Feasibility of an ecosystem-based management in an eastern Canadian boreal forest:
Testing for ecological suitability, economic viability, social acceptability and adaptability to wildfire and climate change

Thèse

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Résumé

Dans la quête de la mise en place d'une stratégie d'aménagement écosystémique (AE) dans la forêt boréale de l'est du Canada, nous avons réalisé une étude de faisabilité sur la viabilité économique, l'acceptabilité sociale et la pertinence écologique. À l'aide de modèles d'approvisionnement en bois, nous avons comparé une stratégie AE à une stratégie de normalisation des forêts (*status quo*) afin d'en évaluer sa robustesse et son l'adaptabilité face à l'augmentation du taux de brûlage et des anomalies de croissance induites par les changements climatiques.

Les modèles d'approvisionnement en bois utilisent le plus souvent un modèle de croissance et de rendement calibré à l'échelle du peuplement (tables de production) pour projeter l'évolution du volume marchand des strates d'aménagement. Puisque les stratégies d'aménagement écosystémique ont tendance à repousser l'âge d'exploitabilité, il est pertinent d'évaluer jusqu'à quel degré les tables de production actuelles peuvent être utilisées comme intrant dans un contexte d'aménagement écosystémique. Lorsqu'une table de production est évaluée relativement à un modèle de croissance calibré à l'échelle de l'arbre, nous montrons que bien que le modèle à l'échelle d'arbre semble moins biaisé, aucun modèle performe de manière adéquate pour prédire la croissance en volume marchand dans notre aire d'étude, particulièrement lorsque nous subdivisons les données par les attributs qui peuvent jouer un rôle pendant la mise en place d'AÉ. Pour les deux modèles, la source majeure d'erreur est liée à la densité du peuplement. Grâce à leur simplicité relative, nous avons préféré utiliser les tables de production pour élaborer nos modèles d'approvisionnement en bois.

La programmation linéaire standard a été utilisée pour tester les effets de quatre enjeux clefs sur le niveau d'approvisionnement en bois: (1) tendre vers une structure d'âge forestière établie à partir du régime naturel des feux et de la dynamique forestière, (2) agglomérer les blocs de récolte dans des chantiers de récolte afin de reproduire les patrons de perturbation naturelle à l'échelle du paysage, (3) maintenir les taux cumulés de coupe totale et de perturbation naturelle à l'intérieur du domaine historique de variabilité, et (4) exclure de la récolte les aires d'intérêt potentiel pour les peuples autochtones. Comparé à un scénario de *status quo*, l'inclusion des trois premiers enjeux résulte en une baisse de 3 à 22% de l'approvisionnement périodique et une période de restauration requérant que la coupe totale soit exclue sur 43 à 67% de la superficie productive pour les prochains 50 ans. Une validation des filtres bruts utilisés dans cette étude (les trois premiers enjeux) a été faite en utilisant les besoins en habitats du caribou des bois (*Rangifer tarandus caribou*). Pratiquement tous les scénarios induisaient un taux de perturbation susceptible de permettre le maintien du caribou des bois d'ici 25 ans.

Enfin, nous avons intégré le taux de brûlage et la sensibilité des tables de production au climat dans nos modèles d'approvisionnement afin de quantifier les incertitudes induites par le climat et les feux pour les deux stratégies d'aménagements. Les deux modèles suggèrent une réduction de l'approvisionnement périodique en bois entre 13 et 79%. Même si les indicateurs écologiques favorisent l'AE par rapport la normalisation des forêts, juste un changement de stratégie n'est pas suffisant faire face aux impacts du risque de feu et des changements climatiques en forêt boréale.

Mots clefs: Forêt boréale, aménagement écosystémique, croissance et rendement, faisabilité, adaptation, feux de la forêt, changements climatiques

Abstract

In the quest of implementing an ecosystem-based management (EBM) in a boreal forest in eastern Canada, we conducted a feasibility study focusing on ecological suitability, economic viability and social acceptability. Through timber supply models, we compared the outputs of EBM with a business as usual (BAU) management to determine former's robustness and adaptability to the increase in wildfire and growth anomalies induced by climate changes.

Timber supply analyses use yield models, most often at the stand-level to project harvestable volume over the planning horizon. Since EBM tend to delay harvesting age, the question may be raised on to what extent existing yield tables can be used with such strategies. When a yield table is rated against a tree-level model, we show that although the tree-level model is less biased, none of the models performed adequately to predict the volume growth of our study area, especially when subdividing the data by attributes that may have an important role while implementing EBM. For both models, the major source of error was related to stand density. Due to its relative simplicity, we chose stand-level yield tables to build our timber supply models.

We then carried out a feasibility study of implementing an EBM strategy in a boreal forest in eastern Canada. With standard linear programming, we tested four policy issues; age structure, harvest agglomeration; limit of cumulative disturbance, and land base of aboriginal interest. These issues were dealt with 3% – 22% reduction in periodic wood supply and a transition period of 50 years where clear-cut needs to be excluded in 43% – 67% of the productive area. Validation of the outputs through habitat requirement of woodland caribou (*Rangifer tarandus caribou*) as a fine filter showed that most of the scenarios should likely allow a self-sustaining caribou population within next 25-years.

Finally, we integrated climate sensitive fire burn rates and yield tables in the timber supply models to quantify the uncertainty induced by climate and fire under both management strategies. Both models responded with a reduction of periodic wood supply by 13% – 79%. Although ecological indicators are relatively better under EBM, merely switching the management strategy is not enough to address the impacts of fire and climate change in the boreal forests.

Key words: Boreal forest, ecosystem-based management, growth and yield, feasibility, adaptation, wildfire, climate change

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Preface

This thesis entitled "*Feasibility of an ecosystem-based management in an eastern Canadian boreal forest: testing for ecological suitability, economic viability, social acceptability and adaptability to wildfire and climate change*" comprises of five sections; General Introduction, Article 1 as Chapter I, Article 2 as Chapter II, Article 3 as Chapter III, and General Conclusion. All the papers and contexts presented in this thesis are my original contributions aimed at obtaining a Ph D in Forest Science at Université Laval, Québec, Canada. Articles I - III respectively correspond to the following (published, submitted or under preparation) articles:

- Dhital, Narayan, Frédéric Raulier, Pierre Bernier, Marie-Claude Lambert and Xiao-Jing Guo. Adequacy of a stand-level yield model for the ecosystem-based management of boreal black spruce forests (submitted to *The Forestry Chronicle* in March 2013).

- Dhital, Narayan; Frédéric Raulier; Hugo Asselin; Louis Imbeau; Osvaldo Valeria; Yves Bergeron. Emulating boreal forest disturbance dynamics: can we maintain timber supply, aboriginal land use, and woodland caribou habitat? (Published in 2013 in *The Forestry Chronicle* 89:54 - 65).

- Dhital, Narayan; Frédéric Raulier; Pierre Y. Bernier; Marie-Pierre Lapointe-Garant; Yves Bergeron. Adaptability of an ecosystem-based management strategy to climate induced increase in fire burn rate and growth anomalies in an eastern Canadian boreal forest (Manuscript under preparation).

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In addition to the papers mentioned above, a research note entitled *"Timber supply analyses under conventional and ecosystem-based management scenarios in an eastern boreal forest"* (SFN Network Research Note Series No. 75) was also published in 2010 by Sustainable Forest Management Network as part of this study. I authored that note with significant contribution from Dr. Raulier.

General Introduction

The best way to sustain timber supply and other ecological services that we enjoy today from forests is by maintaining viable ecosystems (Jetté et al. 2009). Ecosystems within their historic range of natural variability are likely to be viable in terms of the level of biodiversity at different spatial scales and ecological processes such as hydrological, carbon and nutrient cycling as well as climate regulation (Haeussler and Kneeshaw 2003).

Historically, human have been altering the nature, extent and timing of natural disturbances through logging and other silvicultural activities to produce desired products and services from the forests (Bettinger et al. 2009; Haeussler and Kneeshaw 2003; Davis et al. 2001). Forest management activities focus on reducing the variability and unpredictability of forest ecosystem at all spatial scales and on maximizing the timber production by liquidating the forests beyond the rotation age (Seymour and Hunter 1999).

Clear cutting has been viewed as an equivalent for stand replacing disturbances in boreal forests of North America in that it has similar ecological consequences (Haeussler and Kneeshaw 2003). This resemblance has been used to justify large scale clear cutting in those forests (Bergeron et al. 2002). Although North American boreal forests are believed to be adapted to stand replacing fire (Haeussler and Kneeshaw 2003; Burton et al. 1999), the role of fire has been overemphasized and the actual situation is considerably more complex (Bergeron et al. 2002; Bergeron et al. 1998). Small scale disturbances caused by low severity fire, insect defoliation, wind throw and diseases play a significant role in shaping boreal forests (Gauthier et al. 2009; Bergeron et al. 2006; Bergeron et al. 2002). Considerable differences between the ecological attributes (e. g., age structure, species composition, variability in fire cycle etc.) created by stand replacing fire and large scale industrial harvest have been widely documented in the past decades (e. g., Kimmins 2004; Haeussler and Kneeshaw 2003; Bergeron et al. 2002; Seymour and Hunter 1999; Burton et al. 1999; Bergeron et al. 1998). All these factors determine the natural range of ecosystem variability. This variability should be maintained to ensure the conservation of biodiversity (Landres et al. 1999; Bergeron et al. 1998). With the failure of traditional sustained yield timber production regime to maintain this natural range of variability, a quest for its alternatives started in late 1980's and early 1990's (Seymour and Hunter 1999). Ecosystem-based management (EBM) is suggested to provide such an avenue through reducing the ecological differences between natural and managed ecosystems (Gauthier et al. 2004; Harvey et al. 2002; Bergeron et al. 2002; Seymour and Hunter 1999; Burton et al. 1999).

EBM is a forest management template guided by natural disturbances (Gauthier et al. 2009). It aims to maintain the biological diversity and sustain the ecological process at all spatial scales by limiting the management related activities within the historical range of variability of the natural disturbances (Gauthier et al. 2009; Bettinger et al. 2009; Davis et al. 2001).

Increasing interest on EBM of boreal forest has been demonstrated by population, managers and scientific community alike over the past decades (Perera et al. 2004; Bergeron et al. 2002). Despite such interest, few studies go beyond theoretical principles and provide implementation guidelines for the EBM (e. g., Gauthier et al. 2009; Belleau et al. 2007; Gauthier et al. 2004; Bergeron et al. 2002; Burton et al. 1999; Seymour and Hunter 1999; Bergeron et al. 1999). One such guideline is to manage the boreal forest under varying rotation ages based on the understanding of the natural disturbance pattern (Seymour and Hunter 1999; Burton et al. 1999). However, longevity of the most species found in boreal forest is not enough to have the commercial rotation of 200 or 300 years as suggested by these authors without reducing the timber flow from the forest significantly. Using silvicultural treatments that are effective in maintaining habitat diversity without reducing the periodic timber supply significantly is more feasible in boreal forests where historical fire cycle, which corresponds to the time required to burn an area equivalent to the study area, is considerably shorter than the current one (Bergeron et al. 2010) and longevity of tree species is shorter than the mean fire cycle (Bergeron et al. 2002). Based on the information on mean fire cycle and maximum harvestable age of the species present in the forest, this model divides the forest landscape into three distinct development types (or cohorts) of roughly 100 years and different silvicultural treatments are used to recruit different cohorts (Nguyn-Xuan 2002). This is particularly suitable in eastern Canadian boreal forest where past and current burn rates are significantly different (Bergeron et al. 2004). If fire burn rate approaches the upper boundary of its historical variability, space for managers to emulate fire by harvesting is limited and harvesting is unsustainable where burn rate and logging begin to compete (Bergeron et al. 2010).

Feasibility of a forest management regime depends on its economic viability, ecological suitability and social acceptability (Weber and Stocks 1998). Although a template of EBM based on multiple cohort management has been suggested (Gauthier et al. 2004) and size and distribution of different cohorts over the landscape has been provided (e. g., Belleau et al. 2007), sustainability aspect of this regime has not been studied sufficiently. Therefore the overall goal of this study is to determine the feasibility of an EBM in a boreal forest in the context of climate change.

Availability of merchantable volume in a sustained basis is central to any forest management regime (Bettinger et al. 2009; Fortin and DeBlois 2010). EBM is a compromise between natural forest condition and sustained timber production (Kimmins 2004; Seymour and Hunter 1999). Empirical growth and yield models are used to project periodic timber supply. Existing growth and yield models were developed with the objective of maximizing volume production. Therefore, their capability of predicting the volume increment beyond the rotation age is not clear. Most of them are built on data collected from pure stands (e. g., Pothier and Savard 1998). Moreover, these models assume that site productivity is constant over a rotation. Although forest mosaic under EBM is expected to be increasingly complex, analogous to natural stand, and assumption of constant site productivity is questionable particularly under future climate scenarios (Lapointe-Garant et al. 2010; Anyomi et al. 2012), the adequacy of the existing growth and yield models to address issues specific to EBM have not been tested and models developed considering EBM as a management strategy are almost non-existent (Saucier and Groot 2009).

Weather elements associated with climate are always changing due to changes in orbital parameters of our planet, output of solar energy and composition of atmosphere (Zhang et al. 2011; Flannigan et al. 2005). Rate of change of such elements has been accelerated recently due to increase in the level of greenhouse gases in the atmosphere resulted from the anthropogenic activities (Zhang et al. 2011; Meehl et al. 2007; Flannigan et al. 2005; Zhang et al. 2000). However, there are large regional and seasonal differences in the magnitude and direction of the effects currently underway. North America saw an increase in average annual temperature between 0.4^0C and 1.2^0C in the last five decades with larger increase in winter temperature (Zhang et al. 2011; Yagouti et al. 2006). There has been a significant increase in precipitation between 1900 and 2005 (IPCC 2007). Average additional warming of about 3.5^0C but up to 5^0C in winter with 10% – 25% increase in precipitation by the end of this century has been projected for this region (Meehl et al. 2007; Plummer et al. 2006; Hulme and Shread 1999). Forest ecosystems respond to change in temperature, precipitation and concentration of CO_2 in the atmosphere directly through changes in growth patterns, species distribution and migration, and indirectly through changes in natural disturbance dynamics such as fire regime, insect epidemics and diseases (Bergeron et al. 2010; Girardin et al. 2008; Kirilenko and Sedjo 2007; Soja et al. 2007; Bergeron et al. 2004).

Depending on the region and species considered, positive as well as negative response to climate change by forest growth pattern has been documented. In ecology, forest productivity is measured in terms of net primary productivity (NPP) (Rustad et al. 2012). Longer growing season, warmer temperature, elevated CO2 concentration in atmosphere and increased precipitation are expected to increase the NPP and expansion of forest coverage northward in high latitude areas such as Canada (Kirilenko and Sedjo 2007; Soja et al. 2007). However, several studies report recent decrease in boreal forest productivity due to asymmetry in increase of temperature and available soil moisture. Bunn et al. (2007) reported increase in productivity until late nineties and wide spread decline recently due to moisture constraint in high latitude areas. Inter-annual growth variation of trembling aspen (*Populus tremuloides* Michx.) in western Canada is dependent on moisture availability, insect defoliation (Hogg et al. 2005) and timing of spring thaw (Bergeron et al. 2007; Kimbal 2000) suggesting that productivity would be reduced significantly if climate leads to dryer condition. Lapointe-Garant et al. (2010) reported increase in growth of trembling aspen in high latitude area of north eastern boreal forest of Canada. Response of growth on climate change is explained mainly by stand species composition and age structure in boreal region of North America. Girardin et al. (2012) suggested that yo*ung*er stands in general and older black spruce (*Picea mariana* (Mill.) B. S. P.) stands should benefit from climate change due to increased growth provided there is no moisture stress. Several species found in northern latitude, however, experienced periods of lower productivity in the past century. Some examples include decline of red spruce (*Picea rubens*) growth after 1960 due to winter injury attributed to rapid freezing and ash (*Fraxinus americana*) dieback after 1920 attributed to climate factors particularly drought and freezing damage (Mohan et al. 2009). Although hardwood species are projected to increase the productivity under projected climate scenarios, spruce-fir forests are projected to do otherwise (Rustad et al. 2012).

It has been reported that fire burn rate in eastern Canadian boreal forest has been decreasing since 1850 from 0.69% to 0.22% per year (Bergeron et al. 2010; Bergeron et al. 2004). However, such rate is projected to increase in the region by almost two times and three times under 2^*CO_2 and 3^*CO_2 climate forcing respectively by the end of this century (Bergeron et al. 2010; Flannigan et al. 2005). Changes in fire burn rates may have important implications in forest and forestry activities and boreal forest carbon budgets (Flannigan et al. 2005). Increased fire burn rates produce younger forest stands and younger forests produce less harvestable timber (IPCC 2007). Therefore, increased fire burn rates will have ramifications in wood supply.

Cumulative impact of climate and fire over this century could increase the productivity in north eastern North American boreal forest by shifting forest distribution to younger age classes and expanding the range of jack pine (*Pinus banksiana* Lamb.) (Girardin et al. 2012). Jack pine is a favorable species that can colonize after fire and its growth is expected to be enhanced under projected climate scenarios (Girardin et al. 2012). For the forestry sector in general, indirect impact of climate change through modification of fire regime is more important than direct impact on growth through changes in precipitation, temperature and CO2 concentration (Kirilenko and Sedjo 2007). This interaction of climate change and change in burn rate may easily overshadow the direct impact of climate change on plant growth, species distribution and migration (Flannigan et al. 2005).

Although natural variability of a forest ecosystem is determined by the frequency and severity of disturbance (e. g., fire which is determined by the climate) and there exists a strong interaction between disturbance regime and successional pathways that follow, these two important aspects of landscape ecology are still to be incorporated in forest management planning (Coulombe Commission 2004). Concentration of forest management activities at stand scale leads to this situation (Monserud 2003). Moreover, tools are not well developed to take into account uncertainty generated by climate change in forest management planning which directly influences the forest growth and indirectly influences the fire regime (Girardin et al. 2008).

Given this context, we formulated a research project for the partial fulfillment of my doctoral degree where we carried out a feasibility study of implementing ecosystem-based management (EBM) in an eastern Canadian boreal forest by testing its ecological suitability, economic viability, social acceptability and its adaptability to climate induced increase in fire burn rate and growth anomalies over this century. The study was divided into three sections, each section being a chapter or an article in this thesis. In the first section, we tested the adequacy of using standard yield tables in black spruce dominated boreal forests while implementing an EBM against indicators used for ecosystem-based management planning. This study was followed by a feasibility study of an EBM strategy in an eastern Canadian boreal forest against economic, social and environmental criteria represented respectively by the level of periodic timber supply, aboriginal land use potential and the persistence of woodland caribou habitat. In the last section, we quantified the likely impacts of climate induced increase in fire burn rate and growth anomalies in long term timber supply under EBM. Adaptability of EBM to such changes was also tested against ecological indicators. Finally, the outputs of the EBM scenarios were compared with a business as usual (BAU) scenario to determine the robustness of EBM in the changing context.

This dissertation has been developed in the form of a collection of three articles already published, submitted or in the process of submission for publication in peer reviewed scientific journals. Therefore, there are some repetitions in the texts.

Chapter 1

Adequacy of a stand-level yield model for the ecosystem-based management of boreal black spruce (*Picea mariana* (Mill.) B.S.P.) forests

Narayan Dhital, Frédéric Raulier, Pierre Y. Bernier, Marie-Claude Lambert, and Xiao-Jing Guo

A version of this chapter is submitted to *The Forestry Chronicle* (March 2013)

1.01 Résumé

L'analyse d'approvisionnement en bois utilise de l'information concernant le rendement estimé à partir de modèles de croissance et de rendement à l'échelle de l'arbre ou du peuplement. Les tables de production actuelles ont été élaborées pour des peuplements issus de coupe à blanc et dont l'âge avant coupe était inférieur à l'âge de rotation. Puisque les stratégies d'aménagement écosystémique ont tendance à repousser l'âge d'exploitabilité, il est pertinent d'évaluer jusqu'à quel degré les tables de production actuelles peuvent être utilisées comme intrant dans un contexte d'aménagement écosystémique. Dans cette étude, nous avons quantifié l'erreur de prédiction associée à l'accroissement net en volume du peuplement provenant d'une part d'un modèle d'accroissement à l'échelle de l'arbre et d'autre part d'un modèle à l'échelle de peuplement. Le but de cet exercice était de quantifier l'erreur de prédiction et le biais associé à chaque modèle dans des peuplements purs d'épinette noire (*Picea mariana* (Mill.) B.S.P.) situés dans la forêt boréale continue du Québec, Canada. Nous avons testé deux hypothèses: 1) un modèle de croissance à l'échelle de l'arbre a des meilleures capacités prédictives qu'un modèle à l'échelle du peuplement en termes de croissance et de production forestière; et 2) la corrélation entre, d'une part, les erreurs de prédiction de l'accroissement en volume et, d'autre part, des indices structuraux du peuplement est plus élevée pour un modèle à l'échelle du peuplement que pour un modèle à l'échelle de l'arbre. Nous avons démontré que la performance des deux modèle n'était pas adéquate à travers la région d'étude et qu'aucun de ces deux modèles n'avait une performance satisfaisante. Pour chacun des deux modèles, la source d'erreur la plus importante était étroitement liée au volume perdu par mortalité, un phénomène en partie relié à la sénescence des peuplements. Puisque la sénescence n'a été évaluée dans aucun des deux modèles, notre étude met en évidence que la modélisation de la sénescence dans ce type de modèle est nécessaire si l'on souhaite améliorer les prédictions de ce type de modèle.

Mots clefs: modèles de croissance et de rendement, accroissement en volume marchand, mortalité, l'aménagement écosystémique, épinette noire, forêt boréale

1.02 Abstract

Timber supply analyses use yield information derived from tree- or stand-level growth and yield models. Existing yield tables were designed to be used on stands created through the clear-cutting of forest stands older than rotation age. Since ecosystem-based management strategies tend to delay harvesting age, the question may be raised on to what extent existing yield tables can be used with such strategies. We quantified the prediction error of stand net volume increment for one stand- and one tree-level yield model to compare the magnitude of prediction error and model bias in pure stands of black spruce (*Picea mariana* (Mill.) B.S.P.) across the continuous boreal forest of Québec, Canada. Two hypotheses were tested: "an individual tree growth model has a better predictive capability of forest growth and yield than a stand-level model" and "correlation between errors in the prediction of volume increment and stand structural indices are higher for a stand-level model than for a tree-level model". We demonstrated that performance of both models was inadequate across the study area and that neither model performed satisfactorily. For both models, the most important source of error was linked to the volume loss due to mortality, in part related to stand senescence. Since senescence was not accounted for by either model, further research needs to be concentrated on improving these models to account for it.

Key words: growth and yield model, merchantable volume increment, mortality, ecosystem-based management, black spruce, boreal forest

1.03 Introduction

Forest management aims at producing a certain quantity of wood from a given area on a sustained basis. To achieve this goal, the age structure of the forest is manipulated. Mainly two contrasting families of desired structures exist; first, the “regulated structure” which aims to maximize the commodity products through harvest, and second, the “structure guided by natural disturbances” that aims to sustain the ecological processes by limiting management activities within the historical range of natural variability (Bettinger *et al.* 2009; page 207). Forest management inspired by the latter concept is called ecosystem-based (Gauthier *et al.* 2009) or natural disturbance-based management (Harvey *et al.* 2002). One clear distinction between a regulated structure and a structure guided by natural disturbance is that the former eliminates all the stands older than rotation age whereas the latter focuses on maintaining at least a representative portion of older stands (Gauthier *et al.* 2009; Haeussler and Kneeshaw 2003; Bergeron *et al.* 2002; Bergeron *et al.* 1998).

Public and scientific interest in ecosystem-based management (EBM) is growing in Canada (Perera and Buse 2004), with a number of EBM approaches being proposed or tested. One such example is the multi-cohort management approach designed for eastern Canadian boreal forests which attempts to emulate the landscape-level age class structure generated by fires through a compromise between rotation age and expected longevity of trees (Bergeron *et al.* 2002). As a general rule, EBM represents a tradeoff between harvest maximization and reproduction of an age class distribution closer to that of forests submitted under a natural disturbance regime (Seymour and Hunter 1999; Haeussler and Kneeshaw 2003).

Timber supply analyses use growth and yield information derived from empirical tree or stand-level models to project future harvestable volume. Stand-level yield tables fit more readily into timber supply analyses than do tree-level growth and mortality models whose development and use are generally more resource-intensive and whose predictions are more prone to error compounding (Weiskittel *et al.* 2011, page 69). Tree-level models, however, usually do not need age as input, except when estimating site index (Burkhart and Tomé 2012). Such a feature provides the ability to predict the growth and the probability of mortality of trees older than the rotation age.

In either stand or tree resolution, most growth and yield models are designed for mono-specific, even-aged stands (Porté and Batterlink 2002) created through complete stand-replacing disturbances such as clear-cutting (Bergeron *et al.* 2002). However, in EBM, different silvicultural tools are needed to emulate other patterns of disturbance such as individual tree mortality and gap dynamics. Clear-cutting should be used only to emulate that portion of the landscape where stand replacement would naturally occur (Bergeron, 2004). The rest of that landscape would, in this case, be treated with partial harvesting to emulate old growth dynamics. As ecosystem-based management envisages modifying the forest age structure to mimic the age class distribution of a natural forest, a given proportion of forest stands need to be older than the rotation age. The question then arises as to what extent the tools such as yield tables designed for regulating forests and predict the growth and yield up to the rotation age can be used in the context of ecosystem-based management strategies.

The objective of this study was to determine the adequacy of certain existing stand-level and tree-level growth and yield models for applications to EBM. We quantified the prediction error (observed minus predicted values) of stand net volume increment of pure stands for a stand-level and a tree-level yield model. The performance of both models was then assessed against attributes potentially used under EBM such as homogeneous forest types, potential vegetation, cohort types, stand density and stand structure (Grondin *et al.* 2007; Gauthier *et al.* 2004; Boucher *et al.* 2003). We then used these results to test two hypotheses. The first was that an individual tree growth model has a better predictive capability of forest growth and yield than a stand-level model. Our second hypothesis was that correlation between errors in the prediction of volume increment and stand structural indices would be higher for a stand-level model than for a tree-level model. The rationale behind the first hypothesis is that more information is used to calibrate a tree-level model relative to a stand-level model. The rationale behind the second hypothesis is that a tree-level model implicitly accounts for stand structural changes as it uses the information at the tree level.

1.04 Materials and Methods

1.04.01 Study area

We selected the continuous boreal forest zone of Quebec as study area (Figure 1.1). Grondin *et al.* (2007) classified this area into five homogenous forest units based on groups of species and dominant vegetation as descriptive variables and biophysical environment (e.g. climate, drainage), natural disturbances and human disturbances (e.g., harvesting) as explicative variables (Figure 1.1). The western softwood unit is dominated by the black spruce - moss vegetation where black spruce (*Picea mariana* (Mill.) BSP) grows habitually as pure monospecific stands. The eastern softwood unit witnesses more precipitation, has longer fire cycle and black spruce is predominantly associated to balsam fir (*Abies balsamea* (L.) Mill.) (De Grandpré *et al.* 2000). Pure black spruce stands are regularly found in early stages of development and on poorer sites in the eastern and central softwood units (Boucher *et al.* 2006).

