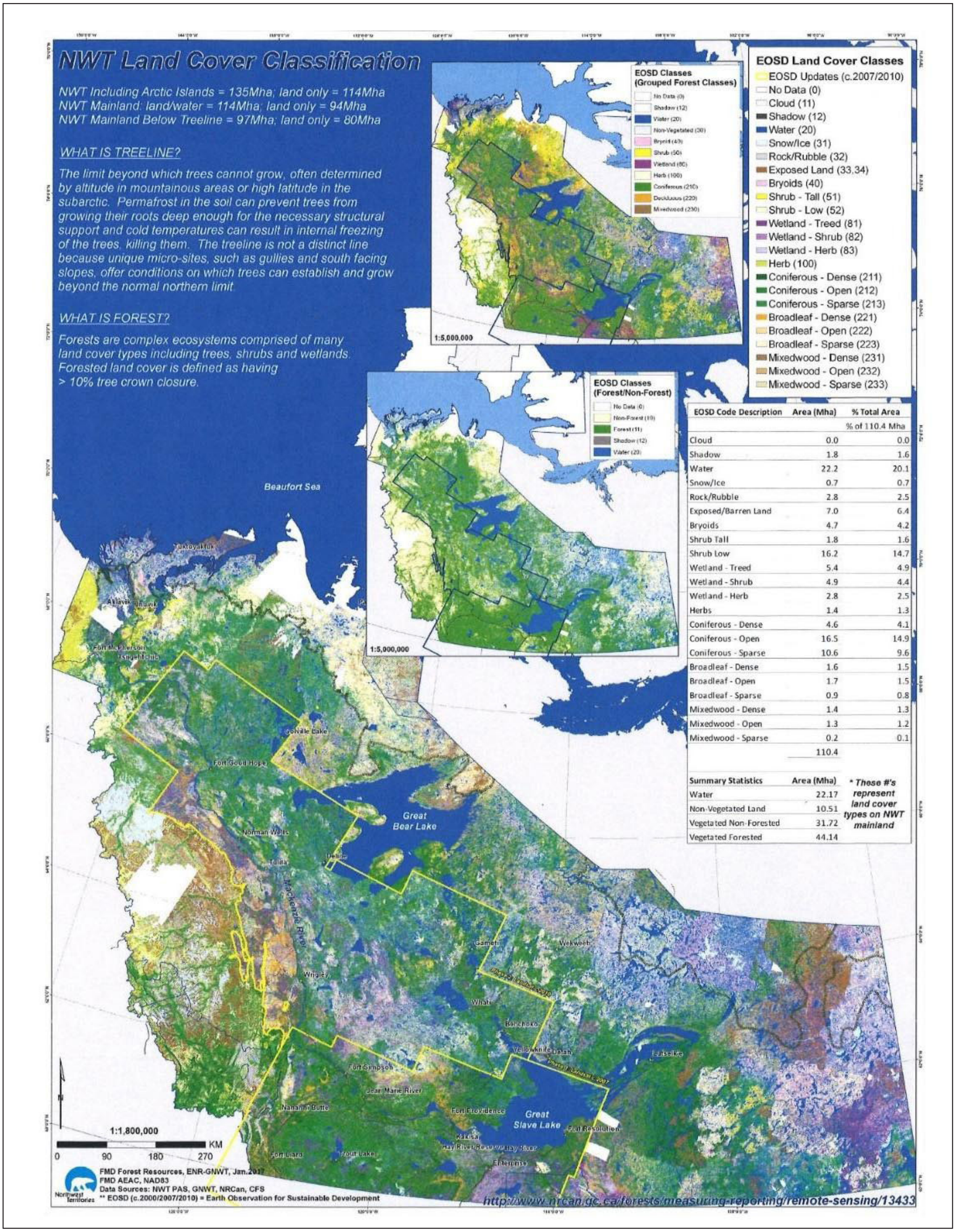
3.2 Forest Inventory

Forest inventories of Canada’s commercially managed forests and forests within protected areas benefit from validation by means of detailed ground-based inventory data (Stinson et al. 2011). However, because most Northwest Territories forests are not managed for commercial purposes, this type of detailed forest inventory is not widely available, which means that forest managers often must rely on less accurate satellite imagery for land cover information (Castilla et al. 2016). Nevertheless, several research projects in the Northwest Territories are now collecting detailed plot-level data about the forests, including fire return intervals and postfire vegetation recovery trajectories; these studies may be able to provide detailed plot-based forest inventory data for some portions of the forested area (see chapter 5 for more information on fire research projects currently underway in the Northwest Territories).

In 2006, a land cover map of Canada’s forested areas was produced through a collaboration of CFS, the Canadian Space Agency, and the provincial and territorial governments, as part of the Earth Observation for the Sustainable Development of Forests (EOSD) project, using Landsat 7 satellite images taken about the year 2000 (Wulder et al. 2008) (Figure 13). This land cover map was later updated for portions of the Northwest Territories using Landsat images from about 2007 (southern area in Figure 13) and about 2010 (northern area in Figure 13) (EOSD 2017). These EOSD maps still represent the most spatially complete picture of Northwest Territories forests (Mahoney et al. 2018). The latest EOSD map shows that the Northwest Territories has 44 Mha of forested area, defined as land with greater than 10% crown closure, spread over 80 Mha of forest lands below the tree line. Based upon these results, the forested area can be classified as 17.2% dense forest (>60% crown closure), 44.1% open forest (26%–60% crown closure), and 26.5% sparse forest (10%–25% crown closure), with the remaining 12.2% of the area classified as treed wetlands (Table 10). These forests are predominantly made up of coniferous stands (about 72%), followed by deciduous stands (about 10%), mixed-species stands (about 6%), and treed wetlands (about 12%) (Table 10) (Wulder and Nelson 2003; EOSD 2017).

In addition to the EOSD land cover map, three other land cover data sets have been created for Canada from satellite imagery: the SPOT 4/5 Medium Resolution Land Cover Mapping of Canada; the Moderate Resolution Imaging Spectroradiometer (MODIS) Land Cover Time Series; and a map created by CFS researchers combining photographic plots from the National Forest Inventory (NFI), MODIS data, and k-nearest neighbor computation to reflect forest conditions in 2001 (Beaudoin et al. 2014). An update of the 2001 map and a new map for 2011 have also recently been created (Beaudoin et al. 2018). Each of the four data sets has estimated accuracy of about 75%–80%, and agreement between them is about 60%, with most of the discrepancies being due to misclassification (Castilla et al. 2016). Thus, none of the maps provides a sufficiently accurate picture of forest cover in the Northwest Territories, particularly for uses such as estimating carbon fluxes (Castilla et al. 2016).

Currently, it is estimated that only 10% of Northwest Territories forests have been inventoried at a spatial resolution high enough for assessing and monitoring the state of the forests at an operational level (unpublished data, provided by G. Castilla, 2018). Filling this gap will be difficult for an area as large and remote as the Northwest



NWT Land Cover Classification

NWT Including Arctic Islands = 135Mha; land only = 114Mha
NWT Mainland: land/water = 114Mha; land only = 94Mha
NWT Mainland Below Treeline = 97Mha; land only = 80Mha

WHAT IS TREELINE?

The limit beyond which trees cannot grow, often determined by altitude in mountainous areas or high latitude in the subarctic. Permafrost in the soil can prevent trees from growing their roots deep enough for the necessary structural support and cold temperatures can result in internal freezing of the trees, killing them. The treeline is not a distinct line because unique micro-sites, such as gullies and south facing slopes, offer conditions on which trees can establish and grow beyond the normal northern limit.

WHAT IS FOREST?

Forests are complex ecosystems comprised of many land cover types including trees, shrubs and wetlands. Forested land cover is defined as having > 10% tree crown closure.

EOSD Code Description	Area (Mha)	% Total Area
		% of 110.4 Mha
Cloud	0.0	0.0
Shadow	1.8	1.6
Water	22.2	20.1
Snow/Ice	0.7	0.7
Rock/Rubble	2.8	2.5
Exposed/Barren Land	7.0	6.4
Bryoids	4.7	4.2
Shrub Tall	1.8	1.6
Shrub Low	16.2	14.7
Wetland - Treed	5.4	4.9
Wetland - Shrub	4.9	4.4
Wetland - Herb	2.8	2.5
Herbs	1.4	1.3
Coniferous - Dense	4.6	4.1
Coniferous - Open	16.5	14.9
Coniferous - Sparse	10.6	9.6
Broadleaf - Dense	1.6	1.5
Broadleaf - Open	1.7	1.5
Broadleaf - Sparse	0.9	0.8
Mixedwood - Dense	1.4	1.3
Mixedwood - Open	1.3	1.2
Mixedwood - Sparse	0.2	0.1
	110.4	

Summary Statistics	Area (Mha)
Water	22.17
Non-Vegetated Land	10.51
Vegetated Non-Forested	31.72
Vegetated Forested	44.14

* These #'s represent land cover types on NWT mainland

FMD Forest Resources, ENR-GNWT, Jan.2017
FMD AEAC, NAD83
Data Sources: NWT PAS, GNWT, NRCan, CFS
** EOSD (c.2000/2007/2010) = Earth Observation for Sustainable Development

http://www.nrcan.gc.ca/forests/measuring-reporting/remote-sensing/13433

Figure 13. Earth Observation for the Sustainable Development of Forests land cover map for the forested regions of the Northwest Territories. (Source: Natural Resources Canada, http://www.nrcan.gc.ca/forests/measuring-reporting/remote-sensing/13433, accessed 22 Apr. 2020.)

Table 10. Area and proportion of forested landscape[a] in the Northwest Territories by stand type,[b] based on the EOSD[c] land cover map. (Table based on EOSD [2017].)

	Coniferous		Deciduous		Mixed		Treed wetland		Total	
Stand type	Area (Mha)	% of total	Area (Mha)	% of total	Area (Mha)	% of total	Area (Mha)	% of total	Area (Mha)	% of total
Dense	4.6	10.4	1.6	3.6	1.4	3.2	–	–	7.6	17.2
Open	16.5	37.3	1.7	3.8	1.3	2.9	–	–	19.5	44.1
Sparse	10.6	24.0	0.9	2.0	0.2	0.5	–	–	11.7	26.5
Wetland	–	–	–	–	–	–	5.4	12.2	5.4	12.2
Total	31.7	71.7	4.2	9.5	2.9	6.6	5.4	12.1	44.2	100

[a] *Definitions of forested landscapes: coniferous forest has >75% basal area consisting of coniferous trees, deciduous forest has >75% basal area consisting of deciduous trees, mixed forest has neither coniferous nor deciduous trees accounting for >75% of basal area, treed wetland has trees accounting for most of its vegetation.*

[b] *Definitions of stand types: dense forest has >60% crown closure, open forest has 26%–60% crown closure, sparse forest has 10%–25% crown closure,*

[c] *EOSD = Earth Observation for the Sustainable Development of Forests.*

Territories, where field data are expensive to collect, few previous remote sensing inventories have been conducted, and the landscape is highly diverse (Skakun et al. 2007). Nevertheless, a collaborative project between CFS and the Department of Environment and Natural Resources of GNWT is in progress to create a forest inventory covering most forests in the Taiga Plains ecoregion, an area of 44 Mha. This Multi-source Vegetation Inventory (MVI) project will use field plots (Figure 14), light detection and ranging (LiDAR) images, radar images, Landsat satellite images, and existing forest inventory data to derive full-coverage 30 × 30 m rasters estimating stand height, crown closure, stand volume, aboveground biomass, and stand age (Natural Resources Canada 2018b). Once completed, this inventory will have many uses, including development of future carbon budget models, planning for postfire salvage logging, modeling of habitat suitability for caribou, and ecological land classification. In addition, the MVI initiative will produce a polygon vector layer resembling a traditional forest inventory map, for use by forest managers. Work is also underway to update forest inventory information to reflect current conditions, which is important because previous mapping did not incorporate changes to the landscape due to the large fires that occurred in 2013 and 2014 in the Northwest Territories.

On the basis of work completed on this project so far, several important forest stand characteristics can be summarized by level III ecoregion (Table 11, Table 12). Key results include the following:

- Average stand age: 71–77 years (range 10–171 years across all sites).
- Average stand height: 6–10 m (range 4–32 m across all sites).
- Average stand volume: 18–51 $\text{m}^3\cdot\text{ha}^{-1}$ (range 7–357 $\text{m}^3\cdot\text{ha}^{-1}$ across all sites).

As could be expected, the taller and higher-volume stands tend to occur in more southern ecoregions, whereas the trees in more northern ecoregions are typically shorter and smaller. Stand ages are variable across all assessed ecoregions (Table 12).

The CFS, in collaboration with provincial and territorial governments, oversees the NFI, which produces annual statistics on forest type, volume, and biomass across Canada's forests. These data are produced from 4 km^2 aerial and satellite photo plots; in the Northwest Territories, these plots exist along a 20 × 20 km grid and cover 1% of the territory's area. Ground plots are also part of the NFI system, and these are installed on roughly 8% of the photo plot locations to collect information not available from imagery (Figure 15). Both plot types are re-measured every 10-years (National Forest Inventory 2016a, National Forest Inventory 2016b). The quality of information available from the NFI photo plots in the Northwest Territories during the baseline period (2000 – 2006) was low, largely because of the lower resolution imagery available for northern Canada at that time, so earlier data should be used with caution (Beaudoin et al. 2014). However, during the first NFI re-measurement period (2007 – 2017), new high resolution, high quality imagery was collected throughout northern Canada. New reports utilizing these data have been published, but are not included in this report due to timing constraints. The NFI organizes its data by ecozone, rather than by provincial or territorial jurisdiction, so information is presented here only for the Taiga Plains ecozone, almost all of which falls within the Northwest Territories.

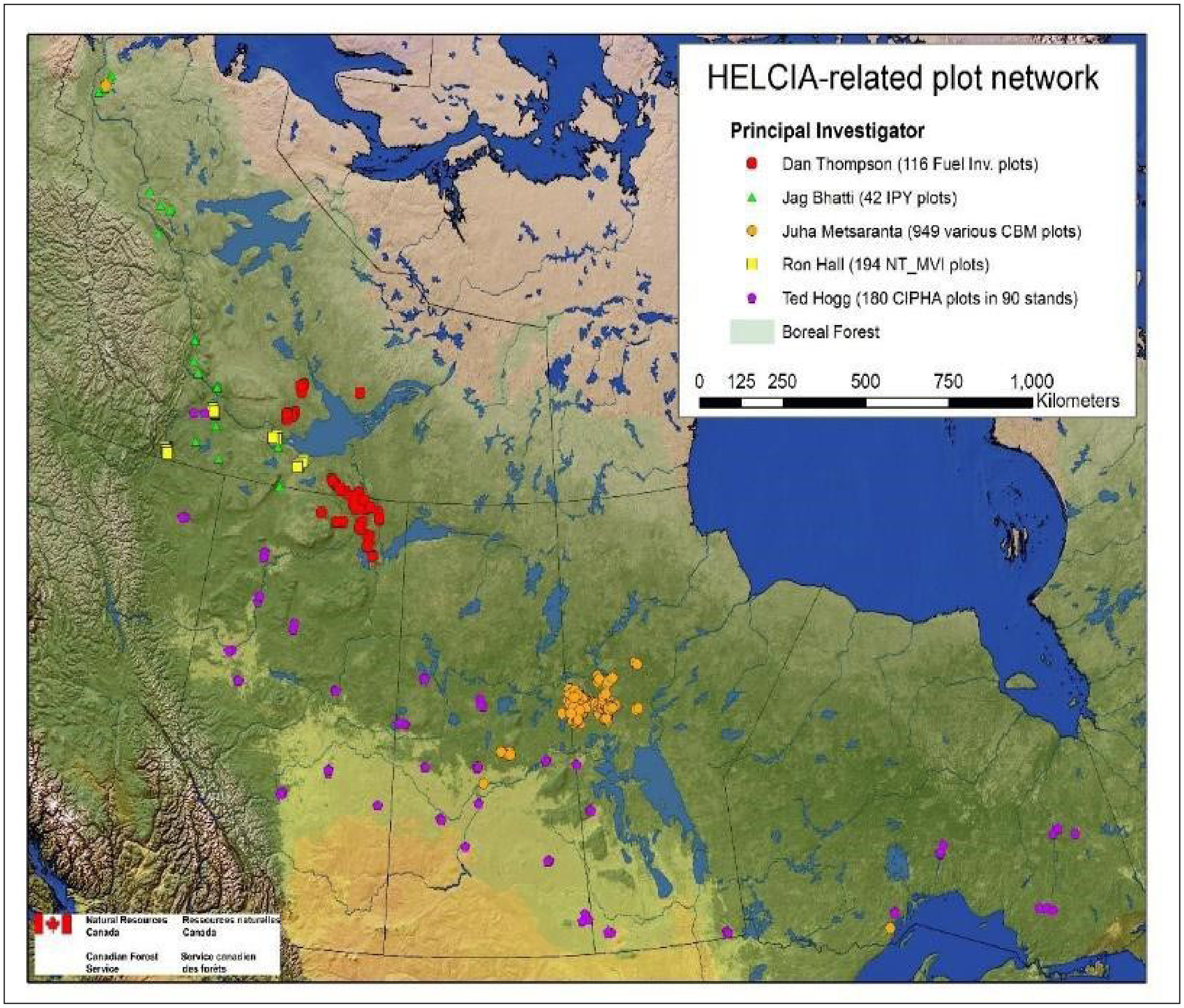


Figure 14. Location of plots established as part of the High Elevation and Latitude Climate Change Impacts and Adaptation (HELCIA) project, including those part of the Multi-source Vegetation Inventory (MVI) and Climate Impacts on Productivity and Health of Aspen (CIPHA) projects. (Source: Natural Resources Canada.)

Table 11. Coverage statistics, by region, from the Multi-source Vegetation Inventory (MVI) project. (Data supplied by R. Skakun, Natural Resources Canada.)

Region	Total area (ha)	% covered by MVI	% upland forest	% treed wetland
Taiga Plains Mid-Boreal	10 717 498	92.00	59.91	7.35
Taiga Plains High Boreal	4 219 858	100.00	66.87	8.60
Taiga Plains Low Subarctic	16 173 735	91.07	42.99	9.57
Taiga Plains High Subarctic	16 953 941	22.77	42.07	7.41
Dehcho	15 868 823	72.84	57.63	10.01
South Slave	20 805 880	36.15	51.42	6.98

Table 12. Mean stand characteristics, by region, from the Multi-source Vegetation Inventory (MVI) project. (Data supplied by R. Skakun, Natural Resources Canada.)

	Stand age (yr)		Stand height (m)		Crown closure (%)		Stand volume ($m^3\cdot ha^{-1}$)		Aboveground biomass ($t\cdot ha^{-1}$)	
Region	Mean ± SD[a]	Range	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD	Range
Taiga Plains Mid-Boreal	76.55 ± 19.71	10.0–171.0	9.85 ± 4.53	4.0–32.0	40.59 ± 8.61	18.0–70.0	50.32 ± 44.47	7.0–357.0	52.29 ± 36.72	1.0–279.0
Taiga Plains High Boreal	73.72 ± 12.57	10.0–149.0	8.48 ± 3.42	4.0–32.0	39.10 ± 7.26	18.0–70.0	36.49 ± 31.88	7.0–357.0	42.05 ± 26.98	1.0–279.0
Taiga Plains Low Subarctic	–[b]	10.0–137.0	6.52 ± 1.65	4.0–29.0	34.87 ± 5.71	17.0–67.0	20.69 ± 12.10	7.0–300.0	30.16 ± 11.21	1.0–232.0
Taiga Plains High Subarctic	–[b]	12.0–114.0	6.16 ± 1.37	4.0–23.0	33.57 ± 5.290	21.0–65.0	18.37 ± 9.16	8.0–195.0	28.47 ± 8.71	17.0–169.0
Dehcho	77.15 ± 24.45	10.0–171.0	9.82 ± 4.99	4.0–32.0	40.83 ± 8.82	18.0–70.0	51.14 ± 50.53	7.0–357.0	52.14 ± 40.05	1.0–279.0
South Slave	71.89 ± 15.85	10.0–149.0	8.77 ± 3.79	4.0–31.0	38.69 ± 8.34	17.0–69.0	40.08 ± 35.13	7.0–341.0	44.04 ± 29.94	1.0–267.0

[a] *SD = standard deviation.*

[b] *Stand age is not reported for the subarctic regions because it was found to be unreliable.*

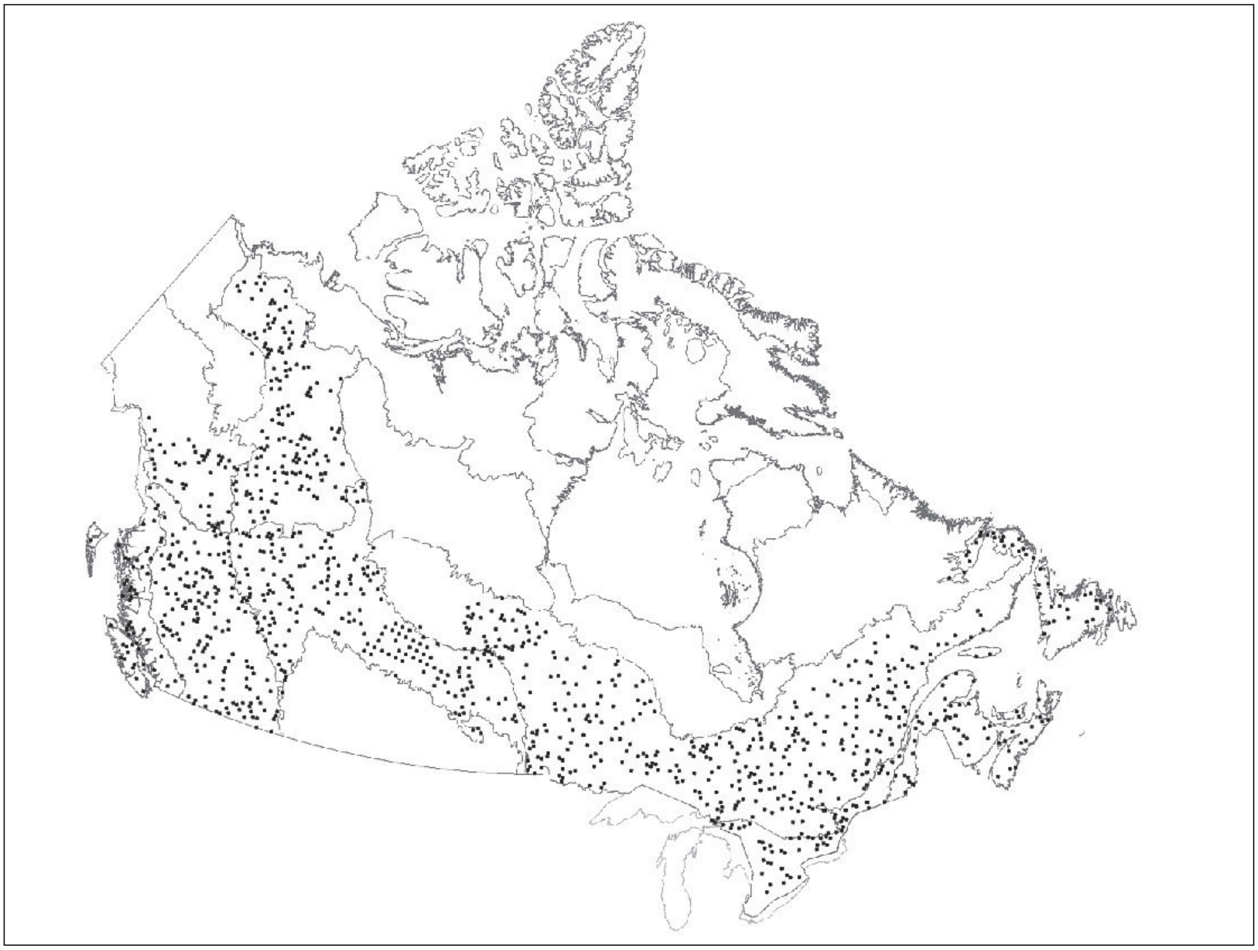


Figure 15. National Forest Inventory ground plot locations. (Source: National Forest Inventory, https://nfi.nfis.org/resources/general/NFI%20 Brochure.pdf, accessed 22 Apr. 2020.)

According to current NFI information, Taiga Plains forests (total 33.6 Mha) are composed primarily of spruce (84.8%, 28.5 Mha), with much smaller components of poplar (12.2%, 4.1 Mha), pine (1.5%, 0.5 Mha), birch (1.0%, 0.34 Mha), and larch (0.3%, 0.09 Mha) (National Forest Inventory 2018) (Table 13). Total aboveground tree biomass in the Taiga Plains follows the same pattern, with most biomass contained in spruce forests, and lesser amounts in poplar, pine, birch, and larch forests (Table 13) (National Forest Inventory 2013a). The dominant age class for coniferous and mixed forests in the Tagia Plains is 81–100 years; deciduous forests in the ecoregion are slightly younger, with the dominant age class in this forest type being 61–80 years (National Forest Inventory 2013b) (Table 14). This pattern reflects the fact that coniferous stands in the boreal forest are often at a later successional stage than broadleaf stands.

Table 13. Taiga Plains forested area and aboveground biomass by tree species. (Data source National Forest Inventory [2013a], National Forest Inventory [2018].)

Tree species	Area (× 1000 ha)	Aboveground biomass (Mt)
Spruce	28 510	2322
Poplar	4092	415
Pine	499	42
Birch	340	25
Larch	94	4
Unclassified	67	0.2
Fir	0.3	0.01
Total	33 602.3	2808.2

3.3 Upland Forests

Upland areas are raised above the surrounding landscape and are therefore moderately to well drained (Ecosystem Classification Group 2007 [rev. 2009]) (Figure 16). Upland soils are mineral soils, although they may also include a thick organic layer (Startsev et al. 2016). In the Taiga Plains ecoregion of the Northwest Territories, upland sites support more productive forests than do peatlands, with a mean biomass of 76.17 $t \cdot ha^{-1}$ in the Mid-Boreal, 67.62 $t \cdot ha^{-1}$ in the High Boreal, 20.21 $t \cdot ha^{-1}$ in the Low Subarctic, and 12.96 $t \cdot ha^{-1}$ in the High Subarctic (Errington et al. 2010) (Figure 17), and generally reaching at least 25% canopy closure and a mean tree height of 5 m (Bona et al. 2013). Upland forests include mixed-wood forests of white spruce and trembling aspen, open to closed black spruce stands, and jack pine stands. Black spruce upland forests have particularly thick organic layers and may be considered intermediate between true upland forests and forested peatlands (Bona et al. 2013). The specific characteristics of upland forest stands, such as species, density, and height, depend on site and microsite conditions, including latitude, elevation, aspect, soil texture, drainage, and fire history. Chapter 1, above, includes brief descriptions of various types of upland forest for each ecoregion.

The presence of permafrost under upland forests in the discontinuous permafrost zones of the Northwest Territories has not been well studied. However, because mineral soil is less insulating than peat, permafrost is generally less likely to be encountered under upland sites than under peatlands. Nevertheless, where permafrost does occur under upland forests, it may be found at deeper depths than in nearby peat plateaus

Table 14. Taiga Plains forested area (× 1000 ha) by forest type and age class. (Data source National Forest Inventory [2013b].)

	Age class (yr)											
Forest type	1–20	21–40	41–60	61–80	81–100	101–120	121–140	141–160	161–180	181–200	≥201	Total
Coniferous	87.3	110.6	546.1	1215.9	22 594.3	605.6	530.3	361.3	85.2	26.4	49.3	26 212.3
Broadleaf	11.2	97.2	671.3	2276.6	361.0	255.6	75.3	17.4	0.4	0	0	3766.0
Mixed-wood	17.0	36.1	371.5	296.5	2462.3	145.9	137.9	79.3	5.1	2.6	2.2	3556.4
Total	115.5	243.9	1588.9	3789.0	25 417.6	1007.1	743.5	458.0	90.7	29.0	51.5	33 534.7

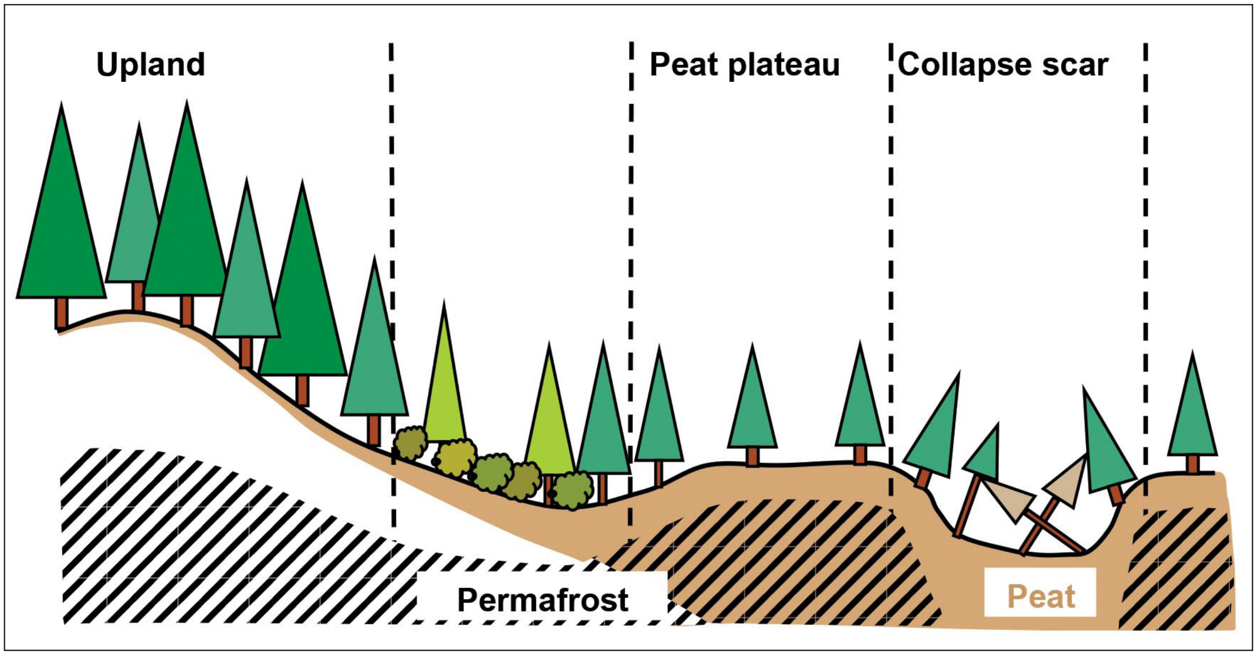


Figure 16. Cross-section of a typical landscape in the Mackenzie Valley, Northwest Territories, showing upland productive forest, a forested peat plateau, and a collapse scar. (Reproduced, with permission, from Errington et al. 2010.)

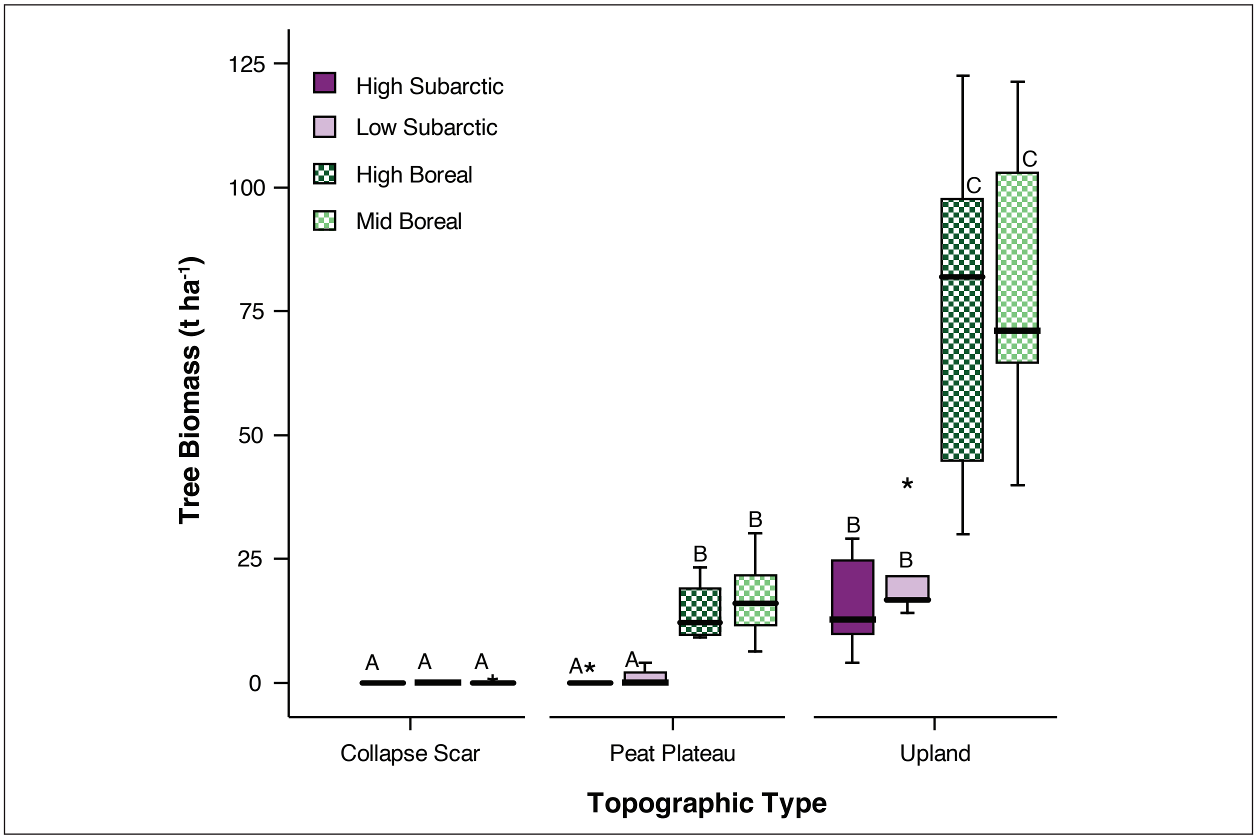


Figure 17. Tree biomass by level III ecoregion and land feature along the Mackenzie Valley, Northwest Territories. Horizontal black bars indicate mean values; bars with different letters indicate significant differences in tree biomass. (Reproduced, with permission, from Errington et al. 2010.)

(Errington et al. 2010). In their study of permafrost in the forested portions of the Mackenzie Valley, Errington et al. (2010) were unable to measure depth to permafrost in upland soils in the discontinuous permafrost zones, but they could not discern whether that was because permafrost was absent from that specific area or because it lay deeper than the 60-cm soil pits (Figure 18).

In a study carried out in the discontinuous permafrost zones of interior, low-elevation Alaska, permafrost was found under upland forests, but only on northern aspects and in valley bottoms (Douglas et al. 2014), which is typical of the discontinuous permafrost zones. That study also noted that the permafrost under upland forests had a lower ice (moisture) content than permafrost underlying peatland forests, a characteristic that makes upland permafrost more thaw-stable than permafrost underlying peatlands (Douglas et al. 2014). Whether this is also true for the discontinuous permafrost zones of the Northwest Territories is unknown.

In the continuous permafrost zone of the Taiga Plains High Subarctic ecoregion, Errington et al. (2010) recorded a median depth to permafrost across six upland plots of 37.5 cm, whereas across two plots in the Taiga Plains Low Subarctic ecoregion, median depth to permafrost was 54.7 cm (Figure 18).

3.4 Forested Peatlands

A peatland is any wetland with more than 40 cm of peat (partially decomposed plant matter accumulated in an oxygen-deprived, saturated environment [Shaw et al. 2016]), and forested peatlands occur all across Canada (Figure 19). In the Northwest Territories, the most common types of peatland are fens and bogs, which are differentiated primarily by the source of their water supply. Fens receive the bulk of their water from surface water or groundwater and are therefore typically rich in dissolved minerals (National Wetlands Working Group 1997). In contrast, bogs receive most of their water from precipitation, which is mineral-poor; as a result, bogs tend to be acidic, with pH between 4.0 and 4.8 (National Wetlands Working Group 1997).

Additional defining features of fens include fluctuating water levels and surface water flow through channels, pools, and other open water

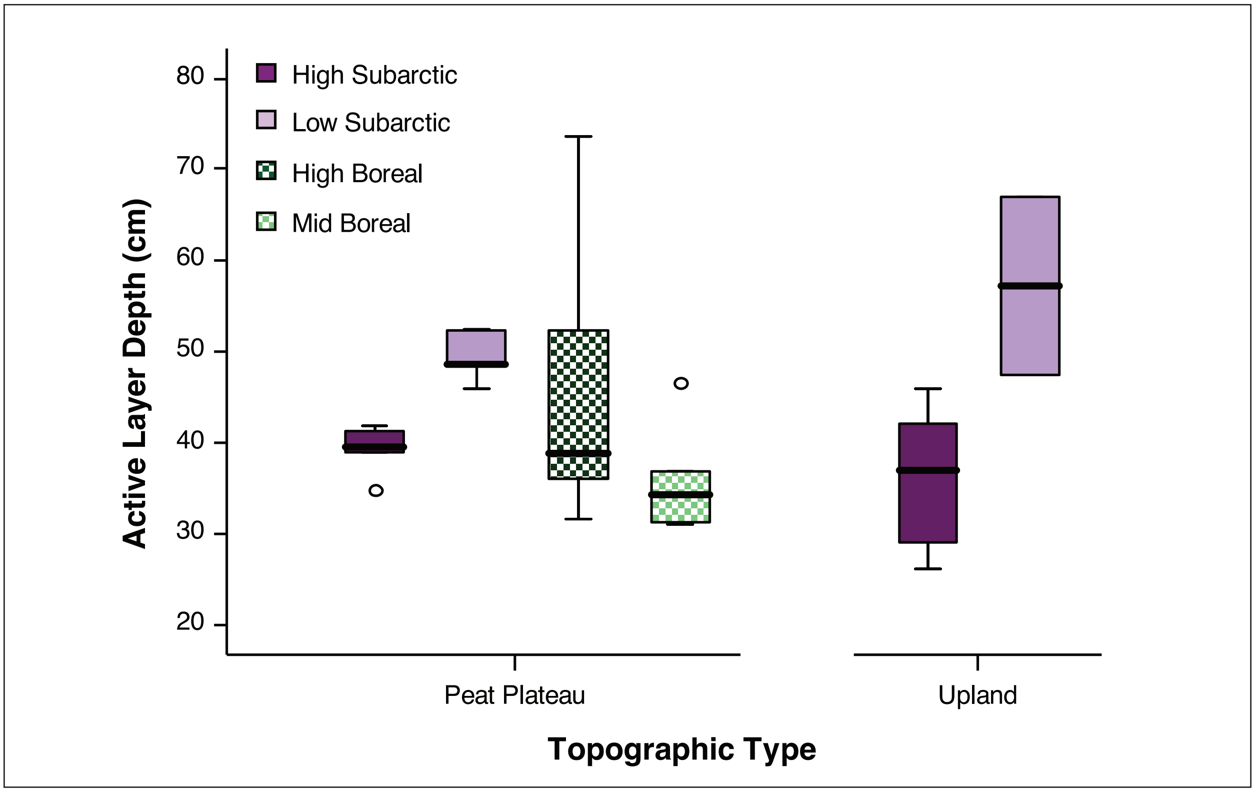


Figure 18. Active-layer depth by level III ecoregion and land feature in the Taiga Plains ecoregion along the Mackenzie Valley, Northwest Territories. Black horizontal bars indicate median values. (Reproduced, with permission, from Errington et al. 2010.)

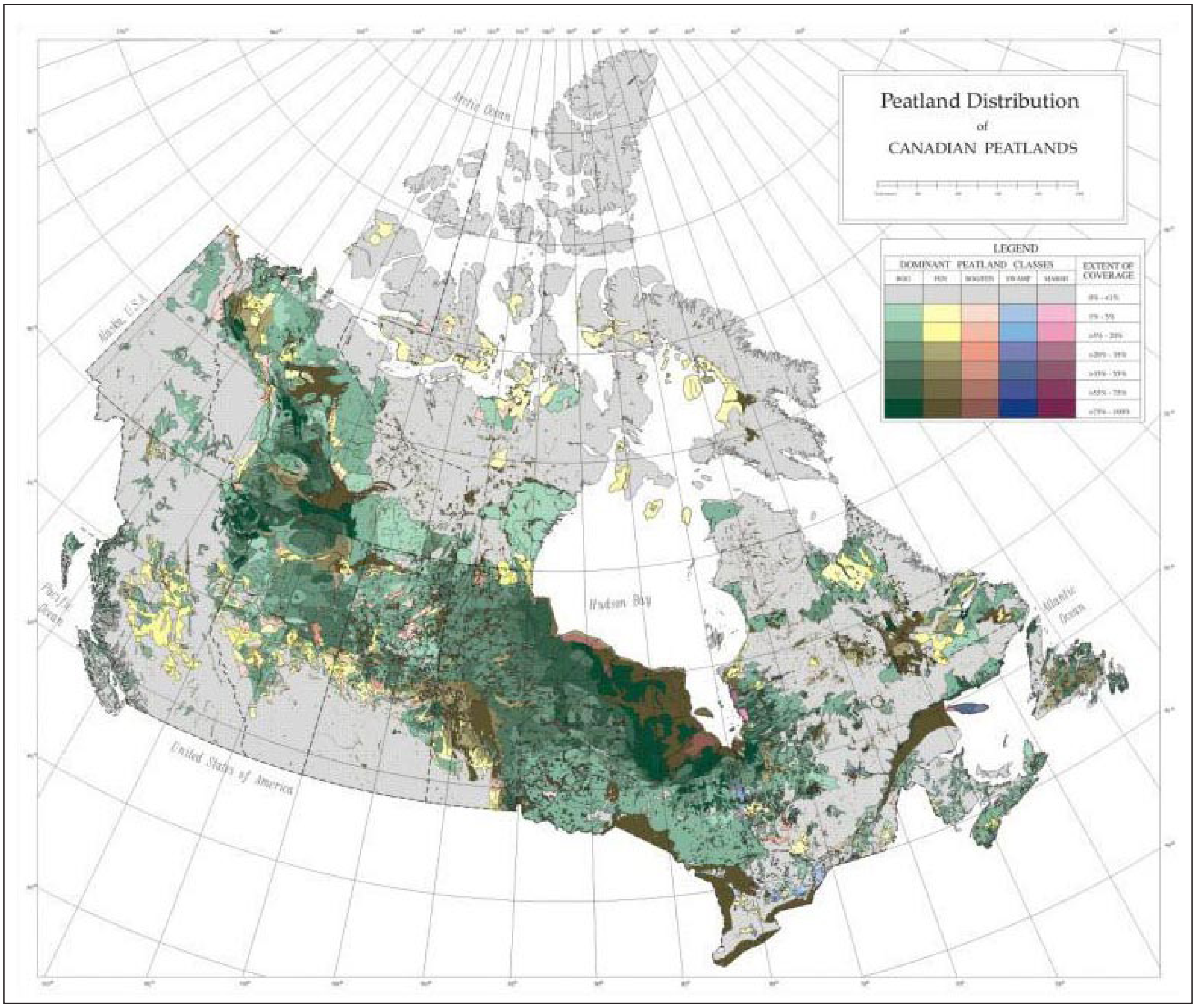


Figure 19. Map of the peatlands of Canada. (Reproduced, with permission, from Tarnocai et al. 2011.)

bodies, which often form characteristic surface flow patterns. Fen vegetation communities are closely related to the depth to the water table and the water chemistry of the fen (National Wetlands Working Group 1997). In general, grassy vegetation and some mosses dominate wet fens, where the water table is at the surface; shrub species dominate drier fens, where the water table is lower; and trees can be present on the driest fens, where microtopography provides a growing environment as much as 20 cm above the water table (National Wetlands Working Group 1997).

Fens can be further differentiated into subforms according to surface pattern, surface relief, proximity to water bodies, and basin topography; some additional fen subforms that are prevalent in the Northwest Territories include basin fens, channel fens, and horizontal fens (National Wetlands Working Group 1997). All of these fen types have flat or concave surfaces and are not adjacent to open bodies of water. Basin fens are topographically isolated and are not considered part of the regional drainage system. Channel fens occupy former stream channels, and small streams may persist and flow slowly through these types of fen. Horizontal fens occupy broader, less defined depressions and can occur across gentle slopes.

Bogs are characterized by a variety of shapes and sizes, but are usually raised or level with the surrounding landscape, with water levels at or slightly below the bog surface (National Wetlands Working Group 1997). Another defining feature of bogs is the prevalence of *Sphagnum* mosses and ericaceous shrubs, but bogs that are especially dry can support dwarf shrub, tree, and lichen plant communities (National Wetlands Working Group

1997). Bogs are typically formed when vertical peat growth raises the surface of the peat above the water table and allows *Sphagnum* mosses to become established (Robinson and Moore 2000). These mosses have high vertical growth rates, further raising the surface of the bog, and they contain acids that create adverse conditions for the survival of most other plants; these species therefore come to dominate bog features (National Wetlands Working Group 1997; Robinson and Moore 2000).

Like fens, bogs can also be further differentiated into subforms on the basis of surface form, relief, and proximity to water bodies (National Wetlands Working Group 1997). In the Northwest Territories, a particularly important and prevalent bog subform is the peat plateau, a small mound of frozen peat with permafrost core that rises 1–3 m above the surrounding frozen landscape (Figure 16, Figure 20). Peat plateaus form because dry peat is an excellent insulator, especially when thawed, whereas frozen peat is an excellent heat conductor (Smith and Riseborough 2002). These characteristics mean that in winter, cold temperatures can penetrate deeply into frozen peat and soil, whereas the thawed and drier raised peat areas insulate the underlying frozen peat from warmer summer temperatures (Robinson and Moore 2000). A negative heat balance throughout the year will maintain existing permafrost or lead to the creation of new permafrost, or both. When permafrost forms, any moisture in the ground freezes, which causes an expansion of about 9% as the ice forms. This expansion pushes the ground surface above the surrounding landscape and creates a peat plateau (Robinson and Moore 2000). In a study carried out in the Mackenzie Valley, the median depth to permafrost across several plots on peat plateaus in the Taiga Plains ecoregion was 34.3 cm in the Mid-Boreal, 38.7 cm in the High Boreal, 48.5 cm in the Low Subarctic, and 39.6 cm in the High Subarctic (Figure 17) (Errington et al. 2010).

Because peat plateaus are raised above the surrounding landscape, they are drier than other types of bogs or fens, which allows drought-tolerant lichens to replace Sphagnum mosses as the dominant ground cover (Robinson and Moore 2000), and these features will eventually support trees and other woody biomass (Figure 16, Figure 20). Errington et al. (2010) noted that mean tree biomass of peat plateaus in the Taiga Plains ecoregion was 15.33 t·ha^{-1} in the Mid-Boreal, 13.36 t·ha^{-1} in the High Boreal, 0.77 t·ha^{-1} in the Low Subarctic, and 0.28 t·ha^{-1} in the High Subarctic.

Palsas, another bog subform, similarly have a permafrost core but are more like mounds, with a convex surface up to 5 m higher than the surrounding landscape (National Wetlands Working Group 1997). Peat plateaus can cover several square kilometers, whereas palsas are smaller, ranging from a few square meters to 0.5 km^2.

Fens, bogs, and peat plateaus are all common in the Northwest Territories. For example, the 22 km^2 area of the Scotty Creek Research Station in the lower Liard Valley (Taiga Plains Mid-Boreal ecoregion, Dehcho administrative region), 50 km south of Fort Simpson, consists of 43% peat plateau, 4% flat bog isolated within peat plateaus, 23% flat bog connected to fen networks, 21% channel fen, with the remaining 9% classified as lakes (Quinton et al. 2009).

3.5 Productivity across Ecoregions

Of the three forested level II ecoregions in the Northwest Territories, the most productive is the Taiga Plains (Matasci et al. 2018). Matasci et al. (2018) found that the mean 95th percentile canopy height of undisturbed forests was 13 m in the Taiga Plains, 8 m in the western Taiga Shield, and 10 m in the Taiga Cordillera (Figure 21). Percent canopy cover above 2 m was 60% in the Taiga Plains, 35% in the Taiga Shield, and 40% in the Taiga Cordillera (Figure 21) (Matasci et al. 2018). As a result of the greater productivity of the Taiga Plains, it is the only level II ecoregion in the Northwest Territories for which data on forest harvesting have been collected. In general, the stands that are harvested in the Northwest Territories are among the most productive in the territory; for example, harvested stands in the Taiga Plains had a mean 95th percentile canopy height of 17.5 m before harvest and percent canopy cover above 2 m of 75% (Figure 22). Thirty years after harvest, the mean 95th percentile canopy height was 15.5 m, whereas canopy cover reached 75% after 22 years, when the 95th percentile canopy height was 8 m (Figure 22). Aboveground biomass was an average of 125 t·ha^{-1} before harvest and reached 120 t·ha^{-1} after 30 years (Figure 22). Canopy cover tends to return to predisturbance levels faster than height and biomass because of the greater foliage growth of residual trees (Matasci et al. 2018).

As part of the most recent International Polar Year (2007–2008), Errington et al. (2010) established

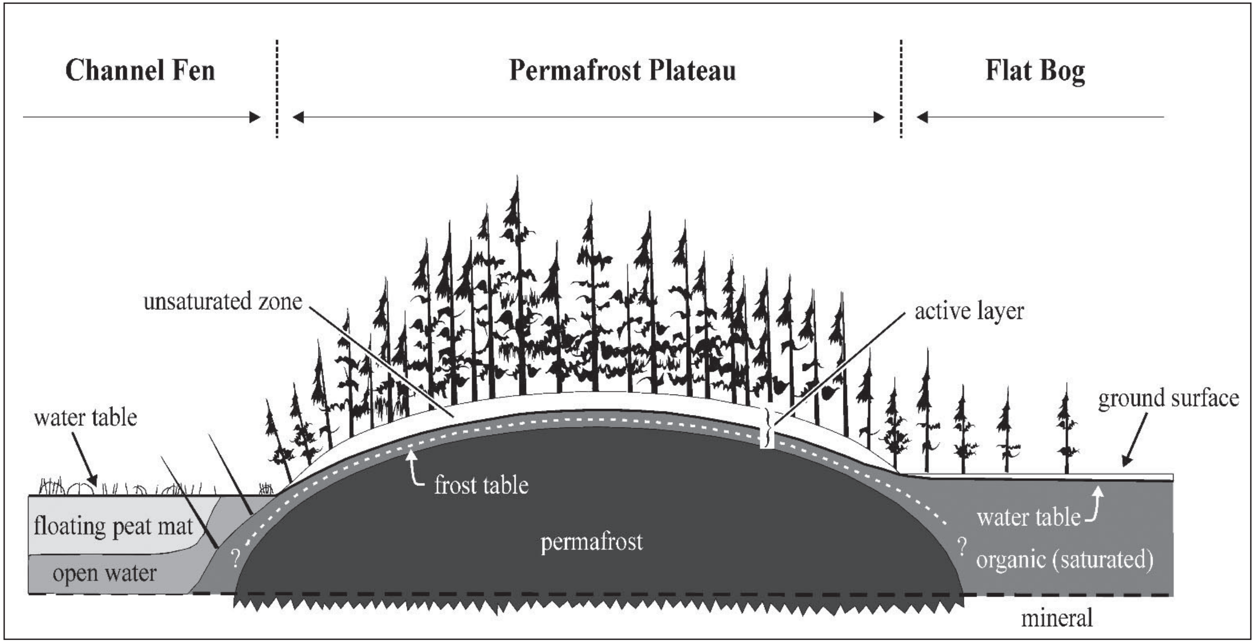


Figure 20. **Cross-section of a permafrost plateau, flanked by a channel fen on one side and a flat bog on the other.** (Reproduced, with permission, from Quinton et al. 2009.)

