UNIVERSITÉ DU QUÉBEC EN ABITIBI-TÉMISCAMINGUE

LA DYNAMIQUE DU PAYSAGE FORESTIER BORÉAL MIXTE EN RÉPONSE AUX FEUX ET À L'AMÉNAGEMENT FORESTIER SOUS L'INFLUENCE DES CHANGEMENTS CLIMATIQUES

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MIXEDWOOD BOREAL FOREST LANDSCAPE DYNAMICS IN RESPONSE TO FIRE AND FOREST MANAGEMENT UNDER THE INFLUENCE OF CLIMATE CHANGE

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I dedicate this thesis to all my big family. To my son Emilio, my mother Beatriz, and my siblings Cata, Natalia, and Mile.

PREFACE

This thesis contains five chapters, three of which are written in manuscript format (chapters 2 to 4). ***Chapter 1*** presents the background and justification of the study. In the ***Chapter 2***, through Landsat imagery from 1985 to 2013, the evaluation of the effect of forest management on landscape heterogeneity of mixedwood boreal forests of northeast Canada (Abitibi Plain) during the last decades was undertaken. In ***Chapter 3***, the forest landscape model LANDIS-II was used to simulate the forest composition, aboveground biomass (AGB) and aboveground net primary production (ANPP) of the boreal forests of northeast Canada under different climate warming scenarios and forest management intensities. In ***Chapter 4***, the spatially explicit model Landis II was used again to simulate successional trajectories in response to fire and forest management under climate change scenarios, as well as to identify the mixed effect of fire and forest management on the forests' spatial distributions; and finally, in the ***Chapter 5*** results were summarized and possible applications for boreal forest management are discussed.

The three chapters (chapters 2, 3 and 4) of this thesis are presented in a scientific journal article format. The second chapter is already published in the *Canadian Journal of Remote Sensing*.

The first author led field work, laboratory work, data analysis, and writing. Dr. Osvaldo Valeria was actively involved in the acquisition and processing of the spatial data and designed all analyses developed, and actively supervised all work conducted by the first author in the three chapters. Furthermore, Dr. Osvaldo Valeria and Dr. Louis de Grandpre participated in manuscript writing. Dr. Yan Boulanger and Dr. Dominic Cyr provided essential advice on the modelling study with Landis II. Dr. Jorge Ramirez participated in the data processing and manuscript writing of chapters 3 and 4.

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RÉSUMÉ

Le paysage forestier boréal dominé par la forêt mixte est le résultat des interactions complexes des conditions abiotiques, de la succession et des régimes de perturbations, qu'influencent les processus écologiques qui opèrent au niveau du peuplement et à l'échelle du paysage. Les changements climatiques prévus pour les prochains siècles devraient modifier les régimes de perturbations naturelles tels que les incendies de forêt, affectant le flux de bois vers les marchés de produits ligneux. C'est pour cette raison que la gestion des forêts boréales mixtes a suscité une préoccupation généralisée concernant le maintenant de sa biodiversité, sa résilience et sa capacité d'adaptation à conserver les avantages sociaux et économiques qu'elles procurent à la société. Cette préoccupation a conduit à proposer une modification de l'exploitation dite traditionnelle vers une approche d'aménagement forestier écosystémique, qui tente d'imiter les patrons de perturbations naturelles afin de conserver la forêt à l'intérieur des limites historiques de variation. En dépit de tous ces efforts, au cours des dernières décennies dans l'est du Canada, la forêt mixte est passée d'un paysage dominé par la forêt mature vers un paysage fragmenté avec une quantité croissante de peuplements à prédominance de feuillus. Toutefois, notre compréhension des variations spatiotemporelles des paysages forestiers et des caractéristiques des peuplements issus après feu et après récolte demeure insuffisante. L'objectif de cette thèse était de caractériser les changements à moyen terme des mosaïques de forêts mixtes de l'est du Canada et d'améliorer notre connaissance des relations entre ces changements et les variations attendues des feux de forêt sous l'effet des changements climatiques ainsi que du régime de perturbations anthropiques. La première étape pour atteindre cet objectif a été de comprendre la dynamique historique du paysage dans un gradient nord-sud de forêts mixtes de l'est du Canada. Pour cette raison, le chapitre 2 présente une étude historique de l'hétérogénéité du paysage forestier, mesuré en termes de

composition et configuration du paysage ainsi que leur interaction avec les feux de forêt historiques dans un paysage aménagé. En utilisant les images Landsat de 1985 à 2013, la cartographie et des mesures spatiales nous avions suivi l'évolution de la composition dans le temps des forêts selon 4 classes : résineux, résineux mixtes à dominance résineuse, mixte à dominance feuillue et feuillus. Cette étude montre que la composition résineuse a dominé la mosaïque en 1985 et représentait un tiers de la superficie de l'étude. De plus, la classe résineuse a enregistré la plus forte diminution avec un taux de 1,7% par an par rapport aux autres couvertures forestières. Les mesures indiquent que les forêts résineuses matures, qui dominaient auparavant le paysage dans l'est du Canada, ont été principalement transformées par les pratiques forestières en un paysage plus hétérogène et fragmenté. Le chapitre 3 détaille le modèle de paysage forestier LANDIS-II utilisé pour évaluer les relations entre les feux de forêt et l'aménagement forestier sous scénarios de réchauffement climatique futur (RCP 2.6, RCP 4.5 et RCP 8.5), via la biomasse forestière (AGB) et la productivité primaire nette (ANPP). Les projections du modèle ont démontré que les régimes de perturbation sont les variables les plus significatives qui déterminent les variations de l'AGB et de l'ANPP. L'aménagement forestier apparait comme le facteur le plus important des variations observées dans les forêts du sud du gradient comparativement au nord, probablement parce que cette région présente des forêts plus âgées et avec une composition d'intérêt commercial pour respecter la possibilité forestière. À l'opposé, les forêts du nord du gradient présentaient un effet mixte du changement climatique et de l'aménagement forestier sur l'AGB et l'ANPP, probablement parce qu'un grand nombre de zones propices à la récolte avaient déjà été brulées, limitant ainsi la quantité du territoire disponible pour la récolte. En général, bien que l'on s'attende à une augmentation des superficies brulées en raison du changement climatique, l'intensification de l'aménagement forestier semble être le principal facteur de l'augmentation des feuillus et des peuplements mixtes et de la diminution des

peuplements résineux, ainsi que de la diminution de la AGB et ANPP, principalement dans les forêts du sud. Le chapitre 4 détaille l'utilisation de LANDIS-II pour modéliser les trajectoires de succession de la gestion post-feu et post-aménagement suite aux scénarios de changement climatique. L'évolution de l'AGB post-perturbations nous a permis de constater que, contrairement aux perturbations causées par les feux, l'aménagement forestier a modifié les voies de succession conduisant à des changements de composition allant de forêts à prédominance de feuillus à une forêt mixte favorisant la prévalence d'essences de feuillus, même après 300 ans. Cette tendance est exacerbée par les scénarios de changement climatique, qui donneront un avantage aux forêts dominées par des espèces intolérantes à l'ombre, en particulier dans les écorégions où elles sont peu présentes (forêts mixtes centrales et septentrionales de la zone d'étude). De plus, ces pratiques d'aménagement conduisent à des indices de formes de forêt plus sinueuses au niveau spatial du paysage, indiquant une augmentation de la fragmentation. Les résultats obtenus mettent en évidence l'échec des pratiques d'aménagement actuelles à imiter la succession naturelle après feu et compromettent le maintien des biens et des services de ces écosystèmes. Les changements climatiques prévus pour le prochain siècle devraient modifier les régimes de perturbations naturelles (tels que les feux de forêt). En outre, il existe des activités d'aménagement telles que la récupération du bois à des fins industrielles qui perturbent les forêts pour satisfaire la demande croissante de produits ligneux. Nos résultats suggèrent que l'approvisionnement en bois à long terme serait menacé dans l'est du Canada. Par conséquent, certaines stratégies devraient être mises en œuvre pour adapter l'aménagement des forêts aux changements climatiques attendus et à maintenir l'avenir des forêts.

MOTS-CLÉS : Forêt boréale et forêts mixtes, incendies, aménagement forestier, hétérogénéité du paysage, composition forestière, configuration du paysage, biomasse aérienne, productivité primaire nette, succession, métriques du paysage, Landis II.

ABSTRACT

Ecological processes that operate from stand to landscape in mixedwood boreal forest landscape is the result of the interactions of abiotic conditions, succession, and disturbance regimes. Projected climate change for the next century is expected to alter natural disturbance regimes such as wildfires, along with the continuous industrial harvesting of forests to supply a growing demand of woody products. Parallel to the expected effect of a changing fire regime and an intensification of forest management on the mixedwood boreal forests, there has been a generalized concern about the maintenance of forest ecosystem functions by maintaining its biodiversity, resilience, and adaptive capacity to retain the social and economic benefits that they provide. The above-mentioned concern has led to propose the modification of traditional harvesting towards a Forest Ecosystem Management approach that attempts to emulate natural disturbances to mimic the forest characteristics produced by such disturbances. Despite all these efforts, during the last decades in eastern Canada, the mixedwood forest landscape has changed from a mature forest landscape to a fragmented one with a growing quantity of hardwood dominated stands. Nevertheless, our understanding of the spatial and temporal variations of mixedwood forest landscapes and the characteristics of the second growth post-fire and post-harvesting stands is still insufficient. The objective of this thesis was to characterize the mid-term changes of mixedwood forest mosaics in eastern Canada and to improve our knowledge regarding the relationships between such changes and the expected change in fire frequency as a consequence of climate change and forest management. The first step to achieve this objective was to understand the landscape history in the north-south gradient of mixedwood forest mosaics in eastern Canada (western Quebec), for that reason Chapter 2 presents an historical study of the forest landscape heterogeneity, understood as landscape composition and configuration, in interaction with historical fires and forest

management. Using Landsat imagery from 1985 to 2013, maps and spatial metrics were obtained to track Conifer, Mixed-Conifer, Mixed-Hardwood, and Hardwood covers through time. This study evidences that the Conifer-dominant cover dominated the mosaic in 1985 and accounted for one-third of the study area. Nevertheless, this class showed the greatest decrease (1.7%/yr) in comparison with the other forest covers. The metrics indicated that the fire-influenced, old-growth conifer forests that previously dominated the landscape in eastern Canada were transformed by forestry practices into a more heterogeneous and fragmented landscape. Chapter 3 evaluates the relationships between fires and forest management in future climate warming scenarios (RCP 2.6, RCP 4.5 and RCP 8.5), and forest aboveground biomass (AGB) and aboveground net primary productivity (ANPP) using the forest landscape model Landis-II. The model projections suggested that disturbance regimes are the most significant variables that determine variations on AGB and ANPP. Between them, forest management will be the most important factor in the southern forests, probably because this region shows more stands with the age and composition required by each harvesting prescription to deal with the annual allowable cut volume than the northern forests. On the contrary, the northern forests presented a mixed effect of climate change and forest management on AGB and ANPP, probably because many of the areas suitable for harvesting were previously burned limiting in this way the amount of area available for harvesting. In general, although an increase in burned area is expected due to climate change, the intensification of forest management seems to be the most important driver of the increase in hardwoods and mixed stands and the decrease in conifers stands, as well as the decrease of AGB and ANPP, mainly in the southern forests. Chapter 4 models with Landis-II the post-fire and post-harvesting successional pathways under climate change scenarios (post-disturbance AGB by species and time). It was found that contrary to fire, forest management modifies the successional pathways leading to composition changes from initial hardwood predominance forests

to a mixed forest favoring the prevalence of hardwood species, even after 300 years. This trend is exacerbated under climate change scenarios, which will give advantage to forests dominated by shade-intolerant species, especially in the ecoregions where they have low presence (central and northern mixedwood forests). Additionally, these management practices are leading to more sinuous forest shapes at landscape spatial level indicating an increase in fragmentation. The obtained results highlight the failure of the current forest management practices to emulate natural postfire succession and the risk that these activities imply for the maintenance of ecosystem goods and services. Projected climate change for the next century is expected to alter natural disturbance regimes (such as wildfires). Additionally, there are anthropic activities such as wood harvesting for industrial purposes disturbing forests to supply the increasing demand for woody products. The aforementioned scenario suggests that timber supply would be at risk in eastern Canada, therefore, some strategies should be implemented to adapt forest management to climate change and the future of forests.

KEYWORDS: Mixedwood boreal forest, Fires, Forest management, Landscape heterogeneity, Forest composition, Landscape configuration, Aboveground biomass, Aboveground net primary productivity, Successional pathways, Landscape metrics, Landis II.

CHAPTER I

INTRODUCTION

1.1 Problem statement

Mixedwood forest occupies 24% of the total boreal forest in Canada (Baldwin et al. 2012). This area plays a prominent environmental, economic, and social role in the northwest of Quebec. This forest supports the cycles of carbon, water, plant life, and wildlife. Furthermore, this forest is one of the main economic drivers in several regions of Quebec, with forestry and wood processing industry, as well as the base of recreational and tourism activities (Coulombe et al. 2004). Mixedwood boreal forest and its successional pathways are driven by disturbance regimes and climatic, topographic, and edaphic conditions that favor the formation of closed canopies dominated by trembling aspen (Populus tremuloides Michx.) or white birch (Betula papyrifera Marshall) in early successional stages, jack pine (Pinus banksiana Lamb.), black spruce (Picea mariana Mill.) or white spruce (Picea glauca (Moench) A. Voss) in mid-successional stages, and balsam fir (Abies balsamea (L.) Mill.) in mature successional stages (Chen and Popadiouk, 2002).

The mixedwood forest dynamics have been mainly driven during last decades by wildfires and forest management when compared with the other natural and anthropogenic disturbances. There are in average 7,500 fires per year in the boreal forest of Canada, which at the landscape scale, burn about 2.4 million ha (Canada 2017). Most of these fires are stand-replacing events, so their frequency, severity, and area disturbed largely affect forest succession and shaped the forest landscape (Burton et al. 2003, De Grandpré, Gagnon, and Bergeron 1993, Payette 1992, Seedre 2009). For example, Bergeron et al. (2004) showed that high fire frequency was associated with the high proportion of young forest stands dominated by early-successional species (hardwoods) in southern Quebec, in contrast to the old-growth matrix that was shaped by low fire frequency in the conifer forests in the north of Quebec.

Also, forest management changes the forest landscape because it diminish mature forest habitats and increases young stands as a result of the harvesting of mature and overmature stands (Bergeron, Engelmark, et al. 1998). Traditional harvesting practices used treatments such as clearcutting stands of old-growth conifers that generated landscapes dominated by extent areas of hardwood stands. Therefore, during the last decades Forest Ecosystem Management (FEM) has emerged as a new paradigm for forest management that has tried to imitate the forest structure and dynamics created by natural disturbances to conserve most of the ecological processes, habitats and species diversity (Bergeron and Harvey 1997, Pothier, Raulier, and Riopel 2004, Jones, Domke, and Thomas 2009). Despite the efforts to narrow the differential effect of forest management on succession and landscape dynamics, the current forest management unlike the fire, has modified the stand age distribution of the landscapes producing a large-scale shift from coniferous to mixed and deciduous forests (Boucher et al. 2009), and an unavoidable diminution of the mature forest landscape which is becoming increasingly fragmented (Bouchard and Pothier 2011, Boucher et al. 2015b, Molina, Valeria, and De Grandpre 2018).

At a landscape scale, the spatial configuration of post-harvesting landscape differs in distribution, size, and connectivity between stands (structural and functional continuity between similar stands) from mosaics formed by natural disturbances (Wang and Cumming 2010). These patterns of fragmentation modify the quality of habitat, travel, and dispersion of species. At a stand scale, young post-harvesting stands generally have a low proportion of later successional species, high tree density, less heterogeneous vertical structure and tree productivity than respective stands originated by natural disturbances (Brassard and Chen 2010). Nonetheless, structure and composition of stands originated from natural disturbance and harvesting could converge in intermediate or advanced successional states (Jetté et al. 2009).

Climate change is one of the main factors that can alter the dynamics of mixedwood boreal forests. Some scenarios are predicting that the temperature will increase 2-4˚C by the end of the current century, with the most significant warming in boreal areas (Solomon et al. 2007). Effects of these climate changes are reflected in the intensification of fire regimes, and then, the modifications in competitiveness of dominant species according to their fire adaptations (Boulanger et al. 2016), as well as changes in forest processes at all scales, such as successional pathways, landscape diversity, and heterogeneity, forest distribution, composition, and structure. For example, burn rate and catastrophic fires have decreased significantly during the last 300 years in mixedwood forests in the Abitibi region. As a consequence, the proportion of mature stands has increased (Bergeron et al. 2001). However, it is expected that more substantial warming and drying will further enhance fire frequency and burned area in many boreal forests by the end of this century (Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, Lesieur, et al. 2006, Bergeron et al. 2011). Therefore, the frequency of stand-replacing fires will increase the proportion of fire-adapted species as well. However, according to Bergeron et al. (2011), future burn rates by itself will not move this ecosystem to a condition not encountered in the past; but the increasing fire incidence plus the current rates of harvesting may reduce the ecological variability of the ecosystem in time and space.

Despite the importance of mixedwood forests, during the last decades in eastern Canada, the mixedwood forest landscape has changed from a mature forest landscape to a fragmented landscape with a growing quantity of young stands (Jetté et al. 2009) During this landscape fragmentation process, the proportion of hardwood-dominated forest communities has increased, with a diminution of conifer dominance. This landscape pattern is closely related to changes in disturbances regimes, principally related to the fire regime as a consequence of climate change and forest management (Simard et al. 2009). However, our understanding of the spatial and temporal variations of mixedwood forest landscapes and the characteristics of the second growth postfire and post-harvesting stands is still insufficient. The objective of this thesis was to characterize the mid-term changes of mixedwood forest mosaics in eastern Canada and improve our knowledge about relationships between these changes and the dynamics of natural and anthropogenic disturbance regimes. The main hypotheses that were tested regard (i) the changes in disturbance regimes during last 70 years, specifically, the diminution of burn rate and high rates of forest harvesting have modified the mixedwood forest landscape in western Quebec, specifically the heterogeneity, configuration, and productivity; (ii) under climate change scenarios aboveground biomass will decrease dramatically and productivity will increase under the most severe climate warming scenario, represented by an increase on fire frequency, and high-intensity forest management. These changes will be accompanied for an increase of hardwood and mixed stand and a decrease of conifer stands in all the study area. In addition, it is expected that all these changes in aboveground biomass, productivity, and forest composition will be caused mostly by the intensification of forest management than by the increase in burned area; and (iii) after fire the stands will be dominated by shade-intolerant and fire-adapted species that will be replaced by a mixture of hardwoods and conifers with mid-tolerance or tolerance to fire, and eventually, the forest will be dominated by conifers. On the contrary, it is expected that after forest management the stands will be composed by a mixture of hardwoods and conifers with different levels of shade-tolerance. Moreover, it is expected that the forest distribution at the landscape scale will change, under extreme climate change (higher fire frequency) and high-intensity forest management (twice-actual harvest intensity) scenarios, the south of the study area will be dominated by stands of shade intolerant and fire-adapted species, while the north of the study area will be dominated by mixed stands of shade intolerant or mid-tolerant species. Those changes in forest composition will be accompanied by an increase in forest fragmentation, more complex patch shapes, and more isolated patches in respect to the initial landscape.

1.2 Theoretical framework

1.2.1 Ecology of mixedwood forest and successional pathways

In natural forest landscapes, stand diversity occurs mainly as a result of three components: abiotic conditions (physical, environment, and climate), succession (dispersion, pre-disturbance composition), and disturbance regimes (fire, insect outbreak, harvesting, etc.) (Gauthier, Leduc, and Bergeron 1996). In Quebec's territory, the diversity and distribution of ecosystems are classified through a hierarchical ecological classification (ecological districts) as a function of relief, geology, and geomorphology. This classification helps to identify the susceptibility of stands to natural disturbance occurrences as a function of soil characteristics and vegetal composition. Related to forest disturbances, in Abitibi the mixedwood forest landscape may be characterized by a mixed influence of fire and forest management (Reyes et al. 2010, Wang and Cumming 2010).

Stand and landscape composition in the mixedwood boreal forests are particular to each ecological region as an effect of abiotic condition. For instance, in the northern border of the mixedwood boreal forests, the spruce-feathermoss bioclimatic domain composed by the ecological regions 6a and 6c have a landscape dominated by black spruce stands and occasional balsam fir; while in the center of the mixedwood boreal forests the balsam fir-white birch bioclimatic domain, composed of the ecological regions 5a and 5b, have a landscape dominated by hardwood or mixed stands with intolerant hardwood as trembling aspen, white birch and jack pine. In the south of the mixedwood boreal forest the balsam fir-yellow birch bioclimatic domain composed of the ecological regions 4a and 4c, have a landscape dominated by mixed stands of yellow birch (Betula alleghaniensis Britt.) and softwoods (Bergeron 2000, Saucier et al. 2011, Seedre and Chen 2010).

In relationship to the effect of the succession on stand and landscape composition, the distribution of species depends on the strength of the natural succession process and the degree to which disturbances affect a stand. In particular, succession in the boreal forest is driven by the interaction between factors such as the type and means of regeneration, species shade tolerance, species longevity and growth rate, and the species adaptation to thrive from disturbances (Bergeron 2000, Seedre and Chen 2010). Although the course of the succession process tends to converge towards dominance by conifers in an almost predictable manner, the succession pathways may give rise to a wide variety of stand dynamics (stand structure and composition), because the development of the succession process is a function of species life history, biotic interactions and abiotic conditions, the aforementioned regeneration process, and even insects outbreaks or another secondary natural disturbances (Bergeron et al. 2014).

Regarding the influence of disturbances regimes on species composition, disturbances affect the stand structure and forest succession dynamics, causing a variety of pathways. Disturbance typically initiates succession by creating suitable conditions for the predominance of shade-intolerant species. In the boreal forest, trembling aspen and white birch are the first species to be established (Ilisson and Chen 2009) due to their high light demand to germinate (Scheller and Mladenoff 2005). However, conifers shade-intolerants such as jack pine can be established with the early successional hardwoods when a seed source is nearby (Bergeron and Charron 1994). In mid-succession stages and given long enough time since fire, the canopy transition allows to shade mid-tolerant conifer such as black spruce and white spruce to grow from the understory and replace the hardwood shade-tolerant pioneers (Bergeron and Dubuc 1989, Brassard and Chen 2010). Finally, in the late succession stage, tree recruitment occurs mostly in small openings created by gap dynamics (MacDonald and Weingartner 1995). In this stage black spruce, white cedar (Thuya occidentalis L.) or balsam fir are the typical dominant species. The simple replacement by the understory coniferous trees of the overstory cohort dominated by deciduous trees is not the most likely scenario. The presence of mixed sub-canopy layers could lead to the development of mixed stands (Bergeron and Charron 1994). Additionally, in mixedwood forests, stands rarely endure as climax stands, because fire, harvesting or other disturbances return these forests to early or mid-successional stages before they reach mature states (Bergeron 2000). Therefore, the equilibrium of these forests can be considered only at landscape level where a relatively stable stand age distribution can be observed (Bergeron and Dubuc 1989).

The disturbance regime of mixedwood boreal forests, which is the sum of all disturbances affecting the forests (Mori 2011), is characterized by the occurrence of periodic and severe natural disturbances, such as fires, insect outbreaks, and small-scale disturbances such as windthrow and gap dynamics, and anthropogenic disturbances such as forest management. Altogether these disturbances model forest composition and structure (Burns and Honkala 1990a, Greene and Johnson 1999, Kasischke, Christensen, and Stocks 1995). However, among all those disturbances fire is the most important natural one that modifies the ecological processes and successional pathways by changing their frequency, intensity, and size (Johnson 1996, Johnstone et al. 2010, Rowe 1961, De Grandpré, Gagnon, and Bergeron 1993).

At the landscape level, fire frequency is probably the most important aspect of fire regime (Wiltshire and Archibald 1998). The fire rate helps to determine the landscape age structure. For instance, short fire cycles limit tree succession because forests do not have enough time to replace species and develop mature stands at the end. Therefore, the landscape age structure and stand composition tend to change toward early and mid-succession stand dominance (Jetté et al. 2009). Otherwise, under long fire cycles, forest structure and composition could be considerably affected by secondary disturbances (Bergeron et al. 2001), and landscape could be dominated by late succession stands. Similarly, the stand succession is different under long or short fire frequency. For example, the fire-adapted pioneers trembling aspen, white birch, and jack pine may regenerate rapidly after fire by root suckering, stump or sprouts, and serotinous cones respectively, therefore, when high-frequent fires occur the stands tend to be dominated by these species. On the contrary, in absence of fire, shade-tolerant species such as black spruce and cedar may regenerate easily by layering, hence, under low-frequent fire the landscape is composed of a mixture of all those species instead of dominated only by fire adapted ones (Burns and Honkala 1990a, Greene et al. 1999).

1.2.2 Effect of climate change on natural disturbance regimes

Climate change will affect boreal forest ecosystems through changes in forest fire regimes (Fleming and Candau 1998). Particularly, in eastern Canada a significant decrease in fire frequency was observed during the years 1530 to 1996 when the average annual burn rate decreased between 0.29-0.35% (Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, Lesieur, et al. 2006, Bergeron et al. 2004). Apparently, this decrease in the frequency of fires has been due, at least in part, to a warming trend that began at the end of the Little Ice Age (ca. 1850) (Bergeron et al. 2001). Consequently, although in the northwest of Abitibi the mean stand age was less than 150 years in that period, 55% of mixedwood forest in Abitibi are now old stands, and their succession may continue after 224 years (Bergeron and Dubuc 1989, Bergeron et al. 2004). However, the predicted rise on the temperature in around 2-5˚C by the end of the current century across Canada's boreal forests (Pachauri et al. 2014), will modify fire regime by increasing the burn rate (Bergeron et al. 2011, Boulanger, Gauthier, and Burton 2014, Wang et al. 2015) and larger fires' frequency (Flannigan et al. 2005, Kasischke and Turetsky 2006), affecting then stand and landscape composition, structure, and configuration (Keane et al. 2013, Stuart-Smith 1997). That burning rate increase will impose a change in the composition of high-productive pioneer tree species (hardwoods) to less productive species that dominate during late-succession stages (conifers) (Chen and Luo 2015, Girardin, Bernier, and Gauthier 2011, Seedre et al. 2014).

1.2.3 Forest management on mixedwood boreal forests

Harvesting over the last century has been a key factor influencing the landscape in mixedwood forests in eastern Canada. Since the 1960s, modern forest industry began with the mechanization of forest operations (Vincent 1995). During this period, clearcutting was the most commonly used treatment. It contributed to change the stand age distribution producing a large-scale shift from the initial predominance of coniferous to a mixed and deciduous forests (Fenton et al. 2009), and also to increase the landscape heterogeneity (Boucher et al. 2015b, Brassard et al. 2008, Molina, Valeria, and De Grandpre 2018, Pickell, Andison, and Coops 2013). Generally, the practice of large-scale harvesting like clearcutting has a significant impact that favors the increase of deciduous species rather than conifers, since in addition to extracting conifer species (balsam fir, spruce, pine, and larch), it eliminates seed-bearing trees and destroy potential and pre-stablished regeneration (Boucher et al. 2009).

