UNIVERSITÉ DU QUÉBEC EN ABITIBI-TÉMISCAMINGUE

AMÉLIORER LE BILAN CARBONE DES FORÊTS BORÉALES QUÉBÉCOISES : STRATÉGIES D'AMÉNAGEMENT À LONG TERME POUR L'ATTÉNUATION DES CHANGEMENTS CLIMATIQUES

THÈSE

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DÉDICACE

I dedicate this thesis to my parents (Hammou & Adjou) and my wife Mounia.

“Science is collaboration”

AVANT-PROPOS

Cette thèse de doctorat est présentée dans le cadre du programme de doctorat en sciences de l'environnement de l’Institut de recherche sur les forêts de l'UQAT. Elle est organisée en cinq chapitres, dont les trois articles (chapitres II, III et IV) sont rédigés en anglais, tandis que les chapitres I et V constituent respectivement l'introduction et la conclusion générale de ma thèse, rédigées en français. Les chapitres II, III et IV sont les trois articles suivants, qui constituent les trois chapitres principaux de cette thèse :

-Ameray, A., Cavard, X., & Bergeron, Y. (2023). Climate change may increase Quebec boreal forest productivity in high latitudes by shifting its current composition. Frontiers in Forests and Global Change, 6, 20.

-Ameray, A., Bergeron, Y. & Cavard, X. Modelling the potential of forest management to mitigate climate change in Eastern Canadian forests. Scientific Reports 13, 14506 (2023).

-Ameray, A., Cavard, X., Cyr, D., Valeria, O., Girona, M.M., & Bergeron, Y. (2023). Two centuries of carbon budget projection in Eastern boreal forest under different management strategies and climate change. (Soumis au journal Forest ecology and management)

Tous ces manuscrits ont été rédigés par AA dans le cadre de sa recherche de thèse doctorale à l'UQAT (Université du Québec en Abitibi-Témiscamingue) sous la supervision de YB et XC. AA, YB et XV ont contribué à la conceptualisation de l'étude et à la conception des scénarios. La calibration et le paramétrage des modèles ont été réalisés par AA. Dans le 3e article, DC a assisté AA dans le paramétrage du Modèle ForCS. Les simulations, l'analyse des données et la visualisation ont été réalisées par AA. Aussi, DC, OV et MMG ont également été invités à améliorer la rédaction de troisième article comme experts dans le domaine. Tous les co-auteurs de chaque article ont lu, commenté et approuvé le manuscrit final. YB et XC ont financé le projet.

Ce projet de doctorat a été mené pour améliorer notre compréhension de l'intensité de l'exploitation forestière et des changements climatiques sur la dynamique du carbone, en tenant compte de différentes perturbations naturelles (incendies, chablis et épidémies). Dans le premier article, le modèle PnET-Succession et LANDIS-II ont été calibrés pour la première fois dans les hautes latitudes de la forêt boréale Québécoise, puis l'impact des incendies de forêt sur le stockage du carbone dans la biomasse et la composition de la forêt a été évalué. Dans le deuxième article, deux perturbations naturelles (tordeuse des bourgeons de l'épinette et les chablis) ont été ajoutées aux incendies afin de simuler une évolution naturelle de l'écosystème. Les effets des stratégies d'aménagement sur la composition forestière, la structure d'âge et la séquestration et le stockage du carbone ont ensuite été comparés à des scénarios naturels. Enfin, dans l'article 3, le modèle ForCS-Succession a été utilisé pour évaluer et projeter les réservoirs et les flux du carbone sous différents scénarios de changement climatique et régimes de gestion, y compris la productivité primaire nette, la respiration hétérotrophe, la production nette de l'écosystème, la production nette du biome et le carbone transféré aux produits du bois. Ce projet pourrait contribuer à la gestion durable des forêts dans l'est du Canada dans le contexte des changements climatiques.

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LISTE DES ABRÉVIATIONS

AAC: Allowable Annual Cut
AD: Absolute Difference
AGB: Aboveground Biomass
BAU: Business-As-Usual
BGB: Belowground Biomass
CanESM2: Second-Generation Canadian Earth System Model
CC: Clear Cutting
CLAAG: Careful Logging Around Advanced Growth
CPRS: Cutting with the Protection of Regeneration and Soil
CRI: Canopy Removal Intensities
DOM: Dead Organic Matter
EBFM: Ecosystem-Based Forest Management
ECPs: Extended Concentration Pathways
ForCS: Forest Carbon Succession Extension
GSA: Global Sensitivity Analysis
IPCC: Intergovernmental Panel on Climate Change
LAI: Leaf Area Index
LSA: Local Sensitivity Analysis
maxAGB: Maximum Aboveground Biomass
maxANPP: Maximum Aboveground NPP
MNRF: Ministry of Naturelle’s Resources and Forest
MU: Management Units
NBP: Net Biome Productivity
NEP: Net Ecosystem Productivity
NFI: National Forest Inventories
NPP: Net Primary Productivity
PAR: Photosynthetically Active Radiation
PBM: Process-Based Models
PC: Partial Cuts
PnET: Photosynthetic/Evapotranspiration Model
RMSE: Root Mean Square Error
RCP: Representative Concentration Pathway
R_{h}: Heterotrophic Respiration

SBW: Spruce Budworm Outbreaks
SEP: Species Establishment Probability
SI: Site Index
SOC: Soil Organic Carbon
Tmax: Average of Maximum Monthly Temperature
Tmin: Average of Minimum Monthly Temperature
VPD: Vapour-Pressure Deficit

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RÉSUMÉ

La séquestration du carbone par les écosystèmes forestiers fait partie de la stratégie d'atténuation du changement climatique définie par le groupe d'experts intergouvernemental sur l'évolution du climat. Pourtant, les forêts boréales subiront des impacts significatifs sous l'effet des changements climatiques, principalement des modifications aux cycles de perturbations naturelles (incendies de forêt, épidémies, chablis). L'augmentation de la séquestration du carbone dans les forêts boréales et les produits du bois grâce à des méthodes d'exploitation appropriées pourrait contribuer efficacement à l'atténuation du changement climatique à l'échelle régionale. Afin de stabiliser et d'augmenter la capacité de séquestration et de stockage du carbone à l'échelle du peuplement, les traitements des coupes partielles (CP) basées sur des intensités de récolte (IR) faibles et modérées ont été proposés comme alternative à la coupe totale (CT) (correspondant à 95-100 % de IR). Cependant, dans la forêt boréale du Québec, il n’existe pas assez d’information sur la proportion appropriée de la superficie annuelle aménagée par traitement au niveau du paysage pour augmenter/stabiliser la séquestration et le stockage du carbone, maintenir l'habitat forestier, et répondre aux besoins industriels, principalement dans le cadre du changement climatique. Sous différents scénarios de changement climatique, y compris le climat actuel, ce projet vise à projeter et à évaluer la dynamique du bilan de carbone sur 300 ans (2010-2310) dans les forêts aménagées du Québec le long d'un gradient longitudinal, de l'ouest vers l'est : Nord-du-Québec (MU1), Saguenay-Lac-Saint-Jean (MU2) et Côte-Nord (MU3). Plusieurs stratégies d'aménagement variant dans leur utilisation de coupes partielles, des coupes totales et de taux de reboisement, incluant le scénario de référence (CT appliquée sur plus de 95 % de la superficie aménagée) ont été testées et comparées au scénario d'évolution naturelle (pas de récolte). Au cours des simulations, le modèle LANDIS-II a été utilisé comme modèle mécaniste combiné avec les extensions de succession PnET et ForCs dans différents chapitres. À l'échelle du peuplement, nous avons constaté que le changement climatique pourrait augmenter le stock de carbone de la biomasse globale et la productivité primaire nette dans le cadre des scénarios RCP par rapport au climat actuel pour toutes les espèces étudiées, à l'exception des espèces de conifères dans MU1 dans le cadre du RCP8.5 après 2100. Cette augmentation peut s'expliquer par l'extension de la saison de croissance et la réduction des blessures potentielles dues au froid. À l'échelle du paysage, les résultats de l'évolution naturelle montrent que le stock du carbone de la biomasse des feuillus augmentera sous différents scénarios de changement climatique au cours de la période d'étude, ce qui pourrait s'expliquer par l'expansion de la proportion de la forêt mixte, principalement entre l’épinette noire et les feuillus. En outre, notre simulation du scénario d'évolution naturelle (conservation totale) a révélé une augmentation globale des stocks de carbone de la matière organique morte (MOM) et de la biomasse, à l'exception de MU1 sous RCP8.5 où ces stocks ont diminué après 2100. Cette réduction est liée aux effets du stress climatique sur la productivité des conifères. La mortalité des conifères au sein des trois unités a été significativement affectée par la compétition avec les feuillus, les feux et le changement climatique. Le scénario de non-récolte semble conduire à une productivité nette de l'écosystème (PNE) négative ou nulle. Les stratégies basées sur les CT (telles que le scénario de référence) pourraient diminuer le stockage total de carbone de l'écosystème (MOM+biomasse) de -10 tC ha^{-1} an^{-1} par rapport au scénario naturel. Aussi, les stratégies fondées sur les CT ont augmenté l'abondance des jeunes forêts de plus de +25 % et ont diminué les forêts matures et anciennes de -10 % et -16 % respectivement par rapport au scénario naturel, tout en augmentant l'abondance des espèces pionnières de feuillus (peuplier faux-tremble et bouleau blanc). Par contre, les stratégies basées sur les CP (appliqué sur plus de 75 % ou 50 % de la surface annuellement aménagée) permettent un stockage du carbone plus proche ou meilleur que le scénario naturel avec une plus grande rétention de la couverture de conifères avec des forêts matures et anciennes. Pour tous les scénarios climatiques, l'application de stratégies basées sur les CP a imité le scénario naturel, en reproduisant des schémas de végétation naturelle et une structure d'âge similaire. Néanmoins, les stratégies basées sur les CP ont montré un PNE plus faible (~0) à long terme par rapport aux stratégies basées sur les CT. Les scénarios basés sur les CT ont le potentiel d'augmenter le PNE (>0) à long terme et d'augmenter le stockage du carbone de la biomasse disponible pour les produits du bois par rapport à ceux basés sur les CP (PNE~0). Dans les hautes latitudes de la forêt boréale Québécoise, un compromis entre les traitements de CT, CPRS et CP est nécessaire pour atteindre de multiples objectifs, y compris l'augmentation de la séquestration et du stockage du carbone et la satisfaction des besoins industriels. À ces fins, nous avons proposé d'augmenter la superficie récoltée par CP à plus de 50 % ou 75 % de la superficie annuellement aménagée comme stratégies alternatives pour guider la gestion durable des forêts au Québec dans le contexte du changement climatique.

Mots clés : Changement climatique, forêt boréale, séquestration et stockage du carbone, produits du bois, structure d'âge, composition, coupes partielles, coupe totale, LANDIS-II

ABSTRACT

Carbon sequestration by forest ecosystems is a part of the climate change mitigation strategy identified by the intergovernmental panel on climate change. However, boreal forests will be experiencing significant impacts under climate changes, mainly the alterations of natural disturbances cycles (wildfires, epidemics, windthrow). Increasing carbon sequestration in boreal forests and wood products through appropriate harvesting methods could effectively contribute to climate change mitigation at regional scales. In order to stabilize and boost the carbon sequestration and storage capacity at stand scale, partial cuts (PC) based on lower and moderate canopy removal intensities (CRI) have been proposed as an alternative to clear-cutting (CC, corresponding to 95-100% of CRI). Yet, in the Eastern Boreal Forest (Quebec), there is a large gap in the suitable proportion of annual area managed by each CRI at the landscape to maintain forest habitat, maximize carbon sequestration and storage, and fulfill industrial needs, mainly under global change. Under climate change scenarios, this project aims to project and assess the carbon budget dynamics for 300 years (2010-2310) in Quebec-managed forests along a longitudinal gradient, from west to east: North-of-Quebec (MU1), Saguenay-Lac-Saint-Jean (MU2), and Côte-Nord region (MU3). Several management strategies varying in their use of partial cuts, clear cuts, and reforestation rate, including the business as usual (BAU) scenario (clear-cut applied on more than 90% of the managed area) were tested and compared to the natural dynamics scenario (no-harvest). During the simulations, the LANDIS-II model was used as a mechanistic model combined with PnET and ForCs succession extensions in different chapters. At a stand scale, we found that climate change might increase overall biomass carbon stock and net primary productivity under RCP scenarios compared to the baseline for studied species, except coniferous species in MU1 under RCP8.5 after 100 years. This may be explained by the extension of the growing season and the reduction of potential cold temperature injuries. At the landscape scale, natural dynamics results show that the biomass carbon stock of broadleaves will increase under different climate change scenarios during the study period, this could be explained by the expansion of the proportion of the mixed forest, mainly between black spruce and broadleaves. Additionally, our simulation of the natural succession scenario (total conservation) revealed an overall increase in the carbon stocks of dead organic matter (DOM) and biomass except in MU1 under RCP8.5 where these stocks decreased after the period 2100. This reduction was related to climate stress' effects on coniferous species' productivity. Also, the no-harvest scenario appeared to lead to negative or null net ecosystem productivity (NEP). CC-based strategies (such as BAU) could decrease total ecosystem carbon storage (DOM+biomaass) by -10 tC ha^{-1} yr^{-1} compared to the natural scenario. Furthermore, CC-based strategies increased the abundance of young forests by more than +25% and decreased mature and old-growth forests by -10% and -16% respectively compared to the natural scenario, as well as increased the abundance of pioneer broadleaf species (trembling aspen and white birch). Oppositely, PC-based strategies (PC applied on more than 75% or 50% of the managed area) carbon storage is closer or better than the natural scenario with more coniferous cover retention with mature and old-growth forests. The application of PC-based mimicked the natural scenario, by emulating similar natural vegetation patterns and age structure under all climate scenarios. Still, PC-based strategies showed a lower NEP (~0) in the long term compared to CC-based strategies. Nevertheless, scenarios based on CC have the potential to increase NEP (>0) in the long term and increase biomass C storage available for wood products compared to those based on PC (NEP~0). In high latitudes of the Eastern Boreal Forest, a compromise between different CRI (PC and CC) is needed to achieve multiple goals, including increasing carbon sequestration and storage, and fulfilling industrial needs. For these purposes, we proposed to increase the harvested area by PCs to more than 50% or 75% of the annual managed area as alternative strategies to guide sustainable forest management in Quebec in the context of climate change.

Key words: Climate change, boreal forest, carbon sequestration and storage, harvest wood products, age structure, composition, partial cuts, clear cuts LANDIS-II, PnET succession, ForCs succession

CHAPITRE I : INTRODUCTION

1.1 Contexte et problématique

Les forêts canadiennes représentent environ 25 % du carbone de la zone boréale et 10 % de la superficie forestière mondiale, et ils sont considérés comme le biome forestier le plus étendu au monde (FAO, 2020a). Les forêts boréales contribuent de manière significative au cycle du carbone à l'échelle mondiale, et leur stockage de carbone est estimé à 272 PgC, avec un puits moyen constant de 0,5 PgC an^{-1} (Pan et al., 2011), tandis que 60 % de ce stock se trouve dans les réservoirs du sol (163 PgC) (Dixon et al., 1994; Pan et al., 2011). L'accumulation progressive de matière organique est le principal facteur à l'origine de cet important stockage de carbone organique du sol (COS). Ce potentiel élevé de séquestration et de stockage du carbone est le résultat net de la dynamique du carbone dans les différentes régions boréales, associée aux conditions environnementales (y compris le sol et le climat), à l’aménagement forestier et aux perturbations naturelles (Bhatti et al., 2002; Kurz et al., 2013).

Les différentes trajectoires d'émissions et leurs forçages radiatifs respectifs impliquent une variété d'impacts sur les écosystèmes forestiers à différentes échelles spatio-temporelles (Boulanger et al., 2019; Boulanger & Puigdevall, 2021; Taylor et al., 2008; Tian et al., 2015). L'augmentation des températures, les changements dans les régimes de précipitations et la concentration atmosphérique élevée de CO_2 peuvent altérer la croissance et la productivité des forêts, augmenter la mortalité et entraîner des changements dans la composition des espèces (Gustafson et al., 2018). Selon le rapport récent (R6) du Groupe d'experts intergouvernemental sur l'évolution du climat (GIEC), récent (R6) du Groupe d'experts intergouvernemental sur l'évolution du climat (GIEC), les forêts boréales devraient subir des impacts plus importants du changement climatique (IPCC, 2014), y compris l'augmentation de la fréquence et de l'intensité des incendies de forêt, l'augmentation de la fréquence des chablis (Bergeron et al., 2006), et l'apparition plus fréquente de maladies et de ravageurs forestiers tels que la tordeuse des bourgeons de l'épinette [Choristoneura fumiferana (Clemens)] (TBE) (Boulanger et al., 2012; Navarro et al., 2018); ainsi que des changements dans la croissance et la mortalité des espèces (Boulanger & Puigdevall, 2021). En outre, l'abondance des espèces pionnières intolérantes à l'ombre, telles que le peuplier faux-tremble et le bouleau blanc, augmentera sous l'effet du changement climatique, de même que celle des jeunes forêts (Bergeron et al., 2017; Boulanger et al., 2022; Tremblay et al., 2018). Cette situation pourrait devenir encore plus critique en cas d'application des fortes intensités de récolte (IR), comme les coupes totales (CT) (Bergeron et al., 2017). Ces aspects nécessiteront une variété d'actions de gestion afin de garantir une gestion durable de la forêt québécoise et de contribuer à la réalisation du plan 2030 pour une économie verte (y compris la neutralité carbone d'ici 2050) (Kouchaki-Penchah et al., 2022). L’aménagement forestier influence le bilan carbone au niveau de la biomasse, du sol et des produits ligneux récoltés. Selon plusieurs études, les pratiques sylvicoles, notamment les CP, peuvent contribuer à augmenter la séquestration future du carbone dans les forêts boréales (Ameray et al., 2021; Dixon, 2009; Simard et al., 2020; Taylor et al., 2008). Au Québec, l’aménagement inéquienne basée sur les coupes partielles (CP) avec différentes IR inférieures à 80%, et l’aménagement équienne basée sur les CT (100% d’IR) et la coupe avec protection de la régénération et des sols (CPRS) avec 95% d’IR sont les deux systèmes sylvicoles les plus utilisés (MRNF, 2010 ; Simard et al., 2020). Afin de garantir une gestion durable des forêts, il est nécessaire de comprendre la dynamique du carbone de la forêt boréale en relation avec le climat, les perturbations naturelles et l'aménagement (équienne *vs* inéquienne). Cependant, malgré l'importance des stocks de carbone de ces forêts, les données et la compréhension des impacts de ces facteurs sur le bilan du carbone et la manière dont ils interagissent les uns avec les autres à court et à long terme sont limitées.

Les paramètres climatiques (précipitations et températures) ont de nombreux effets potentiels sur la croissance des forêts, la mortalité des arbres, les perturbations, le cycle des nutriments du sol et tous les autres processus qui modulent le fonctionnement des écosystèmes forestiers et, par conséquent, leur dynamique du carbone (Davidson & Janssens, 2006; Peng et al., 2011). White et al. (1999) et D'Orangeville et al. (2018) prévoient que la productivité primaire nette (PPN) de la forêt boréale au Québec augmentera sous l'effet du changement climatique. De plus, le réchauffement climatique augmentera la respiration hétérotrophe de l'écosystème (R_h), ce qui diminuera la production nette de l'écosystème (PNE). D'Orangeville et al. (2018) s’attendent à ce que les forêts boréales accumulent davantage de carbone dans la biomasse vivante sous l'effet du changement climatique. En revanche, le réchauffement climatique augmentera les émissions de carbone provenant des sols en accélérant le processus de décomposition et les émissions directes de CO_2 dues aux perturbations naturelles (principalement les incendies de forêt). Également, sous l'effet du changement climatique, les incendies, les chablis et les épidémies de TBE sont les principales perturbations naturelles au Québec qui joueront un rôle majeur dans la modification de la structure d'âge et la composition future des forêts (Bergeron et al., 2006; Boulanger et al., 2014; Navarro et al., 2018).

Pour une gestion durable des forêts québécoises, il est nécessaire de comprendre les liens et l'équilibre entre la séquestration du carbone et les pratiques d’aménagement, ainsi que leurs interactions avec les perturbations naturelles. Le MRNF du Québec a mis en œuvre des politiques visant à soutenir la gestion durable des forêts tout en favorisant une industrie forestière robuste (Gouvernement du Québec, 2022). Cela inclut des plans pour l'exploitation forestière, soutenir la certification forestière en forêt privée, la recherche sur les pratiques de sylviculture durable, et la promotion des technologies innovantes dans le secteur forestier. Face aux défis posés par le changement climatique, le MRNF a élaboré des stratégies visant à renforcer la résilience des écosystèmes forestiers; telle que la promotion de pratiques sylvicoles adaptées aux conditions changeantes, la surveillance continue des impacts climatiques sur les forêts, et l'intégration des connaissances scientifiques actuelles dans la prise de décision.

Actuellement, les modèles empiriques, notamment CBM-CFS3, dominent dans la modélisation de la dynamique du carbone en réaction à diverses perturbations naturelles et anthropogéniques. Ce n'est que récemment que l'évaluation des impacts des changements climatiques sur le cycle du carbone forestier est devenue possible. Cette évaluation s'effectue grâce à l'utilisation de modèles mécanistiques de simulation qui intègrent les cycles du carbone, de l'azote et de l'eau entre les arbres et le sol. Ces modèles capturent les interactions complexes entre l'atmosphère et la végétation, ainsi que les processus compétitifs entre les arbres (Alvarez et al., 2016; Gustafson et al., 2015; Wu et al., 2017; Zhuo et al., 2020). Bien que des modèles empiriques tels que CBM-CFS3 puissent partiellement répondre à ces questions et analyser la dynamique du carbone forestier (Kurz et al., 2009), ils ne tiennent pas compte de certains processus cruciaux liés à la photosynthèse, à la compétition pour la lumière et l'eau, ainsi qu'à d'autres facteurs climatiques et paysagers influençant la productivité et la succession des forêts (Kim et al., 2015). Notre recherche se fonde sur l'utilisation du modèle de paysage forestier Landis-II conjointement avec le modèle écophysiologique et mécanistique PnET (Photosynthetic / Evapotranspiration model). Cette approche nous permet d'étudier la productivité forestière, y compris la PPN et la biomasse, sur une échelle temporelle et spatiale étendue, à court (2010-2110), moyen (2111-2210) et long terme (2211-2310) à grande échelle (>0.5 mha). De plus, le troisième chapitre de notre recherche utilise l'extension de succession ForCS et Landis-II pour examiner tous les pools et flux de carbone, y compris le sol, la biomasse et le transfert de carbone vers le bois sur une période de 200 ans (2010-2210). Ces analyses sont menées sous divers scénarios de changement climatique et de stratégies d'aménagement forestier.

L'objectif général de cette thèse était d'évaluer et de prédire les effets à long terme des changements climatiques sur la séquestration et le stockage du carbone, en tenant compte de différentes stratégies d'aménagement forestier ainsi que des perturbations naturelles telles que les incendies, le chablis et la TBE. Dans le cadre de la stratégie du ministère sur les pratiques sylvicoles adaptées aux conditions changeantes, nous avons examiné si l'inclusion des CP améliorera la séquestration et le stockage du carbone à l'échelle du paysage sous différents scénarios climatiques. Cette thèse propose de nouvelles stratégies pour maintenir et augmenter la capacité de production des forêts boréales du Québec et leur durabilité en considérant l'importance d'adapter les pratiques forestières aux impacts du changement climatique dans le temps à l'échelle régionale.

1.2 Dynamique du carbone forestier et perturbations naturelles

Les incendies de forêt sont le principal agent de changement dans la zone boréale, affectant environ 2.4 Mha an^{-1} (NRC, 2022). Leurs impacts sur les écosystèmes forestiers sont généralement bien décrits par Bergeron et al. (2004, 2006). Ils modifient les flux et les réservoirs de carbone et entraînent des conséquences à grande échelle en raison de leur contribution aux émissions de carbone dans l'atmosphère, libérant environ 27 TgC an^{-1} (Amiro et al., 2001). De même, ils affectent la croissance, la décomposition, la régénération et la succession des forêts (Bergeron, 2000; Bergeron et al., 2004). En fonction de leur sévérité, les graines d'espèces sérotineuses peuvent être stimulées, ce qui permet l'établissement d'un nouveau peuplement avec un potentiel de croissance différent du précédent (Mladenoff & He, 1999). Ce processus implique une forêt jeune avec une PPN élevée, comme celles dominées par le pin gris (*Pinus banksiana*), qui se régénère même après de sévères incendies. Bergeron et al.

(1999) et Molina et al. (2021) affirment que la taille et la sévérité des incendies contrôlent la transition des forêts mixtes vers les forêts de conifères, en modifiant la composition et la structure des peuplements. Ces modifications ont une influence significative principalement sur la PPN et le COS, puisqu'elles modifient la composition et la diversité des espèces (Cavard et al., 2010). Les régions plus humides de l'est du Québec (par exemple la Côte-Nord) sont souvent moins touchées par les incendies que les régions de l'ouest et du centre (par exemple l'Abitibi, la région du lac Saint-Jean), à la suite de l'augmentation des températures estivales et des sécheresses (Bergeron et al., 2010; Boulanger et al., 2014).

Les épidémies de TBE constituent l'une des principales perturbations naturelles dans l'est du Canada (Québec) (Boulanger et al., 2012; Navarro et al., 2018), et entraînent une perte considérable de la biomasse aérienne, ce qui a un effet négatif sur le bilan du carbone séquestré (Volney & Fleming, 2000). Dans l’est du Québec (région de la Côte-Nord), plus de 50 % de la perte de volume annuelle est causée par la défoliation des arbres due à la TBE (Navarro et al., 2018), tandis que les conséquences de la défoliation restent relativement modérées dans les régions de l’ouest. La TBE est plus fréquente du côté nord-est du Québec en raison de plusieurs facteurs liés à l'environnement, au climat et à la dynamique des populations d'insectes. Par exemple, les hivers froids et les étés tempérés de la région nord-est du Québec créent un environnement propice au cycle de vie de la TBE, favorisant sa reproduction et sa survie. La TBE est responsable de pertes économiques importantes pour l'industrie forestière en raison de la mortalité élevée des arbres et de la perte de productivité des forêts (MacLean, 2016). En 2018, les populations de TBE sont en hausse et des signes de défoliation ont été observés, les superficies touchées par la tordeuse des bourgeons de l’épinette ont varié de 8,2 à 13,5 Mha de forêt boréale québécoise (MRNF, 2018). Cependant, leur impact sur le bilan carbone dépend de l’échelle spatio-temporelle et de leur cycle. Boulanger et al. (2012) signalent que la TBE a une périodicité d’environ 32 ans dans les forêts du Québec. L'étendue spatiale d'une épidémie sera également influencée par la dynamique des populations d'insectes en fonction des influences climatiques et/ou biotiques, de l'abondance des espèces hôtes et des pratiques sylvicoles (Bouchard et al., 2017). Les cycles temporels de TBE et leurs amplitudes (par exemple, leur sévérité) varient d'un paysage à l'autre, car ils sont influencés par l'abondance des espèces hôtes (sapin baumier, épinette blanche, épinette noire) (Robert et al., 2018). Les épidémies de TBE diminuent la quantité de carbone dans la biomasse et augmentent celle dans la litière et les débris ligneux, ce qui augmente la respiration hétérotrophe (R_h) et réduit le PNE (Liu et al., 2019).

Les chablis représentent aussi l'une des perturbations forestières les plus critiques au Québec, provoquant la mortalité des arbres par leur déracinement (ou leur cassure par le vent). Ils ont un impact potentiel sur le cycle du carbone (De Grandpré et al., 2018), mais leurs effets sont moindres que ceux des incendies et de la TBE, puisqu'ils ont touché en moyenne 0,0255% de la superficie par an entre 1971 et 2000 (Bouchard et al., 2009a). Après un chablis, la respiration de l'écosystème augmente principalement en raison de la décomposition des débris ligneux grossiers résiduels, tandis que la PPN annuelle diminue (Yamanoi et al., 2015). En outre, l'augmentation de la respiration hétérotrophe du sol entraîne une diminution des stocks de carbone organique du sol (Mayer et al., 2017). Bouchard et al. (2009b) et Rich et al. (2007) affirment que le schéma de mortalité des arbres par chablis dépend des espèces, de l'âge du peuplement et des pratiques sylvicoles (Girona et al., 2019a). La mortalité causée par le TBE et les chablis est plus importante dans les régions de l'est du Québec (Bergeron & Leduc, 1998; Perry et al., 2008).

Afin de déterminer avec précision la dynamique du carbone à long terme, une approche à l'échelle du paysage est nécessaire pour prendre en compte ces perturbations. En général, la sévérité élevée de toutes les perturbations décrites ci-dessus peut retarder le rétablissement de la PPN d'un peuplement. En fait, l'évolution de l'activité photosynthétique et l'augmentation de la PPN de la forêt après une perturbation sont liées aux nouvelles informations structurelles du peuplement (par exemple, la composition des espèces et l'indice de surface foliaire), à l'humidité du sol, à la température, aux nutriments et au microclimat (Liu et al., 2011; Peng et al., 2011). La TBE et les chablis entraînent une perte de carbone par la mortalité des arbres et transfèrent le carbone des réserves de biomasse vivant aux réserves de bois mort, où la décomposition (augmentation du R_h) libère progressivement une fraction (F) du carbone dans l'atmosphère, et (1-F) dans la réserve de sol (sol minéral de 5 cm à 10 cm) (Don et al., 2012; Dos Santos et al., 2016; Kurz et al., 2009).

1.3 Dynamique du carbone forestier et pratiques d’aménagement

Au Québec, l'exploitation forestière dépend du milieu physique, de la capacité de production de bois, de la vulnérabilité de la forêt aux incendies et de la conservation de la biodiversité (Jobidon et al., 2015). Une approche analytique a été développée par Jobidon et al. (2015) pour évaluer l'aptitude des unités de territoire à la production de bois, en considérant ces critères pour assurer l'aménagement forestier durable (AFD). Cette approche permet de classer le territoire québécois en zones peu sensibles (AFD possible), moyennement sensibles (AFD possible sous certaines conditions) et très sensibles. Notre recherche a été menée dans la zone très sensible près de la limite nordique d’exploitation forestière, le long d'un gradient longitudinal où différents traitements sylvicoles sont appliqués, y compris les CT, CPRS, et CP avec différentes IR (25 %, 50 %, 75 %). Depuis 1990, la CPRS est appliquée en forêt boréale québécoise, ce traitement est similaire au CT, sauf qu'il protège la régénération avec un impact moindre sur les sols (Gauthier & Vaillancourt, 2008). Selon la géodatabase de l'inventaire forestier (MRNF, 2010), dans la zone d’étude, plus de 95 % de la superficie annuelle récoltée pour l'approvisionnement en bois est actuellement gérée par les CT et les CPRS. L'effet des CPRS sur la dynamique du carbone est similaire à celui du CT, lorsque le stock semencier du sol et le taux de régénération sont plus faibles.

La récolte pourrait avoir un impact sur les flux et le stockage du carbone, en fonction de la durée de rotation et de l’intensité de récolte (IR) (Ameray et al., 2021). Price et al. (2005) indiquent qu'une rotation plus courte réduit la densité moyenne de la biomasse (par unité de superficie forestière) et que l'augmentation de l’intensité de récolte entraîne une réduction du stockage total du carbone dans l'écosystème, l'ampleur de ces pertes étant corrélée aux conditions climatiques (Simard et al., 2020). Aussi, Dixon (2009) rapporte que les pratiques sylvicoles (par exemple, l'éclaircie commerciale) peuvent être employées pour séquestrer 1 à 64 MgC ha^{-1} dans les biomes boréaux, tempérés et tropicaux. Peng et al. (2002) ont conclu que les CP pourront augmenter la séquestration du carbone d'environ 36 à 40% dans la région de la forêt boréale à long terme, car elle réduit la concurrence pour la lumière, l'eau et les nutriments, améliorant ainsi la capacité de séquestration du carbone des peuplements résiduels (Carroll et al., 2012). Il existe différentes formes des CP appliquées en forêt boréale québécoise, telles que l'éclaircie commerciale, la coupe sélective, la coupe progressive (régulière ou irrégulière) et la coupe avec protection de petites tiges marchandes (appelée CPPTM au Québec) (Girona et al., 2016, 2017). Les CP consistent à récolter un pourcentage du volume forestier commercialisable, les taux de rétention des arbres après CP pouvant aller jusqu'à 80% (Ameray et al., 2021).

D'autre part, 95 à 100 % de l'IR associé aux traitements des CT et CPRS peuvent augmenter les quantités de rayonnement solaire et de précipitations qui atteignent la surface du sol, augmentant ainsi la température et l'humidité du sol, et par conséquent augmentant la respiration hétérotrophe et la réduction de la PNE (Ameray et al., 2021). Taylor el al. (2008) ont constaté qu'au niveau des peuplements, les CP augmentaient le

C total de l'écosystème des peuplements de 308.9 à 327.3 Mg ha^{-1} le long de la simulation de 240 ans, alors que les CT diminuaient ce stock à 305.8 Mg ha^{-1}. Cependant, à l'échelle du paysage, le CT peut accélérer l'abondance de jeunes forêts, augmentant ainsi la NPP et la PNE par rapport aux CP (Ameray et al., 2021). Dans certaines zones présentant des conditions spécifiques telles que les forêts paludifiées, le CPRS favorise l'accumulation de matière organique en baissant la décomposition, réduisant ainsi la PPN à long terme (Fenton et al., 2010; Lafleur et al., 2010; M. Lavoie et al., 2005). En ce qui concerne le COS, les coupes ont un impact faible ou moindre (Simard et al., 2008; Taylor et al., 2008).

Les forêts ne sont pas gérées sur la base d'un seul peuplement, mais à l'échelle du paysage (Ameray et al., 2021), exigeant qu'une stratégie d'aménagement forestier adaptative visant à augmenter ou stabiliser la capacité de séquestration du carbone doive tenir compte de la surface annuellement récoltée par traitement. Au cours de la dernière décennie, pour assurer le maintien de la biodiversité ainsi que la viabilité et la fonctionnalité des écosystèmes, le gouvernement du Québec a adopté l'aménagement forestier écosystémique (AFE) comme outil d'aménagement durable, en réduisant les écarts entre les forêts naturelles et aménagées (Bergeron et al., 1999, 2001; Gauthier & Vaillancourt, 2008). L'AFE s'inspire principalement des perturbations naturelles et suppose que les pratiques sylvicoles peuvent avoir des impacts similaires à ceux causés par les perturbations naturelles, selon leur ampleur et leur sévérité (Bergeron et al., 1999; Dhital et al., 2015). Afin d'atteindre l'AFE, Bergeron et al. (1999) a développé l'approche par cohorte, qui vise à maintenir la composition et la structure naturelle du peuplement et à produire une composition identique au scénario naturel. Les stratégies basées sur les CP peuvent maintenir une structure de végétation et d'âge similaire à l'évolution naturelle (Girona et al., 2023; Martin et al., 2022), mais leurs effets dans le cadre du changement global ne sont pas bien étudiés.

1.4 Méthode générale, Objectifs et hypothèses

Dans notre étude, nous avons utilisé le modèle Landis-II, un modèle basé sur les processus (MBP) largement reconnu pour sa capacité à intégrer les relations fonctionnelles au sein des écosystèmes (Mladenoff & He, 1999; Scheller et al., 2007). Les MBP, tels que Landis-II, fonctionnent selon le principe de la modélisation mécaniste, dans lequel le comportement d'un système est simulé par une série de sous-modèles décrivant les processus de causalité (Landsberg et al. 2011). Ces processus, qui englobent les interactions arbre-sol-climat, visent à élucider la dynamique d'un système donné. La spécificité et le pouvoir prédictif des MBP, comme Landis-II, résident dans leur description détaillée des mécanismes liés à la croissance, ce qui permet une compréhension globale des changements dans l'état des arbres, des peuplements ou des cohortes (Mladenoff & He, 1999). Au niveau de l'écosystème, divers processus entrent en jeu, et les MBP offrent un cadre précieux pour tester des hypothèses et prédire les interactions en cas de changements environnementaux. Dans le contexte de notre étude, LANDIS-II est apparu comme un puissant outil de prévision du paysage, capable de simuler des forêts sur des échelles de temps allant de la décennie à plusieurs siècles et sur des échelles spatiales allant de centaines à des millions d'hectares sous différents scénarios de changements climatiques (Scheller et al., 2007). Ce modèle excelle à saisir la succession forestière, les perturbations (telles que les incendies, le vent et les épidémies), l’aménagement forestier, la dispersion des graines, la dynamique du carbone et les impacts du changement climatique (Figure 1.1). La polyvalence de LANDIS-II est encore renforcée par l'intégration de sous-modèles (ou d'extensions), facilitant la simulation de divers processus et perturbations à l'échelle du paysage.

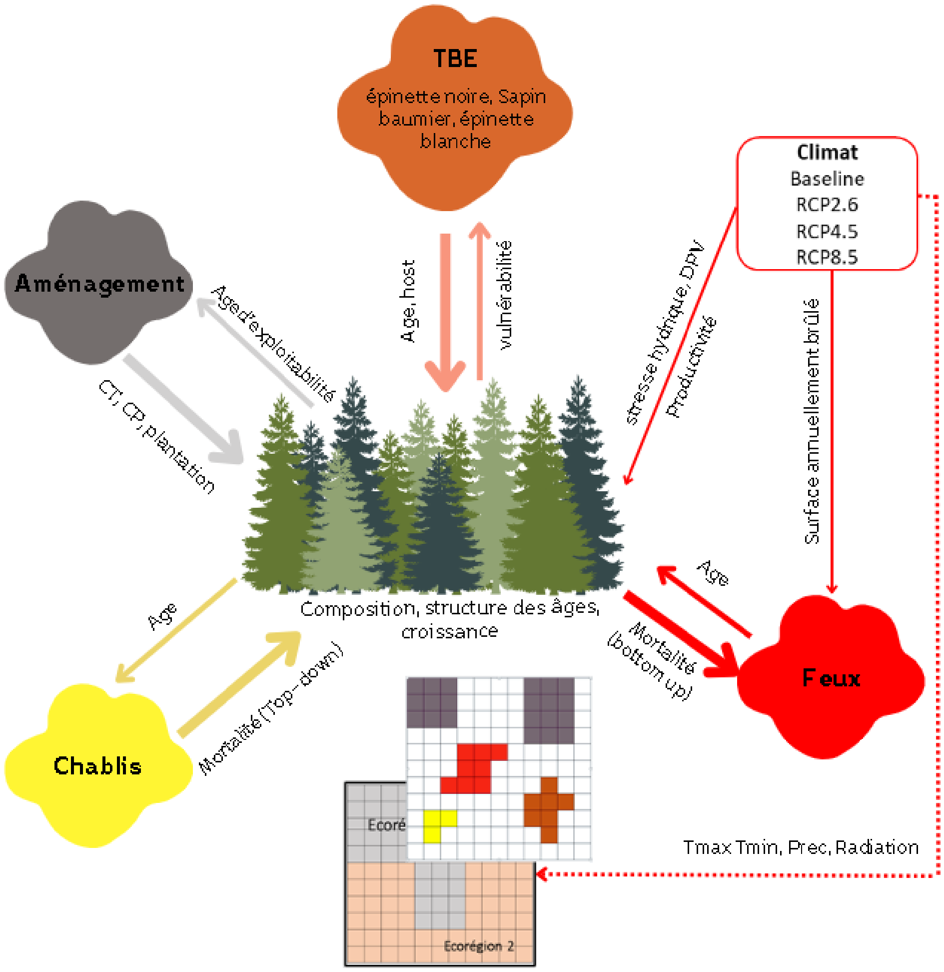

Figure 1.1 Description générale du modèle LANDIS-II utilisé pour modéliser l'impact des perturbations naturelles et anthropiques sur la séquestration et le stockage du carbone.

Dans le premier article de notre étude (chapitre 2), nos objectifs spécifiques étaient de projeter la composition future (2010-2310) de la forêt boréale selon différents scénarios de changements climatiques, de modéliser la séquestration du carbone à l'échelle du peuplement et du paysage dans le contexte du changement climatique, et de modéliser l'effet de la dispersion/régénération des feuillus sur la composition et la capacité de stockage du carbone. L'extension PnET a été utilisée pour simuler la croissance et la succession, permettant ainsi le suivi de la biomasse et de la composition forestière au niveau du paysage. En se basant sur la littérature, nous avons émis l'hypothèse initiale selon laquelle, sous l'impact du changement climatique, les forêts boréales pourraient accumuler davantage de carbone dans la biomasse vivante des feuillus tout en réduisant celle des conifères (Dixon 2009; D'Orangeville et al., 2018 ; Molina et al., 2021 ; Wang et al., 2023).

Dans le deuxième article (chapitre 3), nos objectifs spécifiques étaient de prédire l'impact du feu, de la TBE et du chablis sur le stockage du carbone dans la biomasse en fonction des scénarios de changement climatique, de projeter l'effet de l'aménagement sur la composition et la structure d'âge dans le cadre de ces scénarios, et de comparer les stratégies d'aménagement forestier pour stabiliser et maximiser le stockage du carbone. La différence entre les systèmes équiens (en utilisant CT et CPRS) et les systèmes inéquiens (en utilisant CP) a été étudiée. L'extension PnET avec LANDIS-II a été réutilisée pour la croissance et la succession, permettant le suivi de la biomasse, de la composition, de la structure des âges, et du volume de bois récolté. Notre hypothèse suggère que l'inclusion des CP dans l'aménagement à l'échelle du paysage pourrait améliorer le stockage du carbone dans la biomasse, avec une augmentation potentielle du capacite de séquestration du carbone dans la région de la forêt boréale (D'Orangeville et al., 2018 ; Peng et al., 2002 ; Taylor et al., 2008). De plus, les CP pourrait maintenir davantage de vieilles cohortes et une plus grande rétention de la couverture coniférienne, tandis que les CT pourraient augmenter l'abondance des jeunes forêts et des feuillus.

Enfin, dans le troisième article (chapitre 4), étant donné que le modèle utilisé dans les deux articles précédents ne permettait pas de suivre la dynamique du carbone au niveau du sol, nous avons utilisé une autre extension appelée Forest Carbon Succession. Nos objectifs spécifiques étaient de modéliser l'impact du changement climatique et de l'aménagement sur le carbone du sol, de quantifier les effets du changement climatique sur la dynamique du carbone en simulant un scénario dynamique naturel, et d'étudier l'effet des stratégies d'aménagement (équiennes vs inéquiennes) sur la dynamique du carbone (flux et réservoirs). Nous avons suivi des variables telles que les flux de carbone, les réservoirs de carbone, la composition, la structure des âges et le volume de bois récolté. Les hypothèses formulées pour ce chapitre sont les suivantes : a) Les scénarios de conservation augmenteraient le carbone total de l'écosystème en maintenant des vielles forêts et une plus grande abondance d'espèces de conifères ; b) Les scénarios basés sur les CT pourraient réduire le carbone total de l'écosystème et améliorer la production nette de l'écosystème en augmentant l'abondance des jeunes forêts et des espèces feuillus ; c) Les scénarios basés sur les CP peuvent imiter les scénarios de conservation en augmentant le carbone total de l'écosystème et l'abondance des vielles forêts, bien que la production nette de l'écosystème soit inférieure à celle des scénarios basés sur les CT.

CHAPTER II: CLIMATE CHANGE MAY INCREASE QUEBEC BOREAL FOREST PRODUCTIVITY IN HIGH LATITUDES BY SHIFTING ITS CURRENT COMPOSITION

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CHAPITRE II : LE CHANGEMENT CLIMATIQUE POURRAIT AUGMENTER LA PRODUCTIVITÉ DE LA FORÊT BORÉALE QUÉBÉCOISE AUX HAUTES LATITUDES EN MODIFIANT SA COMPOSITION ACTUELLE

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Abstract

Several recent studies point out that climate change is expected to influence boreal forest succession, disturbances, productivity, and mortality. However, the effect of climate change on those processes and their interactions is poorly understood. We used an ecophysiological-based mechanistic landscape model to study those processes and their interactions and predict the future productivity and composition under climate change scenarios (RCP) for 300 years (2010-2310). The effects of climate change and wildfires on forest composition, biomass carbon sequestration and storage, and mortality were assessed in three management units of Quebec boreal forest, distributed along a longitudinal gradient from west to east: North-of-Quebec (MU1), Saguenay-Lac-Saint-Jean (MU2), and Côte-Nord region (MU3). Coniferous mortality variation was explained by competitive exclusion and wildfires, which are related to climate change. In the studied MU, we found a decrease in coniferous pure occupancy at the landscape scale and an increase in mixed deciduous forests in MU1 and MU2, and an increase in mixed coniferous, mainly black spruce and balsam fir in MU3. On the other hand, for extreme scenarios (RCP8.5), in the absence of broadleaves dispersal, the open woodland occupancy could increase to more than 8%, 22%, and 10% in MU1, MU2, and MU3 respectively. Also, climate change might increase overall biomass carbon stock two times for RCP2.6 and RCP4.5 scenarios compared to the baseline this may be explained by the extension of the growing season and the reduction of potential cold-temperature injuries. Generally, western regions were more sensitive to climate changes than the eastern regions (MU3), in fact under RCP8.5 biomass carbon stock will be decreasing in the long term for MU1 compared to the current climate. This study provides a good starting point to support future research on the multiple factors affecting forest C budget under global change.

Keywords: forest carbon, climate change, wildfires, PnET- Succession, LANDIS-II, competition, mortality

Résumé

Plusieurs études récentes soulignent que le changement climatique devrait influencer la succession, les perturbations, la productivité et la mortalité des forêts boréales. Cependant, l'effet du changement climatique sur ces processus et leurs interactions est mal compris. Nous avons utilisé un modèle de paysage mécaniste basé sur l'écophysiologie pour étudier ces processus et leurs interactions et prédire la productivité et la composition future en fonction des scénarios de changement climatique (RCP) pour 300 ans (2010-2310). Les effets du changement climatique et des feux de forêt sur la composition forestière, la séquestration et le stockage du carbone dans la biomasse et la mortalité ont été évalués dans trois unités d’aménagement (MU) de la forêt boréale québécoise, réparties le long d'un gradient longitudinal d'ouest en est : Nord du Québec (MU1), Saguenay-Lac-Saint-Jean (MU2) et Côte-Nord (MU3). La variation de la mortalité des conifères a été expliquée par les incendies de forêt et par la compétition avec les feuillus, qui sont liés au changement climatique. Dans les MU étudiés, une diminution de l'occupation des conifères pure à l'échelle du paysage été observé et une augmentation des forêts mixtes entre les feuillus dans MU1 et MU2, ainsi qu'une augmentation des conifères mixtes, principalement des épinettes noires et des sapins baumiers dans MU3. En revanche, pour les scénarios extrêmes (RCP4.5 et RCP8.5), en l'absence de dispersion des feuillus, l'occupation des forêts ouvertes pourrait augmenter jusqu'à plus de 8 %, 22 % et 10 % dans les zones MU1, MU2 et MU3 respectivement. En outre, le changement climatique pourrait doubler le stock global de carbone de la biomasse pour les scénarios RCP2.6 et RCP4.5 par rapport à la situation de référence, ce qui peut s'expliquer par l'extension de la période de croissance et la réduction des blessures potentielles dues aux températures froides. En général, les régions d'Ouest (MU1) sont plus sensibles aux changements climatiques que les régions d'Est (MU2 et MU3). En fait, dans le cadre du scénario RCP8.5, le stock de carbone de la biomasse diminuera à long terme pour MU1 par rapport au climat actuel. Cette étude constitue un bon point de départ pour soutenir les recherches futures sur les multiples facteurs qui affectent le bilan de carbone des forêts dans le cadre du changement climatique.

Mots clés : carbone forestier, changement climatique, incendies, PnET-Succession, LANDIS-II, compétition, mortalité

2.1 Introduction

The boreal forest is the second largest biome on Earth and is the subject of increasing interest for its role in the terrestrial carbon cycle. Those ecosystems occupy 27% of world forests in 2020, dominating the subarctic northern latitudes of Eurasia and America, providing 37% of global wood consumption, and other ecosystem services that benefit society at levels ranging from local to global (Ameray et al., 2021; FAO, 2020a; Gauthier et al., 2015). They store 88 Pg of carbon in living biomass (aboveground and belowground), 471 Pg in soil and sequester 0.5 Pg yr^{-1}, and contain 60% of the world's soil organic carbon (SOC) (Dixon et al., 1994; Pan et al., 2011). However, numerous studies carried out in the boreal forest and elsewhere have shown that climate change will have considerable impacts on forest mortality, composition, and carbon storage (Bergeron et al., 2006; Boulanger et al., 2014; D'Orangeville et al., 2018; Molina et al., 2021; Pan et al., 2011).

Recently, the climate has been warming as a result of augmentation in radiatively active gases in the atmosphere caused by human activities (IPCC, 2014). The exchanges of carbon between vegetation and the atmosphere are essential determinants of regional climate and the global carbon budget (Schimel et al., 2001). In Quebec boreal forests, climate change reveals a large positive effect of increasing thermal energy on forest productivity leading to 20.5 to 22.7% projected gains in growth with climate change under RCP 4.5 and RCP8.5 (Wang et al., 2023). In the high latitudes where the boreal forests are situated, increasing temperatures directly affect tree growth and SOC decomposition through effects on photosynthetic and heterotrophic respiration rates respectively, and it is one of the main direct results of climate change which has been observed (Curiel et al., 2007; D'Orangeville et al., 2018). An increase in temperatures may stimulate vegetation growth by increasing the growing season length, but in areas more prone to droughts an increase in temperature might reduce productivity when the tree foliage closes its stomata by regulation process, or even cause increased mortality rates (Peng et al., 2011). Also, higher rates of temperatures and drought frequency may increase the intensity and frequency of wildfires, mainly in the boreal forest (Bergeron et al., 2006; Boulanger et al., 2014).

Wildfires are one of the most significant natural disturbances in the Canadian boreal zone, as there are annually an average of 7,500 fires, burning approximately 2.4 million ha (NRC, 2022). In addition, they affect carbon pools, mostly aboveground biomass, litter, and soil, and any change in fire regimes can have large-scale consequences through their contribution to atmospheric carbon emissions (Peng & Apps, 2000). Similarly, they affect forest growth, decomposition, forest regeneration and succession, and control stand age class distribution (Bergeron et al., 1999, 2002; Peng & Apps, 2000). Also, fire frequency and intensity affect vegetative reproduction and the regeneration of dispersed or serotinous species, so when the intensity of fires is high, the regeneration opportunities for widely dispersed or serotinous species are favored (Mladenoff & He, 1999). Generally, when the intervals between fires are longer, forests are mainly dominated by conifers while when the intervals between fires are shorter, forests are dominated by early successional species (mostly broadleaves and some conifers like Jack pine and black spruce) (Bergeron et al., 2014). Johnstone et al. (2010) state also that when fire intervals are short, black spruce tends to become less dominant.

Landscape models for forest carbon modelling are usually based on empirical models rather than mechanistic ones, and may not be robust to global changes, which are producing novel conditions that forests have not experienced historically. Landsberg et al.(2011) and Kim et al.(2015) have extensively discussed the advantages and disadvantages of using empirical and mechanistic process-based models in spatial forest management. The empirical models based on national forest inventories are widely used for the assessment of carbon sequestration (e.g., CBM-CFS3, CO2FIX, CASMOFOR, and EFISCEN) (Kim et al., 2015; Kurz et al., 2009; Landsberg et al., 2011). In boreal forests, CBM-CFS3 is commonly used to provide relatively more detailed-level compartments and volume-to-biomass conversions in the processes of growth (Kurz et al., 2009). However, these models do not treat explicitly all of the processes occurring in the ecosystem, as they are mainly based on statistical relationships to estimate forest development (Kim et al., 2015; Kurz et al., 2009). On the other hand, process-based models (PBMs) integrate a set of functional relationships (tree-soil-climate), and their interactions with each other, in order to describe the behavior of a given system (Huang et al., 2018). There are multiple processes such as succession, disturbances, and land-use changes, occurring at the ecosystem level; PBMs offer a framework for testing alternative hypotheses and help us accurately describe how those processes will interact under a given environmental change (Cook et al., 2008).

In the boreal forest climate change and wildfires will affect forest composition and structure at the landscape scale. Consequently, young forests dominated by pioneer broadleaf species (e.g. *Populus tremuloides*) are projected to increase in relative abundance, while the coniferous boreal forest is expected to decrease significantly mainly in transition zones (boreal-temperate) (Boulanger et al., 2017; Molina et al., 2021). In the Quebec spruce-feathermoss bioclimatic zone, increases in droughts and frequent vast fires will likely drive a transition to open lichen woodlands (Augustin et al. 2022; Stralberg et al., 2018). However, species growth and establishment are related to site conditions. This process depends on 1) how far seeds disperse to new sites within a given timeframe, 2) the new site soil and climate conditions, 3) how readily new seeds can produce established cohorts, 4) how quickly newly established cohorts reach sexual maturity, 5) competition with resident and other migrating species, and 6) natural and anthropogenic disturbances (e.g., harvest, wildfire, insects, and disease) that interact to affect competition for light and water (Liang et al., 2018).

In this study, we used PnET-Succession of LANDIS-II as PBMs to improve our understanding of the short, medium, and long-term effects of climate change and wildfires on boreal forest composition, mortality, and biomass carbon storage at the landscape scale (Gustafson et al., 2015; Mladenoff & He, 1999; Scheller et al., 2007). This PBM uses physiological first principles to mechanistically account for the effects of temperature, light, and water availability on photosynthesis (competition and growth) and includes simulation of seed dispersal and establishment. Based on the literature, we first hypothesized that in the future boreal forests might accumulate more C in living tree biomass for broadleaves and decrease those of coniferous under climate change (Bergeron et al., 2014; D'Orangeville et al., 2018; Molina et al., 2021; Wang et al., 2023). Also, coniferous mortality will increase under climate change, because of wildfires and competition with broadleaves (Boulanger & Puigdevall, 2021). Therefore, near the Québec northern logging limit (the limit beyond which forests are not managed because of their lower productivity), along a longitudinal gradient in three management units (MU) and under different scenarios of climate change (RCP) and wildfires frequency and severity, we aim to 1) project the future boreal forest composition, 2) forecast biomass carbon storage at the landscape scale, 3) assess the effect of climate and the cause (competitive exclusion or wildfires) on forest mortality, and finally, 4) carbon sequestration was evaluated under different climate change scenarios. The scope of this study did not include human interventions (e.g., reforestation, management) and disturbances other than fire (disease and pests). We highlighted this study's aims as well to calibrate the model and perform an extensive sensibility analysis. This study identifies the parameters that mostly influence the outputs of LANDIS-II/PnET-Succession and consider interactive effects as well. Consequently, it offers to the users a more complete understanding of model uncertainty and responses to variations in the inputs.

2.2 Materials and Methods

2.2.1 Study area

The study was conducted along a longitudinal gradient of Québec’s boreal forest (Figure 2.1). Forest landscapes were simulated within three management units (MU): UM08551 in Nord-du-Québec (MU1), UM2471 in Saguenay-Lac-Saint-Jean (MU2), and UM09351 in the Côte-Nord region (MU3). Each MU encompasses a wide variety of forest types, soils, slopes, and local climate conditions. Also, those MU are located in several sensitive areas near the northern limit of Québec managed forest and belong to the spruce-feathermoss and balsam fir- white birch bioclimatic domains of the Boreal Shield, dominated by black spruce (*Picea mariana*), jack pine (*Pinus banksiana*), balsam fir (*Abies balsamea*), white birch (*Betula papyrifera*), trembling aspen (*Populus tremuloides*) and other species (MRNF, 2010). All commercial tree species present in these landscapes are summarized in Table 2.1 and simulated during the experiments. From the western to the eastern region the annual precipitation average ranges from 824 mm to 970 mm with a high longitude gradient effect (Wang et al., 2016). In each MU, the MRNF (Ministère des Ressources naturelles et des Forêts) ecoregions were considered (Figure 2.1), and the unproductive forest land was excluded for the analysis including water bodies, wetlands, islands, and other no-commercial species (inactive cells). On the other hand, the productive cells that were recently disturbed in 2010 (after logging or wildfires) were activated.

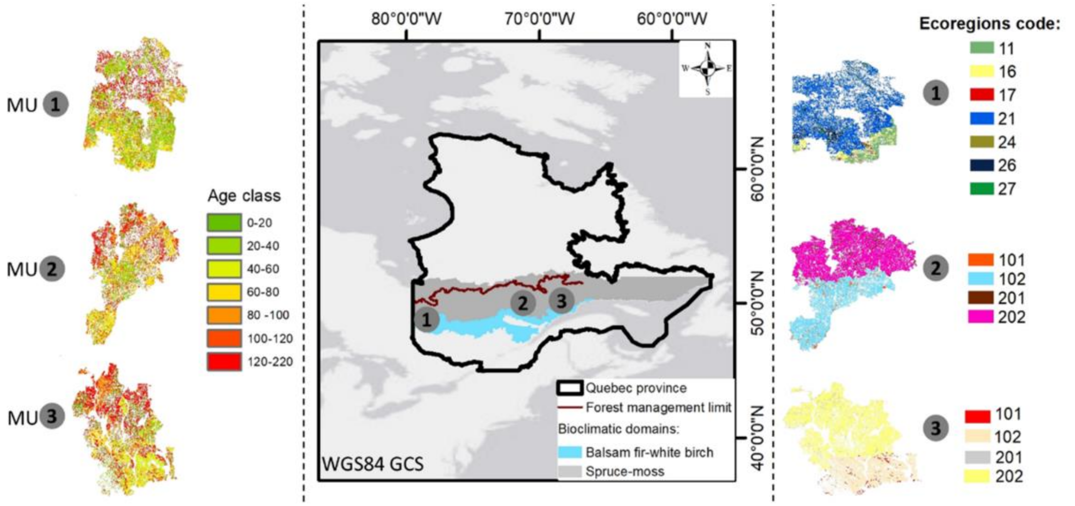

Figure 2.1 Study area (three management units: MU1, MU2, and MU3) with their ecoregions (see Table 2.2). The age-class structure of each MU in 2010 (the starting year 0).

2.2.2 Forest simulation model

We conducted landscape-scale simulations at a spatial resolution of 200x200 m^2 (4 ha) for forest growth, succession, and disturbance across the three MU, using LANDIS-II (V7.1). The simulated area was 0.62×10^6 ha, 1.00×10^6 ha, 1.15×10^6 ha in MU1, MU2, and MU3 respectively. LANDIS-II is a process-based forest landscape disturbance and succession model, stochastic and spatially explicit, it simulates the forest development processes (e.g., succession, growth, dispersal), forest disturbances (management, wildfires, windthrow, insect outbreaks), and other forest degenerative processes (senescence and mortality) at large spatial and temporal scales (Mladenoff & He, 1999; Scheller et al., 2007). These landscape processes are dependent on neighboring locations in the surrounding landscape and on neighborhood interactions from focal cells, including adjacent neighbors (e.g., fire spread) and non-adjacent neighbors (e.g., seed dispersal). We used the default seed dispersal algorithm of Brendan Ward's where two negative exponential distribution are used to calculate the probability of a seed landing at a site (Scheller et al., 2007; Ward et al., 2005). LANDIS-II uses a probability decay function to simulate seed dispersal from surrounding cells - i.e., the probability of arriving seeds decreases with increased seeding distance, with distance drawn from two negative exponential distributions defined by a species' effective and maximum seed dispersal distances, and the total probability that seeds will be present is equal to the sum of the probabilities from each source cell (Ward et al., 2005).

Because our study investigates novel conditions of climate, we used a LANDIS-II succession extension (PnET) with very direct links between climate drivers and species establishment and growth (Aber & Federer, 1992; Gustafson & Miranda, 2019). The PnET-SUCCESSION v4.1 extension uses the physiological first principles incorporated in the PnET-II ecophysiology model (Aber & Federer, 1992; De Bruijn et al., 2014). PnET-Succession mechanistically projects growth and competition of tree species cohorts on each landscape grid cell (De Bruijn et al., 2014). The cohort photosynthesis and growth are simulated as competition for light and water among all the cohorts at each grid cell with a monthly time step (De Bruijn et al., 2014; Gustafson et al., 2015). These estimations are dependent on multiple reduction factors: water stress, suboptimal radiation, vapor pressure deficit, and departure from optimal temperature. Respiration increases with temperature using a Q_{10} relationship, where a 10°C increase in temperature results in a tenfold increase in respiration rate. Each species has optimal, minimum, and maximum temperatures for photosynthesis, causing its phenology to respond to the climate inputs each month and year. Competition for light is modeled by tracking solar radiation through canopy layers (related to cohort age) according to a standard Lambert-Beer formula (Aber & Federer, 1992). Available soil water depends on soil texture, inputs from precipitation, and losses from interception, evaporation, runoff, transpiration by cohorts, and percolation out of the rooting zone. The establishment probability (*Pest*) is a function of the light and water photosynthesis reduction factors for the species at the time of establishment. In PnET-Succession, species-specific life-history traits reflect the competitive ability for resources (*e.g*., light, water), which have implications for simulated tree species range boundary shifts. PnET-Succession and LANDIS-II have been widely used in North America including Quebec boreal forest (Boulanger et al., 2017; Molina et al., 2021; Mina et al., 2021; 2022). Complete details of the PnET-Succession extension are found in De Bruijn et al. (2014) and Gustafson& Miranda (2019). All simulations were run for 300 years (2010–2310) at 20-year step intervals and a 200×200 m resolution grid (4 ha: minimum area to define a productive stand, used as a standard in forest inventories).

2.2.3 Models' parametrization and calibration

2.2.3.1 Initial communities and ecoregions parameters

For each MU, an input map of initial communities that contains a unique integer for each combination of species and cohorts was created from the fourth national forest inventories (NFI) geodatabase (MRNF, 2010). The trees of each forest stand are grouped into species-age classes, each class being represented by a cohort. The species were grouped into 344, 377, and 339 initial communities in MU1, MU2, and MU3 respectively. Each tree species requires a set of parameters related to the cells as attributes, including longevity, sexual maturity, shade tolerance, seed dispersal distance, sprouting, and post-fire regeneration, as well as shade parameters, which affect the ability of tree species to reproduce via seeding and resprouting (Table 2.1). The life-history attributes determine the successional strategy for each species. Also, the PnET-Succession extension modifies the calculation of shade based on the leaf area index (LAI). The HalfSat (half saturation light level for photosynthesis) was calibrated to reflect shade tolerance because the LANDIS-II shade tolerance parameter in Table 2.1 has no effect in PnET-Succession. The species shade tolerance and the stand shade are used to determine whether light is sufficient for the species to seed or resprout. Also, species *Pest* is calculated at each time step as a function of abiotic conditions (water and light).

The LANDIS-II model requires an input map that divides the landscape into ecoregions, where the soil properties and climate conditions (precipitation and temperatures) are assumed to be homogenous (Table 2.2). The ecoregions were defined by Quebec's *Ministry of naturelle's resources and forest* (MNRF) and integrated with Duchesne & Ouimet (2021) dataset to delimit more refined ecoregions per management unit based on soil types (textures) which correspond with FAO categories (Gustafson & Miranda, 2019). Duchesne & Ouimet (2021) modelled and mapped particle size composition for the entire managed forests area in the province of Quebec, and these data were used to classify soil's texture for all MUs using clay, silt, and sand percentages for each cell of 4 ha based on a decision tree algorithm. The soil type is used in the PnET model to determine the water retention curve of the soil. The precipitation lost fraction which represents the proportion of precipitation that does not enter the soil (e.g., runoff) was adjusted for topographic slope per ecoregion. Also, the rooting depth was assumed to be lower in Cote-Nord (MU3) than in other units, because of sand loam soil texture.

Table 2.1 The life-history attributes for the 8 species according to Boulanger et al. (2017), the ages were updated considering site index per species from Pothier and Savard (1998) yield tables. (L: longevity (years), SM: sexual maturity, ST shade tolerance, SDD: seeds dispersal distance, VRP: vegetation reproduction probability, PFR: post-fire regeneration)

Species	**L**	**SM**	**ST**	**FT**	**SDD**		**VRP**	**VRP min age**	**VRP max age**	**PFR**
					effective	**Max**				
Abies balsamea	150	30	5	1	25	160	0	0	0	none
Betula alleghaniensis	220	40	3	1	100	400	0.1	10	180	resprout
Betula papyrifera	140	20	2	1	100	1000	0.5	10	70	resprout
Larix laricina	160	40	1	1	50	200	0	0	0	none
Picea glauca	200	30	3	2	100	300	0	0	0	none
Picea marian	220	30	4	2	80	200	0	0	0	serotiny
Pinus banksiana	140	20	1	2	30	100	0	0	0	serotiny
Populus tremuloides	130	20	1	2	500	5000	0.9	10	130	resprout

Table 2.2 The characteristics of used ecoregions during the simulations. The climatic annual averages of maximum and minimum temperature and precipitation (P) were calculated from the historic (1900-2010).

