

# Chapter 8

## Ecological Classification in Forest Ecosystem Management: Links Between Current Practices and Future Climate Change in a Québec Case Study



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**Abstract** Climate change is expected to profoundly impact boreal forests, ranging from changes in forest composition and productivity to modifications in disturbance regimes. These climate-induced changes represent a major challenge for forest ecosystem management, as information based on ecological classification may no longer provide a straightforward guide for attaining management goals in the future. In this chapter, we examine how climate change could influence the use of ecological classification and by what means this approach can continue to be relevant for guiding the ongoing development of management practices. We address these questions by first describing ecological classification, using the example of Québec's classification system, and then showing its importance in forest ecosystem management. Using a forest landscape in Québec as a case study, we then look at how climate change could affect boreal forest ecosystems by presenting a detailed, multi-step analysis that considers climate analogs, habitat suitability, and changes in forest composition. We show that at the end of the century, the vegetation of the *Abies-Betula* western subdomain will not change sufficiently to resemble that of its climate analog, currently located ~500 km to the south. Changes in fire frequency and severity could

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significantly modify forest dynamics and composition. Consequently, the potential vegetation and the successional pathways defined under the current climate could change and follow new successional trajectories. This possible reality forces us to question some fundamental aspects of ecological classification. However, we argue that ecological classification can still provide a valuable framework for future forest management, particularly in continuing to recognize the various types of ecosystems present along toposequences. Given the changes expected in forest vegetation composition and dynamics, future variability and uncertainty must be integrated into the current stable classification units and predictable successional trajectories of ecological classification.

## 8.1 Introduction

Ecological classification provides complete descriptions of biophysical forest features and their historical variability at various spatial scales (global, regional, local). This ecological information is necessary to inform forest ecosystem management, which consists of implementing silvicultural strategies that aim to ensure the long-term maintenance of ecosystem functions and, consequently, the social and economic benefits that forest ecosystems provide to society (Chap. 1; Gauthier et al., 2009). However, climate change is expected to affect growth rates, disturbance regimes, species distributions, and, ultimately, biodiversity (Boulanger et al., 2014, 2016; Périé & de Blois, 2016; Périé et al., 2014; Price et al., 2013). These changes raise serious concerns for forest ecosystem management if knowledge from ecological classification can no longer provide a reliable guide for meeting future management objectives (Millar et al., 2007; Nagel et al., 2017; Puettmann, 2011).

In this chapter, we first introduce ecological classification and the concept of potential vegetation (Sect. 8.2.1) and then characterize the principles of the Québec hierarchical classification system (Sect. 8.2.2). We then consider historical variability

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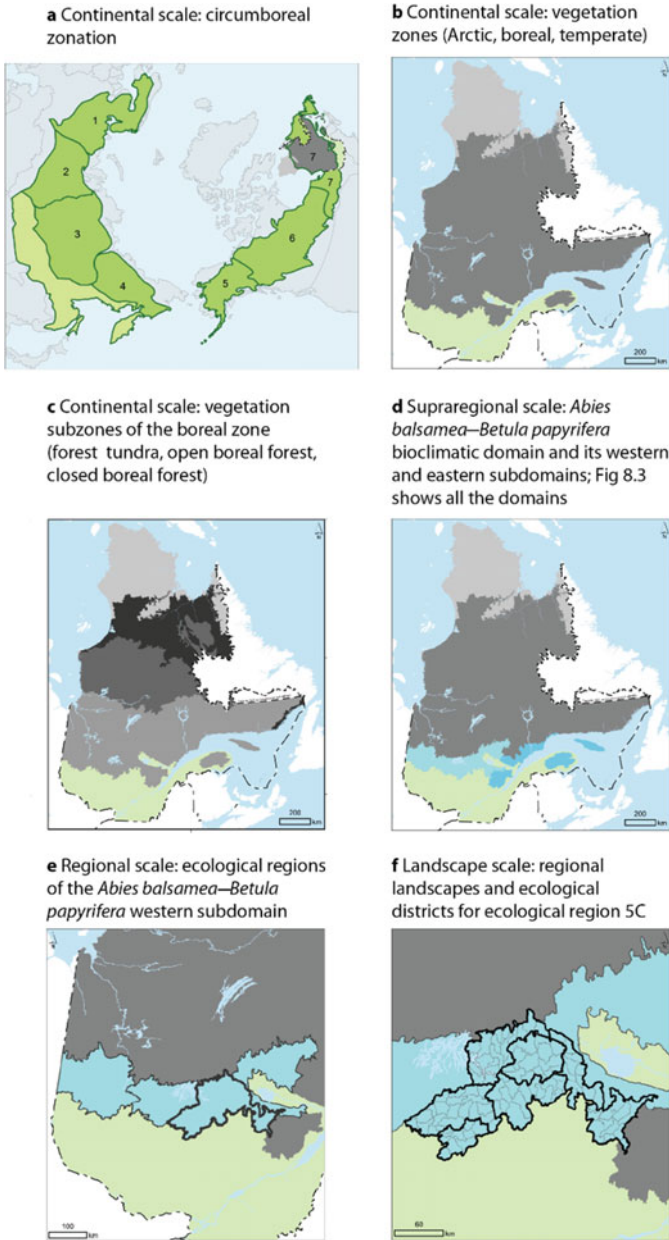
at regional and local scales (Sect. 8.2.3) and the importance of ecological classification in forest ecosystem management (Sect. 8.2.4). Next, using a three-step analysis, we indicate how climate change could modify boreal forest ecosystems and ecological classification (Sect. 8.3.1). In the final section (Sect. 8.3.2), we discuss the issues and challenges raised by climate change and illustrate how the ecological classification framework could integrate future variability in forest dynamics in the context of risk management. Throughout the chapter, we focus on the boreal biome, using the case study of the *Abies balsamea*–*Betula papyrifera* western subdomain (84,567 km<sup>2</sup>) in Québec, Canada (hereafter the *Abies-Betula* w. subdomain).

## 8.2 Ecological Classification of Ecosystems

We define ecological classification as “an analytical process that consists of delineating and defining ecosystems, at different spatial scales, on the basis of the abiotic and biotic factors that govern their development (climate, disturbances, physical environment, plant succession) along ecological gradients for the purposes of land resource management” (Bailey, 2009; Barnes et al., 1982; Rowe & Sheard, 1981; Sims et al., 1996; Whittaker, 1962). The ecological classification of ecosystems has developed progressively in parts of the world, including in Canada, e.g., the provinces of Québec (Fig. 8.1; MFFP 2021; Saucier et al., 2009, 2010), British Columbia (Banner et al., 1993; MacKenzie & Meidinger, 2018; Pojar et al., 1987) and others (Baldwin et al., 2020), as well as in the United States (Bailey, 2009), Scandinavia (Engelmark & Hytteborn, 1999; Sjörs, 1999), and northeastern Asia (Krestov et al., 2009).