Because of the negative effects of traditional harvesting practices on forest composition and structure, and also on the landscape configuration, in recent decades an Forest Ecosystem Management (FEM) have been proposed as the new forest management approach (Simard et al. 2009). FEM try to emulate natural disturbances and then, maintain biodiversity, resilience, and adaptive capacity of forest ecosystems by reducing the gap between managed and natural landscapes to ensure long-term maintenance of ecosystem functions and thereby retain the social and economic benefits they provide to society (Gauthier et al. 2009). Currently, the most widely used FEM harvesting strategies in eastern Canada include partial cutting (it comprises commercial thinning, shelterwood, and selection cutting).

In theory, by adjusting the forest management to resemble the patterns observed in natural disturbances, most of the natural habitat structures, processes, and species of the ecosystem could be safeguarded. However, none of these new approaches has been able to resemble natural disturbances, and currently there are differences in structure and composition during the initiation stage of stand development between post-fire and post-harvesting stands (Brassard and Chen 2010). For instance, harvesting has not yet been able to reproduce the distribution of forest age classes observed after natural disturbances, because harvesting continues being undertaken systematically in mature and overmature stands, and it is occurring much faster than natural disturbances would (Bergeron, Richard, et al. 1998, Boucher et al. 2009, Cyr et al. 2009). Therefore, mature and overmature stands continue decreasing considerably in forest landscapes, and a simplification of stand structure has been observed (Brassard and Chen 2010). Furthermore, conifer cover is declining and making way for hardwood and mixedwood forests (Bouchard and Pothier 2011). In terms of spatial configuration, the mosaic formed by harvesting is more fragmented than the one created by natural disturbances (Wang and Cumming 2010). This pattern is associated with the variability in size, form, and time of the harvested areas. These patterns of fragmentation modify the amount of interior habitats and connectivity between mature stands, which in turn influence the movement and dispersion of species (Bergeron and Charron 1994, Bergeron and Dubuc 1989, Fleming and Candau 1998, NCASI 2006). The main objective of this thesis was to characterize the mid-term changes of mixedwood boreal forest landscape in north-west Quebec and improving our knowledge regarding the relationships between such changes and the expected variations on fires as a consequence of the future climate change and forest management.

CHAPTER II

TWENTY-EIGHT YEARS OF CHANGES IN LANDSCAPE HETEROGENEITY OF MIXEDWOOD BOREAL FOREST UNDER MANAGEMENT IN QUEBEC, CANADA

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2.1 Abstract

Changes in natural and anthropogenic disturbances in the mixedwood boreal forests of northwest Quebec have modified the landscape heterogeneity in terms of its composition and configuration. The aim of this study was to evaluate the heterogeneity (composition and configuration) of 78,000 km2 of mixedwood boreal forest landscape during recent decades (using Landsat imagery from 1985 to 2013) in areas affected by forest management. The classes Conifer, Mixed-Conifer, Mixed-Hardwood, and Hardwood were differentiated with an object-based classification (accuracy >80.2, Kappa coefficient >0.7). In addition, 5 metrics (mean area, largest patch index, percentage of the landscape that corresponds to core area, perimeter-area fractal dimension, and aggregation index) were calculated. This study showed that conifer-dominant cover dominates the mosaic and accounted for one-third of the study area. Nevertheless, the conifer-dominant class showed the greatest decrease (reduction of 35% of its initial area at a rate of 1.7% per year). The metrics indicated that forest management in recent years produced a more heterogeneous landscape in 2013 unlike the landscape in 1985. The fire-influenced, old-growth conifer forests that previously dominated the landscape in northwestern Quebec were transformed by forestry practices into a more heterogeneous landscape.

Keywords: Landscape heterogeneity, landscape composition, landscape configuration, mixedwood boreal forest, GEOBIA, landscape metrics, wildfires, forest management.

Résumé

Les régimes des perturbations naturelles et anthropiques dans les forêts boréales mixtes du nord-ouest du Québec ont modifié l'hétérogénéité du paysage en termes de composition et de configuration. L'objectif de cette étude était d'évaluer l'hétérogénéité (composition et configuration) d'un territoire de 78,000 km^2 de la forêt boréale mixte aménagée au cours des dernières décennies en utilisant des images Landsat de 1985 à 2013. Les classes de couvertures forestières Résineux, Résineux mélangés, Feuillus mélangés et Feuillus ont été différenciées à l'aide d'une méthode de classification basée sur l'identification d'objets (précision >80.2, coefficient Kappa >0.7). En outre, cinq paramètres (superficie moyenne, indice de la taille de la plus grande parcelle, pourcentage du paysage de l'aire centrale, dimension fractale entre périmètre et surface et indice d'agrégation) ont été calculés. Les résultats de cette étude ont montré que la classe Résineux, domine la mosaïque et représente un tiers de la zone d'étude. Malgré, cette même classe a enregistré la plus forte diminution au cours de la période d'évaluation avec une réduction de 35% de sa superficie initiale au taux de 1,7% par an. Les paramètres indiquent que l'aménagement forestier au cours des dernières années a produit un paysage plus hétérogène en 2013, contrairement au paysage en 1985. Ces résultats indiquent que les forêts anciennes résineuses et influencées par le feu qui dominaient auparavant le paysage dans le nord-ouest du Québec ont été transformées par les pratiques forestières en un paysage plus hétérogène.

Mots clés: hétérogénéité du paysage, composition et configuration du paysage, forêt boréale mixte, GEOBIA, métriques du paysage, feu, aménagement forestière.

2.2 Introduction

Forest Landscape heterogeneity is the degree of spatial variability of different elements within the landscape in space and time (Li and Reynolds 1995, Wiens 1989). This may be expressed as a combination of two underlying components: composition and configuration. Composition indicates what habitats and how many are present in a landscape by measuring the area ocupied by each habitat, and configuration describes how these habitats are arranged spatially by measuring the spatial characteristics, arrangement, position, and/or geometrics of the landscape units or patches (Fahrig et al. 2011, Trani and Giles 1999, Turner 2005).

The main drivers of landscape heterogeneity are climate, the physical environment, and disturbance regimes (Grondin et al. 2014). Disturbance regime characteristics (fires, windthrows, insect outbreaks, forest management), and especially their frequency, spatial extent, and severity, are the main drivers of forest composition and configuration in space and time (Romme et al. 2011, Turner 1989). For example, Turner et al. (1994) found that severe and extensive fires increased landscape heterogeneity in Yellowstone National Park, as they generated a mosaic of stands with different severity of burning. Opposite trends were observed in the northeastern Canadian boreal forest where fire recurrence is very low and natural landscapes are very homogeneous, mainly dominated by old-growth forests with a low proportion of young forest patches (Boucher et al. 2015b).

Mixedwood forest occupies 24% of the total boreal forest in Canada (Baldwin et al. 2012). This forest zone is characterized by disturbance regimes and climatic, topographic, and edaphic conditions that favor the formation of closed canopies dominated by trembling aspen (Populus tremuloides) or white birch (Betula papyrifera) in early successional stages, black spruce (Picea mariana) or white spruce (Picea glauca) in mid-successional stages, and balsam fir (Abies balsamea) in mature successional stages. Historically, the heterogeneity of the mixedwood forest landscape has been related to wildfire regimes. For example, (Bergeron et al. 2004) showed that high fire frequency was associated with the high proportion of young forest stands in the mixedwood forests of southern Quebec, in contrast to the old-growth matrix that was shaped by a low frequency of fires in the conifer forests in the north of Quebec.

During the last decades, intensification of forest management has modified the heterogeneity of mixedwood forests at the landscape scale. Since the 1970s, the modern forestry industry began the mechanization of forest operations in the study region (Vincent 1995). During this period, clearcutting of old growth conifer stands was the most commonly used treatment. This contributed to the change in the age class distribution of forests at the landscape level by increasing the proportion of young stands of hardwoods (Fenton et al. 2009). For many years, it was suggested that clearcutting was emulating wildfire disturbance patterns (Keenan and Kimmins 1993). However, recent studies have shown that the landscapes that have resulted from clearcutting differed considerably in patch shape and distribution from those typically generated by fires (Boucher et al. 2015b, Gosselin 2002, McRae et al. 2001, Pickell, Andison, and Coops 2013). For instance, in contrast with unmanaged forests, managed landscapes with clearcutting had a lower mean patch area and more intricate shapes (Mladenoff 2004, Jetté et al. 2009).

Recently, traditional forest management (clearcutting of large areas) has changed toward Forest Ecosystem Management (FEM), where harvesting tries to emulate natural disturbances by retaining forest structural characteristics. The objective is to maintain forest ecosystem resilience and integrity and consequently reduce the negative effects of traditional forest management (Hunter Jr. 1990). In the context of

FEM, new approaches, such as partial cutting and commercial thinning (Simard et al. 2009), have been implemented in some portions of the mixedwood forest. However, because of the heritage of past forest management and that harvesting is undertaken systematically in mature and overmature conifer forests, FEM cannot reproduce forest age classes distribution similar to what is generated by natural disturbances (Kneeshaw and Bergeron 1998). Therefore, mature and overmature forest cover has considerably decreased in the landscape. In addition, conifer cover is declining and making way for hardwood and mixedwood forests (Bouchard and Pothier 2011). In terms of spatial configuration, the mosaic resulting from forest management is more homogeneous and fragmented than the one created by natural disturbances (Wang and Cumming 2010). This pattern is associated with the variability in size, form, and time of the harvested areas. Such habitat fragmentation modifies the number of interior habitats and connectivity between mature stands (Bergeron and Charron 1994, Bergeron and Dubuc 1989).

One promising method to measure landscape heterogeneity is based on the analysis of remote sensing imagery such as Landsat images to obtain landscape coverages (Banskota et al. 2014, Boyd and Danson 2005, Wulder et al. 2004) along with the analysis of metrics to measure the arrangement, position, and geometrics of the landscape units or patches (Uuemaa, Mander, and Marja 2013). Among the methods employed to extract landscape coverages from remotely sensed imagery, object-based image analysis (GEOBIA) has been increasingly used due to its capacity to classify objects (polygons composed of multiple neighbour pixels equivalent to landscape units or patches) rather than single pixels, doing so by combining image segmentation and classification and by integrating radiometric and textural image attributes (Benz et al. 2004, Qin et al. 2013). An advantage of GEOBIA is the inclusion of texture attributes during segmentation that avoid the inclusion of errors related to single pixels (Blaschke

2010). In addition, landscape metrics are very useful for quantifying landscape heterogeneity. By measuring many geometric relationships at different spatial scales, landscape indices or metrics allows to describe spatial patter within a landscape mosaic (McGarigal and Marks 1995).

The aim of this study was to evaluate the heterogeneity of mixedwood boreal forests of Quebec during the last decades (from 1985 to 2013). We hypothesize that these forests are now more heterogeneous than 28 years ago as a consequence of the forest management. Therefore, because the conifer species are preferentially harvested, a decline in abundance of conifer and mixed conifer forests and an increase in hardwood and mixed hardwood forests is expected. Similarly, it is probable that the configuration has changed from consisting of larger patches to smaller and less compact patches of conifer and mixed conifer forests; however, it is likely that as forest management has not change significantly, the spatial distribution and average size of harvested patches, the patch sizes and aggregation of hardwood and mixed hardwood have not changed during the time considered in this study.

2.3 Methodology

2.3.1 Study area

The study area is located in the north-west of Quebec. It encompasses portions of six ecological regions (4a, 4b, 5a, 5b, 6a, and 6c) in northwestern Quebec (Table 2.1) (Saucier et al. 1998). The area covers about 146000 km2 and is located between 47°30'-49°30' N and 76°30'-79°30' W (Figure 2.1), however the final area after excluding some areas (agriculture, water, clouds, etc) is about 78000 km2. The climate is subpolar and sub-humid continental. In the northern part of the study area, the average annual temperature is 1.2 ºC and the annual precipitation is 917 mm (La Sarre station). In the southern part, the average annual temperature is 3.4 ºC. The average annual precipitation is 831 mm (Ville-Marie station) (Environment Canada 2016). Soils are gray luvisols that originated from clay deposits left by proglacial Lake Ojibway, and the drainage is moderate to imperfect (Vincent and Hardy 1977).

Table 2.1 Description of the ecological regions covered in the study. The Total Area represents the portion of the Ecological Region within the study region.

	Ecological region	Bioclimatic subdomain	Dominant forest cover	Area	
				Total (km^2)	Studied area (%)
4a	Plains and hills of Simard lake	Balsam fir-yellow birch domain	Mixed stands of yellow birch and softwoods	5943	79
4b	Cabonga watershed hillside			27429	52
5a	Abitibi plains	Balsam fir-white birch domain	Hardwood or mixed stands with intolerant hardwoods (trembling aspen, white birch and jack pine)	26842	89
5b	Gouin watershed hillside		Balsam fir and white spruce stands mixed with white birch	15758	51
6a	Matagami lake plains	Spruce-moss domain	Black spruce with occasional balsam fir	48842	18
6c	Opémisca lake plane		Black spruce	21428	37

Following fires, even-aged stands are dominated by trembling aspen and paper birch (Ilisson and Chen 2009). Conifers, such as jack pine (Pinus banksiana Lamb.), can be present with the early successional hardwoods if a seed source is nearby (Bergeron and Charron 1994). Canopy transition allows for the presence of mid-shade-tolerant conifers, such as spruce species (mainly Picea mariana and Picea glauca (Moench) A. Voss) with deciduous species in mid-succession (Bergeron and Dubuc 1989, Kneeshaw and Bergeron 1998). In the late successional stage, stand dynamics are driven by windthrows and insect outbreaks (Bergeron et al. 2014). Forest composition is characterised by the dominance of late-successional species such as balsam fir and eastern white-cedar (Thuya occidentalis) and, less commonly, trembling aspen, paper birch, and white spruce (Kneeshaw and Bergeron 1998).

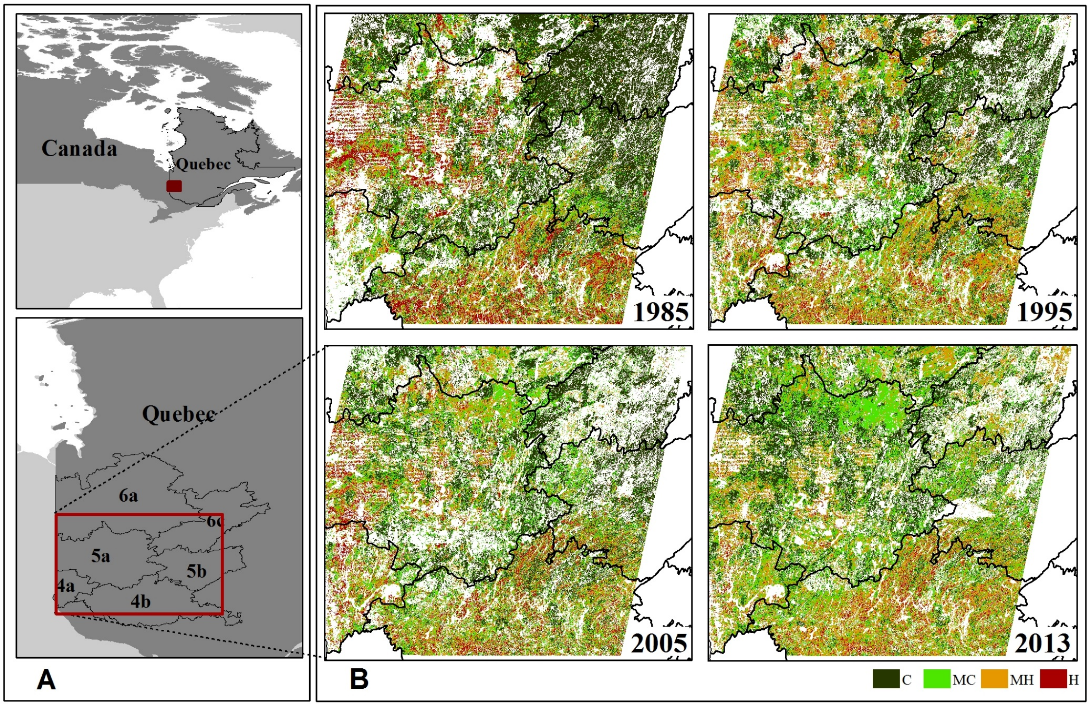

Figure 2.1 Study area in North West Quebec, and B: Changes in the abundance and distribution of forest classes between 1985 and 2013 in the Abitibi region, Northwest Quebec. C: conifer, MC: mixed conifer, MH: mixed hardwood, and H: hardwood classes. The black line represents the limit of ecological regions.

2.3.2 Methods

2.3.2.1 Data

Landscape heterogeneity was evaluated according to the dynamics of four forest classes and one non-forest class from 1985 to 2013, which was evaluated in three periods (1985-1995, 1995-2005, and 2005-2013) through analysis of Landsat imagery. Given all of the remotely sensed imagery available, Landsat images were chosen for use because they have many advantages for landscape-scale analyses and for monitoring (Frazier et al. 2014, Valeria, Laamrani, and Beaudoin 2012, Wulder et al. 2011). For example, this sensor produces images that cover large geographic areas (185 x 185 km) and has adequate spatial resolution (30 m), quality, and temporal frequency to develop forest landscape monitoring; in addition, they are freely available from the U.S. Geological Survey (USGS).

These four classes were differentiated according to tree-species composition as follows: Conifer-Dominant, where >70% of species in the stand were conifers; Mixed Conifer, where conifers occupied 50-70% of the stand; Mixed Hardwood, where hardwood species occupied 50-70% of the stand; and Hardwood-Dominant, where >70% of the species present in the stand were hardwoods. The Non-Forest class was included in order to describe the rural area without forest cover, including agriculture, burned and harvested areas but excluding water bodies, rivers, roads, and urban areas.

Image pre-processing

To develop the land cover mosaics for each year, a set of four Landsat images (paths 17-18, rows 26-27 of Worldwide Reference System II) were selected for 1985, 1995, and 2013. For 2005, it was necessary to use six images to eliminate a cloudy area from the final mosaic (Table 2.2). The images were selected under relatively clear sky conditions (less than 10% cloud cover) during the summer and early fall to avoid classification errors that would be associated with differences in phenological conditions.

Table 2.2 Landsat time-series of imagery used in the study.

Year	Path/Row	Adquisition date	Sensor	Spectral bands	Resolution
1985	17/26	22-Jun-84	Landsat Thematic Mapper (TM)	TM1, TM2, TM3, TM4, TM5, TM7	30 m
	17/27	22-Jun-84			
	18/26	21-Jul-86			
	18/27	21-Jul-86			
1995	17/26	4-Jul-94	Landsat Thematic Mapper (TM)	TM1, TM2, TM3, TM4, TM5, TM7	30 m
	17/27	4-Jul-94			
	18/26	30-Jul-95			
	18/27	30-Jul-95			
2005	17/26	13-Jun-04	Landsat Enhanced Thematic Mapper (ETM)	ETM1, ETM2, ETM3, ETM4, ETM5, ETM7	30 m
	17/26	4-Sep-05			
	17/27	3-Aug-05			
	17/27	7-Sep-06			
	18/26	26-Aug-05			
	18/27	25-Jul-05			
2013	17/26	26-Sep-13	Landsat 8 Operational Land Imager (OLI)	OLI2, OLI3, OLI4, OLI5, OLE6, OLI7	30 m
	17/27	26-Sep-13			
	18/26	17-Sep-13			
	18/27	17-Sep-13			

For each image, haze was eliminated and the clouds were masked by applying the function "Calculation of cloud/haze mask" from the PCI software (PCI Geomatics Enterprises Inc 2012). Furthermore, where the image had gaps, they were filled with segments of other, cloud-free images from the same year. Also, small clouds and cloud shadows – as well as water bodies, rivers, roads, and urban areas - were excluded from the mosaic. Water (water bodies and rivers) was identified by comparing the reflectance values in the Near Infrared band (band 4) and short-wave infrared bands (5 and 7), as described by (Frazier and Page. 2000). Roads and urban areas were masked using the database of the Ministry of Forests, Fauna and Parks of Quebec (Quebec Ministry of Forests 2015).

Cloud-free images were radiometrically corrected in respect to a reference image to minimize variations in atmospheric conditions. The high-quality, cloud-free image path 17, Row 26 from 1985 was used as the radiometric reference point. The radiometric reference image was calibrated using the function "Atmospheric correction for flat areas" (ATCOR2) in the software PCI. This function creates corrected reflectance images for each band using known atmospheric conditions for the image acquisition date (PCI Geomatics Enterprises Inc 2012). Mosaics for each year were then built with the module Orto-Engine in PCI.

Finally, principal component analysis (PCA) and Normalized Difference Vegetation Index (NDVI) were undertaken for the mosaic of each year to evaluate changes in vegetation along the temporal sequence. The PCA was calculated with Landsat bands 1 through 7 (excluding band 6) (USGS 2016). The Normalized Difference Vegetation Index (NDVI) was computed with the red and infrared bands (bands 3 and 4). These are common for multi-temporal land cover analysis because of their simplicity and capability to enhance information to identify changes (Deng et al. 2008). PCA is a linear transformation of a set of image bands (six bands in our case) that creates a new orthogonal band set, which is uncorrelated and ordered in terms of the amount of variance explained in the original data. Therefore, PCA reduces the dimensionality and eliminates redundant information in the initial set of bands, but it retains the maximum variation present in the bands (Byrne, Crapper, and Mayo 1980). The final product of a PCA is a new raster with the same number of components (raster bands) as the input raster. The first principal component (PC1) has the greatest variance, the second (PC2) contains the second-most variance not described by the first, and so forth (Demšar et al. 2013). The variance explained by the first three components of the four mosaics was higher than 90%. NDVI combines information contained in two spectral bands, the Red and NIR (NDVI = (NIR - red) / (NIR + red)), to identify the fraction of absorbed, photosynthetically-active radiation or vegetation (Rouse Jr et al. 1974).

Forest classification

The mosaics were classified by the GEOBIA method with the software eCognition Developer 8 (Trimble 2011). GEOBIA classification included two steps: image segmentation and object classification. Image segmentation is the process of dividing the image into segments that contain pixels with similar spectral and textural values to produce a polygon vector with those segments (Qin et al. 2013). A multi-resolution, bottom up, region-merging segmentation technique was applied. This method identified single image objects of one pixel from any place in the image and merged them with their neighbors through numerous clustering processes that were based on relative homogeneity criteria such as scale, shape, and compactness of the inputs (Benz et al. 2004). The final product is a polygon vector file with polygons that correspond to the segments separated during segmentation, with polygons covering the entire raster area, with the exception of the masked areas. To perform the segmentation, the set of mosaic bands (bands 1 through 7, excluding band 6), the PC bands, and the NDVI layer were introduced. The values of the weight scale, shape, and compactness parameters were obtained by iterative segmentations until an accurate division of land covers that were well-delimited was achieved, such as stands of forest inside big areas of non-forest or vice versa. This accurate division was evaluated by visual inspection, comparing the GEOBIA segmentation with the “ecoforestal” map produced by the Ministry of Forest, Fauna, and Parks (Quebec Ministry of Natural Resources and

Wildlife 2015a). The final values retained of these parameters were 10, 0.1, and 0.5, for weight scale, shape, and compactness, respectively.

During the classification process, the “assign class” algorithm was performed to determine whether an image object or polygon fit a particular landcover class. This algorithm allows for the use of spectral, textural, and contextual values as threshold conditions. These values are obtained by averaging the values from all pixels that were merged into a polygon. For this study, the classification criteria to identify forest classes (Conifer-Dominant, Mixed Conifer, Mixed Hardwood, and Hardwood-Dominant) and Non-Forest were the average values of NDVI and PC1.

A database of 2664 independent, geo-referenced plots (215, 2219, 140, and 51 for the years 1985, 1995, 2005, and 2013 respectively) was used to calibrate the threshold values during classification and to test the accuracy of the classification. The plot database was stratified into the four forest classes that were identified in the classification by calculating the proportion of the basal area of hardwood vs conifer species. 30% of this database was used during classification, and the other 70% was used in the accuracy estimation. This database is composed of a selection from the permanent and temporary plots (circular plots of 400 m2) of the Ministery of Forest, Fauna and Parks, with stand age between 0 and >120 years for the years 1985, 1995, and 2005 (Quebec Ministry of Natural Resources and Wildlife 2015a), and of the dataset of 51 temporary measured during the summer of that year.

Classification accuracy

A confusion matrix (Congalton, Oderwald, and Mead 1983) and the Kappa coefficient of agreement (Cohen 1960) were calculated to test the classification accuracy. A confusion matrix is a table where rows contain the classification that was generated, and columns contain the reference database. This matrix indicates the number of sample plots that were assigned to a particular category in the classification relative to the actual land cover category, and it describes the accuracy of each category along with commission and omission errors. The overall accuracy is calculated based on the data in the matrix diagonal. The Kappa coefficient is a discrete, multivariate technique that measures the agreement between two databases that were classified into the same categories (Lunetta and Lyon 2004). Indirectly, Kappa incorporates the off-diagonal elements into the accuracy estimation (Congalton 1991). From 70% of the plot database, the number of plots that matched with each forest class were established by superimposing the plots database.

2.3.2.2 Analysis

Landscape metrics

A metric analysis of the forest class maps with the software FRAGSTAT v4.2 was performed (McGarigal, Cushman, and Ene 2012) for each studied year. FRAGSTAT metrics are computed at three levels according to the landscape elements analyzed at different scales: the patch level measures all individual patches, the class level measures the structure of all patches that compose each class, and the landscape level measures the structure of the habitat mosaic. Furthermore, FRAGSTAT groups the metrics in the four main groups of metric indices according to what landscape characteristic is being described: *i)* Area / Edge, *ii)* Shape, *iii)* Core area, and *iv)* Aggregation. In this study, the metric analysis was done at the class level, where each class corresponds to one of the four forest types described above.

For each of the four forest classes (Conifer-dominant, Mixed Conifer, Mixed Hardwood, and Hardwood-dominant) in each of the four mosaics (1985, 1995, 2005, and 2013), 82 metrics were computed. However, because many of the metrics were partially or completely redundant (McGarigal and Marks 1995) and because no single metric captures the configuration of a landscape adequately (Turner 2005), the computed metrics database was simplified and reduced in number by a PCA. After computing the PCA for each year, all metrics with a smaller eigenvector were eliminated until the smallest possible set of metrics that both minimize redundancy and correctly describe the landscape configuration was obtained. To do so, the conservation of at least one metric of the main metric groups proposed by FRAGSTAT was attempted: *i)* Area / Edge, *ii)* Shape, *iii)* Core area, and *iv)* Aggregation.