Region	MU	MRNF ecoregion	Lat	Bioclimatic domains	Climatic annual averages			Refined ecoregions	
					Tmax (°C)	Tmin C)	P (mm)	Soils Texture	Area (%)
Quebec Nord	**MU1**	5a	49	Balsam fir-white birch	6.4	-5.6	824.0	clay	12
								clay loam	06
								Loam	01
		6a	50	Spruce-moss	5.8	-6.5	807.8	Clay	72
								Sand clay loam	02
								Clay loam	06
								Loam	02
Saguenay lac-saint Jean	**MU2**	5d	50	Balsam fir-white birch	6.1	-5.7	970.1	Sand	03
								Sand loam	35
		6h	51	Spruce-moss	3.6	-7.4	923.0	Sand	04
								Sand loam	57
Côte-Nord	**MU3**	5g	49	Balsam fir-white birch	5.6	-5.3	956.8	Sand	02
								Sand loam	30
		6i	50	Spruce-moss	3.8	-8.2	969.9	Sand	04
								Sand loam	64

2.2.3.2 Climate and atmosphere composition

PnET-Succession extension requires average monthly temperature (Tmin, Tmax), precipitation (mm), photosynthetically active radiation (PAR), and atmospheric CO_2 and O_3 (optional) concentrations as inputs. The climatic data for each ecoregion were extracted from the climateNA model (Wang et al., 2016), a local downscaling model which facilitates extracting climate data for specific locations (longitude, latitude, elevation). The second-generation Canadian Earth System Model (CanESM2) data from climateNA spatial model were used (Wang et al., 2016). In order to represent ecoregion climate conditions, an average of each climate parameter was calculated from 10 locations randomly distributed inside each MRNF ecoregion (Table 2.2). ClimateNA model provides data until 2100, so the climate scenarios per month were extrapolated until 2310, using Extended Concentration Pathways (ECPs) rules, recommended by Meinshausen et al (2011) and IPCC group (Collins et al., 2013), which describe the RCPs from 2100 to 2500 (Figure A2.1, Figure A2.2). The purpose of these extrapolations was to simulate for the medium and long-term (more than species longevity) and to catch the landscape productivity and composition equilibrium. CO_2 concentration data was acquired from the RCP database (https://tntcat.iiasa.ac.at/RcpDb), which covers data representing a time frame of every 10 years until 2310. For the baseline climatic scenario, we assumed that CO_2 remained constant at 389 ppm (2010) during the simulation, while in the case of RCP scenarios, the CO_2 concentration was projected to 360.67 ppm under RCP2.6, 542.96 ppm under RCP4.5, and 1961.58 ppm under RCP8.5 in 2310 (Figure A2.2).

2.2.3.3 PnET-Succession parametrization and calibration

The model was tested and calibrated using one single cell, based on local biomass estimations and literature information (Duveneck & Thompson, 2019; Gustafson & Miranda, 2019; Paré et al., 2013). Our goal was to match the empirical curve (above-ground biomass (g m-2) as a function of age) as much as possible. PnET succession parameters were calibrated based on Gustafson's guidance (Gustafson & Miranda, 2019). For each MU, the species were calibrated for 120 years, following a single cohort initialized without regeneration, using the PnET-Succession output-sites option to produce the cohort growth output files. All species were calibrated under optimal conditions (Gustafson & Miranda, 2019), using a fixed annual weather stream (long-term monthly averages) of the historic climate (1900-2010). Also, sandy loam soil type was used for calibration since it has relatively a high-water holding capacity. For species generic parameters, the default values from the PnET-Succession user guide were used (Gustafson & Miranda, 2019).

In order to improve our calibration, 19 specific parameters were tuned within the empirical bounds of literature values (Table 2.3). Regarding each specific parameter's confidence interval (McKenzie et al., 2019), other values were assessed, thus increasing the number (n) of tested combinations of all parameters. In order to test several combinations ($n>10^4$), a python script was built (using NumPy and Pandas libraries), allowing the selection of the best one (set of the 19 parameters). The closeness of fit was quantified by the *Least-squares approximation* method as one of the most used objective function, which allows finding a single optimum parameter set that provides model results fitting the empirical data as closely as possible. The root means square error (RMSE) between predicted and simulated curves was calculated and only the output with the lowest RMSE and close to the empirical values (AGB) reported by Pothier & Savard (Pothier & Savard, 1998) was saved. To select the appropriate empirical yield curve (optimal) per species from Pothier & Savard (1998), we used the mean site index (SI) from NFI, which reflects the abiotic conditions (soil and climate) of each MU. To ensure that units were the same as outputs by PnET-Succession, we converted volumes to biomass using density values (g m^{-3}) (MRNF). In addition, as the leaf area index (LAI) determines the ability of a cohort to compete for light, LAI must approximate empirical values. We used the values reported by Chen et *al.* (1997) for boreal forest species ([1.56≤ LAI ≤4.81 m^{2} m^{-2} for the old stand).

Table 2.3 An example of the tested values of 19 parameters for black spruce in MU1. Values from literature and other proposed values were tested. For the species generic parameters, we used default values from the PnET succession user guide. To make competitive interactions more predictable, we minimized species differences in parameters par MU (such as PsnAgeRed, MaintResp, DNSC, FracFol, and FracActWd, see PnET succession user guide V4.1).

Parameter	Brief Description	Tested values
MaintResp	Loss of non-structural carbon due to maintenance respiration	0.001^{b}; 0.0015; 0.002^{a}
FolN^{c}	Foliar nitrogen content (%).	0.6; 0.65, 0.7, 0.75; 0.9^{c}; 1^{a}; 1.2; 1.25
SLWmax	Maximum specific leaf weight at the top of canopy	180, 200^{a}; 210; 220
TOfol	Fraction (%) of foliage biomass lost per year	0.21, 0.22, 0.23, 0.24, 0.25^{a}, 0.26
HalfSat	Half saturation light level for photosynthesis. Lower values reflect more shade tolerance (µmol/m^{2}/sec; similar unit to PAR).	180, 200^{a}, 220
BFolResp	Base Foliar Respiration Fraction (%)	0.1^{a}, 0.08^{b},0.09
EstMoist^{d}	Tuning parameter to control the sensitivity of establishment (Pest) to soil moisture.	Typically, equal to 1^{a}
EstRad^{d}	Tuning parameter to control the sensitivity of establishment (Pest) to light level (radiation)	0.8-0.94^{a}
PsnTOpt	Optimal temperature for photosynthesis (°C)	18, 19.5^{b}, 19^{a}
PsnTMin	Minimum temperature for photosynthesis (°C)	1.6^{a}, 2
PsnTmax	Maximum average daytime temperature for photosynthesis. Typically, not greater than 37°C.	28, 29^{a}, 30
FracFol	Fraction of the amount of active woody biomass (above and belowground) that is allocated to foliage per year	0.08^{b}, 0.09, 0.1^{a}

Table 2.3 continued

FrActWd	Shape parameter of negative exponential function that calculates the amount of woody biomass that has active xylem capable of supporting foliage	0.00004[a, b], 0.00005, 0.00006
KWdLit	Annual decomposition rate (decay constant, k) of woody litter (% yr^{-1})	0.043[a], 0.125[a],
H3	Water stress parameters. Black spruce was calibrated as drought intolerant and waterlogging tolerant species (see inputs parameters in GitHub)	100,105,110, 115, 111[a]
H4		140, 145, 147[a]
H1/H2		0/4 for most species [a]. Use -3.3/2 for species that are very waterlogging tolerant [a].
FracBelowG	Below ground biomass fraction	0.3[b], 0.33[a] 0.35, 0.4[c]

[a] Gustafson & Miranda (2019) values, EstRad was calibrated for each species from calibration output files as the maximum value observed.

[b] Duveneck and Thompson.(2019) values,

[c] Paré et al. (2013) FolN values for boreal forest, those values work well for all species.

[d] These parameters control the Pest and the total number of cohorts at the landscape scale. We set the EstRad establishment modifier equal to the highest expected reduction factor value after running the model during the calibration. However, using a MaxPest value of 0.8 (time step=20) with the baseline scenario that reflects recent history, we look to see that the total number of cohorts reaches an equilibrium that is not wildly different from that of national forest inventory (Gustafson & Miranda, 2019), take into account that we expect a relative increase of the number since the empty cells after harvest and wildfires were activated and other disturbances were not considered.

2.2.3.4 Wildfire's disturbances calibration

The Wildfires effect was assessed using the extension Base-Fire (v4.7), this extension simulates fire regimes through stochastic fire events depending on fire ignition, initiation, and spread by ecological region, using as input data: ignition probability, fire regions map, fire size (min, mean, and max), fire severity and the *k* parameter that determines the strength of the association between fire spread probability and fuel age (Scheller & Domingo, 2018). As the wildfire regime depends on climate change, the burned area (land disturbed annually) was calculated per MRNF ecoregion (5a, 5d, 5g, 6a, 6h, 6i) for each climatic scenario from the literature (Table A2.1) (Bergeron et al., 2006; Boucher et al., 2017; Boulanger et al., 2014; Gauthier et al., 2015; Molina et al., 2021; Tremblay et al., 2018). Fire simulations were not parameterized as a process emerging from dynamic changes in vegetation as well as from climate change, but fire regime data (annual area burned, fire occurrence, min, max, and mean fire size) were first compiled into "fire regions" corresponding to the Canadian Homogeneous Fire Regime zones (Boulanger et al., 2014), and they were updated to account for changing climate conditions under the different RCP scenarios (see Table A2.1) (Bergeron et al., 2006; Gauthier et al., 2015; Tremblay et al., 2018). Also, the probability of ignition for each ecoregion as well as the severity was used from the literature (Molina et al., 2021; Tremblay et al., 2018). The fire rotation period was short in MU2 compared to MU1 and MU3 (see input files for more details). In addition, the initial fire regions map parameter is the input map showing where the fire regions are located on the landscape, we assumed that the fire could occur stochastically across all activated cells.

2.2.4 Sensitivity analysis

To better understand the increased mechanistic detail provided by the model, we studied the effect of the uncertainty of the parameters on the outputs based on local and global sensitivity analysis. In order to reduce the simulation time and outputs size during the sensitivity analysis, we simulated a small landscape in each MU of 46-by-46 cells grid landscape totalling 2116 cells, each representing 4 ha of forested land. This experiment grid contains all the studied species. As the PnET model is more mechanistic, more input parameters are required, which increases model parameter uncertainty. We used a local sensitivity analysis (LSA) on 12 specific parameters (MaintResp, FolN, SLWmax, TOfol, HalfSat, BFolResp, FracFol, FrActWd, KWdLit, AmaxA, AmaxB, H3, H4, K, Slwdel, FracBelowG) and the climatic inputs using 20 replications. The strength of LSA is that it is easy to perform since just one input parameter is varied for each test. Using the initial calibrated values, we varied all the parameters including radiation and CO_2 atmospheric composition by 5%. Then, the variation percentages (*V*; %) between the biomass outputs of each iteration and the initial values (reference parameters) were calculated for each time step. However, interpretations of LSA are limited when several parameters might interact dynamically through the range of individual values (Saltelli & Annoni, 2010). Consequently, we used a global sensitivity analysis (GSA) to capture the model sensitivity throughout all of the input parameters, using the *Sobol* algorithm. The *Sobol* indices is a variance-based method very popular in recent literature (McKenzie et al., 2019). *Salib* library in python was used to build the *Sobol* model algorithm using 100 as the sampling size, resulting in 2600 replications. This algorithm is an extensive GSA; it calculates total sensitivity indices for each parameter, including influence due to interaction effects (McKenzie et al., 2019). The objective of LSA and GSA was to identify the most sensitive parameters and to reduce the tested combinations in the other MU (2 and 3) later during the calibration.

2.2.5 Model validation

The annual NPP (below and Aboveground) was validated in the three MU using MODIS imagery (Moderate Resolution Imaging Spectroradiometer). MOD17A3 is a product provided by MODIS that allows studying mainly carbon sequestration by vegetation, the product MOD17A3 allows the calculation of annual NPP (Running et al., 2000). The MOD17A3 was resampled to match LANDIS-II NPP grid outputs resolution (200x200). From the NFI geodatabase, 10^4 sites per MU with mature (80-120 years old) high-density stand were randomly selected, followed by PnET-output sites extension, and compared to MOD17 NPP in the year 0 (2010) (*t*-test). High-density stands were chosen to avoid any considerable effect of understory vegetation on annual NPP estimation from MODIS imagery.

On the other hand, the LAI during summer (growing season) was validated as well using remote sensing. The monthly LAI estimations were derived only in MU1 since all the studied MU were dominated by similar species, using 500 samples (cells). The LAI simulated values were provided by PnET-output sites extension. The empiric LAI was extracted from Sentinel-2 imagery and compared to the estimated LAI by PnET succession, using *t*-test in *R* software. The images were downloaded from Open Access Hub during the growing season of 2020 (08 August, zone path: T17UPQ), processed, rescaled and co-registered to match the spatial resolution of the used grid (200x200m), using Castro et al. (2020) method in SNAP software. The obtained values by both methods were compared to Chen et al. (1997) optical measurements.

2.2.6 Scenarios design and data analysis

In order to separately assess wildfires and climate effects on forest biomass carbon storage, we designed 8 scenarios for all the studied MU. We firstly ran the model without wildfires (only succession) with the four climate scenarios (baseline and RCP). Then, we ran it with all the climate scenarios under wildfire disturbance. We assessed the differences in biomass accumulation, forest composition, and mortality rate among the scenarios in the short (ST: 2010–2110), medium- (MT: 2110–2210), and long- (LT: 2210–2310) terms. The mortality rate reflected the number of cohorts of each species killed by the major sources of mortality succession (competitive exclusion) and wildfires. The percentage area of the different tree species and forest type was used to quantify forest composition occupancy in percentage, dividing the forest into six types (BSPF: black spruce pure forests, OcPF: Other coniferous pure forests, BPF: broadleaves pure forests, BsJP: black spruce and jack pine, BsBF: black spruce and balsam fir, BsOC: black spruce and other coniferous, OCMF: Other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, OCBMF: Other coniferous and broadleaves mixed forests; Em: open woodlands). We highlight that for each scenario, we used three replicates, which reflected the minimum required number to capture stochastic variation in the response variables (Murphy & Myors 2023; Zhuo et al., 2020). The relative effects of mortality cause (wildfires/competitive exclusion), climate, period (ST, MT, LT), and the interaction of climate with both factors on mortality were calculated using a repeated-measures analysis of variance (ANOVA-3). All the statistical analyses in our study were performed in RStudio (v3.6.3). In addition, in order to understand the model behavior and climate effect on carbon sequestration, the species' annual NPP (above and belowground) under different climate scenarios were assessed during their longevity period (starting from their establishment until their mortality).

LANDIS-II could not account for the fact that most broadleaves have several limitations for dispersal near the nordic limit of Québec managed boreal forest (*e.g.,* organic soils, nutrient deficit). Consequently, we assessed and discussed the effect of restricting the broadleaves dispersal on forest composition and biomass carbon storage. An extreme scenario was therefore added in all MU, where under wildfire disturbance, the broadleaves effective and maximum dispersal value was set to 1 m (minimum value). It's important to highlight that this scenario, where there is no regeneration/dispersion of broadleaves, is highly conservative. It assumes that the existing distribution of broadleaves cannot extend further.

2.3 Results

2.3.1 Calibration, validation, and sensitivity analysis

After calibration, modelled projections of aboveground biomass (AGB) under the current climate, showed excellent agreement with empirical data (Figure 2.2). The calibrated curves closely emulated the empirical ones for the most abundant species such as black spruce, jack pine, balsam fir, and trembling aspen (AD $\leq$ 5 t ha^{-1} and RMSE $\leq$ 10 t ha^{-1}), and moderately emulated them for the other species (e.g., white birch) in MU1 (10 t ha^{-1}< RMSE $\leq$ 15 t ha^{-1}). The highest RMSE was observed for white spruce and larch (Larix laricina) species (RMSE >15 t ha^{-1}). The correlation coefficient between measured and predicted biomass was 91%, but there was a slight bias in the model predictions for the age between 40-80 years with an absolute difference (AD) higher than 15 t ha^{-1} for larch tree and white spruce, mainly in MU2 and MU3.

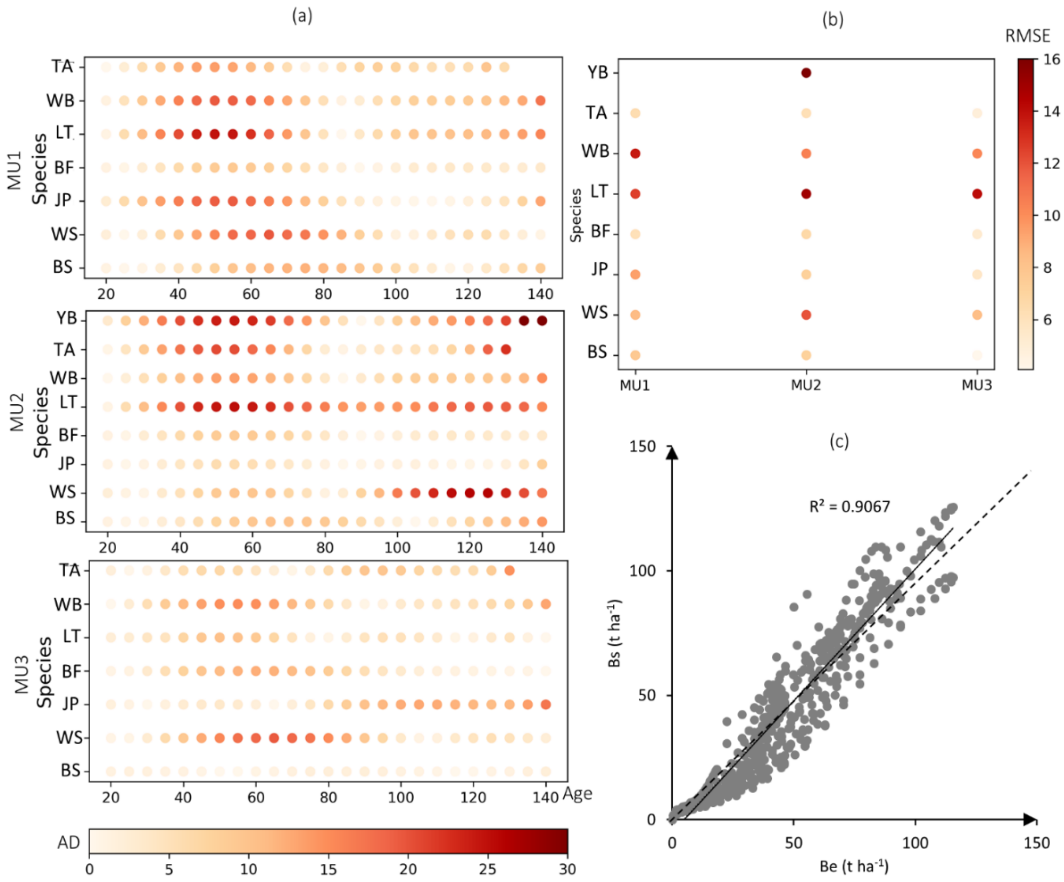

Figure 2.2 a) Calibration results of all species in the three MUs (MU1, MU2, MU3), the color-bars present the absolute difference (AD) between empirical biomass (Be) and simulated biomass (Bs) for 120 years per studied species (BS: black spruce, WS: white spruce, JP: jack pine, BF: balsam fir, LT: larch tree, WB: white birch, TA: trembling aspen, YB: yellow birch). b) The root-mean-square error (RMSE) between Be and Bs in the three MU. c) The correlation between all simulated and empirical values used for calibration, the dotted line represents the 1:1 ratio, and the continuous line represents the adjusted model.

Our validation of both LAI and NPP was informative (Figure 2.3). We found a LAI value of 2.36 and 2.78 m^2 m^{-2} from Sentinel-2 and PnET-succession respectively (Figure 2.3 (a)). The *t*-test reflects that the mean LAI of PnET-succession was significantly different from that observed by Sentinel-2 (*P*-value <0.05). However, the values founded by both methods are very close to those reported in the literature in the boreal forest. For NPP validation, the estimated NPP values from MODIS and

LANDIS-II models were very similar, but also significantly different ($P<0.05$). The highest NPP average estimated in 2010 by our model was observed in MU2 (606.67 g m^{-2} yr^{-1}) while a moderate value was observed in MU1 (575.22 g m^{-2} yr^{-1}) and lower values in MU3 (483.62 g m^{-2} yr^{-1}) (Figure 2.3 (b)). However, the RMSE between both methods (MODIS vs Landis-II) is moderately high (>100 g m^{-2} yr^{-1}).

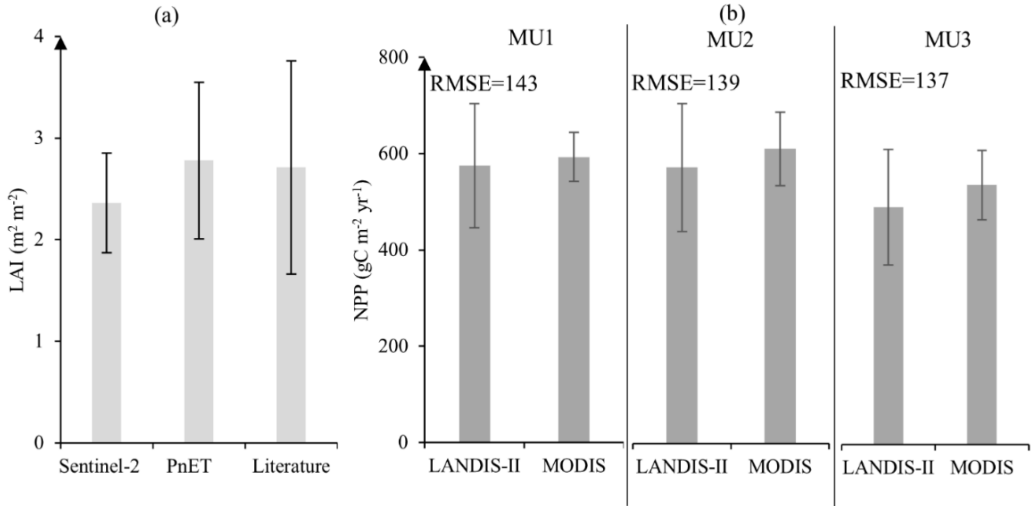

Figure 2.3 a) Comparison for MU1 between monthly LAI during the growing season per unit ground surface area m^2 m^{-2}) at landscape scale of PnET succession in 2020 with Sentinel-2 and literature values. b) Model validation through annual NPP estimated by MODIS imagery and the predicted data from LANDIS-II. The error bar presents the standard deviation at a significance level of 0.05.

Our LSA analysis indicates that the biomass module was sensitive to all PnET parameters tested (V>5%) (Figure 2.4 (a)). The LSA showed that the model was highly sensitive to four parameters (FolN, Halfsat, Fracfol, and TOfol), where the V tended to 100% when we increased the parameter by 20%. The LSA supported the direction and magnitude of the influence of each parameter on total biomass (Figure 2.4 (a)), for instance, increased FolN and temperature inputs increased landscape biomass variation. On the other hand, the precipitation did not have a substantial effect on the biomass variation in the study area, while increasing radiation (rad) and atmospheric CO_2 concentration by 50% did increase the biomass by 30% (Figure 2.4 (b)). Based on the *Sobol* results (Figure 2.4 (c)), all the parameters exhibited first-order sensitivities

(S1>0.01), except KWdlit, H3, and H4. The total landscape biomass was especially sensitive to Fracfol, FolN, followed by MainResp. Also, GSA showed that there were likely higher-order interactions occurring. The GSA showed that MaintResp was either strongly or moderately interacting with all of the tested parameters. A strong interaction was also observed between FolN, MaintResp, SLWmax, TOfol, HalfSat, BFolResp, FractWd and KWdlit. However, it seemed that the PnET model mostly required a careful calibration of FolN and MaintResp since they are interacting with other parameters.

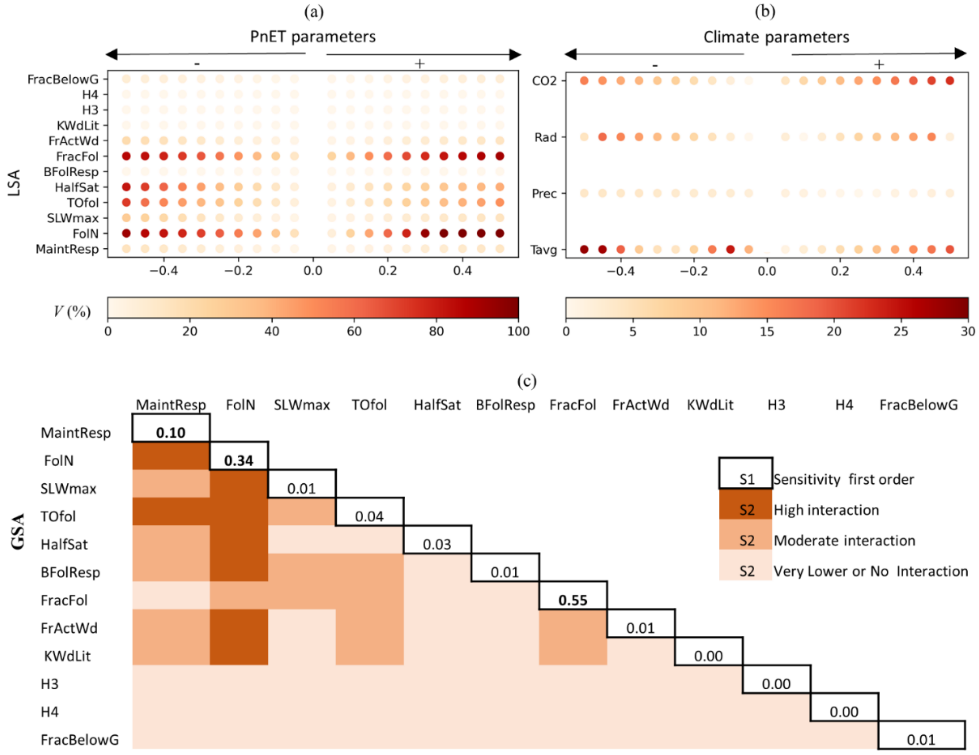

Figure 2.4 a) Local sensitivity analysis (LSA) of the 12 PnET model parameters in 2050, where *V* is the variation percentage compared to the reference values, the signs -/+ reflect the decreasing/increasing in biomass variation. b) LSA of CO_2 concentration and climate parameters (P: precipitation, Tavg: average temperature of Tmin and Tmax, Rad: radiation), c) global sensitivity analysis using *Sobol* algorithm, the diagonal values represent the sensitivity of first order, well the other values reflect the sensitivity of the second-order (S2) (interaction), (lower: S2< 0.05) (moderate: 0.05<S2 <0.10) (high: S2> 0.10).

2.3.2 Forest carbon

2.3.2.1 Species NPP

In order to isolate climate effects on species in the absence of regeneration, disturbance, and competition, our simulation from single-cell and single-cohort experiments showed that under all scenarios and MU, most species exhibited an increase in NPP under climate change by 2100. Then it decreased to baseline or below baseline conditions by 2200 in MU1. Indeed, boreal forest species NPP near the northern limit of Quebec managed forest benefited from climate change in the next 100 years mainly for intermediate scenarios (RCP2.6 and RCP4.5) (Figure 2.5, Figure A2.3). While, under RCP8.5, the NPP of coniferous was lower than that observed for the baseline climate in MU1, mainly for black spruce, white spruce and jack pine. On the other hand, coniferous forest NPP was higher than that of the baseline in MU3 under all climate scenarios during their next succession (longevity). Broadleaves forests on the other hand benefited from climate change in all scenarios, potentially doubling their NPP compared to the baseline. The length of the growing season might affect a species' NPP and biomass carbon storage. In fact, the growing season increased under climate change for all species, except black spruce in MU1 after 2100 under RCP8.5 (Figure A2.4).

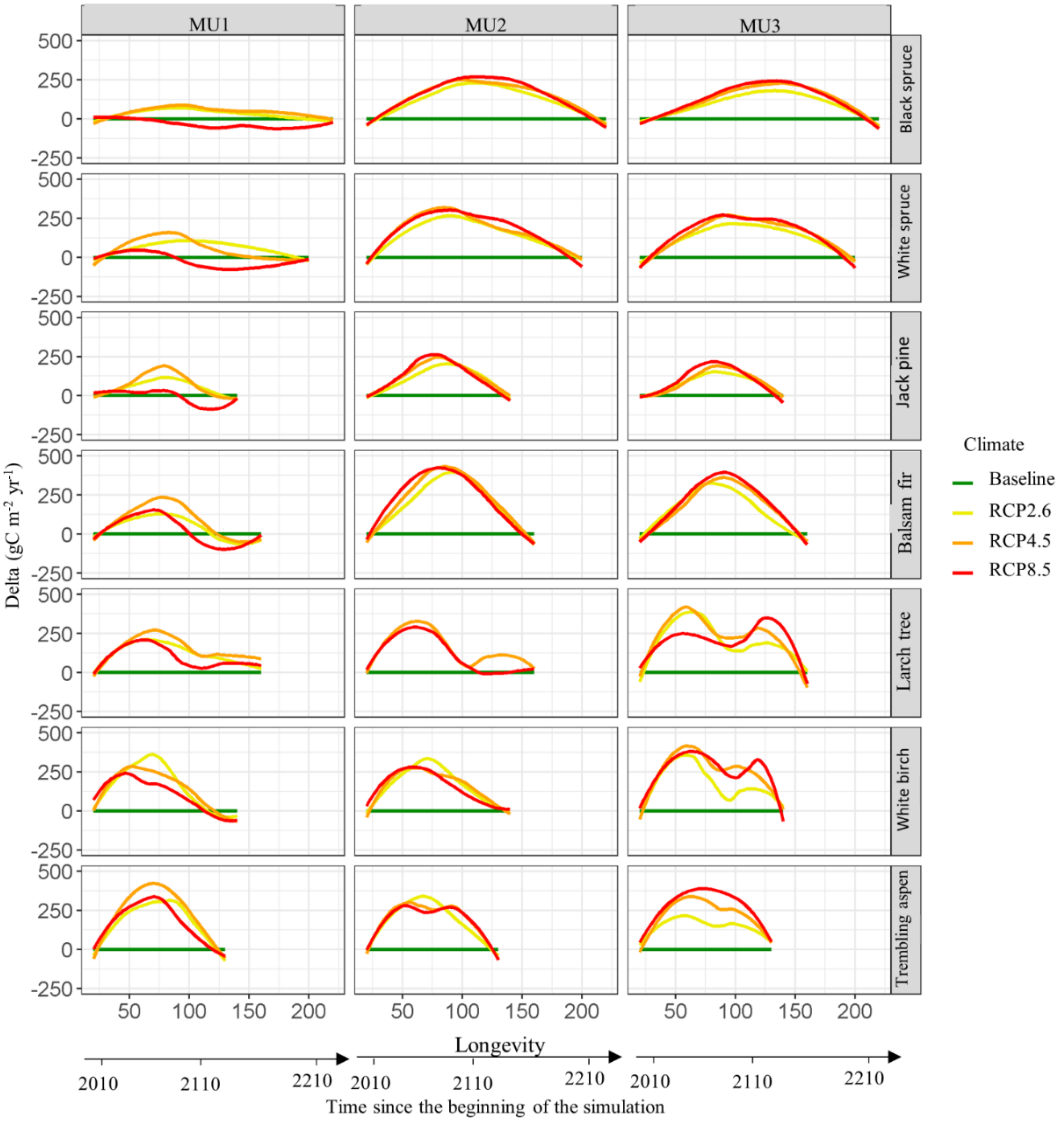

Figure 2.5 The difference in species' annual NPP (Delta; gC m-^{2} yr^{-1}) average between climate change scenarios and current climate (baseline) during their longevity age in the three management units (see Figure A2.3 for more details about the NPP per species and its confident interval). The simulations were started in 2010 with a single cohort (20 years old, monoculture) using the dominant soil texture per management unit (Clay in MU1, and sandy-loam texture in MU2 and MU3). During the simulations, the growing season was increased under climate change effect from 90-120 days to 150-180 days (see Figure A2.4).

2.3.2.2 Biomass carbon storage

Forest total living biomass (AGB and BGB) carbon storage responded to climate change and integrated disturbances (Figure 2.6). In the short-term (2010-2110), in all MU and for all RCP scenarios, the simulated carbon storage in biomass increased, by about twice compared to the baseline. Under wildfires scenarios, in MU1, the biomass carbon storage showed an increase from 37 tC ha^{-1} at the beginning of the modeling period to about 59 tC ha^{-1} for the baseline, 71 tC ha^{-1} for RCP2.6, and 83 tC ha^{-1} for RCP4.5. Similarly, in MU2, carbon storage in biomass showed an increase from 28 tC ha^{-1} to 39, 65, 73 for the baseline, RCP 2.6 and RCP4.5 respectively, but in the long-term, it declined to 44 tC ha^{-1} in MU1 and 59 tC ha^{-1} in MU2 under RCP8.5. On the other hand, high stocks were observed for RCP8.5 and RCP4.5 during the study period in MU3 for all periods with or without wildfires, in fact, the biomass carbon storage increased from 32 to around 90 tC ha^{-1} for both climate scenarios, and to 61 tC ha^{-1} for RCP2.6. In the MT, the stocks stabilized under all scenarios with high values observed for RCP4.5 in MU1 and MU2, and for RCP8.5 in MU3. In the long term, the RCP scenarios continued to be more productive than the baseline in MU2 and MU3, while for MU1 the baseline was more productive than RCP8.5.

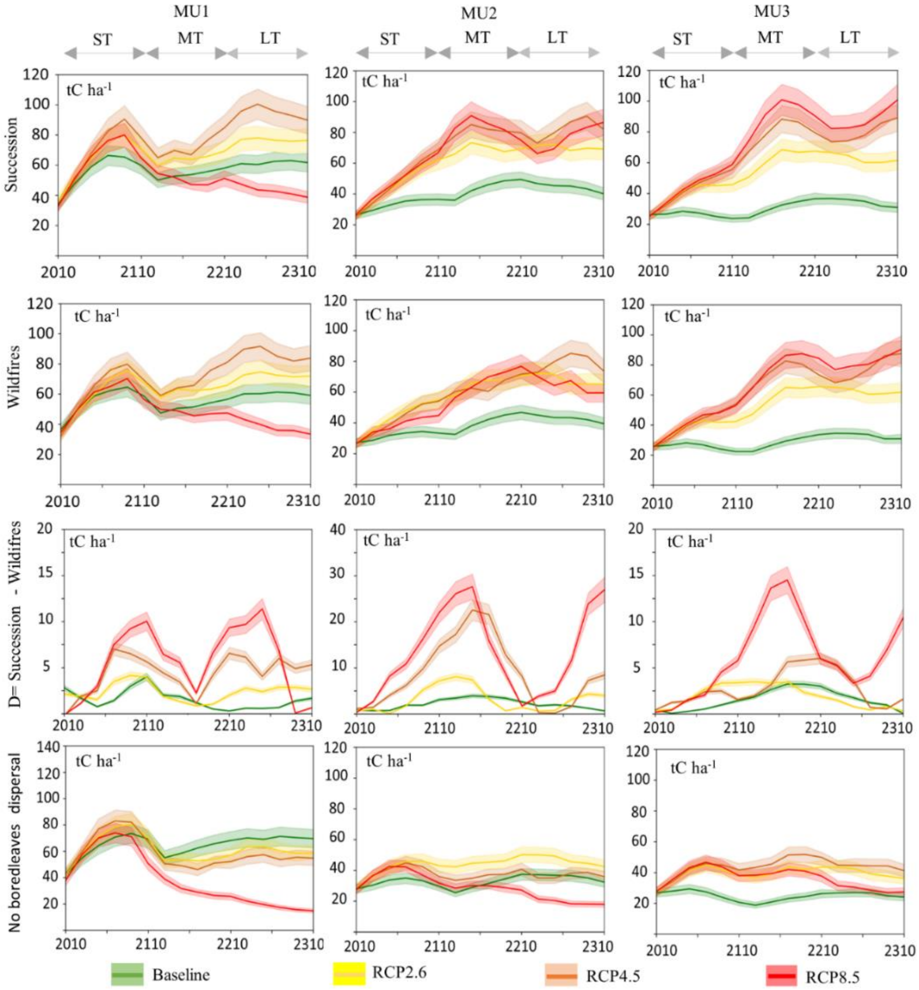

Figure 2.6 Mean simulated annual total biomass (stem, roots) (tC ha^{-1}) projection for 300 years (2010-2310) under RCP and current climate (baseline) scenarios for three periods ((ST:2010-2110); (MT:2110-2210); (LT: 2210-2310)), with and without wildfires disturbance (see Figure 2.7 for more details per species). Difference (D) in carbon (C) loss due to wildfires integrated across the studied MU over time, relative to the no-disturbance scenario (only succession). The confident intervals were estimated from the three repetitions per time step to assess the stochastic effect of the model on the outputs.

The growth of the five studied coniferous species will decline in future climates, mainly in the medium- and long-term in the western parts of the Québec boreal forest (Figure 2.7). Under global warming, white birch and trembling aspen offset the coniferous species' biomass decline (Figure 2.7). The total biomass of coniferous species decreased mainly in the long term, while that of broadleaves increased. Carbon losses related to fire had increased under climate change and increased dramatically if there is no broadleaves dispersal (Figure A2.5). On the other hand, climate change enhanced biomass carbon storage for all MUs under different RCP scenarios, except in MU1 under RCP8.5 (Figure 2.6, Figure 2.7). Furthermore, after 100 years of simulation and under no dispersal of broadleaves, total biomass under RCP 8.5 dropped to 20 and 15 tC ha^{-1} in MU1 and MU2 respectively (more than 30%) relative to the current climate. For RCP8.5 in MU2, this reduction was related to wildfires, which reduced the stocks by around -60%, while in MU1 both climate change and wildfires contributed to this reduction by -27% and -29% respectively (Figure A2.5).

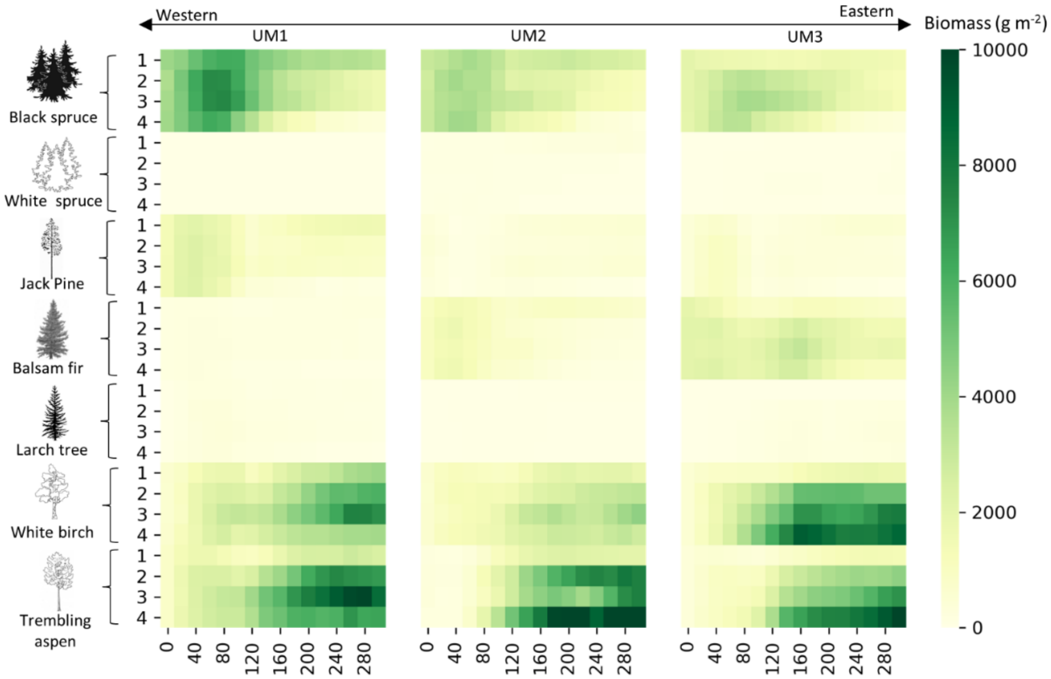

Figure 2.7 Mean simulated annual biomass (wood+roots) (tC ha^{-1}) of three replicates per species projection for 300 years (2010-2310) under four climate scenarios (1: Baseline, 2: RCP2.6, 3: RCP4.5, 4: RCP8.5).

On the other hand, compared to the baseline the carbon losses from the biomass pool under wildfires (D) were lower for all climate scenarios in MU3 and MU1 compared to MU2 (Figure 2.6). However, the increase in the burned area increased carbon losses from biomass pool under RCP scenarios compared to the baseline, with high fluctuation under RCP4.5 and RCP8.5, thus related the fluctuation in total area burnt (Figure A2.6). The losses could reach 12, 30 and 15 tC ha^{-1} yr^{-1} in MU1, MU2, and MU3 respectively. In scenarios where there was broadleaves dispersal restriction, the forest biomass carbon storage dropped to less than 20 tC ha^{-1} under RCP8.5 in MU1 and MU2 in the MT and LT. Biomass carbon storage has remained higher than that of the baseline in MU3 for all RCP scenarios. Consequently, MU3 was less vulnerable to climate change than MU1 and MU2 even without broadleaves dispersal or regeneration since coniferous species' NPP increased (Figure 2.5), with more abundance of black spruce and balsam fir.

2.3.3 Forest composition dynamics

Generally, in all MUs, there is a decrease in the proportion of pure coniferous species occupancy at the landscape scale and an increase in the mixed deciduous forest (Figure 2.8). In the ST, under different climate change scenarios, BSPF accounted for the largest proportion of the area in all MU. In the MT and LT, mainly in MU1 and MU2, the area of BsBMF, BMF, and BPF increased, mainly for RCP2.6 and RCP4.5, while the area of BSPF and OcMF decreased. Under RCP8.5 BMF and BPF dominated the landscape in MU2 and MU1. In MU3 the coniferous occupancy was less sensitive to climate change scenarios, the BsBF, OCMF, and BsBMF will continue to dominate the landscape, with 70%, 63%, 44%, for RCP2.6, RCP4.5, and RCP8.5 respectively. On the other hand, mixed forest of BsBMF remained the dominating composition in MU1 with 46%, 48%, 42%, and 32% for the baseline, RCP2.6, RCP4.5, and RCP8.5 respectively. In all MU, our results showed that OCMF (other species except black spruce) is the most stable and resilient composition under all scenarios. Also, it was very likely that the Côte-Nord region (MU3) will be a refuge site for coniferous species.

2.3.4 Broadleaves dispersal limitation effect

The dispersal constraint considerably affected the forest composition (Figure 2.9). Under broadleaves dispersal limitation scenario, in MU1 and MU2, BSPF, CPF, BsJP, and OCMF dominated the landscape during the study period (300 years), with lower productivity for RCP8.5 (Figure 2.6). In the medium- and long-terms, in both MU (1 and 2) the Em which represents the open forest woodlands was increasing for both RCP4.5 and RCP8.5 scenarios, with more than 10 and 20% in MU1 and MU2 respectively. On the other hand, in MU3 the Em was around 10% for RCP4.5 and RCP8.5, a little higher than that of the current climate. However, in MU3, BsBF and OCMF were the dominant compositions for all scenarios, with more than 50% of the total area for RCP4.5 and RCP8.5. In this unit, the coniferous species (mainly black spruce and balsam fir) did not appear to be under climate stress and remained more productive than the baseline (Figure A2.3).

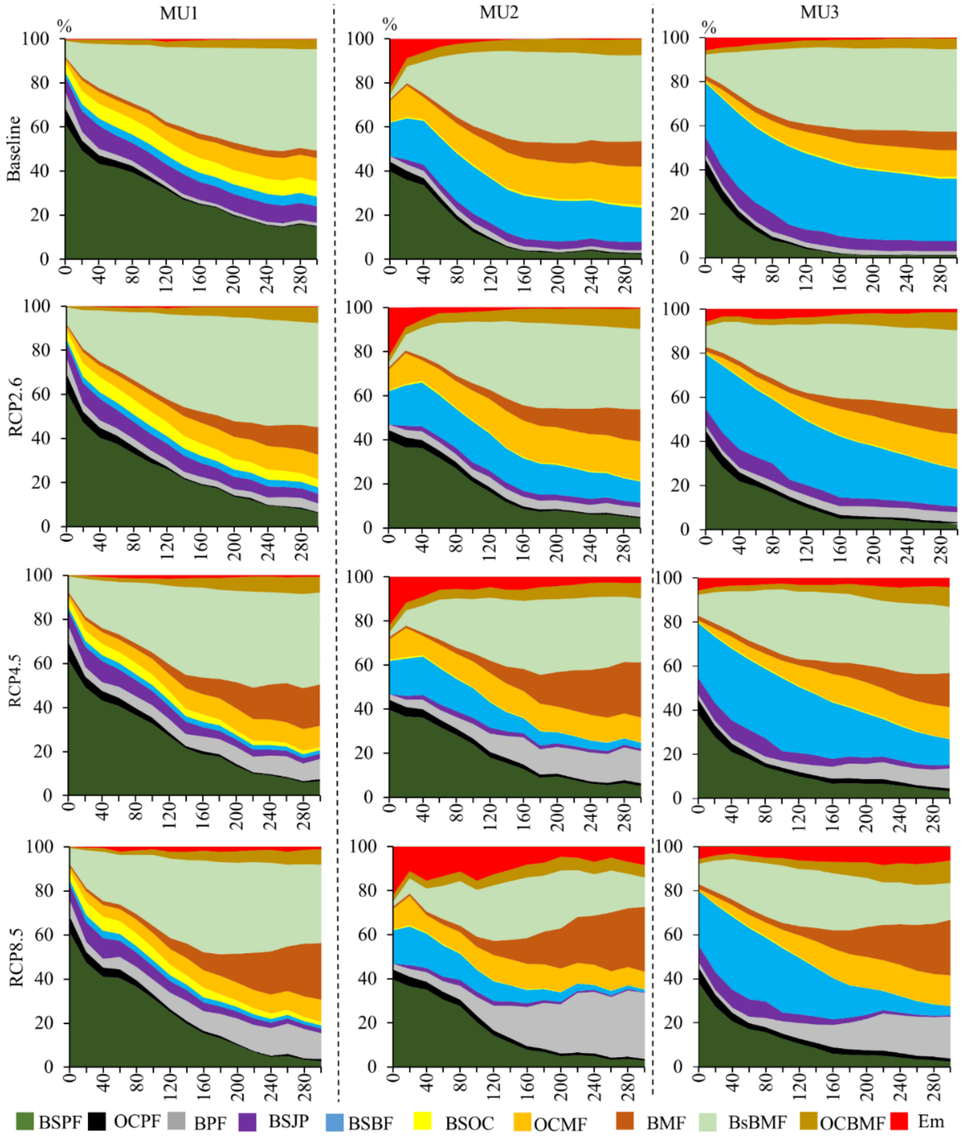

Figure 2.8 The occupancy area (%) of forest type in the three MU under climate change and wildfires scenarios between 2010 (year 0) and 2310 (year 300) (BSPF: black spruce pure forests, OcPF: Other coniferous pure forests, BPF: broadleaves pure forests, BsJP: black spruce and jack pine, BsBF: black spruce and balsam fir, BsOC: black spruce and other coniferous, CMF: Other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, OCBMF: Other coniferous and broadleaves mixed forests; Em: empty land which represents open forest woodland).

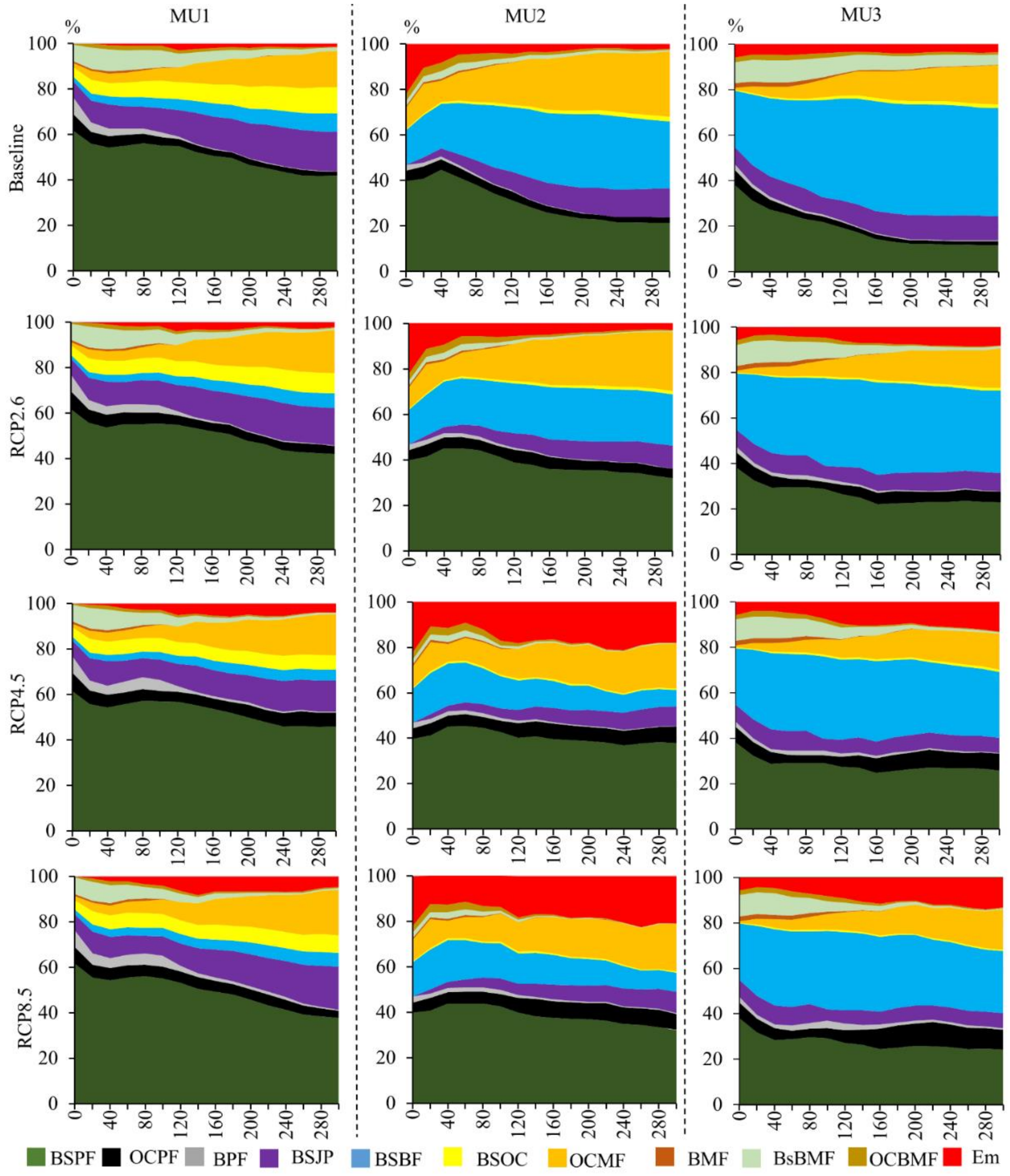

Figure 2.9 The occupancy area of each forest type (%) in the three MU under the four climate scenarios, where we assumed that broadleaves will not disperse across all the landscape because of soil limitation (legend: BSPF: black spruce pure forests, OCPF: Other coniferous pure forests, BPF: broadleaves pure forests, BSJP: black spruce and jack pine, BSBF: black spruce and balsam fir, BSOC: black spruce and other coniferous, OCMF: Other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, OCBMF: Other coniferous and broadleaves mixed forests; Em: empty land which presents open forest woodland).

2.3.5 Mortality

The total number of cohorts killed by wildfires during the study period was lower than that of competitive exclusion (senescence not included) but increased under climate change scenarios for both MU1 and MU2 (Figure 2.10). Indeed, in MU2 under RCP4.5 and RCP8.5 pathways, this total could reach a similar value of mortality as that from the competition (Figure 2.10). According to the ANOVA test, forest mortality within the different forest types was significantly affected by its cause (fire and competition), climate change, period (succession), and the interaction between climate and cause. The presence of a very significant interaction ($P<0.001$) between climate change and cause indicates that the mortality caused by competition and wildfires were depending on climate change. The variation explained by this interaction was higher for coniferous than broadleaves mainly in MU1 and MU2 (Table 2.4). In MU1, the period factor explained 41% of the variation within the total cohort killed for broadleaves, while the mortality cause factor explained 44% for coniferous. In MU2, period factor explained 32% and 19% of mortality variation for broadleaves and coniferous respectively. In MU3, this variation was explained mainly by mortality cause, 28% and 73% for broadleaves and coniferous, respectively. Also, we highlight that the variation part explained by the interaction climate×cause and climate×period for coniferous was higher than that of broadleaves. In conclusion, the ANOVA test showed that broadleaves mortality was related to period factor (succession), while the mortality of conifers was related to the cause factor (competition or wildfires), with high interaction with climate change.

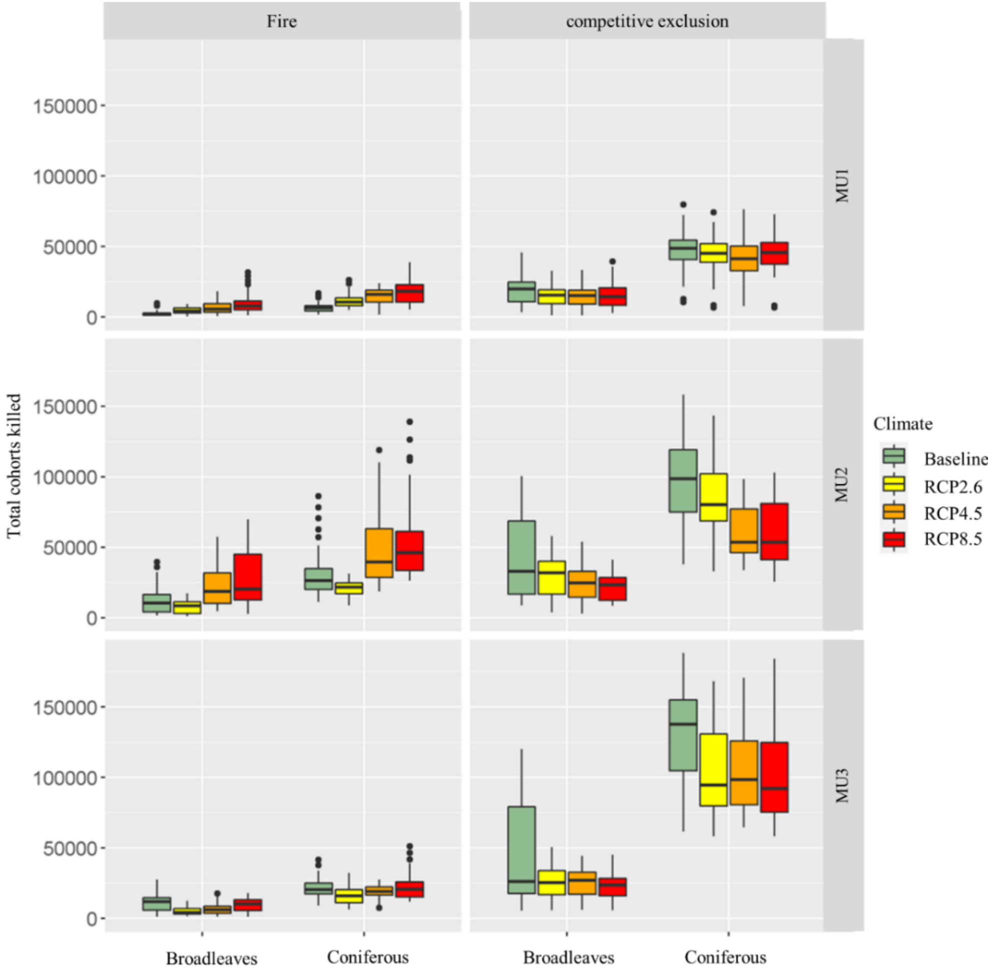

Figure 2.10 Fires and competitive exclusion mortality boxplots and averages for the entire study period (2010-2310) under climate change scenarios in the three management units.

Table 2.4 Variation explained (*V*) and significance of the effects of climate, mortality cause (competitive exclusion vs. wildfires), period (succession), and the interaction of climate × cause in the three MU from 2010 to 2310.

		Climate		**Cause**		**Period**		**Climate*cause**		**Climate*period**		**Cause*period**	
Species	**MU**	**V (%)**	***p***	**V (%)**	***p***	**V (%)**	***p***	**V (%)**	***p***	**V (%)**	***p***	**V (%)**	***p***
Broad-leaves	**MU1**	1	<0.001	19	<0.001	41	<0.001	3	<0.001	0	>0.05	6	<0.001
	MU2	2	<0.001	14	<0.001	32	<0.001	1	<0.001	2	<0.001	5	<0.001
	MU3	11	<0.001	28	<0.001	23	<0.001	6	<0.001	9	<0.001	10	<0.001
Coniferous	**MU1**	0	<0.01	44	<0.001	o.6	<0.001	4	<0.001	19	>0.05	1	<0.001
	MU2	2	<0.001	19	<0.001	11	<0.001	15	<0.001	7	<0.001	3	<0.001
	MU3	2	<0.001	73	<0.001	3	<0.001	6	<0.001	2	<0.001	3	<0.001

2.4 Discussion

2.4.1 Future Boreal Forest composition

Wildfire’s activity and changes in environmental conditions (higher temperatures, more precipitation in studies area) under climate change will affect forest species composition. Our results are consistent with Boulanger & Puigdevall (2021) findings, climate change will favour white birch and trembling aspen, which will strongly increase at the expense of coniferous species. Likewise, more broadleaved species are now observed in the regeneration layer after fire rather than conifer species, apparently due to global warming (Walker et al., 2017). Similarly, in the next 100 years, Augustin et al. (2022) report that the abundance of black spruce and jack pine will decline by around 9%, 40%, and 58% in Central, West, and East MU, respectively. Also, Molina et al. (2021) stated that an increase in fire events is accompanied by an increase in the mixed broadleaves stands proportion in the landscape for western regions. All of our results are consistent with other projections for the boreal forest of Canada (Augustin et al., 2022; Boulanger & Puigdevall, 2021; Molina et al., 2021), showing that spruce-broadleaves mixed forests will be resilient to climate change and wildfires.

The climate-induced changes in growth directly alter the competitive abilities of boreal coniferous species, making black spruce less productive when mixed with broadleaves in extreme scenarios (Brecka et al., 2020; Fisichelli et al., 2014). However, in our simulations, this co-occurrence allowed white birch and trembling aspen species to partially compensate for the decline in coniferous species biomass carbon stock. Overall, the mortality of forests dominated by shade-tolerant conifers is positively related to dry conditions (warm temperature) (Sánchez-Pinillos et al., 2022). Our model overestimated the dispersal of white birch and trembling aspen since there are several edaphic limitations for their regeneration and establishment. This study suggests that in all scenarios for 300 years, coniferous species may not disappear from the northern limit of forest management, but they could be at risk of becoming less productive than they currently are in the western regions compared to the Côte-Nord region. In contrast, broadleaves might take over the landscape where the soil conditions are favourable for their establishment. Regarding that, a high proportion of the landscape in MU1 is dominated by paludifed and organic soils, and the other management units are characterized by lower soils nutrient content, such factors preventing broadleaves regeneration (Greene et al., 1999). On the other hand, although black spruce is less productive than other boreal species, it can grow under conditions of low nutrient availability and paludified soils and can therefore outcompete other species on nutrient-poor sites (Lavoie et al., 2006).

On the other hand, similar to our results Stralberg et al. (2018) confirm that wildfires could increase the areas of open woodlands (unproductive forest area or), with lower productivity in the absence of broadleaves dispersal. Augustin et al. (2022) state that unproductive forest area will exceed 30% in all MUs under RCP4.5 and RCP8.5 mainly in central units (MU2), which could be explained by the occurrence of shorter fire return intervals, resulting in a higher rate of fire-induced regeneration failure compared to the other units (Augustin et al., 2022; Baltzer et al., 2021). Cyr et al. (2021) propose a post-fire reforestation with jack pine as the most effective strategy to maintain potential production under climate change effect, the post-fire reforestation using mixed stands of black spruce and jack pine in open forest woodland could be an opportunity for carbon offsets in Québec boreal forest, mainly in MU2 where the fire cycle is short. Diversified forests were shown to be more resilient under increased anthropogenic climate forcing (Ameray et al., 2021).

2.4.2 Climate change and forest carbon sequestration

To our knowledge, this is the first study based on an ecophysiological model to project the impact of wildfires and climate change on Québec boreal forest composition and productivity at the landscape scale, near the Québec northern limit of forest management (Figure 2.1). Our study indicates that the projected impacts of global warming and wildfires are substantial and subject to major differences between the eastern and western parts of the Québec boreal forest. In the next 100 years, it is expected that climate change will increase annual temperature averages by more than 2.5 ° C, 4°C, and 7°C for RCP2.6, RCP4.5, and RCP8.5 respectively (Wang et al., 2016). Our models projected an increase in boreal forest productivity and mixed forest area in the short term. Such an increase in productivity can be explained both by climate change-induced increases in potential growth and by the dominance of mixed forests (Ameray et al., 2021; Cavard et al., 2010; Molina et al., 2021). Warming extends the growing season from 90-120 days to 150-180 days (Figure A2.4) and increases growth rates while reducing potential cold-temperature injuries (D'Orangeville et al., 2018), which could explain the increase of NPP for both coniferous and broadleaves, mainly for intermediate scenarios RCP2.6 and RCP4.5 in all MU. Under RCP8.5 in MU1, climate change is expected to increase summer ecosystem respiration, which may increase drought and contributes to reducing NPP and biomass gains (Boulanger & Puigdevall, 2021). Besides, after 2100 the NPP of conifers (mainly Black spruce) under RCP8.5 in MU1 during both July and August started to drop to 0, because tree foliage closes its stomata due to an increase in temperature above the upper temperature limit for photosynthesis (as defined for each species). Also, as we did not consider the harvesting and more than 50% of the age structure is less than 60 years in MU1, consequently, the stocks increased even in the baseline. Our results were consistent with D'Orangeville et al. (2018), which find that 2 °C of warming may increase overall boreal forest productivity by 13 ± 3%. On the other hand, Boulanger et al. (2017) report opposite results in transition zones (boreal-temperate) in southern regions (lower latitudes), where the potential growth will decline. In the medium and long-term, we emphasize that conifer biomass was reduced under the effect of extreme climate change scenarios in the western, drier regions (Boulanger & Puigdevall, 2021).

Under intermediate climate change scenarios, coniferous species have shown increases in photosynthesis as temperatures approached the upper limit of their optimal range (27-29 C, see input files for species calibrated parameters), which could reflect the thermal acclimation of the photosynthesis process. However, only few studies (*e.g.*, Benomar et al., 2018; Dusenge et al., 2020; Way & Sage, 2008) showed moderate thermal acclimation of respiration and no evidence for thermal acclimation of photosynthesis. Boreal coniferous species, which are well adapted to cold temperature, were assumed to be limited in their capacity to adapt to warm conditions (Girardin et al., 2016). In addition to temperature effects, Gustafson et al. (2018) found that elevated CO_2 stimulates productivity such that it dwarfs the negative effect caused by elevated temperature. Also, Charney et al. (2016) used continent-wide empirical observations of tree sensitivity to climate and CO_2 to predict increases in growth across much of North America of up to 60% under the RCP8.5 emissions scenario, while in our case we found that an increase of 50% in atmospheric CO_2 could increase the growth by 30%. According to our simulations, the growth of coniferous species will decline in future climates, mainly in the medium and long-terms in both MU1 and MU2. Indeed, after 100 years of simulation and under no dispersal of broadleaves, the carbon stock in biomass under RCP 8.5 had decreased by 50% relative to the baseline climate.

Several studies showed that future warming will create climate conditions that are more prone to fire occurrence (Boulanger et al., 2017). Based on the literature, our study assumed that burned areas strongly differed between climate change scenarios and the baseline, increasing with climate change severity (Boulanger et al., 2014; Molina et al., 2021; Tremblay et al., 2018). Our findings are consistent with previous studies (Boulanger & Puigdevall, 2021; Molina et al., 2021), which report that an increase in wildfire activity could explain these declines in biomass. Indeed, in MU1 and MU2 area burned per year by 2100 under RCP85 reach 1% and 1.25 % of the landscape respectively (Bergeron et al., 2006), where the losses reach more than 15 tC yr^{-1}. Fire cycles are generally shorter in the western regions, as a result, the forest matrix tends to be dominated by a younger post-fire forest with patches of older forest dispersed throughout (Bergeron et al., 2017), which comprise much lower biomass carbon stock (Boulanger & Puigdevall, 2021; Johnson et al., 1998), thus could explain the drop of the stocks under RCP8.5 in MU1. However, our study project that the losses in forest area under wildfires disturbance will have more fluctuation for extreme scenarios (RCP4.5 and RCP8.5) with high values compared to that of the baseline (current burn rate) in MU2, which could reach 9 10^{3} ha yr^{-1} (Figure A2.6). These fluctuations could be explained by the model behaviour, in fact the fire sizes in the model follow a log-normal distribution with small fire occurring more frequently than large fires (Scheller & Domingo, 2018). For instance, in the calibration of the fire model, the mean annual area burned per year in MU2 under RCP8.5 may reach 1.26% (Table A2.1), reflecting a larger fire compared to the current rate.