### 8.2.1 Ecological Classification: Potential Vegetation

Ecological classification is based on the science of phytosociology, which originated more than a century ago when Flahault and Schröter (1910) defined a plant association or community as “a vegetal grouping of determined floristic composition, presenting a uniform physiognomy that grows in uniform site conditions.” From the school of phytosociology emerged the concept of potential natural vegetation (hereafter potential vegetation). This concept refers to a *stable state* or *late-successional vegetation stage* at a particular site under the existing climatic and edaphic conditions (Baldwin et al., 2020; Blouin & Grandtner, 1971; Braun-Blanquet, 1972; Clements, 1936; Daubenmire, 1968; Küchler, 1964; Tuxen 1956 in Härdtle, 1995). Gradually, although mainly in the early 1970s, interest focused on understanding natural disturbances, particularly fire frequency and severity, and the impacts of disturbance on vegetation dynamics (Damman, 1964; Heinselman, 1973; Rowe & Scotter, 1973; White, 1979). For example, in some cases, the late-successional stage is never reached because of recurrent disturbances, such as fire, and only the early- or mid-successional



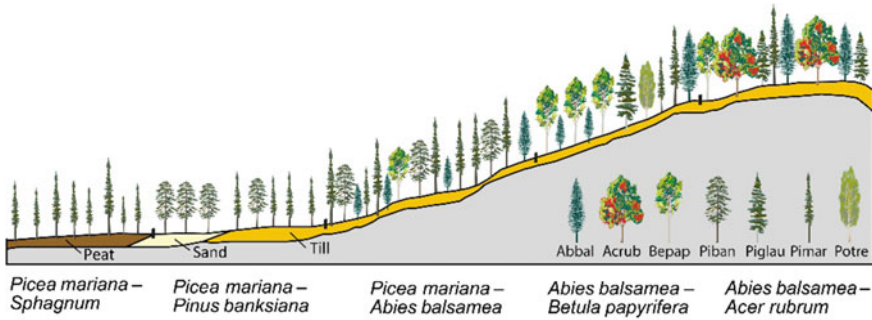
**Fig. 8.1** a Circumboreal zonation modified by permission of Taylor & Francis Group from Saucier et al. (2015) and Baldwin et al. (2020). Only the largest subdivisions are considered (1–7, dark green). 1 North European boreal, 2 Western Siberian boreal, 3 Central Siberian boreal, 4 North-eastern Siberian boreal, 5 Alaska–Yukon boreal, 6 West-central North American boreal, 7 Eastern North American boreal. b–f Ecological classifications of the ministère des Forêts, de la Faune et des Parcs du Québec (MFFP 2021 [CC-BY 4.0]; Saucier et al., 2009, 2010, with permission from Multimondes and AgroParisTech, respectively)

stage is attained after disturbances (Bergeron et al., 2014; Cogbill, 1985; Couillard et al. 2016; Dansereau, 1957; Frégeau et al., 2015; Frelich & Reich, 1995; Grondin & Gosselin, 2013; Grandtner, 1966; White, 1979). Anthropogenic disturbances are also considered an important factor that can modify the dynamics of potential vegetation (Danneyrolles et al., 2020; Härdtle, 1995; Laflamme et al., 2016; Seastedt et al., 2008). Québec's ecological classification system defines potential vegetation as “a classification unit that includes the complete set of communities associated to the late-successional stage on a given site,” thus integrating vegetation dynamics within the concept (Saucier et al., 2009). Potential vegetation is defined on the basis of a particular set of tree species (early- and late-successional species) and understory indicator species that grow together on similar site conditions. Potential vegetation is considered permanent if soil and climatic conditions and the disturbance regime do not change (Saucier et al., 2009). It is mainly on the basis of this predictable successional trajectory that the concept has been challenged (Loidi & Fernández-González, 2012).

The concept of potential vegetation is widely used worldwide as the basic unit of vegetation classification systems (Härdtle, 1995; Loidi & Fernández-González, 2012; Sims et al., 1996). The distribution of potential vegetation (Fig. 8.2) can be illustrated along a toposequence, which divides the landscape into relatively homogeneous sections with respect to topography (slope, elevation), microclimate, drainage, surficial deposits, nutrient regime, soil type, disturbance regime, and forest dynamics (Banner et al., 1993; Barnes et al., 1982; Blouin et al., 2008; Rey, 1960; Rowe & Sheard, 1981; White 1973). This combined use of the physical environment and forest dynamics is termed a toposequence–chronosequence approach (Damman, 1964). Potential vegetation typical of mesic sites—zonal soils sensu Baldwin et al. (2020) and Pojar et al. (1987)—reflects the regional climate (temperature and precipitation). Other potential vegetation characterizing the toposequence is associated with regional climate and reflects the influence of specific local environmental characteristics (e.g., *Picea-Sphagnum* potential vegetation, which grows on poorly drained sites). Toposequences present a synthetic view of the ecological information about a landscape that can be used to produce an ecological map of potential vegetation. Such maps present a global view of the distribution of vegetation over a specific territory (Blouin & Grandtner, 1971; Küchler, 1964).

### 8.2.2 Ecological Classification: A Hierarchical Approach

Various hierarchical classifications have been developed for describing and managing ecosystems at different spatial scales, and each has its own nomenclature and levels (Bailey, 2009; Baldwin et al., 2020; Banner et al., 1993; Klijn & Udo de Haes, 1994; MacKenzie and Meidinger 2017; Powell, 2000; Sims et al., 1996). The ecological classification system developed in Québec (MFFP 2021; Saucier et al., 2009, 2010) was influenced by European phytosociological classification (Grandtner, 1966) and



**Fig. 8.2** Zonation of potential vegetation illustrated along a toposequence characterizing the southern part of the *Abies balsamea*–*Betula papyrifera* western subdomain, Québec. The species codes are defined in Fig. 8.7

Australian landscape classification (Jurdant et al., 1977). Québec’s ecological classification system consists of multiple hierarchy levels corresponding to various spatial scales. At the *continental scale* (~1,000,000 km<sup>2</sup>), vegetation zones and subzones correspond to very large territories characterized by a vegetation physiognomy and floristic composition that reflect macroclimatic conditions controlled by latitude and continentality. The boreal zone in Québec extends from the northern temperate to Arctic zones and is part of the eastern North American boreal floristic subdivision (Fig. 8.1a and b). It is subdivided from south to north into three subzones based on vegetation structure: closed boreal forest, open boreal forest, and forest–tundra (Fig. 8.1c). This latitudinal zonation is common in the boreal biome, which forms a forest belt around the Northern Hemisphere (Fig. 8.1a). At the *supraregional scale* (~100,000 km<sup>2</sup>), the bioclimatic domains and subdomains of Québec encompass large areas characterized by relatively homogeneous mesoclimatic conditions that are associated with the dominant potential vegetation of mesic sites and a natural disturbance regime (Fig. 8.1d). At the *regional scale* (~10,000 km<sup>2</sup>), each ecological region is characterized by the relative abundance of a potential vegetation and their respective physical environments (ecological types); for example, in the *Abies balsamea*–*Betula papyrifera* domain, ecological region 5B differs from region 5C by having a less rugged relief and much larger areas of organic deposits. This physical environment leads to a greater abundance of *Picea*–*Sphagnum* potential vegetation on wet organic soils and a lower abundance of *Abies*–*Betula* potential vegetation, which is normally typical of mesic sites in this bioclimatic domain. At the *landscape scale* (~100–1,000 km<sup>2</sup>), the regional landscape unit is a relatively homogeneous portion of land in terms of relief, altitude, surficial deposits, hydrography, and potential vegetation (Fig. 8.1f). The ecological district, delimited according to the same criteria as the regional landscapes but at a finer scale, is the basic mapping unit at all higher levels described above. The *local scale* (0.1–1 km<sup>2</sup>) is characterized by the potential vegetation, ecological type, and forest type. The ecological type combines a potential vegetation type and a physical environment type and is considered a stable