The final group was composed of five metrics from each of the four groups that showed the greatest statistical weights from the PCA. In this smaller group, 95.3% of the total cumulative variance of the landscape configuration was explained with the two first components (Table 2.3). The metrics in this group were the mean area "AREA" as the area-weighted, mean patch size of patches of the corresponding class, where the proportional area of each patch was based on total class area; largest patch index "LPI", as the percentage of the landscape comprised by the largest patch of a class; core area percentage of landscape "CAP", the percentage of the landscape of the core area within the corresponding patch class; perimeter-area fractal dimension "PFAD", as the degree of complexity of the patch shapes at different scales, where the minimum value 1 described very simple perimeters such as squares and approached a value of 2 for shapes with highly convoluted, plane-filling perimeters; and the aggregation index

"AI", obtained from the frequency with which different pairs of patch types including like adjacencies between the same patch type appeared side-by-side on the map. AI equaled 0 when the patch types were maximally disaggregated and increased as the landscape became increasingly aggregated, with a value of 100 when the landscape consisted of a single compact patch (McGarigal, Cushman, and Ene 2012).

2.4 Results

2.4.1 Landcover classification and accuracy

The classification rules that were proposed yielded successful cover maps for each of the evaluated periods (the rules are in the Figure 2.2 and Table 2.4, and the map in Figure 2.1b). However, the thresholds in the rule sets were adjusted for each mosaic because they were not spectrally consistent enough for the same algorithm to be applied to all (Figure 2.2). For instance, NDVI was a better classifier for the 1985 mosaic, but CP1 was the better classifier for 1995, 2005, and 2013 (Table 2.4). The overall classification accuracy was greater than 80%, and the Kappa coefficient was greater than 0.7 (lowest value in 2013), but these values varied widely according to the forest type. All cover maps showed the lowest classification accuracy for the mixed forests (Mixed Conifer and Mixed Hardwood), and the largest values of accuracy were obtained from the pure forests (Conifer and Hardwood) (Table 2.4).

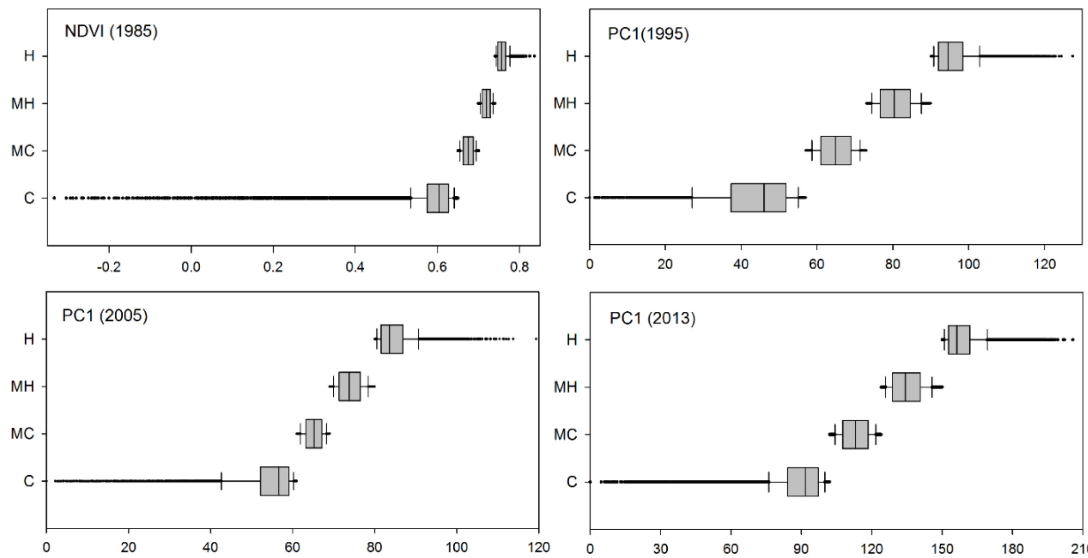

Figure 2.2 Distribution of NDVI and PC1 values after GEOBIA segmentation and classification of four forest classes from Landsat images between 1985 and 2013 in the Abitibi region, Northwest Quebec. The main vertical line shows the median, boxes represent quartiles, and whiskers depict either the maximum or 1.5 times the interquartile range of the data (whichever is smaller). Points are outliers. C: conifer, MC: mixed conifer, Mixed Hardwood: mixed hardwood, and H: hardwood.

2.4.2 Changes in landscape composition

Although all forest types were present throughout the period, their relative abundance and distribution throughout the landscape and over time changed a lot (Figure 2.1 and Figure 2.3). Covering 68% to 77% of the area, forest remained the dominant cover in the landscape between 1985 and 2013, with the remaining area composed of Non-Forest. Among the forest classes, the conifer-dominant class dominated the mosaic between 1985 and 2005 and accounted for a third of the study area. This class showed the greatest decrease in area during the evaluation period (a 35% reduction in its initial area at a rate of 1.7% per year). Mixed Conifer was the second most abundant cover, and in contrast to conifer-dominant, its area increased by 73% relative to its initial area at a rate of 2.6% per year, becoming the dominant class in 2013. Among the other classes, Mixed Hardwood increased by 63%, and Hardwood-dominant decreased by 59% relative to their respective initial areas (Figure 2.3).

Table 2.3 Group of metrics selected by Principal Component Analysis (PCA) to evaluate landscape configuration.

Group	Metrics selected	Cum % of variance		*ChiSq*
		PC1	PC2	
Area-Edge	Mean area (AREA)	69.6	95.3	<.0001
	Largest Patch Index (LPI)			
Shape	Perimeter-area Fractal dimension (PFRAC)			
Core area	Core area percentage of landscape (CPLAND)			
Aggregation	Aggregation Index (AI)			

Table 2.4 Classification rules set, and accuracy assessment obtained by GEOBIA classification on Landsat mosaic for 1985, 1995, 2005, and 2013 in the Abitibi region, Northwest Quebec. PC1: first principal component; NDVI: Normalized Difference Vegetation Index; C: conifer; MC: mixed conifer; MH: mixed hardwood; and H: hardwood classes. Accuracy was not calculated for Class H in 2013 because there were no validation plots.

Year	Classification rules set	Forest class	Accuracy	Validation plots	Kappa coef.
1985	0.65≥NDVI	C	97.8%	91	
	0.7≥NDVI>0.65	MC	70.0%	40	
	0.74≥NDVI>0.7	MH	63.6%	33	
	NDVI≥0.74	H	98.0%	51	
	Layer 6≤1 21≤Layer 3	NF	–	–	
	Total		*87.44%*	*215*	0.82
1995	PC1≤58	C	95.4%	1001	
	58<PC1≤73	MC	60.8%	393	
	73<PC1<90	MH	55.2%	509	
	PC1≥90	H	96.5%	316	
	Layer 6≤1 30≤Layer 3	NF	–	–	
	Total		*80.22*	*2219*	0.80
2005	PC1≤58	C	98.8%	85	
	58<PC1≤72	MC	88.6%	35	
	72<PC1≤85	MH	87.5%	16	
	85<PC1	H	90.0%	10	
	Layer 6≤1 24≤Layer 3	NF	–	–	
	Total		*94.52*	*146*	0.95
2013	PC1≤70	C	100%	20	
	70<PC1≤88	MC	65.0%	20	
	88<PC1<110	MH	72.7%	11	
	110≤PC1	H	NA	0	
	Layer 6≤1 38≤Layer 6	NF	–	–	
	Total		*80.39*	*51*	0.70

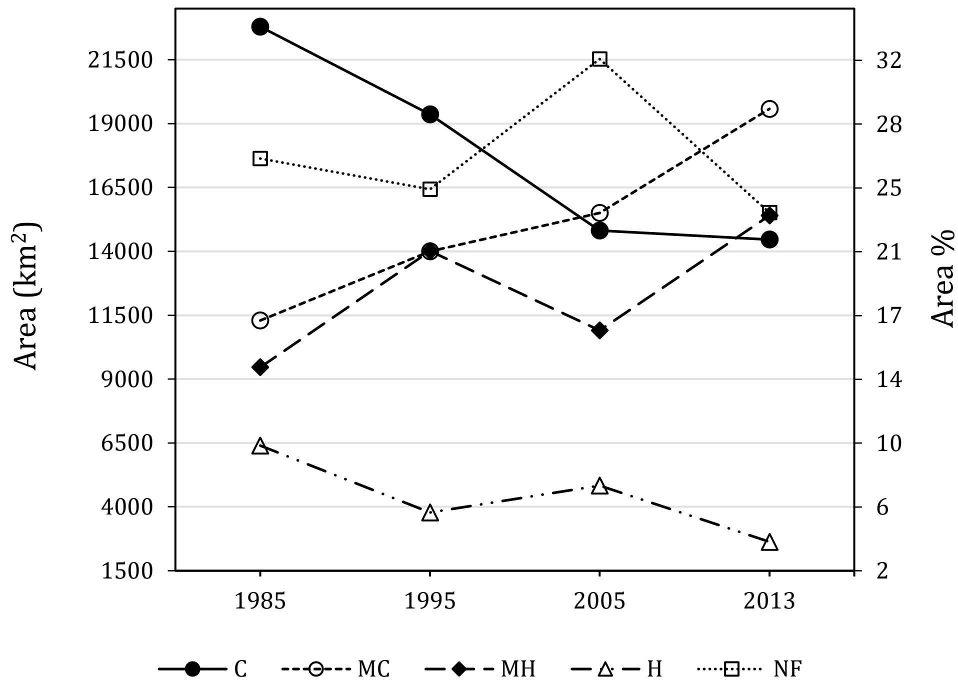

Figure 2.3 Change in composition of forest classes between 1985 and 2013 in the Abitibi region, Northwest Quebec. C: conifer, MC: mixed conifer, MH: mixed hardwood, H: hardwood, and NF: non-forest classes.

Changes in the area followed a very different dynamic between classes (Table 2.5). The Conifer-dominant class exhibited the most significant changes relative to Non-Forest or Mixed Conifer, but the contribution of the Mixed Hardwood and Hardwood-dominant to the increase/decrease of Conifer-dominant class was negligible (<1%). In the periods 1985-1995 and 1995-2005, the Conifer-dominant class changed to Non-Forest by 6.3% and 6.0% of the study area, respectively, and Conifer-dominant forests changed to the Mixed Conifer class by 5.4% and 5.1%, respectively. Simultaneously, Mixed Conifer changed to Conifer-dominant in 4.9% and 4.6% of the area during the same periods mentioned above, respectively, indicating that the changes between Conifer-dominant and Mixed Conifer classes were concurrent and more or less equivalent. Regarding the Mixed Conifer class, the decrease of Non-Forest and Mixed

Hardwood classes was equivalent to the sustained increase of Mixed Conifer class during the entire study period. New stands of Mixed Hardwood appeared at the expense of Non-Forest and Hardwood-dominant classes in the periods 1985-1995 and 2005-2013. However, during the 1995-2005 period, changes in the area of the Mixed Hardwood class were explained by the increase in the Mixed Conifer (5.8%) and Hardwood-dominant (3.6%) classes. The successive increases and decreases of the Hardwood-dominant class were linked to the increases/decreases of the Non-Forest class (1.2%, -0.4%, and 0.3% for the periods 1985-1995, 1995-2005, and 2005-2013, respectively) and the Mixed Hardwood class (-4.2%, 1.7%, and -3.3% for the periods 1985-1995, 1995-2005, and 2005-2013, respectively).

Changes in forest composition were not uniform throughout the study area (Figure 2.1 and Figure 2.4). In general, the ecological region 6c in the north of the study area experienced the highest forest changes in all the evaluated time periods, mainly due to the decrease of the Conifer-dominant class area. Ecological regions 4b, 5b, and 6a in the north and east of the study area show a significant decrease in the Conifer-dominant class area. Ecological regions 4a and 5a in the west of the study area showed a small decrease in Non-Forest and an increase in forest area (combination of all forest classes) but showed no change in the Conifer-dominant class, which was more or less constant during that time.

Table 2.5 Changes in four forest classes (% of total landscape area) over time. Row and column totals indicate the total proportion of the study area occupied by each forest class in the Abitibi region, Northwest Quebec. C: conifer, MC: mixed conifer, MH: mixed hardwood, and H: hardwood classes.

Year		1995					Total 1985
	Cover	NF	C	MC	MH	H	
1985	NF	14.3	1.4	3.7	4.7	1.9	26.1
	C	6.3	21.3	5.4	0.7	0.1	33.7
	MC	1.9	4.9	6.4	3.3	0.2	16.7
	MH	1.1	0.9	4.1	6.7	1.1	14.0
	H	0.7	0.1	1.1	5.3	2.3	9.5
Total 1995		24.3	28.6	20.7	20.7	5.6	100

Year		2005					Total 1995
	Cover	NF	C	MC	MH	H	
1995	NF	22.3	0.3	0.8	0.8	0.2	24.3
	C	6.0	16.6	5.1	0.7	0.2	28.6
	MC	1.9	4.6	11.1	2.7	0.4	20.7
	MH	1.1	0.4	5.8	9.9	3.6	20.7
	H	0.6	0.0	0.3	1.9	2.8	5.6
Total 2005		31.9	21.9	22.9	16.1	7.2	100

Year		2013					Total 2005
	Cover	NF	C	MC	MH	H	
2005	NF	18.4	1.4	5.7	5.5	0.8	31.9
	C	2.4	14.2	4.8	0.5	0.0	21.9
	MC	1.1	5.4	12.5	3.8	0.1	22.9
	MH	0.5	0.4	5.5	9.0	0.6	16.1
	H	0.5	0.0	0.4	3.9	2.4	7.2
Total 2013		23.0	21.4	29.0	22.8	3.9	100

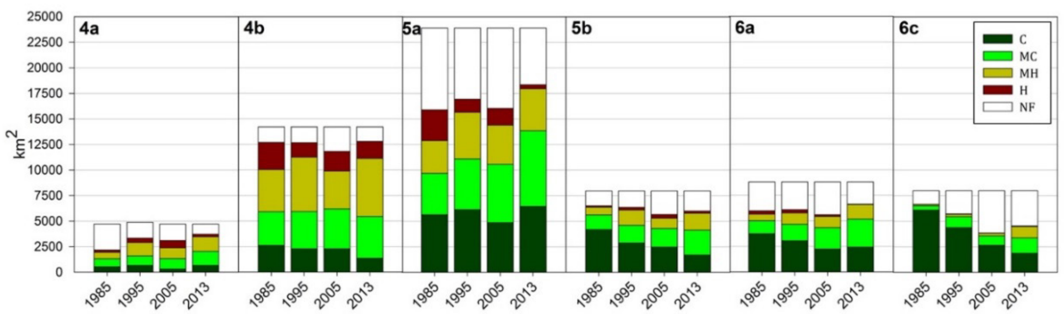

Figure 2.4 Changes in composition of forest classes by ecological regions between 1985 and 2013 in the Abitibi region, Northwest Quebec. C: conifer, MC: mixed conifer, MH: mixed hardwood, H: hardwood, and NF: non-forest classes.

2.4.3 Analysis of forest metrics

The initial landscape in 1985 showed that the Conifer-dominant class had the largest and most-aggregated patches (AREA, AI, Figure 2.4). It also had the largest patch in the landscape (LPI) and the largest interior area of a patch (CAP). The other forest classes had approximately the same, relatively low patch area, largest patch in the landscape, and patch interior area values. At the same time, Conifer-dominant and Mixed Conifer classes exhibited more complex patch shapes (PFAD) than did Hardwood-dominant and Mixed Hardwood classes in 1985.

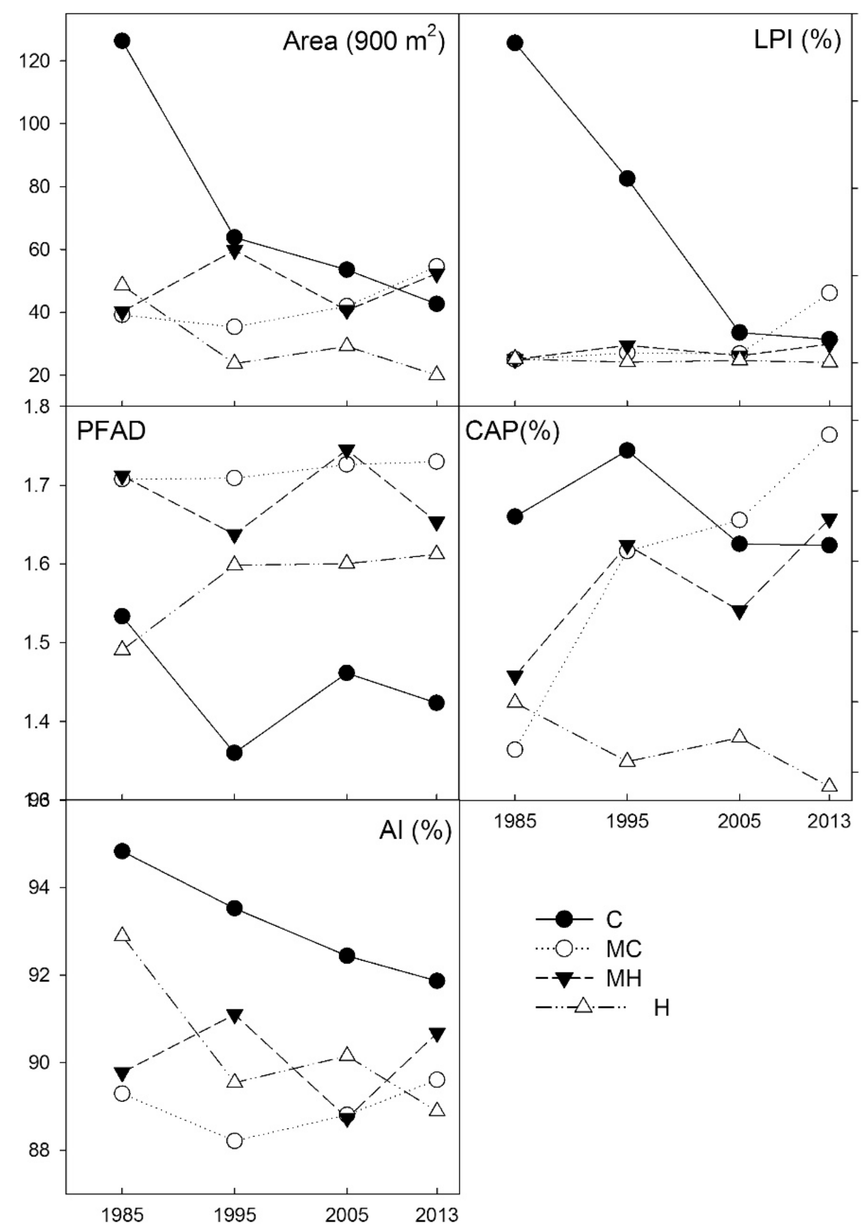

Figure 2.5 Variation in landscape configuration during 1985-2013 (AREA: patch mean area, LPI: largest patch index, PFAD perimeter-area fractal dimension, CAP: core area percentage of landscape, and AI: aggregation index) in the Abitibi region, Northwest Quebec. H hardwood classes, MH: mixed hardwood, MC: mixed conifer, and C: conifer

The Conifer-dominant class showed the most evident changes during the study. AREA and LPI declined at a higher rate between 1985 and 2005, CAP declined overall relative to 1985, AI decreased steadily during all the study period, and PFAD experienced periodic intervals of increase and decrease. Conversely, metrics for the Mixed Conifer class increased during the 1985-2013 period as follows: AREA, PFAD, and AI increased at a slow, constant rate; CPLAND, CAP increased drastically; and PLI showed an evident increase only during the 2005-2013 period. For the Mixed Hardwood forest class, although AREA, LPI, PFAD, and AI exhibited periodic intervals of increase and decrease, their values were more or less stable during all study periods; only CAP showed a net increase. For the Hardwood dominant forest class, AREA, CAP, and AI followed the same pattern of decrease, although PFAD showed a net increase, with a larger rate from 1985 to 1995, and LPI was stable (Figure 2.4).

2.5 Discussion

2.5.1 The forest landscape dynamic

The high abundance of mature forests and low landscape heterogeneity at the beginning of the study period (in comparison with the rest of the study period), suggested that natural forest dynamics involved low frequency wildfires accompanied by many low-impact disturbances, such as windthrows and minor outbreaks of insects (Boucher et al. 2009).

As hypotetized in this study, the subsequent loss of conifer forests and the increase in landscape heterogeneity that we observed during the study period are consistent with the cumulative effects of forest management. The mechanization of forest operations that started in the Abitibi region after the 1970s (Vincent 1995) and the effects of forestry on landscape composition and configuration became more evident. Clear-cutting of overmature and old-growth conifer and mixed-conifer forests was the most commonly used harvesting method, and it accounted for a decrease of approximately 324000 ha of these forest classes (Quebec Ministry of Natural Resources and Wildlife 2015b). Forestry between 1960 and 1985 was concentrated in the southern part of the study area (ecological regions 4b and 5a), especially along a railroad where most pulp and sawmill companies were located (Vincent 1995). This contributed to an increase in the proportion of young stands of hardwood and mixed-hardwood in the subsequent decades (Bergeron et al. 2002) in this part of our study area, which showed the largest increase in hardwood and mixed-hardwood forests. The rise in timber demand and the scarcity of large contiguous areas of conifers close to pulp and sawmill factories resulted in the displacement of forestry activity to the northern ecological regions 5b and 6a-c (Coulombe et al. 2004), which explains in part the decrease of conifer forest area after 1985 in the center and north of our study area.

The group of metrics analyzed indicated that the landscape became progressively more heterogeneous from 1985 to 2013, probably because of the cumulative impact of forest management. Boucher et al. (2015b) found similar effects of management on forest configuration in the north-eastern boreal forest of Quebec. The size distribution range of clear-cut areas was less variable than the clearings that would result from fire, are typically larger patches with larger core areas (McRae et al. 2001, Schroeder and Perera 2002, Wang and Cumming 2010). In addition, different disturbance types create different patch shapes; wildfires tend to burn with irregular and rounded edges, which follow natural contours leaving many residual forests or unburned patches of vegetation within the burned area. In contrast, clearcuts are characterized by straight edges (more regular borders) with low abundance of residual trees, producing more complex shapes in the landscape than exist in the post-wildfire landscape (Schroeder and Perera 2002). Thus, harvesting creates more fragmented habitat and a patch structure that is less spatially aggregated compared to wildfire (Turner et al. 2003, Wang and Cumming 2010).

The large amount of stands that were without forest cover or that were covered by early successional communities in 1985 were progressively moving toward more complex communities, compositionally evolving from hardwood to mixed forests and, finally, to conifer dominant, as described by Brassard et al. (2008). Similarly, the areas that were in an early or intermediate successional stages continued to develop through more complex successional stages; for example, a large amount of mixed-hardwood and mixed-conifer stands passed to mixed-conifer (5,1% of total landscape area for all the study period) and conifer (5% of total landscape area for all the study period), respectively. Additionally, some minor disturbances, such as insect outbreaks and windthrows, and natural succession were probably responsible for the equivalent proportion of conifers that passed to mixed conifer (5,1% of total landscape area for all the study period) and from mixed conifer to conifer (5% of total landscape area for all the study period) (Boucher et al. 2009).

In contrast to conifer and hardwood forests, mixed conifers showed an increasing area and spatial homogenization. This forest class showed a rise in mean area of patches and the aggregation and consolidation of larger patches that were related to the natural succession of hardwood and mixed hardwood forest classes (Turner, Gardner, and O'Neil 2001). Thus, approximately 29% of the landscape, which was an area covered by mixed conifer forests, shows homogenization, and the remaining 71%, which is mainly conifer, mixed hardwood, hardwood, and non-forested classes, was experiencing strong heterogenization.

2.5.2 Implications for forest management

In the context of FEM, during recent decades, harvesting methods such as, partial cutting and commercial thinning are applied to extensive areas in parallel of clear-cutting. Evidence of this change was the decrease in the rate of reduction of conifer-dominant class after 1985 throughout the study area, and the deceleration after 1995 in the decrease in mean area and core area of patches and the decline in the intricacy of patch shapes (Figure 2.4). Nevertheless, none of these metrics showed a stabilization or recovery during the period of the present study (Figure 2.4). Additionally, some metrics, such as patch aggregation, decreased at a constant rate during the study, which indicated that thus far FEM has not fully mitigated the impacts of forestry practices on this important landscape trait. All this results are evidence that support the hypotesis of this study about the changes in landscape configuration, specifically the change from larger patches to smaller and less compact patches of conifer and mixed conifer forests.

FEM aims to maintain forests within their limits of natural variability and that would require a change in the current spatial distribution of harvesting. For instance, some studies in Quebec have recommended implementing functional zoning of forests, where the forest is divided into areas of conservation, intensive forest harvesting, and multiple uses (Coulombe et al. 2004, Messier et al. 2009). Therefore, if forest harvesting is concentrated in specific areas, this may limit the spatial extent of the human footprint on the landscape. Indeed, Tittler, Messier, and Fall (2012) demonstrated that intensive management of a small part of the landscape is better than extensive management distributed over the entire landscape. Specifically, the landscape created by this zoning is less fragmented and the forests are more aggregated than the ones created by the influence of current practices.

2.5.3 Methodological approach and sources of error

Our results show that GEOBIA is an efficient classifier of forest in mixedwood boreal lands. In general, this method allowed us to develop an effective and replicable method, and the replicability may be the highest advantage of GEOBIA over other methods reported (Lübker and Schaab 2010). Compared with our method, pixel-based approaches, such as the enhancement-classification method that was previously used to classify the forests in our study area, achieved a lower accuracy for Conifer-dominant and Hardwood-dominant stands, and a similar accuracy for Mixed stands (Valeria, Laamrani, and Beaudoin 2012).

With respect to classification accuracy, the lower values obtained for mixed forest classes in comparison with unmixed classes may be related to the low discrimination capability of spectral values to separate these forest classes. The spectral response of mixed stands is influenced by the species mixture, canopy closure, and the contribution of the understory (Wulder et al. 1998). In our case, stands composed of mixtures of trembling aspen and white birch with jack pine, balsam fir, white spruce, and black spruce were classified without considering variability of stand structure (i.e., density, canopy closure, stand height, and understory composition). These excluded variables produced a broad range of spectral values and textures that were related to the physical structure of the stand instead of its composition (ratio hardwood/conifers). Other studies have faced the same problem. For example, Wolter et al. (1995) obtained low accuracy for classifying mixed stands from Landsat images, especially for balsam fir-aspen and mixed conifer stands. Similarly, Gerylo et al. (2002) found that stand age and height influenced the overall canopy and understory reflectance values. One way to improve the accuracy in classification may be to add textural information during the classification process, as was suggested by Franklin et al. (2000) and Moskal and

Franklin (2002); they demonstrated that incorporattion of textural information into the classification of high-resolution imagery can increase classification accuracy by 12% or more.