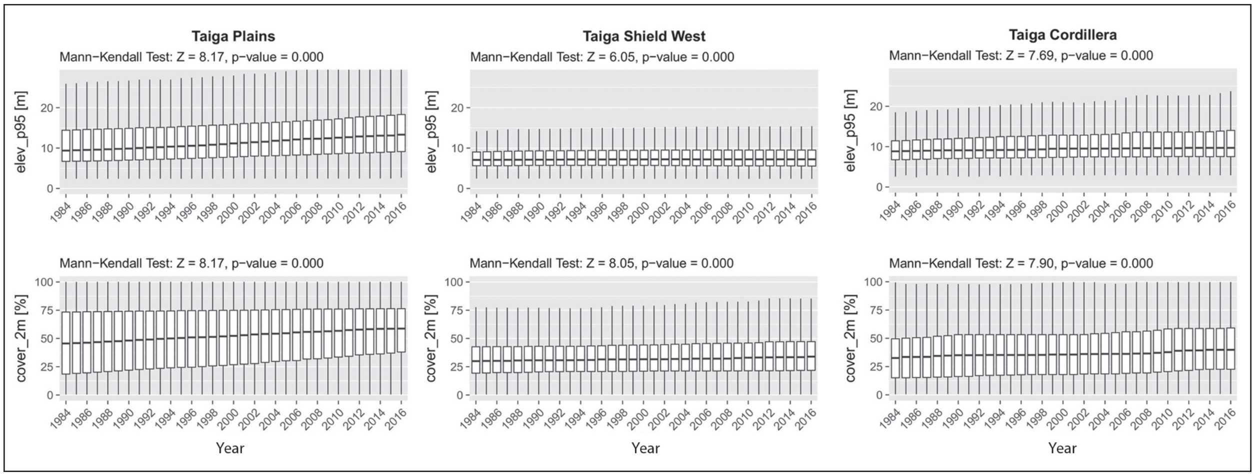


Figure 21. **Ninety-fifth percentile of stand height (elev_p95) and canopy cover of trees more than 2 m tall (cover_2m) in undisturbed forests in the Taiga Plains, western Taiga Shield, and Taiga Cordillera ecoregions of Canada, 1984–2016.** (Adapted, with permission, from Matasci et al. 2018.)

a series of plots along the Mackenzie Valley on a south–north gradient from the Taiga Plains Mid-Boreal ecoregion to the Taiga Plains High Subarctic ecoregion to investigate differences in tree biomass on upland forests, peat plateaus, and collapse scars (Figure 23). This work showed significant differences in tree biomass among these different landforms, as well as among level III ecoregions. The highest aboveground tree biomass, 45–100 $t{\cdot}ha^{-1}$, was found in Mid-Boreal and High Boreal uplands (Figure 17) and was associated with a deeper active-layer depth than that of the other site types that were assessed. Low Subarctic and High Subarctic uplands and Mid-Boreal and High Boreal peat plateaus had shallower active-layer depths and intermediate levels of tree biomass (10–30 $t{\cdot}ha^{-1}$) (Errington et al. 2010). Low Subarctic and High Subarctic peat plateaus and all collapse scars had tree biomass levels close to zero (Errington et al. 2010) (Figure 17). Analysis of data collected during a 10-year resampling of plots is currently underway.

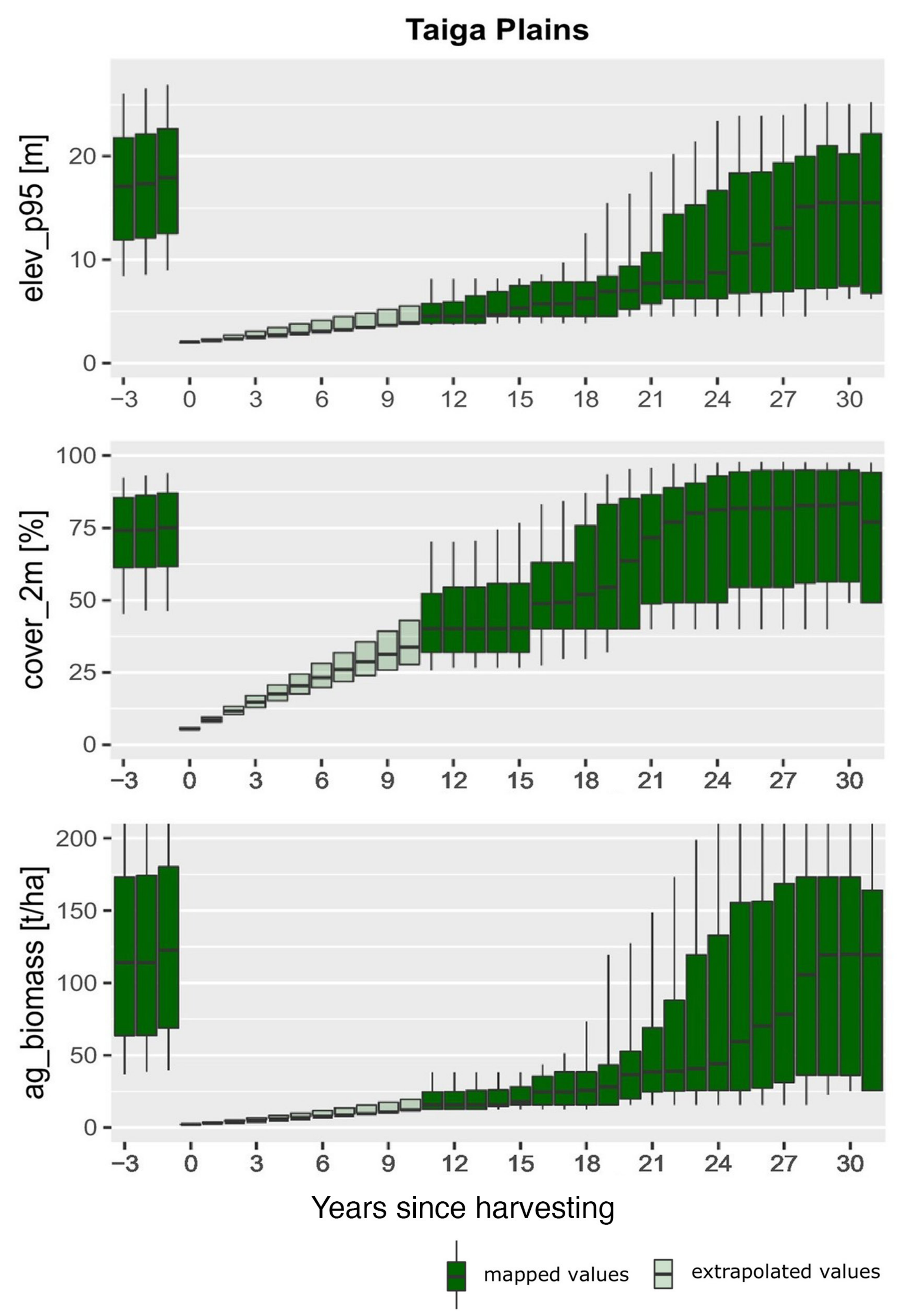


Figure 22. Ninety-fifth percentile of stand height (elev_p95), canopy cover of trees more than 2 m tall (cover_2m), and aboveground biomass (ag_biomass) before and after stand harvest in the Taiga Plains ecoregion, 1984–2016. (Adapted, with permission, from Matasci et al. 2018.)

The effects of climate change on tree productivity in the Taiga Plains are expected to be complex. Upland forest biomass can be expected to increase because of increasing temperatures. However, changes in forest biomass on peat plateaus are expected to be variable and will likely be highly dependent on whether warming temperatures cause permafrost thaw: in locations where such thaw does not occur, biomass could increase, but where warming temperatures thaw the permafrost and transform peat plateaus into collapse scars, biomass will likely decline (Errington et al. 2010). As a result, net biomass change in Northwest Territories forests is expected to depend largely on the extent of permafrost decline and whether these effects outweigh any potential increased growth in upland or remaining peat plateau areas. Indeed, as the prevalence of collapse scars increases across the landscape, these features may become connected with other collapse scars or with channel fens, potentially allowing the collapse scars to drain and further change the hydrology of the area. Landscapes of peat plateaus and collapse scars may then become a series of bogs and fens, completely changing forest growth dynamics. Both vegetation and soils buffer climatic changes to some degree, and time lags must therefore be taken into consideration when studying and projecting impacts on this type of landscape (Errington et al. 2010).

3.6 Productivity Trends

Girardin et al. (2016a) studied trends in the productivity of Canada's boreal forest for the years 1950–2002 but did not detect any significant changes over that period. Nevertheless, among the small changes that were detected, the

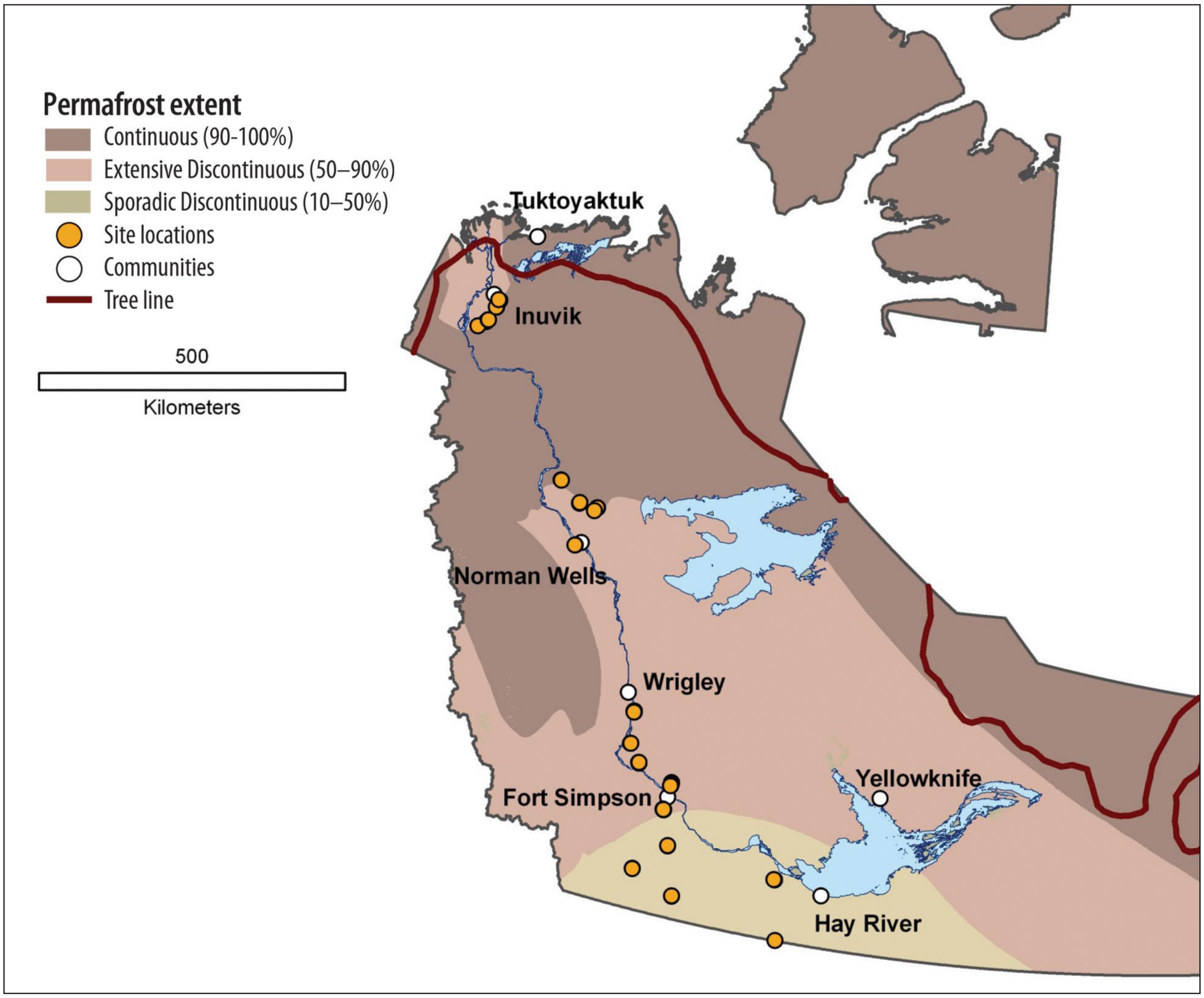


Figure 23. Permafrost map of the Northwest Territories showing International Polar Year tree biomass study sites. (Reproduced, with permission, from Errington et al. (2010), based on original permafrost map data provided by the Atlas of Canada, http://atlas.gc.ca © 2007.)

greatest productivity declines were observed in northwestern forests located east of the Rocky Mountains, including the Taiga Plains. In this zone, annual declines in growth were 0.1% for black spruce, 0.12% for white spruce, 0.21% for trembling aspen, and 0.26% for larch (Girardin et al. 2016a). Both temperature and soil moisture were identified as important drivers of tree growth in that study, with positive correlations observed between summer soil moisture and tree growth and negative correlations observed between summer temperatures and tree growth for black spruce, white spruce, lodgepole pine, jack pine, and trembling aspen, and for the Taiga Plains overall (Girardin et al. 2016a). These results suggest that higher summer temperatures may lead to some level of moisture stress, and this relationship seems to be a key driver of forest productivity across the boreal forest, although particular relationships varied slightly by region and species (Girardin et al. 2016a).

Indeed, in a study examining the relationship between climate and black spruce productivity, Girardin et al. (2016b) noted a positive correlation between tree growth and soil moisture for trees growing on well-drained mineral soil sites, but not for trees growing on poorly drained peatland areas. On the peatland sites, reduced soil moisture was thought to be beneficial for black spruce because the sites transitioned from poor drainage to a drainage class more conducive to tree growth.

The relationships among temperature, soil moisture, and tree growth can become even more complicated when permafrost is present on the landscape. In a study of 445 trees across seven peat plateaus in the Scotty Creek area of the Northwest Territories, black spruce productivity declined steadily over several decades, and the relationship between temperature and tree growth actually shifted from a positive correlation before 1970 to a negative correlation for the period 1971–2011, despite an increase in precipitation over the same period at the study sites (Sniderhan and Baltzer 2016). When these results were further examined to include evaluations of microsite, the authors observed that greater productivity declines occurred in trees located closer to peat plateau edges than in those located in peat plateau interiors, although trees growing in both types of microsite exhibited some level of productivity decline during this period (Sniderhan and Baltzer 2016). The greater productivity losses observed near peat plateau edges were attributed to vertical permafrost thaw leading to increased active-layer depth, which lowered the water table, dried out the surface soil, and likely resulted in some level of moisture stress for the trees (Sniderhan and Baltzer 2016).

On the basis of these results and others, it is apparent that the relationships among summer temperatures, soil moisture, and tree productivity are complex, and it should not be assumed that warmer summers will necessarily result in greater tree growth rates, even in northern climates. For instance, the rates of both autotrophic respiration and evapotranspiration increase with warmer temperatures, which means that trees will require greater carbohydrate reserves and greater access to soil moisture to fuel these processes (Helbig et al. 2017a). If these resources are not available in the quantities needed to accommodate the trees' increased respiration and evapotranspiration rates under warmer temperature conditions, tree growth will decline. Indeed, Helbig et al. (2017a) found that at the landscape scale in the Scotty Creek watershed, positive carbon sequestration (tree growth) began at a mean daily temperature of 2°C and reached a maximum at a mean daily temperature of 15°C. As the number of days above 2°C increases (currently 50% of days in nearby Fort Simpson), there may be an opportunity for increased tree productivity. However, this opportunity may be countered by decreased productivity when temperatures are above 15°C (currently 15% of days at Fort Simpson).

3.7 Tree Line

The forest–tundra ecotone in the Northwest Territories runs along a northwest–southeast axis and is on average 112 km wide (Figure 24). In the past, it has roughly followed the mean July air temperature isotherm of 10°C to 12.8°C, although recent warming may have altered that pattern (Timoney et al. 1992). In the Taiga Plains, where the ecotone lies farther north, the dominant species at the tree line is white spruce, while in the Taiga Shield, where the ecotone lies farther south, it is black spruce. The latter has an optimal germination temperature of 20°C, whereas for white spruce, the optimal germination temperature is 12.8°C to 15.6°C. Thus, white spruce is better adapted to more northerly latitudes and therefore dominates the tree line in the Taiga Plains. However, white spruce does not grow as well in the nutrient-deficient Taiga Shield; in that ecoregion, black spruce dominates the tree line, which consequently

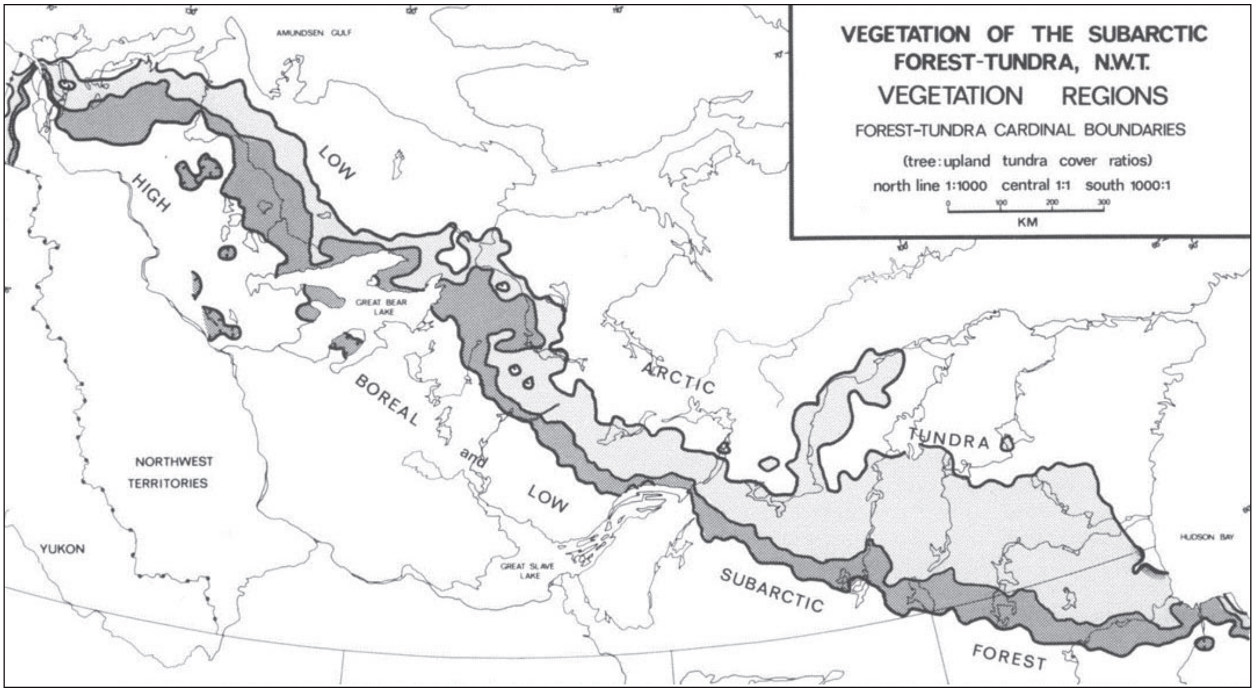


Figure 24. Northern, central, and southern boundaries between forest and tundra, according to tree–tundra cover ratios. The northern boundary has a cover ratio of 1:1000, the central boundary has a cover ratio of 1:1, and the southern boundary has a cover ratio of 1000:1. (Reproduced, with permission, from Timoney et al. 1992.)

lies farther south than in the Taiga Plains (Timoney et al. 1992).

The location of the tree line is particularly influenced by the sexual reproductive capacity of individual trees, which is most strongly affected by summer temperatures (Walker et al. 2012). Given that the location of the tree line is strongly influenced by temperature, warming temperatures due to climate change are predicted to cause a northward shift of the tree line. However, between 1990 and 2009, Walker et al. (2012) found no increase in seed production of white spruce within the forest–tundra ecotone at the Mackenzie Delta, likely because most warming has occurred in the winter, with little change in summer temperatures to date. Furthermore, these authors found no seedling recruitment in the study area, although viable seed was present and transplanted see dlings survived. They attributed the lack of white spruce seedling establishment to a lack of suitable substrate for seed germination; the lichen that is a common ground cover in the forest–tundra ecotone is a poor substrate for white spruce germination; bare mineral soil is best for this tree species. Thus, the exposure of mineral soil through natural disturbances such as fire may increase white spruce germination farther north, especially once summer temperatures begin to warm more significantly (Walker et al. 2012).

In a study carried out in 1990, Landhäusser and Wein (1993) examined several sites near Inuvik that experienced burning in 1968, followed by a prolonged but slight warming and drying trend that may have been analogous to contemporary climate change. They found that deciduous species such as paper birch and trembling aspen were able to take advantage of the increased temperatures and exposed mineral soil to become established in areas of former tundra. In contrast, white and black spruce were not found in any of the former tundra areas examined, and the density of these species actually declined in previously treed areas that had burned during the 1968 fire. The reason for this discrepancy appears to be that deciduous species have greater seed production and longer dispersal distances, and can easily reproduce vegetatively, whereas spruce species produce fewer seed and have shorter dispersal capabilities. Thus, the northerly spread of white spruce may continue to be restricted after fire even as climate change progresses (Landhäusser and Wein 1993). Black spruce in the Taiga Shield ecoregion may be slightly more successful in spreading north than white spruce in the Taiga Plains ecoregion because black spruce cones are semiserotinous, which allows seeds of this species to survive fire. As such, after a fire, black spruce seeds will have a shorter distance to travel to take advantage of exposed mineral soil than is the case for white spruce, which must rely on the presence of unburned trees (Brandt 2009).

Key Points

- According to the EOSD land cover map, there is 44 Mha of forested area (>10% crown closure) in the Northwest Territories, across 80 Mha of land below the tree line.
- Coniferous forest makes up about 72% of the forested area, with the remainder a combination of broadleaf forest, mixed-wood forest, and treed wetland. Open forest (20%–60% crown closure) constitutes the largest stand structure type, accounting for about 44% of the forested area.
- Data from the NFI show that 86% of coniferous stands in the Taiga Plains ecoregion are in the 81–100 year age class, and 60% of broadleaf stands are in the 61–80 year age class.
- Data from the MVI show that among all component regions in the Taiga Plains ecoregion, forest stands in the Mid-Boreal have the greatest height (9.9 m), crown closure (41%), stand volume (50 $m^3 \cdot ha^{-1}$), and aboveground biomass (52 $t \cdot ha^{-1}$).
- In the Taiga Plains ecoregion, uplands have the greatest tree biomass, with a mean of 76.17 $t \cdot ha^{-1}$ in the Mid-Boreal, 67.62 $t \cdot ha^{-1}$ in the High Boreal, 20.21 $t \cdot ha^{-1}$ in the Low Subarctic, and 12.96 $t \cdot ha^{-1}$ in the High Subarctic. Also in the Taiga Plains ecoregion, peat plateaus have a mean tree biomass of 15.33 $t \cdot ha^{-1}$ in the Mid-Boreal, 13.36 $t \cdot ha^{-1}$ in the High Boreal, and insignificant biomass in the Low and High Subarctic. Collapse scars cannot support tree growth because the rooting zones are saturated.
- Between 1950 and 2002, tree productivity in the Taiga Plains declined by up to 0.8% annually. Among individual species, average annual decline was 0.1% for black spruce, 0.12% for white spruce, 0.21% for trembling aspen, and 0.26% for larch.

Knowledge Gaps

- Data on long-term growth rates and patterns of trees in the Northwest Territories are limited.
- The effects of warming on tree productivity are uncertain because of complex interactions among increased rates of photosynthesis, respiration, and evapotranspiration, as well as permafrost thaw.
- It is unknown whether increasing levels of CO_2 will have a positive effect on tree growth in the Northwest Territories.

CHAPTER 4. PERMAFROST

4.1 Introduction

Permafrost is ground that remains frozen year-round, for 2 years or more, in northern or mountainous regions. The extent and thickness of permafrost increases northward as mean annual air temperature decreases. Located above permafrost is the active layer, which undergoes freezing in winter and thawing in summer. The depth of the active layer generally increases southward, but it varies with soil condition, slope, aspect, and vegetation (Wolfe and Kokelj 2019). Permafrost extent is categorized as continuous (90%–100% of ground underlain by permafrost), extensive discontinuous (50%–90% underlain by permafrost), or sporadic discontinuous (10%–50% underlain by permafrost) (Natural Resources Canada 2009). All three of these permafrost categories occur in the Northwest Territories. Sporadic discontinuous permafrost is found in the southern Taiga Plains ecoregion from the British Columbia–Alberta border to south of Fort Simpson, whereas extensive discontinuous permafrost underlies the area between Great Slave and Great Bear Lakes and along the central Mackenzie Valley (Figure 23) (Natural Resources Canada 2009). Because more bedrock is present in the Great Slave Upland region north of Great Slave Lake, permafrost may be sporadic discontinous there as well (Morse et al. 2016). Continuous permafrost is found north and east of Great Bear Lake and throughout much of the Taiga Cordillera ecoregion (Natural Resources Canada 2009). The latest permafrost map of Canada shows a pocket of extensive discontinuous permafrost in the Mackenzie Delta, although other data suggests that the delta is underlain by continuous permafrost (Nguyen et al. 2009).

In the Northwest Territories, permafrost thickness can range from 400 m in the continuous zone to less than 50 m in the extensive discontinuous zone and only a few meters in the sporadic discontinuous zone (Wolfe and Kokelj 2019). Depth to permafrost, or the active-layer depth, depends on local soil, vegetation, and topographic variables. Permafrost can occur in both mineral and organic soils; however, a thick organic soil layer generally results in a shallower active layer, because of summer insulation. Conversely, seasonal flooding or greater snow cover can increase the active-layer depth. For example, Smith et al. (2009) found that, because of differences in the thickness of organic soil, active-layer depths averaged 76 cm in the forested portion of the Mackenzie Delta, but only 45 cm farther south in the Mackenzie Plain near Norman Wells. In contrast, an active-layer thickness of 84 cm near Fort Simpson was attributed to a warmer climate (Smith et al. 2009). Errington et al. (2010) found that the median active-layer depths of peat plateaus in the Taiga Plains ecoregion were 34 cm in the Mid-Boreal ecoregion and 39 cm in the High Boreal ecoregion, while median active-layer depths of mineral soil uplands in the same ecoregions were greater than the 60-cm soil pit. In the Taiga Plains Low Subarctic ecoregion near Norman Wells, the median active-layer depths were 49 cm for peat plateaus and 58 cm for uplands. In the Taiga Plains High Subarctic ecoregion of the Mackenzie Delta, the median active-layer depths were 40 cm for peat plateaus and 38 cm for uplands (Figure 18) (Errington et al. 2010). In the Great Slave Lowland and Great Slave Upland ecoregions north of Great Slave Lake, permafrost was found beneath all landforms except bedrock. However, the active-layer depth was greatest beneath paper birch stands (105 cm), least beneath peatlands (63 cm), and intermediate beneath black spruce uplands (86 cm). Paper birch stands typically have the least organic soil accumulation and the best drainage, whereas black spruce stands are intermediate in terms of both these characteristics (Morse et al. 2016).

Continuous permafrost is mostly driven by climate and is found north of the −6°C to −8°C isotherm of mean annual air temperature (Smith and Riseborough 2002). Where mean annual air temperature is above this isotherm, the presence of permafrost depends on environmental conditions such as snowpack, soil type, water table, slope, aspect, vegetation, and proximity to water bodies, and the resulting permafrost is discontinuous. One reason why permafrost is not necessarily present where mean annual air temperatures are below 0°C is the insulating effect of snow cover. In the Fort Simpson area, for example, the mean annual air temperature was −4°C, while the annual ground-surface temperature was 4.7°C to 5.3°C warmer because of snow insulation during the winter (Smith and Riseborough 2002). The thicker the snowpack, the greater the chance that the ground will be insulated from cold winter temperatures.

However, between −6°C and −8°C, permafrost will form regardless of snowpack depth (Smith and Riseborough 2002). Thus, at these temperatures, even bedrock and very dry mineral soil can maintain permafrost in all conditions across the landscape, and permafrost is therefore continuous (Smith and Riseborough 2002).

In the discontinuous permafrost zones, climate alone generally cannot create or sustain permafrost. In addition to snow cover, another factor controlling the formation of permafrost is soil type. The thermal conduction of soils is lower during the thaw season than during winter because the thermal conductivity of ice is four times that of water. As such, frozen soils conduct heat away from lower depths of the soil profile in winter to a greater extent than thawed soils conduct heat to those depths in summer (Smith and Riseborough 2002). This difference in conductivity between the seasons, referred to as the thermal offset, depends on the amount of moisture in the soil: organic soils, which tend to be poorly drained, have a larger thermal offset than mineral soils (Jorgenson et al. 2010). Mineral soils can be 0.5°C to 2°C colder at lower positions in the soil profile than at the surface, depending on soil texture, whereas organic soils can be 0.5°C to 3.5°C colder at lower positions in the soil profile than at the surface. Thus, most mineral soils will not contain permafrost where the mean annual air temperature is above −2°C, but permafrost can exist below wet organic soil where the mean annual air temperature is 1°C to 1.5°C (Smith and Riseborough 2002). Permafrost under mineral soils that are particularly dry, or in areas of the discontinuous permafrost zones where the air temperature is above −2°C, will likely be confined to north-facing slopes (Jorgenson et al. 2010). In general, permafrost in the discontinuous zones, especially the sporadic discontinuous zone, is most common and is closest to the surface under peatlands (Smith et al. 2008; Quinton et al. 2009).

Permafrost thaw may result from a disturbance to the insulating surface of the peat plateau, from the thinning or disappearance of the tree canopy and subsequent increase in solar radiation reaching the surface, from wildfires, or from warming temperatures. Permafrost in the discontinuous zones is particularly sensitive to climate change, and even small changes in the surface energy balance can cause serious thawing (Quinton et al. 2009). In addition, because permafrost in this area is maintained by the insulating properties of the soil, changes to the soil through the creation of human infrastructure, wildfires, or loss of forest cover can cause permafrost degradation (Shur and Jorgenson 2007).

When ice-rich permafrost thaws, the loss of ice volume within the peat plateau can cause the surface to collapse 1–3 m into a wet depression, known as a thermokarst bog or collapse scar bog (Olefeldt et al. 2016). Because the collapse scar is isolated within and surrounded by a higher-elevation peat plateau, the water cannot drain, and the resulting saturated conditions drown tree roots, causing the spruce–lichen complex to be replaced by peat mosses (Figure 16) (Quinton et al. 2009; Errington et al. 2010). Collapse scar bogs are generally 0.5–10 ha in size, but can reach up to 100 ha, and are common in the Mackenzie Valley (Olefeldt et al. 2016). The wet collapse scar creates good conditions for peat accumulation, eventually resulting in peat growth above the water table, which once again forms a dry insulating layer and allows for the reestablishment of permafrost (Zoltai 1993). However, although permafrost creation and destruction are cyclic, the reestablishment of permafrost in a former collapse scar is believed to take at least 600 years, and even then, only under climatic conditions conducive to permafrost formation (Zoltai 1993). Climate warming may shift the balance toward degradation and a net loss of permafrost (Robinson and Moore 2000).

When permafrost thaws in hilly terrain where ice content in the soil is high, the hillslope can fail, with mass wasting of the soil and vegetation overlying the permafrost. These disturbances, known as hillslope thermokarsts, are generally smaller than collapse scars, but can reach up to 10 ha in area (Olefeldt et al. 2016). In addition to land subsidence, lateral soil transport resulting in a slide can occur on these slopes. These “thaw slumps” are also found in the Mackenzie Valley, although not as commonly as collapse scars (Olefeldt et al. 2016).

4.2 Monitoring Permafrost: Temperature Change

The Geological Survey of Canada has maintained a monitoring network of permafrost sites in the Mackenzie Valley and the Mackenzie Delta since the 1980s. In the summer and fall of 2016, ground temperatures and active-layer information were collected at sites from Wrigley to the Mackenzie Delta. Mean annual ground temperature was measured at a depth where seasonal variation in temperature is less than 0.1°C. Throughout the discontinuous permafrost zones, mean annual ground temperature was generally above −2°C, while in the continuous permafrost zone it was generally below −4°C (Figure 25) (Smith et al. 2017, Smith et al. 2018). Colder permafrost sites had warming of about 2°C since the 1970s (Wolfe and Kokelj 2019), including an increase of 0.6°C between 2007 and 2016 (Smith et al. 2010). Warmer permafrost sites, especially those close to 0°C and having ice-rich soils, had little temperature change. The reason for this difference between sites is that as the ice reaches thaw temperature, latent heat effects occur, and heat is used to carry out the phase change rather than increase the temperature (Smith et al. 2010). In contrast, permafrost thaw and the formation of thermokarst bogs are much more prevalent in the sporadic discontinuous zone, where thin permafrost layers are expected to eventually disappear (Wolfe and Kokelj 2019). The active-layer depth responds to short-term climatic variation and therefore displays much greater variation than deeper ground temperatures (Smith et al. 2010, Smith et al. 2017). Active-layer depths along the Mackenzie Valley generally increased between 2008 and 2016, exceeding the long-term mean in 2009, but are still less than the maximum depth documented in 1998, which was one of the warmest years on record (Smith et al. 2017).

4.3 Monitoring Permafrost: Thaw

This section focuses on the discontinuous permafrost zones that underlie the more extensively forested regions of the Northwest Territories, where permafrost is thawing at higher rates. The following descriptions of permafrost and permafrost thaw may therefore not apply to more northerly regions of the Northwest Territories.

Measurements in the Scotty Creek basin, which is in the sporadic discontinuous permafrost zone showed that the mean annual active-layer temperature increased consistently between 2001 and 2009, causing a deepening of the active layer (Quinton et al. 2009). How this translates into permafrost thaw can be estimated by using changes in canopy cover as a proxy, given that densely treed areas are found on peat plateaus but not collapse scars. Satellite and LiDAR images of a 1.6 km² study area within the Scotty Creek watershed detected changes in the forest canopy between 2000 and 2008 that suggested a 12% loss of permafrost during that period (Quinton et al. 2009) (Table 15). The study area was small, however, and other studies have yielded more conservative estimates. For example, Carpino et al. (2018), using historical aerial photographs and satellite images of vegetation cover to compare the relative proportions of plateau and wetland, found that permafrost thaw rates ranged from 9.5% to 11.6% over the period 1970–2010 at 8 sites in the discontinuous permafrost zones of the Scotty Creek

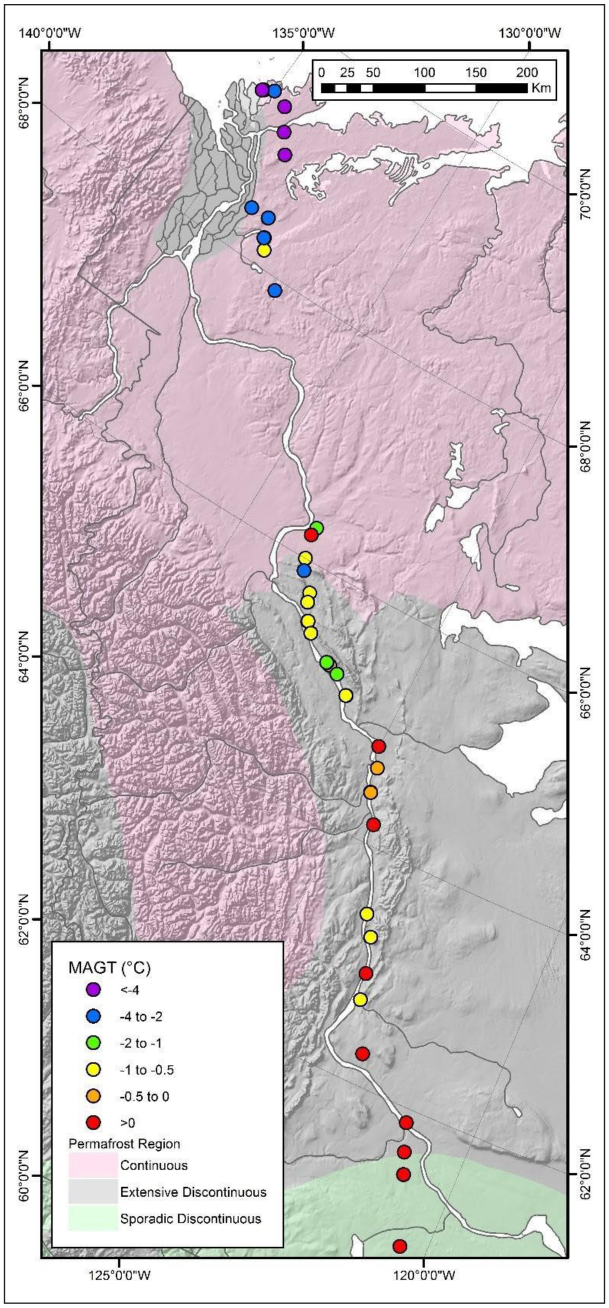


Figure 25. Mean annual ground temperature (MAGT) from surveys conducted in 2016 and 2017 in the Mackenzie Valley corridor. (Reproduced, with permission, from Smith et al. 2018.)

Table 15. Permafrost thaw rates from three studies within the discontinuous permafrost zone of the Taiga Plains, Northwest Territories. (Data from sources as listed.)

		Thaw rate (%)				
Study	Period	For total period	Annual	Detection methods	Permafrost proxy	Area
Quinton et al. (2009)	2000–2008	12	1.5	Satellite images, LiDAR[a]	Land cover	1.6 km^2 within Scotty Creek watershed
Carpino et al. (2018)	1970–2010	9.5–11.6	0.24–0.29	Satellite images, aerial photographs	Land cover	8 sites from Scotty Creek watershed to British Columbia border
Baltzer et al. (2014)	1977–2010	6.3–12.5	0.19–0.38	Satellite images, aerial photographs, LiDAR	Land cover and terrain	7 × 0.25 km^2 peat plateaus within Scotty Creek watershed
	1977–2000	1.6–7.6	0.07–0.33	Satellite images, aerial photographs, LiDAR	Land cover and terrain	7 × 0.25 km^2 peat plateaus within Scotty Creek watershed
	2000–2010	2.6–6.3	0.26–0.63	Satellite images, LiDAR	Land cover and terrain	7 × 0.25 km^2 peat plateaus within Scotty Creek watershed

[a] *LiDAR = light detection and ranging.*

watershed (Table 15). Baltzer et al. (2014), using the same methods, found that the mean annual rate of permafrost loss (and corresponding loss of forest) between 1977 and 2010 was 0.26% in a 152 km^2 study area in the Scotty Creek watershed, for a total loss of 8.6% during the study period. These authors also calculated loss rates for shorter periods within the overall study period and found that the rate between 2000 and 2010 was three times that between 1977 and 2000, such that the loss of permafrost for the later subperiod was 4.7% (Table 15) (Baltzer et al. 2014).

Thaw rates at the margins of peat plateaus were greater than in the interior of the plateaus. Wetlands adjacent to the peat plateaus are treeless, and transition edges have lower canopy cover. Therefore, plateau edges receive more sunlight. In addition, the wetlands are warmer and wetter than the plateaus, and because heat conduction in peat increases with moisture content, wetlands warm up the edges of adjacent plateaus (Baltzer et al. 2014). As peat plateaus thaw and shrink, the ratio of edge to interior area increases, and thaw subsequently accelerates (Baltzer et al. 2014). Thus, permafrost thaw in peat plateaus creates positive feedback and increases thawing.

In addition to the active layer that undergoes a freeze-and-thaw cycle each year, there may be a layer of perennially thawed soil between the permafrost and the active layer, known as a talik. Together, the active layer and the talik layer are known as the suprapermafrost layer (Connon et al. 2018). Taliks form when summer ground thaw penetrates deeper than the active layer, and winter freezing does not reach sufficient soil depths to refreeze this soil layer. In the discontinuous permafrost zones, taliks are more likely to form under sites with sparse tree cover. The lack of shade increases solar radiation in the warm months and also increases the snowpack and hence insulation in the cold months, limiting the depth of winter refreezing. Once formed, taliks can accelerate thaw both upward and downward in the soil profile, because they are warmer than the active layer above during the winter and warmer than the permafrost below all year (Figure 26). Connon et al. (2018) tested this concept at a peat plateau in the Scotty Creek area. Measurements taken in April 2016 showed that as the suprapermafrost layer deepened beyond 80 cm, the active layer actually began to shrink (Figure 27) (Connon et al. 2018). Because the suprapermafrost layer was so deep, a

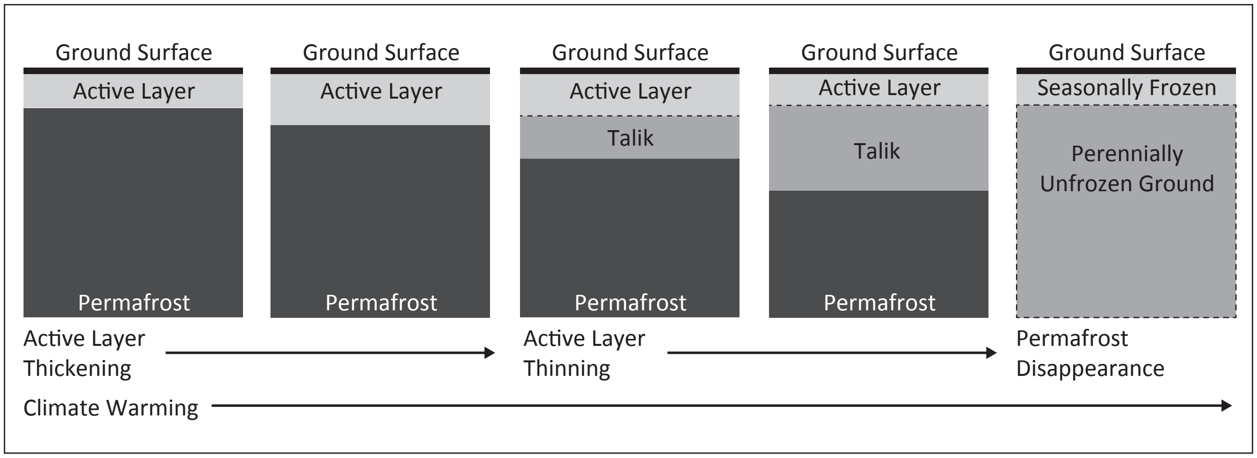


Figure 26. Changes in depth of the active layer in response to climate warming. Active-layer depth reaches a maximum before talik development. Depth of the suprapermafrost layer is the combined thickness of the active layer and the talik (Reproduced, with permission, from Connon et al. 2018.)

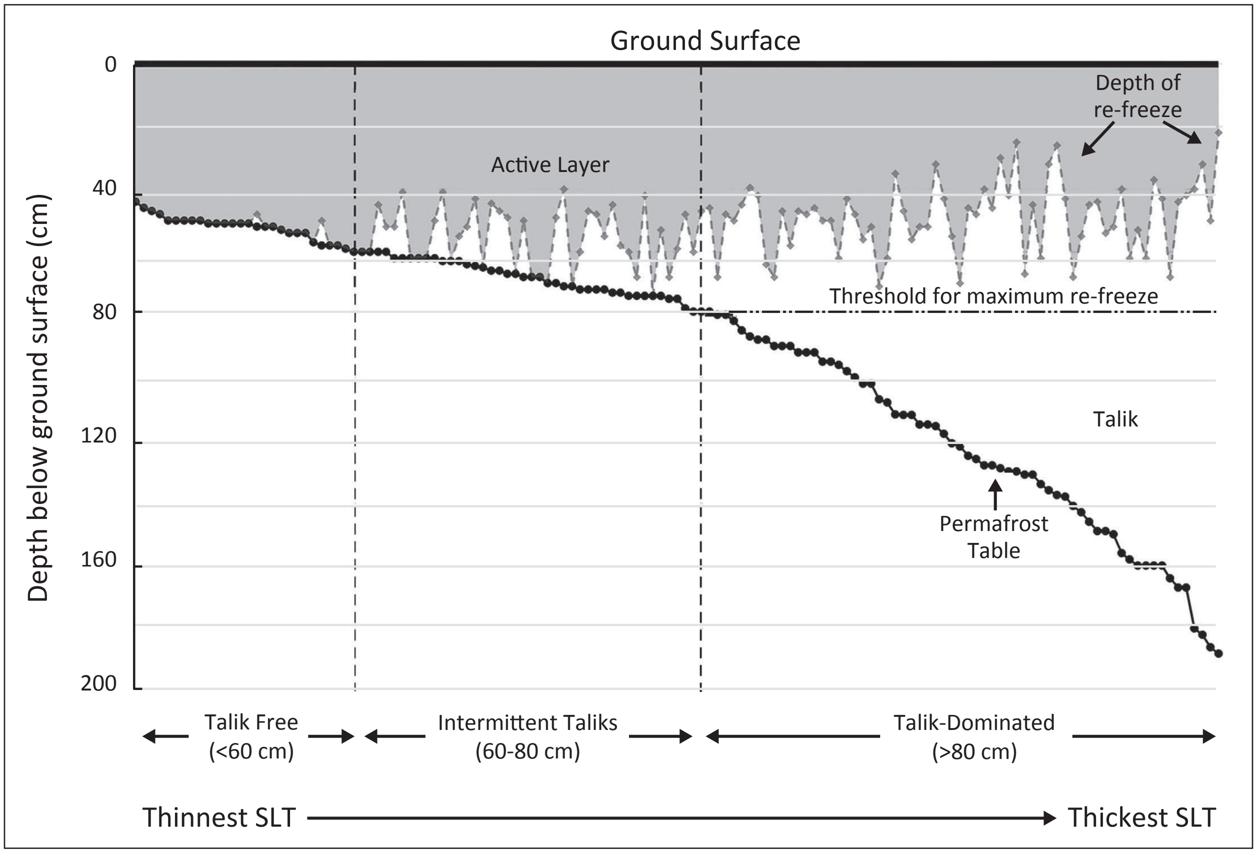


Figure 27. Measurements of suprapermafrost layer thickness (SLT), ranked from thinnest (left) to thickest (right), and associated measurements of depth of thaw. (Reproduced, with permission, from Connon et al. 2018.)

talik formed below the depth of the winter freeze, which then warmed the soil above it, keeping it from freezing during the winter and thinning the active layer. Taliks also reduced the thickness of permafrost. Between 2011 and 2016, permafrost thickness along study transects decreased by a median of 37 cm at points with a talik and by only 8 cm at points without a talik. The presence of taliks has been increasing in recent years. In 2011, only 20% of transect points had a suprapermafrost layer greater than 80 cm, at which point the existence of a talik can be assumed, whereas in 2015, the frequency had risen to 48%. Furthermore, a grid placed over the interior of the peat plateau had taliks under only 8% of its area in 2011 but under 33% of its area in 2013. Transect points had a greater proportion of taliks than the grid because they traversed the edges of the plateau where thaw is likely to be greater, while the grid was entirely within the plateau interior (Figure 28) (Connon et al. 2018).

Soil thermal properties and permafrost thaw of palsas were studied by Mamet et al. (2017) in the Boreal Cordillera Mid-Boreal ecoregion of the Northwest Territories near the Yukon border, an area with extensive discontinuous permafrost. Across five study sites, palsa area decreased by an average of 27% between 1940 and 2016. The higher-elevation sites (at 1473, 1477, and 1621 m) lost 31%–63% of palsa area between 1940 and 1981 and 15%–21% of palsa area between 1981 and 2016. The lower-elevation sites, in valley bottoms (1260 and 1272 m), lost 11%–20% of palsa area between 1940 and 1981 and 22%–64% of palsa area between 1981 and 2016 (Mamet et al. 2017).

The relationship between palsa area and climate is complex, combining the effects of seasonal, annual, and longer time scales. In the study by Mamet et al. (2017), higher-elevation sites had colder mean annual and thaw season temperatures, but lower-elevation sites had colder mean winter temperatures because of temperature inversions. These findings indicate that air temperature has a greater effect on palsas located at lower elevations because of cold air drainage, whereas ground temperature has a greater effect on higher-elevation palsas, although there was a significant delay in the response. The permafrost thawing observed at the lowest-elevation palsas was correlated with an increase in thaw season precipitation, because the saturated soil in the valley bottom increased heat conduction (Mamet et al. 2017).