ecological unit. The forest type describes the observed vegetation using physiognomy, overstory tree species composition, and understory indicator species of site conditions.

In Québec, the boreal zone is subdivided into four bioclimatic domains distributed north to south along a temperature gradient (Fig. 8.3; Grondin et al., 2007, 2014; MFFP 2021; Richard, 1987; Saucier et al., 2009). The southernmost *Abies balsamea*–*Betula papyrifera* domain comprises mainly mixed boreal forests. The *Picea mariana*–moss domain, located to the north of this southernmost domain, is characterized by closed coniferous boreal forests. Further north, the *Picea mariana*–lichen domain is characterized by open forests, whereas the northernmost forest–tundra domain consists of a mosaic of forest and treeless communities where, with increasing latitude, forests are increasingly scattered and confined to sheltered locations up to the Arctic tree line (MFFP 2021; Payette, 1992; Rowe & Scotter, 1973). Most bioclimatic domains in Québec are subdivided further into two subdomains (west and east) because of differences in precipitation, physical features, and the dominant natural disturbance regimes, which produce differences in the dominant vegetation (Couillard et al., 2019; Grondin et al., 2007). The climate of the western subdomains is more continental and drier, therefore more susceptible to fire than the eastern subdomains, which have a maritime influence, as illustrated by ombrothermal areas (Fig. 8.4a; Rey, 1960; Richard, 1978).

At the local scale, the domain, subdomain, and ecological region are all characterized by a range of potential vegetation distributed along toposequences. In the toposequence of the southern part of the *Abies*–*Betula* w. subdomain (Fig. 8.2), the warmer hilltops host a mixed boreal vegetation (*Betula papyrifera*, *Abies balsamea*, *Picea glauca*, *P. mariana*) with the presence of *Acer rubrum*, which forms the *Abies balsamea*–*Acer rubrum* potential vegetation (hereafter *Abies*–*Acer*). The mesic midslopes (zonal soils) are occupied by the potential vegetation that best reflects the climatic conditions of the *Abies balsamea*–*Betula papyrifera* bioclimatic domain, which is the *Abies*–*Betula* potential vegetation. The lower subhydric, and often stoniest, slopes are occupied by the *Picea mariana*–*Abies balsamea* potential vegetation (*Picea*–*Abies*), and *Pinus banksiana* may also be present. Finally, the flatter areas host the *Picea mariana*–*Pinus banksiana* potential vegetation (*Picea*–*Pinus*) on till or sand, whereas the wet organic soils support the *Picea*–*Sphagnum* potential vegetation.

### 8.2.3 Ecological Classification: A Historical Perspective

A historical perspective (millennial scale) that considers the range of natural variability of vegetation can enhance the description of ecological classification units and forest dynamics. The concept of natural variability was introduced in the 1990s to incorporate an understanding of past spatial and temporal variability into ecosystem management (Cissel et al., 1999; Morgan et al., 1994; Powell, 2000; Swanson et al., 1994). Keane et al. (2009) defined the natural range of variability as “the variation of

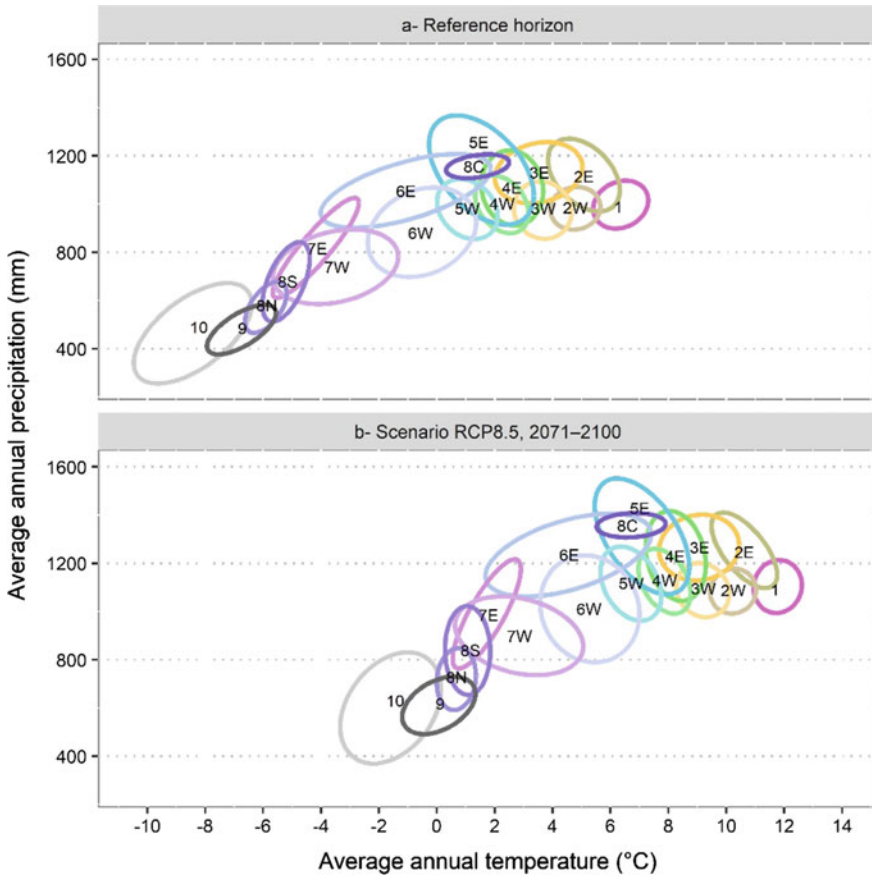




**Fig. 8.3** Vegetation zonation in Québec according to the level of zone, domain (1–10) and subdomain (W, west; E, east; N, north; S, south; C, coastal) (MFFP 2021 [CC-BY 4.0]; Saucier et al., 2009, with permission from Multimondes). The codes (1–10) are used in Fig. 8.4 to define the ombrothermal area of the bioclimatic subdomains. The red line is the northern limit of commercially productive forest (MRN 2013a)

historical ecosystem characteristics and processes (vegetation, disturbance regimes, climate) over time and space scales that are appropriate for management application.” An approach that considers the natural range of variability assumes (1) that ecosystems are complex and have a range of conditions that are self-sustaining. Beyond that range they move into disequilibrium; and (2) historical conditions can indicate ecosystem health (Keane et al., 2009; Kuuluvainen, 2002; Morgan et al., 1994).