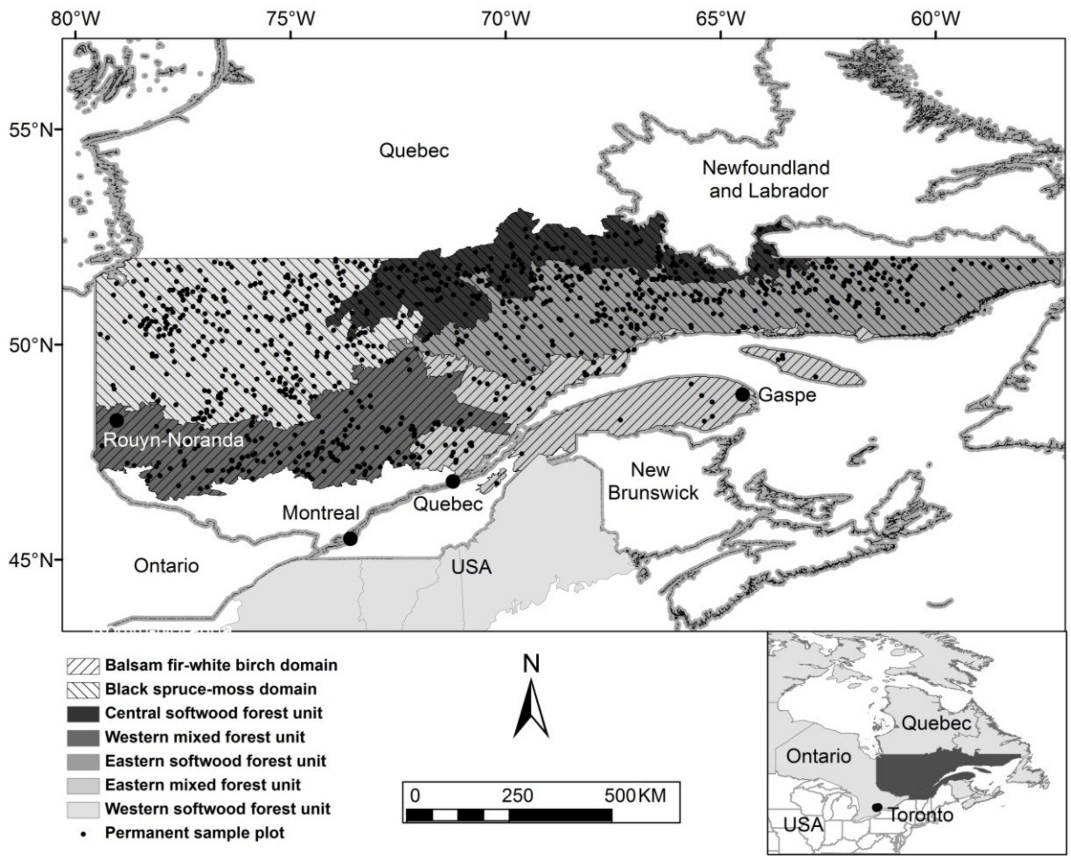

Figure 1.1 Study Areas. Polygons in different tones correspond to homogeneous forest units as classified by Grondin et al. (2007)

Mixed forests are dominated by balsam fir and white spruce (*Picea glauca* (Moench) Voss) stands. Major companion species are white birch (*Betula papyrifera* Marsh.), black spruce, jack pine (*Pinus banksiana* Lamb.), larch (*Larix laricina* (Du Roi) K. Kocl) and trembling aspen (*Populus tremuloides* Michx.). The key driver of forest dynamics in the mixed forest zone is spruce budworm (*Choristoneura fumiferana* (Clemens)) due to the heavier presence of balsam fir, but fire also plays an important role (Saucier *et al.* 2009). The abundance of hardwoods or mixed stands with intolerant hardwoods in the west is explained by lower precipitation and shorter fire cycle (Bergeron and Dansereau 1993). Pure black spruce stands are rare and limited either to xeric or hydric sites or to highlands (Saucier *et al.* 2009).

1.04.02 Data

The Ministère des Ressources naturelles du Québec maintains a network of about 5700 permanent sample plots (PSPs) in the continuous boreal zone of Quebec for which inventory data with at least two consecutive measurements are available. The inventory consists of numbering and measuring the diameter at breast height (DBH) of each tree with a DBH greater than 9 cm within 400 m^2 circular plots. Height (to the nearest 10 cm) and age are measured for a selected number of dominant or co-dominant trees. A status code indicating whether a tree is dead or alive during measurement is also recorded. Moreover, the number of saplings (having DBH between 1 cm and 9 cm) is counted for each species within a 40 m^2 sub-plot located in the center of the plot and grouped in 2-cm diameter classes. Re-measurement interval ranges between 4 and 23 years with a mean of 11 years.

We used these repeatedly measured PSP data to estimate the observed stand net volume increment and calculate its error of prediction for a stand-level yield model (Pothier and Savard 1998) and a tree level model (Teck and Hilt 1990; 1991). Stand net volume increment was selected as the variable to rate the performance of one model against another.

The plots were partitioned into cohort types, types of stand structure, potential vegetation types, stand density classes and homogenous forest units. A cohort corresponds to a developmental stage reflecting stand composition as well as their vertical and horizontal structures. Cohort type was assigned with the classification tree provided by Nguyen-Xuan (2002), determining that trees with DBH 2-6 cm, 7-14 cm and more than 14 cm form the lower, middle and upper stories (Nguyen-Xuan 2002). The type of stand structure was determined for each plot to be regular, irregular or uneven-aged as a function of its diameter distribution using the model of Boucher *et al.* (2003). Stated briefly, this model considers five structural indices: Shannon evenness index (Shannon 1948), coefficient of asymmetry (skewness), coefficient of variation and stem densities with a DBH larger than 10 cm or 14 cm (expressed as a percentage relative to the merchantable stem density). These indices serve to estimate a score that is used to assign a stand structural type. Potential vegetation is the vegetation of a reference point in future (climax of a succession pathway, MRN 2003b) as opposed to the actual vegetation (Henderson *et al.* 2011). It is predicted as a function of indicator species groups, actual vegetation, regeneration and physiographic variables (MRN 2003b). The potential vegetation describes the successional relationship between vegetation and its environment and answers the questions regarding the growth potential of a particular species group in a given environment (Henderson *et al.* 2011). We followed the definition of density classes by Pothier and Savard (1998, their Table 10) (dense, intermediate or sparse) as a function of site and relative density indices. When required, the strength of association between ordinal variables was estimated with the gamma measure of association (Γ) (PROC LOGISTIC, SAS Institute, Cary, NC).

1.04.03 Growth and yield models

To single out the effect of modeling resolution during the model comparison (and not that of including different explanatory variables – such as crown length or the spatial distribution of trees – e.g. Mailly *et al.* 2003), we chose two models that use similar input variables (diameter, height, stem density and site quality): the Pothier and Savard (1998) stand-level yield model and the NE-TWIGS (Teck and Hilt 1990; 1991) tree-level growth model. These models are typical examples and both have also been widely used and tested. For the past 14 years, the Pothier and Savard (1998) yield model was the official tool used by the province of Quebec for timber supply analyses. The NE-TWIGS is a basic component of the Forest Vegetation Simulator for Northeastern US (Crookston and Dixon 2005) and has also been used or tested in eastern Canada (Payandeh and Huynh 1991; Lacerte *et al.* 2006). Other individual tree-level models (e.g. Fortin and Langevin 2009) could have been used but NE-TWIGS has a limited number of parameters, making its calibration much simple (Teck and Hilt 1990).

Table 1.1 Systems of equations (Eqs. 1.1 to 1.4) of the stand-level growth and yield model (Pothier and Savard 1998). Equations 1.5 to 1.7 serve to estimate SI and ρ_{100}

(1.1)	$H_d = b_{10} + b_{11}\, SI^{b_{12}} \left(1 - \exp(-b_{13} A_c)\right)^{b_{14} SI^{-b_{15}}}$
(1.2)	$\overline{D}_q = b_{21}\, b_{22}^{H_d} A_c^{b_{23}} \rho_r^{b_{24}}$
(1.3)	$G = b_{31} H_d^{b_{32}} b_{33}^{H_d} A_c^{b_{34}} \rho_r^{b_{35}} \exp\left(\frac{b_{36}}{A_c}\right)$
(1.4)	$V = b_{41} H_d^{b_{42}} G^{b_{43}} \overline{D}_q^{b_{44}}$
(1.5)	$SI = b_1 H_d^{b_2} \left(1 - \exp(-b_3 A_c)\right)^{b_4 H_d^{-b_5}}$
(1.6)	$\rho_r = \frac{N_0}{\left[\frac{\overline{D}_q}{10^{b_1}}\right]^{-1/b_2}}$
(1.7)	$\rho_{100} = \frac{\left(\ln(\rho_r) A_c - b_{10}\right)}{b_{11}}$

H_d: dominant height (m) of trees in a plot, SI: Site index (m), A_c: Corrected age (years), $\overline{D}_q$: mean quadratic diameter (cm), ρ_r: relative density index, G: basal area (m^2 ha^{-1}), V: Volume (m^3 ha^{-1}), N_0: number of stems (per hectare), ρ_{100}: relative density index at the age of 100 years, b_is are parameters to be estimated.

The stand-level model is composed of four interrelated equations that estimate dominant height, mean quadratic diameter, basal area and merchantable volume of stands (equations 1.1 to 1.4, respectively, Table 1.1). The input variables for this system of equations are mean age of plot dominant/co-dominant trees, site index (*SI*) and relative density index ρ_{100}.

SI (Equation 1.5, Table 1.1) is calculated as a function of stand dominant height and stand age and referenced at the age of 50 years. ρ_{100} is calculated as the ratio of observed to maximum stem densities referenced at the age of 100 years (Equations 1.6 and 1.7, Table 1.1). Maximum stem density is related to the self-thinning line (Drew and Flewelling 1979) and is consequently a function of the stand mean quadratic diameter. Stand age corresponds to the mean age of sampled canopy trees in the stand corrected at the height of 1 m from base to avoid the variation introduced by potential abnormal growth of sample trees at their sapling stage. For this study, we used the parameter values provided by Pothier and Savard (1998). These values were derived from a large temporary sample plot dataset whose geographical coverage included our study area.

The NE-TWIGS model is defined by two equations that project the basal area increment (Equation 1.1; Table 1.2) and the survival probability (Equations 1.2; Table 1.2) of individual trees. For calibration purposes, the survival probability was estimated and modeled by dividing the tree population into definite cells of similar values of site index, competition index and DBH. We had a total of 125 cells with varying number of trees (3 – 1661) in each cell. The tree-level model was calibrated with 2006 black spruce dominated PSPs. Dominant species contained at least 50% of the merchantable basal area. This data covered a wide range of ages, site classes and growing stocks. Seventeen species were found associated with black spruce including balsam fir, jack pine, white birch, trembling aspen, and sugar maple (*Acer saccharum* Marsh.). PROC NLMIXED (for equation of basal area increment) and PROC MODEL (for the equation of survival probability) (SAS Institute, Cary, NC) were used to estimate the parameters (Table 1.3). These two equations are used to project the tree list of a plot from one measurement date to another. The volume table of Perron (2003), also used by the stand-level yield table of Pothier and Savard (1998), was used to estimate standing living and dead volumes and consequently stand volume increments.

To estimate the prediction error of net volume increment for both models, only pure black spruce stands were considered as Pothier and Savard (1998) model was originally calibrated with pure stands from temporary sample plots. We selected plots with at least 75% of total merchantable basal area in black spruce, defined as a pure black spruce stand (Pothier and Savard 1998). There were 1311 such plots.

Table 1.2 Equations of the individual tree-level growth model (Teck and Hilt 1990, 1991)

(1.1) $i_{g(t+1),t} = b_1\, SI\, (1 - \exp(-b_2\, d_{1.3}))\, \exp(-b_3\, G_L)$

(1.2) $s = 1 - (1 + \exp(n))^{-1}$

with $n = b_1 + b_2 (d_{1.3} + 1)^{b_3} \exp(-b_4\, d_{1.3} - b_5 G_L + b_6 SI)$

i_g: basal area increment ($cm^2\ y^{-1}$) of subject tree between time t and t+1, SI: site index (m), $d_{1.3}$: diameter (cm) at breast height (1.3 m) of subject tree, G_L: Basal area of trees larger than subject tree, s: annual survival probability of a subject tree, b_i are parameters to be estimated.

Table 1.3 Measures of fit and performance of equation 1.1 and 1.2 of the table 1.2. Multiple correlation coefficient and degree of freedom for the equation 1.1 and 1.2 of table 1.2 are 0.92, 49000 and 0.26, 78283 respectively.

Equation (Table 1.2)	Parameter	Estimates	Standard Error
1.1	b_1	0.30	7.47E-03
	b_2	0.06	2.88E-03
	b_3	0.02	5.27E-04
1.2	b_1	0	0
	b_2	2.30	0.03
	b_3	0.39	5.98E-03
	b_4	0.02	3.56E-04
	b_5	7.95E-04	3.90E-05
	b_6	-1.98E-03	1.32E-04

1.04.04 Assessment of model performances

(i) Model precision and biases

Performance of stand- and tree-level models was assessed by regressing observed vs. predicted net volume increments. Annual gross volume increment is the difference between the volume of living trees (i.e. of trees surviving between two measurements and of trees newly exceeding the merchantable DBH threshold) between two consecutive plot measurements divided by the measurement interval. Annual net volume increment is the difference between annual gross volume increment and annual volume loss due to mortality. We chose the coefficient of determination (R^2) to measure the proportion of variance explained by the model (Nagelkerke 1991). The significance of differences between the coefficients of determination of the both models was also tested (Draper and Smith 1998). Since an unbiased model has an intercept and slope not significantly different from zero and unity respectively, we carried out simultaneous F-tests (equation 1.1) using PROC REG (SAS Institute, Cary, NC) to assess the bias of both models;

eq. 1.1 $$F_{2,n-2} = \frac{\mathrm{n}b_0^2+2\sum\hat{y}_i b_0(b_1-1)+\sum\hat{y}_i^2(b_1-1)^2}{2\sum(y_i-\tilde{y}_i)/(n-2)}$$

where y_i and $\hat{y}_i$ are observed and predicted values of the model and $\tilde{y}_i$ is the predicted value from $y_i = b_0 + b_1\hat{y}_i$. We chose the root mean square error (RMSE) to assess the deviance of predicted values from the observed values (Mayer and Butler 1993). To assess the model behavior across the classes of attributes (i. e., cohort, stand density, stand structure, potential vegetation and homogenous forest units), we split the data into the respective sub-classes and treated each separately.

Before carrying out further analyses of the model residuals, a robust scale statistic was calculated to detect the potential outliers in the residuals (Rousseeuw and Hubert 2011):

eq.1.2 $$\mathrm{z} = \frac{\mathrm{x} - \mathrm{median}\ (\mathrm{x})}{\mathrm{mad}(\mathrm{x})}$$

where z is a robust scale statistics (z-score), x is a vector of residuals and “mad” is median of all absolute deviations from median. SAS PROC IML (SAS 9.3, SAS, CARY, NC, USA) was used to calculate the “mad” function and multiplied by a constant (1.483) to have a robust estimation of the standard deviation (Rousseeuw and Hubert 2011). A cutoff value of 3 was fixed and the observations having a value higher than the cutoff value were not considered for fitting the regression.

We also assessed the performance of both models with the prediction errors of stand net volume increment. We calculated the mean bias of the residuals ($\bar{E}$) as given by Reynolds (1984). An unbiased model should have a mean bias not significantly different from zero. We calculated the first and the second critical errors (e* and e**) from Reynolds (1984) for both models. The e* is the smallest value of $\bar{E}$ to conclude the acceptance of the null hypothesis that mean bias is not significantly different from zero. The model with smaller critical errors (e*) is considered to be better (Reynolds 1984; Rauscher 1986). The second critical error (e**) corresponds to the smallest value of $\bar{E}$ beyond which a null hypothesis that $\bar{E}$ is not significantly different from zero is rejected. Since e* is smaller than e**, the difference between these two quantities is equivalent to a confidence bound for the upper (1 – α) quantile of the distribution of the mean bias (Reynolds 1984).

Users of both models may be interested not only in the model meeting certain standards such as R^2 and RMSE but also on the magnitude of the expected error when they use one of the models to predict volume increments (Reynolds 1984). To provide the range of model accuracy, we further calculated confidence, prediction and tolerance intervals of the mean bias ($\bar{E}$). A confidence interval (CI) is the indication of uncertainty based on the current sample (Rauscher 1986):

eq. 1.3 $$\mathrm{CI} = \bar{\mathrm{E}} \pm St_{(1-\frac{\alpha}{2}, n-1)} / \sqrt{n}$$

where S is standard deviation of the residuals and ($t_{1-\alpha/2}$) is the (1 – α/2) quantile of the t distribution with (n – 1) degrees of freedom. Since a confidence interval indicates the uncertainty based on the current sample, the uncertainty related to a future prediction may be given by the prediction interval. The prediction interval gives the probability that k number of future values of E_i will fall within its range. A (1 – α) % prediction interval (PI) to contain the value of E_i is given by (Reynolds 1984);

eq. 1.4 $$\mathrm{PI} = \bar{\mathrm{E}} \pm \sqrt{(\frac{1}{k} + \frac{1}{n})} St_{(1-\frac{\alpha}{2}, n-1)}$$

For providing the limits to contain a specified proportion of the distribution of E_i, a tolerance limit needs to be constructed (Reynolds 1984). If a user wants to contain the 95% of the distribution of E_i with a 95% confidence; a tolerance interval (TI) will be of the form;

eq. 1.5 $$\mathrm{TI} = \bar{\mathrm{E}} \pm k_{(1-\gamma, n, 1-\alpha)} S$$

where $\alpha = \gamma = 0.05$ and value of the k-factor with n degrees of freedom is given in Table 2.1 of Eisenhart *et al.* (1947). These intervals are sensitive when data is not normal, particularly in cases where sample size is small (Rauscher 1986). Our study had 1311 sample plots which provide assurance over the issue of non-normality.

(ii) Sources of error

Correlation analyses (Spearman rank correlation) were performed between model residuals and stand structural indices (Shannon evenness index, coefficient of asymmetry, coefficient of variation) as well as stand attributes such as site index, mean quadratic diameter, relative density index and initial merchantable volume to identify the sources of error in the model. Relative density and site indices are basic inputs of the stand-level model and are considered constant through time (Pothier and Savard 1998). An important source of volume growth prediction error could thus be related to the derivation error of these indices (Pothier *et al.* 2004). Therefore, change in SI and ρ_{100} between measurement intervals was calculated and correlation analyses between these variables and model residuals was performed. To ease the interpretation of the results, the following scale was used to rate correlations (r_P and r_S for Pearson and Spearman correlations, respectively): $r < 0.33$ (weak), $0.33 \leq r < 0.66$ (moderate), $r \geq 0.66$ (strong).

Correlation analyses were also performed with the square of residuals to identify sources of variability for both models. An error model (Gregoire and Dyer 1989) was built with the variables most correlated to the square of residuals:

eq. 1.6 $$e^2 = \nu \, \Pi_{i=1}^{n} X_i^{\omega_i}$$

where e^2 is the squared residuals of predicted net volume increments for one model, ν and ω_i are parameters to be estimated (PROC NLIN, SAS Institute, Cary, NC). Significance of inclusion of a variable X_i into eq. 1.6 (i.e., $H_0 : \omega_i \neq 0$) was tested with a likelihood ratio test for nested models (Bates and Watts 1988). Order of inclusion followed the importance of correlation.

1.05 Results

1.05.01 Performance assessment of stand- and tree-level yield models

(i) Degree of fit and biases

Both the stand- and the tree-level models perform moderately well as coefficients of determination (R^2) between observed and predicted volume increments are significant but low (Table 1.4). Simultaneous F-test revealed that both models are biased as the hypothesis that intercept and slope of the relationship between observed and predicted volume increment is not significantly different from zero and unity respectively was rejected in both cases. However, magnitude of the bias (F-statistic) is higher with stand-level model (Table 1.4). Root mean square error (RMSE) is also slightly higher in the stand-level model (1.65 vs 1.68 m^3/ha/year; Table 1.4).

Mean bias revealed that the stand-level model underpredicts annual net volume increments of black spruce plots in our study area by 3%, compared to 39% for the tree-level model (Table 1.5). Analyses of mean bias after splitting the data into classes of EBM associated attributes (e. g., cohort, stand density, potential vegetation, homogenous forest units and stand structure) revealed that mean bias of the stand-level model is strongly associated with stand density (Table 1.5). Coefficient of determination between observed and predicted volume increments is the lowest for intermediate stand densities, where stand structure types often vary. In the eastern softwood unit where the stand-level model has the lowest coefficient of determination (0%, Table 1.4), a great majority of plots (89%, 245 out of 275 plots) are found in the balsam fir - black spruce potential vegetation with an intermediate plot density and an uneven-aged stand structure. In the central softwood unit where coefficient of determination for the stand-level model is slightly better, pure black spruce stands are also encountered in the same dominating potential vegetation type, but with stand structure more often regular.

Table 1.4 Prediction errors by attributes potentially used in ecosystem-based management

Attributes	R^2		F- Statistic		RMSE		e*		e**		Plot frequency
	PS	TWIGS	PS	TWIGS	PS	TWIGS	PS	TWIGS	PS	TWIGS	(n=1311)
Study area	**10%**	**13%**	**166**	**33**	**1.68**	**1.65**	**3.73**	**3.21**	**3.98**	**3.43**	**100%**
Homogenous forest units											
Eastern Softwood	0%	1%	127	3	1.52	1.44	3.50	2.75	3.89	3.06	21%
Central Softwood	14%	14%	49	4NS	0.83	0.83	2.11	1.54	2.42	1.77	23%
Western Softwood	6%	16%	126	13NS	1.79	1.79	3.78	3.33	4.32	3.81	37%
Eastern Mixedwood	11%	8%	22	21	2.39	2.32	4.94	4.08	6.48	5.34	12%
Western Mixedwood	0%	5%	17	2NS	2.22	2.26	4.49	4.56	5.42	5.50	6%
Potential vegetation types											
Balsam fir-white birch	29%	16%	4	4NS	2.05	2.23	3.66	3.97	5.31	5.76	3%
Black spruce-moss	12%	8%	78	9NS	1.65	1.67	3.62	3.17	4.08	3.57	29%
Black spruce-sphagnum	26%	11%	18	1NS	1.68	1.85	3.55	3.19	4.70	4.23	5%
Balsam fir-black spruce	5%	14%	169	17	1.51	1.44	3.43	2.77	3.73	3.02	57%
Stand density											
Dense	17%	21%	74	24	2.41	2.36	5.47	4.67	6.30	5.38	21%
Medium	8%	14%	97	8NS	1.32	1.27	2.84	2.41	3.13	2.65	44%
Sparse	11%	5%	90	2NS	1.44	1.48	3.14	2.77	3.51	3.09	35%
Stand structure											
Regular	3%	6%	124	16	1.71	1.66	3.89	3.19	4.32	3.54	48%
Irregular	14%	11%	20	3NS	2.49	2.54	4.97	4.64	5.93	5.54	13%
Uneven-aged	15%	17%	152	28	2.28	1.27	2.94	2.49	3.22	2.74	39%
Cohort types											
1	11%	13%	83	12	2.22	2.19	4.70	4.17	5.18	4.59	45%
2	19%	10%	29	24	1.41	1.48	2.90	2.99	3.44	3.55	14%
3	7%	1%	90	11	0.72	0.75	2.37	1.43	2.62	1.58	41%

1. NS Indicate intercepts and slope not significantly different from 0 and 1 respectively at 95% confidence interval.
2. PS denotes the result from Pothier and Savard (1998) and TWIGS belongs to the results from Teck and Hilt (1990; 1991).
3. e* and e** are the first and second critical errors of mean bias ($\bar{E}$) calculated based on Student's T-statistic (Reynolds 1984).

Table 1.5 Confidences, prediction and tolerance intervals of the model mean

Attributes	Mean bias ($\bar{E}$) ($m^3ha^{-1}year^{-1}$)		Half confidence interval of $\bar{E}$		Half prediction interval of $\bar{E}$		Half tolerance interval of $\bar{E}$		Plot frequency
	PS	TWIGS	PS	TWIGS	PS	TWIGS	PS	TWIGS	(n=1311)
Study area	**0.01**	**0.26**	**±0.11**	**±0.09**	**±2.14**	**±1.97**	**±3.86**	**±3.28**	**100%**
Homogenous forest units									
Eastern Softwood	0.23	0.24	±0.17	±0.09	±3.66	±2.87	±3.86	±3.03	21%
Central Softwood	-0.10	-0.02	±0.13	±0.12	±2.25	±1.65	±2.42	±1.77	23%
Western Softwood	-0.24	-0.04	±0.23	±0.11	±4.01	±3.57	±4.31	±3.83	37%
Eastern Mixedwood	0.15	0.49	±0.65	±0.23	±5.66	±4.58	±6.54	±5.29	12%
Western Mixedwood	1.53E-03	1.16	±0.4	±0.16	±4.94	±4.46	±5.46	±4.93	6%
Potential vegetation types									
Balsam fir-white birch	0.40	0.83	±0.68	±0.31	±4.50	±4.66	±5.37	±5.56	3%
Black spruce-moss	-0.04	0.31	±0.20	±0.10	±3.85	±3.31	±4.09	±3.52	29%
Black spruce-sphagnum	-0.90	-0.30	±0.44	±0.24	±3.69	±3.71	±4.28	±4.23	5%
Balsam fir-black spruce	0.13	0.21	±0.13	±0.07	±3.57	±2.86	±3.73	±2.99	57%
Stand density									
Dense	1.11	0.76	±0.33	±0.12	±5.45	±4.79	±5.87	±5.16	21%
Medium	-0.06	0.10	±0.12	±0.08	±2.98	±2.52	±3.13	±2.65	44%
Sparse	-0.55	0.16	±0.15	±0.09	±3.14	±2.91	±3.30	±3.06	35%
Stand structure									
Regular	0.39	0.37	±0.18	±0.09	±4.03	±3.28	±4.25	±3.46	48%
Irregular	0.18	-0.33	±0.41	±0.15	±5.42	±5.04	±5.95	±5.53	13%
Uneven-aged	-0.36	0.32	±0.12	±0.08	±2.30	±2.54	±3.14	±2.66	39%
Cohort types									
1	0.39	0.24	±0.20	±0.18	±4.88	±4.35	±5.12	±4.57	45%
2	0.16	0.72	±0.23	±0.22	±3.15	±2.93	±3.45	±3.21	14%
3	-0.46	0.11	±0.10	±0.06	±2.33	±1.48	±2.44	±1.56	41%

1. Negative and positive values of the mean bias indicate over and under prediction of volume increment respectively.
2. PS denotes the result from Pothier and Savard (1998) and TWIGS belongs to the results from Teck and Hilt (1990; 1991).

Contrary to expectation, the stand-level model has its lowest coefficient of determination between observed and predicted volume increments in regular stands (3%, Table 1.4). This is because stand density explains most of the residual variance of net volume increment for the stand-level model and that stand density is weakly associated to stand structure (Boucher *et al.* 2006) (degree of association Γ between stand density and stand structure is 0.16). Dense stands (even-aged) more often tend to have a regular structure (46%, 126 out of 275 plots) than an uneven-aged structure (32%, 88 plots). Sparse stands have usually an uneven-aged or irregular structure (64%, 294 out of 459 plots).

For both models, the coefficient of determination between observed and predicted volume increments is the lowest in cohort 3 stands (last developmental stage) (Table 1.4), where plot density varies between sparse (50%, 269 out of 538 plots) and intermediate (47%, 253 plots). Contrary to the stand-level model, mean bias of the tree-level model is the highest in western mixed forest unit, where pure black spruce stands are most often found in the black-spruce moss potential vegetation and where most of the plots (63%, 99 out of 157 plots) have a sparse density.

Simultaneous F-test of the split data showed that contrary to the stand-level model, the tree-level model is not biased in central and western softwood forests (60%, 787 out of 1311 plots), balsam fir – white birch, black spruce – moss and black spruce – sphagnum vegetation types (37%, 485 plots), medium and sparse density stands (79%, 1036 plots) and irregular stands (13%, 170 plots). Magnitude of RMSE, on the other hand, is generally higher in the stand types where tree-level model was not biased (Tables 1.4 and 1.5). Finally, models are highly biased when data is split by cohorts, stand density and stand structure (Tables 1.4 and 1.5), variables potentially used under ecosystem-based management. However, relative bias of the two models differs markedly (Table 1.4).

(ii) Confidence, prediction and tolerance intervals of mean bias

The critical errors (Reynolds 1984) of the mean bias for tree-level model (e^*=3.21 m^3/ha/year and e^{**}=3.43 m^3/ha/year) are slightly smaller than those of the stand-level model (e^*=3.50 m^3/ha/year and e^{**}=3.89 m^3/ha/year) (Tables 1.4 and 1.5). Analyses of the limits of the mean biases (confidence, prediction and tolerance interval) after splitting data into different classes of the stand attributes (cohort, stand structure, stand density, potential vegetation and homogenous forest unit) revealed that tree-level model has smaller magnitude of critical error in a majority of stand types (except for western mixed forests (6%, 79 plots), balsam fir-white birch potential vegetation types (3%, 40 plots) and cohort 2 stands (14%, 184 plots) (Table 1.4). However, the magnitude of differences in the critical errors between both models remains small in all cases (Tables 1.4 and 1.5).