2.4.3 Model limitation and future improvements

Generally, our model underestimated the biomass of young stands (40-80 years), which may be due to the numerous parameters (Table 2.3) considered by PnET and Landis-II models (Duveneck & Thompson, 2019; Gustafson & Miranda, 2019), but mainly to the data used for calibration (yield curves shape). However, Pothier & Savard (1998) yield curves should be accurate enough for our purpose since they are based on site index, thus reflecting the regional potential growth of the studied species. The bias could be reduced using monthly and annual dendrochronological data, improving the calibrated parameters. However, the PnET model allows us to adequately assess the effect of climate on species productivity and competition. It was also very useful in estimating biomass variation and composition under climate change. In MU1, our results could be overestimated, because the lowland sites are characterized by a thick organic layer, in fact, black spruce productivity declines by 50–80% with increased paludification, particularly during the first centuries after fire (Simard et al., 2007). However, the species waterlogging tolerance mainly in MU1 (clay soils) was considered during the calibration. So far, the model is not associated with soil pools to simulate SOC dynamics. Since the PnET model mechanistically simulates litter and deadwood across the landscape, a dynamic soil model could be associated with both to follow the soil organic carbon pools. Also, the integration of a dispersal model based on soil condition and nutrient content could be a good improvement in the future. In addition, the species' drought sensitivity may be underestimated by the current PnET parameters, requiring more ecophysiological research to adjust them if necessary.

There are known problems with the model we used (v.4) and a new version is due to be released shortly that will fix some known bugs. Models are constantly being improved upon, and we used the latest official release of the model. Some of the known issues with the model we used are an underestimation of drought effect on forest growth and other problems related to soil texture that causes more water to be retained in coarse soils (*e.g.,* sand compared to loam). The updated beta version could result in more mortality due to drought and competition, and less biomass for conifers, particularly in MU1 under RCP8.5. Any switch to this new version requires verification and potential calibration and adjustment of our parameters to achieve similar empirical yield curves per species”. Moreover, the V5 beta version is not yet finalized and still needs some improvements to be compatible with the LANDIS-II output extensions.

2.5 Conclusions

Our findings are consistent with previous studies, which state that climate change will considerably affect boreal forest landscapes in the future. In that context, the use of forest simulators such as PnET and LANDIS-II models makes it a useful tool for integrating ecophysiological and ecological processes and testing climate change with multiple simultaneous objectives. The use of a process-based model allowed us to mechanistically simulate the changes in forest stand productivity and composition. In this study, we draw four main conclusions which could be useful in future decision-making: i) the current productivity of boreal forests in high latitudes will increase from lower to high values in the short-term (2110), which could reach more than 80 tC ha^{-1}. ii) the dominance of black spruce decreased in favor of mixed forest and broadleaves pure forests. iii) in the absence of broadleaves dispersal because of soil limitations, the risk of forests conversion into open forest woodlands was high due to wildfires activity, which may require more reforestation of mixed coniferous to offset the lower regeneration rate in these zones, iv) coniferous mortality was strongly explained by competitive exclusion and wildfires with high interaction with climate change, while for broadleaved species it was related to forest succession. The impacts of global warming were much more important under RCP 8.5 after 2100, suggesting that western Québec forests might experience important stress under strong anthropogenic climate forcing, while eastern regions could be less venerable to climate change and a refuge site for coniferous. The results obtained in this study provide a good reference to guide sustainable forest management as well as a reasonable starting point, which can be improved and support future work on carbon budget modelling.

CHAPTER III: MODELLING THE POTENTIAL OF FOREST MANAGEMENT TO MITIGATE CLIMATE CHANGE IN EASTERN BOREAL FORESTS.

Abderrahmane Ameray, Yves Bergeron, Xavier Cavard

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CHAPITRE III : MODÉLISATION DU POTENTIEL D'AMÉNAGEMENT FORESTIER POUR ATTÉNUER LE CHANGEMENT CLIMATIQUE DANS LES FORÊTS BORÉALES ORIENTALE

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Abstract

Climate change poses a serious risk to sustainable forest management, particularly in boreal forests where natural disturbances have been projected to become more severe. In three Québec boreal forest management units, biomass carbon storage under various climate change and management scenarios was projected over 300 years (2010-2310) with a process-based dynamic landscape model (PnET-succession for Landis-II). Several strategies varying in their use of partial cuts and clear cuts, including business as usual (BAU) (clear-cut applied on more than 95% of the managed area), were tested and compared to conservation scenarios (no-harvest). Based on simulation results at the landscape scale, the clearcut-based scenarios such as BAU could result in a decrease of biomass carbon stock by 10 tC ha^{-1} yr^{-1} compared to the natural scenario. However, this reduction in carbon stock could be offset in the long term through changes in composition, as clearcut based strategies promote the expansion of trembling aspen and white birch. In contrast, the use of strategies based on partial cuts on more than 75% or 50% of the managed area was closer to or better than the natural scenario and resulted in greater coniferous cover retention. These strategies seemed to be the best to maximize and stabilize biomass carbon storage and ensure wood supply under different climate change scenarios, yet they would require further access and appropriate infrastructure. Furthermore, these strategies could maintain species compositions and age structures similar to natural scenarios, and thus may consequently help achieve forest ecosystem-based management targets. This study presents promising strategies to guide sustainable forest management in Eastern Canada in the context of climate change.

Keywords: forest carbon, climate change, PnET- Succession, LANDIS-II, biomass, harvest, wildfires, windthrow, spruce budworm

Résumé

Le changement climatique représente un risque sérieux pour la gestion durable des forêts, en particulier dans les forêts boréales où l'on prévoit que les perturbations naturelles deviendront plus sévères. Dans trois unités de gestion de la forêt boréale québécoise, le stockage du carbone de la biomasse sous divers scénarios de changement climatique et de gestion a été projeté sur 300 ans (2010-2310) à l'aide d'un modèle de paysage dynamique basé sur les processus écophysiologique (PnET-succession pour Landis-II). Plusieurs stratégies variant dans leur utilisation de coupes partielles et de coupes totale, y compris la référence (BAU) (coupes totales appliquées sur plus de 95 % de la superficie récoltée annuellement), ont été testées et comparées à des scénarios naturel (pas de récolte). D'après les résultats des simulations à l'échelle du paysage, les scénarios fondés sur la coupe total, tels que le scénario BAU, pourraient entraîner une diminution du stock de carbone de la biomasse de 10 tC ha^{-1} an^{-1} par rapport au scénario naturel. Toutefois, cette réduction du stock de carbone pourrait être compensée à long terme par des changements de composition, car les traitements des coupes totales favorisent l'expansion du peuplier faux-tremble et du bouleau blanc. En revanche, l'utilisation de stratégies basées sur des coupes partielles sur plus de 75 % ou 50 % de la zone gérée annuellement était plus proche ou meilleure que le scénario naturel et permettait une plus grande rétention de la couverture de conifères. Ces stratégies semblent être les meilleures pour maximiser et stabiliser le stockage du carbone dans la biomasse et garantir l'approvisionnement en bois dans différents scénarios de changement climatique, mais elles nécessiteraient un accès plus large et des infrastructures appropriées. En outre, ces stratégies pourraient maintenir des compositions d'espèces et des structures d'âge similaires aux scénarios naturels, et donc contribuer à atteindre les objectifs de gestion écosystémique. Cette étude présente des stratégies prometteuses pour guider la gestion durable des forêts dans l'est du Canada dans le contexte du changement climatique.

Mots clés : Carbone forestier, changement climatique, PnET- Succession, LANDIS-II, biomasse, récolte, feux de forêt, chablis, tordeuse des bourgeons de l'épinette.

3.1 Introduction

Boreal forest ecosystems provide a wide range of ecosystem services. Forests offer a multitude of benefits beyond just providing wood products, as they also play an important role in filtering water supplies, controlling floods and erosion, sustaining biodiversity and genetic resources, and providing opportunities to mitigate climate change by reducing atmospheric CO_2 (Dixon et al., 1994; Gauthier et al., 2015; Landry et al., 2021; Pan et al., 2011). The boreal forest sequesters 500 TgC yr^{-1} as a net carbon sink and stores 25% and 60% of the world's carbon biomass and soils, respectively. Furthermore, Smyth et al. (2014) point out that forest management and harvest of wood products may increase the mitigation potential of Canada's forest sector by 68, 320 TgC in 2030 and 2050, respectively. Forest management could help to improve the forest carbon sequestration capacity in both landscapes and harvested wood products (Ameray et al., 2021).

Climate change is increasing both the frequency and severity of natural disturbances in Eastern boreal forests, which has an impact on all carbon pools and several processes such as decomposition, forest regeneration, and succession (Bergeron et al., 2001, 2004, 2006). Wildfires, windthrows, and spruce budworm (SBW) outbreaks are the primary natural disturbances in Québec, which play an important role in shaping forest age structure and composition (Bergeron et al., 2006; Bouchard et al., 2009a; Navarro et al., 2018). In this part of the boreal biome, those disturbances could be classified according to the disturbed area, severity, and carbon impacts in the following order. First are wildfires, which are the most extensive disturbance, with the current annual burn rate varying between 0.04% and 0.26% in Québec. It is expected that this rate will increase under climate change, as indicated by studies (Bergeron et al., 2006; Boulanger et al., 2014). Recent studies (e.g., Ameray et al., 2023a; Augustin et al., 2022; Boulanger et al., 2014) predict an increasing burned area trend in Québec forests over the next 100 years, with inter-annual variability, mainly under RCP4.5 and RCP8.5 climate change pathways, thus increasing carbon losses from the biomass carbon pool (Ameray et al., 2023a). Second is SBW (spruce budworm) outbreaks, which are characterized by a rotation period of 32 years in the eastern regions of Québec (Boulanger et al., 2012). The last outbreaks started in 1992 and decreased the average aboveground biomass and belowground biomass by 5.96% and 6.94%, respectively (Liu et al., 2019). Dymond et al. (2010) state that SBW significantly reduced ecosystem carbon stock enough to change the landscape from a sink (4.6 ± 2.7 gC m^{-2} y^{-1} in 2018) to a source (−16.8 ± 3.0 gC m^{-2} y^{-1} in 2018). Lastly, windthrows have a potential impact on carbon cycle by increasing tree mortality (Mayer et al., 2017). The amount of damage caused by wind varies by tree size classes; while large trees may be relatively well anchored and fail via crown or stem break, smaller trees are frequently broken when large trees fall on them (Mitchell, 2012). In addition, their effects are related to the species root architecture and soil properties (Mitchell, 2012). However, the effects of wind impacts are less than those of fires and SBW, as they affected, on average, 0.0255% of the area per year between 1971 and 2000 (Bouchard et al., 2009a). All of the three cited disturbances cause carbon loss, increases in tree mortality, and a greater amount of carbon being transferred from the live biomass pool to deadwood (Don et al., 2012; Kurz et al., 2009; Mayer et al., 2017).

The net primary productivity (NPP) and biomass carbon storage are directly influenced by local abiotic factors such as temperature, precipitation, atmospheric CO_2 concentration, and solar radiation (Ameray et al., 2023a; Gustafson et al., 2018; Schuur et al., 2008). Since the late 19th century, the climate has been warming due to an increase in radiatively active gases in the atmosphere resulting from human activities. This warming trend is expected to have particularly pronounced impacts on higher latitudes, with anticipated increases in both precipitation and temperatures (IPCC, 2014). An increase in temperatures may stimulate vegetation growth up to a certain limit, beyond which it might reduce productivity under drought conditions or even cause increased mortality rates (Peng et al., 2011). It is only in the last few years that it has been possible to assess the effect of climate change on the forest carbon cycle, using simulation models which couple plant-soil carbon with the nitrogen and water cycles, integrate atmosphere-vegetation interactions, and represent competitive processes (De Bruijn et al., 2014; Gustafson et al., 2018; Gustafson et al., 2015; Zhuo et al., 2020). Several models report that boreal forests are expected to accumulate more carbon in living biomass under global warming effects, mainly under intermediate scenarios (RCP2.6 and RCP4.5) (Taylor et al., 2008; Tian et al., 2015). However, climate (i.e., precipitation and temperatures) has many potential effects on forest growth, mortality, disturbances, and establishment (Taylor et al., 2008).

In addition to natural disturbances and local abiotic conditions, harvest has a high effect on the forest carbon dynamics (Goulden et al., 2011; Liu et al., 2011). This effect is related to applied canopy removal intensity (CRI). Indeed, 100 % CRI using clear cutting (CC) at the stand scale achieves negative net ecosystem production 20 years following harvest because of high respiration and a lower NPP of the replacing stand (Ameray et al., 2021; Goulden et al., 2011; Liu et al., 2011). On the other hand, lower or moderate CRI using partial cuts (PC) and selective cutting maintains the uneven-aged forest system and thus may maximize forest carbon sequestration and storage compared to clear cuts in the long term (Lee et al., 2002; Noormets et al., 2015). Several studies have used a variety of methodologies (empirical data (Pamerleau-Couture et al., 2015), or simulation (Taylor et al., 2008)) to examine the CRI on forest carbon from empirical data at the stand level and the majority of their findings reflect that PC with lower and/or moderate CRI could increase stand net growth and stabilize biomass carbon storage as well as improve the soil carbon sink (Ameray et al., 2021). In Quebec boreal forest, CC treatments including CC and careful logging (known as CPRS or CLAAG which cuts around 95% of the canopy and aims to conserve advanced regeneration), and PC treatments (such as shelterwood cutting systems, precommercial thinning, and commercial thinning with the protection of small merchantable stems) are applied (MRNF, 2010). Currently, in Québec, more than 95% of the annual harvest area is managed using CC and CPRS (MRNF, 2010). However, their impacts at the landscape scale have not been well evaluated.

Forest planning and management decisions are always made at the regional (landscape) scale (Ameray et al., 2021). Consequently, we must assess the impact of various harvesting intensities at the landscape scale, where different CRI such as PC and CC or CPRS could be used. The annual harvest area percentage per treatment that ensures maximum carbon sequestration and storage, wood supply for industries, and conservation of current habitat (composition and age structure) is unknown. As a hypothesis, we expect that the inclusion of more PC treatments at the landscape scale will improve biomass carbon storage since lower/moderate CRI could increase carbon sequestration by 36%-40% in the boreal forest region (Peng et al., 2002). In this paper, our main objectives were: i) Firstly, to project the effect of forest management and climate change on forest composition and age structure under climate change scenarios for 300 years (2010-2310). ii) Secondly, to predict the accumulative impacts of SBW, wildfires, and windthrow on the biomass carbon pool along with their interactions with climate change. iii) Thirdly, to investigate the potential impact of forest management on carbon biomass storage, as well as annual harvested biomass. To achieve these objectives, a portfolio of strategies including variable proportions of low- and high CRI treatments (PC and CC), including the business-as-usual (BAU), were tested and compared to natural dynamics (no-harvest) under different climate change scenarios. Using a mechanistic model (PnET succession for LANDIS-II), this study introduces a new methodology to evaluate the combined effects of forest management and climate change (through alterations of growth and mortality as well as of natural disturbances regimes) on carbon dynamics at the regional scale in the high latitudes of the eastern boreal forest of Canada.

3.2 Materials and methods

3.2.1 Study area

The study was conducted in three management units (MUs) near Quebec's northern forest management limit, i.e., the limit of the territories where it was estimated that logging could be carried out profitably, and beyond this limit (~latitudes>51°), where forests are not managed because of their lower productivity (Jobidon & Bergeron, 2015). These three MUs are distributed along a longitudinal gradient from west to east: North-du-Québec (MU1; UM08551), Saguenay-Lac-Saint-Jean (MU2; UM2471), and Côte-Nord regions (MU3; UM09351) (Figure 3.1). These regions experience a boreal climate characterized by long, cold winters with short, mild summers and moderate, seasonally distributed precipitation (Wang et al., 2016). Under current climate conditions, rainfall increases from east to west from about 800 mm to about 1000 mm, reflecting a longitudinal gradient between MU, while within each MU, a latitudinal temperature gradient that increased by around 2°C from south to north was observed between ecoregions (Wang et al., 2016). These MUs belong to the spruce-feathermoss and balsam fir-white birch bioclimatic domains of the Boreal Shield, dominated by black spruce (*Picea mariana*), jack pine (*Pinus banksiana*), balsam fir (*Abies balsamea*), white birch (*Betula papyrifera*), trembling aspen (*Populus tremuloides*) and other species (MRNF, 2010). The soils are characterized by a high clay content in MU1 and a high sand-silt percentage in MU2 and MU3.

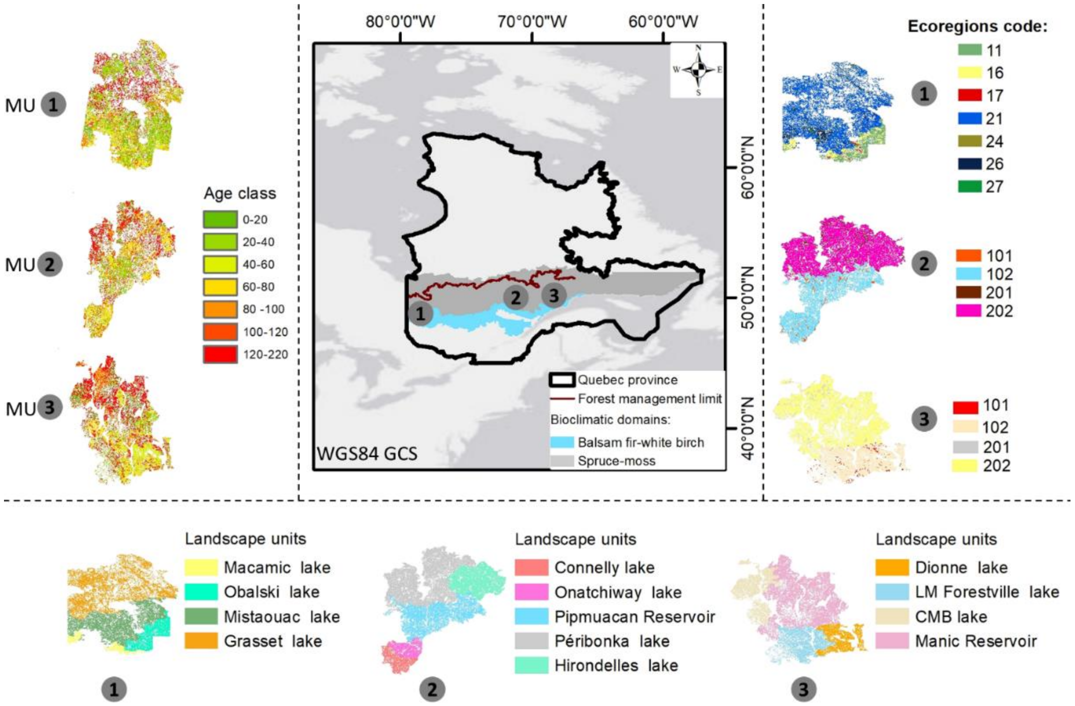

Figure 3.1 Study area (three MUs: 1, 2, and 3), with the forest age class structure at the beginning of the simulation (year 2010), as well as the landscape units inside each MU used as management area. Each ecoregion per MU was characterized by a code map during the simulation, the soil texture of each one of them is in Table 3.1.

3.2.2 Climate data

The monthly climate data (temperatures, precipitation, radiation) for each ecoregion were extracted from the climateNA model and the second-generation Canadian Earth System Model (CanESM2) data was used (Wang et al., 2016). Our analysis includes four climate scenarios: baseline (historic), RCP2.6, RCP4.5, and RCP8.5. The baseline climate scenario from 2010 to 2310 was constant around the monthly means of the historic climate (1900-2010). For RCP scenarios, climateNA model provides data until 2100, so they were extrapolated at monthly intervals until 2310, using Extended

Concentration Pathways (ECPs) rules from Meinshausen et al. (2016) and Collins et al. (2013), which describe the RCPs from 2100 to 2500 (for more details see Ameray et al., (2023a)). CO_2 concentration for RCP scenarios was acquired from the RCP database (https://tntcat.iiasa.ac.at/RcpDb), which covers data representing 10-year time frames until 2310. We assumed that CO_2 remained constant at 389 ppm (2010) during the simulation for the baseline climatic scenario.

3.2.3 Simulation Models

3.2.3.1 Forest growth and succession model

LANDIS-II is a process-based, stochastic and spatially explicit forest landscape model that integrates disturbance and succession models (Figure 3.2 (b)). The landscape in Landis-II is defined as a grid of cells, where each one of them is assumed to be homogeneous in terms of stand, belongs to a homogenous ecoregion (soil and climate), and can contain one or multiple species cohorts that can be killed by age-related mortality, disturbances, and competition. In our study, we used PnET-Succession extension (V4.1), which embeds elements of the PnET ecophysiology model, and allows us to assess species establishment, growth, mortality, and competition for available light and water and links those processes to climate drivers (Aber & Federer, 1992; De Bruijn et al., 2014; E. Gustafson & Miranda, 2019). The PnET extension is convenient for forest landscape modeling in a changing environment integrating precipitation, temperature, radiation, and atmospheric CO_2 concentration (Figure 3.2(a)) (De Bruijn et al., 2014). Furthermore, this model scales leaf-level processes such as respiration, transpiration, and photosynthesis with a monthly time step to the grid cell by incorporating light extinction and water consumption in stacked canopy layers and computing a dynamic soil water balance (Figure 3.2 (a)) (Gustafson et al., 2023). PnET extension parameters such as foliar N concentration, maintenance respiration, half saturation light level for photosynthesis, and maximum and minimum temperatures for photosynthesis were calibrated and validated for the same MUs in previous works (Ameray et al., 2023a), using Pothier and Savard empirical yield curves (Pothier & Savard, 1998). All simulations were run for 300 years (2010–2310) at 20-year step intervals and a 200×200 m resolution grid (4 ha). The simulated area was 0.62, 1.16 and 1.15 Mha in MU1, MU2, and MU3, respectively.

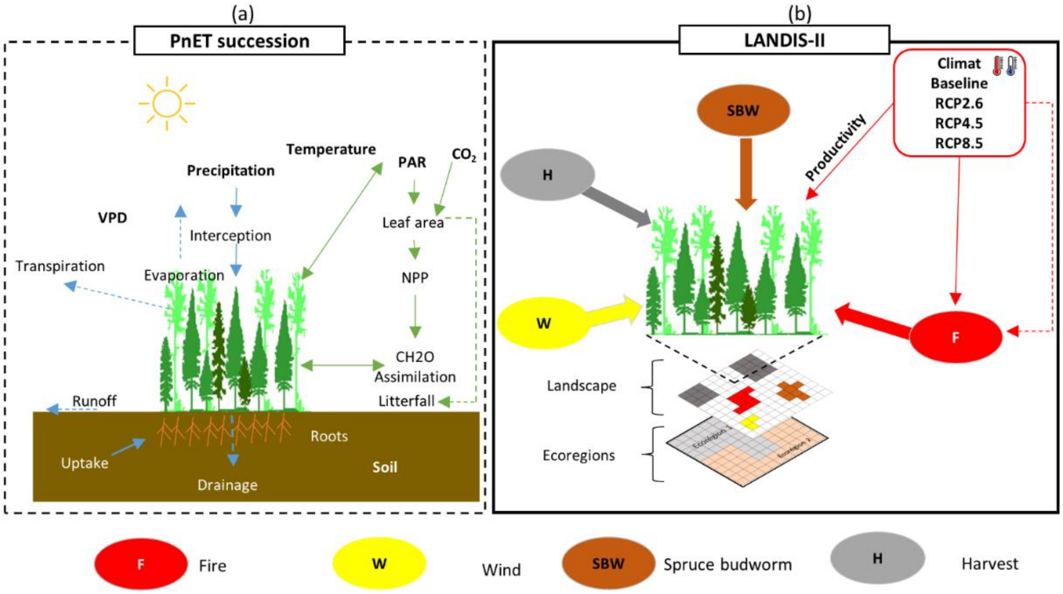

Figure 3.2 General methodology framework. a) PnET model used for succession simulates simultaneously water and carbon cycles and integrates environmental factors such as soil texture, precipitation, temperature, radiation (PAR), and vapour-pressure deficit (VPD) to estimate the net primary productivity. b) An overview of all the disturbance extensions from Landis-II considered in this study and their interaction at landscape scale per ecoregions.

3.2.3.2 Initial landscape: species and ecoregions

Regarding the LANDIS-II inputs, life history traits of studied dominant tree species were retrieved from previous studies (Boulanger et al., 2017), and included longevity, sexual maturity, shade tolerance, fire tolerance, seed dispersal distance, sprouting, and post-fire regeneration (Table A3.1). For each MU, the initial forest composition and spatial distribution were derived from "Ministère des Ressources naturelles et des Forêts" (MRNF) geodatabase (MRNF, 2010) and rasterized to a fine spatial resolution (200 m), where each cell included the tree species and age-class information. At the beginning of the simulation (2010), the initial dominant age class was 20-40 years in MU1 and 120-200 years in both MU2 and MU3 (Figure 3.1). The ecoregions were used from Ameray et al. (2023a), where MRNF ecoregions were intersected with Duchesne and Ouimet's (2021) soil map and MRNF's bioclimate shapefile to have more refined ecoregions (Table 3.1). Each ecoregion was associated with a code map in raster format (Figure 3.1). Water bodies, wetlands, islands, and other non-commercial species were excluded from our analysis (inactive cells).

Table 3.1 Simulated management units (MU) and their ecoregions.

Region	MU	Bioclimatic domains	ecoregion code	Soils texture	Area (%)
Quebec Nord-West	MU1	Balsam fir-white birch	11	Clay	12
			16	Clay loam	06
			17	Loam	01
		spruce-moss	21	Clay	72
			24	Sand clay loam	02
			26	Clay loam	06
			27	Loam	02
Saguenay lac-saint Jean	MU2	Balsam fir-white birch	101	Sand	03
			102	Sandy loam	35
		spruce-moss	201	Sand	04
			202	Sandy loam	57
Côte-Nord	MU3	Balsam fir-white birch	101	Sand	02
			102	Sandy loam	30
		spruce-moss	201	Sand	04
			202	Sandy loam	64

3.2.3.3 Natural disturbances

In order to model forest biomass carbon storage response to natural disturbances and different management scenarios under climate change (Figure 3.2 (b)), several extensions from the Landis-II library were used, including “Base Fire v4.0” (Scheller & Domingo, 2018), “Base Wind v3.1” (Scheller et al., 2018), “Biomass Harvest v4.4” (Scheller et al., 2019) and “Biological Disturbance Agent (BDA) extension v4.0.1”

(Sturtevant et al., 2019). Wildfires were included using the Base-Fire extension. This extension simulates fire regimes through stochastic fire events depending on ignition probability, fire size (min, mean and max), fire severity, and the *K* parameter that determines the strength of the association between fire spread probability and fuel age (Scheller & Domingo, 2018). Fire regime input parameters were used from previous works in the Québec boreal forest and calibrated for each climate scenario (Bergeron et al., 2006; Boulanger et al., 2014; Gauthier et al., 2015; Molina et al., 2021; Tremblay et al., 2018). In addition, the Base Wind extension stochastically simulates windthrow disturbance based on their intensity, size, spread, severity, and rotation period (Scheller et al., 2018). The wind size and rotation per ecoregion were parameterized based on historical data from the forest inventory geodatabase (MRNF, 2010). Similarly, the BDA extension stochastically introduces periodic defoliation events uniquely parameterized by defoliator species; SBW disturbance (Sturtevant et al., 2019). Host tree species for SBW included, from the most to least vulnerable, balsam fir (*Abies balsamea*), white spruce (*Picea glauca*), and black spruce (*Picea mariana*). Parameters used in this study for BDA extension were calibrated and validated using various sources for the boreal forest (Boulanger et al., 2017, 2019). We assumed that climate change did not directly affect the wind and SBW disturbance regimes, but rather altered forest composition and structure which influenced their spatial and temporal pattern.

3.2.3.4 Harvest disturbance

The biomass harvesting extension was used to model different management strategies (Scheller et al., 2019). This extension requires dividing the landscape into management areas (i.e., landscape units in Figure 3.1), specifying stands to be harvested based on species and age criteria, and defining the order in which they will be harvested. We used the proposed annual harvested volume (converted to biomass) necessary to cover the timber supply from the 2023-2028 management plan (Table A3.2) (Forestier en chef, 2022), corresponding to the current allowable annual cut (AAC), i.e., the maximum volume that can be harvested annually without reducing future forest production capacity, set by the Chief Forester of Québec (Forestier en chef, 2022). At the first-time step, all the scenarios were calibrated to cover the current AAC using the current climate (baseline). Firstly, we designed the prescriptions which reflected the silvicultural treatments used for harvest at stand scale (MRNF, 2010), including clear cut (100% CRI), CPRS (CRI was fixed at 95% and the cohorts of 1-20 years were conserved) and three forms of PC (with 25%, 50%, and 75% CRI). The stand ranking was based on species' economic value and their minimum age criteria for exploitability (Multiple Repeat was disactivated). Secondly, for the BAU scenario, we used a geodatabase of the harvest history (1970-2010) to calculate the harvested managed area per silvicultural treatment (Table A3.3). Finally, we designed and compared other strategies to BAU, varying the harvested area per treatment (Table 3.2). In the Québec boreal forest, where reforestation could follow harvest mainly after CC or CPRS, our prescriptions respected the historic planted ratio of each species (70%, 25%, 3%, and 2% of black spruce, jack pine, larch tree, and white spruce, respectively). Furthermore, in our proposed scenarios the annual level of the replanted area was variable (Table 3.2).

Table 3.2 Tested scenarios and their description. Treatments used at stand scale with different harvesting intensities and the percentage of annually managed area per treatment. CC+Aff is the percentage of the replanted area after harvest by CC or CPRS, due to a lower soil seed bank and regeneration rate.

Scenarios	**Description**	**Code**	**Used treatment at stand scale and % of managed area per treatment**					
			CC	**CC+Aff**	**CPRS**	**PC75%**	**PC50%**	**PC25%**
Scenario-0	No harvest scenario under natural disturbances	S0	0.0	0.0	0.0	0.0	0.0	0.0
Scenario-1	All the annually harvested area is managed using high CRI (CC and CPRS). The establishment is based only on regeneration.	S1	50.0	0.0	50.0	0.0	0.0	0.0
Scenario-2	BAU. Currently used scenario, where CPRS and CC are used for more than 95% of annually harvested area and 10% is managed using PCs with 25%, 50%, and 75% of CRI.	S2	From the historic, See Table A3.3					
Scenario-3	We used 75% of the annually harvested area for high CRI (CC and CPRS) and 25% for low-removal ones (PC)	S3	25.0	25.0	25.0	8.3	8.3	8.3
Scenario-4	We used 50% of the annually harvested area for high-removal treatments and 50% for low-removal ones.	S4	16.7	16.7	16.7	16.7	16.7	16.7
Scenario-5	We used 25% of annually harvested area for high-removal treatments and 75% for low-removal ones.	S5	8.3	8.3	8.3	25.0	25.0	25.0
Scenario-6	Extreme use of PCs (100% of annually harvested area), the opposite of scenario 1	S6	0.0	0.0	0.00	33.3	33.3	33.3

3.2.4 Simulation settings

Simulations were carried out for 300 years, with three replicates to estimate uncertainties in the simulation results. We began our simulation with an only-succession scenario, where there are no disturbances. After we assessed the accumulative impact of natural disturbances, this was done by running the following scenarios: 1) only succession and winds; 2) only succession, winds, and fires; and 3) only succession, winds, fires, and SBW, corresponding to natural scenario (S0). Also, we aimed to compare CC based scenarios (S1, S2, S3) and PC based scenarios (S4, S5, S6) in the selected management units with natural dynamics scenario (S0) (CRI=0 or total conservation) (Table 3.2). All the scenarios were designed to assess whether, and to what degree, different harvest practices, in combination with wind, wildfires, and SBW disturbances affect the changes in living biomass carbon pool and harvested biomass under climate change. In this study only living and harvested biomass were investigated. The total biomass harvested per management scenario was converted to merchantable proportion (Landry et al., 2021) and compared to ACC. The differences in the biomass carbon pools between the management scenarios and the no-harvest scenario ($\Delta_{i,j}$) were calculated in order to assess the biomass carbon balance of each management strategy relative to the natural baseline (equation (1)) (Krofcheck et al., 2019).

$$\Delta_{i,j} = S_{i,j} - S_{i,0} \quad (1)$$

were *i* climate scenario: baseline (*i*=1), RCP2.6 (*i*=2), RCP4.5(*i*=3), RCP8.5(*i*=4), and *j* management scenario from 1 to 6 (Table 3.2).

The biomass community output extension was used to track each cell stand composition and the following forest type descriptions: BSPF: black spruce pure forests, OCPF: other coniferous pure forests (jack pine, balsam fir, white spruce, larch tree), BPF: broadleaves pure forests, BSJP: black spruce and jack pine, BSBF: black spruce and balsam fir, BSOC: black spruce and other coniferous, OCMF: other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, and OCBMF: other coniferous and broadleaves mixed forests. To catch the effect of management on forest composition, the differences between each composition per management scenario and its correspondence in the natural scenario (no harvest) were calculated. All analyses were carried out in Python environment.

3.3 Results

3.3.1 Forest management effect on composition and age structure

According to our simulations at the landscape scale in all management units (MUs), the following compositions show the most significant alterations due to management and climate change when compared to natural dynamics scenario (S0): BSPF, BSOC, BSBF, BsBMF, OCMF, BPF, and BMF (Figure 3.3, Figure A3.1). In MU1, under the current climate, the differences between all scenarios and S0 showed that the strategies based on CC (S1, S2 (BAU), and S3) may reduce BSPF, BSOC, BSBF, and BsBMF by 10%, 5%, 2%, and 15%, and generally increased BPF, BMF, and OCMF by 10%, 5% and 13%, respectively. In MU1, BPF may rise by 20% under RCP8.5 and S1 because this scenario was based on the regeneration rather than reforestation of coniferous species. In MU2, CC-based strategies (S1, S2 and S3) reduce BSPF, BSBF, and BsBMF by more than 8%, 10% and 15%, respectively, independently of climate scenarios (Figure 3.3, Figure A3.1). In contrast, in this unit BPF may attain more than 10%, 7% and 5% under S1, S2 and S3, respectively; similarly, BMF had increased to more than 5% for all CC-based scenarios. These BPF thresholds may double in MU2 as a result of climate change (Figure 3.3, Figure A3.2), in particular for RCP8.5. The most altered composition in MU3 under the current climate is BSBF, which decreased by 18%, 17%, and 18% under S1, S2, and S3, respectively, compared to S0. These three scenarios in MU3 may decrease BSBF, BSPF, and BsBMF by 10%, 5%, and 15% under RCP8.5. Briefly, CC-based strategies increased the abundance of broadleaved species and decreased coniferous cover in all MUs, primarily black spruce, regardless of climate scenarios.

On the other hand, PC inclusion (S4, S5 and S6) diminished the gap between management and natural dynamics scenario (S0) (Figure 3.3, Figure A3.1). Generally, the differences of all compositions when compared to S0 under S6 scenarios were less than 5% in all MUs under all climate change scenarios. For instance, in MU1 under current climate, S4, S5 and S6 had decreased BSPF by 8%, 7%, 4%, and BsBMF by 15%, 9% and 1%, respectively. Similarly, under RCP8.5 in MU1, BSPF dropped by 4%, 3% and 2% under S4, S5, and S6 respectively; in MU2, for the baseline climate scenario, BSBF decreased by 8% under S4 and S5, but increased by 2% under S6. BSPF decreased by 2% under S4 and S5, but only 1% was observed under S6. Additionally, BsBMF differences were predicted to be less than 12% for the S4 and S5 scenarios and less than 5% for the S6 scenario; in MU3, BSPF composition under current climate was stable (~0% of the difference compared to S0) under PC-based strategies (S4, S5 and S6). BSBF dropped by 12% and 10% under S4 and S5, respectively, and increased by 1% for S6 under current climate.

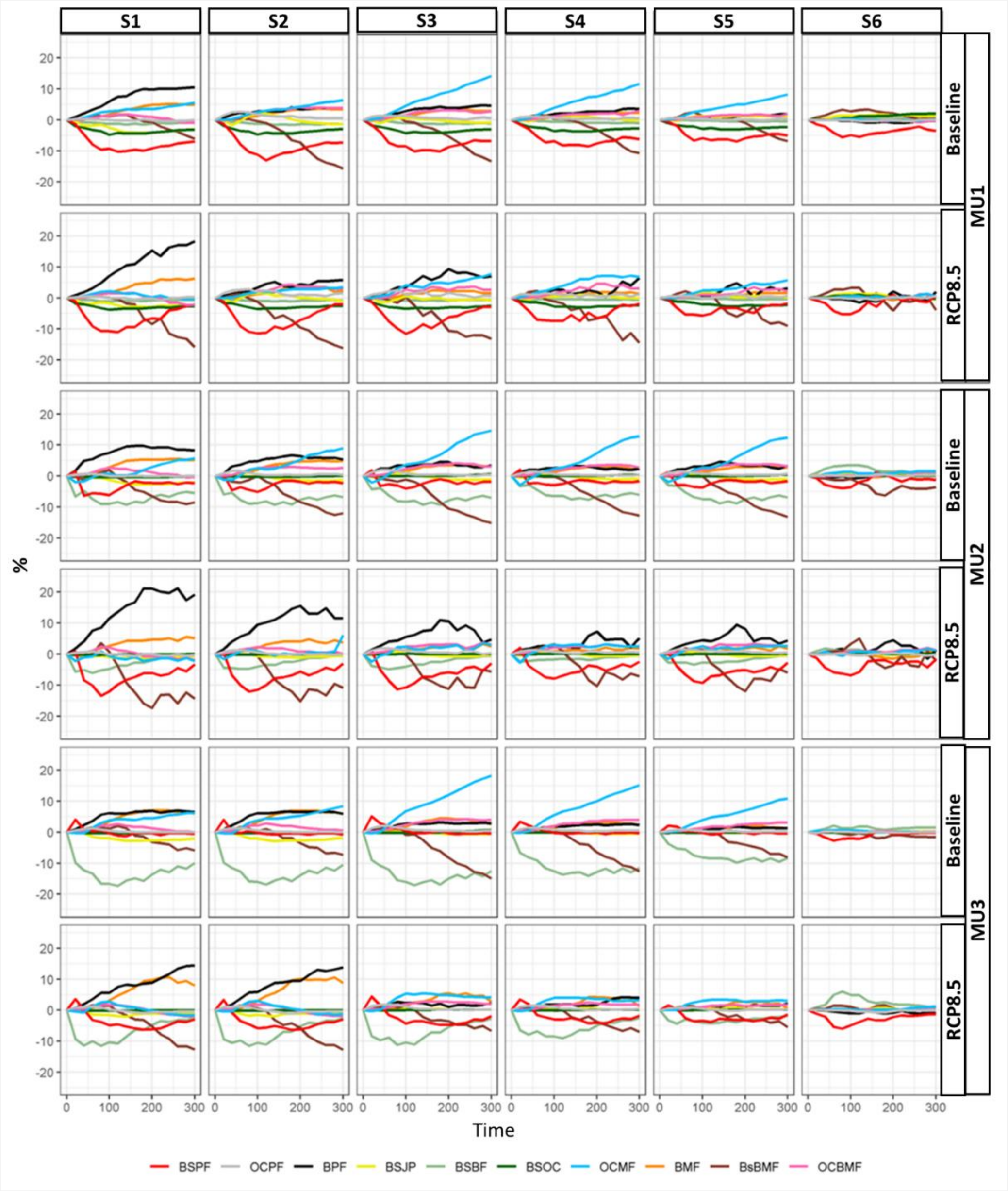

Figure 3.3 The difference (Δ; %) of composition percentage between natural scenario and management scenarios in the three MUs between 2010 (year 0) and 2310 (year 300) under current climate scenarios and RCP8.5. (BSPF: black spruce pure forests, OCPF: other coniferous pure forests, BPF: broadleaved pure forests, BsJP: black spruce and jack pine, BsBF: black spruce and balsam fir, BsOC: black spruce and other coniferous, OCMF: other coniferous mixed forests, BMF: broadleaved mixed forests; BsBMF: black spruce and broadleaved mixed forests, OCBMF: other coniferous and broadleaved mixed forests). The difference results under RCP2.6 and RCP4.5 are in Figure A3.1.

Regarding the age structure, independently of climate scenario in all MUs, the abundance of young forests in S1 and S2 (BAU) increased by more than +25% and mature and old-growth forests decreased by -10% and -16%, respectively, compared to the natural scenario (S0) (Figure 3.4, Figure A3.3). The inclusion of PC could help to reverse the decline in mature and old-growth forests in all MUs. For instance, in MU1, the differences in young forest from S0 were approximately +20%, +15%, +10%, and -2% under S3, S4, S5, and S6, respectively, whereas those differences were approximately -5%, -3%, -2%, and 0% for mature forests in the same order. Additionally, according to our simulation, old growth forest under S3, S4, S5, and S6 varied by -14%, -10%, -7%, and -4%, respectively. MU2 and MU3 showed a similar age-structure pattern under various management scenarios. In conclusion, the application of PC-based strategies (lower and moderate CRI) on more than 75% or 50% of the managed area (S4, S5 and S6) mimicked the natural scenario by emulating similar natural vegetation patterns and age structures (Figure 3.3, Figure 3.4) under all climate scenarios.

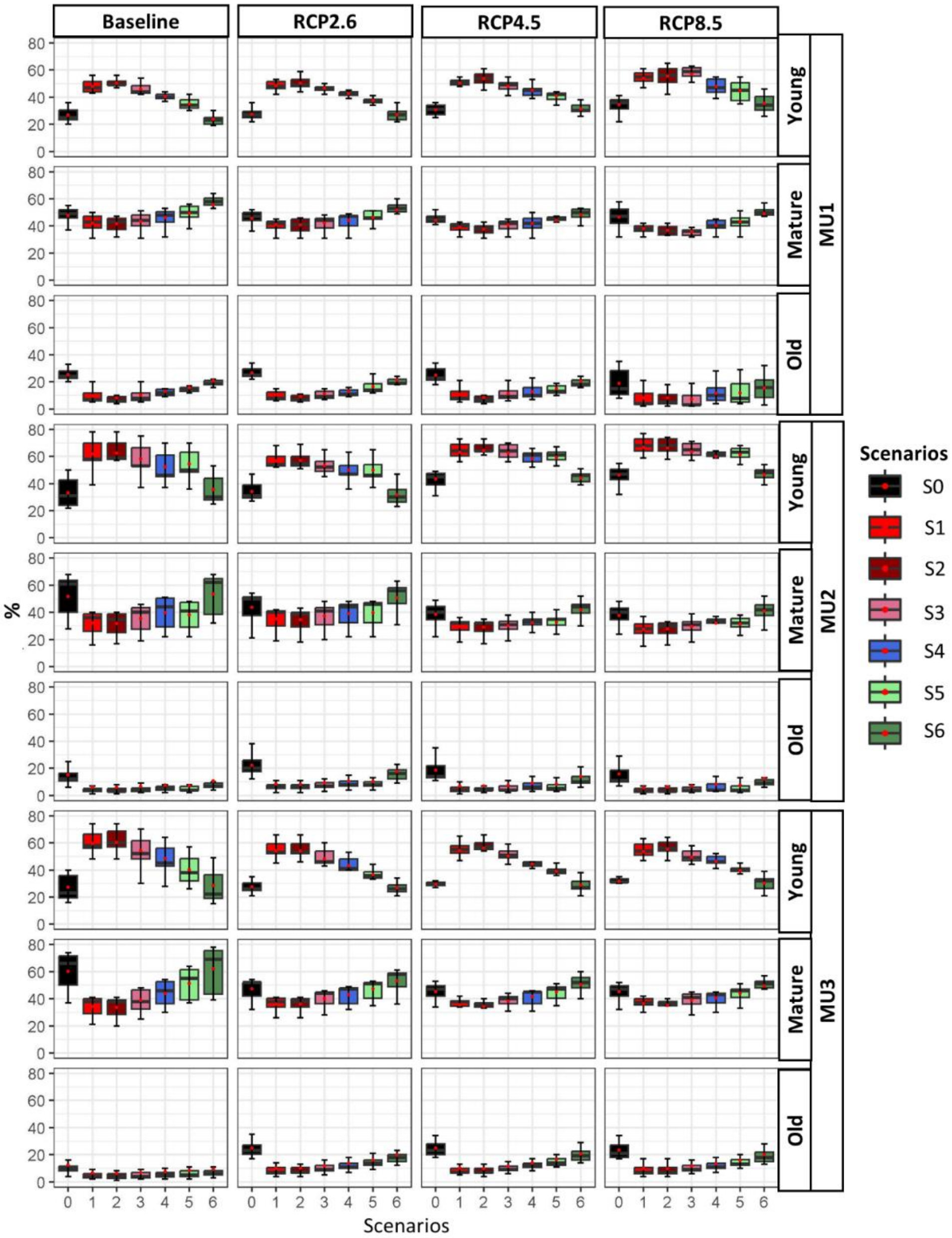

Figure 3.4 Age structure average of young forest (age≤40), mature forest (40<age≤100), and old growth forest (age>100) calculated for cells across the landscape during the entire period under climate change and all management scenarios.

3.3.2 Cumulative impact of natural disturbances

Under the only succession scenario (no disturbances) and current climate (baseline), coniferous biomass carbon stock increased in MU1 from 27 tC ha^{-1} to 37 tC ha^{-1} and remained stable in both MU2 and MU3 at around 26 tC ha^{-1} (Figure 3.5). However, under RCP8.5 their biomass decreased by 72% and 25% in MU1 and MU2, respectively, in the simulation when compared to the baseline. Meanwhile, broadleaved species biomass carbon stock increased by around 200% in all MUs under RCP2.6 and RCP4.5 and could reached 300% in MU3 under RCP8.5 at the end of simulation. Windthrow (S+W scenario) had a lower effect compared to the other disturbances (fire and SBW) (Figure 3.5, Figure A3.4). For instance, under current climate the average coniferous carbon biomass losses were -0.03, -0.12 and -0.13 tC ha^{-1} yr^{-1} during the study period in MU1, MU2 and MU3, respectively. Fires (S+W+F scenario) had a considerable impact mainly on the coniferous biomass of both MU1 and MU2, due to shorter fire cycles. The biomass carbon storage of broadleaved species was close to that of the succession-only scenario for S+W and S+W+F scenarios. When SBW was considered (S+W+F+SBW scenario), the biomass carbon stock of coniferous considerably dropped mainly in MU3 compared to other scenarios and an increase in that of broadleaved species was noticed (Figure 3.5, Figure A3.4). For example, in MU3 where SBW is more frequent, the average annual losses of coniferous biomass carbon storage due to SBW were -3.89, -5.77, -7.30, and -6.26 tC ha^{-1} yr^{-1} for the baseline RCP2.6, RCP4.5, and RCP8.5, respectively (Figure A3.4). On the other hand, the biomass carbon storage of broadleaved species was increased by 3.38, 8.27, 12.02, and 13.24 tC ha^{-1} yr^{-1} for baseline, RCP2.6, RCP4.5, and RCP8.5 respectively (Figure A3.4). For the entire study period, the average reduction rate in biomass carbon storage for coniferous species vulnerable to SBW was 5.25%, 19%, and 23% in MU1, MU2 and MU3, respectively.

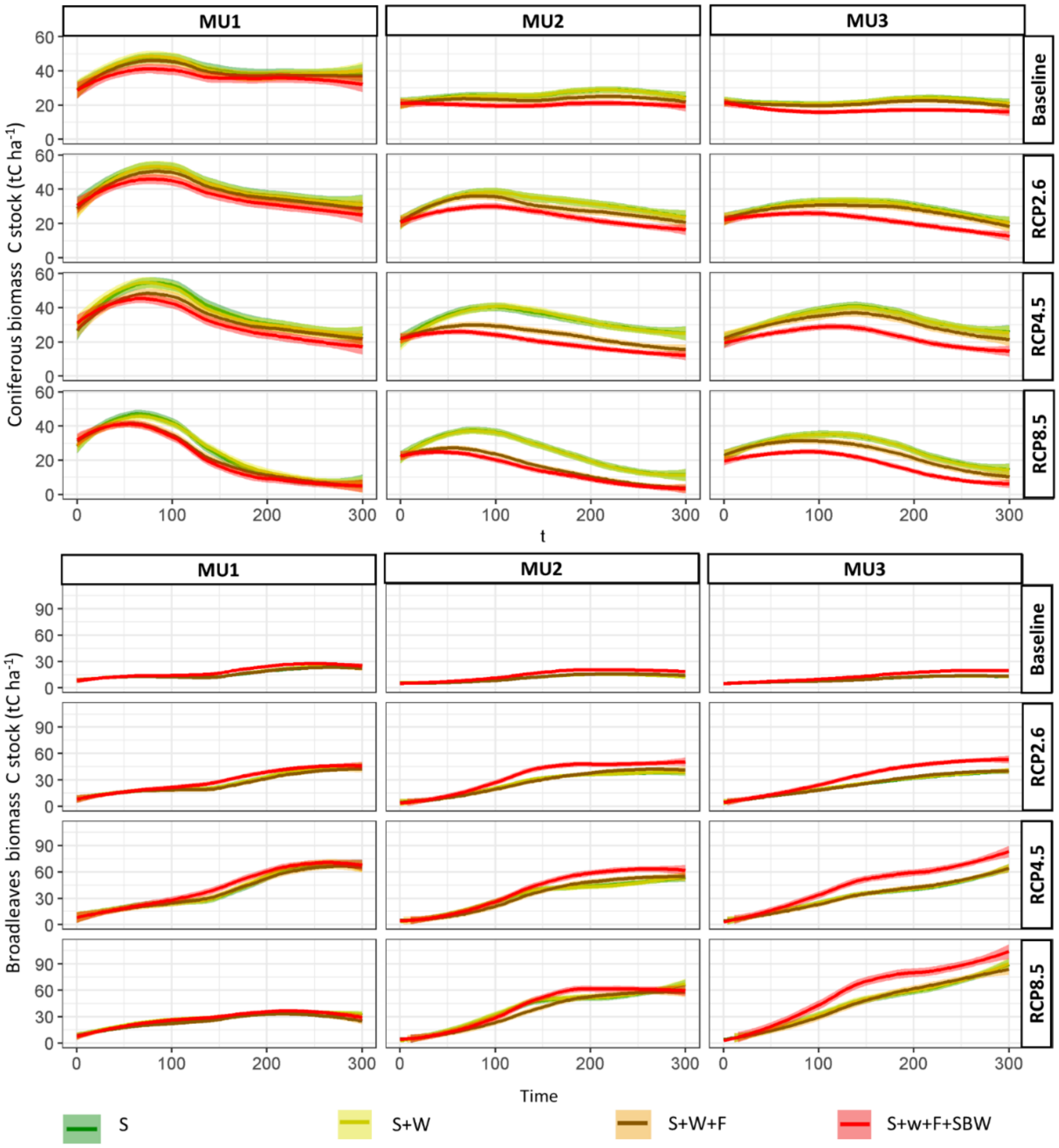

Figure 3.5 Accumulative impact of natural disturbances (legend: S= only-succession, W=winds, F= fire, SBW= spruce budworm) by 20-time step on biomass carbon storage for both coniferous and broadleaved species in the management unit under four climate scenarios (baseline and three RCP scenarios) during the study period 2010 (year 0) and 2310 (year 300).

3.3.3 Management effect on forest carbon

CC-based scenarios (S1, S2 and S3) deceased the total biomass carbon storage by around 10 tC ha^{-1} yr^{-1} over the next 100 years compared to the no harvest scenario (Figure 3.6) under current climate (baseline) in all MUs. On the other hand, PC-based scenarios (S4, S5 and S6) showed a better performance than CC-based scenarios, particularly for S5 and S6, which were close to natural scenarios (S0) with a difference less than 2 tC ha^{-1} yr^{-1} in all MUs under baseline climate scenarios. Under climate change, the losses under S1 and S2 attained 12 tC ha^{-1} yr^{-1} for MU1 and 15 tC ha^{-1} yr^{-1} in both MU2 and MU3. Nevertheless, those reductions in biomass carbon stock under strategies based on high CRI (i.e, S1, S2) could be compensated after 200 years (2210), because these scenarios accelerated the abundance of intolerant shade species (trembling aspen and white birch) compared to natural scenarios (Figure A3.2, Figure 3.3), which captured more carbon and offset previous years' losses. The use of PCs on more than 50% (S4), or 75% (S5) or 100% (S6) of the managed area was the best strategy to stabilize biomass carbon storage under current climate and may achieve high stocks under RCP scenarios compared to S0 in the next century (Figure 3.6).

Regarding the reforestation effect, S3 with a 25% replanting rate after CC had lower biomass carbon storage than S4 and S5 which were based on 16.7% and 8.3% replanting rates, respectively, under all climate scenarios (Figure 3.6). For example, under current climate, S3, S4 and S5 reduced the biomass carbon storage by 9, 6, and 5 tC ha^{-1} yr^{-1}, respectively, in MU1. Similarly in the same order, those carbon stocks dropped by 7, 7, and 4 tC ha^{-1} yr^{-1} in MU2. Also, the difference in carbon stocks between these three scenarios was considerable in MU3, where a reduction of 6, 4, and 2 tC ha^{-1} yr^{-1} was found for S3, S4 and S5, respectively. However, in all MUs, in terms of biomass carbon storage, S4 and S5 showed a higher performance than BAU (S2), while the effect of S3 was almost similar to BAU.

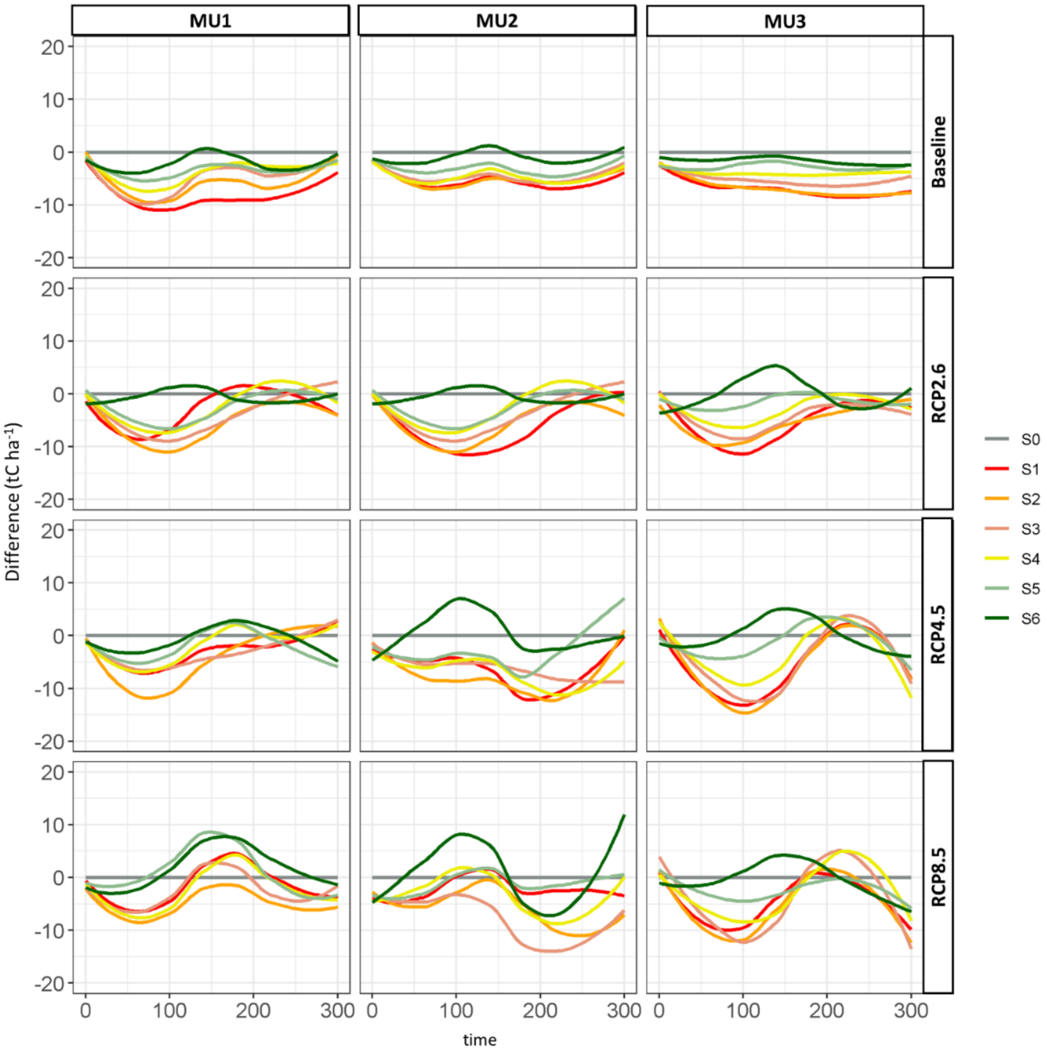

Figure 3.6 The living biomass carbon storage differences ($\Delta_{i,j}$Δi,j) from 2010 (year 0) to 2310 (year 300) between management scenarios described in Table 3.2 and no harvest scenario (S0: natural disturbances only). The relative changes compared to the natural scenario (S0) expressed in percentages and the confident interval of $\Delta_{i,j}$ are in Figure A3.5 and Figure A3.6 respectively.

3.3.4 Harvested Biomass carbon storage

The estimated AAC of 2023/2028 was around 2.3 105 Mg yr^{-1}, 3.5 105 Mg yr^{-1}, and 4.0 105 Mg yr^{-1} in MU1, MU2, and MU3, respectively (Figure 3.7). Under baseline climate scenarios for all MUs, all the tested management strategies satisfied the AAC of 2023/2028 except S6 where the total landscape was managed using only PC (Figure 3.7). In fact, S6 fulfilled around 80% of ACC under baseline climate scenario. In the short-term (2010-2110), the coniferous species presented more than 50% of annual harvested biomass, with mainly black spruce and jack pine in MU1, and black spruce with balsam fir in MU2 and MU3. After 110 years, more broadleaved species were harvested than coniferous (Figure A3.7). Also under climate change, broadleaved species were the most harvested species in the medium and long term (period 110-310). Strategies with more PCs were able to keep greater coniferous contributions in AAC under the climate change effect (Figure A3.7), since they maintained their cover retention over the medium and long-terms (Figure 3.3). The annual harvested biomass was expected to increase in MU1 and decrease in the next 100 years for both MU2 and MU3 under the current climate. However, under RCP scenarios the annual harvested biomass increased for all MUs, and all the strategies fulfilled the industrial demand. For instance, under RCP2.6 the annual harvested biomass could reach 5 105 Mg yr^{-1}, 106 Mg yr^{-1}, and 9 105 Mg yr^{-1} for MU1, MU2 and MU3, respectively. Under RCP4.5 those amounts could reach 7 105 Mg yr^{-1} in MU1 and more than 7 105 Mg yr^{-1} in MU2 and MU3. Nevertheless, the annual harvested biomass dropped considerably under RCP8.5 in MU1 after 200 years.

In order to fulfill the current AAC of 2023/2028, the harvested area using the BAU scenario (S2) was 1.32%, 2.04%, and 1.80% of the annual managed area in MU1, MU2, and MU3, respectively (Figure 3.8). When PCs were used more than CC and CPRS, the harvested area to cover current AAC increased and doubled that of S1 and S2. For instance, under the S6 strategy it reached 2.89% yr^{-1} in MU1, 4.27% yr^{-1} in MU2, and 3.55% yr^{-1} in MU3. Nevertheless, because the annual harvested biomass is projected to rise as a result of climate change (Figure 3.7), it is likely that the harvested area needed to fulfill the current AAC using S6 will be less than that indicated for the baseline climate.

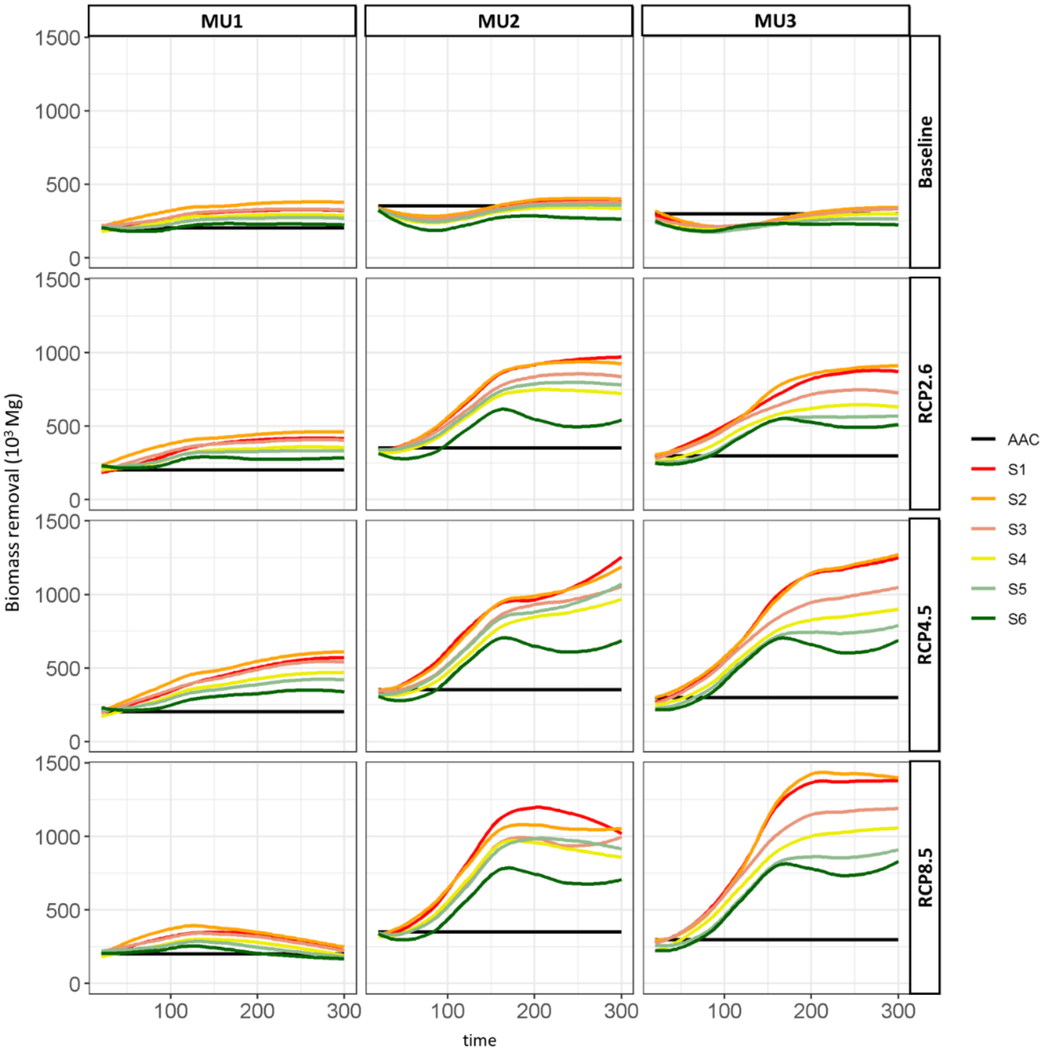

Figure 3.7 Annual harvested biomass (10^3 Mg) per management scenario under climate change effect, compared to the allowable annual Cut (AAC of 2023/2028) from 2010 (year 0) to 2310 (year 300). All the scenarios were calibrated at the beginning of simulations to cover the timber supply analysis for the period 2023-2028 (AAC).

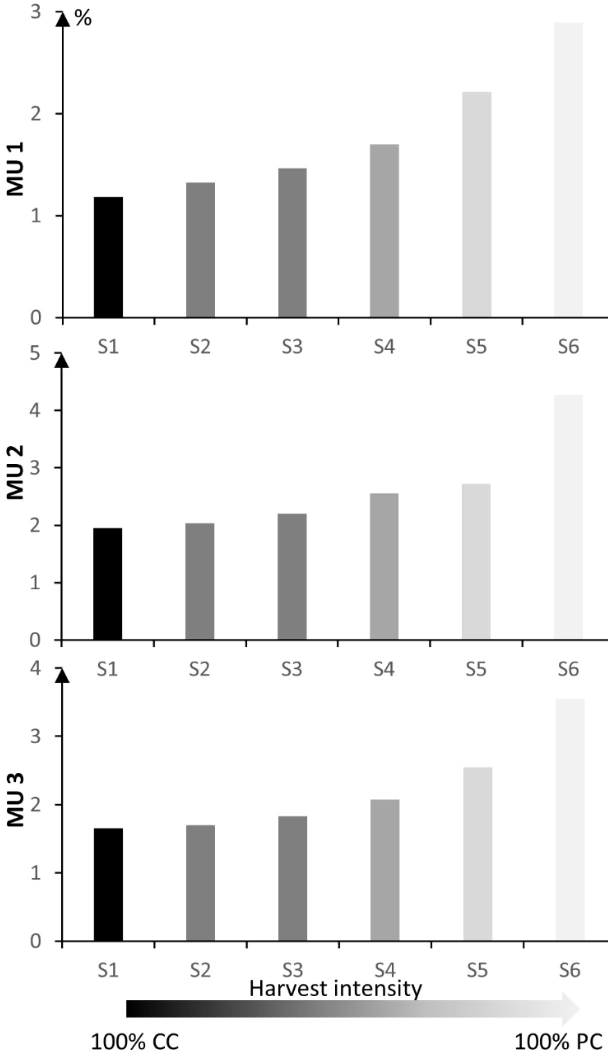

Figure 3.8 Annual harvested area (expressed in %) per management scenario. The threshold of S2 corresponds to the required annual managed area (%) to fulfill the timber supply analysis for the period 2023-2028 (AAC) under BAU scenario.