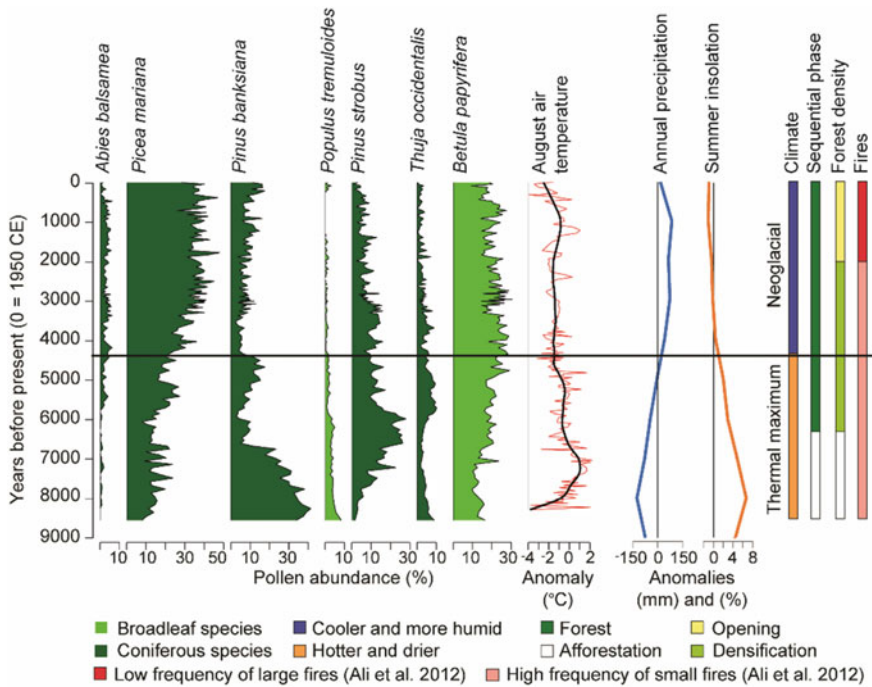
**Fig. 8.4** Ombrothermal area of the bioclimatic subdomains of Québec’s boreal forest (Fig. 8.3) for **a** the reference horizon (1969–1990) and **b** the scenario under RCP8.5 for 2100. The subdomain codes are composed of the latitudinal gradient (1–10) and the longitudinal gradient (E: east or W: west). Ellipses trace the 95% confidence level with a multivariate *t*-distribution of the data generated from global climate simulations (<https://www.climateinteractive.org/tools/world-climate-simulation/>). Some subdomains exhibit a wide climatic range because of their large area (6–10) or the highly variable habitat conditions (5E). The climate of the coastal forest–tundra subdomain (8C) overlaps with the southern portion of the *Picea mariana*–mosses subdomain because its vegetation is strongly influenced by rocky substrate and strong winds

Spatial variability is generally based on ecological units of the ecological classification determined at regional (Shorohova et al., 2011) to local scales (MacLean et al., 2021). Temporal scales define the short-, medium-, and long-term variability of forest composition, natural disturbances (mainly fire), and climate. This variability is reconstructed using dendrochronology and paleoecology (Chap. 2). Such information of historical conditions helps justify and validate management strategies elaborated in the context of climate change (Gillson & Marchant, 2014; Grondin et al., 2020;

Marcisz et al., 2018). In Québec, information regarding the long-term natural variability of ecosystems is derived from studies of postglacial vegetation, disturbance regimes, and climate (Payette, 1993; Richard, 1993).

At the *regional scale*, the Holocene history of the *Abies-Betula* w. subdomain was first characterized by an afforestation period, followed by the Holocene climatic optimum, marked by the expansion of the temperate species *Pinus strobus* and *Thuja occidentalis* (Fig. 8.5). Subsequently, the abundance of these species was reduced as their range contracted over the following 3,000 years in response to a cooler climate and possibly a relatively lower frequency of large fires (Ali et al., 2012). With time, the modern-day boreal forest became established. The last millennium has been characterized by a slight increase in *P. banksiana* and an opening of the forest cover (Ali et al., 2012; Asselin et al., 2016; Bajolle, 2019; Bajolle et al., 2018; Carcaillet et al., 2001; Fréchette et al., 2018; Hennebelle et al., 2018; Larochelle et al., 2018; Richard et al., 2020). Today, this region experiences recurrent spruce budworm outbreaks every 30–40 years (Saucier et al., 2009), and the fire cycle has been approximately 275 years for the 1890–2020 period (Couillard et al., 2022).

Studies of recent forest dynamics at the *local scale* (toposequence, Fig. 8.2) indicate that the *Abies-Betula* potential vegetation follows a pathway characterized by a high abundance of *B. papyrifera* in the early postfire successional stages followed by an increasing abundance of *A. balsamea* over time. Mature stands of *A. balsamea* are subject to gap dynamics, and some stands are severely affected by spruce budworm (*Choristoneura fumiferana*) outbreaks (Bergeron et al., 2014; Couillard et al., 2012; Navarro et al., 2018; Saucier et al., 2009). The dynamics of the *Picea-Abies* potential vegetation can follow one of two pathways. The first pathway has fires maintaining a stand dynamic dominated by *P. mariana*, with scattered *A. balsamea* in regeneration. The second pathway has a sufficiently long fire interval to support a successional dynamic toward *Picea mariana-Abies balsamea* stands. The *Picea-Pinus* potential vegetation is controlled primarily by frequent fires that generally favor the cycling of the same plant communities (recurrence dynamics), i.e., fires enable the post-disturbance recovery process to produce stands resembling those of the predisturbance state (Fig. 8.2; Couillard et al. 2016; Frégeau et al., 2015; Martin et al., 2018). This potential vegetation can be subject to a regeneration failure that shifts closed-canopy stands toward lichen woodlands. Various causal factors can explain this transformation; these factors include fire intensity (severe or light), successive fires within a short period, and the cumulative effect of an insect outbreak or logging immediately followed by a fire. For these areas, the return of closed-canopy stands is not possible without direct intervention, such as planting (Couillard et al., 2021; Girard et al., 2008; Pinno et al., 2013; Schab et al., 2021; Splawinski et al., 2019).