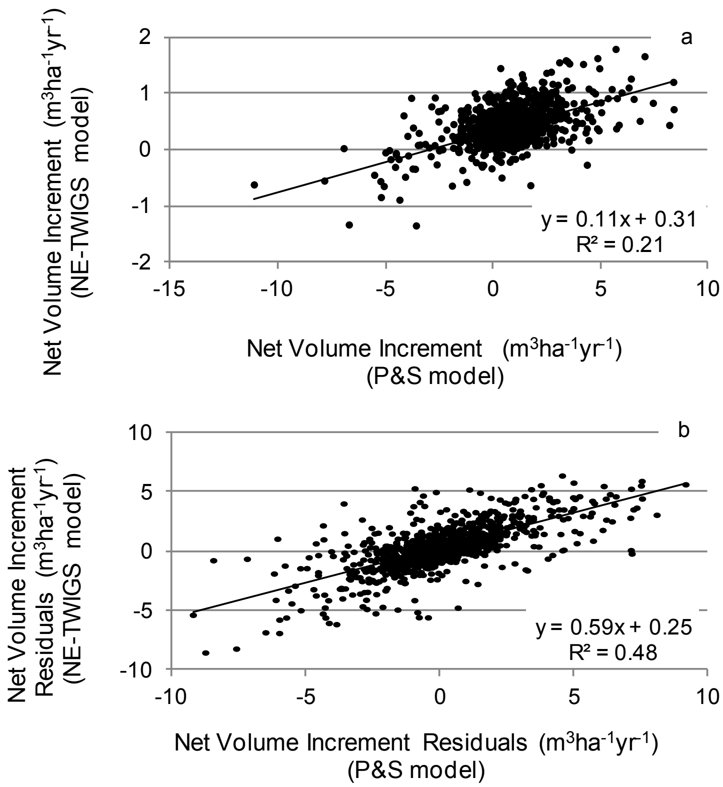

Figure 1.2 Net volume predictions between the tree-level (NE-TWIGS) and stand-level models (Pothier and Savard 1998) are moderately correlated (a) but their residuals are strongly correlated (b) indicating that sources of error associated with both the models are related

1.05.02 Identifying the main sources of error

The predictions of volume increment of both models are correlated (R^2 = 0.21), but residuals correlate more strongly (R^2 = 0.48), suggesting that the sources of error are related for both models (Figure 1.2). For both models, residuals of predicted net volume increment are at most weakly correlated to stand structural indices and stand attributes (results not shown), suggesting that only minor improvements may be gained by including these variables in the models. Residuals of both models are however moderately correlated (r_P varied between 0.40 and 0.48; $p < 0.05$) to changes in site index and relative density index (see section 1.04.04.ii) over the period between two measurements (Figure 1.3). The change in site index is at most weakly correlated to stand structural indices and stand attributes (not shown). On the other hand, the change in relative density index is moderately correlated to the observed volume loss by mortality (r_P = -0.43; $p < 0.05$) but weakly or not correlated to any other considered variables, suggesting that natural mortality is not accounted for accurately in either model.

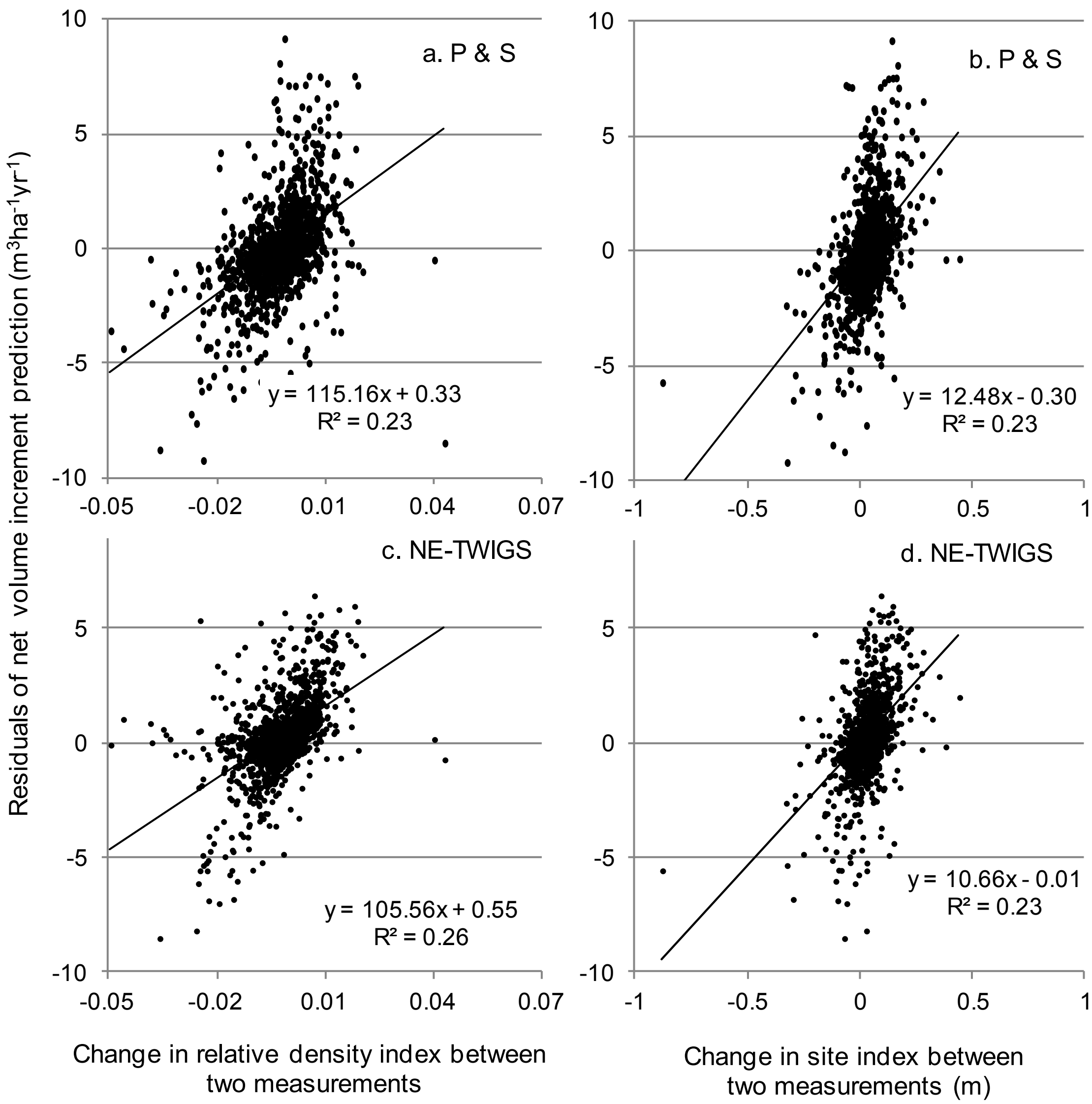

Figure 1.3 Residuals of stand- (a, b) and tree-level (c, d) volume increment predictions correlate moderately with changes of relative density and site indices between two measurements.

For the tree-level model, residuals of net volume increment are moderately correlated to changes in site index and relative density index (r_P = 0.59 and 0.36 respectively, $p < 0.05$) and residuals of predicted volume losses by mortality are moderately correlated to the change of relative density index (r_P = -0.47). The residuals of net volume increment predicted by the stand yield model are also weakly correlated to the volume loss by mortality (r_P = 0.22, $p < 0.05$) and moderately correlated with the residuals of volume loss by mortality (r_P = -0.52, $p < 0.05$). One primary source of error for both models is thus the change in stand density between two measurements, linked to the volume loss by mortality (Figure 1.4), in part in relation with mean tree age (Figure 1.5), a factor not included in the model.

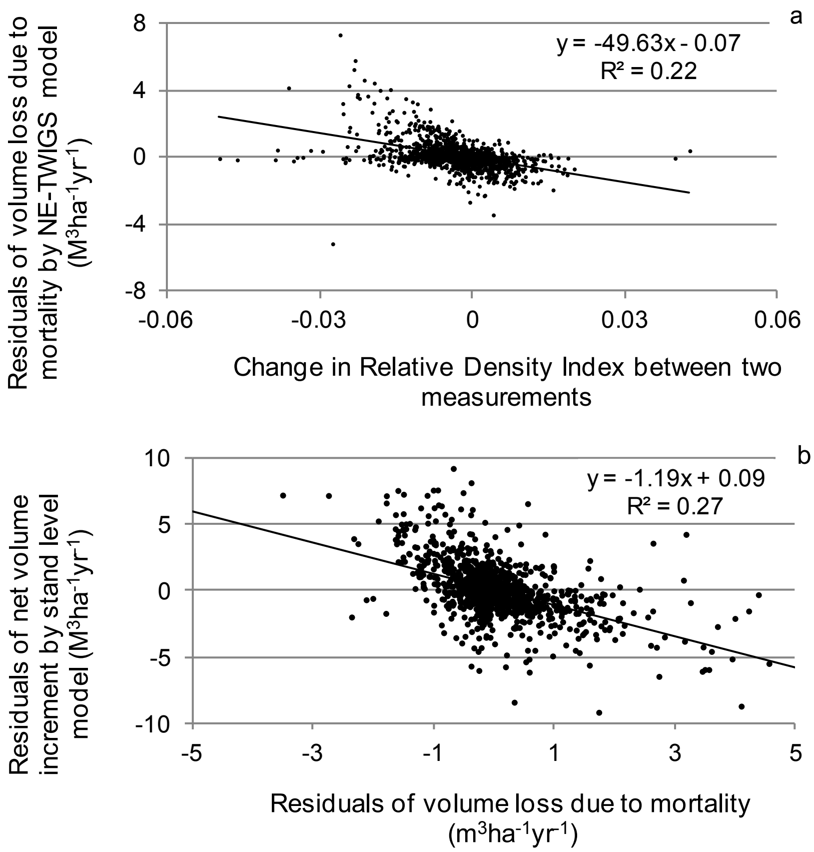

Figure 1.4 Residuals of volume loss due to mortality by NE-TWIGS model correlate moderately with the change of relative density index between two measurements (a). Residuals of net volume increment of the stand-level model correlate moderately with the residuals of volume loss due to mortality

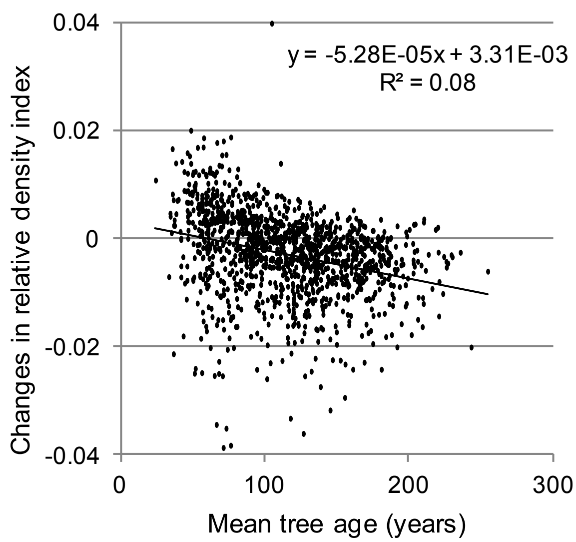

Figure 1.5 Correlation between changes in relative density index from one measurement to the next and mean age of dominant/co-dominant trees.

Table 1.6 Spearman rank correlations between different stand attributes and squared residuals of net volume increment (e^2_{iv}) for the stand- and tree-level models and of volume loss due to mortality (e^2_m) for the tree-level model (all values are significantly different from 0, p<0.01, except for the skewness coefficient).

Attributes	e^2_{iv} (stand-level)	e^2_{iv} (tree-level)	e^2_m
Shannon-Wienner Index	0.30	0.37	0.50
Skewness coefficient	NS*	NS*	NS*
Coefficient of variation	0.25	0.35	0.36
Relative density index	**0.45**	0.48	0.53
Initial standing volume (m^3ha^{-1})	0.42	0.50	**0.63**
Site Index	0.43	**0.59**	0.57

* NS=Not significant at $p < 0.05$

Squared residuals of net volume increment of both models are weakly correlated with stand structural indices and moderately correlated with site and relative density indices and initial standing volume (r_S varies between 0.42 and 0.59, Table 1.6). Once the relative density index is included into the error model of either growth and yield model (Eq. 1.6), no other variables were significant. Relative density index is however strongly correlated to standing volume (r_P = 0.85) and moderately correlated to site index (r_P = 0.62), Shannon evenness index (r_P = 0.65), and mean stem size (r_P = 0.45) (Figure 1.6).

1.06 Discussion

1.06.01 Which model is best?

The first hypothesis of this study; "individual tree growth model has a better predictive capability of net volume increment than that of stand-level model" cannot be completely rejected by the results. The stand-level model has higher biases when rated with F-statistic but its errors, when rated with correlation, are comparable to that of the tree-level model (Table 1.4 and 1.5). Porté and Batterlink (2002) suggested that stand-level models are better for regular and even-aged stands whereas tree-level models predict better in irregular stands. Tree-level model was, indeed, not biased in irregular stands in our study (Table 1.5) but the stand-level model performed as well as the tree-level model in the two other types of structure identified by Boucher *et al.* (2006) which represents 87% of the sample plots. Both models performed poorly as rated by correlation coefficient in cohort 3 stands which are more often uneven-aged with a sparse or intermediate density. Moreover, both models were significantly biased in all cases of cohorts as rated by simultaneous F-test.

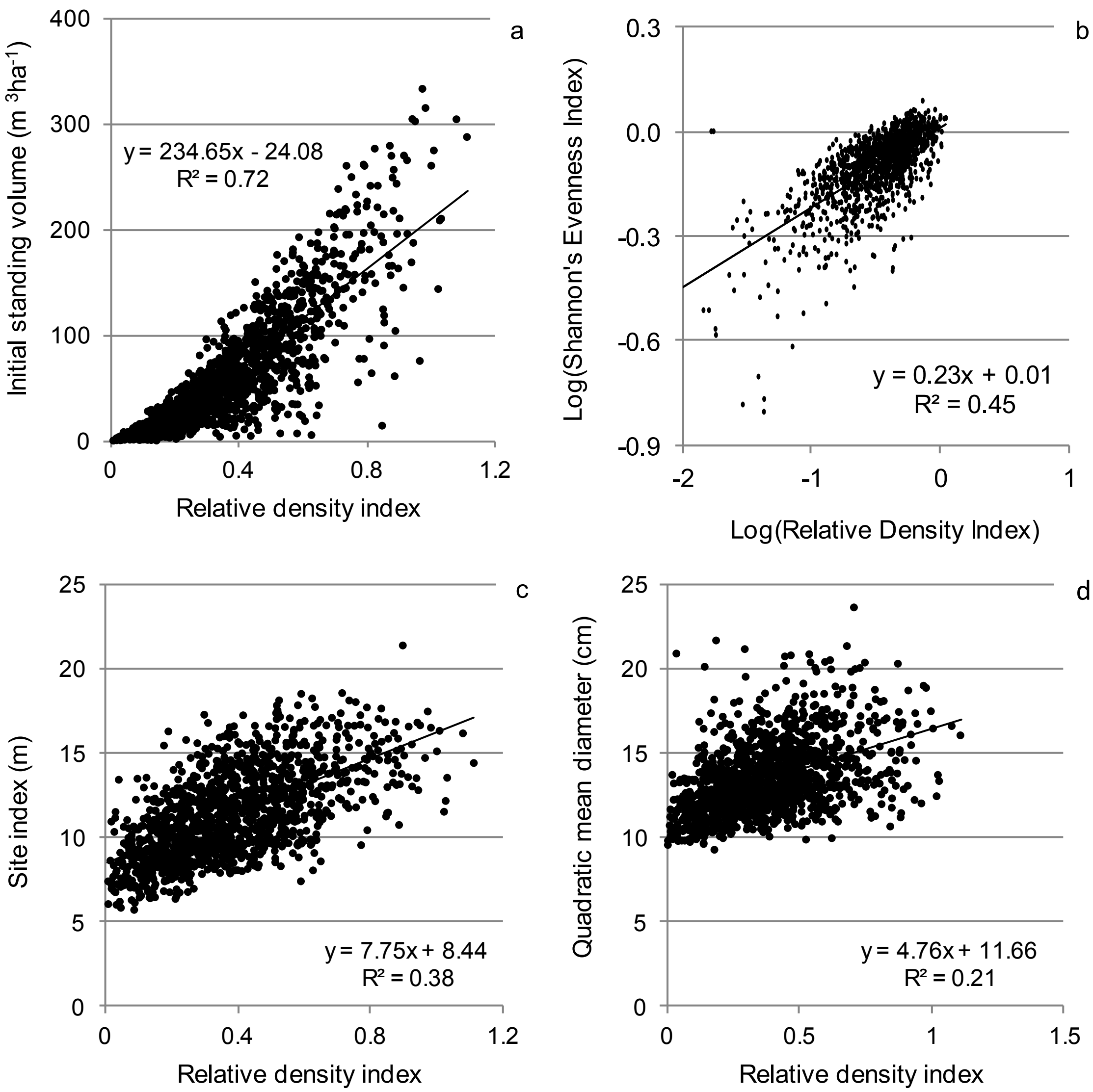

Figure 1.6 Correlation of relative density index with different stand attributes: a. Initial standing volume, b. Shannon's evenness index, c. Site index and d. Quadratic mean diameter.

Our second hypothesis "correlation between errors in the prediction of volume increment and stand structural indices is higher for a stand-level model than for a tree-level model" was not supported by the results. Residuals and square of residuals of volume increment predictions of both models behaved in a similar fashion with stand structural indices showing only non-significant or weak correlations.

Compared to stand-level models, tree-level models seem to be more flexible and more accurate under different growth conditions, provide high resolution outputs and appear better at characterizing wildlife habitats and the impacts of disturbing agents (Weiskittel *et al.* 2011, page 69). Yet, a model with less parameters and simpler assumptions is beneficial when it is able to pick up the essence of data set (Burkhart and Tomé 2012, page 238). Since the performance of both models is of moderate quality, the stand-level model still seems to be a better choice. Requirement of data at the level of individual trees and time and resources required for calculation further make the tree-level model less attractive in its current form for the tree species considered in this study. Despite all these elements, we consider that none of the models performed adequately to predict the volume growth of our study area especially when subdividing the data by attributes that will be important in ecosystem-based management. The two models selected in this study did not perform well in any of the sub-classes of the data considered in this study. They need to be improved notably in accounting for the mortality.

1.06.02 Mortality as a source of error in growth and yield models

For both models, an important source of error was linked to the volume loss due to mortality (Figure 1.4). An accurate prediction of mortality is an essential feature of any growth prediction system. A significant part of uncertainty related to growth prediction is related to mortality (Eid and Oyen 2003). For instance, Gertner (1991) used an error propagation method to calculate the variance of growth and yield predictions made with STEMS (a tree-level growth model) and found that 68% of basal area growth variability was associated with the mortality model. However, mortality is a rare, irregular and stochastic phenomenon and is therefore difficult to predict (Lee 1971). Maybe due to the difficulty in modeling mortality as an ecological process, most growth models do not simulate mortality. Out of over 150 plant growth models reviewed by Hawkes (2000), less than 50% included mortality equations.

Some progress has been made in understanding of the natural tree mortality in temperate as well as boreal regions recently. Van Mantgem *et al.* (2009) suggested two categorical factors for the increase in dominant and co-dominant tree mortality in temperate forests; endogenous factors (e. g., competition for resources and space, changes in forest structure) and exogenous factors (e. g., environmental changes such as regional temperature rise and consequent increase in drought stress). A recent increase in natural tree mortality in temperate forests was in fact attributed to rapid environmental changes (van Mantgem *et al.* 2009). Peng *et al.* (2011) also reported a widespread increase in tree mortality of natural boreal forests across Canada in the last 50 years and suggested that water stress created by regional drought was behind such increase which may also have wider consequences on forest structure, carbon storage, habitat types and vulnerability to natural disturbances such as fire. However, the physiological causes of climate-induced tree mortality are still poorly understood, which limits our ability to predict the impact of increase in mortality on carbon, energy and water fluxes as well as linking the pattern of mortality with extreme climate events (Peng *et al.* 2011). In the specific case of black spruce, Pothier *et al.* (2012) have shown that severity of past defoliation by spruce budworm (*Choristoneura fumiferana* (Clem.)), that usually affects balsam fir stands, was significantly correlated to black spruce mortality.

Several approaches have been suggested or applied to improve mortality models, particularly by considering more information related to physiology as well as climate (Gertner 1991; Peng et. al. 2011). Increasing the size of the plots to collect information on tree mortality may be an option. For instance, other provinces in Canada, such as Ontario and Alberta, use bigger plot size to assess the tree mortality: 6400 m^2 (Sharma *et al.* 2008) and 1000 m^2 (ASRD 2005), respectively. Changing the plot size has, however, strong implications in terms of available budget, sampling intensity, and compatibility with past measurement campaigns.

1.06.03 The missing feature for both models: accounting for forest succession

It is also possible that important factors are not accounted for by any of the two models, forest succession being one of them notably. Forest succession is expressed by pure black spruce stands with stand senescence (Garet *et al.* 2009, their Fig. 2) and changes in forest composition (Bouchard *et al.* 2008). Stand senescence starts to occur once the stand age exceeds the species longevity (Robichaud and Methven 1993) and is difficult to identify with mean tree age alone (Garet *et al.* 2012). Stand senescence is not accounted for with the self-thinning concept (Vanderschaaf and Burkhart 2008; Sturtevant *et al.* 1997) on which the stand-level yield model of Pothier and Savard (1998) relies upon. In natural forests, stand age, as defined by the time since last stand replacing disturbance, is often much longer than mean tree age. Senescence and succession are therefore impossible to capture in models such as Pothier and Savard (1998) which are fitted to plot data where mean tree age is used to represent stand age.

In the NE-TWIGS model, stem DBH is included in the mortality model (Table 1.2, eq. 1.2), but no distinction can be made between the mortality caused by competition and mortality due to age, since stems smaller than the average stem size tend to die more frequently (assuming that tree size is always related to age). In fact, Garet *et al.* (2009) have shown for black spruce stands that the change in the relative density index, the most important source of error for both models (Figure 1.4), is related to the time since the last fire and is in interaction with site index and time interval between two successive measurements of permanent sample plots. This relationship is weakly apparent in Figure 1.5 and deserves further investigation. We evaluated the model residuals against the interval between two measurements for both models but the correlation was very weak (0.005 for P & S model and 0.01 for NE-TWIGS model, graph not shown).The most important variable missing is the time since the last stand-initiating disturbance (Garet *et al.* 2012) which is still not available for the whole study area (Gauthier et al. 2009a). In the context of ecosystem-based management, this variable corresponds to the key temporal variable for assigning cohorts as well as for determining forest age structure over the territory (Bergeron 2004; Bergeron *et al.* 2001).

In reviews of growth and yield models (except Porté and Bartelink 2002), forest succession models (gap or landscape models, e.g. Botkin *et al.* 1972; Yemshanov and Perera 2002) are either treated apart from growth and yields models (e. g., Monserud 2003) or not treated at all (e. g., Weiskittel *et al.* 2011; Burkhart and Tomé 2012). Even-aged and irregular stands or pure and mixed stands may be linked through time by forest succession. In the lengthy transitional stages between these states, net growth is controlled more by senescence than by competition (Franklin *et al.* 2002; Gauthier *et al.* 2004; Bouchard *et al.* 2008) and cannot be explained by most growth and yield models. Accounting for forest succession in growth and yield modeling could imply a change of scale or the consideration of scales usually ignored by such models .It could also imply a stratification of calibration data (i.e. temporary or permanent sample plots) as a function of expected chronosequences with a forest succession model (e.g. Bond-Lamberty *et al.* 2005). Forest succession is under the control of disturbances such as fire, insects and windstorms and is by essence a stochastic process very difficult to predict at a very fine scale such as that of a plot or even a stand. For long-term timber supply planning, however, accuracy is essentially required at coarse scales such as whole forest or operating units where planning problems are solved (e.g. Bettinger *et al.* 2009). Such models could thus be evaluated differently.

1.07 Conclusion

The growth and yield models analyzed in this study gave mixed results over the stand attributes that might be relevant to the ecosystem-based management. Both models were highly biased to predict volume growth at the level of cohorts, a key variable in the context of EBM. Nevertheless, models inspired by EBM remain nonexistent and we still need to choose from existing ones. This analysis showed that we need to use these models with caution. Our results suggest that the use of the simpler stand-level model is still a valid option for the majority of the area covered by this study.

The tree-level model has allowed us to identify the different error components associated with both models. An important part of the total error was associated with the volume loss due to mortality. This analysis further revealed that change in stand density over time, which is linked to natural mortality of trees in a stand, is one of the major sources of error in the stand-level model. In particular, both models did not consider tree senescence notably of the first canopy tree cohort. With this and similar studies, it is clear that future research should still be directed at improving non-catastrophic mortality predictions both at the tree and stand levels. Different approaches could be followed for such an improvement, notably the inclusion of physiological and climate variables. Yet, efforts are needed to integrate elements of forest succession in growth and yield models.

1.08 Acknowledgments

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Chapter 2

Emulating boreal forest disturbance dynamics: can we maintain timber supply, aboriginal land use, and woodland caribou habitat?

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2.01 Résumé

Les effets sur l'approvisionnement en bois de la mise en place d'une stratégie d'aménagement écosystémique ont été évalués dans une Unité d'Aménagement Forestier en zone boréale de l'est du Canada. La programmation linéaire standard a été utilisée pour tester les effets de quatre enjeux clefs : (1) tendre vers une structure d'âge forestière établie à partir du régime naturel des feux et de la dynamique forestière (approche multi-cohorte), (2) agglomérer les blocs de récolte dans des chantiers de récolte afin de reproduire les patrons de perturbation naturelle à l'échelle du paysage, (3) maintenir les taux cumulés de coupe totale et de perturbation naturelle à l'intérieur du domaine historique de variabilité, et (4) exclure de la récolte les aires d'intérêt potentiel pour les peuples autochtones. La structure d'âge forestière ciblée peut être atteinte avec une réduction minime de l'approvisionnement périodique en bois, mais seulement après 50 ans. Comparé à un scénario de statu quo, l'inclusion des trois premiers enjeux résulte en une baisse de 3 à 22% de l'approvisionnement périodique et une période de restauration requérant que la coupe totale soit exclue sur 43 à 67% de la superficie productive pour les prochains 50 ans. De tels résultats requièrent que la coupe partielle ne soit pas limitée aux chantiers de récolte ouverts à la coupe totale. L'exclusion supplémentaire des aires forestières d'intérêt potentiel pour les peuples autochtones entraîne une baisse additionnelle de 4 à 10% dans l'approvisionnement. Une validation des filtres bruts utilisés dans cette étude (les trois premiers enjeux) a été faite en utilisant les besoins en habitats du caribou des bois (*Rangifer tarandus caribou*). Pratiquement tous les scénarios induisaient un taux de perturbation susceptible de permettre le maintien du caribou des bois d'ici 25 ans.

Mots clés : aménagement écosystémique, aménagement avec des cohortes multiples, approvisionnement en bois, forêt boréale, peuple autochtone, scénarios d'aménagement forestier, programmation linéaire, caribou des bois

2.02 Abstract

The effects on timber supply incurred by implementing an ecosystem-based management strategy were evaluated in an eastern Canadian boreal forest management unit. Standard linear programming was used to test the effects of four key policy issues: (1) aim for a targeted forest age structure inspired by natural fire regime and forest dynamics (multi-cohort approach), (2) agglomerate harvest blocks in operating areas to reproduce natural disturbance patterns at the landscape-scale, (3) maintain disturbance rates (clear cutting + fire) inside the historical range of variability, and (4) exclude from harvest the areas of potential interest to aboriginal people. The targeted forest age structure was achieved with a minimum reduction of periodic timber supply, but only after 50 years. Compared to a "business-as-usual" scenario, inclusion of the first three policy issues resulted in a 3 to 22% reduction in planned timber supply and a restoration period requiring that 43 to 67% of the productive area be excluded from clear-cutting activities for the next 50 years. Such results require that partial cutting not to be confined to operating areas eligible for clear-cutting. Further exclusion of forest areas of potential interest to aboriginal people resulted in an additional 4 to 10% decrease in planned timber supply. Validation of the coarse filters used in this study (first three policy issues) was done using habitat requirements of woodland caribou (*Rangifer tarandus caribou*). Almost all scenarios induced a disturbance rate likely to allow a self-sustaining woodland caribou population within 25 years.