2.6 Conclusion

The results of this study showed that the previous, fire-influenced, old- growth conifer forests that dominated the landscape in northwestern Quebec were transformed by forestry practices between 1985 and 2013 to produce more heterogeneous (composition and configuration) landscape. Traditionally, forest management involved harvesting extensive areas of conifers and changed the landscape composition affecting relative abundance of the forest classes, resulting in the decrease in large, contiguous areas of conifer class (1,400 km^2 equivalent to 12.4% of the initial conifer area). At the same time, the landscape became more fragmented, with more complex patch shapes, lower core areas, and more isolated patches. Except for the loss of forest aggregation, the change of these metrics showed a deceleration after 1995, which might be a consequence of the change in forestry practices at that time from traditional forest management (clearcutting of large areas) toward ecosystem-based management. These findings support our hypothesis that forest heterogeneity has changed with a decline in the relative abundance and landscape configuration of conifer forests over a period of 28 years. Although there are difficulties in classifying mixedwood stands, we found that GEOBIA was an efficient (accuracy > 80.2, Kappa coefficient > 0.7), objective, and replicable method to classify forest landcovers in mixedwood boreal forests. More research is needed to establish the causal relationships between different forest cover classes, disturbances, and landscape heterogeneity, as well as the ecological and economic implications of this heterogeneous landscape dynamic.

2.7 Acknowledgement

We wish to thank the NSERC, Tembec, and Norbord industries for supporting this research. We would like to thank the Ministère des forêts, de la faune et des parcs, and his Forest inventory department for their support to provide the digital inventory data. NSERC/UQAT/UQAM Industrial Chair in Sustainable Forest Management (SFM Chair) also helped us in this project. We are also grateful to Carolina Rodriguez for the standardization of the databases, and Jorge Ramirez and 3 anonymous reviewers for critically reading the manuscript and helping us to improve it.

CHAPTER III

PROJECTING FUTURE ABOVEGROUND BIOMASS AND PRODUCTIVITY OF MANAGED EASTERN CANADIAN MIXEDWOOD BOREAL FOREST IN RESPONSE TO CLIMATE CHANGE

Eliana Molina, Osvaldo Valeria, Louis De Grandpre, Jorge Andres Ramirez, Dominic Cyr and Yan Boulanger.

3.1 Abstract

Eastern Canada boreal forests are mainly modulated by natural wildfires and forest management activities. To evaluate forest dynamics under possible interactions among fire and timber harvest in a future climate warming scenario (RCP 2.6, RCP 4.5 and RCP 8.5) the forest landscape model LANDIS-II was used to simulate the dynamics of the 78000 km^2 of boreal forests in eastern Canada. Forest management intensity scenarios were modeled considering the changes in the annual harvested area (0.5%, 1%, and 2%) and the age that conifers and hardwoods can be harvested (50 and 30 years, 70 and 50 years, and 90 and 70 years). The results of the 300-year model projections showed that both forest management intensity and climatic scenarios explained most of the variability in aboveground biomass, aboveground net primary productivity and forest composition. Forest management seems to be the most important factor that modeled the landscape in the southern forests because there were more stands with the age and composition required by each harvesting prescription to deal with the annual allowable cut volume. On the contrary, in the northern forests there was a mixed effect of climate change and forest management because many of the areas suitable for harvesting were previously burned limiting the amount of area available for harvesting. Thus, although it is expected an increase in burned area due to climate change, the intensification of forest management seems to be the most important driver of the increase of hardwoods and mixed stands and the decrease of conifers stands on the mixedwood boreal landscape, mainly in the southern forests. These results suggest that timber supply would be at risk in the Abitibi Plain, therefore, some strategies should be applied to adapt forest management to climate change.

Keywords: Climate change, Forest management, Fire, LANDIS-II, Boreal forest, Aboveground Biomass, Aboveground Net Primary Productivity, forest composition.

Résumé

Les forêts boréales de l'est du Canada sont principalement modulées par les feux de forêt naturels et les activités de gestion forestière. Pour évaluer la dynamique des forêts sous les interactions possibles entre le feu et la récolte de bois dans un futur scénario de réchauffement climatique (RCP 2.6, RCP 4.5 et RCP 8.5), le modèle de paysage forestier LANDIS-II a été utilisé pour simuler la dynamique des 78000 km^2 de forêts boréales de l'est Canada. Des scénarios d'intensité de gestion forestière ont été modélisés en tenant compte des changements dans la superficie récoltée annuelle (0,5%, 1% et 2%) et de l'âge auquel les conifères et les feuillus peuvent être récoltés (50 et 30 ans, 70 et 50 ans, et 90 et 70 ans). Les résultats des projections du modèle sur 300 ans ont montré que l'intensité de la gestion forestière et les scénarios climatiques expliquaient la plupart de la variabilité de la biomasse aérienne, de la productivité primaire nette aérienne et de la composition de la forêt. La gestion forestière semble être le facteur le plus important ayant modélisé le paysage dans les forêts du sud en raison de la présence de plus de peuplements ayant atteint l'âge et la composition requis par les prescriptions de récolte afin de respecter le volume de coupe annuel autorisé. Au contraire, dans les forêts du nord, le changement climatique et la gestion forestière ont eu un effet mixte, car de nombreuses zones propices à la récolte étaient auparavant brûlées, ce qui limitait la superficie disponible pour la récolte. Ainsi, bien qu'on s'attende à une augmentation de la superficie brûlée en raison du changement climatique, l'intensification de la gestion forestière semble être le principal moteur de l'augmentation des feuillus et des peuplements mixtes et de la diminution des peuplements de conifères dans le paysage boréal mixte, principalement dans les forêts du sud. Ces résultats suggèrent que l'approvisionnement en bois serait menacé dans la plaine de l'Abitibi, par conséquent, certaines stratégies devraient être appliquées pour adapter la gestion forestière au changement climatique.

Mots clés: *changement climatique, aménagement forestière, feux, LANDIS-II, foret boréal, Biomasse aérienne, Productivité Primaire Nette aérienne, composition forestière.*

3.2 Introduction

Boreal forests are naturally modulated by an array of periodic disturbances such as wildfires, insect outbreaks, and windthrow that shape forest composition and structure, and consequently, biomass and productivity (Burns and Honkala 1990a, Greene and Johnson 1999, Kasischke, Christensen, and Stocks 1995). Wildfire is one of the main disturbances that drives the forest dynamics when comparing with the other natural disturbances. There are on average 7,500 fires per year in the Canadian boreal forest burning about 2.4 million ha, which were responsible for the emission of 247 Mt CO2e in 2015 (Natural Resources Canada 2017). Moreover, according to the Global Carbon Budget projections (Le Quéré et al. 2016), under the most severe anthropogenic climate warming scenario, it is projected that boreal regions could experience temperature increases up to 7.5°C accompanied by an increase on aridity by the end of this century (Representative Concentration Pathway, RCP 8.5) (IPCC 2013, Moss et al. 2008, Price et al. 2013, Wang et al. 2014). In such climate warming scenario, forest productivity will decrease (D'Orangeville et al. 2018), while fire frequency and extent may increase due to longer, warmer and drier summers in some parts of the boreal forests (Flannigan et al. 2016, Krawchuk, Cumming, and Flannigan 2009, Wotton, Flannigan, and Marshall 2017).

Usually, fires are stand-replacing events, their frequency, severity, and area disturbed largely affect forest succession and shape forest landscape (Burton et al. 2003, Payette 1992). For instance, when the intervals between fires are longer, forests are mainly dominated by late-successional species (conifers); while when the intervals between fires are shorter, forest are dominated by early-successional species (mostly hardwoods and some conifers like Jack pine and black spruce) (Bergeron et al. 2004). Therefore, under extreme climate change scenarios the increase on fire activity may promote the recruitment of pioneer species at the expense of longer-lived ones typical of late-successional stages, shifting forest age structure and composition to higher proportions of hardwood or mixed forests, affecting productivity and aboveground biomass(AGB) (Gauthier, Bernier, Boulanger, et al. 2015, Gustafson et al. 2000a). Consequently, indirect effect of climate warming represented as changes in natural disturbances might become as or more important in the ecosystem dynamics than the direct effects of climate warming to shape boreal forest landscape, and AGB stocks and productivity (Bergeron, Engelmark, et al. 1998, Schumacher and Bugmann 2006).

Most of boreal forests are also subject to management in eastern Canada. This management also modulates forest composition, (AGB), and productivity. For example, extensive historical harvesting contributed to decrease the proportion of forests in late-successional stages in southern boreal regions (Boucher et al. 2015b, Cyr et al. 2009, Molina, Valeria, and De Grandpre 2018, Valeria, Laamrani, and Beaudoin 2012, Boucher et al. 2017). Additionally, these harvesting practices have altered the spatial distribution of species, their woody biomass stocks and accumulation across the landscape. Several reports have found that forest management has gradually increased the proportion of hardwood or mixed early-successional forest (with lower biological and structural diversity than late-successional forest), which is simultaneously modifying the successional patterns and accelerating the species turnover in mixedwood boreal forests (He, Mladenoff, and Gustafson 2002, Kuuluvainen and Aakala 2011, Molina, Valeria, and De Grandpre 2018, Shorohova et al. 2011, Valeria, Laamrani, and Beaudoin 2012, Venier, Thompson, Fleming, Malcolm, Aubin, Trofymow, Langor, Sturrock, Patry, Outerbridge, et al. 2014).

Evaluating the future interactions among fire and timber harvest in a climate change context is necessary since they may lead to changes in composition, succession, and AGB and productivity (Gustafson et al. 2010, He, Mladenoff, and Gustafson 2002, Pastor and Mladenoff 1992). These changes may have no historical analogs and their study could help provide the basis for upcoming forest management policies considering future climate warming. In this study, a forest landscape model (Landis II) (Scheller and Mladenoff 2007) was used to project landscape composition, AGB and productivity in response to intensification on wildfire regimes by climate warming and forest management. A simulation experiment was conducted in a region widely managed by forest industry over several decades in the transition between mixedwoods and conifer forest. This area allows to study the expected change on forest composition, biomass and productivity through a landscape composed by mixedwood and pure stands (conifer and hardwoods). Landis II was used because it allows the integration of a key stand-level elements such as cohorts' regeneration, aging, and death, and landscape-level processes such as natural and anthropogenic disturbances to create novel communities that emerge from stand-level species interactions. Also, this model is sensitive to climate, so several climate change scenarios may be included in the modeled landscape.

Since it is expected that the forest landscape responds to future climate warming, represented as changes in fire regimes, and forest management interactions, this study aims to (i) identify the effect that fire and forest management scenarios will have on AGB, productivity, and forest composition of boreal forest located in eastern Canada (Abitibi Plain), and (ii) identify the role that the forest management will play on AGB, and productivity of boreal forest under expected climate change.

3.3 Methodology

3.3.1 Study area

The study area is located in the boreal forests of the Abitibi Plain in eastern Canada. The area covers about 78000 km2 between coordinates 47°30' - 49°30' N and 76°30' - 79°30' W (Figure 3.1). The climate is subpolar and sub-humid continental. In the northern part of the study area, the average annual temperature is 1.2 ºC and annual precipitation is 917 mm (La Sarre meteorological station. Environment and Climate Change Canada - Meteorological Service of Canada). In the southern part of the study area, the average annual temperature is 3.4ºC, and the average annual precipitation is 831 mm (Ville-Marie meteorological station) (Environment Canada 2016). The study area encompasses portions of the ecological regions 4a, 4b, 5a, 5b, 6a, and 6c as defined by the Quebec's Ministères de la Forêt, de la Faune et des Parcs. These areas are characterized by their topography, physiography, climate and variable proportions of superficial deposit types in eastern Canada (Quebec's northwestern part, Table 1) (Robitaille 1988). In the north, the spruce-feathermoss bioclimatic domain (ecological regions 6a and 6c) is dominated by black spruce (Picea mariana) stands with occasional balsam fir (Abies balsamea). The balsam fir-white birch (Betula papyrifera) bioclimatic domain (the ecological regions 5a and 5b), in the center of the study area, is dominated by hardwood or mixed stands with intolerant species as trembling aspen (Populus tremuloides), white birch and jack pine (Pinus banksiana). In the south, the balsam fir-yellow birch (Betula alleghaniensis) bioclimatic domain (ecological regions 4a and 4c) is dominated by mixed stands of yellow birch and different conifers (Saucier et al. 1998).

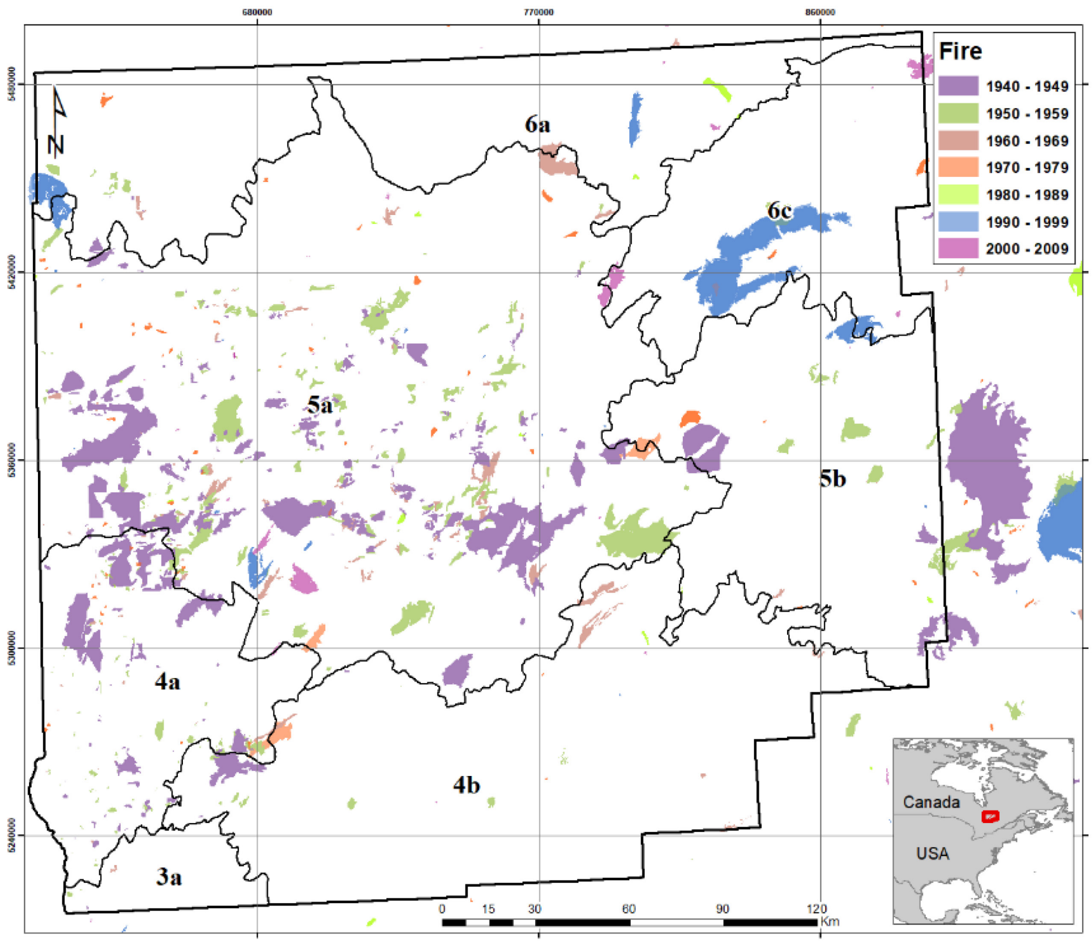

Figure 3.1 Study area with the ecoregions and historical fires. The black line shows the delimitation of the ecological regions 4a, 4b, 5a, 5b, 6a, and 6c. Colored polygons represent the burned areas by decade.

3.3.2 Landis II model

Biomass (AGB) and Net Primary Productivity (ANPP) of the aboveground forest component were projected using the model Landis-II 6.1, a stochastic, spatially explicit, and temporally discrete forest landscape modeling framework. This model simulates ecosystem processes at both stand and landscape scale, including processes such as forest succession, seed dispersal, forest management, and natural disturbances (fire, wind, insect outbreaks, drought) (Scheller and Domingo 2012, Scheller and Mladenoff 2007, Xu, Gertner, and Scheller 2009). During simulations, cohorts regenerate, age, and die according to user-defined time steps and simulation duration. In our case, simulations were run for 300 years (2010-2310) at a 20-year step intervals and 200 m resolution grid (4 ha).

Landis-II has been widely used in boreal forest landscapes for modeling succession, natural and anthropogenic disturbances (Gustafson et al. 2000a, Mladenoff 2004). In addition to many other similar models, this one can consider the effects of climate change on tree species ecological patterns (establishment probability, maximum and aboveground biomass, and maximum productivity) and simulate over long time periods how these factors interact with the landscape as well as natural and anthropogenic disturbances by creating communities to emerge from stand-level species interactions (He, Mladenoff, and Crow 1999). These model characteristics allowed us to identify the mixed and separate effects of fire and forest management on the forest composition, aboveground biomass, and productivity.

The landscape is represented in the model as a grid of interacting cells, each cell containing multiple species-age cohorts with their associated aboveground biomass. Each grid cell was assigned to a spatial unit (i.e. a "landtype") in which local soil and climate conditions were assumed homogeneous across all cells in each landtype, each landtype was big enough to construe changes in landscape and reduce simulation time. In our study, each ecoregion was defined by the Quebec's Ministry of Forest, Wildfire and Parks and considered as one landtype, this level of detail on the landtypes enabled us to identify differences in AGB, productivity, and forest composition according to the fire and forest management scenarios designed for our scale of analysis (Table 3.1). Forest community structure of each cell was initialized using forest attribute data derived from the fourth forest inventory of the Quebec database (Ministère des Ressources Naturelles du Québec 2013).

Table 3.1 Ecoregions in the study area.

	Ecological region	Bioclimatic subdomain	Dominant forest cover	Area	
				Total (km)	**Studied area (%)**
4a	Plains and hills of Simard lake	Balsam fir-Yellow birch domain	Mixed stands of Yellow birch and softwoods	5943	79
4b	Cabonga watershed hillside			27429	52
5a	Abitibi plains	Balsam fir-White birch domain	Hardwoods or mixed stands with intolerant hardwoods (Trembling aspen, White birch, and Jack pine)	26842	89
5b	Gouin watershed hillside		Balsam fir and White spruce stands mixed with White birch	15758	51
6a	Matagami lake plains	Spruce-Moss domain	Black spruce with occasional balsam fir	48842	18
6c	Opémisca lake plane		Black spruce	21428	37

3.3.2.1 Main inputs of Landis II

The Landis II architecture consists of several extensions where various ecological processes are simulated, which interact though a core module (Scheller et al. 2007). In this study the extensions “Biomass Succession 3.2.1” (Scheller and Mladenoff 2004), “Base Fire 3.0.3” (He and Mladenoff 1999) and “Base Harvesting 3.0” (Gustafson et al. 2000a) were used to model the forest response to natural and anthropogenic disturbances.

3.3.2.1.1 Forest succession modeling

Forest succession was simulated using the extension Biomass Succession (Scheller and Mladenoff 2004). This extension project changes in the cohort AGB and ANPP over time as each cohort regenerates, ages, and dies. Tree growth considers tree species' life history traits (Table 3.2) and species-specific responses to the environmental conditions unique to each landtype. Species life-history traits information for 17 species was collected from several sources and previous studies involving Landis II for North American forest landscapes (Qualtiere 2012, Scheller et al. 2007, Scheller et al. 2008, USDA 2014, Xu, Gertner, and Scheller 2009). Three climate-sensitive dynamic growth parameters, i.e., species establishment probability (SEP), maximum aboveground NPP (maxANPP) and maximum aboveground biomass (maxAGB) were parametrized for each species according to the values reported in the literature. These species were grouped in 262 initial communities on a raster as cohorts based on basal area and age from the fourth forest inventory of Quebec database according to the Landis II inputs requirements (Ministère des Ressources Naturelles du Québec 2013). Finally, species-specific establishment probability for each ecoregion was estimated from the proportion of area occupied by each species and ecoregion from the available 4th forest inventory of Quebec (Annex 3-A) (Ministère des Ressources Naturelles du Québec 2013).

Table 3.2 Life-history for the 17 species included in this study

Specie	Longevity (years)			Sexual Maturity (years)			ST	FT	ED (m)		MD (m)		VRP	VRP min age (years)	VRP max age (years)	RPF
	Min	Mean	Max	Min	Mean	Max			Min	Max	Min	Max				
Gray birch	20	20	20	8	8	8	1	1	60	60	80	100	0,5	2*	16*	Sprout
Yellow birch	150	225	300	20	40	70	2	1	213	250	400	400	0	0	0	None
White birch	80	110	140	15	15	40	1	2	60	100	5000	5000	1	40	125	Sprout
White spruce	100	211	250	15	30	40	3	3	64	100	200	400	0	0	0	None
Black spruce	150	180	250	10	20	30	4	2	50	80	150	300	0	0	0	Serotiny
Red spruce	250	350	400	20	30	40	4	1	50	50	61	100	0	0	0	None
Tamarack	150	180	230	15	30	40	1	3	14	21	40	60	0	0	0	None
Eastern white pine	200	200	450	10	20	30	4	3	60	60	210	210	0	0	0	None
Jack pine	75	140	200	5	10	15	1	4	20	40	60	100	0	0	0	Serotiny
Red pine	200	300	400	15	25	50	2	4	12	12	275	300	0	0	0	None
Balsam fir	80	150	200	20	25	30	5	1	25	60	100	160	0	0	0	None
Red maple	80	100	150	4	10	10	4	1	100	100	200	1000	1	10	150	Sprout
Sugar maple	300	400	500	30	40	60	4	1	15	15	100	200	0,5	40*	240	Sprout
Balsam poplar	120	140	150	8	10	20	1	2	200	1000	5000	5000	1	0	100	Sprout
Largetooth aspen	50	70	100	10	15	20	1	1	200	200	5000	10000	1	7*	56*	Sprout
Trembling aspen	60	130	200	10	15	20	1	2	500	1000	5000	10000	1	0	100	Sprout
Eastern white-cedar	300	350	400	6	30	35	4	1	45	45	60	60	0	0	0	None

ST: Shade tolerance, FT: fire tolerance, ED: effective seed dispersal distance in meters, MD: maximum seed dispersal distance in meters, VRP: vegetative reproduction probability, RPF: post-fire regeneration. Vegetative reproduction minimum and maximum values were estimated as the 10% and 80% of the mean longevity respectively.

3.3.2.1.2 Forest wildfire modeling

Fire was included using the extension Base-Fire. This extension simulates fire regimes through stochastic fire events depending on fire ignition, initiation and spread by ecological region using as input data fire size (min, mean and max), ignition probability, and the parameter *K* that determines the strength of the association between fire spread probability and fuel age. To parameterize the fire regimes and fire size, historical forest fire between 1941 and 2006 from SOPFEU database (*Société de Protection des Forêts Contre le Feu*) were used (Figure 3.1) and the ignition probability and *K* parameter were adjusted following values reported by Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, Lesieur, et al. (2006).

3.3.2.1.3 Forest management modeling

Forest management was simulated using the extension Base-Harvest (Gustafson et al. 2000a). The study area was divided into 14 forest management units available from the Ministry of Forests, Wildlife and Parks (MFFP) where specific harvesting prescriptions were planned (Ministère des Ressources Naturelles et de la Faune 2012). The harvesting prescriptions are a combination of temporal, spatial, and species components applied at the stand level. For each forest management unit, two common harvesting prescriptions were designed: careful logging around advance growth (CLAAG) and partial cutting (it comprises commercial thinning, shelterwood logging, and selection cutting), and applied following the annual allowable cut (AAC) volume calculation for Quebec authority in 2008 (Bureau du Forestier en Chef 2013). The economic rank by species, the minimum age to harvest each specie, the percentage of cohort harvested, and the maximum area to be harvested (50 ha) were extracted from the AAC calculation 2013-2018 (Annex 3-B) (Bureau du Forestier en Chef 2013).

3.3.2.2 Modeling scenarios

A baseline scenario was established, three fire regimes according to projected climate change scenarios (RCPs 2.6, 4.5 and 8.5 climate change scenarios) and three scenarios of forest management intensity. The baseline scenario models a landscape where the fire and forest management intensity were set with the information corresponding to the year 2010. For the baseline, the fire regimes for each ecological region were set as it follows: the higher ignition probability values and lower *K* values are in the northern ecological regions (6a and 6c), and the lower values of ignition probability and higher values of *K* are in the southern ecoregions (4a and 4b). For the forest management regime (baseline scenario), the minimum age to harvest conifers was set at 70 years, and 50 years for hardwoods. Furthermore, the maximum area to be harvested annually was set as 1% (ratio harvest/AAC for Quebec province was 0.7% in average between 1990 and 2013 (Bureau du forestier en chef 2015). Three fire regimes scenarios were elaborated based on the RCPs climate change scenarios (Van Vuuren et al. 2011): a very low emission scenario with an increase of 0.9-2.3°C by 2100 (RCP 2.6) (Riahi, Grübler, and Nakicenovic 2007); an intermediate stabilization scenario with an increase of 1.7-3.2°C by 2100 (RCO 4.5) (Clarke et al. 2007a); and a low mitigation scenario with an increase of 3.2-5.4°C by 2100 (RCP 8.5) (Van Vuuren et al. 2007). To model these scenarios, the changes in fire frequency were simulated according to Bergeron et al. (2011). The forest management intensity scenarios were designed based on two variables: the maximum allowable harvest area and the minimum age at which the conifers and hardwoods can be harvested. The allowable harvest area had three categories: low (0.5%), current (around 1%), and high (2%); and the minimum age to harvest conifers and hardwoods had three categories: a conservative scenario was set to 90 and 70 years respectively (refer here as 9070), and an extreme scenario was set to 50 and 30 years respectively (refer here as 5030). Each scenario was coded according to the climate change scenario first (RCP 2.6, RCP 4.5, and RCP 8.5), and then with the forest management scenario (for example: "5030_2%", where 50 indicates the age conifer may be harvested, 30 the age hardwood may be harvested, and 2% is the maximum area that may be harvested).

3.3.2.3 Model validation and verification

Our model was validated by comparing the biomass for the year 0 from the modeling exercise and from the fourth forest inventory plot dataset (Annex 3C). Also, our model was verified by comparing the baseline scenario (current climate and disturbance) outputs with the current AGB and ANPP available for the eastern boreal forest (Annex 3-D).

3.3.3 Experimental design and data analyses

Model outputs generated by both climate change and forest management scenarios were replicated five times accounting for the stochastic components of LANDIS-II. For each ecoregion and scenario, the temporal trends in total AGB and ANPP. AGB and ANPP were determined and reported here as mean values since the variation among simulated replicates in LANDIS-II were small. The effect of the ecoregion, forest management intensity and climate change scenarios on AGB and ANPP were evaluated using analyses of variance (ANOVA). A separate analysis was performed for AGB and ANPP at 160 and 300 simulation years (actual years 2170 and 2310) since the response variables varied over time. The explained variation of forest management intensity and climate change scenario by ecoregion were quantified as the percent of the total variation. Results were mainly assessed through visual inspection of trends since stochastic variation among replicates was minimal. Statistical analyses were performed in R 3.4.1 (R Development Core Team 2014) using the library MASS.