**Fig. 8.5** Postglacial vegetation and environmental data of the *Abies balsamea*–*Betula papyrifera* western subdomain, Québec. Pollen curves for selected coniferous (5) and broadleaf (2) tree taxa from Lake Lili (Bajolle, 2019) are coupled with a chironomid-based reconstruction of August air temperature at Lake Aurélie (Bajolle et al. 2018), a millennial reconstruction of annual precipitation and summer sunshine for the entire subdomain (Fréchette et al., 2018), and a regional summary of past climate, sequential cover type, forest density, and fires according to Bajolle (2019). Note that some taxa of the diagram are greatly underrepresented by the pollen they shed compared to their abundance in the field (e.g., *A. balsamea*, *P. tremuloides*, *T. occidentalis*), whereas others are highly overrepresented (e.g., *P. strobus*, *B. papyrifera*). The maximum values of the *P. banksiana* pollen curve at the base of the sequence (7,000–8,500 cal yr BP) reflect mainly a long-distance, northerly transport of extraregional pollen during the afforestation stage

### 8.2.4 Ecological Classification: A Foundation for Forest Ecosystem Management

Ecological classification is a framework that provides users with a straightforward structure to practice ecosystem-based management, facilitate modeling of the spatial distribution of ecosystems, and evaluate the impacts of changing climate and environmental conditions on vegetation (MacKenzie & Meidinger, 2018; Sims et al., 1996). For example, MacLean et al. (2021) present a new approach for establishing management guidelines defined for each potential vegetation in Nova Scotia. The forest management system developed in Québec also relies on potential vegetation

(Grondin et al., 2003; Saucier et al., 2010). More specifically, a specific silvicultural scenario is developed for each ecological type or groups of ecological type (MFFP 2020; MRN 2013b). Moreover, Québec forestry is oriented toward calculating the annual allowable cut for each management unit and for which potential vegetation plays a crucial role (Fortin & Langevin, 2010). Recently, a northern limit of commercially productive forest was established on the basis of relationships between fire regime, regeneration, and the time required for trees and forests to reach a merchantable volume, as calculated by ecological district and considering the potential vegetation (Fig. 8.3; Gauthier et al., 2015b; MRN 2013a).

Some management activities also consider reference conditions that serve to ensure long-term ecosystem health. In Québec, reference conditions for defining targets for old-growth proportions and forest composition were established at the spatial scale of bioclimatic subdomains (Fig. 8.3; Boucher et al., 2011). These reference conditions are based on studies of natural variability, mainly related to the fire regime of the last century or sometimes an even longer period (~200 years). The fire regime is documented from stand origin maps—which date the last fire—and from which a fire cycle and a proportion of old-growth forest are calculated. The long-term range of natural variability does not determine reference conditions because the links between long-term history and forest management must be clarified, as per Hennebelle et al. (2018). They showed that a relatively high proportion of old-growth forest existed throughout the Holocene in Québec. Such information would suggest that estimates based on the recent past are similar to those based on long-term history.

By using ecological classification information to manage Québec's forest ecosystems, forest managers (1) describe the natural forest, (2) compare it to the managed forest to highlight the differences induced by forest management, (3) translate this information into ecological issues to be resolved and (4) integrate these issues into management plans to reduce differences between natural and managed forests (Harvey et al., 2009; MFFP 2017). In the *Abies-Betula* w. subdomain, for example, the reference conditions indicate that even-aged mature stands (older than 80 years) represent 30% of the subdomain compared with 31% for uneven-aged old-growth forests (Boucher et al., 2011). These thresholds serve as guidelines to define targets for ecosystem management plans (MFFP 2015). However, Martin et al. (2021) showed that in the *Picea mariana*-mosses domain, when local features of potential vegetations are not sufficiently considered in forest landscapes, old-growth forests are preserved essentially on the least productive sites, such as forested wetlands (*Picea-Sphagnum* potential vegetation); the most productive sites are harvested. The system of protected areas in Québec, in which a variety of forest stands are protected, and relatively small, protected areas (i.e., biological shelters), which protect in particular old-growth forest, partially avoid the pattern of old-growth forests being found only on the least productive sites. It is clear that local conditions must be better integrated within targets set at the landscape scale, particularly with respect to complex forest structures and their attributes (e.g., deadwood). In the context of ecosystem management, adequate protection and adapted silviculture (partial cutting) of old-growth forests specific to each potential vegetation are key to ensuring the resilience and

adaptive capacity of boreal forest ecosystems under global change (Martin et al., 2018; Shorohova et al., 2011).

Overall, ecological classification is a valuable tool for summarizing spatiotemporal information to help guide effective conservation and forest management practices (Keane et al., 2009; Loidi & Fernández-González, 2012; MacLean et al., 2021). However, ecological classification was developed assuming a relatively stable climate and that potential vegetation recovers its initial structure and composition after a disturbance (Loidi & Fernández-González, 2012; Powell, 2000; Saucier et al., 2009). Future climate change challenges the validity of predictable successional pathways and stable potential vegetation dynamics. This assumed predictability requires a reanalysis of ecological classification in light of the uncertainties associated with climate change by considering different successional trajectories for a given potential vegetation. Some mechanisms already exist in Québec's ecological classification system, particularly in forest mapping activities, to account for observed changes in potential vegetation, such as shifts from closed-canopy stands (*Picea-Pinus* potential vegetation) to lichen woodlands (*Picea mariana*–lichens) caused by regeneration failure. Over the past 15 years, several studies have attempted to improve our understanding of natural variability at regional and local scales (Sect. 8.2.3; e.g., Couillard et al. 2016; Fréchette et al., 2018). These studies have greatly contributed to our knowledge of forest dynamics and the concepts of stability and predictability associated with Québec's ecological classification. However, vegetation changes that can be anticipated because of ongoing climate change have yet to be considered. The following section reflects on strategies for bringing ecological classification into the future.

### 8.3 Ecological Classification and Climate Change

Future environmental conditions are expected to differ markedly from present-day conditions. This shift will occur faster and at a greater magnitude than at any time since the end of the last glaciation (IPCC 2021). Certain projections have ecosystems falling beyond their natural range of variability (Seastedt et al., 2008). In North America, Keane et al. (2020) projected significant changes for potential vegetation in Montana, United States, along an altitudinal gradient. Wang et al. (2012) showed that suitable habitats in British Columbia, Canada, for grasslands, dry forests, and moist continental cedar–hemlock forests would expand and habitats for boreal, subalpine, and alpine ecosystems would decrease. Moreover, in the adjacent province of Alberta, wildfire activity is projected to accelerate the conversion of about half the province's upland mixed and coniferous forests to more climate-adapted deciduous woodlands and grasslands by 2100 (Stralberg et al., 2018). In Siberia, the expansion of dark coniferous forest—currently found in the southern boreal forest—into light coniferous forest (*Larix laricina* northern forest, often underlain by permafrost) has also been documented (Gauthier et al., 2015a; Kharuk et al., 2007).