Key words: Ecosystem-based Management, Multi-cohort Management, Timber Supply, Boreal Forest, Aboriginal People, Forest Management Scenarios, Linear Programming, Woodland Caribou

2.03 Introduction

Forest ecosystems are shaped by environmental conditions and natural disturbances such as fire, insect epidemics, and diseases (Seymour and Hunter 1999, Stocks *et al*. 2002, Flannigan *et al*. 2005). The fire regime of the North American boreal forest is characterized by a predominance of frequent and severe fires that reset the successional clock over large areas (Payette 1992, Flannigan *et al*. 2005). While the widespread occurrence of catastrophic wildfire has long been used to justify even-aged management of boreal forests (Bergeron *et al.* 2002, Gauthier et al. 2009), the natural disturbance regime over much of this forest is not driven solely by stand-replacing fires. Secondary disturbances also play an important role by altering internal stand structure and the spatiotemporal distribution of stands across the boreal landscape. Hence, secondary disturbances should also be taken into account when designing management strategies that are aimed at emulating natural disturbances (Bergeron *et al.* 2001, Gauthier et al. 2009).

Until recently, timber supply was determined by following the concept of sustained yield. The forest was "regulated," meaning that an equal volume of wood could be harvested each year in perpetuity, eventually leading to all age-classes occupying equal areas (Davis *et al.* 2001, Buongiorno and Gilless 2003). This unrealistic assumption has led to simplified forest ecosystems and reduced biodiversity (Seymour and Hunter 1999). Consequently, even-aged management fails to emulate many aspects of natural disturbance processes (Kimmins 2004). Moreover, a fully regulated forest landscape has a truncated age-class distribution with no over-mature or old-growth stands. Such a structure is outside the range of natural variability (Bergeron *et al.* 2002, Bergeron 2004, Cyr *et al.* 2009). Reducing the differences between the spatiotemporal patterns that are created by management activities and natural disturbances can therefore help maintain the integrity of forest ecosystems (Haeussler and Kneeshaw 2003, Gauthier *et al.* 2009).

Ecosystem-based management (EBM), which is based on natural disturbance dynamics, offers an alternative to even-aged management for preserving forest diversity and function (Harvey *et al.* 2002, Gauthier *et al.* 2009). EBM implies the use of silvicultural strategies that seek to reproduce the frequency, severity, and spatial distribution of natural disturbances (Haeussler and Kneeshaw 2003, Gauthier *et al.* 2009). The theoretical framework of EBM has been described extensively (Landres *et al.* 1999, Seymour and Hunter 1999, Bergeron *et al.* 2002, Harvey *et al.* 2002) and proposals have been made for its application in the field. One example is the multi-cohort management approach designed for implementation in eastern Canadian boreal forests (Bergeron *et al.* 2002).

The aforementioned approach is based on a compromise between rotation age and the expected longevity of black spruce trees (*Picea mariana* (Mill.) BSP), and is implemented in several steps. First, the forest age-class distribution is divided into three cohorts, which correspond to natural stand development stages (Franklin *et al.* 2002). Second, different silvicultural strategies are proposed, either to maintain the stands in the same cohort or to shift them to another cohort. Third, proportions of each cohort in the boreal landscape, together with cohort transition rates, are based on a modeled disturbance regime (Bergeron *et al.* 2002) and forest successional dynamics (Harvey *et al.* 2002). However, the applicability of the cohort approach in terms of its ecological suitability, economic viability, and social acceptability has yet to be evaluated in an operational context.

One of the main challenges that is associated with the social acceptability of forestry practices is the recognition and inclusion of traditional ecological knowledge (Cheveau *et al*. 2008, Wyatt 2008, Beaudoin 2012, Jacqmain *et al.* 2012). With the intensification of industrial forestry that followed the colonization of rural regions of boreal Canada, aboriginal people have felt that their land had been unfairly expropriated, which led to conflicts with forest product companies and governments (Treseder and Krogman 1999, Wyatt 2008). For example, the Kitcisakik Anicinapek (Algonquin) community of Quebec maintains the claim that timber extraction over the last century has resulted in the loss of biodiversity and has failed to protect their relationship with the forest environment (Saint-Arnaud *et al.* 2009). Yet, it is widely recognized that the success of public forest management depends mainly upon the active participation of local communities, including indigenous peoples (Brunson 1996, FSC 2004, Saint-Arnaud *et al.* 2009). Forest management is more acceptable to local communities if they feel that their concerns have been addressed and that they have been treated fairly during the planning process (Brunson 1996, Wyatt *et al.* 2011).

In EBM, the natural forest age-structure is reproduced in a non-spatial, coarse-filter approach to maintain the habitat requirements of most species present (Haufler *et al.* 1996, Seymour and Hunter 1999). Major coarse filters are the age structure, the size and spatial distribution of harvest agglomerations and the rate of disturbances. Agglomeration of harvesting activities allows for the adjustment of frequency distributions of forest patch sizes and disturbance rates (two other coarse filters), which minimize fragmentation and maximize connectivity between habitat patches at the landscape scale (Bergeron *et al*. 1999, Belleau *et al.* 2007). Fine filters are complementary to coarse filters and focus on the conservation of specific elements that are not captured by the latter (Haufler *et al.* 1996). For example, agglomerating harvesting activities can limit the extent of anthropogenic disturbances and could thus potentially protect key habitat attributes of endangered species such as the woodland caribou (*Rangifer tarandus caribou*) (Faille *et al.* 2010, Moreau *et al.* 2012). However, timber harvesting has a greater negative impact than fire on the probability of conserving a sustainable caribou population (Vors *et al.* 2007, Wittmer *et al.* 2007, Environment Canada 2011). Therefore, the management of this endangered species is likely to require a fine-filter approach.

Our objective was to quantify the sensitivity of timber supply to EBM implementation within an eastern Canadian boreal forest management unit. Results were compared with a business-as-usual (BAU) scenario. We hypothesized that 1) timber supply would not decrease as a result of implementing EBM, and that 2) it is possible to take into account aboriginal considerations through the coarse-filter approach offered by EBM without drastically reducing timber supply. We further hypothesized that coarse-filter measures would increase the likelihood of maintaining viable woodland caribou populations by providing larger intact forest tracts.

2.04 Methodology

2.04.01 Study area

The study area was Forest Management Unit 085-51 in western Quebec, which covers an area of 1.08 million ha (Figure 2.1). About half of the territory (542 000 ha) was considered productive, i.e., having the capacity to produce more than 50 m^3 ha^{-1} of wood in one rotation (Tembec 2007). The company responsible for the forest management unit wished to apply EBM to meet certification standards (FSC 2004).

A substantial portion of the ancestral territory of the Pikogan Anicinapek (Algonquin) community (ca. 800 members) is located within the study area. People from Pikogan extensively use the territory for several cultural activities such as hunting, trapping and collecting various forest products (Germain 2012).

The study area is located entirely in the boreal bioclimatic zone, and mostly within the black spruce-feather moss bioclimatic sub-domain (Robitaille and Saucier 1998). Plains dominate the topography with an elevation averaging 280 m a.s.l. Soils of the northern half of the study area are characterized by poorly drained clay, whereas better-drained tills are more frequent in the southern half, even though clay deposits remain abundant. Mean annual temperature varies from -2.5 °C to 0 °C, the length of the growing season is 150-160 days, and total precipitation is 700-800 mm (Robitaille and Saucier 1998). Forest landscapes are fairly uniform, dominated by extensive pure black spruce stands. Black spruce is sometimes accompanied by other species such as jack pine (*Pinus banksiana* Lamb.), balsam fir (*Abies balsamea* (L.) Mill.), paper birch (*Betula papyrifera* Marsh.), trembling aspen (*Populus tremuloides* Michx.), and, to a lesser extent, balsam poplar (*Populus balsamifera* L.). Bryophytes and dwarf ericaceous shrubs form the understory. Herbaceous species are sparse (MRN 2003, Fenton and Bergeron 2006).

The natural disturbance regime of the study area is dominated by large crown fires (Bergeron *et al.* 2004), but gap-phase dynamics and wind throw are also important, as the present fire cycle (ca. 400 years, Bergeron *et al.* 2004) is longer than the mean longevity of black spruce (100-150 years, Robichaud and Methven 1993).

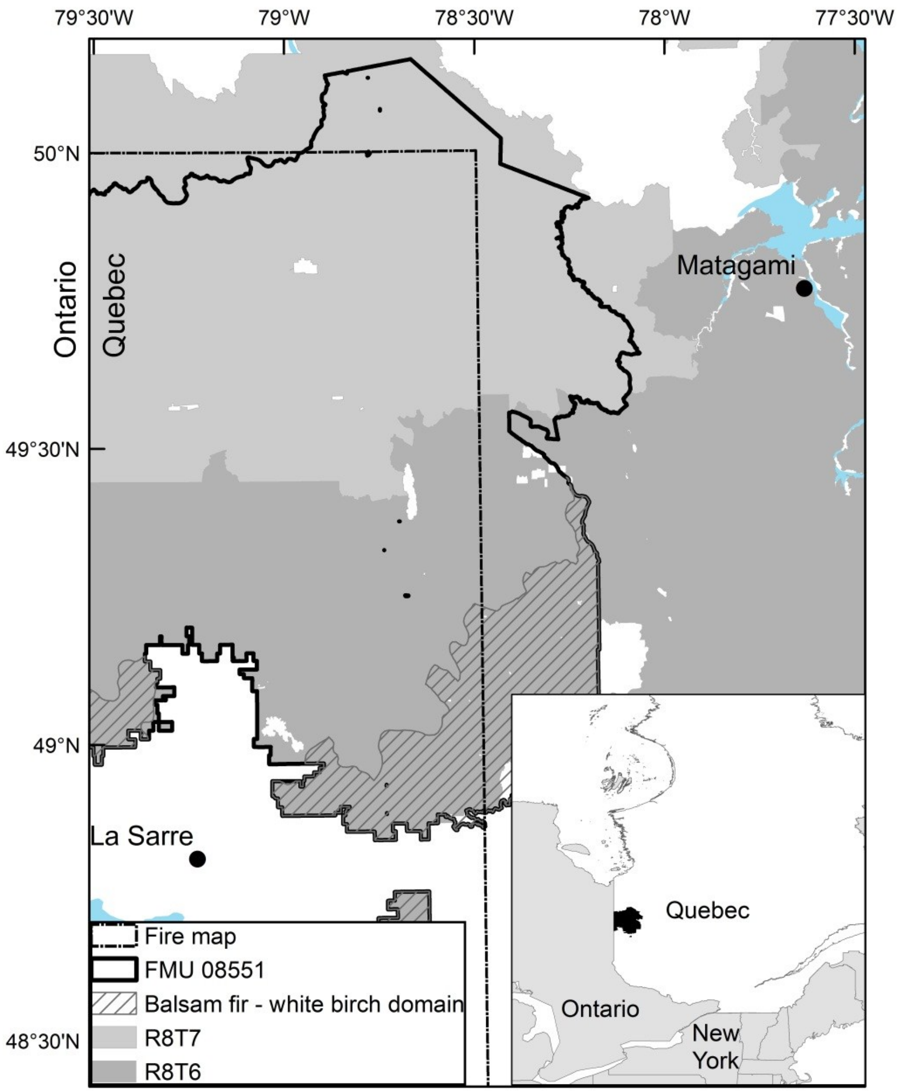

Figure 2.1 Study area (Forest Management Unit 085-51). R8T7 and R8T6 are two distinct forest inventory units.

Burn rate in the study area has declined sharply from 0.68% year^{-1} between 1850 and 1920 to 0.25% year^{-1} since 1920 (Bergeron *et al.* 2004). In the prolonged absence of fire, the productivity of black spruce stands gradually declines due to paludification (Belleau et al. 2011, Lecompte et al. 2006). Jack pine and aspen colonize post-fire stands in well-drained areas (Belleau and Légaré 2009).

The forest dynamics in the region can be simplified to three main successional pathways, which are dominated by black spruce, jack pine, or trembling aspen (Belleau et al. 2011, Lecompte et al. 2006, Bergeron *et al.* 2002, Nguyen-Xuan 2002, Gauthier *et al.* 2004). These pathways can be further subdivided into two to three developmental stages or cohorts. The first developmental stage (cohort 1) corresponds to the stands that were recently initiated by a stand-replacing fire. Stands in cohort 1 are dense and closed, with a simple vertical structure. In the absence of stand-reinitiating disturbances, cohort 1 evolves into cohort 2, which is characterized by a semi-open and irregular canopy. At this developmental stage, shade-intolerant species are gradually replaced by black spruce. In the absence of a stand-reinitiating disturbance, stands continue to evolve into a third cohort, with complete replacement of the individuals established during the first and second cohorts (Belleau and Légaré 2009). Third cohort stands are relatively open and uneven-aged, with a well-developed vertical structure.

2.04.02 Forest stratification and timber yield

Analyses were based on the data gathered for the last timber supply analysis that was performed by the Ministère des Ressources Naturelles (MRN) for the 2008-2013 planning period. From the forest maps used for the timber supply analysis, we grouped forest polygons into inventory strata based on cover type, stand density and height, age class, and ecological type. Inventory strata with a canopy height > 7 m (467 strata) were characterized using 400 m^2 circular temporary sample plots (2696 plots). Within sample plots, species and diameter at breast height (DBH, 1.3 m; 2 cm diameter classes) were recorded for every tree with a DBH > 9 cm. Height and age were also noted for four to nine randomly selected canopy-dominant or co-dominant trees per plot. About 40% of these plots were sampled in 1996 or 1997 within two forest inventory units covering the study area (Figure 2.1). Plots that had been sampled earlier (between 1986 and 1996) but were located in forest polygons that had remained undisturbed between their initial measurement and 1996 were aged to 1996 with an empirical growth model created by the MRN. To complement this dataset, the MRN added a further 1050 plots outside of these two inventory units, based on similarity in their cartographic attributes (Bernier *et al.* 2010). To simplify timber supply analyses, inventory strata were grouped into 328 management strata with the following method:

1. Inventory strata with stand age < 30 years (canopy height < 7 m) were considered under regeneration, irrespective of their origin (729 inventory strata). Considering their ecological types and age classes, these regeneration strata were grouped into 124 separate management strata based on their regeneration mode (including secondary anthropogenic disturbances such as commercial and pre-commercial thinning).
2. Cluster analysis grouped 467 inventory strata into 180 management strata for which sample plot data were available. The precision in merchantable volume estimation for an inventory stratum was calculated as the ratio of half the 95 % confidence interval of the mean on the mean volume. The Mahalanobis distance (Mahalanobis 1936) was calculated with vectors of merchantable volume per

species groups and DBH classes (5 cm classes) between the stratum with the lowest precision and the remaining strata with similar potential vegetation (Robitaille and Saucier 1998). Species groups were based on softwood/hardwood distinction and shade-tolerance/intolerance. An *F*-test determined if there was a significant difference with the closest stratum; strata were grouped when the *F*-test was not significant. The procedure was iteratively repeated until strata could no longer be grouped. The CANDISC procedure (SAS Institute, Cary, NC) was used to perform the cluster analysis.

3. To group inventory strata that were neither regenerating nor had sample plot data available, a key was developed that was based on cartographic information using age, species group, ecological type, canopy height, and density class available from the last forest inventory. The 209 inventory strata having attributes similar to those of management strata, which had been obtained with the cluster analysis, were included in these management strata. The remaining 104 inventory strata (15 051 ha; < 3 % of the total productive area) were grouped into 24 new management strata.

The yield trajectory of each management stratum was based on area-based weighted averages of the yield tables that were compiled by the MRN during their last timber supply analysis. These yield curves were calculated using the model developed by Pothier and Savard (1998). Species volumes were also aggregated on the basis of softwood/hardwood distinction and shade tolerance/intolerance information.

2.04.03 Forest development stages

Forest age-class structure is the main coarse-filter indicator used in the multi-cohort management approach. A target age structure is expressed in terms of areal proportions of developmental stages or cohorts and is derived from a theoretical age-class distribution. Assuming a random probability of fire, Van Wagner (1978) showed that these estimates follow a negative exponential distribution. The calculation of areal proportions by cohort requires a mean time since the last stand-initiating fire and estimates of ages at which transitions are expected to occur from one cohort to another. For this purpose, the fire history of the study area was extracted from the fire history map produced by Bergeron *et al.* (2004) to compute a mean time since the last fire (147 years, with lower and upper 95% confidence limits of 125 and 170 years, respectively). Gauthier *et al*. (2004) estimated transition ages of roughly 150 years between first and second cohorts, and 275 years between second and third cohorts. Consequently, cohort proportions were estimated using equation 3 of Van Wagner (1978), as 63%, 21%, and 16% of the forest area for cohorts 1, 2, and 3, respectively.

To monitor cohort proportions during timber supply analyses, cohort characterization had to be linked to yield curves. Stand age normally corresponds to the mean age of canopy trees, and differs from time since the last fire when mean tree longevity is exceeded. Gauthier *et al.* (2004) linked cohort transitions with changes in stand structural and compositional attributes. For the successional pathways that were dominated by jack pine (intolerant softwood) and trembling aspen (intolerant hardwood), cohort transition is linked to changes in species composition, while it is a function of stand structural changes for the pathway dominated by black spruce (tolerant softwood). Consequently, successional pathways first had to be identified, and then, two different strategies had to be followed to identify cohort number with yield curves (i.e., one for the pathways dominated by jack pine or trembling aspen, and one for the black spruce pathway). An identification key was constructed with the yield curves of each management stratum specifying the volume proportions of each species and, using the method of Nguyen-Xuan (2002), identifying the successional pathway for each sample plot of the corresponding stratum. Minimum confusion between the assignments of successional pathways to strata with plots or yield curves was attained with the pathway assigned to black spruce when the volume proportion of tolerant softwoods was > 70 %, to trembling aspen when the volume proportion of intolerant hardwoods was > 30 %, and to jack pine otherwise. The concordance level between the two methods of assigning the successional pathways was 84%.

For jack pine and trembling aspen successional pathways, two identification keys (one for each successional pathway) were developed to determine cohort number, based on the volume proportion of each species and age class. With these keys, confusion matrices of cohort number prediction between sample plots and yield curves showed concordances of 56% (jack pine) and 62% (trembling aspen). For the black spruce pathway, a different analysis was carried out, with the cohort number derived directly from yield tables rather than from sample plots. Nguyen-Xuan (2002) distinguished cohorts, based on stem densities of the middle- and upper-canopy layers. Pothier and Savard (1998) provided yield tables for different minimum merchantable DBHs (i.e., 9 cm to 17 cm), which allowed stem density estimation for both layers (i.e., 9 cm-15 cm, and > 15 cm). For each age step of a set of black spruce yield curves that covered the observed range of site index and stand density in the study area, the number of stems per hectare was thus estimated for two DBH classes, which were respectively assigned to the middle and upper layers, consistent with the Nguyen-Xuan (2002) method. A general logistic model (PROC LOGISTIC, SAS Institute, Cary, NC) was then used to estimate the probability of a stratum being in a specific cohort as a function of standing volume of black spruce and mean age of canopy trees. The Nagelkerke pseudo-R^2 of the logistic regression was 0.86.

2.04.04 Silvicultural strategies

In the BAU scenario, careful logging around advanced regeneration (CLAAG) (Groot *et al.* 2005) was the only available silvicultural option for timber harvesting. No hypotheses were made regarding forest succession. Hence, the species composition of any regenerated stand was assumed to be equivalent to that of the pre-harvest stand. In the EBM scenario, more diverse silvicultural strategies were followed to reproduce natural succession. CLAAG was used to initiate regeneration of black spruce stands, while clear-cutting followed by planting was used to renew jack pine stands, and clear-cutting followed by natural regeneration was used to regenerate trembling aspen stands. Partial cuts were used to emulate the transition of even-aged stands to irregular stands, and to convert irregular stands to uneven-aged forests. Cut with protection of small merchantable stems (in Quebec, CPPTM: *coupe avec protection des petites tiges marchandes*) was used as a partial cut option. CPPTM is a diameter-limit cut where merchantable stems < 15 cm DBH are left standing (Thorpe and Thomas 2007, Simard *et al.* 2009), although it is not necessarily considered a regeneration cut as it frees the undergrowth (Bouchard 2009). CPPTM was used to accelerate the cohort transition. It removes about 80% of the standing volume. The age of residual stands was assumed to be ten years. Irregular shelterwood cuts (50% removal of merchantable volume, Raymond *et al.* 2009) were used as a potential alternative to CPPTM. Clear cutting followed by scarification and clear cutting followed by scarification and plantation was applied to convert the black spruce dominated cohort 3 stands either to trembling aspen or jack pine stands.

2.04.05 Harvest agglomeration

To agglomerate harvest blocks, the territory was divided into different spatially organized compartments (operating areas) as a function of canopy closure, species composition and landscape patterns (Annie Belleau, Biologist MRN, pers. comm. 2009). Operating areas varied in size between 30 km^2 and 150 km^2 (Belleau and Légaré 2009), to emulate the observed range of fire sizes (Bergeron *et al.* 2004, their Fig. 8). Each operating area was required to have more than a given percentage of its productive area (e.g., 30, 50 or 70%) eligible for harvest (i.e., older than the minimum harvesting age) before any harvesting could occur. Once an operating area was open for harvest, it remained so for the rest of the planning horizon. The planning horizon was 150 years (30 periods of 5 years). Model was required to maintain 21% and 16% of the productive area under cohort 2 and 3 respectively. Adjacency constraints, green-up delays and minimum cut-block sizes were not considered in this study.

2.04.06 Timber supply calculation (baseline scenario)

The baseline scenarios were first constructed for both BAU and EBM. Standard versions of an optimal timber supply problem (e.g., Bettinger *et al.* 2009) were elaborated in Woodstock (Remsoft Inc., Fredericton, NB) and solved by linear programming with Mosek 5.0 (Mosek ApS, Copenhagen, Denmark) for the BAU and EBM baseline scenarios. In either case, the general objective was to maximize the volume planned for harvest, subject to an even-flow of harvest volume. Woodstock adds default constraints of non-negative harvested areas and resource availability.

For the EBM scenario, jack pine plantations were limited to less than the actual plantation level (<= 7500 ha per period, Tembec 2007) to constrain their potential effect on the allowable cut. Furthermore, the maximization problem was solved iteratively to reach targeted area proportions under cohorts 1, 2 and 3 (63%, 21%, and 16%, respectively) in a minimum amount of time. Since the present areas under cohorts 1, 2, and 3 were 79%, 19%, and 2%, respectively, the starting period for applying these constraints had to be delayed to find a feasible solution for the baseline EBM scenario.

2.04.07 Sensitivity analyses

A first type of sensitivity analysis was conducted with shadow prices. Shadow prices are provided with the resolution of an optimization problem and correspond to the change in the value of the objective function should a particular constraint be changed by one unit (Davis *et al.* 2001). Shadow prices help identify binding constraints and the intensity of the binding. In this study, maximum values of shadow prices for clear-cutting and partial-cutting harvests by period were used to identify the most constraining periods in terms of the availability of the resources.

The volume planned for harvest was used to assess the economic aspect of the scenarios and to rate the sensitivity of the analyses with respect to four key policy issues: (1) aim for a forest age structure targeted with the multi-cohort approach, (2) agglomerate harvest blocks in operating areas to reproduce natural disturbance patterns at the landscape scale, (3) maintain cumulated clear-cutting and natural disturbance rates below the historical range of disturbance rates, and (4) exclude forest areas of potential interest to aboriginal people by creating different buffer zones. Different target forest age structures were considered for the EBM scenarios, by assuming that the mean time since last fire was equal to either its lower (125 years) or upper (170 years) confidence limit to represent shorter and longer fire cycles, respectively. These fire regimes led to corresponding target age structures of 70%: 19% : 11% and 58% : 22% : 20% in cohorts 1, 2, and 3, respectively.

Three different intensities of harvest agglomeration were tested. As previously stated, the percentage of productive area that was eligible for harvesting was used as a criterion to open an operating area to harvest activities. The minimum percentage varied between 30% and 70%. This constraint was specified in the timber supply problems with the help of time-dependent opening curves having either 0 (closed) or 1 (open) values. The values were specified for each operating area by first simulating the baseline timber supply problem without harvest. Operating areas were kept closed until the minimum percentage of stands eligible to harvest was reached and were then left open for the rest of the planning horizon. Harvest agglomeration should have an important effect on the timber supply level and, therefore, two alternative scenarios were considered. The first scenario consisted of agglomerating all harvesting activities, as inspired by natural disturbance sizes as an integral part of our EBM approach. The second type of scenario assumed that partial cutting, which is considered a small-scale secondary disturbance (e.g., Harvey *et al.* 2002), could be less detrimental to species that were associated with late-successional forest stages (Vanderwell et al. 2007). In this latter scenario, only clear-cutting activities were spatially concentrated.

Clear-cutting rates should not exceed the difference between historical and present burn rates (Gauthier *et al.* 2009). The present (1920 and later) burn rate is 0.251% y^{-1} (= 1/398 years) (Bergeron *et al.* 2004). Considering an historical rate of 0.680% y^{-1} (= 1/147 years), the clear-cutting rate should therefore not exceed 0.429 % y^{-1}. To express the area that was harvested by clear-cutting as a rate, harvested areas were divided by the area corresponding to the terrestrial portion of the forest management unit (1.08 million ha). Harvest agglomeration tends to control the clear-cutting rate and, as a result, acceptable agglomeration intensities (expressed by the percentage of stands eligible for harvest) had to be higher than the level where the clear-cutting rate equaled the aforementioned maximum rate.

Lastly, Germain and Asselin (2010) showed that forest areas adjacent to roads (< 100 m) and water bodies (< 60 m) were used by the Pikogan aboriginal community for cultural activities such as hunting, trapping, and the collection of various forest products. Zones of aboriginal interest in the study area were thus excluded from the timber production area in the sensitivity analysis by delimiting them as buffer zones. Two types of scenarios were constructed, one that excluded all of these buffer zones from the timber production area, and one that excluded only buffers on all-season roads and water bodies having an area > 0.5 ha. The first 20 m that surrounded water bodies (riparian buffer zones) was excluded from harvest in all scenarios, as required by provincial forest regulations.

2.04.08 Risk assessment for woodland caribou persistence

Environment Canada (2011) has shown that the proportion of forest that is younger than 40 –years-old is strongly related to the mean recruitment of caribou calves throughout the species range. Recruitment is related to the probability of observing stable or increasing caribou populations over a 20-year period. Compared to burned areas, however, harvested areas apparently exert a detrimental influence on habitat suitability for woodland caribou up to a distance of 500 m from their borders. The 500-m border represents a minimum distance and may be challenged (e.g., Vors *et al.* 2007, Moreau *et al.* 2012). Nevertheless, Environment Canada (2011) showed that increasing the buffer size up to 2000 m did not significantly improve the correlation between the proportion of cumulated disturbed area (fire and harvest) and recruitment of caribou calves.

Since the timber supply models that were used in the present study are non-spatial and since agglomeration of harvesting blocks has an impact on the disturbed area for woodland caribou because of overlapping buffers, we evaluated the strength of the relationship between the proportions of disturbed area with and without a buffer at the level of operating areas. We created 500 m buffers from the margin of forest polygons harvested over the last 40 years in Arc GIS 9.3 (ESRI, Redlands, CA). Overlap of disturbed areas was removed before estimating the total disturbed area, as required by Environment Canada (2011). A Gompertz function was selected on the basis of its minimum root mean square error (RMSE) and lack of bias. Parameters were estimated by nonlinear least squares (PROC NLIN, SAS Institute, Cary, NC):

[2.1]
$$p_{bd} = \beta_1\left(1 - e^{-\beta_2 p_d}\right)^{\beta_3}$$

where p_{bd} and p_d are the proportions of disturbed area in an operating area, with (bd) or without (d) buffers, and β_1, β_2 and β_3 are parameters to be estimated.