In addition, a map with four forest types; conifer (C), mixed conifer (MC), mixed hardwood (MH), and hardwood (H) was elaborated by ecoregion and scenarios at the 160 and 300 simulation years. The forest type was assigned according to the proportion of AGB of conifers and hardwoods following the same rule set of (Molina, Valeria, and De Grandpre 2018) as follows: Conifer, where >70% of AGB in the stand were conifers; Mixed Conifer, where conifers AGB occupied 50-70% of the stand; Mixed Hardwood, where hardwoods AGB occupied 50-70% of the stand; and Hardwood, where >70% of AGB in the stand were hardwoods.

3.4 Results

3.4.1 Disturbed areas under fire and forest management scenarios

The burned area in the baseline reached 10824 ha year^{-1} after 300 years in the study area (0.3% of the total area annually). After 300 years, burned area was significantly different between climate change scenarios, varying from the smaller values under RCP 2.6 (14557 ha year^{-1}), then increasing under RCP 4.5 (17485 ha year^{-1}), and the higher values under RCP 8.5 (23590 ha year^{-1}). Overall, forest management intensity scenarios (5030_2%, 7050_1%, and 9070_0.5%) did not have any influence in the total burned area behavior through the time and under the different climate change scenario (Figure 3.2).

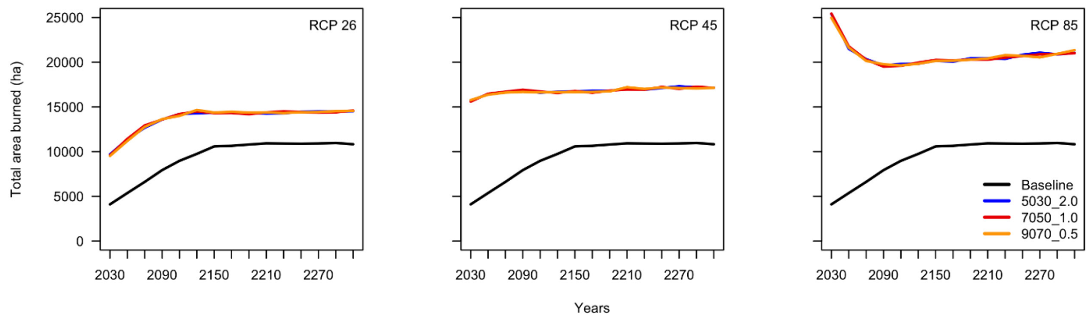

Figure 3.2 Burned area under three climate change scenarios (RCP 2.6, RCP 4.5, and RCP 8.5), and three forest management intensity regimes scenarios (9070_0.5%, 7050_1.0 %, 5030_2.0%. Where the first two numbers indicate the age at which the conifers may be harvested, the subsequent two numbers the age at which the hardwood species may be harvested, and the percentage indicates the percentage of the area that may be harvested).

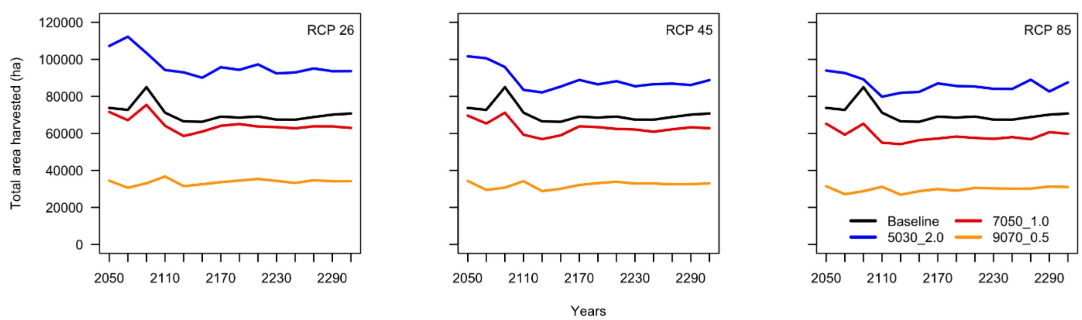

Figure 3.3 Total harvested area under climate change (RCP 2.6, RCP 4.5, RCP 8.5), and forest management intensity regimes scenarios (9070_0.5%, 7050_1.0 %, 5030_2.0%; where the first two numbers indicate the age at which the conifers may harvested, the subsequent two numbers the age at which the hardwood species may be harvested, and the percentage indicates the percentage of the area that may be harvested).

Total harvested area reached in the baseline 70770 ha year^{-1} after 300 years (2% of the total area). In general, the forest management intensity scenario 5030_2% presented a harvested area larger than the baseline, compared to the scenarios 7050_1% and

9070_0.5% that had both a lower harvested area than the baseline. The total harvested area for the management intensity scenarios 9070_0.5% and 7050_1% was similar under the three climate change scenarios, for the scenario 5030_2% the total harvested area decreased when the climate change scenario was more intense (Figure 3.3).

3.4.2 Forest AGB and ANPP following fire and forest management intensity scenarios

Forest management intensity and climatic scenarios explained most of the variability in AGB and ANPP. The AGB for the ecoregions 4a, 4b and 5b was related mostly to the forest management intensity (more than 87% of the variance), while for the ecoregions 5a and 6a the AGB was related in a similar proportion to the climate and forest management (Table 3.3). In respect to ANPP, the ecoregions 4a, 4b and 5b had more than 80% of variance explained by the forest management, while the ecoregions 5a, 6a and 6c had more than 80% of variance explained by the climate. The interaction between climate and forest management intensity explained less than 1% of the variability in AGB and ANPP. However, the ecoregion 6c showed the highest variance explained by this interaction with a 4.7% and 2.3% of the variability of AGB and ANPP respectively (Table 3.3).

Table 3.3 Evaluation of climate change and forest management effects on AGB and ANPP by ecoregion for the simulated years 2170 and 2310, by an analysis of variance (ANOVA).

Variable	Simulation year	Ecoregion	Climate		Forest Management		Climate*Forest Management	
			Variance explained (%)	P-value	Variance explained (%)	P-value	Variance explained (%)	P-value
AGB	160 (2170)	4a	0.43	< 0.001	98.73	< 0.001	0.06	0.63
		4b	0.07	< 0.05	99.53	< 0.001	0.04	0.48
		5a	38.97	< 0.001	57.59	< 0.001	0.40	0.34
		5b	0.27	0.15	97.11	< 0.001	0.21	0.55
		6a	54.16	< 0.001	40.09	< 0.001	0.08	0.97
		6c	15.47	< 0.001	58.73	< 0.001	3.76	0.21
	300 (2310)	4a	9.06	< 0.001	87.87	< 0.001	0.34	0.36
		4b	1.21	< 0.001	97.85	< 0.001	0.20	0.06
		5a	40.61	< 0.001	52.27	< 0.001	0.95	0.26
		5b	0.08	0.57	97.01	< 0.001	0.37	0.28
		6a	62.97	< 0.001	33.25	< 0.001	0.05	0.97
		6c	12.03	< 0.001	67.15	< 0.001	4.75	< 0.05
ANPP	160 (2170)	4a	0.64	< 0.001	98.83	<0.001	0.03	0.71
		4b	0.11	< 0.001	99.77	<0.001	0.03	0.05
		5a	88.38	< 0.001	6.75	<0.001	0.58	0.32
		5b	1.35	< 0.001	97.74	<0.001	0.14	0.18
		6a	89.42	< 0.001	0.21	0.70	0.08	0.99
		6c	78.10	< 0.001	1.95	0.15	2.34	0.33
	300 (2310)	4a	4.99	< 0.001	93.89	< 0.001	0.18	0.18
		4b	1.04	< 0.001	98.78	< 0.001	0.03	0.15
		5a	88.36	< 0.001	8.26	< 0.001	0.49	0.21
		5b	16.92	< 0.001	80.26	< 0.001	0.26	0.48
		6a	92.25	< 0.001	0.20	0.63	0.02	1.00
		6c	85.08	< 0.001	6.17	< 0.001	1.00	0.34

The AGB showed a reduction from 121 ton ha^{-1} at the beginning of the modeling period to about 69-75ton ha^{-1} at the end of the modeling period depending on the ecoregion, climate change, and forest management intensity scenarios (Figure 3.4). At the end of the modeling period, the highest AGB was observed in the ecoregion 6a with 107 ton ha^{-1} for the baseline, and the minimum AGB was in the ecoregion 5b and 6c with 85-

86 ton ha^{-1} for the scenario RCP 8.5_5030_2%. For all ecoregions, the difference in the AGB between the baseline and climate change scenarios was higher when the climate moved toward a lower mitigation scenario. Thus, the AGB decreased respect to the baseline in around 5%, 6% and 8% under RCP 2.6, RCP 4.5, and RCP 8.5 scenarios, respectively. For the forest management intensity scenarios, the AGB decreased respect to the baseline in around 8% for most intense scenario (5030_2%) and 5% for the less intense scenarios (7050_1%, 9070_0.5%). The spatial distribution of AGB showed a clear north-to-south and west-to-east gradient under the climate scenarios considered. The AGB was significantly higher in the ecoregions 4a and 4b (south of the study area) where the proportion of hardwoods is higher, intermediate in the ecoregions 5a and 6a (west of study area), and lower in the ecoregions 5b and 6c (northeast of study area) where the proportion of hardwoods is lower (Figure 3.4).

The ANPP was 7.4 ton ha^{-1} yr^{-1} at the beginning of the modeling period and was increased to values between 7.5 - 8.1 ton ha^{-1} yr^{-1} depending on the climate change and forest management intensity scenarios. The ANPP decreased in respect to the baseline in around 1.2%, 1.9% and 3% under RCP 2.6, RCP 4.5, and RCP 8.5 scenarios, respectively, and 3%, 1.5% and 0.7% under 5030_2%, 7050_1%, and 9070_0.5% scenarios. The baseline also showed a reduction in the ANPP along the latitudinal gradient, presenting a small reduction in the south (ecoregions 4a and 4b) and a large reduction in the northeast ecoregions (6a and 6c) confirming that the forests in the north grow slower than the forests in the south (Figure 3.5).

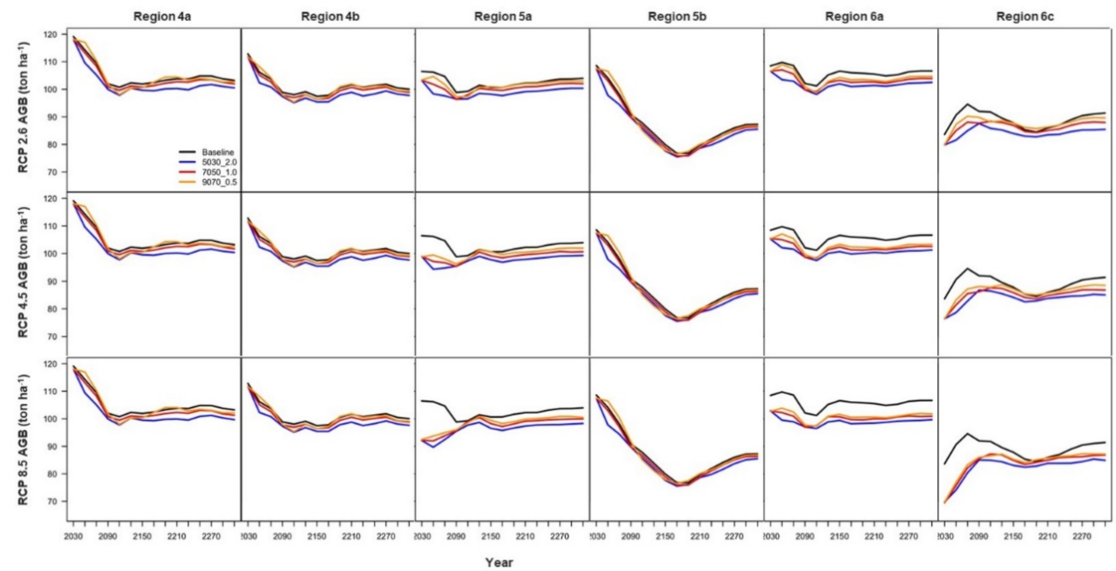

Figure 3.4 AGB (ton ha^{-1}) by the ecoregions 4a, 4b, 5a, 5b, 6a, and 6c under climate change scenarios (RCP 2.6, RCP 4.5, RCP 8.5) and the forest management intensity scenarios (9070_0.5%, 7050_1.0%, 5030_2.0 %; where the first two numbers indicate the age at which the conifers may be harvested, the subsequent two numbers the age at which the hardwood species may be harvested, and the percentage indicates the percentage of the area that may be harvested).

3.4.3 Forest type area

The area classified according to their composition within the forest landscape was stable through time in the baseline scenario. Conifers occupied between 50-60% of the landscape, the maximum values were present in the north (ecological regions 6a, 6c, 5a), and the minimum values were present in the south of the study area (ecological regions 4a, 4b, 5b). Hardwoods and mixedwoods occupied around 25-30% and 10-25% of the landscape respectively, with the maximum values present in the south and the minimum in the north of the study area (Figure 3.6, Annex 3-D).

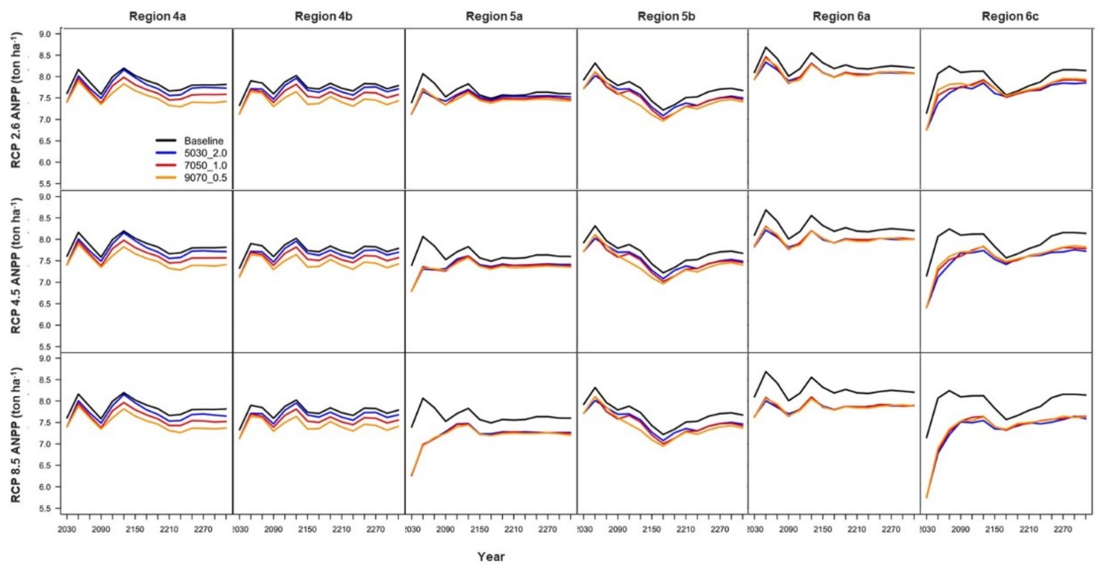

Figure 3.5 ANPP (ton ha^{-1}) by the ecoregions 4a, 4b, 5a, 5b, 6a, and 6c under climate change scenarios (RCP 2.6, RCP 4.5, RCP 8.5) and the forest management intensity scenarios (9070_0.5%, 7050_1.0%, 5030_2.0%; where the first two numbers indicate the age at what the conifers may harvested, the subsequent two numbers the age at what the hardwood species may be harvested, and the percentage indicates the percentage of the area that may be harvested).

The landscape composition changed dramatically with the intensification of climate change scenarios through the time. Under RCP 2.6 and RCP 4.5 the pattern was more similar to the baseline, although for the RCP 4.5 the proportion of conifer area decreased by 4%. The RCP 8.5 presented a dramatic decrease of conifers coverage of 15% in the landscape. Although the forest management scenarios did not have any significant influence in the temporal pattern of landscape composition, the proportion of hardwoods and mixedwood increased between 1% and 4% when the forest management was more intense (Figure 3.6).

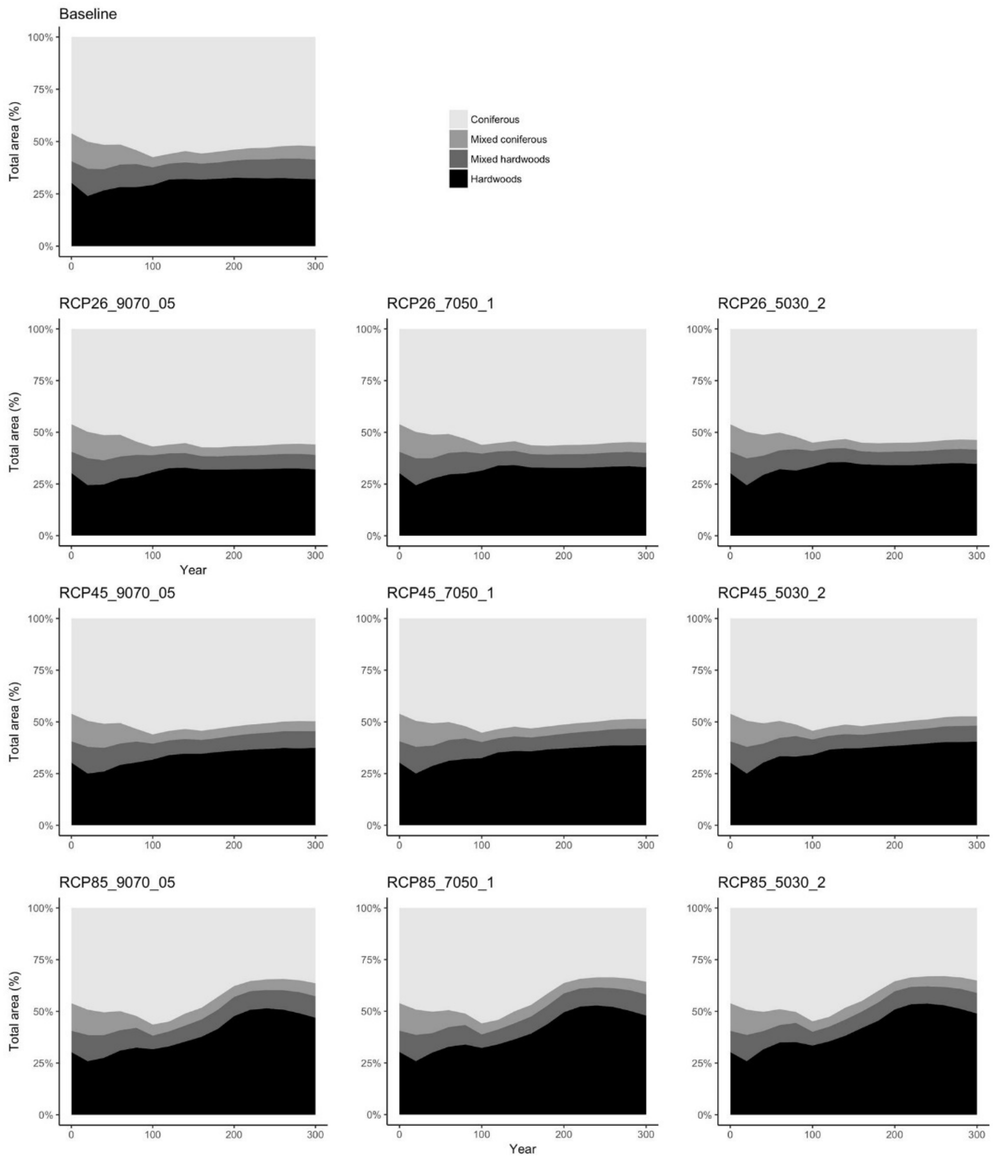

Figure 3.6 Forest landscape distribution according to forest composition type (%) through time by climate change scenarios (RCP 2.6, RCP 4.5, RCP 8.5) and forest management intensity scenarios (9070_0.5%, 7050_1.0%, 5030_2.0%; where the first two numbers indicate the age at which the conifers may be harvested, the subsequent two numbers the age at which the hardwood species may be harvested, and the last number indicates the percentage of the area that may be harvested).

3.5 Discussion

This study modeled the effects of climate change (expressed as variations in fire regimes) and forest management (expressed as intensity and age values) on boreal forest landscapes located in the eastern Canada (Abitibi Plain). The AGB, ANPP, and forest composition were simulated in the baseline scenario within the range of values reported by the literature for similar ecosystems and latitudes (Molina, Valeria, and De Grandpre 2018) (Annex 3-C). The results of the model projections for the different climate change and forest management scenarios showed that both disturbances considered have effects on AGB, ANPP and forest composition.

3.5.1 Fire regime and forest composition, AGB and ANPP

Burned area strongly differed between baseline and the climate change scenarios evaluated, increasing according to the severity of climate change scenarios (Figure 3.2). Our results, showed a maximum increase on burned area in 217% in the extreme climate change scenario, which is consistent with several models that project an increase in the burned area up to 300% due to higher fire occurrence and severity in some parts of Canada (Boulanger, Gauthier, and Burton 2014, Gauthier, Bernier, Boulanger, et al. 2015). Total area burned slightly fluctuated after 100 years even when climate warming conditions continued increasing. AGB accumulates rapidly after the fire. Both AGB and ANPP reached the maximum stand values during the early and middle successional stages responding to the age of the stand and the species composition. Then, ANPP declines with stand aging while the accumulated AGB remained more or less constant through the time (Gower, McMurtrie, and Murty 1996, Taylor, Wang, and Chen 2007, Wang, Bond-Lamberty, and Gower 2003). The deceleration of ANPP in middle and late-successional stages is related with a compositional change from fast-growing pioneer tree species (mostly hardwoods) to slower-growing ones that dominate during late-successional stages (conifers) (Seedre et al. 2014). Under climate change scenarios, the increase in the proportion of young stands because of the intensification of fire regimes produced a small decline in AGB stocks since fire lead to lose of a high proportion of mature stands with the biggest biomass stocks. This effect is more evident in the northern forests, where the proportion of mature and overmature stands is higher (stands with higher AGB stocks), as well as under the most extreme climatic scenario (RCP 8.5) where the fire regime will present the higher increase of fire frequency, extent, and severity.

An increase in young stands following fire events is accompanied by an increase in the hardwood and mixed hardwood stands proportion in the landscape and influenced the ANPP. Under the most severe climate change scenarios (with higher fire severity), most of the mature and overmature burned stands restarted in early or mid-successional stages with higher ANPP increasing the landscape productivity (Goulden et al. 2011). Our models showed that under more severe fire regimes most of the stands returned to the earliest successional stages with very low initial values of ANPP that increased very fast (Gower, McMurtrie, and Murty 1996). All of our results are consistent with the boreal forest projections in Canada that used similar fire models to ours and found that forest volume decreased due to increased fire incidence on the landscape (Girardin et al. 2013, Gustafson et al. 2010, Terrier et al. 2013). Temperature, humidity, and CO2 fertilization effects on ANPP were not included in this study, which may interact with other global warming effects and produce bigger differences between climate change scenarios (Girardin, Bernier, and Gauthier 2011, Hyvönen et al. 2007). For instance, a moderate increase in temperature (RCP 2.6) may have a temporary positive effect on boreal tree growth because the warming would extend the growing season and increase humidity, increasing or at least maintaining forests AGB and ANPP despite increased burn rate (D'Orangeville et al. 2018).

3.5.2 Forest management and AGB, ANPP and forest type spatial distribution

Forest management can produce an abrupt and significant change in forest composition, especially in the age class distributions and thus, in the AGB and ANPP (Boulanger et al. 2016, Gustafson et al. 2010, Seedre et al. 2014). It was found in our simulations that there is a significant effect of forest management on AGB and ANPP. Regardless it is expected that the burned area increases by 100% under the RCP 8.5, while under the most intense forest management scenario the harvested area is expected increase only by 2%, forest management seems to be the most significant disturbance in the southern forests (ecoregions 4a, 4b, and 5b), probably because there were more stands with the age and composition required by each harvesting prescription to deal with the annual allowable cut volume. Furthermore, there were no restrictions to harvest the maximum area even for the more intense forest management scenario. On the contrary, in the northern forests (ecoregions 5a, 6a, and 6c), there was a mixed effect of climate change and forest management. Probably, for those forests many areas suitable for harvesting were burned limiting the amount of area available for harvesting, and therefore, the harvested area could not achieve the maximum area possible according to the harvesting prescriptions (Boulanger, Girardin, et al. 2017).

According to our results, under the climate change scenarios and forest management strategies and prescription evaluated, timber supply will be at risk in the study area. The vulnerability of timber supply under a rise of 1°C (RCP 2.6) will be low due to a slight decrease of mature and overmature stands. However, under a rise of 3.7°C (RCP 8.5) timber supply will be more vulnerable due to a decrease of the ANPP as well as the amount of mature and overmature stands. In those extreme cases, some strategies should be applied to adapt forest management to climate change as described by Girardin et al. (2013), Spittlehouse and Stewart (2003), and Ogden and Innes (2007). For instance, it has been strongly suggested developing fire-smart landscapes by the manipulation of forest stand structure and composition. As hardwoods stands are less flammable than coniferous stands (Hély et al. 2010), these could be used combined with roads, lakes, and rivers to create strategic barriers to decrease the landscape susceptibility to fire, especially in large stands of mature or overmature conifer stands. Second, the reduction of fuel load, fuel continuity, or stand age distribution across a landscape by changing tree spacing and density and reducing standing dead trees or coarse woody debris on the forest floor could reduce the risk and extent of fires (Dale et al. 2001). Third, to mitigate the loss of harvestable wood volumes due to fire, the salvage logging in burned stands has been strongly recommended.

3.5.3 Modelling limitations and uncertainties

There are numerous sources of bias in Landis II (Gustafson et al. 2010) but in general, Landis II overlooks the vegetation response to climate change. Sensitivity analyses of Landis II showed that ANPP and AGB are highly sensitive to maximum ANPP and maximum AGB (Simons-Legaard, Legaard, and Weiskittel 2015). Thus, evaluating different climate warming scenarios possibly would have changed our results to some degree, but our conclusions would remain the same since we inferred the direction of trends rather than the magnitude of changes.

Climate change influences forest landscapes through different ways (i.e. tree species growth, mortality and migration (Neilson et al. 2005), nutrient cycles, shifts in atmospheric concentration of CO2 and soil nitrogen deposition, and their fertilizing effect (Girardin, Bernier, and Gauthier 2011, Thornton et al. 2007), changes in natural disturbance regimes such as fires (Flannigan et al. 2005) and insect outbreaks (Fleming and Volney 1995), variations in seasonal weather patterns -timing of spring thaw- (Kimball et al. 2000), etc.). However, despite that all those variables have direct effect on AGB, ANPP and forest composition, at the moment of this modelling exercise, Landis II did not have any extension to include them on the modeling. Variables such as temperature increase, precipitation variations, atmospheric concentration of CO2, nitrogen concentration, etc., could be included directly in a climatic extension to produce a more accurate AGB, ANPP trends under climate warming scenarios. Additionally, other natural disturbances such as insect outbreaks may have some impacts on the forest carbon balance (Fleming and Volney 1995), and their impact may increase in the future with increases in fire regimes (Fleming, Candau, and McAlpine 2002). However, regardless of the potential importance of secondary natural disturbances on AGB or ANPP, changes in the fire regimes were only incorporated in our modeling for determining the exact role of climate on all the natural disturbances, and then, the potential impact on AGB and ANPP was not part of our objectives.