3.4 Discussion

3.4.1 Climate change and natural disturbances impacts on biomass carbon storage

Evaluating the net effects of climate change on regional carbon dynamics requires the use of forest landscape models, such as the one we present in this article, which include the main driving processes and their interactions with local conditions (soil and climate). The results of this study are in agreement with previous studies (Ameray et al., 2023a; D’Orangeville et al., 2018; Molina et al., 2021), which report that climate change will increase forest productivity in the high latitude of boreal forests. Other studies underline an increase in drought and mortality risk, mainly under RCP8.5 in the drier, western regions (Ameray et al., 2023a). According to our projections, moderate climate change scenarios (RCP2.6 and RCP4.5) will likely increase and probably double the current productivity because of the shift in forest composition toward broadleaved species and changes in the capacity of trees to grow and sequester atmospheric carbon for both coniferous and broadleaved species (Ameray et al., 2023a). This positive effect could be explained by the extension of the growing season and the reduction of potential cold-temperature late-frosts (D’Orangeville et al., 2018). Our simulated short-term (2010-2110) responses in coniferous biomass carbon showed an increase in all MUs near the Québec Forest management limit, although a reduction after 2110 was predicted in the western region. These findings are consistent with other empirical studies and model projections (*e.g.*, (Beck et al., 2011; Dhital et al., 2015)), which concluded that increases in surface air temperature could hinder the capacity of coniferous species to grow and assimilate atmospheric carbon (D’Orangeville et al., 2018). In addition, the increase in mixed forest occupancy, mostly between black spruce and balsam fir, black spruce and broadleaved species, and broadleaved pioneer forests of birch or aspen, might improve biomass carbon stock (Ameray et al., 2021; Cavard et al., 2010, 2011; Stuenzi & Schaepman-Strub, 2020). In the long term, there are a few studies that project carbon dynamics for time periods greater than 100 years in the boreal forest zone (Boulanger & Puigdevall, 2021; Paradis et al., 2019; Peng et al., 2002). Our simulations suggest either a long-term decline or a stable trend over time of coniferous biomass stock in eastern and western regions, respectively, while that of broadleaved species will increase for all RCP scenarios. Likewise, boreal conifer biomass declines are projected to be more significant with increasing anthropogenic climate forcing and with decreasing latitude (Boulanger & Puigdevall, 2021; Pau et al., 2023).

Regarding composition, all of our results are consistent with previous projections in Québec’s boreal forests (Ameray et al., 2023a; Augustin et al., 2022; Boulanger & Puigdevall, 2021; Molina et al., 2021), which state that the future boreal forest composition will feature less pure black spruce forests and higher occurrences of : a) black spruce mixed with white birch and trembling aspen, b) mixed forest of broadleaved species, c) mixed forest between black spruce and other coniferous mainly jack pine and balsam fir. Our findings suggest that climate change will favor increased abundances of white birch and trembling aspen (Boulanger & Puigdevall, 2021; Walker et al., 2017). In contrast, extreme climate scenarios (RCP8.5) will possibly result in a more significant decline in coniferous communities compared to RCP2.6 and RCP4.5 after 2100 in the western regions of Québec’s boreal forest (Ameray et al., 2023a). Similarly, Boulanger & Puigdevall (2021) found that pioneer species proportions in the boreal forest could increase by 10–50% relative to the baseline climate scenario. In addition, natural disturbances could alter forest composition. For instance, Molina et al. (2021) pointed out that an increase in fire events is accompanied by an increase in mixed broadleaved species stand proportions in the landscape for western regions. Additionally, in MU3 under current climate and the no-harvest scenario, compared to Ameray et al. (2023a) where SBW was not considered, black spruce and balsam fir occupancy was around 30% in the medium and long term, but in this study this composition was reduced to 21% because of the effect SBW (Figure

A3.2).

Carbon dynamics in North American boreal forest ecosystems are strongly affected by tree mortality that occurs during SBW outbreaks, fire events and windthrows. Our simulations show an increase in broadleaved species biomass carbon storage under SBW outbreaks. In fact, SBW killed the host tree species, i.e., balsam fir, white spruce, and black spruce, and increased empty cells which regenerated later with broadleaved species, thus explaining these increases. Similar to the findings Liu et al. (2019), SBW defoliation and related mortality decreased the average coniferous biomass carbon stock by around 5%-6 % under current climate. SBW outbreaks typically result in an average 42–50% stand mortality and a loss of biomass production of 32–48% (Navarro et al., 2018; Paixao et al., 2019), with a value of 20%- 30% being found for the eastern units. Our findings are consistent with previous studies which report that an increase in wildfire activity could explain the declines in biomass (Boulanger & Puigdevall, 2021; Molina et al., 2021). The carbon losses from the biomass pool under fire events may reach 10tC yr^{-1} in MU1 and MU2 by 2100 under RCP8.5 (Figure A3.4) (Ameray et al., 2023a), which could be explained by the increase in annual burn rate that may reach more than 1% and 1.25 % in MU1 and MU2, respectively (Bergeron et al., 2006). Fire cycles are generally shorter in the western regions, resulting in a younger post-fire forest with patches of older forest dispersed throughout with lower biomass carbon stock (Bergeron et al., 2017; Boulanger & Puigdevall, 2021). Under the current and extreme climate change scenario (RCP8.5), Splawinski et al., (2019) predict that over the next 50 years, the amount of forest area affected by natural regeneration failure will gradually increase because of increasing fire intensity, meaning the open woodland areas in our study might be underestimated, primarily in MU1 and MU2.

Windthrows are likely to further reduce carbon sink strength in specific regions of the boreal zone, but are not well quantified in previous works. In our simulations, windthrows were a minor driver compared with other natural disturbances and had a minor effect on biomass carbon dynamics compared to wildfires and SBW. Windthrow impacts in Canada’s boreal forest appear to be limited to occasional local events (Bouchard et al., 2009a; Kurz et al., 2013). We emphasize that there was a high synergy between forest species to offset carbon losses under natural disturbances, as broadleaved species could offset the carbon losses from coniferous species (Figure A3.4). In fact, a reduction in coniferous productivity is synchronized with an increase in that of broadleaved pioneer forests after disturbance (Dulamsuren, 2021). The occurrence of birch (or aspen) as pioneer trees after disturbance is widely observed in the boreal forest biome (Boulanger et al., 2019; Molina et al., 2021; Stuenzi & Schaepman-Strub, 2020).

3.4.2 Future management strategies

The impact of PCs has been identified as one of the major knowledge gaps in regional and global carbon accounting. In the last decade, PCs have started gaining interest as alternatives to CC to mitigate climate change and several studies (e.g., Ameray et al., 2021; Peng et al., 2002; Taylor et al., 2008) show that they could enhance carbon sequestration rates. Based on our results at stand scale (1 cell of 4ha), PC treatments were close to natural dynamic scenarios and could stabilize biomass carbon storage in the long term at stand scale (Figure A3.8). This effect could be explained mainly by the reduction in competition between cohorts for light and water since PC opens the canopy and increases available light and water for growth. Furthermore, PCs could reduce regeneration failure by conserving seed-trees and decreasing biotic and abiotic stresses through microhabitat modification under the tree canopy, resulting in a higher rate of germination and greater seedling survival (Guignabert et al., 2020). Therefore, at the landscape scale the PC based scenarios could be favoured in respect to those based on CC treatments (Boulanger & Puigdevall, 2021; Lee et al., 2002; Molina et al., 2021; Taylor et al., 2008). Similarly, Lee et al. (2002) and Taylor et al. (2008) affirms that the positive effect of PC on forest carbon sequestration depends upon CRI. In addition, Simard et al. (2020) found that the high intensity 1-year post-harvest decrease in total biomass carbon stocks and the magnitude of these losses were negatively correlated with climatic aridity. From a long-term study, Peng et al. (2002) concluded that PC could increase carbon sequestration by about 36%-40% in the boreal forest region. On the other hand, a few recent studies showed that there is a high interaction between PCs forms (e.g., shelterwood cutting, selection (distant or close), retention systems, and seed-tree systems) and increased risks of disturbances, mainly windthrow (Girona et al., 2019b). Girona et al. (2019b) stated that 60% of residual trees were dead in seed-tree treatments, compared to 30% for shelterwood cuts. Therefore, the success of the PC approach requires consideration of not only its intensity but also its form (Ameray et al., 2021). Our study did not consider such interactions between PCs and windthrow as Landis-II is not spatially explicit within a cell.

In the eastern boreal forests of Canada, transient changes could be largely tackled through management interventions, such as changes in harvesting intensity and using more PCs at the landscape scale than CC. Our analyses showed that the BAU scenario will strongly decrease forest inertia and will interact with anthropogenic climate forcing to further alter forest landscapes in all MUs. Furthermore, strategies based on more CC and CPRS could accelerate the abundance of pioneer species and decrease coniferous communities at the landscape scale, which could explain the accelerated increase in broadleaved species biomass and the decrease in that of coniferous species for the scenarios which used more of those treatments. This study provides two promising strategies (S4 and S5) with different harvesting intensities, which could be the best direction for increasing forest carbon sequestration capacity and maintaining other ecosystem services (Ameray et al., 2021). These two strategies can maximize forest carbon biomass at the landscape scale and fulfill industrial needs, which ensures a positive trade-off between carbon sequestration and harvested wood products.

Reforestation is used in Quebec boreal forests mainly after CC and CPRS where regeneration is insufficient (Boucher et al., 2012; Gaboury et al., 2009). Our study shows that strategies with high reforestation rates may not achieve greater carbon sequestration than the natural scenario where carbon sequestered in harvested wood products is not taken into account, e.g., strategy S3 where the rate was 25% of the annually managed area. It must also be considered that CC based scenarios (S1, S2, S3) harvest less area in addition to the high carbon transfer to harvested wood products (Figure 3.7), compared to other PC based strategies. Furthermore, we emphasize that strategies with high reforestation rates, such as like S3 (25% on annual harvest area), cannot achieve as high a cumulative photosynthesis value compared to the natural scenario, and their effect is almost similar to the BAU under different climate change scenarios. In terms of forest carbon stabilization at the ecosystem scale, our findings showed that scenarios S4 and S5 seemed to be the best ones in MU1 and MU2 under the current climate. The annual reforestation for the BAU is 15% and 10% in MU1 and MU2, respectively, which means that currently we are close to the S4 and S5 reforestation rates in these units (Table A3.3). Therefore, S4 and S5 could be the best ones for MU1 and MU2. In the MU3 region, the increase in the reforestation rate is recommended to be at least 8.3%, which corresponds to S5 strategy. Still, using S4 in MU1 with a 16.7% coniferous reforestation could show similar results to the BAU under RCP8.5, because their net primary productivity and biomass stocks were projected to be lower under extreme climate scenarios in this region (Ameray et al., 2023a).

Forests are currently managed for multiple goals and benefits and not only to improve carbon sequestration (FAO, 2020b), and, because climate change is a dynamic and complex phenomenon with high uncertainty, any future strategy should at least maintain similar vegetation patterns and age structures to the natural scenarios under different climate change scenarios. With increasing pressures from global changes, sustainable forest management can probably be accomplished with an efficient combination of PCs, CC, and CPRS. Increasing the share of protected forest areas for biodiversity conservation is necessary, but has proven to be difficult, given mounting land-use pressures. In these circumstances, promoting strategies such as S4 and S5 that could maintain the old-growth forest, uneven-aged forest structures, and deadwood (enriching soil organic carbon) may be required to halt further degradation of biodiversity (Harvey et al., 2002; Martin et al., 2022). Martin et al. (2022) states that silvicultural alternatives such as continuous cover by using PCs or retention forestry have the potential to restore and protect the habitats and functions of boreal forests. Our results show that in order to fulfill the current AAC level, the BAU scenario harvested 1.32%, 2.04%, and 1.80% of the annual managed area in MU1, MU2, and MU3 respectively, while those thresholds could be double under S4 and S5. All previously mentioned evidence indicates that PCs could help achieve sustainable management, yet the big challenge will be operational barriers. In fact, in order to harvest the same volume using PC as with CC, PC must be applied in different places and requires greater access, as compared to CC, thereby making PC more costly, as well as resulting in a potential loss of carbon stocks due to the extent of the required road network (Ameray et al., 2021).

The current goal of forest management in Québec and in many jurisdictions in the world is to be ecosystem-based, which aims to maintain historical forest composition and structure in an attempt to imitate natural disturbance regimes and preserve natural vegetation patterns (Ameray et al., 2021; Bergeron et al., 1999). Accordingly, our two proposed strategies, S4 and S5, may achieve ecosystem-based management targets since they deviate the least from the vegetation pattern and age structure of the natural scenario. However, the inclusion of more PCs requires increased roads networks with consequent losses in forest area and associated carbon sinks, leading to lower landscape connectivity and increased habitat fragmentation for fauna (e.g. caribou) (Prima et al., 2019), thus requiring additional effort to consider more ecosystem services in forest landscape models. On the other hand, it seems that using PC based strategies could maintain greater coniferous cover and better preserve current habitats than those based on CC and CPRS. PC also provides quality wood with added value because of higher stem diameters and lower impacts on wildlife at a local scale (St-Laurent et al., 2022). In fact, PC maintained mature and old-growth forests, which are distinguished by large-diameter trees, whereas CC increased young forests, which are characterized by lower-diameter trees. However, the implementation of strategies with more PCs requires further road access and appropriate infrastructure, meaning a higher investment cost (Ameray et al., 2021, 2023a) and increased fragmentation of fauna habitats at landscape scale.

3.4.3 Modelling limitations and improvements

Our study examines a gradient of forest management strategies running from high to lower harvest intensities at the stand scale and does not account for all of the many site-level conditions. This study does not consider paludified soil in MU1. Although the waterlogging and drought tolerance parameters (H1–H4) were considered, nutrient limitations also reduce productivity, and this is not currently modelled in PnET-Succession. The PnET model simulates stand scale dynamics and extrapolates the results at the landscape scale, hence the integration of the nutrient-stand relationship in the model could be a promising improvement. In addition, the model was optimistic about broadleaved species regeneration under the climate change effect. In fact, the current soil conditions (paludification in MU1, organic and nutrients-poor soils) may limit dispersal for most commercial broadleaved species (Gewehr et al., 2014). In the same MU, Ameray et al (2023a) report that under a broadleaved species dispersal restriction scenario, the open forest woodlands will increase for both the RCP4.5 and RCP8.5 scenarios, with more than 10 and 20% in MU1 and MU2, respectively. In contrast, in MU3 the open forest woodland will be around 10% for RCP4.5 and RCP8.5. Consequently, it is possible that some of the increases in projected productivity under the RCP 2.6, 4.5, and 8.5 emissions scenarios may not be realized because of nutrient and soil limitations.

In the boreal forest, the carbon cycle depends on the long-term balance between vegetative carbon inputs from litterfall and root turnover and carbon outputs derived from organic matter decomposition (Laganière et al., 2013). PCs may reduce decomposition and increase soil carbon storage. In the PC system, higher productivity (*i.e.*, more litterfall input) and lower decomposition rates could be achieved by maintaining a constant level of growing biomass (Taylor et al., 2008). Future assessments need to integrate the soil carbon pool and estimate both net ecosystem and net biome production, as well as the potential for the forest sector to mitigate climate change, by incorporating harvested wood products in the balance. The PCs positive effect on productivity could be overestimated because our study did not consider the forest edge effect and the interactions with windthrow and SBW. We also highlight that the base harvest model used from LANDIS-II utilizes annual harvested areas rather than volumes, which explains why the model removes less or more biomass than the AAC (Figure 3.8). However, our experiment showed the long-term consequences of alternative forest management strategies at the landscape scale compared to the current BAU under climate change effects. Despite current limitations, our study provides proof-of-concept assessment of the ability of a mechanistic forest landscape model to conduct simulations in order to reduce the uncertainty surrounding the ability of climate-adaptive silvicultural strategies to achieve their stated objectives (Gustafson et al., 2020, 2022).

There are known issues with the model we used (V.4.1) and a new version (beta V5) is due to be released, where these problems are addressed. Some of the known issues with the model we used are an underestimation of drought effects on forest growth and other problems related to soil texture that causes more water to be retained in coarse soils (*e.g.*, sand compared to loam). However, the LANDIS-II Model is constantly being improved, and we used the latest official model release (V4.1). Any switch to the Beta V5 version requires a potential adjustment of our parameters to achieve similar yield curves for each species during the calibration step (Ameray et al., 2023a). The use of Version 5 may raise the mortality rate, which means that the 4.1 version could lead to an overestimation of biomass carbon storage and an underestimation of the mortality rate caused by drought or competition in this work, particularly for late-successional conifers under extreme climate scenarios (RCP8.5) (Ameray et al., 2023a).

3.5 Conclusions

Boreal forests in eastern Canada are undergoing intensive harvesting using CC and CPRS, requiring new strategies to achieve sustainability and contribute to climate change mitigation and adaptation. This study indicates that the inclusion of more partial cuts could stabilize forest carbon under the current climate and may even exceed that of a natural scenario (no-harvest) under climate change. Furthermore, in our simulations, PC-based scenarios (S4, S5 and S6) reduced the occupancy of broadleaved species, maintained greater coniferous cover, and allowed a larger contribution of coniferous trees in the future merchantable harvested biomass, mainly after 2110. Under different emission pathways, the losses from CC-based scenarios (S1, S2 and S3) could be compensated in the long term, since it accelerated the expansion of trembling aspen and white birch, while those based on PC- treatments increased coniferous cover retention, but that shift in composition could prove challenging to the forest sector. Under climate change effect, the industrial sector must prepare for shifts in forest composition, this involves transitioning to innovative products based on broadleaves instead of conifers.

The application of PC-based scenarios under current climate conditions requires forest infrastructural reorganization which will be a big challenge in the future. In addition, the S6 scenario where the entire annual managed area is undergoing PCs treatments might not fulfill industrial needs under the current climate. The application of S4 and S5 requires increasing the coniferous reforestation rate in Eastern regions (MU3) to 8.3-16.7 % of the annual harvested area. In addition, CC-based scenarios harvest less area with high annual harvested biomass compared to those based on PC treatments, requiring a better coupling of harvested wood products' life-cycle analyses with landscape dynamics models in future research.

CHAPTER IV: TWO CENTURIES OF CARBON DYNAMICS IN THE EASTERN CANADIAN BOREAL FOREST UNDER VARIOUS MANAGEMENT STRATEGIES AND CLIMATE CHANGE PROJECTIONS

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CHAPITRE IV : DEUX SIÈCLES DE DYNAMIQUE DU CARBONE DANS LA FORÊT BORÉALE DE L'EST DU CANADA SELON DIVERSES STRATÉGIES D'AMÉNAGEMENT ET PROJECTIONS DU CHANGEMENT

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Abstract

Partial cutting has lower canopy removal intensities than clearcutting and has been proposed as an alternative harvesting approach to enhance ecosystem services, including carbon sequestration and storage. However, the ideal partial cutting/clearcutting proportion that should be applied to managed areas of the eastern Canadian boreal forest to optimize long-term carbon sequestration and storage at the landscape scale remains uncertain. Our study projected carbon dynamics over 200 years (2010–2210) under a portfolio of management strategies and future climate scenarios within three boreal forest management units in Quebec, Canada, distributed along an east–west gradient. To model future carbon dynamics, we used LANDIS-II, its Forest Carbon Succession extension, and several extensions that account for natural disturbances in the boreal forest (wind, fire, spruce budworm). We simulated the effects of several management strategies on carbon dynamics, including a business-as-usual strategy (clearcutting applied to more than 95% of the annually managed area), and compared these projections against a no-harvest natural dynamics scenario. We projected an overall increase in biomass and dead organic matter carbon stocks, mostly because of increased productivity and broadleaf presence under limited climate change. The drier western region under climate scenario RCP8.5 was an exception, as stocks decreased after 2100 because of the direct negative effects of extreme climate change on coniferous species' productivity. The natural dynamics scenario projected a negative or null net ecosystem productivity (~0 tC ha^{-1} yr^{-1}). Applying partial cutting to more than 50% of the managed forest area mitigated the negative impacts of climate change on carbon balance and minimized differences in terms of stand composition, age structure, and carbon sequestration and storage between forests in the natural dynamics scenario and forests managed for timber provisioning. Clearcutting-based scenarios, including the business-as-usual strategy, reduced total annual ecosystem carbon storage by approximately double (10 tC ha^{-1} yr^{-1}) that of partial cutting–dominated scenarios (<5 tC ha^{-1} yr^{-1}). Nevertheless, relative to partial cutting based strategies, clearcutting can potentially increase net ecosystem productivity over the long term by increasing the abundance of young forests and broadleaf species. Our findings illustrate the need to balance diverse canopy removal intensities in managed boreal forests to enhance carbon sequestration and storage while preserving other ecosystem qualities in the context of present and future climate change.

Keywords: Landis-II, boreal forest, carbon sequestration and storage, forest management, heterotrophic respiration, net biome production, Partial cuts, clear cuts

Résumé

Les coupes partielles basées sur des intensités de récolte inférieure à celle de la coupe totale et a été proposée comme une méthode de récolte alternative pour améliorer les services écosystémiques, y compris la séquestration et le stockage du carbone. Cependant, la proportion appropriée de surface annuellement récoltée par traitement aux forêts boréales de l'est du Canada pour optimiser la séquestration et le stockage du carbone à long terme à l'échelle du paysage reste incertaine. Notre étude a projeté la dynamique du carbone sur 200 ans (2010-2210) en fonction d'un portefeuille de stratégies de gestion et de scénarios climatiques futurs au sein de trois unités d'aménagement au Québec, Canada, réparties le long d'un gradient est-ouest. Pour modéliser la dynamique future du carbone, nous avons utilisé LANDIS-II, son extension Forest Carbon Succession et plusieurs extensions qui tiennent compte des perturbations naturelles (vents, feux, tordeuse des bourgeons d'épinette noire). Nous avons simulé les effets de plusieurs stratégies de gestion sur la dynamique du carbone, y compris la référence (stratégie BAU : traitements des coupes totales appliquées sur plus de 95 % de la superficie annuelle gérée), et nous avons comparé ces projections à un scénario de dynamique naturelle sans récolte. Nous avons projeté une augmentation globale des stocks de carbone dans la biomasse et la matière organique morte, principalement en raison de l'augmentation de la productivité et de la présence des feuillus. Les régions occidentales plus sèches dans le cadre du scénario climatique RCP8.5 constitue une exception, les stocks ayant diminué après 2100 en raison des effets négatifs directs du changement climatique extrême sur la productivité des espèces de conifères. Le scénario dynamique naturelle prévoyait une productivité nette de l'écosystème négative ou nulle (~0 tC-ha^{-1}-an^{-1}). L'application de coupes partielles à plus de 50 % de la zone forestière gérée a atténué les effets négatifs du changement climatique sur le bilan carbone et minimisé les différences en termes de composition, structure d'âge, séquestration et stockage du carbone entre les forêts du scénario de dynamique naturelle et les forêts aménagées pour l'approvisionnement en bois. Les scénarios basés sur les coupes totales, y compris BAU, ont réduit le stockage annuel total de carbone de l'écosystème d'environ le double (10 tC ha^{-1} an^{-1}) de celui des scénarios dominés par les coupes partielles (<5 tC ha^{-1} an^{-1}). Néanmoins, par rapport aux stratégies basée sur des coupes partielles, les coupes totales peuvent potentiellement augmenter la productivité nette de l'écosystème à long terme en augmentant l'abondance des jeunes forêts et des espèces des feuillus. Nos résultats illustrent la nécessité d'équilibrer et de diversifier les intensités de récoltes dans les forêts boréales Québécoise afin d'améliorer la séquestration et le stockage du carbone, tout en préservant d'autres qualités de l'écosystème dans le contexte du changement climatique actuel et futur.

Mots-clés : Landis-II, forêt boréale, séquestration et stockage du carbone, aménagement forestier, respiration hétérotrophe, production nette du biome, coupes partielles, coupes totales.

4.1 Introduction

Forests store about 50% of the world’s terrestrial carbon, making these biomes important for climate change mitigation (Batjes, 1996). Forest ecosystems remove nearly 2 PgC yr^{-1} of carbon from the atmosphere through photosynthesis, absorbing about 30% of anthropogenic CO2 emissions (Bellassen & Luyssaert, 2014; Köhl et al., 2015). The boreal forest is the second-largest terrestrial biome, represents 27% of the world’s forests in 2020, and provides 7% of the global wood supply (Dixon et al., 1994; Gauthier et al., 2015; Grace, 2005). It stores 88 GtC in living biomass and 471 GtC in soils (Dixon et al., 1994), the latter corresponding to 60% of the world’s soil organic carbon (Pan et al., 2011). Moreover, boreal peatlands hold an additional estimated soil carbon stock of 260–600 PgC (Apps et al., 1993; Tarnocai et al., 2009). However, carbon sequestration and storage in the boreal forest are sensitive to climate change (direct climate effects on growth, mortality, and decay rates), natural disturbance (the frequency and severity of these events will be altered by climate change), and forest management practices, all of which could alter boreal forest composition and structure and switch areas of the boreal forest from carbon sinks to sources (Ameray et al., 2021; Boulanger et al., 2019; Girona et al., 2023; Moreau et al., 2022).

Forest management has the potential to alter the carbon pool and flux dynamics; the extent of this influence depends on the applied canopy removal intensity (CRI). Partial cutting (PC), having a low–moderate CRI, reduces competition and favors a gradual shift in stand structure from stands with many trees of low individual biomass toward stands having fewer but larger trees (Ameray et al., 2021; Moussaoui et al., 2020; Taylor et al., 2008). PC also maintains uneven age structures and continuous cover, thereby ensuring some degree of carbon storage; the actual amount depends on the CRI (Ameray et al., 2023b). In contrast, clearcutting (CC) leads to a decrease in total ecosystem carbon storage over decades. In some cases, it may enhance the decomposition process and heterotrophic respiration (R_h) over a shorter period because CC exposes deadwood and litter to light and stimulates microbial activity (Campbell et al., 2009). Moreover, at a stand scale, CC can temporarily shift forests from being carbon sinks to sources by increasing R_h and reducing the photosynthetic biomass in the post-harvest period (Taylor et al., 2008). If regeneration occurs without any unusual delays, these stands return to their carbon sink status generally 20 years after harvest (Covington, 1981; Goulden et al., 2011). The effects of PC and CC on soil carbon stocks appear minor (Ameray et al., 2021; Mayer et al., 2020). However, CC has been shown to reduce total soil organic carbon storage relative to PC (Goulden et al., 2011; Jandl et al., 2007), although the degree to which CC affects this carbon pool relates to site conditions, mainly temperature which may increase R_h, as well as carbon transfer from the litter (e.g., leaves, lifted branches post-cutting) to the soil.

The managed forests in Quebec’s boreal zone (70% of the total Quebec boreal forest area) are harvested for wood production (NFD, 2023). These forests are subject to high-CRI silvicultural practices, such as CC and careful logging around advanced growth (CLAAG). The latter is also known as cutting with the protection of regeneration and soil (CPRS) and reduces the canopy by about 95% while attempting to preserve advanced regeneration (Girona et al., 2023; MRNF, 2010). The even-aged management systems using CC and CPRS are currently used in more than 95% of the annual harvested area (MRNF, 2010). on the other hand, uneven-aged systems using PC are applied across less than 5% of the annually harvested area. These PC treatments include shelterwood cutting and commercial thinning with the protection of small merchantable stems (MRNF, 2010).

Currently in Quebec, sustainable forest management is the primary goal of ecosystem-based forest management (EBFM), which aims to minimize the differences between managed and natural forests with the underlying goal that the applied EBFM approaches preserve biodiversity and ensure the supply of ecosystem goods and services (Girona et al., 2023). EBFM strategies must also be designed to offer the capacity for climate change mitigation (Girona et al., 2023). Smyth et al. (2014) identified that reduced harvesting levels and an improved selection of trees to produce long-lived wood products could offer an optimal mitigation strategy for forests within the Eastern Boreal Shield region of Canada. However, the potential long-term impacts of harvesting on the overall carbon state of boreal forests in the context of climate change remain unknown. Moreover, novel silvicultural approaches must be evaluated to address the challenges facing the eastern Canadian boreal forest under future environmental conditions, including warmer temperatures, altered natural disturbance regimes, greater forest fragmentation, and a reduced extent of old-growth forest (Girona et al., 2023).

Quebec has set ambitious greenhouse gas reduction targets to achieve carbon neutrality by 2050. Achieving this goal may be facilitated by the presence of natural sinks that can be enhanced further by placing a particular emphasis on forest management (Krug, 2018). Therefore, it is crucial to understand the short- and long-term forest carbon dynamics at a regional scale and include the effect of various forest management strategies and silvicultural practices within the context of climate change. Moreover, the short- and long-term impacts of fire, windthrow, and spruce budworm (SBW; Choristoneura fumiferana) outbreaks on carbon dynamics must be considered in eastern Canadian forests (Boulanger et al., 2012; MacLean, 2016). In these forests, climate change is expected to shorten fire-return intervals and heighten fire sizes and intensities (Boulanger et al., 2014). Additional studies are needed to fully understand the cumulative effects of natural and anthropogenic disturbances on carbon stocks and fluxes under climate change and how these vary along a longitudinal gradient.

Here, we run a forest landscape model over two centuries (2010–2210) and examine forest carbon dynamics under various climate change (representative concentration pathway; RCP) and management scenarios. We also integrate more frequent natural disturbances and direct climate impacts on growth and mortality. We also vary the CRI and the annually harvested area per treatment to identify the optimal strategies for increasing forest carbon sequestration and storage capacity at the landscape scale. Specifically, we aim to i) quantify the isolated effects of climate change on carbon dynamics under natural dynamic scenario; and ii) investigate the effect of management strategies on carbon dynamics under these same climate projections. This study will improve our understanding of the driving factors behind the carbon balance of Quebec's managed boreal forests, evaluate the potential of these forests to mitigate future climate change, and help guide forest managers in achieving sustainability in the boreal forest.

4.2 Materials and Methods

4.2.1 Study area

Our study area comprises three management units (MUs) located in several sensitive areas near the Quebec northern limit of commercial forestry (Jobidon et al., 2015)—beyond this northern limit, forests are not managed for timber production because of their lower productivity. The three MUs—Nord-du-Quebec (MU1), Saguenay-Lac-Saint-Jean (MU2), and Côte-Nord (MU3)—lie within the spruce–feathermoss and balsam fir–white birch bioclimatic domains of the Boreal Shield along an east–west gradient (Figure 4.1). These landscapes are dominated by black spruce (*Picea mariana*), jack pine (*Pinus banksiana*), balsam fir (*Abies balsamea*), white birch (*Betula papyrifera*), and trembling aspen (*Populus tremuloides*) (MRNF, 2010). The dominant age class is 20–40 years in MU1 and 120–200 years in MU2 and MU3 (Figure 4.1). Annual precipitation averages 800 mm to 1000 mm with a strong longitudinal gradient with greater precipitation moving eastward (Wang et al., 2016).

Each MU has a marked north–south temperature gradient (Wang et al., 2016). The dominant soil texture is clay in MU1 and sandy loam in MU2 and MU3 (Ameray et al., 2023a; Duchesne & Ouimet, 2021). Our simulations did not consider unproductive areas, including wetlands, islands, and noncommercial species.

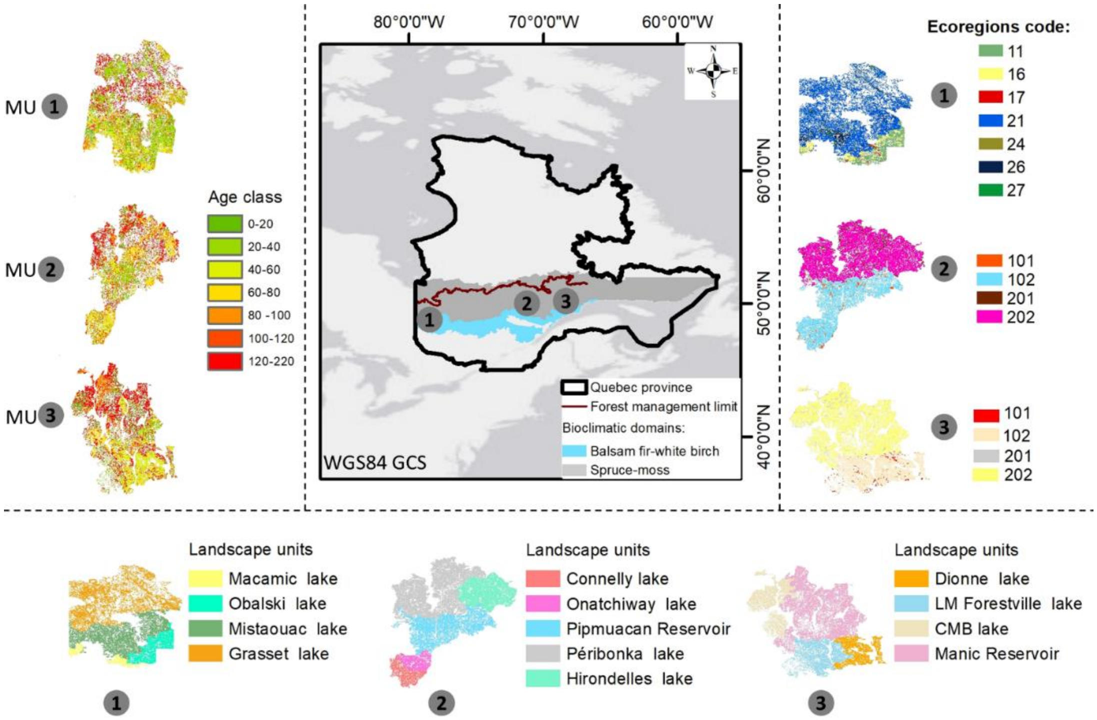

Figure 4.1 Location of the three studied management units in Quebec, Canada: Nord-du-Quebec (MU1), Saguenay-Lac-Saint-Jean (MU2), and Côte-Nord (MU3)). For each management unit (MU), we present the forest age class structure at the beginning of the simulation (year 2010), the landscape units used as managed areas (MRNF, 2010), and the ecoregion code used during the simulation.

4.2.2 Simulation Models

4.2.2.1 LANDIS-II and extensions

We simulated the forest dynamics using LANDIS-II, a stochastic and spatially explicit forest landscape model widely used to integrate forest succession, management, and natural disturbances and to simulate forest degenerative processes (senescence and mortality) at spatial (~10^5–10^7 ha) and temporal (≥100 years) scales (Mladenoff & He, 1999; Scheller et al., 2007). LANDIS-II can project changes in species composition, biomass, carbon stocks, natural disturbances, etc. (Mladenoff & He, 1999). We conducted our simulations at a spatial resolution of 200×200 m^2 (4 ha) and at a 1-year time step over 200 years (2010–2210). The simulated areas covered 0.62 Mha, 1.16 Mha, and 1.15 Mha for MU1, MU2, and MU3, respectively. LANDIS-II simulates forest succession and productivity at both the stand and landscape scales under different environmental conditions and disturbances using a variety of extensions. Each extension requires independent parameterization and calibration to reflect the current and future forest state. We used the extensions Forest Carbon Succession (ForCS) v 3.1 (Dymond et al., 2016), Base Fire V4.0 (Scheller & Domingo, 2018), Biomass Harvest V4.4 (Gustafson et al., 2000), Base Biological Disturbance Agent V4.0.1 (Sturtevant et al., 2004), and Base Wind V3.1 (Scheller et al., 2018).

4.2.2.2 Initial landscape: climate, species and ecoregions

The 2010 spatial forest inventory data set maintained by the MRNF (Quebec's Ministère des Resources naturelles et des Forêts) provided the species and age information for the initial communities mapped at a 200 m resolution. Each species in the data set is associated with its life-history attributes collected from previous studies (Boulanger et al. 2017, 2022), including longevity, sexual maturity, shade tolerance, fire tolerance, seed dispersal distance, sprouting, and post-fire regeneration (Table 4.1) (Ameray et al., 2023a; Boulanger et al., 2017; Molina et al., 2021). Each cell (4 ha) in the landscape is assigned to a single land type where soil and climate conditions are assumed to be homogeneous, and growth and reproduction functions are unique.

For each ecoregion, we collected historical monthly weather data and RCP scenarios (RCP2.6, RCP4.5, RCP8.5) from the ClimateNA model (CanESM2 projections), a local downscaling model that facilitates extracting climate data for specific locations (longitude, latitude, elevation) in North America (Wang et al., 2016). For the current baseline climate, we extrapolated the historical monthly climatic data from 1991 to 2010, including minimum and maximum temperatures and precipitation, on the basis of their Gaussian distribution around the mean so that the baseline climate would be constant (Ameray et al., 2023a). The CanESM2 projections have mean annual temperatures increasing respectively by about 2.5, 4, and 7 °C for RCP2.6, RCP4.5, and RCP8.5 by 2100 relative to the current baseline climate (Figure 4.2). Boulanger and Puigdevall (2021) found that CanESM2 projections tended to be warmer and drier relative to 26 other models used to project climate change in Quebec. We used ecoregions from Ameray et al. (A2023a), which are delimited according to the Duchesne and Ouimet (2021) soil map and the bioclimate regions of MRNF geodatabase (Figure 4.1, Table A4.1). Duchesne and Ouimet (2021) modeled and mapped particle size composition (clay, silt, and sand) for the entire managed forest in Quebec. Relying on a decision tree algorithm, Ameray et al. (2023a) applied these soil data to categorize the soil texture for all MUs using clay, silt, and sand percentages for each 4 ha cell; the obtained soil textures were overlapped with the MRNF bioclimatic regions.

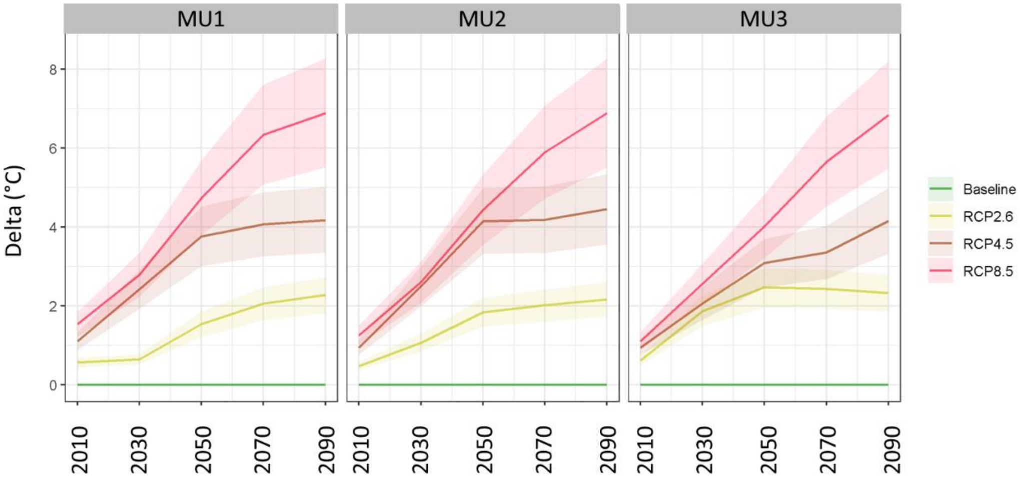

Figure 4.2 Differences in mean annual temperature (delta; °C) between climate change scenarios (RCP2.6, RCP4.5, and RCP8.5) and current climate from 2010 to 2090 for the zone between 49° and 50° lying within the boreal forest in Quebec (Wang et al., 2016).

Table 4.1 The life-history attributes for the 8 species (L: longevity (years), SM: sexual maturity (years), ST shade tolerance, SDD: seeds dispersal distance (m), VRP: vegetation reproduction probability and age, PFR post-fire regeneration strategy (Boulanger et al., 2017; Molina et al., 2021).

Species	CODE	L	SM	ST	FT	SDD		VRP	VRP min age	VRP max age	PFR
						effective	max				
Abies balsamea	BF	150	30	5	1	25	160	0	0	0	none
Betula alleghaniensis	YB	220	40	3	1	100	400	0.1	10	180	resprout
Betula papyrifera	WB	140	20	2	1	100	1000	0.5	10	70	resprout
Larix laricina	LT	160	40	1	1	50	200	0	0	0	none
Picea glauca	WS	200	30	3	2	100	300	0	0	0	none
Picea mariana	BS	220	30	4	2	80	200	0	0	0	serotiny
Pinus banksiana	JP	140	20	1	2	30	100	0	0	0	serotiny
Populus tremuloides	TA	130	20	1	2	500	5000	0.9	10	130	resprout

4.2.2.3 ForCS Calibration and parameterization

The ForCS (v3.1) extension for LANDIS-II calculates how cohorts of trees reproduce, age, grow, and die (Dymond et al., 2016). The accumulation of biomass carbon through growth and reproduction generally follows the Biomass Succession (v5.7) extension and the methods outlined in Scheller and Mladenoff (2004). This extension also tracks the evolution of forest stands and carbon dynamics, including carbon turnover, net growth, net primary production (NPP), heterotrophic respiration (R_h), net ecosystem productivity (NEP), net biome productivity (NBP), transfers between pools, losses from the ecosystem because of logging, and carbon emissions because of decay or combustion (Dymond et al., 2016; Hof et al., 2017). Moreover, as described in Dymond et al. (2016), the ForCS DOM and soil dynamics are built from the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) model. CBM-CFS3 implements a Tier 3 approach of the Intergovernmental Panel on Climate Change (IPCC) good practice guidance for reporting on carbon stocks and changes resulting from land-use change and forestry. Consequently, we used the default parameters from CBM-CFS3, particularly for DOM carbon transfers and disturbance matrices (Kurz et al., 2009).

ForCS requires species establishment probability (*SEP*), maximum aboveground NPP (*maxANPP*), and maximum aboveground biomass (*maxAGB*) as inputs (Figure 4.3). The *SEP*, *maxANPP*, and *maxAGB* were derived directly from the PnET-output data using the PnET-output site extension. First, the calibrated parameters of PnET-Succession version 4.1 were adjusted for the new version Beta V5 (Ameray et al., 2023a) (Figure A4.1). PnET-Succession is a mechanistic model based on ecophysiological parameters. To adjust the parameters, we relied on our previous work for the same MUs, run using Pothier and Savard's (1998) yield curves for the historical climate. For each climate change scenario, we used PnET-Succession version Beta V5 to estimate ForCS input parameters (*SEP*, *maxANPP*, *maxAGB*) for each monospecific stand per species for all land types (ecoregions) using a 10-year time step (Figure A4.2). The *SEP* is simulated in the PnET model as a function of light and water photosynthesis reduction factors for the species at the time of establishment. Establishment probabilities were adjusted for an annual time step using the properties of Bernoulli's trials, assuming that the species could establish at least one time or more [*SEP* $(X \geq 1)$] in 10 successive trials. All simulations were run in PnET-Succession on a single cell to drive each simulation for 140 years, starting from a single 20-year-old cohort on bare ground. According to Tremblay et al. (2018), most of the studied species' aboveground biomass reaches a plateau around 100–140 years.

For the baseline climate scenario, we estimated the parameters *SEP*, *maxANPP*, and *maxAGB* using the extrapolated annual weather stream from the historic monthly time series (1991–2010) at a constant CO2 atmospheric concentration of 389 ppm (2010 value) (Ameray et al., 2023a). We applied the obtained values for the historic climate scenario (1991–2010) for the starting year 0 (2010). For all climate change scenarios (baseline, RCP2.6, RCP4.5, and RCP8.5), we modeled *SEP*, *maxANPP*, and *maxAGB* for each 20-year time series of climate data for five periods (2010–2030, 2030–2050, 2050–2070, 2070–2090, and 2090–2110) and assumed these parameters remain constant after 2110 until 2210. For each species and land type, we ran ten replicate runs to isolate the stochastic effect of PnET-Succession on the *SEP*, *maxANPP*, and *maxAGB* outputs. Before starting the simulation at the landscape scale, we used the LANDIS-Site extension to verify model behavior at the stand level (1 cell) and the successional pathways emerging from these parameters (Figure A 4.3). We based this verification on Tremblay et al. (2018) and ran the model at the stand level for 1000 years using a 1-year time step and 20 repetitions. At the landscape scale in all MUs, the simulations were run for 200 years (2010–2210). *MaxAGB* and *MaxANPP* and *establishment* parameters derived from PnET were updated every 20 years to account for climate change until 2110. Beyond 2110, we assumed a constant climate until 2210.

We assessed and validated the growth parameters from PnET-Succession in our previous work in the same study areas, using yield curves and remote sensing (MODIS and Sentinel-2 images) (Ameray et al., 2023a). Natural regeneration for each site (grid cell) depended on neighboring species composition, seed dispersal distances, available light, and species’ shade tolerance. For a species to seed a site or resprout on a site, sufficient light must be available, determined by comparing the species’ shade tolerance with the shade at the site. ForCS iterates the number of time steps equal to the maximum cohort age for each site (cell) (Dymond et al. 2016). Beginning at time t-oldest cohort age, cohorts are added at each time step corresponding to the time when the existing cohorts were established. Thus, each cohort undergoes growth and mortality for the number of years equal to its current age, and its initial biomass value reflects competition among cohorts. For the initial DOM, ForCS implements the same approach as CBM-CFS3 (Kurz et al. 2009), and relies on the soil pool initialization process (spin up), where the model operates the abovementioned biomass spin up multiple times, each run assuming that a high-severity fire has killed all cohorts present at the end but then regrow exactly as before. This process is repeated until the slow soil pools have stabilized (Dymond et al., 2016), and we used the resulting values for our simulations at year 0 (2010).

4.2.2.4 Disturbances parameterization and simulated scenarios

The natural dynamics scenario (no-harvest) reflected forest succession under fire, windthrow, and spruce budworm (SBW) disturbances as the three major natural disturbances shaping the boreal forest in the study areas (Aakala et al., 2023). The extensions used for these disturbances were already calibrated and parameterized in our previous works (Ameray et al., 2023a, 2023b). Briefly, the Base Fire extension simulates fire regimes through stochastic fire events that depend on fire ignition, initiation, and spread by ecoregion, using the input data of ignition probability, map of fire regions, fire size (minimum, mean, and maximum), and fire severity (Scheller & Domingo, 2018). As the wildfire regime depends on climate, we calibrated the burn rate (% of land disturbed annually) per MRNF ecoregion (5a, 5d, 5g, 6a, 6h, 6i) for each climate change scenario, including the current baseline climate from the literature (Bergeron et al., 2006; Boulanger et al., 2014; Molina et al., 2021; Tremblay et al., 2018). Climate change is expected to increase the burn rate and fire return interval, particularly in MU1 and MU2 (Boulanger et al., 2014). The Base Wind extension was used to stochastically simulate windthrow disturbance on the basis of windthrow intensity, size, spread, severity, and rotation period (Scheller et al., 2018). Windthrow size and period per ecoregion were parameterized using the historical data from the forest inventory geodatabase (1970–2010). Similarly, the Biological Disturbance Agent (BDA) extension stochastically introduces periodic defoliation events parameterized solely by defoliation during SBW outbreaks (Sturtevant et al., 2019). SBW host species included, from most to least vulnerable, balsam fir, white spruce, and black spruce. The BDA extension parameters were calibrated and validated in other studies for similar landscapes in the Quebec boreal forest (Boulanger et al., 2017, 2019). We relied on a 400-year dendrochronological reconstruction of SBW outbreaks in southern Quebec to set an average of 32 years between outbreaks (Boulanger et al., 2012; Navarro et al., 2018). After the SBW event, we assumed that all biomass of the killed cohorts transferred to the DOM pool. This can produce an immediate increase in the DOM pool and a decrease in living biomass. Finally, salvage logging was not considered in this study.

We used the Biomass Harvest extension v4.4 (Gustafson et al., 2000) to simulate harvest disturbance. First, for all scenarios, annually harvested area—expressed as a percentage (Table A4.2)—was calibrated to match as close as possible to the allowable annual cut determined for the current planning cycle (2023–2028), given that the model’s target is area-based whereas the allowable annual cut is volume (biomass)-based. This model requires dividing the landscape into management areas, specifying the order in which stands are to be harvested. Stands were deemed eligible for harvesting on the basis of their exploitability age and the species’ economic importance. The Biomass Harvest extension modelled the various management scenarios described in Figure 4.3 and Table A4.3 (Scheller et al., 2019). We designed prescriptions on the basis of current silvicultural treatments and implemented them in varying proportions through different management scenarios (MRNF, 2010). These scenarios included clearcutting (100% CRI), CPRS (CRI fixed at 95% and the cohorts of 1–20 years being avoided), and three forms of PC having 25%, 50%, and 75% CRI (see Table A4.3 for more details). For all management scenarios, we applied stem-only harvest processing (also known as short-wood or cut-to-length logging in North America), assuming that only merchantable wood was transferred to the forest industry, whereas foliage, branches, and coarse and fine roots were left on site and transferred to DOM. Moreover, all PCs were based on commercial thinning from above (only cohorts older than the economic age of operability were removed). In the Quebec boreal forest, reforestation/replanting generally occurs after CC or CPRS in areas having a poor soil seed bank and a low regeneration rate; our prescriptions ensured that the historic replanted ratio of each species was respected, at 70% black spruce, 25% jack pine, 3% larch, and 2% white spruce. In addition, our proposed scenarios featured varying levels of annual replanted area, as outlined in Table A4.3.

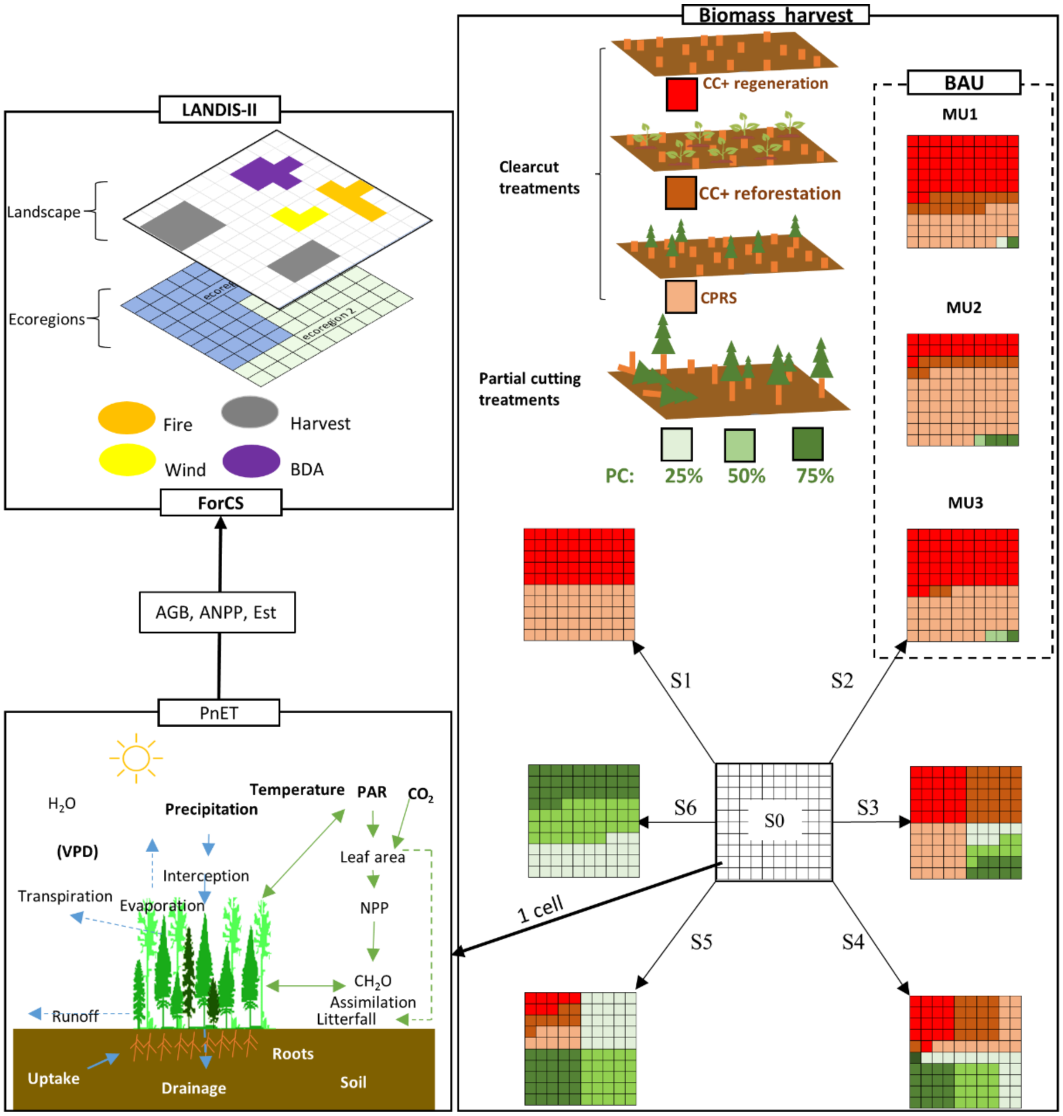

Figure 4.3 General methodology framework. The PnET-Succession model used for succession simulates simultaneously water and carbon cycles and integrates environmental factors such as soil texture, precipitation temperature radiation (PAR), and vapor pressure deficit (VPD). The model estimated the above net primary productivity (ANPP), aboveground biomass (AGB), and establishment (Est). These parameters were then integrated into ForCS and other disturbance extensions [harvest, winds, biological disturbance agent (BDA), fire] within the LANDIS-II model, and management scenarios (Biomass harvest) were tested at the landscape scale (see Table A4.3 for more details). S2 reflects the BAU scenario derived from the 1970–2010 historic harvested geodatabase from the Quebec forest inventory of each MU.

4.2.3 Data analysis

To account for the variability among simulations, we repeated each management scenario four times per climate change pathway, including the natural dynamics scenario (a total of 336 simulations in all MUs). This number of repetitions was sufficient to identify the stochastic effect of LANDIS-II on the outputs (Ameray et al., 2023a; Zhuo et al., 2020). We aimed to compare the BAU scenario in the selected MUs against other management scenarios (Figure 4.3). First, for the no-harvest (S0) natural dynamics scenario, we assessed and visualized carbon pools (biomass and DOM) and fluxes (NPP, R_h, NEP, NPB) per climate scenario (baseline, RCP2.6, RCP4.5, and RCP8.5). Second, to identify the effect of management selection under climate change scenarios, we visualized various forest carbon pools (biomass and DOM) and fluxes, including NPP, R_h, NEP, and NBP averages and their confidence intervals. To identify whether the effect was positive or negative, we calculated the magnitude of change for each variable (V: biomass, DOM, NPP, R_h, NEP, and NBP) as the difference (Δi,j) between the averages (entire period) of each management scenario (Vi,j) and S0 (Vi,0) for each climate projection (i) (Eq. 1). White et al. (2014) proposed that evaluating the magnitude of differences between simulations was a better approach than relying on statistical tests within the simulation models. We compared CC-based scenarios (S1, S2, and S3)—in which CC and CPRS were applied to more than 50% of the harvested area during the year—with PC-based scenarios (S4, S5, and S6) (Figure 4.3).

$$\Delta_{i,j} = V_{i,j} - V_{i,0} \text{ (Eq. 1)}$$

were *i* climate change scenario: baseline (*i*=1), RCP2.6 (*i*=2), RCP4.5(*i*=3), RCP8.5(*i*=4), and *j* management scenario from 1 to 6 (Figure 4.3).

4.3 Results

4.3.1 Carbon pools and fluxes under natural scenario

For all climate change scenarios involving the natural dynamics scenario (S0) (Figure 4.4), our simulations indicated an overall average increase in carbon stocks of biomass (AGB and BGB) at the landscape scale for all MUs, except for MU1 under RCP8.5. The baseline climate scenario had the total biomass carbon storage (AGB and BGB) in all MUs increase slightly but remain stable after 2100. Under the current baseline climate, both AGB and BGB increased from 27 tC ha^{-1} to 42 tC ha^{-1} in MU1, from 25 tC ha^{-1} to 41 tC ha^{-1} in MU2, and from 24 tC ha^{-1} to 26 tC ha^{-1} in MU3.

Relative to the current baseline climate, RCP2.6 produced an increase of 8%, 6%, and 25% in average biomass carbon storage (AGB and BGB) over the 200-year simulation for MU1, MU2, and MU3, respectively. RCP4.5 led to an increased average biomass carbon storage of 14%, 5%, and 41% in MU1, MU2, and MU3, respectively. In contrast, under RCP8.5 in MU1, biomass carbon storage dropped by 9%; it increased by 6% and 43% in MU2 and MU3, respectively. After 2090, the biomass carbon storage in MU1 decreased under RCP8.5 and eventually fell below that of the current baseline climate (42 tC ha^{-1}).

Simulations indicated that the carbon stocks in the DOM pools, which includes carbon in deadwood, litter, humus, and mineral soil, were higher than biomass carbon storage and that DOM variations had a significant impact on total ecosystem carbon stocks at the landscape scale (Figure 4.4). Relative to the current baseline climate, a persistent reduction in DOM stocks was observed for the DOM carbon pool in MU1 and MU2. In MU1, DOM stocks decreased on average −2%, −4%, and −15% for RCP2.6, RCP4.5, and RCP8.5, respectively, and by −3%, −11%, and −13% in MU2. For MU3, DOM carbon storage increased by 1%, 2%, and 4% under RCP2.6, RCP4.5, and

RCP8.5, respectively. Regardless of the climate change scenario applied to S0, DOM stocks continued to increase over time, relative to the observed values in 2010, until the end of the simulation (2210), where it reached more than 120 tC ha^{-1} in all MUs, except in MU1. In MU1, after 2090, DOM stocks dropped to under 100 tC ha^{-1} in RCP8.5 and then stabilized around this value until 2210 because of the increase in broadleaf abundance and productivity (Figure A4.4), and a slight stabilization in annual carbon turnover around 2.2 tC ha^{-1} yr^{-1}. This carbon turnover reflects the carbon transferred annually from biomass (above- and belowground) to the DOM and soil pools (Figure A4.5). Moreover, the periodic losses from AGB every 32 years because of SBW outbreaks can explain the increased DOM. Indeed, under baseline climate, AGB carbon storage was reduced by 2.5, 5, and 7 tC ha^{-1} during SBW outbreaks in MU1, MU2, and MU3, respectively (Figure A4.5). These reductions in AGB were similar for RCP2.6 in all MU, whereas they were less than the baseline conditions in MU1 and MU2 under RCP4.5 and RCP8.5. In MU3, the reduction from SBW outbreaks could reach more than 7.5 tC ha^{-1} yr^{-1} under all RCP scenarios.

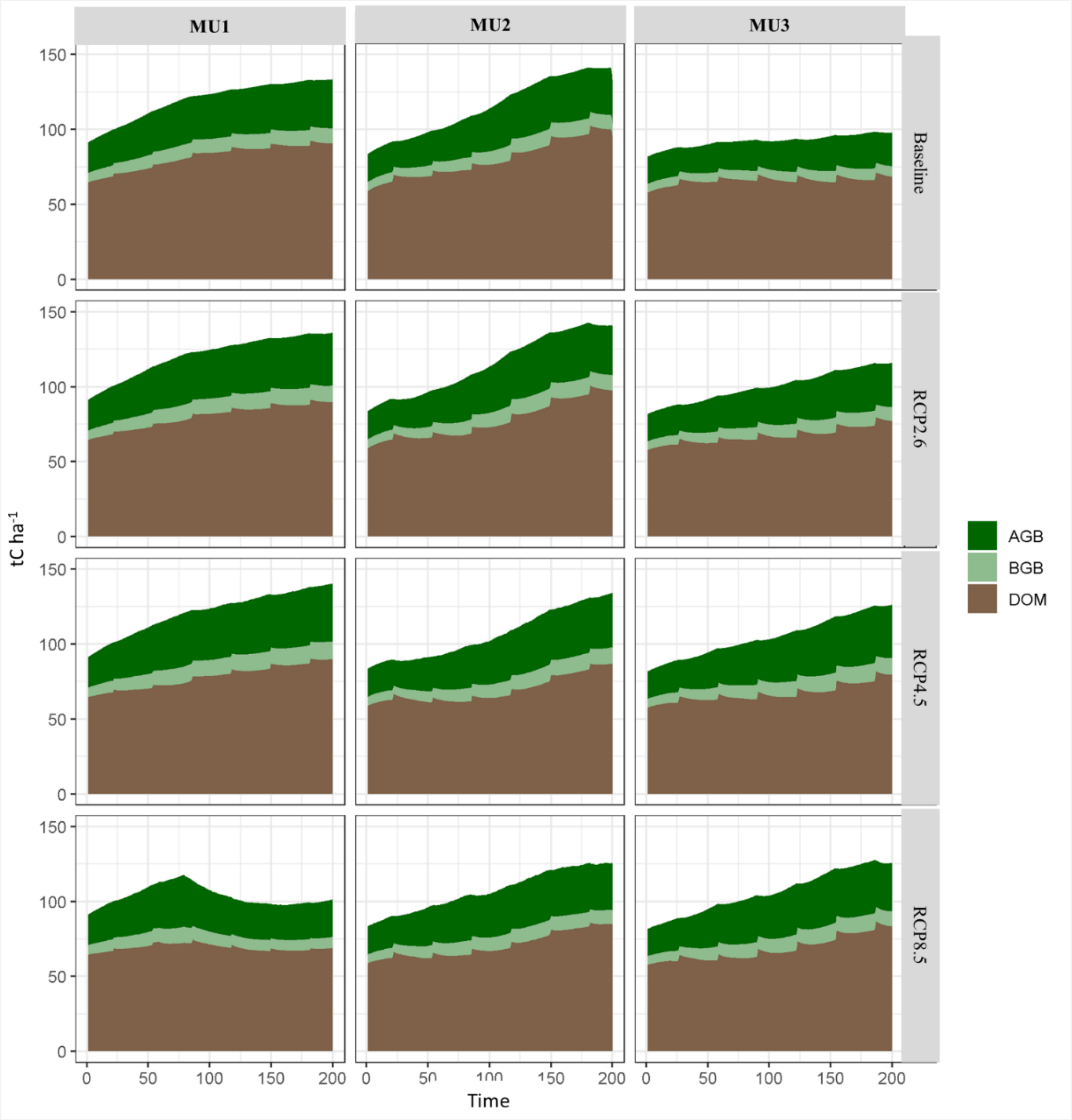

Figure 4.4 Average carbon stocks (tC ha^{-1}) under the natural dynamics scenario (S0) for aboveground biomass (AGB), belowground biomass (BGB), and dead organic matter (DOM) in three management units within the Quebec boreal forest (MU1, MU2, and MU3) under four climate change scenarios (baseline, RCP2.6, RCP4.5, and RCP8.5) from 2010 (year 0) to 2210 (year 200). The average carbon stocks were calculated from four runs. DOM includes deadwood, litter, humus, and mineral soil.

Under the current baseline climate and natural dynamics scenario (S0) scenarios, NPP is shown to increase in both MU1 and MU2, reaching around 150 and 200 gC m^{-2} yr^{-1}, respectively, whereas it remains constant at around 100 gC m^{-2} yr^{-1} in MU3. Over the following 80 years (2010−2090), forest NPP is projected to be greater than R_h, resulting in a positive NEP around 20 gC m^{-2} yr^{-1}. After 2090, NEP would become negative because of a high R_h. In both MU2 and MU3, R_h approached NPP because of a high mortality explained by the dominance of old-growth forests. We observed a decreasing pattern for NEP in MU1 under RCP8.5 around 2090, explained by a reduced NPP and increased R_h, (Figure 4.5). For the S0 scenario from 2090 to the end of the simulations, the NEP increase in MU1 could be explained by a greater broadleaf abundance (Figure A4.4, Figure 4.5).

SBW outbreaks appeared responsible for the temporary increase in R_h and decrease in NEP for S0 (Figure 4.5, Figure A4.6). For instance, in 2032, under the current climate scenario, the area disturbed by SBW was around 50,000 ha in MU1, and the NEP dropped from 25 to 0 gC m^{-2} yr^{-1}. In all MUs under all climate change scenarios, we observed a positive NBP at the beginning of the simulation until the first SBW outbreak (Figure 4.5). For all climate scenarios, NEP had positive values at the start of the simulations before the first SBW outbreak in MU2 and MU3, thus indicating an overall increase in the total ecosystem carbon. However, during the first outbreak in these two MUs, the biomass carbon storage decreased instantly, whereas DOM increased because of a higher turnover. In MU1, NEP remained positive until the third outbreak, indicating a considerable increase in total ecosystem carbon, after which both values began to attain negative values. In all MUs, the post-outbreak DOM carbon storage decreased/stabilized because of a high R_h (>NPP), leading to negative NEP, whereas biomass carbon storage began to recover.