For each period of the timber planning scenarios that were considered in this study, the total proportion of area disturbed by harvest was estimated by summing the proportions calculated with eq. 2.1 and weighed by the operating area's terrestrial coverage. Two levels of disturbance were then considered to provide a disturbance range. The minimum level only considered the area disturbed by harvest. The maximum level added the expected proportion of area burned in 40 years ($1 - e^{-40 \text{ x } 0.00251} = 0.096$).

2.05 Results

In 2007, the Chief Forester's Office of Quebec (CFOQ) recommended a timber supply of 3.45 Mm^3 for the period 2008-2013 in the study area (Tembec 2007). The periodic timber supply level for the BAU baseline scenario was 9% higher than the CFOQ's recommendation, mostly because different planning techniques were used (optimization *vs*. simulation). Under the EBM scenario, the periodic supply level of the baseline scenario was 14% higher than the CFOQ's recommendation. The difference between BAU and EBM scenarios is mostly due to the possibility of partial cutting in EBM. The possibility, in EBM, of converting black spruce to jack pine through planting (Table 2.1) had only a slight effect (1%) on timber supply. Diversification of harvesting strategies in the EBM scenario led to diversification of harvesting ages. For the successional pathway that was dominated by black spruce, the mean harvest age scheduled by optimization of the timber planning problem was 121 years for CLAAG, 164 years for CPPTM, and 100 years for the irregular shelterwood cut. Conversion of black spruce stands to either jack pine or trembling aspen was planned to occur, on average, between 155 and 170 years from now. Contrary to this result, mean harvest age varied only between 96 and 123 years in the alternative BAU baseline scenario.

Table 2.1 Silvicultural strategies used for ecosystem-based management scenarios

		Natural stand dynamics		Ecosystem-based management	
Dominant species	Initial cohort	Disturbance	Resulting cohort	Silvicultural strategy	Resulting cohort
Black spruce	1	Absence	2	No intervention	2
	2, 3	Gap dynamics	3	Partial cut	3
	1, 2, 3	Fire	1	CLAAG	1
				Clear cut + scarification	1(Trembling aspen)
				Clear cut + scarification + plantation	1(Jack pine)
Jack pine	1	Absence	2	No intervention	2
	1, 2	Fire	1	Clear cut + plantation	1
	2	Gap dynamics	3*	Partial cut	3*
Trembling aspen	1	Absence	2	No intervention	2
	1, 2	Fire	1	Clear cut	1
	2	Gap dynamics	3*	Partial cut	3*

*Cohort 3 is always dominated by black spruce.

For baseline scenarios (BAU and EBM), the critical period (when available volume for harvest equals the harvested volume) lay between 45 and 50 years from now. A shadow price analysis confirmed that the first 50 years provide the greatest constraint for both scenarios in terms of area available for harvest (resource availability constraint, Figure 2.2), but constraints were more restrictive during the first seven 5-year periods for the EBM baseline scenario. This was due to the cohort proportion constraints that indirectly affected the area available for harvest. Periodic timber supply was only reduced by 1% when a targeted forest structure was required, but a solution could only be obtained when the cohort proportion constraints were applied from the 11th period onward, and not before.

The first element considered for sensitivity analyses was the proportion of territory under different cohorts. The first scenario considered a shorter fire cycle that required proportions of 70%, 19%, and 11% under cohorts 1, 2, and 3, respectively. Such a target could be met at the 11th 5-year period of simulation, with only a slight loss in periodic timber supply (4%). The second scenario corresponded to a longer fire cycle, involving 58%, 22%, and 20% of the territory under cohorts 1, 2, and 3, respectively. Again, such a target was met at the 11th period of simulation, but a more severe drop was incurred in periodic timber supply (28%). Periodic timber supply thus decreases when the target age structure departs from the present age class distribution, but a longer fire cycle has a higher cost because it requires a higher proportion of cohort 3 stands.

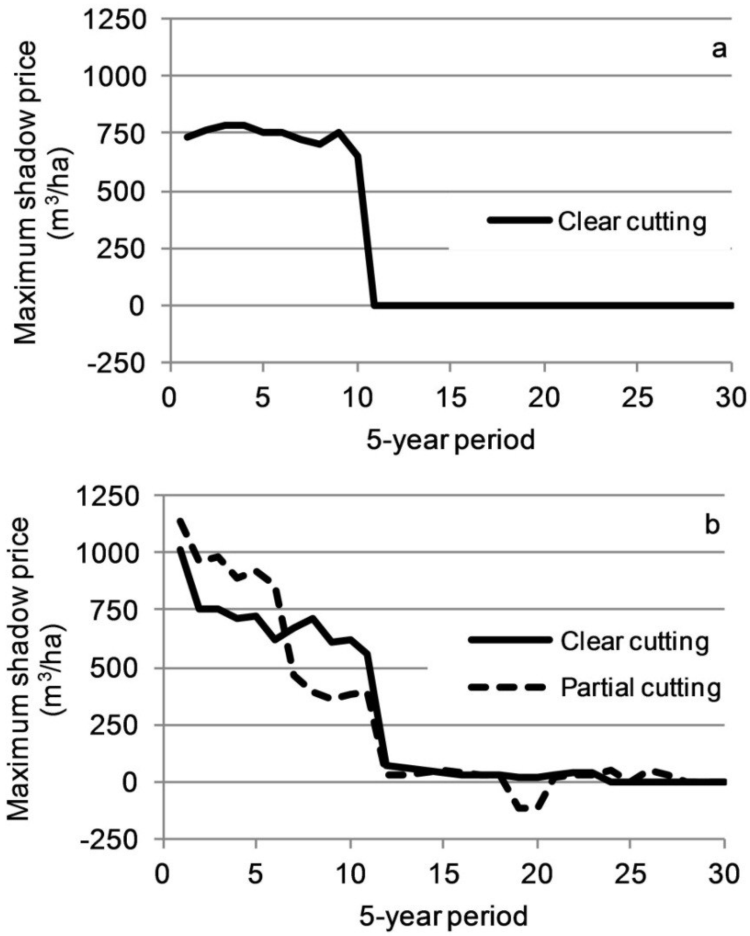

Figure 2.2 Maximum values of shadow prices for clear-cutting and partial cutting harvest by period under (a) business-as-usual (BAU) scenario and (b) ecosystem-based management (EBM) scenario.

The second element considered in the sensitivity analyses was harvest agglomeration. It was emulated by requiring that 30%, 50%, or 70% of the productive area was eligible for harvest before opening an operating area to clear-cutting. This resulted in a sharp drop in periodic timber supply for both BAU and EBM scenarios (Table 2.2), but the drop remained consistently lower for the EBM scenario, by 40% to 70%. This difference was caused by the possibility, only in the EBM scenario, of partial cutting outside of the operating areas that had been opened for clear-cutting.

Indeed, both scenarios behaved similarly when partial cutting was constrained in the EBM scenarios to the operating areas that had been opened for clear-cutting (Table 2.2). With these considerable drops in planned harvest, clear-cutting rates tended to adjust to the difference between historical and current burn rates between 30% and 50% (Figure 2.3). Applying a harvest agglomeration constraint implies that 43% to 67% of the productive area of the forest is closed to clear-cutting for about 50 years (Figure 2.4). This could be viewed as a restoration strategy, as the percentage of area eligible for clear-cutting during the first planning period is well-correlated with recent harvest history in each operating area (Figure 2.5; $r = -0.854$, $p < 0.05$).

Inclusion of a social criterion (excluding buffer zones around roads and water bodies from the productive land base to protect areas of aboriginal interest) resulted in an 11% decrease in periodic timber supply in both the BAU and EBM scenarios. When only buffers on all-season roads and water bodies > 0.5 ha were excluded, the periodic timber supply level dropped by 3 to 4% relative to BAU and EBM baseline scenarios.

Table 2.2 Change in timber supply level (%) relative to BAU or EBM baseline scenario under different spatial constraints and cutting types, and proportional volume harvested by clear-cutting under the EBM scenario.

Type of harvest spatially restricted	Proportion of eligible stands required before opening an operating area to harvest	Change (%) relative to the BAU baseline scenario (3.76 Mm3/period)	Change (%) relative to the EBM baseline scenario (3.93 Mm3/period)	% of harvest volume realized with clear cut in EBM scenarios
None	0% [a]	baseline scenario	baseline scenario	81%
Clear-cutting	30%	-11%	-3%	82%
	50%	-36%	-11%	78%
	70%	-57%	-22%	71%
Clear-cutting and partial cutting	30%	-	-8%	88%
	50%	-	-30%	83%
	70%	-	-58%	81%

[a]Baseline scenarios

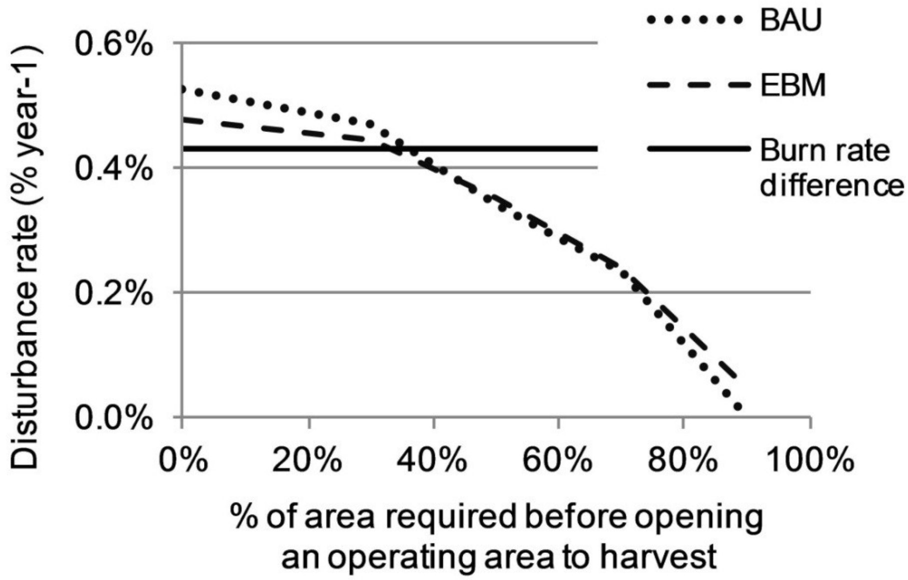

Figure 2.3 Variation of the clear-cutting rate of business-as-usual (BAU) and ecosystem-based management (EBM) scenarios as a function of the minimum percentage of productive area for an operating area that is required to be eligible for clear-cutting before opening it. The horizontal black line shows the difference between present and historical fire disturbance rates in the study area.

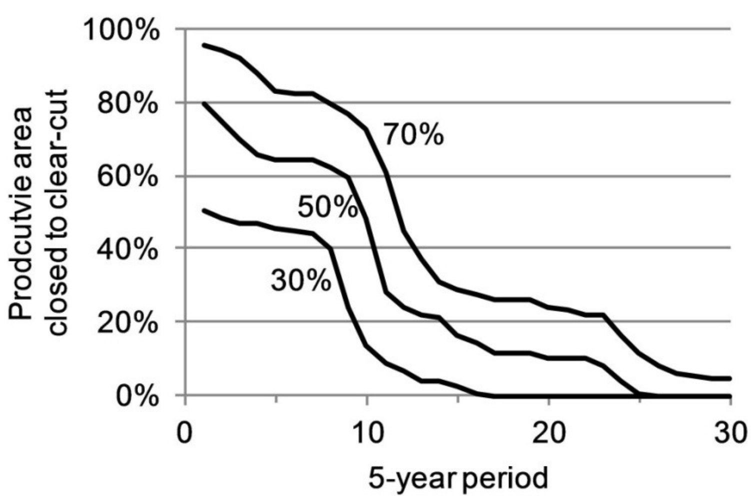

Figure 2.4 Proportion of productive area closed to clear-cutting as a function of three minimum percentages of the productive area for an operating area to be eligible for clear-cutting before opening it.

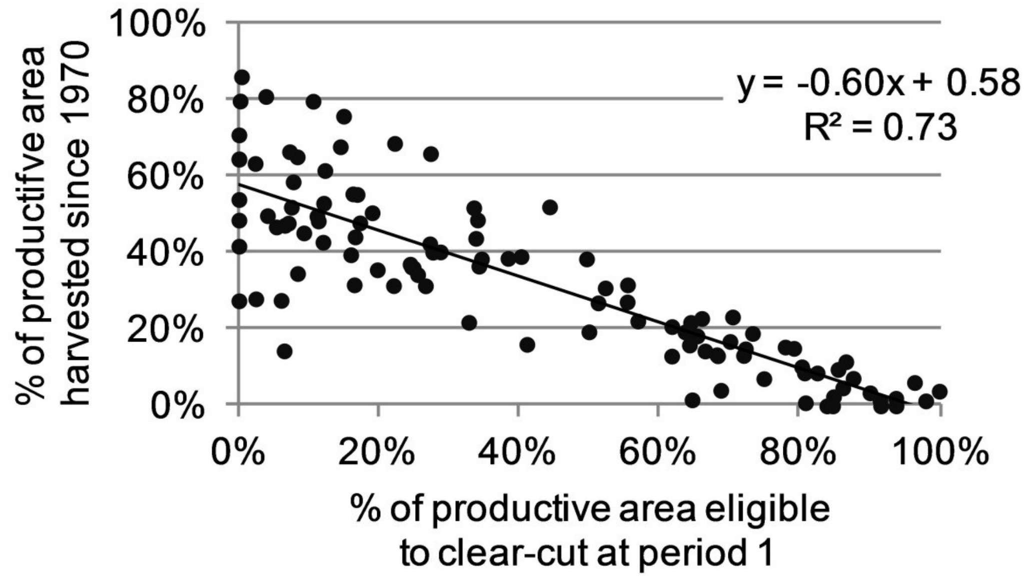

Figure 2.5 Relationship between the proportion of productive area harvested since 1970 and the productive area eligible to be clear-cut at period 1 in 107 operating areas of the 085-51 Forest Management Unit.

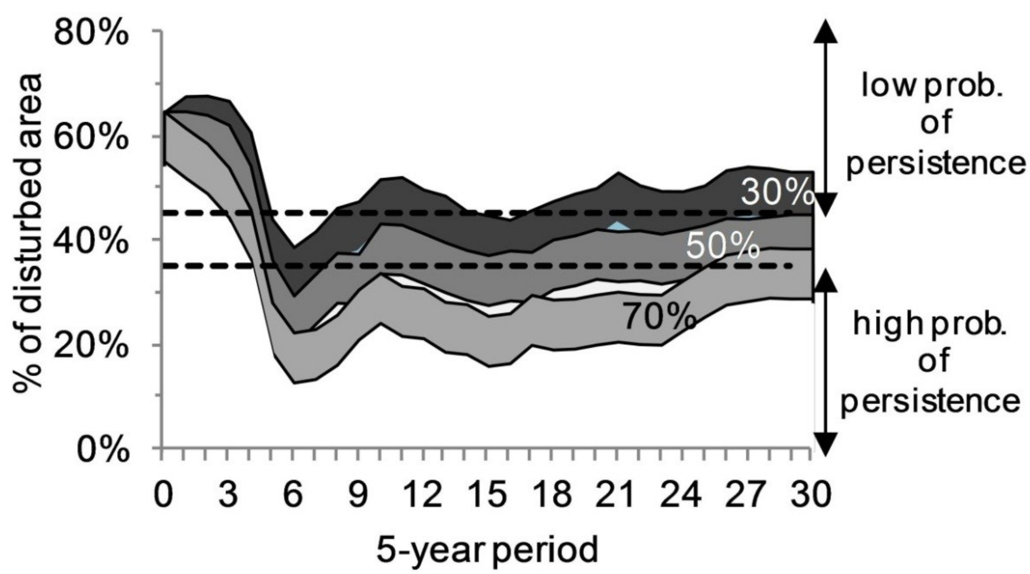

Figure 2.6 Proportion of disturbed area (total area less than 40-years-old, taking into account either harvest disturbance or cumulated harvest and fire disturbances). Three EBM scenarios are considered, constraining clear-cutting in operating areas that had more than 30%, 50%, or 70% of their productive area that was eligible for harvest. Probability of persistence for woodland caribou follows Environment Canada estimates (2011, their Table 9).

In our sensitivity analyses, we assumed that a coarse-filter approach constraining the clear-cutting rate to the difference between the historical and current burn rates could satisfy the habitat needs of woodland caribou. All scenarios in which harvests were agglomerated tended to lower the risk associated with a low probability of persistence of a woodland caribou population (Figure 2.6). These scenarios had their peak area proportion under cumulated disturbances during the first 5 periods of the planning horizon. This proportion dropped below 45% after 25 years, a threshold below which the probability of woodland caribou persistence is higher than 40% (Environment Canada 2011). It even fell below 35% for the two most constrained scenarios (50% and 70%, Figure 2.6), a level where woodland caribou populations are considered self-sustaining (Environment Canada 2011). However, this favorable level for caribou is not maintained throughout the planning horizon (Figure 2.6).

2.06 Discussion

Gauthier *et al.* (2004) evaluated the possibility of achieving objectives of landscape level cohort proportions based on the model of Bergeron *et al.* (2002) in the same forest management unit. They found that such objectives could be achieved without significantly reducing the periodic timber supply. Our study confirms this result but also highlights the fact that the next 50 years will be critical for establishing a multi-cohort management strategy in this study area. The targeted forest age structure could be reached after 55 years (11^{th} simulation period), but not before. Consequently, harvesting activities that are realized in the near future will have a critical effect on the probability of success in establishing a multi-cohort management strategy (Figure 2.2b). Yet, this can be achieved without reducing the timber supply level.

Harvest spatialization placed a severe constraint on timber supply in both BAU and EBM scenarios. The diverse silvicultural practices that were included in the EBM scenario gave it a comparative advantage over BAU. Although the use of partial cuts helped compensate for harvest spatialization in the EBM scenario, the consequences of their implementation on the growth and mortality of residual trees should be evaluated to determine the long-term effects of a multi-cohort management strategy. Commercially important boreal species respond positively to reduced competition through partial cutting (Thorpe and Thomas 2007). However, post-harvest mortality of residual trees remains a concern, since post-harvest mortality increases with the proportion of individuals harvested (Thorpe and Thomas 2007). For more than a decade, experimental partial cut designs have been implemented in Quebec (Ruel *et al.* 2007, Fenton *et al.* 2009) and elsewhere in the Canadian boreal forest (e.g., Thorpe and Thomas 2007). These will help better define the operability of such treatments, but they have only started to provide information regarding the mortality and growth of residual stands. We did not consider the economic impact of dispersing partial cutting, and this is probably not the best alternative. Cost-benefit analysis of such a partial cutting strategy was beyond the scope of this study but should point to other scenarios of spatial dispersion or aggregation of partial cuts.

Larger and more aggregated harvesting blocks should create landscape patterns that can accommodate edge-sensitive wildlife species (Rempel and Kaufmann 2003). Two key strategies were proposed by Belleau *and Légaré* (2009) to protect such habitats in this study area: (1) using partial cuts that favor the maintenance of spruce-lichen woodlands, and (2) agglomerating harvesting activities to limit habitat fragmentation. Timber supply and habitat conservation objectives often conflict regarding the optimal size of harvesting blocks and the dispersed or aggregated distribution of these blocks (Rempel and Kaufman 2003, Tittler *et al.* 2012). Hence, we tested the effects of these two strategies. Our simulations showed that the proportion of the area under clear-cutting could be reduced to attain a disturbance rate equivalent to the difference between historical and current burn rates, a situation probably more favourable to species such as woodland caribou (Hovington *et al.* 2010). In contrast to Rempel and Kaufman (2003), we did not consider spatial adjacency constraints in our planning models. Belleau *et al*. (2007), in a simulation study including our study area, observed that individual fires often agglomerate. They suggested that harvesting blocks could be agglomerated, as long as the maximum proportion of first cohort stands was respected. Harvest agglomeration would lead to a 3% to 11% diminution of the planned harvest and necessitate a 50-year restoration period for 43% to 67% of the productive area. Partial cutting compensated for part of the loss of periodic timber supply caused by harvest agglomeration, but partial cuts had to be spread over the territory, thereby potentially increasing habitat fragmentation by roads. Functional habitat loss for woodland caribou has been shown to occur within 750 m to 1250 m from these linear features (Leblond *et al*. 2011), and their negative effect was not taken into account in the present study. Otherwise, limiting partial cutting to operating areas eligible for clear-cutting aggravated the drop in periodic timber supply (Table 2.2). Given the importance of partial cutting in maintaining timber supply, different agglomeration strategies of partial cutting (e.g., Bergeron *et al.* 2004) should be considered in further analyses.

The habitat requirement of woodland caribou was used to evaluate whether or not the coarse-filter approach (maintaining forest age structure, agglomerating cut blocks, and limiting disturbance rate) could induce a disturbance rate capable of maintaining woodland caribou within the study area. The EBM strategies did seem to improve the likelihood of maintaining woodland caribou, especially when the percentage of stands eligible for harvest before opening an operating area to clear-cutting was higher than 30% (Figure 2.6, 50% or 70%). This effect was only temporary, however, but this was due to the limited adequacy of the approach used in the timber supply problem of agglomerating harvest with opening curves. This approach was used to avoid requiring optimization techniques other than linear programming that are much more difficult to solve (Weintraub and Murray 2006) or post-processing the results with spatial blocking (Rempel and Kaufmann

2003). In any case, this result points to the need for more sophisticated strategies of opening and closing operating areas for harvest, and an in-depth analysis of their impact on timber supply.

Taking aboriginal values into account has become a central issue in Canadian forest management (Saint-Arnaud *et al.* 2009, Germain and Asselin 2010). When management decisions are taken regarding state-controlled resources (e.g., forests), their effects on the general public should be considered (Brunson 1996). Aboriginal communities depend on forests for the goods and services that they provide and for cultural activities. Thus, management decisions have obvious impacts on their livelihoods. While it is recognized that the needs of local communities should receive the highest priority when making forest management decisions (Shindler *et al.* 1993, Brunson and Steel 1994), taking into account the needs of the Pikogan Anicinapek community resulted in a 4 to 11% decrease in planned timber supply level. Identical impacts of the social criterion in both BAU and EBM scenarios can be explained by the similar way in which this issue was taken into account in both models (i.e., reducing the timber production area). It should also be understood that each aboriginal community has its own relationship with the land (Germain and Asselin 2010) and, consequently, the effects incurred on timber supply by taking their needs into account will vary considerably. This effect on timber supply level could potentially be reduced by allowing partial cutting in buffer zones that are reserved for aboriginal activities, since aboriginal people often prefer selective or retention harvesting over clear-cutting, as the former maintain ecological and cultural functions of the forests (Larouche 2008, Saint-Arnaud *et al.* 2009).

Taking into account all key policies simultaneously led to reductions of 7% (criterion of harvest agglomeration fixed at 30%, buffer zones on all-season roads and water bodies greater than 0.5 ha excluded from the productive area) to 21% (criterion of agglomeration fixed at 50%, all buffer zones excluded). In absolute terms, such reductions are equivalent to periodic timber supplies of 3.11 to 3.66 Mm^3 $period^{-1}$. The CFOQ (2011) expects a timber supply of 2.93 Mm^3 $period^{-1}$ for 2013-2018, to account for the effects of new forest legislation (Government of Quebec 2010). For the purposes of comparison, this value has to be adjusted to remove the effect of creating 59 000 ha of protected areas within the forest management unit between 2007 and 2009 (3.10 Mm^3 $period^{-1}$). These values are lower than the volume harvested between 2000 and 2005 (3.52 Mm^3 $period^{-1}$), but are substantially higher than the volume harvested between 2005 and 2010 (1.98 Mm^3 $period^{-1}$, Louis Dumas, pers. comm. 2011).

2.07 Conclusion

Conventional management (BAU) and ecosystem-based management (EBM) are two forest management practices used in Canada. The main difference is that, under EBM, partial cutting is used in addition to clear-cutting and harvesting activities are spatialized to reproduce natural forest dynamics more closely.

Consideration of Aboriginal values and maintenance of coarse-filter indicators of biodiversity reduced the periodic timber supply. However, this decrease was lower with EBM thanks to the availability of several silvicultural strategies. An economic analysis of partial cuts and plantations should be undertaken, as they greatly influence timber supply solutions, affect operational costs, and have social impacts. The effects of harvest agglomeration, scattering of partial cuts and their associated road networks on habitat availability for woodland caribou should also be further investigated.

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Chapter 3

Adaptability of an ecosystem-based management strategy to climate induced increase in the fire burn rate and growth anomalies in an eastern Canadian boreal forest

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Manuscript under preparation

3.01 Résumé

L'adaptabilité d'une stratégie daménagement écosystemique sur l'impact potentiel des changements climatiques sur l'approvisionnement du bois a été évaluée en tenant compte du rôle du climat sur la croissance des forêts et des feux dans une unité d'aménagement forestier dans la zone boréale de l'est du Canada. Un modèle de l'indice de qualité de station sensible au climat a été calibré pour trois espèces commerciales, l'épinette noire (*Picea mariana* (Mill) BSP), le pin gris (*Pinus banksiana* Lamb.) et le peuplier faux-tremble (*Populus tremuloides* Michx.). Ce modèle a été utilisé pour projeter l'évolution du volume marchand de ces espèces dans le cadre d'un calcul de possibilité en visant soit la normalisation des forêts, soit la mise en œuvre d'une stratégie d'aménagement écosystémique selon deux scénarios climatiques. Le taux de brûlage actuel ainsi que les taux de brûlage projetés sous les scénarios climatiques futurs ont été intégrés dans les modèles d'approvisionnement du bois. Un outil d'aide à la décision a aussi été conçu afin d'évaluer les risques encourus.

Les scénarios climatiques projetés semblent causer une réduction de l'approvisionnement périodique en bois qui pourrait atteindre 79%. Une interaction entre l'activité des feux et la croissance des peuplements en fonction des changements climatiques a aussi été observée. La mise en œuvre de l'approche écosystémique semble être une meilleure stratégie d'aménagement dans le contexte de différents scénarios climatiques, car elle permettait de maintenir un âge moyen plus élevé des peuplements sur pied, une plus faible proportion de superficies dans les jeunes classes d'âge ainsi qu'un plus fort niveau d'approvisionnement. Toutefois, d'autres stratégies d'adaptation semblent également être requises tout en changeant de stratégie d'aménagement afin de tenir compte des scénarios climatiques projetés et leur impacts potentiels sur la dynamique de croissance et le régime de feu.

Mots clés : Approvisionnement du bois, rendement, changements climatiques, feu, adaptation, forêt boréale

3.02 Abstract

Adaptability of an ecosystem-based management to the potential impact of climate change was evaluated considering the role of climate on forest growth and fire activity in a forest management unit in the boreal zone of eastern Canada. A climate sensitive growth index model was calibrated for three commercial species (black spruce (*Picea mariana* (Mill) B.S.P.), jack pine (*Pinus banksiana* Lamb.) and trembling aspen (*Populus tremuloides* Michx.). The model was used to project the evolution of merchantable volume of these species over time under conventional sustained yield timber production and ecosystem based management (EBM) under two climate scenarios. Current burn rate and burn rates under future climate scenarios were also considered in the timber supply model. A risk assessment tool was presented as a decision support.

Under the projected climate scenarios, the periodic timber supply responded with long-term reduction by up to 79%. An interaction between the response of growth and fire to the projected climate scenarios was also revealed. EBM emerged to be a better management strategy in the context of projected climate scenarios as it maintained a higher mean standing inventory age over time and a lower proportion of area under younger age classes in addition to the higher level of periodic timber supply. However, further adaptation strategies may be needed in addition to switching management strategy to EBM to deal with the projected climate scenarios and their potential impact on growth and disturbance dynamics.

Key words: Timber supply analysis, growth and yield, climate change, fire, adaptation, boreal forest

3.03 Introduction

The Intergovernmental Panel on Climate Change (IPCC 2007) projects a global mean temperature rise between 1.4˚C and 5.8˚C by the end of this century in addition to the increase of 0.74 ˚C that has been observed between 1906 and 2005. During this period, a pronounced long-term trend in the amount of precipitation with regional variability has also been observed (Trenberth et al. 2007). Eastern North America is one of the regions to see the greatest warming and a significant increase in amount of precipitation during this period (IPCC 2007). In the south-western part of the boreal forest in Quebec, for example, an increase in temperature of 0.5 to 1.2˚C was recorded between 1960 and 2003 (Yagouti et al. 2006). With such an increase in temperature in the region, recent climate simulations indicate the possibility of drought and increasing forest fire activity over this century (Bergeron et al. 2006; Flannigan et al. 2005).