Regardless of the modelling limitations described above, our results are considered reliable. One of the reasons is that Landis II simulations have been widely used in many studies and its validity reported in boreal forests worldwide (Dymond et al. 2016, Li et al. 2013, Scheller and Mladenoff 2004, Simons-Legaard, Legaard, and Weiskittel 2015). In addition, because this study was focused on studying the patterns, trends, and interaction in boreal forest under climate warming (fire regimes) and forest management intensities based on our current understanding of the processes that drive its landscape dynamics, instead of the analysis of the specific output values of AGB and ANPP.

3.6 Conclusions

Climate change described here as fire regime intensification and forest management intensity will have a significant effect on the AGB, ANPP and forest landscape composition in boreal forests of the study area. Thus, although under climate change scenarios the extent of burned area could be up to five times bigger than harvested area, the intensification of forest management seems to be the most important driver of the increase of hardwoods and mixed stands and the decrease of conifers stands, as well as the decrease of AGB and ANPP on the mixedwood boreal landscape, mainly in the southern forests.

CHAPTER IV

LONG-TERM LIMITATIONS OF FOREST MANAGEMENT TO EMULATE NATURAL POSTFIRE SUCCESSIONAL PATHWAYS IN NORTHEASTERN CANADIAN BOREAL FOREST. A MODELING APPROACH

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4.1 Abstract

Along with fire and its response to climate change, forest management is a crucial disturbance in boreal forests. Contrary to fire disturbances, forest management has been modifying the successional pathways leading to composition changes from initial conifer predominance forests to a mixed and/or hardwood forest. In this study, the LANDIS-II spatial explicit landscape model was used to evaluate the effects of future predicted fire regimes and forest management under climate change on the composition, succession, and spatial configuration in the northeastern Canadian boreal forests. This study found that contrary to successional pathways after fire, successional pathways after forest management favored mixed forest with prevalence of broadleaf species, even after 300 years. This trend is exacerbated under climate change scenarios, which will give advantage to forests dominated by shade-intolerant species, especially in the ecoregions where they have low presence (center and north of the study area). Additionally, forest management leads to more sinuous forest shapes at a landscape level, indicating an increase in forest fragmentation. Our results highlight the failure of the current forest management regime to emulate the effects of the natural disturbance regime on the landscape composition and the risk that it implies for the maintenance of ecosystem goods and services.

Keywords: Climate change, Forest management, Fire, LANDIS-II, Mixedwood Boreal forest, Forest composition, Successional pathways.

Résumé

Avec les feux de forêt et notamment leur comportement au sein du changement climatique, la gestion forestière constitue une perturbation principale dans les forêts boréales. Contrairement aux perturbations causées par les feux e forêt, la gestion forestière a modifié la succession des espèces forestières, entrainant des changements de feux de forêts, de prédominance de conifères à forêt mixte et / ou feuillue. Dans cette étude, le modèle spatial de paysage LANDIS-II a été utilisé pour évaluer les effets des futurs régimes de feux de forêt et de la gestion des forêts, soumis au changement climatique, sur la composition, la succession et la configuration spatiale dans les forêts boréales du nord-est du Canada. Cette étude a révélé que contrairement à de composition après un feu de forêt, la gestion forestière favorisait la forêt mixte avec une prévalence d'espèces feuillues, même après 300 ans. Cette tendance est exacerbée par les scénarios de changement climatique, qui profiteront aux forêts dominées par des espèces intolérantes à l'ombre, en particulier dans les écorégions où elles sont peu présentes (centre et nord de la zone d'étude). De plus, la gestion forestière conduit à des formes forestières plus sinueuses au niveau du paysage, indiquant une augmentation de la fragmentation des forêts. Nos résultats mettent en évidence l'échec du régime actuel de gestion forestière à imiter les effets du régime de perturbation naturelle sur la composition du paysage et le risque qu'il implique pour le maintien des biens et services écosystémiques.

Mots clés: *changement climatique, aménagement forestière, feux, LANDIS-II, foret boréal mélangée, composition forestière, Biomasse aérienne, composition forestière.*

4.2 Introduction

Natural disturbances are responsible for modeling the stand structure and the dynamics of the forest succession in the mixedwood boreal forest. For instance, when stand-replacing fires occur, four developmental stages are typipcally observed (Chen and Popadiouk 2002) i) stand initiation, where pioneer species colonize the area; ii) stem exclusion, where inter- and intra-specific competition occur as individual trees grow up, restricting the establishment of new stems; iii) canopy transition, where trees start to decline and die because of longevity or damage from non-stand-replacing disturbances, and where shade-tolerant coniferous trees from the understory and intermediate canopy take over the main canopy; and iv) gap dynamics, where conifer trees dominate the stand and individual or groups of trees create stands dominated by hardwoods or conifer species. However, in stands hardwoods can be maintained in large gaps, where light conditions are favorable for their establishment. These successional steps produce multi-cohort stands instead of a simple replacement of hardwoods by conifers species, and therefore, it results in a high proportion of mixed stands of hardwood and conifers species (Bergeron et al. 2014).

In eastern Canada, primary natural disturbances, such as forest fires, and secondary disturbances, such as insect outbreaks, are the main drivers of forest dynamics. However, for the past 50 years mechanised harvesting has played an increasingly important role in forest dynamics (Bergeron et al. 2001). After fire stands typically go through the following stages: 1) the dominance of shade-intolerant species, usually hardwoods such as aspen (*Populus sp.*), birch (*Betula sp.*), and occasionally Jack pine (*Pinus banksiana* Lamb.), immediately after fire and last 100 years approximately (Harvey et al. 2002); (2) the decline of these shade-intolerant species and the recruitment of subsequent intermediate shade-tolerant species mixed with conifer species (with abundant spruce (*Picea* sp.)) between 75 and 175 years after the fire, and, (3) the replacement of old hardwood by coniferous species dominated by black spruce (*Picea mariana* Mill.), balsam fir (*Abies balsamea* (L.) Mill.) and white cedar (*Thuja occidentalis* L.) after 150 years after fire (Bergeron 2000, Brassard and Chen 2006).

In northestern Canada, there is a wide variety of boreal forests associated to the south-to-north climate gradient, to the mosaic of edaphic conditions, and to the prevalence or frequency of various natural disturbances such as fire, insect outbreaks and gap dynamics (Bergeron et al. 2014, Bergeron et al. 2004). In these forests, it is common to observe a variety of stands type according to the tree species composition related to stand age dynamics. For instance, in the north of the climate gradient, the higher proportion of stands are dominated by black spruce accompanied occasionally by balsam fir, whereas the center of this gradient is dominated by hardwood or mixed stands with trembling aspen (*Populus tremuloides* Michx.), white birch (*Betula papyrifera* Marshall) and jack pin. Finally, the southernmost part of the gradient the higher proportion of stands are dominated by yellow birch (*Betula alleghaniensis* Britt.) and conifers (Bergeron et al. 2014).

Thus, although the course of the successional process usually converges towards mixed stands or stands dominated by conifers in almost a predictable way, the succession pathways can result in a wide variety of stands in terms of structure and composition, because the successional process is also function of the interaction of species traits such as shade tolerance, fire adaptation, mode of regeneration, species longevity, and the growth rate (Seedre et al. 2014, Frelich and Reich 1995, Bergeron 2000). Additionally, biotic interactions, abiotic conditions, and the type of disturbance drive successional pathways (Bergeron et al. 2014, Chen and Popadiouk 2002). For instance, where a high-burn rate occurs, a mixture of hardwood and conifer species such as aspen, birch, and spruce dominate and limit species replacement over time, while where a lower burn rate occurs, the proportion of old-growth forests dominated by conifers increases, and the forests remain in a dynamic equilibrium maintained by secondary disturbances such as low intensity fires, insect outbreaks, or small canopy disturbances (Bergeron and Dubue 1988, De Grandpré, Gagnon, and Bergeron 1993). Under global warming scenarios, it is expected that burn rate will increase, impacting considerably the structure, composition, and functioning of forest landscapes (Seidl, Rammer, and Spies 2014).

Along with fire disturbance and its response to climate change, forest management is today the second most important disturbance that annually affects the larger areas in the boreal forest. During the last decade Forest Ecosystem Management (FEM) has emerged as a new paradigm for forest management that aims to reproduce forest structure and composition created by natural disturbances to reproduce the mosaic of stands in terms of composition, structure, and age (Bergeron and Harvey 1997, Landres, Morgan, and Swanson 1999, Pothier, Raulier, and Riopel 2004, Jones, Domke, and Thomas 2009). Despite all the efforts to promote FEM, the mixture of traditional and FEM practices is producing a large-scale change from initial conifer-dominated forests to mixed and/or hardwood-dominated forests, which are simultaneously modifying the frequency of stand type at the landscape level and accelerating forest rotation at a landscape scale (Venier, Thompson, Fleming, Malcolm, Aubin, Trofymow, Langor, Sturrock, Patry, and Outerbridge 2014, Boucher et al. 2009).

This study aimed to evaluate the effect of fire regimes (burn rate) and forest management on the composition, succession, and spatial configuration of the mixedwood boreal forests under climate change and forest management scenarios. It is expected that after fire the landscape will be composed of stands dominated by shade-intolerant and fire-adapted species that will be replaced by a mixture of hardwoods and conifers with mid-tolerance or tolerant to fire, and eventually, the forest will be dominated by conifers. On the contrary, it is expected that after forest management the landscape will be composed of stand with a mixture of hardwoods and conifers with different levels of shade tolerance. With respect to stand type distribution at a landscape scale, it is expected that under extreme climate change (higher fire frequency) and high-intensity forest management (twice-actual the harvest intensity) scenarios, the southern part of the study area will be dominated by stands of shade-intolerant and fire-adapted species, while the northern part of the study area will be dominated by mixed stands of shade-intolerant or mid-tolerant species. Those changes in forest composition will be accompanied by an increase in forest fragmentation, more complex patch shapes, and more isolated patches compared to the initial landscape. In this context, Landis II, a spatially explicit model was used to simulate the landscape dynamics in response to fire and forest management under climate change scenarios, as well as to identify the influence on the spatial distribution of the various stand types.

4.2.1 Study area

This study covers an area of 67600 km2 of boreal forests located at the northeast of Canada (Abitibi Plain) between 47°30'-49°30' N and 76°30'-79°30' W (Figure 4.1). The climate is subpolar and sub-humid continental, with warm and short summers, and cold, long, and snowy winters. Temperature and precipitation differ across the study area, in the north, the average annual temperature is 1.2ºC, and annual precipitation is 917 mm (La Sarre meteorological station), while in the south the average annual temperature is 3.4ºC, and the average annual precipitation is 831 mm (Ville-Marie meteorological station). The annual burn rate in the study area moves from 0.239% of the forest area in the north to 0.036% in the south (Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, Lesieur, et al. 2006).

The study area is mostly located within the clay belt deposit left by the pro-glacial lake Ojibway (Veillette 1994). This area includes portions of six ecological regions (Table 4.1): 4a and 4b (temperate mixedwood forests), 5a and 5b (mixedwood boreal forests), 6a and 6c (Boreal conifer forests) (Ministère de la Forêt, de la Faune et des Parcs du Quebec - MFFP)(Robitaille 1988). In this study, the ecoregions layer was delimitated according to the ecological regions defined by the Quebec's MFFP (Table 4.1) (Saucier et al. 2011).

Table 4.1 Ecoregions in the study area

Ecological region		Bioclimatic subdomain	Dominant forest cover	Area		Current burn rate (%)
				Total (km)	Studied area (%)	
4a	Plains and hills of Simard lake	Balsam fir-Yellow birch domain	Mixed stands of Yellow birch and other softwoods	5943	79	0.048
4b	Cabonga watershed hillside			27429	52	0.036
5a	Abitibi plains	Balsam fir-White birch domain	Hardwoods or mixed stands with intolerant hardwoods (Trembling Aspen, White birch, and Jackpine)	26842	89	0.258
5b	Gouin watershed hillside		Balsam fir and White spruce stands mixed with White birch	15758	51	0.048
6a	Matagami lake plains	Spruce-Moss domain	Black spruce with occasional balsam fir	48842	18	0.239
6c	Opémisca lake plane		Black spruce	21428	37	0.239

In the north, the temperate mixedwood forests (spruce-feathermoss bioclimatic domain, ecological regions 6a and 6c) are dominated by conifer stands of black spruce with occasional balsam fir distributed in the homogeneous landscape of mature stands. The mixedwood boreal forests, (balsam fir-white birch bioclimatic domain, ecological regions 5a and 5b), in the center of the study area, are dominated by hardwood or mixed stands with intolerant hardwood as trembling aspen, white birch and jack pine configuring a more heterogeneous (less aggregated) landscape. In the south, the temperate mixedwood forests (balsam fir-yellow birch bioclimatic domain, ecological regions 4a and 4c) are dominated by mixed stands of yellow birch and conifers in a more fragmented landscape (Molina, Valeria, and De Grandpre 2018, Saucier et al. 1998).

4.2.2 Modeling of succession after fire and harvesting, and under different climate change scenarios

4.2.2.1 Inputs setting

LANDIS-II 6.1 (Scheller et al. 2007), a spatially explicit landscape model with a core extension structure was used in this study to simulate tree species dynamics under natural and anthropogenic disturbances (Gustafson et al. 2000a). Aboveground biomass (AGB) stocks by species and by pixel was simulated on the entire study area (Scheller and Domingo 2012). Five repetitions of each model scenario were run for 300 years (2010-2310) at a 20-year time step and 200 m resolution pixels (4 ha).

The LANDIS-II core was fed with four data sets: first, an ecological regions layer that divides the landscape by similarity of soil and climatic conditions according to the delimitation defined by the Quebec's MFFP (ecological regions 4a, 4b, 5a, 5b, 6a, and 6c, see Table 4.1). Second, the life-history traits of the species with higher dominance in our study area (13 species, Annex 4-A) (Qualtiere 2012, Scheller et al. 2007, Scheller et al. 2008, USDA 2014, Xu, Gertner, and Scheller 2009). Third, an initial communities' layer that represents the actual distribution of species by age-cohorts across the study area elaborated from the fourth decennial forest inventory map of

Quebec (Ministère des Ressources Naturelles du Québec 2013). And fourth, a species-specific establishment probability by ecoregion estimated according to the proportion of area occupied by each species by ecoregion from the fourth decennial forest inventory of Quebec (Ministère des Ressources Naturelles du Québec 2013).

Fire and forest management were simulated using the extensions Base-Fire and Base-Harvest, respectively. The fire extension Base-Fire (He and Mladenoff 1999) was calibrated with a layer that compiled historical forest fire events registered by the forest fire agency (Société de Protection des Forêts Contre le Feu, SOPFEU) between 1941 and 2006, and from Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, and Lesieur (2006), who did an estimation of the burn rate and ignition probability in our study area. The extension Base-Harvest (Gustafson et al. 2000b) considered that the study area was divided into 14 forest management units (FMU) with two harvesting prescriptions, i.e. careful logging around advanced growth (CLAAG) and partial cutting (which includes commercial thinning, shelterwood, and selection cutting). Prescription was established following the annual allowable cut (AAC) volume calculation for the 2013-2018 period established by Quebec's chief forester (Bureau du Forestier en Chef 2013).

4.2.2.2 Modeling scenarios

A baseline scenario with the burn and forest management intensity that took place in 2010 was designed, i.e., burn annual rate between 0.048 and 0.239% of the study area annually (Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, and Lesieur 2006) and forest harvesting allowance of around 1% of the study area (Bureau du Forestier en Chef 2013). Also, three climate change scenarios and three harvesting scenarios were considered. The fire regime scenarios were designed following three different RCPs climate change scenarios. The RCP 2.6 is a very low emission scenario with an increase of 0.9-2.3°C by 2100 (Van Vuuren et al. 2007), the RCP 4.5 is an intermediate stabilization scenario with an increase of 1.7-3.2°C by 2011 (Clarke et al. 2007b), and the RCP 8.5 is a low mitigation scenario with an increase of 3.2-5.4°C by 2100 (Riahi et al. 2011). The input values of fire regimes, the net primary productivity (ANPP) and aboveground biomass (AGB) necessary to run the model were extracted from Bergeron et al. (2011) and Boulanger, Taylor, et al. (2017). The three forest management intensity scenarios were designed according to the maximum area that can be harvested under actual AAC values by FMU and the minimum age where conifers and hardwoods may be harvested. Forest management intensity scenarios were: 1) low-intensity, with harvest ratio 0.5% of the forest per period and set to 90 and 70 years the minimum age to harvest conifers and hardwoods respectively (refer here as 9070_0.5%); 2) mid-intensity, with harvesting ratio (close to the actual) 1% of the forest per period and set to 70 and 50 years the minimum age to harvest conifers and hardwoods respectively (refered here as 7050_1%); and 3) high-intensity, with harvest ratio 2% of the forest per period and set to 50 and 30 years the minimum age to harvest conifers and hardwoods respectively, to indicate that they will not have an age or tree size restriction to be harvested (refered here as 5030_2%). Each scenario was coded according to the climate change scenario (RCP 2.6, RCP 4.5, RCP 8.5), followed by the forest management regime, for instance, the more extreme scenario corresponds to RCP8.5_5030_2%. Each scenario was replicated five times to meet the minimum sample number for statistical analysis.

4.2.3 Successional pathways after fire and forest management by pixel

For the two disturbances origin fire and forest management, successional pathways curves were created by accounting the AGB accumulated by species by year and by each individual pixel disturbed by fire or harvested with the maximum intensity at the beginning of the disturbance modeling time. For each disturbance origin, five disturbed pixels were randomly selected by ecological region (6), scenario (9), and scenario replication (5), for a total sample of 1350 pixels (54 km2) by disturbance origin. At the same time, the 13 species were regrouped according to their 1) fire adaptation (adapted or not adapted), 2) shade tolerance (intolerant, mid-tolerant, tolerant, or very tolerant), and 3) functional type (conifers or hardwoods). For the three grouping, succession pathways curves were created with the sum of the biomass of the all species that compose each category, by grouping and averaged by disturbance origin.

To identify the grouping which most closely explains the variability in AGB as a function of disturbance, scenarios, ecoregions, and time; an analysis of variance partitioning was performed (Borcard, Legendre, and Drapeau 1992, Peres-Neto et al. 2006).

4.2.4 Landscape maps construction

Maps were created according to the AGB accumulated by pixel and each category of each grouping as follows: For group 1, pixels were classified as fire adapted when ≥ 70% of the AGB corresponded to fire-adapted species, and not fire adapted when ≥ 70% of the AGB corresponded to not fire adapted species, and mixed when the AGB was not classified as adapted or non-adapted. For group 2, pixels were classified as intolerant when ≥70% of the AGB corresponds to shade-intolerant species, mid-tolerant when ≥70% of the AGB corresponds to shade mid-tolerant species, tolerant when ≥70% of the AGB corresponds to shade-tolerant species, and very tolerant when ≥ 70% of the AGB corresponds to very shade tolerant species. The pixels that did not meet any of these conditions were excluded. These maps were created for the years 0 and 300 to be able to identify the main changes between scenarios for the complete study period.

4.2.5 Landscape metrics of the study area

A metric analysis of the reclassified forest maps was performed with the software FRAGSTAT v4.2 (McGarigal, Cushman, and Ene 2012) for the year 0 and 300 (current and maximum year modeled), and for the baseline and the most extreme scenario (RCP 8.5 5030_2%) to obtain the most contrasting changes throuth the time and between scenarios. Three metrics were used to explain the spatial variability of AGB of the area (mean area “AREA”), shape (perimeter-area ratio “PARA”), and aggregation (aggregation index “AI”). AREA describes the mean stand area in hectares by class. PARA is a simple measure of shape complexity by dividing the perimeter by the area; PARA is equal to 1 when the shape is a square and increases as the forms become less regular or more sinuous, it means when there are more border effect and less core area. AI shows the frequency with which different pairs of patch types appear side-by-side on the map, measuring the level of aggregation of each class stands.

4.3 Results

4.3.1 Succession pathways after fire and forest management

Figure 4.1 shows the projections of AGB across a 300-year period. After fire the species adapted to this disturbance increased their AGB until 120 years (maximum 115 ton /ha), then the AGB decreased rapidly (55 ton/ha). On the contrary, species not adapted to fire increased their AGB slowly but steadily during the evaluation period (5 ton/ha) until reaching the same AGB as species adapted to fire at 300 years of modelling (50 ton/ha) (Figure 4.1a). For the other groupings, the intolerant and mid-tolerant to shade species, as well as the hardwood species, presented a similar pattern to those species adapted to fire (Figure 4.1b, c).

Contrary to the successional pattern presented after the fire, harvesting favored the accumulation of AGB from species not adapted to fire during the evaluation period (Figure 4.1d). Contrary to fire, after harvesting, shade-intolerant species decreased their proportion of AGB after the disturbance, while medium-tolerant, tolerant, and very tolerant species began to increase their AGB slowly but steadily after 100 years until the end of the evaluation period (Figure e, f).

The test of significance of the partition of variance indicates that between the grouping of three species analyzed fire adaptation is the best descriptor to explain AGB and composition in post-fire stands (7% of variance explained), while shade tolerance describes better AGB and composition in post-harvested stands (4% of variance explained) (Annex 4-B). The composition grouping explained less than 1% of the variance, less than the other two grouping systems for both disturbances of origin. The other variables such as ecological regions and the modeling scenarios (three for climate change and three for forest management) explained altogether less than 1% of the variance.

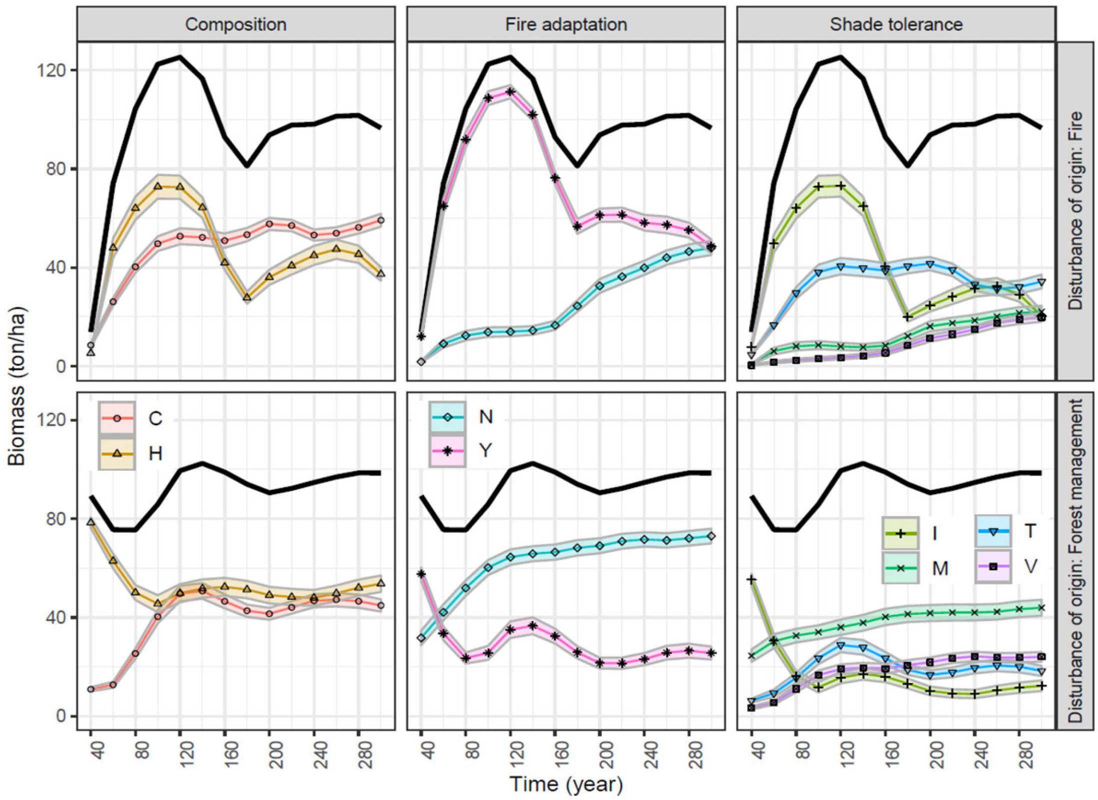


Figure 4.1 Successional pathways of AGB (ton/ha) under different disturbances of origin and for three species grouping over the simulation period. 1) Composition: H: a group of hardwoods species, and C: a group of conifer species. 2) Fire adaptation: Y: a group of species with fire adaptations, and N: group of species without fire adaptations. 3) Shade tolerance: I: a group of shade-intolerant species, M: a group of species with mid-tolerance to shade, T: a group of shade-tolerant species, and V: a group of species very tolerant to shade. The solid black line represents the total AGB (ton/ha).

4.3.2 Spatial changes in the successional pathways under climate change and forest management scenarios

The spatial distribution of the groupings and their classes through time in the baseline and RCP 8.5_5030_2% scenarios are shown in Figure 4.2. At the beginning of the analysis period stands dominated by species adapted to fire dominated the entire study area (Figure 4.2a). After 300 years, in the baseline and in the RCP 8.5_5032_2% scenarios, the proportion of stands dominated by species not adapted to fire increased in the south of the study area (ecological regions 4a, 4b and 5b), while the species adapted to fire increased in the north (ecological regions 5a, 6a and 6c). This trend is more pronounced under the extreme climate change and forest management scenario, where the proportion of stands dominated by fire-adapted species in the north was even higher than under the baseline (Figure 4.2 b, c).

At the beginning of the analysis period in the shade tolerance grouping, the south and east of the study area had a higher proportion of stands dominated by shade-intolerant species, while the north had a higher proportion of stands dominated by shade-tolerant and shade very tolerant species (Figure 4.2 d). After 300 years, the proportion of stands dominated by intolerant and medium shade-tolerant species in the baseline increased in the south (ecological regions 4a, 4b, and 5b), and tolerant in the north of the study area (ecological regions 5a, 6a, and 6c) (Figure 4.2e). Under the RCP 8.5_5032_2% scenario, a higher proportion of stands dominated by shade-intolerant species was observed in the south and center of the study area (ecological regions 4a, 4b, 5a, and 5b), while the north of the study area remained dominated by shade-tolerant stands (ecological regions 5a, 6a, and 6c) (Figure 4.2 f).