The complexity of palsa shape also affected the rate of permafrost thawing, with more complex shapes exhibiting greater rates of thawing, likely because their greater perimeters provided more area for

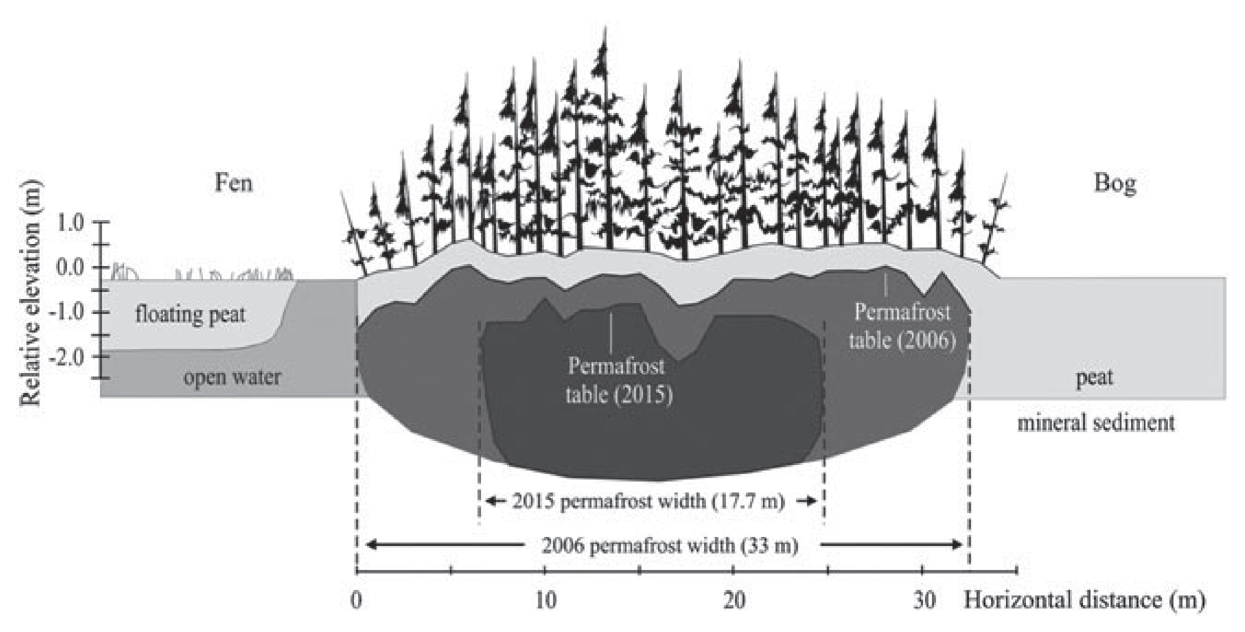


Figure 28. Cross-section of the Scotty Creek study peat plateau, showing lateral and vertical loss of permafrost between 2006 and 2015. (Reproduced, with permission, from Connon et al. 2018.)

lateral erosion. As palsas thaw, their shape tends to become simpler, which decreases the proportion of edge relative to interior area and slows the rate of thawing (Mamet et al. 2017). This relationship can be confounded in valley bottoms, where the typically higher water table is associated with greater peat accumulation and therefore greater insulation of permafrost.

Aside from warming temperatures, other aspects of climate change may increase permafrost thaw or lead to positive feedbacks (Figure 3, Figure 4). For example, the amount of nonsnow precipitation in the Northwest Territories has been increasing, and the seasonal patterns of this precipitation have been changing. These changes have increased the number of rain-on-snow events and have caused earlier snow melt, both of which increase permafrost thaw by removing insulating snow during warmer periods (Carpino et al. 2018). In addition, decreasing winter precipitation is causing a shallower snowpack. This can mean less insulation from winter lows, and therefore a colder permafrost with a lower thaw rate; however, it may also mean earlier snow melt and therefore less insulation from warmer spring temperatures (Carpino et al. 2018).

Jones et al. (2014) modeled the susceptibility of permafrost to thaw within the extensive discontinuous permafrost zone (Taiga Plains Mid-Boreal ecoregion). They found that depth of winter freeze and persistence of permafrost are greater when the snowpack is reduced. For example, under maximum snowpack conditions for the Fort Simpson area, permafrost would not have developed under peatlands if temperatures had been 2°C to 3°C higher than the historical records for 1964–2008. Without any snow cover, permafrost would not have developed had temperatures been 5°C to 6°C higher (Jones et al. 2014).

4.4 Impacts of Fire on Permafrost

Gibson et al. (2018) studied the effects of wildfires on soil thermal properties and permafrost thaw in peat plateaus of the sporadic and extensive discontinuous permafrost zones of the Taiga Plains in northern Alberta and southern Northwest Territories (Figure 29). Ten sites that had burned 2–49 years before the study were compared with six unburned sites. All had similar peat depths and current or prefire tree densities, and mean annual variation in air temperature among the unburned sites did not explain the variations among them in soil thermal properties. Therefore, differences in soil thermal properties were considered to be a result of differing fire histories. This work showed that sites with recent fires (< 5 years) had a greater than 60% increase in the active-layer depth; in addition, according to data collected at sites with a longer time since fire, it appeared to take about 20 years for the active-layer depth to return to prefire levels. Differences in depth to the permafrost layer between burned and unburned sites were already noticeable in June, with the difference between them increasing through summer and fall. Increased net surface radiation caused by the loss of shade from black spruce and decreased albedo caused by the conversion of light-colored lichens to burned char could be the cause. Other effects took longer to manifest. For instance, sites that burned 10–20 years previously had a greater incidence of taliks than unburned sites: burned sites had approximately 20% talik coverage, compared with 70%–100% coverage at the unburned sites. Examination of sites with a longer time since fire suggested that talik coverage should return to prefire levels after about 50 years. Maximum annual soil temperatures at 40 cm depth were monitored for a full year during this study, and the data indicated that soil temperatures increased by 2°C to 5°C at unburned sites and by 9°C to 11°C at sites that burned 10–20 years previously. According to measurements at sites with a longer time since fire, it appears that soil temperatures returned to prefire values after approximately 40 years, which coincided with vegetation recovery. This finding highlights the role that vegetation plays in shading and albedo (Gibson et al. 2018).

The same authors studied the effect of fire on the formation of collapse scars by comparing burned and unburned sites less than 10 km apart on four large peatlands that were partially burned 20–30 years previously (Gibson et al. 2018). Most of the effects of fire on the soil thermal regime lasted up to 30 years, so sites burned 30 years previous were assumed to capture most of the effects of fire on collapse scar formation. This work showed that burned sites formed collapse scar bogs at triple the rate of unburned sites and that wildfires had caused about 23% of the conversion of peat plateau to collapse bog within the study area in the previous 30 years. These findings highlight the important role of wildfire in permafrost thaw and formation of collapse scar bogs in the Northwest Territories (Gibson et al. 2018).

4.5 Impacts of Seismic Lines on Permafrost

Linear features such as seismic lines (narrow linear clearings used in oil and gas exploration activities) and roads may also contribute to permafrost

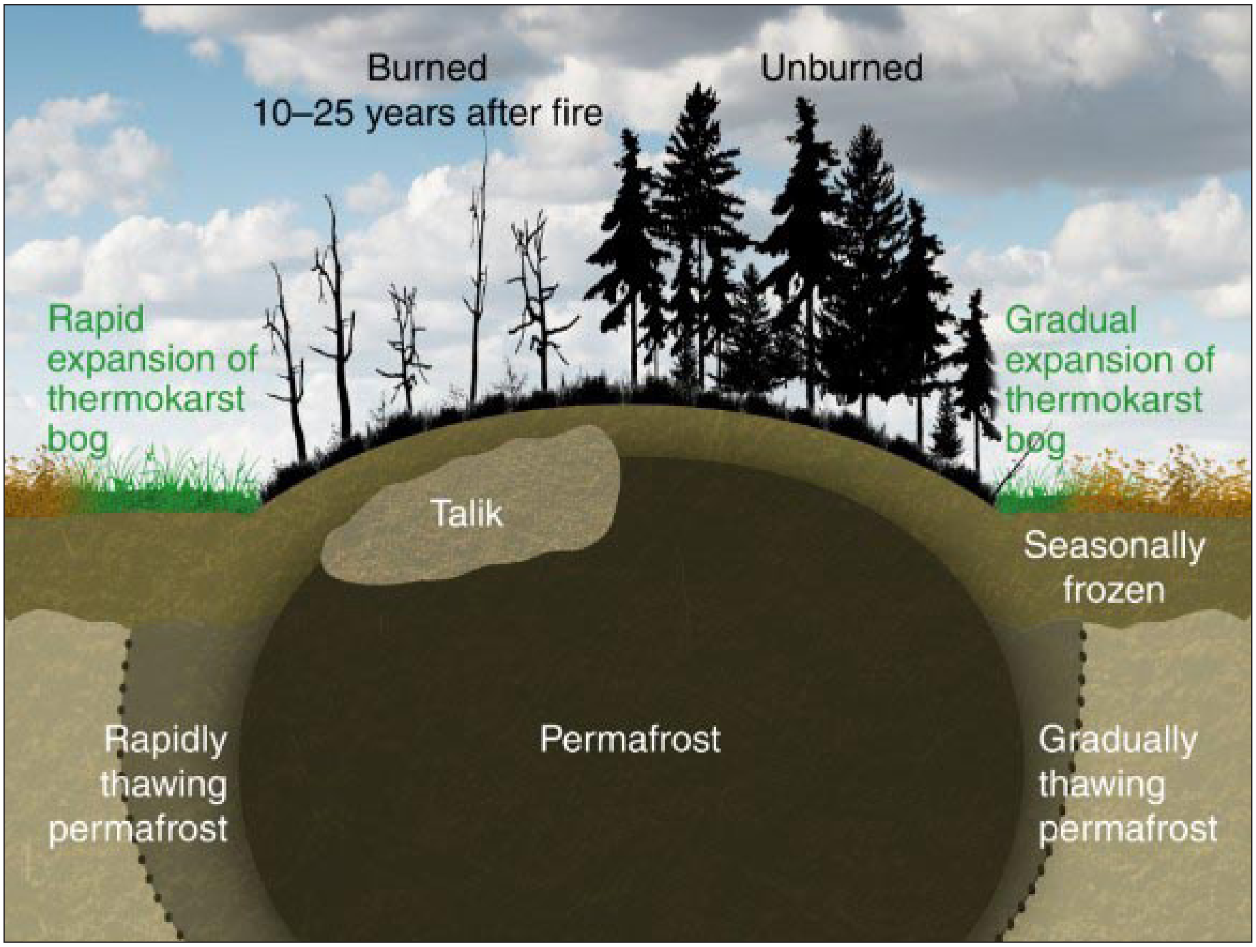


Figure 29. Simplified illustration of soil thermal states for peat profiles along transects from thermokarst bogs to peat plateaus, within and outside historical burn areas. (Reproduced, with permission, from Gibson et al. 2018.)

thaw through complex interactions of changes to vegetation, soil compaction, and changes to soil hydrology, all of which affect the thermal properties of soil (Dabros et al. 2018). In the Scotty Creek area south of Fort Simpson, the density of seismic lines and roads is 0.875 km·km^{-2}, which has been associated with significant permafrost thaw in the peat plateaus of the area (Williams et al. 2013). Permafrost thaw occurs along seismic lines because the removal of vegetation along the line increases summer solar radiation and winter snowpack (which insulates the soil from freezing temperatures), while soil compaction along the line increases soil moisture and heat conductivity. Once thaw occurs, linear depressions are created along the seismic lines, which act as drainage corridors connecting peat plateaus to lowland wetlands and allowing bogs that would otherwise store water to drain into fens and basin outlets (Braverman and Quinton 2016). Seismic lines have seven times the density of natural drainage in the Scotty Creek area and so may significantly change the region's hydrology by increasing the area contributing to runoff and shortening runoff transit times by increasing the paths such runoff can take. The full consequences of these effects are not yet understood (Braverman and Quinton 2016).

Experiments using mulch to cover seismic lines have been conducted with artificial soil columns that imitate the soil conditions found along seismic lines and the seasonal freeze-and-thaw cycles that occur, as well as with models that additionally incorporate the effects of solar radiation, snowpack, and precipitation (Mohammed et al. 2017). Mulching significantly reduced permafrost thaw by providing insulation and absorbing moisture that would otherwise increase heat conduction through the soil profile. Ground thaw was delayed by 57, 90, and 102 days for mulch layers of 10, 20, and 30 cm thickness, respectively, under wet conditions and by 63, 96, and 108 days, respectively, under dry conditions. Similarly, maximum thaw depth was reduced by 42%, 55%, and 71% for mulch layers of 10, 20, and 30 cm thickness, respectively, under wet conditions and by 12% and 37% for mulch layers of 20 and 30 cm, respectively, under dry conditions; a mulch layer 10 cm thick had no positive effects under dry conditions (Mohammed et al. 2017). Questions remain about the persistence of mulch over time, how its properties change through decomposition, and the effects of different topographic features on the ability of mulch to absorb moisture. In addition, mulch may hinder the regrowth of vegetation that would, presumably, be the long-term solution for rehabilitation of these features. Field studies are needed to further investigate whether mulching should be recommended as a best practice for industry (Mohammed et al. 2017).

4.6 Impacts of Permafrost Thaw on Forests

Subsidence of the ground caused by permafrost thaw can cause forest stands to lean or fall over, a phenomenon known as "drunken forests" (Department of Environment and Natural Resources [N.W.T.] 2018a). Permafrost thaw beneath peat plateaus causes saturated conditions in the rooting zone of trees, which decreases oxygen levels. The most common tree species on peat plateaus is black spruce, which (like other conifers) cannot tolerate low oxygen levels because of a limited ability to transport oxygen (Baltzer et al. 2014). Thus, permafrost thaw drowns tree roots and causes the spruce–lichen complex to be replaced with a collapse scar of shrub and moss (Quinton et al. 2009; Errington et al. 2010).

Peat plateaus drain to lower-lying wetlands, mostly isolated bogs where the water is trapped by the surrounding higher-elevation peat plateaus with impervious permafrost. With continuing thaw and subsidence of peat plateaus, these formerly isolated bogs will connect to each other and to the channel fens that are part of the regional drainage system, allowing the water in the bogs and collapse scars to drain. This phenomenon is likely increasing runoff rates in the short term, but may decrease runoff in the future as water sources from the plateaus and thawing permafrost disappear (Quinton et al. 2009).

Although the transition from peat plateau to collapse scar decreases tree biomass, the re-formation of a bog ecosystem in the collapse scar may increase carbon accumulation rates. However, were the water table to drop significantly, newly thawed peat might become available for decomposition—especially considering the warmer temperatures and increased soil aeration—whereas increasing wildfires would further decrease peat accumulation (Robinson and Moore 2000).

Warming temperatures that result in permafrost thaw and forest loss may also lead to afforestation and greater rates of tree growth in wetlands. Although this effect has not yet been seen in sites within the Northwest Territories, it has been observed at two sites along the territory's border with British Columbia (Carpino et al. 2018). However, despite forest loss often being the cause of permafrost thaw through increased solar radiation, the reverse effect, with permafrost returning to newly forested areas, has not occurred because of the currently warmer temperatures (Carpino et al. 2018). In addition to the direct effects of warmer temperatures on tree growth in cool wetlands, increased drainage from the expansion of permafrost thaw may eventually result in drier peatlands and an increase in forest cover (Carpino et al. 2018).

Key Points

- Permafrost thaw can cause lichen–spruce peat plateaus to transition to moss-dominated collapse scar bogs.
- Although peat plateaus and collapse scars have historically existed as part of a freeze-and-thaw cycle, peat plateaus in the discontinuous permafrost zones appear to now exist in a climate that is too warm for the re-formation of permafrost and are experiencing net thaw.
- Studies using land cover changes as a proxy for permafrost thaw have found average annual thaw rates of 0.07%–0.33% over the past few decades in the discontinuous permafrost zones of the Scotty Creek watershed (Taiga Plains Mid-Boreal ecoregion). In the decade of 2000–2010, rates of 0.26%–1.5% were seen, indicating acceleration of thaw rates.
- Warming temperatures may also increase the presence of taliks, pockets of year-round thawed soil between the permafrost and the seasonally frozen active layer. Once taliks have formed, they accelerate permafrost thaw and thin the active layer. Taliks increased by more than 300% in a study grid in the Scotty Creek area between 2011 and 2013 and by 140% along study transects between 2011 and 2015. Permafrost thaw between 2011 and 2016 was more than 360% greater at points underlying a talik than at points without a talik.
- Wildfires in northern Alberta and southern Northwest Territories warmed the soil and caused permafrost thaw for up to 20 years and increased talik coverage for up to 50 years. Collapse scars form at triple the rate in areas that burned within the previous 30 years, and 23% of all peat plateau loss in the Fort Simpson area can be attributed to wildfires.

Knowledge Gaps

- Plots for permafrost and active-layer monitoring are mostly confined to the Mackenzie Valley and Mackenzie Delta.
- Research on permafrost thaw has been limited to specific plot sites. There is a need for landscape-level knowledge of rates of permafrost thaw, rates of formation of collapse scars, and the rate at which thawing is moving northward.
- Spatially distributed soil temperature data and models that can predict soil temperature and active-layer thickness are lacking.
- Data on the rate at which collapse scars connect with each other and with the regional drainage system, and the subsequent impacts on runoff rates over both the short and the longer term, are also lacking.
- Poorly drained and organic-rich lowland forests are the most sensitive to permafrost thaw and should undergo large-scale mapping.
- More research is needed to map the extent and depth of permafrost beneath forested uplands in the discontinuous permafrost zones.
- The effects of changing amounts and seasonal patterns of precipitation on permafrost thaw rates is currently not well understood. For example, less winter precipitation means less snow and less insulation for the soil during the cold months, leading to a colder permafrost layer. However, it may also mean earlier snow melt and less insulation during warmer spring months.
- The impact of seismic lines on drainage after permafrost thaw is not well understood, nor is the effectiveness or desirability of mulching linear disturbances to decrease thaw rates over the long term.

CHAPTER 5. WILDFIRE

5.1 Introduction

A history of natural and human-caused disturbance has shaped the forest ecosystems across Canada today. For example, sites that have historically been prone to droughts tend to support tree and other vegetation species that survive well under dry conditions, whereas sites that have experienced frequent wildfires tend to support species with an ability to survive fires or quickly recolonize a site after fire (Dale et al. 2001). In the boreal forest, wildfire has historically been one of the most important drivers affecting forest distribution and composition (Bernier et al. 2016). Indeed, tree species have evolved with, and have adapted to, local fire regimes over thousands of years, and key differences exist even within different parts of the boreal forest. For example, the fire regime in the Russian boreal forest is primarily driven by surface fires, and as a result, tree species in that region commonly have a thick-barked stem trait that enables them to survive these low-intensity fires (de Groot et al. 2013). In the North American boreal forest, however, the fire regime has historically been driven by stand-replacing crown fires (Bernier et al. 2016); as a result, tree species in this region have developed adaptations, such as serotinous and semiserotinous cones that are grouped at the tops of trees, to ensure seeds can survive these intense crown fires and then successfully regenerate afterward (de Groot et al. 2013). Climate change will affect these relationships, and both ecological and fire management implications should be expected into the future.

5.2 Fire Occurrence

Wildfire occurrence is determined by three environmental factors: ignition source, fuel, and weather conditions (Parisien and Moritz 2009). These factors interact in complex ways to create the fire regime of a given location (Parisien et al. 2011). Anthropogenic influences further complicate matters by increasing the number of ignitions and also introducing the effects of fire suppression (Stocks et al 2002; Parisien et al. 2011).

Tree-ring, charcoal, and pollen analyses indicate that the boreal forest of northwestern Canada (northern Alberta, northern British Columbia, and southern Northwest Territories) had a historical fire return interval of 50–100 years (Larsen and MacDonald 1998; Wallenius et al. 2011). Fire frequency in Canada’s northwestern boreal forest began to decline in the mid-19th century because of fire suppression, reaching a low of about every 300 years in the second half of the 20th century. Modern, mechanized fire suppression in this area began only in the 1950s and therefore does not fully account for the longer-term decline (Wallenius et al. 2011), although the prevention and suppression of human-caused wildfires along river corridors in the Northwest Territories began about 1910 (Pyne 2008). It is believed that most historical fires in the area were anthropogenic and that the decline in fire frequency is due to a decline in human-caused ignitions. These ignitions once accounted for about 95% of fires, but currently account for only about 54% (Wallenius et al. 2011), and in northern Canada as few as 30% of fires annually are human-caused (Stocks et al. 2002). The recent increase in fire frequency that has been attributed to climate change (Kasischke and Turetsky 2006) is small relative to the historical decline in fire occurrence, and the current rate of burning is still lower than historical levels (Wallenius et al. 2011).

The occurrence of wildfire is affected by many environmental factors, including geographic location, topography, and forest characteristics, which may differ in importance depending on spatial scale (Bernier et al. 2016). Parisien et al. (2011) studied the relative importance of various environmental factors on area burned at four spatial scales across Canada’s boreal region: 100, 1000, 10 000, and 100 000 km^2 (Table 16). They found that the temperature extreme, as measured by the average 99th percentile summer temperature, had one of the strongest positive correlations with area burned at all spatial scales (Figure 30). This relationship was not linear, because some of the warmest parts of the boreal forest, such as aspen parkland, had less flammable vegetation. The importance of summer temperature suggests that area burned will likely increase with climate change (Parisien et al. 2011). Estimates for increases in annual area burned in the Northwest Territories by the year 2100 range from 1.5 to 5.5 times baseline levels (Flannigan et al. 2005; Balshi et al. 2009; Boulanger et al. 2014).

Table 16. Dependent and exploratory variables used to model the spatial distribution of area burned in the boreal forest of Canada from 1980 to 2005. (Adapted, with permission, from Parisien et al. [2011].)

Variable	Description	Unit of measure
Fire		
AAB_Pct	Average annual area burned by fire, 1980–2005	%
Ignitions		
Ltg_Dens	Average annual density of lightning strikes per unit area, 1995–2005	strikes·km^{-2}·yr^{-1}
Road_Dens	Average road density in 2005	km·km^{-2}
HumFoot	Average human footprint value (index of human influence) in 2005	Unitless, 0 = lowest; 100 = highest
Vegetation		
Conif_Pct	Land cover, coniferous forest	%
DecidMixed_Pct	Land cover, deciduous and mixed forest	%
Nonfuel_Pct	Land cover, nonfuel (e.g., rock, glaciers)	%
Water_Pct	Land cover, bodies of water	%
CMI	Average annual climate moisture index (precipitation minus potential evapotranspiration), calculated monthly from 1 September to 31 August over the period 1980–2005	mm
GrowDays	Average annual growing season duration, starting when mean daily temperature is ≥5°C for 5 consecutive days (after 1 March) and ending when mean minimum temperature is ≤ −2°C, for the period 1980–2005	No. of days
Climate		
Temp99	Average of annual 99th percentile values of temperature between 15 May and 15 August, for the period 1980–2005	°C
Wind95	Average of annual 95th percentile values of wind speed between 15 May and 15 August, for the period 1980–2005	km·h^{-1}
ISI90	Average of annual 90th percentile values of Initial Spread Index (Canadian Forest Fire Danger Rating System index of ease of fire spread) between 15 May and 15 August, for the period 1980–2005	Unitless, open-ended
BUI99	Average of annual 99th percentile values of Build-Up Index (Canadian Forest Fire Danger Rating System index of drought severity) between 15 May and 15 August, for the period 1980–2005	Unitless, open-ended
Topography		
SurfArea_Ratio	Ratio of surface to area, an index of topographic roughness	Unitless, 1 = flat

5.3 Fire Potential

The ease of fire spread as measured by wind was also important at all scales, whereas the influence of the CMI, a measure of soil moisture, declined with increasing scale (Figure 30). The greater influence of CMI at smaller scales may also be related to low CMI increasing fuel flammability through drying in the short term, while low CMI influences vegetation patterns toward less flammable fuels over the longer term (Parisien et al. 2011). It is expected that climate change will lead to low moisture conditions (drought) occurring more frequently and over greater areas than in the past, which will alter fire regimes and behavior in the coming decades.

Most of the area burned in the boreal forest has been through high-intensity crown fires, with coniferous stands typically burning at higher intensities than deciduous stands. However, the intensity within a given fire is often highly variable (Bothwell et al. 2004). Mature black spruce stands are highly flammable, burning more readily than other vegetation or fuel types on the boreal landscape (Cumming 2001). Roughly 25% of peat plateaus dominated by black spruce have burned since the 1980s (Gibson et al. 2018).

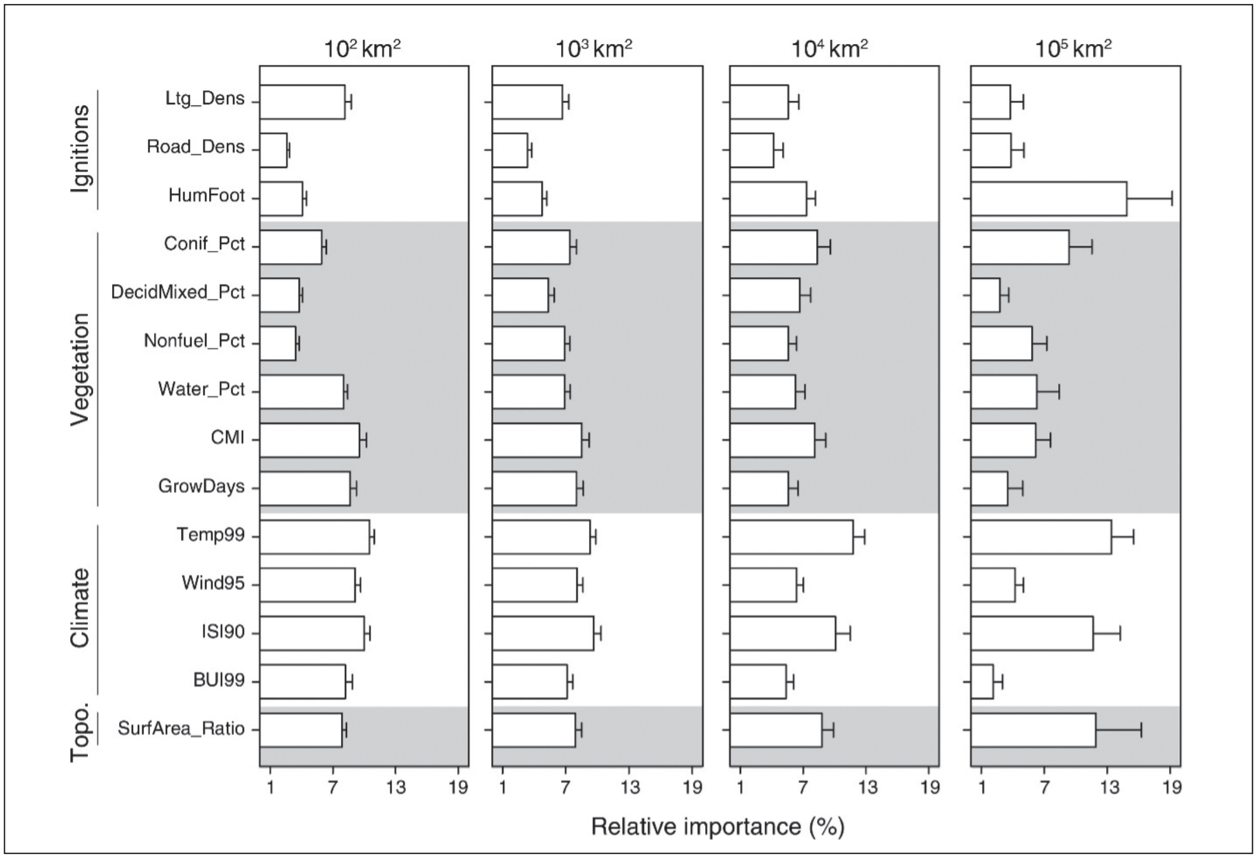


Figure 30. Relative contributions of explanatory variables to boosted regression tree models of area burned at four spatial scales in the boreal forest of Canada. See Table 16 for definitions of variables (Reproduced, with permission, from Parisien et al. 2011.)

Although the Canadian boreal forest has certain similarities throughout, regional differences in fire regime exist because of broad-scale gradients of climate, vegetation, and soils (Parisien et al. 2011). In addition, opposing factors may make characterizing a given fire regime difficult. For example, the northern boreal forest has a shorter fire season, but longer summer days and a greater proportion of more flammable conifer fuels than in areas farther south. Finer-scale regimes are even more difficult to predict because of complex interactions between and among factors (Parisien et al. 2011).

In the study by Parisien et al. (2011), the correlation between area burned and proportion of conifer fuel was strongly positive at the 10^5 km^2 scale but negative at the 10^2 and 10^3 km^2 scales, likely because wetlands are dominated by conifers. The human footprint on the landscape was negatively correlated with area burned, because of fire suppression, and this effect was strongest at the largest scale (Figure 30) (Parisien et al. 2011). Overall, climate variables contributed 35.7%–41.0% to the variability of area burned over the various scales, vegetation contributed 38.7%–46.8%, and ignition sources contributed 16.1%–25.6%, with the strongest influence of ignitions found at the 10^5 km^2 scale (Parisien et al. 2011).

Thompson et al. (2017) studied the fuel load (amount of flammable material per unit area) of 66 ground plots across a gradient of soil moisture and stand age in Wood Buffalo National Park in southern Northwest Territories (Tathlina Plain, within the Taiga Plains Mid-Boreal ecoregion) and northern Alberta (Figure 31). The highest fuel load was found in poorly drained upland conifer forests older than 97 years, with an average load almost four times that of the other sites (Table 17). However, most of the sites with high fuel loading were found in areas sheltered from fire by surrounding rich fens or in the lee of lakes and containing white spruce, which together indicate a long fire-free period. Discounting these wet upland conifer forests, there was no significant difference in the fuel load of wetland and upland sites (Table 17) (Thompson et al. 2017).

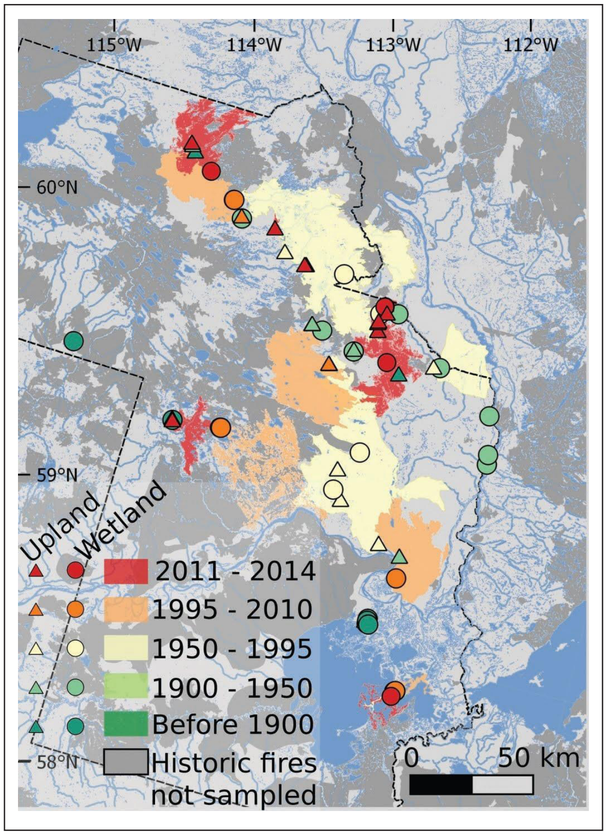


Figure 31. Measurement locations for wildfire fuel-loading study in Wood Buffalo National Park, Northwest Territories (Reproduced, with permission, from Thompson et al. 2017.)

Parisien et al. (2006) estimated fire potential characteristics for the Canadian level II ecoregions that experience significant fire activity, including the Taiga Plains, Taiga Shield, and Taiga Cordillera. The highest Seasonal Severity Rating (SSR), a measure of fire weather averaged over the fire season and derived from the Canadian Forest Fire Weather Index on the basis of weather observations from 1959 to 1997, was found in the Taiga Shield (2.8), followed by the Taiga Plains (2.5), and the Taiga Cordillera (1.7); the average across all Canadian ecoregions with a described fire regime was 2.23 (Parisien et al. 2006) (Table 18). However, the seasonal average is not necessarily the best indicator of a region's fire potential, as the number of high-intensity fires is influenced more by extreme fire weather than by the seasonal average (Parisien et al. 2006).

As with most fire metrics, SSR is predicted to increase with climate change, given that the severity of fire weather (increased evapotranspiration and changing rainfall patterns), the frequency of extreme weather events, and the length of fire seasons have all increased in recent years (Podur and Wotton 2010; de Groot et al. 2013). As such, suppressing wildfires will likely become much more difficult in the future, given that traditional methods of fighting wildfires can be ineffective for extremely intense fires. Furthermore, several researchers have predicted a significant increase in the prevalence of lightning-caused fires in the future because of changing weather patterns (Wotton et al. 2005; Krawchuck et al. 2009; Podur and Wotton 2010). Lightning-caused fires can be problematic because they burn a disproportionate amount of forested area; in addition, they often occur in remote areas, so detection and appropriate action can take longer, and they also typically occur in clusters (Podur and Wotton 2010). For instance, an examination of Ontario fire records showed that lightning-caused wildfires accounted for approximately 40% of all fires detected between 1976 and 2004, but 66% of the area burned during that period (Podur and Wotton 2010). The combination of intense fire behavior and more fire ignitions likely means there will be years when fire suppression capacity becomes overwhelmed in Canada, which will further increase the amount of area burned annually (Podur and Wotton 2010). Indeed, extreme fire years are predicted to become more commonplace in the future, which is expected to significantly increase expenditures on fire suppression (Hope et al. 2016). An analysis of future wildfire suppression costs predicted that by 2100, on a national scale, average annual fire management costs across Canada will increase by 60% under a low climate change scenario and by as much as 120% under a high climate change scenario (Hope et al. 2016). The researchers stated that in many provinces and territories, annual fire suppression costs that are currently considered extreme (occurring once every approximately 10 years) are projected to become much more commonplace and by 2100 may occur every year or two; furthermore, most Canadian provinces are predicted to start experiencing these increases in area burned and suppression costs by 2050 (Hope et al. 2016).

Regions of the Northwest Territories considered to have active fire suppression were derived from maps produced by fire management agencies. The Taiga Plains ecoregion had 20% of its area within fire suppression zones, while the Taiga Shield ecoregion had only 2% and the Taiga Cordillera ecoregion had none; the overall Canadian average was 37.7%.

Table 17. Fuel load by ecosite and age class in the Northwest Territories. (Reproduced, with permission, from Thompson et al. [2017].)

Time since fire (yr)	Ecosite name	Ecosite code[a]	*n*	Fuel load ($kg{\cdot}m^{-2}$) Litter	WD[b]	Live[c]	Shrub	Under[d]	Total fuel load ($kg{\cdot}m^{-2}$) Σ surface	Σ crown	Σ fuel
Uplands											
1–95	Dry pine–lichen	a	7	0.31	0.03	0.03	0.00	0.10	0.48	0.73	1.21
95–128	Dry pine–lichen	a	1	0.10	0.14	0.02	0.00	0.00	0.26	1.52	1.78
1–95	Pine–mixed-wood	b	5	0.24	0.03	0.11	0.00	0.01	0.39	1.09	1.48
95–146	Pine–mixed-wood	b	3	0.23	0.02	0.05	0.00	0.01	0.31	0.94	1.25
1–95	Mesic upland	d	5	0.18	0.03	0.12	0.00	0.00	0.34	0.91	1.25
95–250	Mesic upland	d	3	0.27	0.03	0.07	0.00	0.00	0.38	1.63	2.01
1–95	Moist upland conifer	g	3	0.29	0.06	0.07	0.00	0.23	0.65	0.35	1.01
95–155	Moist upland conifer	g	4	0.21	0.04	0.06	0.00	0.13	0.43	4.11	4.54
Wetlands											
1–97	Bog	l	7	0.16	0.02	0.14	0.00	0.08	0.39	0.09	0.48
1–97	Treed fen	j	4	0.08	0.00	0.10	0.02	0.03	0.23	0.59	0.81
97–113	Treed fen	j	1	0.06	0.04	0.10	0.00	0.02	0.22	0.55	0.77
1–97	Open fen	k	11	0.35	0.00	0.18	0.03	0.00	0.56	0.00	0.56
97–155	Open fen	k	5	0.29	0.02	0.15	1.62	0.01	0.50	0.00	0.53
1–97	Saline wet meadow	l	4	0.18	0.00	0.20	0.00	0.00	0.38	0.00	0.38
97–250	Saline wet meadow	l	1	0.21	0.05	0.16	0.00	0.05	0.47	1.61	2.08

[a] *Ecosite codes as set out by Beckingham and Archibald (1996).*

[b] *WD = woody debris with diameter <1 cm.*

[c] *Live = sum of live nontree surface fuels such as forbs and small shrubs.*

[d] *Under = understory trees with diameter at breast height <3 cm.*

Table 18. Factors influencing the fire regime for the three forested ecoregions of the Northwest Territories. (Data source Parisien et al. [2006].)

Ecoregion	Seasonal Severity Rating	Fuel ratio conifer–nonconifer	Proportion of ecoregion made up of lakes and nonfuels (%)	Proportion of ecoregion with intense fire suppression (%)	Proportion of area with human land use (%)	Ignitions ratio human–lightning
Taiga Plains	2.5	1.21	23	20	22	0.23
Taiga Shield	2.8	3.32	47.1	2	2	0.26
Taiga Cordillera	1.7	0.38	55.1	0	14	0.09
Average of Canadian ecoregions	2.23	2.83	28.82	37.7	24.4	0.7

The extent of human land use was calculated by drawing a 5-km buffer around all roads and taking the percentage of buffer area over total area. The percentage of human land use was 22% in the Taiga Plains, 14% in the Taiga Cordillera, and 2% in the Taiga Shield, with the Canadian average being 24.4%. The ratio of human- to lightning-caused fires for the period 1980–1999 was calculated from a Canadian database of reported fires. The Taiga Plains had a ratio of 0.23, the Taiga Shield a ratio of 0.26, and the Taiga Cordillera a ratio of 0.09; the Canadian average was 0.7 (Table 18) (Parisien et al. 2006).

In summary, the Taiga Shield ecoregion had a higher SSR, a higher proportion of more flammable coniferous fuel (Figure 32), and less area with

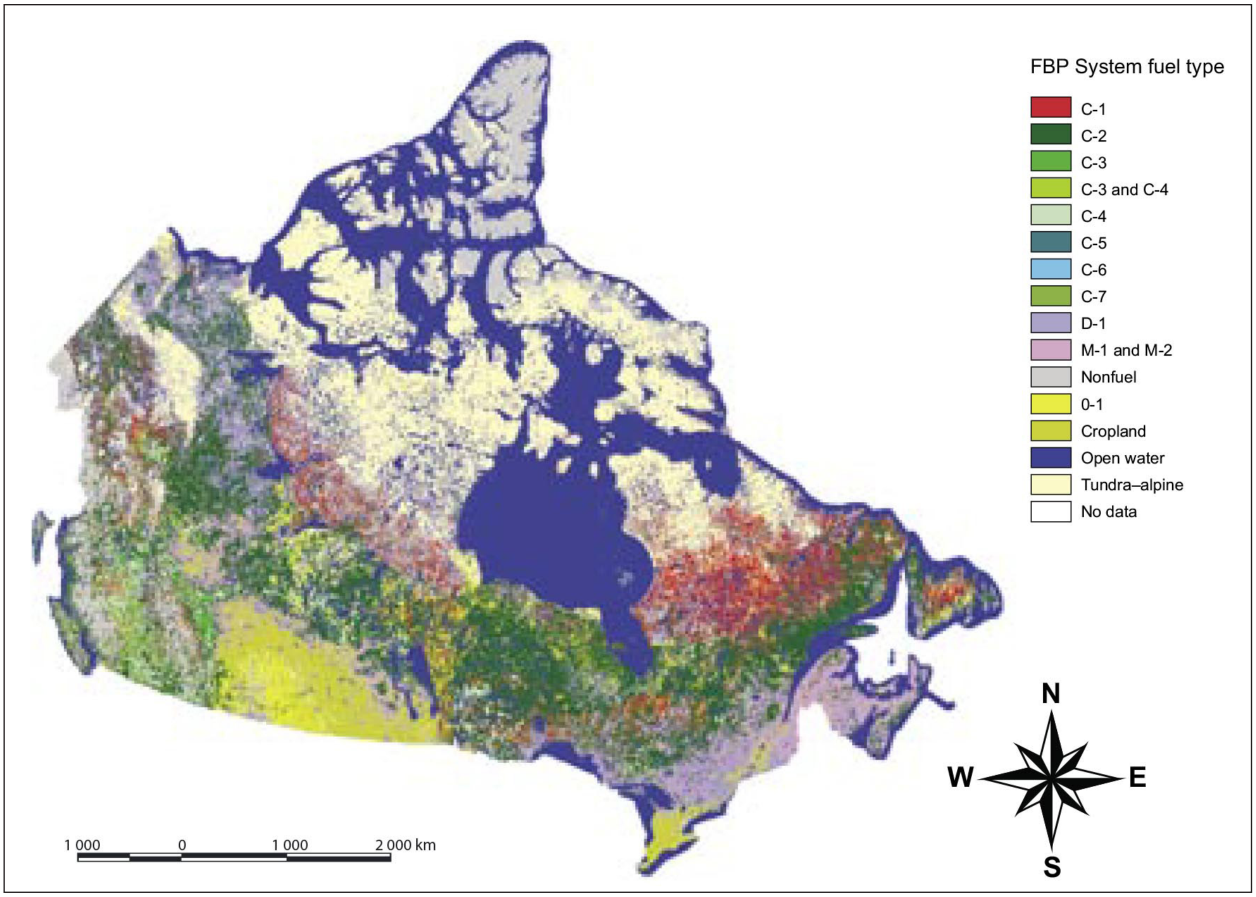


Figure 32. Canadian fuel-type map. C-1 = spruce–lichen woodland; C-2 = boreal spruce; C-3 = mature jack or lodgepole pine; C-4 = immature jack or lodgepole pine; C-5 = red and white pine; C-6 = conifer plantation; C-7 = ponderosa pine/Douglas-fir; D-1 = leafless aspen; M-1, M-2 = boreal mixed-wood; M-3, M-4 = dead balsam fir/mixed-wood; S-1 = jack or lodgepole pine slash; S-2 = spruce/balsam slash; S-3 = coastal cedar/hemlock/Douglas-fir slash; O-1 = grassland. (Reproduced, with permission, from Nadeau et al. 2005.)

fire suppression than the Taiga Plains ecoregion. Conversely, the Taiga Plains had a smaller nonfuel area (Figure 32) and greater area of human use. The Taiga Cordillera ecoregion had significantly lower fire potential than the other two ecoregions.

5.4 Fire History

Using an analysis that compared the fire potential characteristics of level II ecoregions with fire data contained in the CFS Large Fire Database for the period 1980–1999, Parisien et al. (2006) found the mean annual number of fires for that period in the Taiga Plains ecoregion was 5.3 per 10 Mha and the mean annual area burned was 433 507 ha. The Taiga Shield ecoregion had an annual average of 5.4 fires per 10 Mha and an annual area burned of 311 222 ha. The Taiga Cordillera ecoregion had an annual average of 3.0 fires per 10 Mha and an annual area burned of 44 473 ha. For all Canadian ecoregions, the annual average was 4.3 fires per 10 Mha and an annual area burned of 207 436 ha (Table 19) (Parisien et al. 2006). The interannual variation in area burned was large, and the standard deviations for these three ecoregions were all greater than their means. The maximum fire size at the time of the study was 887 804 ha in the Taiga Plains ecoregion, 289 236 ha in the Taiga Shield ecoregion, and 61 715 ha in the Taiga Cordillera ecoregion. The maximum fire size for the Taiga Plains was also the largest among all ecoregions (Table 19) (Parisien et al. 2006).

Thus, the Taiga Plains ecoregion had the largest area burned and the largest fire size of the three ecoregions, despite having a slightly lower SSR and substantially lower ratio of coniferous to other fuels than the Taiga Shield ecoregion. These findings may be due to the low cover of nonfuel landforms and a higher level of human land use in the Taiga Plains ecoregion, although the proportion of human-caused fires was higher in the Taiga

Table 19. **Fire data for the three forested ecoregions of the Northwest Territories from the Large Fire Database of the Canadian Forest Service, 1980–1999.** (Data source Parisien et al. [2006].)

Ecoregion	Annual no. of fires (mean ± SD[a])	Annual area burned (ha) (mean ± SD)	Maximum fire size (ha)	Eccentricity (mean ± SD)	Shape complexity (mean ± SD)	Clustering (NN[b])
Taiga Plains	26.4 ± 15.4	433 507 ± 524 466	887 804	1.77 ± 0.58	2.27 ± 0.77	0.278
Taiga Shield West	26.9 ± 24.9	311 222 ± 568 119	289 236	1.76 ± 0.57	2.24 ± 0.86	0.185
Taiga Cordillera	15.2 ± 11.6	44 473 ± 51 256	61 715	1.76 ± 0.67	1.88 ± 0.63	0.288
Average of Canadian ecoregions	21.4 ± 12.0	207 436 ± 197 903	887 804	1.91± 0.61	1.96 ± 0.75	0.362

[a] *SD = standard deviation.*
[b] *NN = Nearest neighbor statistic.*

Shield ecoregion (Parisien et al. 2006). Principal components analysis also showed that the Taiga Plains and Taiga Shield ecoregions had similar fire regimes that differed greatly from that of the Taiga Cordillera ecoregion. In addition to the described fire factors on which the Taiga Cordillera had low scores, the mountainous terrain serves to limit horizontal fuel continuity (Parisien et al. 2006).

Fire clustering represents areas that consistently see more fire activity. All three ecoregions had a high degree of fire clustering, with the most clustering in the Taiga Shield (0.185) and less in the Taiga Plains (0.278) and the Taiga Cordillera (0.288); the average across all Canadian ecoregions was 0.362 (Table 19). The northern part of the Taiga Shield supports tundra vegetation with exposed rock that hinders fire spread, thus concentrating fire activity in the south and increasing the extent of clustering. Other reasons for clustering include weather patterns (increased lightning) and anthropogenic activity (Parisien et al. 2006).

Earlier Canadian fire maps and databases have recently been improved by the National Burned Area Composite (NBAC) database, developed by the CFS (Figure 33) (Guindon et al. 2014; Natural Resources Canada 2018a). The NBAC database contains data on area burned since 1986 and is an improvement over previous fire maps because it uses finer-resolution Landsat satellite images, removes bodies of water and unburned patches from burn polygons, and includes previously missed fires (R.S. Skakun, Natural Resources Canada, personal communication, 2018). Annual area burned and annual fire frequency have been summarized by level III ecoregion for the period 1986–2018 (Figure 34 and Figure 35, respectively). The mean annual area burned in each level III ecoregion is shown in Table 20, and can be summarized as follows:

- Taiga Plains ecoregion: 246 837 ha, or 0.51% of the entire area, for a fire return interval of 200 years.
- Taiga Shield ecoregion: 189 718 ha, or 0.57% of the entire area, for a fire return interval of 167 years.
- Boreal and Taiga Cordillera ecoregions: 50 939 ha, or 0.33% of the entire area, for a fire return interval of 333 years.

The annual number of fires was 9.2 per 10 Mha in the Taiga Plains ecoregion, 13.6 per 10 Mha in the Taiga Shield ecoregion, and 5.6 per 10 Mha in the Taiga Cordillera ecoregion (Table 21). The entire forested area of the Northwest Territories had an annual average of 10.1 fires per 10 Mha and an average annual area burned of 0.5%.

A comparison of the fire regimes of the three ecoregions confirms the conclusion of Parisien et al. (2006) that the Taiga Plains and Taiga Shield share a similar fire regime that is more severe than the Boreal and Taiga Cordillera. However, the NBAC values of mean annual area burned for the Taiga Plains and Taiga Shield are far lower than the values obtained using the Large Fire Database because the NBAC values omit unburned patches, whereas the annual number of fires is significantly higher because the NBAC values include smaller fires and other previously unrecorded fires.

To evaluate changes over time in number of fires and area burned, the period 1986–2018 was divided into two periods (1986–2002 and 2003–

2018). Trends differed by ecoregion. For example, percent annual area burned (PAAB) in the Taiga Plains decreased from 0.60% during the period 1986–2002 to 0.42% during the period 2003–2018, whereas in the Taiga Shield it increased from 0.48% to 0.68% (Table 20). Variable values and trends were seen between level III ecoregions within each level II ecoregion as well. Overall, the average annual area burned in the forested area of the Northwest Territories decreased from 520 163 ha for 1986–2002 to 452 785 ha for 2003–2018, or from 0.54% to 0.47% (Table 20). Although the total area burned decreased between the periods 1986–2002 and 2003–2018, the average annual number of fires in the forested area of the Northwest Territories increased from 77.6 to 118.7 (Table 21).

As discussed above, ecoregions have traditionally been used to define fire regimes because these units are defined by the interaction of broad-scale climate, vegetation, geology, soils, human activity, and physiographic characteristics (Ecological Stratification Working Group 1996), which are known to also affect the occurrence and behavior of wildfires. However, Boulanger et al. (2012) evaluated the data contained in the Canadian National Fire Database maintained by the CFS and mapped the number of fires, the percent area burned, and the fire start date data across a Canada-wide grid of 40 × 40 km cells. The results from each cell were then combined to create regions with maximum internal homogeneity, which resulted in 33 Homogeneous Fire Regime (HFR) zones (Boulanger et al. 2012). These HFR zones

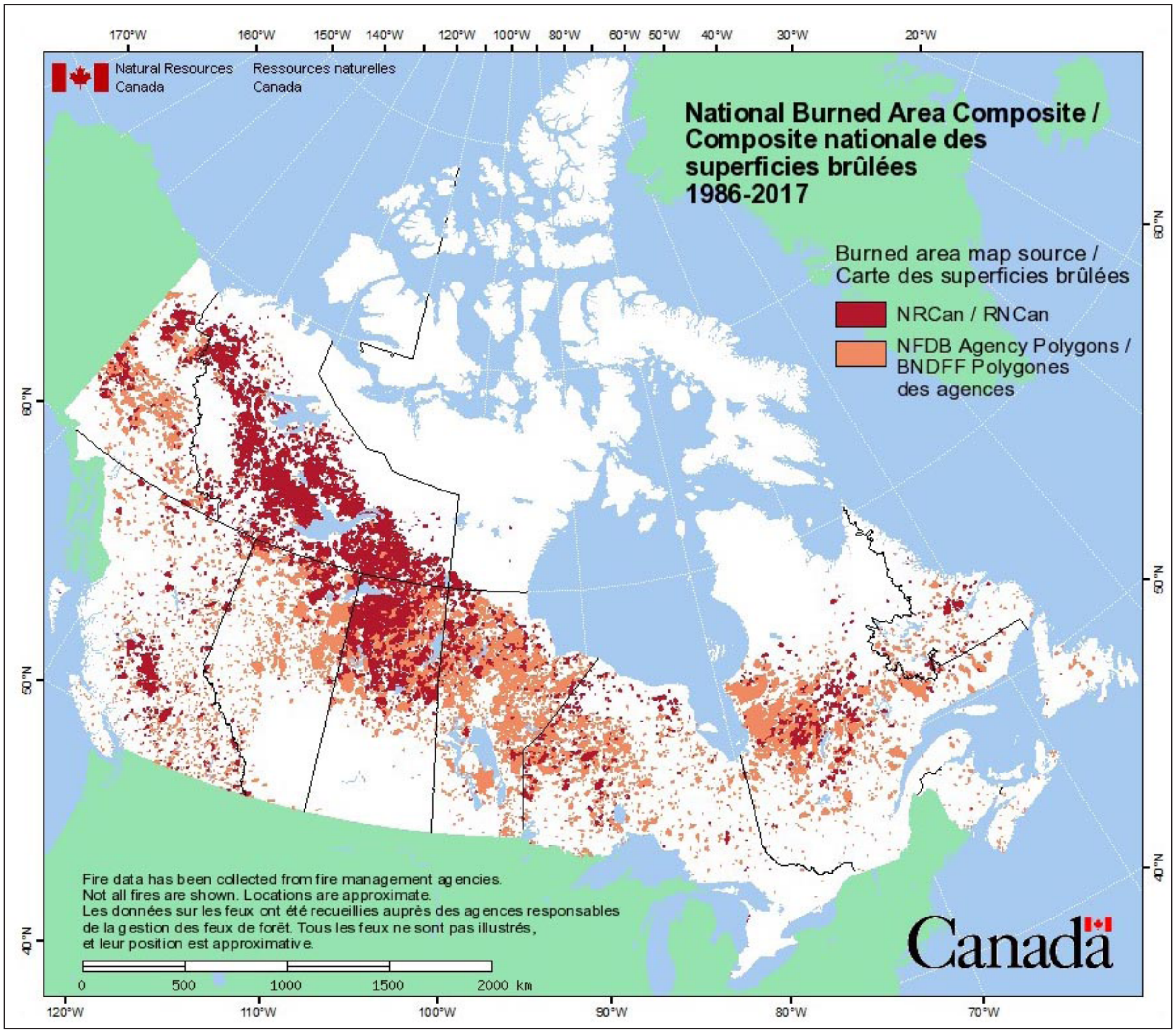


Figure 33. National Burned Area Composite map showing areas burned, 1986–2017. (Source: Natural Resources Canada (NRCan), Canadian National Fire Database (NFDB), http://cwfis.cfs.nrcan.gc.ca/ha/nfdb?type=nbac&year=9999, accessed 24 Apr. 2020.)