Forest ecosystems are shaped by environmental conditions such as climate, soils and natural disturbances (Seymour and Hunter 1999; Stocks et al. 2003, Flannigan et al. 2005). Climate directly affects ecosystem functions such as species growth, reproduction and migration whereas it indirectly controls natural disturbance regimes such as fire, insect outbreaks and diseases (Bergeron et al. 2010; Girardin et al. 2008).

A tree growth rate is directly determined by temperature, soil moisture and radiant energy. Changes in these variables will have repercussion on forest stand development and eventually on forest yield (Papadopol, 2000). There is a growing body of evidence on non-intuitive responses of ecosystems to climate change. These include a decline in tree ring width in some northern locations of Alaska (d'Arrigo et al. 2008), a decline in the boreal forest productivity in western Canada (Goetz et al. 2005) and carbon neutrality of some old black spruce forest stands in Québec (Bergeron et al. 2007). A rising temperature increased the productivity of boreal forests of northern latitudes in the 1980s and 1990s but more recently, drought conditions have done the opposite (Bunn et al. 2007).

A sustained even flow of merchantable volume is a central objective of forest management (Bettinger et al. 2009). Timber supply analyses (TSA) estimate future long-term harvest of such volume using growth and yield models to satisfy often conflicting objectives of timber harvest and other non-timber requirements. Different jurisdictions use different growth and yield models to estimate future harvest. For example, in the province of Ontario, normal yield tables (Plonski 1981) and its variants are used to forecast future volume evolution for the majority of commercial species (Sharma et al. 2008). In the province of Alberta, a stand level growth and yield projection model called growth and yield projection system (GYPSY) is used to forecast future evolution of volume in commercial forests (Huang et al. 2009). In British Columbia, tree and stand simulator (TASS) and variable density yield prediction (VDYP) models are used for pure, even aged conifer forests and mixed, uneven aged forests respectively (Bettinger et al. 2009). TASS is a distance dependent individual tree growth model and spatially explicit tree map is required to simulate tree growth. In the province of Quebec, a stand level model (Pothier and Savard 1998) has been used to project future growth and yield at the scale of the province for last several years. All the models (discussed here) except TASS are stand level models with similar input variables such as site index, stand density, basal area, and age of stand dominant trees. These models assume past growth condition will apply for the future.

Forest management is sustainable when managed ecosystems remain within their natural range of variability (Haeussler and Kneeshaw 2003; Landres et al. 1999). Such range of variability is determined by the frequency and severity of disturbances. Within an ecosystem, there exists a strong interaction between succession trajectory and type and severity of the disturbances (Desponts et al. 2004). These factors are not taken into account quantitatively while elaborating forest management strategies in Quebec, Canada (Coulombe Commission 2004). Two main reasons for this are: 1) tools used in Canada for timber supply analyses are designed to establish a desired condition of forest in a deterministic way and do not take into account the random nature of fire, forest succession and uncertainty in climate and 2) forest management models are mostly at the scale of stands (Monserud 2003; Raulier et al. 2004) and not at the scale of landscapes. One way of dealing with the uncertainty stemmed from fire or succession is to take into account fire activity while elaborating periodic timber supply (e. g., Savage et al. 2010; Armstrong 2004; Martell 1994; Van Wagner 1983). Integrating a stochastic fire model into a harvest-scheduling model may give a more realistic picture of the impact of fire on periodic timber supply. However, taking fire frequency in a deterministic way also gives an insight on impact of fire on planned forest management strategy (Boychuk and Martell 1996; Davis et al. 2001).

Traditional sustained yield timber management is blamed for the loss of the resilience of a forest against the disturbances by simplifying the ecosystem (Seymour and Hunter 1999; Haeussler and Kneesahw 2003). Resilience is the capacity of a forest to resist change or recover following a disturbance (Thompson et al. 2011). It is dependent on the biodiversity at multiple scales and on legacies following disturbances (Chapin III et al. 2004). Since forests with a higher diversity are more resilient to climate change (Thompson 2011; Chapin III et al. 2010; Chapin III et al. 2004), management strategies that close the ecological differences between an unmanaged and a managed forest (e g., Bergeron et al. 2002) should be favorable in the context of climate change. Ecosystem-based management, which is designed to implement harvesting activities at the landscape level with landscape level variables, can provide such an avenue (Harvey et al. 2002; Gauthier et al. 2009a).

In this study, we used a meta-modeling approach to account for the response of yield and fire activities to climate forcing and their impact on periodic timber supply. A meta-model is a model built from other models that keeps the essences of those models intact (Urban et al. 1999). The objective was to quantify the response of growth and disturbance of a boreal ecosystem to the projected future climate scenarios. First, we quantified the potential impact of climate change on the growth and yield of commercial boreal species typical of eastern Canada (black spruce, jack pine and trembling aspen) and the consequences of such changes on timber supply. An empirical growth and yield model was made climate sensitive by relating its explanatory variables (site index and relative density index) to climate in order to provide the ability to build climate-sensitive yield curves. Then, we determined the impact of fire frequency on timber supply by integrating possible future fire scenarios in harvest scheduling models. We estimated the progressive impact of fire at five year time step through Monte-Carlo simulation following a 5-year rolling planning exercise until the year 2100. Finally, we compared the results of timber supply simulations under conventional sustained yield timber production (business-as-usual, BAU) and ecosystem-based management (EBM) strategies against ecological criteria to determine the adaptability of EBM in light of climate induced increase in fire frequency and growth anomalies. We first hypothesized that a changing climate will have a positive impact on growth and productivity of the boreal forest. Our second hypothesis was that ecosystem-based management has higher adaptability to the impacts of climate change than a BAU management.

3.04 **Methodology**

3.04.01 Study area

The study area is located between 48°50'N and 50°09'N latitude, and 78°05'W and 79°31'W longitude (Figure 3.1). Total area of this landscape is about 1 million hectares. It is designated as the forest management unit (FMU) 085-51 by the government of Quebec for regulatory and administrative purposes. Approximately half of the territory is considered productive. A forest stand is productive if it is capable of producing 50 m^3 ha^{-1} of timber over a rotation. Tembec Inc. (www.tembec.com) is responsible for the management of the FMU. The FMU is located within the continuous boreal forest zone of Quebec (MRN 2003) in the clay belt at the border between the provinces of Quebec and Ontario (Figure 3.1). Almost 85% of the FMU lies in the black spruce–feather moss bioclimatic sub-domain (Robitaille and Saucier 1998). Black spruce dominates the canopy normally as pure stands but is occasionally accompanied by trembling aspen (*Populus tremuloides* Michx.), balsam fir (*Abies balsamea* (L.) Mill.), white birch (*Betula papyrifera* Marsh.) or balsam poplar (*Populus balsamifera* L.). The southern portion of the FMU lies within the balsam fir-white birch bioclimatic sub-domain which is also referred to as mixed-wood ecological region (Bergeron et al. 2004). Soil deposit in the forest area is predominantly clay (60%), followed by Cochrane till (22%) and organic deposits (18%).

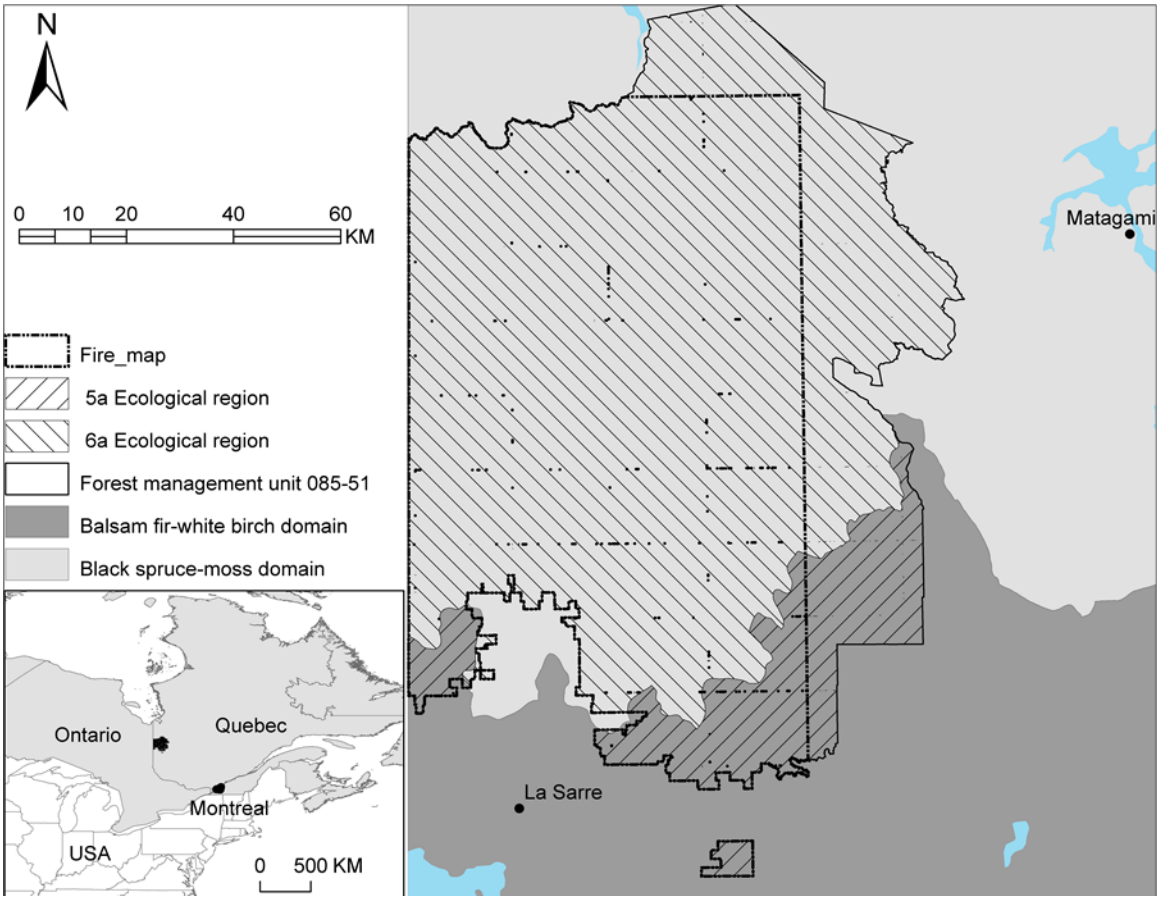

Figure 3.1 Study area (085-51 forest management unit in western Quebec, Canada). 5a is the Abitibi lowland ecological region and 6a is the Lake Matagami lowland ecological region

The study area has cold-dry winters and warm-moist summers (Bergeron et al. 2004). The mean annual temperature varies between -2.5°C to 0°C with a mean January temperature and a mean July temperature of 20^{0}C and 17^{0}C respectively. The length of the growing season is 150-160 days (above 5^{0}C). The sum of growing degree days is 1300 to 1400 distributed between May and September, and total precipitation is 700-800 mm (Robitaille and Saucier 1998).

Fire dominates the disturbance regime. Gap-dynamics and wind-throw also play an important role as the current fire cycle (i. e., 400 years, Bergeron et al. 2004) is longer than the longevity of the tree species present in the area. The successional dynamics of this area has been simplified into three pathways defined by the population dynamics of black spruce, jack pine, or trembling aspen (Simard et al. 2009; Lecomte et al. 2006; Gauthier et al. 2004; Nguyen-Xuan 2002). Jack pine and trembling aspen are found on stands with better drainage condition, but black spruce largely dominates the other areas with unfavorable drainage conditions (Belleau and Légaré 2009). These pathways are further subdivided into two to three developmental stages or cohorts depending on the species that leads the pathway or the stand vertical structure (Gauthier et al. 2004). The first cohort (cohort 1), which is dense and has little canopy differentiation, is composed of trees established right after a stand-replacing fire. In the absence of severe disturbances, cohort 1 develops into a second tree cohort characterized by an irregular canopy. At this stage, pioneer species such as trembling aspen and jack pine are gradually replaced by black spruce. A prolonged absence of a stand-reinitiating disturbance leads to the continual evolution of stands into a third cohort characterized by open and uneven-aged stands of a well-developed vertical structure. At this stage, species established during cohorts 1 and 2 are usually replaced by black spruce (Belleau and Légaré 2009).

3.04.02 Forest growth and yield projection

(i) Land classification

Temporary sample plot (TSP) data (forest inventory data) from Quebec (1992-2002) and timber supply analysis data of our study area for the period 2008-2013 were provided by Ministère des Ressources naturelles, Québec (MRN) for the present study. A TSP is a circular plot of 400 m^2 where species and diameter at breast height (DBH) are recorded for all stems with a DBH greater than 9 cm. Age and height of few selected dominant/co-dominant trees are also recorded during the inventory. Saplings (DBH ≤ 9cm) are counted by 2-cm diameter classes in a sub-plot of 40 m^2 located at the centre of the TSP.

The MRN regrouped stand polygons of similar cover type, cover density, cover height, age classes and ecological types into inventory strata. There were 1509 inventory strata in our study area. To simplify the analysis, we regrouped these inventory strata into 328 management strata through cluster analysis when TSP data was available or through similarity in stand cartographic attributes (Dhital et al. 2013). Stand development types or cohorts were assigned to each management stratum following a cohort classification tree (Nguyen-Xuan 2002, for the cohort classification details, refer to chapter 2). Yield curves for each stratum were assigned as an area-weighted average of the yield curves of inventory strata represented within the management strata. These curves were built by the MRN with the stand yield model of Pothier and Savard (1998).

We aggregated the species available in our study area into three species groups (intolerant hardwood, TA; intolerant softwood, JP and tolerant softwood, BS) lead by trembling aspen, jack pine and black spruce respectively. To reflect the change in cohort number, one key each for species groups dominated by TA and JP was developed based on the proportion of volume of each species and age class. For the species group dominated by BS, a logistic model was developed based on total volume and stand age. More details on strata regrouping as well as cohort assignment of each stratum and yield curve are provided by Dhital et al. (2013).

(ii) Integrating projected climate scenarios into growth and yield projection

To cover a wide spectrum of alternative futures, we used two scenarios of projected changes in greenhouse gas emissions (Nakicenovic et al. 2000). Each scenario represents different development stages in the future such as global demographic, technological, socio-economic and environmental conditions. The storylines describe the relationships between the forces driving greenhouse gas and aerosol emissions and their trajectories over the 21st century. The A2 storyline represents a very heterogeneous world with continuously increasing global population and regionally oriented economic growth that is more fragmented and slower than in other storylines. The B1 storyline represents a convergent world with rapid changes in the economic structures towards a service and information economy, with reductions in material intensity, and the introduction of clean and resource-efficient technologies. A2 and B1 storylines were considered as the worst- and the best-case scenarios respectively. A detailed description of these climate scenarios is found in Nakicenovic et al. (2000, chapter 5). Future climate data for the study area were obtained from the Ouranos Consortium, a public entity of the provincial government of Quebec, Canada which is specialized in adaptation to climate change in the region. The consortium generated regional daily maximum and minimum temperatures and precipitation data from three global circulation models (Canadian general circulation model version 3, Hadley climate model version 2 and ECHAM version 4) for each storyline. In our study area, these scenarios ensemble cover a range of 1.07^{0}C to 3.77^{0}C increase in mean annual temperature and 10% to 25% increase in annual precipitation by 2100 (Hulme and Shread 1999; Plummer et al. 2006).

Site index (SI), relative density index (RDI_{100}) and mean age of plot dominant trees are the key input variables in the yield projection models of Pothier and Savard (1998). SI and RDI_{100} are assumed to remain constant over a rotation. This may not hold, particularly in a changing climate. We replaced SI by a growth index (Lapoint-Garant et al. 2010, Lapointe-Garant 2010 personal communication). This growth index (GI) is a function of climate variable and changes in accordance with the temporal scale of SI (i. e., 50 years).

To integrate the climate variables in the GI, basal area growth equation of NE-TWIGS (Teck and Hilt 1991) was modified as follow:

Eq. 1 $$\Delta BA_{t+1} = GI * (b_1)(1 - \exp[(-1) * (b_2) * dbh_t])$$

where $\Delta BA_{(t+1)}$ is basal area growth at time t + 1, dbh_t is diameter at breast height in time t and b_1 and b_2 are parameters. These parameter values for black spruce, jack pine and trembling aspen were taken from Raulier *et al.* (2004). Climate variables to be integrated in equation 1 were as identified by Huang et al. 2009. Equations 2, 3 and 4 respectively give the estimation of climate functions for black spruce, jack pine and trembling aspen respectively as linear mixed function estimated by using PROC MIXED of SAS (SAS, Cary NC):

Eq. 2
$$GI_{Blackcspruce} = \beta_0 + \beta_1 * T_{\max Fev} + \beta_2 * (T_{\max Fev})^2 + \beta_3 * T_{\min Aug} + \beta_4 * T_{\min Jun-Aug} + \beta_5 * (T_{\min Jun-Aug})^2 + \beta_6 * P_{Sept_{t-1}} + \beta_7 * P_{mai-Aug}$$

Eq. 3
$$GI_{jackpine} = \beta_0 + \beta_1 * GDD + \beta_2 * P_{oct-fev} + \beta_3 * (P_{oct-fev})^2 + \beta_4 * T_{\min April-Aug}$$

Eq. 4
$$GI_{trembling\ aspen} = \beta_0 + \beta_1 * GDD + \beta_2 * (GDD)^2 + \beta_3 * P_{Oct_{t-1}} + \beta_4 * T_{\min Jun} + \beta_5 * (T_{\min Jun})^2 + \beta_6 * T_{\min Fev} + \beta_7 * (T_{\min Fev})^2$$

where T is monthly or periodic temperature (maximum or minimum), P is sum of precipitation in given month or period, t is current year and t - 1 is year before, GDD is annual sum of growing degree days (average of the daily sum of maximum and minimum temperature over a year with cutoff of minimum temperatures being 5^0C) and $\beta_0 - \beta_7$ are species specific parameters and their estimates are given in table 3.1.

We replaced SI in the yield model of Pothier and Savard (1998) with the growth index derived from the above equations for each species (Figure 3.2). Several other studies have also replaced SI with climate sensitive indices (e. g., Ung et al. 2001; Lapointe-Garant et al. 2010; Anyomi et al. 2012).

Table 3.1 Parameters of equations 1, 2 and 3

Equation	β_0	β_1	β_2	β_3	β_4	β_5	β_6	β_7
2	62.796	0.2357	0.01003	0.3663	1.426	-0.129	0.00942	0.00483
3	91.366	-0.237	0.02824	-5E-05	1.194	-	-	-
4	18.742	1.2485	-0.0404	0.00966	0.551	0.044	0.4414	0.0103

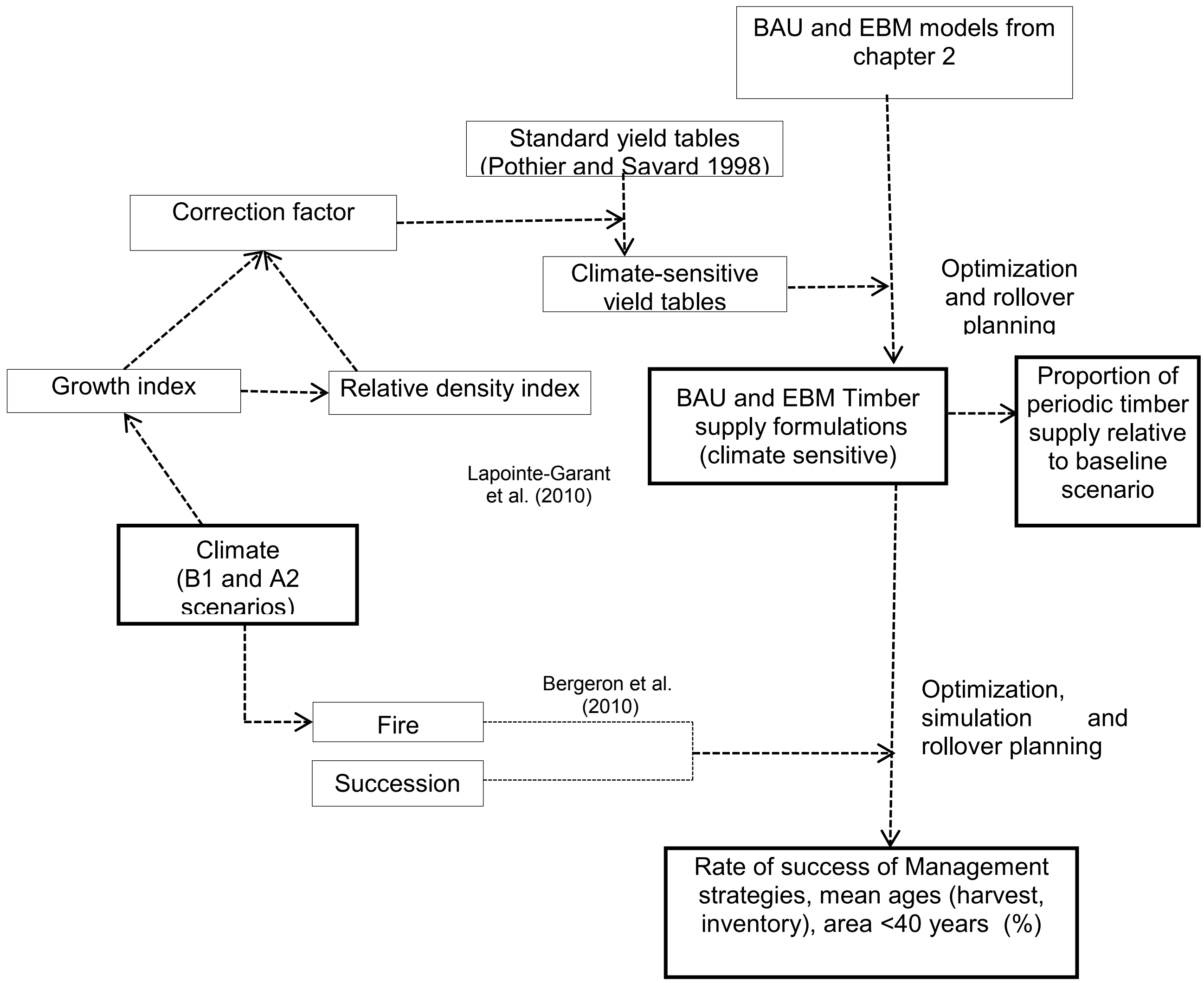

Figure 3.2 Methodological Framework (BAU = business as usual; EBM = ecosystem-based management)

Growth simulations with the A2 and B1 climate scenarios were run using equation 3 over 2000-2100 for the three species groups. Then, fifty-year weighted moving averages of the growth indices were calculated to obtain the surrogate of SI as SI is defined by the mean height of dominant trees at 50 years of age. The average value of such indices simulated under B1 and A2 climate scenarios with the range of uncertainty is given in Figure 3.3. The growth indices of trembling aspen and jack pine seem to slightly benefit from projected climate scenario but black spruce seems to suffer severely due to such changes (Figure 3.3). This reduction in growth of black spruce is attributable to the increase in mean August temperature of previous year (Girardin et al. 2011; Girardin et al. 2006) and similar increase in mean July temperature of previous year (Lapointe-Garant et al. 2010). It is particularly a concern because such temperature is expected to increase in the future (IPCC 2007).

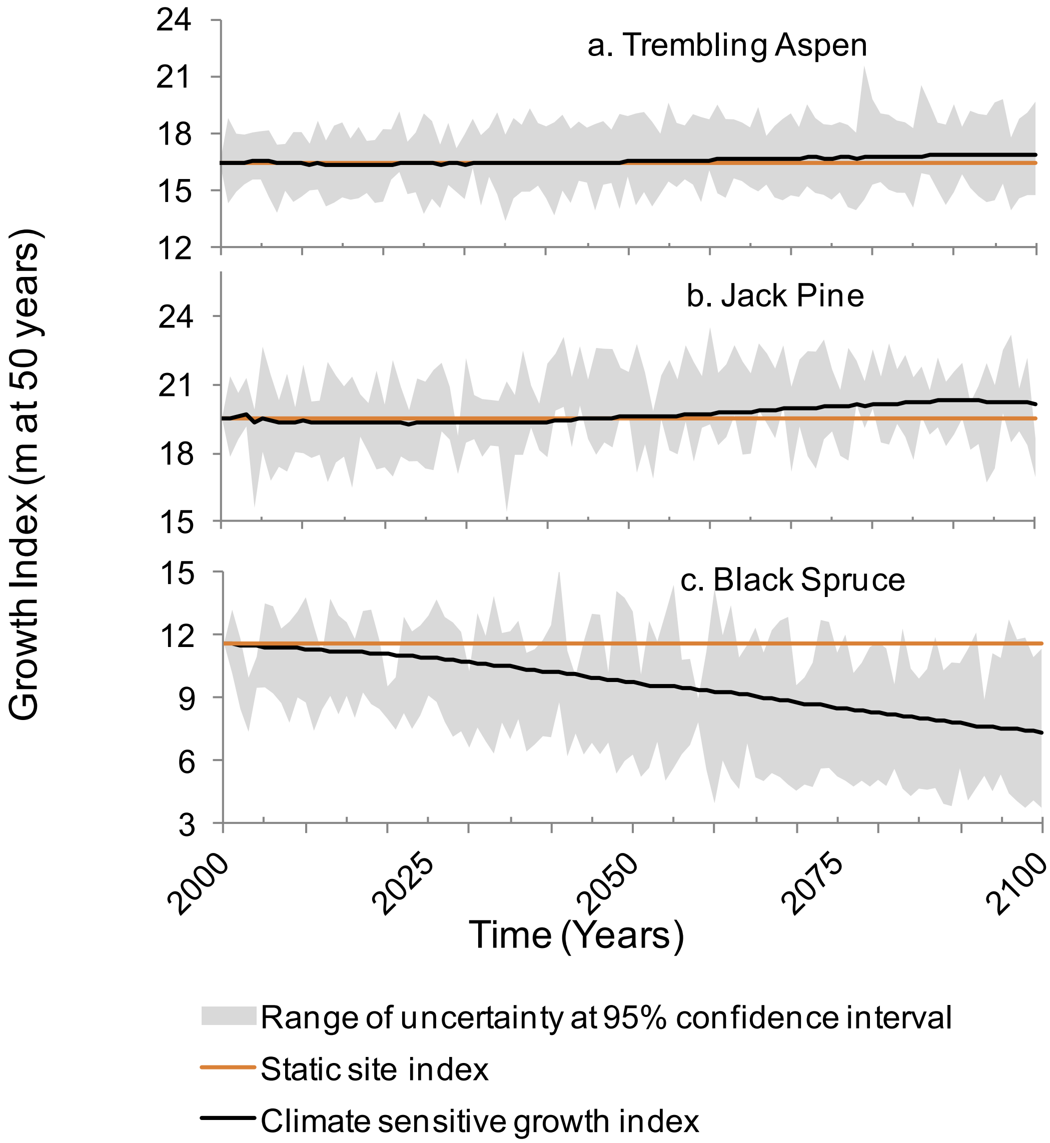

Figure 3.3 Fifty year moving averages of climate sensitive growth indices (dark lines) calibrated from climate and tree ring data in the study area under A2 and B1 projected climate scenarios.

RDI is one of the major sources of error in yield projection by Pothier and Savard (1998) model (Pothier et al. 2004; Dhital et al. submitted). To improve the yield projections, Lapointe-Garant (pers. comm. 2010) rebuilt their RDI_{100} equation by making it dependent to climate sensitive growth index using temporary sample plot data of the study area from the last forest inventory (Equation 3.5; Figure 3.2);

Eq. (3.5) $RDI_{variable} = \exp((-b_1 + b_2 * GI) + (b_3 + b_4 * GI) * RDI_{100}) * 1/age$

where $RDI_{variable}$ is a species-specific climate sensitive RDI_{100}, GI is a climate sensitive growth index derived for the study area, RDI_{100} is relative density index at the age of 100, age is the mean age of selected trees in the plot, b_1-b_4 are parameters to be estimated. PROC MIXED (SAS, Cary, North Carolina, USA) was used for parameter estimates and values are given in Table 3.2:

Table 3.2 Species specific parameter values for equation 3.1.