The consequences of climate change for Canada's boreal forests will be numerous; these impacts include changes in forest composition and growth rates, shifts in disturbance regimes, and, ultimately, altered biodiversity levels (Price et al., 2013). These climate-induced changes in forest ecosystems thus represent a major challenge for forest management. Climate projections must be translated into concrete information and guidelines to prepare and assist managers in adapting forest management plans to these changes (Forestier en chef, 2020; Gauthier et al., 2014, 2015a; Millar et al., 2007; Nagel et al., 2017; Puettmann, 2011; Thiffault et al., 2021). In this context, climate-induced changes could be forecast from the information and models available at the different levels of the ecological classification system already in use in forest ecosystem management. The following questions are thus appropriate to consider in the discussion below:

- Question 1. Will climate change modify vegetation along the various ecological classification scales, i.e., from bioclimatic subdomains to potential vegetation?
- Question 2. Can knowledge derived from ecological classification help respond to the challenges inherent to climate change?

### **8.3.1 A Three-Step Analysis to Characterize Changes in Climate, Species Sustainability, and Forest Dynamics**

In the following section, we present an analytical approach that will address these two questions using information generated at different levels of ecological classification. This approach proceeds through several steps that we consider to be prerequisites for determining silvicultural strategies in the context of climate change. Our method can be adapted according to specific regional and local contexts. The aim is to illustrate how climate change may impact forests at the relevant scale by focusing on the *Abies balsamea*–*Betula papyrifera* w. subdomain in the province of Québec and comparing current vegetation patterns with those developed under the RCP8.5 scenario from now to 2150. As explained by the IPCC (2021),

Representative Concentration Pathways (RCP) are scenarios that include time series of emissions and concentrations of the full suite of greenhouse gases (GHGs) and aerosols and chemically active gases, as well as land use/land cover ... RCPs usually refer to the portion of the concentration pathway extending up to 2100, for which Integrated Assessment Models produced corresponding emission scenarios.

The RCP8.5 scenario is generally considered the most severe anthropogenic climate forcing scenario. Indeed, under this pathway, the forcing reaches  $8.5 \text{ W}\cdot\text{m}^{-2}$  in 2100 and continues to increase for some time afterward. Under this scenario, the temperature in the studied bioclimatic subdomain will increase 7–8 °C and precipitation 7% by 2100 (Boulanger & Pascual Puigdevall, 2021). The *Abies*–*Betula* w. subdomain will thus likely experience a longer fire season and more fire-prone weather conditions, which could increase the annual area burned by 2 to 4 times by the end of the century, as projected for various regions in Canada (Boulanger

et al., 2014). Ecological processes including tree growth, regeneration, interspecific competition, and forest productivity could also be affected by these climate changes; these altered processes could ultimately modify the forest assemblage currently defining each potential vegetation (Boulanger & Pascual Puigdevall, 2021). Our approach involves acquiring information about future climate analogs (Step 1), defining potential impacts on tree species–habitat suitability at the subdomain scale (Step 2), and finally considering local information at the potential vegetation scale to assess changes in forest dynamics (Step 3).

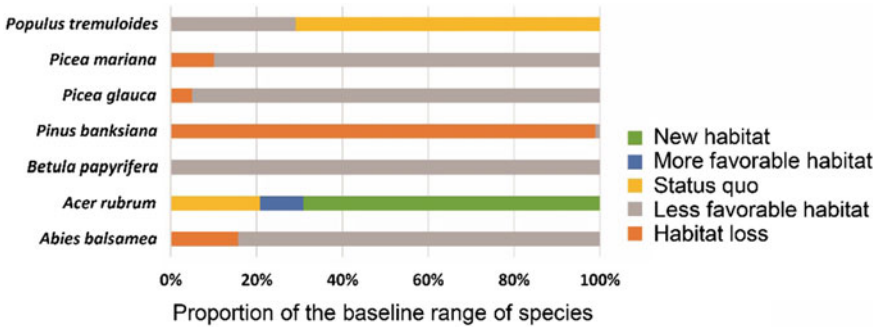
### 8.3.1.1 Step 1. Identifying Contemporary Climate Analogs

A climate analog is a simple tool for visualizing the magnitude of climate change at a relatively large spatial scale. This tool “identifies locations for which historical climate is similar to the anticipated future climate at a reference location” (Grenier et al., 2013). Here, the approach shows how the temperature and precipitation of the *Abies-Betula* w. subdomain (Fig. 8.3) will be modified under RCP8.5 during the 2071–2100 period (code 5 W in Fig. 8.4b). Under this scenario, this subdomain will experience a climate analogous to that currently experienced by the *Acer saccharum–Carya cordiformis* bioclimatic subdomain (code 1 in Fig. 8.4b), located ~500 km to the south. Northern temperate hardwoods characterize this latter area, with several thermophilous species (e.g., *Quercus*, *Carya*, *Tilia*) reaching the northernmost limit of their range and small-gap dynamics being the most common natural disturbance. Therefore, this climatically analogous region includes very different tree species assemblages and experiences markedly different disturbance dynamics than those currently observed in the *Abies-Betula* w. subdomain.

### 8.3.1.2 Step 2. Assessing Future Tree Species Habitat Suitability at the Regional Scale

Because the climate analog provides only partial insight into the impacts of climate change on forest ecosystems, it is important to identify to what extent the projected climate in the *Abies-Betula* w. subdomain will be suitable for species currently observed in the area and those taxa that could potentially migrate into the subdomain. Niche models (Boisvert-Marsh et al., 2014; Périé & de Blois, 2016; Périé et al., 2014) are used to assess the climatic vulnerability of the dominant tree species of the subdomain (Fig. 8.6). Under the RCP8.5 climate scenario for 2071–2100, niche models project that much of this subdomain will become a less favorable habitat for most conifers and *B. papyrifera*. *Populus tremuloides* habitat will remain present at a similar abundance (status quo), whereas the situation appears more critical for *P. banksiana*, as the model indicates a less favorable habitat over much of the subdomain. *Acer rubrum*, a thermophilous species that favors the warmer hilltops (Fig. 8.2) of the southern part of the western subdomain, has a distinct profile from the other species as new suitable habitats will become available. These changes in habitat





**Fig. 8.6** The impact of climate change on tree habitat suitability under scenario RCP8.5 (2071–2100) as defined by niche models. The baseline (1969–1990) range of a species is the total area (km<sup>2</sup>) of all cells where the baseline average model predicted a suitable habitat for that species in the *Abies balsamea*–*Betula papyrifera* western subdomain and within the *Abies balsamea*–*Betula papyrifera* potential vegetation

suitability suggest that species that have traditionally defined entire regional vegetation assemblages could become less adapted to their particular regions, resulting in significant impacts on ecosystems.