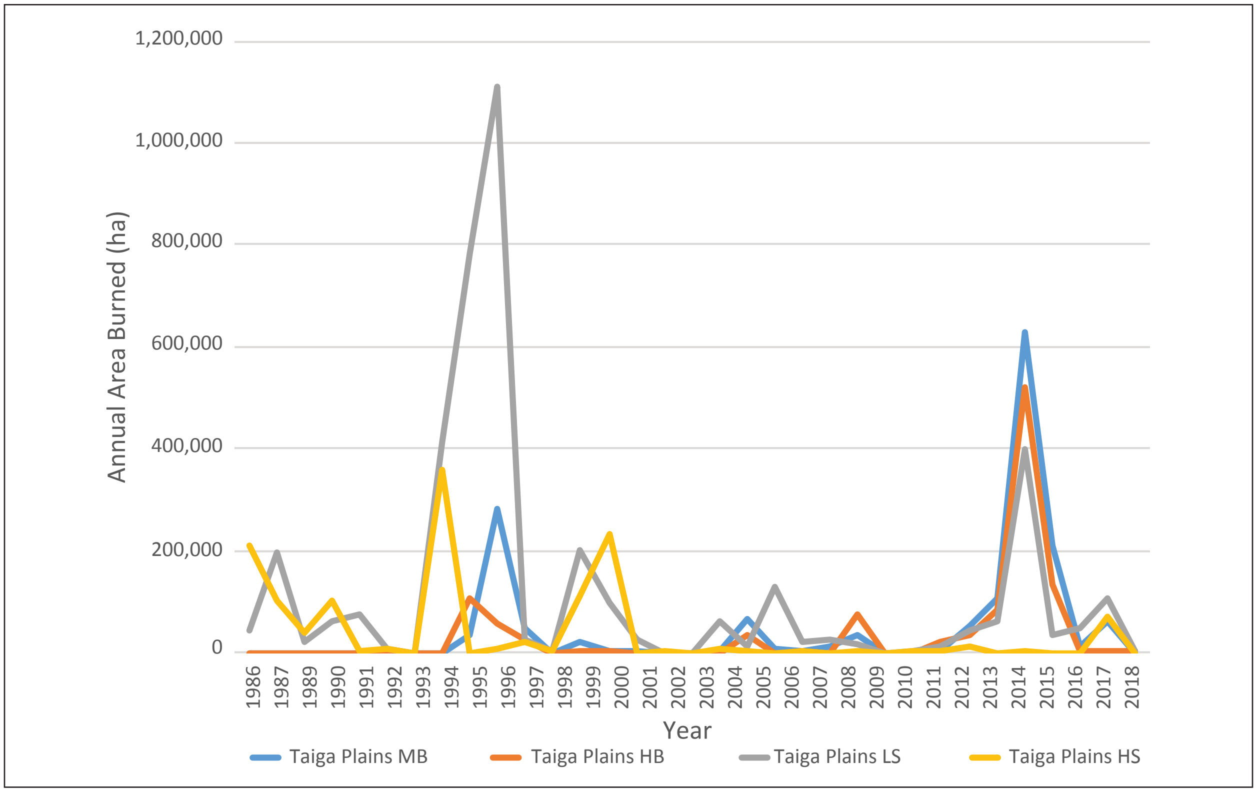


Figure 34. Annual area burned (AAB) in level III ecoregions of the Taiga Plains, Northwest Territories, 1986–2018. MB = Mid-Boreal, HB = High Boreal, LS = Low Subarctic, HS = High Subarctic. (Data from National Burned Area Composite fire maps, supplied by R. Skakun, Natural Resources Canada.)

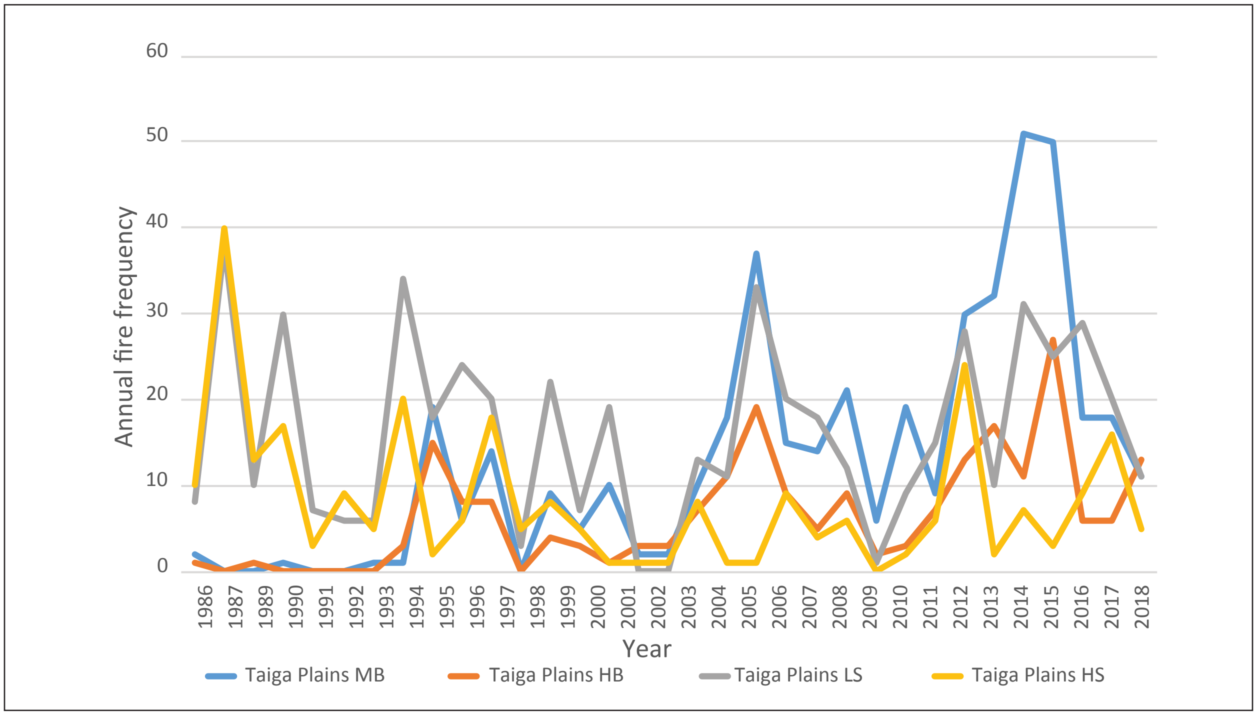


Figure 35. Annual fire frequency in level III ecoregions of the Taiga Plains, Northwest Territories, 1986–2018. MB = Mid-Boreal, HB = High Boreal, LS = Low Subarctic, HS = High Subarctic. (Data from National Burned Area Composite fire maps, supplied by R. Skakun, Natural Resources Canada.)

Table 20. Annual area burned and percent of total area burned, by level III ecoregion, from National Burned Area Composite fire maps. (Data supplied by R. Skakun, Natural Resources Canada.)

		1986–2002		2003–2018		1986–2018	
Level II ecoregion	Level III ecoregion	Mean area burned (ha)	Mean proportion burned (%)	Mean area burned (ha)	Mean proportion burned %	Mean area burned (ha)	Mean proportion burned %
Taiga Plains	Mid-Boreal	23 687	0.22	75 705	0.71	48 908	0.46
	High Boreal	11 926	0.28	58 064	1.38	34 296	0.81
	Low Subarctic	180 882	1.12	61 932	0.38	123 209	0.76
	High Subarctic	71 236	0.42	7 686	0.05	40 424	0.24
	Subtotal	287 731	0.60	203 387	0.42	246 837	0.51
Taiga Shield	Mid-Boreal	821	0.12	14 761	2.22	7 580	1.14
	High Boreal	50 490	0.60	119 727	1.41	84 060	0.99
	Low Subarctic	97 139	0.85	69 447	0.61	83 712	0.73
	High Subarctic	9 445	0.08	19 595	0.16	14 366	0.12
	Subtotal	157 895	0.48	223 530	0.68	189 718	0.57
Boreal Cordillera	Mid-Boreal	2 216	0.09	2 244	0.09	2 229	0.09
	High Boreal	17 830	0.54	11 728	0.35	14 871	0.45
Taiga Cordillera	Low Subarctic	54 388	0.72	11 834	0.16	33 756	0.45
	High Subarctic	103	0.004	62	0.003	83	0.003
	Subtotal	74 537	0.48	25 868	0.17	50 939	0.33
Total		520 163	0.54	452 785	0.47	487 494	0.50

Table 21. Annual number of fires (absolute) and number of fires per 10 Mha, by level III ecoregion, from National Burned Area Composite fire maps. (Data supplied by R. Skakun, Natural Resources Canada.)

		1986–2002		2003–2018		1986–2018	
Level II ecoregion	Level III ecoregion	Mean no. of fires	Mean no. of fires per 10 Mha	Mean no. of fires	Mean no. of fires per 10 Mha	Mean no. of fires	Mean no. of fires per 10 Mha
Taiga Plains	Mid-Boreal	4.2	3.9	22.4	21.0	13.1	12.3
	High Boreal	2.9	6.9	10.3	24.5	6.5	15.4
	Low Subarctic	14.8	9.2	17.9	11.1	16.3	10.1
	High Subarctic	9.6	5.6	6.4	3.8	8.1	4.8
	Subtotal	31.5	6.6	57.0	11.9	44.0	9.2
Taiga Shield	Mid-Boreal	0.5	7.6	4.0	60.6	2.2	33.3
	High Boreal	10.8	12.7	20.8	24.5	15.6	18.4
	Low Subarctic	23.5	20.5	20.1	17.6	21.8	19.1
	High Subarctic	3.8	3.1	6.7	5.4	5.2	4.2
	Subtotal	38.6	11.7	51.6	15.6	44.8	13.6
Boreal Cordillera	Mid-Boreal	1.0	4.2	1.4	5.9	1.2	5.0
	High Boreal	2.3	6.9	4.3	13.0	3.2	9.6

Table 21. Concluded

		1986–2002		2003–2018		1986–2018	
Level II ecoregion	Level III ecoregion	Mean no. of fires	Mean no. of fires per 10 Mha	Mean no. of fires	Mean no. of fires per 10 Mha	Mean no. of fires	Mean no. of fires per 10 Mha
Taiga Cordillera	Low Subarctic	4.1	5.5	4.2	5.6	4.1	5.5
	High Subarctic	0.1	0.4	0.3	1.2	0.2	0.8
	Subtotal	7.5	4.8	10.2	6.5	8.7	5.6
Total		77.6	8.0	118.8	12.3	97.5	10.1

captured significantly more spatial variability in fire attributes than the traditionally used ecoregions (Boulanger et al. 2012). Further refinement of this system incorporated vegetation classes and fire data from other databases (Boulanger et al. 2014) and resulted in 16 HFR zones across Canada (Figure 36). The PAAB and the annual number of large fires calculated for the 16 HFR zones (Figure 37, Figure 38) showed substantial deviations from the corresponding metrics calculated according to ecoregion (Boulanger et al. 2014). Seven HFR zones are found at least in part in the Northwest Territories (Figure 36) (Boulanger et al. 2014):

- The Pacific HFR zone includes the center of the western Northwest Territories, roughly corresponding to the Taiga Cordillera ecoregion. That zone had a mild fire regime with an annual mean of 17 fires per 10 Mha and a PAAB of 0.04%.
- The Interior Cordillera HFR zone includes the western portion of the Northwest Territories south of the Pacific HFR zone and had an annual mean of 9.2 fires per 10 Mha and a PAAB of 0.32%.
- The Southern Prairies HFR zone includes the southwestern part of the Northwest Territories. It had an annual mean of 4.0 fires per 10 Mha and a PAAB of 0.20%.
- The Great Slave Lake HFR zone includes the central Northwest Territories, from the Alberta border to the area south of Great

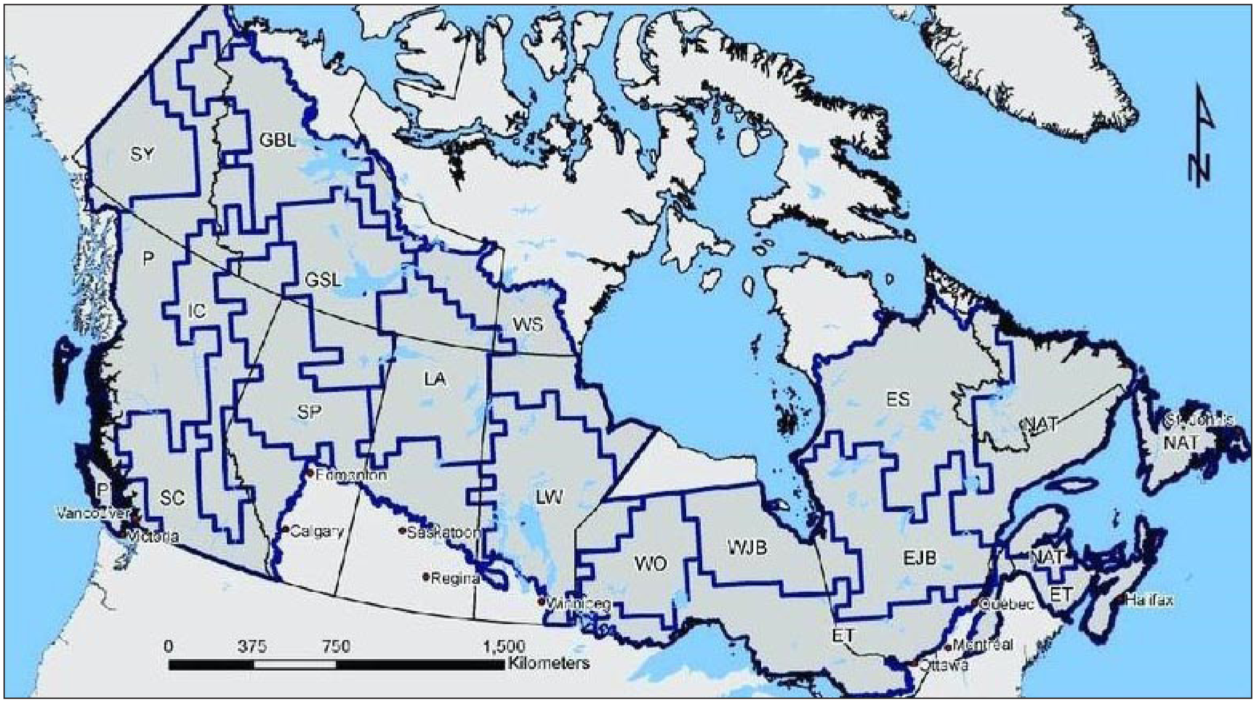


Figure 36. Canadian Homogeneous Fire Regime zones. Zones within the Northwest Territories are P = Pacific, IC = Interior Cordillera, GBL = Great Bear Lake, GSL = Great Slave Lake, SP = Southern Prairies, LA = Lake Athabasca, and WS = Western Subarctic. (Reproduced, with permission, from Bernier et al. 2016.)

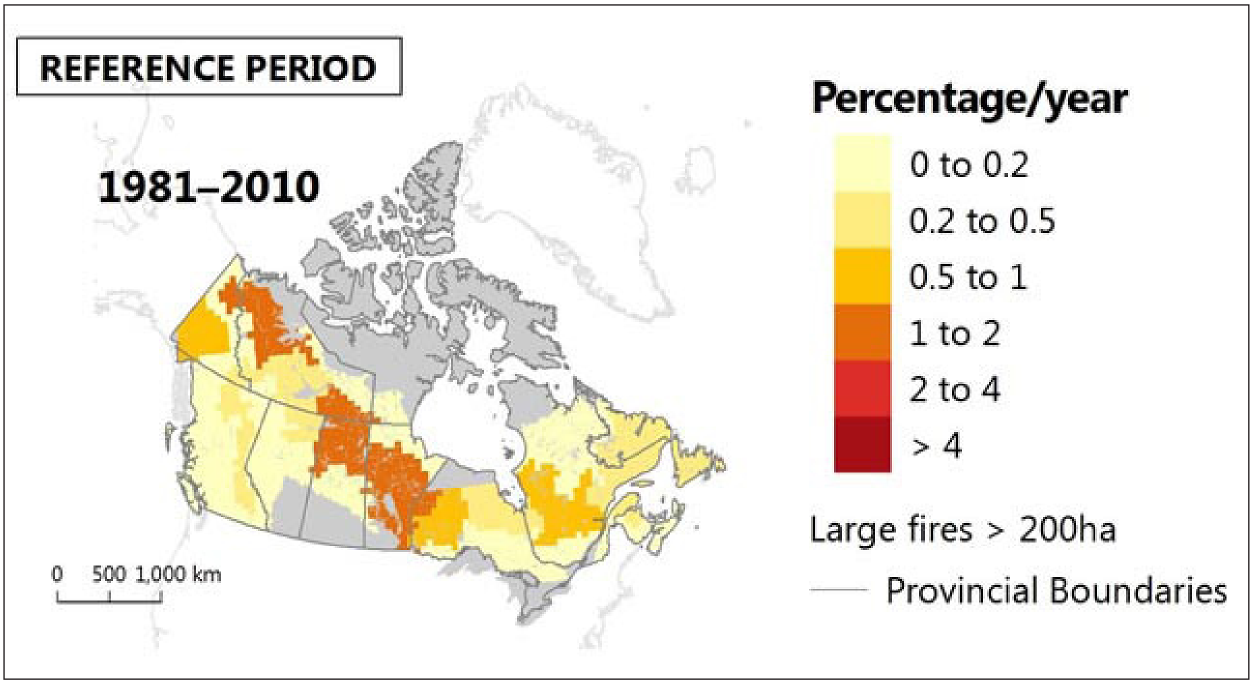


Figure 37. Annual percent area burned, averaged by Homogeneous Fire Regime zones. (Source: Natural Resources Canada, https://www.nrcan.gc.ca/climate-change/impacts-adaptations/climate-change-impacts-forests/forest-change-indicators/fire-regime/17780, accessed 15 May 2020.)

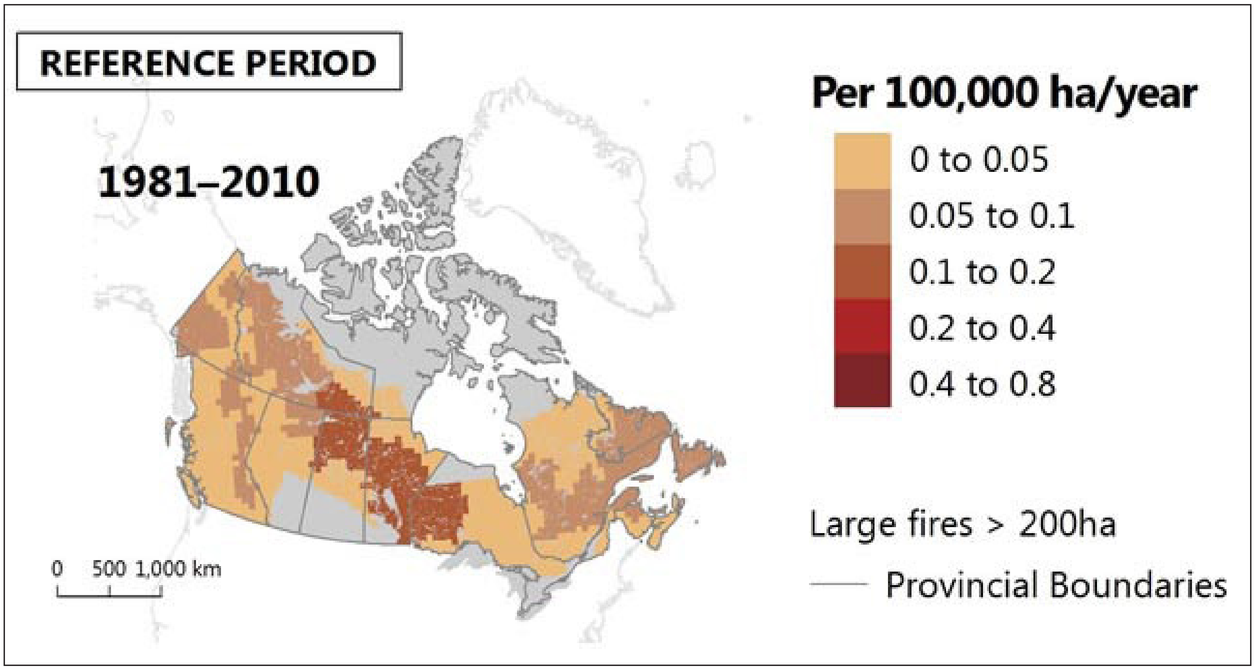


Figure 38. Annual number of large fires per 100 000 ha, averaged by Homogeneous Fire Regime zones. (Source: Natural Resources Canada, https://www.nrcan.gc.ca/climate-change/impacts-adaptations/climate-change-impacts-forests/forest-change-indicators/fire-regime/17780, accessed 15 May 2020.)

Bear Lake. It had an annual mean of 8.2 fires per 10 Mha and a PAAB of 1.02%.

- The Great Bear Lake HFR zone stretches from Great Bear Lake to Inuvik and had an annual mean of 7.3 fires per 10 Mha and a PAAB of 0.64%.
- The Lake Athabasca HFR zone includes the area of the Northwest Territories along the Saskatchewan border sloping northwest to the east arm of Great Slave Lake. It had both the highest number of annual fires of all 16 zones, with a mean of 14.6 fires per 10 Mha, and the highest PAAB, at 1.48%.
- The Western Subarctic HFR zone includes much of the Taiga Shield Subarctic ecoregions of the Northwest Territories. It has a mild fire regime, with an annual mean of 0.98 fires per 10 Mha and a PAAB of 0.10%.

Fire data from 1959 to 1995 were correlated with monthly climate variables for each of these HFR zones to build a model of annual number of fires and PAAB as a function of climate; the model then underwent validation with data from 1996 to 2011. The model was used to project changes to the fire regime within the HFR zones, as caused by future climate change, for the periods 2011–2040, 2041–2070, and 2071–2100 using the Special Report Emmissions Scenario A2 of the Intergovernmental Panel on Climate Change (Boulanger et al. 2014). This modeling predicted 3.7- and 3.0-fold increases in annual area burned and annual number of fires, respectively, by 2100 (Boulanger et al. 2014).

Although the HFR zones explain much of the variability in fire regimes across Canada's boreal forest, finer-scale variability also exists within HFR zones because of changes to forest composition and either forest age or biomass over the landscape. Bernier et al. (2016) used forest attribute maps from Beaudoin et al. (2014) and Canada-wide maps of area burned from Guindon et al. (2014) to study the effects of forest characteristics on fire frequency. They found that conifer forests and older or higher-biomass forests were more likely to burn than deciduous and young or lower-biomass forests (Figure 39). This approach allowed them to create a map showing finer-scale differences in fire interval within each broader-scale HFR zone (Figure 40).

Several field-based studies of fire history have also been conducted at specific locations in the Northwest Territories over the past few decades. This fieldwork (based on the presence of fire scars) indicated that the fire return interval is 22–28 years in the Nahanni National Park Reserve and 23–27 years in the Mackenzie Bison Sanctuary (Bothwell et al. 2004). However, when these results were compared with data in the CFS Large Fire Database, enormous discrepancies were discovered. According to information in the database, the fire return intervals are estimated to be 1142 years for the Nahanni National Park Reserve and 199 years for the Mackenzie Bison Sanctuary (Bothwell et al. 2004). There are several possible explanations for these discrepancies, including a difference in the periods covered by the two data sources; the assumption of the researchers who performed the fire scar study that any evidence of fire scars on a site meant that the entire site had burned; the tendency for fire scars to heal over time, such that they are not visible decades after a fire; the fact that the CFS database contains information only for fires larger than 200 ha; and the smaller area covered by the study sites evaluated by Bothwell et al. (2004). The combination of these factors likely means the estimate of fire return interval based on field studies is too short, and the estimate of fire return interval based on the database is too long (Bothwell et al. 2004).

A CFS research team led by M.-A. Parisien is currently reconstructing the fire history of the past 200 years along a 300-km transect in the Northwest Territories using field evidence of fire-scarred trees. Estimates of burn rates, fire sizes, and fire intervals will be derived from the scars, providing evidence of any changes to the fire regime over that period. Feedback mechanisms between fire and vegetation will also be studied to understand the time required for a stand to re-burn after a fire.

5.5 Fire Severity

Fire severity refers to the extent of fire effects on an ecosystem, such as biomass loss or changes to soil properties (Keeley 2009). Various field-collected and remotely sensed metrics of fire severity have been developed to measure combustion, tree mortality, or a combination of the two for various forest strata. However, their accuracy for the boreal forest is sometimes debated (Whitman et al. 2018b). In 2014, the Northwest Territories experienced a particularly severe fire season, with more than 380 fires burning a total area of about 2.9 Mha. The 2014 fires were therefore used to

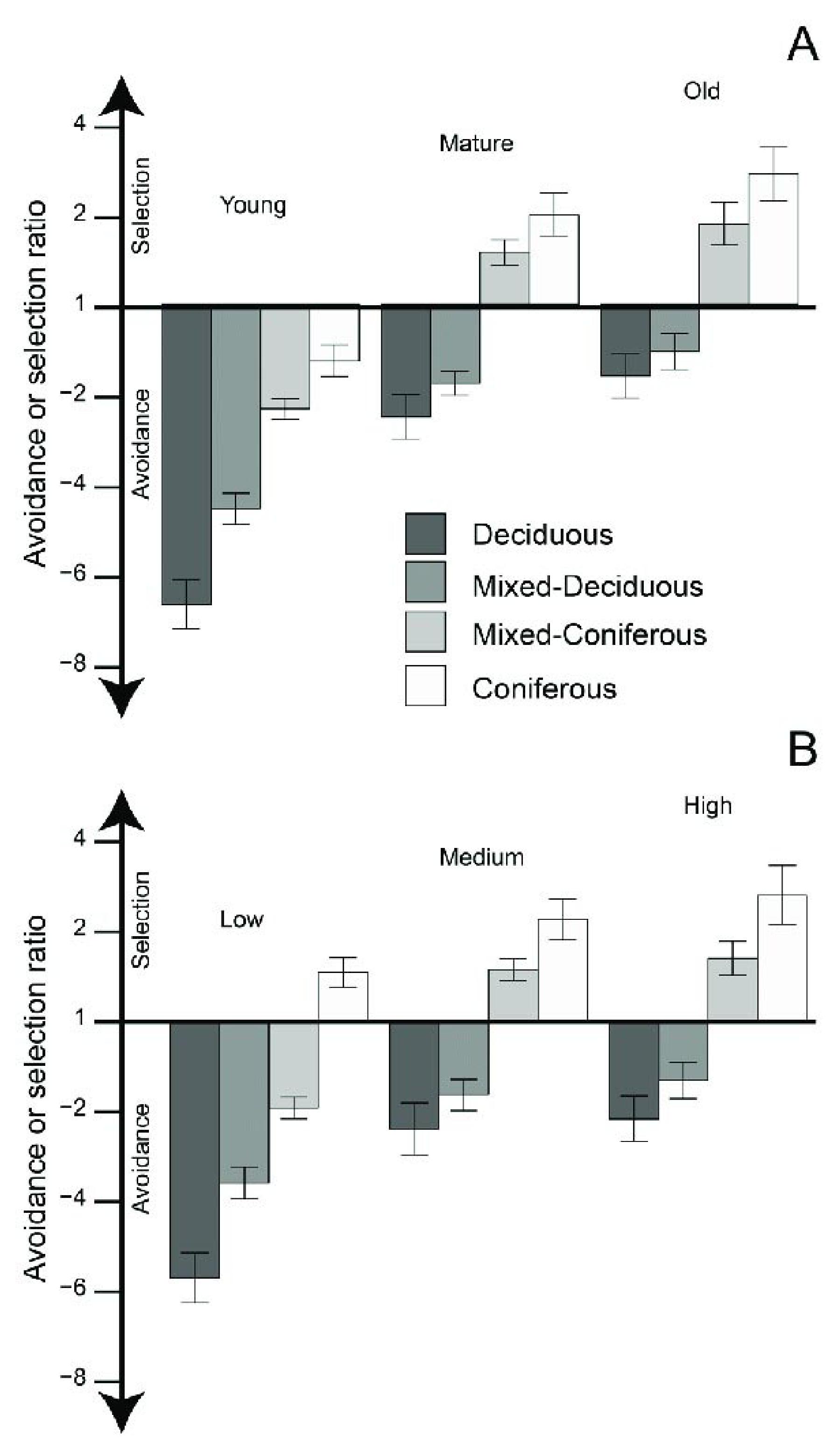


Figure 39. **Mean relative likelihood of fire across the 16 Homogeneous Fire Regime zones for (A) classes of composition × age and (B) classes of composition × biomass.** A value of 1 indicates that fire selectivity is random. Values greater than 1 indicate areas that are more likely to burn than by random chance, and values less than 1 indicate areas that are less likely to burn than by random chance. Vertical bars indicate standard deviation. (Reproduced, with permission, from Bernier et al. 2016.)

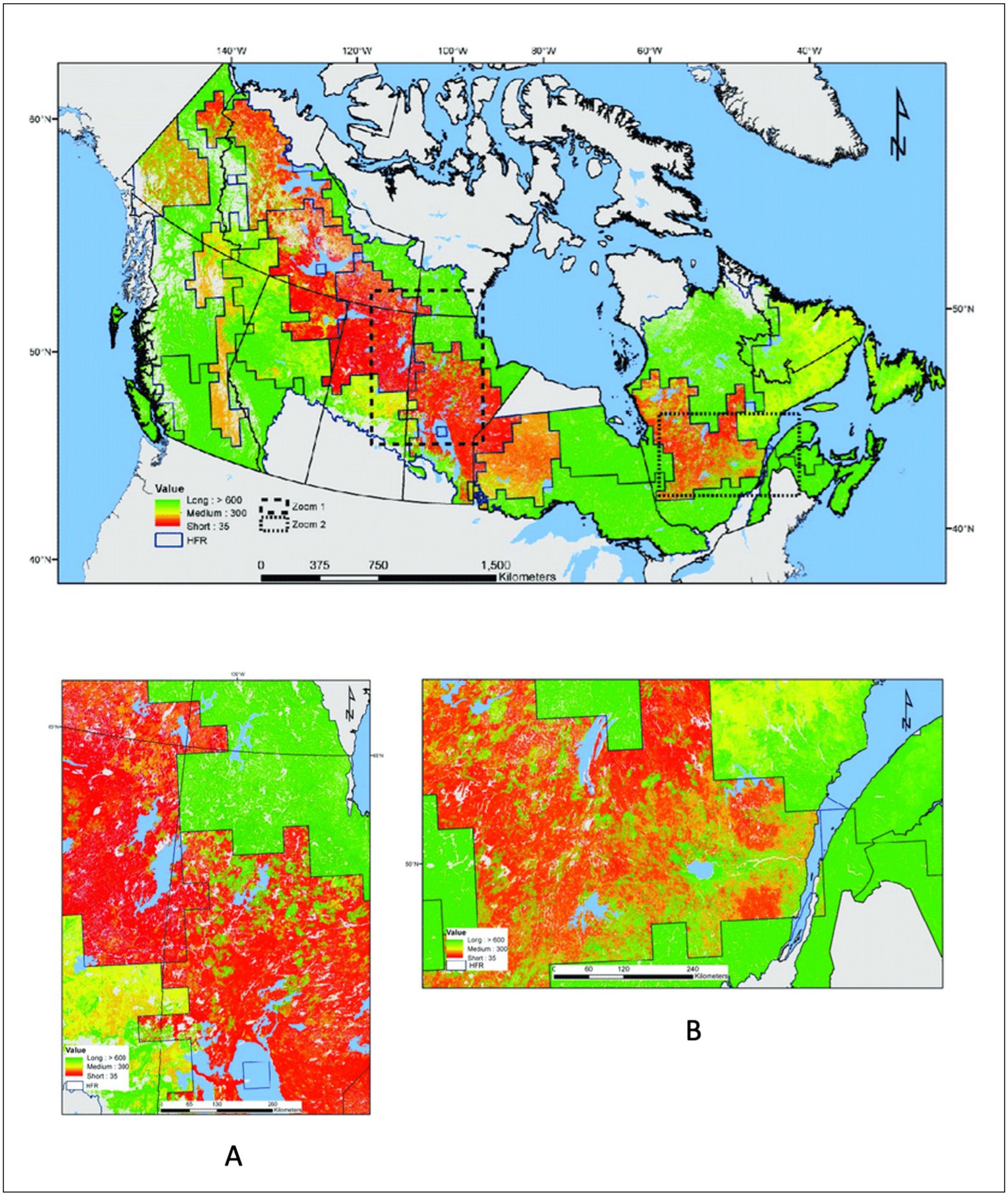


Figure 40. Map of local (pixel-level) fire risk within Homogeneous Fire Regime (HFR) zones. Maps A and B represent areas labelled "zoom 1" and "zoom 2" in the main map. The HFR-level risk was modulated at the pixel level by fire selectivity on the basis of each pixel's composition and age. Fire risk is presented in terms of the fire return interval (1/burn rate); long intervals indicate low fire risk. (Reproduced, with permission, from Bernier et al. 2016.)

assess the strength of several fire severity metrics for the northwestern boreal forest. Fifty-one sites affected by six wildfires between 14 000 and 700 000 ha in size were studied from the north shore of Great Slave Lake to Wood Buffalo National Park on the border between the Northwest Territories and Alberta. Field metrics used to classify the burned areas included the following (Table 22):

- Percent overstory mortality of mature trees.
- Composite Burn Index, a combination of mortality and combustion values across all canopy levels.
- Canopy Fire Severity Index, a measure of combustion in the tree canopy.
- Burn Severity Index, a measure of burn severity of the forest floor and soil.

Remotely sensed burn severity metrics were measured using satellite images that captured changes in the light reflectance of vegetation and soil after fire (Whitman et al. 2018b). This work showed that different fire severity levels could be detected at the sites of the 2014 fires. Approximately 40% of the burned area experienced severe, stand-replacing fires when considered in terms of overstory mortality. These severely burned (high mortality) patches tended to be large, aggregated, and of simple form. This metric is of importance to nonserotinous trees that depend on living trees for postfire recruitment (Whitman et al. 2018b).

However, when the metric for canopy combustion (Canopy Fire Severity Index) was used, 45% of the burned area was shown to have experienced moderate-severity burning, and most of the severely burned areas were small. This metric may better represent the effects of fire on serotinous species that can regenerate from trees that have been killed by fire, as long as their cones were not fully burned and the seeds remain viable (Whitman et al. 2018b).

The metric for forest floor and soil burn severity, the Burn Severity Index, indicated that most of the landscape had burned with moderate severity during the 2014 fires. This finding is important for the survival of seed banks in the soil and for the quality of seedbeds, the latter characterized by the presence of organic soils in peatlands and the availability of some exposed mineral soils in uplands (Whitman et al. 2018b). About 15% of the area within the 2014 fire perimeter suffered no damage, and these unchanged areas could provide seed sources for the nonserotinuous white spruce and habitat for species requiring mature forest. Thus, when modeling and mapping fire severity, metrics that are meaningful to the local fire regime and postfire seedling recruitment should be chosen (Whitman et al. 2018b).

Characteristic average fire severity levels were found for sites with differing topoedaphic attributes, vegetation types, and stand structure, although substantial variation in severity was observed within each vegetation type, perhaps because of prefire site variations or weather, or both (Figure 41) (Whitman et al. 2018b). Although upland jack pine stands have a smaller surface fuel load than black spruce stands (Thompson et al. 2017) and therefore experience less combustion in absolute terms (Walker et al. 2018), the proportion of the surface burned was highest for jack pine stands when burn severity of the forest floor was included. However, the only statistically significant difference was between upland jack pine stands and upland mixed-wood stands (Whitman et al. 2018b). Because jack pine stands are the driest stand type, a greater proportion of the forest floor surface tends to burn (Walker et al. 2018). Black spruce, by contrast, had the highest canopy combustion, although differences between stand types were not significant. Open wetland sites had the lowest fire severity, likely due to increased soil moisture and a lower fuel load, whereas treed wetland sites were not significantly different from upland sites. Mixed-wood upland stands had the greatest variability in burn severity because of differing proportions of coniferous and deciduous trees across sites (Whitman et al. 2018b).

Table 22. Fire severity field metrics. (Data source Whitman et al. [2018a].)

Fire severity metric	Measures	Forest strata affected	Effect on soil
Composite burn index	Combustion + mortality	All	Surface combustion
Percent overstory mortality	Mortality	Mature trees	None
Canopy fire severity index	Combustion	Mature trees	None
Burn severity index	Combustion	Forest floor	Surface combustion

Generally, the risk of a crown fire increases with lower height to the base of the live crown, because surface fires can more easily jump to the tree's crown (Agee and Skinner 2005). In the study by Whitman et al. (2018b), however, trees with a lower estimated prefire height to the base of the crown were more often found in sparsely treed wetlands or mixed-wood stands with a suppressed conifer understory, and fire severity was therefore positively correlated with height to the crown. A greater tree density allows for the easier spread of crown fires from tree to tree (Agee and Skinner 2005), and fire severity therefore increased with increased stem density (Whitman et al. 2018b). Large mature trees tend to be more fire resistant because they have the highest crowns and the thickest bark (Agee and Skinner 2005). The findings of Whitman et al. (2018b) echoed that conclusion, in that the basal area of mature trees was negatively correlated with burn severity because of the more open structure and higher crown-base heights of older stands with large trees, as well as their resistance to mortality from fire.

Unburned and low-severity burned areas were more commonly found in jack pine and mixed-wood stands, as well as in stands with a mature or open stand structure with large-diameter trees. Harvesting that is intended to mimic natural disturbance through variable retention must therefore be planned to incorporate consideration of why certain sites survive fire and others do not (Whitman et al. 2018b). Although weather conditions are ordinarily a strong driver of fire severity, the extreme fire weather of 2014 meant that variations in weather between different fires were not as great as usual, and weather did not appear to significantly contribute to the differences in fire severity among sites (Whitman et al. 2018b).

Although the NBAC project of the CFS has led to improved maps of area burned, they do not show the burn severity for different areas. The differenced normalized burn ratio has been used to create Canada-wide fire severity maps (Guindon et al. 2021). Work is underway to incorporate these newly derived maps of fire severity into other national wildfire and carbon products (unpublished data provided by E. Whitman).

The effects of forest fuel management on mitigation of wildfire risk, such as thinning treatments to reduce tree density and fuel load, have been studied extensively in many different forest types. One such study in the boreal forest of Alaska evaluated the impacts of several landscape-level fuel reduction treatments on fire behavior in mature spruce-dominated forests. The study assessed whether prefire thinning or shear-blading (removal of all standing material at ground level with a blade attached to a large piece of machinery) affected fuel consumption, burn severity, or fire behavior during a prescribed burn (Butler et al. 2013). This work showed that thinning the forest stands significantly reduced the amount of tree crown consumed during the fire, the fire intensity, and the rate of fire spread among trees relative to the other test conditions, particularly the untreated control area (Butler et al. 2013). This treatment response was largely attributed to the more open canopy and higher live-needle moisture content measured in thinned areas, which likely resulted from the reduced competition among trees in these more widely spaced forest stands (Butler et al. 2013). How or whether these outcomes would change in future decades as a result of changing fire regimes and fire behavior from climate change is currently unknown.

5.6 Postfire Forest Recovery

It is generally believed that regenerating forests in the northern boreal forest return to a composition and density similar to what was present before the fire (Ilisson and Chen 2009). This belief may be reasonable, given that the short fire return interval does not allow enough time for the successional stages necessary for development of true "old growth" stands (Johnstone et al. 2010).

Whitman et al. (2018a) studied postfire seedling recruitment and understory vegetation communities of Northwest Territories sites that were burned in the 2014 fires. All 51 sites were studied 1 year after the fires, and 30 of the sites were studied for a period of 3 years after the fires. Although a longer-term study would provide additional insight into postfire recruitment, it is believed that the first few years of recruitment determine the future composition of stands in the boreal forest (Johnstone et al. 2004, Johnstone et al. 2020). At the Northwest Territories sites, the most significant drivers of postfire understory regeneration and seedling recruitment were climate, site moisture, and prefire forest composition, rather than variables related to the fire itself (Whitman et al. 2018a).

Fire severity and fire history had strong secondary influences, however, especially on seedling

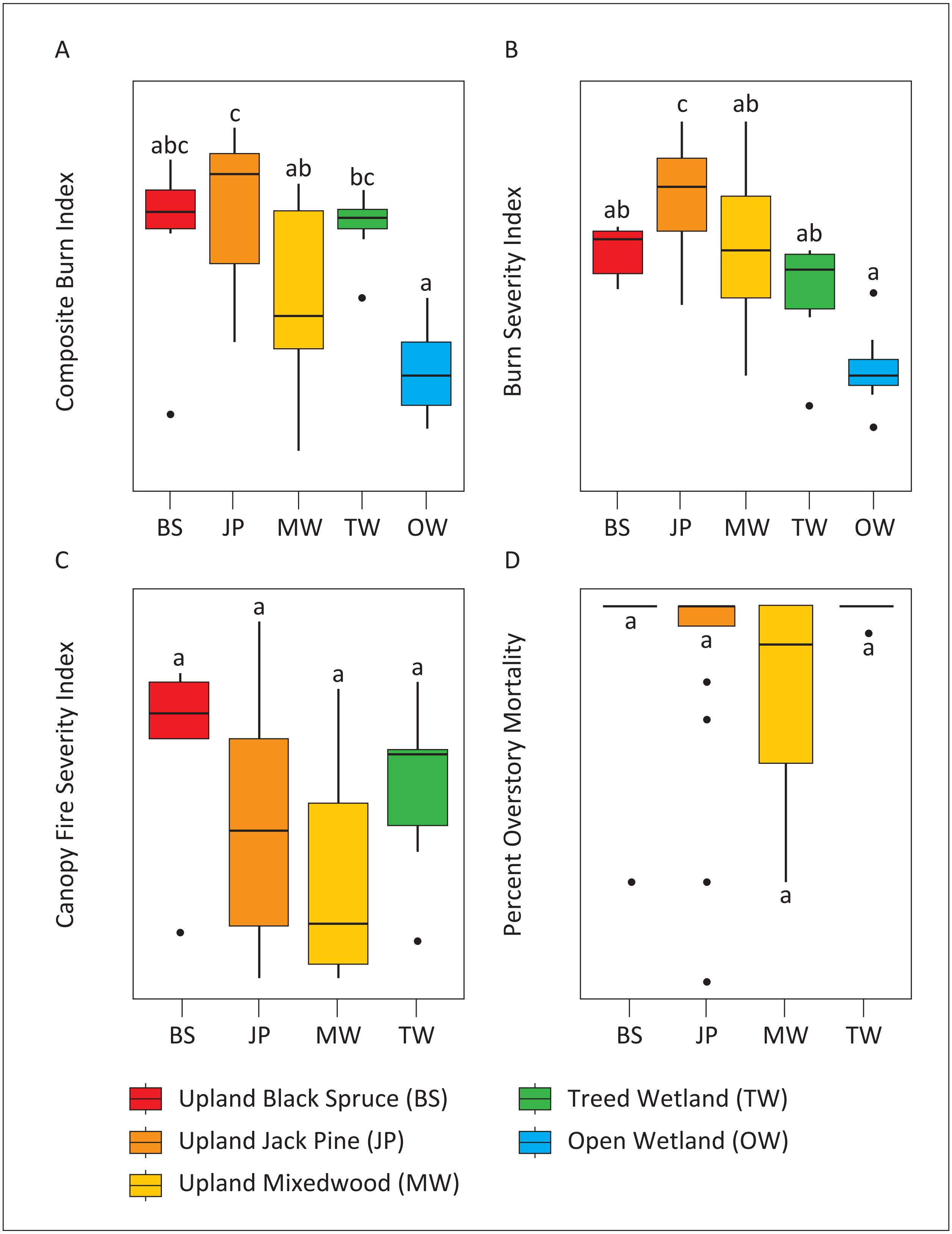


Figure 41. Burn severity metrics for 2014 wildfires in various vegetation communities of the Northwest Territories. Differing letters above or below each box plot indicate statistically significant differences ($P \leq 0.05$); individual solid circles indicate outliers. (Reproduced, with permission, from Whitman et al. 2018b.)

recruitment, and changes in species dominance were detected. The proportion of black spruce declined in upland sites with a fire return interval of less than 100 years, and both white and black spruce declined where high-severity crown fires occurred (Figure 42). White spruce remained stable at moister sites, and black spruce dominance remained stable in treed wetlands, although seedling density was generally low in wetlands after a fire. Such declines in black spruce after severe fire have also been documented in Alaska, Yukon, and the boreal forest of eastern Canada (Whitman et al. 2018a). Although black spruce is adapted to stand-replacing fires in the boreal forest, it is possible that increasing fire frequency and severity are challenging the ability of this species to recover

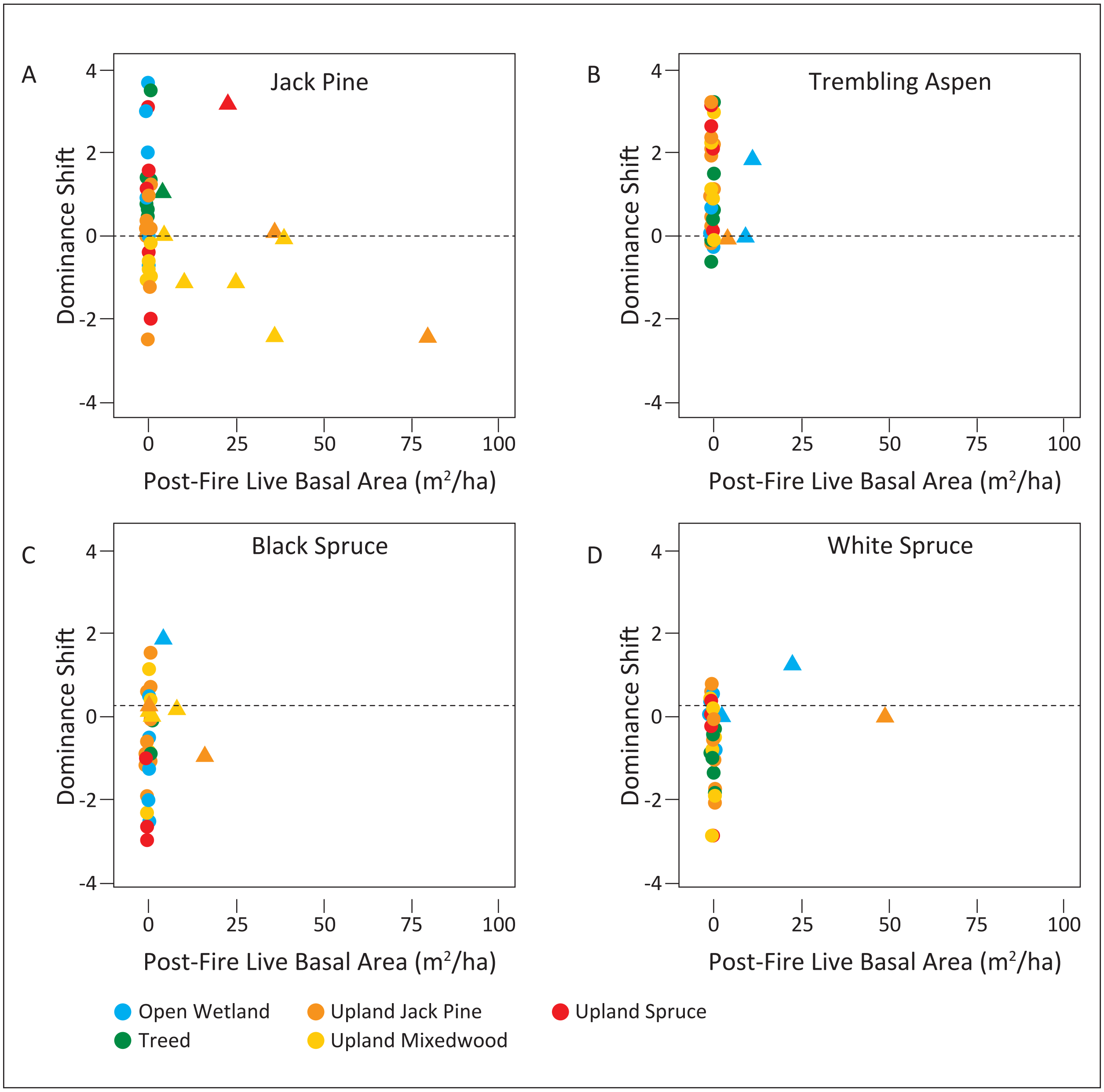


Figure 42. Changes in postfire dominance of (A) jack pine, (B) trembling aspen, (C) black spruce, and (D) white spruce, plotted against postfire live basal area of each species. Circles indicate sites where the species either experienced complete death due to the fire or was not present before the fire. Triangles indicate sites with live residual basal area after the fire. Points are offset ("jittered") to reduce overlap. Dashed horizontal lines indicate no change in species dominance after a fire (relative to what was present before the fire); points above this line represent sites where postfire dominance increased, and points below this line represent sites where the species dominance declined after the fire. (Reproduced, with permission, from Whitman et al. 2018a.)

fully. Jack pine increased in stand dominance after fire, except for slight decreases due to suckering trembling aspen where the latter had already been present before the fire. Aspen increased at all sites (Figure 42) (Whitman et al. 2018a).