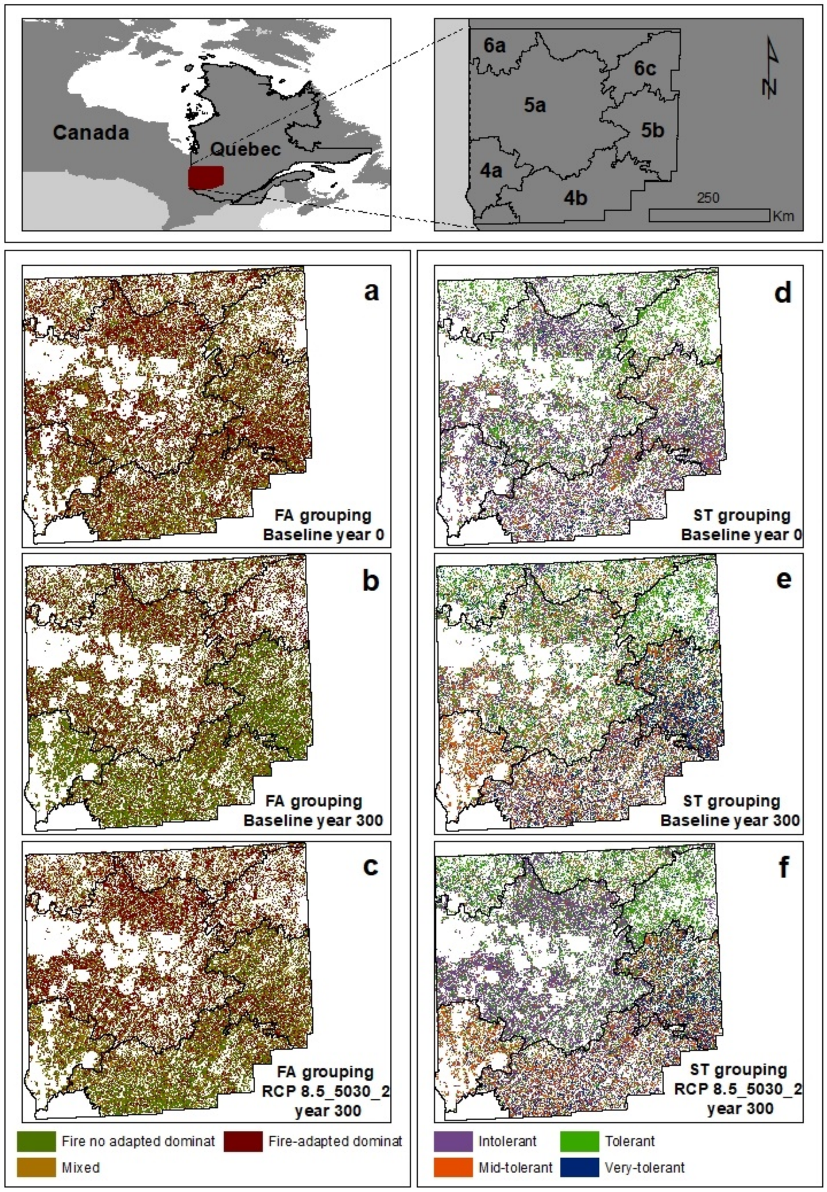


Figure 42 Study area location map, and maps for fire adaptation and shade tolerance groupings under Baseline and RCP 8.5_5032_2% scenarios, at the beginning (0) and at the end of simulation time (300). White areas represent agricultural areas or water bodies.

Under the baseline scenario (burn rate 0.048-0.239% of the study area, 7050_1%), the landscape metrics showed that the stands dominated by fire non-adapted species increased in mean area (AREA) through the time while mixed and adapted species stands decreased. Those changes are simultaneously presented with the increase through the simulation time of the stands shape complexity (PARA) which indicates an increase of stand border area, and a decreased of the aggregation (AI) of the stands by class. Metrics for the shade tolerance grouping show that stands dominated by intolerant species decreased in mean area (AREA) until the year 160, then all the classes (I, M, T, and V) remained more or less steady. Similar to fire adaptation, the border area (PARA) increased through the time while stand aggregation (AI) decreased (Figure 4.3).

Under extreme climate change and forest management modeling scenario (RCP 8.5_5030-2%), stands dominated by fire adapted and non-adapted species slightly increased its mean area through the time, while mixed stands (stands with a proportion of fire-adapted or non-adapted species is lower than 70%) decreased its mean area. Also, the shape complexity (PARA) increased for the two groupings and plotted classes, but especially for the mixed species stands (stands with a proportion of fire-adapted or non-adapted species is lower than 70%). The opposite trend was shown by the stand aggregation (AI). With respect to shade tolerance, mean stand area (AREA) of shade-intolerant stands showed an opposite trend than the one observed in the baseline, increasing the value through the simulation period. Similar to the baseline, the shape complexity (PARA) increased through the time while stand aggregation (AI) decreased (Figure 4.3)

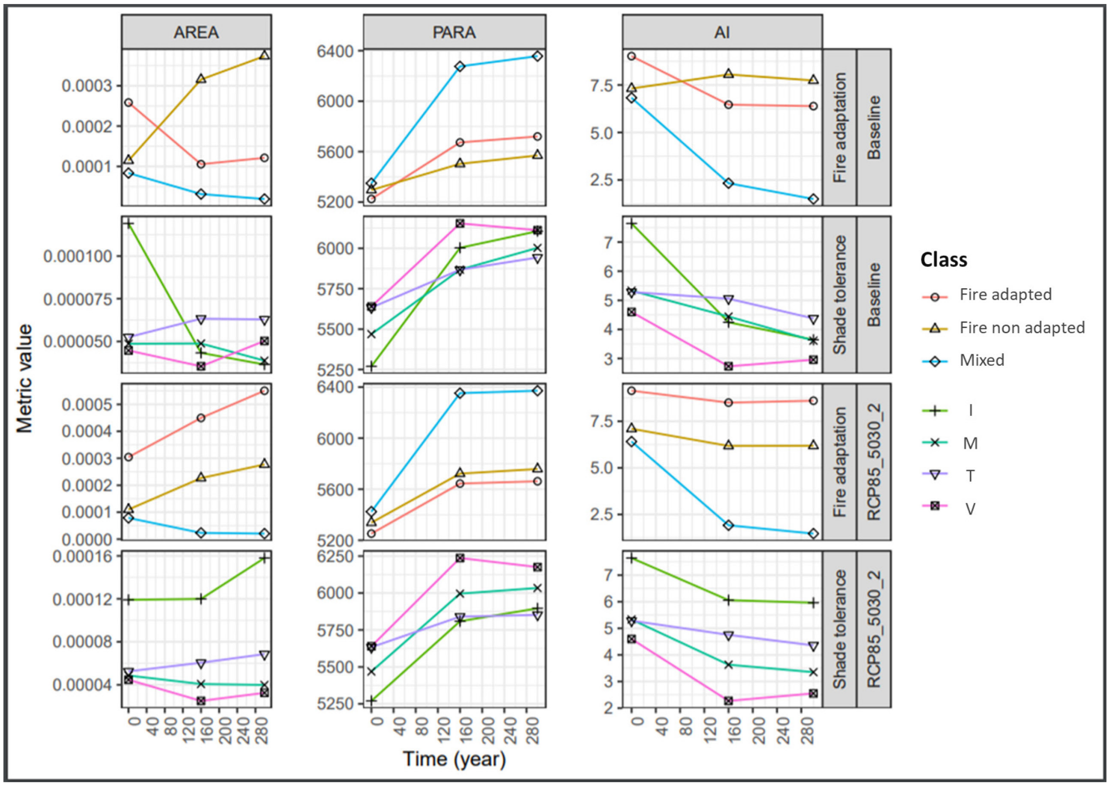


Figure 4.3 Landscape metrics (Area, PARA and AI) for the groupings 1) fire adaptation and 2) shade tolerance under modeling scenarios Baseline and the most severe climate change and forest management scenario RCP 8.5_5030-2%.

4.4 Discussion

At the landscape level, natural disturbances reinitiate forest succession at different temporal and spatial scales. Fire is the dominant natural disturbance in boreal forests, which, at the landscape level, creates a mosaic of forest stands with different ages, structure and composition since last burning (Johnson 1992, Payette 1992). Overall, fire induces changes in forest stands in composition through time, from broadleaf deciduous trees (hardwoods) or mixed stand to a mixture of deciduous trees with some conifers (mixedwoods), and finally to a dominated by conifers (Bergeron and Fenton 2012), however many different pathways are possible according to pre-fire stand composition and site conditions (Bergeron et al. 2002, Bergeron et al. 2014). This process mainly results by the different shade tolerances of the species that dominate each stand and their different growth rates, as well as their fire adaptations (Simard, Bergeron, and Sirois 2003, Bergeron and Fenton 2012). For example, the shade-intolerant trembling aspen and white birch may regenerate rapidly after fire or harvesting by root suckering, and stump or sprouts respectively, while the pines, spruces, fir, and white cedar regenerate basically by seeds respectively, indeed, jack pine and black spruce are fire-adapted species through serotinous and semi-serotinous cones, and white spruce and white cedar seeds show clear preference for downed dead wood for germination. However, in absence of fire shade-tolerant species such as black spruce may regenerate by layering (Burns and Honkala 1990b, Greene et al. 1999).

Current forest management was thought as a system for timber production that tries to reproduce stand composition and structure at spatial and temporal scales with the aim of reducing the ecological differences between natural and managed landscapes preserving in the landscape spatial extent and distribution of disturbed areas. This type of management should maintain the forest landscape structure (Gauthier et al. 2009). Nevertheless, this study found that contrary to postfire successional pathways, after forest management, succession favored mixed forest with higher prevalence of hardwoods stands, even after 300 years. This change in forest composition, compared to that caused by fire, suggests the alteration of two key attributes that contribute to resilience and resistance of this ecosystem: biodiversity and biological legacies (Kuuluvainen and Gauthier 2018). A reduction of the forest biodiversity implies lower ecosystem redundancy, which is necessary to respond to disturbances and stresses. Whereas the change in the resulting species indicates an effect on the successional pathway due to loss of biological legacies (Gauthier et al. 2009).

The occurrence of fire gives an advantage to the species with fire adaptation over species without fire adaptations. Thus, fire adapted species have an advantage at early and mid-successional stage stands, while species without fire adaption are only able to colonize and occupy some space just after the first successional stages (Bergeron and Dansereau 1993, De Grandpre et al. 2003). For instance, after fire it is usual to find species such as aspen and jack pine colonizing quickly burned areas, since they are shade-intolerant species adapted to fire, while fir and spruces can establish at the same time than aspen, but as they have a lower growth rate they stay in the understory for a longer time, until canopy gaps create conditions that increase their growth rate (Vaillancourt et al. 2009). On the contrary, sometimes forest management favors the preestablished regeneration of shade-tolerant conifers, regardless their fire adaptations (Groot et al. 2005, Cimon-Morin, Ruel, and Darveau 2010). In the early successional stages, after forest management, the shade-tolerant species established under the canopy or residual trees have an advantage over the shade-intolerant species, which began to compete at the end of the early succession stages of the stands or in the intermediate successional stage (Vepakomma, St-Onge, and Kneeshaw 2011, Bose et al. 2013). Thus, depending on pre-disturbance stand composition and type of disturbance forest, stands do not return the forest to its initial successional stage because the protected regeneration contains a bigger proportion of balsam fir and conifers (Boucher et al. 2015a), which leads to later hardwoods such as aspen that dominates or co-dominates the stands due to their competitive advantage over conifers to colonize open areas (Madoui et al. 2015, Bergeron and Harvey 1997). However, the stand compositional response to harvesting will depend on pre-harvesting stand composition and harvesting method. Aspen-dominated, mixed and conifer-dominated stands are likely to respond differently to partial cutting and clearcutting.

This difference at stand level is seen reflected on a landscape scale. Forest management has increased spatial fragmentation and will increase this trend in the future (Figure 4.3). In particular, the landscape of northwestern Quebec has become progressively more heterogeneous since the beginning of forest management (Boucher et al. 2015a, Molina, Valeria, and De Grandpre 2018). This fragmentation has produced more complex patch shapes, lower core areas, and more isolated patches that have changed the landscape composition and affected the relative abundance of the conifer dominated, mixedwood and hardwood dominated stands Figure 4.3 (Molina, Valeria, and De Grandpre 2018).

Climate change has been impacting boreal forest by direct mechanisms such as the increase of temperature and by indirect mechanisms that modify natural disturbance regimes such as fires, insect epidemics, diseases, and windfall (Le Goff et al. 2009). It is predicted that the temperature will increase 2 to 5˚C by the end of the current century, with a great impact on boreal forest (Team, Pachauri, and Meyer 2014). This change will lead to drying conditions and the subsequent rise of fire frequency and burned area in wide areas of boreal forests and also create drought stress (and related dieback and increased vulnerability to pests and diseases) in some species. (Bergeron, Cyr, Drever, Flannigan, Gauthier, Kneeshaw, Lauzon, Leduc, Goff, Lesieur, et al. 2006, Bergeron et al. 2011). This study considered the effects of climate change as a function of the changes on fire frequency. Future higher burn rate of boreal forest will give advantage to forests dominated by shade-intolerant and fire-adapted species, especially in the ecoregions where they have lower presence (center and north of the study area). For example, frequent fires will favor the recurrence of jack pine or birch dominated stands where jack pine may be present with or without black spruce (Bergeron and Dubue 1988). Additionally, climate change will exacerbate the effects of forest management producing more sinuous forest shapes and, at a landscape level, more fragmented areas (Figure 4.3) (Li et al. 2013).

4.4.1 Implication for forest management

As it has been described, the current forest management has not been able to reproduce the postfire successional pathways. The increment of the harvested forests under current forest management will lead to a diminution of the abundance of species typical of mid or late successional stages, which are today the species most used by the timber industry. These changes will be accompanied by change in the abundance of species with higher economic value such as spruce and balsam fir, while the abundance of the lower economic value species will increase like birch and aspen. Current forest management allows the harvesting of timber from balsam fir, spruce, pine and larch principally (MFFP 2017), however, under all climate change and forest management scenarios, most of these species will be less abundant. It implies that the timber industry will have a higher proportion of less valuable timbers and then, they will need to find new solutions for sustainable revenue production. To mitigate the depletion of conifer timbers Kruhlov et al. (2018) recommend adapting the abundance of the species at landscape scales, considering changing climate and landscape gradients by enriching the understory with the proper species to ensure the appearance of conifers in early and mid-successional stages. Regeneration enrichment with conifers will shorten the time at which conifers appear in the stands, and maybe be able to dominate in mid and late successional stages, moment in which postfire stands are usually dominated by conifers. Therefore, the conifers timber volume probably will reach current values by applying this kind of strategies. Also, development of alternative silvicultural interventions (different types of partial cuttings) that would emulate secondary disturbances (e.g. wind, insects) rather than fire would maintain pre-industrial forest characteristics (e.g. composition and age class distribution), and then allow the maintenance of a conifer forest cover (Girardin et al. 2013).

4.4.2 Modeling limitations and uncertainties

There are numerous limitations associated to the use of Landis II, and accordingly, our results were found to be limited because of two reasons. First, climate change has multiple direct impacts on species because the physiological responses are altered by the changing pattern of the environment. As a consequence, AGB, ANPP, and in general the forest composition is strongly modified by climate change. Unfortunately, Landis II did not have any extension to include this kind of environmental parametrizations when the modeling exercise was developed. For this reason, variables such as temperature increase, precipitation variations, atmospheric concentration of CO2 and nitrogen concentration were not included explicitly. Additionally, tree species distribution is also influenced by the fertilizing effect associated with soil nitrogen deposition and atmospheric concentration of CO2 (Thornton et al. 2007, Girardin et al. 2011), however these variables cannot be included in Landis II modeling.

Secondly, forest management prescription where not drawn with all the details needed to evaluate in detail the real effect of them in contrast with the fire. For instance, the regeneration enrichment post-harvesting cannot be included as a post-disturbance strategy, also post fire logging was not included despite its importance in the timber supply of high value species. Additionally, there is not a way to introduce the adaptive advantage some species to mechanical disturbances, for example, aspen, white birch, and red maple are species well adapted to regenerate after mechanical disturbance favoring the regeneration of these species after harvesting when they were present before disturbance.

Landis II demonstrated to be a very useful tool to conduct controlled experiment to discover general trends of boreal ecosystem response to several spatial changes at stand and landscape level, given our current understanding of the ecological processes that structure the boreal forests and landscapes (Gustafson et al. 2010) Therefore, all the bias described above do not influence the relative change between species, ecological regions, and scenarios, because it was assumed that all scenarios contained the same bias, and also because the relative changes between scenarios were evaluated rather than the magnitude of the changes provided by Landis II.

4.5 Conclusions

Our modeling results indicate that landscapes after forest management practices undertaken for wood production will have a negative impact on the structure, composition, and in the patch size distribution at landscape level. Overall, management practices, combined with a changing climate will imply a risk for forest health and biodiversity, and hence, for the maintenance of ecosystem goods and services. Our results highlight the limitation of the current forest management to emulate natural postfire succession and suggest the need to modify the system to a one inspired by natural forest dynamics and disturbances and at the same time maintain economic and social sustainability, mitigate and adapt to climate change, and safeguard its ecosystem services (Gauthier, Bernier, Kuuluvainen, et al. 2015).

4.6 Acknowledgments

We wish to thank the NSERC, RYAM, and Norbord industries for supporting this research. We would like to thank the Ministère des Forêts, de la Faune et des Parcs, and its Forest inventory department for their support to provide the digital inventory data to calibrate Landis II. NSERC/UQAT/UQAM Industrial Chair in Sustainable Forest Management (SFM Chair) also helped us in this project by providing all the technical support. We are also grateful to Johana Herrera and Javier Peinado for the compilation and construction of some input's datasets, and 3 anonymous reviewers for critically reading the manuscript and helping us to improve it.

CHAPTER V

5.1 General conclusions

This study was conducted to enhance our understanding of the potential effect that the climate change, understood as changes in fire regimes and forest management will have on stands and landscape structure and composition in the Canadian northeastern boreal forests. Compared to most previous studies done in similar forest types, this thesis studies the landscape historical response to the simultaneous effects of the two most important stand replacement disturbances, wildfire and forest management (*Chapter 2*), and includes as well a modeling exercise to evaluate the longer-term stand and landscape responses to those disturbances, specifically the response of AGB and ANPP (*Chapter 3*), and successional pathways (*Chapter 4*) were evaluated.

Initially, in *Chapter 1*, a literature review about the ecology of boreal mixedwood forest and the implications of forest management was presented. In the *Chapter 2*, a characterization of the landscape heterogeneity (composition and configuration) during recent decades was presented using Landsat imagery from 1985 to 2013. The results of this study showed that the previous, fire-influenced, old-growth conifer forests that dominated the mixedwood boreal forest landscape in eastern Canada were transformed by forestry practices between 1985 and 2013 toward a more heterogeneous one. Traditionally, forest management involved the harvesting of extensive areas of conifers changing the relative abundance of the forest classes, resulting in the decrease in large, contiguous areas of conifer class (12.4% of the initial conifer area). At the same time, the landscape became more fragmented, with more complex patch shapes, lower core areas, and more isolated patches. Except for the loss of forest aggregation, the change of these metrics showed a deceleration after 1995, which might be a consequence of the change in forestry practices at that time from traditional forest management (clearcutting of large areas) to FEM. Indeed, our modeling indicates that FEM will have a negative impact on the structure, composition and in the patch size distribution at a landscape level in comparison with postfire natural landscapes.

Using the landscape explicit model LANDIS-II, in *Chapter 3* the effects that fire and forest management will have on aboveground biomass, productivity, and forest composition under expected climate change were presented. According to our modeling exercise, the climate change understood here as fire regime intensification, and forest management will have a significant effect on the AGB, ANPP and forest landscape composition. Thus, although under climate change scenarios the extent of burned area will be about five times bigger than harvested area, the intensification of forest management seems to be the most important driver of the increase of hardwoods and mixed stands, as well the decrease of the AGB and ANPP, mainly in the southern forests.

In *Chapter 4* LANDIS-II was used to simulate successional trajectories in response to fire and forest management under climate change scenarios, as well as to identify the influence on the spatial stands' distribution. Our modeling results indicate that the current forest management conduce to a diminution of the abundance of fire non-adapted species typical of mid or late successional stages, which corresponds to the species with actual higher economic value such as spruce and balsam fir, while the abundance of the lower economic value species like birch and aspen increase after this type of disturbance. The forest management and the regeneration enrichment after harvesting with conifers will shorten the time at which the conifers appear in the stands, but because of the first successional stages are missed these conifers species are not able to dominate the stand even after 300 years, moment in which postfire stands are dominated by conifers. Also, the landscape configurated by forest management practices will have a negative impact on the structure, composition and in the patch size distribution at landscape level in comparison with postfire forests.

5.2 Forest management recommendations

Due to the limitations of the LANDIS-II model, this study cannot express the real magnitude of the impact of the forest management under the climate change on the mixedwood boreal forests. However, our results indicated that forest management had strong negative effect on the stands and landscape scale. The forest landscape product of decades of clearcuttings is now more heterogeneous and fragmented that the postfire one, and with lower proportion of conifer mature stands. Additionally, the current forest management combined with a changing climate will imply a risk for forest health and biodiversity, and hence, for the maintenance of ecosystem goods and services.

Our results highlight the incapacity of current forest management to emulate natural postfire succession and landscape and suggest the need to modify the system to a one inspired by natural forest dynamics and disturbances at stand and landscape scales, and at the same time maintain economic and social sustainability, mitigate and adapt to climate change, and safeguard biodiversity and related ecosystem services. For instance, FEM slowed down the loss of Conifer cover detriment in the past. Nonetheless, landscape metrics did not stabilize or recovery. Even more, FEM did not and will not slow down landscape fragmentation. The forest management applied in the study area has been designed at Forest Management Unit scale, as it was demonstrated in this study, if the dynamics of future fires regimes under climate change scenarios are desired to be considered, this scale of planification seems to be inappropriate. It will be necessary to manage to forest landscape at broader scales. In that case, it could be possible design the forests management according functional zoning at regional scale where the forest is divided into areas of conservation, intensive forest harvesting, and multiple uses help to limit the spatial extent of the human footprint on the landscape, and producing in that case more aggregated forest landscape and limit the spatial extent of the human footprint.

Furthermore, to maintain a sustainable timber production will imply alternative silvicultural interventions additional to the currents. These alternative interventions should help to shorten the time at what the conifers appears in the stands, and the proportion of C stands in landscape. For instance, the partial cutting should be applied in more than 10% of the harvested areas. In that way partial cuttings could emulate secondary disturbances (e.g. wind, insects) rather than fire and it could maintain pre-industrial forest characteristics (e.g. composition and age class distribution), and then allow the maintenance of a conifer forest cover. Also, fast growing conifer plantation should be stablished to ensure the supply of a proportion of high value timber to the market and decrease the pressure on the natural forest.

ANNEXES

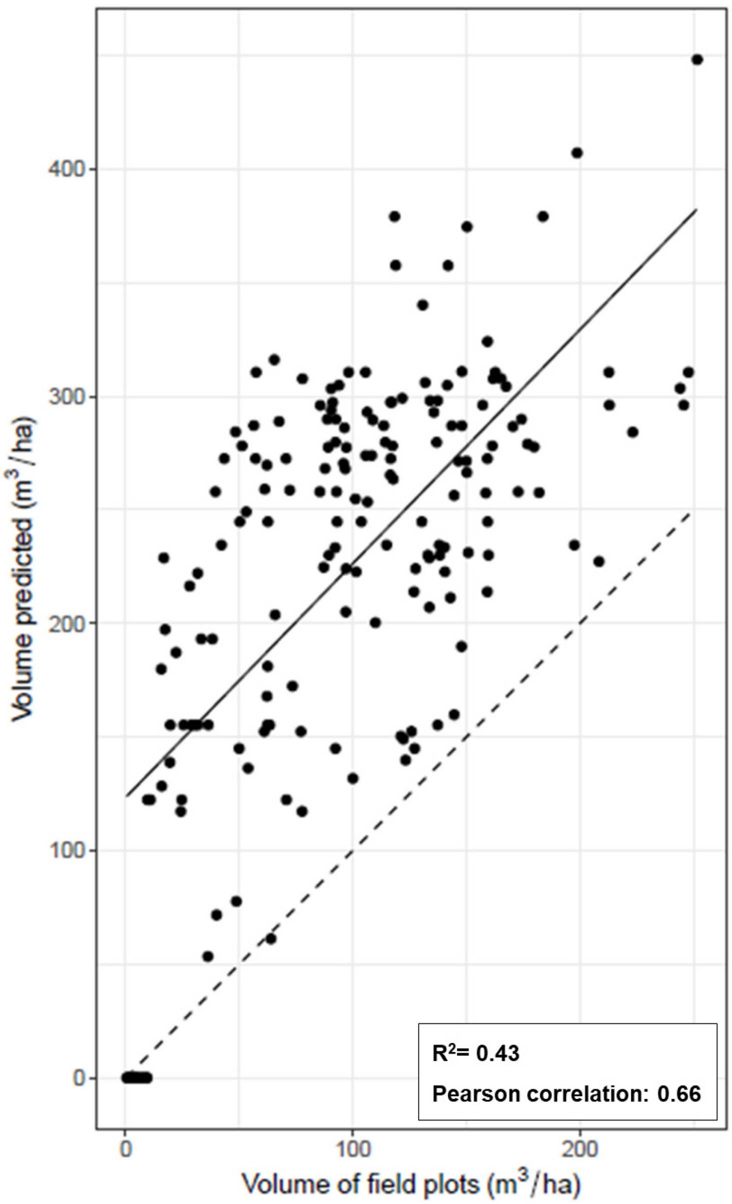
Annex 3-A Establishment probability by species and ecoregion

Specie	4a	4b	5a	5b	6a	6c
Gray birch	0.017	0.005	0.014	0.015	0.074	0.008
Yellow birch	0.029	0.142	0.001	0.009	0.000	0.000
White birch	0.369	0.597	0.271	0.438	0.154	0.121
White spruce	0.020	0.016	0.015	0.005	0.019	0.028
Black spruce	0.361	0.373	0.523	0.571	0.574	0.807
Red spruce	0.213	0.255	0.001	0.059	0.000	0.000
Red maple	0.049	0.125	0.004	0.015	0.013	0.000
Sugar maple	0.002	0.020	0.000	0.000	0.000	0.000
Tamarack	0.028	0.012	0.056	0.020	0.004	0.002
Balsam poplar	0.246	0.097	0.217	0.085	0.181	0.047
Largetooth aspen	0.243	0.097	0.217	0.085	0.181	0.047
Trembling aspen	0.352	0.141	0.321	0.125	0.287	0.085
Eastern white pine	0.024	0.021	0.000	0.000	0.000	0.000
Jack pine	0.151	0.140	0.216	0.272	0.303	0.303
Red pine	0.009	0.002	0.000	0.000	0.000	0.000
Balsam fir	0.249	0.178	0.157	0.193	0.111	0.128
Eastern white-cedar	0.016	0.044	0.002	0.000	0.000	0.000

Annex 3-B Forest management intensity scenarios. CPRS: cutting with protection of regeneration and soils, CT: partial cuttings prescriptions

Forest management unit - Prescription	**Allowable cutting area scenarios**			
	Historical (Baseline)	**Less intense**	**Current conditions**	**More intense**
1-085-51-CPRS	1.78%	0.50%	1%	2%
2-085-51-CP	0.12%			
3-086-52-CPRS	1.91%	0.50%	1%	2%
4-086-52-CP	0.19%			
5-087-51-CPRS	1.81%	0.50%	1%	2%
6-087-51-CP	0.09%			
7-087-63-CPRS	1.46%	0.50%	1%	2%
8-087-63-CP	0.15%			
9-087-62-CPRS	1.40%	0.50%	1%	2%
10-087-62-CP	0.10%			
11-087-64-CPRS	1.59%	0.50%	1%	2%
12-087-64-CP	0.11%			
13-084-62-CPRS	1.82%	0.50%	1%	2%
14-084-62-CP	0.09%			
15-084-51-CPRS	1.90%	0.50%	1%	2%
16-084-51-CP	0.10%			
17-086-51-CPRS	1.97%	0.50%	1%	2%
18-086-51-CP	0.13%			
19-082-51-CPRS	1.62%	0.50%	1%	2%
20-082-51-CP	0.18%			
21-083-51-CPRS	1.51%	0.50%	1%	2%
22-083-51-CP	0.09%			
23-081-52-CPRS	0.65%	0.50%	1%	2%
24-081-52-CP	0.45%			
25-074-51-CPRS	1.08%	0.50%	1%	2%
26-074-51-CP	0.32%			
27-073-52-CPRS	0.77%	0.50%	1%	2%
28-073-52-CP	0.33%			
Minimum age to harvest				
Conifers	70	90	70	50
Hardwoods	50	70	50	30

Annex 3-C Model validation. The validation was done by comparing the biomass reported by the fourth forest inventory plot dataset and the predicted data from LANDIS II. The continuous line represents the adjusted model and the dotted line represents the 1:1 ratio.