### 8.3.1.3 Step 3. Assessing Local Change in Forest Composition and Dynamics

How will the climate-induced vulnerability of these species affect the dynamics of specific potential vegetation? Evaluating such impacts requires a more thorough assessment of climate-induced changes on processes at the stand- (e.g., interspecific competition, soil characteristics, productivity) and landscape- (e.g., seed dispersal, natural disturbances) scales, which govern finer-scale forest dynamics. Differences in soil characteristics, topographical position, and current tree species assemblages may thus impact how specific potential vegetation will respond to climate change. Climate-induced changes in natural disturbance regimes are another key component to consider. Simulating changes in the disturbance regime enables assessing possible future alterations in the natural range of variability (Fig. 8.5), which is not explicitly accounted for within niche models. Using a forest landscape model (LANDIS-II; Scheller et al., 2007), we projected the future long-term compositional change of two contrasting potential vegetation types characteristic of the *Abies*–*Betula* w. subdomain, i.e., *Abies*–*Betula* and *Picea*–*Pinus* potential vegetation. We simulated 130 years, starting in 2020, under baseline and RCP8.5 climate scenarios. To assess the impact of changes in the natural disturbance regime, we simulated future forest landscapes under RCP8.5, considering the current fire regime (status quo) and a future fire regime under climate change projections.

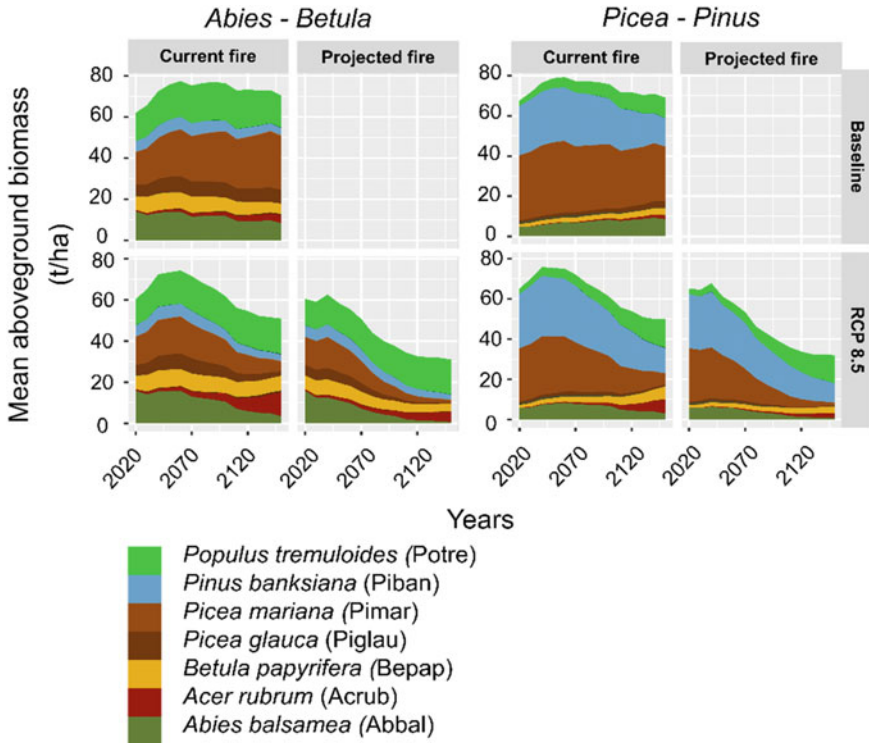
We found that under the RCP8.5 projected climate conditions and regardless of the fire regime, i.e., current or projected fires (Fig. 8.7), there is a significant decrease

in total biomass after 2070 for both potential vegetation types, primarily caused by a large decrease in the biomass of *A. balsamea* and/or *P. mariana*, the two dominant boreal species. These decreases in total biomass are likely to be even greater when considering the concomitant climate-induced change in fire regimes (Boulanger et al., 2014, 2016). Our analysis distinguishes different alterations in forest dynamics specific to each potential vegetation. For instance, *A. rubrum* and *B. papyrifera* are projected to sharply increase in the *Abies-Betula* potential vegetation, whereas *P. tremuloides* biomass will remain stable or increase only slightly. The increase of *A. rubrum* is greater in the current fire regime than in the projected fire regime. In the *Picea-Pinus* potential vegetation, however, *P. tremuloides* sharply increases, whereas *A. rubrum* remains a minor component of the forest landscape. Still here, *A. rubrum* is greater in the current fire regime and after 2100. Therefore, by 2100, the *Abies-Betula* potential vegetation will be dominated mainly by conifers accompanied by *B. papyrifera*, *P. tremuloides*, and *A. rubrum*. Beyond 2100, and given the projected fires, broadleaf species (*P. tremuloides*, *B. papyrifera*, *A. rubrum*) will dominate. In the *Picea-Pinus* potential vegetation, *P. tremuloides* and *P. banksiana* will be the most common species. The projected dynamics for both potential vegetation types are consistent with contemporary trends found in other studies (Boisvert-Marsh et al., 2014; Brice et al., 2019; Fisichelli et al., 2014; Iverson et al., 2008; Jain et al., 2021).

### 8.3.2 *Ecological Classification and Climate Change: The Main Issues*

Several conclusions can be drawn from this multistep analysis to answer both questions posed above. First, we are interested in specifying how climate change will modify the current vegetation (Question 1). At the regional scale, the future vegetation of the *Abies-Betula* w. subdomain will not correspond to that identified by its climate analog by the end of the century (Fig. 8.4 and Step 1). Given dispersal limitations (Iverson et al., 2011) and forest inertia (Brice et al., 2020), it is unlikely that the thermophilous species associated with the *Acer saccharum*–*Carya cordiformis* subdomain will keep pace with the >500 km northward migration of the ombrothermal area (Boulanger & Pascual Puigdevall, 2021; Taylor et al., 2017). With climate niches of most boreal species deteriorating within the *Abies-Betula* w. subdomain, this scenario will likely introduce a *climate debt* (Taylor et al., 2017), with current species assemblages being strongly maladapted to future climate conditions. Under such conditions, it is improbable that regional vegetation will be at a climate equilibrium, at least for the next several decades.

At the local scale (Fig. 8.7), our model (LANDIS-II) also suggests that increased disturbance rates will cause a decline in the biomass of coniferous species and provide pioneer deciduous tree species a competitive advantage because they can reproduce vegetatively after a disturbance (Boulanger et al., 2019; Landhäuser et al., 2010). If stands of *Abies-Betula* potential vegetation become dominated by *P. tremuloides*



**Fig. 8.7** Mean stand aboveground biomass of forest species in the *Abies balsamea*-*Betula papyrifera* western subdomain according to the LANDIS-II model and projections using the RCP8.5 scenario. The panels represent the *Abies-Betula* potential vegetation in midslope (left panel) and the *Picea-Pinus* potential vegetation (right panel) in flat areas (Fig. 8.2). In the bottom left chart of each panel, the current fire regime remains relatively unchanged in the future. In the bottom right chart of each panel, the fire regime is altered in relation to projected climate change

and *A. rubrum*, will this community be considered (1) a mid-successional stage of the *Abies-Betula* potential vegetation with a greater abundance of *A. rubrum* than expected under current climate conditions, (2) a transition stage toward an *Abies-Acer* potential vegetation, or (3) a new potential vegetation? Knowledge of forest dynamics under climate change is not sufficiently advanced to project which of the three possibilities will prevail in the future.