Climate change is likely driving long-term changes to tree composition in parts of the western boreal forest, and increased frequency and severity of fire would accelerate those changes, possibly overwhelming topoedaphic site attributes and historical forest compositions that contribute to forest resilience (de Groot et al. 2013; Whitman et al. 2018a). A long-lasting shift to aspen dominance would provide negative feedback to both fire likelihood (because of the low combustibility of aspen) and climate change (due to increased albedo). However, such a shift may be limited by the sensitivity of aspen to drought. Given that black spruce is also susceptible to drought, increased drought induced by climate change would benefit the more drought-tolerant jack pine (Whitman et al. 2018a). However, increased fire severity may reduce the ability of jack pine to regenerate after fire if serotinous cones are burned during intense wildfires (de Groot et al. 2013). Indeed, Pinno et al. (2013) found that postfire densities of jack pine in a northeastern Alberta forest were substantially lower than prefire densities, and these researchers suggested that large fires may shift forest structure to a more open canopy woodland. In particular, young jack pine stands had difficulty with recruitment, likely because younger trees produced fewer serotinous cones (Pinno et al. 2013). Johnstone and Chapin (2006) also found that while stands in the boreal forest of northern British Columbia and southern Yukon generally returned to prefire composition, younger stands (<25 years) contained a greater proportion of deciduous species than before the fire.

Matasci et al. (2018) produced forest structure maps for Canada from 1986 to 2016, showing changes in stand height, biomass, percent cover, basal area, and volume of forests in the years after a fire. These values were derived from LiDAR images, which were correlated with 30-m spatial resolution Landast images and then averaged by ecoregion. This work showed that after stand-replacing fires, forests in the Taiga Plains ecoregion reached an average 95th percentile height of 10 m after 25 years, which is the average for forests across the landscape, and continued to grow to 13 m by 31 years (the exent of the study period). This study also showed that aboveground biomass reached the landscape average of 35 t·ha^{-1} after 20 years and grew to 65 t·ha^{-1} by year 31. Percent canopy cover reached the landscape average of 37% by year 18, grew to 75% by 30 years after the fire, and then began to decline (Matasci et al. 2018). Taiga Shield forests returned to the landscape average for 95th percentile tree height of 7 m after about 11 years and then remained steady beyond 30 years. Canopy cover in this ecoregion reached 30% and aboveground biomass reached 20 t·ha^{-1} after 31 years, below the landscape averages of 38% and 27 t·ha^{-1}, respectively. Taiga Cordillera forests returned to the landscape average for 95th percentile tree height of 9 m after about 20 years and then remained steady. Canopy cover in this ecoregion reached the landscape average of 32% after 20 years, grew to 41% by year 30, and then began to decline. Aboveground biomass reached the landscape average of 28 t·ha^{-1} after 25 years and then remained steady (Figure 43) (Matasci et al. 2018).

CFS and University of Alberta researchers have developed partnerships to study forest recovery after the 2014 Northwest Territories fires. They note that most of the burning occurred in areas that had not burned since 1980, and biomass accumulation in the intervening years made these areas more susceptible to large fires. Areas that had experienced fire in the period 1980–2013 and then reburned in 2014 were more likely to have had all remaining vegetation on the site burned, which produced desert-like conditions, with large reductions in conifer seedlings alongside modest reductions in aspen and other broadleaf seedlings, resulting in a likely future forest of greatly reduced stem density (Whitman et al. 2019). At the same time, the fires left some forested areas, even those that had not burned for a long time, untouched or only lightly burned, creating fire refugia that could act as seed sources or critical habitat for fire-sensitive species (Whitman et al. 2018b). Recently released burn severity trends for Canada, calibrated in part with data from the Northwest Territories (Guindon et al. 2021), showed no meaningful trend in fire severity from 1985 to 2015, with small decreases in median fire severity observed in the Taiga Plains and Taiga Shield ecoregions across all forest types, and a decrease in fire severity in conifer forests in the Taiga Plains ecoregion. Nevertheless, projections of future fire regimes indicate significant changes in future decades, particularly in terms of fire return interval (de Groot et al. 2013). If the mean fire return interval

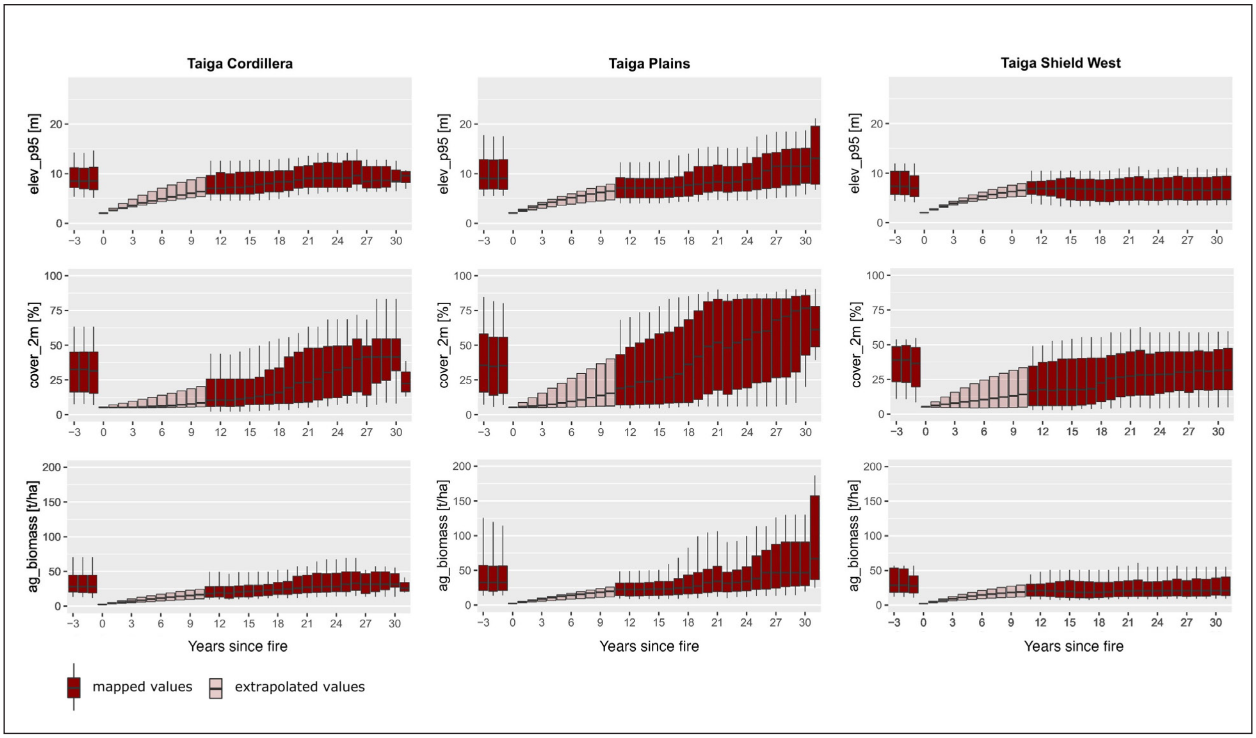


Figure 43. Ninety-fifth percentile of stand height (elev_p95), canopy cover of trees more than 2 m tall (cover_2m), and aboveground biomass (ag_biomass) before and after stand-replacing wildfires, by ecoregion, 1984–2016. (Adapted, with permission, from Matasci et al. 2018.)

becomes shorter, future fire regimes will favor species that can quickly recolonize a site after fire through mechanisms such as root sprouting (de Groot et al. 2013) over species that take longer to regenerate, such as white spruce. Increases to fire intensity are also predicted to lead to changes in postfire forest composition: intense fires reduce the amount of viable seed that is available after a fire, because a greater proportion of seeds stored in tree canopies and forest floor seedbanks is burned during such fires (de Groot et al. 2013).

5.7 Interactions of Wildfire with Other Disturbances

Although the HFR zones provide a more accurate picture of fire disturbance than ecoregion divisions, the omission of other disturbances means that these zones do not fully reflect observed changes to forest composition and structure (Figure 3, Figure 4) (Burton and Boulanger 2018). Burton and Boulanger (2018) therefore studied fire and insect outbreaks in British Columbia to determine whether there was a consistent pattern between these types of disturbances.They found no significant correlation between fire history and insect outbreaks and concluded that interactions between the two disturbance regimes create emergent properties on the landscape (Burton and Boulanger 2018). Whether this is also the case in the Northwest Territories is unknown.

Fleming et al. (2002) studied the interaction between outbreaks of spruce budworm (Choristoneura fumiferana) and forest fires in Ontario and found that fires occurred disproportionately often in the period 3–9 years after a spruce budworm outbreak. This pattern was attributed to the buildup of dead branches and tree tops on the ground, as well as the increase in dead lower branches, which together create a fuel ladder allowing small surface fires to reach the canopy and spread quickly through the stand. Once the ground fuel begins to decompose and the herbaceous understory grows denser, the fire risk declines (Fleming et al. 2002). Western Ontario had a longer and greater range of time lag between spruce budworm outbreaks and fire than was the case in Eastern Ontario because of a drier summer climate, which is more conducive to large fires (Fleming et al. 2002). Spruce budworm is common in the Northwest Territories, but interactions between outbreaks and fires have not been studied there.

Boucher et al. (2018) created Homogeneous Disturbance Regime zones that map zones of timber volume vulnerable to overlapping disturbances of fire, spruce budworm, mountain pine beetle (*Dendroctonus ponderosae*), drought, and harvest, using both exposure and sensitivity to each disturbance. The simple overlap of disturbances was modeled, without accounting for interactions between them, but care was taken to avoid double counting timber that might have been affected by more than one disturbance (Boucher et al. 2018). The area mapped did not include the Taiga ecoregions and thus covered only two small areas of the Northwest Territories: a strip between Hay River and Fort Resolution on the south bank of Great Slave Lake to the Alberta border (Boreal Plains ecoregion) and a small part of the southwest corner of the Northwest Territories (Boreal Cordillera ecoregion) (where ecozones are defined by the Canadian Soil Information Service [2013]). For the period 1981–2010, both areas were estimated to have 10%–20% of wood volume at risk from cumulative impacts of fire, spruce budworm, mountain pine beetle, and drought, and less than 10% of the volume not at risk was harvested (Boucher et al. 2018). However, climate projections based on Representative Concentration Pathway (RCP) 8.5 show that for the period 2011–2040, 20%–40% of wood volume in these areas could be at risk from at least one of the four disturbances, a proportion that could reach 90%–100% by 2071 (Boucher et al. 2018).

Fires also contribute to permafrost degradation (Figure 3, Figure 4). A study conducted in the Taiga Plains ecoregion found that active-layer depths increased by 60% for 5 years after fire, likely because of increased albedo from the conversion of light-colored lichens to black char, and returned to prefire depths after about 20 years (Gibson et al. 2018). Talik coverage increased from an average of 20% on unburned sites to 70%–100% on burned sites 10–20 years after the fire and returned to prefire levels after about 50 years (Gibson et al. 2018). Soil temperature at 40 cm depth increased from 2°C to 5°C at unburned sites to 9°C to 11 °C at sites that burned 10–20 years previously and returned to prefire temperatures after about 40 years. Recovery of soil thermal regimes coincided with vegetation recovery, highlighting the role that vegetation plays in shading and albedo (Gibson et al. 2018).

Sites that burned within the previous 30 years formed collapse scars at triple the rate of unburned sites. As a result, wildfires can be considered to have caused about 23% of the loss of peat plateaus over the past 30 years (Gibson et al. 2018).

Key Points

- Fire regimes are influenced by many factors that create complex interactions. At the scale of level II and level III ecoregions, the five most important contributors to area burned are human footprint, extreme summer temperatures, wind, topography, and ratio of conifer (most flammable) to other fuels.
- The Taiga Plains and Taiga Shield ecoregions have similar fire weather severity (measured as Seasonal Severity Rating or SSR) and a similar ratio of human-caused to lightning-caused ignitions, although the Taiga Plains has a significantly lower ratio of conifer to other fuels, a higher area of fire suppression, and lower cover of nonfuel materials. The Boreal and Taiga Cordillera ecoregions had the least severe values for SSR, the lowest ratio of conifer to other fuels, the lowest cover of nonfuel materials, and the fewest human-caused ignitions; however, these ecoregions also had no fire suppression.
- Data from the National Burned Area Composite database show that from 1986 to 2018, the average percent annual area burned within the Northwest Territories was 0.51% for the Taiga Plains ecoregion, 0.57% for the Taiga Shield ecoregion, and 0.33% for the Boreal and Taiga Cordillera ecoregions.
- The similarity of fire regimes in the Taiga Plains and Taiga Shield ecoregions and the less severe fire regime in the Cordillera ecoregions are consistent with the respective fire potential characteristics of these ecoregions.
- Homogeneous Fire Regime (HFR) zones were created for Canada on the basis of fire history and vegetation cover. Seven of these zones are found in the Northwest Territories. The Pacific HFR zone (approximately equivalent to the Taiga Cordillera ecoregion) had the lowest percent annual area burned (0.04%). The Lake Athabasca zone (located along the Saskatchewan border and toward the east arm of Great Slave Lake) had the highest percent annual area burned (1.48%).
- The fuel loads in upland and wetland sites in Wood Buffalo National Park, along the border between the Northwest Territories and Alberta, did not differ significantly, except for the fuel load in sheltered and moist white spruce stands, which rarely burn. However, wetlands must experience drought conditions before they can burn.
- About 40% of the burned area in six of the large 2014 fires in the Great Slave Lake area experienced stand-replacing fires (causing severe canopy mortality). However, overstory combustion was moderate over 45% of the area, with severe combustion limited to small patches, which allowed serotinous cones of jack pine and black spruce to reseed most of the area. Forest floor combustion was mostly moderate, which suggested adequate seedbeds for seedling recruitment. About 15% of the area within the fire perimeter suffered no damage at all; these areas could provide seed sources for white spruce and habitat for fire-intolerant species.
- Postfire recruitment is mostly influenced by the prefire stand composition. However, the 2014 fires showed that particularly severe fires can decrease the density of spruce and increase the proportions of jack pine and aspen. Repeated fires lead to the regeneration of more open, broadleaf-dominated forests.

Knowledge Gaps

- The fire regime of wetlands is not well understood because of the interaction between rapid fuel accumulation after fire and the requirement for drought conditions (for flammability).
- Knowledge of interactions between disturbances such as wildfire and insect outbreaks in the Northwest Territories is limited.

CHAPTER 6. FOREST INSECTS AND DISEASES

6.1 Forest Insects

The most serious forest insect pest in the Northwest Territories is the spruce budworm (Figure 44, Figure 45). In this region, the preferred host of the spruce budworm is white spruce. Black spruce is nutritionally acceptable, but a 2-week delay in bud break in this species kills off larvae when they are emerging in the spring and looking to feed (Pureswaran et al. 2015). The current range of white spruce in the Northwest Territories extends north of the spruce budworm's range, making this region susceptible to range expansion of the insect with climate warming (Régnière et al. 2012). In general, host species that currently inhabit regions beyond the climactic tolerance of the insect species that feed on them may consist of genotypes that lack defense mechanisms and are therefore more vulnerable to outbreaks if the insect range expands (Price et al. 2013). In addition, changing tree phenologies may increase compatibility between spruce budworm and black spruce, further increasing the extent of future outbreaks (Pureswaran et al. 2015). Temperature is the single most important and well understood factor influencing survival of spruce budworm. South of its range, warmer winter temperatures lead to exhaustion of energy reserves for the overwintering larvae in diapause, and the larvae starve. North of its range, autumn temperatures are too low for the larvae to grow and reach the cold-hardy stage needed for winter survival in diapause (Régnière et al. 2012).

Over the past few hundred years, spruce budworm has exhibited a 30- to 40-year cycle of outbreak followed by population crash. During outbreaks, populations may be so high that up to a few hundred larvae can be observed on each branch; in contrast, during crash years, it may be difficult to find a single larva within several hundred branches (Gray 2008). Outbreaks last anywhere from 1 to 20 years (Gray 2008), and they can cover large areas: in Ontario from 1941 to 1996, defoliation by this insect covered an area 185 times larger than that affected by wildfire (Fleming et al. 2002). However, because the infestation shifts locality from year to year and because it takes multiple years of moderate to severe infestations to kill trees, most trees survive defoliation. For example, between 1941 and 1996 in Ontario, only about 9.1% of trees defoliated by spruce budworm were killed (Fleming et al. 2002).

The GNWT (specifically, the Forest Management Division of the Department of Environment and Natural Resources) and the CFS jointly lead annual forest health surveys focused on insect and disease outbreaks. Annual aerial surveys are conducted for 8 days in July, coinciding with peak spruce budworm outbreak; in addition, aerial surveys focused on aspen defoliation are carried out in June, and surveys for detecting mountain pine beetle occur in September. Ground surveys conducted along highways complement the aerial surveys by detecting emerging trends not visible from the air, especially early defoliation, and by verifying signs of disturbance seen from the air (Figure 46, Figure 47) (Olesinski and Brett 2017).

Although spruce budworm has not yet caused significant damage to Northwest Territories forests, this insect has historically been found throughout the Dehcho and South Slave administrative regions (Table 23). In the Dehcho administrative region, a recent outbreak doubled in size in 2017 relative to 2016, causing defoliation throughout 165 585 ha (Olesinski and Brett 2017). The expansion was most significant along the Liard River between Fort Liard and Nahanni Butte (Taiga Plains Mid-Boreal ecoregion) and along the Mackenzie River between North Nahanni River and Wrigley (Boreal Cordillera High Boreal ecoregion). In the South Slave administrative region, spruce budworm increased in the Cameron Upland (Taiga Plains High Boreal ecoregion) but declined elsewhere (Figure 47) (Olesinski and Brett 2017). Mortality caused by this insect has been detected, but the data have not been compiled. In addition to direct detection, the extent of mortality can be modeled by correlating severity and duration of defoliation with mortality for different regions, but this has not yet been done. Generally, moderate to severe defoliation over a period of 5–7 years will kill affected trees (Olesinski and Brett 2018).

The first recorded outbreak of spruce budworm in the Mackenzie Delta in the Inuvik administrative region occurred in 2015, affecting 100 000 ha, but the population collapsed in 2016, perhaps because of rain and freezing spring temperatures. It was the most northerly known outbreak in Canada

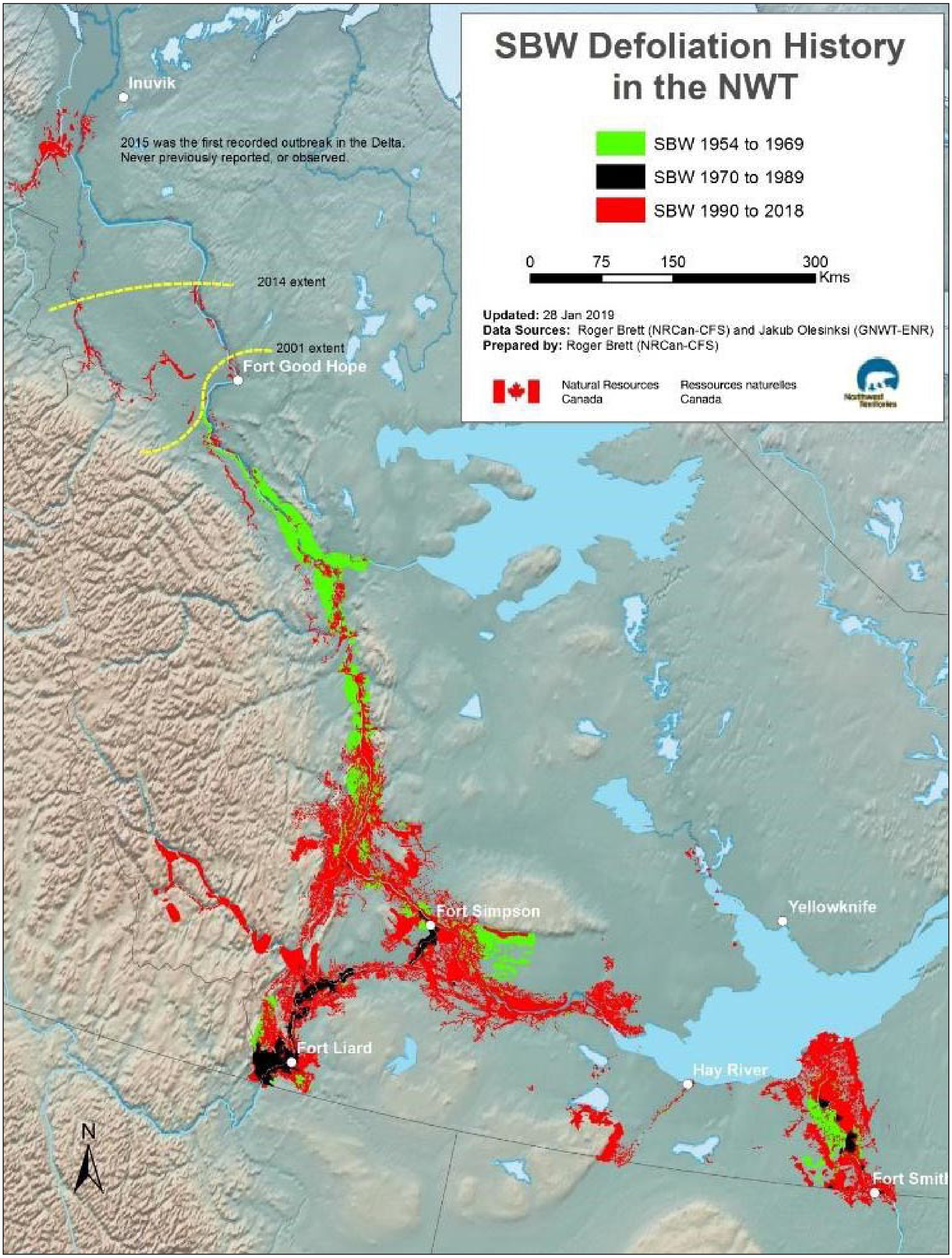


Figure 44. History of spruce budworm (SBW) defoliation in the Northwest Territories (NWT) during the periods 1954–1969, 1970–1989, and 1990–2018. (Data supplied by R. Brett.)

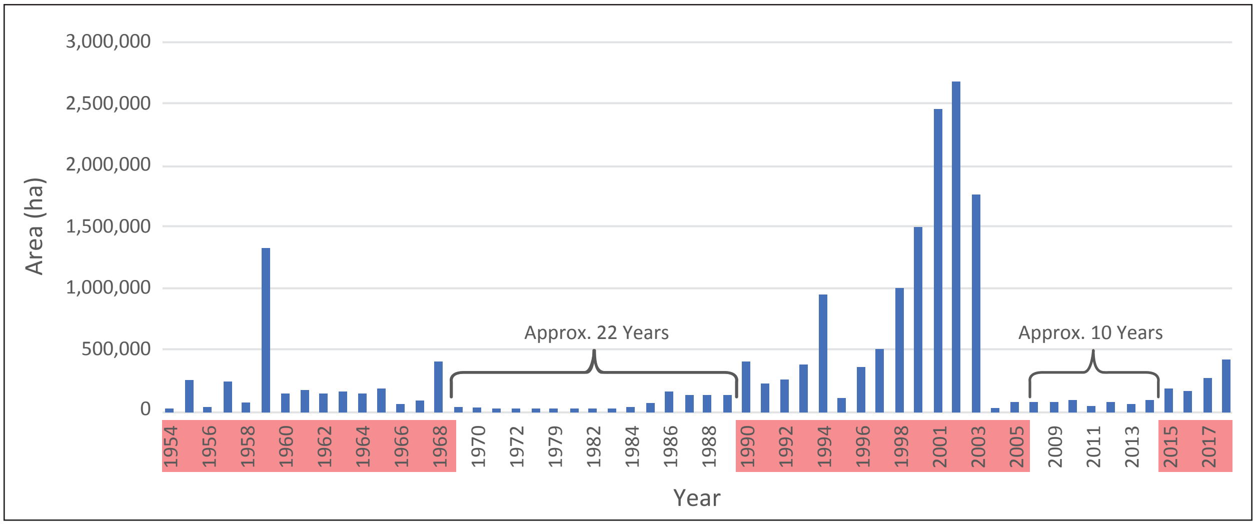


Figure 45. Area affected by spruce budworm defoliation, 1954–2018. Red shading along the horizontal axis indicates outbreak years, whereas absence of shading indicates non-outbreak years. The reduction in periods between outbreaks points to an increase in outbreak frequency. (Data supplied by R. Brett.)

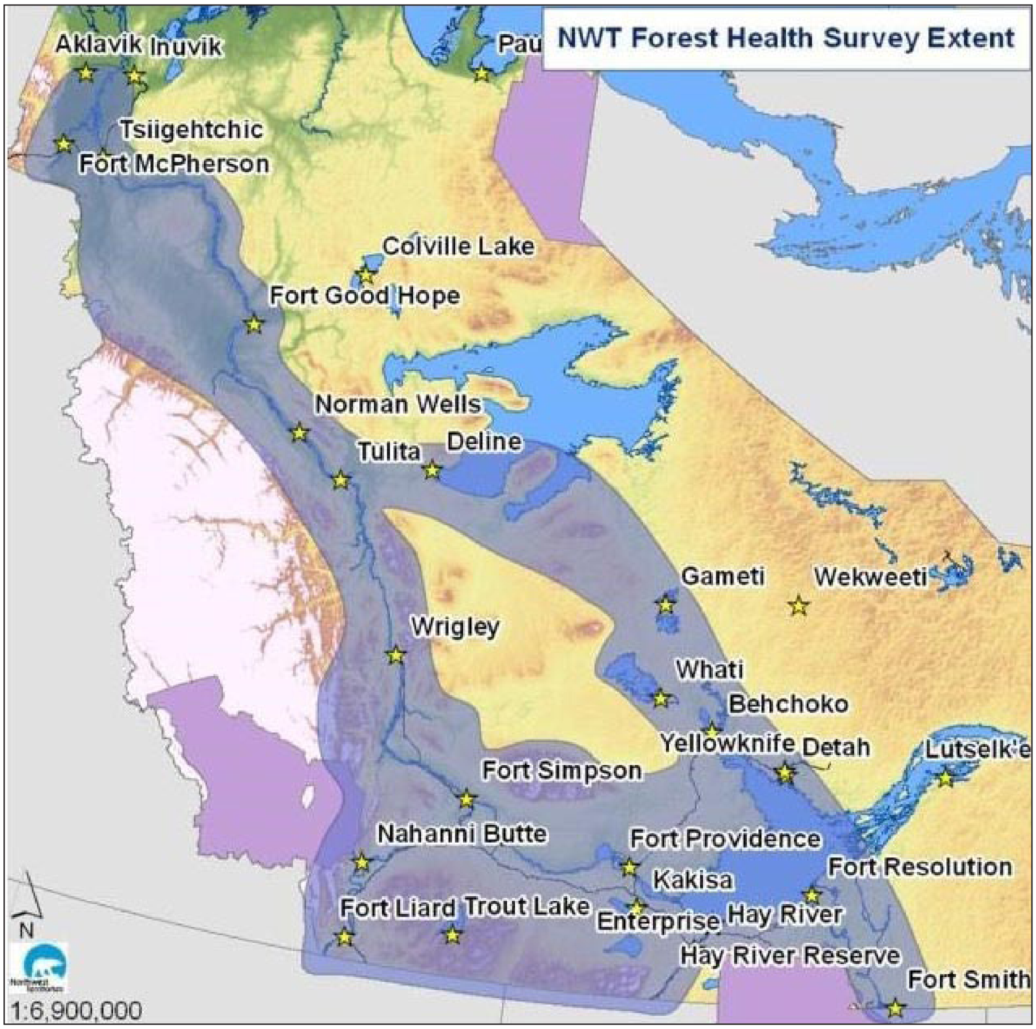


Figure 46. Extent of the 2018 Northwest Territories forest health survey. (Reproduced, with permission, from Olesinski and Brett [2018].)

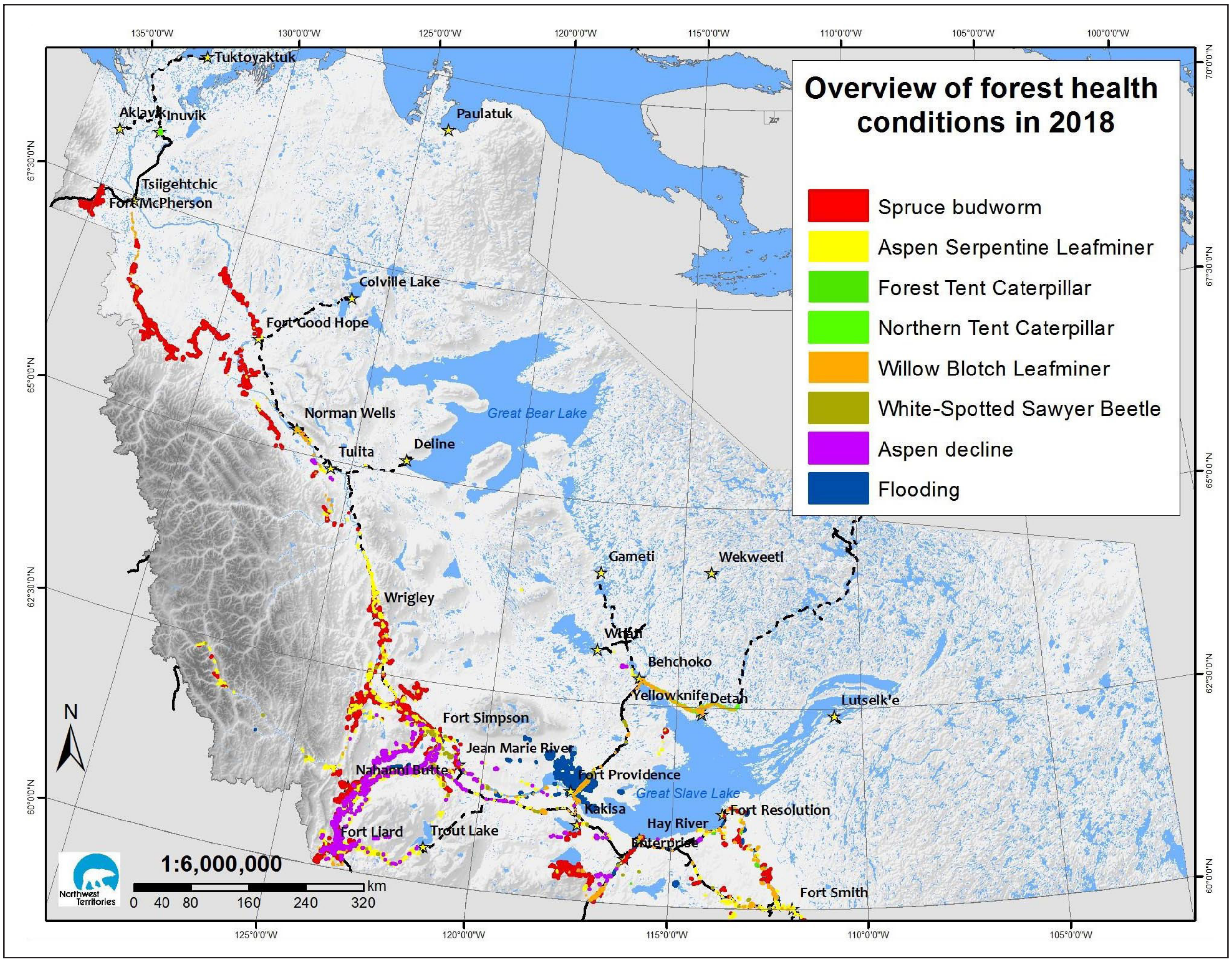


Figure 47. **Biotic and abiotic damage detected in the 2018 Northwest Territories forest health survey.** (Reproduced, with permission, from Olesinski and Brett [2018].)

Table 23. **Area affected by biotic and abiotic disturbances across administrative regions of the Northwest Territories, as detected in the 2018 forest health survey.** (Adapted, with permission, from Olesinski and Brett [2018].)

	Administrative region and area affected (ha)					
Type of disturbance	Dehcho	Inuvik	North Slave	Sahtú	South Slave	Grand total
Biotic disturbances						
Aspen serpentine leafminer	288 765	n/a	1304	6499	39 419	335 987
Scarab leaf beetle	n/a[a]	n/a	n/a	n/a	1397	1397
Eastern larch beetle	644	n/a	409	n/a	699	1752
Forest tent caterpillar	n/a	n/a	n/a	n/a	3141	3141
Gray willow leaf beetle	1758	n/a	n/a	n/a	n/a	1758
Northern tent caterpillar	n/a	1206	10 037	206	n/a	11 449
Spruce budworm	245 765	47 998	1304	59 649	63 143	417 859
Willow blotch leafminer	8202	2544	27 127	4222	52 535	94 630
Rusty tussock moth	n/a	1330	n/a	n/a	n/a	1330
Birch leafminer	n/a	n/a	237	n/a	n/a	237
Western balsam bark beetle	923	n/a	n/a	n/a	n/a	923

Table 23. Concluded.

	Administrative region and area affected (ha)					
Type of disturbance	Dehcho	Inuvik	North Slave	Sahtú	South Slave	Grand total
Biotic disturbances						
Yellow-headed spruce sawfly	n/a	n/a	8	n/a	204	212
White-spotted sawyer beetle	3935	n/a	286	n/a	826	5047
Spruce needle rust	n/a	39	n/a	n/a	n/a	39
Subtotal of biotic disturbances	549 992	53 117	40 712	70 576	161 364	798 887
Abiotic disturbances						
Flooding	2648	n/a	n/a	78	10 253	12 979
Drought	n/a	n/a	n/a	n/a	5	5
Red belt	1077	n/a	n/a	n/a	n/a	1077
Slumping	500	n/a	n/a	176	n/a	676
Aspen decline	90 475	n/a	1224	922	6403	99 024
Blowdown	485	n/a	447	n/a	48	980
Subtotal of abiotic disturbances	95 185	n/a	1671	1176	16 709	114 741
Grand total	645 177	53 117	42 383	71 752	178 073	990 502

[a] *n/a = not applicable.*

and perhaps North America (Olesinski and Brett 2016). A total of 25 753 ha was affected by spruce budworm defoliation in Inuvik in 2017 in the areas of Fort McPherson and the southern Arctic Red River (Figure 44) (Olesinski and Brett 2017). The presence of spruce budworm this far north indicates a changing climate that is becoming more favorable to the spread of this insect, and the link between insect phenology and climate requires further study (Olesinski and Brett 2016). The Mackenzie Delta is a unique ecosystem within the boreal forest because it has a long fire return interval and contains old-growth forests that are hundreds of years old. The northern expansion of large insect outbreaks into this region is therefore particularly devastating (Olesinski and Brett 2018). Pheromone trapping to detect the presence of spruce budworm moths and to monitor population size was conducted in Inuvik between 2007 and 2012, and trapping was reestablished there in 2016 because of the outbreak (Olesinski and Brett 2017, Olesinski and Brett 2018).

Records from 1954 to 2017 suggest that the number of years between outbreak episodes is becoming shorter (such that outbreaks are becoming more frequent) (Figure 45). As of the 2018 summer survey, the spruce budworm outbreak is expanding in all regions of the Northwest Territories (Olesinski and Brett 2018).

Forest tent caterpillar (*Malacosoma disstria*) is a serious defoliator of aspen. An outbreak of this insect in the Northwest Territories was first recorded along the Liard River in 1995, with 32 459 ha of defoliation. The outbreak increased to 224 830 ha in 1996 (Brandt 1997). The next outbreak began in the South Slave administrative region along the Slave and Hay Rivers in 2015, defoliating more than 100 000 ha of aspen. The affected area increased by 24% in 2016 but declined by 58% in 2017, and this decline is expected to continue. Aerial surveys of aspen defoliation in the South Slave administrative region took place in June 2016 and 2017 (Figure 47) (Olesinski and Brett 2017). The northern tent caterpillar (*Malacosoma californicum pluviale*) was documented in 2018 as far north as Inuvik (Taiga Plains High Subarctic ecoregion) (Table 23), whereas the previously recorded limit of its range had been Norman Wells (Taiga Plains Low Subarctic ecoregion) (unpublished data provided by R. Brett).

Aspen serpentine leafminer (*Phyllocnistis populiella*) is ordinarily a secondary invader causing minor growth loss and no permanent damage. However, an outbreak in the Northwest Territories has been occurring for the past 20 years throughout the entire aspen range, including damage to 250 000 ha in both 2016 and 2017. The duration and severity of the outbreak is likely causing more significant, although as-yet-unknown, damage (Figure 47) (Olesinski and Brett 2017).

Willow blotch leafminer (*Micrurapteryx salicifoliella*) damages willow leaves throughout the Northwest Territories. In 2017, damage from this insect was recorded farther north than ever previously recorded (Figure 47). The area affected by the leafminer is likely underestimated, given that most of the willow range was not surveyed (Olesinski and Brett 2017).

Gray willow leaf beetle (*Tricholochmaea decora*) is a native insect but reached outbreak levels in the Northwest Territories in 2015 around Fort Liard (Liard Upland within the Taiga Plains Mid-Boreal ecoregion) and the Cameron Upland (Taiga Plains High Boreal ecoregion) (Table 23) (Olesinski and Brett 2017).

Mountain pine beetle (*Dendroctonus ponderosae*) has caused widespread mortality of pines in British Columbia and Alberta over the past two decades, but to date has not caused significant tree mortality in the Northwest Territories. Since 2006, the mountain pine beetle has moved from primarily lodgepole pine forests in the interior of British Columbia and the Rocky Mountains to jack pine stands in the northern boreal forest of Alberta (Hall et al. 2016). In 2012, mountain pine beetle attacked bait trees in a single stand in the Northwest Territories just north of the Alberta–British Columbia border. The affected bait trees were cut down and burned. The use of bait trees serves to both detect initial outbreaks and treat initial infestations (Olesinski and Brett 2018). An aerial survey looking for signs of mountain pine beetle in the most sensitive areas of the Dehcho and South Slave administrative regions took place in September 2017; during that survey, one pine stand with suspected insect activity was identified approximately 40 km southeast of Trout Lake (Figure 48) (Olesinski and Brett 2017). However, when the site was ground-checked in early 2018, no evidence of mountain pine beetle was found in the stand (Olesinski and Brett 2018). Nevertheless, pheromone baiting stations have been established along Highway 1 in the Taiga Plains Mid-Boreal ecoregion, between the Northwest Territories–Alberta border and the community of Enterprise and between Jean Marie River and Kakisa; these baiting stations continued to operate through 2019 (Figure 48) (Olesinski and Brett 2017, Olesinski and Brett 2018). No evidence of mountain pine beetle has been recorded at any baiting station to date (Olesinski and Brett 2018).

Mountain pine beetle cannot survive under-bark winter temperatures below −40°C. Outbreaks also require a mean maximum August temperature of at least 18.3°C for peak emergence and flight, as well as more than 305 degree days above 5.5°C from 1 August to the end of the growing season and more than 833 degree days above 5.5°C between 1 August and 31 July to complete the life cycle (Carroll et al. 2006). Temperatures that are too warm can also disrupt the life cycle, including effects on the timing of cold-hardiness and synchronicity between emergence and bud break (Régnière et al. 2012). Climate change may already have led to the expansion of the mountain pine beetle to Alberta's northern boreal forest and will likely lead to more northward shifts in this insect's range (Carroll et al. 2006; Régnière et al. 2012). However, jack pine stands in the boreal forest are less susceptible to outbreaks of mountain pine beetle than lodgepole pine stands in British Columbia because smaller trees and less continuous pine stands lead to lower volume of pine in the boreal forests (Safranyik et al. 2010).

Forests in many areas of the Northwest Territories experienced drought conditions during the period 2012–2016 (Olesinski and Brett 2016) and are now in recovery (Olesinski and Brett 2017). In this situation, it is more difficult for trees to defend themselves against insects and diseases (Figure 3, Figure 4), which enables ordinarily secondary invaders of little concern to cause significant damage (Olesinski and Brett 2017). In total, more than 573 000 ha were affected by insects and diseases in the Northwest Territories in 2017, an 8.5% increase over 2016 (Olesinski and Brett 2017).

A project initiated by the CFS to investigate the health of trembling aspen forests, the Climate Impacts on Productivity and Health of Aspen (CIPHA), found a 35% aspen mortality rate between 2012 and 2016 at study sites on the Poplar River southwest of Fort Simpson (Taiga Plains Mid-Boreal ecoregion) (Figure 14). The MVI project of the CFS, using tree measurements and tree-ring analysis, found 42% aspen mortality and 11% biomass reduction between 2008 and 2016 in study plots near Fort Liard (Taiga Plains Mid-Boreal ecoregion). For both sites, the likely cause was a combination of severe drought in 2013 and 2014 and subsequent insect damage from forest tent caterpillars and leaf miners (Hogg 2017).

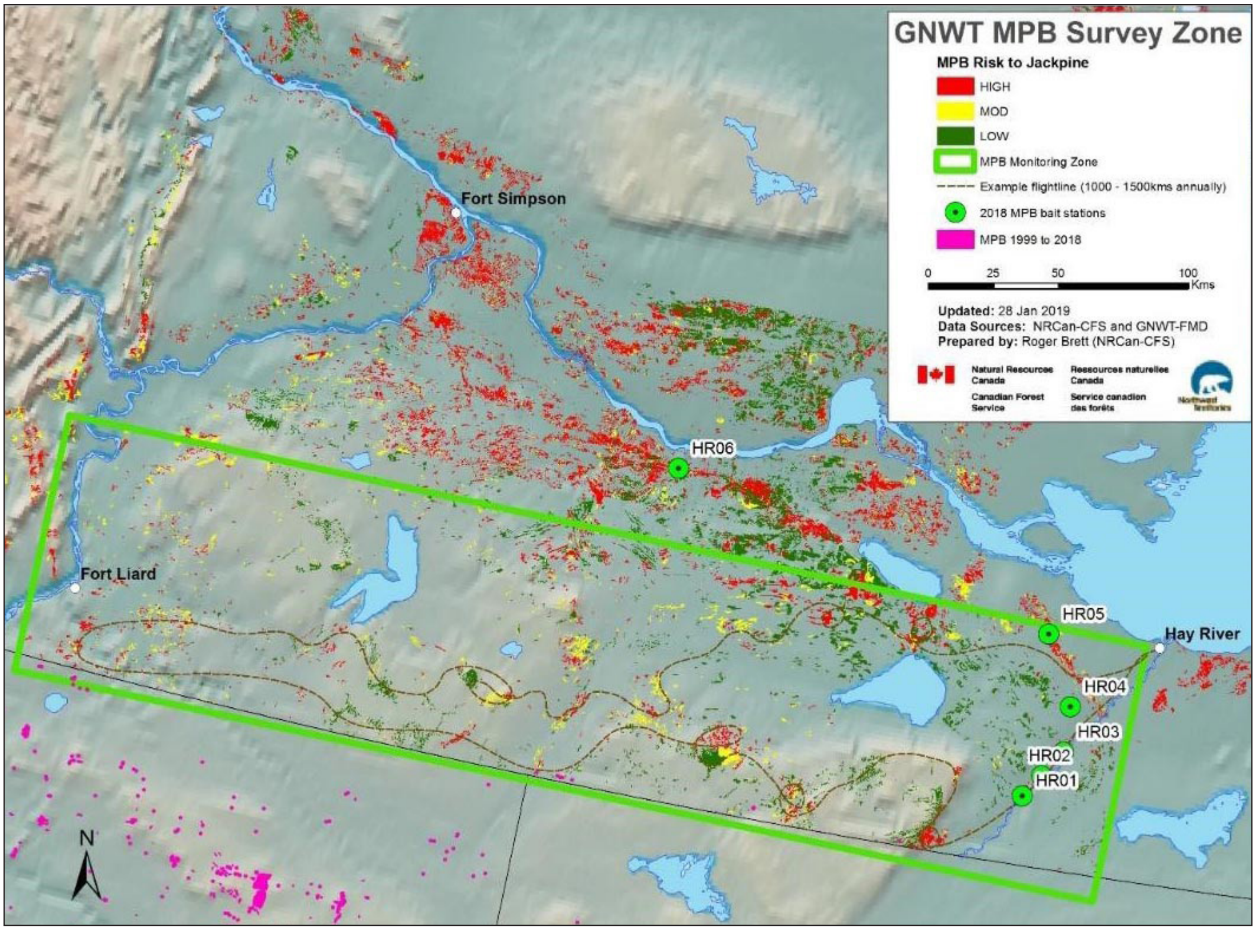


Figure 48. Government of Northwest Territories (NWT) forest health survey monitoring area for mountain pine beetle (MPB). (Data supplied by R. Brett.)

Historical forest health information is available for the Northwest Territories from the Forest Insect and Disease Survey program, which was run by the CFS from 1936 to 1995. This program gathered data on forest insect outbreaks and diseases across Canada and published the results in annual reports that have since been compiled (Hall 1996) and digitized (Power 1986). Data for the Northwest Territories were included for the first time in 1954 as part of the Alberta report. A spruce budworm infestation was recorded that year along the Mackenzie River and for most years after that (Department of Agriculture Canada 1954). More than 370 000 ha in the Northwest Territories along the Slave and Mackenzie Rivers suffered moderate to severe defoliation from the spruce budworm in 1994 (Hall 1996). These annual reports provide an excellent historical resource for forest insects and diseases found in the Northwest Territories between 1954 and 1995, and information for the Northwest Territories is currently being extracted and compiled digitally in a spreadsheet.

6.2 Forest Diseases

Fungi are the most common source of disease in forests. These organisms rely on three factors for spread, which together are known as the disease triangle: the fungal species, the host, and environmental conditions (primarily temperature and relative humidity) (Scholthof 2007). The effects of changing environmental conditions due to climate change on the spread of fungal pathogens in the northern boreal forest are uncertain. While warming temperatures are now well known, less clear is the effect of climate change on precipitation (Meehl et al. 2007). Thus, projections of fungal disease spread should be made both for a warmer and wetter climate and for a warmer and drier climate. Sturrock et al. (2011) has found that pathogens directly affected by temperature and humidity, such as *Endocronartium harknessii* (the causal agent of western gall rust), are likely to have greater impact under warmer and wetter conditions, but less impact under warmer and drier conditions. Conversely, root disease fungi, such as *Armillaria* spp. are expected to have increased impacts under warmer and drier conditions because these conditions increase stress on the host and decrease its ability to resist the pathogen (Sturrock et al. 2011).

Although many fungal diseases require moist conditions to spread, drought weakens a tree's defenses and can make it more susceptible to disease in the period following the drought, when conditions necessary for fungal colonization are present. Dry seasons in 2014 and 2015, followed by increased moisture during 2016 and 2017, may have increased the incidence of disease across the Northwest Territories (Olesinski and Brett 2017). Western gall rust affects lodgepole and jack pine and caused increased branch mortality, possibly made worse by drought, near Fort Simpson and at the junction of the Mackenzie and Liard Highways in 2015, with expansion of the affected area in 2017 (Olesinski and Brett 2017). The fungus causes galls to form on branches, within which spores form; these galls later burst, allowing the spores to infect other pines. The galls weaken branches, which can lead to breakage (Hiratsuka and Maruyama 1991).

Armillaria sinapina was believed to be the only species of *Armillaria* root rot found in the Northwest Territories, until *A. ostoyae* was found in the South Slave administrative region in 2019. *A. sinapina* mostly infects poplars, but has also been found on conifers. Compared with other species of *Armillaria*, *A. sinapina* is less virulent (Mallett 1995; unpublished data provided by T. Ramsfield).

Other diseases found in the 2017 Northwest Territories forest health survey include spruce-Labrador tea needle rust (*Chrysomyxa ledicola*) in the Inuvik region, comandra blister rust (*Cronartium comandrae*) causing seedling and branch mortality from stem cankers in jack pine near Yellowknife, spruce broom rust (*Chrysomyxa arctostaphyli*) along the Mackenzie River, sweetfern blister rust (*Cronartium comptoniae*) on pine trees near Yellowknife, and marssonina leaf spot (*Marssonina populi*) in aspen stands across the South Slave administrative region (Olesinski and Brett 2017). Many of these fungal disease infestations have been endemic to the Northwest Territories for decades, including comandra blister rust on pine near Yellowknife and the spruce needle rusts *Chrysomyxa ledi* and *C. ledicola* along the Nahanni River; the presence of these diseases were first recorded in reports from the Forest Insect and Disease Survey in 1961 (Department of Forestry [Canada] 1961).

Key Points

- Spruce budworm is the most serious forest insect pest in the Northwest Territories. Temperature is the single most important factor influencing its survival, and a warming climate will likely cause range expansion within the territory. A 2015 outbreak in Inuvik was the most northerly known outbreak in Canada. In addition, short-term records suggest that outbreaks are becoming more frequent.
- The forest tent caterpillar is the most serious aspen defoliator, and its range appears to be expanding in the Northwest Territories. An outbreak of aspen serpentine leafminer has been in progress for the past 20 years, likely shifting its impact from minor growth loss to more significant damage.
- The mountain pine beetle has not yet appeared in the Northwest Territories, aside from a solitary incident near the Alberta border that was quickly contained. However, warming temperatures will allow it to expand northward, and it has already jumped from lodgepole pine–dominated stands to jack pine stands in Alberta's northern boreal forest.
- Drought causes stress for trees, making it more difficult for them to fight off insect and fungal attacks.

Knowledge Gaps

- The lack of long-term records and studies of insect and disease outbreaks in the Northwest Territories makes it difficult to know for certain whether outbreaks expanding farther north are truly novel.

CHAPTER 7. FLOODING

Little baseline information is available concerning abiotic disturbances such as flooding (used here to mean a persistent shift to saturated soils, not temporary river flooding) in the Northwest Territories, which makes the detection of climate-induced changes difficult. For this reason, abiotic disturbances have been systematically included in the annual forest health aerial surveys of the GNWT and CFS since 2009 (Olesinski and Brett 2017). Forest mortality, particularly of aspen, from extensive flooding near Fort Providence (Taiga Plains Mid-Boreal ecoregion) in the South Slave administrative region was first noticed in aerial surveys in 2015 but has likely been present for up to two decades. The flooding has caused large-scale mortality of aspen and, to a lesser extent, conifers through drowning of roots and destabilization of the soil (Olesinski and Brett 2018). Mapping in 2017 revealed that the flooded area amounted to 117 400 ha (Figure 47) (Olesinski and Brett 2017). The flooding is thought to be a result of hydrological changes, including destabilization of permafrost, from forest fires that occurred in the mid-1990s and again in 2014 and 2015 in the Horn Plateau, which drains into the Fort Providence area (Olesinski and Brett 2016). Aspen groves in this area are found on raised hummocks surrounded by wetter lowlands. As flood waters rise, aspen trees on hummock edges are the first to be affected, an example of edge effects (Olesinski and Brett 2017).