The NBP was positive from 2010 to 2032 in MU1 and MU2 and from 2010 to 2037 for MU3; thus, these regions act as carbon sinks in the present and near future (Figure 4.5). After 2032, however, NBP fell below 0 in MU1 and MU2 because of high carbon emissions related to SBW-disturbed forest areas—we observe a comparable situation for MU3 after 2037. After each outbreak, the NBP increased with more fluctuations under RCP4.5 and RCP8.5 because of increased burned areas (Figure A4.7), especially in MU1 and MU2. Positive NBP values remained until the following SBW outbreak 32 years later (Figure 4.5).

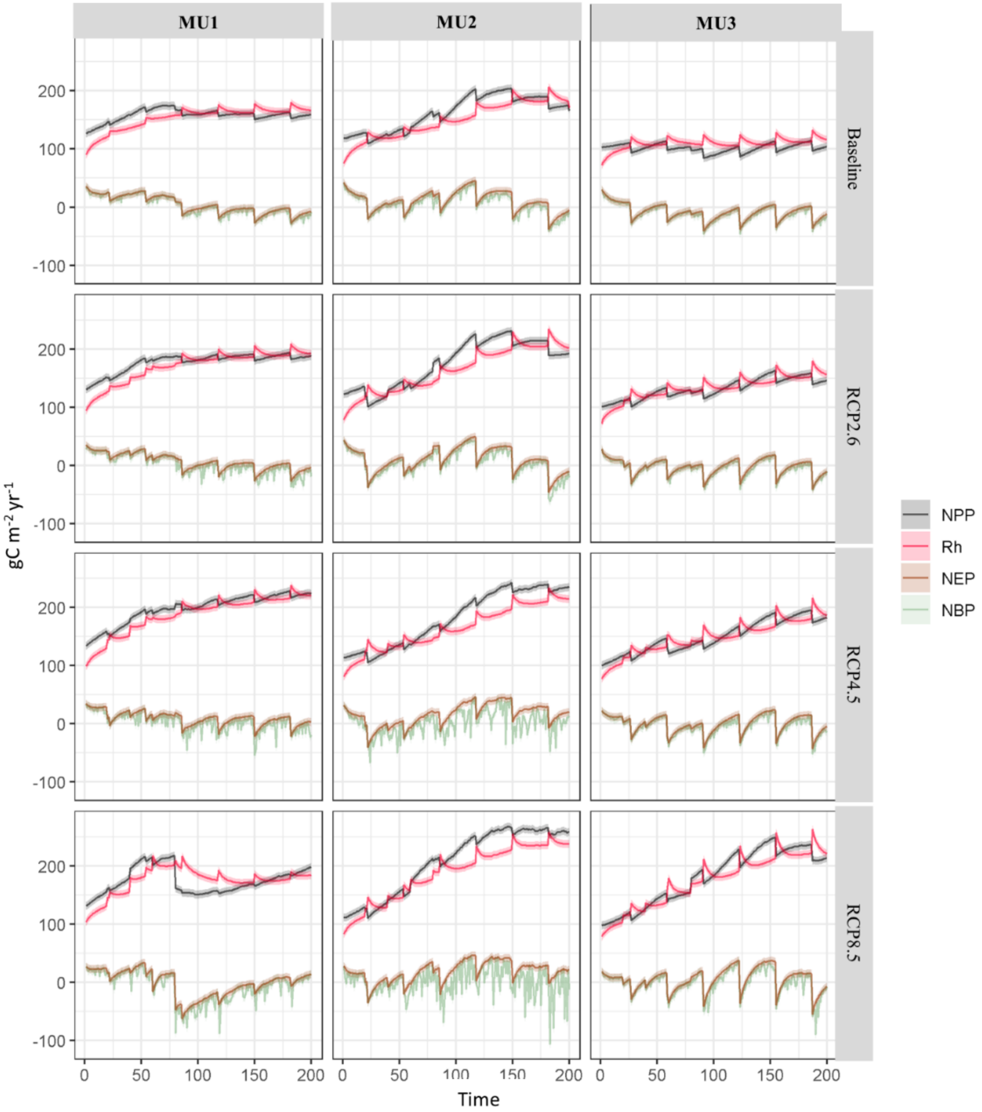

Figure 4.5 Average ecosystem carbon fluxes (gC m^{-2} yr^{-1}), including confidence intervals, for the S0 scenarios, including net primary production (NPP), heterotrophic respiration (R_h), net ecosystem productivity (NEP = NPP − R_h) and net biome productivity (NBP = NEP − losses due to disturbances) determined by four simulation runs for three management units in the Quebec boreal forest (MU1, MU2, and MU3) under the baseline, RCP2.6, RCP4.5, and RCP8.5 climate scenarios.

4.3.2 Managements effect on Carbon dynamics

4.3.2.1 Effect on Biomass and DOM pool

Our simulations at the landscape scale for all climate scenarios showed that total biomass and DOM carbon storage increased regardless of the applied management approach; however, the S0 scenario showed the largest increase (Figure 4.6). In all MU under all climate change scenarios (except MU1 under RCP8.5), PC-based scenarios (S4, S5, S6) reduced total biomass and DOM carbon storage by approximately 10% relative to S0, whereas these CC-based treatments (S1, S2, and S3) reduced DOM and carbon storage by approximately 25% (Figure 4.6, Figure 4.7). Under RCP8.5 in MU1, CC-based scenarios (S1, S2, and S3) began to store more biomass and DOM carbon than in S0 after 2100 (Figure 4.6) because of greater coniferous tree mortality and increasing broadleaf abundance (Figure A4.4 and Figure A4.8). Over the entire 200-year period, CC-based scenarios reduced total biomass and DOM carbon storage to nearly 10 tC ha^{-1} 1 in all MU under all climate scenarios, corresponding to a 25% reduction relative to S0 (Figure 4.7). In MU1 under RCP8.5, these scenarios decreased total biomass and DOM carbon storage by 5 tC ha^{-1} relative to S0 because of lower conifer productivity, especially between 2090 and 2210 (Figure 4.7; Figure A4.4). In contrast, this reduction was about 2–5 tC ha^{-1} for the PC-based scenarios relative to S0, reflecting a 10% reduction (Figure 4.7). Therefore, only PC-based strategies maintained long-term carbon storage in biomass and DOM.

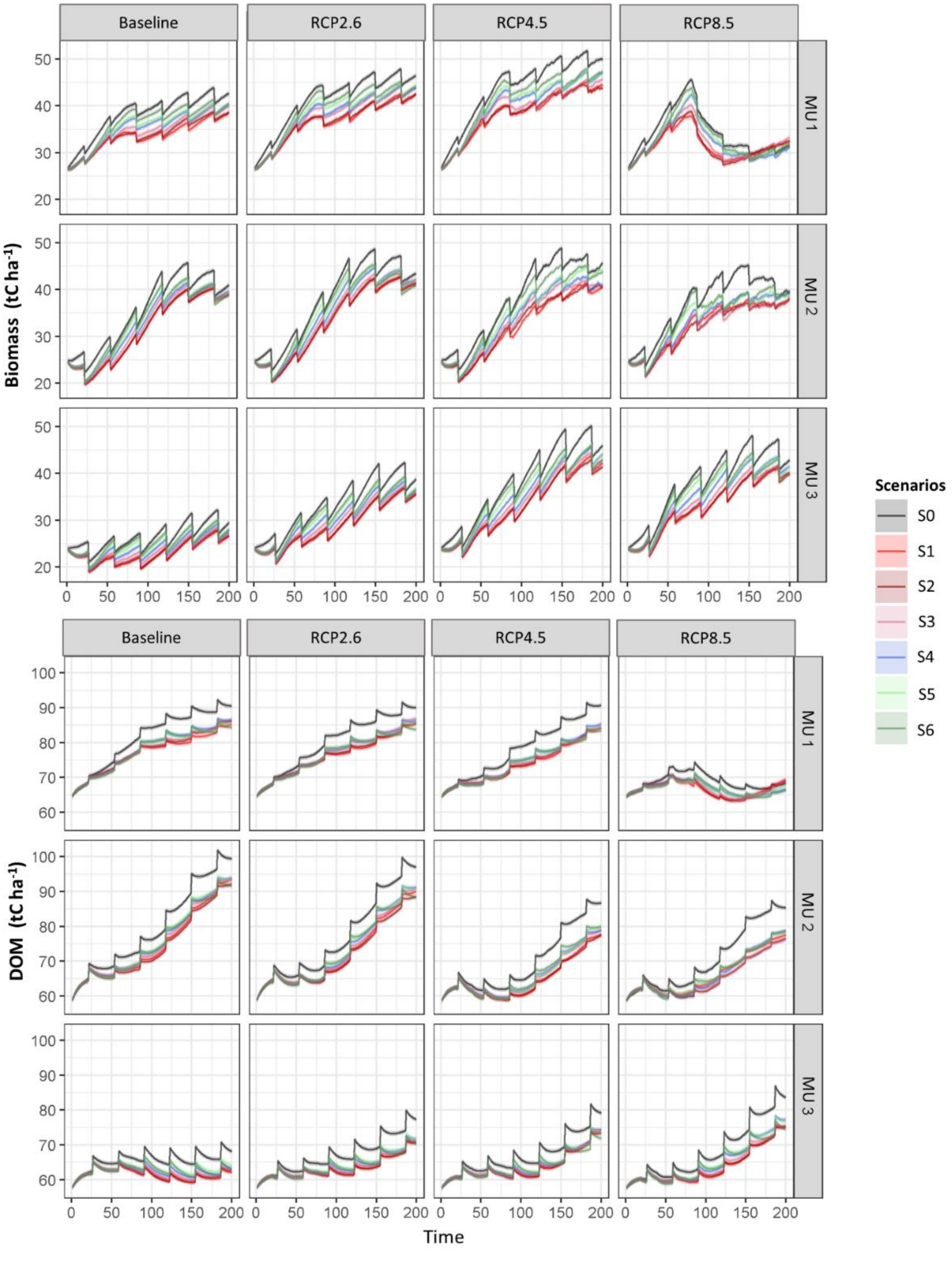

Figure 4.6 Average biomass and dead organic matter (DOM) carbon storage (tC ha^{-1}), including confidence intervals, calculated from four simulation runs per management scenario (S0, S1, S2, S3, S4, S5, and S6) and climate change scenario (baseline, RCP2.6, RCP4.5, and RCP8.5) for three management units of the Quebec boreal forest (MU1, MU2, and MU3) over 200 years (2010–2210).

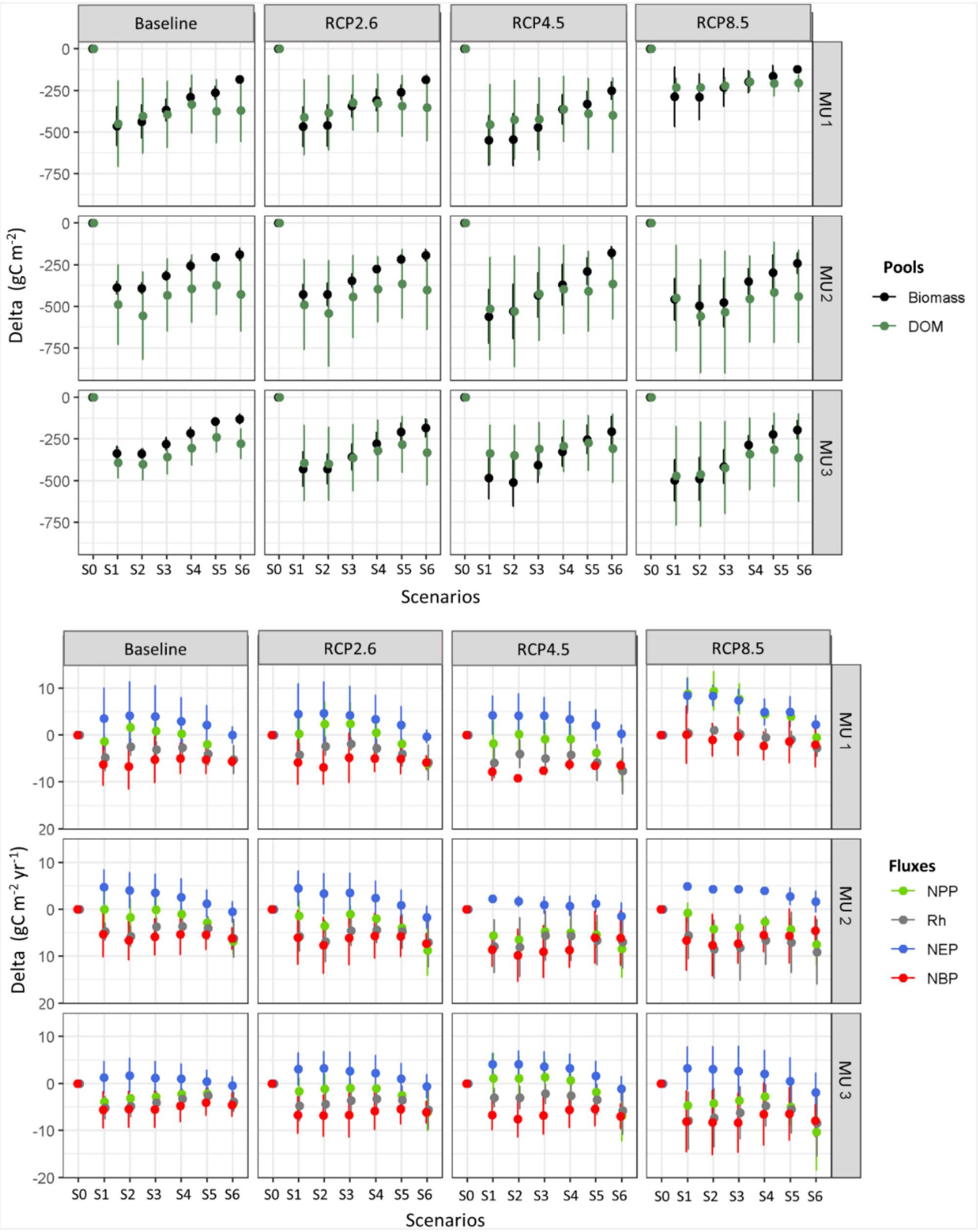

Figure 4.7 Average delta ($\Delta_{i,j}$), including confidence intervals, of management (S1, S2, S3, S4, S5, and S6) and climate scenarios (baseline, RCP2.6, RCP4.5, and RCP8.5) relative to the no-harvest scenario (S0) for three management units of the Quebec boreal forest (MU1, MU2, and MU3) for carbon pools (DOM) and biomass [aboveground biomass (AGB) + belowground biomass (BGB)] and fluxes (net primary production (NPP), heterotrophic respiration (R_h), net ecosystem productivity (NEP = NPP − R_h), and net biome productivity (NBP = NEP − losses because of disturbances) over the entire simulation period (2010−2210).

4.3.2.2 Effect on carbon fluxes

In all MUs under the scenarios of baseline climate, RCP2.6, and RCP4.5, carbon sequestration (NPP) and ecosystem R_h increased for all management scenarios (Figure 4.8). Under RCP8.5 in MU1, NPP dropped by 31% (from 220 gC m^{-2} $yr^{-1}$to 100 gC m^{-2} yr^{-1}) after 2091; however, NPP increased after 2100 under all scenarios and mainly for the CC-based scenarios because they increased the area covered by trembling aspen and white birch with a high percentage of young forest (Figure A4.8 and Figure A4.9). Indeed, relative to scenario S2 (BAU) in MU1 under baseline climate, scenarios S4, S5, and S6 saw in an expansion of mature and old-growth forests by over 10%, 20%, and 35%, respectively, whereas S2 maintained 50% young forest (Figure A4.9). In MU2 and MU3, NPP increased under all climate change and management scenarios, doubling its values under RCP4.5 and RCP8.5, particularly in MU3.

Ecosystem respiration, R_h, showed a similar NPP pattern and trend, and its values approached those of NPP for the S0 and PC-based scenarios. We observed a considerable difference in carbon fluxes of the S0 and PC-based (S4, S5, S6) scenarios against those of the CC-based scenarios (S1, S2, and S3). Moreover, some intermediate scenarios, such as S3 and S4, shared similar values of NPP and R_h. However, when assessing the differences between all management scenarios and S0 for the entire period (Figure 4.7), we note that CC-based scenarios had a more positive NEP (approx. 10 gC m^{-2} yr^{-1}) relative to PC-based scenarios (S4, S5, S6), having a NEP approaching 0. This difference related to the increased NPP under the S1 and S2 management scenarios, which was greater than R_h. On the other hand, when a greater proportion of PC was applied to a managed area, NPP approached R_h, i.e., as observed for the S6 management scenario (Figure 4.9). CC-based scenarios increased the expansion of pioneer species and the abundance of young forests (Figure A4.8 and Figure A4.9) relative to the PC-based scenarios (S4, S5, and S6), which could explain the differences in NEP. It is important to consider the impact of climate change on carbon fluxes, as it appears to have a greater effect than the chosen management approach. Depending on the given RCP climate scenario, NPP and R_h could potentially double relative to the current climate, particularly in MU2 and MU3 (Figure 4.8). However, it is worth noting that carbon pools are influenced by both climate change and management approach (as shown in Figure 4.7 and Figure A4.10).

NBP accumulation differed considerably among the management scenarios (including S0) and MUs (Figure 4.9). Under S0, the MU1 landscape yielded a positive cumulative carbon balance for all climate scenarios. This carbon accumulation reached 2000 gC m^{-2} yr^{-1} for the baseline and RCP2.6 scenarios. In all management scenarios, the cumulative carbon balance was less than that for S0. PC-based scenarios maintained a positive carbon accumulation of around 1000 gC m^{-2} yr^{-1} for up to 150 years in MU1 (Figure 4.9). While CC-based scenarios led to MU1 shifting from a carbon sink to a source after 100 years under all climate scenarios. This shift was particularly strong under RCP8.5, with the accumulative carbon balance dropping to −2000 gC m^{-2} yr^{-1}. In MU2, all management scenarios (including S0) produced a positive cumulative carbon balance of around 2000 gC m^{-2} yr^{-1} until the first SBW outbreak (2032), after which S0 showed no accumulation and the other scenarios showed negative values (Figure 4.9). Moreover, in MU2, PC-based scenarios showed a higher carbon accumulation than CC-based ones. Finally, for MU3 the accumulative carbon balance was negative after the second SBW outbreak; the losses were higher under the current climate than for the RCP scenarios because of lower productivity (Figure 4.5), and more extensive areas disturbed by SBW under RCP projections (Figure A4.6). In summary, S0 had the greatest accumulation across all MUs, followed by the PC-based scenarios and then the CC-based ones. CC-based scenarios export a significant amount of carbon to wood products, whereas the PC-based scenarios produce higher carbon emissions to the atmosphere because of windthrow and SBW outbreaks, and less carbon is transferred to wood products (Figure 4.10 and Figure A4.6). Also, The NBP losses under SBW were greater under the PC-based scenarios than the CC-based scenarios because PC retained more SBW host species, mainly black spruce and balsam fir, over the long term (Figure A4.6 and 4.10).

4.3.2.3 C transfers to wood products

At the landscape level, annual carbon transfer to wood products rates (*H*) for all management scenarios remained constant in MU2 and declined in MU1 and MU3 until around 2030, after which they increased again in all MUs (Figure 4.10). For all MUs, those rates were generally higher under the RCP scenarios than the current baseline scenario. As with other indicators, annual carbon transfer to wood products differed among all management scenarios across all MUs and climate change scenarios (Figure 4.10). The *H* values were lower in PC-based scenarios. For instance, in MU1 under the current baseline climate scenario, S1 and S2 transferred around 200 gC m^{-2} yr^{-1}, whereas S6 transferred less than 100 gC m^{-2} yr^{-1}. However, when comparing CC and PC management scenarios, we observed that PC-based scenarios harvested a larger area while transferring less carbon to harvested wood products (Figure 4.10).

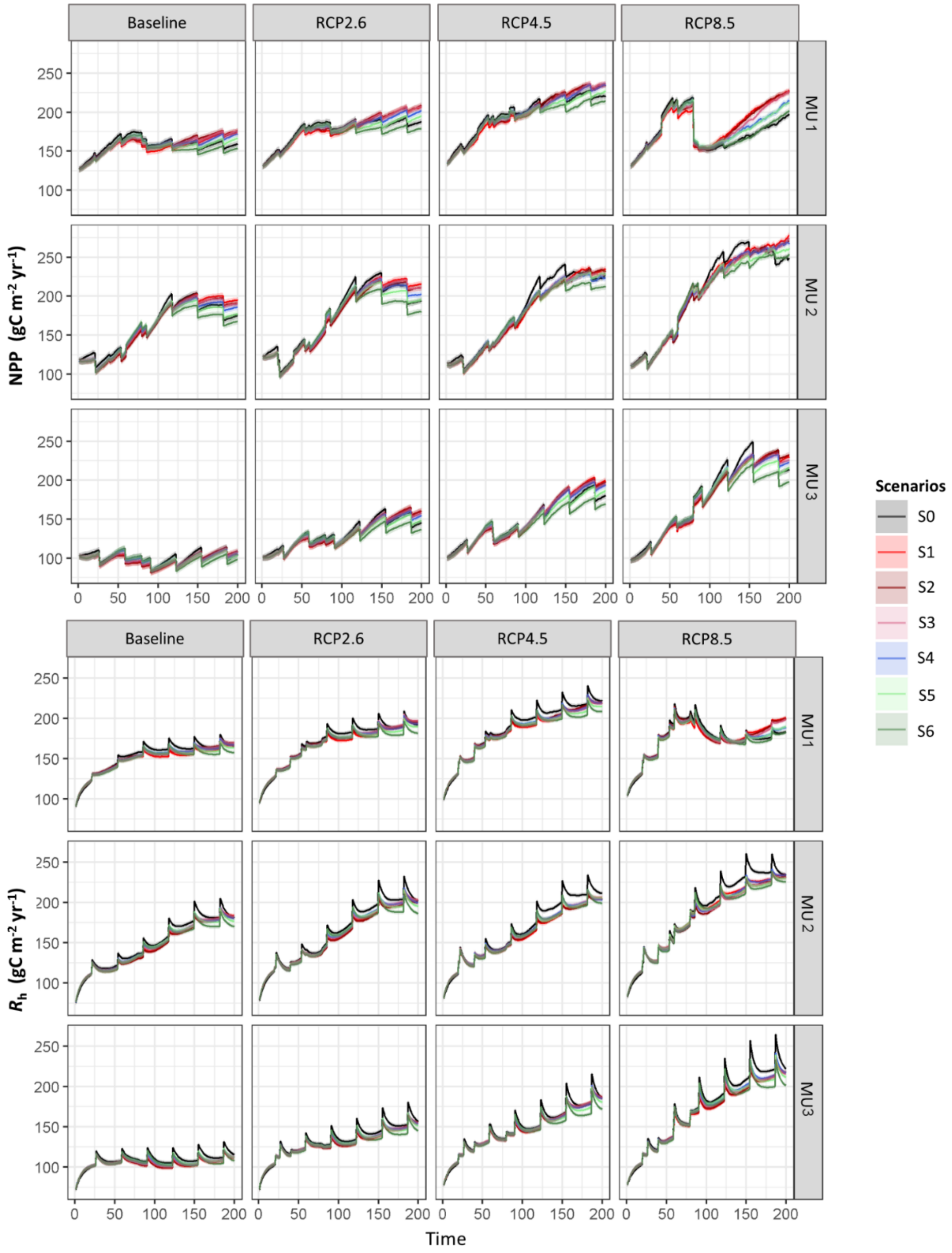

Figure 4.8 Average ecosystem carbon fluxes, including confidence intervals, for net primary production (NPP) and heterotrophic respiration (R_h) (gC m^{-2} yr^{-1}) per management scenario (S0, S1, S2, S3, S4, S5, and S6) and climate scenario (baseline, RCP2.6, RCP4.5, and RCP8.5) for three management units of the Quebec boreal forest (MU1, MU2, and MU3) over 200 years (2010–2210).

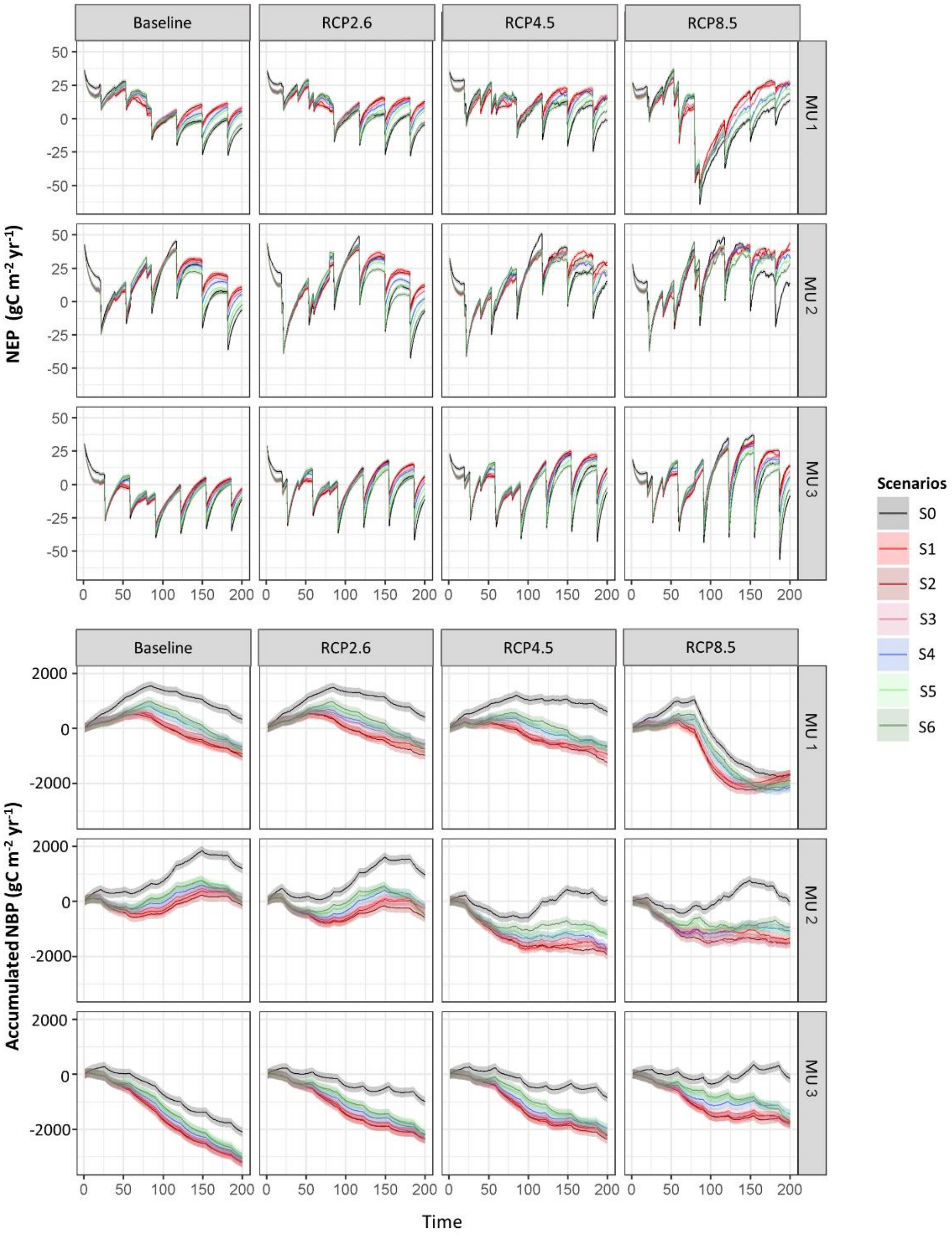

Figure 4.9 Average net ecosystem production (gC m^{-2} yr^{-1}) and the accumulated net biome production (NBP) (gC m^{-2} yr^{-1}), including confidence intervals, per management (S0, S1, S2, S3, S4, S5, and S6) and climate scenario (baseline, RCP2.6, RCP4.5, and RCP8.5) for three management units of the Quebec boreal forest (MU1, MU2, and MU3) over 200 years (2010–2210). NBP corresponds to NEP minus losses from the ecosystem because of disturbances (both emissions to the atmosphere from combustion and losses related to the forest products sector).

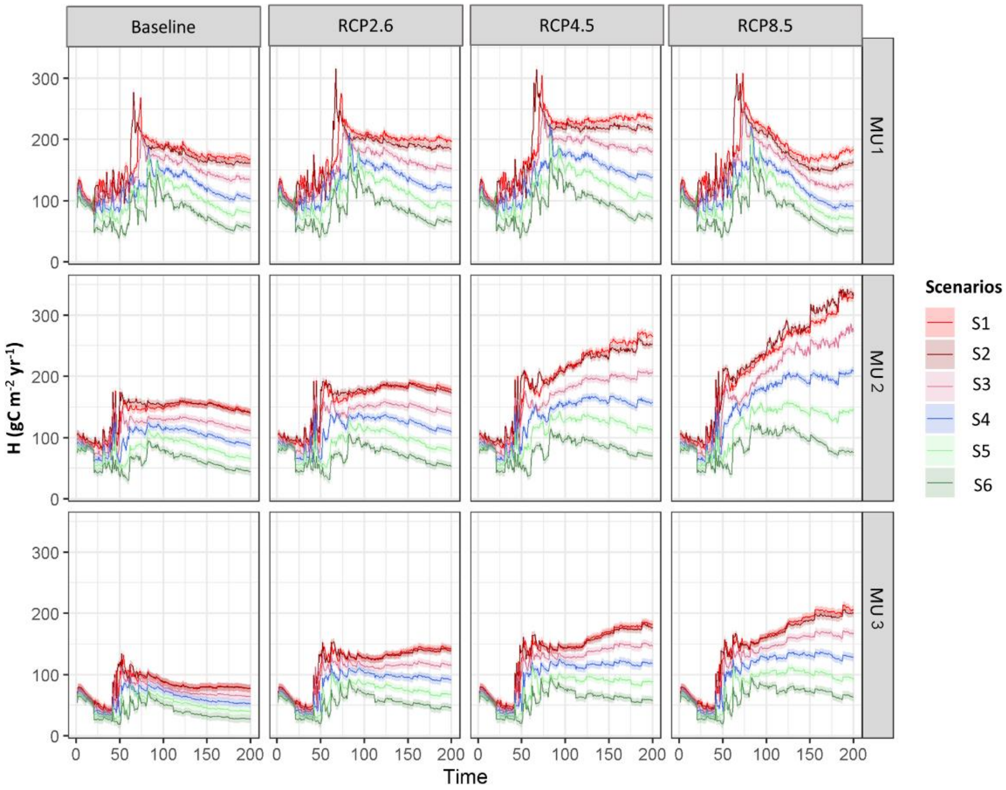

Figure 4.10 Average amount of carbon transferred to harvested wood products (*H*; gC m^{-2} yr^{-1}), including confidence intervals, per management (S0, S1, S2, S3, S4, S5, and S6) and climate change scenario (baseline, RCP2.6, RCP4.5, and RCP8.5) for three management units in the Quebec boreal forest (MU1, MU2, and MU3) over 200 years (2010–2210).

4.4 Discussion

4.4.1 Carbon dynamic under climate change and management effect

The boreal forests of Canada are currently experiencing climate change, including altered precipitation patterns, increased temperatures, and changes in the frequency and severity of climate-influenced natural disturbances, such as forest fires and insect outbreaks (SBW) (Achim et al., 2022; Ameray et al., 2023a; Boulanger et al., 2019; Molina et al., 2021; Tremblay et al., 2018; Wang et al., 2016). Our study aimed to improve our understanding of carbon dynamics in the Quebec boreal forest ecosystem near the northern limit of the commercial forest under various climate change and management scenarios. This region's forests are currently characterized by relatively low productivity and severe natural disturbances, mainly wildfires. We determined that future climate change, natural disturbances, and management will affect forest ecosystem carbon fluxes and stocks considerably, confirming previous research, i.e., (Ameray et al., 2023a; Boulanger et al., 2022; Molina et al., 2021). Our study further details how the direction and magnitude of the responses of carbon fluxes and stocks depend on the particular climate and management scenario.

Natural disturbances are anticipated to significantly affect future carbon dynamics and alter the size and location of the disturbance-affected areas in the eastern Canadian boreal forest. Our simulations and previous work within the same geographical region (Ameray et al., 2023b, 2023a), allow us to compare these disturbances on the basis of the disturbed area and resulting carbon losses, particularly from biomass:

1) Wildfires: We found that the average burned area per year remained similar across all management scenarios and was primarily influenced by climate change rather than the management approach (Figure A4.7). In MU2, for example, our simulations indicate an increase in the annual burned area from approximately 0.2% yr^{-1} under the current climate conditions to 0.6 % yr^{-1} under RCP8.5, and these rates are similar regardless of the applied management approach. Previous research had projected an increase in burn rates to 1 % yr^{-1}, 1.3 % yr^{-1}, and 0.38 % yr^{-1} by 2100 in MU1, MU2, and MU3, respectively, under RCP8.5 (Ameray et al., 2023a; Boulanger et al., 2014). Fire has a considerable impact in both MU1 and MU2, particularly on biomass carbon storage, because of the shorter fire cycles in these two MUs; biomass losses from broadleaf and coniferous trees vary between 5 and 10 tC ha^{-1} yr^{-1}, respectively

(Ameray et al., 2023b). In addition to the loss of carbon storage, fire also reduces NEP because of an immediate post-fire decrease in NPP and an increase in R_h.

2) SBW outbreaks: Our simulations under current climate conditions projected increases in the SBW total-affected area of 0.1, 0.3, and 0.35 Mha during outbreaks by 2100 in MU1, MU2, and MU3, respectively (Figure A4.6). Relative to current climate scenarios, RCP2.6 projected similar area sizes being affected by SBW. However, RCP4.5 and RCP8.5 led to smaller SBW-disturbed areas in all MU. This decrease in extent may relate to the increased abundance of broadleaf trees under RCP4.5 and RCP8.5 and, thus, fewer available coniferous host species. Nonetheless, climate change is a major driver of increased defoliation severity observed during recent outbreaks at the stand scale (Subedi et al., 2023). When SBW outbreaks occur, biomass carbon storage losses are estimated at 4, 5, and 7 tC ha^{-1} yr^{-1} in MU1, MU2, and MU3, respectively (Ameray et al., 2023b).

3) Windthrow: This disturbance had less of an effect than fire and SBW. For instance, under the current climate, the average coniferous carbon biomass losses were 0.03, 0.12, and 0.13 tC ha^{-1} yr^{-1} in MU1, MU2, and MU3, respectively (Ameray et al., 2023b). In regard to the effects of forest management on disturbance regimes, PC-based scenarios may increase the amount of SBW- and wind-affected areas (Figure A4.6). These scenarios maintain a more vulnerable age class of tree to both winds and SBW and promote tree species that act as hosts for SBW (Ameray et al., 2023b).

We observed that future climate change is projected to increase forest productivity (as estimated by NPP and biomass) in the Quebec boreal forest. These results are similar to those obtained previously in the same management units using the PnET-Succession extension for LANDIS-II (Ameray et al., 2023a, 2023b). Furthermore, D’Orangeville et al. (2018) found that a 2 °C increase would cause an increase in forest productivity of up to 13%. This increased productivity arises from warmer conditions extending the vegetative season and reducing the potential for cold-temperature injuries, thereby promoting greater tree growth (Ameray et al., 2023a). However, warmer conditions also create more favorable fire conditions, increasing disturbance severity (Pau et al., 2023). Moreover, relying on dendrochronological data, Marchand et al. (2021) reported lower productivity for black spruce under drought conditions. Under the current climate scenario, MU1 is dominated by young forest (40–60 years old, Figure 4.1), which could explain the increased NPP, whereas in MU2, the age structure is dominated by old forest (>100 years, Figure 4.1). This increased NPP for MU1 could be related to a greater abundance of trembling aspen and white birch (Figure A4.4). On the other hand, MU3 is characterized by old-growth forest age structures (Figure 4.1) with a lower abundance of trembling aspen and white birch (Figure A4.4). This old-growth species assemblage could explain the stable forest NPP under the current climate. Other studies using CBM-CFS3 and LANDIS-II found that productivity in the Quebec boreal forest will benefit from increased temperatures in northern zones (including our study area), whereas lower coniferous productivity is expected in the more southern regions and transition zones (boreal–temperate forest) (Boulanger et al., 2017; Boulanger & Puigdevall, 2021; Landry et al., 2021; Moreau et al., 2022; Taylor et al., 2008). We observed an exception to this pattern in the westernmost region (MU1) under RCP8.5, where productivity will be reduced after 2090; similar results have also been reported (Ameray et al., 2023a; Molina et al., 2021). Moreover, the increased NPP in MU2 and MU3 and NPP recovery in MU1 after 2090 appear to be related to a greater abundance of broadleaf trees in these areas.

In regard to the biomass pool, Duchesne et al. (2016) estimated that the AGB carbon storage in the Quebec boreal forest ranged between 10 and 30 tC ha^{-1} on the basis of the third forest inventory (1990–2002), whereas we estimated an AGB carbon storage of approximately 20 tC ha^{-1} at the beginning of simulation (2010) under current baseline climate (1991–2010 historical averages) because of the lower productivity in the study area. Boudreau et al. (2008) combined data from airborne and spaceborne LiDAR, Landsat ETM, and land cover maps to estimate AGB carbon stocks between

10 and 30 tC ha^{-1} (in 2008) in the Quebec boreal forest. We estimated an initial DOM carbon storage of 65, 60, and 58 tC ha^{-1} in MU1, MU2, and MU3, respectively, estimates within the lower range of those of Paré et al. (2011) (2008 estimate: 61–131 tC ha^{-1}) in the same study region (i.e., spruce–feathermoss bioclimatic region). Our model assumed that a high-severity fire kills all the oldest cohorts present. Considering that some stands (cells) are potentially older than the age of the oldest cohort, this could partly explain why DOM initial values are at the lower limit of the data range reported by Paré et al. (2011).

The species' AGB can explain DOM carbon storage dynamics over time. Indeed, under the current baseline climate and natural dynamics scenario, all the studied species increase markedly in AGB over the landscape, particularly in MU2 (Figure A4.4). Empirical data (Laganière et al., 2015) show that DOM carbon storage is higher under coniferous cover than under broadleaf trees, whereas broadleaf DOM decay rates are higher than for coniferous trees (Hüblová & Frouz, 2021). In our simulations, we assumed that both groups shared a similar decay rate, which is a limitation of our work. In our scenarios, we classified the decay rate of all species among ten compartments defined by Kurz et al. (2009) (1: Very fast aboveground, 2: Very fast belowground, 3: Fast aboveground, 4: Fast belowground, 5: Medium, 6: Slow aboveground, 7: Slow belowground, 8: Snag stem, 9: Snag other, 10: Extra). Our assumption may therefore underestimate the actual difference in decomposition rates between coniferous and broadleaf stands. Consequently, the R_h estimated for CC-based scenarios, which increased with a greater abundance of broadleaf trees, may be underestimated. Total AGB increased continually under RCP2.6 and RCP4.5 in all MU, except for MU1 under RCP8.5, which saw the largest reductions of coniferous AGB after 2090. Relative to the initial conditions in MU1, the total AGB of black spruce under RCP8.5 decreased 63% by the end of the simulation, whereas trembling aspen and white birch stands increased 263% and 241%, respectively. Boulanger et al. (2017) and Molina et al. (2021) found that the AGB for black spruce significantly declined in this region, regardless of the applied RCP scenario. In contrast, coniferous species in MU2 and MU3 maintained or increased their productivity under all future climate scenarios, whereas broadleaf species doubled (or more) their AGB relative to the initial (2010) values (Figure A4.4), which could explain the observed increase in the DOM carbon pool.

Our simulations were designed to forecast the effect of including PC-based scenarios on carbon dynamics and to compare the effects against CC- and CPRS-based ones. At the stand scale, high CRI (CC, CPRS) induced net emissions to the atmosphere in the years following harvest because emissions from decomposition were substantially higher than sequestration through photosynthesis (Paradis et al., 2019). In contrast, PC-based continuous-cover forestry was associated with lower NPP, as it maintains tree cover but accrues less carbon debt and provides a more constant turnover of organic matter. Combining these processes can explain the increased DOM under PC-based scenarios (Paradis et al., 2019).

Previous stand-scale carbon flux studies have shown that PC-based scenarios maintain a greater long-term carbon sequestration than CC-based scenarios with a high NBP (Ameray et al., 2021; Goulden et al., 2011; Liu et al., 2011; Paradis et al., 2019). However, these studies did not consider forest dynamics at the landscape scale and stand replacement by other species; these newly arrived species may be more productive than the original trees. At the landscape scale, our CC-based scenarios accelerated the expansion of trembling aspen and white birch (and young forest) under all RCP scenarios, thereby increasing forest NPP and NEP above those of PC-based scenarios. CC-based scenarios may mitigate the impacts of SBW and wind activity by decreasing the area of SBW host species and targeting the oldest cohorts, which are more vulnerable to wind disturbance. The NPP under the CC-based scenarios counterbalances the carbon losses under R_h over time, as NPP and biomass accumulation increase in the youngest stands to produce an extended period of carbon accumulation and a positive NEP (Ameray et al., 2021). On the other hand, PC-based scenarios (S4, S5, and S6) expanded the abundance of oldest stands, which increased deadwood input to DOM and favored greater decomposition. This improves stocks in the landscape but causes NEP to approach zero (Goulden et al., 2011).

Our study shows that the net biome productivity (NBP) at the landscape scale under all management scenarios was lower than for the natural dynamics scenario. Moreover, the CC-based scenarios occasionally had lower NBP than that of PC-based ones, particularly for RCP4.5 and RCP8.5, although the differences are relatively minor. The lower NEP and the higher area disturbed by winds and SBW could explain the reduced NBP for PC-based scenarios. For CC-based scenarios, the lower NBP relates to the carbon transferred to harvested wood products. PC-based scenarios offer the benefit of increased overall carbon storage within the ecosystem all the while preserving the diversity in age structure and the abundance of old-growth forests and current species (e.g., black spruce) (Table 4.2) (Ameray et al., 2023b). In contrast, CC-based scenarios produce a positive NEP because of the abundance of broadleaf trees and young forests, leading to a greater harvest volume than that of PC-based scenarios. An increased harvest volume may result in greater carbon storage beyond the ecosystem boundary under CC-based scenarios, depending on the life cycle of the various wood product types (Moreau et al., 2022). Nevertheless, the drawback of CC-based scenarios lies in the loss of old-growth forests and the associated ecosystem services (Table 4.2), such as biodiversity conservation (Ameray et al., 2021; Martin et al., 2022).

Table 4.2 A summary of advantages and disadvantages of conservation, partial cutting (PC)–based scenarios, and clearcutting (CC)-based scenarios in terms of age structure, composition, carbon sequestration and carbon storage, and disturbance (+++: high, ++: medium, +: low) under current climate and the additional effect of future climate scenarios (RCP): positive (+ increase) or negative (- decrease) or minimal change (~) relative to the baseline conditions.

Component		**Strategy under current climate**			**Additional effect of climate change**		
		Conservation	**PC-based scenarios**	**CC-based scenarios**	**RCP2.6**	**RCP4.5**	**RCP8.5**
Age structure abundance	**Old**	+++	++	+	~	-	-
	Mature	++	+++	+	~	~	-
	Young	+	++	+++	~	+	+
Composition abundance	**Broadleaf**	+	++	+++	+	+	+
	Coniferous	+++	++	+	+	+	-
Carbon sequestration and storage	**NEP**	+	++	+++	+	+	+
	Biomass	+++	++	+	+	+	+ except in MU1
	DOM	+++	++	+	+	+	+ except in MU1
	HWP	No-harvest	++	+++	+	+	+ except in MU1
Disturbances	**Fire**	+++	+++	+++	+	+	+
	SBW	+++	++	+	~	~	-
	Winds	+++	++	+	~	-	-

Climate-induced increases in forest productivity, thus affecting the total amount of carbon transferred to harvested wood products, have also been modeled by Dymond et al. (2020) in northwestern Canada and by Ameray et al. (2023b) in eastern Canadian boreal zones. Ameray et al. (2023b) reported that climate change and CC-based scenarios would markedly affect softwoods and cold-tolerant species, particularly black spruce, and result in their decline over the next 100 years. A greater proportion of hardwoods in harvested volumes under climate change has also been observed in other studies (e.g., (Ameray et al., 2023b; Boulanger et al., 2023; McKenney, 2016)). By 2100, strong climate forcing could alter boreal coniferous species growth (spruce, pine, and balsam fir) in Quebec, particularly in the southern part of the forest; thermophilous species, species associated with relatively warmer conditions (trembling aspen and white birch), are expected to experience increased potential productivity (Ameray et al., 2023a; Boulanger et al., 2023), barring any soil limitations.

4.4.2 Management implications

Global change has and will continue to alter boreal forest dynamics and all ecological processes (Ameray et al., 2023a; Bergeron et al., 2006; Boulanger et al., 2022, 2023; Girona et al., 2023; Gustafson et al., 2020, 2023). Consequently, there is a need to implement forest management strategies that consider multiple goals, including carbon sequestration and storage, maintaining old-growth forest cover for ecosystem services, and wood supply for industrial needs (Ameray et al., 2023a; Boulanger et al., 2023; Gauthier et al., 2023; Girona et al., 2023). In this context, our study assesses carbon sequestration and storage capacity, forest composition, and forest age structure under multiple management and climate change scenarios. The main purpose is to help attain these multiple goals at a regional scale. Our study confirmed that the natural dynamics scenario, which corresponds to conservation and intact forest, stores more carbon in DOM and biomass in the various future climate scenarios; however, the NEP of these scenarios is null because of natural disturbances, mainly wildfires and SBW outbreaks. Under RCP4.5 and RCP8.5, the NBP could be positive because of an increased abundance of young and broadleaf forests, especially in MU2 and MU3. In our model, fire size and intensity were updated for each climate change scenario and are projected to increase under global change (Boulanger et al., 2014). Bergeron et al. (2002) stated that a shorter fire return interval may increase young forests and reduce the abundance and extent of mature and old-growth forests (Bergeron et al., 2001); this change was indeed observed under RCP8.5, particularly in MU2 (Figure A4.9).

Under CC-based scenarios (S1, S2, and S3), we observed an increased carbon sequestration capacity (NPP, NEP) and reduced total ecosystem carbon storage although with some carbon transferred to harvested wood products. Perez-Garcia et al. (2007) found similar results when they modeled the carbon balance of stands in the West Cascade region of the Pacific Northwest (USA). They concluded that a shorter clearcut rotation (45 years, which is much shorter than that required in the eastern boreal forest) would provide the greatest reduction of atmospheric CO_2 because CC-based scenarios increase the abundance of young forest and replace current coniferous species with more productive broadleaf species (Ameray et al., 2023b, 2023a). Moreover, CC-based scenarios can indirectly contribute to post-fire regeneration failure by increasing the number of young and vulnerable stands across the landscape (Cyr et al., 2021). Likewise, Boulanger et al. (2023) also found that maintaining current harvesting strategies (S2) or using the other CC-based scenarios (S1, S3) could exacerbate the negative impacts on forest habitats (i.e., composition, abundance, and age structure) and positive impacts on productivity, except for the westernmost unit (MU1) under RCP8.5. Other studies (e.g., (Ameray et al., 2023b; Stralberg et al., 2015; Tremblay et al., 2018)) have also found that using CC-based scenarios could reduce existing habitat for fauna conservation. For instance, Stralberg et al. (2015) and

Tremblay et al. (2018), both using species distribution models, reported that mature forests might suffer dramatic reductions in suitable habitat for bird conservation (e.g., woodpeckers) over the next century under intensive management and extreme climate change. Furthermore, small stem size is often discarded during CC treatments, whereas PC treatments have the advantage of producing larger stems, which can increase carbon sequestration and substitution benefits in longer-lived wood products, which might otherwise not be as obtainable in the slow-growing northern reaches of the boreal forests (Auty et al., 2014; Paradis et al., 2019).

On the other hand, extreme PC-based scenarios (i.e., harvesting 100% of annual managed area by PC), could emulate the natural dynamics scenario by maintaining similar natural vegetation patterns and age structure, which has also been observed in previous simulations based on PnET-Succession (Ameray et al., 2023b). However, although this strategy has the highest total ecosystem carbon storage, it also has the lowest global net ecosystem productivity and may not fulfill industrial needs (Ameray et al., 2023b). From our discussion above, it appears a compromise is needed between PC and CC treatments (S4: 50%PC/50%CC or S5: 75%PC/25%CC) approaches at the landscape scale to achieve the multiple goals and sustainable management aims under global change. Bergeron et al. (2002) and Ameray et al. (2021) suggest that the diversification of these strategies at the landscape scale, designed to maintain a spectrum of forest composition and structure at different scales, could be the best direction for increasing forest carbon sequestration capacity, rather than relying on a single strategy. This approach was also suggested by Seymour and Hunter (1992) and is denoted as “functional zoning” (TRIAD approach). It has been used in parts of boreal Canada as a potential strategy for achieving diverse objectives by dividing the landscape into three zones: conservation, intensive management, and extensive management (Ameray et al., 2021). In this context, our study proposed scenarios S4 (50% PC treatments / 50% CC treatments) and S5 (75% PC treatments / 25% CC treatments) as a means of attaining multiple goals (conservation, carbon sequestration and storage, industrial needs). Given the reforestation level considered in our study per scenario, the application of S4 and S5 requires increasing the coniferous reforestation rate in eastern regions to 8% and 17% of the annually harvested area; in MU1 and MU2, the suitable rate is already applied (Table A4.3). However, to maintain the current allowable annual cut, these proposed strategies (S4 and S5) require harvesting more area than CC-based scenarios and require improved road infrastructure, which could be a limiting factor and represent a considerable investment (Cyr et al., 2021).

In addition to adapting the TRIAD approach at the landscape scale as a suitable strategy, the wood industry sector of Quebec is confronting considerable challenges linked to climate change, requiring a proactive preparation to anticipate shifts in forest composition (Ameray et al., 2023a; 2023b). The transition towards innovative products, focusing on broadleaf species rather than conifers, emerges as a crucial strategy. This transition towards innovative products not only addresses the challenges of climate change but also seizes economic opportunities, ensuring the continuity of industrial operations and contributing to the preservation of forest resources.

4.4.3 Models improvements and limitations

We determined that ForCS, an extension of LANDIS-II, can produce reliable and valuable projections of future carbon dynamics. It is important to note that the accuracy and precision (or reliability) of ForCS depends strongly on the input parameters of growth and establishment probability, to which our results were highly sensitive (Dymond et al., 2016; Hof et al., 2017). We calibrated our inputs using Pothier and Savard's growth and yield curves (Pothier & Savard, 1998), which are based on a 140-year chronosequence. These parameters were updated every 20 years until 2110, and then we assumed them to remain constant until the end of the simulation for the model to stabilize. In ForCS, the growth and reproduction generally followed the Biomass Succession (v5.7) extension and the methods outlined in Scheller and Mladenoff (2004), whereas the modeling of DOM and decay generally followed the methods outlined in Kurz et al. (2009) using the CBM-CFS3 model. It must be noted, however, that CBM-CFS3, upon which the Forest Carbon Succession extension is based, does not always adequately model the evolution of DOM, especially in mineral soils (Sage et al., 2019; Shaw et al., 2008). These DOM-related issues may be arguably worse when run with future climate scenarios because DOM is based on the CBM-CFS3 empirical model and not a process-based model, which also integrates factors other than temperature (e.g., precipitation and soil texture). A great improvement would be if the input parameters (AGB, ANPP, SEP) were associated with climate input parameters as for PnET-Succession model (De Bruijn et al., 2014). Accordingly, integrating CBM-CFS3 model DOM equations used in ForCS with PnET-Succession could be particularly useful for ForCS rather than following Biomass Succession extension.

Our study did not consider carbon storage in paludified areas (forests turning into peatlands), particularly in MU1, which may underestimate our observed soil carbon in this MU. Several studies show that an additional 10–150 Mg·ha−1 is stored at 15–45 cm depth in boreal forest soils (Deluca & Boisvenue, 2012; Lavoie et al., 2005), despite these sites being characterized by lower productivity (Simard et al., 2007). Moreover, there are additional processes that the model does not consider but can affect the storage of carbon in the DOM. These processes include the amount and chemical composition of litter, the input of organic carbon, and the regulation of both microclimate and soil microbial activity, which could increase in importance in a warmer climate (Feng et al., 2006).

Broadleaf species encounter numerous obstacles to regenerating and establishing in many sites within the study area (e.g., organic soils and soil nutrient deficiencies).

Moreover, we did not consider regeneration failure because of increased fire activity, although this has been identified as the most important driver of forest productivity at the landscape level in fire-prone landscapes (Augustin et al., 2022; Cyr et al., 2021) In the same study area, PnET-Succession revealed a higher abundance of pure broadleaf forest than that found in ForCS (Ameray et al., 2023b). The observed difference could be attributed to the distinction that PnET considers competition for water and light, whereas ForCS succession addresses competition for light exclusively. Additionally, one of the major limitations of our study is the SBW model, which causes immediate carbon losses and reduces NEP to less than zero. In reality, the negative impacts of SBW are not always immediate and can persist for more than a decade (Boulanger et al., 2012). One aspect that has been neglected in our analysis concerns salvage harvesting following disturbance. Indeed, recovery of disturbed biomass, especially after TBE, could mitigate the high R_h observed in our study for S4, S5 and S6 scenarios.

PC-based scenarios were proposed as appropriate solutions to increase/stabilize the total ecosystem carbon storage and maintain the current habitat for biodiversity (Ameray et al., 2023b; Boulanger et al., 2023; Eyvindson et al., 2021); however, their success requires considering not only their intensity but also their type, e.g., shelterwood cutting, selection (distant or close), retention systems, and seed-tree systems (Ameray et al., 2021). Some forms of PC could increase post-cutting mortality incurred by disturbance, particularly windthrow (Girona et al., 2019), thereby affecting NBP. Several studies have estimated that post-cutting mortality following PC ranges from 15% to 74%, this mortality being attributed to windthrow (Girona et al., 2019; Wallentin & Nilsson, 2013). Girona et al. (2019) showed that experimental uniform shelterwood cuttings with 50% harvest intensity reduced the tree loss proportion relative to the seed-tree system currently used in Quebec's boreal forest. However, our study did not consider the interactions between disturbance and harvest during the simulation, even though windthrow after silvicultural intervention is one of the major issues with partial harvesting (Girona et al., 2023). Nonetheless, harvesting alters the stand age structure and composition, indirectly affecting areas disturbed by windthrow and SBW (Lavoie et al., 2021). Our study is a proof-of-concept evaluation of a forest landscape model’s capability to conduct simulations. Despite the limitations discussed above, this model can reduce uncertainty about the effectiveness of climate-adaptive silvicultural practices. The results obtained here will prove valuable for managers when making well-informed decisions related to sustainable forest management under global change.

4.5 Conclusions

Our results suggest that climate change may increase future forest net primary productivity, heterotrophic respiration, and carbon emissions (mainly because of wildfires and SBW) in the boreal forests of Quebec. These changes occur in large part because over the next 100 years, climate change and clearcutting will increase the abundance of young forest and pioneer species and thus heighten carbon sequestration yet also carbon emissions from decomposition. We identified the positive and negative impacts of various forest management strategies in relation to multiple climate change scenarios. The natural dynamics scenario (conservation) would maintain the highest amount of carbon in the DOM and biomass pool. CC-based scenarios project a reduced carbon pool storage in the ecosystem, although they favor a higher NEP and more carbon transferred in harvested wood products. Nonetheless, it is uncertain whether clearcutting will be an effective means of storing carbon over the long term because it favors the abundance of young and broadleaf forests characterized by short-lived products relative to the wood products from coniferous forests.

PC-based scenarios produce similar conditions as the natural dynamics scenario in terms of total ecosystem carbon storage, composition, and age structure (maintaining more old-growth and mature forests); nonetheless, the PC-based scenarios result in a lower NEP and less carbon transferred to harvested wood products than the natural scenario. Consequently, a compromise between the various canopy removal intensities is required to achieve forest management's multiple goals and maintain carbon dynamics. We propose applying the S5 strategy (harvest 75% of the managed area by PC treatments and 25% by CC treatments) or S4 (harvest 50% of the managed area by PC treatments and 50% by CC treatments) as strong possibilities. However, it is essential to note that implementing the S5 and S4 strategies may require additional access routes and infrastructure, which could potentially pose significant challenges and issues such as reduced landscape connectivity.

CHAPITRE V : CONCLUSION GÉNÉRALE

5.1 Recommandations pour l’aménagement forestier

Cette thèse présente des conclusions précieuses qui peuvent être incorporées dans les plans futurs d'aménagement forestier afin de promouvoir la gestion durable des forêts du Québec. Notre évaluation a comparé plusieurs stratégies alternatives, incluant le scénario actuel (« business-as-usual ») et un scénario de conservation (sans récolte) : i) les stratégies basées sur un aménagement équienne, où les CT et CPRS sont appliqués sur plus de 50% de la surface annuelle aménagée, ii) les stratégies basées sur aménagement inéquienne, où le CP est utilisé sur plus de 50% de la surface annuelle aménagée, et trois intensités de récolte sont considérées (25%, 50%, 75%), et iii) la stratégie de conservation, correspondant à la succession naturelle sans récolte. Nos résultats révèlent que dans la forêt boréale de l'Est (Québec), des stratégies alternatives qui font un compromis entre CP et CT sont nécessaires pour atteindre plusieurs objectifs, notamment la conservation de la biodiversité, les besoins industriels et l'atténuation du changement climatique par l'augmentation de la capacité de séquestration et de stockage du carbone.

Les forêts boréales du Québec, qui sont actuellement dominées par des espèces de conifères comme l'épinette, le sapin et le pin, pourraient connaître une augmentation de l'abondance des espèces de feuillus comme le bouleau et le tremble, principalement sous RCP2.6 et RCP4.5 dans le cadre de l'évolution naturelle. En outre, après 100 ans, l'abondance des espèces de conifères sera réduite dans les régions de l’ouest du Québec (région de l'Abitibi) en vertu du RCP8.5, tandis que les régions de l'est comme la Côte Nord seront un refuge pour les conifères dans tous les scénarios de changement climatique. De plus, les stratégies de gestion basées sur des traitements de forte intensité de récolte de la canopée, telles que CT et CPRS, peuvent accélérer l'abondance des espèces pionnières de feuillus (peuplier faux-tremble et bouleau blanc), tandis que les stratégies basées sur CP maintiendraient une plus grande rétention de la couverture de conifères. Cependant, la situation sera plus critique en l'absence de régénération des feuillus, en raison de la limitation des sols (p. ex. sols organiques, déficit en éléments nutritifs) et les accidents de régénération, où les forêts ouvertes atteindront plus de 20 % principalement selon le RCP8.5, notamment dans les régions de Saguenay-Lac-Saint-Jean. En ce qui concerne la structure d'âge, le principal facteur de changement dans la répartition des classes d'âge est la stratégie d'aménagement. Les stratégies basées sur les CT et CPRS diminuent l’abondance des forêts matures et anciennes de 10 % et 16 % respectivement et augmentent celle des jeunes forêts de 25 % par rapport à l'évolution naturelle. Le seuil des jeunes forêts peut atteindre 40 % par rapport au scénario naturel dans le cadre du RCP8.5, en raison de l'activité des incendies de forêt.

Dans les hautes latitudes de la forêt boréale québécoise où notre étude a été menée, la gestion peut modifier non seulement la composition et la structure d'âge de la forêt, mais aussi la capacité de séquestration et de stockage du carbone. Le scénario de conservation (pas de récolte) a permis d'atteindre le plus haut niveau de stockage total du carbone de l'écosystème, incluant la biomasse et la matière organique morte. Les stratégies basées sur les CP étaient plus proches du scénario de conservation en termes de stockage total du carbone, avec une différence de moins de 5tC ha^{-1} an^{-1} en moyenne sur la durée de la simulation, tandis que les stratégies basées sur le CT et CPRS présentaient la plus grande différence, qui pouvait atteindre plus de 10tC ha^{-1} an^{-1}. Néanmoins, les stratégies basées sur les CT et CPRS se caractérisent par un potentiel de séquestration du carbone élevé (Productivité Nette de l’Écosystème >0), en raison de l'abondance de jeunes forêts et d'espèces pionnières des feuillus. D'autre part, les stratégies basées sur les CP avaient un PNE similaire à celui de la conservation, qui était plus proche de zéro. En ce qui concerne la production nette du biome (PNB), les forêts aménagées du Québec réagiront comme des puits de carbone (PNB>0) entre 2010-2032 dans les régions de l'Ouest et du Centre, ainsi que dans l'Est entre 2010 et 2037. Après 2032, elles réagissent comme une source de carbone dans les régions du centre et de l'ouest, en raison des fortes émissions de carbone liées aux zones forestières perturbées par les épidémies de TBE, un résultat similaire étant observé dans l'Est en 2037. Toutefois, aucune différence considérable n'a été observée entre toutes les stratégies d'aménagement testées en termes de PNB à la fin des simulations. En fait, si l'on applique des stratégies basées sur les CT et CPRS, une grande quantité de carbone est transférée aux bois récoltés par rapport à celles basées sur les CP, alors que les CP pourront doubler la zone perturbée par les vents et les TBE, et transfèrent moins du carbone vers les produits de bois. Par conséquent, si les peuplements ne sont pas récoltés totalement, ils seront plus vulnérables aux perturbations et à la mortalité, ce qui pourrait accélérer le processus de décomposition et augmenter les émissions de carbone. Dans cette perspective, il est préférable d'éviter tous les scénarios extrêmes impliquant l'utilisation de 100% de CP (aménagement inéquienne) sur la surface annuellement récoltée. D’autre part, un aspect qui a été négligé dans notre analyse concerne les coupes de récupération suite aux perturbations. En effet, la récupération de la biomasse perturbée pourrait atténuer les fortes respirations hétérotopiques que nous avons observées dans notre étude, surtout après la TBE.

Le reboisement est couramment pratiqué après une CT ou CPRS et en cas de régénération naturelle insuffisante. Notre étude démontre que les stratégies avec des taux de reboisement plus élevés ne se traduisent pas nécessairement par une séquestration accrue du carbone par rapport au scénario naturel. Par exemple, la stratégie S3, qui prévoyait un taux de reboisement de 25 % sur la surface annuellement récoltée, n’a pas entraîné une séquestration supérieure du carbone par rapport aux scénarios basés sur CP ou l’évolution naturelle. De plus, les stratégies avec des taux de reboisement élevés, comme la stratégie S3 montrent des effets similaires à la référence "business as usual" (BAU) sous différents scénarios de changement climatique. D'autre part, nos résultats indiquent que les scénarios base sur les CP comme S4 et S5 (avec 8.3% et 16.3% de reboisement) sont les plus performants dans toutes les unités.

En conclusion, notre recherche indique qu'un compromis entre les intensités de récolte de la canopée, y compris CT, CPRS et CP (25%, 50%, 75%) est nécessaire pour atteindre plusieurs objectifs : la conservation de la biodiversité, la séquestration et le stockage du carbone, et l'approvisionnement industriel. Pour y parvenir, nous suggérons d'appliquer le CP à 50 % ou 75 % de la zone gérée annuellement et le CT/CPRS à la fraction restante. Ces stratégies nécessitent également d'augmenter le taux de reboisement de conifères dans les régions de l'Ouest (Abitibi) pour qu'il atteigne 8,3 à 16,7 % de la superficie annuelle récoltée après le CT. Néanmoins, le succès des CP dépend de leur intensité et de leur forme, telle que la coupe de jardinage (CJ), éclaircie commerciale, la coupe progressive irrégulière (CPI). Il est important de noter que certaines formes de CP peuvent augmenter les taux de mortalité après la coupe en raison de perturbations telles que le chablis, ce qui peut avoir un impact sur le bilan total du carbone.

Le changement climatique affecte les écosystèmes à travers le monde. Au cœur de cette préoccupation, la forêt boréale du Québec se trouve confrontée à des défis majeurs. Nos recherches examinent de près les stratégies visant à atténuer le forçage radiatif dans cette région. Plusieurs approches ont été étudiées, avec un accent particulier sur les types des coupes (CT vs CP). D’après nos résultats, les stratégies suivantes sont parmi les stratégies les plus prometteuses : les reboisements dans les régions où les accidents de régénération sont élevés à cause des feux, la diversification des espèces, l’adaptation de l’approche TRIAD qui consiste à créer des zones pour un aménagement intensif, des zones pour l’aménagement extensif, et d’autres pour la conservation. La diversification des stratégies au niveau du paysage est une approche solide pour s'adapter au changement climatique et répondre aux demandes industrielles. Il s'agit d'adopter une perspective écologique globale et une planification stratégique pour garantir la résilience des forêts face à l'évolution des conditions climatiques. Face aux défis du changement climatique, le secteur industriel doit se préparer de manière proactive à des changements dans la composition forestière. Un élément crucial de cette préparation est la transition vers des produits innovants qui tirent parti des espèces feuillues, en s'éloignant d'une dépendance prédominante à l'égard des conifères. Cette approche avant-gardiste permettrait au secteur industriel de prospérer dans un paysage changeant, reflétant un engagement en faveur de la durabilité, de la résilience et de la gestion responsable des ressources forestières dans un climat changeant.