Species	Parameters			
	B_1	B_2	B_3	B_4
Black spruce	-214.2	8.8	262.1	-13.2
Trembling aspen	-152.6	4.9	162.9	-5.66
Jack pine	-187.8	8.1	215	-10.5

Now, SI and RDI_{100} change through time. The impact of climate change on yield depends not only on time but also on stand age (eq. 3.5) which requires building several yield curves for each stratum. To reduce the volume of work without avoiding the essence of the approach, we first examined how the ratios between periodic annual increments in volume calculated from yield curves with and without climate change (i.e., using variable or fixed SI and RDI_{100}) changed by age classes and time. First, for each species group, we used a mean site index estimated for the whole study area with the growth index of Lapointe-Garant et al. (2010) between 1950 to 2000 as initial site index and an area-weighted mean of actual RDI_{100} of the forest management unit as initial RDI_{100}. Initial values of SI and RDI_{100} were 11.5, 19.5, 16.4 m and 0.67, 0.61, 0.64 for black spruce, jack pine and trembling aspen respectively. We simulated these variables with or without considering climate impact. We then calculated the ratios of annual volume increments using variable and fixed SI and RDI for each period and used it as a correction factor. Although this ratio varied widely for very young (less than 40 years) and very old (more than 160 years) stands, it was quite stable across young and mature stands which constitute the majority of our study area (Figure 3.4). Hence, we used a mean correction factor independent of age classes to correct the yield curves for each period (2001 – 2100 at 5-year time step). Therefore, for each stratum, there were 120 yield curves (1 stratum * 2 climate scenarios * 3 species groups * 20 time steps of 5 year each).

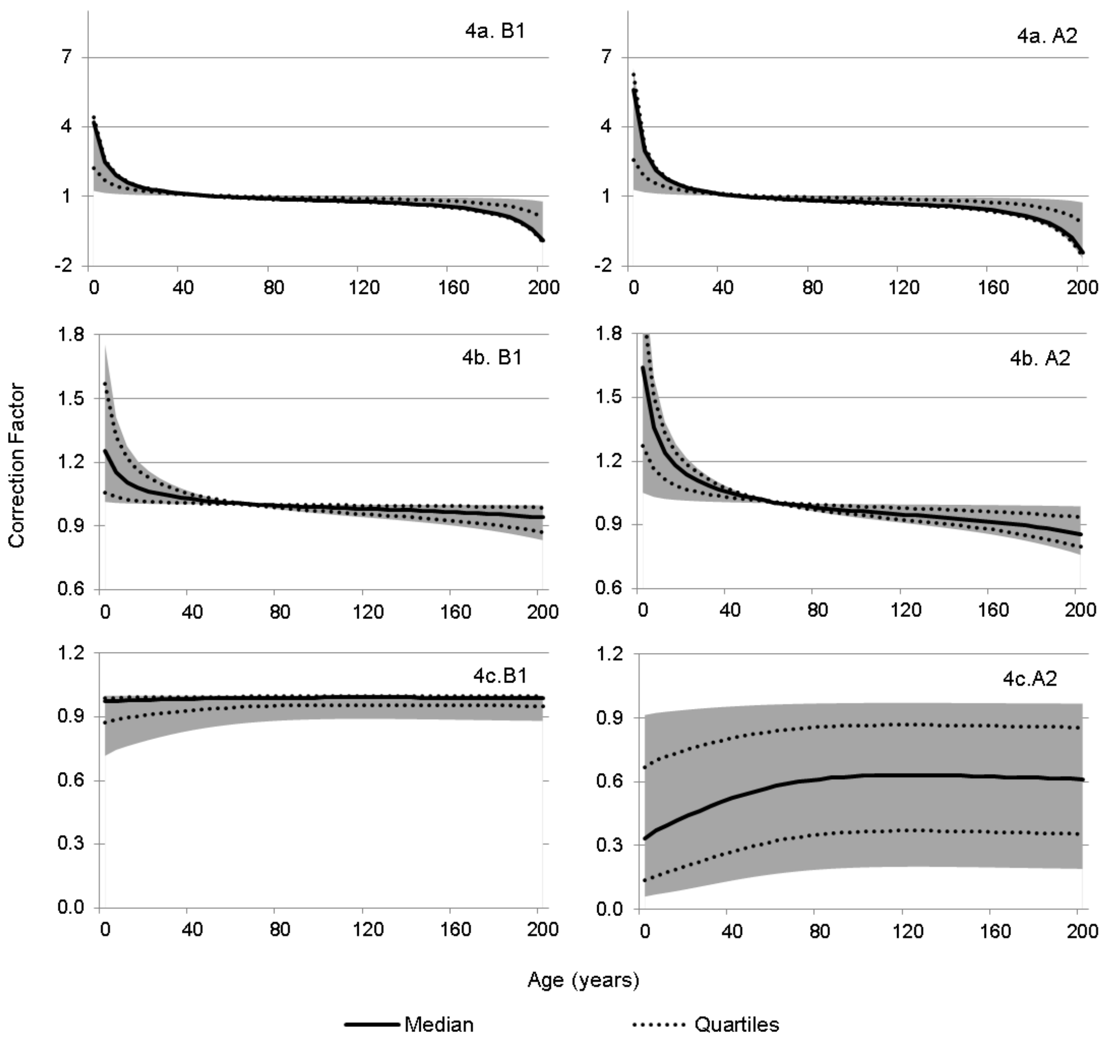

Figure 3.4 Percentiles of correction factor under different age classes of a. Aspen, b. Jack pine and c. Black spruce stands under B1 (left) and A2 (right) climate scenarios. Shaded area is the uncertainty band between 5th and 95th percentiles.

3.04.03 Strategic forest management planning

Two forest management strategies were considered: an ecosystem-based management strategy (EBM) and a forest regulation strategy (business-as-usual – BAU). To reduce the overall complexity of the study, we only considered the most important coarse filter element of EBM (Gauthier et al. 2004, Dhital et al. 2013), namely a targeted forest age structure.

(i) Targeted forest age structure

Assuming that fire is a random event and that the probability of a stand to burn is independent of its species composition and age, a target forest age structure under EBM is expressed in terms of areal proportions of cohorts derived from the theoretical age-class distribution of Van Wagner (1978). The proportion of area under each cohort was calculated based on an average fire cycle and expected transition ages from one cohort to another. Gauthier et al. (2004) estimated 150 and 275 years as transition ages from cohort 1 to cohort 2 and cohort 2 to cohort 3. Providing an average fire cycle of 147 years (lower and upper 95% confidence limits of 125 and 170 years, respectively) (Bergeron et al. 2004), our target age structure was determined to be 63 %, 21 %, and 16 % of the forest area under cohorts 1, 2, and 3. The actual forest age structure was 79%, 19% and 2% under cohorts 1, 2 and 3 respectively. BAU targets to maintain an equal area for all age classes younger than the rotation age in the FMU.

(ii) Silvicultural strategies

In the BAU scenario, careful logging around advanced growth (CLAAG) (Groot et al. 2005) was the only silvicultural option available for timber harvesting. We assumed that the species composition of any regenerated stand would be equivalent to that of the pre-harvest stand. The silvicultural strategies under EBM are inspired by stand dynamics. To emulate large scale disturbances, essentially under cohort 1, we used clear cutting. To emulate the small scale disturbances and natural mortality (gap dynamics) under cohort 2 and 3 respectively, we used partial cutting. CLAAG was used to reinitiate succession of black spruce stands. Clear-cutting followed by planting was used to regenerate jack pine stands. Finally, trembling aspen stands was reinitiated with clear-cutting. Partial cutting was used to emulate the transition of even-aged stands (including cohort 1 stands) to irregular stands, and to convert irregular stands to uneven-aged forests. Irregular shelter wood cut (removal of up to 50 % of merchantable volume, Raymond et al. 2009) was used as partial cutting to emulate cohort transitions.

(iii) Succession trajectories after fire

We used the fire map provided by Bergeron et al. (2004) to identify the time since last fire of the study area. However, this map did not entirely cover our study area (Figure 3.1). Therefore, we used temporary sample plot data (see § 3.04.02) to calibrate a regression model predicting time since last fire from cartographic attributes. We modeled the fire date as a function of different stand attributes (equation 3.6);

Eq. (3.6) $$TSF = \beta_0 + \beta_i X_i + \beta_j X_j + \cdots + \beta_n X_n + \beta_{ij} X_i X_j + \cdots + \beta_{n-1*n} X_{n-1} X_n$$

where TSF is time since fire, β_0 is model intercept and $\beta_i - \beta_n$ are parameters to be estimated (Table 3.2). X_i – X_n are categorical or continuous variables: stand density, species group, surface deposit, potential vegetation, ecological region and mean age of plot dominant trees. Stand density was classified into two classes (A-B as high density stands and C-D as low density stands, MRN 2009). Species groups are explained in section

Table 3.3 Parameter values of equation 3.6.

Variables	Classification	Estimate	Standard Error	P-Value
Model intercept		1928.83	39.86	<.0001
Mean age of dominant trees		-0.75	0.04	<.0001
Density	Low	-16.28	3.70	<.0001
Species group				
	Tolerant softwood	35.85	26.33	0.1736*
	Intolerant hardwood	26.41	26.89	0.3262*
	Intolerant softwood	54.38	26.80	<.0427
Surficial deposit				
	Clay	16.12	3.85	<.0001
	Coarse sandy deposit	33.44	6.19	<.0001
Potential vegetation				
	ME1	-29.54	29.58	0.3181*
	MS2	-173.69	46.80	<.0002
	RE2	-50.25	29.42	0.0879*
	RE3	-35.88	29.36	0.2219*
	RS2	-43.82	30.49	0.151*
Interaction (ecological region * potential vegetation)				
	5a * ME1	-50.23	8.82	<.0001
	5a * MS2	66.79	38.39	0.0822*
	5a * RE2	9.32	11.46	0.4165*
	5a * RE3	10.80	9.65	0.2629*
	5a * RS2	-23.05	11.61	<.0473

***Not significant at P<0.05**
ME1 and MS2 respectively belong to mixed forest of "black spruce – trembling aspen" and "balsam fir – white birch" vegetation types at the climax of succession. RE2, RE3 and RS2 respectively belong to softwood forest of "black spruce – moss" or "black spruce – ericaceous shrubs", "black spruce – sphagnum" and "balsam fir-black spruce" vegetation types at the climax stage. 5a is Abitibi lowland and 6a is Lake Matagami lowland ecological regions.

3.02.02. Surface deposits were grouped into Cochrane till, fine textured glacio-lacustrine clay and coarse textured deposits following Belleau et al. (2011). Potential vegetation is the vegetation of a reference point in future (climax of a succession pathway, MRN 2003) as opposed to the actual vegetation (Henderson et al. 2011). It is predicted as a function of indicator species groups, actual vegetation, regeneration and physiographic variables (MRN 2003). Mean age corresponds to the mean age of plot dominant or co-dominant trees from which a core at 1 m is extracted for ring counting during the inventory (MRN 2003). SAS PROC MIXED procedure (SAS, Cary, USA) was used for parameter estimates. There were 1162 temporary sample plots available for which time since last fire could be identified from the fire map. Overall R^2 was 43% and mean age of stand dominant trees explained the highest proportion of variance (15%) followed by the interaction of ecological region and potential vegetation.

Once time since last fire was estimated for all the polygons in the study area, succession pathways were identified by regrouping forest polygons on the basis of similarity in ecological region, soil type (Belleau et al. 2011), potential vegetation type, species group, density classes and time since last fire. Time since last fire was classified into three classes (0 -150 years, 150-275 years and >275 years). These limits correspond to the approximate transition ages of cohorts 1, 2 and 3 respectively (Gauthier et al. 2004). Then we calculated the area covered by each of these groups and only considered the most representative succession pathway (the trajectory that covered the highest proportion of area) in each class of time since last fire. Table 3.4 provides the succession trajectories after fire. We used these trajectories to reinitiate the stands burned by fire in our timber supply model.

(iv) Baseline timber supply formulations

We first elaborated standard versions of timber supply problems (e.g. Bettinger *et al.* 2009) in a 150 year planning horizon (30 periods of 5-year each) in Woodstock (Remsoft Inc., Fredericton, NB) as baseline scenarios for both BAU and EBM strategies without climate change effects. Under BAU, the model was required to provide an even flow over the planning horizon. EBM required an additional constraint of maintaining 63%, 21% and 16% of area under cohort 1, 2 and 3 respectively over the planning horizon. Plantation of jack pine was limited below 7500 ha per period as specified in the actual forest management plan to avoid a potential allowable cut effect. Baseline models for BAU and EBM were solved by the Mosek linear programming solver (Mosek ApS, Copenhagen, Denmark). The general objective was to maximize the periodic harvest. Woodstock, by default, adds non-negativity and resource availability constraints.

A sensitivity analysis was conducted with shadow prices. A shadow price is the change of the value of the objective function, should a particular constraint be changed by one unit in a linear programming optimization problem (Davis et al. 2001). Shadow prices identify the binding constraints and their magnitude. In this study, maximum values of shadow prices were used to identify the most constraining periods for each 5-year periodic re-planning exercise with updated climate sensitive yield curves. We also used maximum and minimum shadow prices to identify the magnitude of constraints imposed by plantation and cohort requirements under EBM.

Table 3.4 Succession trajectories of study area observed after 75, 175 and 275 years since last stand replacing fire (TSF) on different surficial deposits in Abitibi lowland (5a) and Lake Matagami lowland (6a) ecological regions.

TSF	Potential vegetation	Succession pathway	5a_clay	5a_coarse sandy deposit	6a_clay	6a_coarse sandy deposit	6a_till
75	ME1	IH	30%		17%		
		IS	18%		12%		
		TS			26%	21%	
	MS2	IH		17%			
	RE2	IS		17%			42%
		TS	44%		9%	69%	24%
	RE3	TS			33%	8%	
	RS2	IH		25%			8%
		IS					7%
175	ME1	IH	7%		15%		
		TS	22%		14%	7%	
	MS2	IH		16%			
	RE1	TS		19%			
	RE2	TS	12%	20%	9%	27%	
	RE3	TS	35%		54%	66%	
	RS2	IH		7%			
		TS		25%			86%
275	ME1	TS	39%		13%		
	MS2	TS		8%			
	RE1	TS		22%			13%
	RE2	TS	7%		11%	23%	70%
	RE3	TS	34%		65%	67%	
	RS2	IH		18%			17%
		TS		39%			

IH=Intolerant hardwood, IS=Intolerant softwood and TS=Tolerant softwood. Potential vegetation types are explained in Table 3.3. The values correspond to the percentage of the area of a surficial deposit in a ecoregion with a specific potential vegetation type after a given time since last stand replacing fire. Smaller proportions of area (<5%) were not included in the table. Therefore, areas do not sum exactly to 100%.

(v) Accounting for climate in timber supply formulations

Climate sensitivity was integrated in timber supply analyses by updating the climate sensitive yield curves under a five year rolling planning framework as the practice in Quebec. To update the age class distribution after each rollover planning, the following procedure was followed:

1. Woodstock (Remsoft Inc. NB, Canada) was used to generate harvest schedule of our study area for next 150 years (30 periods of 5 years each).
2. We then used Stanley (Remsoft Inc. NB, Canada) to allocate treatments to the polygons as specified in the harvest schedule for the first period of the planning horizon. Stands treated by clear cutting (e. g., CLAAG or clear cutting followed by plantation) were updated with their post treatment age class set to zero. For the stands treated by partial cutting, post treatment age was determined based on the proportion of volume removed during partial cut (50%) and the age for the stand to achieve the volume equivalent to the residual volume after partial cutting in a particular stand type. It was assumed that the whole polygon was treated if Stanley indicated that it underwent a treatment in a planning period.
3. Area module of the Woodstock was rebuilt with new age structure and new harvest schedule was built.
4. This procedure was repeated 20 times as we had climate data available until the year 2100.

A total of four timber supply analyses scenarios (two management strategies times two projected climate scenarios) were developed.

(vi) Accounting for fire in timber supply formulations

Fire burn rate is projected to increase progressively over the period 2001-2100 under B1 as well as A2 climate scenarios in our study area (Bergeron et al. 2010). Current burn rate and burn rate under B1 and A2 climate forcing was reported as 0.22% year^{-1}, 0.35% year^{-1} and 0.55% year^{-1} by the year 2100 respectively (Bergeron et al. 2010). We used these results. As in integrating climate sensitivity (section 3.03.02.ii), we calculated the five year average burn rate for each scenario considered (1 burn rate for each time step (total 20) over 100 years period) and simulated the timber supply in simulation platform of Woodstock (Remsoft Inc. NB, Canada) drawing randomly the burn proportion of that time step in a five year rolling planning framework. A total of six scenarios were developed to account for three burning scenarios (3 burn rates * 2 management scenarios).

While accounting for the impact of fire on periodic timber supply, following assumptions were made; 1. Flammability of forest is equal regardless of age, structure or species composition; 2. Burning or clear cutting (e. g., CLAAG) converts the age class of the stand to zero; 3. Burned stands regenerate without delay. 4. A stand hit by fire was assumed to burn completely, hence salvage logging was not considered. Stands to be burned were selected randomly. Stands with the highest volume were selected first for harvesting treatments during the simulations.

Since BAU scenario has CLAAG as the only option for timber harvest, 100% of the target output in the simulation was assigned for this treatment. However, EBM had several silvicultural treatments as a means of timber harvest. Therefore, the order of treatment was set based on the treatment providing the lowest to the highest proportion of timber harvested under the optimization mode of the model in the baseline scenario. In other words, silvicultural treatment contributing the lowest proportion of volume was ordered first in the sequence to avoid all the volume coming from a single treatment.

To update the inventory after fire, we followed Armstrong 2004 and Savage et al. 2010. We actually summed the area burned by development type and transferred it to the youngest age class of a given succession pathway depending on the stand potential vegetation type (Table 3.4). Harvested areas were updated as explained in section 3.04.03.v.

To account for the cumulative impact of climate on fire and growth, we simulated fire and harvest in the 5-year periodic re-planning updating the age class distribution and yield table. Four scenarios (2 climate scenarios * 2 management scenarios) were developed to account for the cumulative impact of projected climate scenarios on wood supply.

To reduce the variability of wood supply while accounting for fire, Armstrong (2004) and Savage et al. (2010) ran 1000 simulations, whereas Didion et al. (2007) ran 30 simulations. We simulated up to 1000 runs to visualize the amplitude of variance of wood supply between the simulations and observed that the amplitude was relatively stable after about 100 simulations. Therefore, we simulated each scenario that included fire as a random event 100 times. From 100 replications, 5^{th}, 10^{th}, 25^{th}, 50^{th}, 75^{th}, 90^{th} and 95^{th} percentiles of the empirical distribution function of periodic total volume harvest were calculated.

We used the method developed by Armstrong (2004) and Savage et al. (2010) to quantify the disruption risk in future timber supply due to fire and changing climate. An arbitrary desired level of periodic timber supply was fixed from 0 to baseline timber supply (4.1 million m^3/period in our case) with an interval of 0.5 million m^3. A binary variable with a value of 1 was assigned if the minimum harvest level was greater than or equal to an arbitrarily fixed desired volume level and 0 otherwise. Then, the proportion of runs that met the arbitrarily fixed desired harvest level was calculated. The probability of achieving a fixed harvest level was obtained by this proportion. This empirical distribution function of periodic total volume harvest was then plotted against the proportion of arbitrarily fixed harvest level to assess the risks related to fire and climate change.

Finally, to test the robustness of the management strategies (i. e., BAU vs. EBM), we used three ecological criteria: mean harvest age, mean inventory age and proportion of productive area under 40 years of age over time in addition to the level of wood supply. Proportion of area under age of 40 years is an important indicator of the probability that a woodland caribou population is self sustaining (Enviornment Canada 2011).

The following naming convention was used when referring to the different scenarios considered in this study: Base – No consideration of climate or fire; CC_B1 – Impact of B1 climate scenario on yield (fire not considered); CC_A2 – impact of A2 climate scenario on yield (fire not considered); Fire_C – Impact of current burn rate of fire (no impact of climate change on yield); Fire_B1 – Impact of fire under B1 climate scenario (no climate impact on yield); Fire_A2 – Impact of fire under A2 climate scenario (no climate impact on yield); Fire_CC_B1 – Cumulative impact of B1 climate scenario on yield and fire; Fire CC_A2 - Cumulative impact of A2 climate scenario on yield and fire.

3.05 Results

3.05.01 Climate change impact on periodic timber supply

Projected climate changes will have a negative impact on the periodic timber supply in the long term. When we used the climate sensitive yield tables under projected B1 and A2 climate scenarios, the minimum periodic timber supply decreased significantly over the 21st century under BAU and EBM. In the short term, EBM showed greater robustness to changes in yield when compared to BAU (Figure 3.5). Projected climate scenarios made the critical period more vulnerable. The critical period corresponds to the period in the planning horizon in which harvestable volume is available at its lowest level. Timber supplies for EBM and BAU recur nearly to their original level after a critical period under B1 but not under the A2 climate scenario (Figure 3.5).

Shadow price analyses revealed that the first five planning periods under EBM as well as BAU are crucial to achieve the desired harvest level, as shadow prices are the highest in these periods (Figure 3.6). Increasing the plantation area of jack pine could help maintain timber supply (Figure 3.7a). Requiring reaching a desired forest age structure in EBM further constrains the level of periodic timber supply as average shadow prices for the minimum area of cohort 2 and cohort 3 are negative; -25 m^3ha^{-1} and -125 m^3ha^{-1} respectively (Figure 3.7b). Shadow prices (Figure 3.6, 3.7) explain the resource availability constraints. Positive values indicate that increasing the silvicultural activities such as plantation area of jack pine or of clear cutting could increase the periodic wood supply. Negative values for the cohort requirements indicate that maintaining a specified area under cohort 2 and 3 (old forest structure) further reduces the periodic wood supply.

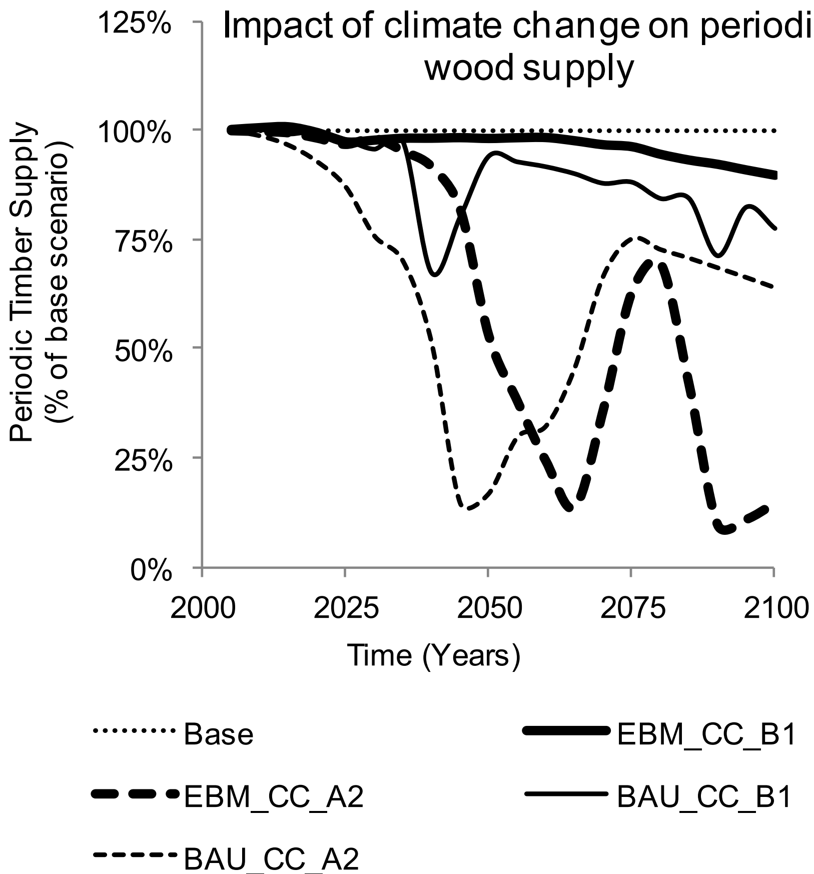

Figure 3.5 Impact of projected climate scenarios on timber supply. Base scenario is common to both (BAU and EBM) management strategies as difference is not significant. EBM_CC_B1 and EBM_CC_A2 are future wood supply with ecosystem-based management under B1 and A2 climate scenarios respectively. BAU_CC_B1 and BAU_CC_A2 are future wood supply with business as usual strategy under B1 and A2 climate scenarios respectively.

3.05.02 Accounting for fire in timber supply analyses

Integrating fire impact with the actual burn rate into timber supply model under BAU resulted in a 19% median loss of periodic harvest at the end of the century. Burn rates under projected B1 and A2 climate scenarios resulted in further median losses of 11% and 19% periodic harvest respectively by the year 2100 (Figure 3.8 a, b, c). Integrating actual burn rate and burn rates under projected B1 and A2 climate scenarios into timber supply model with EBM resulted in a median losses of 13, 22 and 29 percent of periodic harvest respectively by the year 2100 (Figure 3.8 d, e, f). As time elapsed, the range of uncertainty (grey area in Figure 8 as 5th and 95th percentiles of simulated timber supply) also grew.

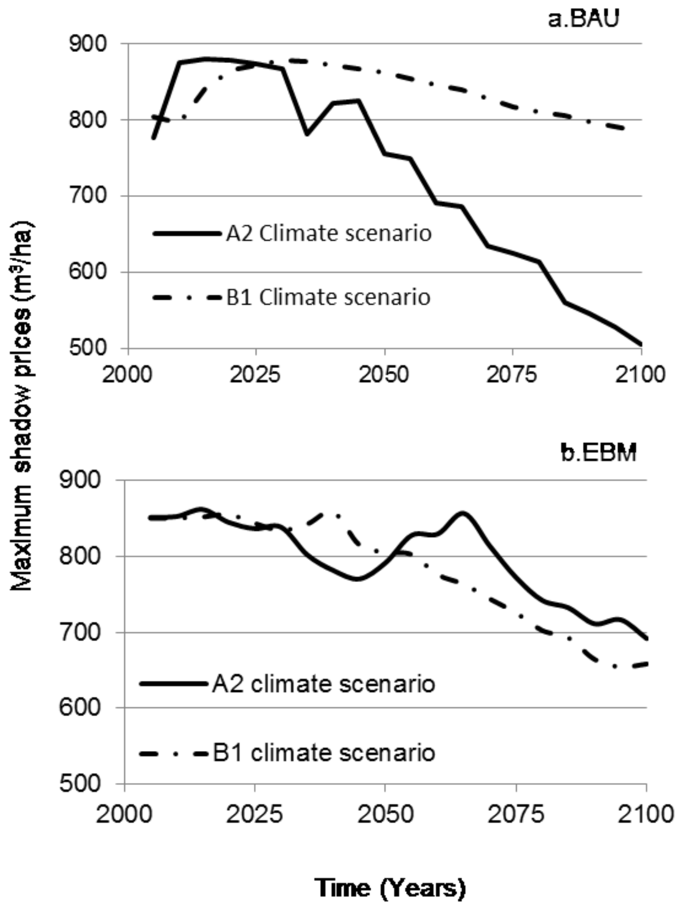

Figure 3.6 Maximum shadow prices of harvesting activities in response to projected A2 and B1 climate scenarios under business as usual (a) and ecosystem-based management (b) strategies.

3.05.03 Cumulative impact on periodic timber supply

Under the BAU, the cumulative impact of fire and climate change was 44% and 79% loss of median periodic harvest under B1 and A2 climate scenario respectively (Figure 3.9 a, b). For EBM scenarios, median loss was smaller (35% and 64% respectively) compared to that of BAU (Figure 3.9 c, d). Under A2, range of uncertainty was also wider under BAU (Figure 3.9). For both management strategies, available volume for harvest in the critical period is severely constrained (Table 3.5).