The 2017 survey found that high water tables were causing decline of aspen stands in the Liard Valley and along the highway from Fort Simpson to Wrigley in the Dehcho administrative region (Olesinski and Brett 2017).

A subsequent survey, in 2018, detected tree mortality due to flooding in many areas, whereas aspen decline due to high water tables was seen throughout the Northwest Territories, though mostly in the Dehcho and South Slave regions (Olesinski and Brett 2018).

The relationship between permafrost thaw and runoff rates in the discontinuous permafrost zones is not yet known with certainty. It is thought that as thaw increases, collapse scars could connect and drain, increasing runoff rates—and thus flooding—in the short term, while decreasing rates in the long term as the permafrost disappears (Figure 3, Figure 4) (Quinton et al. 2009). The amount of water draining through channels connecting collapse bogs has not yet been quantified, but it is expected to increase as permafrost thaw continues and to play an important role in the more general hydrology of the landscape (Connon et al. 2015).

Key Points

- Large-scale flooding is occurring in the Fort Providence area as a result of large-scale fires in both 1995 and 2012. Permafrost thawing in these burned areas may also be contributing to the flooding, which is leading to extensive tree mortality, especially of aspen. These incidents highlight the complex interactions between disturbances.
- High water tables throughout the Dehcho and South Slave administrative regions are causing aspen dieback

Knowledge Gaps

- Flood damage has only recently been included systematically in forest health surveys, and there is a lack of data on historical flood regimes and impacts.
- The effect of large-scale permafrost thaw on regional hydrology is uncertain. As collapse scars grow larger, they may connect with each other and to regional drainage outlets, increasing runoff rates and flooding.
- Historical air photographs and satellite imagery could be used to further explore the timing and extent of this flooding and dieback.

CHAPTER 8. DROUGHT

8.1 Soil Moisture

Drought has been defined as "an interval of time, generally on the order of months or years, during which the actual moisture supply at a given place rather consistently falls short of the climatically expected or climatically appropriate moisture supply" (Palmer 1965). As described in section 2.4, the CMI, representing the moisture available to plants, is calculated on a monthly basis by subtracting potential evapotranspiration from mean precipitation, where potential evapotranspiration is calculated from monthly means of daily minimum and maximum temperatures, as well as elevation at the particular location (Hogg 1997). Long-term averages of annual CMI, calculated as the sum of the monthly values from 1 August to 31 July, correlate well with the southern extent of the boreal forest in Canada. The southern limit of the boreal forest closely follows the zero isoline of the CMI; south of that line, annual evapotranspiration is greater than mean annual precipitation, and the forest gives way to open parkland (Hogg 1994).

In tree-ring studies of aspen, the strongest correlation between CMI and tree growth was observed when annual CMI was calculated for a "water year," defined as 1 August of the previous year to 31 July of the current year (Hogg et al. 2005). However, a lag was found between the annual CMI values for the water year and changes to tree growth, likely because of the trees' deep rooting zone. Thus, there was a correlation between aspen growth on CIPHA study sites (locations shown in Figure 14) and the CMI of the previous 4 years, as well as the CMI of the current growing year (Hogg et al. 2005). To examine the cause of the growth lag, Hogg et al. (2013) developed the concept of Soil Moisture Index (SMI), which represents the amount of soil water available in the rooting zone of trees. The SMI is calculated using CMI inputs and a simple model of soil water balance. It can be calculated from existing historical observations of weather stations and can be easily mapped over large areas and time scales, so is a potentially useful metric. When SMI values were examined in relation to aspen growth on the same sites as the previous CMI study, correlation with growth was found only for that year's SMI, thus eliminating the lag. This finding was attributed to SMI capturing the drawdown of soil moisture reserves during drought and the subsequent recharging of reserves over the following years, meaning that SMI experiences less annual fluctuation than does CMI (Hogg et al. 2013).

According to annual CMI over the period 1951–2010, the two driest decades in the Canadian boreal forest were 1951–1960 and 2001–2010, with drying trends being noted in the northern Taiga Plains and Taiga Cordillera ecoregions (Wang et al. 2014).

Aspen growth rates at six sites near Fort Smith in the Northwest Territories (Taiga Plains Mid-Boreal ecoregion), which had not experienced significant insect defoliation for the previous 87 years, were compared in terms of CMI and SMI to more confidently isolate the effects of soil moisture on growth. Growth rates again showed a correlation with the CMI of the current year and of the previous 4 years, but correlation with the SMI occurred only for the current year. The slight lag that exists even when using SMI can be explained by xylem cavitation or depletion of carbohydrate reserves during drought, or both (Hogg et al. 2013). At the Fort Smith sites, both CMI and SMI regression equations were successful in explaining years of exceptionally low growth, but were not successful for years with exceptionally high growth, making these indexes good predictors of the effects of drought (Hogg et al. 2013).

Soil moisture is an important driver of tree growth and is positively correlated with the growth of black spruce, lodgepole and jack pine, and trembling aspen throughout the Taiga Plains ecoregion (Girardin et al. 2016b). Furthermore, data accumulated from sites across western Canada, including the southern Northwest Territories, show that the probability of tree survival increases with increasing CMI for black and white spruce, aspen, and balsam poplar, but decreases with increasing CMI for jack pine (Cortini et al. 2017). SMI trends, according to the method of Hogg et al. (2013), as described above, were negative for the Taiga Plains ecoregion from 1950 to 2002 and can thus be correlated with the decrease in growth observed during that period (see section 3.6, above) (Girardin et al. 2016b).

The positive correlation observed between black spruce growth and soil moisture may hold only for trees growing on well-drained mineral soil sites. For black spruce growing in poorly drained peatlands, the observed correlation was actually negative, whereby lowering the water table causes increased growth (Girardin et al. 2016b). However, this relationship can be further confounded by permafrost thaw and the resultant deepening of the active layer; a study by Sniderhan and Baltzer (2016) suggested that a lower water table resulting from vertical permafrost thaw decreased annual black spruce growth.

Although a global study found that drought typically had a greater effect on the growth and mortality of larger trees (Bennett et al. 2015), tree-ring analysis conducted in the boreal forests of Alberta and Saskatchewan showed that drought caused a greater decline in aboveground biomass accumulation in younger white spruce stands than in older stands, an effect that was attributed to their higher productivity (Hogg et al. 2017).

Aspen decline has been observed over the past number of years in the Dehcho region, but the cause is uncertain. The extent of the area of decline is now being mapped by the Forest Management Division of the GNWT Department of Environment and Natural Resources, with ground checking and sample collection to follow (Olesinski and Brett 2017). However, long-term studies of aspen decline in the Prairies, characterized by similar symptoms, suggests that it is correlated with drought conditions lowering the soil moisture levels (Hogg et al. 2013; Olesinski and Brett 2017). The CIPHA project of the CFS found 35% aspen mortality between 2012 and 2016 on study sites along the Poplar River southwest of Fort Simpson (Taiga Plains Mid-Boreal ecoregion) (Figure 14). The MVI project of the CFS found 42% aspen mortality and 11% biomass reduction between 2008 and 2016 on study plots near Fort Liard (Taiga Plains Mid-Boreal ecoregion) using tree measurements and tree-ring analysis. For both sites, the likely cause is a combination of severe drought in 2013 and 2014 and insect damage from forest tent caterpillars and leafminers (Hogg 2017).

Drought causes a drying of vegetation and an increased risk of fire occurrence and severity, including the burning of deeper peat layers in peatlands (Price et al. 2013). In addition, as described in section 4.4, fire increases the rate of permafrost thaw, which may also contribute to flooding. Linear trends of CMI from the climate station in Fort Simpson (Taiga Plains Mid-Boreal ecoregion) show severe drought years in 1994, 2013, and 2014 (Figure 49). The low CMI in those years can be correlated with severe fire years in 1995 and 2014 in the Taiga Plains Mid-Boreal ecoregion (Figure 34).

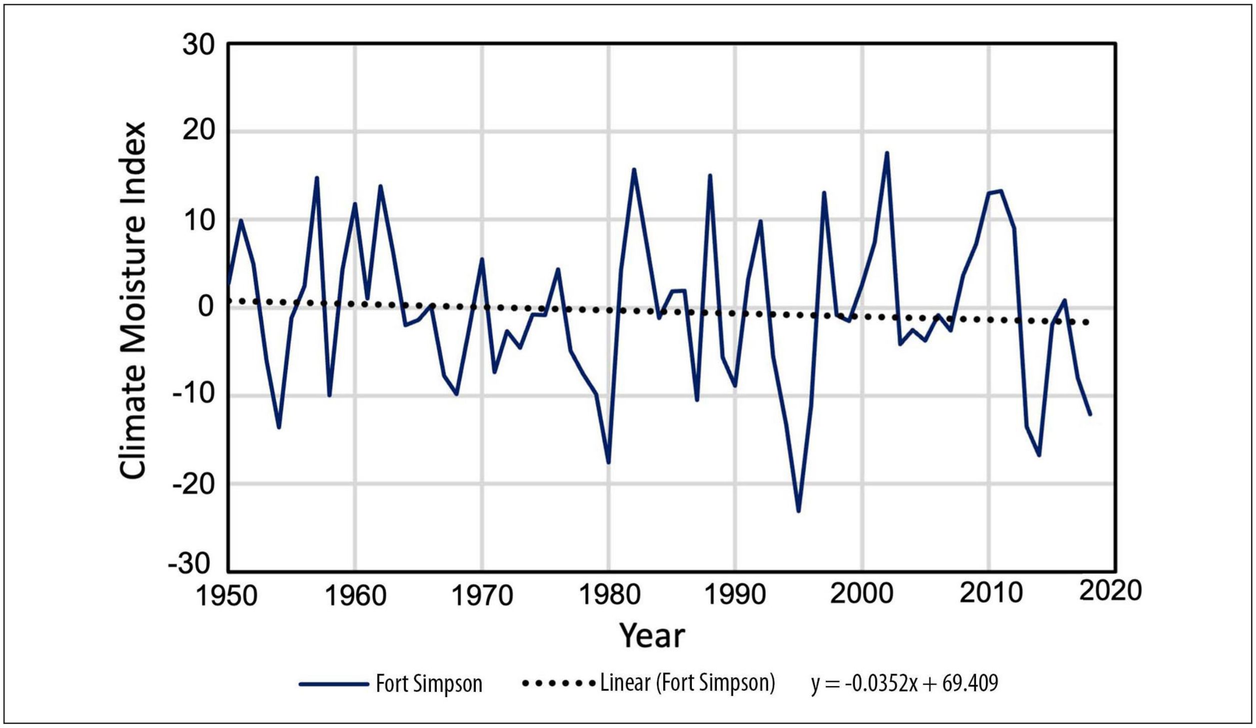


Figure 49. Trend in climate moisture index for 1950–2018 based on data collected from the climate station in Fort Simpson, Northwest Territories (Taiga Plains Mid-Boreal ecoregion). (Data supplied by E.H. Hogg.)

8.2 Drought Vulnerability

Aubin et al. (2018) studied vulnerability to drought for 22 tree species found in Canada's boreal forest. Vulnerability is a result of the combination of three factors: exposure to environmental change, the sensitivity of the species (the degree to which it is affected by the change), and the adaptive capacity of the species (its ability to respond to impacts from the change).

The exposure in this case was defined as changes to the CMI in regions where a given species is found. A relative scale sensitivity index was created using drought-related species traits such as rooting depth, root sensitivity to damage, and resistance of the xylem to cavitation. Adaptive capacity, measured as a species' ability to track its climatic niche through migration, combines distance between current and ideal or future distribution with life-history traits that determine migration capacity, such as seed production, seed dispersal, and colonization potential. For the purpose of calculating distance to the future ideal habitat, the climatic niche was based on a combination of various measures of mean temperature and precipitation (Aubin et al. 2018).

This work yielded two color-coded maps. For the first map, the color legend depicts biomass under drought conditions on the *x*-axis and drought sensitivity on the *y*-axis (Figure 50). Map colors matching those in the top right corner of the color legend represent areas with high biomass under drought conditions and high drought sensitivity. For the second map, the color legend depicts distance to a suitable climatic niche on the x-axis and migration capacity on the y-axis (Figure 51). Map colors matching those in the top right corner of the color legend represent areas containing species that are far from a suitable climate and have low migration capacity (Aubin et al. 2018). Each of the two maps was also projected into the future for the periods 2041 to 2070 and 2071 to 2100 on the basis of future CMI projections.

Drought sensitivity index values ranged from −19 (high sensitivity) to 120 (low sensitivity), and migration capacity index values ranged from 15 (low capacity) to 105 (high capacity). Values of sensitivity and migration capacity were calculated for seven common tree species in the Northwest Territories (Table 24). The deciduous species of trembling aspen and paper birch have high drought sensitivity but also high migration capacity, ensuring the likelihood that they will maintain viable populations into the future, even if current distributions are hit by drought. Black spruce has above-average drought sensitivity and below-average migration capacity, which means it may suffer biomass diebacks if its current range is hit by drought. White spruce has slightly lower drought sensitivity than black spruce, but it too may be vulnerable under drought conditions. Jack pine has both low drought sensitivity and high migration capacity, ensuring continued viability (Aubin et al. 2018).

Results of hydric envelope mapping show that areas west of Great Slave Lake and around the northern Mackenzie Valley will likely have moderate levels of biomass consisting of drought-sensitive species existing under drought conditions during the period 2011–2040 (Figure 50), with this situation becoming more widespread through the periods 2041–2070 and 2071–2100. However, within the Northwest Territories, there is only a small area in the southwest with low migration capacity and moderate distance to a suitable climatic niche during the period 2011–2040 (Figure 51); even by the period 2071–2100, this combination of distance to climatic niche and low migration capacity remains largely confined to the southwest area of the territory (Aubin et al. 2018). These findings suggest that in the coming decades the vulnerability of forests in the Northwest Territories to drought (whereby the forest have large numbers of drought-sensitive species with low migration capacity existing under drought conditions) will be small. Nevertheless, it should be emphasized that even mild drought may interact with or potentially amplify other impacts of climate change, such as insect and disease outbreaks or wildfire occurrence and severity (Figure 3, Figure 4), and the potential impacts of these interactions are currently not well understood.

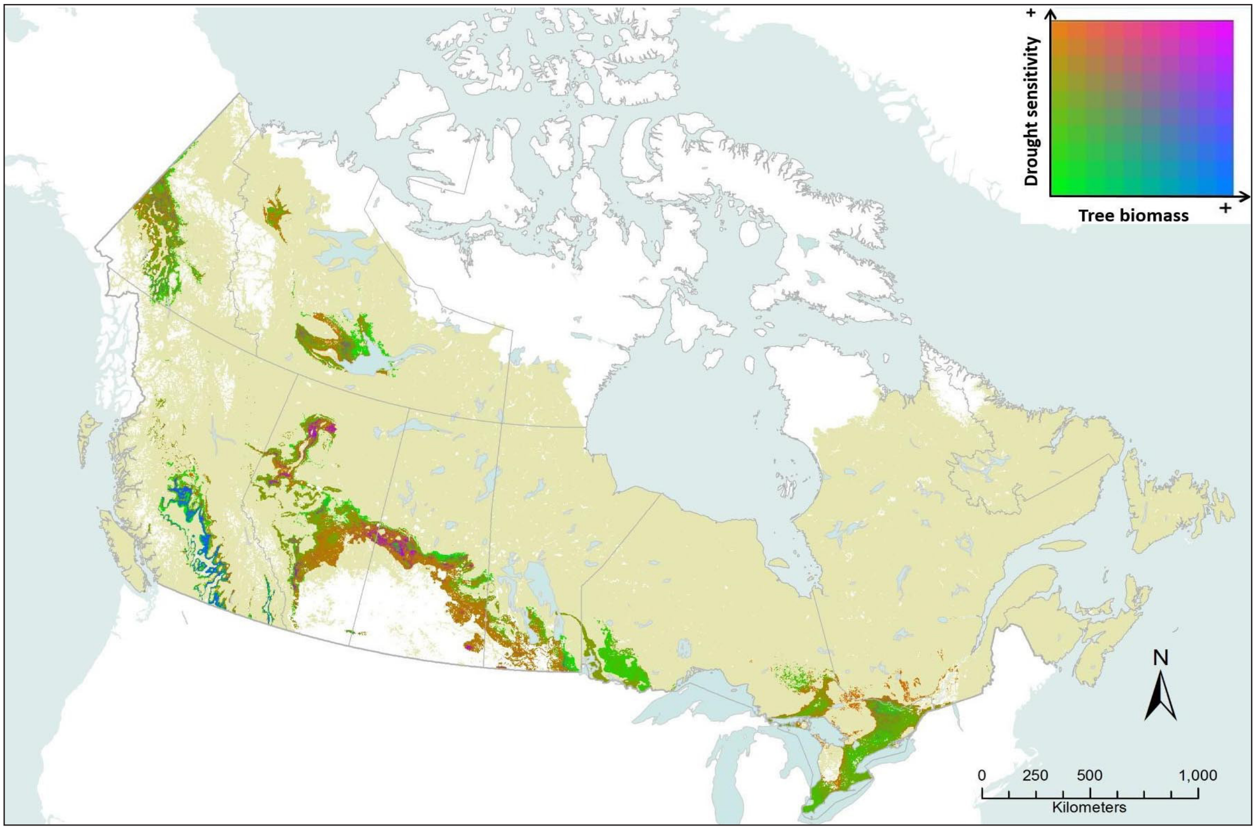


Figure 50. Canadian forest sensitivity to drought projected for the period 2011–2040. The x axis of the legend indicates stand biomass exposed to drought (in tonnes per hectare), referring to the forest biomass that is projected to experience conditions below the current hydric envelope. The y axis represents the drought sensitivity of the stand type, which corresponds to the weighted average of the species' sensitivity index values. Areas of high exposure and high sensitivity are shown in fuchsia (upper right corner of legend). Areas where forest stands are projected to stay within the current hydric envelope are shown in beige. (Reproduced, with permission, from Aubin et al. [2018].)

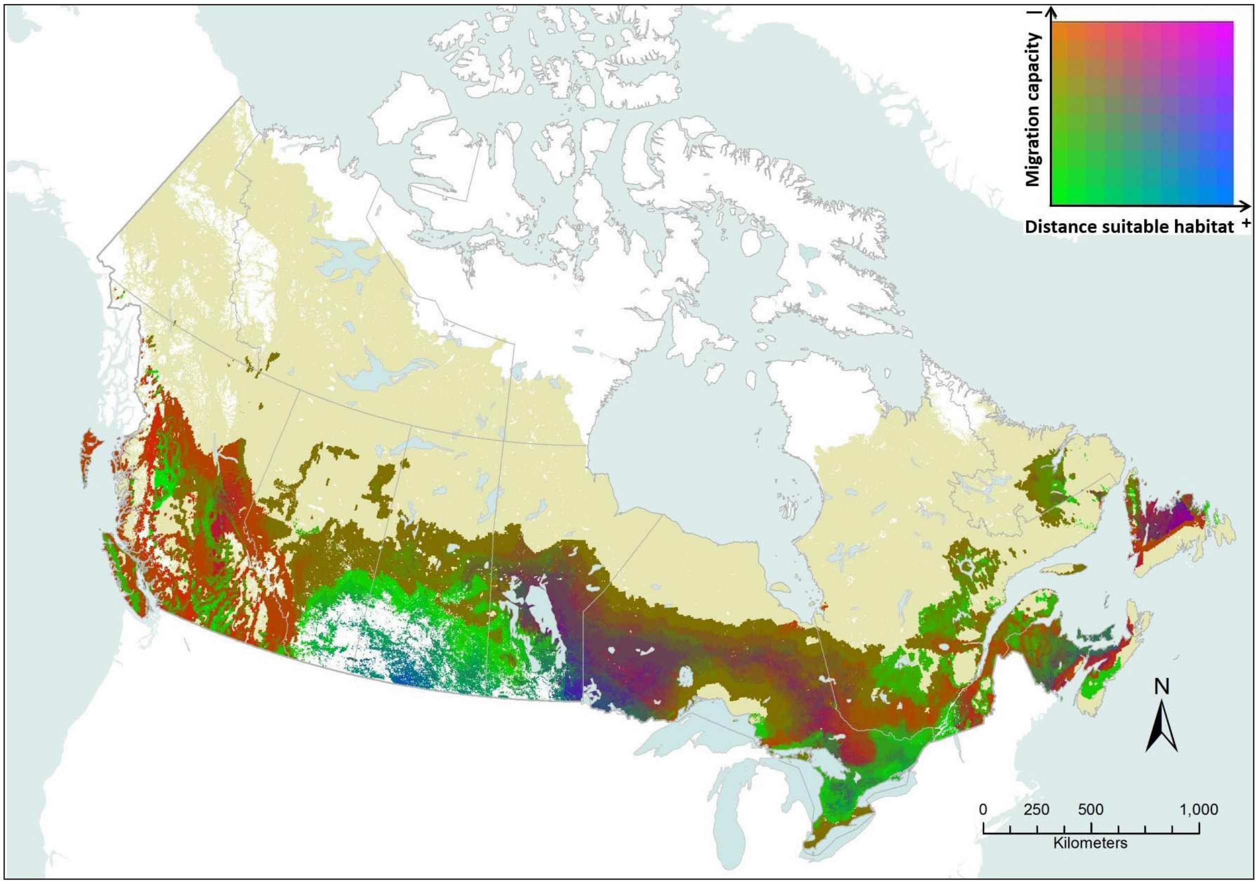


Figure 51. **Canadian forest vulnerability to migration failure for the period 2011–2040.** The x axis of the legend indicates distance to suitable climatic habitat (in kilometers) for species expected to experience conditions outside their current climatic envelope. The y axis represents the stand migration capacity, which corresponds to the weighted average of the species' migration index values. Areas of long distance to suitable habitat and low migration capacity are shown in fuchsia (upper right corner of legend). Areas where forest stands are projected to stay within the current climatic envelope are shown in beige. (Reproduced, with permission, from Aubin et al. [2018].)

Table 24. **Drought sensitivity and migration capacity indices for 7 boreal tree species found in the Northwest Territories.** (Adapted, with permission, from Aubin et al. [2018].)

	Drought sensitivity[b]		Migration capacity index[c]	
Species[a]	Value	Rank[d]	Value	Rank[d]
Betula papyrifera (paper birch)	–15	1	100	4
Larix laricina (larch)	35	15	70	14
Picea glauca (white spruce)	30	13	70	14
Picea mariana (black spruce)	20	7	70	14
Pinus banksiana (jack pine)	54	19	95	5
Pinus contorta (lodgepole pine)	55	20	55	20
Populus tremuloides (trembling aspen)	8	4	105	1

[a] *Listed in alphabetical order of scientific names.*

[b] *Drought sensitivity index values range from –19 (high sensitivity) to 120 (low sensitivity).*

[c] *Migration capacity index values range from 15 (low capacity) to 105 (high capacity).*

[d] *Ranking is from 1 (most sensitive) to 22 (least sensitive); rank numbers omitted from this table refer to species included in the analysis of Aubin et al. (2018) but not found in the Northwest Territories.*

Key Points

- Tree-ring studies have shown that forest growth is often limited by soil moisture, even under the cold climatic conditions of the region. Recent severe droughts may be an important cause of forest decline episodes in the Dehcho and South Slave administrative regions.
- A negative trend of modeled soil moisture conditions in the Taiga Plains ecoregion for 1950 to 2002 corresponds with declining tree growth during that period. Trembling aspen, balsam poplar, lodgepole pine, jack pine, white spruce, and black spruce survival rates have all been negatively affected by drought conditions.
- The sensitivity of trees to drought, expected exposure to drought conditions, migration capacity, and distance to suitable climates have been mapped for Canada. Black spruce has above-average drought sensitivity and below-average migration capacity, making it more susceptible to drought than other species.

Knowledge Gaps

- Causes of aspen decline in the Dehcho administrative region are uncertain, although similar symptoms in the Prairies were the result of drought.
- Climate stations in the Northwest Territories, especially those that can provide reliable observations of precipitation, are sparse. There are plans to establish additional climate stations in the future, but the number of stations and their locations are uncertain.

CHAPTER 9. CARBON BALANCE

9.1 Introduction

The boreal forest contains 18%–23% of the world's terrestrial carbon stocks (338 000–471 000 Tg [teragrams, 10^{12} g)—mostly in its soil—and has a greater soil carbon density than any other biome in the world (Prentice et al. 2001). The boreal region accounts for 77% of Canada's forests and about 30% of the global boreal forest (Natural Resources Canada 2020). Thus, the condition of carbon stocks in Canada's boreal forests plays a significant role in global carbon sequestration.

The forest carbon balance is a product of two basic processes: uptake of atmospheric CO_2 through net primary production (NPP) and release of CO_2 and methane (CH_4) back to the atmosphere through decomposition by heterotrophic and anaerobic respiration (Figure 52). Added to these continuous processes are anthropogenic and natural disturbances such as wildfire, insect infestations, and harvesting. These disturbances transfer carbon from live biomass directly into the atmosphere through oxidation, into dead organic matter (DOM) that decompose slowly over years through respiration, and into wood products with variable lifespans (Kurz et al. 2013).

9.2 Measurement and Modeling

The carbon cycle consists of complex processes over large scales, making it difficult to measure directly (Kurz et al. 2013). Kurz et al. (2013) outlined various methods for estimating carbon flux from independent data sources, creating an opportunity for comparison and verification. Field measurements include using eddy covariances to measure vertical air fluxes and detect CO_2 exchanges between the forest and atmosphere over small areas. However, eddy covariance measurements do not capture periods of disturbance during which large CO_2 fluxes can occur over short periods. Yield curves—predictions of tree growth for a given species in different locations and environmental conditions—exist for particular stand types within the ecosystems across the boreal forest of Canada. When combined with knowledge about DOM and soil carbon pools, these curves can be used to estimate the stand-level carbon balance. Finally, tree-ring analysis can be used to estimate growth rate and mortality and to extend the spatial and temporal knowledge of ecosystem productivity. Improving technology may soon allow the use of remote sensing to measure carbon biomass from the reflectance of different vegetation types or structural characteristics (Kurz et al. 2013). Atmospheric inversion models measure CO_2 fluxes over large areas and generate estimates based on entirely independent data. However, they cannot provide an estimate of forest carbon balance in regions consisting of a mix of forested and nonforested landscapes (Kurz et al. 2013).

The National Forest Carbon Monitoring, Accounting and Reporting System (NFCMARS) is the CFS program responsible for calculating Canada's forest carbon balance. It incorporates forest inventories, empirical yield curves for specific sites, process modeling of DOM and soil carbon, statistics on forest management, and remote sensing of disturbance and land use changes into the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) to estimate the carbon balance at regional or national scales (Kurz et al. 2013). The network of NFI ground plots was used as an independent data source to validate the model (Shaw et al. 2014). The CBM-CFS3 was devised for upland managed forests, for which inventory data were available. Managed forests are defined for the purpose of the CBM-CFS3 as those managed for sustainable harvest of wood products or biofuels, those actively protected from natural disturbances, and those in protected areas managed for ecological values. As such, managed forests in this model do not correspond to the more common concept of managed forests that have a greater degree of human management activity. Managed forests included in the CBM-CFS3 account for about half of the forested upland area in the Canadian boreal forest, but only 30.2% of upland forests in the Northwest Territories (Figure 53) (Stinson et al. 2011; Kurz et al. 2013).

The MOSS-C module has been designed to incorporate deep organic layers of upland peaty forests into the CBM-CFS3 (Shaw et al. 2016). Peaty forests have a forest floor or organic soil layer up to 40 cm thick, mostly composed of living and decomposing bryophytes, which may add up to 20% in upland forest productivity (Kurz et al. 2013). Before inclusion of the MOSS-C module, the CBM-CFS3 underestimated the carbon stocks

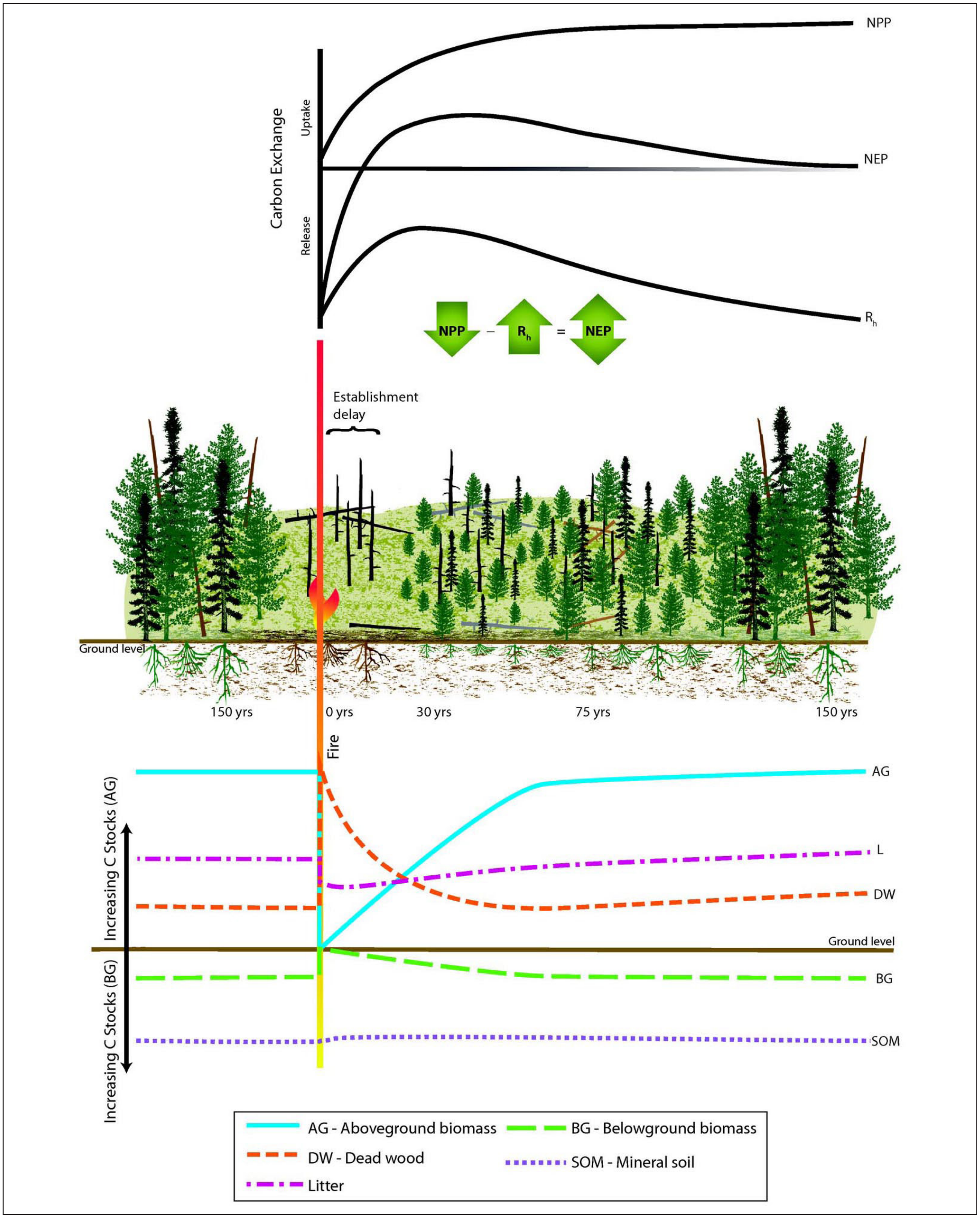


Figure 52. Generalized schematic of the dynamics of stand-level net primary production (NPP), heterotrophic respiration (Rh, shown here as a negative flux and including in this graph large direct fire emissions in the year of the disturbance), and their net balance, net ecosystem production (NEP) after a fire disturbance (upper graph). The generalized carbon stock dynamics after a fire disturbance are shown in the lower graph. For simplicity, dead belowground (coarse root) wood is included in the aboveground dead wood (DW) pool (Reproduced, with permission, from Kurz et al. [2013].)

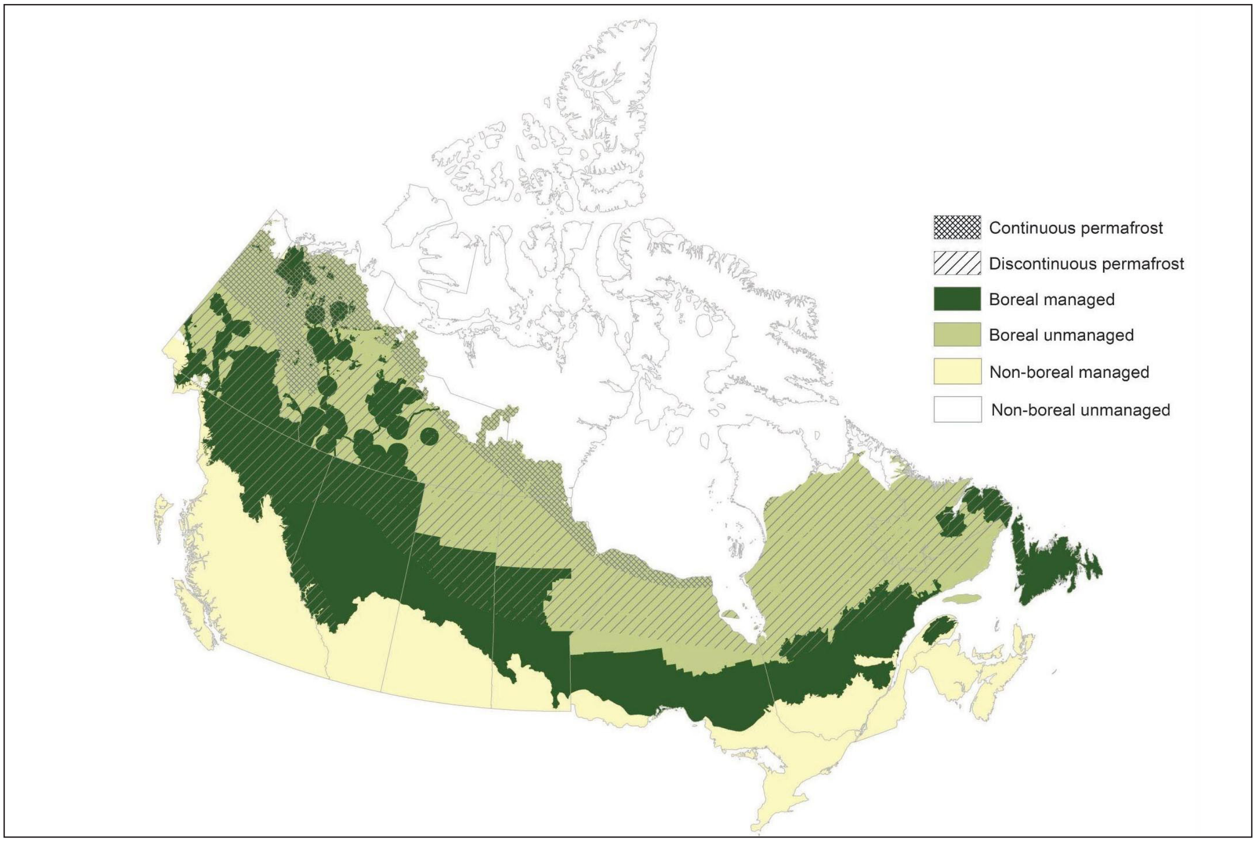


Figure 53. Managed forest area in Canada. In the Northwest Territories, managed forests comprise 13.32 Mha (13.32×10^6 hectares), or 30.2% of the territory's forested land base (Reproduced, with permission, from Kurz et al. [2013].)

of peaty forests because it excluded the forest floor. This was particularly problematic for black spruce–dominated boreal forests, which are often transitional between upland and peatland forests. Moss can contribute 31%–49% of ecosystem carbon stocks in black spruce forests, especially on poorly drained sites (Bona et al. 2013). The MOSS-C module includes carbon pools from both feather and sphagnum mosses and is now in operation. Initial results show that it increases the accuracy of the CBM-CFS3 (Bona et al. 2016).

The Canadian Model for Peatlands (CaMP) module was built for the Generic Carbon Budget Model, the newly developed, spatially explicit version of the CBM-CFS3. The CaMP module is designed to track greenhouse gas emissions and removals from lands classified by the Intergovernmental Panel on Climate Change as wetlands or forest land on organic soils (forested wetlands) for regional and national accounting (Bona et al. 2020). The "peatland site" is the basic modeling unit within the CaMP module, and each such site is defined by peatland category, ecoregion, and political boundary. The peatland categories tracked in the module follow the Canadian wetland classification categories (National Wetlands Working Group 1997) of bog and fen, which are then divided into categories of forested (tree height 5 m, 25% cover), treed (tree height < 5 m, 10%–25% cover), and open (tree height < 5 m, <10% cover) (Shaw et al. 2016; Bona et al. 2020). Model carbon pools include biomass (mosses, sedges, shrubs, and trees), DOM, the peat zone within which the water table fluctuates and aerobic decomposition can take place (the acrotelm), and the peat zone below the static water table within which anaerobic decomposition takes place (the catotelm) (Bona et al. 2020). Default rates of productivity and mortality are based on literature-cited values from 186 peatland study sites compiled in a parameter database (Bona et al. 2018). The CaMP module uses a Q_{10} relationship with mean annual temperature to model variation in decay and a CH_4 flux response to deviations in annual water table depth. The model has been evaluated against site-level observed flux data and was found to provide an appropriate framework for large spatial- and temporal-scale estimation; however, further work is needed to explain finer-scale variation. In the Northwest Territories, the model might be validated by sites in the Mackenzie Valley comprising bog, peat plateau, and poor fen, where data on CO_2 and CH_4 exchange, net ecosystem exchange (NEE), water table levels, and biomass production were collected between 2007 and 2010 (Shaw et al. 2016).

9.3 Upland Forests

Carbon stocks in managed upland forests over the period 1990–2008 were calculated for Canadian ecoregions using the CBM-CFS3. This analysis included all aboveground and belowground biomass, dead wood, and soil carbon. The Taiga Plains ecoregion had carbon stocks of about 3000 Tg (or 3 Pg) and carbon density of about 150 Mg $C·ha^{-1}$ (150 megagrams [10^6 g] of carbon per hectare) in its upland forests, the western Taiga Shield ecoregion had stocks of about 250 Tg and density of 135 Mg $C·ha^{-1}$, and the Taiga Cordillera ecoregion had stocks of about 100 Tg and density of 240 Mg $C·ha^{-1}$ (Table 25) (Kurz et al. 2013).

Preliminary results from a CFS pilot study focusing on an area of 17.8 Mha within Taiga Plains forests of the Northwest Territories over the period 1990–2018 showed carbon stocks of 3755 Tg and carbon density of about 210 Mg $C·ha^{-1}$. Total ecosystem carbon stocks were estimated to be increasing, from 3614 Tg in 1990 to 3755 Tg in 2018. This represents an estimated increase in total ecosystem carbon stocks of about 140 Mg or 515 Mg CO_2^e (17.78 Mg $CO_2^e·yr^{-1}$), where CO_2^e refers to "CO_2 equivalent" (unpublished data provided by M. Voicu) (To convert the mass of carbon to the mass of CO_2, the mass of carbon is multipled by the mass ratio CO_2/C: mass of CO_2 = mass of C × [44/12]). Between 1990 and 2017, the managed upland forests of the Northwest Territories had carbon stocks of about 1832 Tg and carbon density of about 138 Mg $C·ha^{-1}$ (Environment and Climate Change Canada 2019).

The carbon flux of Canadian boreal forests was calculated at a national scale. It was found that after a stand-replacing disturbance, upland boreal forests in Canada are a net source of carbon for 5–30 years (10–20 years after harvest), because during that period, the accumulated carbon decomposes faster than carbon is taken up by the forest. Eventually, net ecosystem production (NEP), the difference between a tree's carbon absorption and its heterotrophic respiration, begins to outstrip decomposition, and the forest again becomes a carbon sink (Figure 52) (Kurz et al. 2013). The period during which the forest is a net carbon source depends on the intensity of disturbance, site productivity, and management actions. In one study, stand-replacing disturbances in the boreal forest resulted in maximum emission rates (with NEP of 733 g $CO_2·m^{-2}·yr^{-1}$ [grams of CO_2 per square meter per year]) in the first years after disturbance, followed by maximum carbon uptake (with NEP 770 g $CO_2·m^{-2}·yr^{-1}$) when the new stands reached their middle age (Amiro et al. 2010). In mature stands, biomass may remain constant or decline, while carbon pools in the DOM and litter continue to increase. As ecosystem carbon storage reaches its limit, these stands may become a carbon source or remain as a small sink (Figure 52) (Kurz et al. 2013).

At the landscape level, the carbon balance is affected mostly by the proportion of different stand ages over the landscape: frequent disturbance leads to a greater proportion of younger forests with lower amounts of stored carbon, in which case the system might be a net carbon source over the disturbance cycle, whereas an infrequent disturbance regime leads to a greater proportion of older forests with greater amounts of stored carbon, in which case the system would be a net

Table 25. Carbon stocks, carbon density, and net ecosystem exchange (NEE) of carbon dioxide (CO_2) for managed forests in Canadian ecoregions, 1990–2008, using the Carbon Budget Model of the Canadian Forest Service (CBM-CFS3)[a]. (Data source Stinson et al. [2011].)

Ecoregion	Carbon stocks (Pg C)	Carbon density (Mg $C·ha^{-1}$)	NEE_{CO2} (g $CO_2·m^{-2}·yr^{-1}$)
Taiga Plains	3	150	−147 to −73
Taiga Shield West	0.25	135	−73 to 0
Taiga Cordillera	0.1	240	+147[b]

[a]High carbon density in combination with low carbon stocks, as reported in this table, is a result of these ecoregions having little area of managed forest. A negative value for NEE (expressed as grams of carbon dioxide per meter squared per year) signifies net sequestration; a positive value signifies net emissions.

[b]Net emissions in the Taiga Cordillera was a result of the high wildfire rate during the study period.

carbon sink over the disturbance cycle (Chetrov et al. 2009; Kurz et al. 2013).

The NPP for Canada's boreal forest between 1990 and 2008 was estimated using the CBM-CFS3 as 438 Tg $C{\cdot}yr^{-1}$, of which 391 Tg $C{\cdot}yr^{-1}$ was returned to the atmosphere through heterotrophic respiration (Rh), while 19 Tg $C{\cdot}yr^{-1}$ was returned to the atmosphere through wildfire. The NEP (where NEP = NPP – Rh) was therefore 47 Tg $C{\cdot}yr^{-1}$. Disturbances (D) were represented by the sum of emissions from wildfire (19 Tg $C{\cdot}yr^{-1}$) and harvesting (17 Tg $C{\cdot}yr^{-1}$). Net biome production (where NBP = NEP – D) therefore amounted to 11 Tg $C{\cdot}yr^{-1}$ removed from the atmosphere by Canada's boreal forest between 1990 and 2008 (Kurz et al. 2013). Carbon mass, rather than CO_2 mass, was used for these calculations, because the carbon flux includes CH_4 and CO emissions from wildfires and, given that the proportions of different greenhouse gases contributing to the total is not clear, it is not possible to convert carbon flux to a CO_2 equivalent (CO_2^e; see section 9.4, below). Current accounting guidelines of the United Nations Framework Convention on Climate Change treat all harvested wood products as replacing previously harvested wood that has decomposed. Thus, carbon removed from the forest through harvest is treated as if it is immediately released to the atmosphere. In reality, the global pool of harvested wood is increasing. An estimated 56.9% of harvested carbon in the period 1990–2008 has not yet been released, but how much of that replaced previously harvested carbon and how much was added to the existing pool is not known (Kurz et al. 2013). Thus, roughly only 2.5% of NPP remains in the forest ecosystem (11/438 $Tg{\cdot}yr^{-1}$), which is explained as being due to the old age of these forests and the high natural disturbance rate during those years. Canada's boreal forests are on average 85 years old and their carbon density is high relative to the ecosystem's carbon carrying capacity, which limits their ability to sequester additional carbon (Kurz et al. 2013).

The CBM-CFS3 was also used to calculate the carbon flux of managed upland forests of the Northwest Territories over the period 1990–2008. The model generated a CO_2 sequestration rate of 73–147 g $CO_2{\cdot}m^{-2}{\cdot}yr^{-1}$ for forests in the Taiga Plains ecoregion, a sequestration rate of 0–73 g $CO_2{\cdot}m^{-2}{\cdot}yr^{-1}$ for forests in the western Taiga Shield ecoregion, and an *emission* rate of more than 147 g $CO_2{\cdot}m^{-2}{\cdot}yr^{-1}$ for forests in the Taiga Cordillera ecoregion, the latter due to the high incidence of fire during the model period (Table 25) (Stinson et al. 2011). These estimates of CO_2 sequestration and emissions were determined from the NEE, the difference between total ecosystem photosynthetic activity and ecosystem-wide respiration and decomposition from natural disturbances such as wildfire. About 17.45 Mha of upland forest exists in the Northwest Territories portion of the Taiga Plains ecoregion (Ecosystem Classification Group 2007 [rev. 2009]). Extrapolating the sequestration rate of managed forests to all upland forests in the ecoregion would give an annual sequestration rate of 12.74–25.65 Tg $CO_2{\cdot}yr^{-1}$.

Preliminary results from the CFS pilot study mentioned above, which focused on carbon stocks and flux for an area of 17.8 Mha within Taiga Plains forests of the Northwest Territories over the period 1990–2018, estimated NPP as 324.5 g $C{\cdot}m^{-2}{\cdot}yr^{-1}$, of which 267.8 g $C{\cdot}m^{-2}{\cdot}yr^{-1}$ was returned to the atmosphere through heterotrophic respiration (Rh) of DOM and soil organic matter, while 21.5 g $C{\cdot}m^{-2}{\cdot}yr^{-1}$ was returned to the atmosphere through disturbances. The NEP (where NEP = NPP – Rh) was therefore 56.7 g $C{\cdot}m^{-2}{\cdot}yr^{-1}$. Disturbances (D) were again represented by the sum of emissions from wildfire and harvesting (27.3 g $C{\cdot}m^{-2}{\cdot}yr^{-1}$). The NBP (where NBP = NEP – D) therefore amounted to 29.4 g $C{\cdot}m^{-2}{\cdot}y^{-1}$ removed from the atmosphere by the forests of the Taiga Plains ecoregion between 1990 and 2018 (unpublished data provided by M. Voicu).

Between 1990 and 2017, the NPP for the managed forests of the Northwest Territories was estimated as 202.8 g $C{\cdot}m^{-2}{\cdot}y^{-1}$, of which 165.4 g $C{\cdot}m^{-2}{\cdot}y^{-1}$ was returned to the atmosphere through heterotrophic respiration (Rh) of DOM and soil organic matter, while 21.5 g $C{\cdot}m^{-2}{\cdot}y^{-1}$ was returned to the atmosphere through disturbances. The NEP (where NEP = NPP – Rh) was therefore 37.4 g $C{\cdot}m^{-2}{\cdot}y^{-1}$. In this analysis, disturbances (D) were represented by the sum of emissions from wildfire (21.4 g $C{\cdot}m^{-2}{\cdot}y^{-1}$), slash burning (0.5 g $C{\cdot}m^{-2}{\cdot}y^{-1}$), and harvesting (0.1 g $C{\cdot}m^{-2}{\cdot}y^{-1}$). The NBP (where NBP = NEP – D) amounted to 15.4 g $C{\cdot}m^{-2}{\cdot}yr^{-1}$ removed from the atmosphere by the managed forest of the Northwest Territories between 1990 and 2017 (Environment and Climate Change Canada 2019).

Despite the modeled sequestration rates for managed upland forests in the Taiga Plains and Taiga Shield ecoregions, the near-constant amount of carbon in their soils indicates that they may have been carbon neutral over a long period, with tree-canopy carbon uptake balancing ground-surface decomposition (Startsev et al. 2016). Investigation of an upland study site in the Taiga Plains High Boreal ecoregion near Fort Simpson showed that the ground surface and near-ground vegetation (not including the carbon accumulation of trees or larger vegetation) emitted about 1100 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ because of high respiration rates of the well-drained soil, large input of organic matter from the tree canopy, and low CO_2 uptake by shaded mosses (Startsev et al. 2016). Lower temperatures farther north decreased the respiration rates. In the same study, a Low Subarctic upland site near Norman Wells emitted about 360 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ and a High Subarctic site near Inuvik emitted about 139 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ (Startsev et al. 2016). Increasing temperatures would likely increase both biomass production and decomposition rates, and the net result in terms of carbon storage is unclear (Startsev et al. 2016).

CFS researchers are currently looking at the carbon stocks and flux of Taiga Plains forests in the Northwest Territories over the period 1990–2018. Spatially explicit modeling such as that employed through this research suggests that the 17.8 Mha within the area of interest is a net carbon sink during most years, except for severe fire seasons (such as 2014) (unpublished data provided by M. Voicu). Such a spatially explicit inventory (of biomass) might be used to estimate future supply for mitigation activities aimed at enhancing forest carbon sinks and contributing toward net negative emissions.

9.4 Peatlands

Boreal peatlands in Canada contain carbon stocks of about 99 000 Tg (Tarnocai 2006). The Peatlands of Canada Database estimates there to be 72.6 Mha of peatland in Canada's boreal regions (Tarnocai 2006). Thus, the average carbon density of peatlands in Canada's boreal regions would be 1364 Mg $C \cdot ha^{-1}$. Model simulations of boreal peatlands at a study site within the southern Mackenzie River basin in northeast Alberta, an ecosystem similar to southern Northwest Territories, estimates carbon stocks to be 530–1650 $Mg \cdot ha^{-1}$ (Beilman et al. 2008).

Rates of carbon sequestration in peatlands in Canada have been estimated at 88 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ over the Holocene period (the past 11 650 years) (Vasander and Kettunen 2006) and 73–110 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ over the past 6000–8000 years (Webster et al. 2018). However, it is uncertain whether these areas are still accumulating carbon today (Wieder et al. 2009). In particular, although peat plateaus have historically accumulated large amounts of carbon, studies have shown that sink capabilities are now much lower and peat plateaus may fluctuate between being small sinks and being sources from year to year. These observations imply either that warming temperatures have already changed the carbon dynamics of peat plateaus or that the average plateau has reached a mature and stable stage in development (Startsev et al. 2016). Collapse scars, by contrast, have high CO_2 sequestration rates because of greater *Sphagnum* growth under waterlogged conditions, and these features may constitute the greatest CO_2 sink in the north (Startsev et al. 2016).