Cependant, il est crucial de reconnaître que l'atténuation nécessite une approche multidimensionnelle, combinant l’aménagement forestier avec des politiques de réduction des émissions de gaz à effet de serre. La récupération du bois peut jouer un rôle crucial dans l’aménagement forestier durable et stockage du carbone au niveau du bois. La valorisation des produits ligneux tout en minimisant les émissions associées au transport doit être priorisée, notamment si les CP sont utilisés, car elles nécessitent un développement adéquat de l'infrastructure forestière. D'autres approches telles que le "fire-smarting", le contrôle TBE, et des traitements visant à rendre les peuplements moins vulnérables au chablis sont essentielles. Notre recherche a ouvert la voie à de nouvelles pistes, particulièrement l'évaluation de la biodiversité, l'impact du changement climatique sur les espèces clés, et l'optimisation des techniques de séquestration du carbone.

5.2 Limites du modèle

Notre projet montre aussi que Landis-II présente plusieurs limitations, lesquelles pourraient être améliorées à l'avenir. Cette étude ne prend pas en compte le sol paludifié dans la MU1, bien que dans les chapitres 1 et 2 les paramètres de tolérance à l'engorgement et à la sécheresse (H1-H4) aient été pris en compte. Aussi, les limitations en nutriments réduisent également la productivité, ce qui n'est pas actuellement modélisé dans Landis-II. Landis-II simule la dynamique du carbone à l'échelle du peuplement et extrapole les résultats à l'échelle du paysage. L'intégration de la relation nutriments/peuplement dans les modèles de succession pourrait donc constituer une amélioration prometteuse. D'autres limitations peuvent être prises en compte dans le modèle Landis-II, comme la dispersion des feuillus, puisque ces espèces rencontrent de nombreux obstacles à la régénération dans la zone d'étude (par exemple, sols organiques, carence en nutriments). Nous soulignons également que le modèle de récolte dans LANDIS-II utilise les surfaces récoltées annuellement plutôt que les volumes, ce qui explique pourquoi le modèle prélève plus de biomasse dans les scénarios basés sur les CT. Il serait donc préférable que le modèle de récolte cible les volumes plutôt que les surfaces. En outre, l'une des principales limites de cette étude est le modèle utilisé pour la TBE, qui entraîne des pertes de carbone immédiates et réduit le bilan carbone à moins de zéro. Les effets négatifs de la TBE ne sont pas nécessairement immédiats et peuvent persister pendant plus d'une décennie. Nous proposons de répartir les pertes dans le cadre de la TBE sur une période de 10 à 20 ans, plutôt que d’attribuer les pertes sur une seule année. Malgré ces limitations, LANDIS-II s’est révélé être un outil prometteur qui permet de simuler la dynamique du carbone forestier et explorer les conséquences à long terme de stratégies alternatives d’aménagement forestier à l'échelle du paysage par rapport à la référence actuelle (BAU) sous l'impact du changement climatique.

5.3 Impacts des coupes sur la biodiversité : exemple du caribou

Au cours des dernières années, les populations boréales de caribous déclinent dans la plupart de l'aire de répartition dans l'est du Canada. Cependant, le débat sur la contribution relative des différents moteurs du changement de composition, tels que les activités anthropiques, les perturbations naturelles et le changement climatique, persiste entre les industries, les politiciens, les organismes environnementaux, les communautés autochtones et les scientifiques. Plusieurs recherches révélant que les caribous évitent les zones récemment perturbées (par le feu ou les coupes), les peuplements feuillus, les grands plans d'eau et les polygones non boisés (en raison de l'échec de la régénération post-perturbation (Moreau, et al. 2012 ; Leclerc et al., 2014; Lafontaine et al., 2019 ; Gagné et al., 2016 ; St-Laurent et al. 2022), tandis qu'ils sélectionnent les zones humides, landes boisées à lichens et les vieux peuplements conifériens (Leclerc et al., 2014; Gagné et al., 2016 ; St-Laurent et al. 2022). Notre recherche démontre que les stratégies basées sur les CT et CPRS augmentent les pertes d'espèces d'arbres spécifiques et la réduction des surfaces des forêts matures et vieilles, en conséquence elles peuvent perturber l'habitat du caribou, alors que les stratégies basées sur les CP peuvent permettre d'atteindre les objectifs de conservation en maintenant de vieux arbres et le couvert des conifères. Les deux stratégies proposées, S4 et S5, peuvent permettre d'atteindre les objectifs de la gestion écosystémique, car elles maintiennent un schéma de végétation et une structure d'âge similaires à ceux de l'évolution naturelle.

D’autre part les caribous évitent presque toujours la proximité des routes. Alors que la mise en œuvre des stratégies basées sur les CP nécessite des infrastructures et un accès supplémentaire, ce qui pourrait constituer un défi important à l'avenir. En effet, il convient de noter qu'elles nécessitent l'expansion des réseaux routiers, ce qui entraîne la perte de zones forestières et une diminution de la connectivité du paysage, donc à une augmentation de la fragmentation de l'habitat pour le caribou. Généralement, le défi de la fragmentation et de la perte d'habitat constitue une menace importante pour la biodiversité en limitant le mouvement des espèces et en diminuant la connectivité au sein de l'écosystème. Afin de préserver les avantages des stratégies basées sur les CP, il est impératif que le développement de l'infrastructure forestière prenne en considération les implications écologiques plus larges. Cela implique de veiller à la préservation des corridors et à maintenir l'interconnexion des habitats.

5.4 Faisabilité économique des coupes partielles

Cette étude préconise l'adoption des CP dans l’aménagement forestier, promouvant une approche plus durable de l'exploitation des ressources en bois. Néanmoins, le Forestier en chef (2022) souligne que les CP sont plus coûteuses que les CT. En effet, pour obtenir le même volume que les CT, les CP doivent être mises en œuvre à divers endroits, nécessitant un accès plus important à la forêt et présentant des obstacles opérationnels qui contribuent au coût global. Toutefois, le coût des CT suivi du reboisement, pourrait dépasser le double de celui des CP. La problématique centrale soulevée ici concerne la faisabilité économique des stratégies proposées. Il est impératif de mener une analyse approfondie de la faisabilité économique des stratégies basées sur les CP et d'intégrer les externalités et les mesures de compensation du carbone dans les évaluations futures. Il est primordial de comparer dans les futures recherches les profits nets entre les traitements des CT, les CP et les initiatives de conservation, garantissant une compréhension complète des implications économiques de chaque approche, et favorisant ainsi une prise de décision éclairée dans le domaine de l'aménagement forestier.

ANNEXE A

MATÉRIELS SUPPLÉMENTAIRE DE L’ARTICLE 1 (CHAPITRE II)

Table A2.1 Annual burnt rate (%) per MU for each RCP scenario, the historic was used as a baseline. Those rates were calibrated from the literature. The wildfires were interactive using 3 fire maps per period, the rates of the period 2080-2100 were used as constant after 2100.

Period	RCP2.6			RCP4.5			RCP8.5		
	MU1	MU2	MU3	MU1	MU2	MU3	MU1	MU2	MU3
Historic	0.22	0.25	0.15	0.22	0.25	0.15	0.22	0.25	0.15
2010-2040	0.20	0.25	0.15	0.40	0.45	0.15	0.50	0.60	0.15
2040-2080	0.25	0.26	0.17	0.60	0.70	0.20	0.75	0.90	0.25
2080-2100	0.30	0.40	0.18	0.80	1.00	0.30	1.00	1.26	0.38

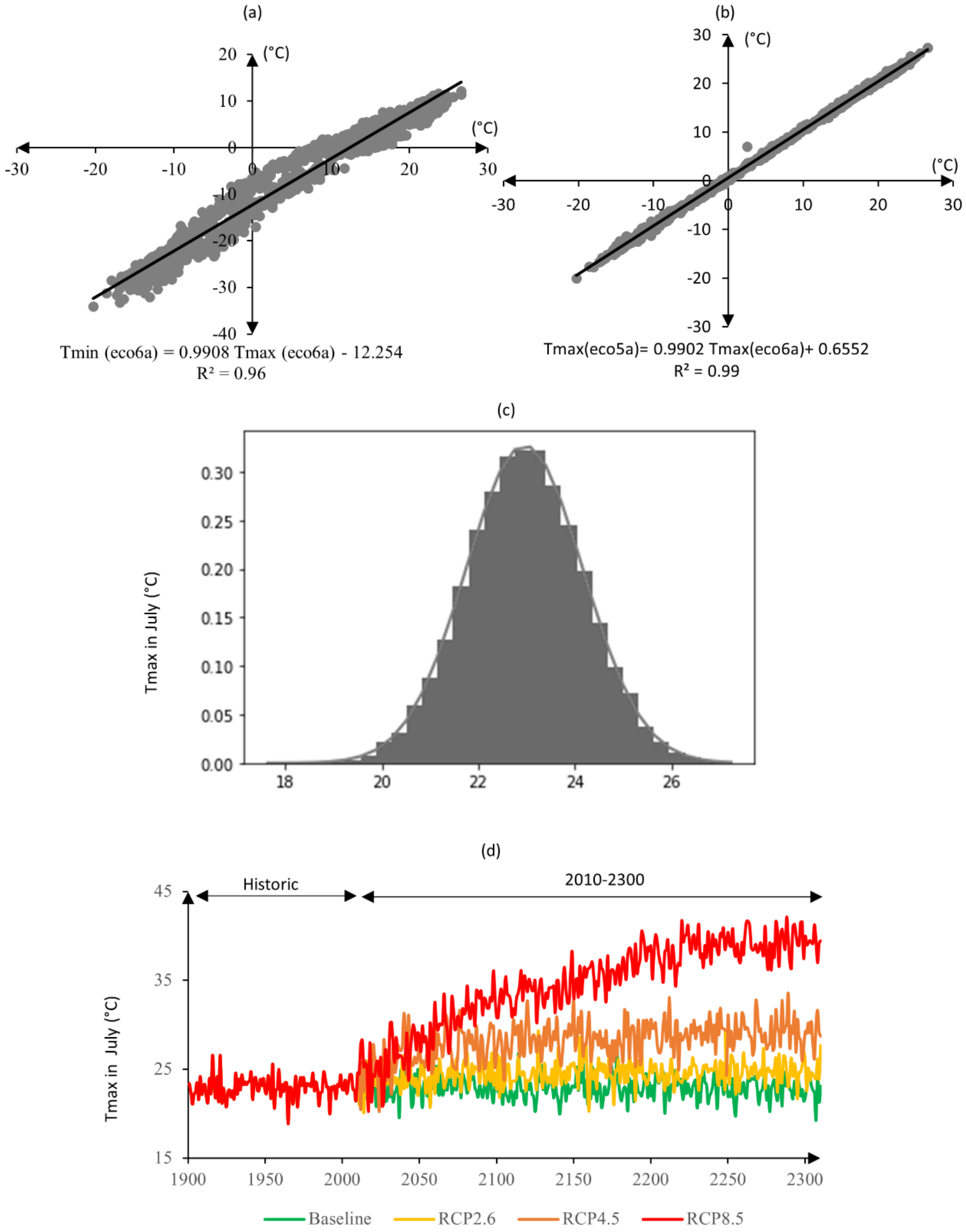

Figure A2.1 RCP and baseline climate scenarios and their Extension from 1900 to 2310 near the north limit of forest logging in Quebec boreal forest: an example of $T_{\max}$ of July in MU1 for ecoregion (6a) (explications below).

The ClimateNA model provides the historic of 1900-2010. For the baseline, each parameter was extended for 300 years using random values per month based on it is gaussian distribution around the mean (Monte Carlo Simulation method) observed from the historic (Figure A2.1 (c)). The Supplementary Material (Figure A2.1) above provides an example in MU1 of selected values for July maximum temperature. The Tmin was highly correlated to Tmax (R2>0.95), so Tmin was linearly extrapolated using generated Tmax values (Figure A2.1 (a)). This method provides good data for the baseline. The extended climate must respect the spatial effect, indeed, for each MU the climate of each ecoregion needs to be correlated to the others, based on their observed relationship from the historic. For instance, the maximum temperature in the ecoregion (5a) is positively correlated to those of eco6a (R2=0.99) (Figure A2.1 (b)). For all climate change scenarios, the ClimateNA model offers RCP data until 2100. For the RCP scenarios extrapolation, we used the Extended Concentration Pathways (ECPs), recommended by Meinshausen et al. (2011) and IPCC group (Collins et al. 2013), which describe the RCPs from 2100 to 2500 that were calculated using simple rules (Figure A2.1 (d)):

RCP2.6 corresponding ECP assuming constant emissions after 2100. So, we assumed and generated a constant and similar climate to 2081-2100 (last 20 years) period for the period 2100-2310.

RCP4.5 corresponding ECPs assuming constant concentrations after 2150, with an augmentation of 4°C every 100 years in the study area. So, we extrapolated the climate increasing the observed average of 2081-2100 by 2°C for all months until 2150 then we generated constant climate data until 2310 similar to the 2130-2150 period, based on its gaussian distribution.

For RCP8.5 the historic average was increased by 5.2 °C for all months per 100 years in the study area compared to the historic average, this high pathway for which radiative forcing reaches greater than 8.5 W m^{-2} by 2100, corresponding ECP assuming constant concentrations after 2250). Therefore, for the next 150 years (2100-2250) we increased the average by 7.8 °C, then a constant climate was generated for the period 2250-2310 similar to the 2230-2250 period.

The precipitation and PAR were extended during the period 2100-2310 using their Gaussian distribution around the mean of the last 20 years of the period 2010-2100, assuming 10% as standard error and considered constant for all scenarios after 2100; those two parameters were provided by ClimateNA model as well for the period (1900-2100).

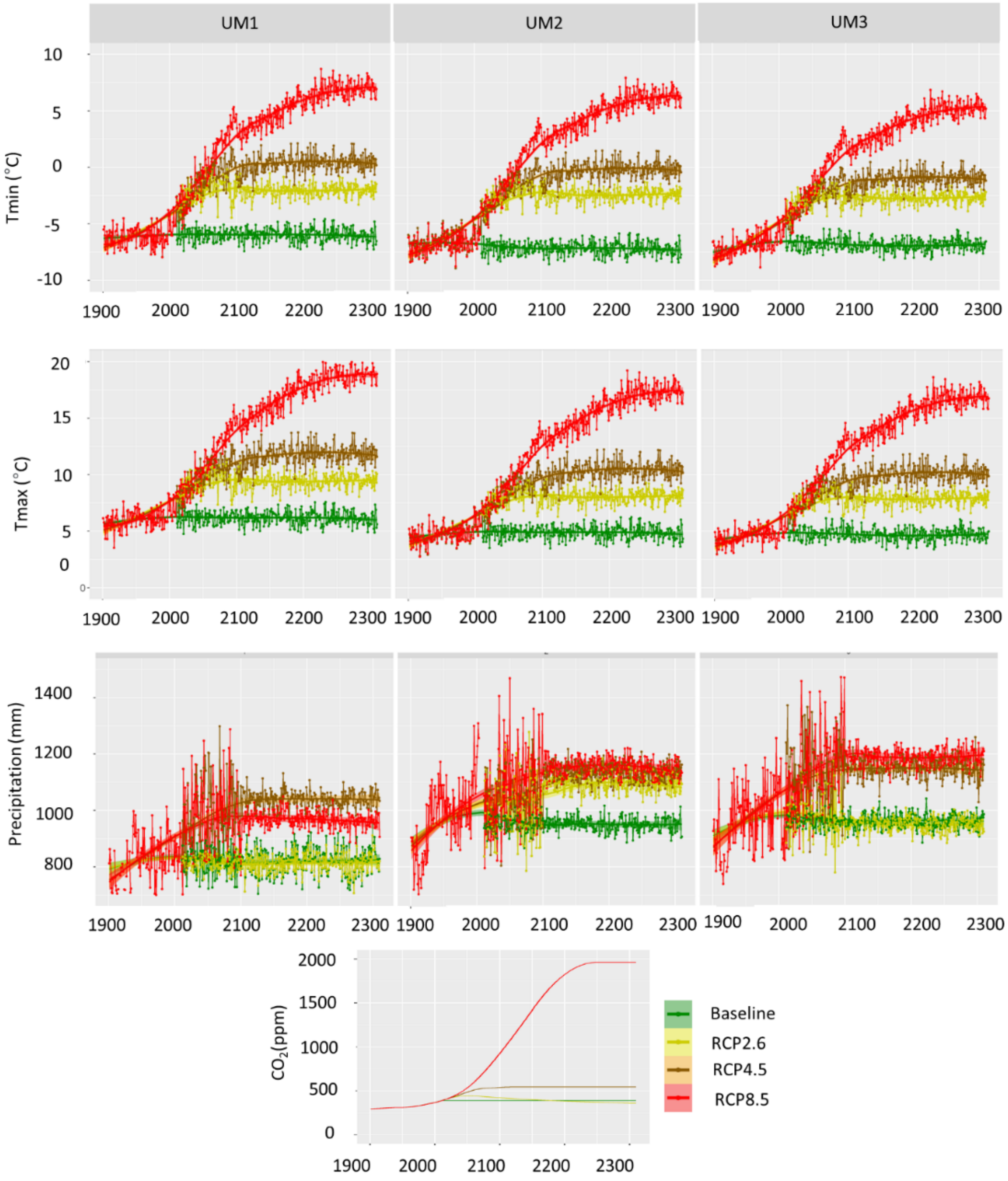

Figure A2.2 The projection of the annual average of Tmax, Tmin, and precipitation, calculated as the mean for all months, and the level of atmospheric CO_2 concentration for RCP and baseline climatic scenarios.

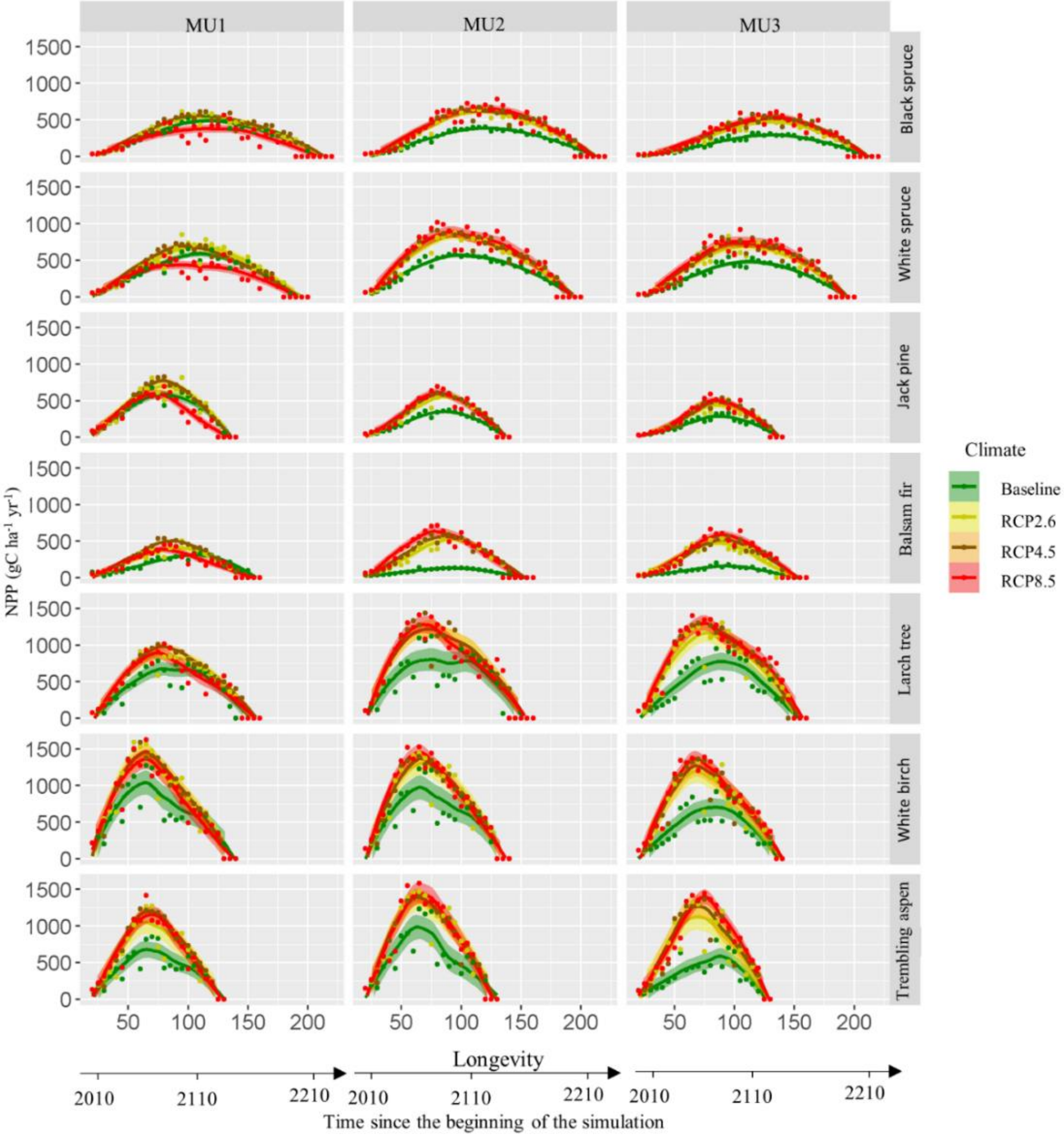

Figure A2.3 Species NPP during their longevity age in the three management units under climate change scenarios and current climate (baseline).

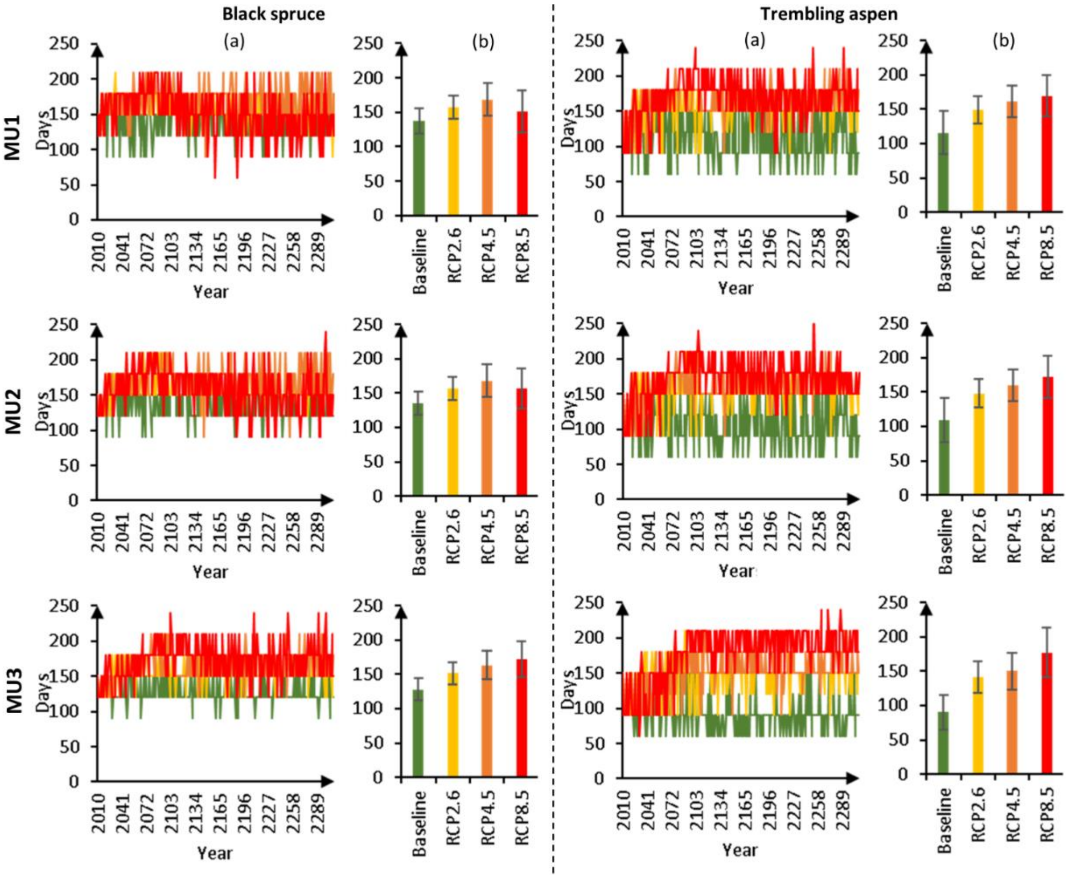

Figure A2.4 Growing season time series of black spruce and trembling aspen in the three MU. b) the growing season average with its standard error observed during the study period per climate change scenarios.

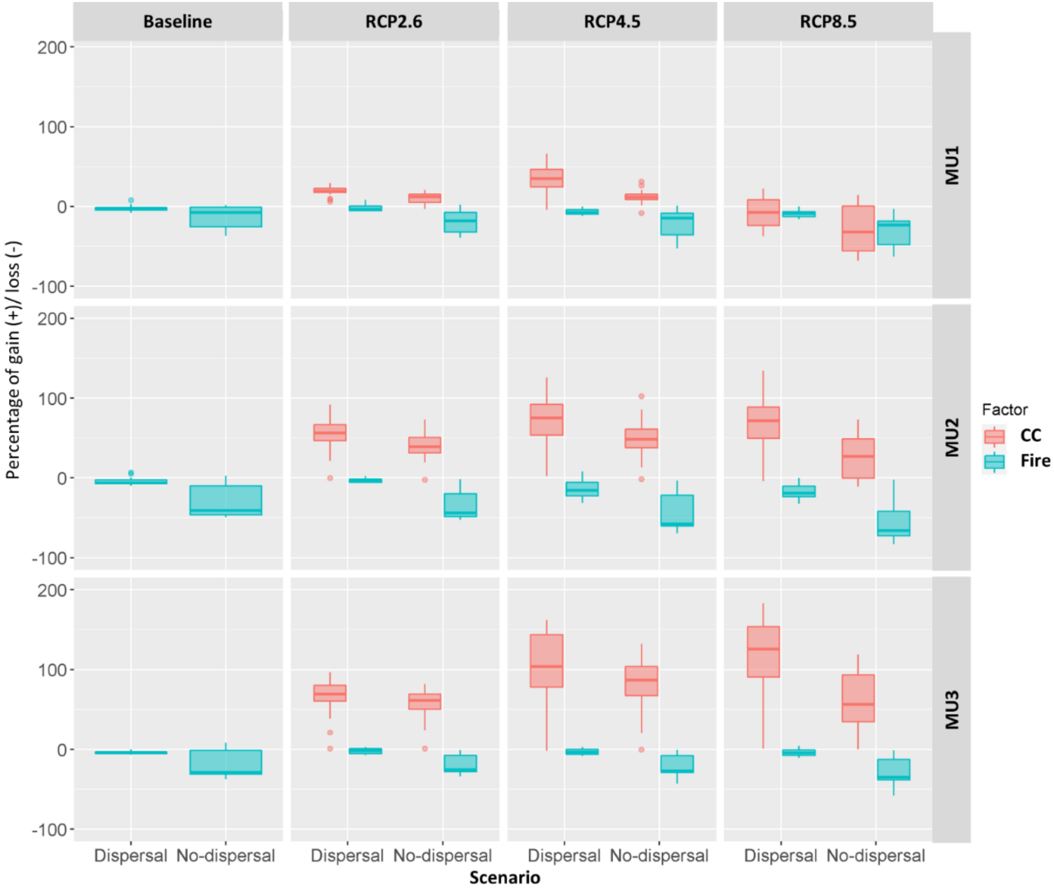

Figure A2.5 The averages of the relative percentage of biomass carbon storage losses and gains during the entire study period for both scenarios of broadleaves dispersal. It was calculated for climate change (CC) scenarios (RCP) relatively to the baseline, while it was calculated for fire relatively to only succession scenario (no disturbance).

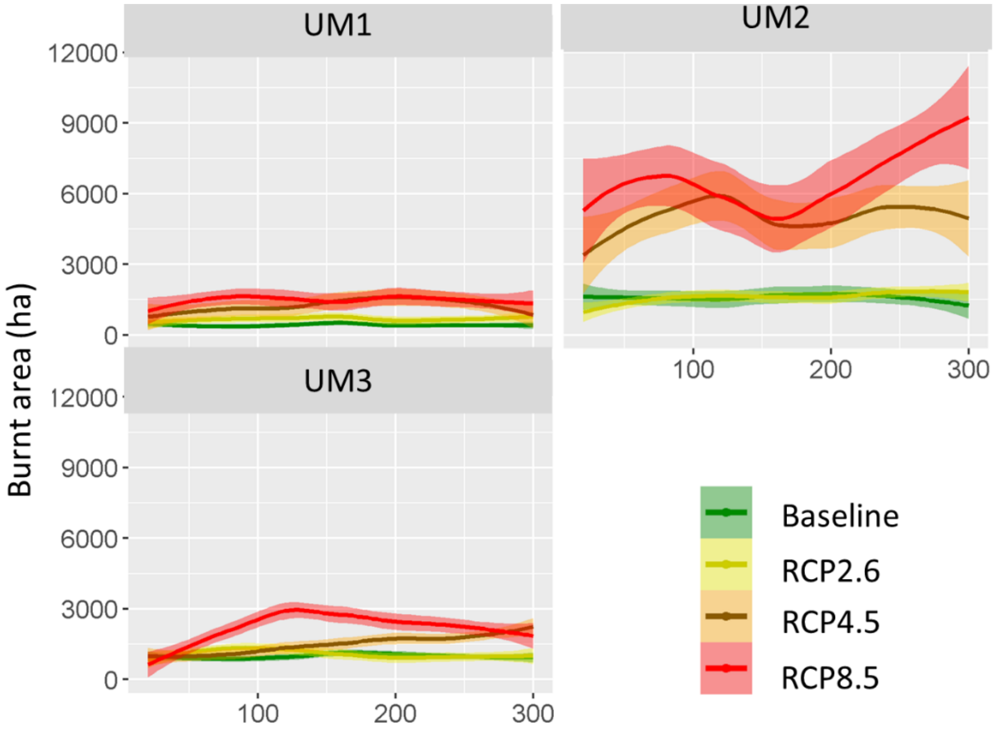

Figure A2.6 Total annual Burned area projection for 300 years (2010-2310) in all MUs (mean of three replications). The losses in forest area under wildfires disturbance will have more fluctuation for extreme scenarios (RCP4.5 and RCP8.5) in MU2 with high values compared to that of the baseline (current burn rate).

ANNEXE B

MATÉRIEL SUPPLÉMENTAIRE DE L’ARTICLE 2 (CHAPITRE III)

Table A3.1 The life-history attributes for the 8 species according to Boulanger et *al.* (2017) (L: longevity (years), SM: sexual maturity, ST: shade tolerance, SDD: seeds dispersal distance, VRP: vegetation reproduction probability, PFR: post-fire regeneration)

Species	**L**	**SM**	**ST**	**FT**	**SDD effective**	**SDD Max**	**VRP**	**VRP min age**	**VRP max age**	**PFR**
Abies balsamea	150	30	5	1	25	160	0	0	0	none
Betula alleghaniensis	220	40	3	1	100	400	0.1	10	180	resprout
Betula papyrifera	140	20	2	1	100	1000	0.5	10	70	resprout
Larix laricina	160	40	1	1	50	200	0	0	0	none
Picea glauca	200	30	3	2	100	300	0	0	0	none
Picea marian	220	30	4	2	80	200	0	0	0	serotiny
Pinus banksiana	140	20	1	2	30	100	0	0	0	serotiny
Populus tremuloides	130	20	1	2	500	5000	0.9	10	130	resprout

Table A3.2 The total area of each management unit (three MUs: North-of-Quebec (MU1), Saguenay-Lac-Saint-Jean (MU2), and Côte-Nord (MU3)), with allowable annual cut (AAC) determined for the current planning cycle (2023-2028). Annual harvested area (%) per management scenario to fulfill the AAC.

MU	**Total area (Mha)**	**AAC (Mg)**	**Annual managed area (%) per scenario**					
			S1	**S2**	**S3**	**S4**	**S5**	**S6**
MU1	0.62	245808	1.18	1.32	1.48	1.70	2.21	2.89
MU2	1.16	511644	1.95	2.04	2.20	2.56	2.72	4.27
MU3	1.14	405547	1.70	1.80	1.83	2.07	2.55	3.55

Table A3.3 Used treatment at stand scale and % of managed area per treatment for the BAU (historic of 1970-2010) per MU and their management areas. CC + reforestation reflects that clear-cut or CPRS is followed by replanting, due to a lower soil seed banks and regeneration rate.

MU	**Management area**	**Map code**	**area (10^3)**	**Used treatment at stand scale and % of managed area per treatment under BAU**					
				CC	**CC+ref**	**CPRS95**	**PC75**	**PC50**	**PC25**
1	Macamic lake	1	31.7	40.91	9.40	44.80	0.88	0.00	4.02
	Obalski lake	2	82.3	73.70	17.82	6.39	0.18	0.00	1.91
	Mistaouac lake	3	225.2	39.74	23.55	33.07	1.97	0.28	1.39
	Grasset lake	4	279.2	34.16	8.49	56.37	0.10	0.02	0.86
	Weighted average			41.80	15.26	40.64	0.83	0.11	1.36
2	Connelly lake	1	80.0	28.03	5.61	58.17	0.32	6.39	1.48
	Onatchiway lake	2	72.1	40.31	4.77	47.08	0.00	6.34	1.49
	Pipmuacan Reservoir	3	300.3	37.28	17.06	44.30	0.16	0.94	0.25
	Péribonka lake	4	465.8	10.35	12.96	72.46	3.72	0.52	0.00
	Hirondelles lake	5	232.3	0.89	0.72	92.78	4.62	1.00	0.00
	Weighted average			18.57	10.54	66.63	2.50	1.50	0.26
3	Dionne lake	1	149.9	86.30	0.07	11.74	0.00	1.60	0.30
	LM Forestville Lake	2	222.7	93.40	1.82	2.61	0.01	2.14	0.01
	CBM lake	3	213.7	0.01	1.42	97.31	0.00	1.26	0.00
	Manic Reservoir	4	562.1	47.12	2.22	47.07	0.54	3.03	0.02
	Weighted average			52.44	1.71	43.19	0.27	2.34	0.05

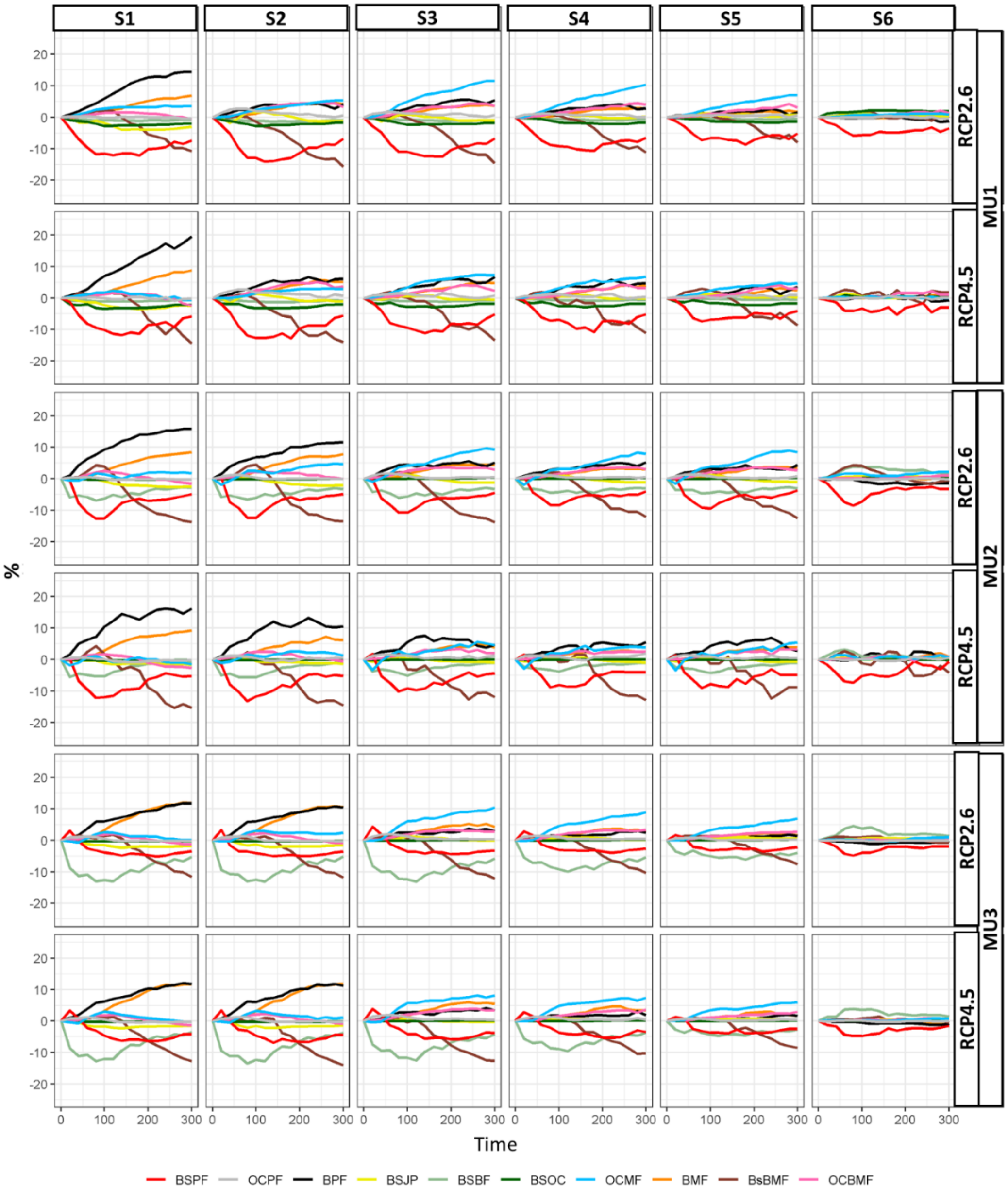

Figure A3.1The difference (*Δ*; %) of composition percentage between natural scenario and management scenarios in the three MU between 2010 (year 0) and 2310 (year 300) under RCP2.6 and RCP4.5 climate change scenario. (BsPF: black spruce pure forests, OcPF: Other coniferous pure forests, BPF: broadleaves pure forests, BsJP: black spruce and jack pine, BsAB: black spruce and balsam fir, BsOC: black spruce and other coniferous, OCMF: Other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, OCBMF: Other coniferous and broadleaves mixed forests).

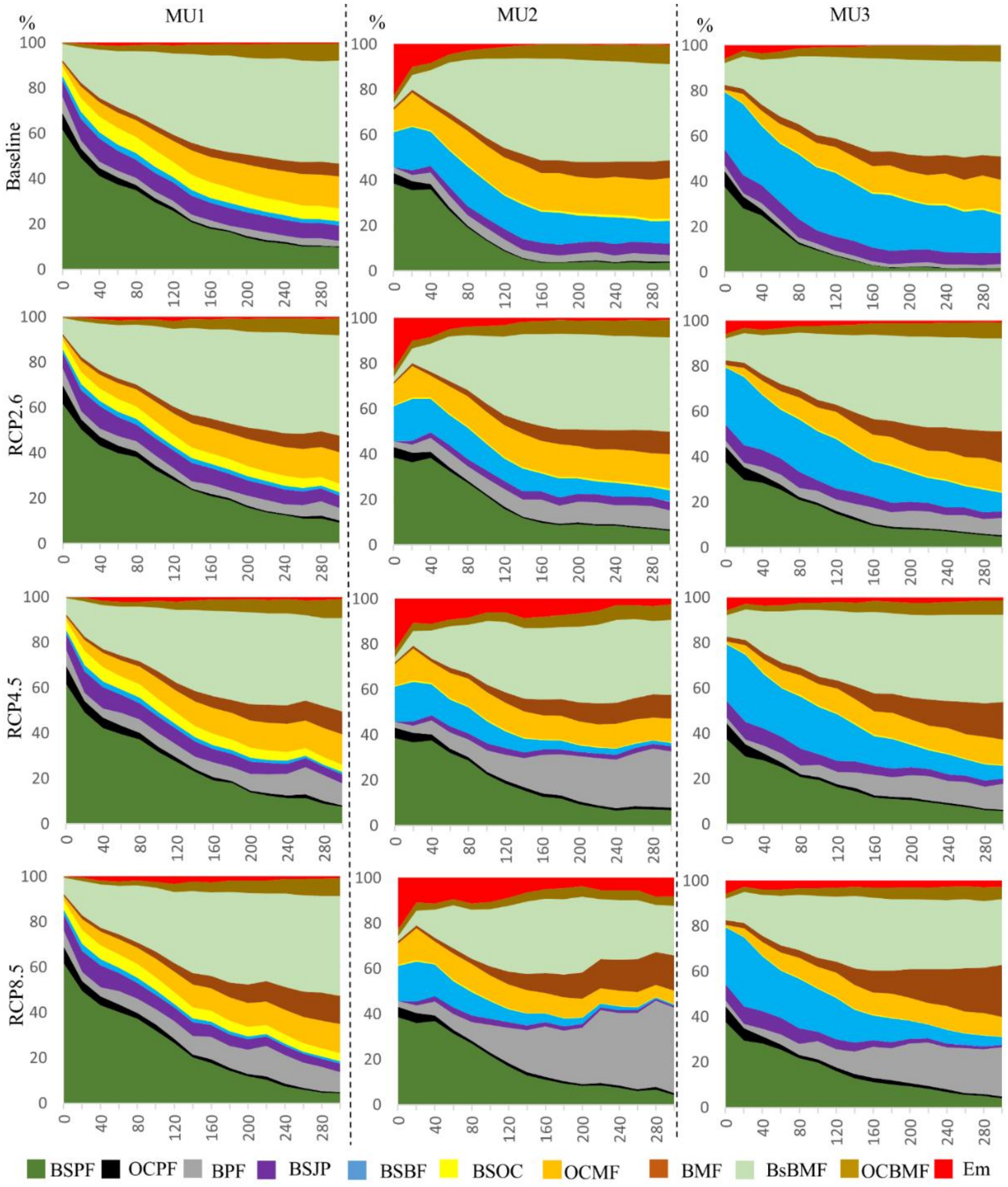

Figure A3.2 The occupancy area (%) of forest type in the three MU between 2010 (year 0) and 2310 (year 300) under climate change scenarios and no-harvest scenario, where the natural disturbances (SBW, wind, fire) were considered (BsPF: black spruce pure forests, OcPF: Other coniferous pure forests, BPF: broadleaves pure forests, BsJP: black spruce and jack pine, BsBF: black spruce and balsam fir, BsOC: black spruce and other coniferous, OCMF: Other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, OCBMF: Other coniferous and broadleaves mixed forests; Em: empty land which represents open forest woodland).

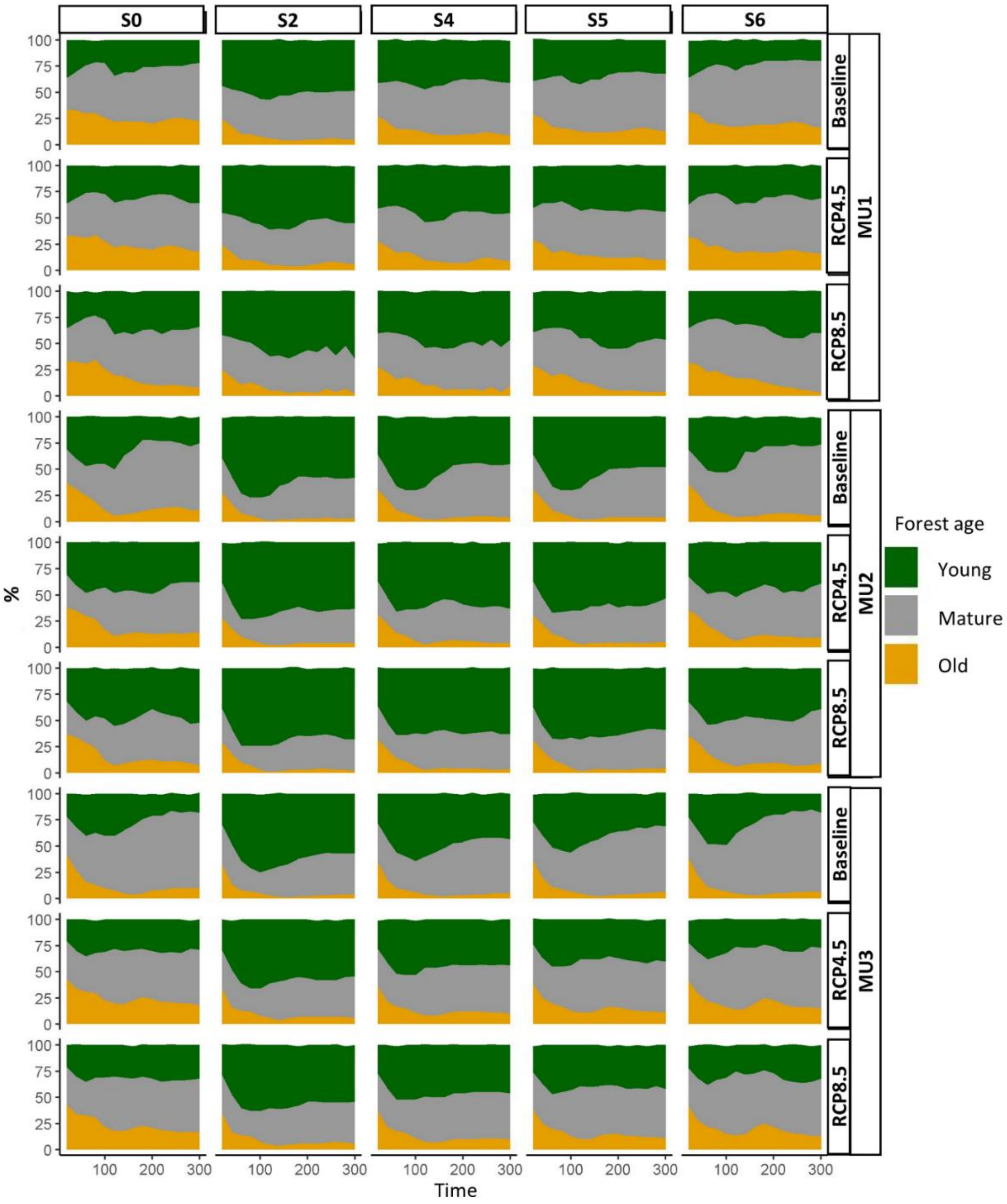

Figure A3.3 Age structure (young forest ≤40; 40<mature forest≤100; 100< old growth forest) under current climate and RCP8.5 scenarios, for natural dynamics scenario (S0; no-harvest), BAU (S2) and PC-based scenarios (S4, S5, S6).

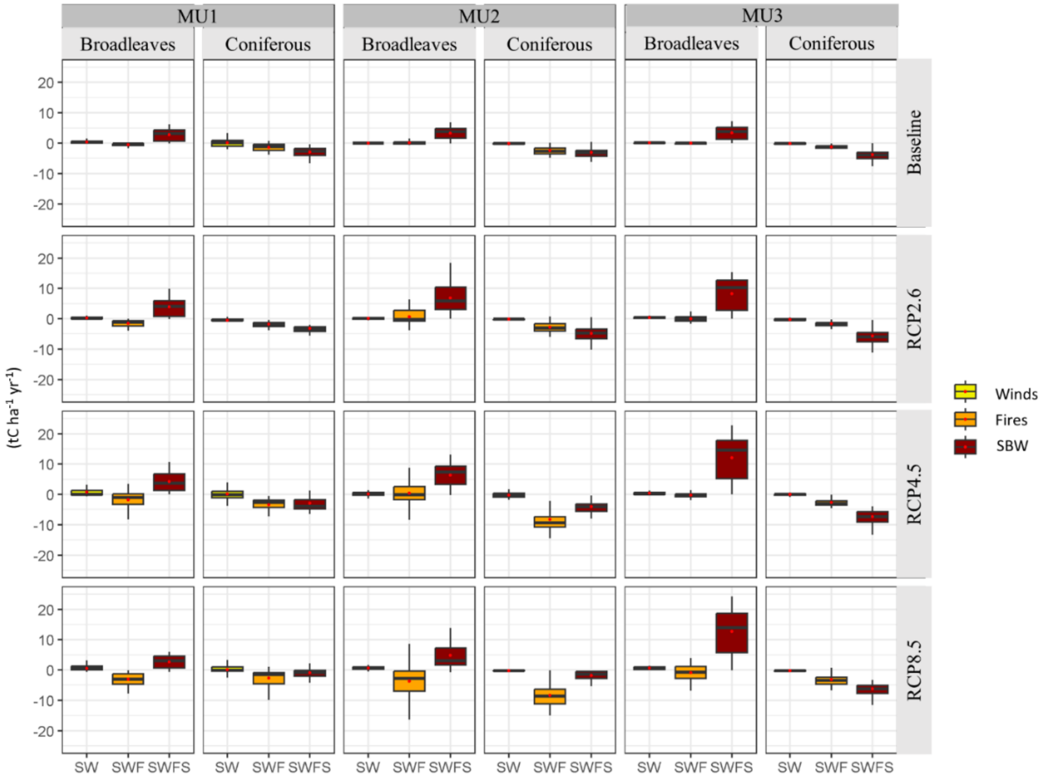

Figure A3.4 Carbon losses (-) or gain (+) (tC ha^{-1} yr^{-1}) average projection coniferous and broadleaves species in the three management units (1, 2, 3) under different natural disturbances (Winds, fires, SBW) and climate change scenarios. The averages were estimated.

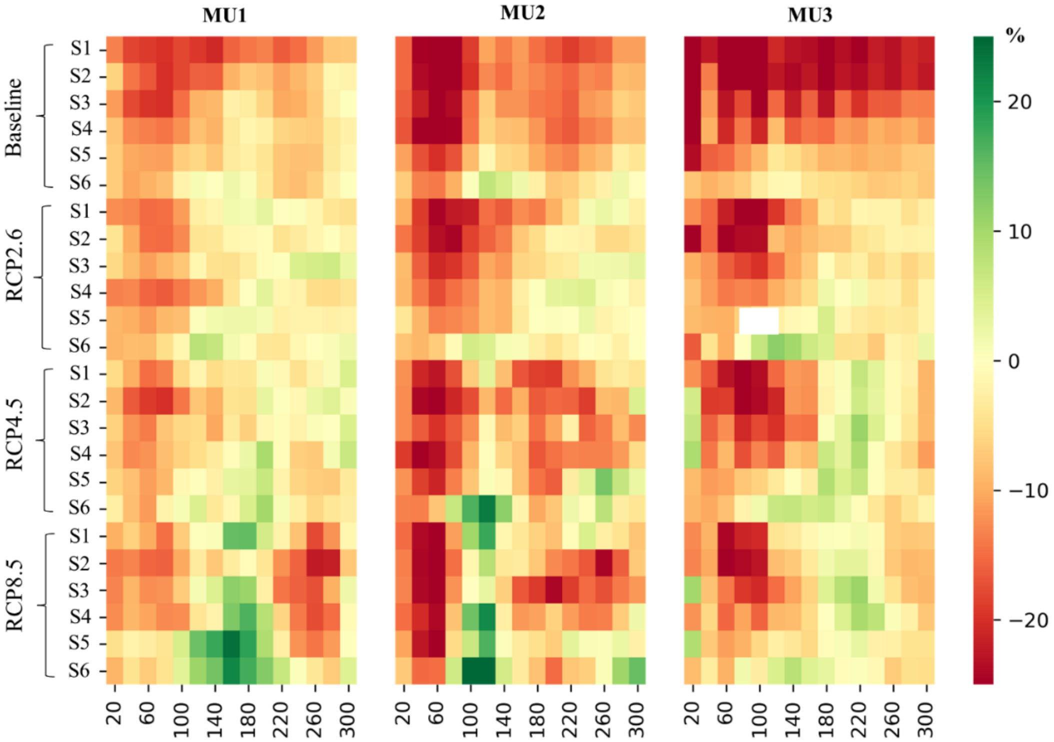

Figure A3.5 The relative changes in biomass carbon storage compared to the natural scenario (S0) expressed in percentage.

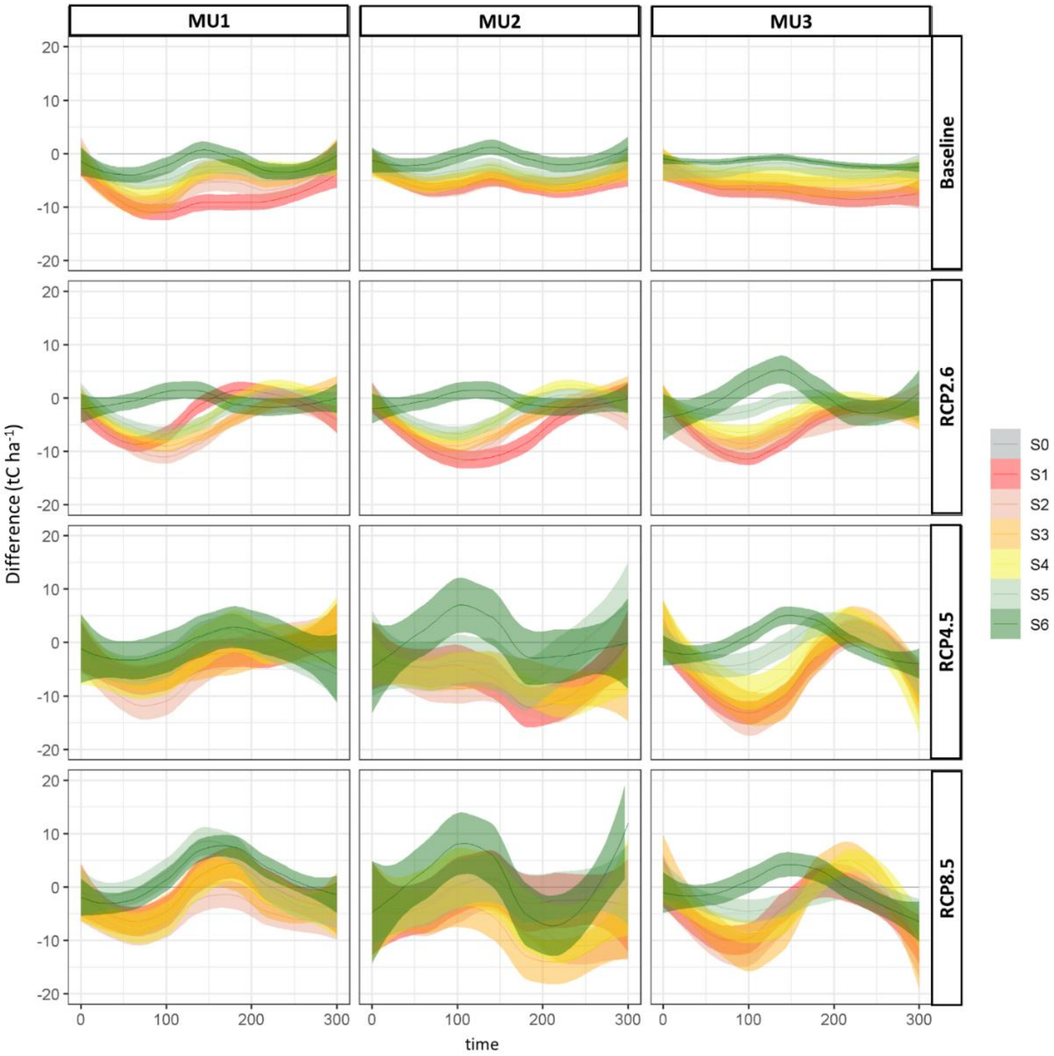

Figure A3.6 The living biomass carbon storage differences (Δi,j) from 2010 (year 0) to 2310 (year 300) between management scenarios and no harvest scenario (S0: natural disturbances only) with their confident interval.

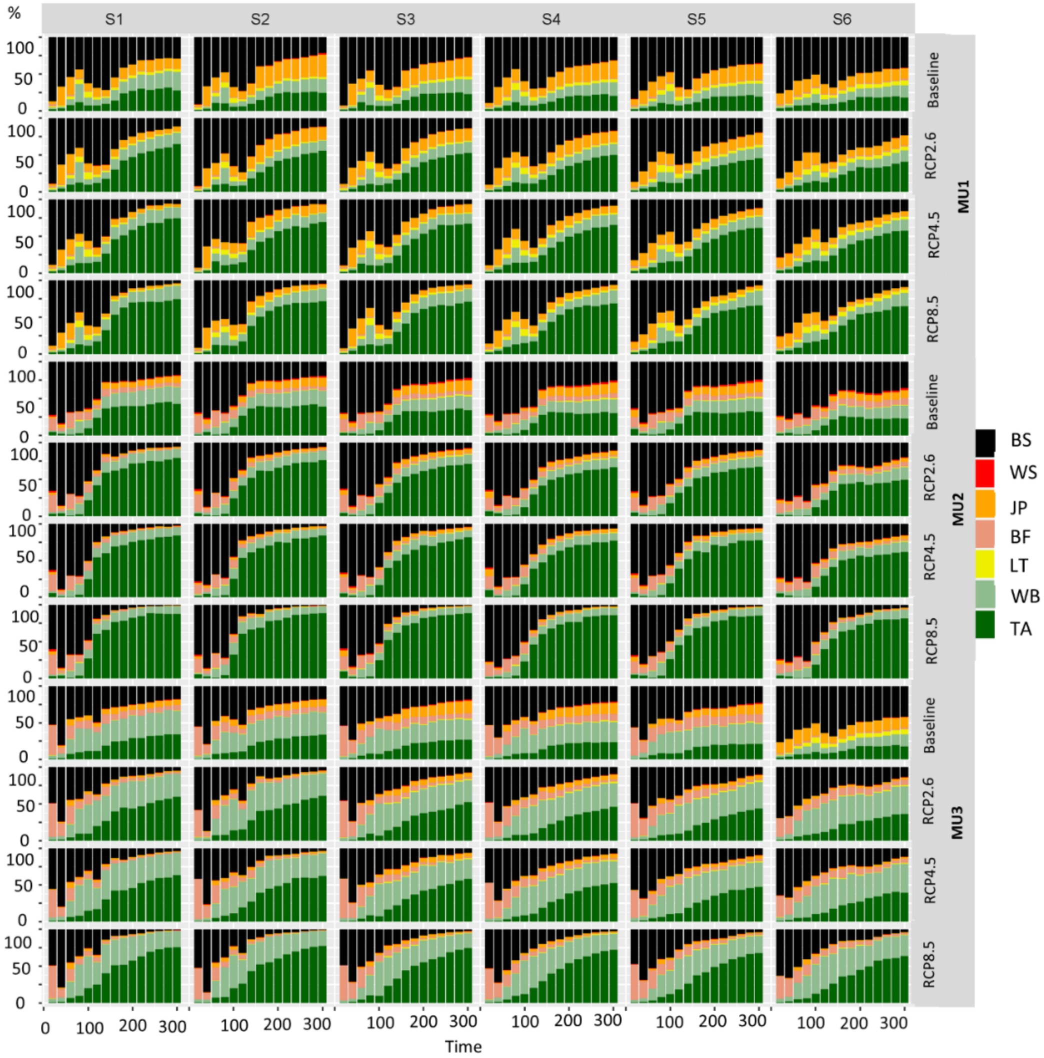

Figure A3.7 The relative percentage of ACC per species in the three management units under climate change scenarios and six management strategies from 2010 (year 0) to 2310 (year 300) (legend: BS: black spruce, WS: white spruce, JP: jack pine, BF: balsam fir, LT: larch tree, WB: white birch, TA: trembling aspen). This SM reflects the contribution of each species in annual harvested biomass.

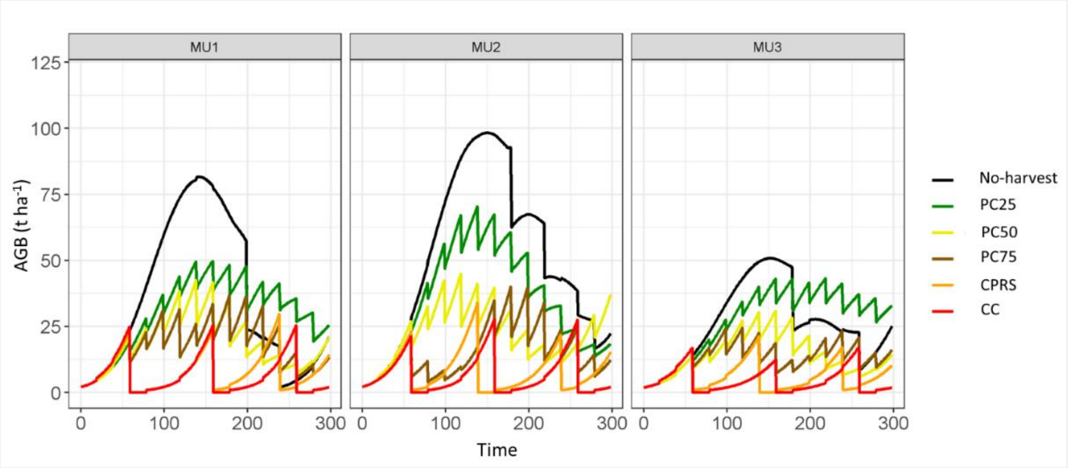

Figure A3.8 Above-ground biomass (AGB) response to silvicultural treatment at stand scale of the black spruce pure forest. PCs stabilized biomass carbon pool compared to CC and CPRS in the long term.

ANNEXE C

MATÉRIEL SUPPLÉMENTAIRE DE L’ARTICLE 3 (CHAPITRE VI)

Table A4.1 Simulated management units and their ecoregions (homogeneous zone based on climate and soil).

Region	MU	Bioclimatic domains	ecoregion code	Soils texture	Area (%)
Quebec Nord-West	MU 1	Balsam fir-white brich	11	clay	12
			16	clay loam	06
			17	Loam	01
		spruce-moss	21	Clay	72
			24	Sand clay loam	02
			26	Clay loam	06
			27	Loam	02
Saguenay lac-saint Jean	MU 2	Balsam fir-white brich	101	Sand	03
			102	Sandy loam	35
		spruce-moss	201	Sand	04
			202	Sandy loam	57
Côte-Nord	MU 3	Balsam fir-white brich	101	Sand	02
			102	Sandy loam	30
		spruce-moss	201	Sand	04
			202	Sandy loam	64

Table A4.2 The total area of each management unit (three MUs: North-of-Quebec (MU1), Saguenay-Lac-Saint-Jean (MU2), and Côte-Nord (MU3)), with allowable annual cut (AAC) determined for the current planning cycle (2023-2028). Annual harvested area (%) per management scenario to fulfill the AAC.

MU	Total area (Mha)	AAC (Mg)	Annual managed area (%) per scenario					
			S1	S2	S3	S4	S5	S6
MU1	0.62	245808	1.40	1.45	1.48	1.60	1.90	2.20
MU2	1.06	511644	2.20	2.30	2.37	2.56	2.80	3.50
MU3	1.15	405547	1.90	2.00	2.10	2.22	2.35	2.90

Table A4.3 Tested scenarios and their description. Used treatment at stand scale with different harvesting intensity and the percentage of annually managed area per treatment.

Scenarios	Code	Description	Used treatment at stand scale and % of annual managed area per treatment					
			CC	CC+Aff	CPRS	PC75%	PC50%	PC25%
Scenario-0	S0	No harvest scenario under natural disturbances	0.0	0.0	0.0	0.0	0.0	0.0
Scenario-1	S1	All the annually harvested area (AHA) is managed using high CRI (CC and CPRS). The establishment is based only on regeneration.	50.0	0.0	50.0	0.0	0.0	0.0
Scenario-2	S2	BAU. Currently used scenario, where CPRS and CC are used for more than 90% of AHA, and 10% of AHA is managed using PCs with 25%, 50%, 75% of CRI.	Historic % per MU from forest inventory geodatabase.					
Scenario-3	S3	we used 75% of AHA for high CRI (CC and CPRS) and 25% for low-removal ones (PC)	25.0	25.0	25.0	8.3	8.3	8.3
Scenario-4	S4	we used 50% of AHA for high-removal treatments and 50% for low-removal ones.	16.7	16.7	16.7	16.7	16.7	16.7
Scenario-5	S5	we used 25% of AHA for high-removal treatments and 75% for low-removal ones.	8.3	8.3	8.3	25.0	25.0	25.0
Scenario-6	S6	Extreme use of PCs (100% of AHA), the opposite of scenario 1	0.0	0.0	0.00	33.3	33.3	33.3

-Each scenario needs to be parameterized in biomass harvest and calibrated to fulfill the allowable annual cut under the current climate.

-The reforestation level (CC+ref) is the percentage of the afforested area after harvest by CC and CPRS with a lower regeneration rate. From the historic 14.8%, 8.9%, and 1.4% of AHA were afforested in MU1, MU2, and MU3 respectively.