Furthermore, we project that warmer conditions should favor the growth and reproduction of thermophilous species (e.g., *Acer* spp.), which will then be more likely to outcompete boreal species in the potential vegetation where they are currently thriving. Such conditions could favor the migration of *Abies-Acer* potential vegetation to mid-hillside, a topographic position where conditions are not currently suitable to support such potential vegetation in the *Abies-Betula* w. bioclimatic subdomain (Fig. 8.2).

For *Picea-Pinus* potential vegetation, dynamic modeling (LANDIS-II) suggests a significant amount of *P. banksiana* will be maintained (Fig. 8.7), even when niche models predict a less favorable habitat for this species. This divergence relates to the niche models relying only on climate, whereas LANDIS-II considers several other elements, including the impacts of the fire regime.

Climate change may therefore affect vegetation as we know it today at different scales of ecological classification, from the bioclimatic subdomain to the potential vegetation level. At the same time, a great deal of uncertainty remains regarding future climate and vegetation changes. The high uncertainty means that anticipated changes in vegetation under climate change will likely be intermediate, falling between those projected under baseline conditions (minimum change) and those of RCP8.5 (maximum change). The variability of climate scenario projections remains great given the highly unpredictable global, political, and social conditions that will exert a strong influence (IPCC 2021). Moreover, changes in climate conditions are likely to evolve over several decades; therefore, it could take decades or even centuries for highly *inertial* forest ecosystems, i.e., those resistant to change, to modify significantly under future climate conditions. Finally, climate and forest models each have their predictive limitations and hence intrinsic uncertainty in addition to the uncertainty of climate scenarios.

How can knowledge of natural variability (Sect. 8.2.3; Fig. 8.5) help us better interpret future vegetation changes? Past patterns in vegetation illustrate previous dynamics that can recur in the future and, therefore, help researchers formulate hypotheses about the limits of variability that species have experienced in the past and that may be exceeded in the future (Gillson & Marchant, 2014; Grondin et al., 2020; Marcisz et al., 2018).

1. In the *Abies-Betula* w. subdomain, both types of modeling, corresponding to the niche models (Step 2) and the LANDIS-II model (Step 3), suggest that by 2100 there will be fewer favorable habitats for conifers and a gradual decline in their biomass. If the abundance of conifers becomes relatively low, as predicted for RCP8.5, then there will be fewer conifers in the landscape than over the past several thousand years (Fig. 8.5). It is difficult to compare the Holocene abundance of coniferous species with that expected in the future; thus, we cannot determine whether future landscapes will move beyond the range of their past natural variability.
2. *Populus tremuloides* and *Pinus banksiana* are predicted to be abundant in the future (Fig. 8.7), albeit within a different context compared with the early Holocene when fires were frequent, and vegetation was in the afforestation stage (Fig. 8.5). After the afforestation period, and mainly from 5,000 to 2,000 yr BP, broadleaf species (*B. papyrifera*) remained abundant; the future *Abies-Betula* w. subdomain may present a similar vegetation pattern (Fréchette et al., 2018).
3. Long-term regional natural variability indicates that *P. banksiana* has increased over the last millennium in western Québec (Fig. 8.5). This trend should continue under climate change with an increased fire incidence given the great ability of *P. banksiana* to renew itself in a recurrence dynamic (Couillard et al. 2016;

Frégeau et al., 2015; Payette et al., 2012). LANDIS-II modeling also supports this hypothesis, although there is a significant decrease in biomass, indicating that the stands will be younger. It is difficult to comment on the spatial variation of *Pinus banksiana* stands; however, one hypothesis holds that this species will increase in geographic extent (Schab et al., 2021; Splawinski et al., 2019). All these dynamics depend on the initial species composition at each site (greater or lower abundance of *P. banksiana*), the fire cycle, and the species' response to a potentially shortened cycle.

4. Finally, the Holocene history of the *Abies-Betula* w. subdomain shows that *P. strobus* and *T. occidentalis* were historically well represented in the landscape during warmer climatic periods (Fig. 8.5). Both species are not considered in contemporary modeling because of their low current abundance. On the other hand, habitat models suggest that habitats will be suitable for the expansion of these species (Périé et al., 2009). This migration will nonetheless depend on multiple factors, e.g., the inertia of the vegetation.

Overall, comparing past and future scenarios allows us to better situate the potential magnitude of future vegetation change (e.g., the decreased abundance of coniferous species), base future projections on specific patterns observed during the Holocene, e.g., broadleaf species such as *Betula papyrifera*, and propose hypotheses regarding future vegetation change, e.g., *P. banksiana*, *P. strobus*, *T. occidentalis*.

Can knowledge generated by ecological classification (Sect. 8.1) be used to address the challenges inherent to climate change (Sect. 8.2)? (Question 2). Despite projected changes in forest ecosystem composition and dynamics at various spatial scales, we argue that current ecological classification can still provide a useful framework to inform future forest management (Loidi & Fernández-González, 2012). The characteristics that currently define potential vegetation at the local scale (soil, surficial deposits, position, and slope) will continue to develop independently in future ecological domains, e.g., *Abies-Betula* versus *Picea-Pinus*. The assemblages within each potential vegetation will evolve according to the competitive abilities of its various species. The future toposequence will continue to show a gradation of potential vegetation along a physiographic gradient. Each potential vegetation will have a specific future successional pathway, albeit different from the current pathway (Fig. 8.7). Moreover, each potential vegetation distributed along the toposequence will continue to be distinct from other assemblages, and the links with the current vegetation will likely remain for a long duration. Studies on how ecosystem inertia could limit future vegetation dynamics must be undertaken. Overall, the hierarchy of ecological classification (regional vs. local) and the specificity of each potential vegetation will continue to be paramount for strategic forest planning under climate change.

## 8.4 Conclusions

It is evident from our first question (Question 1) that climate change will modify the principles of ecological classification. Given the expected modifications in forest vegetation composition and dynamics, we must integrate the expected variability and uncertainty of forest vegetation composition and dynamics into the current stable and predictable potential vegetation dynamics. The results presented in this chapter, through a three-step analysis, provide a preliminary view of the links between current ecological classification and forest management under climate change. This information is limited in scope and should be interpreted with caution. New knowledge, more complete than that currently available, will be added rapidly, especially through projects that address potential vegetation dynamics under climate change. In regard to the possible use of ecological classification in the context of climate change (Question 2), we documented that each potential vegetation could change and be characterized by its own successional trajectory. Thus, we must maintain and manage this diversity. Ecological classification can and must assist forest managers in estimating potential future changes despite inherent variability and uncertainty. The challenge is to select target species and silvicultural scenarios at the regional and local levels on the basis of the different trajectories that current forest stands are projected to follow under increased temperatures and precipitation amounts and changing disturbance regimes. “Decision makers within any institution, therefore, have to find their own way through sometimes conflicting information and face the prospect of planning with and for uncertainty” (Gauthier et al., 2014).

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