Wetlands are the largest natural source of CH_4 to the atmosphere, contributing about a third of the global total, through anaerobic decomposition under saturated conditions (Helbig et al. 2017b). According to the latest estimates, which use the global warming potential conversion metric, CH_4 has a greenhouse effect 28 times greater than that of CO_2 over a 100-year period (Myhre et al. 2013). Thus, even if peatlands remain a net carbon sink, the fact that wetland portions emit CH_4 means these landforms may still be a net contributor of greenhouse gases, measured as the CO_2 equivalent (CO_2^e) (where conversion from CH_4 to CO_2^e entails multiplying the value for CH_4 emissions by 28).

Boreal peatlands currently contribute about 20% of CH_4 emissions from global wetlands. That this proportion is not higher is due to low temperatures limiting microbial decomposition. As temperatures increase in the boreal region and thawing permafrost creates more widespread saturated conditions, while also encouraging the growth of easily decomposable sedges, CH_4 emissions will likely rise (Helbig et al. 2017b).

9.4.1 CO_2 Flux by Landform

Helbig et al. (2017a) measured the CO_2 flux of a collapse scar in the Scotty Creek watershed (Taiga Plains Mid-Boreal ecoregion) between June 2015 and August 2016 using a 1.9-m eddy covariance tower and found a sequestration rate of 86 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$. Startsev et al. (2016) also found a sequestration rate of 86 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ for a collapse scar near Fort Simpson (Taiga Plains High Boreal ecoregion) using a portable gas analyzer during the summers of 2008–2010 (Table 26). Although Startsev et al. (2016) measured flux only in the growing season, winter fluxes at the site were believed to be negligible. However, Webster et al. (2018) estimated Canada's peatland NEE for days outside the growing season using an average rate (calculated from six different studies) of 0.9 g $CO_2 \cdot m^{-2} \cdot day^{-1}$. Eventually, Canada's peatland NEE for days outside the growing season amounted to 184.56 g $CO_2 \cdot m^{-2}$ each winter, while peatland NEE for the growing season amounted to −185.81 g $CO_2 \cdot m^{-2}$ per season (Webster et al. 2018). The sequestration rate increased along a latitudinal gradient, with a site in the Low Subarctic ecoregion sequestering about 190 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ and a site in the High Subarctic ecoregion about 312 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ (Startsev et al. 2016).

Helbig et al. (2017a) also measured the CO_2 flux for a general landscape unit at Scotty Creek that included both a forested peat plateau and a collapse scar wetland between March 2015 and August 2016 using a 15.2-m eddy covariance tower. The landscape unit was shown to sequester an average of 74 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$. The wetland and forested peat plateau contributed equally to the landscape flux, so the values for the two landscape units were averaged in the final calculation of CO_2 flux for the landscape unit (Helbig et al. 2017a). Thus, a forested peat plateau may be said to sequester 62 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ ([86 + 62]/2 = 74). In any case, Helbig et al. (2017a) concluded, from the small difference in CO_2 flux between the collapse scar and the landscape unit, that permafrost thaw may have a negligible effect on CO_2 flux. In contrast, Startsev et al. (2016) noted that most studies have shown peat plateaus to vary from year to year as minor CO_2 sinks or sources, whereas collapse scars act as significant CO_2 sinks (Table 26).

9.4.2 CH_4 Flux by Landform

Helbig et al. (2017b) measured the CH_4 emissions of a collapse scar bog in the Scotty Creek watershed (Taiga Plains Mid-Boreal ecoregion) between April 2014 and May 2016 using a 2-m eddy covariance tower and found emissions of 13 g $CH_4 \cdot m^{-2}$ per growing season. Liblik et al. (1997) used a static chamber method and found an average CH_4 emission rate of 13.5 g $CH_4 \cdot m^{-2}$ per growing season for four collapse bogs north of Fort Simpson.

Peat plateaus were not directly measured for CH_4 emissions by Helbig et al. (2017b); instead, a larger landscape unit, which included both a forested peat plateau and a collapse scar wetland, was measured between May 2013 and May 2016 by a 15-m eddy covariance tower, and a rate of 7 g $CH_4 \cdot m^{-2}$ per growing season was observed. Linear regression analysis revealed that most emissions in the general landscape were from wetlands, with peat plateaus contributing little, whereas other studies have shown forested peatlands to be either small sinks or sources of CH_4 (Table 26) (Helbig et al. 2017b). Liblik et al. (1997) found an average sequestration rate of 0.06 g $CH_4 \cdot m^{-2}$ per growing season for three sites on a peat plateau neat Fort Simpson.

Table 26. Greenhouse gas fluxes[a] for peatland landforms in the discontinuous permafrost zone of the Northwest Territories (Scotty Creek watershed and Fort Simpson)[b]. (Data sources Liblik et al. [1997], Startsev et al. [2016], Helbig et al. [2017a], Helbig et al. [2017b].)

Landform	CO_2 (g $CO_2 \cdot m^{-2} \cdot y^{-1}$)	CH_4 (g $CH_4 \cdot m^{-2} \cdot y^{-1}$)	Net (g $CO_2^{e} \cdot m^{-2} \cdot y^{-1}$)
Peat plateau	0/−62[c]	0	0/−62[c]
Collapse scar bog	−86	+16	+362
Poor fen	n/a[d]	+22.1 per growing season	n/a
Rich fen			
Graminoid	n/a	+8.4 per growing season	n/a
Low shrub	n/a	+2.5 per growing season	n/a
Treed	n/a	+0.5 per growing season	n/a
Landscape unit (Scotty Creek)	−74	+8.5	+164

[a] *CO_2 = carbon dioxide, CH_4 = methane, CO_2^{e} = carbon dioxide equivalent.*

[b] *Positive sign denotes emissions, negative sign denotes sequestration.*

[c] *indicates variable results across studies*

[d] *n/a = not available.*

These CH_4 emission rates did not include winter emissions, which may contribute more than 10% of annual emissions (Helbig et al. 2017b). Helbig et al. (2017b) estimated that emissions were 3 g $CH_4 \cdot m^{-2}$ per winter season (November–April) in wetlands and 1.5 g $CH_4 \cdot m^{-2}$ per winter season over the larger landscape, for a total of 16 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$ (13 g $CH_4 \cdot m^{-2}$ per growing season + 3 g $CH_4 \cdot m^{-2}$ per winter season) and 8.5 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$ (7 g $CH_4 \cdot m^{-2}$ per growing season + 1.5 g $CH_4 \cdot m^{-2}$ per winter season), respectively (Table 26).

In addition to the collapse scars and peat plateaus mentioned above, Liblik et al. (1997) also measured the CH_4 flux in other wetland categories along the Mackenzie Valley near Fort Simpson during the summer of 1995 using a static chamber method (Table 26). Four open poor-fens had an average emission rate of 22.1 g $CH_4 \cdot m^{-2}$ per growing season. Four open rich-fens dominated by graminoids had an average emission rate of 8.4 g $CH_4 \cdot m^{-2}$ per growing season. Four open rich-fens dominated by low shrubs had an average emission rate of 2.5 g $CH_4 \cdot m^{-2}$ per growing season. Two treed rich-fens had an average emission rate of 0.5 g $CH_4 \cdot m^{-2}$ per growing season.

9.4.3 Net Landscape Emissions

As mentioned above, Helbig et al. (2017a) found that the larger peatland landscape at Scotty Creek incorporating both peat plateaus and collapse scars sequestered 74 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ and emitted 8.5 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$. The net greenhouse gas emissions for that landscape would therefore be about 164 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$ ([8.5 × 28] – 74) (Table 26). CO_2 sequestration rates of more than 200 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ would be necessary to balance the rates of CH_4 emissions. However, the long-term rates of CO_2 sequestration for boreal peatlands similar to that in the Scotty Creek watershed range from 49 to 157 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ (Helbig et al. 2017a).

9.4.4 Climate Change and Permafrost Thaw

Helbig et al. (2017b) found that CH_4 emissions increased with warming soil temperatures. Thus, warming temperatures would increase CH_4 emission rates not only by transforming peat plateaus to collapse scars, but also by increasing emission rates within collapse scars (Figure 3, Figure 4). They found that emissions were only weakly correlated with water table depth once the correlation between warmer temperatures and lower water table depth was taken into account. However, this would only be the case so long as the water table is maintained completely above the rooting zone. Once the water table falls below this zone, root exudates can decompose aerobically and would no longer produce CH_4 (Helbig et al. 2017b). Thus, Liblik et al. (1997) found that water table depth was the strongest driver of CH_4 emissions, with a strong positive correlation between water tables and emissions. In fact, they found a positive correlation between temperature and CH_4 emissions only when water table position was taken into account (Liblik et al. 1997). Thus, the impacts of widespread permafrost thaw on drainage, and subsequently on the water table depth, will likely influence the levels of CH_4 emissions. Some models predict that permafrost thaw will cause lower CH_4 emissions due to improved drainage over the landscape, although ground subsidence from thaw may bring the water table closer to the surface, which would counteract the effects of drainage (Helbig et al. 2017b). Nevertheless, these effects should be considered transitory responses to permafrost thawing, with the expectation that eventually the system will transition to a different plant community (sedge or moss), where productivity will increase and offset the CH_4 emissions from permafrost thawing.

As noted above (section 9.4.1), peat plateaus in the Taiga Plains Mid-Boreal ecoregion have a CO_2 sequestration rate of about 62 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ and a negligible CH_4 emission rate, for a net sequestration rate of 62 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$. Collapse scar bogs in the Taiga Plains Mid-Boreal ecoregion have a sequestration rate of about 86 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ and an emission rate of about 16 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$. The net emission rate of collapse scars would therefore be about 362 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$ ([16 × 28] – 86). A complete thaw of permafrost that converts a section of peat plateau to collapse scar bog would therefore create a net increase in emissions of about 424 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$ (362 + 62). However, if peat plateaus are carbon neutral, as stated by Startsev et al. (2016), then the net increase in emissions from the conversion of peat plateau to collapse bog would be only 362 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$.

Conversion of peat plateau to collapse scar in the Scotty Creek area was estimated to contribute additional CH_4 emissions, averaged out over the entire watershed, of about 0.034 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$ (0.95 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$) (Helbig et al. 2017b). Because the conversion of peat plateau to collapse scar causes an increase of 16 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$ at the location of conversion, the average effect of permafrost thaw over the entire watershed would be 0.21% ([0.034/16] × 100) of the local effect at the place of thaw. This percentage value can be used to translate the effects of peat plateau conversion on CO_2 sequestration to the entire watershed. Thus, peat plateau thaw would increase CO_2 sequestration over the entire watershed by 0.05 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ (0.0021 × [86 – 62 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$]) or by 0.18 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ (0.0021 × [86 – 0]) if peat plateaus are carbon neutral. The landscape-wide increase in emissions due to the conversion of peat plateaus to collapse scars in the Scotty Creek watershed would then average 0.77–0.90 g $CO_2^{e} \cdot m^{-2} \cdot yr^{-1}$ (0.95 g $CO_2^{e} \cdot m^{-2} \cdot yr^{-1}$ – 0.05 to 0.18 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$).

9.5 Effects of Fire on Carbon Balance

Wildfires have the greatest impact of all disturbances on the carbon cycle in the boreal forest (Wieder et al. 2009) (Figure 3, Figure 4). The NFCMARS of the CFS incorporates the Fire Monitoring, Accounting and Reporting System (FireMARS), which estimates carbon emissions from wildfires in Canada. The NFCMARS calculates carbon emissions as the product of area burned, fuel load, fraction of fuel load consumed, and an emission factor. The area burned is derived from the NBAC map. The fraction of fuel load consumed by fire is derived from a combination of fire weather information from the Canadian Wildland Fire Information System and the CanFIRE fire behavior and impacts model. The emissions factor is set at 0.5 t of carbon per tonne of oven-dry biomass, the same as in the CBM-CFS3 (Anderson et al. 2015).

One of the greatest uncertainties in calculating carbon emissions from fires is the fuel load and vegetation classes of the area burned (Anderson et al. 2015). Currently, NFCMARS uses an average fuel load for the level II ecoregion where the fire took place and does not link the vegetation classes of a specific area to the location of the fire (S. Kull, Natural Resources Canada, personal communication). Anderson et al. (2015) compared fuel load data in the NFCMARS to three spatial fuel maps in an effort to determine whether finer-scale fuel maps matched to the fire location would improve estimates of carbon emissions. The three maps used for this analysis were the Canadian national fuel map (1-km resolution) (Nadeau et al. 2005), regional data sets from Parks Canada (Landsat images, 30-m resolution), and the EOSD land cover map. The EOSD map was also combined with both the national and regional maps, for a total of five fuel type maps. The EOSD map had a finer scale than the national or regional maps and was better able to differentiate open from closed spaces. Open spaces have a much lower fuel load than closed forest, and differentiating between them should increase the accuracy of emissions estimates. The national and regional maps differentiated between conifer species, which the EOSD map did not. Black spruce stands have a higher fuel load than jack pine stands, and black spruce trees often have low-hanging branches that act as a ladder to crown fires, whereas jack pines have high crowns. Thus, combining the finer-scale EOSD land cover map and maps using forest inventory information that differentiates among conifer species should yield the best estimates (Anderson et al. 2015).

The fuel maps were overlaid first with daily fire progression maps, to assess how much of each fuel type burned on each day, and then with fire conditions derived from daily fire weather, to assess how those fuel types would have burned. The model was then applied to six large fires in the western boreal forest, including a 2004 fire in Wood Buffalo National Park in southern Northwest Territories that burned 51 968 ha. The fuel type burned in the Wood Buffalo fire differed substantially among the various maps. According to the national map, 99% of the burn area was composed of boreal spruce (C-2 fuel type) and 1% of leafless aspen (D-1 fuel type). The EOSD map showed 72% boreal spruce, 2% leafless aspen, 6% leafless mixed-wood (M-1 fuel type), and 21% grassland (O-1 fuel type). The regional map showed 29% boreal spruce, 60% mature jack pine (C-3 fuel type), 1% immature jack pine (C-4 fuel type), 1% leafless aspen, 7% leafless mixed-wood, and 3% grassland (Anderson et al. 2015). As a result of these differences in modeled land cover, carbon emissions for the 2004 Wood Buffalo fire based on the three maps differed substantially, from 1.7 kg $C \cdot m^{-2}$ according to the combined EOSD and regional maps to 3.8 kg $C \cdot m^{-2}$ according to the Canadian national fuel map. Total estimated emissions over the entire burn area ranged from 882 000 Mg of carbon according to the combined EOSD and regional maps to 1 971 000 Mg of carbon according to the national map (Anderson et al. 2015).

For all maps, the standard deviations of the estimates were much less than that of the estimate obtained with current NFCMARS methods, illustrating the increased precision of the methods used by Anderson et al. (2015). However, the large differences among the estimates based on the five maps illustrates the necessity of acquiring accurate and consistent fuel maps that combine high-resolution land cover images with accurate identification of tree species (Anderson et al. 2015).

In 2014, large fires in the Northwest Territories burned 2.81–3.57 Mha, more than eight times the annual mean for the area. Walker et al. (2018), as part of the Arctic-Boreal Vulnerability Experiment, sponsored by the US National Aeronautics and Space Administration, measured the carbon emissions from seven of those fires within the Taiga Plains and Taiga Shield ecoregions in the Yellowknife area using a combination of ground sites and remote sensing data. The researchers accounted for spatial heterogeneity by means of stratified random sampling using land cover characteristics, sampling of plots within the sites along a moisture regime gradient, and use of remote sensing images at the same scale as the ground plots. Models were used to extrapolate from plot emissions to the entire 2014 burn area according to covariates associated with landform, vegetation, and fire characteristics, all of which were shown to have predictive value (Walker et al. 2018). The estimated total emissions from the 2014 fires were 346 Tg CO_2, an amount equal to almost 50% of the mean annual NEP of all forests in Canada (Walker et al. 2018).

Almost 90% of these emissions were from the soil organic layer, mainly due to the flammability of black spruce stands, which also tend to have a particularly thick organic soil layer (Walker et al. 2018). The results indicate that carbon emissions were affected most by more stable characteristics, such as soil moisture and the relative abundance of black spruce, rather than by temporary factors, such as fire weather or date of burn. For example, black spruce–dominated stands with intermediate drainage contributed the most emissions relative to other stand and drainage types. Areas with intermediate drainage had the ideal combination of soil organic layer depth and combustibility, whereas drier sites had a thinner soil organic layer, and less of the organic layer burned at wetter sites (Walker et al. 2018). Black spruce–dominated stands have a thicker soil organic layer than jack pine stands; therefore, mean CO_2 emissions in jack pine stands amounted to only 2.6 kg $CO_2 \cdot m^{-2}$, 69% of which was from the soil organic layer, whereas mean CO_2 emissions in black spruce stands amounted to 14.2 kg $CO_2 \cdot m^{-2}$, 90.5% of which was from the soil organic layer (Walker et al. 2018).

Given that vegetation cover and soil drainage are characteristics that show great variability at a fine scale, these results show the importance of accounting for landscape heterogeneity. Fire weather and date of burn have elsewhere been used to successfully predict emissions from wildfires, but the fact that the fires in this analysis took place over a short time span and within an extreme fire season probably explains why those factors were of less importance in this case (Walker et al. 2018).

Using spatial modeling, Walker et al. (2018) estimated the area burned to be 2.85 Mha, similar to the estimate of 2.81 Mha generated from the NBAC database. Other estimates used data with coarser resolution and would have included bodies of water and other unburned areas smaller than a 500-m pixel; the resulting estimates were higher, at 3.11–3.57 Mha (van der Werf et al. 2017; Veraverbeke et al. 2017; Walker et al. 2018). This demonstrates the effectiveness of higher-resolution mapping (Walker et al. 2018). However, this study did have high uncertainty at the pixel level. Because soil moisture and prefire tree species were the most important emission predictors at the site level, improving spatial layer accuracy would greatly improve the model. Nonetheless, the total emissions estimate had much lower uncertainty, which shows that field sampling was representative and that modeling at the landscape level is more robust than for any given pixel (Walker et al. 2018).

9.6 Effects of Insect Outbreaks on Carbon Balance

Insect outbreaks affect the carbon balance in forests by defoliating trees, thereby reducing photosynthesis; by killing trees, thereby increasing the decomposition of DOM; and by changing forest age-class and structure (Fleming et al. 2002; Kurz et al. 2008) (Figure 3, Figure 4). The insects with the greatest effect on carbon emissions from Canada's forests are the spruce budworm and mountain pine beetle. During outbreaks, these insects can shift the entire forest carbon budget toward being a greater source. However, their effects on carbon emissions are indirect and therefore difficult to model (Kurz et al. 2008). The latest carbon balance for Canada's managed forest, obtained with the CBM-CFS3, accounted for the effects on tree growth and mortality from mountain pine beetle, spruce beetle (*Dendroctonus rufipennis*), eastern hemlock looper (*Lambdina fiscellaria*), forest tent caterpillar, and large aspen tortrix (*Choristoneura conflictana*), but not from the spruce budworm (Stinson et al. 2011).

9.7 Effects of Temperature Increase on Carbon Balance

Helbig et al. (2017a) found that at the landscape scale in the Scotty Creek watershed, CO_2 sequestration began at a mean daily temperature of 2°C and reached a maximum at a mean daily temperature of 15°C. At nearby Fort Simpson, about 50% of the days each year have a mean temperature below 2°C and about 15% of the days have a mean temperature above 15°C. As temperatures warm, the number of days above 2°C will rise, and more days will have a mean daily temperature that rises toward 15°C, increasing productivity. At the same time, the number of days with a mean daily temperature above 15°C will also rise, decreasing productivity (Helbig et al. 2017a).

Landscape productivity and CO_2 sequestration increase with warmer temperatures only up to a point and then decrease with further warming, for several reasons. First, warmer temperatures increase the rate of autotrophic respiration in trees, whereby carbohydrate reserves are burned and CO_2 is emitted (Helbig et al. 2017a). Second, evapotranspiration increases with warmer temperatures, which increases the potential for water deficits that limit tree growth (Helbig et al. 2017a). Third, at a certain point, photosynthesis in the boreal forest switches from being limited by temperature to being limited by light, in which case further increases in temperature cannot lead to increased tree growth without a concurrent increase in light availability. Photosynthesis is limited by light more often in the fall than in the spring, and the latter season therefore sees greater productivity gains from increasing temperatures (Helbig et al. 2017a).

Trees do have the ability to acclimate their rates of respiration to warming temperatures. For example, acclimated boreal tree species (specifically, balsam fir, white spruce, jack pine, paper birch, and trembling aspen) in the boreal–temperate ecotone of Minnesota increased leaf respiration by an average of 5% across species when the temperature was experimentally increased from 20°C to 23.4°C, and by 6% under natural temporal variation of those same temperature shifts, whereas nonacclimated trees increased respiration by 23% under the experimental conditions and by 25% under natural variation. Thus, acclimation eliminated almost 80% of the increase in respiration under both experimental and natural conditions, with greater acclimation among conifer species than broadleaf species (Reich et al. 2016). However, whether respiration acclimation can help trees to maintain their carbon balance in the northern boreal forest remains unknown (Girardin et al. 2016b).

Temperature also influences CH_4 emissions. For example, Helbig et al. (2017c) found that the mean temperature in May, the month when snow melt is completed, strongly affected seasonal CH_4 emissions in permafrost-free wetlands in the Scotty Creek watershed. The mean May temperatures in 2013, 2015, and 2016 were 2.4°C to 4.1°C higher than in 2014, which had a mean May temperature equivalent to the long-term (1951–2010) mean. Subsequently, the mean soil temperatures in June were 6°C warmer in the warmer years than in 2014, and emissions of CH_4 during May–October were then about 42% higher in the warmer years than in 2014 (Helbig et al. 2017c). Half of the increase was generated between May and June and the other half between July and October. The higher July and August emissions occured despite mean soil temperatures during those months being similar across all years. The increased emissions later in the summer likely resulted from an increase in vegetation productivity during the years with a warmer spring, which contributed to greater amounts of decomposition under anaerobic conditions later in the season (Helbig et al. 2017c).

Key Points

- The carbon balance of 30% of upland forests of the Northwest Territories was estimated using the Carbon Budget Model of the Canadian Forest Sector, developed by CFS. Forests in the Taiga Plains had a total of 3000 Tg of stored carbon and a density of 150 Mg $C \cdot ha^{-1}$ and were estimated to have a carbon dioxide (CO_2) sequestration rate of 73–147 g CO_2 $m^{-2} \cdot yr^{-1}$. However, a limited and stable amount of carbon in the soil suggests that these forests have been historically carbon neutral, with canopy sequestration through photosynthesis equaling emissions from ground-surface respiration.
- Carbon estimation with the Carbon Budget Model of the Canadian Forest Sector is being improved with the MOSS-C module, designed for upland forests with a thick organic layer, and the Canadian Model for Peatlands (CaMP) module, designed for peatlands.
- Peat plateaus are believed to be CO_2 neutral, with tree sequestration equaling surface emissions, although some researchers have estimated a CO_2 sequestration rate of 62 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$. Methane ($CH_4$) emissions from peat plateaus are believed to be negligible.
- Collapse scars sequester CO_2 and emit CH_4. Sequestration rates of CO_2 on collapse scars were measured as 86 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$ in the Taiga Plains High Boreal and Mid-Boreal ecoregions. Emissions of CH_4 from collapse scars in the Taiga Plains Mid-Boreal ecoregion were measured as 16 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$, or 448 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$. If peat plateaus are CO_2 and CH_4 neutral, then the increase in emissions from a transition of peat plateau to collapse scar in the Taiga Plains Mid-Boreal ecoregion can be estimated as 362 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$. The landscape-wide increase in emissions from permafrost thaw in the Scotty Creek area is estimated to be 0.77 g $CO_2^e \cdot m^{-2} \cdot yr^{-1}$.
- Wildfires contribute more carbon emissions than any other disturbance. The 2014 fires in the Yellowknife area were estimated to emit 346 Tg CO_2, almost half the amount of the Net Ecosystem Production of all Canada's forests. Almost 90% of the emissions were derived from the soil organic layer because of the flammability of black spruce forests, which also tend to have a thick organic soil layer.
- Maximum CO_2 sequestration over the entire landscape in the Scotty Creek watershed is reached at a mean daily temperature of 15°C. Increasing temperatures have been found to increase sequestration early in the summer and decrease sequestration later in the summer.

Knowledge Gaps

- Mapping of peatland types and permafrost features in the Northwest Territories has been limited.
- The peatland carbon balance in the Northwest Territories has been measured only at the plot level, and plots are placed only along major highways. Studies at a larger landscape level are necessary.
- The Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) has large gaps related to inter-annual and site-specific responses to disturbance and changing environmental conditions, including temperature and precipitation regimes, increasing CO_2 concentration, and deposition of nitrogen.
- The CBM-CFS3 is currently suitable only for managed forests that have inventory data, which describes about 30% of upland forests in the Northwest Territories, including virtually all peatland forests.
- Thawed permafrost may act as a net carbon sink because of greater rates of carbon accumulation in wetlands, or it may act as a net source if previously frozen carbon deposits begin to degrade when thawed. Which scenario occurs may depend on whether enough of the discontinuous permafrost landscape thaws to enable wetlands to drain, causing accumulated peat to sit above the water.
- Field measurements of CO_2 and CH_4 fluxes are lacking from the western part of the Taiga Shield ecoregion, a significant portion of which exists in the Northwest Territories.
- The CO_2 and CH_4 balance of channel fens are generally poorly studied. In addition, knowledge about winter emissions of CO_2 and CH_4 is limited.
- The greatest uncertainty in estimating the emissions from fires is the fuel load at the burned sites.

CHAPTER 10. FOREST CARIBOU HABITAT

10.1 Introduction

Caribou in the Northwest Territories comprise five ecological types (Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2011), two of which spend at least some of the year within the boreal forest: migratory barren-ground caribou (*Rangifer tarandus groenlandicus*), found east of the Mackenzie River in the Taiga Plains, Taiga Shield, and Southern Arctic ecoregions, and nonmigratory boreal woodland caribou (*Rangifer tarandus caribou*), found throughout the Taiga Plains ecoregion (Gunn et al. 2011; ESTR Secretariat 2013).

10.2 Barren-Ground Caribou

Barren-ground caribou are distinguished by herds that are based on geographic distribution. Calving grounds and summer ranges are distinct for each herd. Collared cows normally return year after year to calving grounds with 95%–98% fidelity, and herds are named on the basis of calving ground locations. Winter ranges are generally larger and more variable from year to year, and they often overlap with those of neighboring herds. Within the Northwest Territories, most of the barren-ground caribou herds spend the calving season in the Southern Arctic ecoregion and the winter farther south in the Taiga Plains and Taiga Shield ecoregions (Figure 54), although some of the herds, especially the smaller ones for which ranges have contracted, may winter on the tundra as well (Gunn et al. 2011).

The level of knowledge regarding distribution and population trends differs among herds because of uneven monitoring, research, methodologies, and access to past surveys. In general, northern caribou numbers were low through the 1970s, increased until the mid-1990s, and have declined since then, with some herds stabilizing (Figure 55, Figure 56, Figure 57) (Gunn et al. 2011). Barren-ground caribou populations are naturally cyclic, driven by decadal climate patterns, forage availability, predation, and pathogens. Current population trends are likely the result of a combination of the effects of these natural cycles, increasing human presence in the caribou ranges, and climate change (Gunn et al. 2011). The effects of climate change on caribou populations are likely multiple, because weather affects caribou in all seasons (Mallory et al. 2018). For example, there are well-documented relationships whereby a warmer climate and longer growing seasons result in increased availability of summer forage plants, which is associated with better caribou body condition going into the winter months and higher winter survival rates (Mallory et al. 2018). However, warmer summer weather has also been demonstrated to increase parasitic insect activity, which can directly decrease caribou body condition (Mallory and Boyce 2018). Additionally, the parasitic insects can harass caribou to the point where they stop feeding during the summer months and instead focus their energy on trying to escape insect swarms, which can reduce their body condition going into the winter months, thus reducing or eliminating the potential positive impacts of increased summer forage availability (Mallory and Boyce 2018). As such, the impacts of climate change on caribou are complex and varied and will likely be different in different parts of this species' range.

Eight barren-ground caribou herds in the Northwest Territories have at least part of their range below the tree line: Bathurst, Cape Bathurst, Bluenose-West, Bluenose-East, Beverly, Qamanirjuaq, Ahiak, and Porcupine. Summarized below is the most current information regarding the state of each of these herds. The length and depth of monitoring, such as population surveys, use of satellite collars, and surveys to assess calf survival and sex ratios, have varied among the herds, with the Bathurst herd having generally been monitored the most closely and over the longest interval.

Bathurst: The Bathurst herd has historically wintered in the Northwest Territories from the eastern and southern shores of Great Bear Lake to the Saskatchewan border, with a core habitat area in the Taiga Shield ecoregion northeast of Yellowknife (Figure 54) (Department of Environment and Natural Resources [N.W.T.] 2018b). Photographic surveys of calving grounds near Bathurst Inlet, Nunavut, show that the herd has decreased to about 8200 individuals from 19 769 in 2015 and from a high of about 470 000 in the mid-1980s and now winters near the tree line or on the tundra (Figure 55, Figure 57) (Adamczewski et al. 2019). Traditional knowledge of Tłįchǫ elders and estimates of caribou abundance based on spruce

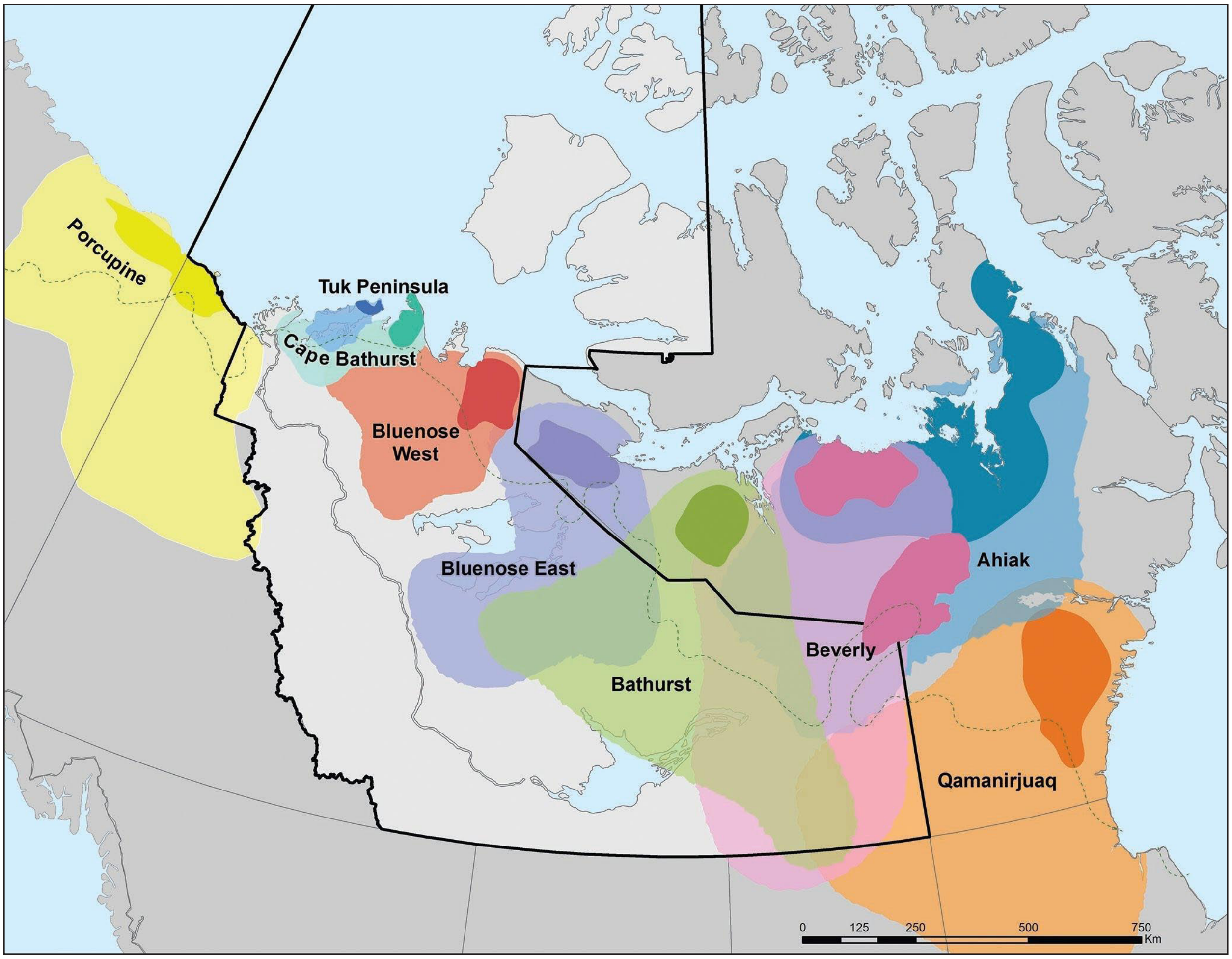


Figure 54. Historical barren-ground caribou herd ranges and calving areas in the Northwest Territories, 1996–2018. (Source: Government of Northwest Territories, Department of Environment and Natural Resources, https://www.enr.gov.nt.ca/en/services/barren-ground-caribou [accessed 4 February 2022].)

root scars on caribou migration trails indicate that the Bathurst herd previously reached high numbers in the 1940s, with low numbers before and after this peak (Zalatan et al. 2006). Commercially guided and resident hunting has been suspended on this herd's range since 2010, and the Indigenous hunt has been closed since winter 2015. A Bathurst Caribou Range Plan based on scientific and traditional knowledge was prepared between 2014 and 2018 through a collaborative process with multiple partners to define limits on industrial development across the herd's range (Department of Environment and Natural Resources [N.W.T.] 2018b). Since 2015, a mobile Bathurst Mobile Conservation Core Area has been created each winter, as a way of enforcing a zero harvest in the Northwest Territories. The mobile zone is defined by the locations of all satellite-collared Bathurst caribou, with a buffer; within this zone, no caribou harvest is allowed (Department of Environment and Natural Resources [N.W.T.] 2016b). An overall management plan for the Bathurst herd, under the collaborative multi-partner Bathurst Caribou Advisory Committee, was released in late 2020.

Cape Bathurst: The winter range of the Cape Bathurst herd extends into the Taiga Plains High Subarctic ecoregion east of the Mackenzie Delta (Figure 54). The herd has decreased from a high of more than 19 000 individuals in the 1990s to a stable population of 1900–2600 individuals between 2005 and 2015 (Figure 55). In 2018, the herd appeared to have increased to 4500 individuals (Department of Environment and Natural Resources [N.W.T.] 2016c). Hunting has been banned since 2007 as result of recommendations from the Wildlife Management Advisory Council (Department of Environment and Natural Resources [N.W.T.] 2016c).

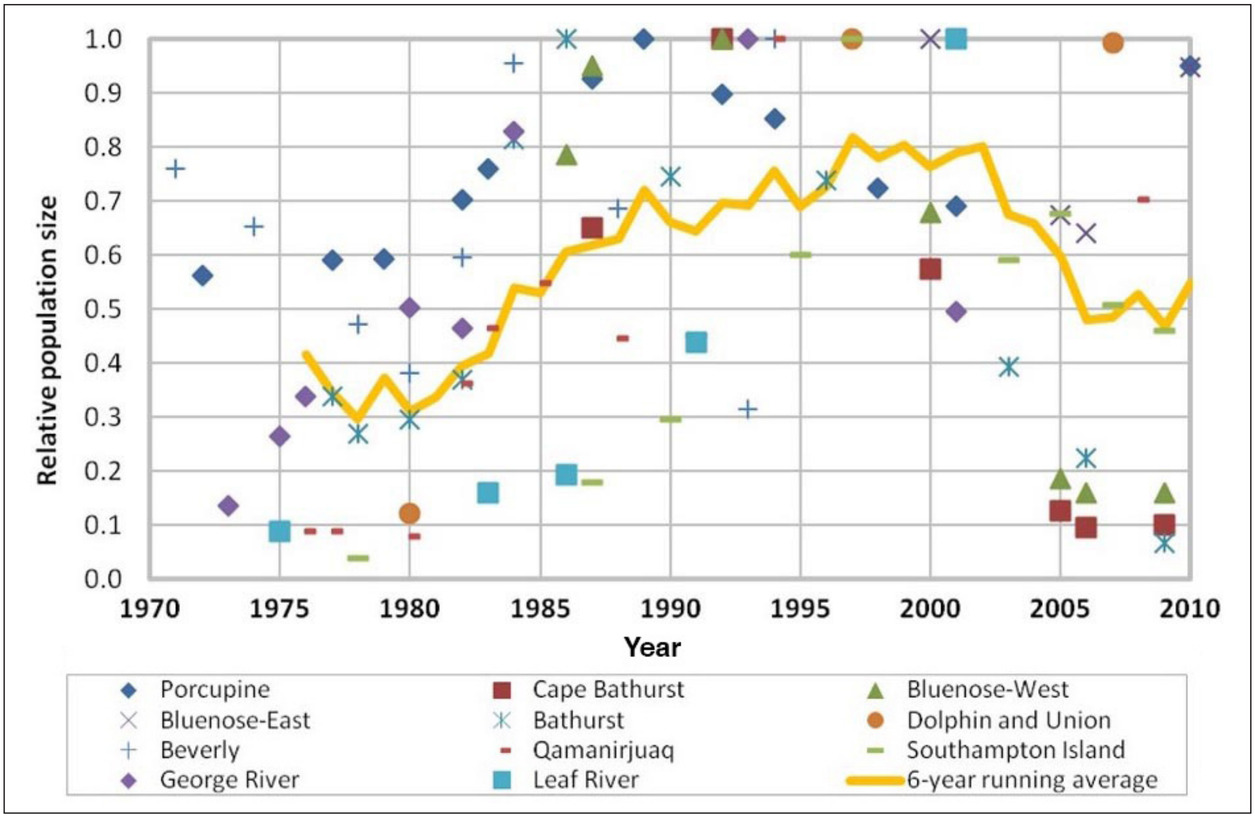


Figure 55. Relative population trends of barren-ground caribou herds in Canada, 1970–2010. Relative population size is the proportion of the maximum recorded estimate. Note that the maximum recorded estimate is not necessarily the peak population over this timeframe, because surveys usually did not cover the entire period and were not conducted every year (Reproduced, with permission, from Gunn et al. [2011].)

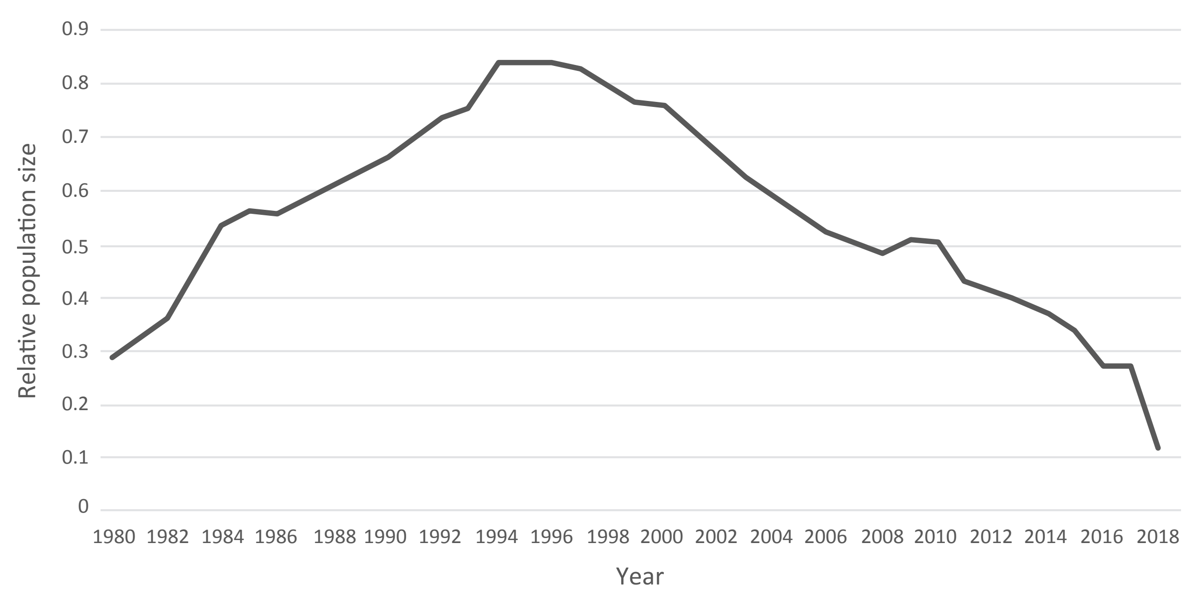


Figure 56. Average relative population size for all barren-ground caribou herds in Canada, 1980–2018. Relative population size is the proportion of the maximum recorded estimate. Graph created by D. Russell, Yukon College, using CARMA data. (carma.caff.is [accessed 15 May 2020].)

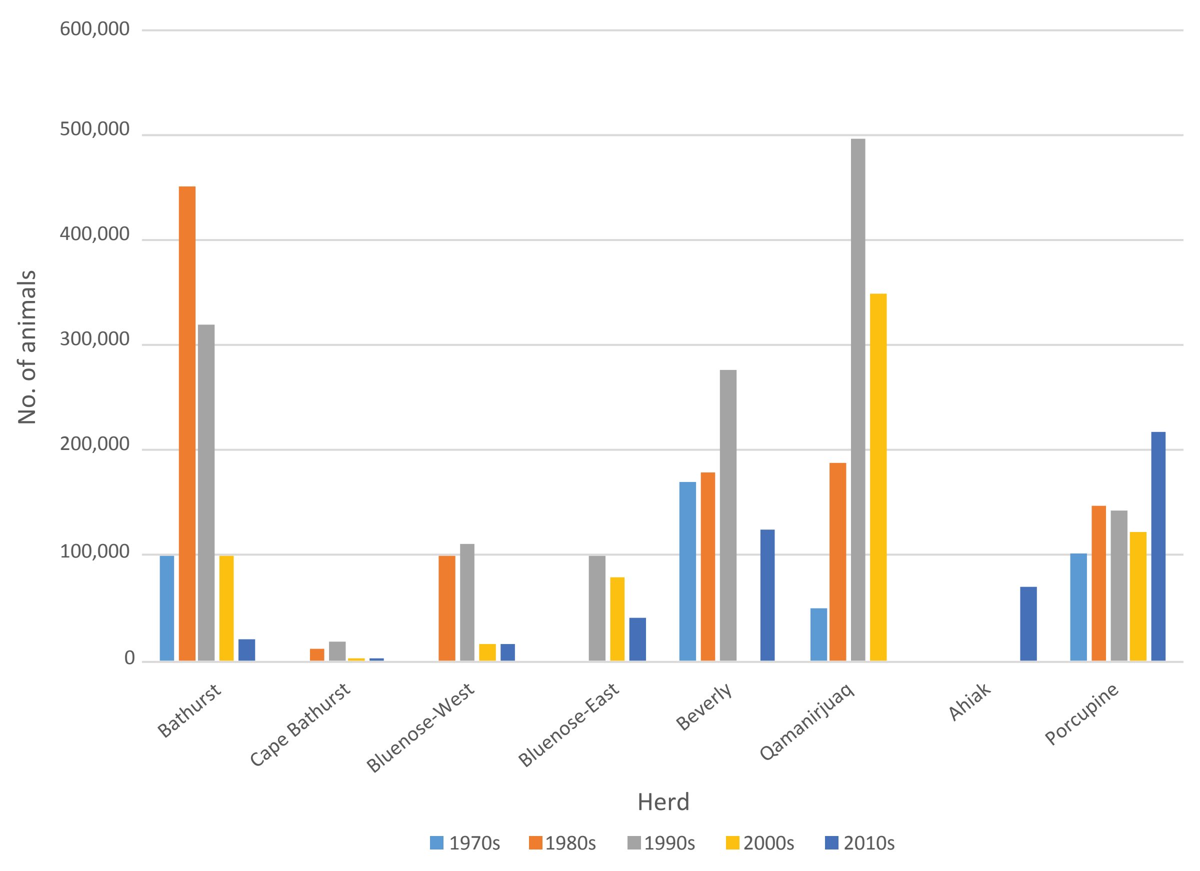


Figure 57. Population trends for barren-ground caribou herds in the Northwest Territories. Missing decades for particular herds indicate lack of surveys. Graph created with data from Beverly and Qamanirjuaq Caribou Management Board (2014); Department of Environment and Natural Resources [N.W.T.] (2016a); Department of Environment and Natural Resources [N.W.T.] (2016c); Porcupine Caribou Technical Committee (2016); Porcupine Caribou Management Board (2018); Adamczewski et al. (2019); Boulanger et al. (2019); Campbell et al. (2019).

Bluenose-West: The winter range of the Bluenose-West herd extends into the Taiga Plains Low Subarctic and High Subarctic ecoregions north of Great Bear Lake (Figure 54). The herd has declined from highs of about 112 000 individuals in the 1990s to a stable population of about 20 000 between 2005 and 2018 (Figure 55) (Department of Environment and Natural Resources [N.W.T.] 2016c). Non-Indigenous hunting has been suspended since 2006, and Indigenous harvest is limited to 4% of the herd, 80% of which must be bulls, as a result of management recommendations from the Wildlife Management Advisory Council and the Gwich’in and Sahtú Renewable Resources Boards in 2006–2007 (Department of Environment and Natural Resources [N.W.T.] 2016c).

Bluenose-East: The winter range of the Bluenose-East herd extends from the Nunavut border east of Great Bear Lake to the Taiga Shield and Taiga Plains Low Subarctic ecoregions south of Great Bear Lake (Figure 54). The herd’s population was more than 100 000 in 2000 but had dropped to 19 300 individuals by 2018 (Figure 55) (Boulanger et al. 2019). The non-Indigenous hunt has been suspended since 2007. Restrictions on the Indigenous harvest were put in place in 2016, with limits on the number of caribou (and the percentage that must be bulls) set for each region.

A combined management plan for the Cape Bathurst, Bluenose-West, and Bluenose-East herds was produced in 2014 by a group of co-management boards (Advisory Committee for Cooperation on Wildlife Management 2014; Davison 2016). The boards of the Advisory Committee for Cooperation on Wildlife Management hold annual status meetings for the three herds to review new information and re-evaluate their management.

Beverly: In the Northwest Territories, the winter range of the Beverly herd extends from the Nunavut border northeast of Great Slave Lake to the Saskatchewan border within the Southern Arctic and Taiga Shield ecoregions (Figure 54). The latest population survey, in 2018, estimated 103 000 individuals (Campbell et al. 2019). The previous survey, in 1994, estimated 276 000 individuals, and little is known about the herd in the intervening period between these surveys (Figure 55, Figure 57) (Beverly and Qamanirjuaq Caribou Management Board 2014). As of 2014, hunting by Northwest Territories residents is limited to one bull each, with no restrictions for Indigenous peoples (Department of Environment and Natural Resources [N.W.T.] 2016a). A management plan for the Beverly and Qamanirjuaq herds for the period 2013–2022 has been created by the Beverly and Qamanirjuaq Caribou Management Board, whose membership consists of representatives of the federal government and relevant provincial and territorial governments, as well as community and Indigenous representitives (Beverly and Qamanirjuaq Caribou Management Board 2014).

Qamanirjuaq: The winter range of the Qamanirjuaq herd extends into the southeastern corner of the Northwest Territories (Figure 54). A 2017 survey estimated 280 000 individuals, down from 349 000 individuals estimated by a 2008 photographic survey and 496 000 individuals estimated by a 1994 survey (Figure 55, Figure 57) (Beverly and Qamanirjuaq Caribou Management Board 2014). As noted above, the Beverly and Qamanirjuaq Caribou Management Board has created a management plan for the Beverly and Qamanirjuaq herds for the period 2013–2022.

Ahiak: The Ahiak caribou migrate from Nunavut through the Thelon Wildlife Sanctuary into the Taiga Shield Low Subarctic and High Subarctic ecoregions northeast of Great Slave Lake (Figure 54). Their winter range thus overlaps with that of the Bathurst herd on the west of its range and with that of the Beverly and Qamanirjuaq herds on the east. Population trends are unknown, as there has been only one survey, in 2011, which showed their numbers to be about 71 000 (Figure 57) (Department of Environment and Natural Resources [N.W.T.] 2016a). As of 2014, the resident hunt in the Northwest Territories is limited to one bull each, with no restrictions on the Indigenous hunt (Department of Environment and Natural Resources [N.W.T.] 2016a).

Porcupine: Most of the Porcupine herd's range lies within Alaska and the Yukon, but the winter range extends across the Yukon border into the eastern portion of the Mackenzie Delta in the Northwest Territories (Figure 54). A photographic survey in 2017 estimated a record high population of 218 457 individuals, up from a 12-year low of 123 000 in 2001 (Figure 55, Figure 57) (Porcupine Caribou Technical Committee 2016; Porcupine Caribou Management Board 2018). The herd is co-managed by the International Porcupine Caribou Board and the Porcupine Caribou Management Board (Canada), and annual reports on the state of the herd are published (Department of Environment and Natural Resources [N.W.T.] 2016d). There is currently a tag limit of two bulls per licensed hunter (Porcupine Caribou Management Board 2018).

Barren-ground caribou wintering south of the tree line prefer a winter habitat of mature coniferous forest abundant with mat-forming lichens such as the reindeer lichens *Cladonia rangiferina* and *Cladonia mitis*. In addition to the lichen abundance in mature forests, the snowpack is also softer and less deep, making it easier for the animals to dig feeding craters to access the ground-dwelling lichens (Barrier and Johnson 2012).

A 2-year study of the Bathurst herd, conducted over 2008–2009, compared unburned sites (both used and unused by caribou) and recently burned sites. Less than 1% of recently burned sites were used by caribou, and these were all within their winter range in the boreal forest north and northeast of Yellowknife. Sites that were used had the greatest lichen volume and ground cover, although lichen biomass did not significantly differ between used and unused unburned sites. *Cladonia mitis* and *C. rangiferina* made up a combined 45% of total lichen volume on used sites. Lichen abundance at used sites may have been underestimated because of the caribou cropping or trampling the lichen mat (Barrier and Johnson 2012). Among the unburned sites, those that were used by caribou had an average stand age of 143 years whereas those not used by caribou had an average stand age of 96 years. Unused sites had greater tree density, as measured by basal area. Caribou were disproportionally found in windswept forest openings and ridges, where snow depth was shallower (Barrier and Johnson 2012). Burned sites had the lowest lichen volume, and most of that was comprised of *Cladonia* species not favored by caribou. Thus, an increase in wildfires may negatively affect barren-ground caribou winter foraging area. However, the Beverly herd has traditionally maintained a high reproductive rate and decent body condition despite a high-frequency fire regime on its winter range (Barrier and Johnson 2012).