Annex 3-D AGB and ANPP values reported by the literature for ecosystems similar to the ones evaluated in this study.

Location	**Biomass (T/ha)**	**Reference**
Forest dominated by Populus tremuloides Quebec.	173	David P. and Bergeron, Y. (Dec, 1995)
Quebec (sites with more than 80% of the biomass being *Populus tremuloides*).	294	David P. and Bergeron, Y. (Dec, 1995)
Ontario Canada.	202.14	Liam 2004
Western Newfoundland in Quebec (model with images).	68-178	Fournier 2003
Lower Laurentian Mountains in Quebec (model with images).	33-177	Fournier 2003
Quebec	70	Penner, M., Power, K., Muhairwe, C., Tellier, R., and Wang, Y. 1997. Inventario Canada.
Ontario	87	
Southern Arctic	43	
Taiga plains	67	
Shield	40	
Boreal shield	72	
Atlantic Maritime	84	
Mixedwood plains	91	
Boreal Plains	78	
Prairies	76	
Taiga cordillera	56	
Boreal cordillera	83	
Pacific Maritime	238	
Montane Cordillera	152	
Hudson Plains	60	
Northern Sweden about 50 km Northwest of Umeå - Krycklan River.	94.98	Biosar. 2008
Northen Wisconsin	54	Zheng et al, 2004
Newfoundland	63.6	Luther et al, in press
Alberta Canada	114	Hall. 2006. Modelling Biomass
Trois-Rivières	127.8	Boudreau. 2008. Regional Forest Biomass Quebec
Mont-Laurier	63.3	
Rivière-du-Loup	84.2	
Lac St-Jean	61.4	
Baie-Comeau	81.9	

Annex 3-D AGB and ANPP values reported by the literature for ecosystems similar to the ones evaluated in this study.

Location	**Biomass (T/ha)**	**Reference**
Chibougamau	81.2	
Radisson	34.5	
Québec commercial forests (MRNFPQ Inventory).	83.7	
North American boreal foresta	41.8	Botkin, D, B., & Simpson, L, G, (1990)
Laurentian highlands	68	Botkin, D, B., Simpson, L, G. & Nisbet, R, A, (1993)
St, Lawrence Lowlands	58	Botkin, D, B., Simpson, L, G. & Nisbet, R, A, (1993)
Eastern North America Temperate Deciduous Forest.	80.5	Botkin, D, B., Simpson, L, G. & Nisbet, R, A, (1993)
Canada's commercial forests	90.9	Penner, M., Power, K., Muhairwe, C., Tellier, R. & Wang, Y, (1997)
Québec's commercial forests	70	Penner, M., Power, K., Muhairwe, C., Tellier, R. & Wang, Y., (1997)
Vancouver, Canada Regenerating stands and mature forest	20 - 550	Tsui, 2012
Canadian Boreal Zones (Atlantic Maritime, Boreal Cordillera, Boreal Plains, Boreal Shield East, Boreal Shield West, Hudson Plains, Taiga Cordillera, Taiga Plains, Taiga Shield East, Taiga Shield West)	52.62	Matasci_2018

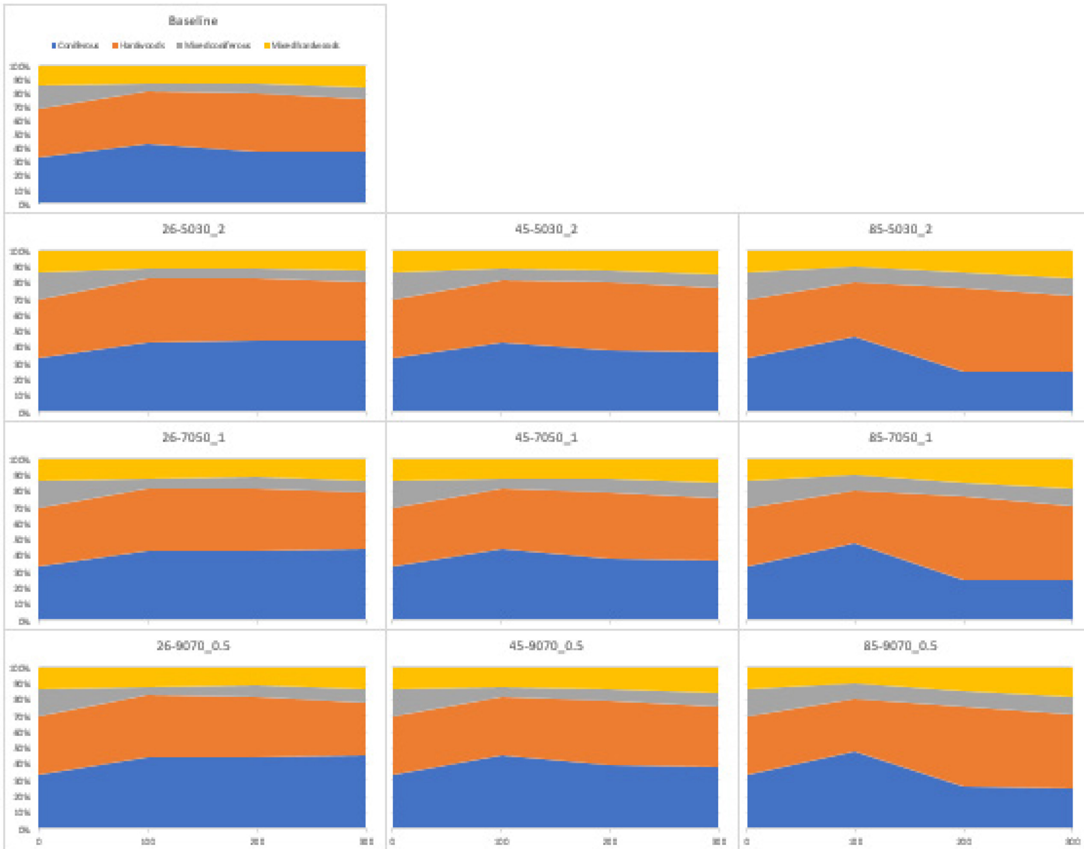
Location	**Species**	**g C /m2/ yr**	**NPPT** **g C /m2/ yr**	**Reference**
Saskatchewan, Canada	*Picea mariana*	146		
Saskatchewan, Canada Manitoba, Canada	*Pinus banksiana*	121	265	Gower, S. T., Krankina, O., Olson, R. J., Apps, M., Linder, S., & Wang, C. (2001). Ecological applications, 1
	Populus tremuloides	352	226	
	Picea mariana	129	394	
Manitoba, Canada Russia	*Pinus banksiana*	121	218	
	Populus tremuloides	349	221	
	Evergreen conifer forest Class I	359	415	
Russia Nordic Country (Sweden-Finland)	Evergreen conifer forest Class II	309	628	
	Deciduous Stand	65		

Annex 3-D AGB and ANPP values reported by the literature for ecosystems similar to the ones evaluated in this study.

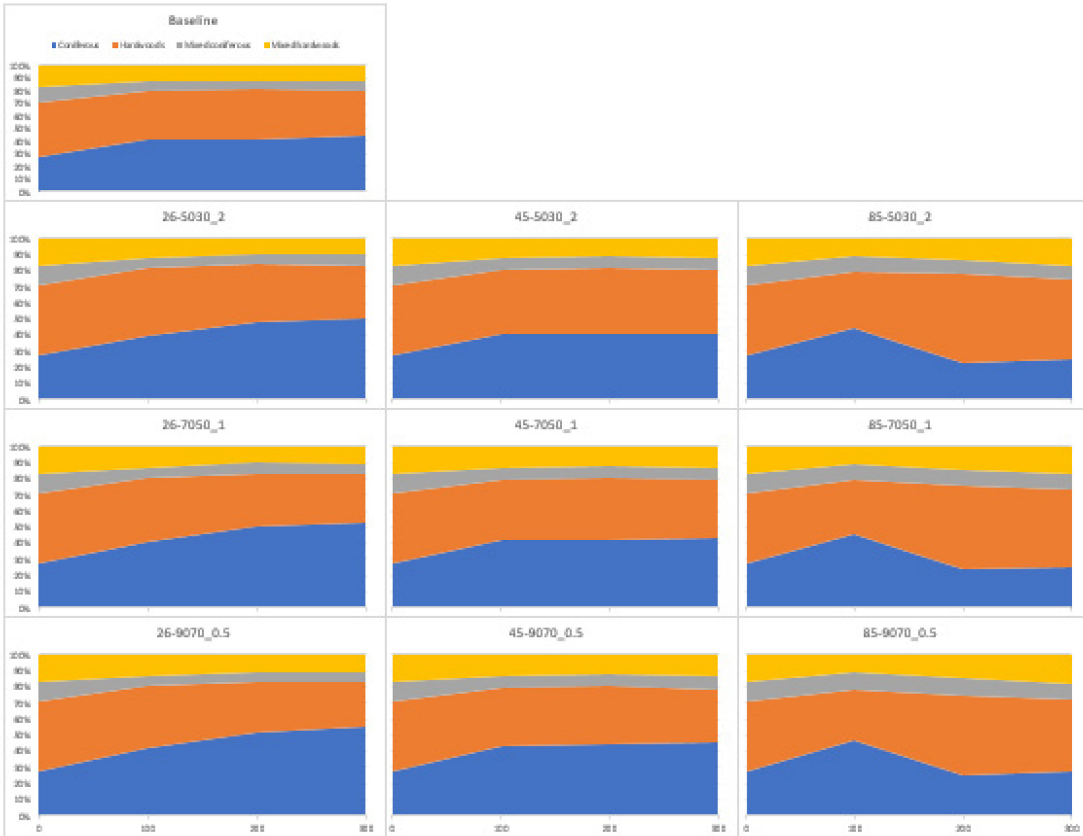
Location		**Biomass (T/ha)**	**Reference**	
	Evergreen conifer forest Class I	177		1(5), 1395-1411.
Nordic Country (Sweden-Finland) United states	Evergreen conifer forest Class II	100	321	
	Evergreen conifer forest Class II	94		
China	*Larix gmelinii*-Deciduous stand	269	NA	
Thomson, Canada		122-387	323	
Prince Albert, Canada		117-380		Peng, C., & Apps, M. J. (1999). Ecological Modelling, 122(3), 175-193.
Manitoba, Canada	*Picea mariana*	132		
Manitoba, Canada Saskatchewan, Canada	*Pinus banksiana*	115	252	Ryan, M., Lavigne, M.B. & Gower, S.T. (1997). Journal of Geophysical Research, 102, 28871–28883.
	Populus tremuloides	342	229	
	Picea mariana	147	426	
Saskatchewan, Canada Ontario, Canada	*Pinus banksiana*	115	307	
	Populus tremuloides	361	237	
	Black spruce (Picea mariana (Mill.)		440	

Annex 3-E Forest type by scenarios and ecological regions

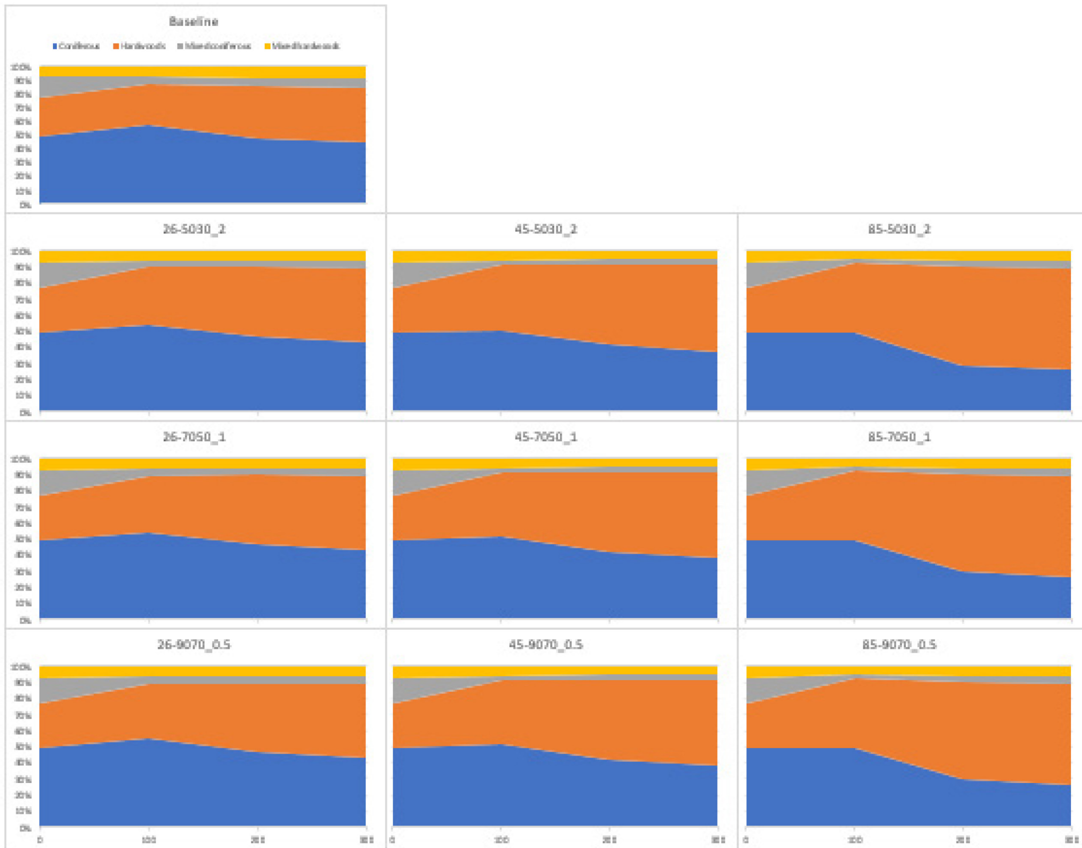
Ecological region 4a



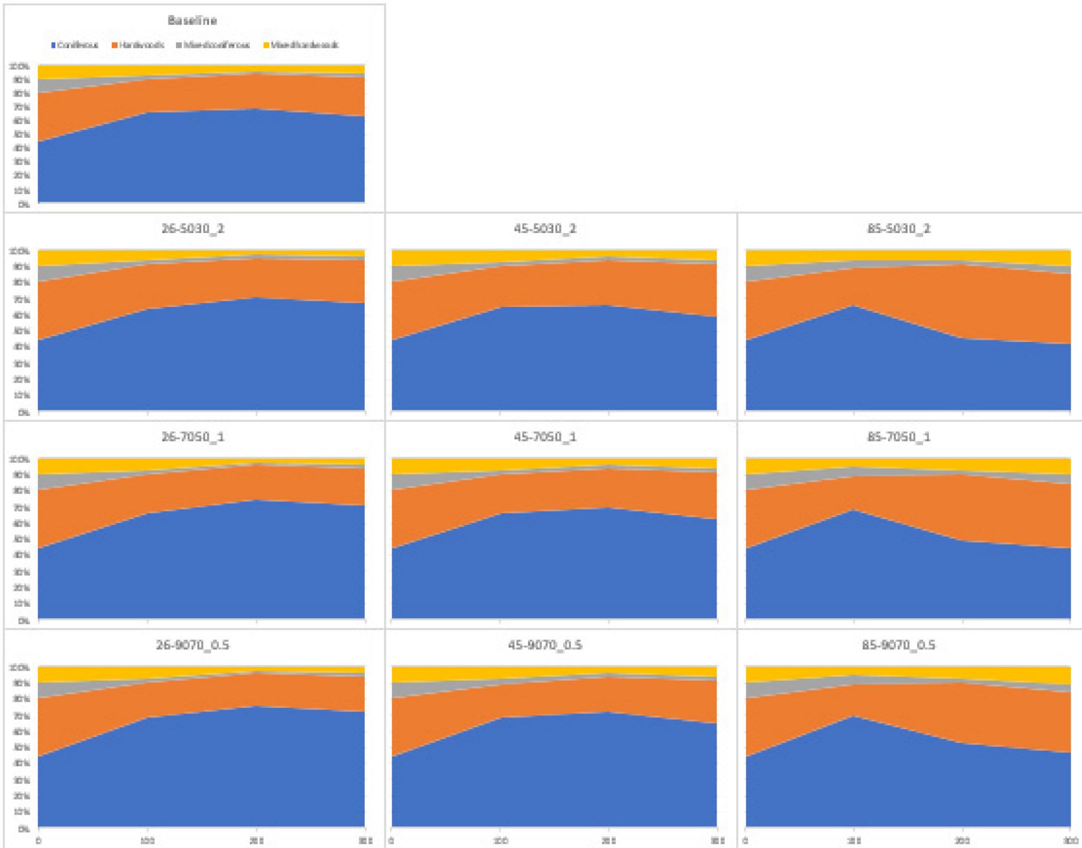
Ecological region 4b



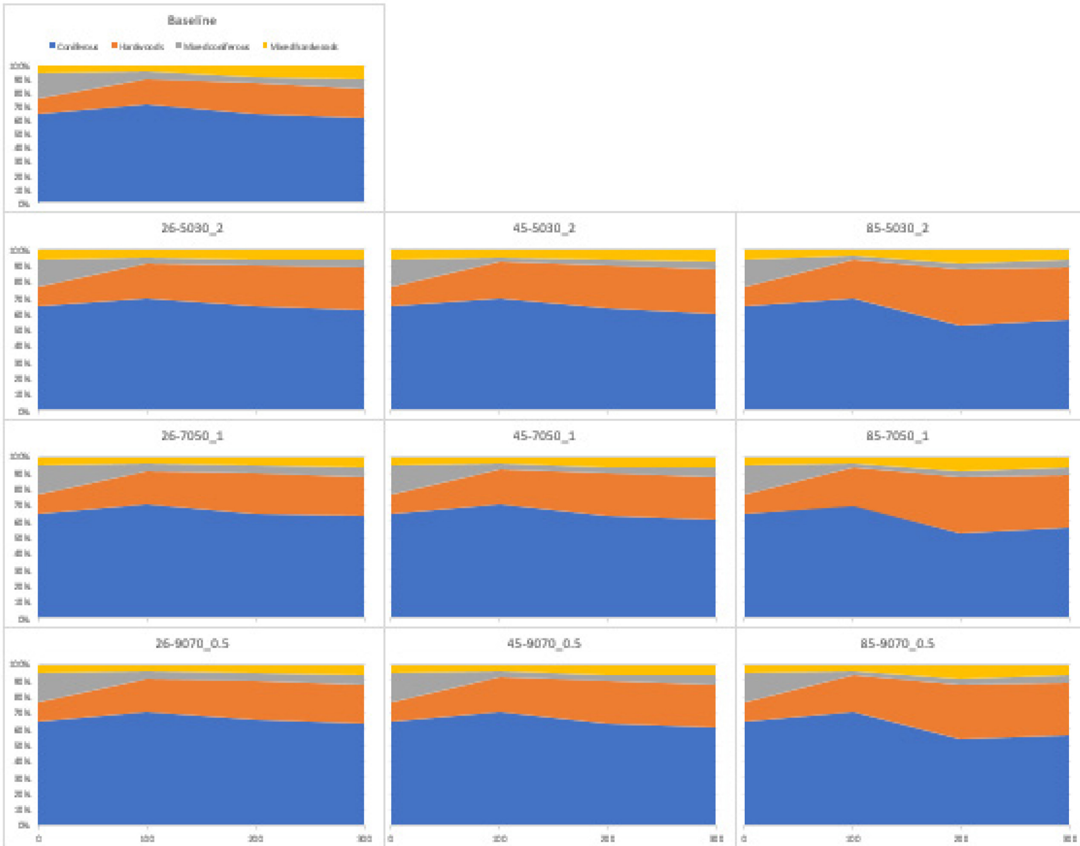
Ecological region 5a



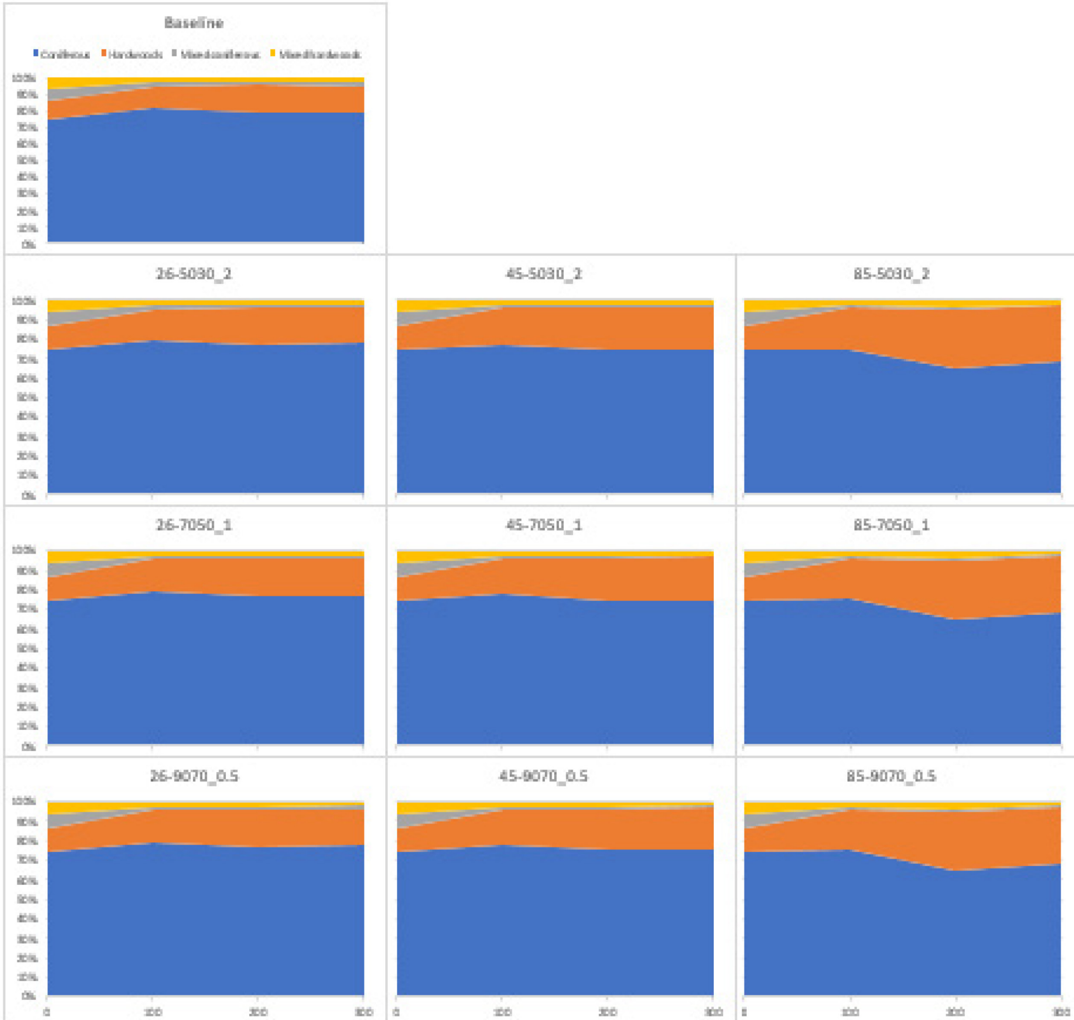
Ecological region 5b



Ecological region 6a



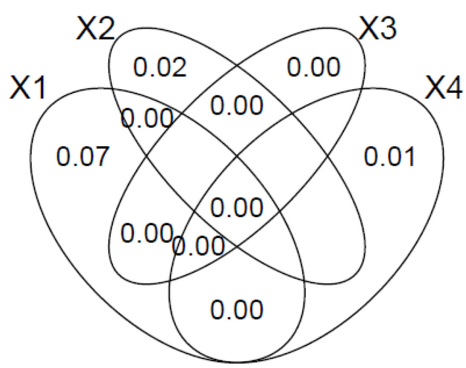
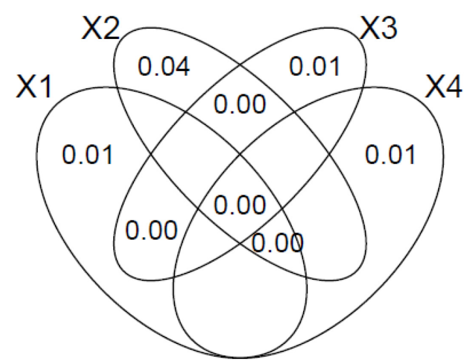
Ecological region 6c



Annex 4-A List of species included in the modeling and their functional traits, where N: non-adapted to fire, Y: fire-adapted, I: shade-intolerant, M: shade mid-tolerant, T: shade-tolerant, V: shade very-tolerant, C: conifers, and H: hardwoods.

Specie	**Fire adaptation**	**Shade tolerance**	**Conifer-Hardwood**
White_birch	Y	I	H
Trembling_aspen	Y	I	H
Jack_pine	Y	I	C
Black_spruce	Y	T	C
Tamarack	N	I	C
Red_pine	N	I	C
Red_maple	N	M	H
Yellow_birch	N	M	H
East_whitepine	N	M	C
Sugar_maple	N	T	H
White_spruce	N	T	C
Balsam_fir	N	V	C
East_whitecedar	N	V	C

Annex 4-B Test of significant of partition of variance of AGB according to species grouping, type of disturbance of origin, ecological regions, scenarios, and time. X1: species grouping according to their fire adaptation, X2: species grouping according to their shade tolerance, X3: species grouping if they are conifer or broadleaf, and X4: Time+ecological region+scenario. Values <0 were not shown.

Disturbance of origin: Fire	Disturbance of origin: Forest management
	

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