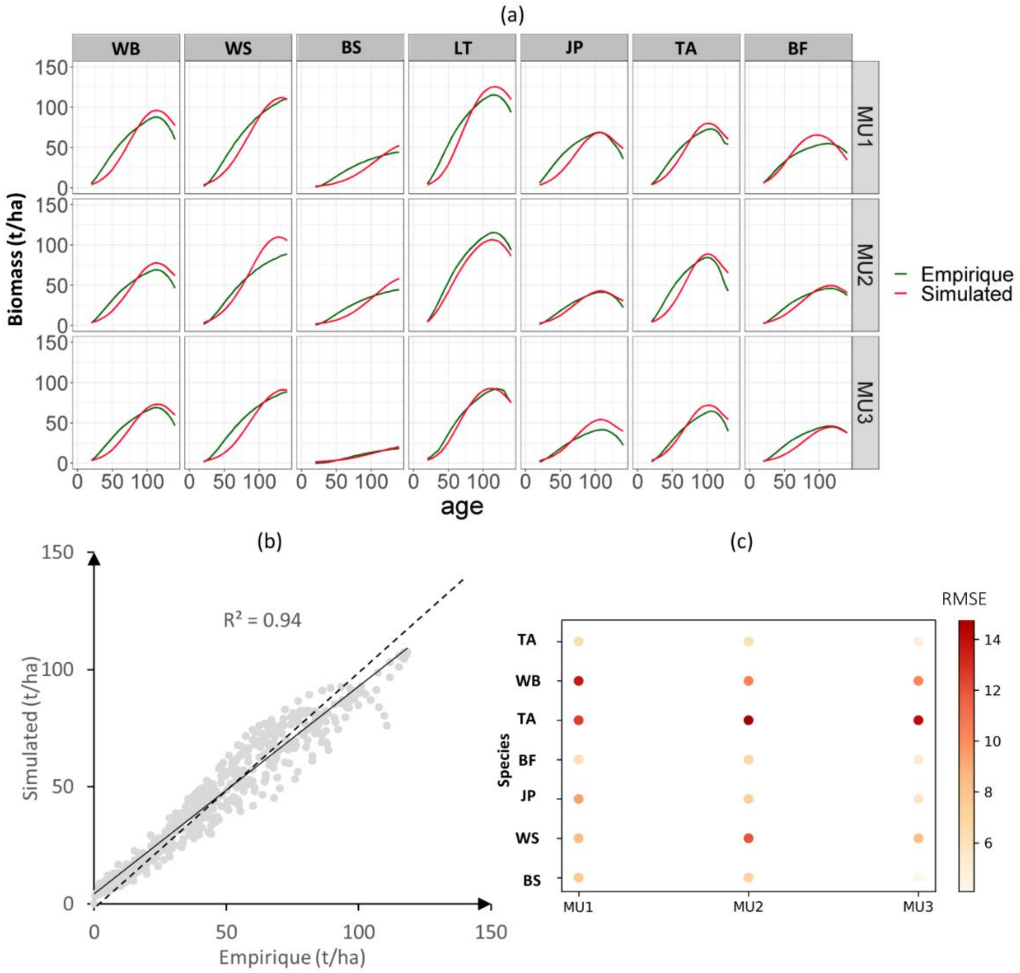

Figure A4.1 a) PnET succession calibration based on yield curves (biomass as a function of age) from Pothier and Savard (1998). Fig b) The correlation between all simulated and empirical values used for calibration, the dotted line represents the 1:1 ratio, and the continuous line represents the adjusted model. Fig c) The root-mean-square error (RMSE) between Be and Bs in the three MU. The calibration of PnET succession Beta version parameters was based on Gustafson's guidance (Gustafson & Miranda, 2019) and Ameray et al.(2023) method using Python (legend: BS: black spruce, WS: white spruce, JP: jack pine, BF: balsam fir, LT: larch tree, WB: white birch, TA: trembling aspen).

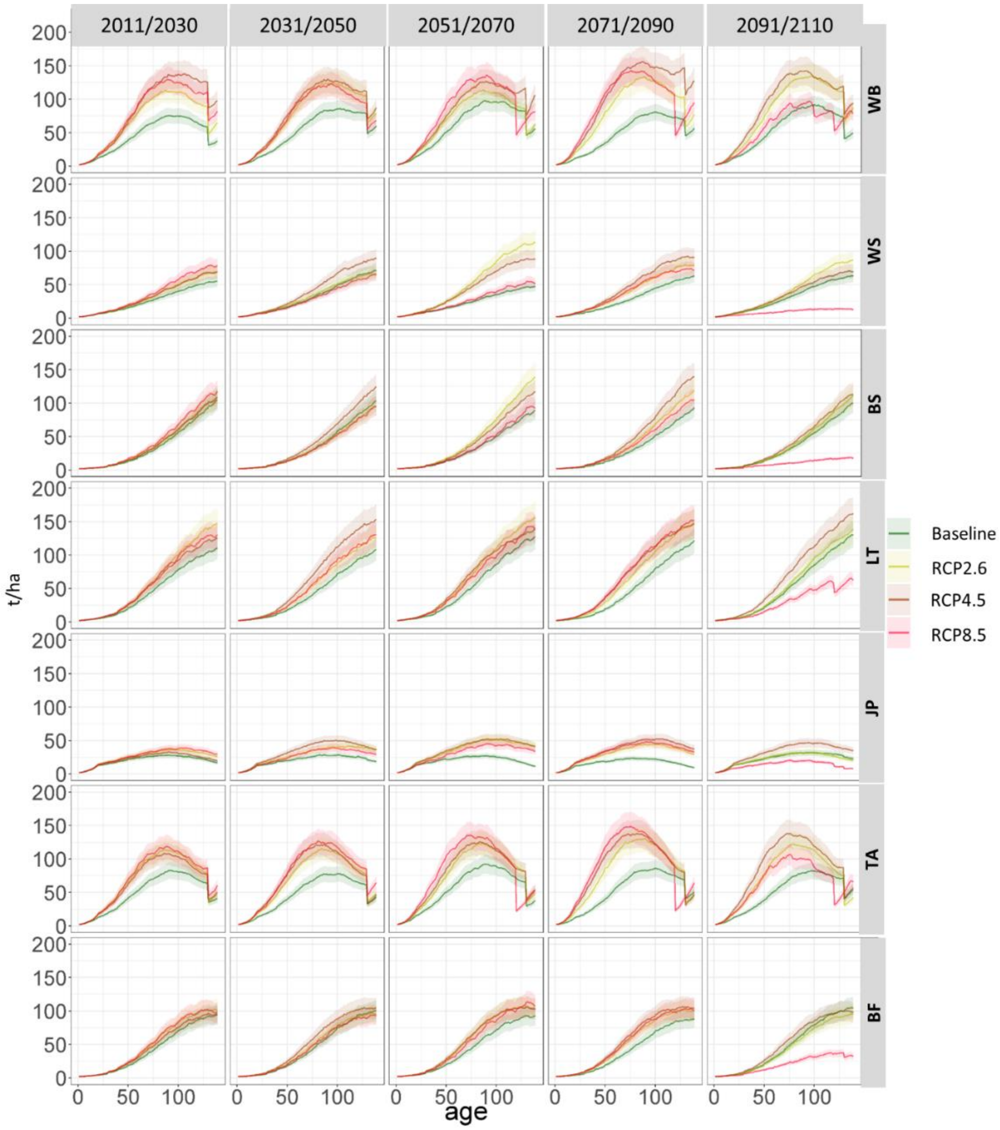

Figure A4.2 PnET succession model was run using 10 years time step and provides AGB, ANPP, and establishment probabilities (Est). Here, in the following figures, we provide examples of those three parameters in the dominant ecoregion per MU (ecoregion 21 in MU1, ecoregion 202 in MU2, and ecoregion 202 in MU3, see Table A4.1) for all periods under climate change scenarios. (Legend: BS: black spruce, WS: white spruce, JP: jack pine, BF: balsam fir, LT: larch tree, WB: white birch, TA: trembling aspen, YB: yellow birch). The Figure above presents Figure A4.2.1(a) which reflects AGB in MU1.

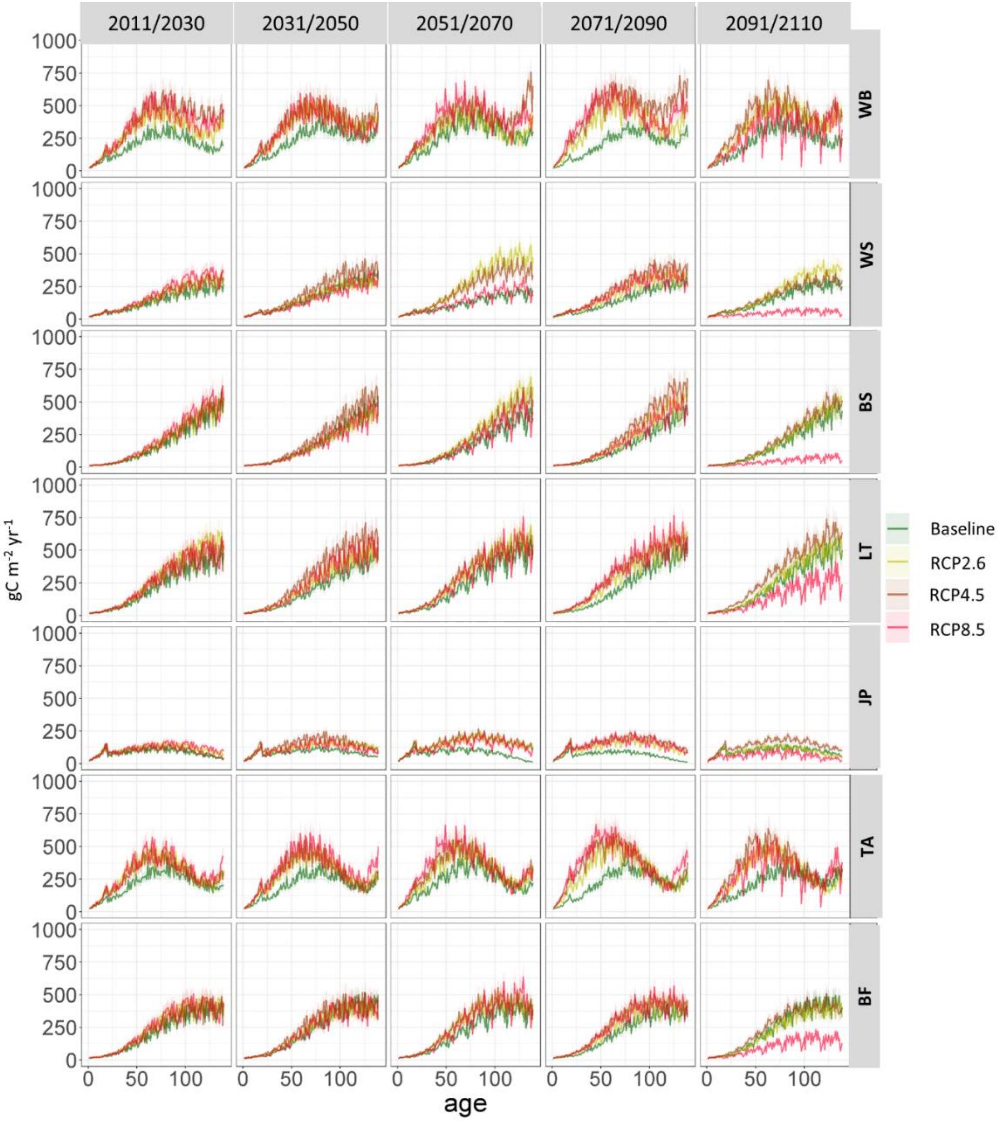

Figure A4.2.1(b) ANPP in MU1.

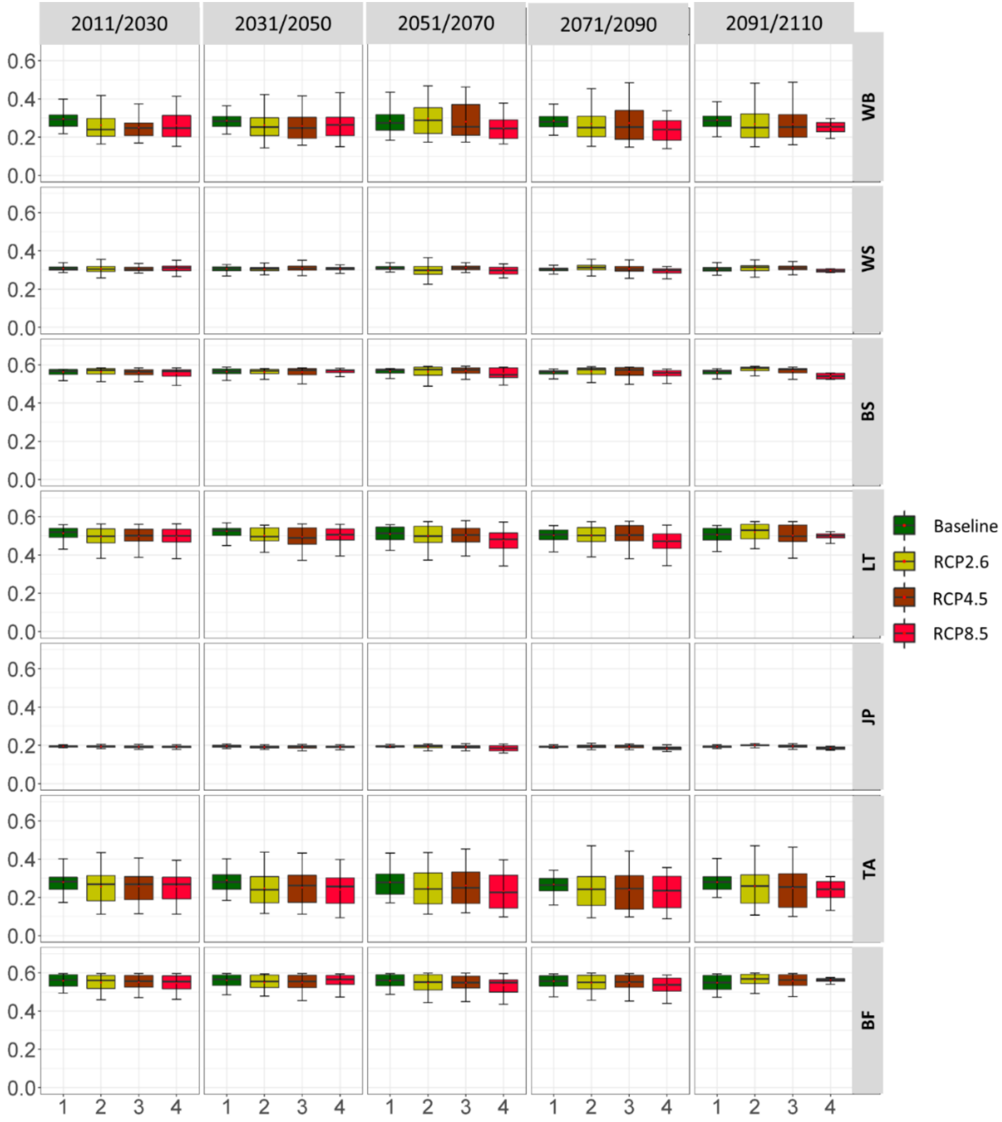

Figure A4.2.1(c) Est in MU1.

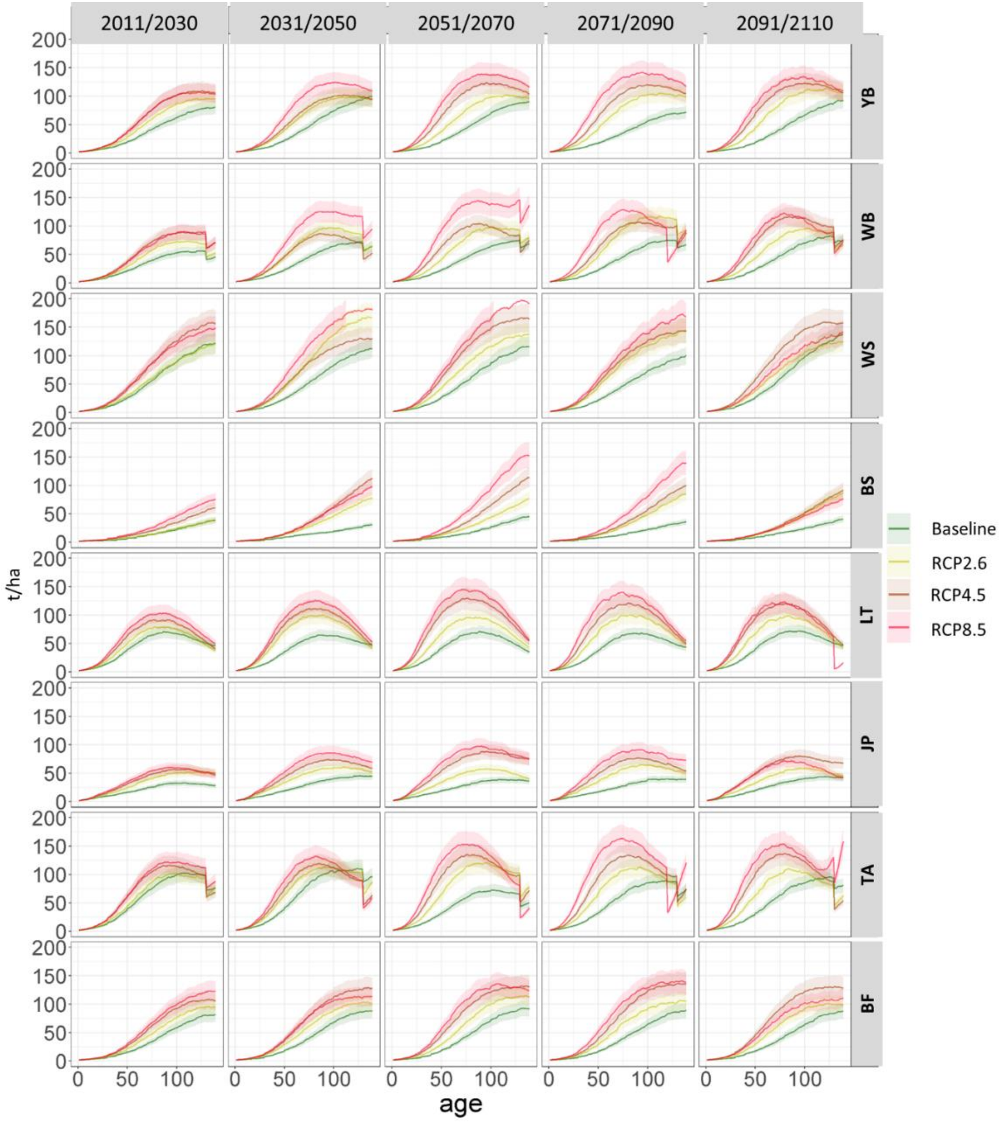

Figure A4.2.2(a) AGB in MU2.

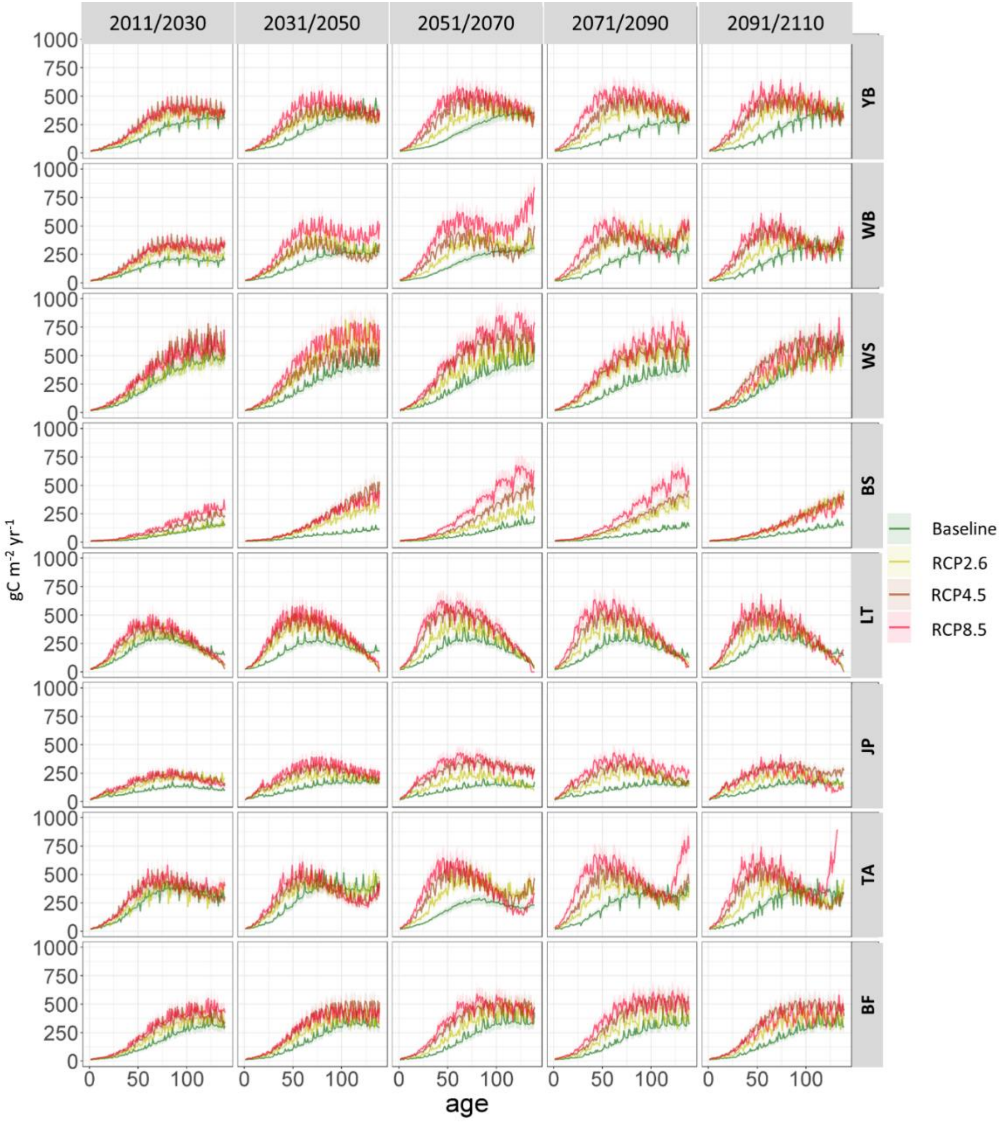

Figure A4.2.2(b) ANPP in MU2.

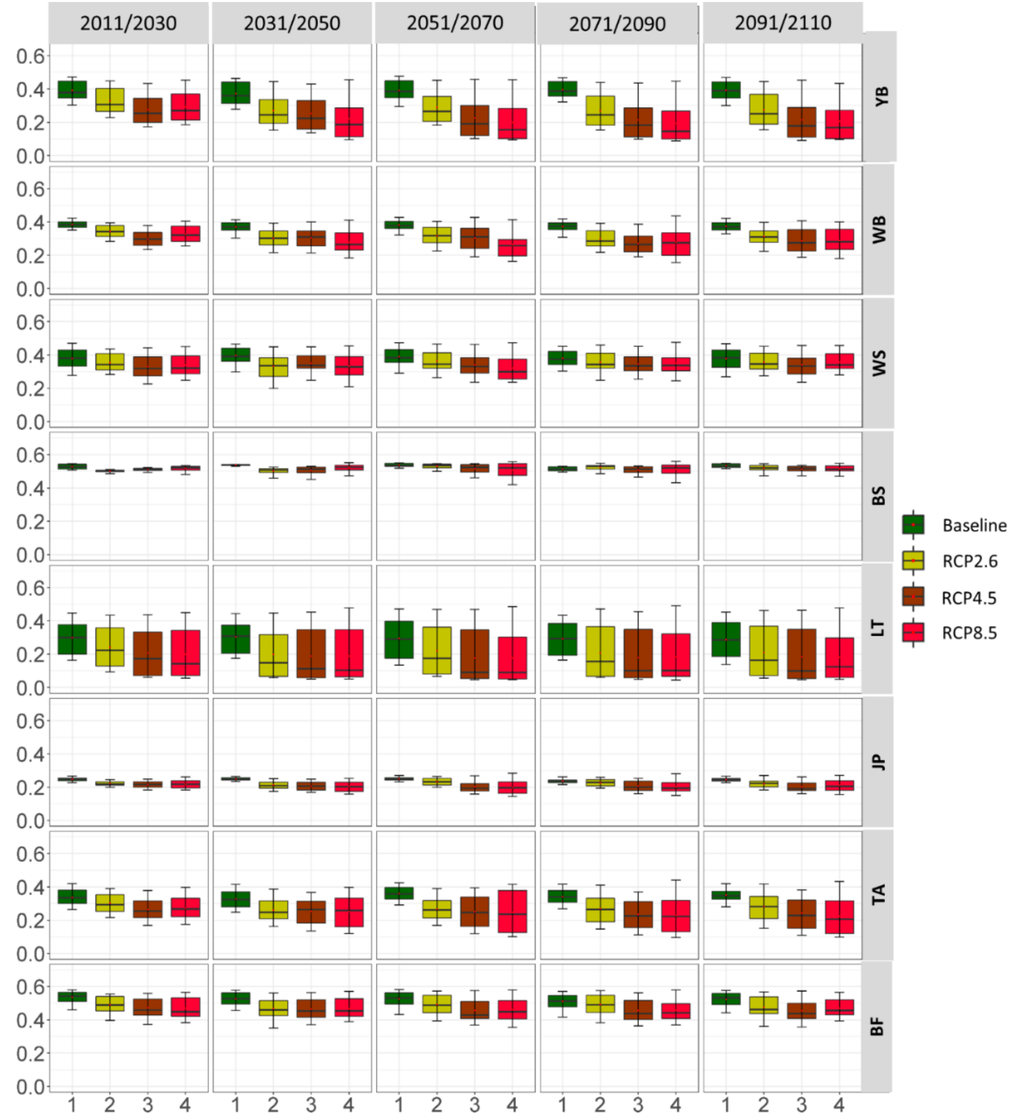

Figure A4.2.2 (c) Est in MU2.

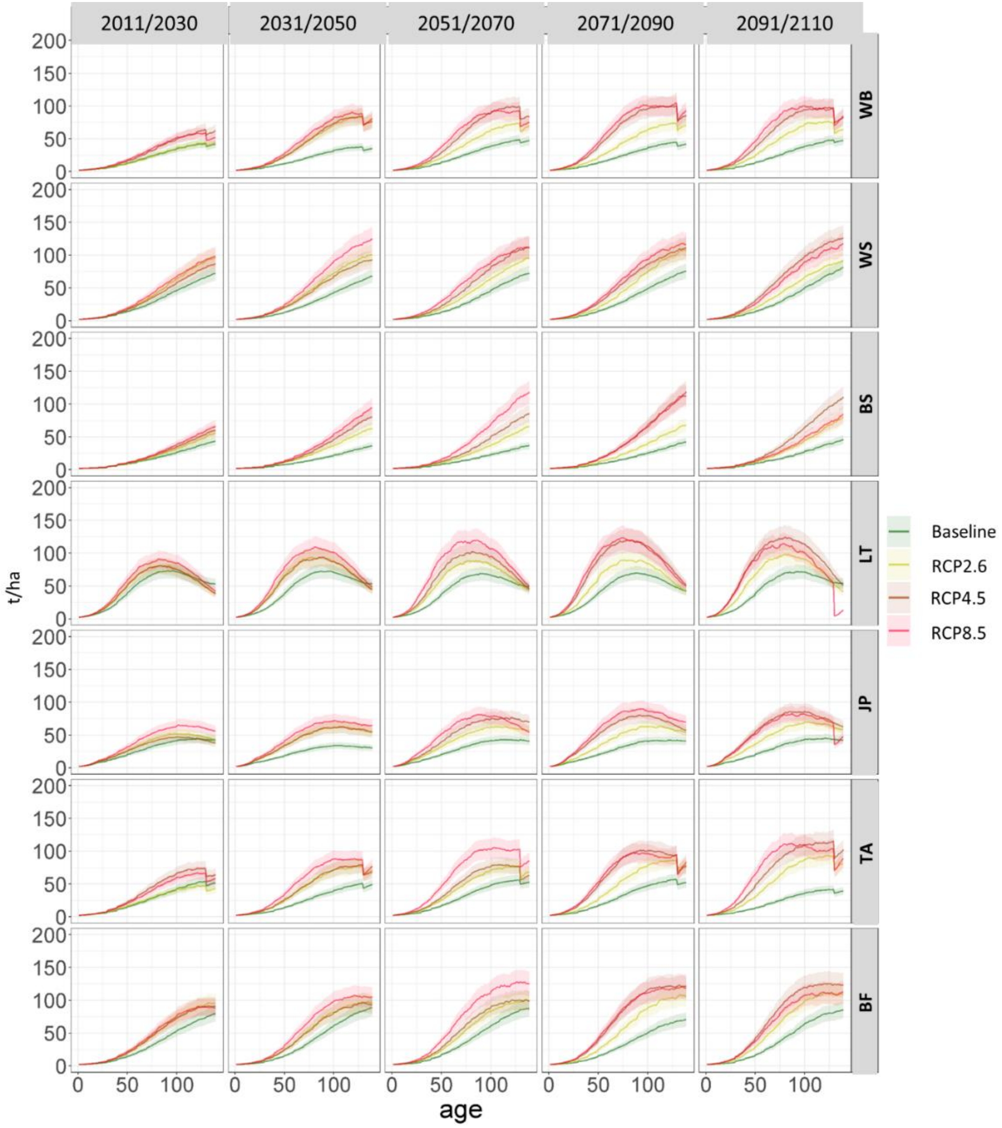

Figure A4.2.3(a) AGB in MU3.

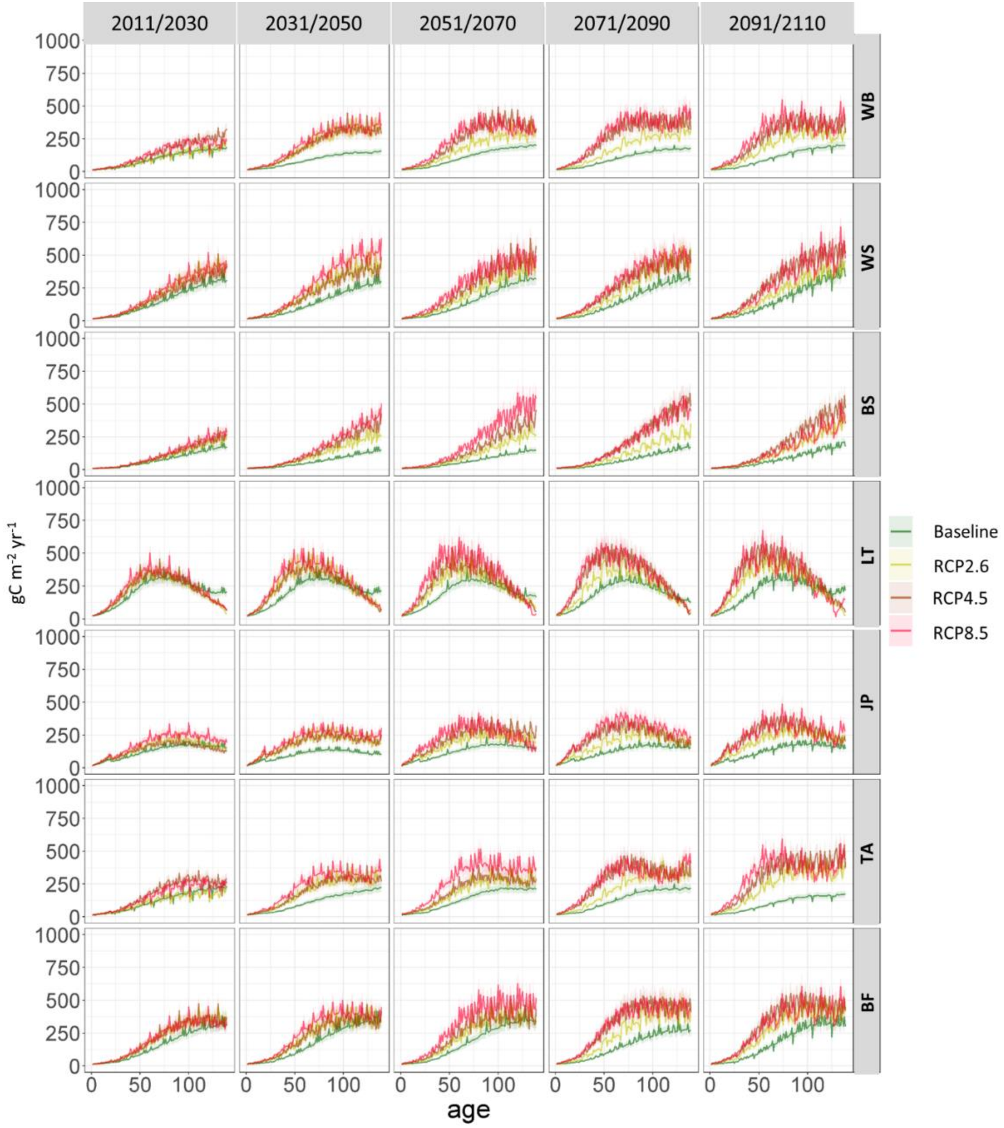

Figure A4.2.3 (b) ANPP in MU3.

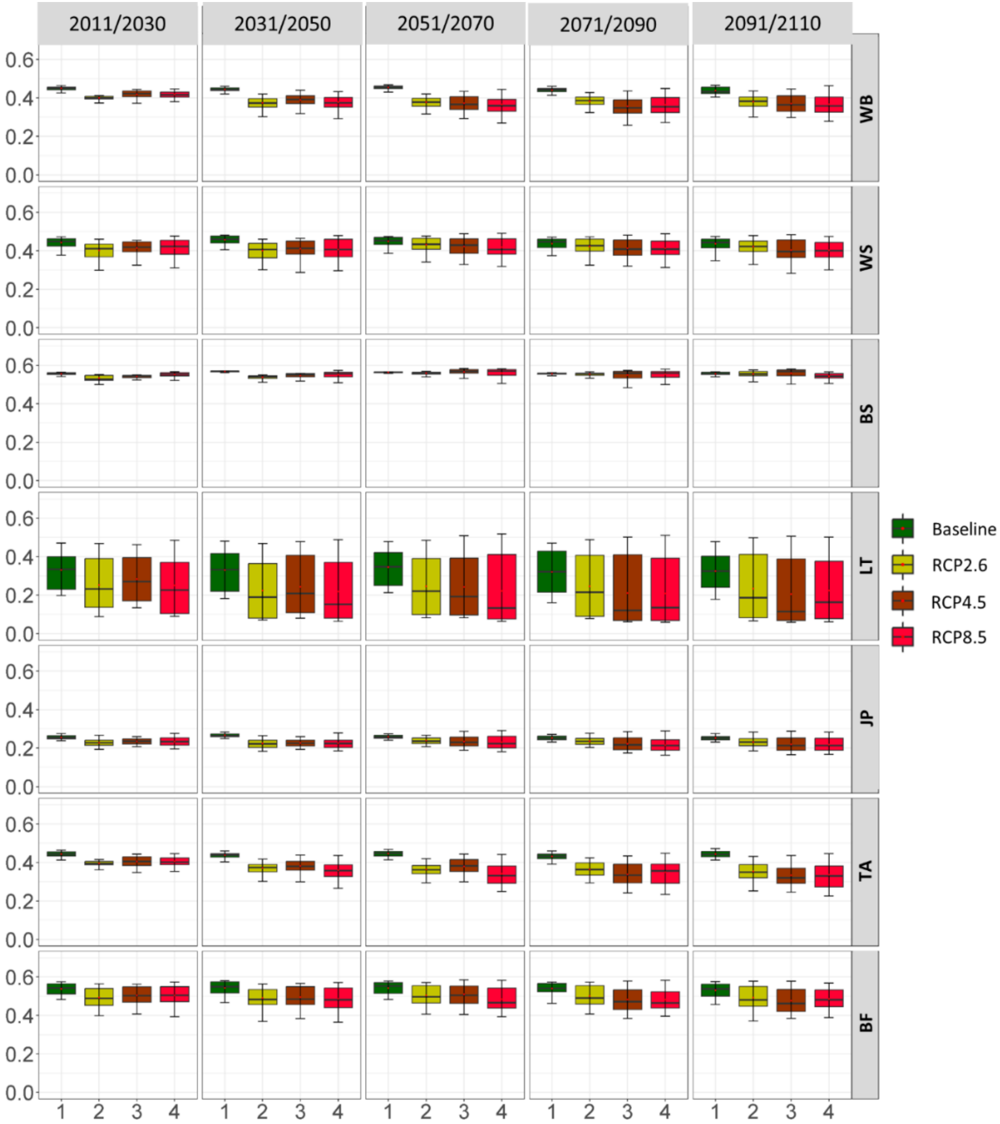

Figure A4.2.3 (c) Est in MU3.

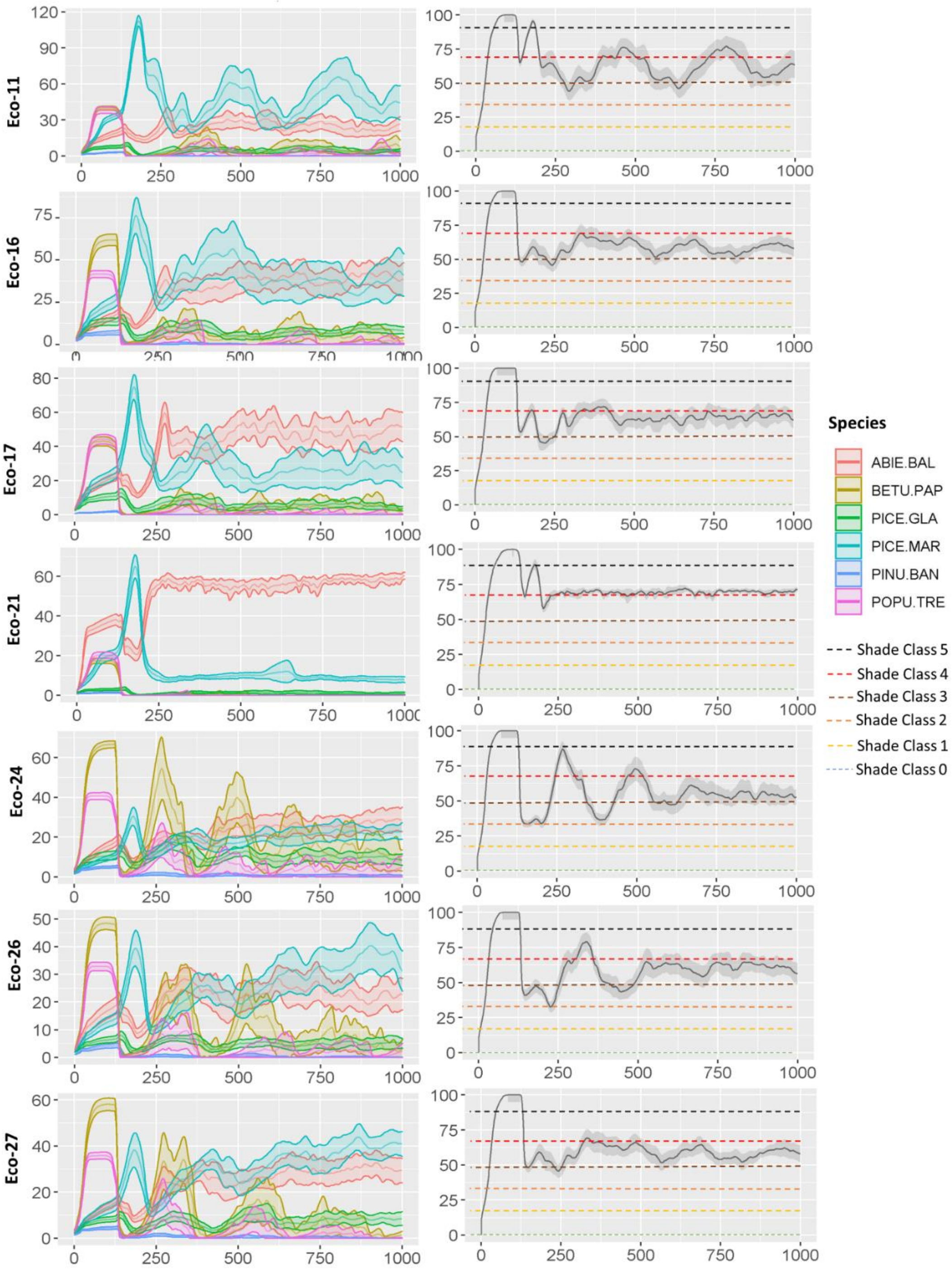

Figure A4.3.1 Verification of Succession emerging successional pathways: of each land-type (ecoregions) in MU1.

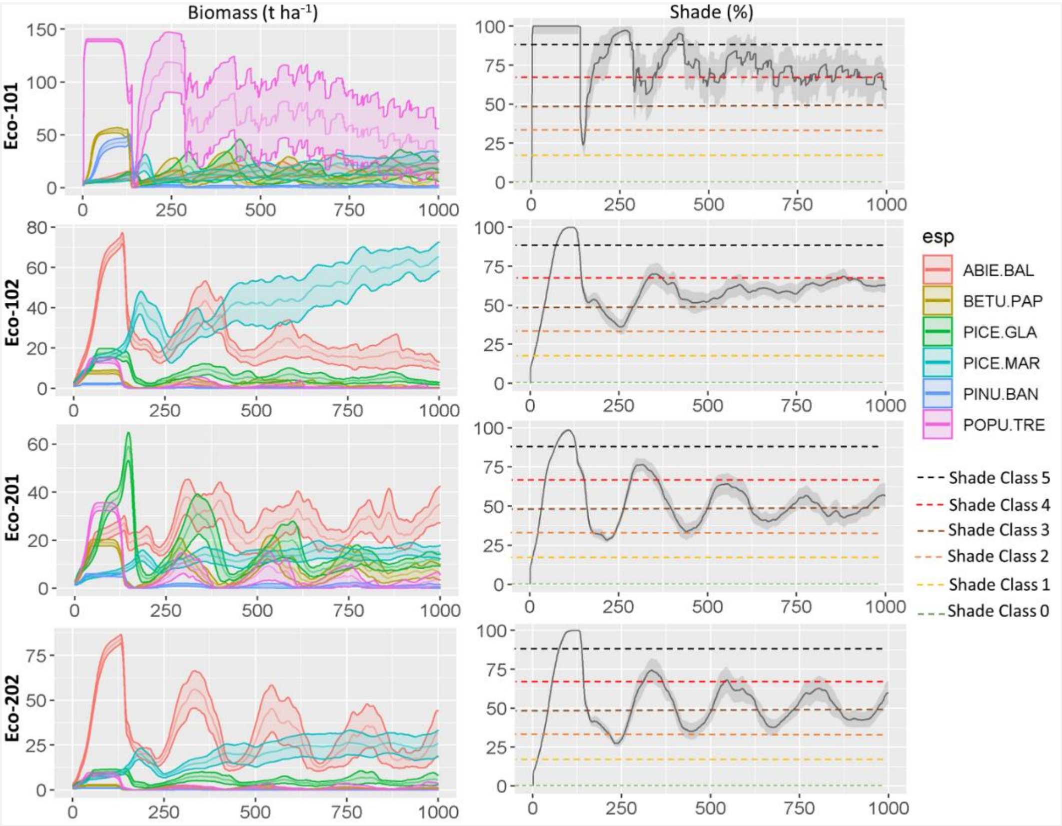

Figure A4.3.2 Verification of Succession emerging successional pathways: of each land-type (ecoregions) in MU2.

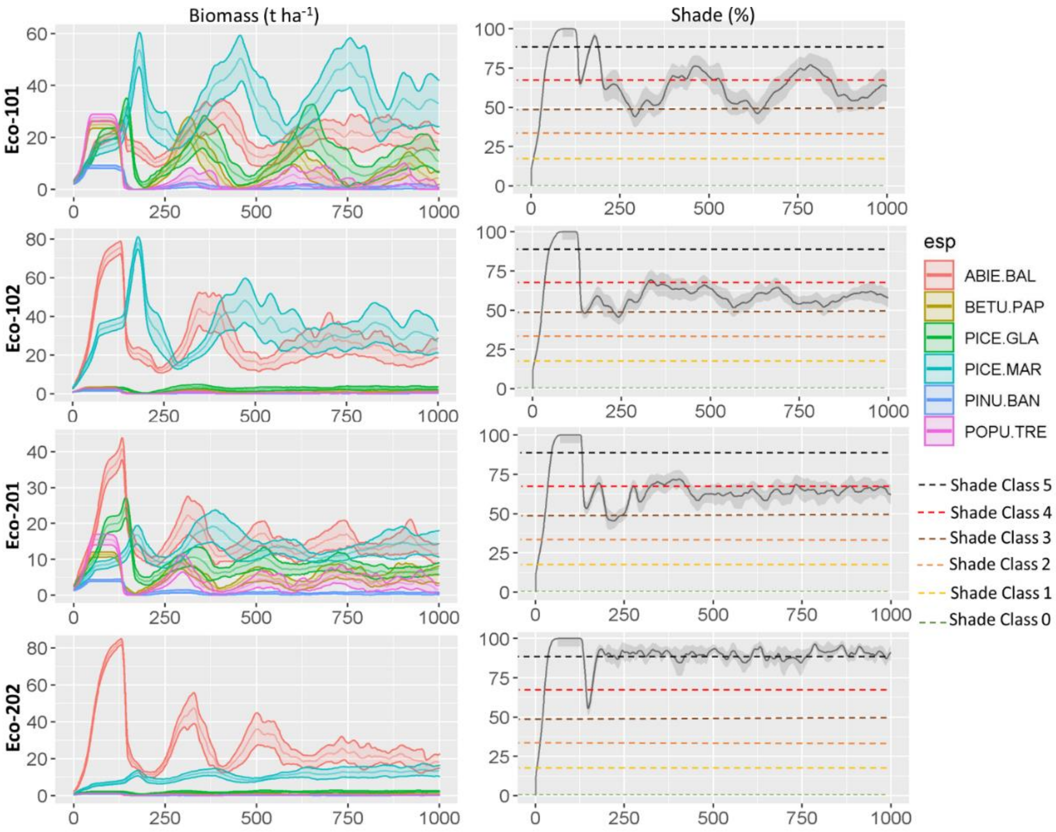

Figure A4.3.3 Verification of Succession emerging successional pathways: of each land-type (ecoregions) in MU3.

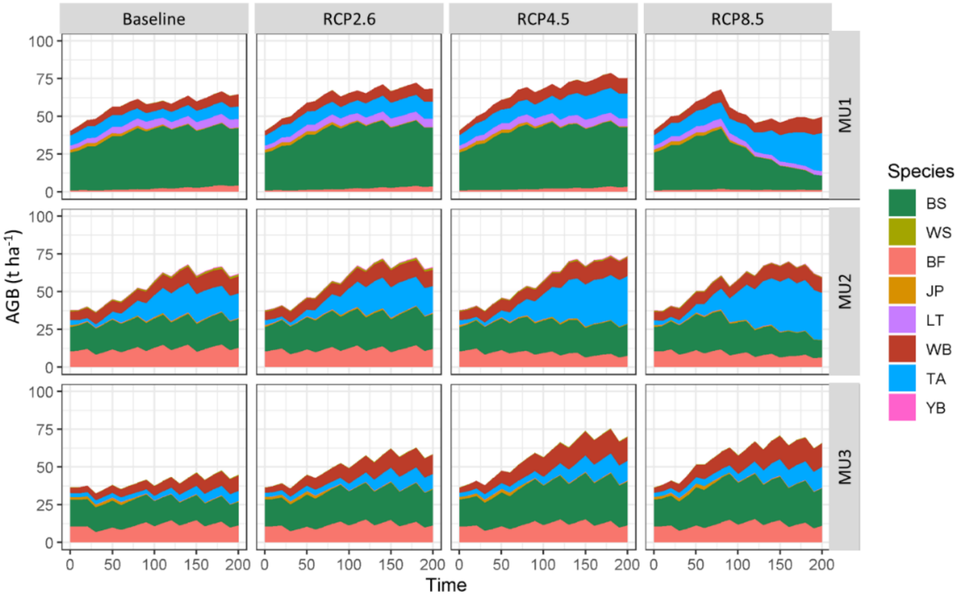

Figure A4.4 Projected changes above ground biomass AGB (t ha^{-1}) in tree species ((legend: BS: black spruce, WS: white spruce, BF: balsam fir, JP: jack pine, LT: larch tree, WB: white birch, TA: trembling aspen, YB: yellow birch), under S0 scenario and all climate change scenarios: baseline, RCP 2.6, RCP 4.5 and RCP 8.5.

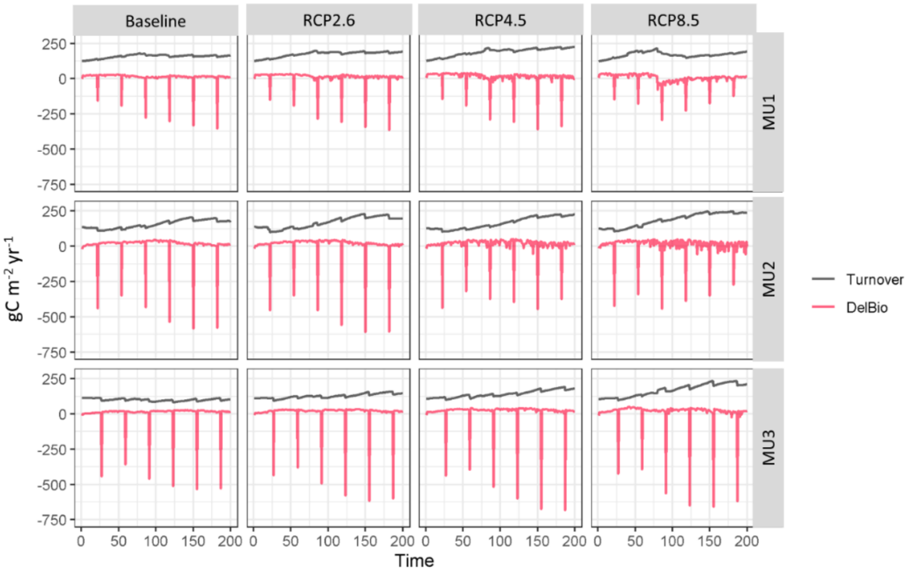

Figure A4.5 Annual change in biomass stocks (DelBio) under all disturbances, and annual turnover under natural dynamics scenario (S0) for all climate change scenarios.

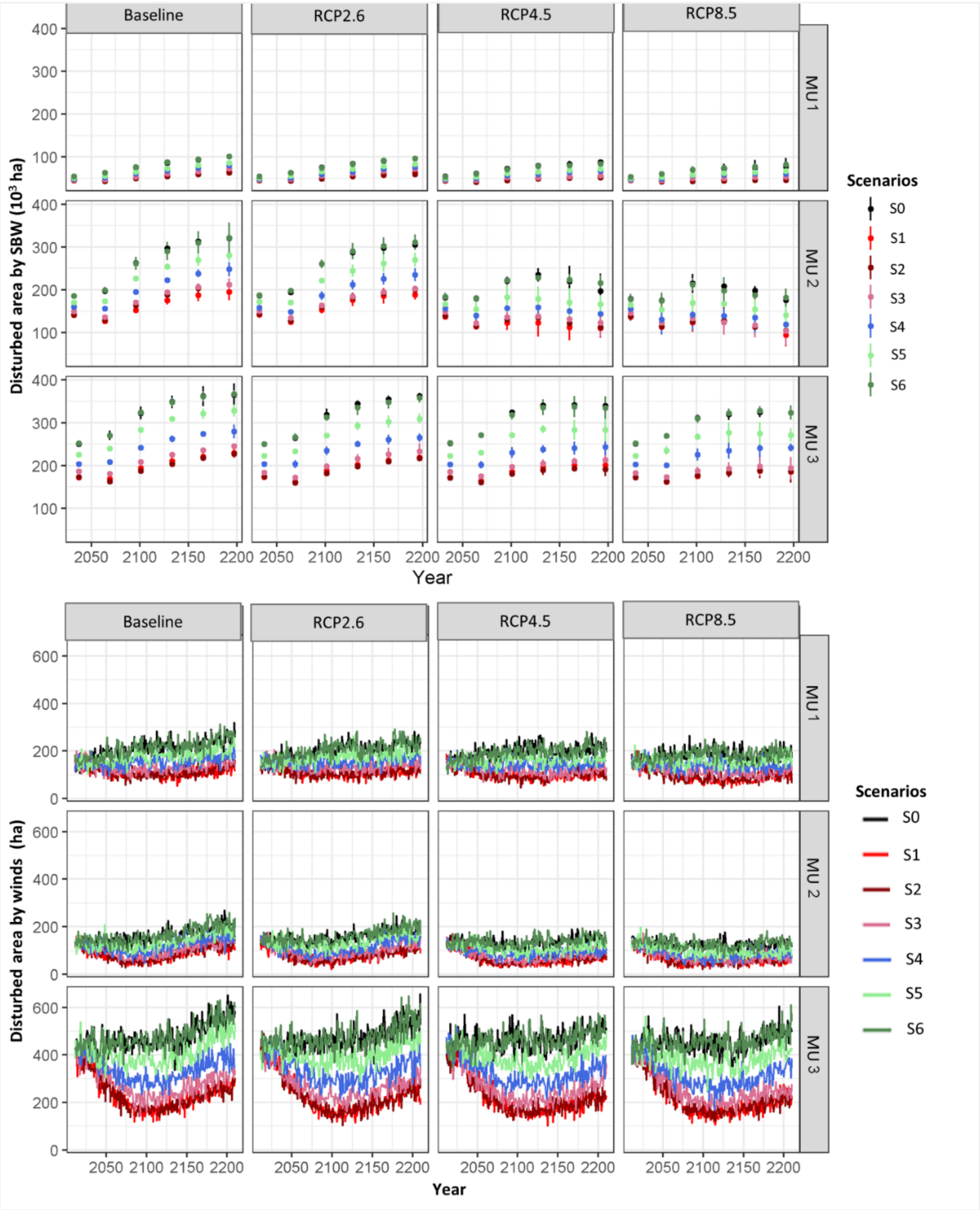

Figure A4.6 Annual affected area by winds and spruce budworm from 2010 (year 0) to 2210 (year 200).

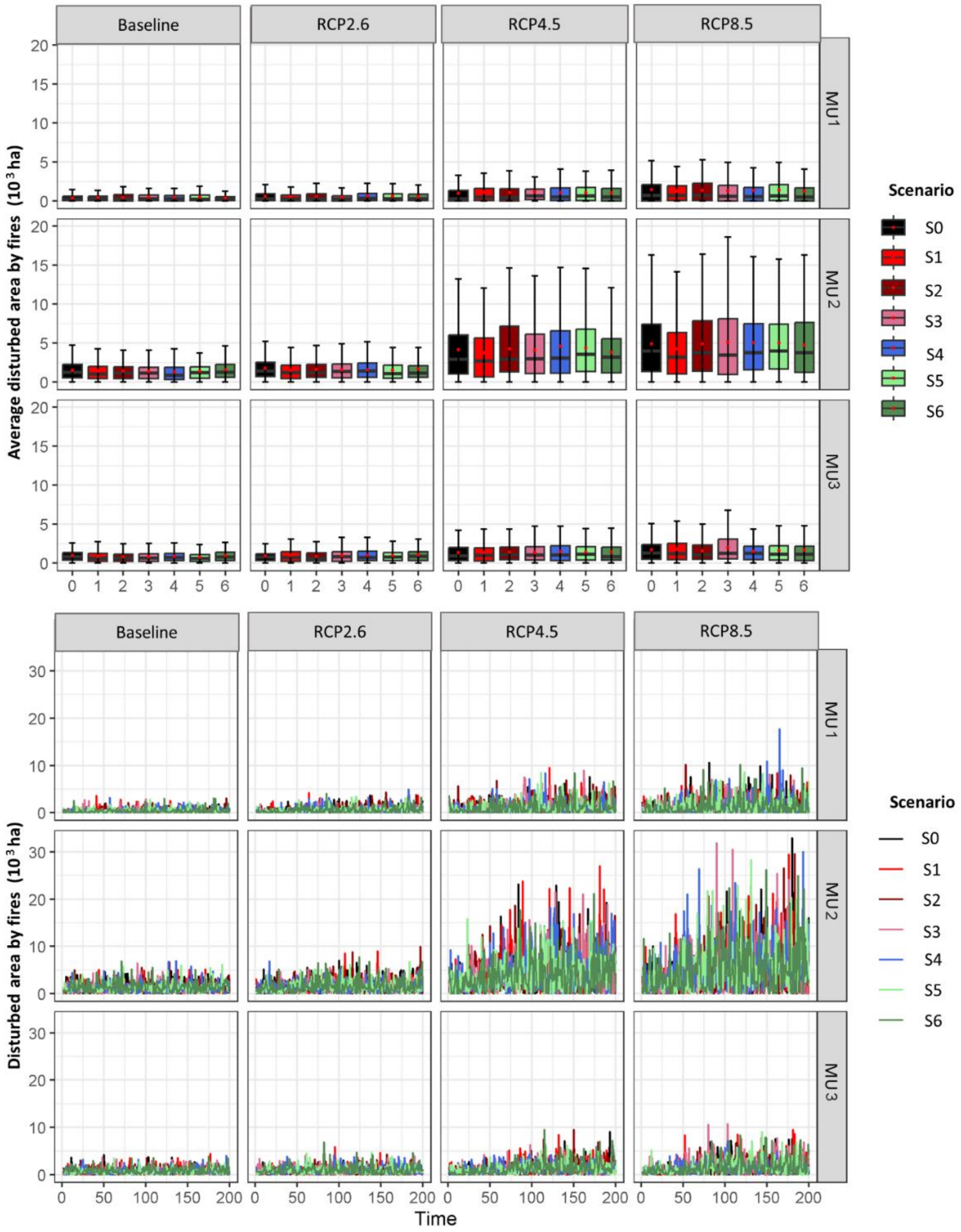

Figure A4.7 Boxplot of annual burnt area average for the entire simulation period (2010-2210), with the annual observed values from 2010 (year 0) to 2210 (year 200).

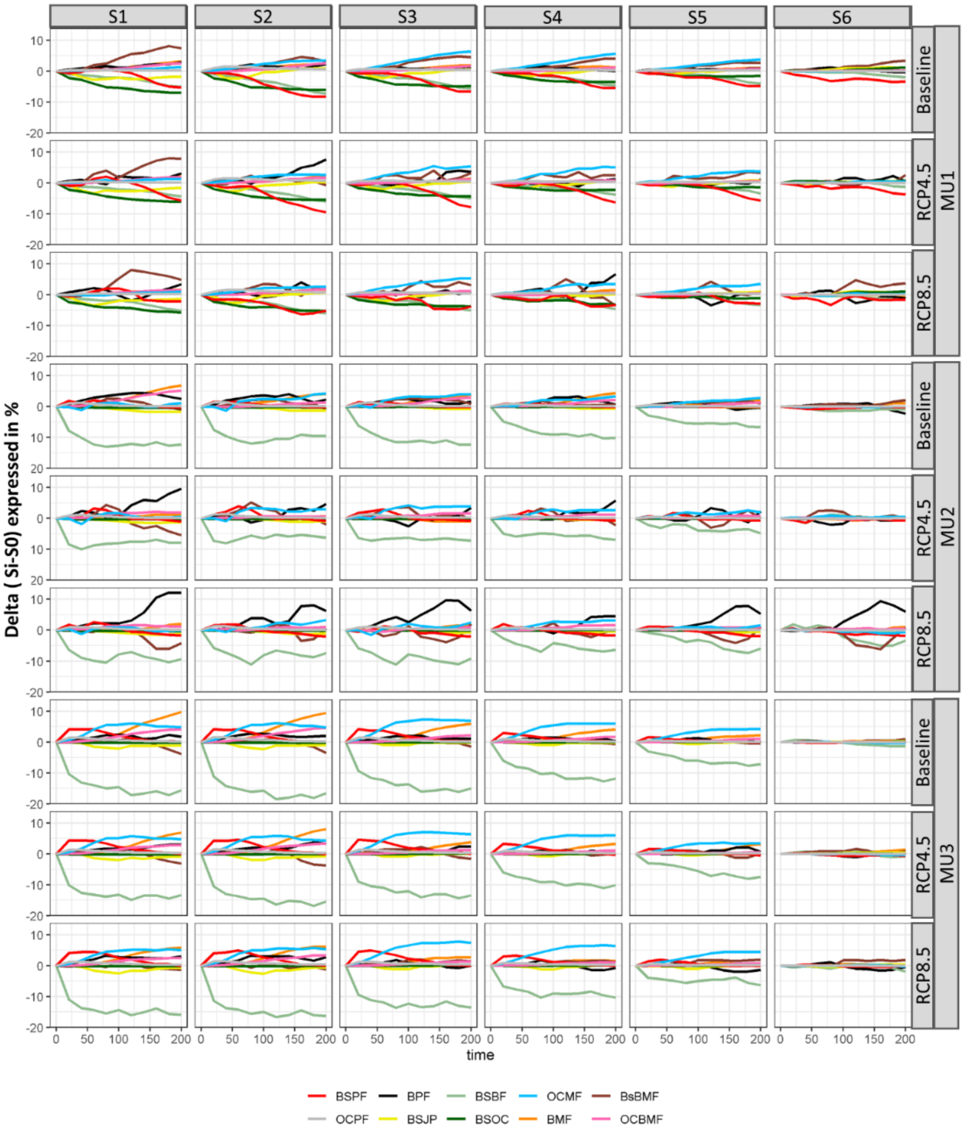

Figure A4.8 The difference (Δ; %) of composition percentage between management scenarios and natural evolution scenario (S0) in the three MU between 2010 (year 0) and 2210 (year 200) under the current climate, RCP4.5, and RCP8.5 scenarios. (BsPF: black spruce pure forests, OcPF: Other coniferous pure forests, BPF: broadleaves pure forests, BSJP: black spruce and jack pine, BSBF: black spruce and balsam fir, BSOC: black spruce and other coniferous, OCMF: Other coniferous mixed forests, BMF: broadleaves mixed forests; BsBMF: black spruce and broadleaves mixed forests, OCBMF: Other coniferous and broadleaves mixed forests).

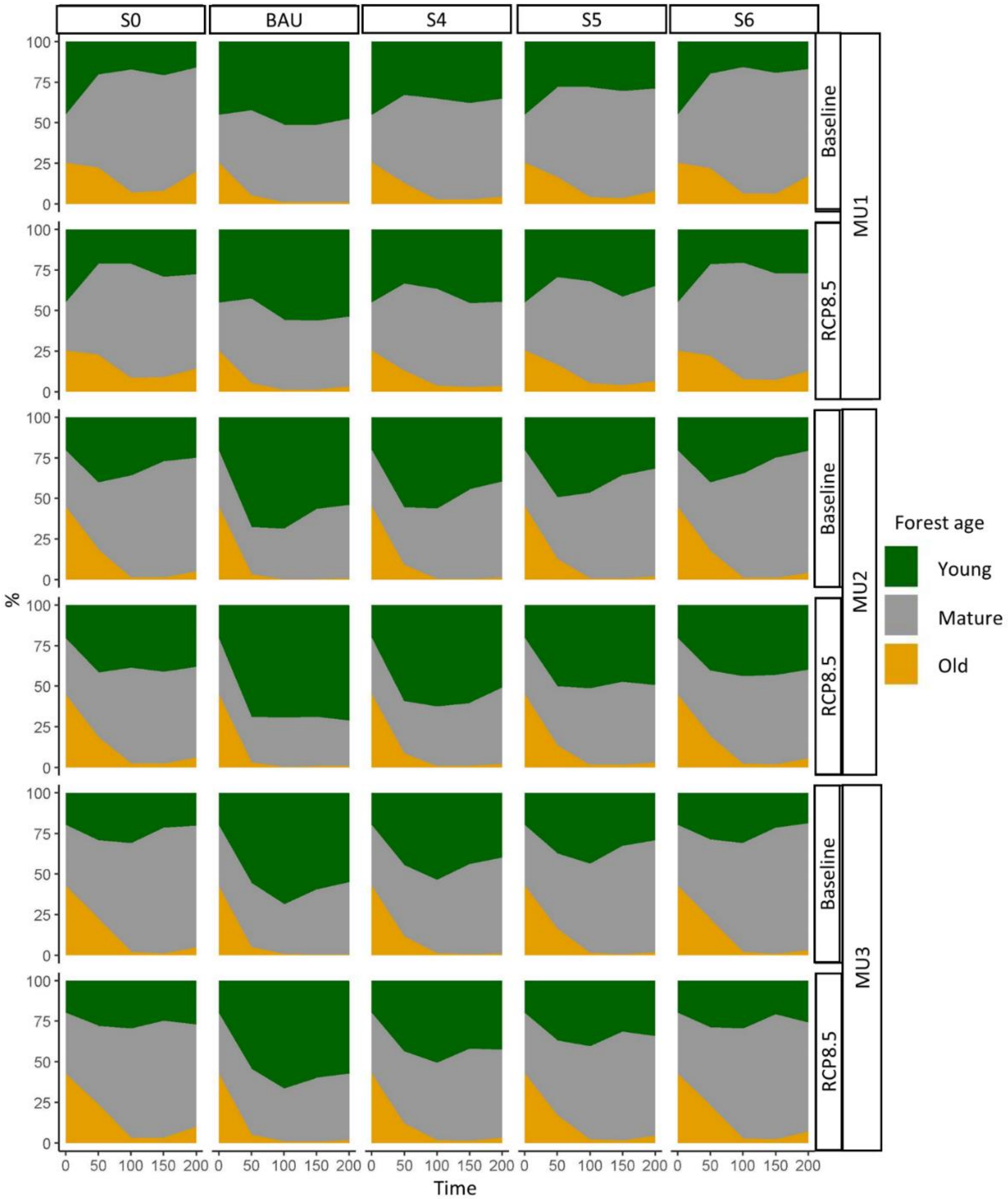

Figure A4.9 Age structure (young forest ≤40; 40<mature forest≤100; 100< old growth forest) under current climate and RCP8.5 scenarios, for natural evolution (S0; no-harvest), BAU, PC-based scenarios (S4, S5, S6).