With a fruticose (shrubby or bushy growth structure) lichen biomass of 2464 kg·ha^{-1} across an area of 240 186 km^2 of mature forest within the Bathurst herd winter range, it is estimated that the Bathurst winter range can support from 240 186 to 480 372 individuals (Barrier and Johnson 2012). The relationship between herd size and lichen availability, however, is complex and involves distribution, abundance in nearby non-traditional ranges, and availability of alternative forage (Barrier and Johnson 2012).

Results obtained by Barrier and Johnson (2012) show that caribou prefer older stands with a low tree volume and high ground cover of lichens. This finding accords with other studies of barren-ground caribou, including studies of the Beverly herd within the Northwest Territories (Thomas et al. 1996; Barrier and Johnson 2012).

Barrier and Johnson (2012) created a model of caribou presence based on the Bathurst herd study that incorporated covariates of coniferous tree basal area and percent ground cover by lichen, moss, litter, and rock. The model had a predictive accuracy of 87.3% for caribou presence at a given site. The model might be improved by including snow depth and hardness (Barrier and Johnson 2012).

10.3 Woodland Caribou

Woodland caribou are found in six populations, two of which are present in the Northwest Territories: the boreal and the northern mountain populations (Environment Canada 2012b).

10.3.1 Boreal Caribou

Boreal caribou are nonmigratory. Local populations are defined by their use of a single range: an area where the caribou are subject to similar influences on demographic characteristics and where they carry out their entire life history. Boreal caribou in the Northwest Territories are widespread throughout the Taiga Plains, and the entire ecoregion is defined as a single range, known as NT1 (Figure 58) (Species at Risk Committee [N.W.T.] 2012). This range is continuous with ranges in northern British Columbia and Alberta, but is delineated as a separate range for political and management purposes (Environment Canada 2012b). Current research is investigating whether subpopulations within the NT1 can be defined according to patterns of movement, genetic analyses, and tradtional knowledge (Department of Environment and Natural Resources [N.W.T.] 2019). The NT1 covers 44 Mha, and caribou within it move freely over greater distances than is the case in most boreal caribou ranges (Environment Canada 2012b).

The latest estimate of the NT1 population is 6000–7000 individuals, but this is only a rough estimate, and a better survey is needed (Department of Environment and Natural Resources [N.W.T.] 2019). Despite the uncertainty in this estimate, the population is almost certainly below 10 000 individuals (Species at Risk Committee [N.W.T.] 2012). Boreal caribou live individually or in small groups of often fewer than 15 individuals. The small herd size combined with the large area over which the animals occur makes it difficult to carry out precise population surveys (Environment Canada 2012b).

The NT1 boreal caribou population is believed to be self-sustaining, a state defined as having at least a 60% to 90% probability (Environment Canada 2011) of the population being stable or growing in the short term (<20 years) and being able to withstand stochastic events in the longer term (>50 years) (Figure 58). Three criteria are required for a population to be considered self-sustaining: it comprises at least 300 individuals, it has a territorial extent that allows for a density of two or three individuals per 100 km^2, and at least 65% of its range is undisturbed (Environment Canada 2012b). The 65% minimum provides a 60% probability that the population is self-sustaining. For the purpose of this definition, disturbance includes fires that burned in the past 40 years and a 500-m buffer around human disturbances visible in Landsat images at a 1:50 000 scale. Thus, an NT1 population of 6500 would require 21.7–32.5 Mha to be self-sustaining; its current territory of 44 Mha fits this criterion (Environment Canada 2012b). As of 2017, the NT1 range had 23.7% of its area disturbed by fire and 9.1% by anthropogenic causes. The fire disturbance was calculated using the latest data from the NBAC. Areas disturbed by wildfire and humans are not counted twice, so the total disturbed area came to 33%, for an undisturbed area of

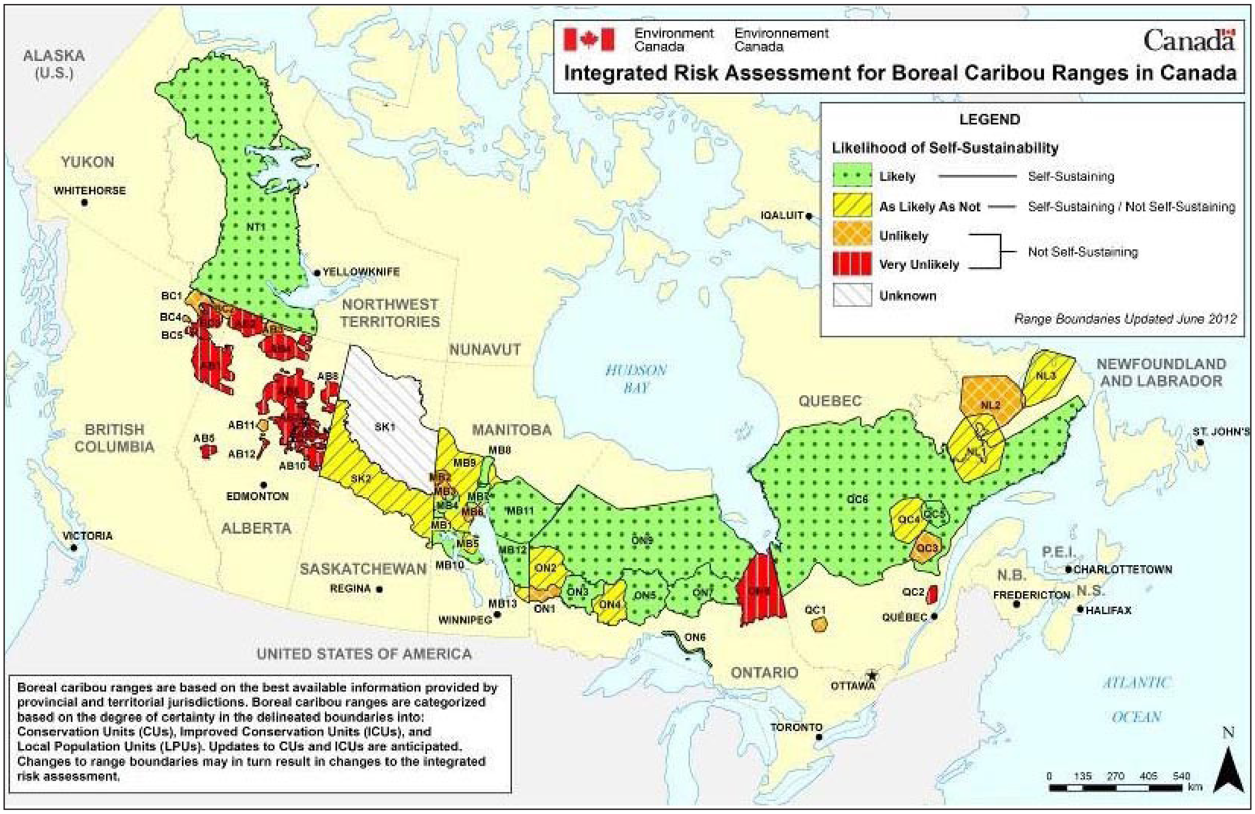


Figure 58. Boreal caribou ranges in Canada, including the NT1 herd in the Northwest Territories, and the likelihood of each herd being self-sustaining. Likelihood of self-sustainability is defined as having a stable or growing population within the next 20 years and being capable of surviving stochastic events over the long term. (Reproduced, with permission, from Environment Canada 2012b.)

67%. However, the southern part of the territory where most of the NT1 population lives most likely has a level of undisturbed habitat below the 65% threshold (Department of Environment and Natural Resources [N.W.T.] 2019): recent surveys estimate undisturbed area at approximately 51% (Conference of Management Authorities 2017).

A study of anthropogenic and natural disturbances and simulations of future disturbance over the next 100 years was conducted in the Dehcho and South Slave portions of the Northwest Territories boreal caribou range by Blyth et al. (2016). About 53% of the NT1 boreal caribou population lives in these two regions (Department of Environment and Natural Resources [N.W.T.] 2019). At the time of the study in 2015, about 66% of the boreal caribou range within the Northwest Territories was considered to be undisturbed. However, in these two southern regions, this study showed that only 52% was undisturbed (Blyth et al. 2016). The total amount of disturbance calculated from the various sources described below sums to about 53%, but because some of these disturbances overlap, only 48% of the landscape was estimated to be affected by disturbance (Blyth et al. 2016).

Seismic lines were found to be the largest anthropogenic disturbance in the Dehcho and South Slave administrative regions, with 25 000 km of linear disturbance covering 12% of the area. However, 80% of the seismic lines were expected to recover within 50 years. Two timber harvest planning areas (Fort Providence and Fort Resolution) covered 11 208 km^2 or 7.1% of the area, although only 1% of the entire study area was open for harvesting. As of 2015, there were 7000 ha of regenerating cutblocks (<40 years old) in these regions, and forest harvesting (including logging roads) led to disturbance of about 1% of the caribou range in these regions. This proportion was expected to increase to 3% over the next 100 years, with estimates of annual combined harvest rates in the timber harvest planning areas of 1700 ha and 250 000 m^3. A peak of 55 000–60 000 ha of regenerating cutblocks was estimated to be reached in 50 years, after which vegetation recovery rates would balance rates of new harvesting. Only a small percentage of logging roads were expected to recover within the simulation period (10% by year 50 and 15%–20% by year 100) because of periodic reuse of road networks and long recovery periods. In addition to cutblocks, about 13 500 ha of other anthropogenic disturbances existed that were not projected to recover over the next 100 years. These included settlements, mines, and oil and gas infrastructure. The area covered by these features, however, was expected to remain stable. Total anthropogenic disturbance amounted to 15% of the area of caribou habitat in these regions (Blyth et al. 2016).

Wildfire is the dominant disturbance on the landscape, and burned areas covered 38% of the study area as of 2015. From studies of caribou populations across Canada, it was found that burned landscapes negatively affect caribou populations for up to 40 years. Therefore, 40 years is the amount of time assumed to be necessary for fire recovery in the boreal caribou recovery strategy. However, this assumption has not been tested for boreal caribou in the Northwest Territories. Vegetation in the Northwest Territories recovers more slowly than farther south, and research within this study area suggests a recovery time of 50 years (Blyth et al. 2016). However, wildfire creates less edge habitat per hectare disturbed than anthropogenic disturbances, and it does not create linear disturbance features. Thus, fire does not promote alternative prey species or create corridors for wolf movement to the same extent as anthropogenic disturbance. Nevertheless, the recovery strategy treats anthropogenic and natural disturbance equally. Because fire dominates the total amount of disturbance to boreal caribou habitat in the Northwest Territories, taking into consideration its lower risk to caribou could result in lower estimates of habitat disturbance. However, current data are insufficient to model this effect (Blyth et al. 2016).

Various combinations of future projections for fire, vegetation recovery, and forest harvesting were used to estimate future levels of undisturbed area in the Dehcho and South Slave administrative regions. All of these scenarios projected that the amount of undisturbed area will increase with the recovery of seismic lines and recently burned areas. Most scenarios project at least 65% undisturbed area in 50 years and more than 70% in 100 years. Uncertainties regarding future fire activity and vegetation recovery are believed to result in possible deviations of ±15% around estimates of undisturbed habitat (Blyth et al. 2016). However, because fire frequency and intensity are both expected to increase in future decades because of climate change, it is possible that deviations from this estimate could be significantly higher. More frequent and more intense fires would disturb a greater proportion of caribou habitat (leaving less habitat undisturbed) and could also change the successional pathways for vegetation away from coniferous-dominated forests toward those better adapted to frequent fires, such as deciduous forests and shrublands (see chapter 5 for more information on the potential changes to fire regime in the future and their possible implications). In addition, although forest harvesting was projected to contribute little to the disturbance of caribou habitat, it has a high level of disturbance per unit area where harvesting does take place. Thus, potential increases in future harvesting rates would substantially increase the amount of habitat disturbance. Nevertheless, these projections did not include potential increases in future harvesting rates (Blyth et al. 2016).

There is evidence of a slight decline of the caribou population in the Dehcho and South Slave administrative regions. Boreal caribou are increasing in the Gwich’in region, where 8% of the population lives. Trends for the remaining 39% of the NT1 population are currently unknown (Department of Environment and Natural Resources [N.W.T.] 2019).

Boreal caribou were listed as threatened under the federal *Species at Risk Act* in 2002 because of a reduction in population greater than 30% over three generations (Environment Canada 2012b). In 2012, the GNWT Species at Risk Committee assessed the NT1 population as threatened because of small population size and expected habitat declines, and the population was listed as such under the *Species at Risk (NWT) Act* in 2014. This listing means that the population is likely to become endangered if action is not taken to reverse factors leading to its extirpation from the Northwest Territories. As a result of this listing, there is a legal requirement to create a recovery strategy that recommends objectives and strategies for the population’s conservation. This recovery strategy was produced in 2017 by the Conference of Management Authorities, a partnership of wildlife comanagement boards and various levels of government that share responsibility for species at risk in the Northwest Territories (Conference of Management Authorities 2017).

In 2019, the GNWT signed a Species at Risk Act Conservation Agreement for the Conservation of the Boreal Caribou with the federal government. The agreement commits to complete range planning within 5 years and includes funding to support the conservation work required by the agreement and engagement with Indigenous partners to ensure these plans reflect their values and interests (Government of Northwest Territories 2019). The Framework for Boreal Caribou Range Planning produced by the GNWT will guide the development of regional range plans. It divides the NT1 territory into five ranges and specifies three levels of land management and specific management actions for each range (Department of Environment and Natural Resources [N.W.T.] 2019).

Given that the NT1 population currently meets the definition of a self-sustaining population, recovery efforts will focus on maintaining both the population and the minimum 65% of undisturbed habitat (Environment and Climate Change Canada 2018a). Identifying and protecting critical habitat constitute an essential step that has yet to be completed. However, five protected areas overlapping the NT1 range already exist and offer a measure of protection from human activity. An additional seven protected areas have been proposed and currently have interim protection (Environment and Climate Change Canada 2018b). Permanent protected areas currently comprise 29 800 km^2 or 6.8% of the NT1. Additional "candidate protected areas" comprise 12 800 km^2 or 2.9% of the NT1. Conservation zones prohibit most industrial development and comprise 21 845 km^2 or 4.9% of the NT1. Interim land withdrawals until land claim negotiations and land use planning are complete comprise another 59 404 km^2 or 13.4% of the NT1. In total, 28% of the NT1 range is protected by some measure from new human disturbances. The southern region of the Northwest Territories, which has the highest level of human-caused disturbance, also has the highest proportion of habitat protection, at 51%. Outside of these areas, there are other mechanisms to manage the cumulative impacts of human development on boreal caribou habitat. For example, Special Management Zones comprise 60 926 km^2 or 13.8% of the NT1, and they have special requirements related to boreal caribou protection. Land use plans have been approved for the Gwich'in Settlement Area, Sahtú Settlement Area, and Tłı̨chǫ Lands (covering 45% of the NT1), and a draft interim land use plan for the Dehcho administrative region is under review (covering an additional 34% of the NT1); these plans all include further requirements for addressing impacts on boreal caribou habitat. Community conservation plans for the Inuvialuit Settlement Region cover 7% of the NT1 and formalize conservation priorities for the region (Government of Canada and Government of Northwest Territories 2019).

Boreal caribou require a large range with continuous undisturbed habitat that allows them to reduce predation by maintaining low population densities and avoiding areas with a high density of predators or alternative prey species. Although boreal caribou do not migrate over long distances, as the barren-ground caribou do, they still require habitat connectivity for seasonal movements between different habitats and for movement in response to disturbance—especially wildfire—and changing environmental conditions, including climate change. Connectivity between ranges is also important to increase gene flow and to allow extirpated ranges to be resettled (Environment Canada 2012b; Species at Risk Committee [N.W.T.] 2012).

Boreal caribou generally prefer older coniferous forest high in lichen abundance or peatlands intermixed with upland areas (Environment Canada 2012b). Lichen is a particularly important winter food. In winter, caribou gather in larger groups in dense forests with an abundance of arboreal lichen and a shallower snowpack. These conditions make it easier for caribou to dig for ground lichens and to move around, and this overwintering habitat is therefore critical for caribou survival (Environment Canada 2012b; Species at Risk Committee [N.W.T.] 2012). Spring is calving season, and habitat during these months is chosen primarily for predator avoidance. Females disperse and often seek out islands. Other preferred calving habitat includes wet areas and high ridges (Species at Risk Committee [N.W.T.] 2012). Summer habitat is generally similar to that of the spring. Deciduous shrubs, rather than lichens, were shown to dominate the diet of female caribou in British Columbia during the summer (Denryter et al. 2017), and traditional knowledge in the Northwest Territories also testifies to caribou eating willow leaves, sedges, and grass in the summer and aquatic plants in the spring (Species at Risk Committee [N.W.T.] 2012). Boreal caribou use a variety of habitats as they move around during the fall rutting season. They often move to higher elevations to escape low-lying fog and seek out trees on which to rub their antlers (Species at Risk Committee [N.W.T.] 2012). During this time, they prefer more open forest and areas of low or sparse vegetation, including regenerating and recently burned areas (Environment Canada 2012b).

Boreal caribou females first reproduce at 3 years of age and give birth to only one calf per year after that. Calf survival to 1 year of age is low because of predation (Environment Canada 2012b). In fact, predation is the limiting factor for boreal caribou populations, and the most important threat to boreal caribou in the Northwest Territories is increased predation from habitat disturbance (Conference of Management Authorities 2017). Linear disturbances related to seismic lines, pipelines, and roads create travel and sight corridors for wolves, giving them access to caribou in places where the caribou once found refuge. Forest disturbances such as fire and harvesting transform mature forest to young seral forest. Elsewhere in Canada, this process has created better habitat for alternative prey, such as deer and moose, and increased the predation rate of caribou through bycatch. Unlike deer, caribou have a low birth rate and cannot sustain the higher levels of predation. It is unknown whether a similar dynamic exists in the Northwest Territories, where predator and alternative prey densities are low (Conference of Management Authorities 2017). However, there is a concern that the combination of habitat disturbance and warming temperatures will lead to an increase in alternative prey densities in the Northwest Territories and a subsequent increase in caribou predation (Environment Canada 2012b).

In addition to the increase in alternative prey densities, climate change is expected to have other negative effects on boreal caribou in the Northwest Territories (Figure 3, Figure 4). Forest fires are expected to increase in frequency and severity in Canada's western boreal forest (Parisien et al. 2011; Stralberg et al. 2018), and area burned in the Northwest Territories is also expected to increase (Flannigan et al. 2005). These changes are expected to reduce the amount of usable caribou habitat (Environment Canada 2012b), although there are uncertainties regarding how soon after a burn caribou will return to an area (Species at Risk Committee [N.W.T.] 2012); certainly caribou have been observed using recently burned areas, particluarly in the fall (Environment Canada 2012b). Permafrost thaw transforms lichen-rich peat plateaus to collapse scars dominated by mosses and sedges, decreasing food availability for caribou (Errington 2019). More frequent ice-on-snow events will make travel and foraging difficult.

It is also important to consider the cumulative effects of multiple threats to boreal caribou (Conference of Management Authorities 2017). In a study assessing the potential long-term impacts of climate change and wildfire on woodland caribou habitat in northern Alberta, Barber et al. (2018) stated that because of a changing fire regime, early-seral deciduous forests are projected to expand by 2050, and the authors suggested that by 2080 large areas of coniferous upland forest could be replaced by grasslands. In fact, these authors predicted that 20%–50% of the study area could be covered by grassland ecosystems by 2080 (Barber et al. 2018). Given that there is little grassland in this region currently, it is difficult to appreciate how this change may alter the area's ecology and fire behavior, but this shift would undoubtedly represent a significant change to the nature and quality of habitat in that area (Barber et al. 2018). Considering that caribou rely on forested habitat, they would not be expected to persist in a grassland ecosystem over the long term (Barber et al. 2018). Nevertheless, forested peatlands are expected to persist in this region under future climate projections, as they have been observed to be more resilient to climate fluctuations because of their ability to retain large volumes of water. Whether these refugia would be sufficient to maintain woodland caribou populations into the future is unknown. It is also not known how applicable these findings from Alberta would be to the caribou of the Northwest Territories.

The priority of the recovery strategy for the NT1 population is to create region-specific and overall NT1 range plans that take into account anthropogenic and natural habitat disturbance, as well as predation. However, the following significant knowledge gaps, among others, impede the ability to create such a plan: human impacts, whether in the past or present, have not always been monitored; the location of preferred habitat on the landscape is often unknown; it is unknown whether alternative prey densities in the Northwest Territories are high enough to cause an increase in caribou bycatch; accurate caribou harvest information is lacking; the cumulative effects of multiple threats is not understood; the rate of habitat regeneration following disturbance is not known for the north; and the long-term effects of climate change are unknown (Conference of Management Authorities 2017). Questions also remain regarding the impacts of wildfire on habitat quality. For example, it is not known whether caribou use areas with low-severity burns or unburned retention within burned areas (Department of Environment and Natural Resources [N.W.T.] 2019).

Woodland caribou harvesting in the Northwest Territories is currently restricted to one boreal and one northern mountain caribou per resident hunter per year, with no restrictions on the Indigenous hunt. The harvest season for boreal caribou has been shortened and is restricted to bulls only (Government of Northwest Territories 2019). It was previously estimated that 22 boreal caribou are on average harvested by resident hunters per year, but there is concern that this number has been underestimated (Environment and Climate Change Canada 2017). While still at low levels, hunting of boreal caribou is believed to be increasing, necessitating more accurate assessments of harvest levels (Conference of Management Authorities 2017). The GNWT currently estimates harvest levels through its annual Resident Hunter Survey and is developing a reporting system for Indigenous harvest that respects treaty and Indigenous rights (Environment and Climate Change Canada 2017).

Under the federal *Species at Risk Act*, it is the responsibility of Environment and Climate Change Canada to identify and protect critical habitat for listed species, including boreal caribou (Environment Canada 2011; Environment and Climate Change Canada 2018b). However, given that most of the land is managed by the GNWT, development of caribou range plans is also the responsibility of the territorial government.

The identification of critical habitat by Environment and Climate Change Canada has focused on human disturbance and the identification of undisturbed habitat (Environment Canada 2012b). Research is now ongoing at the NoFC to identify areas of differing habitat quality in terms of meeting the nutritional, security, and climatic needs of caribou. Currently, a broad-scale, Canada-wide model of caribou occupancy is being created. The next stage will focus on regional simulations to test the ability of management to increase caribou viability under climate change (unpublished data provided by E. Neilson).

Whitman et al. (2017) created a habitat quality model for woodland caribou in northeastern Alberta (Boreal Plains ecoregion) and eastern Quebec, which could potentially be used with data for boreal caribou in the Northwest Territories as well. Nutritional resources, predation risk, and landscape connectivity were combined using three different methods to produce an overall habitat quality value for each pixel on the landscape. Each method produced different maps of habitat quality, but habitats of very high and very low quality were consistently placed in the same locations by all three methods. Mature conifer forests, bogs, and fens were consistently identified as high-quality habitat in both Alberta and Quebec, whereas areas with extensive seismic lines were classified as having the worst quality because of increased predation (Whitman et al. 2017). Simulated wildfire disturbance was added to the model to determine how wildfire affects the distribution of habitat quality. In Alberta, the influence of fire was neutral over the entire landscape, but had a large effect on individual habitats, particularly on those of medium quality, by increasing nutrition in regenerating mixed-wood and deciduous stands, while increasing predation risk and decreasing winter forage in burned dense coniferous stands (Whitman et al. 2017).

Lichen biomass is higher in peat plateaus than in upland forests and collapse scars. However, 10-year remeasurements of plots along the Mackenzie River Valley revealed a decline in lichen biomass in peat plateaus, along with expansion of collapse scars (Errington 2019). In the coming years, the results of the above calculations of lichen biomass decline will be extrapolated over the entire landscape. Correlation of the distribution and vegetation of peat plateaus with caribou behavior has shown that caribou preferentially select peat plateaus where lichen biomass is greatest. Future work will extend this analysis to determine whether there are correlations between the presence of particular landforms (peat plateaus and collapse scars) and caribou occupancy and population trends (unpublished data provided by E. Neilson).

For the past decade and a half, the movements and behavior of individual boreal caribou in the Dehcho and South Slave administrative regions have been studied through radio collar tracking. The results of these studies are summarized below.

Dehcho: Since 2004, boreal caribou have been collared and studied in the Dehcho administrative region by the GNWT (Figure 59). To date, 168 female and 10 male caribou have been collared, and information has been gathered on the movements of 116 females and one male over at least 12 months (Larter and Allaire 2018). The mean female range size was 3118.7 km^2, while the male's annual range was 1958.93 km^2.

Movement of females decreased from 6 km·day^{-1} at 2 days before calving to 0.2 km·day^{-1} on the day of calving and to less than 1 km·day^{-1} for a week afterward. Females spread out before calving to avoid predation. Some used the same general area in subsequent years to calve, but others did not, so that one group of 10 females had an average of less than 6 km separating four calving locations, while another group of six females had an average distance greater than 30 km (Larter and Allaire 2018). Movement toward calving grounds took place in small groups in April, and calving tended to occur on islands in May, with peak calving on May 15 (Larter and Allaire 2018). During the peak breeding season of 20 September to 4 October, the males reduced their movements, waiting in areas averaging 285 km^2 for the females to come to them (Larter and Allaire 2018).

Pregnancy and birth rates of female caribou were well above 90% between 2004 and 2018. The high birth rate offsets the low calf survival rate. Surveys of female caribou in March have shown that averages of 28.2% of collared females and 32.6% of all observed females were accompanied by calves. This puts calf survival at only 30%–35%. Similarly, only eight of the 24 calves born in May 2017 survived to March 2018, a survival rate of 33% (Figure 60) (Larter and Allaire 2018). Pregnancy rates are affected by the body fat condition of female caribou in the previous late winter. Late-winter body fat condition of female caribou remained high and stable during the 4 years this metric was assessed, and individuals that entered winter in good condition were typically observed to enter spring in good condition as well (Larter and Allaire 2018).

The average rate of population growth between 2005–2006 and 2018–2019 in Dehcho South was 0.97, with an annual range of 0.72–1.28. The rate of population growth for the same years in Dehcho North was 0.94, with an annual range of 0.72–1.60 (Department of Environment and Natural Resources [N.W.T.] 2019). Although this is close to the self-replacing rate of 1, it means that a population starting at 1000 adult females would have only 687 adult females after 13 years (Larter and Allaire 2018). A stable population is believed to require a calf–cow ratio of 0.29:1 and an annual cow survival rate of 85%, with a decrease in one of these values requiring an increase in the other to maintain a stable population. The average annual adult female survival rate between 2005–2006 and 2018–2019 was 80%, while the mean calf–cow ratio was 0.33:1 (Table 27) (Kelly and Cox 2011).

Most adult female deaths occurred between late March and mid-July (73%). Of the 57 death sites investigated, 75% were from suspected wolf predation, 12% from hunting, 9% from malnutrition, and 1% from black bear predation. Two male deaths were of unknown causes, one in April and one in December. The age at death ranged from 4 to 22 years, with a mean of 10.3 years (Larter and Allaire 2018). Anthropogenic linear features are believed to facilitate predator access and increase caribou predation. This theory was supported by the observation that 39% of predator kills took place within 400 m of an anthropogenic linear feature (Larter and Allaire 2018). Seismic lines and roads in the Scotty Creek area south of Fort Simpson exist at a density of 0.875 km·km^{-2} (Dabros et al. 2018). In addition to allowing easier access for wolves, they create early succession habitats that attract primary wolf prey such as moose and deer, increasing incidental predation of caribou. Woodland caribou have been found to avoid using seismic lines, thereby losing access to otherwise suitable habitat (Dabros et al. 2018).

Wolves' hunting efficiency has been shown to increase with increasing snow depth, whereas for ungulates other than caribou, there is a negative correlation between calf survival and the previous winter's snow depth, perhaps because of the negative effects of snow depth on the physical condition of the pregnant mother (Larter et al. 2017). Larter et al. (2017) therefore tested whether snow depth in the current or previous winter was negatively correlated with caribou calf survival using data from the surveys in the Dehcho administrative region. However, they found no correlation between calf survival and snow depth in either year. The lack of impact from the previous winter's snow depth could be explained by the fact that winter forage is not a limiting factor within the study area, and caribou could therefore compensate for lower calf birth weight by a greater investment in rearing. The lack of impact from the present year's snow depth was perhaps due to the low density of both caribou and wolves in the study area. As well, although snow depth over the study period ranged from 41 to 85 cm—a twofold difference—these snow depths are far below the variation in snow depth found over the entire boreal caribou range. For example, mean snow depth in the boreal caribou range of Labrador is 177 cm. It is therefore possible that snow depths in the Dehcho administrative region are not deep enough, even at their maximum, to increase wolf predation (Larter et al. 2017).

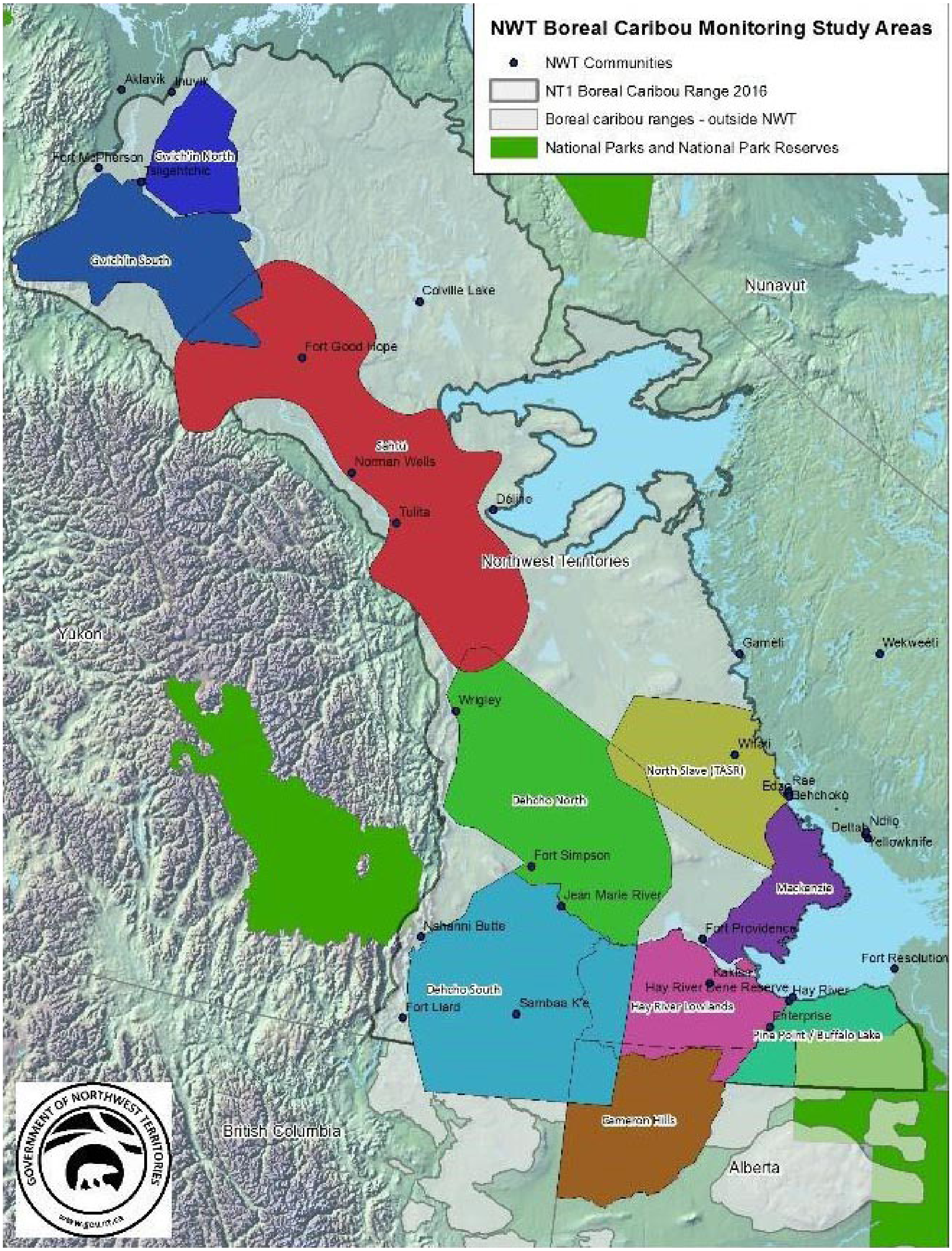


Figure 59. Boreal caribou study areas within the Northwest Territories. The Sahtú, Gwich'in North and South, and Cameron Hills study areas are no longer active. Map provided by Government of Northwest Territories, May 2020. For further information on boreal caribou monitoring study areas in the Northwest Territories, see Appendix A of A Framework for Boreal Caribou Range Planning. (Reproduced, with permission, from Department of Environment and Natural Resources [N.W.T.] 2019.)

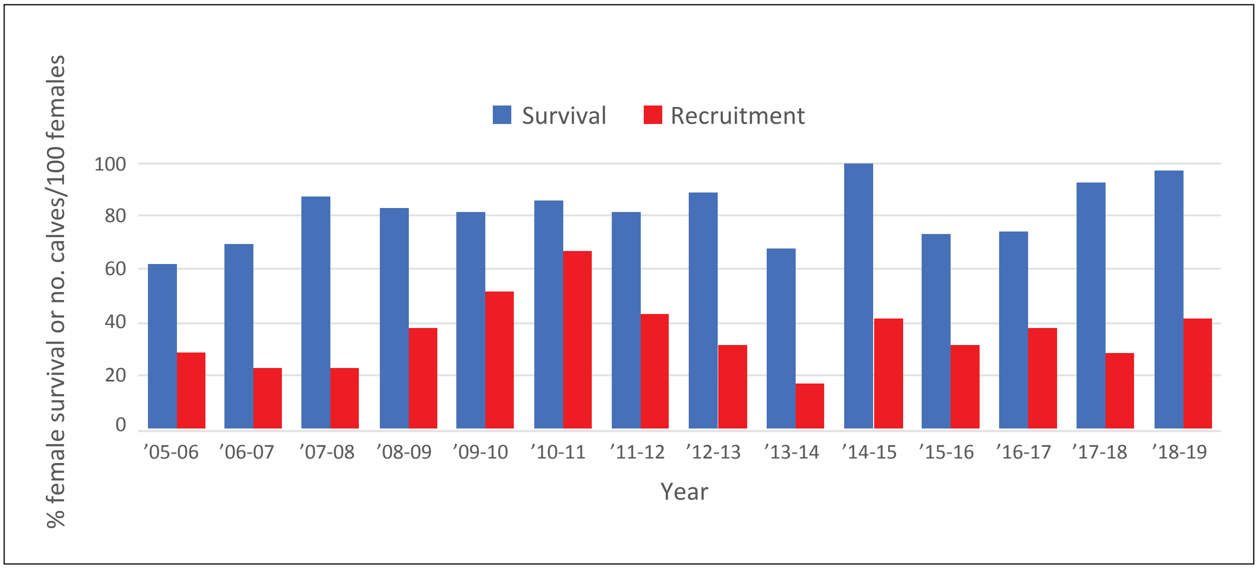


Figure 60. Adult female survival (%) and number of calves per 100 adult females (recruitment) of boreal woodland caribou in the Dehcho study area from 2005/06 to 2018/19, based on radio collar data. (Reproduced, with permission, from Larter et al. 2019.)

Table 27. Estimated annual rate of increase (λ) of the population of boreal woodland caribou in the South Dehcho administrative region for 14 successive years. (Reproduced, with permission, from Larter et al. [2019].)

		Calves per 100 females		Annual rate of increase (λ)	
Year	Female survival rate	Collared caribou	All caribou	Collared caribou	All caribou
2005–2006[a]	0.62	28.81	28.81	0.72	0.72
2006–2007[a]	0.69	22.86	22.86	0.78	0.78
2007–2008[a]	0.87	23.35	23.35	0.99	0.99
2008–2009	0.81	37.83	31.25	1.00	0.97
2009–2010	0.81	51.28	35.16	1.09	0.98
2010–2011	0.86	66.67	44.59	1.29	1.10
2011–2012	0.81	43.33	38.46	1.03	1.00
2012–2013	0.88	31.43	28.11	1.05	1.03
2013–2014	0.68	16.67	27.36	0.74	0.79
2014–2015	1.00	41.03	45.32	1.26	1.29
2015–2016	0.74	32.35	31.09	0.88	0.87
2016–2017	0.74	37.93	41.33	0.92	0.94
2017–2018	0.92	28.21	32.56	1.07	1.10
2018–2019	0.97	41.67	38.09	1.22	1.20
Average	0.81	35.96	33.46	1.00	0.98

[a]Because of the small number of collared animals in these years, the ratios and rates were determined from all caribou surveyed.

South Slave: Boreal caribou of the South Slave administrative region have been tracked in the Hay River Lowlands south of the Mackenzie River and Great Slave Lake since 2003, in the Cameron Upland spanning the Northwest Territories–Alberta border between 2004 and 2011, in the Mackenzie Bison Sanctuary and at Pine Point and Buffalo Lake since 2015, and in the North Slave region since 2017 (Figure 59) (Department of Environment and Natural Resources [N.W.T.] 2019). Population demographics, seasonal range use, and habitat selection in response to human and natural disturbances were all monitored. The Hay River Lowlands overlap three secure habitat patches of at least 500 km^2 that are not fragmented by anthropogenic linear features and have not had fires in the past 50 years. These constitute 15% of the entire area and are used by caribou 92% more often than would be expected by chance. Total secure habitat in the Hay River Lowlands accounts for 52% of the area. Such areas are associated with greater caribou survival. The entire area has had little industrial development and has an average seismic line density of 0.36 km·km^{-2} (Kelly and Cox 2011, Kelly and Cox 2013). The Cameron Upland, by contrast, have no secure habitat patches larger than 500 km^2, and secure habitat covers only 16% of the area, with 25% of that consisting of small patches (less than 2.5 km^2). It has a seismic line density of 3.32 km·km^{-2} and a higher level of ongoing industrial development. Studying both areas thus allows comparisons between areas of high and low human impact (Kelly and Cox 2011).

The mean annual home range of caribou that had collars for at least 1 year was 2847 km^2 in the Hay River Lowlands and 4140 km^2 in the Cameron Upland (Kelly and Cox 2011). Collared caribou have been seen moving between the Dehcho and South Slave study areas (Larter and Allaire 2018).

Caribou in the Hay River Lowlands and Cameron Upland avoided seismic lines during precalving, calving, and summer rearing periods, during which they were most vulnerable to predators. In the Hay River Lowlands they avoided areas within 400 m of seismic lines, whereas in the Cameron Upland they avoided areas within 100 m of seismic lines. The smaller buffer area in the Cameron Upland was probably due to the greater density of seismic lines and hence their unavoidability. Caribou in both areas also crossed seismic lines less often than would be expected by random movement, and when doing so they traveled faster than usual, thus expending extra energy (Kelly and Cox 2011). Cows spread out over the landscape for calving and did not show fidelity to calving sites from year to year. They remained alone with their calves through the summer. Birth rates ranged from 87% to 95% in the Hay River Lowlands between 2003 and 2007 and from 79% to 90% in the Cameron Upland between 2005 and 2007 (Kelly and Cox 2011).

The Hay River Lowlands had mean annual adult female survival of 86% between 2004 and 2010 and a calf–cow ratio of 0.24:1 between 2004 and 2013. Between 2006 and 2010, the Cameron Upland had mean annual adult female survival of 81% and a calf–cow ratio of 0.15:1 (Kelly and Cox 2011, 2013). Average population growth in the Hay River Lowlands was 0.96, with a range of 0.72–1.14, while in the Cameron Upland it was 0.87, with a range of 0.74–1.00 (Kelly and Cox 2011). The Hay River Lowlands population declined by about 20% between 2003 and 2004 and by a further 10% between 2004 and 2010, although the overlap among confidence intervals could mean that the trend for 2004–2010 was stable. The Cameron Upland population declined by about 50% between 2005 and 2010. As with the Dehcho population, most deaths occurred in spring and summer (75%), with wolves as the main cause (accounting for 78% of recorded deaths between 2003 and 2008; 9% were from bear predation, 9% from hunting, and 3% of natural causes) (Kelly and Cox 2011).

10.3.2 Northern Mountain Caribou

The second population of woodland caribou in the Northwest Territories is the northern mountain caribou. Six herds have at least part of their range within the Mackenzie Mountains and foothills of the Northwest Territories. From south to north these are the La Biche, Coal River, South Nahanni, Redstone, Tay River, and Bonnet Plume herds (Figure 61). These ranges cover about 12 Mha within the Northwest Territories (Species at Risk Committee [N.W.T.] 2020).

The latest estimate for the entire northern mountain caribou population was 50 000–55 000 individuals (Species at Risk Committee [N.W.T.] 2020). In 2002, COSEWIC listed the population as being of special concern because of increased human presence and development in the area, although most of the range is still remote and has not undergone change (Environment Canada 2012a). The South Nahanni herd was estimated in

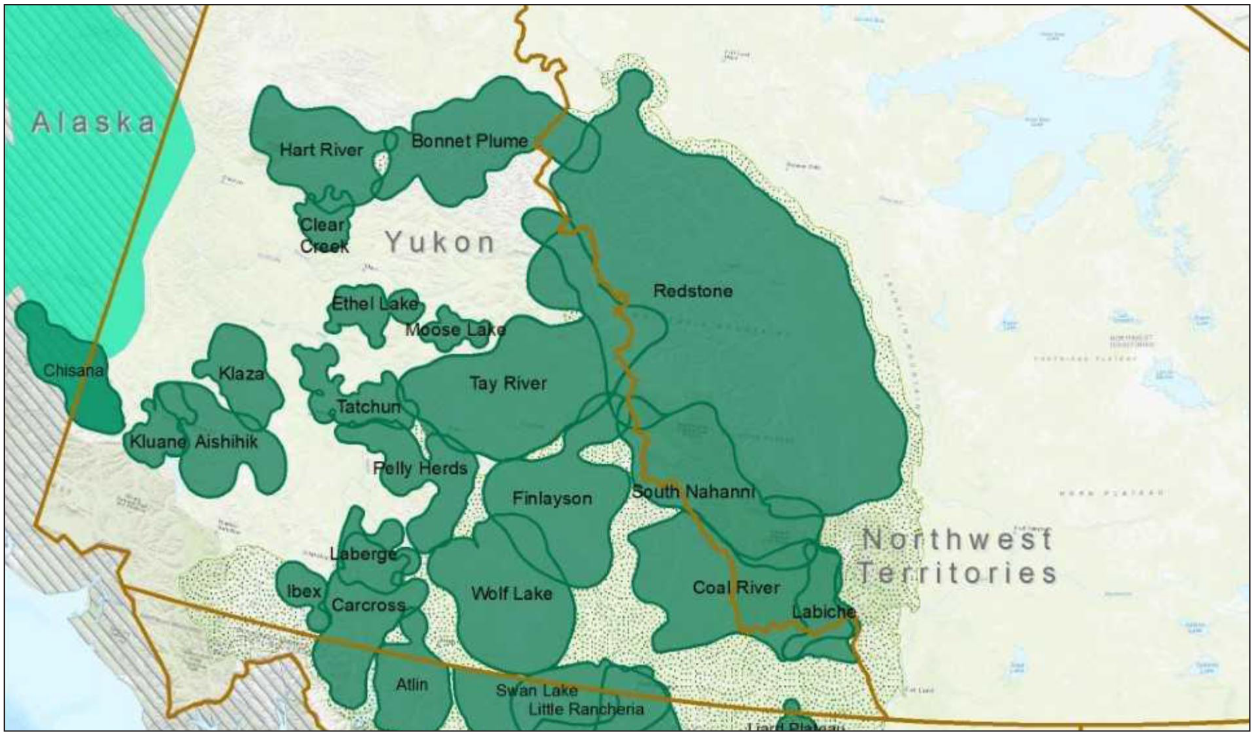


Figure 61. Northern mountain caribou ranges in the Northwest Territories, including the La Biche, Coal River, South Nahanni, Redstone, Tay River, and Bonnet Plume herds. (Reproduced, with permission, from Species at Risk Committee [N.W.T.] 2020.)

2009 to contain about 2105 individuals, the Coal River herd in 2008 to contain 450 individuals, the La Biche herd in 1993 to contain 450 individuals, the Tay River herd in 1991 to contain 3758 individuals, the Bonnet Plume herd in 1982 to contain 5000 individuals, and the Redstone herd in 2012 to contain more than 10 000 individuals. A rough estimate of all northern mountain caribou in the Northwest Territories would therefore be about 21 800 individuals, although most of the component estimates are outdated (Species at Risk Committee 2020). A decrease in the calf–cow ratio in the Redstone and Bonnet Plume herds between 1991 and 2016 signifies that the populations may be decreasing. The South Nahanni herd appears to be stable (Species at Risk Committee [N.W.T.] 2020). Harvest within the Northwest Territories is low, at 300–350 individuals per year, but may be increasing in some areas (Environment Canada 2012a).

Northern mountain caribou often make seasonal altitudinal migrations from alpine and subalpine areas in the summer to lower subalpine and coniferous forest areas in the winter, travelling up to 250 km between them (Species at Risk Committee [N.W.T.] 2020), although some herds remain in alpine areas for the winter. Human barriers to this movement would hinder their ability to find seasonal forage (Environment Canada 2012a). The most common winter habitat is open spruce forests in valley bottoms (Species at Risk Committee [N.W.T.] 2020), where mountain ridges or divides create a rain shadow and a shallower snowpack. Northern mountain caribou rely mostly on terrestrial lichens where the snow depth is less than 50–100 cm and on arboreal lichens where snow depth is deeper (Environment Canada 2012a). The locations of preferred winter ranges for specific herds are found in the 2020 species status report (Species at Risk Committee [N.W.T.] 2020). In the summer, caribou favor birch–sedge communities, where they feed on willow leaves and sedges, switching to sedge meadows in the fall. Alpine snow patches are used in the summer to escape insects and the heat (Environment Canada 2012a). Females calve in May and June, during which time they disperse individually to higher mountainous terrain to escape predators (Species at Risk Committee [N.W.T.] 2020).

Fire strongly affects northern mountain caribou habitat (Figure 3, Figure 4). Within their range in the Northwest Territories, the fire return interval is 150–

300 years (Environment Canada 2012a). Northern mountain caribou in Alaska were observed to avoid burned areas for up to 60 years, perhaps because of the lack of lichen for winter forage, although they did feed on regenerating vegetation in the short term immediately after a fire. The caribou expanded their winter range to compensate for burned areas and successional shifts in vegetation. Although fires are necessary to reduce the growth of moss and maintain lichen dominance, increasing wildfires may shift the landscape to a greater proportion of early successional stands and cause a decline in northern mountain caribou habitat (Environment Canada 2012a). Fires mainly occur at lower elevations at the eastern boundary of their range, and seismic lines are mostly found at lower elevations along the eastern and northern boundaries (Species at Risk Committee [N.W.T.] 2020). Forestry was listed as the greatest concern for northern mountain caribou in the COSEWIC assessment, but this designation does not currently apply in the Northwest Territories, where the main threats are wolf predation, mineral exploration and development (with their associated roads), hunting, and climate change. Roads and seismic lines create travel corridors for wolves and can increase caribou predation. Moose densities are low in the Northwest Territories, and there is less risk of moose causing an increase in caribou bycatch (Species at Risk Committee [N.W.T.] 2020).

Most industrial activity within the northern mountain caribou range in the Northwest Territories occurs on the periphery, leaving large interior areas that are inaccessible by road. The Nahanni National Park Reserve and Nááts’į̨hch’oh National Park Reserve together protect almost 3.5 Mha within the ranges of the South Nahanni, Coal River, La Biche, and Redstone herds (Species at Risk Committee [N.W.T.] 2020).

Key Points

- Eight barren-ground caribou herds winter within the boreal forest of the Northwest Territories. The level of monitoring and time since the most recent census differ for each herd, but the average population across all herds was low during the 1970s, increased until the mid-1990s, and has declined since then. The recent decline is probably a combination of a natural population cycle and increasing human pressure on caribou habitat. Management plans have been created for several of these herds, and hunting restrictions specific to each one are in place.
- Barren-ground caribou prefer mature coniferous forest as winter habitat, because these forests have a greater abundance of mat-forming lichens and a softer, shallower snowpack that allows easier access to the lichen. Burned sites have a lower volume of lichen and are used less often by the herds.
- Boreal caribou have a single range delineated in the Northwest Territories, known as the NT1. They are non-migratory and range throughout the Taiga Plains ecoregion, either individually or in small groups. The population is believed to consist of 6000–7000 individuals and is considered to be self-sustaining.
- One criterion for a self-sustaining population requires that 65% of the population’s range being undisturbed. The NT1 range currently has about 69% undisturbed habitat, but only 52% of land is undisturbed in the southern portion of the range, where 53% of the population resides.
- The NT1 population was listed as threatened under the Species at Risk (NWT) Act in 2014 and recovery efforts will focus on maintaining the 65% threshold for undisturbed habitat.
- Boreal caribou prefer areas high in lichen abundance, including mature coniferous forest and peat plateaus. They require a large range to maintain low density and avoid areas with high predator densities. Females disperse to even more isolated areas in May to give birth, often on islands. Habitat connectivity is important for access to seasonal foods and to escape disturbance, particularly wildfires.
- Radio collar studies in the Dehcho and South Slave administrative regions have tracked the movements of female and male caribou since 2003–2004 and provide estimates on survival, birth rate, and population trends. A large proportion (75%–78%) of mortality was caused by wolf predation. Average population growth was 0.97 in the Dehcho region, 0.96 in the Hay River Lowlands, and 0.87 in the Cameron Upland.
- The transition of peat plateau to collapse scar because of permafrost thaw has caused a decline in lichen biomass in plots along the Mackenzie Valley.

Knowledge Gaps

- Population trends for many barren-ground caribou herds are unknown or have not been recently updated. Population estimates for boreal caribou need to be improved.
- Locations of high-quality habitat are often unknown. Existing models can be used to create maps for boreal caribou in the Northwest Territories that indicate habitat quality in relation to nutritional needs, connectivity, and predator avoidance.
- It is not known whether alternative prey densities are high enough in the Northwest Territories to cause an increase in wolf populations and incidental predation of caribou.
- Landscape-scale effects of permafrost thaw on lichen biomass have not been calculated.
- The effects of fire on habitat quality and how these effects differ from anthropogenic disturbances are not completely understood.
- The impacts and implications of changing fire regimes for critical caribou habitat are also not currently understood.

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