Table 3.5 Impact of climate and fire on periodic wood supply

Strategies	Scenarios				
	Fire_C	Fire_B1	Fire_A2	Fire_CC_B1	Fire_CC_A2
BAU	19%	30%	38%	44%	79%
EBM	13%	22%	29%	35%	64%

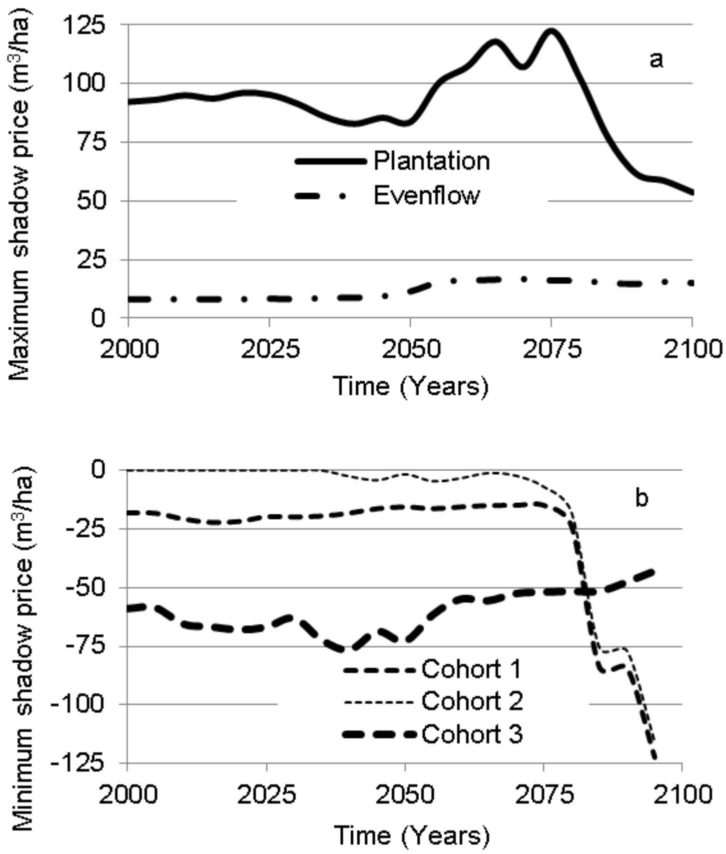

Figure 3.7 Maximum shadow prices under the requirements such as plantation and even flow over the periods (panel a) and minimum shadow prices under the requirement of the proportion of desired age-class distribution (cohort) over the territory (panel b) under ecosystem-based management scenario.

Table 3.6 Age structure and species composition under the influence of climate change

Scenarios	Ecological indicators					
	EBM			BAU		
	Stand Age (±)*	CPRS Age(±)	Proportion of Area Under Age 40 (±)	Stand Age (±)*	CPRS Age(±)	Proportion of Area Under Age 40 (±)
Base	50(2)	106(11)	43(2)%	45(2)	84(9)	52(2)%
CC_B1	50(1)	100(2)	44(1)%	47(2)	81(1)	48(2)%
CC_A2	72(11)	119(7)	28(0)%	66(9)	99(7)	35(5)%
Fire_C	49(1)	96(3)	44(0)%	45(0)	79(1)	51(0)%
Fire_B1	49(1)	96(3)	44(1)%	45(0)	79(1)	51(0)%
Fire_A2	48(1)	95(4)	53(1)%	45(0)	80(1)	50(0)%
Fire_CC_B1	50(1)	98(3)	44(1)%	45(0)	79(1)	46(2)%
Fire_CC_A2	65(7)	115(6)	34(4)%	63(7)	103(7)	38(4)%

*Values in the parentheses are ± confidence interval at 95%.

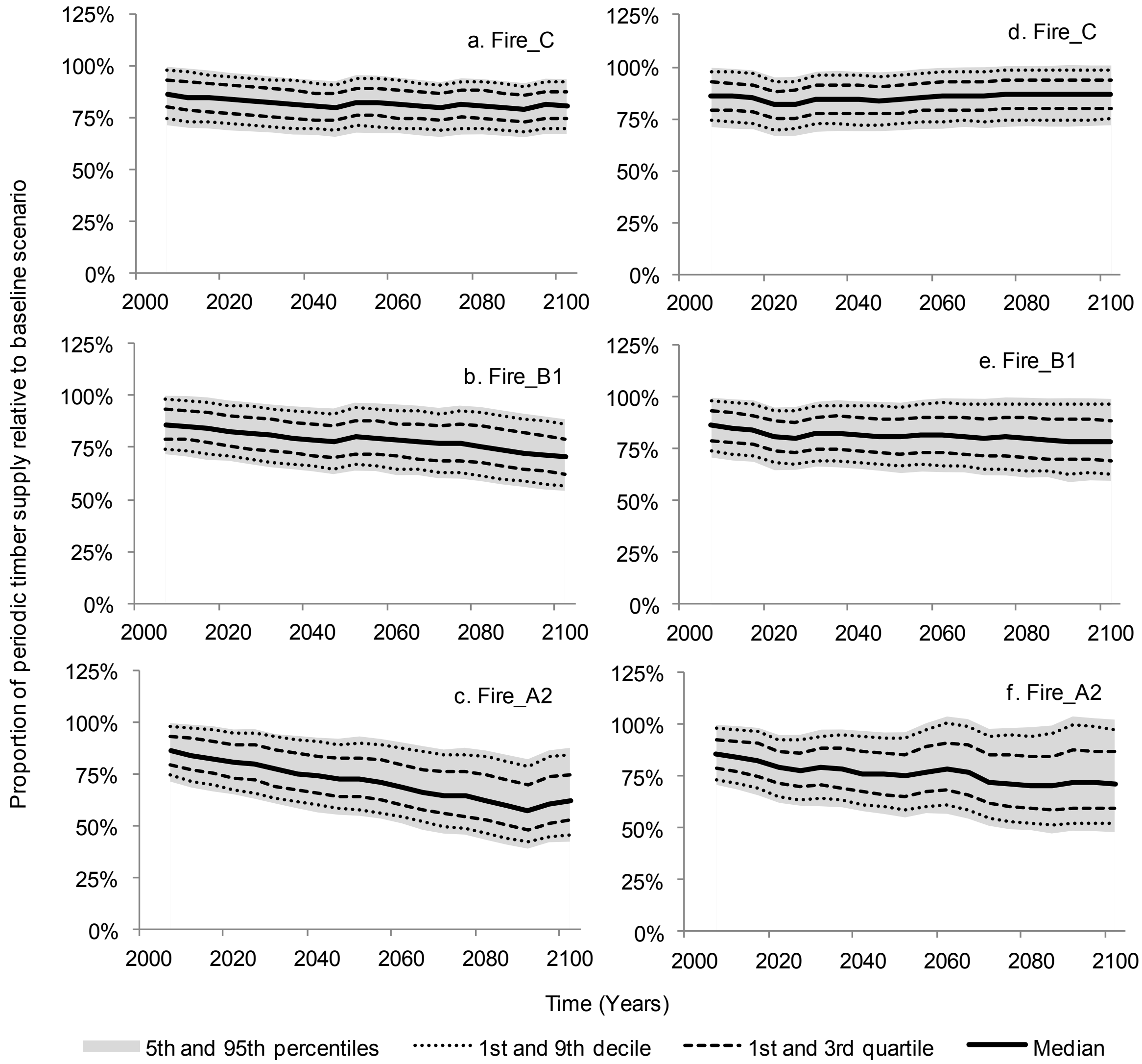

Figure 3.8 Distribution of periodic timber supply calculated using 5-year periodic re-planning accounting for fire under actual burn rate and burn rates under A2 and B1 climate scenarios. Panels in left (a, b and c) and right (d, e and f) respectively show the results under business as usual and ecosystem-based management scenarios. Grey area is the uncertainty band as 95th and 5th percentile timber supply.

It is impossible to maintain the current level of periodic timber supply when fire is accounted for in the planning models (Figure 10. a, b). The cumulative impact under A2 climate scenario had an extreme impact on the periodic timber supply under each management scenario.

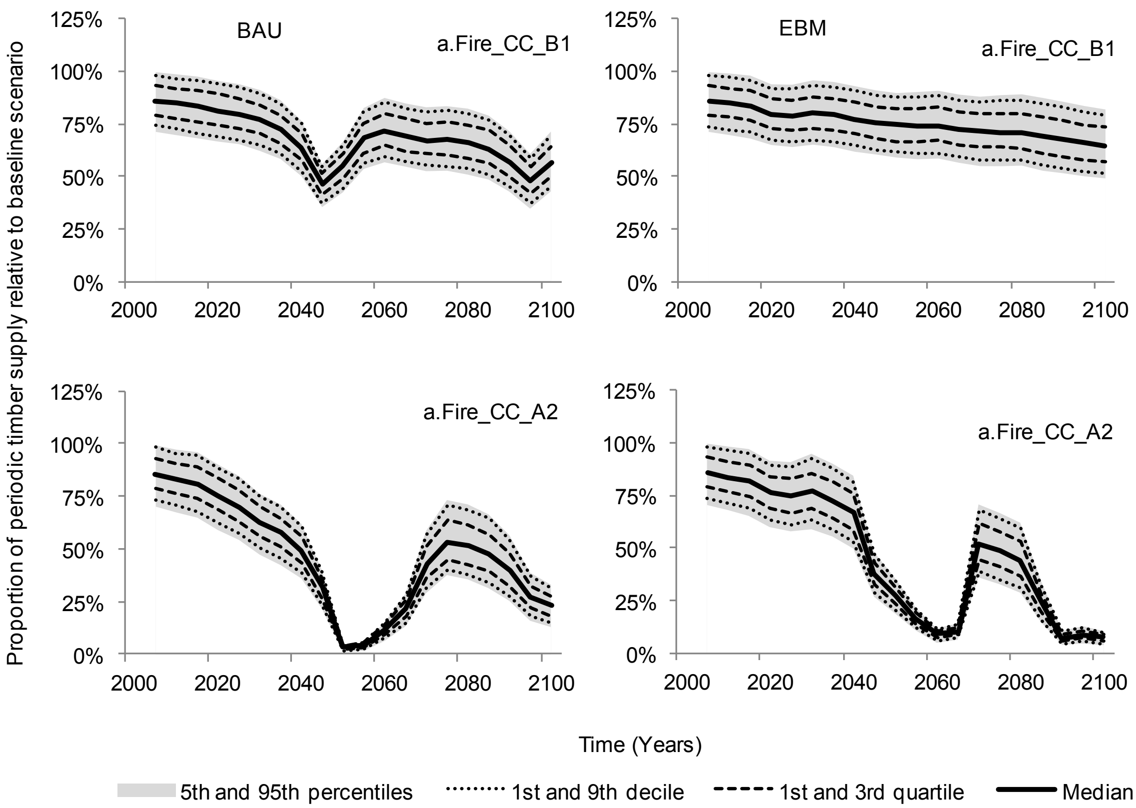

Figure 3.9 Distribution of periodic timber supply calculated using 5-year periodic re-planning accounting for the cumulative impacts of climate forcing scenarios on growth and fire. B1 and A2 are two climate scenarios accounted for. Panels on left (a and b) present the results under business as usual and panels on right (c and d) demonstrate that of ecosystem-based management strategies. Grey area is the uncertainty band as 95th and 5th percentile timber supply.

3.05.04 Age structures and species composition

Average age of harvest with careful logging around advanced growth (CLAAG) over the planning horizon under baseline scenario was higher under EBM compared to BAU (Table 3.6). EBM had also a higher stand inventory age under baseline scenario and the scenarios developed to account for the impact of climate change on yield and fire (Table 3.6). However, accounting for the impact of climate change on yield and fire burn rates resulted in the reduction of mean harvest age and mean inventory age under both management strategies (Table 3.6). Accounting for fire (current burn rate and such rates under B1 and A2 climate scenarios) under EBM and BAU scenarios reduced the mean harvest age by up to 11(±4)[1] years and 5(±1) years respectively. It reduced the mean inventory age under EBM by 1(±1) years. No such change was observed under BAU.

[1] Values in the parentheses are confidence interval of mean at 95%.

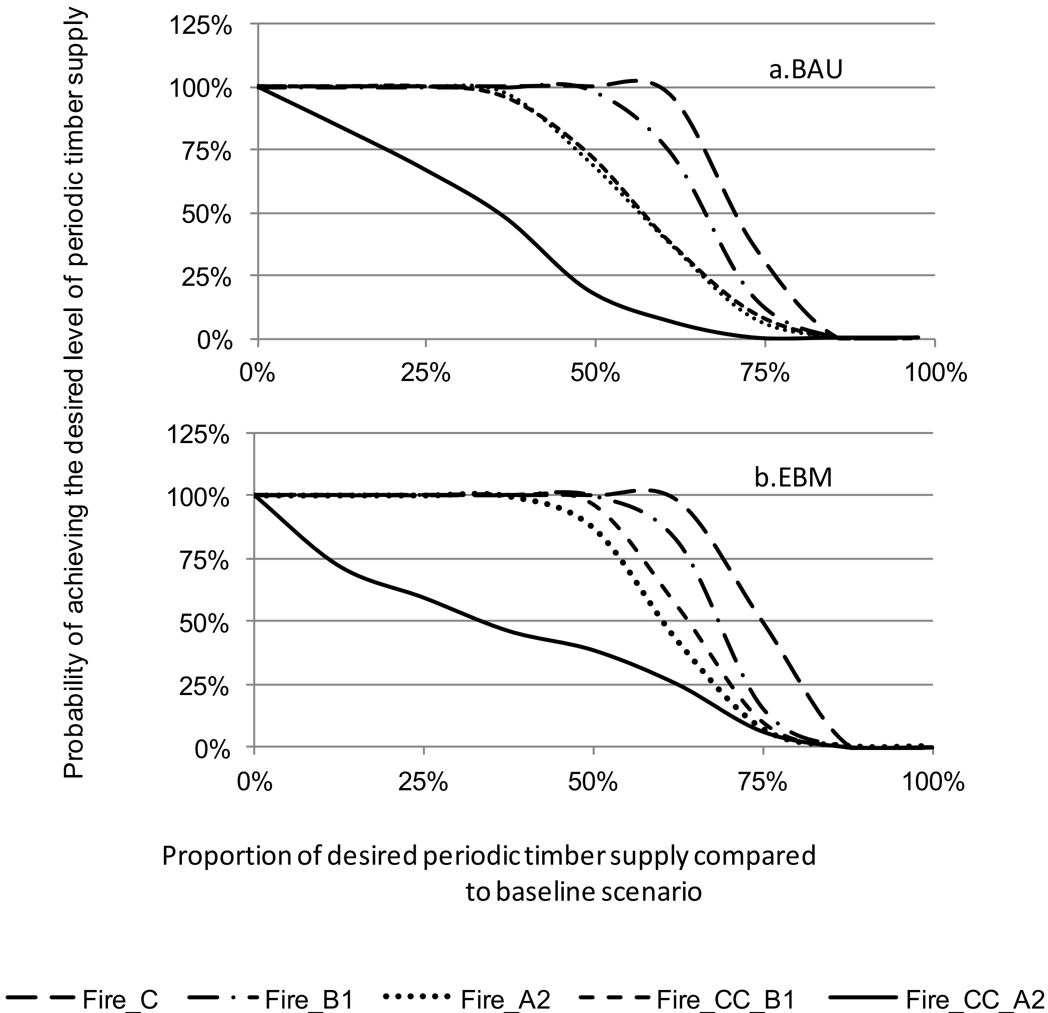

Figure 3.10 Risk assessment tool as a probability of achieving the minimum harvest volume against the baseline scenario over the year 2100. Panels a and b respectively show such probability under business and usual and ecosystem-based management strategies.

Accounting for the impact of climate scenarios on yield also reduced the mean harvest age by up to 6(±2) years and 4(±1) years under EBM and BAU respectively. Cumulative impact of climate scenarios on yield and fire resulted in reduction of mean harvest age by 8(±3) and 5(±1) years under EBM and BAU respectively.

We calculated the proportion of area under age of 40 to evaluate the effect of each management strategy on the quantity of regenerating forest under different climate scenarios. The analysis showed that proportion of such area was increased under EBM while considering the impact of climate on fire and yield (except CC_A2 and fire_CC_A2). Such proportion was decreased under BAU (Table 3.6). However, the change in the proportion of area under 40 years of age was minimum (except EBM_Fire_A2 where the proportion increased by 10%) and EBM had consistently lower proportion of area under age of 40 for the most of the scenarios (Table 3.6).

The baseline proportion of area covered by trembling aspen, jack pine and black spruce was 13%, 33% and 54% respectively. Accounting for the impact of projected climate scenarios on yield did not change the proportion of area covered by different species groups significantly. Although jack pine increased its area coverage by 1% (which was within the 95% confidence limit) over the planning horizon under projected B1 climate scenario, such gain was not evident under projected A2 climate scenario (graph not shown). There was no effect of management strategy.

3.06 Discussion

3.06.01 Combined effects of fire and climate

Simulation of fire events in a harvest scheduling model suggested that a reduction in the target of desired harvest level is required (Figures 3.8, 3.9, 3.10) to ensure the long term sustainability of timber supply to wood processing industries. This result is consistent with the findings of other investigators (e. g., Armstrong 2004; Peter and Nelson 2005). Although Savage et al. 2010, Martell (1994) and Van Wagner (1983) suggested that low burn fractions do not impact on periodic timber supply significantly, our study suggests that even with a 0.22% annual burn rate, the median periodic timber supply should be reduced by at least 19% (Figure 3.8) under the a BAU. Such reduction is slightly less under EBM (13%).

The real impact of fire on timber supply is not limited to the sum of volume harvested or area burned. Van Wagner (1983) urged that it has to be measured by the effect on the harvestable timber supply. When mature and old growth stands are limited in the landscape due to harvesting, fire affects younger stands thereby making even salvage logging unprofitable (Gauthier et al. 2009b). Therefore, an interaction mechanism between harvest and fire provides a better understanding of the impact of fire activities on future wood supply (Savage et al. 2010).

Fire plays a key role in shaping forest ecosystems (Seymour and Hunter 1999; Stocks et al. 2003; Flannigan et al. 2005). A larger proportion of recently burned area is colonized by pioneer species such as trembling aspen and jack pine (Bergeron and Dansereau 1993; Girardin et al. 2012). Both management strategies (BAU and EBM) responded to different climate scenarios with fairly unchanged relative area over time under succession pathways lead by black spruce, jack pine and trembling aspen. Jack pine is expected to expand its coverage under projected climate scenarios in the cost of the succession pathways led by black spruce and trembling aspen (e. g., Girardin et al. 2012). However, such gain was uncertain under projected climate scenario in our study area. This is because our succession model was not sensitive to projected climate scenarios.

Effect of climate change on the boreal forest may be direct through altering plant growth and indirect through changing disturbance activity. The projected climate scenarios seem to play a deciding role in growth and disturbance dynamics of the forest in our study area. Although the change in growth pattern of jack pine and trembling aspen under the projected climate scenarios seems insignificant, black spruce growth rates decline markedly (Figure 3.3). Periodic timber supply in the early half of this century showed a minimum change in response to climate change. Consequent impacts should occur by the end of this century (Figure 3.5). This reduction is attributable to the reduced growth of black spruce (Figure 3) which covers about 60% of our study area.

Increase in the mean summer temperature and lack of synchrony in the increase of temperature and precipitation explains the reduction in growth of black spruce in eastern Canada (Girardin et al. 2006) and white spruce at tree line areas in Alaska (Wilmking et al. 2004). Warmer temperatures increase the moisture holding capacity of the atmosphere, thereby imposing an additional evaporative demand to low productivity boreal ecosystems (Bunn et al. 2007; Falge et al. 2002). These indications are preoccupying because temperature is expected increase at higher latitudes (IPCC 2007) where our study area is located.

Several studies have suggested that fire regime has significantly changed in response to climate change in the recent years with more spatial and temporal variability in the boreal region (Flannigan et al. 1998; Flannigan et al. 2005). Since burn rate is largely determined by climate conditions (Flannigan et al. 2005), change in climate may have significant impacts on forest composition. Although it is difficult to predict the exact development of forest composition after fire given the current state of knowledge, the dynamics of fire on existing forest composition can be estimated. Bergeron et al. (2002) concluded that the presence of strong deciduous components in the mixed wood forest contributes to the presence of smaller fire of lower intensity making it possible to maintain mixed-wood associated species whereas larger tracts of coniferous forests favor larger fires of higher intensity. Therefore, it is critical to understand how disturbance regimes and changes in such regimes due to climate change may affect the vegetation distribution and forest composition after disturbances (Bergeron et al. 2004). IPCC (2007) projects a shorter fire return period and younger stands in boreal forests. Such changes might have a significant impact on wood supply as harvest in younger stand produces less merchantable timber per unit area (Gauthier et al. 2009b).

Cumulative impacts of climate scenarios (on growth and fire regime) resulted in a significant drop of median periodic timber supply in this study (Figure 3.9) and an increase in the uncertainty of achieving a desired level of timber supply under both management strategies (Figure 3.10). The study revealed that EBM has a relative advantage over BAU in terms of wood supply under cumulative impact of climate and fire (Figure 3.9; section 3.04.03). We also noticed that there is an interaction between direct and indirect impact of climate change as the cumulative impact of climate change on periodic timber supply was lower than the sum of direct and indirect impacts. This positive feedback is explained by the higher frequency of younger stands under increased temperature and shorter fire cycle (Girardin et al. 2012).

3.06.02 Adaptability of the forest management strategies to climate change

Rating the robustness of the management strategies through ecological criteria such as mean harvest age, mean inventory age and proportion of area under younger age classes suggested that ecosystem-based management was arguably the most robust strategy as it maintained a higher rotation age, an older standing inventory and a lower proportion of territory below forty years of age over time. Mean harvest age is important because a management strategy with extended rotation can absorb the variation in fire regime (Didion et al. 2007) which, in essence, is a function of climate. The proportion of productive area under 40 years of age can be used to estimate the critical disturbance area in the landscape beyond which the population of woodland caribou (*Rangifer tarandus caribou*) may not persist (Environment Canada 2011.). A decrease in mean inventory age means a decrease in the mean volume per hectare requiring more area to be harvested to meet the wood supply target (Didion et al. 2007) even though younger stands have a higher growth rate (Girardin et al. 2012).

3.06.03 Management implications

The projected increase in fire regime and resulting decrease in periodic timber supply may have significant consequences on public safety, timber protection and regional economy (Martell 1994; Bergeron et al. 2010). The combined effect of fire and harvest may lead the forest ecosystem beyond its historical range of variability, leaving little space for managers to avert the situation (Bergeron et al. 2010). Adaptation strategies to deal with these socio-economic and ecological challenges are needed to avoid such undesirable conditions. The maintenance of ecosystem resilience is a pre-requisite to avoid such situation (Bergeron et al. 2010). Boreal forests are generally resilient to fire and recover fully over a number of years (Thompson 2011). Based on the natural disturbance dynamics, ecosystem-based management may provide such an avenue to maintain the ecosystem resilience as it aims to reduce the gap between natural and managed ecosystems (Harvey et al. 2002; Gauthier et al. 2009a). Emulation of large scale disturbances such as stand replacing fire through clear cutting and small scale disturbances such as low severity fire and natural mortality through partial cutting and selection cutting respectively (Bergeron et al. 2002) could contribute to maintaining biodiversity and essential ecological functions.

Another way of maintaining the resilience of ecosystem is through reducing the environmental stresses (IPCC 2007). Such stresses include habitat fragmentation, pollution, introducing exotic species and overexploitation of forest resources. Increasing the network of protected areas and connecting individual protected areas and refuges through biological corridors may be useful in maintaining ecosystem resilience on a wider scale (Rustad et al. 2012). After all, it is the resilience of ecosystem that needs to be maintained by the adaptation strategies targeted to deal with climate change.

Probability of attaining a desired level of periodic wood supply can be determined with a risk assessment tool such as one provided in Figure 3.10. Adapted from Savage et al. (2010) and Armstrong (2004), information provided in our tool may be useful in estimating the probability of achieving a desired level of harvest in future under projected climate scenarios as well as under potential fire burn rates with EBM and BAU management strategies. This kind of information can also be used to reflect other uncertain decision making problems such as the probability of profits pertaining to a specific sawmill, impacts of market fluctuations, etc. (Savage et al. 2010).

3.07 Conclusion

The novelty in this study is that direct and indirect impacts of climate change were integrated in a single strategic forest management planning model which revealed the interaction between the two. A risk assessment tool with the information on probability of attaining desired wood supply in future under projected climate and fire scenarios with EBM and BAU was also developed. Our results show that decline in yield of black spruce dominated boreal zone could affect more on future wood supplies than succession or disturbance dynamics.

The results demonstrate the risk posed by fire aggravated by climate change in future timber supply from boreal forests. However, uncertainty is high among projected scenarios and some of the relationships presented here are nonlinear. The succession model used in this study was not climate sensitive. The present study was limited to the non-spatial nature of the problem. This study could be extended to address ecological values such as habitat of woodland caribou under uncertain circumstances.

Our simulations with projected climate scenarios and its impact on tree growth and fire activities suggest a substantial decrease in future timber supply. This adds in the growing body of researches that urge for adaptation strategies to deal with the impact of climate change which is already evident in the boreal forests. Although advantage of EBM over BAU in terms of wood supply was minimal, EBM seemed to provide several other advantages over BAU, which are crucial in maintaining ecosystem resilience and adaptability to environmental changes in uncertain times. However, it can be concluded from this study that merely switching forest management strategy from conventional sustained yield timber production to ecosystem-based management is not sufficient to ensure the sustainability of boreal forest and further adaptation strategies to minimize the impact of climate change on growth and disturbance regime in boreal forests are needed.

3.08 **Acknowledgements**

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3.09 References

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General Conclusion

Regulation of maximum sustained even flow of forests is simplifying the forest ecosystems and is questioned for its alleged role in the loss of biodiversity in managed landscape (Seymour and Hunter 1999; Bergeron et al. 2002; Heussleur and Kneeshaw 2003). Although ecosystem-based management is being considered to bridge the gap between managed and natural landscape (Harvey et al. 2002; Gauthier et al. 2009), feasibility of this strategies in the context of boreal forest has not been sufficiently studied. Moreover, tools used in forest management are designed for the maximization of timber harvest and do not account for the notion of uncertainty induced by fire, subsequent succession and changing climatic scenarios. Therefore, it is important to understand the impact of implementing ecosystem-based management strategy and its adaptability to the natural disturbances and climate changes.

Although ecosystem-based management (EBM) is drawing attention of planners, policy makers and academia over the past several years, tools specialized for this new regime of forest management are still to be developed. Until the tools such as growth and yield tables tested in the context of EBM are available, managers are obliged to choose from existing ones which were generally developed for the maximization of wood supply. We tested the adequacy of a stand-level and a tree-level model in predicting volume increment in an EBM context through repeatedly measured permanent sample plot data from the continual boreal zone of Quebec province, Canada (chapter I). Since stand-level model was found relatively better for our study area, we built timber supply models using stand-level yield tables and assessed the feasibility of an EBM strategies (chapter II) against sustainability criteria (e. g., social, economic and ecological) comparing the outputs with a business as usual (BAU) scenario. EBM objectives were achievable with 3 – 22% reduction in periodic wood supply and a transition period of 50 years during which the clear-cutting needs to be restricted to selected areas. The work of chapter II was extended to test the adaptability of EBM with accounting for the uncertainty induced by fire and projected climate scenarios in periodic wood supply (chapter III). We show that although EBM has relative merit over forest regulation, simply switching the management strategy is not enough to adapt to the anomalies in forest growth and wildfire regime induced by the climate change.

Although we covered a wide range of issues in this thesis, important issues still need to be considered to improve the knowledge in this domain. The growth projection model we used here to build timber supply models did not project the yield appropriately across all the variables we considered in this study. It still needs to be improved. Although we concentrated our harvest within operating areas, we did not consider other spatial issues such as green-up delays, adjacency constraints etc. Economic analysis of partial cutting needs to be carried out to demonstrate the feasibility of EBM in a boreal forest. We had limited climate data to calculate the growth index in our study area. It may have led to the higher uncertainty to the growth prediction of black spruce under A2-climate scenario. It needs to be improved. Our succession model was not climate sensitive. It may be the reason for jack pine not to expand itself under changing climate scenarios as expected. Therefore, future research needs to focus on improving the growth and yield projection models by improving the mortality component of the model, more pilot studies on the response of different forest stands in the boreal region to ecosystem-based management and improvement of growth index model with finer scale climate data to remove the uncertainty related to the impact of climate change on forest growth. Moreover, spatial analysis of the harvest, fire and their interaction in the context of changing climate may further improve the required knowledge significantly.

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