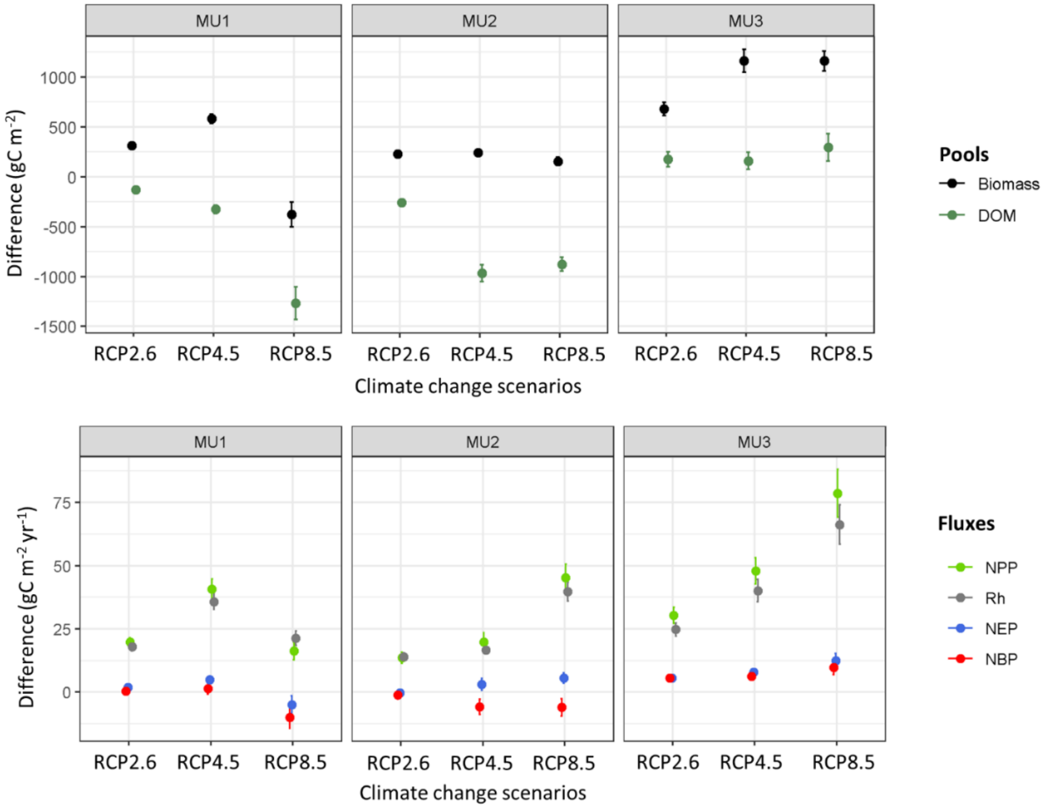

Figure A4.10 Differences in carbon pools and fluxes between RCP scenarios and current climate, under natural evolution (S0).

RÉFÉRENCES

Aakala, T., Remy, C. C., Arseneault, D., Morin, H., Girardin, M. P., Gennaretti, F., Navarro, L., Kuosmanen, N., Ali, A. A., & Boucher, É. (2023). Millennial-Scale Disturbance History of the Boreal Zone. In: Girona, M.M., Morin, H., Gauthier, S., Bergeron, Y. (eds) Boreal Forests in the Face of Climate Change, pp. 53-87. *Advances in Global Change Research*. Cham, Springer. https://doi.org/10.1007/978-3-031-15988-6_2.

Aber, J.D., & Federer, C.A. (1992). A generalized, lumped-parameter model of photosynthesis, evapotranspiration and net primary production in temperate and boreal forest ecosystems. *Oecologia*, 92(4), 463–474.

Achim, A., Moreau, G., Coops, N. C., Axelson, J. N., Barrette, J., Bédard, S., Byrne, K. E., Caspersen, J., Dick, A. R., D'Orangeville, L., Drolet, G., Eskelson, B. N. I., Filipescu, C. N., Flamand-Hubert, M., Goodbody, T. R. H., Griess, V. C., Hagerman, S. M., Keys, K., Lafleur, B., … White, J. C. (2022). The changing culture of silviculture. Forestry: *An International Journal of Forest Research*, 95(2), 143–152. https://doi.org/10.1093/forestry/cpab047.

Alvarez, S., Ortiz, C., Díaz-Pinés, E., & Rubio, A. (2016). Influence of tree species composition, thinning intensity and climate change on carbon sequestration in Mediterranean mountain forests: a case study using the CO2Fix model. *Mitigation and Adaptation Strategies for Global Change*, 21(7), 1045–1058.

Ameray, A., Bergeron, Y., Valeria, O., Girona, M., & Cavard, X. (2021). Ameray, A., Bergeron, Y., Valeria, O., Montoro Girona, M. and Cavard, X., 2021. Forest carbon management: A review of silvicultural practices and management strategies across boreal, temperate and tropical forests. *Current Forestry Reports*, pp.1-22. https://doi.org/10.1007/s40725-021-00151-w.

Ameray, A., Bergeron, Y., & Cavard, X. (2023a). Climate change may increase Quebec boreal forest productivity in high latitudes by shifting its current composition. *Frontiers in Forests and Global Change*, 6, 20. https://doi.org/10.3389/ffgc.2023.1020305.

Ameray, A., Bergeron, Y., & Cavard, X. (2023b). Modelling the potential of forest management to mitigate climate change in Eastern Canadian forests. *Scientific Reports*, 13(1), 14506.

Amiro, B. D., Todd, J. B., Wotton, B. M., Logan, K. A., Flannigan, M. D., Stocks, B. J., Mason, J. A., Martell, D. L., & Hirsch, K. G. (2001). Direct carbon emissions from Canadian forest fires, 1959-1999. Canadian Journal of Forest Research, 31(3), 512–525.Apps, M. J., Kurz, W. A., Luxmoore, R. J., Nilsson, L. O., Sedjo, R. A., Schmidt, R., Simpson, L. G., & Vinson, T. S. (1993). Boreal forests and tundra. *Water, Air, and Soil Pollution*, 70(1–4), 39–53.

Augustin, F., Girardin, M. P., Terrier, A., Grondin, P., Lambert, M.-C., Leduc, A., & Bergeron, Y. (2022). Projected changes in fire activity and severity feedback in the spruce–feather moss forest of western Quebec, Canada. *Trees, Forests and People*, 8, 100229. https://doi.org/https://doi.org/10.1016/j.tfp.2022.100229.

Auty, D., Achim, A., Bédard, P., & Pothier, D. (2014). StatSAW: modelling lumber product assortment using zero-inflated Poisson regression. *Canadian Journal of Forest Research*, 44(6), 638–647. https://doi.org/10.1139/cjfr-2013-0500.

Baltzer, J. L., Day, N. J., Walker, X. J., Greene, D., Mack, M. C., Alexander, H. D., & Johnstone, J. F. (2021). Increasing fire and the decline of fire adapted black spruce in the boreal forest. *Proceedings of the National Academy of Sciences*, 118(45), e2024872118. https://doi.org/10.1073/pnas.2024872118.

Batjes, N. H. (1996). Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, 47(2), 151–163.

Beck, P. S. A., Juday, G. P., Alix, C., Barber, V. A., Winslow, S. E., Sousa, E. E., Heiser, P., Herriges, J. D., & Goetz, S. J. (2011). Changes in forest productivity across Alaska consistent with biome shift. *Ecology Letters*, 14(4), 373–379. https://doi.org/https://doi.org/10.1111/j.1461-0248.2011.01598.x.

Bellassen, V., & Luyssaert, S. (2014). Carbon sequestration: Managing forests in uncertain times. *Nature*, 506(7487), 153–155.

Benomar, L., Lamhamedi, M. S., Pepin, S., Rainville, A., Lambert, M.-C., Margolis, H. A., Bousquet, J., & Beaulieu, J. (2018). Thermal acclimation of photosynthesis and respiration of southern and northern white spruce seed sources tested along a regional climatic gradient indicates limited potential to cope with temperature warming. *Annals of Botany*, 121(3), 443–457.

Bergeron, Y. (2000). Species and stand dynamics in the mixed woods of quebec's southern boreal forest. *Ecology*, 81(6), 1500–1516. https://doi.org/10.2307/177302 .

Bergeron, Y., Chen, H. Y. H., Kenkel, N. C., Leduc, A. L., & Macdonald, S. E. (2014). Boreal mixedwood stand dynamics: ecological processes underlying multiple pathways. *The Forestry Chronicle,* 90(02), 202–213. https://doi.org/10.5558/tfc2014-039.

Bergeron, Y., Cyr, D., Drever, C., Flannigan, M., Gauthier, S., Kneeshaw, D., Lauzon, È., Leduc, A., Goff, H., Lesieur, D., & Logan, K. (2006). Past, current, and future fire frequencies in Quebec's commercial forests: Implications for the cumulative effects of harvesting and fire on age-class structure and natural disturbance-based management. *Canadian Journal of Forest Research*, 36, 2737–2744. https://doi.org/10.1139/X06-177.

Bergeron, Y., Cyr, D., Girardin, M. P., & Carcaillet, C. (2010). Will climate change drive 21st century burn rates in Canadian boreal forest outside of its natural variability: Collating global climate model experiments with sedimentary charcoal data. *International Journal of Wildland Fire*, 19(8), 1127–1139. https://doi.org/10.1071/WF09092.

Bergeron, Y., Gauthier, S., Flannigan, M., & Kafka, V. (2004). Fire regimes at the transition between mixedwood and coniferous boreal forest in northwestern quebec. *Ecology*, 85(7), 1916–1932. https://doi.org/10.1890/02-0716.

Bergeron, Y., Gauthier, S., Kafka, V., Lefort, P., & Lesieur, D. (2001). Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. *Canadian Journal of Forest Research*, 31(3), 384–391.

Bergeron, Y., Harvey, B., Leduc, A., & Gauthier, S. (1999). Forest management guidelines based on natural disturbance dynamics: stand-and forest-level considerations. *The Forestry Chronicle*, 75(1), 49–54.

Bergeron, Y., Irulappa Pillai Vijayakumar, D. B., Ouzennou, H., Raulier, F., Leduc, A., & Gauthier, S. (2017). Projections of future forest age class structure under the influence of fire and harvesting: implications for forest management in the boreal forest of eastern Canada. Forestry: *An International Journal of Forest Research*, 90(4), 485–495. https://doi.org/10.1093/forestry/cpx022.

Bergeron, Y., & Leduc, A. (1998). Relationships between change in fire frequency and mortality due to spruce budworm outbreak in the southeastern Canadian boreal forest. *Journal of Vegetation Science*, 9(4), 492–500. https://doi.org/10.2307/3237264.

Bergeron, Y., Leduc, A., Harvey, B. D., & Gauthier, S. (2002). Natural fire regime: a guide for sustainable management of the Canadian boreal forest. *Silva Fennica*, 36(1), 81–95.

Bhatti, J. S., Apps, M. J., & Jiang, H. (2002). Influence of nutrients, disturbances and site conditions on carbon stocks along a boreal forest transect in central Canada. *Plant and Soil*, 242(1), 1–14. https://doi.org/10.1023/A:1019670619316.

Bouchard, M., Pothier, D., & Ruel, J.-C. (2009a). Stand-replacing windthrow in the boreal forests of eastern Quebec. *Canadian Journal of Forest Research*, 39(2), 481–487.

Bouchard, M., Pothier, D., & Ruel, J.-C. (2009b). Stand-replacing windthrow in the boreal forest of eastern Quebec. *Canadian Journal of Forest Research*, 39, 481–487. https://doi.org/10.1139/X08-174.

Bouchard, M., Régnière, J., & Therrien, P. (2017). Bottom-up factors contribute to large-scale synchrony in spruce budworm populations. *Canadian Journal of Forest Research*, 48(3), 277–284.

Boucher, J.-F., Tremblay, P., Gaboury, S., & Villeneuve, C. (2012). Can boreal afforestation help offset incompressible GHG emissions from Canadian industries? *Process Safety and Environmental Protection*, 90(6), 459–466. https://doi.org/https://doi.org/10.1016/j.psep.2012.10.011.

Boucher, Y., Perrault-Hébert, M., Fournier, R., Drapeau, P., & Auger, I. (2017). Cumulative patterns of logging and fire (1940–2009): consequences on the structure of the eastern Canadian boreal forest. *Landscape Ecology*, 32(2), 361–375. https://doi.org/10.1007/s10980-016-0448-9.

Boudreau, J., Nelson, R. F., Margolis, H. A., Beaudoin, A., Guindon, L., & Kimes, D. S. (2008). Regional aboveground forest biomass using airborne and spaceborne LiDAR in Québec. *Remote Sensing of Environment*, 112(10), 3876-3890.

Boulanger, Y., Arseneault, D., Boucher, Y., Gauthier, S., Cyr, D., Taylor, A. R., Price, D. T., & Dupuis, S. (2019). Climate change will affect the ability of forest management to reduce gaps between current and presettlement forest composition in southeastern Canada. *Landscape Ecology*, 34(1), 159–174. https://doi.org/10.1007/s10980-018-0761-6.

Boulanger, Y., Arseneault, D., Morin, H., Jardon, Y., Bertrand, P., & Dagneau, C. (2012). Dendrochronological reconstruction of spruce budworm (Choristoneura fumiferana) outbreaks in southern Quebec for the last 400 years1This article is one of a selection of papers from the 7th International Conference on Disturbance Dynamics in Boreal Forest. *Canadian Journal of Forest Research*, 42(7), 1264–1276. https://doi.org/10.1139/x2012-069.

Boulanger, Y., Gauthier, S., & Burton, P. J. (2014). A refinement of models projecting future Canadian fire regimes using homogeneous fire regime zones. *Canadian*

Journal of Forest Research, 44(4), 365–376.

Boulanger, Y., Pascual, J., Bouchard, M., D'Orangeville, L., Périé, C., & Girardin, M. P. (2022). Multi-model projections of tree species performance in Quebec, Canada under future climate change. *Global Change Biology*, 28(5), 1884–1902. https://doi.org/10.1111/gcb.16014.

Boulanger, Y., Puigdevall, J. P., Bélisle, A. C., Bergeron, Y., Brice, M.-H., Cyr, D., De Grandpré, L., Fortin, D., Gauthier, S., Grondin, P., Labadie, G., Leblond, M., Marchand, M., Splawinski, T. B., St-Laurent, M.-H., Thiffault, E., Tremblay, J. A., & Yamasaki, S. H. (2023). A regional integrated assessment of the impacts of climate change and of the potential adaptation avenues for Quebec's forests. *Canadian Journal of Forest Research*. 53(8): 556-578. https://doi.org/10.1139/cjfr-2022-0282.

Boulanger, Y., & Puigdevall, P. J. (2021). Boreal forests will be more severely affected by projected anthropogenic climate forcing than mixedwood and northern hardwood forests in eastern Canada. *Landscape Ecology*, 36(6), 1725–1740. https://doi.org/10.1007/s10980-021-01241-7.

Boulanger, Y., Taylor, A. R., Price, D. T., Cyr, D., McGarrigle, E., Rammer, W., Sainte-Marie, G., Beaudoin, A., Guindon, L., & Mansuy, N. (2017). Climate change impacts on forest landscapes along the Canadian southern boreal forest transition zone. *Landscape Ecology*, 32(7), 1415–1431. https://doi.org/10.1007/s10980-016-0421-7.

Brecka, A. F. J., Boulanger, Y., Searle, E. B., Taylor, A. R., Price, D. T., Zhu, Y., Shahi, C., & Chen, H. Y. H. (2020). Sustainability of Canada's forestry sector may be compromised by impending climate change. *Forest Ecology and Management*, 474, 118352. https://doi.org/https://doi.org/10.1016/j.foreco.2020.118352.

Campbell, J., Alberti, G., Martin, J., & Law, B. E. (2009). Carbon dynamics of a ponderosa pine plantation following a thinning treatment in the northern Sierra Nevada. *Forest Ecology and Management*, 257(2), 453–463.

Carroll, M., Milakovsky, B., Finkral, A., Evans, A., & Ashton, M. S. (2012). Managing carbon sequestration and storage in temperate and boreal forests. In M. S. Ashton, M. L. Tyrrell, D. Spalding, & B. Gentry (Eds.), Managing forest carbon in a changing climate (pp. 205–226). Dordrecht city, The Netherlands: Springer Science & Business Media. https://doi.org/10.1007/978-94-007-2232-3_10.

Castro, M., Ameray, A., & Castro, J. P. (2020). A new approach to quantify grazing pressure under mediterranean pastoral systems using GIS and remote sensing.

International Journal of Remote Sensing, 41(14), 5371–5387.

Cavard, X., Bergeron, Y., Chen, H. Y. H., & Pare, D. (2010). Mixed-species effect on tree aboveground carbon pools in the east-central boreal forests. *Canadian Journal of Forest Research*, 40(1), 37–47.

Cavard, X., Bergeron, Y., Chen, H. Y. H., Paré, D., Laganière, J., & Brassard, B. (2011). Competition and facilitation between tree species change with stand development. *Oikos*, 120(11), 1683–1695. https://doi.org/10.1111/j.1600-0706.2011.19294.x.

Charney, N. D., Babst, F., Poulter, B., Record, S., Trouet, V. M., Frank, D., Enquist, B. J., & Evans, M. E. K. (2016). Observed forest sensitivity to climate implies large changes in 21st century North American forest growth. *Ecology Letters*, 19(9), 1119–1128. https://doi.org/https://doi.org/10.1111/ele.12650.

Chen, J. M., Rich, P. M., Gower, S. T., Norman, J. M., & Plummer, S. (1997). Leaf area index of boreal forests: Theory, techniques, and measurements. *Journal of Geophysical Research: Atmospheres*, 102(D24), 29429–29443.

Collins, M., Knutti, R., Arblaster, J., Dufresne, J.-L., Fichefet, T., Friedlingstein, P., Gao, X., Gutowski, W. J., Johns, T., & Krinner, G. (2013). Long-term climate change: projections, commitments and irreversibility. In Climate change 2013-The physical science basis: Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge city, UK, pp. 1029–1136.

Cook, B. D., Bolstad, P. V, Martin, J. G., Heinsch, F. A., Davis, K. J., Wang, W., Desai, A. R., & Teclaw, R. M. (2008). Using Light-Use and Production Efficiency Models to Predict Photosynthesis and Net Carbon Exchange During Forest Canopy Disturbance. *Ecosystems*, 11(1), 26–44. https://doi.org/10.1007/s10021-007-9105-0.

Covington, W. W. (1981). Changes in Forest Floor Organic Matter and Nutrient Content Following Clear Cutting in Northern Hardwoods. *Ecology*, 62(1), 41–48. https://doi.org/10.2307/1936666.

Curiel, Y, J., Baldocchi, D. D., Gershenson, A., Goldstein, A., Misson, L., & Wong, S. (2007). Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture. *Global Change Biology*, 13(9), 2018–2035. https://doi.org/10.1111/j.1365-2486.2007.01415.x.

Cyr, D., Splawinski, T. B., Pascual Puigdevall, J., Valeria, O., Leduc, A., Thiffault, N., Bergeron, Y., & Gauthier, S. (2021). Mitigating post-fire regeneration failure in boreal landscapes with reforestation and variable retention harvesting: At what cost? *Canadian Journal of Forest Research*, 52(4), 568–581. https://doi.org/10.1139/cjfr-2021-0180.

D'Orangeville, L., Houle, D., Duchesne, L., Phillips, R. P., Bergeron, Y., & Kneeshaw, D. (2018). Beneficial effects of climate warming on boreal tree growth may be transitory. *Nature Communications*, 9(1), 3213. https://doi.org/10.1038/s41467-018-05705-4.

Davidson, E. A., & Janssens, I. A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, 440(7081), 165–173. https://doi.org/10.1038/nature04514.

De Bruijn, A., Gustafson, E. J., Sturtevant, B. R., Foster, J. R., Miranda, B. R., Lichti, N. I., & Jacobs, D. F. (2014). Toward more robust projections of forest landscape dynamics under novel environmental conditions: embedding PnET within LANDIS-II. *Ecological Modelling*, 287, 44–57.

De Grandpré, L., Waldron, K., Bouchard, M., Gauthier, S., Beaudet, M., Ruel, J.-C., Hébert, C., & Kneeshaw, D. D. (2018). Incorporating insect and wind disturbances in a natural disturbance-based management framework for the boreal forest. *Forests*, 9(8), 471.

Deluca, T. H., & Boisvenue, C. (2012). Boreal forest soil carbon: distribution, function and modelling. Forestry: *An International Journal of Forest Research*, 85(2), 161–184. https://doi.org/10.1093/forestry/cps003.

Dhital, N., Raulier, F., Bernier, P. Y., Lapointe-Garant, M.-P., Berninger, F., & Bergeron, Y. (2015). Adaptation potential of ecosystem-based management to climate change in the eastern Canadian boreal forest. *Journal of Environmental Planning and Management*, 58(12), 2228–2249. https://doi.org/10.1080/09640568.2014.978079.

Dixon, R. (2009). Silvicultural options to conserve and sequester carbon in forest systems: Preliminary economic assessment. *Critical Reviews in Environmental Science and Technology*, 27(sup001), 139–149. https://doi.org/10.1080/10643389709388515.

Dixon, R. K., Solomon, A. M., Brown, S., Houghton, R. A., Trexier, M. C., & Wisniewski, J. (1994). Carbon Pools and Flux of Global Forest Ecosystems. *Science*, 263(5144), 185-190.

Don, A., Bärwolff, M., Kalbitz, K., Andruschkewitsch, R., Jungkunst, H. F., & Schulze, E.-D. (2012). No rapid soil carbon loss after a windthrow event in the

High Tatra. *Forest Ecology and Management*, 276, 239–246.

Dos Santos, L. T., Marra, D. M., Trumbore, S., De Camargo, P. B., Negrón-Juárez, R. I., Lima, A. J. N., Ribeiro, G. H. P. M., Dos Santos, J., & Higuchi, N. (2016). Windthrows increase soil carbon stocks in a central Amazon forest. *Biogeosciences*, 13(4), 1299–1308.

Duchesne, L., Houle, D., Ouimet, R., Lambert, M.-C., & Logan, T. (2016). Aboveground carbon in Quebec forests: stock quantification at the provincial scale and assessment of temperature, precipitation and edaphic properties effects on the potential stand-level stocking. *PeerJ*, 4, e1767.

Duchesne, L., & Ouimet, R. (2021). Digital mapping of soil texture in ecoforest polygons in Quebec, Canada. *PeerJ*, 9, e11685.

Dulamsuren, C. (2021). Organic carbon stock losses by disturbance: Comparing broadleaved pioneer and late-successional conifer forests in Mongolia’s boreal forest. *Forest Ecology and Management*, 499, 119636. https://doi.org/https://doi.org/10.1016/j.foreco.2021.119636

Dusenge, M. E., Madhavji, S., & Way, D. A. (2020). Contrasting acclimation responses to elevated CO2 and warming between an evergreen and a deciduous boreal conifer. *Global Change Biology*, 26(6), 3639–3657.

Duveneck, M. J., & Thompson, J. R. (2019). Social and biophysical determinants of future forest conditions in New England: effects of a modern land-use regime. *Global Environmental Change*, 55, 115–129.

Dymond, C., Beukema, S., Nitschke, C. R., Coates, K. D., & Scheller, R. M. (2016). Carbon sequestration in managed temperate coniferous forests under climate change. *Biogeosciences*, 13, no. 6 (2016): 1933-1947.

Dymond, C., Giles-Hansen, K., & Asante, P. (2020). The forest mitigation-adaptation nexus: Economic benefits of novel planting regimes. *Forest Policy and Economics*, 113, 102124.

Dymond, C., Neilson, E. T., Stinson, G., Porter, K., MacLean, D. A., Gray, D. R., Campagna, M., & Kurz, W. A. (2010). Future Spruce Budworm Outbreak May Create a Carbon Source in Eastern Canadian Forests. *Ecosystems*, 13(6), 917–931. https://doi.org/10.1007/s10021-010-9364-z.

Eyvindson, K., Duflot, R., Triviño, M., Blattert, C., Potterf, M., & Mönkkönen, M. (2021). High boreal forest multifunctionality requires continuous cover forestry as a dominant management. *Land Use Policy*, 100, 104918.

https://doi.org/https://doi.org/10.1016/j.landusepol.2020.104918.

FAO. (2020a). The State of the World's Forests 2020. Forests, biodiversity and people.. Rome city. Publisher: FAO and UNEP. Retrieved from: https://doi.org/10.4060/ca8642en.

FAO. (2020b). Global Forest Resources Assessment 2020: Main report. Rome city. Retrieved from: https://www.fao.org/documents/card/en/c/ca9825en.

Feng, R.-F., Yang, W.-Q., & Zhang, J. (2006). Artificial forest management for global change mitigation. *Acta Ecologica Sinica*, 11(26), 3870–3877.

Fenton, N. J., Bergeron, Y., & Paré, D. (2010). Decomposition rates of bryophytes in managed boreal forests: influence of bryophyte species and forest harvesting. *Plant and Soil*, 336(1–2), 499–508.

Fisichelli, N. A., Frelich, L. E., & Reich, P. B. (2014). Temperate tree expansion into adjacent boreal forest patches facilitated by warmer temperatures. *Ecography*, 37(2), 152–161.

Forestier en chef. (2022). Forestier en chef – Possibilités forestières 2023-2028. Retrieved from: https://forestierenchef.gouv.qc.ca/possibilites-forestieres/.

Gaboury, S., Boucher, J.-F., Villeneuve, C., Lord, D., & Gagnon, R. (2009). Estimating the net carbon balance of boreal open woodland afforestation: A case-study in Québec's closed-crown boreal forest. *Forest Ecology and Management*, 257(2), 483–494. https://doi.org/https://doi.org/10.1016/j.foreco.2008.09.037.

Gagné, C., Mainguy, J., & Fortin, D. (2016). The impact of forest harvesting on caribou–moose–wolf interactions decreases along a latitudinal gradient. *Biological Conservation*, 197, 215-222.

Gauthier, S., Bernier, P., Kuuluvainen, T., Shvidenko, A. Z., & Schepaschenko, D. G. (2015). Boreal forest health and global change. *Science*, 349(6250), 819–822.

Gauthier, S., Kuuluvainen, T., Macdonald, S. E., Shorohova, E., Shvidenko, A., Bélisle, A.-C., Vaillancourt, M.-A., Leduc, A., Grosbois, G., & Bergeron, Y. (2023). Ecosystem management of the boreal forest in the era of global change. M.M. Girona, H. Morin, S. Gauthier, Y. Bergeron (Eds.), Boreal Forests in the Face of Climate Change, (pp. 3-49). *Cham, Springer International Publishing*. https://doi.org/10.1007/978-3-031-15988-6_1.

Gauthier, S., M. A. Vaillancourt, A. Leduc, L. De Grandpré, D. Kneeshaw, H. Morin, P. Drapeau, and Y. Bergeron. 2008. Aménagement Écosystémique en Forêt

Boréale. Presses de l'Université du Québec, Québec. 568 p.

Gewehr, S., Drobyshev, I., Berninger, F., & Bergeron, Y. (2014). Soil characteristics mediate the distribution and response of boreal trees to climatic variability. *Canadian Journal of Forest Research*, 44(5), 487–498.

Girardin, M. P., Bouriaud, O., Hogg, E. H., Kurz, W., Zimmermann, N. E., Metsaranta, J. M., de Jong, R., Frank, D. C., Esper, J., & Büntgen, U. (2016). No growth stimulation of Canada’s boreal forest under half-century of combined warming and CO2 fertilization. *Proceedings of the National Academy of Sciences*, 113(52), E8406–E8414.

Girona, M., Aakala, T., Aquilué, N., Bélisle, A., Chaste, E., Danneyrolles, V., Díaz-Yáñez, O., D’Orangeville, L., Grosbois, G., Hester, A., Kim, S., Kulha, N., Martin, M., Moussaoui, L., Pappas, C., Portier, J., Teitelbaum, S., Tremblay, J.-P., Svensson, J., … Gauthier, S. (2023). Challenges for the sustainable management of the boreal forest under climate change. In Boreal Forests in the face of climate change: sustainable management (pp. 773-837). Cham, *Springer International Publishing*. doi.org/10.1007/978-3-031-15988-6_31.

Girona, M.M., Morin, H., Lussier, J.-M., & Ruel, J.-C. (2019a). Post-cutting Mortality Following Experimental Silvicultural Treatments in Unmanaged Boreal Forest Stands. *Frontiers in Forests and Global Change* (Vol. 2, p. 4). https://www.frontiersin.org/article/10.3389/ffgc.2019.00004.

Girona, M.M., Morin, H., Lussier, J.-M., & Ruel, J.-C. (2019b). Post-cutting Mortality Following Experimental Silvicultural Treatments in Unmanaged Boreal Forest Stands. *Frontiers in Forests and Global Change*, 2, 4. https://doi.org/10.3389/ffgc.2019.00004.

Girona, M.M., Morin, H., Lussier, J.-M., & Walsh, D. (2016). Radial Growth Response of Black Spruce Stands Ten Years after Experimental Shelterwoods and Seed-Tree Cuttings in Boreal Forest. *Forests*, 7(12), 240. https://doi.org/10.3390/f7100240.

Girona, M.M., Moussaoui, L., Morin, H., Thiffault, N., Leduc, A., Raymond, P., Bose, A., Bergeron, Y., & Lussier, J.-M. (2023). Innovative silviculture to achieve sustainable forest management in boreal forests: lessons from two large-scale experiments. In Boreal Forests in the face of climate change: sustainable management (pp. 417-440). Cham, *Springer International Publishing*. https://doi.org/10.1007/978-3-031-15988-6_16.

Girona, M.M., Rossi, S., Lussier, J.-M., Walsh, D., & Morin, H. (2017). Understanding tree growth responses after partial cuttings: A new approach. *PLoS One*, 12(2), e0172653. https://doi.org/10.1371/journal.pone.0172653.

Goulden, M. L., McMillan, A. M. S., Winston, G. C., Rocha, A. V, Manies, K. L., Harden, J. W., & Bond-Lamberty, B. P. (2011). Patterns of NPP, GPP, respiration, and NEP during boreal forest succession. *Global Change Biology,* 17(2), 855–871.

Gouvernement du Québec (2022). Aménagement durable des forêts. Retrieved from: https://www.quebec.ca/agriculture-environnement-et-ressources-naturelles/forets/gestion-forets-publiques/amenagement-durable-forets.

Grace, J. (2005). Role of forest biomes in the global carbon balance. In The carbon balance of forest biome. Cambridge city. In symposia-society for experimental biology (Vol: 57, p. 19).

Greene, D. F., Zasada, J. C., Sirois, L., Kneeshaw, D., Morin, H., Charron, I., & Simard, M.-J. (1999). A review of the regeneration dynamics of North American boreal forest tree species. *Canadian Journal of Forest Research*, 29(6), 824–839. https://doi.org/10.1139/x98-112.

Guignabert, A., Augusto, L., Delerue, F., Maugard, F., Gire, C., Magnin, C., Niollet, S., & Gonzalez, M. (2020). Combining partial cutting and direct seeding to overcome regeneration failures in dune forests. *Forest Ecology and Management*, 476, 118466. https://doi.org/https://doi.org/10.1016/j.foreco.2020.118466.

Gustafson, E. J., De Bruijn, A. M. G., Pangle, R. E., Limousin, J.-M., McDowell, N. G., Pockman, W. T., Sturtevant, B. R., Muss, J. D., & Kubiske, M. E. (2015). Integrating ecophysiology and forest landscape models to improve projections of drought effects under climate change. *Global Change Biology*, 21(2), 843–856. https://doi.org/10.1111/gcb.12713.

Gustafson, E. J., Kern, C. C., & Kabrick, J. M. (2023). Can assisted tree migration today sustain forest ecosystem goods and services for the future? *Forest Ecology and Management*, 529, 120723. https://doi.org/https://doi.org/10.1016/j.foreco.2022.120723.

Gustafson, E. J., Kern, C. C., Miranda, B. R., Sturtevant, B. R., Bronson, D. R., & Kabrick, J. M. (2020). Climate adaptive silviculture strategies: How do they impact growth, yield, diversity and value in forested landscapes? *Forest Ecology and Management,* 470–471, 118208. https://doi.org/https://doi.org/10.1016/j.foreco.2020.118208.

Gustafson, E. J., Miranda, B. R., Dreaden, T. J., Pinchot, C. C., & Jacobs, D. F. (2022). Beyond blight: Phytophthora root rot under climate change limits populations of reintroduced American chestnut. *Ecosphere*, 13(2), e3917. https://doi.org/https://doi.org/10.1002/ecs2.3917.

Gustafson, E. J., Shifley, S. R., Mladenoff, D. J., Nimerfro, K. K., & He, H. S. (2000). Spatial simulation of forest succession and timber harvesting using LANDIS. *Canadian Journal of Forest Research*, 30(1), 32–43.

Gustafson, E., & Miranda, B. (2019). PnET-Succession - LANDIS-II. Retrieved from: http://www.landis-ii.org/extensions/pnet-succession.

Gustafson, Miranda, B., & Sturtevant, B. (2018). Can Future CO_2 Concentrations Mitigate the Negative Effects of High Temperature and Longer Droughts on Forest Growth? *Forests* (Vol. 9, Issue 11). https://doi.org/10.3390/f9110664.

Harvey, B. D., Leduc, A., Gauthier, S., & Bergeron, Y. (2002). Stand-landscape integration in natural disturbance-based management of the southern boreal forest. *Forest Ecology and Management*, 155(1), 369–385. https://doi.org/https://doi.org/10.1016/S0378-1127(01)00573-4.

Hof, A. R., Dymond, C. C., & Mladenoff, D. J. (2017). Climate change mitigation through adaptation: the effectiveness of forest diversification by novel tree planting regimes. *Ecosphere*, 8(11), e01981.

Huang, C., He, H. S., Liang, Y., Wu, Z., Hawbaker, T. J., Gong, P., & Zhu, Z. (2018). Long-term effects of fire and harvest on carbon stocks of boreal forests in northeastern China. *Annals of Forest Science*, 75(2), 42. https://doi.org/10.1007/s13595-018-0722-x.

IPCC. (2014). : Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D. W., Minkkinen, K., & Byrne, K. A. (2007). How strongly can forest management influence soil carbon sequestration? *Geoderma*, 137(3), 253–268. https://doi.org/https://doi.org/10.1016/j.geoderma.2006.09.003

Jobidon, R., & Bergeron, Y. (2015). Assessing the biophysical potential for sustainable forest management: a case study from Quebec's boreal forest/Evaluation du potentiel biophysique pour un amenagement durable des forets: le cas de la foret boreale du Quebec. *Canadian Journal of Forest Research*, 45(5).

Jobidon, R., Bergeron, Y., Robitaille, A., Raulier, F., Gauthier, S., Imbeau, L., Saucier, J.-P., & Boudreault, C. (2015). A biophysical approach to delineate a northern limit to commercial forestry: the case of Quebec's boreal forest. *Canadian Journal of Forest Research*, 45(5), 515–528.

Johnson, E. A., Miyanishi, K., & Weir, J. M. H. (1998). Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. *Journal of Vegetation Science*, 9(4), 603–610.

Johnstone, J. F., Hollingsworth, T. N., Chapin III, F. S., & Mack, M. C. (2010). Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. *Global Change Biology*, 16(4), 1281-1295.

Kim, Y.-H., Kim, R., & Park, H. (2015). Reviews of forest carbon dynamics models that use empirical yield curves: CBM-CFS3, CO2FIX, CASMOFOR, EFISCEN AU - Kim, Hojung. *Forest Science and Technology*, 11(4), 212–222. https://doi.org/10.1080/21580103.2014.987325.

Köhl, M., Lasco, R., Cifuentes, M., Jonsson, Ö., Korhonen, K. T., Mundhenk, P., de Jesus Navar, J., & Stinson, G. (2015). Changes in forest production, biomass and carbon: Results from the 2015 UN FAO Global Forest Resource Assessment. *Forest Ecology and Management*, 352, 21–34.

Kouchaki-Penchah, H., Bahn, O., Vaillancourt, K., & Levasseur, A. (2022). The contribution of forest-based bioenergy in achieving deep decarbonization: Insights for Quebec (Canada) using a TIMES approach. *Energy Conversion and Management*, 252, 115081. https://doi.org/https://doi.org/10.1016/j.enconman.2021.115081.

Krofcheck, D. J., Remy, C. C., Keyser, A. R., & Hurteau, M. D. (2019). Optimizing Forest Management Stabilizes Carbon Under Projected Climate and Wildfires. Journal of Geophysical Research: *Biogeosciences*, 124(10), 3075–3087. https://doi.org/10.1029/2019JG005206.

Krug, J. H. A. (2018). Accounting of GHG emissions and removals from forest management: a long road from Kyoto to Paris. *Carbon Balance and Management*, 13(1), 1. https://doi.org/10.1186/s13021-017-0089-6.

Kurz, W. A., Dymond, C. C., White, T. M., Stinson, G., Shaw, C. H., Rampley, G. J., Smyth, C., Simpson, B. N., Neilson, E. T., & Trofymow, J. A. (2009). CBM-CFS3: a model of carbon-dynamics in forestry and land-use change implementing IPCC standards. *Ecological Modelling*, 220(4), 480–504.

Kurz, W. A., Shaw, C. H., Boisvenue, C., Stinson, G., Metsaranta, J., Leckie, D., Dyk, A., Smyth, C., & Neilson, E. T. (2013). Carbon in Canada's boreal forest — A synthesis. *Environmental Reviews*, 21(4), 260–292. https://doi.org/10.1139/er-2013-0041.

Lacerte, R., Leblond, M., & St-Laurent, M. H. (2021). Determinants of vegetation regeneration on forest roads following restoration treatments: implications for boreal caribou conservation. *Restoration Ecology*, 29(7), e13414.

Lafleur, B., Paré, D., Fenton, N. J., & Bergeron, Y. (2010). Do harvest methods and soil type impact the regeneration and growth of black spruce stands in northwestern Quebec? *Canadian Journal of Forest Research*, 40(9), 1843–1851.

Lafontaine, A., Drapeau, P., Fortin, D., Gauthier, S., Boulanger, Y., & St-Laurent, M. H. (2019). Exposure to historical burn rates shapes the response of boreal caribou to timber harvesting. *Ecosphere*, 10(5), e02739.

Laganière, J., Paré, D., Bergeron, Y., Chen, H. Y. H., Brassard, B. W., & Cavard, X. (2013). Stability of Soil Carbon Stocks Varies with Forest Composition in the Canadian Boreal Biome. *Ecosystems*, 16(5), 852–865. https://doi.org/10.1007/s10021-013-9658-z.

Landry, G., Thiffault, E., Cyr, D., Moreau, L., Boulanger, Y., & Dymond, C. (2021). Mitigation Potential of Ecosystem-Based Forest Management under Climate Change: A Case Study in the Boreal-Temperate Forest Ecotone. *Forests*, Vol. 12, Issue 12. https://doi.org/10.3390/f12121667

Landsberg, J. J., Sands, P. J., Landsberg, J., & Sands, P. (2011). Physiological ecology of forest production: principles, processes and models. Elsevier/Academic Press London, Amsterdam city (Vol. 4), p352.

Lavoie, J., Girona, M.M., Grosbois, G., & Morin, H. (2021). Does the type of silvicultural practice influence spruce budworm defoliation of seedlings? *Ecosphere*, 12(4), e03506.

Lavoie, M., Paré, D., & Bergeron, Y. (2005). Impact of global change and forest management on carbon sequestration in northern forested peatlands. *Environmental Reviews*, 13(4), 199–240.

Lavoie, M., Paré, D., & Bergeron, Y. (2006). Relationships between microsite type and the growth and nutrition of young black spruce on post-disturbed lowland black spruce sites in eastern Canada. *Canadian Journal of Forest Research*, 37(1), 62–73.

Leclerc, M., Dussault, C., & St-Laurent, M. H. (2014). Behavioural strategies towards human disturbances explain individual performance in woodland caribou.

Oecologia, 176, 297-306.

Lee, J., Morrison, I. K., Leblanc, J.-D., Dumas, M. T., & Cameron, D. A. (2002). Carbon sequestration in trees and regrowth vegetation as affected by clearcut and partial cut harvesting in a second-growth boreal mixedwood. *Forest Ecology and Management*, 169(1), 83–101. https://doi.org/https://doi.org/10.1016/S0378-1127(02)00300-6.

Liang, Y., Duveneck, M. J., Gustafson, E. J., Serra-Diaz, J. M., & Thompson, J. R. (2018). How disturbance, competition, and dispersal interact to prevent tree range boundaries from keeping pace with climate change. *Global Change Biology*, 24(1), e335–e351.

Liu, S., Bond-Lamberty, B., Hicke, J. A., Vargas, R., Zhao, S., Chen, J., Edburg, S. L., Hu, Y., Liu, J., & McGuire, A. D. (2011). Simulating the impacts of disturbances on forest carbon cycling in North America: Processes, data, models, and challenges. *Journal of Geophysical Research*. Biogeosciences, 116(G4).

Liu, Z., Peng, C., De Grandpré, L., Candau, J.-N., Work, T., Huang, C., & Kneeshaw, D. (2019). Simulation and Analysis of the Effect of a Spruce Budworm Outbreak on Carbon Dynamics in Boreal Forests of Quebec. *Ecosystems*, 22(8), 1838–1851. https://doi.org/10.1007/s10021-019-00377-7.

MacLean, D. A. (2016). Impacts of insect outbreaks on tree mortality, productivity, and stand development. *The Canadian Entomologist*, 148(S1), S138–S159. https://doi.org/DOI: 10.4039/tce.2015.24.

Martin, M., Leduc, A., Fenton, N. J., Girona, M.M., Bergeron, Y., & Valeria, O. (2022). Irregular forest structures originating after fire: An opportunity to promote alternatives to even-aged management in boreal forests. *Journal of Applied Ecology*, 59(7), 1792–1803. https://doi.org/https://doi.org/10.1111/1365-2664.14186.

Mayer, M., Prescott, C. E., Abaker, W. E. A., Augusto, L., Cécillon, L., Ferreira, G. W. D., James, J., Jandl, R., Katzensteiner, K., Laclau, J.-P., Laganière, J., Nouvellon, Y., Paré, D., Stanturf, J. A., Vanguelova, E. I., & Vesterdal, L. (2020). Tamm Review: Influence of forest management activities on soil organic carbon stocks: A knowledge synthesis. *Forest Ecology and Management*, 466, 118127. https://doi.org/https://doi.org/10.1016/j.foreco.2020.118127.

Mayer, M., Sandén, H., Rewald, B., Godbold, D. L., & Katzensteiner, K. (2017). Increase in heterotrophic soil respiration by temperature drives decline in soil organic carbon stocks after forest windthrow in a mountainous ecosystem. *Functional Ecology*, 31(5), 1163–1172. https://doi.org/10.1111/1365-2435.12805

McKenney, D.W., Yemshanov, D., Pedlar, J., Allen, D., Lawrence, K., Hope, E., Lu, B., & Eddy, B (2016). Canada's timber supply: current status and future prospects under a changing climate. Natural Resources Canada, Canadian Forest Service. Great Lakes Forestry Centre, Sault Ste. Marie, Ontario. 75p. Information Report GLC-X-15. https://cfs.nrcan.gc.ca/publications?id=37783.

McKenzie, P. F., Duveneck, M. J., Morreale, L. L., & Thompson, J. R. (2019). Local and global parameter sensitivity within an ecophysiologically based forest landscape model. *Environmental Modelling & Software*, 117, 1–13.

Meinshausen, M., Smith, S. J., Calvin, K., Daniel, J. S., Kainuma, M. L. T., Lamarque, J.-F., Matsumoto, K., Montzka, S. A., Raper, S. C. B., & Riahi, K. (2011). The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Climatic Change*, 109(1), 213–241.

Mitchell, S. J. (2012). Wind as a natural disturbance agent in forests: a synthesis. *Forestry*: *An International Journal of Forest Research*, 86(2), 147–157. https://doi.org/10.1093/forestry/cps058.

Mladenoff, D. J., & He, H. S. (1999). Design, behavior and application of LANDIS, an object-oriented model of forest landscape disturbance and succession. Spatial Modeling of Forest Landscape Change: Approaches and Applications. Cambridge University Press, Cambridge city, UK, 125–162.

Molina, E., Valeria, O., De Grandpre, L., Ramirez, J. A., Cyr, D., & Boulanger, Y. (2021). Projecting future aboveground biomass and productivity of managed eastern Canadian mixedwood boreal forest in response to climate change. *Forest Ecology and Management*, 487, 119016.

Moreau, L., Thiffault, E., Cyr, D., Boulanger, Y., & Beauregard, R. (2022). How can the forest sector mitigate climate change in a changing climate? Case studies of boreal and northern temperate forests in eastern Canada. *Forest Ecosystems*, 9, 100026. https://doi.org/https://doi.org/10.1016/j.fecs.2022.100026.

Moreau, G., Fortin, D., Couturier, S. & Duchesne, T. (2012) Multi-level functional responses for wildlife conservation: the case of threatened caribou in managed boreal forests. *Journal of Applied Ecology*, 49, 611–620.

Moussaoui, L., Leduc, A., Girona, M. M., Bélisle, A. C., Lafleur, B., Fenton, N. J., & Bergeron, Y. (2020). Success Factors for Experimental Partial Harvesting in Unmanaged Boreal Forest: 10-Year Stand Yield Results. *Forests*, 11(11), 1199.

MRNF. (2010). Forêt ouverte: inventaire forestier national. Gouvernement du Québec. Retrieved from: https://www.foretouverte.gouv.qc.ca/.

MRNF. (2018). Insectes, maladies et feux dans les forêts du Québec en 2018. Gouvernement du Québec, Direction de la protection des forêts. Québec city, p72. Retrieved from: https://mffp.gouv.qc.ca/wp-content/uploads/bilan2018-p.pdf.

Murphy, K., & Myors, B. (2023). A Simple and General Model for Power Analysis. In Statistical Power Analysis (5th ed., pp. 24). Routledge. New York city. https://doi.org/10.4324/9781003296225 .

Navarro, L., Morin, H., Bergeron, Y., & Girona, M. M. (2018). Changes in Spatiotemporal Patterns of 20th Century Spruce Budworm Outbreaks in Eastern Canadian Boreal Forests. *Frontiers in Plant Science* (Vol. 9, p. 1905). https://www.frontiersin.org/article/10.3389/fpls.2018.01905.

Noormets, A., Epron, D., Domec, J. C., McNulty, S. G., Fox, T., Sun, G., & King, J. S. (2015). Effects of forest management on productivity and carbon sequestration: A review and hypothesis. *Forest Ecology and Management*, 355, 124–140. https://doi.org/https://doi.org/10.1016/j.foreco.2015.05.019.

NRC. (2022). How does disturbance shape Canada's forests? The state of canada's forests: annual report 2022. Natural Resources Canada, p104. Retrieved from: https://natural-resources.canada.ca/our-natural-resources/forests/state-canadas-forests-report/16496.

Paixao, C., Krause, C., Morin, H., & Achim, A. (2019). Wood quality of black spruce and balsam fir trees defoliated by spruce budworm: A case study in the boreal forest of Quebec, Canada. *Forest Ecology and Management*, 437, 201–210. https://doi.org/https://doi.org/10.1016/j.foreco.2019.01.032.

Pamerleau-Couture, É., Krause, C., Pothier, D., & Weiskittel, A. (2015). Effect of three partial cutting practices on stand structure and growth of residual black spruce trees in north-eastern Quebec. Forestry: *An International Journal of Forest Research*, 88(4), 471–483. https://doi.org/10.1093/forestry/cpv017.

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., Phillips, O. L., Shvidenko, A., Lewis, S. L., Canadell, J. G., Ciais, P., Jackson, R. B., Pacala, S. W., McGuire, A. D., Piao, S., Rautiainen, A., Sitch, S., & Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, 333(6045), 988 LP – 993. https://doi.org/10.1126/science.1201609.

Paradis, L., Thiffault, E., & Achim, A. (2019). Comparison of carbon balance and climate change mitigation potential of forest management strategies in the boreal forest of Quebec (Canada). Forestry: *An International Journal of Forest Research*, 92(3), 264–277. https://doi.org/10.1093/forestry/cpz004.

Paré, D., Banville, J. L., Garneau, M., & Bergeron, Y. (2011). Soil Carbon Stocks and Soil Carbon Quality in the Upland Portion of a Boreal Landscape, James Bay, Quebec. *Ecosystems*, 14(4), 533–546. https://doi.org/10.1007/s10021-011-9429-7.

Paré, D., Bernier, P., Lafleur, B., Titus, B. D., Thiffault, E., Maynard, D. G., & Guo, X. (2013). Estimating stand-scale biomass, nutrient contents, and associated uncertainties for tree species of Canadian forests. *Canadian Journal of Forest Research*, 43(7), 599–608.

Pau, M., Gauthier, S., Boulanger, Y., Ouzennou, H., Girardin, M. P., & Bergeron, Y. (2023). Response of forest productivity to changes in growth and fire regime due to climate change. *Canadian Journal of Forest Research*. 53(9): 663-676. https://doi.org/10.1139/cjfr-2022-0207.

Peng, Chang-hui, & Apps, M. J. (2000). Simulating global soil-CO_2 flux and its response to climate change. *Journal of Environmental Sciences*, 12(3), 257–265.

Peng, Changhui, Jiang, H., Apps, M. J., & Zhang, Y. (2002). Effects of harvesting regimes on carbon and nitrogen dynamics of boreal forests in central Canada: a process model simulation. *Ecological Modelling*, 155(2–3), 177–189.

Peng, Changhui, Liu, J., Dang, Q., Apps, M. J., & Jiang, H. (2002). TRIPLEX: a generic hybrid model for predicting forest growth and carbon and nitrogen dynamics. *Ecological Modelling*, 153(1), 109–130. https://doi.org/https://doi.org/10.1016/S0304-3800(01)00505-1.

Peng, Ma, Z., Lei, X., Zhu, Q., Chen, H., Wang, W., Liu, S., Li, W., Fang, X., & Zhou, X. (2011). A drought-induced pervasive increase in tree mortality across Canada's boreal forests. *Nature Climate Change*, 1(9), 467.

Perez-Garcia, J., Lippke, B., Comnick, J., & Manriquez, C. (2007). An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. Wood and Fiber Science, 37, 140–148.

Perry, D. A., Oren, R., & Hart, S. C. (2008). Forest Ecosystems (2nd ed.). MD: Johns Hopkins University Press, Baltimore city. https://doi.org/10.56021/9780801888403.

Pothier, D., & Savard, F. (1998). Actualisation des tables de production pour les principales espèces du Québec. Gouvernement Du Québec, Ministère Des Ressources Naturelles, Bibliothèque Nationale Du Québec. RN98-3054.

Price, D. T., Halliwell, D. H., Apps, M. J., Kurz, W. A., & Curry, S. R. (2005).

Comprehensive assessment of carbon stocks and fluxes in a Boreal-Cordilleran forest management unit. *Canadian Journal of Forest Research*, 27(12), 2005–2016.

Prima, M.-C., Duchesne, T., Fortin, A., Rivest, L.-P., Drapeau, P., St-Laurent, M.-H., & Fortin, D. (2019). A landscape experiment of spatial network robustness and space-use reorganization following habitat fragmentation. *Functional Ecology*, 33(9), 1663–1673. https://doi.org/https://doi.org/10.1111/1365-2435.13380.

Rich, R. L., Frelich, L. E., & Reich, P. B (2007). Wind-throw mortality in the southern boreal forest: effects of species, diameter and stand age. *Journal of Ecology*, 95(6), 1261–1273. https://doi.org/10.1111/j.1365-2745.2007.01301.x

Robert, L., Sturtevant, B. R., Cooke, B. J., James, P. M. A., Fortin, M., Townsend, P. A., Wolter, P. T., & Kneeshaw, D. (2018). Landscape host abundance and configuration regulate periodic outbreak behavior in spruce budworm Choristoneura fumiferana. *Ecography*, 41(9), 1556–1571.

Rogelj, J., den Elzen, M., Höhne, N., Fransen, T., Fekete, H., Winkler, H., Schaeffer, R., Sha, F., Riahi, K., & Meinshausen, M. (2016). Paris Agreement climate proposals need a boost to keep warming well below 2 °C. *Nature*, 534, 631. https://doi.org/10.1038/nature18307.

Running, S. W., Thornton, P. E., Nemani, R., & Glassy, J. M. (2000). Global Terrestrial Gross and Net Primary Productivity from the Earth Observing System (O. E. Sala, R. B. Jackson, H. A. Mooney, & R. W. Howarth (eds.); pp. 44–57). *Springer*, New York city. https://doi.org/10.1007/978-1-4612-1224-9_4.

Sage, L. K., Smith, C. T., Kurz, W., Thiffault, E., Paré, D., & Bernier, P. (2019). Empirical and Predicted Boreal Forest Carbon Pools Following Stem-Only Harvesting in Quebec, Canada. *Soil Science Society of America Journal*, 83(S1), S59–S81. https://doi.org/https://doi.org/10.2136/sssaj2018.07.0283.

Saltelli, A., & Annoni, P. (2010). How to avoid a perfunctory sensitivity analysis. *Environmental Modelling & Software*, 25(12), 1508–1517. https://doi.org/https://doi.org/10.1016/j.envsoft.2010.04.012.

Sánchez-Pinillos, M., D'Orangeville, L., Boulanger, Y., Comeau, P., Wang, J., Taylor, A. R., & Kneeshaw, D. (2022). Sequential droughts: A silent trigger of boreal forest mortality. *Global Change Biology*, 28(2), 542–556. https://doi.org/https://doi.org/10.1111/gcb.15913

Scheller, R. M., & Domingo, J. B. (2018). LANDIS-II Base Fire v4.0 Extension User Guide. In Reproduction (pp. 0–9). Retrieved from: https://www.landis-ii.org/extensions.

Scheller, R. M., Domingo, J. B., Sturtevant, B. R., Williams, J. S., Rudy, A., Gustafson, E. J., & Mladenoff, D. J. (2007). Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. *Ecological Modelling*, 201(3–4), 409–419.

Scheller, R. M., & Mladenoff, D. J. (2004). A forest growth and biomass module for a landscape simulation model, LANDIS: design, validation, and application. *Ecological Modelling*, 180(1), 211–229. https://doi.org/https://doi.org/10.1016/j.ecolmodel.2004.01.022

Scheller, R. M., Sturtevant, B. R., Gustafson, E. J., Miranda, B. R., & Zollner, P. A. (2019). Biomass Harvest v4.3 LANDIS-II Extension User Guide. 0–9. Retrieved from: https://www.landis-ii.org/extensions

Scheller, R. M., Sturtevant, B. R., Gustafson, E. J., Miranda, B. R., Zollner, P. A., Mladenoff, D. J., & Domingo, J. B. (2018). LANDIS-II Base Wind v3.0 Extension User Guide (pp. 1–11). Retrieved from: https://www.landis-ii.org/extensions.

Schimel, D. S., House, J. I., Hibbard, K. A., Bousquet, P., Ciais, P., Peylin, P., Braswell, B. H., Apps, M. J., Baker, D., & Bondeau, A. (2001). Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. *Nature*, 414(6860), 169.

Schuur, E. A. G., Bockheim, J., Canadell, J. G., Euskirchen, E., Field, C. B., Goryachkin, S. V, Hagemann, S., Kuhry, P., Lafleur, P. M., Lee, H., Mazhitova, G., Nelson, F. E., Rinke, A., Romanovsky, V. E., Shiklomanov, N., Tarnocai, C., Venevsky, S., Vogel, J. G., & Zimov, S. A. (2008). Vulnerability of Permafrost Carbon to Climate Change: Implications for the Global Carbon Cycle. *BioScience*, 58(8), 701–714. https://doi.org/10.1641/B580807.

Seymour, R. S., & Hunter, M. L. (1992). New forestry in eastern spruce-fir forests: principles and applications to Maine. College of Forest Resources, University of Maine, Orono, USA. (Vol. 716).

Shaw, C. H., Banfield, E., & Kurz, W. A. (2008). Stratifying soils into pedogenically similar categories for modeling forest soil carbon. *Canadian Journal of Soil Science*, 88(4), 501–516. https://doi.org/10.4141/CJSS07099

Simard, M., Lecomte, N., Bergeron, Y., Bernier, P., & Paré, D. (2008). Un aménagement écosystémique de la pessière du nord de la ceinture d’argile québécoise. Aménagement Écosystémique En Forêt Boréale. Presses de l’Université du Québec, Québec city, p. 568.

Simard, M., Lecomte, N., Bergeron, Y., Bernier, P. Y., & Paré, D. (2007). Forest productivity decline caused by successional paludification of boreal soils. *Ecological Applications*, 17(6), 1619–1637. https://doi.org/https://doi.org/10.1890/06-1795.1.

Simard, W. S., Roach, W. J., Defrenne, C. E., Pickles, B. J., Snyder, E. N., Robinson, A., & Lavkulich, L. M. (2020). Harvest Intensity Effects on Carbon Stocks and Biodiversity Are Dependent on Regional Climate in Douglas-Fir Forests of British Columbia. *Frontiers in Forests and Global Change* (Vol. 3). https://www.frontiersin.org/articles/10.3389/ffgc.2020.00088.

Smyth, C. E., Stinson, G., Neilson, E., Lemprière, T. C., Hafer, M., Rampley, G. J., & Kurz, W. A. (2014). Quantifying the biophysical climate change mitigation potential of Canada's forest sector. *Biogeosciences*, 11(13), 3515–3529.

Splawinski, T. B., Cyr, D., Gauthier, S., Jetté, J.-P., & Bergeron, Y. (2019). Analyzing risk of regeneration failure in the managed boreal forest of northwestern Quebec. *Canadian Journal of Forest Research*, 49(6), 680–691.

St-Laurent, M.-H., Boulanger, Y., Cyr, D., Manka, F., Drapeau, P., & Gauthier, S. (2022). Lowering the rate of timber harvesting to mitigate impacts of climate change on boreal caribou habitat quality in eastern Canada. *Science of The Total Environment*, 838, 156244. https://doi.org/https://doi.org/10.1016/j.scitotenv.2022.156244.

Stralberg, D., Bayne, E. M., Cumming, S. G., Sólymos, P., Song, S. J., & Schmiegelow, F. K. A. (2015). Conservation of future boreal forest bird communities considering lags in vegetation response to climate change: a modified refugia approach. *Diversity and Distributions*, 21(9), 1112–1128.

Stralberg, D., Wang, X., Parisien, M., Robinne, F., Sólymos, P., Mahon, C. L., Nielsen, S. E., & Bayne, E. M. (2018). Wildfire-mediated vegetation change in boreal forests of Alberta, Canada. *Ecosphere*, 9(3), e02156.

Stuenzi, S. M., & Schaepman-Strub, G. (2020). Vegetation trajectories and shortwave radiative forcing following boreal forest disturbance in eastern Siberia. *Journal of Geophysical Research: Biogeosciences*, 125(6), e2019JG005395.

Sturtevant, B. R., Gustafson, E. J., Li, W., & He, H. S. (2004). Modeling biological disturbances in LANDIS: a module description and demonstration using spruce budworm. *Ecological Modelling*, 180(1), 153–174. https://doi.org/https://doi.org/10.1016/j.ecolmodel.2004.01.021.

Sturtevant, B. R., He, H. S., Scheller, R. M., & Miranda, B. R. (2019). LANDIS-II

Biological Disturbance Agent v2. 0 Extension User Guide, p0–19. Retrieved from: https://www.landis-ii.org/extensions.

St-Laurent, M. H., Boulanger, Y., Cyr, D., Manka, F., Drapeau, P., & Gauthier, S. (2022). Lowering the rate of timber harvesting to mitigate impacts of climate change on boreal caribou habitat quality in eastern Canada. *Science of The Total Environment*, 838, 156244.

Tarnocai, C., Canadell, J. G., Schuur, E. A. G., Kuhry, P., Mazhitova, G., & Zimov, S. (2009). Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, 23(2). https://doi.org/10.1029/2008GB003327.

Taylor, A. R., Wang, J. R., & Kurz, W. A. (2008). Effects of harvesting intensity on carbon stocks in eastern Canadian red spruce (Picea rubens) forests: An exploratory analysis using the CBM-CFS3 simulation model. *Forest Ecology and Management*, 255(10), 3632–3641. https://doi.org/https://doi.org/10.1016/j.foreco.2008.02.052.

Tian, H., Lu, C., Yang, J., Banger, K., Huntzinger, D. N., Schwalm, C. R., Michalak, A. M., Cook, R., Ciais, P., Hayes, D., Huang, M., Ito, A., Jain, A. K., Lei, H., Mao, J., Pan, S., Post, W. M., Peng, S., Poulter, B., … Zeng, N. (2015). Global patterns and controls of soil organic carbon dynamics as simulated by multiple terrestrial biosphere models: Current status and future directions. *Global Biogeochemical Cycles*, 29(6), 775–792. https://doi.org/10.1002/2014GB005021

Tremblay, J. A., Boulanger, Y., Cyr, D., Taylor, A. R., Price, D. T., & St-Laurent, M.-H. (2018). Harvesting interacts with climate change to affect future habitat quality of a focal species in eastern Canada’s boreal forest. *PLoS One*, 13(2), e0191645.

Volney, W. J. A., & Fleming, R. A. (2000). Climate change and impacts of boreal forest insects. Agriculture, Ecosystems & Environment, 82(1–3), 283–294.

Walker, X. J., Mack, M. C., & Johnstone, J. F. (2017). Predicting ecosystem resilience to fire from tree ring analysis in black spruce forests. *Ecosystems*, 20(6), 1137–1150.

Wallentin, C., & Nilsson, U. (2013). Storm and snow damage in a Norway spruce thinning experiment in southern Sweden. *Forestry: An International Journal of Forest Research*, 87(2), 229–238. https://doi.org/10.1093/forestry/cpt046.

Wang, J., Taylor, A. R., & D’Orangeville, L. (2023). Warming-induced tree growth may help offset increasing disturbance across the Canadian boreal forest. *Proceedings of the National Academy of Sciences*, 120(2), e2212780120. https://doi.org/10.1073/pnas.2212780120.

Wang, T., Hamann, A., Spittlehouse, D., & Carroll, C. (2016). Locally Downscaled and Spatially Customizable Climate Data for Historical and Future Periods for North America. *PLoS One*, 11(6), e0156720. https://doi.org/10.1371/journal.pone.0156720.

Ward, B. C., Mladenoff, D. J., & Scheller, R. M. (2005). Simulating landscape-level effects of constraints to public forest regeneration harvests due to adjacent residential development in northern Wisconsin. *Forest Science*, 51(6), 616–632.

Way, D. A., & Sage, R. F. (2008). Thermal acclimation of photosynthesis in black spruce [Picea mariana (Mill.) B.S.P.]. *Plant, Cell & Environment*, 31(9), 1250–1262. https://doi.org/https://doi.org/10.1111/j.1365-3040.2008.01842.x.

White, A., Cannell, M. G. R., & Friend, A. D. (1999). Climate change impacts on ecosystems and the terrestrial carbon sink: a new assessment. *Global Environmental Change*, 9, S21–S30. https://doi.org/https://doi.org/10.1016/S0959-3780(99)00016-3.

Wu, Z., Dai, E., Ge, Q., Xi, W., & Wang, X. (2017). Modelling the integrated effects of land use and climate change scenarios on forest ecosystem aboveground biomass, a case study in Taihe County of China. *Journal of Geographical Sciences*, 27(2), 205–222. https://doi.org/10.1007/s11442-017-1372-x.

Yamanoi, K., Mizoguchi, Y., & Utsugi, H. (2015). Effects of a windthrow disturbance on the carbon balance of a broadleaf deciduous forest in Hokkaido, Japan. *Biogeosciences*, 12(23), 6837–6851.

Zhuo, W., Dai, E., Wu, Z., & Lin, M. (2020). Assessing differences in the response of forest aboveground biomass and composition under climate change in subtropical forest transition zone. *Science of The Total Environment*, 706, 135746.